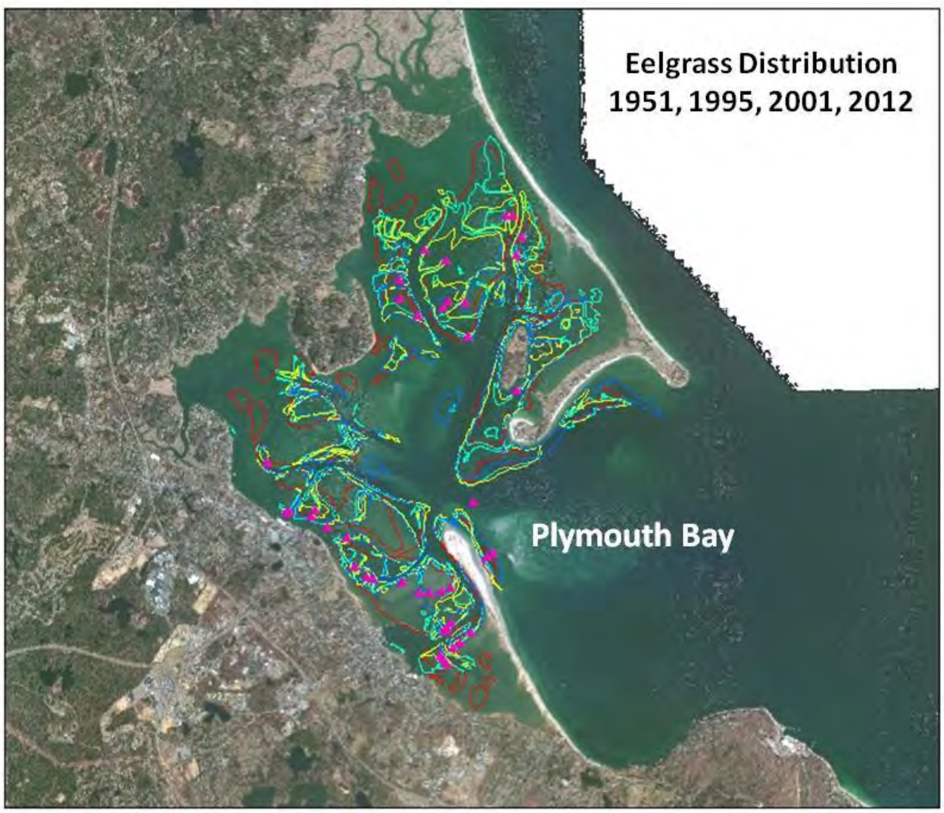
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine the Critical Nitrogen Loading Threshold for the Plymouth Harbor, Kingston Bay and Duxbury Bay Estuarine System
Towns of Plymouth, Kingston, Duxbury, MA.



University of Massachusetts Dartmouth
School of Marine Science and Technology

Massachusetts Department of Environmental Protection

DRAFT REPORT – DECEMBER 2017

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I. INTRODUCTION

The Plymouth Harbor, Kingston Bay and Duxbury Bay (PKD) embayment system is a complex estuary on the southeastern shore of the Massachusetts coastline and is bounded by the Towns of Plymouth, Kingston and Duxbury but whose watershed also contains the Towns of Halifax, Marshfield, Pembroke and Plympton. The PKD embayment system supports a single large eastern facing inlet which receives marine water directly from Cape Cod Bay (Figure I -1). Land-uses closest to an embayment generally have greater impact than those in the upper portions of the watershed, which allows for potential attenuation of nitrogen during transport through natural aquatic systems (e.g. ponds, rivers, wetlands etc.) prior to discharge to the embayment. However, effective nutrient management for protection/restoration of the PKD Embayment System will require consideration of all sources of nitrogen load throughout the entire watershed. That the open water basins are shared among three towns and the entire watershed to the system is contained within an additional four towns will make development and implementation of a comprehensive nutrient management and protection/restoration plan more complex as the challenges are increased due to the level of inter-municipal co-ordination required and potentially conflicting municipal constraints and regulations.

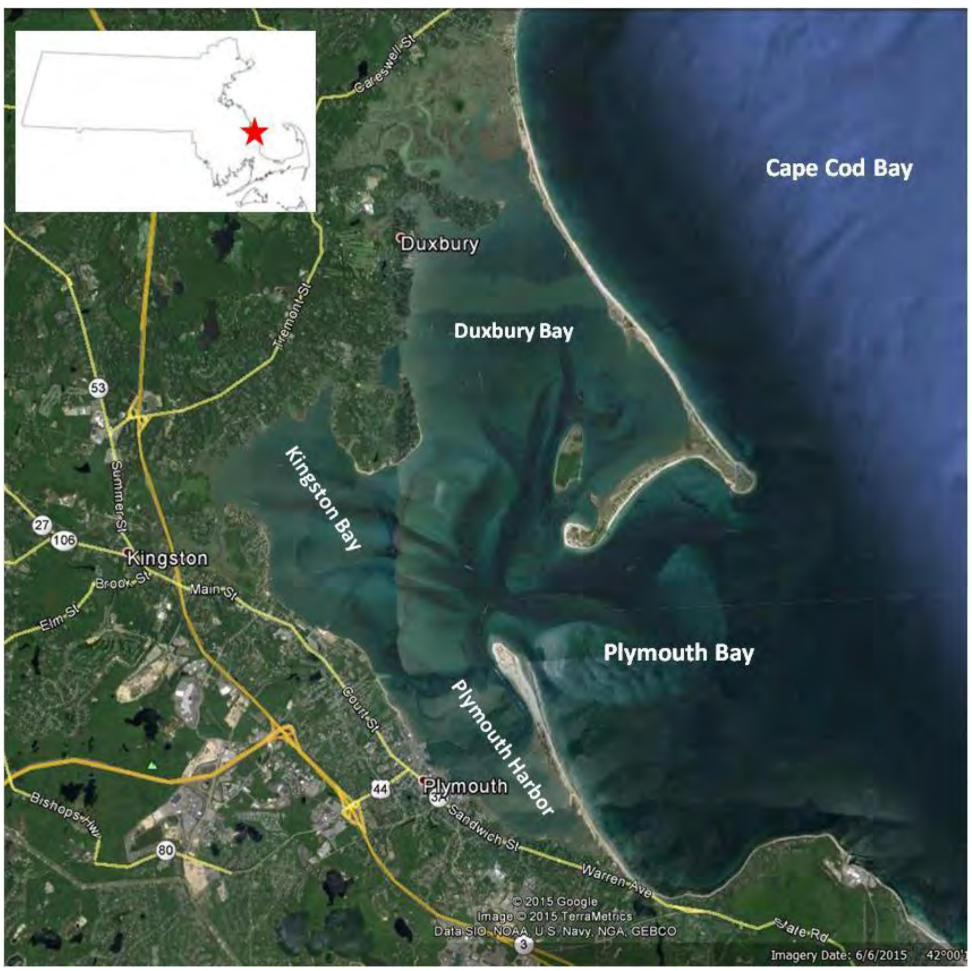


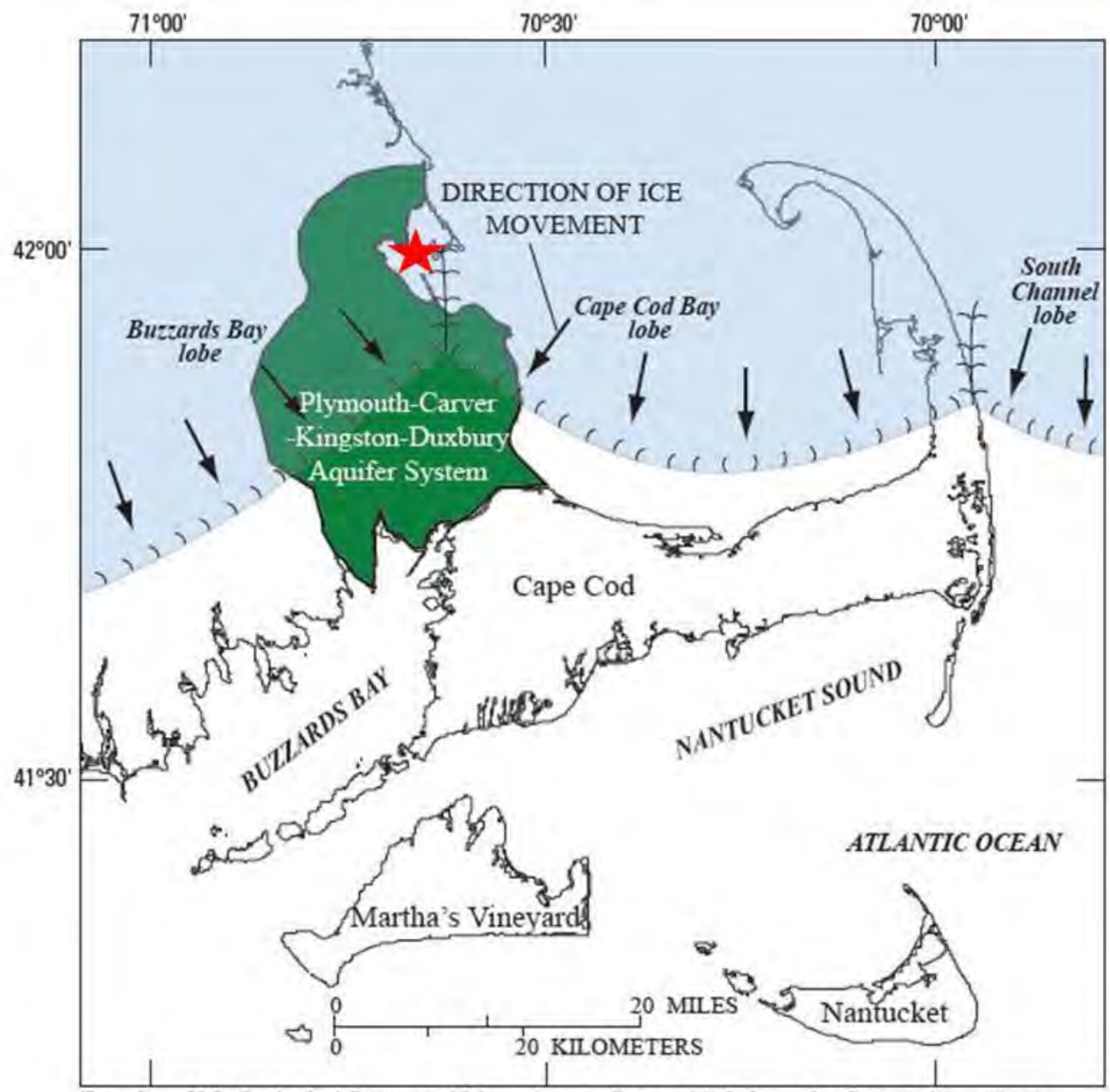
Figure I-1. Location of the Plymouth Harbor, Kingston Bay, Duxbury Bay (PKD) Embayment System, is bounded by the Towns of Plymouth, Kingston and Duxbury. The PKD embayment system is one of the largest estuaries in southeastern Massachusetts with a large inlet that supports free exchange of tidal waters with Cape Cod Bay.

The nature of enclosed embayments in populous regions brings two opposing elements to bare: 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. The multiple coves and sub-embayments to the PKD Embayment System greatly increases its shoreline and decreases the travel time of groundwater (and its entrained pollutants) from the watershed recharge areas to bay regions of discharge. This embayment system also receives direct surfacewater discharge from two significant Rivers (Jones River and Eel River) as well as a large brook (Town Brook), all of which serve as direct conduits of nutrient load from within the watershed to estuarine waters. The Town of Plymouth also operates a wastewater treatment facility that discharges directly to Plymouth Harbor.

The Plymouth-Kingston-Duxbury Bay Embayment System is a complex coastal open water embayment comprised of a northern basin (Duxbury Bay) that supports a relatively large salt marsh above the Powder Point Bridge. The salt marsh connects directly to Duxbury Bay which is joined by Kingston Bay at the level of the inlet to the overall system. The basin is formed behind a 3.6 mile barrier comprised of 2 barrier spits, Long Beach to the south and Duxbury Beach/Saquish Neck to the northeast. Shoreline management began in the 1700's and today a stone dike runs the length of Long Beach (southern spit). Harbor dredging began in the 1800s with the deepening of navigation basin and channels, which has continued and expanded until present. The enclosed basin is continually modified by coastal processes and sediment transport, while the southern spit (Long Island has been undergoing erosion and revetment and seawall failures in recent decades.

Kingston Bay receives the large freshwater inflow from the Jones River and is connected to Plymouth Harbor to the south via a complex network of channels traversing the sand flats. Similar to Kingston Bay, Plymouth Harbor also receives large fresh surfacewater discharges from both the Eel River at the southern most end of the system and Town Brook which discharges at the level of Plymouth Rock and the Plymouth Cultural Center. Duxbury Bay, Kingston Bay and Plymouth Harbor all come together in a common basin adjacent the system's tidal inlet.

The present Plymouth-Kingston-Duxbury Bay Embayment System results from a complex geologic history dominated by glacial processes occurring during the last glaciation of the southeastern Massachusetts region. The late Wisconsinan Laurentide ice sheet reached its maximum extent and southernmost position about 20,000 years before present (BP), as indicated by the presence of terminal moraines on Martha's Vineyard and Nantucket and the southern limit of abundant gravel on the sea floor of Nantucket Sound and Vineyard Sound (Schlee and Pratt, 1970; Oldale, 1992; Uchupi et al., 1996). The glacial deposits within the watershed to the Plymouth-Kingston-Duxbury embayment system consist of sediments that range in size from clay to boulders. These sediments were deposited as a result of a complex series of retreats and advances of two large sheets of ice—the Buzzards Bay and Cape Cod Bay lobes (Mather and others, 1942) (Figure I-2). The predominant glacial features are outwash plains and moraines in the southern Plymouth-Carver area and valley-fill stratified glacial deposits bordered by upland till areas in the northern Duxbury area (USGS, 2009-5063) and are underlain by bedrock.



Base from U.S. Geological Survey and Massachusetts Geographic Information System data sources, Massachusetts State Plane Coordinate System, Mainland Zone

Figure I-2. Generalized geologic map of study region (south coast including Cape Cod and Islands) for the Massachusetts Estuaries Project analysis of the Plymouth-Kingston-Duxbury Bay Embayment System (USGS, 2009-5063).

The watershed to the Plymouth-Kingston-Duxbury Bay Embayment System is composed of a complex surficial aquifer whose main water bearing deposits are large outwash plain deposits, the Wareham and Carver Pitted Plains (USGS, 2009-5063. As presented in the USGS report summarizing the update of the Plymouth-Carver Aquifer Model (USGS, 2009-5063), these outwash plain deposits were formed by meltwater from the retreat of the Buzzards Bay and Cape Cod Bay lobe ice sheets. These meltwater deposits formed as deltas deposited within a large glacial lake that formed from meltwater in the wake of the retreating ice sheets (Larson, 1980). The flat surfaces of the outwash plains were altered by the numerous kettle holes that were formed as collapse structures by the melting of buried blocks of ice stranded by the retreating ice lobes. These ice blocks, stranded directly on basal till and bedrock, subsequently were buried by prograding deltaic sediments. When the buried ice blocks melted, coarse sands and gravels collapsed into the resulting depressions. The kettle holes that intercept the water table now are found as the numerous kettle-hole ponds throughout the region (USGS, 2009-5063). These surfacewater features (kettle holes, bogs, wetlands) form the headwaters of the Jones River, Town Brook and the Eel River, all conduits for direct discharge of freshwater and associated nutrient loads to the estuarine waters.

The basins (Plymouth Harbor, Kingston Bay and Duxbury Bay) comprising the PKD Embayment System were formed by coastal processes that created a barrier beach along the open basin front to Cape Cod Bay. The barrier beach system is comprised of a northern arm, Duxbury Beach extending southward towards the inlet and terminating in Saquish Neck and Saquish Head and a southern arm, Plymouth Beach that extends from the shoreline northward to the inlet and encloses Plymouth Harbor into which the Eel River and Town Brook discharge. These basins are properly termed lagoons (e.g. lagoonal estuarine basins) and run parallel to the coast behind a sandy barrier beach/island. The formation and structure of the Plymouth-Kingston portion of the overall embayment system parallels that of its large neighboring Cape Cod Bay estuaries on Cape Cod, Barnstable Harbor, Wellfleet Harbor and Provincetown Harbor.

Coastal processes formed the Plymouth-Kingston-Duxbury Bay Embayment System and continue to modify its structure. Unlike drown river valley estuaries, the PKD Embayment System is a lagoonal estuary formed solely by the development of the barrier beaches which form the semi-enclosed basin where fresh and salt waters mix (e.g. an estuary). The function of the overall system is further affected by the degree of infilling of the inlet as well as the annual changes in the numerous tidal channels that exist among the shifting sand flats in the more quiescent upper reaches of the estuary. The ecological and biogeochemical structure of the embayment system is likely to have changed over time as the barrier beach has migrated, breached and closed as a function of storm frequency, intensity and sand supply. As such, it is critical that nutrient management within the watershed to the estuary is conducted in the context of the hydrodynamic characteristics of the system, inclusive of its large tidal range.

The primary ecological threat to the Plymouth-Kingston-Duxbury Embayment System as a coastal resource is degradation resulting from nutrient enrichment. Nutrient enrichment generally occurs through increases in watershed nitrogen loading resulting from changing land uses (typically conversion of pine/oak forest to residential development) and/or reduced tidal exchanges with offshore waters. Although it is possible that portions of Plymouth-Kingston-Duxbury System (particularly the smaller coves) can have periodic issues relative to bacterial contamination primarily within the most enclosed regions of each, fecal coliform contamination does not generally result in ecological impacts, rather it is associated with public health concerns related with consumption of potentially contaminated shellfish or contact recreation. The primary impact of bacterial contamination is the closure of shellfish harvest areas (and swimming beaches), rather than the destruction of shellfish and other marine habitats. In contrast, increased loading of the critical eutrophying nutrient (nitrogen) to the Plymouth-Kingston-Duxbury System results in both habitat impairment and loss of the resources themselves. Within the watershed of this complex estuarine system, nitrogen loading has been increasing as land-uses have changed over the past 60 years. The nitrogen loading to this system, like almost all embayments in southeastern Massachusetts and the Islands, results primarily from on-site disposal of wastewater (direct wastewater treatment plant discharge in the case of PKD Bay), agriculture (animal and plant) and fertilizer applications (residential and agricultural), and to a lesser extent stormwater discharges. Nitrogen enrichment of all coastal embayments and restoration of nitrogen impaired habitats can only be managed through lowering inputs or increasing the rate of loss through tidal flushing. This is discussed in detail in Sections IV.1 and VI.

The Towns of the Massachusetts south coast (Plymouth, Kingston, Duxbury) have been among the fastest growing towns in the Commonwealth over the past three decades and unlike many of the towns of southeastern Massachusetts and Cape Cod, the Town of Plymouth does operate a centralized wastewater treatment system with the site of discharge of its treated effluent being located in the Plymouth-Kingston-Duxbury embayment system, with a secondary discharge within the Eel River sub-watershed. Other towns in the watershed, such as the Towns of Kingston and Duxbury, do not have similar wastewater collection and treatment facilities servicing any portion of their watershed areas. Rather, treatment of wastewater within these areas of the watershed is by privately maintained on-site septic systems for treatment and disposal of wastewater. As existing and likely increasing levels of nutrients impact the coastal embayments of the Towns of south coastal Massachusetts, water quality degradation will accelerate, with further harm to valuable aquatic resources of the region.

As the primary stakeholders to the Plymouth-Kingston-Duxbury embayment system, the Towns of Plymouth, Kingston and Duxbury have been among the first communities in southeastern Massachusetts to become concerned over perceived degradation of their coastal embayments, particularly the Town of Plymouth that operates the WWTF that discharges to the estuary. Over the years, this local concern has led to the conduct of several studies (see Section II) of nitrogen loading to this large estuary such as those under taken by Camp Dresser and McKee, Inc. relative to the WWTF. Key in this effort has been the Water Quality Monitoring Program that was initiated by the Town of Plymouth under the 604b grant program with technical assistance by the Coastal Systems Program at SMAST-UMD. This effort provided the quantitative water column nitrogen data (2003, 2004, 2005, 2007 and 2013) required for the implementation of the MEP's Linked Watershed-Embayment Approach used in the present study.

Since the initial results of the historic Water Quality Monitoring Program indicated that parts of the Plymouth-Kingston-Duxbury system were showing signs of nutrient related impairment and reduced water quality, presumably due to elevated land-derived nitrogen inputs. A private entity, Plymouth Rock Studios, also provided support for a detailed land-use and nutrient loading analysis of the Eel River System, following on work on Eel River Water Quality conducted by Camp Dresser and McKee, Inc. and freshwater monitoring by Horsley and Witten Inc (for Pine Hills). Appropriate data from these studies was incorporated into the MEP assessment and modeling effort of the PDK Embayment System[1]. The common focus of the historic work related to the Plymouth-Kingston-Duxbury System has been to gather site-specific data on the current nitrogen related water quality throughout the estuary and determine its relationship to watershed nitrogen loads. The multi-year water quality monitoring effort has provided the baseline information required for calibrating and verifying the water quality model linking watershed nitrogen loading, tidal flushing, and estuarine water quality. The MEP effort builds upon the Water Quality Monitoring Program results and includes higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the restoration of this embayment system. These critical nitrogen threshold levels and the link to specific ecological criteria form the quantitative basis for the nitrogen loading targets necessary for nitrogen management plans and the development of cost-effective alternatives for protection/restoration of habitat impaired by nitrogen enrichment needed by the Towns.

[1] Howes, B.L., G. Mercer, and D.D. Goehringer. 2000 Evaluation of Nutrient Inputs and the Health of the Eel River System, Plymouth MA, in Support of a Nutrient Management Plan. Technical Report to Massachusetts Department of Environmental Protection. 90pp.

While the completion of this complex multi-step process of rigorous site-specific scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of a large number of Town staff and volunteers over many years. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns of Plymouth, Kingston and Duxbury and the other towns in the watershed to develop and evaluate the most cost effective nitrogen management alternatives to protect / restore this valuable coastal resource which is currently being gradually degraded by nitrogen overloading. It is important to note that the Plymouth-Kingston-Duxbury Embayment System and its associated watershed have been altered by human activities over the past ~400 years. As a result, the present nitrogen "overloading" appears to result partly from alterations to its ecological systems. These alterations subsequently diminish nitrogen retention processes within the watershed and influence the degree to which nitrogen loads impact the estuary. Therefore, protection / restoration of this system should focus on managing nitrogen through both management of nitrogen loading within the watershed, restoration/management of processes which serve to lessen the amount or impact of nitrogen entering the estuary (enhanced natural attenuation) and channel maintenance to maximize the rate of nitrogen removal from the estuary via tidal flushing.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

Coastal embayments throughout the Commonwealth of Massachusetts (and along the U.S. eastern seaboard) are becoming increasingly 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 their ability to assimilate additional nutrient inputs without decline in their ecological health. The result is the loss of fisheries habitat, eelgrass beds, and a general disruption of benthic communities and the food chain which they support. At higher levels, nitrogen loading from surrounding watersheds causes aesthetic degradation and inhibits even recreational uses of coastal waters. In addition to nutrient related ecological declines, an increasing number of embayments are being closed to swimming, shellfishing and other activities as a result of bacterial contamination. While bacterial contamination does not generally degrade the habitat, it restricts human uses. However like nutrients, bacterial contamination is frequently related to changes in land-use as watersheds become more developed. The regional effects of both nutrient loading and bacterial contamination span the spectrum from environmental to socio-economic impacts and have direct consequences to the culture, economy, and tax base of Massachusetts's coastal communities.

The primary nutrient causing the increasing impairment of the Commonwealth's coastal embayments is nitrogen and the primary sources of this nitrogen are wastewater disposal, fertilizers, and changes in the freshwater hydrology associated with development. At present there is a critical need for state-of-the-art approaches for evaluating and restoring nitrogen sensitive and impaired embayments. Within Southeastern Massachusetts and the Islands, almost all of the municipalities (as is the case with the Towns of Plymouth, Kingston and Duxbury, as well as "up-gradient" Towns in the watershed) are grappling with Comprehensive Wastewater Planning and/or environmental management issues related to the declining health of their estuaries or lakes/ponds, typically resulting from nutrient over-enrichment.

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 or nitrogen attenuation in watershed transport. 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 in a manner to support planning. Although it may be technically complex to implement, results must be understandable to the regulatory community, town officials, and the general public.

The Massachusetts Estuaries Project represents the next generation of watershed based nitrogen management approaches. The Massachusetts Department of Environmental Protection (MassDEP), the University of Massachusetts – Dartmouth School of Marine Science and Technology (SMAST), and others including the Martha's Vineyard Commission (MVC), the Southeastern Regional Planning and Economic Development District (SRPEDD) and the Cape Cod Commission (CCC) have undertaken the task of providing a quantitative tool for watershed-embayment management for communities throughout Southeastern Massachusetts and the Islands.

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 MassDEP 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) for those estuarine systems that are presently impaired by nitrogen enrichment or which will become impaired as build-out of their watershed continues. 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 MassDEP 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, MassDEP 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.

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

The major MEP nitrogen management goals are to:

- provide technical analysis and supporting documentation to Towns as a basis for sound nutrient management decision making towards embayment restoration
- develop a coastal TMDL working group for coordination and rapid transfer of results,
- determine the nutrient related health and nutrient sensitivity of each of the embayments in southeastern Massachusetts
- 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 ~70 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 their associated cost. In addition, once a model is fully functional it can be "kept alive" and updated for continuing changes in land-use or embayment characteristics (at minimal cost). In addition, since the Model uses a holistic approach (the entire watershed, embayment and tidal source waters), it can be used to evaluate all projects as they relate directly or indirectly to water quality conditions within its geographic boundaries.

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

- Water column 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 attenuated 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

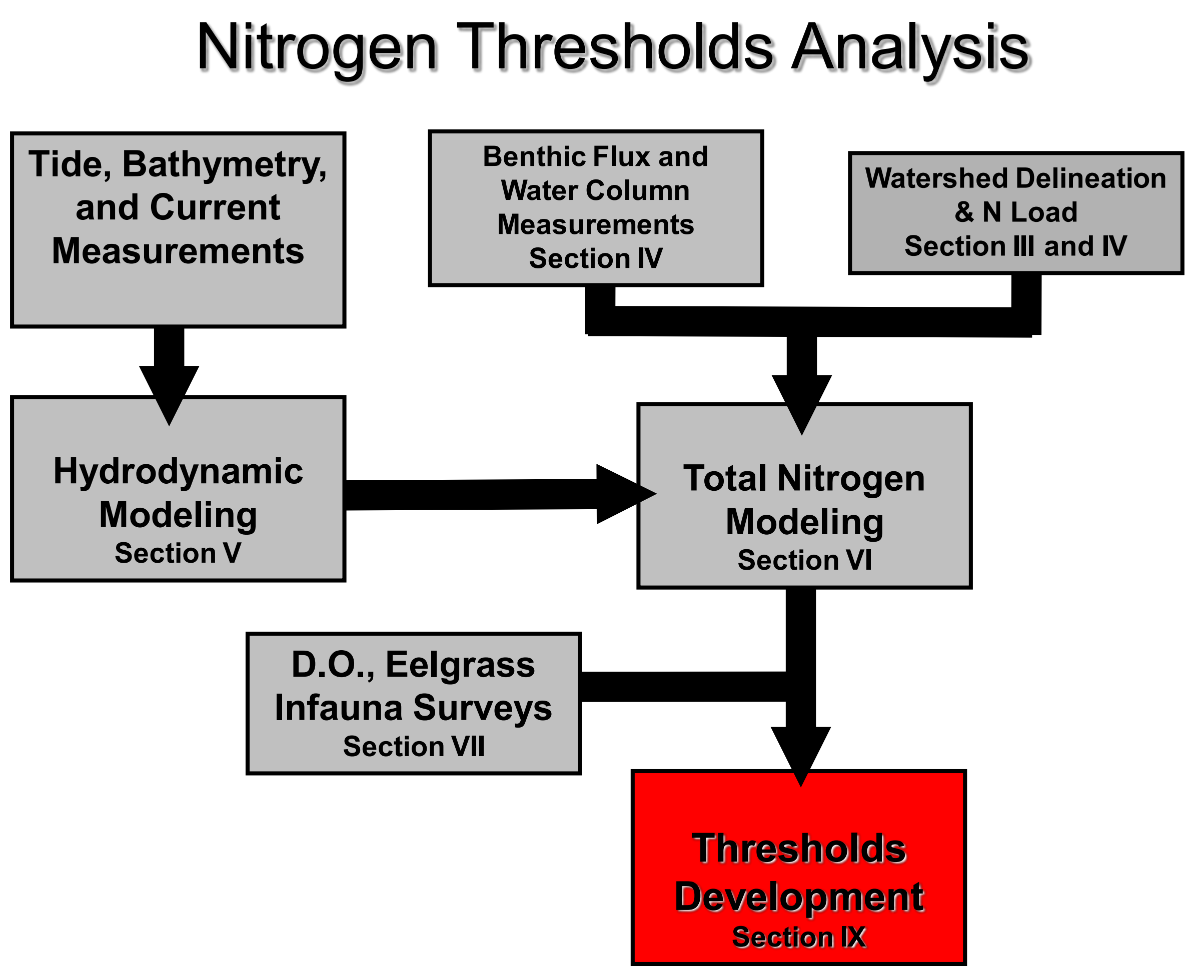


Figure I-3. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach.

I.2 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 large areas of the southern watershed to the Plymouth-Kingston-Duxbury embayment system, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer minerals (Weiskel and Howes, 1992). Since even south coastal rivers as well as those situated on the Islands and 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) and Martha's Vineyard. 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). The estuarine reaches within the Plymouth-Kingston-Duxbury Embayment System follow this general pattern, with an average Redfield Ratio (N/P) of only 3 (<<16) and with total dissolved inorganic nitrogen levels quite low (mean 1.74 uM or 0.024 mg/L) indicating that addition of nitrogen would have a stimulatory effect on plant production.

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

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

Protection and restoration of coastal embayments from nitrogen overloading has resulted in a focus on determining the assimilative capacity of these aquatic systems for nitrogen. While this effort is ongoing (e.g. USEPA TMDL studies), southeastern Massachusetts and the Islands 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, MVC Water Quality Policy and the present Massachusetts Estuaries Project). 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 and its effects on water quality. 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 in the 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 Plymouth-Kingston-Duxbury embayment system monitored by the collaboration between the Town of Plymouth and the UMD-SMAST Coastal Systems Program. The Water Quality Monitoring Program along with site-specific habitat quality data collected by the MEP technical team (D.O., eelgrass, phytoplankton blooms, benthic animals) was utilized to refine general nitrogen thresholds typically used by the Cape Cod Commission, Buzzards Bay Project, and Massachusetts State Regulatory Agencies.

A number of estuarine reaches within the upper portions of the Plymouth-Kingston-Duxbury Embayment System are approaching or slightly beyond their ability to assimilate additional nutrients without impacting their ecological health. In these upper regions and especially upper Duxbury Bay, nitrogen levels are elevated and impairment to eelgrass habitat is becoming evident. Eelgrass coverage within this basin has been declining (1995 to 2012) as indicated by the MassDEP Eelgrass Mapping Program and as confirmed by the MEP Technical Team during the summer and fall of 2007. In addition, nitrogen related habitat impairment within the Plymouth-Kingston-Duxbury system is consistent with the nitrogen levels as well as biologic indicators of habitat health (e.g. benthic infauna samples collected in fall 2007 and 2013. The result is that nitrogen management of the primary sub-embayments to the Plymouth-Kingston-Duxbury system is aimed at restoration of upper regions and protection/maintenance of habitat quality in lower basins,

In general, nutrient over-fertilization is termed “eutrophication” and in certain instances can occur naturally over long periods of time. When the nutrient loading is rapid and primarily from human activities leading to changes in a coastal watershed, nutrient enrichment of coastal waters is termed “cultural eutrophication”. Although the influence of human-induced changes has increased nitrogen loading to this embayment system and contributed to its low-moderate decline in ecological health, the Plymouth-Kingston-Duxbury Embayment, like others analyzed by the MEP such as Barnstable Harbor and Wellfleet Harbor, are especially sensitive to nitrogen inputs in their upper tidal reaches, due to increases in development within the watershed, though the sensitivity is much less in systems with large tidal ranges such as those on Cape Cod Bay). The quantitative role of natural attenuation of watershed derived nutrient loads, changes in circulation from tidal channel constrictions or tidal damping in this system, all as natural processes, were also assessed in the MEP nutrient threshold analysis. As part of future restoration efforts, it is important to understand that it may not be possible to turn each portion of the embayment into a “pristine” system as certain impaired areas may be that way for natural reasons.

I.3 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 Plymouth-Kingston-Duxbury Embayment System; however, a thorough understanding of estuarine circulation is required to accurately determine nitrogen concentrations within each component of the overall 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 an accurate 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 Plymouth-Kingston-Duxbury Embayment System, including the uppermost reaches of Duxbury Bay up-gradient of the Powder Point Bridge linking the Duxbury barrier beach to the mainland. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents at the Plymouth Harbor inlet and water elevations was employed for the system. Once the hydrodynamic properties of each estuarine basin were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates and under present circulation patterns.

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 nitrogen inputs and hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis and based upon USGS/MEP refined watershed and subwatershed delineations. The delineations were developed relative to: 1) an updated version of the Plymouth-Carver, Kingston, Duxbury Aquifer Groundwater Model completed in 2009 (Masterson et. al. 2009) and 2) water table contours measured in specific locations across the aquifer domain and 3) USGS topographic maps as appropriate. Almost all nitrogen entering the Plymouth-Kingston-Duxbury Embayment System is transported by freshwater, predominantly groundwater, with the exception of flows entering the system from the Jones River, Eel River and Town Brook. Concentrations of total nitrogen and salinity of Cape Cod Bay source waters and throughout the Plymouth-Kingston-Duxbury Embayment System were taken from the Water Quality Monitoring Program (a coordinated effort between the Towns of Plymouth, Kingston, Duxbury and the Coastal Systems Program at SMAST {604b funded}). Measurements of the distribution of nitrogen and salinity throughout the estuarine waters of the system (2003, 2004, 2005, 2007, 2013) were used to calibrate and validate the water quality model (under existing loading conditions).

I.4 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Plymouth-Kingston-Duxbury Embayment System for the Towns of Plymouth, Kingston, Duxbury, Halifax, Marshfield, Pembroke and Plympton comprising its watershed. 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.3). Nitrogen loads from the watershed and sub-watersheds surrounding the estuary were derived from the Towns and SRPEDD data and offshore water column nitrogen values were derived from an analysis of monitoring stations in Cape Cod Bay (Section IV and VI respectively). Intrinsic to the calibration and validation of the linked watershed-embayment modeling approach is the collection of background water quality monitoring data (typically conducted by municipalities) as discussed in Section VI. 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 eelgrass surveys and benthic community analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed in Section VIII for protection/restoration of the embayment system. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined threshold for restoration of the estuary. This latter assessment represents only one of many solutions and is produced to assist the Towns in developing a variety of alternative nitrogen management options for this system. Finally, additional analyses of the Plymouth-Kingston-Duxbury Bay System, beyond the standard suite offered by the MEP, are presented in Section IX as requested by the Town of Plymouth.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

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

In most marine and estuarine systems, such as the Plymouth Harbor-Kingston Bay-Duxbury 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. As a result, there has been significant effort to develop tools for predicting how modification of watershed nitrogen loads and changes in tidal flushing quantitatively cause changes in the concentrations of water column nitrogen in the receiving estuary. Further development of these approaches generated specific guidelines as to what is to be considered acceptable water column nitrogen concentrations to achieve desired water quality goals (e.g., see Cape Cod Commission 1991, 1998; Howes et al. 2002).

These tools for predicting loads and concentrations tend to be generic in nature, and overlook some of the specifics for any given water body. In contrast, some approaches can be tailored for each individual estuary of interest, but require large amounts of site-specific information, which increases accuracy but also costs and therefore they are not widely applied. The present Massachusetts Estuaries Project (MEP) effort uses one such site-specific approach. The assessment 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 within individual estuaries. The linked watershed-embayment model is built using embayment specific measurements, thus enabling calibration of the predictions for the specific conditions in each of the estuaries of southeastern Massachusetts, including the Plymouth-Kingston-Duxbury Bay Estuary. As the MEP approach requires substantial amounts of site-specific data collection, part of the program is to review previous data collection and modeling efforts. These reviews are both for purposes of “data mining” and to gather additional information on an estuary’s habitat quality and unique features.

Several studies relating primarily to river restoration efforts, watershed characteristics and to nitrogen loading associated with the Town of Plymouth Wastewater Treatment Facility have been conducted over the past two decades which were examined in an effort to inform the MEP process. Pertinent historical work along with quantitative information on estuarine water column parameters over multiple summers (including nitrogen) has helped advance the MEP effort in regard to the Plymouth-Kingston-Duxbury Bay Embayment System. These studies are summarized below.

Evaluation of Nutrient Inputs and the Health of the Eel River System, Plymouth MA, in Support of a Nutrient Management Plan. Technical Report to Massachusetts Department of Environmental Protection. (Howes, B.L., G. Mercer, and D.D. Goehringer, 90pp., 2000): The Town of Plymouth was planning to construct a new WWTF with on-site disposal within the Eel River Watershed. SMAST scientists were tasked by MassDEP with establishing a stakeholder group which included the Town of Plymouth's wastewater consultants, consultants for the new Pine Hills development as well as NGO's and other stewards of the Eel River Watershed. The purpose was to develop a management strategy for the Eel River focusing on the increasing nitrogen and phosphorus loads and their effects. The effort also reviewed all relevant data, identified data gaps and designed additional sampling to fill the gaps. The plan was immediately implemented so the effort had the necessary data for making site-specific management decisions. The overall result was the finding that the multiple ponds associated with the Eel River were removing nitrogen (natural attenuation) from river water and that the system could not be managed for nitrogen, as nitrogen levels had been increasing and would certainly increase further even if nitrogen management actions were implemented. Instead, it was determined that the Eel River System was limited by phosphorus and given the projected nitrogen potential increases, phosphorus needed to be prevented through rigorous phosphorus management. To this end the Town of Plymouth took steps to prevent phosphorus from its new WWTF from entering the freshwater system and subsequently a cranberry bog (identified by the monitoring program) was taken out of production and restored by the Eel River Headwaters Restoration Project (see below).

Basin-Scale Modeling of Nutrient Impacts in the Eel River Watershed, Plymouth, Massachusetts (MIT Master's Thesis 2002): In this Master's Thesis, a surface water hydrologic model was developed to assess nutrient impacts in the Eel River Watershed. The study was performed on behalf of the Eel River Watershed Association, in response to specific concerns regarding eutrophication in the watershed. These concerns are focused on increased nutrient loading caused by the construction of the Waste Water Treatment Facility, as well as increased development (i.e., golf courses and residential).

The degree candidate selected the surface water model HSPF for its comprehensive hydrologic simulation capabilities, deemed necessary for the heterogeneous, baseflow-dominated nature of the watershed. Hydrologic calibration of the model was successful in replicating observed stream flow to the resolution of the daily storm hydrograph. Detailed total nitrogen and total phosphorous loading estimates were then calculated to provide a screening tool for the extent of nutrient impacts. The results indicate increased loads of 167% and 171% for total nitrogen and total phosphorous, respectively, in the watershed.

These loading estimates were integrated within the HSPF hydrologic model via the build-up / wash-off algorithms. Nitrate transport was simulated under baseline conditions, with modeled results indicating a strong correlation to measured concentrations. Forecasting of the impacts from the WWTF effluent discharge was performed. According to the author, a significant localized increase in nitrate concentrations on the order of 15-20% was modeled, indicating the potential for increased eutrophication in associated water bodies. This finding, however, was not supported by Eel River studies indicating that phosphorus was the nutrient of management concern within the freshwater Eel River system, not nitrogen. However, the projections of increases in nitrogen to the Eel River were consistent with most other assessments

Within the thesis historic streamflow values for the Eel River were reported as determined by both the USGS and Camp, Dresser & McKee, Inc (above study). at the same location as the MEP gauge deployed in 2003-2004. MEP flows compared well to the flows determined by the USGS (1969-71) and CDM (1998-2000) and are further described in Section IV-2.

Town of Plymouth Eel River Headwaters Restoration Project (2009-2010) - As summarized by the Town of Plymouth, the Eel River Headwaters Restoration Project transformed approximately 60-acres of formerly commercial cranberry farm into self-sustaining freshwater wetlands. A broad range of stakeholders supported the multi-year effort driven by the Town of Plymouth to permanently protect the land, design and implement restoration activities, and provide public access and recreational opportunities.

Through a coordinated series of restoration actions, the project addressed a number of stressors limiting ecological potential, including: barriers to fish and wildlife passage, altered hydrology and degraded wetland soils (buried under a century of farm-applied sand), and simplified channel and floodplain structure. While not specifically designed or undertaken to address nutrient management in the watershed, this sort of restoration project is likely to stimulate enhanced natural attenuation of nutrients being transported to the Plymouth Harbor portion of the estuary and certainly removes fertilization related losses to the river system. As such is a clear sign of the Town's commitment to improving water quality in its freshwater and estuarine resources.

Specific actions taken under this project included: Removal of the Sawmill Pond Dam (a complete barrier to upstream fish migration); replacement of two undersized culverts and removal of a third; removal of seven water control structures (essentially small dams); re-construction of 1.7 miles of stream channel and floodplain; installation of 1,000+ pieces of large wood, removal of 48,000 cubic yards of fill, and installation of approximately 20,000 plants, including 17,000+ Atlantic white cedar trees. Monitoring, evaluation, and adaptive management are ongoing and lessons learned from this restoration of a former cranberry farm can provide guidance to other towns in the MEP study region that could benefit from this approach to nutrient management and is an approach supported by the MEP in appropriate watershed.

***Eel River Land-use and Nutrient Loading Analysis to Support Green Development - Plymouth Rock Studios (PRS) 2010*:** The Plymouth Rock Studio was planning a green development project that would minimize inputs of nitrogen and phosphorous to, and alteration of, freshwater flow within the Eel River System. As part of this effort, detailed information and quantitative modeling of nitrogen and inputs from PRS relative to other sources within the watershed was assessed, as well as potential freshwater flow alterations within the Eel River. The approach followed the SMAST Technical Team's watershed analyses for the region-wide Massachusetts Estuaries Project. However, the phosphorus and freshwater flow analysis represented extra tasks that are not part of the standard MEP Nutrient Threshold Report.

The watershed analysis used a detailed, watershed-specific land-use loading model which is generally similar in form to traditional approaches by the Cape Cod Commission and the Buzzards Bay Project, but which was upgraded to include parcel-specific data from the Town of Plymouth Assessors database as well as utilizing Town water use records. Quantification of the total nitrogen and phosphorous input to the Eel River system was based on watershed land use analysis with direct input from the Town of Plymouth Water and Sewer Department and Planning Department. This analysis was used to project current and future (build-out) nitrogen and phosphorus loads.

The watershed to the overall Eel River system was divided into regions: a) contributing directly via groundwater to the river, b) contributing to freshwater lakes and ponds, and c) contributing to freshwater wetlands. A full land-use nutrient loading model was applied to each of these sub-watersheds (where they exist) to determine the spatial distribution and amount of nitrogen and phosphorous loading to the Eel River system and the down-gradient embayment system. In this regard it should be noted that not all nitrogen entering the watershed was determined to reach Plymouth Harbor, this "natural attenuation" was accounted for as possible from available data. Again, all sources entering the watershed will be examined to understand the potential role of each source.

Watershed Action Alliance of Southeastern Massachusetts - South Coastal Watershed Action Plan, 2006: This action plan developed in 2006 as a 5-year plan for all the coastal watersheds of southeastern Massachusetts and was written to guide local, state, and federal environmental efforts within the watersheds of this region from 2007-2011. The plan summarizes the environmental concerns in the region and potential actions to improve the environmental health of the watershed, such as improving water quality, restoring natural flows to rivers, protecting and restoring biodiversity and habitats, improving public access and balanced resource use, improving local capacity, and promoting a shared responsibility for watershed protection and management.

The South Coastal Watershed Action Plan was developed with input from a diverse cross-section of stakeholders including watershed groups, state and federal agencies, municipal officials, Regional Planning Agencies and the public from across the Watershed. The project was the collaboration of Jones River Watershed Association, North and South River Watershed Association, Eel River Watershed Association, Pembroke Watershed Association and the Six Ponds Watershed Association with technical support from a private sector firm, Lenehan Consulting.

Several priorities were articulated in the 2006 Plan and the MEP analysis of the Plymouth-Kingston-Duxbury Bay Embayment System and its associated coastal watershed helps to address some of the plans following priorities:

- Improve water quality by addressing point and non-point sources of pollution
- Protect and restore aquatic habitat
- Protect and Restore the natural hydrology
- Strengthen local capacity to protect and enjoy the South Coastal Watersheds

In the context of the above mentioned priorities, the action plan outlines the necessary steps that residents, watershed associations, businesses and state and municipal officials must take to manage sustainable growth. The action plan called for:

- Bylaw changes to address stormwater impacts on estuaries and rivers.
- Water supply and waste water planning and management to be coordinated at the local level and in the context of what each watershed can sustain without damaging sensitive aquatic habitats.
- Fish passage to be restored and unnecessary obstruction and flow diversion be removed to restore the natural flow to our rivers and streams.

In the action plan the steps are outlined for the South Coastal region as a whole, and more specifically in separate chapters for six sub areas including: the Gulf River and Scituate Harbor Watersheds, the North River Watersheds, the Indian Head River Watersheds, the South

River and Green Harbor River Watersheds, the Jones River and Duxbury Bay Watersheds, and the Plymouth Watersheds.

Town of Plymouth Nutrient Management Data Report - Operational Monitoring Program (WWTF): As part of the Massachusetts Department of Environmental Protection (DEP) approval of Plymouth's Waste Water Treatment Facility (WWTF) Permit, SE# 1-677, a Nutrient Management Plan (NMP) was developed and put in place by the Town's consultant (Camp, Dresser and McKee, Inc.) with technical support from SMAST (see above) and approved by the MassDEP in 2001. As part of the WWTF Permit the NMP consists of surface and groundwater monitoring within the Eel River Watershed in addition to the monitoring required by WWTF plant operations. Included in the monitoring network of stations are two in estuary stations located in the southern most portion of Plymouth Harbor close to the mouth of the Eel River. The NMP monitoring program consists of three parts; the baseline monitoring which occurred from May 1998 through February 2000; the interim monitoring which occurred from May 2000 through November 2001; the operational monitoring which began following the operations of the WWTF in May 2002 and continues to date providing valuable information on the nutrient related water quality of the effluent being discharged by the WWTF.

At a most basic level, the NMP presents a methodology for monitoring changes in the Eel River system and provides specific action levels based on changes in water quality parameters. In addition to the monitoring, the NMP consists of controls and practices, known as the Base Management Plan, which the Town has and will continue to implement to reduce existing nutrient loads to the River and/or help minimize any future increases. As presented in the most recent operational report (2013-2014), the surface water and groundwater monitoring conducted in the Eel River Watershed does not indicate impact from the Wastewater Treatment Facility nor were there any major environmental concerns in 2013-2014. As with 2012, the total nitrogen concentrations discharged into the infiltration basins in 2013 and 2014 are almost half the DEP permitted level of 10 mg/L. The flow to the infiltration basins are still at approximately 15% of the permitted flow of 0.75MGD. Surface water and groundwater monitored under the NMP showed a decreasing trend for both total nitrogen and total phosphorus.

This program is a critical component to overall nutrient management in the watershed to Plymouth Harbor and will be valuable to the Town of Plymouth as it moves towards future implementation strategies to achieve the MEP nutrient threshold for the system.

Plymouth-Kingston-Duxbury Bay Nutrient Related Water Quality Monitoring: The MEP analysis requires accurate and spatially distributed water quality data in order to complete its assessment and modeling approach. The Town of Plymouth Water Quality Monitoring Program (supported by the Massachusetts 604(b) grant program) collected data on nutrient related water quality throughout the Plymouth-Kingston-Duxbury Embayment System. The Town of Plymouth Water Quality Monitoring Program collected the principal baseline water quality data necessary for ecological management of each of the component basins associated with each Town.

The water quality monitoring project goal was to collect and analyze water samples and associated field parameters relevant to the nutrient related water quality of the Plymouth Harbor-Kingston Bay-Duxbury Bay System and adjacent Ellisville Harbor. This water quality monitoring effort was a collaborative effort between the Towns of Kingston, Duxbury, and Plymouth whereby each Town fielded a water sampling team trained and coordinated by University of Massachusetts – Dartmouth, School of Marine Science and Technology (SMAST), Coastal Systems Laboratory Staff. Each water sampling team was responsible for collection of water samples at assigned sampling stations with logistical support from SMAST. Personnel from the Coastal Systems

Program within SMAST were also involved in the field sampling in order to assist in the collection of samples and insure proper transport and delivery of samples to the Coastal Systems Analytical Facility where chemical assays were performed.

The water quality data collected by the combined efforts of each Town's sampling team was required for application of the Linked Watershed-Embayment Approach of the Massachusetts Estuaries Project (MEP). All embayments undergoing MEP analysis require a minimum of three years of high-quality water chemistry and field data related to nitrogen dynamics. Although there was some existing water quality data prior to the MEP that has been considered in this MEP analysis, a complete water quality monitoring effort had to be implemented in order to satisfy the full water quality monitoring data requirements of the MEP and produce a unified picture of water quality conditions in the embayment. In order to initiate the needed data collection for the Plymouth Harbor/Duxbury Harbor/Kingston Bay System, and Ellisville Harbor to support entry into the Estuaries Project and thereby allow full evaluation of protective measures, the Towns received MassDEP 604(b) funding support for collection, processing and analyses of water samples from the overall embayment system. The Town of Plymouth singularly funded additional years of water quality data collection beyond the years funded by the 604b grant program in order to develop as robust a water quality base line as possible for MEP modeling. A summary report was completed per the requirements of the 604b grant program and is a publically available document through the MassDEP.

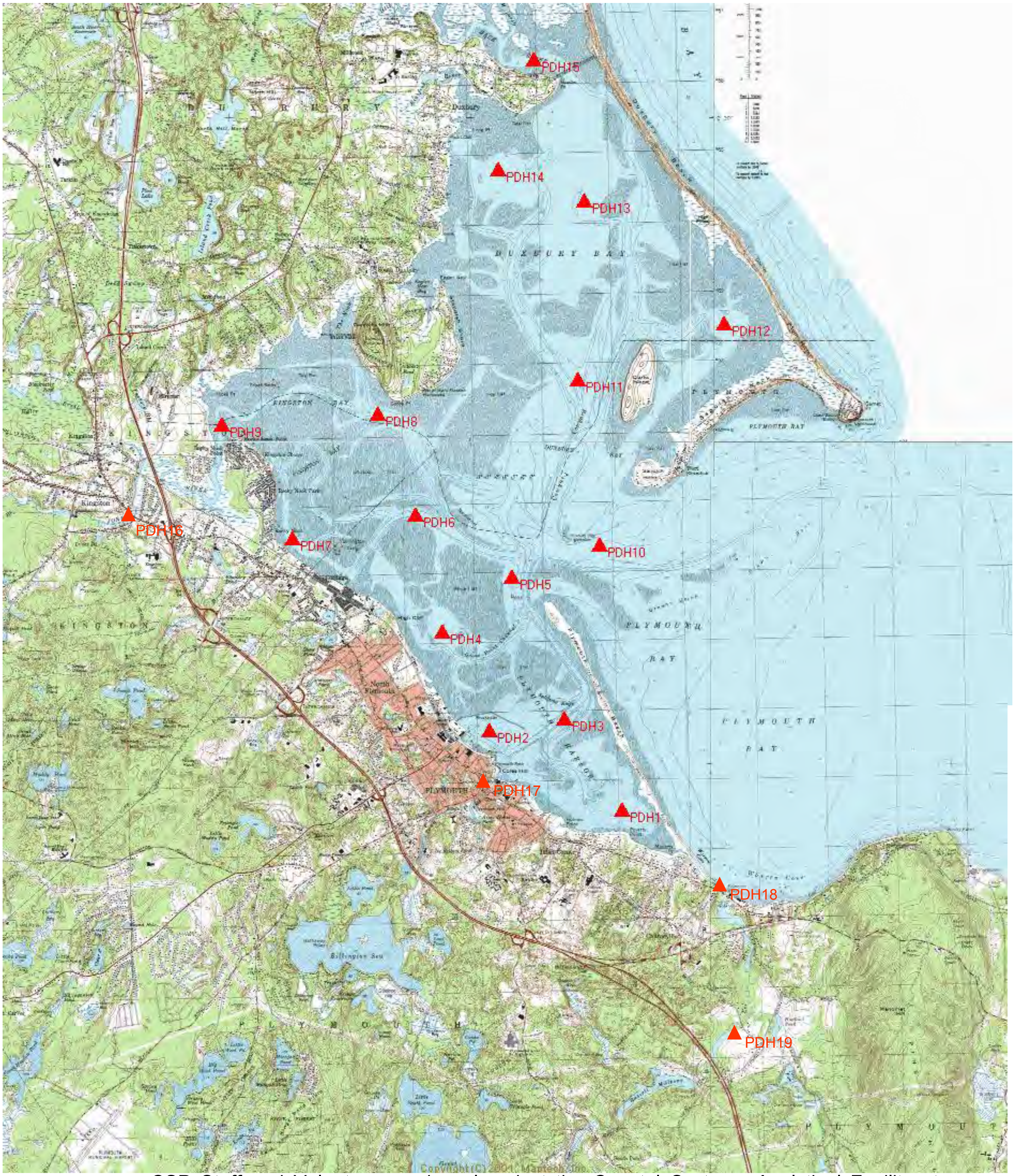
It should be noted that in addition to baseline water quality monitoring undertaken in support of the MEP water quality modeling effort, additional water quality monitoring has been undertaken by the Provincetown Center for Coastal Studies (PCCS) in Cape Cod Bay. Coastal and offshore stations in this monitoring program range from Provincetown Harbor to the Plymouth-Kingston-Duxbury Bay estuary system. Within the PKD estuary system, PCCS monitors approximately 8 stations generally clustered around the freshwater discharges such as the Eel River and Jones River as well as two coastal stations in the upper portion of Duxbury Bay. Results from the PCCS "How's Our Bay" Report (2012) that summarizes 5-years of monitoring were considered in the review of historical water quality data.

Since the results of the Town of Plymouth's long term Water Quality Monitoring Program and the above studies indicate that portions of the Plymouth-Kingston-Duxbury Embayment System could be threatened by the combination of land-derived nitrogen inputs and possible intermittent restriction of tidal exchange from shifting tidal channels, the Town of Plymouth undertook participation in the Massachusetts Estuaries Project. The goal of the MEP is to complete ecological assessment and water quality modeling to develop nutrient thresholds for the Plymouth-Kingston-Duxbury Embayment System for protection of it high quality habitats and restoration of habitats which have become impaired by increased nitrogen loading. It should be noted that the Town of Plymouth generated all the municipal match for completion of the MEP analysis.

Regulatory Assessments of Plymouth-Kingston-Duxbury Bay Resources - In addition to locally generated studies, the Plymouth-Kingston-Duxbury Embayment System is part of the Commonwealth's environmental surveys to support regulatory needs. The overall estuary contains a variety of natural resources of value to the citizens of the south coast region as well as to the Commonwealth. As such, over the years surveys have been conducted to support protection and management of these resources. The MEP also gathers the available information on these resources as part of its assessment, and presents some of them here for reference by those providing stewardship for this estuary and some in Chapter 7 to support the nitrogen thresholds analysis. For the Plymouth-Kingston-Duxbury Estuary these include:

- Designated Shellfish Growing Area – MassDMF (Figure II-2a,b,c,d,e)
- Shellfish Suitability Areas – MassDMF (Figure II-3a,b)
- Anadromous Fish Runs - MassDMF (Figure II-4a,b)
- Priority Habitats for Rare Wildlife and State Protected Rare Species – NHESP (Figure II-5)
- Mouth of Coastal Rivers – MassDEP Wetlands Program (Figure II-6a,b,c,d,e,f,g,h,i,j)

The MEP effort builds upon earlier watershed delineation and land-use analyses, the hydrodynamic modeling, historical eelgrass surveys and water quality surveys discussed above. This information is integrated with MEP higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the overall estuarine system. The MEP has incorporated appropriate and available data from pertinent previous studies to enhance the determination of nitrogen thresholds for the Plymouth-Kingston-Duxbury Embayment System and to reduce costs to the Towns of Plymouth, Kingston and Duxbury as well as other associated up-gradient Towns that constitute the overall watershed.



CSP Staff and Volunteers and analyzed at the Coastal Systems Analytical Facility at SMAST during summers 2003, 2004, 2005 (604b) and 2007 and 2013 (Town of Plymouth).

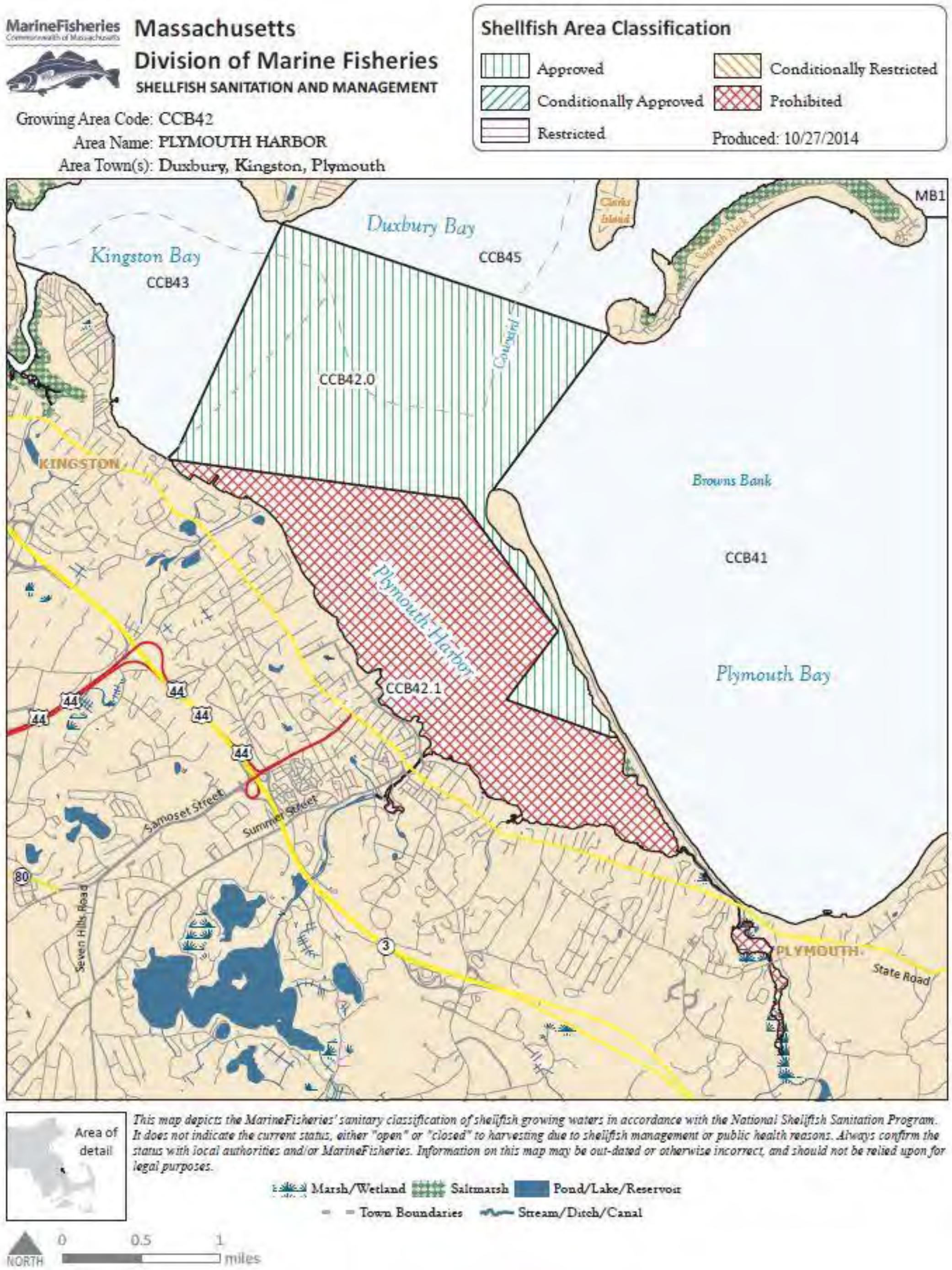


Figure II-2a. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination. The prohibited areas are related to active marina areas such as the downtown Plymouth area or smaller marsh tributary creeks that receive most of the freshwater inflow to the system (Eel River mouth). Wetland areas with persistent fecal coliform levels >14 cfu per 100 mL may be prohibited to shellfishing until the source of contamination (frequently wildlife & birds) is documented.

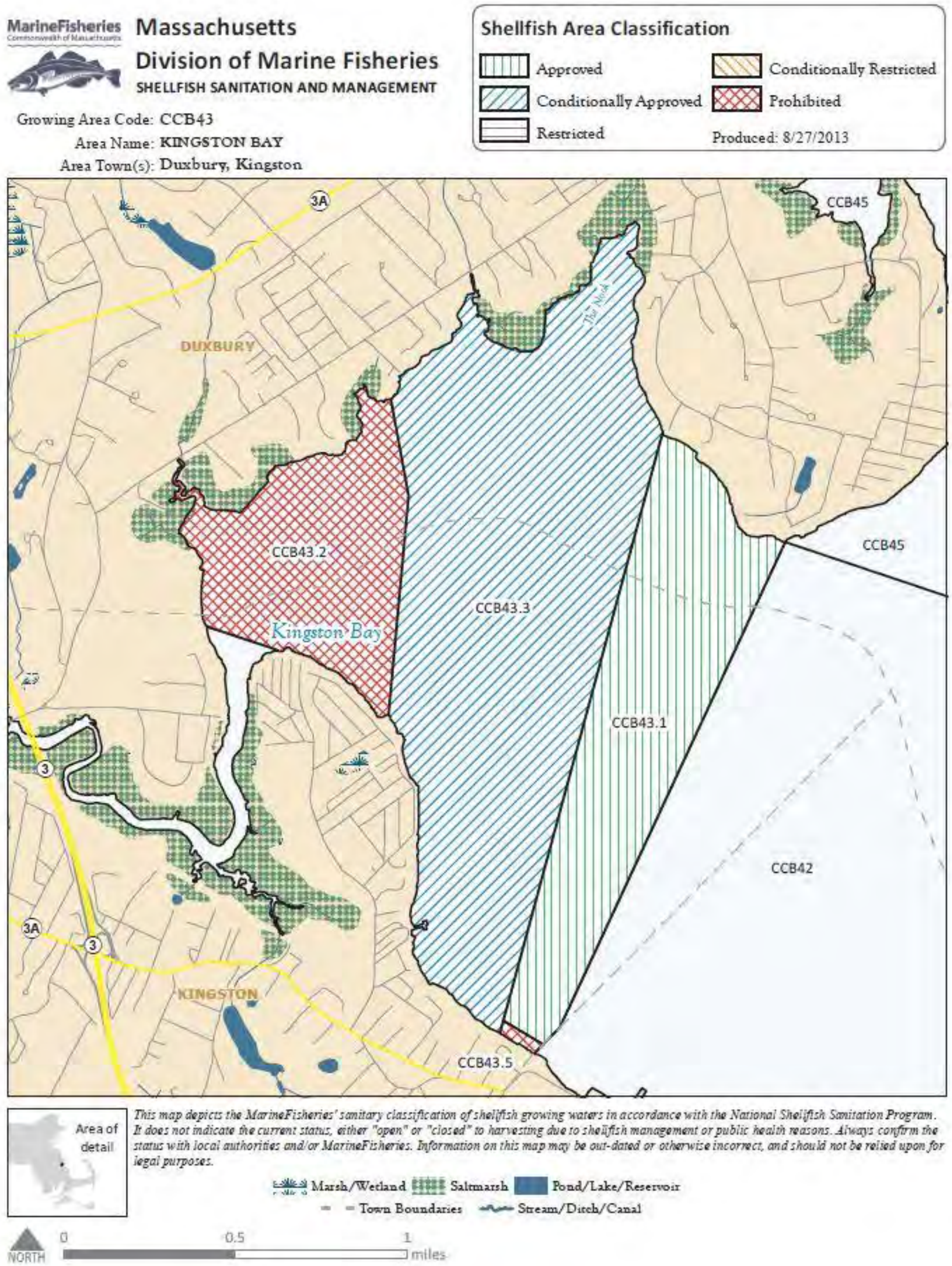


Figure II-2b. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination. The prohibited area at the mouth of the Jones River receives most of the freshwater inflow to the system or active harbor/marina areas, while the "conditionally approved" area is a mixing zone between freshwater inputs to the system and the area that is well flushed with Cape Cod Bay water each tide.

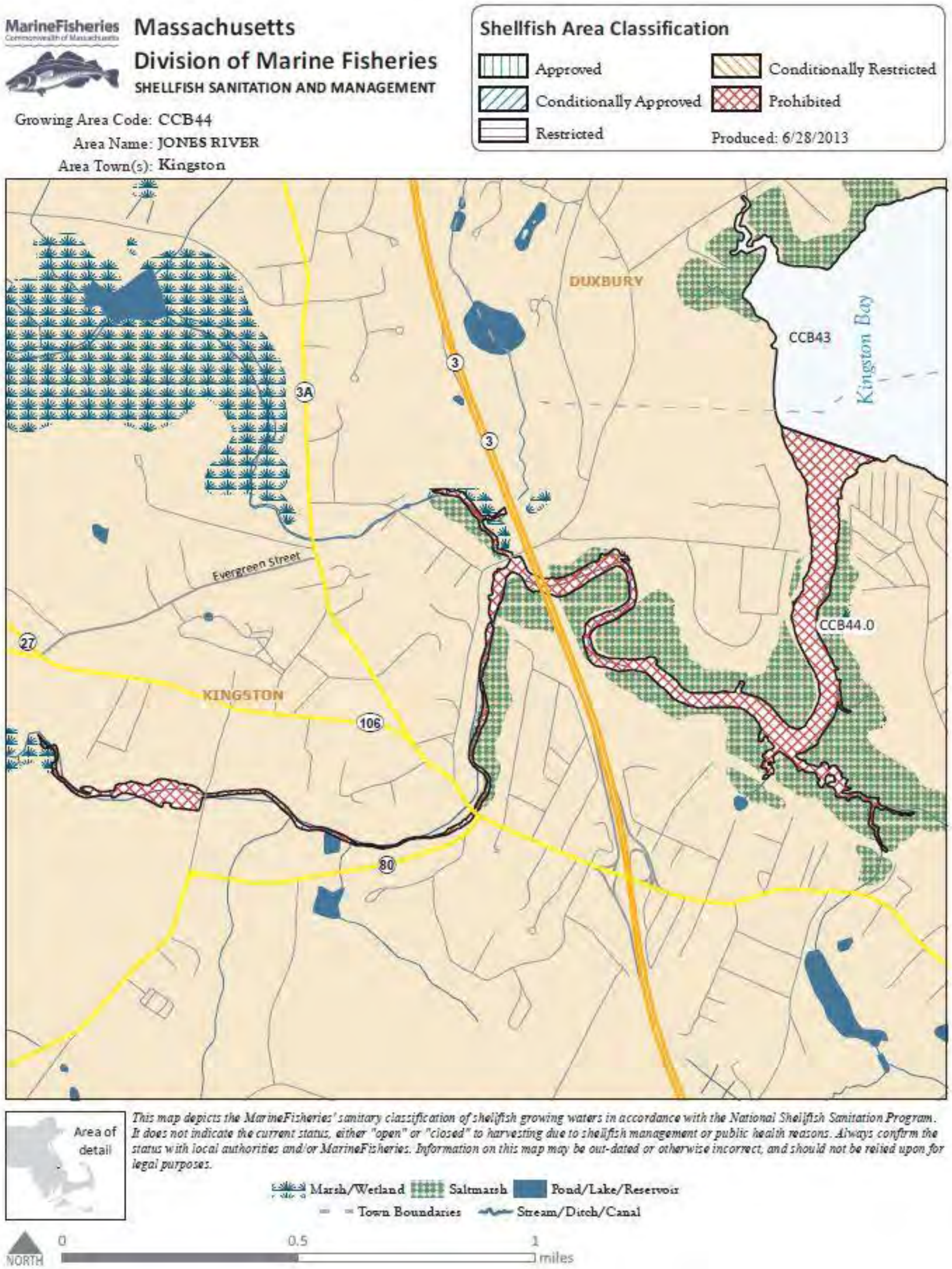


Figure II-2c. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination. The prohibited areas are smaller marsh tributary creeks that receive most of the freshwater inflow to the system such as the tidally influenced reach of the Jones River. Wetland areas with persistent fecal coliform levels >14 cfu per 100 mL may be prohibited to shellfishing until the source of contamination (frequently wildlife & birds) is documented.

Figure II-2d. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries.

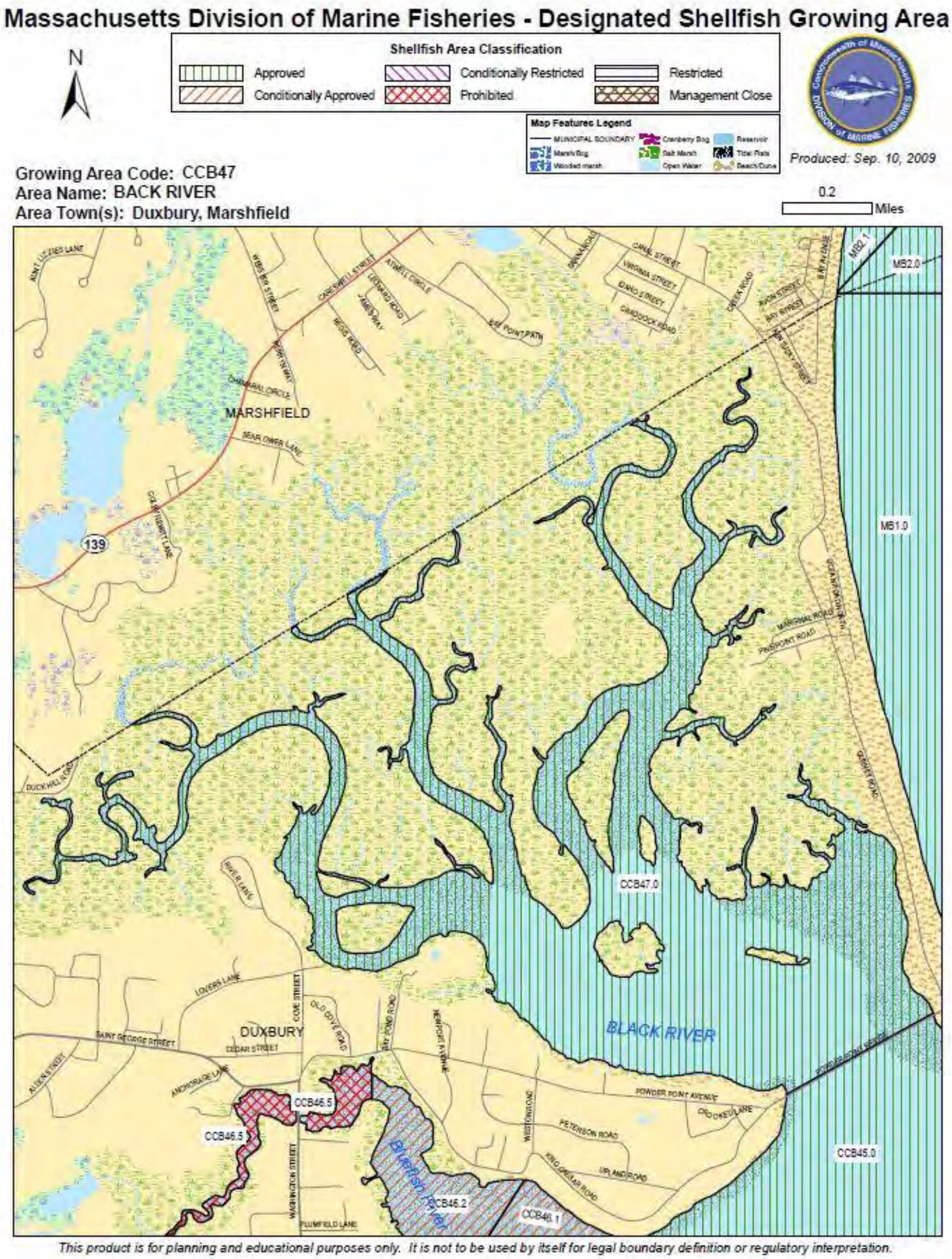


Figure II-2e. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination. The prohibited areas are smaller marsh tributary creeks that receive direct freshwater inflow to the system or may support active harbor/marina areas, while the "conditionally approved areas are in the main tidal creeks that flush with Cape Cod Bay water each tide. Wetland areas with persistent fecal coliform levels >14 cfu per 100 mL may be prohibited to shellfishing until the source of contamination (frequently wildlife & birds) is documented.

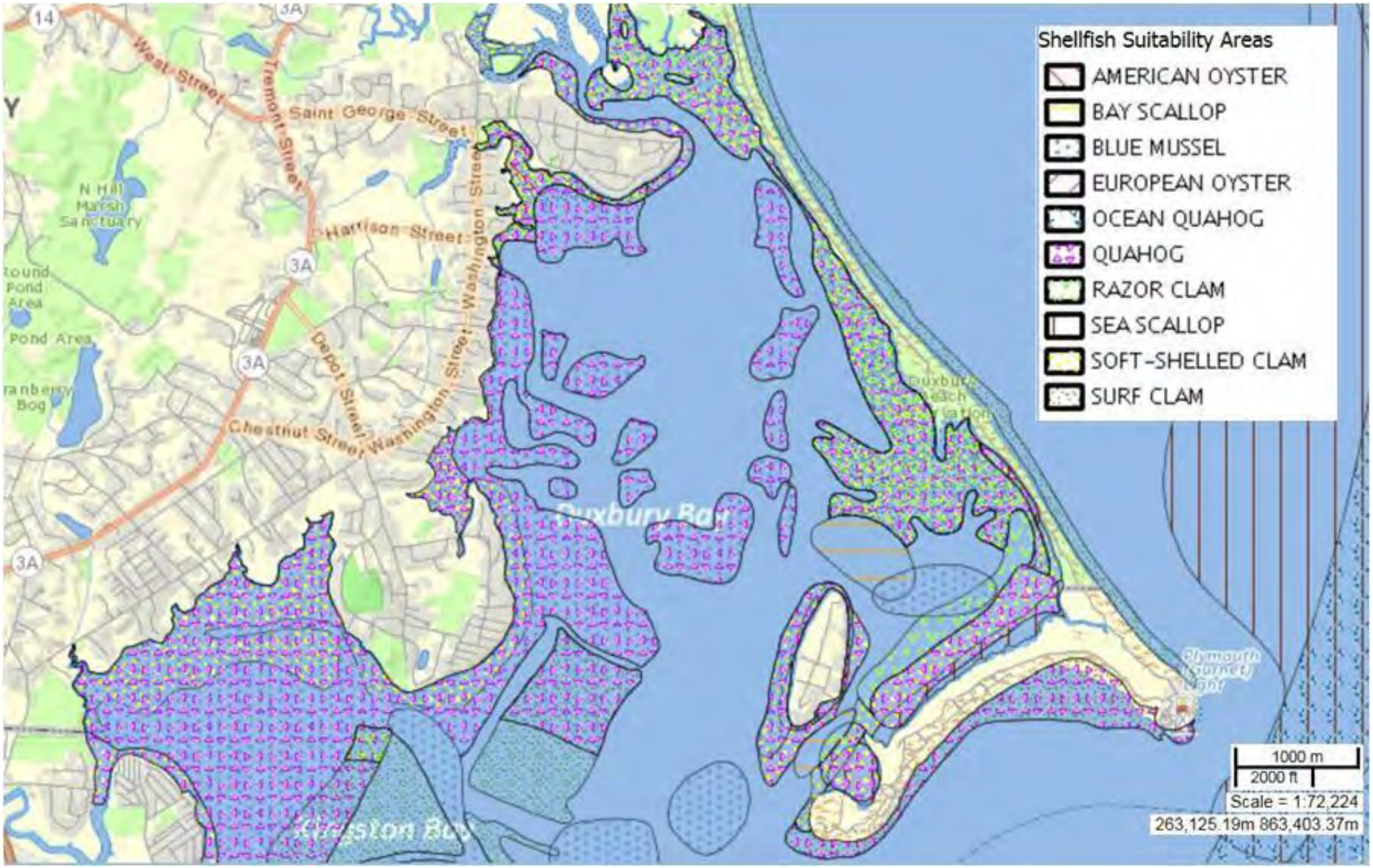


Figure II-3a. Location of shellfish suitability areas within the Kingston and Duxbury Bay portions of the overall estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean that a shellfish population is "present" or that harvest is allowed (see Figure II-2). Duxbury Bay supports active oyster aquaculture and landing at the high end for Massachusetts waters.

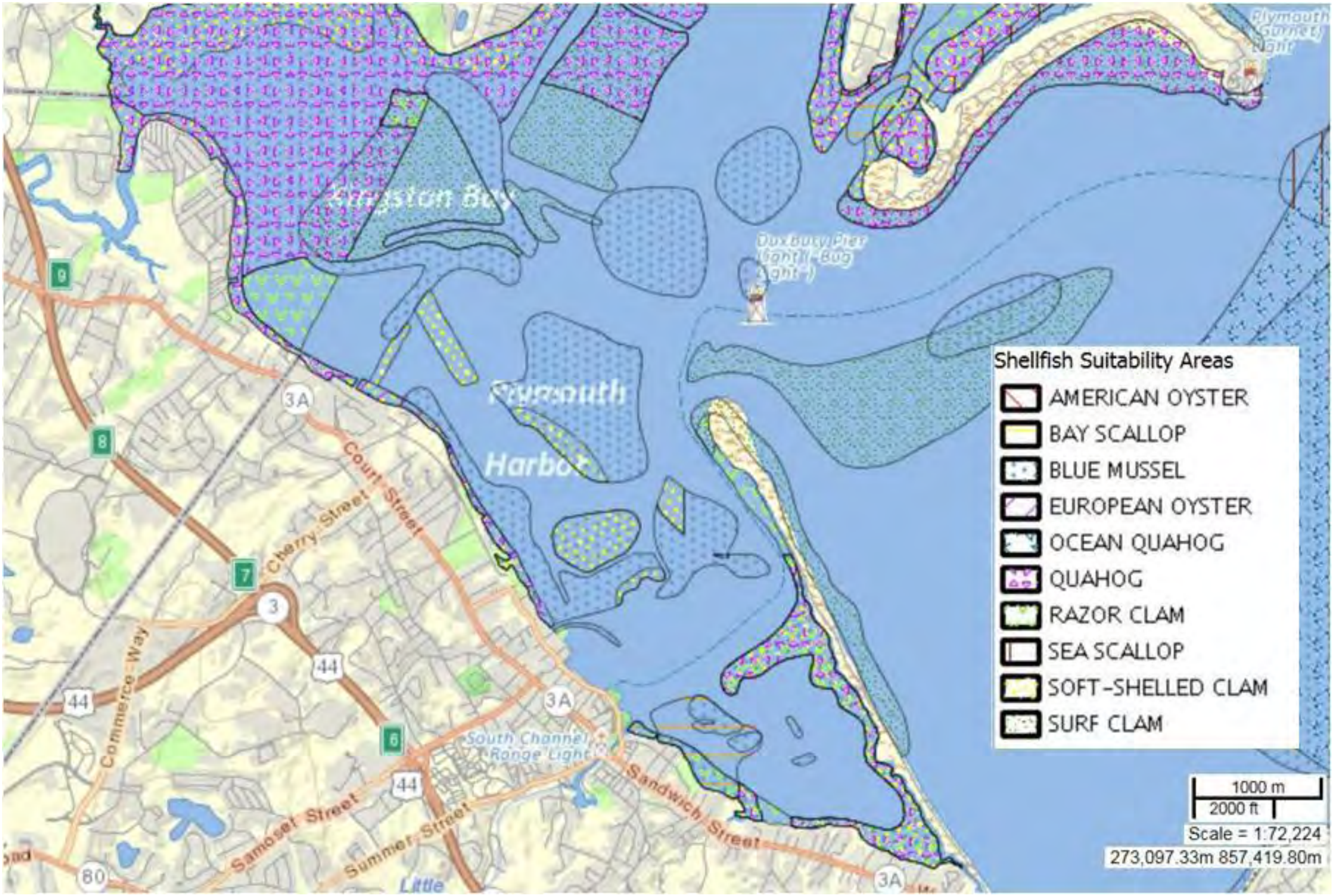


Figure II-3b. Location of shellfish suitability areas within the Plymouth Harbor portion of the overall estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean that a shellfish population is "present" or that harvest is allowed (see Figure II-2).

Figure II-4a. Anadromous fish runs associated with Duxbury Bay, Kingston Bay and the surface freshwater rivers which discharge to these basins as determined by Mass Division of Marine Fisheries. The red symbols show areas where fish were observed.

Figure II-4b. Anadromous fish runs associated with Plymouth Harbor and the surface freshwater rivers (Eel River, Town Brook) which discharges to this basin as determined by Mass Division of Marine Fisheries. The red symbols show areas where fish were observed.

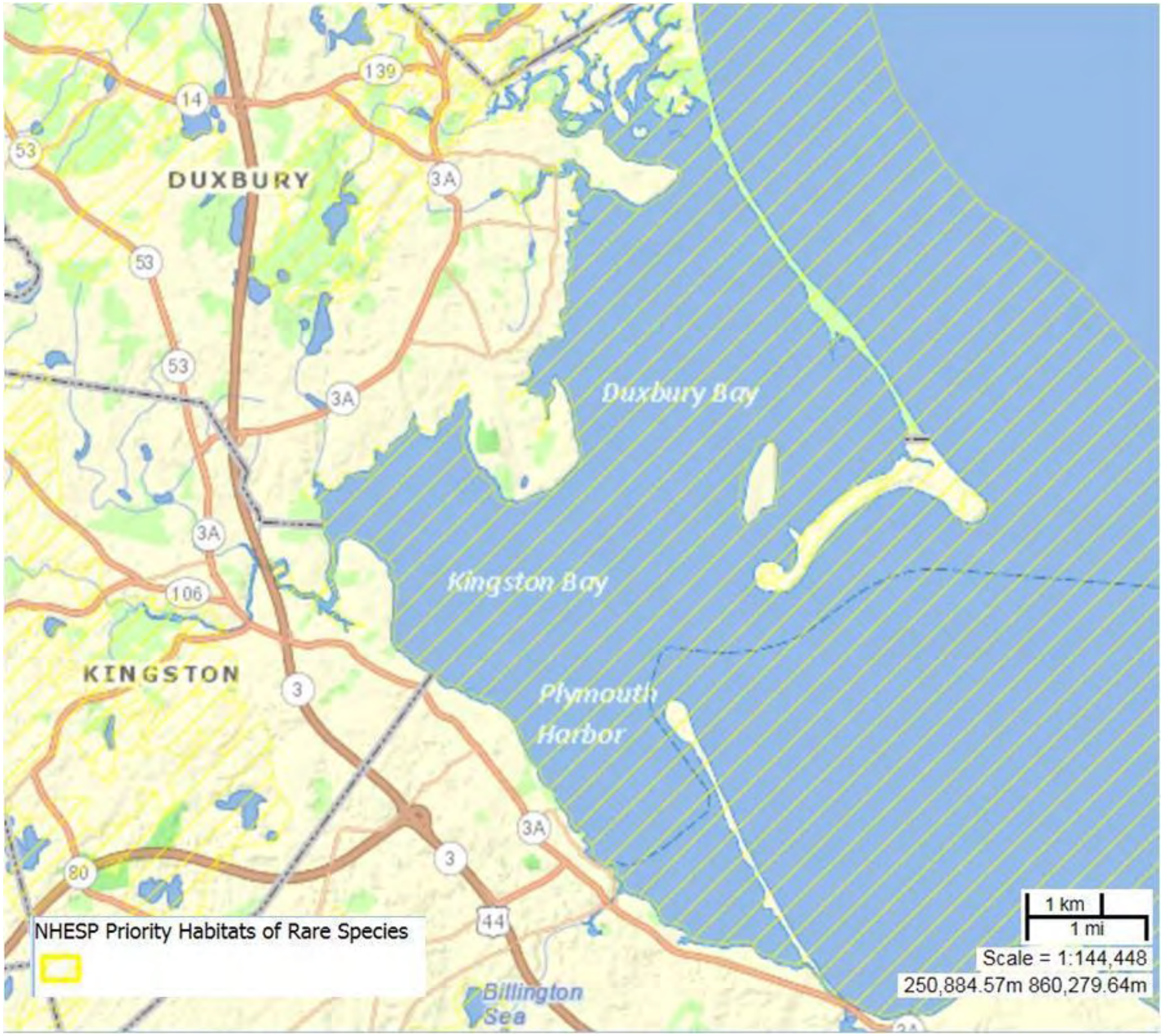


Figure II-5. Estimated Habitats for Rare Wildlife and State Protected Rare Species within the Sandwich Harbor Estuary as determined by the Massachusetts Natural Heritage and Endanger Species Program (NHESP).

Figure II-6a. Mouth of Coastal Rivers designation for Duxbury Bay as determined by – MassDEP Wetlands Program, under the Massachusetts Rivers Protection Act. The open water basins of Plymouth Harbor, Kinston Bay and Duxbury Bay are lagoons and not part of the river systems.

Figure II-6b. Mouth of Coastal Rivers designation for Duxbury Bay as determined by – MassDEP Wetlands Program, under the Massachusetts Rivers Protection Act. The open water basins of Plymouth Harbor, Kinston Bay and Duxbury Bay are lagoons and not part of the river systems.

Figure II-6c. Mouth of Coastal Rivers designation for Duxbury Bay as determined by – MassDEP Wetlands Program, under the Massachusetts Rivers Protection Act. The open water basins of Plymouth Harbor, Kinston Bay and Duxbury Bay are lagoons and not part of the river systems.

Figure II-6d. Mouth of Coastal Rivers designation for Duxbury Bay as determined by – MassDEP Wetlands Program, under the Massachusetts Rivers Protection Act. The open water basins of Plymouth Harbor, Kinston Bay and Duxbury Bay are lagoons and not part of the river systems.

Figure II-6e. Mouth of Coastal Rivers designation for Duxbury Bay as determined by – MassDEP Wetlands Program, under the Massachusetts Rivers Protection Act. The open water basins of Plymouth Harbor, Kinston Bay and Duxbury Bay are lagoons and not part of the river system.

Figure II-6f. Mouth of Coastal Rivers designation for Kingston Bay as determined by – MassDEP Wetlands Program, under the Massachusetts Rivers Protection Act. The open water basins of Plymouth Harbor, Kinston Bay and Duxbury Bay are lagoons and not part of the river systems.

Figure II-6g. Mouth of Coastal Rivers designation for Plymouth Harbor as determined by – MassDEP Wetlands Program, under the Massachusetts Rivers Protection Act. The open water basins of Plymouth Harbor, Kinston Bay and Duxbury Bay are lagoons and not part of the river systems.

Figure II-6h. Mouth of Coastal Rivers designation for Plymouth Harbor as determined by – MassDEP Wetlands Program, under the Massachusetts Rivers Protection Act. The open water basins of Plymouth Harbor, Kinston Bay and Duxbury Bay are lagoons and not part of the river systems.

Figure II-6i. Mouth of Coastal Rivers designation for Plymouth Harbor as determined by – MassDEP Wetlands Program, under the Massachusetts Rivers Protection Act. The open water basins of Plymouth Harbor, Kinston Bay and Duxbury Bay are lagoons and not part of the river systems.

Figure II-6j. Mouth of Coastal Rivers designation for Plymouth Harbor as determined by – MassDEP Wetlands Program, under the Massachusetts Rivers Protection Act. The open water basins of Plymouth Harbor, Kinston Bay and Duxbury Bay are lagoons and not part of the river systems.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Massachusetts Estuaries Project team includes technical staff from the United States Geological Survey (USGS). The USGS groundwater modelers were central to the development of the groundwater modeling approach used by the Estuaries Project. The USGS has a long history of developing regional models, including those for the Plymouth Carver Aquifer. Through the years, advances in computing, continuing data collection on lithology from well installations, water level monitoring, stream flow, and reconstruction of glacial history have allowed the USGS to update and refine the groundwater models. The MODFLOW and MODPATH models utilized by the USGS organize and analyze the available data using up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation. These questions include surface water/groundwater interactions, groundwater travel times, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including the Plymouth Harbor - Duxbury Bay - Kingston Bay (PDK) Embayment System. The PDK watershed covers portions of seven towns: Duxbury, Halifax, Kingston, Marshfield, Pembroke, Plymouth, and Plympton increasing the complexity of watershed nitrogen management planning.

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to initially define the watershed or contributing area to the PDK Embayment System under evaluation by the Project Team. The PDK estuary is a 9.3 square kilometer lagoonal estuary composed of its component, relatively shallow bays connected to a single, relatively deep inlet through which it exchanges tidal waters with Cape Cod Bay. Watershed modeling was undertaken to sub-divide the overall watershed into functional sub-units based upon: (a) defining inputs from contributing areas to each major component basin of the embayment system, (b) defining contributing areas to major freshwater aquatic systems which attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining the land areas with groundwater travel times that are greater and less than 10 years time-of-travel to the estuary. These travel-time distributions within subwatersheds are used as a procedural check to gauge the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters at the time of the MEP analysis. The three-dimensional numerical model employed has also been used to evaluate the contributing areas to public water supply wells in the Plymouth Carver Aquifer (PCA); the PDK watershed is located along the northern edge of the Plymouth Carver Aquifer groundwater lens. USGS model outputs were also compared to surface water discharges measured as part of the MEP stream flow program (2003 to 2005), as well as historic (1969-1971) and more recent (2006-2009) USGS monitoring of the Eel River.

The land adjacent to the PDK Embayment System contains a complex mix of geologic types, which creates a complicated hydrologic environment for watershed delineation. Most of the watershed to Plymouth Harbor is composed of highly transmissive, glacially-derived sands and gravels with some tills (silty gravel) near the hills of the southern boundary. In the northern portion of the watershed, beginning just to the south of the Jones River, a transition to less-transmissive, stratified drift soils occurs (Masterson, *et al.*, 2009). Sediments were deposited approximately 15,000 years ago during the late Wisconsinan glacial stage of the Pleistocene Epoch (Larson, 1980) after the ice had retreated northward from the Martha’s Vineyard, Nantucket and Cape Cod terminal moraines. The southern outwash plain contains numerous kettle ponds

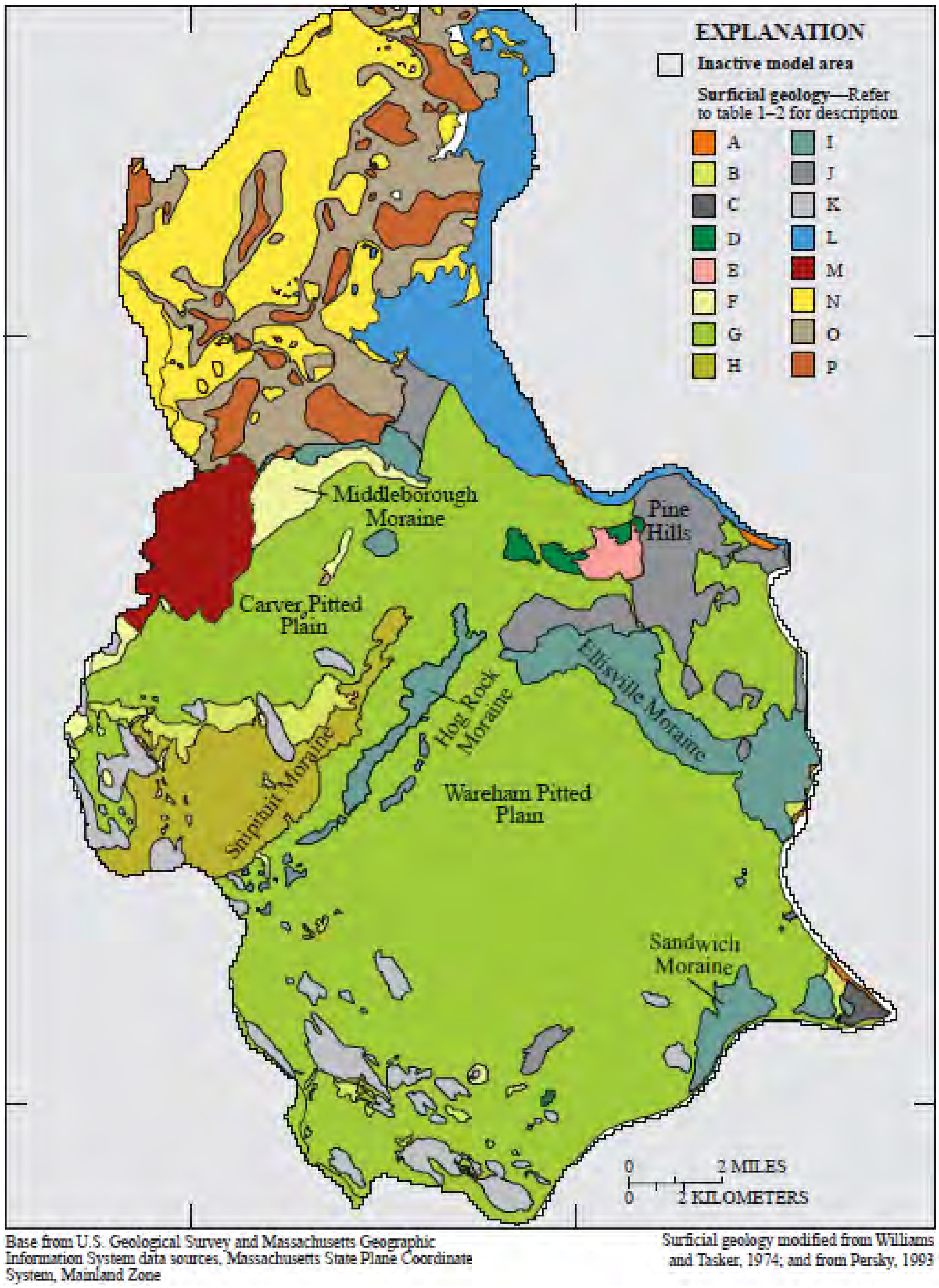


Figure III-1. Surficial geology of the Plymouth-Carver-Kingston-Duxbury aquifer system. The southern portion of Plymouth Harbor - Duxbury Bay - Kingston Bay Embayment System is located in outwash plains (pitted plains), while the northern portion is composed of till. Moraines bracket the outwash plain areas. Details on the composition of the surficial geology in these areas are found in Masterson, *et al*., 2009. This figure is modified from Figure 3 in Masterson, *et al*., 2009.

formed from collapsed sediments over buried ice blocks and is generally bordered on the east and north by a series of moraines (Figure III-1). Another smaller outwash plain is found to the north of these moraines before moving further north into the area of stratified drift.

For the purposes of delineating a watershed, these geologic transitions mean that watershed boundaries in the northern stratified drift area tend to be determined by land surface topography and precipitation on land surfaces tends to generate more surface runoff than infiltration/recharge to groundwater. In contrast, the highly-transmissive sands of the southern watershed areas are part of the Plymouth Carver Aquifer system, which is similar to geology of most of Cape Cod, and has watersheds primarily defined by the elevation of the water table and where precipitation results in recharge to the groundwater aquifer rather than surface runoff (Cambareri and Eichner, 1998; Millham and Howes 1994a,b). Freshwater discharge to estuaries is usually composed of surface water inflow from streams, which receive much of their water from groundwater base flow, and direct groundwater discharge. For a given estuary, differentiating between these two water inputs and tracking the sources of nitrogen that they carry requires determination of the portion of the watershed that contributes directly to streams and the portion of the groundwater system that discharges directly into an estuary as groundwater seepage.

III.2 MODEL DESCRIPTION

Contributing areas to the Plymouth Harbor - Duxbury Bay - Kingston Bay (PDK) Embayment System and its various subwatersheds, such as the Jones River, Town Brook, Eel River, and each of the 45 freshwater ponds were delineated using the updated regional model of the Plymouth Carver Aquifer (PCA) (Masterson, *et al.*, 2009). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al.*, 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 1994), which uses output files from MODFLOW-2000 to track the simulated movement of water within the aquifer, was used to delineate the area at the water table that contributes water to public water supply wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to the PDK Embayment System and its subwatersheds and also to determine portions of recharged water that may flow through fresh water ponds and streams prior to discharging into coastal water bodies.

The 2009 USGS Plymouth Carver Aquifer groundwater model is a second-generation groundwater model, built using the results from a first-generation version that the USGS completed in 1992 (Hansen and Lapham, 1992). The 2009 modeled aquifer area is about 750 square kilometers and consists of a finite-difference grid of uniformly spaced 400 ft squares spread over 355 rows and 270 columns. The model also includes 8 layers designed to accommodate lithology, well screens, and pond and streambed depths. The bottom layer (layer 8) extends 50 ft below the bedrock surface, which ranges in altitude from more than 60 ft above NGVD 29 in the northern portions of the model grid to more than 150 ft below NGVD 29 in the southern portions. The top of the upper layer is the water table. Calibration of the model was an iterative process, incorporating known geology and measurements of water table elevations and stream flows. Average recharge was set at 27 inches per year based on review of historic precipitation and construction of a transient component of the model. The model also contains public water supply wells pumping at known rates with water returned to the aquifer via septic systems and wastewater treatment facilities based on land use patterns and facility siting; 15% consumptive loss is assumed in the return flow.

III.3 PLYMOUTH HARBOR - DUXBURY BAY - KINGSTON BAY CONTRIBUTORY AREAS

The initial watershed and sub-watershed boundaries for the PDK Embayment System were modeled by the United States Geological Survey (USGS). Groundwater models based on the USGS MODFLOW groundwater modeling code, like the PCA version, are developed as grids, which create blocky or saw-toothed representations of natural features such as shorelines, ponds, coastlines and rivers. The MEP Technical Team corrected the model output recharge areas using USGS topographic quadrangles and aerial photographs to better reflect actual shoreline geometry; this is a standard step in all MEP analyses as discussed with the USGS modelers. The recharge area correction includes evaluations to: (a) correct for the model grid spacing, (b) to reflect actual pond and coastal shorelines based on aerial ortho-photos, (c) to include water table data in the lower regions of the watersheds near the coast (as available), (d) to more closely match the sub-estuary segmentation of the tidal hydrodynamic model and (e) to address streamflow measurements collected as part of the MEP. The smoothing efforts rely on the modeled water table contours, as well as more refined, site-specific data collected during the MEP assessment.

The MEP sub-watershed delineations include 10-yr time-of-travel boundaries, subwatersheds to public water supply wells, ponds and portions of the coast, and the three MEP stream gauges within the overall PDK watershed. In total, 87 sub-watershed areas were delineated within the overall PDK watershed (Figure III-2). Among the subwatersheds, there are 56 to ponds, lakes or reservoirs, 10 to public water supply or irrigation wells, 11 to streams or rivers, and the remainder directly discharge to various basins of the PDK estuary via groundwater. Watershed discharge rates based on the USGS modeled recharge rate are shown for each of the subwatersheds in Table III-1.

During the delineation of the MEP watersheds, Technical Team members also compared modeled USGS streamflows to measured streamflow volumes collected as part of the MEP process. As part of the usual MEP efforts, stream gauges are placed in each of the major streams discharging to an estuary. The freshwater flow and nitrogen loads collected at the gauges are used as a check on the watershed delineations and the measured flows and loads are incorporated into the estuary water quality model. As documented in Howes and Samimy (2005), surface fresh water inflows to the PDK embayment system were measured and water quality samples were collected just prior to discharge into estuarine waters at three locations:

- Jones River at Rt. 3A, Monitoring Station PDH-16
- Town Brook at Rt. 3A, Monitoring Station PDH-17
- Eel River down-gradient at Plymouth Harbor, Monitoring Station PDH-18

Another part of the regular MEP approach is to compare MEP-collected data to data available from other sources, including historic streamflow data collected by the USGS. USGS has established numerous long-term stream gauge locations throughout New England, usually on the larger streams. In addition, the USGS often collects spot measurements at other smaller streams in the same general area. USGS readings often have a longer historic record than the MEP records, but the MEP measurements are collected at the time of estuary water quality data collection and are collected more frequently than USGS readings during that time. Therefore, the MEP streamflow readings are more appropriate for comparing to all the other estuary data collected simultaneously, as well for use in the development of an estuary water quality model. However, this distinction means that the MEP flows may be slightly different than the USGS long-term averages, so comparative analysis of USGS and MEP streamflow records is conducted to assess reasonable long-term average flow conditions.

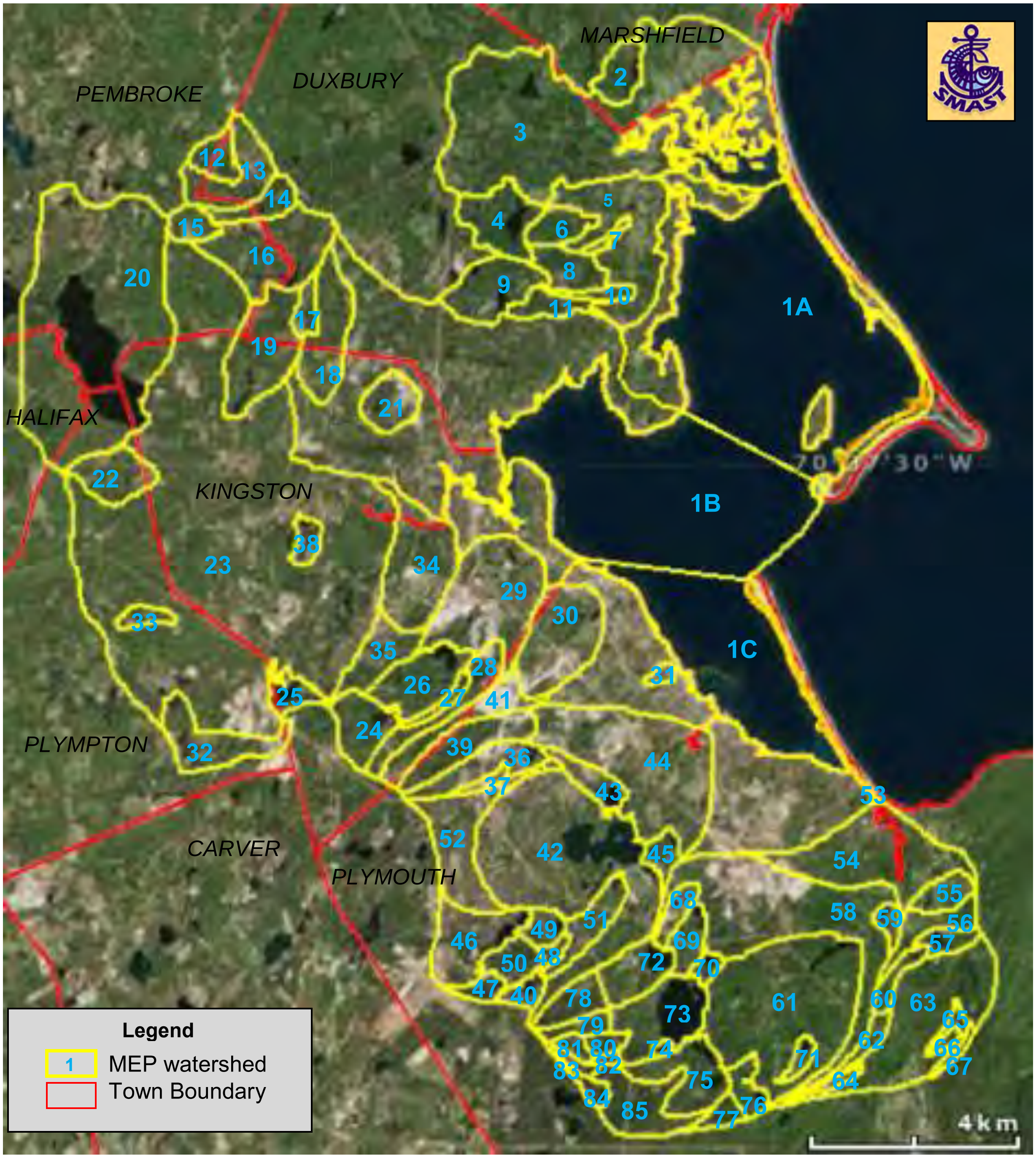


Figure III-2. Watershed delineation for the Plymouth Harbor - Duxbury Bay - Kingston Bay (PDK) Embayment System, which exchanges tidal waters with Cape Cod Bay (outer edge of yellow). Subwatershed delineations (numbered) are based on USGS groundwater model output with modifications to better address pond and estuary shorelines and MEP stream gauge measurements. Ten-year time-of-travel delineations were produced for quality assurance purposes and are designated with a “10” in the watershed names. Names for each of the subwatersheds are listed in Table III-2.

Table III-1. Daily groundwater discharge from each of the sub-watersheds in the overall Plymouth Harbor - Duxbury Bay - Kingston Bay (PDK) Embayment System MEP watershed.

Shed #	Subwatershed Name	Area_m2	Recharge m3/d
1A	Plymouth Harbor LT10N	4,177,134	7,848
1B	Plymouth Harbor LT10Mid	18,509,037	34,777
1C	Plymouth Harbor LT10S	9,918,165	18,635
2	Careswell Pond	791,100	1,486
3	Duxbury Marsh	14,081,999	26,459
4	North Hill Pond	1,515,746	2,848
5	Bluefish River LT10	3,600,103	6,764
6	Duxbury PWS3	719,287	1,351
7	Duxbury PWS1	171,500	322
8	Bluefish River GT10 N	943,596	1,773
9	Island Creek Pond	1,618,640	3,041
10	Duxbury PWS2	331,275	622
11	Bluefish River GT10 S	615,396	1,156
12	Bog Pond N	668,699	1,256
13	Bog Pond S	1,540,343	2,894
14	Upper Chandler Pond	734,700	1,380
15	Hill Pond	543,214	1,021
16	Lower Chandler Pond	2,871,165	5,395
17	Halls Brook Reservoir	411,729	774
18	Bracketts Pond	1,828,639	3,436
19	Pembroke St South Pond	2,376,125	4,465
20	Silver Lake	12,399,509	23,297
21	Blackwater Pond	932,241	1,752
22	Harrobs Corner Bog Pond	1,195,343	2,246
23	Jones River USGS Gauge	31,853,983	59,851
24	Muddy Pond	1,232,468	2,316
25	Indian Pond	574,417	1,079
26	Smelt Pond	1,686,597	3,169
27	Little Smelt Pond	268,985	505
28	Kingston PWS1	1,208,120	2,270
29	Foundry Pond Stream	3,768,350	7,080
30	Spooner Pond Stream LT10	2,529,204	4,752
31	Plymouth PWS2	158,057	297
32	Bay State Comp. Bog Reservoir	1,578,822	2,966
33	Dennets Pond	335,658	631
34	Jones River Gauge LT10	2,792,155	5,246
35	Jones River Gauge GT10	1,185,144	2,227
36	Triangle Pond	1,223,364	2,299
37	Little Muddy Pond	210,502	396
38	Crossman Pond	386,429	726
39	Plymouth PWS1	1,537,446	2,889
40	Kings Pond	390,151	733
41	Spooner Pond Stream GT10	1,324,173	2,488
42	Billington Sea LT10	8,268,306	15,535
43	Little Pond	694,108	1,304
44	Town Brook Gauge	5,686,220	10,684
45	Lout Pond	475,417	893
46	4 Ponds	1,857,006	3,489
47	Ellis Pond	357,169	671

Table III-1 (continued). Daily groundwater discharge from each of the sub-watersheds in the overall Plymouth Harbor - Duxbury Bay - Kingston Bay (PDK) Embayment System MEP watershed.

Shed #	Subwatershed Name	Area_m2	Recharge m3/d
48	Little Micajah Pond	210,299	395
49	Micajah Pond	461,873	868
50	Plymouth PWSS3	753,730	1,416
51	Briggs Reservoir	968,170	1,819
52	Billington Sea GT10	2,332,354	4,382
53	Eel River Gauge	26,182	49
54	Eel River 3A	3,873,527	7,278
55	Howland Pond	625,259	1,175
56	Eel River Mid	893,220	1,678
57	Forge Pond	445,184	836
58	Eel River W	6,337,399	11,907
59	Hayden Mill Pond	280,350	527
60	Cold Bottom Pond LT10	427,713	804
61	Russell Mill Pond	6,813,136	12,801
62	Cold Bottom Pond GT10	432,264	812
63	Eel River S	4,272,541	8,028
64	WELL GC1	183,840	345
65	Valley Road Pond	249,599	469
66	WELL GC2	120,186	226
67	Pine Road Pond	158,798	298
68	WELL	468,043	879
69	Cooks Pond	682,842	1,283
70	South Triangle Pond	247,816	466
71	Island Pond	241,290	453
72	Little South Pond LT10	1,164,323	2,188
73	Great South Pond LT10	2,042,832	3,838
74	Great South Pond Inlet	83,031	156
75	Boot/Ingalls Ponds LT10	1,684,375	3,165
76	Gunners Exchange/Hoyt Ponds LT10	548,075	1,030
77	Gunners Exchange/Hoyt Ponds GT10	387,359	728
78	Little South Pond GT10	855,111	1,607
79	Great South Pond GT10N	453,093	851
80	Powderhorn Pond LT10	284,079	534
81	Powderhorn Pond GT10	241,943	455
82	Great South Pond GT10S	119,287	224
83	Little Widgeon Pond	221,368	416
84	Widgeon Pond	300,000	564
85	Boot/Ingalls Ponds GT10	1,678,895	3,154
	TOTAL	195,646,318	367,601

Notes:
1) Discharge volumes are based on 27 inches of annual recharge on watershed areas.
2) Total recharge flow to Plymouth Harbor is less than the sum of these subwatersheds due to splitting of selected boundary subwatersheds and subsequent flow out of the system.
3) listed flows do not include precipitation on the surface of the estuary
4) totals may not match due to rounding.

In addition, the MEP also usually collects data at streams where the USGS has either not collected flow data or only has collected periodic spot measurements. For example, in the PDK watershed the continuous MEP flow measurements collected between 2003 and 2005 at Town Brook are the most detailed flow dataset available; by comparison, the USGS collected a total of 20 instantaneous flow readings in Town Brook between 1968 and 2010. Overall, the density of data collection for a given estuary/watershed system under the MEP is generally more refined and detailed than any prior assessments and this is certainly the case for the PDK system.

The MEP watershed modeled flows, MEP measured flows and the USGS measured flows generally agree at each of the three gauge locations (Table III-2). The average MEP measured flow at the Eel River gauge in 2003 to 2005 is within 2% of the average USGS measured flow. In addition, the MEP modeled flow based on the watershed recharge area is essentially the same as the average MEP measured flow. It is notable that the most recent USGS monitoring of the Eel River (2006-2009) has a 16% higher average flow at the Eel River gauge than the 1969-1971 average used in the calibration of the USGS regional groundwater model. It is also notable that MEP readings were collected in 2003-2005, a time period when the USGS was not monitoring the river. The coefficient of variation for daily flow in the older USGS time period is 20%, which is indicative of some moderate variability in flow; MEP stage readings indicate regular peaks in measured flow that would be consistent with this variation.

Table III-2. Comparison of MEP and USGS Measured Streamflows and MEP Watershed Flows at Gauge Locations in the Plymouth Harbor Watershed

Gauge Location	MEP Gauge	MEP Measured Flow - Mean[a]	MEP Watershed Flow[b]	USGS Measured Flow[c] - Mean	USGS Measured Flow – Standard Deviation	USGS data points	USGS Measuring Period
		m3/d	m3/d	m3/d	m3/d	N	Years
Jones River	PDH-16	102,201	114,497	86,161[d]	77,673	16,088	1966-2010
Town Brook	PDH-17	52,939	47,136	Regular measurements not collected			
Eel River	PDH-18	68,983	69,036	67,741	13,542	651	1969-1971
Eel River				78,914	19,155	1,173	2006-2009

Notes:

[a] measured flows from MEP monitoring 2003-2005 (Howes, B. and R. Samimy. 2005.)

[b] estimated flow based on recharge of 27 inches/yr and MEP-refined versions of USGS modeled contributing areas (Masterson, J.P., Carlson, C.S., and Walter, D.A. 2009.)

[c] USGS data from http://waterdata.usgs.gov/nwis/

[d] USGS Jones River gauge location is approximately 0.6 miles upstream of the MEP gauge and is expected to have a lower flow

The Jones River MEP flows are adjusted to account for public water supply withdrawals and water level fluctuations in Silver Lake (subwatershed #20). Silver Lake is part of the City of Brockton's water supply system (Gomez and Sullivan, 2013). As such, water levels in Silver Lake fluctuate significantly due to the drinking water withdrawals and additions for water supply management, as well as of natural fluctuations due to precipitation within the watershed. As a result, the Lake may go through significant periods where it does not discharge surface waters to the Jones River. MEP staff reviewed water levels in the Lake during the period that 604b stream gauging occurred (September 2003 to August 2004) and found that the Lake discharged to the Jones River approximately five months during that period (December 2003 to April 2004). Adjusting the watershed discharge from Silver Lake to account for this limitation resulted in excellent balance (<1% difference) between the estimated Jones River flow based on recharge within the USGS recharge area and measured MEP 2003-04 flow.

Comparison of measured flows at the MEP and USGS gauge locations along the Jones River indicates some variability between the MEP and USGS average measured flows, but most of this appears to be reasonable given the difference in gauge locations and the high flow variability. The USGS gauge is ~0.6 miles (1 km) upstream of the MEP gauge. The result is that the USGS gauge captures flow from less of the watershed than the MEP gauge. This difference accounts for a significant portion of the difference in observed flow. Comparison of the variability shows similar variability in both the USGS and MEP recordings. USGS historic flow readings have been collected daily since 1966 and they have an extremely high variability (coefficient of variation = 90%). This variability was also seen in the MEP stage readings where peak flows occur rapidly during storms, but only slowly decreased back to baseflow conditions, often taking weeks to attain pre-storm flows. The net result of the comparisons is that the mean flows generated from the MEP and USGS gauge records are not significantly different ($\rho<0.05$), with the MEP average well within one standard deviation of the USGS mean. Overall, given the flow variability at each of the gauge locations and the differences in gauge placement, the MEP watershed estimated flows for the Jones River are reasonable and show good agreement between the MEP and USGS streamflow readings.

The MEP watershed delineation appears to be the first of its kind for the entire Plymouth Harbor - Duxbury Bay - Kingston Bay (PDK) Embayment System. The Town of Plymouth has utilized USGS groundwater contours from the Hansen and Lapham (1992) model to delineate an outer watershed boundary within the town and has incorporated that boundary into part of an aquifer protection bylaw (Horsley and Witten, 2006). There is also a regional watershed divide between Plymouth Harbor and the Taunton River and Buzzards Bay watersheds that defines the South Coastal watershed used by the MA Water Resources Commission. This was generated by the USGS in 2000 as part of a statewide watershed delineation effort and appears to be based on topography. The MEP team created a draft delineation of the watersheds in this report that did not include the segmentation of the less than 10 year time of travel subwatershed to the main portion of Plymouth Harbor (Eichner and Howes, 2011). The MEP watershed is much more refined than either of the earlier delineations and the updated USGS groundwater modeling has generally been confirmed by MEP streamflows. The MEP watershed delineation also provides separate subwatersheds to component ponds, streams, and well resources that can be further updated as additional water quality data is developed. Overall the MEP watersheds offer a framework to develop holistic watershed water quality management strategies that can incorporate and unify strategies for each of the individual components, as well as the PDK system as a whole.

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

IV. WATERSHED NITROGEN LOADING TO EMBAYMENT: LAND USE, STREAM INPUTS, AND SEDIMENT NITROGEN RECYCLING

IV.1 WATERSHED LAND USE BASED NITROGEN LOADING ANALYSIS

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

In order to determine watershed nitrogen loading inputs to the PDK estuary system, the MEP Technical Team developed nitrogen-loading rates (Section IV.1) to each component of estuary and its watersheds (Section III). The PDK watershed was sub-divided to define contributing areas or subwatersheds to each of the major inland freshwater systems and to each major portion of the estuary. Further sub-divisions were made to identify watershed areas where a nitrogen discharge reaches estuary waters in less than 10 years or greater than 10 years. A total of 87 subwatersheds were delineated in the overall PDK watershed, including subwatersheds to 56 to ponds, lakes or reservoirs, 10 to public water supply or irrigation wells, and 11 to streams or rivers (see Chapter III). The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds and each portion of the estuary.

The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges to the watershed have reached the estuary. This involves a temporal review of land use changes, the time of groundwater travel to the estuary provided by the USGS watershed model, and review of data at natural collections points, such as streams and ponds. Evaluation and delineation of ten-year time of travel zones are a regular part of the watershed analysis. Ten-year time of travel subwatersheds in the PDK watershed have been delineated for ponds, streams and the estuary itself. Review of less than and greater than watersheds indicates that 91% of the unattenuated nitrogen load from the whole watershed is within less than 10 year travel time to the estuary (Table

IV-1). This finding should not be surprising given how streams generally penetrate at least 50% of the distance from the coast to the inland-most edge of the watershed and how most of the watershed development is concentrated closer to the coast. This comparison includes refinements for transfer of load out of the watershed by City of Brockton by the use of Silver Lake as part of the municipal water supply system (discussed in Chapter III). The overall result is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuary (after accounting for natural attenuation, see below) and that the distinction between time of travel in the subwatersheds is not important for modeling existing water quality conditions in the estuary. Overall and based on the review of all this information, it was determined that the PDK estuary is currently in balance with its watershed load.

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

Natural attenuation of nitrogen during transport from land-to-sea within the PDK watershed was determined based upon site-specific study of streamflow at three locations and assumed attenuation in the upgradient freshwater ponds. Streamflow was characterized at: Eel River, Jones River and Town Brook (Howes and Samimy, 2005). Subwatersheds to these stream discharge points allowed assignment of attenuation factors based on comparisons between field collected data from the streams and estimates from the nitrogen-loading sub-model. Nitrogen attenuation in individual ponds is conservatively assumed to equal 50% unless available monitoring and pond physical data is reliable enough to calculate a pond-specific attenuation factor.

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

Based upon the evaluation of the watershed systems, the MEP Technical Team used the Nitrogen Loading Sub-Model estimate of nitrogen loading for the subwatersheds that directly discharge groundwater to the estuary without flowing through one of these interim pond and stream measuring points. Internal nitrogen recycling was also determined throughout the tidal reaches of the PDK Estuarine System; measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying water-column. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

Table IV-1. Percentage of unattenuated nitrogen loads in less than ten year time-of-travel subwatersheds to Plymouth Harbor-Duxbury Bay-Kingston Bay Embayment System MEP watershed.

Shed #	Subwatershed Name	Less than 10 year time-of-travel to Estuary	Greater than 10 year time-of-travel to Estuary	Total	% LT10
		kg/yr	kg/yr	kg/yr	
1A	Plymouth Harbor LT10N	5,884		5,884	100%
1B	Plymouth Harbor LT10Mid	20,603		20,603	100%
1C	Plymouth Harbor LT10S	14,517		14,517	100%
2	Careswell Pond	594		594	100%
3	Duxbury Marsh	11,421		11,421	100%
4	North Hill Pond	910		910	100%
5	Bluefish River LT10	5,143		5,143	100%
6	Duxbury PWS3	345		345	100%
7	Duxbury PWS1	189		189	100%
8	Bluefish River GT10 N		753	753	0%
9	Island Creek Pond	786		786	100%
10	Duxbury PWS2	432		432	100%
11	Bluefish River GT10 S		922	922	0%
12	Bog Pond N	546		546	100%
13	Bog Pond S	1,192		1,192	100%
14	Upper Chandler Pond	704		704	100%
15	Hill Pond	645		645	100%
16	Lower Chandler Pond	3,592		3,592	100%
17	Halls Brook Reservoir	300		300	100%
18	Bracketts Pond	1,409		1,409	100%
19	Pembroke St South Pond	2,147		2,147	100%
20	Silver Lake	5,545		5,545	100%
21	Blackwater Pond	956		956	100%
22	Harrobs Corner Bog Pond	781		781	100%
23	Jones River USGS Gauge	27,417		27,417	100%
24	Muddy Pond	304		304	100%
25	Indian Pond	475		475	100%
26	Smelt Pond	498		498	100%
27	Little Smelt Pond	37		37	100%
28	Kingston PWS1	255		255	100%
29	Foundry Pond Stream	3,187		3,187	100%
30	Spooner Pond Stream LT10	2,078		2,078	100%
31	Plymouth PWS2	279		279	100%
32	Bay State Comp. Bog Res.	855		855	100%
33	Dennets Pond	192		192	100%
34	Jones River Gauge LT10	3,526		3,526	100%
35	Jones River Gauge GT10		1,225	1,225	0%
36	Triangle Pond	2,422		2,422	100%
37	Little Muddy Pond	212		212	100%
38	Crossman Pond	595		595	100%
39	Plymouth PWS1	1,660		1,660	100%
40	Kings Pond		1,101	1,101	0%
41	Spooner Pond Stream GT10		891	891	0%
42	Billington Sea LT10	12,659		12,659	100%
43	Little Pond	1,314		1,314	100%

Table IV-1 (continued). Percentage of unattenuated nitrogen loads in less than ten year time-of-travel subwatersheds to Plymouth Harbor-Duxbury Bay-Kingston Bay Embayment System MEP watershed.					
Shed #	Subwatershed Name	Less than 10 year time-of-travel to Harbor	Greater than 10 year time-of-travel to Harbor	Total	% LT10
		kg/yr	kg/yr	kg/yr	
44	Town Brook Gauge	9,010		9,010	100%
45	Lout Pond	349		349	100%
46	4 Ponds		2,516	2,516	0%
47	Ellis Pond		335	335	0%
48	Little Micajah Pond		377	377	0%
49	Micajah Pond		1,085	1,085	0%
50	Plymouth PWSS3	547		547	100%
51	Briggs Reservoir	1,003		1,003	100%
52	Billington Sea GT10		6,017	6,017	0%
53	Eel River Gauge	8		8	100%
54	Eel River 3A	3,245		3,245	100%
55	Howland Pond	391		391	100%
56	Eel River Mid	293		293	100%
57	Forge Pond	184		184	100%
58	Eel River W	7,396		7,396	100%
59	Hayden Mill Pond	161		161	100%
60	Cold Bottom Pond LT10	102		102	100%
61	Russell Mill Pond	4,737		4,737	100%
62	Cold Bottom Pond GT10		324	324	0%
63	Eel River S	2,453		2,453	100%
64	WELL GC1	362		362	100%
65	Valley Road Pond	185		185	100%
66	WELL GC2	94		94	100%
67	Pine Road Pond	47		47	100%
68	WELL	216		216	100%
69	Cooks Pond	249		249	100%
70	South Triangle Pond	152		152	100%
71	Island Pond	399		399	100%
72	Little South Pond LT10	686		686	100%
73	Great South Pond LT10	1,513		1,513	100%
74	Great South Pond Inlet	67		67	100%
75	Boot/Ingalls Ponds LT10	649		649	100%
76	Gunners Exchange/Hoyt Ponds LT10	308		308	100%
77	Gunners Exchange/Hoyt Ponds GT10		19	19	0%
78	Little South Pond GT10		44	44	0%
79	Great South Pond GT10N		24	24	0%
80	Powderhorn Pond LT10	76		76	100%
81	Powderhorn Pond GT10		12	12	0%
82	Great South Pond GT10S		8	8	0%
83	Little Widgeon Pond		52	52	0%
84	Widgeon Pond		136	136	0%
85	Boot/Ingalls Ponds GT10		179	179	0%
	TOTAL	171,486	16,021	187,507	91%

Table IV-1 Notes:
a) Whole system totals may not add due to rounding.
b) Loads do not include atmospheric loading on the estuary surface waters; if atmospheric loading on the surface of the estuary is included, the percentage of load within a less than 10 year time-of-travel increases to 93% of the total load.
c) Silver Lake loads are corrected for City of Brockton public drinking water withdrawals and average flow over its dam into the Jones River during the MEP stream monitoring (see Chapter III).

IV.1.1 Land Use and Water Use Database Preparation

The PDK watershed comprised of land areas within seven towns: Duxbury (20% of the watershed), Halifax (1%), Kingston (24%), Marshfield (2%), Pembroke (7%), Plymouth (39%), and Plympton (7%). Estuaries Project staff obtained digital parcel and tax assessor's data from the seven towns to serve as a base for the MEP watershed nitrogen loading model (Eichner and Howes, 2011). Using GIS techniques, this data was linked to current zoning areas and available parcel-by-parcel water use information for the towns with public water supply. Table IV-2 lists the data obtained from each of the seven towns in the watershed. The resulting unified watershed land use database also contains traditional information regarding land use classifications (MassDOR, 2012) plus additional information developed by the towns, such as building footprints. The database efforts were completed with the assistance from GIS staff from MAE, Inc. and the Town of Plymouth.

Table IV-2. Land Use information in Plymouth Harbor - Duxbury Bay - Kingston Bay (PDK) Watershed used in the MEP watershed nitrogen loading model. Data was current at the time of the development of the model (2011).

Town	Parcels	Assessor's Land Use Classifications	Zoning	Parcel-by-parcel Water Use	Sewered Parcels
Duxbury	2010	2010	2010	2008 to 2010	2010
Halifax	2010	2010	2010[b]	Town contacted[e]	No sewers
Kingston	2009	2009	2009	2008 to 2010	2009
Marshfield	2010	2010	2010	Town contacted[e]	2010
Pembroke	2010[a]	2010	2010[d]	Town contacted[e]	No sewers
Plymouth	2009	2009	2009	2008 to 2010	2009
Plympton	2009	2009	2009[c]	No public water	No sewers

Notes:

[a] Official Pembroke parcels were unavailable from town. Parcels in PDK watershed were digitized and created by MEP staff using town parcel maps.

[b] Halifax was in the midst of updating parcels and zoning GIS; while parcels were available, only currently available GIS zoning was a MassGIS version; town staff confirmed that current minimum parcel size for all zoning districts was 40,000 sq. ft.

[c] Plympton did not have GIS zoning; only current available GIS zoning was MassGIS version; town staff confirmed that all zoning districts have a current minimum lot size of 60,000 sq. ft. with exception of retreat lots, which were listed in the assessor's data and were required to have 120,000 sq. ft.

[d] Pembroke did not have GIS zoning; only current available GIS zoning was MassGIS version; an updated map of 2010 zoning was included in the most current town zoning bylaws and showed that the entire area of the town within the PDK watershed was in the Residential A zone, which had a 40,000 sq. ft. minimum lot size.

[e] Towns were contacted for water use data, but it was not been received. Towns without parcel-based water use represent 10% of the PDK watershed area.

Figure IV-1 shows the land uses within the PDK estuary watershed. Land uses in the study area are grouped into ten land use categories: 1) residential, 2) commercial, 3) industrial, 4) agricultural, 5) multi-use, 6) recreational 7) undeveloped, 8) public service/government, including road rights-of-way, 9) freshwater, and 10) unclassified (properties without assigned town assessor's land use codes). These land use categories are generally aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MADOR, 2012). "Public service" in the MADOR system is tax-exempt properties, including lands owned by government (*e.g.*, wellfields, schools, open space, roads) and private groups like churches and colleges.

Residential land uses are generally the dominant land use type within the PDK watershed; they occupy the highest percentage area of land use types (35%) within the overall watershed and are the highest percentage in most of the subwatershed groupings shown in Figure IV-2. Examples of residential land uses include single-family residences, condominiums, apartment buildings, and multi-family residences. Public service is generally the second highest percentage area, although it is the highest percentage area of land use types in the Eel River subwatershed. Overall public service lands are 20% to 40% of the land use areas in the subwatershed groupings in Figure IV-2. Examples of these land uses are lands owned by town and state government (including golf courses, open space, and wellhead protection lands), housing authorities, and churches. Undeveloped lands generally are the third highest area in the subwatershed groupings and are 9% of the land area within the overall PDK watershed.

In all the subwatershed groupings, residential parcels are the dominant parcel type (Figure IV-3). Residential parcels are 76% of the parcels in the Jones River subwatershed, 78% of parcels in the Town Brook subwatershed, 62% of all parcels in the Eel River system watershed and 72% of the parcels in the overall PDK watershed. Single-family residences (MassDOR land use code 101) are the dominant type of residential parcel; these represent 85% to 96% of residential parcels counts in the gauged subwatersheds and 92% of the overall residential parcel area throughout the PDK system watershed.

In order to estimate wastewater flows within the PDK study area, MEP staff also obtained parcel-by-parcel water use data from the towns indicated in Table IV-2. Where water use was obtained, it was linked to the available town parcel database and assessor's data. Measured water use is used to estimate wastewater-based nitrogen loading from individual parcels; average water use is used for each parcel with multiple years of data. The final wastewater nitrogen load for each parcel is based upon the measured water-use, wastewater nitrogen concentration, and consumptive loss of water before the remainder is treated in a septic system (see Section IV.1.2). All parcels are assumed to use on-site septic systems unless additional information is available.

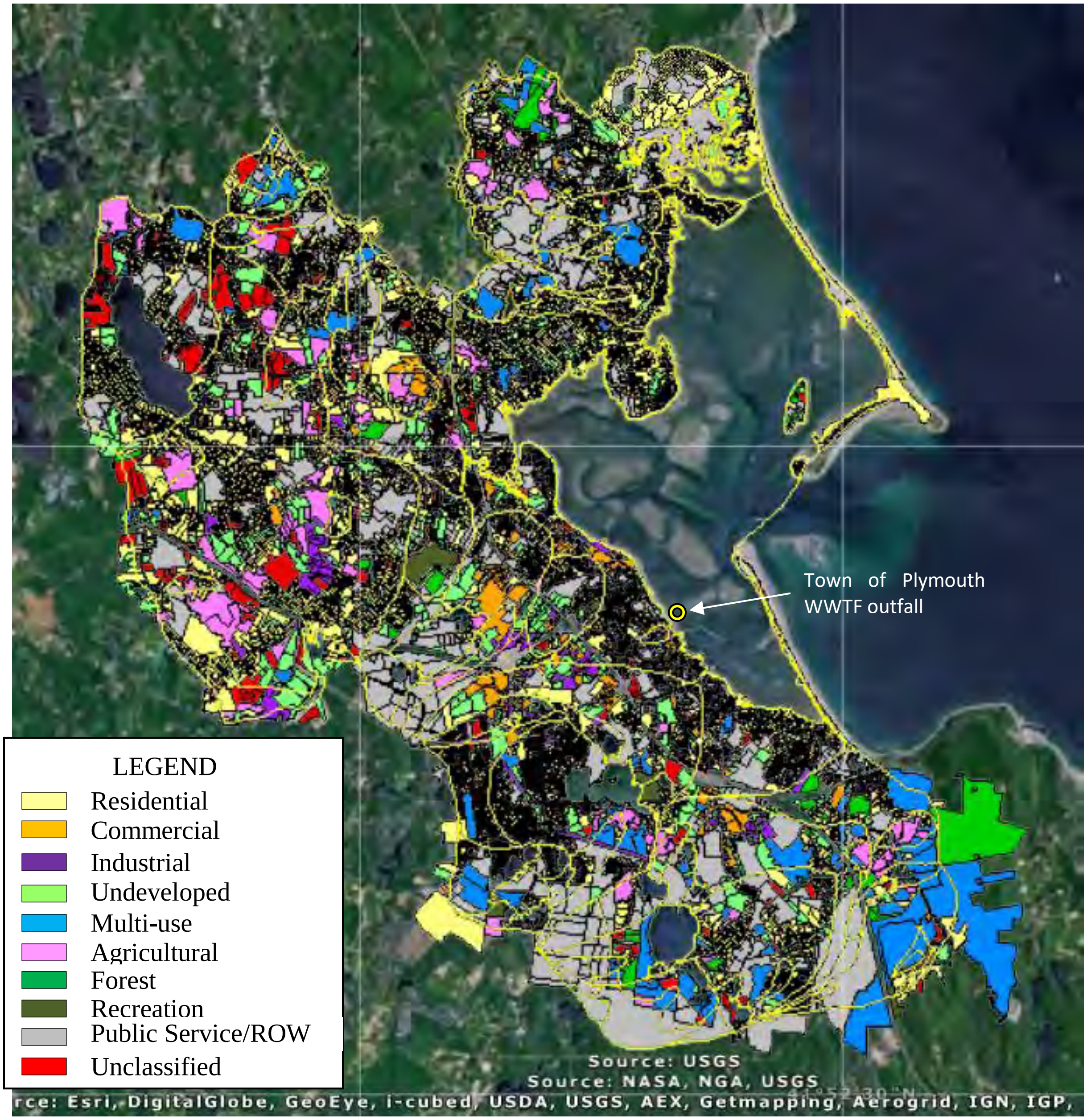


Figure IV-1. Land Uses within the Plymouth Harbor - Duxbury Bay - Kingston Bay Watershed. Land uses are based on town assessors' land use classifications from the seven towns within the watershed. Residential parcels are the dominant land use type both in terms of area and number of parcels. Classifications are aggregated based on the MassDOR general categories (MassDOR, 2012). Undeveloped parcels include parcels classified by the town assessors' as both developable (*e.g.*, land use categories 130, 391, and 441) and undevelopable (*i.e.*, land use categories 132, 392, and 442). Unclassified properties did not have land use category assignments in the assessors' databases used in the assessment. The assessors' databases were current as of the years listed in Table IV-2. The Town of Plymouth WWTF outfall location is also shown.

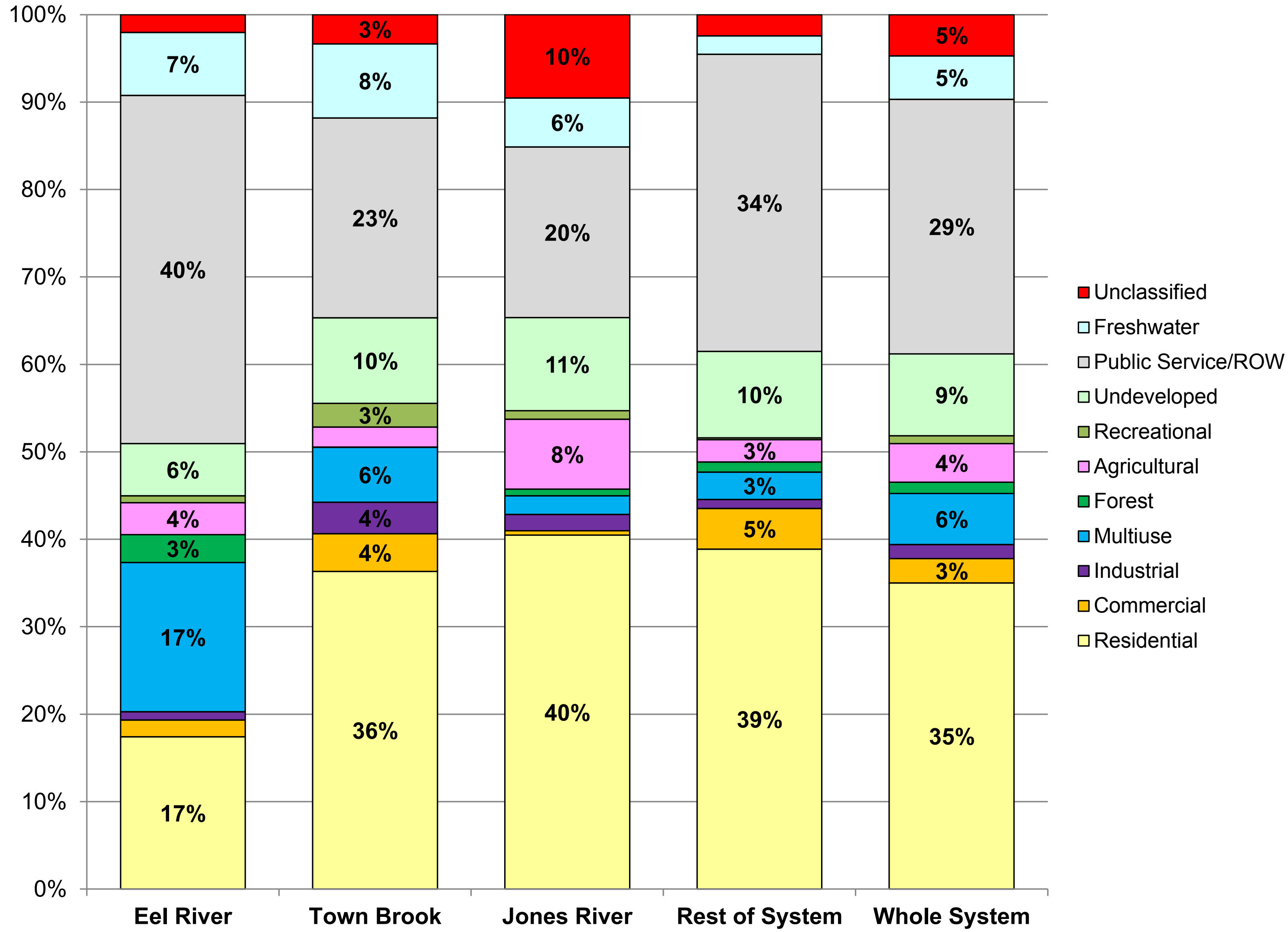


Figure IV-2. Distribution of land-uses by area within the Plymouth Harbor - Duxbury Bay - Kingston Bay system watershed and four component subwatersheds. Land use categories are generally based on town assessors' land use classification and groupings recommended by MADOR (2012). Unclassified parcels do not have an assigned land use code in the town assessors' databases. Only percentages greater than or equal to 3% are shown.

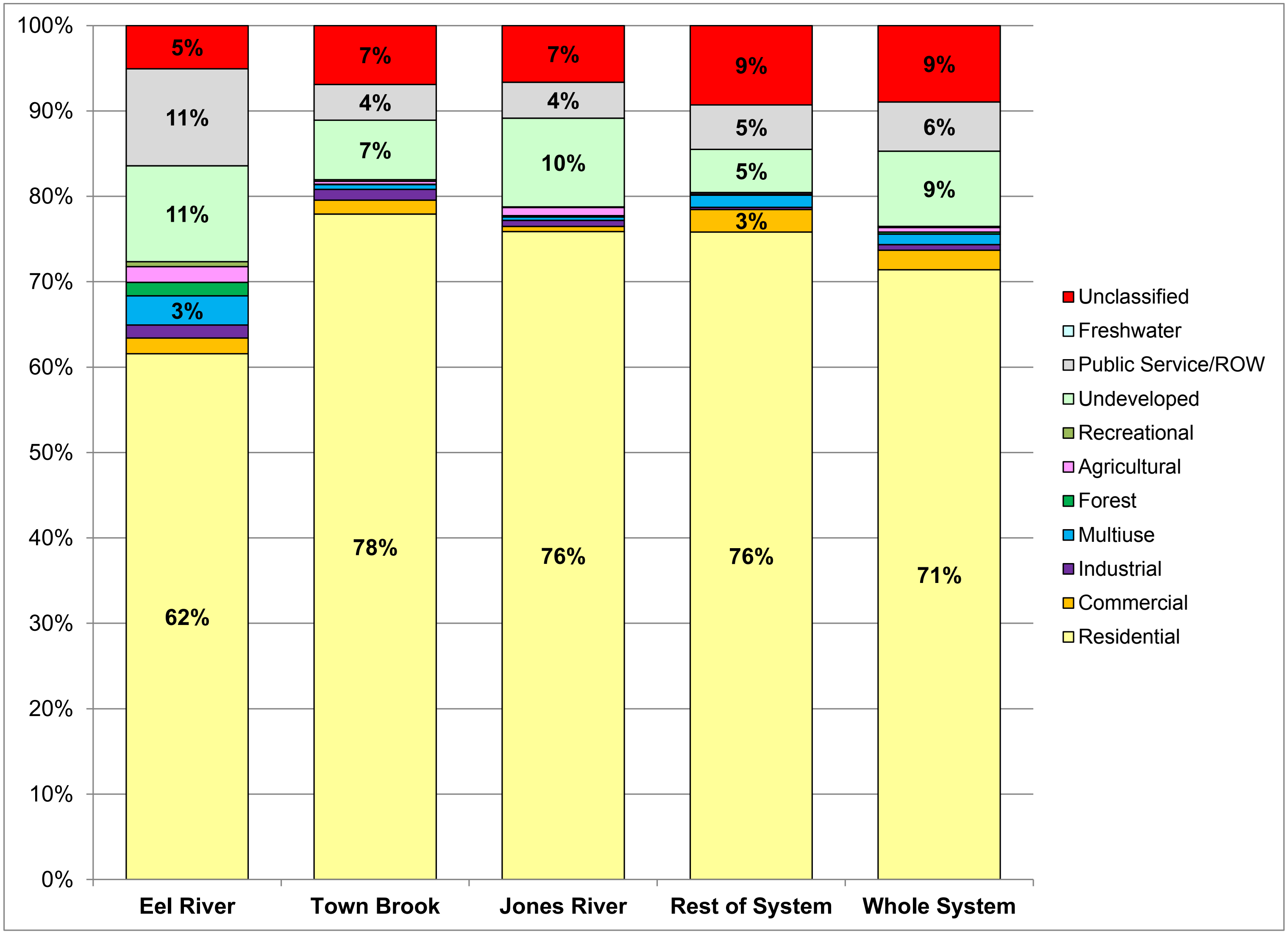


Figure IV-3. Distribution of land-uses by parcel count within the Plymouth Harbor - Duxbury Bay - Kingston Bay system watershed and four component subwatersheds. Land use categories are generally based on town assessors' land use classification and groupings recommended by MADOR (2012). Unclassified parcels do not have an assigned land use code in the town assessors' databases. Only percentages greater than or equal to 3% are shown.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

The Massachusetts Estuaries Project septic system nitrogen loading rate is fundamentally based upon a *per capita* nitrogen load to the receiving aquatic system. Specifically, the MEP septic system wastewater nitrogen loading is based upon a number of studies and additional information that directly measured septic system and per capita loads on Cape Cod or in similar geologic settings (Nelson *et al*. 1998, Weiskel and Howes 1991, 1992, Koppelman 1978, Frimpter *et al*. 1990, Brawley *et al*. 2000, Howes and Ramsey 2000, Costa *et al.* 2002). Variation in *per capita* nitrogen load has been found to be relatively small, with average annual *per capita* nitrogen loads generally between 1.9 to 2.3 kg/person-yr^{-1}.

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

All nitrogen losses within the septic system are incorporated into the MEP analysis. For example, information developed at the MassDEP Alternative Septic System Test Center at the Massachusetts Military Reservation on Title 5 septic systems have shown nitrogen removals between 21% and 25%. Multi-year monitoring from the Test Center has revealed that nitrogen removal within the septic tank was small (1% to 3%), with most (20 to 22%) of the removal occurring within five feet of the soil adsorption system (Costa *et al.*, 2002). Downgradient studies of septic system plumes in similar soils indicate that further nitrogen loss during aquifer transport is negligible (Robertson *et al*. 1991, DeSimone and Howes 1996).

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

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

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

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

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

At the outset of the MEP, project staff decided to utilize the water use approach for determining residential wastewater generation by septic systems because of the inherent difficulty in accurately gauging actual occupancy. Water use information exists for 73% of the 18,335 developed parcels in the PDK watershed. For the purposes of the nitrogen loading assessment, developed parcels without water use accounts are assigned an average water use. These are either assumed to be parcels utilizing private wells for drinking water in towns with available water use databases or utilizing some combination of private wells and public water for towns without water use databases. In all cases, these are properties that are classified with land use codes that should be developed (*e.g.*, 101 or 325), have been confirmed as having buildings on them through a review of aerial photographs, and do not have either a listed account in the water use databases or the town does not have public water or an available water use database. Of the 4,909 developed parcels without water use accounts, 3,980 (81%) are classified as single-family residences (land use code 101).

Developed parcels that do not have measured water use are assigned water use based on the average water use for similarly classified properties. Single-family residences are assigned the watershed average water use of 213 gallons per day (gpd), while other residential properties, which are mostly various classifications of multi-family properties, are assigned a watershed average water use of 411 gpd. Existing flows at commercial and industrial properties have a wide range of water uses, which would be expected given the diversity of uses within these categories (*e.g.*, hotels and fast food restaurants are in the commercial category). Evaluation of the existing Plymouth water use within these categories found that the averages were above the 75th percentile; for this reason, median flows from existing properties with water use were used for existing commercial and industrial properties without water use records throughout the watershed. Commercial properties were assigned 279 gpd, while industrial properties were assigned 408 gpd.

In order to provide an independent validation of the average residential water use within the PDK watershed, MEP staff reviewed US Census population values for the towns in the watershed. The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so for the purposes of Title 5 each person generates 55 gpd of wastewater. Based on data collected during the 2010 US Census, average residential occupancy within the seven towns in the watershed ranged between 2.63 and 2.83 people per housing unit with 86% to 96% year-round occupancy of available housing units. Average water use for single-family residences with municipal water accounts in the PDK MEP study area is 213 gpd. If the PDK average flow is multiplied by 0.9 to account for consumptive use, the study area wastewater average flow for a single-family residence is 192 gpd. If the average census occupancies are multiplied by 55 gpd, the range of wastewater generation is 146 to 156 gpd. Similar previous MEP analyses have indicated that seasonal occupancy can be a key determinant in this comparison. Even though the Census classifies only a small portion of the residential units as seasonal dwellings (0 to 10% of the housing stock in the watershed towns), occupancy of these units at higher intensity (2-3 times the annual average) would result in population estimated average wastewater generation reasonably reflective of the average measured wastewater estimates.

Wastewater Treatment Facilities

When developing watershed nitrogen loading information, MEP project staff typically seeks additional information on enhanced wastewater treatment in the project study area. This information is reviewed and if judged reliable is included in the watershed nitrogen loading model.

MEP staff collected wastewater treatment plant performance data from the Massachusetts Department of Environmental Protection (personal communication, Brian Dudley, MassDEP, 4/10). There are 11 facilities with state Groundwater Discharge Permits (GWDPs) within the PDK watershed (Table IV-3). A GWDP is required under MassDEP regulations for wastewater treatment systems with design flows greater than 10,000 gallons per day. The wastewater treatment facilities are located within the towns of Kingston, Plymouth, and Duxbury, and include municipal facilities for the Towns of Kingston and Plymouth. The Town of Plymouth wastewater treatment facility also utilizes an outfall pipe into Plymouth Harbor. MEP staff received four years-worth of monitoring data (2006, 2007, 2008, and 2009) with average monthly effluent flows and total nitrogen concentrations from MassDEP and additional clarifying data from the Town of Plymouth regarding their division of effluent between the outfall and the discharge beds at the WWTF (Kim Tower, Town of Plymouth, 1/17). These flow and concentrations were used to develop annual loads for each of the treatment facilities. These annual loads were averaged and incorporated into the PDK watershed nitrogen loading model on the wastewater discharge sites for each facility.

Properties within the PDK watershed that were identified through town parcel information as having sewer connections were not assigned a wastewater nitrogen load. All other properties were assumed to utilize on-site septic systems and were assigned a wastewater load based average water uses identified in the town databases.

Table IV-3. Wastewater Treatment Facilities with MassDEP Ground Water Discharge Permits within the Plymouth Harbor - Duxbury Bay - Kingston Bay MEP Watershed.

DEP Facility #	Town	Description	Monitoring Data: Beginning Yr	End Yr	Frequency	Mean Flow (gallon per day)	Permit Flow (gallon per day)
500	Duxbury	Duxbury School Complex	2006	2009	Monthly	9,530	30,500
433	Duxbury	Villages at Duxbury	2006	2009	Monthly	34,502	54,000
191	Kingston	Town & Country Mobile Home	2006	2009	Monthly	17,671	31,400
394	Kingston	Silver Lake Regional HS	2006	2009	Monthly	5,985	30,000
417	Kingston	Independence Mall	2006	2009	Monthly	19,873	90,000
462	Kingston	Evanswood	2006	2009	Monthly	18,122	65,000
494	Kingston	Summer Hill Plaza	2006	2008	Monthly	7,331	10,000
659	Kingston	Kingston WWTF	2006	2009	Monthly	240,855	907,000
226	Plymouth	Summer Hill Condo	2006	2009	Monthly	23,037	48,970
665	Plymouth	Sunrise Assisted Living	2006	2009	Monthly	5,115	13,500
677	Plymouth	Town of Plymouth WWTF	2006	2009	Monthly	1,729,688	3,450,000

Note: flow information for all facilities, except Plymouth WWTF, supplied by MassDEP (personal communication, Brian Dudley, SERO, April 2010). Plymouth WWTF data supplied by Town of Plymouth (Kim Tower, Department of Marine and Environmental Affairs, January 2017).

As noted above, the Town of Plymouth WWTF utilizes both on-site discharge beds and an outfall pipe for treated effluent disposal. The location of the outfall pipe discharge is shown in Figure IV-1. Review of the data supplied by the Town of Plymouth showed that 91% of the 1.7 MGD effluent produced by the WWTF was discharged through the outfall pipe. The associated nitrogen load from the WWTF was divided between the outfall pipe and the on-site discharge beds based on the information supplied by the town.

Nitrogen Loading Input Factors: Fertilized Areas, Golf Courses, and Agriculture

The second largest source of watershed nitrogen loading to estuaries is usually fertilized areas: lawns, golf courses, and cranberry bogs. Residential lawns are usually the predominant source within this category. In order to add this source to the watershed nitrogen loading model for the PDK system, MEP staff reviewed available regional information about residential lawn fertilizing practices and incorporated site-specific information for cranberry bogs, golf courses, and agricultural areas in the watershed. Cranberry bog nitrogen loading was determined based on previous studies conducted in southeastern Massachusetts.

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

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

In addition to residential fertilizer nitrogen within the watershed, there are also eight golf courses. MEP and Town of Plymouth staff tried to contact superintendents at each golf course in order to obtain course and turf specific nitrogen fertilizer application information. Two of the golf courses responded: Plymouth Country Club and Old Sandwich Golf Club. At the Plymouth Country Club, the following annual nitrogen application rates (in lbs/1,000 sq. ft.) were reported for the various turf areas: greens, 1.7; tees, 5.8; fairways, 1.2, and rough, 0.4 (personal communication, W. Weldon, 12/09). At the Old Sandwich Golf Club, application rates were: greens, 2.5; tees, 3.75; fairways, 2.25, and rough, 2.25 (personal communication, S. McCormick, 1/10). MEP staff reviewed the layout of each golf course from aerial photographs, classified the various turf types, determined which subwatershed turf was located in, applied a standard MEP 20% leaching rate, and developed course and subwatershed-specific nitrogen loads.

For the six other golf courses without reported nitrogen application rates, turf types were classified and assigned to various subwatersheds based on review of aerial photographs. Nitrogen application rates and loads for each of these golf courses were determined from average application rates developed from 23 courses collected during the MEP. The average turf application rates applied to the six other golf courses were (in lbs/1,000 sq. ft.): greens, 3.5; tees, 3.5; fairways, 3.2, and rough, 2.4.

Recent quantitative work on local cranberry bogs has indicated that non-flow through bogs lose less nitrogen to downgradient systems than those with regular or continuous flow through the bogs. After reviewing previously existing and new studies of nitrogen export from regional cranberry bogs (*e.g.*, Howes and Teal, 1995; DeMoranville, *et al*., 2009), MEP staff refined the nitrogen loading factors assigned to cranberry bogs based on whether water continuously flowed through the bog or was pumped or diverted onto the bog from an outside source of water (non-flow through bogs). Based on this refinement, non-flow through bogs were assigned a downstream nitrogen loss of 6.95 kg/ha/yr, while flow-through bogs were assigned a nitrogen load of 23.1 kg/ha/yr. In order to distinguish between flow through and non-flow through bogs, MEP staff reviewed available aerial photographs and classified each bog for the purposes of the watershed nitrogen loading model. Review of historic aerial photographs also shows that growers are regularly changing their bogs and many of them have been reconfigured to achieve non-flow through configurations. The areas of each bog are based on a MassDEP GIS coverage that is maintained by MassDEP for Water Management Act permitting (personal communication, Jim McLaughlin, MassDEP SERO, 1/13).

MEP staff also reviewed aerial photographs to determine the area of agricultural fields. This review identified 241 acres of agricultural fields. This review indicated that most of these fields were either pasture or hay. Both of these types of fields were assigned a nitrogen application rate of 5 kg/ha/yr, which is the MEP standard for these types of agricultural fields.

Nitrogen Loading Input Factors: Landfill Nitrogen Loads

MEP staff contacted MassDEP to obtain any nitrogen monitoring data for solid waste sites within the PDK system watershed. MassDEP has seven sites listed in their solid waste database that are located within the PDK watershed. Among these, three have available monitoring data: Duxbury Landfill, Kingston Landfill, and the Plymouth South Street Landfill (Mark Dakers, SERO, personal communication, 9/10). Development of nitrogen loads for each of these sites is based on the available monitoring data that is discussed in this section.

<u>Duxbury Landfill</u>

The Duxbury Landfill is located within the Bluefish River subwatershed (subwatershed #8). According to the MassDEP database, the landfill is capped, but not lined, and occupies 9.7 acres. MassDEP provided six sampling runs of biannual compliance monitoring data between November 2004 and April 2008, as well as groundwater elevation data, a map of well locations, and an interpretative groundwater contour map (Weston and Sampson, 2010a). MEP staff reviewed contaminant concentrations in wells along the prospective downgradient flow path.

The available Duxbury landfill groundwater monitoring data includes nitrate-nitrogen concentrations, but does not include total nitrogen or ammonium-nitrogen data. MEP staff estimated the rest of the dissolved nitrogen concentration during each sampling run based on alkalinity concentrations and the relationship between alkalinity concentrations and ammonium-nitrogen concentrations from groundwater monitoring of the Town of Brewster landfill (Cambareri and Eichner, 1993). After calculation, the estimated ammonium-nitrogen concentrations are added to the measured nitrate-nitrogen concentrations to provide an estimate of dissolved inorganic nitrogen (DIN), which is also used as an estimate of total nitrogen.

Based on the estimates, DIN concentrations in the downgradient wells ranged between 0.42 mg/L and 11.46 mg/L. MEP staff selected the wells with the two highest concentrations and, using the regional PDK recharge rate and the landfill area, determined the annual nitrogen load from the Duxbury Landfill was 247 kg/yr. This load was added to the subwatershed #8 load.

It is acknowledged that this approach for estimating a nitrogen load includes a number of assumptions, but it is appropriate based on the available data. A detailed assessment of all the available data is beyond the scope of the project, but staff balanced reasonable estimates of the various factors based on the general MEP guidance from MassDEP to include conservatism in nitrogen loading estimates when uncertainty exists in the data. A more refined evaluation and assessment of the established monitoring well network, including, at a minimum, analysis of total nitrogen concentrations and well construction details, would help to refine this assessment and future management options.

Kingston Landfill

The Kingston Landfill is located within the Foundry Pond Stream subwatershed (subwatershed #29). According to the MassDEP database, the landfill is capped, but not lined, and occupies 35 acres. MassDEP provided six sampling runs of biannual compliance monitoring data between October 2007 and April 2010, as well as groundwater elevation data, a map of well locations, and an interpretative groundwater contour map (Weston and Sampson, 2010b).

Direction of groundwater flow at this site appears to be complicated. A small stream flows along the south-southeastern edge of the site and water level readings in the stream suggest that groundwater flow should be toward the stream. However, water table elevations on the landfill site show flow away from the stream. With this in mind, MEP staff reviewed all contaminant concentrations in the available monitoring data.

The available Kingston landfill groundwater monitoring data includes nitrate-nitrogen concentrations, but does not include total nitrogen or ammonium-nitrogen data. MEP staff estimated the rest of the dissolved nitrogen concentration during each sampling run based on alkalinity concentrations and the relationship between alkalinity concentrations and ammonium-nitrogen concentrations from groundwater monitoring of the Town of Brewster landfill (Cambareri and Eichner, 1993). After calculation, the estimated ammonium-nitrogen concentrations are added to the measured nitrate-nitrogen concentrations to provide an estimate of dissolved inorganic nitrogen (DIN), which is also used as an estimate of total nitrogen.

Based on the derived estimates, DIN concentrations in the four most contaminated wells ranged between 0.94 mg/L and 1.76 mg/L. In addition, MEP staff reviewed historic aerials and estimated the solid waste area of 13.9 acres. MEP staff selected these four wells and, using the regional PDK recharge rate and the estimated solid area, determined the annual nitrogen load from the Kingston Landfill was 46 kg/yr. This load was added to the subwatershed #29 load.

It is acknowledged that this approach for estimating a nitrogen load includes a number of assumptions, but it is appropriate based on the available data. A detailed assessment of all the available data is beyond the scope of the project, but staff balanced reasonable estimates of the various factors based on the general MEP guidance from MassDEP to include conservatism in nitrogen loading estimates when uncertainty exists in the data. A more refined evaluation and assessment of the established monitoring well network, including, at a minimum, analysis of total nitrogen concentrations and well construction details, would help to refine this assessment and future management options.

Plymouth Landfill

The Plymouth Landfill is located within the Eel River West subwatershed (subwatershed #58). According to the MassDEP database, the landfill is capped, but not lined, and occupies 97.5 acres. Review of current and historic aerial photographs shows that the landfill is located under the Shops at 5 Plaza and that the solid waste area was estimated to cover 49 acres. MassDEP provided two sampling runs of biannual compliance monitoring data from 2009, but did not provide a map of well locations, an interpretative groundwater contour map, or well construction details.

Based on the limited information, MEP staff decided to review the available monitoring data and selected the monitoring well (MW-3) with the highest contaminant concentrations. The available Plymouth landfill groundwater monitoring data includes nitrate-nitrogen concentrations, but does not include total nitrogen or ammonium-nitrogen data. MEP staff estimated the rest of the dissolved nitrogen concentration during each sampling run based on alkalinity concentrations and the relationship between alkalinity concentrations and ammonium-nitrogen concentrations from groundwater monitoring of the Town of Brewster landfill (Cambareri and Eichner, 1993). After calculation, the estimated ammonium-nitrogen concentrations are added to the measured nitrate-nitrogen concentrations to provide an estimate of dissolved inorganic nitrogen (DIN), which is also used as an estimate of total nitrogen.

Based on the estimates, DIN concentrations at MW-3 were 12.32 mg/L on 4/26/09 and 19.71 mg/L on 10/22/09. Using the regional PDK recharge rate and the estimated solid area, MEP staff determined the annual nitrogen load from the Plymouth Landfill was 2,177 kg/yr. This load was added to the subwatershed #58 load.

It is acknowledged that this approach for estimating a nitrogen load includes a number of assumptions, but it is appropriate based on the available data. A detailed assessment of all the available data is beyond the scope of the project, but staff balanced reasonable estimates of the various factors based on the general MEP guidance from MassDEP to include conservatism in nitrogen loading estimates when uncertainty exists in the data. A more refined evaluation and assessment of the established monitoring well network, including, at a minimum, analysis of total nitrogen concentrations and well construction details, would help to refine this assessment and future management options.

Nitrogen Loading Input Factors: Jones River Freshwater Wetlands Nitrogen Loads and Reconciling Flow

Nitrogen loads within the Jones River watershed included loads from freshwater wetlands. During the course of the MEP, staff has found a number of occasions where stream nitrogen loads in relatively high flow rivers in non-outwash hydrogeology have not matched MEP watershed nitrogen loading estimates (*e.g.*, Howes, *et al*., 2012). Since the MEP assessment approach is data-driven, MEP staff began the process of exploring the cause of these higher nitrogen loads by re-reviewed all of the data leading to the preliminary watershed loads, including the watershed delineations, the nitrogen loading inputs, and re-reviewing the streamflow and concentration data. These steps led to additional clarification of the flows in the Jones River and suggested that there was another nitrogen source in the Jones River watershed that was not included in the preliminary model. Evaluations in the Town Brook and Eel River watersheds did not have similar differences.

The Jones River watershed includes Silver Lake (subwatershed #20). Silver Lake has water withdrawals and additions as part of the Town of Brockton's water supply system (Gomez and Sullivan, 2013). Water levels in Silver Lake fluctuate significantly because of natural fluctuations, water supply withdrawals and additions; as a result, the Lake may go through significant periods where it does not discharge surface waters to the Jones River. MEP staff reviewed water levels in the Lake during the period that 604b stream gauging occurred (September 2003 to August 2004) and found that the Lake discharged to the Jones River approximately five months during that period (December 2003 to April 2004). Adjusting the watershed discharge from Silver Lake to account for this limitation resulted in excellent balance (<1% difference) between the estimated Jones River flow based on recharge within the USGS recharge area and measured 2003-04 flow.

While the water flow balanced between measured and estimated for the nitrogen loading model, the initial nitrogen load in the model was lower than the measured load. In the prior evaluations where nitrogen loading estimates were initially less than measured stream nitrogen loads, MEP staff identified extensive wetland and swamp lands surrounding streams and rivers feeding into the estuary as the most likely unaddressed nitrogen source within the watershed (*e.g.*, Howes, *et al*, 2012). Studies have indicated that the ability of river wetlands to attenuate nitrogen is directly related to their hydraulic residence times (*e.g.*, Jansson, *et al*., 1994; Perez, *et al*., 2011; Toet, *et al*., 2005) with longer residence times resulting in greater nitrogen reduction. Direct data in the overall MEP study area generally confirms this relationship with lower flow/longer residence times streams on the eastern portion of the overall MEP study area having greater nitrogen attenuation, as well as even greater attenuation in ponds and lakes, which have even longer residence times than streams (ponds and lakes typically have nitrogen attenuation rates of 50% or more).

This addition of nitrogen loads from surrounding wetlands seems to be associated with the underlying geology and how wetlands occur within different geologic settings. In most of the streams and rivers on Cape Cod, the Islands, eastern Buzzards Bay, and the Plymouth portion of the PDK watershed, the sandy aquifer-dominated, outwash-plain systems leading to MEP stream gauge locations typically have only limited fringing freshwater wetlands, comparatively low streamflows, and streamflow patterns that tend to be less influenced by rainfall events and more influenced by regional groundwater fluctuations. These streams generally produce N attenuation rates of 25 to 30%. In systems that are underlain by bedrock and till, like the Jones River, the groundwater flow paths to the river are shorter and the river flows tend to be flashier and more influenced by rainfall events. These rivers also tend to have higher flows. Because of all these characteristics, residence times in these wetlands are likely to be much shorter than those in the outwash plain stream wetlands.

In addition, reviews of river wetlands have indicated that there are threshold effects like those seen in estuaries and ponds. This means that they can become loaded with nitrogen and act as transformers of nitrogen (changing nitrate+nitrite to organic forms), but not attenuators of nitrogen (*e.g.*, USDA, 2011). This change appears to be related to the amount of nitrogen received, as well as inter-related factors such as hydraulic residence time, temperature, plant surface coverage, and plant density (*e.g.*, Hägg *et al*., 2011; Kröger, *et al*., 2009; Alexander, *et al*., 2008).

Based on insights gathered from previous MEP assessments, staff incorporated nitrogen loading from the wetland areas in the Jones River watershed by assigning the water surface nitrogen loading factor (1.09 mg/L TN) to the wetland areas identified in a MassGIS/MassDEP 1:12000 wetland coverage (available at MassGIS: http://www.mass.gov/mgis/wetdep.htm). The wetland areas in this coverage were corrected to remove surface waters for freshwater ponds and cranberry bogs since loads for each of these are calculated and added separately. For the purposes of the MEP assessment, the treatment of these wetlands as water surfaces is appropriately conservative without further data to refine the spatial differences in residence times, plant communities/densities and the role of seasonal impacts along the various streams and rivers in the Jones River watershed system. Addition of these loads created a reasonable balance between estimated and measured nitrogen loads in the Jones River.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas in the PDK assessment are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes, *et al*., 2001). The factors are similar to those utilized by the CCC's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and MassDEP's Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the USGS groundwater modeling effort (Masterson, *et al*., 2009). Factors used in the MEP nitrogen loading analysis for the PDK watershed are summarized in Table IV-4.

Road areas are based on GIS information developed by the Massachusetts Executive Office of Transportation, which provides road, sidewalk, and road shoulder widths for various road segments (April 2012 GIS coverage available through MassGIS). MEP staff utilized the GIS to sum these segments and their various widths by subwatershed. Project staff also checked this information against parcel-based rights-of-way.

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 subwatershed. Following the assigning of parcels, all large boundary parcels were examined individually and were split (as appropriate) in order to obtain less than a 2% difference between the total land area of each subwatershed and the sum of the area of the parcels within each subwatershed. This effort results in "parcelized" watersheds that can be more easily used during the development of management strategies and subsequent regulatory discussions.

The review of individual parcels straddling watershed boundaries includes corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Building footprints, for example, are generally based on available information contained in the respective town assessor's database. Project staff used the average single-family residence building footprint based on available properties in the Town of Plymouth (1,720 sq ft) for any single-family residential units without footprint information. Commercial and industrial footprints for properties without building footprint information are also based on average building coverage of individual lots with similar land uses within the respective towns. Individualized information for parcels with atypical nitrogen loading (condominiums, golf courses, etc.) was also assigned at this stage. It should be noted that small shifts in nitrogen loading due to the above assignment procedure generally have a negligible effect on the total nitrogen loading to the PDK estuary. The assignment effort is undertaken to better define sub-estuary loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

IV.1.3 Calculating Nitrogen Loads

Following the assignment of all parcels, subwatershed modules were generated for each of the 87 subwatersheds in the PDK study area. These subwatershed modules summarize, among other things: water use, parcel area, frequency, private wells, and road area. All relevant nitrogen loading data was assigned to each subwatershed. Individual sub-watershed information was then integrated to create the PDK Watershed Nitrogen Loading module with summaries for each of the individual 87 subwatersheds. The subwatersheds are generally paired with functional embayment/estuary units for the Linked Watershed-Embayment Model's water quality component.

Table IV-4. Primary Nitrogen Loading Factors used in the Plymouth Harbor - Duxbury Bay - Kingston Bay MEP watershed analyses. General factors are from MEP modeling evaluation (Howes, *et al*., 2001). Site-specific factors are derived from watershed-specific data.

Nitrogen Concentrations:	**mg/l**	**Recharge Rates:**	**in/yr**
Road Run-off	1.5	Impervious Surfaces	40
Roof Run-off	0.75	Natural and Lawn Areas	27.25
Natural Area Recharge	0.072	**Water Use/Wastewater:**	
Direct Precipitation on Embayments and Ponds	1.09	Existing developed single-family residential parcels wo/water accounts and buildout residential parcels:	213 gpd[2]
Wastewater Coefficient	23.63		
Fertilizers:			
Average Residential Lawn Size (sq ft)[1]	5,000	Existing developed parcels w/water accounts:	Measured annual water use
Residential Watershed Nitrogen Rate (lbs/lawn)[1]	1.08	Commercial and Industrial Buildings without/WU and buildout additions[3]	
Leaching rate	20%	Commercial	
Cranberry Bogs nitrogen release – flow through bogs (kg/ha/yr)	23.08	Wastewater flow (gpd/1,000 ft2 of building):	91
Cranberry Bogs nitrogen release – pump on/pump off bogs (kg/ha/yr)	6.95	Building coverage:	17%
Nitrogen Fertilizer Rate for vegetable crop applications based on loads determined in other MEP assessments		Industrial	
		Wastewater flow (gpd/1,000 ft2 of building):	27
		Building coverage:	20%
		Average Single Family Residence Building Size (sq ft)	1,720

Notes:
1) Data from MEP lawn study in Falmouth, Mashpee & Barnstable of over 2,000 lawns (2001).
2) Based on average measured flow in the MEP PDK watershed area
3) Based on characteristics of similarly classified properties with the Town of Plymouth

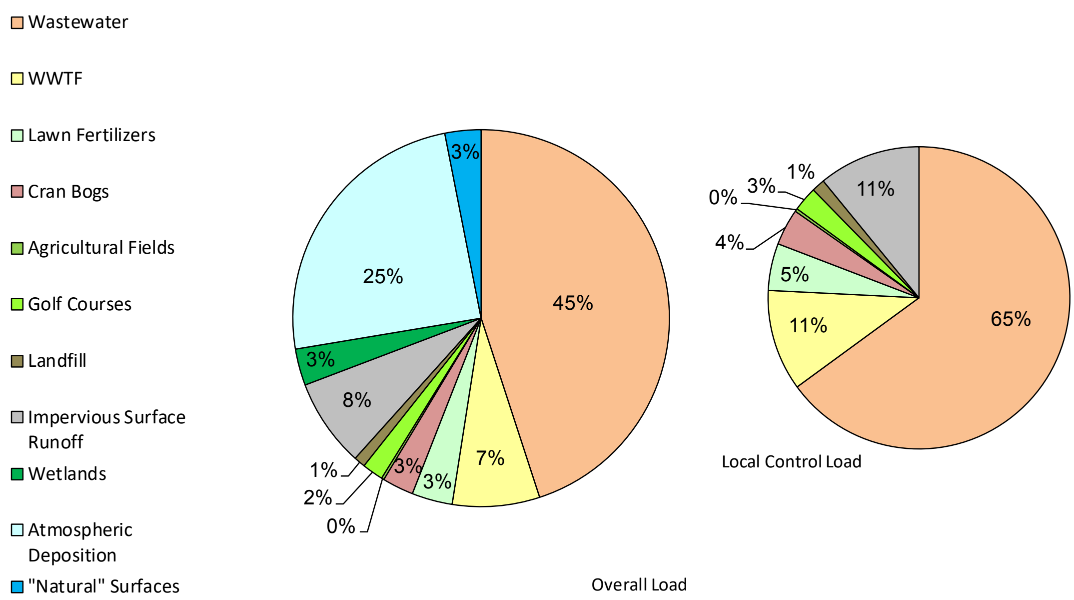
For management purposes, the aggregated estuary watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the PDK study area, the major types of nitrogen loads were: wastewater (*e.g.*, septic systems), wastewater treatment facilities, fertilizers (including contributions from agriculture and golf courses), impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-5). The output of the watershed nitrogen-loading model was the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-4). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation in streams and ponds during transport to the estuarine system before use in the embayment water quality sub-model.

Table IV-5. Plymouth Harbor - Duxbury Bay - Kingston Bay Watershed MEP Nitrogen Loads. Nitrogen loads are listed by various sources and by subwatershed. Unattenuated nitrogen loads are a sum of all sources without including natural nitrogen attenuation in fresh surface waters. Attenuated nitrogen loads are based on measured and assigned attenuation factors for upgradient streams and freshwater ponds. Stream attenuation factors are based on measured loads (see Section IV.2). All nitrogen loads are kg N yr^{-1}.

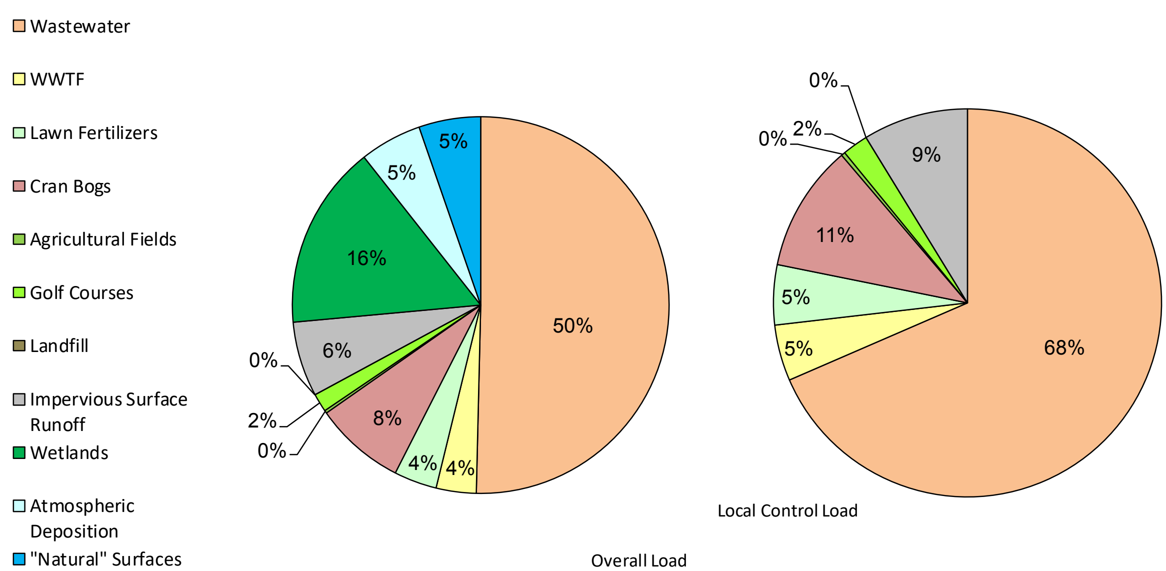
		Plymouth Harbor - Duxbury Bay - Kingston Bay N Loads by Input (kg/y):												% of Pond Outflow	Present N Loads			Buildout N Loads		
Watershed Name	shed ID#	Wastewater	WWTF	Lawn Fertilizers	Cran Bogs	Agricultural Fields	Golf Courses	Landfill	Impervious Surface Runoff	Wetlands	Atmospheric Deposition	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
PDK Whole System		**113,728**	**18,953**	**8,815**	**6,816**	**555**	**4,599**	**2,471**	**19,136**	**7,946**	**62,034**	**7,831**	**42,925**		**252,884**		**216,935**	**295,810**		**252,402**
PDK N TOTAL		19,273	80	1,292	802	-	1,177	247	2,041	-	24,410	1,151	5,161		50,473		50,357	55,634		55,501
Plymouth Harbor LT10N	1A	4,993	-	293	-	-	-	-	416	-	5	177	637		5,884		5,884	6,521		6,521
Duxbury Marsh TOTAL		8,440	80	707	653	-	252	-	1,088	-	2,198	637	3,417		14,055		14,055	17,471		17,471
Careswell Pond	2	361	-	25	-	-	-	-	47	-	129	31	318		594		594	912		912
Duxbury Marsh	3	8,079	80	681	653	-	252	-	1,041	-	29	606	3,098		11,421		11,421	14,519		14,519
Duxbury Marsh Estuary Surface											2,040				2,040		2,040	2,040		2,040
Blue Fish River TOTAL		5,839	-	292	149	-	925	247	537	-	599	336	1,108		8,926		8,811	10,034		9,901
North Hill Pond	4	188	-	14	60	-	98	-	22	-	476	51	62		910		910	972		972
Blue Fish River LT10	5	3,720	-	171	63	-	715	-	284	-	46	144	1,095		5,143		5,143	6,239		6,239
Duxbury PWS1	7	71	-	4	-	-	95	-	13	-	-	6	23		189		189	212		212
Duxbury PWS2	10	354	-	23	-	-	-	-	42	-	-	14	47		432		432	479		479
Duxbury PWS3	6	260	-	11	-	-	18	-	21	-	-	34	16		345		345	361		361
Bluefish River GT10 N	8	376	-	19	8	-	-	247	60	-	-	43	(247)		753		753	506		506
Bluefish River GT10 S	11	760	-	42	-	-	-	-	84	-	10	25	78		922		922	1,000		1,000
Island Creek Pond TOTAL	ICP	110	-	7	18	-	-	-	11	-	66	19	34	29%	231		116	265		133
Plymouth Harbor LT10N Estuary Surface											21,608				21,608		21,608	21,608		21,608
PDK Mid TOTAL		43,187	2,782	3,490	4,548	160	783	46	7,105	7,946	21,598	3,863	17,793		95,508		83,089	113,301		97,844
Plymouth Harbor LT10Mid	1B	14,364	947	1,358	521	15	-	-	2,526	-	135	738	4,195		20,603		20,603	24,799		24,799
Island Creek Pond TOTAL	ICP	264	-	17	43	-	-	-	27	-	159	45	82	71%	555		277	637		318
Blackwater Pond	21	657	-	11	55	-	-	-	52	109	36	36	87		956		956	1,042		1,042
Bracketts Pond TOTAL	BP	1,167	-	92	14	26	-	-	167	6	97	90	643	100%	1,660		767	2,303		1,017
Plymouth Harbor LT10Mid Estuary Surface											17,968				17,968		17,968	17,968		17,968
Jones River TOTAL		26,735	1,835	2,011	3,914	120	783	46	4,334	7,831	3,203	2,954	12,785		53,767		42,518	66,552		52,699
Foundry Pond Stream TOTAL		1,806	134	192	57	-	-	46	1,148	-	569	328	2,016		4,281		3,786	6,297		5,753
Foundry Pond Stream	29	1,642	134	186	35	-	-	46	952	-	51	140	1,777		3,187		3,187	4,964		4,964
Smelt Pond TOTAL	SP	83	-	6	22	-	-	-	76	-	505	128	84	100%	820		326	904		360
Kingston PWS1 TOTAL	PWS1	80	-	-	-	-	-	-	120	-	13	61	155		274		274	429		429
Jones River Gauge TOTAL		24,929	1,701	1,820	3,857	120	783	-	3,186	7,831	2,634	2,626	10,768		49,486		38,731	60,255	-	46,946
Jones River Gauge LT10	34	561	1,501	79	-	-	710	-	262	305	8	99	526		3,526		3,526	4,053		4,053
Jones River Gauge GT10	35	971	-	42	-	-	-	-	113	47	-	51	62		1,225		1,225	1,287		1,287
Jones River USGS Gauge TOTAL		23,397	201	1,699	3,857	120	73	-	2,810	7,479	2,625	2,476	10,180		44,735		33,980	54,915		41,606
Jones River USGS Gauge	23	15,157	201	1,018	3,111	21	73	-	2,126	4,243	273	1,193	6,232		27,417		27,417	33,649		33,649
Silver Lake	20	2,550	-	267	184	83	-	-	-	293	1,287	880	143	42%	5,545	50%	2,772	5,688	50%	2,844
Harrobs Corner Bog Pond	22	342	-	19	38	15	-	-	37	154	135	41	419	100%	781	50%	391	1,201	50%	600
Indian Pond	25	21	-	1	38	-	-	-	7	128	270	9	39	100%	475	50%	238	514	50%	257
Bay State Comp. Bog Reservoir	32	215	-	15	49	-	-	-	38	288	194	55	994	100%	855	50%	427	1,849	50%	924
Dennetts Pond	33	102	-	5	13	-	-	-	24	3	32	13	78	100%	192	50%	96	270	50%	135
Crossman Pond	38	416	-	41	-	-	-	-	40	-	85	12	45	100%	595	50%	297	640	50%	320
Pembroke St South Pond TOTAL	PSSP	4,594	-	332	423	-	-	-	537	2,369	348	272	2,231	100%	8,875	50%	2,342	11,106	50%	2,877

Table IV-5 (continued). Plymouth Harbor - Duxbury Bay - Kingston Bay Watershed MEP Nitrogen Loads. Nitrogen loads are listed by various sources and by subwatershed. Unattenuated nitrogen loads are a sum of all sources without including natural nitrogen attenuation in fresh surface waters. Attenuated nitrogen loads are based on measured and assigned attenuation factors for upgradient streams and freshwater ponds. Stream attenuation factors are based on measured loads (see Section IV.2). All nitrogen loads are kg N yr^{-1}.

		Plymouth Harbor - Duxbury Bay - Kingston Bay N Loads by Input (kg/y):												% of Pond Outflow	***Present N Loads***			***Buildout N Loads***		
Watershed Name	**shed ID#**	**Wastewater**	**WWTF**	**Lawn Fertilizers**	**Cran Bogs**	**Agricultural Fields**	**Golf Courses**	**Landfill**	**Impervious Surface Runoff**	**Wetlands**	**Atmospheric Deposition**	**"Natural" Surfaces**	**Buildout**		**UnAtten N Load**	**Atten %**	**Atten N Load**	**UnAtten N Load**	**Atten %**	**Atten N Load**
PDK S TOTAL		51,268	16,091	4,033	1,467	394	2,638	2,177	9,991	-	16,026	2,818	19,971		106,904		83,489	126,875		99,057
Plymouth Harbor LT10S	1C	10,228	66	1,354	-	-	-	-	2,539	-	-	331	4,000		14,517		14,517	18,517		18,517
Plymouth PWS2	31	160	-	34	-	-	-	-	81	-	-	3	89		279		279	368		368
Plymouth WWTF Outfall			14,412										513		14,412		14,412	14,925		14,925
Plymouth WWTF Deep Discharge (scenarios only)																				
Plymouth Harbor LT10S Estuary Surface											9,349				9,349		9,349	9,349		9,349
Spooner Pond Stream TOTAL		1,817	-	234	-	-	-	-	673	-	92	153	1,657		2,969		2,969	4,626		4,626
Spooner Pond Stream LT10	30	1,159	-	234	-	-	-	-	499	-	92	93	1,494		2,078		2,078	3,572		3,572
Spooner Pond Stream GT10	41	658	-	-	-	-	-	-	174	-	-	60	163		891		891	1,054		1,054
Town Brook Gauge TOTAL		29,414	388	1,898	503	30	206	-	4,371	-	2,923	873	9,209		40,606	0%	25,846	49,815	0%	32,273
Plymouth PWS1	39	975	-	42	-	-	-	-	590	-	-	54	584		1,660		1,660	2,245		2,245
Town Brook Gauge	44	6,920	-	437	41	-	-	-	1,351	-	55	207	3,272		9,010		9,010	12,282		12,282
Triangle Pond TOTAL	TP	1,363	-	50	-	-	-	-	232	-	98	33	564	67%	1,776		852	2,340		1,071
Little Pond TOTAL	LP	689	242	111	-	-	-	-	127	-	168	19	227	62%	1,356		539	1,582		600
Lout Pond TOTAL	Lout	8,850	66	570	212	14	93	-	943	-	1,226	261	2,049	100%	12,234		5,166	14,283		5,998
Billington Sea TOTAL	BS	10,617	80	689	251	17	113	-	1,129	-	1,377	298	2,513	55%	14,571		8,619	17,083		10,078
Eel River TOTAL		9,649	1,224	514	963	364	2,432	2,177	2,327	-	3,662	1,457	4,504		24,771	4%	16,116	29,274	4%	18,999
Eel River Gauge	53	-	-	-	-	-	-	-	5	-	1	1	-		8		8	8		8
Eel River 3A TOTAL		9,649	1,224	514	963	364	2,432	2,177	2,322	-	3,661	1,456	4,504		24,763		16,780	29,267		19,783
Eel River 3A	54	2,195	-	120	-	56	98	-	439	-	184	152	1,085		3,245		3,245	4,330		4,330
Howland Pond TOTAL		989	-	60	103	202	2,218	-	317	-	248	297	2,369		4,434	0%	2,233	6,803	0%	3,979
Howland Pond	55	172	-	7	-	124	-	-	28	-	46	16	295		391	50%	196	686		686
Eel River Mid	56	143	-	9	-	60	-	-	44	-	0	36	443		293		293	735		735
Cold Bottom Pond TOTAL	CBP	86	-	6	-	-	77	-	28	-	22	20	17	56%	239		119	256		128
Forge Pond TOTAL	FP	589	-	39	103	18	2,141	-	216	-	181	225	1,614	100%	3,512		1,626	5,126		2,430
Eel River West TOTAL		6,465	1,224	333	860	106	117	2,177	1,566	-	3,229	1,007	1,049		17,084	0%	11,302	18,134	0%	11,474
Eel River W	58	2,503	1,224	63	-	32	92	2,177	1,020	-	14	270	(652)		7,396		7,396	6,744		6,744
Hayden Mill Pond	59	34	-	-	-	74	-	-	7	-	41	5	93		161	50%	81	255		255
Well	68	7	-	0	123	-	-	-	5	-	63	18	62		216		216	278		278
Russell Mill Pond TOTAL	RMP	3,615	-	251	604	-	25	-	482	-	1,948	557	1,345	100%	7,482		3,005	8,827		3,530
Cooks Pond TOTAL	CP	172	-	11	107	-	-	-	32	-	723	101	122	100%	1,145		329	1,267		361
South Triangle Pond TOTAL	STP	35	-	3	8	-	-	-	8	-	263	19	41	72%	335		109	376		122
Little South Pond TOTAL	LSP	99	-	6	18	-	-	-	12	-	176	37	38	43%	349		166	387		184

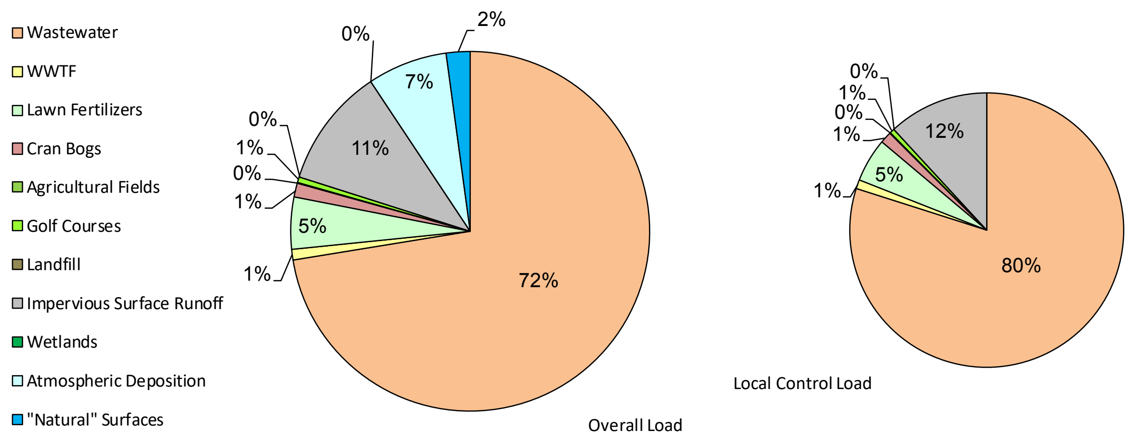


A. Whole Plymouth Harbor - Duxbury Bay - Kingston Bay System

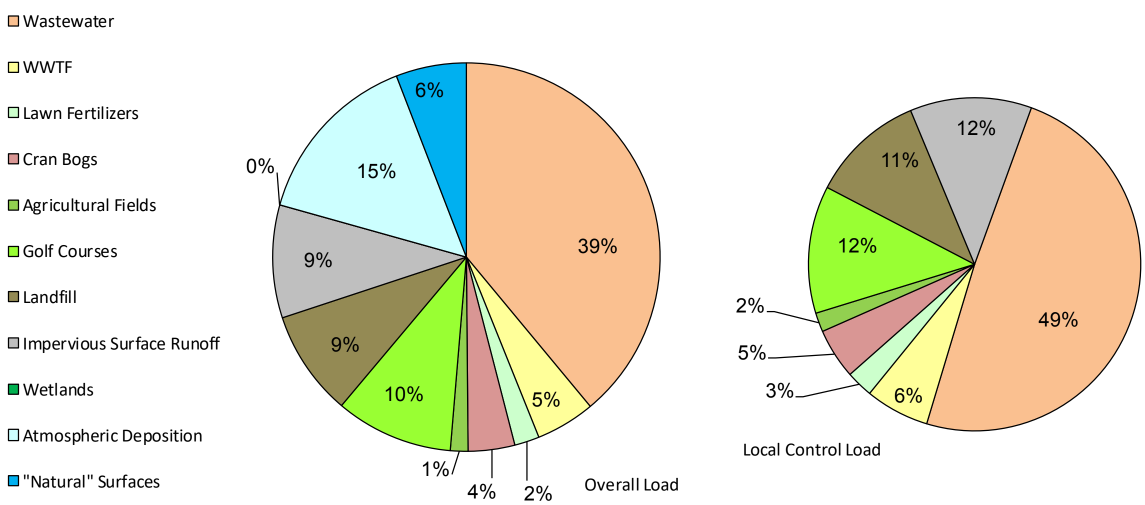


B. Jones River Gauge Subwatershed

Figure IV-4 (A,B). Source-specific unattenuated watershed nitrogen loads (by percent) to the A) whole Plymouth Harbor - Duxbury Bay - Kingston Bay Watershed and B) Jones River Gauge subwatershed. “Overall Load” is the total nitrogen input within the watershed, while the “Local Control Load” represents only those nitrogen sources that could potentially be under local regulatory control. Comparative sizes of pies represent reduction from overall load to local control load.



C. Town Brook Gauge Subwatershed



D. Eel River Gauge Subwatershed

Figure IV-4 (C,D). Source-specific unattenuated watershed nitrogen loads (by percent) to the C) Town Brook Gauges subwatershed and D) Eel River Gauge subwatershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control. Comparative sizes of pies represent reduction from overall load to local control load.

Freshwater Pond Nitrogen Loads

Freshwater ponds are one of the watershed locations where natural nitrogen attenuation occurs and this attenuation is included in the MEP watershed nitrogen loading model. Freshwater ponds in aquifer systems like those in the Plymouth-Carver-Kingston-Duxbury Aquifer are generally kettle-hole depressions that intercept the water table of surrounding groundwater. Groundwater typically flows into the pond along the upgradient shoreline, then lake water flows back into the groundwater system along the downgradient shoreline. Occasionally these ponds will also have a stream outlet or herring run that also acts as a discharge point; many of the ponds in the PDK watershed are connected to each other through streams and rivers, as well as connections that have been developed for cranberry bog operations.

Since watershed nitrogen loads flow into the ponds along with the groundwater, the pond biomass (plants and animals) have the opportunity to incorporate some of the nitrogen, as well as transporting/burying some of it to the pond sediments. As the nitrogen is captured and used in the pond ecosystem, it is also changed amongst its various oxidized and reduced forms. These interactions also allow for some chemical denitrification and release of some of the nitrogen to the atmosphere, as well as permanent burial in the pond sediments of some portion of the load that the pond receives. Through the cumulative effect of these interactions with the pond ecosystem, some of the nitrogen from the pond watershed is removed and is not transferred downgradient or downstream to the estuarine receiving water. If this reduced (or attenuated) load does not encounter any streams or other ponds, it will eventually discharge to the downgradient embayment. If it enters another pond or stream prior to discharge, this load can be further attenuated (see Section IV.2 for stream attenuation). In the nitrogen loading summary in Table IV-3, the unattenuated loads are those without any natural nitrogen attenuation included, while the attenuated loads include the attenuation within ponds, streams, and, in some cases, the cumulative effect of attenuation within a number of ponds and streams as the water moves toward discharge into the estuary.

Nitrogen attenuation in freshwater ponds has generally been found to be at least 50% in MEP analyses, so this value is generally used as a standard MEP default attenuation rate when sufficient pond-specific data is not available. Detailed studies of southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE, 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM, 2001) have supported a 50% attenuation factor as a reasonable, somewhat conservative rate. However, in some cases, if sufficient monitoring information is available, a pond-specific attenuation rate is incorporated into the watershed nitrogen loading modeling [*e.g.*, 87%, Mystic Lake; 40%, Middle Pond; and 52%, Hamblin Pond in the Three Bays MEP Report (Howes, *et al.*, 2006)]. In order to estimate nitrogen attenuation in the ponds, available physical and water quality data for each pond is reviewed. Available bathymetric information is reviewed relative to measured pond temperature profiles to determine whether an epilimnion (*i.e.*, well mixed, uniform temperature, upper portion of the water column) exists in each pond. This step is completed to assess whether available data is influenced significantly by nitrogen regeneration from the pond sediments. Bathymetric information is necessary to develop a residence or turnover time and complete an estimate of nitrogen attenuation. Collectively, a standard 50% nitrogen attenuation rate is assigned to ponds with delineated watersheds in MEP nitrogen loading models unless sufficient information is available regarding the physical structure of the pond and its water quality conditions to reasonably assign a different pond-specific rate.

In the PDK watershed, MEP staff worked with Town of Plymouth staff and reviewed available MassDEP and other data sources for available pond monitoring and physical characterization data. This effort was recently supplemented with the completion of the Plymouth

Pond and Lake Atlas (Eichner, *et al.*, 2015), which included completion of a comprehensive database of Plymouth ponds and lakes and a review of available water quality and bathymetric data. The database indicated that the Town has 450 freshwater ponds with 83 Great Ponds (publicly-owned ponds with areas of 10 acres or more). The creation of the Atlas was also accompanied by the creation of the Plymouth Pond and Lakes Stewardship (PPALS) Program, which began with the snapshot sampling of 38 ponds during the late summer of 2014. Data reviewed during the course of the Atlas preparation and pond data in other towns for the MEP assessment of the PDK watershed found that among the 45 freshwater ponds with delineated subwatersheds, data is generally limited to selected bathymetric maps and limited, snapshot water quality monitoring. This data is insufficient for alternative pond-specific nitrogen attenuation rates. For this reason, the standard MEP 50% attenuation was assigned to all freshwater ponds with delineated subwatersheds.

Since groundwater outflow from a pond can enter more than one downgradient sub-watershed, the length of shoreline on the downgradient side of the pond was used to apportion the pond-attenuated nitrogen load to respective downgradient watersheds. The apportionment was based on the percentage of discharging shoreline bordering each downgradient sub-watershed. So for example, Billington Sea has a downgradient shoreline of 6,264 feet; 45% of that shoreline discharges into the Lout Pond subwatershed (watershed #45 in Figure III-2) and 55% discharges to the Town Brook Gauge subwatershed (watershed #44). The attenuated nitrogen load discharging from Billington Sea is divided among these sub-watersheds based on these percentages of the downgradient shoreline.

Buildout

Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment (or scenario) of potential development and accompanying nitrogen loads within the study area watersheds. The MEP buildout is relatively straightforward and is generally completed in three steps: 1) each residential parcel classified by the town assessor as developable is identified and divided by minimum lot sizes specified in town zoning and the resulting number of new residential units is rounded down, 2) parcels classified as developable commercial and industrial parcels by the town assessor are identified, and 3) residential, commercial and industrial parcels with existing development and areas greater than twice zoning's minimum lot size are identified, divided by the minimum lot size and the resulting number of new units is rounded down.

It should be noted that the initial MEP buildout approach is relatively simple and does not include any modifications/refinements for lot line setbacks, wetlands, road construction, frontage requirements, parcel shape requirements, or other more detailed zoning provisions. The MEP buildout approach also does not include potential impacts associated with the higher densities usually associated with 40B affordable housing projects. The approach includes provisions to maintain current commercial and industrial uses. Chapter 61A lands (land use code 601), which tend to be forest lands in "agricultural use" are assumed to remain in this use at buildout. Data on permanently protection open space is also incorporated if available.

As an example of how the MEP approach might apply, assume an 81,000 square foot lot is classified by the town assessor as a developable residential lot (land use code 130). This lot is divided by the 40,000 square foot minimum lot size specified in town zoning and the result is rounded down to two. As a result, two additional residential lots would be added to the subwatershed in the MEP buildout scenario.

Other provisions of the MEP buildout assessment include undevelopable lots, commercial and industrial properties, and lots less than the minimum areas specified by zoning. Properties classified by the town assessor's as "undevelopable" (*e.g.*, MassDOR codes 132, 392, and 442) are not assigned any development at buildout. Commercial and industrial properties classified as developable are not subdivided; the area of each parcel and the factors in Table IV-4 are used to determine a building size and wastewater flow for these properties. Pre-existing lots classified by the town assessor as developable are also treated as developable even if they are less than the minimum lot size specified in zoning; so, for example, a 10,000 square foot lot classified by the town assessor as a developable residential property (130 land use code) will be assigned an additional residential dwelling in the MEP buildout scenario even if the minimum lot size required by the zoning in the area is 40,000 square feet. Most town zoning bylaws have a lower minimum lot size for pre-existing lots (usually 5,000 square feet) that will minimize instances of regulatory takings. Existing developed residential properties that are larger than zoning's minimum lot sizes are also assigned additional development potential only if enough area is available to accommodate at least one additional lot as specified by the zoning minimum.

All the parcels with additional buildout potential within the overall PDK watershed are shown in Figure IV-5. Overall, this buildout includes a projected 4,692 additional residences at buildout, 10,805,535 square feet of additional commercial properties and 27,131,170 square feet of additional industrial properties. Each additional residential, commercial, or industrial property added at buildout is assigned nitrogen loads for wastewater and impervious surfaces. Residential additions also include lawn fertilizer nitrogen additions. All wastewater loads are assumed to come from standard on-site septic systems unless the parcel is designated as already having a sewer connection (for additional development on existing lots) or identified within an existing sewer service area. Cumulative unattenuated buildout loads for each subwatershed are indicated in a separate column in Table IV-5. Buildout additions within the PDK watersheds will increase the unattenuated loading rate by 17%.

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 relative to the tidal flushing and nitrogen cycling within the embayment basins. This watershed nitrogen input parameter is the primary term used to relate present and future loads (build-out, sewering analysis, enhanced flushing, pond/wetland restoration for natural attenuation, etc.) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the sub-watersheds of the Plymouth Harbor, Kingston Bay and Duxbury Bay system being investigated under this nutrient threshold analysis were based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1).

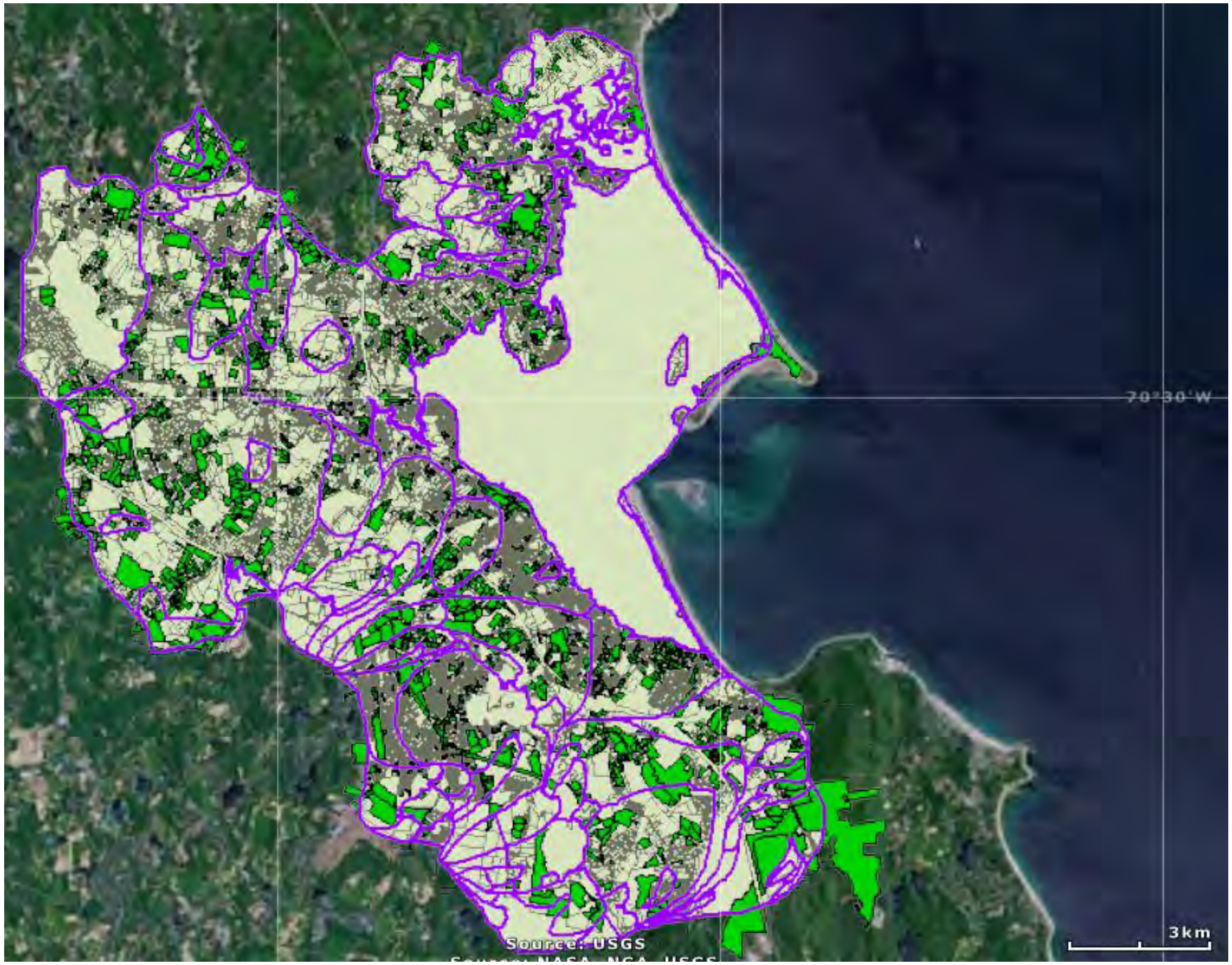


Figure IV-5. Developable Parcels in the Plymouth Harbor - Duxbury Bay - Kingston Bay Watershed. Parcels colored green are parcels with additional development potential based on current zoning. Developable parcels are based on town assessor classifications of developable properties and minimum lot sizes specified in town zoning; these parcels are assigned estimated nitrogen loads in MEP buildout calculations. Details on additional development assigned to individual parcels are available in the MEP Data Disk that accompanies this report.

If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watersheds where nitrogen transport from source to estuarine waters is through groundwater flow in sandy outwash aquifers (such being the case in the developed region of southeastern Massachusetts and Cape Cod). The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, nitrogen passes through a surface water ecosystem (pond, wetland, stream) on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) can be diminished by natural biological processes that represent removal (not just temporary storage). However, this potential natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes to varying degrees based on habitat and residence time. In the watershed for the Plymouth-Kingston-Duxbury Embayment System, a portion of the freshwater flow and transported nitrogen passes through several significant surface water systems (e.g. Jones River, Town Brook and Eel River) prior to entering the estuary, providing the opportunity for significant nitrogen attenuation under appropriate conditions (Figure IV-6).

Failure to determine the attenuation of watershed derived nitrogen overestimates the nitrogen load to receiving estuarine waters. If nitrogen attenuation is significant in one portion of a watershed and insignificant in another the result is that nitrogen management would likely be more effective in achieving water quality improvements if focused on the watershed region having unattenuated nitrogen transport (other factors being equal). In addition to attenuation by freshwater ponds (see Section IV.1.3, above), attenuation in surface water flows is also important. An example of the significance of surface water nitrogen attenuation relating to embayment nitrogen management was seen in the Agawam River, where >50% of nitrogen originating within the upper watershed was attenuated prior to discharge to the Wareham River Estuary (CDM 2000). Similarly, MEP analysis of the Quashnet River (Town of Falmouth, Cape Cod) indicated that in the upland watershed, which has natural attenuation predominantly associated with riverine processes, the integrated attenuation was 39% (Howes et al. 2004). In addition, a preliminary study of Great, Green and Bournes Ponds in Falmouth, measurements indicated a 30% attenuation of nitrogen during stream transport (Howes and Ramsey 2001). An example where natural attenuation played a significant role in nitrogen management can be seen relative to West Falmouth Harbor (Falmouth, MA), where ~40% of the nitrogen discharge to the Harbor originating from the groundwater effluent plume emanating from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Based on MEP assessments of numerous streams/river systems of various sizes across the MEP study region, not only can pond and stream attenuation be a significant load reducing process, it varies significantly from system to system making it all the more important to quantify this term on a site specific basis (Table IV-6) as was done for the Jones River, the Eel River and Town Brook discharging from the overall Plymouth-Kingston-Duxbury embayment watershed. Proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment (attenuated), not just loaded to the watershed (unattenuated).

Figure IV-6. Location of Stream gauges (yellow symbols) in the Plymouth Harbor-Kingston Bay-Duxbury Bay embayment system. Two stream gauge locations (Jones River and Eel River) did have historic stream flow measurements completed by the US Geological Survey for comparative purposes.

Table IV-6 Summary of measured stream attenuation values determined by the MEP in 30 different surfacewater systems across southeastern Massachusetts and of varying annual flow rates ranging from small, medium and large.

Statistics	Un-Attenuated Watershed Loading (kg/yr) (N-Loading Model)	Discharged Load (kg/yr) to Estuary (measured)	Percent Attenuation
	Large River Systems (Flow = 10 x Million m3/yr)		
Min	13537	7541	8%
Max	87956	74397	44%
Average	35610	27434	23%
Std. Deviation	28757	23553	13%
Range	13537 - 87956	7541 - 74397	8% - 44%
N	7	7	7
	Medium River Systems (Flow = 1 x Million m3/yr)		
Min	1624	1095	0.3%
Max	12518	12158	74%
Average	5751	4125	34%
Std. Deviation	3497	3530	26%
Range	1624 - 12518	1095 - 12158	0.3% - 74%
N	13	13	13
	Small River Systems (Flow = 100 x Thousand m3/yr)		
Min	121	105	0%
Max	3719	2209	67%
Average	1252	716	37%
Std. Deviation	1012	576	21%
Range	121 - 3719	105 - 2209	0% - 67%
N	10	10	10

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements of upper watershed attenuation were undertaken as part of the MEP Approach in the Plymouth-Kingston-Duxbury Embayment System. MEP conducted long-term measurements of natural attenuation relating to the most significant surface water discharges to the estuary in addition to the natural attenuation measures by fresh kettle ponds, addressed above (Section IV.1). These additional site-specific studies were conducted in the 3 major surface water flow systems in the overall embayment watershed, 1) Jones River discharging to Kingston Bay, 2) Town Brook discharging to Plymouth Harbor and 3) Eel River discharging to Plymouth Harbor. Measurement of the flow and nutrient load associated with the Jones River (at Route 3A/Main St. and Brook Street), Town Brook (between Main Street and Sandwich Street bridge crossings in downtown Plymouth) and Eel River (immediately down gradient of the Route 3A bridge) provide a direct integrated measure of all of the processes presently attenuating nitrogen in the sub-watersheds upgradient from the gauging sites.

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, corrected for the time of travel as appropriate (Section IV.1). Measurement of the flow and nutrient load associated with the freshwater streams discharging to the estuary provides a direct integrated measure of all of the processes presently attenuating nitrogen in the contributing area up-gradient from the various gauging sites. Flow and nitrogen load were measured at the gauges in each freshwater stream site for ~ 20 months of record. During each stream study period, velocity profiles were completed on each surface water inflow every month to two months. The summation of the products of stream subsection areas of the stream cross-section and the respective measured velocities represent the computation of instantaneous stream flow (Q).

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

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

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

where by:

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

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

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

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

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Jones River Discharge to Kingston Bay

Similar to other surface water features in the MEP study region that typically emanate from a specific pond or wetland, Jones River, which discharges into the Kingston Bay portion of the overall embayment system, does have a network of up-gradient bog/wetland and pond areas that contribute surface water to this significant river of southeastern Massachusetts. Based on numerous previous studies completed by the MEP on other systems in southeastern Massachusetts, the outflow from the bog/wetlands/ponds and the wooded areas up-gradient of the Jones River gauge very likely remove nitrogen from the water passing through them and also provides for a direct measurement of this nitrogen attenuation. The combined rate of nitrogen attenuation by the biological processes that occur in the various surface water features was determined by comparing the present predicted (calculated from the land use model, Section IV.1) nitrogen load to the sub-watershed region contributing to the bog/wetlands and wooded areas above the gauge site and the measured annual Jones River discharge of nitrogen to Kingston Bay, Figures IV-6,IV-7.

At the Jones River gauge site (established at the Main Street bridge crossing), a continuously recording vented calibrated water level gauge was installed to yield the level of water in the channel that carries the flows and associated nitrogen load to the estuarine system. As the lower reach of Jones River is tidally influenced down gradient of Route 3, the stage record from the gauge was checked to make sure there was no tidal influence in the record at low tide. To confirm that freshwater was being measured at low tide, the stage record was analyzed for any semi-diurnal variations indicative of tidal influence and salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average salinity of the water samples taken from Jones River at Main Street at low tide was determined to be 0.1 ppt. Therefore, the gauge location was deemed acceptable for making freshwater flow measurements at low tide. Calibration of the gauge was checked monthly. The gauge on Jones River was installed on July 23, 2003 and was set to operate continuously for a complete hydrologic year (low flow to low flow, ~12 months). Stage data collection continued until February 16, 2005 for a total deployment of 19 months.

Surface freshwater flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for Jones River at the gauge site located immediately up-gradient of the Main Street bridge, based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the estuary and is reflective of the biological processes occurring in the stream channel and the network of bogs/wetlands/ponds and wooded areas contributing to nitrogen attenuation (Figure IV-8b,c and Table IV-7 and IV-8). In addition, a water balance was constructed based upon the U.S. Geological Survey/MEP defined watershed delineations to determine long-term average freshwater discharge expected at each gauge site based on area and average recharge.

Figure IV-7. Location of MEP stream gauges (yellow symbol) for measuring flow and nitrogen loads transported by the Jones River. Jones River receives surfacewater from a network of up-gradient bog/wetland/pond features. USGS gauging location for comparative flow measurements is denoted by red symbol.

The annual freshwater flow record for Jones River as measured by the MEP was compared to the long-term average flows determined by the USGS/MEP modeling effort (Table III-1). The measured freshwater discharge from Jones River at the Route 3A gauge location was nearly identical to (~1% above) the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2003 and ending in August 2004 (low flow to low flow) was 102,201 m^3/day compared to the long term average flows determined by the MEP watershed modeling effort (101,216 m^3/day). It should be noted that daily flows calculated using the rating curve developed by the MEP were confirmed relative to a historical record of daily flows developed by the US Geological Survey at a USGS maintained gauging station approximately 1 km up-gradient of the MEP gauge deployed on the Jones River. As depicted in Figure IV-8a predicted daily flows developed under the 604(b) grant agree within 14 percent of the USGS daily flows for the overlapping period of record. Additionally, predicted daily flows agree favorably with measured flows used in the development of the rating curve. The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Jones River discharging from the sub-watershed indicates that the river is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within the Jones River outflow were high, 1.04 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 106.4 kg/day and a measured total annual TN load of 38,837 kg/yr. By comparison Nitrate + Nitrite (NOx) was on average 0.493 mg N L^{-1}. In the Jones River, nitrate was the predominant form of nitrogen (50 %), 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. Dissolved inorganic nitrogen (DIN) was the next most abundant nitrogen species with an average of 0.521 mg N L^{-1} (51 % of the Total Nitrogen pool) followed by dissolved organic nitrogen (DON) with an average concentration of 0.413 mg N L^{-1} (41 % of the Total Nitrogen pool). Figures IV-8b,c depicts the daily freshwater flow in the Jones River relative to the concentrations of Total Nitrogen (TN) and Nitrate + Nitrite (NOx) as determined from the weekly water quality sampling at the gauge as supported by the MEP.

From the measured nitrogen load discharged by Jones River to the Kingston Bay portion of the overall system and the nitrogen load determined from the watershed based land-use analysis, it appears that there is moderate nitrogen attenuation of upper watershed derived nitrogen during transport to Jones River and the down gradient estuary. Based upon the lower total nitrogen load (38,837 kg yr^{-1}) discharged from Jones River at Route 3A compared to that added by the various land-uses to the associated watershed (49,486 kg yr^{-1}), the integrated attenuation in passage through the stream and up-gradient freshwater ponds and wetlands prior to discharge to the estuary is 22% (i.e. 22% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams evaluated under the MEP is expected given the nature of the up-gradient ponds/wetlands/bogs capable of attenuating nitrogen. The directly measured nitrogen load from Jones River was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

Table IV-7. Comparison of water flow and nitrogen load discharged by surface waters (freshwater) to the Plymouth-Kingston-Duxbury Embayment System. The "Stream" data are from the MEP stream gauging effort. Watershed data are based upon the USGS/MEP watershed modeling effort (Section IV.1). Delineations were reviewed by MEP Technical Team Members and smoothed as described in Section III.

Stream Discharge Parameter	Jones River Discharge[a] Kingston Bay	Town Brook Discharge[a] Plymouth Harbor	Eel River Discharge[a] Plymouth Harbor	Data Source
Total Days of Record	365[b]	365[b]	365[b]	(1)
Flow Characteristics				
Stream Average Discharge (m3/day)	102,201	52,939	68,983	(1)
Contributing Area Average Discharge (m3/day)	101,216	47,136	69,036	(2)
Discharge Stream 2006-07 vs. Long-term Discharge	0.96%	10.96%	-0.08%	
Nitrogen Characteristics				
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.517	0.729	0.144	(1)
Stream Average Total N Concentration (mg N/L)	1.041	1.354	0.642	(1)
Nitrate + Nitrite as Percent of Total N (%)	50%	54%	22%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	106.40	71.69	44.29	(1)
TN Average Contributing UN-attenuated Load (kg/day)	135.58	111.25	110.35	(3)
Attenuation of Nitrogen in Pond/Stream (%)	22%	36%	60%	(4)

(a) Flow and N load to streams discharging to Plymouth Harbor and Kingston Bay include apportionments of Pond contributing areas as appropriate.
(b) Average September 1, 2003 to August 31, 2004.

(1) MEP gage site data
(2) Calculated from MEP watershed delineations to ponds upgradient of specific gages;
the fractional flow path from each sub-watershed which contribute to the flow in the streams to the Plymouth-Kingston-Duxbury system;
and the annual recharge rate.
(3) As in footnote (2), with the addition of pond and stream conservative attentuation rates.
(4) Calculated based upon the measured TN discharge from the rivers vs. the unattenuated watershed load.

Table IV-8. Summary of annual volumetric discharge and nitrogen load from the four major surface water discharges to the Plymouth-Kingston-Duxbury embayment system (based upon the data presented in Figures IV-8b,c ,9a,b,10a,b and Table IV-7.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	**TN**
Kingston Bay Jones River MEP measured	**September 1, 2003 to August 31, 2004**	**37,303,479**	**19,281**	**38,837**
Kingston Bay Jones River	Based on Watershed Area and Recharge	**36,943,840**	--	--
Plymouth Harbor Town Brook MEP measured	**September 1, 2003 to August 31, 2004**	**19,322,561**	**14,085**	**26,166**
Plymouth Harbor Town Brook	Based on Watershed Area and Recharge	**17,204,640**	--	--
Plymouth Harbor Eel River MEP measured	**September 1, 2003 to August 31, 2004**	**25,178,656**	**3,625**	**16,166**
Plymouth Harbor Eel River	Based on Watershed Area and Recharge	**25,198,140**	--	--

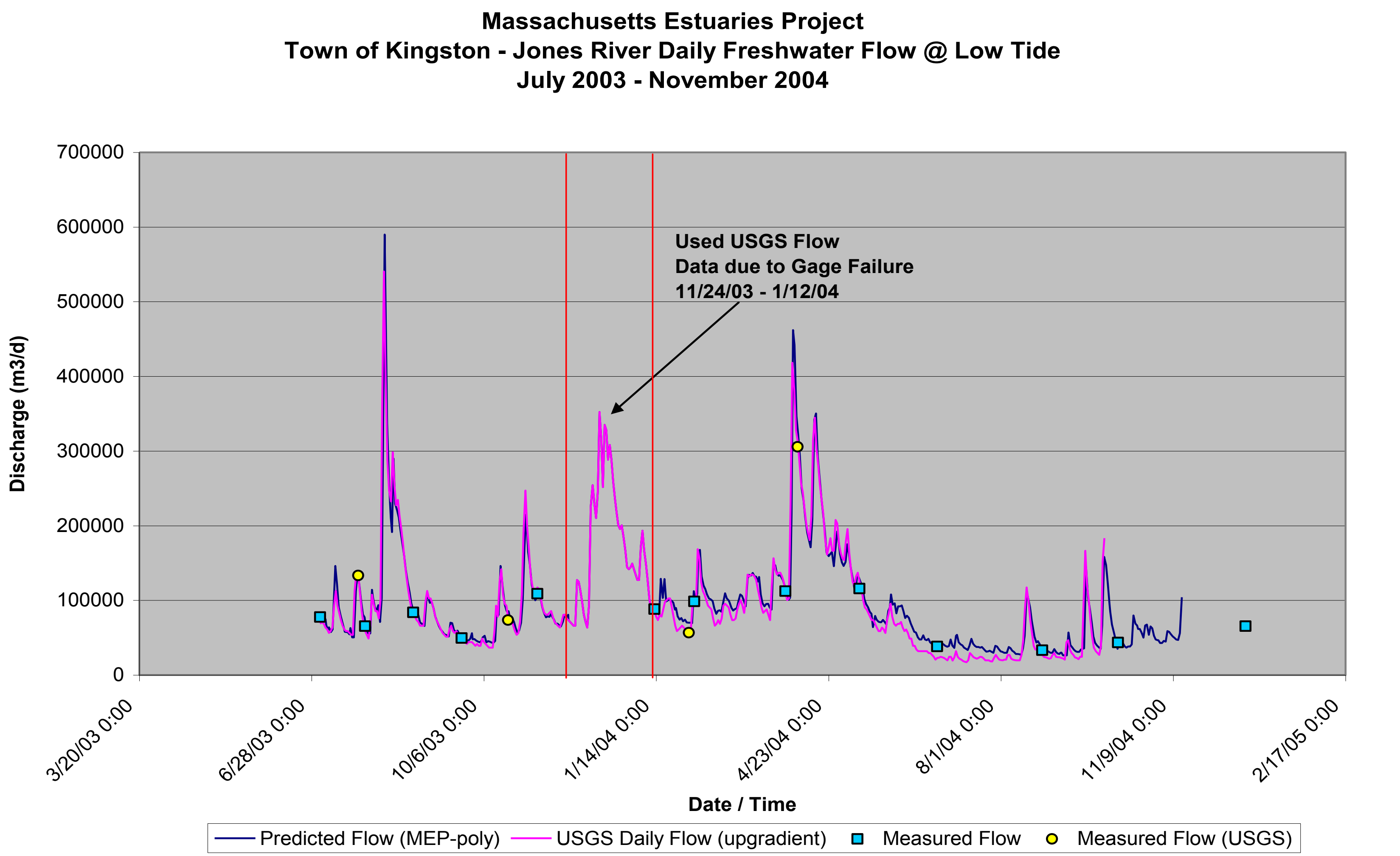


Figure IV-8a. Predicted daily discharge (USGS and MEP) for the Jones River discharging to Kingston Bay. Blue and yellow symbols are measured flows.

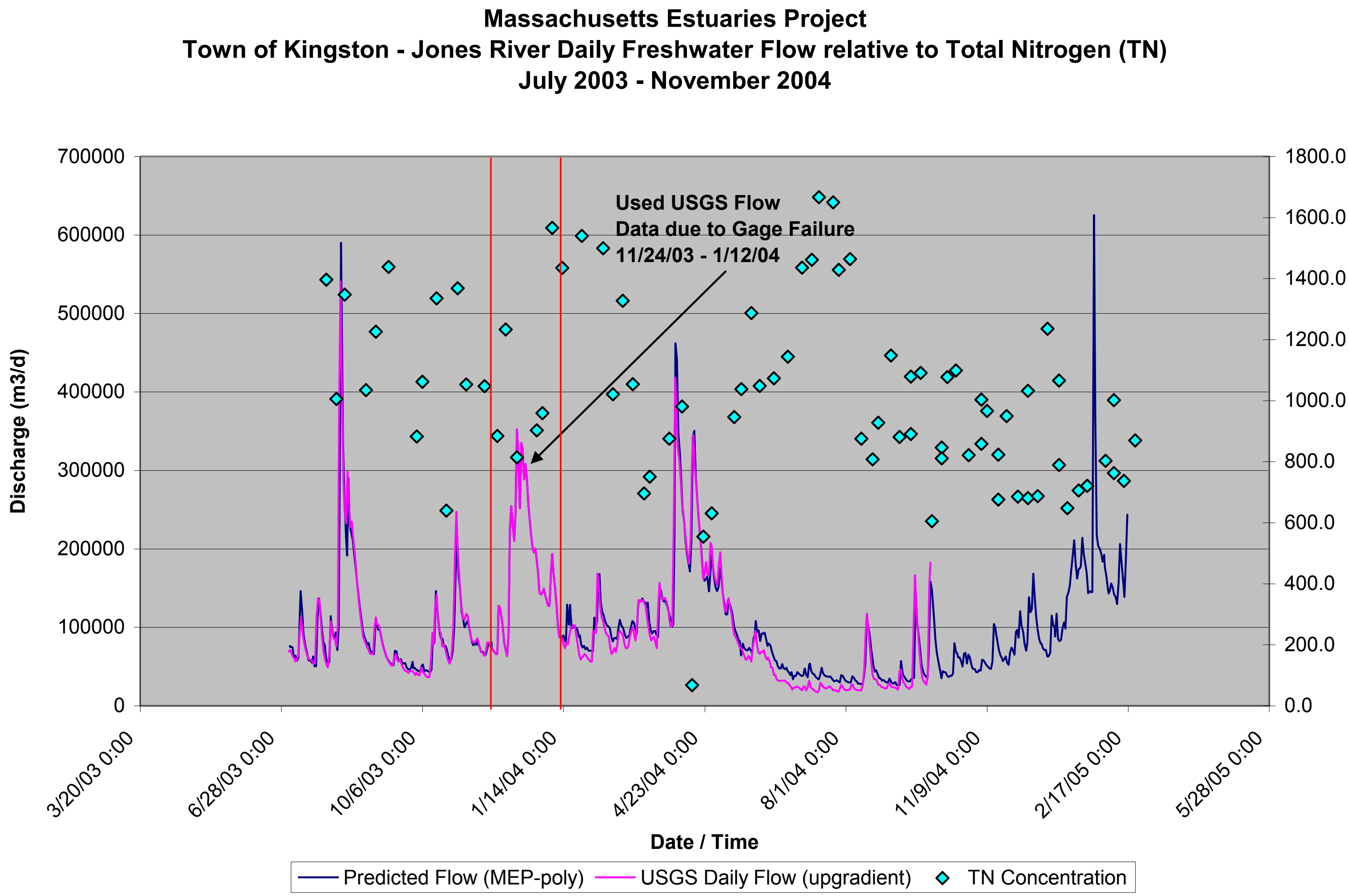


Figure IV-8b. Discharge from Jones River to Kingston Bay (solid blue line) compared to USGS determined flow (pink line). Total nitrogen (blue symbols) concentration (mg/m3) are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-6).

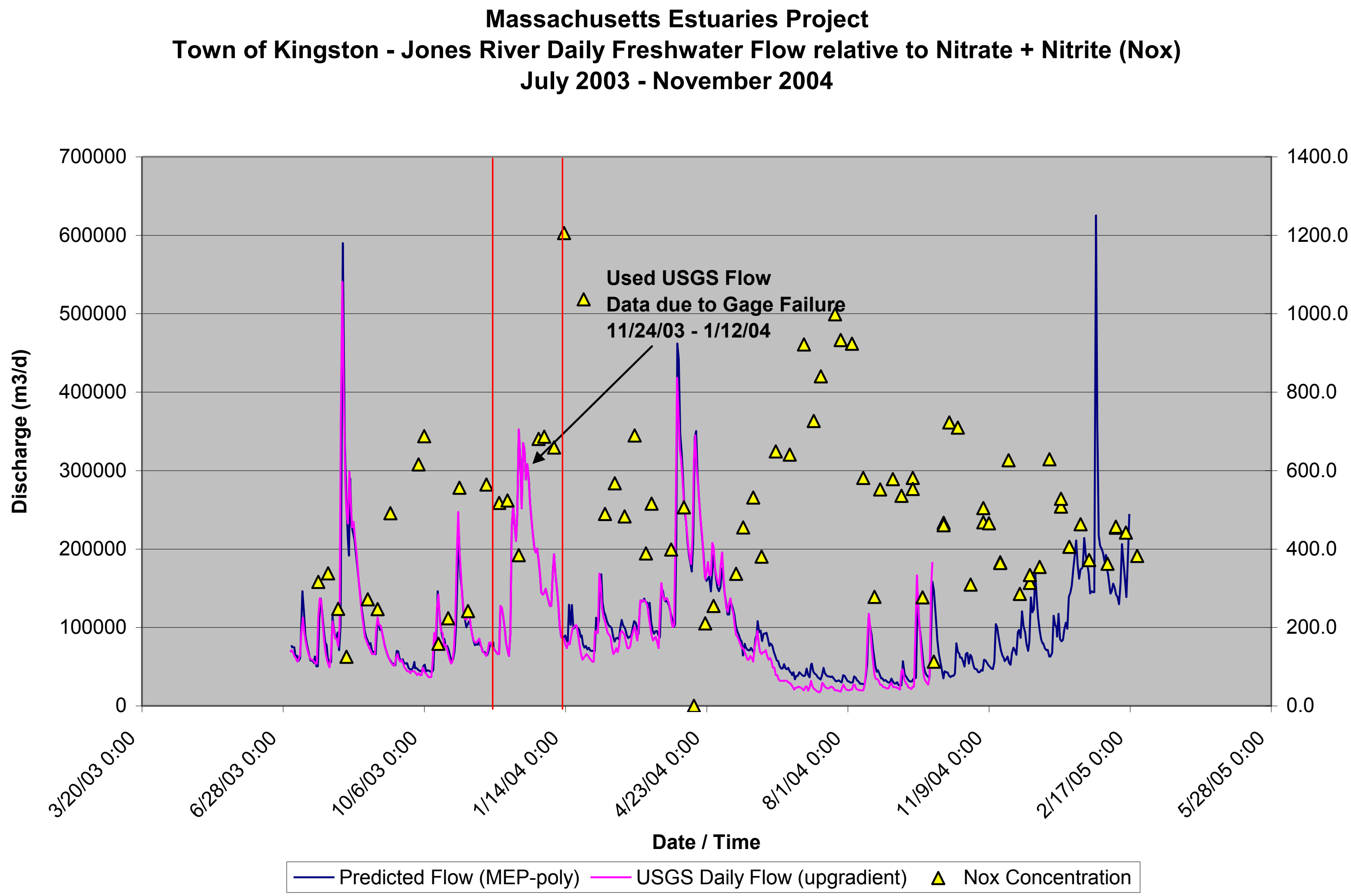


Figure IV-8c. Discharge from Jones River to Kingston Bay (solid blue line) compared to USGS determined flow (pink line). Nitrate + Nitrite (NOx) (yellow symbols) concentrations (mg/m3) are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-6).

Given the large nitrogen load being transported by the Jones River to the estuary and the dominance of nitrate, the opportunity may exist to enhance nitrogen attenuation by freshwater systems. High nitrate concentrations can support denitrification if freshwater systems (ponds, wetlands) of proper structure and sediment organic matter content can be enhanced or constructed to intercept flow and nitrate load. The MEP Technical Team suggests that this be examined by the Towns as they undertake watershed nitrogen management planning.

IV.2.3 Surface water Discharge and Attenuation of Watershed Nitrogen: Eel River Discharge to Plymouth Harbor

Similar to other surface water features in the MEP study region that typically emanate from a specific pond or wetland, the Eel River, which discharges into the Plymouth Harbor portion of the overall embayment system, does have a network of up-gradient bog/wetland and pond areas that contribute surfacewater to this significant river of southeastern Massachusetts. Based on numerous previous studies completed by the MEP on other systems in southeastern Massachusetts, the outflow from the bog/wetlands/ponds and the wooded areas up-gradient of the Eel River gauge very likely contribute to the attenuation of nitrogen and also provides for a direct measurement of the nitrogen attenuation. The combined rate of nitrogen attenuation by the biological processes that occur in the various surface water features was determined by comparing the present predicted (calculated from land use analysis) nitrogen loading to the sub-watershed region contributing to the bog/wetlands and wooded areas above the gauge site and the measured annual Eel River discharge of nitrogen to the southern-most end of Plymouth Harbor, Figures IV-6, IV-9a.

At the Eel River gauge site (immediately down gradient of the Route 3A bridge crossing), a continuously recording vented calibrated water level gauge was installed to yield the level of water in the channel that carries the flows and associated nitrogen load to the estuarine system. As the lower reach of Eel River is tidally influenced down gradient of Route 3A, the stage record from the gauge was checked to make sure there was no tidal influence in the record at low tide. To confirm that freshwater was being measured at low tide, the stage record was analyzed for any semi-diurnal variations indicative of tidal influence and salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average salinity of the water samples taken from Eel River near the Route 3A bridge at low tide was determined to be 0.1 ppt. Therefore, the gauge location was deemed acceptable for making freshwater flow measurements at low tide. Calibration of the gauge was checked monthly. The gauge on Eel River was installed on July 23, 2003 and was set to operate continuously for a complete hydrologic year (low flow to low flow, ~12 months). Stage data collection continued until February 16, 2005 for a total deployment of 19 months.

Surface freshwater flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for Eel River at the gauge site located immediately down-gradient of the Route 3A bridge based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the estuary and is reflective of the biological processes occurring in the stream channel and the network of bogs/wetlands/ponds and wooded areas contributing to nitrogen attenuation (Figure IV-9a,b,c and Table IV-7 and IV-8). In addition, a water balance was constructed based upon the U.S. Geological Survey/MEP defined watershed delineations to determine long-term average freshwater discharge expected at each gauge site based on area and average recharge.

Figure IV-9a. Location of MEP stream gauges (yellow symbol) for measuring flow and nitrogen loads in Eel River. Eel River receives surfacewater from a network of up-gradient bog/wetland/pond features. Historic USGS gauging location (station id. 01105876, 1969-1971) for comparative flow measurements was located ~100 meters up-gradient of the MEP gauge.

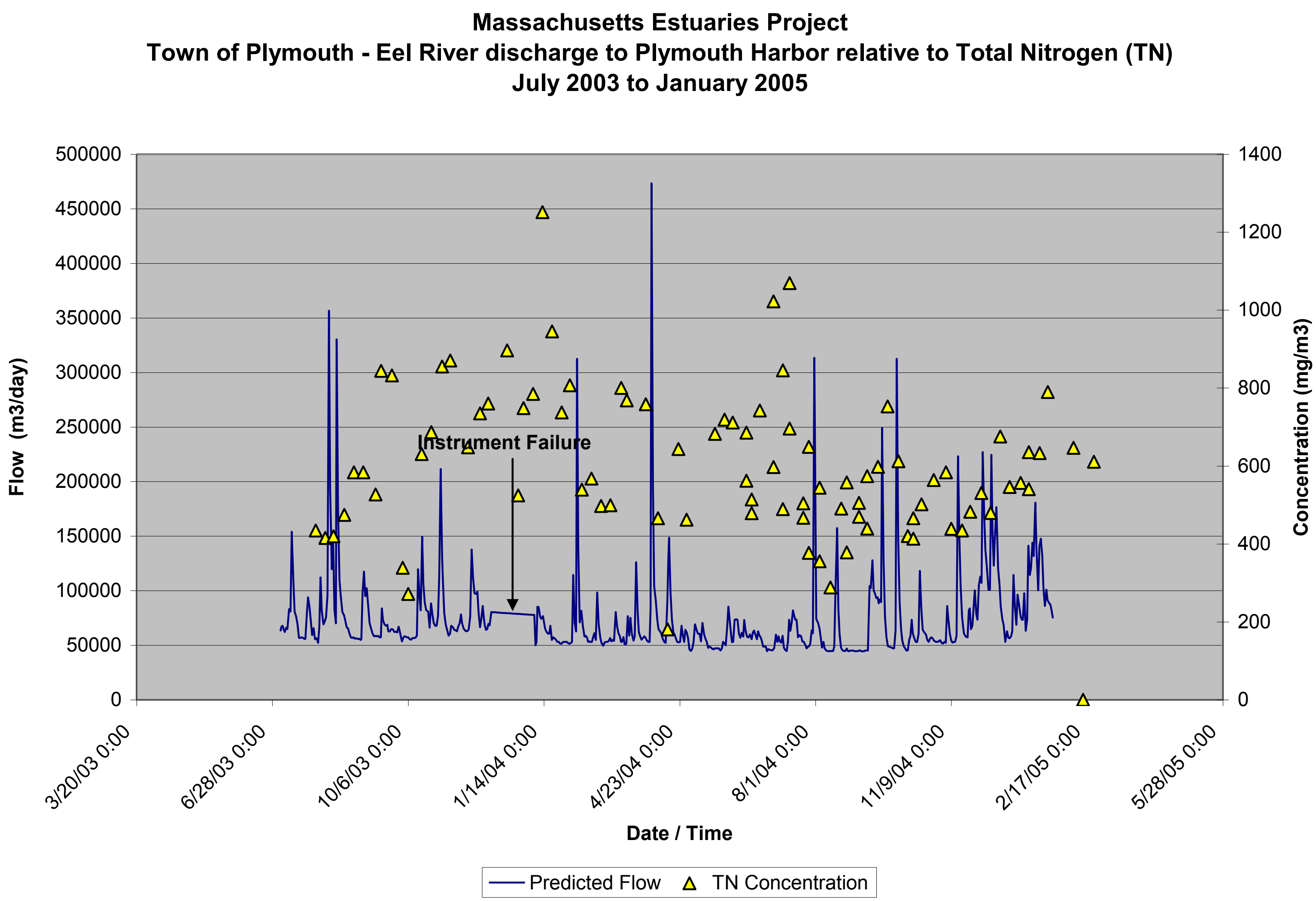


Figure IV-9b. Discharge from Jones River to Kingston Bay (solid blue line). Total nitrogen (yellow symbols) concentration (mg/m3) are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-6).

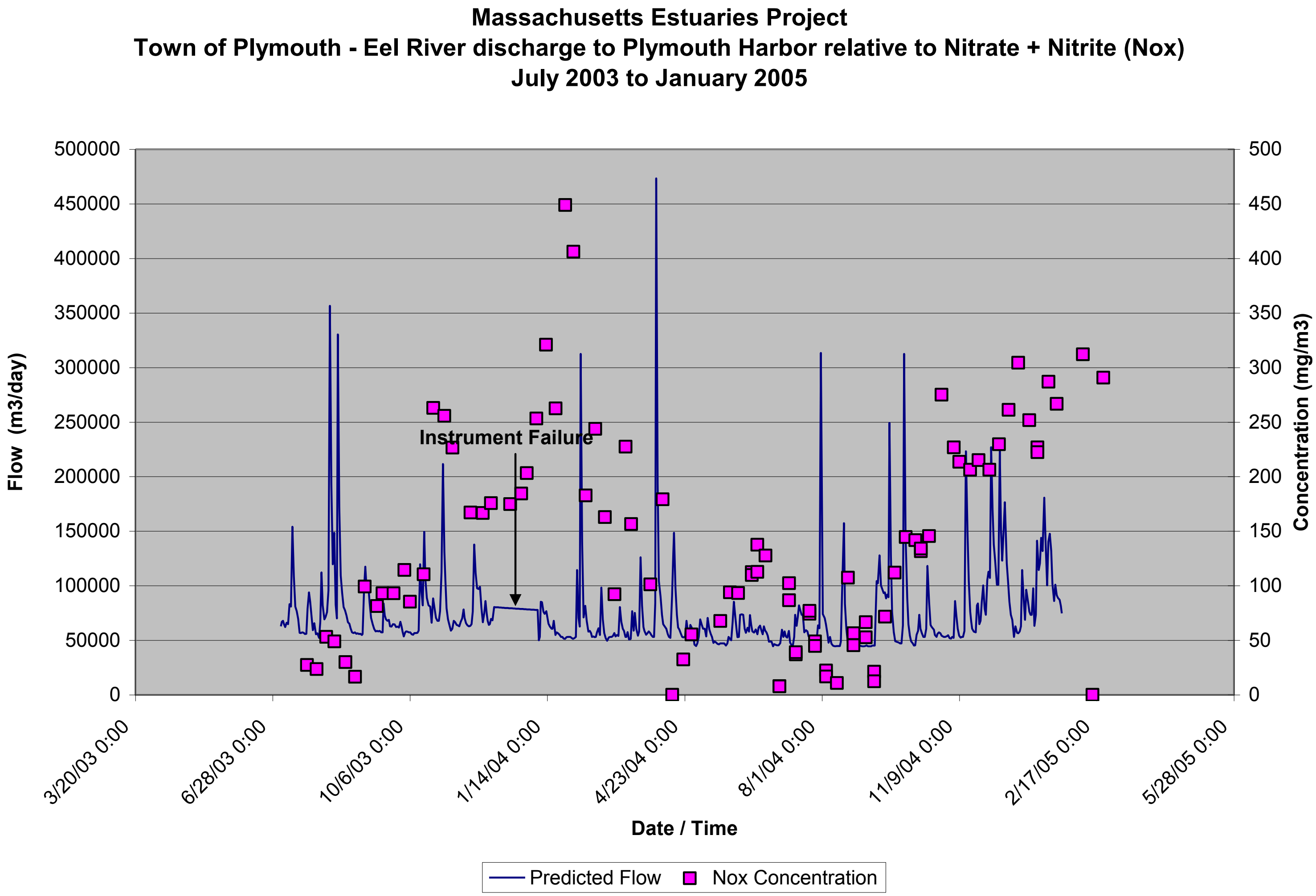


Figure IV-9c. Discharge from Eel River (solid blue line). Nitrate + Nitrite (NOx, pink symbols) concentrations are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-6).

The annual freshwater flow record for Eel River as measured by the MEP was compared to the long-term average flows determined by the USGS/MEP modeling effort (Table III-1). The measured freshwater discharge from Eel River at the Route 3A gauge location was ~1% below the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2003 and ending in August 2004 (low flow to low flow) was 68,983 m^3/day compared to the long term average flows determined by the MEP watershed modeling effort (69,036 m^3/day). It should be noted that daily flows calculated using the rating curve developed by the MEP were confirmed relative to a historical record (1969-1971) of daily flows developed by the US Geological Survey at a USGS maintained gauging station (station id. 01105876) approximately 100m up-gradient of the MEP gauge deployed on the Eel River. The USGS determined average daily flow for the period 1969-1971 was 68,504 m^3/day (28 cfs). A second independent confirmation of the MEP determined flow in the Eel River was available by comparing the MEP flow to Eel River average daily flow determined by a study of the Eel River completed by Camp, Dresser & McKee, Inc. In that study, the average daily flow in the Eel River for the period 1998-2000 was 65,127 m^3/day (26.62 cfs). The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in the Eel River discharging from the sub-watershed indicates that the river is capturing the up-gradient recharge (and loads) accurately. The independent measure of the Eel River flow by the USGS and CDM also serve to confirmation the MEP flows are accurate and reasonable for calculating nutrient loads to the estuary.

Total nitrogen concentrations within the Eel River outflow were moderate, on average 0.642 mg N L^{-1}, whereas average Nitrate + Nitrite (NOx) concentration was 0.144 mg N L^{-1} (22 % of the Total Nitrogen pool). Additionally, particulate organic nitrogen (PON) with an average concentration of 0.120 mg N L^{-1} represented 20% of the total nitrogen pool. In the Eel River, dissolved organic nitrogen (DON) with an average concentration of 0.310 mg N L^{-1} was the predominant form of nitrogen (51% of the Total Nitrogen pool), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was significantly taken up by plants within the pond or stream ecosystems prior to discharging to the Plymouth Harbor system. Figures IV-9b,c) depict the daily freshwater flow in the Eel River relative to the concentrations of Nitrate + Nitrite (NOx) and Total Nitrogen (TN) as determined from the weekly water quality sampling at the MEP gauge.

From the measured nitrogen load discharged by the Eel River to the Plymouth Harbor portion of the overall system and the nitrogen load determined from the watershed based land use analysis, it appears that there is moderate-high nitrogen attenuation of upper watershed derived nitrogen during transport to the Eel River and the down gradient estuary. Based upon the lower total nitrogen load (16,166 kg yr^{-1}) discharged from the Eel River at Route 3A compared to that added by the various land-uses to the associated watershed (40,278 kg yr^{-1}), the integrated attenuation in passage through the stream and up-gradient freshwater ponds and wetlands prior to discharge to the estuary is 60% (i.e. 60% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams evaluated under the MEP is expected given the nature of the up-gradient ponds/wetlands/bogs capable of attenuating nitrogen. The directly measured nitrogen load from the Eel River was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

IV.2.4 Surface water Discharge and Attenuation of Watershed Nitrogen: Town Brook Discharge to Plymouth Harbor

Similar to other surface water features in the MEP study region that typically emanate from a specific pond or wetland, Town Brook, which discharges into the Plymouth Harbor portion of the overall embayment system, does have a network of up-gradient bog/wetland and ponds (most significantly, Billington Sea) areas that contribute surfacewater to this significant river of southeastern Massachusetts. Based on numerous previous studies completed by the MEP on other systems in southeastern Massachusetts, the outflow from the bog/wetlands/ponds and the wooded areas up-gradient of the Town Brook gauge very likely contribute to the attenuation of nitrogen and also provides for a direct measurement of the nitrogen attenuation. The combined rate of nitrogen attenuation by the biological processes that occur in the various surface water features was determined by comparing the present predicted (calculated from land use analysis) nitrogen loading to the sub-watershed region contributing to the bog/wetlands and wooded areas above the gauge site and the measured annual Town Brook discharge of nitrogen to the middle portion of Plymouth Harbor, Figures IV-6,IV-10a.

At the Town Brook gauge site (between Main Street and Sandwich Street bridge crossings, downtown Plymouth), a continuously recording vented calibrated water level gauge was installed to yield the level of water in the channel that carries the flows and associated nitrogen load to the estuarine system. As the lower reach of Town Brook is tidally influenced down gradient of Route 3A, the stage record from the gauge was checked to make sure there was no tidal influence in the record at low tide. To confirm that freshwater was being measured at low tide, the stage record was analyzed for any semi-diurnal variations indicative of tidal influence and salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average salinity of the water samples taken from Town Brook near the Main Street bridge at low tide was determined to be 0.1 ppt. Therefore, the gauge location was deemed acceptable for making freshwater flow measurements at low tide. Calibration of the gauge was checked monthly. The gauge on Town Brook was installed on July 23, 2003 and was set to operate continuously for a complete hydrologic year (low flow to low flow, ~12 months). Stage data collection continued until February 22, 2005 for a total deployment of 19 months.

Surface freshwater flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for Town Brook at the gauge site located immediately down-gradient of the Sandwich Street bridge based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the estuary and is reflective of the biological processes occurring in the stream channel and the network of bogs/wetlands/ponds and wooded areas contributing to nitrogen attenuation (Figure IV-10a,b,c and Table IV-7 and IV-8). In addition, a water balance was constructed based upon the U.S. Geological Survey/MEP defined watershed delineations to determine long-term average freshwater discharge expected at each gauge site based on area and average recharge.

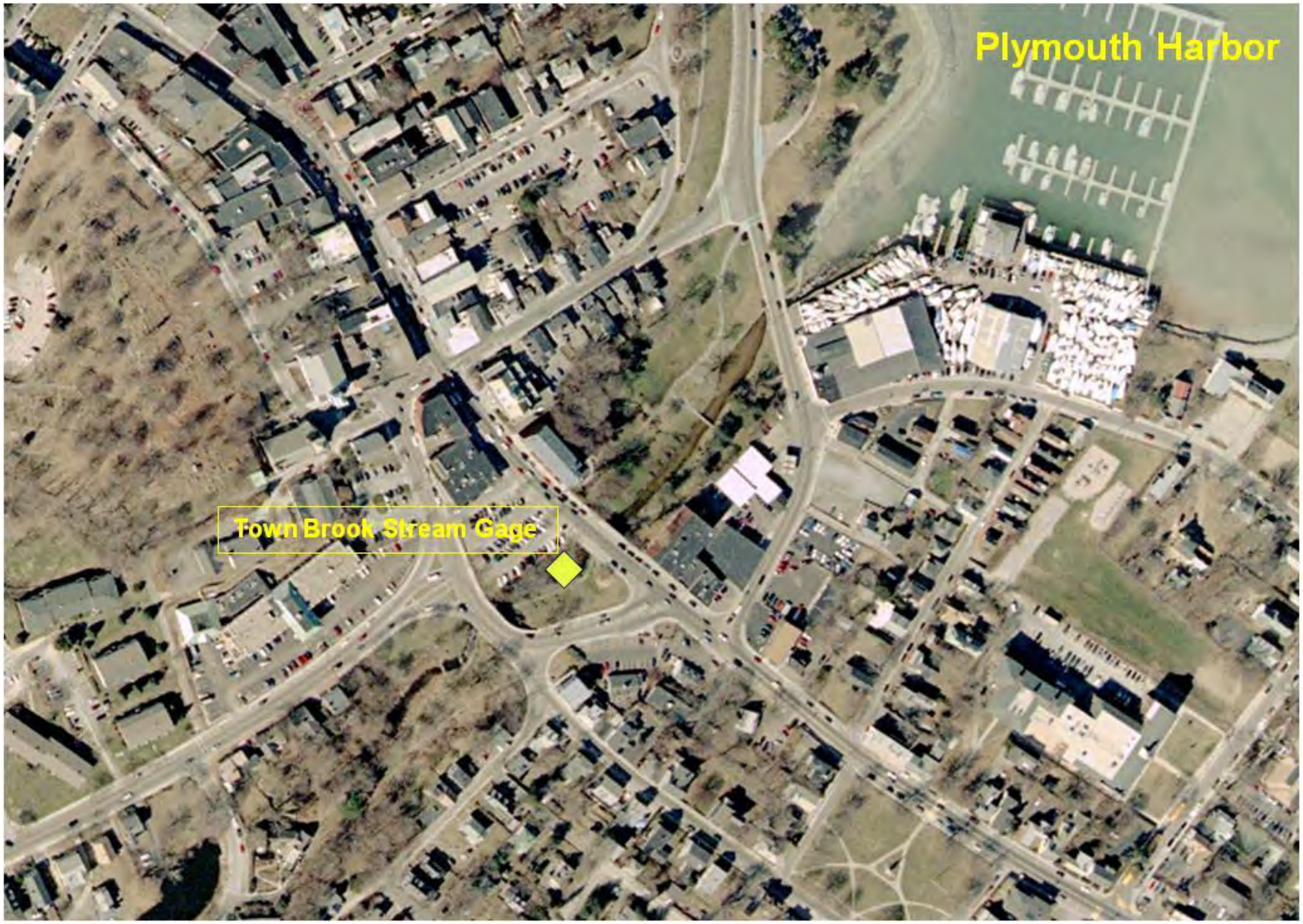


Figure IV-10a. Location of MEP stream gauge (yellow symbol) for measuring flow and nitrogen load in Town Brook. Town Brook receives both groundwater as well as surfacewater flow from Billington Sea (a large up-gradient freshwater pond) and discharges to the down gradient estuarine receiving waters of Plymouth Harbor.

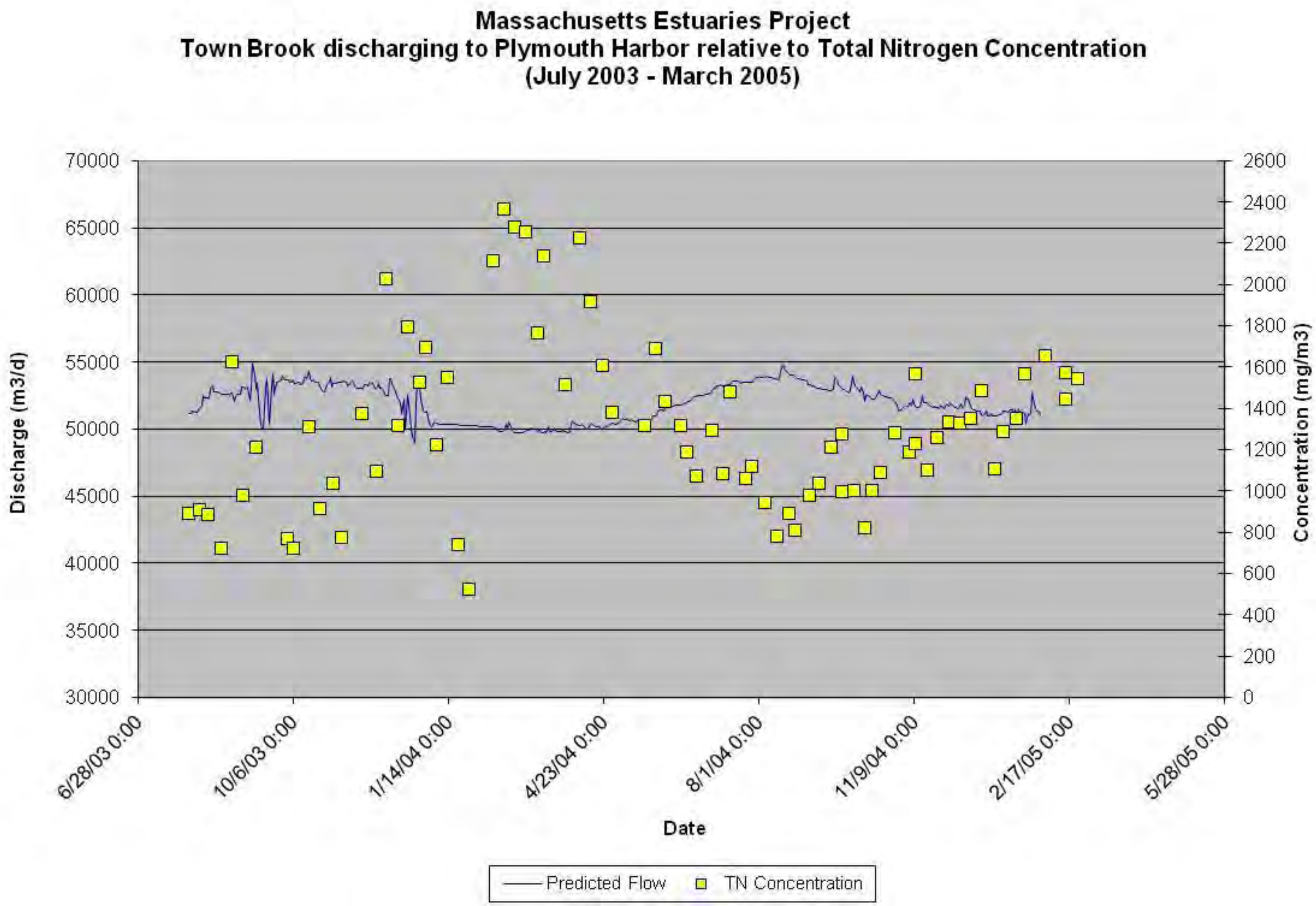


Figure IV-10b. Discharge from Bridge Creek (solid blue line). Total nitrogen (yellow symbols) and Nitrate + Nitrite (NOx, blue symbols) concentrations are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-6).

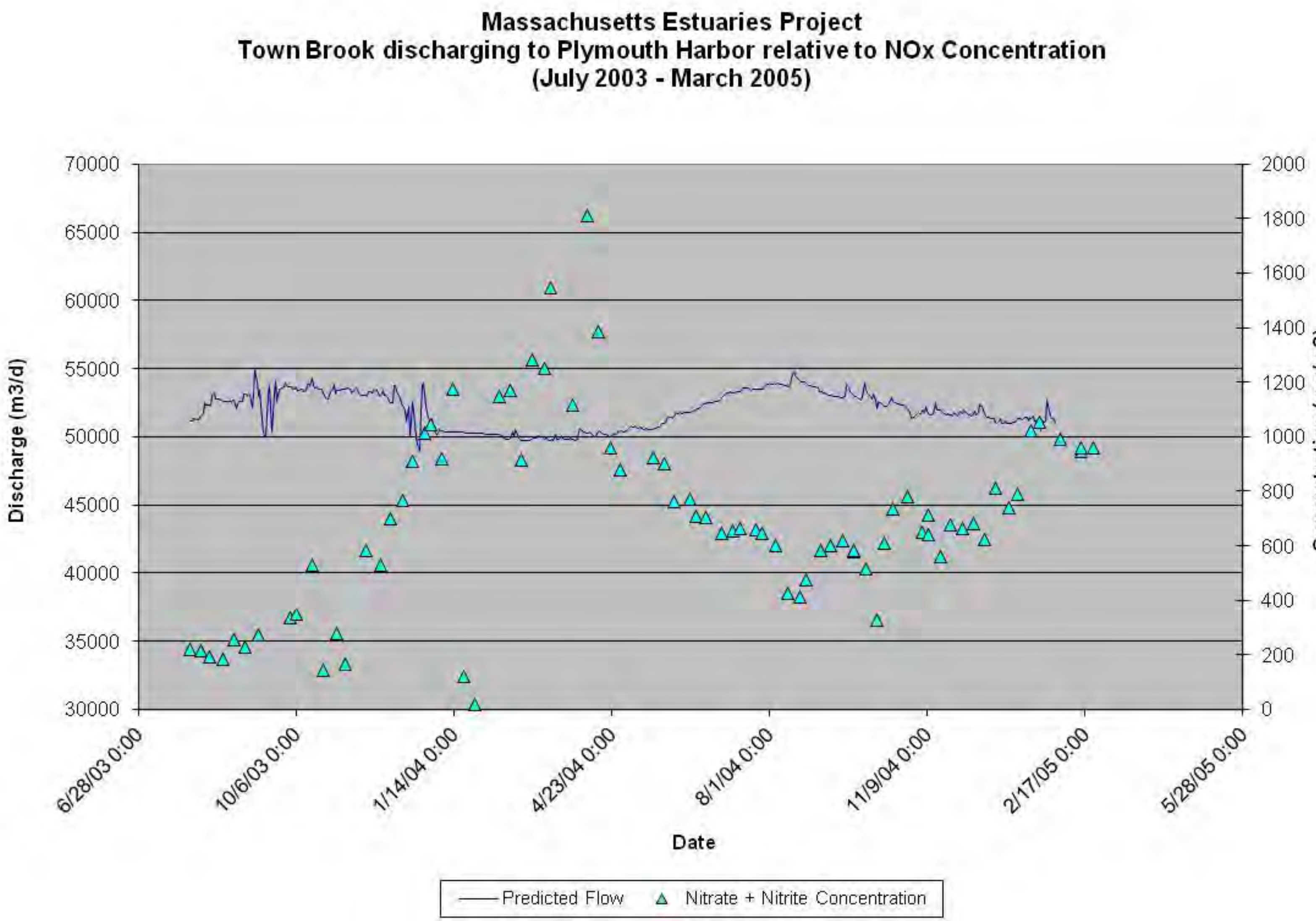


Figure IV-10c. Discharge from Town Brook (solid blue line). Nitrate + Nitrite (NOx, blue symbols) concentrations are used for determination of attenuated nitrogen load from the sub-watershed (Table IV-6).

The annual freshwater flow record for Town Brook as measured by the MEP was compared to the long-term average flows determined by the USGS/MEP modeling effort (Table III-1). The measured freshwater discharge from Town Brook at the Main/Sandwich Street gauge location was ~10% above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2003 and ending in August 2004 (low flow to low flow) was 52,939 m^3/day compared to the long term average flows determined by the MEP watershed modeling effort (47,136 m^3/day). The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Town Brook discharging from the sub-watershed indicates that the river is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within the Town Brook outflow were high, on average 1.354 mg N L^{-1}, whereas average Nitrate + Nitrite (NOx) concentration was 0.729 mg N L^{-1} (54 % of the Total Nitrogen pool). Additionally, particulate organic nitrogen (PON) with an average concentration of 0.184 mg N L^{-1} represented 14% of the total nitrogen pool. In Town Brook, dissolved inorganic nitrogen (DIN) with an average concentration of 0.757 mg N L^{-1} was the predominant form of nitrogen (58% of the Total Nitrogen pool) while dissolved organic nitrogen (DON) with an average concentration of 0.372 mg N L^{-1} (28% of the Total Nitrogen pool), was less significant. This would indicate that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was less significantly taken up by plants within the pond or stream ecosystems prior to discharging to the Plymouth Harbor system. Figures IV-10b,c depict the daily freshwater flow in Town Brook relative to the concentrations of Nitrate + Nitrite (NOx) and Total Nitrogen (TN) as determined from the weekly water quality sampling at the MEP gauge.

From the measured nitrogen load discharged by Town Brook to the Plymouth Harbor portion of the overall system and the nitrogen load determined from the watershed based land use analysis, it appears that there is low to moderate nitrogen attenuation of upper watershed derived nitrogen during transport to Town Brook and the down gradient estuary. Based upon the lower total nitrogen load (26,166 kg yr^{-1}) discharged from Town Brook at the gauge site compared to that added by the various land-uses to the associated watershed (40,606 kg yr^{-1}), the integrated attenuation in passage through the stream and up-gradient freshwater ponds and wetlands prior to discharge to the estuary is 36% (i.e. 36% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams evaluated under the MEP is expected given the nature of the up-gradient ponds/wetlands/bogs capable of attenuating nitrogen. Additionally, given the predominance of DIN in the stream flow, it may be possible to enhance natural attenuation through restoration of Billington Sea thereby reducing total nitrogen load to Plymouth Harbor. The directly measured nitrogen load from Town Brook was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

Similar to the Jones River nutrient speciation were high DIN levels were observed along with low natural attenuation, given the large nitrogen load being transported by Town Brook to the estuary and the dominance of dissolved inorganic nitrogen (DIN, predominantly nitrate), the opportunity may exist to enhance nitrogen attenuation by freshwater systems. High nitrate concentrations can support denitrification if freshwater systems (ponds, wetlands) of proper structure and sediment organic matter content can be enhanced or constructed to intercept flow and nitrate load. The MEP Technical Team suggests that this be examined by the Towns as they undertake watershed nitrogen management planning

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

The overall objective of the benthic nutrient flux survey of the Plymouth Harbor, Kingston Bay and Duxbury Bay embayment system was to quantify the summertime exchange of regenerated nitrogen, between the sediments and overlying waters throughout the overall embayment system. The mass exchange of nitrogen between water column and sediments is a fundamental factor in controlling nitrogen levels within coastal waters. These fluxes and their associated biogeochemical pools relate directly to carbon, nutrient and oxygen dynamics and the nutrient related ecological health of these shallow marine ecosystems. In addition, these data are required for the proper modeling of nitrogen in shallow aquatic systems, both fresh and salt water.

IV.3.1 Sediment-Water column Exchange of Nitrogen

As stated in the above section, 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 Plymouth-Kingston-Duxbury Embayment System predominantly in highly bio-available forms from the surrounding upland watersheds and more refractory forms in the inflowing tidal waters. If all of the nitrogen remained within the water column (once it entered) then predicting water column nitrogen levels would be simply a matter of determining the watershed loads, dispersion, and hydrodynamic flushing. However, as nitrogen enters the embayment from the surrounding watersheds it is predominantly in the bio-available form nitrate. This nitrate and other bio-available forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton “particles”. Most of these “particles” remain in the water column for sufficient time to be flushed out to a down gradient larger water body (like Cape Cod Bay). However, some of these phytoplankton particles are grazed by zooplankton or filtered from the water by shellfish and other benthic animals and subsequently deposited on the bottom. Also, in longer residence time systems (greater than 8 days) these nitrogen rich particles may die and settle to the bottom. In both cases (grazing or senescence), a fraction of the phytoplankton with their associated nitrogen “load” become incorporated into the surficial sediments of the embayment.

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

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bio-available nitrogen is returned to the embayment water column for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that have been investigated by SMAST and the MEP, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. In contrast in some systems, with deep depositional basins or salt marsh tidal creeks, the sediments can be a net sink for nitrogen even during summer (e.g. Namskaket Salt Marsh (lower reach), Mashapaquit Creek Salt Marsh, West Falmouth Harbor; Centerville River Salt Marsh or Sesachacha Pond on the Island of Nantucket). Embayment basins can also be net sinks for nitrogen to the extent that they support relatively oxidized surficial sediments, for example in the margins of the main basin to Lewis Bay (Town of Barnstable, Cape Cod). In contrast, most embayments show low rates of nitrogen release throughout much of a basins area and, in regions of high deposition the anoxic sediments show high release rates during summer months. The consequence of high deposition rates is that the basin sediments are unconsolidated, organic rich and sulfidic nature (MEP field observations).

Failure to account for the site-specific nitrogen balance of the sediments and its spatial variation from the tidal creeks and embayment basins will result in significant errors in determination of the threshold nitrogen loading to the Plymouth-Kingston-Duxbury Bay Embayment System. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

IV.3.2 Method for determining sediment-water column nitrogen exchange

For the Plymouth-Kingston-Duxbury Embayment System, in order to determine the contribution of sediment regeneration to water column nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected from a total of 23 sites (24 cores) spatially distributed across the overall system. All the sediment cores for this system were collected in July-August 2007. Measurements of total dissolved nitrogen, nitrate + nitrite (NOx), ammonium were made in time-series on each incubated core sample.

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

Duxbury Marsh & Bay Benthic Nutrient Regeneration Cores

- PKD-1 1 core (Marsh - above Powder Point Bridge)
- PKD-2 1 core (Open Water)
- PKD-3 1 core (Open Water)
- PKD-4/5 2 cores (Open Water)
- PKD-6 1 core (Open Water)
- PKD-7 1 core (Open Water)
- PKD-8 1 core (Open Water)

Kingston Bay Benthic Nutrient Regeneration Cores

- PKD-9 1 core (Open Water)
- PKD-10 1 core (Open Water)
- PKD-11 1 core (Open Water)
- PKD-12 1 core (Open Water)
- PKD-13 1 core (Open Water)
- PKD-14 1 core (Open Water)

Central Basin - Inlet Benthic Nutrient Regeneration Cores

- PKD-15 1 core (Proximal to inlet)
- PKD-16 1 core (Proximal to inlet)
- PKD-17 1 core (Open Water-mid)

• PKD-22	1 core	(Proximal to inlet)
• PKD-23	1 core	(Proximal to inlet)
• PKD-24	1 core	(Proximal to inlet)

Plymouth Harbor Benthic Nutrient Regeneration Cores

• PKD-18	1 core	(Open Water-southern end)
• PKD-19	1 core	(Open Water-southern end)
• PKD-20	1 core	(Open Water-southern end)
• PKD-21	1 core	(Open Water-mid)

Sediment-water column exchange follows the methods of Jorgensen (1977), Klump and Martens (1983), and Howes *et al.* (1998) for nutrients and metabolism. Upon return to the field laboratory at the Plymouth Boatyard (a private marine facility on the shore of Plymouth Harbor operated by Mr. Todd Jesse), the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and ortho-phosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining samples frozen (-20^{o}C) for assay of nitrate + nitrite (Cd reduction: Lachat Autoanalysis), and DON (D'Elia *et al.* 1977). Rates were determined from linear regression of analyte concentrations through time.

Chemical analyses were performed by the Coastal Systems Analytical Facility at the School for Marine Science and Technology (SMAST) at the University of Massachusetts in New Bedford, MA. (508-910-6325 or d1white@umassd.edu). The laboratory follows standard methods for saltwater analysis and sediment geochemistry.

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

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

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

Figure IV-11. Plymouth Harbor, Kingston Bay, Duxbury Bay Embayment System sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above.

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 levels of organic matter within the sediments, whether the sediments are oxic or 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 lowering nitrogen levels.

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

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 (blooms).

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

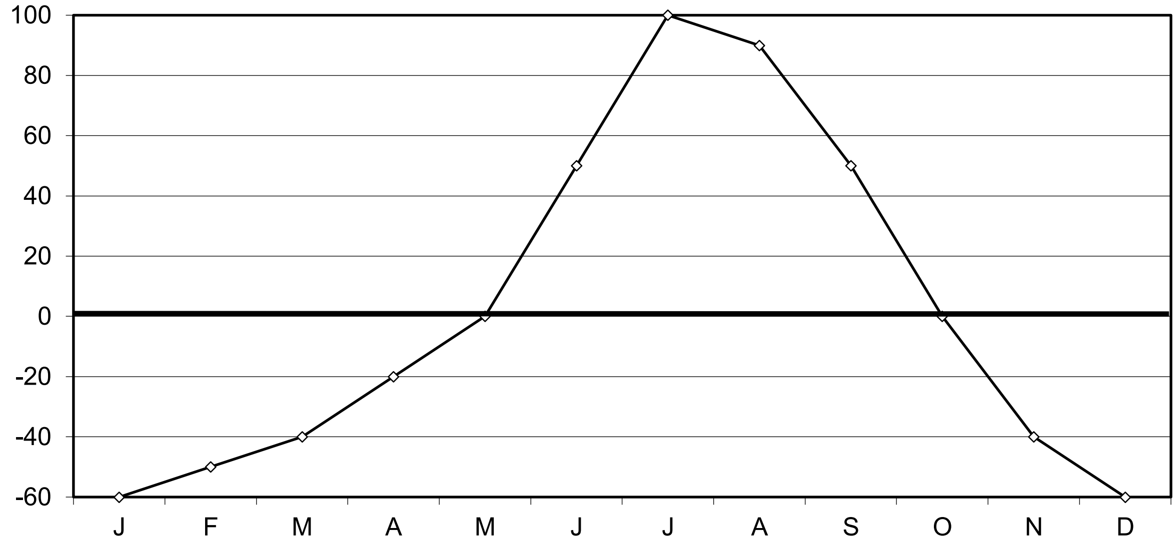


Figure IV-12. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

Sediment Nitrogen Release by Standard Core Approach: Sediment sampling was conducted throughout the main tidal channels of the seagrass/sand flats portions of the system as well as the deeper open water areas of the embayment close to the inlet and the salt marsh basin in Duxbury Bay. The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content, as well as sediment type and an analysis of each site's tidal flow velocities. As expected flow velocities are generally low in the uppermost reaches of the tidal creeks and high in the lower portions of the system situated closer to the tidal inlet. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within each sub-embayment.

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within the basin where regeneration was measured, the average summer particulate carbon and nitrogen concentration within the overlying water and the tidal velocities from the hydrodynamic model (Section V). Two levels of settling were used. If the sediments were organic rich and fine grained, and the hydrodynamic data showed low tidal velocities, then a water column particle residence time of 8 days was used (based upon phytoplankton and particulate carbon studies of poorly flushed basins). If the sediments indicated coarse-grained sediments and low organic content and high velocities, then half this settling rate was used. Adjusting the measured sediment releases was essential in order not to over-estimate the sediment nitrogen source and to account for those sediment areas which are net nitrogen sinks for the aquatic system. This approach has been previously validated in outer Cape Cod embayments (Town of Chatham embayments) by examining the relative fraction of the sediment carbon turnover (total sediment metabolism), which would be accounted for by daily particulate carbon settling. This analysis indicated that sediment metabolism in the highly organic rich sediments of the wetlands and depositional basins is driven primarily by stored organic matter (ca. 90%). Also, in the more open lower portions of larger embayments, storage appears to be low and a large proportion of the daily carbon requirement in summer is met by particle settling (approximately 33% to 67%). This range of values and their distribution is consistent with ecological theory and field data from shallow embayments. Additional, validation has been conducted on deep enclosed basins (with little freshwater inflow), where the fluxes can be determined by multiple methods. In this case the rate of sediment regeneration determined from incubations was comparable to that determined from whole system balance.

Net nitrogen release or uptake from the sediments within the Plymouth-Kingston-Duxbury Bay Embayment System were comparable to other large well flushed open water systems in southeastern Massachusetts The spatial distribution of nitrogen release/uptake by the sediments was slightly higher in the semi enclosed basins of the PDK Estuary: Plymouth Harbor (7.4 mg N m^{-2} d^{-1}) Kingston Harbor (7.8 mg N m^{-2} d^{-1}), than in the open basins: Duxbury Bay (1.0 mg N m^{-2} d^{-1}) and the central basin adjacent the tidal inlet (2.9 mg N m^{-2} d^{-1}). This slight gradient in benthic nitrogen release follows the pattern of particulate nitrogen in the water column of the system and therefore the pattern of particulate deposition. The overall rates (1.0 – 8.0 mg N m^{-2} d^{-1}) were comparable to other similar basins: e.g. the enclosed main basin of Wellfleet Harbor (2.2 – 10.2 mg N m^{-2} d^{-1}) , Phinneys Harbor (2.9 – 9.4 mg N m^{-2} d^{-1}), Lewis Bay main basin (6.9 mg N m^{-2} d^{-1}), Madaket Harbor main basin (6.0 mg N m^{-2} d^{-1}). All of these basins are well flushed with moderate to high tide ranges (6'-10') with low to moderate watershed nitrogen loading and generally sandy sediments in the outer reaches and consolidated oxidized muds in the inner basins.

Duxbury marsh basin (8.0 mg N m^{-2} d^{-1}) was also similar to other salt marsh dominated portions of the Back River (Bourne) and the Slocums and Little River Estuaries (Dartmouth) which support similarly small net release rates of 6.5 mg N m^{-2} d^{-1} and 4.6-9.0 mg N m^{-2} d^{-1}, respectively.

Net nitrogen release rates for use in the water quality modeling effort for the component sub-basins of the Plymouth Harbor, Kingston Bay and Duxbury Bay system (Section VI) are presented in Table IV-9. There was a clear spatial pattern of sediment nitrogen release, with slightly higher rates in the upper/more enclosed basins and lower rates in the more open basins of Duxbury Bay and the inlet basin. The sediments within the Plymouth-Kingston-Duxbury Bay Embayment System showed nitrogen fluxes typical of similarly structured systems with low to moderate watershed nitrogen loading and appear to be in balance with the overlying waters and the nitrogen flux rates consistent with the low nitrogen loading to this system and it relatively high flushing rate. Both the spatial pattern or release, the magnitude of the releases and the consistency of the values within the Plymouth-Kingston-Duxbury Bay Embayment System and the comparability to similar systems in southeastern Massachusetts supports the use of these release rates in the nitrogen modeling of this embayment system.

Table IV-9. Rates of net nitrogen return from sediments to the overlying waters of component basins comprising the Plymouth-Kingston-Duxbury Bay Estuarine System. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (see Section VI). Measurements represent July -August rates.

Location	Sediment Nitrogen Flux (mg N m^{-2} d^{-1})			Station I.D. *
	Mean	S.E.	# sites	PKD-#
Plymouth-Kingston-Duxbury Estuarine System				
Duxbury Marsh	8.0	0.7	6	1
Duxbury Bay	1.0	1.6	7	2-9
Kingston Bay	7.8	0.8	6	9-14
Outer Basin/Inlet	2.9	3.3	6	15-17, 22-24
Plymouth Harbor	7.4	8.5	4	18-21

* Station numbers refer to Figure IV-11.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This hydrodynamic study was performed for the Plymouth Harbor/Plymouth Bay/Kingston Bay/Duxbury Bay estuary system (i.e., the Plymouth Bay system), located on the southwestern shoreline of Cape Cod Bay, within the towns of Plymouth, Kingston and Duxbury, Massachusetts. The hydrodynamic analysis is used to characterize flow and circulation into and out of the system and serves as the basis for nutrient related water quality modeling discussed in Section VI. A topographic map detail in Figure V-1 shows the general study area. The system includes broad areas of sandy flats that are exposed at low tides, and areas of salt marsh, mostly at the northern extent of Duxbury Bay. The main basin of the system is bound to the east by Duxbury Beach and Plymouth Long Beach. The lowest elevations of the system exist in the main natural channel, off the northern tip of Long Beach, where maximum depths of approximately -75 feet NAVD occur. The surface coverage of the Plymouth Bay system including Plymouth Bay is more than 18,000 total acres. The area coverage of Plymouth Harbor, Kingston Bay and Duxbury Bay together is more than 12,000 acres, which includes about 1,300 acres of marsh plain in Duxbury Bay.

Tidal exchange with Cape Cod Bay dominates circulation in the Bay system. From measurements made in the course of this study, the average offshore tide range is 9 feet. Tidal flushing in the system is efficient, which is indicated by the lack of attenuation of the tide range for tide gauge stations located inside the system inlet.

The hydrodynamic study of the Plymouth Bay system proceeded as two main efforts that dealt with data collection and development of the hydrodynamic model. In the first portion of the study, bathymetry, tide data and current measurements were collected in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the study. The bathymetry survey of the Plymouth Bay system was performed to determine the variation of depths throughout the main sub-embayment areas of the system. Tides were recorded for 31 days at four stations positioned around the Harbor system. In addition to the tide records, an ACDP survey of tidal velocities was performed over the course of a tide cycle along two transects. These tide and velocity data were necessary to develop, calibrate and corroborate the hydrodynamic model of the system.

A numerical hydrodynamic model of Plymouth Bay and its attached sub-embayments was developed in the second portion of this study. Using the bathymetry survey data, a model grid mesh was generated for use with the RMA-2 hydrodynamic code. The tide data from the offshore area of Plymouth Bay were used to define the open boundary condition that drives the circulation of the model. Data measured within the system were used to calibrate and verify model performance to ensure that it accurately represents the dynamics of the real, physical system.

The calibrated hydrodynamic model of Plymouth Harbor is an integral piece of the water quality model developed in Chapter VI of this MEP nutrient threshold report. In addition to its use as the hydrodynamic basis for the TN and salinity models, the calibrated hydrodynamic model is a useful tool that can be used to investigate the tidal properties of the system.

Figure V-1. Topographic map detail of the Plymouth Harbor and Duxbury Bay system vicinity.

V.2 DATA COLLECTION AND ANALYSIS

The field data collection portion of this study was performed to characterize the physical properties of Plymouth Harbor. Bathymetry data were collected throughout the system so that it could be accurately represented as a computer hydrodynamic model and flushing rates could be determined for the system. In addition to the bathymetry, tide data were also collected throughout the Harbor in order to run the circulation model with real tides, and also to calibrate and verify model performance.

V.2.1 Bathymetry Data

Bathymetry data collection was conducted on five hydrographic surveys (November 12, 19 and 29; December 3 and 4, 2012). Due to the size and unique characteristics of the Plymouth Bay system, longitudinal survey transects were spaced at 400-meter intervals to provide high resolution coverage. Within Plymouth Harbor proper, survey line spacing was decreased to 150-meter intervals to increase spatial coverage. Particular attention was focused on the numerous channels (150-meter transect spacing interval) that intersect the system in order to capture the variability in the bottom bathymetry within these critical areas.

Survey transects were concentrated in the vicinity of the inlet to the system as well as throughout the channel network were the greatest variability in bottom bathymetry was expected. Survey transects were run along the channel perimeter, the channel midline, and the channel width. Survey transects along the channel width were spaced at periodic intervals throughout the channel area to capture the cross-sectional variation in the bottom bathymetry. Using a single beam precision fathometer (Odom Hydrotrac, 0.01-meter resolution) depth measurements were collected by shallow draft vessel. Global position data (Latitude, Longitude) was also collected using a differential GPS (Leica) with an accuracy 0.05 – 1.0 meters. All depth and position data were recorded into a laptop computer using hydrographic software (HYPACK) integrating the DGPS position and depth measurement into a single data set. Integrating the data in this manner enables the depth measurement to be linked with a precise known position.
The raw measured water depths were corrected for tidal stage (from tide gauges) to yield basin depths throughout the estuary to be referenced to the North American Vertical Datum of 1988 (NAVD88) . Reference to a single datum provides a correction of the bathymetric data for changing water levels due to the tide changes during the lengthy surveys. Once rectified, the finished, processed data were archived as ‘xyz’ files containing x-y horizontal position (in Massachusetts Mainland State Plane 1983 coordinates) and vertical elevation of the bottom (z).
Other sources of bathymetry data used in the development of the model grid are a December 2010 US Army Corps of Engineers survey of the Plymouth Harbor channel and boat basin, and 2013 USGS LiDAR for upland areas. The final processed bathymetric and topographic data are presented in Figure V-2.

Tide data records were collected concurrently at four gauging stations shown in Figure V-2, located offshore in Plymouth Bay (PLY1), in Plymouth Harbor (PLY2), at the inlet of Jones River in Kingston Bay (PLY3), and at the Plymouth Harbor Boat Basin (PLY4). The Temperature Depth Recorders (TDR) used to record the tide data were deployed simultaneously for a 31-day period between October 9 and November 9, 2012. The elevation of each gauge was surveyed relative to the NAVD vertical datum. The Plymouth Bay tide record was used as the open boundary condition of the hydrodynamic model. Data from inside the system were used to calibrate the model.

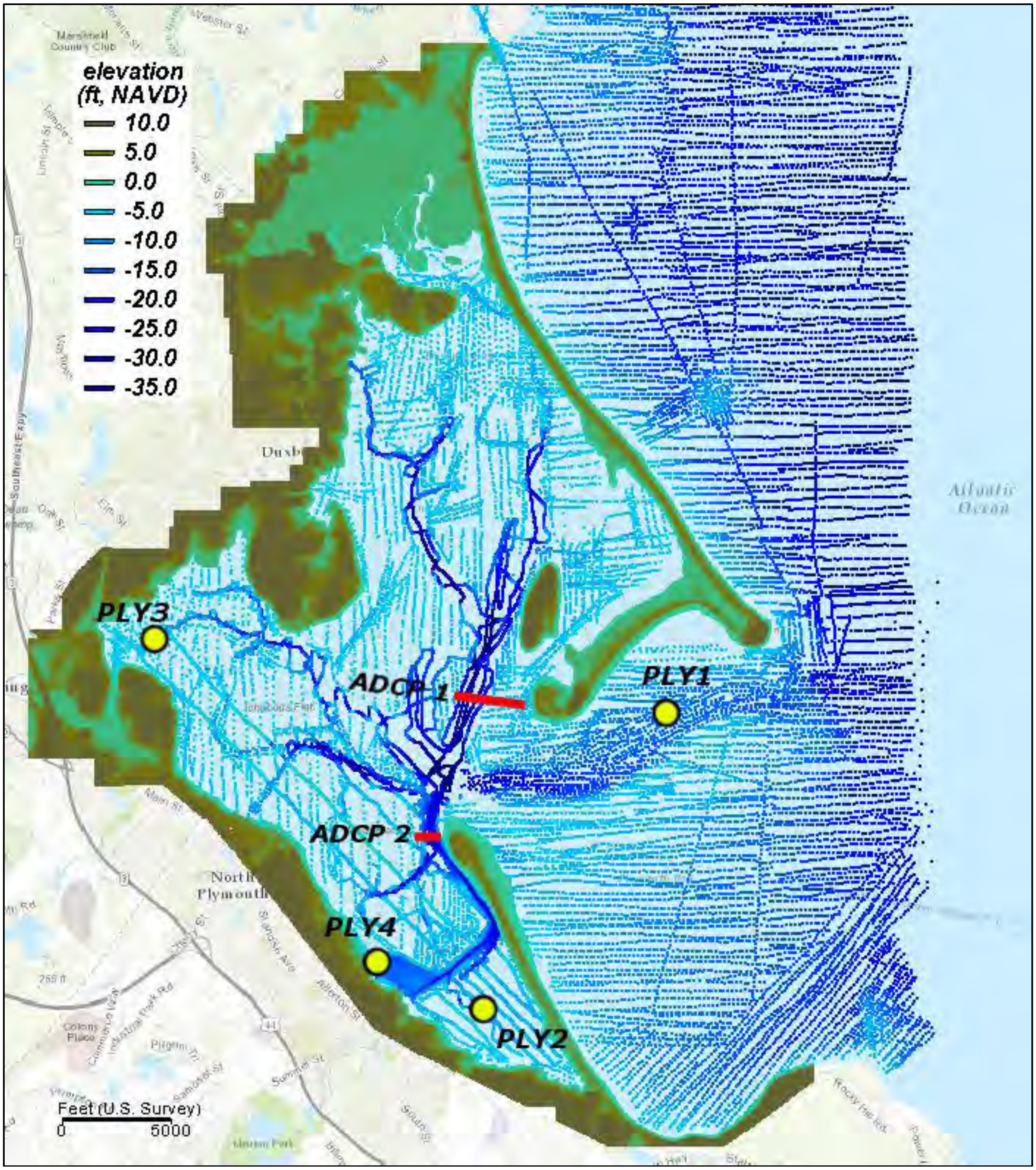


Figure V-2. Bathymetric and topography data used to develop the RMA-2 hydrodynamic model. Points are colored to represent the elevation relative to NAVD. The data sources used to develop the grid mesh are the 2012 bathymetry survey conducted by the MEP Technical Team, NOAA soundings and USGS 2013 LiDAR topography. Location of tide gauges and ADCP transects are also indicated.

V.2.2 Tide Data Collection and Analysis

Tide records longer than 29 days are necessary for a complete evaluation of tidal dynamics within the estuarine system. Although a one-month record likely does not include extreme high or low tides, it does provide an accurate basis for typical tidal conditions governed by both lunar and solar motion. For numerical modeling of hydrodynamics, the typical tide conditions associated with a one-month record are appropriate for driving tidal flows within the estuarine system.

Plots of the tide data from the five gauges are shown in Figure V-3 for the 31-day deployment period. The spring-to-neap variation in tide range is easily recognizable in these plots. The data record begins during a transitional period from neap to spring tides. The new moon occurs October 15, at the beginning of the first period of spring tides. After this, there is a period of neap tides around October 23, which occurs around the time of the first quarter moon of October 21. The minimum neap tide range in the offshore record is 5.7 feet (October 9), while the maximum spring tide rage (occurring about a week later) is 13.0 feet (October 16). A visual comparison between tide elevations offshore and at the different stations in the system (Figure V-4) shows that the tide amplitude varies little across the main embayments of the system.

V.2.2.1 Tide Datums

To better quantify the changes to the tide from the inlet to inside the system, the standard tide datums were computed for the 31-day period of concurrent data from the deployed TDRs. These datums are presented in Table V-1. 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 (MHH) and Mean Lower Low (MLL) 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. There is no evidence of tidal damping in the system, as differences in datum elevations across the system are within the typical range of measurement and survey error.

V.2.2.2 Tide Harmonic Analysis

A more thorough harmonic analysis of the tidal time series was also performed to produce tidal amplitude and phase of the major tidal constituents, and provide assessments of hydrodynamic 'efficiency' of the system in terms of tidal attenuation. This analysis also yielded an assessment of the relative influence of non-tidal, or residual, processes (such as wind forcing) on the hydrodynamic characteristics of each system.

A harmonic analysis was performed on the time series from each gauge location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The observed astronomical tide is the sum of several individual tidal constituents, with a particular amplitude and frequency (Figure V-5). For demonstration purposes, a graphical example of how these constituents computed for the Plymouth Harbor gauge data add together is shown in Figure V-6. The amplitudes and phase of 21 known tidal constituents result from this procedure. Table V-2 presents the amplitudes of eight tidal constituents computed for the Plymouth Harbor gauge records. The M_2, or the familiar twice-a-day lunar semi-diurnal tide, is the strongest contributor to the signal with an offshore amplitude of 4.4 feet. The total range of the M_2 tide is twice the amplitude, or 8.8 feet.

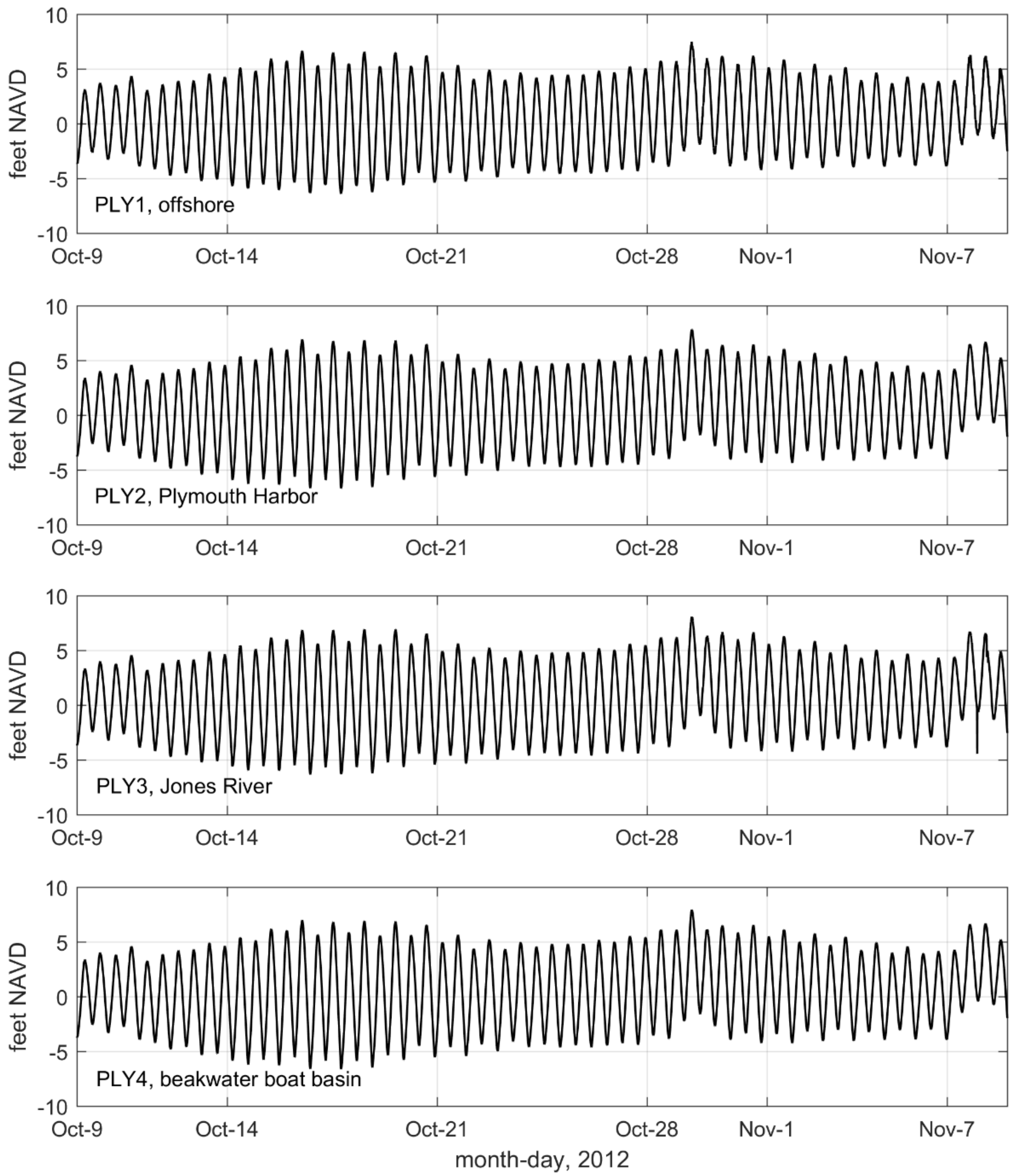


Figure V-3. Plots of observed tides for stations in Plymouth Harbor, for the 31-day period between October 9 and November 9, 2012. All water levels are referenced to the NAVD vertical datum.

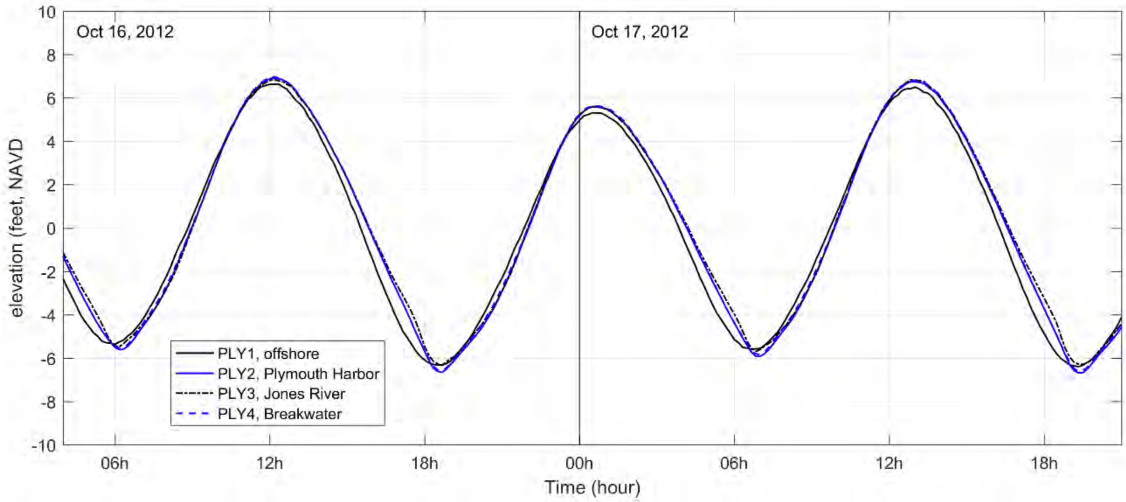


Figure V-4. Two-day tide plot showing tides measured in Plymouth Bay (PLY1) and at stations in the Plymouth Bay estuary system.

Table V-1. Tide datums computed from 31-day records collected offshore and in the Plymouth Bay estuary system in October and November 2012. Datum elevations are given relative to NAVD vertical datum.

Tide Datum	Plymouth Bay	Plymouth Harbor	Jones River	Breakwater
Maximum Tide	7.5	7.8	8.1	7.9
MHHW	5.3	5.6	5.7	5.6
MHW	5.0	5.2	5.3	5.3
MTL	0.4	0.5	0.5	0.5
MLW	-4.2	-4.3	-4.2	-4.2
MLLW	-4.5	-4.7	-4.6	-4.6
Minimum Tide	-6.4	-6.7	-6.3	-6.6
Mean Range	9.1	9.5	9.5	9.5

The diurnal constituents (once daily), K_1 and O_1, have amplitudes of approximately 0.4 feet and 0.3 respectively. Other semi-diurnal tides, the S_2 (12.00-hour period), N_2 (12.66-hour period) and L_2 (12.19-hour period) tides, also contribute to the total tide signal, with amplitudes of 0.7 feet, 1.0 feet and 0.2 feet, respectively.

The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 period for the M_6) and result from frictional attenuation of the M_2 tide in shallow water. The degree of energy transfer from the M_2 and its harmonics is small based on the comparison of amplitudes presented in Table V-2.

The phase change of the tide is also small across the system. Table V-3 shows the delay of the M_2 at different points in Plymouth Harbor relative to the timing of the M_2 constituent in Plymouth Bay, near the harbor entrance. The delay is of the same order of magnitude as the time step of the tide gauge records.

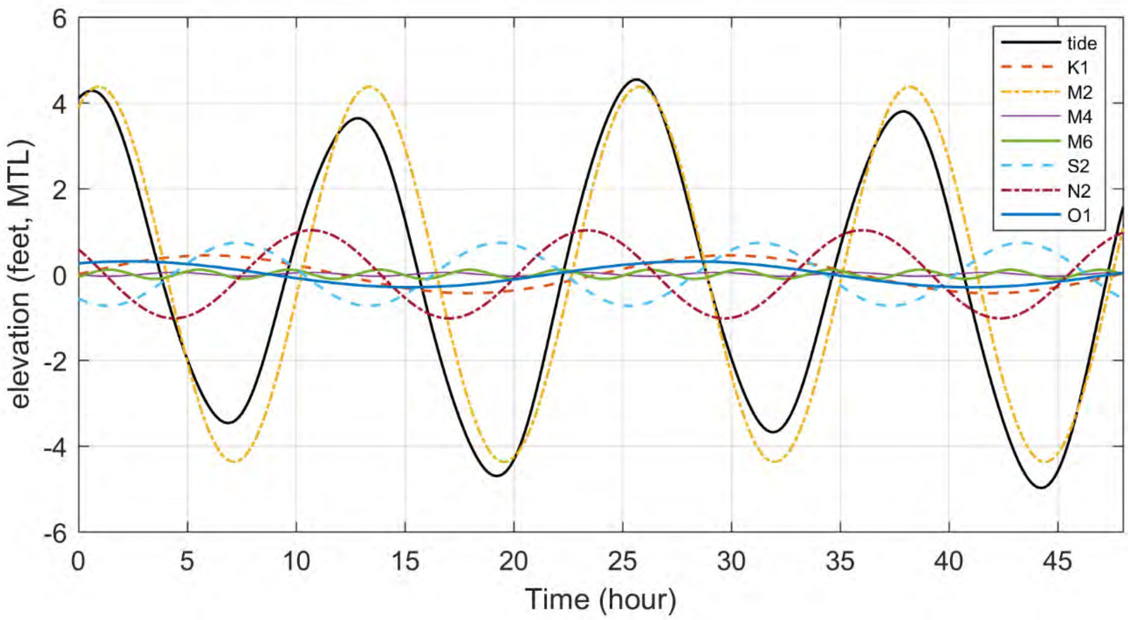


Figure V-5. Example of an observed astronomical tide (solid black lines) as the sum of its primary constituents, using constituents computed from the Plymouth Bay tide gauge record (PLY1) .

Table V-2. Tidal Constituents computed for tide stations in the Plymouth Bay estuary system and offshore in Plymouth Bay, October and November 2012.

	Amplitude (feet)							
Constituent	M_2	M_4	M_6	K_1	N_2	S_2	O_1	L_2
Period (hours)	12.42	6.21	4.14	23.93	12.66	12.00	25.82	12.19
Plymouth Bay	4.38	0.04	0.11	0.44	1.03	0.74	0.30	0.22
Plymouth Harbor	4.46	0.15	0.15	0.45	1.04	0.76	0.30	0.22
Jones River	4.41	0.18	0.14	0.45	1.00	0.74	0.29	0.24
Breakwater	4.44	0.13	0.16	0.45	1.03	0.75	0.30	0.22

In addition to the tidal analysis, the data were further evaluated to determine the importance of tidal versus non-tidal processes to changes in water surface elevation. These other processes include wind forcing (set-up or set-down) within the estuary, as well as sub-tidal oscillations of the sea surface. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large compared to tidal flow.

The results of an analysis to determine the energy distribution (or variance) of the measured water elevation records for the gauge records in Plymouth Harbor compared to the energy content of the astronomical tidal signal (re-created by summing the contributions from the 21 constituents determined by the harmonic analysis) is presented in Table V-4. Subtracting the tidal signal from the original elevation time series resulted in the non-tidal, or residual, portion of the water elevation changes. The energy of this non-tidal signal is compared to the tidal signal, and yields a quantitative measure of how important these non-tidal physical processes can be to hydrodynamic circulation within the estuary. Figure V-6 shows the comparison of the measured tide from Cape Cod Bay, with the computed astronomical tide resulting from the harmonic analysis, and the resulting non-tidal residual.

The analysis shows that tides are responsible for about 95% of the water level changes in Cape Cod Bay and all of Plymouth Harbor for the gauge deployment period. This indicates that the hydrodynamics of the system is influenced predominantly by astronomical tides. The non-tidal residual is typically 5% of the total variance of the observed water level changes. Two storm events captured in the tide records contribute to the residual variance. Hurricane Sandy caused a surge that quickly peaked at about 3.3 feet on October 29. About a week later, the November 2012 northeast storm caused a longer period of water levels, with a maximum surge of about 2.8 feet. Both events are clearly visible in the residual plot of Figure V-6.

Table V-3. M_2 tidal constituent phase delay (relative to the Cape Cod Bay station) for gauge locations in the Plymouth Great Marsh estuary system, determined from measured tide data.

Station	Delay (minutes)
Plymouth Harbor	11.0
Jones River	11.7
Breakwater	10.6

Table V-4. Percentages of Tidal versus Non-Tidal Energy for stations in the Plymouth Bay estuary system and Plymouth Bay, October and November 2012.

TDR Location	Total Variance (ft^2)	Tidal (%)	Non-tidal (%)
Plymouth Bay	10.2	95.5	4.5
Plymouth Harbor	10.6	94.8	5.2
Jones River	10.4	94.7	5.3
Breakwater	10.6	94.7	5.3

V.2.2.3 Tide Flood and Ebb Dominance

An investigation of the flood or ebb dominance of different areas in the Plymouth Bay estuary system was performed using the measured tide data. Estuaries and sub-embayments that are flood dominant are typically areas that collect sediment over time since they have maximum flood tide velocities that are greater than the maximum velocities that occur during the ebb portion of the tide. Salt marshes tend to be flood dominant, as this condition allows them to collect material that is required to maintain healthy marsh resources.

Flood or ebb dominance in channels of a tidal system can be determined by utilizing the results of the harmonic analysis of tidal elevations, or by performing a similar analysis on a time series of tidal currents. A discussion of the method of relative phase determination is presented in Friedrichs and Aubrey (1988). For this method, the same M_2 and M_4 tidal constituents presented in Table V-2 were used as the basis of this analysis.

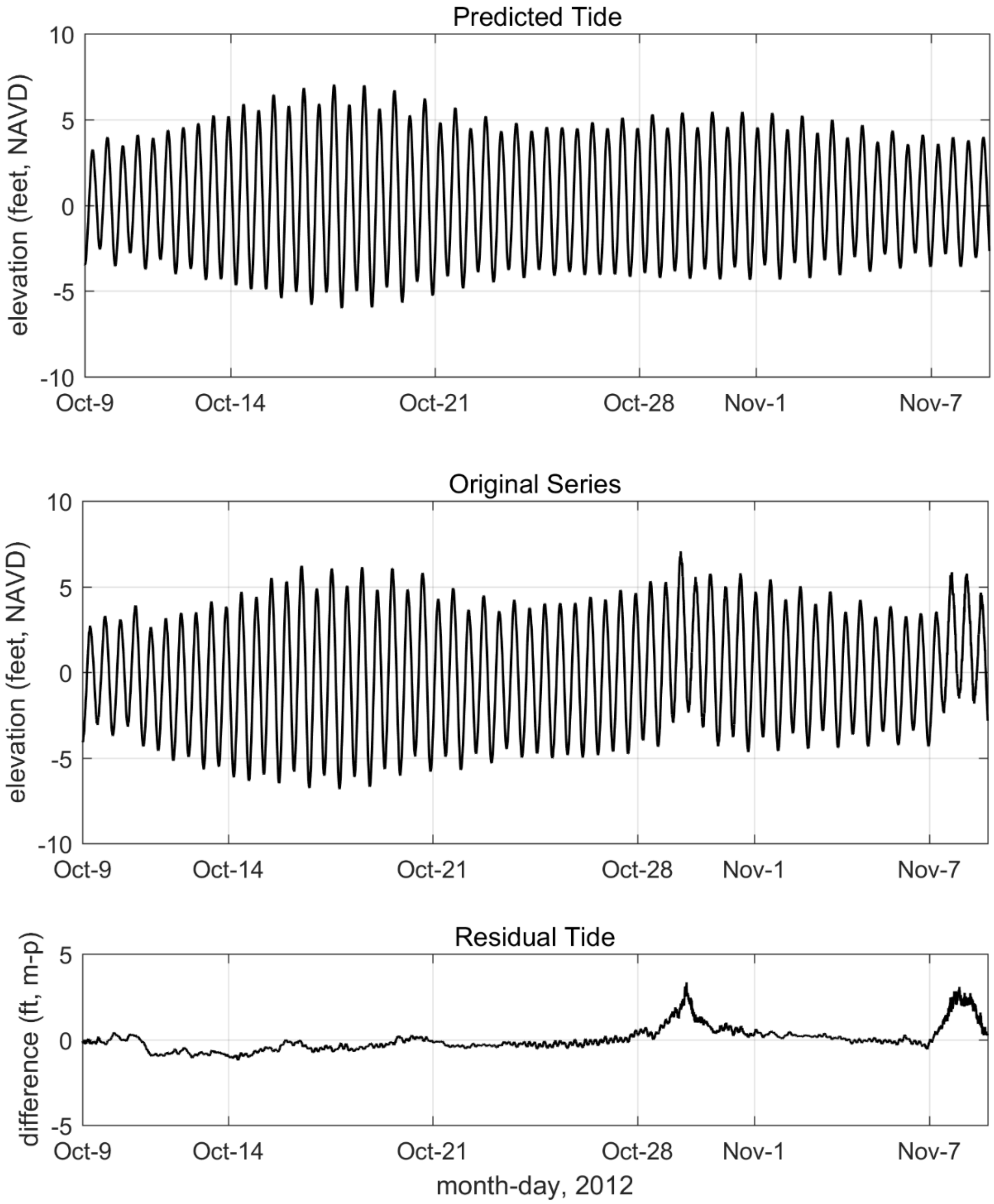


Figure V-6. Plot showing the comparison between the measured tide time series (top plot), and the predicted astronomical tide (middle plot) computed using the 21 individual tide constituents determine in the harmonic analysis of the Plymouth Bay gauge data, collected offshore Plymouth Harbor. The residual tide shown in the bottom plot is computed as the difference between the measured and predicted time series (r=m-p).

For constituents based on tidal elevations, the relative phase difference is computed as the difference between two times, the M_2 phase and the phase of the M_4, expressed as $\Phi=2M_2-M_4$. If Φ is between 0 and 180 degrees ($0<\Phi<180$), then the channel is characterized as being flood dominant, and peak flood velocities will be greater than for peak ebb. Alternately, if Φ were between 180 and 360 degrees ($180<\Phi<360$), then the channel would be ebb dominant. If Φ is exactly 0 or 180 degrees, neither flood nor ebb dominance occurs. For Φ equal to exactly 90 or 270 degrees, maximum tidal distortion occurs and the velocity residuals of a channel are greatest. This relative phase relationship is presented graphically in Figure V-7.

Though this method of tidal constituent analysis provides similar results to a visual inspection of a tidal record (e.g., by comparing peak ebb and flood velocities), it allows a more exact characterization of the tidal processes. By this analysis technique, a channel can be characterized as being strongly, moderately, or weakly flood or ebb dominant.

The four gauge stations in the harbor were used for this analysis. These data make it possible to characterize the flood or ebb dominance of different areas of the system from offshore (PLY1 in Plymouth Bay) through to the upper reaches of the system (e.g., PLY2, in Plymouth Harbor proper). The results of this velocity analysis of the Plymouth Bay system measured tide data show that although the offshore gauge is ebb dominant, all interior gauge stations indicate flood dominance. The computed values of $2M_2$-M_4 are presented in Table V-5.

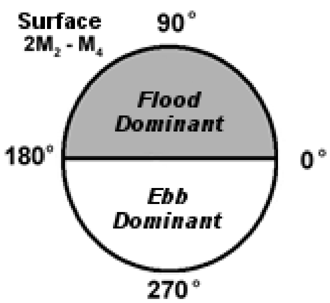


Figure V-7. Relative amplitude phase relationship of M2 and M4 tidal elevation constituents and characteristic dominance, indicated by the unit circle. Relative phase is computed as the difference of two times the M2 phase and the M4 phase (2M2-M4). A relative phase of exactly 0 or 180 degrees indicates a symmetric tide, which is neither flood nor ebb dominant.

Table V-5. Plymouth Harbor relative tidal phase differences of M_2 and M_4 tide constituents, determined using tide elevation record records.

location	$2M_2$-M_4 relative phase (deg)	Characteristic dominance
Plymouth Bay, PLY1	200.9	moderate ebb
Plymouth Harbor, PLY2	127.0	moderate flood
Jones River, PLY3	120.0	moderate flood
Breakwater, PLY4	129.8	moderate flood

V.2.3 ADCP Velocity Data Collection and Analysis

Cross-channel current measurements were surveyed at hourly intervals through a complete tidal cycle at two transect locations close to the inlet to the Plymouth-Kingston-Duxbury embayment system (Figure V-2). Current measurements were completed using an acoustic doppler current profiler (ADCP), which yields water velocities measured in discrete 0.5-meter (1.6-foot) increments between the surface and bay bottom. By conducting cross-channel transects the total volume of water passing across a transect line can be determined. This total volume is also an output of the hydrodynamic model, providing a straight forward comparison between modeled and observed volumes. The ADCP survey as undertaken in early November 2012.

The ADCP survey was conducted at two locations chosen to optimize the velocity measurements of water exiting and entering the embayment system on the both the ebb and flood tides. The first survey transect (T1) was located south of Duxbury Harbor extending from the eastern edge of the Cowyard channel to the western shore of Saquish Head. The second transect was located at the entrance to Plymouth Harbor Channel extending across the main channel east toward Plymouth Beach Point. ADCP transects are typically designed to run bank-to-bank in order to capture the complete volume of water flooding and ebbing at an embayment boundary. In the case of Plymouth Harbor and Duxbury Bay, broad tidal flats made it difficult to run the survey boat and take measurements in areas outside of the main channels at all stages of the tide.

The tidal exchange survey occurred during a single tide cycle. Throughout the survey, hourly velocity and discharge measurements were collected using the ADCP integrated with a differential GPS unit. By integrating the DGPS and ADCP into a single data stream, the velocity/discharge data collected have a precise global position associated with a known velocity/discharge measurement at a specific time.

Measured depth-averaged currents for peak flood and ebb flows are shown in Figure V-8 and V-9. During peak flood, the maximum depth averaged velocities were 2.1 ft/sec at Transect T1 and 2.6 ft/sec at Transect T2. Peak flood discharges across the transect lines were 67,400 ft^3/sec at Transect T1 and 33,100 ft^3/sec at Transect T2. During peak ebb flows, the maximum measured depth-averaged velocity was 2.3 ft/sec at both transects. Maxim ebb flow discharges across the transect lines were 96,100 ft^3/sec across Transect T1 and 32,000 ft^3/sec across Transect T2.

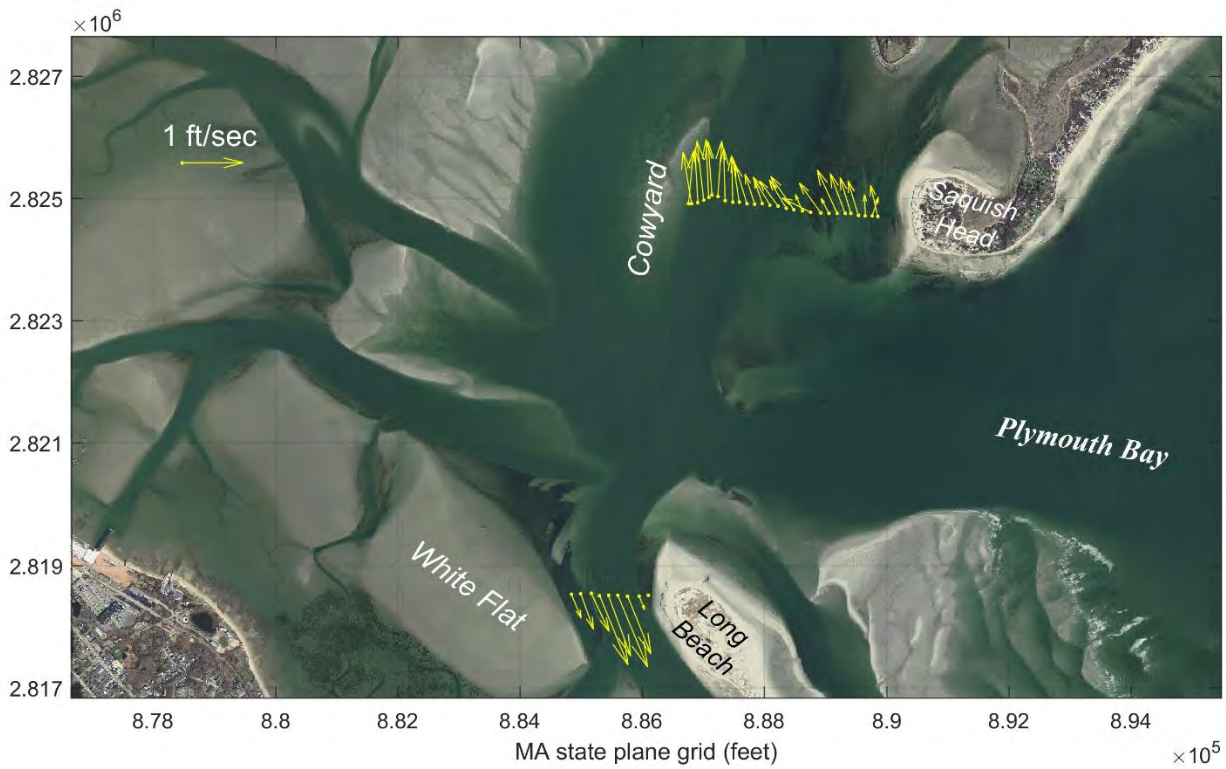


Figure V-8. Vector plot of maximum flood tide currents at the two ADCP transects followed during the November 1, 2012 survey. Transect time is 0952 for Transect 1 at Saquish Head, and 0920 for Transect 2 off Long Beach.

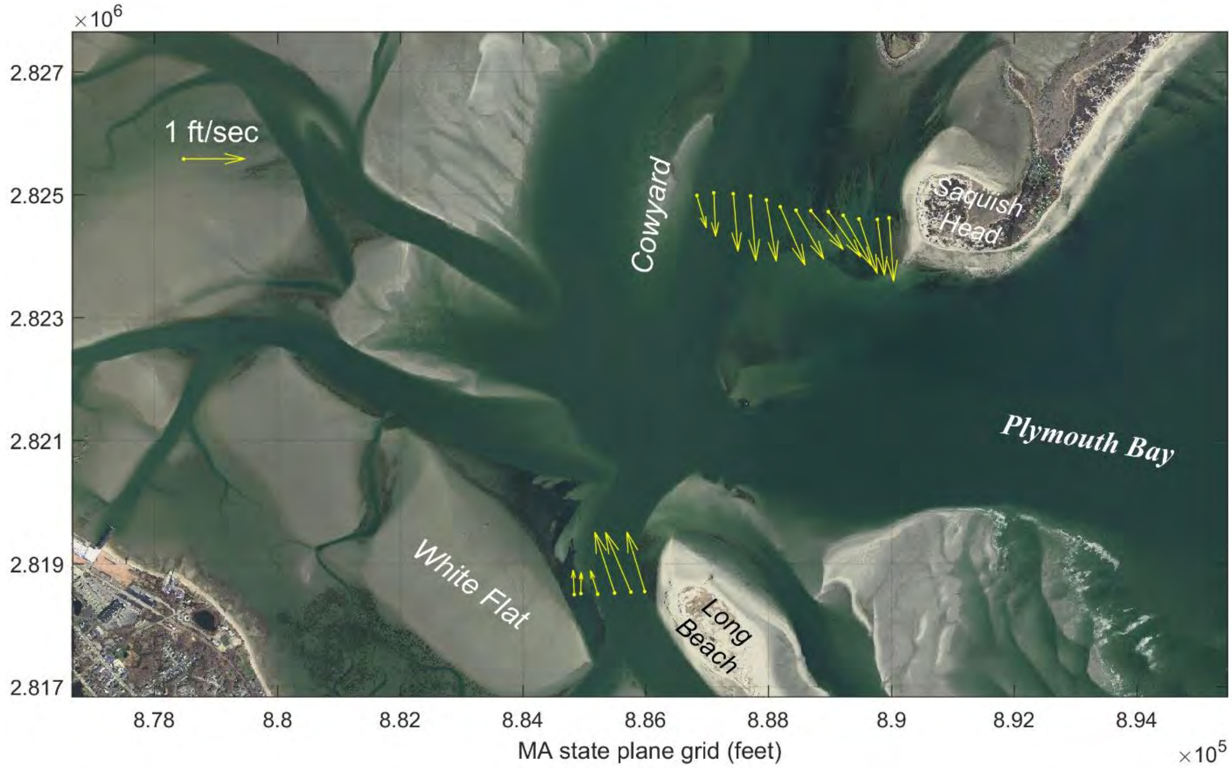


Figure V-9. Vector plot of maximum ebb tide currents at the two ADCP transects followed during the November 1, 2012 survey. Transect time is 1714 for Transect 1 at Saquish Head, and 1734 for Transect 2 off Long Beach.

V.3 HYDRODYNAMIC MODELING

For the modeling of the Plymouth-Kingston-Duxbury estuary system, MEP Technical Team members from Applied Coastal Research and Engineering (ACRE) utilized a hydrodynamic computer model to evaluate tidal circulation and flushing in the Harbor. 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 in southeastern Massachusetts under the MEP umbrella, including Sandwich Harbor, Barnstable Harbor, and Wellfleet Harbor.

V.3.1 Model Theory

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

RMA-2 is a finite element model designed for simulating one- and two-dimensional depth-averaged hydrodynamic systems. The dependent variables are velocity and water depth, and the equations solved are the depth-averaged Navier Stokes equations. Reynolds assumptions are incorporated as an eddy viscosity effect to represent turbulent energy losses. Other terms in the governing equations permit friction losses (approximated either by a Chezy or Manning formulation) for 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 prototypical 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.3.2 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using 2009 and 2013 digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the open water boundary of the grid based on the tide gauge data collected offshore the Harbor in Plymouth Bay. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several model calibration simulations for the system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.3.2.1 Grid generation

The grid generation process was aided by the use of the SMS package. Digital aerial orthophotos, the fall 2012 bathymetry survey data, 2010 USACE survey of the Harbor navigation channel and boat basin, and available 2013 LiDAR topography were imported to SMS, and a finite element grid was generated to represent the estuary. The aerial photograph was used to determine the land boundary of the system, as well as determine the surface coverage of salt marsh. The bathymetry and topography data were interpolated to the developed finite element mesh of the system. The completed grid consists of 6,159 nodes, which describe 2,843 total 2-dimensional (depth averaged) quadratic elements. The maximum nodal depth is -66.0ft (NGVD) in the natural channel of the harbor. The completed grid mesh of the Plymouth Bay estuary system is shown in Figure V-10.

The finite element grid for the system provides the detail necessary to accurately evaluate the variation in hydrodynamic properties of Plymouth Harbor. Areas of marsh were included in the model. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary. Grid resolution is generally governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability of the system. Relatively fine grid resolution is employed where complex flow patterns are expected, generally near the inlet. Appropriate implementation of wider node spacing and larger elements reduces computer run time with no sacrifice of accuracy.

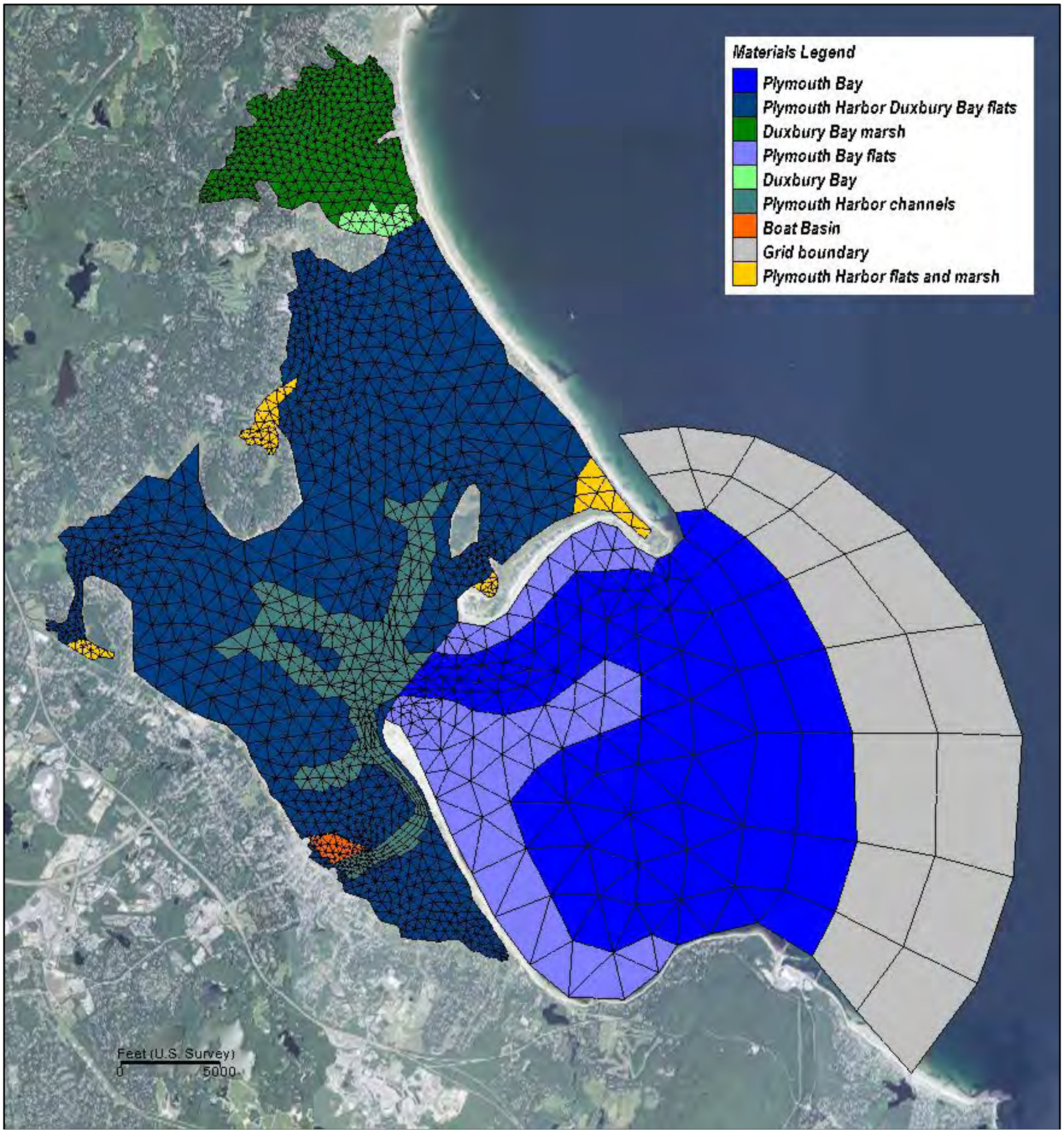


Figure V-10. Plot of hydrodynamic model grid mesh for Plymouth Harbor. Colors are used to designate the different model material types used to vary model calibration parameters and compute flushing rates.

V.3.2.2 Boundary condition specification

Three types of boundary conditions were employed for the RMA-2 model of the Plymouth Bay estuary system: 1) "slip" boundaries, 2) tidal elevation boundaries, and 3) constant flow input boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. A tidal boundary condition was specified using the data collected at the offshore gauge station. TDR measurements provided the required data. The rise and fall of the tide in the Bay is the primary driving force for estuarine circulation in this system. Dynamic (time-varying) model simulations specified a new water surface elevation at the open boundary of the Plymouth Harbor grid every model time step. The model runs of the Harbor used a 10-minute time step, which the same as the 10-minute sampling rate of the measured tide data. Details concerning the constant flow input boundary conditions included in the hydro model are discussed in Section VI.

V.3.2.3 Calibration

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

Calibration of the hydrodynamic model required a close match between the modeled and measured tides from stations inside the system (i.e., from the TDR deployments). Initially, the model was calibrated to obtain visual agreement between modeled and measured tides.

Once visual agreement was achieved, an eight-day period (16 tide cycles) was modeled to calibrate the model based on dominant tidal constituents discussed in Section V.2. The lunar-week period was extracted from a longer simulation to avoid effects of model spin-up, and to focus on average tidal conditions. Modeled tides for the calibration time period were evaluated for time (phase) lag and height damping of dominant tidal constituents. The calibration was performed for the eight-day period beginning October 15, 2012 as 1100 EDT.

After the model was calibrated, an additional model run was made in order corroborate the model performance using the discharge rates based on the ADCP-measured water column velocities. The model corroboration run period is 31 hours long and begins October 31, 2012 at 1200 EDT, which covers the duration of the survey on November 1, and includes a model spin-up period prior to the start of the survey.

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

V.3.2.3.a Friction coefficients

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

To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning's coefficients were matched to bottom type. For example, lower friction coefficients were specified for the main marsh creeks, versus the extensive marsh plain areas of the Harbor, which provide greater flow resistance by the presence of marsh vegetation. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within each estuary. Manning's values for different bottom types were initially selected based on ranges provided by the available engineering references (Chow, 1959). Values were incrementally changed as appropriate to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-6.

Table V-6. Manning's Roughness and eddy viscosity coefficients used in simulations of the Plymouth Bay estuary system. These embayment delineations correspond to the material type areas shown in Figure V-9.

System Embayment	bottom friction	eddy viscosity lb-sec/ft^2
Plymouth Bay	0.020	30.0
Plymouth Harbor/Duxbury Bay tide flats	0.022	27.0
Duxbury Bay marsh	0.030	40.0
Plymouth Bay tide flats	0.021	36.0
Duxbury Bay	0.025	40.0
Plymouth Harbor channels	0.022	23.0
Breakwater boat basin	0.022	28.0
Grid open boundary	0.025	80.0
Plymouth Harbor tide flats and marsh	0.030	50.0

V.3.2.3.b Turbulent exchange coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swifter, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). In most cases, the modeled systems were relatively insensitive to turbulent exchange coefficients because there were no regions of strong turbulent flow. Typically, model turbulence coefficients were set between 20 and 80 lb-sec/ft^2 (Table V-6). A higher value was used in the region of the grid boundary.

V.3.2.3.c Marsh porosity processes

Modeled hydrodynamics were complicated by wetting/drying cycles on the marsh plain included in the model of the Plymouth Bay system. Cyclically wet/dry areas of the marsh prevalent in Duxbury Bay 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 and tide flats. 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, similar to a sponge.

V.3.2.3.d Comparison of modeled tides and measured tide data

A best-fit of model output for the measured data was achieved using the aforementioned values for friction and turbulent exchange. Figures V-11 through V-14 illustrate sections of the eight-day simulation periods for the calibration model. Modeled (solid line) and measured (dotted line) tides are illustrated at each model location with a corresponding TDR.

Although visual calibration achieved reasonable modeled tidal hydrodynamics, further tidal constituent calibration was required to quantify the accuracy of the models. Calibration of the M_2 harmonic was the highest priority since M_2 accounted for a majority of the forcing tide energy throughout the system. Four tidal constituents were selected for constituent comparison: the K_1, M_2, M_4 and M_6.

Measured tidal constituent amplitudes are shown in Table V-7 for the calibration simulation. The constituent amplitudes shown in this table differ from those in Table V-2 because constituents were computed for only the separate seven-day sub-sections of the month-long period represented in Table V-2. In Table V-8 error statistics are shown for the calibration run.

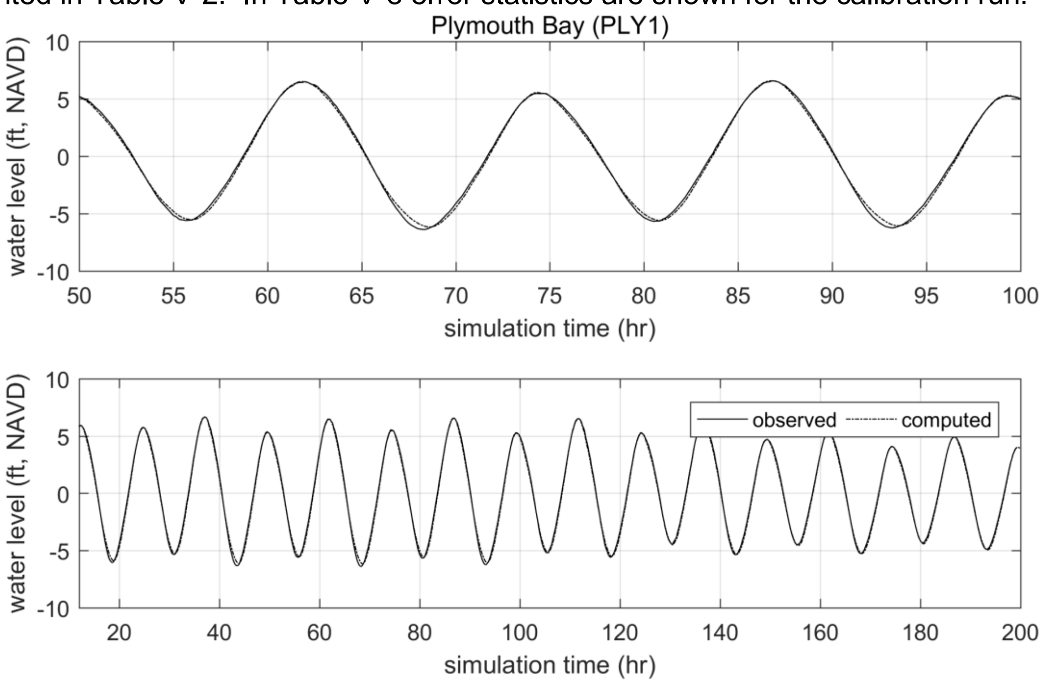


Figure V-11. Comparison of model output and measured tides for the offshore Plymouth Bay TDR station for the calibration model run (beginning October 15, 2012 as 1100 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot.

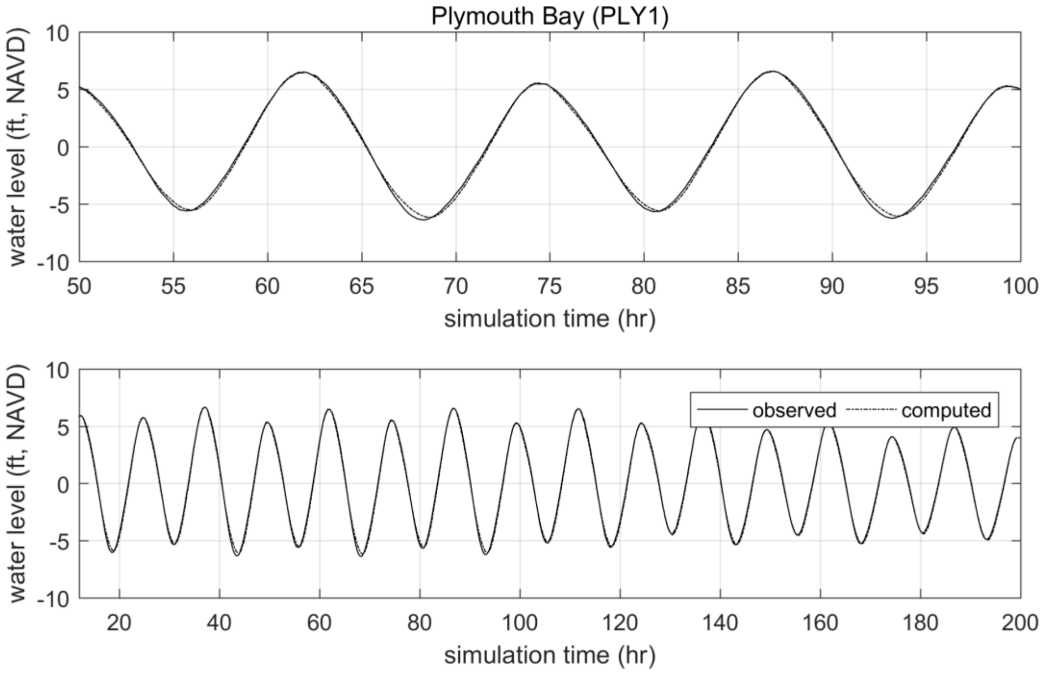


Figure V-12. Comparison of model output and measured tides for the TDR Plymouth Bay (PLY2) for the calibration model run (beginning October 15, 2012 as 1100 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot.

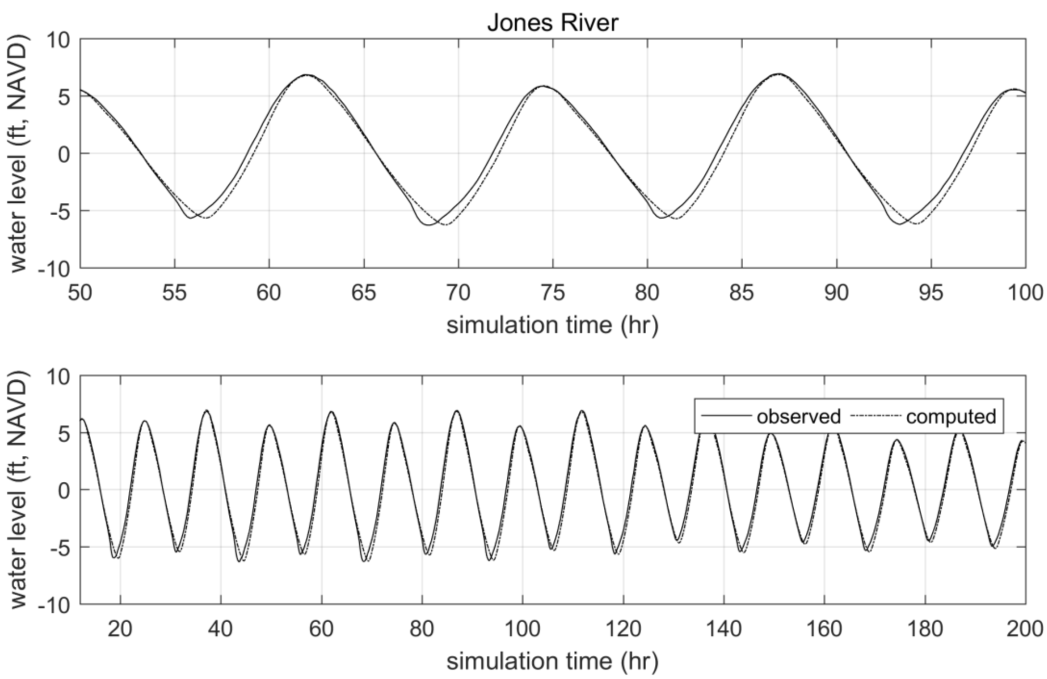


Figure V-13. Comparison of model output and measured tides for the Jones River TDR location (PLY3) for the final calibration model run (beginning October 15, 2012 as 1100 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot.

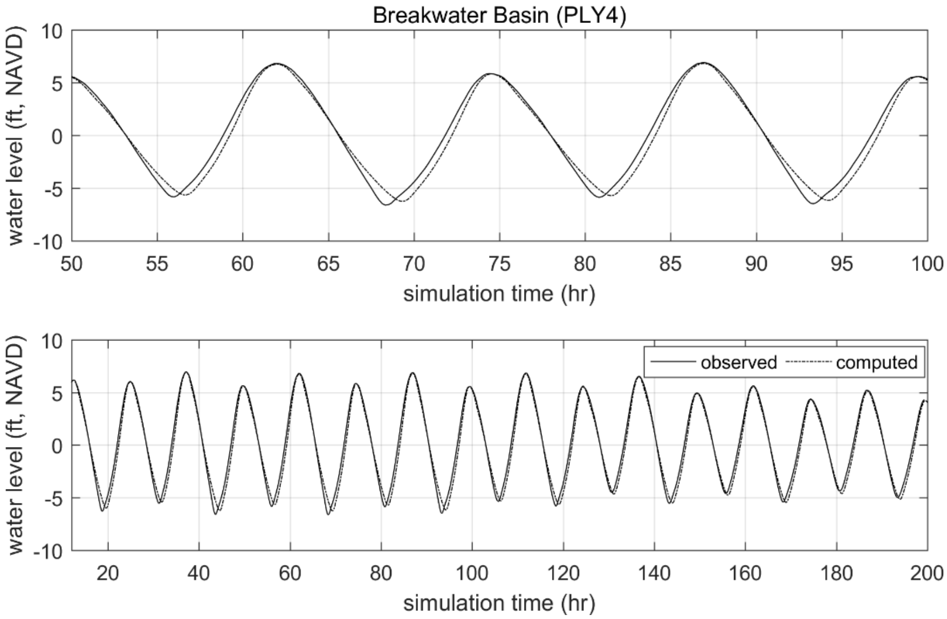


Figure V-14. Comparison of model output and measured tides for the Breakwater Basin TDR station (PLY4) for the final calibration model run (beginning October 15, 2012 as 1100 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot.

The constituent calibration resulted in excellent agreement between modeled and measured tides. The errors associated with tidal constituent amplitude for both the calibration and verification simulations were on the order of 0.1 ft, which is of the same order of magnitude accuracy as that of the tide gauges (0.25 ft). Time lag errors for the main estuary reach were generally less than the time increment resolved by the model and tide data (10 minutes), indicating good agreement between the model and data. The skill of the model calibration is also demonstrated by the high degree of correlation (R^2) and low RMS error shown in Table V-9 for all stations.

V.3.2.4 ADCP corroboration of hydrodynamic model

An additional evaluation of model corroboration with measured data was performed by comparing model flow rates and ADCP field measurements. An ADCP survey of flow velocities along two cross-channel transects at Saquish Head and Long Beach (Figure V-2) was executed on November 1, 2012. During this survey, velocities were measured by a boat-mounted ADCP that traversed the transects over an eight hour and 20-minute period during the course of the survey day. Flow rates were output from the model at continuity lines placed across the channel in the same location as the ADCP transects. The comparison of ADCP measurement-derived flow rates and model output is presented in Figure V-15 and V-16. The comparisons between model output and ADCP flow rates is very good, further indicating that the hydrodynamic model adequately represents the physics of the real system. For Transect 1 (Figure V-15), the R^2 correlation between model output and measurements is 0.91, and the RMS error of the model output is 18,390 ft^3/sec, which is 27% of the maximum measured flow rate. At Transect 2, the R^2 correlation between model output and measurements is 0.89, and the RMS error of the model output is 7,940 ft^3/sec, which is 24% of the maximum measured flow rate. Some of the error observed in the model/measurement comparison is most likely a result of the transects not being bank-to-bank. Because the survey boat did not traverse the full width of the embayments, the total discharge flow in and out of the bays could not be completely measured. The error would result from slight differences in the distribution of flows in the model and the actual conditions in the embayments.

Table V-7. Tidal constituents for measured water level data and calibrated model output, with model error amplitudes, for Plymouth-Kingston-Duxbury embayments, during modeled calibration time period.

Model calibration run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕK_1
Plymouth Bay	5.23	0.14	0.11	0.56	23.3	21.8
Plymouth Harbor	5.21	0.55	0.15	0.58	34.7	31.1
Jones River	5.22	0.55	0.14	0.58	34.0	30.8
Breakwater Basin	5.21	0.54	0.15	0.58	34.7	31.1
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕK_1
5.30	0.07	0.14	0.57	18.1	16.9	5.30
5.39	0.22	0.21	0.58	24.2	22.2	5.39
5.29	0.27	0.20	0.58	24.9	24.3	5.29
5.37	0.20	0.22	0.58	24.0	22.0	5.37
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕK_1
Plymouth Bay	0.07	-0.07	0.03	0.01	-10.9	-19.3
Plymouth Harbor	0.18	-0.33	0.06	0.00	-21.7	-35.8
Jones River	0.07	-0.28	0.06	0.00	-18.9	-25.9
Breakwater Basin	0.16	-0.34	0.07	0.00	-22.1	-36.3

Table V-8. Error statistics for the Plymouth Harbor hydrodynamic model, for model calibration.

	R^2	RMS error (feet)
Plymouth Bay	1.00	0.2
Plymouth Harbor	0.98	0.6
Jones River	0.98	0.5
Breakwater Basin	0.97	0.6

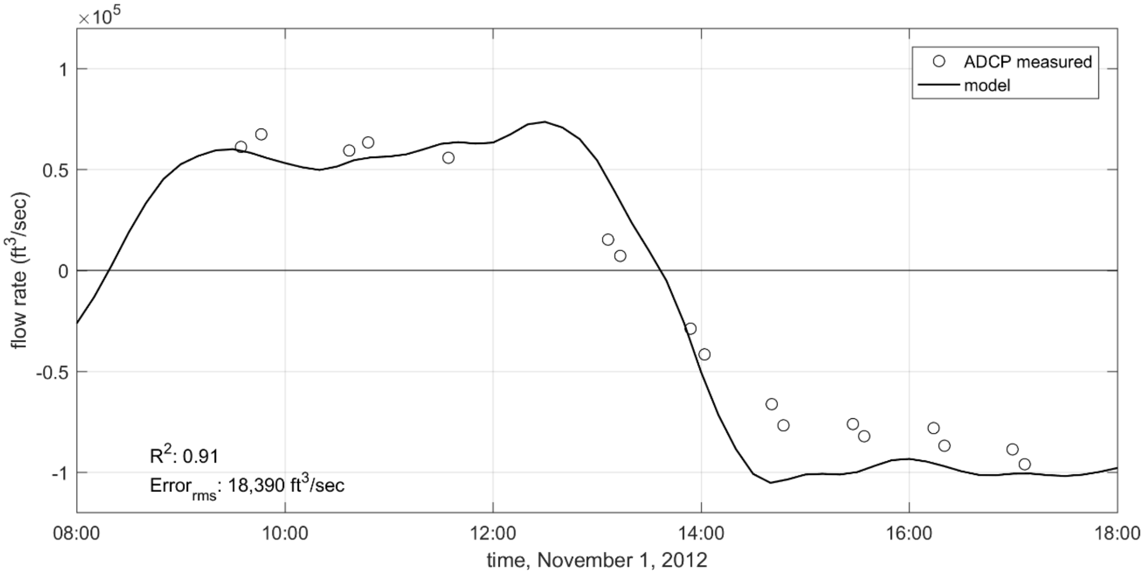


Figure V-15. Comparison of flow rates determined using ADCP velocity data and modeled flow rates at survey Transect 1, in the vicinity of Saquish Head (Figure V-2).

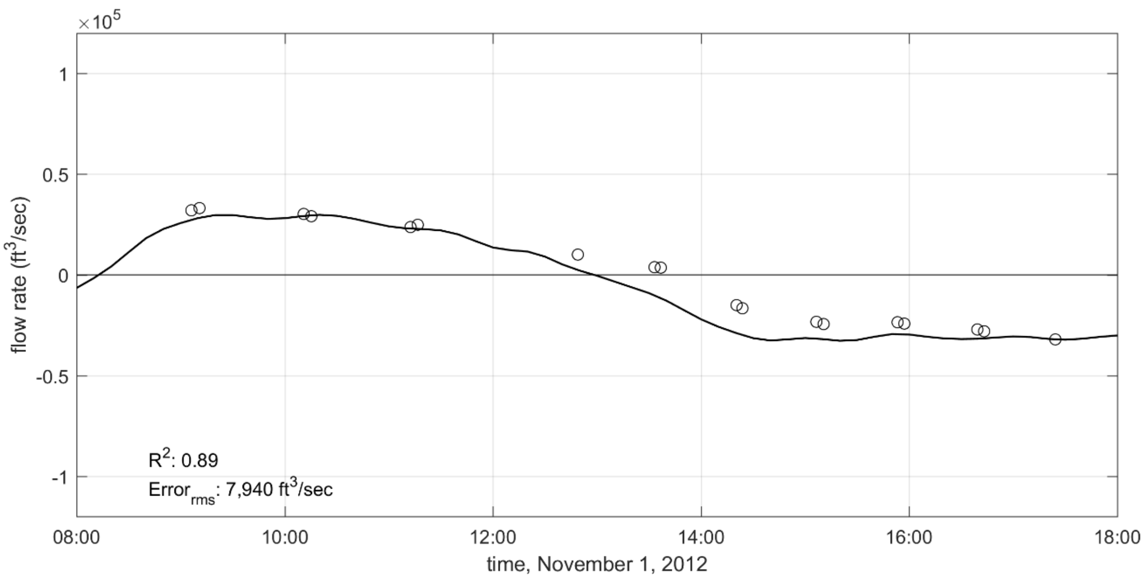


Figure V-16. Comparison of flow rates determined using ADCP velocity data and modeled flow rates at survey Transect 2, in the vicinity of the northern tip of Long Beach (Figure V-2).

V.3.3 Model Circulation Characteristics

The final calibrated model serves as a useful tool in investigating circulation characteristics of the whole Plymouth Bay estuary system. Inputs of bathymetry and tide data can be leveraged to develop further insight into tidal velocities and flow rates 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. As an example, Figure V-17 shows color contours and vectors that indicate velocity during a single model time step, during a period of maximum ebb currents at the entrance to the harbor.

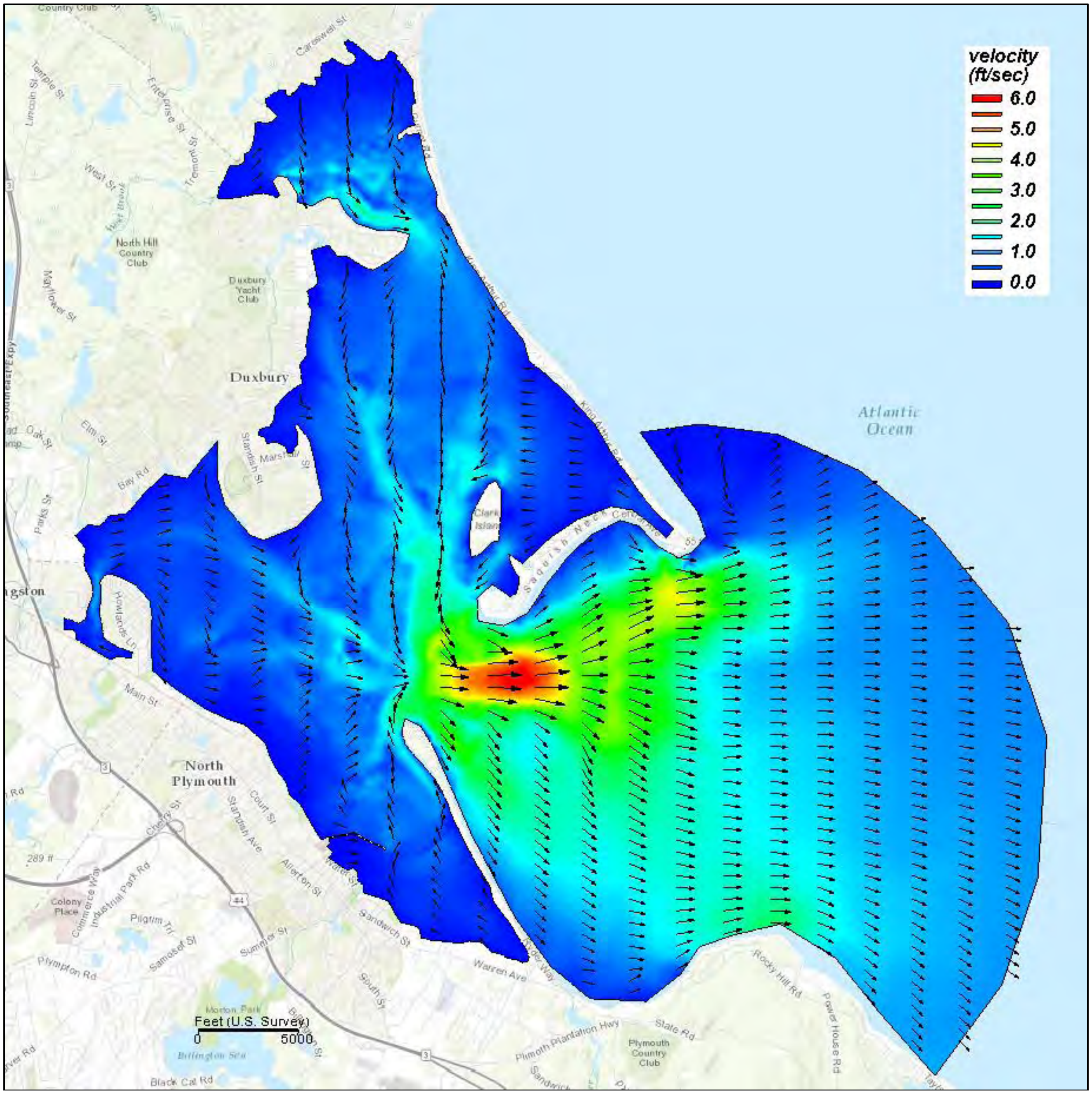


Figure V-17. Example of Plymouth Harbor hydrodynamic model output for a single time step during an ebbing tide. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

As another example, from the calibration model run of the model, the total flow rate of water flowing through the harbor entrance (including all flows into and out of Plymouth Harbor, Duxbury Bay and Kingston Bay and their attached sub-embayments) can be computed with the hydrodynamic model. The variation of flow as the tide floods and ebbs is seen in the plot of system flow rates in Figure V-18. During spring tides, the maximum flood flow rates into the harbor reach 443,200 ft^3/sec. Maximum ebb flow rates during spring tides are smaller about three-quarters of the flood flow rates experienced during spring tides (300,300 ft^3/sec).

V.3.4 Flushing Characteristics

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

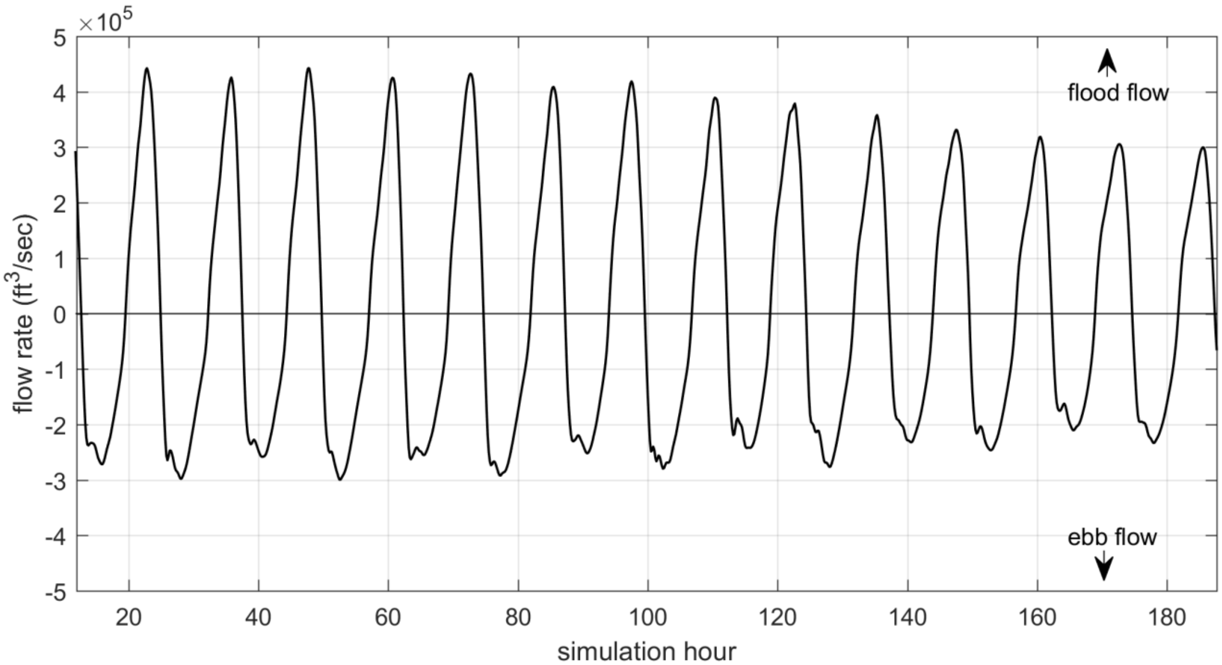


Figure V-18. Time variation of computed flow rates for the whole of the Plymouth Bay estuary system. Model period shown corresponds to spring tide conditions, where the tide range is the largest, and resulting flow rates are correspondingly large compared to neap tide conditions. Positive flow indicated flooding tide flows, while negative flow indicates ebbing tide flows.

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 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 system residence time, V_{system} represents volume of the (entire) system at mean tide level, P equals the system's tidal prism (or volume entering the system through a single tidal cycle), and t_{cycle} the period of the tidal cycle, taken typically as the period of the M_2 tide, or 12.42 hours (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, is 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. In the case of the combined Plymouth-Kingston-Duxbury embayments, the system residence time is the average time required for water to migrate out of the entrance at the tip of Long Beach, then across Plymouth Bay, and finally out into Cape Cod Bay. Alternatively, the local residence time is the average time required for water to migrate from inside the entrance and into Plymouth Bay (not all the way to Cape Cod Bay). 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.

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. It is impossible to evaluate an estuary's health based solely on flushing rates. 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 is obtained from applying the calibrated hydrodynamic model as described in the following section of this report (Section VI) and by extending the model to include pollutant/nutrient dispersion. The water quality model provides an additional valuable tool to evaluate the complex mechanisms governing estuarine water quality in the Harbor 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 as two subdivisions of the system. In addition, system and local residence times were computed to indicate the range of conditions possible for the system.

Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged over a flood tidal cycle (tidal prism). The mean volumes and tide prisms of the portion of the system inside the entrance between Saquish Head and Long Beach, and the whole system including Plymouth Bay that are used in this analysis are presented in Table V-9.

Table V-9. Plymouth Harbor mean volume and average tidal prism during simulation period.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Plymouth Bay/Plymouth Harbor/Kingston Bay/Duxbury Bay	9,920,479,800	8,985,064,300
Plymouth Harbor/Kingston Bay/Duxbury Bay	4,251,849,900	4,745,974,900
Duxbury Bay	1,903,007,500	2,362,917,600
Plymouth Harbor	623,638,800	739,410,300

Residence times represent average values for 8-tidal-day period (16 tide cycles) run for the model calibration period, and are listed in Table V-10. The modeled time period includes the transition between from spring to neap tide conditions. The RMA-2 model calculated flow crossing specified grid continuity lines (similar to an ADCP transect) for each sub-embayment to compute the tidal prism volume. Since the lunar-week period used to compute the flushing rates of the system represent average tidal conditions, it provides an appropriate method for determining mean flushing rates for the system sub-embayments.

Table V-10. Computed System and Local residence times for the Plymouth Bay estuary system.

Embayment	System Residence Time (days)	Local Residence Time (days)
Plymouth Bay/Plymouth Harbor/Kingston Bay/Duxbury Bay	0.6	0.6
Plymouth Harbor/Kingston Bay/Duxbury Bay	1.1	0.5
Duxbury Bay	2.2	0.4
Plymouth Harbor	6.9	0.4

The computed flushing rates for the entire system show that as a whole, the system flushes very well. A flushing time of 0.6 days for the entire estuary shows that on average, water is resident in the system for less than one day. The low local residence times for the whole of the Plymouth Bay estuary system show that water quality in the system is not impacted negatively by tidal flushing. This is a typical result for estuaries dominated by marsh resources or with extensive tidal flats, where the tide prism volume is of a comparable magnitude to the mean volume of the system. For the area of the system inside the entrance between Saquish Head and Long Beach (including Plymouth Harbor, Duxbury Bay and Kingston Bay), the system residence time is only slightly longer, and still about one day.

Local flushing rates are smallest for Duxbury Bay and Plymouth Harbor, which both have a residence time less than one-half day. The sub-embayment with the longest system residence time is Plymouth Harbor, with a flushing time of nearly seven days. This indicates that this area would be more sensitive to watershed N loading, compared to other areas of the estuary.

Based on our knowledge of estuarine processes, we estimate that the combined errors associated with the method applied to compute residence times are within 10% to 15% of "true" residence times, for the Plymouth Bay estuary system. 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 in some of the smaller sub-embayments of the system.

Minor errors may be introduced in residence time calculations by simplifying assumptions. Flushing rate calculations assume that water exiting an estuary or sub-embayment does not return on the following tidal cycle. For regions where a strong littoral drift exists, this assumption is valid. However, water exiting a small sub-embayment on a relatively calm day may not completely mix with estuarine waters. In this case, the "strong littoral drift" assumption would lead to an under-prediction of residence time. Since littoral drift along the shoreline of Cape Cod Bay typically is strong because of the effects of the local winds and tidal induced mixing, the "strong littoral drift" assumption will cause only minor errors in residence time calculations.

VI. WATER QUALITY MODELING

VI.1 DATA SOURCES FOR THE MODEL

Several different data types and calculations are required to support the water quality modeling effort for the Plymouth Bay system (inclusive of Kingston Bay and Duxbury Bay). These include the output from the hydrodynamics model, calculations of external nitrogen loads from the watersheds (surface and ground water inflow and loads), 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 embayment was an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated hydrodynamic model representing the transport of water within the Plymouth Bay system. Files of node locations and node connectivity for the RMA-2V model grids were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic model output used for the water quality model calibration was the lunar-week (14 tide cycles) period beginning October 15, 2012 as 1100 EDT. This period overlaps with the time period used for the hydrodynamic model calibration and also the flushing analysis presented in Section V. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to allow the model to reach a dynamic “steady state”, and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayments

Three primary nitrogen loads to the Plymouth Bay embayment system were utilized 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 system, consisting of the background concentrations of total nitrogen in the water entering from Cape Cod Bay. This load is represented as a constant concentration along the seaward boundary of the model grid.

VI.1.3 Measured Nitrogen Concentrations in the Embayment

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in the area map presented in Figure VI-1. The multi-year averages present the “best” comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data are the minimum required to provide a baseline for MEP analysis. For stations in the Plymouth Bay system, a total of five individual years of water quality data (with data at all stations) were available between 2003 and 2013 in support of the modeling effort.

Table VI-1. Measured data and modeled total nitrogen (TN) concentrations for the Plymouth Bay estuarine system. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of all measurements. Data represented in this table were collected in the summers of 2003, 2004, 2005, 2007 and 2013.

Location	Monitoring station	Data Mean	s.d. all data	N	model min	model max	model average
Plymouth Harbor - south	PDH1	0.365	0.062	22	0.308	0.354	0.325
Plymouth Harbor - boat basin	PDH2	0.343	0.072	27	0.310	0.321	0.315
Plymouth Harbor - mid	PDH3	0.316	0.069	24	0.278	0.318	0.296
Plymouth Harbor - north	PDH4	0.302	0.062	27	0.269	0.299	0.278
Plymouth Harbor - channel	PDH5	0.286	0.061	24	0.262	0.293	0.274
Kingston Bay - east	PDH6	0.271	0.063	24	0.261	0.289	0.274
Kingston Bay - Rocky Nook	PDH7	0.328	0.064	29	0.273	0.290	0.282
Kingston Bay - Goose Point	PDH8	0.324	0.115	27	0.259	0.329	0.281
Jones River	PDH9	0.434	0.082	27	0.321	0.517	0.389
Plymouth Bay	PDH10	0.241	0.065	27	0.251	0.292	0.265
Duxbury Bay - Cowyard	PDH11	0.301	0.063	30	0.247	0.332	0.276
Duxbury Bay - Saquish Neck	PDH12	0.345	0.107	30	0.287	0.324	0.307
Duxbury Bay - mid	PDH13	0.300	0.064	29	0.266	0.470	0.342
Duxbury Bay - west	PDH14	0.372	0.106	27	0.281	0.471	0.347
Duxbury Marsh	PDH15	0.471	0.132	28	0.324	0.527	0.430

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 Plymouth Bay embayment 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 Plymouth Bay and its attached sub-embayments (Kingston Bay and Duxbury Bay). Like the RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. The MEP Technical Team has utilized this model in water quality studies of other embayment systems across southeastern Massachusetts, including but not limited to Sandwich Harbor (Howes *et al*., 2015); Barnstable Harbor (Howes *et al*., 2017); Edgartown Great Pond, MA (Howes *et al*., 2008) and Wellfleet Harbor (Howes et al., 2017).

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 and water quality conditions tend to be the lowest of the year. 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 MEP Technical Team watershed loading analysis, as well as the measured bottom sediment nitrogen fluxes. Water column nitrogen measurements were utilized as model boundaries and as calibration data. Hydrodynamic model output (discussed in Section V) provided the remaining information (tides, currents, and bathymetry) needed to parameterize the water quality model of the Plymouth Bay system.

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

VI.2.1 Model Formulation

The formulation of the model is for two-dimensional depth-averaged systems in which concentration in the vertical direction is assumed uniform. The depth-averaged assumption is justified since vertical mixing by wind and tidal processes prevent significant stratification in the modeled embayment. 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 is the water quality constituent concentration; t is time; u and v are the velocities in the x and y directions, respectively; D_x and D_y are the model dispersion coefficients in the x and y directions; and σ is the constituent source/sink term. Since the model utilizes input from the RMA-2 model, a similar implicit solution technique is employed for the RMA-4 model.

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

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

For each model, an initial total N concentration equal to the concentration at the open boundary was applied to the entire model domain. The model was then run for a simulated month-long (28 day) spin-up period. At the end of the spin-up period, the model was run for an additional 14-day (336 hour) period. Model results were recorded only after the initial spin-up period. The time step used for the water quality computations was 10 minutes, which corresponds to the time step of the hydrodynamics input for the Plymouth Bay hydrodynamic model.

VI.2.3 Boundary Condition Specification

Mass loading of nitrogen into each model included: 1) sources developed from the results of the watershed analysis, 2) estimates of direct atmospheric deposition, and 3) summer benthic regeneration. Nitrogen loads from each separate sub-watershed to the embayment were distributed by watershed. For example, the watershed load for the Jones River was input within the model area that represents this surface water discharge point into the Bay. Benthic loads were distributed in a similar manner.

The loadings used to model present conditions in the Plymouth Bay embayment 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 plain, when present), resulting in a total flux for each portion of the overall 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. Fluxes range between negative (uptake = sink of nitrogen) and positive (release = source of nitrogen) values in different areas of the system. In Plymouth Bay, offshore of Long Beach, the net benthic flux is negative which indicates a net uptake of nitrogen in the bottom sediments. The greatest measured positive fluxes exist in the channels of Duxbury Marsh.

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. The total nitrogen concentration of the incoming water was set at the value designated for the open boundary. The boundary concentration in Cape Cod Bay, offshore the harbor inlet, was set at 0.241 mg/L, based on SMAST data collected at the system inlet (PDH10).

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Duxbury Marsh	32.918	5.589	44.988
Duxbury Bay	16.121	59.200	14.576
Kingston Bay	61.926	49.227	40.756
Plymouth Harbor	48.671	25.614	50.707
Blue Fish River	24.140	-	-
Jones River	116.488	-	-
Town Brook	70.811	-	-
Eel River	44.153	-	-
Plymouth WWTF Outfall	39.485	-	-
System Total	454.712	139.630	151.027

VI.2.4 Model Calibration

The development of the Plymouth Bay water quality model began with the parameterization and calibration of the salinity model. Salinity is a conservative water quality constituent and therefore ideally suited for model calibration. Model dispersion coefficients were adjusted so that model output for salinity matched measured data from the Bay. Generally, several model runs were required to bring the model into agreement with the water column measurements. Dispersion coefficient (*E*) values were varied through the modeled system by setting different values of *E* for each grid material type, as designated in Section V. Observed values of *E* in coastal estuary areas typically range between order 10 and order 0.001 m^2/sec (USACE, 2001). The final values of *E* used in each sub-embayment of the modeled system are presented in Table VI-3. These values were used to develop the "best-fit" salinity model calibration. For the case of salinity 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 the model domain.

The only required inputs into the RMA-4 salinity model of the system, in addition to the RMA-2 hydrodynamic model output, were salinities at the model open boundary, and freshwater inputs (including inputs from rain, surface streams and groundwater). The open boundary salinity in Cape Cod Bay was set at 31.2 ppt. Surface water and groundwater input salinities were set at 0 ppt. Fresh water inputs into the model are provided in Section III.

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

For model calibration, the target modeled salinities were compared to mean measured salinity data values at all water-quality monitoring stations. The calibration target was set between the modeled maximum and tidal averaged concentration at each station, in order to represent samples collected at or after the time of mid-ebb tide offshore in Cape Cod Bay.

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

Embayment Division	E m^2/sec
Cape Cod Bay	5.0
Plymouth Bay	5.0
Kingston Bay	16.0
Duxbury Bay	5.0
Duxbury Marsh	1.0
Plymouth Harbor basin	5.0
Plymouth Harbor	10.0
Jones River	15.0

Also presented in Figure VI-3 are unity plot comparisons of measured data verses modeled target values for each system. The root mean squared (rms) error of the run is 1.1 ppt, which demonstrates good agreement between modeled and measured data for this system.

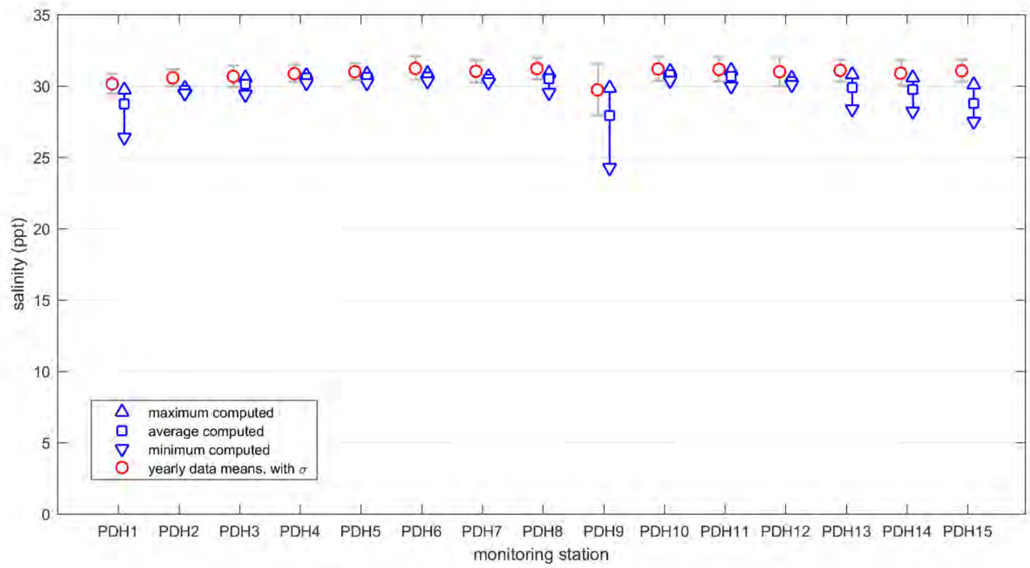


Figure VI-2. Comparison of measured total salinity and calibrated model output at stations in Plymouth Bay. Station labels correspond with the MEP IDs provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle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

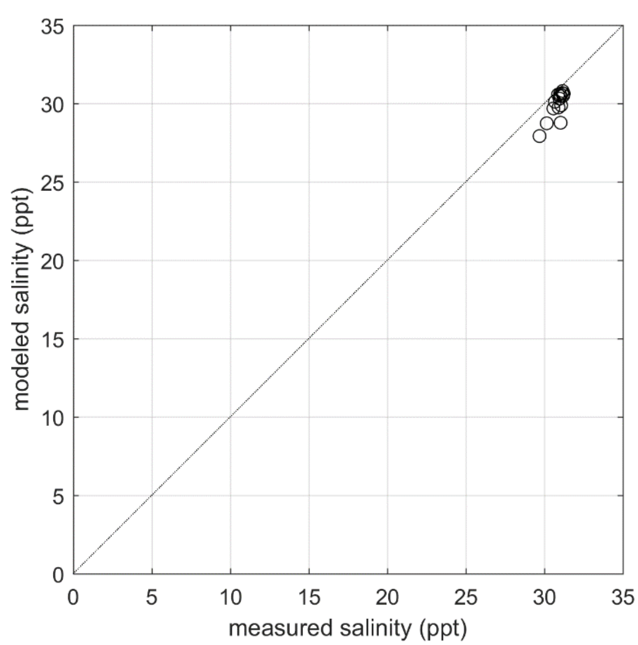


Figure VI-3. Model salinity calibration target values are plotted against measured concentrations, together with the unity line. Computed error (rms) for the model is 1.05 ppt.

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

VI.2.5 Model Verification

In addition to the model calibration based on salinity, the numerical water quality model performance was verified by modeling total nitrogen (TN). This step was performed for the Plymouth Bay system using TN measurements collected at the same stations as the salinity data and N loads from Table VI-2. For the TN verification, none of the model dispersion coefficients were changed from the values used in the salinity calibration. Comparisons of modeled and measured TN concentrations are presented in Figures VI-5 and VI-6, with contour plots of model output shown in Figure VI-7. The R^2 correlation of the model and measurements is 0.82 and the rms error of the model is 0.033 mg/L.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

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

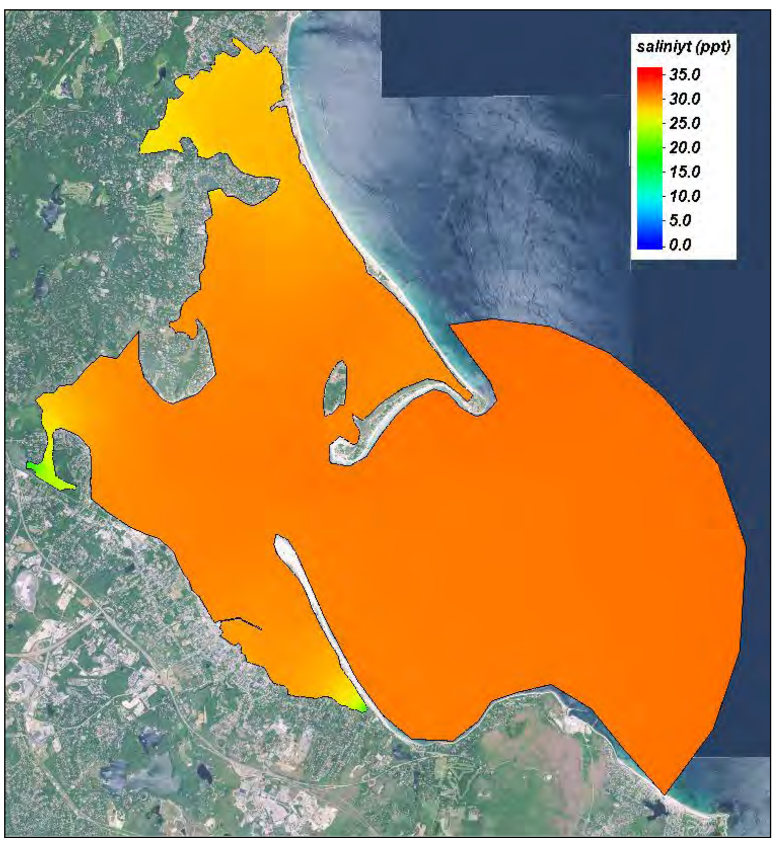


Figure VI-4. Contour Plot of average modeled salinity (ppt) in the Plymouth Bay system.

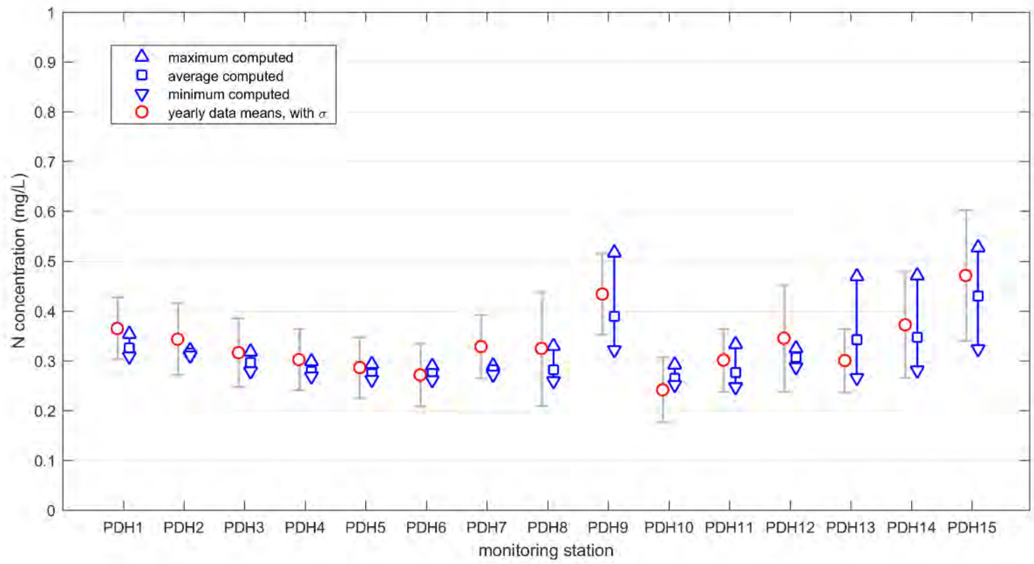


Figure VI-5. Comparison of measured and calibrated TN model output at stations in Plymouth Bay. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with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

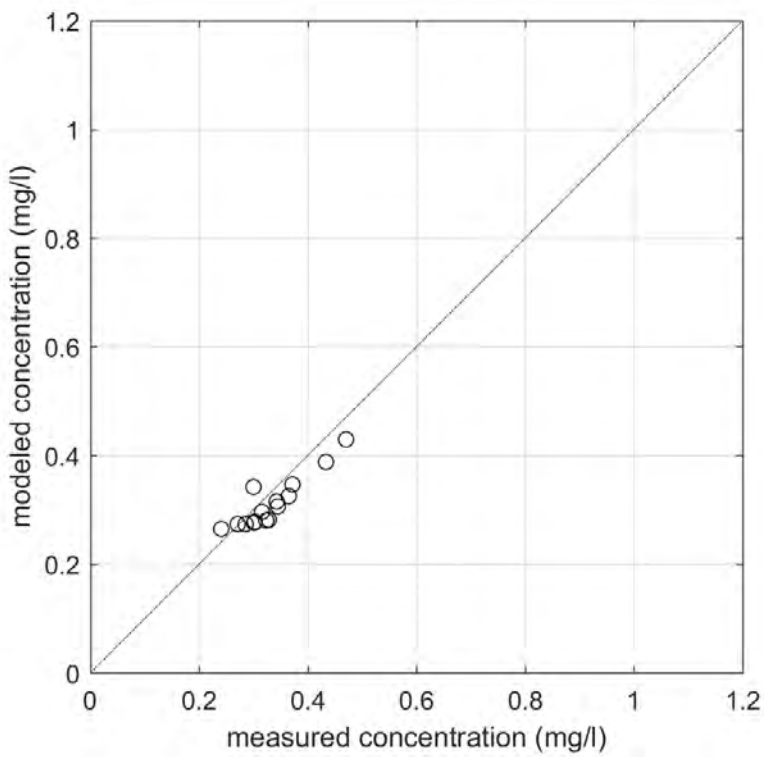


Figure VI-6. Model TN target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) is 0.82 and RMS error for this model verification run is 0.033 mg/L.

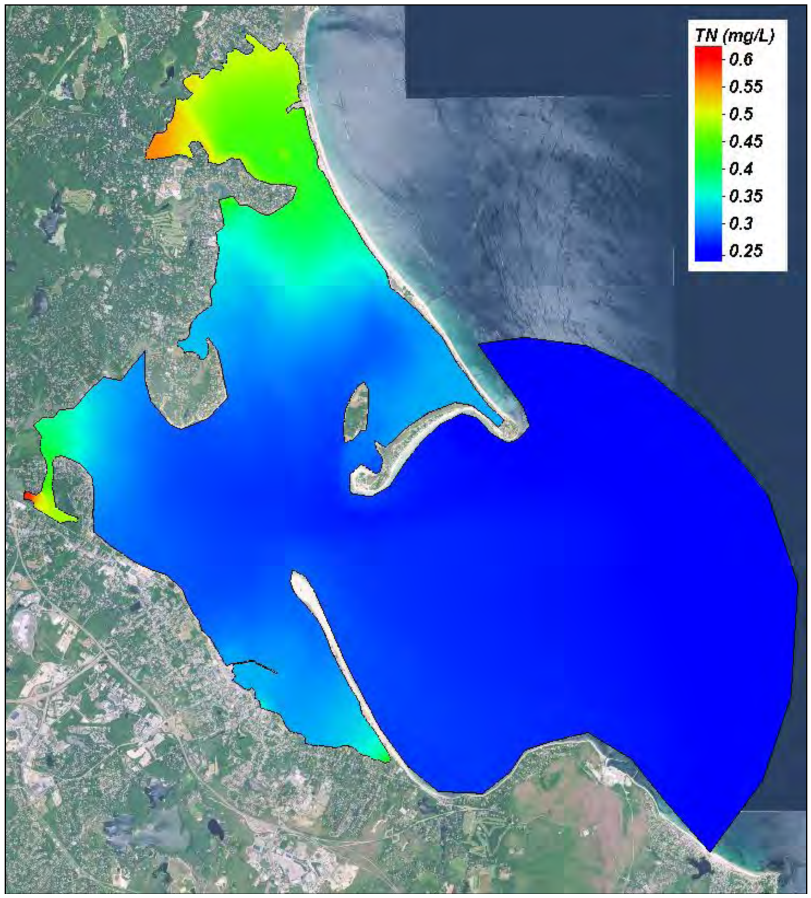


Figure VI-7. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for Plymouth Bay system.

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 Plymouth 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
Duxbury Marsh	32.918	42.277	+28.4%	2.351	-92.9%
Duxbury Bay	16.121	17.866	+10.8%	0.567	-96.5%
Kingston Bay	61.926	74.455	+20.2%	3.805	-93.9%
Plymouth Harbor	48.671	64.414	+32.3%	2.129	-95.6%
Blue Fish River	24.140	27.126	+12.4%	2.532	-89.5%
Jones River	106.488	144.381	+23.9%	26.044	-77.6%
Town Brook	70.811	88.419	+24.9%	6.323	-91.1%
Eel River	44.153	52.052	+17.9%	5.827	-86.8%
Plymouth WWTF Outfall	39.485	40.890	+3.6%	0.000	-100.0%
system total	454.712	551.879	+21.4%	49.578	-89.1%

VI.2.6.1 Build-Out

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

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Following development of the nitrogen loading estimates for the build-out scenario, the water quality models of the system was run to determine TN concentrations within each sub-embayment (Table VI-6). In this table, the percent change P over background presented in this table is calculated as:

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

where N is the total nitrogen concentration at the indicated monitoring station for present conditions and the loading scenario (i.e., build-out in this case), and also in Cape Cod Bay (background). Total nitrogen concentrations in the receiving waters (i.e., Cape Cod Bay) remained identical to the existing conditions modeling scenarios. For build-out, the percent increase in modeled TN concentrations is greatest at the stations near the Jones and Eel Rivers (PDH1 and PDH9). Concentrations increased more than 20% above background at these two monitoring stations. The largest TN magnitude change occurs also at stations PDH1 and PDH9, where average TN increases 0.031 mg/L. A contour plot showing average TN concentrations throughout the harbor system is presented in Figure VI-8 for the model of build-out loading.

Table VI-5. **Build-out** scenario sub-embayment and surface water loads used for total nitrogen modeling of the Plymouth 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 net (kg/day)
Duxbury Marsh	42.277	5.589	49.487
Duxbury Bay	17.866	59.200	16.398
Kingston Bay	74.455	49.227	42.294
Plymouth Harbor	64.414	25.614	53.448
Blue Fish River	27.126	-	-
Jones River	144.381	-	-
Town Brook	88.419	-	-
Eel River	52.052	-	-
Plymouth WWTF Outfall	40.890	-	-
system total	551.879	139.630	161.627

Table VI-6. Comparison of model average TN concentrations from present loading and the **build-out scenario**, with percent change over background in Cape Cod Bay (0.241 mg/L), for the Plymouth Bay system.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	build-out (mg/L)	% change
Plymouth Harbor - south	PDH1	0.325	0.342	+20.3%
Plymouth Harbor - boat basin	PDH2	0.315	0.330	+19.7%
Plymouth Harbor - mid	PDH3	0.296	0.306	+18.1%
Plymouth Harbor - north	PDH4	0.278	0.285	+16.9%
Plymouth Harbor - channel	PDH5	0.274	0.280	+17.1%
Kingston Bay - east	PDH6	0.274	0.279	+17.2%
Kingston Bay - Rocky Nook	PDH7	0.282	0.289	+17.4%
Kingston Bay - Goose Point	PDH8	0.281	0.288	+17.4%
Jones River	PDH9	0.389	0.419	+20.9%
Plymouth Bay	PDH10	0.265	0.269	+16.6%
Duxbury Bay - Cowyard	PDH11	0.276	0.282	+16.0%
Duxbury Bay - Saquish Neck	PDH12	0.307	0.317	+16.0%
Duxbury Bay - mid	PDH13	0.342	0.358	+15.9%
Duxbury Bay - west	PDH14	0.347	0.364	+16.1%
Duxbury Marsh	PDH15	0.430	0.461	+16.5%

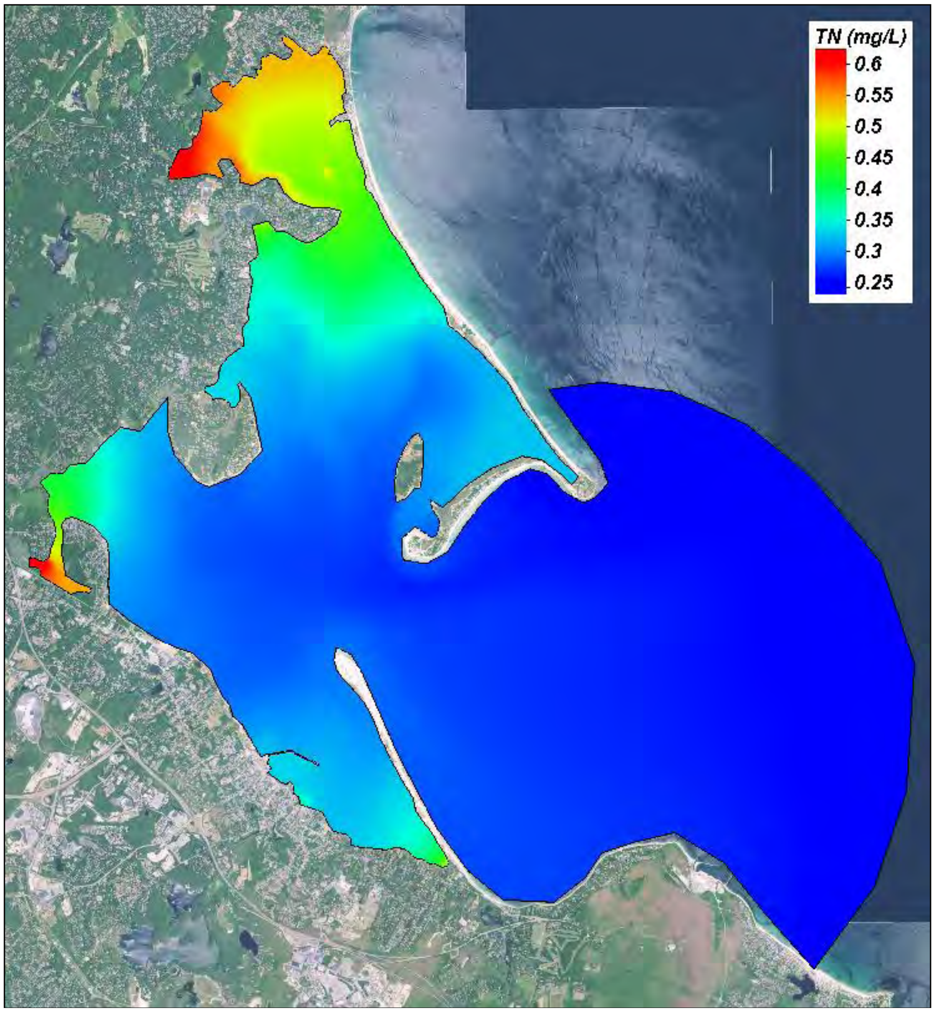


Figure VI-8. Contour plot of modeled total nitrogen concentrations (mg/L) in the Plymouth Bay system, for projected build-out scenario loading conditions.

VI.2.6.2 No Anthropogenic Load

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

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations at each monitoring station. Again, total nitrogen concentrations in the receiving waters (i.e., Cape Cod Bay) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was large, with all areas of the system experiencing reductions greater than 60%, compared to the background concentration of 0.241 mg/L in Cape Cod Bay (Table VI-8). The greatest drop occurs in the southern portion of Plymouth Harbor, near the Eel River inlet. A contour plot showing TN concentrations throughout the system is presented in Figure VI-9.

Table VI-7. "**No anthropogenic loading**" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of the Plymouth 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 net (kg/day)
Duxbury Marsh	2.351	5.589	26.431
Duxbury Bay	0.567	59.200	10.932
Kingston Bay	3.805	49.227	34.604
Plymouth Harbor	2.129	25.614	39.058
Blue Fish River	2.532	-	-
Jones River	26.044	-	-
Town Brook	6.323	-	-
Eel River	5.827	-	-
Plymouth WWTF Outfall	0.000	-	-
system total	49.578	139.630	111.025

Table VI-8. Comparison of model average TN concentrations from present loading and the "**No anthropogenic loading**" ("no load"), with percent change over background in Cape Cod Bay (0.241 mg/L), for the Plymouth Bay system.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	No load (mg/L)	% change
Plymouth Harbor - south	PDH1	0.325	0.254	-84.5%
Plymouth Harbor - boat basin	PDH2	0.315	0.256	-79.3%
Plymouth Harbor - mid	PDH3	0.296	0.256	-73.2%
Plymouth Harbor - north	PDH4	0.278	0.253	-68.1%
Plymouth Harbor - channel	PDH5	0.274	0.252	-67.7%
Kingston Bay - east	PDH6	0.274	0.252	-66.3%
Kingston Bay - Rocky Nook	PDH7	0.282	0.255	-66.8%
Kingston Bay - Goose Point	PDH8	0.281	0.255	-65.7%
Jones River	PDH9	0.389	0.279	-74.3%
Plymouth Bay	PDH10	0.265	0.249	-65.1%
Duxbury Bay - Cowyard	PDH11	0.276	0.254	-62.5%
Duxbury Bay - Saquish Neck	PDH12	0.307	0.266	-61.4%
Duxbury Bay - mid	PDH13	0.342	0.280	-61.0%
Duxbury Bay - west	PDH14	0.347	0.281	-62.4%
Duxbury Marsh	PDH15	0.430	0.313	-61.9%

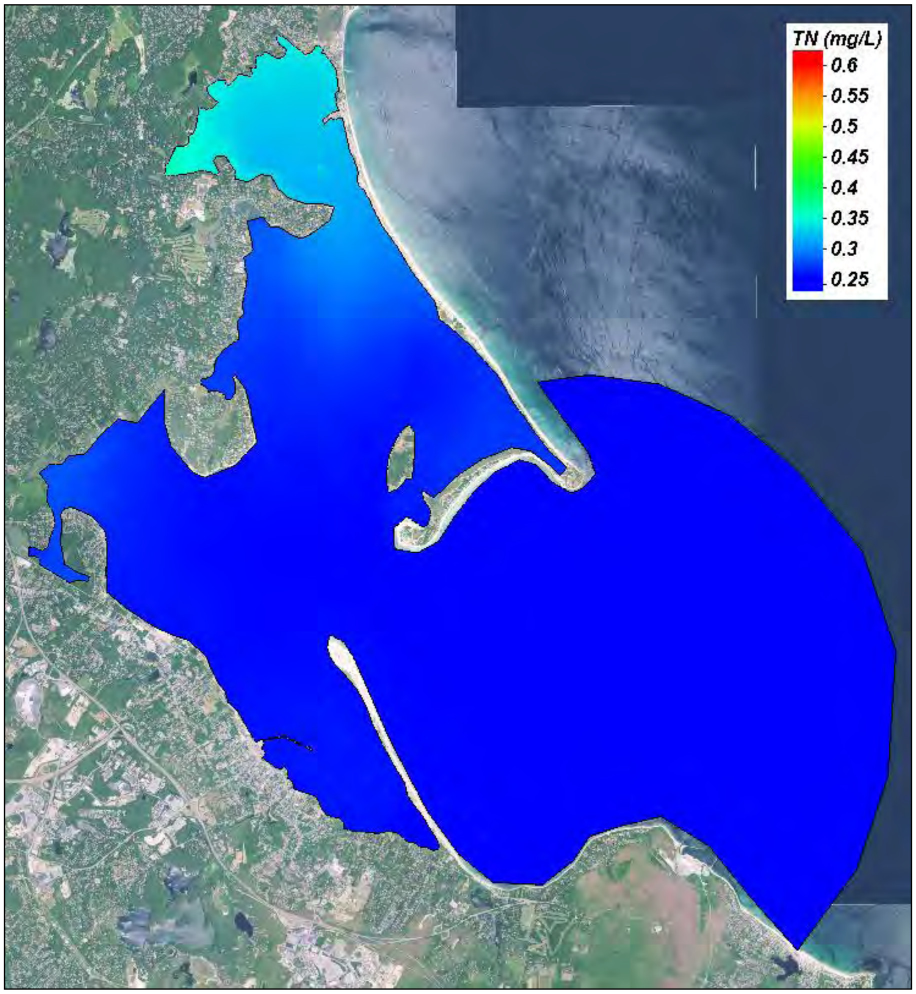


Figure VI-9. Contour plot of modeled total nitrogen concentrations (mg/L) in the Plymouth Bay system, for no anthropogenic loading conditions.

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 Plymouth-Kingston-Duxbury Embayment System, the MEP assessment is based upon data from the water quality monitoring program developed by the Town of Plymouth with technical assistance from SMAST2, as well as field survey and historical data collected under the programmatic umbrella of the Massachusetts Estuaries Project. These data include temporal surveys of eelgrass distribution (1951, 1995, 2001, 2006, 2012); surveys of benthic animal communities and sediment characteristics (2007, 2013); and summer time-series measurements of dissolved oxygen and chlorophyll-a (2007). 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 Massachusetts Estuaries Project (MEP) water quality modeling effort, becomes the basis of the nitrogen threshold development for this system (Section VIII). Part of the MEP assessment necessarily includes confirmation that the critical nutrient for management in any embayment is nitrogen and determination that a system is or is not impaired by nitrogen enrichment. Analysis of inorganic N/P molar ratios within the water column of the Plymouth-Kingston-Duxbury Embayment System support the contention that nitrogen is the nutrient to be managed, as the Redfield Ratio (inorganic N/P) ranges from 1-3, with a system-wide average of 1.8 within Plymouth Harbor, Kingston Bay and Duxbury Bay. Ratios <10 indicate that nitrogen additions will increase phytoplankton production, organic matter levels and turbidity within estuarine waters. Increased phytoplankton and organic matter levels increase oxygen consumption within the waters and sediments and increase the extent of oxygen depletion and habitat impairment. It should be noted that nitrogen enrichment occurs through two primary mechanisms, high rates of nitrogen entering from the surrounding watershed and/or low rates of flushing due to restriction of tidal exchange with low nitrogen offshore waters. Like most coastal watersheds and associated estuaries in southeastern Massachusetts, the Plymouth-Kingston-Duxbury Embayment System has seen increasing nitrogen loading from its watershed from shifting land-uses.

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

[2] Howes, B. and R. Samimy. 2005. Summary of Water Quality Monitoring Program for the Plymouth, Kingston, and Duxbury Harbor Embayment System. Completed for Town of Kingston, Town of Duxbury, Town of Plymouth, and MADEP 604b Program. Coastal Systems Laboratory, School for Marine Science and Technology, University of Massachusetts Dartmouth.

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 throughout the Plymouth-Kingston-Duxbury Embayment at nine (9) critical points in the system 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. Eelgrass loss in southeastern Massachusetts estuaries associated with nitrogen enrichment is generally through decreased light penetration resulting from increased phytoplankton biomass and resulting suspended organic particles, as well as shading by epiphytes (small plants that colonize eelgrass shoots) and sometimes by accumulations of drift macroalgae. Each of these factors is a result of nitrogen enrichment and all result in stress to eelgrass beds.

Changes in the distribution of eelgrass beds within the Plymouth-Kingston-Duxbury Embayment System was evaluated using coverage maps developed by the MassDEP Eelgrass Mapping Program (C. Costello) and relied on aerial photo analysis for the estimate of eelgrass presence in 1951, (with similar analysis in 1995, 2001, 2006 and 2012 that also included on-site verification) Additionally, the MEP Technical Team did conduct a general survey as part of the mooring program (2007) and sediment and infauna surveys in 2007 and 2013. It should be noted that MEP staff did observe eelgrass in the embayment as it was completing its field data collection tasks. Temporal trends in the distribution of eelgrass beds are typically used by the MEP to assess the stability of the habitat and to determine trends potentially related to nutrient enrichment and 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. This is consistent with results from the Water Quality Monitoring Program indicating that phytoplankton production and reduction in light penetration within the basins of the Plymouth-Kingston-Duxbury Estuary System can be enhanced by additional nitrogen loading. This is based upon inorganic nitrogen to phosphorus ratios, where basin averages throughout the embayment system range from 1-3. While this ratio approach (Redfield Ratio) is an approximation, where values <<1 are associated with nitrogen limitation, >>16 phosphorus limitation, the low value of the ratio provides additional site-specific evidence that nitrogen is the appropriate nutrient for management of potential eutrophication in this system.

While a temporal change in eelgrass distribution provides a basis for evaluating increases (nitrogen loading) or decreases (increased flushing) in nutrient enrichment within the Plymouth-Kingston-Duxbury Embayment System. In areas that have not historically supported and do not presently support eelgrass, 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 (Sanders, H.L. 1960, Sanders, H.L. *et.al.*, 1980, Tian, Y.Q., J.J. Wang, J. A. Duff, B.L. Howes and A. Evgenidou. 2009) and New Bedford (Howes, B.L. and C.T. Taylor, 1990), as well as the Woods Hole Oceanographic Institution Nantucket Harbor Study (Howes *et al.* 1997). These data are coupled with the level of diversity (H') and evenness (E) of the benthic community and the total number of individuals to determine the infaunal habitat quality.

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, USEPA[3] suggests that the chronic protective oxygen level to support growth of estuarine animals is 4.8 mg L^{-1}, with a limit for survival of juvenile and adult organisms of 2.3 mg L^{-1}. However, studies have demonstrated that slightly higher oxygen levels, 3.0 mg/L, can be lethal to larval fish and crustaceans (Poucher and Coiro 1997). Massachusetts State Water Quality Classification indicates that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1}. The tidally influenced waters of the Plymouth-Kingston-Duxbury Embayment System are currently listed under this Classification as SA. It should be noted that the classification system represents the water quality that the embayment should support, not the existing level of water quality. It is through the MEP and TMDL processes that management actions are developed and implemented to keep or bring the existing conditions in line with the classification.

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates of oxygen uptake 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 Plymouth-Kingston-Duxbury Embayment 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 typical 28 day deployment within the interval from August through mid-September. The mooring data (DO and CHLA) from the Plymouth-Kingston-Duxbury Embayment System was collected during the summer of 2007.

Similar to other embayments in southeastern Massachusetts, the Plymouth-Kingston-Duxbury Embayment System evaluated in this assessment showed high frequency variation in water-column oxygen and chlorophyll-*a* levels, related to the diurnal light cycle 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 most of the mooring sites, underscores the need for continuous monitoring within these systems.

Dissolved oxygen and chlorophyll-*a* records were evaluated both for temporal trends and to determine the percent of the 23 to 55 day deployment period that these parameters were

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

below/above various benchmark concentrations (Tables VII-1, VII-2). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions.

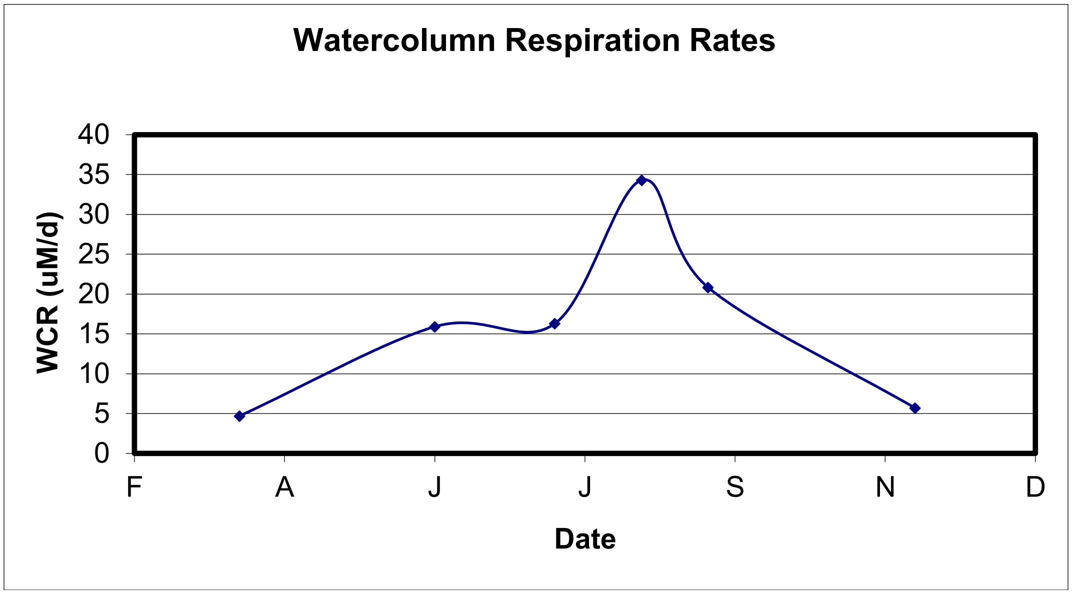


Figure VII-1. Average water column respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll-*a* levels indicate low to moderately nutrient enriched waters with higher nutrient related water quality in the central basin of the overall system that encompasses portions of Plymouth Harbor, Kingston Bay and Duxbury Bay. Greater levels of oxygen depletion and phytoplankton biomass were observed in the uppermost portions of Duxbury Bay, particularly at the Duxbury Marsh station (PDH-8) that is located in the salt marsh dominated area of Duxbury Bay above the Powder Point bridge crossing over to the barrier beach (Long Island). It should be noted that the Water Quality Monitoring Program observed similar levels of chlorophyll and bottom water oxygen depletion, although it did not always capture the minimum oxygen or maximum chlorophyll-*a* conditions at this site. The oxygen data is typical of salt marsh dominated basins which are naturally nutrient and organic matter rich and frequently show hypoxic conditions at night in pristine salt marsh basins. These nighttime depletions are clearly seen in the time-series (Figure VII-17). The measured levels of oxygen depletion are indicative of organic and nutrient rich estuarine systems and in embayments indicates impaired habitat quality. In contrast, salt marshes are naturally nutrient and organic matter enriched as part of their ecological design, which makes them such important nursery areas for adjacent offshore waters. However, a natural consequence of their organic rich sediments is periodic oxygen depletion within the tidal creeks, particularly during the summer. In addition, the elevated chlorophyll a levels are consistent with the observed nitrogen levels. While the levels of chlorophyll a are not sufficient to impair salt marsh habitats, they reflect natural nutrient levels combined with watershed inputs focused in the headwaters of Duxbury Bay.

Overall, the open water basins show levels of oxygen depletion consistent with a low to moderate levels of organic matter enrichment, primarily from phytoplankton biomass as seen in the parallel measurements of chlorophyll-*a*. The measured levels of oxygen depletion and chlorophyll-*a* levels are consistent with the observed nitrogen levels within the various basins and the parallel variation in these water quality parameters is generally consistent with watershed based nitrogen inputs being focused in the upper most portions of this estuarine system (e.g. upper Kingston Bay, upper Duxbury Bay down gradient of Powder Point Bridge and to a lesser extent Plymouth Harbor). In the uppermost reaches of Duxbury Bay, upgradient of the Powder Point Bridge, it is important to recognize that this area is dominated by salt marsh and observed oxygen depletions are characteristic of this ecosystem type.

The mooring data show that overall, both the inner and outer portions of the sub-embayments of the Plymouth-Kingston-Duxbury system can be characterized by oxygen and chlorophyll-a levels supportive of moderately impaired to healthy habitat. This is most likely due to the large tidal range and effective flushing of the system with low nutrient water from Cape Cod Bay. However, specific areas do receive significant watershed nitrogen loads relative to their volumes and turnover rates (particularly in the vicinity of large river discharge points like the Jones and Eel Rivers, Town Brook), have slightly elevated levels of chlorophyll-a and consequently some eelgrass loss.

Interpretation of estuarine oxygen records need to consider both the frequency of oxygen depletion and the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions. The use of only the duration of oxygen below, for example 4 mg L^{-1}, can underestimate the level of habitat impairment in a particular location. The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production, oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally equilibrium was ~7-8 mg L^{-1} at the mooring sites). This was considered in the interpretation of the dissolved oxygen and chlorophyll a records as well as the effect of the changing tide.

Figure VII-2. Aerial Photograph of the Plymouth-Kingston-Duxbury Embayment System in the Towns of Plymouth, Kingston and Duxbury showing locations of Dissolved Oxygen / CHLA mooring deployments conducted in the summer of 2007.

The pattern of low-moderate oxygen depletion, low-moderately elevated chlorophyll-*a* values in specific areas and low-moderate levels of nitrogen enrichment are consistent with the observed moderate loss of eelgrass (Section VII.3) and generally high quality infaunal habitats (Section VII.4) observed throughout the Plymouth-Kingston-Duxbury Embayment System. These assessments indicate an estuarine system that is approaching its ability to assimilate nitrogen loads without impairment. The embayment specific results are as follows:

Plymouth Harbor (PDH1) (Figures VII-3 and VII-4):

The inner Plymouth Harbor (PDH1) mooring was centrally located within the innermost portion of the Plymouth Harbor sub-embayment, down gradient of the Eel River discharge (Figure VII-2). Daily excursions (maximum to minimum) in oxygen levels at this location were small, generally varying only 2 mg L^{-1}. Oxygen levels varied primarily with the tide (semi-diurnal cycle) as this portion of the system is well flushed by high quality water from Cape Cod Bay and does not appear to have restricted tidal exchange (based on stage records, Section V). Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). These cycles also tended to correspond to period of low tide and high tide respectively. In addition, maximum oxygen levels rarely exceeded air equilibration (% air saturation), which occurs when nitrogen enrichment has stimulated phytoplankton production and oxygen release. Both the absence of high oxygen levels (>10 mg L^{-1}) and the small daily excursions suggest that significant organic matter enriched conditions were not extant in this region of the basin during the measurement period.

Oxygen levels were almost always above 6 mg L^{-1} (99% of record) and was always above 5 mg L^{-1} over the 24 day record (Figure VII-3), consistent with the Water Quality Monitoring Program which recorded DO levels >6 mg L^{-1} during all sampling events at this location. These values are comparable to the results from the long-term Water Quality Monitoring Program sampling at this location. Oxygen levels at this site in upper portion of Plymouth Harbor sub-embayment were always >4 mg L^{-1}, the critical threshold for oxygen stress in an estuarine system (Table VII-1). The infrequent oxygen declines were consistent with the low to moderate levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a,* averaged 6.7 ug L^{-1} over the record and only exceeded 10 ug L^{-1} 3% of the deployment period. The *chlorophyll-a* levels were slightly elevated at the beginning of the deployment period, but steadily declined showing only a slight increase potentially indicative of a small bloom towards the end of the 24 day measurement period. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments. Average chlorophyll a measurements from both of the moorings (PDH-1, PDH-2) in Plymouth Harbor (averaging 6.7 and 6.3 ug/L, respectively) and the long-term Water Quality Monitoring Program (water quality monitoring stations PDH1, PDH 2 and PDH 3 average chlorophyll concentrations = 5.1, 5.9 and 4.1 ug L^{-1}, respectively) show similarly low-moderate levels. These levels of chlorophyll-*a* are indicative of an open water basin with low to moderate nitrogen and organic matter enrichment (Table VII-2, Figure VII-4), which is resulting in only minor oxygen depletion.

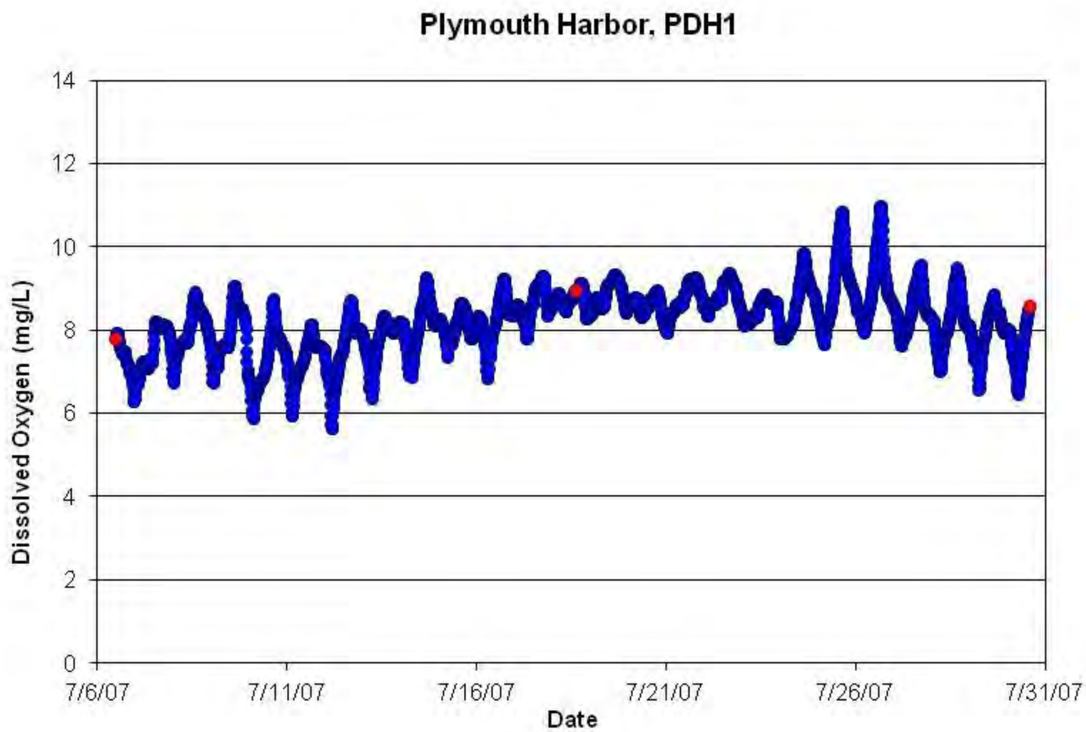


Figure VII-3. Bottom water record of dissolved oxygen at the Plymouth Harbor (PDH1) station, Summer 2007 (location in Figure VII-2). Calibration samples represented by red dots.

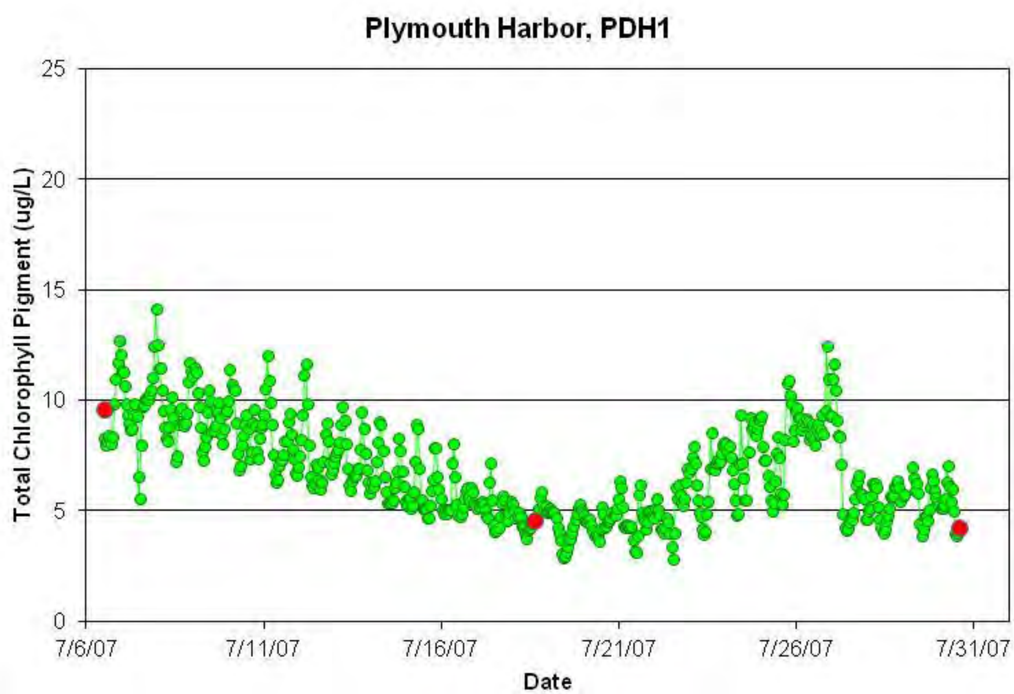


Figure VII-4. Bottom water record of Chlorophyll-*a* in the Plymouth Harbor (PDH1) station, Summer 2007. Calibration samples represented as red dots.

Plymouth Harbor (PDH2) (Figures VII-5 and VII-6):

The Plymouth Harbor (PDH2) mooring was located within the boat basin portion of the Plymouth Harbor sub-embayment and slightly north of the Town Brook discharge (Figure VII-2). Daily excursions (maximum to minimum) in oxygen levels at this location were small, generally varying only 1 to 2 mg L^{-1}. Oxygen levels varied primarily with the tide (semi-diurnal cycle) as this portion of the system is well flushed by high quality water from Cape Cod Bay and does not appear to have restricted tidal exchange (based on stage records, Section V). Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). These cycles also tended to correspond to period of low tide and high tide respectively. In addition, maximum oxygen levels did not exceed air equilibration (% air saturation), which occurs when nitrogen enrichment has stimulated phytoplankton production and oxygen release. Both the absence of high oxygen levels (>10 mg L^{-1}) and the small daily excursion suggest that significant organic matter enriched conditions were not extant in this region of the basin during the measurement period.

Oxygen levels were generally above 6 mg L^{-1} (86% of record) and always >5 mg L^{-1} over the 55 day record (Figure VII-5). These values are comparable to the results from the long-term Water Quality Monitoring Program sampling at this location which found oxygen to be >6 mg/L in all samplings over 5 years.. Oxygen levels at this site in the boat basin portion of Plymouth Harbor sub-embayment were always >4 mg L^{-1}, the critical threshold for oxygen stress in an estuarine system (Table VII-1). The infrequent small oxygen declines were consistent with the moderate to low levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 6.3 ug L^{-1} over the record and only exceeded 10 ug L^{-1} <1% of the deployment period and never reaching 15 ug L^{-1}. The *chlorophyll-a* levels were generally between 5 ug L^{-1} and 10 ug L^{-1} during the entire 55 day measurement period. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments. Average chlorophyll a measurements from both of the moorings (PDH-1, PDH-2) in Plymouth Harbor (averaging 6.7 and 6.3 ug/L, respectively) and the long-term Water Quality Monitoring Program (water quality monitoring station PDH1, PDH 2 and PDH 3 average chlorophyll concentrations = 5.1, 5.9 and 4.1 ug L^{-1}, respectively) show similarly low-moderate levels. These levels of chlorophyll-*a* are indicative of an open water basin with low to moderate nitrogen and organic matter enrichment (Table VII-2, Figure VII-6), which is resulting in only minor oxygen depletion.

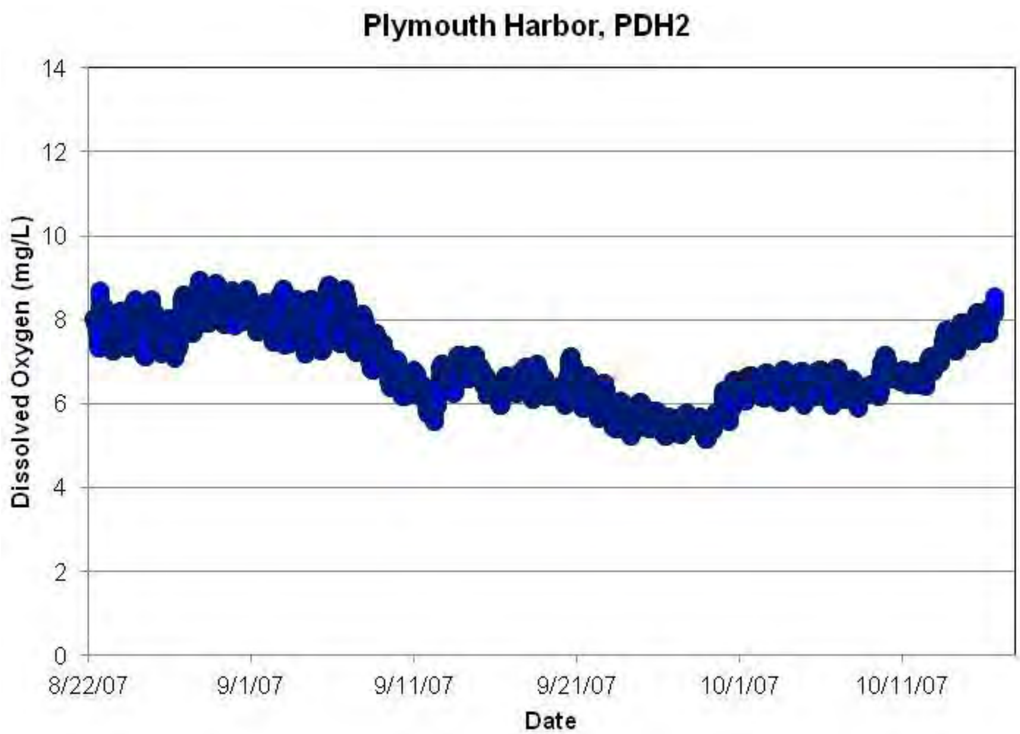


Figure VII-5. Bottom water record of dissolved oxygen recorded within the boat basin area of the Plymouth Harbor portion of the overall system, summer 2007 (location in Figure VII-2). Calibration samples represented as red dots.

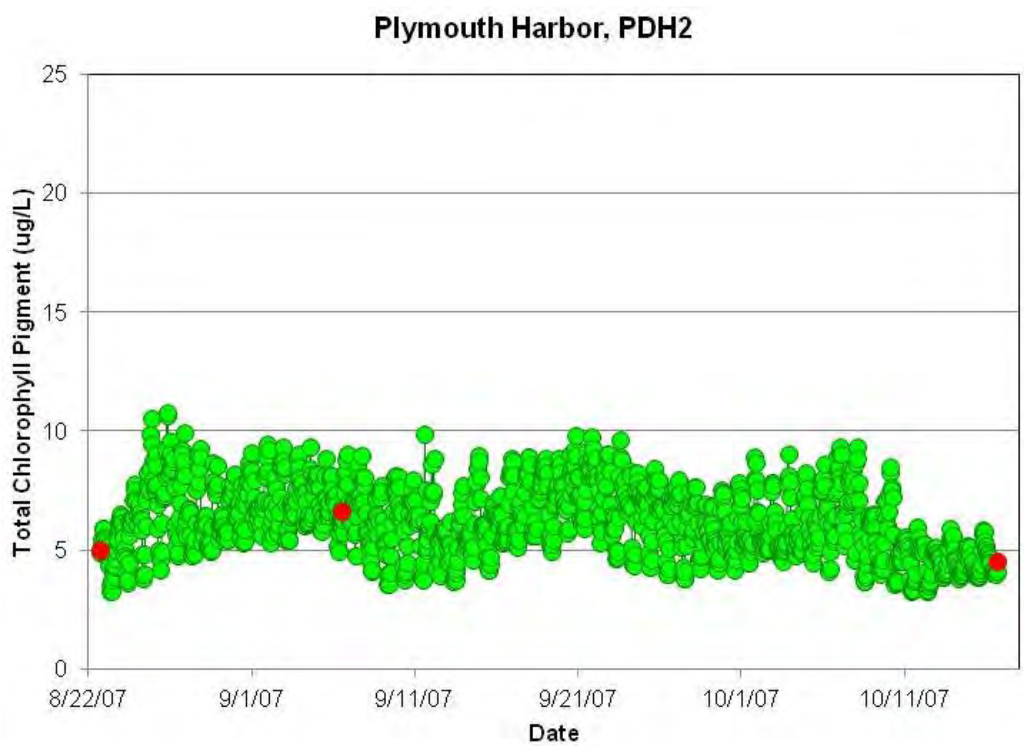


Figure VII-6. Bottom water record of Chlorophyll-*a* recorded within the boat basin area of the Plymouth Harbor portion of the overall system, summer 2007 (location in Figure VII-2). Calibration samples represented as red dots.

Outer Basin-Inlet (PDH3) (Figures VII-7 and VII-8)

The outer basin at the nexus between Duxbury Bay, Kingston Bay, Plymouth Harbor and the tidal inlet had an open water mooring (PDH3) within a deeper region that was centrally located adjacent the inlet (Figure VII-2). Daily excursions (maximum to minimum) in oxygen levels at this location were small, generally varying only 1 to 2 mg L^{-1}. Oxygen levels varied primarily with the tide (semi-diurnal cycle) as this portion of the system is well flushed by high quality water from Cape Cod Bay and does not appear to have restricted tidal exchange (based on stage records, Section V). Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). These cycles also tended to correspond to period of low tide and high tide respectively. In addition, maximum oxygen levels did not exceed air equilibration (% air saturation), which occurs when nitrogen enrichment has stimulated phytoplankton production and oxygen release. Both the absence of high oxygen levels (>10 mg L^{-1}) and the small daily excursion suggest that significant organic matter enriched conditions were not extant in this region of the basin during the measurement period.

Oxygen levels were always above 6 mg L^{-1} (100% of record) for the duration of the 24 day record (Figure VII-7). These values are comparable to the results from the long-term Water Quality Monitoring Program sampling at this location which also found oxygen to be consistently >6 mg L^{-1}. Oxygen levels at this site in the central basin of the outer embayment system were always >4 mg L^{-1}, the critical threshold for oxygen stress in an estuarine system (Table VII-1). It should also be noted that based on the long-term Water Quality Monitoring Program, the minimum oxygen levels recorded for this outer basin (PDH-6, PDH-11) were 7.8 and 7.2 mg L^{-1}, respectively, over the 5 years of sampling. The high oxygen levels were consistent with the low levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 4.2 ug L^{-1} over the time-series record, was <5 ug L^{-1} for 77% of the record and only exceeded 10 ug L^{-1} 1% of the deployment period, never reaching 15 ug L^{-1}. The *chlorophyll-a* levels were slightly elevated at the beginning of the deployment period, but steadily declined remaining consistently below 5 ug L^{-1} for the latter half of the 24 day measurement period. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments. Average chlorophyll a measurements from the mooring (PDH-3) in the central basin – inlet site (averaging 4,2 ug/L) and the long-term Water Quality Monitoring Program (water quality monitoring stations PDH6 and PDH 11 average chlorophyll concentrations = 3.0 and 4.0 ug L^{-1}, respectively over 5 years) show similarly low-moderate levels. These levels of chlorophyll-*a* are indicative of an open water basin with low to moderate nitrogen and organic matter enrichment (Table VII-2, Figure VII-8), which is resulting in only minor oxygen depletion.

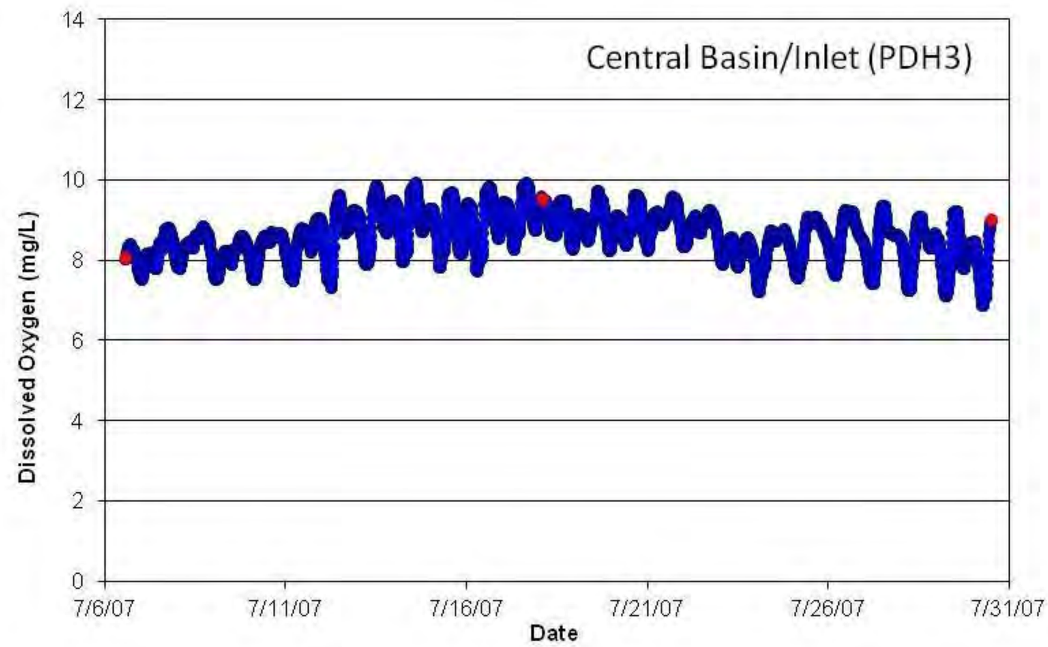


Figure VII-7. Bottom water record of dissolved oxygen within the open water open water central basin adjacent the inlet comprising the outer basin of the overall estuarine system, summer 2007 (Figure VII-2). Calibration samples shown as red dots.

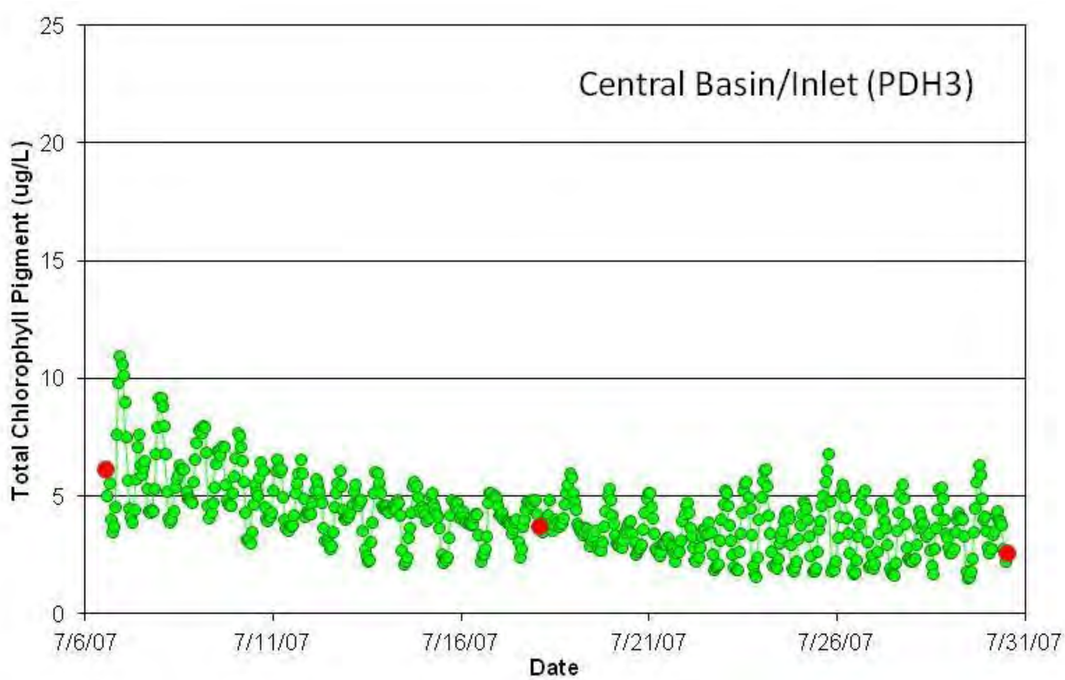


Figure VII-8. Bottom water record of Chlorophyll-*a* recorded within the open water central basin adjacent the inlet comprising the outer basin of the overall estuarine system, summer 2007 (location in Figure VII-2). Calibration samples represented as red dots.

Kingston Bay (PDH4) (Figures VII-9 and VII-10)

The Kingston Bay (PDH4) mooring was located nearshore and in open water in the border area between Kingston Bay and Plymouth Harbor south of where the Jones River discharges (Figure VII-2). Daily excursions (maximum to minimum) in oxygen levels at this location were moderate, generally varying between 2 and 4 mg L^{-1}. Oxygen levels varied primarily with the tide (semi-diurnal cycle) as this portion of the system is well flushed by high quality water from Cape Cod Bay and does not appear to have restricted tidal exchange (based on stage records, Section V). Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). These cycles also tended to correspond to the period of low tide and high tide respectively. In addition, maximum oxygen levels only rarely exceeded air equilibration (% air saturation), which occurs when nitrogen enrichment has stimulated phytoplankton production and oxygen release. Both the presence of high oxygen levels (>10 mg L^{-1}) and the moderate daily excursion suggest that more significant organic matter enriched conditions are extant in this region of the basin during the measurement period compared to other mooring locations. This location is likely being slightly effected by the discharge of Town Brook transporting its watershed nitrogen load..

Oxygen levels were virtually always above 6 mg L^{-1} (>99% of record) and always >5.9 mg L^{-1} over the 24 day record (Figure VII-9). These values are comparable to the results from the long-term Water Quality Monitoring Program sampling at this location (station PDH-4) which had a minimum oxygen level of 6.6 mg L^{-1} over 5 years of monitoring. Oxygen levels at this site in upper portion of Plymouth Harbor sub-embayment were always >4 mg L^{-1}, the critical threshold for oxygen stress in an estuarine system (Table VII-1). The high oxygen levels were consistent with the generally low levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 3.2 ug L^{-1} over the time-series record and never exceeded 10 ug L^{-1}. The *chlorophyll-a* levels were slightly elevated (above 5.0 ug L^{-1}) at the beginning of the deployment period, but quickly declined to below 5.0 ug L^{-1} early in the deployment and remained at that level for the remainder of the 24 day measurement period. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments. Average chlorophyll a measurements from the mooring (PDH-4) at the inland border between Kingston Bay and Plymouth Harbor, adjacent the mouth of Town Brook (averaging 3,2 ug/L) and the long-term Water Quality Monitoring Program (water quality monitoring station PDH-4 average chlorophyll concentration= 4.1 ug L^{-1}, respectively over 5 years) show similarly low-moderate levels. These levels of chlorophyll-*a* are indicative of an open water basin with low to moderate nitrogen and organic matter enrichment (Table VII-2, Figure VII-10), which is resulting in high quality estuarine habitat.

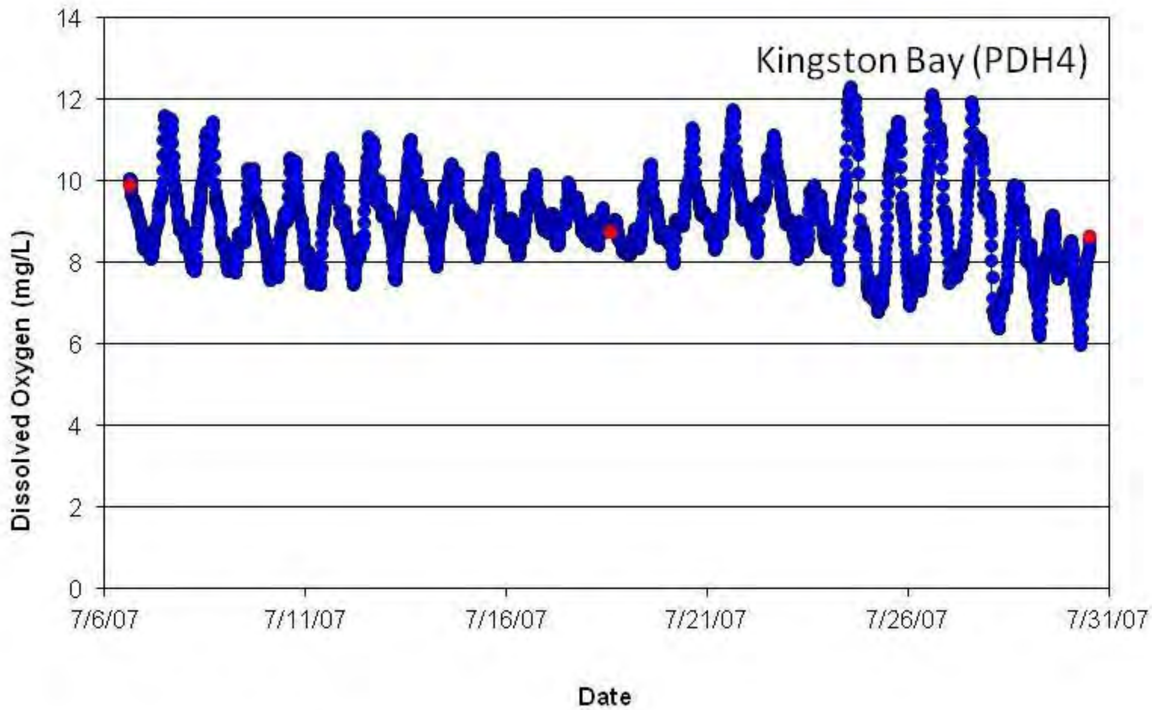


Figure VII-9. Bottom water record of dissolved oxygen within the nearshore open water innermost area of the Kingston Bay portion of the overall estuarine system, summer 2007 (Figure VII-2). Calibration samples shown as red dots.

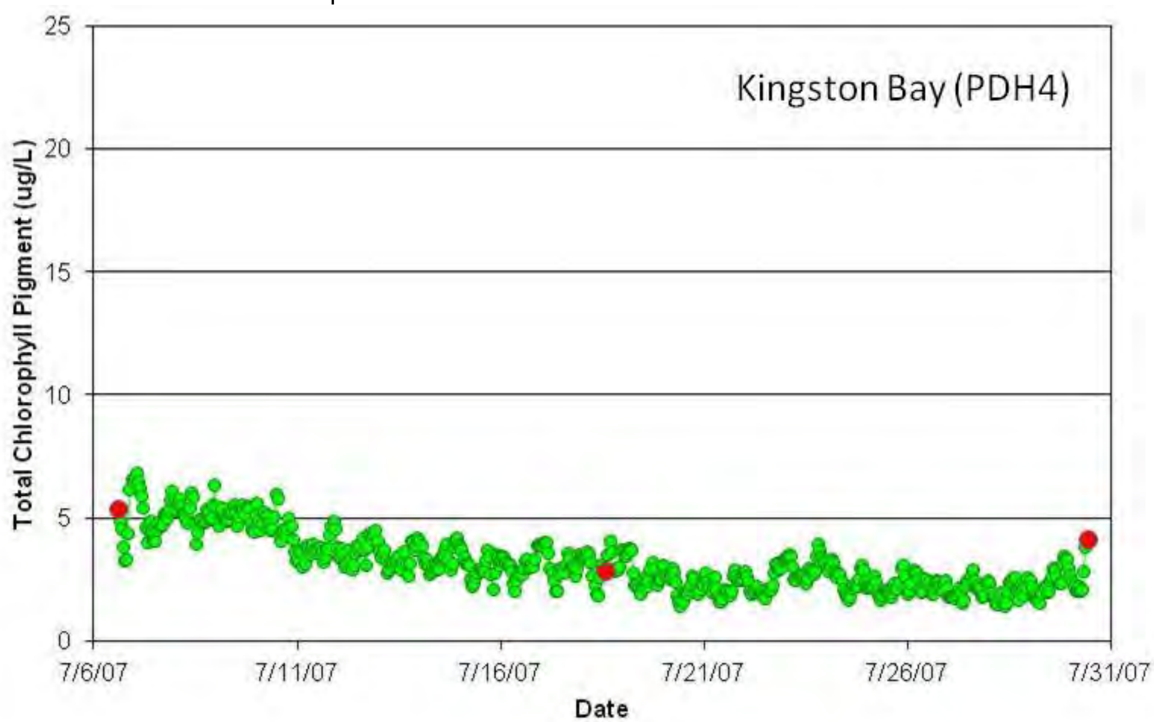


Figure VII-10. Bottom water record of Chlorophyll-*a* within the nearshore open water innermost area of the Kingston Bay portion of the overall estuarine system, summer 2007 (location in Figure VII-2). Calibration samples shown as red dots.

Kingston Bay (PDH5) (Figures VII-11 and VII-12)

The Kingston Bay (PDH5) mooring was located within the central basin in open water, down gradient Jones River mouth (Figure VII-2). Daily excursions (maximum to minimum) in oxygen levels at this location were slight, generally varying only 2 mg L^{-1}. Oxygen levels varied primarily with the tide (semi-diurnal cycle) as this portion of the system is well flushed by high quality water from Cape Cod Bay and does not appear to have restricted tidal exchange (based on stage records, Section V). Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). These cycles also tended to correspond to period of low tide and high tide respectively. In addition, maximum oxygen levels did not exceed air equilibration (% air saturation), which occurs when nitrogen enrichment has stimulated phytoplankton production and oxygen release. Both the absence of high oxygen levels (>10 mg L^{-1}) and the small daily excursions suggest that significant organic matter enriched conditions were not extant in this region of the basin during the measurement period.

Oxygen levels were almost always above 6 mg L^{-1} (99% of record) and did not decline below 5.8 mg L^{-1} over the 24 day record (Figure VII-11). These values are comparable to the results from the long-term Water Quality Monitoring Program sampling in Kingston Bay (stations PDH-7,8,9) which measured oxygen at >6 mg L^{-1} over each of the sampling events over 5 years. Oxygen levels at this site in upper portion of the Kingston Bay sub-embayment were always >4 mg L^{-1}, the critical threshold for oxygen stress in an estuarine system (Table VII-1). The infrequent oxygen declines were consistent with the moderate to low levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 5.2 ug L^{-1} over the time-series record and only exceeded 10 ug L^{-1} 4% of the deployment period, never reaching 15 ug L^{-1}. The *chlorophyll-a* levels were slightly elevated at the beginning of the deployment period, but steadily declined remaining consistently below 5 ug L^{-1} for the latter half of the 24 day measurement period. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments. Average chlorophyll a measurements from the mooring (PDH-5) in central Kingston Bay (averaging 5,2 ug/L) and the long-term Water Quality Monitoring Program (water quality monitoring stations PDH7, PDH 8 and PDH 9 average chlorophyll concentrations = 5.1, 3.6 and 7.1 ug L^{-1}, respectively) show similar low-moderate levels. It should be noted that the long-term water quality station PDH-9 is at the innermost portion of Kingston Bay and shows the effects of nitrogen inputs from the Jones River. However, the low chlorophyll a levels at mid basin suggest that the impacts are spatially limited. The mooring average of 5.2 ug/L within the central basin was similar to the basin average of the water quality stations, PDH-7,8,9 which averaged 5.3 ug/L over 5 years of samplings. These levels of chlorophyll-*a* are indicative of an open water basin with low to moderate nitrogen and organic matter enrichment (Table VII-2, Figure VII-12), which is resulting in only minor oxygen depletion.

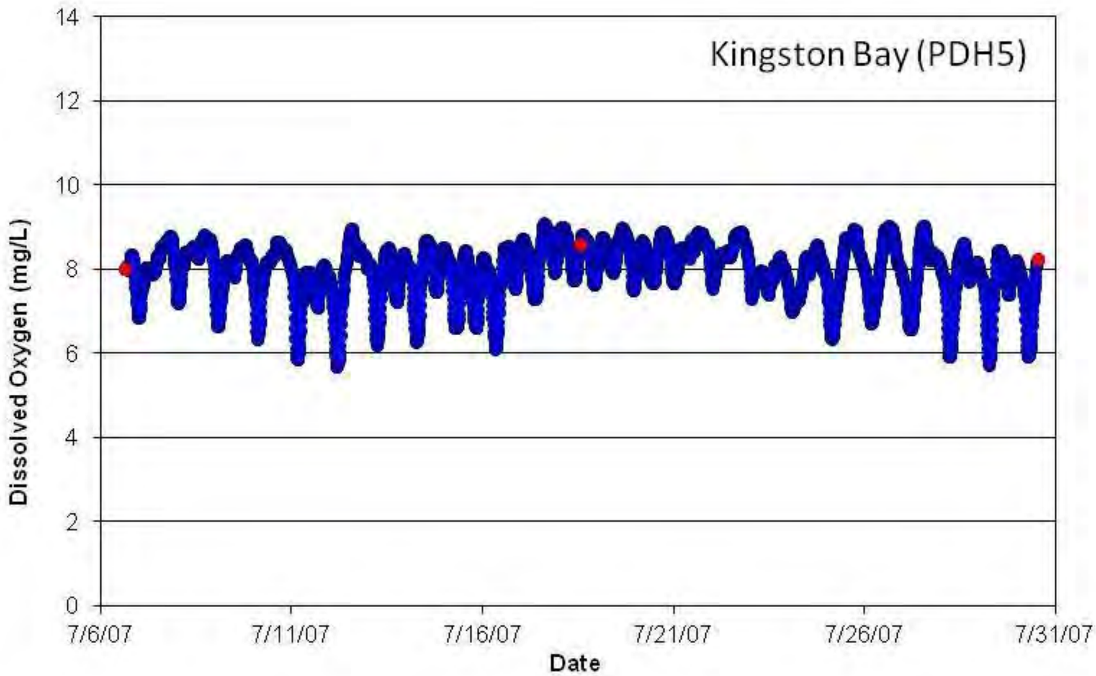


Figure VII-11. Bottom water record of dissolved oxygen within the nearshore open water innermost area (Jones River mouth) of the Kingston Bay portion of the overall estuarine system, summer 2007 (Figure VII-2). Calibration samples shown as red dots.

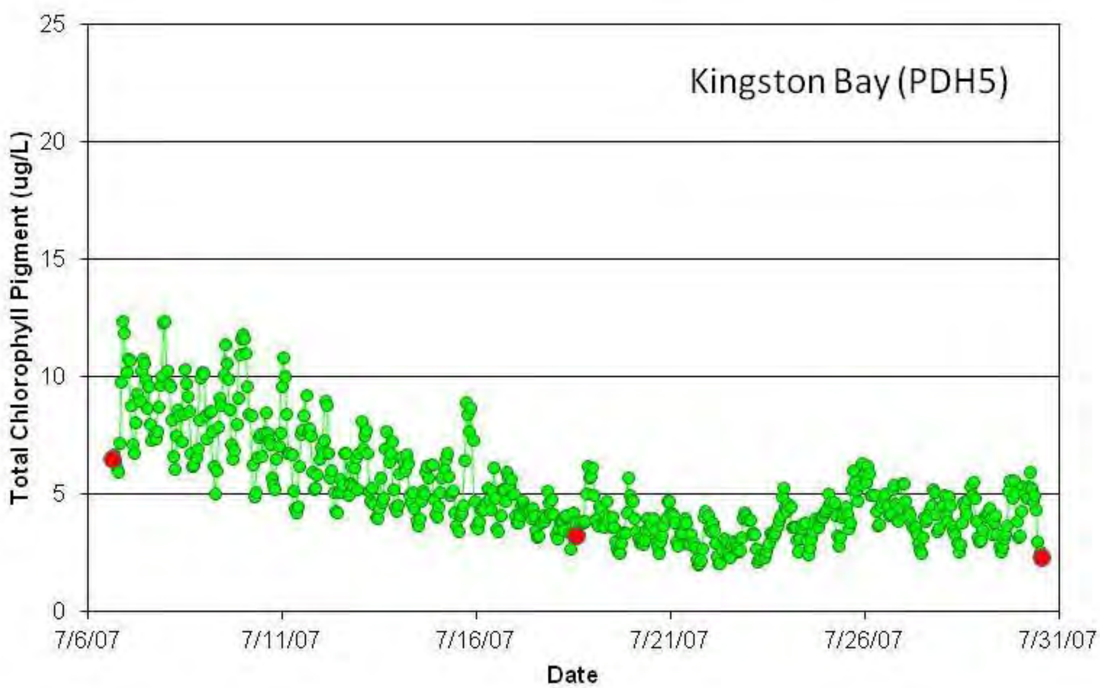


Figure VII-12. Bottom water record of Chlorophyll-*a* within the nearshore open water innermost area (Jones River mouth) of the Kingston Bay portion of the overall estuarine system, summer 2007 (location in Figure VII-2). Calibration samples shown as red dots.

Duxbury Bay (PDH6) (Figures VII-13 and VII-14):

The Duxbury Bay (PDH6) mooring was centrally located within the open water lower basin of the Duxbury Bay portion of the overall estuarine system (Figure VII-2). Daily excursions (maximum to minimum) in oxygen levels at this location were small, generally varying only 1 to 2 mg L^{-1}. Oxygen levels varied primarily with the tide (semi-diurnal cycle) as this portion of the system is near the tidal inlet and is well flushed by high quality water from Cape Cod Bay and does not appear to have restricted tidal exchange (based on stage records, Section V). Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). These cycles also tended to correspond to low tide and high tide respectively. In addition, maximum oxygen levels did not exceed air equilibration (% air saturation), which occurs when nitrogen enrichment has stimulated phytoplankton production and oxygen release. Both the general absence of high oxygen levels (>10 mg L^{-1}) and the small daily excursion suggest that significant organic matter enriched conditions were not extant in this lower region of the basin during the measurement period.

Oxygen levels were above 6 mg L^{-1} (100% of record) and never declined to between 5 and 6 mg L^{-1} during the 24 day record (Figure VII-13). These values are comparable to the results from the long-term Water Quality Monitoring Program sampling at this location (station PDH-11) which had a minimum oxygen level of 7.8 mg L^{-1} over 5 years of monitoring. Oxygen levels at this site in the lower portion of the Duxbury Bay sub-embayment were always >4 mg L^{-1}, the critical threshold for oxygen stress in an estuarine system (Table VII-1). The high oxygen levels were consistent with the generally low levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 4.4 ug L^{-1} over the time-series record and never exceeded 10 ug L^{-1} during the deployment period. The *chlorophyll-a* levels were slightly elevated (above 5.0 ug L^{-1}) at the beginning of the deployment period, but quickly declined to below 5.0 ug L^{-1} early in the deployment and remained at that level for the remainder of the 24 day measurement period. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments. Average chlorophyll a measurements from the mooring (PDH-6) within the lower portion of Duxbury Bay (averaging 4.4 ug/L) and the long-term Water Quality Monitoring Program (water quality monitoring station PDH-11 average chlorophyll concentration= 4.0 ug L^{-1}, respectively over 5 years) show similarly low-moderate levels. These levels of chlorophyll-*a* are indicative of an open water basin with low to moderate nitrogen and organic matter enrichment (Table VII-2, Figure VII-14), which is resulting in only minor oxygen depletion.

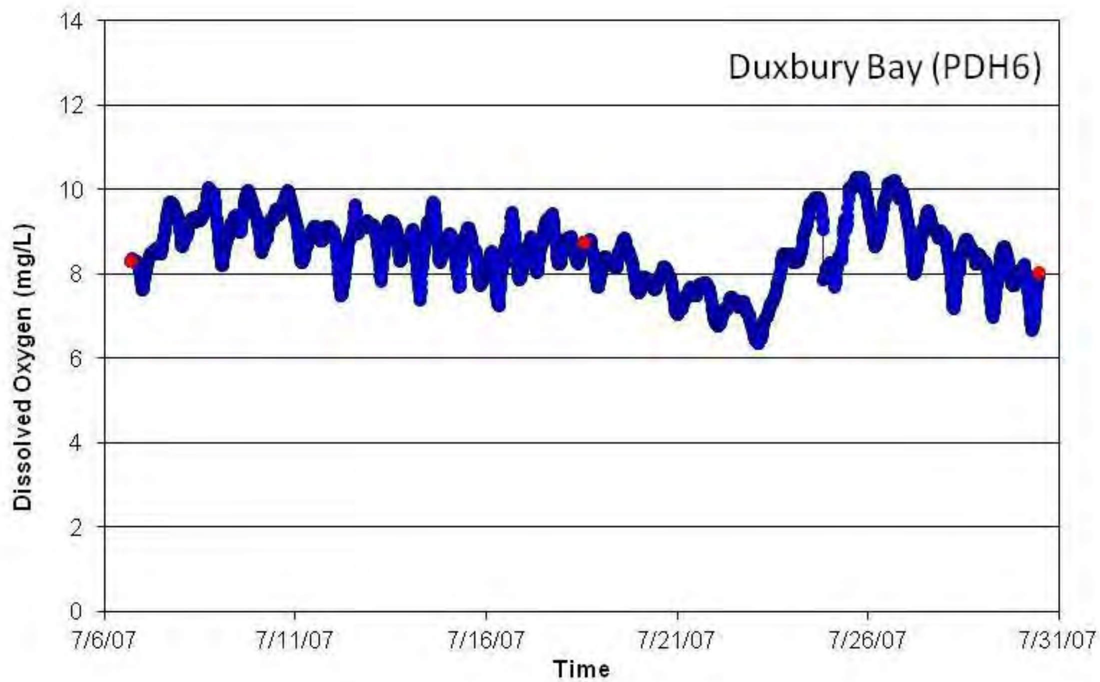


Figure VII-13. Bottom water record of dissolved oxygen within the open water outermost area of the Duxbury Bay portion of the overall estuarine system, summer 2007 (Figure VII-2). Calibration samples shown as red dots.

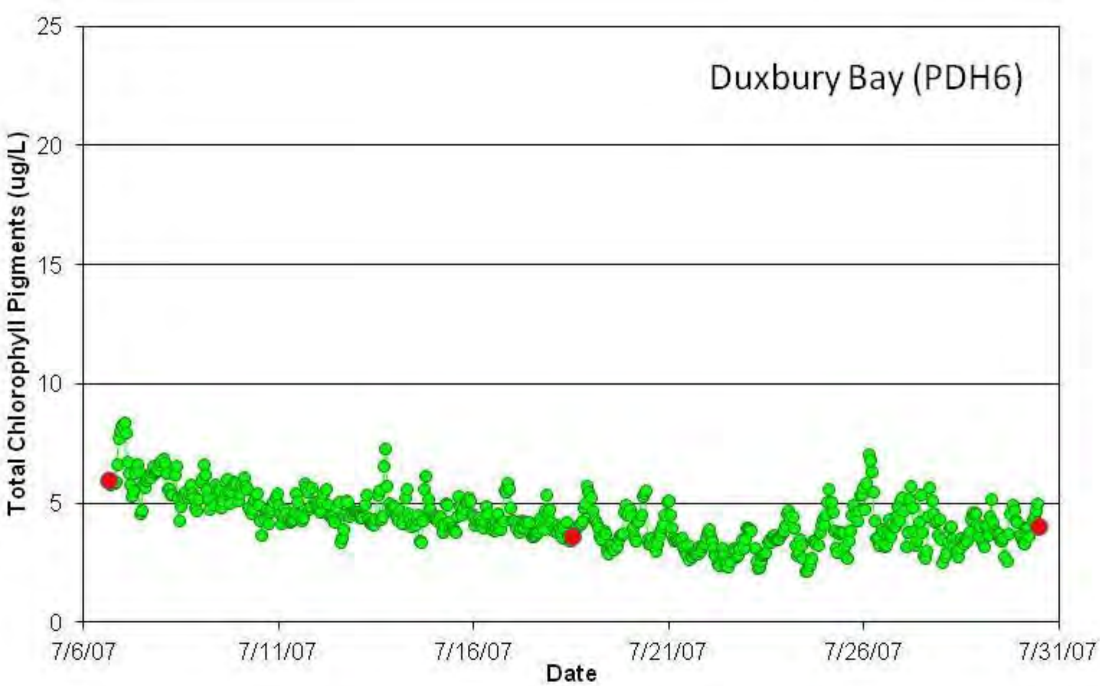


Figure VII-14. Bottom water record of Chlorophyll-*a* within the open water outermost area of the Duxbury Bay portion of the overall estuarine system, summer 2007 (location in Figure VII-2). Calibration samples shown as red dots.

Upper Duxbury Bay (PDH7) (Figures VII-15 and VII-16)

The upper Duxbury Bay (PDH7) mooring was located within the nearshore upper area (but below Powder Point Bridge) of the Duxbury Bay portion of the overall estuarine system (Figure VII-2). Daily excursions (maximum to minimum) in oxygen levels at this location were small <2 mg L^{-1}. Oxygen levels varied primarily with the tide (semi-diurnal cycle) as this portion of the system is less well flushed than the lower bay but does not appear to have restricted tidal exchange (based on stage records, Section V). Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). These cycles also tended to correspond to low tide and high tide respectively. In addition, maximum oxygen levels did not exceed air equilibration (% air saturation), which occurs when nitrogen enrichment has stimulated phytoplankton production and oxygen release. Both the absence of high oxygen levels (>10 mg L^{-1}), the small daily excursion and the high overall oxygen concentrations suggest that significant organic matter enriched conditions were not extant in this region of the basin during the measurement period.

Oxygen levels were above 6 mg L^{-1} (100% of record) and never declined to between 5 and 6 mg L^{-1} during the 24 day record (Figure VII-15). These results are comparable to the long-term Water Quality Monitoring Program sampling at this location (station PDH-13) which had a minimum oxygen level of 6.7 mg L^{-1} over 5 years of monitoring and similarly the nearby middle of the upper basin of Duxbury Bay station (PDH-14), minimum of 6.8 mg L^{-1}. Oxygen levels at this site in upper portion of the Duxbury Bay sub-embayment were always >4 mg L^{-1}, the critical threshold for oxygen stress in an estuarine system (Table VII-1). The high oxygen levels were consistent with the moderate levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 7.8 ug L^{-1} over the time-series record and exceeded 15 ug L^{-1} for 5% of the deployment period. The *chlorophyll-a* levels were elevated at the beginning of the deployment period, but steadily declined showing a second increase potentially indicative of a small bloom towards the tail end of the 24 day measurement period. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments. Average chlorophyll a measurements from the mooring (PDH-7) within the upper portion of Duxbury Bay basin (averaging 7.8 ug/L with maxima >20 ug L^{-1}) and the long-term Water Quality Monitoring Program (water quality monitoring stations PDH-13 and PDH-14 average chlorophyll concentrations= 4.6 and 5.0 ug L^{-1}, respectively over 5 years) indicate a basin with moderate phytoplankton levels. These levels of chlorophyll-*a* are indicative of an open water basin with low to moderate nitrogen and organic matter enrichment (Table VII-2, Figure VII-16), which is resulting in generally high oxygen levels.

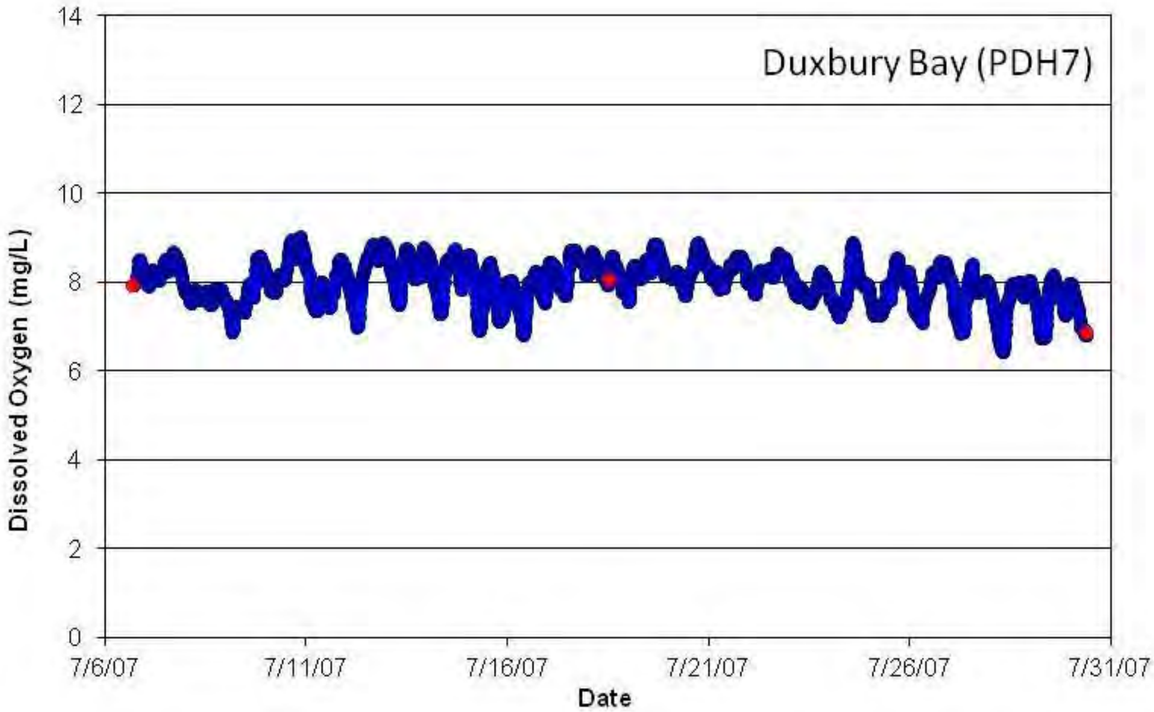


Figure VII-15. Bottom water record of dissolved oxygen within the nearshore upper area of the Duxbury Bay portion of the overall estuarine system, summer 2007 (Figure VII-2). Calibration samples shown as red dots.

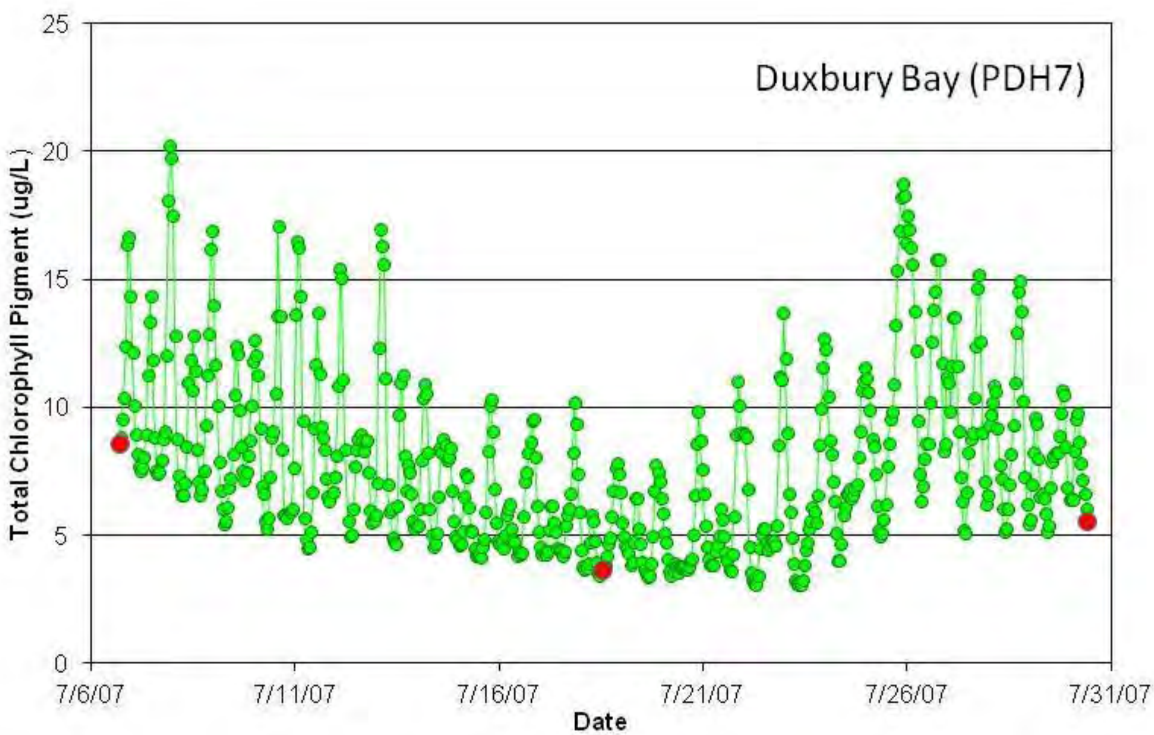


Figure VII-16. Bottom water record of Chlorophyll-*a* within the nearshore upper area of the Duxbury Bay portion of the overall estuarine system, summer 2007 (location in Figure VII-2). Calibration samples shown as red dots.

Duxbury Marsh (PDH8) (Figures VII-17 and VII-18)

The Duxbury Marsh (PD8) mooring was located within the salt marsh basin (above Powder Point Bridge) the uppermost sub-basin of the Duxbury Bay portion of the overall estuarine system (Figure VII-2). This area is dominated by a large salt marsh, which naturally supports a dynamic oxygen balance compared to that of high quality open water estuarine basins. The upper reaches of the Duxbury Marsh has deeply incised narrow creeks surrounded by large mud flats and emergent marsh vegetated with typical New England high and low marsh plants. The lower reach (still upgradient of Powder Point Bridge) has broader creeks with sediments formed from marsh deposits and primarily soft organic mud. The lower reach is transitional from the highly organic sediments of the upper marsh to the sandier near shore sediments of Duxbury Bay. The tide range in adjacent Cape Cod Bay is large, ~10 ft and the salt marsh areas are regularly flooded at high tide and the salt marsh creeks drain nearly completely with each ebb tide.

The large diurnal shifts in dissolved oxygen reflected at the Duxbury Marsh site are consistent with the high productivity within the marsh, high levels of oxygen uptake by the organic matter rich marsh sediments and tidal changes in salinity and temperature which influence oxygen solubility (e.g. incoming tides transport oxygen rich waters). The moderate chlorophyll a concentrations reflect the near complete exchange of tidal waters on each tide in the creeks which prevents the build-up of high chlorophyll levels. The absence of elevated (above air equilibration) oxygen levels in day time, which is typically found in nitrogen enriched embayments (due to stimulation of phytoplankton), supports the concept that the twice-a-day flushing of this tributary basin and the high nighttime oxygen uptake are the primary controls on oxygen dynamics at this site. In fact, the daily average dissolved oxygen concentration varied inversely with the tidal amplitude suggesting that longer residence time and greater areal submergence of the marsh was responsible for the lowest oxygen observed oxygen levels. Further evidence for the dominance of marsh processes is the lack of linkage between the observed variations in chlorophyll and the extent of oxygen depletion. In embayments, oxygen minima are typically observed as a bloom declines (senesces), a pattern not seen at this site.

Oxygen levels were typical of temperate salt marsh basins, generally above 6 mg L^{-1} (64% of record) but frequently declining to less than 4 mg L^{-1} for 7% of the 24 day record (Figure VII-17). These values are comparable to long-term Water Quality Monitoring Program sampling at this location (PDH-15). Oxygen levels at this site in the upper portion of the Duxbury Bay sub-embayment were mostly >4 mg L^{-1}, the critical threshold for oxygen stress in an estuarine system but not salt marshes (Table VII-1). The oxygen declines are consistent with this tributary basin's function as a salt marsh dominated estuarine basin and the moderate levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 7.1 ug L^{-1} over the record and only exceeded 10 ug L^{-1} for 15% of the deployment period, and exceeding 15 ug L^{-1} only 2% of the deployment period. The *chlorophyll-a* levels were generally low (<5 ug L^{-1}) at the beginning of the deployment period, but steadily increased (over 10 ug L^{-1}) showing indication of a small bloom towards the latter part of the 24 day measurement period. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments. Average chlorophyll a measurements from the mooring (PDH-8) within the upper portion of Duxbury Bay basin (averaging 7.1 ug/L with maxima >15 ug L^{-1}) and the long-term Water Quality Monitoring Program (water quality monitoring station PDH-15, average chlorophyll concentrations= 7.6 ug L^{-1}, over 5 years) indicate a basin with moderate but in this basin, not harmful, phytoplankton levels. These levels of oxygen and chlorophyll-*a* are representative of a salt marsh and less so a nutrient enriched open water basin (Table VII-2, Figure VII-18).

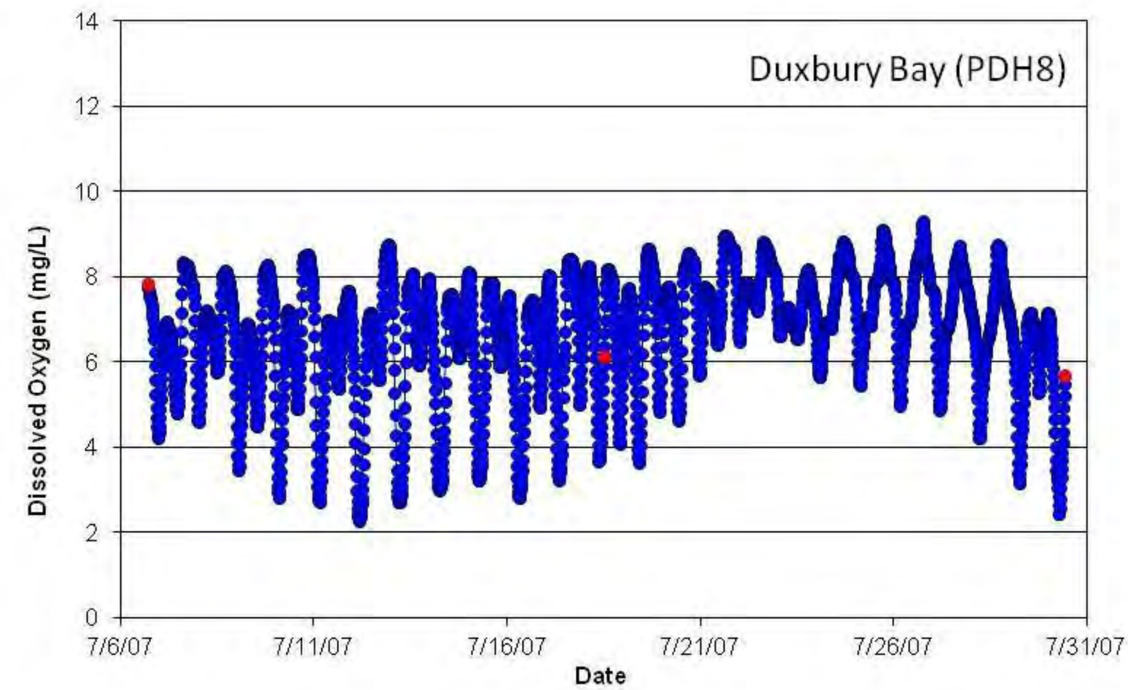


Figure VII-17. Bottom water record of dissolved oxygen within the nearshore upper area of the Duxbury Bay portion of the overall estuarine system, summer 2007 (Figure VII-2). Calibration samples shown as red dots.

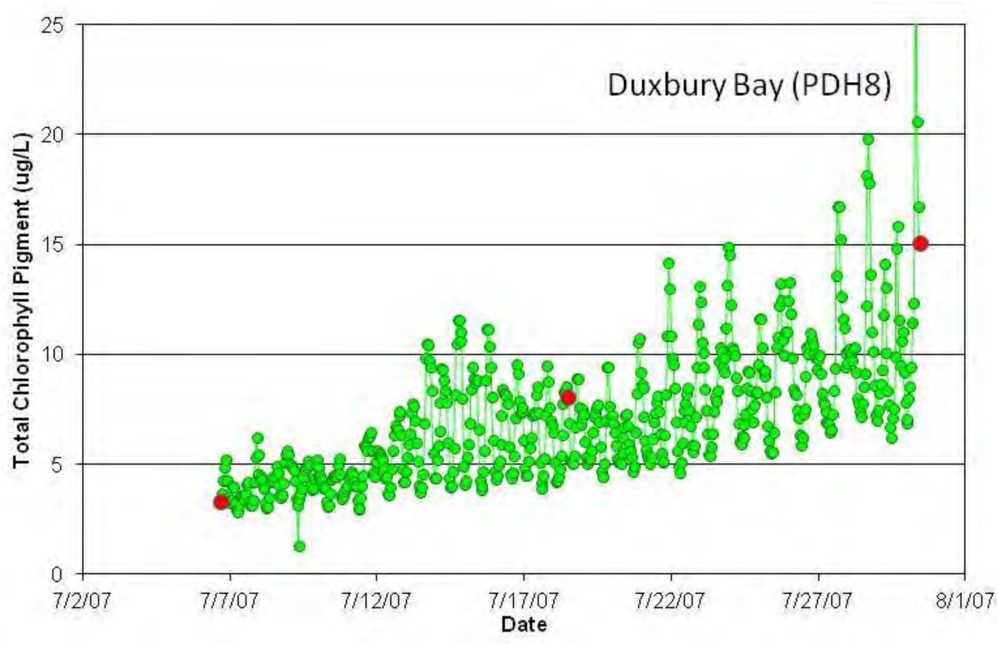


Figure VII-18. Bottom water record of Chlorophyll-*a* within the nearshore upper area of the Duxbury Bay portion of the overall estuarine system, summer 2007 (Figure VII-2). Calibration samples shown as red dots.

Boundary of Duxbury Harbor and the Central Basin-Inlet (PDH9) (Figures VII-19 and VII-20)

The boundary station (PDH9) mooring was located within the lower area of the Duxbury Bay portion of the overall estuarine system (Figure VII-2). This station is bounded by the barrier spit and Clarks Island. This site was evaluated due to its potentially semi-isolated nature, rather than as an indicator of the quality of the greater system. Daily excursions (maximum to minimum) in oxygen levels at this location were low to moderate, generally varying 2 to 4 mg L^{-1}. Oxygen levels varied primarily with the diurnal cycle and less with the tide (semi-diurnal cycle) as this portion of the system is generally well flushed by high quality water from Cape Cod Bay and does to appear to have restricted tidal exchange (based on stage records, Section V). Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). In addition, maximum oxygen levels periodically exceeded air equilibration (% air saturation), which occurs when nitrogen enrichment has stimulated phytoplankton production and oxygen release. Both the periodic high oxygen levels (>10 mg L^{-1}) and the low-moderate daily excursion suggest that a moderate level of organic matter enrichment is extant in this region of the basin during the measurement period, but that the tidal flushing was sufficient to prevent impairments.

Oxygen levels were generally above 6 mg L^{-1} (99% of record) and rarely declined to <6 mg L^{-1} (<1% of record) and remained >5.8 mg L^{-1} throughout the 24 day record (Figure VII-19). These values are comparable to the results from the long-term Water Quality Monitoring Program sampling at this location (station PDH-12) which had a minimum oxygen level of 6.6 mg L^{-1} over 5 years of monitoring. Oxygen levels at this site were always >4 mg L^{-1}, the critical threshold for oxygen stress in an estuarine system (Table VII-1). The high oxygen levels were consistent with the low levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 4.1 ug L^{-1} over the time-series record and did not reach 10 ug L^{-1} over the deployment period. The infrequent oxygen declines were consistent with the moderate to low levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 4.1 ug L^{-1} over the record and never exceeded 10 ug L^{-1} during the deployment period. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments. Average chlorophyll a measurements from the mooring (PDH-8) within the upper portion of Duxbury Bay basin (averaged 4.1 ug/L with maxima >10 ug L^{-1}) and the long-term Water Quality Monitoring Program (water quality monitoring station PDH-12, average chlorophyll concentrations= 4.0 ug L^{-1}, over 5 years) indicate a basin with relatively low phytoplankton levels. These levels of chlorophyll-*a* are indicative of a well flushed basin with low to moderate nitrogen and organic matter enrichment (Table VII-2, Figure VII-20), which is resulting in only minor oxygen depletion.

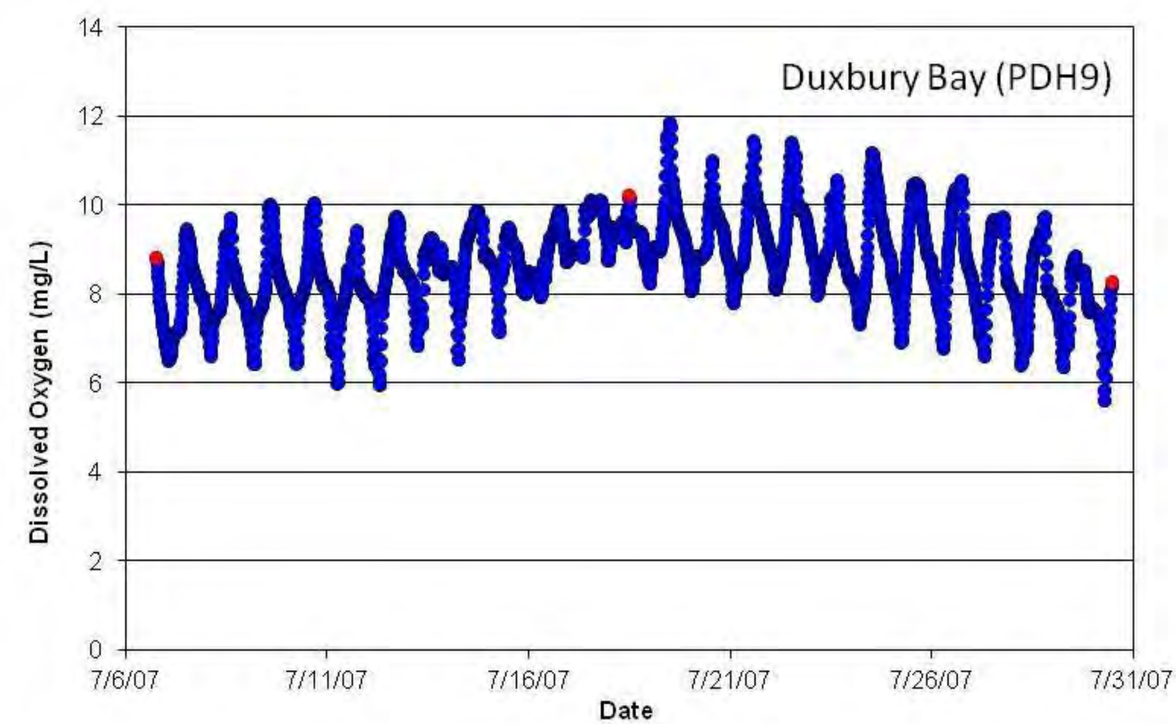


Figure VII-19. Bottom water record of dissolved oxygen within the lower area of the Duxbury Bay portion of the overall estuarine system, summer 2007 (Figure VII-2). Calibration samples shown as red dots.

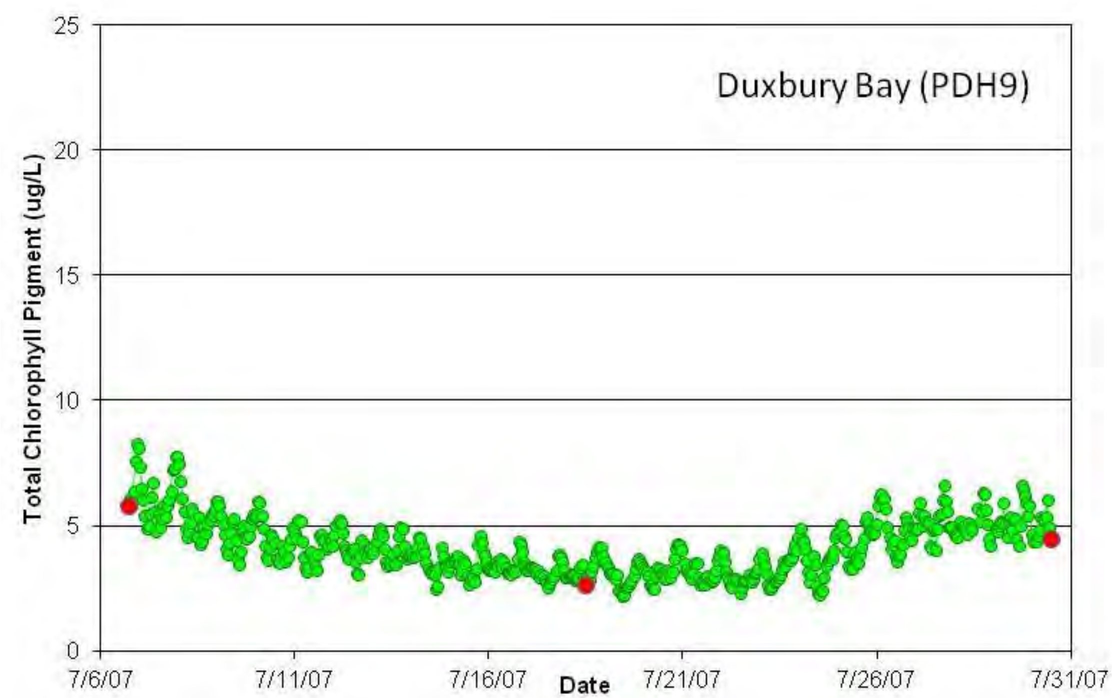


Figure VII-20. Bottom water record of Chlorophyll-*a* within the lower area of the Duxbury Bay portion of the overall estuarine system, summer 2007 (Figure VII-2). Calibration samples shown as red dots.

Table VII-1. Days and percent of time during deployment of in situ sensors that bottom water oxygen levels were below various benchmark oxygen levels.

Mooring Station Id.	Start Date	End Date	Total Deployment (Days)	<6 mg/L Duration (Days)	<5 mg/L Duration (Days)	<4 mg/L Duration (Days)	<3 mg/L Duration (Days)
Plymouth Harbor PDH 1	**7/6/2007**	**7/30/2007**	**24.1**	**1%**	**0%**	**0%**	**0%**
			Mean	0.04	0.04	0.04	0.04
			Min	0.02	0.02	0.02	0.02
			Max	0.06	0.06	0.06	0.06
			S.D.	0.02	0.03	0.03	0.03
Plymouth Harbor PDH2	**8/22/2007**	**10/16/2007**	**55.4**	**14%**	**0%**	**0%**	**0%**
			Mean	0.52	NA	NA	NA
			Min	0.01	0.00	0.00	0.00
			Max	5.05	0.00	0.00	0.00
			S.D.	1.28	NA	NA	NA
Plymouth Harbor PDH 3	**7/6/2007**	**7/30/2007**	**24.0**	**0%**	**0%**	**0%**	**0%**
			Mean	NA	NA	NA	NA
			Min	0.00	0.00	0.00	0.00
			Max	0.00	0.00	0.00	0.00
			S.D.	NA	NA	NA	NA
Plymouth Harbor PDH 4	**7/6/2007**	**7/30/2007**	**23.9**	**0.04%**	**0.00%**	**0.00%**	**0.00%**
			Mean	0.01	NA	NA	NA
			Min	0.01	0.00	0.00	0.00
			Max	0.01	0.00	0.00	0.00
			S.D.	NA	NA	NA	NA
Plymouth Harbor PDH5	**7/6/2007**	**7/30/2007**	**24.0**	**1%**	**0%**	**0%**	**0%**
			Mean	0.07	NA	NA	NA
			Min	0.04	0.00	0.00	0.00
			Max	0.10	0.00	0.00	0.00
			S.D.	0.02	NA	NA	NA
Plymouth Harbor PDH6	**7/6/2007**	**7/30/2007**	**23.8**	**0%**	**0%**	**0%**	**0%**
			Mean	NA	NA	NA	NA
			Min	0.00	0.00	0.00	0.00
			Max	0.00	0.00	0.00	0.00
			S.D.	NA	NA	NA	NA
Plymouth Harbor PDH7	**7/6/2007**	**7/30/2007**	**23.8**	**0%**	**0%**	**0%**	**0%**
			Mean	NA	NA	NA	NA
			Min	0.00	0.00	0.00	0.00
			Max	0.00	0.00	0.00	0.00
			S.D.	NA	NA	NA	NA
Plymouth Harbor PDH8	**7/6/2007**	**7/30/2007**	**23.7**	**26%**	**14%**	**7%**	**2%**
			Mean	0.18	0.14	0.13	0.08
			Min	0.04	0.02	0.06	0.05
			Max	0.33	0.26	0.21	0.14
			S.D.	0.09	0.08	0.05	0.03
Plymouth Harbor PDH9	**7/6/2007**	**7/30/2007**	**23.7**	**0.22%**	**0.00%**	**0.00%**	**0.00%**
			Mean	0.03	NA	NA	NA
			Min	0.01	0.00	0.00	0.00
			Max	0.04	0.00	0.00	0.00
			S.D.	0.02	NA	NA	NA

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

Mooring Station Id	Start Date	End Date	Total Deployment (Days)	>5 ug/L Duration (Days)	>10 ug/L Duration (Days)	>15 ug/L Duration (Days)	>20 ug/L Duration (Days)	>25 ug/L Duration (Days)
Plymouth Harbor PDH1	**7/6/2007**	**7/30/2007**	**24.10**	**65%**	**3%**	**1%**	**0%**	**0%**
Mean Chl Value = 6.7 ug/L			**Mean**	0.33	0.17	0.13	NA	NA
			Min	0.04	0.04	0.04	0.00	0.00
			Max	1.92	0.33	0.21	0.00	0.00
			S.D.	0.34	0.11	0.12	NA	NA
Plymouth Harbor PDH2	**8/22/2007**	**10/16/2007**	**55.4**	**78%**	**0%**	**0%**	**0%**	**0%**
Mean Chl Value = 6.3 ug/L			**Mean**	0.79	0.06	NA	NA	NA
			Min	0.04	0.04	0.00	0.00	0.00
			Max	7.83	0.08	0.00	0.00	0.00
			S.D.	1.19	0.03	NA	NA	NA
Plymouth Harbor PDH 3	**7/6/2007**	**7/30/2007**	**24.0**	**23%**	**1%**	**0%**	**0%**	**0%**
Mean Chl Value = 4.2 ug/L			**Mean**	0.18	0.13	NA	NA	NA
			Min	0.04	0.13	0.00	0.00	0.00
			Max	0.38	0.13	0.00	0.00	0.00
			S.D.	0.10	NA	NA	NA	NA
Plymouth Harbor PDH 4	**7/6/2007**	**7/30/2007**	**23.9**	**9%**	**0%**	**0%**	**0%**	**0%**
Mean Chl Value = 3.2 ug/L			**Mean**	0.19	NA	NA	NA	NA
			Min	0.04	0.00	0.00	0.00	0.00
			Max	0.54	0.00	0.00	0.00	0.00
			S.D.	0.14	NA	NA	NA	NA
Plymouth Harbor PDH5	**7/6/2007**	**7/30/2007**	**24.0**	**41%**	**4%**	**0%**	**0%**	**0%**
Mean Chl Value = 5.2 ug/L			**Mean**	0.29	0.13	NA	NA	NA
			Min	0.04	0.04	0.00	0.00	0.00
			Max	3.67	0.21	0.00	0.00	0.00
			S.D.	0.64	0.07	NA	NA	NA
Plymouth Harbor PDH6	**7/6/2007**	**7/30/2007**	**23.8**	**26%**	**0%**	**0%**	**0%**	**0%**
Mean Chl Value = 4.4 ug/L			**Mean**	0.16	NA	NA	NA	NA
			Min	0.04	0.00	0.00	0.00	0.00
			Max	0.92	0.00	0.00	0.00	0.00
			S.D.	0.19	NA	NA	NA	NA
Plymouth Harbor PDH7	**7/6/2007**	**7/30/2007**	**23.8**	**77%**	**22%**	**5%**	**0%**	**0%**
Mean Chl Value = 7.8 ug/L			**Mean**	0.84	0.18	0.13	0.04	NA
			Min	0.08	0.04	0.04	0.04	0.00
			Max	5.08	0.58	0.42	0.04	0.00
			S.D.	1.36	0.11	0.11	NA	NA
Plymouth Harbor PDH8	**7/6/2007**	**7/30/2007**	**23.7**	**72%**	**15%**	**2%**	**1%**	**0%**
Mean Chl Value = 7.1 ug/L			**Mean**	0.75	0.18	0.13	0.13	0.08
			Min	0.04	0.08	0.04	0.13	0.08
			Max	8.17	0.33	0.21	0.13	0.08
			S.D.	1.66	0.09	0.07	NA	NA
Plymouth Harbor PDH9	**7/6/2007**	**7/30/2007**	**23.7**	**20%**	**0%**	**0%**	**0%**	**0%**
Mean Chl Value = 4.1 ug/L			**Mean**	0.18	NA	NA	NA	NA
			Min	0.04	0.00	0.00	0.00	0.00
			Max	0.63	0.00	0.00	0.00	0.00
			S.D.	0.15	NA	NA	NA	NA

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Analysis of historical eelgrass coverage data was conducted for the Plymouth-Kingston-Duxbury Embayment System by the MassDEP Eelgrass Mapping Program as part of the MEP. Analysis of available aerial photos from 1951 was used to reconstruct the eelgrass distribution prior to any substantial development of the watershed. The 1951 data are for general guidance as they cannot be verified, although general patterns are similar to the verified 1995 coverages. In addition, qualitative field observations of eelgrass was made in 2007 and 2013 by scientists from the Coastal Systems Program (UMASS-SMAST) involved in the Massachusetts Estuaries Project. While these latter observations do not lend themselves to mapping of eelgrass coverage, they provide critical information on the absence/presence of eelgrass within this large embayment system and the general locations, depths and density of eelgrass where present. These data form the basis of the MEP eelgrass assessment for this estuary. It should be noted that the MEP Technical Team also contacted the Town of Plymouth to identify sources of historical eelgrass data, however, none were identified but for a qualitative mention of eelgrass presence as recorded in a report found by the Town of Plymouth, *Status of Eelgrass (Zostera marina) on the North Atlantic Coast, 1937*.

The primary use of the MEP eelgrass assessment for an estuary 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 2012 (Figure VII-21); 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 and the potential recoverable acreages should it be determined that habitat loss has occurred.

Plymouth Harbor Eelgrass Presence

Over the past several decades, eelgrass has generally existed across the Plymouth-Kingston-Duxbury Embayment System. Currently, eelgrass is present within large portions of the Plymouth-Kingston-Duxbury System, indicative of an estuary with high habitat quality areas. These eelgrass beds are generally restricted to the larger open water basins that comprise mid and outer regions of Plymouth Harbor, outer regions of Kingston Bay and the lower portion of Duxbury Bay. Coverage also seems to be most present fringing the tidal flats and associated channels (Figure VII-21a,b). The basins presently supporting eelgrass habitat also supported habitat in the 1951 historical analysis. However, it is clear from the 1995, 2001, 2006 and 2012 temporal sequence that the eelgrass areas are relatively stable except in Duxbury Bay, where the upper portion of the basin that supported eelgrass in 1995 has lost coverage over the following ~20 years. Equally significant is the pattern of coverage loss in this basin, where the uppermost eelgrass beds have gradually “retreated” toward the lower basin nearer the tidal inlet. This pattern of loss from upper tidal reaches with higher phytoplankton and nutrient levels to lower tidal reaches with lower chlorophyll a and higher light penetration due to proximity to offshore high quality flood waters has been seen in estuaries throughout southeastern Massachusetts by the MEP Technical Team.

It is interesting to note that the innermost region of Plymouth Harbor, near the mouth of Eel River has consistently not supported eelgrass apparently since 1937. Also, all studies since 1937 have reported coverage in the upper reaches of Plymouth Harbor as indicated in the excerpted 1937 report text as follows:

Several places have been noted on the coast v/here new growths of Zostera show great progress near the mouths of fresh-water streams. One such growth at Plymouth Harbor, Mass., which is bathed in almost pure fresh water from Eel Hiver at low tide, has progressed remarkably in the past season and has not shown any of the usual signs of disease, such as leaf- streaking or blotching. A similar situation prevails on a still larger scale among the eelgrass of Chesapeake Bay, where conditions have been almost normal for at least two years.

The present distribution of eelgrass coverage is consistent with the results of the oxygen and chlorophyll time-series data (Section VII.2), nitrogen levels within the inner and outer basins (Section VI) and the benthic infauna analysis (Section VII.4). The overall pattern of eelgrass distribution and temporal decline in coverage is consistent with the spatial pattern of nitrogen enrichment (Section VI) and oxygen and chlorophyll levels in the various basins and the water depth over the beds (above). The pattern of decline in coverage is typical of environmental changes wrought by nutrient enrichment. Nutrient enrichment tends to result in loss of eelgrass habitat in the uppermost reaches of the estuarine system which also tend to be the focus areas for watershed nitrogen inputs. Eelgrass loss appears to be most prevalent in the upper portion of the Duxbury Bay sub-basin, down gradient from the Bluefish River discharge. The pattern of loss from the tidal reaches furthest from the inlet can also be seen in the Pleasant Bay System on Cape Cod, where healthy beds remain within the region of the Chatham Harbor basin or in the eelgrass decline in Oyster Pond within the Stage Harbor Estuary. It appears from 1995-2012 field verified coverages that on the order of 330 acres of eelgrass beds has been lost and that the loss is continuing (Table VII-3 bottom). The clear loss of significant eelgrass coverage indicates that this system is slightly above its ability to assimilate additional nitrogen inputs without further habitat impairment. Nitrogen management is needed to recover the lost eelgrass acreage and to prevent further declines.

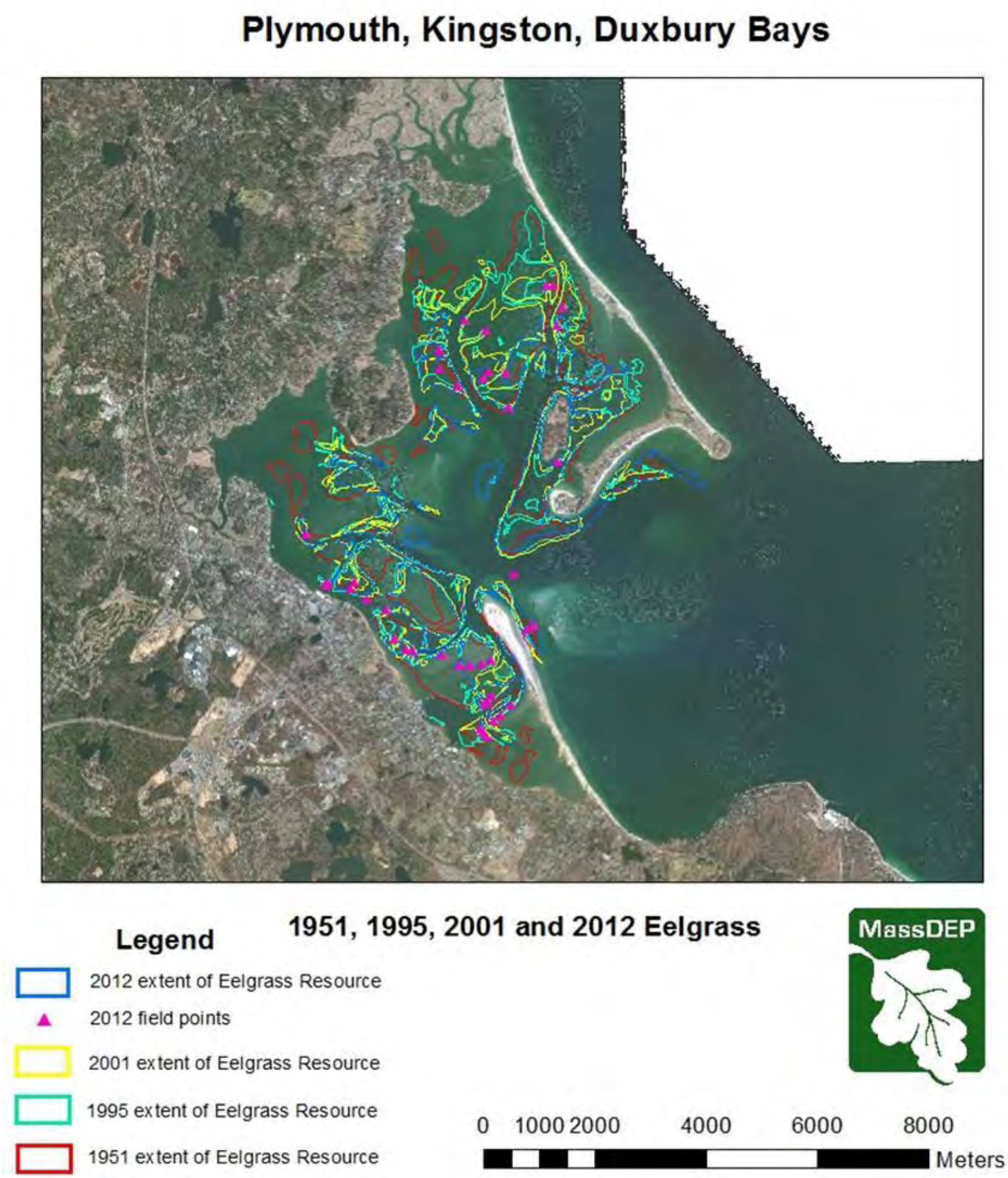


Figure VII-21a. Historical Eelgrass bed distribution within the Plymouth-Kingston-Duxbury Embayment System. The 1951 baseline coverage is outlined in red and was developed by the MassDEP Eelgrass Mapping Program using aerial photography and photo interpretation techniques, but has not been field verified. The 1995, 2001, 2012 (and 2006 not shown) have been field verified.

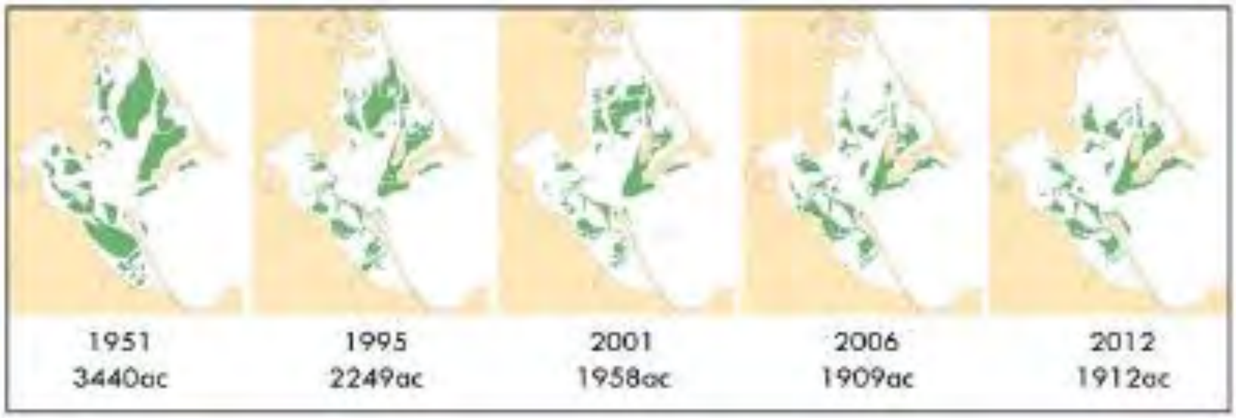


Figure VII-21b. Historical Eelgrass bed distribution within the Plymouth-Kingston-Duxbury Embayment System as determine by the MassDEP Eelgrass Mapping Program using aerial photography and photo interpretation techniques. 1951 has not been field verified but the 1995, 2001, 2006 and 2012 distribution maps have been field verified (Figure 2 in Ford and Carr DMF 2016).

Table VII-3. Eelgrass areal coverage determined by the MassDEP Eelgrass Mapping Program. Changes in area are determined from the coverage maps. It is not known why the values in the tables and in Figure VII-21b are very slightly different. The threshold development (Section VIII-2) used only the verified maps.

Plymouth-Kingston-Duxbury Embayment System: Temporal Change in Eelgrass Coverage					
Embayment	**1951 (acres)**	**1995 (acres)**	**2001 (acres)**	**2012 (acres)**	**Percent Difference (1951 to 2012)**
Lagoon Pond	3440.24	2245.71	1951.15	1912.52	**44%**

Plymouth-Kingston-Duxbury Embayment System: Temporal Change in Eelgrass Coverage					
Embayment	**1951 (acres)**	**1995 (acres)**	**2001 (acres)**	**2012 (acres)**	**Percent Difference (1995 to 2012)**
Lagoon Pond	3440.24	2245.71	1951.15	1912.52	**15%**

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling for benthic community characterization was conducted in two different years for the Plymouth-Kingston-Duxbury embayment system. The first characterization of benthic infauna habitat was completed by the MEP Technical Team in 2007 following the deployment of the above described oxygen/chlorophyll moorings. Subsequently, the same technical team returned to a subset of the 2007 coring locations and re-sampled those locations in 2013 to determine if there had been any significant changes in the benthic species diversity or evenness over the intervening 6 years. Sediment cores were collected in 2007 at 20 locations throughout the Plymouth-Kingston-Duxbury Embayment System (Figure VII-22). Sampling sites were generally distributed evenly throughout the system in order to get an unbiased representation of the benthic infaunal characteristics of the four component sub-embayments. Sampling sites were located in Plymouth Harbor (4), Kingston Bay (5), Duxbury Bay (8) and the main central basin adjacent the inlet (3). More sampling locations were situated in Duxbury Bay as that sub-embayment is the largest of the three sub-embayments and includes diverse basins: upper and lower regions of the Bay and the Duxbury Marsh basin. At each site multiple assays were conducted. No significant differences were observed between the 2007 and 2013 surveys when the specific sites are taken into account.

In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. All benthic animals from each sediment grab sample (Van Veen Grab) are identified to the species level and ranked as to their association with nutrient and organic enrichment related stresses, such as organic matter loading, hypoxia/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 standard community metrics of diversity and evenness.

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-*a* 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.

It should be noted that, given the moderate loss of eelgrass beds, the Plymouth-Kingston-Duxbury Embayment System is showing signs of impairment by nitrogen enrichment and that the system may have reached its limit for assimilating nitrogen without habitat impairment. The benthic infauna analysis is important for determining the level of impairment (healthy→moderately impaired→significantly impaired→severely degraded). Given the generally high water quality and modest loss of eelgrass, it is possible the benthic communities have not yet been impaired, as benthic communities are less sensitive to nitrogen enrichment than eelgrass, which requires high light penetration for growth. The benthic animal assessment coupled with the eelgrass assessment (above) are essential for the establishment of site-specific nitrogen thresholds (Section VIII).

Overall, the infauna survey indicated that most sub-basins comprising the Plymouth-Kingston-Duxbury Embayment System are presently supporting high quality benthic animal habitat. The exception is the upper portion of the Duxbury Bay sub-basin which is supporting lower numbers of species and individuals and lower diversity (2.30) and evenness (0.56) than the other open water basins with mud/fine sand sediments and relatively low water velocities. It is striking that the upper and lower portions of Duxbury Bay support very different benthic habitat quality with the diversity and evenness indicative of high quality habitat in the lower reach, 3.01 and 0.76, respectively, while these metrics indicate a moderate habitat impairment in the upper reach.

Close examination of the benthic community data from the main central basin-inlet did not indicate nitrogen related impairments. There were low numbers of stress indicator species, but the sediments consisted of swept sands which has been found in other estuaries to reduce the ability of benthic communities to persist. For example, Chatham Harbor in Pleasant Bay has very low species and number of individuals due to the high tidal velocities causing unstable sands. The central basin has somewhat lower tidal velocities and supports moderate numbers of species and individual but with only moderate diversity (2.23) and evenness (0.63). All indications are that this is due to natural environmental processes, not related to nitrogen enrichment. Similarly, the salt marsh dominated Duxbury Marsh basin, is supporting benthic habitat quality typical of high quality southeastern Massachusetts marsh basins, particularly the low diversity (1.75) and evenness (0.48) and species numbers. Typical of healthy salt marsh basins, the community has very few stress indicator species (<1%) and is dominated by spionids (*Streblospio benedicti*).

Kingston Bay and Plymouth Harbor both currently support high quality benthic community habitat. Kingston Bay has high numbers of individual and relatively high diversity (2.7) and evenness (0.69) with the uppermost areas likely locally affected by the Jones River discharge. Plymouth Harbor has a lower surface freshwater discharge (Town Brook, Eel River), and support moderate numbers of individuals, and very high diversity (3.28) and evenness (0.79) indicative of a high quality benthic habitat.

Overall there was a clear spatial pattern in habitat quality, with only moderate impairment found in the upper reach of Duxbury Bay (where eelgrass also has been lost), and high quality habitat throughout most of the other basins, with naturally lowered metrics in the main central basin (swept sands) and Duxbury Marsh not associated with nitrogen enrichment. Only Upper Duxbury Bay is currently supporting nitrogen impaired benthic habitat (as seen in the organic rich sediment and elevated chlorophyll levels (highest in system), but the habitat is only moderately impaired as oxygen levels remain high. The Benthic Survey did not reveal any areas of severe degradation, as indicated by low numbers of individuals and species or dominance by opportunistic stress indicator species such as Capitellids and Tubificids. In fact, at low velocity locations throughout the sub-basins of this embayment system, there were high numbers of individuals (>200 per grab sample), moderate to high numbers of species (15 to 18) and low numbers of Capitellids and Tubificids (generally <5% of community), see Table VII-3. While there is little evidence of high levels of nitrogen related impairment of benthic animal communities, upper Duxbury Bay did show clear evidence of low to moderate impairment associated with nitrogen and organic matter enrichment.

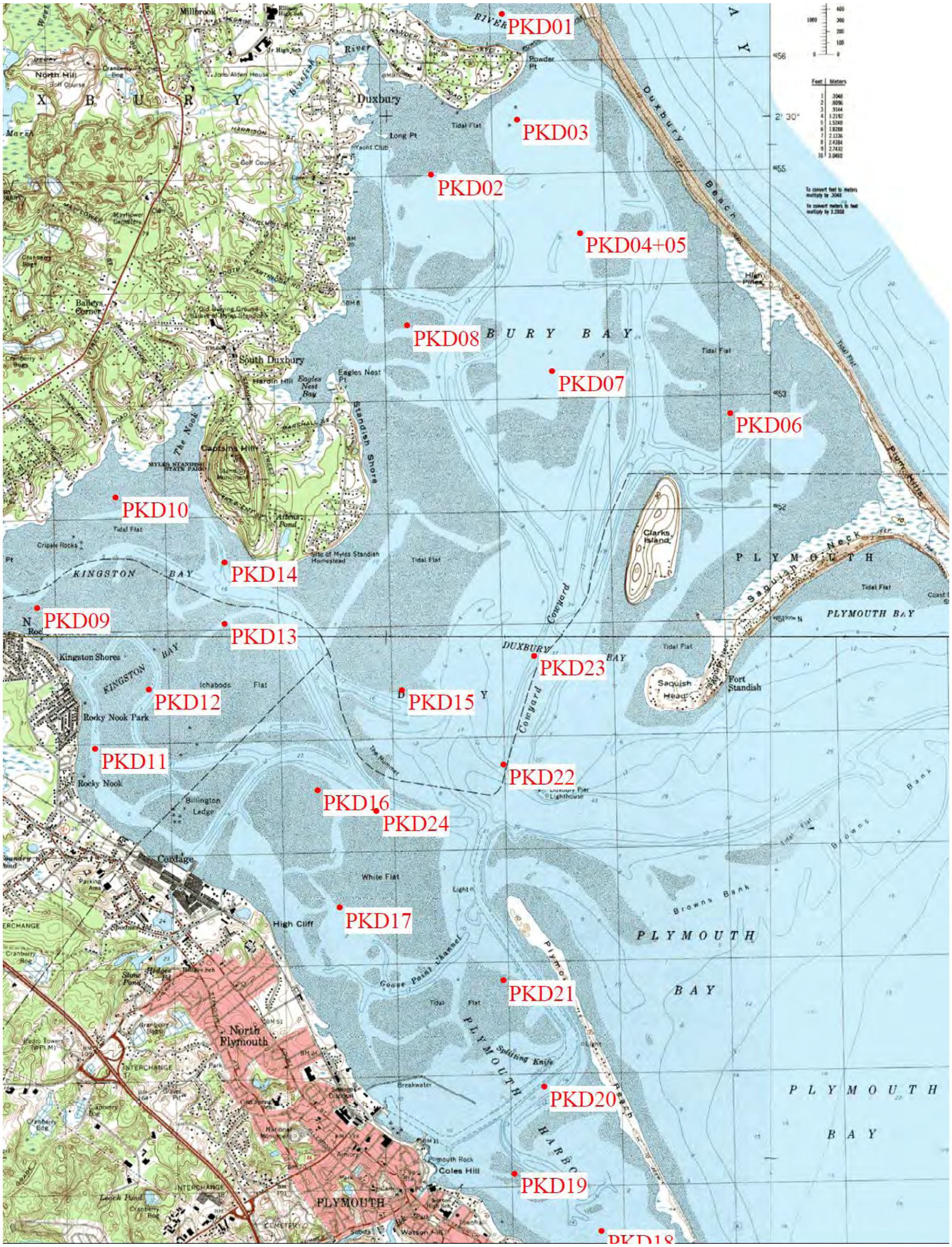


Figure VII-22. Map of the Plymouth-Kingston-Duxbury Embayment System showing location of benthic infaunal sampling stations (red symbol).

Table VII-3a. Benthic infaunal community data (2007) for the Plymouth-Kingston-Duxbury 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 0.0625 m^2. Stations refer to map in figure VII-22, replicate samples were collected at each location. S.E. is the standard error of the mean; N is the number of samples.

2007 Sub-Embayment	Station PKD-#	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)
PLYMOUTH/KINGSTON/DUXBURY ESTUARINE SYSTEM						
Dux Marshes	1	13	1122	6	1.75	0.48
Upper Dux Bay	2,3	17	1544	9	2.30	0.56
Lower Dux Bay	4,5,6,7,8	16	325	11	3.01	0.76
Ply-Dux Central	14,16,23	12	246	10	2.23	0.63
Kingston Bay	9,10,11,12,13	15	905	10	2.67	0.69
Plymouth Hbr	17,18,19,20	18	223	15	3.28	0.79

Table VII-3b. Benthic infaunal community data (2013) for the Plymouth-Kingston-Duxbury 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 0.0625 m^2. Stations refer to map in figure VII-22, replicate samples were collected at each location. S.E. is the standard error of the mean; N is the number of samples.

2013 Sub-Embayment	Station PKD-#	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)
PLYMOUTH/KINGSTON/DUXBURY ESTUARINE SYSTEM						
Dux Marshes	--	--	--	--	--	--
Upper Dux Bay	--	--	--	--	--	--
Lower Dux Bay	6,8	17	284	12	3.31	0.81
Ply-Dux Central	14,16	12	265	11	2.36	0.67
Kingston Bay	--	--	--	--	--	--
Plymouth Hbr	20	16	84	16	3.65	0.91

The results of the infauna survey and partial loss of eelgrass coverage within the Plymouth-Kingston-Duxbury Embayment System indicates that the nitrogen management threshold analysis (Section VIII) needs to aim for lowering nitrogen enrichment, specifically within Duxbury Bay, for restoration of impaired eelgrass and benthic animal habitat.. However, it is important to note that as portions of the Plymouth-Kingston-Duxbury Embayment System are showing nitrogen related impairments, adjacent areas are almost certainly nearing their loading threshold. Since these impairment are localized and relatively recent, it is likely that only limited reductions in nitrogen enrichment are required for restoration and protection of current high quality habitats. It should be emphasized that reducing nitrogen enrichment can be achieved by reducing nitrogen inputs and/or increasing its rate of loss through tidal exchange. However, in this system tidal flushing does not appear to be the main issue as tidal stages do not show signs of dampening.

It is clear that the recent signs of habitat impairment within the Plymouth-Kingston-Duxbury Embayment System are associated with nitrogen enrichment. The loss of the historical eelgrass makes restoration of this resource the primary focus for nitrogen management. Secondarily, the sub-basins that have slightly impaired benthic habitat should be restored as a consequence of management to restore the eelgrass habitat. Restoring these habitats should be the focus of the nitrogen management threshold analysis (Section VIII).

In addition to benthic infaunal community characterization undertaken as part of the MEP field data collection, other biological resources assessments were integrated into the habitat assessment portion of the MEP nutrient threshold development process as developed by the Commonwealth and available to the MEP Technical Team. The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish areas closed to harvest as well as the suitability of a system for the propagation of shellfish (Figure VII-23a,b,c,d). As is the case with some systems in southeastern Massachusetts and on Cape Cod, the majority of the central portion of the Duxbury Bay (as well as the salt marsh dominated area upgradient of Power Point Bridge that is referred to as Back River) and Kingston Bay sub-embayments is approved for the taking of shellfish year round. As one moves upgradient into the Kingston Bay sub-embayment the classification changes to conditionally approved and ultimately the area that represents the mouth of the Jones River is classified as prohibited to shellfishing. A small section of the uppermost portion of Duxbury Bay referred to as Bluefish River (DMF Growing Area Code CCB46) is conditionally approved (CCB46.1 and CCB46.2) to shellfishing during specific times during the year, typically the cold winter months, indicating the system is generally supportive of shellfish communities and slightly further upgradient into Bluefish River (CCB46.5) the area becomes classified as prohibited to shellfishing year round. With regard to the Plymouth Harbor sub-embayment, harvest of shellfish is prohibited year round throughout the entire sub-embayment indicating the presence of a persistent environmental contaminant. This is potentially due to the possibility of bacterial contamination most likely from WWTP outfall to Harbor waters. The major shellfish species with potential habitat within large portions of Duxbury Bay and Kingston Bay are mainly Quahogs. This area that is suitable as Quahog habitat is also interspersed with areas that are suitable for soft shelled clams (*Mya arenaria*) in shallower waters fringing the shore. More open water portions of the overall system appear suitable for surf clam and blue mussel (Figure VII-24).

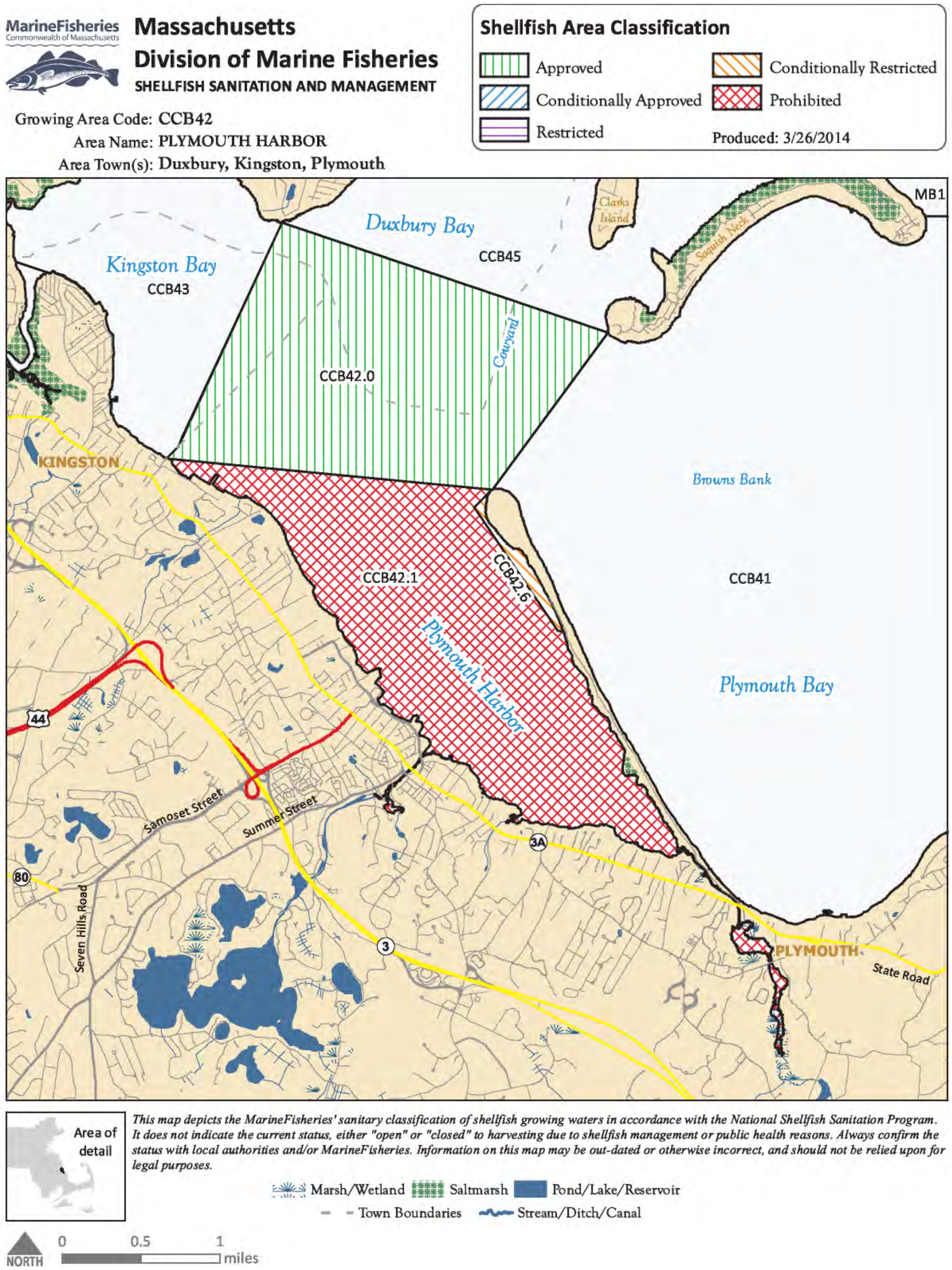


Figure VII-23a. Location of shellfish growing areas in Plymouth Harbor and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.

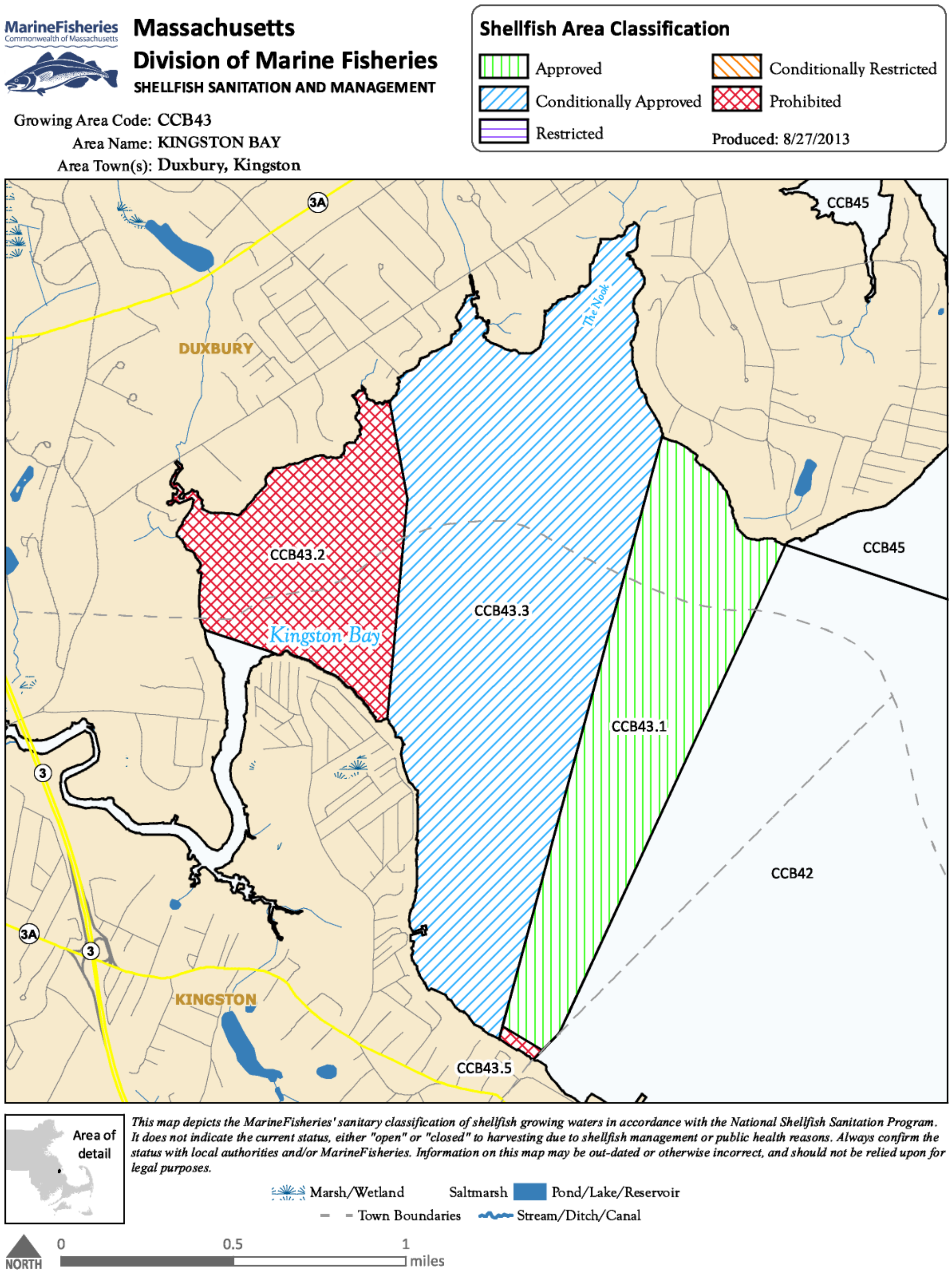


Figure VII-23b. Location of shellfish growing areas in Kingston Bay and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.

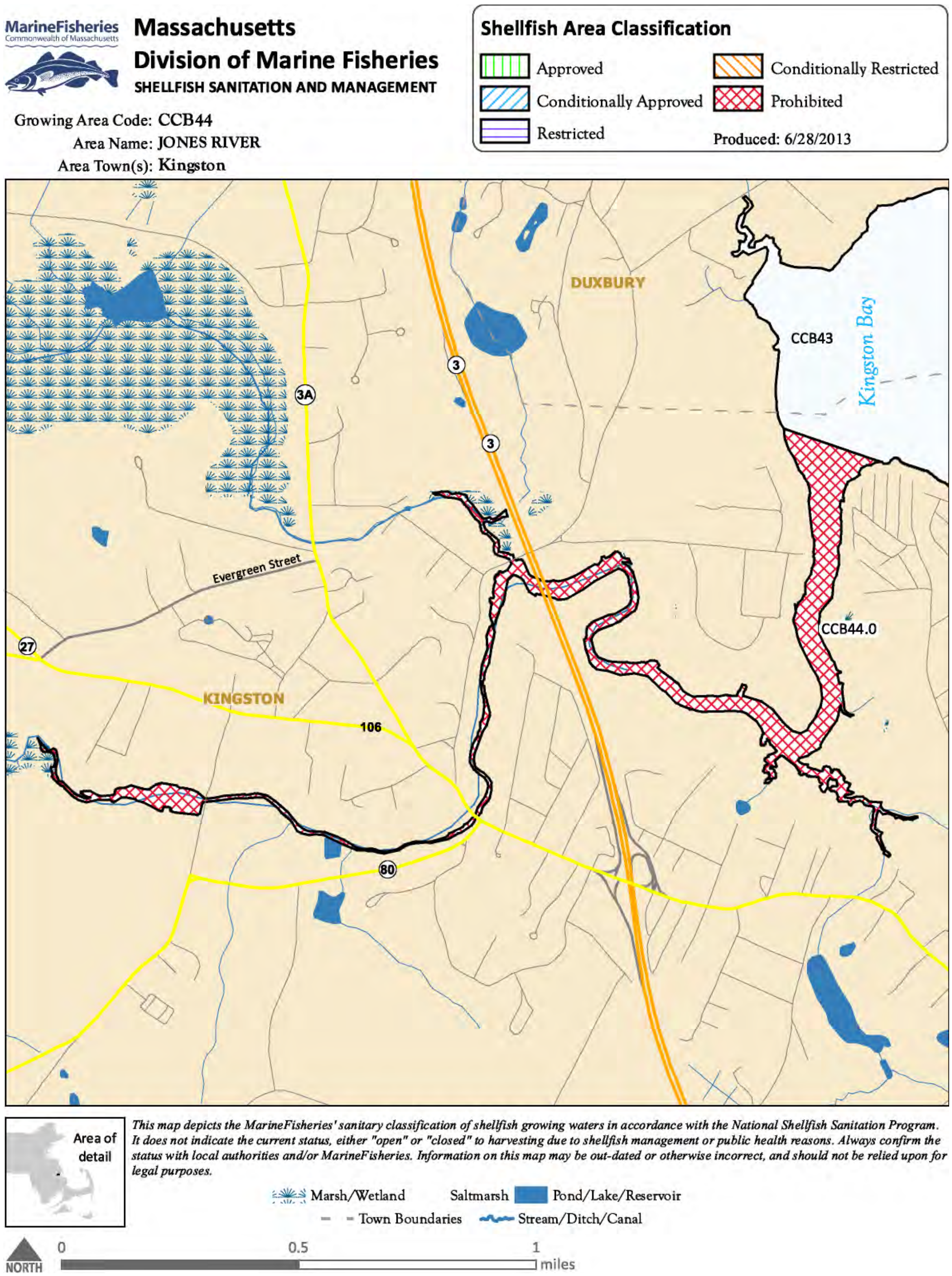


Figure VII-23c. Location of shellfish growing areas in the Jones River and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.

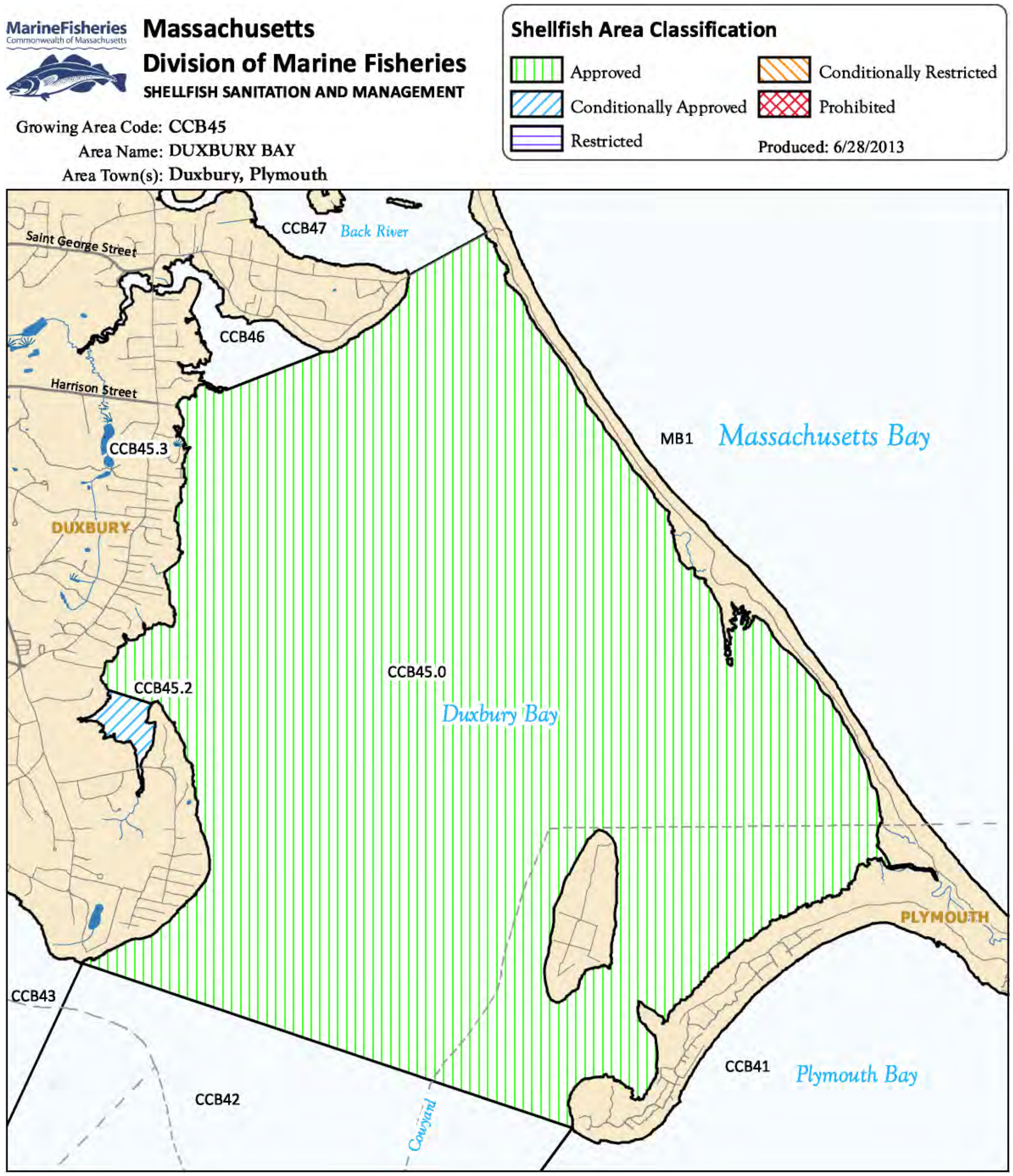


Figure VII-23d. Location of shellfish growing areas in Duxbury Bay and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.

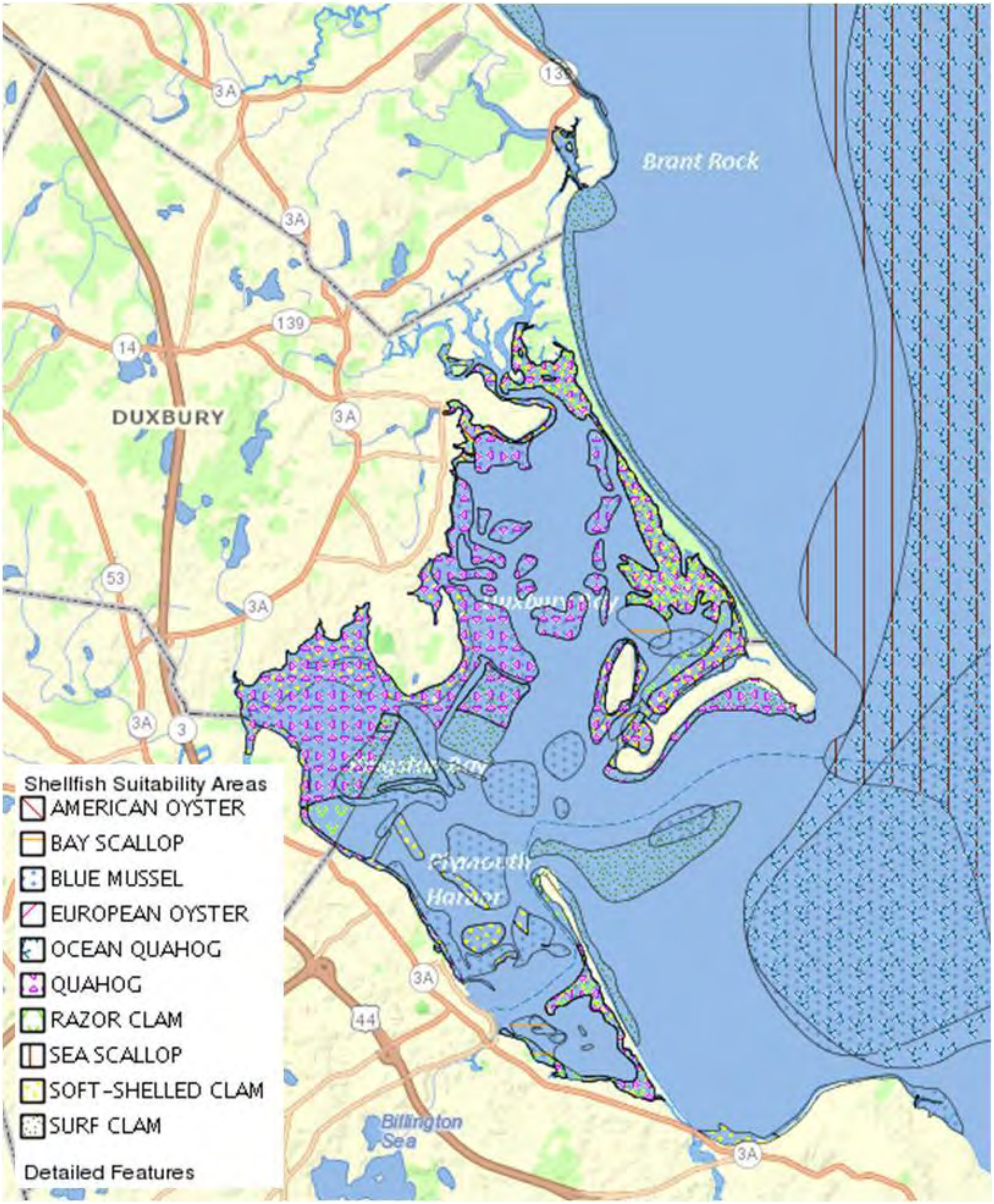


Figure VII-24. Location of shellfish suitability areas within the Plymouth-Kingston-Duxbury Embayment System as determined by the Massachusetts Division of Marine Fisheries. Suitability does not necessarily mean "presence".

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

VIII-1. ASSESSMENT OF NITROGEN-RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires the integration of key habitat parameters (infauna and eelgrass), sediment characteristic data, 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 all collected to support threshold development within the component sub-embayments comprising the Plymouth-Kingston-Duxbury Embayment System by the MEP Team and were discussed in Section VII. Nitrogen threshold development builds on these data and links habitat quality to summer water column nitrogen levels from long-term baseline water quality monitoring (Town of Plymouth Water Quality Monitoring Program, as tidally averaged nitrogen levels (Section VI).

Site-specific data was used for the development of the threshold which included temporal surveys of eelgrass distribution (1951, 1995, 2001, 2006, 2012); surveys of benthic animal communities and sediment characteristics (2007, confirmation in 2013); and summer time-series measurements of dissolved oxygen and chlorophyll-*a* (2007). These data were integrated with the water quality modeling results (Section VI) to quantitatively assess the present health of this embayment system as linked to nitrogen related water quality. The concept is to determine nitrogen levels currently supporting high quality benthic animal and eelgrass habitats, nitrogen levels supporting transitional habitats and levels associated with impaired habitats within the Plymouth-Kingston-Duxbury Embayment System. Many estuaries in s.e. Massachusetts have lost all of their eelgrass habitat and support impaired benthic animal habitat so linkage to nitrogen levels must necessarily rely on comparisons to high quality habitats in similar estuarine settings. Fortunately the Plymouth-Kingston-Duxbury Embayment System currently supports significant high quality benthic animal and eelgrass habitat allowing basin inter-basin comparisons within this single system. This increases the accuracy of the analysis and lowers the uncertainty relative to threshold nitrogen levels and appropriate watershed nitrogen loads.

The MEP habitat analysis uses eelgrass as a keystone 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. Eelgrass loss in southeastern Massachusetts estuaries associated with nitrogen enrichment is generally through decreased light penetration resulting from increased phytoplankton biomass and resulting suspended organic particles, as well as shading by epiphytes (small plants that colonize eelgrass shoots) and sometimes by accumulations of drift macroalgae. Each of these factors is a result of nitrogen enrichment and all result in stress to eelgrass beds. In addition, within the Plymouth-Kingston-Duxbury Embayment System benthic animal habitat is also important and is also part of the threshold analysis as a key habitat. The primary stress to benthic animal habitat is through nitrogen enrichment resulting in organic enrichment of sediments and associated sulfide accumulation and at higher levels, depletion of bottom water dissolved oxygen and smothering by accumulations of drift macroalgae.

The levels of oxygen depletion within the open water basins is consistent with a low to moderate levels of organic matter enrichment, primarily from phytoplankton biomass as seen in the parallel measurements of chlorophyll-*a*. The measured levels of oxygen depletion and chlorophyll-*a* levels are consistent with the observed nitrogen levels within the various basins and the parallel variation in these water quality parameters is generally consistent with watershed based nitrogen inputs being focused in the upper most portions of this estuarine system (e.g. upper Kingston Bay, upper Duxbury Bay down gradient of Powder Point Bridge and to a lesser extent Plymouth Harbor). The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll-*a* levels indicate low to moderately nutrient enriched waters with higher nutrient related water quality in the central basin of the overall system that encompasses portions of Plymouth Harbor, Kingston Bay and Duxbury Bay. Greater levels of oxygen depletion and phytoplankton biomass were observed in the uppermost portions of Duxbury Bay, particularly with Duxbury Marsh, a salt marsh dominated estuarine basin above the Powder Point bridge. The oxygen data is typical of salt marsh dominated basins which are naturally nutrient and organic matter rich and frequently show hypoxic conditions at night even in pristine salt marsh basins. In addition, the elevated chlorophyll a levels are consistent with the observed nitrogen levels. While the levels of chlorophyll a are not sufficient to impair salt marsh habitats, they reflect natural nutrient levels combined with watershed inputs focused in the headwaters of Duxbury Bay.

The pattern of low-moderate oxygen depletion, low-moderately elevated chlorophyll-*a* values in specific areas and low-moderate levels of nitrogen enrichment are consistent with the observed high quality eelgrass and benthic animal habitats throughout most of the open water basins and the moderate loss of eelgrass and moderately impaired benthic animal habitat with the upper portion of Duxbury Bay. At present oxygen depletion is not a major stressor within the component basins of this embayment system although there is some depletion (non-stressful) that suggests that the nitrogen threshold is being approached and that further increases in nitrogen loading will result in ecologically significant levels of oxygen depletion in Kingston Bay and Plymouth Harbor where bottom water was found to periodically decline below 6 mg/L.

Overall, the infauna survey indicated that most sub-basins comprising the Plymouth-Kingston-Duxbury Embayment System are presently supporting high quality benthic animal habitat. The exception is the upper portion of the Duxbury Bay sub-basin which is supporting lower numbers of species and individuals and lower diversity (2.30) and evenness (0.56) than the other open water basins with mud/fine sand sediments and relatively low water velocities. It is striking that the upper and lower portions of Duxbury Bay support very different benthic habitat quality with the diversity and evenness indicative of high quality habitat in the lower reach, 3.01 and 0.76, respectively, while these metrics indicate a moderate habitat impairment in the upper reach.

Close examination of the benthic community data from the main central basin-inlet did not indicate nitrogen related impairments. There were low numbers of stress indicator species, but the sediments consisted of swept sands which has been found in other estuaries across Cape Cod to reduce the ability of benthic communities to persist. For example, Chatham Harbor in Pleasant Bay (Town of Orleans, MA.) has very low species and number of individuals due to the high tidal velocities causing unstable sands. The central basin has somewhat lower tidal velocities and supports moderate numbers of species and individual but with only moderate diversity (2.23) and evenness (0.63). All indications are that this is due to natural environmental processes (shifting sands in high current areas), not related to nitrogen enrichment. Similarly, the salt marsh dominated Duxbury Marsh basin, is supporting benthic habitat quality typical of high quality southeastern Massachusetts marsh basins, particularly the low diversity (1.75) and evenness (0.48) and species numbers. Typical of healthy salt marsh basins, the community has very few stress indicator species (<1%) and is dominated by spionids (*Streblospio benedicti*).

Kingston Bay and Plymouth Harbor both currently support high quality benthic community habitat. Kingston Bay has high numbers of individual and relatively high diversity (2.7) and evenness (0.69) with the uppermost areas likely locally affected by the Jones River discharge. Plymouth Harbor has a lower surface freshwater discharge (Town Brook, Eel River), and support moderate numbers of individuals, and very high diversity (3.28) and evenness (0.79) indicative of a high quality benthic habitat.

Overall there was a clear spatial pattern in habitat quality, with only moderate impairment found in the upper reach of Duxbury Bay (where eelgrass also has been lost), and high quality habitat throughout most of the other basins, with naturally lowered metrics in the main central basin (swept sands) and Duxbury Marsh, not associated with nitrogen enrichment. Only Upper Duxbury Bay is currently supporting nitrogen related impaired benthic habitat (as seen in the organic rich sediment and elevated chlorophyll levels (highest in system), but the habitat is only moderately impaired as oxygen levels remain high. The benthic survey did not reveal any areas of severe degradation, as indicated by low numbers of individuals and species or dominance by opportunistic stress indicator species such as Capitellids and Tubificids. In fact, at low velocity locations throughout the sub-basins of this embayment system, there were high numbers of individuals (>200 per grab sample), moderate to high numbers of species (15 to 18) and low numbers of Capitellids and Tubificids (generally <5% of community). While there is little evidence of high levels of nitrogen related impairment of benthic animal communities, upper Duxbury Bay did show clear evidence of low to moderate impairment associated with the elevated water column nitrogen levels and modest organic matter enrichment.

The spatial distribution and changes in coverage of eelgrass within are consistent with the nitrogen, chlorophyll and oxygen levels. Currently, eelgrass is present within large portions of this embayment system, indicative of an estuary supporting high habitat quality and relatively low nitrogen enrichment. These eelgrass beds are generally restricted to the larger open water basins that comprise the mid and outer regions of Plymouth Harbor, outer regions of Kingston Bay and the lower portion of Duxbury Bay. Coverage also seems to be most present fringing the tidal flats and associated channels (Figure VII-21a,b). However, it is clear from the 1995, 2001, 2006 and 2012 temporal surveys that the eelgrass areas are relatively stable except in Duxbury Bay, where the upper portion of the basin that supported eelgrass in 1995 has lost coverage over the following ~20 years. Equally significant is the pattern of coverage loss in this basin, where the uppermost eelgrass beds have gradually "retreated" toward the lower basin nearer the tidal inlet. This pattern of loss from upper tidal reaches with higher phytoplankton and nutrient levels to lower tidal reaches with lower chlorophyll a and higher light penetration due to proximity to offshore high quality flood waters has been seen by the MEP Technical Team in estuaries throughout southeastern Massachusetts.

Integrating the nitrogen concentration data with the eelgrass coverage maps indicates that the region of eelgrass loss in upper Duxbury Bay currently support the highest total nitrogen levels of all the open water basins comprising the Plymouth-Kingston-Duxbury Embayment System, with the exception of Duxbury Marsh which historically has not supported eelgrass.

The present distribution of eelgrass coverage is consistent with the results of the oxygen and chlorophyll time-series data (Section VII.2), nitrogen levels within the inner and outer basins (Section VI) and the benthic infauna analysis (Section VII.4). The overall pattern of eelgrass distribution and temporal decline in coverage is consistent with: 1) the spatial pattern of nitrogen enrichment (Chapter VI), 2) oxygen and chlorophyll levels in the various basins and 3) the water depth over the beds (above). The pattern of decline in coverage is typical of environmental changes wrought by nutrient enrichment. Nutrient enrichment tends to result in loss of eelgrass habitat in the uppermost reaches of the estuarine system which also tend to have the highest nutrient concentrations and are typically the focus areas for watershed nitrogen inputs. Eelgrass loss appears to be most prevalent in the upper portion of the Duxbury Bay sub-basin, down gradient from the Bluefish River discharge. The pattern of loss from the tidal reaches furthest from the inlet can also be seen in the Pleasant Bay System on Cape Cod, where healthy beds remain within the region of the Chatham Harbor basin or in the eelgrass decline in Oyster Pond within the Stage Harbor Estuary, also on Cape Cod (Town of Chatham).

It appears from 1995-2012 field verified eelgrass coverages that on the order of 330 acres of eelgrass beds have been lost and that the loss is continuing (Section VII). The clear loss of significant eelgrass coverage indicates that this system is slightly above its ability to assimilate additional nitrogen inputs without further habitat impairment. Nitrogen management is needed to recover the lost eelgrass acreage and to prevent further declines. In addition, nitrogen reduction for eelgrass habitat restoration will also improve the moderately impaired benthic animal habitat within this embayment system by lowering organic enrichment of the sediments supporting infaunal communities.

VIII-2. THRESHOLD NITROGEN CONCENTRATIONS

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

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

The Plymouth-Kingston-Duxbury Embayment System Embayment System is presently supporting generally high quality benthic animal and eelgrass habitat throughout its component basins. However, the upper portion of Duxbury Bay is showing clear nitrogen related habitat impairment due to its loss of historic eelgrass beds and the low diversity and evenness of its benthic communities. Other basins appear to be bordering on potential impairment, but they have not yet been realized. Further managing nitrogen levels to restore upper Duxbury Bay habitats will also improve nitrogen related water quality throughout the Plymouth-Kingston-Duxbury Embayment System. Within upper Duxbury Bay, both the location and the temporal trend is consistent with nitrogen enrichment. However, the rate of documented eelgrass loss has been gradual and relatively recent (post 1995) which indicates that this estuary is only just beyond its nitrogen threshold (i.e. the level of nitrogen a system can tolerate without impairment). The presence of stable, dense eelgrass beds throughout the other component basins of the Plymouth-Kingston-Duxbury Embayment System, the generally high quality benthic animal habitat throughout these basins and only moderate impairment in the region of eelgrass loss in upper Duxbury Bay also indicates a system only just beyond its threshold. The indication of impairment to eelgrass and infaunal animal habitat as recently observed, is supported by the observed low levels of oxygen depletion and enhanced total pigment levels in this portion of the overall system. Upper Duxbury Bay is currently supporting moderately nitrogen impaired benthic habitat (as seen in the organic rich sediment and elevated chlorophyll levels (highest in system), but the habitat is only moderately impaired as oxygen levels remain high. The benthic surveys (2007 an 2013) did not reveal any areas of severe degradation, as indicated by low numbers of individuals and species or dominance by opportunistic stress indicator species such as Capitellids and Tubificids. The upper portion of the Duxbury Bay sub-basin supports lower numbers of species and individuals and lower diversity (2.30) and evenness (0.56) than the other open water basins with mud/fine sand sediments and relatively low water velocities. It is striking that the upper and lower portions of Duxbury Bay support very different benthic habitat quality with the diversity and evenness indicative of high quality habitat in the lower reach, 3.01 and 0.76, respectively, while these metrics indicate a moderate habitat impairment within the upper reach.

The spatial distribution of high quality and impaired habitats and associated oxygen and total pigment levels also parallels the gradient in water column total nitrogen levels within this estuary. The tidally averaged total nitrogen levels within the basins in areas supporting both high quality benthic animal and eelgrass habitat were observed to be 0.274 to 0.315 mg N L^{-1}. In areas not historically supporting eelgrass, but with unimpaired benthic habitat the tidally averaged TN levels were higher, 0.325 mg N L^{-1} (inner Plymouth Harbor) and 0.430 mg N L^{-1} mg N L^{-1} in the salt marsh dominated basin of Duxbury Marsh. In contrast the upper portion of the Duxbury Bay basin that is currently supporting moderately impaired benthic animal habitat and has recently lost eelgrass coverage has higher TN than the unimpaired open water basins (0.342 and 0.347 at long-term monitoring stations PDH-13 and PDH-14). These stations are associated with the area that has recently lost eelgrass and have higher observed TN levels than that found in high quality eelgrass areas. This provides additional support for the contention that eelgrass loss and benthic animal habitat impairment is associated with nitrogen related water quality. Based upon the location and nitrogen levels of these stations, the MEP Technical Team determined that PDH-13 and 14 would be ideal for use as sentinel stations for upper Duxbury Bay and the overall Plymouth-Kingston-Duxbury Embayment System. Lowering nitrogen levels at these stations (the average of PDH-13 and PDH-14 is used for comparison to the nitrogen threshold value) will necessarily lower nitrogen levels within other component basins and restore both eelgrass and benthic animal habitats within Duxbury Bay. It appears from the tidally averaged TN levels throughout the Plymouth-Kingston-Duxbury Embayment System that the appropriate threshold value to restore eelgrass in upper Duxbury Bay and protect eelgrass within the other component basins of the system should be 0.33 mg TN L^{-1}. The TN threshold must be less than observed levels where eelgrass habitat is currently impaired (0.345 mg TN L^{-1}) but can be higher than where eelgrass habitat is unpaired (0.315 mg TN L^{-1}) and benthic animal habitat is unimpaired (0.325 mg TN L^{-1}).

In comparison to other similar estuaries in southeastern Massachusetts, the observed TN levels and habitat stability/decline are consistent with persistence and loss of eelgrass at similar depths. For example, with the Nantucket Harbor Estuary, tidally averaged levels in the lower reach of Head of the Harbor (0.340-0.353) were associated with recent loss of eelgrass coverage, while eelgrass was lost from West Falmouth Harbor when tidally averaged TN exceeded 0.35 mg L^{-1}. The recent relatively small loss (as a percentage of total coverage) of eelgrass from Quissett Harbor was associated with tidally averaged nitrogen (total nitrogen, TN) levels of 0.354 mg N L^{-1}, while the Outer Basin high quality eelgrass habitat is at lower TN levels, 0.304 mg N L^{-1}. A threshold for tidally averaged TN at the sentinel station in the Inner Basin of Quissett Harbor (QH-2) of 0.34 mg N L^{-1}, was selected to restore eelgrass habitat. Similarly, healthy eelgrass beds within the Outer Harbor of West Falmouth Harbor exist at tidally averaged total nitrogen levels of 0.33-0.31 mg N L^{-1} (similar to the Plymouth-Kingston-Duxbury Embayment System), whereas eelgrass habitat was found to be impaired at total nitrogen levels of 0.37 mg N L^{-1}.

Restoring the impairments to eelgrass and benthic animal habitat is the focus of the nitrogen management threshold loading analysis (Section VIII.3). As eelgrass is more sensitive to nitrogen enrichment the threshold was selected to support restoration of stable eelgrass habitat which is presently showing moderate to significant impairment within upper Duxbury Bay (Table VIII-1). Nutrient management planning for restoration of the eelgrass habitat associated with the component basins to Plymouth-Kingston-Duxbury Embayment System should focus on reducing the level of nitrogen enrichment in main basin waters through watershed nitrogen management and managing tidal exchange as appropriate.

Based upon the information above and in Chapter VII and the level of eelgrass impairment observed, it appears that the system is presently only slightly beyond its nitrogen threshold for sustainable eelgrass coverage. This assessment is based upon several factors as follows: 1) the distribution of the remaining eelgrass habitat, 2) that the decline has been gradual and relatively recent and 3) that the system is only moderately nitrogen and organic matter enriched. The moderate level of impairment to benthic animal habitat also in upper Duxbury Bay should be restored if the threshold TN level for eelgrass restoration is achieved. Therefore, the focus of nitrogen management within Duxbury Bay should be on meeting the eelgrass threshold (see below) The nitrogen loads associated with the threshold concentration at the upper Duxbury Bay composite sentinel location is discussed in Section VIII.3, below.

VIII.3 DEVELOPMENT OF TARGET NITROGEN LOADS

The tidally averaged total nitrogen thresholds derived in Section VIII-2 were used to determine the amount of total nitrogen mass loading reduction required for restoration of habitat throughout the Plymouth-Kingston-Duxbury Embayment System. The thresholds are used as the target concentrations and the watershed N loading is adjusted in the calibrated constituent transport model (Section VI) until they are met at the sentinel stations. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the selected sentinel stations in Duxbury Bay. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent **only one of a suite of potential reduction approaches** that need to be evaluated by the community. This particular loading scenario is presented to establish the general degree and spatial pattern of watershed nitrogen source reduction that will be required for restoration of nitrogen related impairments throughout the embayment system, including restoration of eelgrass habitat. A comparison between present septic and total watershed loading and the loadings for the modeled threshold scenario is provided in Table VIII-2.

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required 13% removal of septic loads (associated with direct groundwater discharge to the embayment) from watersheds associated with the northern portion of the estuary. The distribution of tidally-averaged nitrogen concentrations associated with the threshold loading is shown in Figure VIII-1.

Tables VIII-3 and VIII-4 provide additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. For example, removal of 50% of the septic load from the Duxbury Bay watershed results in a 28% reduction in total watershed nitrogen load. Table VIII-4 shows the breakdown of threshold sub-embayment and surface water loads used for

Table VIII-1. Summary of Nutrient Related Habitat Health within the Plymouth-Kingston-Duxbury Embayment System, a macro-tidal estuary on Cape Cod Bay, based upon assessment data presented in Chapter VII. D.O. (dissolved oxygen) and Chl a (chlorophyll a) from the mooring data (VII.2). WQMP=Town Water Quality Monitoring Program results.

Sub-Embayment	Nutrient related Health Indicator					
	D.O.	Chl a	Macro-algae	Eelgrass	Infaunal Animals	Overall
Barnstable Great Marshes						
Duxbury Marshes	H[1]	H[5]	--[12]	--[13]	H[18]	**H[22]**
Duxbury Bay – Upper	H[4]	MI[6]	H/MI[11]	SI[15]	MI[16, 17]	**MI/SI[23]**
Duxbury Bay – Lower	H[3]	H[7]	--[12]	H[14]	H[16,19]	**H[24]**
Plymouth-Duxbury Central/Inlet	H[3]	H[8]	--[12]	H[14]	H[21]	**H[25]**
Kingston Bay	H[4]	H[9]	H/MI[11]	H[14]	H[16, 20]	**H[24]**
Plymouth Harbor	H[2]	H[10]	--[12]	H[14]	H[16, 19]	**H[24]**

1) salt marsh dominated sub-basin, natural oxygen depletions (large diurnal variations 8-3 mg/L), typical of salt marsh creeks, which can go anoxic due to naturally high organic sediments and high rates of oxygen uptake.
2) oxygen levels generally above 6 mg L^{-1} (99% and 86%of record) and always above 5 mg L^{-1} and all WQMP samples from 3 stations over 5 years were >6 mg/L in this basin.
3) time-series and WQMP found >6 mg/L 100% of observations
4) time-series >6 mg/L 99% or record, always >5.8 mg/L; WQMP found >6 mg/L 100% of observations
5) moderate chlorophyll a levels 7.1 ug/L with maxima >15 ug/L;WQMP 7.6 ug/L average over 5 years, not harmful in a highly organic salt marsh basin flushed twice per day
6) moderate phytoplankton biomass, chlorophyll a averaging 7.8 ug/L with maxima >20 ug L-1 of Time-series; WQMP average chlorophyll concentrations= 4.6 and 5.0 ug L-1 over 5 years
7) low-moderate chlorophyll a averaged 4.4 ug/L; WQMP = 4.0 ug L-1 over 5 years) 26 dates
8) low average chlorophyll a 4,2 ug/L (time-series); WQMP average of 3.0-4.0 ug L-1, 5 yrs
9) low-moderate chlorophyll a averaged 5.2 ug/L; WQMP 3 stations average 5.3 ug/L, range 3.6-7.1 ug/L over 5 yrs, 26 dates
10) low-moderate time-series chlorophyll of 6.3-6.7 ug/L; WQMP, 4.2-5.9 ug/L at 3 stations over 5 years
11) sparse filamentous or Ulva
12) drift algae not evident
13) no evidence this basin is supportive of eelgrass.
14) eelgrass coverage relatively stable, 1995-2012
15) clear evidence of significant eelgrass loss in pattern consistent with loss due to nitrogen enrichment
16) high numbers of individuals (>200 per grab), moderate to high numbers of species (15 to 18) and low numbers of Capitellids and Tubificids (generally <5% of community)
17) high numbers of species (17) and individuals (1500), but low diversity (2.30) and evenness (0.56) dominated by organic enrichment species, but only 3% Capitellids and Tubificids
18) community typical of high quality s.e. Massachusetts marsh basins, particularly the low diversity (1.75) and evenness (0.48) and species numbers (12) and dominance of Spionids, particulary Streblospio
19) high diversity (>3.0) and evenness (>0.7) indicative of high quality habitat.
20) high numbers of individuals (900/grab) and relatively high diversity (2.7) and evenness (0.69) appears to be near its nitrogen threshold
21) moderate numbers of species and individuals, with only moderate diversity (2.23) and evenness (0.63); appears due to natural processes (swept sands), not related to nitrogen enrichment.
22) all habitat metrics typical of a high quality salt marsh dominated basin in s.e. Massachusetts.
23) metrics indicate moderate of benthic habitat and significant impairment to eelgrass habitat, consistent with elevated chlorophyll a levels and presence of sparse Ulva patches..
24) all habitat metrics typical of a high quality open water estuarine basin in s.e. Massachusetts.
25) high quality eelgrass habitat and moderate infauna habitat due to high velocity flows and swept sands oxygen, chlorophyll indicative of high water quality and no macroalgal accumulations.

H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment;
SD = Severe Degradation; -- = not applicable to this estuarine reach

total nitrogen modeling. In Table VIII-4, loading rates are shown in kilograms per day. Note that 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 watershed nitrogen load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Cape Cod Bay, as discussed in Section VI.2.6.1.

Comparison of model results between existing loading conditions and the selected loading scenario to achieve the target TN concentrations at the sentinel station is shown in Table VIII-4. To achieve the threshold nitrogen concentrations at the sentinel stations, reductions in TN concentrations of about 11% is required in Duxbury Bay, which was achieved with at 7.4% reduction in total watershed nitrogen load.

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

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

sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Duxbury Marsh	23.123	17.342	-25.0%
Duxbury Bay	13.679	6.840	-50.0%
Kingston Bay	42.995	25.797	-40.0%
Plymouth Harbor	33.438	33.438	+0.0%
Blue Fish River	15.792	11.844	-25.0%
Jones River	57.266	57.266	+0.0%
Town Brook	51.616	51.616	+0.0%
Eel River	17.200	17.200	+0.0%
System Total	255.110	221.343	-13.2%

Table VIII-3. “Comparison of sub-embayment ***total watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Plymouth 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
Duxbury Marsh	32.92	27.14	-17.6%
Duxbury Bay	16.12	9.28	-42.4%
Kingston Bay	61.93	44.73	-27.8%
Plymouth Harbor	48.67	48.67	+0.0%
Blue Fish River	24.14	20.19	-16.4%
Jones River	116.49	116.49	+0.0%
Town Brook	70.81	70.81	+0.0%
Eel River	44.15	44.15	+0.0%
Plymouth WWTF Outfall	39.48	39.48	+0.0%
System Total	454.71	420.95	-7.4%

Table VIII-4. Threshold scenario sub-embayment and surface water loads used for total nitrogen modeling of the Plymouth 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 net (kg/day)
Duxbury Marsh	27.137	5.589	41.479
Duxbury Bay	9.281	59.200	13.579
Kingston Bay	44.728	49.227	40.244
Plymouth Harbor	48.671	25.614	50.707
Blue Fish River	20.192	-	-
Jones River	116.488	-	-
Town Brook	70.811	-	-
Eel River	44.153	-	-
Plymouth WWTF Outfall	39.485	-	-
system total	420.946	139.630	146.007

Table VIII-5. Comparison of model average TN concentrations from present loading and the threshold, with percent change over background in Cape Cod Bay (0.241 mg/L), for the Plymouth Bay system. Sentinel stations in bold (average of 2 stations is 0.333 mg N/L.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	threshold (mg/L)	% change
Plymouth Harbor - south	PDH1	0.325	0.323	-3.3%
Plymouth Harbor - boat basin	PDH2	0.315	0.312	-4.1%
Plymouth Harbor - mid	PDH3	0.296	0.293	-5.3%
Plymouth Harbor - north	PDH4	0.278	0.276	-7.2%
Plymouth Harbor - channel	PDH5	0.274	0.272	-7.2%
Kingston Bay - east	PDH6	0.274	0.271	-8.0%
Kingston Bay - Rocky Nook	PDH7	0.282	0.279	-7.6%
Kingston Bay - Goose Point	PDH8	0.281	0.278	-7.7%
Jones River	PDH9	0.389	0.380	-5.7%
Plymouth Bay	PDH10	0.265	0.263	-8.3%
Duxbury Bay - Cowyard	PDH11	0.276	0.273	-9.5%
Duxbury Bay - Saquish Neck	PDH12	0.307	0.300	-10.2%
Duxbury Bay - mid	**PDH13**	**0.342**	**0.331**	**-10.8%**
Duxbury Bay - west	**PDH14**	**0.347**	**0.335**	**-11.0%**
Duxbury Marsh	PDH15	0.430	0.408	-11.5%

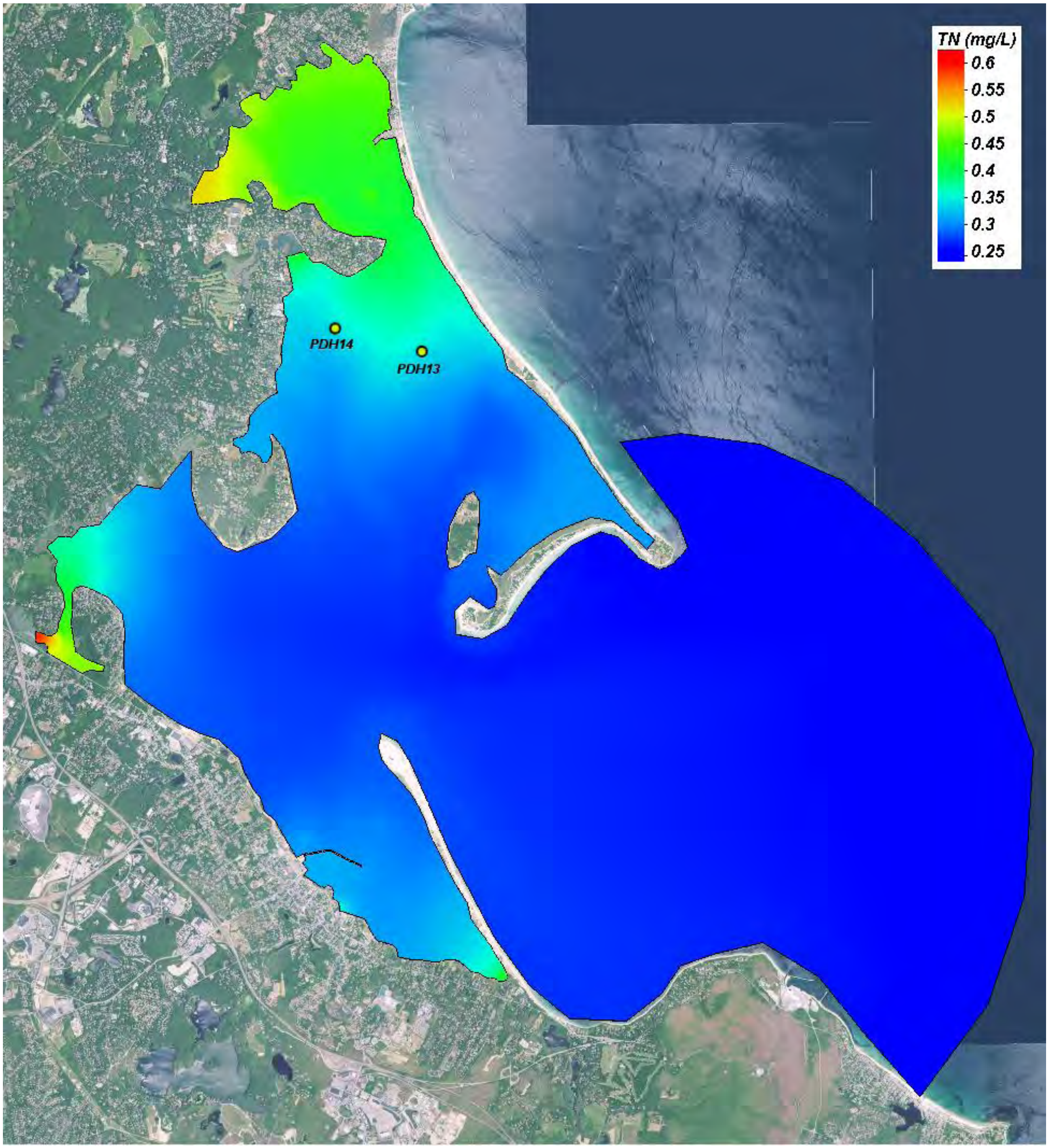


Figure VIII-1. Contour plot of tidally averaged modeled total nitrogen concentrations (mg/L) in the Plymouth Bay system, for threshold. Yellow markers indicate sentinel stations (PDH13 and PDH14) used to determine the threshold (0.33 mg/L)..

IX. ALTERNATIVE WATER QUALITY MODEL SCENARIOS

While discussing potential future water quality management strategies with Town of Plymouth staff, the Town requested four different scenarios. The Warren Avenue scenario included connection of all parcels within the prospective Warren Avenue sewer service area to the municipal wastewater treatment facility (Figure IX-1). This scenario is based on the existing development scenario including the 2006-2009 nitrogen effluent concentrations at the Town of Plymouth WWTF. As a result of connecting the selected properties to the sewer system, the daily nitrogen loading in the Plymouth Harbor LT10S (#1C) and Eel River 3A (#54) subwatersheds decreased by 3.37 kg and 1.15 kg, respectively, and the load within the Eel River W (#58) subwatershed increased by 1.56 kg/d. Incorporating the additional nitrogen attenuation in the Eel River, the overall reduction in existing nitrogen load as a result of sewering the Warren Avenue service area is 3.41 kg/d (or -0.6% from existing conditions).

The other three scenarios requested by the Town of Plymouth involved differing amounts of effluent discharge to the Town WWTF on-site beds and outfall pipe. As mentioned in Section IV, an average of 1.73 MGD was treated at the Town WWTF under existing conditions (2006 to 2009 data averages). The discharge of this treated effluent was divided between the on-site beds at the WWTF (9% of the effluent) and an outfall pipe into Plymouth Harbor (91% of the effluent). All three of the alternative WWTF scenarios had a total effluent flow of 1.75 MGD with varying amounts of the effluent flow discharged to the on-site beds at the WWTF. WWTF Scenario 1 included 1.25 MGD of effluent discharged to the on-site beds at the WWTF with the remainder of the flow discharged through the outfall, while WWTF Scenario 2 had 1.50 MGD discharged at the beds and 0.25 MGD discharged through the outfall. WWTF Scenario 3 had all of the WWTF effluent (1.75 MGD) discharged at the on-site beds.

During the development of the three WWTF scenarios, MEP and Town staff discussed how increased discharge at the WWTF site might alter the flow paths and surface water endpoint discharge locations for flow from the WWTF. The MEP watershed delineations were based on average WWTF discharge at the time of the USGS modeling (Masterson, *et al.*, 2009), which reasonably matched the existing conditions discharges. Because of this, the MEP watersheds represent groundwater conditions with relatively small discharge rates at the WWTF site; WWTF alternative scenarios have substantially larger portions of the WWTF flow discharged at the site. As part of the 1997 Town Facilities Plan/EIR, the Town had evaluated larger discharge flows at the WWTF site. These evaluations used a previous version of the regional USGS groundwater model to focus in on the area around the WWTF and evaluate groundwater mounding and surface water discharge locations for effluent applied at the WWTF site. The modeling alternatives focused on the impact of varying flows at the current WWTF site and a nearby adjacent site. MEP staff reviewed the alternative results and developed an apportionment of flow to nearby surface waters for each of the current WWTF scenarios (Table IX-1). This apportionment included reconciling watershed differences between the previous and current groundwater modeling results and extrapolation of the results. The overall approach and apportionment was necessarily a synthesis of various groundwater modeling results. Results were discussed with Town staff and approved for use in the MEP alternative scenarios.

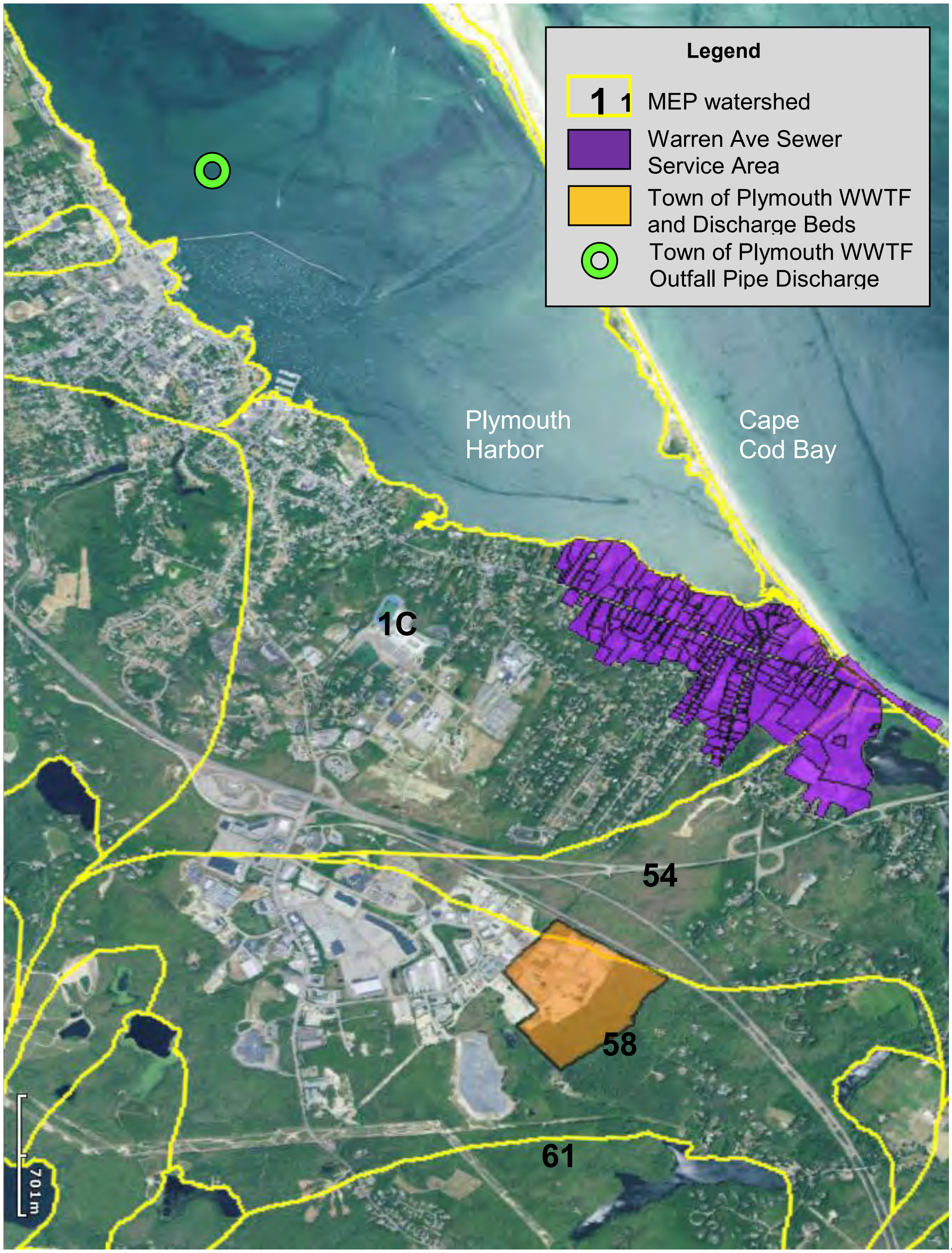


Figure IX-1. Town of Plymouth WWTF, Warren Avenue Sewer Service Area, and Outfall Pipe. Town of Plymouth asked for four alternative scenarios, including three with varying divisions of effluent discharge at the WWTF site and the outfall pipe and another scenario with the connection of properties within the Warren Avenue service area to the town wastewater treatment facility. The scenarios were completed with existing watershed development conditions, including nitrogen performance information for the WWTF.

Table IX-1. Town of Plymouth WWTF Scenarios. All scenarios are based on a total WWTF effluent discharge of 1.75 MGD and existing MEP development conditions in the overall watershed. Each of the scenarios varies the division of the discharge flow between the discharge beds on the WWTF site and the outfall discharge in Plymouth Harbor. Depending on the portion of the discharge directed to the beds, the location of which surface water is impacted varies. The percentage of WWTF bed discharge that arrived at each surface water are shown based on groundwater modeling completed by CDM for the Phase IIIA Facilities Plan/Environmental Impact Report (1997) completed for the Town.

				Surface Water Endpoint Discharge Location for Flow from Beds					
MEP WWTF scenario	Total Flow	Flow to Beds	Flow to Outfall	Warren Wells Brook & Hayden Pond	Russell Mill Pond	Confluence to Route 3A	Route 3A to Harbor	Deep Discharge to Harbor	Total
				MEP shed 58	MEP shed 61	MEP shed 54	MEP shed 1C	MEP shed 1C	
	MGD	MGD	MGD	%	%	%	%	%	%
1	1.75	1.25	0.50	21	9	17	39	14	100
2	1.75	1.50	0.25	24	14	11	32	21	100
3	1.75	1.75	0.00	26	18	4	24	28	100

Note: totals may not match due to rounding.

Each scenario was run using the Plymouth Bay TN model. Model N loading for each scenario are provided in Tables IX-2 through IX-4. The percent change in the watershed loads for each scenario is presented in Tables IX-5.

Similar to the other N loading scenarios, benthic flux is modified from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Cape Cod Bay, as discussed in Section VI.2.6. TN concentrations at the water quality monitoring stations and the resulting percent change from present conditions are presented in Table IX-6.

Table IX-2. WWTF scenario 1 sub-embayment and surface water loads used for total nitrogen modeling of the Plymouth 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 net (kg/day)
Duxbury Marsh	32.918	5.589	44.988
Duxbury Bay	16.121	59.200	14.576
Kingston Bay	61.926	49.227	40.777
Plymouth Harbor	65.049	25.614	50.792
Blue Fish River	24.140	-	-
Jones River	116.488	-	-
Town Brook	70.811	-	-
Eel River	56.277	-	-
Plymouth WWTF Outfall	12.362	-	-
system total	456.090	139.630	151.133

Table IX-3. WWTF scenario 2 sub-embayment and surface water loads used for total nitrogen modeling of the Plymouth 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 net (kg/day)
Duxbury Marsh	32.918	5.589	44.988
Duxbury Bay	16.121	59.200	14.576
Kingston Bay	61.926	49.227	40.759
Plymouth Harbor	68.142	25.614	50.717
Blue Fish River	24.140	-	-
Jones River	116.488	-	-
Town Brook	70.811	-	-
Eel River	58.153	-	-
Plymouth WWTF Outfall	6.181	-	-
system total	454.879	139.630	151.040

Table IX-4. WWTF scenario 2 sub-embayment and surface water loads used for total nitrogen modeling of the Plymouth 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 net (kg/day)
Duxbury Marsh	32.918	5.589	44.988
Duxbury Bay	16.121	59.200	14.576
Kingston Bay	61.926	49.227	40.736
Plymouth Harbor	71.170	25.614	50.625
Blue Fish River	24.140	-	-
Jones River	116.488	-	-
Town Brook	70.811	-	-
Eel River	59.816	-	-
Plymouth WWTF Outfall	0.000	-	-
system total	453.389	139.630	150.925

Table IX-5. Comparison of sub-embayment ***total watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Plymouth 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)	Scen. 1 load (kg/day)	Scen. 1 % change	Scen. 2 load (kg/day)	Scen. 2 % change	Scen. 3 load (kg/day)	Scen. 3 % change
Duxbury Marsh	32.918	32.918	0.0%	32.918	0.0%	32.918	0.0%
Duxbury Bay	16.121	16.121	0.0%	16.121	0.0%	16.121	0.0%
Kingston Bay	61.926	61.926	0.0%	61.926	0.0%	61.926	0.0%
Plymouth Harbor	48.671	65.049	33.7%	68.142	40.0%	71.170	46.2%
Blue Fish River	24.140	24.140	0.0%	24.140	0.0%	24.140	0.0%
Jones River	116.488	116.488	0.0%	116.488	0.0%	116.488	0.0%
Town Brook	70.811	70.811	0.0%	70.811	0.0%	70.811	0.0%
Eel River	44.153	56.277	27.5%	58.153	31.7%	59.816	35.5%
WWTF Outfall	39.485	12.362	-68.7%	6.181	-84.3%	0.000	-100.0%
System Total	454.712	456.090	0.3%	454.879	0.0%	453.389	-0.3%

Table IX-6. Comparison of model average TN concentrations from present loading and the three WWTF scenarios, with percent change over background in Cape Cod Bay (0.241 mg/L), for the Plymouth Bay system. See Figure VI-1 for a map of station locations.

location	monitoring station (MEP ID)	present (mg/L)	Scen. 1 (mg/L)	Scen. 1 % change	Scen. 2 (mg/L)	Scen. 2 % change	Scen. 3 (mg/L)	Scen. 3 % change
Plymouth Harbor	PDH1	0.325	0.336	+12.2%	0.337	+14.0%	0.338	+15.5%
Plymouth Harbor	PDH2	0.315	0.317	+2.3%	0.317	+2.6%	0.317	+2.6%
Plymouth Harbor	PDH3	0.296	0.298	+3.4%	0.298	+3.8%	0.298	+4.0%
Plymouth Harbor	PDH4	0.278	0.278	-1.1%	0.278	-1.3%	0.278	-1.9%
Plymouth Harbor	PDH5	0.274	0.275	+0.6%	0.275	+0.6%	0.275	+0.3%
Kingston Bay	PDH6	0.274	0.274	+0.3%	0.274	+0.3%	0.274	+0.3%
Kingston Bay	PDH7	0.282	0.282	+0.5%	0.282	+0.5%	0.282	+0.2%
Kingston Bay	PDH8	0.281	0.281	+0.2%	0.281	+0.2%	0.281	+0.2%
Jones River	PDH9	0.389	0.389	+0.1%	0.389	+0.1%	0.389	+0.1%
Plymouth Bay	PDH10	0.265	0.265	+0.4%	0.265	+0.4%	0.265	+0.4%
Duxbury Bay	PDH11	0.276	0.276	+0.3%	0.276	+0.3%	0.276	+0.3%
Duxbury Bay	PDH12	0.307	0.307	+0.2%	0.307	+0.2%	0.307	+0.2%
Duxbury Bay	PDH13	0.342	0.342	+0.1%	0.342	+0.1%	0.342	+0.1%
Duxbury Bay	PDH14	0.347	0.347	+0.1%	0.347	+0.1%	0.347	+0.1%
Duxbury Marsh	PDH15	0.430	0.430	+0.1%	0.430	+0.1%	0.430	+0.1%

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