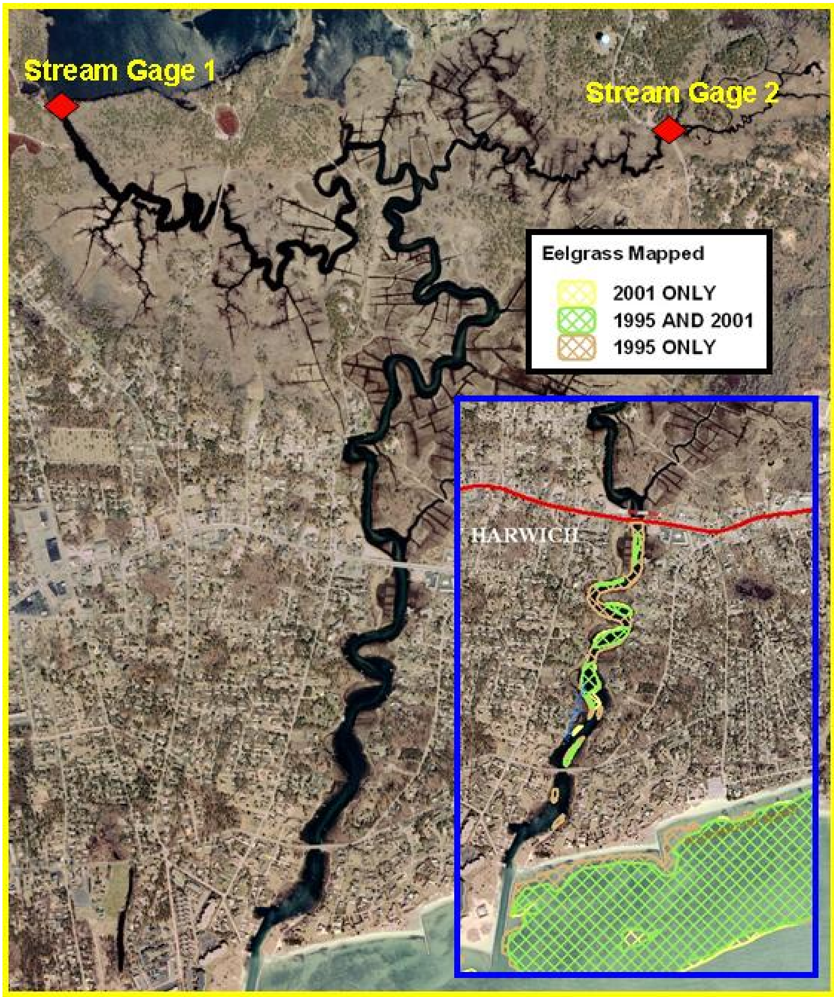
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Herring River Embayment System, Harwich, Massachusetts



University of Massachusetts Dartmouth
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Environmental Protection

DRAFT REPORT – June 2012

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ACKNOWLEDGMENTS

The Massachusetts Estuaries Project Technical Team would like to acknowledge the contributions of the many individuals who have worked tirelessly for the restoration and protection of the critical coastal resources of Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Herring River Embayment System. Without these stewards and their efforts, this project would not have been possible.

First and foremost we would like to recognize and applaud the significant time and effort in data collection and discussion spent by members of the Town of Harwich Water Quality Monitoring Program and SMAST "volunteers". These individuals gave of their time to collect nutrient related water quality from this system, without which the present analysis would not have been possible. Of particular note are Frank Sampson Chairman of the Town-wide Water Quality Management Task Force and Heinz Proft, Coordinator of the Town of Harwich Monitoring Program. Similarly, many in the Town of Harwich helped in this effort, the Harwich Department of Public Works and the Harwich Shellfish Department.

In addition to local contributions, technical, policy and regulatory support has been freely and graciously provided by Paul Niedzwiecki and Tom Cambareri of the Cape Cod Commission. We are also thankful for the long hours in the field and laboratory spent by the technical staff (Jennifer Benson, Michael Bartlett, Sara Sampieri, Camille David and Dahlia Medieros), interns and students within the Coastal Systems Program at SMAST-UMD.

Support for this project was provided by the Town of Harwich, Barnstable County, Three Bays Preservation Trust and the University of Massachusetts-Dartmouth and MassDEP supported initial field work in support of this project.

SUGGESTED CITATION

Howes B., H.E. Ruthven, J. S. Ramsey, R. Samimy, D. Schlezinger, E. Eichner (2012). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Herring River Embayment System, Harwich, Massachusetts, Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

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I. INTRODUCTION

The Herring River Marsh / Embayment System is located within the Town of Harwich, Massachusetts on Cape Cod. While the estuary is located entirely within Harwich, the watershed to the overall system extends into the Town of Brewster and slightly into the Town of Dennis. The Herring River System is comprised of a main tidal channel, a west branch that extends up to a man-made freshwater reservoir and an east branch that extends up into a small terminal brackish marsh. The Herring River System is functionally a wetland, one of the largest on Cape Cod (Strahler 1988), grading from salt marsh in the lower and mid reaches and along the major tidal creeks to brackish to predominantly freshwater marsh on the marsh plain in its upper regions. Although the Herring River System functions primarily as a tidal wetland system, its lower reach close to the inlet is a tidal river with limited wetland vegetation (from inlet to Rt. 28 bridge), rather than being structured as a tidal marsh creek. In this area the tidal channel is relatively wide and navigable thus functioning more like an open water basin than a marsh. Up-gradient of Route 28, the channel narrows and intersects with numerous tidal ditches and smaller tributary marsh creeks. The difference in structure above and below Rt. 28 bridge created historic eelgrass habitat and benthic animal communities of more open water basins in the lower tidal reach and wetland dominated habitats in the upper system of salt marsh and tidal channels. This ecological difference results in a greater sensitivity to nitrogen in the lower tidal river portion than in the upper wetland dominated portions.

This large tidal marsh system situated on the southern shore of Cape Cod receives tidal flood water from Nantucket Sound through a single tidal inlet (Figure I-1). The inlet has been stabilized by a pair of jetties and is bounded by beach to both the east and west. Pleasant Road Beach to the east terminates at the inlet on one side while Inman Road Beach bounds the inlet to the west. In this area the tidal channel is relatively wide and navigable thus functioning more like an embayment rather than a marsh. Up-gradient of Route 28, the channel narrows and penetrates into a portion of the system that is dominated by salt marsh.

Overall, the Herring River Marsh system is typical of a large New England tidal marsh system, with the lower regions composed of predominantly salt marsh dominated by a central tidal creek and the marsh plain colonized by *Spartina alterniflora* (low marsh) and *Spartina patens and Distichlis spicata* (high marsh). The upper regions, furthest from the tidal inlet show the influence of the freshwater inflows from the surrounding watershed with species grading to brackish marsh dominated by *Phragmites* finally shifting to freshwater marsh dominated by *Typha* and other freshwater species on the marsh plain (Figure I-2).

Tidal exchange with the high quality waters of Nantucket Sound is high, given the maintained inlet and the moderate offshore tide range (ca. 6 feet), which has also resulted in tidal creeks which are moderately incised, with near complete drainage of tidal creeks in the upper most portions of the system at low tide. Observations by the USGS and the MEP Technical Team indicate a healthy functioning New England tidal wetland system.

The Herring River Marsh provides both wildlife habitat and a nursery to offshore fisheries, as well as serving as a storm buffer and nutrient sink for watershed derived nitrogen. The upper upland margin of the Herring River Marsh to the west currently supports a conservation area with walking trails and a large herring run that provides a hydraulic connection between the tidally influenced upper marsh system and the up-gradient freshwater reservoir.

Figure I-1. Study region for the Massachusetts Estuaries Project nitrogen thresholds analysis for the Herring River Marsh System. Tidal waters are exchanged with Nantucket Sound through a single tidal inlet, stabilzed by jetties. Freshwaters enter primarily into the uppermost reaches to the east and the west via specific stream flows and more generally throughout via direct groundwater discharge.

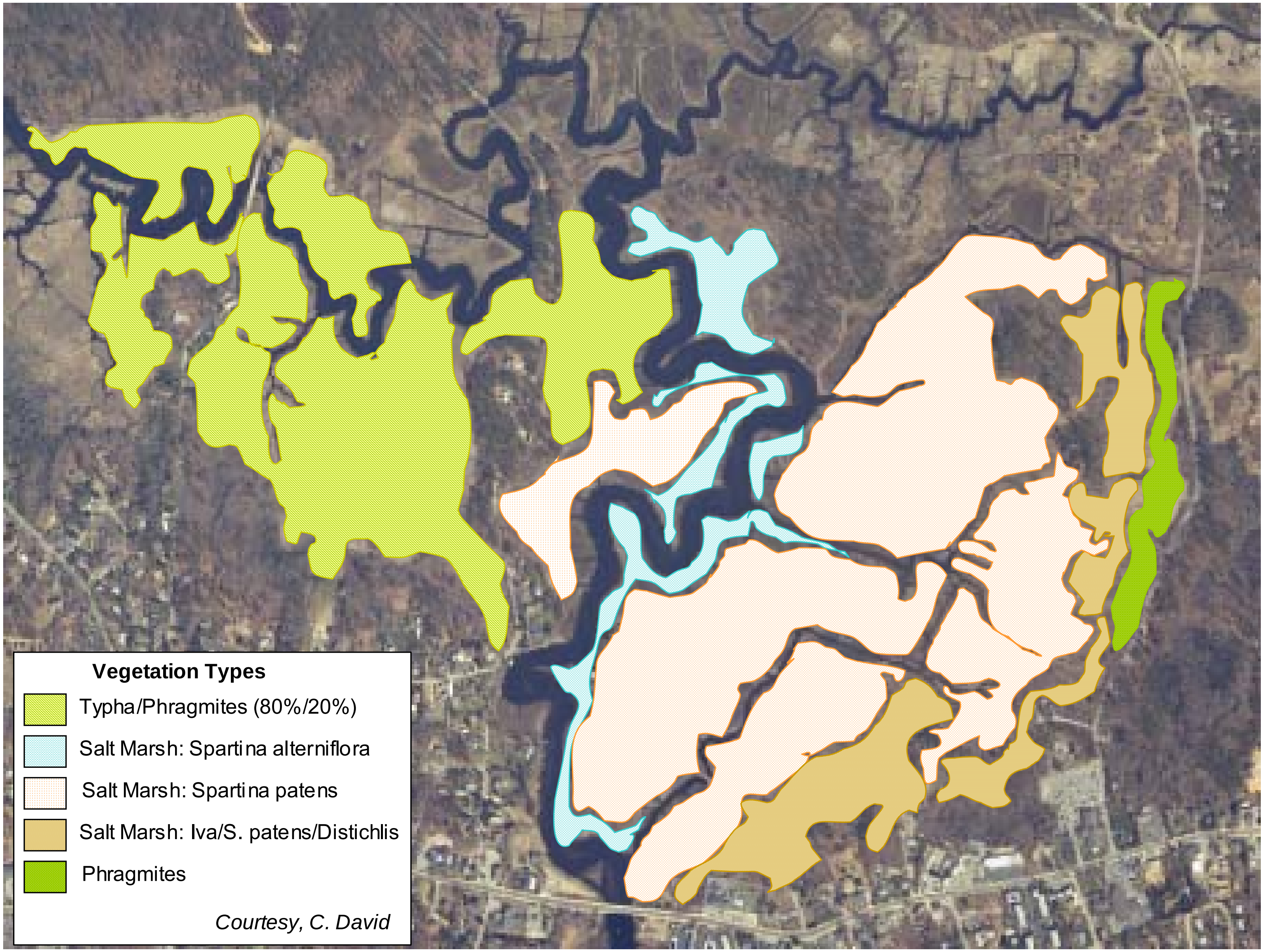

Figure I-2. Wetland vegetation map of upper Herring River wetlands conducted in 2009. Map courtesy of D. Medeiros and C. David.

The Herring River Marsh is a moderately sized estuary behind a barrier beach formed by coastal processes associated with nearshore Nantucket Sound. This estuarine system is a relatively "young" coastal feature that required significant post glaciation sea-level rise and the formation of the barrier beach, occurring on the order of 2500-4000 years b.p. At present, two moderately sized streams are present in the system, one representing discharge from the freshwater reservoir in the western portion of the system and the other discharging from a small terminal marsh in the eastern portion of the system. Both streams discharge only a fraction of the aquifer recharge to the estuary, the rest entering from groundwater flow or direct precipitation onto the marsh surface.

Tidal exchange with Nantucket Sound is through a 100-foot wide main inlet that is armored to the east and the west and bounded by beach on both sides. The beach and the inlet are very dynamic geomorphic features, due to the influence of littoral transport processes. These processes may periodically affect the health of this estuary through changes in hydrodynamics wrought by sedimentation in and around the tidal inlet (see Section V). To the extent that the inlet becomes restricted and tidal flushing is reduced, nitrogen loading impacts will be magnified over present conditions. Equally important, to the extent that tide range may become reduced in the upper portions of the system, the health and productivity of the emergent salt marsh would be reduced. Any long-term habitat management plan for the Herring River System must recognize the importance of inlet dynamics and include options to continue and maintain tidal exchange.

The primary ecological threat to Herring River estuarine resources is degradation resulting from nutrient enrichment, particularly within the lower tidal river reach. Loading of the critical eutrophying nutrient, nitrogen, has been increasing over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to this and other outer Cape estuaries (such as Pleasant Bay, Namskaket Marsh, Little Namskaket Marsh and Nauset Marsh in the Town of Orleans), like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater or disposal of treated effluent from municipal treatment facilities. The Town of Harwich along with other towns on Cape Cod has been among the fastest growing towns in the Commonwealth over the past two decades and does have two small wastewater treatment facilities located within the town boundaries (the town Middle/Elementary School complex and the Cranberry Pointe nursing home facility). Even so, most areas of the Herring River Estuary watershed rely almost entirely on privately maintained on-site septic treatment and disposal of wastewater. As existing and probable increasing levels of nutrients impact the coastal embayments of the Town of Harwich, water quality degradation will accelerate, with further harm to valuable environmental resources.

Fortunately for the resource protection of the Herring River Marsh, its function as a tidal wetland system makes it more tolerant of watershed nitrogen inputs than coastal open-water embayments, like nearby Allens Harbor or Wychmere Harbor. The greater sensitivity of embayments versus wetlands results from their lower rates of water turnover, the fact that there is limited to no exposure of the sediments to the atmosphere at low tide (like the marsh plain), and the fact that these systems have evolved under much lower levels of productivity and organic matter loading than wetlands. For example, the organic carbon content of New England Salt Marsh vegetated sediments can frequently reach 20%, while embayment sediments are generally in the 1%-5% range. Yet another difference between system types is that oxygen depletion in the creeks of *pristine* wetlands can normally occur on summer nights, while embayment bottom waters such as in the lower portion of the Herring River between the inlet and Route 28 can become hypoxic generally as a result of *eutrophic* conditions.

Some additional insight into the nitrogen response by salt marshes can be garnered from long-term chronic nitrogen addition experiments. These have been conducted at multiple sites along the Atlantic coast and specifically in a nearby New England salt marsh, Great Sippewissett Marsh (West Falmouth, MA). This latter project was started by WHOI scientists in 1970 and has been overseen solely by current SMAST Staff since 1985. These studies reveal that nitrogen additions to low marsh (*Spartina alterniflora*) and high marsh (*Spartina patens, Distichlis spicata*) areas, typically results in increased plant production and biomass and secondary production as well. Nitrogen dynamics have been quantified, which show that as nitrogen is added the initial increased nitrogen available is taken up by the plants, but this plant demand is rapidly satisfied and additional load is denitrified *in situ* by soil bacteria. In the Great Sippewissett Marsh fertilization experiments the denitrification capacity of the sediments has not been exhausted in 30 years of N additions and at levels about 7 times the natural background N input (75.6 g N m^{-2} each growing season).

Salt marsh creek bottoms and creek banks (such as those found in the Herring River Marsh system upgradient of Route 28) have developed under nutrient and organic matter rich conditions, as have the organisms that they support. It is the creek bottoms rather than the emergent marsh which are the primary receptors of increased watershed derived nitrogen in Cape Cod salt marshes. Watershed nitrogen predominantly enters these salt marshes through groundwater or headwater streams, as is the case in the eastern and western portions of the Herring River system. Both surface and groundwater entry focuses on the tidal channels. Even groundwater entry through seepage at the upland interface is channeled to creek bottoms. As the tide ebbs in these New England salt marshes (like the tidal creeks in the upper portions of the Herring River Marsh system) the freshwater inflow “freshens” the waters and the nitrogen levels in the tidal creeks increase due to the nitrogen entry from the watershed. At low tide the nitrogen levels in the tidal creeks are dominated by watershed inputs.

Since the predominant form of nitrogen entering from the watershed is inorganic nitrate, the effect on the creek bottom is to stimulate denitrification, hence nitrogen removal. For example, in a salt marsh in West Falmouth Harbor, Mashapaquit Creek, ~40% of the entering watershed nitrogen is denitrified by the creek bottom sediments on an annual basis. This stimulation of denitrification does not negatively affect the salt marsh, but does result in a reduction of nitrogen loading to the adjacent nitrogen sensitive coastal waters such as the lower portion of the system and the nearshore waters of Nantucket Sound. However, analysis by MEP Staff of salt marsh areas receiving wastewater discharges indicates that at very high nitrogen loads (inputs relative to tidal flushing), macroalgal accumulations can occur. These accumulations are generally found in the creek bottoms and flats and also may drift and settle on the creek banks. Large macroalgal accumulations in tidal creeks can cause impairment of benthic animal communities. In the latter case, negative effects on creek bank grasses can occur, which may lead to bank erosion and negative effects on organisms. A part of the focus of the present MEP analysis of the Herring River Marsh System, relates to potential macroalgal issues.

As the primary stakeholder to the Herring River Marsh system, the Town of Harwich and its citizens have been active in promoting restoration of this and the other embayment systems of the town (Saquatucket Harbor, Wychmere Harbor, Allens Harbor and in a collaborative manner Pleasant Bay). This local concern has also led to the conduct of several studies (see Section II) to support restoration and the Town is presently undertaking the development of a Comprehensive Wastewater Master Plan in order to implement a unified nitrogen management scheme in the town that would be restorative of its coastal resources. To this end, the Town through the efforts of its Harbormaster Office, has been active in the development of the needed minimum of three years of water quality baseline data for entry into the Massachusetts Estuaries Project. In 2001, a nitrogen related water quality monitoring program was established in all the coastal embayments of the Town of Harwich. The Town managed Water Quality Monitoring Program was provided technical assistance by the Coastal Systems Program at SMAST-UMD and over the past several years has been extended beyond the minimum three years in order to improve the calibration and validation of the MEP hydrodynamic and water quality models. This water quality monitoring effort provides the quantitative watercolumn nitrogen data (2001-2011) required for the implementation of the MEP's Linked Watershed-Embayment Approach used in the development of nutrient thresholds. Moreover, water quality data generated by the Town of Harwich Water Quality Monitoring Program is consistent with that generated by the Towns of Yarmouth, Dennis, Orleans and Chatham as well as the Pleasant Bay Alliance making the data for all the systems in Harwich cross comparable to other data set from systems in adjacent towns thereby facilitating inter-system comparisons.

The common focus of the water quality monitoring effort undertaken by the Town of Harwich has been to gather site-specific data on the current nitrogen related water quality throughout the Herring River Estuary as well as Saquatucket Harbor, Wychmere Harbor and Allens Harbor and portions of Pleasant Bay as they exist within the Town of Harwich. These data were then utilized to determine the relationship between observed water quality and watershed nitrogen loads. This multi-year effort has provided the baseline information required for determining the link between upland loading, tidal flushing, and estuarine water quality. The combined water quality data sets from the Harwich Water Quality Monitoring Program form a baseline from which to gauge long-term changes as watershed nitrogen management moves forward. This data has already proven to be of high quality and adequate for the development of management thresholds for the Town's coastal systems, as was demonstrated in the MEP nitrogen threshold analysis completed for the Pleasant Bay System (Howes et al. 2006) and the Town's three harbors (Howes et al. 2010). The Harwich Water Quality Monitoring Program efforts allowed the MEP to prioritize all of the Harwich systems for the next step in the restoration/protection and management process.

The critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to develop and implement management alternatives needed by the Town of Harwich for estuarine restoration/protection. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff and volunteers over many years, most notably within the Harbormaster Department. The modeling tools developed as part of this program provide the quantitative information necessary for the Town of Harwich to develop and evaluate the most cost effective management alternatives to restore these three coastal resources.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

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

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

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

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

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

The major Project goals are to:

- develop a coastal TMDL working group for coordination and rapid transfer of results,
- determine the nutrient sensitivity of each of the 89 embayments in Southeastern MA
- provide necessary data collection and analysis required for quantitative modeling,
- conduct quantitative TMDL analysis, outreach, and planning,
- keep each embayment model available to address future regulatory needs.

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

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

The Linked Model has been applied for watershed nitrogen management in ca. 33 embayments throughout Southeastern Massachusetts. In these applications it has become clear that the Linked Model Approach's greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing "what if" scenarios for evaluating watershed nitrogen management options.

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

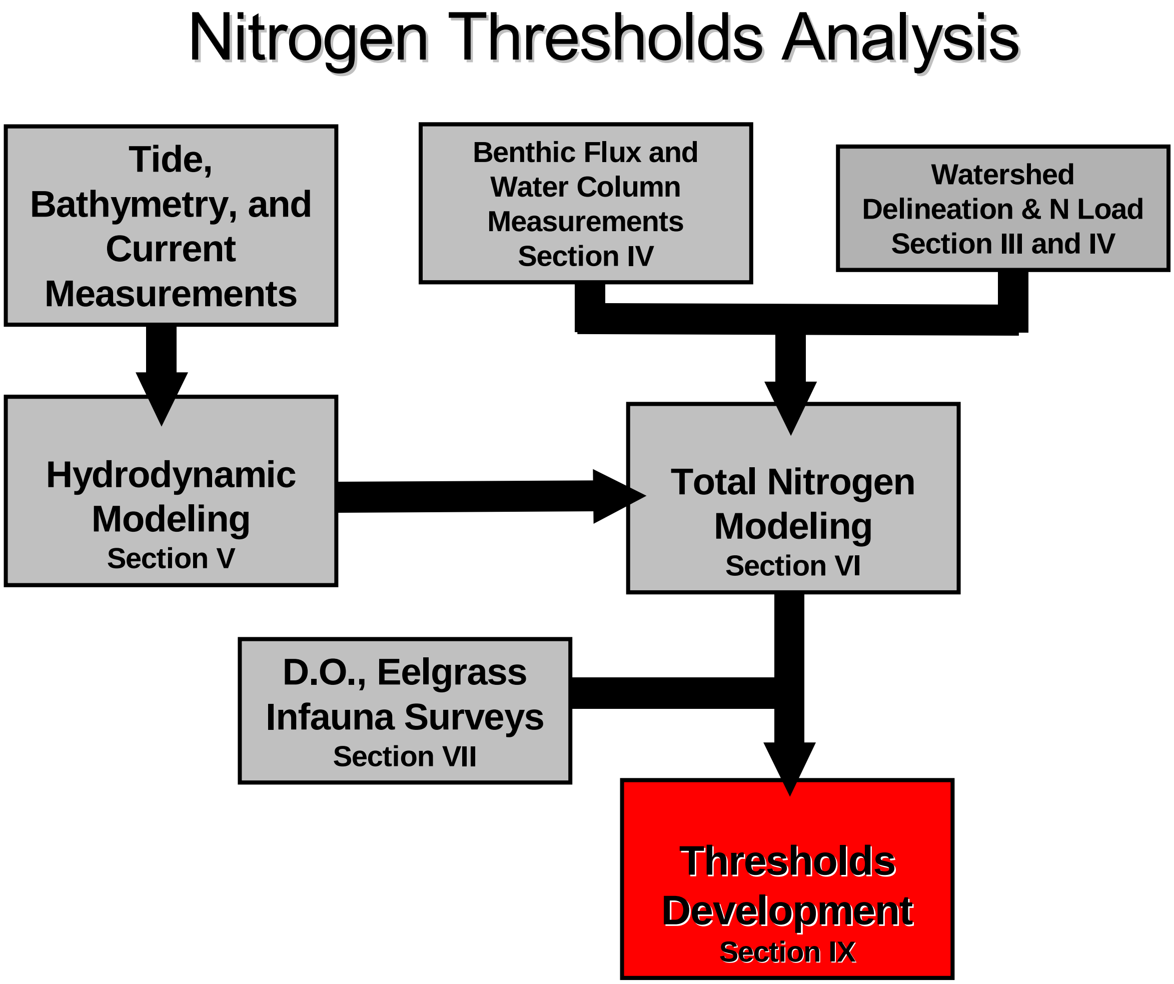

Figure I-3. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach.

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

- Monitoring - multi-year embayment nutrient sampling
- Hydrodynamics -
    - embayment bathymetry
    - site specific tidal record
    - current records (in complex systems only)
  - hydrodynamic model
- Watershed Nitrogen Loading
    - watershed delineation
    - stream flow (Q) and nitrogen load
    - land-use analysis (GIS)

  - watershed N model
- Embayment TMDL - Synthesis
  - linked Watershed-Embayment N Model
  - salinity surveys (for linked model validation)
  - rate of N recycling within embayment
  - D.O record
  - Macrophyte survey
  - Infaunal survey

I.2 SITE DESCRIPTION

The Herring River Marsh estuary, associated primarily with the Town of Harwich along with Dennis and Brewster for portions of the upper watershed to the system is among the larger of the coastal salt marsh systems of Cape Cod. The system is situated on the southern shore of Cape Cod and exchanges tidal waters with Nantucket Sound through a single tidal inlet. The inlet, though armored and stabilized by jetties, can be significantly affected by longshore sand transport (east to west), where shoaling can impede hydrodynamic exchange at the mouth of the system thereby necessitating periodic maintenance dredging of the inlet for navigation. While the navigational channel is maintained, shoals are abundant in the vicinity of the inlet and depths can vary significantly. Depths throughout the Herring River system vary due to the tidal salt marsh characteristics in the upper portions of the system in combination with the moderate tidal range in this area of Nantucket Sound. At low tide portions of the upper marsh become nearly dry tidal creeks.

The present configuration of the Herring River Salt Marsh System results from a combination of glacially dominated geologic processes including the deposition of glacial outwash deposits and tidal flooding of post-glacial valleys formed primarily by post-glacial surface water erosion of the outwash. This estuarine system is a relatively "young" coastal feature that required significant post glaciation sea-level rise and the formation of the barrier beach, occurring on the order of 2500-4000 years b.p. Tidal exchange with Nantucket Sound is through a single inlet through the barrier beach. The nearshore shoals and beach are very dynamic geomorphic features, due to the influence of littoral transport processes. These processes may periodically affect the health of this estuary through changes in hydrodynamics driven by sediment transport and deposition in and around the tidal inlet (see Section V). To the extent that the inlet becomes restricted and tidal flushing is reduced, nitrogen loading impacts will be magnified over present conditions. Equally important, to the extent that tide range may become reduced, the health and productivity of the emergent salt marsh would be reduced. Any long term habitat management plan for the Herring River Estuary must recognize the importance of inlet dynamics and include options to maintain maximal tidal exchange.

Similar to nearby salt marshes in the Town of Orleans (Namskaket and Little Namskaket marshes), the Herring River Marsh is a shallow coastal estuary dominated by salt marsh, as well as being located within a watershed that includes glacial outwash plain (Harwich Outwash Plain) deposits. These subsurface formations consist of material deposited after the retreat of the Laurentide Ice sheet ~15,000 years ago. These deposits, which form the present aquifer soils, are highly permeable and vary in composition from well sorted medium sands to course pebble sands and gravels (Oldale, 1992). As such, direct rainwater run-off is typically rather low and most freshwater inflow to these estuarine systems is via groundwater discharge or groundwater fed surface water flow. Freshwater inflow to the Herring River Estuary enters through 2 direct surface water discharges and enters through groundwater seepage along the estuarine margins and creek bottoms. Unlike many other tidal wetland systems, the Herring

River supports a deeper main tidal channel (tidal river) in the lower part of the system that functions more like an open water embayment than a salt marsh.

The Herring River Marsh acts as a mixing zone for terrestrial freshwater inflow and saline tidal flow from Nantucket Sound (31.6 ppt). Salinity levels vary with the volume of freshwater inflow as well as the effectiveness of tidal exchange. In the lower portion of the system, given the large tidal flows and volumetric exchange, there is presently only minor dilution of salinity throughout the area of the estuary below Route 28 at high tide (26-30 ppt). However, the system is a tidal wetland and as such, the elevation of the tidal creek bottoms is generally higher than the low tide elevation in the adjacent Bay (e.g. the creeks drain nearly completely at low tide). The result is that at low tide in the upper reaches of the system, the salinity of the out flowing water from the western and eastern portions of the system is brackish (7-12 ppt), due to the dominance of the freshwater inflow. As a result salinity variations of the creek waters in the upper marsh are very large with the range decreasing moderately toward the tidal inlet. Organisms associated with these creeks have developed strategies for dealing with these large salinity variations.

Overall, the Herring River Marsh system, is typical of a large New England tidal marsh system, with the lower regions of predominantly salt marsh dominated by a central tidal creek and the marsh plain colonized by *Spartina alterniflora* (low marsh) and *Spartina patens and Distichlis spicata* (high marsh). The upper regions, furthest from the tidal inlet show the influence of the freshwater inflows from the surrounding watershed with species grading to brackish marsh dominated by *Phragmites* finally shifting to freshwater marsh dominated by *Typha* and other freshwater species on the marsh plain (Figure I-2). The result is that the upper wetland portion of this estuary has a relatively high tolerance for nitrogen inputs from its watershed, while the lower tidal river portion is the most sensitive region to nitrogen enrichment.

Overall, Herring River Marsh appears presently to be a healthy functioning New England salt marsh as noted during the MEP Technical Team's field survey. Nitrogen levels within the tidal creeks do show evidence of impairing the resource. However, the primary ecological concern relative to loss of Herring River Estuary's resources is degradation of its tidal river portion, resulting from nutrient enrichment (and possibly restriction of tidal flushing due to coastal processes). Loading of the nitrogen has been increasing over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to this and other outer Cape estuaries, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater or disposal of treated effluent from municipal treatment facilities. The Town of Harwich has been among the fastest growing towns in the Commonwealth over the past two decades and does have centralized wastewater treatment though not all the areas of the town are serviced and as such there is still a large dependence on septic systems. As levels of nitrogen loading to coastal systems continue to increase, concern has grown in outer Cape Cod Towns over associated nutrient impacts. However, as noted above, tidal wetlands are relatively insensitive to degradation by nitrogen inputs from the surrounding watershed and at present the ebbing tidal waters carry only moderate nitrogen levels for Cape Cod wetlands (~0.8 mg N L^{-1}). This results from the structure of the salt marsh within the upland hydrologic system and the natural nitrogen processing by these systems. In addition, the plants and animals within salt marshes have adapted to the high organic matter levels within the marsh sediments and associated waters and the associated biogeochemical effects. Critical to the MEP analysis of this salt marsh system is partitioning of salt marsh and tidal river portions of the estuary (main tidal channel between inlet and Route 28).

I.3 NITROGEN LOADING

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

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

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

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

I.4 WATER QUALITY MODELING

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

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

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS. The groundwater model was originally developed for the Cape Cod Sagamore lens and was then paramaterized and calibrated for watershed delineation of contributing area to drinking water supply wells, freshwater ponds, wetland and estuaries situated within the Monomoy aquifer, designated by the MEP. Almost all nitrogen entering the Herring River Estuary is transported by freshwater, predominantly through a groundwater flow path (either directly to estuary or to a stream). Concentrations of total nitrogen and salinity of Nantucket Sound source waters and the wetland system itself was taken from the water quality monitoring program run by the Town of Harwich (associated with the Coastal Systems Program at SMAST). Measurements of current salinity and nitrogen and salinity distributions throughout the estuarine waters of this marsh system were used to calibrate and corroborate the water quality model (under existing loading conditions).

I.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Herring River Estuary in the Town of Harwich. A review of existing studies related to habitat health or nutrient related water quality is provided in Section II with a more detailed review of prior hydrodynamic investigations in Section V. 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 nitrogen recycling associated with the bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Section IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Town of Harwich and Cape Cod Commission data. Nantucket Sound water quality data was derived from an analysis of available nearshore Harwich monitoring data and summer sampling of the Sound just offshore from the Herring River inlet (Section IV and VI). Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data (conducted by municipalities) as discussed in Section 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 each embayment was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of each embayment in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined threshold for restoration in a given salt pond. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for the marsh system.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

Nutrient additions to aquatic systems cause shifts in a series of biological processes that can result in impaired nutrient related habitat quality. Effects include excessive plankton and macrophyte growth, which in turn lead to reduced water clarity, organic matter enrichment of waters and sediments with the 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 dependant upon these highly productive estuarine systems as a habitat and food resource during migration or during different phases of their life cycles. This process is generally termed “eutrophication” and in embayment systems, unlike in shallow lakes and pond, it is not a necessarily a part of the natural evolution of a system.

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

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

A number of studies relating to nitrogen loading, hydrodynamics and habitat health have been conducted within the Herring River System over the past decade.

Town of Harwich Water Quality Monitoring Program - The Town of Harwich, while being actively engaged in the study and management of municipal infrastructure and natural resources, committed early on to gathering baseline water quality monitoring data from its 4 Nantucket Sound estuaries in support of the MEP, as well as Pleasant Bay, which is shared with the Towns of Orleans, Chatham and Brewster. While all the towns collaborate on sampling of Pleasant Bay via the Pleasant Bay Alliance, each Town operates a Water Quality Monitoring Program collecting water quality data on all of that town’s embayment systems, as is the case with the Herring River. For Harwich and this system specifically, the focus of the effort has been to gather site-specific data on the current nitrogen related water quality to support evaluations of observed water quality and habitat health. The Harwich Water Quality Monitoring Program for Herring River developed the baseline data from sampling stations distributed throughout the tidal reaches of the system (Figure II-1). Water quality monitoring of the Herring River System has been a coordinated effort between the Town of Harwich Harbor Master Department and The Coastal Systems Program at SMAST-UMD. The water quality monitoring program was initiated in 2001 with support from the Town of Harwich and has continued uninterrupted through the summer of 2011. As with other monitoring programs in the study region, monitoring reduced its sampling effort after six years of baseline water quality data were captured, with continued summer time monitoring of key stations thereby generating an extremely robust water quality database for the calibration and corroboration of the MEP water quality model. Additionally, as future remediation plans for these systems are implemented throughout the Town of Harwich, continued monitoring will provide quantitative information to the Town of Harwich relative to the efficacy of remediation efforts.

The joint Town of Harwich / CSP Water Quality Monitoring Program provided the quantitative water column nitrogen data (2001-2011) required for the implementation of the MEP's Linked Watershed-Embayment Approach. The MEP effort also builds upon previous watershed delineation and land-use analyses, the previous embayment hydrodynamic modeling and historical eelgrass surveys. This information is integrated with MEP higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Herring River Estuary. The MEP has incorporated all appropriate data from all previous studies to enhance the determination of nitrogen thresholds for this large wetland dominated estuary and to reduce costs to the Town of Harwich.

A Baseline Hydrodynamic and Water Quality Investigation of the Lower Herring River Harwich, MA, Horsely & Witten Inc. June 2000. - An initial investigation of tide range and stage changes in the upper and lower Herring River was undertaken relative to the ability of the estuary to assimilate nitrogen and to gauge potential water quality impacts. A series of estuarine flushing models were used to evaluate residence time including a tidal prism analysis. The analysis focused primarily on hydrodynamics and as an initial effort was not tasked to develop water quality models or land-use nitrogen loading models for prediction of how changes in watershed loading would quantitatively increase or decrease watercolumn nitrogen concentrations.

Cape Cod Coastal Embayment Project - Cape Cod Commission September 1998. The Cape Cod Commission completed a study that included the first comprehensive Cape-wide estuary watershed delineations, a tidal flushing study of one estuary, and review of watershed nitrogen loading (both existing and future buildout) for 10 selected systems. The study also included two planning metrics for prioritizing future characterization efforts based on readily available Cape-wide information. The nitrogen sensitivity metric based on four criteria: 1) ratio of estuary surface area to width of ocean inlet, 2) ratio of estuary surface area to saltwater wetland area, 3) branching ratio, and 4) tidal range. Herring River was evaluated as 23 out of 52 based on this metric. The planning/remediation metric was based on four criteria: 1) potentially developable acreage in the watershed, 2) the % of watershed that was developable, 3) the % of open shellfish beds, and 4) the ratio of existing watershed nitrogen loading to the surface area of the estuary. Herring River was evaluated as 32 of 52 based on this metric; closer to the "planning to protect" portion of the scale than the "need to remediate" portion.

Figure II-1. Town of Harwich Water Quality Monitoring Program for Herring River. Estuarine water quality monitoring stations sampled by the Town and volunteers. The stream water quality stations associated with the east and west branches of the upper river (depicted in Chapter 4) were sampled weekly by the MEP.

History of the Herring River, Harwich Natural Resources Department Special Report - G. M. Tunison (http://www.vsv.cape.com/~harharb/tunnison1.html). A historical retrospective of the Herring River Estuary and the various activities and changes in land-use of its watershed was developed. It was made clear that water associated industries have significantly altered the Herring River System over the past 350 years. For example, the West Reservoir is a man-made waterbody created by damming the upper freshwater river to create a freshwater pond that still exists today, ~100 years later. In addition, cranberry agriculture was developed both along the upper river and associated with freshwater ponds, used for flooding for harvest and frost control. The river itself was valued as a herring and alewife fishery, the preservation of which is of present concern to managers.

Regulatory Assessments of Herring River Marsh Resources - In addition to locally generated studies, Herring River Marsh is part of the Commonwealth's environmental surveys to support regulatory needs. The Herring River Estuary contains a variety of natural resources of value to the citizens of Harwich 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 Herring River Estuary these include:

- Designated Shellfish Growing Area – MassDMF (Figure II-2)
- Shellfish Suitability Areas – MassDMF (Figure II-3)
- Estimated Habitats for Rare Wildlife and State Protected Rare Species – NHESP (Figure II-4)
- Mouth of River designation - MassDEP (Figure II-5)
- Anadromous Fish Runs - MassDMF (Figure II-6)

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 Herring River Marsh Estuarine System. The MEP has incorporated appropriate data from pertinent previous studies to enhance the determination of nitrogen thresholds for the Herring River Marsh System and to reduce costs to the Towns of Harwich.

Figure II-2. 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 or "activities", such as the location of marinas. Bacterial contamination in tidal wetland areas is frequently associated with wildlife.

Figure II-3. Location of shellfish habitat suitability areas within the Herring River Estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean that the specific shellfish is "present", rather that the habitat should support its colonization and growth.

Figure II-4. Estimated Habitats for Rare Wildlife and State Protected Rare Species within the Herring River Estuary as determined by – NHESP.

Figure II-5. Regulatory designation of the mouth of the Herring River relative to the Massachusetts Rivers Act (MassDEP). Upland adjacent the "river front" inland of the mouth of the river has restrictions specific to the Act.

Figure II-6 Anadromous fish runs within the Herring River Estuary as determined by Mass Division of Marine Fisheries. The red diamonds show areas where fish were observed. The uppermost sites are within the major freshwater ponds Hinckley's Pond, Long Pond and Seymours Pond, smaller ponds and the man-made basin of West Reservoir. These freshwater basins have a long history of supporting herring and alewife habitat. The run also represents a surface water pathway for nutrients to enter the Herring River System.

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 relative to the Herring River Estuary. The USGS has a long history of developing regional models for the six-groundwater flow cells on Cape Cod. Through the years, advances in computing, lithologic information from well installations, water level monitoring, stream flow measurements, and reconstruction of glacial history have allowed the USGS to update and refine the groundwater models for Cape Cod. The MODFLOW and MODPATH models utilized by the USGS to organize and analyze the available data use up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation, surface water/groundwater interaction, groundwater travel time, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including the Herring River Estuary. This estuary is situated along the southern edge of Cape Cod, exchanging tidal waters with and bounded by Nantucket Sound. The Herring River watershed includes portions of the Towns of Harwich, Brewster, and Dennis.

In the present investigation, the USGS applied its groundwater modeling approach to define the watershed or contributing area to the Herring River Estuary under evaluation by the MEP Project Team. The Herring River Estuary receives freshwater through a number of groundwater-fed freshwater streams, including the Herring River, which extends from The Reservoir (also called West Reservoir) to Hinckleys Pond. Hinckleys Pond has additional surface water connections to Long and Seymour ponds. Further modeling of the estuary watershed was undertaken to sub-divide the overall watershed into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion of the embayment system, (b) defining contributing areas to major freshwater aquatic systems which generally attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining 10 year time-of-travel distributions within each sub-watershed as a procedural check to gauge the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in the overall Monomoy groundwater flow cell. Model assumptions for calibration of the Herring River Estuary groundwater model included surface water discharges measured as part of the MEP stream flow program (2004 to 2006) and historic stream gage information.

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

III.2 MODEL DESCRIPTION

Contributing areas to the Herring River Estuary and its various subwatersheds, such as Hinckleys, Flax, and Long Ponds, were delineated using a regional model of the Monomoy Lens flow cell (Walter and Whealan, 2005). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al*., 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, bogs and coastal water bodies. This approach was used to determine the contributing areas to the Herring River system and its subwatersheds and also to determine portions of recharged water that may flow through freshwater ponds and streams prior to discharging into the coastal water bodies.

The Monomoy Flow Model grid consists of 164 rows, 220 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft (Walter and Whealan, 2005). The top of layer 8 resides at NGVD 29 with layers 1-7 stacked above and layers 8-20 below. Since water elevations are less than +40 ft in the portion of the Monomoy Lens in which the Herring River estuary resides, the three uppermost layers of the model are inactive. Layer 18 has a thickness of 40 feet and extends to 140 feet below NGVD 29, while layer 19 extends to 240 feet below NGVD 29. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics (up to 525 feet below NGVD 29); since bedrock is 400 to 500 feet below NGVD 29 in most of the Herring River area, the two lowest model layers were active in this area of the model. The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, were used to simulate the top of the water table, which varies in elevation depending on the location in the flow cell.

The glacial sediments that comprise the aquifer of the Monomoy Lens consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward sequence with sand and gravel deposits deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. The Herring River Estuary watershed is located in the Outwash Plain Deposits (Oldale, 1974). The plains materials are generally composed of sand and gravel (Oldale, *et al*., 1971). Modeling and field measurements of contaminant transport at the Massachusetts Military Reservation have shown that similar deposited materials are highly permeable (*e.g*., Masterson, *et al*., 1996). Given their high permeability, direct rainwater run-off is typically rather low for this type of watershed system. Lithologic data used to determine hydraulic conductivities used in the groundwater model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Town and stream flow data collected in 1989-1990 as well as in 2004-05.

The Monomoy Flow Cell groundwater model simulates steady state, or long-term average, hydrologic conditions including a long-term average recharge rate of 27.25 inches/year and the pumping of public-supply wells at average measured annual withdrawal rates for the period

1995-2000 with a 15% consumptive loss. This recharge rate is based on the most recent USGS information. Large withdrawals of groundwater from pumping wells may have a significant influence on water tables and watershed boundaries and therefore the flow and distribution of nitrogen within the aquifer. After accounting for the consumptive loss, water withdrawn from the modeled aquifer by public drinking water supply wells is evenly returned within designated residential areas utilizing on-site septic systems.

III.3 HERRING RIVER ESTUARY CONTRIBUTORY AREAS

The refined watershed and sub-watershed boundaries for the Herring River embayment system, including Long, Hinckleys, Seymour and Flax ponds and West Reservoir, the fresh portion of the Herring River, and the estuarine sub-basins (Figure III-1) were determined by the United States Geological Survey (USGS). Model outputs of watershed boundaries were “smoothed” to (a) correct for the model grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, (c) to include water table data in the lower regions of the watersheds near the coast (as available), (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 refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. The MEP sub-watershed delineations for these watersheds also include 10 year time-of-travel boundaries. Overall, 29 sub-watershed areas were delineated within the Herring River study area for land-use nitrogen loading and freshwater recharge/discharge analysis.

Table III-1 provides the daily freshwater discharge volumes for each of the 29 subwatersheds comprising the watershed to the Herring River Estuary as calculated by the groundwater model and these volumes were used to assist in the development of the estuarine hydrodynamic model salinity calibration of the estuarine water quality model and to determine hydrologic turnover in the lakes/ponds, as well as for comparison to directly measured surface water discharges. The overall estimated freshwater inflow into the Herring River Estuary from its MEP watersheds is 70,509 m3/d.

The MEP watershed delineation described above is the second watershed delineation completed in recent years for the Herring River Estuary. Figure III-2 compares the delineation completed under the current effort with the delineation completed by the Cape Cod Commission as part of the Coastal Embayment Project (Eichner, *et al*., 1998). The CCC delineation was developed based on regional water table measurements collected from available well data over a number of years and normalized to average conditions. This analysis incorporated included water table data collected in Johnson and Davis (1988). The Commission’s delineation was incorporated into the Commission’s regulations through the three versions of the Regional Policy Plan (CCC, 1996, 2001, and 2009).

The MEP watershed area for the Herring River system as a whole is roughly the same area as the 1998 CCC delineation (9,185 acres vs. 9,171 acres, respectively). The areas that are included in the delineations are slightly different, largely due to the internal subwatershed refinements included in the MEP delineations. For example, the MEP delineation of the West Reservoir expands its watershed compared to the CCC delineation; this expansion brings White Pond and Elbow Pond into the MEP Herring River watershed. The MEP watershed delineation also includes interior sub-watersheds to various components of the Herring River system, such as ponds and public water supply wells that were not fully included in the CCC delineation, as well as subwatersheds to streams gauged during the MEP: the West Reservoir and near

Lothrup Road (Section IV.2). The inclusion of these refinements is another benefit of the update of the regional groundwater model (Walter and Whealan, 2005).

The evolution of the watershed delineation for the Herring River estuary system has allowed increasing accuracy as each new version adds new hydrologic data to that previously collected; the groundwater model allows all this data to be organized and to be brought into congruence with data from 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 the use of this model for the evaluation of nitrogen management alternatives. 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 downgradient estuary. The MEP watershed delineation was used to develop the watershed nitrogen loads to each of the aquatic systems and ultimately to the estuarine waters of the Herring River Estuary (Section V.1).

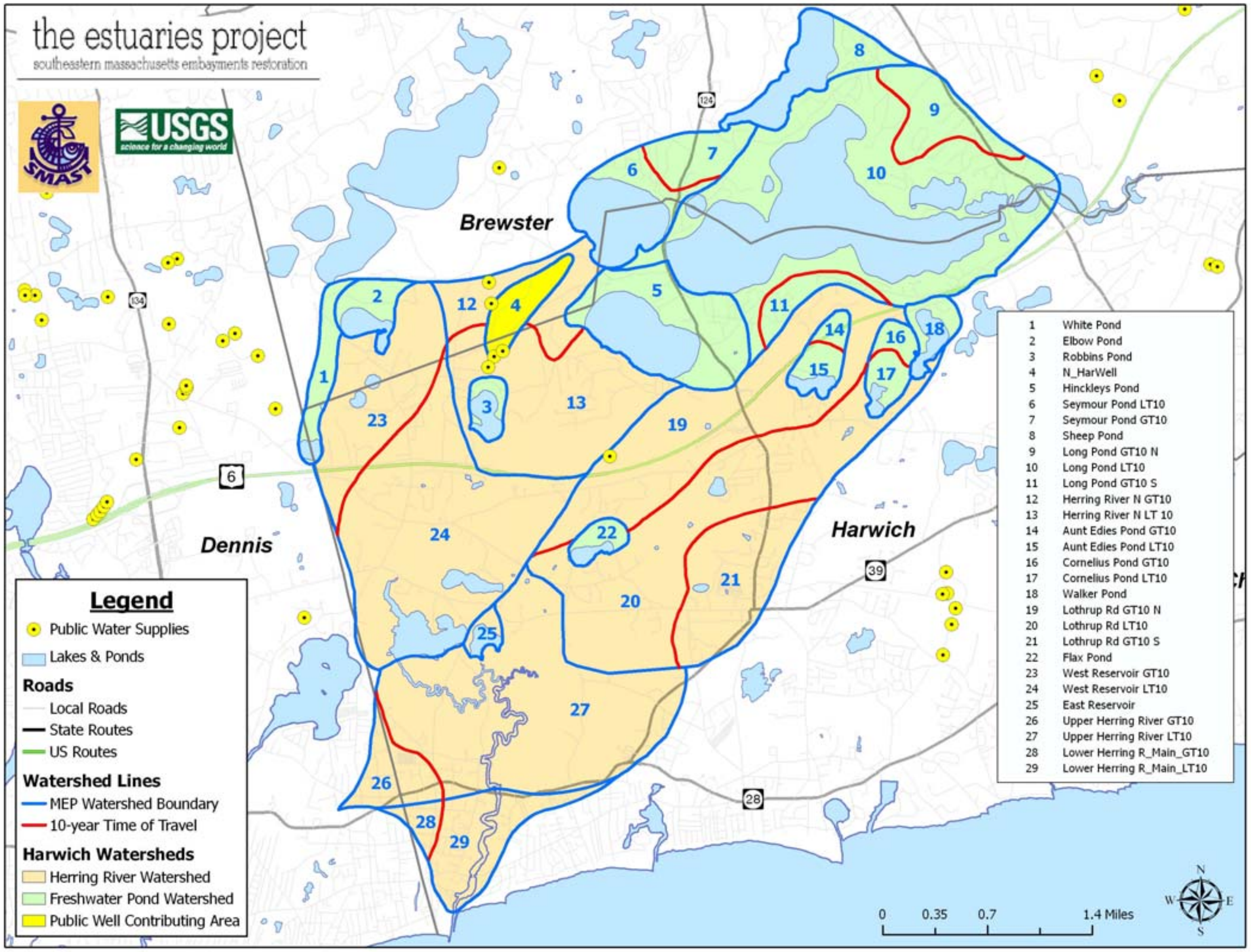

Figure III-1. Watershed delineation for Herring River estuary. Subwatershed delineations are based on USGS groundwater model output with modifications to better address pond and estuary shorelines and MEP stream gauge measurements. Ten-year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the watershed names and are indicated by the red lines (above). Sub-watersheds within the estuary portion of the watershed (*e.g.*, Upper Herring River) were selected based upon functional estuarine sub-units in the water quality model (see Section VI).

Table III-1. Daily groundwater discharge from each of the sub-watersheds in the watershed to the Herring River estuary, as determined from the regional USGS groundwater model.

Watershed	#	Watershed Area (acres)	Discharge	
			m^3/day	ft^3/day
White Pond	1	106	810	28,613
Elbow Pond	2	97	744	26,268
Robbins Pond	3	57	438	15,464
N_HarWell	4	98	755	26,645
Hinckleys Pond	5	435	3,336	117,804
Seymour Pond LT10	6	258	1,979	69,886
Seymour Pond GT10	7	131	1,003	35,418
Sheep Pond	8	214	1,646	58,126
Long Pond GT10 N	9	185	1,418	50,061
Long Pond LT10	10	1,644	12,617	445,555
Long Pond GT10 S	11	85	650	22,952
Herring River N GT10	12	209	1,601	56,554
Herring River N LT 10	13	615	4,722	166,764
Aunt Edies Pond GT10	14	28	216	7,614
Aunt Edies Pond LT10	15	64	491	17,348
Cornelius Pond GT10	16	38	292	10,304
Cornelius Pond LT10	17	62	476	16,792
Walker Pond	18	77	591	20,877
Lothrup Rd GT10 N	19	729	5,596	197,605
Lothrup Rd LT10	20	903	6,927	244,613
Lothrup Rd GT10 S	21	367	2,814	99,373
Flax Pond	22	51	391	13,808
West Reservoir GT10	23	440	3,373	119,109
West Reservoir LT10	24	1,076	8,259	291,662
East Reservoir	25	36	276	9,744
Upper Herring River GT10	26	141	1,083	38,261
Upper Herring River LT10	27	1,085	8,329	294,130
Lower Herring R_Main_GT10	28	46	355	12,554
Lower Herring R_Main_LT10	29	229	1,754	61,942
HERRING RIVER SYSTEM TOTAL			70,509	2,490,001

Notes: 1) discharge volumes are based on 27.25 in of annual recharge over the watershed area; 2) these flows do not include direct precipitation to the surface of the estuary; 3) upgradient ponds often discharge to numerous downgradient subwatersheds including some discharge out of the Herring River system (*e.g.*, Sheep Pond), percentage of outflow is determined by length of downgradient shoreline going to each receiving subwatershed; the sum of these corrections are included in the total system recharge, but not in the recharge for individual subwatersheds.

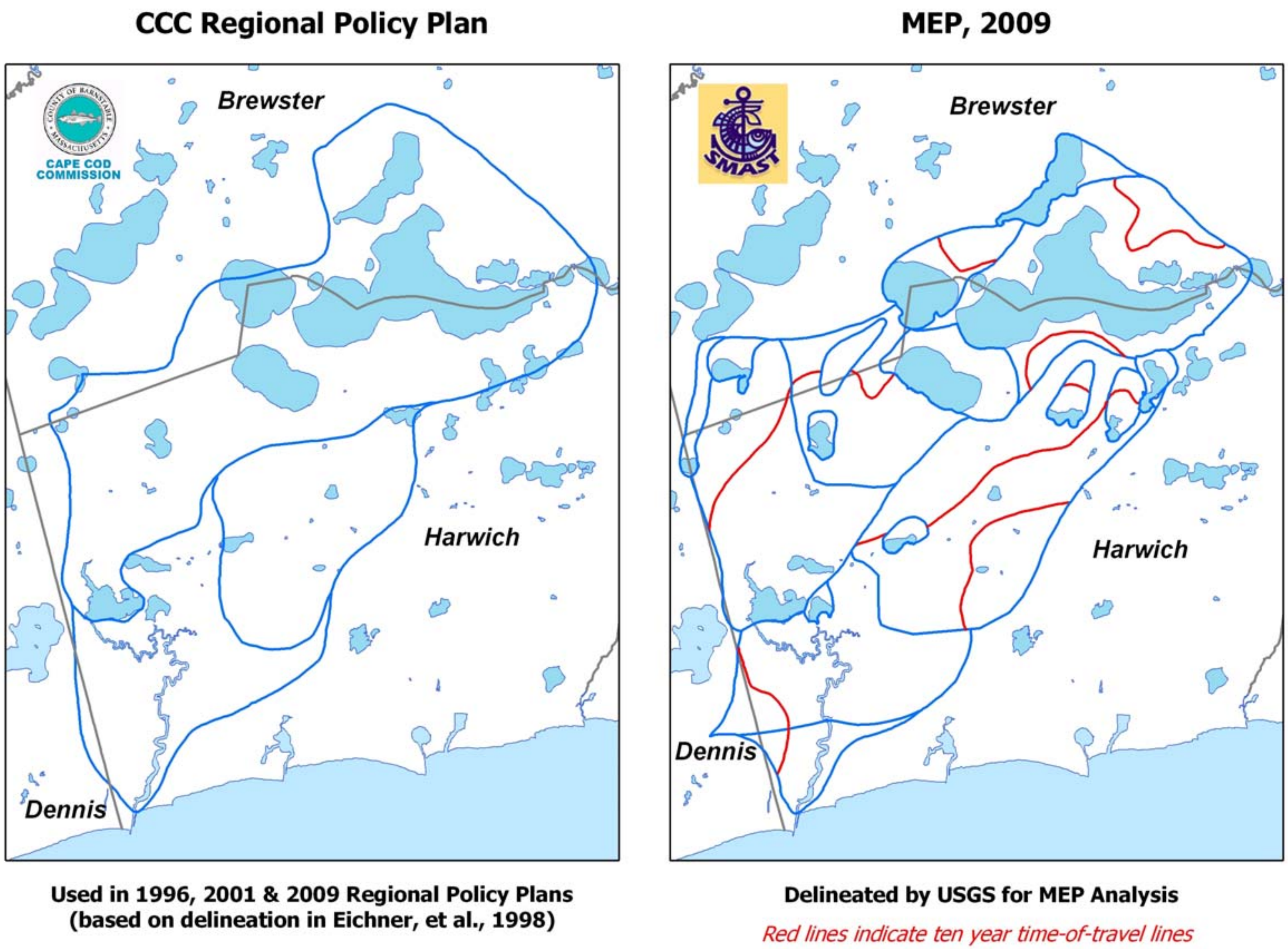

Figure III-2. Comparison of MEP Herring River watershed and subwatershed delineations used in the current assessment and the Cape Cod Commission delineation (Eichner, *et al*., 1998), which has been used in three Barnstable County Regional Policy Plans (CCC, 1996, 2001, and 2009). Changes relate to the inclusion of new data and refinements included in the USGS analysis, which primarily altered the western boundary.

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 Herring River estuary system. Determination of watershed nitrogen inputs to embayment systems requires the (a) identification and quantification of the nutrient sources and associated 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. Burial of nitrogen is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters 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 Herring River Estuary, the MEP Technical Team developed nitrogen-loading rates (Section IV.1) to each component of the estuary and its watersheds (Section III). The Herring River watershed was sub-divided to define contributing areas 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 estuarine waters in less than 10 years or greater than 10 years. A total of 29 sub-watersheds were delineated in the overall Herring River watershed, including watersheds to the following ponds: Sheep, Long, Hinckleys, Seymour, Elbow, White, West Reservoir, East Reservoir, Robbins, Aunt Edies, Cornelius, and Walker. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds and each sub-embayment of the overall estuary (see Chapter III).

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 provided by the USGS watershed model, and review of data at natural collection points, such as streams and ponds. Evaluation and delineation of ten-year time of travel zones are a regular part of the watershed analysis. To support this evaluation, ten-year time of travel sub-watersheds within the overall Herring River watershed were delineated for ponds, streams and the estuary itself. Because of the way the Herring River and its tributaries flow from the uppermost portions of the watershed, flow and nitrogen loads from the upper portions of the watershed reach the estuary within 10 years. For example, nitrogen loads from the Walkers Pond or Long Pond watershed reach the Herring River estuary in less than 10 years. MEP staff also reviewed land use development records for the age of developed properties in the watersheds. Based on the review of all this information, it was determined that 74% of its overall unattenuated watershed nitrogen load is within 10 years of groundwater time of travel to the estuary (Table IV-1). The overall result of integrating the timing of development with the groundwater travel times and the fact that most greater than 10 year time of travel watersheds have some of their nitrogen removed in transport (attenuated) 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 sub-watersheds is not significant for modeling existing conditions.

Table IV-1. Percentage of unattenuated watershed nitrogen loads in less than 10 year time of travel sub-watersheds to Herring River.

WATERSHED	LT10	GT10	TOTAL	%LT10
Name	kg/yr	kg/yr	kg/yr	
West Reservoir	18,803	4,196	22,999	82%
Lothrop Road	5,101	4,747	9,848	52%
Herring River System	29,747	11,358	41,105	72%

Notes: loads have been corrected for portions that leave the watershed from ponds discharging along the overall watershed boundary; loads do not include atmospheric loading on the estuary surface waters

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 and Ramsey, 2001) uses a land-use Nitrogen Loading Sub-Model based upon sub-watershed specific land uses and pre-determined nitrogen loading rates based on regional analyses. For the Herring River Estuary, the model used land-use data from the Towns of Harwich, Brewster, and Dennis transformed to nitrogen loads using both regional nitrogen loading factors and local watershed specific data (such as parcel-by-parcel water use or alternative septic system monitoring). 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 capable of reaching the embayment, since attenuation during transport has not yet been included.

Natural attenuation of nitrogen during transport from land-to-sea (Section IV.2) within the Herring River watershed was determined based upon a site-specific study of streamflow and assumed and measured attenuation in the upgradient freshwater ponds. Stream flow was characterized at the discharge from the West Reservoir and at the stream passing under Lothrop Road. Sub-watersheds to these stream discharge points allowed comparisons between field collected data from the streams and ponds and estimates from the nitrogen-loading sub-model. Nitrogen attenuation in individual ponds is generally estimated based on available information. Attenuation through the ponds is conservatively assumed to equal 50% unless available monitoring and pond physical data is reliable enough to calculate a pond-specific attenuation factor. Stream flow and associated surface water attenuation is included in the MEP's nitrogen attenuation and freshwater flow investigation, presented in Section IV.2.

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. In the present effort, estimates of pond nitrogen attenuation were made in 16 ponds in the Herring River Estuary watershed. All of these ponds have watersheds that were modeled by the USGS when the MEP watersheds were delineated. 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 and the direct measurement of nitrogen loading for most of the watershed area.

Based upon the evaluation of the watershed system, the MEP Technical Team used the Nitrogen Loading Sub-Model estimate of nitrogen loading for the sub-watersheds 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 Herring River 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).

IV.1.1 Land Use and Water Use Database Preparation

Since the overall watershed to the Herring River Estuary includes portions of the Towns of Harwich, Brewster, and Dennis, Estuaries Project staff obtained digital parcel and tax assessor's data from the towns to serve as a base for the watershed nitrogen loading model. Digital parcels and land use/assessors data for Harwich are from 2006, Brewster data are from 2004, and Dennis data are 2007. These land use databases contain traditional information regarding land use classifications (MassDOR, 2009) plus additional information developed by the towns. This effort was completed with the assistance from Cape Cod Commission (CCC) technical staff.

Figure IV-1 shows the land uses within the Herring River estuary watershed. Land uses in the study area are grouped into nine (9) land use categories: 1) residential, 2) commercial, 3) industrial, 4) mixed use/no land use code, 5) recreational (golf course), 6) undeveloped, 7) agricultural, 8) public service/government, including road rights-of-way, and 9) freshwater features (*e.g.* ponds and streams). These land use categories, except the freshwater features, are aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MassDOR, 2008). "Public service" in the MassDOR system is tax-exempt properties, including lands owned by government (*e.g.*, wellfields, schools, golf courses, open space, roads) and private non-profit groups like churches and colleges.

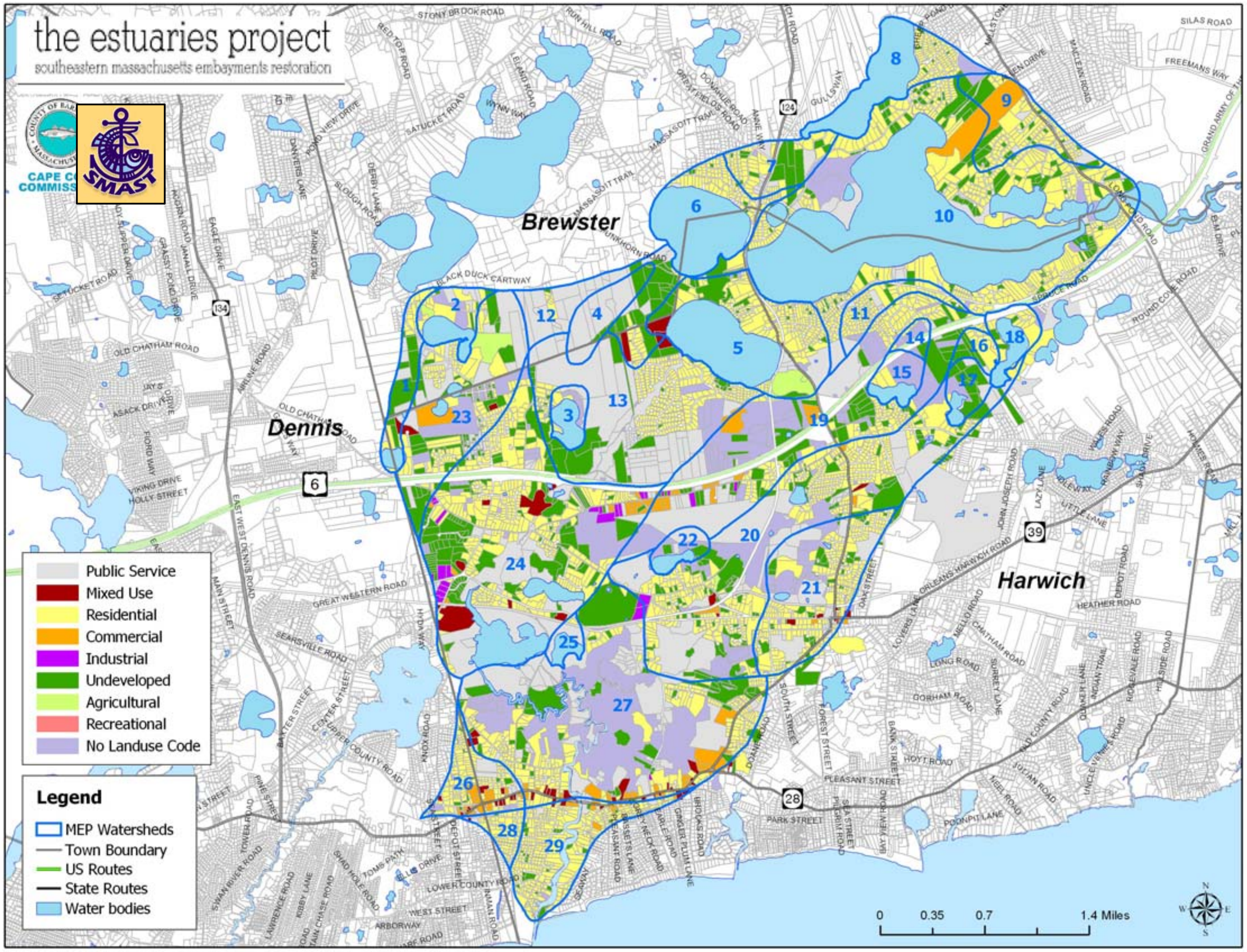

Figure IV-1. Land-use in the Herring River Estuary watershed. Watershed includes portions of the Towns of Harwich, Brewster, and Dennis. Land use classifications are based on 2006, 2004, and 2007 town assessors' records.

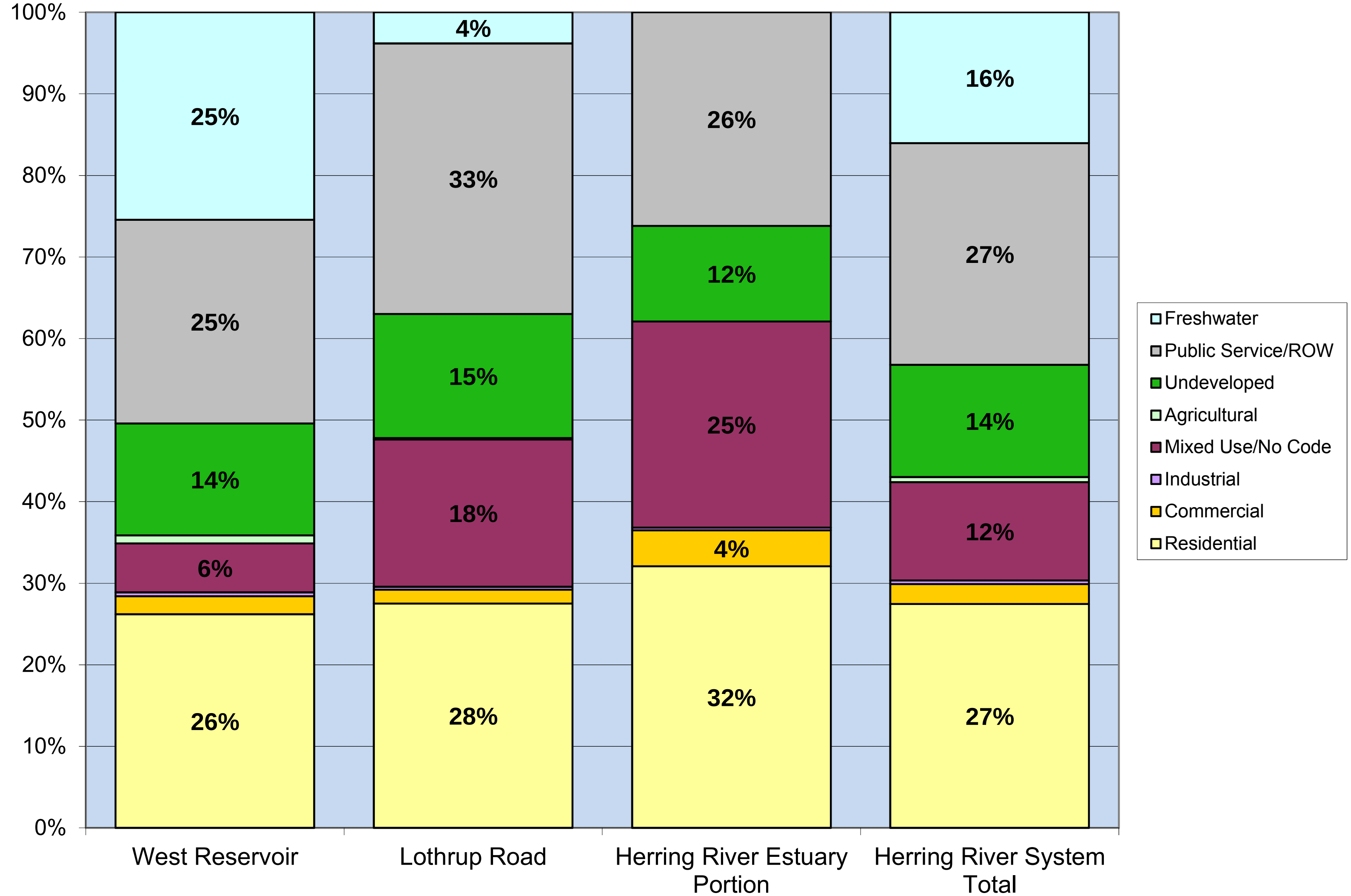

Figure IV-2. Distribution of land-uses by area within the whole Herring River watershed and three component sub-watersheds. Only percentages greater than or equal to 4% are shown. Land use categories are based on town assessor classifications.

Two land use categories, residential and public service, are the dominant land use types in the overall Herring River watershed. These are each 27% of the watershed area (Figure IV-2). Freshwater surface area is the third largest area at 16%, followed closely by undeveloped lots at 14% and mixed/unclassified uses (predominantly salt marsh areas) at 12%. Within the West Reservoir sub-watershed, freshwater ponds, residential, and public service lands all occupy approximately a quarter of the watershed area. In the Lothrop Road sub-watershed, public service (33%) is the dominant land use, which includes the Cranberry Valley Golf Course and the town Middle/Elementary School complex.

In all the sub-watershed groupings shown in Figure IV-2, residential parcels are the dominant parcel type, ranging between 65% and 71% of all parcels and 66% of all the parcels in the whole Herring River watershed. Single-family residences (MassDOR land use code 101) comprise 88% to 96% of the total residential parcels in the sub-watersheds and 94% of the residential parcels throughout the whole Herring River watershed.

In order to estimate wastewater flows within the Herring River study area, MEP staff obtained parcel-by-parcel water use data from the three towns in the watershed. Four years (2004 through 2007) of water use information was obtained from the Town of Harwich Water Department (personal communication, Craig Weigand, Superintendent, 4/08). Three years of water use data (2002 through 2004) were obtained from the Brewster Water Department (personal communication, Paul Hicks, Superintendent). And 2007 water use data was obtained from the Dennis Water District (personal communication, Sheryl McMahon, Treasurer, 2/08). Brewster is utilizing water use from 2009 for their current CWMP efforts (CDM, 2011). The water use data was linked to the respective town parcel databases by the Cape Cod Commission GIS Department staff.

Measured water use is used to estimate wastewater-based nitrogen loading from the 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 or wastewater treatment facility (see Section IV.1.2). All parcels are assumed to use on-site septic systems unless additional information is available.

MEP staff also received state Groundwater Discharge Permit (GWDP) nitrogen effluent data from the MassDEP (personal communication, Brian Dudley, 2/09) and alternative, denitrifying septic system total nitrogen effluent data from the Barnstable County Department of Health and the Environment (personal communication, Sue Rask and Brian Baumgaertel, 2/09). Two wastewater treatment facilities requiring state GWDPs are located in the Herring River watershed: the town Middle/Elementary School complex and the Cranberry Pointe nursing home facility. The BCDHE has 27 innovative/alternative septic systems in their performance database for the Town of Harwich. Four IA systems are in the Herring River watershed. This data was used to develop wastewater nitrogen loads for these sites.

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. 1990, Weiskel & Howes 1991, 1992, Koppelman 1978, Frimpter et al. 1990, Brawley et al. 2000, Howes and Ramsey 2000, Costa et al. 2001). Variation in per capita nitrogen load has been found to be relatively small, with average annual per capita nitrogen loads generally between 1.9 to 2.3 kg $person^{-1}yr^{-1}$.

However, given the seasonal shifts in occupancy and past 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 assessors parcel information using GIS techniques. The parcel specific water use data is converted to septic system nitrogen discharges (to the receiving aquatic systems) by adjusting for consumptive use (e.g. irrigation) and applying a wastewater nitrogen concentration. The water use approach focuses on the nitrogen load, which reaches the aquatic receptors 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. 2001). Downgradient studies of septic system plumes indicate that further nitrogen loss during aquifer transport is negligible (Robertson et al. 1991, DeSimone and Howes 1996).

In its application of the water-use approach to septic system nitrogen loads, the MEP 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, the MEP has derived a combined term for an effective N Loading Coefficient (consumptive use multiplied by 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 $person\text{-}yr^{-1}$ 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 use based on average residential occupancy to measured average water uses. This is performed as a quality assurance check to increase certainty in the final results. This comparison has shown that the larger the watershed the better the match between average water use and occupancy. For example, in the combined Great Pond, Green Pond and Bournes Pond watershed in the Town of Falmouth and the combined Popponesset Bay/Eastern Waquoit Bay watershed, both of which cover large areas and have significant year-round populations, the septic nitrogen loading based upon the census data is within 5% of that determined 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 form town-wide occupancy rates. These analyses provide additional support for the use of the water use approach in the MEP study region.

Overall, the MEP water use approach for determining septic system nitrogen loads has been both calibrated and validated in a variety of watershed settings. The approach: (a) is consistent with a suite of studies on per capita nitrogen loads from septic systems in sandy outwash aquifers; (b) has been validated in studies of the MEP Watershed “Module”, where there has been excellent agreement between the nitrogen load predicted and that observed in direct field measurements corrected to other MEP Nitrogen Loading Coefficients (e.g., stormwater, lawn fertilization); (c) the MEP septic nitrogen loading coefficient agrees in 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 analysis of Herring River 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 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 how the septic system nitrogen mass compares to the nitrogen loads from all other sources that reach the estuary (i.e. attenuated loads).

In order to provide an independent validation of the average residential water use within the Herring River Estuary sub-watersheds, MEP staff reviewed US Census population values for the Towns of Harwich and Brewster. Since Dennis occupies such a small portion of the watershed, its information was not included in this validation analysis. 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 2000 US Census, average occupancy within Harwich is 2.26 people per housing unit with 58% of year-round occupancy of available housing units; 2010 Census results are roughly the same: 2.18 and 55%, respectively. Brewster population numbers are similar: 2000 Census is 2.45 people per household with 56% year-round dwellings, while 2010 Census is 2.24 people per household with 55% year-round dwellings.

Comparison of average measured water use and water use estimates based on Census information show good agreement. Average water use for single-family residences with municipal water accounts in the Harwich MEP study area is 181 gpd. If this flow is multiplied by 0.9 to account for consumptive use, the study area average is 163 gpd. If the state Title 5 estimate of 55 gpd per capita is multiplied by average 2000 Harwich occupancy, the average water use per residence would be estimated at 125 gpd or 120 gpd if the 2010 occupancy is used. These Census-based flows do not include any adjustment for seasonal increases in occupancy. If it is generally assumed that seasonal properties are occupied at twice the year-round occupancy for three months, the average town-wide water use would be 156 or 150 gpd, respectively, which is approximately the same as the study area average water use. Analysis of the Brewster US Census results yields similar results and indicate little difference with Harwich in wastewater generation rates per residence. This analysis supports the approach of deriving average wastewater generation rates from measured average water use.

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 in areas impacted by seasonal population fluctuations such as most of Cape Cod. Estimates of summer populations on Cape Cod derived from a number of approaches (*e.g.*, traffic counts, garbage generation, sewer use) suggest average population increases from two to three times year-round residential populations measured by the US Census. This analysis suggests that water use, on average, is a reasonable estimate of wastewater generation within the study area.

Water use information exists for 79% of the 3,953 developed parcels in the Herring River watershed. The parcels without water use are split roughly between Brewster and Harwich; 58% are located in Brewster. Parcels without water use accounts are assumed to utilize private wells for drinking water. These are properties that were 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 a listed account in the water use databases. Of the 838 parcels, 750 (89%) are classified as single-family residences (land use code 101). These parcels are assumed to utilize private wells and were assigned the study area average water use of 181 gpd in the watershed nitrogen loading modules. Given the preponderance of residential land uses among developed parcels without water use accounts, all of these parcels were assigned 181 gpd as their water use in the watershed nitrogen loading model.

Alternative Site-specific Wastewater Estimates

During MEP assessments, MEP staff typically seek out information on large wastewater treatment facilities and alternative denitrifying septic systems for site-specific modifications. As mentioned previously, there are two wastewater treatment facilities requiring state GWDPs which are located in the Herring River watershed: the town Middle/Elementary School complex and the Cranberry Pointe nursing home facility. Effluent flow and total nitrogen concentration data for both facilities was provided to MEP staff by MassDEP staff (personal communication, Brian Dudley, 2/09). The school complex data covers the period from 2004 through 2007, while the nursing home facility has both effluent flows and nitrogen concentrations in 2006 and intermittently during 2005 and 2007.

The school complex has an average reported effluent discharge of 4,548 gallons per day (gpd), with most of the discharge occurring during the school year, September to June (Figure IV-3). Reported effluent flow ranges between 976 and 8,318 gpd. It should be noted that the average reported effluent discharge is 90% of the average water use at the complex (5,060 gpd), which supports the 10% consumptive use factor used in the general nitrogen loading calculations. Reported effluent total nitrogen concentrations at the school complex range between 2.0 and 48.1 mg/l and have an average concentration of 13.7 mg/l. Based on the monthly effluent and total nitrogen concentrations, MEP project staff calculated annual loads between 2004 and 2007 ranging between 68 and 107 kg with an average annual nitrogen load from the school complex of 90 kg. This average load is used in the annual watershed nitrogen loading estimates.

The nursing home facility has an average reported effluent discharge of 7,975 gallons per day (gpd) for the months where flow was reported; only 21 readings are reported over the four year period (Figure IV-4). Reported effluent flow ranges between 6,528 and 9,917 gpd. There are more (43) reported effluent total nitrogen concentrations than effluent flow readings; these range between 2.0 and 15.2 mg/l with an average concentration of 7.0 mg/l. Since only 2006 has a complete year where both both effluent and total nitrogen concentrations were reported, MEP project staff used this year to calculate a 91 kg annual nitrogen load from the Cranberry Pointe nursing facility.

Barnstable County Department of Health and the Environment maintains a database of selected alternative, denitrifying septic systems and their performance removing nitrogen. Review of the nitrogen effluent discharge from nearly 500 of these systems used on single-family residences throughout the county is only marginally different than the MEP factor (Heufelder, *et al*., 2007). In the Herring River Estuary study area, there are four alternative, denitrifying septic systems that have total nitrogen effluent data in the Barnstable County Department of Health and the Environment database (personal communication, Sue Rask and Brian Baumgaertel, 2/09). These four systems have 16 to 42 measurements and generally have average total nitrogen concentrations between 8.9 and 28 ppm. Project staff used these site-specific, average measured effluent total nitrogen concentrations and the average measured water use from the town Water Department records to calculate average annual loads from each of these four alternative, denitrifying septic systems. These loads were incorporated into the watershed nitrogen loading model for the Herring River Estuary.

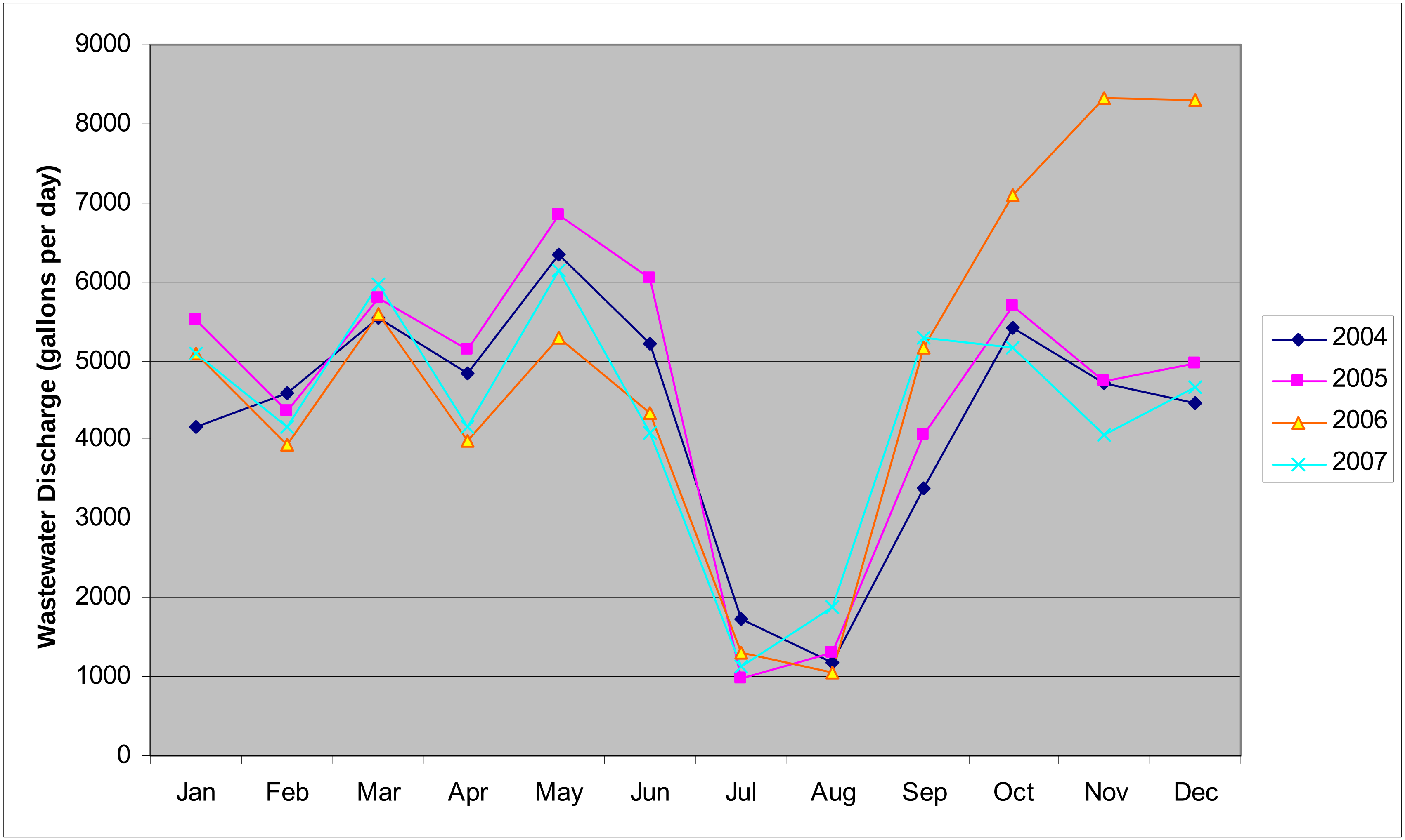

Figure IV-3. Total effluent discharge at the Town of Harwich Middle/Elementary School wastewater treatment facility (2004-2007). Data provided by MassDEP (personal communication, B. Dudley, 2/09). Annual nitrogen loads were determined from reported effluent flow and total nitrogen concentrations and averaged to produce the annual site-specific wastewater load used in the MEP watershed nitrogen loading calculations.

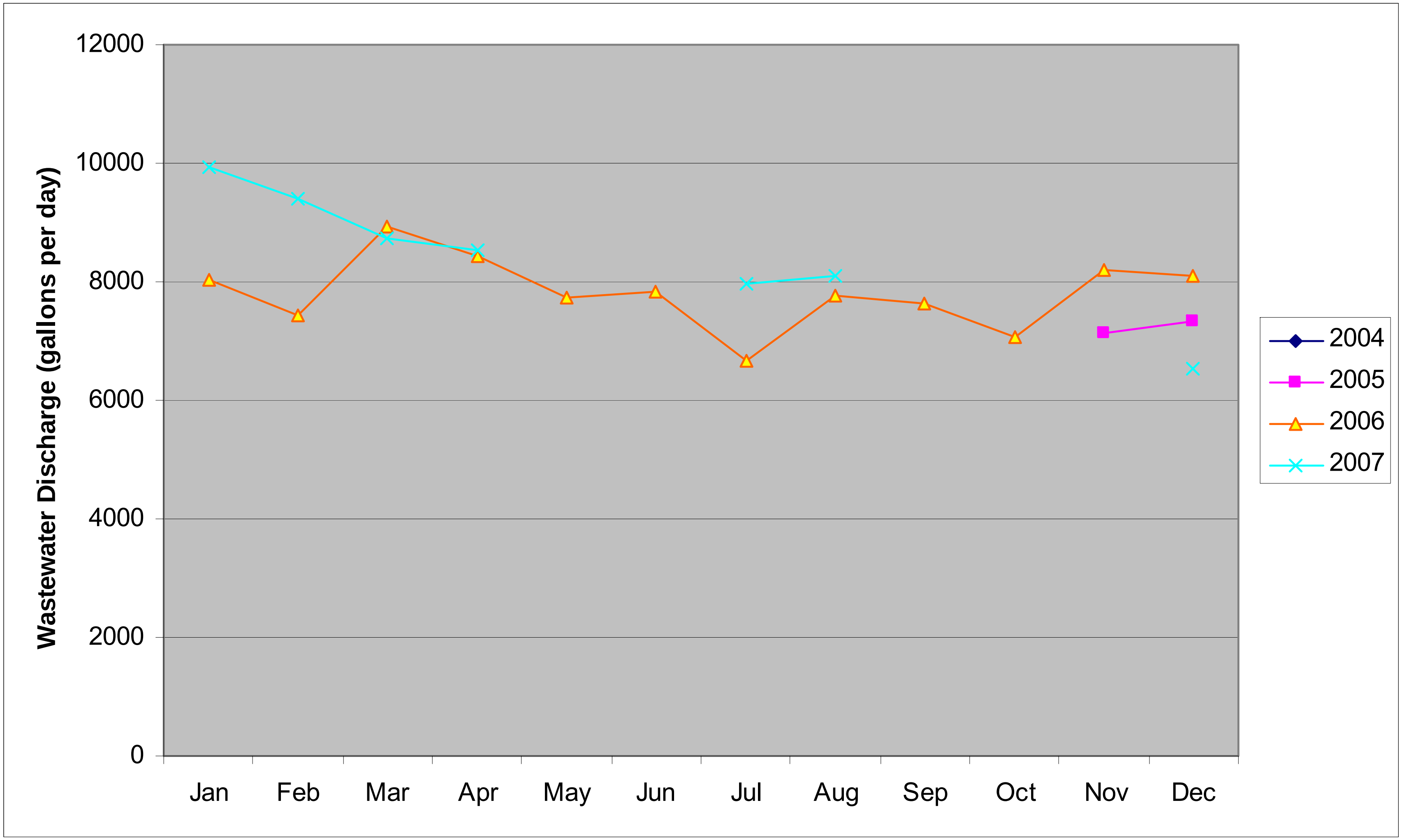

Figure IV-4. Total effluent discharge at the Cranberry Pointe nursing home wastewater treatment facility (2004-2007). Data provided by MassDEP (personal communication, B. Dudley, 2/09). No effluent flow data was reported in 2004, most of 2005, and portions of 2007. Since 2006 was the only year with both effluent flow and complementary total nitrogen concentrations, this year is the basis for the annual site-specific wastewater load from this facility in the MEP watershed nitrogen loading calculations.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of estuary watershed nitrogen loading 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 nitrogen loading model for the Herring River system, MEP staff reviewed available regional information about residential lawn fertilizing practices and incorporated site-specific information for the portion of the Cranberry Valley Golf Course and the cranberry bogs in the watershed. Cranberry bog nitrogen loading was determined based on previous studies conducted in southeastern Massachusetts, while MEP staff contacted the golf course superintendent to determine fertilizer application rates.

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 in over a decade, the MEP Technical Staff undertook an assessment of lawn fertilizer application rates and a review of leaching rates for inclusion in the Watershed Nitrogen Loading Sub-Model.

The initial step in this assessment of nitrogen factors for lawns and other turf was to determine nitrogen fertilization rates for residential lawns in the Towns of Falmouth, Mashpee and Barnstable. The assessment accounted for proximity to fresh ponds and embayments. Based upon ~300 interviews and over 2,000 site surveys, a number of findings emerged: 1) average residential lawn area is ~5000 sq. ft., 2) half of the residences did not apply lawn fertilizer, and 3) the weighted average application rate was 1.44 applications per year, rather than the 4 applications per year recommended on the fertilizer bags. Integrating the average residential fertilizer application rate with a leaching rate of 20% results in a fertilizer contribution of N to groundwater of 1.08 lb N per residential lawn; these factors are used in the MEP nitrogen loading calculations. It should also be noted that a recent data review of lawn fertilizer leaching in settings similar to those on Cape Cod confirmed that the 20% leaching rate is appropriate (HWG, 2009). It is likely that this still represents a conservative estimate of nitrogen load from residential lawns. It should be noted that professionally maintained lawns were found to have the higher rate of fertilizer application and hence higher estimated loss to groundwater of 3 lb/lawn/yr.

MEP staff contacted Sean Fernandez (02/2009) at the publicly-owned Cranberry Valley Golf Course to obtain current information about fertilizer application rates. Golf courses usually have different fertilizer application rates for different turf areas, usually higher annual application rates for tees and greens (~3-4 pounds per 1,000 square feet) and lower rates for fairways and roughs (~2-3.5 pounds per 1,000 square feet). At the Cranberry Valley Golf Course, Mr. Fernandez reported the following annual nitrogen application rates (in lbs/1,000 ft2) for the various turf areas: greens, 2.5; tees, 3.5; fairways, 3.5, and rough, 1.5.

As has been done in all MEP reviews, MEP staff reviewed the layout of the Cranberry Valley Golf Course from aerial photographs, classified the turf types, and, using GIS, assigned these areas to the appropriate sub-watersheds. The respective nitrogen application rates were then applied to these areas, a 20% leaching rate was applied, and annual load by sub-watershed was calculated.

Cranberry bog fertilizer application rate and percent nitrogen attenuation in the bogs is based on the only annual study of nutrient cycling and loss from cranberry agriculture that has been conducted in southeastern Massachusetts (Howes and Teal, 1995). Only the bog loses measurable nitrogen, the forested upland releases only very low amounts. For the watershed nitrogen loading analysis, the areas of active bog surface are digitized based on review of aerial photographs for properties classified as cranberry bogs in the town-supplied land use classifications. This review identified 166 acres of cranberry bogs located throughout the Herring River watershed.

Nitrogen Loading Input Factors: Harwich Landfill and other solid waste site

MEP staff contacted MassDEP and town staff to obtain any nitrogen monitoring data for solid waste sites within the Herring River system watershed. MassDEP has two solid waste sites within the watershed: 1) the Town of Harwich landfill and 2) the Robert B. Our Company, Wood Waste Reclamation Facility. Project staff review data on the town landfill presented by town staff and obtained data from MassDEP files to review the Our Company site (Mark Dakers, SERO, personal communication, 12/15/10). Development of nitrogen loads for each of these sites is based on the available monitoring data that is discussed in this section.

Harwich Landfill: The Harwich Landfill is located upgradient of Flax Pond (subwatershed #22). Groundwater contamination of the landfill has been monitored in at least two studies: 1) Horsley Witten Hegemann (1992) hydrogeologic characterization for a GWDP application for the Harwich Septage Treatment Plant and 2) Stearns and Wheler (2007) post-closure monitoring report. The landfill, which included septage lagoons, was capped in 1999. In addition, the town installed a floating, recirculating system on Flax Pond to try to restore its water quality and monitored the pond water quality between 1993 and 2002. A review of this monitoring data was completed in 2004 (Eichner, 2004).

MEP staff reviewed six (6) sampling runs of groundwater monitoring data for the landfill from 2004 to 2007 (Stearns and Wheler, 2007). Groundwater samples were analyzed for nitrate-nitrogen and ammonia-nitrogen. Staff used the sum of these constituent concentrations to approximate total nitrogen; this is a reasonable assumption provided that none of the nitrogen is bound in organic matrices. There is likely some unaccounted nitrogen since septage lagoons tend to release dissolved organic forms, but all forms would be captured by Flax Pond. Wells on the eastern portion of the landfill (where the septage lagoons were located) averaged 14.3 mg/l of NH4-N + NO3-N, while wells on the western portion of the landfill averaged 1.7 mg/l of NH4-N + NO3-N. MEP staff reviewed the downgradient boundary of the landfill and divided the boundary between an east and west portion based on average water table configuration. Stearns and Wheler (2007) states that there are 28 acres of solid waste on the landfill site. Using the downgradient boundary separation between the east and west portions of the landfill, MEP staff divided the recharge passing through the site by the percentage of each boundary portion and multiplied this by the average NH4-N + NO3-N concentration for each portion. This approach results in total annual nitrogen load from the landfill of 597 kg with 525 kg coming from the eastern portion of the landfill. The total annual load of 597 kg is added to the subwatershed #22 load.

It is acknowledged that this approach for estimating a nitrogen load from the Harwich landfill 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 MEP. MEP staff balanced reasonable estimates of the various factors based on the MassDEP's general guidance to the MEP 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, would help to refine this assessment for future management options.

Robert B. Our Company, Wood Waste Reclamation Facility: The Our Company site is 11.5 acres located north of Great Western Road and abutting the Dennis town boundary. Four and a half (4.5) acres of the site are used for solid waste activities. The site is wholly within the West Reservoir LT10 subwatershed (subwatershed #24). Monitoring data available in MassDEP files is from four (4) sampling runs: May, August, and December 2007 and April 2010. Available data consists of contaminant concentrations from groundwater samples collected from four (4) wells. Unlike the Harwich landfill data, the available Stump Dump groundwater monitoring data includes both nitrate-nitrogen and Kjeldahl nitrogen (ammonium + dissolved organic nitrogen) concentrations, which can be added to determine total nitrogen.

The four monitoring wells at this site are located in a pattern to establish one upgradient site and three downgradient sites. Well logs detailing construction details were not available from MassDEP files. The average total nitrogen concentration of the downgradient wells is 7.7 mg/l, while the upgradient well has an average total nitrogen concentration of 3.7 mg/l. Using MassDEP materials and reviewing aerial photos, project staff determined that there is 4.1 acres of solid waste on the site.

Given the lack of definitive groundwater flow paths and well construction details, it is unclear whether the up gradient well is in the same flow path as the wells installed as the downgradient wells. In order to address this uncertainty and based on MassDEP guidance to be conservative in uncertain characterizations, MEP staff utilized the average total nitrogen concentration of the downgradient wells to estimate the nitrogen load from the R.B. Our Company site. Using this concentration and the standard MEP recharge rate on the 4.1 acres of solid waste on this site results in an annual nitrogen load of 97 kg from the Our Company site. This load is added to the subwatershed #24 load.

It is acknowledged that this approach for estimating a nitrogen load from the R.B. Our Company site includes a number of assumptions and is conservative. A detailed assessment of all the available data is beyond the scope of the MEP, but MEP 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, additional sampling for total nitrogen concentrations, would help to refine this assessment and future management options.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas in the Herring River Estuary assessment are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and the MassDEP Nitrogen Loading Computer Model Guidance Document (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). Factors used in the MEP nitrogen loading analysis for the Herring River Estuary watershed are summarized in Table IV-2.

Table IV-2. Primary Nitrogen Loading Factors used in the Herring River MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Harwich data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.

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
Direct Precipitation on Embayments and Ponds	1.09	Water Use/Wastewater:	
Natural Area Recharge	0.072	Existing developed residential parcels wo/water accounts and buildout residential parcels:	181 gpd
Wastewater Coefficient	23.63		
Town of Harwich Middle/Elementary School wastewater facility			
Average effluent Flow (gallons per day)	4,548		
Effluent Total Nitrogen concentration (mg/l)	13.7	Existing developed parcels w/water accounts:	Measured annual water use
Cranberry Pointe nursing home wastewater facility		Average Single Family Residence Building Size from watershed data (sq ft)	1,216
Average effluent Flow (gallons per day)	7,821		
Effluent Total Nitrogen concentration (mg/l)	8.4	Commercial and Industrial Buildings without/WU and buildout additions	
Fertilizers:		Commercial	
Average Residential Lawn Size (sq ft)*	5,000	Wastewater flow (gpd/1,000 ft2 of building):	236
Residential Watershed Nitrogen Rate (lbs/lawn)*	1.08	Building coverage:	13.2%
Cranberry Bogs nitrogen application (lbs/ac)	31	Industrial	
Cranberry Bogs nitrogen attenuation	34%	Wastewater flow (gpd/1,000 ft2 of building):	78
Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined from site-specific information		Building coverage:	14.5%

Road areas are based on MassHighway GIS information, which provides road width for various road segments. MEP staff utilized the Cape Cod Commission GIS to sum these segments and their various widths by sub-watershed. Project staff also checked this information against parcel-based rights-of-way.

In addition to these other sources, MEP staff also determined nitrogen loading estimates for farm animals. Project staff obtained locations of farms and counts of various animal species from the Town of Harwich Health Director (personal communication, Paula Champagne, 3/08). Using nitrogen loading factors developed for individual farm animals in previous MEP analyses (Howes *et al.*, 2007) and GIS techniques, project staff determined nitrogen loading estimates for each farm and assigned the load to the appropriate sub-watershed. Nine animal types and a total of 292 animals are located in the Herring River watershed: 144 horses, 4 cows, 28 goats, 19 sheep, 3 rabbits, 6 pigs, 9 turkeys, 51 chickens/guinea hens, and 28 ducks. The total unattenuated nitrogen load from all these animals is 2,149 kg/y.

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information was linked to the parcel coverages, parcels were assigned to various sub-watersheds based initially on whether at least 50% or more of the land area of each parcel was located within a respective watershed. Following the assigning of boundary parcels, all large parcels were examined individually and were split (as appropriate) in order to obtain less than a 2% difference between the total land area of each sub-watershed and the sum of the area of the parcels within each sub-watershed. The resulting "parcelized" watersheds to Herring River are shown in Figure IV-5.

The review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Each of the towns provided GIS coverages of building footprints for the roof area calculations; Dennis footprints are from 2009, Brewster's footprints are from 2005, and Harwich footprints are from 2006. 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 Herring River Estuary. The assignment effort was undertaken to better define sub-estuary loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

Following the assignment of all parcels, sub-watershed modules were generated for each of the 29 sub-watersheds in the Herring River study area. These sub-watershed modules summarize, among other things: water use, parcel area, parcel frequency by land use category, private wells, and road area. The individual sub-watershed modules were then integrated to create a Herring River Watershed Nitrogen Loading module with summaries for each of the 29 individual sub-watersheds. The subwatersheds are generally paired with functional embayment/estuary units for the Linked Watershed-Embayment Model's water quality component.

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 Herring River study area, the major types of nitrogen loads are: wastewater (*e.g.*, septic systems), wastewater treatment facilities, fertilizer (with cranberry and golf course loads specified), impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-3). The output of the watershed nitrogen-loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-6). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation during transport to the estuarine system before use in the embayment water quality sub-model.

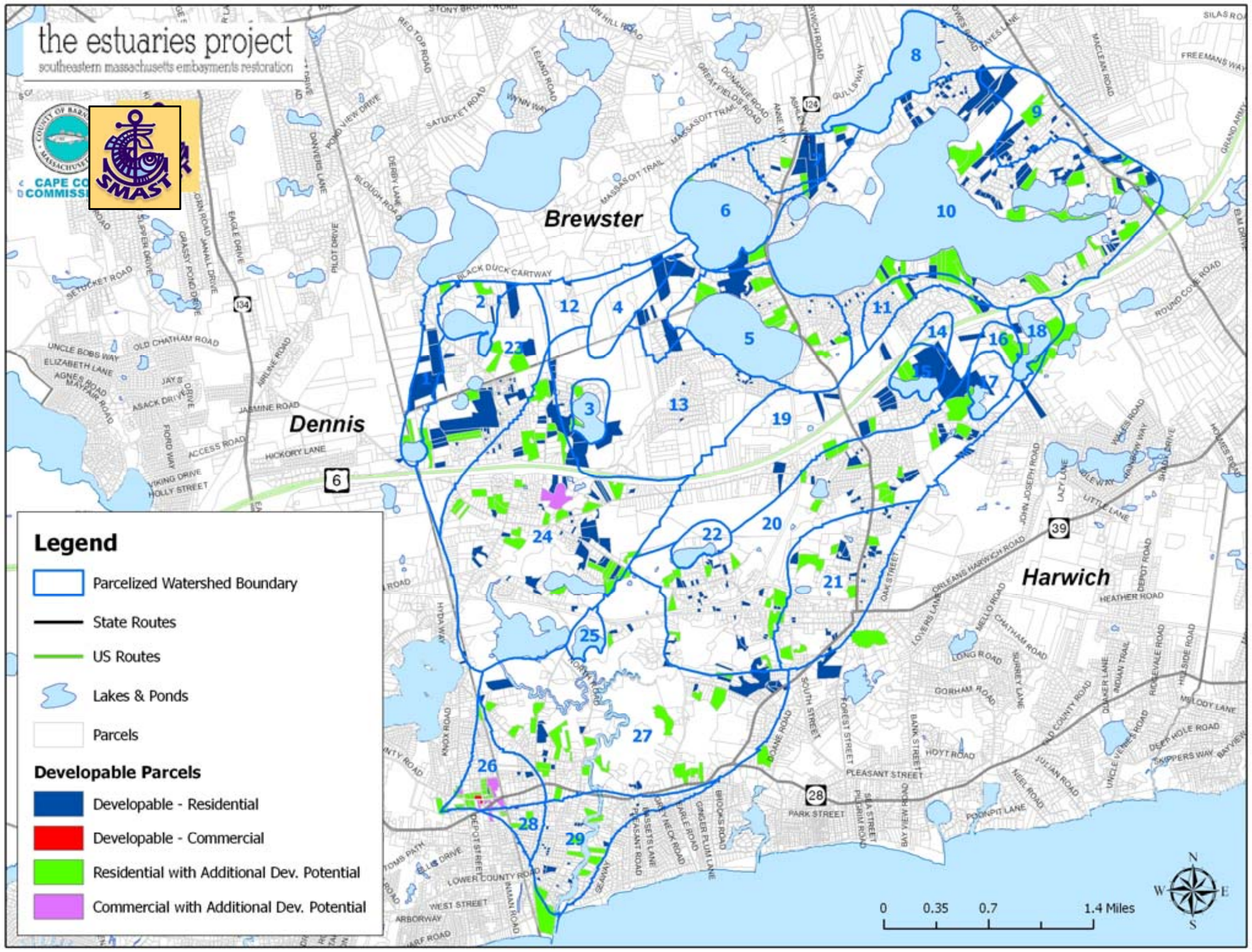

Figure IV-5. Parcels, Parcelized Watersheds, and Developable Parcels in the Herring River watersheds. Parcels colored green and purple are developed parcels (residential and commercial, respectively) with additional development potential based on current zoning, while parcels colored blue and red are corresponding undeveloped parcels classified as developable by the respective town assessors. The parcelized watersheds are drawn to minimize the division of properties for management purposes while achieving a match of area with the modeled watersheds of 2% or less. Developable parcels are based on town assessor classifications and minimum lot sizes specified in town zoning; these parcels are assigned estimated nitrogen loads in MEP buildout calculations. All buildout results were reviewed with respective town staff.

Table IV-3. Herring River Estuary Watershed Nitrogen Loads. Attenuated nitrogen loads are based on measured and attenuation factors assigned to streams and freshwater ponds. Stream attenuation is based on measured loads (see Section IV.2), while pond attenuation factors are either determined from available pond monitoring data or assigned a standard MEP nitrogen attenuation of 50%. Values are kg N yr^{-1}.

		Herring River N Loads by Input (kg/y):												Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	WWTF	Landfill /Solid Waste	Total Fertilizers	Cranberry Fertilizer	Golf Fertilizer	Farm Animal Loads	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout	% of Pond Outflow	UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Herring River Total		**24589**	**158**	**694**	**3434**	**1546**	**77**	**2149**	**2532**	**6432**	**1352**	**9729**		**41340**		**23164**	**51069**		**29210**
Lower Herring River		2578	0		230	0	0	183	274	92	33	292		3390		3390	3682		3682
Lower Herring R_Main_LT10	29	2116	0		183	0	0	183	221	0	28	238		2732		2732	2970		2970
Lower Herring R_Main_GT10	28	462	0		47	0	0		53	0	5	53		566		566	620		620
Lower Herring River Estuary Surface										92				92		92	92		92
Upper Herring River		3839	0		438	200	0	0	381	226	220	1414		5104		5104	6518		6518
East Reservoir	25	17	0		1	0	0		3	82	3	0		107		107	107		107
Upper Herring River LT10	27	2205	0		344	200	0		258	0	197	773		3005		3005	3777		3777
Upper Herring River GT10	26	1616	0		93	0	0		120	0	20	642		1848		1848	2490		2490
Upper Herring River Estuary Surface										144				144		144	144		144
West Reservoir Total		11595	0	97	1914	1025	0	1736	1234	5717	705	6072		22999	25%	10061	29071	25%	13509
West Reservoir LT10	24	2921	0	97	454	278	0	1057	308	109	185	2175		5132		5132	7306		7306
West Reservoir GT10	23	841	0		199	142	0	653	80	2	80	704		1854		1854	2558		2558
White Pond	1	61	0		5	0	0		6	31	10	92	55%	114	50%	57	207	50%	103
Elbow Pond	2	102	0		30	21	0		6	160	11	20		309	50%	154	329	50%	164
USGS Gauge		7670	0		1225	584	0	26	834	5416	419	3081		15590		6218	18671		7880
Herring River N LT 10	13	1865	0		339	200	0	26	166	3	106	853		2504		2504	3358		3358
N_HarWell	4	0	0		0	0	0		0	0	20	47		20		20	66		66
Robbins Pond	3	24	0		2	0	0		3	147	5	20		180	50%	90	200	50%	100
Herring River N GT10	12	187	0		15	1	0		23	0	40	200		266		266	466		466
Seymour Pond	SEP	247	0		25	4	0		28	223	9	64	25%	532	69%	151	595	69%	170
Hinckleys Pond	HP	5347	0		845	379	0		614	5043	239	1897	100%	12088	31%	3187	13985	31%	3720
Lothrup Road Total		6577	158	597	853	320	77	229	643	397	393	1951		9848	50%	4609	11799	50%	5501
Lothrup Rd GT10 N	19	1518	68		81	0	0	52	163	0	137	503		2019		2019	2522		2522
Flax Pond	22	0	0	597	19	19	0		2	70	7	7		696	50%	348	702	50%	351
Lothrup Rd LT10	20	2856	0		600	301	77	177	256	33	152	835		4073		4073	4908		4908
Lothrup Rd GT10 S	21	2007	90		140	0	0		200	0	59	280		2496		2496	2776		2776
Aunt Edies Pond	AEP	65	0		3	0	0		6	95	14	113		183	50%	92	296	50%	148
Cornelius Pond	CP	83	0		7	0	0		9	120	21	125		240	50%	120	365	50%	183
Walker Pond	WP	47	0		4	0	0		6	80	4	88	53%	141	50%	70	229	50%	115

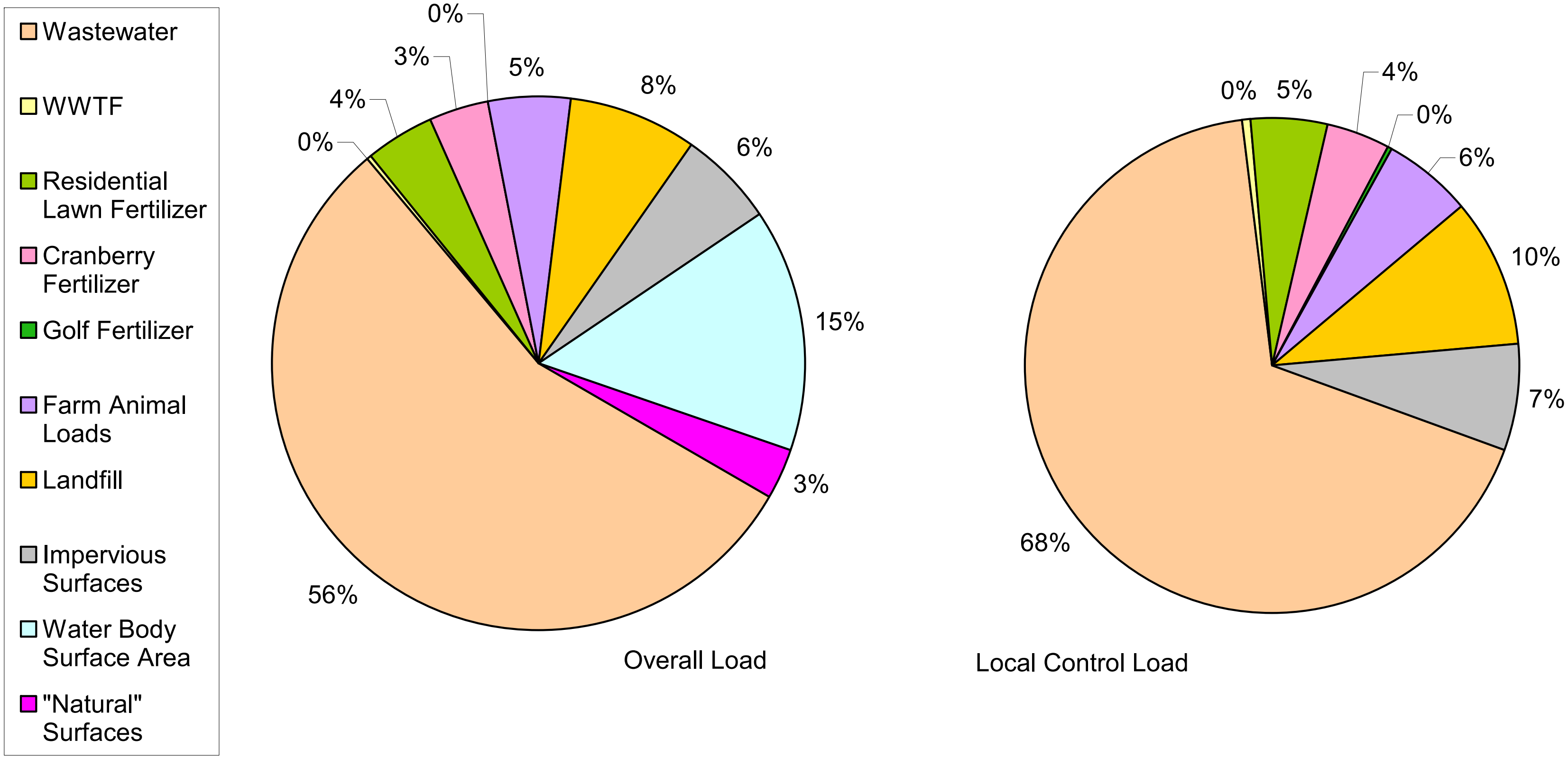

Figure IV-6 Land use-specific unattenuated nitrogen loads (by percent) to the Herring River watershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.

One of these attenuation adjustments occurs in the freshwater ponds. Since groundwater outflow from a pond can enter more than one down gradient sub-watershed, the length of shoreline on the down gradient side of the pond was used to apportion the pond-attenuated nitrogen load to respective down gradient watersheds. The apportionment was based on the percentage of discharging shoreline bordering each downgradient sub-watershed. In the Herring River study area, this primarily occurs in the ponds located along the outer boundary of the Herring River watershed: Sheep, Hinckley, and White. At Sheep Pond, for example, the pond has a down gradient shoreline of 11,828 feet; 30% of that shoreline discharges into the Long Pond sub-watershed (watershed #10 in Figure IV-1), 12.5% discharges to Seymour Pond GT10 sub-watershed (watershed #7 in Figure IV-1) and the remainder is discharged outside of the Herring River watershed. This breakdown of the discharge from Sheep Pond means that 30% of the attenuated nitrogen load that leaves Sheep Pond reaches Long Pond, 12.5% reaches Seymour Pond, and the remainder leaves the watershed. Similar pond-specific calculations were completed wherever pond flows and nitrogen loads were divided among a number of downgradient receiving subwatersheds.

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally watershed sites of natural nitrogen reduction (or attenuation) prior to the watershed nitrogen reaching an estuary. These ponds are kettle hole depressions that intercept the surrounding groundwater table revealing what some call "windows on the aquifer." Groundwater typically flows into the pond along the up-gradient shoreline, then lake water flows back into the groundwater system along the down gradient shoreline. Occasionally a Cape Cod pond will also have a stream outlet, which is often a herring run, that also acts as a discharge point. Since the nitrogen loads flow into the pond with the groundwater, the relatively more productive pond ecosystems incorporate some of the nitrogen, retain some nitrogen in the sediments, and change the nitrogen among its various oxidized and reduced forms. As result of these interactions, some of the nitrogen is removed from the watershed system, mostly through burial in the sediments and denitrification that returns it to the atmosphere. Following these reductions, the remaining attenuated loads flow back into the groundwater system along the downgradient side of the pond and eventually discharge into the estuary via groundwater or discharge through a stream outlet directly to the estuary. The nitrogen load summary in Table IV-3 includes both the unattenuated (nitrogen load to each sub-watershed) and attenuated nitrogen loads.

Nitrogen attenuation in freshwater ponds has generally been found to be at least 50% in MEP analyses, so the watershed model assigns a conservative attenuation rate of 50% to all nitrogen from freshwater pond watersheds unless more detailed monitoring or studies are available. Detailed studies of other southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE, 2000) and the Agawam/Wankinco River Nitrogen Discharges (CDM, 2001) have also supported a 50% attenuation factor. 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 review whether a nitrogen attenuation rate higher than 50% should be used, the MEP Technical Team reviews the available data on each pond, including available nitrogen concentrations, impacts of sediment regeneration, temperature profiles, and bathymetric information.

Bathymetric information is a prerequisite for calculating a pond-specific attenuation rate, since it provides the volume of the pond and, with appropriate nitrogen concentrations, a measure of the nitrogen mass in the water column. Combined with the watershed recharge, this information can provide a residence or turnover time that is necessary to gauge attenuation.

In addition to bathymetry, temperature profiles are useful to help understand whether temperature stratification is occurring in a pond. If the pond has an epilimnion (*i.e.*, a well-mixed, relatively isothermal, warm, upper portion of the water column) and a hypolimnion (i.e., a deeper, colder layer), the stability and volume of these two layers must be accounted for in the nitrogen attenuation calculations. In these stratified lakes, the upper epilimnion is usually the primary discharge for watershed nitrogen loads; the deeper hypolimnion generally does not interact with the upper layer. However, deep lakes with hypolimnions often also have significant sediment regeneration of nitrogen and in lakes with impaired water quality this regenerated nitrogen can impact measured nitrogen concentrations in the upper epilimnion and this impact should also be considered when estimating nitrogen attenuation.

Many ponds on Cape Cod have been sampled through the regional Cape Cod Pond and Lake Stewards (PALS) Snapshots and the initiative of local volunteer pond sampling programs. The PALS Snapshots are regional volunteer pond one-time sampling supported for the last nine years by SMAST and the Cape Cod Commission, with free laboratory services provided by the Coastal Systems Program Laboratory at SMAST. Sampling protocols developed through the PALS program (Eichner *et al*., 2003) have been used for more extensive pond sampling programs in many communities on Cape Cod, including Brewster (Eichner, 2009) and Harwich (Moran, 2009). Sampling under these protocols has included field collection of temperature and dissolved oxygen profiles and sampling of standardized depths that include some evaluation of the impact of sediment nutrient regeneration. PALS water samples are analyzed at the SMAST Coastal Systems Analytical Facility for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH. In some cases town programs have generated sufficient sampling data that modified MEP nitrogen attenuation rates can be reliably assigned to freshwater ponds.

Within the Herring River Estuary watershed, there are twelve major freshwater ponds with delineated watersheds: White, Elbow, West Reservoir, Robbins, Hinckleys, Flax, Aunt Edies, Cornelius, Walkers, Seymour, Sheep, and Long. Among these ponds, seven have available pond-wide bathymetric data and among these water quality data is available for six ponds: Elbow, Hinckleys, Flax, Seymour, Sheep, and Long. The remaining ponds were assigned the standard MEP pond nitrogen attenuation rate of 50% except for West Reservoir, which has a stream monitoring point at its outflow (see Section IV.2).

In order to further evaluate whether an alternative nitrogen attenuation rate could be assigned to the six ponds that have water quality data and pond-wide bathymetric data, MEP staff reviewed the available water quality monitoring data. This data includes sampling results collected through 2008 by town volunteers, including laboratory samples analyzed through the annual Cape Cod Pond and Lake Stewards (PALS) Snapshots and town-funded activities.

In MEP analyses, available nitrogen concentrations from individual ponds are reviewed to establish whether sediment regeneration is a significant factor in a pond and if the pond is stratified such that it can be determined whether or not the entire volume of the pond should used to determine a turnover time. Turnover time is how long it takes the recharge from the up-gradient watershed to completely exchange the water in the pond or, in the case of a thermally stratified pond, exchange just the epilimnion. The total mass of nitrogen in the pond or epilimnion is adjusted using the pond turnover time to determine the annual nitrogen load returned to the aquifer through the down gradient shoreline. This mass is then compared to the nitrogen load coming from the pond's watershed to determine the nitrogen attenuation factor for the pond.

Review of available data in Flax Pond shows that sediment regeneration of nutrients is so significant that it may be impacting nitrogen concentrations throughout the pond. A previous assessment of phosphorus in the pond suggested that observed concentrations were at least twice what they would be if only watershed sources were contributing to the pond (Eichner, 2004). Given this situation, it is difficult to accurately gauge the attenuation of nutrients from the watershed. For this reason, Flax was assigned the standard MEP freshwater pond 50% nitrogen attenuation rate.

Available water quality data for the remaining ponds generally focused on summer months, with significant concentration of data in the months of July and August. Although this time frame is crucial for determining ecological status in surface waters, MEP staff reviewed this data to assess its variability and determined that some of the average conditions in Elbow Pond demonstrated too much variability to use an alternative nitrogen attenuation rate. At this stage, it was determined that the standard MEP pond attenuation rate of 50% should be utilized for nitrogen loads from the watershed to Elbow Pond.

Water quality data for Hinckleys, Seymour, Sheep, and Long was determined to be consistent and abundant enough to develop pond-specific nitrogen attenuation rates. However, since the available datasets do not provide adequate assessment of nitrogen attenuation during seasons other than summer, it was determined that the attenuation rates should be conservatively determined using the average nitrogen concentrations plus one standard deviation (Table IV-4). The average attenuation for these four ponds was 62%.

Buildout

Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment of potential development within the study area watersheds. The MEP buildout is relatively straightforward and is completed in four steps: 1) each residential parcel classified by the town assessor as developable are identified and divided by minimum lot sizes specified in current town zoning and the resulting number of new residential units is rounded down, 2) parcels classified by the town assessor as developable commercial and industrial parcels are identified, 3) residential, commercial and industrial parcels with existing development and lot 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, and 4) results are discussed with town staff and/or planning board members and the analysis results are modified based on local knowledge.

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 Chapter 40B affordable housing projects. The fourth step, including the discussions with town planners, and, occasionally, town boards and wastewater consultants, usually leads to additional insights on developments that are planned, especially developments planned on government or public service parcels, and updates to assessor classifications, including lands purchased by the town as open space. This final step may lead to removal and/or additions to the number of parcels initially identified as developable and application of more detailed zoning provisions.

Table IV-4. Nitrogen attenuation by Freshwater Ponds in the Herring River watershed based upon 2001 through 2008 town volunteer sampling and Cape Cod Pond and Lakes Stewardship (PALS) program sampling. These data were collected to provide a site-specific check on nitrogen attenuation by these systems. All ponds in the watershed are assigned the standard MEP nitrogen attenuation value of 50% except for Hinckleys, Seymour, Sheep, and Long. The listed attenuation rates for these ponds are based on average surface concentrations plus one standard deviation, which provides some conservatism to the attenuation rates and accounts for limited sampling data outside of the summer season.

Pond	PALS ID	Area acres	Maximum Depth m	Upper (U) /Whole (W) volume turnover time yrs	# of TN samples for N Attenuation calculation	N Load Attenuation %
Aunt Edies	HA-376	21.2	2.3	No Bathymetric Info	23	Not calculated due to lack of bathymetry
Cornelius Pond	HA-381	11.0	2.6		29	
Robbins	HA-386	32.8	2.9		33	
White	HA-414	12.1	6.3		26	
Walkers	HA-358	33.4	7.9		22	
West Reservoir	HA-530	22.4	2.4	0.1 (W)	No WQ data	Stream outflow
Flax	HA-507	15.7	6.5	1.5 (W)	18	50% std
Hinckleys	HA-353	170.9	8.4	0.4 (W)	16	31%
Elbow	BR-357	35.6	9.3	1.1 (U)	13	50% std
Seymour	HA-306	182.9	12.8	2.2 (U)	22	80%
Sheep	BR-240	148.0	20.1	6.1 (U)	30	90%
Long	BR-279	742.1	21.0	3.0 (U)	54	73%

Data sources: all areas from CCC GIS; all maximum depths are maximum recorded depths from citizen sampling except for West Reservoir, which is based on bathymetric measurements; number of Total Nitrogen samples available for attenuation calculations are surface concentrations from town monitoring and annual PALS Snapshot provided by SMAST lab; available bathymetry from MassDFW bathymetric maps (www.mass.gov/dfwele/dfw/dfw_pond.htm and Eichner, et al., 2003).

As an example of how the MEP approach might apply, assume an 86,000 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 differentiated treatment of undevelopable lots, commercial and industrial properties, and lots less than the minimum areas specified by zoning. Properties classified by the Town of Dennis, Harwich, and Brewster assessors as “undevelopable” (*e.g.*, MassDOR codes 132, 392, and 442) are not assigned any development at buildout (unless revised by the town review). Commercial and industrial properties classified as developable are not subdivided; the area of each parcel and the factors in Table IV-2 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 130 land use code will be assigned an additional residential dwelling in the MEP buildout scenario even though the minimum lot size 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.

Following the completion of the initial buildout assessment for the Herring River watersheds, MEP staff reviewed the results with respective town officials. A number of Harwich discussions were held with Sue Leven, the town planner at the time of their preparation, and Frank Sampson, Chair of the Town-Wide Water Quality Management Task Force, but buildout results were also discussed with Elizabeth Hude, in her brief time as town planner, and David Spitz, the current town planner. Harwich discussions included an alternative review of clusters of government owned parcels, so-called "Selectmen Parcels".

MEP staff also discussed initial Dennis buildout results with Dan Fortier, Town of Dennis Town Planner, who provided additional refinements on the Dennisport Growth Incentive Zone (GIZ). The core of the GIZ is within the Herring River watershed and Mr. Fortier provided buildout information from the GIZ analysis. Results from these modifications are incorporated into the buildout analysis.

MEP staff discussed initial Brewster buildout results with Jillian Douglas, Town of Brewster Assistant Town Administrator. Modifications from Brewster included incorporation of a zoning bylaw amendment that allows an additional 1-bedroom unit by right on all parcels greater than 125% of the minimum lot size. The analysis completed by the town in support of the amendment was incorporated into the MEP Herring River Estuary buildout assessment.

All the parcels with additional buildout potential within the Herring River watershed are shown in Figure IV-5. 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 additions. All wastewater loads are assumed to come from on-site septic systems. Cumulative unattenuated buildout loads are indicated in a separate column in Table IV-3. Buildout additions within the Herring River watersheds will increase the existing unattenuated loading rate by 24%.

IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT

IV.2.1 Background and Purpose

Modeling and predicting changes in coastal embayment nitrogen related water quality is based, in part, on determination of the inputs of nitrogen from the surrounding contributing land or watershed. This watershed nitrogen input parameter is the primary term used to relate present and future loads (build-out, sewering analysis, enhanced flushing, pond/wetland restoration for natural attenuation, etc.) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the sub-watersheds of the Herring River System being investigated under this nutrient threshold analysis were based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1).

If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watersheds where nitrogen transport from source to estuarine waters is through groundwater flow in sandy outwash aquifers (such as the developed regions of the Herring River watershed). The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, nitrogen passes through a surface water ecosystem (pond, wetland, stream) on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) is diminished by natural biological processes that represent removal (not just temporary storage). However, this natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes. In the watershed for this marsh / embayment system, a portion of the freshwater flow and transported nitrogen passes through several surface water systems (e.g. Herring River discharge from West Reservoir into western portion of the marsh and Herring River discharge under Lothrop Road into eastern portion of the marsh) prior to entering the estuary, producing the opportunity for significant nitrogen attenuation.

Failure to determine the attenuation of watershed-derived nitrogen overestimates the nitrogen load to receiving estuarine waters. If nitrogen attenuation is significant in one portion of a watershed and insignificant in another the result is that nitrogen management would likely be more effective in achieving water quality improvements if focused on the watershed region having unattenuated nitrogen transport (other factors being equal). In addition to attenuation by freshwater ponds (see Section IV.1.3, above), attenuation in surface water flows is also important. An example of the significance of surface water nitrogen attenuation relating to embayment nitrogen management was seen in the Agawam River, where >50% of nitrogen originating within the upper watershed was attenuated prior to discharge to the Wareham River Estuary (CDM 2000). Similarly, MEP analysis of the Quashnet River indicates that in the upland watershed, which has natural attenuation predominantly associated with riverine processes, the integrated attenuation was 39% (Howes et al. 2004). In addition, a preliminary study of Great, Green and Bournes Ponds in Falmouth, measurements indicated a 30% attenuation of nitrogen during stream transport (Howes and Ramsey 2001). An example where natural attenuation played a significant role in nitrogen management can be seen relative to West Falmouth Harbor (Falmouth, MA), where ~40% of the nitrogen discharge to the Harbor originating from the groundwater effluent plume emanating from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Clearly, proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements of upper watershed attenuation were undertaken as part of the MEP Approach for the Herring River Estuary. MEP conducted long-term measurements of natural attenuation relating to surface water discharges to the perimeter of the embayment system in addition to the natural attenuation measures by fresh kettle ponds, addressed above (Section IV.1). These additional site-specific studies were conducted for the 2 major surface water flow systems in their respective sub-watersheds to the overall system: (1) Herring River (fresh) discharging to the western portion of the upper reach of the Herring River Estuary and (2) stream flow passing under the Lothrop Road bridge discharging to the eastern portion of the upper tidal reach (brackish) of the Herring River Estuary (Figure IV-7).

Figure IV-7. Location of Stream gauges (red symbols) for determination of freshwater discharge and nitrogen load from the upper sub-watersheds to the Herring River Estuary.

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the 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 between 15 and 24 months of record depending on the stream gaging location (Figures IV-8 to IV-9). During each study period, velocity profiles were completed on each stream cross-section 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. These hourly stages values where then entered into the stage-discharge relation to compute hourly flow. Hourly flows were summed over a period of 24 hours to obtain daily flow and further, daily flows summed to obtain annual flow. In the case of tidal influence on stream stage, the diurnal low tide stage value was extracted on a day-by-day basis in order to resolve the stage value indicative of strictly freshwater flow. The two low tide stage values for any given day were averaged and the average stage value for a given day was then entered into the stage – discharge relation in order to compute daily flow. A complete annual record of stream flow (365 days) was generated for the surface water discharges flowing into the Herring River embayment system.

The annual flow record for the surface water flow at each gauge was merged with the nutrient data set generated through the water quality sampling performed at the gauge locations to determine nitrogen loading rates to the wetlands filling the upper basin of the Herring River

Estuary. 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 measurements 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 system.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Herring River at Herring Run Outlet of West Reservoir (Stream 1) to Western Portion of the Wetland Dominated Upper Basin of the Herring River Estuary

The West Reservoir, located up gradient of the Herring River gauge site (positioned immediately down gradient of the herring run located in the Herring River Conservation Area) is a moderately sized freshwater “pond” and unlike many of the freshwater “ponds” on Cape Cod, this man-made pond/reservoir has stream outflow rather than discharging solely to the aquifer (e.g. Sand or Flax Pond) along its down-gradient shore. This stream outflow, Herring River (stream 1), may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation taking place in the reservoir and up gradient bogs, wetlands and streambed associated with the sub-watersheds contributing freshwater and nitrogen to the gauge site. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to Herring River (stream 1) above the gauge site and the measured annual discharge of nitrogen to the tidal portion of the Herring River system, Figure IV-7.

Stream gauging on the Herring River (stream 1) was undertaken at two different locations, only one of which was ultimately successful. Initially, the stream gauging was undertaken at a location where the Herring River passes under Bells Neck Road. This location was selected because it was situated in the freshwater marsh portion of the system (Figure I-2) and was reflective of freshwater conditions at low tide. Stream gauging at this initial location would also provide for a measure of the nitrogen attenuation taking place within the freshwater marsh area located up-gradient of Bells Neck Road. Unfortunately, measures of freshwater flow this far down in the system were too influenced by backwater effects from tidal influences lower in the system and a credible stage-discharge relationship could not be developed. Furthermore, the stage record was compromised from frequent vandalism of the water level logger collecting the high frequency stage record. Towards the end of the 16 month deployment of the stream gauge at the Bells Neck location, the MEP Technical Team decided to re-deploy the stream gauge further up-gradient in the system, specifically 50 meters south of the Herring Run and the large culvert discharging freshwater flow from West Reservoir. As this was a second deployment it was initiated immediately upon determining that the first stream gauging effort would not be valid. As such, the second deployment was begun in December of 2005 and continued for 12 months until December 2006 thereby capturing one summer season.

At the Herring River gauge site (stream 1 discharging to western portion of the upper basin), a continuously recording vented calibrated water level gauge was installed to yield the level of water in the freshwater stream channel carrying flow and associated nitrogen load to the estuary and eventually the near shore waters of Nantucket Sound. As portions of the lower segments of the Herring River are tidally influenced, the gauge was located such that freshwater flow could be measured at low tide and a stage-discharge relationship could be developed based on stages at low tide without any back water effects. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average low tide salinity was determined to be 0.1 ppt. Therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked monthly. The re-deployed gauge on this western portion of the Herring River system immediately down gradient from the reservoir was installed on December 24, 2005 and was set to operate continuously for 12 months such that at least one summer season would be captured in the flow record. Stage data collection continued until November 20, 2006 at which time the stream gauge was vandalized making for a total deployment of 11 months. Stream flows for the one month for which there was no stage data had to be interpolated based on the last measured stream flow valued obtained at this site.

Stream flow (volumetric discharge) was directly measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Herring River (stream 1) site based upon these flow measurements and measured water levels from the gauge. 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 allowed for the determination of nitrogen mass discharge to the head of western portion of the Herring River Estuary, reflective of the biological processes occurring in the reservoir and riparian zones and up-gradient ponds contributing to nitrogen attenuation (Figure IV-8 and Table IV-5). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gauge site.

The annual freshwater flow record for the Herring River at the gauge immediately down gradient of the reservoir measured by the MEP was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from Herring River at this location showed good agreement with the long-term average from the USGS, being only 3% above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for December of 2005 to December of 2006 was 42,111 m^3/day, close to the long term average flows determined by the USGS modeling effort (40,919 m^3/day).

The difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in the Herring River in the western portion of the system was considered to be negligible. The insignificant difference between the long-term average flow based on watershed recharge rates and the MEP measured Herring River discharge indicates that the measured Herring River flow is properly capturing the up-gradient recharge (and loads) and provides direct evidence that the delineated watershed areas are correct.

Table IV-5. Comparison of water flow and nitrogen discharges from Rivers and Streams (freshwater) discharging to estuarine reach of Herring River Marsh. The "Stream" data is from the MEP stream gauging effort. Watershed data is based upon the MEP watershed modeling effort by USGS.

Stream Discharge Parameter	(Stream 1) **Herring River Discharge**[(a)] at Herring Run	(Summer) **Herring River Discharge**[(a)] at Herring Run	(Stream 2) **Herring River Discharge**[(a)] at Lothrop Road	(Summer) **Herring River Discharge**[(a)] at Lothrop Road	**Data Source**
Total Days of Record	365[(b)]	81[(c)]	365[(b)]	81[(c)]	(1)
Flow Characteristics					
Stream Average Discharge (m3/day) **	42,111	38,851	20,533	14,409	(1)
Contributing Area Average Discharge (m3/day)	40919		17792		(2)
Discharge Stream 2004-05 vs. Long-term Discharge	3%		13%		
Nitrogen Characteristics					
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.12	0.024	0.148	0.106	(1)
Stream Average Total N Concentration (mg N/L)	0.651	0.703	0.606	0.814	(1)
Nitrate + Nitrite as Percent of Total N (%)	18%	3%	24%	13%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	27.43	27.31	12.45	11.48	(1)
TN Average Contributing UN-attenuated Load (kg/day)	63.01	--	26.98	--	(3)
Attenuation of Nitrogen in Pond/Stream (%)	56%	--	54%	--	(4)

(a) Flow and N load to streams discharging to Herring River includes apportionments of Pond contributing areas.
(b) June 12, 2004 to June 11, 2005 for flow at Lothrop Road;
Dec. 22, 2005 to Dec. 21, 2006 (Herring Run) due to having to redo deployment
** Based on annual flow for 2004-2005 (Lothrop) and 2005-2006 (Herring Run)
(c) Average daily flow for summer months (June, July August) in 2004 (Lothrop) and 2006 (Herring Run).
(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 Herring River;
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.

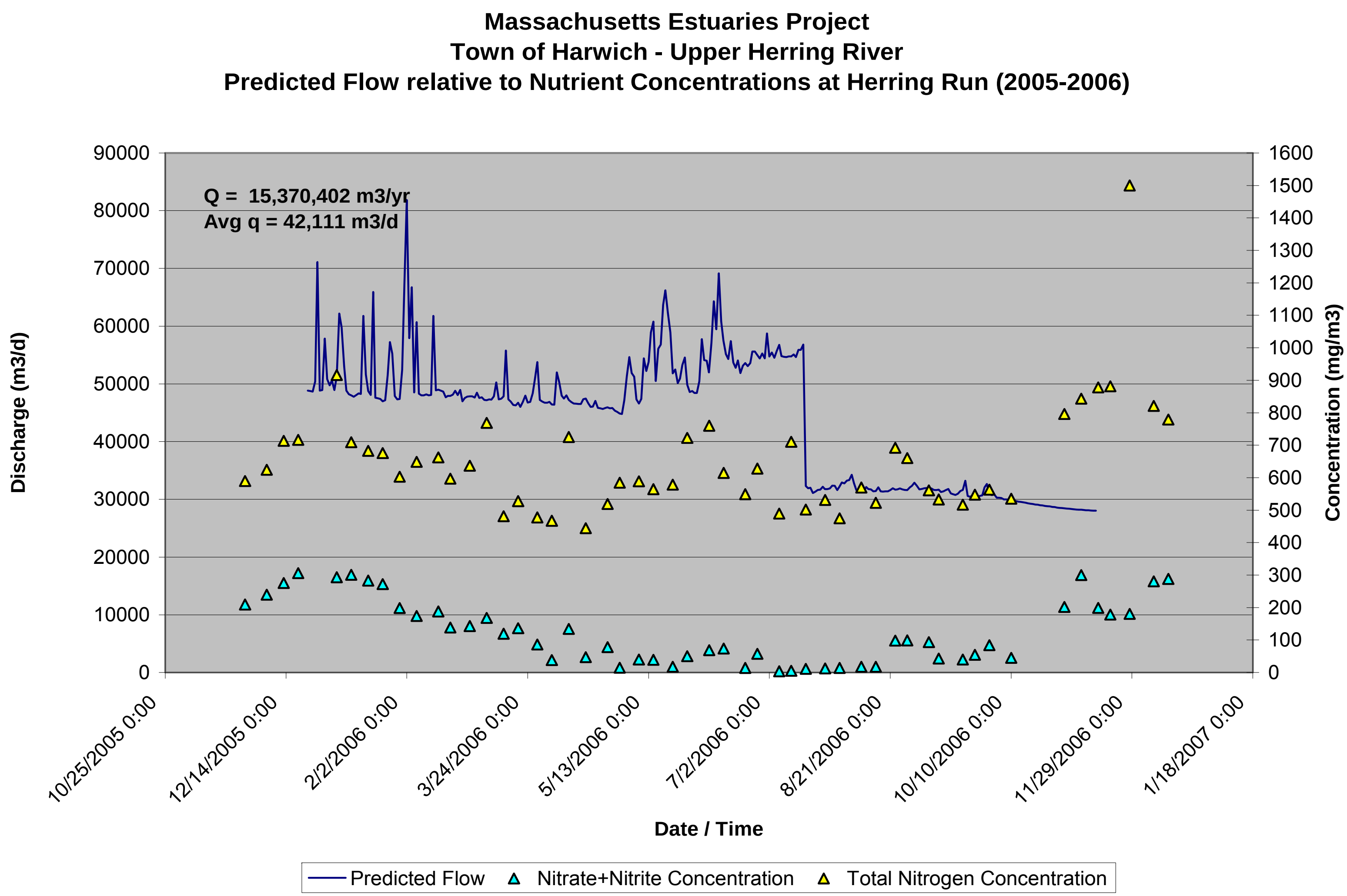

Figure IV-8. Herring River discharge at the herring run down gradient of West Reservoir (Stream 1). Discharge (solid blue line), nitrate+nitrite (blue triangle) and total nitrogen (yellow triangle) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to western portion of the upper wetland basin of the Herring River Estuary (Table IV-6). The large drop in flow in the stream gauge record was related to a significant drop in stage caused by manipulation of an up gradient water control structure.

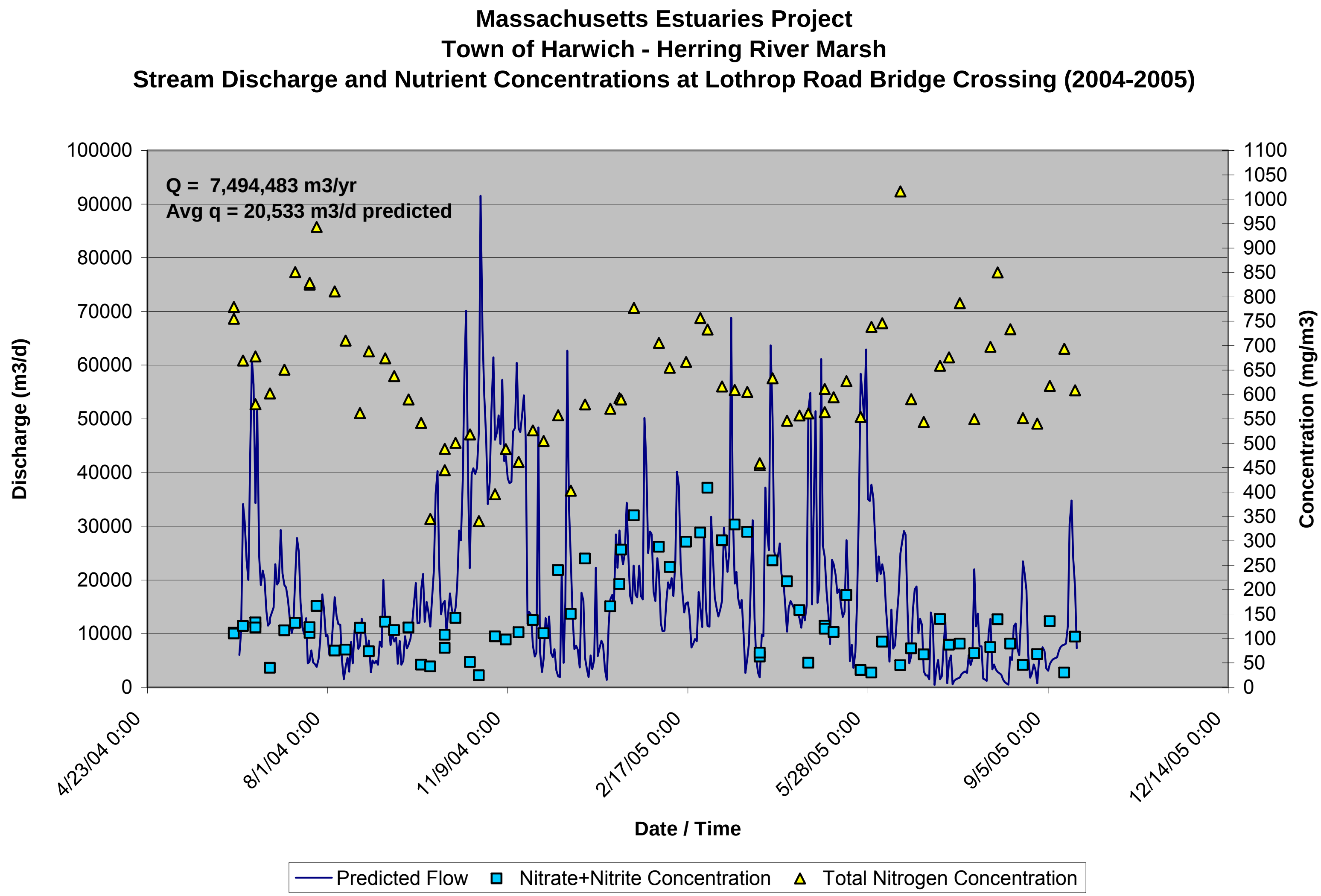

Figure IV-9. Discharge from tributary stream passing under Lothrop Road to Herring River (solid blue line), nitrate+nitrite (yellow squares) and total nitrogen (pink squares) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to the upper reaches of the Herring River marsh (Table IV-5).

Total nitrogen concentrations within the Herring River (stream 1) outflow were low to moderate, 0.651 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 27.43 kg/day and a measured total annual TN load of 10,013 kg/yr. In Herring River (stream 1), nitrate was not the predominant form of nitrogen (18%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was generally being taken up by plants within the reservoir or up gradient stream ecosystems and transformed to organic forms. This is supported by the measured concentrations of dissolved organic nitrogen (DON) and particulate organic nitrogen (PON), which constitute 77 percent of the measured total nitrogen load at the gauge location (5,491 kg/yr and 2,192 kg/yr respectively). However, since nearly one quarter of the nitrogen is in plant available forms, there remains the potential to increase removal during passage through up gradient freshwater systems (e.g. the reservoir or along the freshwater reach of the stream flowing into the freshwater reservoir from the upper portions of the watershed).

From the measured nitrogen load discharged by the Herring River (stream 1) to the estuary and the nitrogen load determined from the watershed based land use analysis, it appears that there is nitrogen attenuation of upper watershed derived nitrogen during transport to the estuary. Based upon lower total nitrogen load (10,013 kg yr^{-1}) discharged from the freshwater Herring River to the gauge site compared to that added by the various land-uses to the associated watershed (22,999 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 56% (i.e. 56% 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 hydraulic nature of the network of up gradient reservoir and ponds capable of attenuating nitrogen. The directly measured nitrogen loads in stream discharge waters was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

IV.2.3 Surface Water Discharge and Attenuation of Watershed Nitrogen: Herring River at Lothrop Road (Stream 2) Discharging to the Eastern Portion of the Wetland Dominated Upper Basin of the Herring River Estuary

Unlike the Herring River discharge to the western portion of the upper wetland basin of the Herring River Estuary, the stream discharge referred to by the MEP as Herring River @ Lothrop Road (e.g. Stream 2) for the purpose of this analysis, does not have an upgradient pond from which the stream discharges. Rather, this small stream appears to be groundwater fed and emanates from a small terminal wetland and associated wooded area northeast of Lothrop Road. The creek outflow leaving the wooded area travels through a small freshwater wetland just prior to passing under Lothrop Road through a relatively wide (~10 feet) bridge opening carrying freshwater and nitrogen to the eastern portion of the upper wetland basin of the Herring River Estuary. The stream outflow from the wetland up-gradient of the gauge provides for a direct measurement of the nitrogen attenuation for nitrogen transported from sources in the associated sub-watersheds. In addition, nitrogen attenuation also can occur within the wetland and must be considered in quantifying the attenuated load of nutrients to the tidal portion of the Herring River system. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the gauge site and the annual discharge of nitrogen and freshwater measured at the gauge site (Figure IV-7).

At the gauge site, a continuously recording vented calibrated water level gauge was installed to yield the level of water in the stream discharge that carries nitrogen load from the watershed to the eastern portion of the wetland dominated upper basin of the Herring River

Estuary. As this stream discharge is tidally influenced the gauge was located as far above the saltwater reach such that freshwater flow could be measured at low tide in the tidal portion of the Herring River. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average low tide salinity was determined to be 2.6 ppt, therefore, while most of the record showed low tide salinities at the gauge of 0.1 ppt, there were times when samples were tidally influenced. While the gauge location was deemed acceptable for making freshwater flow measurements, it was necessary to adjust portions of the flow record for slight tidal influence as has been the case in a few other stream systems in the MEP study area. Calibration of the gauge was checked monthly. The gauge on the stream was installed on June 11, 2004 and operated continuously for 18 months, such that two summer seasons were captured in the flow record.

Stream flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the stream site based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allowed for the determination of nitrogen mass discharged to the eastern portion of the upper basin of the Herring River Estuary (Figure IV-9 and Table IV-5). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at the gauge site.

The annual freshwater flow record for the Lothrup Road Stream (also referred by the MEP as Stream 2) as measured by the MEP was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from the stream at Lothrop Road was 13% above the long-term average modeled flows. Measured flow in the stream was obtained for one hydrologic year (June 2004 to June 2005). The average daily flow based on the MEP measured flow data was 20,533 m^3/day compared to the long term average flows determined by the USGS modeling effort (17,792 m^3/day).

The difference between the long-term average flow based on watershed recharge rates and the MEP measured flow were considered to be insignificant. The small difference between the modeled and measured freshwater flows is consistent with the gauge site capturing the appropriate up-gradient recharge (and loads).

Total nitrogen concentrations within the stream outflow were moderate, 0.606 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 12.45 kg/day and a measured total annual TN load of 4,545 kg/yr, about half that of the Herring River discharge. In the stream surface water system, nitrate was not the predominant form of nitrogen (24%), indicating that groundwater nitrogen (typically dominated by nitrate) is being taken up and transformed to organic forms by plants within the pond, wetland or stream ecosystems which intercept groundwater nitrogen up-gradient of the gauge.

From the measured nitrogen load discharged by the stream to the head of the eastern portion of the Herring River system and the nitrogen load determined from the watershed based land use analysis, it appears that there is significant nitrogen attenuation of upper watershed derived nitrogen during transport to the estuary, similar to the Herring River (fresh stream 1). Based upon lower nitrogen load (4,545 kg yr^{-1}) discharged from the freshwater stream compared to that added by the various land-uses to the associated watershed (9,848 kg yr^{-1}), the integrated attenuation in passage through wetland and wooded portions of the sub-watershed prior to discharge to the estuary is 54% (i.e. 54% of nitrogen input to watershed does not reach the estuary). However, since more than one quarter of the nitrogen is in plant available forms, there remains the potential to increase removal during passage through up-gradient freshwater pond and wetland systems. The measured level of attenuation compared to other streams evaluated under the MEP is expected given the hydrologic characteristics of the up gradient watershed with ponds and wetlands that contribute to nitrogen attenuation. The directly measured nitrogen loads from the stream were used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

Table IV-6. Summary of annual volumetric discharge and nitrogen load from the two streams (freshwater) discharging to the Herring River embayment system based upon the data presented in Figures IV-8 and IV-9 and Table IV-5.

Embayment System	Period of Record	Discharge (M^3/year)	Attenuated Load (Kg/yr)	
			Nox	TN
Herring River @ Herring Run MEP	December 22, 2005 to December 21, 2006	15,370,402	1851	10013
Herring River @ Herring Run CCC	Based on Watershed Area and Recharge	14,935,435	-	-
Herring River @ Lothrop Road MEP	June 12, 2004 to June 11, 2005	7,494,483	1112	4545
Herring River @ Lothrop Road CCC	Based on Watershed Area and Recharge	6,494,080	-	-

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

The overall objective of the benthic nutrient flux Survey was to quantify the summertime exchange of nitrogen, between the sediments and overlying waters throughout the Herring River Estuary. 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 this shallow marine ecosystem. 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 above sections, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the Herring River system predominantly in highly bioavailable forms from the surrounding upland watershed and more refractory forms in the inflowing tidal waters. If all of the nitrogen remained within the 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 watershed it is predominantly in the bioavailable form nitrate. This nitrate and other bioavailable forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton "particles". Most of these "particles" remain in the water column for sufficient time to be flushed out to a down gradient larger water body (like Nantucket Sound). However, some of these phytoplankton particles are grazed by zooplankton or filtered from the water by shellfish and other benthic animals and deposited on the bottom sediments. 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 associated nitrogen “load” become incorporated into the surficial sediments of the harbors.

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 bioavailable 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 salt marsh tidal creeks, like in the upper wetland dominated basin of Herring River (above Rt. 28 bridge), the sediments can be a net sink for nitrogen even during summer (e.g. Mashapaquit Creek Salt Marsh, West Falmouth Harbor; Centerville River Salt Marsh). Embayment basins can also be net sinks for nitrogen to the extent that they support relatively oxidized surficial sediments, such as found within much of the bordering region to the Lewis Bay main basin in nearby Barnstable. In contrast, regions of high deposition like Hyannis Inner Harbor (also in Barnstable), which is essentially a dredged boat basin and channel similar to Saquatucket Harbor and Wychmere Harbor in Harwich, typically support anoxic sediments with elevated rates of nitrogen release during summer months. The consequence of this deposition is that these basin sediments are unconsolidated, organic rich and sulfidic nature (MEP field observations). Similarly, tidal rivers on Cape Cod tend to show a low net release of nitrogen during summer, as seen in the tidal channels of Parker's River, Centerville River and Little and Great Rivers in the Waquoit Bay System, although a slight net nitrogen uptake can also occur.

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 Herring River Estuary. 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 Herring River Estuary, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected from 15 sites throughout the tidal river of the lower estuary and major wetland creeks of the upper estuary in August 2004 (Figure IV-10). In August 2010 four (4) additional sites were sampled within East Reservoir (aka. Herring River Pond) located in the uppermost reach of the eastern portion of the Herring River Estuary to expand the spatial coverage of the sediment nutrient regeneration measurements into this brackish tidal basin (Figure IV-11). All MEP nutrient flux cores were collected, incubated and assayed in the same manner in order to determine sediment-watercolumn exchanges of nitrogen. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample.

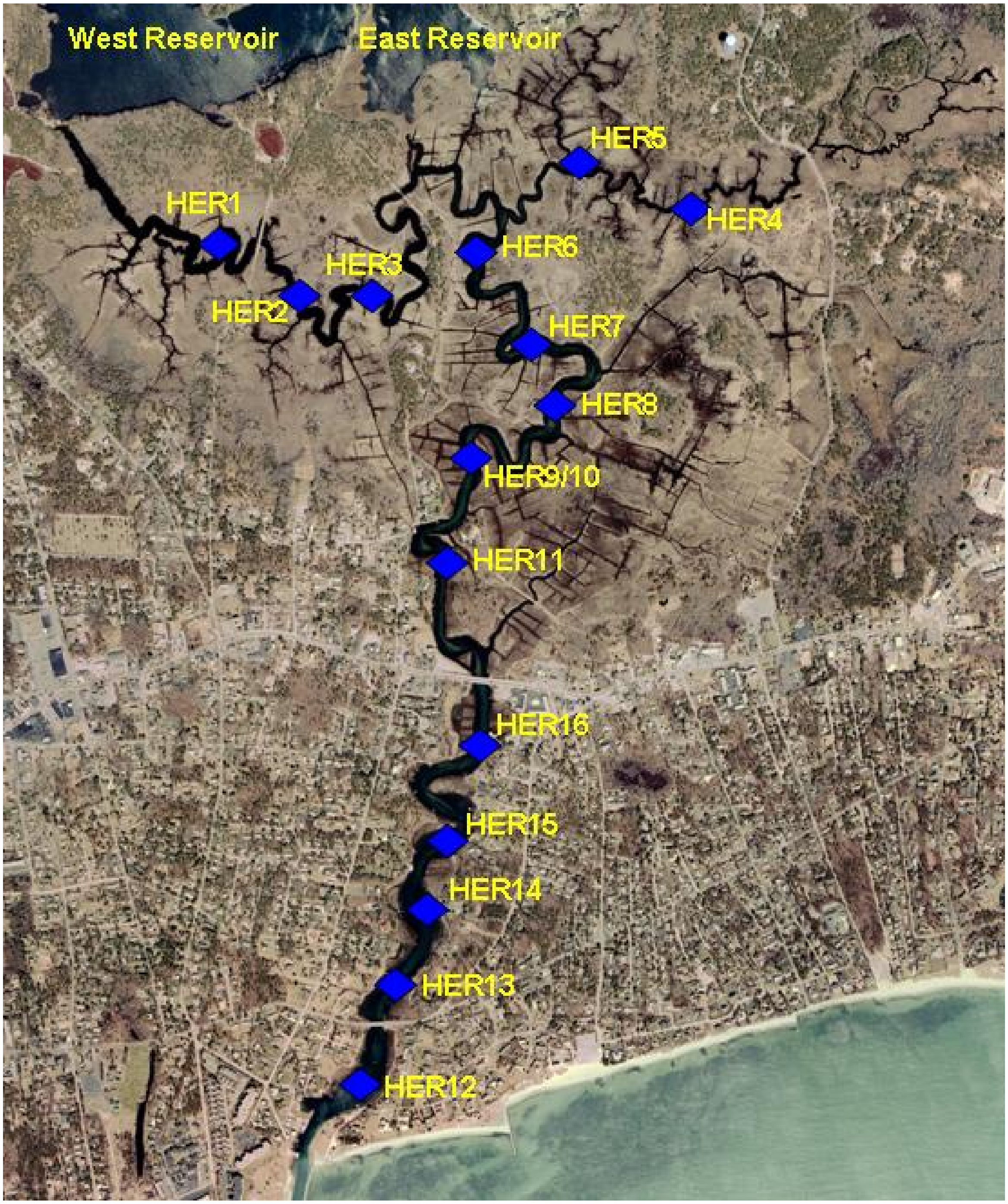

Figure IV-10. Sediment sampling locations (blue diamonds) within the Herring River Estuary, Town of Harwich MA. Sediment cores were collected by SCUBA diver for determination of nitrogen regeneration rates. Station numbers refer to identifications listed above and those in Table IV-7.

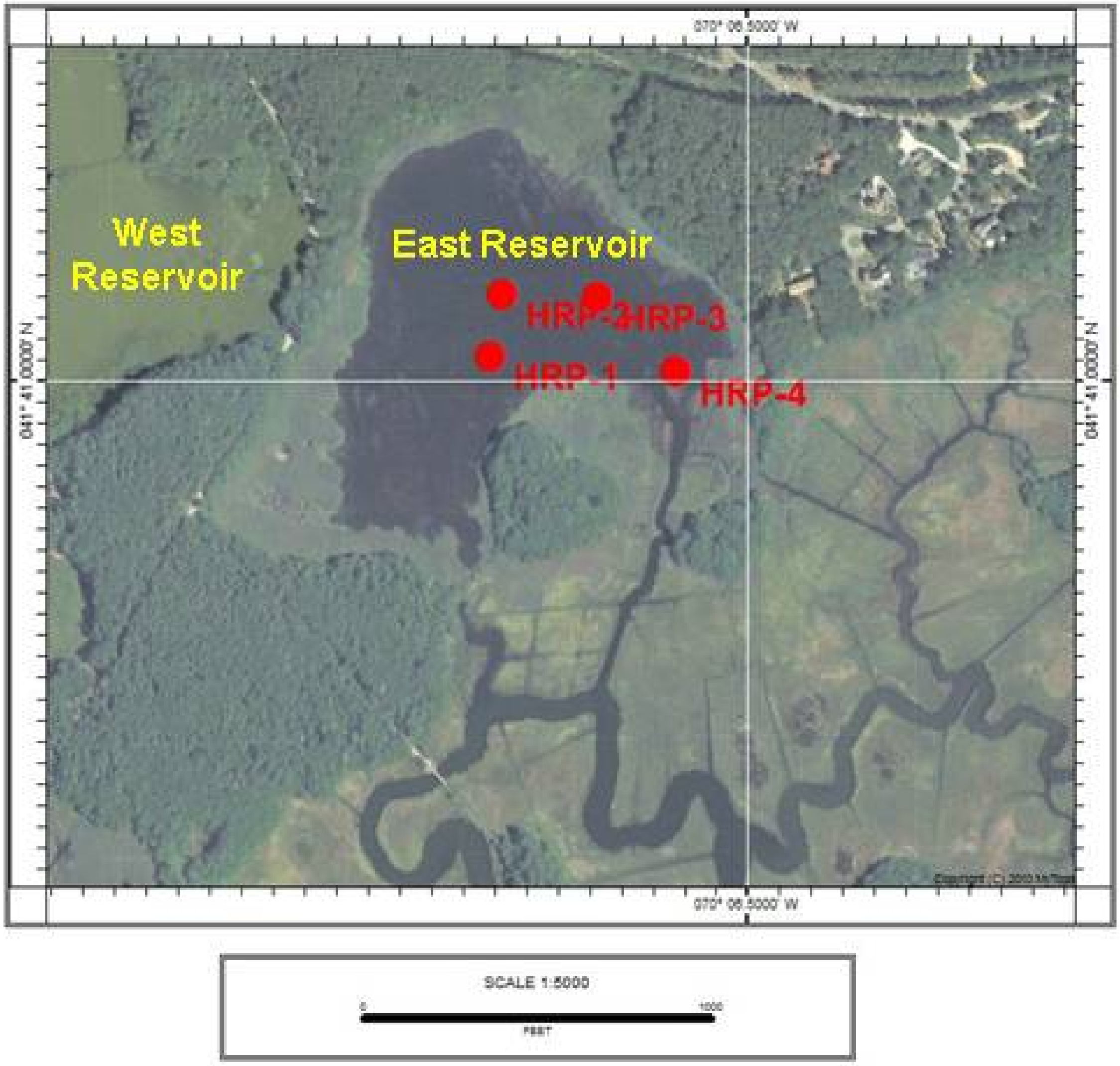

Figure IV-11. East Reservoir (aka. Herring River Pond) located in the uppermost reach of the eastern portion of the Herring River Estuary. Wetlands surrounding the basin are dominated by Phragmites and freshwater wetland plants. Sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above and in Table IV-7. No sediment sampling was undertaken in West Reservoir, as it is a freshwater pond.

Rates of nitrogen release were determined using undisturbed sediment cores incubated for 24 hours in temperature-controlled baths. Sediment cores (15 cm inside diameter) were collected by SCUBA divers and cores transported by small boat to a shore side field lab. Cores were maintained from collection through incubation at *in situ* temperatures. Bottom water was collected and filtered from each core site to replace the headspace water of the flux cores prior to incubation. The number of core samples from each site (Figures IV-10 and IV-11) per incubation were as follows:

Herring River Embayment System Benthic Nutrient Regeneration Cores

- HER-1 1 core *Upper Wetland Creek - West*
- HER-2 1 core *Upper Wetland Creek - West*
- HER-3 1 core *Upper Wetland Creek - West*
- HER-4 1 core *Upper Wetland Creek - East*
- HER-5 1 core *Upper Wetland Creek - East*
- HER-6 1 core *Upper Wetland Basin - Main Creek*
- HER-7 1 core *Upper Wetland Basin - Main Creek*
- HER-8 1 core *Upper Wetland Basin - Main Creek*
- HER-9 2 cores *Upper Wetland Basin - Main Creek*
- HER-10 1 core *Upper Wetland Basin - Main Creek*
- HER-11 1 core *Upper Wetland Basin - Main Creek*
- HER-12 1 core *Tidal River - Lower Main Channel*
- HER-13 1 core *Tidal River - Lower Main Channel*
- HER-14 1 core *Tidal River - Lower Main Channel*
- HER-15 1 core *Tidal River - Lower Main Channel*
- HER-16 1 core *Tidal River - Lower Main Channel*
- HRP-1 1 core *East Reservoir Basin*
- HRP-2 1 core *East Reservoir Basin*
- HRP-3 1 core *East Reservoir Basin*
- HRP-4 1 core *East Reservoir Basin*

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

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 (Allens Harbor Marine) the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and ortho-phosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining samples frozen (-20^{o}C) for assay of nitrate + nitrite (Cd reduction: Lachat Autoanalysis), and DON (D'Elia *et al.* 1977). Rates were determined from linear regression of analyte concentrations through time.

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

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

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 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, 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.

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

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

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

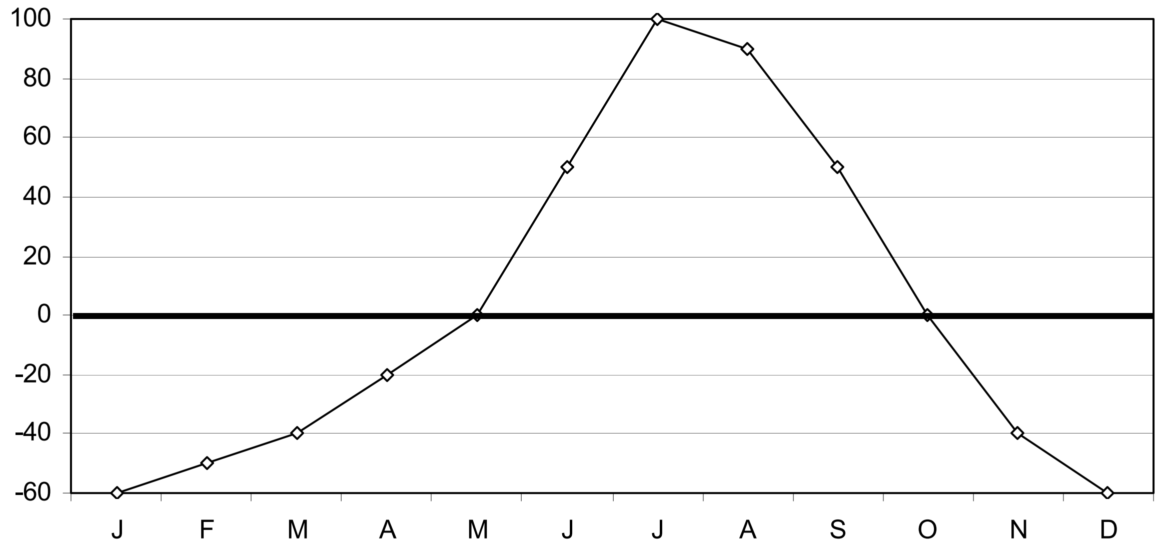

Figure IV-12 Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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

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

Sediment sampling was conducted throughout the main tidal channels and basins of the Herring River Estuary in order to obtain the spatial distribution of nitrogen regeneration rates required for parameterization of the water quality model. The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. For each site the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content and sediment type and an analysis of each site's tidal flow velocities. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within the different portions of the overall system.

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within each sediment site, the average summer particulate carbon and nitrogen concentration within the overlying water and the tidal velocities from the hydrodynamic model (Chapter 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 Herring River Estuary were comparable to other similar basins with similar structure and with similar configuration and flushing rates in southeastern. Massachusetts. In addition, the pattern of sediment N release was also similar to other systems, with the wetland basins and creeks.

The Herring River Estuary functions as a two component system, with the upper basin dominated by wetlands surrounding the tidal creeks and a shallow wetland pond, East Reservoir, and a lower "basin" structured as a tidal river with little bordering wetland and high tidal flows. The large tidal flows in the tidal river are the result of all of the tidal water that floods and ebbs in the large area of the upper basin (above Rt. 28) passing through the channel of the tidal river. The result is that during much of the flood tide, the tidal river waters have water quality very close to adjacent Nantucket Sound. The different components support somewhat different sediment-watercolumn exchanges, as seen in other Cape Cod estuaries. The brackish wetland dominated pond (East Reservoir) and the tidal creeks support a range of net nitrogen releases from a low rate of release in the pond (9.7 mg N m^{-2} d^{-1}) and the eastern wetland creek (10.5 mg N m^{-2} d^{-1}) to a low rate of uptake in the western creek (-10.2 mg N m^{-2} d^{-1}) and moderate rate of uptake in the larger main tidal creek (-17.9 mg N m^{-2} d^{-1}). Net nitrogen release has been measured in the upper reaches of some wetland dominated brackish estuarine basins on Cape Cod and Martha's Vineyard, such as the Back River Estuary (6.5 mg N m^{-2} d^{-1}), Mill Pond in Bass River (6.1 mg N m^{-2} d^{-1}) and Trapps Pond (9.2 mg N m^{-2} d^{-1}). However, a net nitrogen uptake is more typical, as seen in the western creek and main wetland channel (-10.2 & -17.9 mg N m^{-2} d^{-1}), as well as Lewis Pond channel in Parkers River (-6.2 mg N m^{-2} d^{-1}), upper salt marsh channel and Bumps River in Centerville River Estuary (-4.7 & -4.5 mg N m^{-2} d^{-1}), Mill Creek salt marsh basin in Lewis Bay (-14.3 mg N m^{-2} d^{-1}), and others. The tidal river portion of the estuary extending from the inlet to the Rt. 28 bridge showed net nitrogen release typical of other tidal rivers on Cape Cod, such as Parkers River below the wetlands (39.9 mg N m^{-2} d^{-1}), Little River in the Waquoit Bay Estuary (27.8 mg N m^{-2} d^{-1}), main channel of Bass River (1.8 mg N m^{-2} d^{-1}), mid and upper channel of The River in Pleasant Bay Estuary (12.0-14.3 mg N m^{-2} d^{-1}).

These rates are consistent with the depositional nature of these basins and the more nutrient enriched waters of the upper wetland areas and high flows in the tidal river. The overall pattern is consistent a number of estuaries of similar structure within the MEP region and other wetland dominated estuaries (MEP Cockle Cove Technical Memorandum-Howes et al. 2006).

Net nitrogen release rates for use in the water quality modeling effort for the main sections of the Herring River Estuary (Chapter VI) are presented in Table IV-7. There was a clear spatial pattern of sediment nitrogen flux, with net uptake of nitrogen in the main wetland creeks, with a low net release in the upper western wetland basins (which clearly have depositional areas) and in the tidal river with its oxidized sediments and aerobic watercolumn. The sediments within the Herring River Estuary showed nitrogen fluxes typical of similarly structured systems within the region and appear to be in balance with the overlying waters and the nitrogen flux rates consistent with the level of nitrogen loading to this system and its relatively high flushing rate.

Table IV-7. Rates of net nitrogen return from sediments to the overlying waters of the Herring River Estuary. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (see Chapter VI). Measurements represent July -August rates.

Location	Sediment Nitrogen Flux (mg N m^{-2} d^{-1})			i.d. *
	Mean	S.E.	# sites	
Herring River Estuary				
East Reservoir	9.7	9.8	4	HER- 17, 18, 19, 20
Upper Wetland Basin East Branch	10.5	12.2	2	HER- 5, 6
Upper Wetland Basin West Branch	-10.2	6.0	3	HER- 1,2, 3
Upper Wetland Basin Main Channel	-17.9	6.0	6	HER- 6, 7, 8, 9, 10, 11
Tidal River - Lower Basin Main Channel	9.6	12.0	5	HER- 12,13, 14, 15, 16

* Station numbers refer to Figure IV-10 and 11.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This section summarizes field data collection effort and the development of a hydrodynamic model for the Herring River estuary system in Harwich, Massachusetts. The final calibrated model of this system offers an understanding of water movement through the estuary, and provides the first step towards evaluating water quality, as well as a tool for later determining nitrogen loading “thresholds”. Nutrient loading data combined with measured environmental parameters within the system become the basis for an advanced water quality model based on total nitrogen concentrations. This type of model provides a tool for evaluating existing estuarine water quality parameters, as well as determining the likely positive impacts of various alternatives for improving overall estuarine health, facilitating the understanding how pollutant loadings into the estuary will affect the biochemical environment and its ability to sustain a healthy marine habitat.

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

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

A hydrodynamic study was performed for the Herring River system, which is located in the Town of Harwich, on Cape Cod. A section of a topographic map in Figure V-1 shows the general study area. The Herring River system has many attached sub-systems, with the five main sub-divisions: 1) the main river reach, 2) Upper Marsh, 3) West Reservoir, 4) East Reservoir, and 5) Lothrop Road Stream. The entire Herring River system has a surface coverage of 350 acres, including the attached sub-embayments. The upper marsh system is the largest sub-embayment of the estuarine system, with 295-acre coverage.

Circulation in the Herring River system is dominated by tidal exchange with Nantucket Sound. The River is connected to Nantucket Sound through a structured inlet. The western inlet jetty is approximately 930 feet long, extending 160 ft beyond the beach into the Sound. The eastern jetty is offset from the western jetty and is approximately 870 feet long. Over the length of the Herring River estuarine system, there is considerable attenuation of the tide range.

Between the inlet and the upper marsh, the average tide range is reduced from 3.7 feet to 3.0 feet, a reduction of 0.7 feet or 19%. This reduction is caused by frictional losses along the main channel and within the marsh system.

Figure V-1. Topographic map detail of the Herring River, from Nantucket Sound to the head of the system.

This hydrodynamic study proceeded as two component efforts. In the first portion of the study, bathymetry, tide, and circulation velocity data were collected in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the study. The bathymetry survey of Herring River was performed to determine the present variation of embayment and channel depths throughout the system. In addition to bathymetry, tides were recorded at six locations within the River system for at least a complete lunar month (29.5 days). These tide data were necessary to run and calibrate the hydrodynamic model of the system. Finally, an Acoustic Doppler Current Profiler (ADCP) survey was completed during a single tide cycle to measure ebb and flood velocities across two channel transects. The ADCP data were used to compute system flow rates and to provide an independent means of verifying the performance of the hydrodynamic model.

A numerical hydrodynamic model of the Herring River system was developed in the second portion of this analysis. Using the bathymetry survey data, a model grid mesh was generated for use with the RMA-2 hydrodynamic code. The tide data from offshore, in Nantucket Sound, were used to define the open boundary conditions that drive the circulation of the model at the system inlet, and data from the five TDR stations 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 computer model of Herring River was used to compute the flushing rates of selected sub-sections. Though water quality in an embayment cannot be directly inferred by use of computed flushing rates alone, they can serve as useful indicators of embayment flushing performance relative to other areas in the same system. The ultimate utility of this hydrodynamic model is as input into a constituent transport model, where water quality constituents like nitrogen are modeled to determine the real water quality dynamics of a system.

V.2 DATA COLLECTION AND ANALYSIS

The field data collection portion of this study was performed to characterize the physical properties of the Herring River estuary. Bathymetry were collected throughout the system so that it could be accurately represented in the computer hydrodynamic model and water quality model of the system. In addition to the bathymetry, tide data also were collected at six locations, to run the circulation model with real tides, and also to calibrate and verify its performance.

V.2.1 Bathymetry Data Collection

Bathymetry data in Herring River were collected during June of 2005. The June 2005 survey employed a single-beam acoustic fathometer mounted to a motor boat. Positioning data were collected using a differential GPS. The survey transects were densest in the vicinity of the inlets, were the greatest variability in bottom bathymetry was expected. Bathymetry in the inlet is important from the standpoint that it has the most influence on tidal circulation in and out of the estuary. The survey was conducted from a small boat with an installed precision fathometer (with a depth resolution of approximately 0.1 foot), coupled together with a differential GPS to provide position measurements accurate to approximately 1-3 feet. A digital output produced a single data set consisting of water depth as a function of geographic position (Massachusetts State Plane). Survey paths and measured depths are shown in Figure V-2. The resulting bathymetric surface created by interpolating the data to a finite element mesh is shown in Figure V-3. All bathymetry was tide corrected, and referenced to the NGVD 29 vertical datum, using survey benchmarks located in the project area.

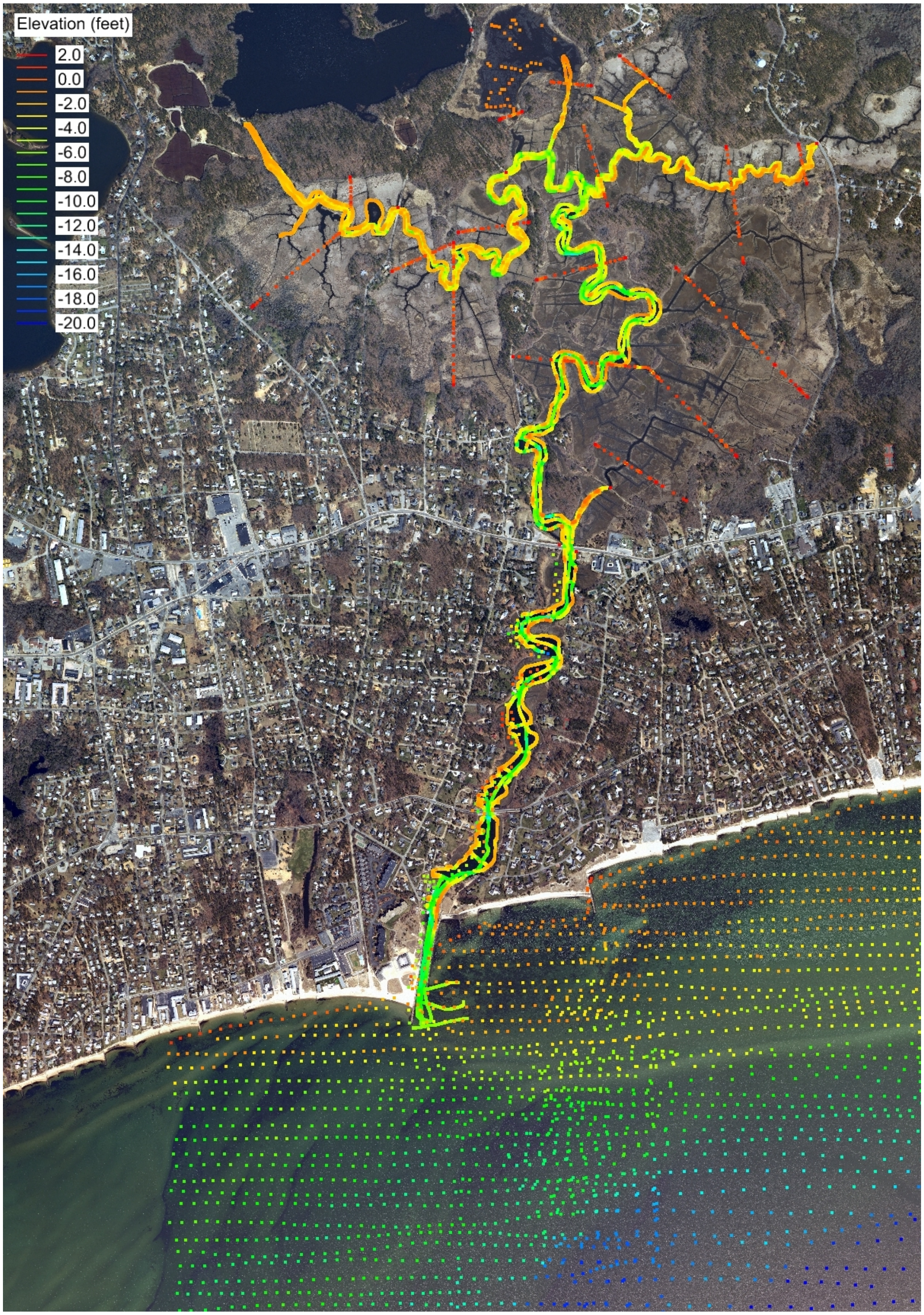

Figure V-2. The data coverage from the bathymetry surveys of Herring River.

The raw measured water depths were merged with water surface elevation measurements to determine bathymetric elevations relative to the NGVD 1929 vertical datum. Once rectified, the finished processed data were archived as 'xyz' files containing x-y horizontal position (in Massachusetts State Plan 1983 coordinates) and vertical elevation of the bottom (z). These xyz files were then interpolated into the finite element mesh used for the hydrodynamic simulations. The final processed bathymetric data from the survey are presented in Figure V-3.

Results from the survey show that the deepest point within the river is located immediately downstream from the Lower County Road Bridge, and is -19.6 ft NGVD. Other deep regions of the River system include the inside bends on the channel where flow accelerate due to the narrowing of the channel. The channels within the system are generally shallow and decrease in depth progressing up the system.

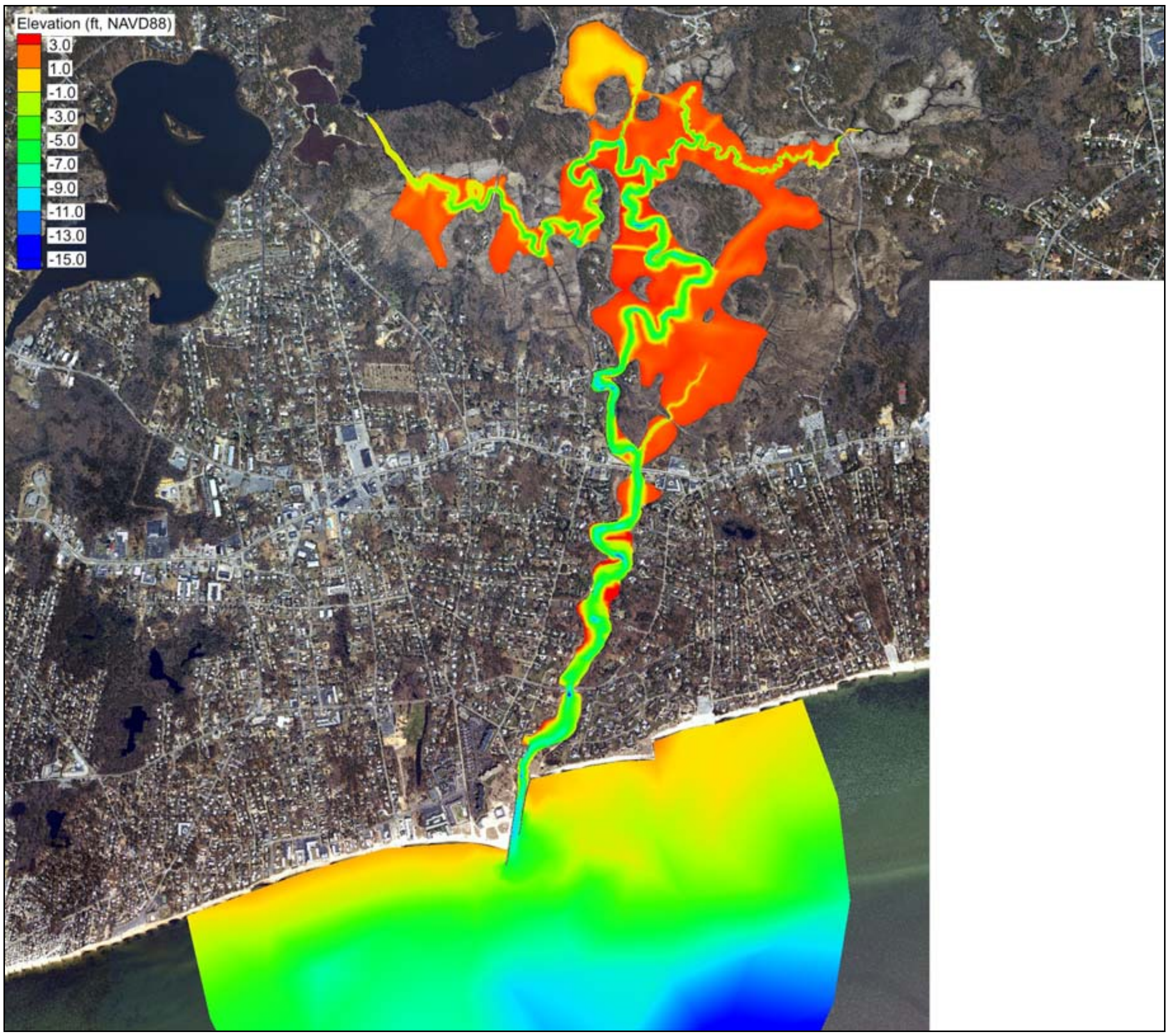

Figure V-3. Plot of interpolated finite-element grid bathymetry of the Herring River system, shown superimposed on 2005 aerial photos of the system locale. Bathymetric contours are shown in color at one-foot intervals.

A secondary topographic survey of the marsh was conducted in December of 2010 to characterize the elevations of the upper marsh system. The survey was conducted by Surveying and Mapping Consultants (SMC) utilizing GPS technology operating in a real-time mode referenced to the KeyNetGPS Virtual Reference System. The data was collected using the horizontal datum referenced to the North American Datum of 1983 (NAD 83/CORS) and the vertical datum was referred to the North American Vertical Datum of 1988 (NAVD88). The nominal horizontal and vertical accuracy for the KeyNetGPS network is ±0.01 meters (0.03 feet) and ±0.02 meters (0.07 feet) respectively. The surveyed transect lines are shown in Figure V-2.

Offshore of Herring River within Nantucket Sound, data from National Oceanic and Atmospheric Administration (NOAA) GEODAS database was utilized to characterize the offshore bathymetry. The data is shown in Figure V-2. Once all that datasets were collected, the vertical datum for the bathymetric and topographic datasets was converted from their base datum to NAVD88 for the development of the hydrodynamic grid.

V.2.2 Tide Data Collection and Analysis

Tide data records were collected at six stations in the Herring River estuary: 1) Lower County Road, 2) Route 28, 3) Salt Meadow Lane, 4) North Road, 5) Bells Neck Road, and 6) Lothrop Road. The locations of the stations are shown in Figure V-4. The Temperature Depth Recorders (TDR) used to record the tide data were deployed for a 30-day period beginning March 24, 2011. The elevation of each gauge was surveyed relative to NAVD88. Data from the gauges inside the system were used to calibrate the model.

Plots of the tide data from six representative gauges are shown in Figure V-5, for the entire 30-day deployment. The neap-to-spring variation in tide can be seen in these plots. From the plot of the data from offshore Herring River, the tide reaches its maximum spring tide range of approximately 5.5 feet around April 18. About seven days earlier the neap tide range is smaller, approximately 3.8 feet.

A visual comparison in Figure V-6 between tide elevations at six stations in Herring River shows that there is a reduction in the tide range as the tide propagates to the upper reaches of the system. The loss of amplitude with distance from the inlet is described as tidal attenuation. Frictional mechanisms dissipate tidal flow energy, resulting in a reduction of the height of the tide. Tide attenuation is accompanied by a time delay (or phase lag) in the time of high and low tide (relative to the offshore tide), which becomes more pronounced farther into an estuary. The tide lag is greatest at Bells Neck Road, as seen in Figure V-6, where high tide occurs approximately 85 minutes after high tide in Nantucket Sound.

Standard tide datums were computed from the 30-day records. 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 Water (MHHW) and Mean Lower Low Water (MLLW) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. The Mean Tide Level (MTL) is simply the mean of MHW and MLW.

As the tide propagates from Nantucket Sound to the upper reaches of the system attenuation of the tide occurs. This is observed as a reduction in the tide range and also as a delay in the time of high and low tide during each tide cycle. The mean tide range in Nantucket Sound is 3.7 feet. At Lothrop Road the mean tide range is reduced to 2.7 feet by frictional losses along the length of the River.

The tides in Nantucket Sound are semi-diurnal, meaning that there are typically two tide cycles in a day. There is usually a small variation in the level of the two daily tides. This variation can be seen in the differences between the MHHW and MHW, as well as the MLLW and MLW levels.

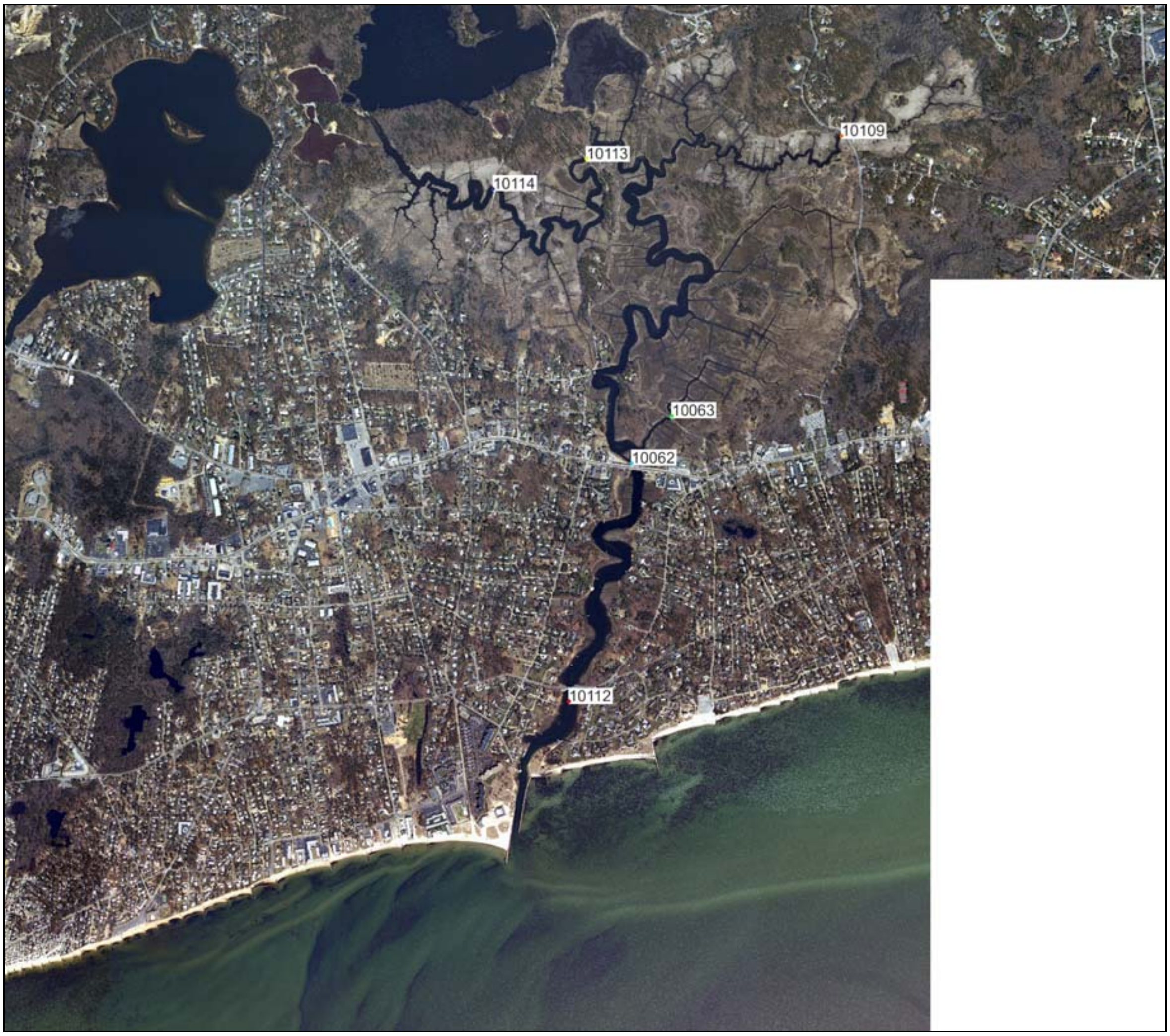

Figure V-4. Map of the study region identifying locations of the tide gauges used to measure water level variations throughout the system. Six (6) gauges were deployed for the 30-day period between March 24, and April 22, 2011. The colored circles represents the approximate locations of the tide gauges: (10112) represents the gage below Lower County Road (Offshore), (10062) above Route 28, (10063) above Salt Meadow Lane, (10113) above North Road, (10114) above Bells Neck Road, and (10109) just below Lothrop Road.

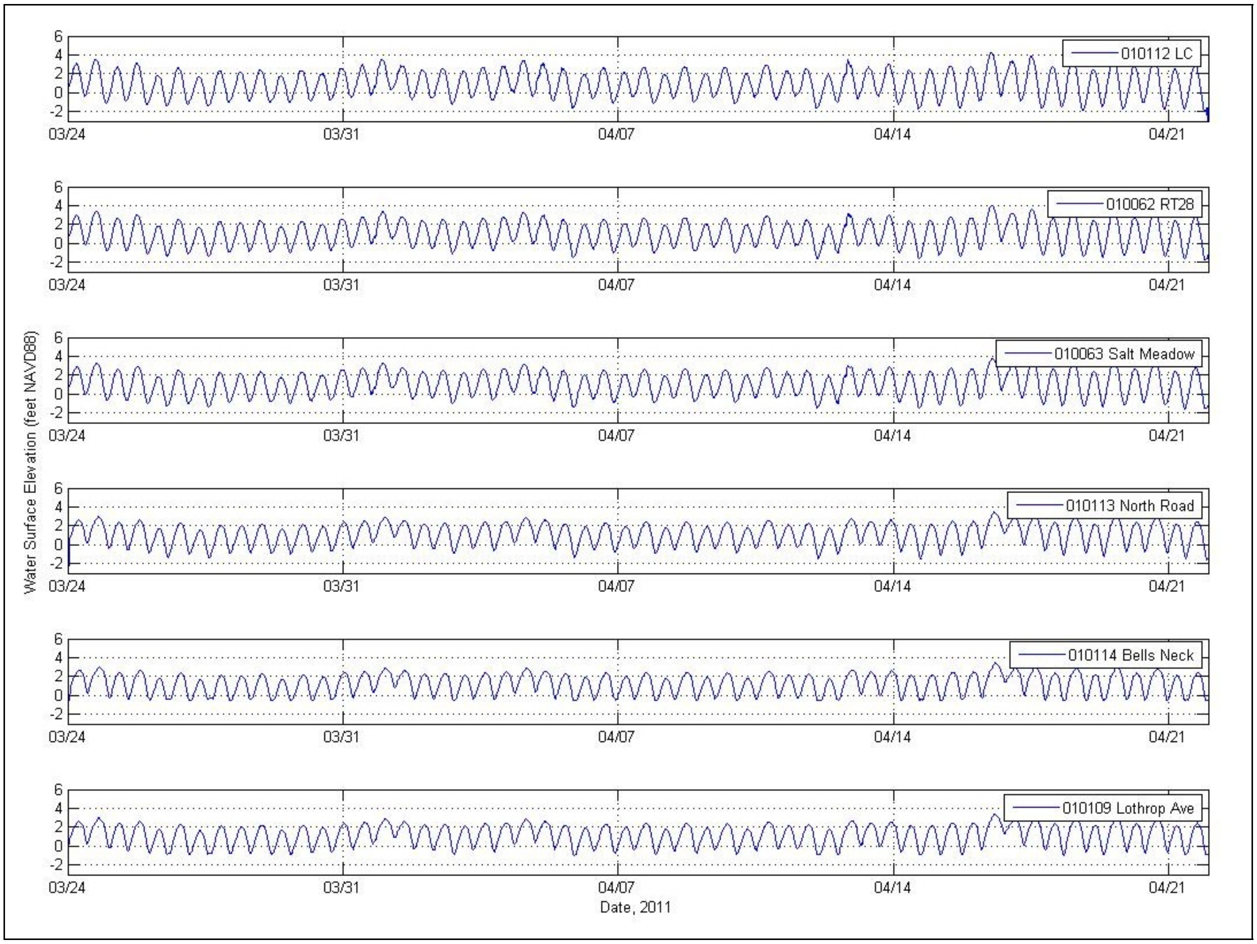

Figure V-5. Plots of observed tides for the Herring River system, for the 30-day period between March 24 and April 22, 2011. The top plot shows tides below the bridge at Lower County Road. Tides recorded above Route 28, Salt Meadow Lane, North Road, Bells Neck Road, and below Lothrop Road are also shown. All water levels are referenced to the North American Vertical Datum of 1988 (NAVD).

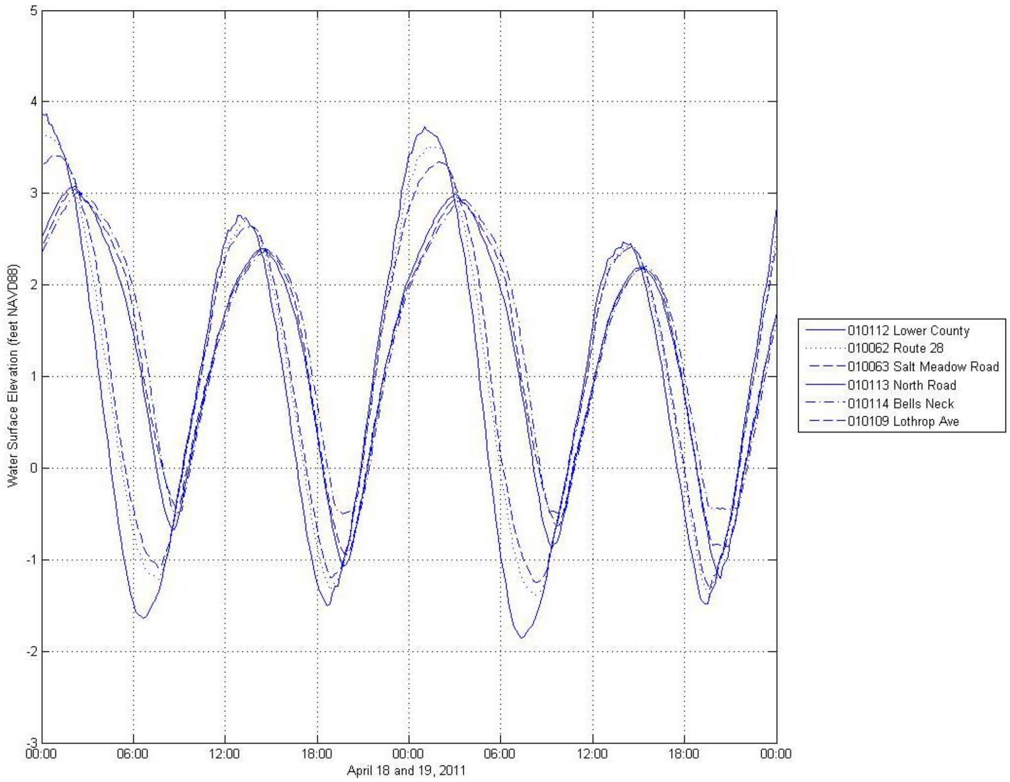

Figure V-6. Plot showing two tide cycles tides at three stations in the Herring River system plotted together. Demonstrated in this plot is the frictional damping effect caused by flow restrictions along the river's length. The damping effects are seen only as a lag in time of high and low tides from Nantucket Sound. The maximum time lag of low tide between the Sound and Bells Neck Road in this plot is 85 minutes.

Table V-1. Tide datums computed from a 28-day period from the tide records collected in the Herring River system. Datum elevations are given relative to NAVD88.

Tide Datum	Lower County Road	Route 28	Salt Meadow Lane	North Road	Bells Neck Road	Lothrop Road
Maximum Tide	4.28	4.04	3.77	3.44	3.51	3.47
MHHW	3.04	2.94	2.86	2.59	2.61	2.60
MHW	2.72	2.66	2.60	2.39	2.38	2.40
MTL	0.89	0.93	0.93	0.94	0.90	1.05
MLW	-0.95	-0.80	-0.73	-0.50	-0.59	-0.31
MLLW	-1.17	-1.01	-0.93	-0.67	-0.84	-0.42
Minimum Tide	-1.86	-1.65	-1.57	-0.98	-1.48	-0.53

A more thorough harmonic analysis of the tidal time series was 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 a quantitative assessment of the relative influence of non-tidal, or residual, processes (such as wind forcing) on the hydrodynamic characteristics of the 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 therefore the sum of several individual tidal constituents, with a particular amplitude and frequency. For demonstration purposes a graphical example of how these constituents add together is shown in Figure V-7. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-2 presents the amplitudes of eight tidal constituents in the Herring River system.

The M_2, or the familiar twice-a-day lunar semi-diurnal tide, is the strongest contributor to the signal with an amplitude of 1.7 ft at Lower County Road. The total range of the M_2 tide is twice the amplitude, or 3.4 ft for Lower County Road. 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), results from frictional attenuation of the M_2 tide in shallow water. The M_4 has an amplitude of 0.11 feet near the system inlet, but is slowly increases progressing towards the head of the system. The M_6 has a very small amplitude in the system (less than 0.1 feet at all gauge stations). The M_2 decreases in the upper reaches of the system as energy is lost through attenuation.

For all the other included constituents, except for the fortnightly M_{sf}, amplitudes decrease with distance into the system. The other major tide constituents also show little variation across the system. The diurnal tides (once daily), K_1 and O_1, possess amplitudes of approximately 0.2 feet. Other semi-diurnal tides, the S_2 (12.00 hour period) and N_2 (12.66-hour period) tides, contribute significantly to the total tide signal, with amplitudes of 0.2 feet and 0.4 feet, respectively. The M_{sf} is a lunarsolar fortnightly constituent with a period of approximately 14 days, and is the result of the periodic conjunction of the sun and moon, and has an amplitude of 0.2 ft.

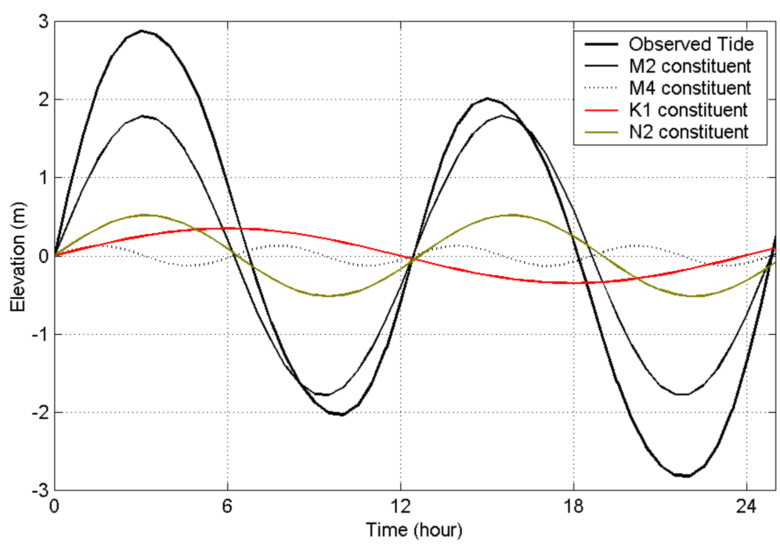

Figure V-7. Example of an observed astronomical tide as the sum of its primary constituents.

Table V-2. Major tidal constituents determined for gauge locations in Herring River, March 24 through April 22, 2011.								
	Amplitude (feet)							
Constituent	M_2	M_4	M_6	S_2	N_2	K_1	O_1	M_{sf}
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Lower County Road	1.70	0.11	0.06	0.24	0.49	0.21	0.26	0.20
Route 28	1.60	0.10	0.04	0.21	0.45	0.19	0.25	0.22
Salt Meadow Lane	1.54	0.12	0.03	0.19	0.43	0.18	0.24	0.22
North Road	1.30	0.16	0.04	0.13	0.33	0.17	0.21	0.26
Bells Neck Road	1.20	0.14	0.02	0.13	0.30	0.19	0.20	0.24
Lothrop Road	1.25	0.17	0.03	0.12	0.31	0.18	0.20	0.25

Though there is little change in constituent amplitudes across the length of the main basin of the River, the phase change of the tide is easily seen from the results of the harmonic analysis. Table V-3 shows the delay of the M_2 at different points in the Herring River system, relative to the timing of the M_2 constituent in Nantucket Sound, offshore the inlet. The analysis of the data from Bells Neck Road show that there is a 85 minute delay between the inlet and the farthest reach of the system. Compared to other locations instrumented in this study, Bells Neck Road shows the greatest tidal attenuation (Figure V-8).

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 original water elevation time series for the Herring River system is presented in Table V-4 compared to the energy content of the astronomical tidal signal (re-created by summing the contributions from the 23 constituents determined by the harmonic analysis). Subtracting the tidal signal from the original elevation time series resulted with the non-tidal, or residual, portion of the water elevation changes. The energy of this non-tidal signal is compared to the tidal signal, and yields a quantitative measure of how important these non-tidal physical processes can be to hydrodynamic circulation within the estuary. Figure V-8 shows the comparison of the measured tide from North Road, with the computed astronomical tide resulting from the harmonic analysis, and the resulting non-tidal residual.

Table V-3. M_2 tidal constituent phase delay for gauge locations in the Herring River system, determined from measured tide data.	
Station	Delay (minutes)
Route 28	15.0
Salt Meadow Lane	20.3
North Road	68.5
Bells Neck Road	85.2
Lothrop Road	73.1

Table V-4 shows that the variance of tidal energy decreases for stations that are farther from the inlet. The analysis also shows that tides are responsible for more than 90% of the water level changes for all gauges in the Bass River system. The remaining variance was the result of atmospheric forcing, due to winds, or barometric pressure gradients.

Table V-4. Percentages of Tidal versus Non-Tidal Energy for stations in Herring River, March to April 2011.

TDR Location	Total Variance (ft^2)	Tidal (%)	Non-tidal (%)
Lower County Road	1.85	93.0	7.0
Route 28	1.66	91.8	8.2
Salt Meadow Lane	1.54	91.2	8.8
North Road	1.15	84.8	15.2
Bells Neck Road	1.21	85.8	14.2
Lothrop Road	1.06	84.8	15.2

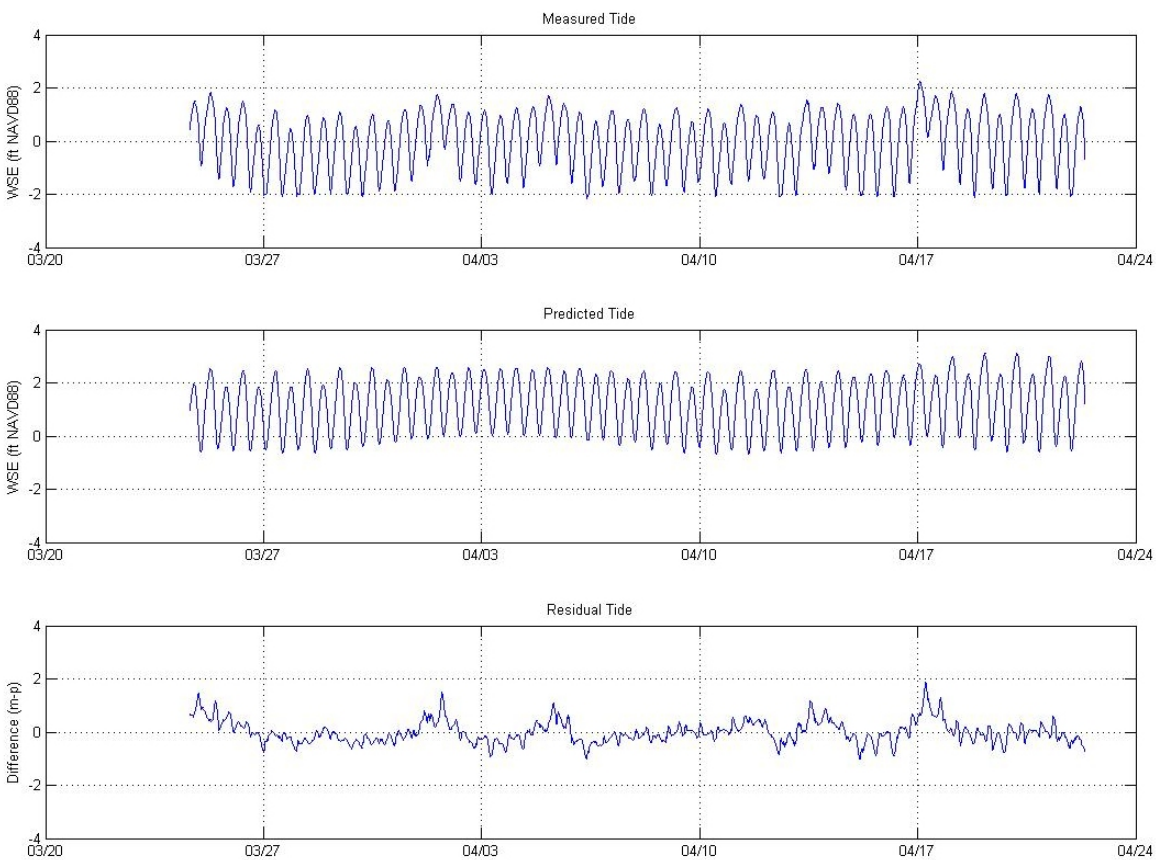

Figure V-8. Plot showing the comparison between the measured tide time series (top plot), and the predicted astronomical tide (middle plot) computed using the 23 individual tide constituents determine in the harmonic analysis of the North Road gauge data. The residual tide shown in the bottom plot is computed as the difference between the measured and predicted time series (r=m-p).

V.3 HYDRODYNAMIC MODELING

For modeling of Herring River, Applied Coastal utilized a state-of-the-art computer model to evaluate tidal circulation and flushing in this system. The particular model employed was the RMA-2 model developed by Resource Management Associates (King, 1990). It is a two-dimensional, depth-averaged finite element model, capable of simulating transient hydrodynamics. The model is widely accepted and tested for analyses of estuaries or rivers. Applied Coastal staff members have utilized RMA-2 for numerous flushing studies on Cape Cod and the Islands, including Bass River, Swan Pond River, Pleasant Bay (Howes, *et al*, 2006), Falmouth "finger" Ponds (Ramsey, *et al*, 2000), and Barnstable Harbor (Wood, *et al*, 1999), and Three Bays (Howes, *et al*, 2005).

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 a Brigham Young University through a package called the Surfacewater Modeling System or SMS (BYU, 1998). Graphics generated in support of this report primarily were generated within the SMS modeling package.

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

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

V.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 2005 color digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition was specified at the inlet to the river system based on the tide gauge data collected at Lower County Road. 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. 2005 digital aerial orthophotos and recent bathymetry survey data were imported to SMS, and a finite element grid was created to represent the estuary. The aerial photographs were used to determine the land boundary of the system. Bathymetry data were interpolated to the developed finite element mesh of the system. The completed grid consists of 13,554 nodes, which describe 5,566 total 2-dimensional (depth averaged) quadratic elements, and covers 1,230 acres. The maximum nodal depth is -32.9 ft (NGVD). This deepest depth occurs at the boundary of the model within Nantucket Sound. The completed grid mesh of the Herring River system is shown in Figure V-9, and grid bathymetry was shown previously in Figure V-3.

The finite element grid for the system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties throughout the Herring River estuarine system. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary. Grid resolution was governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability of the system. Relatively fine grid resolution was employed where complex flow patterns were expected. For example, smaller node spacing in inlet and along the main river channel was designed to provide a more detailed analysis in these regions of rapidly varying flow. Widely spaced nodes were often employed in areas where flow patterns are not likely to change dramatically, such as in the attached marsh plains in the upper portion of the system. Appropriate implementation of wider node spacing and larger elements was used to reduce computer run time with no sacrifice of accuracy.

V.3.2.2 Boundary condition specification

Three types of boundary conditions were employed for the RMA-2 model of the Herring River system: 1) "slip" boundaries, 2) tidal elevation boundaries, and 3) flow boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. Tidal boundary conditions were specified at the inlet from Nantucket Sound. Flow boundaries were used to describe the freshwater flow entering the system from West Reservoir and Lothrop Road Stream.

The rise and fall of the tide in Nantucket Sound is the primary driving force for estuarine circulation in this system. Dynamic (time-varying) model simulations specified a new water surface elevation at the model's offshore open boundary every model time step of 10 minutes, which corresponds to the time step of the TDR data measurements.

V.3.2.3 Calibration

After developing the finite element grid, and specifying boundary conditions, the model for the Herring River system was calibrated. The calibration procedure ensures that the model predicts accurately what was observed in nature during the field measurement program. Numerous model simulations are required (typically 10+) for an estuary model, specifying a range of friction and eddy viscosity coefficients, to calibrate the model.

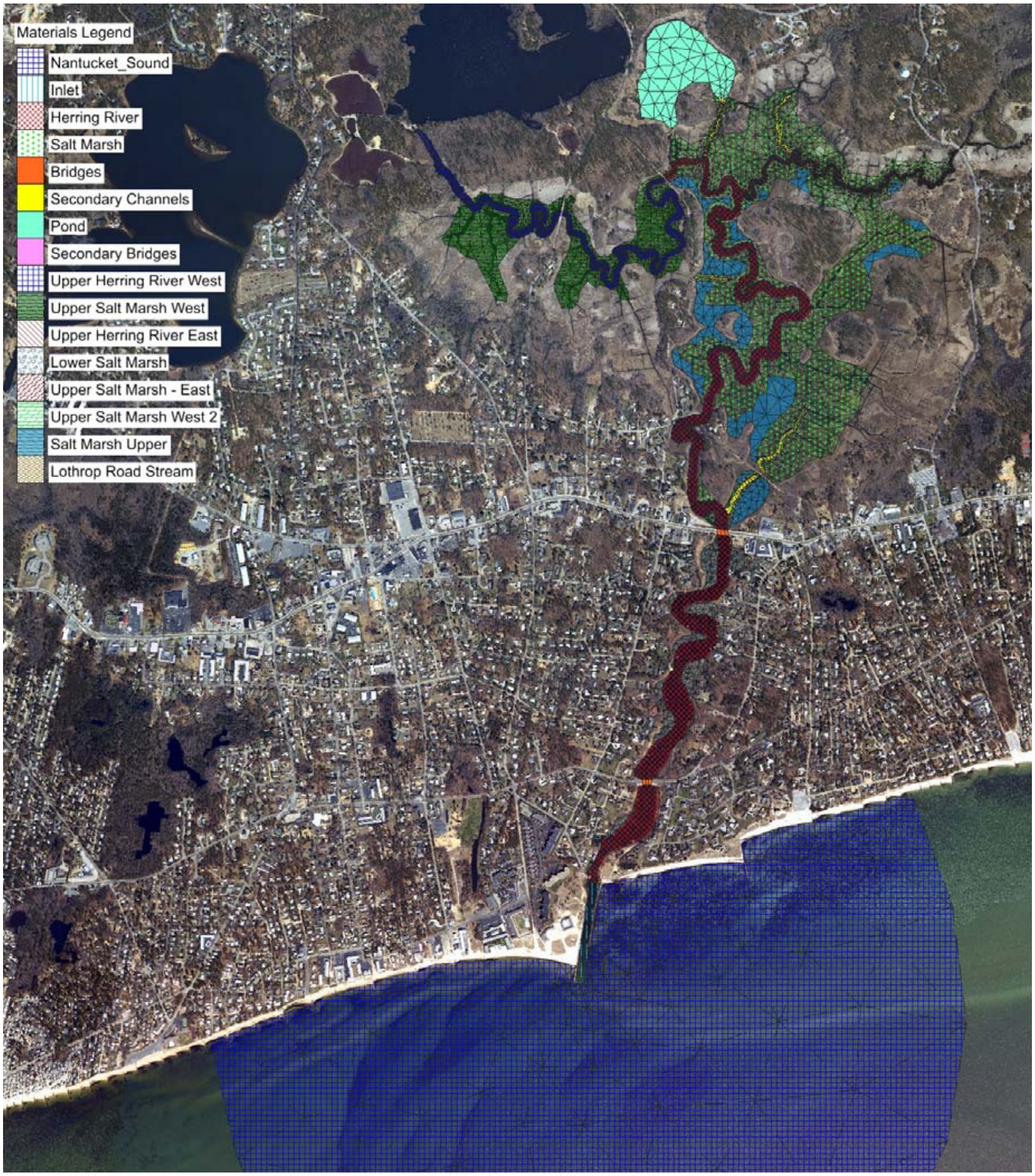

Figure V-9. Plot of hydrodynamic model grid mesh for the Herring River estuarine system of Harwich, Massachusetts. Color patterns designate the different model material types used to vary model calibration parameters and compute flushing rates.

Calibration of the hydrodynamic model required a close match between the modeled and measured tides in each of the sub-embayments where tides were measured (i.e., from the TDR deployments). Initially, the model was calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement was achieved, a six lunar-day period (12 tide cycles) was modeled to calibrate the model based on dominant tidal constituents discussed in Section V.3.2. The five-day period was extracted from a longer simulation to avoid effects of model spin-up, and to focus on average tidal conditions. Modeled tides for the calibration time period were evaluated for time (phase) lag and height damping of dominant tidal constituents.

The calibration was performed for a six-day period beginning April 6, 2011 at 0000 EDT. This representative time period selected represents the transition between neap tide and spring tide range of conditions (bi-weekly minimum and maximum tidal ranges, respectively). The period was selected to provide average tidal forcing conditions for model calibration and the flushing analysis.

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

V.3.2.3.1 Friction coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where velocities are relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude damping and phase delay of the tidal signal. Friction is approximated in RMA-2 as a Manning coefficient, and is applied to grid areas by user specified material types. Initially, Manning's friction coefficient values of 0.025 were specified for all element material types. This values corresponds to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) (Henderson, 1966).

During calibration, friction coefficients were incrementally changed throughout the model domain. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within the estuary system. Manning's values for different bottom types were initially selected based ranges provided by the Civil Engineering Reference Manual (Lindeburg, 1992), and values were incrementally changed when necessary to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-5.

V.3.2.3.2 Turbulent exchange coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swifter, such as inlets and other channel constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). Typically, model turbulence coefficients were set between 75 and 200 lb-sec/ft^2. In most cases, the Herring River system was relatively insensitive to turbulent exchange coefficients. The exception was the marsh plains, where higher exchange coefficient values (100 lb-sec/ft^2) were used to ensure numerical stability in these areas characterized by shallow transitioning flows.

V.3.2.3.3 Marsh porosity processes

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

Table V-5. Manning's Roughness coefficients used in simulations of modeled sub-embayments. These embayment delineations correspond to the material type areas shown in Figure V-9.

System Embayment	Bottom Friction
Nantucket Sound	0.026
Inlet	0.027
Herring River	0.027
Salt Marsh	0.033
Pile supported bridges	0.028
Marsh Channels	0.028
East Reservoir	0.027
Secondary Bridges	0.028
Upper Herring River - West	0.028
Upper Salt Marsh - West	0.033
Upper Herring River - East	0.028
Salt Marsh below Rt. 28	0.037
Upper Salt Marsh - East	0.033
Upper Salt Marsh - West	0.033
High Salt Marsh	0.036
Lothrop Road Stream	0.027

V.3.2.3.4 Comparison of modeled tides and measured tide data

A best-fit of model predictions for the TDR deployment was achieved using the aforementioned values for friction and turbulent exchange. Figures V-10 through V-15 illustrate the six-day calibration simulation along with a 50-hour sub-section. Modeled (dash line) and measured (solid line) tides are illustrated at each model location with a corresponding TDR.

Although visual calibration achieved reasonable modeled tidal hydrodynamics, further tidal constituent calibration was required to quantify the accuracy of the models. Calibration of M_2 (principle lunar semidiurnal constituent) was the highest priority since M_2 accounted for a majority of the forcing tide energy in the modeled system. Due to the duration of the model runs, four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Table V-6 for the calibration period differ from those in Table V-2 because constituents were computed for only the five-day section of the 30-days represented in Table V-2. Table V-6 compares tidal constituent amplitude (height) and relative phase (time) for modeled and measured tides at the TDR locations. The constituent phase shows the relative timing of each separate constituent at a particular location, and also the change (or phase lag) in timing of a single constituent at different locations in an estuary.

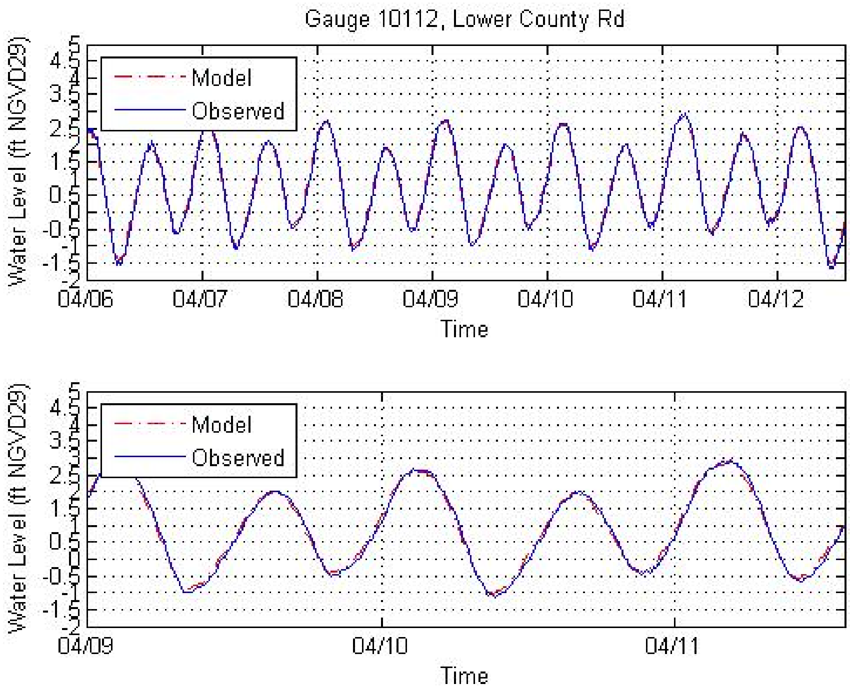

Figure V-10. Comparison of model output and measured tides for the TDR location downstream of Lower County Road. The bottom plot is a 50-hour sub-section of the total modeled time period, shown in the top plot.

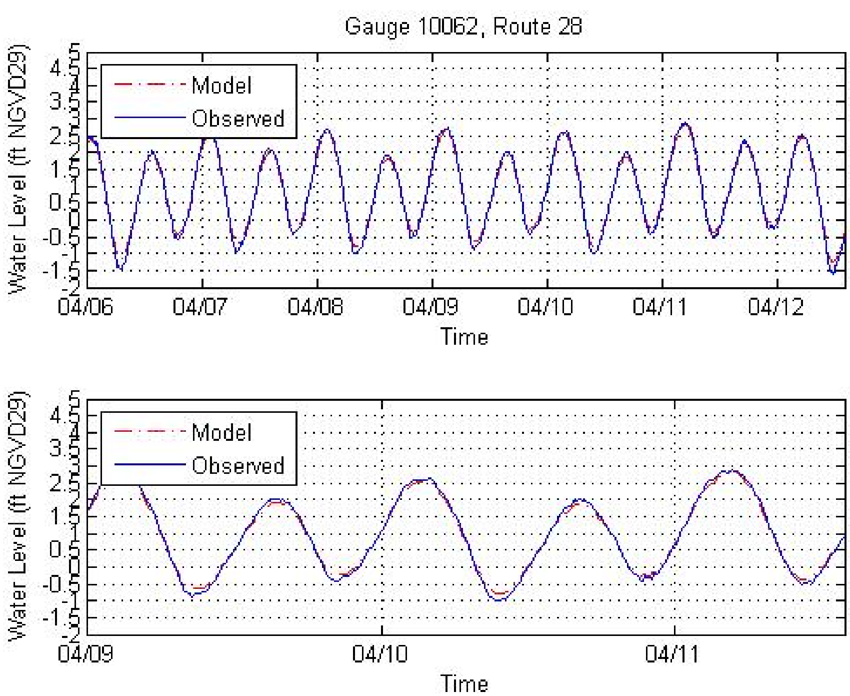

Figure V-11. Comparison of model output and measured tides for the TDR location upstream of Route 28. The bottom plot is a 50-hour sub-section of the total modeled time period, shown in the top plot.

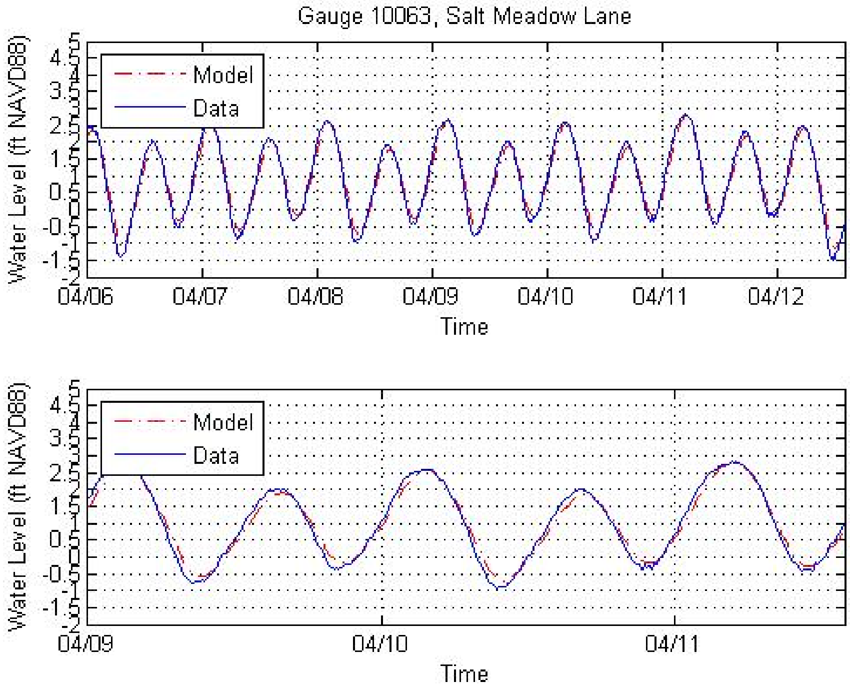

Figure V-12. Comparison of model output and measured tides for the TDR location upstream of Salt Meadow Lane. The bottom plot is a 50-hour sub-section of the total modeled time period, shown in the top plot.

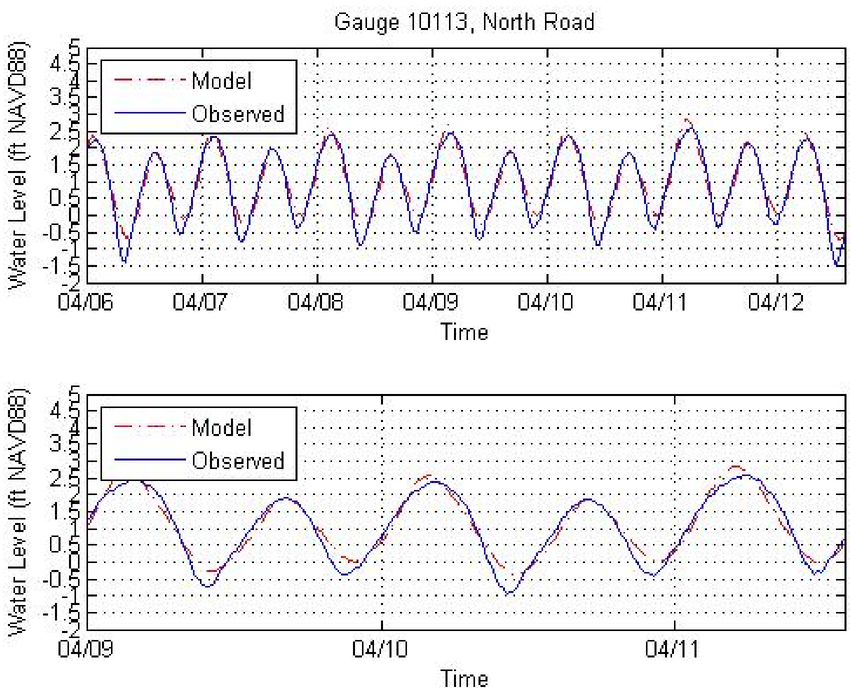

Figure V-13. Comparison of model output and measured tides for the TDR location upstream of North Road. The bottom plot is a 50-hour sub-section of the total modeled time period, shown in the top plot.

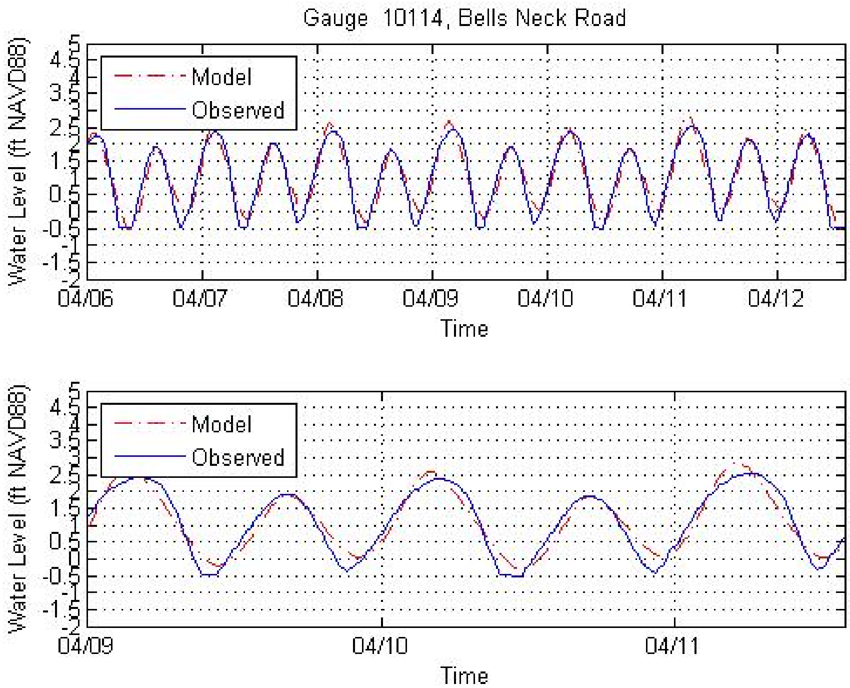

Figure V-14. Comparison of model output and measured tides for the TDR location upstream of the Bells Neck Road. The bottom plot is a 50-hour sub-section of the total modeled time period, shown in the top plot.

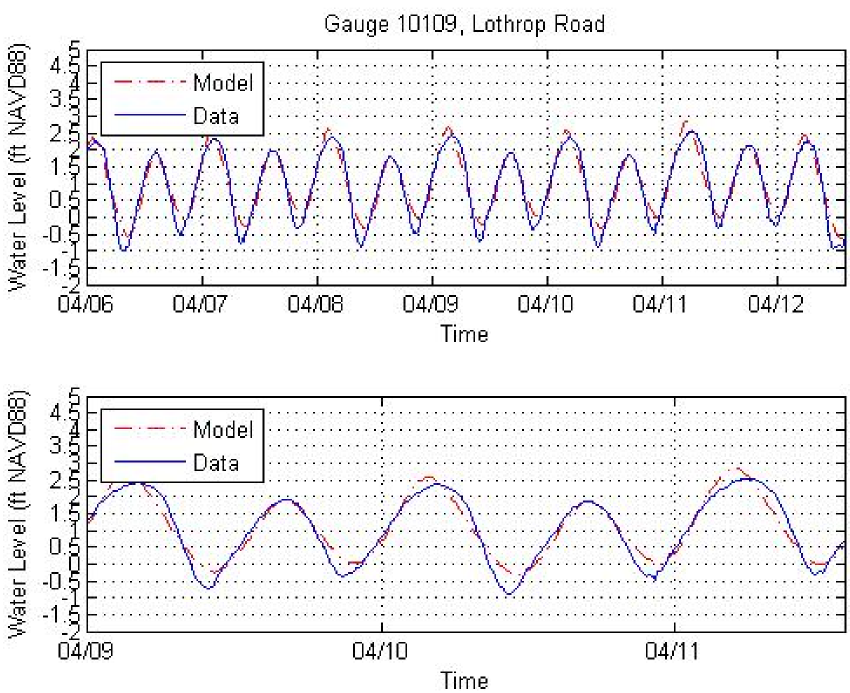

Figure V-15. Comparison of model output and measured tides for the TDR location downstream of Lothrop Road. The bottom plot is a 50-hour sub-section of the total modeled time period, shown in the top plot.

Table V-6. Tidal constituents for measured water level data and calibrated model output, with model error amplitudes, for the Herring River system, during modeled calibration time period.

Location	Constituent Amplitude (ft)				Phase (rad)
Model calibration run					
	M_2	M_4	M_6	K_1	ϕM_2
Lower County Road	1.49	0.08	0.03	0.41	-3.0
Route 28	1.38	0.08	0.02	0.39	-2.8
Salt Meadow Lane	1.31	0.07	0.03	0.38	-2.6
North Road	1.17	0.04	0.03	0.38	-2.3
Bells Neck Road	1.13	0.08	0.03	0.38	-2.2
Lothrop Road	1.16	0.07	0.02	0.38	-2.4
Measured tide during calibration period					
Location	Constituent Amplitude (ft)				Phase (rad)
	M_2	M_4	M_6	K_1	ϕM_2
Lower County Road	1.56	0.10	0.04	0.42	-3.0
Route 28	1.50	0.10	0.04	0.40	-2.9
Salt Meadow Lane	1.46	0.11	0.03	0.39	-2.8
North Road	1.32	0.17	0.05	0.34	-2.5
Bells Neck Road	1.25	0.16	0.04	0.34	-2.4
Lothrop Road	1.30	0.19	0.05	0.33	-2.5
Error					
Location	Error Amplitude (ft)				Phase error (min)
	M_2	M_4	M_6	K_1	ϕM_2
Lower County Road	0.06	0.01	0.01	0.01	1.4
Route 28	0.13	0.02	0.02	0.01	-7.6
Salt Meadow Lane	0.15	0.03	0.00	0.00	-24.6
North Road	0.15	0.13	0.02	-0.04	-20.3
Bells Neck Road	0.11	0.08	0.01	-0.03	-24.1
Lothrop Road	0.14	0.13	0.02	-0.04	-14.1

The constituent calibration resulted in good agreement between modeled and measured tides. The largest errors associated with tidal constituent amplitude were on the order of 0.1 ft, which is better than the order of accuracy of the tide gauges ($\pm$0.12 ft). Time lag errors were approximately within a time step or two of the model (10 minutes), indicating good agreement between the model and data.

V.4.2.5 Model Circulation Characteristics

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

From the model run of the River, maximum ebb velocities in the inlet channels are slightly larger than velocities during maximum flood. Maximum depth-averaged ebb velocities in the model are approximately 2.4 feet/sec at the inlet and 1.6 feet/sec at the Lower County Road Bridge, while maximum flood velocities are about 1.8 feet/sec at the inlet and 1.4 feet/sec at the

Lower County Road Bridge. Close-up views of model output are presented in Figure V-16 and V-17, which show contours of velocity magnitude along with velocity vectors that indicate flow direction, each for a single model time-step, at the portion of the tide where maximum ebb velocities occur (in Figure V-16), and for maximum flood velocities in Figure V-17.

In addition to depth-averaged velocities, the total flow rate of water flowing through a channel can be computed with the hydrodynamic model. The variation of flow as the tide floods and ebbs at the two system inlets is seen in the plot of flow rates in Figure V-18. Maximum flow rates are roughly equal during flood and ebb tides. At Lower County Road, the modeled maximum flow rate is 1,550 ft^3/sec.

Figure V-16. Example of hydrodynamic model output for a single time step where maximum ebb velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

Figure V-17. Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

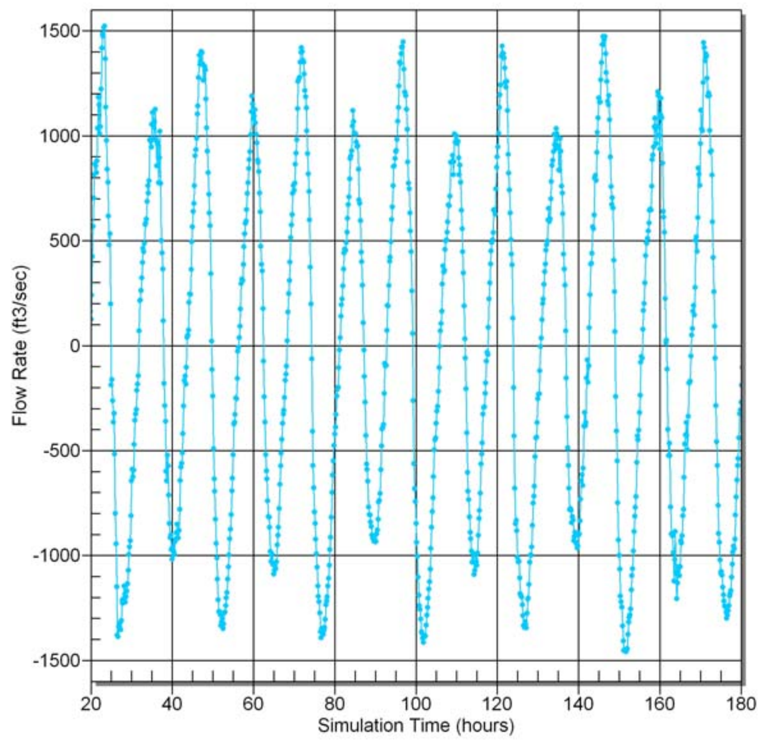

Figure V-18. Time variation of computed flow rates at the inlet to Herring River. Positive flow indicated flooding tide, while negative flow indicates ebbing tide.

V.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 Herring River system is tidal exchange. A rising tide offshore in Nantucket Sound 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 Nantucket Sound 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 evaluate quantitatively tidal flushing of the River system, and was used to compute flushing rates (residence times) and tidal circulation patterns.

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

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

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

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

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

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

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

The rate of pollutant/nutrient loading and the quality of water outside the estuary both must be evaluated in conjunction with residence times to obtain a clear picture of water quality. 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 the calibrated hydrodynamic model in the following section of this report (Section VI) 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 River system.

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in the system, model results were used to compute residence times. Residence times were computed for the entire estuary, as well the six sub-embayments within the system. In addition, **system** and **local residence times** were computed to indicate the range of conditions possible for the system.

Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism). Units then were converted to days. The volume of the entire estuary was computed as cubic feet. Model divisions used to define the system sub-embayments include 1) the entire Herring River system, 2) Herring River above Route 28, 3) East Reservoir, and 4) Herring River west of North Rd. These system divisions follow the model material type areas designated in Figure V-9. Sub-embayment mean volumes and tide prisms are presented in Table V-7.

Residence times were averaged for the tidal cycles comprising a representative 7 lunar day period (14 tide cycles), and are listed in Table V-8. The modeled time period used to compute the flushing rates started April 6, 2011, similar to the model calibration period, and included the transition from neap to spring tide conditions. The RMA-2 model calculated flow crossing specified grid lines for each sub-embayment to compute the tidal prism volume. Since the 7 lunar day period used to compute the flushing rates of the system represent average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the system sub-embayments.

The computed flushing rates for the River system show that as a whole, the system flushes well. A flushing time of 0.77 days for the entire estuary shows that on average, water is resident in the system less than a day. System sub-embayments typically have local flushing times that are less than a day. East Reservoir has the shortest local flushing time, because this embayment has a small mean sub-embayment volume, relative to its tide prism. The highest local flushing rate for the system occurs in upper Herring River Marsh west of North Road. For this sub-embayment, the local flushing rate is 0.72 days due to a small tide range and large volume.

The generally low local residence times in all areas of the Herring River system show that they would likely have good water quality if the system water with which it exchanges also has good water quality. For example, the water quality of East Reservoir would likely be good as long as the water quality of the River was also good. Actual water quality would still also depend upon the total nutrient load to each embayment.

For the smaller sub-embayments of the River system, computed system residence times are typically one or two orders of magnitude longer than their corresponding local residence time. System residence times provide a qualitative measure that helps to identify the relative sensitivity of different sub-embayments to nutrient loading.

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 Herring River 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 Nantucket Sound typically is strong because of the effects of the local winds and tidal induced mixing within Nantucket Sound, the "strong littoral drift" assumption only will cause minor errors in residence time calculations.

Table V-7. Embayment mean volumes and average tidal prism during simulation period.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Herring River	25,703,730	17,432,150
Herring River above RT. 28	18,478,400	14,245,170
East Reservoir	1,319,260	1,708,490
Herring River west of North Rd	4,175,510	3,004,950

Table V-8. Computed System and Local residence times for embayments in the Herring River system.

Embayment	System Residence Time (days)	Local Residence Time (days)
Herring River	0.77	0.77
Herring River above RT. 28	0.93	0.67
East Reservoir	10.13	0.40
Herring River west of North Rd	4.45	0.72

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 Herring River System. These include the output from the hydrodynamics model, calculations of external nitrogen loads from the watersheds, measurements of internal nitrogen loads from the sediment (benthic flux), and measurements of nitrogen in the water column.

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayment

Extensive field measurements and hydrodynamic modeling of the embayment were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated model output representing the transport of water within the system embayment. Files of node locations and node connectivity for the RMA-2 model grid were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic output for the water quality model calibration was a 10-tidal cycle period in April 2011. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to allow the model had reached a dynamic "steady state", and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayment

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

VI.1.3 Measured Nitrogen Concentrations in the Embayment

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in Figure VI-1. The multi-year averages present the "best" comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data is the minimum required to provide a baseline for MEP analysis. Nine years of data (collected between 2001 and 2009) were available for stations monitored by SMAST in the Herring River System.

VI.2 MODEL DESCRIPTION AND APPLICATION

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of nitrogen loading in the Herring River System. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to simulate the fluid dynamics of the Herring River System. Like RMA-2 numerical code, RMA-4 is a two-dimensional depth averaged finite element model capable of simulating time-dependent

Table VI-1. Town of Harwich water quality monitoring data, and modeled Nitrogen concentrations for the Herring River System used in the model calibration plots of Figure VI-2. All concentrations are given in mg/L N. “Data mean” values are calculated as the average of the separate yearly means.

Sub-Embayment	Monitoring station	2001 mean	2002 mean	2003 mean	2004 mean	2005 mean	2006 mean	2007 mean	2008 mean	2009 mean	mean	s.d. all data	N	model min	model max	model average
Wixen Dock	HAR-6	0.760	0.696	0.716	0.567	0.537	0.686	0.475	0.654	0.566	0.853	0.567	0.628	0.323	0.677	0.425
Rt.28 Bridge	HAR-7	0.755	0.756	0.814	0.742	0.768	0.581	0.566	0.625	0.529	0.693	0.712	0.685	0.338	0.767	0.567
North Rd	HAR-9	0.793	0.853	0.919	0.968	0.794	0.873	0.667	0.783	0.636	0.873	0.776	0.810	0.711	0.793	0.776
Lothrop Rd	HAR-8	0.705	0.891	0.910	0.814	0.786	--	--	--	--	--	--	0.827	0.822	0.852	0.840
W. Reservoir	HAR-10	0.732	0.968	0.836	0.654	0.607	0.605	--	--	--	--	--	0.700	0.710	0.712	0.710

constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. Applied Coastal staff have utilized this model in numerous water quality studies of other embayments as part of the MEP Program.

Figure VI-1. Estuarine water quality monitoring station locations in the Herring River System. Station labels correspond to those provided in Table VI-1.

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

VI.2.1 Model Formulation

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

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

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

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

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

Based on measured stream flow rates from SMAST and groundwater recharge rates from the watershed analysis, the hydrodynamic model was set-up to include the latest estimate of surface water flows from Reservoir Stream and Lothrop Road Stream along with ground water flowing into the system from watersheds. The Reservoir Stream has a measure flow rate of 15.8 ft^3/sec (38,852 m^3/day) and Lothrop Road Stream has a measure flow rate of 6.0 ft^3/sec (14,777 m^3/day). Herring River has twenty nine watersheds contributing to the groundwater flow; the combined flow rate into the system is 4.9 ft^3/sec (12,173 m^3/day).

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

VI.2.3 Boundary Condition Specification

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

The loadings used to model present conditions in the Herring River 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 coverage, when present), resulting in a total flux for each embayment (as listed in Table VI-2). Due to the highly variable nature of bottom sediments and other estuarine characteristics of coastal embayments in general, the measured benthic flux for existing conditions also is variable. For present conditions, the benthic flux is negative indicating a net uptake of nitrogen in the bottom sediments in the northwestern area of the upper marsh. Along the main channel the benthic regeneration is positive, and increases from the upper marsh to Nantucket Sound.

In addition to mass loading boundary conditions set within the model domain, a concentration along the model open boundary was specified. The model uses the specified concentration at the open boundary during the flooding tide periods of the model simulations. TN concentration of the incoming water is set at the value designated for the open boundary. The boundary concentration in Nantucket Sound was set at 0.299 mg/L, based on SMAST data from the Nantucket Sound. The open boundary total nitrogen concentration represents long-term average summer concentrations found within Nantucket Sound.

VI.2.4 Model Calibration

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

Table VI-2. Sub-embayment loads used for total nitrogen modeling of the Herring River System, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent **present loading conditions.**			
sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Lower Herring River	9.036	0.252	1.427
East Reservoir	0.293	0.000	0.752
Upper Herring River	13.296	0.395	-1.742
Surface Water Sources			
West Reservoir	27.564	-	-
Lothrop Road Stream	12.627	-	-

Table VI-3. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for Herring River System.	
Embayment Division	E m^2/sec
Nantucket Sound	3.0
Inlet	3.0
Herring River	7.0
Salt Marsh	1.5
Pile supported bridges	7.0
Marsh Channels	6.0
East Reservoir	4.0
Secondary Bridges	4.0
Upper Herring River - West	6.0
Upper Salt Marsh - West	1.0
Upper Herring River - East	6.0
Salt Marsh below Rt. 28	2.0
Upper Salt Marsh - East	1.0
Upper Salt Marsh - West	0.7
High Salt Marsh	0.9
Lothrop Road Stream	5.0
East Reservoir Channel	5.0
Upper Herring River – West Reservoir	1.0
West Reservoir Spillway	2.0
Herring River below Rt. 28	2.0

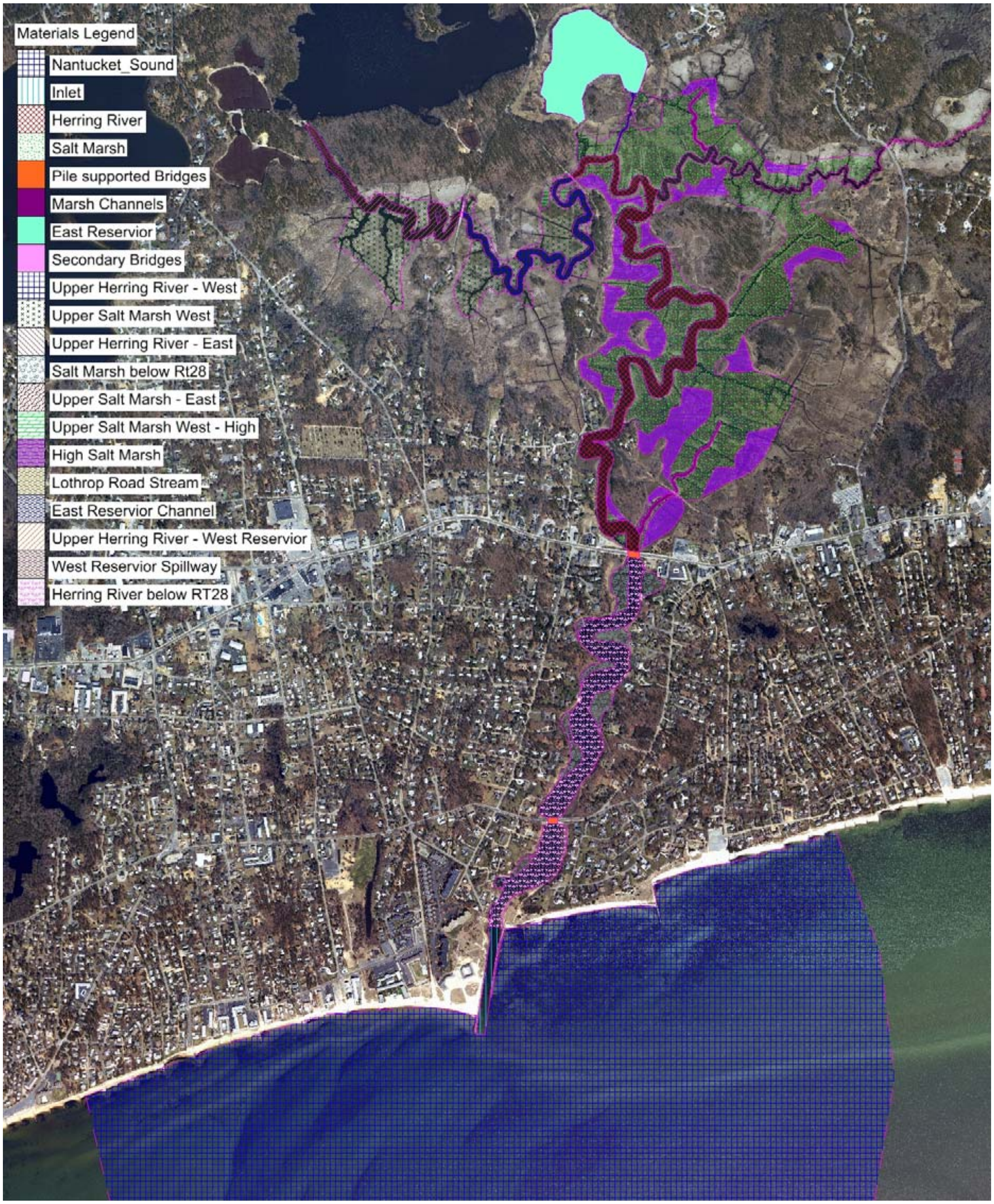

Figure VI-2. Map of Herring River water quality model longitudinal dispersion coefficients. Color patterns designate the different areas used to vary model dispersion coefficient values.

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

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

Also presented in this figure are unity plot comparisons of measured data verses modeled target values for the system. The model fit is exceptional for the Herring River System, with rms error of 0.02 mg/L and an R^2 correlation coefficient of 0.91.

A contour plot of calibrated model output is shown in Figure VI-4 for Herring River System. In the figure, color contours indicate nitrogen concentrations throughout the model domain. The output in the figure show average total nitrogen concentrations, computed using the full 5-tidal-day model simulation output period.

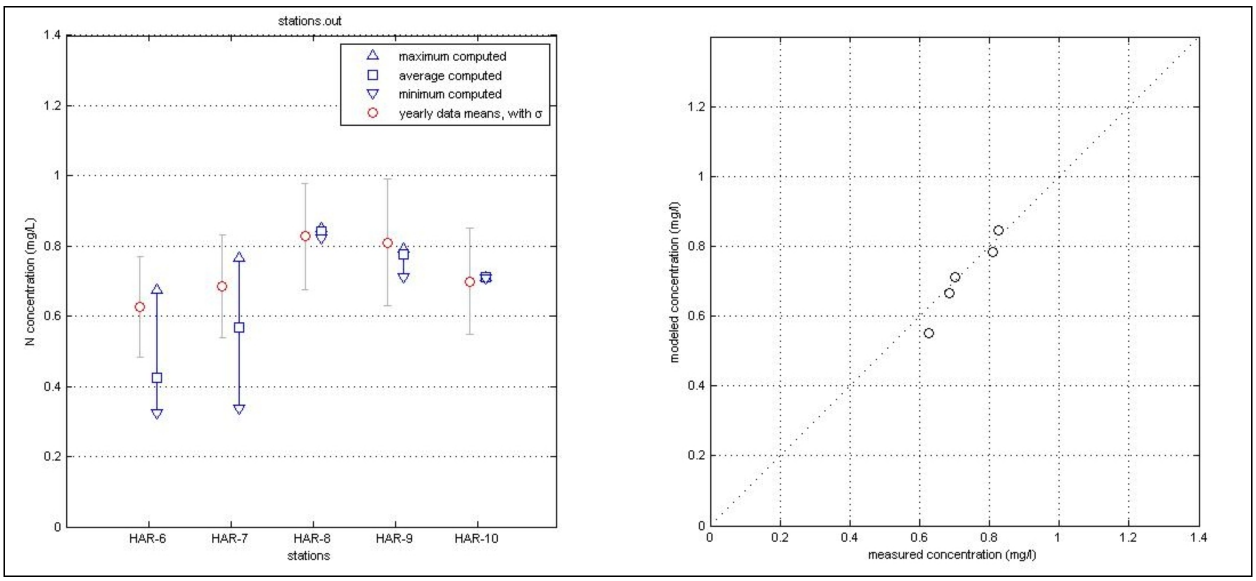

Figure VI-3. Comparison of measured total nitrogen concentrations and calibrated model output at stations in Herring River System. For the left plot, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line.

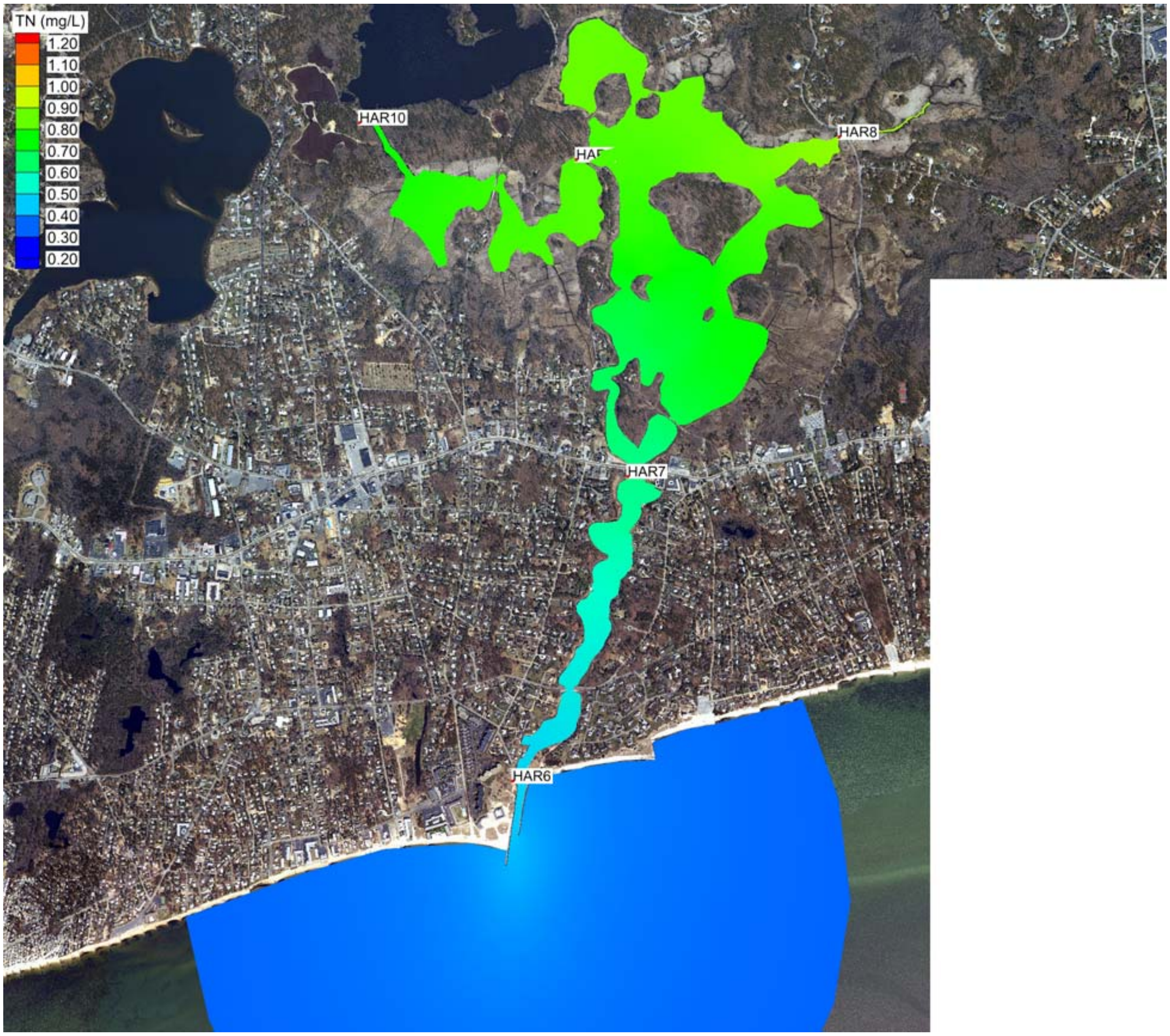

Figure VI-4. Contour plots of average total nitrogen concentrations from results of the present conditions loading scenario, for Herring River System. The approximate location of the sentinel threshold station for Herring River System (HAR-7) is shown.

VI.2.5 Model Salinity Verification

In addition to the model calibration based on nitrogen loading and water column measurements, numerical water quality model performance is typically verified by modeling salinity. This step was performed for the Herring River System using salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of each system, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary, and groundwater inputs. The open boundary salinity was set at 31.6 ppt. For groundwater inputs salinities were set at 0 ppt. The total groundwater input used for the model was 4.9 ft^3/sec (12,173 m^3/day) distributed amongst the watersheds. Groundwater flows were distributed evenly within each watershed through grid cells that formed the perimeter along each watershed's land boundary.

Comparisons of modeled and measured salinities are presented in Figure VI-5, with contour plots of model output shown in Figure VI-6. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model skillfully represents salinity gradients in Herring River System. The rms error of the models was 1.37 ppt, and correlation coefficient was 0.72. The salinity verification provides a further independent confirmation that model dispersion coefficients and represented freshwater inputs to the model correctly simulate the real physical systems.

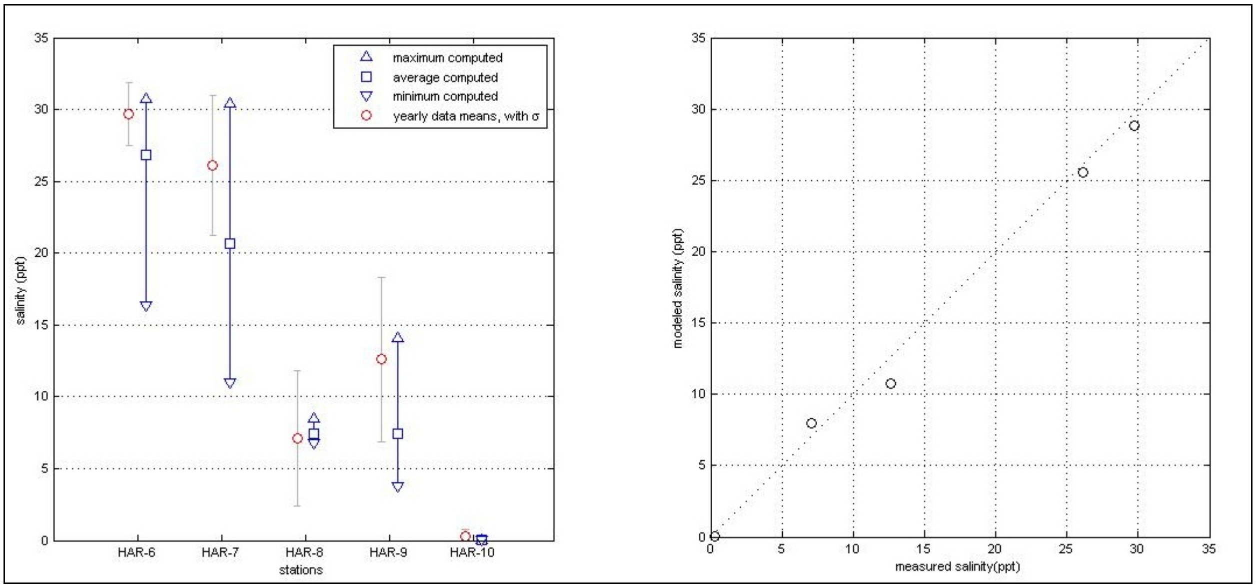

Figure VI-5. Comparison of measured and calibrated model output at stations in Herring River System. For the left plots, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed salinity for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line.

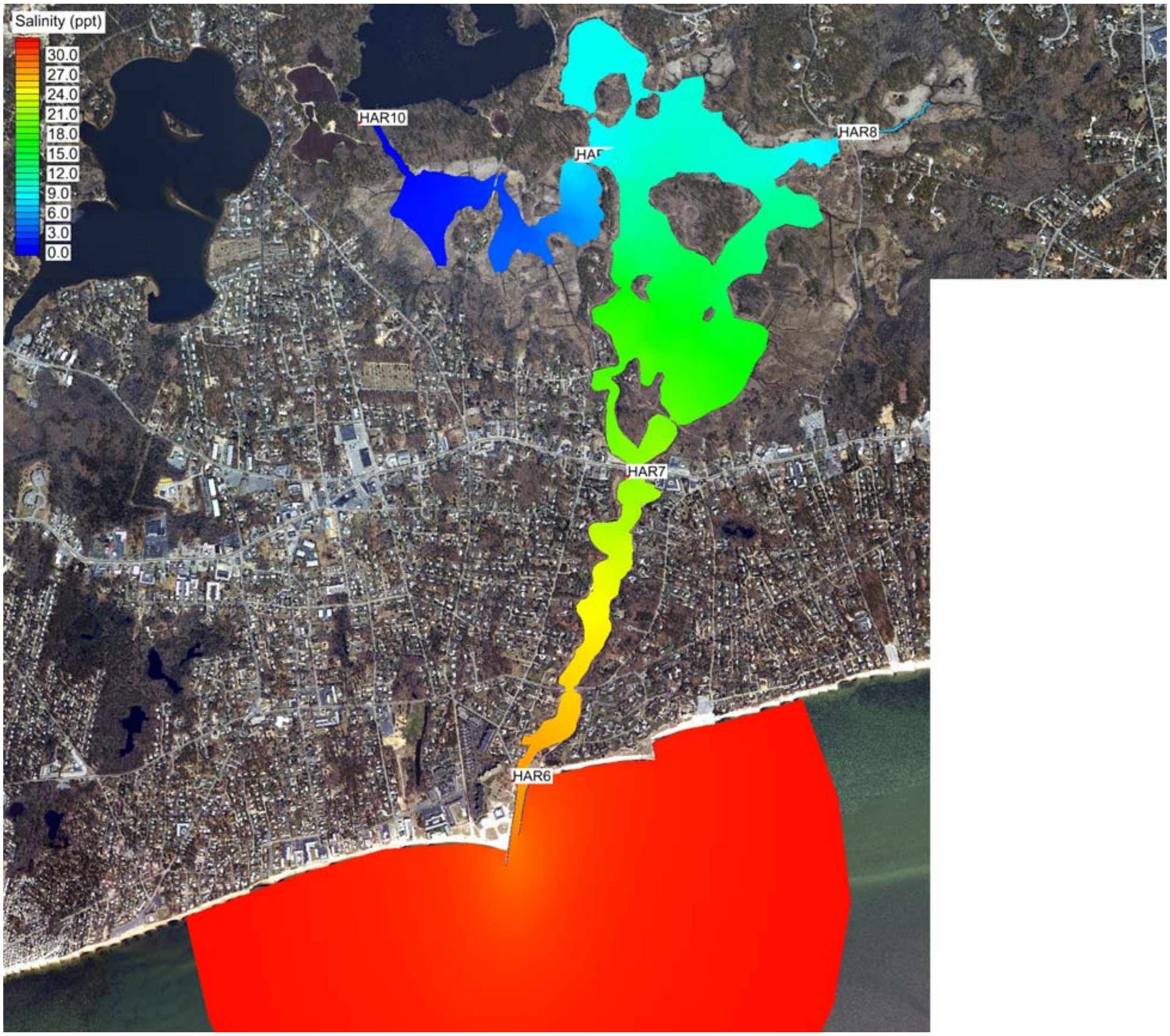

Figure VI-6. Contour plots of modeled salinity (ppt) in Herring River System.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

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

Table VI-4. Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic (“no-load”) loading scenarios of the Herring River System. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	build out (kg/day)	build out % change	no load (kg/day)	no load % change
Lower Herring River	9.036	9.836	+8.9%	0.129	-98.6%
East Reservoir	0.293	0.293	+0.0%	0.236	-19.6%
Upper Herring River	13.296	17.173	+29.2%	0.638	-95.2%
Surface Water Sources					
West Reservoir	27.564	37.011	+34.3%	8.408	-69.5%
Lothrop Road	12.627	15.071	+19.4%	0.844	-93.3%

VI.2.6.1 Build-Out

In general, certain sub-embayments would be impacted more than others. The build-out scenario indicates that there would be a increase in watershed nitrogen load to the Herring River as a result of potential future development. Specific watershed areas would experience large load increases, for example the loads to West Reservoir would increase 34% from the present day loading levels. For the no load scenarios, a majority of the load entering the watershed is removed; therefore, the load is significantly lower than existing conditions.

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

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-5. Build-out sub-embayment and surface water loads used for total nitrogen modeling of the Herring River System, with total watershed N loads, atmospheric N loads, and benthic flux.			
sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Lower Herring River	9.836	0.252	1.650
East Reservoir	0.293	0.000	0.892
Upper Herring River	17.173	0.395	-2.038
Surface Water Sources			
West Reservoir	37.011	-	-
Lothrop Road	15.071	-	-

Following development of the nitrogen loading estimates for the build-out scenario, the water quality model of Herring River System was run to determine nitrogen concentrations within each sub-embayment (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The stations in Herring River show steady increase in nitrogen from the inlet to the head of the system. Color contours of model output for the build-out scenario are present in Figure VI-7. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figure VI-4, which allows direct comparison of nitrogen concentrations between loading scenarios.

Table VI-6. Comparison of model average total N concentrations from present loading and the build-out scenario, with percent change, for the Herring River System. Sentinel threshold station is in bold print.				
Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Wixen Dock	HAR-6	0.425	0.471	+11.0%
Rt. 28 Bridge	**HAR-7**	**0.567**	**0.671**	**+18.4%**
Lothrop Rd	HAR-8	0.840	1.019	+21.3%
North Rd	HAR-9	0.776	0.991	+27.6%
West Reservoir	HAR-10	0.710	0.953	+34.2%

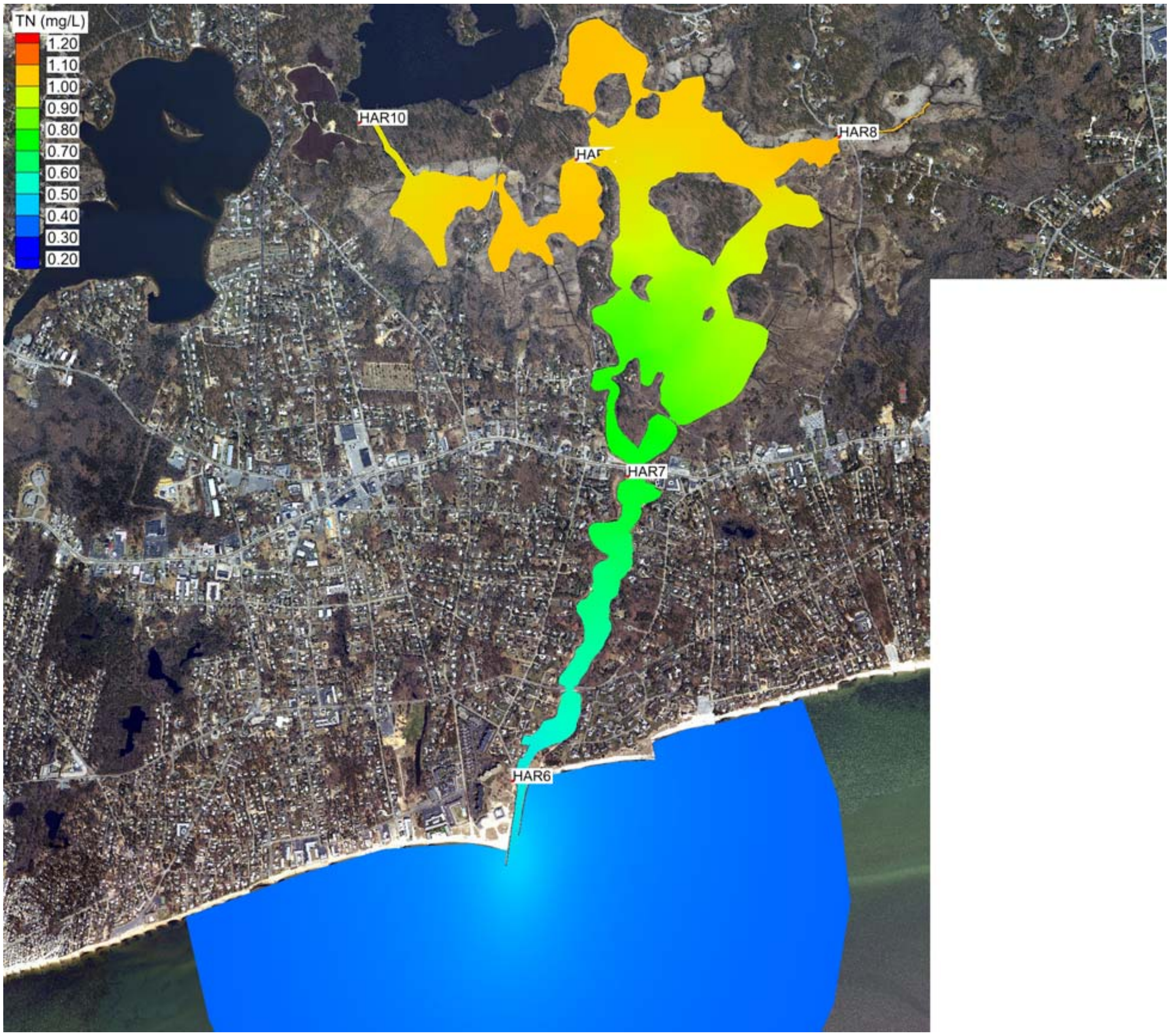

Figure VI-7. Contour plots of modeled total nitrogen concentrations (mg/L) in Herring River System, for projected build-out loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Herring River System (HAR-7) is shown.

VI.2.6.2 No Anthropogenic Load

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

Table VI-7. "No anthropogenic loading" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of Herring River System, with total watershed N loads, atmospheric N loads, and benthic flux			
sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Lower Herring River	0.129	0.252	0.758
East Reservoir	0.236	0.000	0.318
Upper Herring River	0.638	0.395	-0.804
Surface Water Sources			
West Reservoir	8.408	-	-
Lothrop Road	0.844	-	-

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Again, total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was significant as shown in Table VI-8, with reductions ranging from 33% inside the inlet to Herring River with greater than 86% reduction in total nitrogen. Results for each system are shown pictorially in Figure VI-8.

Table VI-8. Comparison of model average total N concentrations from present loading and the no anthropogenic ("no load") scenario, with percent change, for the Herring River System. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). Sentinel threshold station is in bold print.				
Sub-Embayment	monitoring station	present (mg/L)	no-load (mg/L)	% change
Wixen Dock	HAR-6	0.425	0.283	-33.4%
Rt. 28 Bridge	**HAR-7**	**0.567**	**0.261**	**-54.0%**
Lothrop Rd	HAR-8	0.840	0.114	-86.5%
North Rd	HAR-9	0.776	0.222	-71.5%
West Reservoir	HAR-10	0.710	0.216	-69.5%

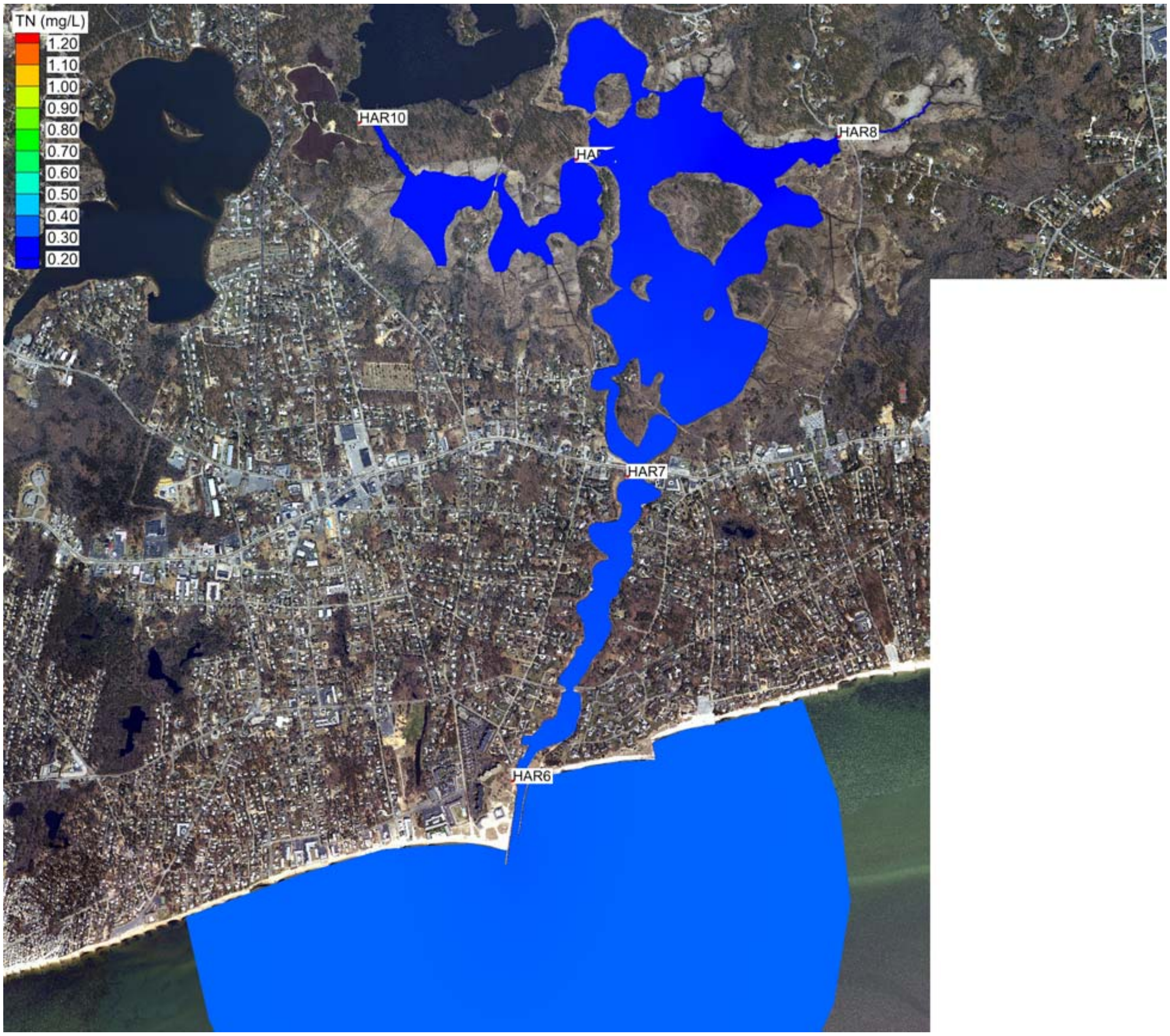

Figure VI-8. Contour plots of modeled total nitrogen concentrations (mg/L) in Herring River System, for no anthropogenic loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Herring River System (HAR-7) is shown.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Herring River Marsh estuarine system in the Town of Harwich, MA, our assessment is based upon data from the water quality monitoring database developed by the Town of Harwich Water Quality Monitoring Program and surveys of eelgrass distribution (1995, 2001, 2004, 2010), benthic animal communities and sediment characteristics, and dissolved oxygen records conducted during the summer and fall of 2004. These data form the basis of an assessment of this system's present health, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for these systems (Chapter VIII).

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

As a basis for a nitrogen thresholds determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed dissolved oxygen sensors within the central region of the tidal river, comprising the lower estuary and in the main wetland tidal creek in the wetland dominated upper estuary to record the frequency and duration of low oxygen conditions during the critical summer period. The MEP habitat analysis uses eelgrass as a sentinel species for indicating nitrogen over-loading to coastal embayments. Eelgrass is a fundamentally important species in the ecology of shallow coastal systems, providing both habitat structure and sediment stabilization. Mapping of the eelgrass beds within the Herring River system was conducted for comparison to historic records (MassDEP Eelgrass Mapping Program, C. Costello). Temporal trends in the distribution of eelgrass beds are used by the MEP to assess the stability of the habitat and to determine trends potentially related to water quality. Eelgrass beds can decrease within embayments in response to a variety of causes, but throughout almost all of the embayments within southeastern Massachusetts, the primary cause appears to be related to increases in embayment nitrogen levels.

In areas that do not support eelgrass beds, benthic animal indicators were used to assess the level of habitat health from "healthy" (low organic matter loading, high D.O.) to "highly stressed" (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stress indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill, benthic population studies in Buzzards Bay (Woods Hole Oceanographic Institution) and New Bedford (SMAST), and more recently the Woods Hole Oceanographic Institution Nantucket Harbor Study (Howes *et al.* 1997). These data are coupled with the level of diversity (H') and evenness (E) of the benthic community and the total number of individuals to determine the infaunal habitat quality. In the analysis of the benthic animal communities it is important to take into account the ecosystem type that contains the habitat. High quality habitat in open water basins support different communities than high quality habitat in tidal wetlands. The differences stem primarily from natural differences in organic matter and nutrient levels. These differences were important to the MEP assessment of the wetland dominated upper region and open water dominated tidal river comprising the lower region of the Herring River Estuary.

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 4.0 mg L^{-1}. Massachusetts State Water Quality Classification indicates that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1}. The tidal waters of the Herring River Estuary are currently listed under this Classification as SA. It should be noted that the Classification system represents the water quality that the embayment should support, not the existing level of water quality. It is through the MEP and TMDL processes that management actions are developed and implemented to keep or bring the existing conditions in line with the Classification.

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (see Figure VII-1 for example). 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, two autonomously recording oxygen sensors were moored 30 cm above the embayment bottom within key regions of the Herring River (Figure VII-2) system. One of the instruments was placed at approximately the mid-point of the tidal river comprising the lower portion of the estuary. The second DO mooring was deployed in the upper wetland dominated portion of the estuary, within the main wetland creek down gradient of the confluence of the secondary wetland creeks entering under Lothrop Road to the east and from West Reservoir to the west.

Before deployment in the Herring River Estuary, the instruments and sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployment. During the deployment period, periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer). The instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a minimum deployment of 30 days within the interval from July through mid-September. All of the mooring data from the Herring River Marsh system was collected during the summer of 2004.

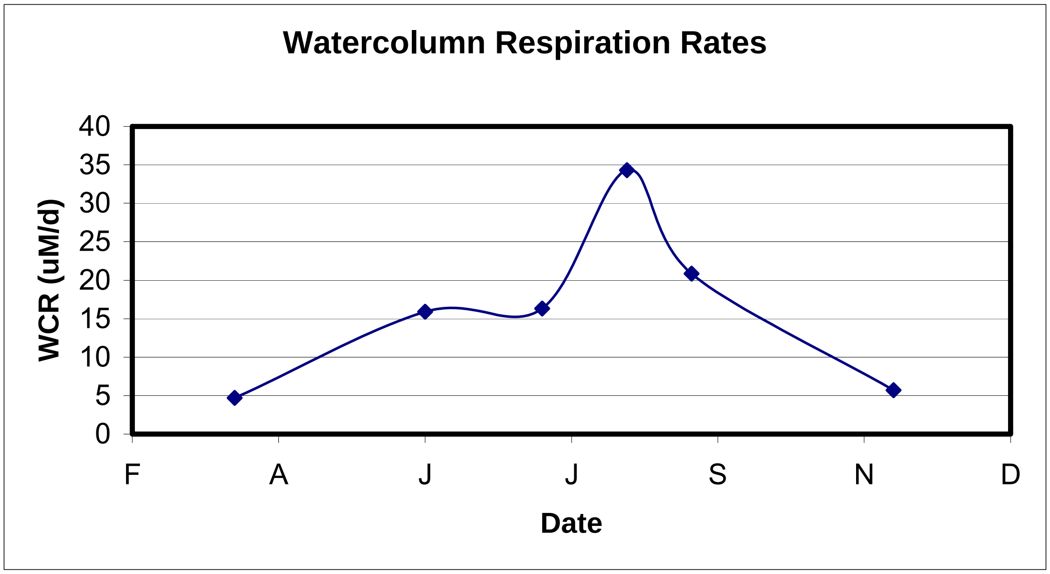

Figure VII-1. Example of average water column respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

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

Dissolved oxygen and chlorophyll a records were examined both for temporal trends and to determine the percent of the 35 day deployment period that these parameters were below/above various benchmark concentrations (Tables VII-1, VII-2). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions. The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate that the upper wetland portion of the Herring River Estuary is organic and nutrient enriched, a natural condition of temperate wetland systems (Figures VII-3, VII-4).The oxygen data is consistent with high organic matter inputs from the surrounding vegetated marsh plain and to a lesser extent from phytoplankton production (chlorophyll a levels, generally ~15 ug/L at the wetland mooring site and ~11 ug/L in the upper wetland creeks as measured by the Harwich Water Quality Monitoring Program). The levels of oxygen depletion in the upper portions of the

Figure VII-2. Aerial Photograph of the Herring River Marsh system in Harwich showing the location of the Dissolved Oxygen mooring deployments conducted in the Summer of 2004.

system are indicative of an organic matter and nutrient rich system and are typical of wetland dominated regions of estuaries throughout southeastern Massachusetts. Although oxygen levels showed less depletion at the lower tidal river mooring site than the upper wetland site (Figures VII-3, VII-5), the daily oxygen excursion was similar. It appears that much of the oxygen depletion at the lower site, resulted from the out-flowing low oxygen water from the up-gradient wetland basin. The absence of significant wetlands bordering the tidal river and the generally low phytoplankton biomass in the watercolumn (chlorophyll a levels of 2 - 8 ug/L at the mooring site and averages of 5 - 7 ug/L, as measured by the Harwich Water Quality Monitoring

Program, HWQMP), supports the contention (Figure VII-6). However, the higher oxygen levels and lower phytoplankton biomass in the tidal river are consistent with an open water basin that until recently supported eelgrass and likely has high quality benthic habitat.

An additional factor used in assessing oxygen levels is the diurnal range of oxygen concentration. The use of only the duration of oxygen below a certain threshold (e.g. 4 mg L^{-1}) can result in underestimates of the level of habitat impairment in a particular location, since nitrogen enrichment also results in increased phytoplankton (or epibenthic algae) production, as evidenced by oxygen levels that rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). The absence of elevated oxygen levels within upper portion of the Herring River Marsh system is consistent with the oxygen variations being the response to the naturally organic rich characteristics of tidal wetland creeks, rather than processes dominated by nutrient driven phytoplankton effects. In these systems the oxygen dynamic is driven by consumption within the tidal creek waters and sediments, with re-oxygenation through phytoplankton production being limited. The creeks drain nearly completely at low tide and the rise in oxygen levels is primarily through the entry of oxygen rich coastal waters on the flooding tide. A nearly identical pattern of oxygen variation was recorded from the nearby and similarly structured tidal creeks of Namskaket and Little Namskaket Marshes. This is further evidence that the oxygen dynamics within these tidal creeks are driven by marsh processes, rather than by watershed nitrogen loading. By contrast, the dissolved oxygen levels recorded at the lower mooring location that has more estuarine qualities suggest that out-flowing low oxygen water from the wetland dominated upper estuary plus uptake within the watercolumn and sediments of the tidal river control the level of oxygen depletion on ebbing tides (to levels of 4 mg/L), while on flooding tides the entry of oxygenated waters from Nantucket Sound generally produce oxygen levels >6 mg/L. This suggests that oxygen conditions within the tidal river portion of the Herring River Estuary are finely balanced and not solely the result of intra-basin processes.

The dissolved oxygen records indicate that the wetland dominated upper estuary has periodic oxygen depletion to ~3 mg L^{-1} (Table VII-1). Such oxygen depletion is typical of organic and nutrient rich wetland dominated systems, although in open water basins it is indicative of impaired habitat quality. However, wetlands are nutrient and organic matter enriched as part of their ecological design, which makes them such important nursery areas for fauna in adjacent offshore waters. However, a natural consequence of the organic rich sediments in these brackish and salt marsh dominated systems is periodic oxygen depletion within the tidal creeks, particularly during the summer. The observed level of oxygen depletion in the upper portion of the Herring River Estuary is expected, as was the nearly identical pattern recorded in the tidal creeks of nearby Namskaket and Little Namskaket Marshes within the Town of Orleans and salt marsh dominated ponds such as in the Parkers River and Centerville River estuaries. Assessment of habitat quality must necessarily consider the natural function and tolerances of the specific estuarine ecosystems being evaluated and therefore the need for multiple mooring deployments in this system. The specific results are as follows:

***Herring River North* (Figures VII-3 and VII-4):**

The upper portion of the Herring River Estuary (above Rt. 28) functions as a large tidal wetland system, dominated by fresh and brackish emergent marsh plants (*Typha, Phragmites*) in the uppermost western branch and salt marsh dominated by high (*Spartina patens*) and low (*Spartina alterniflora*) marsh plants in the eastern and central areas (Figure I-2). The wetland basin has deeply incised narrow creeks generally with organic rich muds intermixed with wetland plant detritus. As with other healthy wetlands, the tidal creeks do not support macroalgae and the sediments generally are oxidized at the surface. At low tide the smaller creeks empty, while some water remains in the main channels. At high tide the emergent salt marsh is frequently inundated, although the fresh/brackish marsh receives little flooding by salt water, due to its elevation and the large freshwater discharge from West Reservoir.

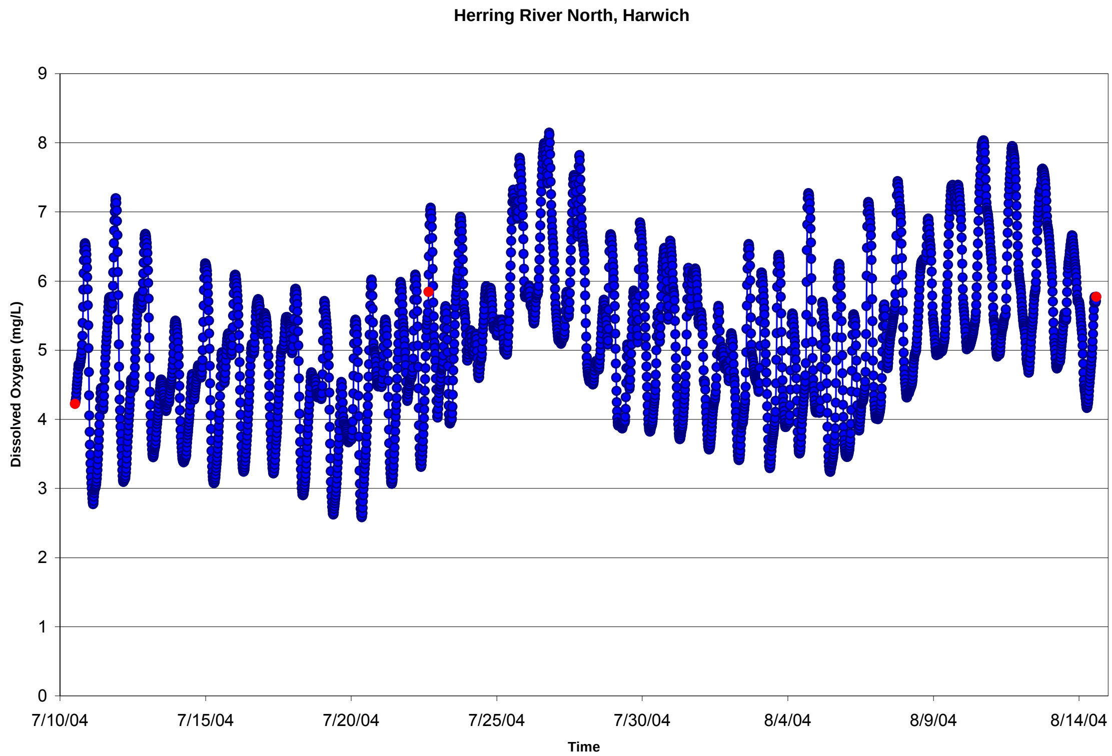

Figure VII-3. Bottom water record of dissolved oxygen in the main channel of the upper wetland dominated portion of the Herring River Estuary (north mooring), Summer 2004. Calibration samples represented as red dots.

The large diurnal shifts in dissolved oxygen within the main tidal creek, with periodic depletion to less than 3 mg L^{-1}, within the upper wetland portion of the Herring River Estuary are consistent with the high productivity within the marsh, high levels of oxygen uptake within the watercolumn and sediment due to the organic matter rich marsh sediments, and tidal changes in salinity and temperature which influence oxygen solubility. That the organic rich nature of wetlands, rather than impacts of nitrogen enrichment are dominated the oxygen dynamics within the wetland creeks is supported by the absence of significant high oxygen levels in excess of air equilibration. Oxygen levels over 7-8 mg/L are frequently observed in nitrogen enriched embayments due to high phytoplankton biomass and day time oxygen production. The generally moderate chlorophyll a levels in the saline reaches and the lack of "excess" oxygen in daytime supports the concept that the marsh processes are the primary control on oxygen dynamics within this basin. Further evidence for the dominance of marsh processes is the lack of linkage between the observed variations in chlorophyll (Table VII-2) 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. However, chlorophyll levels at the mooring site were generally moderate to low (~10 ug/L at the mooring and averaging ~11 ug/L slightly upgradient at water quality monitoring station HER-9, Harwich Water Quality Monitoring Program) and even in open water embayments would not typically result in the levels of oxygen depletion observed within the upper Herring River Estuary. For the above reasons and the similarity in the temporal pattern of oxygen depletion, both in timing and magnitude, between the upper Herring River Estuary and nearby Little Namskaket and Namskaket Marshes it can be concluded that the oxygen dynamics are being driven by processes associated with the naturally organic rich nature of wetlands and their associated tidal creeks and therefore the levels of oxygen depletion do not represent impairment, but rather the are part of the functioning of wetland systems.

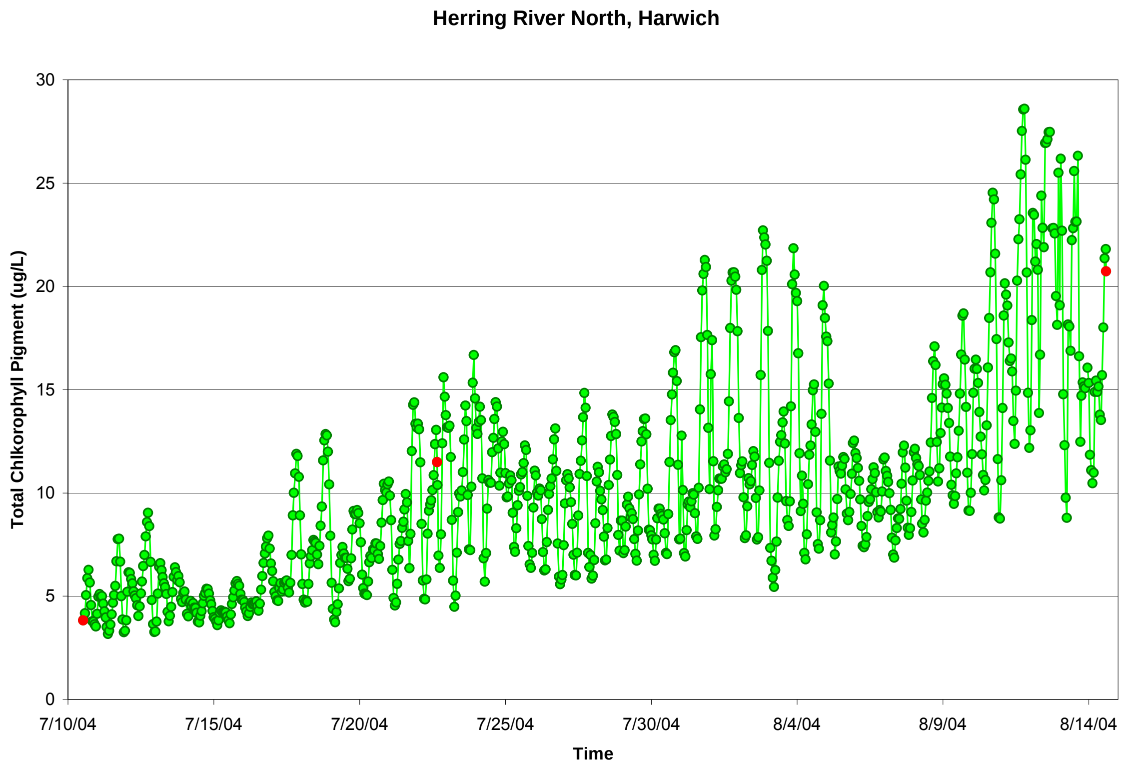

Figure VII-4. Bottom water record of Chlorophyll-*a* in the main channel of the upper wetland dominated portion of the Herring River Estuary (north mooring), Summer 2004. Calibration samples represented as red dots.

***Herring River South* (Figures VII-5 and VII-6):**

The lower reach of the Herring River Estuary functions as an open water tidal river, with little bordering wetland or tidal flats. The tidal river transports water exchanging between the upper wetland basin (above Rt. 28) and Nantucket Sound. The result is that during most of the flood tide, high quality Nantucket Sound water flows through the river, while during ebb tide the river is carrying water ebbing from the wetland basin with its nutrient enriched waters periodically depleted in oxygen. Fortunately, during transport aeration and mixing occurs such that during ebb tide, oxygen levels in the tidal river are significantly (1-2 mg/L) higher than in the wetland basin. However, while the ebbing waters from the upper basin are important to the oxygen dynamics in the tidal river, internal processes also play a role. The pattern of oxygen and chlorophyll a in the lower versus upper basins are not entirely consistent. Most importantly the lower basin has a more stable oxygen cycle and doesn't show the same temporal pattern in chlorophyll levels. However, analysis of the time-series indicates that the rebound of oxygen levels is in part due to the inflow of Nantucket Sound water, while the oxygen minima appear to be related to oxygen dynamics within the upper wetland basin. The magnitude of the maxima and minima within both basins is different due to both internal oxygen processes within each basin and aeration and mixing during transport.

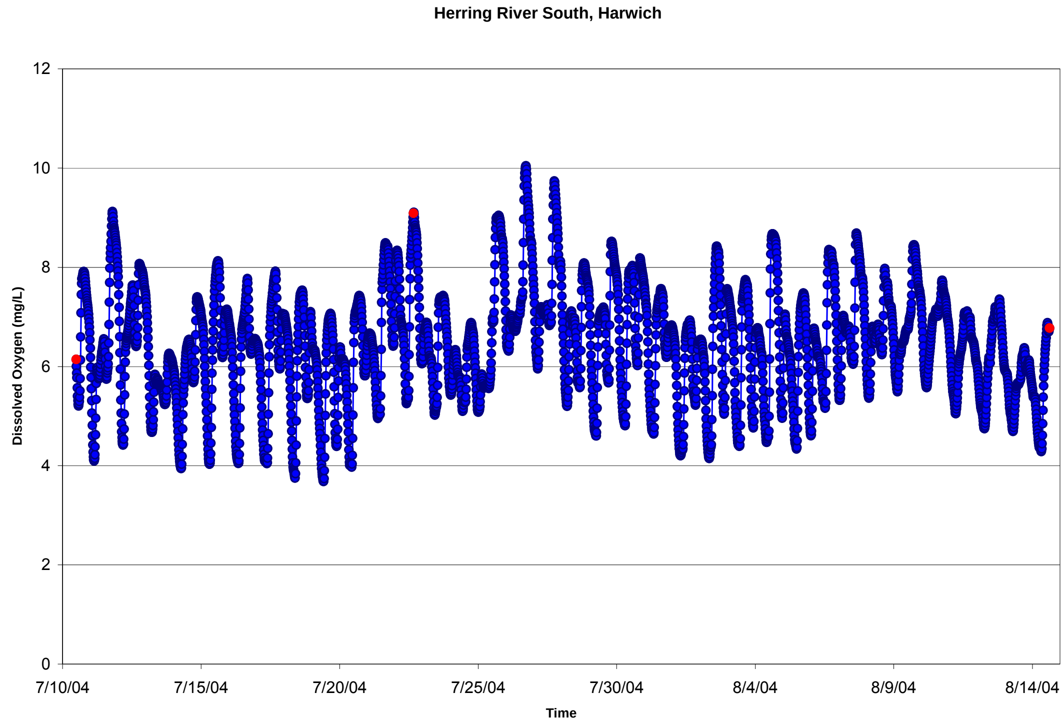

Figure VII-5. Bottom water record of dissolved oxygen in the tidal river portion of the Herring River Estuary (south mooring), Summer 2004. Calibration samples represented as red dots.

Oxygen levels within the tidal river were generally >5 mg/L, ~80% of record, with only a rare decline to 3 mg/L. These levels were similar to those measured by the Harwich Water Quality Monitoring Program (HWQMP), although the station at the Rt. 28 bridge (HER-7) on the ebbing tide periodically showed oxygen levels <4 mg/L, and general D.O. conditions lower than the station in the lower tidal river (HER-6), consistent with expected oxygen dynamics in a tidal river under these conditions. Similarly, chlorophyll a levels were low, averaging 4.7 ug/L, again comparable to the measurements by the HWQMP which averaged 6.7 ug/L and 4.9 ug/L for the upper and lower stations in the tidal river (2002-08). Equally important was the absence of bloom conditions with chlorophyll levels generally <10 ug/L and only rarely >15 ug/L as measured by both approaches (MEP & HWQMP). Overall, the level of oxygen depletion in the tidal river appears to be greater than can be explained by internal processes (nitrogen, phytoplankton) alone (e.g. chlorophyll a levels are generally low, while oxygen reaches 4 mg/L in a highly flushed environment). This discrepancy is clearly the result of the outflowing water from the large upper tidal wetland basin, which due to natural processes periodically releases low oxygen water on the summer ebb tides.

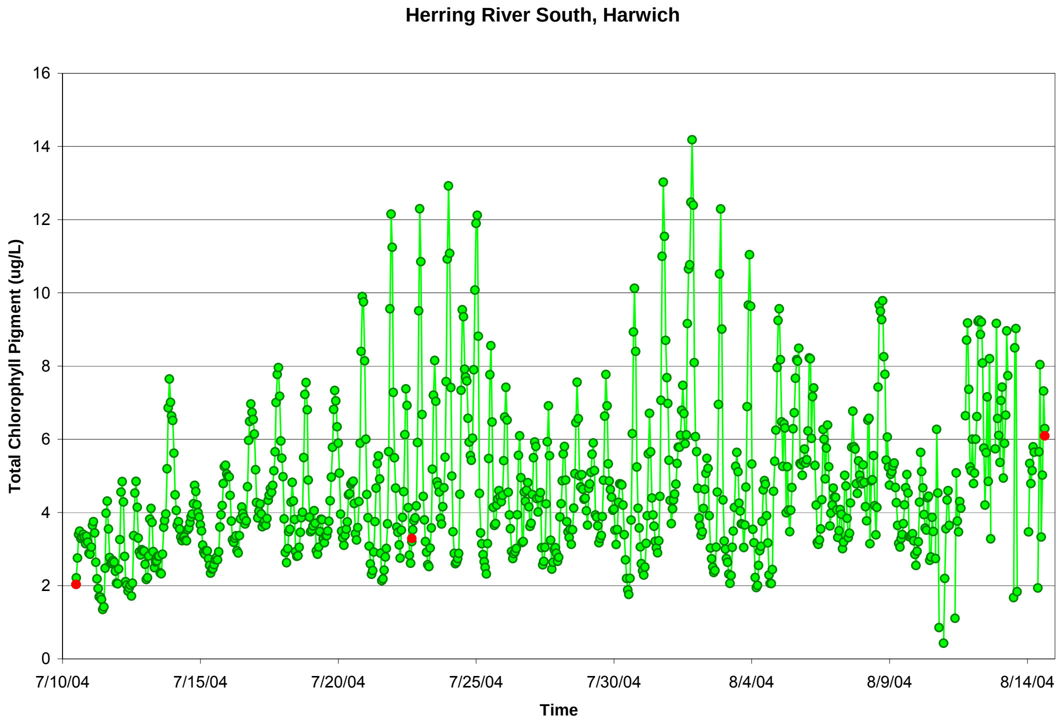

Figure VII-6. Bottom water record of Chlorophyll-a in the tidal river portion of the Herring River Estuary (south mooring), Summer 2004. Calibration samples represented as red dots.

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

Mooring Location	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)
Herring River North	7/10/2004	8/14/2004	**35.1**	**80%**	**47%**	**17%**	**1%**
			Mean	0.87	0.34	0.24	0.12
			Min	0.09	0.04	0.08	0.07
			Max	4.64	0.89	0.34	0.17
			S.D.	0.81	0.21	0.07	0.04
Herring River South	7/10/2004	8/14/2004	**35.1**	**34%**	**11%**	**1%**	**0%**
			Mean	0.24	0.14	0.07	N/A
			Min	0.02	0.02	0.02	0.00
			Max	0.68	0.23	0.10	0.00
			S.D.	0.14	0.06	0.04	N/A

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

Mooring Location	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)
Herring River North	7/10/2004	8/14/2004	**35.1**	**87%**	**45%**	**15%**	**7%**	**2%**
Mean Chl Value = 10.3 ug/L			Mean	1.70	0.39	0.22	0.18	0.10
			Min	0.04	0.04	0.04	0.04	0.04
			Max	22.29	2.17	0.75	0.46	0.21
			S.D.	5.16	0.38	0.16	0.12	0.09
Herring River South	7/10/2004	8/14/2004	**35.1**	**32%**	**3%**	**0%**	**0%**	**0%**
Mean Chl Value = 4.7ug/L			Mean	0.19	0.10	N/A	N/A	N/A
			Min	0.04	0.04	0.00	0.00	0.00
			Max	0.83	0.21	0.00	0.00	0.00
			S.D.	0.16	0.05	N/A	N/A	N/A

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data is key part of the MEP Approach. Surveys were conducted in the vicinity of the Herring River Estuary by the MassDEP Eelgrass Mapping Program as part of the MEP effort in 1995, 2001 and 2010. The primary use of the data is to indicate (a) estuarine regions that have historically or presently support eelgrass habitat, 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 1995 to 2010 (Figure VII-7); the period in which watershed nitrogen loading increased to its present level. This temporal information can be used to determine the stability of the eelgrass community.

Eelgrass surveys were not undertaken for the upper reaches of the main tidal creek (upgradient of Route 28) of the Herring River Marsh by the MassDEP Eelgrass Mapping Program (C.Costello), as this "basin" is the central tidal creek of an extensive wetland system, similar to the salt marshes of the Town of Orleans (Namskaket and Little Namskaket Creeks). Tidal creeks do not generally support eelgrass habitat, particularly when the creek drains significantly during each ebb tide. In addition, MassDEP was unable to perform its 1951 analysis due to the inadequacy of the available aerial photos. However, there is no evidence of eelgrass previously colonizing the wetland creeks of the upper wetland basin, which is dominated by salt marsh transitioning to freshwater marsh in the upper western region receiveing the high volume freshwater discharge from West Reservoir. The MEP Technical Team did confirm the lack of eelgrass in the tidal creeks throughout the upper wetland basin, above the Rt. 28 bridge, of the Herring River Estuary while undertaking field surveys as part of the benthic regeneration and infauna studies and during the deployment and recovery and monitoring of the instrument mooring in the main tidal creek of the upper basin.

In contrast, the lower tidal reach of the Herring River Estuary, which functions as a tidal river carrying tidal exchange between the large upper wetland basin and Nantucket Sound has historically supported eelgrass as indicated in the MassDEP 1995 and 2001 surveys and confirmed in MEP surveys in 2004. Based on the 1995 and 2001 eelgrass surveys conducted by the MassDEP Eelgrass Mapping Program eelgrass coverage within the tidal river up to Rt. 28 had declined significantly and by the 2010 survey the remaining coverage was negligible (Table VII-3).

Based upon all available information, it appears that the tidal creeks of the upper wetland basin of the Herring River Estuary are not structured to support eelgrass habitat, as was also found for adjacent nearby salt marsh systems in the Town of Orleans, Barnstable and Chatham (Cockle Cove). This is not the case in the lower estuarine reach (e.g. tidal river, below Route 28) which supported eelgrass as recently as 2004 and showed a continuing decline in coverage since 1995. At present, there has been a near complete loss of eelgrass coverage within the tidal river of the Herring River Estuary. Therefore, threshold development for protection/restoration of this system will focus on restoration of eelgrass habitat within the tidal river.

The eelgrass data for the Herring River Marsh system are consistent with the results of the benthic infauna analysis and the water quality data for this system (see below).

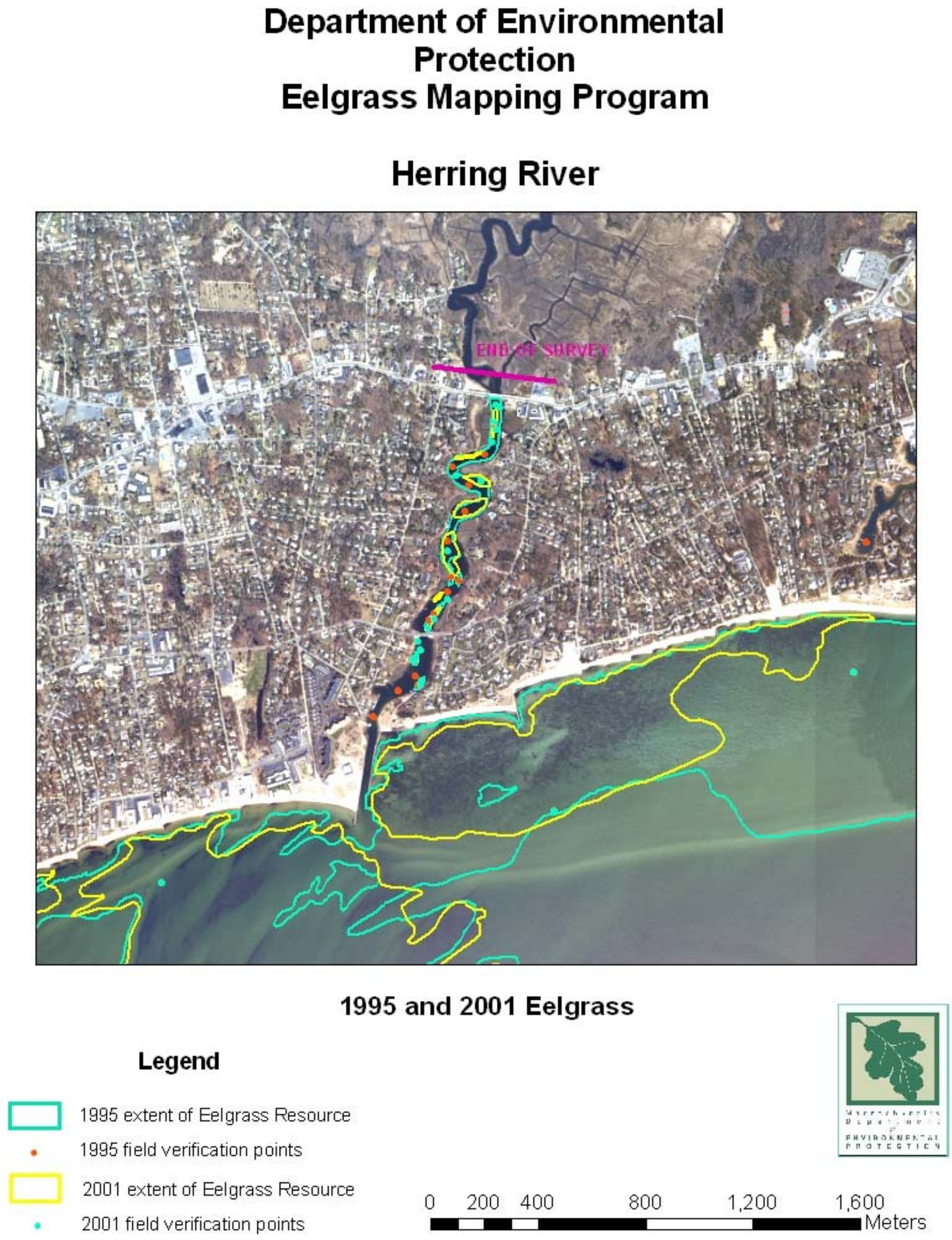

Figure VII-7. Eelgrass bed distribution in the main tidal channel of the Herring River system, Harwich, MA. and immediately offshore the from the inlet. The 1995 coverage is depicted by the green outline and the 2001 coverage the yellow outline, which circumscribes the eelgrass beds as mapped by DEP Eelgrass Mapping Program. There is no evidence that the upper reaches of the system upgradient of Route 28 ever supported eelgrass habitat, as it is primarily tidal salt marshes.

Table VII-3. Temporal change in eelgrass coverage within the lower reach (tidal river) portion of the Herring River Estuary as determined by the MassDEP Eelgrass Mapping Program (C. Costello). Eelgrass was also noted in the 2004 surveys by the MEP.

Estuary	1951 (acres)	1995 (acres)	2001 (acres)	2010 (acres)	% Difference (1995 to 2010)
Herring River	**NA**	**9.3**	**4.5**	**0**	**100%**

VII.4 BENTHIC INFAUNA ANALYSIS

In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that, the absence of eelgrass habitat within the upper wetland basin of the Herring River Estuary results from its function as a wetland, rather than nutrient enrichment from watershed nitrogen loading. Therefore, to the extent that the upper wetland dominated basin supports healthy infaunal communities relative to its specific nutrient and organic matter levels and resulting environmental conditions, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

Analysis of the evenness and diversity of the benthic animal communities was also used to support the density data and the natural history information. The evenness statistic can range from 0-1 (one being most even), while the diversity index does not have a theoretical upper limit. The highest quality habitat areas, as shown by the oxygen and chlorophyll records and eelgrass coverage, have the highest diversity (generally >3) and evenness (~0.7). The converse is also true, with poorest habitat quality found where diversity is <1 and evenness is <0.5 (Table VII-4).

Quantitative sediment sampling was conducted at 13 locations throughout the tidal reach of the Herring River Estuary (Figure VII-8). In some cases multiple assays were conducted. Four of the sampling locations were situated in the lower estuary in the tidal river with tidal water exchanges between the upper wetland basin (above the Rt. 28 bridge) and Nantucket Sound. The remaining 9 infaunal sampling locations were spatially distributed throughout the upper wetland basin, which supports a strong salinity gradient, particularly at low tide, from nearly freshwater to salinities typical of Cape Cod embayments (26 ppt). As discussed above, the upper wetland basin is dominated primarily by salt marsh with fresh/brackish wetland in the upper western region (Figure I-2). All samples were to assess subtidal benthic animal communities either in the tidal river or in the major tidal creeks of the wetland. The assessment

Figure VII-8. Aerial photograph of the Herring River Estuary showing location of benthic infaunal community sampling stations (red symbol), November 2004.

of the habitat quality must take into account the very different environmental conditions resulting from the very different ecological and biogeochemical conditions created by their structure and function (e.g. wetland versus open water). Fundamentally, unimpaired wetlands are naturally nutrient and organic matter enriched and support communities adapted to these conditions. Open water basins, such as the tidal river (lower reach of the estuary) are less tolerant of nitrogen and organic matter enrichment and require moderate to high oxygen levels and moderate to low rates of organic matter for sustaining high quality infaunal animal habitat. Both basins were assessed relative to their ecological structure and function.

The Infauna Survey clearly indicated high quality habitat within both the upper wetland basin and in the tidal river reach of the lower estuary (Table VII-4). The upper wetland basin creeks are typical of salt marshes (upper reach eastern branch and main creek) and fresh/brackish (upper reach western branch). The communities reflect both the wetland dominated status and the influence of the large freshwater inflow from West Reservoir. The salt marsh creeks supported benthic animal communities with moderate to high numbers of individuals (200 to 900) and moderate numbers of species (11 to 13). The communities were dominated by crustaceans (mostly amphipods) and polychaetes with few if any indicators of impairment. Both the Diversity and Evenness were high for wetland creeks (0.63-0.67). Also indicative of a high quality habitat. The fresh/brackish upper western creek supported a highly productive community but distributed among a low number of species including more freshwater tolerant organisms (e.g. *Hypaniola, Cyanthura*).

Table VII-4. Benthic infaunal community data for Herring River Estuary, Town of Harwich, MA. 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-8.

Basin	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)	Stations HER# [1]
Herring River Estuary						
Upper Wetland Basin						
Upper-East	11	873	7.2	2.05	0.63	4, 5, 6
Upper West	8	1447	5.9	1.67	0.58	1,2,3
Main Creek	13	184	12.0	2.46	0.67	7,8,11
Lower Basin: Tidal River						
Tidal River	23	476	15.1	3.30	0.77	12,13,15,16
1- Station ID's refer to locations in Figures VII - 8.						

The lower tidal reach of the estuary, categorized as a tidal river, supported a different infaunal animal community, representative of high quality open water habitat. The tidal river supports a high number of individuals (~500) and a high number of species (20). Species numbers of 20-25 have been found to be typical of high quality habitat in s.e. Massachusetts by the MEP. Similarly, the community showed a high degree of diversity (3.3) and Evenness (0.77) exceeding the benchmarks (noted above) of 3.0 and 0.7, respectively. It should be noted that some organic enrichment species (tubficids, Capitellids) were found in depositional areas where their numbers approached almost 30% of the total individuals. However, the other species present were indicative of high quality habitat, suggesting that nutrient and organic matter loading may be approaching a threshold for supporting unimpaired infaunal habitat, particularly in the upper reach of the tidal river. The present diversity and Evenness indices for the tidal river are higher than the moderately impaired tidal river reach of the Parkers River Estuary, which was found to have diversity and Evenness of 2.94 and 0.61, respectively.

The Infauna Study indicated that the tidal creeks of the upper wetland dominated basin of the Herring River Estuary are presently supporting a typical salt marsh and fresh/brackish marsh infaunal habitat. Infauna communities within the tidal creeks were indicative of the organic rich environment typical of wetlands and consistent with the observed levels of oxygen depletion and watercolumn TN. The communities within the upper reach had moderate to high numbers of individuals, and moderate to low species numbers, with lower numbers of individuals and higher numbers of species within the lower reach of the main wetland creek. The observed communities were typical of New England salt marsh creek bottom environments in the summer. The communities generally contained some organic enrichment tolerant species. However, species like *Capitella* and *Streblospio,* typically observed in impaired embayment habitats did not dominate. The communities were composed of polychaetes, crustaceans and mollusks, with both crustaceans and polychaetes being the predominant taxa. The designation as high quality habitat is also supported by the absence of macroalgal accumulations and algal mats within the creek bottoms, which can result if there is "excessive" external nitrogen loading. The absence of macroalgal accumulations is consistent with the relatively low total nitrogen levels within this salt marsh, 0.71-0.84 mg N L^{-1} (tidally averaged), comparable to the unimpaired habitats of Little Namskaket Marsh (tidally averaged TN, 0.78-1.04 mg N L^{-1}) and also Cockle Cove Creek (a salt marsh in Chatham), which supports high quality habitats, both emergent marsh and creek bottom, at TN levels of 2 mg TN L^{-1}. Based upon all lines of evidence it appears that the upper wetland basin of the Herring River Estuary is presently supporting high quality infaunal habitat and has not exceeded its threshold nitrogen level for assimilating additional nitrogen without impairment. Similarly, the absence of macroalgal accumulations or algal mat development in the tidal river of the lower estuary is also consistent with an unimpaired habitat. However, the tidally averaged TN levels in the region of the Rt. 28 bridge (0.57 mg N L^{-1}) and the moderate numbers (<30%) of organic enrichment species in some samples suggests that the upper portion of the tidal river is near its nutrient and organic matter enrichment threshold, although the lower reach of the tidal river remains clearly below its threshold.

Overall, the pattern of infaunal habitat quality throughout the Herring River Estuary is consistent with measured dissolved oxygen concentrations, chlorophyll, nutrients and organic matter enrichment throughout this system. Classification of habitat quality necessarily includes the structure of the specific estuarine basin, specifically as to whether a basin area is wetland dominated such as the upper basin or tidal embayment dominated, such as the tidal river of the lower estuary. Based upon this analysis it is clear that the upper regions of the Herring River Estuary are presently supporting high quality habitat and are not impaired by their naturally high levels of nitrogen and organic matter enrichment. Similarly, the open water basin of the tidal river is presently supporting high quality benthic animal also consistent with its level of oxygen depletion and organic matter and nutrient enrichment and high water turnover. However, this open water basin does appear to be at or slightly below its threshold level of enrichment relative to benthic animal habitat in its upper most reach.

The results of the Infauna Survey does not indicate clear impairment of benthic habitat within the Herring River Estuary at present levels of organic matter and nitrogen loading. Therefore, it is recommended that nitrogen management threshold analysis (Section VIII), needs to focus on documented eelgrass loss from the lower estuary's tidal river basin, as the upper wetland basin appears to be well below its nitrogen loading threshold level. As infaunal habitat is less sensitive to the effects of nitrogen enrichment than is eelgrass, reducing the level of nitrogen enrichment to restore the impaired eelgrass habitat will also enhance infaunal habitat within the tidal river portion of the estuary.

Other Benthic Resources:

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. The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish area closures as well as the suitability of a system for the propagation of shellfish in a given system. As is the case with many systems on Cape Cod, large portions of the lower and middle reaches of the Herring River Estuary are conditionally approved for shellfishing during specific times during the year, typically the cold winter months, indicating the systems are generally supportive of shellfish communities (Figures VII-9, VII-10). However, in the upper most reaches of Herring River system, specifically in the marsh tidal creeks upgradient of Route 28, shellfishing is prohibited year round indicating persistent environmental contamination. In the case of the closure of shellfish habitat in the wetland dominated upper reach of the Herring River Estuary, that is likely due to a combination of natural bacterial contamination that occurs within a marsh systems due to waterfowl and other fauna and to a lesser extent anthropogenic sources of bacteria from residential septic systems in adjacent upland areas discharging groundwater directly to the marsh margin. The major shellfish species with potential habitat within the lower portion of the Herring River Estuary are quahogs (*Mercenaria*) and bay scallops (Figure VII-10). While the shellfish suitability assessment does not mean that these species currently exist within the system (though the presence of quahogs in the Herring River Estuary was documented by the MEP), should the habitat conditions improve based on nutrient management it is possible that the lower portion of the system would be supportive of these shellfish populations.

Figure VII-9. Division of Marine Fisheries shellfish growing areas and closure status for the Herring River Estuary, Harwich, MA.

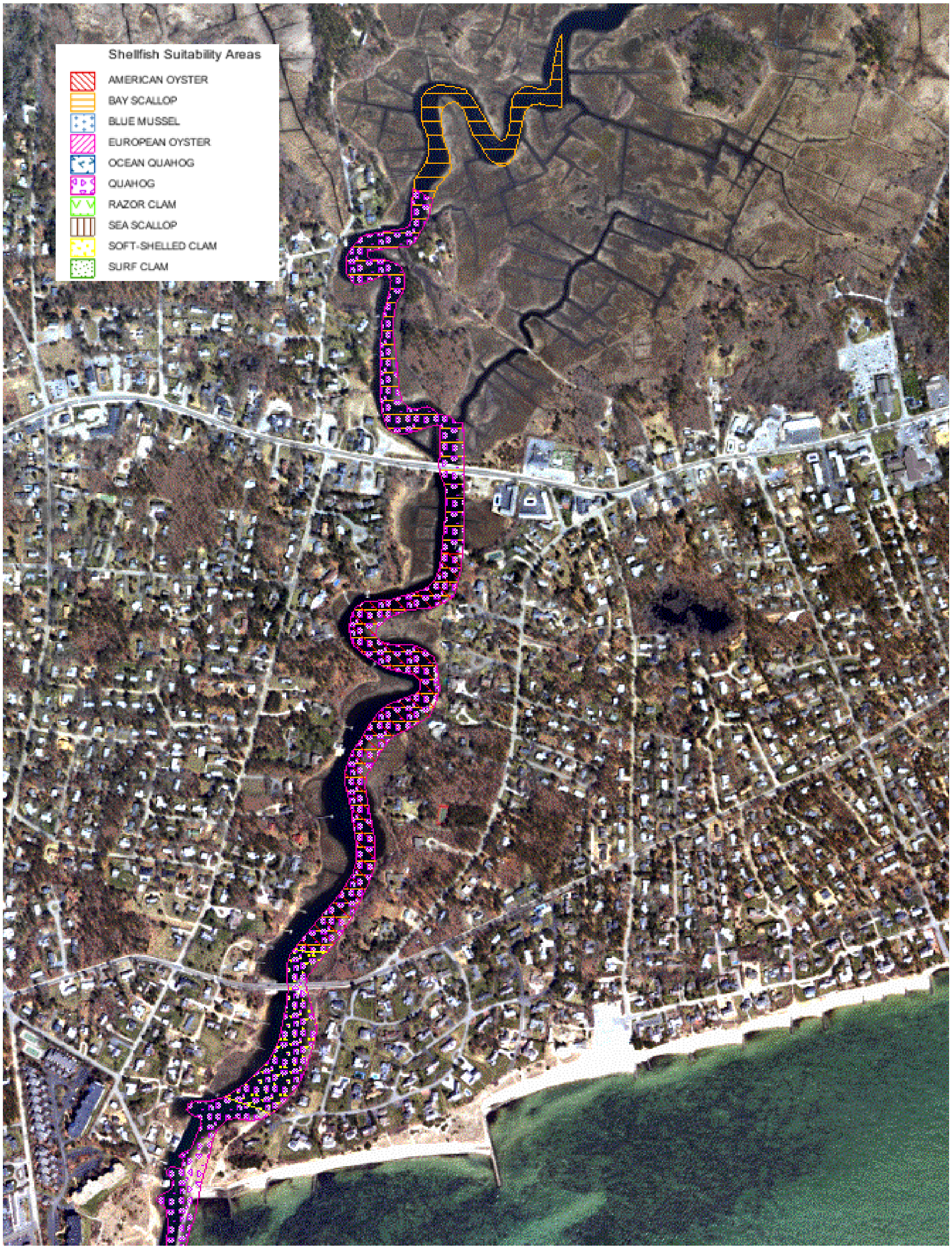

Figure VII-10. Areas within the Herring River Estuary, Harwich, MA. that are suitable habitat for specific shellfish species. Designation of areas as potentially supportive of shellfish propagation does not indicate that shellfish are currently present. Source: Mass GIS.

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

VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires integration of key habitat parameters (infauna and eelgrass), sediment characteristics, and nutrient related water quality information (particularly dissolved oxygen and chlorophyll). Additional information on temporal changes within each sub-embayment of an estuary, its associated watershed nitrogen load and geomorphological considerations of basin depth, stratification and functional type further strengthen the analysis. These data were collected to support threshold development for the Herring River Estuary by the MEP and were discussed in Section VII. Nitrogen threshold development builds on this data and links habitat quality to summer water column nitrogen levels from the baseline Harwich Water Quality Monitoring Program conducted by the Town of Harwich with analytical support from the Coastal Systems Analytical Facility at SMAST-UMass Dartmouth.

The Herring River Estuary is functionally a wetland receiving tidal waters through a open water tidal river. The Herring River wetlands are among the largest on Cape Cod (Strahler 1988), grading from salt marsh in the lower and mid reaches and along the major tidal creeks to brackish, predominantly freshwater marsh on the marsh plain in its upper regions. Although the Herring River Estuary functions primarily as a tidal wetland system, its lower reach close to the inlet is a tidal river with limited wetland vegetation (from inlet to Rt. 28 bridge), rather than being structured as a tidal marsh creek. In this area the tidal channel is relatively wide and navigable thus functioning more like an open water basin than a marsh. Up-gradient of Route 28, the channel narrows and intersects with numerous tidal ditches and smaller tributary marsh creeks. This upper wetland basin is typical of a large New England tidal marsh system, with the lower regions of predominantly salt marsh habitat, dominated by a central tidal creek and the marsh plain colonized by *Spartina alterniflora* (low marsh) and *Spartina patens and Distichlis spicata* (high marsh). The upper regions, furthest from the tidal inlet show the influence of the freshwater inflows, particularly from West Reservoir, with species grading to brackish marsh dominated by *Phragmites* and finally shifting to freshwater marsh dominated by *Typha* and other freshwater species (Figure I-2).

High quality habitat in open water basins support different communities than high quality habitat in tidal wetlands. The differences stem primarily from natural differences in organic matter and nutrient levels. These differences were important to the MEP assessment of the wetland dominated upper region and open water dominated tidal river comprising the lower region of the Herring River Estuary. The difference in structure above and below the Rt. 28 bridge created historic eelgrass habitat and benthic animal communities of more open water basins in the lower tidal reach and wetland dominated habitats in the upper wetland basin. This ecological difference results in a greater sensitivity to nitrogen in the lower tidal river portion than in the upper wetland dominated portions.

The measured levels of oxygen depletion and enhanced chlorophyll a levels are consistent with the observed habitat quality within the functional basin types (wetland/open water) throughout the Herring River Estuary. Particularly within the lower estuary's tidal river, the spatial pattern of total nitrogen levels (Section VI) and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate that the upper wetland portion of the Herring River Estuary is organic and nutrient enriched, a natural condition of temperate wetland systems. The oxygen data is consistent with high organic matter inputs from the surrounding vegetated marsh plain and to a lesser extent from phytoplankton production (chlorophyll a levels, generally ~15 ug/L at the wetland mooring site and ~11 ug/L in the upper wetland creeks. The observed level of oxygen depletion in the upper wetland basin of the Herring River Estuary is comparable to other high quality Cape Cod wetlands, showing a nearly identical pattern as the tidal creeks of nearby Namskaket and Little Namskaket Marshes within the Town of Orleans and salt marsh dominated ponds such as in the Parkers River and Centerville River estuaries.

Oxygen levels showed less depletion within the lower tidal river compared to the upper wetland creeks. It appears that low oxygen levels (to levels of 4 mg/L) within the tidal river during ebb tide are the result of the combined effects of out-flowing low oxygen water from the wetland dominated upper estuary plus uptake within the watercolumn and sediments of the tidal river, while on flooding tides the entry of oxygenated waters from Nantucket Sound generally produce oxygen levels >6 mg/L. This suggests that oxygen conditions within the tidal river portion of the Herring River Estuary are finely balanced and not solely the result of intra-basin processes. This is supported by the absence of significant wetlands bordering the tidal river and the generally low phytoplankton biomass in its watercolumn (chlorophyll a levels of 2 - 8 ug/L at the mooring site and averages of 5 - 7 ug/L, as measured by the Harwich Water Quality Monitoring Program, HWQMP). However, the higher oxygen levels and lower phytoplankton biomass in the tidal river are consistent with an open water basin that until recently supported eelgrass and presently supports high quality benthic habitat.

Overall the oxygen levels observed within the creeks of the upper basin are typical of wetland dominated creeks and are comparable to other similarly structured healthy wetland areas on Cape Cod. Tidal river oxygen conditions did exhibit daily excursions in oxygen levels, but the range of daily oxygen excursion and level of depletion was moderate. Based upon the level of depletion (periodically to 4 mg/L), there should be concern that should nitrogen enrichment increase, causing even greater oxygen depletion, the high quality benthic habitat in the tidal river will become impaired.

Eelgrass habitat has not historically existed within the creeks of the upper wetland dominated basin of the Herring River Estuary, consistent with other large wetland systems. Tidal creeks do not generally support eelgrass habitat, particularly when the creek drains significantly during each ebb tide. Further, the naturally high organic matter and nitrogen levels and low oxygen in large wetlands are not generally supportive of eelgrass development and growth. In contrast, the lower tidal reach of the Herring River Estuary, which functions as a tidal river carrying tidal exchange between the large upper wetland basin and Nantucket Sound has historically supported eelgrass. The MassDEP Eelgrass Mapping Program (C. Costello) documented eelgrass bed coverage in its 1995 and 2001 surveys and was independently confirmed in MEP surveys in 2004. Based on the 1995 and 2001 MassDEP surveys, eelgrass coverage within the tidal river up to Rt. 28 had declined significantly and by the 2010 survey the remaining coverage was negligible (Table VII-3).

Based upon all available information, it appears that the tidal creeks of the upper wetland basin of the Herring River Estuary are not structured to support eelgrass habitat. The lower estuarine reach (e.g. tidal river, below Route 28) is structured to support eelgrass habitat. Significant eelgrass coverage was documented by MassDEP in 1995, with much less coverage found in 2001 and 2004. Finally, at present eelgrass has nearly completely lost within the tidal river portion of the estuary. Therefore, threshold development for protection/restoration of this system should focus on restoration of eelgrass habitat within the tidal river. The temporal changes in eelgrass within the Herring River Estuary are consistent with the results of the benthic infauna analysis and the water quality data for this system.

The infauna survey clearly indicated high quality habitat within both the upper wetland dominated basin and the lower tidal river portion of the Herring River Estuary. This estuary contains a wide variety of habitats, with the upper wetland basin supporting a strong salinity gradient, from nearly freshwater to salinities typical of Cape Cod embayments (26 ppt). This upper basin is dominated primarily by salt marsh with fresh/brackish wetland in the upper western region. In addition unimpaired wetlands are naturally nutrient and organic matter enriched and support communities adapted to these conditions. The lower portion of the estuary is functionally an open water basin (tidal river) with habitats sensitive to nitrogen and organic matter enrichment and requiring moderate to high oxygen levels and moderate to low rates of organic matter for sustaining high quality infaunal animal habitat. Both basins were assessed relative to their ecological structure and function.

The upper wetland creek infaunal habitat quality are typical of healthy salt marshes (upper reach eastern branch and main creek) and fresh/brackish (upper reach western branch). The communities reflect both the wetland dominated status and the influence of the large freshwater inflow from West Reservoir. The salt marsh creeks supported benthic animal communities with moderate to high numbers of individuals (200 to 900) and moderate numbers of species (11 to 13). The communities were dominated by crustaceans (mostly amphipods) and polychaetes with few if any indicators of impairment. Both the diversity and Evenness were high for wetland creeks (0.63-0.67), indicative of a high quality habitat in wetlands. The fresh/brackish upper western creek supported a highly productive community but distributed among a low number of species including more freshwater tolerant organisms (e.g. *Hypaniola, Cyanthura*).

The lower tidal reach of the estuary, categorized as a tidal river, supported a different infaunal animal community, representative of high quality open water habitat. The tidal river supports a high number of individuals (~500) and a high number of species (20). Species numbers of 20-25 have been found to be typical of high quality habitat. Similarly, the community showed a high degree of diversity (3.3) and Evenness (0.77). However, some organic enrichment species (tubificids, Capitellids) were found in depositional areas, although the other species present were indicative of high quality habitat. The low-moderate numbers of organic enrichment tolerant species and moderate oxygen depletion, suggesting that nutrient and organic matter loading to the tidal river basin may be approaching a threshold for supporting unimpaired infaunal habitat, particularly in its upper region. The present diversity and Evenness indices for the tidal river are higher than the moderately impaired tidal river reach of the Parkers River Estuary, which was found to have diversity and Evenness of 2.94 and 0.61, respectively.

The Infauna Study indicated that the tidal creeks of the upper wetland dominated basin of the Herring River Estuary are presently supporting a typical salt marsh and fresh/brackish marsh infaunal habitat. Infauna communities within the tidal creeks were indicative of the organic rich environment typical of wetlands and consistent with the observed levels of oxygen depletion and watercolumn TN. The observed communities were typical of New England salt marsh creek bottom environments in the summer. The communities were composed of polychaetes, crustaceans and mollusks, with both crustaceans and polychaetes being the predominant taxa. The designation as high quality habitat is supported by the absence of macroalgal accumulations and algal mats within the creek bottoms, which can result if there is "excessive" external nitrogen loading. The absence of macroalgal accumulations is consistent with the relatively low total nitrogen levels within this salt marsh, 0.71-0.84 mg N L^{-1} (tidally averaged), comparable to the unimpaired habitats of Little Namskaket Marsh (tidally averaged TN, 0.78-1.04 mg N L^{-1}) and also Cockle Cove Creek (a salt marsh in Chatham), which supports high quality habitats, both emergent marsh and creek bottom, at TN levels of 2 mg TN L^{-1}. Based upon all lines of evidence it appears that the upper wetland basin of the Herring River Estuary is presently supporting high quality infaunal habitat and has not exceeded its threshold nitrogen level for assimilating additional nitrogen without impairment. Similarly, the absence of macroalgal accumulations or algal mat development in the tidal river of the lower estuary is also consistent with an unimpaired habitat. However, the tidally averaged TN levels in the region of the Rt. 28 bridge (0.57 mg N L^{-1}) and the moderate numbers of organic enrichment species in some samples suggests that the upper portion of the tidal river is near its nutrient and organic matter enrichment threshold, although the lower reach of the tidal river remains clearly below its threshold.

Overall, the pattern of infaunal habitat quality throughout the Herring River Estuary is consistent with measured dissolved oxygen concentrations, chlorophyll, nutrients and organic matter enrichment throughout this system. Classification of habitat quality necessarily includes the structure of the specific estuarine basin, specifically as to whether a basin area is wetland dominated such as the upper basin or tidal embayment dominated, such as the tidal river of the lower estuary. Based upon this analysis it is clear that the upper regions of the Herring River Estuary are presently supporting high quality habitat and are not impaired by their naturally high levels of nitrogen and organic matter enrichment. Similarly, the open water basin of the tidal river is presently supporting high quality benthic animal also consistent with its level of oxygen depletion and organic matter and nutrient enrichment and high water turnover. However, this open water basin does appear to be at or slightly below its threshold level of enrichment relative to benthic animal habitat in its upper most reach.

The results of the Infauna Survey does not indicate clear impairment of benthic habitat within the Herring River Estuary at present levels of organic matter and nitrogen loading. Therefore, it is recommended that nitrogen management threshold analysis (Chapter VIII.2), should focus on the recent losses of eelgrass habitat from the lower estuary's tidal river basin as the primary focus for nitrogen management, as the upper wetland basin appears to be well below its nitrogen loading threshold level. As infaunal habitat is less sensitive to the effects of nitrogen enrichment than is eelgrass, reducing the level of nitrogen enrichment to restore the impaired eelgrass habitat will also enhance infaunal habitat within the tidal river portion of the estuary. Determining the nitrogen target to restoring eelgrass habitat is the focus of the nitrogen management threshold analysis, below.

Table VIII-1. Summary of nutrient related habitat quality within the Herring River Estuary within the Town of Harwich, MA, based upon assessments in Section VII. The lower river is a tidal river connected to Nantucket Sound, functioning as an open-water basin (inlet to Rt. 28 bridge). In contrast, the upper wetland areas are predominantly extensive salt marsh and fresh/brackish tidal wetlands with tidal creeks (above Rt. 28 bridge). Note: HWQMP refers to the Harwich Water Quality Monitoring Program.

Health Indicator	Herring River Estuary			
	Tidal Wetlands (Upper Estuary)			Tidal River (Lower Estuary)
	West	Main Creek	East	
Dissolved Oxygen	H[1]	H[1]	H[1]	MI[2]
Chlorophyll	H-MI[3]	H[4]	H[4]	H[5]
Macroalgae	--[6]	--[6]	--[6]	--[7]
Eelgrass	--[8]	--[8]	--[8]	SI[9]
Infaunal Animals	H[10]	H[11]	H[11]	H[12]
Overall:	**H[13]**	**H[13]**	**H[13]**	**SI[14]**

1 – Wetland tidal creek, frequent oxygen depletion to ≤4 mg/L, periodically to <3 mg/L HWQMP; typical of healthy wetland systems that are naturally organic matter rich.
2 – oxygen levels influenced by ebbing low oxygen waters from upper wetland basin, time-series minimum ~4 mg/L and HWQMP (2002-08) periodically <4 mg/L; flooding tide >6 mg/L.
3 – moderate wetland summer chlorophyll levels averaging 18 ug/L (HWQMP, 2002-2008), but represent phytoplankton entering in discharge from West Reservoir, rather than *in situ* production.
4 – low to moderate wetland summer chlorophyll levels averaging 6-11 ug/L (HWQMP, 2002-2008) and 10.3 ug/L (MEP mooring 2004), maxima generally <15 ug/L but periodic blooms to 25 ug/L.
5 – low open-water basin chlorophyll, average 2004 = 4.7ug/L rarely >10 ug/L; HWQMP 2002-08 average at Rt. 28 = 6.7 ug/L, near inlet = 4.9 ug/L rarely >10 ug/L.
6 -- drift algae sparse or absent, no evidence of surface microphyte mat.
7 -- very sparse patches of drift algae, *Ulva*, with small areas of attached *Codium*.
8 – no evidence this basin historically supported eelgrass, typical of wetland dominated tidal creeks.
9 -- MassDEP (C. Costello) indicates that eelgrass lost from this system between 1995-2010, 2004 field survey indicated discontinuous eelgrass beds in tidal river.
10 -- community consistent with healthy fresh/brackish marsh, high numbers of individuals low numbers of species, diversity and Evenness, low salinity tolerant species present in upper reach, species indicative of disturbance not present, amphipods and surface deposit feeders dominant.
11 -- community consistent with healthy salt marsh, high numbers of individuals, moderate species, for salt marshes a high diversity and Evenness, community dominated by crustaceans and polychaetes, with only low numbers of individuals from species indicative of disturbance; the organic enrichment indicators present are typical of salt marsh creeks and ponds.
12 -- high numbers of species (23) and high number of individuals (500), with high diversity (3.3) and Evenness (0.77), dominated by crustaceans, polychaetes and mollusks, some organic and nutrient enrichment species present, but other species indicative of a high quality habitat present.
13 -- High quality habitat as part of an extensive healthy wetland system comprised of fresh/brackish and salt marsh areas. Less oxygen depletion than many wetlands and chlorophyll a generally moderate, no evidence of historic eelgrass coverage, consistent with wetland structure. Infaunal communities with large numbers of individuals and species and indices typical of wetland creeks.
14 -- Significant Impairment due to MassDEP surveys with loss of eelgrass, 1995-2001-2010, infauna habitat near threshold in upper portion, presently supporting high quality habitat overall; consistent with absence of macroalgae, low-moderate chlorophyll, although some moderate oxygen depletion.

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

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 and its 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 Herring River Estuary 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.

As described in detail above, the Herring River Estuary is composed of 2 major basins, an upper large fresh/brackish and salt marsh dominated basin with tidal creeks and an open water tidal river which carries tidal exchanges between the upper wetland dominated basin and Nantucket Sound. The upper wetland basin is typical of a large New England tidal marsh system, with the lower regions of predominantly salt marsh, dominated by a central tidal creek and the marsh plain colonized by *Spartina alterniflora* (low marsh) and *Spartina patens and Distichlis spicata* (high marsh). Within this basin the uppermost regions, furthest from the tidal inlet show the influence of the freshwater inflows, particularly from West Reservoir, with species grading to brackish marsh dominated by *Phragmites* finally shifting to freshwater marsh dominated by *Typha* and other freshwater species (Figure I-2). High quality habitat in open water basins support different communities than high quality habitat in tidal wetlands. The differences stem primarily from natural differences in organic matter and nutrient levels. This ecological difference results in a greater sensitivity to nitrogen in the lower tidal river portion than in the upper wetland dominated portions, which was taken into account in the MEP assessment of the wetland dominated upper region and open water dominated tidal river comprising the lower region of the Herring River Estuary.

Overall the oxygen levels observed within the creeks of the upper basin are typical of wetland dominated creeks and are comparable to other similarly structured healthy wetland areas on Cape Cod. Tidal river oxygen conditions did exhibit daily excursions in oxygen levels, but the range of daily oxygen excursion and level of depletion was moderate. Based upon the level of depletion (periodically to 4 mg/L), there is concern that should nitrogen enrichment increase, causing even greater oxygen depletion, the high quality benthic habitat in the tidal river will become impaired.

Eelgrass habitat has not historically existed within the creeks of the upper wetland dominated basin of the Herring River Estuary, consistent with other large wetland systems analyzed by the MEP. Tidal creeks do not generally support eelgrass habitat, particularly when the creek drains significantly during each ebb tide. Further, the naturally high organic matter and nitrogen levels and low oxygen in large wetlands are not generally supportive of eelgrass development and growth. However, the upper basin currently supports high quality benthic animal habitat throughout the network of tidal creeks. Infauna communities within the tidal creeks were indicative of the organic rich environment typical of wetlands and consistent with the observed levels of oxygen depletion and watercolumn TN concentration. The observed communities were typical of New England salt marsh creek bottom environments in the summer, composed of polychaetes, crustaceans and mollusks, with both crustaceans and polychaetes being the predominant taxa. The designation as high quality habitat is supported by the absence of macroalgal accumulations and algal mats within the creek bottoms, which can result if there is "excessive" external nitrogen loading. The absence of macroalgal accumulations is consistent with the relatively low total nitrogen levels within this salt marsh, 0.71-0.84 mg N L^{-1} (tidally averaged), comparable to the unimpaired habitats of Little Namskaket Marsh (tidally averaged TN, 0.78-1.04 mg N L^{-1}) and also Cockle Cove Creek (a salt marsh in Chatham), which supports high quality habitats, both emergent marsh and creek bottom, at TN levels of 2 mg TN L^{-1}. Based upon all lines of evidence it appears that the upper wetland basin of the Herring River Estuary is presently supporting high quality infaunal habitat, is structurally unable to support eelgrass, and has not exceeded its threshold nitrogen level for assimilating additional nitrogen without impairment.

In contrast, the lower tidal reach of the Herring River Estuary, which functions as a tidal river carrying tidal exchange between the large upper wetland basin and Nantucket Sound has historically supported eelgrass. The MassDEP Eelgrass Mapping Program (C. Costello) documented eelgrass bed coverage in its 1995 and 2001 surveys and eelgrass presence was confirmed in MEP surveys in 2004. Based on the 1995 and 2001 MassDEP surveys, eelgrass coverage within the tidal river up to Rt. 28 had declined significantly and by the 2010 survey the remaining coverage was negligible (Table VII-3). However, this tidal river supports a high number of individuals (~500) and a high number of species (20). Species numbers of 20-25 have been found to be typical of high quality habitat. Similarly, the community showed a high degree of diversity (3.3) and Evenness (0.77). Although there were some organic enrichment species (tubficids, Capitellids) found in depositional areas, the other species present were indicative of high quality habitat. The low-moderate numbers of organic enrichment tolerant species and moderate oxygen depletion suggests that nutrient and organic matter loading to the tidal river basin may be approaching a threshold for supporting unimpaired infaunal habitat, particularly in its upper region. The present Diversity and Evenness indices for the tidal river are higher than the moderately impaired tidal river reach of the Parkers River Estuary, which was found to have diversity and Evenness of 2.94 and 0.61, respectively. The absence of macroalgal accumulations or algal mat development within the tidal river is consistent with an unimpaired habitat. However, the tidally averaged TN levels in the region of the Rt. 28 bridge (0.57 mg N L^{-1}) and the moderate numbers of organic enrichment species in some samples suggests that the upper portion of the tidal river is near its nutrient and organic matter enrichment threshold, although the lower reach of the tidal river remains clearly below its threshold.

It appears that low oxygen levels (to levels of 4 mg/L) within the tidal river during ebb tide are the result of the combined effects of out-flowing low oxygen water from the wetland dominated upper estuary plus uptake within the watercolumn and sediments of the tidal river, while on flooding tides, the entry of oxygenated waters from Nantucket Sound generally produce oxygen levels >6 mg/L. This suggests that oxygen conditions within the tidal river portion of the Herring River Estuary are finely balanced and not solely the result of intra-basin processes. This is supported by the absence of significant wetlands bordering the tidal river and the generally low phytoplankton biomass in its watercolumn (chlorophyll a levels of 2 - 8 ug/L at the mooring site and averages of 5 - 7 ug/L, as measured by the Harwich Water Quality Monitoring Program, HWQMP). However, the higher oxygen levels and lower phytoplankton biomass in the tidal river are consistent with an open water basin that until recently supported eelgrass and presently supports high quality benthic habitat.

The measured levels of oxygen depletion and enhanced chlorophyll a levels are consistent with the observed habitat quality within the functional basin types (wetland/open water) throughout the Herring River Estuary. Particularly within the lower estuary's tidal river, the spatial pattern of total nitrogen levels and the temporal changes in habitat quality are consistent with watershed based nitrogen enrichment. At present there is no clear impairment of benthic habitat within the Herring River Estuary at existing levels of organic matter and nitrogen loading. Therefore, nitrogen management should focus on the recent losses of eelgrass habitat from the lower estuary's tidal river basin, as the upper wetland basin appears to be well below its nitrogen loading threshold level. As infaunal habitat is less sensitive to the effects of nitrogen enrichment than is eelgrass, reducing the level of nitrogen enrichment to restore the impaired eelgrass habitat will also enhance infaunal habitat within the tidal river portion of the estuary.

The results indicate that eelgrass has been lost from the Herring River Estuary in areas that presently support tidally averaged TN levels of 0.57 mg N L^{-1}. The absence of eelgrass within the tidal inlet is attributed to inlet maintenance and unstable sediments in the high velocity zone. At lower nitrogen levels eelgrass was observed in 2004, but with epiphytes and losses of coverage. These sites are associated with ~0.50 mg N L^{-1} (as interpolated by the water quality model). In other similar systems "healthy" beds have been observed at <0.428 mg N L^{-1} and 0.421 mg N L^{-1} in the East and West Branches of the Westport River Estuary, which also has extensive up gradient wetlands. It appears that in the Westport River Estuary, the TN level to support high quality eelgrass habitat may be greater than 0.43 mg N L^{-1}, but less than 0.50 mg N L^{-1}. Given these results and the configuration and depth of the eelgrass areas of the Herring River Estuary system, a comparison to other estuaries was undertaken to refine the threshold.

It appears from the recent loss of eelgrass in the Herring River Estuary that eelgrass habitat was supported at higher TN levels than generally found in high quality eelgrass habitat such as in deeper systems (>2 m) like Stage Harbor (0.38 mg L-1) or West Falmouth Harbor and Phinneys Harbor (0.35 mg L^{-1}). However, in systems with shallow water or where the tidal exchange places clear low nutrient water over the eelgrass for half of the tide, like in the tidal river reach of the Herring River Estuary, eelgrass beds are sustainable at higher tidally averaged TN levels. At shallow depths in Bournes Pond, eelgrass can still be found (although heavy with epiphytes) at the mouth of the upper tributary at a tidally averaged TN concentration of 0.481 mg TN L^{-1}, while the more stable beds in the lower region of Israel's Cove have at a tidally averaged TN of 0.429 mg TN L^{-1}. It should be added that eelgrass can persist at nitrogen levels that are non-supportive of healthy beds, and eelgrass within Hamblin Pond persisted at a high TN level (0.5 mg L-1), long after eelgrass within the central portion of Waquoit Bay had disappeared. But the 0.5 mg N L^{-1} TN level in Hamblin Pond was associated with diminishing eelgrass patches and was just beyond the level supportive of high quality habitat. These higher levels (~0.5 mg L^{-1}) were also associated with impaired eelgrass areas in the Westport River System. All of the eelgrass information for the Herring River Estuary indicates that the nitrogen threshold level supportive of high quality eelgrass habitat is close to, but less than 0.50 mg N L^{-1}. Given the structure of the tidal river, particularly the existence of the extensive upper wetland system, and the recent loss and current TN levels, it appears that TN levels need to be lowered to 0.48 mg L^{-1} at the Sentinel Station (HER-7) at the Rt. 28 bridge. This level is higher than the poorly flushed Bournes Pond eelgrass beds, but the same as for the lower West Branch of the Westport River with its extensive wetlands up-gradient from the eelgrass beds. The threshold is significantly effected by the very high water quality during flood tides (0.32 - 0.34 mg N L^{-1}) which is supportive of eelgrass coverage versus the relatively poor water quality (for open water systems) during the ebbing tides (0.68 - 0.77 mg N L^{-1}) due to out-flow from the extensive upper wetland basin. The result is that the threshold must take into account the daily variation in conditions not just the average condition. It should be noted that lowering the tidally averaged TN level to the threshold will also protect the high quality benthic animal habitat throughout the tidal river.

VIII.3. DEVELOPMENT OF TARGET NITROGEN LOADS

The nitrogen thresholds developed in the previous section were in turn used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Herring River System. Tidally averaged total nitrogen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel stations chosen for the Herring River System (HAR-7 is located approximately at the upper limit of historic eel grass within the system). It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches. Community discussions should review this option and consider evaluation of other alternatives. The presentation below is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required removal of 100% of the septic nitrogen load from two watersheds (100% of the septic nitrogen load removed from watersheds 26 and 27) and 50% of the septic nitrogen load from three watersheds to the Herring River Estuary (50% of the septic nitrogen load removed from watersheds 19, 20, and 21). The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1.

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

sub-embayment	present septic load (kg/day)	threshold septic load (kg/day)	threshold septic load % change
Lower Herring River	7.063	7.063	0.0%
East Reservoir	0.047	0.047	0.0%
Upper Herring River	10.468	0.000	-100.0%
Surface Water Sources			
West Reservoir	12.137	12.137	0.0%
Lothrop Road	8.877	4.504	-49.3%

Tables VIII-3 and VIII-4 provide additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. Removal of septic loads from Herring River watersheds results in the total nitrogen loads presented in Table VIII-4. Table VIII-4 shows the breakdown of threshold sub-embayment and surface water loads used for total nitrogen modeling. In Table VIII-4, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent ‘worst-case’ summertime conditions.

The benthic flux for this modeling effort is reduced from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Nantucket Sound.

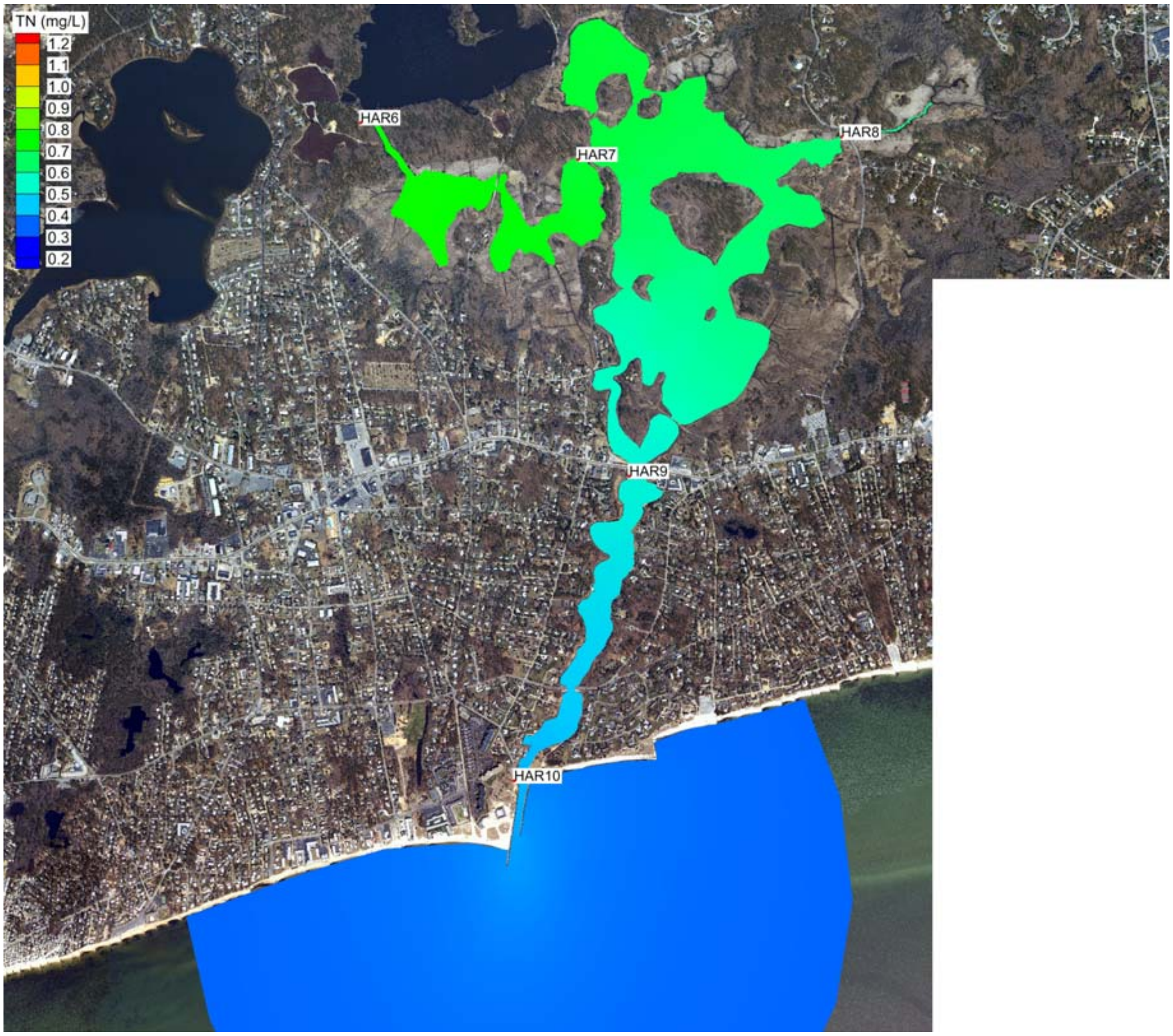

Figure VIII-1. Contour plot of modeled average total nitrogen concentrations (mg/L) in Herring River system, for threshold conditions (0.48 mg/L at water quality monitoring station HAR-7). The approximate location of the sentinel threshold station for Herring River (HAR-7) is shown.

Table VIII-3. Comparison of sub-embayment ***total attenuated watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Herring River system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.			
sub-embayment	present total load (kg/day)	threshold load (kg/day)	threshold % change
Lower Herring River	9.036	9.036	0.0%
East Reservoir	0.293	0.293	0.0%
Upper Herring River	13.296	2.827	-78.7%
Surface Water Sources			
West Reservoir	27.564	27.564	0.0%
Lothrop Road	12.627	8.255	-34.6%

Table VIII-4. Threshold sub-embayment loads and attenuated surface water loads used for total nitrogen modeling of the Herring River system, with total watershed N loads, atmospheric N loads, and benthic flux			
sub-embayment	threshold load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Lower Herring River	9.036	0.252	1.249
East Reservoir	0.293	0.000	0.628
Upper Herring River	2.827	0.395	-1.464
Surface Water Sources			
West Reservoir	27.564	-	-
Lothrop Road Stream	8.255	-	-

Comparison of model results between existing loading conditions and the selected loading scenario to achieve the target TN concentrations at the sentinel stations is shown in Table VIII-5. To achieve the threshold nitrogen concentrations at the sentinel station within Herring River, a reduction in TN concentration of approximately 16% was required at the long-term monitoring station HAR-7 (Sentinel Station).

Table VIII-5. Comparison of model average total N concentrations from present loading and the modeled threshold scenario, with percent change, for the Herring River system. Sentinel threshold station, HAR-7, to restore eelgrass habitat within the lower tidal river is in bold print.				
Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Wixen Dock	HAR-6	0.425	0.384	-9.6%
Rt. 28 Bridge	**HAR-7**	**0.567**	**0.479**	**-15.5%**
Lothrop Rd	HAR-8	0.840	0.583	-30.6%
North Rd	HAR-9	0.776	0.648	-16.6%
West Reservoir	HAR-10	0.710	0.709	-0.1%

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

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.

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