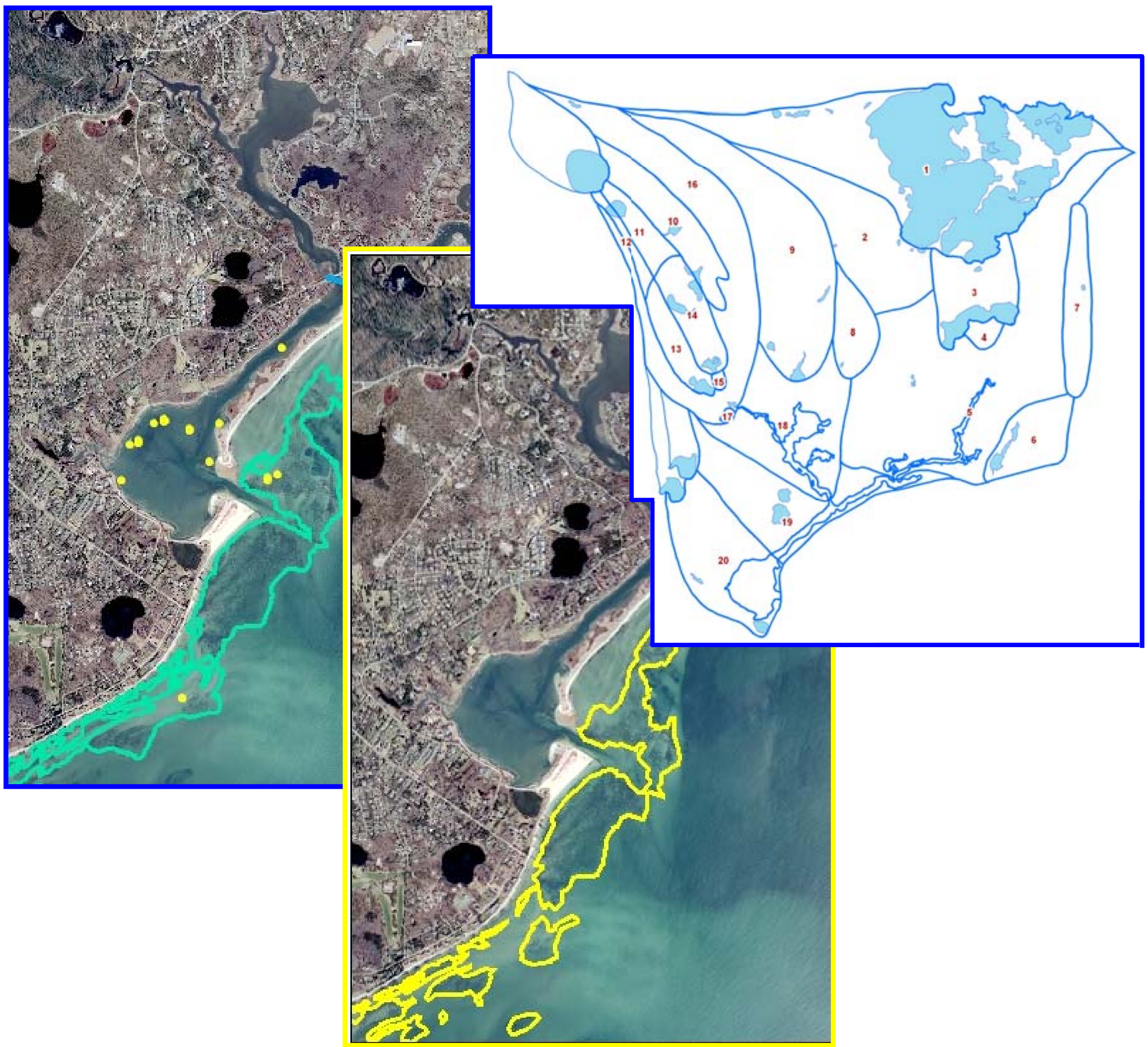
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Centerville River, Town of Barnstable, Massachusetts



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Environmental Protection

FINAL REPORT – NOVEMBER 2006

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I. INTRODUCTION

The Centerville River Embayment System is a complex estuary located within the Town of Barnstable on Cape Cod, Massachusetts with a southern shore bounded by water from Nantucket Sound (Figure I -1). The estuary is composed of a lagoon formed behind a barrier spit and running parallel to the shoreline, comprised of the Centerville River and East Bay, and a drowned river valley estuary, Bumps River/Scudder Bay. The Bay's watershed is distributed entirely within the Town of Barnstable. A large portion of the overall watershed includes the sub-watersheds contributing direct groundwater discharge to the estuary and contributing to the four surface water discharges flowing with to the estuarine reach of the Bumps River (Skunknett River, Bumps River) or directly into Centerville River (stream from Long Pond, stream from Lake Elizabeth). Although land-uses closest to an embayment generally have greater impact than those in the upper portions of the watershed, which are subject to nitrogen attenuation during transport through natural aquatic systems (e.g. ponds, rivers, wetlands etc.) prior to discharge to the embayment, effective restoration of the Centerville River System, will require the Town of Barnstable to be active in nutrient management throughout the watershed to the overall system. This will be made easier by virtue of the fact that the watershed to the Centerville River System resides entirely within the boundary of the Town of Barnstable.

The number of sub-embayments (East Bay, Centerville River, Bumps River, Scudder Bay, Centerville River marshes) to the Centerville River System greatly increases the shoreline and decreases the travel time of groundwater (and its pollutants) from the watershed recharge areas to bay regions of discharge. The nature of enclosed embayments in populous regions brings two opposing elements to bear: as protected marine shoreline they are popular regions for boating, recreation, and land development; as enclosed bodies of water, they may not be readily flushed of the pollutants that they receive due to the proximity and density of development near and along their shores. In particular, the Centerville River system and its sub-embayments along the Barnstable shores are at risk of eutrophication (over enrichment) from high nitrogen loads in the groundwater and runoff from their watersheds.

The Centerville River Embayment System is a complex estuary with one inlet connecting Centerville Harbor-Nantucket Sound to Centerville River and a number of sub-embayments (East Bay, Centerville River, Bumps River, Scudder Bay, Centerville River marshes) as depicted in Figure I-1. Centerville Harbor abuts Nantucket Sound and is bounded to the west by Dowes Beach in the vicinity of the inlet to the estuary and Halls Creek Salt Marsh located along the most eastern boundary of the Harbor. The Centerville River Estuary receives tidal waters from Nantucket Sound which flow into a single lower basin (East Bay) located on the western end of Centerville River. East Bay and Centerville River are separated from Centerville Harbor by a barrier beach. The barrier beach is commonly known as Dowes Beach to the west of the inlet to Centerville River, Long Beach immediately to the east of the inlet and then Long Beach transitions into Craigville Beach. Moving east from Craigville Beach is Coville Beach which terminates in a small tidal inlet which allows Nantucket Sound flood waters to enter the Halls Creek Salt Marsh system (Figure I-2). The inlet connecting Centerville Harbor and East Bay – Centerville River is a feature that has very likely migrated along the barrier beach as a function of longshore transport of sediments and coastal storms. Currently, the inlet is armored and stable and the Town of Barnstable periodically dredges the inlet channel to keep the inlet and East Bay navigable. Centerville River runs west to east behind the barrier beach and terminates in a salt marsh system commonly referred to as the Centerville Marshes. Midway along the west-east axis of the Centerville River, the Centerville River is joined with the Bumps River flowing south from the up reaches of the estuarine system. The confluence of the estuarine Bumps River with Centerville River is a very shallow area of the Centerville River reach and has periodically been dredged to keep the waterway navigable as described in Section II. At the uppermost end of the estuarine reach of Bumps River is located Scudder Bay, a terminal sub-embayment that receives direct fresh surfacewater inflow from the watershed via the Bumps River. The estuarine reach of the Bumps River also receives fresh surfacewater inflow from the watershed via the Skunknett River. Unlike the more upland habitat that characterizes the shores of Scudder Bay, salt marsh dominates the shoreline at the eastern most end of the Centerville River estuarine reach. Centerville River proceeds in a northeasterly direction upwards towards Long Pond and receives direct fresh surface water discharge from Long Pond and Lake Wequaquet. Lower in the salt marsh system is located another fresh surface water input that discharge from Lake Elizabeth to the east. These smaller sub-embayments (including Bumps River and Scudder Bay) constitute important components of the Town's natural and cultural resources.

Figure I-1. Study region for the Massachusetts Estuaries Project analysis of the Centerville River Embayment System. Tidal waters enter the Bay through one inlet from Nantucket Sound. Freshwaters enter from the watershed primarily through 4 surface water discharges (Skunknett River, Bumps River, a stream from Long Pond and a stream from Lake Elizabeth) and direct groundwater discharge.

Figure I-2. Topographic Map of the Centerville River System depicting major geographic features.

The present Centerville River system results from tidal flooding of drowned river valleys formed primarily by the Bumps River discharging to Scudder Bay and Skunknett River discharging to estuarine reach of the Bumps River. In the northeast portion of the Centerville River system, the stream flowing from Long Pond likely contributed to the morphology of this portion of the system as would also be so for the small stream flowing from Lake Elizabeth (a small coastal kettle pond). Drowning of the river valleys occurred gradually as a result of rising sea level following the last glaciation approximately 18,000 years BP. Coastal processes, including the formation of a barrier spit (beach and dune deposits) have altered the positions of the tidal inlet(s) to the Centerville River system and affecting tidal exchange between the open water Centerville Harbor and the enclosed Centerville River. Most of the estuarine reach of the Centerville River and East Bay comprise a lagoon formed behind a barrier spit (Long Beach - Craigville Beach), which separates the estuary from Centerville Harbor and Nantucket Sound. The barrier spit grew from the southeastern shore, and is a very dynamic geomorphic feature.

The primary ecological threat to the Centerville River System as a coastal resource is degradation resulting from nutrient enrichment. Although the enclosed estuarine system has some bacterial contamination issues related to stormwater run-off from the watershed, these do not appear to be having large system-wide impacts. Bacterial contamination causes closures of shellfish harvest areas within the East Bay and Bumps River sub-embayments as well as portions of Centerville River. In contrast, loading of the critical eutrophying nutrient, nitrogen, to the Centerville River System has been greatly increased over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to the Centerville River Estuary, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater.

The Town of Barnstable has been among the fastest growing towns in the Commonwealth over the past two decades and does have a centralized wastewater treatment system located in Hyannis. Even so, the vast majority of the Centerville River System watershed is not connected to any municipal sewerage system and wastewater treatment and disposal is primarily based on privately maintained septic systems. As existing and probable increasing levels of nutrients impact Barnstable's coastal embayments, water quality degradation will accelerate, with further harm to invaluable environmental resources.

As the primary stakeholder to the Centerville River System, the Town of Barnstable was among the first communities to become concerned over perceived degradation of Bay waters. The concern over declining habitat quality led directly to the establishment of a comprehensive water quality monitoring program aimed at determining the degree to which waters of the Town's embayments maybe be impaired. The Town of Barnstable Water Quality Monitoring Program was provided technical assistance by the Coastal Systems Program at SMAST-UMD and over the past several years has had the Water Quality Monitoring Program of the Three Bays Preservation Trust merged into Barnstable's Town-wide embayment monitoring program. This effort provides the quantitative watercolumn nitrogen data (2001-2005) required for the implementation of the MEP's Linked Watershed-Embayment Approach used in the present study.

The common focus of the Barnstable Water Quality Monitoring effort has been to gather site-specific data on the current nitrogen related water quality throughout the Centerville River System and determine its relationship to 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 MEP effort builds upon the Water Quality Monitoring

Program, and previous hydrodynamic and water quality analyses conducted in support of an Environmental Impact Report for the Centerville River Dredging Project that got underway in the early part of 2002. The details of the EIR are presented in Section II. The MEP approach includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for each major sub-embayment. These critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater planning and nitrogen management alternatives development needed by the Town of Barnstable. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff and volunteers over many years, most notably from members of the local non-governmental organization (NGO) Three Bays Preservation. The modeling tools developed as part of this program provide the quantitative information necessary for the Town of Barnstable to develop and evaluate the most cost effective nitrogen management alternatives to restore this valuable coastal resource which is currently being degraded by nitrogen overloading. It is important to note that the Centerville River System has been significantly altered by human activities over the past ~100 years or more (see Section I.2, below). As a result, the present nitrogen "overloading" appears to result partly from alterations to the geomorphology and ecological systems. These alterations subsequently affect nitrogen loading and transport within the watershed and influence the degree to which nitrogen loads impact the estuary. Therefore, restoration of this system should focus on managing nitrogen through both management of nitrogen loading within the watershed and restoration/management of processes which serve to lessen the amount or impact of nitrogen entering the estuary.

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 and the food chain which they support. At higher levels, nitrogen loading from surrounding watersheds causes aesthetic degradation and inhibits even recreational uses of coastal waters. In addition to nutrient related ecological declines, an increasing number of embayments are being closed to swimming, shellfishing and other activities as a result of bacterial contamination. While bacterial contamination does not generally degrade the habitat, it restricts human uses. However like nutrients, bacterial contamination is frequently related to changes in land-use as watersheds become more developed. The regional effects of both nutrient loading and bacterial contamination span the spectrum from environmental to socio-economic impacts and have direct consequences to the culture, economy, and tax base of Massachusetts's coastal communities.

The primary nutrient causing the increasing impairment of the Commonwealth's coastal embayments is nitrogen and the primary sources of this nitrogen are wastewater disposal, fertilizers, and changes in the freshwater hydrology associated with development. At present there is a critical need for state-of-the-art approaches for evaluating and restoring nitrogen sensitive and impaired embayments. Within southeastern Massachusetts alone, almost all of the municipalities (as is the case with the Town of Barnstable) are grappling with

Comprehensive Wastewater Planning and/or environmental management issues related to the declining health of their estuaries.

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

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

In appropriate estuaries, bacterial technical reports will be developed in support of a Cape Cod wide TMDL for bacterial contamination. As possible, these analyses of bacterial contamination will be conducted in concert with the nutrient effort (particularly if there is a 303d listing), as was the case for the Prince’s Cove sub-embayment to the Three Bays system. Currently, the MEP (through SMAST) has not been tasked with a technical assessment of bacterial contamination in the Centerville River System for inclusion of this system into the Cape Cod wide bacterial TMDL that the MassDEP is in the process of producing.

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

The major MEP nitrogen management goals are to:

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

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

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

The Linked Model has been applied for watershed nitrogen management in approximately 23 embayments throughout southeastern Massachusetts. In these applications it has become clear that the Linked Model Approach's greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing "what if" scenarios for evaluating watershed nitrogen management options.

The Linked Watershed-Embayment Model when properly parameterized, calibrated and validated for a given embayment becomes a nitrogen management planning tool, which fully supports TMDL analysis. The Model facilitates the evaluation of nitrogen management alternatives relative to meeting water quality targets within a specific embayment. The Linked Watershed-Embayment Model also enables Towns to evaluate improvements in water quality relative to the associated cost. In addition, once a model is fully functional it can be "kept alive" and updated for continuing changes in land-use or embayment characteristics (at minimal cost). In addition, since the Model uses a holistic approach (the entire watershed, embayment and tidal source waters), it can be used to evaluate all projects as they relate directly or indirectly to water quality conditions within its geographic boundaries.

Linked Watershed-Embayment Model Overview: The Model provides a quantitative approach for determining an embayment's: (1) nitrogen sensitivity, (2) nitrogen threshold loading levels (TMDL) and (3) response to changes in loading rate. The approach is both calibrated and 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:

- Watercolumn 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 Centerville River Embayment System exchanges tidal water with Nantucket Sound through one inlet at the west end of a barrier beach referred to as Long – Craigville Beach. The inlet connecting Centerville Harbor to East Bay was opened by dredging in the early 1900's and is armored on the west side (Dowes Beach) and remains in a "natural" un-stabilized state on the east side (Long Beach). For the MEP analysis, the Centerville River Estuarine System has been partitioned into four general sub-embayment groups: the 1) East Bay 2) Centerville River West 3) Centerville River East, including the marshes northeast of bridge, and 4) Bumps River (inclusive of Scudder Bay) as depicted in Figure I-1.

Within the overall Centerville River System is seen a diversity of estuarine habitats, including the tidal portion of East Bay and Centerville River which operates as an embayment, the Bumps River and Scudder Bay with associated fringing wetlands and the large salt marsh area at the eastern end of the Centerville River. Most of the System's salt marsh area is to the east and associated with Bumps River and has shallow tidal flats and large salinity fluctuations. In contrast, East Bay and Centerville River show more typical embayment characteristics with a mixture of open water areas and channels, small fringing salt marshes and relatively stable salinity gradients. Although the upper sub-embayment system of Bumps River and Scudder Bay up-gradient of Centerville River exhibit different hydrologic characteristics (river dominated versus tidally dominated), the tidal forcing for these systems is generated from Nantucket Sound. Nantucket Sound exhibits a moderate to low tide range, with a mean range of about 2.5 ft. Since the water elevation difference between Nantucket Sound and the Centerville River System is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle (note the tide range off Stage Harbor Chatham is ~4.5 ft, Wellfleet Harbor is ~10 ft).

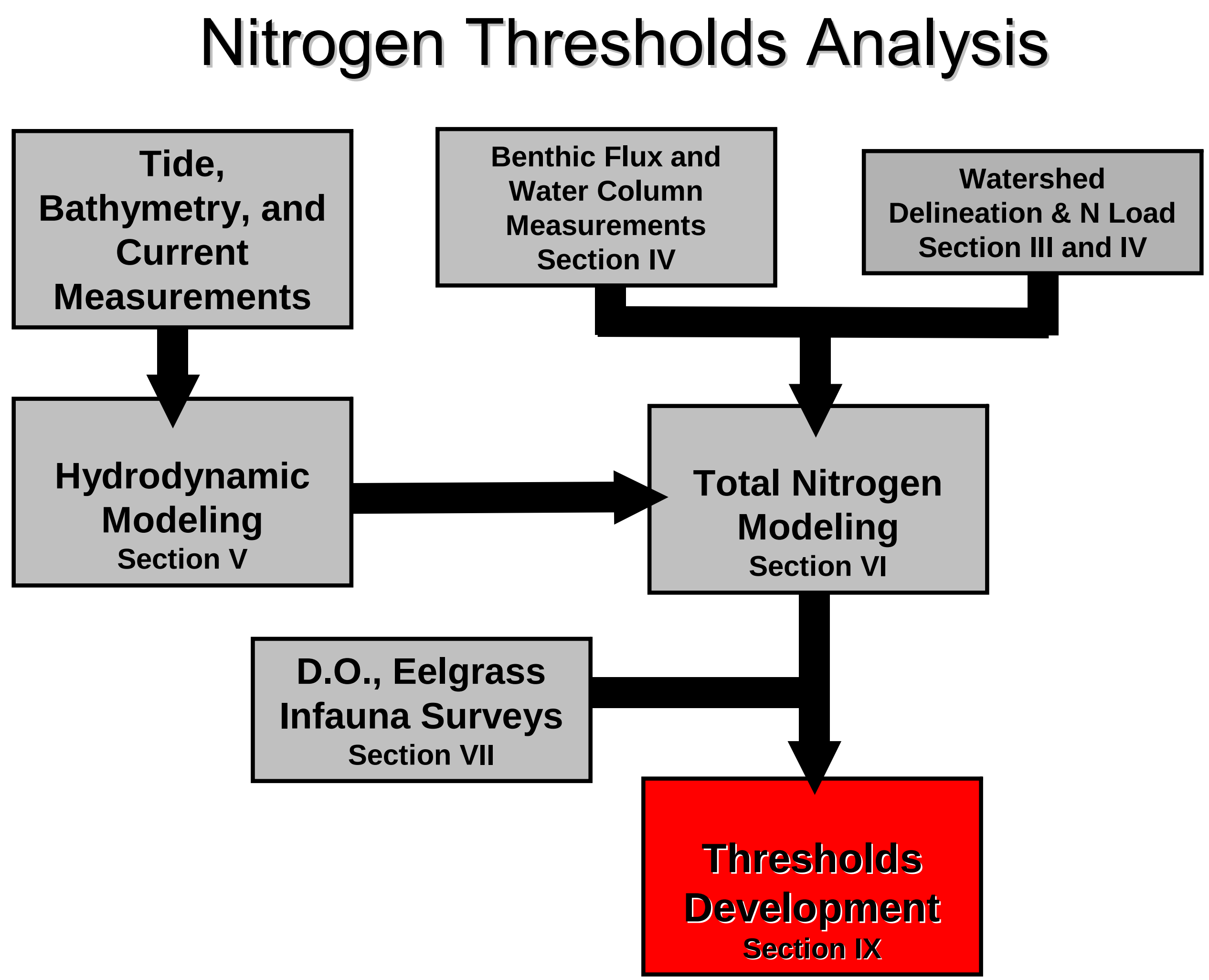


Figure I-3. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach

Tidal damping (reduction in tidal amplitude) through an embayment can range from negligible, indicating "well-flushed" conditions, or show tidal attenuation caused by constricted channels and marsh plains, indicating a "restrictive" system, where tidal flow and the associated flushing are inhibited. Tidal data indicate only minimal tidal damping through the inlet into the East Bay – Centerville River. It appears that the tidal inlet is operating efficiently having recently been dredged in the 2002 to 2003 time frame during the most recent Centerville River Dredging Project. Within the Bumps River and Centerville Marshes portion of the System, the tide propagates to the sub-embayments with negligible attenuation, consistent with generally well-flushed conditions throughout.

I.3 NUTRIENT LOADING

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

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

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

Protection and restoration of coastal embayments from nitrogen overloading has resulted in a focus on determining the assimilative capacity of these aquatic systems for nitrogen. While this effort is ongoing (e.g. USEPA TMDL studies), southeastern Massachusetts has been the site of intensive efforts in this area (Eichner et al., 1998, Costa et al., 1992 and in press, Ramsey et al., 1995, Howes and Taylor, 1990, and the Falmouth Coastal Pond Overlay District 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 Centerville River System monitored by the Town of Barnstable/Three Bays Preservation. Data from the Water Quality Monitoring Program combined with site-specific habitat quality data (D.O., eelgrass, phytoplankton blooms, benthic animals) was utilized to “tune” general nitrogen thresholds typically used by the Cape Cod Commission, Buzzards Bay Project, and Massachusetts State Regulatory Agencies.

Unfortunately, almost all of the estuarine reaches within the Centerville River System are near or beyond their ability to assimilate additional nutrients without impacting their ecological health. Nitrogen levels are elevated throughout the System and eelgrass beds have not been observed within the Centerville system for over a decade. Nitrogen related habitat impairment within the Centerville River Estuary shows a gradient of high to low moving from the inland reaches to the tidal inlet. The result is that nitrogen management of the primary sub-embayments to the Centerville River system is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed “eutrophication” and in certain instances can occur naturally over long periods of time. When the nutrient loading is rapid and primarily from human activities leading to changes in a coastal watershed, nutrient enrichment of coastal waters is termed “cultural eutrophication”. Although the influence of human-induced changes has increased nitrogen loading to the systems and contributed to the degradation in ecological health, it is sometimes possible that eutrophication within the Centerville River sub-embayments (e.g. Scudder Bay) could potentially occur without human influence and must be considered in the nutrient threshold analysis. While this finding would not change the need for restoration, it would change the approach and potential targets for management. As part of future restoration efforts, it is important to understand that it may not be possible to turn each embayment into a “pristine” system.

I.4 WATER QUALITY MODELING

Evaluation of upland nitrogen loading provides important “boundary conditions” (e.g. watershed derived and offshore nutrient inputs) for water quality modeling of the Centerville System; however, a thorough understanding of estuarine circulation is required to accurately determine nitrogen concentrations within each component of 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 Centerville River System, including the tributary sub-embayments of East Bay, Centerville

River, Bumps River, Scudder Bay and the Centerville River Marshes. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for each of the systems. Once the hydrodynamic properties of each estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates.

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape model for sub-watershed areas designated by MEP. Almost all watershed sourced nitrogen entering the Centerville River System is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Nantucket Sound source waters and throughout the Centerville River system were provided by the Town of Barnstable/Three Bays Water Quality Monitoring Program (a coordinated effort between the Town of Barnstable, Three Bays Preservation and the Coastal Systems Program at SMAST). Measurements of the salinity and nitrogen distributions throughout estuarine waters of the Centerville River System (2000-2005) were used to calibrate and validate the water quality model (under existing loading conditions).

I.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Centerville River System for the Town of Barnstable. A review of existing water quality studies is provided (Section II). The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Sections III and IV. In addition, nitrogen input parameters to the water quality model are described. Since benthic flux of nitrogen from bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Section IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Cape Cod Commission data and offshore water column nitrogen values were derived from an analysis of monitoring stations in Nantucket Sound (Section IV). Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data (conducted by municipalities) as discussed in Section IV. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling, as well as an analysis of how the measured nitrogen levels correlate to observed estuarine water quality are described in Section VI. This analysis includes modeling of current conditions, conditions at watershed build-out, and with removal of anthropogenic nitrogen sources. In addition, an ecological assessment of the component sub-embayments was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration/protection of the River in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined system threshold for restoration or protection. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for this system. Finally, analyses of the Centerville River System were undertaken relative to potential alterations of circulation and flushing, including an analysis to identify hydrodynamic restrictions and an examination of dredging options to improve nitrogen related water quality. The results of the nitrogen modeling for each scenario have been presented in Section IX.

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 shellfisherman and to the sport-fishery and offshore finfishery, 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 Centerville River System, the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if nitrogen addition is controlled, then eutrophication is controlled. This approach has been formalized through the development of tools for predicting nitrogen loads from watersheds and the concentrations of water column nitrogen that may result. Additional development of the 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 Centerville River System. 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 (mostly in the context of dredging) and habitat health have been conducted within the Centerville River System over the past 10 years. The most directly applicable study relative to the objectives of the MEP focused on nitrogen fluxes and mitigation strategies in the Audubon Skunknett River Wildlife Sanctuary (Hamersley, 2004). The study is significant relative to elucidating the potential for natural attenuation within the Centerville River watershed as well as being an independent confirmation of the MEP stream gaging effort on the Skunknett River. The study was undertaken by an SMAST-Coastal Systems Program scientist directly involved in researching biogeochemical aspects of nutrient cycling to the direct benefit of the MEP analytical approach utilized in this analysis of the Centerville River System. As described in the Skunknett River Wildlife

Sanctuary Report, increasing development in the watershed of Scudder Bay (tributary sub-embayment to Centerville River) has led to algal blooms and eutrophication of the estuarine receiving water. Two point sources for nutrients entering the bay are the Skunknett and Bumps Rivers. The Skunknett River flows through the Audubon Skunknett River Wildlife Sanctuary (ASRWS), the site of four former or existing ponds. The former ponds were created in the 19th century by dams which washed out in the early 1990's. Restoring these ponds has the potential to support the removal of nitrogen from the Skunknett River via the natural bioremediation processes of denitrification and storage. The removal by denitrification is primarily dependent on contact with organic-rich sediments. Restoring the ponds would increase the contact time of the sediments, as well as their organic content through deposition. Hydrological and nutrient fluxes into and out of the Sanctuary were measured in the summer of 2002 and through most of the year 2003 in order to perform a preliminary evaluation of the feasibility of restoring one or more of the ponds to enhance the removal of nitrogen from the Skunknett River.

The preliminary analysis of the Skunknett River aquatic system (inclusive of the associated pond system) suggested that the Skunknett River represents a significant source of nitrogen to Scudder Bay, and that much of this nitrogen is in the form of nitrate, making it amenable to removal through denitrification. As measured at the time of the study (2002-2003) the Skunknett River generated a total nitrogen load to Scudder Bay of 8,490 kg N/yr. Housing development in the Skunknett River watershed will likely increase that N load over time. Well monitoring by the town in the watershed is showing very high and increasing levels of nitrate (up to 6.2 mg L^{-1}, Craig Crocker, Centerville-Osterville-Marstons Mills Water Superintendent, personal communication) indicating the potential for increased N flows in the future. The Skunknett River currently flows through the ASRWS in 2.2 hours, permitting little time for natural N uptake and removal processes which occur during contact of water with sediments. The rapid flow scours stream sediments of organic material, further decreasing N removal. The present residence time of the river (as determined in the 2004 Hamersley study) in the Sanctuary is too low to allow significant removal. Restoring the former ponds and raising the level of Mill Pond could increase the residence time of the Skunknett River by eleven-fold. Summer total nitrogen removals under these conditions might approach 3,400 kg N y^{-1}. Summer reductions of TN flows into Scudder Bay resulting from pond restoration could be the equivalent of the N output of 565 houses. Although our preliminary study demonstrates the potential feasibility of hydrological manipulations in the ASRWS in mitigating N flows to Scudder Bay, any such manipulations would be require further topographical, wildlife, land use, and water quality data, as well as agreements with the landowner, Massachusetts Audubon.

In addition to the study on nitrogen fluxes and mitigation strategies in the Audubon Skunknett River Wildlife Sanctuary (Hamersley, 2004) up-gradient of the Centerville River System, a Draft Environmental Impact Report (DEIR) was developed by the Woods Hole Group in December 2003. The DEIR was developed in regards to a planned dredging of the Centerville River portion of the System as well as associated Craigville Beach nourishment activities. The DEIR focused on the Town of Barnstable developed two phased project approach to conduct maintenance and improvement dredging in the Centerville River/East Bay estuarine system. The proposed dredge project was initiated in September 2000 with the submittal of an Environmental Notification Form (ENF) to the Massachusetts Environmental Policy Act (MEPA) Unit. The ENF specified a two phased approach with a request for a waiver from the requirements of an EIR for Phase I of the project. The Phase I record of decision waiving the EIR requirements for Phase I of the project was issued in April of 2001.

Phase I of the Centerville River dredge and beach nourishment project including dredging material from the Centerville River west of the confluence of the estuarine reach of the Bumps

River and Centerville River. The Phase I dredging would extend into East Bay including removal of material from the East Bay inlet channel adjacent Dowes Beach. Dredge planning, engineering and permitting was completed by October of 2002 and dredging by the Barnstable County dredge commenced that fall. During the initial part of the Phase I dredge more than 20,000 cubic yards of sand was removed from the Centerville River leading to East Bay and that sand was subsequently used to nourish the Long Beach barrier separating Centerville River from Centerville Harbor. An additional 10,700 cubic yards of silty material was planned for removal from East Bay in the fall/winter of 2003 and subsequent dewatering and disposal.

Phase II of the dredge project involved additional maintenance and improvement dredging of the Centerville River east of the confluence of the Bumps River and Centerville River and extended into the most upgradient salt marsh areas (Centerville Marshes). Phase II of the overall project was the basis of the DEIR as required by the Secretary's Certificate on the ENF issued in March of 2001. The Phase II portion of the project was also required to be reviewed as a District of Regional Impact as specified under the Cape Cod Commission Act, Section 12(i). Under Phase II, an estimated 31,180 cubic yards of sediment would be removed from the eastern end of the Centerville River. Analyses of sediment disposal alternatives as well as environmental impact studies were conducted under the DEIR to elucidate preferred design and construction approaches for the dredge project. Sandy sediments would be used for beach nourishment (Long Beach and Craigville Beach). Silt/sand sediments would be dewatered in basins constructed on Craigville Beach and the sandy fraction of the dewatered sediments would be used for nourishment of Craigville Beach while silty sediments would be transported to a waste recycling facility in Sandwich, MA. The strictly silty sediments removed from the Centerville Marshes area of the system would be transported to the waste recycling facility for reuse.

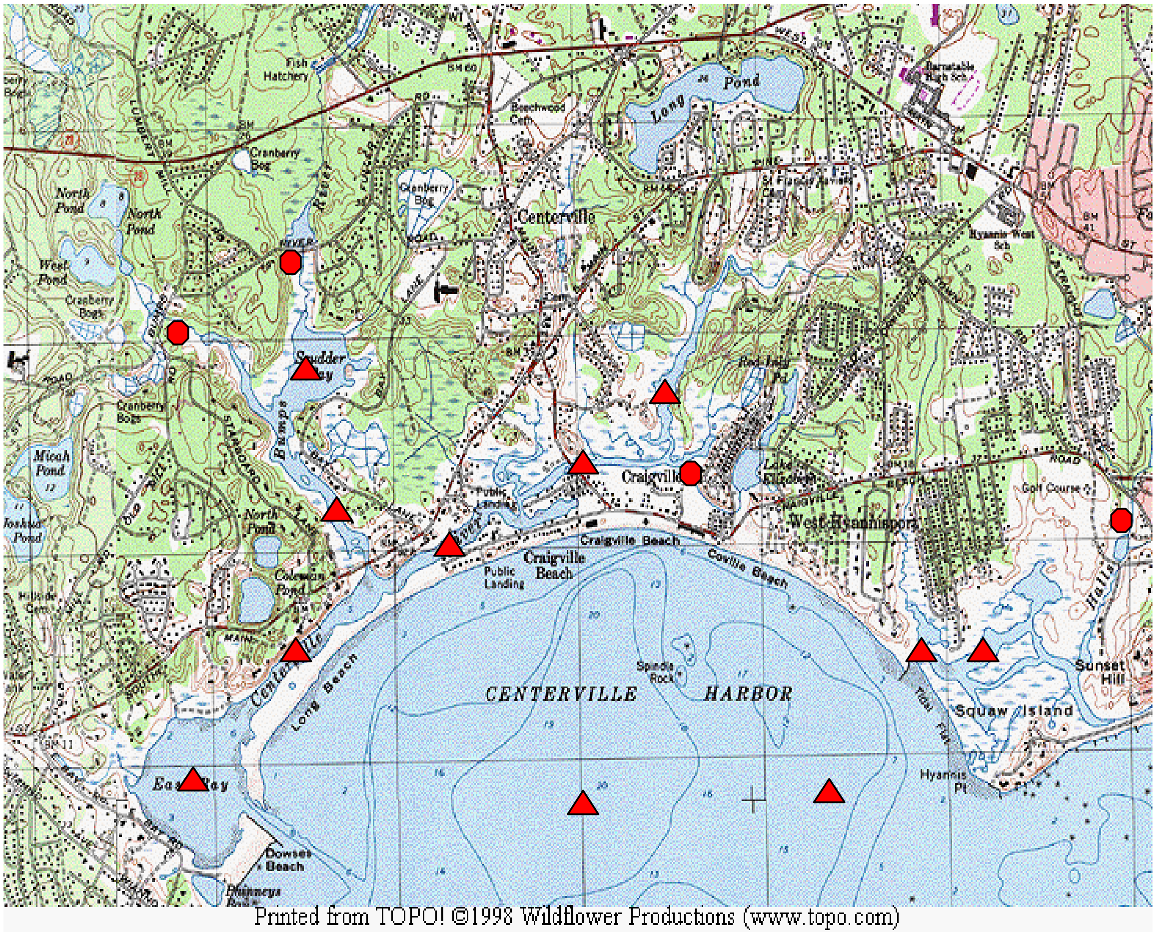
As part of the DEIR, a summary of the existing environment was conducted combining a variety of data findings from studies carried out prior to initiation of the dredge project. Based on the summary provided in the DEIR it is apparent that the bathymetry of the Centerville River system had been characterized on 5 separate occasions between 1930 and 1969. Nearly twenty five years elapsed before the next bathymetric survey was conducted in 1996 as a condition survey for development of a hydrodynamic model of the Centerville River system (ACI, 1996). The 1996 hydro model was developed to examine tidal flushing characteristics within East Bay and Centerville River estuary with specific attention being given to the effects of proposed dredging on existing and future conditions. The MEP Technical Team captured the most recent elements of the historical record on bathymetry of the Centerville River system as well as details of the dredging activity in the system as has occurred in the past several years such that this knowledge could be incorporated into the development of the MEP hydrodynamic model. Moreover, details of the ACI, 1996 hydrodynamic model have also been captured by the MEP Technical Team in order to leverage any pertinent information generated under that effort which may be useful in the MEP analysis of the Centerville River System.

In addition to the summary of bathymetric data collection and description of the results of the 1996 hydrodynamic model runs for the Centerville River system, the DEIR also presents a discussion of the water quality characteristics of the estuarine system. It suffices to say that a large part of the discussion on water quality in the Centerville River system is based on the water quality monitoring program developed by the SMAST-Coastal Systems Program for the Town of Barnstable and in support of the Massachusetts Estuaries Project. An ancillary sampling program undertaken specifically in support of Phase II of the dredging project was developed and described in the DEIR. This dredge related sampling program focused primarily on nutrients within sediments. The sediment nutrient sampling program was developed and executed by Dr. Dror Angel from the Department of Civil Engineering located at the Massachusetts Institute of Technology (MIT). The purpose of the sediment sampling program was to quantify the nutrient content of bottom sediments in the Centerville River system and to evaluate the extent to which nutrients could be released during resuspension of these sediments during a proposed dredge event. While this information is useful to the purpose that it was intended (i.e. dredging impacts), it does not relate to nutrient release associated with the undisturbed sediments of an estuarine basin.

While the DEIR covers a wide variety of other environmental factors that are of general interest, it does include specific information on shellfish and other invertebrates present in the system that was of interest to the MEP given the MEP's data collection on benthic infauna as a biological indicator of habitat health. A shellfish and benthic survey was developed in support of the DEIR and the proposed Phase II dredge project. The shellfish and benthic survey was conducted during August and September of 2003. A series of survey transects were established at 300 foot intervals across the Centerville River system west to east starting at the confluence with the Bumps River and extending up to the Centerville Marshes area. Shellfish and benthic data was collected at a total of 21 transects with three benthic sampling stations located along each transect. Sampling was conducted with rake hauls and clam rakes. As summarized in the DEIR, sections of Centerville River show signs of well oxygenated sediments that contain abundant benthic life including shellfish, *Mercenaria* and other species. Based on the DEIR benthic survey, other areas of the Centerville River system show signs of reduced flushing, anoxic sediments (black and sulphidic) and an embayment bottom with reduced and, in some instances, depopulated of benthic infauna. This information has been captured and considered by the MEP for use as appropriate.

The common focus of the Town of Barnstable Water Quality Monitoring Program effort has been to gather site-specific data on the current nitrogen related water quality throughout all the embayments of the Town (including the Centerville River System) to support evaluations of observed water quality and habitat health. This multi-year effort was initiated in 2001 for the Centerville River, with support from the Town of Barnstable and technical assistance from Three Bays Preservation and the Coastal Systems Programs at SMAST-UMD. The Barnstable Water Quality Monitoring Program in Centerville River developed a data set at sampling stations (Figure II-1) that elucidated the long-term water quality of the river system. Additionally, as remediation plans for this and other various systems are implemented throughout the Town of Barnstable, the continued monitoring is planned to provide quantitative information to the Town relative to the efficacy of remediation efforts. The MEP effort builds upon the water quality monitoring program, previous hydrodynamic evaluations conducted during the development of the EIR developed for the Centerville River Dredging Project and water quality analyses conducted by SMAST. Additionally, the MEP approach includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Centerville River System.

The Town of Barnstable Water Quality Monitoring Program provided the quantitative water column nitrogen data (2001-2005) 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 and the embayment water quality and eelgrass surveys. This information is integrated with MEP higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Centerville River Estuarine System. The MEP has incorporated all appropriate data from all previous studies to enhance the determination of nitrogen thresholds for the Centerville River System and to reduce costs to the Town of Barnstable.



Estuarine WQ Stations   Stream WQ Stations

Figure II-1. Town of Barnstable Water Quality Monitoring Program. Estuarine water quality monitoring stations sampled by the Town and volunteers. Stream water quality stations sampled weekly by the MEP. Halls Creek along the eastern shore of Centerville Harbor will be assessed in a future MEP Technical Report on the Lewis Bay System.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Massachusetts Estuaries Project team includes technical staff from the United States Geological Survey (USGS). These USGS groundwater modelers were central to the development of the groundwater modeling approach used by the Estuaries Project. The USGS has a long history of developing regional models for the six-groundwater flow cells on Cape Cod. Through the years, advances in computing, lithologic information from well installations, water level monitoring, stream flow measurements, and reconstruction of glacial history have allowed the USGS to update and refine the groundwater models. The MODFLOW and MODPATH models utilized by the USGS organize and analyze the available data using up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation, 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 Centerville River/East Bay embayment system located in Barnstable, Massachusetts. The Centerville River/East Bay watershed is situated along the southern edge of Cape Cod and is bounded by Vineyard/Nantucket Sound.

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watershed or contributing area to the Centerville River/East Bay embayment system under evaluation by the Project Team. The Centerville River/East Bay estuarine system is a moderately complex estuary and includes significant wetland dominated eastern portion. Further watershed modeling was undertaken to sub-divide the overall watershed to the Centerville River/East Bay system into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within 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 Sagamore flow cell on Cape Cod. Model assumptions for calibration were matched to surface water inputs and flows from MEP stream flow measurements (2003 to 2004).

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 a stream and the portion of the groundwater system that discharge directly into an estuary as groundwater seepage.

III.2 MODEL DESCRIPTION

Contributing areas to the Centerville River/East Bay system were delineated using a regional model of the Sagamore Lens flow cell (Walter and Whealan, 2004). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al.*, 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program

MODPATH4 (Pollock, 1994), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to Centerville River/East Bay system including a subwatershed to Scudder Bay/Bumps River and also to determine portions of recharged water that may flow through fresh water ponds and streams prior to discharging into coastal water bodies.

The Sagamore Flow Model grid consists of 246 rows, 365 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft. The top of layer 8 resides at NGVD 29 with layers 1-7 stacked above and layers 8-20 below. 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 519 feet below NGVD 29); since bedrock is 300 to 400 feet below NGVD 29 in the Centerville River/East Bay 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, was used to simulate the top of the water table, which varies in elevation depending on the location in the Lens.

The glacial sediments that comprise the aquifer of the Sagamore Lens consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward 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 Centerville River/East Bay watershed (including Scudder Bay/Bumps River) is generally split between the Mashpee Outwash Plain Deposits to the west and Barnstable Plain Deposits to the east; modeling and field measurements of contaminant transport at the MMR has 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 coastal 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 Towns and streamflow data collected in 1989-1990 as well as 2003.

The model simulates steady state, or long-term average, hydrologic conditions including a long-term average recharge rate of 27.25 inches/year and the pumping of public-supply wells at average annual withdrawal rates for the period 1995-2000 with a 15% consumptive loss. This recharge rate is based on the most recent USGS information. Large withdrawals of groundwater from pumping wells may have a significant influence on water tables and watershed boundaries and therefore the flow and distribution of nitrogen within the aquifer. After accounting for the consumptive loss and measured discharge at municipal treatment facilities, water withdrawn from the modeled aquifer by public drinking water supply wells is evenly returned within designated residential areas utilizing on-site septic systems. Since no municipal wastewater treatment facilities discharge within the Centerville River/East Bay watershed, modeled return flow is returned to the groundwater in developed areas as septic system recharge.

III.3 CENTERVILLE RIVER/EAST BAY CONTRIBUTORY AREAS

Newly revised watershed and sub-watershed boundaries were determined by the United States Geological Survey (USGS) for the Centerville River/East Bay embayment system, including Scudder Bay/Bumps River sub-embayment and Lake Wequaquet (Figure III-1). Model outputs of MEP watershed boundaries were "smoothed" to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, (c) to include water table data in the lower regions of the watersheds near the coast (as available), and (d) to more closely match the sub-embayment segmentation of the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. The MEP sub-watershed delineation includes 10 yr time of travel boundaries. Overall, twenty-two (22) sub-watershed areas, including nine freshwater ponds, were delineated within the watershed to the Centerville River/East Bay embayment system.

Table III-1 provides the daily discharge volumes for various sub-watersheds as calculated from the groundwater model; these volumes were used to assist in the salinity calibration of the tidal hydrodynamic models and to determine hydrologic turnover in the lakes/ponds, as well as for comparison to measured surface water discharges. The overall estimated groundwater flow into Centerville River/East Bay from the MEP delineated watershed is 54,416 m3/d.

The delineations completed for the MEP project are the second watershed delineation completed in recent years for portions of the Centerville River/East Bay estuary. Figure III-2 compares the delineation completed under the current effort with the Centerville River/East Bay delineation completed by the Cape Cod Commission in 1998 as part of the Coastal Embayment Project (Eichner, *et al*., 1998). The delineation completed in 1998 was defined based on regional water table measurements collected from available wells over a number of years and normalized to average conditions; delineations based on this previous effort were incorporated into the Commission's regulations through the Regional Policy Plan (CCC, 1996 & 2001).

The MEP watershed area for the Centerville River/East Bay system as a whole is 11% smaller (913 acres) than the 1995 CCC delineation. The differences are largely attributable to a more southern location for the Cape Cod Bay/Vineyard Sound groundwater divide and a more eastern location for the western boundary of the system watershed in the MEP watershed. The change in the western boundary is generally due to the refinements of the Three Bays watershed documented in that system's MEP Technical Report (Howes, *et al*., 2005). It should also be noted that the Three Bays watersheds to Micah and Joshua's ponds were corrected in the Centerville River/East Bay analysis in order to better account for the measured streamflow in Skunknet River (see section IV.2.2). Subwatersheds to individual freshwater ponds were not delineated in the CCC watershed.

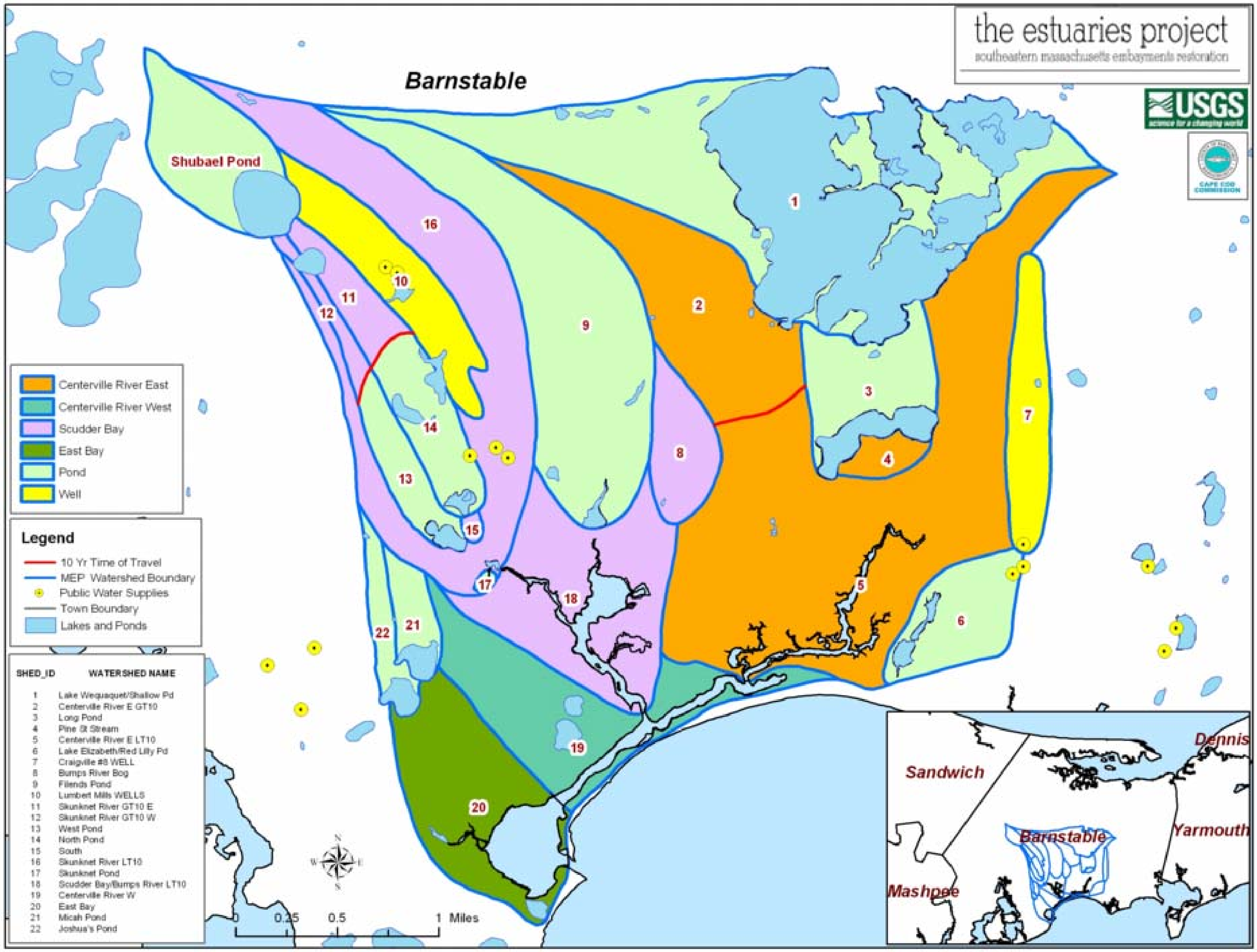


Figure III-1. Watershed and sub-watershed delineations for the Centerville River/East Bay estuary system. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the watershed names (above). Sub-watersheds to embayments were selected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

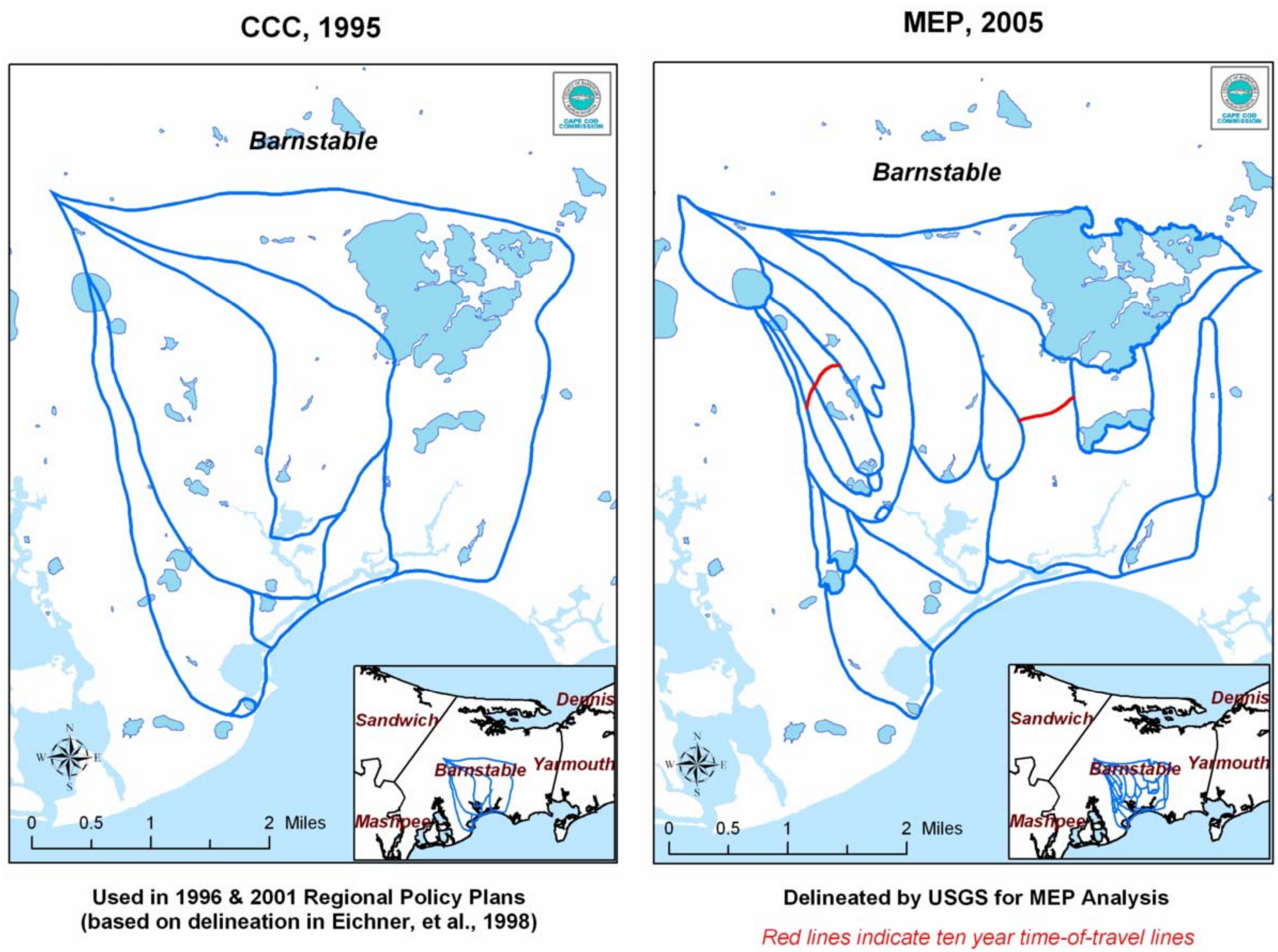


Figure III-2. Comparison of 1998 Cape Cod Commission and current Centerville River/East Bay watershed and subwatershed delineations.

Table III-1. Daily groundwater discharge from each of the sub-watersheds to the Centerville River/East Bay Estuary, as determined from the USGS groundwater model.			
Watershed	**Watershed #**	**Discharge**	
		m^3/day	**ft^3/day**
Lake Wequaquet/Shallow Pond	1	10,637	375,647
Centerville River E GT10	2	3,202	113,086
Long Pond	3	1,804	63,715
Pine Street Stream	4	345	12,174
Centerville River E LT10	5	9,615	339,559
Lake Elizabeth/Red Lilly Pond	6	1,358	47,946
Craigville #8 WELL	7	638	22,514
Bumps River Bog	8	1,006	35,514
Filends Pond	9	5,217	184,254
Lumbert Mills WELLS	10	1,769	62,472
Skunknet River GT10 E	11	695	24,532
Skunknet River GT10 W	12	298	10,523
West Pond	13	794	28,053
North Pond	14	1,160	40,979
South	15	61	2,150
Skunknet River LT10	16	4,894	172,833
Skunknet Pond	17	40	1,405
Scudder Bay/Bumps River LT10	18	3,147	111,121
Centerville River W	19	2,752	97,185
East Bay	20	2,430	85,805
Micah Pond	21	483	17,041
Joshua's Pond	22	430	15,194
Shubael Pond		1,642	57,997
TOTAL		54,416	1,921,697
NOTE: Discharge rates are based on 27.25 inches per year of recharge (Walter and Whealan, 2005); discharge flows from Shubael, Micah, and Joshua's Ponds and Craigville #8 WELL are adjusted to account for flow out of the system watershed.			

The evolution of the Centerville River/East Bay watershed has allowed increasing accuracy as each new version adds new hydrologic data to that previously collected; the model allows all this data to be organized and to be brought into congruence with adjacent watersheds. The evaluation of older data and incorporation of new data during the development of the model is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model used 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 Centerville River/East Bay system (Section V.1).

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

The MEP Technical Team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP Technical Team members, CCC staff developed nitrogen-loading rates (Section IV.1) to the Centerville River/East Bay embayment system. The Centerville River/East Bay watershed was sub-divided to define contributing areas to each of the major inland freshwater systems and to each major sub-embayment to Centerville River/East Bay. Further sub-divisions were made to identify watershed areas where a nitrogen discharge reaches embayment waters in less than 10 years or greater than 10 years. A total of 22 sub-watersheds were delineated for the Centerville River/East Bay Estuarine System. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds and each embayment (see Section III).

The initial task in the MEP land use analysis is to gage whether or not nitrogen discharges to the watershed have reached the embayment. 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 collections points, such as streams and ponds. After reviewing the percentage of nitrogen loading in the less than 10 year time of travel (LT10) and greater than 10 year time of travel (GT10) watersheds (Table IV-1), land use development records, and water quality modeling, it was determined that Centerville River/East Bay is currently in balance with its watershed load. The bulk (86%) of the watershed nitrogen load is within 10 years flow to Centerville River/East Bay and its sub-estuaries. The overall result of the timing of development relative to groundwater travel times is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuaries (after accounting for natural attenuation, see below).

Table IV-1. Percentage of unattenuated nitrogen loads in less than 10-year time of travel subwatersheds to Centerville River/East Bay.

WATERSHED	LT10	GT10	TOTAL	%LT10
Name	kg/yr	kg/yr	kg/yr	
Centerville River E	24,744	6,164	30,908	80%
Scudder Bay	21,399	2,499	23,898	90%
Centerville River W	3,454		3,454	100%
East Bay	3,486		3,486	100%
TOTAL	53,082	8,663	61,745	86%

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 studies is applied to other portions. The Linked Watershed-Embayment Management Model (Howes, Ramsey, Kelley, 2001) uses a land-use Nitrogen Loading Sub-Model based upon subwatershed-specific land uses and pre-determined nitrogen loading rates. For the Centerville River/East Bay embayment system, the model used Town of Barnstable land-use data transformed to nitrogen loads using both regional nitrogen loading factors and local watershed specific data (such as parcel by parcel water use or groundwater monitoring wells). Determination of the nitrogen loads required obtaining watershed specific information regarding wastewater, fertilizers, runoff from impervious surfaces and atmospheric deposition. The primary regional factors were derived for southeastern Massachusetts from direct measurements. The resulting nitrogen loads represent the "potential" or unattenuated nitrogen load to each receiving 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 Centerville River/East Bay System watershed was determined based upon a site-specific study of streamflow from the Skunknett River, Bumps River Bog and flow coming from the Lake Elizabeth/Red Lilly Pond system and the Lake Wequaquet/Long Pond system. Subwatersheds to these various portions allowed comparisons between field collected data from the streams and ponds and estimates from the nitrogen-loading sub-model. Attenuation through the ponds were conservatively assumed to equal 50% based on available monitoring of selected Cape Cod lakes; calculations for individual ponds were also determined. Streamflow 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 affect groundwater flow patterns (area and depth) is a standard part of the data collection effort of the MEP. In the present effort, measurements were made of attenuation in Lake Wequaquet, Long Pond, Shubael Pond, Micah Pond, Joshua's Pond and in the stream complexes to Skunknett River, Bumps River Bog and stream flow coming from the Lake Elizabeth/Red Lilly Pond system and the Lake Wequaquet/Long Pond system. However, if smaller aquatic features that have not been included in this MEP analysis are providing additional attenuation of nitrogen, nitrogen loading to the estuary would only be slightly (~10%) overestimated given the distribution of nitrogen sources within the watershed. Based upon these considerations, the MEP Technical Team used the Nitrogen Loading Sub-Model estimate of nitrogen loading for the seven sub-watersheds that directly discharge groundwater to the estuary without flowing through one of the interim measuring points. Internal nitrogen recycling was also determined throughout the tidal reaches of the Centerville River/East Bay 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

Estuaries Project staff obtained digital parcel and tax assessors data from the Town of Barnstable Geographic Information Systems Department. Digital parcels and land use/assessors data are from 2004. These land use databases contain traditional information regarding land use classifications (MADOR, 2002) plus additional information developed by the town. The parcel data and assessors' databases were combined for the MEP analysis by using the Cape Cod Commission Geographic Information System (GIS).

Figure IV-1 shows the land uses within the Centerville River/East Bay Estuary watershed area. Land uses in the study area are grouped into seven land use categories: 1) residential, 2) commercial, 3) industrial, 4) undeveloped, 5) agricultural, 6) public service/government, including road rights-of-way, and 7) 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 (MADOR, 2002). These categories are common to each town in the watershed. “Public service” in the MADOR system is tax-exempt properties, including lands owned by government (e.g., wellfields, schools, golf courses, open space, roads) and private groups like churches and colleges.

In the overall Centerville River/East Bay System watershed, the predominant land use based on area is residential, which accounts for 55% of the overall watershed area and 88% of the parcels in the system watershed (Figure IV-2). Single-family residences (MADOR land use code 101) are 94% of the residential parcels and single-family residences are 91% of the residential land area. In the individual subwatersheds, residential land uses vary between 51 and 64% of the subwatershed areas. After residential land use, public service is usually the second highest percentage land use, except in the Centerville River East subwatershed, where Lake Wequaquet and Long Pond push the freshwater area percentage higher than public service. Undeveloped parcels are the second highest parcel count after residential with 5 to 14% of the parcel counts in the subwatersheds. Overall, undeveloped land uses account for 7% of the entire Centerville River/East Bay watershed, while commercial properties account for approximately 1% of the watershed area.

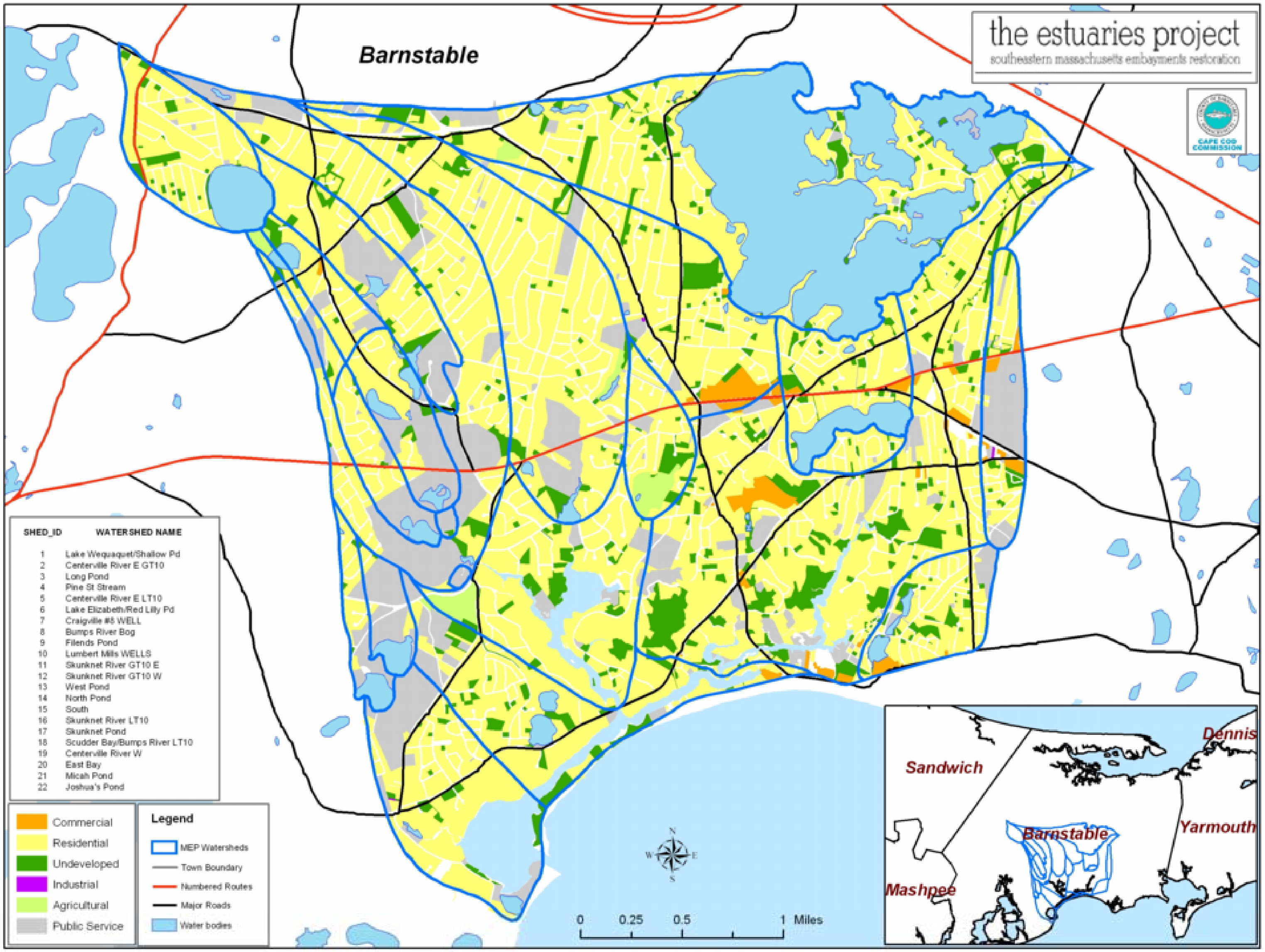


Figure IV-1. Land-use in the Centerville River/East Bay watershed. The watershed is completely contained within the Town of Barnstable. Land use classifications are based on assessors' records provided by the town.

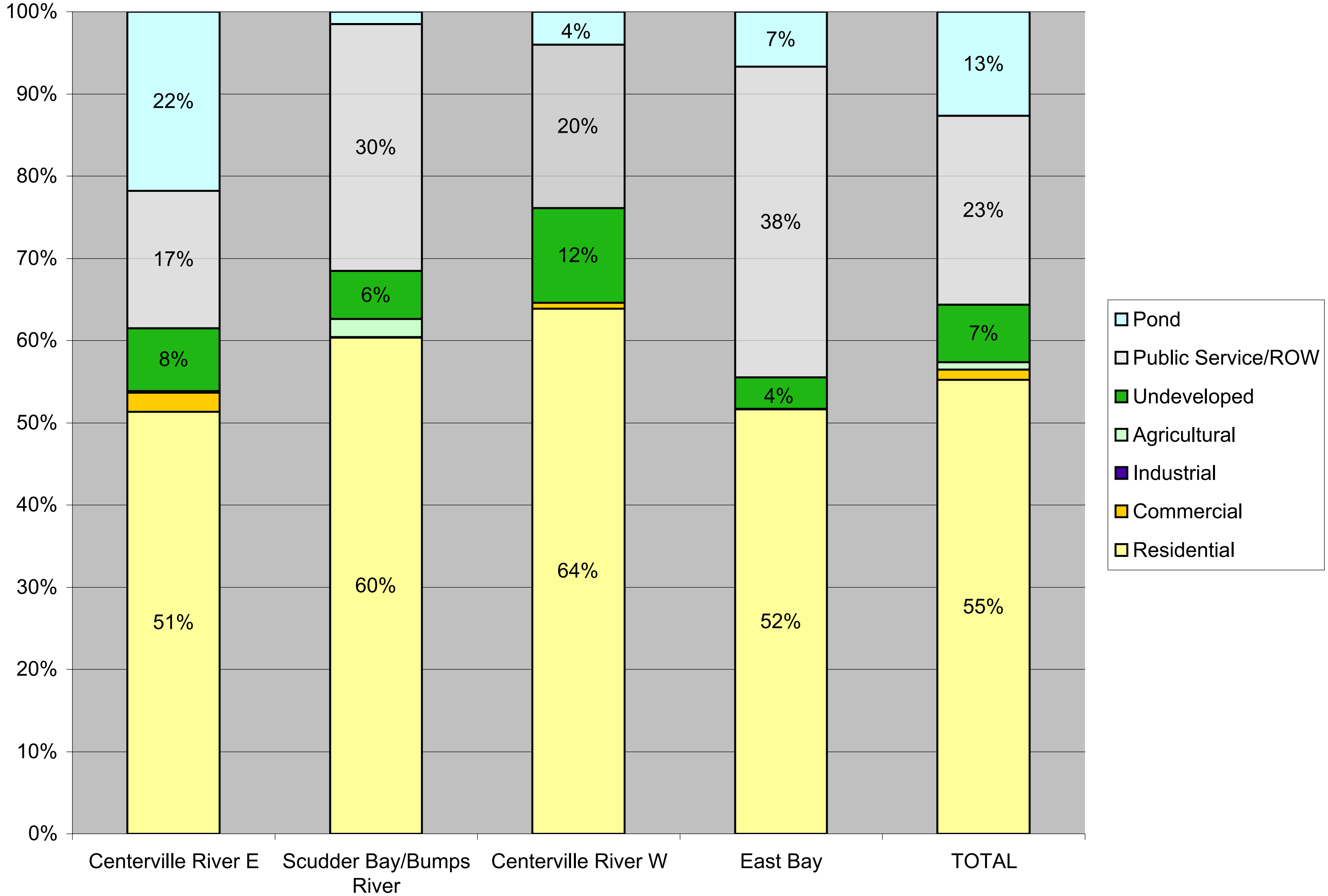


Figure IV-2. Distribution of land-uses within the major subwatersheds and whole watershed to Centerville River/East Bay. Only percentages greater than or equal to 4% are shown.

In order to estimate wastewater flows within the Centerville River/East Bay study area, the Cape Cod Commission obtained parcel-by-parcel water use information from the Town of Barnstable. The water use data includes information from the Centerville, Osterville, Marstons Mills (COMM) Water District, as well as some data from the Town of Barnstable Water Supply Division (WSD), which supplies water to properties on the easternmost edge of the watershed. The information from COMM contains water use from 2001 through 2005, MEP wastewater loads were determined by averages of the number of annual volume at each parcel. The WSD parcels presented an additional challenge; only 1993 water use is available to the Town for these parcels (personal communication, Jim Benoit, Town of Barnstable GIS Division). The town is currently working with contractors to the previous owners of the water supply system to obtain previous years' water use (personal communication, Hans Keijser, Town of Barnstable Water Supply Division). Select parcels within this service area are also connected to the municipal sewer system and the Hyannis WWTF; these parcels have zero wastewater nitrogen loads in the watershed nitrogen loading analysis. MEP staff linked water use information to the parcel and assessors data using GIS techniques. There are no municipal WWTFs in the Centerville River/East Bay watershed, but there are nine innovative/alternative septic systems (personal communication, Sue Rask, Barnstable County Department of Health and the Environment). Wastewater-based nitrogen loading from the individual parcels using on-site septic systems is based upon the measured water-use, nitrogen concentration, and consumptive loss of water before the remainder is treated in a septic system (see Section IV.1.2).

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 2001, Costa et al. 2002). Variation in per capita nitrogen load has been found to be relatively small, with average annual per capita nitrogen loads generally between 1.9 to 2.3 kg person-yr^{-1}.

However, given the seasonal shifts in occupancy and rapid population growth throughout southeastern Massachusetts, decennial census data yields accurate estimates of total population only in selected watersheds. To correct for this uncertainty and more accurately assess current nitrogen loads, the MEP employs a water-use approach. The water-use approach is applied on a parcel-by-parcel basis within a watershed, where annual water meter data is linked to 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 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 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 the 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-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). Modeled and measured nitrogen loads were determined for a small sub-watershed to Mashapaquit Creek in West Falmouth Harbor (Smith and Howes, manuscript in review) where measured nitrogen discharge from the aquifer was within 5% of the modeled N load. Another evaluation was conducted by surveying nitrogen discharge to the Mashpee River in reaches with swept sand channels and in winter when nitrogen attenuation is minimal. The modeled and observed loads showed a difference of less than 8%, easily attributable to the low rate of attenuation expected at that time of year in this type of ecological situation (Samimy and Howes, unpublished data).

While census based population data has limitations in the highly seasonal MEP region, part of the regular MEP analysis is to compare expected water used based on average residential occupancy to measured average water uses. This is performed as a quality assurance check to increase certainty in the final results. This comparison has shown that the larger the watershed the better the match between average water use and occupancy. For example, in the cases of the combined Great Pond, Green Pond and Bournes Pond watershed in the Town of Falmouth and the Popponesset Bay/Eastern Waquoit Bay watershed, which covers large areas and have significant year-round populations, the septic nitrogen loading based upon the census data is within 5% of that from the water use approach. This comparison matches some of the variability seen in census data itself. Census blocks, which are generally smaller areas of any given town, have shown up to a 13% difference in average occupancy 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 analyses and a variety of other MEP embayments. While the MEP septic system nitrogen load is the best estimate possible, to the extent that it may underestimate the nitrogen load from this source reaching receiving waters provides a safety factor relative to other higher loads that are generally used in regulatory situations. The lower concentration results in slightly higher amounts of nitrogen mitigation (estimated at 1% to 5%)) needed to lower embayment nitrogen levels to a nitrogen target (e.g. nitrogen threshold, cf. Section VIII). The additional nitrogen removal is not proportional to the septic system nitrogen level, but is related to the how the septic system nitrogen mass compares to the nitrogen loads from all other sources that reach the estuary (i.e. attenuated loads).

In order to provide an independent validation of the average residential water use within the Centerville River/East Bay System watershed, MEP staff reviewed US Census population values for the Town of Barnstable. 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 Barnstable is 2.44 people per housing unit, while year-round occupancy of available housing units is 78%. Average water use for single-family residences with municipal water accounts in the Centerville River/East Bay watershed is 204 gpd. If this flow is multiplied by 0.9 to account for consumptive use, the watershed average is 183 gpd. If this flow is then divided by 55 gpd, the average estimated occupancy in the watershed is 3.3 people per household.

In most previously completed MEP studies, average population and average water use have generally agreed fairly well. Since review of water use in the Centerville River/East Bay watershed suggests that on average there is approximately one additional person per housing unit (or 30% higher than predicted), MEP staff reviewed more refined US Census information, 1990 Census information and water use information for each parcel within the watershed. Besides reviewing data on town and state levels, the US Census also develops information for smaller areas (i.e., tracts and block groups). Portions of six Census tracts are contained within the watershed to Centerville River/East Bay; year 2000 Census residential occupancy rates in the tracts range from 2.18 to 2.73 people per house. Average occupancy for these tracts reported for the 1990 Census range from 2.22 to 2.83 people per house. While these occupancies suggest that the area is given to a fairly wide range of readings, these occupancies are less than the occupancy expected based on water use.

MEP staff then reviewed the average water uses measured in the subwatersheds of the Centerville River/East Bay system. While the overall average for single-family residences (SFRs) is 202 gpd, averages in the subwatersheds varied widely with a range between 138 and 288 gpd. Review of individual SFR water uses within subwatershed ranged as high as 3,429 gpd, but even this use was consistent across five years of data. The standard deviation among all the watershed averages is 39 gpd; the 134 gpd population estimated average is within 1.4 standard deviations of the 183 gpd measured water use mean.

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. While land use characteristics in the Centerville River/East Bay subwatershed may be unlikely to see summer population increases at the upper end of regional estimates, a doubling of the town occupancy for three months would be sufficient to increase the average annual water use based on Title 5 to 201 gpd, which is the same as the average measured flow in the watershed. The above analysis suggests that additional analysis of water uses within the Centerville River/East Bay watershed should be considered, but review of the water uses on the parcel, Census tract, and watershed scales do not suggest that there is any consistent inaccuracies. Given all the above analysis and the difficulty in accurately gauging seasonal population fluctuations, MEP staff decided to continue to use the Centerville River/East Bay watershed-specific water uses without any additional factors and used the average water use for the residential parcels without water use and for the 405 additional residential parcels included in the buildout analysis.

Although water use information exists for 94% of the approximately 6,936 developed parcels in the Centerville River/East Bay watershed, there are 417 parcels that 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 417 parcels, 83% of them (346) are classified as single-family residences (land use code 101) and another 14% are classified as other types of residential development [e.g. 109 (multiple houses on a single property)]. The remaining 3% of the parcels are commercial and industrial properties (300s and 400s land use codes, respectively). MEP staff used current water use to develop a watershed-specific water use estimate for the residential uses that were assumed to utilize private wells (Table IV-2). Commercial and industrial properties assumed to have private wells were assigned a water use based on percentage of parcel building coverage and average water uses for commercial properties in the Three Bays and Eel Pond/Back River watersheds.

Table IV-2. Average Water Use in Centerville River/East Bay Watershed.

Land Use	State Class Codes	# of Parcels with Water Use in Watershed	Water Use (gallons per day)	
			Watershed Average	Subwatershed Average Range
Residential	101	6077	204	138 to 288
Commercial	300 to 389	43	817	21 to 9,693
Industrial	400 to 439	none	--	--

Note: All data for analysis supplied by Town of Barnstable GIS Services Department and COMM Water District.

As mentioned previously, the eastern edge of the watershed is the boundary between the COMM Water District and the Town of Barnstable Water Supply Division (WSD) public water supply systems. Available WSD water uses exist for 1993, but more up-to-date data is not currently available. MEP staff reviewed the 1993 average residential water use and found that multiplying the 1993 water use by two brought the average into the low end of the subwatershed average water use range. There are 75 parcels in subwatershed #7 (Craigville Well) and 2 parcels in subwatershed #5 (Centerville River East LT10) that utilize this adjusted 1993 water use for estimating the wastewater nitrogen load.

Nine properties within the Centerville River/East Bay watershed have innovative/alternative (I/A) septic systems that are designed to reduce the amount of nitrogen in their effluent. MEP staff obtained a list of properties in the Town of Barnstable from the Barnstable County Department of Health and the Environment (Sue Rask, personal communication). The properties in the project watershed were then identified using the Cape Cod Commission GIS. BCDHE is tracking I/A systems for a number of Cape Cod towns and, in the process, have developed an extensive database of their performance. Based on 368 samples, the average total nitrogen effluent produced by these systems following an initial startup phase is 18.4 mg/l. This concentration was adjusted to a loading factor of 16.6 mg/l based on the similar method used to determine the loading factor for standard Title 5 systems. This factor was applied to the water use for the nine properties identified as using I/A nitrogen reducing septic systems.

It should also be noted that a community septic system collects and treats wastewater around a portion of Red Lilly Pond. The system does not have enhanced treatment according to Town of Barnstable staff (personal communication, John Jacobson), but the leaching field is located in the Centerville River E LT10 subwatershed (#5). Town staff identified the parcels connected to this system and the wastewater loads were added to subwatershed #5 in the nitrogen-loading model.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of estuary watershed nitrogen loading is usually fertilized lawns, golf courses, and cranberry bogs, with lawns being the predominant source within this category. In order to add this source to the nitrogen loading model for the Centerville River/East Bay system, MEP staff reviewed available information about residential lawn fertilizing practices and incorporated site-specific information to determine nitrogen loading from large tracks of turf in the watershed. MEP staff contacted the staff at appropriate organizations regarding the following large turf areas: the municipal Olde Barnstable Fairgrounds Golf Course, playing fields for the Barnstable Public Schools, and the horse paddocks at the Sheriff's Youth Ranch. Cranberry bog nitrogen loading was determined based on previous studies conducted in southeastern Massachusetts.

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

The initial effort in this assessment 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 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 Superintendent Bruce McIntyre to obtain current information for fertilizer practices at the municipal Olde Barnstable Fairgrounds Golf Course. 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 Olde Barnstable Fairgrounds Golf Course average annual nitrogen application rates (lbs/1,000 ft2) for the various turf areas are: greens, 4.5; tees, 4; fairways, 3, and rough, 2. As has been done in all MEP reviews, MEP staff reviewed the layout of the Olde Barnstable Fairgrounds Golf Course, classified the turf types, and assigned these areas to the appropriate subwatersheds. The nitrogen application rates were then applied to these areas and a load was calculated. Only a portion of the golf course is located in the watershed and the entire portion in the watershed is located within subwatershed to the Skunknet River (#16).

MEP staff also contacted Lee Saarkinnen of the Barnstable Public Schools and Kathy Hill, who has a contract to run the horse program at the Sheriff's Youth Ranch. Ms. Hill indicated that the horse paddocks are not fertilized and all horse manure on the site is removed every week. Based on this, no additional nitrogen load was added to the model for these activities at the Ranch. Mr. Saarkinnen indicated that turf at the Barnstable Public Schools playing fields have an annual nitrogen application rate of 0.75 lbs per acre. These playing fields are located at the Barnstable Middle and High School (subwatersheds #5, 7) and the Marstons Mills Horace Mann School (subwatershed #12). Field areas were determined based on review of aerial photographs and loads were determined and assigned to the appropriate subwatersheds on this basis.

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 based on review of aerial photographs for properties classified as cranberry bogs in the town-supplied land use classifications.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and Massachusetts DEP's

Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). Factors used in the MEP nitrogen loading analysis for the Centerville River/East Bay watershed are summarized in Table IV-3.

Table IV-3. Primary Nitrogen Loading Factors used in the Centerville River/East Bay MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Barnstable data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.

<table>
<tr><th>Nitrogen Concentrations:</th><th>mg/l</th><th>Recharge Rates:</th><th>in/yr</th></tr>
<tr><td>Road Run-off</td><td>1.5</td><td>Impervious Surfaces</td><td>40</td></tr>
<tr><td>Roof Run-off</td><td>0.75</td><td>Natural and Lawn Areas</td><td>27.25</td></tr>
<tr><td>Direct Precipitation on Embayments and Ponds</td><td>1.09</td><td colspan="2">Water Use/Wastewater:</td></tr>
<tr><td>Natural Area Recharge</td><td>0.072</td><td rowspan="4">Existing developed residential parcels wo/water accounts and buildout residential parcels:</td><td rowspan="4">204 gpd</td></tr>
<tr><td>Wastewater Coefficient</td><td>23.63</td></tr>
<tr><td>Fertilizers:</td><td></td></tr>
<tr><td>Average Residential Lawn Size (ft^2)*</td><td>5,000</td></tr>
<tr><td>Residential Watershed Nitrogen Rate (lbs/lawn)*</td><td>1.08</td><td>Existing developed parcels w/water accounts:</td><td>Measured annual water use</td></tr>
<tr><td>Cranberry Bogs nitrogen application (lbs/ac)</td><td>31</td><td rowspan="3">Commercial and industrial parcels wo/water accounts and buildout additions:</td><td rowspan="3">21 gpd/1,000 ft2 of building</td></tr>
<tr><td>Cranberry Bogs nitrogen attenuation</td><td>34%</td></tr>
<tr><td colspan="2" rowspan="2">Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined from site-specific information</td></tr>
<tr><td>Commercial and industrial building coverage for parcels wo/water accounts and buildout additions:</td><td>28%</td></tr>
</table>

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information was linked to the parcel coverages, parcels were assigned to various watersheds based initially on whether at least 50% or more of the land area of each parcel was located within a respective watershed. Following the assigning of boundary parcels, all large parcels were examined individually and were split (as appropriate) in order to obtain less than a 2% difference between the total land area of each subwatershed and the sum of the area of the parcels within each subwatershed. The resulting "parcelized" watersheds to Centerville River/East Bay are shown in Figure IV-3.

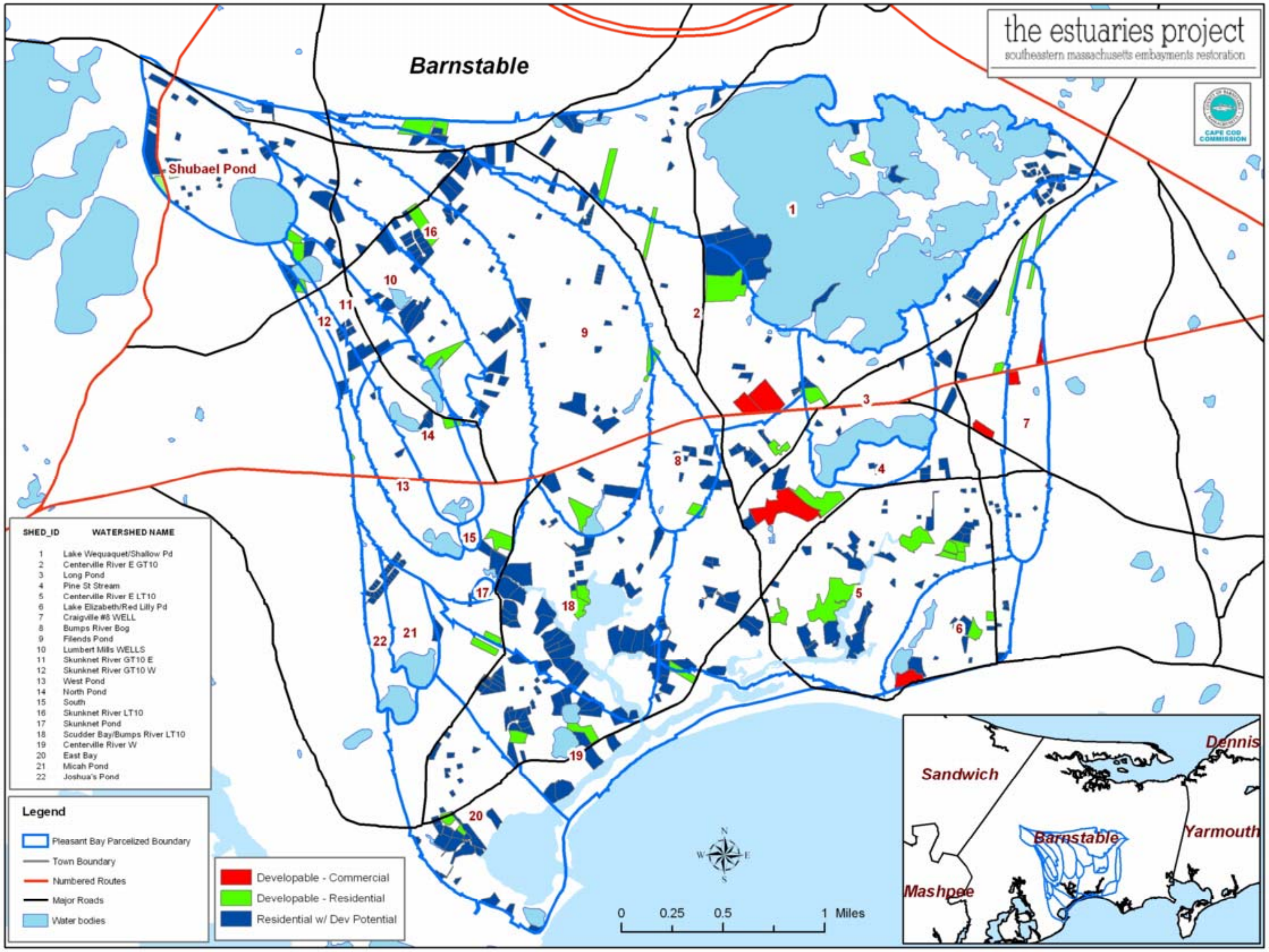


Figure IV-3. Parcels, Parcelized Watersheds, and Developable Parcels in the Centerville River/East Bay watersheds.

Table IV-4. Centerville River/East Bay Nitrogen Loads. Attenuation of Centerville River/East Bay system nitrogen loads occurs as nitrogen moves through upgradient ponds and streams during transport to the estuary. All values are kg N yr^{-1}.

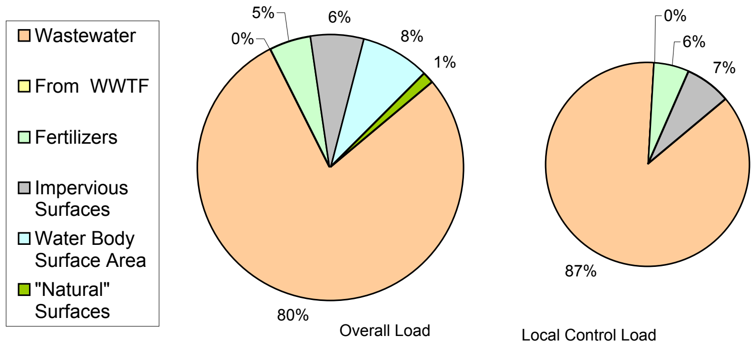
		Centerville River/East Bay N Loads by Input (kg/yr):							% of Pond Outflow	Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	From WWTF	Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
CENTERVILLE RIVER/EAST BAY SYSTEM		**48493**	**47**	**3155**	**3972**	**5156**	**921**	**3107**		**61745**		**48277**	**64852**		**50731**
Centerville River East		**23544**	**47**	**1150**	**2006**	**3759**	**402**	**1735**		**30908**		**22466**	**32643**		**23779**
Centerville R E LT10	5	10577	47	83	841	0	186	797		11733		11733	12530		12530
Craigville #8 WELL	7	451	0	40	53	0	11	60	48%	555		555	616		616
Lake Wequaquet (LW)	LW	1905	0	163	161	1245	37	148	38%	3510	50%	1755	3658	50%	1829
Long Pond (LP)	LP	572	0	47	51	246	10	50	15%	925	50%	315	975	50%	333
Centerville R E GT10	2 + LW	4653	0	353	410	670	78	259		6164		5234	6423		5453
Pine Street Stream	4 + LP,LW	3717	0	298	326	1384	61	286		5786	20%	1882	6072	20%	1972
Lake Elizabeth/Red Lilly Pd (stream att too?)	6, 7	1669	0	167	164	50	20	135	100%	2070	60%	828	2206	60%	882
Centerville River East Estuary Surface		0	0	0	0	164	0	0		164		164	164		164
Scudder Bay		**19614**	**0**	**1699**	**1577**	**611**	**397**	**1027**		**23898**		**19208**	**24925**		**20010**
Bumps River Bog	8	1098	0	230	92	0	19	96		1440		1440	1536		1536
Filends Pond	9	6746	0	544	520	25	92	222	100%	7928	20%	6343	8150	20%	6520
Scudder Bay	18	3100	0	163	212	0	67	163		3542		3542	3704		3704
Scudder Bay Estuary Surface		0	0	0	0	250	0	0		250		250	250		250
Skunknet River		8670	0	762	752	335	218	547		10738	20%	7634	11284	20%	8000
Lumbert Mills WELLS	10 + SP	1837	0	138	128	121	53	277		2276			2553		2006
Skunknet R GT10 E	11 + SP	758	0	64	65	79	21	68		987		739	1055		784
Skunknet R GT10 W	11 + SP	383	0	35	30	51	10	47		509		333	557		364
West Pond	13	712	0	60	66	46	13	15		897	30%	628	912	30%	638
North Pond	14	569	0	45	49	34	25	15		722		722	737		737
South	15	0	0	0	0	0	2	0		2		2	2		2
Skunknet R LT10	16	4411	0	420	414	4	94	126		5343		5343	5468		5468
Skunknet Pond	17	0	0	0	0	0	1	0		1	50%	1	1	50%	1
Centerville River West		**2812**	**0**	**145**	**180**	**262**	**56**	**200**		**3454**		**3454**	**3654**		**3654**
Centerville R W	19	2812	0	145	180	0	56	200		3192		3192	3392		3392
Centerville River West Estuary Surface		0	0	0	0	262	0	0		262		262	262		262
East Bay		**2523**	**0**	**161**	**210**	**525**	**67**	**144**		**3486**		**3149**	**3630**		**3288**
East Bay	20	2117	0	129	166	0	51	133		2464		2464	2597		2597
Joshua's Pond (JP)	JP + MP	340	0	29	37	86	11	11	77%	504	50%	221	515	50%	226
Micah's Pond (MP)	MP	66	0	3	7	28	4	0	40%	107	50%	53	107	50%	53
East Bay Estuary Surface		0	0	0	0	411	0	0		411		411	411		411

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. 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 Centerville River/East Bay estuary. The assignment effort was undertaken to better define the sub-embayment loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

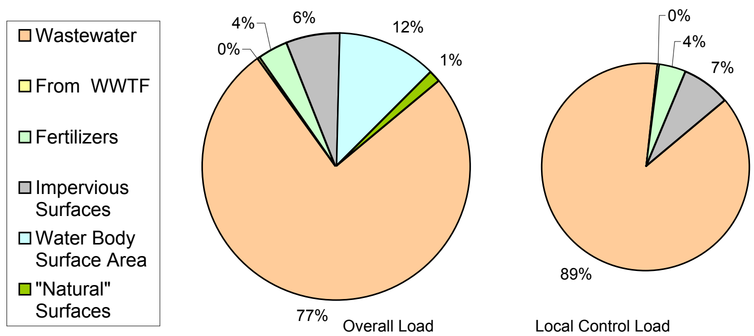
Following the assignment of all parcels, subwatershed modules were generated for each of the 22 sub-watersheds summarizing water use, parcel area, frequency, sewer connections, private wells, and road area. The individual sub-watershed modules were then integrated to create the Centerville River/East Bay Watershed Nitrogen Loading module with summaries for each of the individual subembayments. The subembayments represent the functional embayment units for the Linked Watershed-Embayment Model's water quality component.

For management purposes, the aggregated embayment watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Centerville River/East Bay System, the major types of nitrogen loads are: wastewater (e.g., septic systems), fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-4). The output of the watershed nitrogen loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-4 a-e). 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.

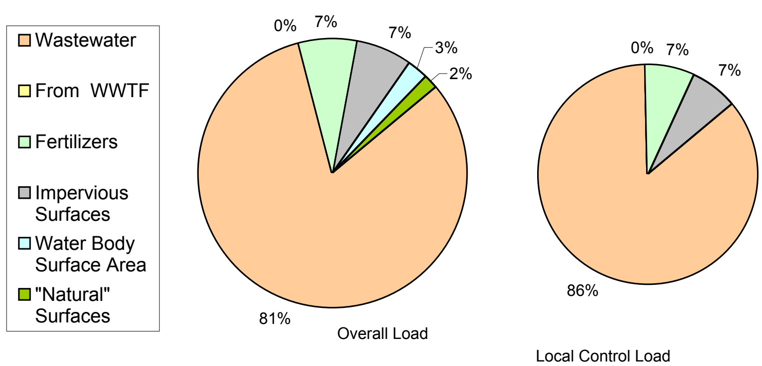
Since groundwater outflow from a pond can enter more than one downgradient sub-watershed, the length of shoreline on the downgradient side of the pond was used to apportion the pond-attenuated nitrogen load to respective downgradient watersheds. The apportionment was based on the percentage of discharging shoreline bordering each downgradient sub-watershed. So for example, Lake Wequaquet has a downgradient shoreline of 9,086 feet; 42% of that shoreline discharges into the Long Pond subwatershed (watershed 3 in Figure IV-1), 38% discharges to the Centerville River E LT10 subwatershed (watershed 6), and 20% discharges to the Centerville River E GT10 subwatershed. The attenuated nitrogen load discharging from Lake Wequaquet is divided among these subwatersheds based on these percentages of the downgradient shoreline. Using this method, 83% of the loads developed in the Three Bays MEP analysis for Shubael Pond are added to the Centerville River/East Bay subwatersheds.



a. Centerville River/East Bay System Overall

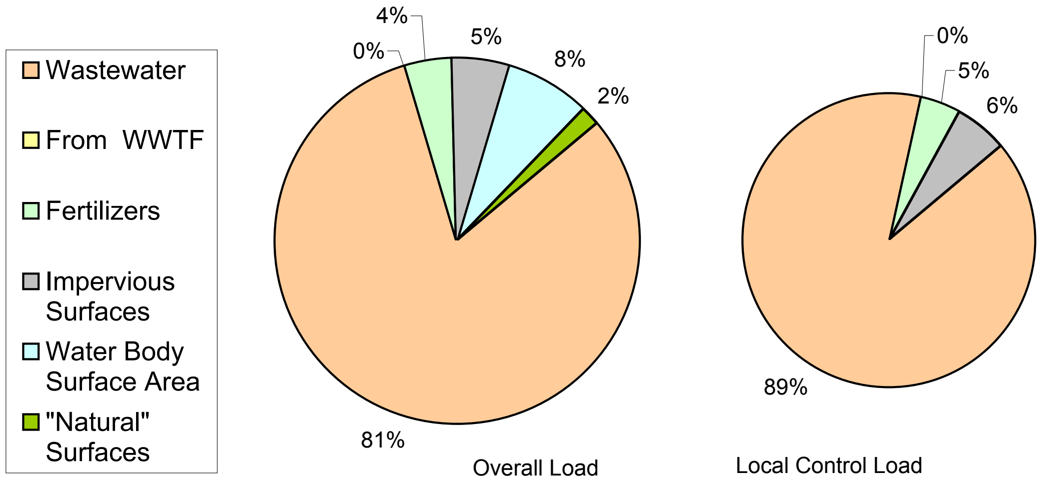


b. Centerville River East

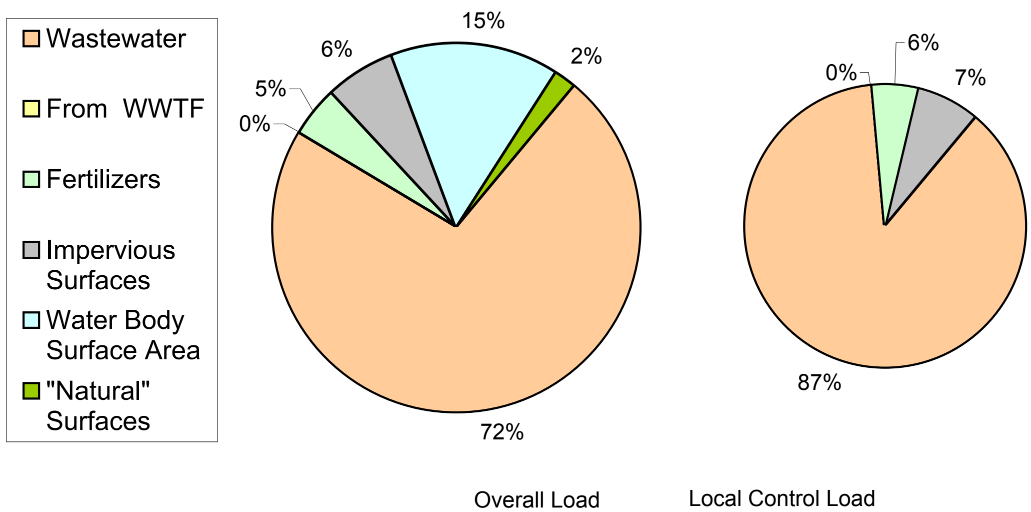


c. Scudder Bay/Bumps River

Figure IV-4 (a-c). Land use-specific unattenuated nitrogen load (by percent) to the (a) overall Centerville River/East Bay System watershed, (b) Centerville River East subwatershed, and (c) Scudder Bay/Bumps River subwatershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.



d. Centerville River West



E. East Bay

Figure IV-4 (d-e). Land use-specific unattenuated nitrogen load (by percent) to the (d) Centerville River West subwatershed, and (e) East Bay subwatershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally kettle hole depressions that intercept the surrounding groundwater table revealing what some call "windows on the aquifer." Groundwater typically flows into the pond along the upgradient shoreline, then lake water flows back into the groundwater system along the downgradient shoreline. Occasionally a Cape Cod pond will also have a stream outlet or 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 (reduced or attenuated) loads flow back into the groundwater system along the downgradient side of the pond or through a stream outlet and eventual discharge into the downgradient embayment. The nitrogen load summary in Table IV-4 includes both the unattenuated (nitrogen load to each subwatershed) and attenuated nitrogen loads.

Pond nitrogen attenuation in freshwater ponds has generally been found to be at least 50% in MEP analyses, so the watershed model contains a conservative attenuation rate of 50%. However, in some cases, if sufficient monitoring information is available, a pond-specific attenuation rate is incorporated into the watershed nitrogen loading modeling (Three Bays MEP Report, 2005). Detailed studies of other southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE, 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM, 2000) have also supported a 50% attenuation factor. In order to estimate nitrogen attenuation in the ponds physical and chemical data for each pond is reviewed. Available bathymetric information is reviewed relative to measured pond temperature profiles to determine whether an epilimnion (*i.e.*, well mixed, homothermic, upper portion of the water column) exists in each pond. Bathymetric information is necessary to develop a residence or turnover time and complete an estimate of nitrogen attenuation. In the Centerville River/East Bay watershed, bathymetric information is available for Lake Wequaquet and the following ponds: Bearses, Joshua's, Long, Micah, Shallow, and lower Red Lilly. Of the ponds with bathymetric information, none are deep enough to develop strong temperature stratification and samples from all depths generally can be used for determining average nitrogen concentrations. Deepest samples must be checked for potential impact by sediment regeneration of nitrogen, especially if low oxygen conditions occur.

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 not, the entire volume of the pond is used to determine a turnover time. Turnover time is how long it takes the recharge from the upgradient 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 downgradient 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. Generally, monitoring is insufficient to support use of a factor different than the standard 50% attenuation.

The standard attenuation assumption for the ponds in the Centerville River/East Bay watershed was checked through the use of pond water quality information collected from the annual Cape Cod Pond and Lake Stewardship (PALS) water quality snapshot. The PALS snapshot is a collaborative Cape Cod Commission/SMAST Program that allows trained, citizen volunteers of each of the 15 Cape Cod towns to collect pond samples in August and September using a standard protocol. Snapshot samples have been collected every year between 2001 and 2005. The standard protocol for the snapshot includes field collection of dissolved oxygen and temperature profiles, Secchi disk depth readings and water samples at various depths depending on the total depth of the pond. PALS snapshot data is available in the Centerville River/East Bay watershed for the following ponds: Bearses, Joshua, Lake Elizabeth, Long, Micah, Red Lilly, Shallow, and Lake Wequaquet. Water samples were analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH. Table IV-5 presents the turnover times and attenuation factors for the ponds in the Centerville River/East Bay watershed.

Table IV-6 summarizes the pond attenuation estimates calculated from land-use modeled nitrogen inflow loads and nitrogen loads recharged to the downgradient aquifer or to outflow streams from each pond based on pond characteristics and measured nitrogen levels. Nitrogen attenuation within these ponds varies between 84 and 96%. However, a caveat to these attenuation estimates is that they are based upon nitrogen outflow loads from water column samples collected at one time during the year, and are not necessarily representative of the annual nitrogen loads that are transferred downgradient.

Table IV-5. Nitrogen attenuation by Freshwater Ponds in the Centerville River/East Bay watershed based upon 2001 through 2005 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. The MEP Linked N Model for Centerville River/East Bay uses a standard value of 50% for the pond systems.

Pond	PALS ID	Area acres	Maximum Depth m	Overall turnover time Yrs	N Load Attenuation %
Bearses	BA-617	66.8	4.9	0.03	
Joshua	BA-807	14.7	7.1	0.51	96%
Lake Elizabeth	BA-795	6.3	1.1	1.00	
Long	BA-737	51.0	6.4	0.74	94%
Micah	BA-797	16.0	10.5	2.40	88%
Red Lilly	BA-782	4.5	0.8	0.78	
Shallow	BA-626	78.4	2.0	0.97	
Wequaquet complex	BA-605	596.3	8.1	2.42	84%
				Mean	91%
				std dev	6%

Data sources: all areas from CCC GIS; Max Depth from MADFW or Cape Cod PALS monitoring; Volume for turnover time calculations from MADFW bathymetric maps (www.mass.gov/dfwele/dfw/dfw_pond.htm); TN concentrations for attenuation calculation from PALS monitoring

Lake Wequaquet is listed as a “complex” in Table IV-5 because of lack of definition between Lake Wequaquet, Bearses Pond, and Shallow Pond in the original groundwater modeling completed for this project. Collected streamflow information at the Pine Street location indicates that flow from this complex must discharge through Long Pond (see Section IV.2.4), but more refined definition of groundwater flow within the complex is necessary to provide more detailed assessment of attenuation factors in the separate basins.

There are a number of “ponds” that are shown in the watershed delineation (see Figure III-1) that are not listed in Table IV-5. Although watersheds were delineated to these waters during the initial watershed modeling, review of aerial photographs has shown that many of these are either not as extensive or more closely approximate traditional wetlands. Attenuation factors of less than 50% are assigned to these components and further discussion is provided in Section IV.2.

Buildout

Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment of potential development within the study area watershed. For the Centerville River/East Bay modeling, MEP staff consulted with Town of Barnstable planners to determine the factors that would be used in the assessment (Thomas Broadrick, personal communication). MEP staff developed the buildout by reviewing the development potential of each property. The buildout procedure used in this subwatershed and generally completed by MEP staff is to evaluate town zoning to determine minimum lot sizes in each of the zoning districts, including overlay districts (*e.g.*, water resource protection districts). Larger lots are subdivided by the minimum lot size to determine the total number of new lots and existing developed properties are reviewed for additional development potential; for example, residential lots that are twice the minimum lot size, but have only one residence. MEP staff also included additional development on residential parcels that are classified as developable residential (state class land use codes 130 and 131) but are less than the minimum lot size and are greater than 5,000 square feet. These parcels are assigned one residence in the buildout; 5,000 square feet is a common minimum buildable lot size in Cape Cod town regulations. Properties classified by the Barnstable assessor as “undevelopable” (e.g., codes 132, 392, and 442) were not assigned any development at buildout. Commercially developable properties were not subdivided; the area of each parcel and the factors in Table IV-3 were used to determine a wastewater flow for these properties. All the parcels included in the buildout assessment of the Centerville River/East Bay watershed are shown in Figure IV-3.

Overall, a nitrogen load for each additional residence or business is included in the cumulative unattenuated buildout indicated in a separate column in Table IV-4. Buildout additions within the overall Centerville River/East Bay System watershed will increase the unattenuated loading rate by 5%.

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 Centerville River System (inclusive of Scudder Bay and Bumps River estuarine reaches) being investigated under this nutrient threshold analysis was based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1). If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watersheds where nitrogen transport from source to estuarine waters is through groundwater flow in sandy outwash aquifers (such as the developed region of the Centerville River System 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 case of the Centerville River embayment system watersheds, a portion of the freshwater flow and transported nitrogen passes through several surface water systems (Skunknett River, Bumps River, stream from Long Pond, stream from Lake Elizabeth) 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. MEP conducted long-term measurements of natural attenuation relating to surface water discharges to the head of the embayment system (e.g. Scudder Bay, estuarine reach of Bumps River) in addition to the natural attenuation measures by fresh kettle ponds, addressed above (Section IV.1). These additional site-specific studies were conducted in the 4 major surface water flow systems in the watershed to Centerville River, 1) the Skunknett River discharging to the estuarine reach of Bumps River, 2) Bumps River discharging to Scudder Bay, 3) the stream from Long Pond discharging to Centerville River and 4) the stream from Lake Elizabeth discharging to Centerville River (Figure IV-5).

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 upgradient from the various gauging sites. Flow and nitrogen load were measured at the gages in each freshwater stream site for between 15 and 26 months of record depending on the stream gaging location (Figures IV-6 to IV-9). During each study period, velocity profiles were completed on each river every month to two months. The summation of the products of stream subsection areas of the stream cross-section and the respective measured velocities represent the computation of instantaneous stream flow (Q).

Figure IV-5. Location of Stream gage (red symbols) in the Centerville River/Harbor embayment system.

Determination of stream flow at each gage 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 gage 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 gages. 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 estuarine portion of Centerville River/Harbor system.

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

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Stream Discharge from Skunknett River to Estuarine Reach of Bumps River

North Pond, West Pond and South Pond, located upgradient of the Skunknett River gage site is a complex of small freshwater ponds and unlike many of the freshwater ponds, this network of ponds has stream outflow rather than discharging solely to the aquifer along its down-gradient shore. This stream outflow, the Skunknett River, may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation. In addition, nitrogen attenuation also occurs within the wetlands and streambed associated with the Skunknett River. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the Skunknett River above the gage site and the measured annual discharge of nitrogen to the tidal portion of the Bumps River, Figure IV-5.

At the Skunknett River gage site, a continuously recording vented calibrated water level gage was installed to yield the level of water in the freshwater portion of the Skunknett River-Bumps River estuarine reach system that carries the flows and associated nitrogen load to the head of the upper portion of the estuarine reach of the Bumps River. As the Skunknett River is tidally influenced the gage was located above the saltwater reach such that freshwater flow could be measured without tidal influence. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gage site. Average low tide salinity was determined to be <0.9 ppt. Therefore, the gage location was deemed acceptable for making freshwater flow measurements. Calibration of the gage was checked monthly. The gage on the Skunknett River was installed on March 27, 2003 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until September 8, 2005 for a total deployment of 26 months. Two 12-month uninterrupted records used in this analysis encompass both the summer 2004 and 2005 field season.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Skunknett River site based upon these flow measurements and measured water levels at the gage 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 discharge to the estuarine portion of the Bumps River flowing to Centerville River/Harbor (Figure IV-6 and Table IV-6). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gage site.

The annual freshwater flow record for the Skunknett River measured by the MEP was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from the Skunknett River was 18% above the long-term average modeled flows. Measured flow in the Skunknett River was obtained for two distinct yet consecutive hydrologic years (September 2003 to August 2004 and September 2004 to August 2005). The average flow for the two hydrologic years measured were used for the purpose of modeling the estuarine system. The average daily flow based on the MEP measured flow data was 13,925 m^3/day compared to the long term average flows determined by the USGS modeling effort (11,353 m^3/day). Additional confirmation of the MEP measured discharge for the Skunknett River was sought in comparing the MEP measured flow to measured flows for the Skunknett River which were obtained independent of the MEP effort during a study for the Town of Barnstable of the Audubon Skunknett River Wildlife Sanctuary (Hamersley, 2004). The study focused on nitrogen fluxes and mitigation strategies in the pond and river system as was conducted by SMAST/Coastal Systems Group scientists. Based on flow measurements conducted in 2002 to 2003 period, the flow from the Skunknett River at the same location as the MEP gage was determined to be 13,317 m^3/day. This flow is consistent with the MEP determined flow in the following years and it was concluded that the MEP flow for the Skunknett River was justified.

Table IV-6. Comparison of water flow and nitrogen discharges from Rivers and Streams (freshwater) discharging to estuarine reach of Centerville River. 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	Skunknett River Discharge[(a)]	Bumps River Discharge[(a)]	Long Pond Stream Discharge[(a)]	Lake Elizabeth Stream Discharge[(a)]	Data Source
Total Days of Record	365[(b)]	365[(b)]	365[(b)]	365[(b)]	(1)
Flow Characteristics					
Stream Average Discharge (m3/day) **	13925	5679	6518	1547	(1)
Contributing Area Average Discharge (m3/day)	11353	5217	6254	1381	(2)
Discharge Stream 2002-03 vs. Long-term Discharge	18%	8%	4%	11%	
Nitrogen Characteristics					
Stream Average Nitrate + Nitrite Concentration (mg N/L)	1.113	2.027	0.199	0.937	(1)
Stream Average Total N Concentration (mg N/L)	1.495	2.978	0.530	1.470	(1)
Nitrate + Nitrite as Percent of Total N (%)	74%	68%	38%	64%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	20.81	16.91	3.45	2.27	(1)
TN Average Contributing UN-attenuated Load (kg/day)	29.42	21.72	10.34	5.67	(3)
Attenuation of Nitrogen in Pond/Stream (%)	29%	22%	67%	60%	(4)

(a) Flow and N load to streams discharging to Centerville includes apportionments of Pond contributing area.
(b) September 1, 2003 to August 31, 2004.
** Flow is an average of annual flow for 2003-2005

(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 Centerville 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.

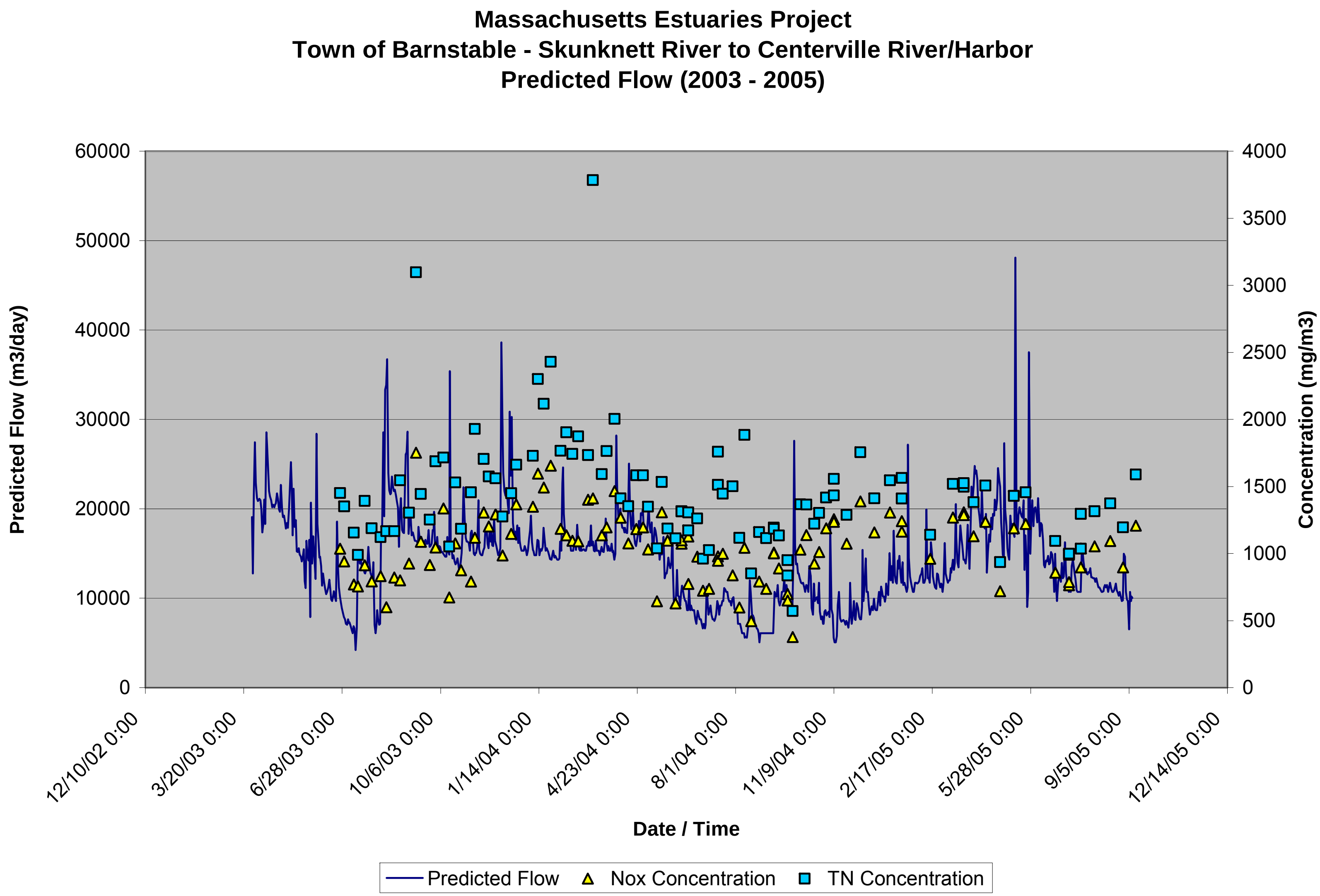


Figure IV-6. Skunknett River discharge (solid blue line), nitrate+nitrite (yellow triangle) and total nitrogen (blue box) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Scudder Bay (Table IV-7).

The difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in the Skunknett River may in part be due to above average rainfall during the stream gage deployment period based on rainfall records obtained from a rain gage in the Town of Hyannis. Ten years of rainfall data (1993-2004) indicate that the average rainfall in the vicinity of the Centerville River system was 38.76 inches. By comparison, rainfall in 2003 and 2004 was 50.07 and 34.62 inches respectively. Rainfall in 2002 was 48.28 inches (above long term average). This was in contrast to rainfall amounts totaling 50.07 inches in 2003. It should be recognized that 2002 and 2003 rainfall was above average thus the water table is likely to have been higher than usual due to the 2 years of higher rainfall. This is significant relative to measured flow in the Skunknett River surface water system as it is essentially a groundwater fed feature. Based upon the rainfall and groundwater levels associated with the stream measurement (suggesting a higher flow than the long-term average) and the some what higher measured stream discharge then predicted (+18%) it appears that the stream is capturing the upgradient recharge (and loads) accurately.

Total nitrogen concentrations within the Skunknett River outflow were moderate, 1.495 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 20.81 kg/day and a measured total annual TN load of 5657 kg/yr. In the Skunknett River, nitrate was the predominant form of nitrogen (74%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was not completely taken up by plants within the pond or stream ecosystems. The high concentration of inorganic nitrogen in the outflowing stream waters also suggests that plant production within the upgradient freshwater ecosystems is not nitrogen limited. In addition, the high nitrate level suggests the possibility for additional uptake by freshwater systems might be accomplished in this system either within the North Pond, West Pond and South Pond network or along the freshwater reach of the Skunknett River.

From the measured nitrogen load discharged by the Skunknett River 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 (7596 kg yr^{-1}) discharged from the freshwater Skunknett River compared to that added by the various land-uses to the associated watershed (10738 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 29% (i.e. 29% of nitrogen input to watershed does not reach the estuary). This slightly lower level of attenuation compared to other streams evaluated under the MEP is expected given the hydraulic nature of the network of upgradient ponds which are essentially shallow flow through systems. The directly measured nitrogen loads from the river was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

IV.2.3 Surface water Discharge and Attenuation of Watershed Nitrogen: Stream Discharge from Bumps River to Scudder Bay

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

At the Bumps River gage site, a continuously recording vented calibrated water level gage was installed to yield the level of water in the freshwater portion of the Bumps River that carries the flows and associated nitrogen load to the head of Scudder Bay and the estuarine reach of the Bumps River. As the Bumps River is tidally influenced the gage was located above the saltwater reach such that freshwater flow could be measured without tidal influence. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gage site. Average low tide salinity was determined to be <0.1 ppt. Therefore, the gage location was deemed acceptable for making freshwater flow measurements. Calibration of the gage was checked monthly. The gage on the Bumps River was installed on March 27, 2003 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until November 8, 2004 for a total deployment of 21 months. The 12-month uninterrupted record used in this analysis encompasses the summer 2003 and 2004 field season.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Bumps River site based upon these flow measurements and measured water levels at the gage 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 discharge to the estuarine portion of Scudder Bay and the Bumps River (Figure IV-7 and Table IV-6). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gage site.

The annual freshwater flow record for the Bumps River measured by the MEP was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from the Bumps River was 8% above the long-term average modeled flows. Measured flow in the Bumps River was obtained for one hydrologic year (September 2003 to August 2004). The average daily flow based on the MEP measured flow data was 5,679 m^3/day compared to the long term average flows determined by the USGS modeling effort (5,217 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 Bumps River are in part be due to above average rainfall during the stream gage deployment period based on rainfall records obtained from a rain gage in the Town of Hyannis. Based on ten years of rainfall data (1993-2004) the average rainfall in the vicinity of the Centerville River system was 38.76 inches. By comparison, rainfall in 2003 and 2004 was 50.07 and 34.62 inches respectively. Rainfall in 2002 was 48.28 inches (above long term average). This was in contrast to rainfall amounts totaling 50.07 inches in 2003. It should be recognized that 2002 and 2003 rainfall was above average thus the water table is likely to have been higher than usual due to the 2 years of higher rainfall. This is significant relative to measured flow in the Bumps River surface water system as it is essentially a groundwater fed feature. Based upon the rainfall and groundwater levels associated with the stream measurement (suggesting a higher flow than the long-term average) and the some what higher measured stream discharge then predicted (+8%) it appears that the stream is capturing the upgradient recharge (and loads) accurately.

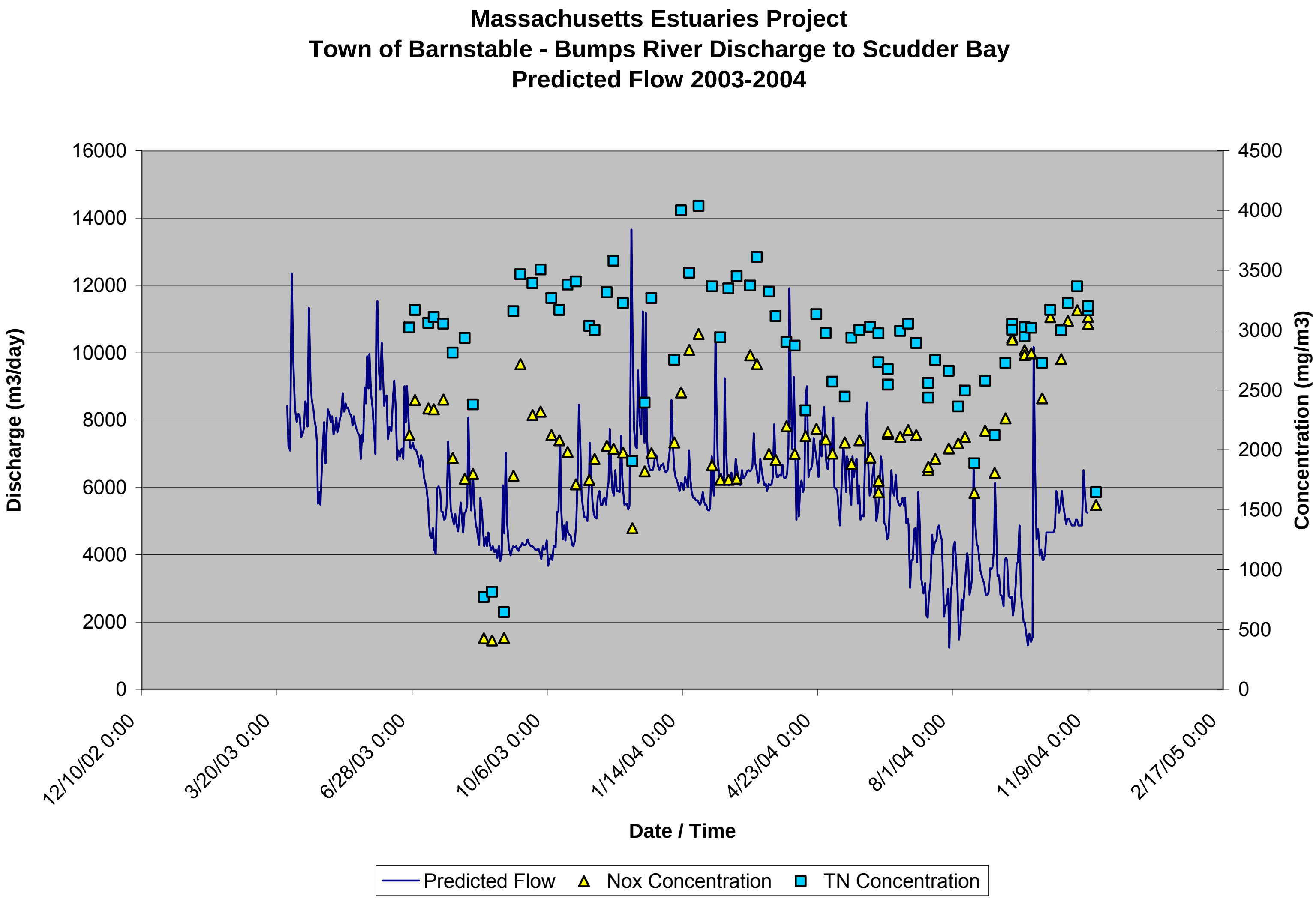


Figure IV-7. Bumps River discharge (solid blue line), nitrate+nitrite (yellow triangle) and total nitrogen (blue box) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Scudder Bay (Table IV-7).

Total nitrogen concentrations within the Bumps River outflow were high, 2.978 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 16.91 kg/day and a measured total annual TN load of 6173 kg/yr. In the Bumps River, nitrate was the predominant form of nitrogen (68%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was not completely taken up by plants within the pond or stream ecosystems. The high concentration of inorganic nitrogen in the outflowing stream waters also suggests that plant production within the upgradient freshwater ecosystems is not nitrogen limited. In addition, the high nitrate level suggests the possibility for additional uptake by freshwater systems might be accomplished in this system either within Filends Pond or along the freshwater reach of the Bumps River.

From the measured nitrogen load discharged by the Bumps River 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 nitrogen load (6173 kg yr^{-1}) discharged from the freshwater Bumps River compared to that added by the various land-uses to the associated watershed (7928 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 22% (i.e. 22% of nitrogen input to watershed does not reach the estuary). This slightly lower level of attenuation compared to other streams evaluated under the MEP is expected given the hydraulic nature of the upgradient pond which is essentially a shallow flow through system. The directly measured nitrogen loads from the river was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

IV.2.4 Surface water Discharge and Attenuation of Watershed Nitrogen: Stream Discharge from Long Pond to Centerville River

Long Pond located upgradient of the Pine Street gage site is a moderately sized freshwater pond and unlike many of the freshwater ponds, this pond has stream outflow rather than discharging solely to the aquifer along its down-gradient shore. This stream outflow, the stream from Long Pond to Centerville River, may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation. In addition, nitrogen attenuation also occurs within the wetlands and streambed associated with the Long Pond Stream. 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 Long Pond and stream system above the gage site and the measured annual discharge of nitrogen to the tidal portion of Centerville River, Figure IV-5.

At the Pine Street gage site, a continuously recording vented calibrated water level gage was installed to yield the level of water in the freshwater portion of the stream from Long Pond that carries the flows and associated nitrogen load to the estuarine reach of the Centerville River. As Centerville River is tidally influenced the gage was located above the saltwater reach such that freshwater flow could be measured without tidal influence. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gage site. Average low tide salinity was determined to be <0.1 ppt. Therefore, the gage location was deemed acceptable for making freshwater flow measurements. Calibration of the gage was checked monthly. The gage on the stream from Long Pond was installed on April 22, 2004 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until November 8, 2005 for a total deployment of 18 months. The 12-month uninterrupted record used in this analysis encompasses the summer 2004 and 2005 field season.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Pine Street gage site based upon these flow measurements and measured water levels at the gage 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 discharge to the estuarine portion of the Centerville River (Figure IV-8 and Table IV-6). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gage site.

The annual freshwater flow record for the stream from Long Pond 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 from Long Pond was 4% above the long-term average modeled flows. Measured flow in the Long Pond stream was obtained for one hydrologic year (September 2004 to August 2005). The average daily flow based on the MEP measured flow data was 6,518 m^3/day compared to the long term average flows determined by the USGS modeling effort (6,254 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 Long Pond stream may in part be due to above average rainfall during the stream gage deployment period based on rainfall records obtained from a rain gage in the Town of Hyannis. Based on ten years of rainfall data (1993-2004) the average rainfall in the vicinity of the Centerville River system was 38.76 inches. By comparison, rainfall in 2003 and 2004 was 50.07 and 34.62 inches respectively. Rainfall for the period August 2004 to July 2005 was 36.50 inches, slightly below the long term average of 38.76 inches. Rainfall in 2002 was 48.28 inches (above long term average). This was in contrast to rainfall amounts totaling 50.07 inches in 2003. It should be recognized that 2002 and 2003 rainfall was above average thus the water table is likely to have been higher than usual due to the 2 years of higher rainfall. This is significant relative to measured flow in the Long Pond stream surface water system as it is essentially a groundwater fed feature. Based upon the rainfall and groundwater levels associated with the stream measurement (suggesting a slightly higher flow than the long-term average) and the some what higher measured stream discharge then predicted (+4%) it appears that the stream is capturing the upgradient recharge (and loads) accurately.

Total nitrogen concentrations within the Long Pond stream outflow were moderate, 0.530 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 3.45 kg/day and a measured total annual TN load of 1,260 kg/yr. In the Long Pond stream, nitrate was not the predominant form of nitrogen (38%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was more completely taken up by plants within the pond or stream ecosystems. The lower concentration of inorganic nitrogen in the outflowing stream waters also suggests that plant production within the upgradient freshwater ecosystems may be slightly nitrogen limited. In addition, the lower nitrate levels in outflowing waters from Long Pond suggests slightly less possibility for additional uptake by freshwater systems might be accomplished in this system either within Long Pond or along the freshwater reach of the stream from Long Pond.

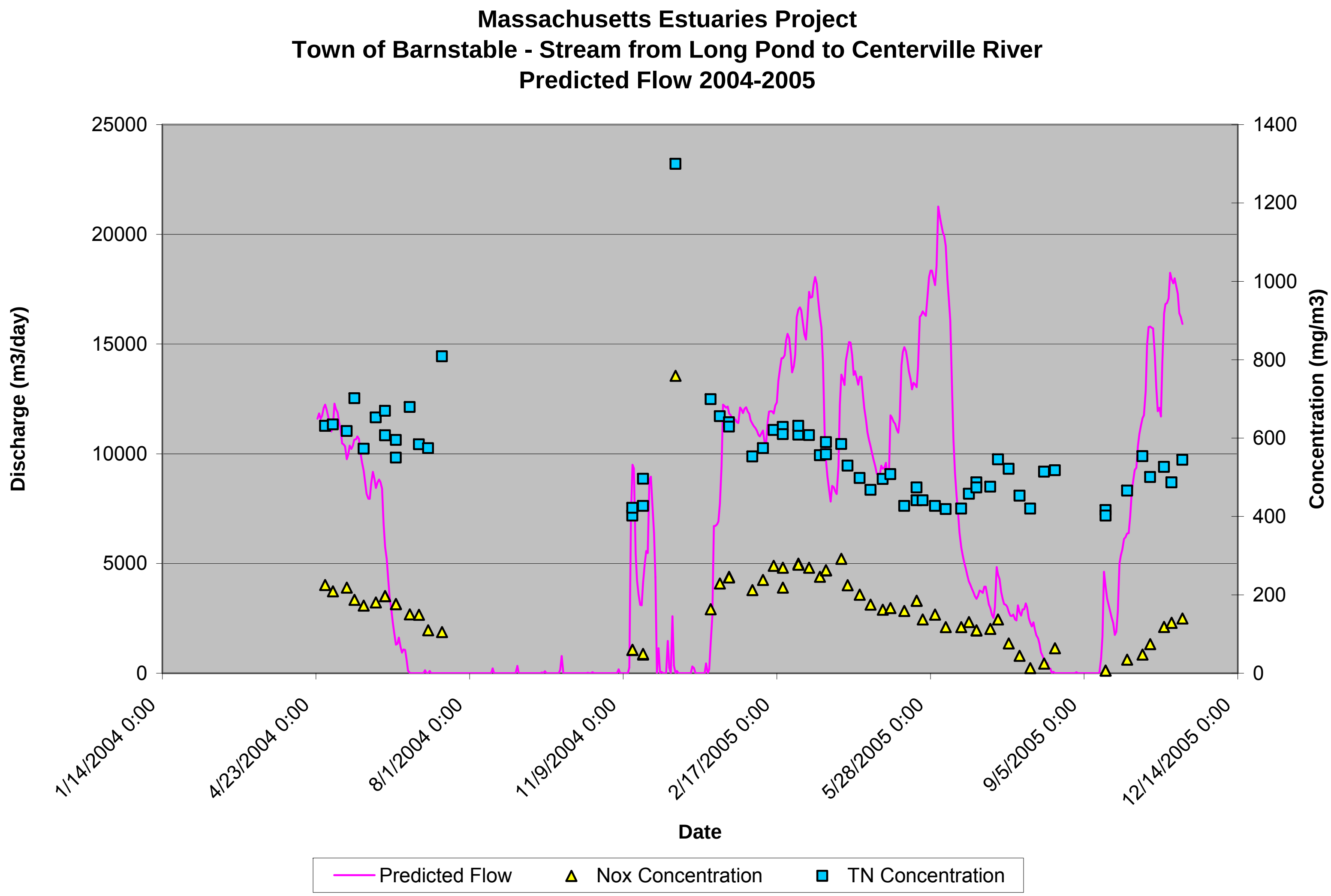


Figure IV-8. Stream discharge from Long Pond(solid pink line), nitrate+nitrite (yellow triangle) and total nitrogen (blue box) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Centerville River (Table IV-7).

From the measured nitrogen load discharged by the stream from Long Pond discharging to Centerville River 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 nitrogen load (1,260 kg yr^{-1}) discharged from the freshwater Back River compared to that added by the various land-uses to the associated watershed (3,773 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 67% (i.e. 67% 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 upgradient pond which is essentially a relatively deep kettle pond system with substantial residence time. The directly measured nitrogen loads from the stream was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

IV.2.5 Surface water Discharge and Attenuation of Watershed Nitrogen: Stream Discharge from Lake Elizabeth to Centerville River

Lake Elizabeth located upgradient of the stream gage site is a small freshwater pond and unlike many of the freshwater ponds, this pond has stream outflow rather than discharging solely to the aquifer along its down-gradient shore. This stream outflow, the Lake Elizabeth stream, may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation. In addition, nitrogen attenuation also occurs within the wetlands and streambed associated with the lake and its outflowing stream. 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 lake and stream above the gage site and the measured annual discharge of nitrogen to the tidal portion of Centerville River, Figure IV-5.

At the Lake Elizabeth stream gage site, a continuously recording vented calibrated water level gage was installed to yield the level of water in the freshwater portion of the stream that carries the flows and associated nitrogen load to the estuarine reach of Centerville River. As the Lake Elizabeth stream is tidally influenced the gage was located above the saltwater reach such that freshwater flow could be measured without tidal influence. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gage site. Average low tide salinity was determined to be <0.2 ppt. Therefore, the gage location was deemed acceptable for making freshwater flow measurements. Calibration of the gage was checked monthly. The gage on the stream from Lake Elizabeth was installed on May 15, 2003 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until November 8, 2004 for a total deployment of 18 months. The 12-month uninterrupted record used in this analysis encompasses the summer 2003 and 2004 field seasons.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Lake Elizabeth stream site based upon these flow measurements and measured water levels at the gage 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 discharge to the estuarine portion of Centerville River (Figure IV-9 and Table IV-6). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gage site.

The annual freshwater flow record for the stream from Lake Elizabeth 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 Lake Elizabeth stream was 11% above the long-term average modeled flows. Measured flow in the Lake Elizabeth stream was obtained for one hydrologic year (September 2003 to August 2004). The average daily flow based on the MEP measured flow data was 1,547 m^3/day compared to the long term average flows determined by the USGS modeling effort (1,381 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 Lake Elizabeth stream may in part be due to above average rainfall during the stream gage deployment period based on rainfall records obtained from a rain gage in the Town of Hyannis. Based on ten years of rainfall data (1993-2004) the average rainfall in the vicinity of the Centerville River system was 38.76 inches. By comparison, rainfall in 2003 and 2004 was 50.07 and 34.62 inches respectively. Rainfall in 2002 was 48.28 inches (above long term average). This was in contrast to rainfall amounts totaling 50.07 inches in 2003. It should be recognized that 2002 and 2003 rainfall was above average thus the water table is likely to have been higher than usual due to the 2 years of higher rainfall. This is significant relative to measured flow in the Lake Elizabeth surface water system as it is essentially a groundwater fed feature. Based upon the rainfall and groundwater levels associated with the stream measurement (suggesting a higher flow than the long-term average) and the some what higher measured stream discharge then predicted (+11%) it appears that the stream is capturing the upgradient recharge (and loads) accurately.

Total nitrogen concentrations within the Lake Elizabeth stream outflow were moderate, 1.470 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 2.27 kg/day and a measured total annual TN load of 830 kg/yr. In the Lake Elizabeth stream, nitrate was the predominant form of nitrogen (64%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was not completely taken up by plants within the pond or stream ecosystems. The high concentration of inorganic nitrogen in the outflowing stream waters also suggests that plant production within the upgradient freshwater ecosystems is not nitrogen limited. In addition, the high nitrate level suggests the possibility for additional uptake by freshwater systems might be accomplished in this system either within Lake Elizabeth or along the freshwater reach of the stream.

From the measured nitrogen load discharged by the Lake Elizabeth stream discharge 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 the lower nitrogen load (830 kg yr^{-1}) discharged from the freshwater stream compared to that added by the various land-uses to the associated watershed (2,070 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 60% (i.e. 60% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams evaluated under the MEP is expected given the hydraulic nature of the upgradient pond which is essentially a coastal kettle pond system with substantial residence time. The directly measured nitrogen loads from the river was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

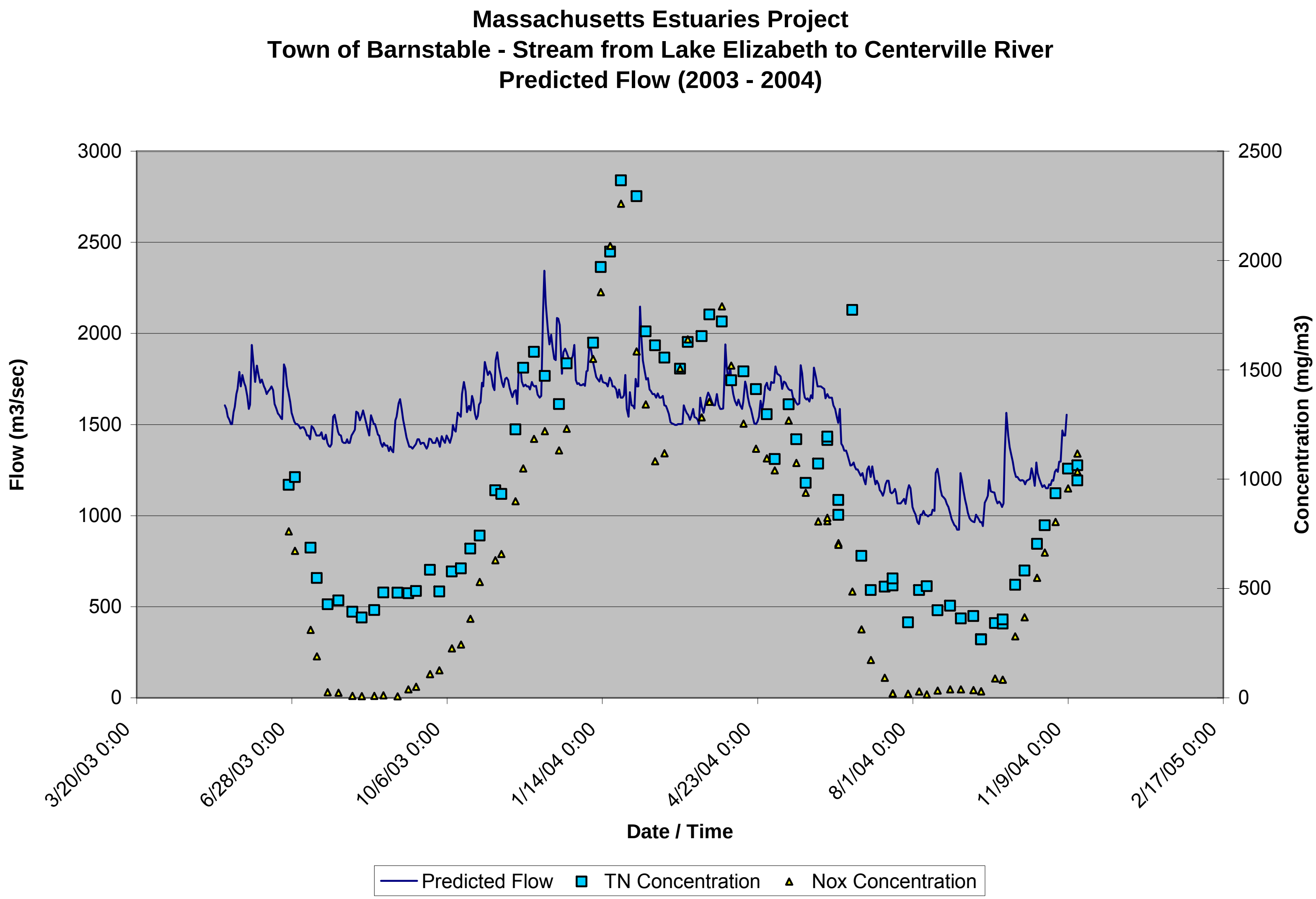


Figure IV-9. Stream discharge from Lake Elizabeth (solid blue line), nitrate+nitrite (yellow triangle) and total nitrogen (blue box) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Centerville River (Table IV-7).

Table IV-7. Summary of annual volumetric discharge and nitrogen load from the Rivers and Streams (freshwater) discharging to the Centerville River based upon the data presented in Figures IV-6 through IV-9 and Table IV-6.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Skunknett River (MEP)	**September 1, 2003 to August 31, 2004**	5391034	6043	8715
	September 1, 2004 to August 31, 2005	4774129	5271	6477
	Average 2003 - 2005	**5082582**	**5657**	**7596**
Skunknett River (SMAST)	**2002 - 2003**	4860705		
Skunknett River (CCC)	Based on Watershed Area and Recharge	4144021	--	--
Bumps River (Freshwater) MEP	**September 1, 2003 to August 31, 2004**	**2072872**	**4201**	**6173**
Bumps River (Freshwater) CCC	Based on Watershed Area and Recharge	1904385	--	--
Long Pond Stream Discharge (MEP)	**September 1, 2004 to August 31, 2005**	**2379169**	**473**	**1260**
Long Pond Stream Discharge (CCC)	Based on Watershed Area and Recharge	2282678	--	--
Lake Elizabeth Stream Discharge (MEP)	**September 1, 2003 to August 31, 2004**	**564615**	**529**	**830**
Lake Elizabeth Stream Discharge (CCC)	Based on Watershed Area and Recharge	503984	--	--

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

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

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

As stated in above sections, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the Centerville 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 watersheds it is predominantly in the bioavailable form nitrate. This nitrate and other bioavailable forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton “particles”. Most of these “particles” remain in the water column for sufficient time to be flushed out to a down gradient larger water body (like Vineyard/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. Also, in longer residence time systems (greater than 8 days) these nitrogen rich particles may die and settle to the bottom. In both cases (grazing or senescence), a fraction of the phytoplankton with their associated nitrogen “load” become incorporated into the surficial sediments of the bays.

In general the fraction of the phytoplankton population which enters the surficial sediments of a shallow embayment: (1) increases with decreased hydrodynamic flushing, (2) increases in low velocity settings, (3) increases within enclosed tributary basins, particularly if they are deeper than the adjacent embayment. 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 deep depositional basins or salt marsh tidal creeks, the sediments can be a net sink for nitrogen even during summer. These latter salt marsh channels and ponds can be seen in the Bumps River sub-system and the upper most reach of the Centerville River, which is essentially a tidal salt marsh. Failure to account for the nitrogen balance of the sediments generally results in significant errors in determination of threshold nitrogen loadings. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

IV.3.2 Method for determining sediment-watercolumn nitrogen exchange

For the Centerville River system, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected from 20 sites throughout the Centerville River Estuary (Figure IV-10) in August 2003. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample.

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

Centerville River Benthic Nutrient Regeneration Cores

- Station CVH-9 1 core (Lower Region)
- Station CVH-10 1 core (Lower Region)
- Station CVH-11 1 core (Lower Region)
- Station CVH-13 1 core (Lower Region)
- Station CVH-14 1 core (Lower Region)
- Station CVH-22 1 core (East Bay)
- Station CVH-23 1 core (East Bay)
- Station CVH-24 2 cores (East Bay)
- Station CVH-25 1 core (East Bay)
- Station CVH-26 1 core (East Bay)
- Station CVH-27 1 core (East Bay)
- Station CVH-28 1 core (Lower Region)
- Station CVH-29 1 core (Lower Region)

Bumps River/Scudder Bay Benthic Nutrient Regeneration Cores

- Station CVH-15 1 core (Lower Region)
- Station CVH-16 1 core (Lower Region)
- Station CVH-17 1 core (Upper Region)
- Station CVH-18/19 2 cores (Upper Region)
- Station CVH-20 1 core (Upper Region)
- Station CVH-21 1 core (Upper Region)

Note: there was no core CVH 12; cores CVH 1-4 & 5-8 were external to Centerville River.

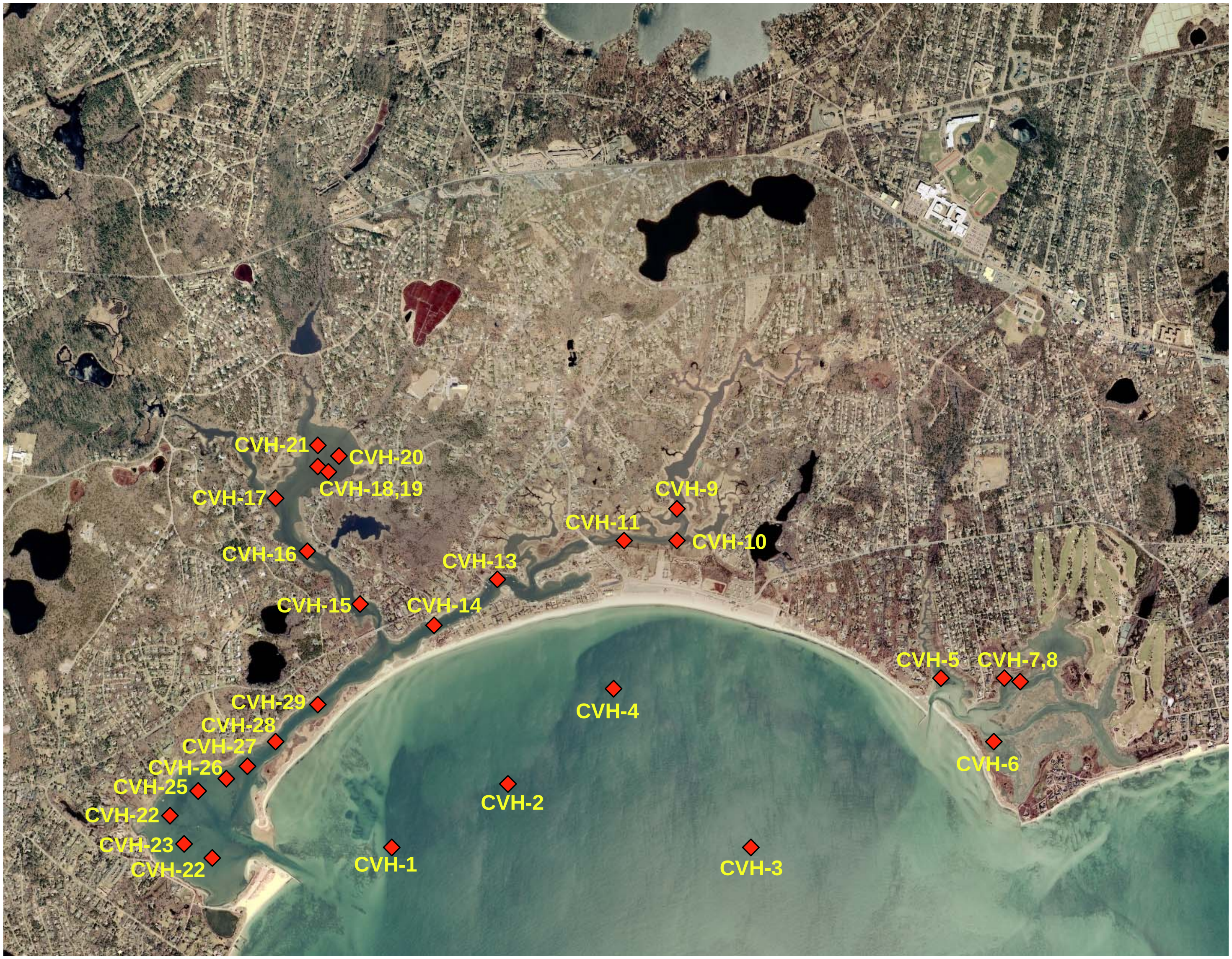


Figure IV-10. Centerville River embayment system sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above.

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

Sediment-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 (Harbormasters Office) 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-11).

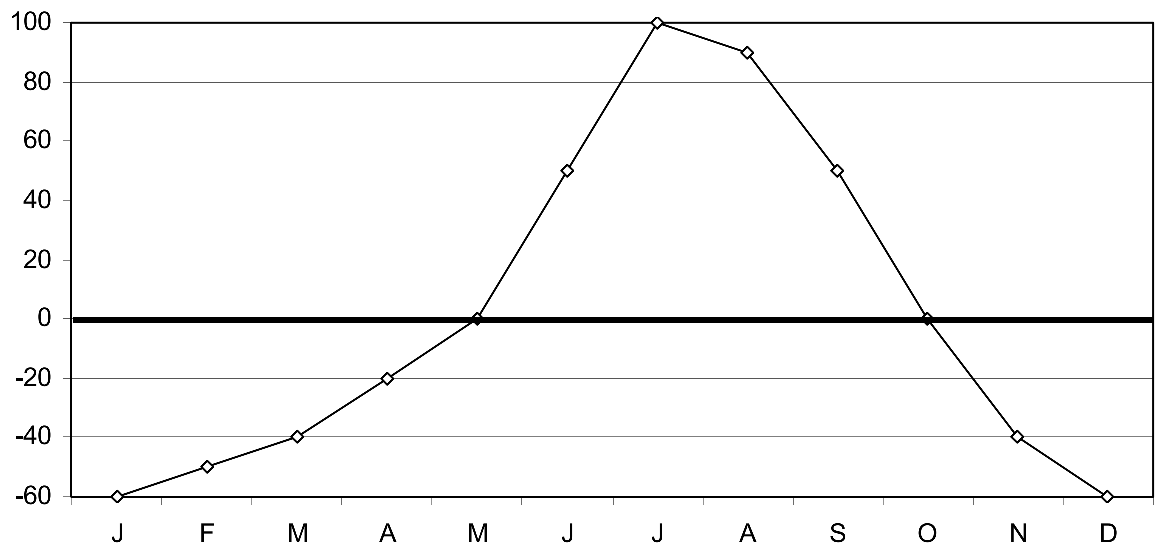


Figure IV-11. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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

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

Sediment sampling was conducted within the Centerville River Estuary in order to obtain the 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 core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content and sediment type and an analysis of each site’s tidal flow velocities. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within each sub-embayment.

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within each sediment site, 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 Centerville River System were comparable to other similar embayments with similar configuration and flushing rates. Overall, sediment nitrogen release was generally low or negative 36.6 to –13.2 mg N m^{-1} d^{-1}. Only the main depositional basin of East Bay showed a moderate level of net nitrogen release, 59 mg N m^{-1} d^{-1}. The rates of nitrogen release were much less than in heavily nitrogen loaded sub-embayments within the Pleasant Bay Estuary (~100 mg N m^{-1} d^{-1}), but very similar to the River (Upper Pleasant Bay) with its similar configuration (-10.9 to 34.2 mg N m^{-1} d^{-1}). The general pattern is consistent with other estuaries, with nitrogen loss from the sediments in the main depositional basin and from the Centerville River sediments, and net uptake in the shallow creeks and salt marsh basins is typical of southeastern Massachusetts estuaries. Net nitrogen uptake was observed in each of the salt marsh dominated sub-systems (-4.5 to -13.2 mg N m^{-1} d^{-1}), specifically the salt marsh areas within the Centerville River System, the upper reach of the

Centerville River which functions as a tidal salt marsh, the shallow Bumps River channel with fringing marshes, and the shallow basin of Scudder Bay which is bordered by tidal marshes and has the characteristics of a salt marsh pond. Net nitrogen uptake by estuarine basins influenced by salt marshes has been well documented within the MEP region.

Net nitrogen release rates for use in the water quality modeling effort for the component sub-basins of the Centerville River System (Chapter VI) are presented in Table IV-8. There was a clear spatial pattern of sediment nitrogen flux, with a strong gradient ranging from the more organic rich inner salt marsh basins showing uptake and the deeper lower reaches of the Centerville River showing net nitrogen release and the main depositional basin at the terminus of the River (East Bay) showing the highest rate of release. Even so, the sediments within the Centerville River System showed only a moderate range in nitrogen fluxes compared to other systems in the region and appear to be in balance with the overlying waters and the nitrogen flux rates consistent with the moderate nitrogen loading to this system and it relatively high flushing rate.

Table IV-8. Rates of net nitrogen return from sediments to the overlying waters of the Centerville River Estuarine System. 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	
Centerville River				
Upper Salt Marsh Reach	-4.7	0.4	3	CVH 9,10,11
Mid River Reach	32.3	3.6	2	CVH 13,14
Lower Reach	36.6	2.0	2	CVH 28,29
East Bay				
Bay/River mouth	-8.9	0.9	2	CVH 26,27
Main Basin	59.1	3.0	4	CVH 22,23,24,25
Bumps River Sub-estuary				
Scudder Bay	-13.2	0.2	4	CVH 18,19,20,21
Bumps River	-4.5	0.8	3	CVH 15,16,17
Station numbers refer to Figures IV-7.				

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

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 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.

Estuarine water quality is dependent upon nutrient and pollutant loading and the processes that help flush nutrients and pollutants from the estuary (e.g., tides and biological processes). Relatively low nutrient and pollutant loading and efficient tidal flushing are indicators of high water quality. The ability of an estuary to flush nutrients and pollutants is proportional to the volume of water exchanged with a high quality water body (i.e. Nantucket Sound). Several embayment-specific parameters influence tidal flushing and the associated residence time of water within an estuary. For the Centerville River system, the most important parameters are the tide attenuation along with the shape, length and depth of the estuary and its attached sub-systems.

Shallow coastal embayments 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.

To understand the dynamics of the Centerville River, a hydrodynamic study was performed. The system is located in the Village of Centerville along the south coast of Cape Cod, Massachusetts. A site map showing the general study area is shown in Figure V-1. The system consists of East Bay which is the entrance to the system from Nantucket Sound, Centerville River, Bumps River, which attaches to Scudder Bay, as well as numerous other smaller coves, creeks, and marshes. It is relatively shallow on average, the exceptions being deeper channel that provides a path between the Nantucket Sound and the river. The approximate tidal range within the system is 4 feet, with Nantucket Sound tidal variations providing the hydraulic forcing that drives water movement throughout the system.

Centerville River is a shallow tidal estuary, with a mean water depth of only 2.5 feet. Although Centerville River contains a larger overall area of salt marsh compared to other systems (approximately 230 acres), this marsh area only accounts for 37 percent of the estuary surface area.

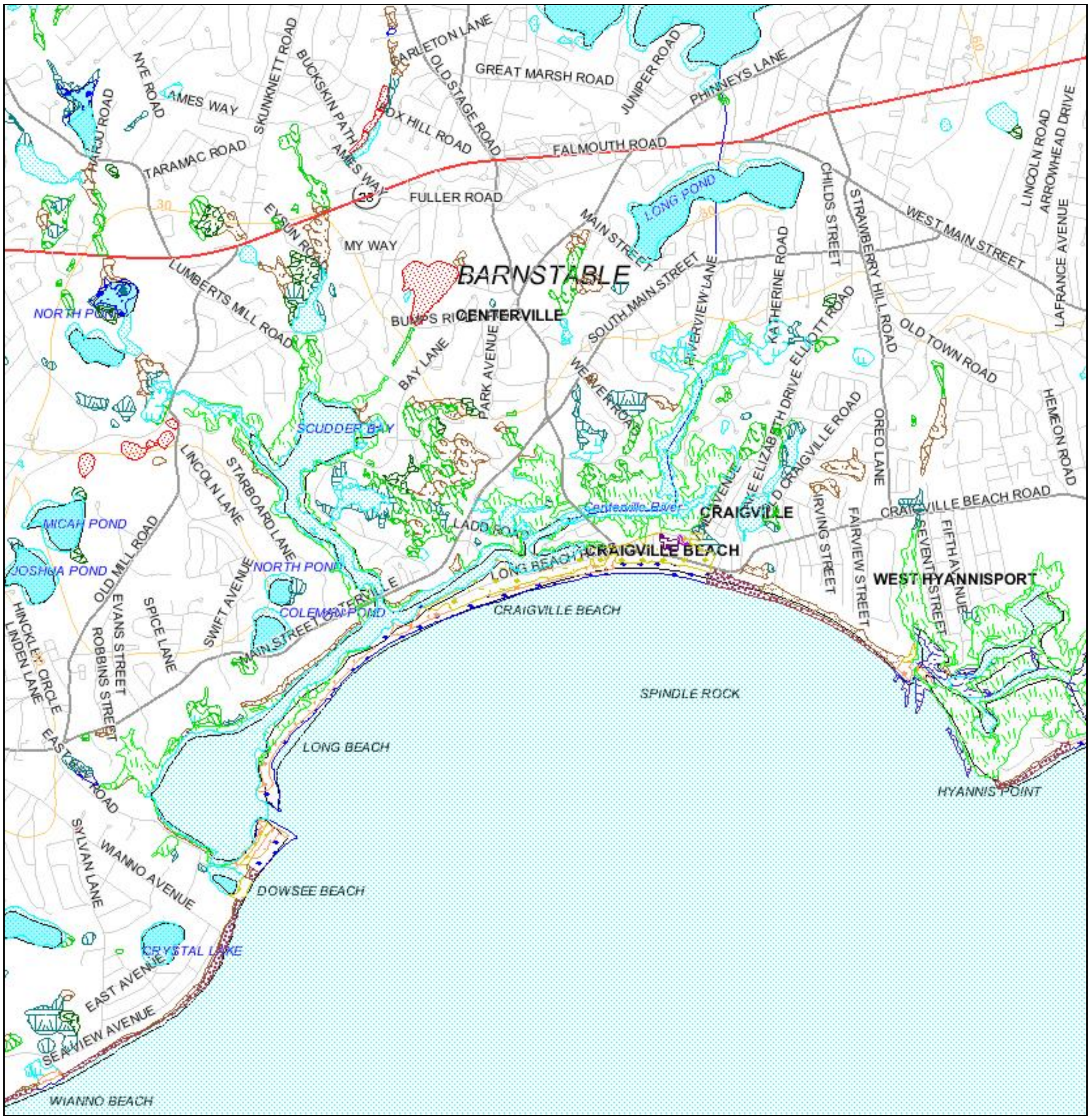


Figure V-1. Map of the Centerville River estuary (from Massachusetts Office of Geographic and Environmental Information).

Circulation in the Centerville River system is dominated by tidal exchange with Nantucket Sound. From measurements made in the course of this study, the average tide range at the entrance to Centerville River is approximately 2.66 feet. Flow restrictions caused by narrow channels, bridge abutments, and frictions losses, reduce the tide range in upper Centerville River to approximately 2.52 feet. Freshwater inflow is relatively small in comparison to tidal waters entering through the inlets, and does not have a significant impact on the hydraulic response of the system.

The hydrodynamic study consisted of two major components. In the first portion of the study, bathymetry, Acoustic Doppler Current Profile (ADCP) measurements, and tide data were collected in order to accurately characterize the physical system, and to provide data necessary for the hydrodynamic modeling portion of the study. The bathymetry survey of Centerville River was performed to determine the variation of embayment and channel depths throughout the system. This survey addressed the previous lack of adequate bathymetry data for this area. In addition to the survey, tides were recorded for 44 days at four locations within Centerville River and at an offshore gage. This tide data were necessary to run and calibrate the hydrodynamic model of the system.

A numerical hydrodynamic model of the Centerville River system was developed in the second portion of this study. Using the bathymetry survey data, a finite element model grid was generated for use with the RMA-2 hydrodynamic code. The tide data from the offshore gage was used to define the open boundary condition that drives the circulation of the model, and data from the five locations within the system were used to calibrate and verify model performance to ensure that it accurately represents the dynamics of the real, physical system. In addition to the calibration process, the ADCP current measurements supplied the data needed as an independent verification of the hydrodynamic model results.

The calibrated computer model of the Centerville River system was used to compute the flushing rates of each of the sub-embayments of the system. Though water quality in an embayment cannot be directly inferred by use of the computed flushing rate alone, it can serve as a useful indicator of an embayments flushing performance relative to other similar systems. 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 water quality dynamics of a system.

For this system, the final calibrated model offers an understanding of water movement through the estuary, and provides the first step towards evaluating the water quality of these estuarine systems, as well as understanding nitrogen loading “thresholds” for each system. Tidal flushing information will be utilized as the basis for a quantitative evaluation of water quality. Nutrient loading data combined with measured environmental parameters within the various sub-embayments become the basis for an advanced water quality model based on total nitrogen concentrations. This type of model provides a tool for evaluating existing estuarine water quality, as well as determining the likely positive impacts of various alternatives for improving overall estuarine health, enabling the bordering towns to understand how pollutant loadings into the estuary will affect the biochemical environment and its ability to sustain a healthy marine habitat.

V.2 GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE ESTUARINE SYSTEM

The southern coast of Cape Cod in the vicinity of Centerville River is a relatively quiescent region (Figure V-2). The only exposure to waves directly from the Atlantic Ocean is through the 4 mile wide opening of Vineyard Sound to the southwest, the 10 mile wide Nantucket Sound opening to the east (between Monomoy Point and Nantucket Island) and the 7 mile gap of Muskeget Channel to the south. As a result of this sheltering, the study area is mostly limited to exposure to short period waves which are locally generated. Although natural wave and tidal forces continue to reshape the shoreline, day-to-day conditions have limited impact on the shoreline migration and/or inlet stability. For typical wave conditions, longshore transport of sand is from west-to-east along the south coast of Barnstable, due primarily to the local wind-driven waves. Tidal currents within Nantucket Sound flood to the northeast and ebb to the southwest.

Figure V-2. Aerial photograph of the Centerville River.

In contrast to the mild day-to-day conditions, infrequent hurricane events such as the hurricanes of 1938, 1944, and 1954, as well as Hurricane Bob in 1991, all caused significant overwash and transport of beach sediments. The 1944 hurricane broke through the beach opposite the entrance to Bumps River. This breach closed naturally over a two year period. (USACE, 1977) In addition to the hurricane events in the region, northeast storm events (causing waves to approach the shoreline from the east and southeast) create a sediment transport reversal from typical conditions, where the longshore sediment transport is generally from east-to-west for a short time.

The single most dominant factor for shoreline change in the study area is the construction of the 400 foot long stone jetty which defines the western edge of the entrance to East Bay and also serves as the eastern edge of Dowses Beach. The history of the jetty construction is unclear, although shoreline maps from the U.S. Coast and Geodetic Survey show that the jetty was constructed sometime between 1893 and 1939 (Figure V-3). The accretion of Dowses Beach and accompanying offset of Long Beach clearly reveal that the construction of the west jetty occurred at some point after the first map but before the second.

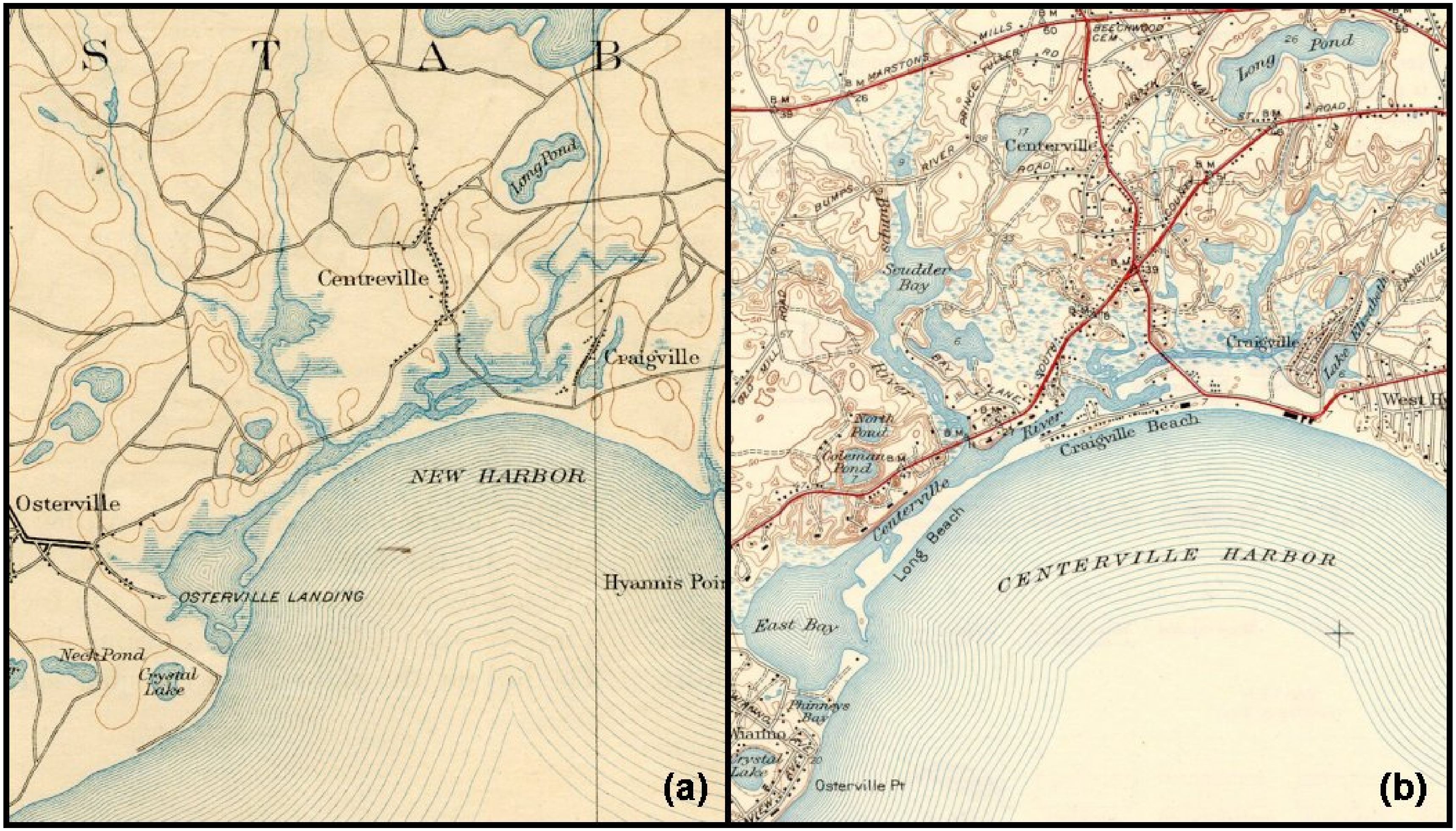


Figure V-3. Historical maps of the Centerville River study area from (a) 1893 and (b) 1939.

The new jetty served to impound a majority of the sand which would normally have continued to the east onto Long Beach. This trapping of sand by the jetty, and to a lesser extent the inlet itself, starved the downdrift section of shoreline, leading to the erosion and retreat of Long Beach. Dowses Beach is shown in Figure V-4, both in 1971 and 2005. The position of the shoreline with respect to the jetty looks to have advanced only a small bit seaward, suggesting that the jetty was close to being filled to impoundment in 1971. In this condition, sediment is no longer interrupted by the structure itself, but is bypassed into the inlet and carried to the flood shoals inside East Bay or offshore. Some of this sand continues to Long Beach, while some is likely lost to deep water and not returned to the nearshore system.

The entrance channel to East Bay was dredged in 1971 and about 30,000 cubic yards of the material was placed on the west end of Dowses Beach where there had been erosion issues. About 20,000 cubic yards of material 1971 dredging was placed on Long Beach to the east. This small amount of sand likely provided only a small benefit to Long Beach, which has experienced a retreat of well over 100 feet from its pre-jetty location. In addition to nourishment related to navigational dredging, residents have nourished Long Beach using private funds.

V.3 FIELD DATA COLLECTION AND ANALYSIS

Accurate modeling of system hydrodynamics is dependent upon measured conditions within the estuary for two important reasons. To accurately define the system geometry and boundary conditions for the numerical model and to provide ‘real’ observations of hydrodynamic behavior to calibrate and verify the model results.

To support hydrodynamic and future water quality modeling efforts in Centerville River, bathymetry, tidal currents, and water elevation variations of the embayments were measured. Cross-channel current measurements were surveyed through a complete tidal cycle at two locations in Centerville River.

The system geometry was defined using the bathymetry data collected and aerial photographs. The bathymetric data collection effort was focused on areas of flow constrictions, near inlets, channels, and narrow sections of the estuaries. The bathymetric information was utilized to develop the computational grid of the system geometry for the hydrodynamic modeling effort. Tidal elevation measurements within the embayment were used for both forcing conditions and to evaluate tidal attenuation through the estuarine system.

V.3.1 Data Acquisition

V.3.1.1 Water Elevation

Changes in water surface elevation were measured using internal recording tide gages. These tide gages were installed on fixed platforms (such as pier pilings or screw anchors secured to the seabed) to record changes in water pressure over time. Variations in the water surface can be due to tides, wind set-up, or other low frequency oscillations of the sea surface. The tide gages were installed in 5 locations in Centerville River estuary (Figure V-5) in April 2004 and recovered mid-May 2004. Data records span at least 29 days to yield an adequate time period for resolving the primary tidal constituents.

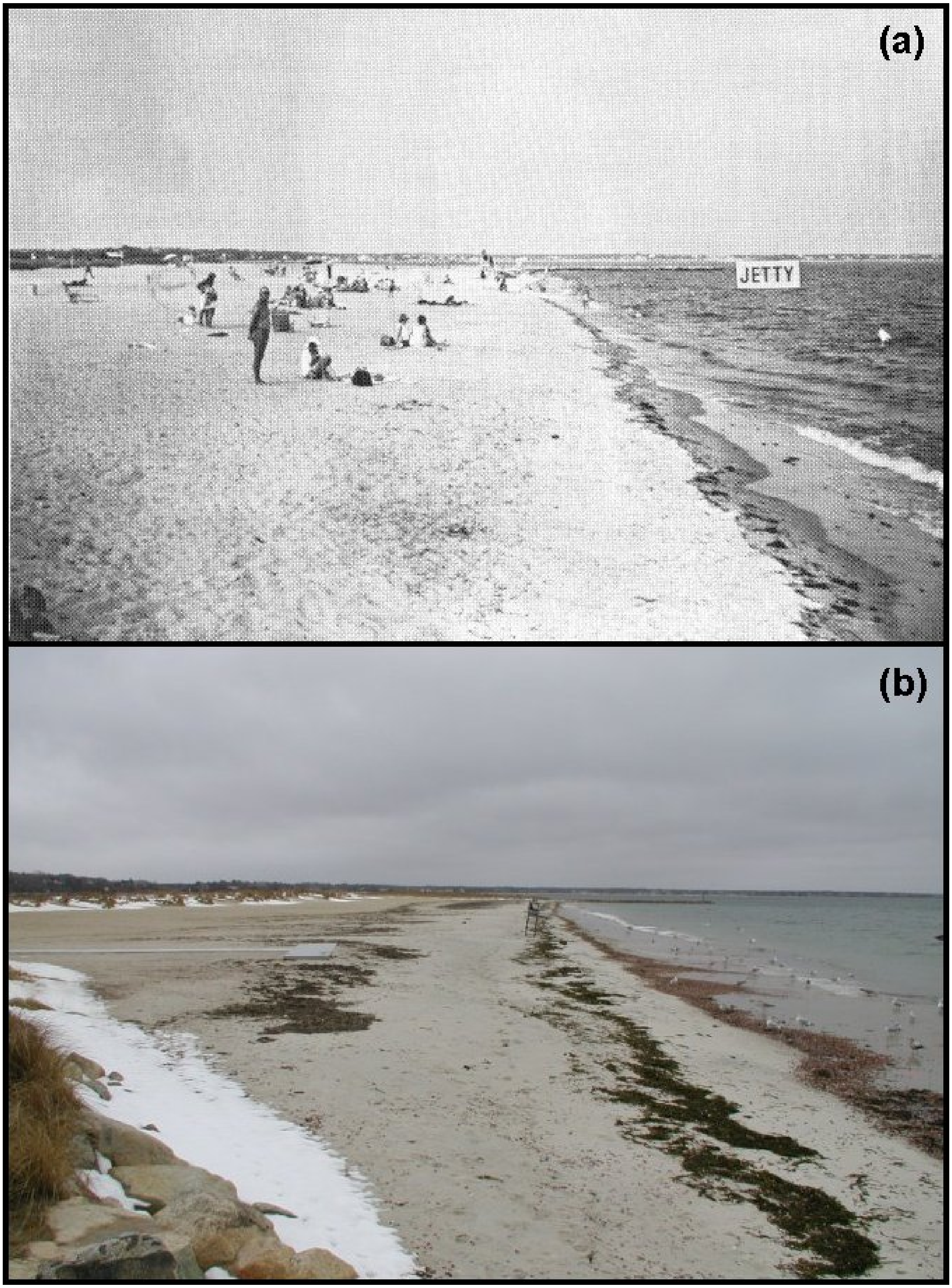


Figure V-4. Photographs of Dowses Beach looking to the east towards the jetty in (a) 1971 and (b) 2005.

The tide gages used for the study were Brancker XR-420 instruments. Data collections parameters were set for 10-minute intervals, with each 10-minute observation resulting from a 16-second burst of measurements that are averaged for each observation. Each of these instruments use strain gage transducers to sense variations in pressure, with resolution of 0.001% full scale and a pressure accuracy of 0.01% full scale. Each gage was calibrated prior to installation to assure accuracy.

Once the data were downloaded from each instrument, the water pressure readings were corrected for variations in atmospheric pressure. Hourly atmospheric readings were obtained from the NOAA buoy in Buzzards Bay (site BUZM3), interpolated to 10-minute intervals, and subtracted from the pressure readings, resulting in water pressure above the instrument. Further, a (constant) water density value of 1025 kg/m^3 was applied to the readings to convert from pressure units (psi) to head units (i.e., feet of water above the tide gage). Several of the sensors were surveyed into local benchmarks to provide vertical rectification of the water level; these survey values were used to adjust the water surface to a known vertical datum. The result from each gage is a time series representing the variations in water surface elevation relative to NAVD88. Figure V-6 present the water levels at each gage location.

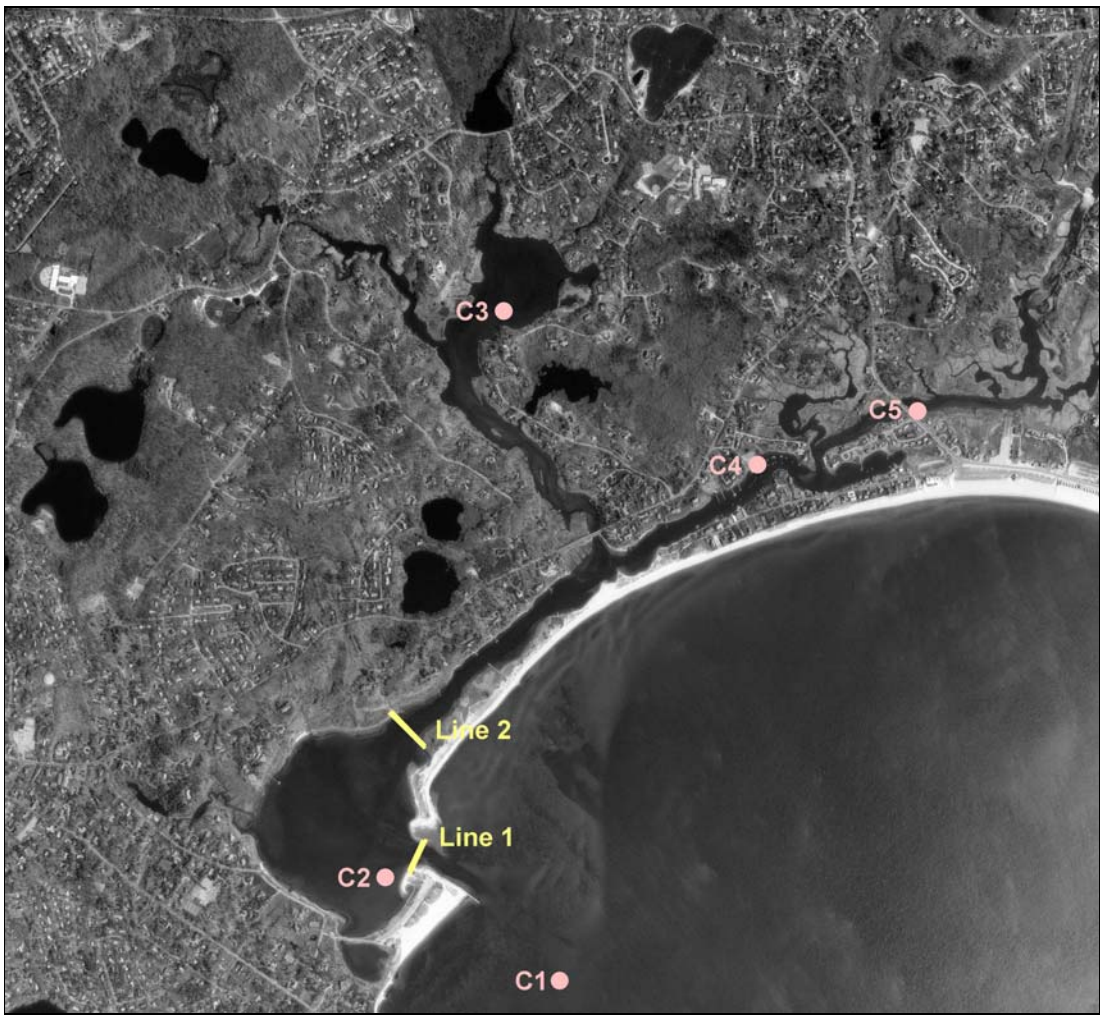


Figure V-5. Tide gage and ADCP transect locations in Centerville River C1 to C5 are tide gage locations. Yellow lines 1 and 2 are ADCP transect locations.

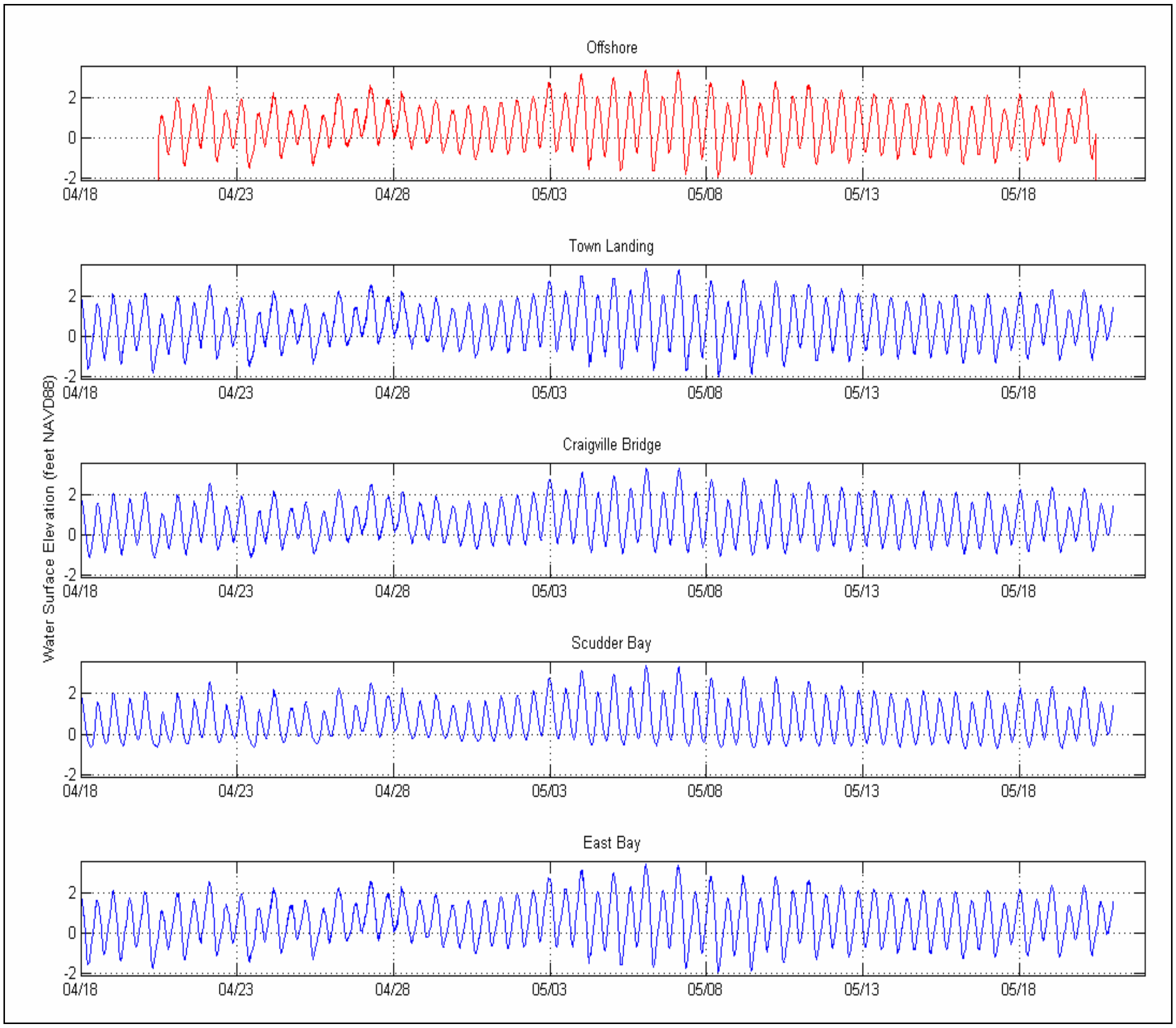


Figure V-6. Tidal elevation observations for Nantucket Sound (C1 of Figure V-5), Town Landing (location C4), Craigville Beach Bridge (location C5), Scudder Bay (location C3), and East Bay (location C2).

V.3.1.2 Bathymetry

The bathymetric, or depth, survey of Centerville River was conducted by Applied Coastal in October 2003. The surveys were completed using a small vessel equipped with a precision fathometer interfaced to a differential GPS receiver. The fathometer has a depth resolution of approximately 0.1 foot and the differential GPS provides x-y position measurements accurate to approximately 1-3 feet. Digital data output from both the echo sounder and GPS were logged to a laptop computer in Hypack.

GPS positions and echo sounder measurements were merged to produce data sets consisting of water depth as a function of x-y horizontal position (in Massachusetts Mainland State Plane, 1983). The data were combined with water surface elevations to obtain the vertical elevation of the bottom (z) relative to the NAVD 1988 vertical datum (NAVD88). The resulting xyz files (Figure V-7) were input to mapping software to calculate depth contours for the system shown in Figure V-8.

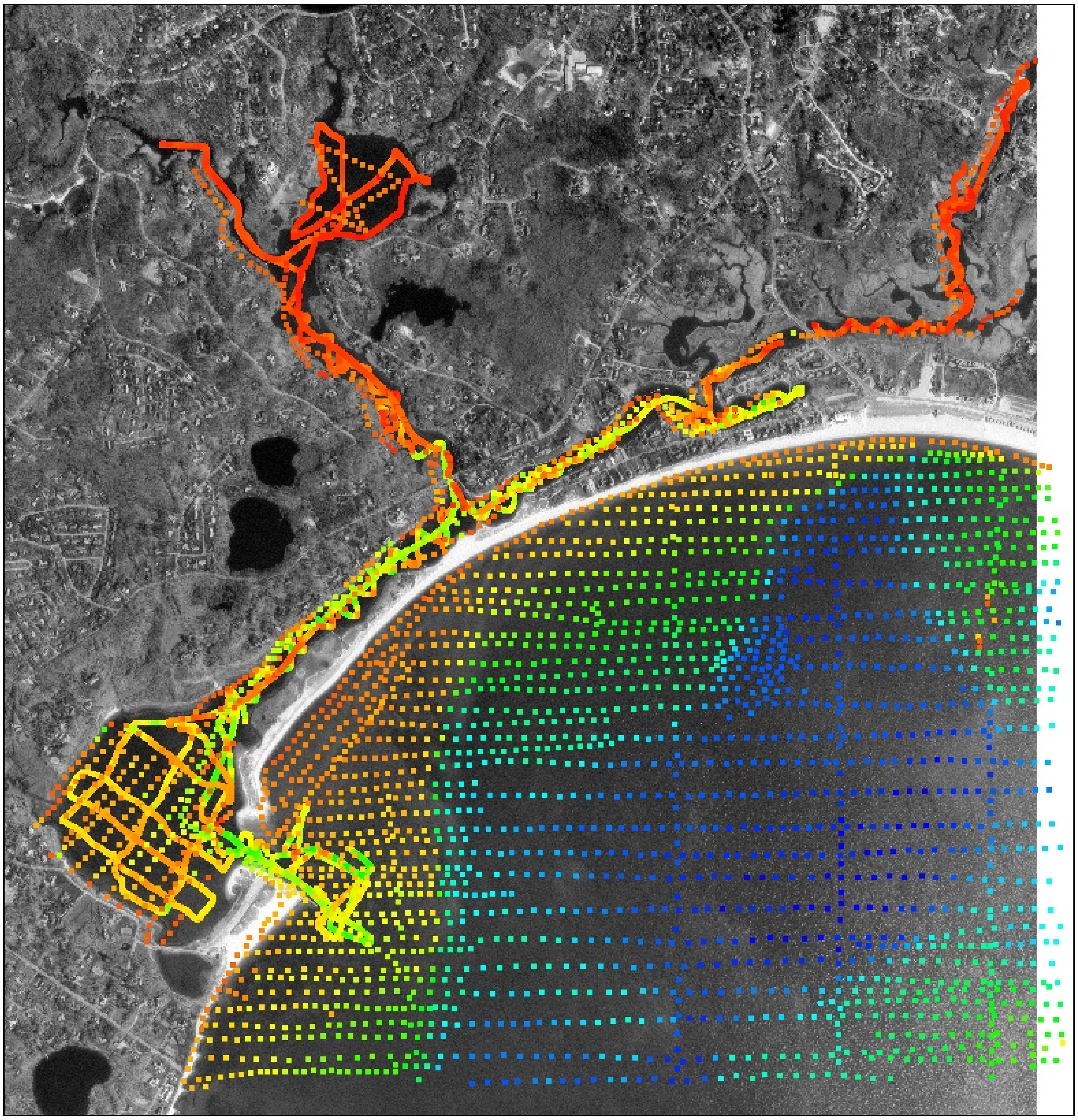


Figure V-7. Bathymetry points collected within Centerville River. Color indicates water depth relative to the NAVD88 vertical datum.

V.3.1.3 Current Measurements

The measurements were collected using an Acoustic Doppler Current Profiler (ADCP) mounted aboard a small survey vessel. The boat repeatedly navigated a pre-defined set of transect lines through the area, approximately every 30 minutes, with the ADCP continuously collecting current profiles. This pattern was repeated for approximately 10-hours to ensure measurements over the entire tidal cycle. The results of the data collection effort are high-resolution observations of the spatial and temporal variations in tidal current patterns throughout the survey area.

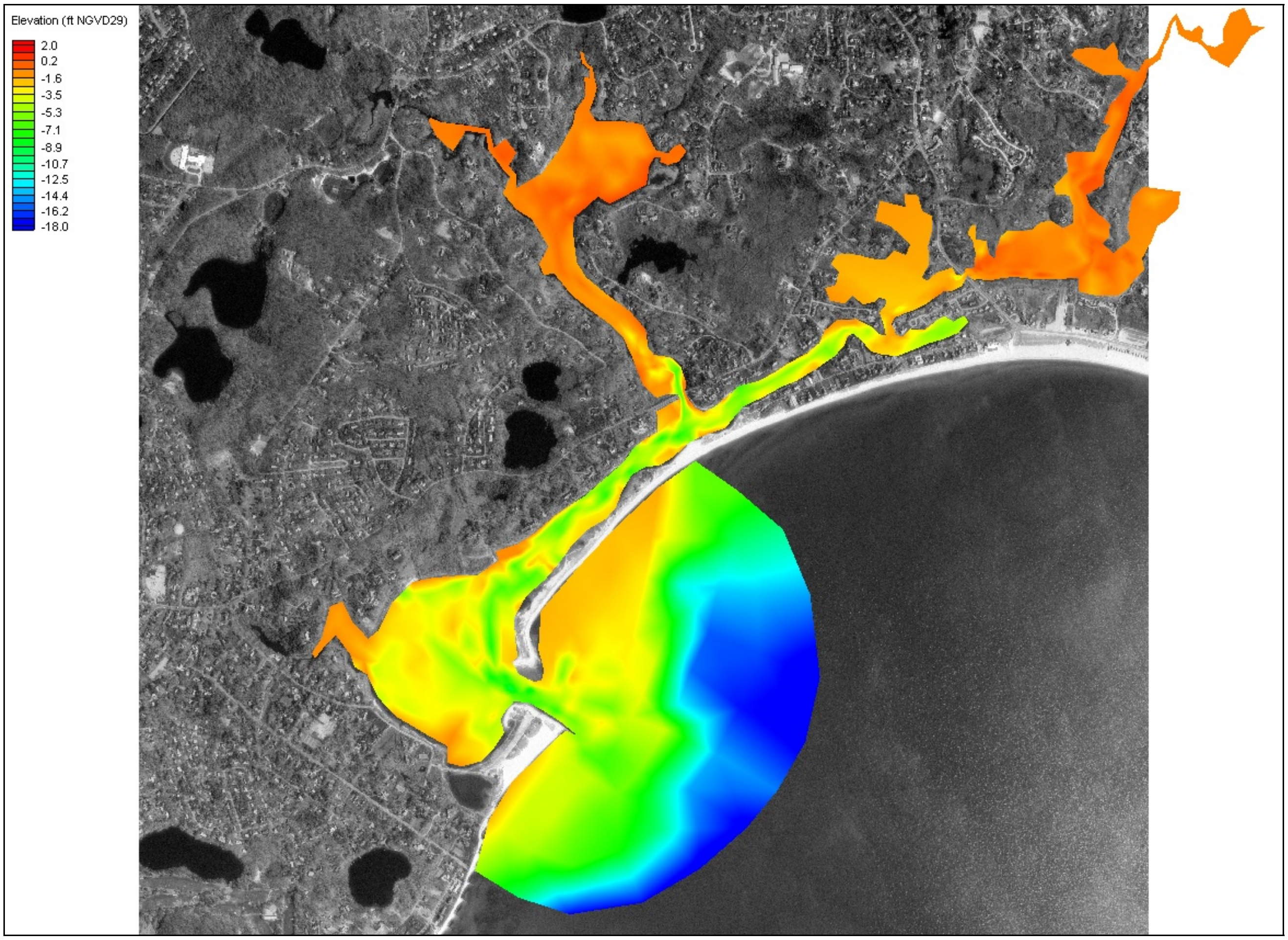


Figure V-8. Bathymetric map of Centerville River. Color indicates water depth relative to the NAVD88 vertical datum.

Measurements were obtained with a BroadBand 1200 kHz Acoustic Doppler Current Profiler (ADCP) manufactured by RD Instruments (RDI) of San Diego, CA. The ADCP was mounted to a specially constructed mast, which was rigidly attached to the rail of the survey vessel. The ADCP was oriented to look downward into the water column, with the sensors located approximately 1 foot below the water surface. The mounting technique assured no flow disturbance due to vessel wake.

The ADCP emits individual acoustic pulses from four angled transducers (at 20° from the vertical) in the instrument. The instrument then listens to the backscattered echoes from discrete depth layers in the water column. The difference in time between the emitted pulses and the returned echoes, reflected from ambient particles (plankton, debris, sediment, etc.), is the time delay. BroadBand ADCPs measure the change in travel times from successive pulses. As particles move further away from the transducers sound takes longer to travel back and forth. The change in travel time, or propagation delay, corresponds to a change in distance between the transducer and the sound scatter, due to a Doppler shift. The propagation delay, the time lag between emitted pulses, and the speed of sound in water are used to compute the velocity of the particle relative to the transducer. By combining the velocity components for at least three of the four directional beams, the current velocities are transformed using the unit's internal compass readings to an orthogonal earth coordinate system in terms of east, north, and vertical components of current velocity.

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

The collection of accurate current data with an ADCP requires the removal of the speed of the transducer (mounted to the vessel) from the estimates of current velocity. ‘Bottom tracking’ is the strongest echo return from the emission of an additional, longer pulse to simultaneously measure the velocity of the transducer relative to the bottom. Bottom tracking allows the ADCP to record absolute versus relative velocities beneath the transducer. In addition, the accuracy of the current measurements can be compromised by random errors (or noise) inherent to this technique. Improvements in the accuracy of the measurement for each bin are achieved by averaging several velocity measurements together in time. These averaged results are termed ‘ensembles’; the more pings used in the average, the lower the standard deviation of the random error.

For this study, the standard deviation (or accuracy) of current estimates (resulting from an ensemble average of 8 individual pulses) was approximately 0.30 ft/sec. Each ensemble took approximately 5-6 seconds to collect. Averaging parameters resulted in a horizontal resolution of approximately 10 feet along the transect line. For example, ADCP transect 1 (Figure V-5) at the inlet to Centerville River was approximately 250 feet across, resulting in approximately 20 to 25 independent velocity profiles per transect. The vertical resolution was set to 0.79 ft, or one velocity observation per every 9.48 inches of water depth. The first measurement bin was centered 1.77 feet from the surface, allowing for the transducer draft as well as an appropriate blanking distance between the transducer and the first measurement.

Position information was collected by Hypack, an integrated navigation software package running on a PC computer, linked to a differential GPS. The position data were read from the device in the WGS-84 coordinate system, and transformed to NAD 1983 Massachusetts Mainland State Plane coordinates. Position updates were available every 1 second. Clock synchronization between the GPS and ADCP laptop computers allowed each ADCP ensemble to be assigned an accurate GPS position during post-processing.

Current measurements were collected by the ADCP as the vessel navigated repeatedly a series of two (2) pre-defined transect lines in Centerville River (Figure V-5). The line-cycles were repeated every half-hour throughout the survey. The first cycle was begun at 07:29 hours (Eastern Daylight Time, EDT) and the final cycle was completed at 17:36 hours (EDT), for a survey duration of approximately 10 hours on October 24, 2003.

The transect lines 1 and 2 were run in ascending order. These lines were designed to measure as accurately as possible the volume flux through the constrictions during a complete tidal cycle. Line 1 ran across the throat of Centerville River Inlet in a west-to-east direction. Line 2 ran south-to-north across inlet to East Bay along Centerville River along the eastern shore of East Bay.

V.3.2 ADCP Data Processing Techniques

Data processing consisted of the following:

- Convert raw ADCP (binary) files to ASCII data
- Merge ADCP vertical profile data with GPS position data
- QA/QC procedures to verify the accuracy of both ADCP and position data
- Manipulate the ADCP data to calculate spatial averages and cross section discharge values

The data files were converted from raw binary format to ASCII values using RDI's BBLIST conversion program. The command set for this conversion process is described in greater detail in the RDI ADCP manual, and consists of developing a user-defined output file format, through which all conversions are defined.

The output data file from this procedure consists of multiple ensemble data 'packets'. The ensemble 'packet' consists of a single line containing the time of the profile, the ensemble number, and the measured water temperature (measured by the ADCP's internal temperature sensor) followed by consecutive rows and columns of the profile data. Each row of profile data corresponds to one bin, or depth layer, with succeeding columns representing east and north components of velocity, error velocity, speed, direction, echo amplitudes (for 4 beams), and correlation magnitudes (for 4 beams). Each ensemble, collected approximately every 5-6 seconds, has 30 rows corresponding to each discrete depth layer, starting at 1.77 feet. A single data file consists of multiple ensembles, as few as 25-30 to as many as 100. A single data file was recorded for each transect.

The next step in the processing was the assignment of an accurate x-y position pair to each ensemble. This was accomplished using the time stamp of both the ADCP data file and the position data file. Prior to the survey, the clocks used for each system were synchronized to assure this operation was valid. The procedure finds the time of each ADCP ensemble, then searches the position data file for the nearest corresponding time. When the nearest time is found, subject to a 'neighborhood' limit of 1 second, the x-y pair for that time is assigned to the ADCP ensemble. This method produces some inaccuracies; however for this survey the error in position definition was less than approximately 3.5 feet (calculated as vessel speed of 2 knots times the neighborhood value of 1 second for this survey). If no time is found within 1 second of the ADCP time, then a position is calculated using the ADCP bottom track velocity for that ensemble, and the time interval between ensembles.

Once each ensemble was assigned a valid x-y position, the data were reduced to calculate vertical averages as well as total discharge. A mean value of each east and north component of velocity is calculated for each vertical profile. These component mean values are then used to determine the mean speed and mean direction.

The total discharge time series represents the total volumetric flow through a waterway cross-section over the duration of the tidal cycle. Discharge calculations were performed on velocity components normal and tangential to the transect azimuth, which in most cases was perpendicular to the channel axis. To determine accurately the discharge normal to the channel cross-section (i.e. along-stream), the east and north velocity components were rotated into normal (along-stream) and tangential (cross-stream) components. Only the along-stream component was used to calculate total discharge.

The discharge through a cross section, Q_t, is the product of the upstream velocity, $V_{upstream}$, multiplied by the cross sectional area, A_{cs}, or

$$\Sigma Q_t = \Sigma_{i=1...N} (V_{upstream}*A_{cs})$$

where the cross sectional area is the water depth times the lateral (cross-stream) distance from the previous ensemble profile. The summation occurs over i, where i represents each individual ensemble profile from 1 to N, with 1 representing the top (surface) bin and N representing the deepest (near-bottom) bin.

Data recorded for the bottom-most bins in the water column can be contaminated by side lobe reflections from the transducer. At times, the measurements can be invalid. Validity of the bottom bin measurements is determined by comparing the standard deviation of bottom values to the standard deviation of mid-column measurements. If the standard deviation at the bottom was more than twice the standard deviation of mid-column measurements, the bottom bin was discarded from the discharge calculation. If the bottom value was within the limits defined by adjacent measurements, the value was included in the calculation.

The total discharge calculations assume a linear extrapolation of velocity from the surface to the first measurement bin. Since the ADCP cannot directly measure the surface velocity, it is assumed the surface layer discharge is equivalent to the discharge in the first depth layer. The same linear assumption was applied to bottom bins when the bin measurement was declared invalid; that is, the bottom bin value was assumed equivalent to the overlying bin velocity value.

V.3.3 Discussion of Results

V.3.3.1 Tidal Harmonic Analysis

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

Figure V-6 shows the tidal elevation for the period April 18 through May 21, 2004 at five locations in Centerville River: Offshore Centerville River in Nantucket Sound (Location C1), East Bay (Location C2), Scudder Bay (location C3), Centerville River at the town landing (Location C4), and Centerville River on the eastern side of Craigville Beach Bridge (Location C5). The curves have a predominant 12.42-hour variation around the lunar semi-diurnal (twice-a-day), or M_2, tidal constituent. There was also a strong modulation of the lunar and solar tides, resulting in the familiar spring-neap fortnightly cycle. The spring (maximum) tide range was approximately 4.5 feet, and occurred on April 26. The neap (or minimum) tide range was 1.9 feet, occurring May 5.

To better quantify the changes to the tide from the inlet to inside the system, the standard tide datums were computed from the 29-day records. These datums are presented in Table V-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 average of the MHW and MLW.

Table V-1. Tide datums computed from 29-day records collected in Centerville River in April/May 2004. Datum elevations are given relative to NAVD 88.

Tide Datum	Offshore (feet)	East Bay (feet)	Scudder Bay (feet)	Centerville River-Town Landing (feet)	Centerville River-Craigville Beach (feet)
Maximum Tide	3.379	3.427	3.379	3.355	3.325
MHHW	2.376	2.396	2.375	2.371	2.373
MHW	1.931	2.012	1.996	1.989	2.012
MTL	0.600	0.604	0.829	0.586	0.750
MLW	-0.732	-0.803	-0.338	-0.817	-0.512
MLLW	-1.194	-1.169	-0.500	-1.201	-0.799
Minimum Tide	-1.952	-1.939	-0.734	-1.987	-1.103

Harmonic analyses were performed on the time series from each gage location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-2 presents the amplitudes of the eight largest tidal constituents. The M_2, or the familiar twice-a-day lunar semi-diurnal, tide is the strongest contributor to the signal with an amplitude of 1.6 feet in Nantucket Sound. The range of the M_2 tide is twice the amplitude, or 3.2 feet. The diurnal tides, K_1 and O_1, possess amplitudes of approximately 0.3 feet and 0.2 feet respectively, throughout the Centerville River system. Other semi-diurnal tides strongly contribute to the observed tide; the S_2 (12.00 hour period) and N_2 (12.66-hour period) tides both have amplitudes of 0.4 feet offshore of Centerville River.

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

Table V-2 also shows how the constituents vary as the tide propagates into the upper reaches of the two tidal rivers. Note the reduction in the M_2 amplitude from Nantucket Sound to the inlets, and the further reduction at the upper portions of Centerville and Little Rivers. The amplitude reduction is greatest at the upper reaches of Little River, where the M_2 amplitude is 0.33 feet smaller than offshore. The decrease in the amplitude of M_2 constituent is evidence of frictional damping. Usually, a portion of the energy lost from the M_2 tide is transferred to higher harmonics (i.e., the M_4 and M_6), and is observed as an increase in amplitude of these constituents over the length of an estuary. This effect is observed in the analysis of the Centerville River tides, where a maximum 0.1 ft increase occurs in the M_4.

Table V-3 presents the phase delay of the M_2 tide at all tide gage locations compared to the offshore gage in Nantucket Sound. Phase delay is another indication of tidal damping, and results with a later high tide at inland locations. The greater the frictional effects, the longer the delay between locations.

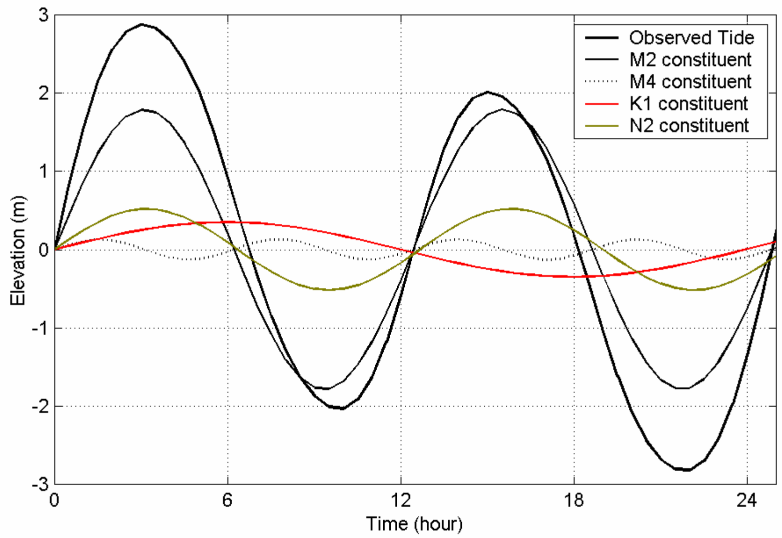


Figure V-9. Example of an observed astronomical tide as the sum of its primary constituents.

Table V-2. Tidal Constituents, Centerville River, April-May 2004.								
	AMPLITUDE (feet)							
	M2	M4	M6	S2	N2	K1	O1	Msf
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Offshore	1.59	0.18	0.01	0.41	0.44	0.24	0.18	0.07
East Bay	1.48	0.20	0.02	0.35	0.41	0.25	0.17	0.04
Scudder Bay	1.34	0.28	0.03	0.28	0.39	0.24	0.16	0.12
Centerville River-Town Landing	1.47	0.24	0.02	0.35	0.40	0.25	0.17	0.09
Centerville River-Craigville Beach	1.26	0.26	0.03	0.27	0.36	0.25	0.15	0.14

Table V-3. M_2 Tidal Attenuation, Centerville River, April-May 2004 (Delay in minutes relative to Offshore).	
Location	Delay (minutes)
Offshore	--
East Bay	21.45
Scudder Bay	30.80
Centerville River -Town Landing	38.73
Centerville River -Craigville Beach	56.74

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. This analysis calculated the energy (or variance) of the original water elevation time series, and compared these energy values to that of the purely tidal signal (re-created by summing the contributions from the 23 known harmonic constituents). Subtracting the tidal signal from the original elevation time series resulted with the non-tidal, or residual, portion of the water elevation changes. The energy of this non-tidal signal is compared to the tidal signal, and yields a quantitative measure of how important these non-tidal physical processes can be to hydrodynamic circulation within the estuary. The results of this analysis for the Centerville River region are posted in Table V-4.

Table V-4. Percentages of Tidal versus Non-Tidal Energy, Centerville River, April to May 2004.

	Total Variance ($ft^2 \cdot sec$)	Tidal (%)	Non-tidal (%)
Offshore	1.55	92.4	7.6
East Bay	1.36	91.8	8.2
Scudder Bay	1.14	90.9	9.1
Centerville River - Town Landing	1.37	91.0	9.0
Centerville River - Craigville Beach	1.04	89.3	10.7

Table V-4 shows that the percentage of tidal energy was largest in the offshore signal in Nantucket Sound; as should be expected given the tidal attenuation through the system. In general, the energy of the signal decreases with distance from the offshore gage, with the lowest energy found in upper regions of the estuarine systems. The analysis also shows that tides are responsible for approximately 90% of the water level changes in Centerville River. Meteorological effects in this data set were significant (approximately 10%) contributors to the total observed water level changes. However, the change in the non-tidal variance from offshore to the systems' upper reaches (approximately 3%) indicates that the offshore tide is adequate for use as the forcing time series of the computer hydrodynamic model of these systems. This relative increase in non-tidal energy within this system is likely due to the decrease in tidal energy as a result of frictional forces rather than actual growth of residual forces. The damping can be seen in Figure V-10.

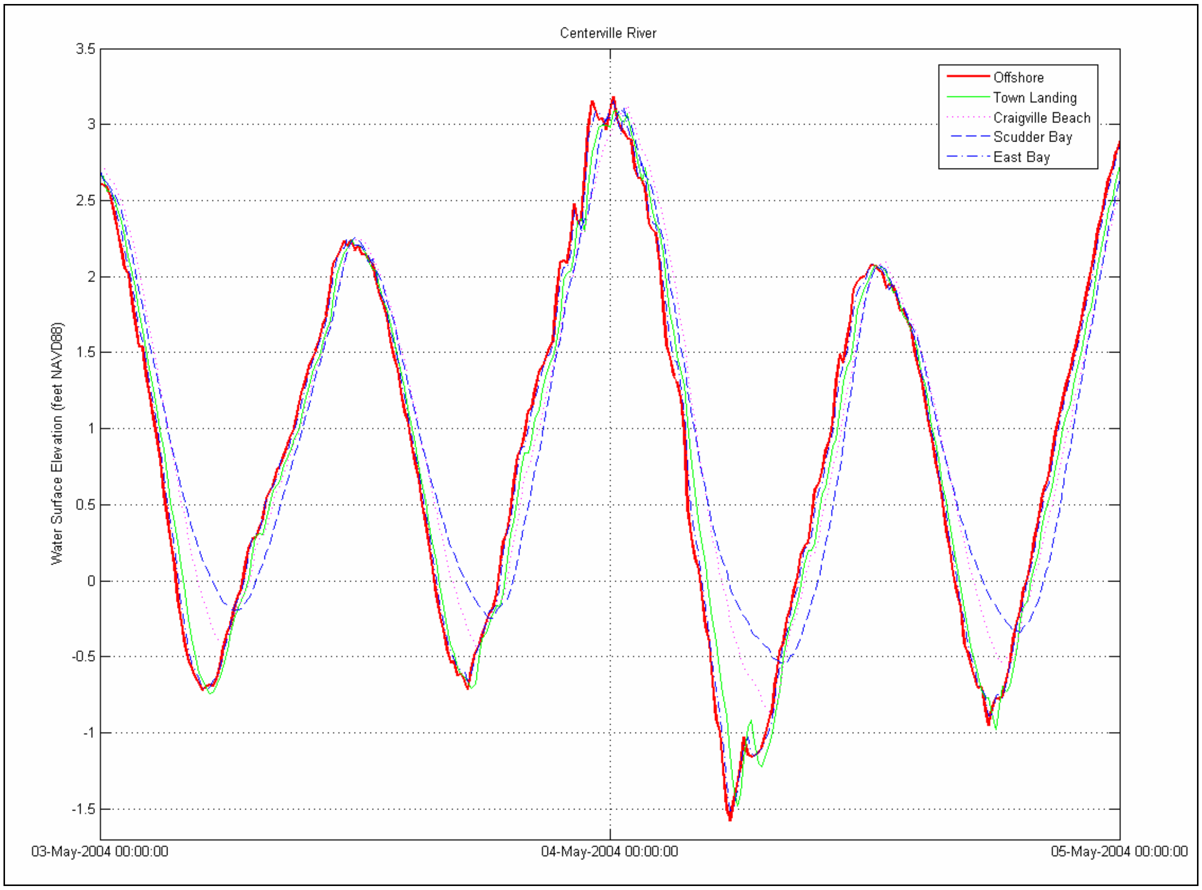


Figure V-10. Water elevation variations for a 2-day period in the Centerville River estuary. Notice the reduced amplitude as well as the delay in times of high- and low- tide relative to offshore (Nantucket Sound) due to frictional damping through the estuary.

V.3.3.2 Current Measurements

Current measurements in Centerville River, surveyed on October 24, 2003, provided observation of the temporal and spatial variability of the flow regime during a tidal cycle. The survey was designed to observe tidal flow through the Centerville River inlet, and attenuation by frictional damping through upstream constrictions at hourly intervals. The current measurements observed during the flood and ebb tides at each constriction can be seen in Figures V-11 through V-14. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. For example, at the Centerville River inlet (line A1), positive along-channel is in the direction of northwest, and positive cross-channel is in the direction of northeast. In the lower left panel of the figures, the mean current or average currents across the channel are shown relative to the shoreline. The lower right panel indicates the stage of the tide during the transect illustrated (shown by a vertical line through the water elevation curve).

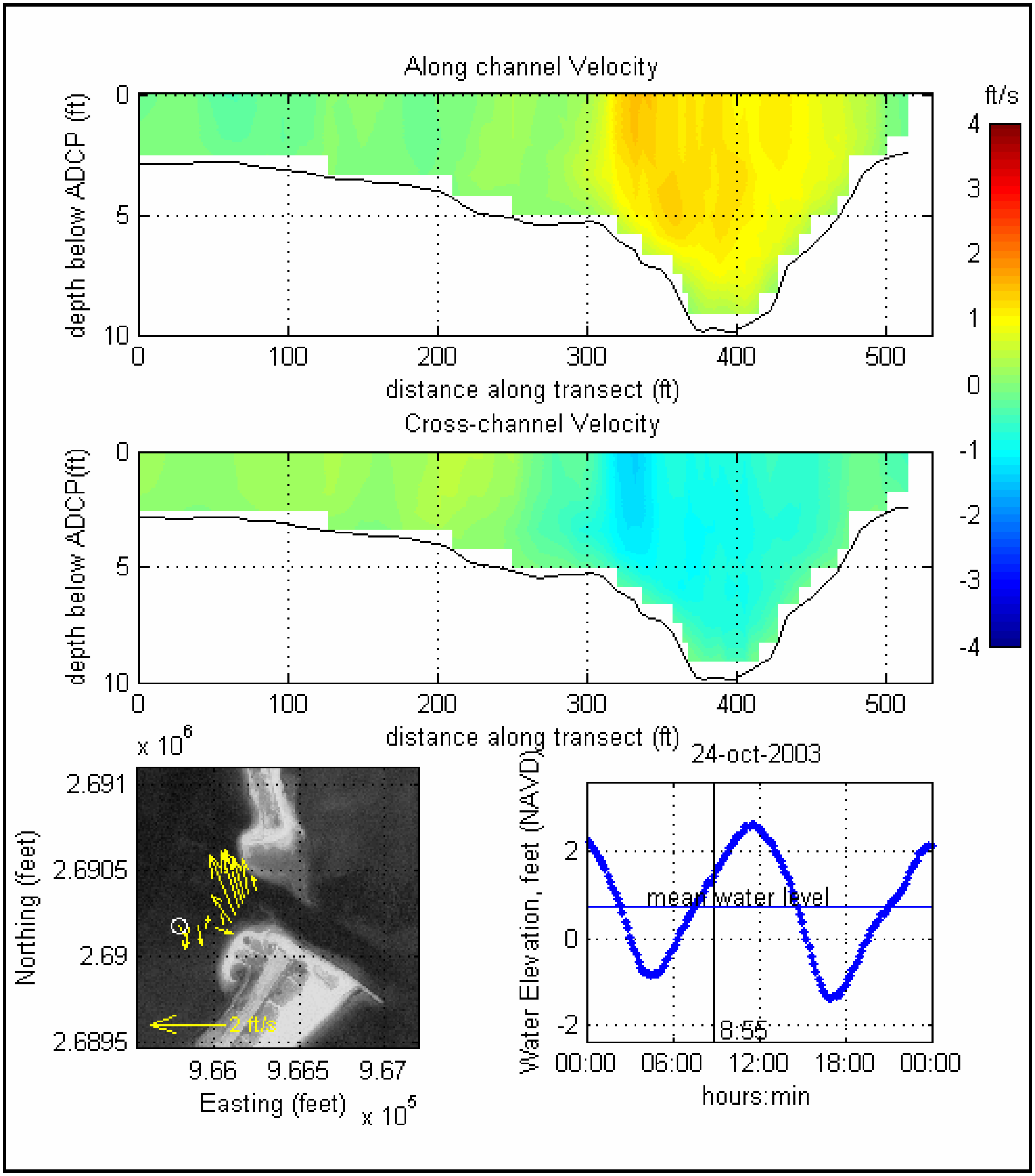


Figure V-11. Color contour plots of along-channel and cross-channel velocity components for transect line A1 across the Centerville River inlet measured at 8:55 on October 24, 2003 during the flood tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

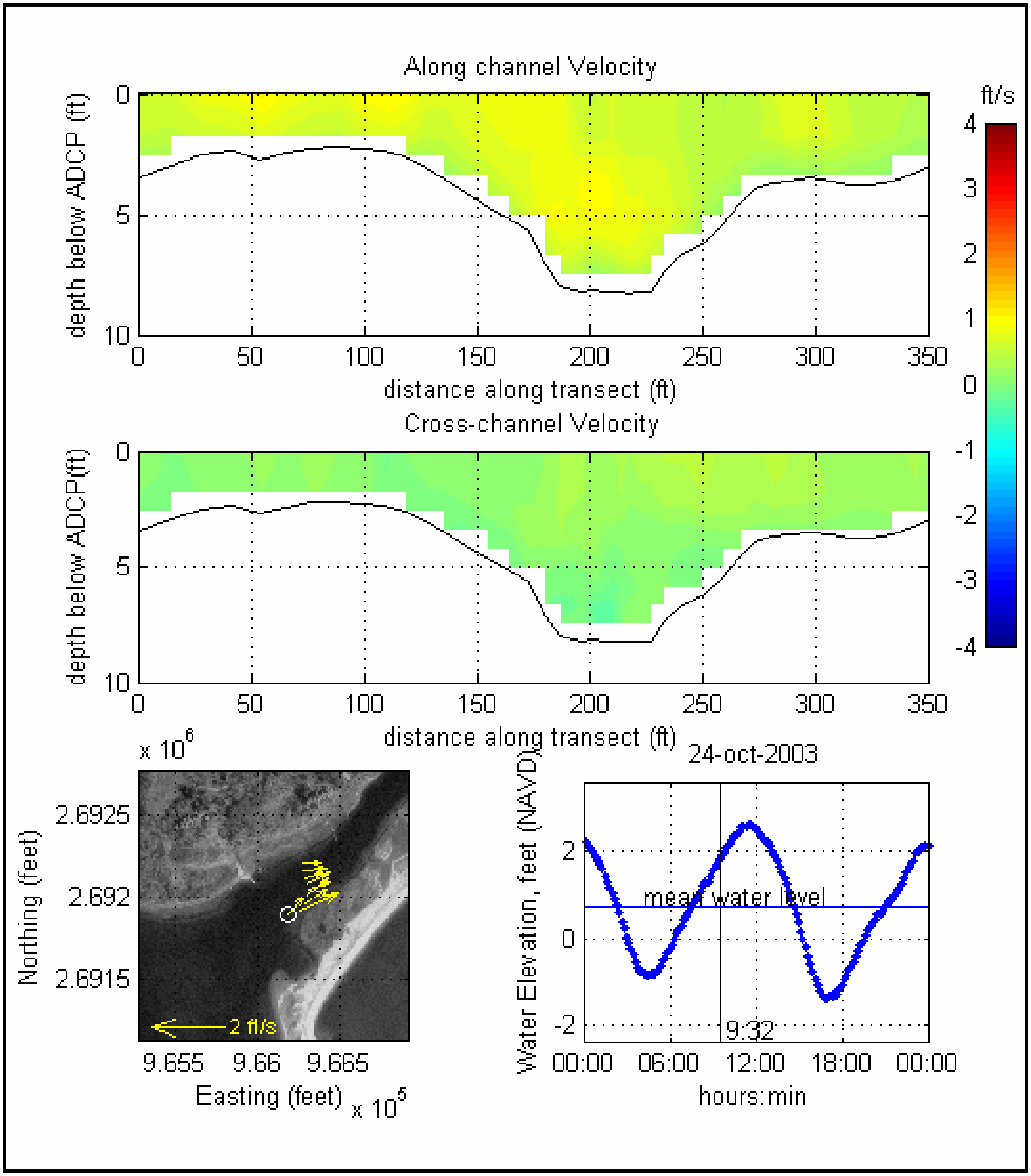


Figure V-12. Color contour plots of along-channel and cross-channel velocity components for transect line A2 across Centerville River measured at 9:32 on October 24, 2003 during the flood tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

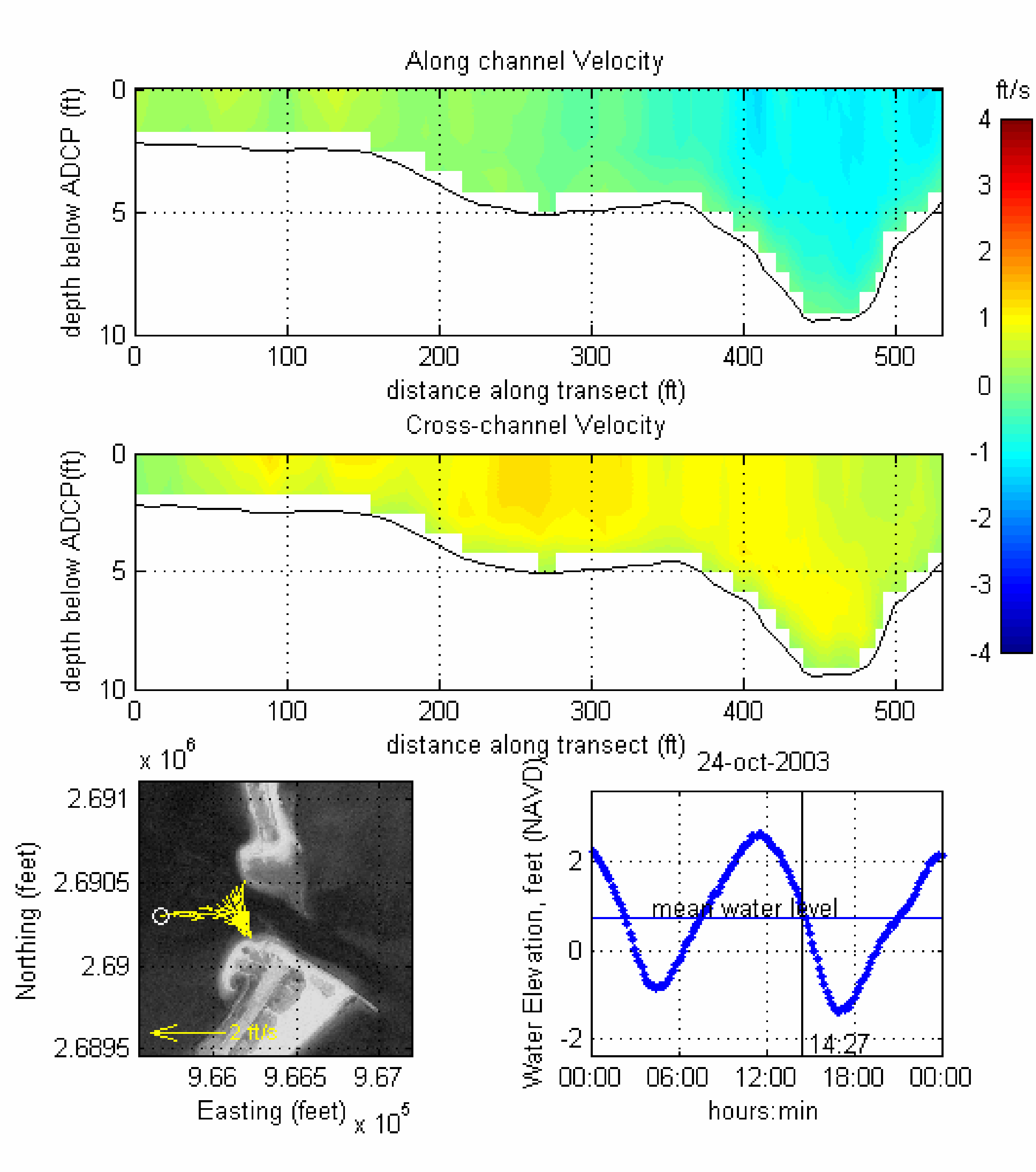


Figure V-13. Color contour plots of along-channel and cross-channel velocity components for transect line A1 across Centerville River inlet measured at 14:27 on October 24, 2003 during the ebb tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

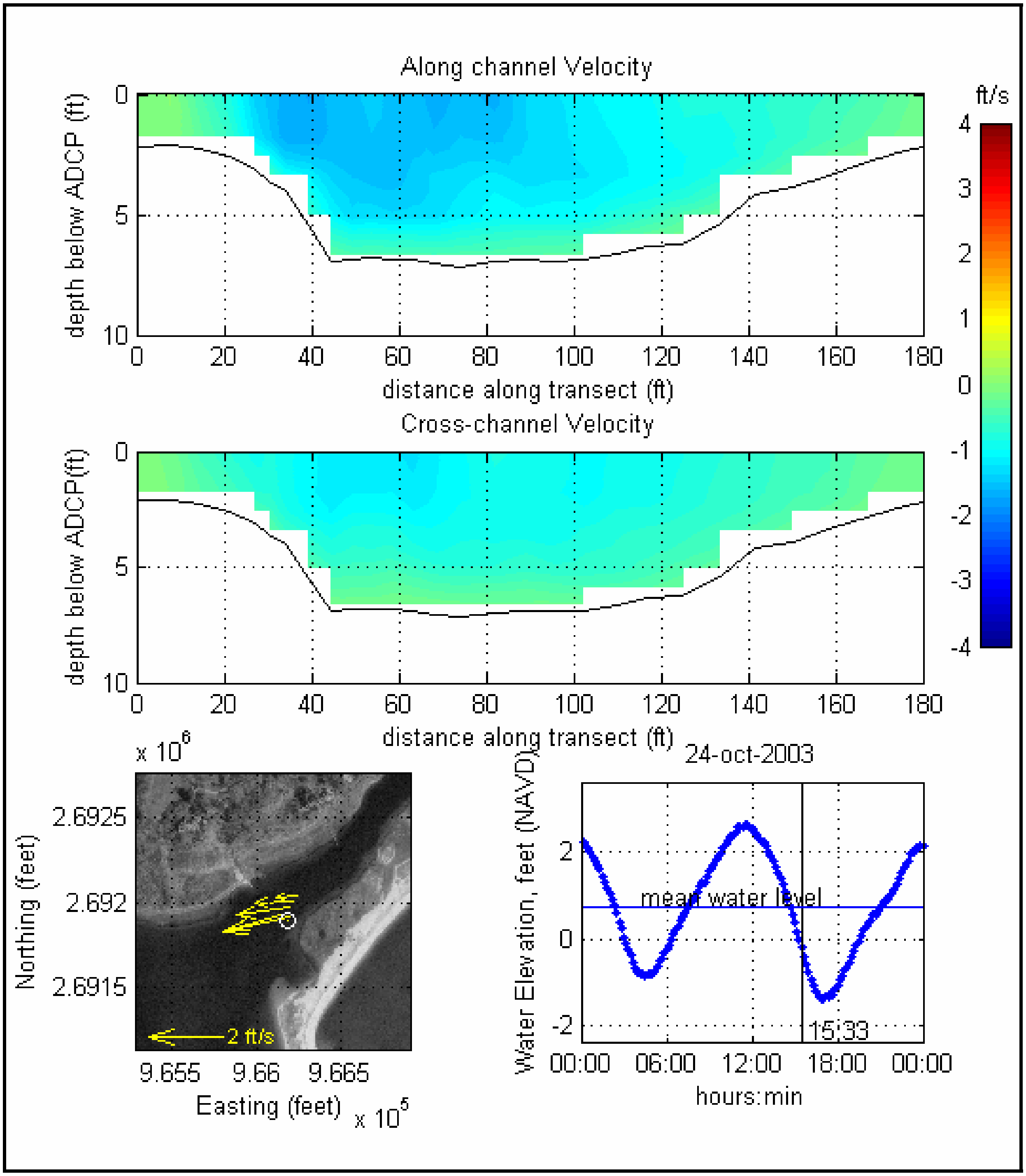


Figure V-14. Color contour plots of along-channel and cross-channel velocity components for transect line A2 across Centerville River measured at 15:33 on October 24, 2003 during the ebb tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

The inlet to Centerville River is relatively unimpeded by large shoals, although the almost 90° turn from the inlet to Centerville River does decrease the hydraulic efficiency of the opening. Tidal currents through Centerville River inlet (line A1) reached maximum speeds of approximately 2 ft/sec on the flood tide. During periods of maximum currents (flood and ebb) the inlet tidal flows are strongest through the main channel (Figures V-11 and V-13). During slack-water periods, currents were vertically coherent, with negligible stratification in the water column. Maximum volume flux through the Centerville River inlet during flood tide was 2,050 ft^3/sec, while the maximum flux during ebb conditions was slightly higher, at -2,360 ft^3/sec.

ADCP Survey line A2, was measured upstream of Centerville inlet entrance, at the mouth of Centerville River. Measured currents across this transect reached maximum speeds of approximately 2.0 ft/sec on the ebb tide (Figure V-14). During flood tide, the volume flow rate was 1,185 ft^3/sec across line A2, and –1,220 ft^3/sec during ebb tide.

V.4 HYDRODYNAMIC MODELING

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

V.4.1 Model Theory

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

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

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

V.4.2 Model Setup

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

- Grid generation

- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using 1994 digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified on the southern boundary of the grid within Nantucket Sound, based on the tide gage data collected offshore of Centerville River within Nantucket Sound (see Figure V-5 for gage location). 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 hydrodynamic information for future detailed water quality modeling.

V.4.2.1 Grid Generation

The grid generation process was aided by the use of the SMS package. A 1994 digital aerial orthophoto and the bathymetry survey data were imported to SMS, and a finite element grid was generated to represent the embayments and waterways within the estuary. The aerial photograph was used to determine the land boundary of the system, as well as determine the surface coverage of salt marsh. The bathymetry data was interpolated to the developed finite element mesh of the system. The completed grid consists of 3,224 nodes, which describe 1,127 total 2-dimensional (depth averaged) quadratic elements. The maximum nodal depth was -19.85 ft (NAVD88), along the open boundary to Nantucket Sound. The maximum modeled marsh plain elevation was +0.71 ft. In the model grid, an average marsh plain elevation of +0.0 ft was used, based on spot surveys across the marsh. The model marsh topography was varied to provide a monotonically sloping surface, in order to enhance the stability of the hydrodynamic model. The completed grid mesh of the Centerville River system is shown in Figure V-15.

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

Grid resolution was governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability of the system. Relatively fine grid resolution was employed where complex flow patterns were expected. For example, smaller node spacing in marsh creeks and channels was designed to provide a more detailed analysis in these regions of rapidly varying flow. Widely spaced nodes were often employed in areas where flow patterns are not likely to change dramatically, such as in outer portion of the bay, along the channels, and on the marsh plain. Appropriate implementation of wider node spacing and larger elements reduced computer run time with no sacrifice of accuracy.

V.4.2.2 Boundary Condition Specification

Two types of boundary conditions were employed for the RMA-2 model of the Centerville River system: 1) "slip" boundaries, and 2) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. A tidal boundary condition was specified at the offshore boundary of the sound. TDR measurements provided the required data. The rise and fall of the tide in Nantucket Sound is the primary driving force for estuarine circulation in this system. Dynamic (time-varying) model simulations specified a new water surface elevation at the boundary to the bay every model time step (10 minutes). Although freshwater enters Centerville Rivers via groundwater, the rate of inflow can be considered negligible relative to the tidal flow that dominates the hydrodynamic processes.

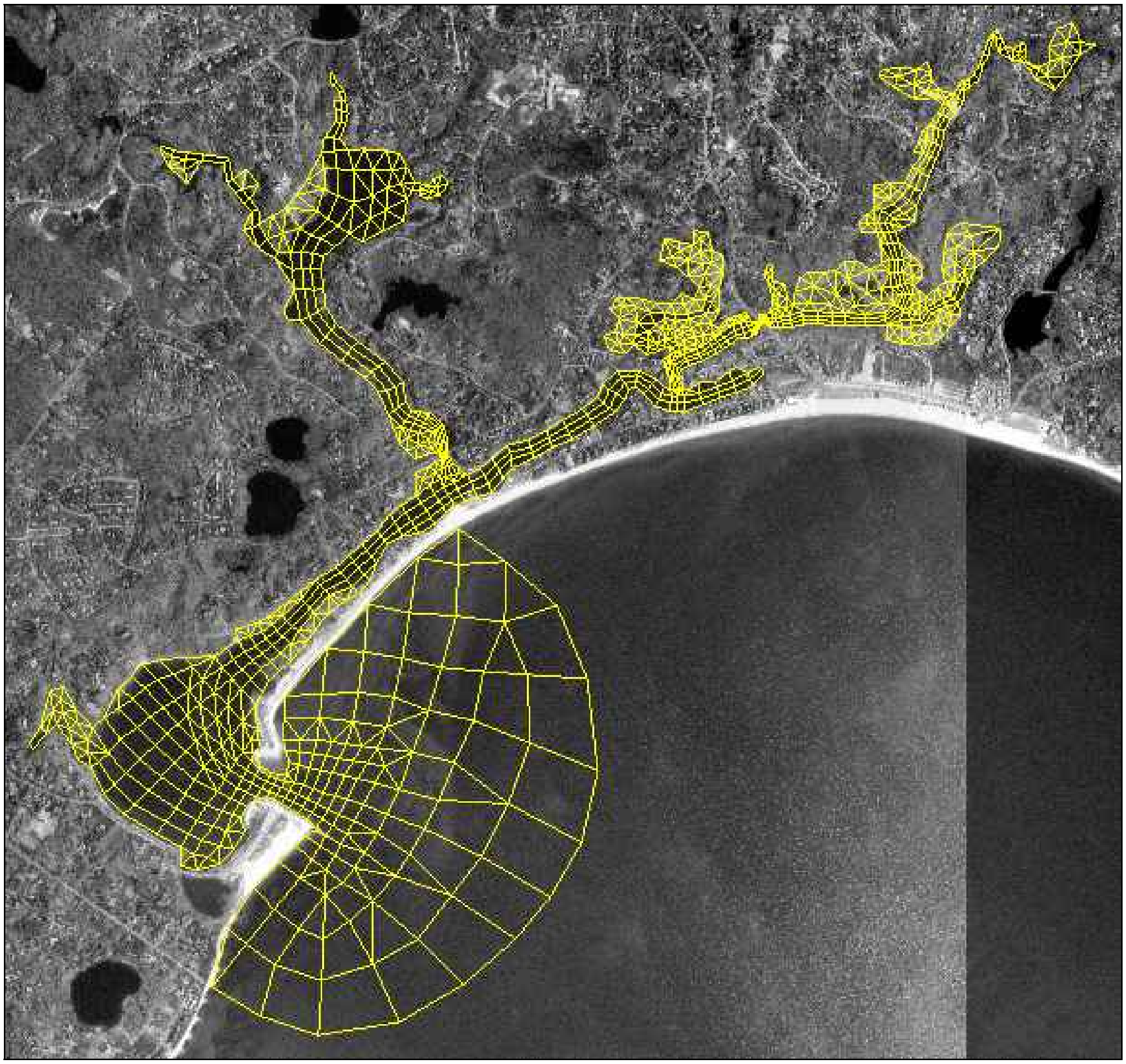


Figure V-15. Plot of hydrodynamic model finite element mesh for the Centerville River system.

V.4.2.3 Calibration

After developing the finite element grid, and specifying boundary conditions, the model for the Centerville 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 for an estuary model, specifying a range of friction and eddy viscosity coefficients, to calibrate the model.

Calibration of the hydrodynamic model required a close match between the modeled and measured tides in each of the sub-embayments where tides were measured (i.e., from the TDR deployments). Initially, the model was calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement was achieved, an approximately seven-day period (14 tide cycles) was modeled to calibrate the model based on dominant tidal constituents discussed in Section V.3.3.1. The seven-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 seven-day period beginning May 1, 2004. This time period represents the transition from neap to spring tide conditions, or a period of average tidal conditions for forcing conditions for use in model verification and flushing analysis.

The calibrated model was used to analyze existing detailed flow patterns and compute residence times. The ability to model a range of flow conditions is a primary advantage of a numerical tidal flushing model. For instance, average residence times were computed over the entire 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.4.2.3.1 Friction Coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where velocities are relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude damping and phase delay of tidal signals. Friction is approximated in RMA-2 as a Manning coefficient, and is applied to grid areas by user specified material types. Initially, Manning's friction coefficients between 0.023 and 0.045 were specified for all element material types. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels and marsh plains with higher friction (Henderson, 1966).

To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning’s coefficients were matched to bottom type. For example, lower friction coefficients were specified for the smooth sandy channels found in the lower portions of the Centerville River, versus the heavily vegetated marsh plains in upper Centerville River, which provide greater flow resistance. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within each estuary. Manning's values for different bottom types were initially selected based ranges provided by the Civil Engineering Reference Manual (Lindeburg, 1992), and values were incrementally changed when necessary to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-5.

Table V-5. Manning's Roughness coefficients used in model simulations. These delineations correspond to the material type areas shown in Figure V-16.	
System Embayment	Bottom Friction
Offshore	0.025
East Bay	0.025
East Bay Marsh Plain	0.045
Bumps River and Scudder Bay	0.025
Bumps River and Scudder Bay Marsh Plain	0.045
Centerville River	0.025
Centerville River Marsh Plain	0.045
Upper Centerville River	0.023
Upper Centerville River Marsh Plain	0.040

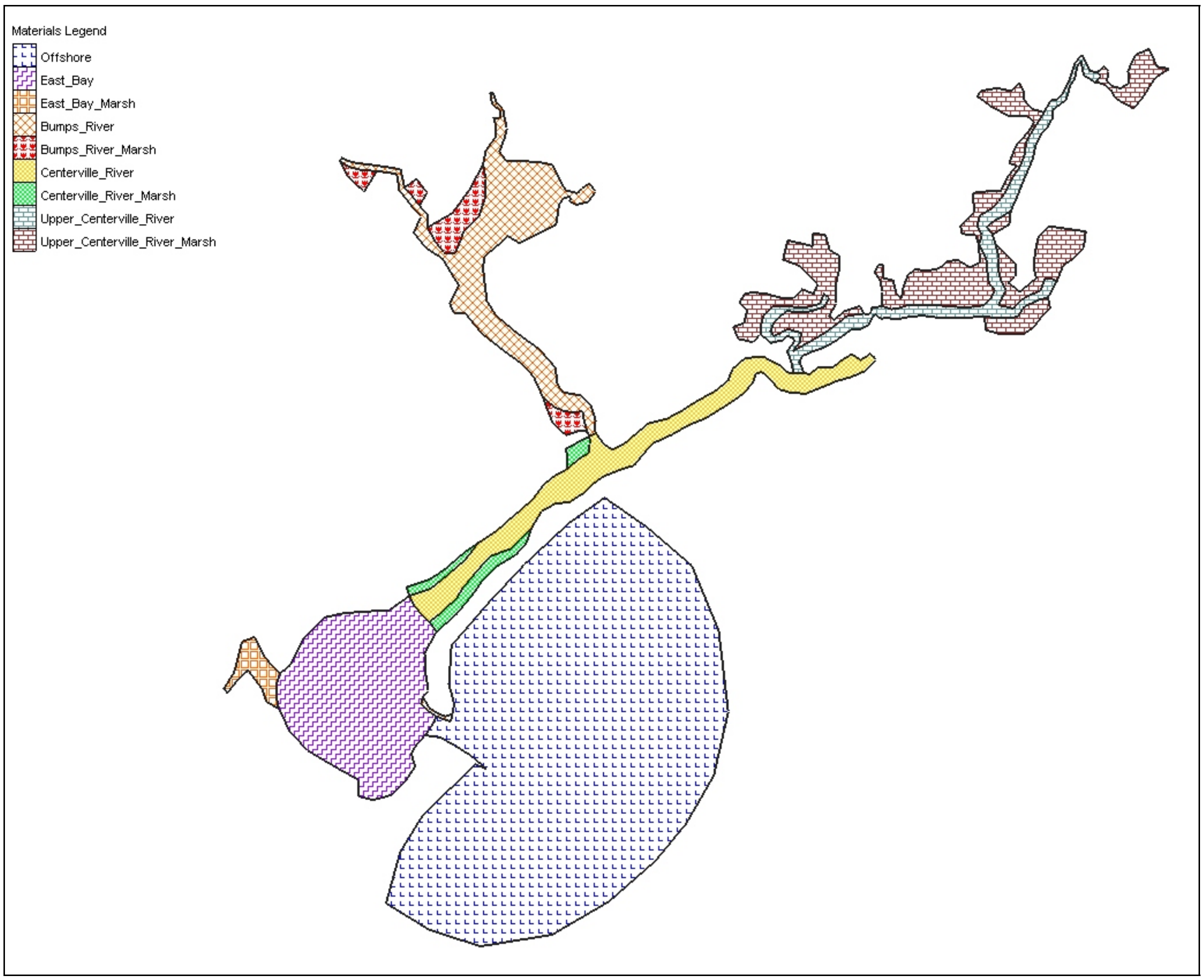


Figure V-16. Hydrodynamic model grid material properties. Color patterns designate the different model material types used to vary model calibration parameters and compute flushing rates.

V.4.2.3.2 Turbulent Exchange Coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swifter, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). In most cases, the modeled systems were relatively insensitive to turbulent exchange coefficients because there were no regions of strong turbulent flow. Typically, model turbulence coefficients were set between 40 and 60 lb-sec/ft^2. Higher values (up to 70 lb-sec/ft^2) were used on the marsh plain, to ensure solution stability.

V.4.2.3.3 Marsh Porosity Processes

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

V.4.2.3.4 Comparison of Modeled Tides and Measured Tide Data

A best-fit of model predictions for the first TDR deployment was achieved using the aforementioned values for friction and turbulent exchange. Figures V-17 though V-20 illustrate the seven-day calibration simulation along with 72-hour sub-section, for East Bay, Scudder Bay, Centerville River at the Town Landing, and Centerville River at the Craigville Beach Bridge. Modeled (dashed 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 was the highest priority since M_2 accounted for a majority of the forcing tide energy in the modeled systems. Due to the duration of the model runs, four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Table V-6 for the calibration period differ from those in Table V-2 because constituents were computed for only the seven-day section of the 29-days represented in Table V-2. Table V-6 compares tidal constituent height and time lag for modeled and measured tides at the TDR locations.

The constituent calibration resulted in excellent agreement between modeled and measured tides. The largest errors associated with tidal constituent amplitude were on the order of 0.1 ft, or just greater than one inch. Time lag errors were typically less than the time increment resolved by the model (0.1667 hours or 10 minutes), indicating good agreement between the model and data. The largest errors were in Centerville River at Craigville Beach Bridge, but the model still very closely agrees with the TDR measurement, see Figure V-20.

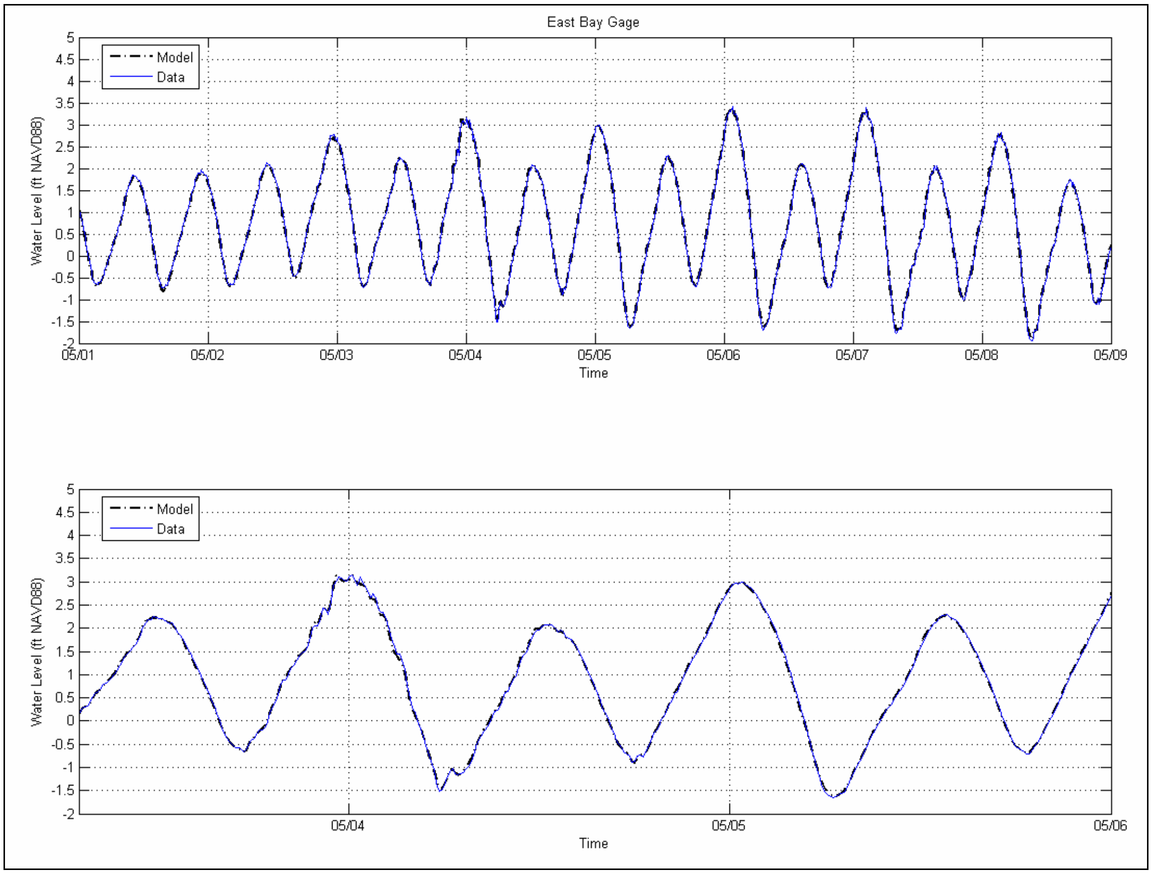


Figure V-17. Comparison of model output and measured tides for the TDR location in East Bay. The bottom plot is a 72-hour sub-section of the total modeled time period, shown in the top plot.

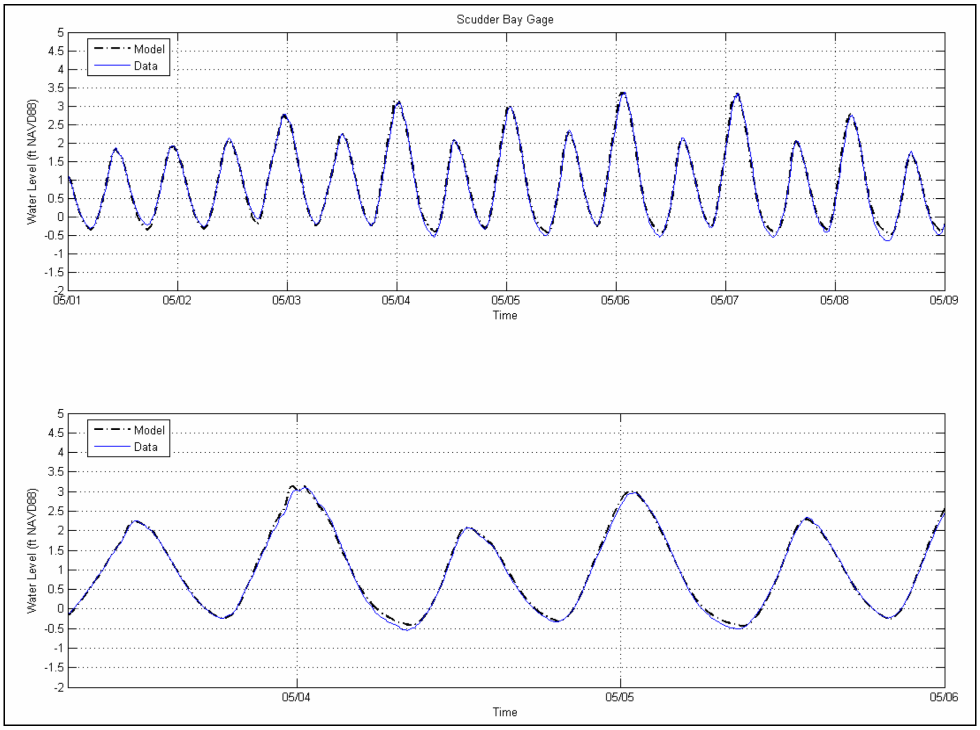


Figure V-18. Comparison of model output and measured tides for the TDR location in Scudder Bay. The bottom plot is a 72-hour sub-section of the total modeled time period, shown in the top plot.

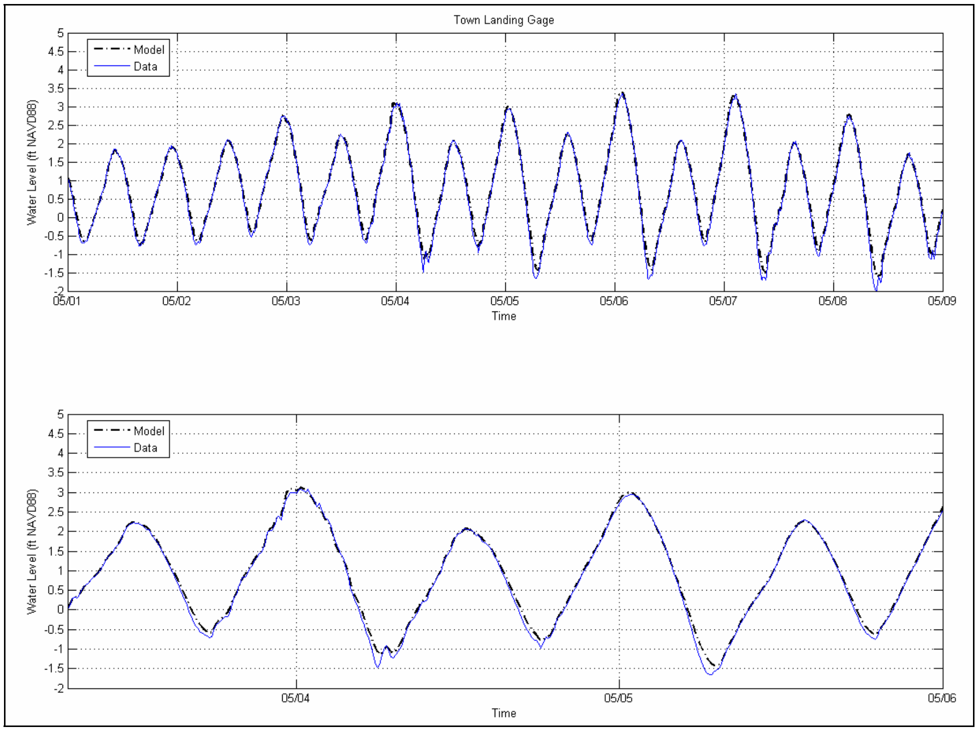


Figure V-19. Comparison of model output and measured tides for the TDR location at the Town Landing in Centerville River. The bottom plot is a 72-hour sub-section of the total modeled time period, shown in the top plot.

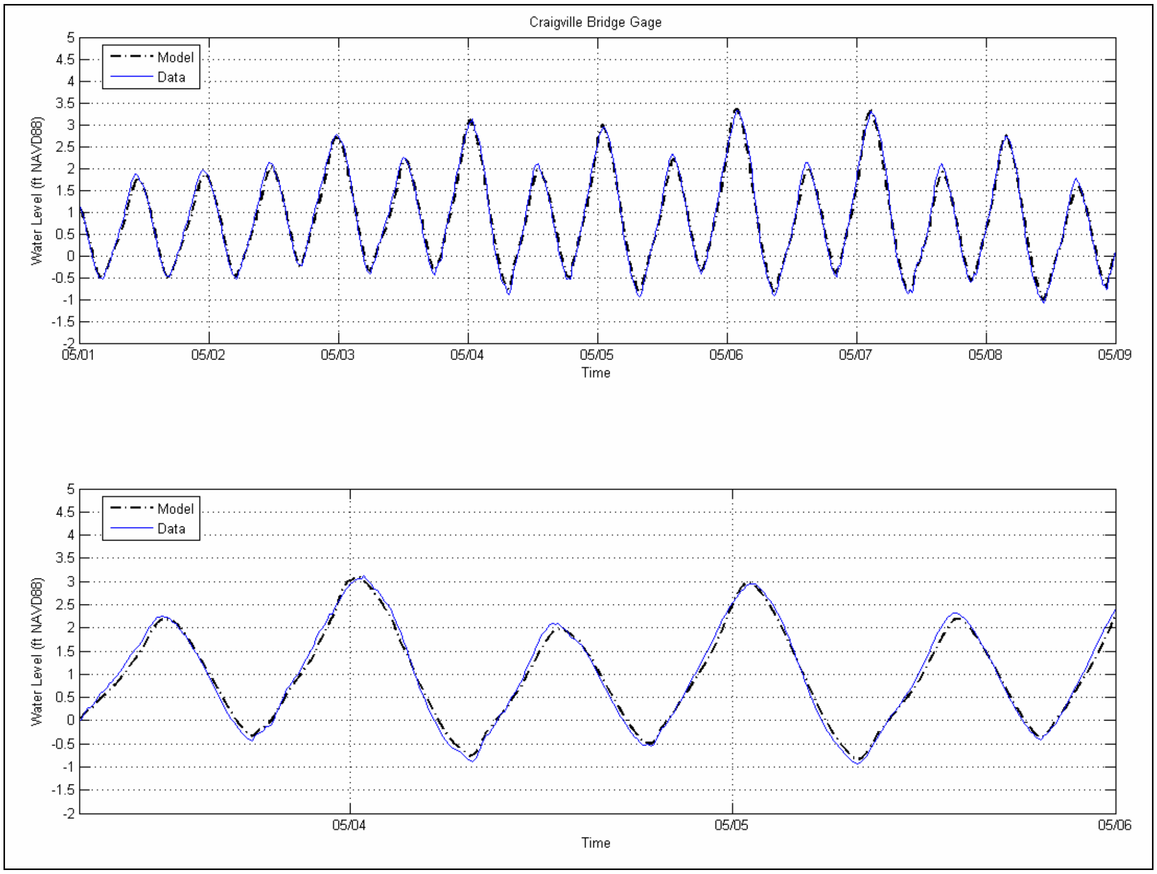


Figure V-20. Comparison of model output and measured tides for the TDR location at the Craigville Beach Bridge in Centerville River. The bottom plot is a 72-hour sub-section of the total modeled time period, shown in the top plot.

V.4.2.4 Model Verification

A verification of the model was then conducted for the period of October 21, 2003 to October 29, 2003 to simulate the time period when the ADCP measurements were taken (October 24, 2003). The verification compared the measured and modeled tide and compared flow rates computed from ADCP measurements to flow rates extracted from the hydrodynamic model. Flow measurements were extracted from the model along two lines in Centerville River which corresponds to the ADCP measurement transects (see Figure V-5 for transect locations).

Comparisons of the modeled and measured flow rates for the ADCP transects are shown in Figures V-21 and V-22. The graphs show that the model follows the trends and characteristics of the ADCP data. However, the model slightly over-predicts the volume of water flow across the transect lines. To quantify the error, an R square error analysis was performed on the results. The results, shown in Table V-7, indicate that the error in the flows rates was small with R-squared values above 0.9.

Table V-6. Tidal constituents for measured water level data and calibrated model output for Centerville River.

Model calibration run						
Location	Constituent Amplitude (ft)				Phase (rad)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Offshore	1.64	0.22	0.09	0.49	-1.31	-0.79
East Bay	1.63	0.21	0.09	0.49	-1.27	-0.72
Scudder Bay	1.33	0.14	0.03	0.43	-0.93	-2.12
Centerville River-Town Landing	1.54	0.15	0.10	0.47	-1.14	-0.53
Centerville River-Craigville Beach	1.33	0.10	0.09	0.42	-0.93	-0.75
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (rad)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Offshore	1.64	0.22	0.09	0.49	-1.31	-0.79
East Bay	1.63	0.22	0.09	0.49	-1.26	-0.71
Scudder Bay	1.34	0.13	0.04	0.42	-0.93	-1.97
Centerville River-Town Landing	1.61	0.20	0.10	0.48	-1.17	-0.55
Centerville River-Craigville Beach	1.43	0.10	0.07	0.43	-0.99	-0.66
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Offshore	0.00	0.00	0.00	0.00	0.07	-0.04
East Bay	0.01	0.01	0.00	0.00	1.50	1.00
Scudder Bay	0.02	-0.01	0.01	-0.01	0.73	9.30
Centerville River-Town Landing	0.06	0.05	0.00	0.01	-3.36	-1.39
Centerville River-Craigville Beach	0.09	0.00	+0.02	0.01	-7.40	5.37

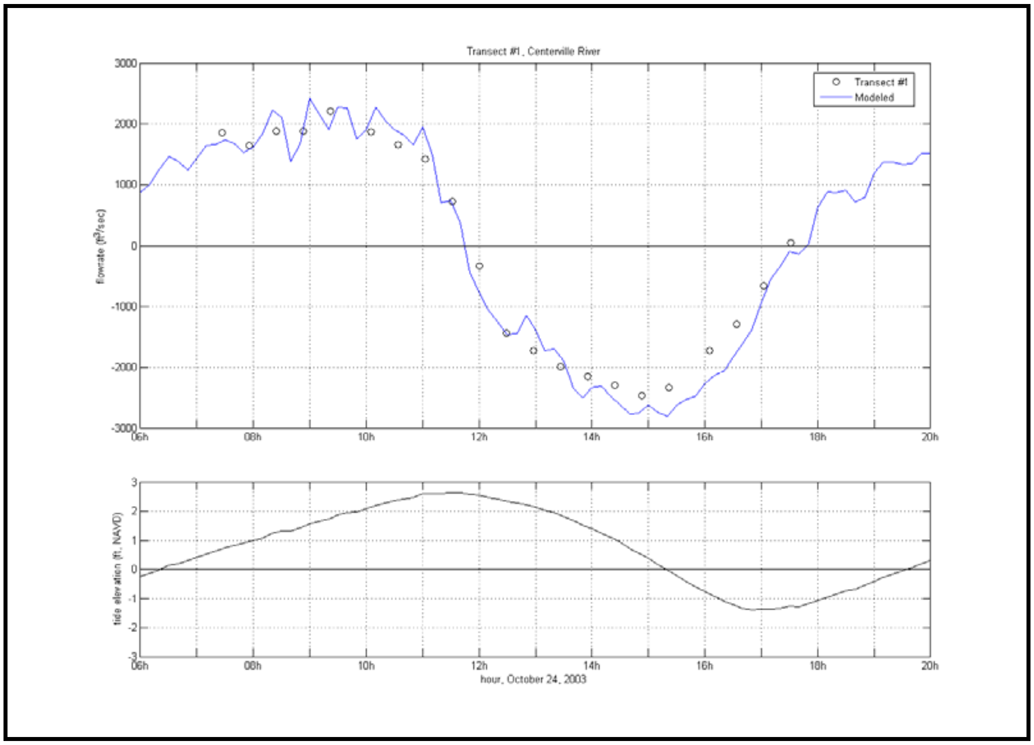


Figure V-21. Comparison of computed flow rates to ADCP Transect 1 across the entrance of East Bay. Model period shown corresponds to transition from high to low tide. Positive flow indicated flooding tide, while negative flow indicates ebbing tide.

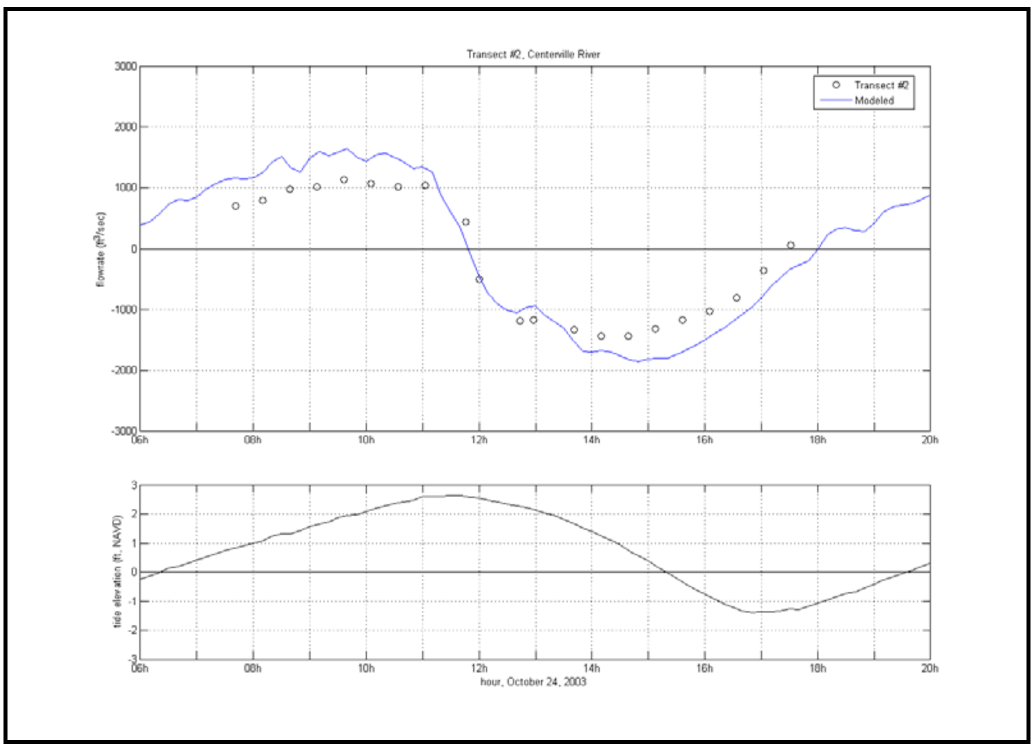


Figure V-22. Comparison of computed flow rates to the ADCP transect at Transect 2. Model period shown corresponds to transition from high to low tide. Positive flow indicated flooding tide, while negative flow indicates ebbing tide.

There are several possible reasons for the model over-predicting the flow measurements. The primary limitation of the ADCP measurements was the inability to capture the outer edges of the channel as a result of depth limitations with the boat and the ADCP. The size of this gap was dependent of the side slopes of the channel. For Transect 1 this ranged between 8 to 12 feet and 15 to 20 feet for Transect 2; therefore, the measurements do not account for the flow along the outer most edges of the channels. Thus, the measured flow rates are assumed to be 10 to 15 percent lower than actual flow rates. Secondly, the ADCP is unable to measure velocities in the first 1 to 2 feet of the water column, due to the ADCP transducer being suspended below the water surface and signal blanking across the first measurement cell. The ADCP cannot take measurements across the first measurement cell since a time gap is required between the transmission and receipt of the acoustic signal (this allows measurement of the Doppler shift). To account for the unmeasured portion of the water column, velocities from second measurement cell were used to represent the portion of water column above. This resulted in a slight under prediction in surface currents and thus adds to the under-prediction of flow rates. Although the measured flow rates were approximately 15 percent less than the modeled flows, the current measurement limitations (primarily the loss of data near the shallow channel edges) provide a reasonable explanation for this magnitude of error. Therefore, the ADCP measurements within Centerville River provided adequate measurements to verify the results of the hydrodynamic model.

Table V-7. Least square error results on the flow analysis for Centerville River.

Transect	R Squared	R Square Error (ft^3/s)
Transect 1	0.978	421
Transect 2	0.913	385

V.4.2.5 Model Circulation Characteristics

The final calibrated and validated model serves as a useful tool in investigating the circulation characteristics of the Centerville 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.

Examining the results from the model run shows ebb velocities in the entrance and lower channels are slightly larger than velocities during maximum flood. The highest velocities occur at the bridge constrictions at Craigville Beach Road on the Centerville River, where the channel width is constrained between the bridge abutments. Similar velocity magnitudes occur along the Centerville River and at the entrance to East Bay. In areas with wider channels, the peak velocities are slightly lower. The maximum velocities in the entrance to East Bay peaks at approximately 2.7 feet/sec during the ebb tide, while maximum ebb velocities are about 1.8 feet/sec. A close-up of the model output is presented in Figure V-23, showing contours of velocity magnitude, along with velocity vectors that indicate the magnitude and direction of flow, for a single model time-step, at the portion of the tide cycle where maximum ebb velocities occur at the entrance to East Bay.

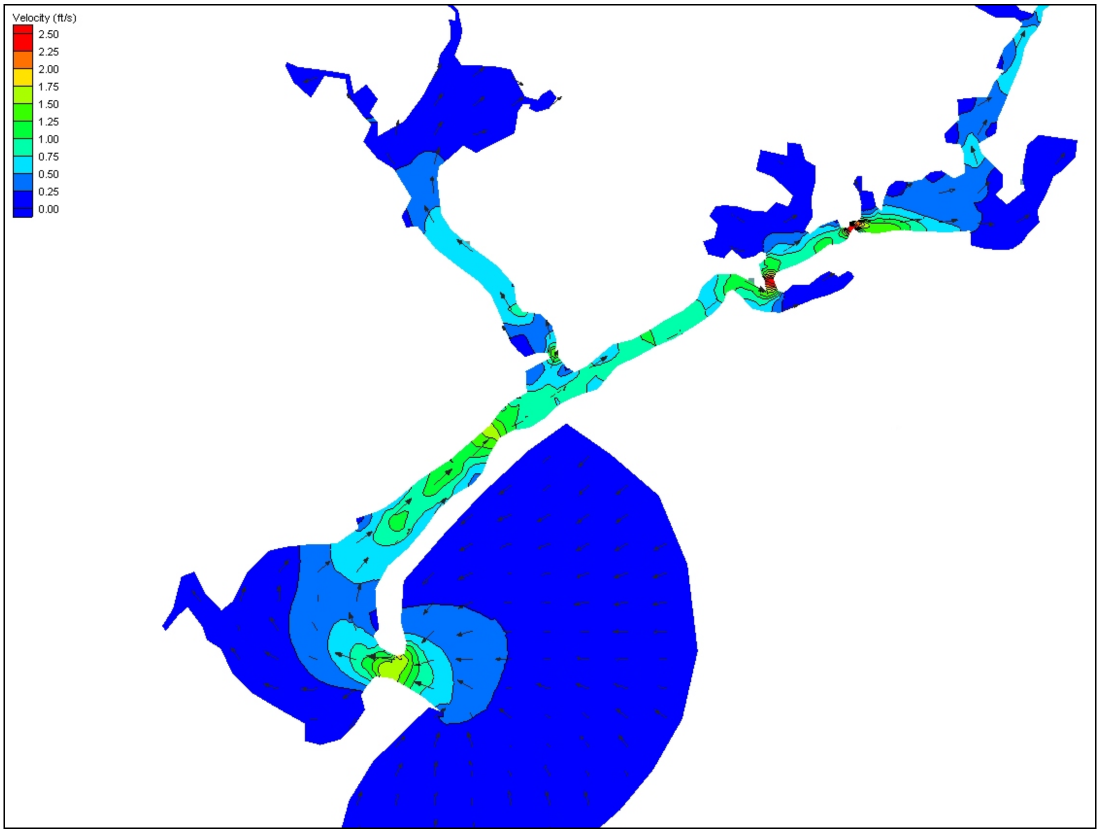


Figure V-23. Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

V.4 FLUSHING CHARACTERISTICS

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

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

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

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

In addition to system residence times, a second residence, the **local residence time**, was defined as the average time required for a water parcel to migrate from a location within a sub-embayment to a point outside the sub-embayment. Using Scudder Bay as an example, the **system residence time** is the average time required for water to migrate from Scudder Bay and Bumps River, through Centerville River, into East Bay, and into Nantucket Sound, where the **local residence time** is the average time required for water to migrate from Scudder Bay to just Centerville 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 Centerville River system this approach is applicable, since it assumes the main system has relatively low quality water relative to Nantucket Sound.

The rate of pollutant/nutrient loading and the quality of water outside the estuary both must be evaluated in conjunction with residence times to obtain a clear picture of water quality. Efficient tidal flushing (low residence time) is not an indication of high water quality if pollutants and nutrients are loaded into the estuary faster than the tidal circulation can flush the system. Neither are low residence times an indicator of high water quality if the water flushed into the estuary is of poor quality. Advanced understanding of water quality will be obtained from the calibrated hydrodynamic model by extending the model to include pollutant/nutrient dispersion. The water quality model will provide a valuable tool to evaluate the complex mechanisms governing estuarine water quality in the Centerville 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 two main sub-embayments within the system. In addition, **system** and **local residence times** were computed to indicate the range of conditions possible for the system. Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism). Units then were converted to days. The volume of the entire estuary was computed as cubic feet. Sub-embayment mean volumes and average tide prisms computer for the Centerville River system are presented in Table V-8.

Residence times were averaged for the tidal cycles comprising a representative 7.25 day period (14 tide cycles), and are listed in Table V-9. The modeled time period used to compute the flushing rates was different from the modeled calibration period, and included the transition from neap to spring tide conditions. Model divisions used to define the system sub-embayments include 1) East Bay, 2) Lower Centerville River, (3) Upper Centerville River, and 4) Scudder Bay and Bumps River (corresponds to divisions shown in Figure V-16). The model calculated flow crossing specified grid lines for each sub-embayment to compute the tidal prism volume. Since the 7.25-day period used to compute the flushing rates of the system represent average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the system sub-embayments.

Computed flushing rates for the Centerville River system show that as a whole, the system flushes well. A flushing time of 0.51 days for the entire estuary shows that on average, water is resident in the system for approximately a half of a day. This is also evident by the fact that the tidal prism of the whole estuary is approximately equal to its mean volume. Scudder Bay and Bumps River have the greatest system residence time. However the local residence time for Scudder Bay and Bumps River is less than a day. As a whole the system flushes well due to shallow basin depths and relatively large forcing tide.

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

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Centerville River (whole system)	39,774,950	39,337,036
East Bay	15,481,617	13,749,703
Centerville River (Lower)	10,511,886	7,966,521
Scudder Bay and Bumps River	5,645,965	8,135,937
Centerville River (Upper)	7,715,137	10,348,940

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

Embayment	System Residence Time (days)	Local Residence Time (days)
Centerville River (whole system)	0.53	0.53
East Bay	1.50	0.59
Centerville River (Lower)	1.98	0.69
Scudder Bay and Bumps River	3.66	0.36
Centerville River (Upper)	2.68	0.39

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

VI. WATER QUALITY MODELING

VI.1 DATA SOURCES FOR THE MODEL

Several different data types and calculations are required to support the water quality modeling effort for the Centerville 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-2V 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 11-tidal cycle period in May 2004. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to enable the model to reach a dynamic “steady state”, and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the 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 Centerville 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. Five years of data (collected between 2001 and 2005) were available for stations monitored by SMAST in the Centerville River System.

Table VI-1. Pond-Watcher measured data, and modeled Nitrogen concentrations for the Centerville 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	Scudder Bay	Bumps River	Centerville River	Centerville River	Centerville River	Centerville River	East Bay
Monitoring station	BC-3	BC-4	BC-5	BC-7	BC-8	BC-9	BC-10
2001 mean	0.593	0.325	0.697	0.484	0.423	0.332	0.330
2002 mean	0.628	0.505	0.759	0.618	0.513	0.460	0.466
2003 mean	0.661	0.542	0.793	0.589	0.588	0.433	0.413
2004 mean	0.569	0.485	0.710	0.536	0.552	0.453	0.390
2005 mean	0.610	0.423	0.720	0.467	0.474	0.399	0.382
mean	0.619	0.481	0.745	0.551	0.526	0.430	0.408
s.d. all data	0.105	0.113	0.147	0.117	0.125	0.112	0.086
N	28	28	29	29	57	55	57
model min	0.386	0.333	0.457	0.385	0.335	0.316	0.310
model max	0.695	0.616	0.749	0.670	0.584	0.514	0.442
model average	0.524	0.451	0.609	0.526	0.454	0.389	0.349

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 Centerville 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 Centerville River System. Like RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. Members of the MEP Technical Team have utilized this model in water quality studies of other Cape Cod embayments, including systems in Falmouth (Ramsey *et al*., 2000); Mashpee, MA (Howes *et al*., 2004) and Chatham, MA (Howes *et al*., 2003).

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

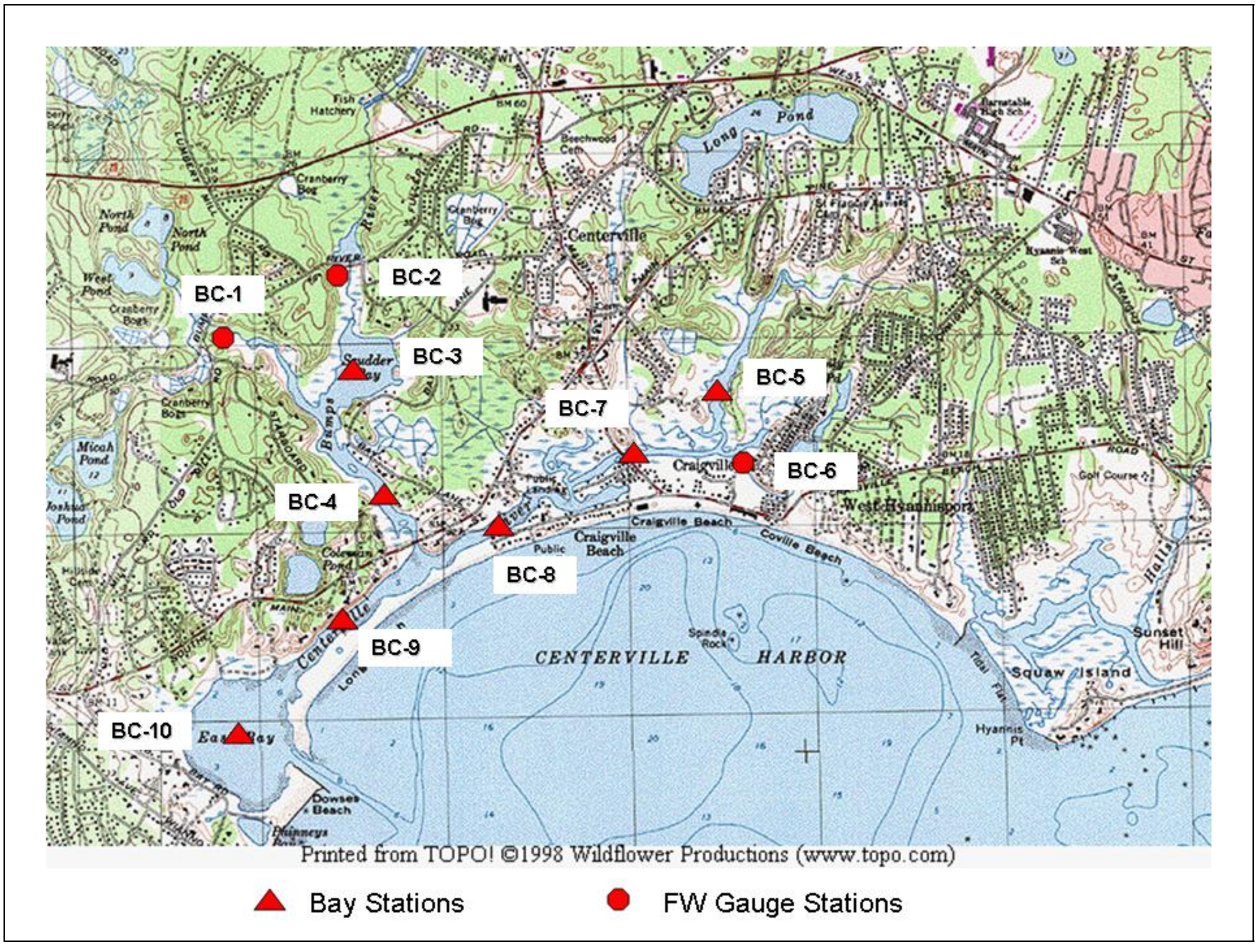


Figure VI-1. Estuarine water quality monitoring station locations in the Centerville River System. Station labels correspond to those provided in Table VI-1.

VI.2.1 Model Formulation

The formulation of the model is for two-dimensional depth-averaged systems in which concentration in the vertical direction is assumed uniform. The depth-averaged assumption is justified since vertical mixing by wind and tidal processes prevent significant stratification in the modeled 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 Centerville 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 Centerville River System was used for the water quality constituent modeling portion of this study.

Based on measured flow rates from SMAST and groundwater recharge rates from the USGS, the hydrodynamic model was set-up to include the latest estimate of surface water flows from Pine Street stream, Lake Elizabeth/Red Lilly Pond stream, Bumps River, and Skunknet River along with ground water flowing into the system from watersheds. The Pine Street stream has a measured flow rate of 2.66 ft^3/sec (6,518 m^3/day), Lake Elizabeth/Red Lilly Pond stream has a measured flow rate of 0.63 ft^3/sec (1,547 m^3/day), Bumps River has a measured flow rate of 2.39 ft^3/sec (5,847 m^3/day), and Skunknet River has a measured flow rate of 5.69 ft^3/sec (13,925 m^3/day). The overall groundwater flow rate into the system is 22.26 ft^3/sec (54,449 m^3/day) distributed amongst the watersheds.

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 Centerville 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) point source input developed from measurements of the Pine Street stream, Lake Elizabeth/Red Lilly Pond stream, Bumps River, and Skunknet River. Nitrogen loads from each separate sub-embayment watershed were distributed across the sub-embayment. For example, the combined watershed direct atmospheric deposition load for Scudder Bay 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 Centerville 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 coverages, when present), resulting in a total flux for each embayment (as listed in Table VI-2). Due to the highly variable nature of bottom sediments and other estuarine characteristics of coastal embayments in general, the measured benthic flux for existing conditions also is variable. For present conditions, some sub-embayments have almost twice the loading rate from benthic regeneration as from watershed loads. For other sub-embayments, the benthic flux is relatively low or negative indicating a net uptake of nitrogen in the bottom sediments.

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. TN concentrations of the incoming water are set at the value designated for the open boundary. The boundary concentration in Nantucket Sound was set at 0.33 mg/L, based on SMAST data from the Sound. The open boundary total nitrogen concentration represents long-term average summer concentrations found within Nantucket Sound.

Table VI-2. Sub-embayment loads used for total nitrogen modeling of the Centerville 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)
Centerville River East	55.737	0.449	5.394
Scudder Bay	14.452	0.685	-2.125
Centerville River West	9.463	0.718	3.497
East Bay	8.627	1.126	12.694
Surface Water Sources			
Pine Street Stream	3.452	-	-
Lake Elizabeth Stream	2.274	-	-
Bumps River	16.912	-	-
Skunknet River	21.260	-	-

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 Centerville River require values of E that are lower compared to the riverine estuary systems evaluated by Fischer, *et al.*, (1979). Observed values of E in these calmer areas typically range between order 10 and order 0.001 m^2/sec (USACE, 2001). The final values of *E* used in each sub-embayment of the modeled systems are presented in Table VI-3. These values were used to develop the "best-fit" total nitrogen model calibration. For the case of TN modeling, "best fit" can be defined as minimizing the error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within each sub-embayment.

Table VI-3. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for Centerville River System.

Embayment Division	E m^2/sec
Offshore	50.0
East Bay	100.0
East Bay Marsh	1.0
Scudder Bay	1.5
Bumps River/ Scudder Bay Marsh	0.9
Centerville River	45.0
Centerville River Marsh	1.0
Upper Centerville River	4.0
Upper Centerville River Marsh	0.7
Centerville River West	100.0
Stream Inputs	0.4
Bumps River	25.0
Centerville River East	15.0
Centerville River East Marsh	7.0

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

Also presented in this figure are unity plot comparisons of measured data verses modeled target values for the system. The model fit is exceptional for the Centerville River System, with rms error of 0.04 mg/L and an R^2 correlation coefficient of 0.88.

A contour plot of calibrated model output is shown in Figure VI-4 for Centerville 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.

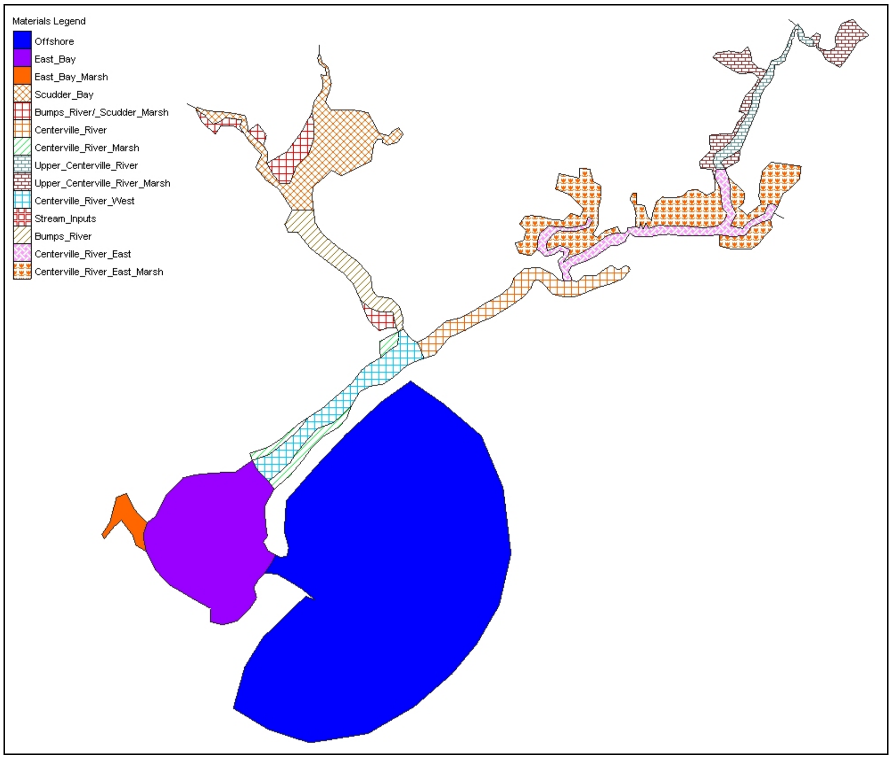


Figure VI-2. Map of Centerville River water quality model longitudinal dispersion coefficients. Color patterns designate the different areas used to vary model dispersion coefficient values.

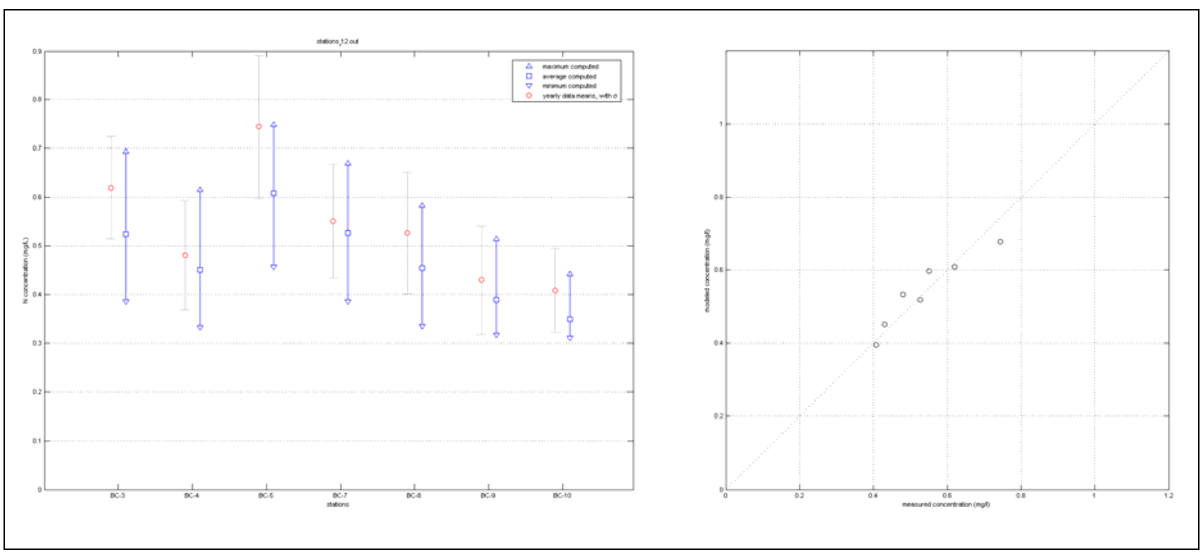


Figure VI-3. Comparison of measured total nitrogen concentrations and calibrated model output at stations in Centerville River System. For the left plot, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed 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. Computed correlation (R^2) and error (rms) for each model are also presented.

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 Centerville 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 30.2 ppt. For groundwater inputs salinities were set at 0 ppt. Groundwater input used for the model was 22.26 ft^3/sec (54,449 m^3/day) distributed amongst the watersheds. Groundwater flows were distributed evenly in each model through the use of several 1-D element input points positioned along each model's land boundary.

Comparisons of modeled and measured salinities are presented in Figure VI-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 Centerville River System. The rms error of the models was 1.44 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.

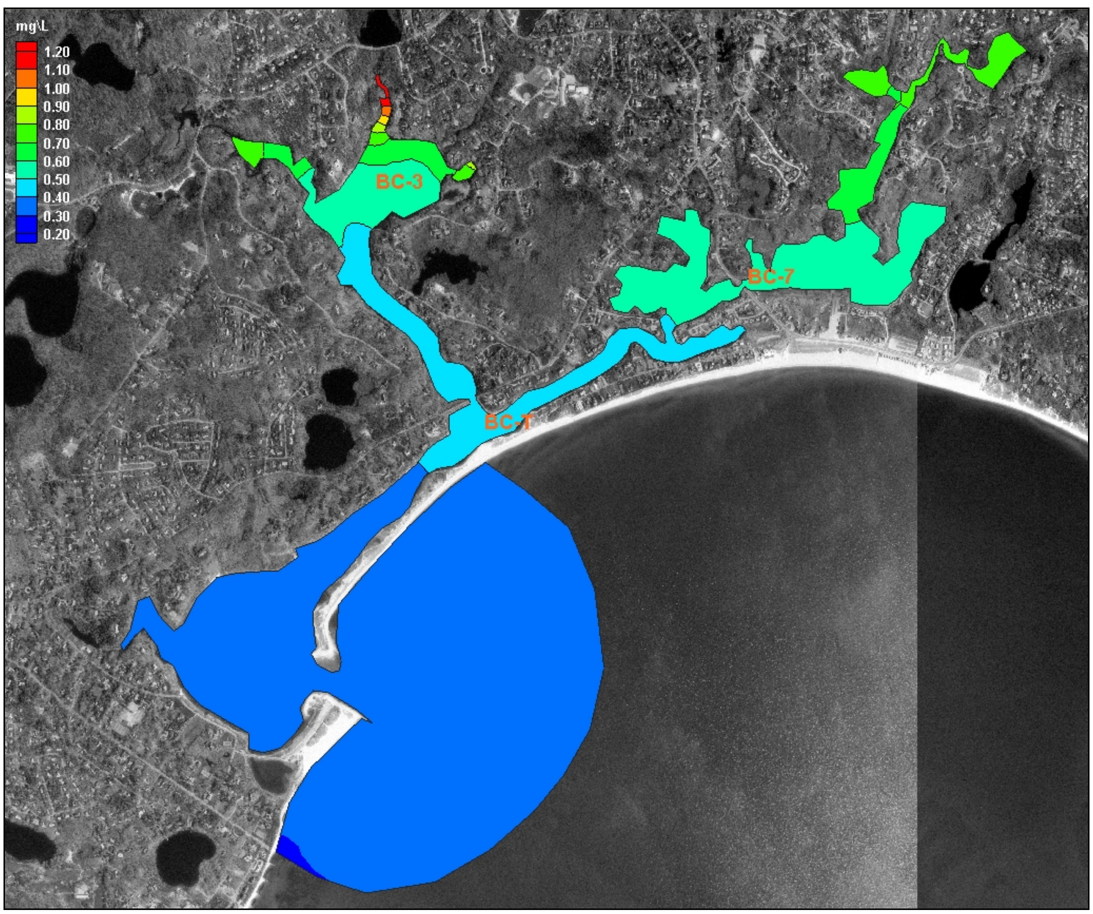


Figure VI-4. Contour plots of average total nitrogen concentrations from results of the present conditions loading scenario, for Centerville River System. The approximate location of the sentinel threshold station for Centerville River System (BC-T) is shown.

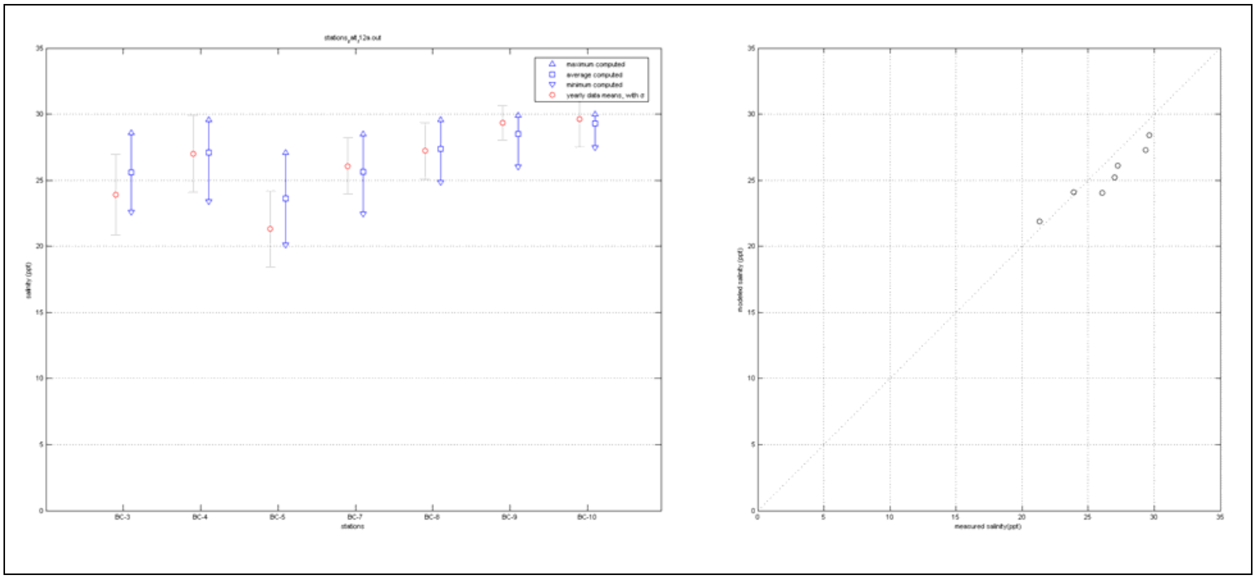


Figure VI-5. Comparison of measured and calibrated model output at stations in Centerville River System. For the left plots, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (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. Computed correlation (R^2) and error (rms) for each model are also presented.

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.

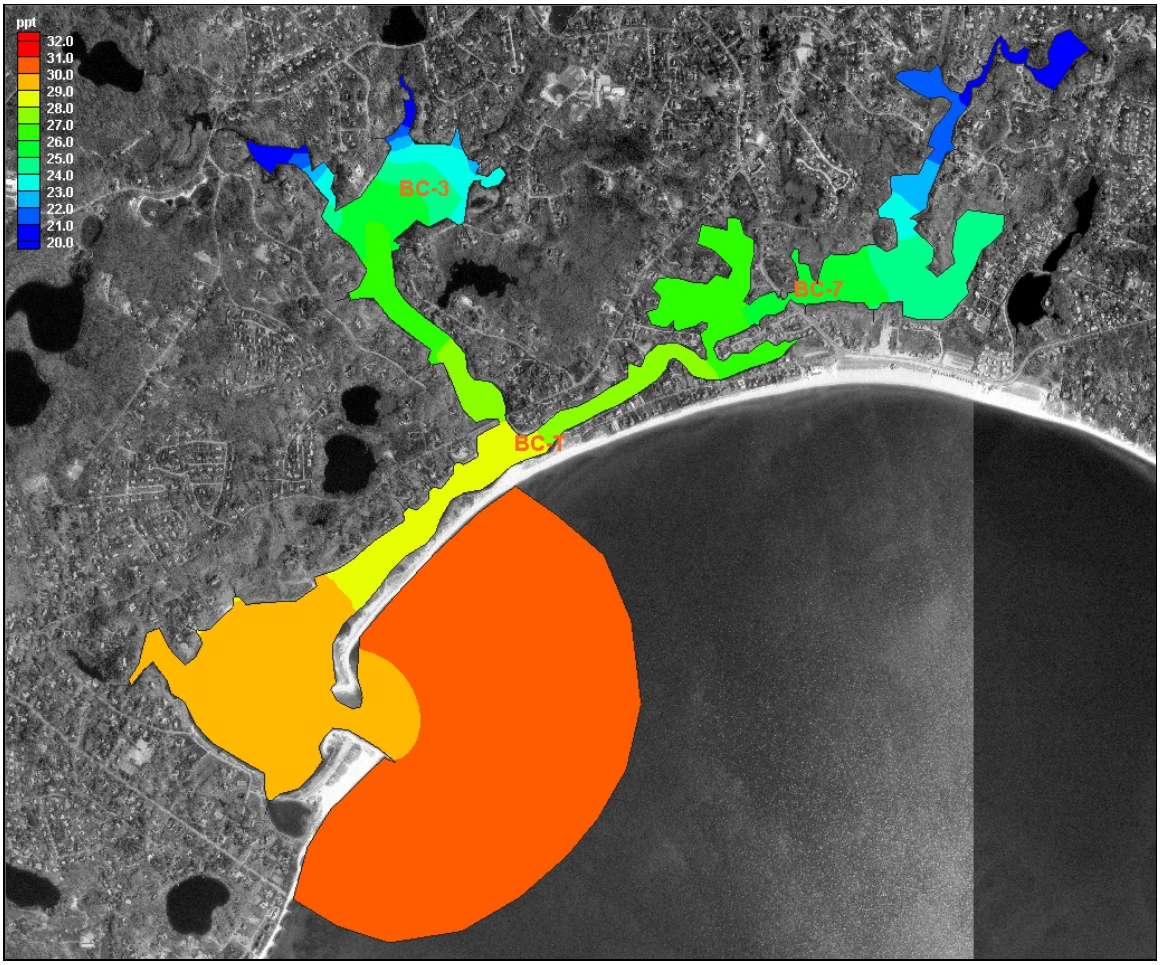


Figure VI-6. Contour plots of modeled salinity (ppt) in Centerville River System.

Table VI-4. Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic ("no-load") loading scenarios of the Centerville 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
Centerville River East	55.737	58.907	+5.7%	4.551	-91.8%
Scudder Bay	14.452	15.041	+4.1%	1.025	-92.9%
Centerville River West	9.463	10.011	+5.8%	0.907	-90.4%
East Bay	8.627	9.008	+4.4%	1.458	-83.1%
Surface Water Sources					
Pine Street Stream	3.452	3.742	+8.4%	0.597	-82.7%
Lake Elizabeth Stream	2.274	2.252	-0.0	0.093	-95.9%
Bumps River	16.912	17.863	+5.6%	0.364	-97.8%
Skunknet River	21.260	21.918	+3.1%	1.044	-95.1%

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 more than a 6% increase in watershed nitrogen load to the western end of Centerville River as a result of potential future development. Other watershed areas would experience similar load increases, for example the loads to East Bay would increase 4% 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 generally lower than existing conditions by over 80% overall and 90% in most areas.

For the build-out scenario, a breakdown of the total nitrogen load entering the Centerville 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:

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-5. Build-out sub-embayment and surface water loads used for total nitrogen modeling of the Centerville 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)
Centerville River East	58.907	0.449	5.469
Scudder Bay	15.041	0.685	-2.170
Centerville River West	10.011	0.718	3.526
East Bay	9.008	1.126	12.799
Surface Water Sources			
Pine Street Stream	3.742	-	-
Lake Elizabeth Stream	2.252	-	-
Bumps River	17.863	-	-
Skunknet River	21.918	-	-

Following development of the nitrogen loading estimates for the build-out scenario, the water quality model of Centerville 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. Total

N concentrations increased the most in the upper portion of the system, with the largest change at the upper portion of Centerville River (2.8%) and the least change occurred in East Bay (0.6%) which exchanges directly with Nantucket Sound. 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 Centerville River System. Sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Scudder Bay	**BC-3**	**0.524**	**0.536**	**+2.1%**
Bumps River	BC-4	0.451	0.458	+1.6%
Centerville River	BC-5	0.609	0.626	+2.8%
Centerville River	**BC-7**	**0.526**	**0.538**	**+2.2%**
Centerville River	BC-8	0.454	0.462	+1.7%
Centerville River	BC-9	0.389	0.394	+1.1%
East Bay	BC-10	0.349	0.351	+0.6%
Confluence of Bumps River and Centerville River	**BC-T**	**0.412**	**0.418**	**+1.4%**

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.

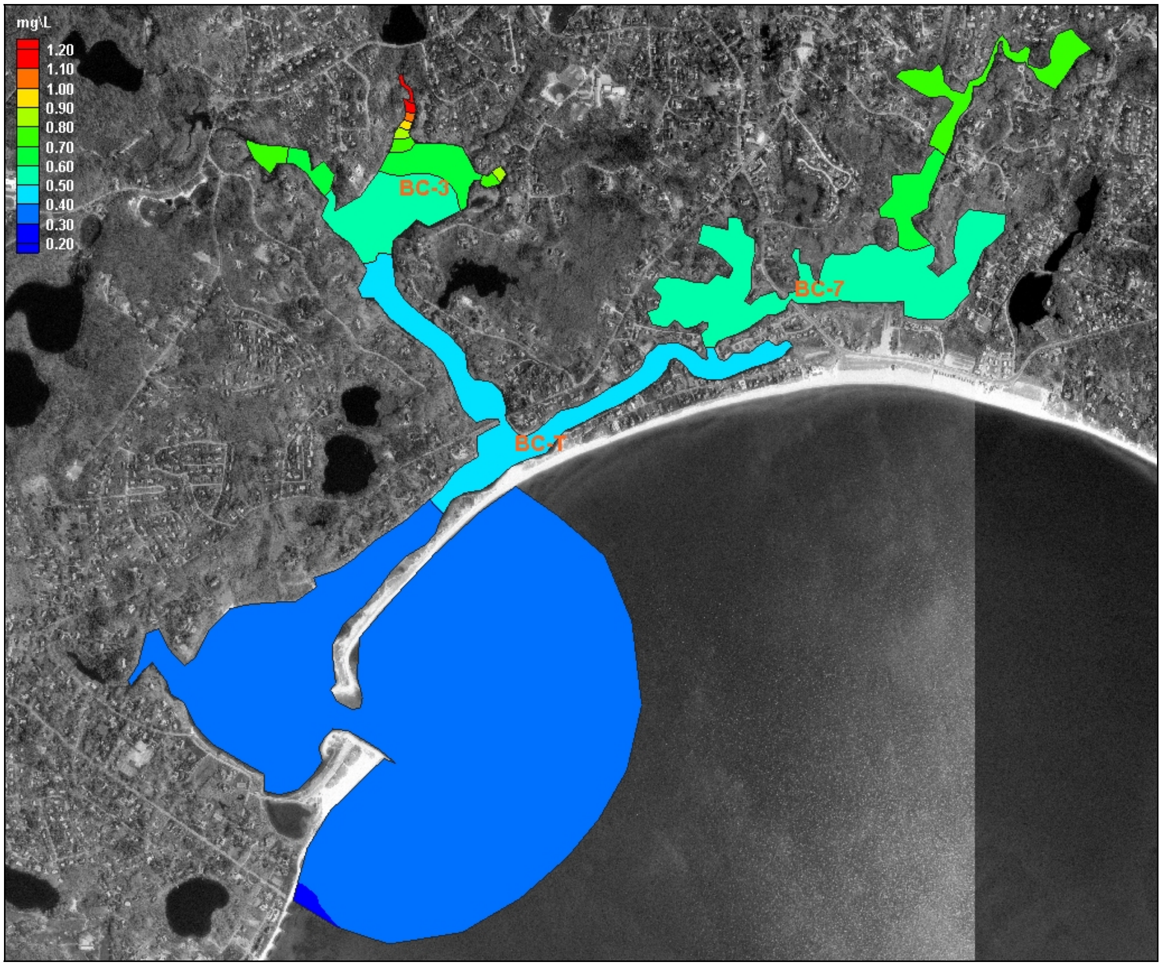


Figure VI-7. Contour plots of modeled total nitrogen concentrations (mg/L) in Centerville River System, for projected build-out loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Centerville River System (BC-T) is shown.

Table VI-7. "No anthropogenic loading" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of Centerville 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)
Centerville River East	4.551	0.449	3.847
Scudder Bay	1.025	0.685	-1.432
Centerville River West	0.907	0.718	3.038
East Bay	1.458	1.126	11.125
Surface Water Sources			
Pine Street Stream	0.597	-	-
Lake Elizabeth Stream	0.093	-	-
Bumps River	0.364	-	-
Skunknet River	1.044	-	-

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Again, total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was significant as shown in Table VI-8, with reductions greater than 40% occurring in Scudder Bay and the upper portions of Centerville River. 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 Centerville River System. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). Sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	no-load (mg/L)	% change
Scudder Bay	**BC-3**	**0.524**	**0.312**	**-40.5%**
Bumps River	BC-4	0.451	0.311	-30.9%
Centerville River	BC-5	0.609	0.328	-46.2%
Centerville River	**BC-7**	**0.526**	**0.326**	**-38.1%**
Centerville River	BC-8	0.454	0.320	-29.5%
Centerville River	BC-9	0.389	0.312	-19.8%
East Bay	BC-10	0.349	0.309	-11.6%
Confluence of Bumps River and Centerville River	**BC-T**	**0.412**	**0.314**	**-23.7%**

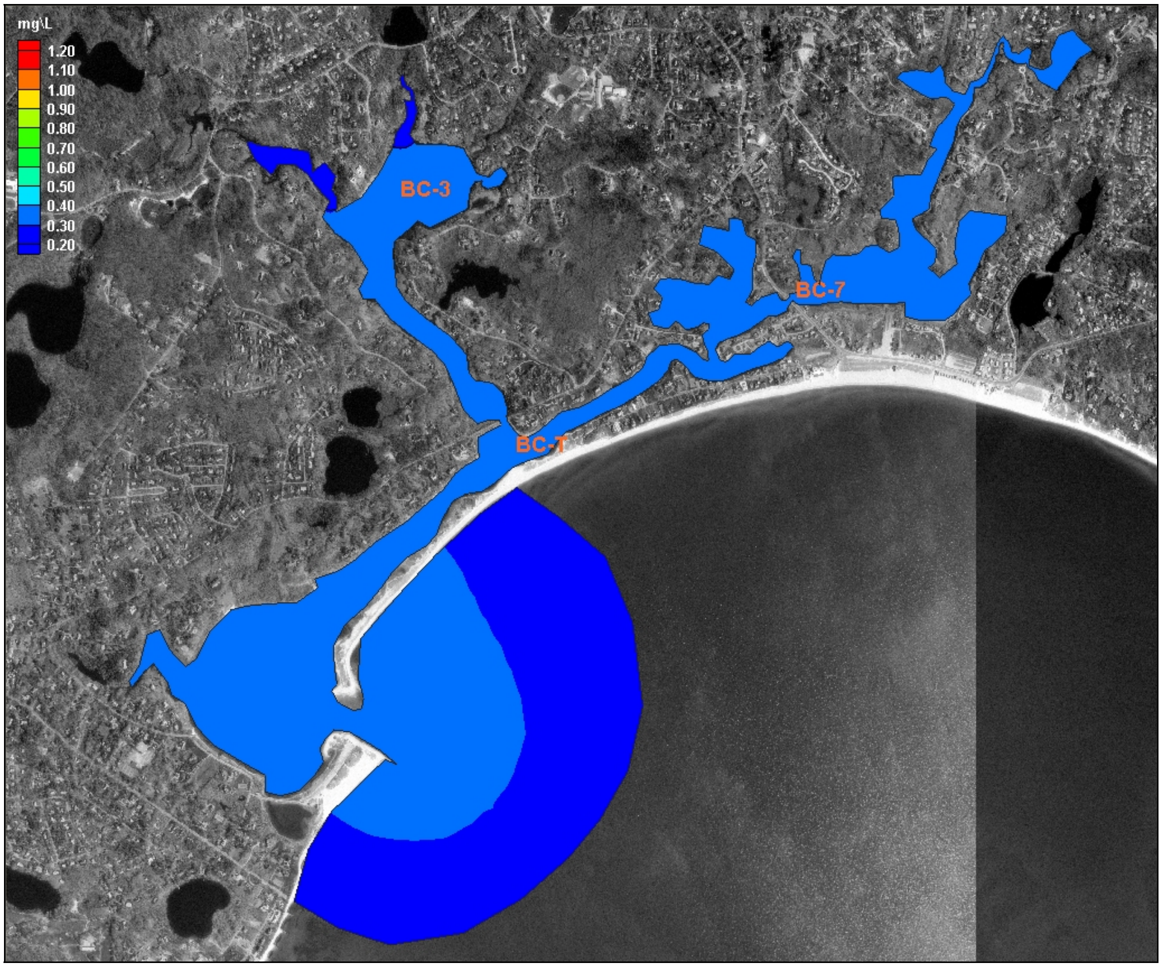


Figure VI-8. Contour plots of modeled total nitrogen concentrations (mg/L) in Centerville River System, for no anthropogenic loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Centerville River System (BC-T) is shown.

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 Centerville River embayment system, the MEP assessment is based upon data from the water quality monitoring database and MEP surveys of eelgrass distribution, benthic animal communities and sediment characteristics, and dissolved oxygen records conducted during the summer of 2003. These data form the basis of an assessment of this system's present health, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for these systems (Chapter VIII).

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

As a basis for a nitrogen thresholds determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed dissolved oxygen sensors within the upper portion of the Centerville River System, Scudder Bay, as well as closer to the inlet to the Centerville River system in East Bay and at the Town Landing in the mid River reach to record the frequency and duration of low oxygen conditions during the critical summer period. This work was conducted in 2003 in association with the Town of Barnstable.

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 Centerville River System was conducted for comparison to historic records (DEP Eelgrass Mapping Program, C. Costello). Temporal trends in the distribution of eelgrass beds are used by the MEP to assess the stability of the habitat and to determine trends potentially related to nutrient related water quality. Eelgrass beds can decrease within embayments in response to a variety of causes, but throughout almost all of the embayments within southeastern Massachusetts, the primary cause appears to be related to increases in embayment nitrogen levels. Within the Centerville River System, temporal changes in eelgrass distribution provides a strong basis for evaluating recent increases (nitrogen loading) or decreases (increased flushing-new inlet) in nutrient enrichment.

In areas that do not naturally support eelgrass beds, benthic animal indicators were used to assess the level of habitat health from "healthy" (low organic matter loading, high D.O.) to "highly stressed" (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stressed indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill, benthic population studies in Buzzards Bay (Woods Hole Oceanographic Institution) and New Bedford (SMAST), and more recently the Woods Hole Oceanographic Institution Nantucket Harbor Study (Howes *et al.* 1997). These data are coupled with the level of diversity (H') and evenness (E) of the benthic community and the total number of individuals to determine the infaunal habitat quality.

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 3.8 mg L^{-1}. Massachusetts State Water Quality Classification indicates that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1}. The tidal waters of the Centerville River System are currently listed under this Classification as SA. It should be noted that the Classification system represents the water quality that the embayment should support, not the existing level of water quality. It is through the MEP and TMDL processes that management actions are developed and implemented to keep or bring the existing conditions in line with the Classification.

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (Figure VII-1). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the embayment bottom within key regions of the Centerville River System (Figure VII-2). The sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployment. In addition periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a minimum deployment of 30 days within the interval from July through mid-September. All of the mooring data from the Centerville River embayment system was collected during the summer of 2003, specifically within the July to mid September period.

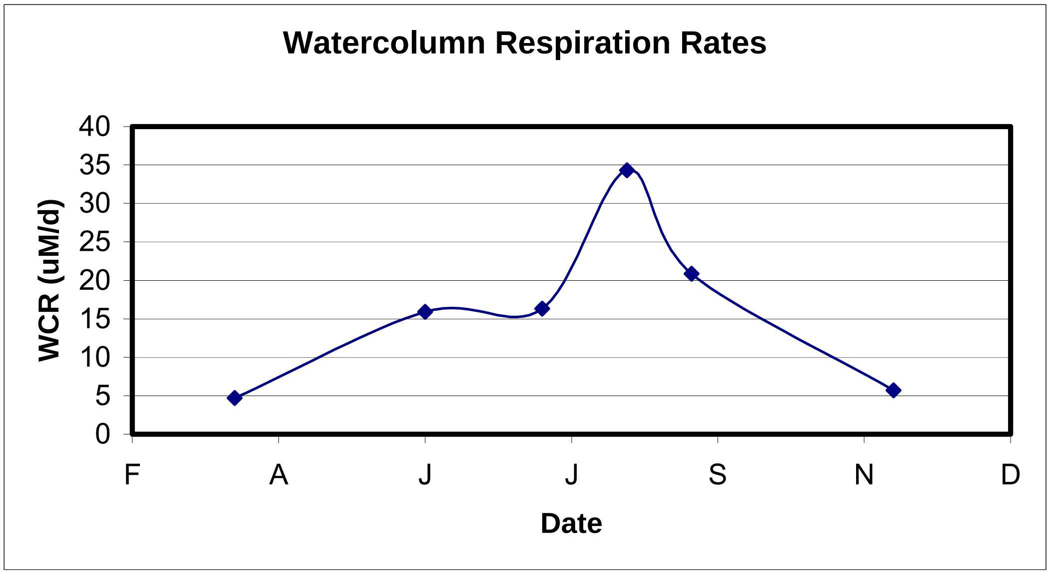


Figure VII-1. Average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

Dissolved oxygen moorings were deployed in the Centerville River estuary (Figure VII-2) during 2003 to assess summertime fluctuations. Similar to other embayments in southeastern Massachusetts, the Centerville River System evaluated in this assessment showed high frequency variation, apparently related to diurnal and sometimes tidal influences. Nitrogen enrichment of embayment waters generally manifests itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. The high degree of temporal variation in bottom water dissolved oxygen concentration at each mooring site, underscores the need for continuous monitoring within these systems.

Dissolved oxygen and chlorophyll a records were examined both for temporal trends and to determine the percent of the 37-40 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 nitrogen enriched conditions and some habitat quality impairment at each of the mooring sites within the estuary (Figures VII-3 through VII-5). The oxygen data throughout the estuary is consistent with elevated organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of estuarine systems

Figure VII-2. Aerial Photograph of the Centerville River embayment system in the Town of Barnstable showing locations of Dissolved Oxygen mooring deployments conducted in the summer of 2003.

Table VII-1a. Number of days during deployment of in situ sensors that bottomwater oxygen was below various benchmark levels.

Mooring ID.	Start Date	End Date	Total Deployment (Days)	<6 mg/L Duration (Days)	<5 mg/L Duration (Days)	<4 mg/L Duration (Days)	<3 mg/L Duration (Days)
Centerville Town Landing	8/9/2003	9/18/2003	39.9	14.20	6.57	3.02	1.01
			Mean	0.24	0.15	0.16	0.10
			Min	0.01	0.01	0.01	0.01
			Max	1.70	0.84	0.39	0.18
			S.D.	0.29	0.18	0.12	0.05
Centerville Scudder Bay	8/9/2003	9/18/2003	40.0	7.56	2.82	0.48	0.10
			Mean	0.18	0.11	0.12	0.05
			Min	0.01	0.01	0.02	0.02
			Max	0.74	0.40	0.32	0.08
			S.D.	0.14	0.09	0.14	0.04
Centerville East Bay	8/9/2003	9/15/2003	36.8	20.21	6.67	0.14	0.00
			Mean	0.47	0.17	0.03	N/A
			Min	0.01	0.01	0.01	0.00
			Max	1.92	0.74	0.06	0.00
			S.D.	0.50	0.14	0.02	N/A

Table VII-1b. Frequency distribution from Water Quality Monitoring grab sampling of bottom water oxygen. Number represent the number of field dates that oxygen was observed within the noted range (<2 mg L^{-1}, 2-3 mg L^{-1}, etc.).

Location		Range of Measured Bottomwater DO Levels (mg L^{-1})							
		<2	2-3	3-4	4-5	5-6	>6	total	Min
Scudder Bay	BC-3	0	0	2	7	12	8	29	3.4
Bumps River Estuary	BC-4	0	0	0	3	11	14	28	4.5
Upper N Centerville River	BC-5	0	1	10	9	4	5	29	2.7
Upper Centerville River	BC-7	0	0	0	2	16	12	30	4.8
Mid Centerville River	BC-8	0	0	0	4	8	13	25	4.5
Lower Centerville River	BC-9	0	0	0	0	5	25	30	5.2
East Bay	BC-10	0	0	0	0	6	25	31	5.7
Centerville Harbor	BC-11	0	0	0	0	0	32	32	6.2

Table VII-2. Duration (number of days during deployment period) 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. The mean in the final column is the average level over the deployment.

Mooring Id.	Start Date	End Date	Total Deployment (Days)	>5 ug/L Duration (Days)	>10 ug/L Duration (Days)	>15 ug/L Duration (Days)	>20 ug/L Duration (Days)	>25 ug/L Duration (Days)
Centerville Town Landing	8/9/2003	9/18/2003	31.7	30.79	22.50	16.00	12.92	11.17
Mean Chl Value = 22.0 ug/L			Mean	2.20	0.51	0.33	0.30	0.27
			Min	0.04	0.04	0.04	0.04	0.04
			Max	14.25	7.96	4.54	3.17	2.58
			S.D.	4.28	1.17	0.72	0.61	0.50
Centerville Scudder Bay	8/9/2003	9/18/2003	32.2	32.04	22.67	14.38	9.92	6.88
Mean Chl Value = 18.02 ug/L			Mean	8.01	0.45	0.28	0.20	0.17
			Min	0.04	0.04	0.04	0.04	0.04
			Max	17.71	7.42	3.29	0.63	0.54
			S.D.	7.63	1.05	0.46	0.17	0.15
Centerville East Bay	8/9/2003	9/18/2003	39.3	16.71	3.79	1.08	0.00	0.00
Mean Chl Value = 5.10 ug/L			Mean	0.19	0.29	0.08	N/A	N/A
			Min	0.04	0.04	0.04	0.00	0.00
			Max	0.71	0.54	0.25	0.00	0.00
			S.D.	0.16	0.16	0.06	N/A	N/A

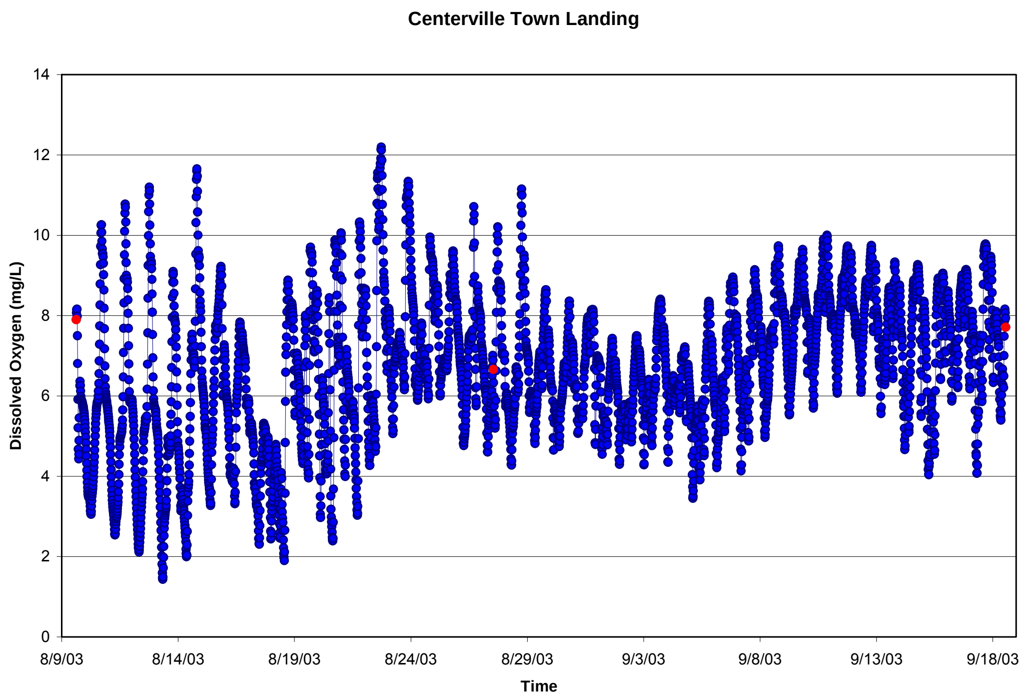


Figure VII-3. Bottom water record of dissolved oxygen at the Centerville Town Landing station, summer 2003. Calibration samples represented as red dots.

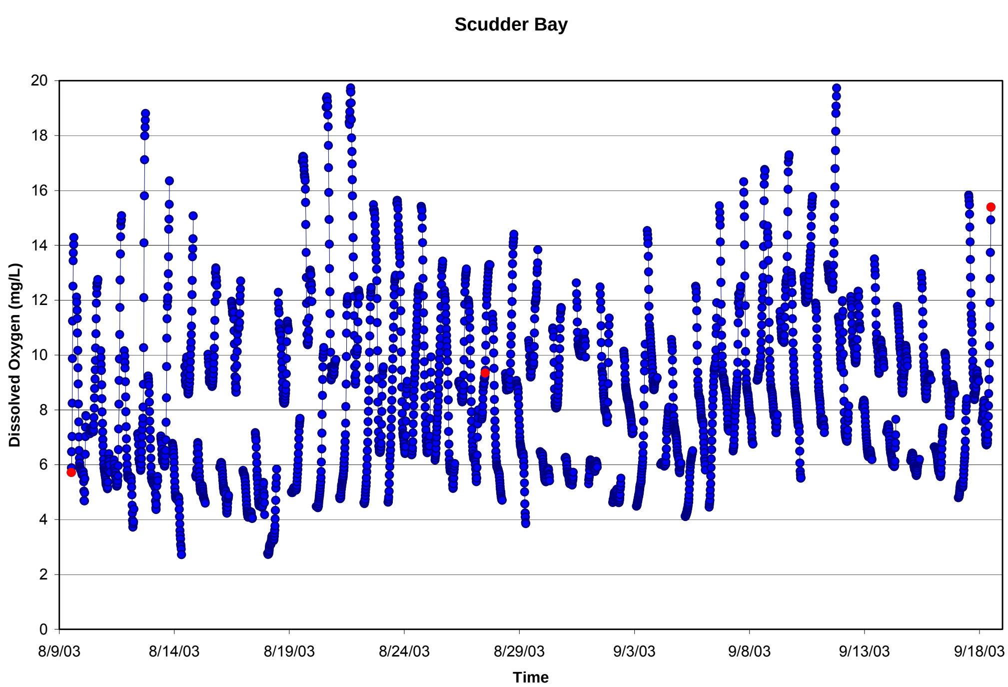


Figure VII-4. Bottom water record of dissolved oxygen in the Scudder Bay station, summer 2003. Calibration samples represented as red dots.

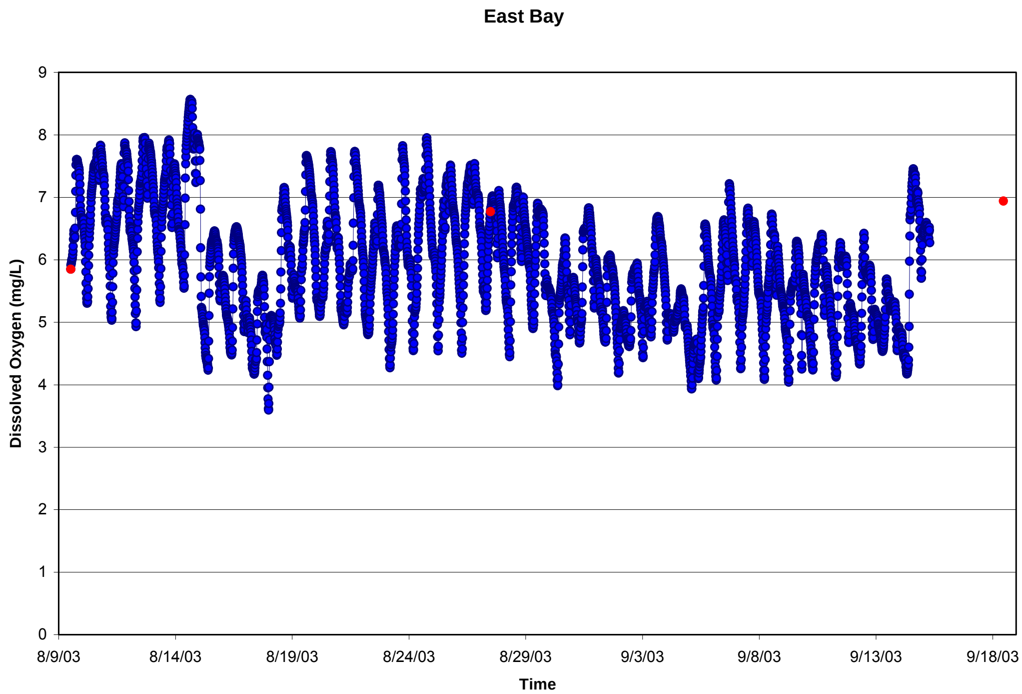


Figure VII-5. Bottom water record of dissolved oxygen in the East Bay station, summer 2003. Calibration samples represented as red dots.

There is a clear gradient in the extent of oxygen depletion from the upper to the lower portions of the system, as shown in both the mooring and Water Quality Monitoring data (Table VII-1). The lowest oxygen levels were seen in the upper Centerville River salt marsh and in Scudder Bay and oxygen levels increased moving seaward to the lower reach of the Centerville River and East Bay. However, the oxygen levels within the Centerville River System had daily minima lower than the offshore basin of Centerville Harbor which had a minimum recorded bottom water oxygen level of 6.2 mg L^{-1} over 22 sampling dates. This latter comparison is added support for the contention that the Centerville Estuary is nitrogen enriched over offshore waters.

Overall, oxygen depletion was observed at each of the mooring sites (Table VII-1a), but the degree of depletion varied between sites. East Bay generally maintained oxygen levels above 4 mg L-1, but with frequent excursions below 5 mg L^{-1}. These oxygen conditions represent a moderate level of impairment to benthic infaunal communities associated with embayments. Oxygen levels in the upper reach of the Centerville River showed even lower levels periodically into the 2-3 mg L^{-1} range, although generally with daily minima above 4 mg L^{-1}. The central region of Scudder Bay was clearly nitrogen enriched, but generally maintained daily oxygen minima above 4 mg L^{-1} and typically above 5 mg L^{-1}. This basin showed its enrichment mainly through the large daily excursions with daily maxima typically >11 mg L^{-1}.

The use of only the duration of oxygen below, for example 4 mg L^{-1}, can underestimate the level of habitat impairment in these locations. The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production, oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). This pattern was seen in both of the upper system sites, further supporting the contention that the upper basins are currently nitrogen enriched. The upper tidal reaches of each estuary have the largest daily oxygen excursion, with daily excursions in excess of 6 mg L^{-1} common, whereas the in the lower system, East Bay, showed more modest excursions of ~3 mg L^{-1} . This further supports the assessment of nitrogen enrichment.

The observations by the continuously recording oxygen sensors were supported by the more spatially distributed grab sample data collected by the Town of Barnstable Water Quality Monitoring Program (Table VII-1b). These data support general observation that oxygen levels throughout the embayment basins of the Centerville River System are generally >4 mg L^{-1} and typically >5 mg L^{-1}, during the summer months. In contrast the salt marsh influenced sites of upper Centerville River and Scudder Bay showed greater depletions, with levels in the 3-4 mg L^{-1} range on 34% and 7% of the dates (N=29), respectively.

The spatial pattern of oxygen depletion within the Estuary was consistent with the measured chlorophyll a levels (Table VII-2, Figures VII-6 to VII-8). The level of oxygen depletion was directly related to the amount of chlorophyll a in the watercolumn. This is consistent with nitrogen enrichment resulting in increased phytoplankton biomass and subsequent organic matter deposition and decay being the process controlling oxygen level in the Centerville River System. Chlorophyll a levels in East Bay were indicative of a relatively healthy to moderately enriched basin. In East Bay chlorophyll a averaged 5.1 ug L^{-1} and was almost always <10 ug L^{-1}. The upper reach of the Centerville River and Scudder Bay supported higher phytoplankton biomass with average levels ~20 ug L^{-1} and blooms producing consistent levels of 30 ugL^{-1}.

It is important to evaluate the level of oxygen excursion and depletion relative to the functional habitats involved. In the upper Centerville River the tidal river serves as the central salt marsh creek to a rather large and healthy salt marsh which is just upgradient of the upper river oxygen mooring. In addition, there is a smaller separate salt marsh adjacent the mooring at this specific location. Dissolved oxygen levels recorded by this mooring reflect the oxygen conditions of the ebbing tidal waters from these wetlands areas. The low oxygen levels are consistent with a salt marsh tidal creek, where the organic matter enriched sediments support high levels of oxygen uptake at night and deplete the overlying waters. While oxygen depletion to 3 mg/L would indicate impairment in an embayment like the East Bay basin, it is consistent with the organically enriched nature of smaller salt marsh creeks.

The results of the summer oxygen and chlorophyll a studies are consistent with the absence of eelgrass throughout the Centerville River/Harbor System and the spatial distribution of infaunal community types and quality of habitat.

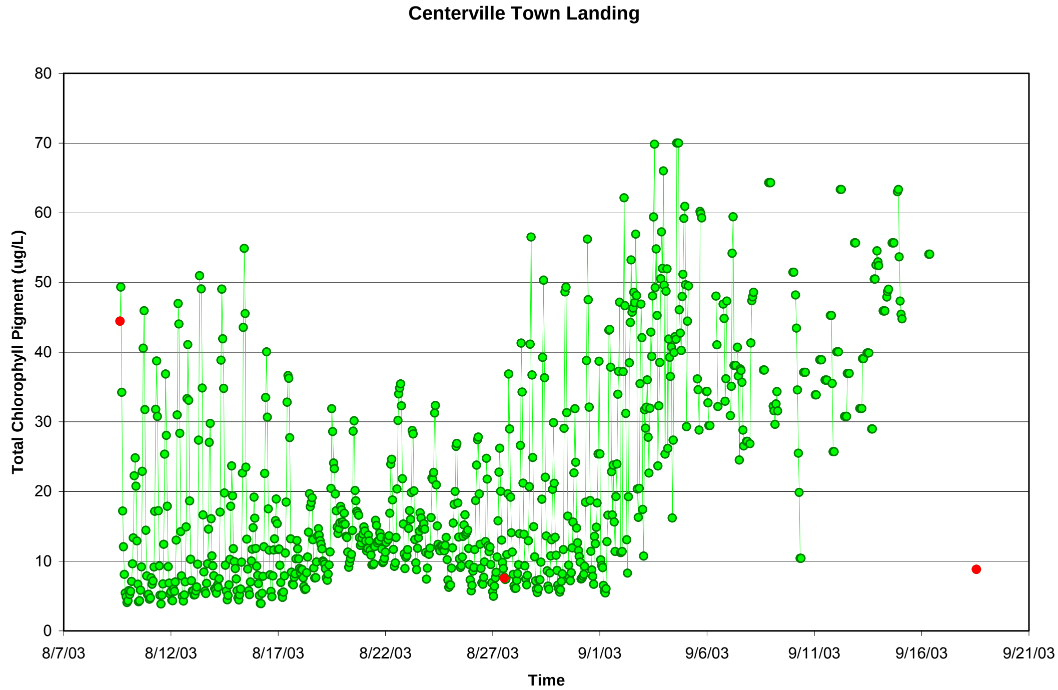


Figure VII-6. Bottom water record of Chlorophyll-a in the Centerville Town Landing station, summer 2003. Calibration samples represented as red dots.

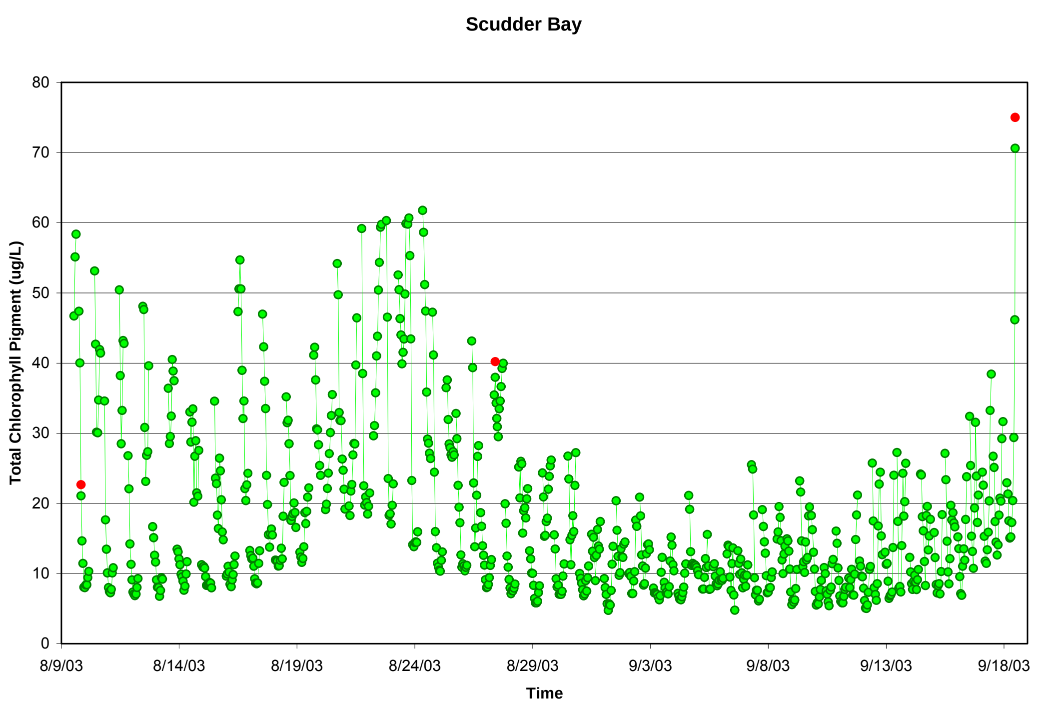


Figure VII-7. Bottom water record of Chlorophyll-a in the Scudder Bay station, summer 2003. Calibration samples represented as red dots.

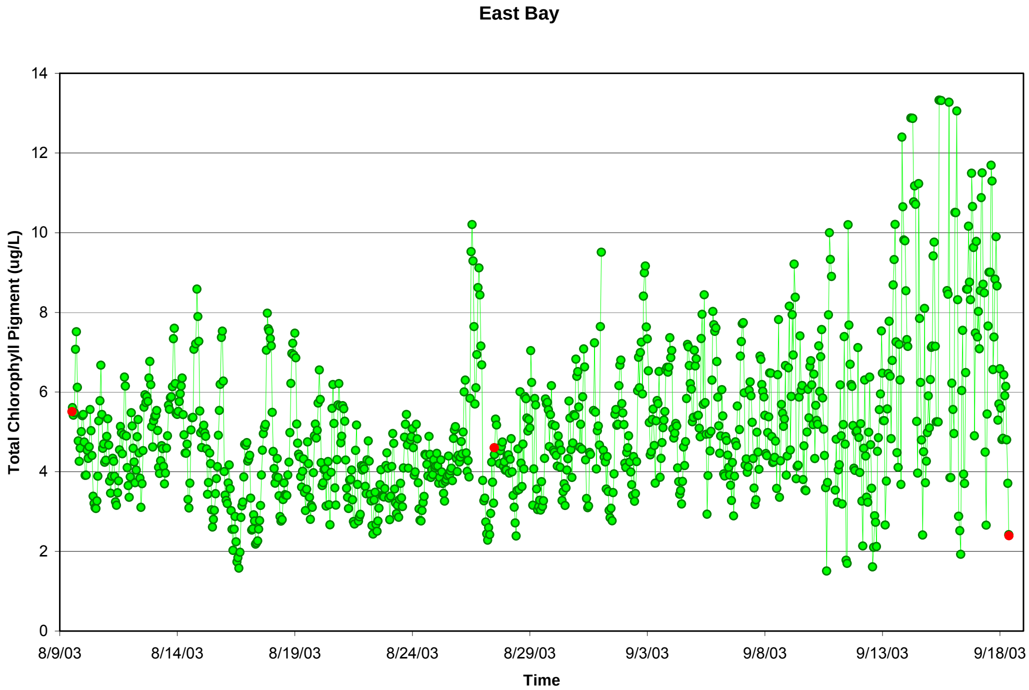


Figure VII-8. Bottom water record of Chlorophyll-*a* at the East Bay station, summer 2003. Calibration samples represented as red dots.

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data was conducted for the Centerville River System by the DEP Eelgrass Mapping Program as part of the MEP Technical Team. Surveys were conducted in 1995 and 2001, as part of this program. Additional analysis of available aerial photos from 1951 was used to reconstruct the eelgrass distribution prior to any substantial development of the watershed. The 1951 data were only anecdotally validated, while the 1995 and 2001 maps were field validated. The primary use of the data is to indicate (a) if eelgrass once or currently colonizes a basin and (b) if large-scale system-wide shifts have occurred or are presently underway. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 1995 to 2001 (Figure VII-9 and VII-10); the period in which watershed nitrogen loading significantly increased to its present level. This temporal information can be used to determine the stability of the eelgrass community.

At present, eelgrass beds are not present within the Centerville River System. In addition, to the DEP mapping, this has been confirmed by the multiple MEP staff conducting the infaunal and sediment sampling and the mooring studies and by the Town of Barnstable Shellfish Department. It should be noted that although the eelgrass beds present in Centerville River as observed in the 1951 aerial photography are no longer apparent, there does appear to be eelgrass remaining just outside the inlet to Centerville River at the discharge point to the Harbor. The current lack of eelgrass beds in Centerville River is expected given the high chlorophyll a and low dissolved oxygen levels and water column nitrogen concentrations within this system. However, it appears that the lower portions of the Centerville Harbor system (principally Centerville River) had water quality conditions capable of supporting eelgrass (except in the deeper channels and basin depths) in 1951. It is important to note that very stable eelgrass beds were observed in each survey (1951, 1995, and 2001) outside the tidal inlet to the east and west.

The present absence of eelgrass throughout the Centerville River System is consistent with the observed oxygen depletions in each basin and the chlorophyll levels as well as functional basin types. Eelgrass in the upper reaches is unlikely as these areas are strongly influenced by surrounding wetlands that do not typically support eelgrass habitat. However, basins like the Centerville River channel (from the Town Landing to East Bay) and especially subembayments like East Bay do generally support eelgrass habitat under low to moderate nitrogen loading conditions. The distribution of eelgrass in 1951 is fully consistent with this functional analysis. It appears that the eelgrass beds which have persisted just outside of the tidal inlet extended into East Bay and up the Centerville River to the mouth of Bumps River in 1951. This is consistent with the lower nitrogen loading at that time and the resultant higher sustained oxygen levels and lower chlorophyll levels (high light penetration) that should have existed at that time based upon population data. Note that nitrogen loads originating in the Bumps River sub-watershed <u>and</u> in <u>the Centerville River east of the mouth of Bumps River also affect the lower reach of the River and East Bay on the ebbing tide. As a result, the entire watershed of the Centerville River system affects conditions in East Bay.</u>

The presence of eelgrass beds just outside of the tidal inlet in each of the DEP assessments supports the presence of eelgrass beds within East Bay at earlier and lower nitrogen levels. The beds just outside of the inlet and much of East Bay have the same water depth and same tidal range, so the major environmental differences between the sites are related to nitrogen enrichment. It appears from the eelgrass and water quality information that eelgrass beds within East Bay and the lower Centerville River should be the target for restoration and that this habitat should be recovered with appropriate nitrogen management.

Other factors which influence eelgrass bed loss in embayments can also be at play in the Centerville River System, though the recent loss seems completely in-line with nitrogen enrichment. Nevertheless, a brief listing of non-nitrogen related factors is useful. Eelgrass bed loss does not seem to be directly related to mooring density, as loss in East Bay was from both mooring and non-mooring areas and the lower Centerville River is not a boat mooring area. Similarly, pier construction and boating pressure may be adding additional stress in nutrient enriched areas, but do not seem to be the overarching factor. It is not possible at this time to determine the potential effect of shellfishing on eelgrass bed distribution, although it should be noted that the System has been an important shellfish area. At present the Centerville River System (Centerville River Segment MA96-04_2004; Bumps River Segment MA96-02_2004) are listed on the MassDEP 2004 Integrated List of waters impaired by pathogens), which typically reduces shellfishing pressure on an embayment.

However, the lack of eelgrass within the main tidal channel into East Bay is certainly associated with the periodic maintenance dredging for navigation. Dredging was recently performed, however, this represents a small region within the much larger area that has lost eelgrass and some dredging will likely always be required to maintain the tidal flushing of this system at its highest level to maintain healthy habitat quality, even after nitrogen management strategies have been implemented. Under present conditions, a reduction in tidal flushing will result in magnifying the eutrophic conditions of the entire system.

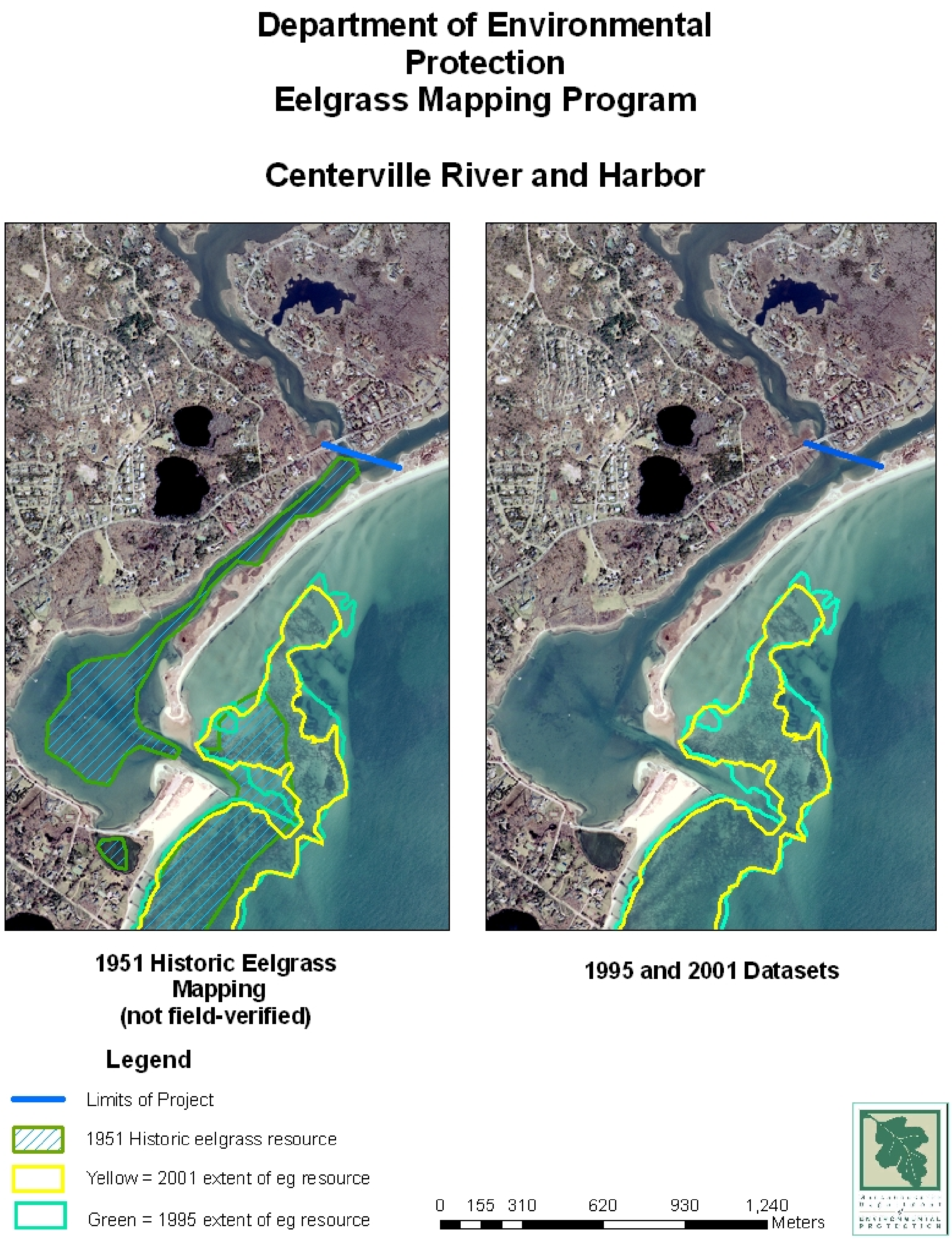


Figure VII-9. Eelgrass bed distribution within the Centerville Harbor System. The 1951 coverage is depicted by the green thatched outline inside of which circumscribes the eelgrass beds. The green (1995) and yellow (2001) areas were mapped by DEP. All data was provided by the DEP Eelgrass Mapping Program.

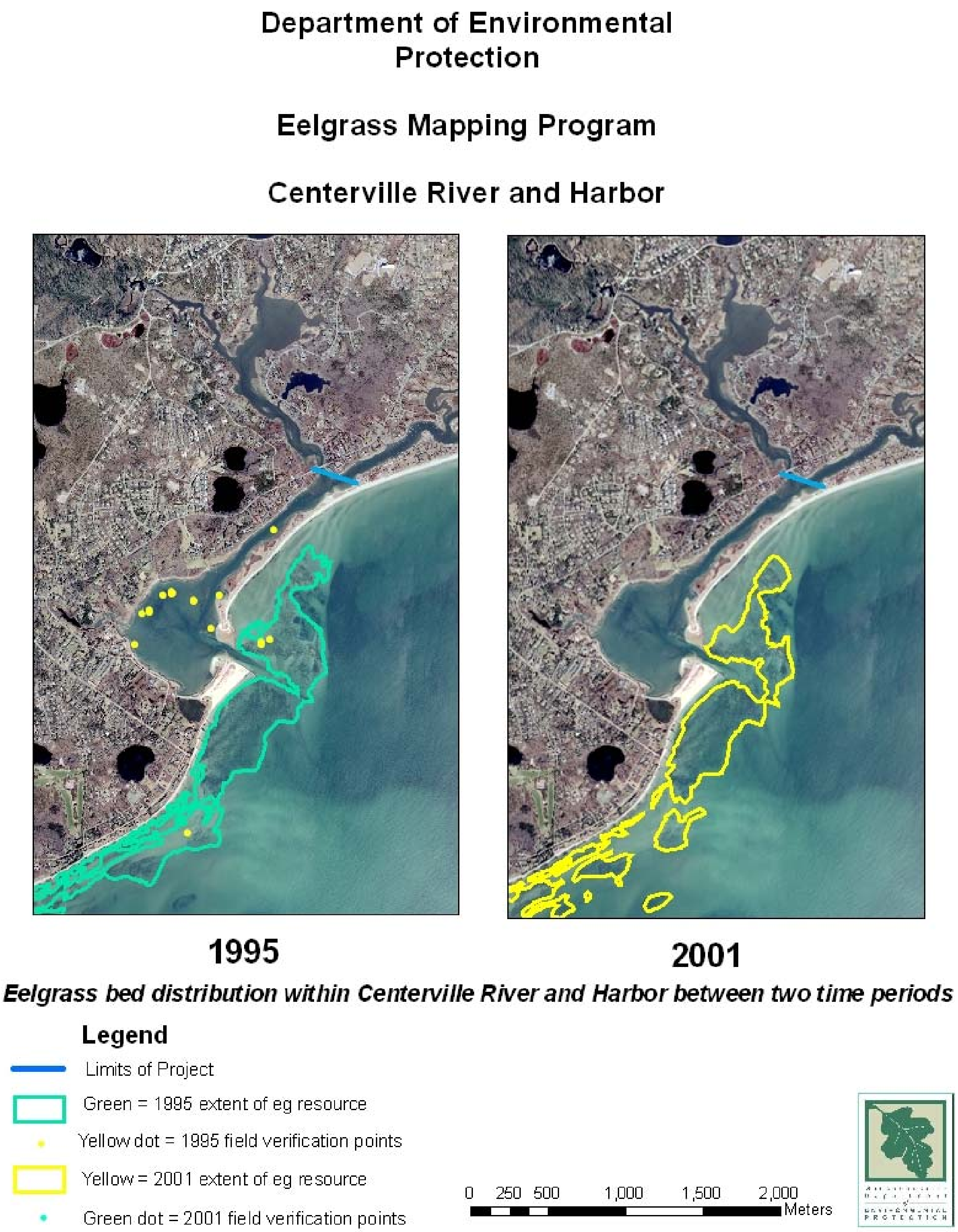


Figure VII-10. Eelgrass bed distribution within the Centerville Harbor System in addition to field verification points. The 1995 and 2001 coverage is depicted by the green and yellow thatched outlines inside of which circumscribes the eelgrass beds. The green (1995) and yellow (2001) areas were mapped by DEP. All data was provided by the DEP Eelgrass Mapping Program.

It is not possible to determine a general idea of short- and long-term rates of change in eelgrass coverage from the mapping data, since there is only limited temporal data with no eelgrass found in the recent surveys. However, it is possible to utilize the 1951 coverage data as an indication that a minimum eelgrass bed area that might be recovered, on the order of 52 acres, if nitrogen management alternatives were implemented (Table VII-3). Note that restoration of this habitat will necessarily result in restoration of other resources throughout the Centerville River/Harbor System and in the region of Scudder Bay and the Town Landing located low in the system, but distant from the inlet and influence of strong tidal flushing. Since East Bay is influenced by waters ebbing from both branches of the entire up-gradient basins, its nitrogen management will de facto result in a lowering of nitrogen enrichment throughout the Centerville River System and therefore an improvement of infaunal habitats in Scudder Bay and the upper Centerville River, which have traditionally only supported infaunal habitat, will support eelgrass after restoration, given its distance from the inlet and limited circulation,

The relative pattern of these data is consistent with the results of the oxygen and chlorophyll a patterns described in the previous section and the benthic infauna analysis, below.

Table VII-3. Changes in eelgrass coverage in the Centerville River portion of the Centerville Harbor system within the Town of Barnstable over the past half century (C. Costello).

EMBAYMENT	1951 (acres)	1995 (acres)	2001 (acres)	% Difference (1951 to 2001)
Centerville River	52.18	0.00	0.00	100%
There is presently no eelgrass in the Centerville River portion of the Centerville Harbor system.				

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 21 locations throughout the Centerville River System (Figure VII-11). In some cases multiple assays were conducted. 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, given the loss of eelgrass beds, portions of the Centerville River System are clearly impaired by nutrient overloading. However, to the extent that it can still support healthy infaunal communities, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

Analysis of the evenness and diversity of the benthic animal communities was also used to support the density data and the natural history information. The evenness statistic can range from 0-1 (one being most even), while the diversity index does not have a theoretical upper limit. The highest quality habitat areas, as shown by the oxygen and chlorophyll records , 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. However, the number of species and individuals must also be taken into account as a high diversity can be achieved in a population decimated by organic loading. Also, as stated above, the specific species must also be examined as a large number of stress indicator species (e.g. Capitellids) or intermediate quality species (*Amplesca*) would be indicative of a significantly or moderately impaired environment, respectively, even if the number of individuals and species is relatively high.

The Infauna Study indicated that most of the infaunal habitat within the Centerville River System is either presently either healthy or only moderately impaired, when evaluated based upon the function type of the basins (i.e. embayment versus salt marsh creek/pond) Table VII-4).

The Bumps River estuarine reach and the lower Centerville River currently support healthy infaunal animal habitat for a coastal embayment/tidal river on Cape Cod. These areas support large numbers of individuals (generally >500) and species (up to 32/sample), with very high Diversity (H' 3.2-4.3) and Eveness (>0.75). The basin of East Bay, which is depositional and receives the ebb tidal waters from the entire estuarine system, is presently showing moderate impairment. This moderate impairment is seen primarily in the dominance of amphipod mats throughout the central basin (*Amplesca*). While the community is supportive of generally high numbers of individuals, the number of species, diversity and Eveness are reduced. Amphipod mats represent a valuable and productive resource, but are indicative of a habitat transitioning from healthy to stressful conditions, particularly as relates to organic matter loading from nitrogen enrichment. The upper Centerville River and the associated mid reach of the Centerville River are dominated by salt marsh conditions. Under these conditions the number of individuals and species is high and the Diversity and Eveness approximates healthy conditions even for an embayment. The species present are typical of salt marsh creeks, which generally have organic matter tolerant species (e.g. *Streblospio*) and crustaceans and mollusks. The upper reaches of the Centerville River appear to be currently supporting healthy infaunal habitat. Similarly, Scudder Bay exhibits infauna habitat quality indicative of organic matter enrichment. However, this basin supports salt marsh areas around its margins and a shallow central basin. It appears that this basin is functioning at least partially as a salt marsh pond and was evaluated as such by the MEP Technical Team. Therefore, this basin appears to be presently moderately impaired, based upon its moderate-low number of species and moderate-low (100) numbers of individuals. The nutrient enriched conditions within this basin appear to be beyond the accommodation of the infaunal community and nitrogen management will be needed to restore this benthic resource.

The MEP Surveys did detect a small localized area of degraded infaunal habitat within the Centerville River System, a small lagoon altered for boats directly inland of the western stretch of Craigville Beach. This small basin contains a shallow sill across its entrance to the River and a "deep" basin (Figure V-6). The basin configuration and sill create a localized depositional area with localized bottom water hypoxia. The result is a significantly degraded yet small altered basin, which requires a local rather than system-wide solution (restoring proper basin configuration and flow). In advance of completion of the MEP analysis, the information regarding the specifics of this sill and associated impacts to the associated deep basin were communicated to the Town of Barnstable by the MEP Technical Team. In a proactive fashion, the Town of Barnstable subsequently invoked the necessary process to dredge the area of concern in order to restore a proper basin configuration and restore effective exchange of tidal waters. Dredging of the sill was completed in the winter of 2005 going into 2006 (December – January timeframe).

The infaunal study indicated an overall system supporting generally healthy to moderately impaired infaunal habitat relative to the ecosystem types represented. The Centerville River System is a complex estuary composed of 4 functional types of component basins: an embayment (East Bay, a salt marsh pond/embayment (Scudder Bay), a tidal salt marsh (upper Centerville River) and a tidal river (mid-lower Centerville River). Each of these 4 functional components has different natural sensitivities to nitrogen enrichment and organic matter loading. Evaluation of infaunal habitat quality must consider the natural structure of each system and the types of infaunal communities that they support.

These results of the MEP benthic analysis are integrated into the assessment of habitat quality throughout the Centerville River System relative to nitrogen levels and thresholds presented in Chapter VIII.

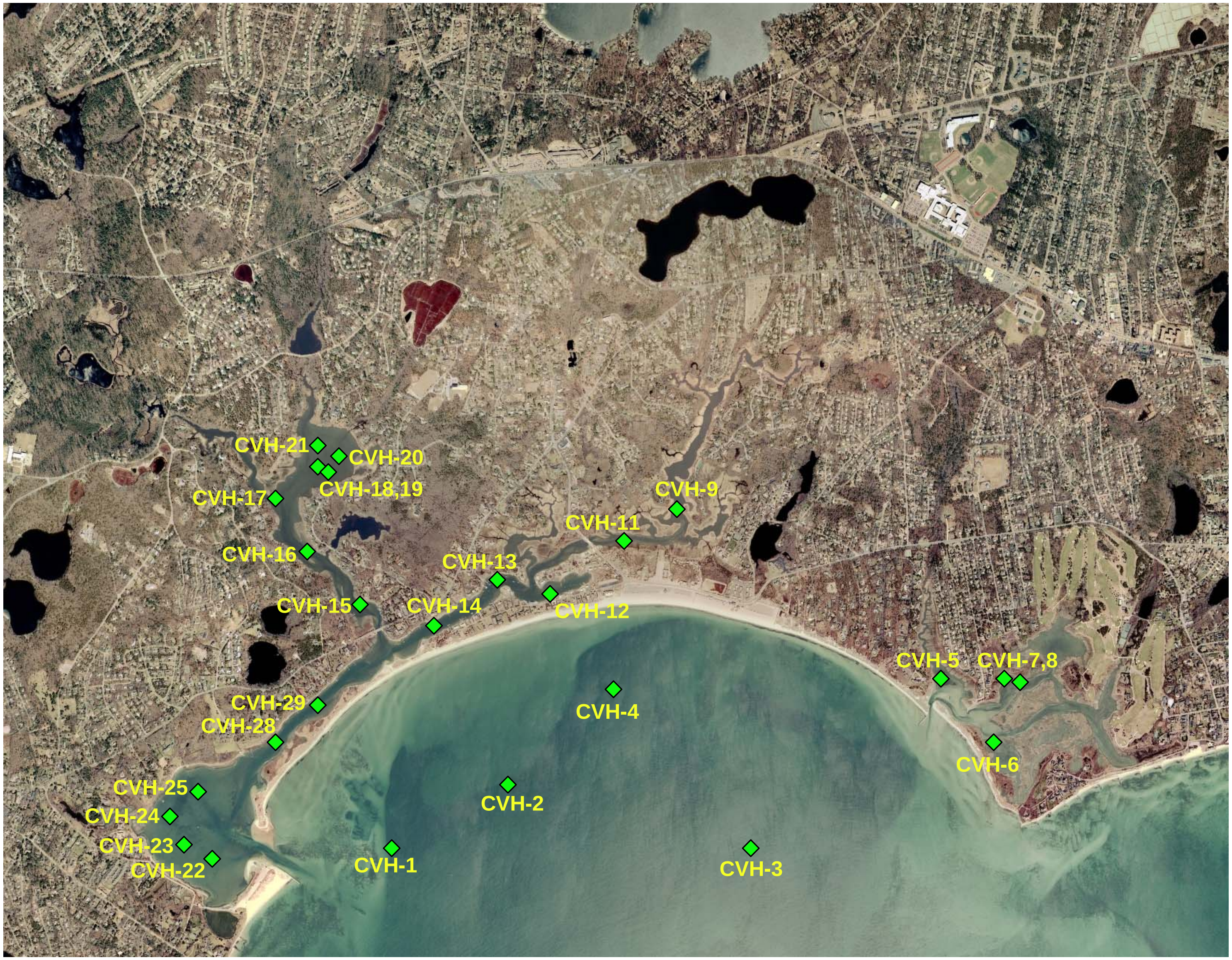


Figure VII-11. Aerial photograph of the Centerville River embayment system showing location of benthic infaunal sampling stations (green symbol).

Table VII-4. Benthic infaunal community data for the Centerville River Estuarine System. Estimates of the number of species adjusted to the number of individuals and community diversity (H') and Evenness (E) to allow comparison between locations (sample surface area, 0.0625 m^2). Station ID.'s with a "D" represent a second grab sample from the same general location.

Sub-Embayment		Sta I.D.	Total Actual Species	Total Actual Individuals	Species Calculated @ 75 Indiv.	Weiner Diversity (H')	Evenness (E)
Centerville River-East Bay/Bumps River-Scudder Bay System							
Scudder Bay		Sta. 18	6	104	6	2.51	0.97
		Sta.18D	4	72	N/A	1.66	0.83
		Sta. 20	5	144	5	1.68	0.72
		Sta. 21	3	88	3	1.24	0.78
Bumps River	Main River	Sta. 15	30	682	18	3.79	0.77
	Main River	Sta. 16	32	727	21	4.28	0.86
	Confluence with Scudder B.	Sta. 17	14	720	13	3.17	0.83
Upper Centerville River Marsh		Sta. 9	17	815	11	2.70	0.66
		Sta. 9D	14	641	12	3.19	0.84
		Sta. 11	17	1643	12	2.33	0.57
		Sta. 11D	12	412	9	2.14	0.60
East River lagoon*		Sta. 12	10	39	N/A	2.80	0.84
		Sta. 12D	8	33	N/A	2.41	0.80
Mid Centerville River		Sta. 13	8	184	8	2.66	0.89
		Sta. 14	11	256	11	2.58	0.75
		Sta. 14D	17	174	14	2.99	0.73
Lower Centerville River		Sta. 28	32	457	22	4.30	0.86
		Sta. 29	31	708	18	3.70	0.75
		Sta. 29D	20	325	15	3.44	0.80
East Bay		Sta. 22	9	144	9	2.91	0.92
		Sta. 23	6	264	6	1.34	0.52
		Sta. 24	5	944	4	1.27	0.55
		Sta. 25	4	256	4	1.55	0.77
		Sta. 25D	5	112	5	2.13	0.92

* Dredged lagoon deep basin and sill at entrance, structurally depositional with organic rich sediments and depleted

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

VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires integration of key habitat parameters (infauna and eelgrass), sediment characteristics, and nutrient related water quality information (particularly dissolved oxygen and chlorophyll) a). Additional information on temporal changes within each sub-embayment and its associated watershed nitrogen load further strengthen the analysis. These data were collected to support threshold development for the Centerville River System by MEP Team and were discussed in Chapter VII. Nitrogen threshold development builds on this data and links habitat quality to summer water column nitrogen levels from the baseline Town of Barnstable Water Quality Monitoring Program, conducted with assistance from Three Bays Preservation and technical support from the Coastal Systems Program at SMAST.

The Centerville River System is a complex estuary composed of 4 functional types of component basins: an embayment (East Bay), a salt marsh pond/embayment (Scudder Bay), a tidal salt marsh (upper Centerville River) and a tidal river (mid-lower Centerville River). Each of these 4 functional components has different natural sensitivities to nitrogen enrichment and organic matter loading. Evaluation of eelgrass and infaunal habitat quality must consider the natural structure of each system and the ability to support eelgrass beds and the types of infaunal communities that they support. At present, the Centerville River System is showing variations in nitrogen enrichment and habitat quality among its various component basins. In general the system is showing healthy to moderately impaired benthic habitat. However, the lower basins (e.g. lower Centerville River, East Bay) are clearly significantly impaired based on eelgrass criteria, as historical eelgrass beds have been lost and eelgrass is no longer present within the System.

Eelgrass: The present lack of eelgrass throughout the Centerville River System is consistent with the observed oxygen depletions in each basin and the chlorophyll levels and functional basin types comprising this estuary. The upper estuarine reaches, which are strongly influenced by surrounding wetlands, do not typically support eelgrass habitat, due to their naturally nutrient enriched shallow waters and salt marsh function. However, basins like the Centerville River channel (from the Town Landing to East Bay), and especially the sub-embayment of East Bay, typically do support eelgrass habitat under low to moderate nitrogen loading conditions. The distribution of eelgrass in 1951 is fully consistent with this functional analysis and the conclusion that the lower region of this Estuary (e.g. downstream of Bumps River mouth) is currently over its nitrogen threshold level that supports healthy eelgrass habitat.

Analysis of the eelgrass beds which have persisted just outside of the tidal inlet and extended in 1951 into East Bay and up the Centerville River to the mouth of Bumps River, supports the contention that the recent loss of eelgrass is the result of nitrogen enrichment, as the well flushed outer beds have been extremely stable over the past half century. These beds are at similar water depths and have the same tidal excursion as the historical bed areas within the lower estuary. Therefore, the major environmental differences between the sites appear to be directly related to nitrogen enrichment. The recent loss of beds from within the Centerville River System is also consistent with the lower nitrogen loading and the resultant higher sustained oxygen levels and lower chlorophyll levels (high light penetration) that should have existed at that time, based upon population data.

It appears from the eelgrass and water quality information that eelgrass beds within East Bay and the lower Centerville River should be the target for restoration and that this habitat should be recovered with appropriate nitrogen management. From the historical analysis, it appears that on the order of 52 acres of eelgrass habitat could be recovered, if nitrogen management alternatives were implemented. Note that restoration of this habitat will necessarily result in restoration of other resources throughout the Centerville River/Harbor System and in the region of Scudder Bay and the Town Landing located low in the system, but distant from the inlet and influence of strong tidal flushing. Since East Bay is influenced by waters ebbing from both branches of the entire up-gradient basins, its nitrogen management will de facto result in a lowering of nitrogen levels throughout the Centerville River System and therefore an improvement of infaunal habitats in Scudder Bay and the upper Centerville River, which have traditionally only supported infaunal habitat. Based upon the above analysis, eelgrass habitat should be the primary nitrogen management goal for the lower Centerville River System and infaunal habitat quality the management target for the upper reaches. These goals are the focus of the MEP management alternatives analysis presented in Chapter IX.

Water Quality: Overall, the oxygen levels within the major sub-basins to the Centerville River System are indicative of relatively healthy or only moderately impaired conditions, since the upper reaches are defined as infaunal habitats (e.g. historically have not supported eelgrass) and when their physical structure and natural biogeochemical cycling is considered. Similar to other embayments in southeastern Massachusetts, the upper Centerville River, Scudder Bay and East Bay basins of the Centerville River System evaluated in this assessment showed high frequency variation, apparently related to diurnal and sometimes tidal influences. Nitrogen enrichment of embayment waters generally manifests itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. The high degree of temporal variation in bottom water dissolved oxygen concentration at each mooring site, underscores the need for continuous monitoring within these systems.

Overall, oxygen depletion was observed at each of the mooring sites (Table VII-1a), but the degree of depletion varied between sites. East Bay generally maintained oxygen levels above 4 mg L-1, but with frequent excursions below 5 mg L^{-1}. These oxygen conditions represent a moderate level of impairment to benthic infaunal communities associated with embayments. Oxygen levels in the upper reach of the Centerville River showed even lower levels periodically into the 2-3 mg L^{-1} range, although generally with daily minima above 4 mg L^{-1}. The central region of Scudder Bay was clearly nitrogen enriched, but generally maintained daily oxygen minima above 4 mg L^{-1} and typically above 5 mg L^{-1}. This basin showed its enrichment mainly through the large daily excursions with daily maxima typically >11 mg L^{-1}.

The observations obtained by the continuously recording oxygen sensors were supported by the more spatially distributed grab sample data collected by the Town of Barnstable Water Quality Monitoring Program (Table VII-1b). These data support general observation that oxygen levels throughout the embayment basins of the Centerville River System are generally >4 mg L^{-1} and typically >5 mg L^{-1}, during the summer months. In contrast the salt marsh influenced sites of upper Centerville River and Scudder Bay showed greater depletions, with levels in the 3-4 mg L^{-1} range on 34% and 7% of the dates, respectively.

The spatial pattern of oxygen depletion within the Estuary was consistent with the measured chlorophyll a levels (Table VII-2, Figures VII-6 –VII-8). The level of oxygen depletion was directly related to the amount of chlorophyll a in the watercolumn. This is consistent with nitrogen enrichment resulting in increased phytoplankton biomass and subsequent organic matter deposition and decay being the process controlling oxygen level in the Centerville River

System. Chlorophyll a levels in East Bay were indicative of a relatively healthy to moderately enriched basin. In East Bay chlorophyll a averaged on 5.1 ug L^{-1} and was almost always <10 ug L^{-1}. The upper reach of the Centerville River and Scudder Bay supported higher phytoplankton biomass with average levels ~20 ug L^{-1} and blooms producing consistent levels of 30 ugL^{-1}.

Tidally averaged total nitrogen (TN) levels showed a similar pattern, consistent with the extent of oxygen depletion at each site and the chlorophyll a levels. Tidally averaged TN (Chapter VI) showed moderate enrichment in uppermost Bumps River (i.e. Scudder Bay, 0.53 mg N L^{-1}) and the salt marshes of upper Centerville River (0.63 mg N L^{-1}) and decline to levels <0.4 mg N L^{-1} in the lower Centerville River/East Bay portion of the system.

It is important to evaluate the level of oxygen excursion and depletion relative to the functional habitats involved. In the upper Centerville River the tidal river serves as the central salt marsh creek to a rather large and healthy salt marsh. Dissolved oxygen levels (and infaunal habitat quality) is consistent <u>with a salt marsh tidal creek, where the organic matter enriched sediments support high levels of oxygen uptake at night and deplete the overlying waters. While oxygen depletion to 3 mg/L would indicate impairment in an embayment like the East Bay basin, it is consistent with the organically enriched nature of smaller salt marsh creeks.</u>

The results of the summer oxygen and chlorophyll a studies are consistent with the absence of eelgrass (Section VII-3) throughout the Centerville River/Harbor System and the spatial distribution of infaunal community types and quality of habitat (Section VII-4).

Infaunal Communities: The infaunal study indicated an overall system supporting generally healthy to only moderately impaired infaunal habitat relative to the ecosystem types represented (i.e. embayment versus salt marsh creek/pond).

The Bumps River estuarine reach and the lower Centerville River currently support healthy infaunal animal habitat for a coastal embayment/tidal river on Cape Cod. These areas support large numbers of individuals (generally >500) and species (up to 32/sample), with very high diversity (H' 3.2-4.3) and Eveness (>0.75). The basin of East Bay, which is depositional and receives the ebb tidal waters from the entire Estuarine System, is presently showing moderate impairment. This moderate impairment is seen primarily in the dominance of amphipod mats throughout the central basin (*Amplesca*). While the community is supportive of generally high numbers of individuals the number of species, diversity and Eveness are reduced. Amphipod mats represent a valuable and productive resource, but transitional between healthy and stressful conditions, particularly as relates to organic matter loading from nitrogen enrichment. The upper Centerville River and the associated mid reach of the Centerville River are dominated by salt marsh conditions. Under these conditions the number of individuals and species is high and the diversity and Eveness approximates healthy conditions even for an embayment. The species present are typical of salt marsh creeks, which generally have organic matter tolerant species (e.g. *Streblospio*) and crustaceans and mollusks. The upper reaches of the Centerville River appear to be currently supporting healthy infaunal habitat. Similarly, Scudder Bay exhibits infauna habitat quality indicative of organic matter enrichment. However, this basin supports salt marsh areas around its margins and a shallow central basin. It appears that this basin is functioning at least partially as a salt marsh pond and was evaluated as such by the MEP Technical Team. Therefore, this basin appears to be presently moderately impaired, based upon its moderate-low number of species and moderate-low (100) numbers of individuals. The nutrient enriched conditions within this basin appear to be beyond the accommodation of the infaunal community and nitrogen management will be needed to restore this benthic resource.

The MEP Surveys did detect a small localized area of degraded infaunal habitat within the Centerville River System, a small lagoon altered for boats directly inland of the western stretch of Craigville Beach. This small basin contains a shallow sill across its entrance to the River and a "deep" basin (Figure V-6). The basin configuration and sill create a localized depositional area with localized bottom water hypoxia. The result is a significantly degraded small altered basin, which will require a local solution (restoring proper basin configuration and flow) in addition to the system-wide effort which is the focus of the present analysis.

The overall results indicate a system generally supportive of diverse healthy communities appropriate to each of the 4 component functional basin types as shown in Table VIII-1. The infaunal habitat quality within each of the basins of the Centerville River System is fully consistent with the oxygen and chlorophyll measurements, temporal trend in eelgrass (i.e. loss from lower estuary) and relatively tidally averaged total nitrogen concentration for each basin. The healthy infauna habitats within the Bumps River, Lower Centerville River at TN levels <0.46 are also consistent with other systems. Similarly, the moderately impaired infaunal habitat in East Bay reflects the basins' depositional nature and depth, indicating a sensitivity of this basin to the negative effects of nitrogen enrichment. The healthy habitat within the upper Centerville marshes at 0.63 mg N L-1, also reflects the tolerance of salt marsh communities to nitrogen enrichment (e.g. Cockle Cove Salt Marsh, Chatham).

VIII.2. THRESHOLD NITROGEN CONCENTRATIONS

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

Determination of the critical nitrogen threshold for maintaining high quality habitat within Centerville River Estuarine System is based primarily upon the nutrient and oxygen levels, temporal trends in eelgrass distribution and current benthic community indicators. Given the database available for this threshold analysis, it is possible to develop a site-specific threshold which is a refinement upon general threshold analysis frequently employed.

The Centerville River System is presently supportive of infaunal habitat throughout its 4 component basins. However, there is a moderate level of infaunal habitat impairment within Scudder Bay and the mid region of the Centerville River, requiring nitrogen management for restoration. The primary habitat issue within the Centerville River System relates to the loss of eelgrass from the lower estuary, specifically from the Centerville River west of the entrance to Bumps River and in East Bay. This loss of eelgrass classifies these areas as "significantly impaired", although they presently support healthy to moderately healthy infaunal communities. Further impairment to both the infaunal habitat in Scudder Bay and the eelgrass habitat in the lower estuary are supported by the variety of other indicators which support the conclusion that these impairments are the result of nitrogen enrichment, primarily from watershed nitrogen loading.

Table VIII-1. Summary of Nutrient Related Habitat Health within the Centerville River Estuary on Nantucket Sound within the Town of Barnstable, MA., based upon assessment data presented in Chapter VII. The upper estuarine reach of the Centerville River is presently a tidal salt marsh. The Centerville River mid reach (from the bridge at Craigville Beach to the mouth Bumps River) and lower reach (Bumps River mouth to East Bay) and East Bay are sub-embayments, while the Bumps River and Scudder Bay support significant salt marsh areas.

Health Indicator	Centerville River Embayment System					
	Centerville River Reach			East Bay	Bumps River	Scudder Bay
	Upper	Mid	Lower			
Dissolved Oxygen	H[1]	MI[3]	H/MI[4]	MI[2]	MI[3]	MI[12]
Chlorophyll	MI[6]	--[13]	H[5]	H[5]	--[13]	MI[6]
Macroalgae	--[8]	--[8]	--[7]	--[7]	--[7]	--[8]
Eelgrass	--[10]	--[10]	SI[9]	SI[9]	--[10]	--[10]
Infaunal Animals	H[17]	H/ MI[16]	H[11]	MI[14]	H[11]	MI[15]
Overall:	**H**	**H/MI**	**H[10]-SI[8]**	**MI[14]-SI[8]**	**H**	**MI**

1 – salt marsh tidal creek, periodic oxygen depletions to 3-4 mg/L.
2 – oxygen depletions frequently to 4-5 mg/L., levels generally >5 mg/L.
3 – Monitoring Program grab sample data, periodically 4.5-5 mg DO/L generally >5 mg DO/L.
4 -- Monitoring Program grab sample data, >5 mg DO/L, but may follow East Bay time-series.
5 – modest chlorophyll a levels generally 4-8 ug/L , average 5 ug/L.
6 – elevated chlorophyll levels, mean 18-20 ug/L.
7 – very sparse or absence of drift algae, no surficial microphyte mat
8-- no drift algae, but benthic algae forming a surficial mat, microphytes
9-- MassDEP (C. Costello) indicates that eelgrass lost from this system between 1951-2000.
10 – no evidence this basin is supportive of eelgrass.
11 -- infauna: high numbers of individuals and species, high diversity and Eveness.
12 -- basin supports fringing salt marsh areas.
13 -- insufficient data
14 -- Infauna: moderate numbers of individuals, moderate-low species, dense amphipod mat.
15 -- Infauna: moderate numbers of individuals, moderate-low species, organic enrichment indicator species typical of salt marsh ponds.
16 -- moderate numbers of individuals and species, high diversity and Eveness, with polychaetes, mollusks and crustaceans, estuarine reach is transitional between saltmarsh creek and tidal river..
17 -- high numbers of individuals and moderate numbers of species. High diversity and eveness. Indicators of organic matter enrichment typical of salt marsh sediments.

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

The present lack of eelgrass throughout the Centerville River System is consistent with the observed oxygen depletions in each basin and the chlorophyll levels and functional basin types comprising this estuary. The basins like the Centerville River channel (from the Town Landing to East Bay), and especially sub-embayment of East Bay typically, do support eelgrass habitat in other embayments with low to moderate nitrogen levels. These basins supported eelgrass in the 1951 analysis. The earlier presence of beds from within the lower reaches of the Centerville River System is consistent with the lower nitrogen loading and the resultant higher sustained oxygen levels and lower chlorophyll levels (high light penetration) that should have existed at that time, based upon population data.

The eelgrass and water quality information supports the conclusion that eelgrass beds within East Bay and the lower Centerville River should be the target for restoration of the Centerville River Estuary and that restoration requires appropriate nitrogen management. From the historical analysis, it appears that on the order of 52 acres of eelgrass habitat could be recovered, if nitrogen management alternatives are implemented. Therefore the sentinel station (BC-T) for the Centerville River System was placed just seaward of the mouth of the Bumps River within the Centerville River and upgradient from the Town of Barnstable Water Quality Monitoring Station BC-9. The station was located at within the uppermost extent of the known previous eelgrass coverage in this System.

The target nitrogen concentration (tidally averaged TN) for restoration of eelgrass at the sentinel location within the lower reach of the Centerville River (region seaward of the mouth of the Bumps River) was determined to be 0.37 mg TN L^{-1}. This nitrogen level is based upon the absence of eelgrass in the Lower Centerville River at a tidally averaged TN of 0.395 mg N l^{-1} and comparison to a stable eelgrass system in a similarly configured basin, the lower Oyster River (Chatham) at 0.37 mg N L^{-1}. Note that this level is only slightly lower than that determined by the MEP Technical Team for nearby Popponesset Bay (0.38 mg N L^{-1}). This difference relates to the much shallower water in Popponesset Bay then in the Centerville River. Water depth is important as the same phytoplankton concentration that results in shading of eelgrass in deep water, will allow sufficient light to support eelgrass in shallow water. The need for a lower threshold in deeper versus shallower water was seen in the MEP eelgrass habitat assessment for Bournes Pond, Falmouth.

The threshold nitrogen level at the sentinel station within the Centerville River System is within the range found for other complex systems such as 0.38 mg N L^{-1} for Stage Harbor, 0.38 N/L^{-1} for Bournes Pond and nearby Popponesset Bay and 0.35 mg N L^{-1} for West Falmouth Harbor and Phinneys Harbor. The sentinel station under present loading conditions supports a tidally corrected average concentration of 0.395 mg TN L^{-1}, so watershed nitrogen management will be required for restoration of the estuarine habitats within this system.

Note that achieving the nitrogen threshold at the sentinel station will necessarily result in restoration of other resources throughout the Centerville River/Harbor System distant from the inlet and the influence of strong tidal flushing. Since East Bay is influenced by waters ebbing from both branches of the entire up-gradient basins, its nitrogen management will de facto require a lowering of nitrogen levels throughout the Centerville River System and therefore an improvement of infaunal habitats in Scudder Bay and the middle reach of the Centerville River, which have traditionally only supported infaunal habitat. Based upon the above analysis, eelgrass habitat should be the primary nitrogen management goal for the lower Centerville River System and infaunal habitat quality the management target for the upper reaches. These goals are the focus of the MEP management threshold loading analysis (Section VIII.3) and alternatives analysis presented (Chapter IX).

Although the nitrogen management target is restoration of eelgrass habitat (and associated water clarity, shellfish and fisheries resources), benthic infaunal habitat quality must also be supported as a secondary condition. At present, in the regions with impaired infaunal habitat, the tidally averaged total nitrogen (TN) level under existing conditions is 0.526 mg N L^{-1} in Scudder Bay and between 0.543-0.465 mg N L^{-1} in the middle reach of the Centerville River (bridge to Bumps River). The observed moderate impairment at these sites is consistent with observations by the MEP Technical Team in other enclosed basins along Nantucket Sound (e.g. Perch Pond, Bournes Pond, Popponesset Bay) where levels <0.5 mg N L^{-1} were found to be supportive of healthy infaunal habitat and in deeper enclosed basins in Buzzards Bay (e.g. Eel Pond in Bourne) where healthy infaunal habitat had a slightly lower threshold level, 0.45 mg N L^{-1}, due to those being a "deep" depositional basin. The higher TN levels observed in the upper Centerville River salt marshes are within the nitrogen threshold to support the observed healthy infaunal habitat in this estuarine reach. To ensure that meeting the nitrogen threshold at the sentinel station (BC-T, just seaward of the mouth of the Bumps River within the Centerville River, upgradient from BC-9) results in restoration of the moderately impaired infaunal habitats in Scudder Bay and the middle reach of the Centerville River, nitrogen criteria for secondary infaunal "check" stations were developed by the MEP Technical Team. Based upon the Centerville River system showing moderate impairment at tidally averaged TN levels of 0.526 mg N L^{-1} in Scudder Bay (BC-3) and 0.543 at the inland end of the middle reach of the Centerville River (BC-7) and the results from nearby embayments to Nantucket Sound (noted above), it was concluded that an upper limit of 0.50 mg N L-1 tidally averaged TN would support healthy infaunal habitat in these inner regions.

It must be stressed that the nitrogen threshold for the Centerville River Estuarine System is at the sentinel location. The secondary criteria (infaunal habitat) should be met when the threshold is met at the sentinel station used for setting the nitrogen threshold and serve as a “check”. The nitrogen loads associated with the threshold concentration at the sentinel location and secondary infaunal check stations are discussed in Section VIII.3, below.

VIII.3. DEVELOPMENT OF TARGET NITROGEN LOADS

The nitrogen thresholds developed in the previous section were used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Centerville River estuary 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 Centerville River and at the secondary stations in Scudder Bay and Centerville River East. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required 80% removal of septic load (associated with direct groundwater discharge to the embayment) for the Centerville River East watershed. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1.

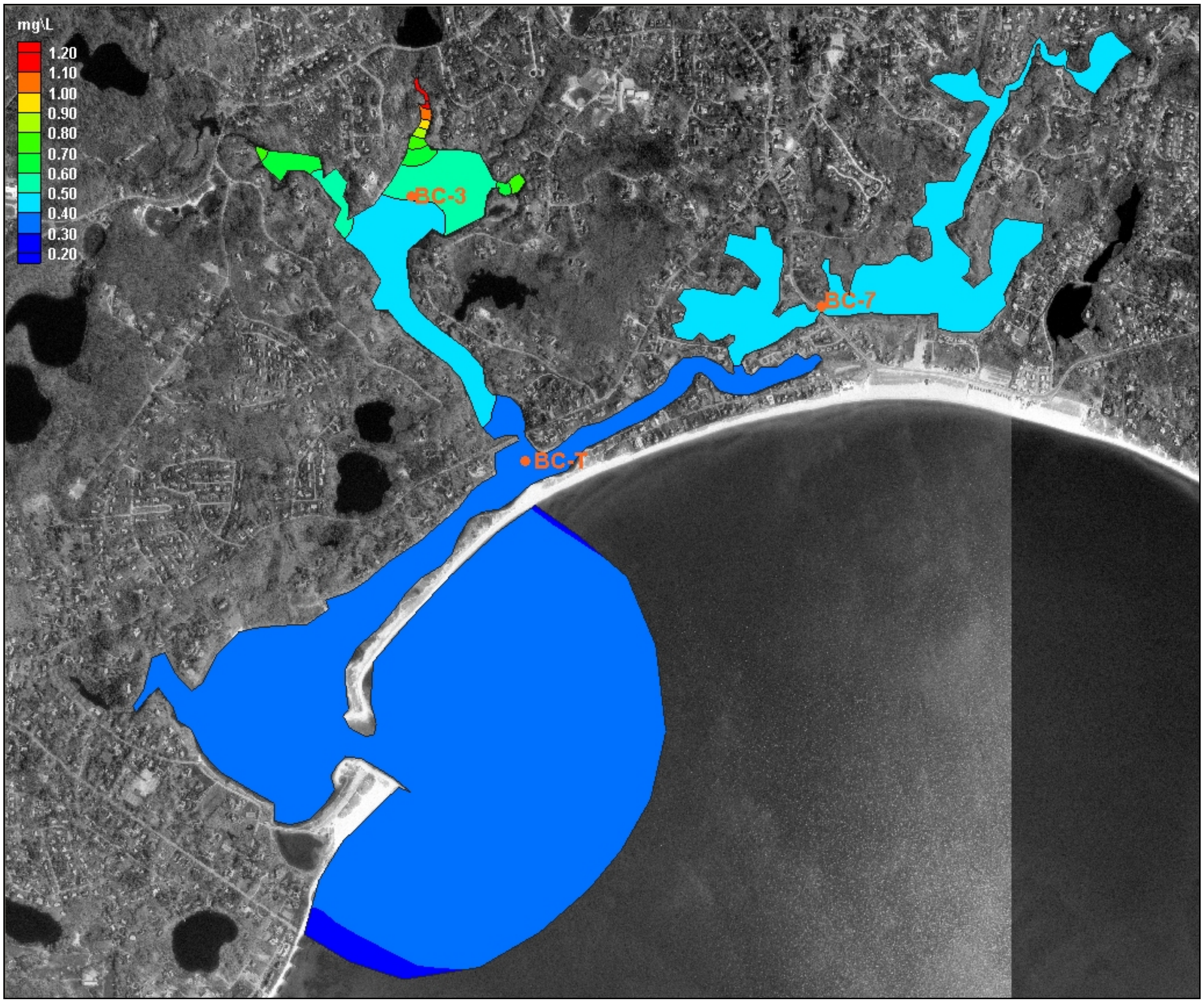


Figure VIII-1. Contour plot of modeled average total nitrogen concentrations (mg/L) in the Centerville River estuary system, for threshold conditions (0.37 mg/L at water quality monitoring station BC-T, and 0.4-0.5 at water quality monitoring stations BC-3 and BC-7). The approximate location of the sentinel threshold station for Centerville River (BC-T) is shown.

Table VIII-2. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading scenarios of the Centerville River estuary 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
Centerville River East	45.929	9.184	-80.0%
Scudder Bay	11.619	11.619	+0.0%
Centerville River West	7.704	7.704	+0.0%
East Bay	6.301	6.301	+0.0%
Surface Water Sources			
Pine Street Stream	2.512	2.512	+0.0%
Lake Elizabeth Stream	1.836	1.836	+0.0%
Bumps River	14.321	14.321	+0.0%
Skunknett River	17.337	17.337	+0.0%

Tables VIII-3 and VIII-4 provide additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. Removal of 80% of the septic load from the Centerville River East watershed results in a 66% reduction in total nitrogen load. 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.

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 Centerville River estuary system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.			
sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Centerville River East	55.737	18.992	-65.9%
Scudder Bay	14.452	14.452	+0.0%
Centerville River West	9.463	9.463	+0.0%
East Bay	8.627	8.627	+0.0%
Surface Water Sources			
Pine Street Stream	3.452	3.452	+0.0%
Lake Elizabeth Stream	2.274	2.274	+0.0%
Bumps River	16.912	16.912	+0.0%
Skunknett River	21.260	21.260	+0.0%

Table VIII-4. Threshold sub-embayment loads and attenuated surface water loads used for total nitrogen modeling of the Centerville River estuary 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)
Centerville River East	18.992	0.449	4.284
Scudder Bay	14.452	0.685	-2.125
Centerville River West	9.463	0.718	3.497
East Bay	8.627	1.126	12.694
Surface Water Sources			
Pine Street Stream	3.452	-	-
Lake Elizabeth Stream	2.274	-	-
Bumps River	16.912	-	-
Skunknett River	21.260	-	-

Comparison of model results between existing loading conditions and the selected loading scenario to achieve the target TN concentrations at the sentinel station is shown in Table VIII-5. To achieve the threshold nitrogen concentrations at the sentinel station, a reduction in TN concentration of approximately 10% is required at station BC-T. The meet secondary threshold requirement for stations BC-3 and BC-7, a reduction in TN concentration of approximately 8% and 21% were required, respectively.

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

Table VIII-5. Comparison of model average total N concentrations from present loading and the modeled threshold scenario, with percent change, for the Centerville River estuary system. Sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Scudder Bay	**BC-3**	**0.524**	**0.480**	**-8.4%**
Bumps River	BC-4	0.451	0.415	-7.9%
Centerville River	BC-5	0.609	0.449	-26.3%
Centerville River	**BC-7**	**0.526**	**0.417**	**-20.8%**
Centerville River	BC-8	0.454	0.386	-15.0%
Centerville River	BC-9	0.389	0.358	-8.0%
East Bay	BC-10	0.349	0.333	-4.6%
Confluence of Bumps River-Centerville River	**BC-T**	**0.412**	**0.372**	**-9.8%**

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.

IX. ALTERNATIVES TO IMPROVE WATER QUALITY

IX.1 PRESENT LOADING WITH SEWERING PORTIONS OF CENTERVILLE RIVER EAST WATERSHED AND INCREASING THE NATURAL NITROGEN ATTENUATION ALONG SKUNKNET RIVER

The Centerville River East watershed contributes the largest amount of Nitrogen to the Centerville River System relative to the other watersheds. Therefore sewering within watershed will have the greatest impact upon the Nitrogen levels entering the system. However the possibility exists to combine sewering with measures to naturally increase Nitrogen attenuation along surface water tributaries within the system. For this alternative the natural Nitrogen attenuation along Skunknet River will be increased by 20-percent to evaluate the magnitude of impact it will have upon reducing the amount of sewering required. To demonstrate this, an alternative was developed to assess impact of removing 75-percent of the septic load from the Centerville River East watershed, which is a 5-percent reduction compared to the threshold analysis in Chapter VIII, along with increasing the natural Nitrogen attenuation along Skunknet River by 20-percent. The present loading conditions will be used for the rest of the system. Table IX-1 and Table IX-2 illustrate the overall change to septic and watershed loads resulting from this alternative. Septic removal from Centerville River East watershed results in significant reductions in the watershed loads in the sub-embayment. Based on the assumptions developed for this alternative, Table IX-3 presents the various components of nitrogen loading for the Centerville River system.

Table IX-1. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling present loading conditions with a 75-percent of the septic load removed from the Centerville River East watersheds and 20% increase to natural Nitrogen attenuation along Skunknet River. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.

sub-embayment	present septic load (kg/day)	scenario septic load (kg/day)	threshold septic load % change
Centerville River East	45.929	11.479	-75.0%
Scudder Bay	11.619	11.619	+0.0%
Centerville River West	7.704	7.704	+0.0%
East Bay	6.301	6.301	+0.0%
Surface Water Sources			
Pine Street Stream	2.512	2.512	+0.0%
Lake Elizabeth Stream	1.836	1.836	+0.0%
Bumps River	14.321	14.321	+0.0%
Skunknet River	17.337	11.762	-32.2%

Table IX-2. Comparison of sub-embayment ***total attenuated watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present conditions in Centerville River with present loading conditions with a 75-percent of the septic load removed from the Centerville River East watersheds and 20% increase to natural Nitrogen attenuation along Skunknet River. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	scenario load (kg/day)	threshold % change
Centerville River East	55.737	21.288	-61.8%
Scudder Bay	14.452	14.452	+0.0%
Centerville River West	9.463	9.463	+0.0%
East Bay	8.627	8.627	+0.0%
Surface Water Sources			
Pine Street Stream	3.452	3.452	+0.0%
Lake Elizabeth Stream	2.274	2.274	+0.0%
Bumps River	16.912	16.912	+0.0%
Skunknet River	21.260	15.685	-26.2%

Table IX -3. Sub-embayment loads used for total nitrogen modeling of the Centerville River system for present loading scenario with present loading conditions with a 75-percent of the septic load removed from the Centerville River East watersheds and 20% increase to natural Nitrogen attenuation along Skunknet River, 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)
Centerville River East	21.288	0.449	4.353
Scudder Bay	14.452	0.685	-2.125
Centerville River West	9.463	0.718	3.497
East Bay	8.627	1.126	12.694
Surface Water Sources			
Pine Street Stream	3.452	-	-
Lake Elizabeth Stream	2.274	-	-
Bumps River	16.912	-	-
Skunknet River	15.685	-	-

Total nitrogen modeling results for existing conditions with the reduced septic loads for Centerville River East watershed and increase in natural attenuation to Skunknet River indicate that the Centerville River would meet the nitrogen threshold target at Station BC-T (Table IX-4 and Figure IX-1), along with meeting the secondary goals at Stations BC-3 and BC-7. It results in reductions in nitrogen concentration in all of the sub-embayments. Nitrogen concentration reductions range from approximately 4% in East Bay to 25% in upper reaches of Centerville River. Overall, this scenario indicates that to reduce the overall nitrogen load effectively, removing septic loads from the Centerville River East watershed is necessary, but that measures to increase natural attenuation are an important component to controlling the percentage of septic removal that will be necessary in the upper watershed.

Table IX-4. Comparison of model average total N concentrations from present loading scenarios (with and without the reduction of septic loads Centerville River East watershed and 20% increase to natural Nitrogen attenuation along Skunknet River), with percent change, for the Centerville River system. The threshold station is shown in bold print.

Sub-Embayment	monitoring station	present (mg/L)	scenario (mg/L)	% change
Scudder Bay	**BC-3**	**0.524**	**0.478**	**-8.8%**
Bumps River	BC-4	0.451	0.412	-8.5%
Centerville River	BC-5	0.609	0.457	-25.0%
Centerville River	**BC-7**	**0.526**	**0.422**	**-19.9%**
Centerville River	BC-8	0.454	0.389	-14.4%
Centerville River	BC-9	0.389	0.359	-7.9%
East Bay	BC-10	0.349	0.333	-4.4%
Confluence of Bumps River and Centerville River	**BC-T**	**0.412**	**0.373**	**-9.6%**

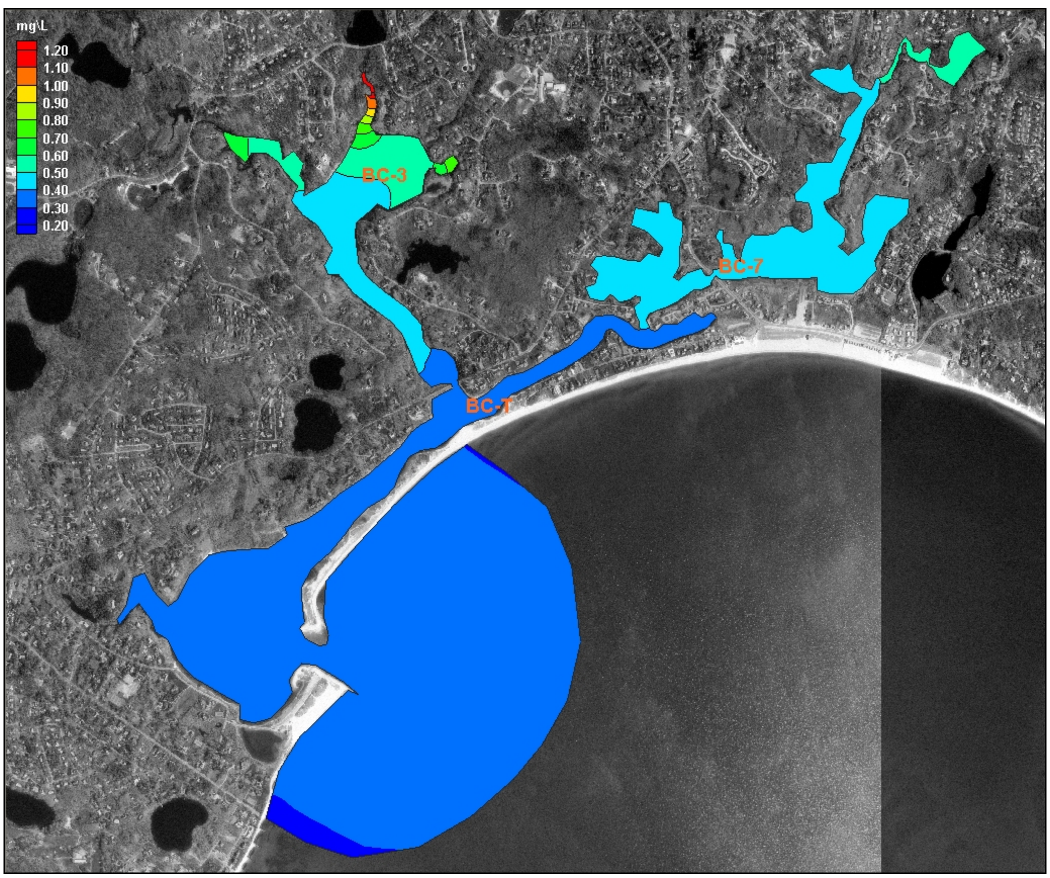


Figure IX-1. Contour plot of modeled total nitrogen concentrations (mg/L) in the Phinney's Harbor system, for present loading conditions with a 75-percent of the septic load removed from the Centerville River East watersheds and 20% increase to natural Nitrogen attenuation along Skunknet River.

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