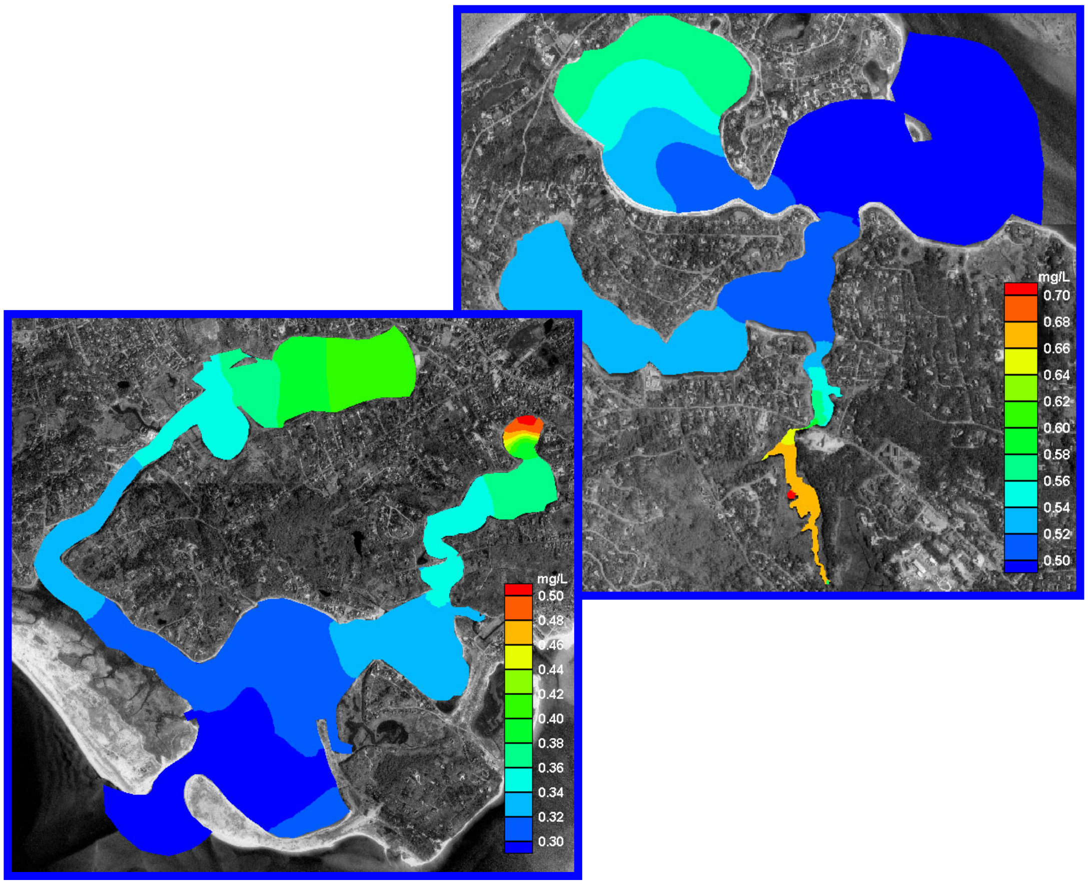
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Stage Harbor, Sulphur Springs, Taylors Pond, Bassing Harbor, and Muddy Creek, Chatham, Massachusetts



FINAL REPORT – December 2003

Massachusetts Department of Environmental Protection

University of Massachusetts Dartmouth School of Marine Science and Technology

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I. INTRODUCTION

Chatham Massachusetts, at the eastern end of Cape Cod, is surrounded by water on three sides, with Nantucket Sound to the south, the Atlantic Ocean and Chatham Harbor to the east, and Pleasant Bay to the north (Figure I-1). Much of the shoreline, especially to the north and south, consists of a number of small embayments of varying size and complexity. These embayments constitute important components of the Town's natural and cultural resources. 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 embayments along Chatham's shore are at risk of eutrophication from high nitrogen loads in the groundwater and runoff from their watersheds.

As existing and potentially increasing levels of nutrients impact Chatham's coastal embayments, water quality degradation will continue to harm invaluable environmental resources. As described in the Town's Wastewater Management Planning Study (CWMP), the primary nitrogen source to Chatham's coastal embayments is on-site septic systems via groundwater flow. Although the CWMP provided a cursory analysis of acceptable nitrogen loading to the local estuarine systems based on the methodology developed by the Buzzards Bay Project (USEPA and Massachusetts EOEA, 1991), ecological indictors contradicted many of the results of this analysis.

Since site-specific data have been lacking on existing water quality in the embayments and its relationship to calculated nitrogen loads from their watersheds, the Town implemented a multi-disciplinary approach to resolving estuarine water quality issues. First, the Town's water Quality Laboratory in conjunction with the Chatham Water Watchers (a citizen volunteer organization) implemented a monitoring program of water column nitrogen in 1999. This four-year evaluation has provided the baseline information required for determining the link between upland loading, tidal flushing, and estuarine water quality. Subsequent to the development of a multi-year data set establishing background water quality monitoring for each of the Chatham Embayment systems, and building on previous hydrodynamic and water quality analyses, additional high order biogeochemical analyses and water quality modeling was necessary to develop critical nitrogen targets for each embayment system. These critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater master planning and nitrogen management alternatives development in the Town of Chatham. The completion of this complex multi-step process of rigorous scientific investigation supporting watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project. The modeling tools developed as part of this program provide the quantitative information necessary for the CWMP Team to predict the impacts on water quality from a variety of proposed management scenarios.

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

Figure I-1. Study region for the tidal flushing study including the estuarine systems in the Stage Harbor System (outlined in red), the South Coast Embayments (outlined in yellow), and the Pleasant Bay Region (outlined in blue).

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

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

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

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

The Massachusetts Estuary Project is founded upon science-based management. The Project will use a consistent, state-of-the-art approach throughout the region's coastal waters and provide technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the Massachusetts Estuaries Project is to provide the DEP with technical guidance to support policies on nitrogen loading to embayments. In addition, the technical reports prepared for each embayment system will serve as the 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 (the "303d list"). TMDLs must identify sources of the pollutant of concern (in this case nitrogen) from both point and non-point sources, the allowable load to meet the state water quality standards and then allocate that load to all sources taking into consideration a margin of safety, seasonal variations, and several other factors. In addition, each TMDL must contain an implementation plan. That plan must identify, among other things, the required activities to achieve the allowable load to meet the allowable loading target, the time line for those activities to take place, and reasonable assurances that the actions will be taken.
In appropriate estuaries, TMDL's for bacterial contamination will also be conducted in concert with the nutrient effort (particularly if there is a 303d listing). However, the goal of the bacterial program is to provide information to guide targeted sampling for specific source identification and remediation. As part of the overall effort, the evaluation and modeling approach will be

used to assess available options for meeting selected nitrogen goals, protective of embayment health.

The major Project goals are to:

- 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 regulatory needs.

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

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

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

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

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

- Monitoring - multi-year embayment nutrient sampling
- Hydrodynamics -
  - embayment bathymetry
  - site specific tidal record
  - current records (in complex systems only)
  - hydrodynamic model
- Watershed Nitrogen Loading
  - watershed delineation
  - stream flow (Q) and nitrogen load
  - land-use analysis (GIS)
  - watershed N model
- Embayment TMDL - Synthesis
  - linked Watershed-Embayment N Model
  - salinity surveys (for linked model validation)
  - rate of N recycling within embayment
  - D.O record
  - Macrophyte survey
  - Infaunal survey (in complex systems)

I.2 SITE DESCRIPTION

The coast of Chatham is bordered to the south by Nantucket Sound, the east by the Atlantic Ocean, and the north by Pleasant Bay. For this study, Chatham's estuarine systems have been separated into three general groups: the 1) Stage Harbor System, 2) the South Coast Embayments and the 3) Pleasant Bay Region Embayments (see Figure I-1).

Although the three estuarine systems along the south shore (Stage Harbor, Sulphur Springs, and Taylors Pond) exhibit different hydrologic characteristics, ranging from expansive salt marshes to flooded kettle ponds, the tidal forcing for these systems is generated from Nantucket Sound. In contrast, water propagating through the Chatham Harbor/Pleasant Bay system is derived from the Atlantic Ocean.

The south shore of Chatham exhibits a moderate tide range, with a mean range of about 4.5 ft. Since the water elevation difference between Nantucket Sound and each of the estuarine systems is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle. Tidal damping (reduction in tidal amplitude) through the Stage Harbor system is negligible indicating "well-flushed" systems. In contrast, the tidal attenuation caused by the restrictive channels and marsh plains within the South Coast Embayments of Mill Creek/Taylors Pond is indicative of a "restrictive" system, where tidal flow and the associated flushing are inhibited. Based on the tidal characteristics alone, this might indicate that the Stage Harbor embayments (e.g. Little Mill Pond) are "healthy" relative to the embayments further the west; however, land development in the southeastern portion of Chatham likely provides a substantially higher nutrient load to the Stage Harbor embayments. Consequently, estuarine water quality may be more dependent on nutrient loading than tidal characteristics for these systems.

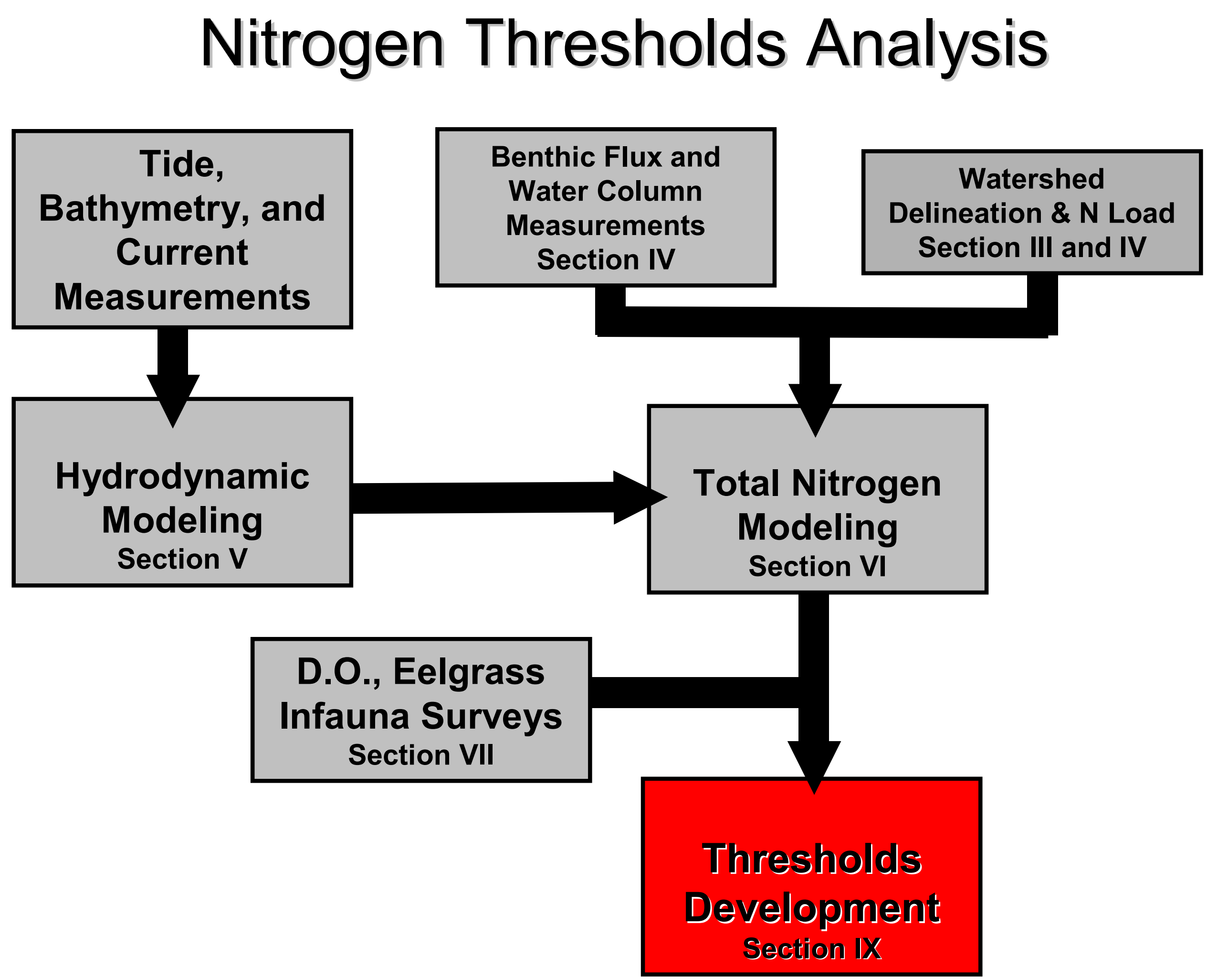


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach

The Stage Harbor System consists of six (6) embayments: Stage Harbor, Oyster Pond River, Oyster Pond, Mitchell River, Mill Pond, and Little Mill Pond. The watershed for this estuarine system contains approximately 1,700 acres dominated by single family residences. As stated above, land development in the southeastern portion of Chatham creates a large nutrient load to the Stage Harbor System. Based on watersheds developed by the Cape Cod Commission (Stearns & Wheler, 1999), the nitrogen loading from the more heavily populated areas of the village and the area to the west is focused on the northern reaches of the estuarine system. For example, approximately 80% of the nitrogen load from single-family dwellings enter the Stage Harbor System along the shorelines of Oyster Pond, the northern portion of Oyster Pond River, Little Mill Pond, and Mill Pond.

The South Coast Embayments exhibit similar nitrogen loading characteristics, where much of the nutrient loading enters the system along the northern limits. Due to the relatively narrow channels that connect the upper portions of these embayments to Nantucket Sound, flushing characteristics are relatively poor. However, the large expanses of salt marsh in Mill Creek and the Cockle Cove Creek/Sulphur Springs systems allow these water bodies to be more tolerant of high nitrogen loads.

Within Pleasant Bay, the tide propagating through New Inlet and Chatham Harbor is significantly attenuated by the series of flood tidal shoals within the inlet throat. The mean tide range drops from just under 8 feet in the Atlantic Ocean to around 5 feet at the Chatham Fish Pier. Only minor attenuation occurs between the Fish Pier and Pleasant Bay; however, smaller sub-embayments separated from the main system by culverts exhibit significant additional tidal attenuation. Both Muddy Creek and Frost Fish Creek have mean tide ranges of less than 1 ft. For the Bassing Harbor system, nitrogen loading is primarily focused in the Frost Fish Creek and Ryder Cove watersheds.

In addition to tidal forcing characteristics, the regional geomorphology influences flushing characteristics and the associated water quality within embayments along the south shore, as well as for the Pleasant Bay system. Shoaling along the south shore of Chatham has caused the opening and closing of several inlets to the Sulphur Springs/Bucks Creek/Cockle Cove Creek system during the past 50 years. In addition, stability issues concerning the Stage Harbor navigation channel required repositioning of the inlet in 1965 as a result of regional shoaling. The most dramatic recent change in local geomorphology occurred in early 1987, when New Inlet formed east of the Chatham Lighthouse. From a tidal flushing and water quality perspective, the resulting increase in tide range within Pleasant Bay of approximately 1 ft caused a substantial improvement of regional tidal exchange.

I.3 NITROGEN LOADING

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

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

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

Unfortunately, almost all of Chatham’s estuarine systems are near or beyond their ability to assimilate additional nutrients without impacting their ecological health. The effect is that nitrogen management of these systems is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed “eutrophication” and when the nutrient loading is primarily from human activities, “cultural eutrophication”. Although the influence of man-induced changes has increased nitrogen loading to the systems and contributed to the degradation in ecological health, eutrophication of several Chatham embayments would occur without man’s influence. 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 (Stearns & Wheler, 1999 and more recent updates to watershed boundaries by USGS and nitrogen loading by the CCC) provides important "boundary conditions" for water quality analyses of Chatham's coastal embayments; however, a thorough understanding of estuarine circulation is required to accurately determine nitrogen concentrations within each system. Therefore, water quality modeling of tidally influenced estuaries must include a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Numerical models provide a cost-effective method for evaluating tidal hydrodynamics since they require limited data collection and may be utilized to numerically assess a range of management alternatives. Once the hydrodynamics of an estuary system are understood, computations regarding the related coastal processes become relatively straightforward extensions to the hydrodynamic modeling. The spread of pollutants may be analyzed from tidal current information developed by the numerical models.

The water quality evaluation examined the potential impacts of nitrogen loading into the Stage Harbor System, the South Coast Embayments, and the Pleasant Bay Region. 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. Almost all nitrogen entering Chatham's coastal embayments is transported by freshwater, predominantly groundwater. Concentrations in Nantucket Sound and Pleasant Bay source waters were taken from Chatham Water Watchers and Pleasant Bay Alliance data. Measurements of current nitrogen distributions throughout estuarine waters were used to calibrate 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 Town of Chatham coastal embayment systems. 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 watersheds surrounding each estuary were derived from Cape Cod Commission data and offshore water column nitrogen values were derived from an analysis of monitoring stations in Chatham Harbor and 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. In addition, an ecological assessment of all coastal embayments was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). This assessment can be used by the Town to develop a baseline for future management and estuary restoration efforts.

Analyses of Chatham’s coastal embayments were performed to assist the Town with future management decisions, beginning with those embayments where flushing improvements were considered (e.g. Muddy Creek). The results of the nitrogen modeling for each scenario have been presented (Section IX).

II. PREVIOUS NITROGEN MANAGEMENT STUDIES

Nutrient additions to aquatic systems can lead to a series of processes in a water body that result in impaired water quality. Effects include excess plankton and macrophyte growth, which in turn lead to reduced water clarity, excess organic matter, the development of lowered dissolved oxygen, especially in bottom waters, and the limitation of the growth of desirable species such as eelgrass. In most marine and estuarine systems, such as those that make up the coastal embayments of Chatham, 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).

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 is an attempt to link water quality model predictions to actual measured values for specific nutrient species thereby enabling calibration of the prediction process for specific conditions in each of the coastal embayments of southeastern Massachusetts, beginning with the embayment systems located in the Town of Chatham.

The first steps of the MEP nutrient analysis process implemented in the Town of Chatham were to measure physical conditions in the various water bodies and to develop hydrodynamic models to simulate and quantify the transport of water in and out of the embayments. This allowed tidal flushing to be evaluated. The results of this work are reported in Kelley, *et al.* (2001). Based on those findings, and on additional biological and chemical measurements made within the embayments, a water quality model was developed that used the tidal flushing inputs and simulated the calculated and measured nitrogen loads to the embayments. This model was then calibrated in a process that rationalizes the resulting calculated water column concentrations with measured values from monitoring programs over the past four years. The water quality model then becomes a predictive tool for evaluating the effects of various nitrogen loading scenarios on nitrogen concentrations in the embayments.

The concern about excessive nitrogen loading to the water bodies in the Chatham study area is evidenced by the number of studies and analyses conducted over the past 10 years. This section summarizes these studies in chronological order to help put the present study in historical perspective.

The first identified study that addresses nitrogen problems in Chatham is the Comprehensive Harbor Management Plan (HWH, 1992). The harbor plan focuses on the Stage Harbor system consisting of Stage Harbor, Mitchell River, Mill Pond (and Little Mill Pond), Oyster Pond and Oyster River. The water quality section inventories the existing water quality and presents an analysis of threats to water quality. The only existing water quality data presented in Comprehensive Harbor Management Plan were fecal coliform measurements at a series of stations in the Stage Harbor system. The analyses of threats to water quality in the system were broken down into six general source types: stormwater runoff, sewage, fertilizers and pesticides, animal waste, household hazardous waste and marinas. The first four are capable of increasing nitrogen levels in the ponds, contributing to eutrophication in this system.

The analysis presented in HWH (1992) predicts levels of nitrogen entering the Stage Harbor system in historical, present and future land use development. This approach requires that a "buildout" analysis be performed to estimate the potential number of additional residences that could be constructed under present zoning regulations. These residences generate additional nitrogen loadings that reach the Stage Harbor system through both surface runoff and groundwater. The study then examined the quantity of nitrogen, expressed as total nitrogen (TN), coming from each source. For the effluent emanating from individual septic systems the analysis used a TN concentration of 40 mg/L and a flow rate of 55 gallons per day for two occupancy rates, depending on the time of year: 1.86 and 3 people per unit. A review was also conducted on leaching rates for fertilizer resulting in an average estimate of 3 pounds per 1000 ft^2 applied to an average lawn size of 6000 ft^2 with approximately 18 inches/year of precipitation entering the groundwater. Pavement and roof runoff TN concentrations were estimated as 2 mg/L and 0.75 mg/L, respectively, with direct runoff flow of 40 inches/year. Precipitation causes direct deposition of nitrogen to the system watershed. Since vegetation removes most of the dissolved nitrogen, a background source concentration of 0.05 mg/L was used for groundwater while direct precipitation to the water bodies was estimated as 0.3 mg/L of dissolved inorganic nitrogen.

Using estimates of flushing from each of the water bodies in the Stage Harbor system an evaluation was performed on resulting nitrogen loadings to be expected in the estuaries. Oyster Pond, Oyster River and Mitchell River were found to approach or exceed the Buzzards Bay recommended limits for shallow, rapidly flushed water bodies for present and future buildout conditions. Stage Harbor, being directly connected to Nantucket Sound with a higher flushing rate, and Mill Pond, being deeper and supposedly able to assimilate more nitrogen, were found to be within the limits. Calculations were also performed for dredging and shoaling alternatives.

The Cape Cod Commission (CCC) conducted a nitrogen loading study for the Pleasant Bay system to determine the maximum allowable loads that 16 subembayments could tolerate based on a series of regulatory limits (CCC, 1998). The CCC began the study by delineating the watersheds that drain into the various subembayments and that provide the nitrogen loads. Land use was determined using data within the CCC's GIS system and then modified as needed in consultation with the local communities. The CCC staff then used their loading protocol as defined in Technical Bulletin 91-001 (CCC, 1991). This protocol assigns loading from a variety of land use types in a generally similar manner as was done by HWH (1992) for the Stage Harbor system. Total nitrogen concentrations from wastewater were assumed to be 35 mg/L; 1.5 mg/L for road runoff; 0.75 mg/L for roof runoff and direct precipitation; and 0.05 mg/L for natural area runoff. Average residential lawn size was assumed to be 5000 ft^2 with a fertilizer application rate of 3 lb/1000 ft^2. Recharge rates used were 40 in/yr for impervious surfaces and 16 in/yr (Brewster, Harwich) or 17 in/yr (Chatham, Orleans, Eastham, Wellfleet, Truro, Provincetown) for natural areas. Both existing and buildout conditions were analyzed. Flushing times were determined for each embayment for both existing and pre-break inlet configurations.

The resulting nitrogen loads were compared to critical levels, here defined as the Buzzards Bay Project Outstanding Resource Waters (BBP ORW) and Outstanding Resource Waters – Nitrogen (ORW-N) limits. Within the Chatham part of the study area, it was found that Muddy Creek exceeded both the nitrogen limits for both configurations while Ryder Cove exceeded the ORW-N limit with the pre-break configuration. This pattern was repeated for the same water bodies under the buildout scenario but with greater exceedences. In addition, difficulties in predicting the change in offshore nitrogen concentrations as New Inlet migrated south to its pre-breach condition (directed toward Nantucket Sound rather than the Atlantic Ocean) made future evaluation of critical nitrogen loads questionable.

The Pleasant Bay Resource Management Plan was prepared by the Pleasant Bay Technical Advisory Committee and Ridley & Associates, Inc. (PBTSC and Ridley & Associates, 1998). The purpose of the plan was not only to reconcile both sustainability and restoration of the Pleasant Bay ecosystem but also to enhance public access and enjoyment of the bay, encouraging recreational, residential and commercial use consistent with resource sustainability. The management plan referred to the CCC study for analyses of nutrient loading and water quality and advocated continued monitoring of the water body.

The most recent study of nitrogen loading to the Chatham study area was performed by Stearns & Wheler as part of its needs assessment for the Chatham wastewater management planning study (Stearns & Wheler, 1999). The study area was divided into three groups that were analyzed separately: Pleasant Bay Region, Stage Harbor System, and the South Coast Embayments (see Figure I-1). The study followed a similar protocol as the earlier studies: use of existing subwatersheds information, calculation of existing and future nitrogen loading to each water body based on land use in its subwatershed, calculation of steady-state nitrogen concentration to be expected based on flushing rate estimates, and finally, comparison of calculated loading to critical nitrogen loading limits to determine if exceedences should be expected, or at what point exceedences may occur as a result of buildout.

An analysis of existing loading to the Pleasant Bay systems embayments was based on the previous Pleasant Bay study by the CCC (1998). An analysis of the existing loading to the Stage Harbor system embayments was based on the previous Stage Harbor study by HWH (1992), and included additional estimates modified to incorporate actual 1997 water use in the watersheds. The south coast embayments had not been previously studied. Therefore these embayment loadings were determined from the CCC protocol using three approaches: Technical Bulletin 91-001 (CCC, 1991), actual 1997 water consumption, and estimates from Title 5 design flows. The loadings based on actual water consumption were lowest of the three and thought to be the most accurate. It was found that the existing nitrogen loadings for all embayments are lower than the critical nitrogen loading for the BBP-SA standard. Taylor Pond and Sulphur Springs exceeded the more stringent ORW-N standard. The analysis was repeated for future seasonal and year round buildout conditions.

Similar to previous studies, the 1999 Stearns & Wheler analysis utilized the Buzzards Bay Project methodology (EPA, 1991) that incorporated a simplistic approach aimed at general planning analyses that was based on “local” residence times. First, this method assumes that tidal waters exiting from a sub-embayment during the ebb cycle are totally replaced with “pristine” water from the downstream sub-embayment. While this assumption may be valid for the main portion of Stage Harbor, where tidal waters are exchanged directly with Nantucket Sound, it is not valid for sub-embayments such as Little Mill Pond, where tidal waters are exchanged with nutrient over-loaded Mill Pond. Secondly, the absence of eelgrass in much of Oyster Pond, Little Mill Pond, and Mill Pond (MassGIS, 1994) indicate embayments exhibiting ecological stress. The existence of sparse/patchy eelgrass beds in portions of these embayments indicates a long-term decline in water quality.

Signs of ecological deterioration and overall habitat stress within all of the Chatham embayment systems prompted the actual measurement of nitrogen concentrations in these embayment systems as initiated in 1998 (Duncanson, 2000; Howes and Schlezinger, 2000). The results of the multi-year water quality monitoring effort begun in 1998 were combined with additional levels of analysis including embayment specific hydrodynamic modeling, water quality modeling, and habitat assessment (Kelley *et al*., 2001 and Applied Coastal *et al*., 2001). Based on the site-specific nutrient analysis for the coastal embayment nutrient threshold development, it appeared that most of the sub-embayments in Chatham already exceeded some or all of the total nitrogen-based water quality criteria used to evaluate critical nitrogen loads.

The water quality analysis and modeling effort in 2001 (Kelley *et al*., 2001) represented an initial effort at the linked water quality modeling approach; however, limitations in the embayment water quality data set and data gaps precluded accurate calibration of the water quality model. Specifically, major shortcomings that limited the utility of the analysis included inconsistent water column nitrogen concentrations in the Bassing Harbor system with regards to the ecological health of the system and incorrect watershed loading to the Mill Creek/Taylors Pond computed by the Town's wastewater engineering consultant (Stearns and Wheler).

To address some of the shortcomings inherent in the 2001 study, the Town funded EarthTech to model the impact of drinking water wells and the existing wastewater treatment facility on the Mill Creek/Taylors Pond watershed. In addition, water column nitrogen measurements have continued since 2001 and updated benthic flux measurements were obtained within the Bassing Harbor system. This additional information was incorporated into the MEP study to improve the water quality analysis of Chatham's coastal embayments.

The indication of a long term decline in water quality and habitat health throughout the Chatham embayment systems is fully explored in this MEP critical nutrient threshold report and incorporates a detailed discussion of historic changes in benthic communities (Section VII). The 4-year water quality monitoring effort combined with the historic information on the benthic community forms the basis for determining appropriate site-specific nutrient thresholds.

Although some researchers, including the CCC, have utilized the Buzzards Bay Project methodology as a general planning tool for determining critical nitrogen loads, it is inappropriate for developing site-specific guidelines regarding nitrogen loading limits. For the Pleasant Bay Region, the Stage Harbor System, and the South Coast Embayments (Figure I-1), water column nitrogen data indicate that all of Chatham's systems are over the State's limits for Outstanding Resource Waters. In addition, limits indicative of maintaining healthy shellfish resources also are exceeded in most systems, where the nitrogen level is higher than 0.15 mg/L over background concentrations in Nantucket Sound or the Atlantic Ocean (Cape Cod Commission, 1998). Since the site-specific data supercedes information obtained from the more generic calculations utilizing the Buzzards Bay Project methodology, future nitrogen management decisions should incorporate information obtained directly from Chatham's coastal embayments. The MEP approach presented in this report develops site-specific critical nutrient thresholds for the five Town of Chatham embayment systems addressed in this report.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

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

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watersheds or contributing areas to the five Chatham estuaries under evaluation by the Project Team. The five estuarine systems are: Muddy Creek, Bassing Harbor/Ryder Cove/Frost Fish Creek/Crows Pond, Stage Harbor, Sulphur Springs/Bucks Creek, and Taylor's Pond. The watersheds to each embayment were divided into functional sub-units based upon: (a) defining inputs from contributing areas to each major sub-embayment within each embayment system (for example Oyster Pond in the Stage Harbor System or Ryder Cove in the Bassing Harbor System), (b) defining contributing areas to major aquatic systems which might 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 in order to gauge the potential mass of nitrogen from "new" development, which has not yet reached the receiving estuarine waters. The three-dimensional numerical model employed is also being used to define the contributing areas to public water supply wells on the Monomoy flow cell on Cape Cod. Model assumptions for calibration were matched to surface water inputs and flows from current (2002) and historical stream gage information.

The relatively transmissive sand and gravel deposits that comprise most of Cape Cod create a hydrologic environment where watershed boundaries are usually better defined by elevation of the groundwater and its direction of flow, rather than by the land surface topography (Cambareri and Eichner 1998, Millham and Howes 1994a,b). 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 source of nitrogen that they carry requires determination of the portion of the watershed that contributes directly to the stream and the portion of the groundwater system that discharges directly into the estuary as groundwater.

Biological attenuation of nitrogen (natural attenuation) occurs primarily within surface aquatic ecosystems (streams, wetlands, ponds) with little occurring within the main aquifer. The freshwater ponds on Cape Cod also provide important environments for the biological attenuation of nitrogen entering them and therefore also require that their contributing areas be delineated. Fresh ponds are hydrologic features directly connected to the groundwater system, which receive groundwater inflow in upgradient areas and discharge water into the aquifer in downgradient areas. The residence time of water within the ponds is a function of pond volume and inflow/outflow rates.

III.2 MODEL DESCRIPTION

Contributing areas to the Chatham estuaries and local freshwater bodies were delineated using a regional model of the Monomoy flow cell. The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al*., 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to Chatham's estuaries and also to determine portions of recharged water that may flow through ponds and streams prior to discharging into coastal water bodies.

The Monomoy Flow Model grid consists of 164 rows, 220 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below sea level and have a uniform thickness of 10 ft. The top of layer 8 resides at sea level with layers 1-7 stacked above sea level to a maximum elevation of +70 feet. In regions like the Monomoy Lens in which Chatham resides, water elevations are less than +40 ft and therefore the uppermost layers are inactive. Layer 18 has a thickness of 40 feet and layer 19 extends to 240 feet below sea level. The bottom layer, 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics.

The glacial sediments that comprise the aquifer of the Monomoy flow cell consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward sequence with sand and gravel deposits deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. Lithologic data used to determine hydraulic conductivities used in the model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and water-level and streamflow data collected in May 2002.

The model simulates steady state, or long-term average, hydrologic conditions including a long-term average recharge rate of about 26 inches/year and the pumping of public-supply wells at average annual withdrawal rates for the period 1995-2000 with a 15% consumptive loss. Large withdrawals of groundwater from pumping wells may have a significant influence on water tables and watershed boundaries and therefore the flow and distribution of nitrogen within the aquifer. Since most of Chatham is unsewered, 85% of the water pumped from wells was modeled as being returned to the ground via on-site septic systems.

III.3 CHATHAM CONTRIBUTORY AREAS

Revised watershed boundaries were determined by the United States Geological Survey (USGS) for each of the five major embayment systems within the Town of Chatham (Muddy Creek, Bassing Harbor/Ryder Cove/Frost Fish Creek/Crows Pond, Stage Harbor, Sulphur Springs/Bucks Creek, and Taylor's Pond) (Figure III-1). Table III-1 summarizes the percent difference in embayment watershed between watershed delineations utilized in previous Chatham assessments (e.g., Stearns and Wheler, 1999) and the newly delineated watersheds obtained using the USGS Cape Cod Groundwater Model. The overall areas of the watersheds to the majority of the embayment systems generally do not change significantly. However, the watershed areas to Little Mill Pond and Frostfish Creek are significantly reduced (36 and 63%, respectively). Ten-year groundwater time-of-travel areas, and contributing areas to selected "large" ponds within each of the five embayment watersheds were also determined (Ponds: Bassing, Emery, Goose, Lovers, Mill, Newty, Schoolhouse, Stillwater, Trout, White). Contributing areas for fresh ponds were delineated if the pond was larger than 3 model grid cells (400 ft X 400 ft each).

Model outputs of watershed boundaries were "smoothed" to correct for the grid spacing, more accurate characterization of the shoreline, and refinement of the embayment segmentation to more closely match the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the Cape Cod Commission, USGS and the rest of the MEP Technical Team. Overall, 52 sub-watershed areas were delineated relating to the 5 embayment systems within the Town of Chatham. Final watershed boundaries are depicted in Figure III-2 (watershed map). Table III-2 provides the daily discharge volumes for various watersheds as calculated by the groundwater model; these volumes were used to assist in the salinity calibration of the tidal hydrodynamic models.

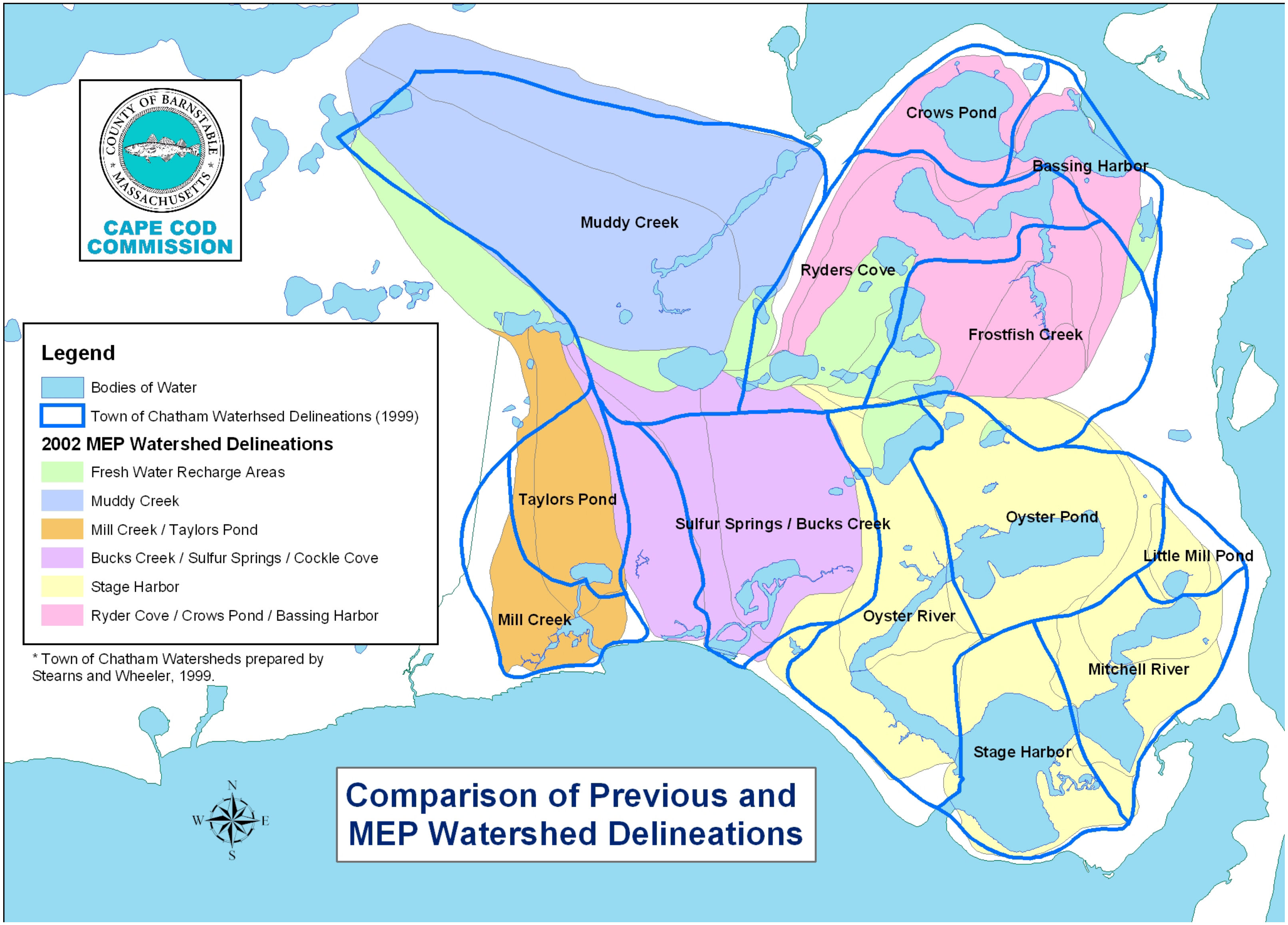


Figure III-1. USGS Cape Cod Groundwater Model revised Town of Chatham embayment watershed delineations

Table III-1. Percent difference in delineated embayment watershed areas between old and newly revised delineations.

Chatham System	*New Watershed (acres)*	*Old Watershed (acres)*	*% Difference*
Muddy Creek System			
Muddy Creek	1871	1855	1%
Stage Harbor System			
Little Mill Pond	83	129	-36%
Mitchell River	521	501	4%
Stage Harbor	554	522	6%
Oyster River	761	772	-1%
Oyster Pond	782	757	3%
Taylor's Pond System			
Taylor's Pond	350	349	0%
Mill Creek	288	340	-15%
Sulfur Springs System			
Sulfur Springs / Bucks Creek	828	793	4%
Basing Harbor System			
Ryders Cove	966	843	15%
Crows Pond	358	306	17%
Frostfish Creek	313	849	-63%
Bassing Harbor	244	287	-15%

Sub-System Name	ID#
Mill Pond Fresh	1
Goose Pond	2
Trout Pond	3
Schoolhouse Pond	4
Stillwater Pond	5
Lovers Lake	6
Emery Pond	7
White Pond	8
Newty Pond	9
Bassing Pond	10
Muddy Creek Well	11
Lower Muddy Creek	12
Lower Muddy Creek 10 E	13
Lower Muddy Creek 10 W	14
Upper Muddy Creek	15
Upper Muddy Creek 10 E	16
Upper Muddy Creek 10 W	17
Mill Creek	18
Mill Creek 10	19
Taylors Pond	20
Taylors Pond 10	21
Cockle Cove	22
Cockle Cove 10	23
Bucks Creek	24
Bucks Creek 10	25
Sulfur Springs	26
Sulfur Springs 10	27
Stage Harbor	28
Stage Harbor 10	29
Lower Oyster River	30
Oyster River	31
Oyster River 10	32
Oyster Pond	33
Oyster Pond 10 S	34
Oyster Pond 10 N	35
Oyster Pond 10 W	36
Mitchell River	37
Mill Pond Salt	38
Mill Pond Salt 10 E	39
Mill Pond Salt 10 W	40
Little Mill Pond	41
Ryders Cove	42
Ryders Cove 10 S	43
Ryders Cove 10 E	44
Ryders Cove 10 W	45
Crows Pond	46
Crows Pond 10	47
Bassing Harbor	48
Bassing Harbor 10	49
Frostfish Creek	50
Frostfish Creek 10	51
Upper Frostfish Creek	52

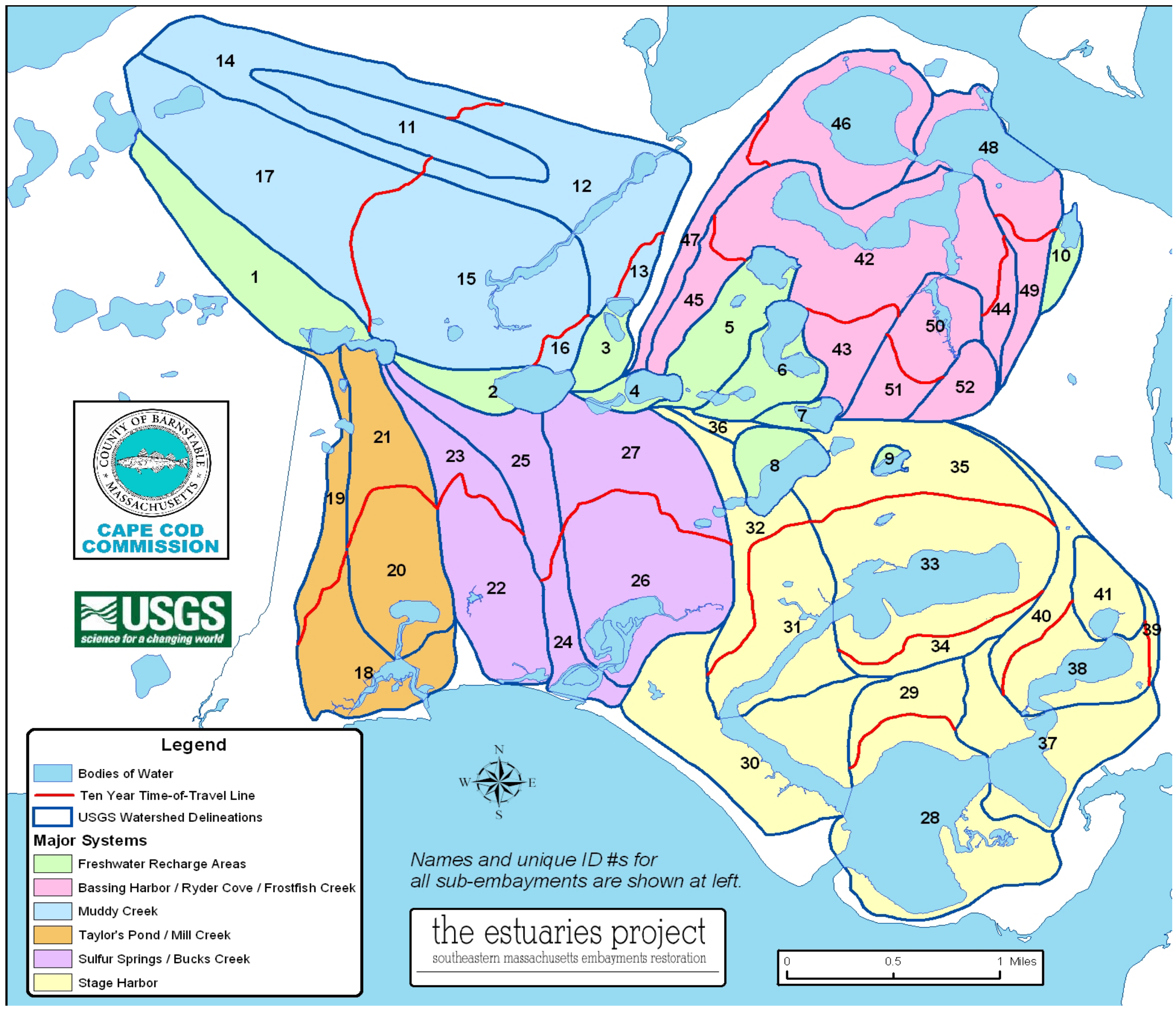


Figure III-2. Watershed and sub-watershed delineations for each of the five major embayment systems within the Town of Chatham, MA. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the figure legend (left). Sub-watersheds to embayments were selected based upon the functional estuarine sub-units in the water quality model (see section VI).

Table III-2. Daily groundwater discharge to each of the major sub-embayments to the five major embayments within the Town of Chatham, MA, as determined from the groundwater model.

Watershed	Discharge		Watershed	Discharge	
	ft^3/day	m^3/day		ft^3/day	m^3/day
Bassing Harbor	49,835	1411	Mitchell River	24,945	706
Bucks Creek	19,037	539	Muddy Creek	190,410	5392
Cockle Cove Creek	82,466	2335	Oyster Pond	185,580	5255
Crows Pond	71,255	2018	Oyster River	117,080	3315
Frost Fish Creek	47,728	1352	Ryder Cove	191,530	5424
Little Mill Pond	18,710	530	Stage Harbor	44,180	1251
Mill Creek	80,189	2271	Sulphur Springs	127,830	3619
Mill Pond	55,140	1561	Taylor Pond	29,448	834
			Upper Muddy Creek	291,190	8246

III.3.1 Well Pumping Effects: Taylors Pond / Mill Creek Watershed

During the review of the Town of Chatham's Comprehensive Wastewater Management Plan effort, concerns were raised as to the effect of the drinking water withdrawal wells (located northwest of Taylors Pond) on the groundwater flow to the Taylors Pond System. These wells operate seasonally to meet summer water-use demand. The issue of the wells was raised when preliminary nitrogen loading assessments completed prior to the initiation of the MEP had difficulty reconciling observed nitrogen concentrations in the Taylor's Pond with estimates based on watershed land uses (Applied Coastal, *et al*., 2001). During a review (by the Town Wastewater Technical Committee and MEP staff) of information used to develop the watershed nitrogen loads under the town's facility plan (Stearns and Wheler, 1999), it was determined that the impact of pumping from nearby Harwich and seasonal Chatham municipal drinking water supply wells was not considered in the initial assessment. As a result, the Town initiated a further investigation of the effects of the water withdrawals on groundwater flow and contributing area to the Taylors Pond System (Earth Tech 2002).

Given the results of the Earth Tech study which indicated that water withdrawals could be influencing groundwater flow patterns in the region of the Taylors Pond System, the MEP Team undertook further modeling studies. The newly constructed USGS groundwater model was used to address the following questions: (a) to what extent are the Harwich and seasonal Chatham municipal drinking water supply wells altering the watershed boundaries to Taylor's Pond and Mill Creek and (b) are these well withdrawals causing seasonal changes in groundwater discharge rates to the receiving estuarine system.

USGS staff conducted modeling runs based upon the Town's recorded winter and summer pumping extremes. These pumping data were taken from a review of monthly withdrawal records between 1995 and 2000. These monthly extremes were then treated as steady-state conditions in order to evaluate the impact on the Taylor's Pond and Mill Creek watershed delineations. The concept was to constrain the maximum extent of seasonal shift in watershed boundary resulting from seasonal water withdrawal.

The results of the MEP modeling effort were qualitatively consistent with the previous study showing a shift in watershed boundary. However, the MEP study indicated that the shift was small and would have little effect on the nitrogen discharge rate from the watershed to either Taylors Pond or Mill Creek (Figure III-3). Further examination of the results indicated that while well withdrawals produced little effect on nitrogen loading to the Taylors Pond System, the spatial coverage of the MEP watersheds differed significantly from the boundaries used in the earlier nitrogen loading study. Therefore, it appears that the difficulties reconciling the monitoring data with the nitrogen loading estimates in the previous nitrogen modeling studies resulted primarily from the areal coverage of the contributing area rather than a seasonality of withdrawal.

Figure III-3 compares the model watershed outputs from the winter and summer conditions and the average conditions. This comparison shows that the summer pumping of the wells causes a slight movement of the watersheds toward the east, but that the average condition watersheds are appropriate for the subsequent land use and nitrogen loading analysis.

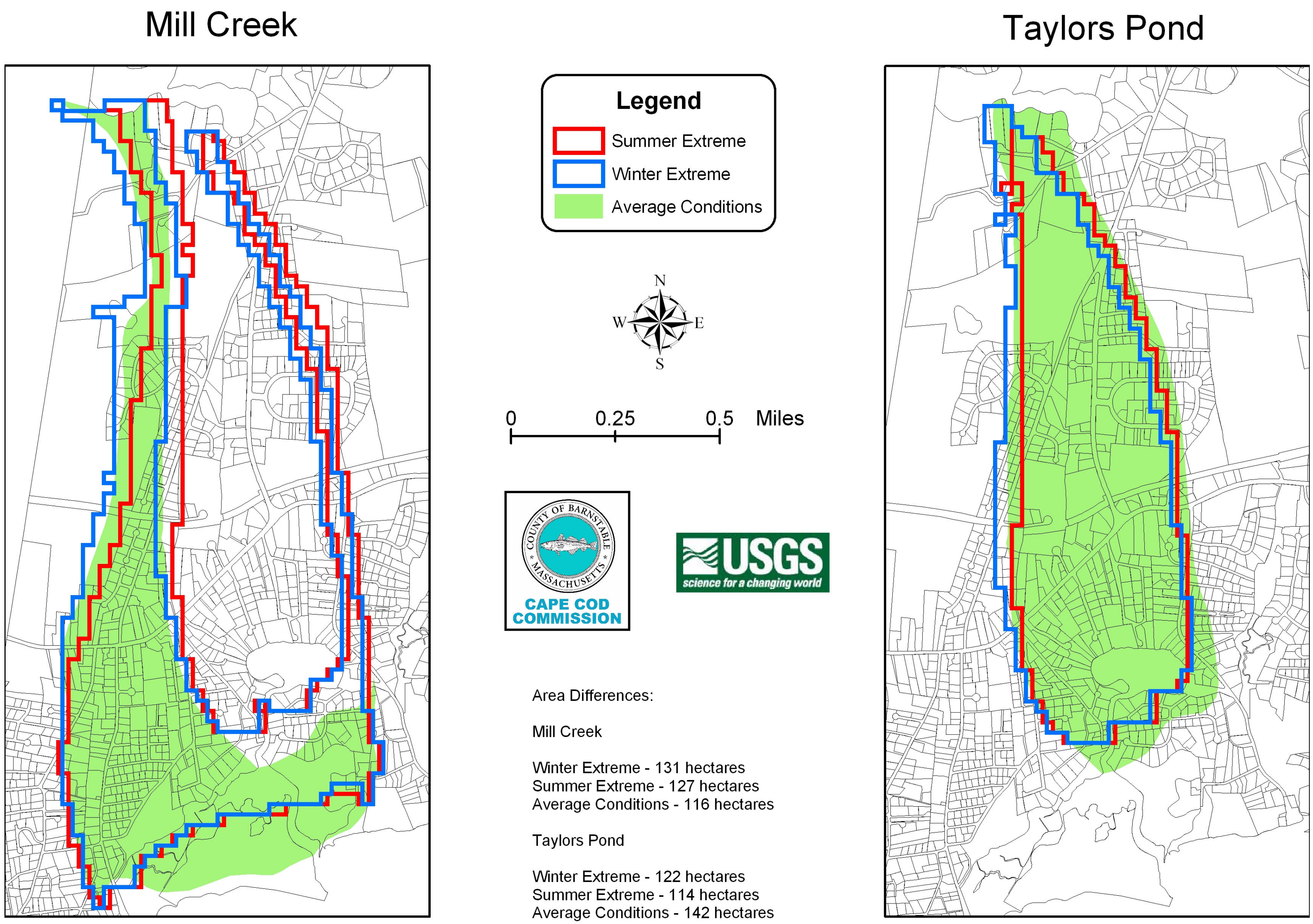


Figure III-3. USGS Cape Cod Model watershed outputs illustrating winter and summer pumping conditions relative to average conditions.

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

IV.1 WATERSHED LAND USE BASED NITROGEN LOADING ANALYSIS

Management of nutrient related water quality and habitat health in coastal waters requires determination of the amount of nitrogen transported by freshwaters (surface water flow, groundwater flow) from the surrounding watershed to the receiving embayment of interest. In southeastern Massachusetts, the nutrient of management concern for estuarine systems is nitrogen and this is true for the embayments within the Town of Chatham. Determination of watershed nitrogen inputs to Chatham's embayments requires the (a) identification and quantification of the nutrient sources and their loading rates to the land or aquifer, (b) confirmation that a groundwater transported load has reached the embayment at the time of analysis, and (c) quantification of nitrogen attenuation that can occur during travel through lakes, ponds, streams and marshes. This latter natural attenuation process is conducted by biological systems which 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 (specifically nitrogen regeneration from sediments). Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters and leads to errors in predicting water quality if it is not included in determination of summertime nitrogen load.

The MEP project team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP technical team staff, CCC staff developed nitrogen loading rates (Section IV.1) within each of the 52 subwatersheds to the 5 embayment systems (Section III). After completing a quality check of land use and reviewing water quality modeling, the 10 year time of travel subwatersheds were eliminated and the number of subwatersheds was reduced to 29. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to ponds and embayments.

In order to determine nitrogen loads from large watersheds, it is not possible to conduct measurements of individual lot-by-lot nitrogen loading. Instead, the Linked Watershed-Embayment Management Model (Howes & Ramsey 2001) uses a land-use Nitrogen Loading Sub-Model based upon subwatershed-specific land-uses and pre-determined nitrogen loading rates. The model used Chatham and Harwich specific land-use data transformed to nitrogen loads using both regional nitrogen load factors and local site-specific data (such as water use). Determination of the nitrogen loads required obtaining site-specific information regarding the wastewater, fertilizers, runoff from impervious surfaces and atmospheric deposition. The primary regional factors were derived for southeastern Massachusetts from direct measurements. The resulting nitrogen loads represent the "potential" nitrogen load to each receiving embayment, since attenuation during transport has not yet been included.

Natural attenuation of nitrogen during transport from land-to-sea (Section IV.2) was determined based upon site-specific studies within the Lovers Lake/Stillwater Pond discharge to Ryder Cove and within Frost Fish Creek. Attenuation during transport through each of the major fresh ponds, within the 5 embayment watersheds, was determined through (a) comparison with other Cape Cod lake studies and (b) data collected on each pond. Nitrogen recycling was also determined within each of the 5 embayment systems. Measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying watercolumn. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

IV.1.1 Land Use and Database Preparation

MEP Technical Staff obtained digital parcel and tax assessors' data from the Towns of Chatham and Harwich. Chatham land use data is from 2002, while Harwich data is from 1999. These two databases were combined by using Geographic Information System (GIS) analysis by the MEP (Cape Cod Commission GIS Department).

Figure IV-1 shows the land uses within the study area; assessors land uses classifications (MADOR, 2002) are aggregated into eight land use categories: 1) residential, 2) commercial, 3) industrial, 4) undeveloped, 5) cranberry bog, 6) golf course, 7) public service, and 8) road right-of-way. Within the five main watersheds considered, the predominant land use is residential, most of which are single family residences. Single family residences occupy approximately 68% of the total land area and 89% of the total parcels (Figure IV-2). Commercial properties are generally concentrated along Route 28, which loops through the Town of Chatham.

In order to estimate wastewater flows within the study area, MEP staff also obtained 2001 water use information from the Town of Chatham and 2000 water use information from the Harwich Water Department. In addition, information on flow, effluent quality, and the service area delineation for the Chatham Wastewater Treatment Facility (WWTF) were obtained. The water use information was linked to the parcel and assessors data using GIS techniques.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

All wastewater is returned to the aquifer underlying Chatham either through the Town's municipal WWTF or individual on-site septic systems. The wastewater in Chatham is predominantly treated through on-site septic systems. Only 4% (266 of the 6,926) of the parcels within the study area are connected to the Town of Chatham wastewater treatment facility. The parcels connected to the WWTF are predominantly located within the watershed to the Stage Harbor System (Figure IV-6).

In order to check the reliability of parcel water use as a proxy for wastewater flow, influent flow at the WWTF was compared to parcel water use within the service area. Previous assessment of WWTF had found that measured water use within the service area closely matched influent at the WWTF. This previous assessment assumed that 90% of the water use throughout Chatham was returned to the aquifer via septic systems (Stearns and Wheler, 1999). Comparison of the 2001 water use with influent flow at the WWTF revealed that influent flow was 71% of the measured water use within the service area.

In order to address this observed difference, WWTF flows and other factors that might have caused water uses to increase in 2001 were investigated. WWTF influent flows between 1998 to 2002 were made available by the Chatham Department of Public Works. Review of these flows shows that average annual influent flow at the WWTF during the period is 40.32 million gallons (MG) with a range of 38.46 (2001) to 42.48 (2000) MG. Annual influent flows showed only about a 10% range over the five years reviewed. Since there is little change in

Sub-Embayment	ID#
Mill Pond (Fresh)	1
Goose Pond	2
Trout Pond	3
Schoolhouse Pond	4
Stillwater Pond	5
Lovers Lake	6
Emery Pond	7
White Pond	8
Newty Pond	9
Bassing Pond	10
Lower Muddy Creek	11
Upper Muddy Creek	12
Mill Creek	13
Taylors Pond	14
Cockle Cove	15
Bucks Creek	16
Sulfur Springs	17
Stage Harbor	18
Lower Oyster River	19
Oyster River	20
Oyster Pond	21
Mitchell River	22
Mill Pond	23
Little Mill Pond	24
Ryders Cove	25
Crows Pond	26
Bassing Harbor	27
Frostfish Creek	28
Upper Frostfish Creek	29

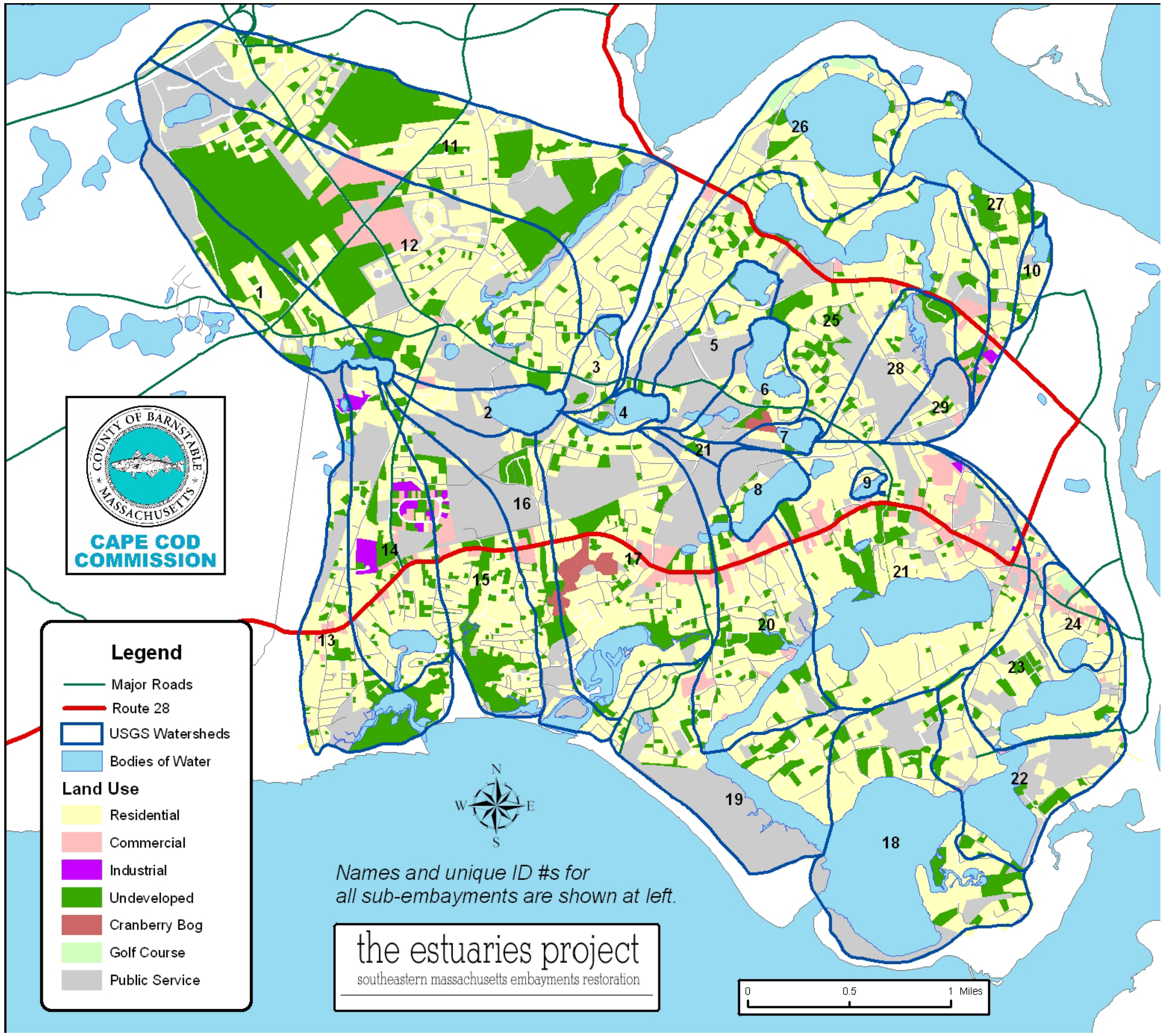


Figure IV-1. Land-use by parcel for the 5 embayment systems. Watershed data encompasses parts of the Towns of Chatham and Harwich, MA.

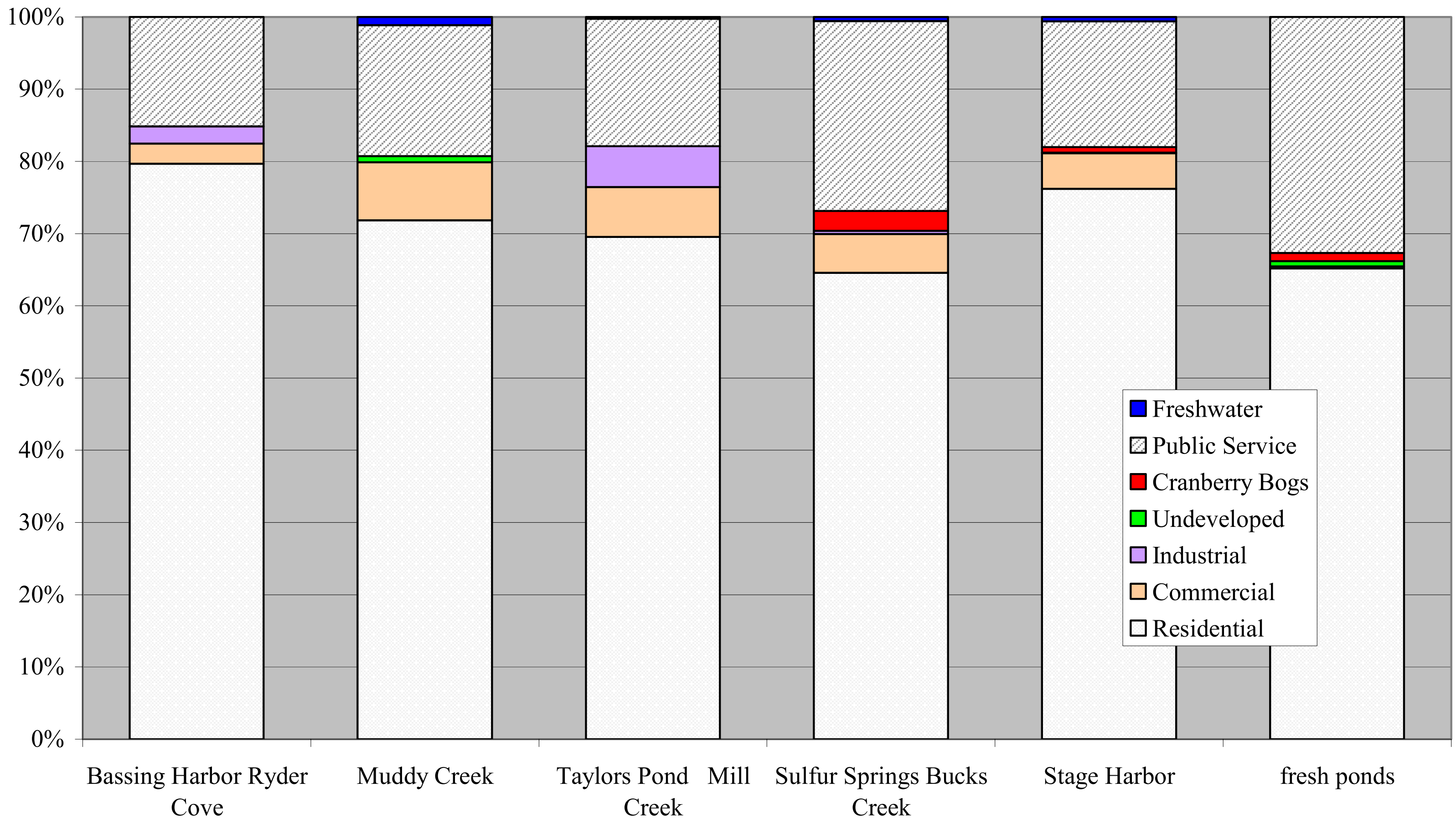


Figure IV-2. Distribution of land-uses within the watersheds to the 5 embayment systems and their freshwater ponds.

water use during this period and flow during 2001 was at the low end of the observed range, it seemed likely that that the observed difference in water use to WWTF influent volume within the service area might represent a shift in water use to purposes other than wastewater, probably lawn and shrub irrigation.

Between 1999 to 2001, annual precipitation in Chatham was cumulatively 15.4 inches below average (Figure IV-3). Since precipitation is the sole source of groundwater on Cape Cod, corresponding regional groundwater levels declined, with winter high elevations barely reaching long-term average conditions (Figure IV-4). Winter is usually the period of greatest recharge and, thus, replenishment of the aquifer and corresponding water levels. The Chatham DPW responded to the 1999 to 2001 drought by instituting a voluntary water ban in July 2001 and a mandatory ban in August 2002.

Review of annual pumping and precipitation records between 1993 and 2001 (Figure IV-5 shows that more precipitation generally results in less pumping; statistical review shows a fairly good linear relationship (R^2 = 0.55). In 2001, water pumping from municipal wells was 18% higher than in 1997. Given that the previous nitrogen loading assessment (Stearns and Wheler 1999)assumed a 10% consumptive loss in their nitrogen loading calculations, the observed 29% difference between water use and wastewater influent (71% return) appears to closely match the combination of a 10% normal consumptive loss plus an 18% increase in non-wastewater associated water use. Based on this analysis, MEP staff concluded that for the 2001 water-use data, the most appropriate breakdown of measured water use is 71% associated with wastewater and 29% for normal consumptive loss and drought associated activities (e.g. irrigation). Correspondingly, wastewater estimates for parcels with water use information were determined by multiplying water use by 0.71.

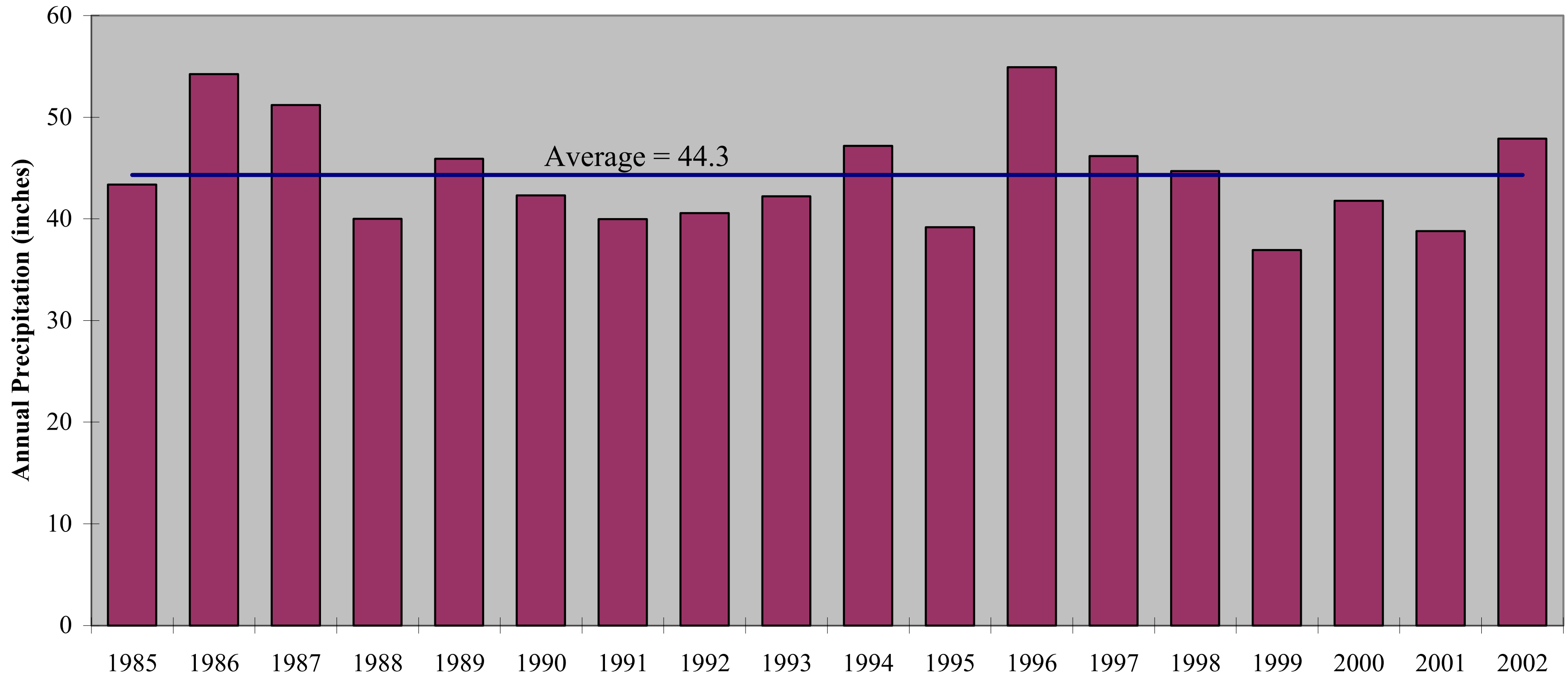
Although this estimate is appropriate for parcels with measured water use, 821 (14%) of the parcels in the study area do not have water use in the available database. These parcels are assumed to utilize private wells. A water use estimate for these parcels was developed based on measured water use from similar land uses. Of these 821 parcels without water use data, 97% are classified as residential parcels (land use codes 101 to 112) or condominium parcels and the remainder are commercial (land use codes 300 to 389). In order to address these parcels, MEP reviewed water use for residential and commercial properties in Chatham with water supply accounts (Table IV-1).

Table IV-1. Water Use in Town of Chatham

Land Use	State Class Codes	# of Parcels	Water Use (gallons per day)		
			Average	Median	Range
Residential*	101	4,420	210	154	4 to 3,077
Commercial	300 to 389	137	580	186	4 to 6,915
Industrial	400 to 433	12	522	123	18 to 2,656

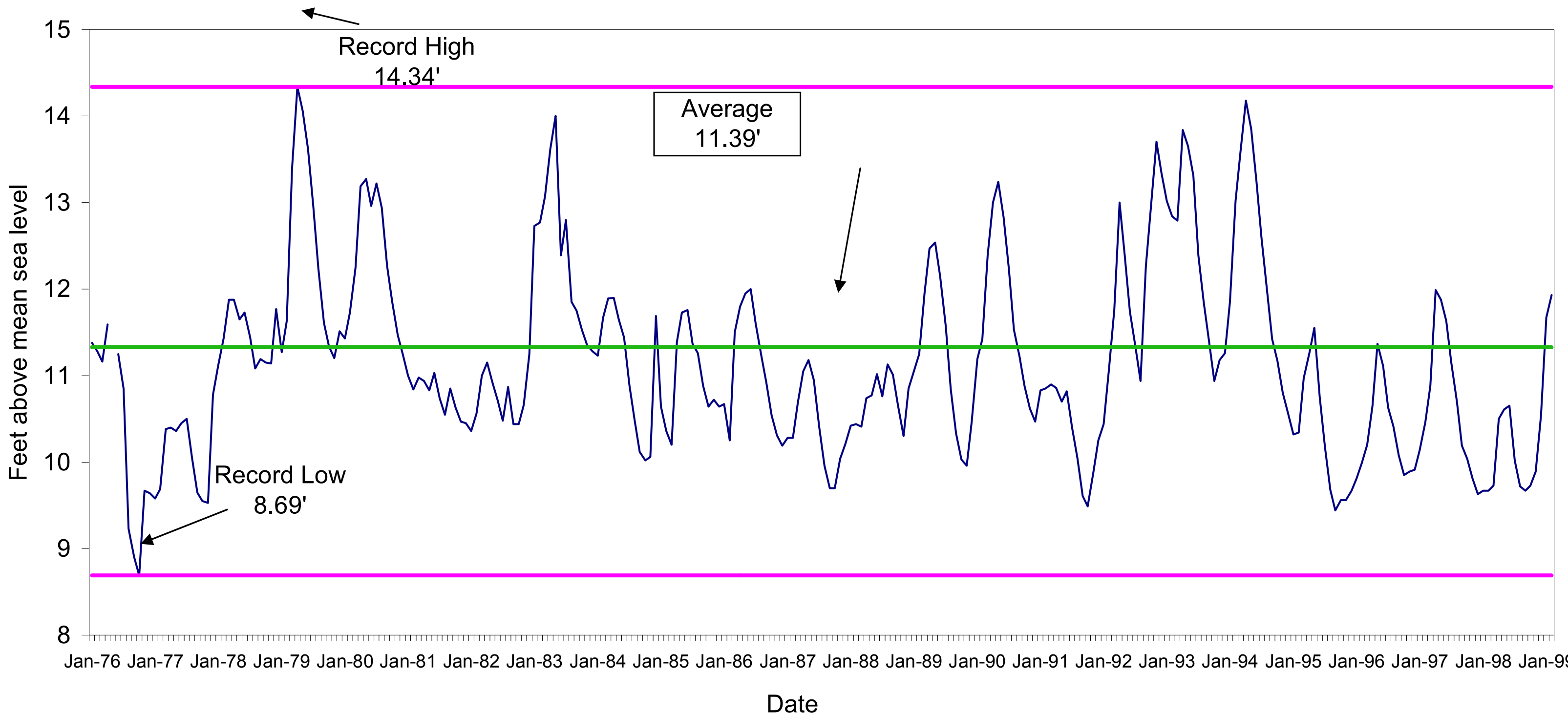
*All values are based on land use from entire town

Review of Chatham water use found that significant differences existed between average and median water use flows in the various land use categories. The average water uses for commercial and industrial parcels are more than double the median, while the residential average is nearly 60 gpd greater than the median. In order to evaluate whether the average or median use data was more appropriate for determining residential wastewater flows for developed parcels without water use information.



source: 1985 thru 2/95 data obtained from National Weather Service Chatham Station; remaining data to present recorded at Town of Chatham Water Quality Laboratory

Figure IV-3. Annual Precipitation Chatham, Massachusetts.



Data from Cape Cod Commission records

Figure IV-4. CGW138 Hydrograph. Trace indicates the water table elevation at the well site from 1980-2002.

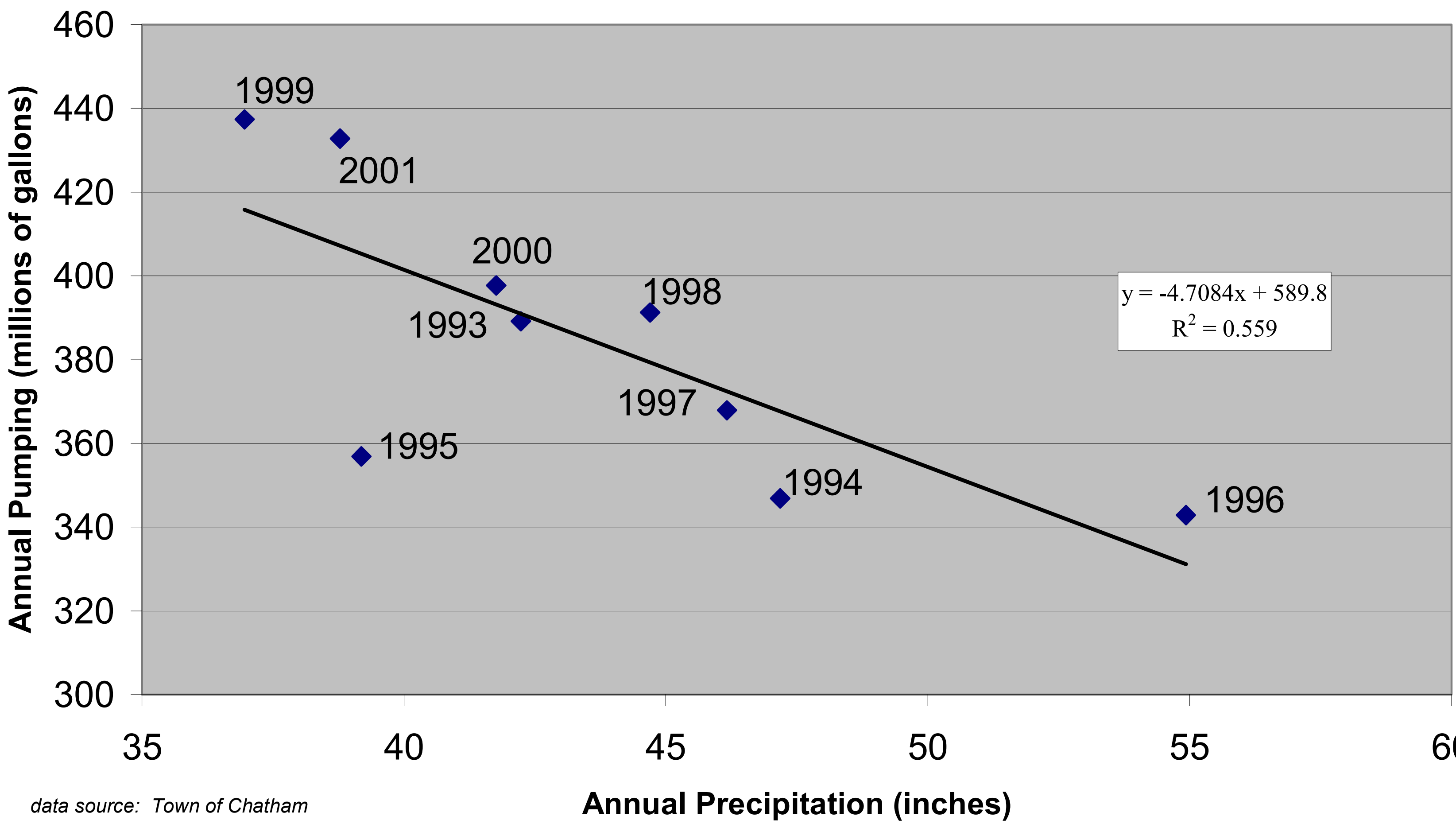


Figure IV-5. Water Supply relative to Precipitation Amounts (annual)

The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd). Therefore, based on these regulations each person would generate 55 gpd. Average occupancy within the town of Chatham during the 2000 US Census was 2.1 people per household. If 2.1 is multiplied by 55 gpd, the average household would generate 115 gpd of wastewater, which is nearly equal to the median residential estimate of 108 gpd based on 2001 water use (154 gpd water use multiplied by 0.71).

Because the water use is measured on an annual basis, seasonal occupancy rates for residences are indirectly accounted for in the annual water uses. In order to provide an additional check of whether the water use agreed with other measures of seasonality, 2000 US Census information was examined. The 2000 Census estimates that 3,147 of the 6,743 housing units (46.7%) in Chatham were occupied for “seasonal, recreational, or occasional use.” Previous estimates of summer population increases have estimated that the Cape’s population triples during the summer. In order for Chatham’s summer population to triple, the seasonal housing units would need to be occupied at twice the year-round occupancy or 4.2 people per household. Average household water use during the summer using the Title 5 flow of 55 gpd/person would be 232 gpd. If this use is assumed to occur for three months and is averaged with 115 gpd for the housing units occupied throughout the year, the resulting annual residential average is 144 gpd. This flow is remarkably close to 155 gpd, the median water use flow in 2001 (see Table IV-1-Water Use).

Based on this analysis, project staff felt that the median residential water use was most appropriate for use in the nitrogen loading calculations for developed residential parcels without water use information and for new residential parcels determined from the buildout assessment. Similar comparisons were not available for the commercial or industrial water uses, which have a much wider range of land uses. Average water use derived from existing commercial and industrial sites were assigned to similar land uses without water use information and for new parcels determined from the build-out assessment.

Nitrogen Loading Input Factors: Residential Lawns

In most southeastern Massachusetts watersheds, nitrogen applied to the land to fertilize residential lawns is the second major source of nitrogen to receiving coastal waters after wastewater associated nitrogen discharges. However, residential lawn fertilizer use has rarely been directly measured in previous watershed-based nitrogen loading investigations. Instead, lawn fertilizer nitrogen loads have been estimated based upon a number of assumptions: a) each household applies fertilizer, b) cumulative annual applications are at3 pounds per 1,000 ft^2, c) each lawn is 5000 sq. ft., and d) only 25% of the nitrogen applied reaches the groundwater (leaching rate). Because many of these assumptions had not been rigorously reviewed in over a decade, the MEP undertook an assessment of lawn fertilizer application rates and a review of leaching rates for inclusion in the land-use Nitrogen Loading Sub-Model.

The initial effort was to determine nitrogen fertilization rates for residential lawns in the Towns of Falmouth, Mashpee and Bourne, and related to inland, fresh ponds and embayments sub-watershed regions. Based upon ~300 interviews and over 2,000 surveys, a number of findings emerged: 1) average residential lawn area is ~5000 sq. ft., 2) half of the residences did not fertilize at all, and 3) the weighted average rate was 1.44 applications per year, rather than the 4 applications per year recommended on the fertilizer bags. Integrating the average residential fertilizer application rate with a leaching rate of 20% results in a fertilizer contribution of N to groundwater of 1.08 lb N per residential lawn for use in the nitrogen loading calculations. It is likely that this still represents a conservative estimate of nitrogen load from residential lawns. It should be noted that professionally maintained lawns were found to have the higher rate of fertilization (loss to groundwater of 3 lb/lawn/yr).

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for impervious surfaces and natural areas are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner, *et al*., 1992) and Massachusetts DEP's Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). Factors used in the nitrogen loading analyses for Chatham's embayments are listed in Table IV-2.

<table>
<tr><td colspan="4">Table IV-2. Primary Nitrogen Loading Factors used in Chatham MEP analysis. General factors are from the MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Chatham data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.</td></tr>
<tr><td>Nitrogen Concentrations:</td><td>mg/l</td><td>Recharge Rates:</td><td>in/yr</td></tr>
<tr><td>Wastewater</td><td>35</td><td>Impervious Surfaces</td><td>40</td></tr>
<tr><td>Road Run-off</td><td>1.5</td><td>Natural and Lawn Areas</td><td>26.5</td></tr>
<tr><td>Roof Run-off</td><td>0.75</td><td colspan="2">Water Use:</td></tr>
<tr><td>Direct Precipitation on Embayments and Ponds</td><td>1.09</td><td>For Parcels wo/water accounts:</td><td>gpd</td></tr>
<tr><td>Natural Area Recharge</td><td>0.072</td><td>Single Family Residence</td><td>154</td></tr>
<tr><td>Fertilizer:</td><td></td><td>Commercial Properties</td><td>580</td></tr>
<tr><td>Average Residential Lawn Size (ft^2)*</td><td>5,000</td><td>Industrial Properties</td><td>522</td></tr>
<tr><td>Residential Watershed Nitrogen Rate (lbs/lawn)*</td><td>1.08</td><td rowspan="2">For Parcels w/water accounts:</td><td rowspan="2">Measured annual water use</td></tr>
<tr><td colspan="2">Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined by site-specific information</td></tr>
<tr><td colspan="2">Town of Chatham Municipal WWTF:</td><td colspan="2" rowspan="2">Wastewater Estimates:</td></tr>
<tr><td>Annual Flow (million gallons)</td><td>38.46</td></tr>
<tr><td>Total Nitrogen Effluent Concentration (mg/l)</td><td>7.44</td><td colspan="2">Wastewater determined by multiplying water use by 0.71</td></tr>
</table>

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information was linked to the parcel coverages, parcels were assigned to various watersheds based initially on whether at least 50% or more of the land area of each parcel was located within a respective watershed. Following the assigning of boundary parcels, all large parcels were examined separately and were split (as appropriate) in order to obtain less than a 2% difference between the total land area of each watershed and the sum of the area of the parcels within each watershed. The resulting "parcelized" watersheds are shown in Figure IV-6. This review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Individualized information for parcels with atypical nitrogen loading (small public water supplies, golf courses, etc.) were also assigned at this stage. DEP and Town of Chatham records were reviewed to determine water use for small public water supplies (*e.g.*, non-community public water supplies) and golf course superintendents for two golf courses in the study area were contacted to determine fertilizer application rates.

Following the assignment of all parcels to individual watersheds, tables were generated for each of 29 sub-watersheds to summarize water use, parcel area, frequency, sewer connections, private wells, and road area.

The 29 individual sub-watershed assessments were then integrated to generate nitrogen loading tables relating to each of the sub-embayments within each of the 5 major embayment systems. The sub-embayments represent the functional embayment units for the Linked Watershed-Embayment Model's water quality component.

For management purposes, the aggregated sub-embayment watershed nitrogen loads are separated into various nitrogen sources to support potential nitrogen mitigation alternative development: wastewater (septic systems and the WWTF), fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge from natural areas (Table IV-3 N Load summary). The output of the watershed nitrogen loading effort is the kg N per year (or day) loaded into each sub-embayment's contributing area, by land use category (Figures 7a-e) which is then adjusted for natural nitrogen attenuation during transport before use in the Linked Model.

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally kettle hole depressions that intercept the surrounding groundwater table revealing what some call "windows on the aquifer." Since the ponds are connected to the aquifer, the ecosystems in these ponds have the opportunity to alter the nitrogen loads flowing into them via groundwater flow. This change to the nitrogen load taking place as a result of the hydraulic interaction with the pond occurs before the loads flow back into the groundwater system through the down gradient side of the pond or stream outlet and eventual discharge into an embayment. Table IV-3 N Load summary includes both the unattenuated (nitrogen load to each subwatershed) and attenuated nitrogen loads. The attenuated loads include site-specific studies within the Lovers Lake/Stillwater Pond system and within Frost Fish Creek (see Section IV.2). Except for the site-specific studies, nitrogen attenuation in the ponds was assumed to be 40%.

This assumption was checked through the use of pond water quality information collected during late August 2001 under the Cape Cod Pond and Lake Stewardship (PALS) program, which is a collaborative Cape Cod Commission/SMAST Program. The Town of Chatham Water Quality Laboratory collected dissolved oxygen and temperature profiles, Secchi disk depth readings and water samples at various depths within the following ponds: Emery, Goose, Lovers, Mill, Schoolhouse, Stillwater, White, Trout, and Newty (Figure IV-1). Water samples were analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH.

In order to estimate nitrogen attenuation in the ponds physical and chemical data for each pond was assessed. Available bathymetric information was reviewed relative to measured pond temperature profiles to determine the epilimnion (*i.e.*, well mixed, homothermic, upper portion of the water column) in each pond. Following this determination, the volume of this portion was determined and compared to the annual volume of recharge from each pond's watershed in order to determine how long it takes the aquifer to completely exchange the water in this portion of the pond (*i.e.*, turnover time). Using the total nitrogen concentrations collected within the epilimnion, the total mass of nitrogen within this portion of the pond was determined and, using

Sub-Embayment	ID#
Mill Pond (Fresh)	1
Goose Pond	2
Trout Pond	3
Schoolhouse Pond	4
Stillwater Pond	5
Lovers Lake	6
Emery Pond	7
White Pond	8
Newty Pond	9
Basing Pond	10
Lower Muddy Creek	11
Upper Muddy Creek	12
Mill Creek	13
Taylors Pond	14
Cockle Cove	15
Bucks Creek	16
Sulfur Springs	17
Stage Harbor	18
Lower Oyster River	19
Oyster River	20
Oyster Pond	21
Mitchell River	22
Mill Pond	23
Little Mill Pond	24
Ryders Cove	25
Crows Pond	26
Bassing Harbor	27
Frostfish Creek	28
Upper Frostfish Creek	29

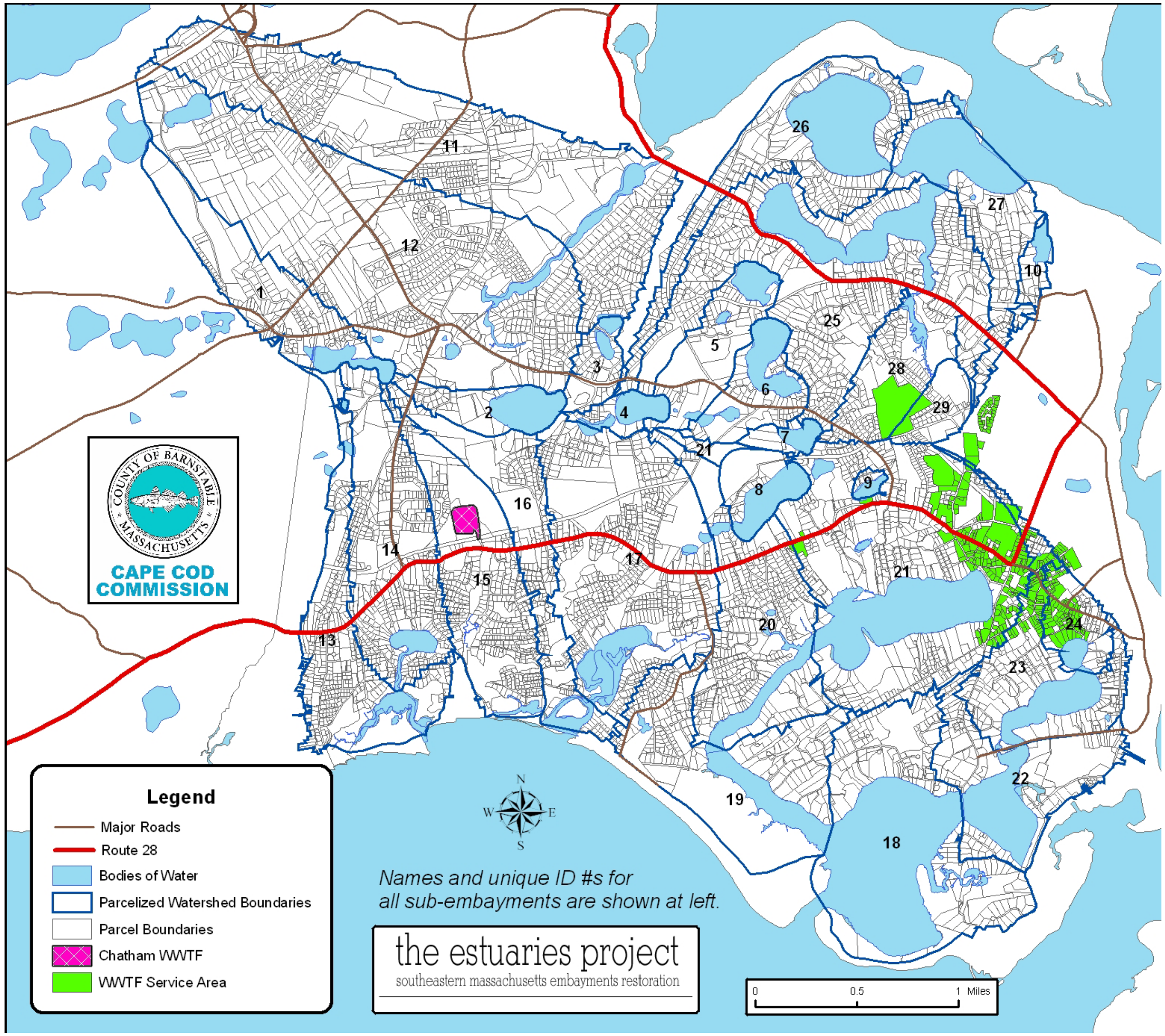


Figure IV-6. Parcel distribution within the watersheds to the 5 embayment systems.

Table IV-3a. Oyster Pond/Stage Harbor System Nitrogen Loads.

*All values in kilograms/year		Chatham N Loads by Input**:							Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout	% of Pond Outflow	UnAtten N Load	Atten %	Atten N Loads	UnAtten N Load	Atten %	Atten N Loads
Oyster Pond / Stage Harbor	7, 8, 9, 18, 19, 20, 21,	11556	791	684	2535	347	1615		15913		15714	17528		17324
Oyster Pond	**7, 8, 9, 21**	**4076**	**251**	**230**	**274**	**106**	**713**		**4936**		**4805**	**5649**		**5516**
Oyster Pond	21	3933	242	209	126	100	707		4610		4610	5316		5316
Newty Pond	9	58	4	1	28	1	0	100%	92	40%	55	92	40%	55
Emery Pond	7	28	3	2	63	3	0	38%	37	40%	22	37	40%	22
White Pond	8	140	7	35	180	7	12	53%	197	40%	118	203	40%	122
Oyster River	**8, 20**	**2169**	**154**	**135**	**252**	**56**	**331**		**2767**		**2698**	**3098**		**3027**
Oyster River	20	2104	151	119	169	52	326		2595		2595	2921		2921
White Pond	8	140	7	35	180	7	12	47%	172	40%	103	177	40%	106
Lower Oyster Rive	**19**	**1361**	**107**	**81**	**230**	**59**	**134**		**1838**		**1838**	**1972**		**1972**
Little Mill Pond	**24**	**494**	**52**	**41**	**44**	**10**	**58**		**642**		**642**	**700**		**700**
Mill Pond Salt	**23**	**565**	**36**	**29**	**229**	**18**	**111**		**877**		**877**	**988**		**988**
Mitchell River	**22**	**2043**	**132**	**117**	**322**	**46**	**93**		**2660**		**2660**	**2754**		**2754**
Stage Harbor	**18**	**848**	**58**	**50**	**1184**	**52**	**175**		**2193**		**2193**	**2368**		**2368**

** sums of unattenuated loads adjusted for pond shore percentages

Note that the N Loads by input show the total nitrogen input to each sub-watershed. However, the fresh ponds sub-watersheds typically contribute to more than 1 sub-embayment and therefore their sub-watershed contribution if apportioned based upon the proportion of shoreline. The total nitrogen load to each fresh pond (e.g. # 7,8,9) are adjusted to their contribution to the receiving sub-embayment in both the "roll-ups" for each sub-embayment and in the present and build-out Nitrogen Loading (right sections of table)

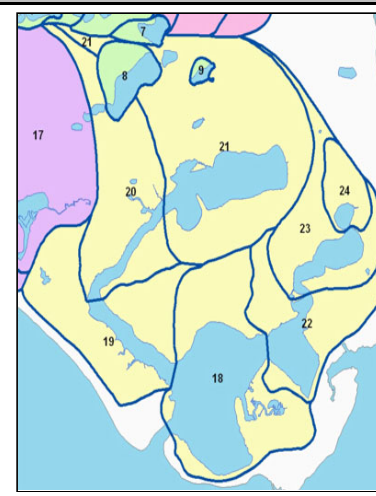


Table IV-3b. Sulphur Springs/Bucks Creek/Cockle Cove System Nitrogen Loads.

*All values in kilograms/year		Chatham N Loads by Input**:								Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	Chatham WWTF	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout	% of Pond Outflow	UnAtten N Load	Atten %	Atten N Loads	UnAtten N Load	Atten %	Atten N Loads
Sulfur Springs / Bucks	1, 2, 15, 16, 17	8257	1107	484	374	334	186	1444		10743		10606	12187		12031
Bucks Creek	**1, 2, 16**	**1303**	**0**	**81**	**61**	**94**	**42**	**284**		**1582**		**1514**	**1866**		**1788**
Bucks Creek	16	1197	0	75	56	48	38	258		1413		1413	1671		1671
Mill Pond Fresh	1	1152	0	57	48	105	37	420	6%	87	40%	52	114	40%	68
Goose Pond	2	155	0	15	8	179	9	0	22%	81	40%	49	81	40%	49
Cockle Cove	**1, 15**	**2094**	**1107**	**163**	**116**	**28**	**51**	**490**		**3560**		**3530**	**4050**		**4011**
Cockle Cove	15	2034	1107	160	114	22	49	468		3486		3486	3954		3954
Mill Pond Fresh	1	1152	0	57	48	105	37	420	5%	74	40%	44	96	40%	57
Sulfur Springs	**2, 17**	**4860**	**0**	**240**	**197**	**212**	**93**	**670**		**5601**		**5561**	**6272**		**6232**
Sulfur Springs	17	4817	0	235	195	163	91	670		5501		5501	6172		6172
Goose Pond	2	155	0	15	8	179	9	0	27%	100	40%	60	100	40%	60

** sums of unattenuated loads adjusted for pond shore percentages

Note that the N Loads by input show the total nitrogen input to each sub-watershed. However, the fresh ponds sub-watersheds typically contribute to more than 1 sub-embayment and therefore their sub-watershed contribution if apportioned based upon the proportion of shoreline. The total nitrogen load to each fresh pond (e.g. # 1, 2) are adjusted to their contribution to the receiving sub-embayment in both the "roll-ups" for each sub-embayment and in the present and build-out Nitrogen Loading (right sections of table)

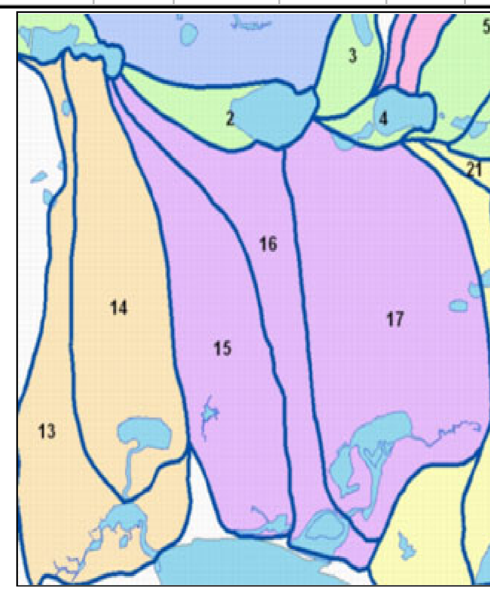


Table IV-3c. Taylors Pond/Mill Creek System Nitrogen Loads.

*All values in kilograms/year		Chatham N Loads by Input**:						% of Pond Outflow	Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Loads	UnAtten N Load	Atten %	Atten N Loads
Taylors Pond/ Mill Creek	**1, 13, 14**	4763	344	258	198	116	1418		5679		5344	7097		6661
Mill Creek	**1, 13**	**2034**	**157**	**111**	**90**	**50**	**382**		**2442**		**2320**	**2824**		**2665**
Mill Creek	13	1783	145	100	67	42	291		2136		2136	2427		2427
Mill Pond Fresh	1	1152	57	48	105	37	420	22%	305	40%	183	397	40%	238
Taylors Pond	**1, 14**	**2728**	**187**	**147**	**108**	**67**	**1035**		**3237**		**3024**	**4273**		**3995**
Taylors Pond	14	2289	166	129	68	53	875		2704		2704	3579		3579
Mill Pond Fresh	1	1152	57	48	105	37	420	38%	533	40%	320	693	40%	416

** sums of unattenuated loads adjusted for pond shore percentages

Note that the N Loads by input show the total nitrogen input to each sub-watershed. However, the fresh ponds sub-watersheds typically contribute to more than 1 sub-embayment and therefore their sub-watershed contribution if apportioned based upon the proportion of shoreline. The total nitrogen load to each fresh pond (e.g. # 1) are adjusted to their contribution to the receiving sub-embayment in both the "roll-ups" for each sub-embayment and in the present and build-out Nitrogen Loading (right sections of table)

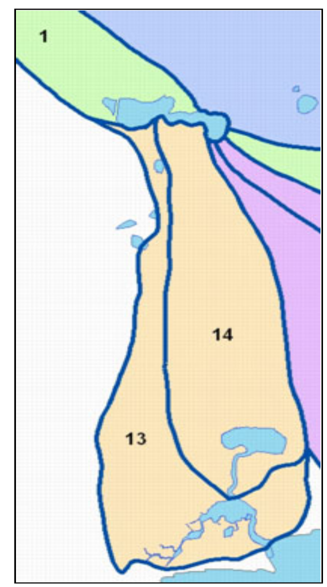


Table IV-3d. Muddy Creek System Nitrogen Loads.

*All values in kilograms/year		Chatham N Loads by Input**:							Present N Loads			Buildout N	
Name	Watershed ID#	Wastewater	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout	% of Pond Outflow	UnAtten N Load	Atten %	Atten N Loads	UnAtten N Load	Atten %
Muddy Creek	**1, 2, 3, 11, 12**	10509	560	572	334	318	1695		12293		11988	13988	
Lower Muddy Creek	**3, 11**	**4291**	**265**	**283**	**118**	**135**	**327**		**5092**		**4953**	**5419**	
Lower Muddy Creek	11	4048	243	249	75	128	309		4744		4744	5053	
Trout Pond	3	243	22	34	43	7	17	100%	349	40%	209	366	40%
Upper Muddy Creek	**1, 2, 12**	**6217**	**295**	**289**	**216**	**183**	**1368**		**7201**		**7035**	**8569**	
Upper Muddy Creek	12	5904	277	276	135	172	1272		6764		6764	8035	
Mill Pond Fresh to MC	1	1152	57	48	105	37	420	23%	321	40%	193	418	40%
Goose Pond	1, 2	155	15	8	179	9	0	32%	131	40%	78	135	40%
Goose Pond	*2*	*155*	*15*	*8*	*179*	*9*	*0*	*100%*	*366*			*366*	
Mill Pond Fresh to GP	*1*	*1152*	*57*	*48*	*105*	*37*	*420*	**6%**	*79*	*40%*	*47*	*102*	*40%*

** sums of unattenuated loads adjusted for pond shore percentages

Note that the N Loads by input show the total nitrogen input to each sub-watershed. However, the fresh ponds sub-watersheds typically contribute to more than 1 sub-embayment and therefore their sub-watershed contribution if apportioned based upon the proportion of shoreline. The total nitrogen load to each fresh pond (e.g. # 1, 2, 3) are adjusted to their contribution to the receiving sub-embayment in both the “roll-ups” for each sub-embayment and in the present and build-out Nitrogen Loading (right sections of table)

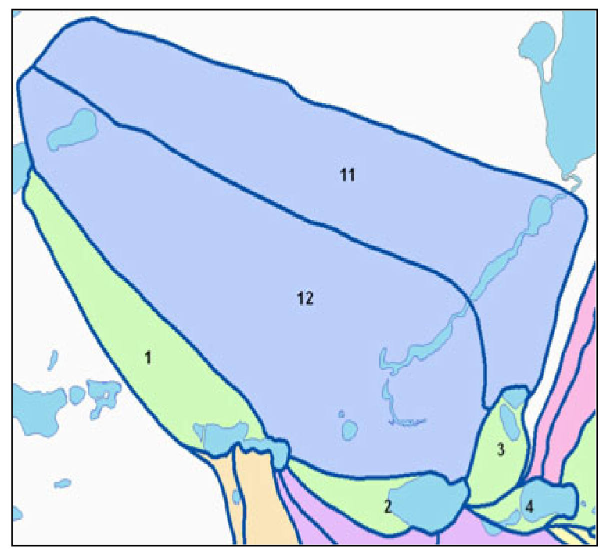
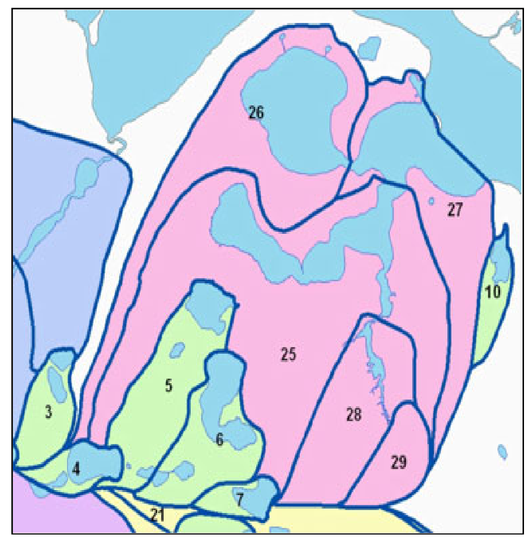


Table IV-3 e. Ryder Cove/Bassing Harbor System Nitrogen Loads.

*All values in kilograms/year		Chatham N Loads by Input**:						% of Pond Outflow	Present N Loads			Buildout N	
Name	Watershed ID#	Wastewater	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Loads	UnAtten N Load	Atten %
Ryders Cove/Bassing	4, 5, 6, 7, 10, 25, 26, 27, 28, 29	8137	465	477	1868	235	1109		11183		10330	12292	
Crows Pond	4, 26	**1871**	**111**	**89**	**512**	**40**	**94**		**2622**		**2618**	**2716**	
Crows Pond	26	1866	111	88	507	40	93		2612		2612	2705	
Schoolhouse Pond	4	95	9	4	112	3	12	5%	11	40%	6	11	40%
Ryders Cove	4, 5, 6, 7, 25, 28, 29	**5384**	**341**	**341**	**957**	**166**	**811**		**7190**		**6357**	**8000**	
Ryder Cove GW	4, 7, 25	3344	201	209	542	92	587		4388		4340	4975	
Ryders Cove	25	3302	197	207	474	89	584		4269		4269	4852	
Schoolhouse Pond	4	95	9	4	112	3	12	26%	58	40%	35	61	40%
Emery Pond	7	28	3	2	63	3	0	62%	62	40%	37	62	40%
Stillwater Pond	4, 5, 6	686	56	59	380	38	96	100%	1218	14%	717	893	14%
Stillwater Pond	5	264	24	29	105	22	23		444			468	
Lovers Lake	6	356	25	26	197	13	64	100%	618	52%	296	682	52%
Schoolhouse Pond	4	95	9	4	112	3	12	69%	156	40%	93	164	40%
Frostfish Creek	28, 29	1353	84	74	35	37	128		1584	18%	1299	1712	18%
Bassing Harbor	10, 27	**883**	**13**	**47**	**399**	**29**	**204**		**1371**		**1355**	**1575**	
Bassing Harbor	27	853	11	46	393	28	202		1332		1332	1533	
Bassing Pond	10	183	11	7	41	5	17	16%	39	40%	24	42	40%

** sums of unattenuated loads adjusted for pond shore percentages

Note that the N Loads by input show the total nitrogen input to each sub-watershed. However, the fresh ponds sub-watersheds typically contribute to more than 1 sub-embayment and therefore their sub-watershed contribution if apportioned based upon the proportion of shoreline. The total nitrogen load to each fresh pond (e.g. # 4, 5, 6, 7, 10) are adjusted to their contribution to the receiving sub-embayment in both the "roll-ups" for each sub-embayment and in the present and build-out Nitrogen Loading (right sections of table)



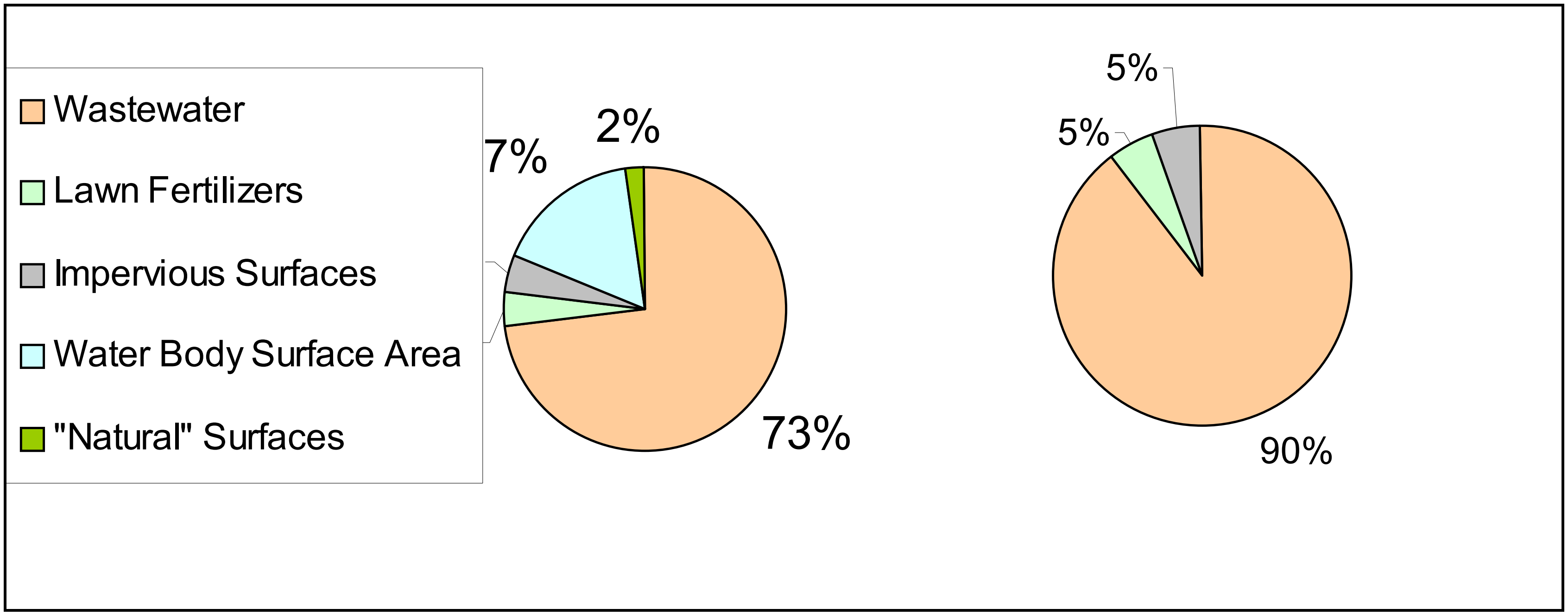


Figure IV-7a. Land use specific unattenuated watershed based nitrogen load (by percent) to Bassing Harbor embayment system.

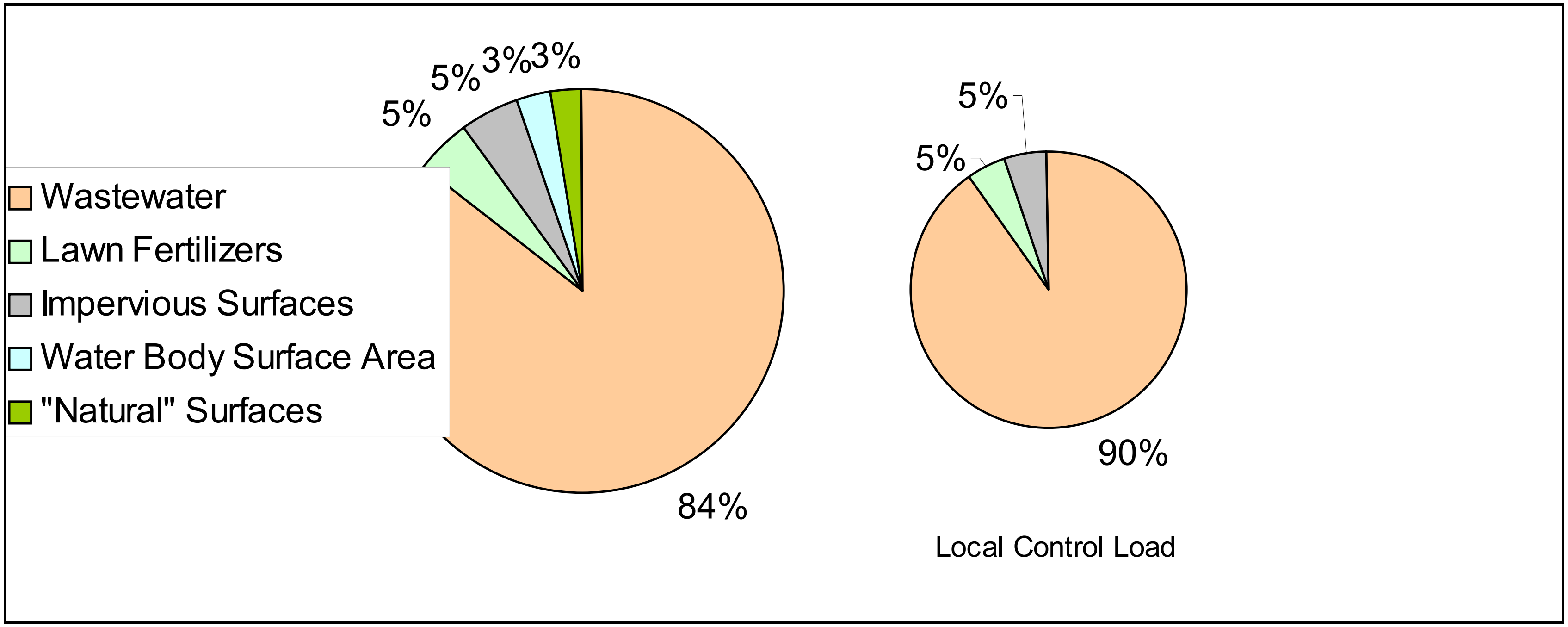


Figure IV-7b. Land use specific unattenuated watershed based nitrogen load (by percent) to Muddy Creek embayment system.

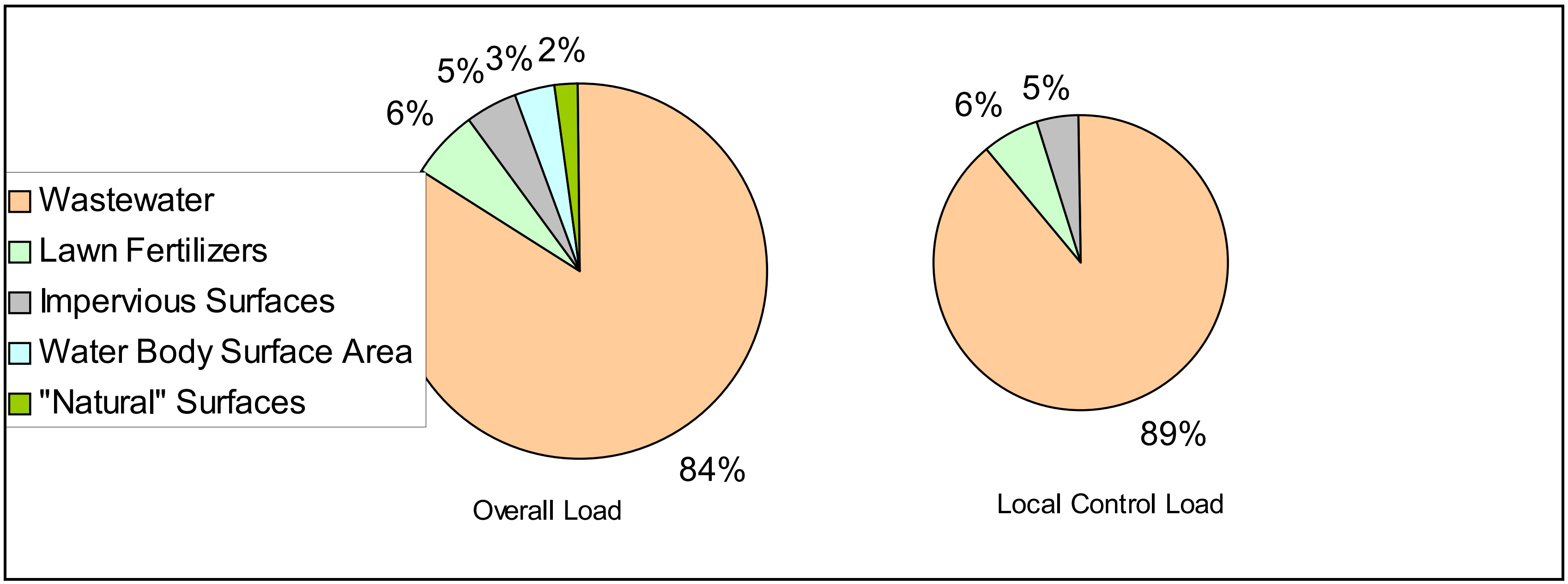


Figure IV-7c. Land use specific unattenuated watershed based nitrogen load (by percent) to Taylors Pond embayment system.

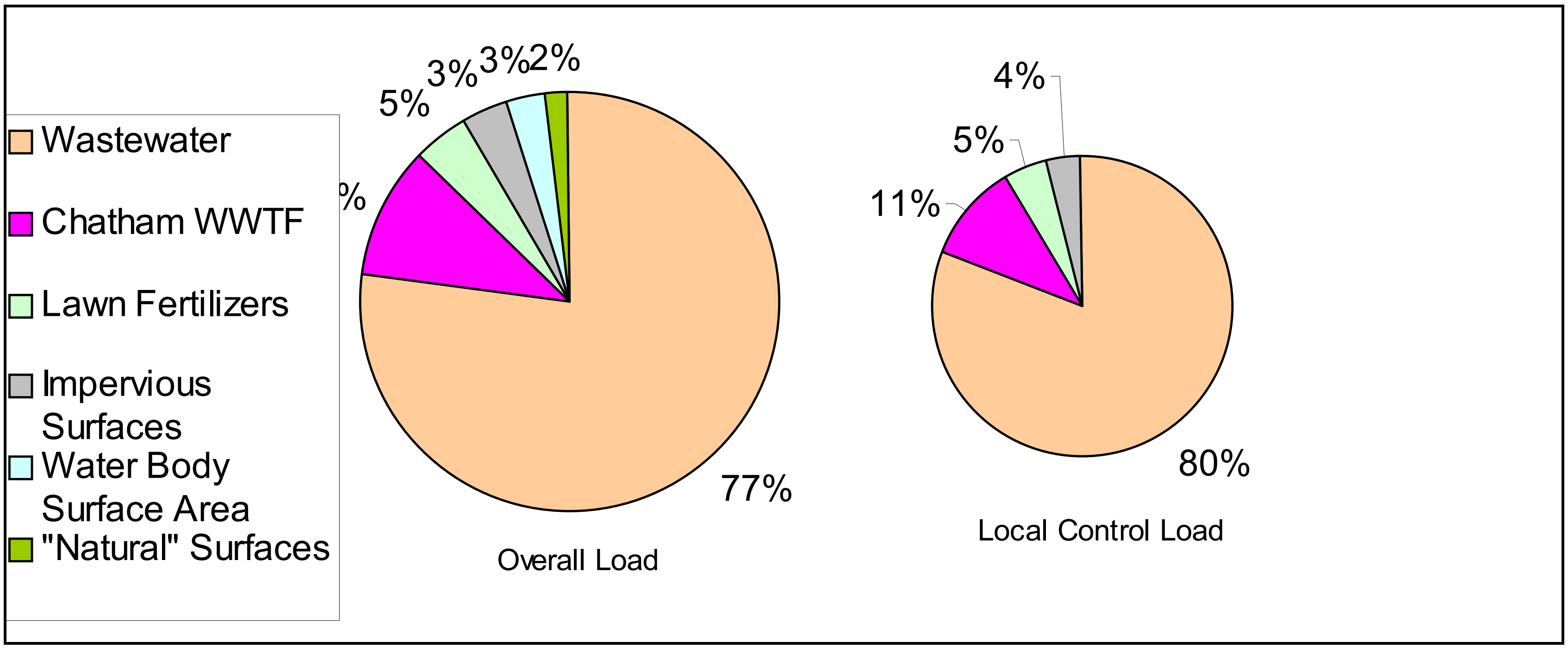


Figure IV-7d. Land use specific unattenuated watershed based nitrogen load (by percent) to Sulphur Springs embayment system.

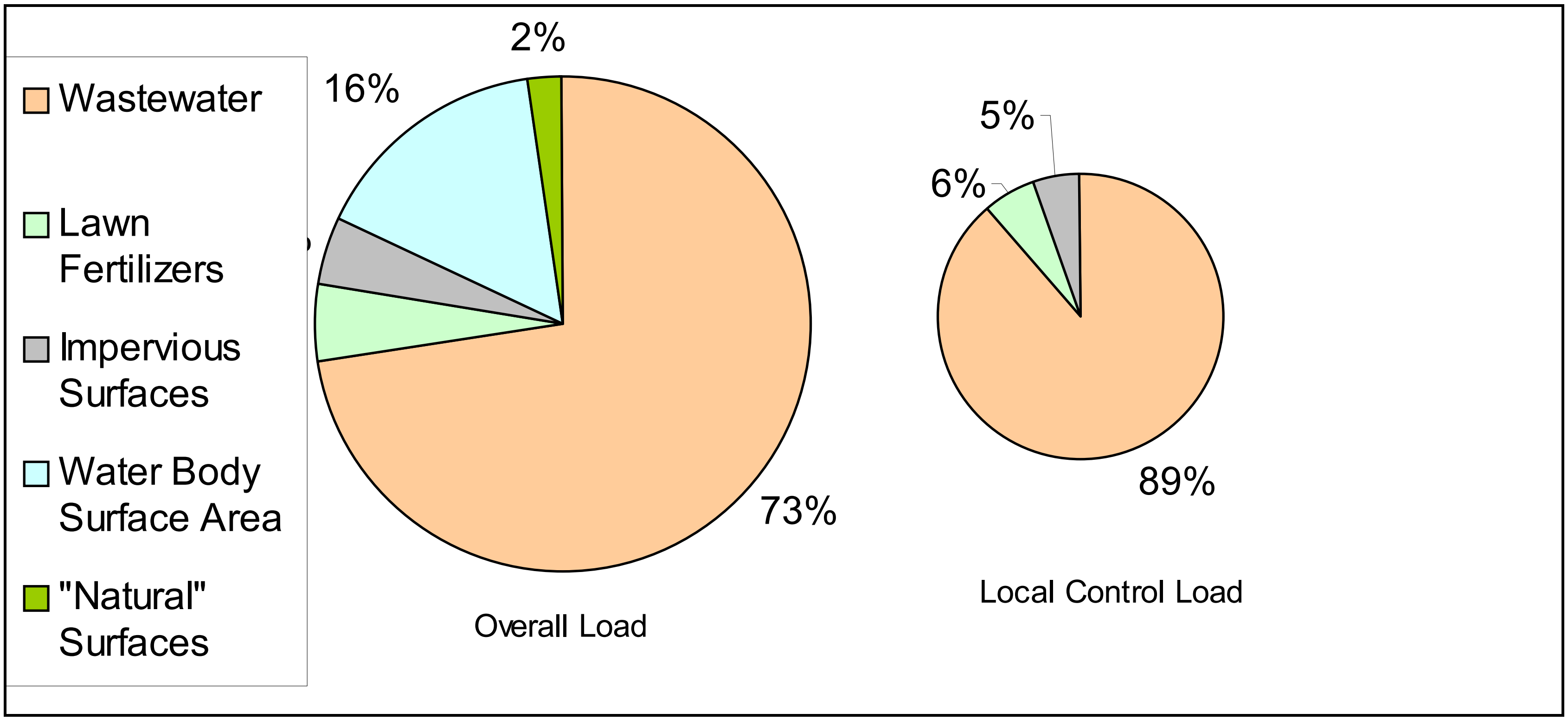


Figure IV-7e. Land use specific unattenuated watershed based nitrogen load (by percent) to Stage Harbor embayment system.

the turnover time, how much of the nitrogen is returned to the aquifer through the downgradient discharge of pond water was determined. In ponds with homothermic water columns, the nitrogen mass within the pond was based on the entire water volume.

Table IV-4 summarizes the pond attenuation estimates calculated from land-use modeled nitrogen inflow loads and nitrogen loads which appear to be recharged to the downgradient aquifer or to outflow streams from each pond based on pond characteristics and measured nitrogen levels. Nitrogen attenuation within these ponds appears to vary between 39 and 95%. However, a caveat to these attenuation estimates is that they are based upon nitrogen outflow loads from summer water column samples, and are not necessarily representative of the annual nitrogen loads that are transferred downgradient. More detailed studies of other southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM 2001) have supported a 40% attenuation factor. Within the Chatham study area, the pond outflows from Lovers Lake allowed a more detailed analysis (Section IV.2) of nitrogen attenuation in this system and attenuation was found to be 52% of total nitrogen input (watershed + atmosphere). This factor is also consistent with the freshwater pond attenuation factors used for the nitrogen balance for Great, Green and Bournes Ponds (embayments) in the Town of Falmouth (Howes and Ramsey 2001).

Table IV-4. Nitrogen attenuation by Chatham Freshwater Ponds based upon late summer 2001 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. Stillwater Pond and Lovers Lake had annual nitrogen and discharge measurements to determine attenuation; only Lovers Lake has full discharge through surface water flow, which yielded an attenuation of 52% (Table IV-5). The MEP Linked N Model uses a value of 40% for the non-stream discharge systems.

Pond	ID	Area acres	Total Depth m	Overall turnover time yrs	N Load Attenuation %
Emery	CH-491	14.11	6.2	3.5	39%
Goose	CH-458	41.25	11.0	8.7	90%
Lovers	CH-428	37.73	9.6	2.9	69%
Mill	CH-440	23.45	2.8	0.3	95%
Schoolhouse	CH-463	22.78	13.2	9.4	93%
Stillwater	CH-396	18.71	13.8	1.3	65%
White	CH-516	40.53	16.2	7.1	88%
Trout	CH-425	4.88	4.8	0.3	94%
Newty	CH-522	5.47	1.7	0.9	81%
				Mean	79%
				s.d.	19%

Since groundwater outflow from a pond can enter more than one down gradient sub-watershed, the length of shoreline on the down gradient side of the pond was used to apportion the attenuated nitrogen load to respective down gradient watersheds. The apportionment was based on the percentage of pond discharging shoreline bordering each down gradient sub-watershed. The percentages of shoreline are shown in Table IV-3 N Load Summary.

Buildout

In order to gauge potential future nitrogen loads resulting from continuing development, the potential number of residential, commercial, and industrial lots within each subwatershed was determined from the GIS database. Assessment began with the state class codes to determine all parcels that are classified as developable: residential land use codes 130 and 131, commercial codes 390 and 391, and industrial codes 440 and 441 (Figure IV-8). Existing zoning maps from the Towns of Chatham and Harwich (Figure IV-9) were then combined with the developable parcels through GIS. Build-out of parcels classified as developable were based on sub-divisions using minimum lot size within each zoning district. All municipal overlay districts (e.g., Districts of Critical Planning Concern, water resource protection districts) were considered in the determination of minimum lot sizes. A nitrogen load for each parcel was determined for the existing development using the factors presented in Table IV-2 and discussed above. A summary of potential additional nitrogen loading from build-out is presented as unattenuated and attenuated loads in Table IV-3.

IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT

IV.2.1 Background and Purpose

Modeling and predicting changes in coastal embayment nitrogen related water quality is based, in part, on determination of the inputs of nitrogen from the surrounding contributing land or watershed. This watershed nitrogen input parameter is the primary term used to relate present and future loads (build-out or sewering analysis) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the sub-watersheds of each sub-embayment of the 5 embayment systems under study was based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1). If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watershed in which nitrogen transport is through groundwater in sandy outwash aquifers. The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, nitrogen passes through a surface water ecosystem on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) is diminished by natural biological processes which represent removal (not just temporary storage). However, this natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes within the Town of Chatham.

Failure to determine the attenuation of watershed derived nitrogen overestimates the nitrogen load to receiving waters. If nitrogen attenuation is significant in one portion of a watershed and insignificant in another the result is that nitrogen management would likely be more effective in achieving water quality improvements if focused on the watershed region having unattenuated nitrogen transport (other factors being equal). An example of the significance of nitrogen attenuation relating to embayment nitrogen management was seen in West Falmouth Harbor (Falmouth, MA), where ~40% of the nitrogen discharge to the Harbor originating from the groundwater discharge from the WWTF was attenuated by a small salt

Sub-Embayment	ID#
Mill Pond (Fresh)	1
Goose Pond	2
Trout Pond	3
Schoolhouse Pond	4
Stillwater Pond	5
Lovers Lake	6
Emery Pond	7
White Pond	8
Newty Pond	9
Bassing Pond	10
Lower Muddy Creek	11
Upper Muddy Creek	12
Mill Creek	13
Taylors Pond	14
Cockle Cove	15
Bucks Creek	16
Sulfur Springs	17
Stage Harbor	18
Lower Oyster River	19
Oyster River	20
Oyster Pond	21
Mitchell River	22
Mill Pond	23
Little Mill Pond	24
Ryders Cove	25
Crows Pond	26
Bassing Harbor	27
Frostfish Creek	28
Upper Frostfish Creek	29

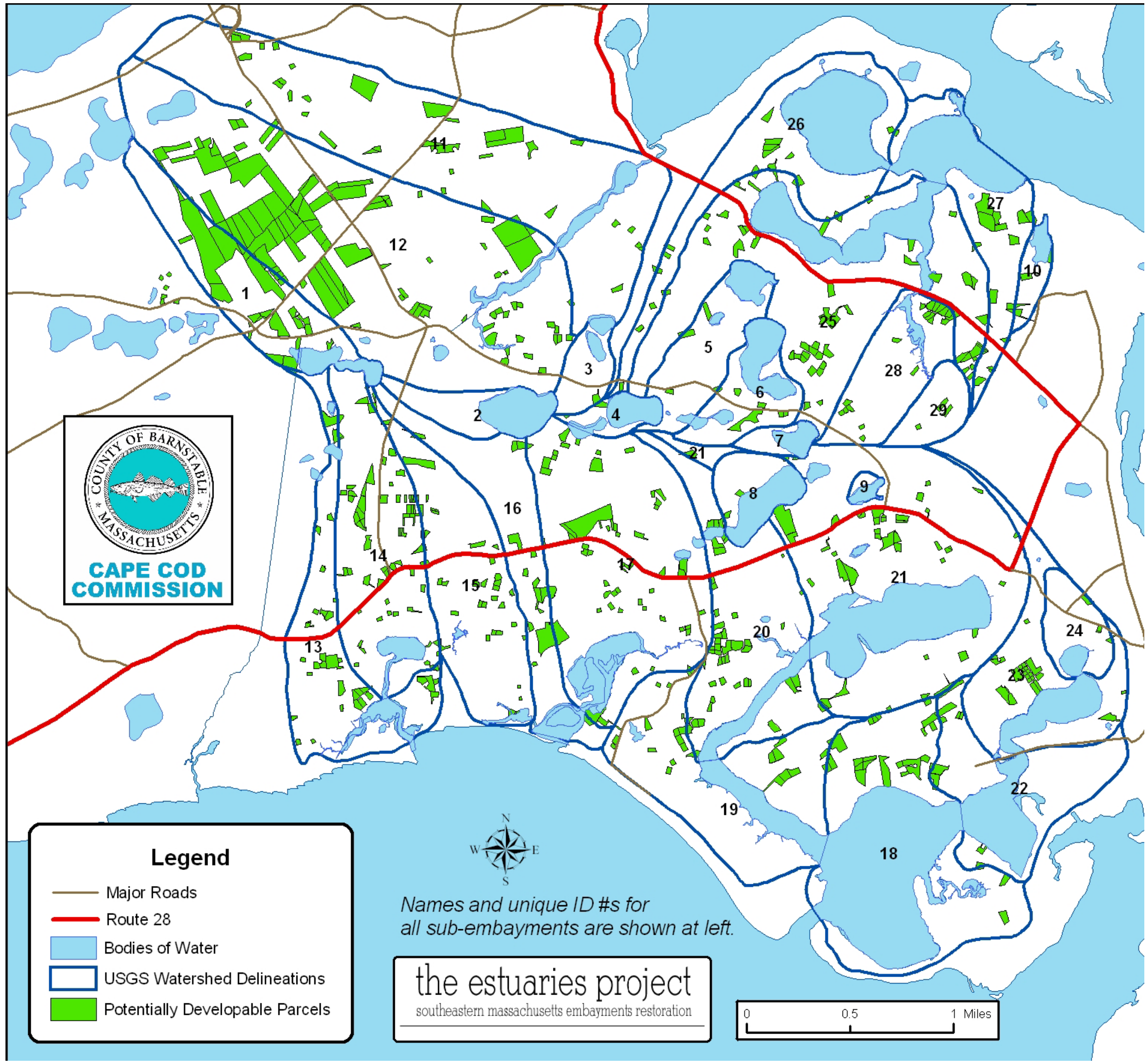


Figure IV-8. Distribution of present parcels which are potentially developable within the watersheds to the 5 embayment systems.

Sub-Embayment	ID#
Mill Pond (Fresh)	1
Goose Pond	2
Trout Pond	3
Schoolhouse Pond	4
Stillwater Pond	5
Lovers Lake	6
Emery Pond	7
White Pond	8
Newty Pond	9
Bassing Pond	10
Lower Muddy Creek	11
Upper Muddy Creek	12
Mill Creek	13
Taylors Pond	14
Cockle Cove	15
Bucks Creek	16
Sulfur Springs	17
Stage Harbor	18
Lower Oyster River	19
Oyster River	20
Oyster Pond	21
Mitchell River	22
Mill Pond	23
Little Mill Pond	24
Ryders Cove	25
Crows Pond	26
Bassing Harbor	27
Frostfish Creek	28
Upper Frostfish Creek	29

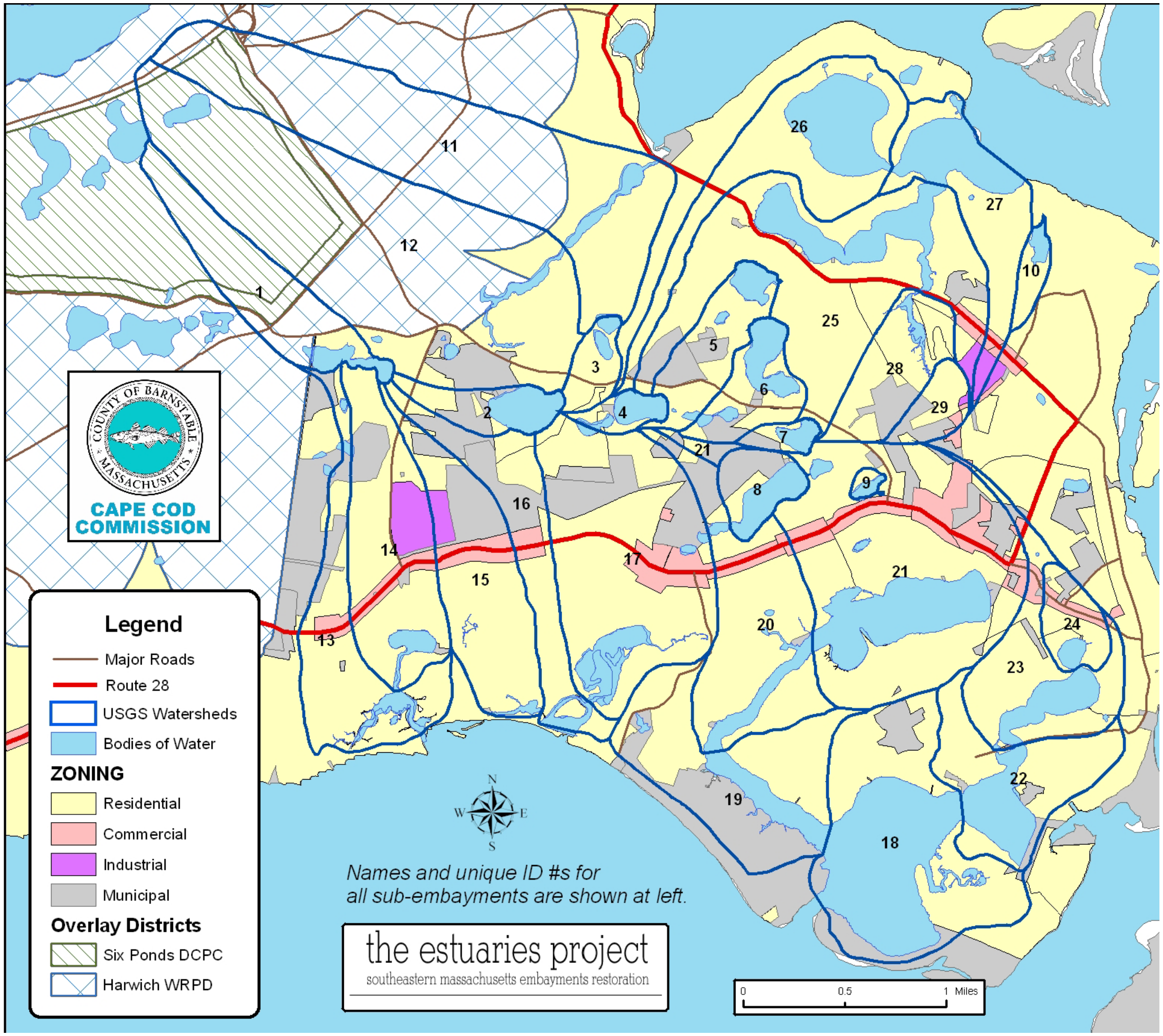


Figure IV-9. Spatial coverage of existing zoning and special overlay districts (DCPC, WRPD) in Harwich and Chatham, MA.

marsh prior to reaching Harbor waters. Proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

The input of nitrogen to Chatham's embayments from the surrounding watersheds is based upon knowing the land area contributing to a particular embayment, quantifying the land-uses, and calculating the nitrogen loading based upon regional measures of nitrogen loading for each land-use. Previous investigations by the Town of Chatham to determine the watershed nitrogen loads indicated that natural attenuation might be occurring in some sub-watersheds. This was based upon Cape Cod Commission watershed nitrogen loading for Chatham embayments presented in the Stearns & Wheler August 1999 Final Needs Assessment Report (updated for the Pleasant Bay embayments in a Memorandum of April 20, 2001). In a study by Applied Coastal Research and Engineering, Inc (2000), both direct observations (Stillwater Pond) and nitrogen modeling indicated that nitrogen attenuation was likely in the Cockle Cove sub-watershed and associated with the Bassing Harbor System.

In the previous watershed loading studies the watershed delineation's were made by the Cape Cod Commission by surveying watertable elevations in available wells. While this is a powerful approach, it is limited by the distribution of existing wells. A review of the watershed delineation's by the Project Team and Cape Cod Commission staff indicated that a revision of watershed and sub-watershed delineations would be necessary in order to accurately quantify watershed based nitrogen load and associated attenuations. Partnership with the United States Geological Survey has allowed for a complete revision of all of the delineations for the hydrologic features contained in Town of Chatham, including all of its coastal embayments. The USGS re-delineation effort is described above in Section III. Based on revised delineations a comprehensive analysis was conducted for nitrogen load determination based on watershed land-use (Section IV.1).

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, the MEP conducted multiple studies on natural attenuation relating to the 5 embayment systems in the study. Natural attenuation by fresh kettle ponds was addressed above. However, additional site-specific studies were conducted in each of the major pond and marsh systems which have significant streams (Lovers Lake and Stillwater Pond discharge to Ryder Cove) or tidal exchanges (Frost Fish Creek). In addition, a screening approach was applied within Stage Harbor, and Cockle Cove Systems (Section IV.2.4.).

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 net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). The development of a nitrogen attenuation term for freshwater transport through streams prior to discharge to marine waters was undertaken on two of the more significant surface water features in the Town of Chatham. Flow was measured at three different surface water locations (Figure IV-10) for their nitrogen loading and attenuation effects on Ryder Cove (creek between Lovers Lake and Stillwater Pond, creek between Stillwater Pond and Ryder Cove) and Bassing Harbor (Frost Fish Creek). Stage (water depth in the creek or stream) was monitored continuously for 16 months in the outflow streams from Lovers Lake to Stillwater Pond and Stillwater Pond to Ryder Cove (controlled outflow). Surface water flows in Frost Fish creek as well as nitrogen loading were measured and analyzed in order to refine the unique nitrogen

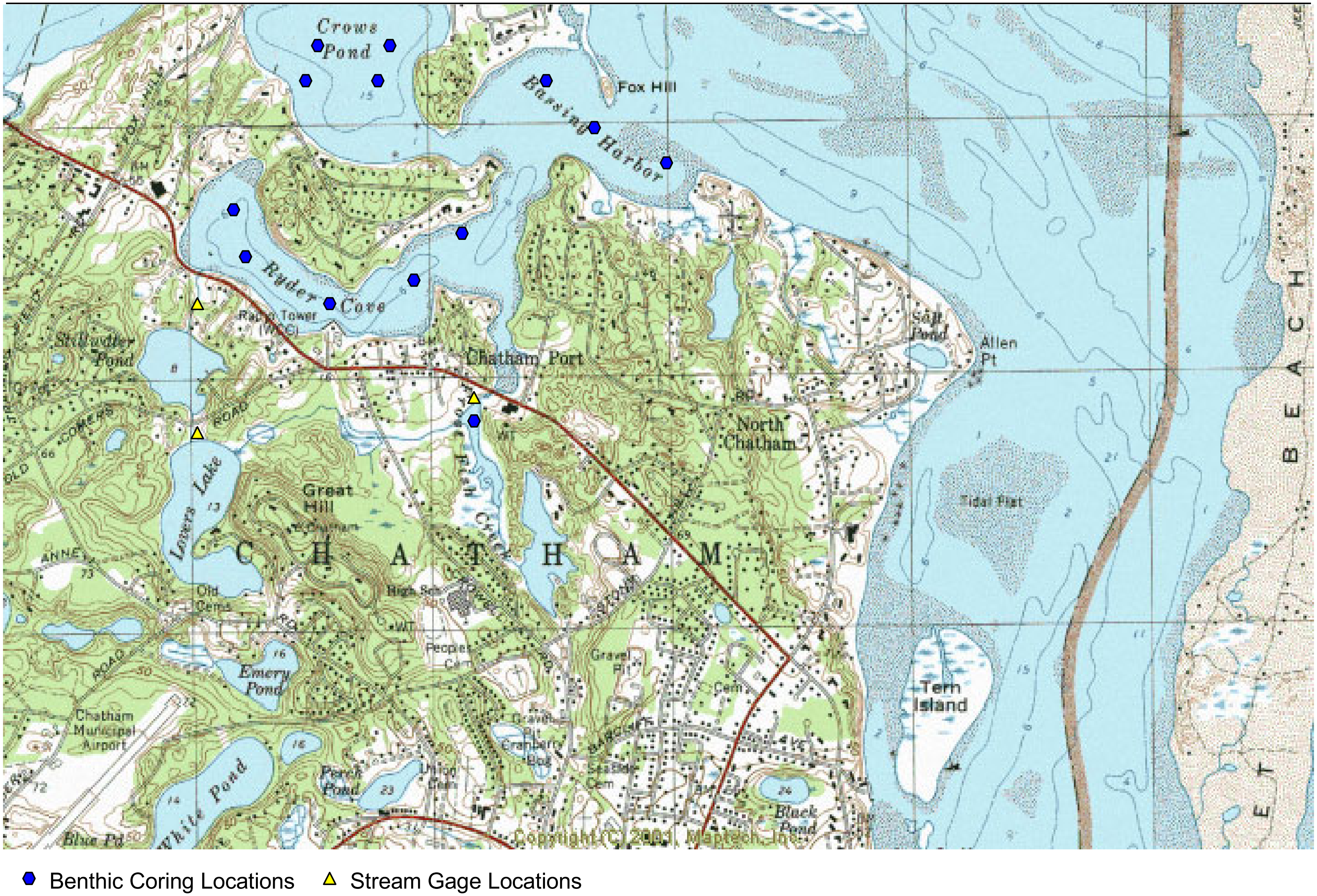


Figure IV-10. Location of Stream gages and benthic coring locations in the Ryder Cove / Bassing Harbor System.

attenuation capacities of this system discharging to Bassing Harbor. Analysis of nitrogen attenuation resulting from biological processes in Frost Fish Creek was based on four separate tidal flux studies performed in July, August, and September 2002.

The Ryder Cove watershed was targeted because it contains surface water bodies which are generally associated with nitrogen attenuation and which previous studies (Applied Coastal Research and Engineering, 2001) indicated are subject to attenuation. In addition, rerouted outflows from Lovers Lake previously to Frost Fish Creek and Stillwater Pond and now to only to Stillwater Pond might provide a potential nitrogen management “soft solution” for Ryder Cove. Surface water samples were collected about weekly by the Chatham Water Quality Laboratory (R. Duncanson) and assayed by the SMAST Coastal Systems Analytical Laboratory.

In addition to the surface water field study within the Ryder Cove watershed, samples of surface water were collected by the water quality monitoring program from a variety of watersheds in order to screen watersheds for significant nitrogen attenuation of the watershed loading estimates. The watershed nitrogen loading and freshwater discharge estimates in this attenuation study were those derived in Section IV.1.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Lovers Lake to Stillwater Pond to Ryder Cove

Lovers Lake and Stillwater Pond are 2 of the larger ponds within the study area and unlike many of the freshwater ponds, these have stream outflows rather than discharging solely to the aquifer on down-gradient shores. These stream outflows may serve to decrease their attenuation of nitrogen, but they also allow for a direct measurement of the nitrogen attenuation. Nitrogen attenuation was calculated in both Lovers Lake and Stillwater Pond from nitrogen loading rate estimates within respective watersheds and measured annual discharge of nitrogen through stream outflows of both ponds.

Stream gauging and nitrogen sampling stations were established within each of the two outflow streams, within the Ryder Cove sub-watershed. An upper station was placed at the discharge from Lovers Lake to Stillwater Pond and a lower station at the outlet of Stillwater Pond to Ryder Cove (Figure IV-10). The upper station was installed to evaluate results of the historical re-routing of discharge from Lovers Lake to Frost Fish Creek, as opposed to present discharge to Stillwater Pond. The lower station was to evaluate the surface water flow and nitrogen load to Ryder Cove from the sub-watersheds to Stillwater Pond + Lovers Lake + a portion of Schoolhouse Pond.

At each sampling site, a continuously recording vented water level gauge was installed and calibrated to yield the level of water in the discharge culvert that carries the flows and associated nitrogen load under roadways. Flow was periodically measured using a Marsh-McBirney electromagnetic flow meter. Periodic (~ weekly) water samples were collected for nitrogen analysis. These measurements allowed for the determination of both total volumetric discharge and nitrogen mass discharge to down-gradient systems. In addition, a water balance was constructed based upon the groundwater flow model to determine freshwater discharge expected at each gauge site. Comparison of measured and predicted discharge is used to confirm that the stream is capturing the entire recharge to its up-gradient contributing area. This comparison also can be used to indicate if pond outflow is through a combination of stream and groundwater outflow. This freshwater balance is necessary to support the attenuation calculations.

The gauges were installed on November 8, 2000 and were set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Due to multiple instrument failures during the period May 2001 to February, 2002, meaningful data was not collected. As a result, the field deployment period for the stream gaging was extended to include the summer 2002 field season. Water samples were collected approximately biweekly with an increase in sampling frequency to weekly during critical summer periods.

The stream gauge records available for this analysis of freshwater stream flow and associated attenuated nitrogen load covers a period of 361 days for the discharge to Ryder Cove and 470 days for the discharge from Lovers Lake to Stillwater Pond. The Ryder Cove gauge was damaged at 111 days and replaced to continue the long term recording of stage. Using the available flow measurements a composite year for each site was constructed from which annual and average daily freshwater flow from Lovers Lake to Stillwater Pond and from Stillwater Pond into Ryder Cove were determined (Figures IV-11, IV-12, Table IV-5). Both stream flow records show a similar seasonal pattern of high flow in spring and lowest flow during summer. This seasonal pattern reflects the annual variation of groundwater levels (Section IV.1), which is a major driver to streamflow in this hydrological setting. The nitrogen concentration measurements indicate the opposite pattern with higher levels in summer.

Total nitrogen concentrations within both streams outflows were relatively high, with Stillwater Pond outflow (0.851 mg N L^{-1}) higher than Lovers Lake outflow (0.732 mg N L^{-1}). This likely represents the higher nitrogen loading to Stillwater Pond (2465 g N d^{-1}) compared to Lovers Lake (1693 g N d^{-1}). In both streams, organic nitrogen forms dominated the total nitrogen pool, indicating that groundwater nitrogen (presumably dominated by nitrate) entering the ponds is taken up by plants within the pond system prior to export to the streams. However, nitrate was still a major fraction of the total nitrogen pool being 17% and 31% of the Lovers Lake and Stillwater Pond outflow nitrogen pools, respectively. The high concentration of inorganic nitrogen in the outflowing stream waters suggest that plant production within these ponds is not nitrogen limited. In the case of Stillwater Pond outflow water, the average nitrate concentration was >0.25 mg N L-1, representing a source of readily available nitrogen for stimulation of phytoplankton production within the receiving waters of Ryder Cove.

Annual flow measured within the Lovers Lake to Stillwater Pond stream agreed well (91%) with the predicted groundwater inflow to Lovers Lake from its watershed (Table IV-5). The slightly lower measured discharge likely results from the lower than average groundwater levels during the study period (Figure IV-4). From these data it appears that Lovers Lake discharges primarily through this stream. Therefore, the much lower nitrogen load (812 g N d^{-1}) discharged from Lovers Lake in this stream outflow relative to the nitrogen mass entering the Lake from its watershed (1693 g N d^{-1}) should be a direct measure of nitrogen attenuation by the pond ecosystem. Therefore, rate of natural attenuation of nitrogen moving through Lovers Lake is 52%, within the 39%-95% range determined from the pond survey method (see above) and consistent with use of a 40% attenuation factor for the survey ponds.

It should be noted that the discharge from Lovers Lake to Stillwater Pond, being the sole surface water drain for Lovers Lake, is a relatively recent phenomenon. The historic discharge from Lovers Lake was to both Stillwater Pond and to Frost Fish Creek (note 1943 USGS Topographic Map). However, one of the outflows, from Lovers Lake to Frostfish Creek, was discontinued since the 1980's (Duncanson, personal communication). This shift in outflow from Lovers Lake, increased the freshwater flow through and therefore decreased the residence time of water within Stillwater Pond (although the extent is currently unknown). This decreased residence time in Stillwater Pond, likely reduces the level of nitrogen attenuation. The effects of restoring the historic dual flow paths on distribution and total load to upper and lower Ryders Cove and the potential for increased nitrogen removal in passage through Stillwater Pond and Frost Fish Creek should be considered by the Town as it develops nitrogen management alternatives for the Bassing Harbor System. In this evaluation, it should be considered that outflow from Lovers Lake could be seasonally shifted between Stillwater Pond and Frost Fish Creek to maximize natural attenuation to "relocate" the site of nitrogen input to the estuary, while still providing for herring migration. While any such analysis must take into account existing aquatic uses of the fresh and saltwater systems being modified, it should be noted that the Frost Fish Creek system is primarily salt marsh with a relatively high salinity and that the flow change is not expected to shift this saltwater system significantly.

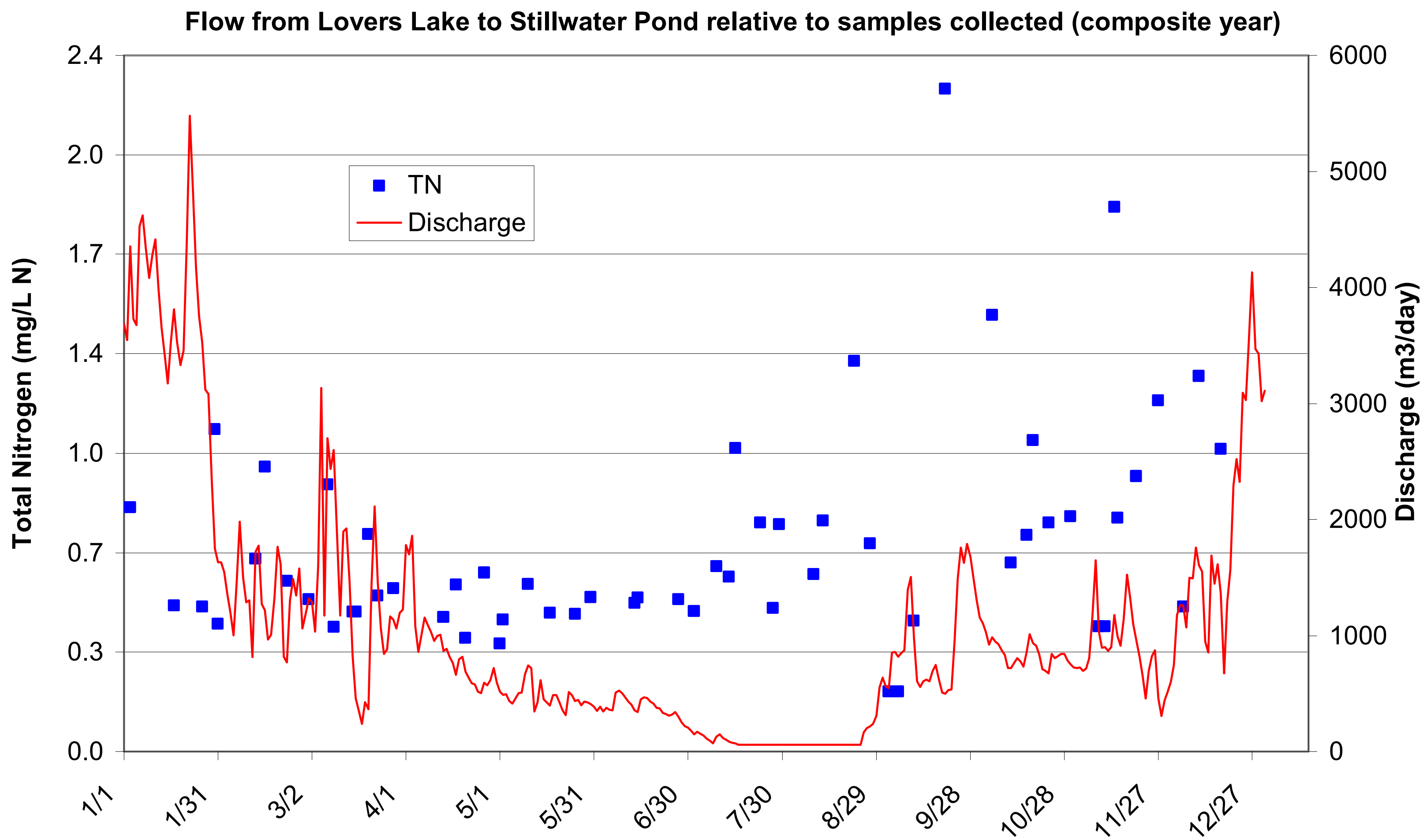


Figure IV-11. Annual composite developed from a stream gauge maintained in the outflow stream from Lovers Lake discharging to Stillwater Pond. Nutrient samples were collected approximately weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow and total nitrogen transport for determining nitrogen attenuation (see Table IV-5).

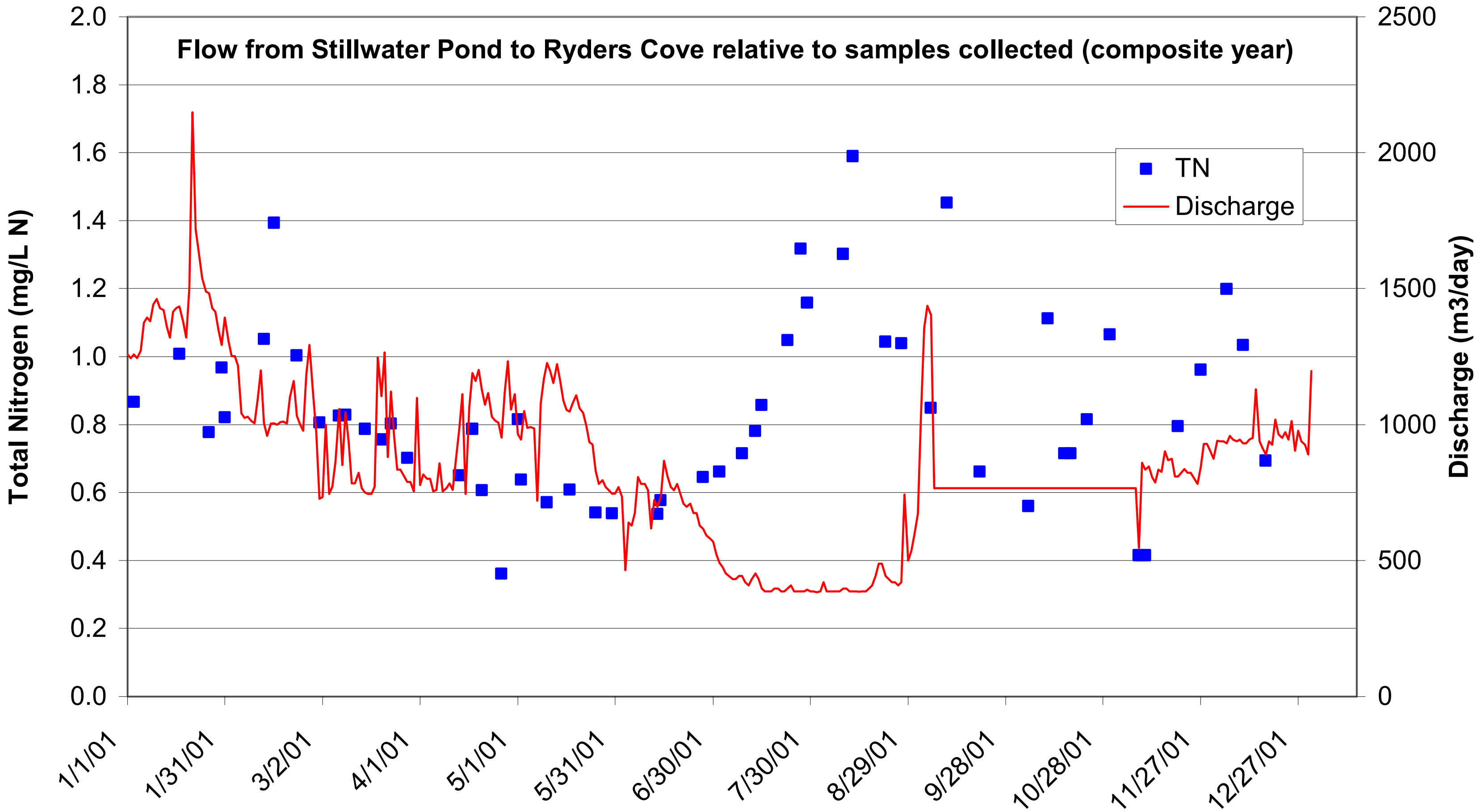


Figure IV-12. Annual composite developed from a stream gauge maintained in the outflow stream from Stillwater Pond discharging to Ryders Cove. Nutrient samples were collected approximately weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow and total nitrogen transport for determining nitrogen attenuation (see Table IV-5).

In contrast to Lovers Lake, the annual flow measured at the stream outflow from Stillwater Pond suggested that only a portion of the groundwater (and nitrogen) inflows from the watershed and Lovers Lake were exiting via the stream (34%). In fact less water was outflowing via Stillwater Pond stream (853 $m^3 d^{-1}$) than entering from Lovers Lake (1079 $m^3 d^{-1}$). In previous preliminary investigation at this site, there was concern that the lower than predicted flows from Stillwater Pond might result from an underestimate of the watershed area (Applied Coastal 2000). This does not appear to be the cause in the present case (even the Lovers Lake inflow is greater than Stillwater outflow). The most likely explanation for this observed water imbalance is that the elevation of the outflow weir from Stillwater Pond results in pond water outflow to the aquifer on the down-gradient shore, as in kettle ponds without stream outflows. In this case it is still possible to estimate nitrogen attenuation by Stillwater Pond. By correcting the nitrogen outflow relative to the proportion leaving via the stream and assuming that the outflowing groundwater has the same nitrogen concentration as the streamwater (conservative estimate), the total mass leaving the pond can be determined. This total discharging nitrogen mass when compared to the predicted watershed nitrogen inflow yields an attenuation of 14% for Stillwater Pond. If it is further assumed that lower groundwater levels are causing lower flows and the ratio from Lovers Lake (0.91) is used to adjust the predicted flow rate, then the calculated attenuation factor rises to 23%. These relatively low nitrogen attenuation rates may result from the relatively high nitrogen load to this system which enters from Lovers Lake, Schoolhouse Pond watershed and the adjacent Stillwater Pond watershed. The high nitrate levels in the outflowing water appear to support a lower attenuation rate for this pond. Given the uncertainties due to the hydrologic balance, the attenuation rate for this system should be considered to be a minimum.

IV.2.3 Freshwater Discharge and Attenuation of Watershed Nitrogen: Frost Fish Creek

Frost Fish Creek (above the Rt. 28 culverts) is a tidal basin with fringing salt marsh (see also Section V for hydrodynamics). Given its tidal flow, continuous stream gauging could not be conducted in the Frost Fish Creek discharge to the Bassing Harbor system. Instead, intensive discrete tidal flux analyses were conducted on four separate occasions (Summer 2002) in order to quantify freshwater inflow to Frost Fish Creek and nitrogen attenuation by this tributary system to Bassing Harbor.

Freshwater and tidal flows were measured over complete tidal cycles. Direct flow measurements were made at the weir near the mouth of Frost Fish Creek (Figure IV-10) combined with high frequency (hourly during ebb and flood, every half hour around the turn of each tide) water quality sampling for nutrients. The combination of both records allowed for the calculation of nitrogen load into and out of the embayment for each of the four tidal periods analyzed in July (1), August (2), and September (1) of 2002. Comparison of measured nitrogen loads resulting from the freshwater fraction of the Frost Fish Creek flow enabled the calculation of a nitrogen attenuation term applicable to the calculated watershed based nitrogen loads for the Frost Fish Creek sub-watershed.

Each of the tidal flux studies performed on Frost Fish Creek were completed over a complete tidal cycle, beginning approximately one hour prior to low tide and continuing through the high tide, ending approximately one hour past the time of the following low tide. The tidal flux studies were conducted with at least two days of no precipitation such that flow measurements, water quality sampling and subsequent nitrogen loading calculations would not be biased by storm related flows.

Table IV-5. Comparison of water flow and nitrogen discharges to Ryder Cove and from School House Pond, Lovers Lake and Stillwater Pond watershed through Stillwater Pond Stream. The "Stream" data is from previous SMAST studies with the Town of Chatham and the MEP stream gauging effort. Watershed data is based upon the MEP watershed modeling effort by USGS.

Stream Discharge Parameter	Stream flow to Ryder Cove	Steam flow into Stillwater Pond	Data Source
Total Days of Record [a]	361	470	(1)
Flow Characteristics:			
Stream Average Discharge (m3/d)	853	1079	(1)
Contributing Area Average Discharge (m3/d)	2488[b]	1185[c]	(2)
Proportion Discharge Stream vs. Contributing Area (%)	34%	91%	
Nitrogen Characteristics:			
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.263	0.127	(1)
Stream Average Total N Concentration (mg N/L)	0.851	0.732	(1)
Nitrate + Nitrite as Percent of Total N (%)	31%	17%	
Stream Average Nitrate + Nitrite Discharge (g/d)	207	192	(1)
Stream Average Total Nitrogen Discharge (g/d)	717	812	(1)
Contributing Area Average Total Nitrogen Discharge (g/d)	2465	1693	(2)
Proportion Total Nitrogen Stream vs. Contributing Area (%)	N/A	48%	
Attenuation (Total) of Nitrogen in Pond/Stream (%)	14%*	52%	

[a] from 11/8/00 to September 2002 (Ryder gage) and December 2002 (Stillwater Pond gage)
[b] flow and N load to Stillwater Pond include Lovers Lake Contributing Area, with correction for low flow using Lovers Lake Outflow %
[c] flow and N load to Lovers Lake represent only the Lovers Lake Contributing Area
* attenuation based upon expected nitrogen in measured volume discharge.
N/A = data not available
(1) MEP data, collected Amendment to present study
(2) Calculated from MEP watershed delineations to School House Pond, Lovers Lake and Stillwater Pond; the fractional flow path from each sub-watershed which contribute to Stillwater Stream Flow; and the annual recharge rate.

All four of the Frost Fish Creek tidal flux studies were conducted at the weir/culvert just up-gradient of Route 28 in Chatham. This culvert separates the main body of Frost Fish Creek from a small impoundment that receives Frost Fish Creek flows prior to final discharge to the Bassing Harbor embayment. Ebb and flood tide velocities were all measured at the same end of the culvert and generally taken concurrently with the water quality samples. In the instances when velocities were obtained at slightly different times than the water quality sample taken, a linear interpolation was utilized to match a flood or ebb tide velocity with the appropriate time of the water quality sample. Completing the linear interpolation on velocity for the complete tidal period yield a detailed record of flow out and in (ebb/flood) that related directly to changes in tidal stage (Figures IV-13A-D) The tidal flux volume results for Frost Fish Creek served the dual purpose of being a means to quantify attenuation of watershed based nitrogen loading to Frost Fish Creek as well as cross check for the RMA-2 hydrodynamic model. With the exception of the tidal study conducted on July 21, 2002, modeled and measured tidal flux volumes differed by only 2 and 6 percent.

As described above, each nutrient water quality sample was paired with a flow rate such that nitrogen and other constituent fluxes in Frost Fish Creek could be calculated for each of the tidal cycles studied. Tidal volume for each study was determined over the period from ebb slack to flood slack tide (Flood) and from flood slack to ebb slack (Ebb). In cases where tidal asymmetry resulted in a change in the water volume stored within the Frost Fish Creek basin (during a tidal cycle), the appropriate flood or ebb interval (time) was adjusted to ensure a zero change in storage volume within the basin, by keeping the measured tidal elevation at the end of a study equal to that at the start. Net tidal flux volume for the system was then determined by the difference in total volume inflow versus outflow over a tidal cycle, positive (+) indicating a net inflow into the system on the flood versus a negative (-) a net discharge from the system (Table IV-6). Determining freshwater inflow to a basin from differences in inflow/outflow at the tidal inlet is an acceptable approach in cases like Frost Fish Creek, where changes in storage can be controlled and where the freshwater outflow is a large fraction of the total outflow volume (Millham and Howes 1994). In the present study, freshwater outflow represented about one-third of the total ebb tide volume, a very large proportion compared to the larger estuarine systems of Chatham.

The measurements of freshwater discharge to Frost Fish Creek from its watershed ranged from 1258 m^3d^{-1} to 900 m^3d^{-1}, with an average (1097 m^3d^{-1}) close to that predicted (1274 m^3d^{-1}) from the groundwater flow model (Section III). Given that measurements were conducted during the summer period when flows are lower than the annual average, the measured and modeled freshwater flows are in excellent agreement. This agreement supports a straight-forward determination of nitrogen attenuation for this system.

Nitrogen mass on each inflowing and outgoing tide was calculated from the tidal sampling data by integrating over the flood and ebb tides. A net nitrogen outflow from Frost Fish Creek to lower Ryder Cove was observed in each event (Table IV-6). In fact, Frost Fish Creek was a net exporter of each of the major nitrogen related water quality constituents assayed. These exports result from the inflow and biological transformation of watershed derived nitrogen in Frost Fish Creek. Nitrogen attenuation was determined as the difference between the predicted watershed nitrogen input (Section IV.1) and the observed net loss of nitrogen to lower Ryder Cove. Comparing the observed mean net nitrogen tidal export of 1.82 kg N $tide^{-1}$ and the predicted watershed nitrogen load of 2.24 kg N $tide^{-1}$, natural attenuation of watershed derived nitrogen within Frost Fish Creek is 19%. This is a lower attenuation rate than the 40% observed in the Mashapaquit Creek Marsh in the West Falmouth Harbor System (Howes and Smith 1999). However, the Frost Fish Creek basin results in a dilution of inflowing groundwater nitrogen which can reduce the rate of denitrification of externally derived nitrate. In Mashapaquit Creek, groundwater flow during ebb tide was directly over creekbottom sediments, enhancing nitrogen removal by denitrification. The lower rate in Frost Fish Creek compared to Mashapaquit Creek is consistent with differences in factors related to denitrification in the 2 systems. In summary, the mass of nitrogen entering lower Ryder Cove from Frost Fish Creek is approximately 19 percent lower than the nitrogen load calculated from the sub-watershed land use analysis (which have been adjusted accordingly for development of management alternatives).

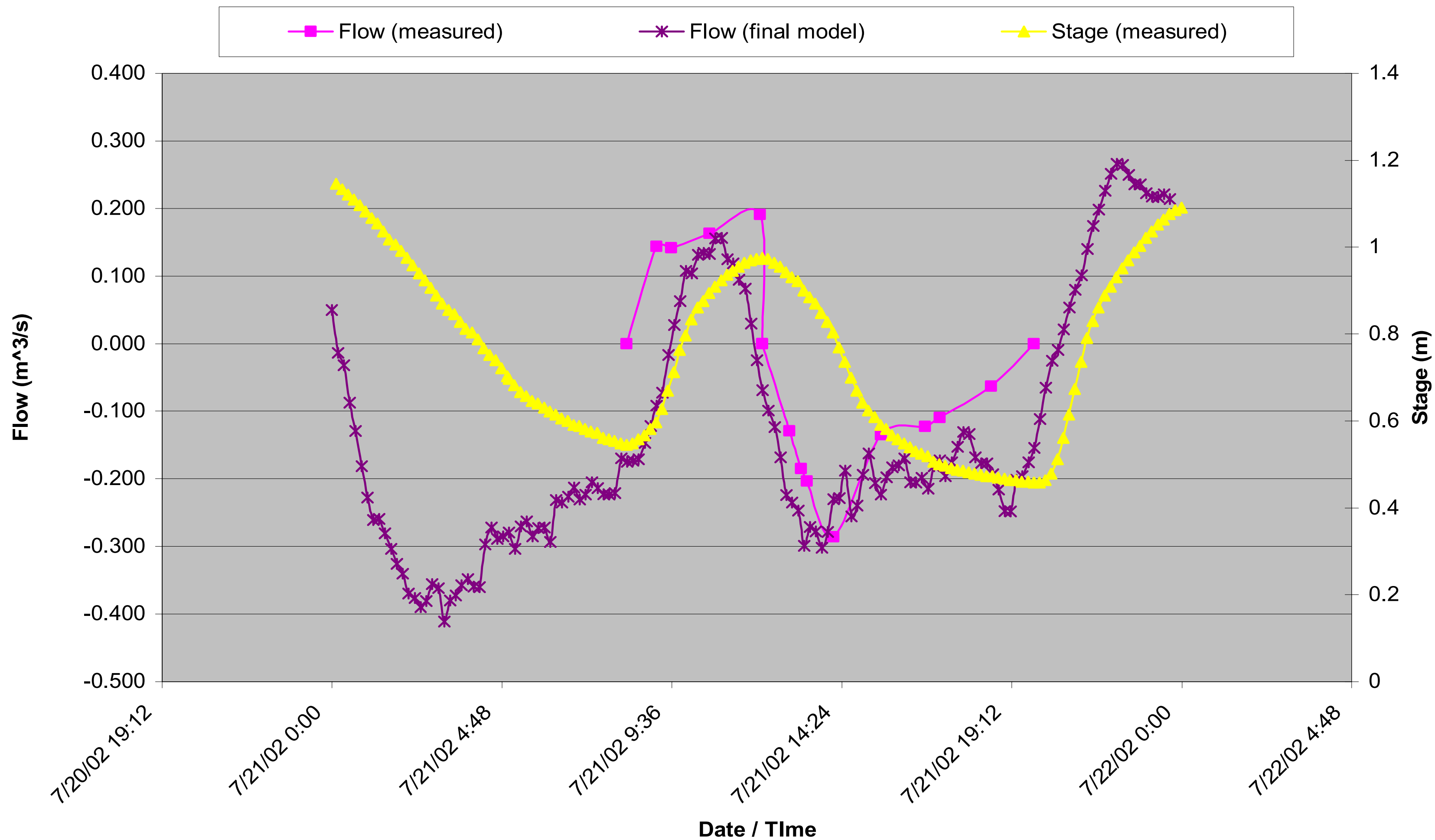


Figure IV-13a. Frost Fish Creek Tidal Study 1 (July 21, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

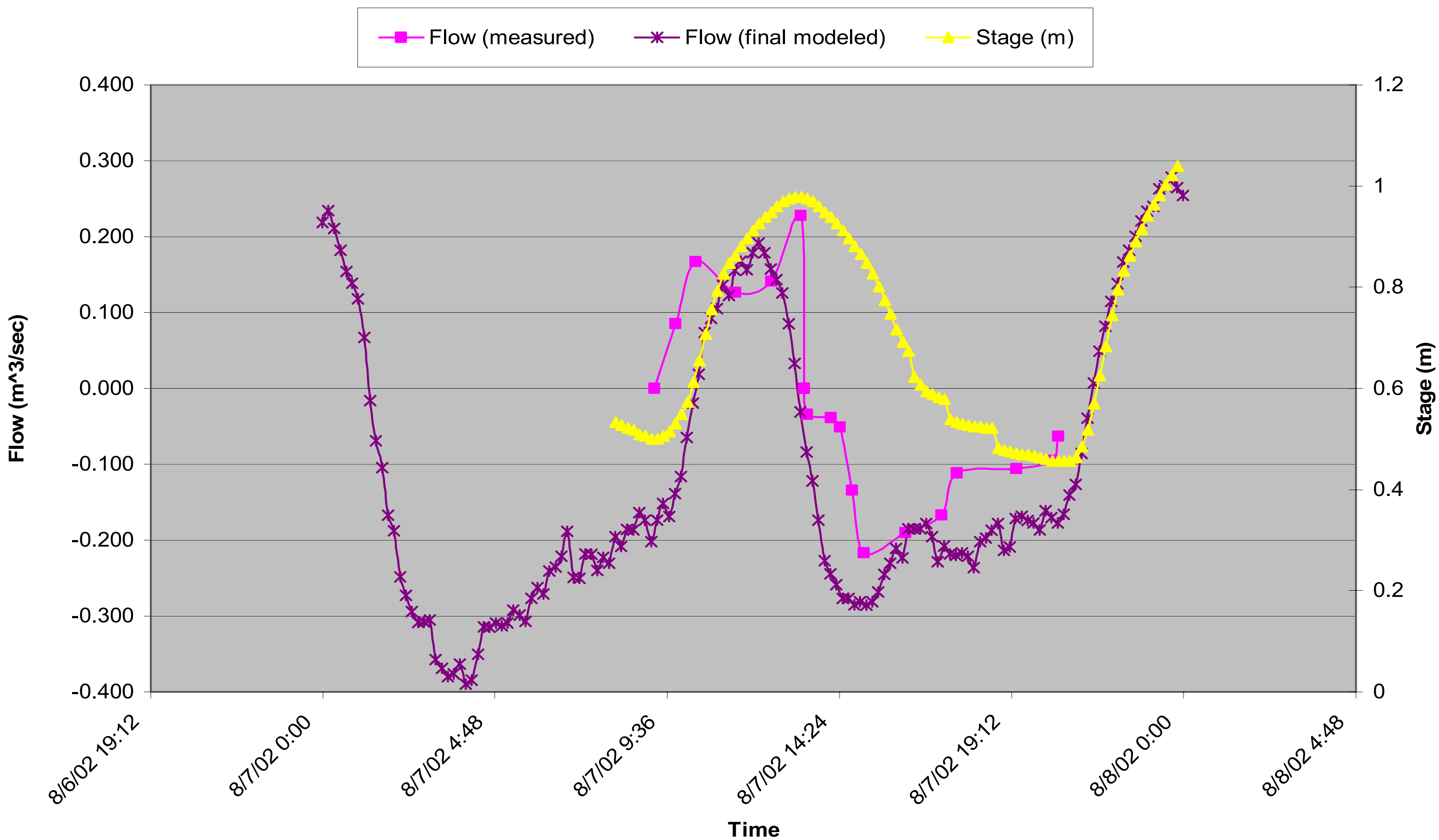


Figure IV-13b. Frost Fish Creek Tidal Study 2 (August 8, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

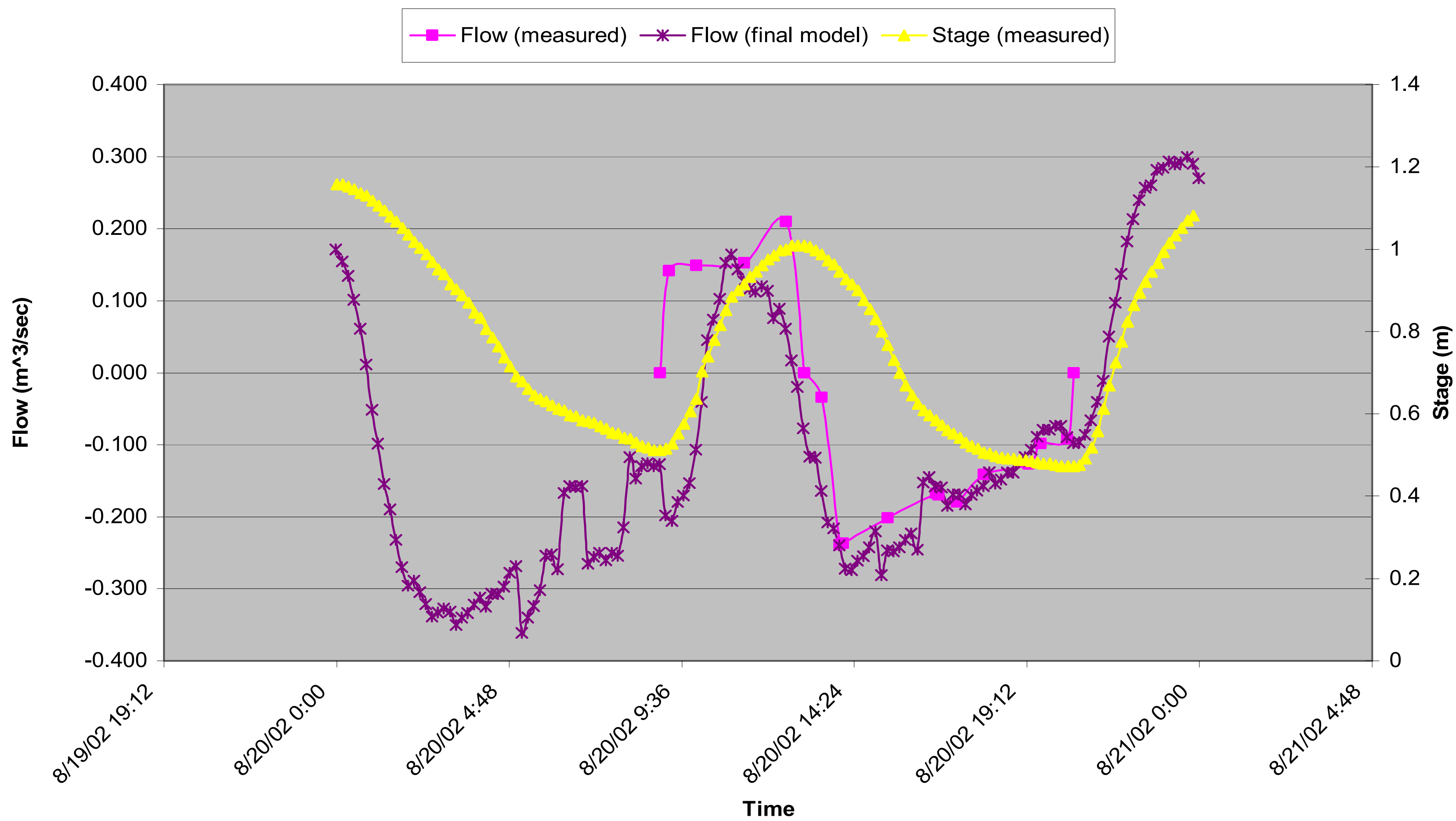


Figure IV-13c. Frost Fish Creek Tidal Study 3 (August 20, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

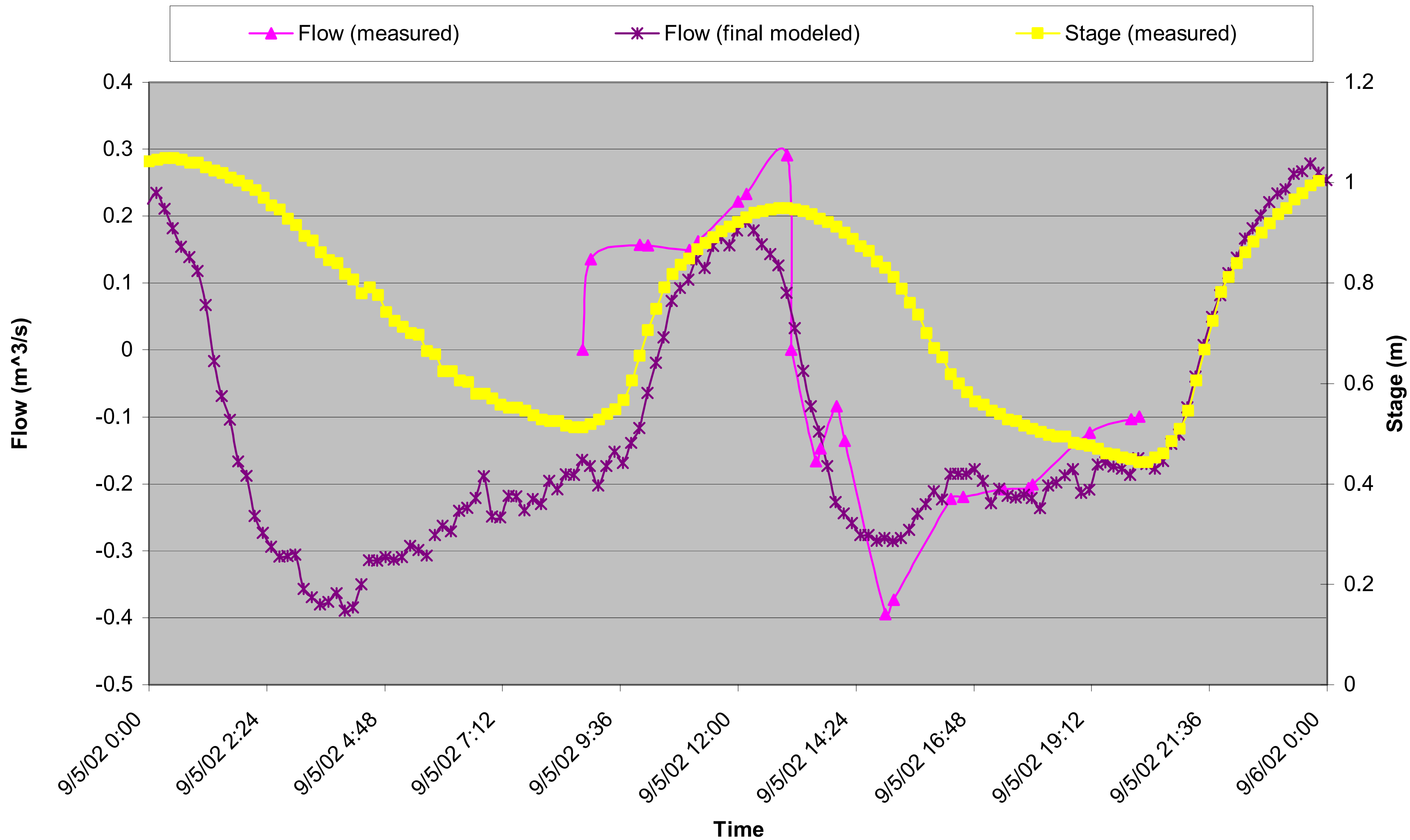


Figure IV-13d. Frost Fish Creek Tidal Study 4 (September 5, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

Table IV-6. Measurement of nitrogen attenuation, flow and water quality constituents within Frost Fish Creek during summer 2002. The total freshwater discharge to Frost Fish Creek from the watershed as determined from the USGS groundwater model (Section III) was 1274 m^3 per day based upon the annual average, compared to the 1097 m^3 per day determined by the RMA-2 model (Section V) and the 1054 m^3 per day from the 4 Tidal Studies. Nitrogen attenuation is calculated as the difference in measured nitrogen mass in tidal outflow from Frost Fish Creek to Ryder Cove versus the nitrogen load entering from the watershed and within the inflowing tidal waters.

Study/Date	Tide	Tidal Flux RMA-2 Modeled m^3/day	Tidal Flux Measured m^3/day	NOX Kg N per tide	Total N kg N per tide	TON Kg N per tide	POC Kg C per tide	DIN Kg N per tide	Pigment g Pig per tide
Study 1	**Flood (+)**		1952	0.03	1.67	1.51	3.88	0.16	20.8
July 21, 2002	**Ebb (-)**		-2903	-0.88	-3.47	-2.51	-5.68	-0.96	-176.7
	Net Flux	**-1258**	**-951**	**-0.85**	**-1.81**	**-1.00**	**-1.80**	**-0.80**	**-155.9**
Study 2	**Flood (+)**		1999	0.04	2.57	2.47	7.88	0.10	44.1
August 7, 2002	**Ebb (-)**		-3222	-0.26	-5.16	-4.85	-15.93	-0.31	-92.7
	Net Flux	**-1155**	**-1223**	**-0.22**	**-2.59**	**-2.38**	**-8.05**	**-0.21**	**-48.6**
Study 3	**Flood (+)**		2128	0.04	1.97	1.88	6.46	0.09	52.6
August 20, 2002	**Ebb (-)**		-3019	-0.02	-2.99	-2.92	-10.32	-0.07	-93.0
	Net Flux	**-900**	**-891**	**0.02**	**-1.02**	**-1.04**	**-3.86**	**0.02**	**-40.4**
Study 4	**Flood (+)**		2756	0.18	5.86	3.39	6.64	2.47	45.1
September 5, 2002	**Ebb (-)**		-3906	-0.51	-7.71	-5.18	-11.97	-2.53	-105.9
	Net Flux	**-1075**	**-1150**	**-0.33**	**-1.85**	**-1.79**	**-5.34**	**-0.06**	**-60.7**
Mean Flux (N kg/tide)		**-1097**	**-1054**	**-0.34**	**-1.82**	**-1.55**	**-4.76**	**-0.26**	**-76.4**
S.E. (N kg/tide)		**75**	**79**	**0.18**	**0.32**	**0.33**	**1.32**	**0.19**	**26.8**
CV%		**-7%**	**-8%**	**-53%**	**-18%**	**-21%**	**-28%**	**-70%**	**-35%**
Total Land + Atmos. Inputs N kg/tide					**2.24**				
Attenuation (calculated)					**19%**				

IV.2.4 Confirmation of Watershed Nitrogen Discharge: Town-wide.

The third approach employed for evaluation of watershed nitrogen attenuation was to examine the nitrogen levels in the small or intermittent surface water discharges to the Town's embayments. The data were collected by the Chatham Water Quality Laboratory at the sites shown in Figure IV-3. Water samples were collected primarily during the summer months from flowing surface waters. Surface flows that were tidal, brackish, and exhibited dilution of nitrogen by salt water required a correction of the data. The dilution by salt water was accounted for based upon the mean concentration of salt and total nitrogen within the water column of the adjacent embayment region. The embayment data was from the water quality monitoring database. This allowed for a site-specific correction and increased the accuracy of the analysis.

The surface water flows are fed by groundwater formed within the watersheds to the embayment's, and therefore, reflect the nitrogen levels in groundwater from a portion of an embayments watershed. These measured nitrogen levels can be compared to the nitrogen levels in freshwater discharging to the Town's embayments. This analysis is a diagnostic tool only.

Nitrogen levels in discharging waters in small streams can be lower than predicted from watershed analysis due to less loading to their contributing area, as opposed to the overall embayment watershed for which land-use nitrogen loading data is provided. The larger the watershed is to the stream, the more representative the comparison and results. Nitrogen levels can also be lower due to attenuation of nitrogen during transport.

The results of this screening indicated that the predicted and observed nitrogen concentrations for various watershed regions compared well for the Stage Harbor System. The results are relatively consistent for Oyster Pond, 2.75 mg N/L (predicted) versus 1.6 - 3.1 mg N/L observed. A similar result was observed from site CM-A in Stage Harbor where the predicted and observed total nitrogen values were 1.95 and 1.40 mg N/L, respectively. These results are consistent with the absence of major upland ponds and lakes within the watershed to the Stage Harbor System.

The apparent nitrogen attenuation within the Cockle Cove Creek system relative to predicted watershed nitrogen levels is likely due in part to stimulation of denitrification within this system. Measurements of nitrate uptake in Cockle Cove Creek made as part of the Sediment Nitrogen Regeneration Study (see below) indicated a large uptake by the Creek sediments. Additional data collection would have to be conducted in order to quantitatively determine the mass of nitrogen removed from the Creek System prior to discharge to Buck Creek. However, nitrogen attenuation by the tidal creek sediments is clearly demonstrated. Additional evaluation of Cockle Cove is relevant only to the nitrogen loading to the Bucks Creek System and macrophyte issues within the near shore region (Harding Beach area).

It appears that those embayment watersheds within Chatham that have significant surface water flows and water bodies have significant amounts of the watershed nitrogen load removed prior to discharge to the adjacent embayments.

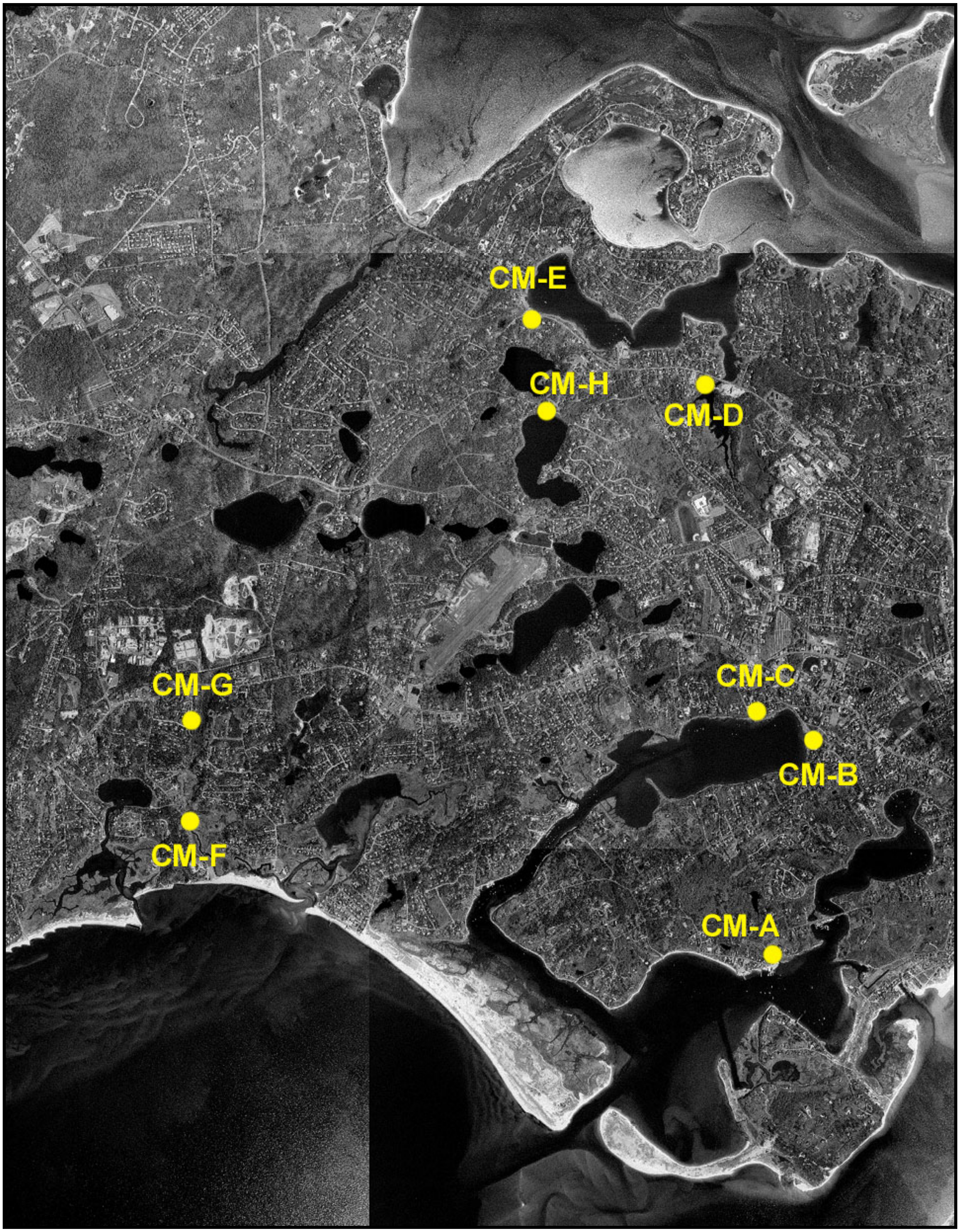


Figure IV-14. Map of freshwater discharge water quality monitoring stations. CM-E & H are the outflow from Stillwater Pond and Lovers Lake, respectively.

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

The overall objective of the Benthic Nutrient Flux Task was to quantify the summertime exchange of nitrogen, between the sediments and overlying waters within each of the 5 embayments in Chatham. The mass exchange of nitrogen between watercolumn and sediments is a fundamental factor in controlling nitrogen levels within coastal waters. These fluxes and their associated biogeochemical pools relate directly to carbon, nutrient and oxygen dynamics and the nutrient related ecological health of these shallow marine ecosystems. In addition, these data are required for the proper modeling of nitrogen in shallow aquatic systems, both fresh and salt water.

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

As stated in above sections, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the embayments of Chatham predominantly in highly bioavailable forms from the surrounding upland watershed and in flooding tidal waters. If all of the nitrogen remained within the watercolumn (once it entered), then predicting watercolumn nitrogen levels would be simply a matter of determining the watershed loads, dispersion, and hydrodynamic flushing. However, as nitrogen enters the embayments from the surrounding watersheds it is predominantly in the bioavailable form nitrate. This nitrate and other bioavailable forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton “particles”. Most of these “particles” remain in the watercolumn for sufficient time to be flushed out to a downgradient larger waterbody (like Pleasant Bay or Nantucket Sound). However, some of these phytoplankton particles are grazed by zooplankton or filtered from the water by shellfish and other benthic animals. Also, in longer residence time systems (greater than 8 days) these nitrogen rich particles may die and settle to the bottom. In both cases (grazing or senescence), a fraction of the phytoplankton with their associated nitrogen “load” become incorporated into the surficial sediments of the bays.

In general the fraction of the phytoplankton population which enters the surficial sediments of a shallow embayment: (1) increases with decreased hydrodynamic flushing, (2) increases in low velocity settings, (3) increases within small basins (e.g. Mill Pond, Taylors Pond, etc). To some extent, the settling characteristics can be evaluated by observation of the grain-size and organic content of sediments within an estuary.

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both in oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content, that bioavailable nitrogen is returned to the embayment watercolumn for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that we have investigated, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. Failure to account for this recycled nitrogen generally results in significant errors in determination of threshold nitrogen loadings. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

IV.3.2 Method for determining sediment-watercolumn nitrogen exchange

For the 5 Chatham embayments 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 from 46 sites (Figure IV-15) were collected in late July 2000, with additional sampling of the Bassing Harbor Systems in 2001. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium and ortho-phosphate were made in time-series on each incubated core sample. As part of a separate research investigation, the rate of oxygen uptake was also determined and measurements of sediment bulk density, organic nitrogen, and carbon content were made.

Rates of nutrient release (and oxygen uptake) were made using undisturbed sediment cores incubated for 24-36 hours in temperature controlled baths. Sediment cores (15 cm inside diameter) were collected by SCUBA divers and cores transported by a small boat. Cores are 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. Cores were collected from the 5 embayments as follows: Stage Harbor System - 18 cores, Bassing Harbor System - 16 cores, Muddy Creek - 4 cores, Taylors Pond/Mill Creek - 5 cores, Sulphur Springs/Cockle Cove/Bucks Creek - 7 cores. Sampling was distributed throughout each embayment system and the core results combined for calculating the net nitrogen regeneration rates for the water quality modeling effort.

Sediment-watercolumn exchange follow the methods of Jorgensen (1977), Klump and Martens (1983), and Howes *et al.* (1995) for nutrients and metabolism. Upon return to the field laboratory (Chatham Water Quality Laboratory Annex) the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Oxygen consumption was determined in time-course incubations up to 24 hours. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and ortho-phosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining sample frozen (-20^{o}C) for assay of nitrate + nitrite (Cd reduction: Lachat Autoanalysis), and DON (D'Elia *et al.* 1977). Rates were determined from linear regression of analyte concentrations through time.

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 salt water analysis and sediment geochemistry.

IV.3.3 Determination of Summer Nitrogen Regeneration from Sediments

Watercolumn nitrogen levels are the balance of inputs from direct sources (land, rain etc), losses (denitrification, burial), regeneration (watercolumn and benthic), and uptake (e.g. photosynthesis). As stated above, during the warmer summer months the sediments of shallow embayments typically act as a net source of nitrogen to the overlying waters and help to stimulate eutrophication in organic rich systems. However, some sediments may be net sinks for nitrogen and some may be in “balance” (organic N particle settling = nitrogen release). Sediments may also take up dissolved nitrate directly from the watercolumn and convert it to dinitrogen gas, hence effectively removing it from the ecosystem. This process can be very effective in removing nitrogen loads, particularly in salt marshes and is termed “denitrification”.

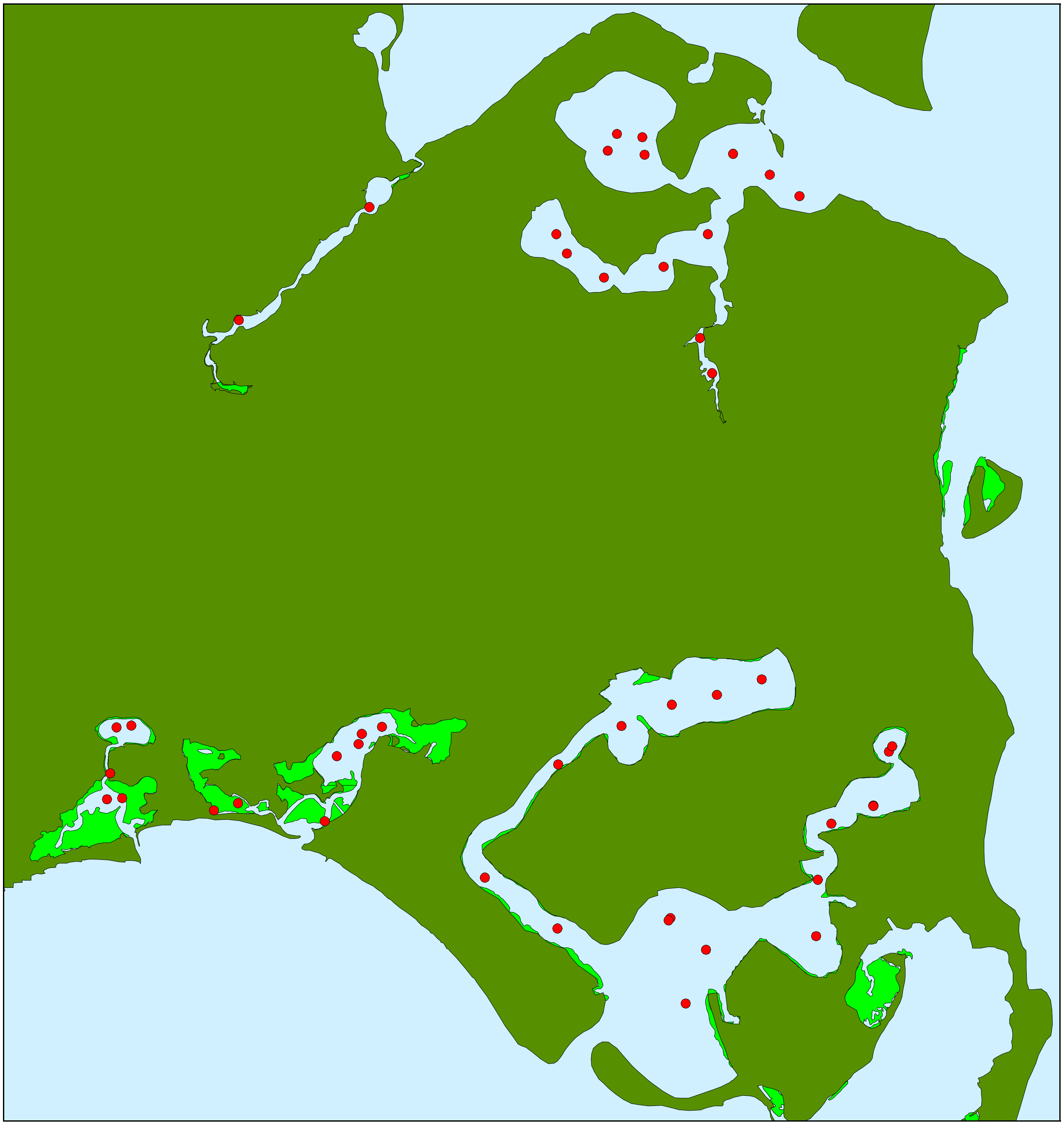


Figure IV-15. Chatham shoreline with locations of sediment core sampling stations shown as red filled circles. Some locations are sites of more than one sample. All sites were assayed in 2000 with Bassing Harbor having additional data collected in 2001.

In addition to nitrogen cycling, there are ecological consequences to habitat quality of organic matter settling and mineralization within sediments which relate primarily to sediment and watercolumn oxygen status. However, for the modeling of nitrogen within an embayment it is the relative balance of nitrogen input from watercolumn to sediment versus regeneration which is critical. 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-16).

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

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

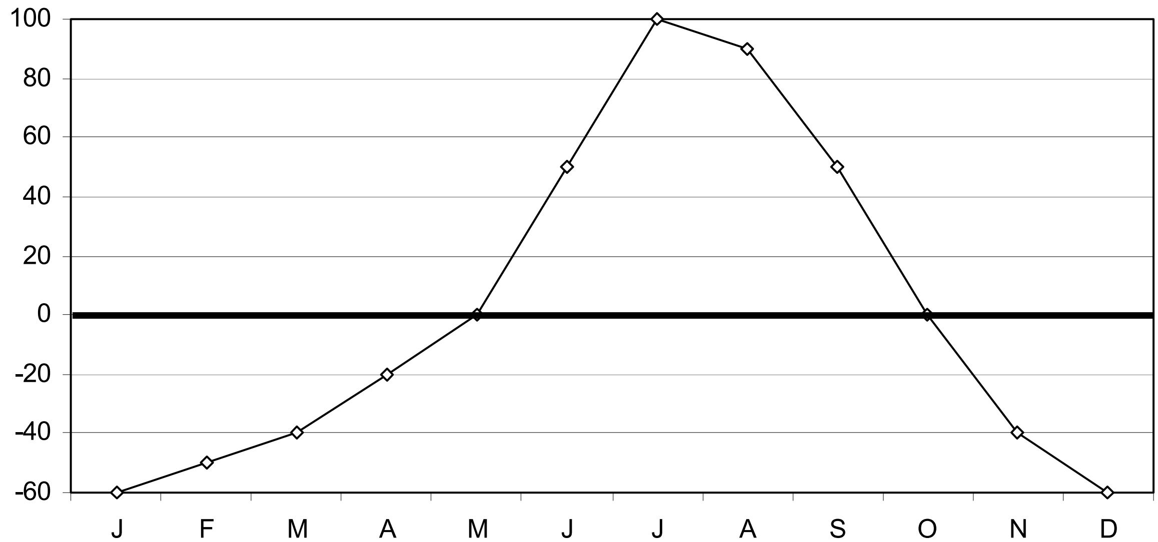


Figure IV-16. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

In order to obtain the net nitrogen balance of each embayments sediments, 48 cores were collected at 46 locations throughout the 5 embayments of Chatham (Figure IV-15). The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. Multiple cores were typically collected per sub-embayment and the results were averaged within an embayment for parameterizing the water quality model. For each core the nitrogen flux from the core incubations (described in the section above) were combined with measurements of the sediment organic carbon and nitrogen content and bulk density and an analysis of the sites tidal flow velocities. The maximum bottom water flow velocity at each coring site was determined from the calibrated and validated hydrodynamic model. The rate of organic nitrogen in particle settling was based upon measured particulate carbon and nitrogen concentrations measured during the appropriate summer, 2000 or 2001, by the Chatham Water Watchers and Pleasant Bay Alliance. These data were then used to determine the nitrogen balance of a sediment system.

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within each sediment site and the average summer particulate carbon and nitrogen concentration within the overlying water (from the monitoring program database). Two levels of settling were used. If the bulk density of the sediments indicated a fine grained substrate and data indicated a high carbon content and low velocities, then a water column particle residence time of 8 days was used (based upon phytoplankton studies of poorly flushed basins). If the sediments indicated a coarse grained sediments and low organic content and high velocities, then half this settling rate was used.

Adjusting the measured sediment releases was essential in order not to over-estimate the sediment nitrogen source and to account for those sediment areas which are net nitrogen sinks for the aquatic system. These results can be validated 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 indicates that sediment metabolism in the highly organic rich sediments of the wetlands and depositional basins was driven primarily by stored organic matter (ca. 90%). Also, in the more open lower portions of the 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 (Figure IV-17). As depicted in figure IV-17, with the exception of Frost Fish Creek, sediment nitrogen to organic carbon ratios indicate that phytoplankton is the prime source of carbon deposited in these sediments.

Net nitrogen release or uptake from the sediments of the 5 embayment systems used in the water quality modeling effort (Section VI) are presented in Table IV-7. There were concerns that the benthic regeneration rates measured in 2000 in the Bassing Harbor System were anomalously high (cf. Applied Coastal 2001), so the system was re-assayed in 2001. Evaluation of the 2000 and 2001 rates indicated that while the oxygen uptake rates were similar (<10% overall) in both years, the rates of nitrogen release in each of the sub-embayments were significantly higher in 2000 versus 2001. Additionally, the 2000 nitrogen release rates where higher than any observed rate in other systems. In contrast, the 2001 data shows nitrogen releases in line with other systems and oxygen uptake to nitrogen release ratios similar to other coastal systems (i.e. about 2 times the Redfield Ratio of 6.7 compared to 1.1 times in 2000). Only the 2001 data for the Bassing Harbor System was used. The variation for each embayment system encompasses the spatial variation within each sub-basin, due to organic matter deposition, water depth, sediment type, etc. Basins with small release rates (near zero) will have proportionally larger variation, however, since the release is low this variation is not generally ecologically significant. The critical way to view the data relates to the inter-basin differences, which typically indicate that the upper basins have the largest nitrogen release rates and the narrower flow regions have lowest release rates (e.g. Oyster Pond versus Oyster River, Mill Pond/Little Mill Pond versus Mitchell River).

Table IV-7. Rates of net nitrogen return from sediments to overlying waters based on sub-embayment area coverage and core flux measurements.

Sub-embayment	Net N Efflux (kg/day)	Standard Error
Stage Harbor		
Oyster Pond	26.8	3.4
Oyster River	0.7	0.9
Stage Harbor	12.8	2.9
Mitchell River	-3.4	2.9
Mill Pond	3.7	1.6
Little Mill Pond	2.0	0.5
Sulphur Springs		
Sulphur Springs	-3.6	1.1
Bucks Creek	2.9	--
Cockle Cove Creek	-0.9	0.2
Taylors Pond		
Mill Creek	-0.3	0.2
Taylors Pond	1.7	1.1
Bassing Harbor		
Crows Pond	3.5	1.7
Ryder Cove	7.4	2.8
Frost Fish Creek	-0.2	0.2
Bassing Harbor	-0.1	0.6
Muddy Creek		
Muddy Creek –lower	-1.9	0.6
Muddy Creek - upper	4.7	0.1

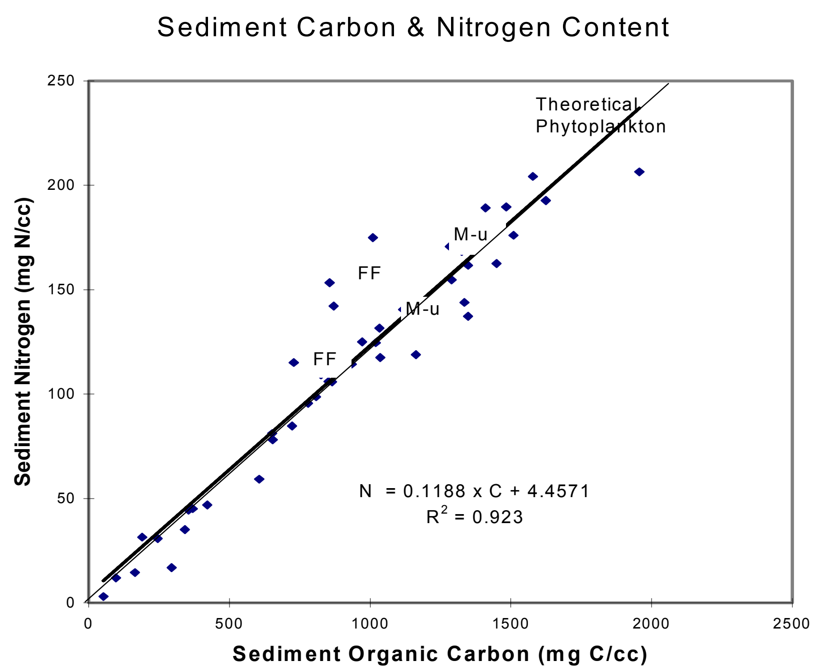


Figure IV-17. Sediment carbon vs. sediment nitrogen content for core samples taken from Chatham subembayments.

V. HYDRODYNAMIC MODELING

V.1. INTRODUCTION

To support the Town with their Comprehensive Wastewater Management Planning (CWMP), an evaluation of tidal flushing has been performed for the coastal embayments within the Town Limits of Chatham. The field data collection and hydrodynamic modeling effort contained in this report, provides the first step towards evaluating the water quality of these estuarine systems, as well as understanding nitrogen loading “thresholds” for each system. The hydrodynamic modeling effort serves as the basis for the total nitrogen (water quality) model, which will incorporate upland nitrogen load, as well as benthic regeneration within bottom sediments. In addition to the tidal flushing evaluation for these estuarine systems, alternatives analyses of tidal flushing improvement strategies have been performed for selected sub-embayments.

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

Estuarine water quality is dependent upon nutrient and pollutant loading and the processes that help flush nutrients and pollutants from the estuary (e.g., tides and biological processes). Relatively low nutrient and pollutant loading and efficient tidal flushing are indicators of high water quality. The ability of an estuary to flush nutrients and pollutants is proportional to the volume of water exchanged with a high quality water body (i.e. Nantucket Sound). Several embayment-specific parameters influence tidal flushing and the associated residence time of water within an estuary. For coastal embayments within Pleasant Bay and along the south coast of Chatham, the most important parameters are:

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

For this study, Chatham’s estuarine systems have been separated into three general groups: the 1) Stage Harbor System, 2) the South Coast Embayments and the 3) Pleasant Bay Region Embayments (see Figure V-1). Although the three estuarine systems along the south shore (Stage Harbor, Sulphur Springs, and Taylors Pond) exhibit different hydrologic characteristics, ranging from expansive salt marshes to flooded kettle ponds, the tidal forcing for these systems is generated from Nantucket Sound. In contrast, water propagating through the Chatham Harbor/Pleasant Bay system is derived from the Atlantic Ocean.

The south shore of Chatham exhibits a moderate tide range, with a mean range of about 4.5 ft. Since the water elevation difference between Nantucket Sound and each of the estuarine

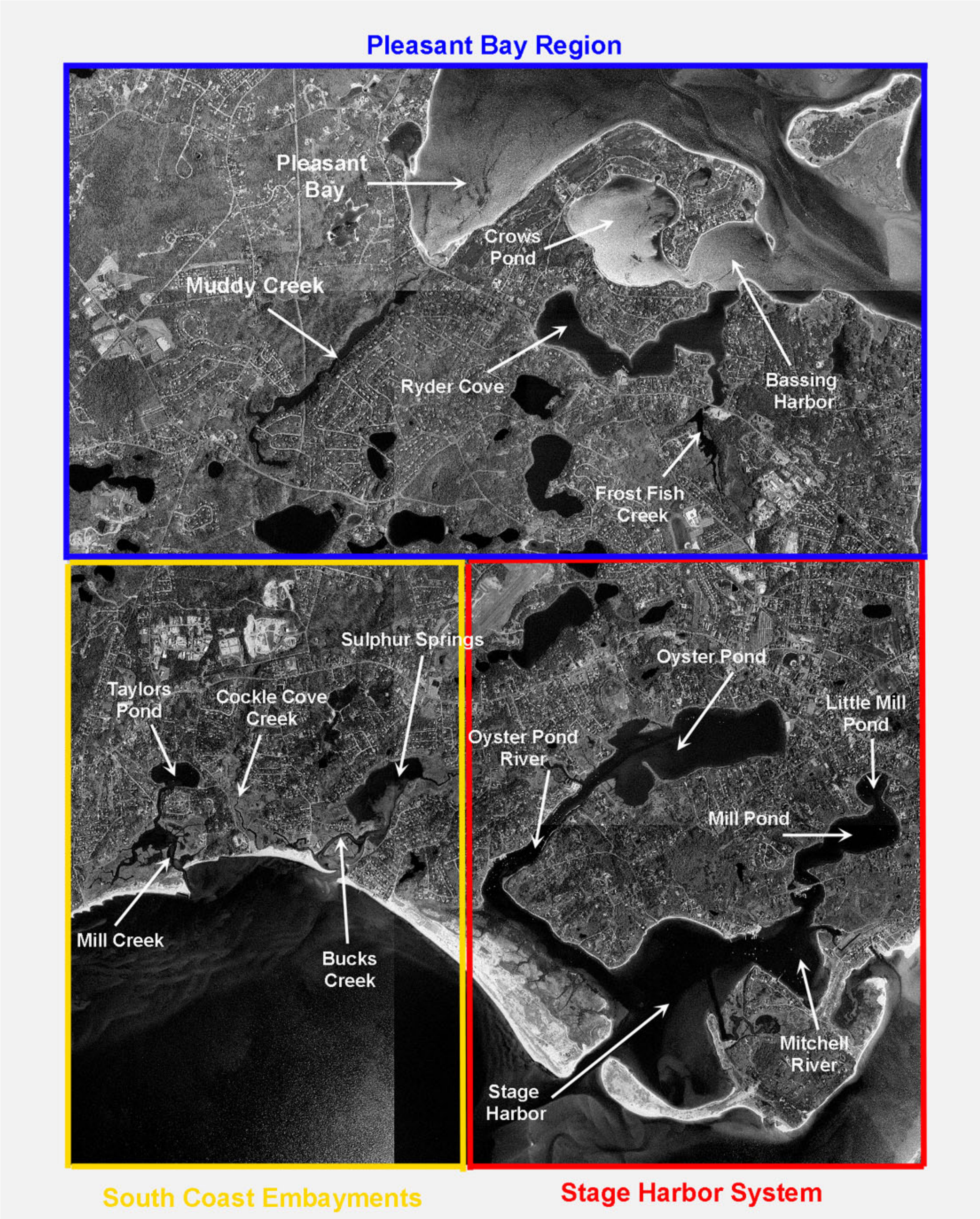


Figure V-1. Study region for the tidal flushing study including the estuarine systems in the Stage Harbor System (outlined in red), the South Coast Embayments (outlined in yellow), and the Pleasant Bay Region (outlined in blue).

systems is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle. Tidal damping (reduction in tidal amplitude) through the Stage Harbor system is negligible indicating "well-flushed" systems. In contrast, the tidal attenuation caused by the restrictive channels and marsh plains within the South Coast Embayments of Mill Creek/Taylors Pond is indicative of a "restrictive" system, where tidal flow and the associated flushing are inhibited. Based on the tidal characteristics alone, this might indicate that the Stage Harbor embayments (e.g. Little Mill Pond) are "healthy" relative to the embayments further the west; however, land development in the southeastern portion of Chatham likely provides a substantially higher nutrient load to the Stage Harbor embayments. Consequently, estuarine water quality may be more dependent on nutrient loading than tidal characteristics for these systems.

Within Pleasant Bay, the tide propagating through New Inlet and Chatham Harbor is significantly attenuated by the series of flood tidal shoals within the inlet throat. The mean tide range drops from just under 8 feet in the Atlantic Ocean to around 5 feet at the Chatham Fish Pier. Only minor attenuation occurs between the Fish Pier and Pleasant Bay; however, smaller sub-embayments separated from the main system by culverts exhibit significant additional tidal attenuation. Both Muddy Creek and Frost Fish Creek have mean tide ranges of less than 1 ft.

In addition to tidal forcing characteristics, the regional geomorphology influences flushing characteristics within embayments along the south shore, as well as for the Pleasant Bay system. Shoaling along the south shore of Chatham has caused the opening and closing of several inlets to the Sulphur Springs/Bucks Creek/Cockle Cove Creek system during the past 50 years. In addition, stability issues concerning the Stage Harbor navigation channel required repositioning of the inlet in 1965 as a result of regional shoaling. The most dramatic recent change in local geomorphology occurred in early 1987, when New Inlet formed east of the Chatham Lighthouse. From a tidal flushing and water quality perspective, the resulting increase in tide range within Pleasant Bay of approximately 1 ft caused a substantial improvement of regional tidal exchange.

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

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

To calibrate the hydrodynamic model, field measurements of water elevations and bathymetry were required. For the Stage Harbor System and the two South Coast Embayments, tide data was acquired within Nantucket Sound (two gauges were installed offshore of Cockle Cove Beach), Oyster Pond, Mill Pond, Little Mill Pond, Sulphur Springs, and Taylors Pond. For the Pleasant Bay Region, tide data was acquired within Pleasant Bay (two gauges were installed at the Chatham Yacht Club in Pleasant Bay), Crows Pond, Ryder Cove, Frost Fish Creek, and Muddy Creek. All 13 temperature-depth recorders (TDRs) were installed for a 30-day period to measure tidal variations through an entire neap-spring cycle. In this manner, attenuation of the tidal signal as it propagates through the various sub-embayments was evaluated accurately. In addition, currents were measured through a tidal cycle within the Stage Harbor and Bassing Harbor systems. These current measurements provided model verification data.

V.2. GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE ESTUARINE SYSTEM

The coast of Chatham is a highly dynamic region, where natural forces continue to reshape the shoreline. As beaches continue to migrate, episodic breaching of the barrier beach system creates new inlets that alter the pathways of water entering the series of local estuaries. Storm-driven inlet formation often leads to hydraulically efficient estuarine systems, where seawater exchanges more rapidly with water inside the estuary. However, this episodic inlet formation is balanced by the gradual wave-driven migration of the barrier beach separating the estuaries from the ocean. As beaches elongate, the inlet channels to the estuaries often become long, sinuous, and hydraulically inefficient. Periodically, overwash from storm events will erode the barrier beach enough at a point to allow again the formation of a new inlet. It is then possible that the new inlet will stabilize and become the main inlet for the system, while the old inlet eventually fills in. Several examples of this process along the Massachusetts coast include Allen's Pond (Westport), New Inlet/Chatham Harbor/Pleasant Bay (Chatham), and Nauset inlet (Orleans). In addition, alterations to the sediment transport patterns in the Cockle Cove/Ridgevale Beach region have altered the tidal inlets to both the Sulphur Springs and Cockle Cove Creek estuaries during the past 50 years.

In addition to natural phenomena affecting estuarine hydrodynamics, man-made alterations have impacted tidal exchange in Chatham's coastal embayments. Examples of anthropogenic modifications range from repositioning of the Stage Harbor inlet in the 1960's to the construction of culverts under Route 28 that restrict tidal exchange between two sub-embayments and Pleasant Bay. Manmade coastal/estuarine structures consist of jetties, groins, and dikes. Many of these structures were utilized to maintain the position of navigation channels; however, alterations to longshore sediment transport patterns may have influenced the stability of inlets to the region's estuaries. Also, dikes and weirs were constructed in the upper portions of some estuarine bodies to prohibit tidal exchange. In these cases, tidelands were reclaimed for agricultural purposes (primarily cranberry bogs).

Although man has modified much of the Chatham coastline, most of the large-scale changes to the estuarine systems have been caused by nature. For example, the 1987 breach of Nauset Beach caused a substantial increase in the tide range of Pleasant Bay (an increase of approximately 1.0 feet). In addition to increasing the tide range, this natural alteration to the system caused a substantial increase in tidal exchange with the Atlantic Ocean and improved the water quality in the upper portions of the estuary. Most of the manmade modifications to coastal embayments have caused small changes to overall estuarine health. While the culverts restricting tidal flow under Route 28 have had a negative influence on water quality within Muddy and Frost Fish Creeks, this influence is minor relative to natural large-scale changes.

V.2.1 Stage Harbor

As a result of regional littoral drift patterns, the location of the Stage Harbor entrance channel, as well as other channels influencing system circulation, have been altered over the past 100 years. Natural migration of the Nauset Beach system and Monomoy Island/Shoals has caused extensive periods of channel shoaling. Following the 1987 breach of Nauset Beach, abrupt changes to the circulation and sediment transport patterns within Chatham Harbor and Pleasant Bay occurred. Although alterations to the sediment supply to the southern portion of the Nauset Beach and Monomoy Island system have occurred less dramatically, landward migration of the southern remnants of Nauset Beach from the 1846 breach influenced circulation patterns within Chatham and Stage Harbors.

Figure V-2, from Geise (1988) illustrates the historic shoreline change of the Nauset Beach system over the 200-year period between 1770 and 1970. Following the 1846 breach, the barrier north of the inlet extended southward and the barrier beach south of Morris Island reattached to Morris Island (Figure V-2, between 1850 and 1950). By 1940, the same general form of 1800 had returned. Southward growth of Nauset Beach until it reached south of Morris Island and the separation of the southern barrier from Morris Island (forming Monomoy Island) occurred after 1940. This process continued until the 1987 breach of Nauset Beach initiated the cyclical pattern in a similar fashion as the 1846 breach.

Periodic breaching of the spit connecting Morris Island to Monomoy Island has caused local shifts in sediment transport patterns and associated shoaling in the vicinity of the Stage Harbor entrance. Prior to the mid-1960s, the inlet to Stage Harbor was located along the western edge of Morris Island, approximately 2,000 feet east of its present location. According to Geise (1988), Harding Beach was artificially breached in 1965 to create the inlet that exists today. As part of the inlet relocation project, the U.S. Army Corps of Engineers constructed a sand dike between the new inlet and Morris Island.

The proximity of the pre-1965 Stage Harbor entrance to Morris Island may have made it difficult to maintain a stable navigation channel once the spit south of Morris Island separated and formed Monomoy Island. In addition, migration of the barrier island remnants following the 1846 breach caused the separation of Amos Point from the mainland, forming Morris Island. Figures V-3, V-4, and V-5 illustrate the changes in the Morris Island region between 1893 and 1943. Prior to 1910, the Morris Island dike was protected from the Atlantic Ocean by Nauset Beach. As the barrier beach remnant migrated landward, it eventually joined the mainland. By 1917 (Figure V-3), natural overwash of the Morris Island dike created a hydraulic connection between Stage and Chatham Harbors. In 1943, the connection between Stage and Chatham Harbors allows circulation at all stages of the tide. By the 1960’s, the Morris Island dike was constructed, blocking the connection between the two harbors. The 1994 aerial photograph (Figure V-6) shows the present form of the Stage Harbor system, with the pre-1965 inlet location and the previous connection to Chatham Harbor shown.

The Morris Island dike will continue to prohibit tidal exchange between Stage and Chatham Harbors for the foreseeable future. In addition, the 1965 location of the Stage Harbor inlet likely will remain in its present location, which will require maintenance dredging on an as-needed basis to ensure safe navigation. Although a series of geomorphic and man-induced changes have occurred within the Stage Harbor estuary during the past 100 years, the existing system appears to be in a state of equilibrium.

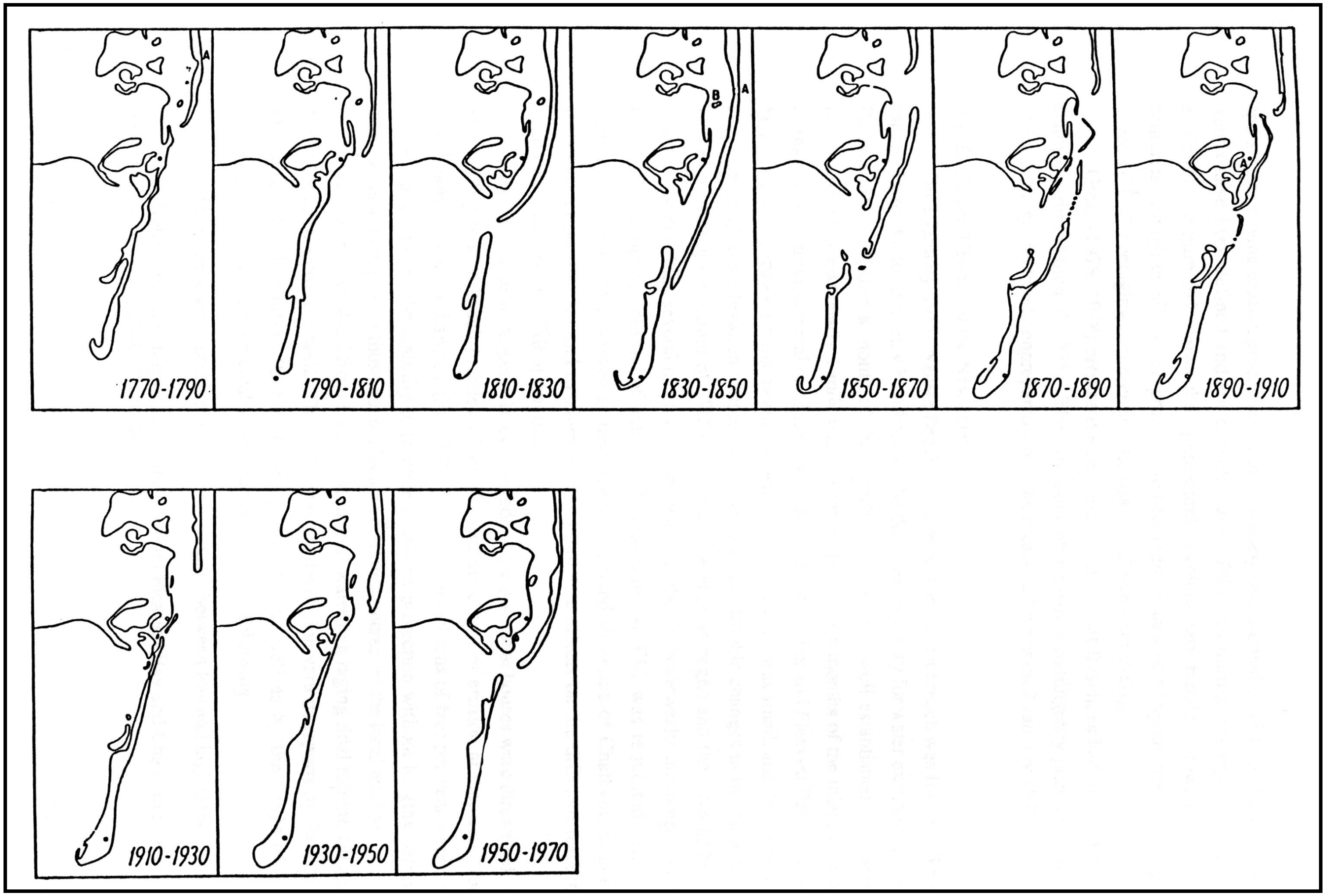


Figure V-2. Historical changes in the Nauset Beach-Monomoy barrier system illustrated by generalized 20-year diagrams from 1770-1790 to 1950-1970 (from Geise, 1987).

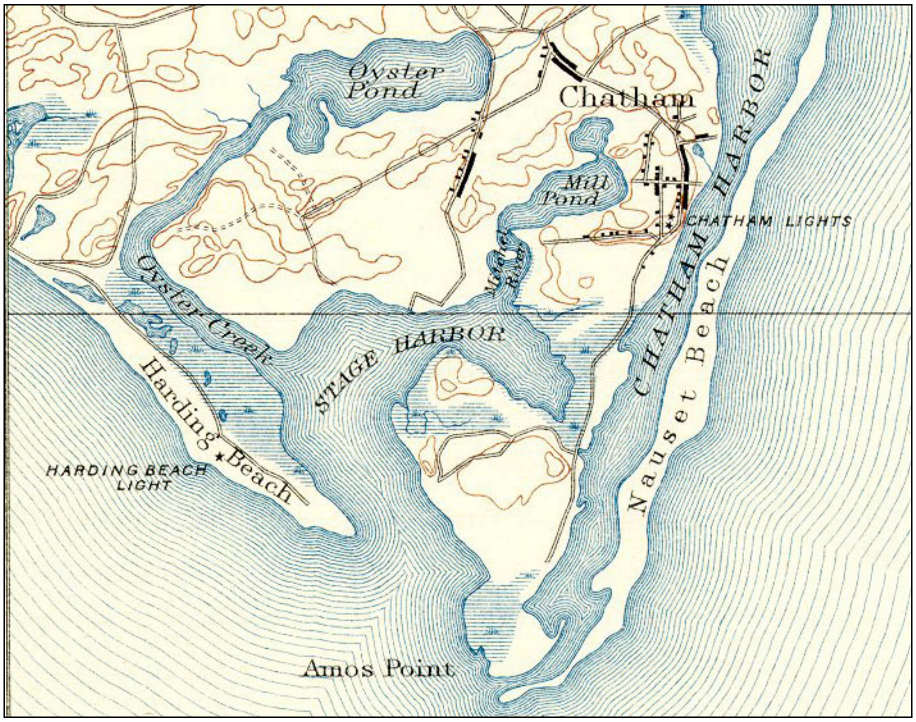


Figure V-3. Topographic map from 1893 showing the old Stage Harbor inlet location and a roadway to Morris Island/Amos Point.

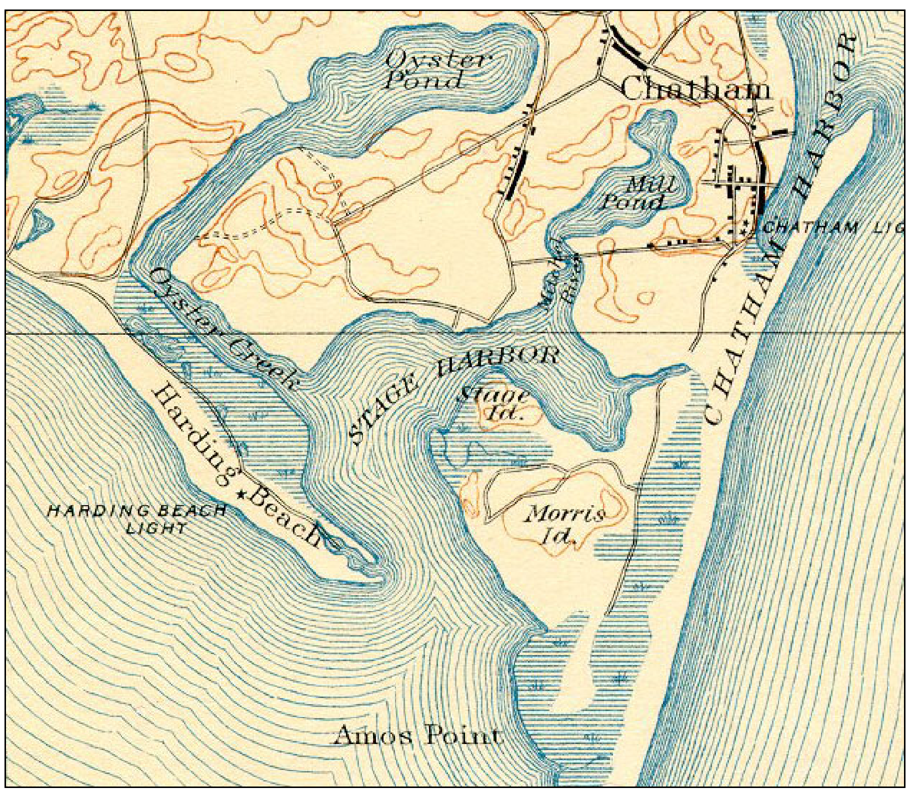


Figure V-4. Topographic map from 1917 showing the region of overwash between Stage and Chatham Harbors.

Figure V-5. Topographic map from 1943 showing the hydraulic connection between Stage and Chatham Harbors.

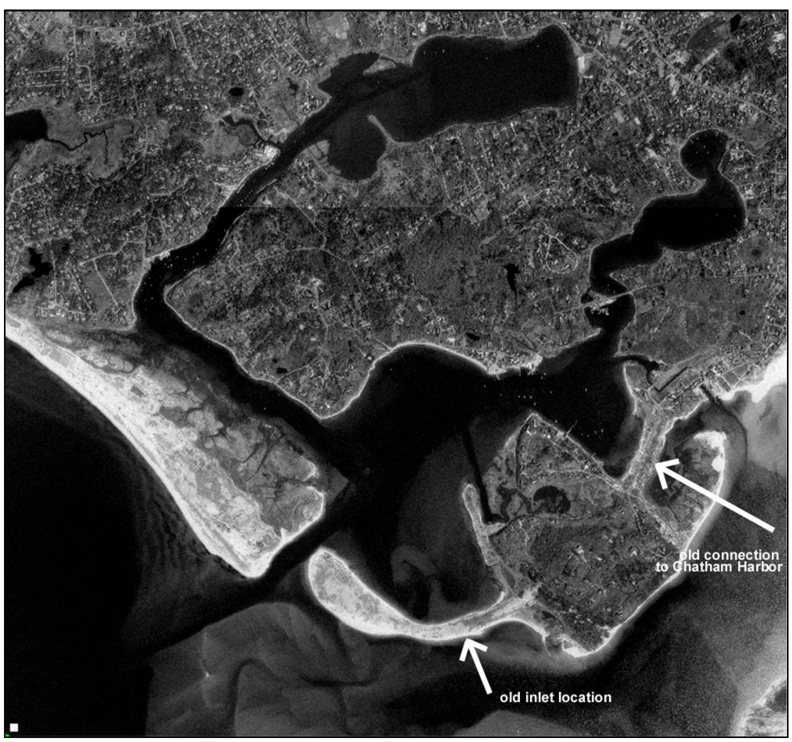


Figure V-6. Aerial photograph from 1994 showing the present system and the location of historic inlet features.

V.2.2 South Coast Embayments

A series of groins were constructed along the south coast of Harwich and Chatham during the 1950's. These structures primarily were constructed to the west of the Mill Creek inlet as a means of trapping littoral drift to stabilize the shoreline. Although these structures often succeeded in protecting beaches updrift (to the west) of the groin fields, downdrift beaches were often starved of sediments. As illustrated in the 1943 topographic map (Figure V-7), a natural sand spit extended from the Mill Creek inlet to the east approximately 1,500 feet, forming Cockle Cove. Taylors Pond/Mill Creek, Cockle Cove Creek (referred to as Bucks Creek on the map), and Sulphur Springs were all connected directly to Cockle Cove.

Following the construction of the groin field in the 1950's, the spit forming Cockle Cove began migrating shoreward as the sediment source for this feature disappeared. The spit was overwashed during storm events, and formed a series of shore parallel bars in the remnants of Cockle Cove. An aerial photograph from the 1950's (Figure V-8) illustrates the series of migratory sandbars. In addition, overwash processes closed the Buck's Creek entrance, forcing tidal waters to enter Sulphur Springs via Cockle Cove Creek.

By the late 1970's, the barrier spit remnants had attached to the shoreline and the three estuarine systems each had a direct connection with Nantucket Sound. Figures V-9, V-10, and V-11 show a series of oblique aerial photographs of the Taylors Pond, Cockle Cove Creek, and Sulphur Springs systems taken in 1979. The Taylors Pond estuary is similar in form to the system that exists today. Figure V-10 illustrates a barrier dividing Cockle Cove and Buck's Creeks, maintaining the hydraulic separation of these estuaries. By this time period, the "training" groins to the east of the Buck's Creek entrance had been constructed. Training groins are used to direct tidal currents from an inlet away from the shoreline in order to limit shoreline erosion and the natural migration of this inlet (Figure V-11).

The 1994 aerial photograph (Figure V-12) illustrates the general form of the estuaries as they appear today. The Cockle Cove Creek inlet has closed and the connection to the Buck's Creek system was reestablished. Although it is possible that a storm could breach the barrier beach to the east of the Cockle Cove Beach parking lot and reopen the Cockle Cove Creek entrance, ongoing beach nourishment efforts along Cockle Cove Beach should continue to stabilize the beach in its present location. Since the sediment supply to beaches in this region is relatively small, it is likely that the system will retain its general form of one inlet servicing both Cockle Cove Creek and the Buck's Creek/Sulphur Springs system.

V.2.3 Pleasant Bay Region

Many of the regional barrier beach systems in Chatham formed after a rise in relative sea level during the Holocene. Approximately 5,000 years before present, relative sea-level was about 15-20 feet below the level existing today. As relative sea level increased over the past 5,000 years, continued erosion of the cliffs in Orleans, Eastham, Wellfleet, and Truro provided sediment to downdrift beaches, modifying the form of the nearshore area. Nauset Beach formed from the erosion of these cliffs and the predominant southerly littoral drift. As relative sea-level continued to increase, the bluffs along the eastern shore of Cape Cod continued to erode and the shoreline moved to the west. Nauset Beach has migrated to the west as a result of episodic overwash events in a process referred to as barrier beach rollover. The "Halloween Storm" of 1991 was an example of this rollover process, where the barrier beach was steepened and large volumes of sand were deposited into Pleasant Bay.

The formation of New Inlet in 1987 altered the hydrodynamics within the Pleasant Bay Estuary. As a result of the inlet, the tide range in Pleasant Bay has increased by approximately 1 ft, with a corresponding improvement to tidal flushing within the northern portions of the estuary. The inlet continues migrating south and Nauset Beach will return to a morphology similar to the pre-breach form. This pattern of inlet formation and southerly growth of Nauset Beach is cyclical. The two most recent breaches through the Nauset barrier occurred in 1846 east of Allen Point and 1987 east of the Chatham Lighthouse. The anticipated cyclical behavior of the inlet system is based on the work of Geise (1988) who described the historical 1846 breach and the subsequent re-formation of Nauset Beach during the next 140 years. Figure V-2 illustrates the cyclical behavior of the Chatham Harbor/Pleasant Bay system between 1770 and 1970.

Following the 1987 breach, the beach system returned to its 1846 form and the cycle started again. As Nauset Beach continues to grow in a southerly direction, the estuarine system becomes less hydraulically efficient, and the phase lag between high tide in the Atlantic Ocean and high tide in the estuary becomes greater. Once a small breach forms during storm overwash conditions, the difference in water elevations between the ocean and the estuary will quickly scour a more efficient channel that will eventually widen to an inlet. For example, the 1987 breach occurred on January 2nd; within one month the breach was well established and within four months the inlet was nearly one mile wide.

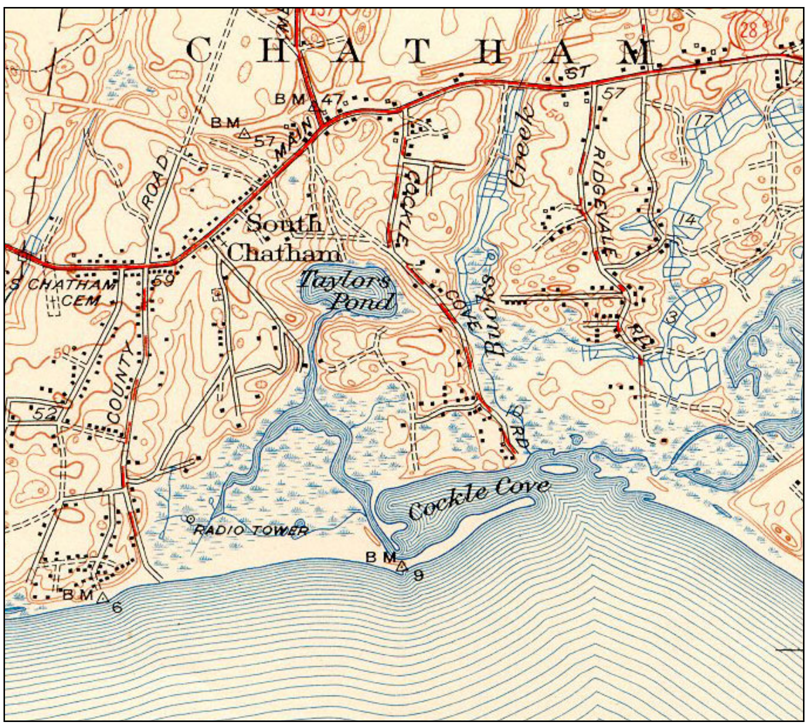


Figure V-7. Topographic map from 1943 showing the Taylors Pond, Cockle Cove Creek (designated as Bucks Creek on this map), Sulphur Springs, and the sand spit forming Cockle Cove.

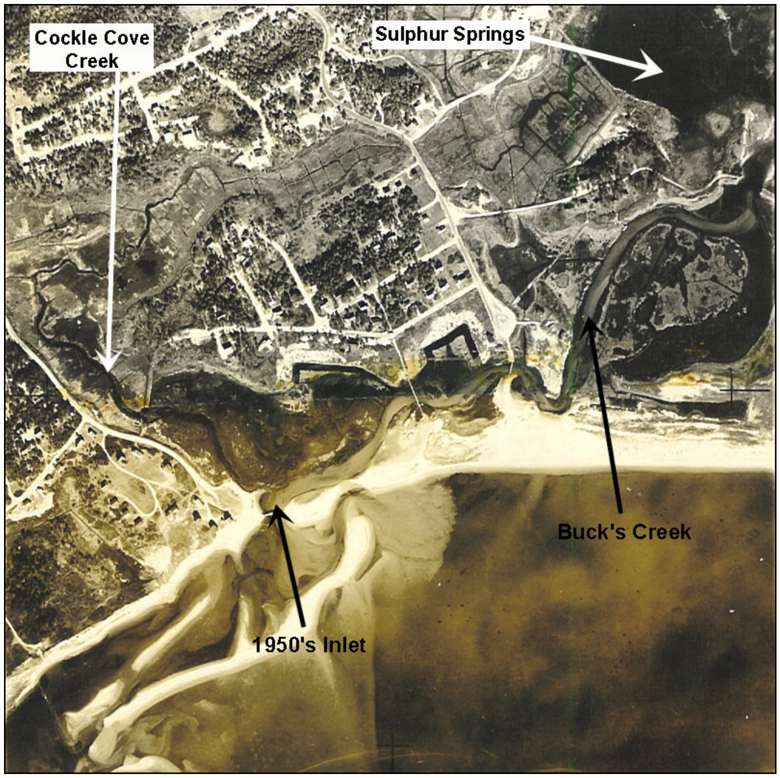


Figure V-8. Aerial photograph from the 1950's showing the shore parallel bars along Cockle Cove Beach and the location of the inlet servicing the Sulphur Springs/Buck's Creek/Cockle Cove Creek system.

Over the past 13 years, littoral drift (wave-driven transport along the outer coast) has caused several shifts in the inlet position of the Chatham Harbor system. Immediately following the 1987 breach, a two-inlet system existed, where Chatham Harbor was connected directly to the Atlantic Ocean east of Chatham Lighthouse and the historic inlet south of Morris Island remained open. By the early 1990's the more efficient inlet across from the lighthouse had captured the tidal prism and the southern inlet had closed. The primary navigation channel migrated to the south and ran along the beach fronting the lighthouse. Between 1999 and 2001, a second navigation channel opened across the swash platform to the north. Each of these changes has altered the pathway of tidal waters entering the Chatham Harbor/Pleasant Bay estuary, as well as the tide range and flushing characteristics of the system.

As Nauset Beach continues its southerly growth, the inlet naturally will become less efficient. Although this process will be gradual over the next 50-to-100 years, the Town of Chatham should consider the impact of this cyclic inlet behavior on estuarine flushing and the associated water quality. For this reason, the previous flushing study of the Pleasant Bay (ACI, 1997) evaluated existing conditions in 1997 and the less hydraulically efficient pre-breach conditions. The flushing analysis performed for the present study also incorporates existing conditions and the 'worst-case' pre-breach conditions.

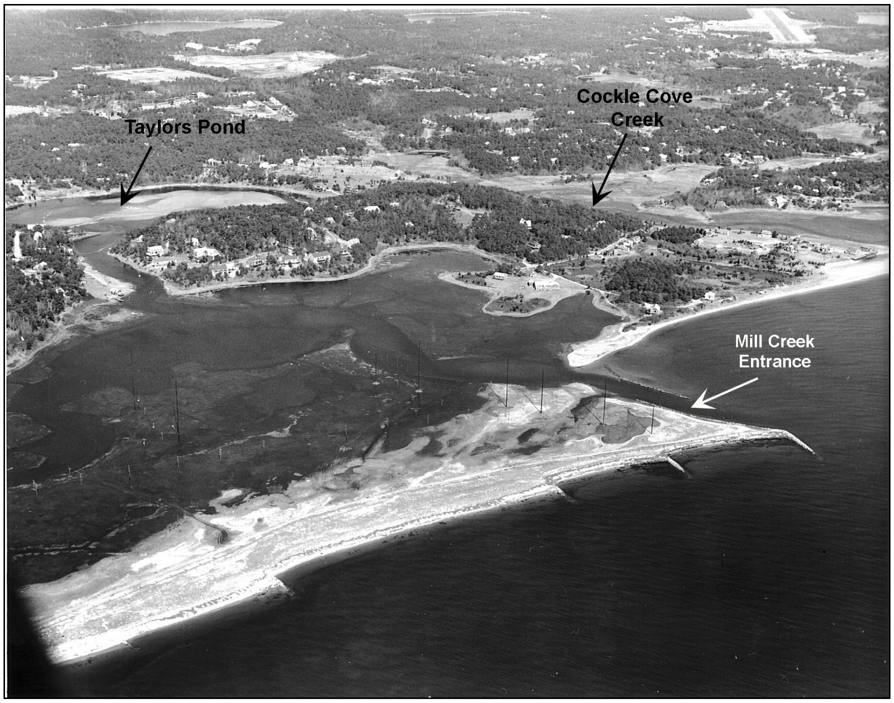


Figure V-9. Oblique aerial photograph from 1979 showing the Taylors Pond and Mill Creek system, with the groin field established updrift of the Mill Creek entrance.

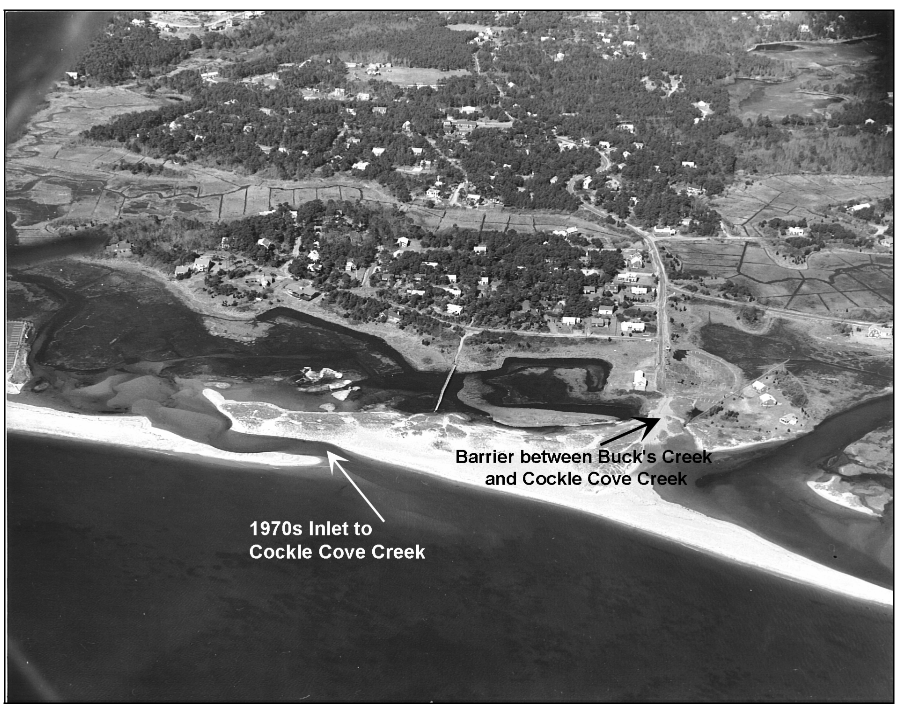


Figure V-10. Oblique aerial photograph from 1979 showing the inlet to Cockle Cove Creek, as well as the barrier separating the Buck’s Creek and Cockle Cove Creek systems.

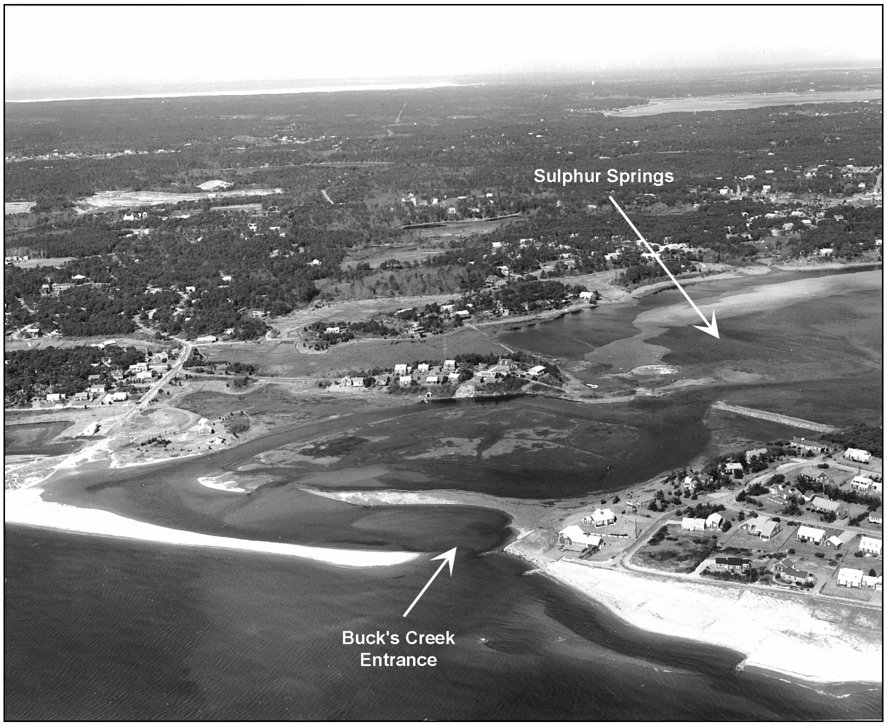


Figure V-11. Oblique aerial photograph from 1979 showing the inlet to Buck's Creek, as well as the barrier separating the Buck's Creek and Cockle Cove Creek systems.

Figure V-12. Aerial photograph from 1994 showing the Taylors Pond, Cockle Cove Creek, and Sulphur Springs estuarine systems as they exist today.

V.2.3.1 Basssing Harbor, Ryder Cove, and Crows Pond

Although the formation of New Inlet in 1987 increased the tide range in all of the sub-embayments surrounding Pleasant Bay, alterations to flow patterns were relatively minor in Bassing Harbor, Ryder Cove, and Crows Pond. The shoals at the entrance of Bassing Harbor have continued to shift over the past decade; however, the cross-sectional area of the channel appears to be stable. Therefore, tidal flushing has not been affected by the minor inlet shoaling.

V.2.3.2 Muddy Creek and Frost Fish Creek

Construction of a roadway along the Route 28 corridor has inhibited tidal exchange between Pleasant Bay and two sub-embayments within Chatham (Muddy Creek and Frost Fish Creek). For Muddy Creek, structures intended to control water levels have been in use since the turn of the century (Duncanson, 2000), and have included, at different times, tide gates, a dam, and the present earthwork structure and culvert under Route 28. In their present condition, the culverts at both Frost Fish Creek and Muddy Creek cause more than a three-fold decrease in the tide range. The location of these culverts is shown on Figure V-13.

The two culverts running under Route 28 at Muddy Creek each have a height of approximately 2.6 feet and a width of 3.7 feet. Since the surface area of Muddy Creek is relatively large, these culverts are not of sufficient size to allow complete tidal exchange between Pleasant Bay into Muddy Creek. This poor tidal exchange is likely responsible for the water quality concerns for the Muddy Creek system. Alternatives to improve tidal flushing and water quality within Muddy Creek are discussed in Section VI.

Two types of flow control structures exist at Frost Fish Creek. First, three partially-blocked 1.5 feet diameter culverts run under Route 28. Approximately 100 feet upstream of these culverts, a single large culvert and a dilapidated weir structure maintain the Creek level well above the mean tide elevation in adjoining Ryder Cove. Since the weir structure likely maintained Frost Fish Creek as a freshwater system, the culverts were adequate for handling the freshwater outflow from the Frost Fish Creek watershed. Following removal of the weir boards, Frost Fish Creek became a salt marsh system with a tide range of less than 0.5 feet. Based on an interpretation of watersheds delineated by the Cape Cod Commission, the freshwater recharge into Frost Fish Creek represents approximately 20% of the flow through the Route 28 culverts. Similar to Muddy Creek, the size of the culverts limits tidal exchange with Ryder Cove and the rest of the Pleasant Bay estuary. The poor tidal exchange is likely responsible for the water quality concerns within Frost Fish Creek. Alternatives to improve tidal flushing in Frost Fish Creek are discussed in Section VI.

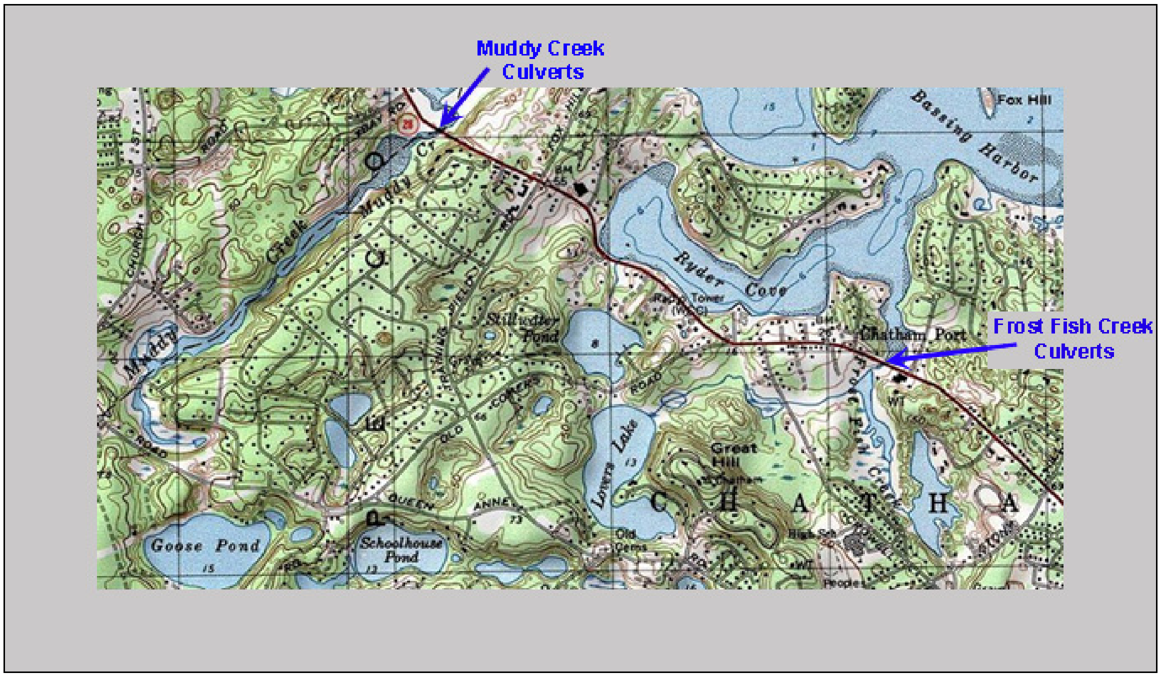


Figure V-13. Topographic map indicating the location of the Muddy Creek and Frost Fish Creek culverts inhibiting tidal exchange with the Pleasant Bay estuary.

V.3 FIELD DATA COLLECTION AND ANALYSIS

A precise description of embayment geometries and hydrodynamic forcing processes is required for the development of numerical models. To support the hydrodynamic and water quality modeling effort in the Stage Harbor, South Coast, and Pleasant Bay regions, tidal currents, water elevation variations, and bathymetry of the embayments were measured. For the purposes of this study, the Stage Harbor geographic region consists of Stage Harbor, Oyster Pond River, Oyster Pond, Mitchell River, Mill Pond, Little Mill Pond (Figure V-14). The South Coast Embayments consist of Cockle Cove Creek, Bucks Creek, Sulphur Springs, Mill Creek, and Taylors Pond (Figure V-14). The following embayments are grouped together in the Pleasant Bay region in this study: Bassing Harbor, Crows Pond, Ryder Cove, Frost Fish Creek, and Muddy Creek (Figure V-15). Cross-channel current measurements were surveyed through a complete tidal cycle at three locations in each system. Tidal elevation measurements within selected embayments were used for both forcing conditions and to evaluate tidal attenuation through each estuarine system. Bathymetry data was collected in regions where the coverage of previous work lacked accuracy and/or detail necessary for evaluation of tidal hydrodynamics. In the Stage Harbor and South Coast regions, bathymetric surveys were conducted in Stage Harbor, Oyster Pond, Mitchell River, Mill Pond, Bucks Creek, and Mill Creek. Bathymetry was collected in Bassing Harbor, Ryder Cove, Crows Pond, Frost Fish Creek, and Muddy Creek for the Pleasant Bay region. The depth measurements were supplemented by bathymetry from past surveys for use in the construction of computational grids for each modeled 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) 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 6 locations in the Stage Harbor and South Coast regions (Figure V-14) in late July 2000 and recovered in late August 2000. The gages were re-deployed in 5 locations in the Pleasant Bay region (Figure V-15) in late August 2000 and recovered in late September 2000. Data records span at least 29 days to yield an adequate time period for resolving the primary tidal constituents at each site.

The Temperature-Depth Recorders (TDR) used to measure tide levels for this study were the Brancker TG-205, Coastal Leasing Macrotide, and Global Water WL-14 instruments. Temperature and pressure data were recorded at 10-minute intervals, with each 10-minute observation resulting from an average of 60 1-second pressure measurements. Each of these instruments use strain gage transducers to sense variations in pressure, with resolution on the order of 1 cm (0.39 inches) head of water. The proper calibration of each gage was verified 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. The Global Water WL- 14 gage is vented to atmosphere, so it records pressure from the water column above the instrument only, and therefore does not need atmospheric correction. The atmospheric pressure record used to correct the other TDR data was derived by subtracting the Global Water WL-14 pressure time series from a second gauge deployed at the same location, which recorded the sum of water column and atmospheric pressure. Further, a (constant) water density value of 1025 kg/m^3 was applied to the readings to convert from pressure units (psi) to head units (for example, feet of water above the tide gage). This density assumption is appropriate for even Muddy Creek and Frost Fish Creek where brackish water (salinity less than 15-20 ppt) is possible, because density variations are small (at most 2.5% between fresh water and sea water), and the gages were positioned in relatively shallow water (less than 3ft). All TDR gages were surveyed into local benchmarks to provide vertical rectification of the water level; these survey values were used to adjust the water surface to a known vertical datum.

The result from each gage is a time series representing the variations in water surface elevation relative to NGVD29 (National Geodetic Vertical Datum of 1929). NGVD29 is a standard fixed vertical reference. Though it is based on the 1929 mean sea level at several stations in the US, there is no consistent relationship between NGVD29 and mean sea level at different locations. FiguresV-16 toV-18 present the water levels at each gage location shown in FiguresV-14 andV-15. The only irregularity experienced during both deployments of the TDRs in Chatham was for the gage in Muddy Creek. Failure of the primary battery caused the unit to stop recording 29.8 days into the deployment, as can be seen in the early termination of the plot in Figure V-18 for Muddy Creek. Data quality is not impacted by primary battery failure. Because the Muddy Creek data record is longer than 29 days, the data record is sufficient to perform a harmonic analysis of its significant tidal constituents.

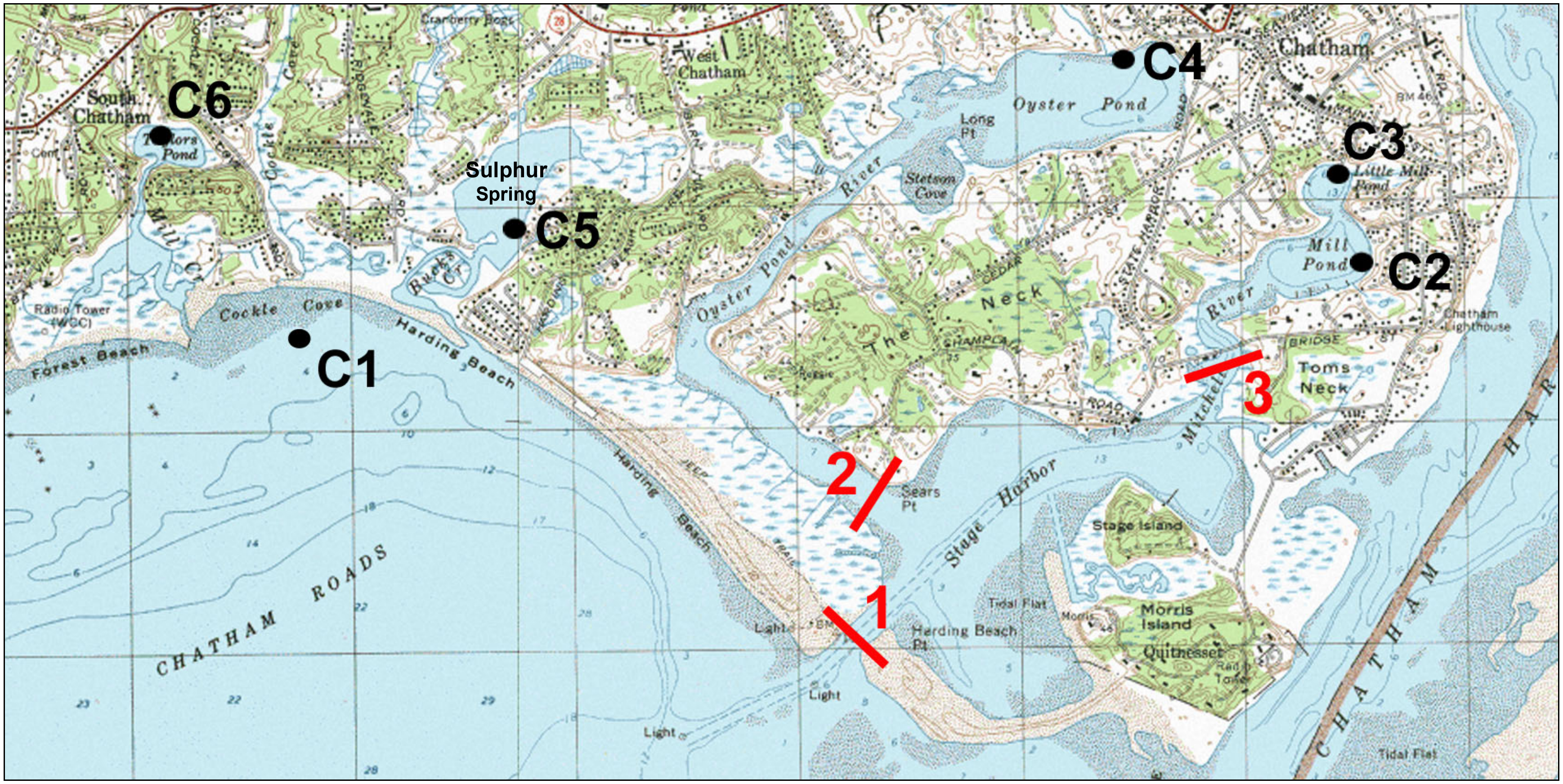


Figure V-14. Tide gage and ADCP transect locations in the Stage Harbor region (C1-C6 are tide gage locations and heavy red lines 1-3 are ADCP transects)

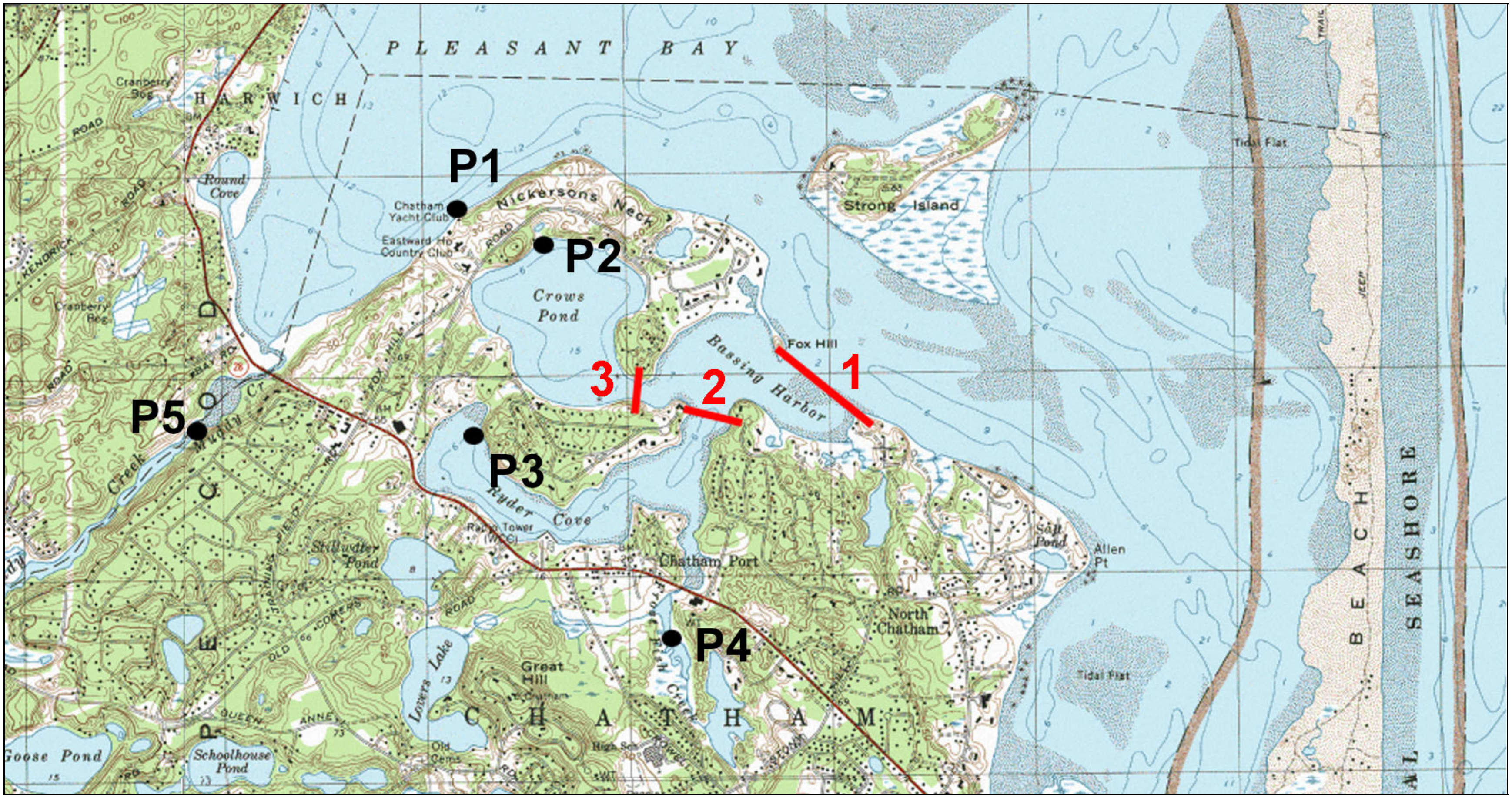


Figure V-15. Tide gage and ADCP transect locations in the Pleasant Bay region (P1-P5 are tide gage locations and heavy red lines 1-3 are ADCP transects).

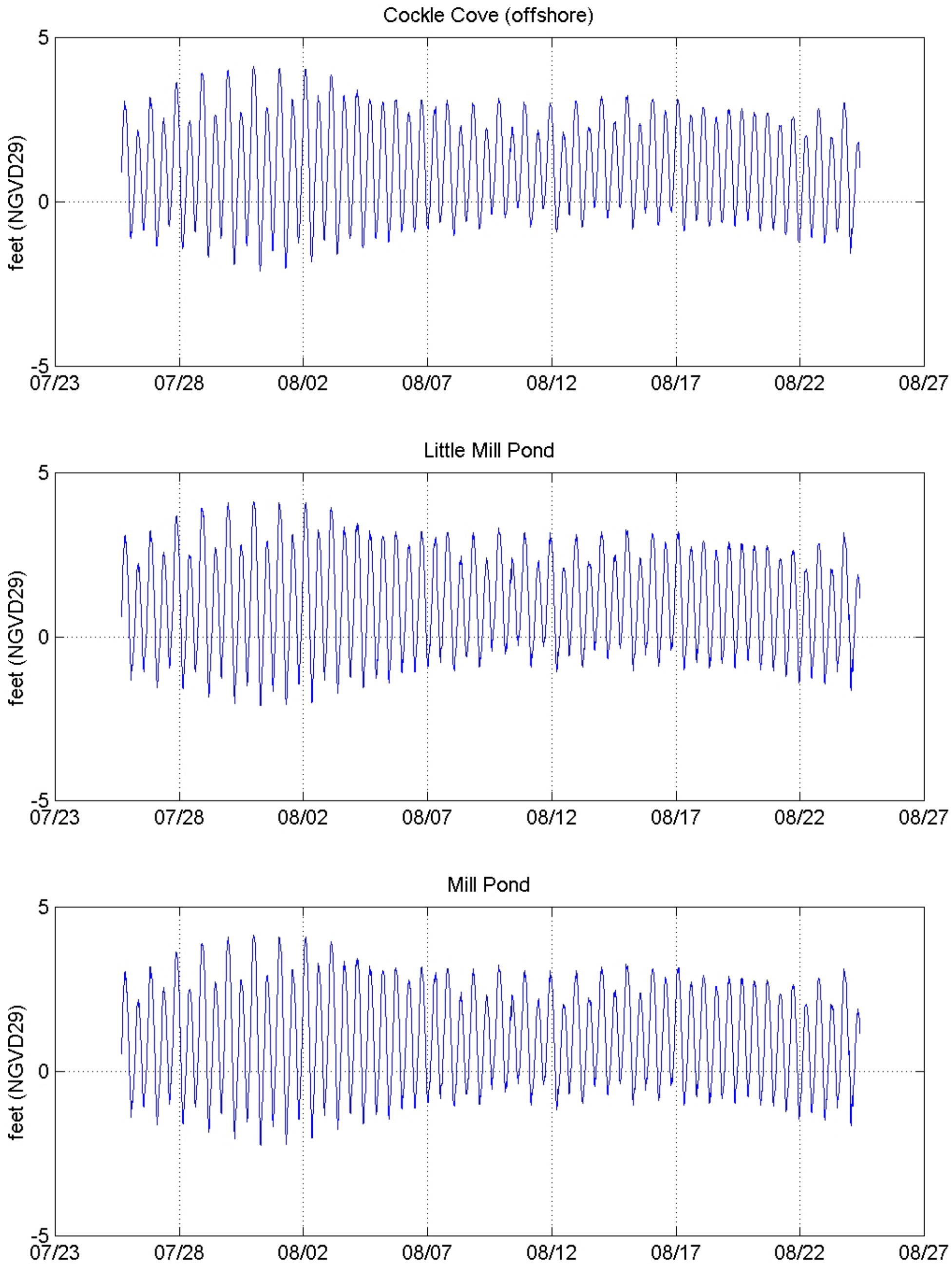


Figure V-16. Tidal elevation observations for offshore Cockle Cove Beach (location C1 of Figure V-14), Little Mill Pond (location C2), and Mill Pond (location C3).

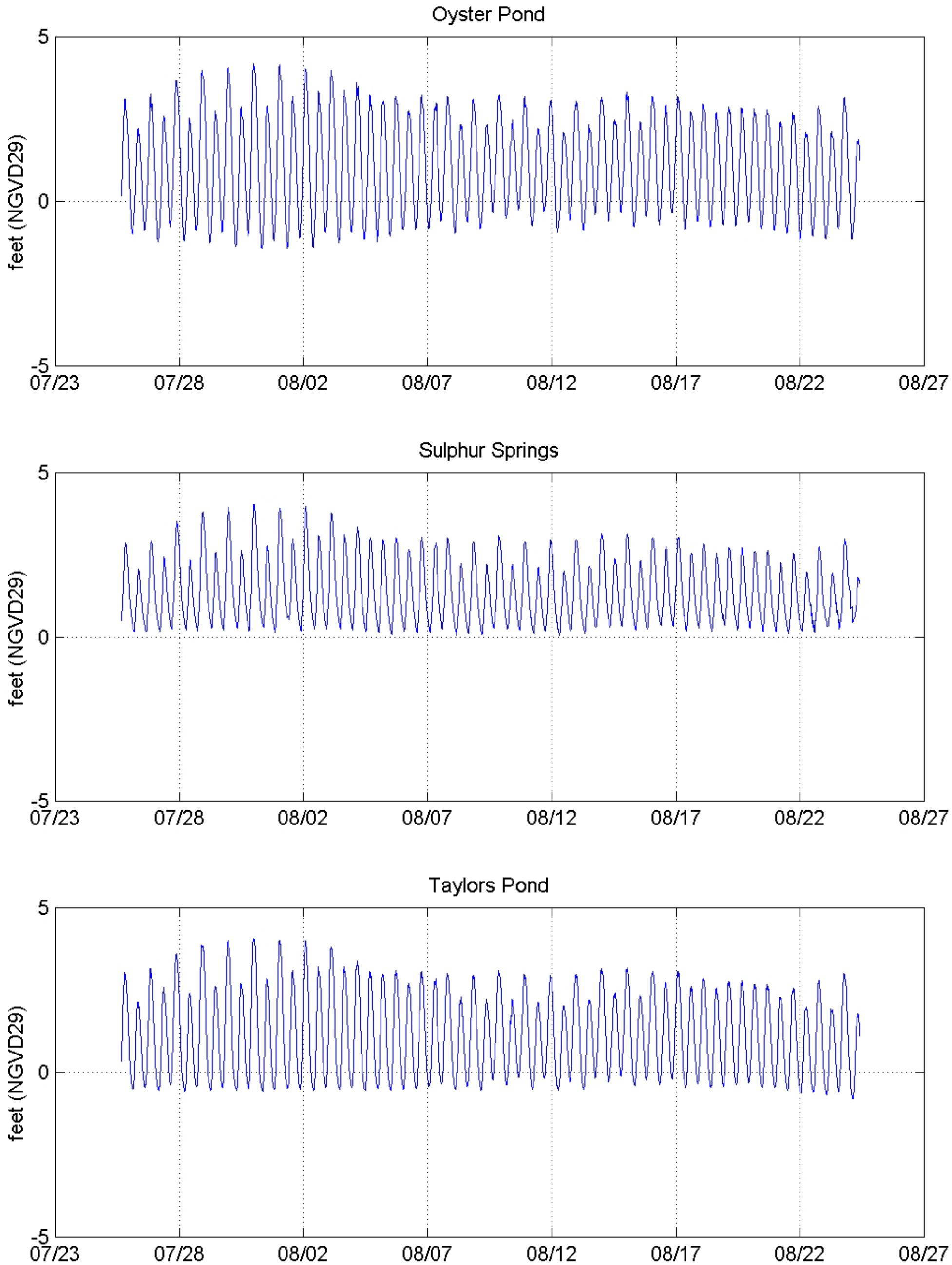


Figure V-17. Tidal elevation observations for Oyster Pond (location C4 of Figure V-14), Sulphur Springs (location C5), and Taylors Pond (location C6).

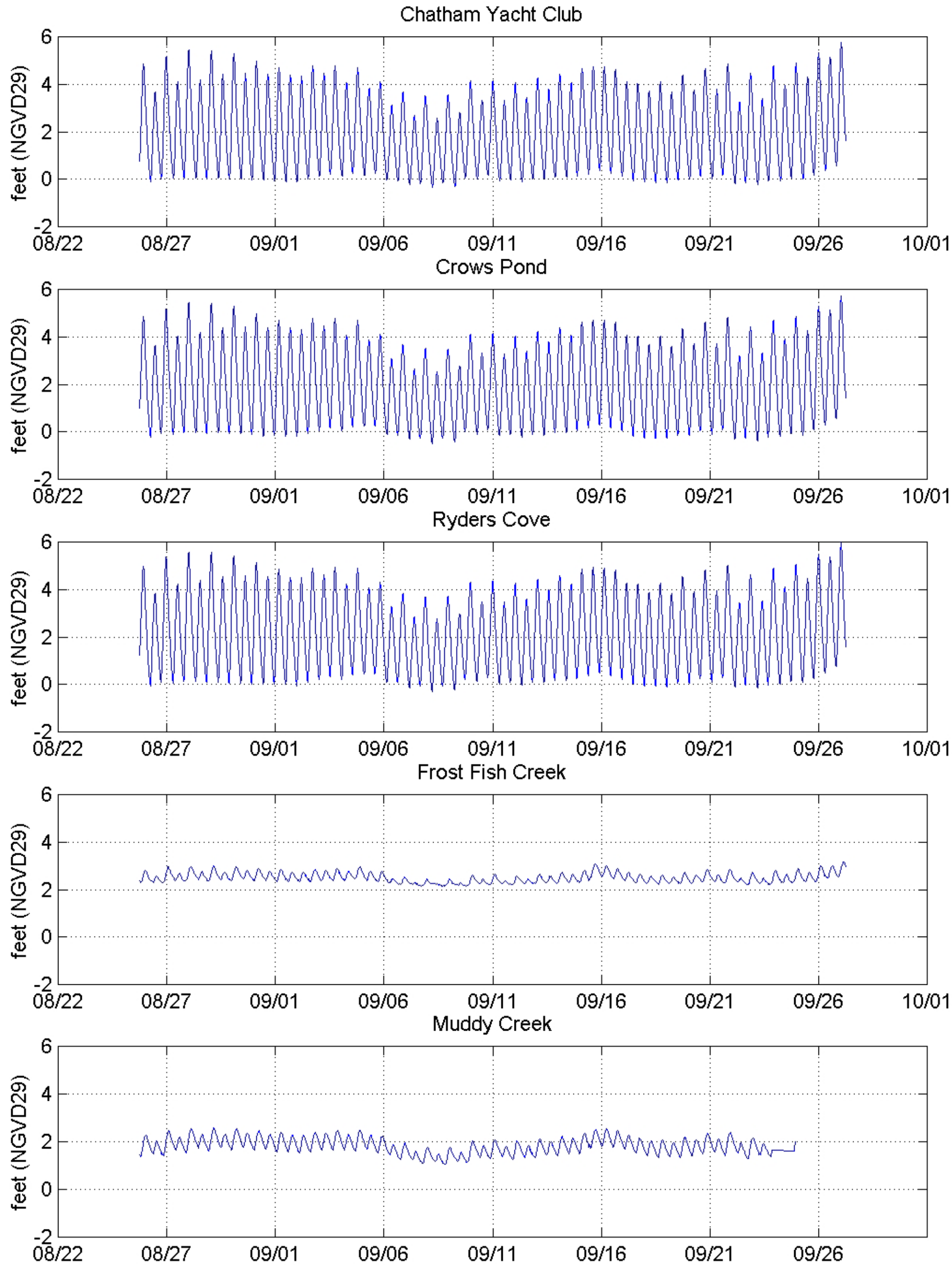


Figure V-18. Tidal elevation observations for Chatham Yacht Club (location P1 of Figure V-15), Crows Pond (location P2), Ryder Cove (location P3), Frost Fish Creek (location P4), Muddy Creek (location P5).

V.3.1.2 Bathymetry

V.3.1.2.1 Stage Harbor and South Coast Embayments

Bathymetry, or depth, of Stage Harbor, Mill Pond, Oyster Pond, and Mill Creek was measured during field surveys in August 2000. 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 echosounder and GPS were logged to a laptop computer.

GPS positions and echosounder measurements were merged to produce data sets consisting of water depth as a function of x-y horizontal position (in Massachusetts Mainland State Plane, 1983). The data were combined with water surface elevations to obtain the vertical elevation of the bottom (z) relative to the NGVD 1929 vertical datum (NGVD29). The resulting xyz files were input to mapping software to calculate depth contours for the system shown in Figures V-19 andV-20. Where necessary, bathymetry collected by Applied Coastal was supplemented by existing data from NOAA collected in 1956. The 1956 NOAA data were not used in southern Stage Harbor, where significant changes have occurred as a result of the relocation of the harbor inlet in 1965.

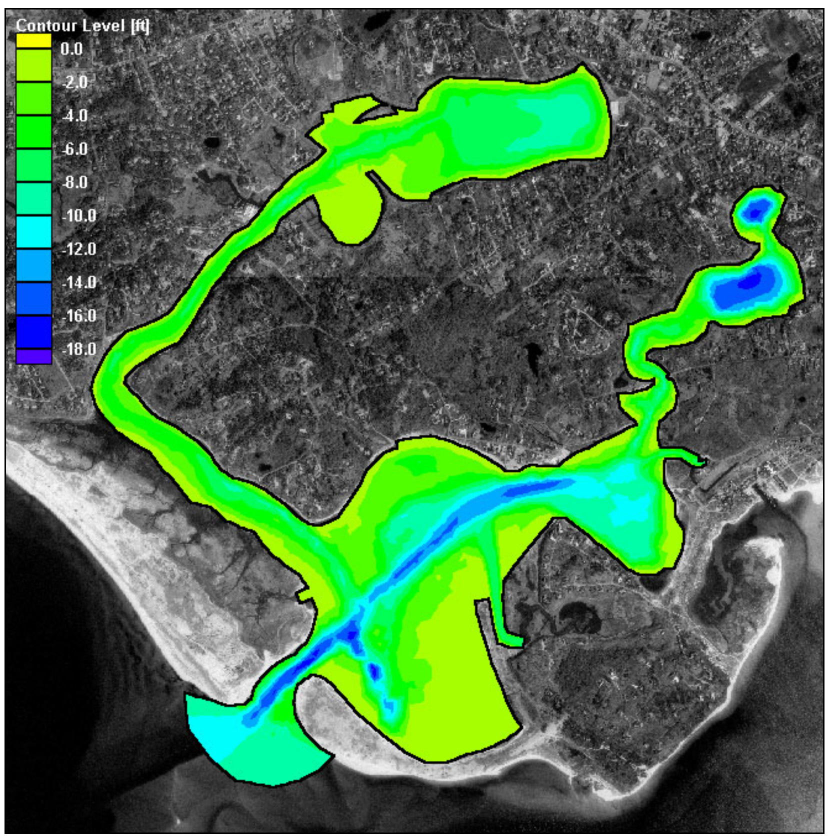


Figure V-19. Depth contour plots of the numerical grid for the Stage Harbor system at 2-foot contour intervals relative to NGVD29 (August 2000 survey, supplemented with 1956 NOAA data in the upper reaches).

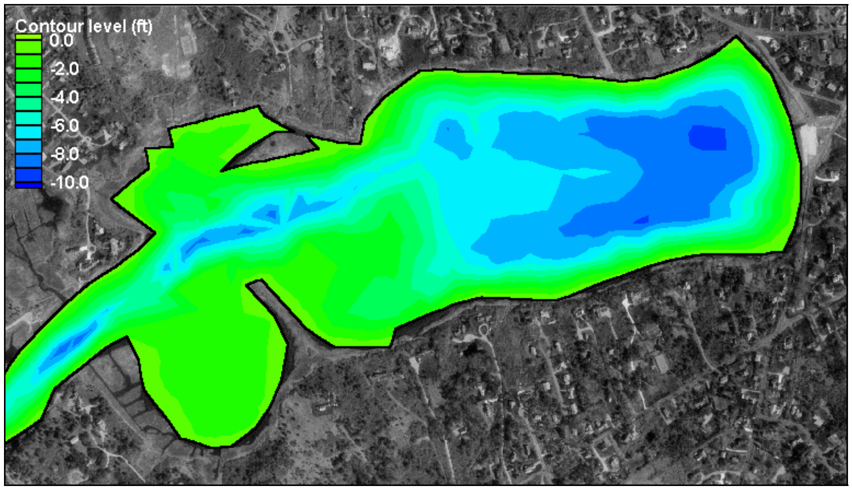


Figure V-20. Depth contour plots of the numerical grid for Oyster Pond (Stage Harbor system) region at 2-foot contour intervals relative to NGVD29 (August 2000 survey, supplemented with 1956 NOAA data).

Bathymetry shown in Figure V-21 for Taylors Pond and Figure V-22 for Sulphur Springs were collected by Applied Science Associates, Inc. in 1998. Applied Coastal took additional measurements in the Sulphur Springs/Buck Creek's system on August 23, 2000 with standard surveying equipment. Elevation measurements were surveyed across seven, shore perpendicular transects at the mouth of Bucks Creek using an automatic digital level sighting on a stadia rod. These measurements were adjusted to NGVD29 from local benchmarks.

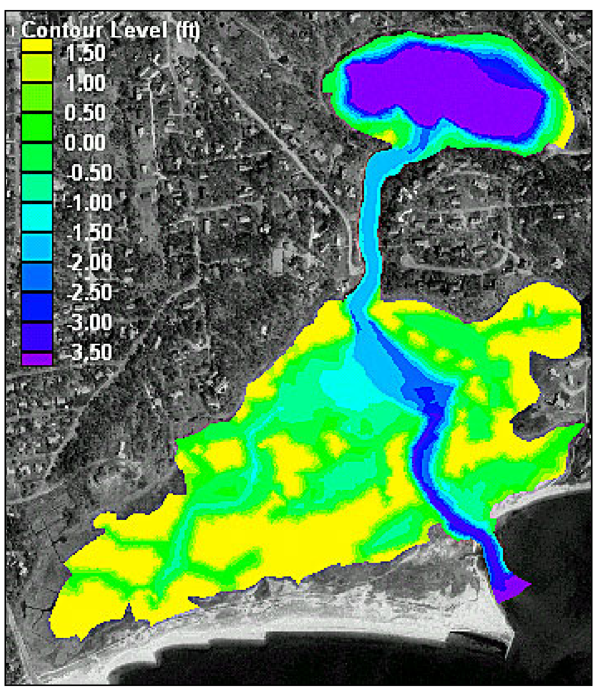


Figure V-21. Depth contour plot of the numerical grid for Taylors Pond and Mill Creek at 0.5 foot contour intervals relative to NGVD29 (August 2000 survey). The yellow to light green indicates marsh plain elevation, and the darker blues to indigo indicate the depth of the creeks and pond basin.

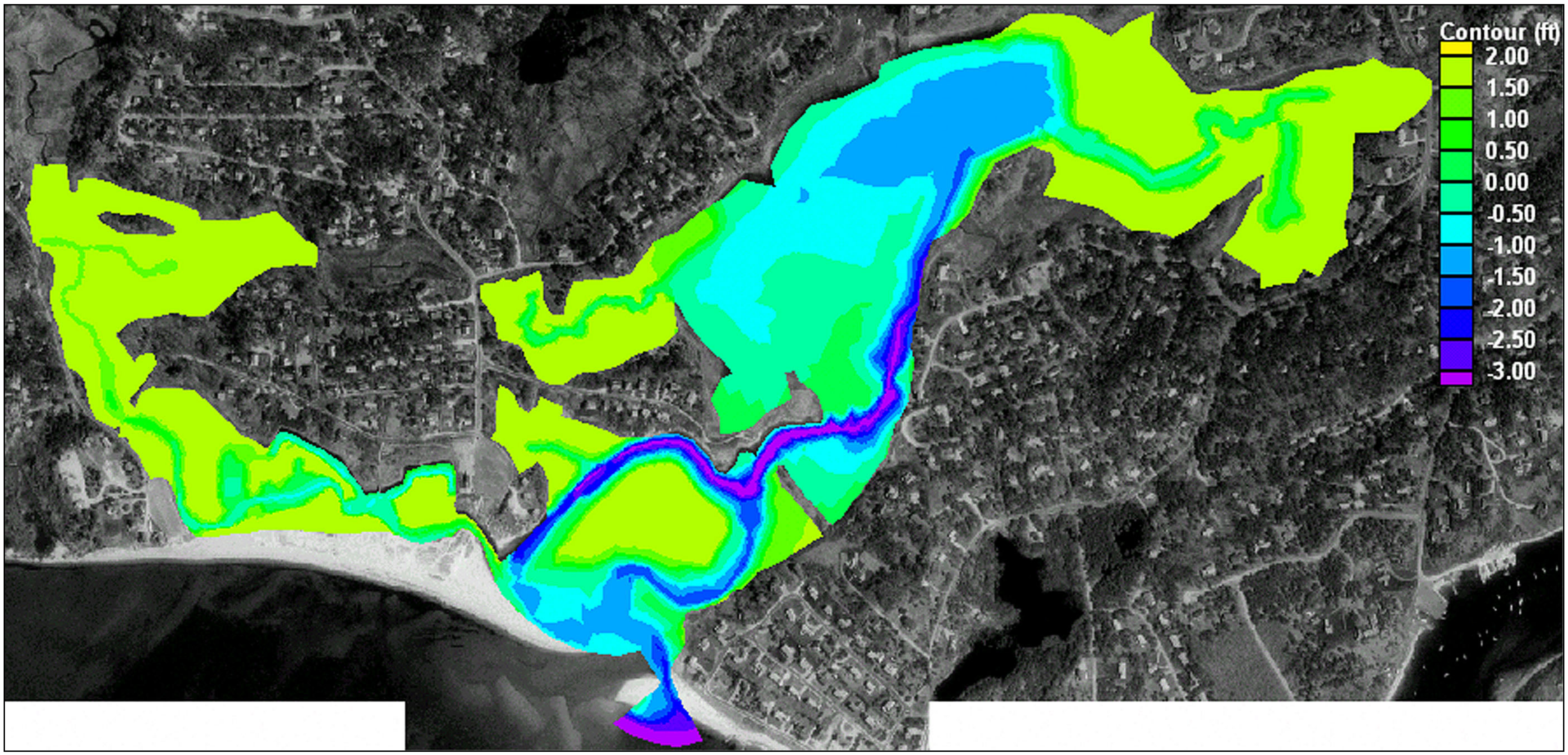


Figure V-22. Depth contour plot of the numerical grid for Sulphur Springs, Bucks Creek, and Cockle Cove Creek at 0.5-foot contour intervals relative to NGVD29 (August 2000 survey). The light green indicates marsh plain elevation, and the darker blues to indigo indicate the depth of the pond basin and creeks.

V.3.1.2.2 Pleasant Bay Region

In the Bassing Harbor system, bathymetry data were collected in September 2000 using an Acoustic Doppler Current Profiler (ADCP) with bottom tracking capability. Measurements were taken in Bassing Harbor, Ryder Cove, lower Frost Fish Creek, and Crows Pond to supplement existing bathymetry collected by Aubrey Consulting, Inc. in 1997. The surveys were conducted in the same manner as the current measurements (described in Section 3), on a small vessel with the ADCP transducer rigidly mounted and a differential GPS collecting position data. In addition to velocity, the bottom tracking process of the ADCP calculates the depth of the seabed below the unit.

The depths were merged with water surface elevations to obtain the vertical elevation of the bottom (z) relative to the NGVD29 vertical datum. The vertical elevations were combined with the x-y horizontal positions from the GPS (Massachusetts Mainland State Plane, 1983), producing xyz files that were input to mapping software. Depth contours for the Bassing Harbor system are depicted in Figure V-23.

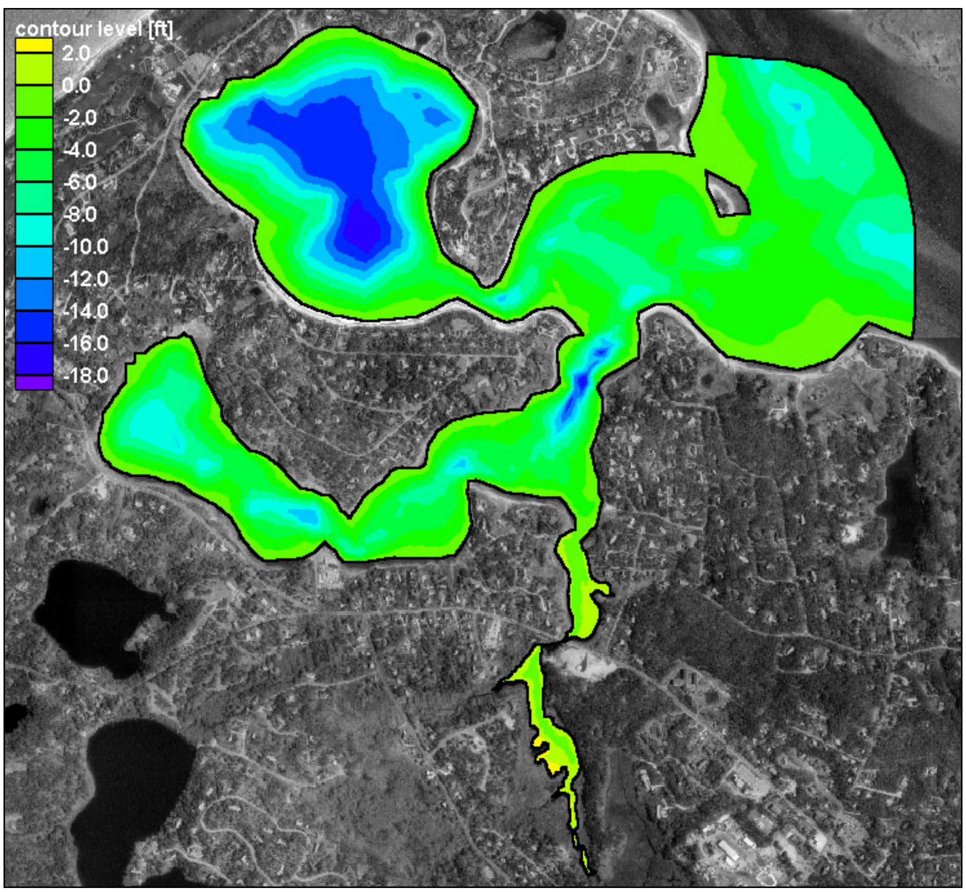


Figure V-23. Depth contour plot of the numerical grid for Bassing Harbor, Ryder Cove, Crows Pond, and Frost Fish Creek at 2-foot contour intervals relative to NGVD29 (September 2000 survey).

Bathymetry was collected in Upper Frost Fish Creek and Muddy Creek from a 16-foot canoe with a hand held differential GPS and stadia rod. Transects were measured cross-creek spaced at 200-300 foot intervals along the length of the creek. Water elevation and GPS x-y positions were simultaneously recorded by hand every 30-50 feet across each transect. The x-y positions were transferred to Massachusetts Mainland State Plane coordinates, and elevation data were combined with tidal water surface elevations to obtain the vertical elevation of the bottom (z) relative to NGVD29. Depth contours were produced for these two systems by the input of the xyz files into mapping software (Figure V-24 and V-25).

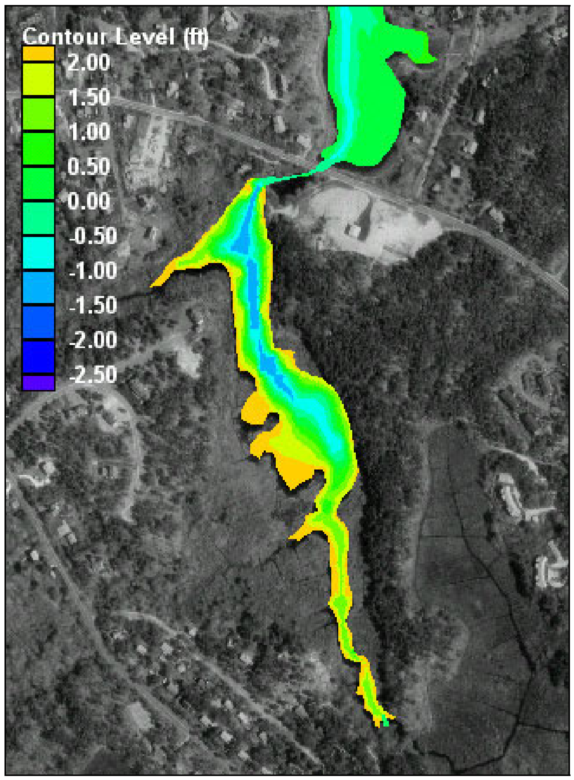


Figure V-24. Depth contour plot of the numerical grid for Frost Fish Creek at 0.5 foot contour intervals relative to NGVD29 (September 2000 survey).

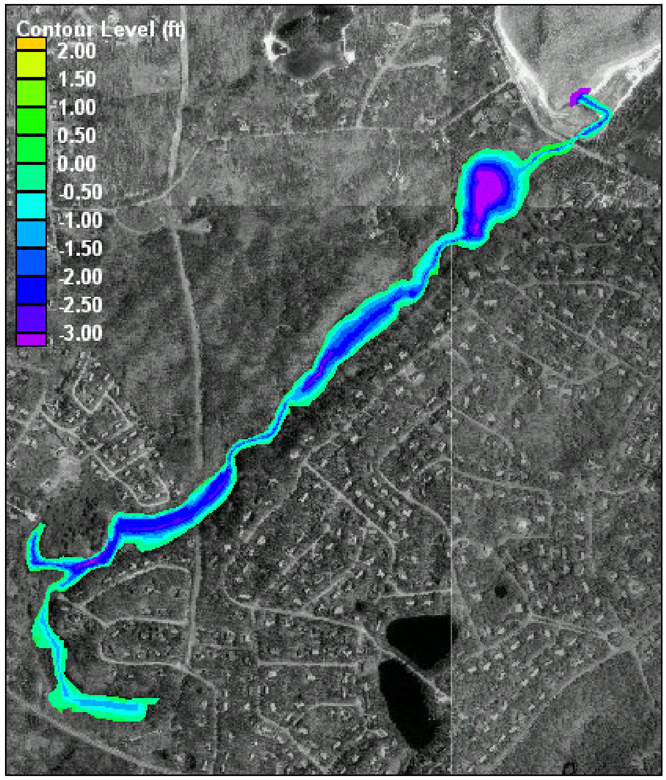


Figure V-25. Depth contour plot of the numerical grid for Muddy Creek at 0.5 foot contour intervals relative to NGVD29 (September 2000 survey).

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 60 minutes, with the ADCP continuously collecting current profiles. This pattern was repeated for an approximate 12.6-hour duration 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.

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 directly downward into the water column, with the sensors located 9 inches below the water surface. The mounting technique assured no flow disturbance due to vessel wake.

The ADCP emits individual acoustic pulses from four angled (at 20° from the vertical) transducers 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 sound scatters (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 particle, 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 13 feet along the transect line. For example, Stage Harbor Inlet was approximately 350 feet wide, resulting in approximately 25-30 independent velocity profiles per transect. The vertical resolution was set to 0.82 ft, or one velocity observation per every 9.8 inches of water depth. The first measurement bin was centered 2.8 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.

V.3.1.3.1 Description of Survey Technique

Stage Harbor System

Current measurements were collected by the ADCP as the vessel navigated repeatedly a series of three (3) pre-defined transect lines through the survey area (Figure V-14). The line-cycles were repeated every hour throughout the survey. The first cycle was begun at 06:48 hours (Eastern Daylight Time, EDT) and the final cycle was completed at 19:36 hours (EDT), for a survey duration of approximately 12.8 hours on August 16, 2000. Each individual transect line was surveyed through a time span of approximately 12.4 hours, for example, transect Line 1 was crossed initially at 06:48 hours and crossed for the final cycle at 19:10 hours.

The transect lines were numbered sequentially 1 through 3, and run in ascending order. These lines were designed to measure as accurately as possible the volume flux through the inlets during a complete tidal cycle. Line 1 ran across the throat of the Stage Harbor Inlet in a southeast-to-northwest direction. Line 2 ran southeast-to-northeast at the throat of Oyster Pond River Inlet, and line 3 ran across the mouth of Mill Pond on the south side of the bridge beginning along the bank on the west flank of the entrance running from southwest-to-northeast along the bridge.

Bassing Harbor System

A series of three (3) pre-defined transect lines in the Bassing Harbor study area were surveyed (Figure V-15) in the same manner. The first cycle was begun at 06:29 hours (Eastern Daylight Time, EDT) and the final cycle was completed at 18:54 hours (EDT), for a survey duration of approximately 12.5 hours on September 1, 2000. Each individual transect line was surveyed through a time span of approximately 12.2 hours.

Line 1 ran across the throat of Bassing Harbor Inlet in a north-to-south direction. Line 2 ran east-to-west at the mouth of Ryder Cove, and line 3 ran across the mouth of Crows Pond from south-to-north.

V.3.1.3.2 Data Processing Techniques

- Data processing consisted of the following:
- Convert raw ADCP (binary) files to engineering units
- 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 engineering 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 2.8 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 total discharge through a channel, Q_t, is computed as the summation of the individual ensemble flows, which is in turn the summation of the product of bin velocity, V, and bin cross sectional area, A, using the velocity measurements through the complete water column, expressed as:

$$Q_t = \sum_{j=1}^{m} \sum_{i=1}^{n} \left(V_i A_i \right)_j$$

where the cross sectional area is the bin depth times the lateral (cross-stream) distance from the previous ensemble profile. The summation occurs over *i*, where *i* represents each individual bin measurement from 1 to *n*, with 1 representing the top (surface) bin and *n* representing the deepest good (near-bottom) bin, and then over *j*, where *j* represents the ensembles along the survey transect.

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 (centered at 3.2 feet). Since the ADCP cannot directly measure the surface velocity, the surface layer current is set to current in the first measured 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.3.2 Discussion of Results

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

V.3.2.1 Stage Harbor System and South Coast Embayments

V.3.2.1.1 Tidal Harmonic Analysis

Figure V-16 shows the tidal elevation for the period July 23 through August 27, 2000 at three locations: offshore Cockle Cove Beach in Nantucket Sound (Location C1), Little Mill Pond (Location C2), and Mill Pond (Location C3). 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 6 feet, and occurred on July 30. The neap (or minimum) tide range was 2.2 feet, occurring August 13th.

Tidal elevations are shown for the next three locations in Figure V-17: Oyster Pond (Location C4), Sulphur Springs (Locations C5), Taylors Pond (Locations C6). Oyster Pond closely follows the tidal elevations for the previous three locations. Sulphur Springs and Taylors Pond drain to Nantucket Sound through shallow, narrow meandering creeks (Bucks Creek and Mill Creek, respectively). The tide signals in these two estuaries, as compared to the offshore signal, show the effects of significant frictional damping. The frictional damping is indicated by the substantial reduction in tide range between the offshore gage and locations within each estuarine system.

Harmonic analyses were performed on the time series from each gage location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-1 presents the amplitudes of the eight largest tidal constituents. The M_2, or the familiar twice-a-day lunar semi-diurnal, tide is the strongest contributor to the signal with an amplitude of 1.79 feet in Nantucket Sound (offshore Cockle Cove). The range of the M_2 tide is twice the amplitude, or 3.58 feet. The diurnal tides, K_1 and O_1, possess amplitudes of approximately 0.3 feet. Other semi-diurnal tides, the S_2 (12.00 hour period) and N_2 (12.66-hour period) tides, strongly contribute with amplitudes of 0.19 feet and 0.52 feet, respectively. The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 period for the M_6), results from frictional attenuation of the M_2 tide in shallow water. The M_4 and M_6 have a very small amplitude in the offshore gage (about 0.13 feet and 0.08 feet. The M_{sf} is a lunarsolar fortnightly constituent with a period of approximately 14 days, and is the result of the periodic conjunction of the sun and moon.

Table V-1. Tidal Constituents, Stage Harbor and South Coast Embayments of Chatham, July-August 2000								
	Amplitude (feet)							
Constituent	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	1.79	0.13	0.08	0.19	0.52	0.35	0.29	0.14
Mill Pond	1.86	0.09	0.08	0.19	0.52	0.35	0.29	0.14
Little Mill Pond	1.85	0.09	0.08	0.19	0.52	0.35	0.29	0.12
Oyster Pond	1.80	0.07	0.04	0.18	0.47	0.35	0.30	0.13
Sulphur Springs	1.17	0.21	0.02	0.12	0.25	0.29	0.27	0.16
Taylors Pond	1.60	0.11	0.04	0.14	0.40	0.32	0.29	0.18

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-26.

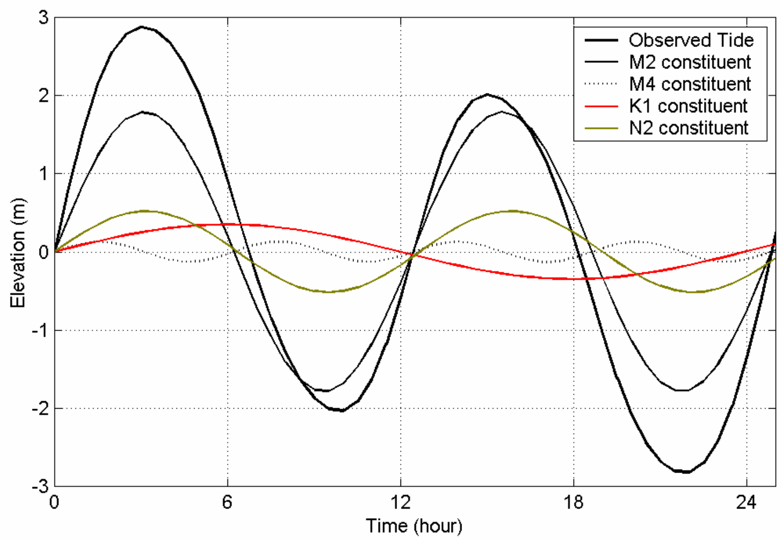


Figure V-26. Example of observed astronomical tide as the sum of its primary constituents. Constituents for offshore Cockle Cove Beach were used in this example.

Table V-1 also shows how the constituents vary as the tide propagates into the estuaries. Note the reduction in the M_2 amplitude between Nantucket Sound and the upper reaches of Sulphur Springs and Taylors Pond. The loss of amplitude with distance from the inlet is described as tidal attenuation. Frictional mechanisms dissipate tidal flow energy, resulting in a reduction of the height of the tide. Usually, frictional damping is most evident as a decrease in the amplitude of M_2 constituent. 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 is seen in the tide at Sulphur Springs, where there is a significant growth of the M_4 constituent from offshore. In contrast, an apparent reduction in the M_2 constituent occurs as the tide propagates from Nantucket Sound in the Stage Harbor System (especially into Mill and Little Mill Ponds). This growth in the M_2 constituent is likely due to a transfer of energy from other constituents.

Table V-2 presents the phase delay of the M_2 tide at all tide gage locations compared to the offshore gage at Cockle Cove Beach. 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. The delays in Mill Pond and Little Mill Pond are nearly equal, as a result of their proximity to each other. More significant damping is seen in another area of Stage Harbor, at Oyster Pond, with a delay of 36.5 minutes, which is more than twice the delay of the Mill Ponds. This difference is primarily due to the flooding of tidal flats in the Oyster Pond River/Oyster Pond system, as well as the shallow constricted channel of Oyster Pond River. Similar to the behavior of the amplitude of M_2 tide, the largest delay of one hour occurs in Sulphur Springs (location C5) relative to offshore. Taylors Pond also exhibits a significant delay of almost a half hour.

Analysis of the data shows that the Oyster Pond, Sulphur Springs, and Taylors Pond systems, with shallow intertidal flats, expansive salt marsh regions, and winding channels and creeks, distorts the tide significantly relative to offshore Cockle Cove Beach, in Nantucket Sound. This distortion of the tide is evidenced as reduction in M_2 tide amplitude and M_2 phase delays.

Table V-2. M_2 Tidal Attenuation Stage Harbor and South Coast Embayments, Chatham, July-August 2000 (Delay in minutes relative to offshore of Cockle Cove Beach).

TDR Location	Delay (minutes)
Offshore	--
Little Mill Pond	12.89
Mill Pond	13.95
Taylors Pond	27.39
Oyster Pond	36.47
Sulphur Springs	59.22

For Oyster Pond and the South Coast Embayments flow restrictions modify the duration of the ebb and flood tide stages to a longer ebb duration (a slower, gradual draining of the estuary) and a briefer flood tide (Figure V-27). Nantucket Sound tides have a flood duration of approximately 7 hours, with an ebb duration of approximately 5 hours. The top plot of Figure V-27 demonstrates the modification to the tide signal in Oyster Pond relative to the remainder of the Stage Harbor complex and the offshore tide. In the lower two plots, the shorter duration of the flood tide in Sulphur Springs and Taylors Pond is more clearly seen with a flood duration of approximately 4 hours and an ebb of approximately 8 hours in duration.

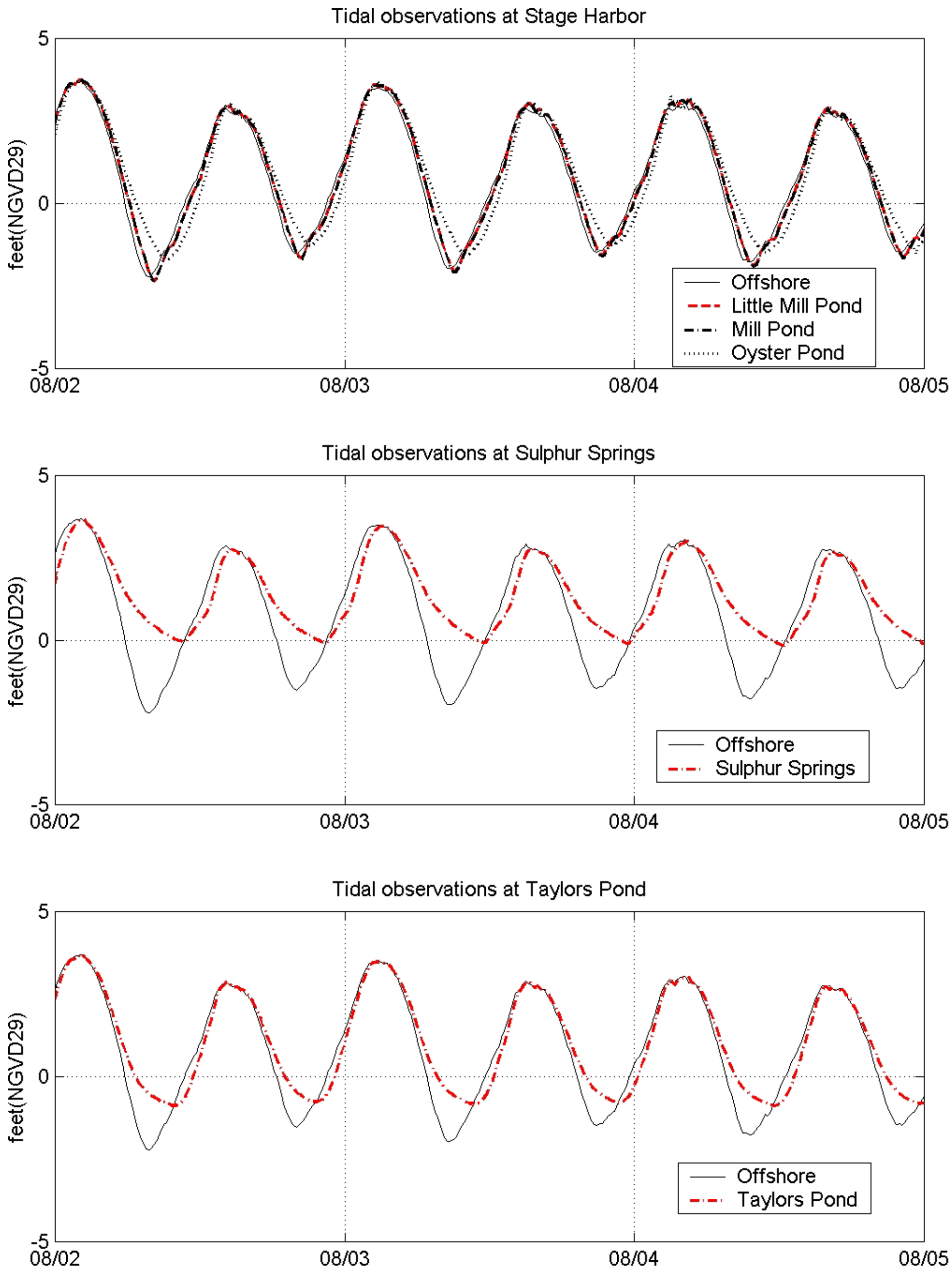


Figure V-27. Water elevation variations for a 3-day period in the Stage Harbor, Sulphur Springs, and Taylors Pond systems. Each plot depicts the Nantucket sound signal (offshore) overlaid with measurements obtained in the inland estuaries. Notice the reduced amplitude as well as the delay in times of high- and low- tide relative to offshore due to frictional damping through the systems.

For locations where the flood tide is shorter in duration than the ebb tide (e.g., Sulphur Springs), currents during the flood stage will be greater than during the ebb, since approximately the same amount of water volume must enter the system on the flood as leaves on the ebb. An example of this phenomenon is the Oyster Pond River inlet, where measured tidal currents were greater during the flood cycle than the ebb cycle. Estuarine systems of this type are referred to as 'flood-dominant'. Flood dominant systems tend to be sediment traps, where sediments transported into the system during flood tide often settle, causing long-term accretion or shoaling. The characteristics of the tidal signal can be utilized to indicate whether an estuarine system is likely to be a sediment trap for fine-grained materials, since trapping of organic sediments is often related to increased benthic flux during the summer months.

In addition to the tidal analysis, the data were further evaluated to determine the importance of tidal versus non-tidal processes to changes in water surface elevation. These other processes include wind forcing (set-up or set-down) within the estuary, as well as sub-tidal oscillations of the sea surface. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large compared to tidal flow. The results of an analysis to determine the energy distribution (or variance) of the original water elevation time series for Stage Harbor and the South Coast Embayments is presented in Table V-3, compared to the energy content the astronomical 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.

Table V-3. Percentages of Tidal versus Non-Tidal Energy, Stage Harbor and South Coast Embayments, July to August 2000

TDR Location	Total Variance ($ft^2 \cdot sec$)	Total(%)	Tidal (%)	Non-tidal (%)
Offshore	1.929	100%	98.9	1.1
Little Mill Pond	2.042	100%	98.6	1.4
Mill Pond	2.057	100%	98.7	1.3
Oyster Pond	1.909	100%	98.8	1.2
Sulphur Springs	0.874	100%	97.3	2.7
Taylors Pond	1.530	100%	98.7	1.3

Table V-3 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 ponds. The analysis also shows that tides are responsible for almost 99% of the water level changes in the Stage Harbor system and South Coast Embayments; wind effects in these data sets were negligible. In Sulphur Springs, tides are still responsible for over 97% of the elevation variation, with inputs from other sources accounting for the remaining energy. 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 a growth of residual forces.

V.3.2.1.2 Current Measurements

Current measurements in the Stage Harbor region, surveyed on August 16, 2000, 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 Stage Harbor inlet, and how it was divided between the Oyster Pond River inlet, and the mouth of Mill Pond at hourly intervals. The current measurements observed during the flood and ebb tides at each constriction can be seen in FiguresV-28 throughV-33. 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 Stage Harbor inlet, positive along-channel is in the direction of northeast, and positive cross-channel is in the direction of southeast. 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).

Tidal currents through the Stage Harbor inlet reached maximum speeds of approximately 3.3 ft/sec directed out of the estuary. During periods of maximum currents (flood and ebb) the inlet tidal flows were vertically coherent, with negligible stratification (FiguresV-28 andV-29) and ran parallel to the main navigation channel. During slack-water periods, there was an indication of mildly stratified flows, evidenced by an abrupt change in horizontal current direction in the water column. Maximum volume flux through the Stage Harbor inlet during flood tide was 5,750 ft^3/sec, while the maximum flux during ebb conditions was slightly more, -7,250 ft^3/sec. These flow measurements are consistent with the tide plots in Figure V-28, which show a longer duration flood stage relative to the duration of the ebb stage of the tide at Stage Harbor Inlet.

The channel through the Oyster Pond River inlet is along the north bank of the river, on average resulting in stronger currents along the northern side of the inlet reaching a maximum of approximately 2.0 ft/sec. During flood conditions, the currents are focused through the channel (Figure V-30), compared to ebb conditions where the flow is distributed relatively evenly across the inlet (Figure V-31). Volume flow rates reached a maximum of 2,310 ft^3/sec during flood conditions, and a maximum ebb flow rate of –1,790 ft^3/sec at the mouth of the Oyster Pond River. These flow measurements are consistent with the tide plots in Figure V-27, which show a longer duration ebb stage relative to the duration of the flood stage of the tide at Oyster Pond.

The third transect, measured south of the bridge at the mouth of Mill Pond showed the most variability in the along-channel tidal currents. The bridge begins approximately 210 feet along the transect (denoted by black line across top panel in Figure V-32), however the flow under the bridge is attenuated at the surface by wooden boards across the first two sets of pilings on the western side. As a result, the tidal currents are focused towards the center of the bridge through the largest opening, which is apparent during the ebb tide in Figure V-33, with a maximum current of approximately 2.0 ft/sec. At the mouth of Mill Pond, the maximum flood flow rate was 870 ft^3/sec, and the maximum ebb flow rate was –890 ft^3/sec. Again, these flow measurements are consistent with the tide plots in Figure V-27, which show a longer duration flood stage relative to the duration of the ebb stage of the tide at Mill Pond.

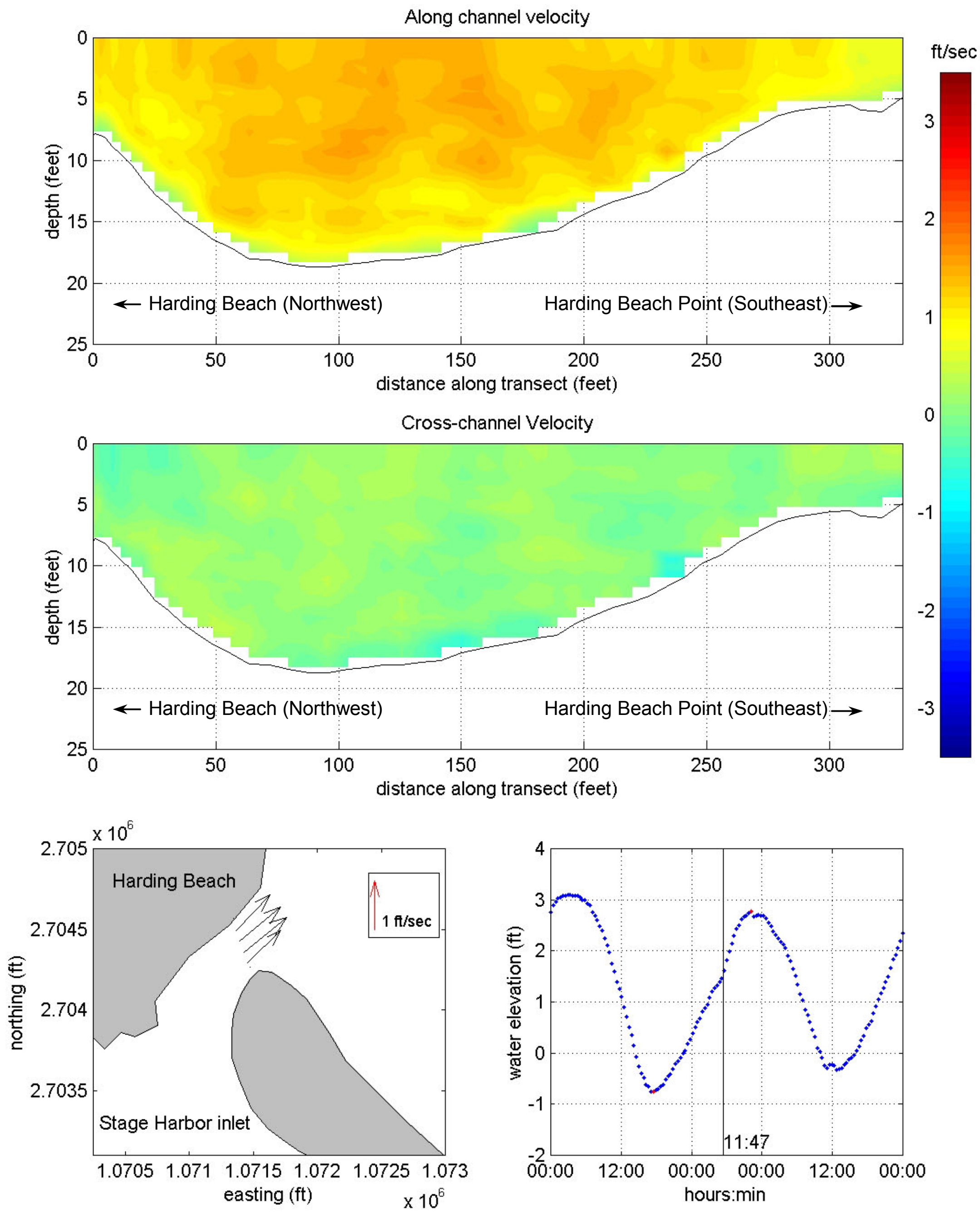


Figure V-28. Color contour plots of along-channel and cross-channel velocity components for transect line 1 run northwest-to-southeast across the Stage Harbor inlet measured at 11:47 on August 16, 2000 during the flood tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

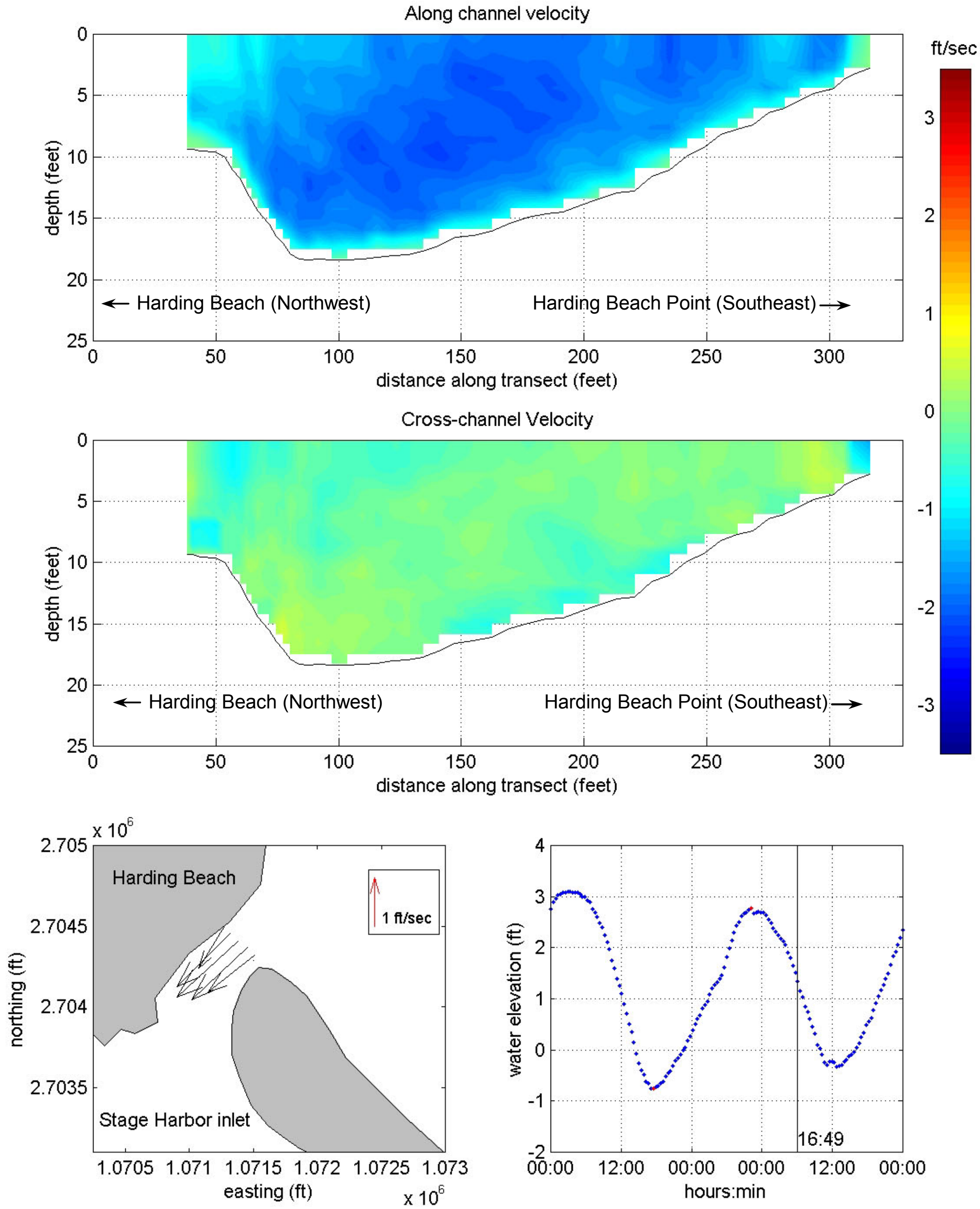


Figure V-29. Color contour plots of along-channel and cross-channel velocity components for transect line 1 run northwest-to-southeast across the Stage Harbor inlet measured at 16:49 on August 16, 2000 during the ebb tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

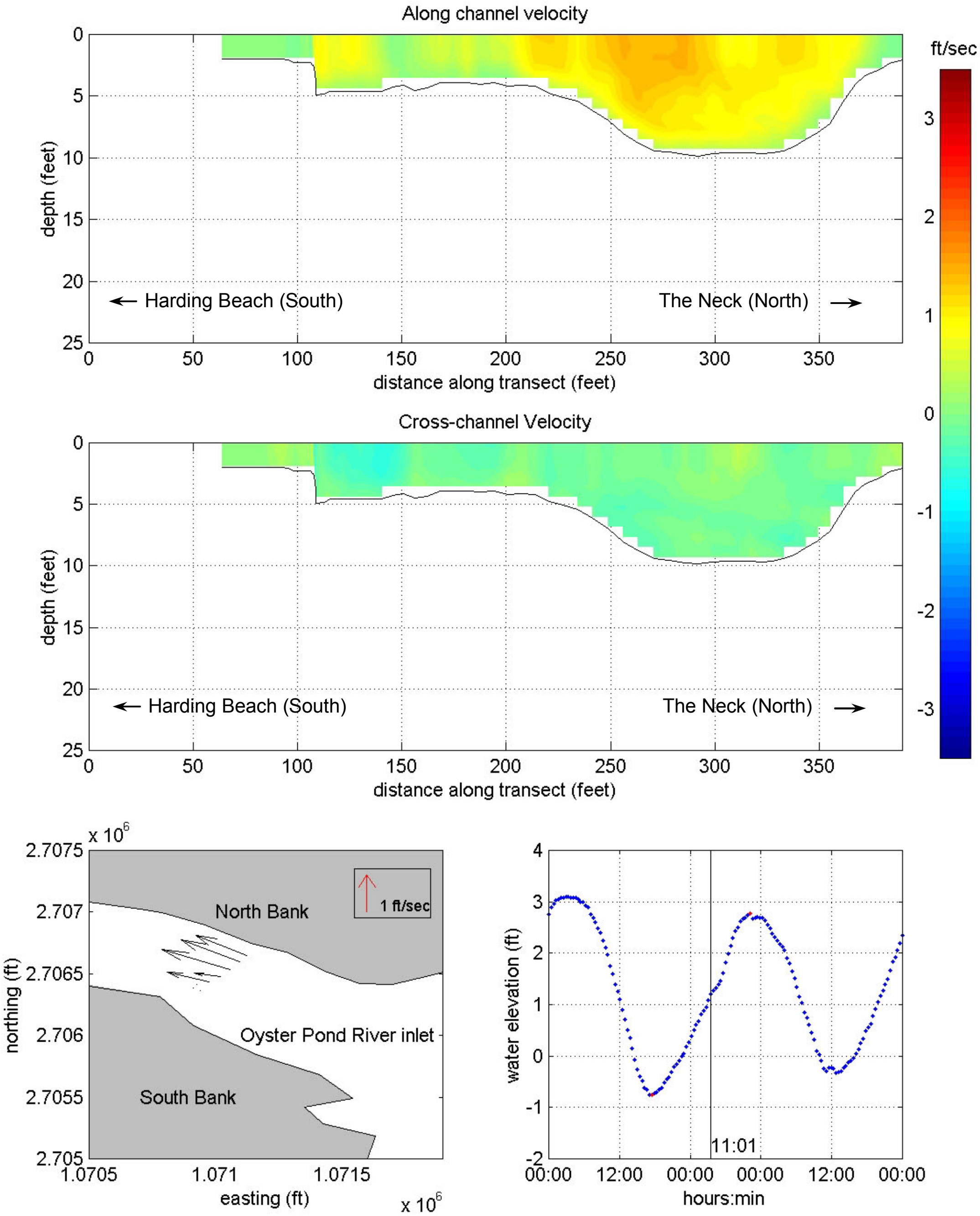


Figure V-30. Color contour plots of along-channel and cross-channel velocity components for transect line 2 run south-to-north across the Oyster Pond River inlet measured at 11:01 on August 16, 2000 during the flood tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

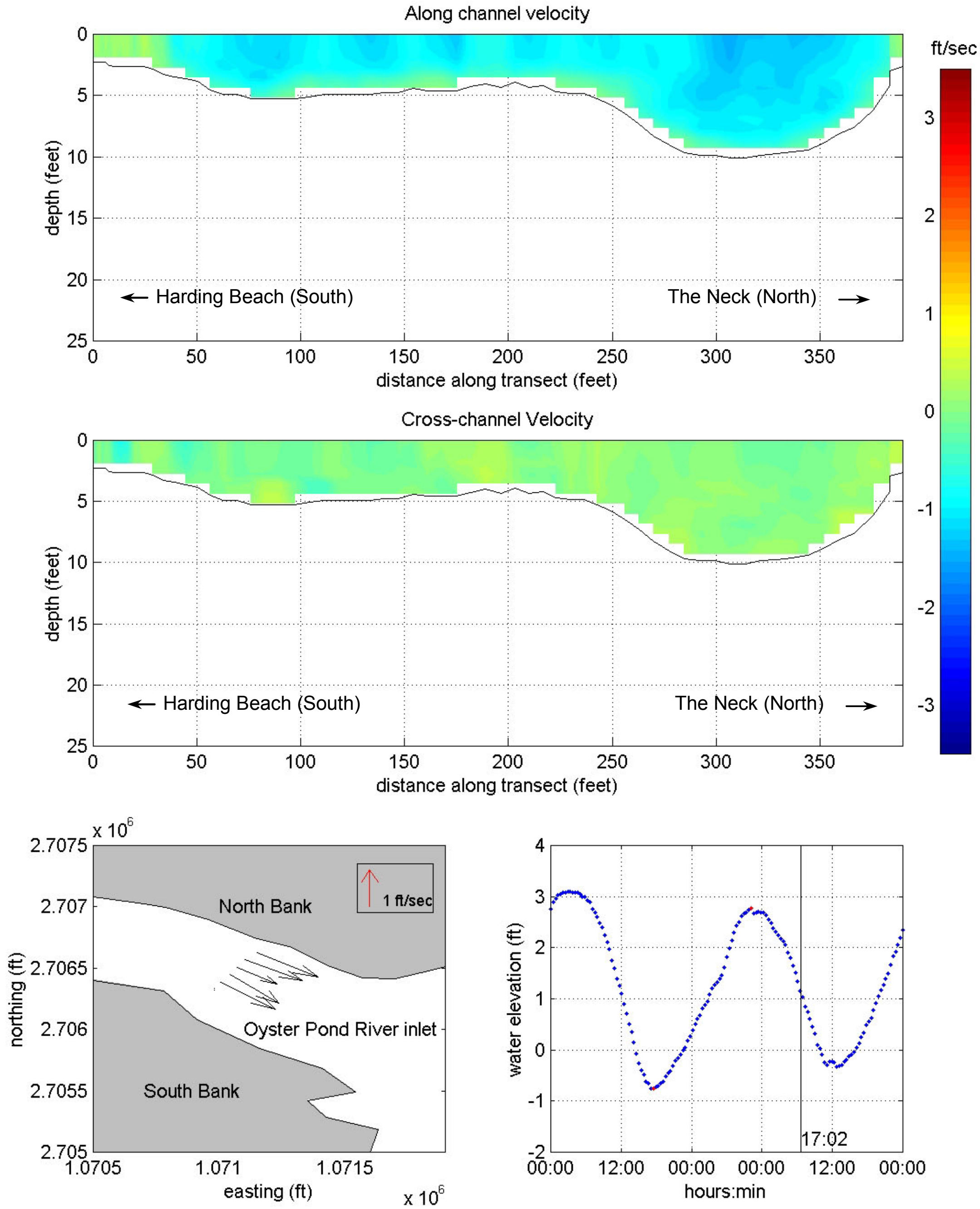


Figure V-31. Color contour plots of along-channel and cross-channel velocity components for transect line 2 run south-to-north across the Oyster Pond River inlet measured at 17:02 on August 16, 2000 during the ebb tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

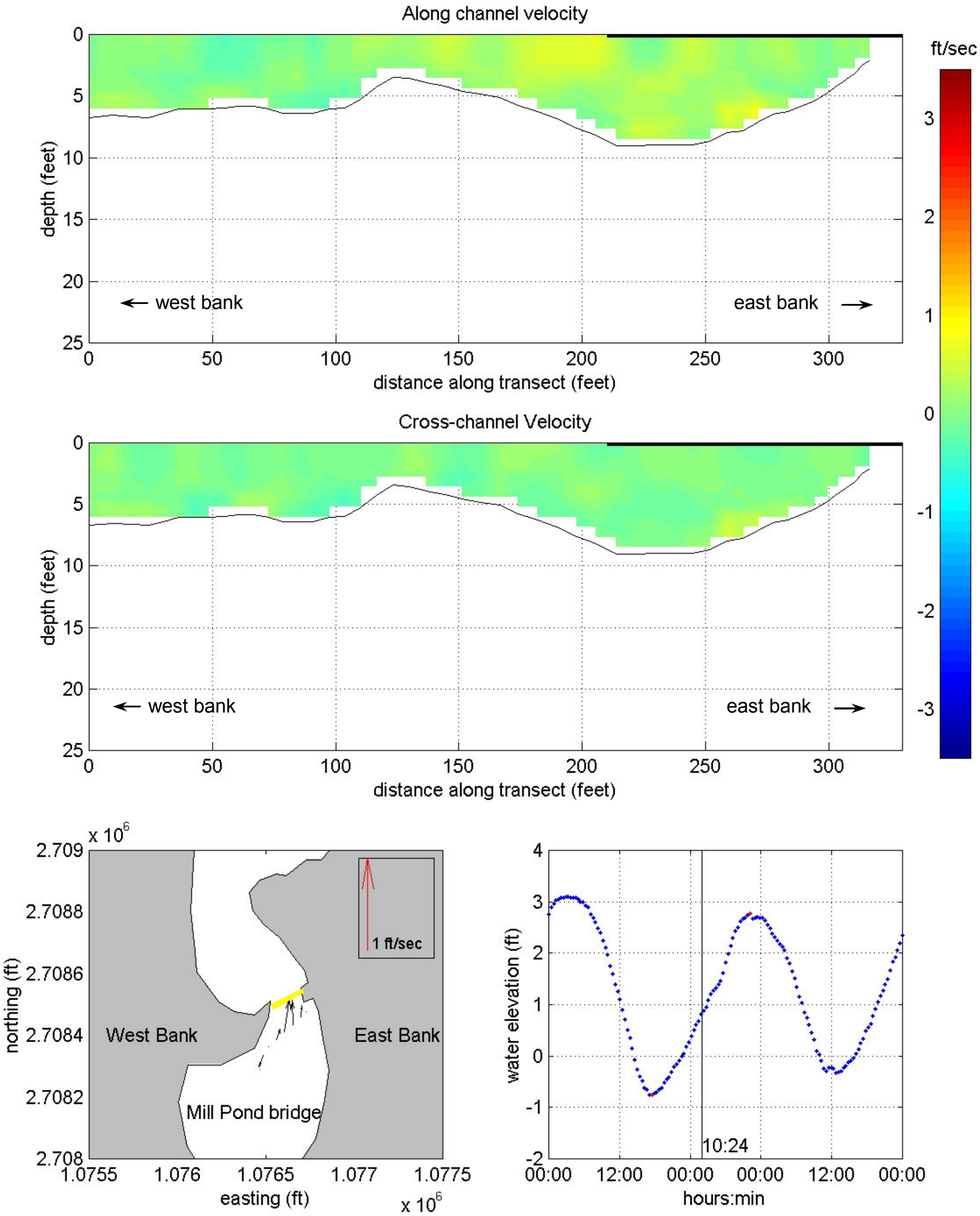


Figure V-32. Color contour plots of along-channel and cross-channel velocity components for transect line 3 run southwest-to-northeast across the south side of Mill Pond bridge measured at 10:24 on August 16, 2000 during the flood tide. The location of the bridge is indicated by a heavy black line at 0 ft depth from 210 to 350 feet along the transect in the top and middle panels, and a yellow line in the lower left panel.

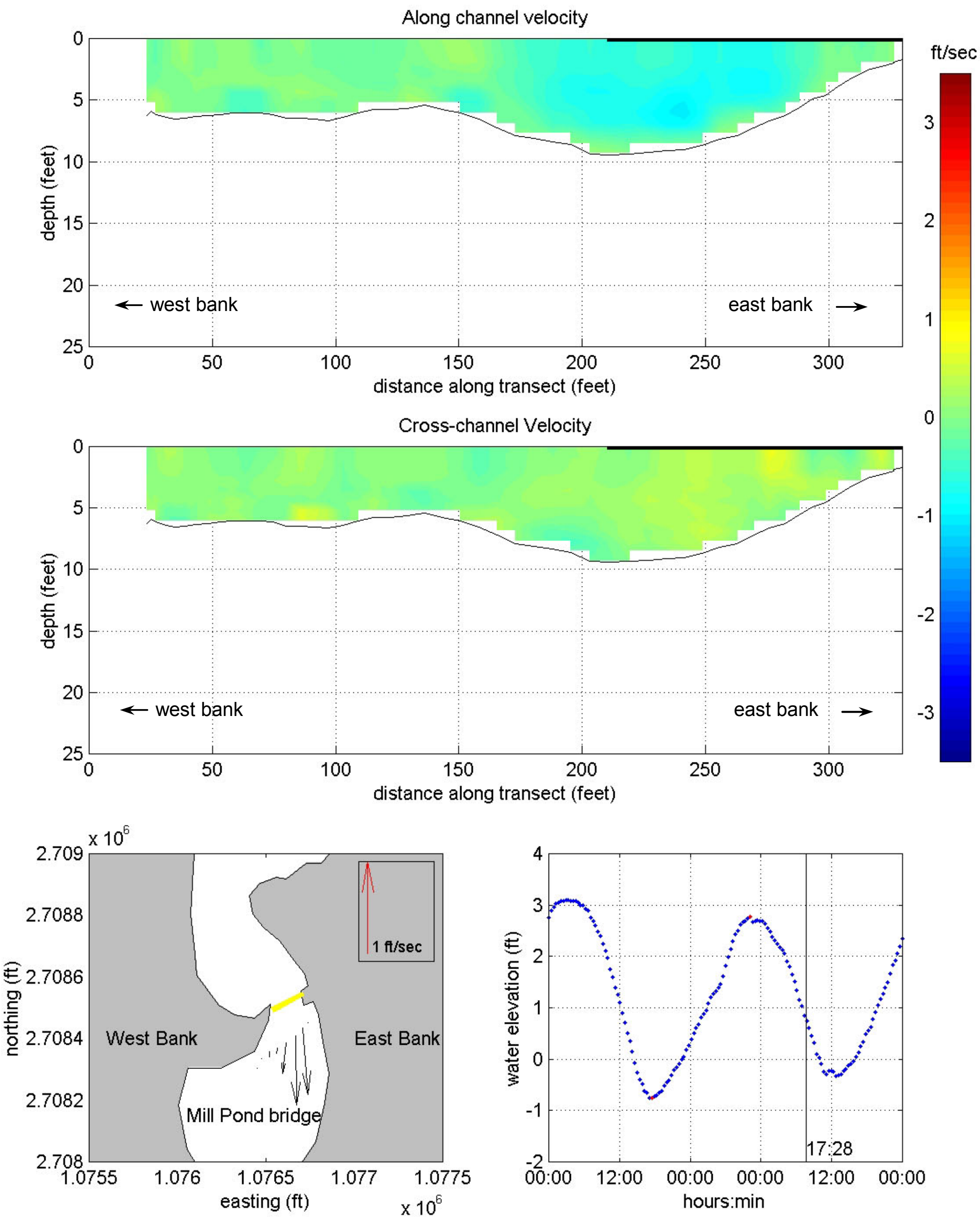


Figure V-33. Color contour plots of along-channel and cross-channel velocity components for transect line 3 run southwest-to-northeast across the south side of Mill Pond bridge measured at 17:28 on August 16, 2000 during the ebb tide. The location of the bridge is indicated by a heavy black line at 0 ft depth from 210 to 350 feet along the transect in the top and middle panels, and a yellow line in the lower left panel.

V.3.2.2 Pleasant Bay Region

V.3.2.2.1 Tidal Harmonic Analysis

Figure V-18 shows tidal elevations for the period August 22 through September 27, 2000 at five locations: Chatham Yacht Club (location P1 in Figure V-15), Crows Pond (location P2), Ryder Cove (location P3), Frost Fish Creek (location P4), and Muddy Creek (location P5). The curves have a predominant 12.42-hour variation around the lunar semi-diurnal (twice-a-day), or M_2, tidal constituent. The maximum (spring) tide range occurred on August 28 with a magnitude of approximately 5.5 feet, and the minimum (neap) tide range of approximately 2.6 feet occurred on September 8. Muddy Creek and Frost Fish Creek are long, narrow estuaries that are constricted at their entrances to the larger estuaries by culverts. The flow control features (i.e., culverts and weirs) of these two systems substantially attenuate the tidal signal.

The amplitudes of the eight largest tidal constituents from the tidal harmonic analysis are presented in Table V-4. The strongest contributor to the signal is the M_2 tide with an amplitude of 1.81 feet in Pleasant Bay (Chatham Yacht Club) and a range (twice the amplitude) of 3.62 feet. The diurnal tides, K_1 and O_1, possess amplitudes of approximately 0.25 feet. Other semi-diurnal tides, the S_2 (12.00 hour period) and N_2 (12.66-hour period) tides, contribute with amplitudes of 0.23 feet and 0.36 feet, respectively.

Table V-5 presents the phase delay of the M_2 tide at all of the tide gage locations in the Pleasant Bay region compared to the Chatham Yacht Club gage. This comparison reinforces the lack of attenuation in Bassing Harbor, Ryder Cove and Crows Pond relative to Pleasant Bay. The flow restriction by culverts in Frost Fish Creek and Muddy Creek result in significant phase delays of the M_2 tide of over 2 hours in both systems. Muddy creek and the upper reaches of Frost Fish Creek drain to the larger estuaries through culverts, which inhibits the tidal flow, resulting in damping of the tidal signal.

Although the Chatham Yacht Club gage originally was intended as the forcing tide for both the Bassing Harbor and Muddy Creek systems, the tide phase at the Yacht Club indicated that it would be inappropriate to act as the model forcing for the Bassing Harbor system. Instead, the Ryder Cove data was utilized as the forcing data for Bassing Harbor. A more complete description of this approach is discussed in the modeling section.

Table V-4. Tidal constituents for Bassing Harbor/Muddy Creek, Chatham, August-September 2000

	Amplitude (feet)							
Constituent	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
Chatham Yacht Club	1.811	0.277	0.035	0.231	0.362	0.248	0.273	0.244
Ryder Cove	1.862	0.203	0.060	0.242	0.374	0.245	0.275	0.245
Crows Pond	1.860	0.228	0.049	0.243	0.377	0.250	0.275	0.256
Frost Fish Creek	0.161	0.036	0.004	0.027	0.041	0.052	0.064	0.114
Muddy Creek	0.296	0.051	0.003	0.037	0.055	0.077	0.088	0.205

Table V-5. M$_2$ Tidal Attenuation Bassing Harbor/Muddy Creek systems August-September 2000 (Delay in minutes relative to Chatham Yacht Club)	
TDR location	Delay (minutes)
Ryder Cove	-15.95
Crows Pond	-12.12
Chatham Yacht Club	--
Frost Fish Creek	122.13
Muddy Creek	141.81

Table V-6 shows the relative energy of tidal versus non-tidal processes at the tide gage locations in the Pleasant Bay region. The signal variance (or energy) of each time series was computed to calculate the percentages. Tides are responsible for approximately 97% of the water level changes at Chatham Yacht Club, Ryder Cove, and Crows Pond; wind effects in the data for these three estuaries were negligible. In the creeks, tides are only responsible for approximately 70% of the elevation variation, with inputs from other sources accounting for the remaining energy. This relative increase in non-tidal energy within the creeks is likely due to the decrease in tidal energy as a result of attenuation of the flow through the constrictions rather than a growth of residual forces. Variations in the water surface elevation of the creeks are also affected by freshwater discharge into the systems.

Table V-6. Percentages of tidal versus non-tidal energy Bassing Harbor/Muddy Creek August-September 2000.				
TDR location	Total Variance ($ft^2 \cdot sec$)	Total(%)	Tidal (%)	Non-tidal (%)
Chatham Yacht Club	1.998	100%	97.0	3.0
Ryder Cove	2.082	100%	97.1	2.9
Crows Pond	2.092	100%	97.0	3.0
Frost Fish Creek	0.037	100%	70.7	29.3
Muddy Creek	0.105	100%	73.9	26.1

V.3.2.2.2 Current Measurements

Cross-channel current measurements were surveyed through a tidal cycle in the Bassing Harbor system on September 1, 2000 to resolve spatial and temporal variations in tidal current patterns. The survey was designed to observe tidal flow through Bassing Harbor inlet, and the division of flow between Crows Pond and Ryder Cove/Frost Fish Creek at hourly intervals. FiguresV-34 throughV-39 show color contours of current measurements observed during the flood and ebb tides at each of the three transects. 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 Bassing Harbor inlet, positive along-channel flow is westerly, and positive cross-channel flow is moving north. 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 that the survey was taken by a vertical line through the water elevation curve.

The complex geometry of Bassing Harbor inlet results in an unequal distribution of currents across the entrance. During flood tide, maximum currents were focused through the primary channel in a west-southwest direction (Figure V-34), with additional flow across the southern portion of the inlet. The curvature of the shoreline of Bassing Harbor and the presence of Fox Hill appears to modify the flow out of the harbor on the ebb tide. In Figure V-35 (lower left panel), ebb currents are distributed more evenly across the inlet, and flow almost perpendicular to the direction of maximum flood currents. In a system with a linear channel, dominant flood and ebb currents flow along the same line, but in opposite directions (180°). In Bassing Harbor, the curvature of the channel with the shoreline complicates the rotation of currents into along-channel and cross-channel components. As a result of the rotation based on the assumption of a linear channel, the color contours in Figure V-35 (top and middle panel) show that a large portion of the tidal flow energy in contained in the cross-channel component during the ebb tide. Tidal currents through the Bassing Harbor inlet reached a maximum of approximately 3.2 ft/sec directed out of the harbor during ebb tide. The maximum volume flow rate through the Bassing Harbor inlet was 4,880 ft^3/sec during the flood tide, and the maximum ebb volume flow rate was –1,670 ft^3/sec.

The second transect was measured from east-to-west across the mouth of Ryder Cove, which also connects to Frost Fish Creek. The dominant flood tidal currents were focused through the channel at a maximum velocity of approximately 2.3 ft/sec (Figure V-36). The transect was begun to the south of a pier on the east side of the inlet, which may account for the presence of currents directed out of the estuary along the east bank. During the ebb tide, the current profiles were moderately stratified in the channel (Figure V-37), with stronger out-estuary velocities in the upper layers of the water column, and weaker in-estuary flow near the bottom of the channel. At the entrance to Ryder Cove/Frost Fish Creek, the volume flow rate reached a maximum during flood tide of 1,800 ft^3/sec, and a maximum of -990 ft^3/sec during the ebb tide.

The geometry of the shoreline at the mouth of Crows Pond shifts the cross channel position of the strongest currents from one side to the other during the flood-ebb tidal cycle. At the wide entrance to Crows Pond, the strongest flood currents into the estuary occur on the north side (Figure V-38), and the strongest ebb currents out of the estuary occur on the south side of the channel (Figure V-39). Maximum along-channel velocities were observed during the ebb tide at magnitudes of 3.2 ft/sec, but strong cross-channel components of ebb tidal currents are also seen as a result of rotation of currents relative to a linear channel. Volume flow rates reach a maximum of 1,600 ft^3/sec during the flood tide, and a maximum of -1,240 ft^3/sec during the ebb tide at the entrance to Crows Pond.

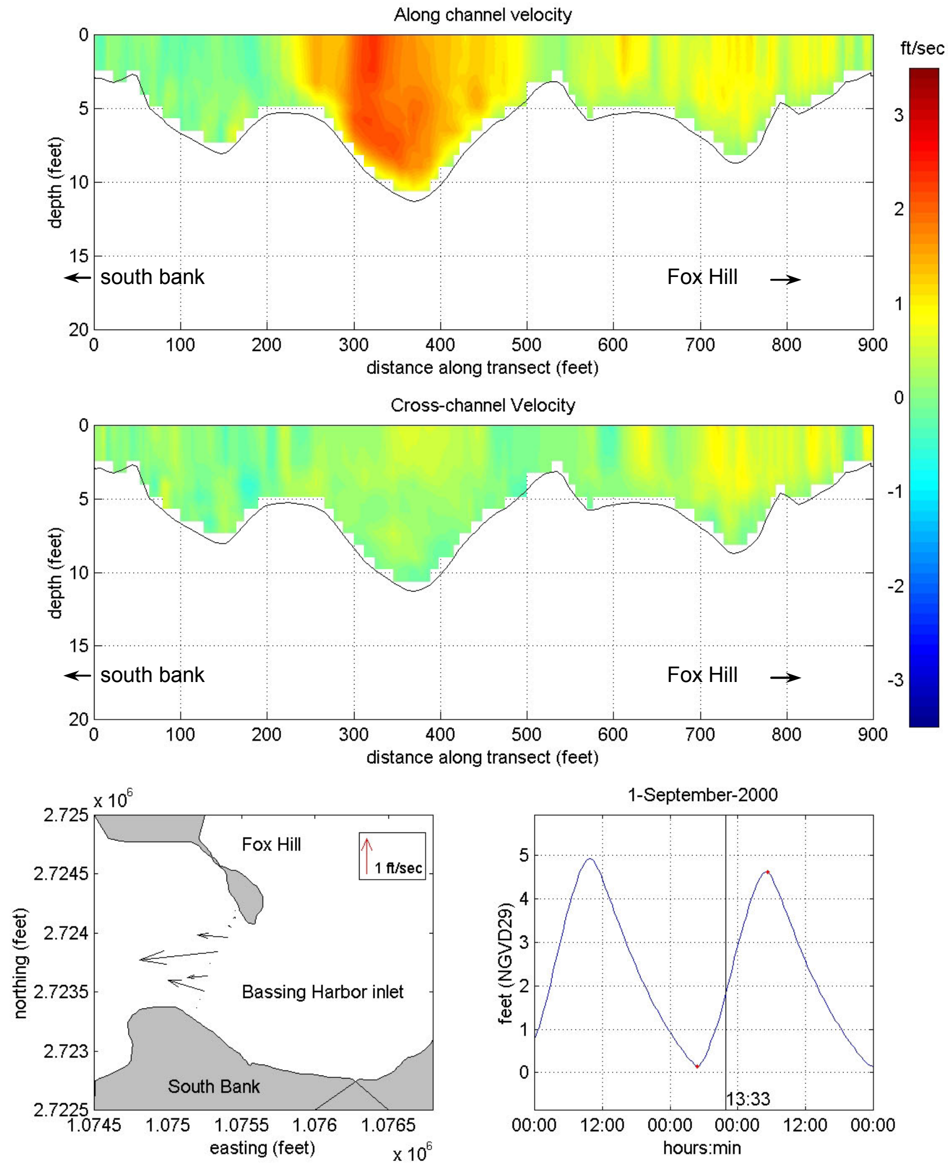


Figure V-34. Color contour plots of along-channel and cross-channel velocity components for transect line run south-to-north across the Bassing Harbor inlet measured at 13:33 on September 1, 2000 during the flood tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

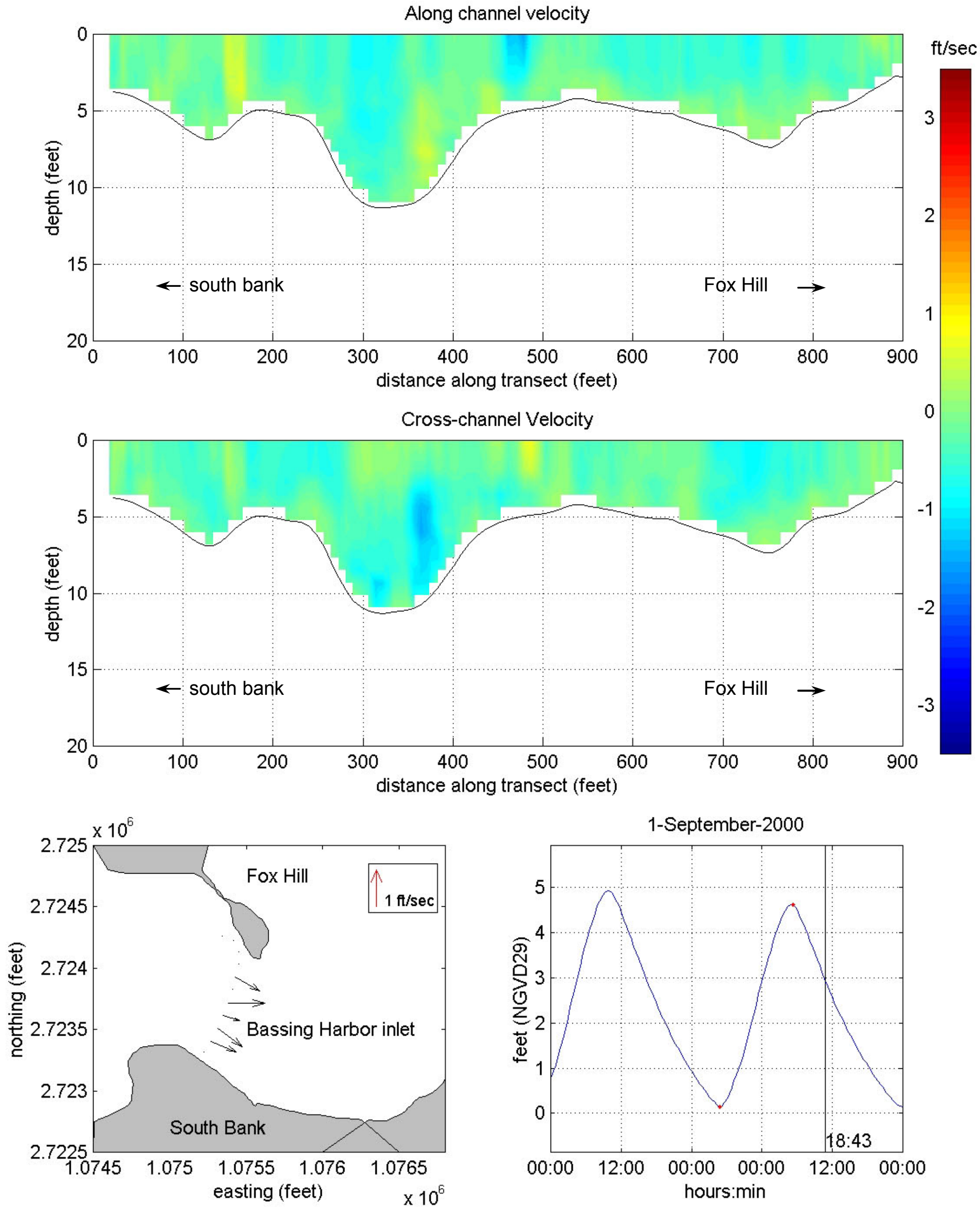


Figure V-35. Color contour plots of along-channel and cross-channel velocity components for transect line run south-to-north across the Bassing Harbor inlet measured at 18:43 on September 1, 2000 during the ebb tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

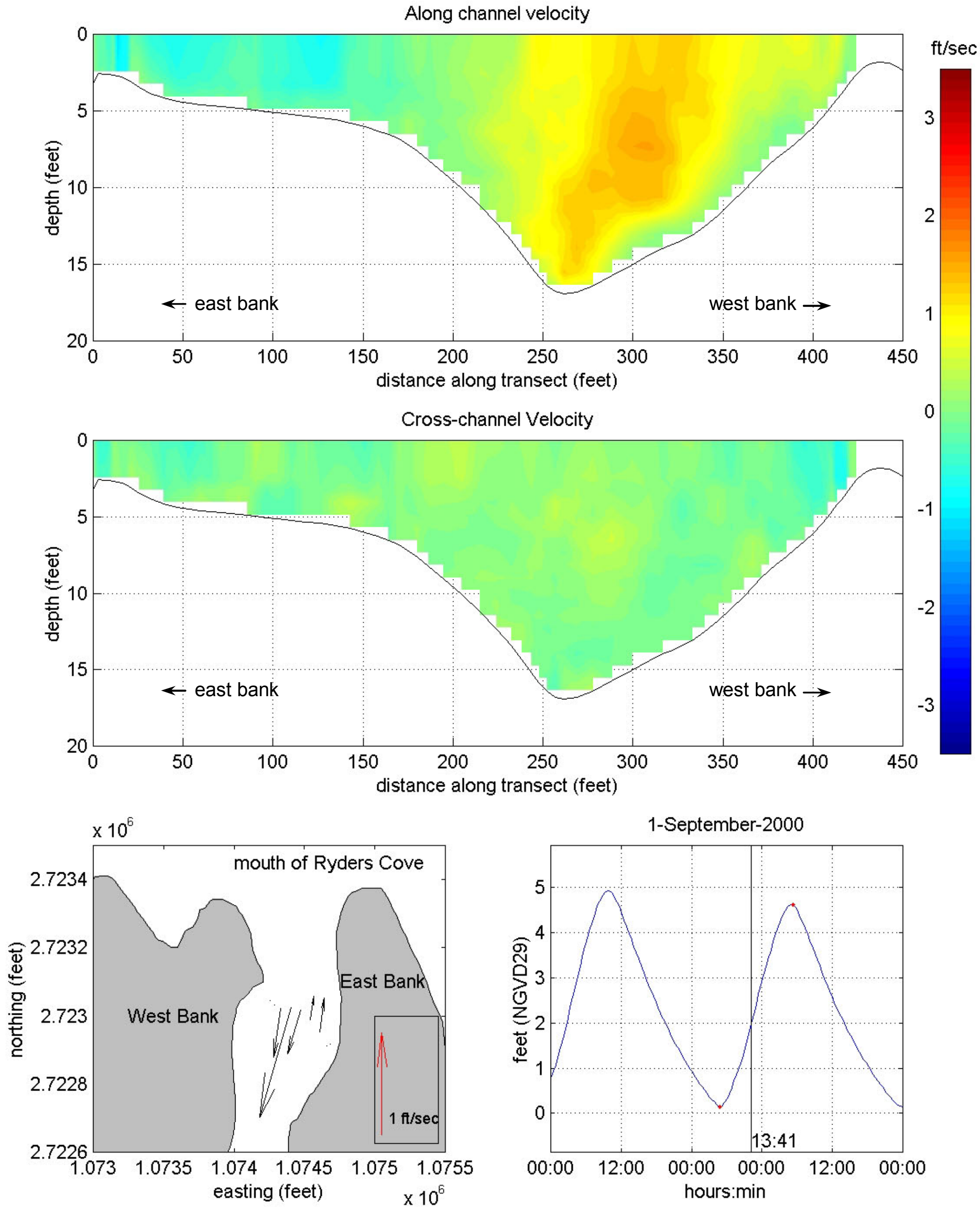


Figure V-36. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across the mouth of Ryder Cove measured at 13:41 on September 1, 2000 during the flood tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

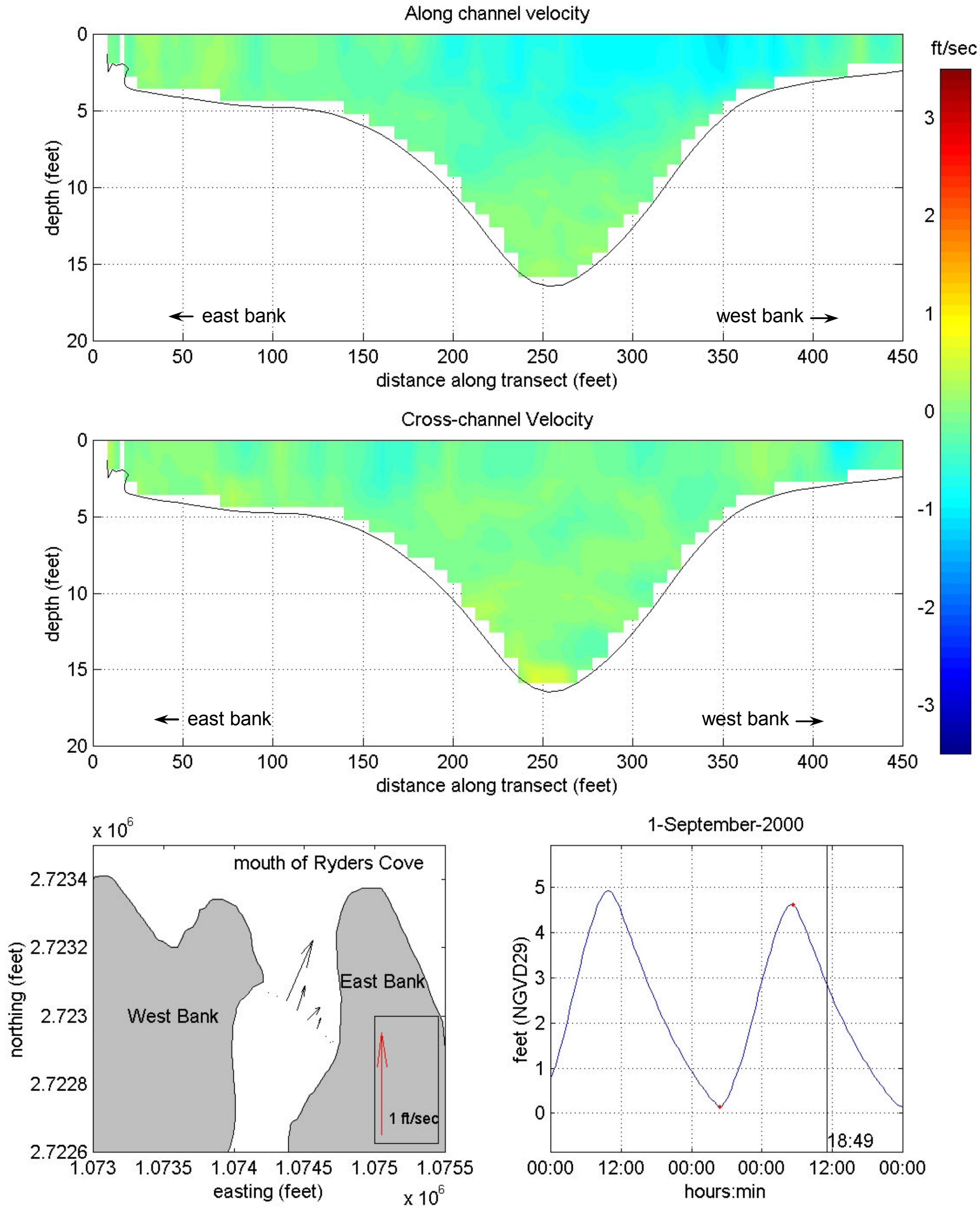


Figure V-37. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across the mouth of Ryder Cove measured at 18:49 on September 1, 2000 during the ebb tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

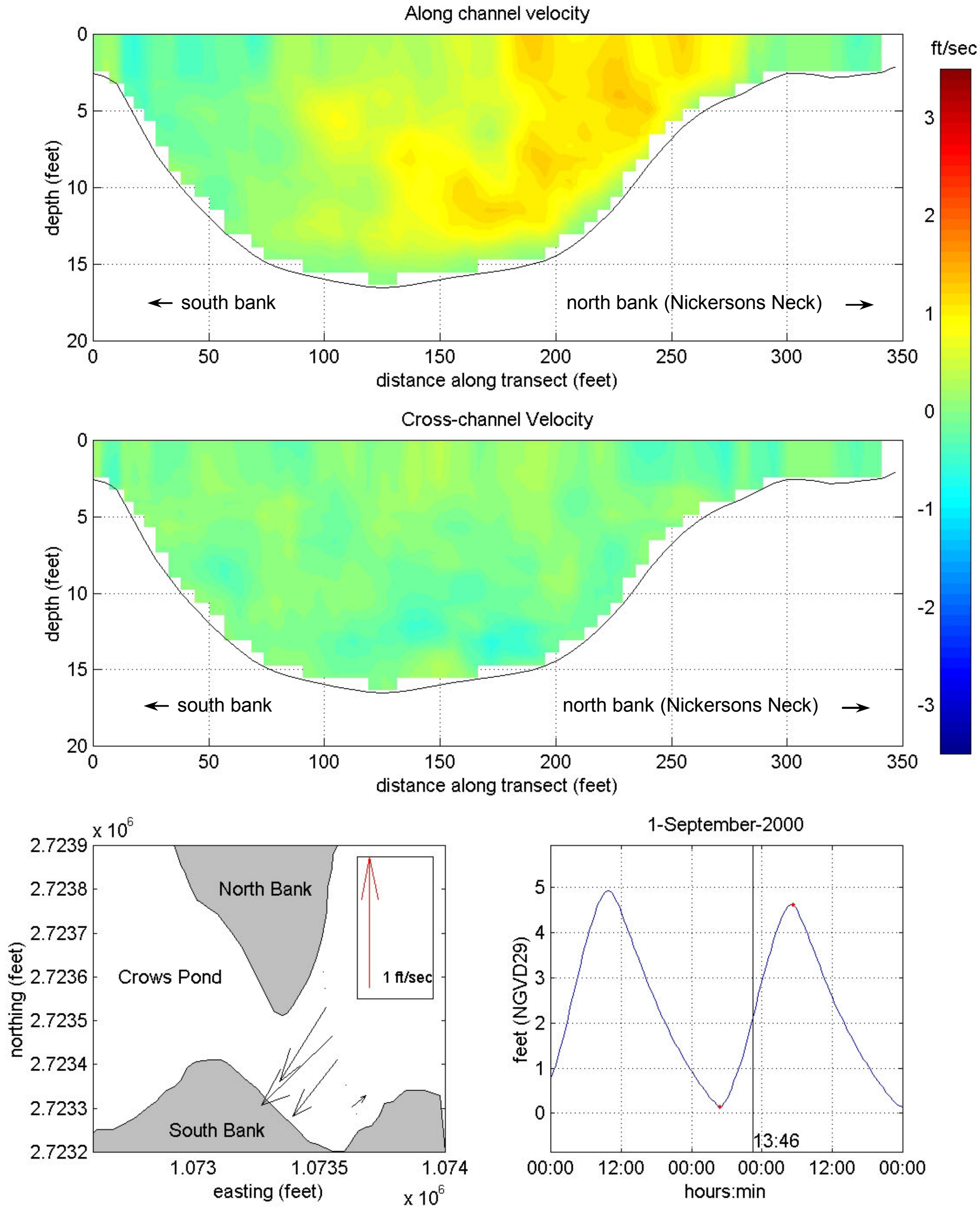


Figure V-38. Color contour plots of along-channel and cross-channel velocity components for transect line run south-to-north across the mouth of Crows Pond measured at 13:46 on September 1, 2000 during the flood tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

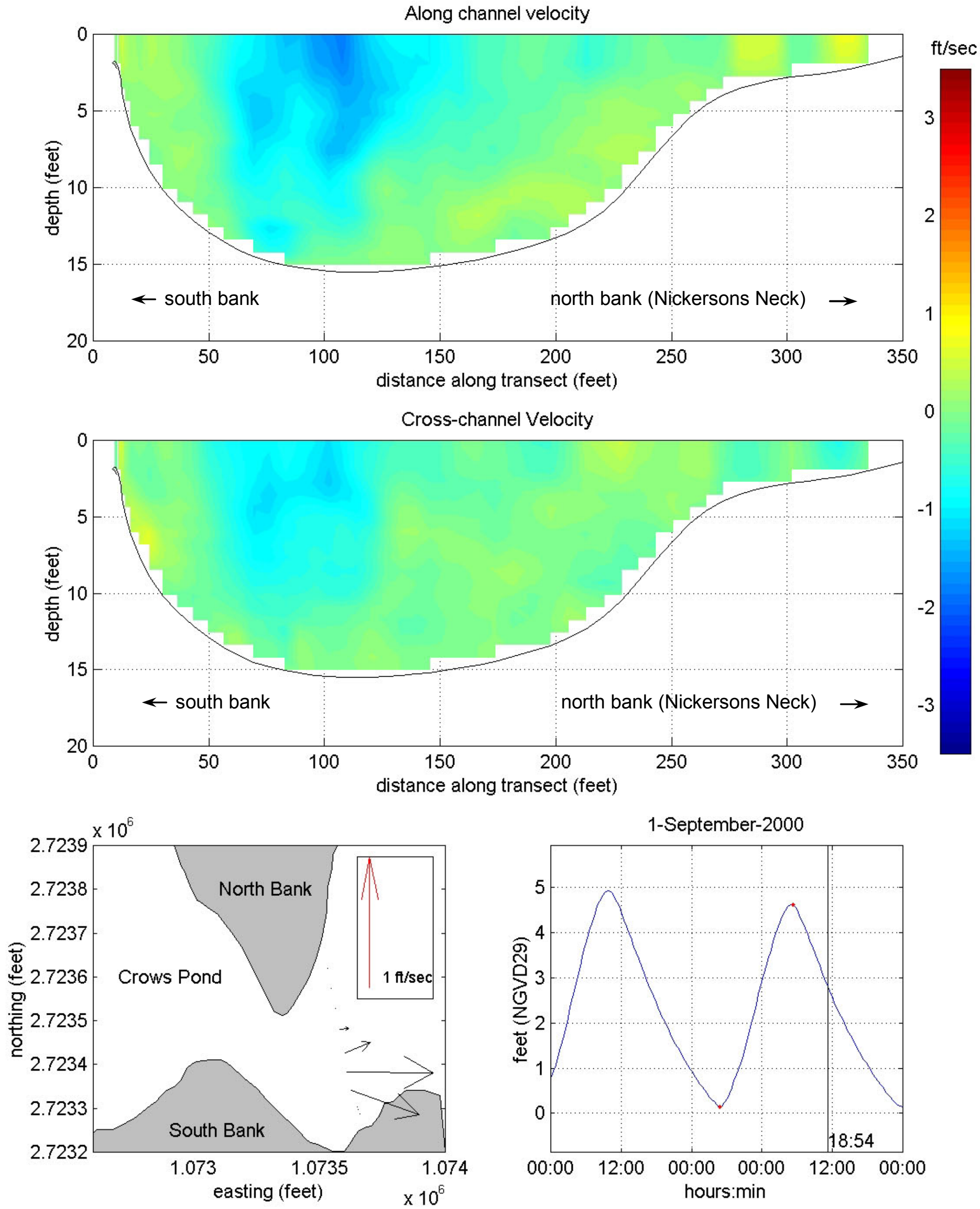


Figure V-39. Color contour plots of along-channel and cross-channel velocity components for transect line run south-to-north across the mouth of Crows Pond measured at 18:54 on September 1, 2000 during the ebb tide. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel.

V.4 HYDRODYNAMIC MODELING

This study focuses on five individual estuarine systems in Chatham, Massachusetts: Stage Harbor, Bassing Harbor, Sulphur Springs, Taylors Pond, and Muddy Creek. 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-2V model developed by Resource Management Associates (King, 1990). It is a two-dimensional, depth-averaged finite element model, capable of simulating transient hydrodynamics. The model is widely accepted and tested for analyses of estuaries or rivers. Applied Coastal staff members have utilized RMA-2V for numerous flushing studies on Cape Cod, including West Falmouth Harbor, Popponesset Bay, Pleasant Bay, Falmouth "finger" Ponds, and Barnstable Harbor.

V.4.1 Model Theory

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

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

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

V.4.2 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using contour data developed for the Town's Geographic Information System (GIS), as well as 1994 digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of each system based on the tide gauge data collected in Nantucket Sound and Pleasant Bay. Freshwater recharge boundary conditions for Muddy Creek and Frost Fish Creek were specified to approximate average fresh water inputs to the systems. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several (15+) model calibration simulations for each system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.4.2.1 Grid generation

The grid generation process was simplified by the use of the SMS package. The digitized shoreline and bathymetry data were imported to SMS, and a finite element grid was generated to represent the estuary. Information about each grid is provided in Table V-7. Figures V-40 through V-44 illustrate the finite element grids for each system modeled: Stage Harbor, Bassing Harbor, Sulphur Springs, Taylors Pond, and Muddy Creek. With the exception of groundwater inputs entering Muddy Creek and Frost Fish Creek, the embayments were represented by two-dimensional (depth-averaged) elements.

The finite element grid for each system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties of each estuary. Fine resolution was required to simulate the numerous channel constrictions that significantly impact the estuarine hydrodynamics. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary. Reference water depths at each node of the model were interpreted from bathymetry data obtained from a combination of sources, including 1) recent fathometer and/or ADCP surveys in Stage Harbor, Bassing Harbor, and Taylors Pond; 2) recent manual surveys of Muddy Creek, Upper Frost Fish Creek and Cockle Cove Creek; 3) existing NOAA data for Stage Harbor; 4) previous bathymetric survey of Bassing Harbor (ACI, 1997); and previous bathymetric surveys of the Sulphur Springs/Bucks Creek and Taylors Pond Systems (Stearns and Wheler, 1999)

Grid resolution was governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability in each 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. Also, elements through deep channels (e.g., Stage Harbor Inlet channel) were designed to account for rapid changes in bathymetry caused by inlet shoaling and scour processes. Widely spaced nodes were often employed in areas where flow patterns are not likely to change dramatically, such as Crows Pond, or Sulphur Springs. Appropriate implementation of wider node spacing and larger elements reduced computer run time with no sacrifice of accuracy.

Areas of marsh in the South Coastal Embayments (i.e., Sulphur Springs/Bucks Creek, Cockle Cove Creek, and Taylors Pond/ Mill Creek) were included in the models because these marsh areas are a large portion of the total area of these systems, and have a significant effect on the hydrodynamics of these embayments. In the other modeled systems, marsh areas were not included in order to simplify the modeling effort without impacting model accuracy. This is justified by the fact that the models calibrated and verified well without the inclusion of areas of marsh.

Table V-7. Characteristics of numerical grids developed for hydrodynamic analyses.					
System	Nodes	Elements	Max. Depth (ft, NGVD)	Min. Depth (ft, NGVD)	location of max. depth
Stage Harbor	3973	1466	-18.4	0.0	Stage Harbor Inlet
Bassing Harbor	4419	1443	-18.3	2.3	Crows Pond
Sulphur Springs	7882	2728	-4.5	2.0	Bucks Creek
Taylors Pond	5202	1853	-11.4	2.0	Taylors Pond
Muddy Creek	2872	874	-4.1	0.7	Lower Muddy Creek

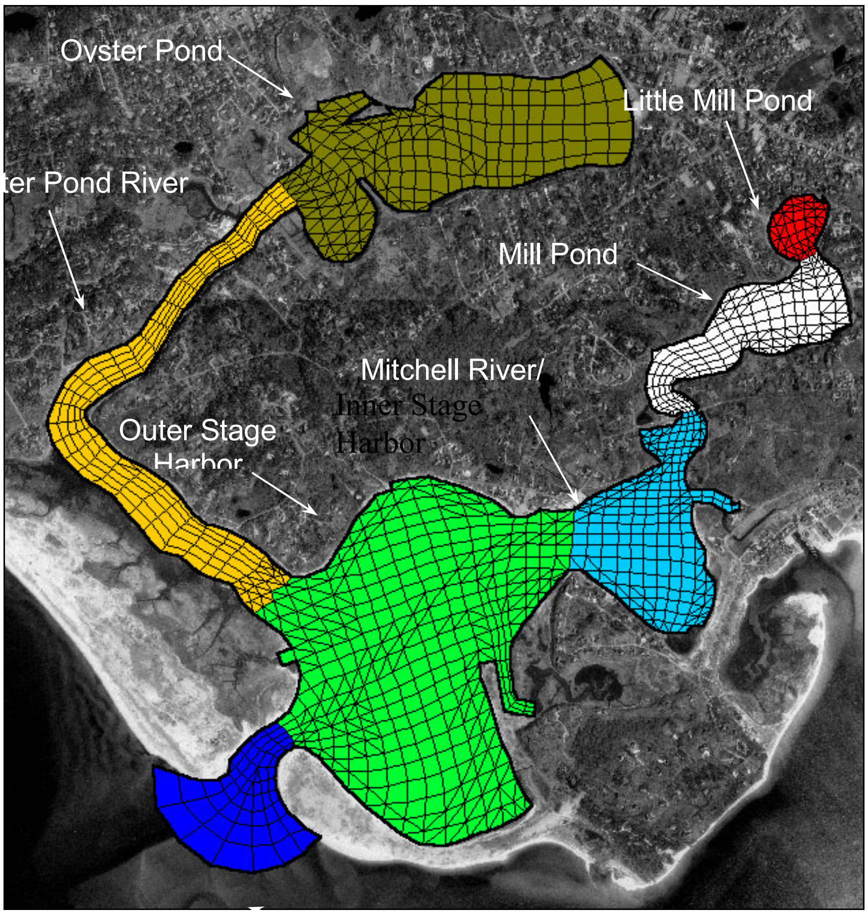


Figure V-40. Plot of numerical grid used for hydrodynamic modeling of Stage Harbor system. Colored divisions indicate boundaries of different grid material types, as well as volumes used to compute flushing rates for individual embayments.

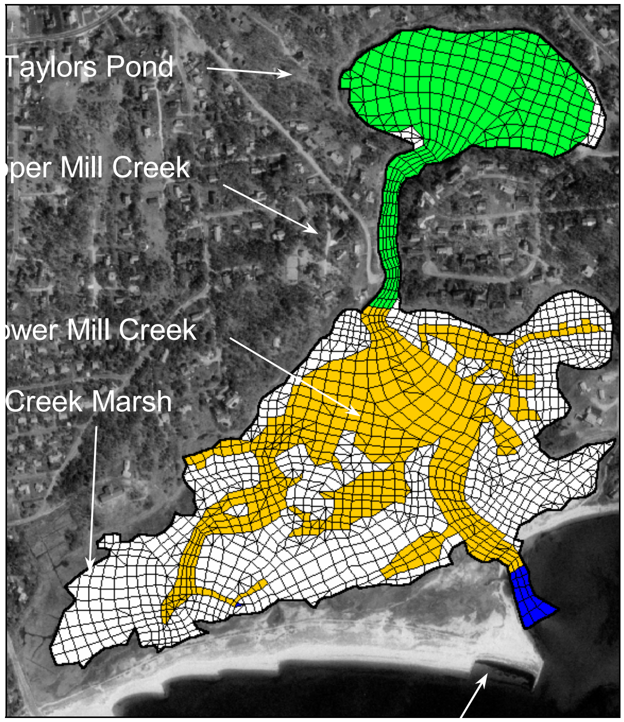


Figure V-41. Plot of numerical grid used for hydrodynamic modeling of Taylors Pond/Mill Creek system. Colored divisions indicate boundaries of different grid material types, as well as volumes used to compute flushing rates for individual embayments.

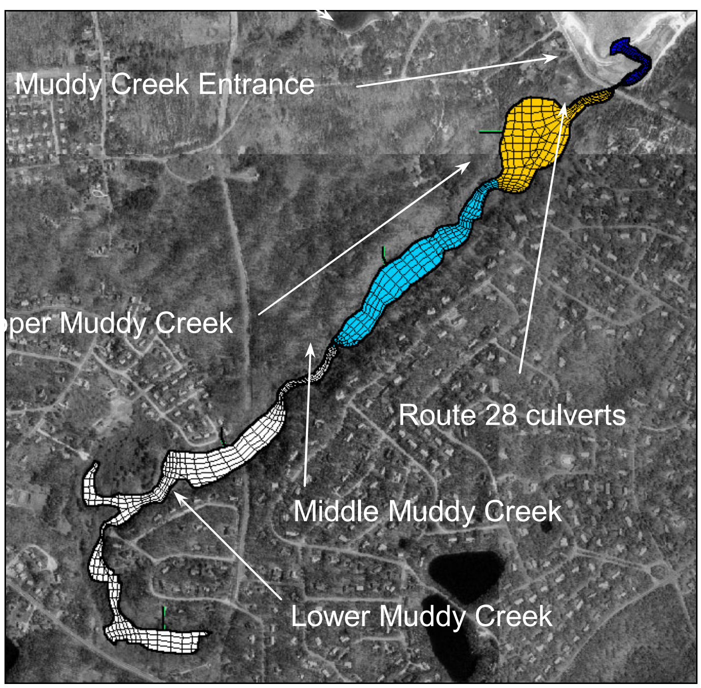


Figure V-42. Plot of numerical grid used for hydrodynamic modeling of Muddy Creek system. Colored divisions indicate boundaries of different grid material types, as well as volumes used to compute flushing rates for the system.

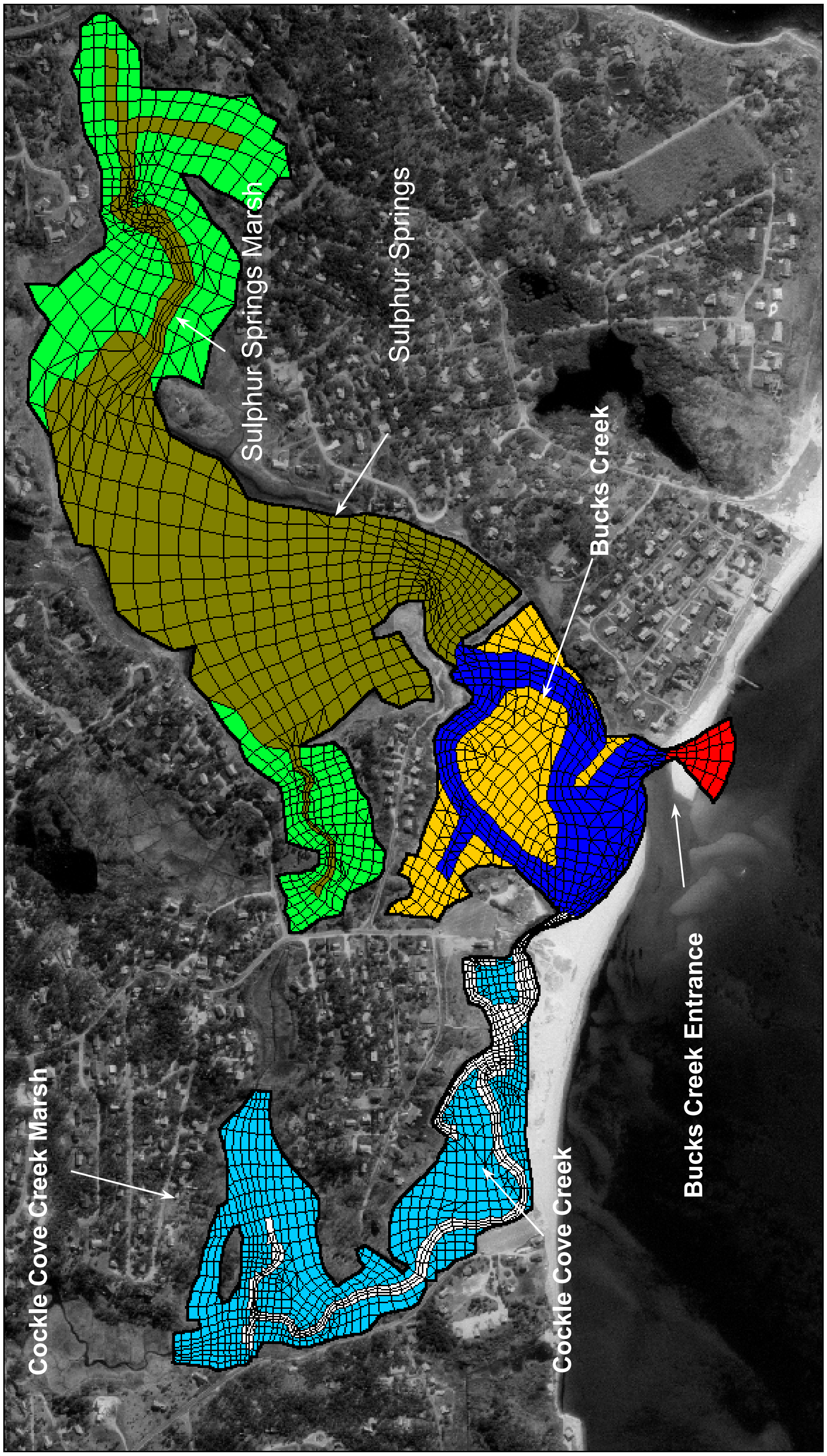


Figure V-43. Rotated view of numerical grid used for hydrodynamic modeling of Stage Harbor system. Colored divisions indicate boundaries of different grid material types, as well as volumes used to compute flushing rates for individual embayments.

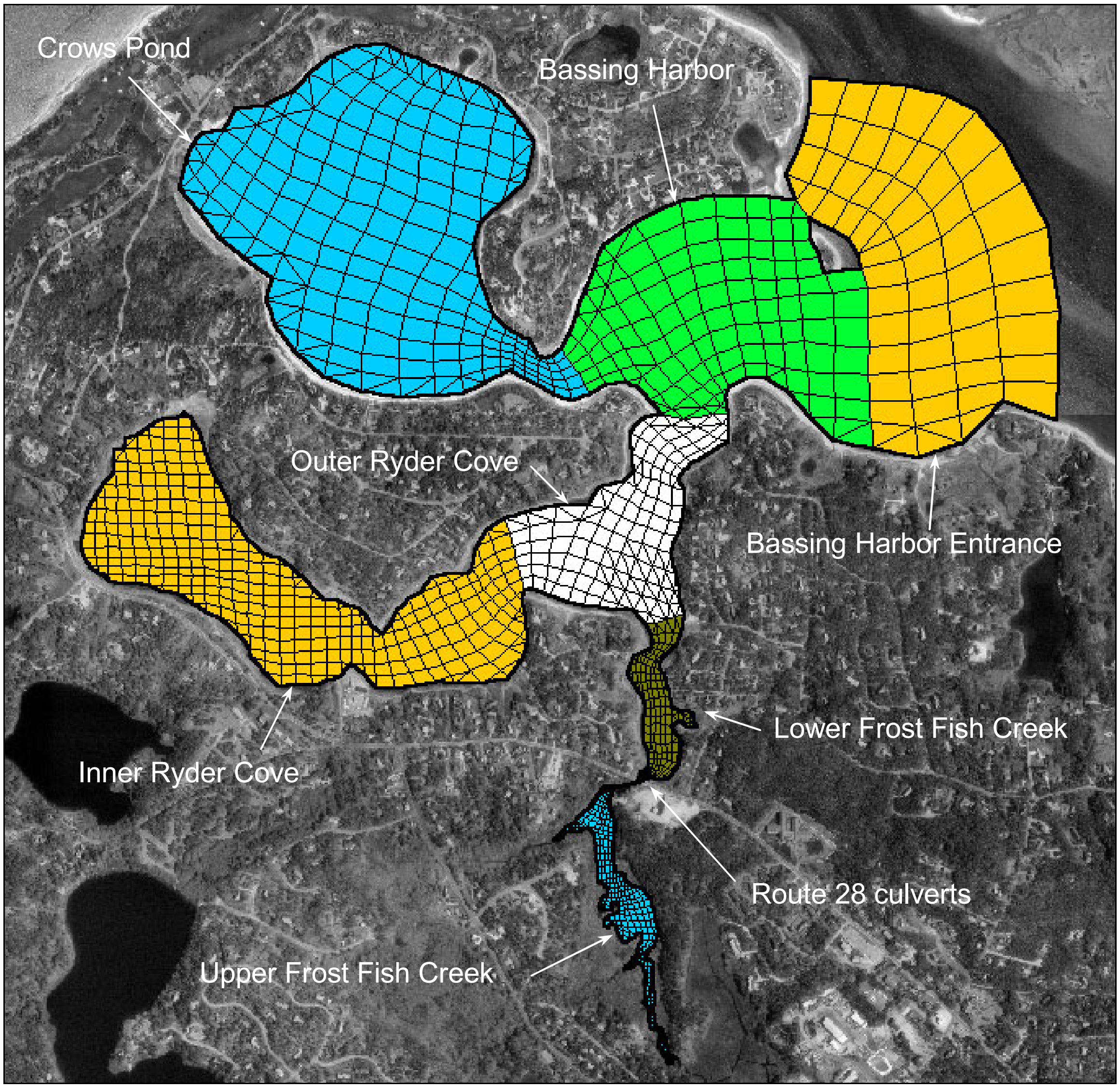


Figure V-44. Plot of numerical grid used for hydrodynamic modeling of Bassing Harbor system. Colored divisions indicate boundaries of different grid material types, as well as volumes used to compute flushing rates for individual embayments.

V.4.2.2 Boundary condition specification

Three types of boundary conditions were employed for the RMA-2V model: 1) "slip" boundaries, 2) freshwater inflow, and 3) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. Based on watershed areas and average rainfall, freshwater recharge (surface and ground water flows) was specified for Muddy Creek and Frost Fish Creek. The flow rates used in the model are 1.56 ft^3/sec for Frost Fish Creek and 3.43 ft^3/sec for Muddy Creek, based on an average rainfall of 16 inch/year (Cape Cod Commission, 1998) and watershed areas determined using the Town GIS (849 acres for Frost Fish Creek and 1863 acres for Muddy Creek). A tidal boundary condition was specified seaward of the inlet to each system. TDR measurements provided the required data. The rise and fall of the tide in Nantucket Sound and

Pleasant Bay is the primary driving force for estuarine circulation. Dynamic (time-varying) model simulations specified a new water surface elevation in Nantucket Sound (for Stage Harbor, Sulphur Springs, and Taylors Pond), and Pleasant Bay (for Bassing Harbor and Muddy Creek) every model time step minutes (12 minutes).

V.4.2.3 Calibration

After developing the finite element grids, and specifying boundary conditions, the model for each system was calibrated. The calibration procedure ensures that the model predicts accurately what was observed in nature during the field measurement program. Calibrated models provide a diagnostic tool to evaluate other scenarios (e.g., the effects of increasing the size of the Frost Fish Creek culverts to improve flushing). Numerous model simulations were required (typically 15+) for each estuary, specifying a range of friction and eddy viscosity coefficients, to calibrate the model.

Calibration of the flushing model required a close match between the modeled and measured tides in each of the sub-embayments where tides were measured (i.e., from the TDR deployments). Initially, a two-day period was calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement was achieved, a seven-day period was modeled to calibrate the model based on dominant tidal constituents discussed in Section III. The seven-day period was extracted from a longer simulation to avoid effects of model spin-up, and to focus on average tidal conditions.

The calibration was performed for a seven-day period beginning August 25, 2000 at 1800 EDT for the Pleasant Bay systems (i.e., Bassing Harbor and Muddy Creek), and beginning July 25, 2000 at 1600 EDT for the systems on Nantucket Sound (i.e., Stage Harbor, Sulphur Springs, and Taylors Pond). These representative time periods include the spring tide range of conditions, when the tide range largest, and resulting tidal currents are greater as well. To provide average tidal forcing conditions for the flushing analyses, a separate time period was chosen that spanned the transition between spring and neap tide ranges (bi-weekly maximum and minimum tidal ranges, respectively). For the flushing analysis the 7.25 day period (14 tide cycles) beginning July 31 2000, at 1300 EDT was used for the systems on Nantucket Sound, and a similar period beginning August 31 2000 at 0300 EDT was selected for the systems on Pleasant Bay.

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. Modeled tides for the calibration time period were evaluated for time (phase) lag and height damping of dominant tidal constituents. The calibrated model was used to analyze existing detailed flow patterns and compute residence times.

V.4.2.3.1 Friction coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where velocities are relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude damping and phase delay of the tidal signal. Friction is approximated in RMA-2V as a Manning coefficient. Initially, Manning's friction coefficients between 0.02 and 0.07 were specified for all elements. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels 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 in the entrance channel of each Pond, versus the silty bottom of the shallow regions in the upper portions of each Pond, which provided 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-8.

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 50 and 100 lb-sec/ft^2. Higher values (up to 300 lb-sec/ft^2) were used on the marsh plain and in culverts.

V.4.2.3.3 Wetting and Drying/marsh porosity processes

Modeled hydrodynamics were complicated by wetting/drying cycles on the marsh plain as well as in intertidal regions in each of the systems. In the case of the marsh plains that are a part of the Sulphur Springs/Cockle Cove Creek and Mill Creek systems, wet/dry areas 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-2V model which 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 RM-2V to change the ability of an element to hold water, like squeezing a sponge. The marsh porosity feature of RMA-2V is typically utilized in estuarine systems where the marsh plain has a significant impact on the hydrodynamics of a system, such as Sulphur Springs and Mill Creek.

Table V-8. Manning's Roughness coefficients used in simulations of modeled embayments.

System	Embayment	Bottom Friction
Stage Harbor	Stage Harbor Entrance	0.030
	Lower Stage Harbor	0.030
	Mitchell River / Upper Stage Harbor	0.030
	Mill Pond	0.025
	Little Mill Pond	0.025
	Oyster Pond River	0.030
	Oyster Pond	0.030
Sulphur Springs	Bucks Creek Entrance	0.030
	Bucks Creek	0.030
	Sulphur Springs	0.030
	Sulphur Springs Marsh Plain	0.100
	Cockle Cove Creek	0.040
	Cockle Cove Creek Marsh Plain	0.070
Taylors Pond	Mill Creek Entrance	0.030
	Taylors Pond	0.025
	Mill Creek	0.027
	Mill Creek Marsh Plain	0.100
Bassing Harbor	Bassing Harbor Entrance	0.030
	Bassing Harbor	0.031
	Outer Ryder Cove	0.015
	Crows Pond	0.030
	Inner Ryder Cove	0.015
	Upper Frost Fish Creek	0.030
	Frost Fish Creek culverts	0.500
	Lower Frost Fish Creek	0.030
Muddy Creek	Muddy Creek Entrance	0.030
	Muddy Creek Culverts	0.150
	Muddy Creek	0.025

For Stage Harbor and Bassing Harbor, an alternate method was employed to simulate the periodic inundation and drying of tidal flats in these systems. Nodal wetting and drying is a feature of RMA-2V that allows grid elements to be removed and re-inserted during the course of the model run. Figure V-45 presents an example of how the computational grid is modified by element elimination. This figure shows the Stage Harbor model at a point just after low tide. White areas within the boundary of the mesh are elements that have gone dry, and as a result, have been removed temporarily from the model solution. The wetting and drying feature has two key benefits for the simulation, 1) it enhances the stability of the model by eliminating nodes that have bottom elevations that are higher than the water surface elevation at that time, and 2) it reduces total model run time because node elimination can reduce the size of the computational grid significantly during periods of a model run. Wetting and drying is employed for estuarine systems with relatively shallow borders and/or tidal flats.

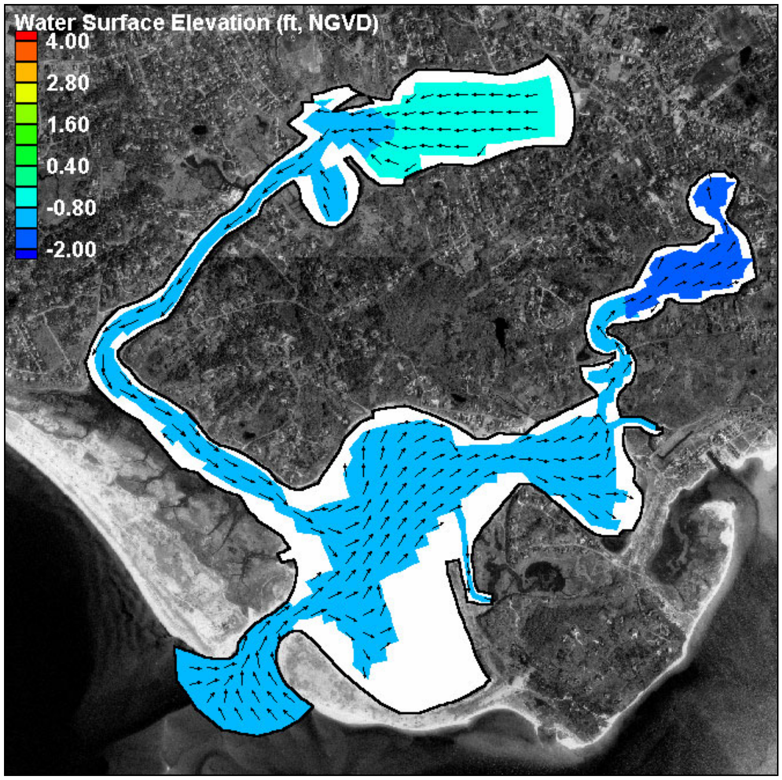


Figure V-45. Stage Harbor model at the inception of a flood tide, with white areas indicating dry elements.

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-46 through V-54 illustrate the seven-day calibration simulation along with a two-day sub-section. Modeled (solid line) and measured (dotted line) tides are illustrated at each model location with a corresponding TDR.

Although visual calibration achieved reasonable modeled tidal hydrodynamics, further tidal constituent calibration was required to quantify the accuracy of the models. Calibration of M_2 was the highest priority since M_2 accounted for a majority of the forcing tide energy in the modeled systems. Due to the duration of the model runs, four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Tables V-9 and V-4 for the calibration period differ from those in Table V-7 because constituents were computed for only the seven-day section of the thirty-days represented in Table V-7. Tables V-9 and V-10 compare tidal constituent height and time lag for modeled and measured tides at the TDR locations. Time lag represents the time required for a constituent to propagate from offshore (Nantucket Sound or Pleasant Bay) to each TDR location.

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, which was only slightly larger than the accuracy of the tide gages (0.032 ft). Generally, errors in modeled constituent amplitudes were of the order 0.01 ft. Time lag errors were typically less than the time increment resolved by the model (0.20 hours or 12 minutes), indicating good agreement between the model and data.

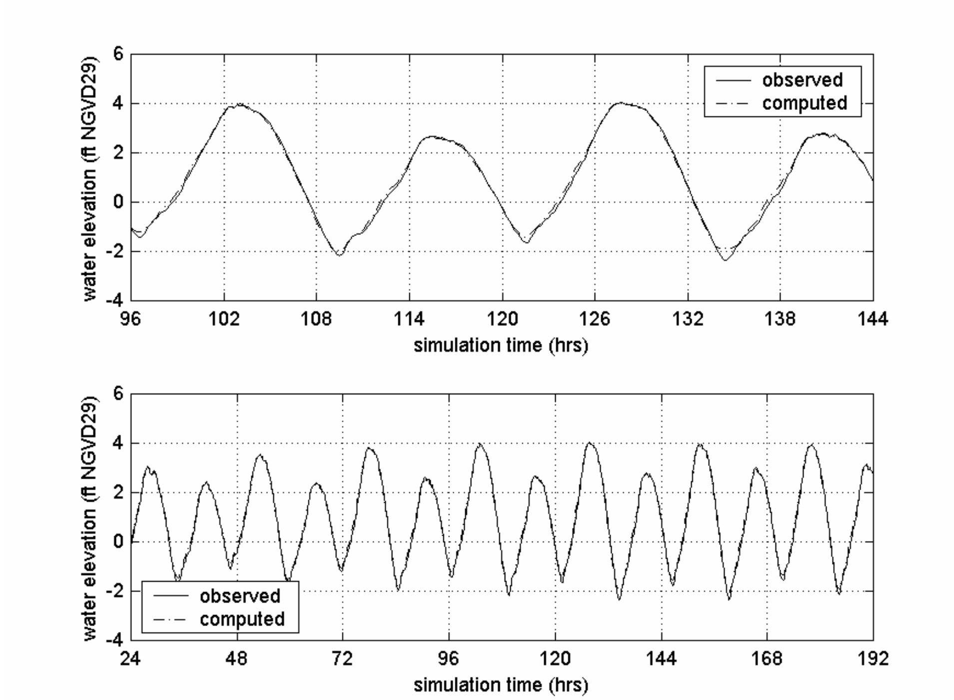


Figure V-46. Observed and computed water surface elevations during calibration time period, for Mill Pond.

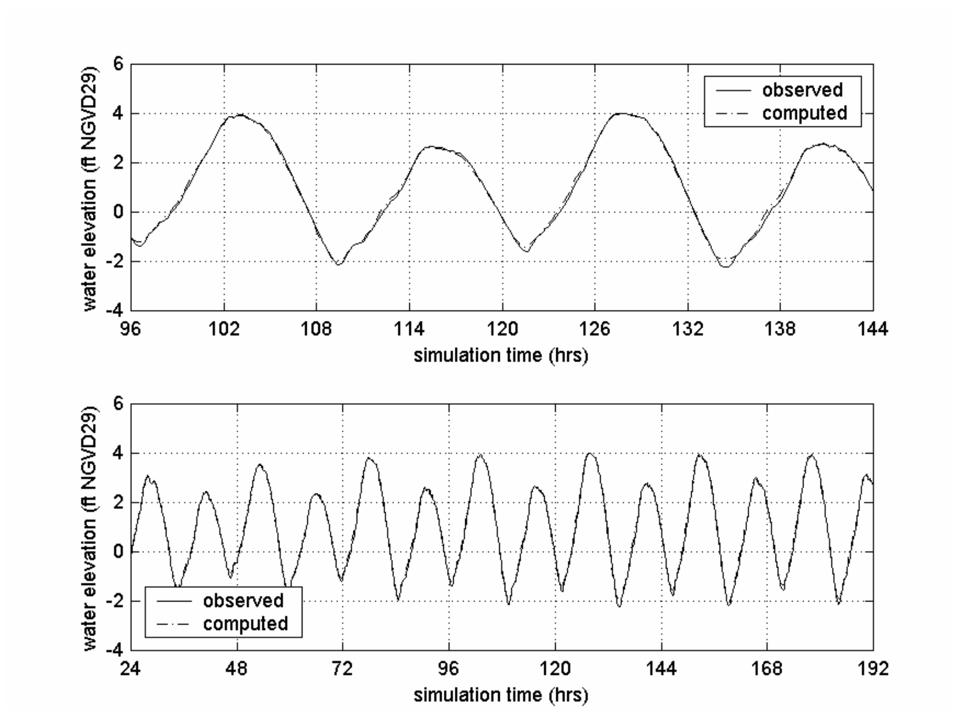


Figure V-47. Observed and computed water surface elevations during calibration time period, for Little Mill Pond.

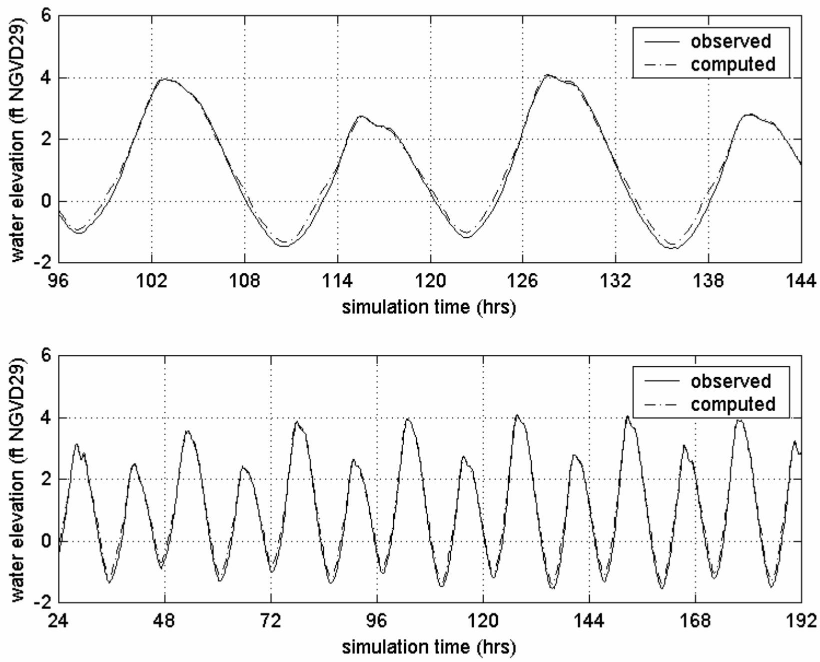


Figure V-48. Observed and computed water surface elevations during calibration time period, for Oyster Pond.

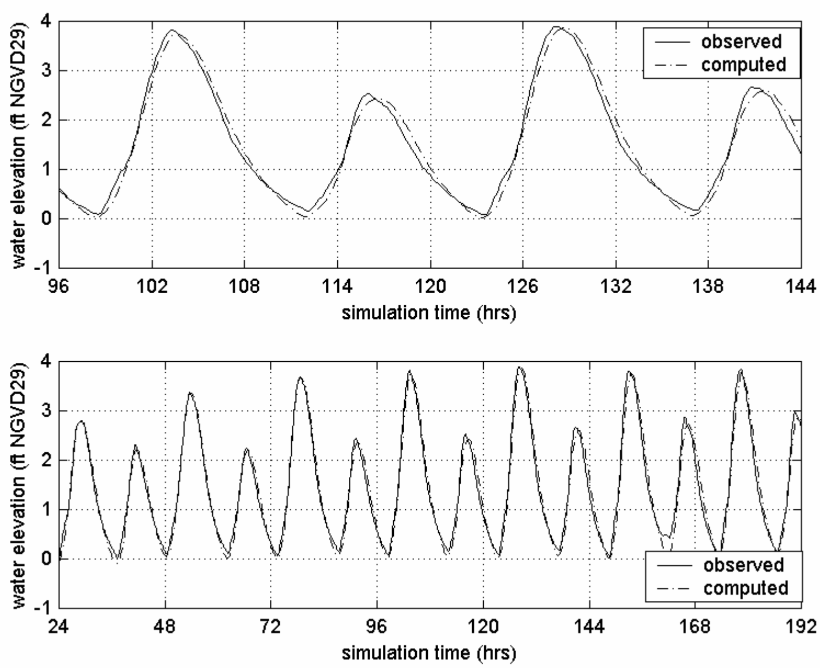


Figure V-49. Observed and computed water surface elevations during calibration time period, for Sulphur Springs.

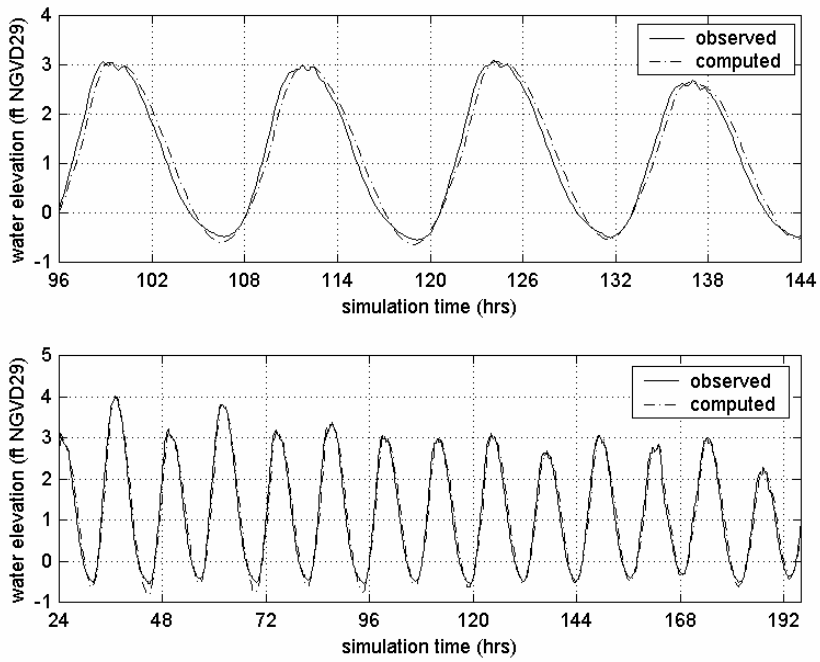


Figure V-50. Observed and computed water surface elevations during calibration time period, for Taylors Pond.

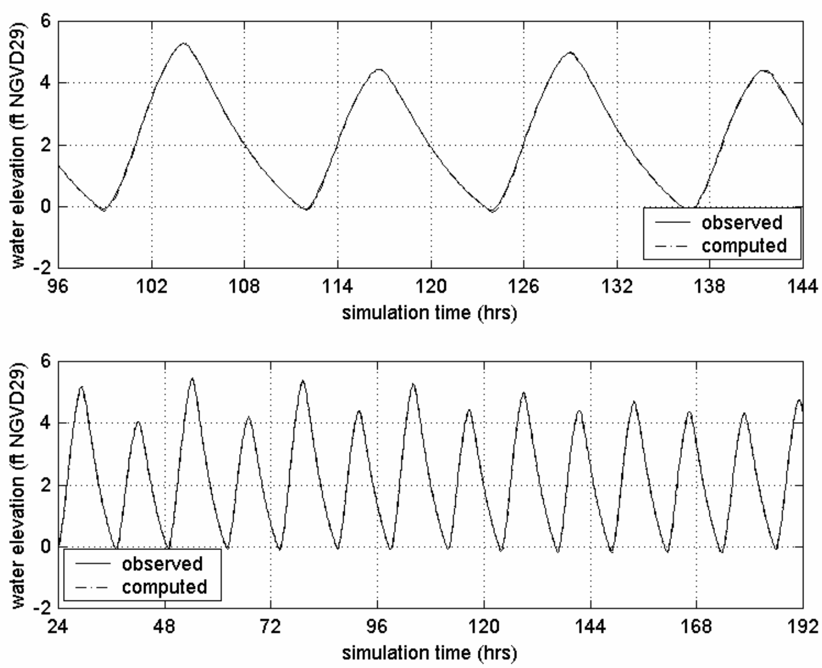


Figure V-51. Observed and computed water surface elevations during calibration time period, for Crows Pond.

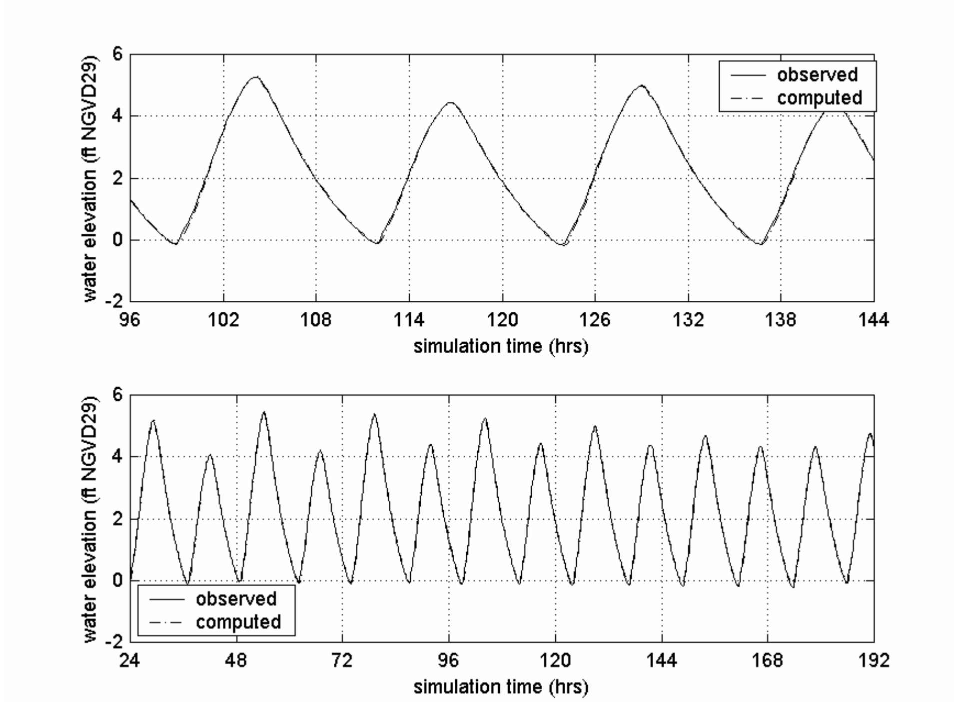


Figure V-52. Observed and computed water surface elevations during calibration time period, for Ryder Cove.

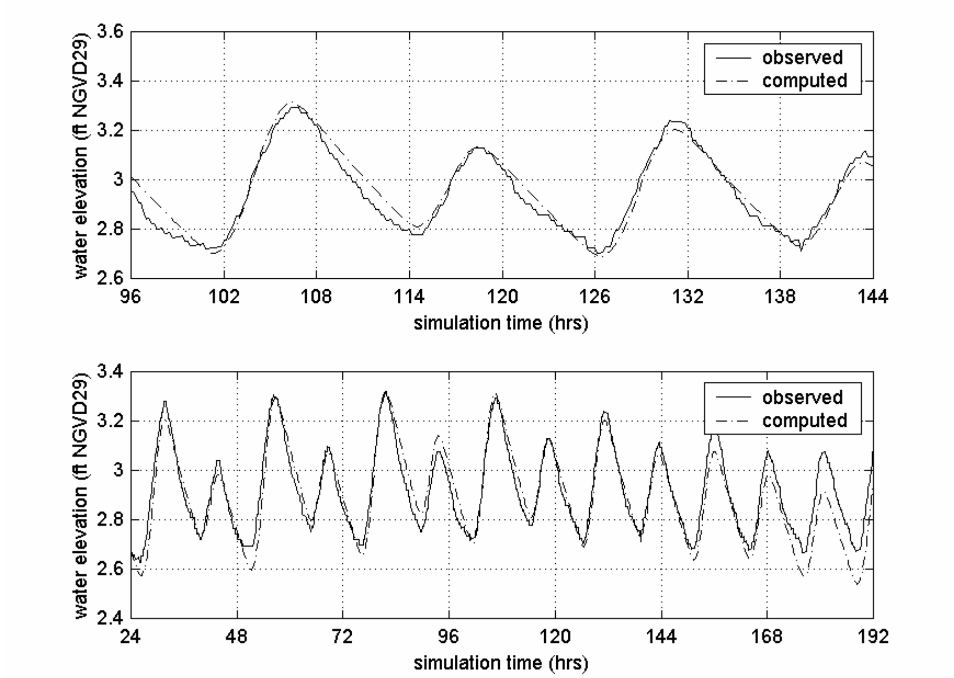
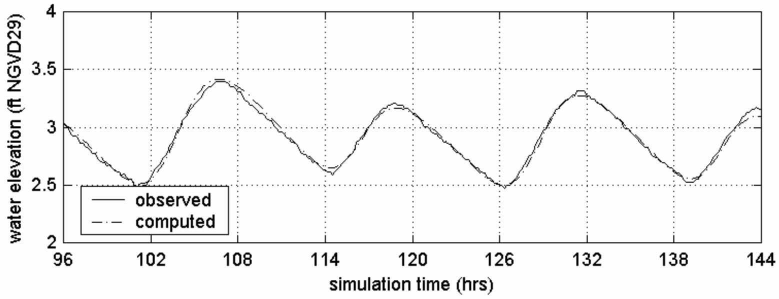


Figure V-53. Observed and computed water surface elevations during calibration time period, for Frost Fish Creek.



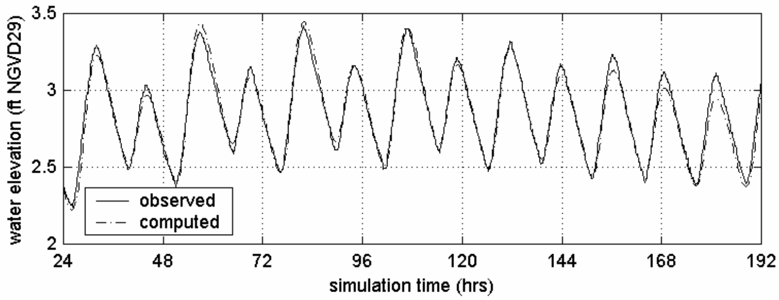


Figure V-54. Observed and computed water surface elevations during calibration time period, for Muddy Creek.

Table V-9. Tidal constituents for measured water level data and calibrated model output for northern embayments.

Model calibration run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Crows Pond	2.17	0.37	0.06	0.30	171.2	267.3
Ryder Cove	2.17	0.35	0.07	0.30	170.2	265.4
Frost Fish Creek	0.20	0.05	0.01	0.08	247.0	34.3
Muddy Creek	0.33	0.05	0.01	0.11	251.0	19.3
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Crows Pond	2.16	0.36	0.05	0.30	170.9	266.9
Ryder Cove	2.16	0.32	0.07	0.30	168.9	263.6
Frost Fish Creek	0.20	0.04	0.01	0.08	237.8	43.6
Muddy Creek	0.33	0.06	0.01	0.10	246.9	25.5
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Crows Pond	0.01	0.01	0.01	0.00	0.6	0.4
Ryder Cove	0.01	0.02	0.00	0.00	2.8	1.8
Frost Fish Creek	0.00	0.01	0.00	0.00	19.0	9.7
Muddy Creek	0.00	0.01	0.00	0.01	8.6	6.4

Table V-10. Tidal constituents for measured water level data and calibrated model output for Stage Harbor and South Coast Embayments.

Model calibration run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Mill Pond	2.24	0.13	0.12	0.57	140.6	81.0
Little Mill Pond	2.24	0.13	0.12	0.57	140.7	82.1
Oyster Pond	2.16	0.14	0.07	0.56	153.9	214.7
Sulphur Springs	1.40	0.23	0.03	0.48	171.7	278.3
Taylors Pond	1.80	0.18	0.06	0.16	152.4	245.7
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Mill Pond	2.30	0.13	0.13	0.57	142.4	85.7
Little Mill Pond	2.31	0.13	0.14	0.57	142.7	86.3
Oyster Pond	2.03	0.14	0.07	0.57	155.1	222.1
Sulphur Springs	1.35	0.28	0.05	0.48	164.6	277.8
Taylors Pond	1.82	0.14	0.04	0.17	149.3	243.2
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Mill Pond	0.06	0.00	0.01	0.00	3.7	4.9
Little Mill Pond	0.07	0.00	0.02	0.00	4.2	4.3
Oyster Pond	0.13	0.00	0.00	0.01	2.5	7.7
Sulphur Springs	0.05	0.05	0.02	0.00	14.8	0.4
Taylors Pond	0.02	0.04	0.02	0.01	6.5	10.5

The hydrodynamic model's ability to predict propagation of the secondary non-linear constituents through the estuary is important for understanding the attenuation of the tidal signal and the impact this has on estuarine circulation. Of primary interest is the M_4 constituent, which can be used to determine the flood dominance (sediment trapping characteristics) of an estuarine system. Proper prediction of M_4 provides confidence in the model's accuracy, since this indicates that the model is capable of simulating the tidal wave form and size. Similar to the model predictions for M_2, comparison of the information from Tables V-9 and V-10 indicates that the modeled phase of M_4 falls within one time step of the observed value.

V.4.3 Model Verification Using ADCP Measurements

The calibration procedure used in the development of the five separate finite-element models required a match between measured and modeled tides. To verify the performance of the Stage Harbor and Bassing Harbor models, computed flow rates were compared to flow rates measure using an ADCP. The ADCP data survey efforts are described in Chapter III. For model verification, both models were run for the period covered during each ADCP survey, on August 16 for Stage Harbor, and September 1 for Bassing Harbor. Model flow rates were computed in RMA-2V at continuity lines (channel cross-sections) that correspond to the actual ADCP transects followed in each survey.

V.4.3.1 Stage Harbor

A comparison of the measured and computed volume flow rates at the Stage Harbor Inlet is shown in Figure V-55 in the top plot, and the tide curve for the same time period is shown in the lower plot. Each ADCP point is a summation of flow measured along the ADCP transect. The ‘bumps’ and ‘skips’ of the flow rate curve can be attributed to the effects of winds (i.e., atmospheric effects) on the water surface and friction across the seabed periodically retarding or accelerating the flow through the inlet, and in the harbor. If water surface elevations changed smoothly as a sinusoid, the volume flow rate would also appear as a smooth curve. However, since the rate at which water surface elevations change does not vary smoothly, the flow rate curve is expected to show short-period fluctuations.

Figure V-55 for the Stage Harbor inlet shows a remarkably good agreement with the model predictions. The calibrated model accurately describes the general conditions and the irregularities of the discharge through the Stage Harbor inlet. Again, at the mouth of Oyster Pond River (Figure IV-56) and the Mill Pond Bridge (Figure V-57), computed volume flow rates agree well with the field measurements, even though the flows are an order of magnitude (~10 times) smaller at the Mill Pond Bridge than in Stage Harbor. Currents are more difficult to accurately measure with the ADCP along the Mill Pond Bridge and Oyster Pond River transects, since these areas are considerably shallower than the harbor inlet. Therefore, portions of the channels are not covered by the ADCP because 1) the ADCP draft (no measurements in top layer), and 2) tide flats too shallow to safely navigate the survey boat, become a much more significant source of measurement error.

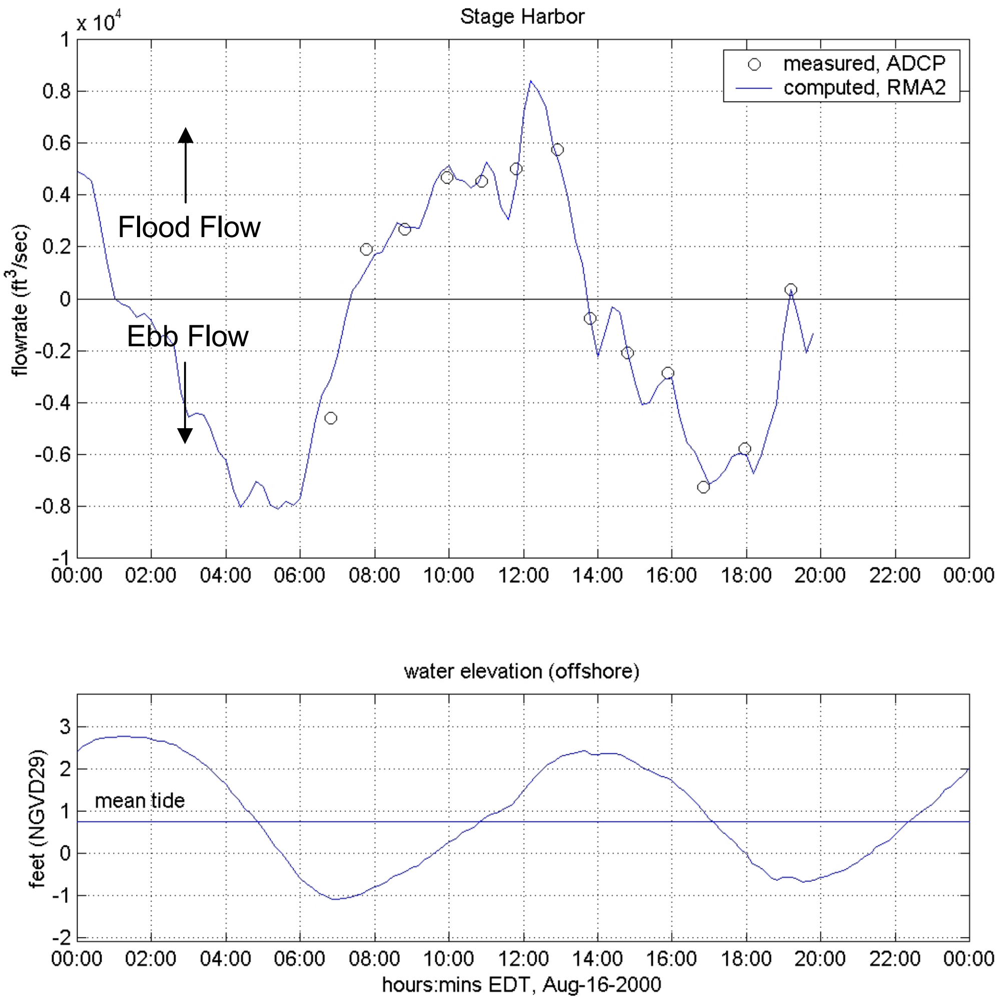


Figure V-55. Comparison of measured volume flow rates versus modeled flow rates through the Stage Harbor Inlet over a tidal cycle on August 16, 2000. Flood flows into the harbor are positive (+), and ebb flows out of the harbor are negative (-).

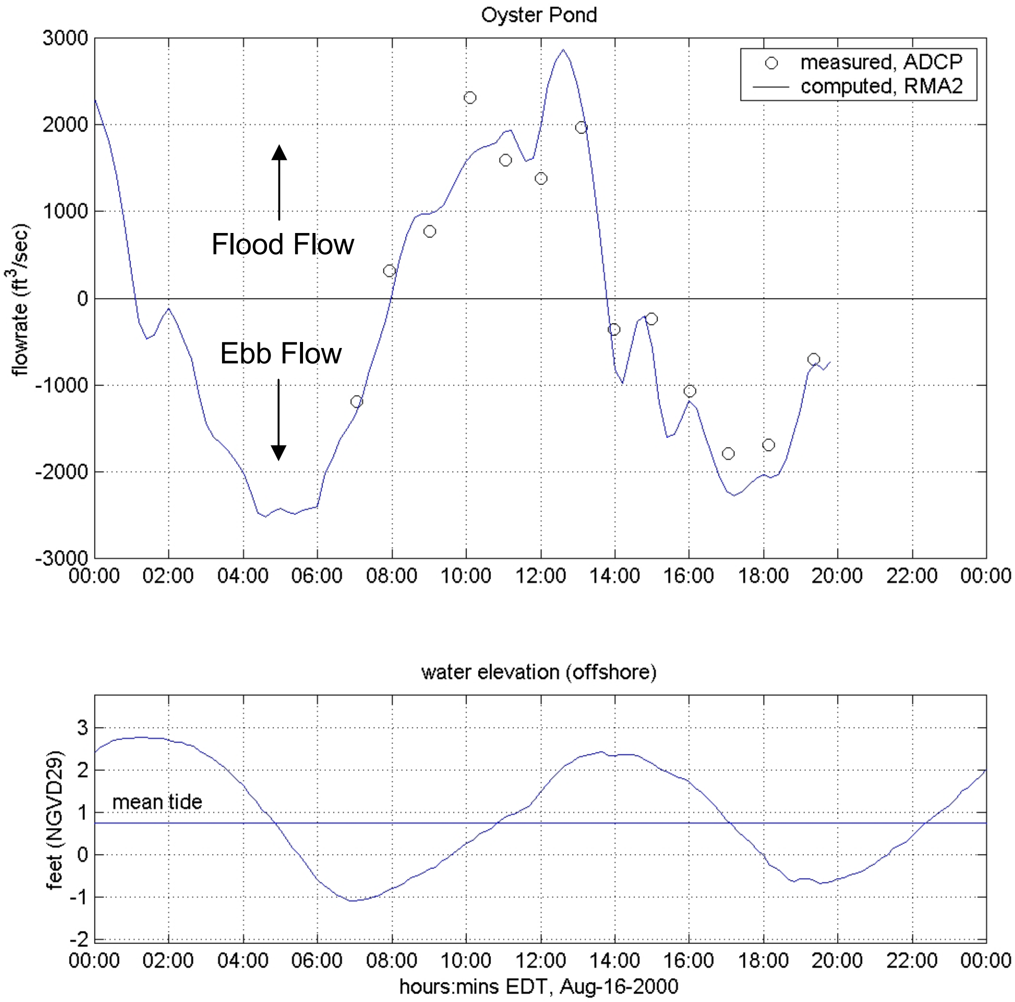


Figure V-56. Comparison of measured volume flow rates versus modeled flow rates through the mouth of Oyster Pond River over a tidal cycle on August 16, 2000. Flood flows into the river are positive (+), and ebb flows out of the river are negative (-).

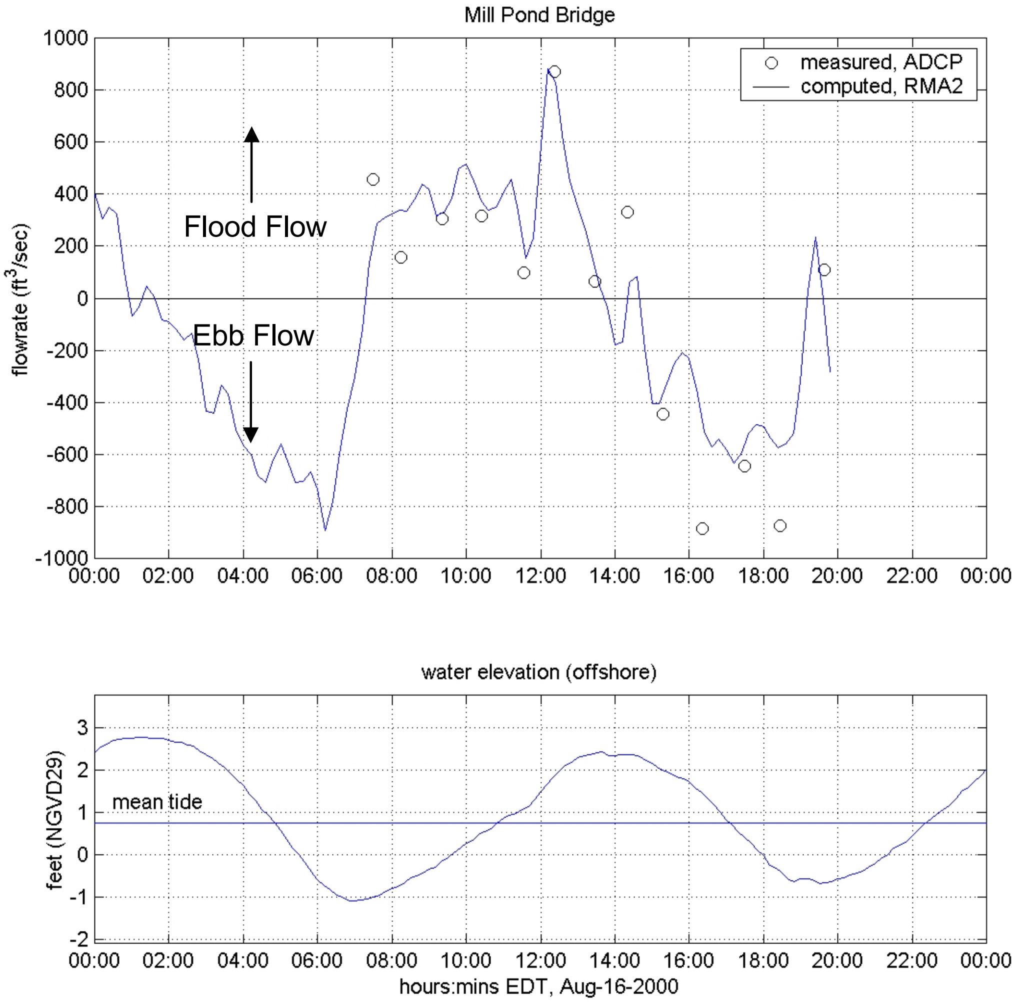


Figure V-57. Comparison of measured volume flow rates versus modeled flow rates through the Mill Pond Bridge over a tidal cycle on August 16, 2000. Flood flows into the pond are positive (+), and ebb flows out of the pond are negative (-).

V.4.3.2 Bassing Harbor

The calibrated Bassing Harbor model was utilized to compute volume flow rates for the mouth of Bassing Harbor, Ryder Cove (including Frost Fish Creek), and Crows Pond. The overall shape of the volume flow curve at the entrance to Bassing Harbor is relatively smooth compared to the Stage Harbor curve, suggesting that wind had less influence on water level changes for this system during the survey period. Flow rates at the Bassing Harbor inlet were noticeably over-predicted during ebb flows (Figure V-58).

The apparent large difference (~20%) during ebbing flow may result from the fact that the ADCP survey transect at the mouth of Bassing Harbor crossed between the southern shore of the inlet and Fox Hill to the north, and not completely across the harbor entrance. Fox Hill is an island, and is connected to the northern shore of the harbor mouth by a sand spit, which is submerged during much of the tide cycle. During the period of the tide cycle following high tide, water can flow easily over this spit. However, during the period following low tide, the spit is barely submerged, resulting in much less flow in this area of the harbor mouth. Therefore, measured and modeled flow rates agree better during the flood flow, when nearly all the flow into Bassing Harbor occurs between Fox Hill and the southern shore of the harbor entrance.

Water moving through Bassing Harbor is divided between Ryder Cove/Frost Fish Creek and Crows Pond. The computed volume flow rates from the calibrated model closely reflect the measured flow rates in the sub-embayments of Bassing Harbor (Figure V-59 and V-60).

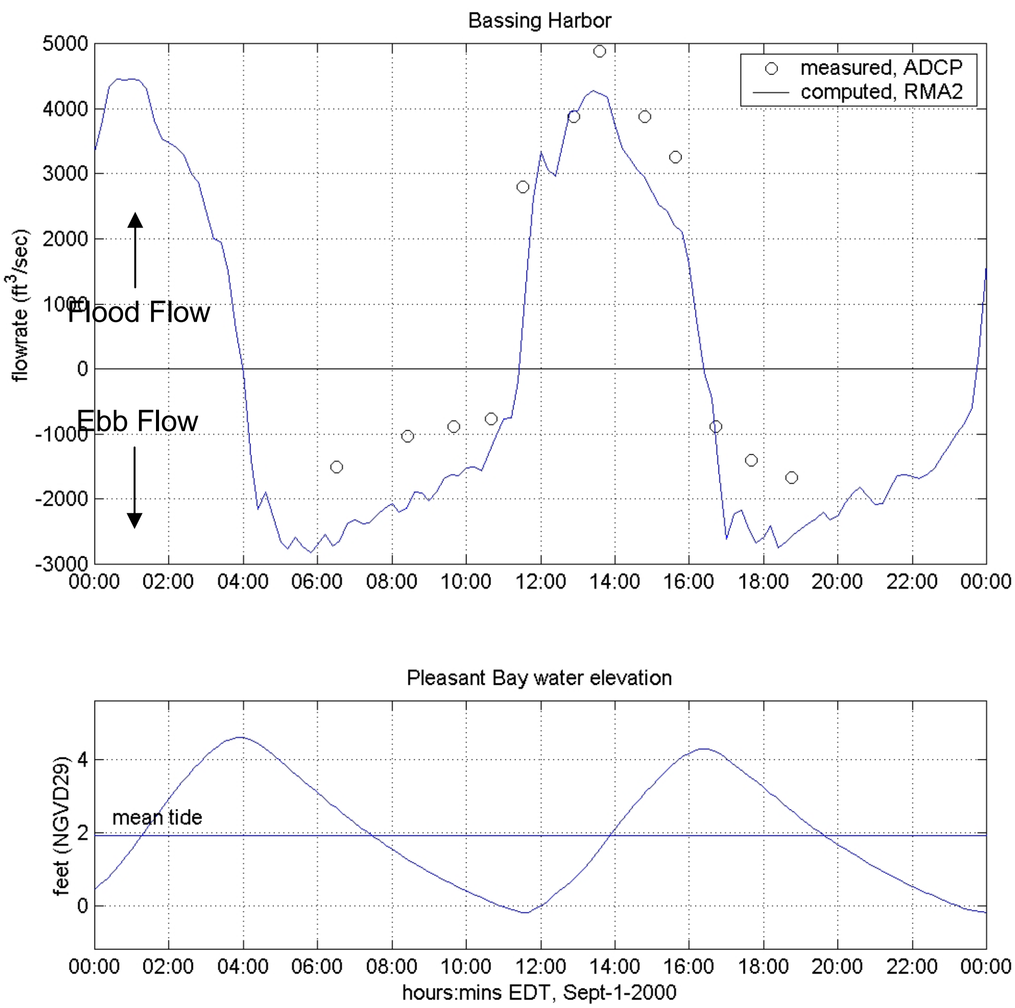


Figure V-58. Comparison of measured volume flow rates versus modeled flow rates through the Bassing Harbor Inlet over a tidal cycle on September 1, 2000. Flood flows into the harbor are positive (+), and ebb flows out of the harbor are negative (-).

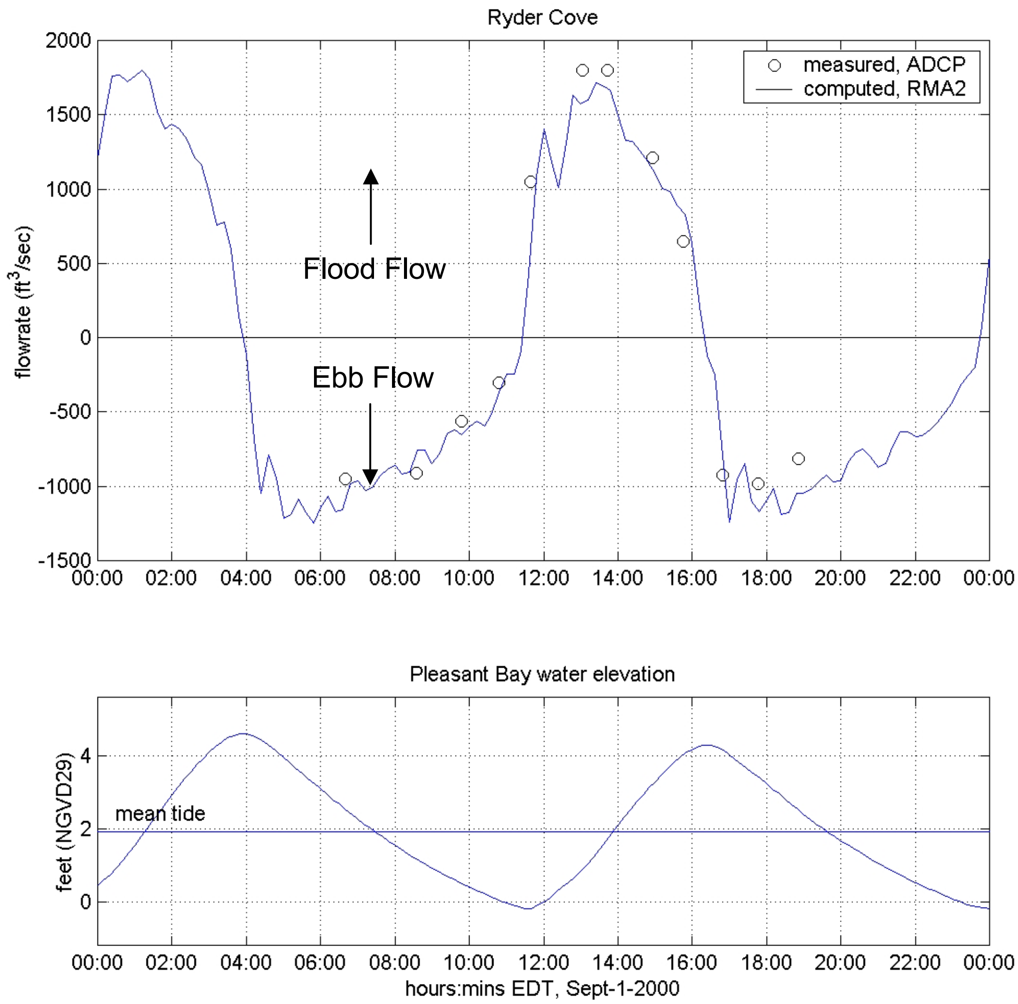


Figure V-59. Comparison of measured volume flow rates versus modeled flow rates through the entrance to Ryder Cove/Frost Fish Creek over a tidal cycle on September 1, 2000. Flood flows into the cove are positive (+), and ebb flows out of the cove are negative (-).

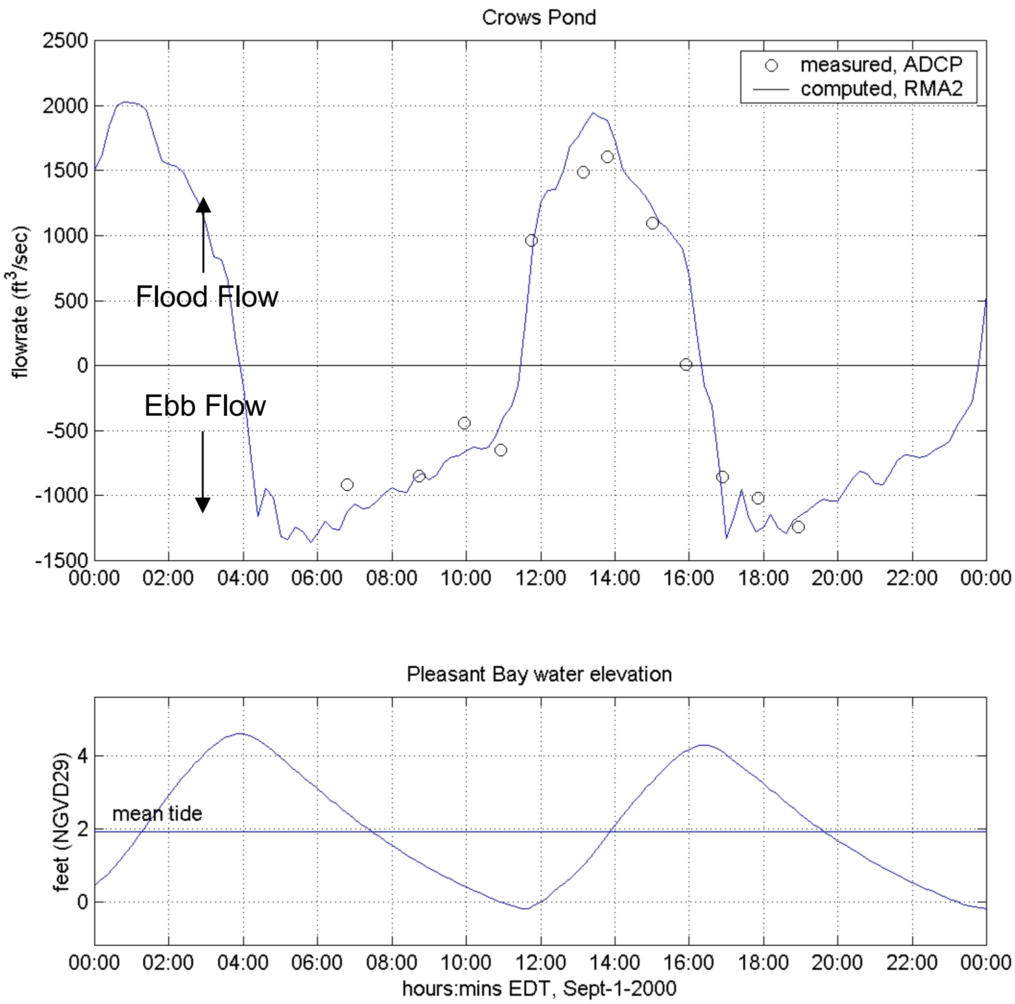


Figure V-60. Comparison of measured volume flow rates versus modeled flow rates through the mouth of Crows Pond over a tidal cycle on September 1, 2000. Flood flows into the pond are positive (+), and ebb flows out of the pond are negative (-).

V.5 FLUSHING CHARACTERISTICS

Since the magnitude of freshwater inflow is much smaller in comparison to the tidal exchange through each inlet, the primary mechanism controlling estuarine water quality within each of the modeled systems is tidal exchange. An exception in this study is Frost Fish Creek, where estimated groundwater inflow into the creek is slightly greater than 50% of the average tidal exchange through the Route 28 culverts, based on the average tidal flow 2.8 ft^3/sec (125,200 ft^3 per tide cycle) and estimated freshwater input of 1.6 ft^3/sec (annual average). A rising tide offshore in Nantucket Sound or Pleasant Bay 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 or Pleasant Bay 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.

V.5.1 Residence Times

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 Crows Pond as an example, the **system residence time** is the average time required for water to migrate from Crows Pond, through Bassing Harbor, and into Pleasant Bay, where the **local residence time** is the average time required for water to migrate from Crows Pond to Bassing Harbor. 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 Stage Harbor Region estuaries 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 were 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 future water quality model will provide a valuable tool to evaluate the complex mechanisms governing estuarine water quality in the Stage Harbor System, South Coast Embayments, and Pleasant Bay Region estuaries.

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in each estuary, model results were used to compute residence times. Residence times were computed for the entire estuary, as well as several sub-embayments within the estuary. In addition, **system** and **local residence times** were computed to indicate the range of conditions possible for each of the estuarine systems. Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism). Units then were converted to days. The volume of the entire estuary was computed as cubic feet. Residence times were averaged for the tidal cycles comprising the representative 7.25 day period (14 tide cycles), and are listed in Table V-12. Model divisions used to define the system sub-embayments listed in Tables V-11 and V-12 are shown in Figures V-61 through Figure V-65, in the previous section. The model calculated flow crossing specified grid lines for each sub-embayment to compute the tidal prism volume.

Generally, errors in computed residence times can be linked to two sources: the bathymetry information and simplifications employed to calculate residence time. Since the calibration period represented average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the various sub-embayments. The bathymetry data collection effort focused on regions of rapidly changing flow conditions (flow constrictions). This methodology provided an efficient and economical technique to measure bathymetric fluctuations affecting tidal flushing; however, the limited bathymetry survey associated with this study may have missed some shoals and/or deep holes introducing minor errors into the residence time calculations. In addition, limited topographic measurements were available on the extensive marsh plains of the South Coast Embayments.

Minor errors may be introduced in residence time calculations by simplifying assumptions. Flushing rate calculations assume that water exiting an estuary or sub-embayment does not return on the following tidal cycle. For regions where a strong littoral drift exists, this assumption is valid. However, water exiting a small sub-embayment on a relatively calm day may not completely mix with estuarine waters. In this case, the “strong littoral drift” assumption would lead to an under-prediction of residence time. Since littoral drift along the Nantucket Sound and Pleasant Bay shorelines in Chatham typically is strong and local winds induce tidal mixing within the regional estuarine systems, the "strong littoral drift" assumption only will cause minor errors in residence time calculations. Based on our knowledge of estuarine processes, we estimate that the combined errors due to bathymetric inaccuracies represented in the model grid and the "strong littoral drift" assumption are within 10% to 15% of "true" residence times.

Table V-11. Embayment mean volumes and average tidal prism during simulation period.

System	Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Stage Harbor	Stage Harbor (system)	142,825,500	107,176,900
	Mitchell R. / Upper Stage H.	40,210,100	20,729,200
	Mill Pond	19,067,900	8,349,300
	Little Mill Pond	3,394,400	1,312,400
	Oyster Pond River	42,797,000	35,598,500
	Oyster Pond	28,218,000	17,925,400
Sulphur Springs	Bucks Creek (system)	7,426,200	10,311,800
	Sulphur Springs	4,885,700	6,747,900
	Cockle Cove Creek	818,700	1,133,200
Taylors Pond	Mill Creek (system)	6,973,900	9,341,300
	Taylors Pond	3,145,600	2,003,100
Bassing Harbor	Bassing Harbor (system)	102,152,200	51,252,700
	Crows Pond	53,345,200	20,699,300
	Ryder Cove	19,385,600	12,805,800
	Frost Fish Creek	1,414,500	1,230,000
	Upper Frost Fish Creek	727,800	125,200
	Ryder Cove / Frost Fish Creek	30,338,500	18,967,900
Muddy Creek	Muddy Creek (system)	5,699,300	982,900

The relatively long residence time for a sub-embayment such as Cockle Cove Creek reveals the inadequacy of using system residence time alone to evaluate water quality. The system residence time is computed as 3.4 days, even though this marsh creek nearly goes dry at low tide. By the definition of system residence time, smaller sub-embayments have longer residence times; therefore, residence times may be misleading for small, remote parts of the estuary. Instead, it is useful to compute a local residence time for each sub-embayment. A local residence time represents the time required for a water parcel to leave the particular sub-embayment. For instance, the local residence time for Upper Frost Fish Creek represents the time required for a water parcel to be flushed from the upper portion of the creek into lower Frost Fish Creek. Local residence times are computed as the volume of the sub-embayment divided by the tidal prism of that sub-embayment, and units are converted to days. Table V-12 lists local residence times for several areas within each of the modeled systems.

Local residence times in Table V-12 are significantly lower than residence times based on the volume of the entire estuary. For example, flow entering Little Mill Pond on an average tidal cycle flushes through Stage Harbor inlet in 56.3 days, but flushes into Mill Pond in 1.3 days.

Generally, a local residence time is only useful where the adjacent embayment has high water quality. For some of the embayments located in the upper reaches of each system (again, Mill Pond and Frost Fish Creek), the receiving waters that exchange tidal flow with the various sub-embayments show signs of ecological stress, indicative of poor water quality. Therefore, system residence times may be more appropriate for future planning scenarios.

Table V-12. System and Local residence times (flushing rates) for Chatham sub-embayments.

System	Embayment	System Residence Time (days)	Local Residence Time (days)
Stage Harbor	Stage Harbor (system)	0.7	0.7
	Mitchell R. / Upper Stage H.	3.6	1.0
	Mill Pond	8.9	1.2
	Little Mill Pond	56.3	1.3
	Oyster Pond River	2.0	0.6
	Oyster Pond	4.1	0.8
Sulphur Springs	Bucks Creek (system)	0.4	0.4
	Sulphur Springs	0.6	0.4
	Cockle Cove Creek	3.4	0.4
Taylors Pond	Mill Creek (system)	0.4	0.4
	Taylors Pond	1.8	0.8
Bassing Harbor	Bassing Harbor (system)	1.0	1.0
	Crows Pond	2.6	1.3
	Ryder Cove	4.1	0.8
	Frost Fish Creek	43.0	0.6
	Upper Frost Fish Creek	422.3	3.0
	Ryder Cove / Frost Fish Creek	2.8	0.8
Muddy Creek	Muddy Creek (system)	3.0	3.0

Another important characteristic of system residence times is that values determined for each sub-embayment are directly dependent on what exactly the total system volume includes. This is readily apparent when a comparison of system residence time from the current report is made to values presented in previous flushing calculations for all of Pleasant Bay (ACI, 1997). For example, in the present study the system residence time for Crows Pond (in the Bassing Harbor system) is calculated to be *2.6* days, but from the earlier study the system residence time for Crows Pond is *68.6* days. The difference is due to the different system volumes used to compute each numbers, i.e., only the volume of the Bassing Harbor system (102,152,200 ft^3) in this study, and the volume of the entire Pleasant Bay (1,997,780,000 ft^3) for the earlier study. Alternatively, local residence times from these two studies show much closer agreement (1.3 days and 1.8 days, for this study and ACI, 1997 respectively), because these numbers are based on the volume of the same sub-embayment, Crows Pond in this case.

V.5.2 Pre-Breach Conditions

The formation of New Inlet in 1987 altered the hydrodynamics within the Pleasant Bay Estuary. As a result of the inlet, the tide range in Pleasant Bay has increased by approximately 1 ft, with a corresponding improvement to tidal flushing within the northern portions of the estuary. The inlet continues to migrate south and Nauset Beach will return to a morphology similar to the pre-breach form. This pattern of inlet formation and southerly growth of Nauset Beach is cyclical. The two most recent breaches through the Nauset barrier occurred in 1846 east of Allen Point and 1987 east of the Chatham Lighthouse. The anticipated cyclical behavior of the inlet system is based on the work of Geise (1988) who described the historical 1846 breach and the subsequent re-formation of Nauset Beach during the following 140 years. For comparison purposes, the pre-breach 1970's form of Nauset Beach is shown in Figure V-61 and the more efficient 1996 system is shown in Figure V-62.

The modeling effort presented above was performed for the existing (post-breach) conditions based on recently obtained bathymetric and tidal data, as well as information from a previous study of regional hydrodynamics (ACI, 1997). To simulate pre-1987 conditions when the Chatham Harbor/Pleasant Bay system was less hydraulically efficient, a revised model grid was developed as part of this previous modeling effort to simulate the pre-breach estuary (ACI, 1997). As a basis for the model grid, digital data obtained from historic NOAA surveys of the region were utilized to supplement the 1997 bathymetry data. Due to the orientation of the historic inlet, the pre-breach estuary was served by a combination of tides from the Atlantic Ocean and Nantucket Sound. The modeling analysis for the pre-breach estuary utilized Atlantic Ocean tides only (the measured 1997 Atlantic Ocean tide data was used to drive the model); however, an attempt was made to "calibrate" the model to the predicted amplitude damping and phase lags presented in the pre-breach NOAA Tide Tables.

To "calibrate" the pre-breach model, ACI matched the modeled tides to the historic amplitude damping and phase lag presented in the 1986 NOAA Tide Tables. For example, the mean tide range in the Atlantic Ocean offshore of Chatham was predicted to be 6.7 ft, with the tide range reducing to 3.6 ft in Chatham Harbor and 3.2 ft in Pleasant Bay. In general, the modeled pre-breach conditions compared well with the NOAA tide information.

The less efficient pre-breach inlet causes the mean tide range within the system to be reduced by approximately 1 ft (ACI, 1997). The reduction in tide range has a corresponding reduction in flow velocities and volume of water moved through the estuary and its sub-embayments. Pre-breach hydrodynamic characteristics were computed utilizing the models developed for Chatham's Pleasant Bay embayments, and a forcing tide generated from the ACI 1997 pre-breach model scenario. Figure V-63 shows the predicted 1997 Pleasant Bay tide, with the corresponding tide curves for the Bassing Harbor system and Muddy Creek developed as part of this study. A calculation of residence times was performed to evaluate the magnitude of the worst-case pre-breach scenario on tidal flushing. The results of this analysis are shown in Tables V-13 and V-4.

The information presented in Tables V-14 and V-15 indicates between a 10% and 88% increase in residence times for the sub-embayments within the Pleasant Bay Estuary. For most of the estuary, the increase in residence times was between 50% and 70%. There are two primary causes for the substantial increase in residence times for the Pleasant Bay systems: 1) an increase in mean sub-embayment volumes for pre-breach conditions, and 2) reduction in the tide range.

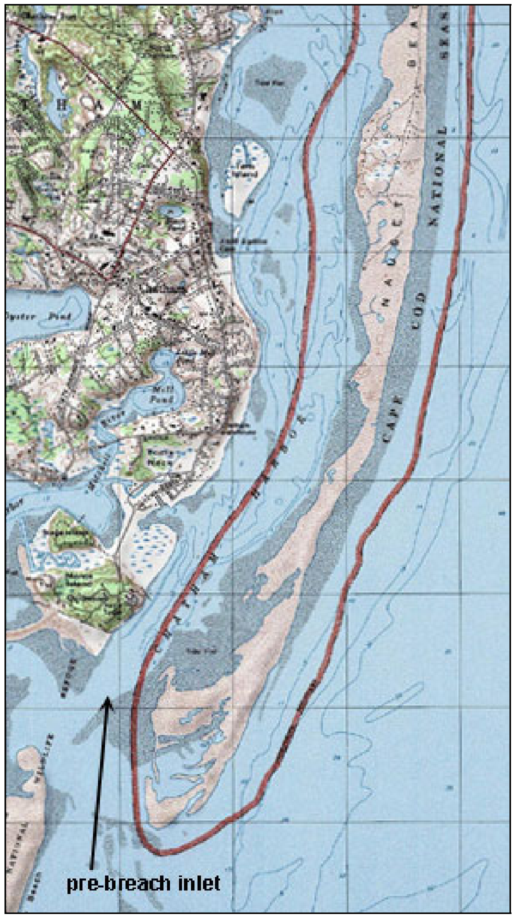


Figure V-61. Topographic map from the 1970's indicating the pre-breach inlet between Morris Island and Nauset Beach.

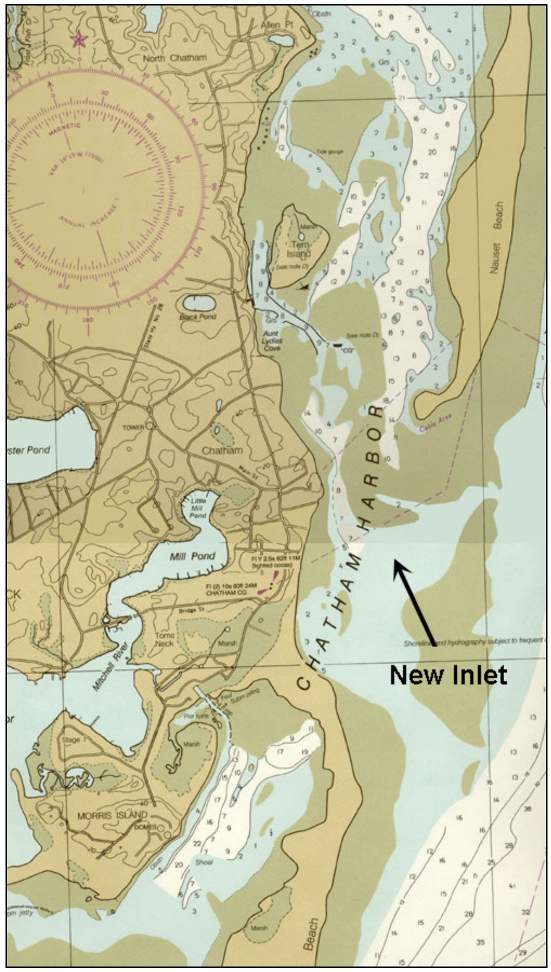


Figure V-62. Recent nautical chart indicating the location of New Inlet at the breach in Nauset Beach

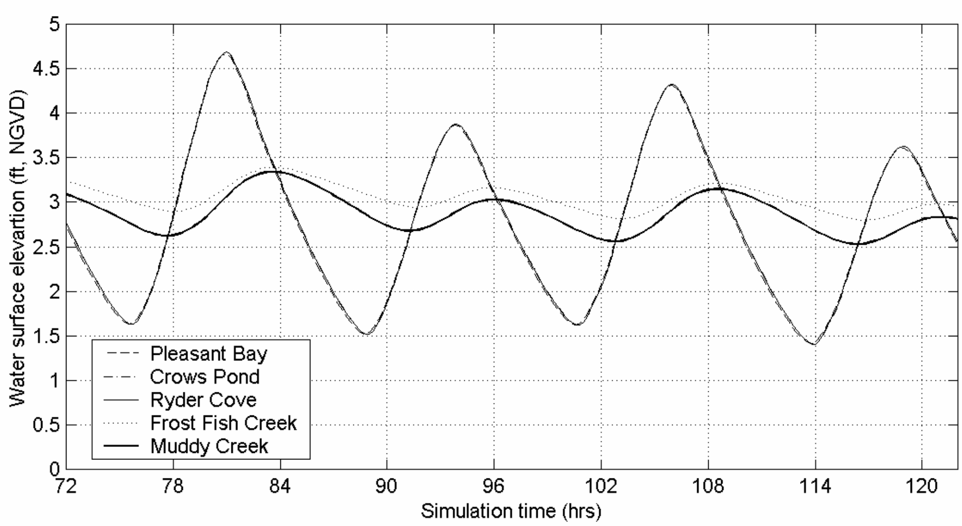


Figure V-63. Plot of two tide cycles of model run results for pre-breach conditions at Muddy creek and Bassing Harbor sub-embayments.

Table V-13. Embayment mean volumes and average tidal prism during simulation period for modeled pre-breach conditions in Pleasant Bay.			
System	Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Bassing Harbor	Bassing Harbor (system)	114,689,900	33,724,000
	Crows Pond	58,302,100	13,602,600
	Inner Ryder Cove	22,267,400	8,437,200
	Frost Fish Creek	1,744,700	898,300
	Upper Frost Fish Creek	768,600	119,021
	Ryder Cove / Frost Fish Creek	34,894,300	12,442,600
Muddy Creek	Muddy Creek (system)	6,496,875	870,315

Table V-14. System and Local residence times (flushing rates) for Pleasant Bay sub-embayments for modeled pre-breach conditions.			
System	Embayment	System Residence Time (days)	Local Residence Time (days)
Bassing Harbor	Bassing Harbor (system)	1.8	1.8
	Crows Pond	4.4	2.2
	Inner Ryder Cove	7.0	1.4
	Frost Fish Creek	66.1	1.0
	Upper Frost Fish Creek	498.7	3.3
	Ryder Cove / Frost Fish Creek	4.8	1.5
Muddy Creek	Muddy Creek (system)	3.9	3.9

Table V-15. Percent change in residence times from present conditions for Pleasant Bay sub-embayments for modeled pre-breach conditions.			
System	Embayment	System Residence Time change (%)	Local Residence Time change (%)
Bassing Harbor	Bassing Harbor (system)	80.0	80.0
	Crows Pond	69.2	69.2
	Inner Ryder Cove	70.7	75.0
	Frost Fish Creek	53.7	66.7
	Upper Frost Fish Creek	18.1	10.0
	Ryder Cove / Frost Fish Creek	71.4	87.5
Muddy Creek	Muddy Creek (system)	30.0	30.0

The sub-embayment mean volumes change due to the increased pre-breach mean tide level (approximately 2.1 ft NGVD for present conditions, and 3.1 ft NGVD for pre-breach conditions, from Table V-16). The reduction in tide range is the greatest of the two factors affecting flushing rates. Table V-16 shows mean-high-water and mean-low-water datums for pre- and post-breach conditions. The post-breach datums were computed using the TDR data collected in August and September 2000 for Ryder Cove. High water levels are similar, but low water levels differ by about 1 ft, therefore the mean tide range of the pre-breach condition is only about 68% of the mean tide range measured in this study. Finally, system and local residence times for each sub-embayment change by different percentages because the change in mean sub-embayment volumes verses mean system volumes is not equivalent. For example, the mean volume of the entire Bassing Harbor System (use for computing system residence times for all sub-embayments in the system) increases by 12% for pre-breach conditions, but the mean volume of Inner Ryder Cove (used to compute local residence time for Inner Ryder Cove) increases by 15%.

Within Muddy Creek, an anticipated increase in residence time of 30% is predicted by the model for the pre-breach conditions. Since the tide range within Pleasant Bay is reduced by approximately 1 ft for pre-breach conditions, the tidal exchange is greatly retarded through the Route 28 culverts. Larger culverts would allow better exchange of tidal waters between Muddy Creek and Pleasant Bay would limit the anticipated increase in residence times as the estuary returns to a pre-breach morphology.

Table V-16. Comparison of tide datums and mean tide levels for pre- and post-breach conditions, for Inner Ryder Cove. Elevations are relative to NGVD 29. Datums for present conditions were computed using TDR data collected in August and September 2000 in Ryder Cove.

Frost Fish Creek	Mean High Water (ft)	Mean Low Water (ft)	Mean Tide Level (ft)
Present conditions	4.2	0.0	2.1
Pre-breach conditions	4.5	1.7	3.1

V.6 ALTERNATIVES TO IMPROVE TIDAL FLUSHING

The two sub-embayments linked to the Pleasant Bay estuary by culverts (Muddy Creek and Frost Fish Creek) exhibit relatively poor tidal flushing. Water quality improvements to these systems likely can be achieved through either resizing of culverts or turning upper portions of the coastal embayments into freshwater ponds. Evaluation of potential alternatives is critical to achieve water quality goals, as well as to avoid adverse environmental impacts.

The hydrodynamic models utilized to evaluate tidal flushing provide the basis for *quantitatively* analyzing the effects of various alternatives on tidal exchange. Using the calibrated models for each system, the model grids were modified to reflect alterations in culvert dimensions and/or bathymetry. Numerical models provided a cost-effective means for evaluating several water quality improvement scenarios. Incorporating hydrodynamic and water quality models was utilized to streamline the alternative selection process.

V.6.1 Muddy Creek

The two culverts running under Route 28 at Muddy Creek each have a height of approximately 2.6 feet and a width of 3.7 feet. Since the surface area of Muddy Creek is relatively large, these culverts are not of sufficient size to allow complete tidal exchange between Pleasant Bay and Muddy Creek. This poor tidal exchange is likely responsible for the water quality concerns for the Muddy Creek system. In addition, replacement of these culverts will likely be an expensive alternative due to the large roadway embankment overlying the flow control structures.

Due to the elevation of Route 28 in this region, the roadway embankment prevents storm surge from overtopping the road and “shocking” the ecosystem in Muddy Creek with a pulse of higher salinity Pleasant Bay water. Therefore, turning Muddy Creek into a completely freshwater system is a viable alternative. Other alternatives considered include turning a portion of the system to freshwater and enlarging the culverts to improve tidal exchange.

V.6.1.1 Alternative M1 – Muddy Creek as a Freshwater System

Gates could be installed on the Pleasant Bay end of the existing culverts to convert the estuarine system to completely freshwater. As mentioned above, the Route 28 embankment would prevent floodwaters from overtopping the road; therefore, the freshwater ecosystem would remain stable during severe conditions. The gates would allow only unidirectional flow from Muddy Creek into Pleasant Bay. Periodic maintenance of the culvert gates would be required, due to their open exposure within Pleasant Bay. A potential environmental drawback to this alternative is the loss of salt marsh that exists within approximately the northern third of the estuary. Since this alternative would eliminate tidal exchange between Muddy Creek and Pleasant Bay, no modeling was performed to evaluate the effect of the gates on local hydrodynamics.

V.6.1.2 Alternative M2 – Muddy Creek as a Partial Freshwater System

To preserve the salt marsh in the lower portion of Muddy Creek and improve tidal flushing characteristics without altering the culvert configuration, a dike could be placed approximately ½ mile upstream from the roadway embankment (see Figure V-64). The region upstream of the dike would be maintained as a freshwater pond, again with a gate that only allowed unidirectional flow from the upper portion of Muddy Creek to the lower estuarine portion. Since the poor tidal exchange through the existing culverts is caused by the small cross-sectional area of the culverts relative to the surface area of Muddy Creek estuary, reducing the estuarine surface area will improve flushing characteristics. For example, hydrodynamic model simulations of dike placement as shown in Figure V-64, reduces the mean-tide estuarine volume by 55%; however, it causes very little reduction in tidal prism. The increase in tide range resulting from Alternative M2 is shown in Figure V-65. In addition, a comparison of tidal flushing improvements is shown in Table V-17.

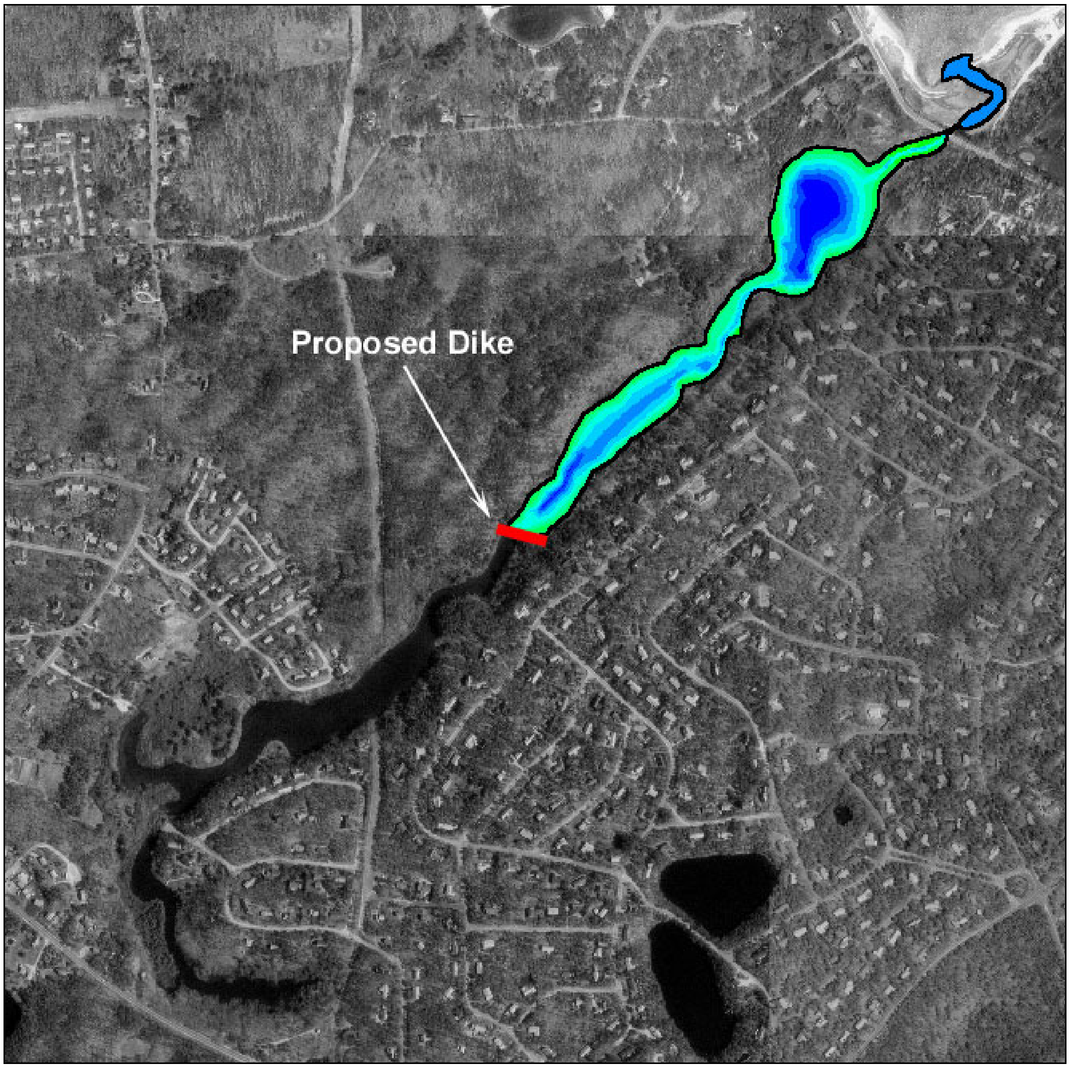


Figure V-64. Muddy Creek Alternative M2 illustrating the approximate position of the dike separating the freshwater and brackish regions.

Design considerations for the dike should include sufficient elevation to minimize potential overtopping during storm conditions. In addition, the freshwater pond level should be set at least 1.0 feet above the anticipated mean tide level in the estuarine section (about 3.5 feet NGVD according to Figure V-64) to ensure flow exits the freshwater section during all phases of the tide. A simple adjustable weir could be designed to fine-tune the water elevation in the freshwater section.

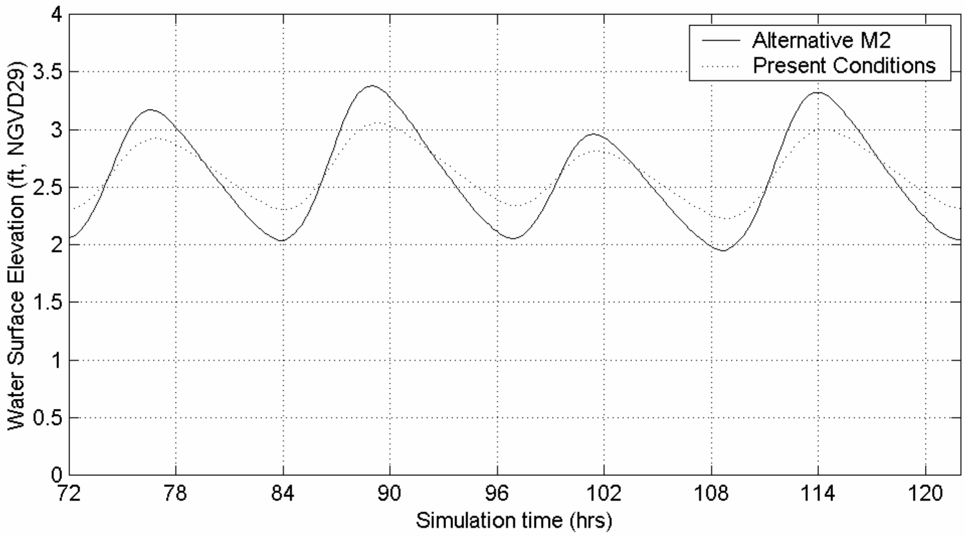


Figure V-65. Modeled tide range for Alternative M2 compared with present conditions.

V.6.1.3 Alternatives M3 and M4 – Increase Size of Route 28 Culverts

Although the Muddy Creek culverts are in good structural shape, it is possible that the Massachusetts Highway Department would consider culvert upgrading as part of the planned Route 28 improvements, if it clearly can be demonstrated that larger culverts are necessary to improve water quality. To assess tidal flushing improvements associated with larger culverts, two alternative culvert sizes were considered: a width of 8 feet and a width of 16 feet. Unlike the existing culverts, the culverts would be designed with a height similar to the tide range in Pleasant Bay (approximately 4.5 feet) to prevent the additional frictional drag associated with totally submerged culverts.

Table V-17 illustrates the change in tidal flushing associated with the two culvert alternatives. The smaller culvert alternative (Alternative M3) provided a similar tide range to Alternative M2. However, the residence time for Alternative M3 is similar to existing conditions, since the tidal prism increases by only about 20% and the mean-tide volume remains similar. Though the larger culvert alternative (Alternative M4) provided a significantly larger tide range, the reduction in residence time was not significantly greater than Alternative M2. The tidal curves for Alternatives M3 and M4 relative to existing conditions are shown in Figure V-66.

Table V-17. Comparison of system volume, tide prism, and residence tides for Muddy Creek for alternatives M2, M3, and M4.

Muddy Creek	system mean volume (ft^3)	tide prism volume (ft^3)	local residence time (days)
Present conditions	5,699,300	982,900	3.0
Alternative M2	3,150,700	957,500	1.7
Alternative M3	5,573,700	1,170,300	2.5
Alternative M4	5,404,600	2,816,100	1.0

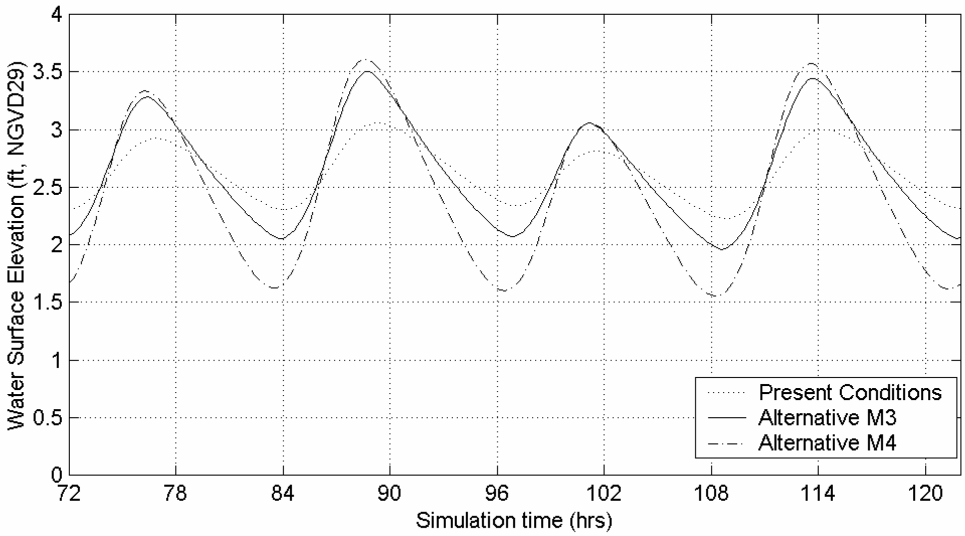


Figure V-66. Modeled tide range for Alternatives M3 and M4 compared with present conditions.

V.6.2 Frost Fish Creek

Two types of flow control structures exist at Frost Fish Creek. First, three partially-blocked 1.5 feet diameter culverts run under Route 28. Approximately 100 feet upstream of these culverts, a single large culvert and a dilapidated weir structure maintain the Creek level well above the mean tide elevation in adjoining Ryder Cove. Since the weir structure likely maintained Frost Fish Creek as a freshwater system, the culverts were adequate for handling the freshwater outflow from the Frost Fish Creek watershed. Following removal of the weir boards, Frost Fish Creek became a salt marsh system with a tide range of less than 0.5 feet. Similar to Muddy Creek, the size of the culverts limits tidal exchange with Ryder Cove and the rest of the Pleasant Bay estuary. The poor tidal exchange is likely responsible for the water quality concerns within Frost Fish Creek.

Since Route 28 in the vicinity of the creek culverts is below the predicted 100-year storm level, occasional overtopping of the roadway is anticipated. If the pond were maintained as a freshwater system, flooding would cause episodic increases in the pond salinity level, with the associated environmental impacts to wetland species. In addition, Frost Fish Creek presently supports a relatively healthy salt marsh system that would be destroyed by converting the system to freshwater. For these reasons, conversion of Frost Fish Creek back to a freshwater pond does not appear to be a feasible alternative. Instead, culvert options were considered to improve tidal exchange and enhance the existing salt marsh.

Since the existing culverts are partially clogged, the Massachusetts Highway Department has indicated a willingness to improve these structures as part of proposed work along Route 28. Two culvert alternatives were evaluated with the hydrodynamic model: Alternative F1 increased the tide range upstream of Route 28 to approximately 1.0 feet by installing a box culvert with a width of 5 ft and a height that allows the top of the culvert to remain above the water surface under most conditions; and Alternative F2 increased the tide range to approximately 1.5 feet by installing a box culvert with a width of 7 ft and a height that again allows the top of the culvert to remain above the water surface. An increase in tide range of greater than 1.5 feet may result in negative impacts to the marsh, because a greater portion of the marsh will be more frequently inundated with salt water; therefore, alternatives with larger culverts were not modeled. However, it may be feasible to reconstruct the weir upstream of Route 28 and utilize this structure to control tidal exchange and water elevations. Adjustment of the weir boards would allow “fine tuning” of the tide range within Frost Fish Creek. In this manner, culverts larger than those presented in Alternatives F1 and F2 below could be installed without impacting the marsh system.

Table V-18 illustrates the change in tidal flushing associated with the two culvert alternatives. The smaller culvert alternative (Alternative F1) provided a tide range of about 1.0 feet, with a significantly reduced local residence time of 1.3 days. The larger culvert alternative (Alternative F2) provided approximately a 1.5 ft tide range, as well as a lower residence time than Alternative F1. The tidal curves for Alternatives F1 and F2 relative to existing conditions are shown in Figure V-67. Due to the substantial tidal attenuation caused by the existing (partially blocked) culverts, the model indicated installation of larger culverts would significantly reduce the mean tide level with a negligible increase in the high tide elevation.

Table V-18. Comparison of system volume, tide prism, and residence tides for Frost Fish Creek for alternatives F1 and F2.

Frost Fish Creek	system mean volume (ft^3)	tide prism volume (ft^3)	local residence time (days)
Present conditions	727,800	125,200	3.0
Alternative F1	618,300	232,800	1.3
Alternative F2	596,000	358,600	0.9

V.6.3 Environmental Effects of Flushing Improvement Strategies

Concerns may arise regarding the potential of increased saltwater intrusion associated with enhancing tidal exchange to Muddy and Frost Fish Creeks. However, tidal embayments with poor tidal flushing characteristics generally have a mean tide level higher than the embayments closer to the ocean. For example, the mean tide level in the Atlantic Ocean offshore of Chatham is between 0.0 and 0.5 feet above NGVD, the mean tide level in Pleasant Bay is approximately 1.7 feet NGVD, and the mean tide level in Muddy Creek is about 2.5 feet NGVD. The hydrology of the estuarine system requires a sloping surface, with the highest long-term mean water level in the upper portions of the estuary and the lowest mean water levels in the ocean. For estuarine systems exhibiting little tidal attenuation, the change in mean water level through the system generally is small. As Figures V-65, V-66and V-67 indicate, the mean tide level is similar or lower than the existing mean tide level for each alternative.

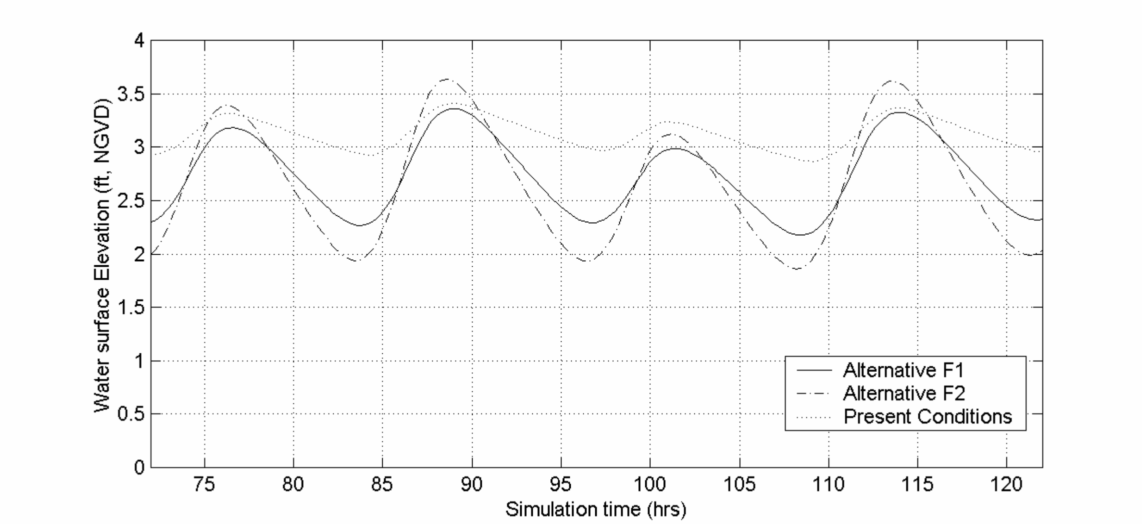


Figure V-67. Modeled tide range for Alternatives F1 and F2 compared with present conditions.

Due to the substantial tidal attenuation caused by the existing Frost Fish Creek culverts, the high tide level for the alternatives also remains similar to the existing high tide level. Only an increase in mean tide level will cause a measurable alteration to saltwater intrusion; therefore, the proposed tidal flushing improvements will have no negative impacts related to increased saltwater intrusion.

Creation of a freshwater system within Muddy Creek will enhance nitrogen attenuation. Since freshwater ponds and/or wetlands are often incorporated into nitrogen "removal" strategies, conversion of a portion of Muddy Creek to a freshwater system will provide two water quality improvement mechanisms: tidal exchange will be enhanced and the freshwater portion will provide natural attenuation of nitrogen. Prior to adopting this alternative for Muddy Creek, an evaluation of impacts to the brackish upper estuary needs to be performed. In addition, the future water quality modeling will analyze the improvements to total nitrogen concentrations that can be anticipated for each alternative.

V.7. SUMMARY

V.7.1 Conclusions

Tidal flushing of estuarine systems within the Stage Harbor System, the South Coast Embayments, and Pleasant Bay Region was evaluated using field measurements (Section V.3) and a calibrated hydrodynamic computer model (Section V.4). Field data included measured tides at eleven (11) locations, detailed depth measurements to augment previous bathymetric survey information, and current measurements taken along cross-channel transects. Field measurements of offshore tides in Pleasant Bay and Nantucket Sound, as well as depth measurements throughout the estuarine systems, provided input data to the computer models. Tide data collected within each sub-embayment were used to confirm the accuracy of the model simulations. For the Bassing Harbor and Stage Harbor systems, current measurements were used to verify the models calibrated with tide data. The computer model simulated water circulation in the estuary, including tides and currents. Two-dimensional current patterns, and water surface elevation were simulated by the model every twelve (12) minutes at thousands of points within each estuarine system. The modeled tides and currents were used to evaluate tidal flushing based on residence times and tidal circulation patterns.

A computer model was developed to simulate accurate tidal hydrodynamics in the Stage Harbor and Pleasant Bay Regions. The accuracy of model simulations was calibrated and verified by comparison to field data. The calibrated model provides a diagnostic tool for future analyses of water quality.

Based on the *local* residence time predictions alone, all of the embayments studied as part of the Stage Harbor system and South Coast Embayments (Stage Harbor, Sulphur Springs, and Taylors Pond) may be considered rapidly flushing systems. The rapid flushing rate of each system typically is an indicator of good relative water quality; however, each system has sub-embayments that exhibit signs of ecological stress, indicative of poor water quality. Therefore, the levels of nutrient loading likely controls water quality within the embayments (especially the upper portions of each system) to a greater degree than the hydrodynamic characteristics of each pond. In addition, it may be more appropriate to utilize *system* residence times to indicate estuarine health in the upper sub-embayments (e.g. Little Mill Pond), since the sub-embayments supplying these upper regions may have relatively poor water quality. For example, Little Mill Pond is flushed by waters traveling through Mill Pond, which exhibits signs of ecological stress.

Based on the *local* residence time predictions alone, much of the Bassing Harbor system may be considered rapidly flushing. Again, the rapid flushing rate of each system typically is an indicator of good relative water quality. The exception to the general rapid flushing of the Bassing Harbor system is Upper Frost Fish Creek (upstream of the Route 28 culverts). Substantial tidal attenuation occurs as a result of the flow restriction caused by under-sized culverts.

Similar to Upper Frost Fish Creek, Muddy Creek also shows substantial tidal attenuation as a result of the flow restriction created by culverts under Route 28. Although the Muddy Creek culverts are significantly larger than the Frost Fish Creek culverts, the greater surface area of the Muddy Creek estuarine system demands a much larger volume of water to raise the water level within the estuary.

The models were used to compute system and local residence times for existing conditions (Table V-12) in each estuarine system. Although tidal amplitude damping was greater across the Bucks Creek and Mill Creek systems than the Stage Harbor system, the limited water depth of these marsh-dominated estuaries (Bucks and Mill Creeks) produced lower overall residence times. Local residence times for the Pleasant Bay Region estuaries were similar to the Stage Harbor Region estuaries, with the exception of Muddy Creek and Frost Fish Creek. Local residence times for Muddy and Frost Fish Creeks (3.0 days for each) indicated reduced flushing for these areas.

Analysis of two-dimensional current patterns revealed that maximum currents within each estuary occurred within the inlets. For example, maximum flood currents were approximately 3.3 and 3.2 feet per second for the Stage Harbor and Bassing Harbor entrances, respectively.

Due to the rapidly changing geomorphology of the Chatham Harbor/Pleasant Bay entrance (New Inlet), a "worst-case" flushing analysis was performed utilizing historic pre-breach morphology and bathymetry. This analysis indicated that residence times would increase between 10 and 88 percent as the system returns to its pre-breach form.

The analysis of alternatives to improve tidal flushing in Frost Fish and Muddy Creeks indicated that a variety of options are available to dramatically improve tidal exchange through the Route 28 culverts. For Muddy Creek, placement of a dike at the approximate mid-point of the system (Figure V-64) would convert the upper half of the system into freshwater. Reduction in the surface area of the tidal portion would reduce the residence time by approximately 50%. Other options for Muddy Creek include increasing the size of the culverts and conversion of the entire estuary to a freshwater system. A modest increase in culvert size at Frost Fish Creek would more than double the tidal exchange. For both Muddy and Frost Fish Creeks, a more complete analysis of environmental impacts associated with improved tidal flushing should be performed prior to implementing project design.

VI. WATER QUALITY MODELING

VI.1 DATA SOURCES FOR THE MODEL

Several different data types and calculations are required to support the water quality modeling effort. These include the output from the hydrodynamics model, calculations of external nitrogen loads from the watersheds, measurements of internal nitrogen loads from the sediment (benthic flux), and measurements of nitrogen in the water column.

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayments

Extensive field measurements and hydrodynamic modeling of the embayments were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a set of five files of calibrated model output representing the transport of water within each of the five embayment systems. Files of node locations and node connectivity for the RMA-2V model grids were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic output for the water quality model calibration was a 14-tidal cycle period in summer 2000 that included both the neap and spring cycles.

VI.1.2 Nitrogen Loading to the Embayments

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

VI.1.3 Measured Nitrogen Concentrations in the Embayments

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The Town of Chatham Water Quality Laboratory, in conjunction with the Chatham Water Watchers (citizen volunteers), initiated a water quality monitoring program in the Stage Harbor system in the fall of 1998, and continued it through the summer of 1999. In 2000, sampling stations were added in the Sulphur Springs, Taylors Pond, Muddy Creek and Bassing Harbor systems (Duncanson, 2000). The sampling continued during 2001 and 2002. The goals of this program were to monitor existing water quality conditions, to provide data on the extent to which water quality was meeting goals or criteria, to compare conditions in the different embayments and their watersheds for targeting remedial actions, to help focus future studies on areas perceived as degraded, and to provide a long term data set for monitoring the success of remediation activities (Duncanson, 2000).

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Table VI-1A. Measured and modeled Nitrogen concentrations for Bassing Harbor and Muddy Creek, used in the model calibration plots of Figures VI-3 (Bassing Harbor total N),VI-4 (Bassing Harbor bio-active N), and VI-5 (Muddy Creek). All concentrations are given in mg/L N. “Data mean” values are calculated as the average of the separate yearly means.

System	Embayment	1999 mean	2000 mean	2001 mean	2002 mean	Overall mean	s.d.	N	model min	model average	model max
Bassing Harbor (TOTAL N)	Ryder Cove (inner)	-	0.465	0.634	0.653	0.569	0.183	46	0.556	0.564	0.573
	Ryder Cove (outer)	-	0.437	0.391	0.427	0.419	0.067	47	0.493	0.522	0.551
	Frost Fish Cr. (inner)	-	0.915	0.684	0.788	0.809	0.218	18	0.676	0.724	0.792
	Frost Fish Cr. (outer)	-	1.244	0.867	1.379	1.187	0.435	23	0.535	0.605	0.818
	Crows Pond	-	0.755	0.936	1.135	0.929	0.346	44	0.576	0.585	0.591
	Bassing Harbor	-	0.543	0.462	0.482	0.499	0.172	23	0.480	0.497	0.532
Bassing Harbor (Bio-Active N)	Ryder Cove (inner)	-	0.178	0.168	0.242	0.189	0.067	46	0.192	0.200	0.208
	Ryder Cove (outer)	-	0.167	0.139	0.191	0.163	0.036	47	0.129	0.158	0.187
	Frost Fish Cr. (inner)	-	-	0.364	0.409	0.387	0.065	10	0.312	0.360	0.428
	Frost Fish Cr. (outer)	-	0.391	0.307	0.290	0.338	0.173	23	0.171	0.241	0.454
	Crows Pond	-	0.220	0.200	0.232	0.218	0.095	44	0.212	0.221	0.227
	Bassing Harbor	-	0.156	0.108	0.131	0.133	0.037	23	0.116	0.133	0.168
Muddy Creek	Lower Muddy Cr.	-	0.569	0.591	0.622	0.586	0.092	21	0.557	0.597	0.658
	Upper Muddy Cr.	-	-	-	1.184	1.184	0.501	6	1.179	1.205	1.232

Table VI-1B. Measured and modeled Nitrogen concentrations for Stage Harbor, Sulphur Springs, and Taylors Pond, used in the model calibration plots of Figures VI-6 (Stage Harbor total N),VI-7 (Sulphur Springs), and VI-8 (Taylors Pond). All concentrations are given in mg/L N. "Data mean" values are calculated as the average of the separate yearly means.

System	Embayment	1999 mean	2000 mean	2001 mean	2002 mean	data mean	s.d.	N	model min	model average	model max
Stage Harbor*	Oyster Pond	0.597	0.786	0.708	0.604	0.667	0.252	63	0.671	0.678	0.687
	Lower Oyster Pond	-	-	0.552	0.498	0.505	0.083	8	0.371	0.547	0.658
	Oyster River	0.451	0.457	0.386	0.536	0.457	0.103	28	0.286	0.374	0.568
	Stage Harbor	0.425	0.664	0.632	0.677	0.597	0.182	58	0.288	0.339	0.427
	Upper Stage Harbor	0.418	0.457	0.503	0.548	0.474	0.116	62	0.382	0.401	0.423
	Mitchell River	-	-	0.429	0.487	0.451	0.092	13	0.403	0.432	0.467
	Mill Pond	0.471	0.503	0.418	0.507	0.463	0.102	70	0.466	0.473	0.485
	Little Mill Pond	0.792	0.690	0.742	0.741	0.733	0.226	60	0.696	0.711	0.723
Sulphur Springs	Mid Cockle Cove Cr.	-	1.492	2.043	1.613	1.685	0.698	18	0.704	1.378	2.493
	Cockle C. Cr. mouth	-	0.890	0.687	0.636	0.742	0.213	23	0.286	0.472	0.988
	Bucks Creek	-	0.401	0.479	0.576	0.473	0.139	20	0.285	0.337	0.508
	Sulphur Springs	-	0.360	0.453	0.584	0.451	0.123	23	0.288	0.369	0.498
Taylors Pond	Mill Creek	-	0.491	0.508	0.530	0.507	0.105	23	0.284	0.326	0.584
	Taylors Pond	-	0.509	0.487	0.530	0.508	0.122	48	0.424	0.467	0.517

* Stage Harbor also included the limited sampling data (N=4) from 1998.

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 five Chatham embayment systems. 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 Chatham embayments. Like RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. Applied Coastal staff have utilized this model in water quality studies of other Cape Cod embayments, including West Falmouth Harbor and the Falmouth "finger" ponds (Ramsey et al., 2000).

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 revised USGS watersheds), as well as the measured bottom sediment nitrogen fluxes. Water column nitrogen measurements by the Chatham Water Watchers 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.

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 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. The dispersion coefficients used in the sub-embayments of each of the five modeled systems were developed during the calibration process. During the calibration procedure, the dispersion coefficients were incrementally changed until model concentration outputs matched measured data.

The depth-averaged assumption is justified since vertical mixing by wind and tidal processes prevent significant stratification in these systems, even in the relatively deep kettle sub-embayments that are part of some of the Chatham embayments. This lack of stratification is evident in the temperature and salinity profiles of three such estuarine kettle ponds in Chatham, shown in Figure VI-1 and VI-2.

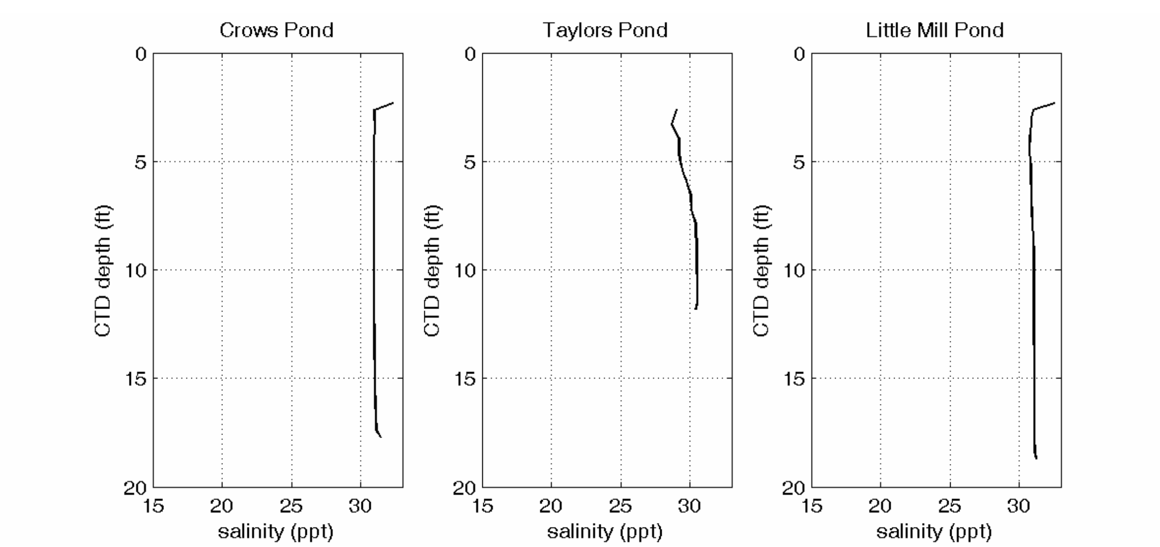


Figure VI-1. CTD cast salinity profiles for Crows Pond (Bassing Harbor), Taylors Pond, and Little Mill Pond (Stage Harbor). Cast data were recorded at 0.66 ft increments (0.2 m), during July 18 (Crows Pond), July 19 (Taylors Pond), and July 20 (Little Mill Pond) of 2000.

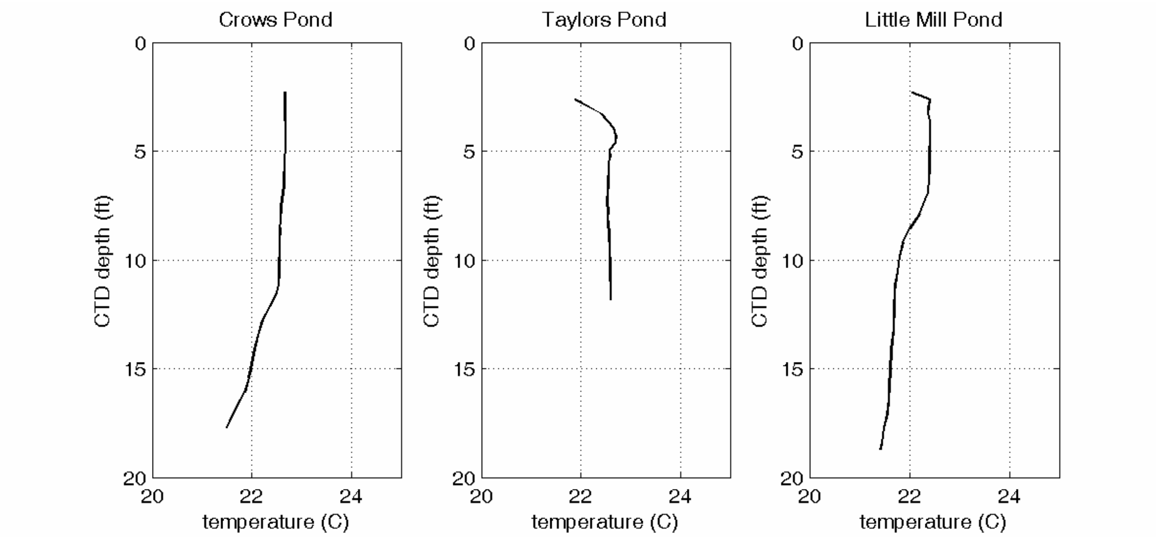


Figure VI-2. CTD cast temperature profiles for Crows Pond (Bassing Harbor), Taylors Pond, and Little Mill Pond (Stage Harbor). Cast data were recorded at 0.66 ft increments (0.2 m), during July 18 (Crows Pond), July 19 (Taylors Pond), and July 20 (Little Mill Pond) of 2000.

RMA-4 model can be utilized to predict both spatial and temporal variations in total 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 12-minute time intervals throughout the grid system. Therefore, the nitrogen concentrations within the coastal pond systems. For this application, the RMA-4 model was used to predict tidally averaged total nitrogen concentrations throughout the five estuarine systems in Chatham.

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 five Chatham sub-embayments also were used for the water quality constituent modeling portion of this study.

Based on updated groundwater recharge rates from the USGS, the Muddy Creek and Bassing Harbor hydrodynamic models were re-run. Muddy Creek and Frost Fish Creek (in the Bassing Harbor system) are the two sub-embayments where freshwater input is significant compared to the volume of water exchanged during a typical tide cycle. From the USGS, freshwater flux into Muddy Creek is 481,600 cubic feet/day, and 47,728 cubic feet/day for Frost Fish Creek. For Muddy Creek, the freshwater input during a single tide cycle (12.42 hours) is 25% of the tidal prism. In Frost Fish Creek, the freshwater recharge is 20% of the average tidal prism.

For each model, an initial total N concentration equal to the concentration at the open boundary was applied to the entire model domain. The model was then run for a simulated month-long (30 day) spin-up period. At the end of the spin-up period, the model was run for an additional 5 tidal-day (124 hour) period. Model results were recorded only after the initial spin-up. The time step used for the water quality computations was 12 minutes, which corresponds to the time step of the hydrodynamics input to each of the five Chatham systems.

VI.2.3 Boundary Condition Specification

Mass loading of nitrogen into each model included 1) sources developed from the results of the watershed analysis, 2) estimates of direct atmospheric deposition, and 3) summer benthic regeneration. Nitrogen loads from each separate sub-embayment watershed were distributed across the sub-embayment. For example, the loads from the Little Mill Pond watershed were evenly distributed at the grid cells that formed the perimeter of the pond. Similarly, benthic flux loads were distributed among grid cells in the central portions of each sub-embayment.

The loadings used to model present conditions in the five Chatham embayments are given in Table VI-2 for the South Coastal embayments and Stage Harbor, and Table VI-3 for the Pleasant Bay embayment systems. Watershed and depositional loads were taken from the results of the analysis of Section IV. Summertime benthic flux loads were computed based on the analysis of sediment cores in Section IV. The area rate (g/sec/m^2) of nitrogen flux from that analysis was applied to the surface area coverage computed for each sub-embayment (excluding marsh coverages, when present), resulting in a total flux for each embayment (as listed in Tables VI-2 and VI-3).

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundaries were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. Constituent concentrations of the incoming water are set at the value designated for the open boundary. For the south coast embayments (Taylors Pond and Sulphur Springs) and Stage Harbor, the boundary concentration in Nantucket Sound was set at 0.29 mg/L, based on Chatham Water Watchers data from the Sound (station CM-7). The open boundary condition for Bassing Harbor was set at 0.48 mg/L in Pleasant Bay (based on station PBA-20). For Muddy Creek, farther into Pleasant Bay, the boundary concentration was set at 0.50 mg/L (based on station PBA-6 and PBA-20). These total nitrogen concentration represent long-term average summer concentrations found within Nantucket Sound and appropriate regions of Pleasant Bay.

Table VI-2. Sub-embayment loads used for total nitrogen modeling of the Stage Harbor and South Coastal embayment systems, with total watershed N loads, atmospheric N loads, and benthic flux. These load represent present loading conditions for the listed sub-embayments.

sub-embayment	watershed load (kg/day)	atmospheric deposition (kg/day)	benthic flux (kg/day)
Stage Harbor			
Oyster Pond	13.03	0.29	26.8
Oyster River	11.47	1.05	0.7
Stage Harbor	2.76	3.25	12.8
Mitchell River	6.38	0.88	-3.4
Mill Pond	1.78	0.63	3.7
Little Mill Pond	1.64	0.12	2.0
Sulphur Springs			
Sulphur Springs	15.33	0.38	-3.6
Bucks Creek	4.08	0.13	2.9
Cockle Cove Creek	6.66	0.06	-0.9
Waste Water TF	3.03	-	-
Taylors Pond			
Mill Creek	6.22	0.17	-0.3
Taylors Pond	8.21	0.19	1.7

VI.2.4 Model Calibration

Calibration of each of the five Chatham embayment systems proceeded by changing model dispersion coefficients so that model output of nitrogen concentrations matched measured data. Generally, several model runs of each system were required to match the water column measurements. Dispersion coefficient (*E*) values were varied through the modeled systems by setting different values of *E* for each grid material type, as designated in Section V. Observed values of *E* (Fischer, *et al.*, 1979) vary between order 10 and order 1000 m^2/sec for riverine estuary systems characterized by relatively wide channels (compared to channel depth) with moderate currents. Coefficients in this range are appropriate for embayments with these characteristics, such as Oyster River (Stage Harbor) and Muddy Creek. Generally, the embayments of Chatham are small compared to the riverine estuary systems evaluated by Fischer, *et al.*, (1979); therefore the values of *E* also are relatively lower for Chatham. Smaller values of *E* occur in deeper and narrower, relatively quiescent sub-embayments, such as Taylors Pond and Crows Pond. 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 Tables VI-4 and VI-5. 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. Sub-embayment loads used for total nitrogen modeling of the Bassing Harbor and Muddy Creek systems of Pleasant Bay, with total watershed N loads, atmospheric N loads, and benthic flux. These load represent present loading conditions for the listed sub-embayments.			
sub-embayment	watershed load (kg/day)	atmospheric deposition (kg/day)	benthic flux (kg/day)
Bassing Harbor			
Crows Pond	5.79	1.39	3.5
Ryder Cove	12.35	1.30	7.4
Frost Fish Creek	3.59	0.10	-0.2
Bassing Harbor	2.66	1.08	-0.1
Muddy Creek			
Muddy Creek –lower	13.36	0.21	-1.9
Muddy Creek - upper	19.05	0.20	4.7

Table VI-4. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for the South Coastal embayments and Stage Harbor.	
Embayment Division	E m^2/sec
Stage Harbor System	
Oyster Pond - upper	1.5
Oyster Pond - lower	2.5
Oyster River	25.0
Little Mill Pond	0.01
Mill Pond	1.0
Mitchell River	10.0
Stage Harbor - upper	4.0
Stage Harbor – main basin	2.0
Stage Harbor - inlet	5.0
Sulphur Springs System	
Cockle Cove Creek – marsh	1.0
Cockle Cove Creek – channel	1.0
Sulphur Springs – basin	0.75
Sulphur Springs – marsh	2.0
Bucks Creek – marsh	2.0
Bucks Creek – channel	2.0
Bucks Creek – inlet to Nantucket Sound	1.0
Taylors Pond System	
Taylors Pond – basin	0.15
Mill Creek – upper channel	0.2
Mill Creek – lower channel	0.5
Mill Creek – marsh	0.05
Mill Creek – inlet to Nantucket Sound	1.0

Table VI-5. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for Bassing Harbor and Muddy Creek.

Embayment Division	E m^2/sec
Bassing Harbor System	
Ryder Cove - inner	10.0
Ryder Cove – outer	10.0
Crows Pond	0.1
Frost Fish Creek – upper (above culverts)	25.0
Frost Fish Creek - lower	10.0
Bassing Harbor – main basin	10.0
Bassing Harbor – Pleasant B. entrance	10.0
Muddy Creek System	
Muddy Creek – upper	10.0
Muddy Creek – mid	15.0
Muddy Creek – lower	90.0
Route 28 culvert	150.0
Entrance to Pleasant Bay	100.0

Comparisons between model output and measured nitrogen concentrations are shown in Figures VI-3 through VI-8 for each of the five modeled embayment systems. In each plot, annual means of the Water Watcher data, and the mean value of all the data at each individual station are plotted against the modeled maximum, mean, and minimum concentrations output from the model at locations which corresponds to the Water Watcher stations. Because the water samples are taken during ebbing tides, calibration targets in each sub-embayment were set such that the means of the measured data would fall within the range between the modeled maximum and modeled mean concentration, for stations where there is a wide range of modeled concentrations. This is demonstrated in plots of results from Frost Fish Creek (Figure VI-3) and Oyster River (Figure VI-6). At other locations (e.g., Ryder Cove and Muddy Creek), where the model exhibited less variability than the measured data, a calibration target near the mean of the Chatham Water Watcher data was selected.

For Bassing Harbor, an alternate calibration technique was employed (Figure VI-3) due to difficulties calibrating the model based on total N concentrations. Bio-active N (DIN+PON, without DON) concentrations were used for calibration due to elevated DON concentrations (relative to other sub-embayments in Bassing Harbor and in the other Chatham system) that exist in outer Frost Fish Creek and Crows Pond. The elevated DON concentrations in these sub-embayments are due to N fluxes not included in the N loading analysis from sources within the water column and from fresh water aquatic plants (more important for Frost Fish Creek). The water column DON pool is refractory, and therefore does not contribute significantly to phytoplankton production. Further discussion of the reasoning for using bio-active N concentrations for Bassing Harbor is given in Section VIII.

Calibrated model output is shown in Figures VI-9 through VI-13 for Stage Harbor, Sulphur Springs/Cockle Cove Creek, Taylors Pond/Mill Creek, Bassing Harbor, and Muddy Creek. In these figures, color contours indicate nitrogen concentrations throughout the model domain. The output in these figures show average total nitrogen concentrations, computed using the full

5-tidal-day model simulation output period. The range of the color scale used to indicate total N concentrations is the same for all five of these figures, to show conditions that exist in each system relative to the complete range of nitrogen concentrations observed in Chatham's embayments.

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 additional modeling step was not feasible in the modeled Chatham embayment systems because measured salinity data show only a slight gradient through to the uppermost reaches of each system (<1 ppt). The only exceptions are in Muddy Creek, Frost Fish Creek, and Cockle Cove Creek, which are brackish to fresh in their upper portions. Salinity modeling was not performed for these systems, however, because the existing salinity data does not provide enough information for adequate model verification. Also, modeling salinity requires extensive knowledge of freshwater inflow to the estuary. For systems where freshwater inflow is dominated by surface flows (e.g., rivers), direct measurement of the inflow is possible and salinity measurements can be utilized to assess dispersion of the freshwater into the estuary. Since Muddy Creek and Frost Fish Creek freshwater inputs are dominated by groundwater flow, no direct measurement of freshwater flow is available. Instead, the groundwater flow rate is assumed to be the long-term average and the freshwater input is evenly distributed around the shoreline. These simplifying, but necessary, assumptions prohibit use of salinity data to evaluate dispersion coefficients.

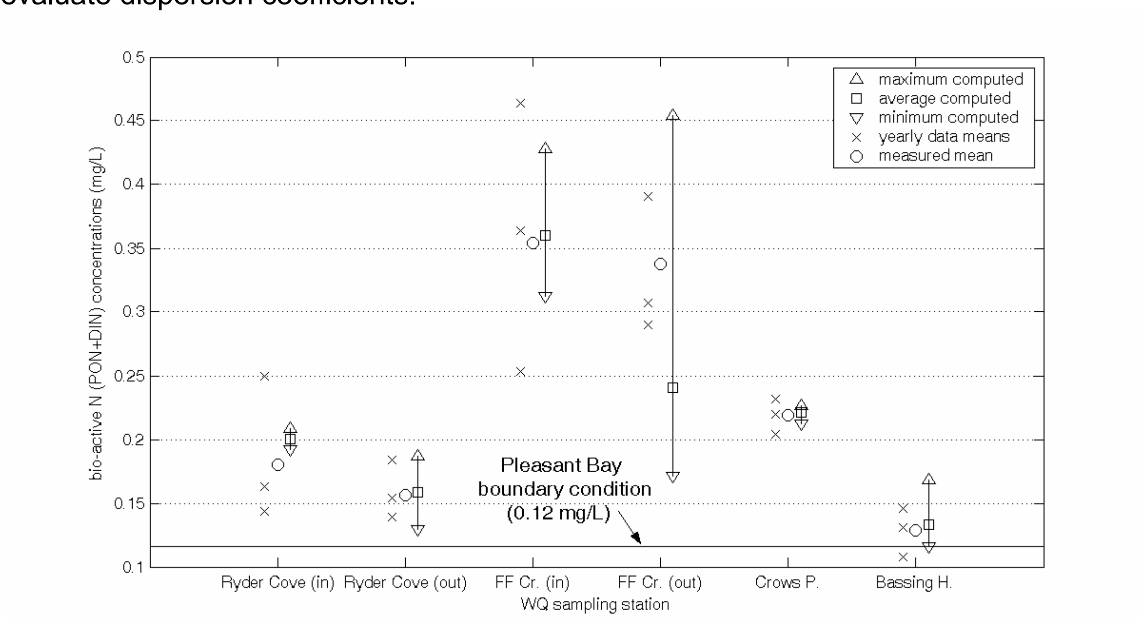


Figure VI-3. Comparison of measured bio-active nitrogen (PON+DIN) concentrations (means for individual years and means of all data together) and calibrated model output at stations in the Bassing Harbor system (with Frost Fish Creek, FF Cr.). Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). The background concentration (0.12 mg/L) in Pleasant Bay is indicated using a solid line.

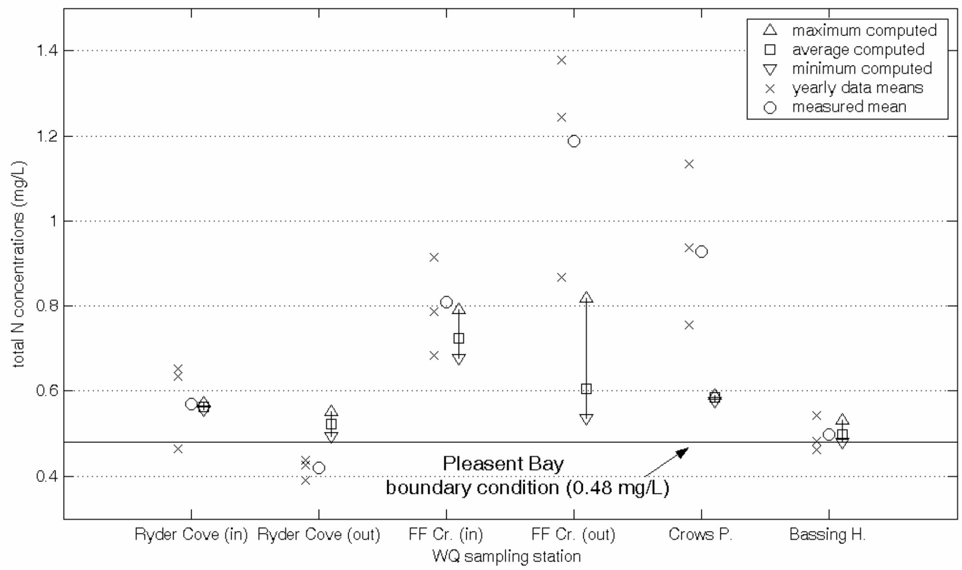


Figure VI-4. Comparison of measured total nitrogen (PON+DIN+DON) concentrations (means for individual years and means of all data together) and calibrated model output at stations in the Bassing Harbor system. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). The background concentration (0.48 mg/L) in Pleasant Bay is indicated using a solid line.

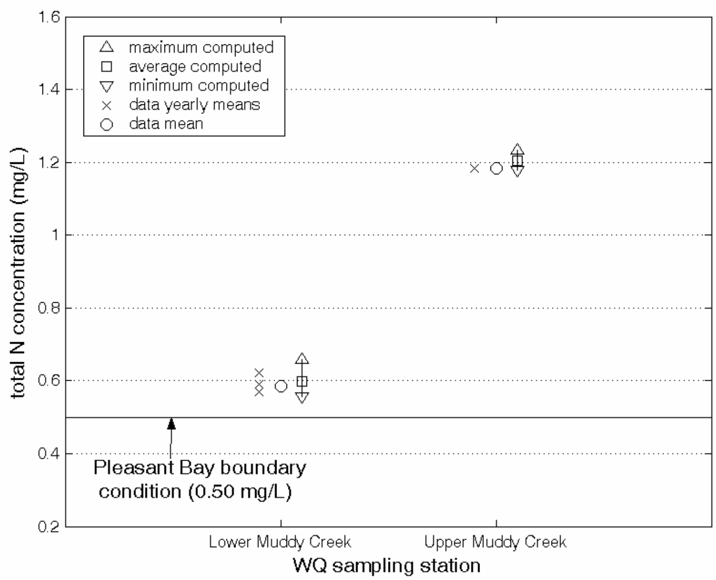


Figure VI-5. Comparison of measured total nitrogen concentrations (means for individual years and means of all data together) and calibrated model output at stations in the Muddy Creek system. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). The background concentration (0.50 mg/L) in Pleasant Bay is indicated using a solid line.

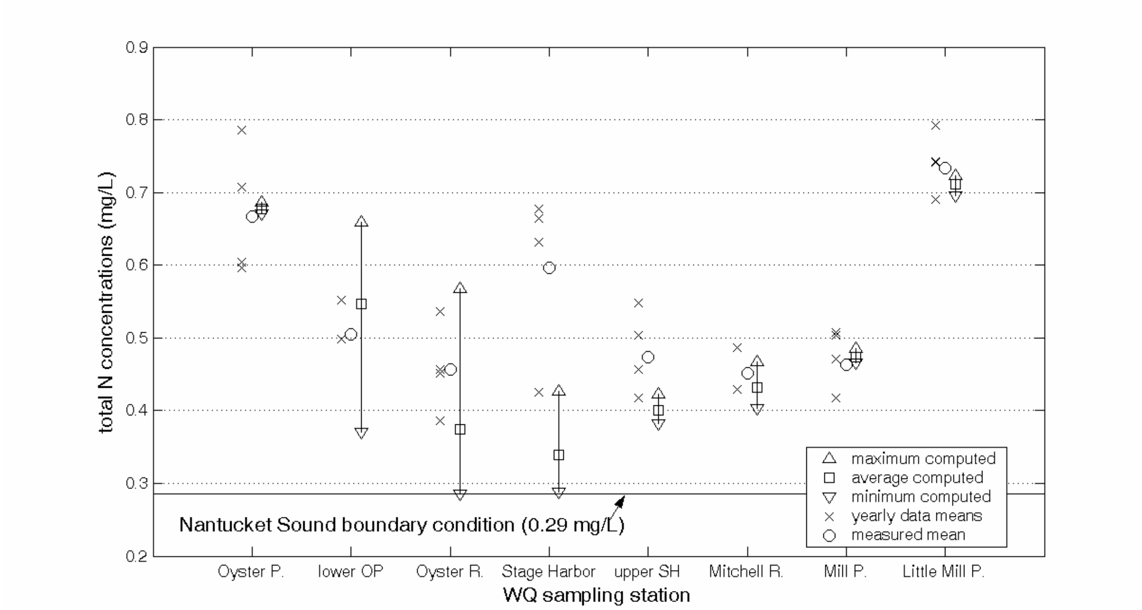


Figure VI-6. Comparison of measured total nitrogen concentrations (means for individual years and means of all data together) and calibrated model output at stations in the Stage Harbor system. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). The background concentration (0.29 mg/L) in Nantucket Sound is indicated using a solid line.

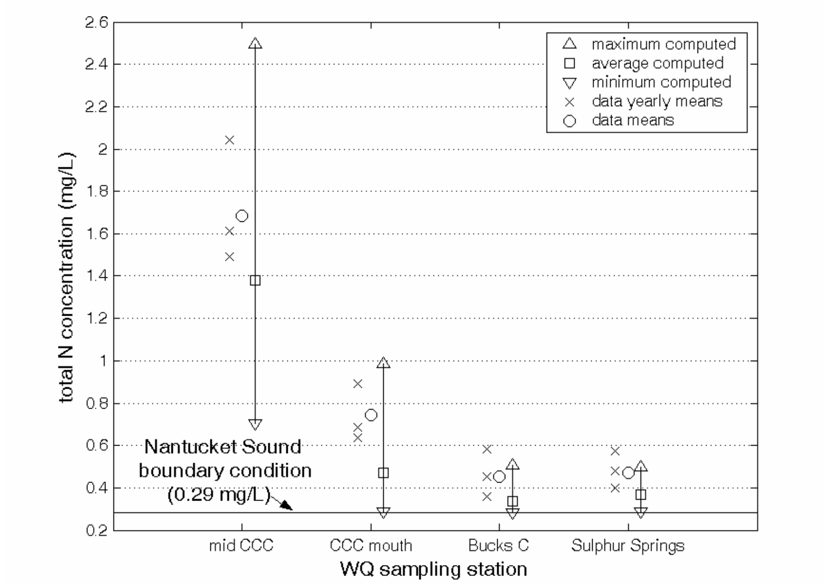


Figure VI-7. Comparison of measured total nitrogen concentrations (means for individual years and means of all data together) and calibrated model output at stations in the Sulphur Springs system, with Cockle Cove Creek (CCC). Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). The background concentration (0.29 mg/L) in Nantucket Sound is indicated using a solid line.

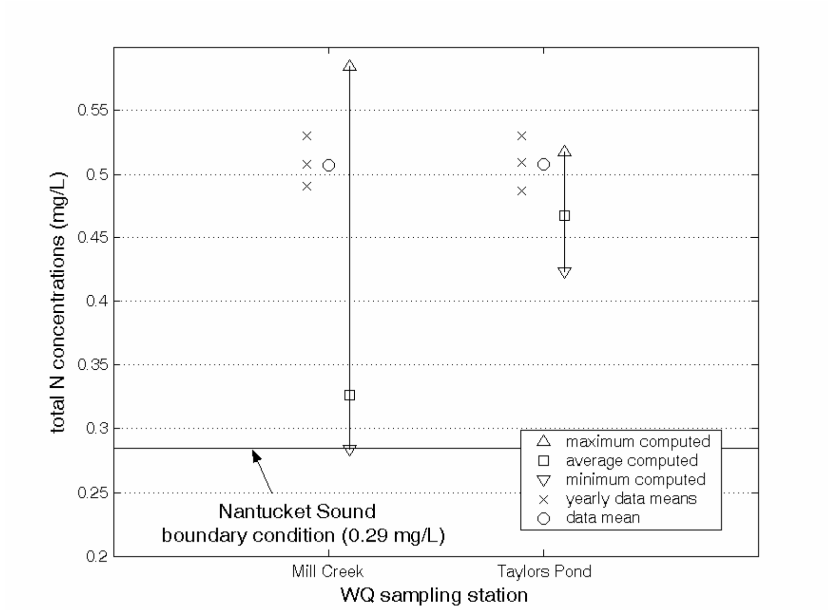


Figure VI-8. Comparison of measured total nitrogen concentrations (means for individual years and means of all data together) and calibrated model output at stations in the Taylors Pond system, with Mill Creek. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). The background concentration (0.29 mg/L) in Nantucket Sound is indicated using a solid line.

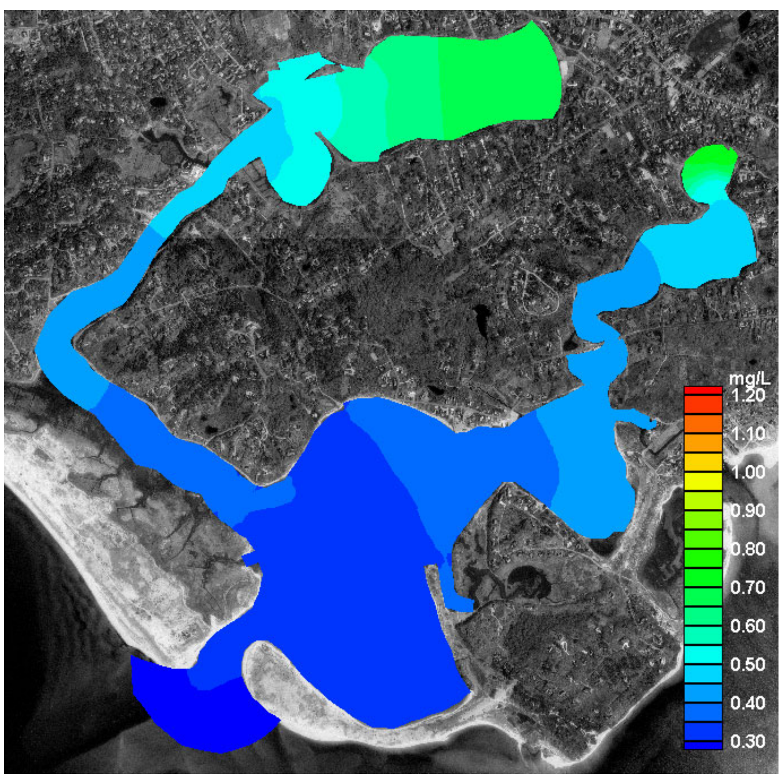


Figure VI-9. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for the Stage Harbor system.

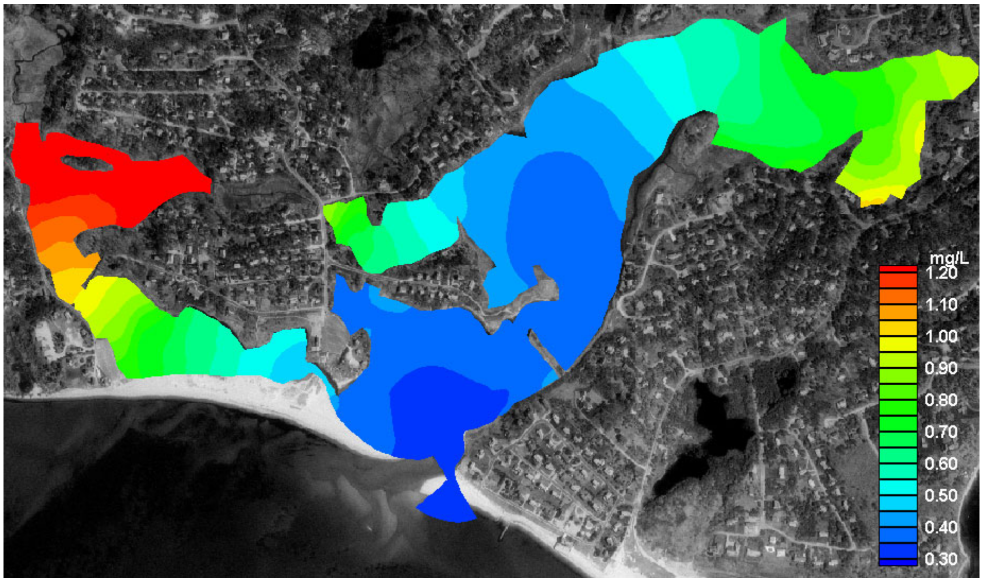


Figure VI-10. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Sulphur Springs/Cockle Cove Creek system, for present loading conditions.

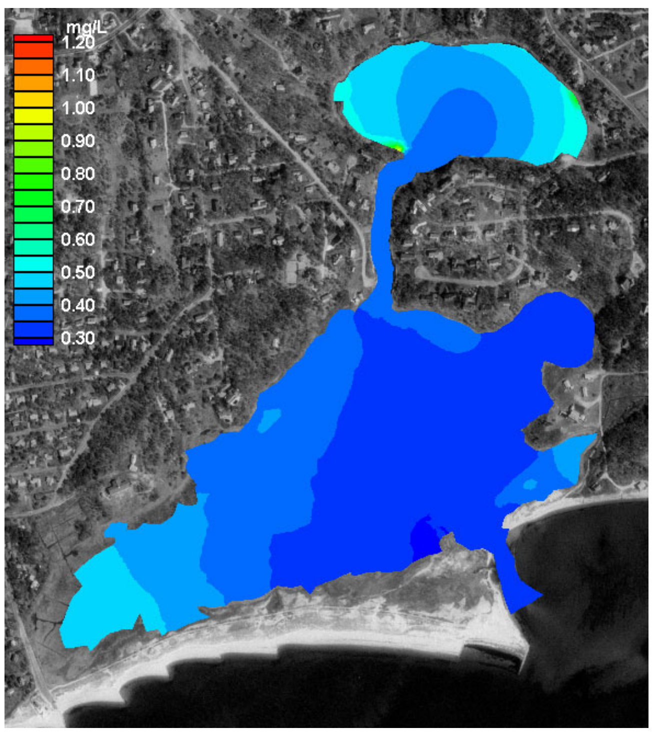


Figure VI-11. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Taylors Pond/Mill Creek system, for present loading conditions.

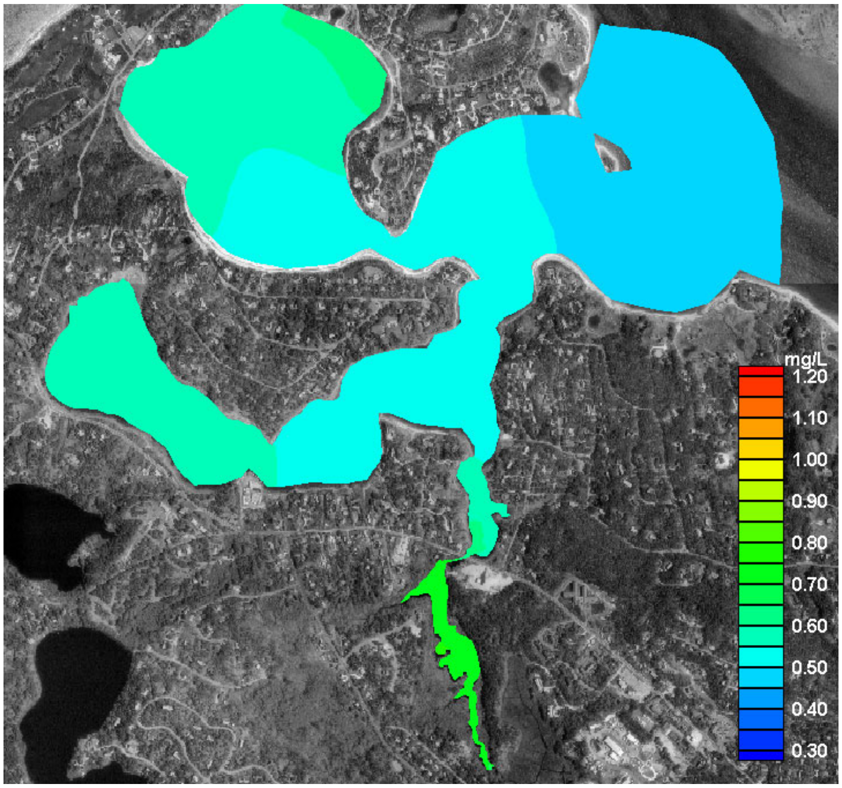


Figure VI-12. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Bassing Harbor system, for present loading conditions, and present background N concentration at the entrance to Pleasant Bay (0.48 mg/L).

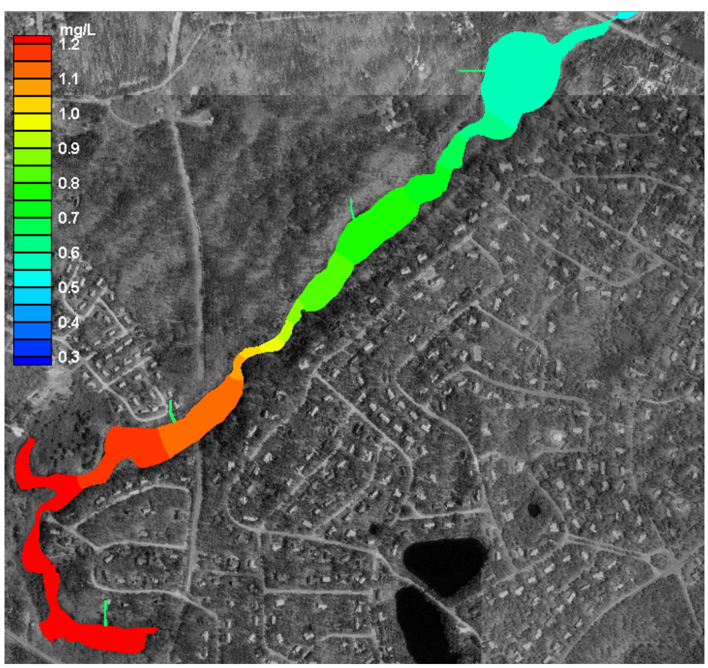


Figure VI-13. Contour plot of modeled total nitrogen concentrations in Muddy Creek, for present loading conditions, and present total nitrogen concentration in Pleasant Bay (0.50 mg/L).

VI.2.5 Build-Out and No Anthropogenic Load Scenarios

To assess the influence of nitrogen loading on total nitrogen concentrations within each of the embayment systems, two standard water quality modeling scenarios were run: a "build-out" scenario based on potential development (described in more detail in Section IV) and a "no anthropogenic load" or "no load" scenario assuming only atmospheric deposition on the watershed and sub-embayment, as well as a natural forest within each watershed. Comparisons of the watershed loading analyses are shown in Tables VI-6 and VI-7. Loads are presented in kilograms per day (kg/day) in this Section, since it is inappropriate to show benthic flux loads in kilograms per year due to seasonal variability. In general, the build-out scenario indicates that there would be less than a 20% increase in watershed nitrogen load as a result of potential future development. However, certain sub-embayments would be impacted more than others. A maximum increase in watershed loading resulting from future development would occur in the Taylors Pond watershed, where the increase would be 32.4%. For the no load scenarios, almost all of the load entering the watershed is removed; therefore, the load is generally lower than existing conditions by over 95%.

Table VI-6. Comparison of sub-embayment watershed loads used for modeling of present, build out, and no-anthropogenic ("no-load") loading scenarios of the Stage Harbor and South Coastal embayment systems. 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
Stage Harbor					
Oyster Pond	13.03	14.98	14.9%	0.64	-95.1%
Oyster River	11.47	12.74	11.1%	0.54	-95.3%
Stage Harbor	2.76	3.24	17.3%	0.16	-94.4%
Mitchell River	6.38	6.64	4.0%	0.16	-97.5%
Mill Pond	1.78	2.08	17.1%	0.06	-96.8%
Little Mill Pond	1.64	1.79	9.7%	0.04	-97.7%
Sulphur Springs					
Sulphur Springs	15.33	17.17	12.0%	0.45	-97.0%
Bucks Creek	4.08	4.83	18.4%	0.21	-95.0%
Cockle Cove Creek	6.66	7.98	19.8%	0.18	-97.3%
Waste Water TF	3.03	3.03	0.0%	0.00	-100.0%
Taylors Pond					
Mill Creek	6.22	7.17	15.2%	0.21	-96.6%
Taylors Pond	8.21	10.87	32.4%	0.27	-96.7%

For the build out scenario, a breakdown of the total nitrogen load entering each sub-embayment is shown in Tables VI-8 and VI-9. The benthic flux for the build-out scenarios is assumed to vary in a linear fashion, where an increase in watershed load will result in the same percentage increase (positive) in benthic flux. Due to the highly variable nature of bottom sediments and other estuarine characteristics of Chatham's coastal embayments, the measured benthic flux for existing conditions also is variable. For build-out conditions, some sub-embayments have approximately twice the benthic flux as total watershed load (e.g. Oyster Pond and Mill Pond). For other sub-embayments, the benthic flux is relatively low or negative (indicating a net uptake of nitrogen in the bottom sediments).

Table VI-7. Comparison of sub-embayment watershed loads used for modeling of present, build out, and no-anthropogenic ("no-load") loading scenarios of the Pleasant Bay embayment systems. These loads do not include atmospheric deposition and benthic flux loading terms.

sub-embayment	present load (kg/day)	build out (kg/day)	build out % change	no load (kg/dy)	no load % change
Bassing Harbor					
Crows Pond	5.79	6.04	4.4%	0.14	-97.6%
Ryder Cove	12.35	14.06	13.9%	0.45	-95.2%
Frost Fish Creek	3.59	3.88	8.0%	0.08	-97.7%
Bassing Harbor	2.66	3.22	20.9%	0.10	-96.4%
Muddy Creek					
Muddy Creek -lower	13.36	14.24	6.6%	0.50	-96.3%
Muddy Creek - upper	19.05	22.69	19.1%	0.87	-95.5%

Table VI-8. Sub-embayment loads used for modeling of buildout out scenarios in the Bassing Harbor and Muddy Creek systems of Pleasant Bay, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	atmospheric deposition (kg/day)	benthic flux (kg/day)
Bassing Harbor			
Crows Pond	6.04	1.39	3.9
Ryder Cove	14.06	1.30	8.1
Frost Fish Creek	3.88	0.10	-0.2
Bassing Harbor	3.22	1.08	-0.1
Muddy Creek			
Muddy Creek –lower	14.24	0.21	-2.1
Muddy Creek - upper	22.69	0.20	5.3

Following development of the various nitrogen loading estimates for the build out scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Total nitrogen concentrations in the receiving waters (Nantucket Sound or Pleasant Bay) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from build out was relatively small as shown in Tables VI-10 and VI-11. These results are shown pictorially in Figures VI-14 to VI-18. Again, the range of nitrogen concentrations shown represent the complete range of total nitrogen values observed in Chatham's coastal embayments. This allows direct comparison of nitrogen concentrations between regional embayment systems.

Table VI-9. Sub-embayment loads used for modeling of build out scenarios of the Stage Harbor and South Coastal embayment systems, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	atmospheric deposition (kg/day)	benthic flux (kg/day)
Stage Harbor			
Oyster Pond	14.98	0.29	29.3
Oyster River	12.74	1.05	0.7
Stage Harbor	3.24	3.25	14.0
Mitchell River	6.64	0.88	-3.8
Mill Pond	2.08	0.63	4.0
Little Mill Pond	1.79	0.12	2.2
Sulphur Springs			
Sulphur Springs	17.17	0.38	-4.1
Bucks Creek	4.83	0.13	3.3
Cockle Cove Creek	7.98	0.06	-1.0
Waste Water TF	3.03	-	-
Taylors Pond			
Mill Creek	7.17	0.17	-0.4
Taylors Pond	10.87	0.19	2.2

Table VI-10. Comparison of model average total N concentrations from present loading and build out scenario, with percent change, for South Coastal embayments and Stage Harbor.

sub-embayment	present (mg/L)	build out (mg/L)	% change
Stage Harbor			
Oyster Pond –upper	0.68	0.72	6.4%
Oyster Pond – lower	0.55	0.58	5.2%
Oyster River	0.37	0.38	2.5%
Stage Harbor – main	0.34	0.34	1.6%
Stage Harbor – upper	0.40	0.41	2.7%
Mitchell River	0.43	0.45	3.2%
Mill Pond	0.47	0.49	3.8%
Little Mill Pond	0.71	0.75	5.6%
Sulphur Springs			
Cockle Cove Cr. – mid	1.38	1.49	8.3%
Cockle Cove Cr. – low	0.47	0.50	5.1%
Bucks Creek	0.34	0.34	2.0%
Sulphur Springs	0.37	0.38	2.7%
Taylors Pond			
Mill Creek	0.33	0.33	2.3%
Taylors Pond	0.47	0.52	11.6%

Table VI-11. Comparison of model average total N concentrations from present loading and build out scenario, with percent change, for Pleasant Bay embayment systems.

sub-embayment	present (mg/L)	build out (mg/L)	% change
Bassing Harbor			
Ryder Cove – inner	0.56	0.57	1.6%
Ryder Cove – outer	0.52	0.53	0.8%
Frost Fish Creek - out	0.72	0.74	2.8%
Frost Fish Creek – in	0.60	0.62	2.0%
Crows Pond	0.59	0.59	1.2%
Bassing Harbor	0.50	0.50	0.3%
Muddy Creek			
Muddy Creek –lower	0.60	0.61	2.4%
Muddy Creek - upper	1.21	1.32	9.9%

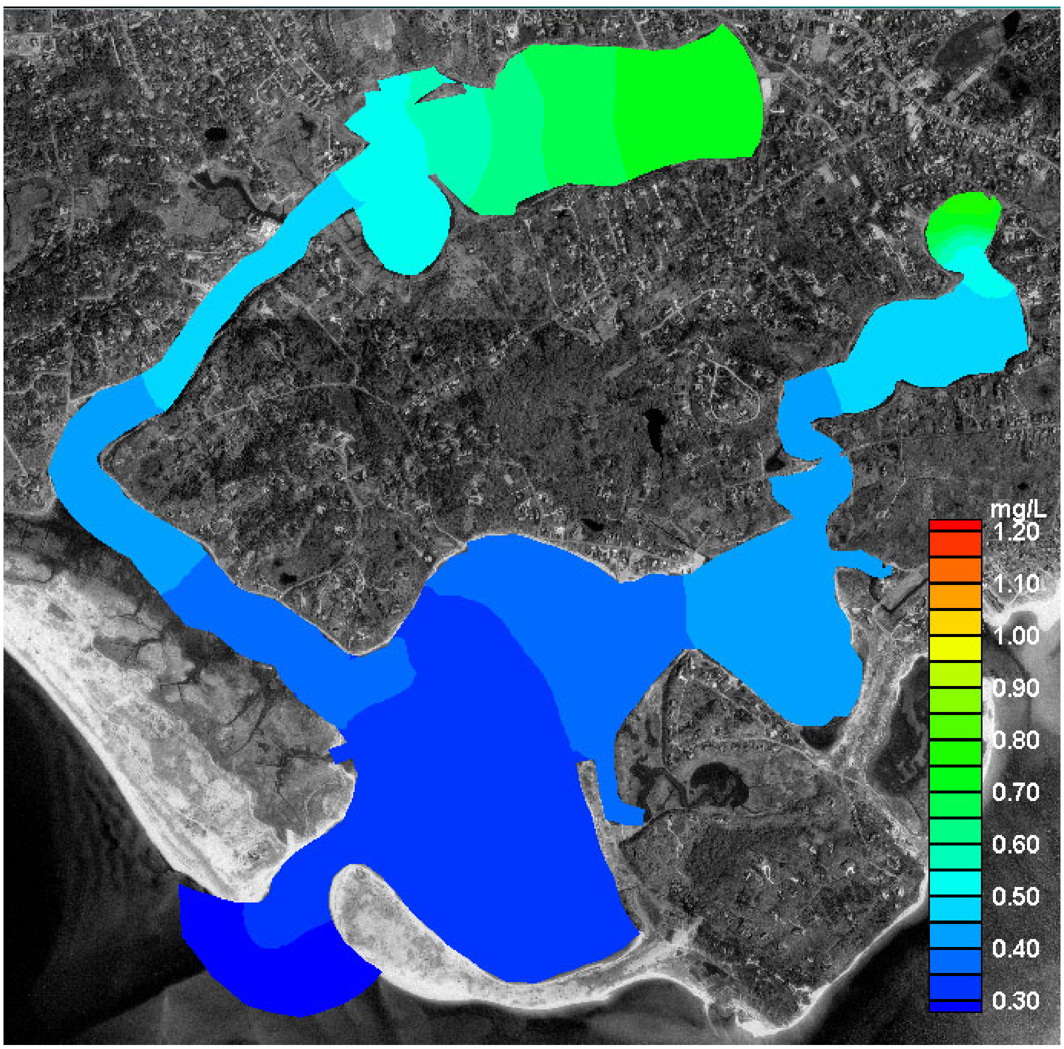


Figure VI-14. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Stage Harbor system, for projected build out loading conditions.

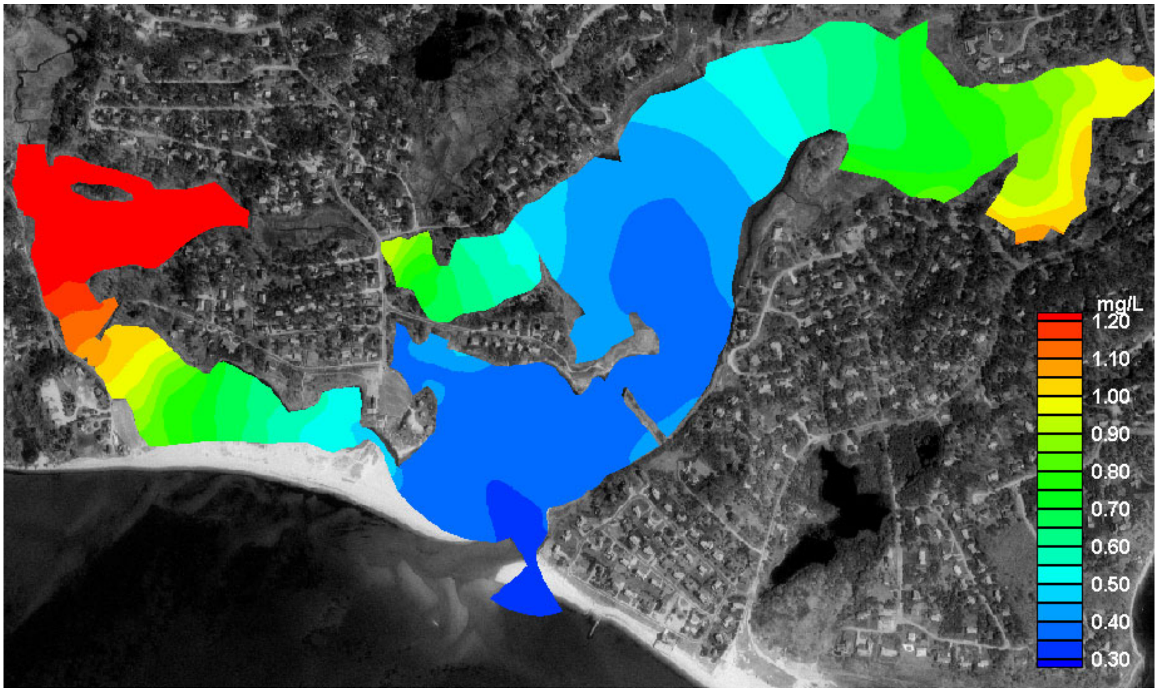


Figure VI-15. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Sulphur Springs/Cockle Cove Creek system, for projected build out loading conditions

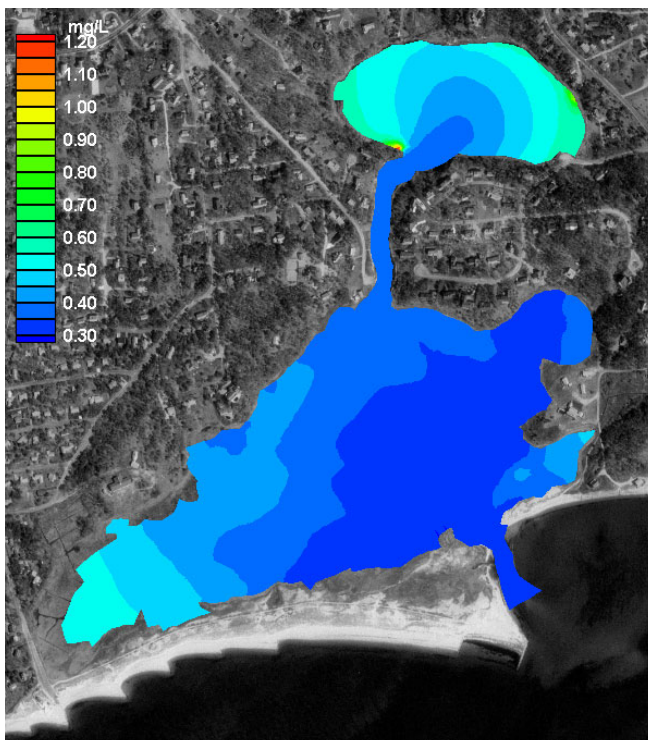


Figure VI-16. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Taylors Pond/Mill Creek system, for projected build out loading conditions.

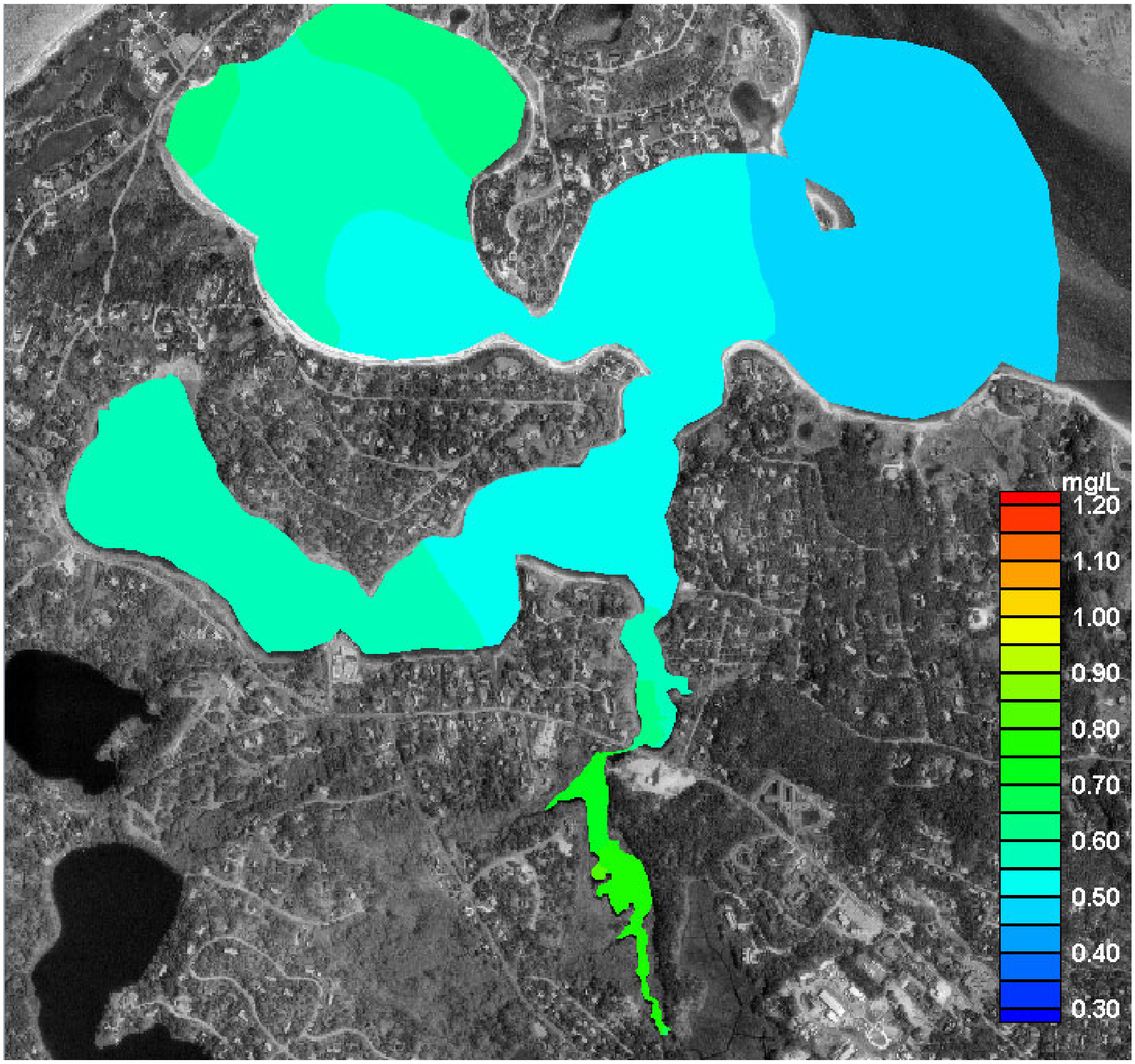


Figure VI-17. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Bassing Harbor system, for projected build out loading conditions, and present background N concentration at the entrance to Pleasant Bay (0.48 mg/L).

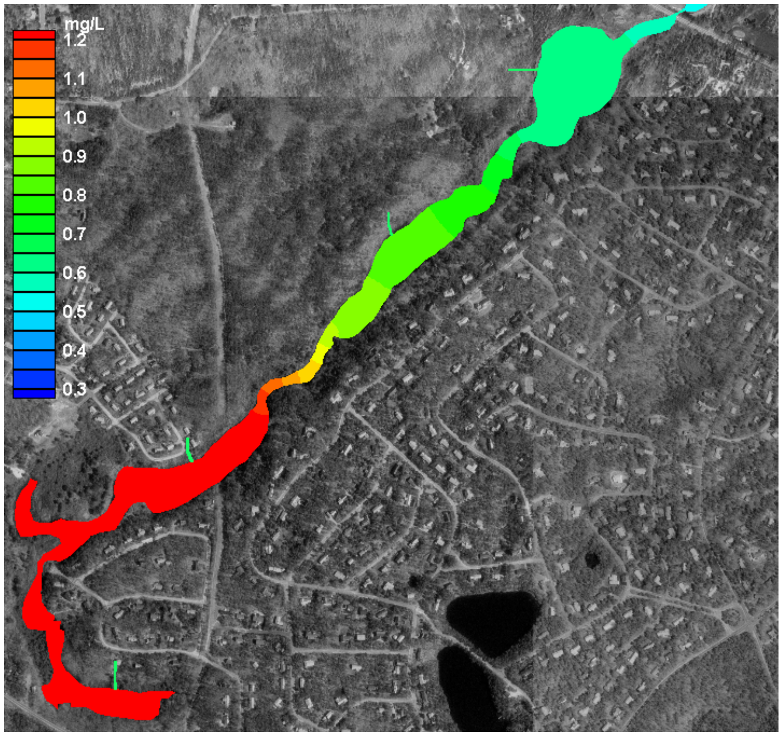


Figure VI-18. Contour plot of modeled total nitrogen concentrations in Muddy Creek, for projected build out loading conditions, and present total nitrogen concentration in Pleasant Bay (0.50 mg/L).

A breakdown of the total nitrogen load entering each sub-embayment for the no anthropogenic load scenarios is shown in Tables VI-12 and VI-13. The benthic flux for the "no load" scenarios is assumed to vary in a linear fashion, where a decrease in watershed load will result in the same percentage decrease in benthic flux. Due to the highly variable nature of bottom sediments and other estuarine characteristics of Chatham's coastal embayments, the measured benthic flux for existing conditions also is variable. For no load conditions, some sub-embayments have a benthic load that is significantly larger than the watershed load (e.g. Oyster Pond and Stage Harbor). Additionally, atmospheric deposition directly to each sub-embayment becomes a greater percentage of the total nitrogen load as the watershed load and related benthic flux decrease.

Following development of the various 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 (Nantucket Sound or Pleasant Bay) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was relatively significant as shown in Tables VI-14 and VI-15. These results are shown pictorially in Figures VI-19 to VI-23. Again, the range of nitrogen concentrations shown represent the complete range of total nitrogen values observed in Chatham's coastal embayments. This allows direct comparison of nitrogen concentrations between regional embayment systems. For the no load scenario, the sub-embayment concentrations are generally governed by the total nitrogen concentrations observed in the local receiving waters, where the concentrations in Stage Harbor, Sulphur Springs/Cockle Cove Creek, and Taylors Pond/Mill Creek are dictated by Nantucket Sound, and the concentrations in Bassing Harbor and Muddy Creek are dictated by Pleasant Bay. For the embayment systems serviced by Nantucket Sound waters, total nitrogen concentrations were below 0.35 mg/L..

Table VI-12. Sub-embayment loads used for modeling of no-anthropogenic loading scenarios of the Stage Harbor and South Coastal embayment systems, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	atmospheric deposition (kg/day)	benthic flux (kg/day)
Stage Harbor			
Oyster Pond	0.64	0.29	4.8
Oyster River	0.54	1.05	0.1
Stage Harbor	0.16	3.25	2.3
Mitchell River	0.16	0.88	-0.6
Mill Pond	0.06	0.63	0.7
Little Mill Pond	0.04	0.12	0.4
Sulphur Springs			
Sulphur Springs	0.45	0.38	-0.2
Bucks Creek	0.21	0.13	0.2
Cockle Cove Creek	0.18	0.06	-0.1
Waste Water TF	0.00	-	-
Taylors Pond			
Mill Creek	0.21	0.17	0.0
Taylors Pond	0.27	0.19	0.1

Table VI-13. Sub-embayment loads used for modeling of no-anthropogenic loading scenarios in the Bassing Harbor and Muddy Creek systems of Pleasant Bay, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	atmospheric deposition (kg/day)	benthic flux (kg/day)
Bassing Harbor			
Crows Pond	0.14	1.39	0.6
Ryder Cove	0.45	1.30	1.4
Frost Fish Creek	0.08	0.10	0.0
Bassing Harbor	0.10	1.08	0.0
Muddy Creek			
Muddy Creek –lower	0.50	0.21	-0.1
Muddy Creek - upper	0.87	0.20	0.3

Table VI-14. Comparison of model average total N concentrations from present loading and the no anthropogenic ("no load") scenario, with percent change, for South Coastal embayments and Stage Harbor. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).

sub-embayment	present (mg/L)	no load (mg/L)	% change
Stage Harbor			
Oyster Pond –upper	0.68	0.34	-49.6%
Oyster Pond – lower	0.55	0.32	-40.9%
Oyster River	0.37	0.30	-20.3%
Stage Harbor – main	0.34	0.29	-13.2%
Stage Harbor – upper	0.40	0.31	-23.9%
Mitchell River	0.43	0.31	-28.0%
Mill Pond	0.47	0.32	-32.9%
Little Mill Pond	0.71	0.35	-51.2%
Sulphur Springs			
Cockle Cove Cr. – mid	1.38	0.30	-77.9%
Cockle Cove Cr. – low	0.47	0.29	-38.7%
Bucks Creek	0.34	0.29	-14.7%
Sulphur Springs	0.37	0.29	-21.6%
Taylors Pond			
Mill Creek	0.33	0.29	-11.9%
Taylors Pond	0.47	0.30	-36.8%

Table VI-15. Comparison of model average total N concentrations from present loading and the no anthropogenic ("no load") scenario, with percent change, for Pleasant Bay embayment systems. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).

sub-embayment	present (mg/L)	no load (mg/L)	% change
Bassing Harbor			
Ryder Cove – inner	0.56	0.49	-12.7%
Ryder Cove – outer	0.52	0.49	-6.9%
Frost Fish Creek - out	0.72	0.50	-31.3%
Frost Fish Creek – in	0.60	0.49	-18.5%
Crows Pond	0.59	0.50	-14.5%
Bassing Harbor	0.50	0.48	-2.9%
Muddy Creek			
Muddy Creek –lower	0.60	0.50	-16.2%
Muddy Creek - upper	1.21	0.53	-55.7%

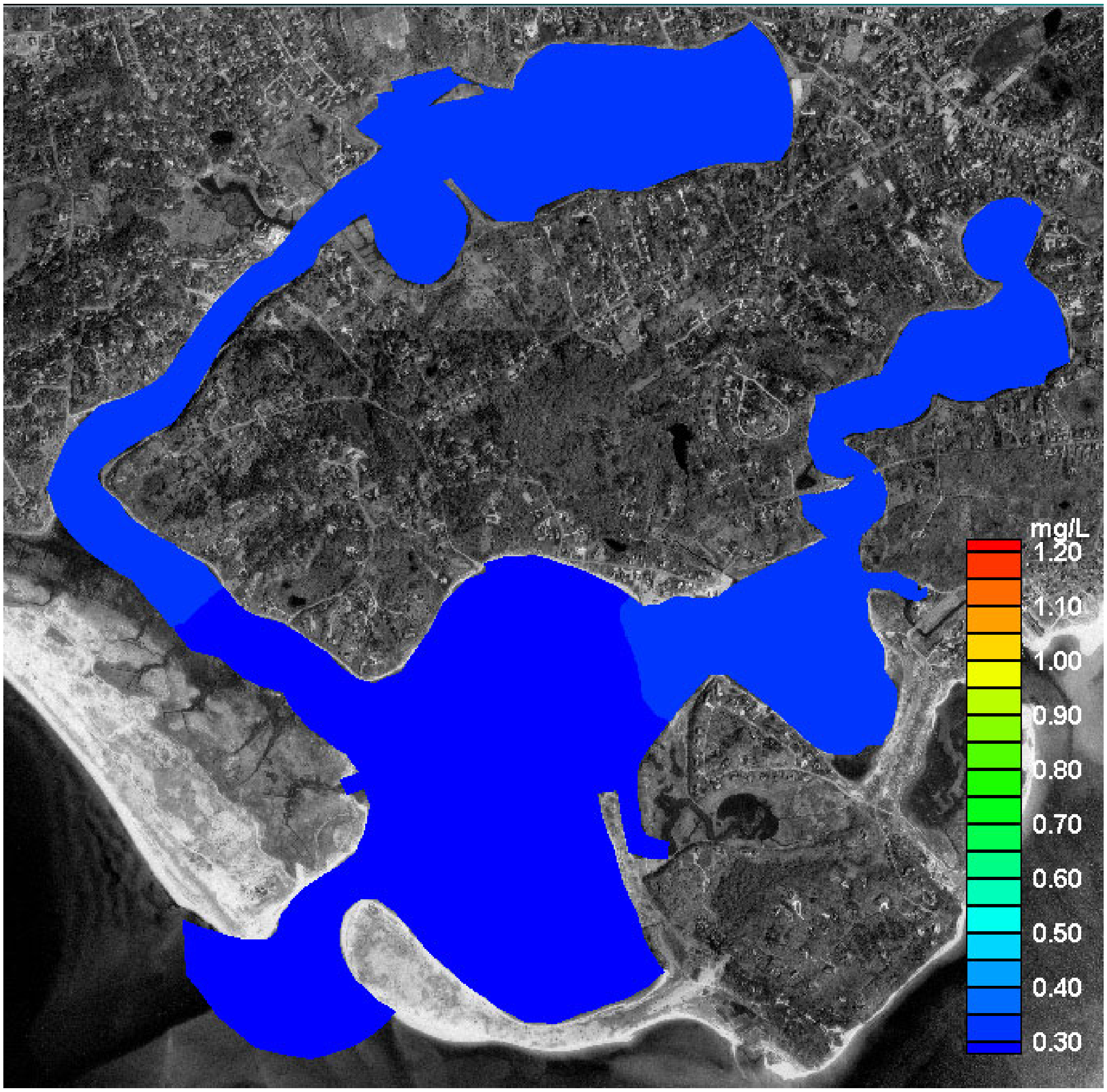


Figure VI-19. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Stage Harbor system, for no anthropogenic loading conditions.

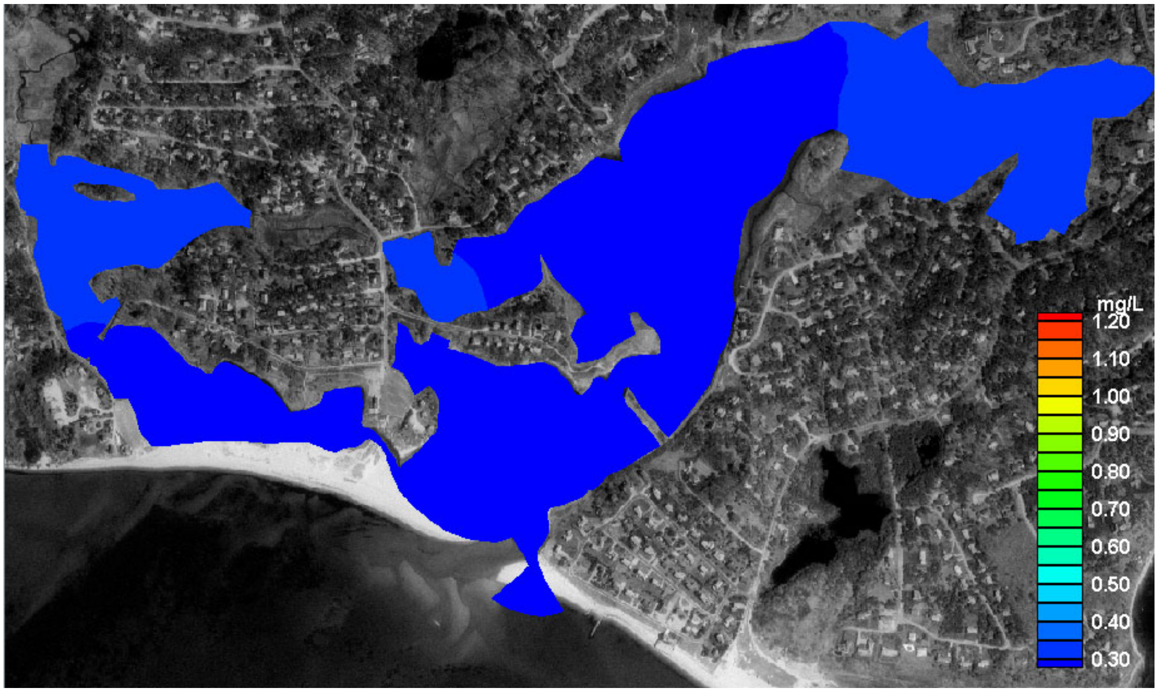


Figure VI-20. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Sulphur Springs/Cockle Cove Creek system, for no anthropogenic loading conditions.

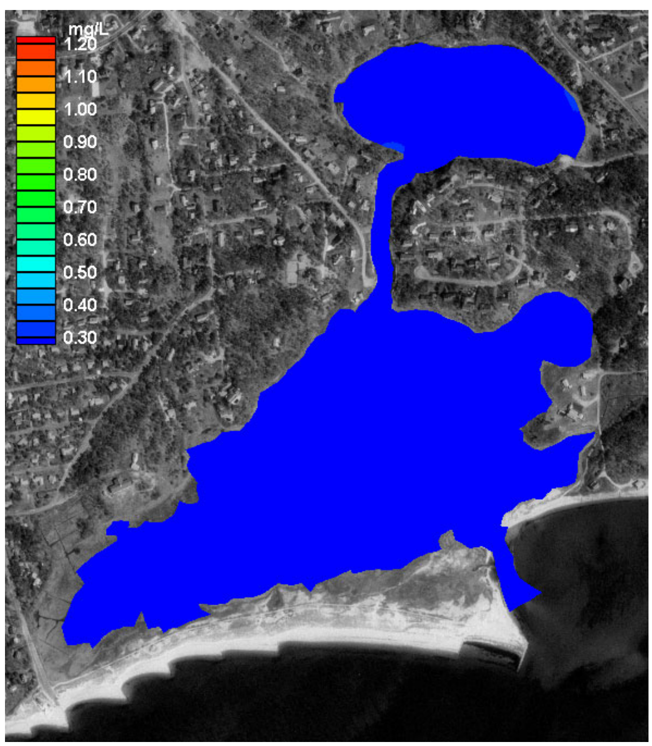


Figure VI-21. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Taylors Pond/Mill Creek system, for no anthropogenic loading conditions.

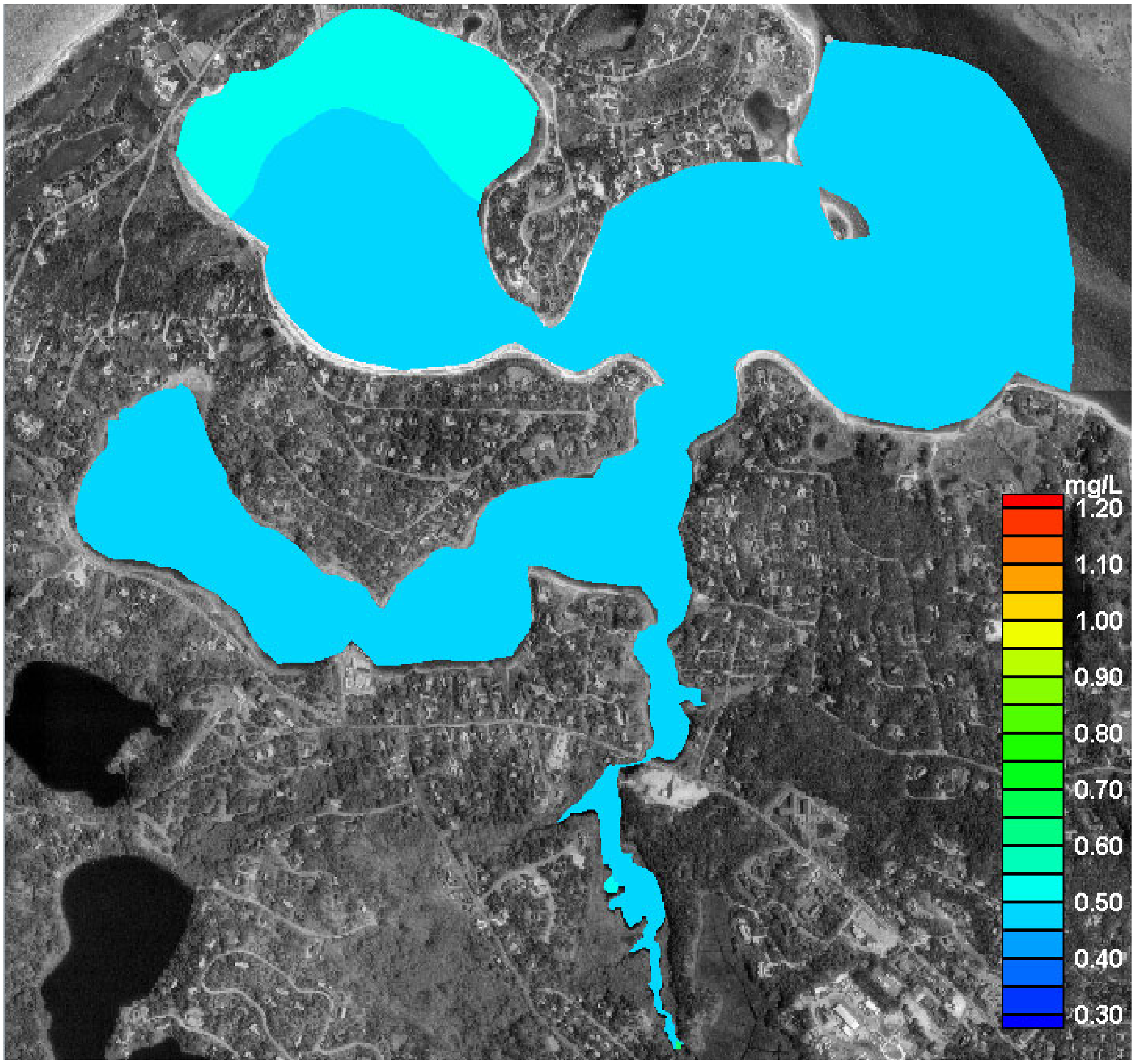


Figure VI-22. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Bassing Harbor system, for no anthropogenic loading conditions, and present background N concentration at the entrance to Pleasant Bay (0.48 mg/L).

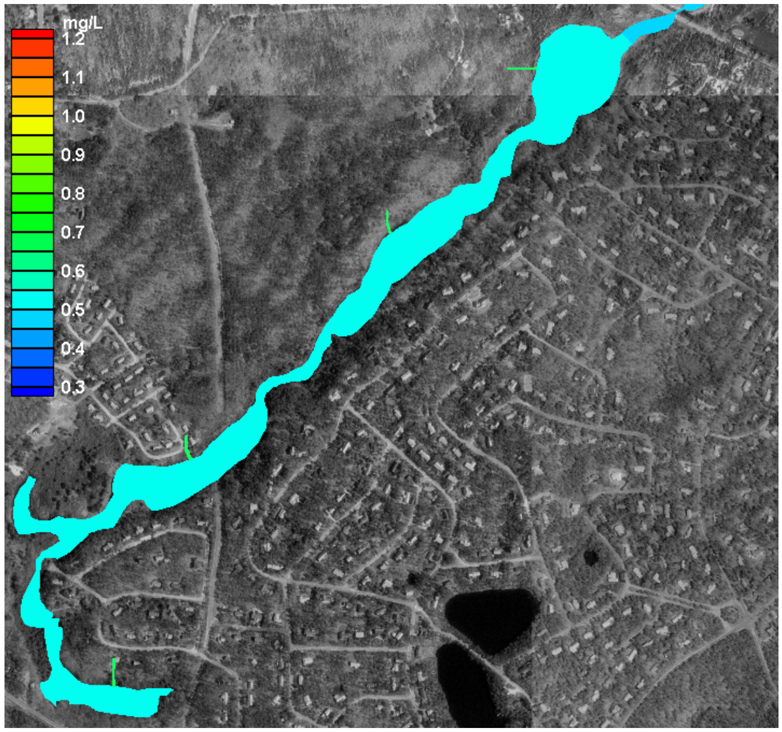


Figure VI-23. Contour plot of modeled total nitrogen concentrations in Muddy Creek, for no anthropogenic loading conditions, and present total nitrogen concentration in Pleasant Bay (0.50 mg/L).

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 Chatham's five embayment systems our assessment is based upon data from the water quality monitoring database and our surveys of eelgrass distribution, benthic animal communities and sediment characteristics conducted during the summer and fall of 2000. These data form the basis of an assessment of these systems' 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.

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 (Section VII.2), (2) eelgrass vs. macroalgal distribution (Section VII.3) and (2) 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, MEP deployed dissolved oxygen sensors within the upper regions of the embayments to record the frequency and duration of low oxygen conditions during the critical summer period. Eelgrass is a sentinel species for indicating nitrogen over-loading to a coastal embayment. It is also a fundamentally important species in the ecology of shallow coastal systems, providing both habitat structure and sediment stabilization. Mapping of each embayment's eelgrass beds was conducted for comparison to historic records. Temporal trends in habitat quality were determined by comparison with previous eelgrass distribution data collected in the Chatham embayment systems by DEP (C. Costello, personal communication). Temporal changes in eelgrass distribution provides a strong basis for evaluating recent increases (nitrogen loading) or decreases (increased flushing-new inlet) in nutrient enrichment.

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

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}.

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. The result is that lowest oxygen levels (mg L^{-1}) are found in the warmest summer months. In addition, biological processes which consume oxygen from the watercolumn vary directly with temperature. The result is that the highest rates of oxygen uptake are in the summer. It is not surprising, then, that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were placed within key sub-embayments to the 5 embayment systems. The sensors (YSI 6600) were first calibrated in the laboratory and checked with standard oxygen mixtures, then placed in the field with calibration samples collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer). Each mooring was serviced and field oxygen samples collected at the sensor, at least biweekly and sometimes weekly during a minimum deployment of 30 days during July and August. All of the mooring data from the 5 embayment systems is from summer 2002.

In addition to the oxygen sensors, chlorophyll a sensors (fluorescence) were also part of the moorings (YSI 6600). The chlorophyll a sensors were maintained as for the oxygen sensors, except that field samples were collected for chlorophyll a and pheophytin analysis by cold acetone (90%) extraction and fluorometric assay (Turner AU10). Like oxygen levels, chlorophyll a is an indicator of habitat health relating to nitrogen loading. Chlorophyll a serves as a proxy for phytoplankton biomass.

Similar to other embayments in southeastern Massachusetts, the 5 embayment systems in this assessment showed high frequency variation, apparently related to diurnal and sometimes tidal influences. 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.

Nitrogen enrichment of embayment waters can manifest itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. This phenomenon is best seen in the upper Muddy Creek record., where dissolved oxygen levels drop to less than 1 mg L-1 during the night and reach levels in excess of atmospheric saturation during the day time (Figure VII-1a). A confirmation that the low dissolved oxygen levels result from nitrogen enrichment of embayment waters is seen in many of the records where the temporal pattern of oxygen depletion is inversely correlated with the timing of phytoplankton blooms (chlorophyll a levels). This is relationship was seen in the Upper Muddy Creek (Figure VIII-1a), Mill Pond (Figure VIII-2)and to a lesser extent in Oyster Pond (Figure VIII-3), Stage Harbor (Figure VIII-4), Sulphur Springs (Figure (VIII-6). In addition, systems which generally had lower chlorophyll levels (<15 ug L^{-1}), tended to show less oxygen depletion. This is clearly seen in the comparison of the Bassing Harbor System (Figures VII-7,8,9,10) to Muddy Creek, Mill Pond, Oyster Pond, and Sulphur Springs sub-embayments (Figures VII-1,2,3,6). It is also seen within the Bassing Harbor System, which show an inverse gradient in oxygen minima to chlorophyll levels moving from Ryder Cove to Crows Pond to Bassing Harbor.

The dissolved oxygen and chlorophyll a records were analyzed to determine the percent of the deployment time (29-64 days) that oxygen was below various benchmark concentrations (Table VII-1) or above various chlorophyll concentrations (Table VII-2). These data indicate not just the minimum or maximum levels of these critical nutrient related constituents, but the intensity of the low oxygen circumstances or of the phytoplankton blooms. It is clear that systems with higher chlorophyll had lower and more prolonged oxygen depletion.

Muddy Creek (upper and lower) are clearly eutrophic with frequent and prolonged oxygen declines below 3 mg L^{-1} (half of the record) and chlorophyll a levels exceeding 25 ug L^{-1} on over half of the days. In addition, it appears that upper Muddy Creek built and sustained a large late summer bloom with exceedingly high chlorophyll a levels, >80 ug L^{-1}.

Within Stage Harbor System, only Mill Pond showed very low oxygen levels (<3 mg L^{-1}), Oyster Pond and upper Stage Harbor (lower Mitchell River) consistently had oxygen levels >5 mg L^{-1} and chlorophyll a levels < 15 u L^{-1} (generally <10 mg L^{-1}). None of these systems showed the very high bloom conditions of Muddy Creek. However, both parameters clearly indicate nutrient enrichment in Mill Pond and to a lesser extent in the other 2 sub-embayments.

A single mooring was placed in the terminal drowned kettle pond, Taylors Pond, in the Taylors Pond System. Mill Creek is very shallow with parts becoming emergent at low tide. In addition, Mill Creek functions primarily as a salt marsh a high proportion of the tidal reach being vegetated by *Spartina* grasses. Taylors Pond also showed indications of nitrogen enrichment, with dissolved oxygen levels declining below 5 mg L^{-1} almost 10% of the time (and <4 mg L^{-1} 2% of the time) and chlorophyll a levels exceeding 10 ug L^{-1} almost 10% of the deployment period.

Sulphur Springs showed a similar level of nitrogen related habitat quality to Mill Pond, both exchanging tidal waters with Nantucket Sound. Sulphur Springs is much shallower than Mill Pond, but still showed significant oxygen depletion, <3 mg L^{-1} on 6% of time and with chlorophyll a levels exceeding 25 ug L^{-1}. Sulphur Springs is the shallow upper basin within the Sulphur Springs, Cockle Cove, Bucks Creek composite embayment. There are signs that Sulphur Springs is currently transitioning to salt marsh.

The Bassing Harbor System is part of the Pleasant Bay Estuary. Bassing Harbor receives nitrogen inputs from its adjacent watershed as well as some nitrogen on the incoming tide which originated within the greater watershed to Pleasant Bay. At present it appears that the Bassing Harbor System overall supports relatively high oxygen levels and moderate chlorophyll a levels, except for the upper reach of Ryder Cove. Ryder Cove receives the highest nitrogen load from its watershed of the sub-embayments to this system. Upper Ryder Cove is approaching Mill Pond relative to its nitrogen response. The difference is that upper Ryder Cove still supports eelgrass, whereas Mill Pond has lost its beds.

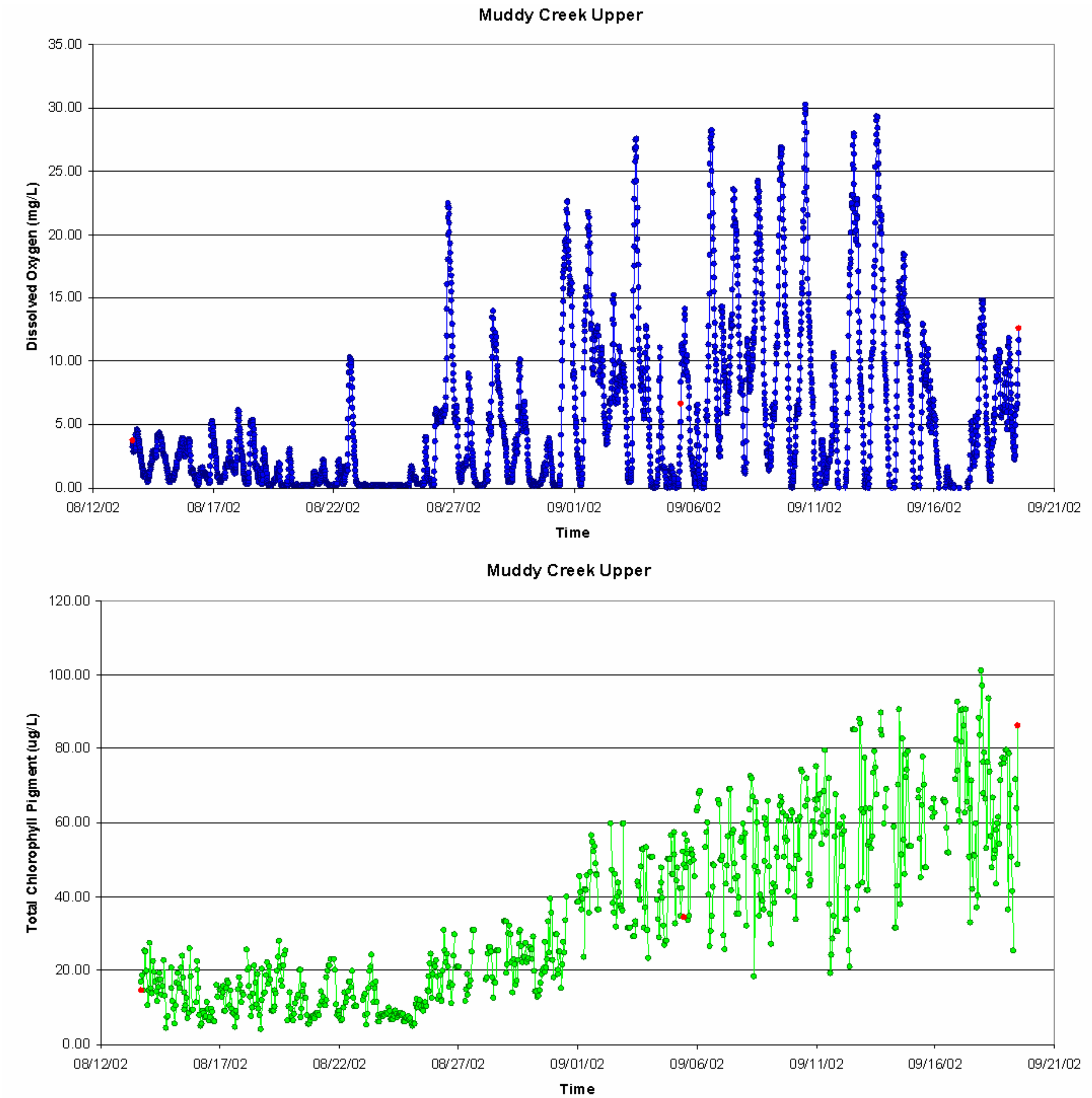


Figure VII-1a. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Upper Muddy Creek, Summer 2002. Calibration samples represented as red dots.

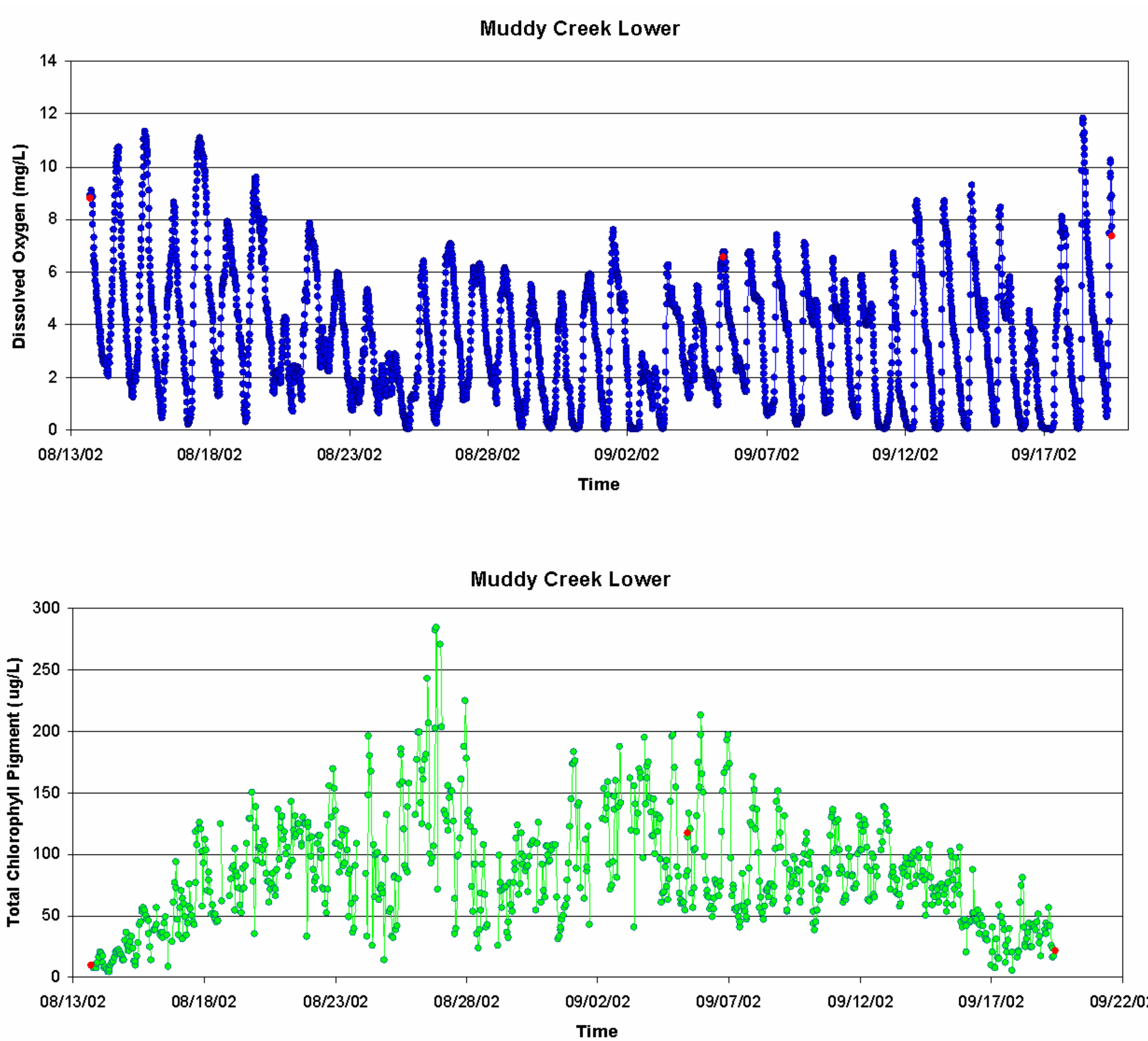
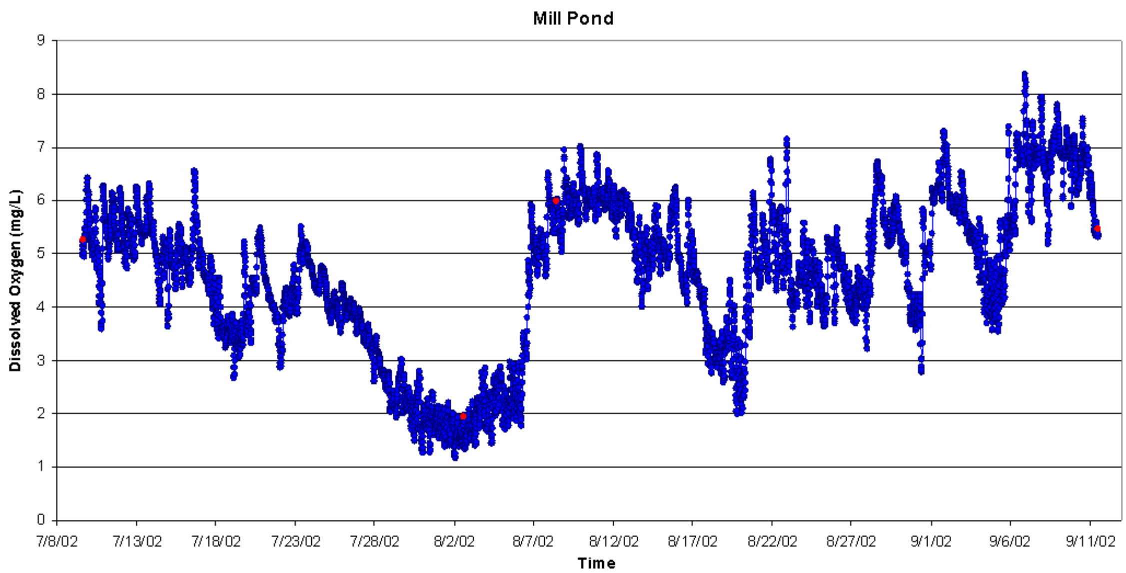


Figure VII-1b. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Lower Muddy Creek, Summer 2002. Calibration samples represented as red dots.



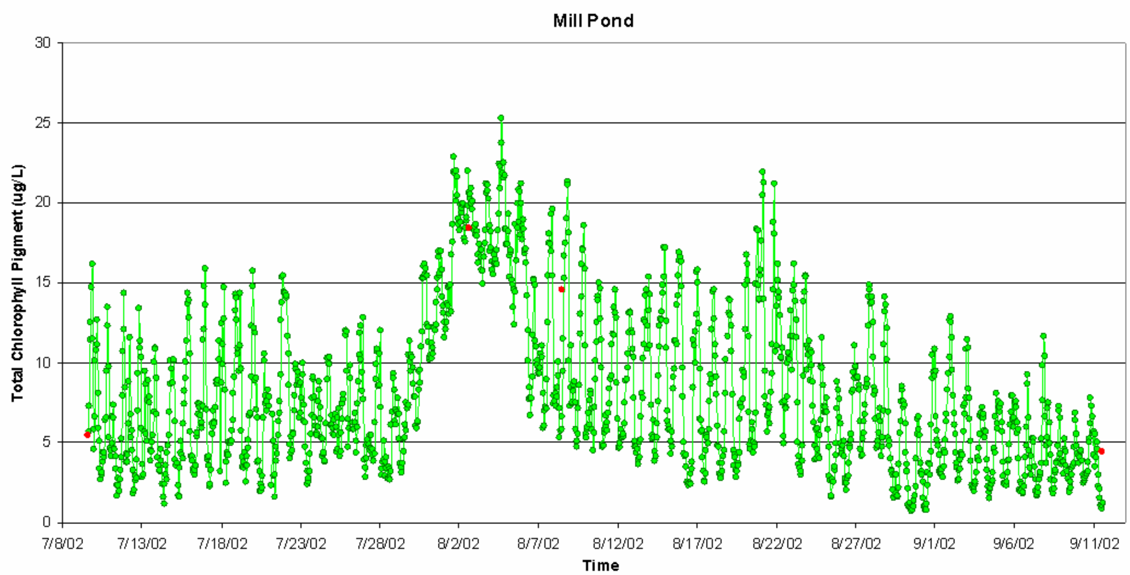


Figure VII-2. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Mill Pond (Stage Harbor System), Summer 2002. Calibration samples represented as red dots.

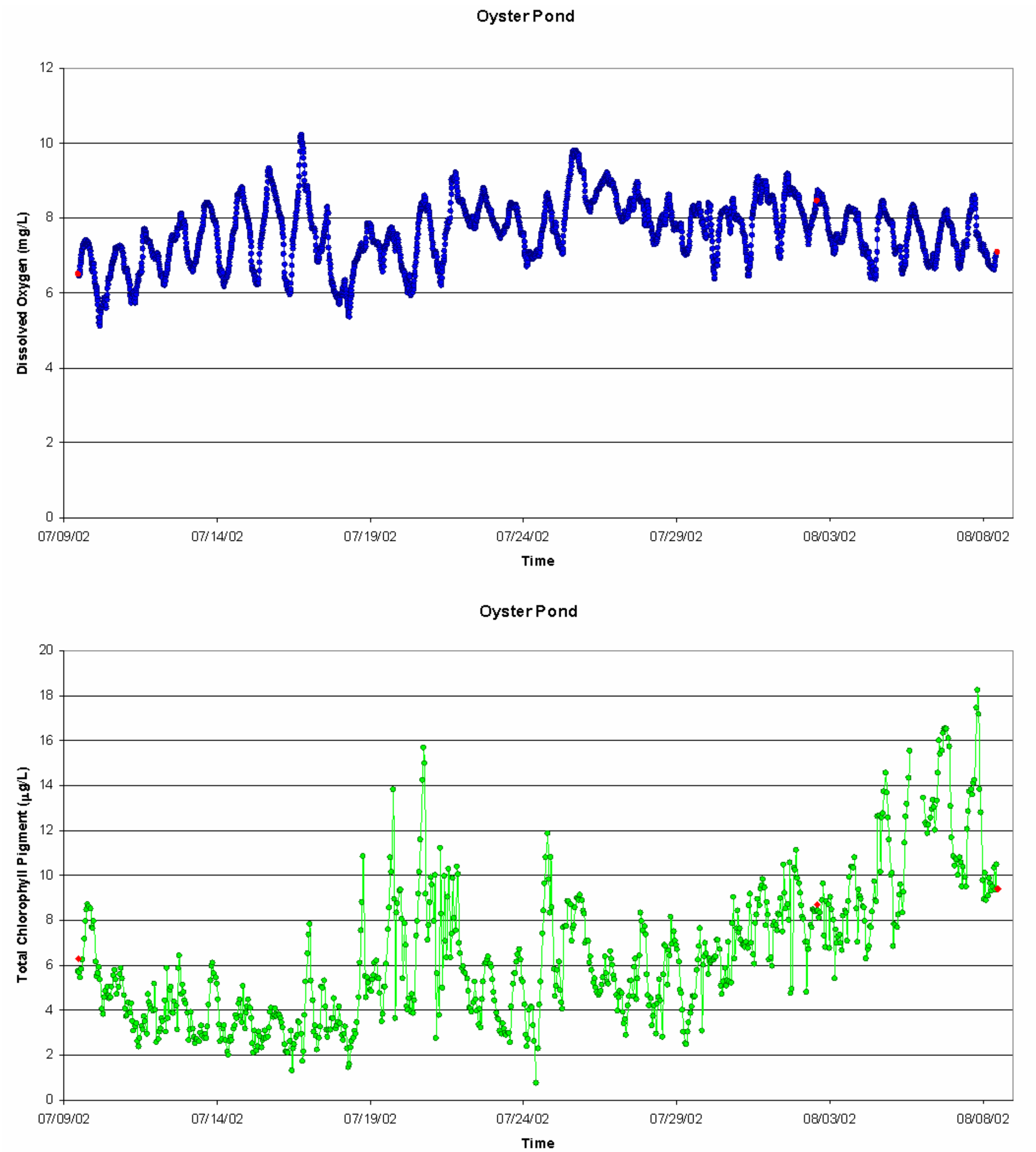


Figure VII-3. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Oyster Pond (Stage Harbor System), Summer 2002. Calibration samples represented as red dots.

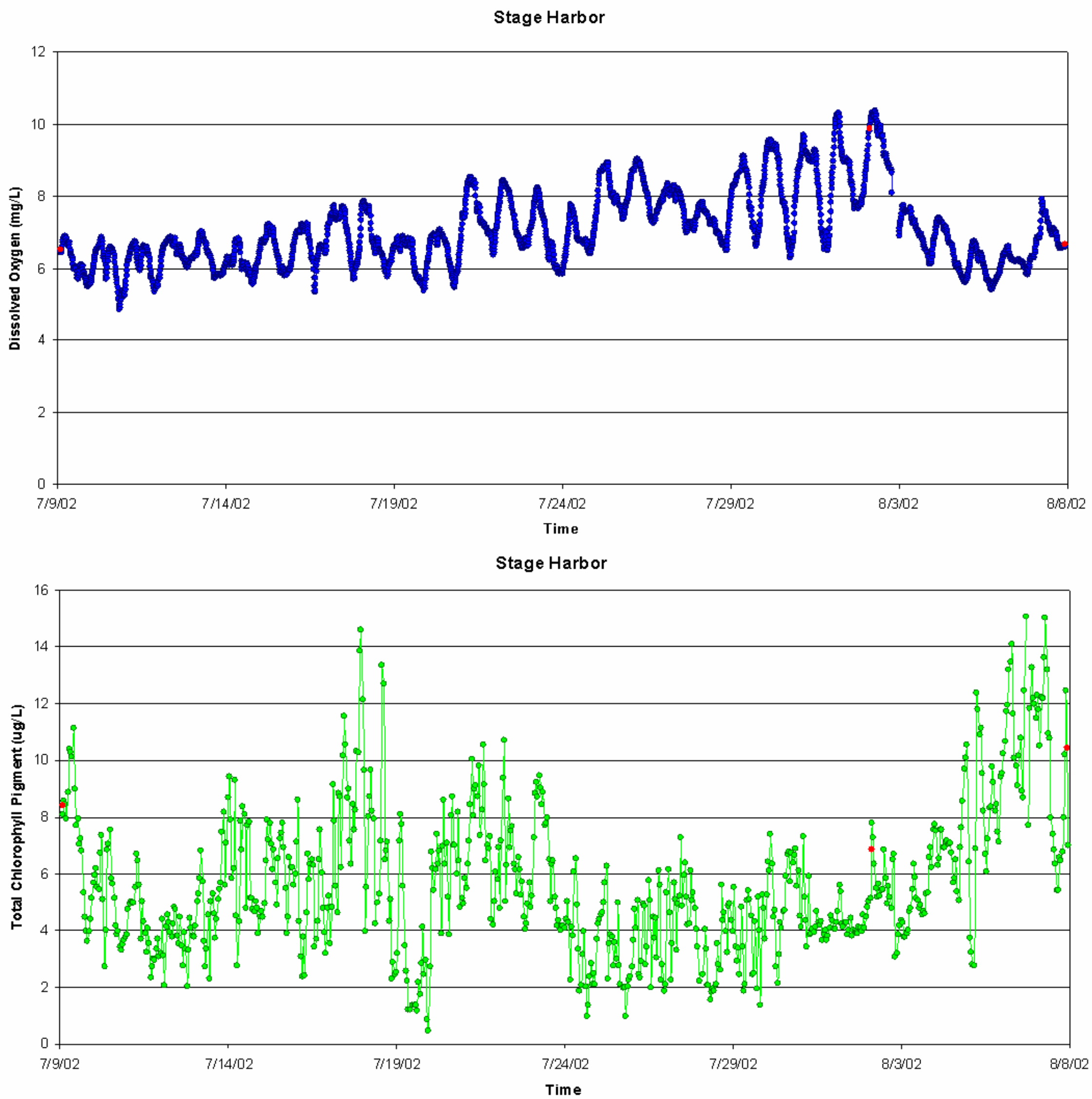


Figure VII-4. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Stage Harbor (Stage Harbor System), Summer 2002. Calibration samples represented as red dots.

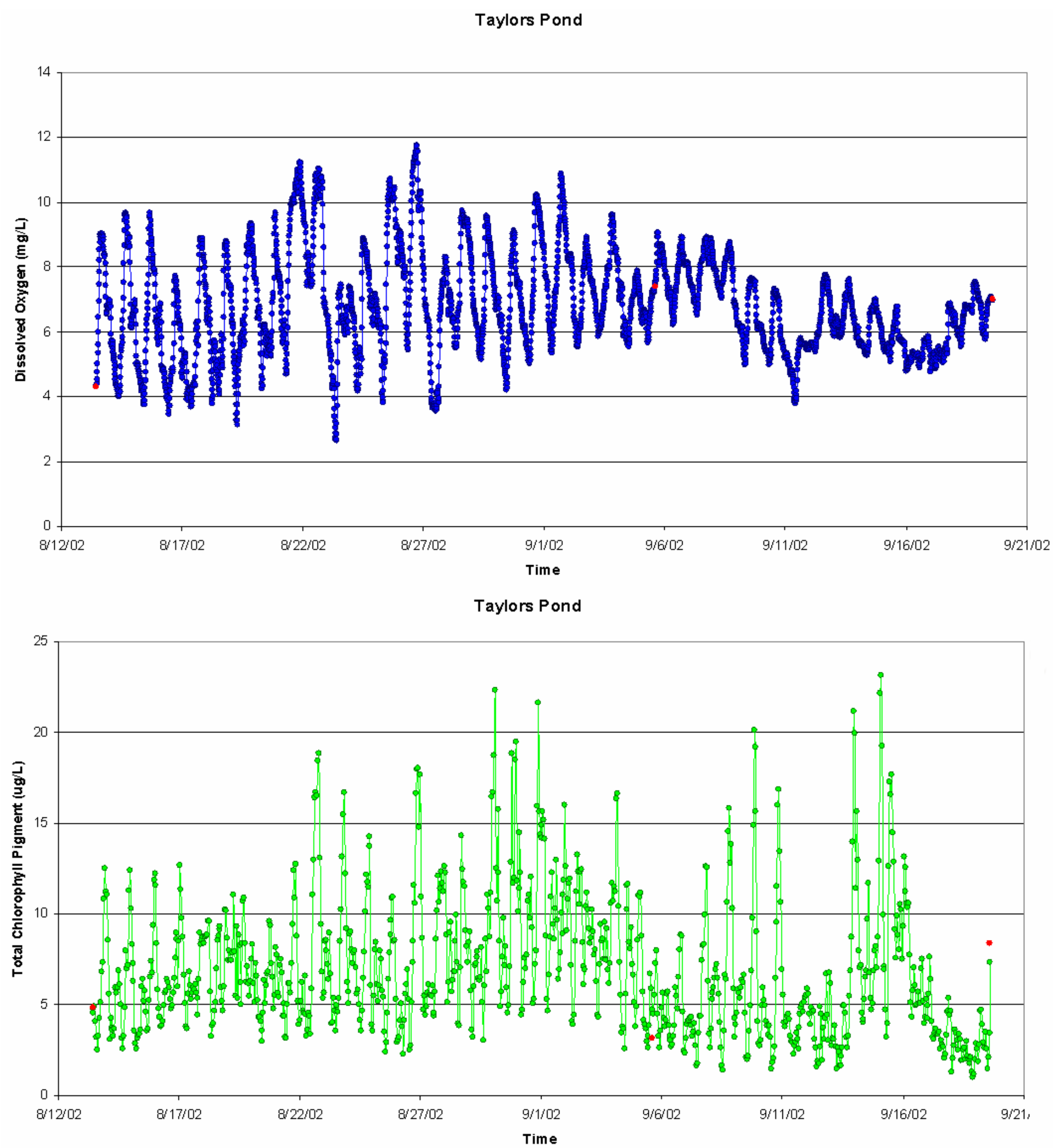


Figure VII-5. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Taylors Pond, Summer 2002. Calibration samples represented as red dots.

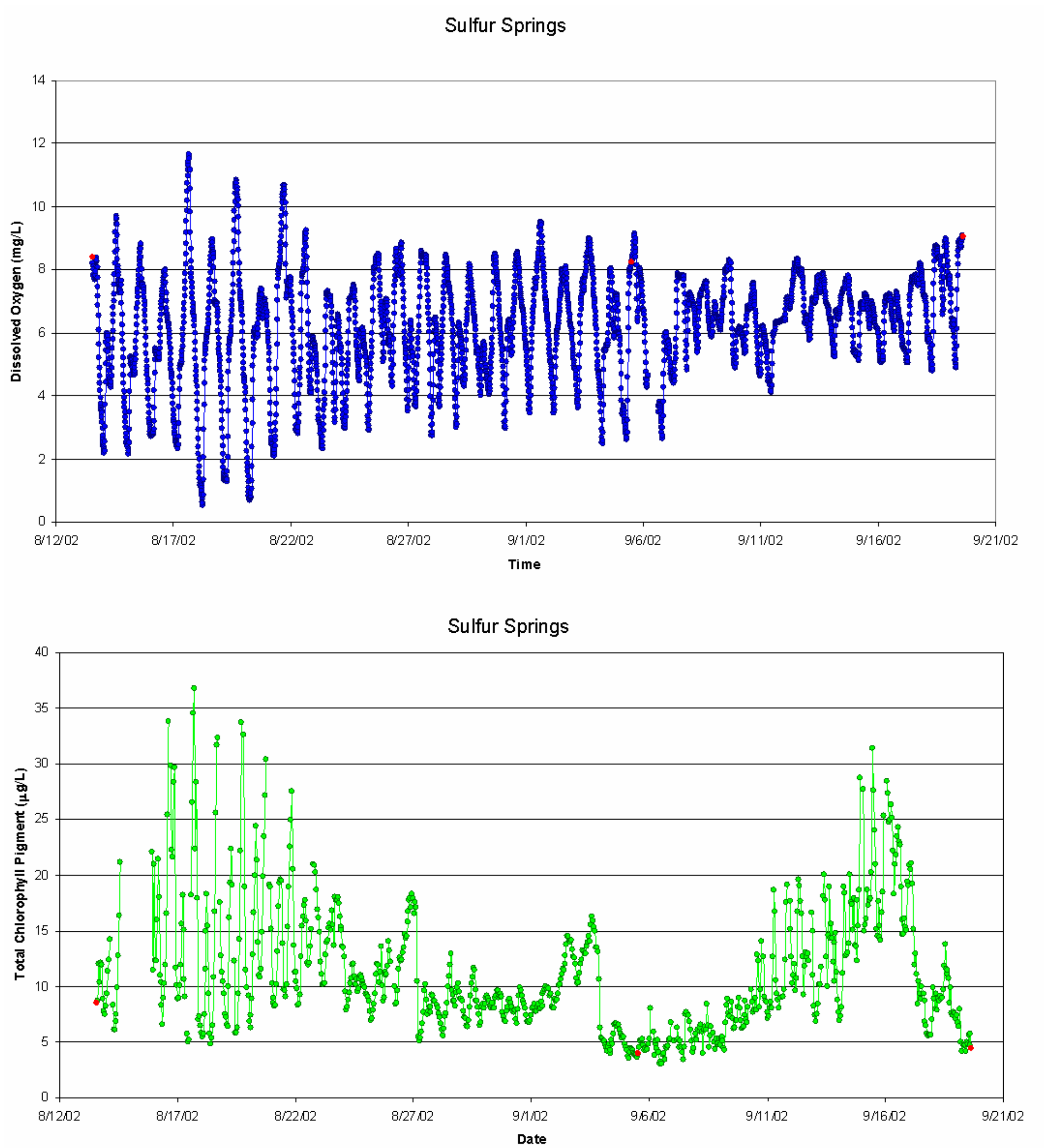


Figure VII-6. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Sulphur Springs, Summer 2002. Calibration samples represented as red dots.

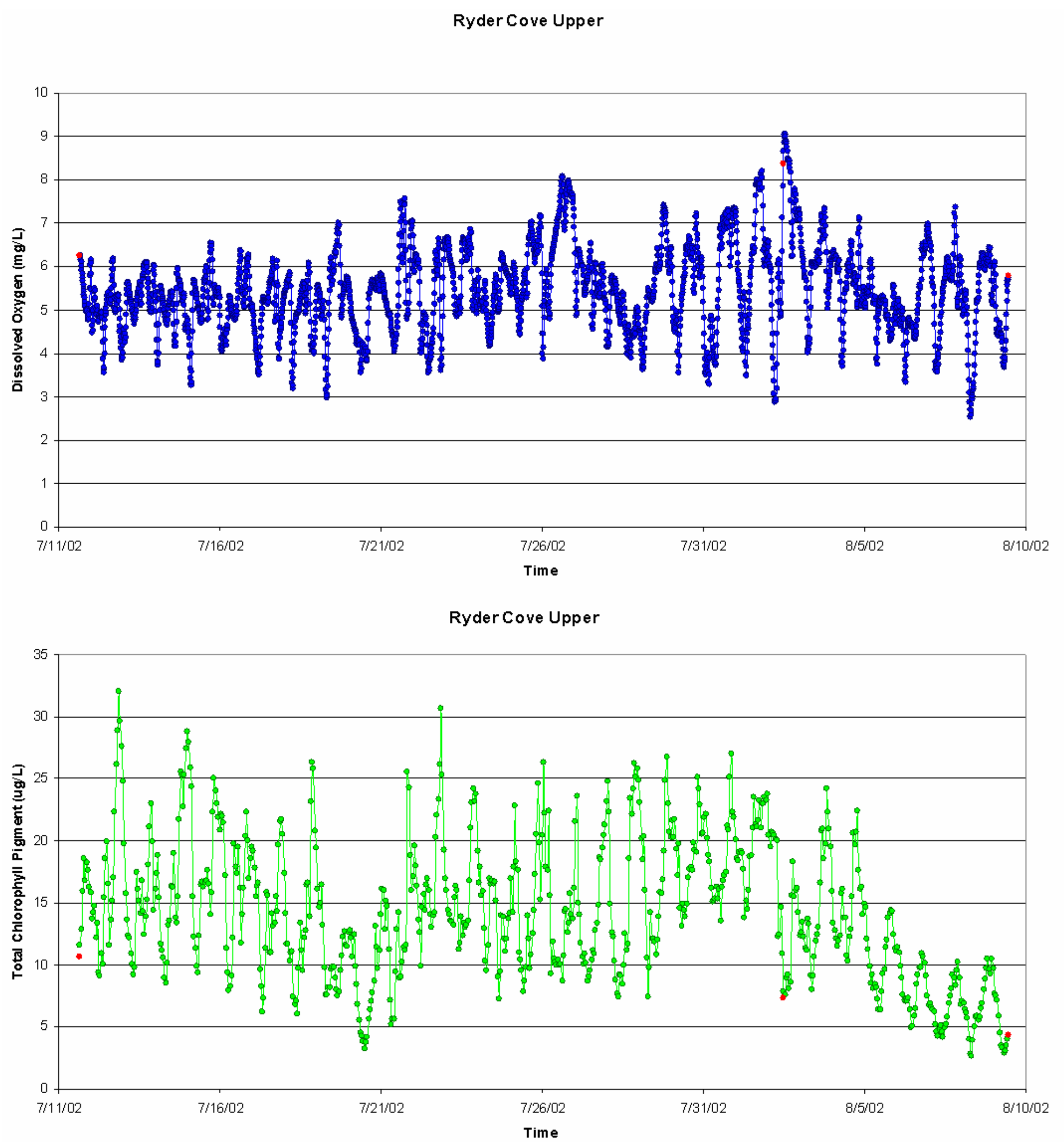
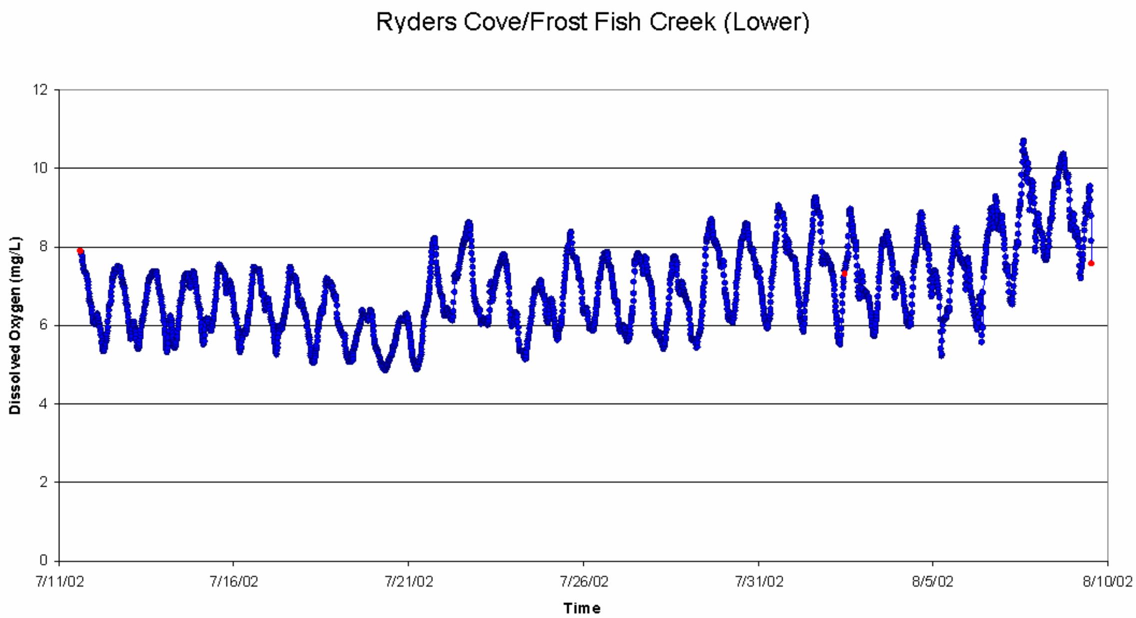


Figure VII-7. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Upper Ryder Cove (Bassing Harbor System), Summer 2002. Calibration samples represented as red dots.



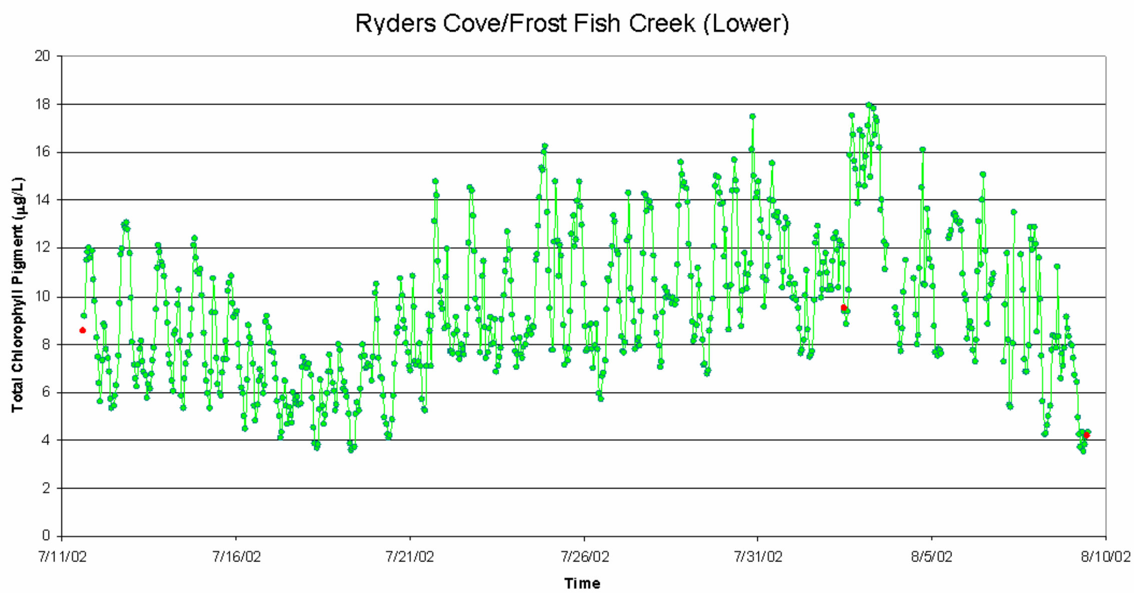


Figure VII-8. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Lower Ryder Cove (Bassing Harbor System), Summer 2002. Calibration samples represented as red dots.

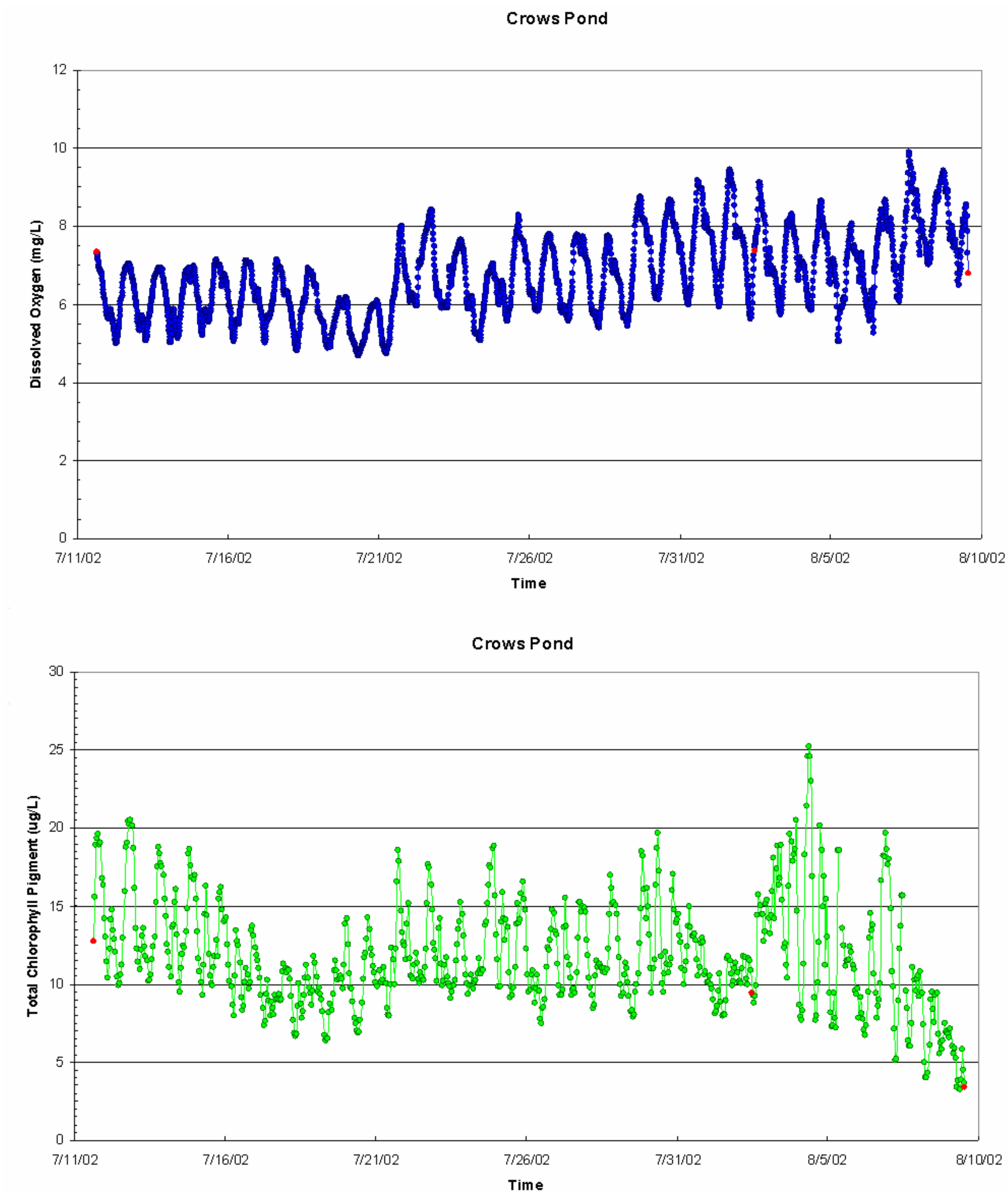


Figure VII-9. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Crows Pond (Bassing Harbor System), Summer 2002. Calibration samples represented as red dots.

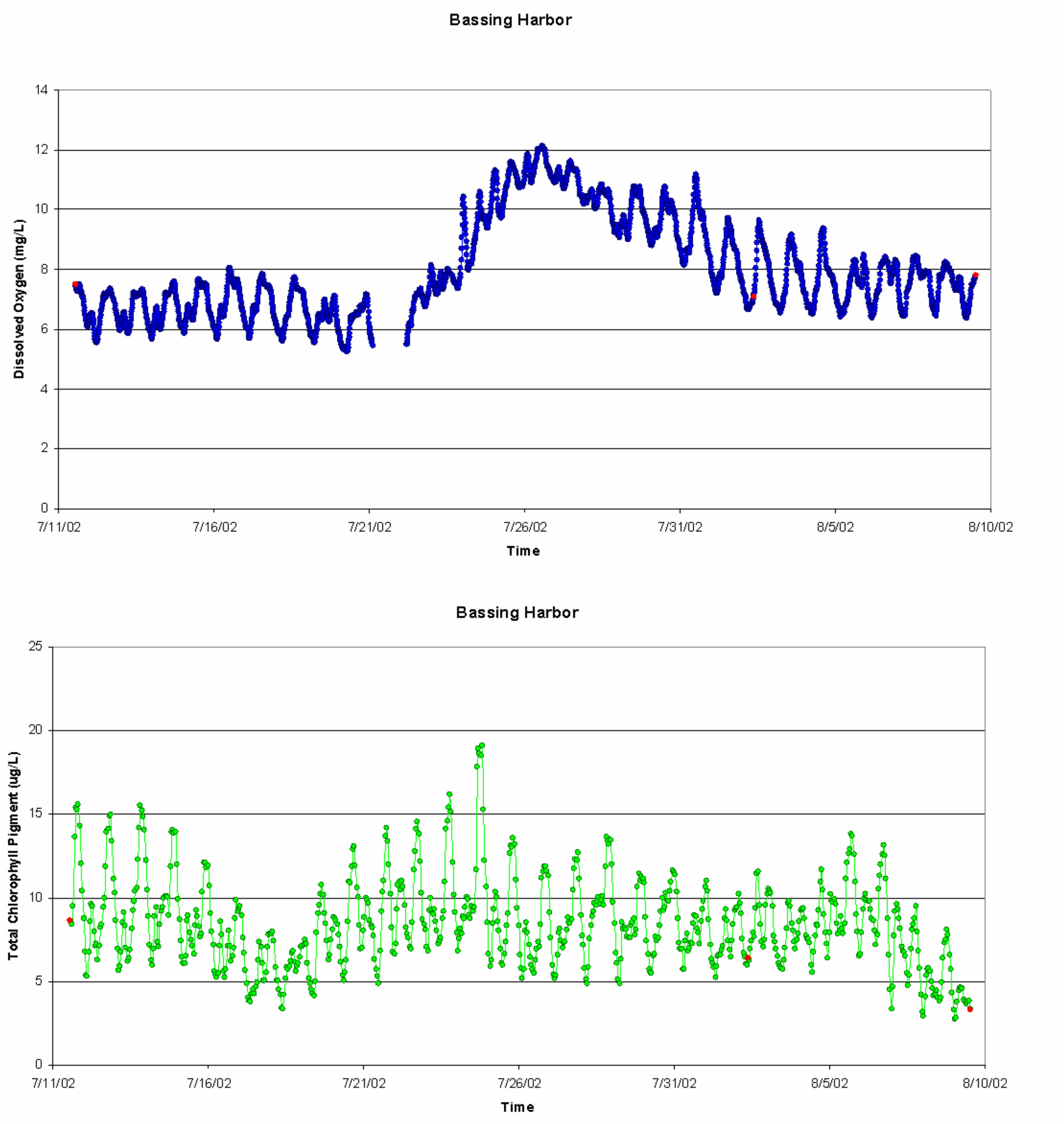


Figure VII-10. Bottom water record of dissolved oxygen (top panel) and chlorophyll-a (bottom panel) in Bassing Harbor, Summer 2002. Calibration samples represented as red dots.

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

Massachusetts Estuaries Project Town of Chatham: 2002	Dissolved Oxygen: Continuous Record, Summer 2002				
	Deployment Days	<6 mg/L (% of days)	<5 mg/L (% of days)	<4 mg/L (% of days)	<3 mg/L (% of days)
Muddy Creek System:					
Muddy Creek-Upper	29	88%	81%	76%	69%
Muddy Creek-Lower	37	85%	74%	60%	49%
Stage Harbor System:					
Mill Pond	64	84%	56%	30%	16%
Oyster Pond	64	6%	1%	0%	0%
Stage Harbor-Upper	30	14%	0%	0%	0%
Taylor's Pond System:					
Taylor's Pond	37	21%	9%	2%	0%
Sulphur Springs System:					
Sulphur Springs-Basin	37	41%	22%	12%	6%
Basing Harbor System:					
Ryder Cove-Upper	29	73%	32%	7%	1%
Ryder Cove-Lower	29	21%	1%	0%	0%
Crows Pond	29	28%	3%	0%	0%
Bassing Harbor	29	7%	0%	0%	0%

Table VII-2. Frequency (number of events during deployment) and duration (total number of days over deployment) of chlorophyll a levels above various benchmark levels within the 5 embayment systems.

Embayment System	Start Date	End Date	Total Deployment (Days)	Duration (cumulative days) >5 ug/L (Days)	>10 ug/L (Days)	>15 ug/L (Days)	>20 ug/L (Days)	>25 ug/L (Days)	Frequency (# events) >5 ug/L (#)	>10 ug/L (#)	>15 ug/L (#)	>20 ug/L (#)	>25 ug/L (#)
Bassing Harbor System													
Bassing Harbor	7/11/02	8/9/02	29	26.833	6.625	0.583	0.000	0.000	11	33	4	0	0
			Mean	2.439	0.201	0.146	N/A	N/A					
			Min	0.208	0.042	0.083	0.000	0.000					
			Max	8.667	0.417	0.250	0.000	0.000					
			S.D.	3.053	0.116	0.072	N/A	N/A					
Ryder's Cove Up	7/11/02	8/9/02	28.8	27.833	21.333	12.167	5.167	1.125	6	33	44	27	12
			Mean	4.639	0.646	0.277	0.191	0.094					
			Min	0.042	0.042	0.042	0.042	0.042					
			Max	16.833	4.125	1.000	0.542	0.208					
			S.D.	6.808	0.779	0.234	0.136	0.062					
Ryder Cove Low	7/11/02	8/9/02	28.9	26.458	10.833	1.292	0.000	0.000	11	44	12	0	0
			Mean	2.405	0.246	0.108	N/A	N/A					
			Min	0.083	0.042	0.042	0.000	0.000					
			Max	17.708	1.125	0.250	0.000	0.000					
			S.D.	5.234	0.224	0.072	N/A	N/A					
Crows Pond	7/11/02	8/9/02	28.9	28.375	19.833	4.917	0.458	0.042	3	49	34	4	1
			Mean	9.458	0.405	0.145	0.115	0.042					
			Min	0.042	0.042	0.042	0.042	0.042					
			Max	27.417	2.000	0.375	0.208	0.042					
			S.D.	15.559	0.400	0.107	0.086	N/A					

Table VII-2. (continued)

Embayment System	Start Date	End Date	Total Deployment (Days)	Duration (cumulative days) >5 ug/L (Days)	>10 ug/L (Days)	>15 ug/L (Days)	>20 ug/L (Days)	>25 ug/L (Days)	Frequency (# events) >5 ug/L (#)	>10 ug/L (#)	>15 ug/L (#)	>20 ug/L (#)	>25 ug/L (#)
Muddy Creek System													
Muddy Creek Low	8/13/02	9/19/02	36.8	32.833	32.292	31.667	31.042	30.333	2	7	12	13	16
			Mean	16.417	4.613	2.639	2.388	1.896					
			Min	0.625	0.042	0.083	0.042	0.042					
			Max	32.208	28.042	20.542	20.542	16.708					
			S.D.	22.333	10.341	5.992	5.796	4.366					
Muddy Creek Up	8/13/02	9/11/02	29	30.958	26.458	22.625	19.417	16.667	6	23	36	35	28
			Mean	5.160	1.150	0.628	0.555	0.595					
			Min	0.958	0.042	0.042	0.042	0.042					
			Max	20.583	20.125	16.167	6.250	5.500					
			S.D.	7.787	4.143	2.670	1.476	1.308					
Taylors Pond System													
Taylor's Pond	8/13/02	9/19/02	37.2	23.833	7.250	1.750	0.250	0.000	66	42	19	5	0
			Mean	0.361	0.173	0.092	0.050	N/A					
			Min	0.042	0.042	0.042	0.042	0.000					
			Max	1.417	0.458	0.208	0.083	0.000					
			S.D.	0.299	0.110	0.047	0.019	N/A					
Mill Pond	7/9/02	9/11/02	63.9	43.125	19.208	7.500	1.500	0.042	78	64	24	10	1
			Mean	0.553	0.300	0.313	0.150	0.042					
			Min	0.042	0.042	0.042	0.042	0.042					
			Max	10.917	6.792	1.958	0.375	0.042					
			S.D.	1.228	0.838	0.510	0.111	N/A					

Table VII-2. (continued)

Embayment System	Start Date	End Date	Total Deployment (Days)	Duration (cumulative days) >5 ug/L (Days)	>10 ug/L (Days)	>15 ug/L (Days)	>20 ug/L (Days)	>25 ug/L (Days)	Frequency (# events) >5 ug/L (#)	>10 ug/L (#)	>15 ug/L (#)	>20 ug/L (#)	>25 ug/L (#)
Sulphur Springs System													
Sulphur Springs	8/13/02	9/19/02	37.1	31.208	15.667	7.208	2.542	1.083	22	36	35	19	12
			Mean	1.419	0.435	0.206	0.134	0.090					
			Min	0.042	0.042	0.042	0.042	0.042					
			Max	16.167	3.000	0.750	0.333	0.125					
			S.D.	3.854	0.588	0.150	0.088	0.030					
Stage Harbor System													
Stage Harbor	7/9/02	8/8/02	29.9	15.875	2.208	0.083	0.000	0.000	60	14	2	0	0
			Mean	0.265	0.158	0.042	N/A	N/A					
			Min	0.042	0.042	0.042	0.000	0.000					
			Max	2.750	0.625	0.042	0.000	0.000					
			S.D.	0.407	0.161	0.000	N/A	N/A					
Oyster Pond	7/9/02	9/11/02	64	17.958	3.625	0.583	0.000	0.000	40	22	4	0	0
			Mean	0.449	0.165	0.146	N/A	N/A					
			Min	0.042	0.042	0.042	0.000	0.000					
			Max	5.792	1.333	0.375	0.000	0.000					
			S.D.	0.953	0.289	0.158	N/A	N/A					
Mill Pond	7/9/02	9/11/02	63.9	43.125	19.208	7.500	1.500	0.042	78	64	24	10	1
			Mean	0.553	0.300	0.313	0.150	0.042					
			Min	0.042	0.042	0.042	0.042	0.042					
			Max	10.917	6.792	1.958	0.375	0.042					
			S.D.	1.228	0.838	0.510	0.111	N/A					

VII.3 EELGRASS ANALYSIS

A detailed, eelgrass survey was conducted of the five embayments of the Town of Chatham in the Fall of 2000. The survey was conducted by shallow draft boat with direct observation of the embayment bottom. In addition to coverage information (presence or absence), the density of the eelgrass beds were assessed in order to determine the role of this resource in system function. Density relates to the amount of bottom covered with eelgrass within the boundary of region of eelgrass bed colonization. This latter density value allows for future tracking of changes in eelgrass bed health, which is frequently not possible from bed delineation alone. This detailed study, when combined with the mapping program by DEP in support of MEP (C. Costello), provides a view of temporal trends in eelgrass distribution from 1951 to 1994/5 to 2000. This temporal information can be used to determine the stability of the eelgrass community.

The fact that each of the eelgrass data sets was collected by a different method reduces the extent to which quantitative rates of change in eelgrass coverage within a basin can be determined. However, 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. The historical eelgrass data (presence/absence) was derived from 1951 aerial photos, but with only anecdotal validation, while the 1994/5 and 2000 data had field validation. Furthermore, the fact that the trend from 1951 to 1994/5 was consistent with the trend from 1994/5 to 2000 lends credence to the earlier data set.

In 2000 only the larger embayment systems contained notable eelgrass coverage. Eelgrass was not observed within Taylors Pond/Mill or Creek, Cockle Cove/Sulphur Springs/Bucks Creek. Muddy Creek was devoid of eelgrass except for a small patch (about 10% density) adjacent the inlet. The eelgrass survey data from the Stage Harbor and Bassing Harbor Systems was used to produce the eelgrass coverage maps shown in Figures VII-11 and VII-12. Within these 2 larger systems, eelgrass was not observed within the upper regions of the Oyster Pond and Little Mill Pond/Mill Pond/Mitchell River sub-embayments in the Stage Harbor System and in Frost Fish Creek in the Bassing Harbor System.

Due to our concern over potential recent changes in nutrient conditions within the major embayment systems resulting from watershed loading and changes in flushing (inlet shifts), we examined Massachusetts DEP eelgrass mapping data collected in 1994 for Chatham's coastal waters. These data confirmed the absence of eelgrass within the smaller embayments and agreed in general distribution within the two large embayment systems. Figure VII-13, VII-14, and VII-16 show the distribution of eelgrass coverage in 1994/5.

The 1951 eelgrass distribution maps for the Stage Harbor System (Figure VII-15) and Bassing Harbor System (Figure VII-16) suggest that eelgrass coverage was significantly greater in some of the sub-embayments compared to present conditions. Most notably both Oyster Pond and Mill Pond had extensive coverage in 1951. These systems still had coverage in 1994 and the near complete loss by 2000. In fact, it appears that most of these 2 embayment systems was capable of supporting relatively dense eelgrass stands in 1951.

It is possible to determine a general idea of short and long term rates of change in eelgrass coverage from the mapping data. However, since the 2000 mapping program was done fully by on-site transect surveys it was able to detect sparse eelgrass beds, not typically seen by aerial mapping (Table VII-3). Therefore, while the 2000 study may represent more fully the eelgrass situation, it is not directly comparable to the historical data. Therefore, to determine historical changes we used the distributions shown in Figures VII-15, VII-16, which were all generally collected using a similar approach (Table VII-4). The latter data represent relatively established beds and therefore the areal coverage's are less than observed in the transect study. None-the-less, it is clear that each of the sub-embayments to the Stage Harbor (Figure VII-15) and Bassing Harbor (Figure VII-16) Systems have lost coverage. Comparison of coverage's based upon maps derived from aerial surveys suggests that there has been significant reduction in eelgrass coverage over the past 50 years in both embayment systems (Table VII-4). That this change is still occurring is seen in the aerial mapping data (Table VII-4) and by comparing the 1994/5 and 2000 maps for each system. Since the 2000 maps (Figures VII-11, 12) use a more sensitive technique than the 1994/5 maps (Figures VII-14, 16), the lower coverage in 2000 suggests a "true" loss of bed area.

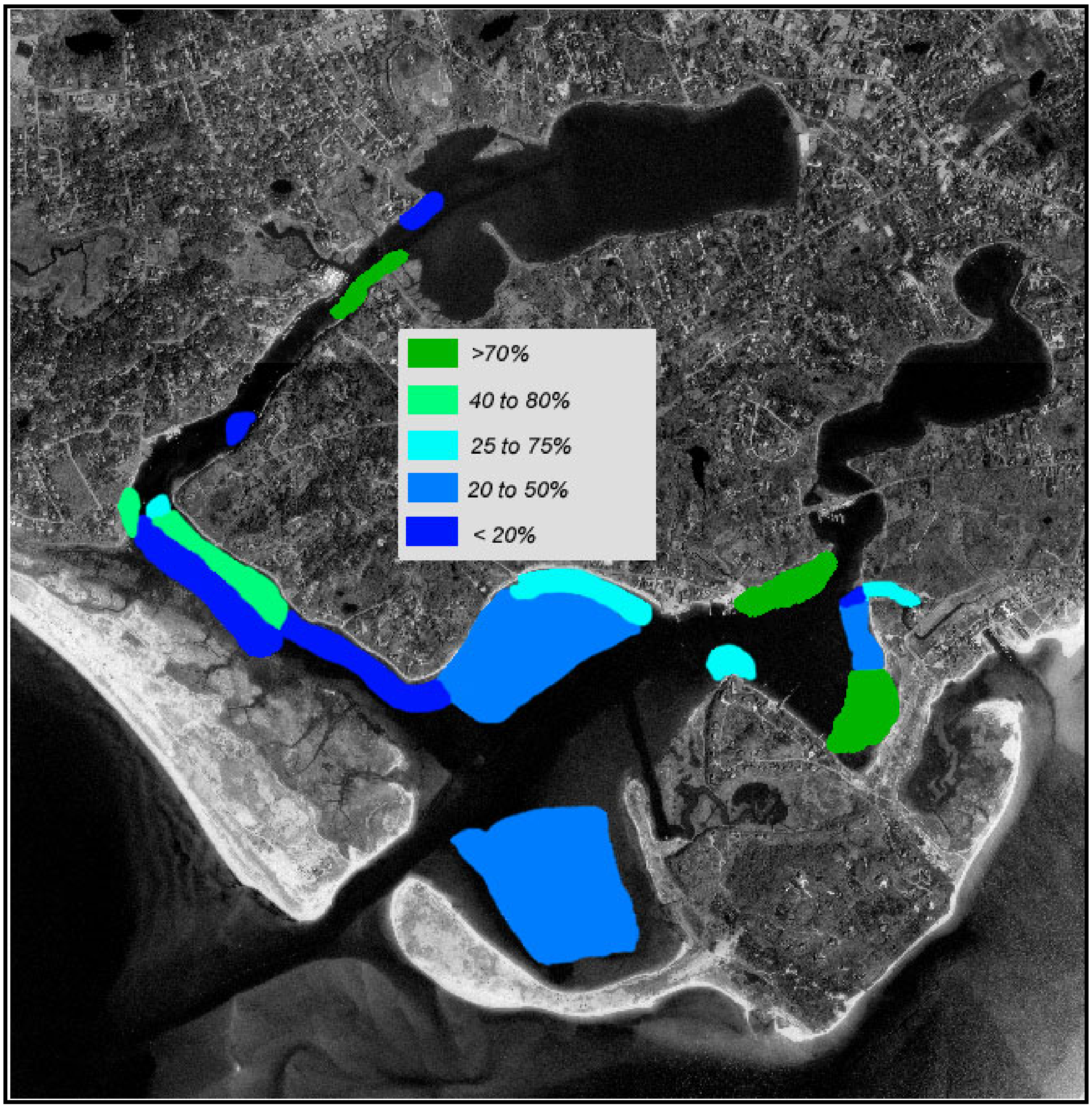


Figure VII-11. Map of Stage Harbor eelgrass distribution as observed in 2000.

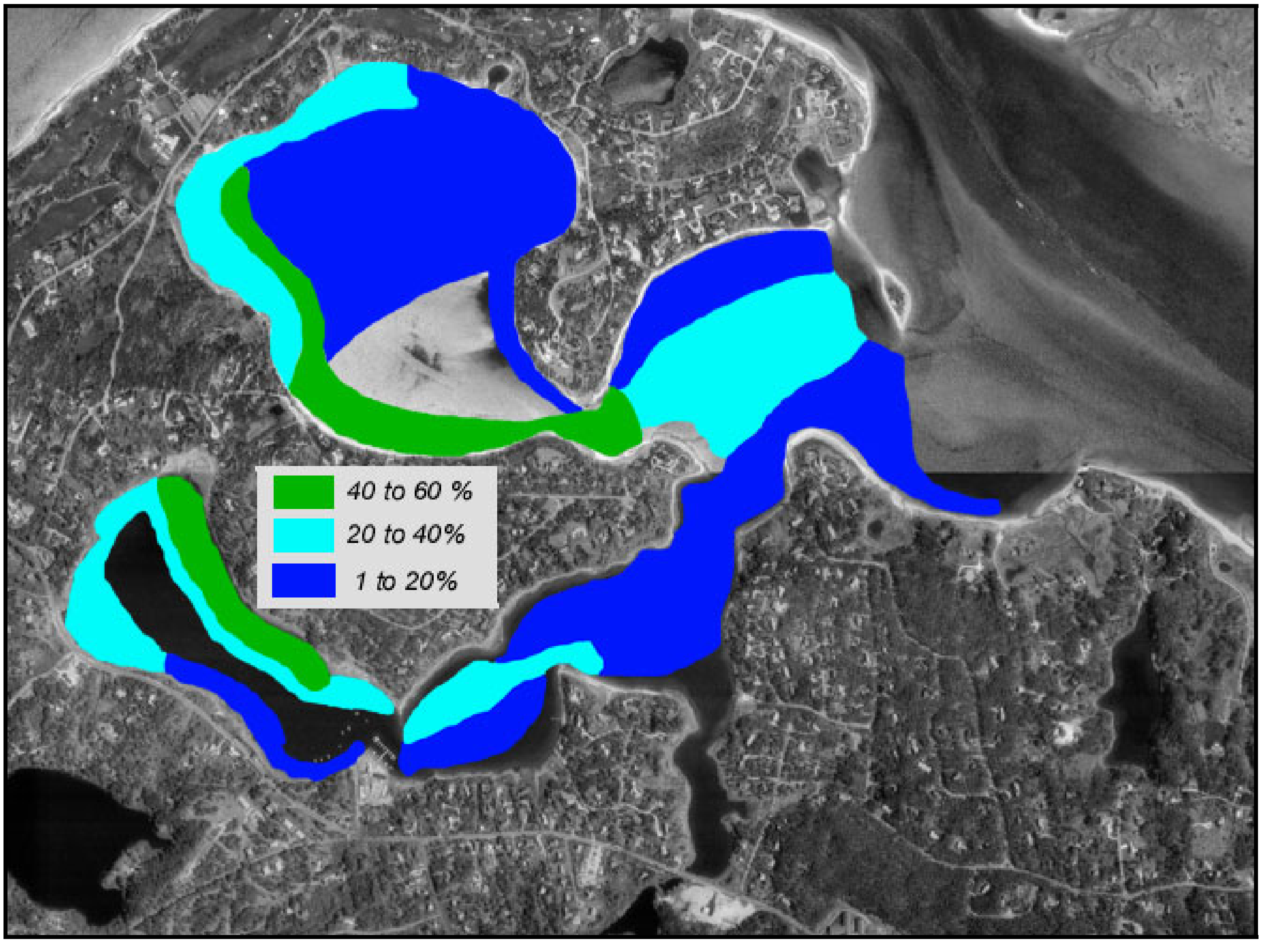


Figure VII-12. Map of Bassing Harbor eelgrass distribution as observed in 2000.

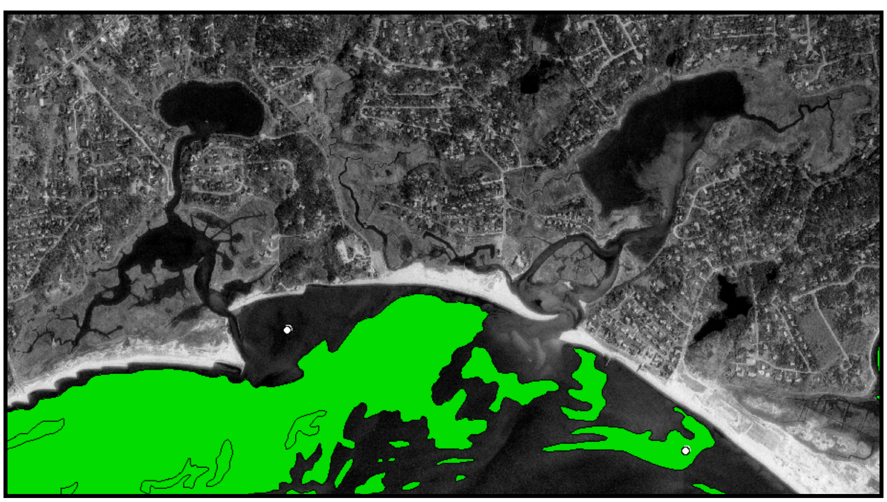


Figure VII-13. Map of Taylors Pond and Sulphur Springs area eelgrass distribution (green shaded area) as determined by Massachusetts DEP in 1994 by analysis of aerial photographs. White circles indicate sites where eel grass coverage was field-confirmed.

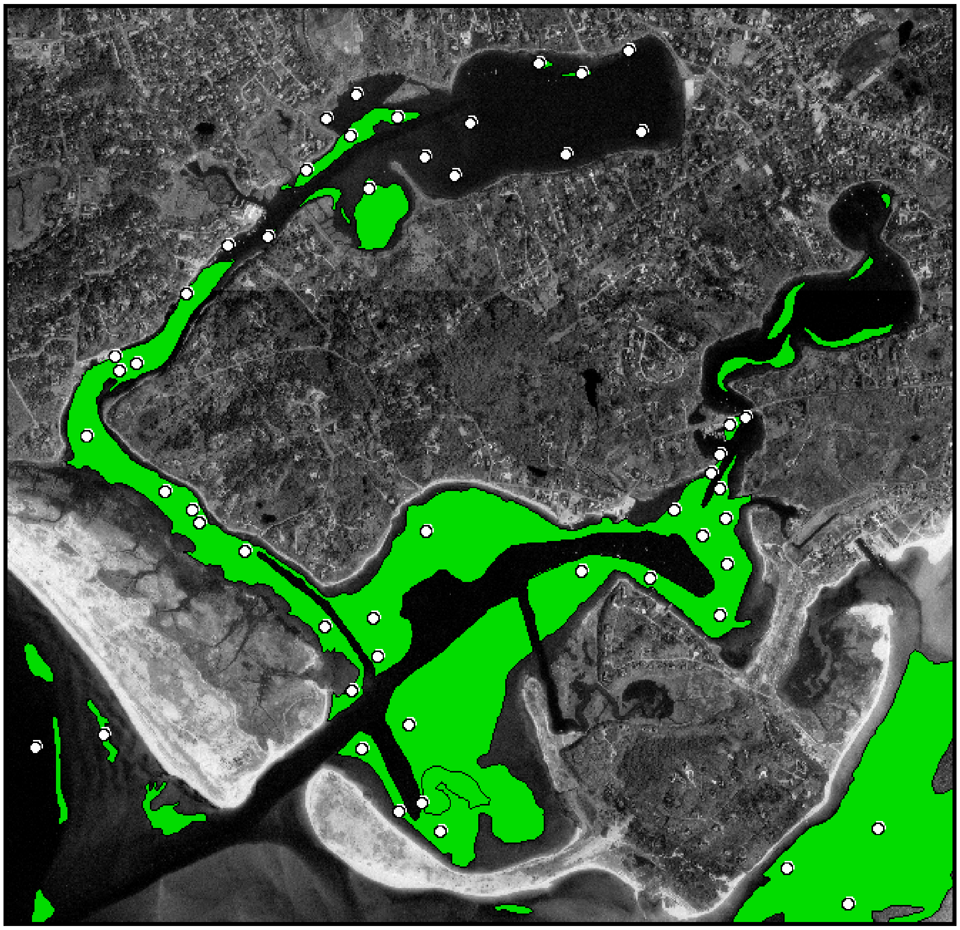


Figure VII-14. Map of Stage Harbor area eelgrass distribution (green shaded area) as determined by Massachusetts DEP in 1994 by analysis of aerial photographs. White circles indicate sites where eel grass coverage was field-confirmed.

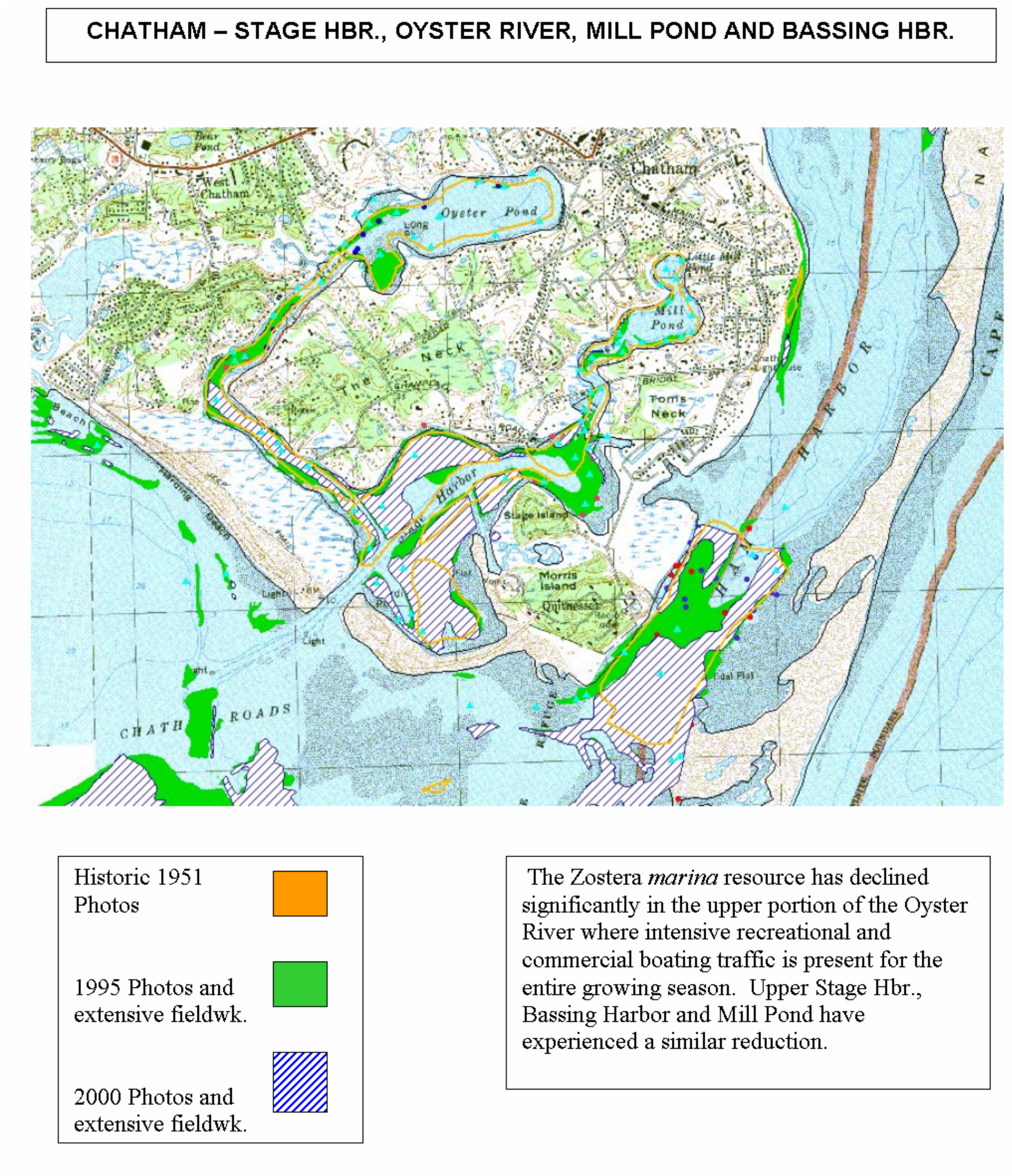
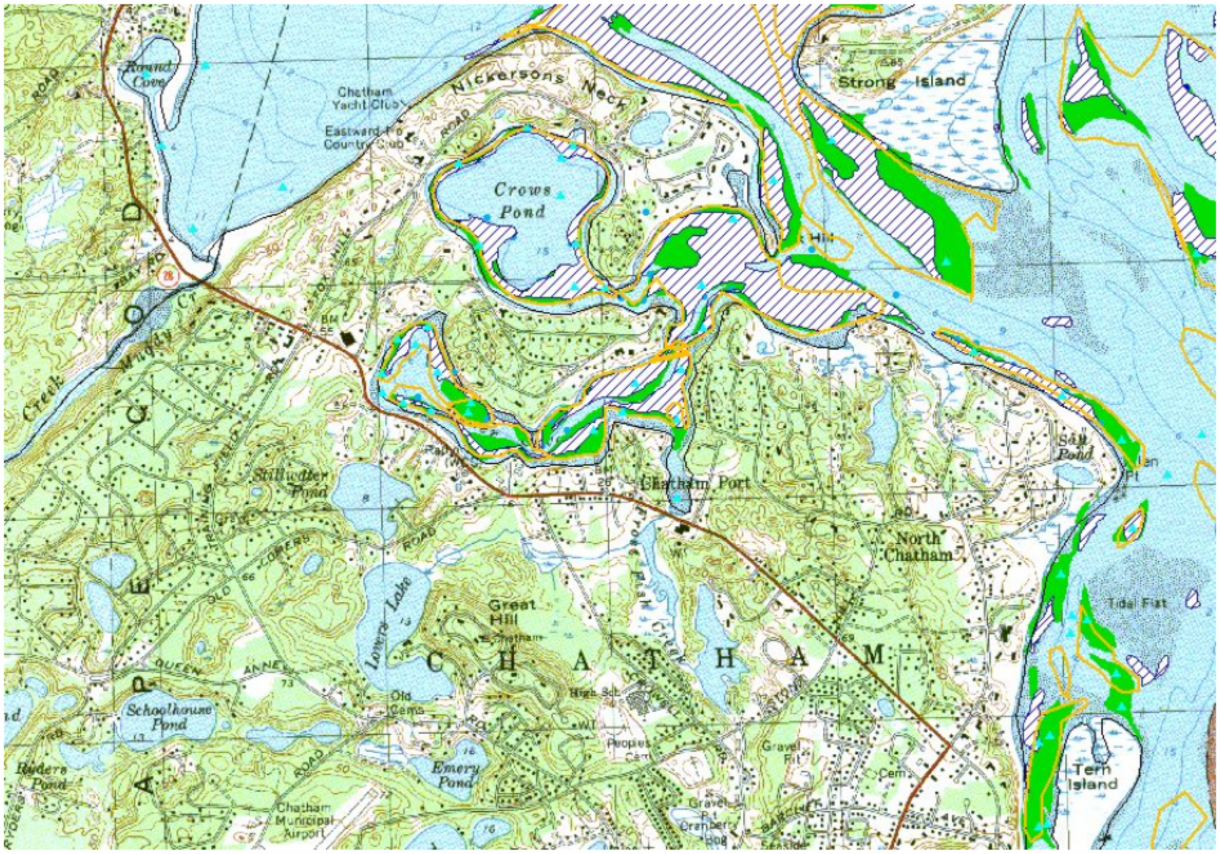


Figure VII-15. Historical eelgrass coverages with the Stage Harbor System. The 1951 coverage is depicted by the orange outline inside of which is the eelgrass beds. The green solid and blue hatched areas depict the bed areas in 1995 and 2000, respectively.

CHATHAM – RYDERS COVE



Historic 1951 Photos

1995 Photos and extensive fieldwk.

2000 Photos and extensive fieldwk.

The Zostera *marina* resource has been relatively stable in the Ryder's Cove area. The Frost Fish Creek area was not included in our survey. Present resources seem to be close to what the historic imagery revealed.

Figure VII-16. Historical eelgrass coverages with the Bassing Harbor System. The 1951 coverage is depicted by the orange outline inside of which is the eelgrass beds. The green solid and blue hatched areas depict the bed areas in 1995 and 2000, respectively.

Table VII-3. Eelgrass coverage in Chatham embayments in 2000 assayed by visual transect surveys. This approach can record the distribution of eelgrass at low density. Therefore the values represent maximum areal coverage.

Embayment (total surface area)	Eel Grass Density	Area (ac)	Coverage Area percentage of total embayment area
Stage Harbor System			
Inner Stage Harbor	> 70%	20.3	26.6
(76.1 ac)	25 to 75%	5.9	7.8
	20 to 50%	4.8	6.4
	< 20%	0.8	1.1
Stage Harbor	25 to 75%	9.6	3.6
(268.2 ac)	20 to 50%	97.5	36.4
	< 20%	2.8	1.0
Oyster Pond River	> 70%	3.9	4.4
(88.1 ac)	40 to 80%	13.2	15.0
	25 to 75%	1.1	1.3
	< 20%	31.3	35.6
Stage Harbor system Total Surface area: 640 ac Stage Harbor system total Eel grass coverage: 191 ac Percent coverage total system: 29.9%			
Bassing Harbor System			
Crows Pond	40 to 60%	17.2	14.8
(115.7 ac)	20 to 40%	17.3	14.9
	1 to 20%	65.4	56.5
Ryder Cove	40 to 60%	9.5	20.3
(46.9 ac)	20 to 40%	15.1	32.1
	1 to 20%	5.1	10.9
Outer Ryder Cover	20 to 40%	6.9	12.8
(54.2 ac)	1 to 20%	34.1	62.9
Bassing Harbor	40 to 60%	3.7	4.3
(86.5 ac)	20 to 40%	26.1	30.1
	1 to 20%	30.8	35.6
Bassing Harbor system Total Surface area: 320 ac Bassing Harbor system total Eel grass coverage: 231 ac Percent coverage total system: 72.2%			

Table VII-4. Changes in eelgrass coverage in the 2 major embayment systems within the Town of Chatham over the past half century (C. Costello). Note: data from Table VII-3 collected by different approach not included.

Embayment*	1951 (acres)	1995 (acres)	2000 (acres)	% Difference (1951 to 2000)
Stage Harbor System	**320**	**267**	**162**	**51%**
Bassing Harbor System	**246**	**153**	**114**	**46%**

*No Eelgrass in the Following Embayment Areas: Sulphur Springs, Muddy Creek, Taylors Pond, Frost Fish Creek.

The pattern of eelgrass loss in these systems is consistent with bed loss from nutrient enrichment. As embayments receive increasing nitrogen inputs from their watersheds, there is typically a resulting gradient in nitrogen levels within embayment waters. In systems like those in Chatham, the general pattern is for highest nitrogen levels to be found within the innermost basins with concentrations declining moving toward the tidal inlet. This pattern is also observed in nutrient related habitat quality parameters, like phytoplankton, turbidity, oxygen depletion, etc. The consequence is that eelgrass bed decline typically follows a pattern of loss in the innermost basins (and sometimes also from the deeper waters of deep basins) first. The temporal pattern is a “retreat” of beds toward the region of the tidal inlet. This is the pattern observed in the 2 major systems in the Town of Chatham.

Other factors which influence eelgrass bed loss in embayments may also be at play in Chatham waters, although the pattern of loss seems diagnostic of nitrogen enrichment. However, a brief listing of non-nitrogen related factors is useful. Eelgrass bed loss does not seem to be directly related to mooring density, as some of the highest mooring areas still support eelgrass, while other areas of low mooring density have lost eelgrass. 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 the loss of eelgrass from the smaller shallower embayments, which do not support significant shellfishing pressure would suggest again that this is not the overarching stress. In fact both the loss from the smaller embayments and pattern of loss within the larger embayments is consistent with nitrogen enrichment as the primary stressor for eelgrass throughout these five of Chatham’s estuaries.

There are several additional conclusions relative to nutrient related habitat quality which can be derived from an examination and comparison of the Year 2000, Year 1994, and Year 1951 eelgrass maps and coverage data (Tables VII-3 and VII-4 show changes to eelgrass coverage). They can be summarized as follows:

- Eelgrass does not presently colonize the smaller embayment systems, most likely due to their high nitrogen levels and periodic depletion of oxygen in these systems. These conditions existed prior to 1994.

- Eelgrass coverage is declining within the Stage Harbor System. Oyster Pond and Oyster Pond River appear to have had bed loss between 1994 and 2000. It is likely that the eelgrass beds within Oyster Pond were relatively extensive in recent times (1970's or 1980's) based upon the apparent rapid rate of loss in other parts of the system and coverage in 1951. Similar to Oyster Pond the Mill Pond tributary to Stage Harbor also appears to be losing eelgrass. The pattern of loss is also similar, with loss beginning in the innermost reaches with migration toward the lower parts of the System. The loss of eelgrass from 1994 to 2000 from Mill Pond, Mitchell River and upper Stage Harbor mirrors the loss from Oyster Pond and Oyster River over the same period.

- It is almost certain that a primary cause of the observed eelgrass decline results from increasing watercolumn nitrogen levels within these environments over the past decades. Areas of loss are generally associated with the higher chlorophyll sites recorded by the moored instruments (Section VII-2).

- Eelgrass coverage does appear to be declining within the overall Bassing Harbor System. Although no eelgrass bed density data was available from the 1994 mapping study, comparison of similar approaches for determining bed coverage indicates a decline from 1951 to 1994 to 2000.

- Eelgrass within portions of Bassing Harbor (near Bassing Island) are colonized by 2 species of tunicates which appear to be causing localized damage to the beds. It appears that both may be introduced bioinvasive organisms (*Botrylloides diegensis* and *Diplosoma sp.*). These beds need to be monitored to the extent that this biological interaction effects their distribution.

- It should be noted that the density of eelgrass in many of the existing coverage areas is relatively sparse (less than 20%). This may indicate a thinning of beds.

- The Sulphur Springs region of the Sulphur Springs/Bucks Creek System (or Cockle Cove System) is currently a region of high production and accumulation of macro-algae. The basin bottom is completely covered during summer with dense accumulations. In addition, the shallow nature of the system has resulted in the colonization of even the main basin by clumps of *Spartina alterniflora*. It appears that this system is beginning to transition to salt marsh.

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

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 15 locations within 4 of the embayment systems. Tidal salt marsh creeks and shallow pools were excluded. Samples were collected from: Ryder Cove, Bassing Harbor, Frost Fish Creek, Crows Pond, Muddy Creek, Stage Harbor, Oyster Pond, Mill Pond, Little Mill Pond, and Taylors Pond. Figure VII-17 shows the benthic infauna sampling stations. 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, dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of excellent or healthy conditions, intermediate in stress, or highly stressed conditions. Both the distribution of species and the overall population density are taken into account. The assemblage was then classified as representative of pristine or healthy conditions, intermediate in stress, or highly stressed conditions. Both the distribution of species and the overall population density were taken into account.

The Infauna Study indicated that most of the upper regions of the embayments are currently supporting habitats under either intermediate or high stress (Table VII-5, VII-6). The lower regions (those nearest the inlets) show higher habitat quality, intermediate to low stress, most likely as a result of the greater dilution of watershed nitrogen inputs by tidal source waters.

The inner “deep” basins, apparently drowned kettle ponds, showed the poorest habitat conditions. Little Mill Pond, Mill Pond (and upper Mitchell River) and Taylors Pond were dominated by stress indicator species. In addition, these systems were supporting low numbers of individuals (except nematodes), indicative of poor nutrient related water quality.

Similar to the “deep” basins, the tidally restricted systems of Muddy Creek and Frost Fish Creek showed very poor habitat quality. This was evidenced by the species present and their low numbers. These systems are heavily nutrient and organic matter loaded. The sediments of Frost Fish Creek and upper Muddy Creek are fluid organic-rich muds, and the assemblages are typical of this type of condition.

The larger basins within the Stage Harbor and Bassing Harbor Systems generally registered as intermediate habitat quality. Only the upper Stage Harbor region and a portion of Crows Pond approached healthy conditions.

Analysis of the evenness and diversity of the benthic animal communities yields a similar evaluation to the natural history information and the evaluation of the number of individuals. The evenness statistic can range from 0-1 (one being most even), while the diversity index does not have a theoretical upper limit. The highest quality habitat areas, as shown by the oxygen and chlorophyll records and eelgrass coverage, have the highest diversity (generally ~3) and evenness (~0.7). These areas are found in the lower regions of the Stage Harbor and Bassing Harbor Systems (for example Crows Pond, Lower Mitchell River, Bassing Harbor). The converse is also true, with poorest habitat quality found in upper Muddy Creek (H’=1.35, E=0.52), Taylors Pond (H’=1.46, E=0.52), Frost Fish Creek (H’=1.53, E=0.66) and Oyster Pond ((H’=1.42, E=0.40)

These results indicate a moderate to high level of nutrient related stress throughout almost all upper regions of Chatham’s embayments (Cockle Cove/Sulphur Springs System not measured). These infauna indicator analysis results are consistent with the levels of nitrogen and oxygen depletion within these systems. In addition, the sediment survey results generally supported the concept of high organic matter loading within the upper poor quality regions of these embayments. The majority of the area within the 2 major embayment systems (Stage Harbor, Bassing Harbor) appear to be experiencing only a moderate level of ecological stress and are supportive of productive and diverse benthic animal communities. These results are also consistent with the water quality monitoring and sediment characteristics data sets.

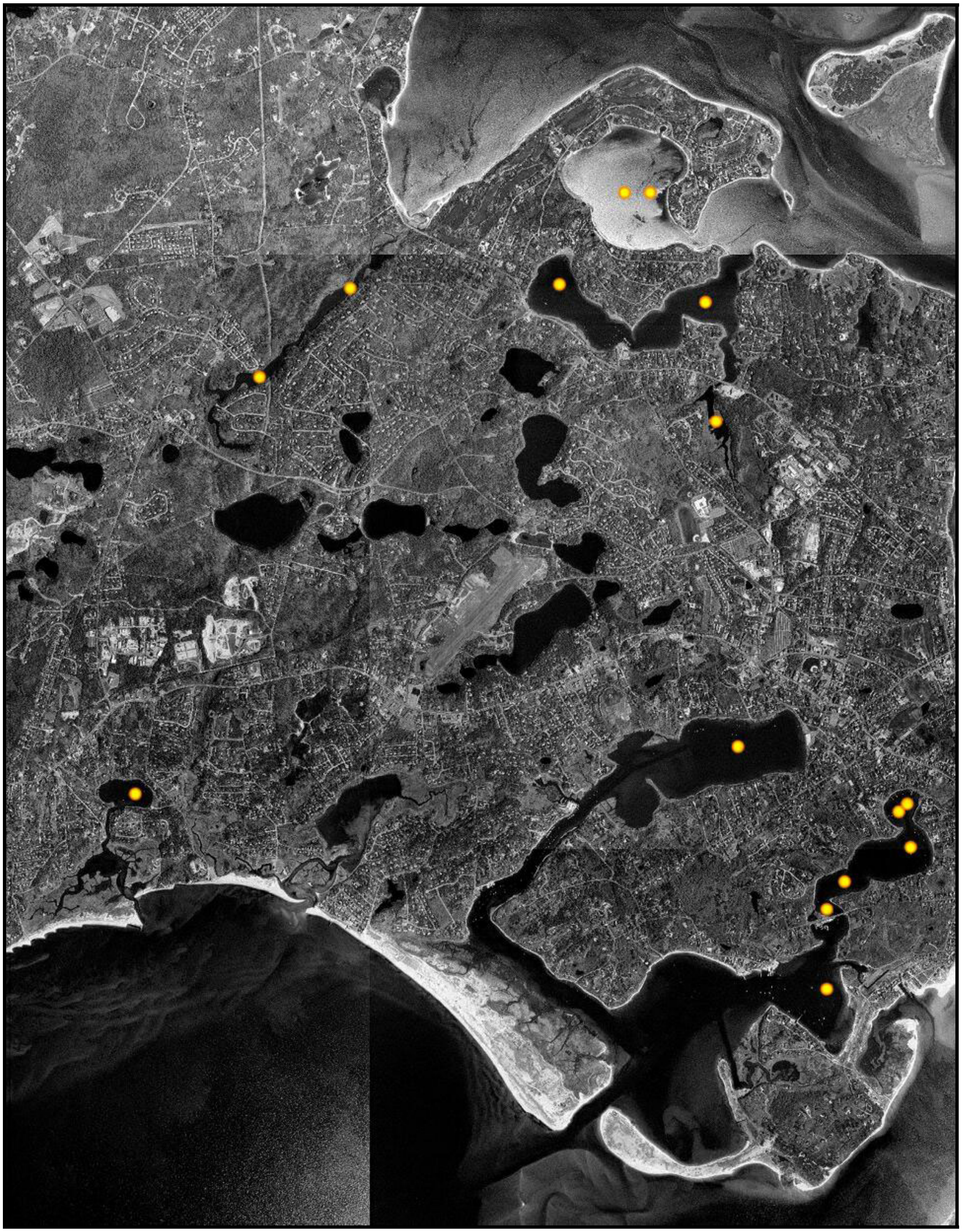


Figure VII-17. Aerial photograph of Chatham showing location of benthic infaunal sampling stations (yellow circles).

Table VII-5. Benthic Infaunal Community Assessment for Chatham Embayments. Samples collected Summer and Fall of 2000. All data is represented as per 1/25 m2. Indicator assessment based upon life history information (G.R. Hampson, SMAST).

Embayment	Total #Species	Total #Individuals	Healthy #Species	Healthy #Individuals	Intermediate #Species	Intermediate #Individuals	Stressed #Species	Stressed #Individuals	Classification
			Benthic Infaunal Community - Indicators						
Stage Harbor System:									
Oyster Pond	12	1090	6	109	3	962	3	19	Intermediate
Little Mill Pond - Rep 1	2	>600	0	0	1	Abundant	1	Abundant	Stressed
Little Mill Pond - Rep 2	2	>600	0	0	1	Abundant	1	Abundant	Stressed
Mill Pond	2	317	0	0	0	0	2	317	Stressed
Mitchell river Upper	17	506	9	55	4	435	4	16	Intermediate
Mitchell river Lower	23	1037	15	469	6	555	2	13	Intermediate/Healthy
Stage Harbor Upper	20	470	12	62	4	337	4	71	Intermediate/Healthy
Basing Harbor System:									
Bassing Harbor	17	137	11	28	4	73	2	36	Intermediate
Crows Pond – I	29	287	23	186	4	87	2	14	Intermediate/Healthy
Crows Pond – O	29	373	22	180	4	28	3	165	Intermediate
Ryder Cove	19	634	14	131	3	339	2	164	Intermediate
Frost Fish Creek	5	125	1	1	3	64	1	60	Stressed
Taylors Pond	6	43	3	3	3	40	0	0	Stressed

Table VII-6. Benthic infaunal community data for the 5 embayment systems. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations.

System	Location	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Muddy Creek System						
Muddy Creek	Upper	6	77	6	1.35	0.52
Muddy Creek	Lower	8	200	7	2.02	0.67
Stage Harbor System						
Little Mill Pond	Rep 1	1	17	NA	0.00	NA
	Rep 2	No Infauna	NA	NA	NA	NA
Mill Pond	Mid	2	317			
Mitchell River	Upper	18	520	11	1.91	0.46
	Lower	23	1037	14	3.10	0.69
Stage Harbor	Upper	20	470	10	1.86	0.43
Oyster Pond	Mid	12	1090	6	1.42	0.40
Bassing Harbor System						
Ryder's Cove		18	633	11	1.81	0.43
Bassing Is.		16	136	13	3.06	0.77
Crows Pond	Inner	29	287	18	3.76	0.77
Crows Pond	Outer	30	374	18	3.63	0.74
Frost Fish Creek		5	125	15	1.53	0.66
Taylor's Pond System						
Taylor's Pond	Basin	7	44	NA	1.46	0.52

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

VIII-1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site specific nitrogen thresholds for an embayment requires the integration of key habitat parameters (infauna and eelgrass), sediment characteristic data, and nutrient related water quality information, (particularly dissolved oxygen and chlorophyll a). Additional information on temporal changes within each sub-embayment and its watershed further strengthen the analysis. These data were all collected to support threshold development in the Stage Harbor, Bassing Harbor, Muddy Creek, Sulphur Springs and Taylor Pond Systems by the MEP Team and were discussed in Section VII. Nitrogen threshold development builds on these data and links habitat quality to summer water column nitrogen levels from long-term baseline water quality monitoring (Chatham Water Watchers, Pleasant Bay Alliance, and MEP Team; Table VIII-1).

The five embayment systems in this study displayed a range of habitat quality, both between systems and along the longitudinal axis of the larger systems. In general, sub-embayments show decline in habitat quality moving from the inlet to the inland-most tidal reach. This trend is seen in both the nitrogen levels (highest inland), eelgrass distribution, infaunal community stress indicators and community properties, as well as summer dissolved oxygen and chlorophyll a records. The following is a brief synopsis of the present habitat quality within each of the five embayment systems. The underlying quantitative data is presented on nitrogen (Section VI), oxygen and chlorophyll a (Section VII-1), eelgrass (Section VII-2), and benthic infauna (Section VII-3).

Stage Harbor System – Little Mill Pond, Mill Pond, and Oyster Pond have elevated nitrogen levels and have lost historic eelgrass beds which once covered most of their respective basins. Oxygen depletion is observed during summer in each system with Mill Pond (and presumably Little Mill Pond) having ecologically significant declines (<3 mg L^{-1}). Oyster Pond had less oxygen depletion possibly due to its greater fetch for ventilation with the atmosphere. Chlorophyll a levels were consistent with the observed oxygen depletion. The lower reaches of the Oyster River and Upper Stage Harbor show good habitat quality as evidenced by their persistent eelgrass beds, infaunal community structure and oxygen and chlorophyll a levels. The inner-most high quality habitat is found in the lower Mitchell River/upper Stage Harbor.

Sulphur Springs System – Cockle Cove consists primarily of a salt marsh and central tidal creek. This system contains little water at low tide and has a high assimilative capacity for nitrogen as do other New England salt marshes. Sulphur Springs is a shallow basin containing significant macroalgal accumulations, no eelgrass, and appears to be transitioning to salt marsh. However, Sulphur Springs basin is still functioning as an embayment, but a eutrophic one. Nitrogen levels are high (Section VI), oxygen levels become significantly depleted (6% of time <3 mg L^{-1}) and phytoplankton blooms are common and large (chlorophyll a levels >20 ug L^{-1}). Eelgrass has not been observed for over a decade.

Table VIII-1. Assessment of nitrogen related habitat quality within the embayments of the Town of Chatham. Water quality stations and benthic stations were in the same basin, but not in the exact same locations. Data for this comparison is from 2000, when the eelgrass mapping and benthic infauna were assayed. Note that the 1998-2002 water quality data was used in the validation of the water quality model and that the moored instrumentation captured a greater range of dissolved oxygen and chlorophyll a than the water quality sampling programs. Ecological Assessment Classification (SMAST) attempts to integrate water quality and habitat indicators, as well as any temporal trends which have been identified. No data is represented by "-- --".

Embayment System	Depth m	Salinity ppt	Minimum D.O. Mg/L	Secchi depth m	Nitrogen DIN mg N/L	Nitrogen TN mgN/L	Phytoplankton Tot-Pig ug/L	Sediment Type	Sediment Carbon mgC/cc	MacroAlgae Abundance	Eelgrass* Cover/Density/Status	Infaunal** Community Classification	Ecological*** Assessment Class/Status
Stage Harbor System:													
Oyster Pond	2.85	29.8	7.06	2.13	0.05	0.79	5.18	Mud/Sand	737	Low	Sparse/Low/Decline	Intermediate	Mod-Fair/Decline
Oyster River	1	30.1	6.23	2.13	0.04	0.46	4.43	Sand	236	-- --	Mod/Mod/Decline	-- --	Mod-High/Decline
Stage Harbor	1.6	30.2	7.09	1.84	0.04	0.66	5.63	Sand	546	-- --	Mod/Mod/Decline	-- --	Mod-High/Decline
Stage Harbor - Upper	1.6	30	7.45	2.22	0.04	0.46	4.2	Sand/mud	950	-- --	Mod/High/Decline	Intermed/Healthy	Mod-High/Decline
Mitchell River								Sand	294	-- --	Low/Low/Decline	Intermediate	Mod/Decline
Mill Pond	4.46	30	6.57	2.12	0.04	0.50	5.2	Mud	815	-- --	0	Stressed	Poor
Little Mill Pond	4.15	29.8	5.63	2.34	0.09	0.69	6.74	Mud	1334	-- --	0	Stressed	Poor
Taylors Pond System:													
Taylors Pond	2.18	28.3	5.85	1.76	0.06	0.51	7.03	Mud	1624	Moderate	0	Stressed	Poor
Mill Creek	1.01	28.3	5.26	1	0.06	0.49	6.35	Sand	702	Moderate	-- --	-- --	-- --
Cockle Cove System:													
Sulphur Springs	1.03	28.6	4.8	1.03	0.04	0.36	5.56	Mud/Sand	1246	High	0	-- --	Poor
Bucks Creek	0.82	28.1	5.86	0.8	0.05	0.40	4.66	Sand	853	Moderate	0	-- --	Moderate
Cockle Cove Cr Mid	NA	0	NA	-- --	0.25	1.49	-- --	-- --	-- --	-- --	-- --	-- --	-- --
Cockle Cove Cr Low	0.31	24.5	2.78	0.31	0.20	0.89	6.35	Sand	300	-- --	-- --	-- --	Poor
Basing Harbor System													
Bassing Harbor	1.8	28.7	6.78	1.42	0.05	0.54	5.63	Sand/mud	1186	-- --	High/Mod/Stable-Incr.	Intermediate	Mod-High
Crows Pond	4.98	29.2	6.58	1.97	0.11	0.76	5.92	Sand/mud	1292	-- --	Mod/Mod/Decline	Intermediate	Moderate
Ryder Cove - Inner	2.34	29.1	6.04	2	0.06	0.47	6.29	Mud/Sand	899	-- --	Mod/High/Decline.	Intermediate	Moderate
Ryder Cove - Outer	3.5	28.2	6.55	2.35	0.04	0.44	6.45	Mud/Sand	1210	-- --	High/Low/Stable	-- --	Moderate
Frost Fish Outer	1.1	28.5	5.48	1.1	0.16	1.24	11.02	-- --	-- --	-- --	0	-- --	Poor
Frost Fish Inner	0.6	15.3	NA	-- --	0.30	0.92	-- --	Mud	792	-- --	0	Stressed	Poor
Muddy Creek System:													
Upper	-- --	-- --	-- --	-- --	-- --	-- --	-- --	Mud	940	-- --	0	Stressed	Poor
Lower	1.37	25.6	6.33	1.18	0.04	0.57	9.99	Mud	1447	-- --	Sparse Patch	Stressed	Poor

* Eelgrass coverage was classified as High, Moderate (Mod), Low and Absent (0); the stability of the beds was based upon areal changes since the DEP survey of 1994 (Costello).
** Infaunal communities classification is based upon the composition and number of species representative of "healthy", "Intermediate", and "stressed" conditions.
*** Classification is based upon High - low nutrient stress & high habitat quality; Moderate - moderate to fair nutrient stress & moderate habitat quality; Low - high nutrient related stress and poor habitat quality.

Taylors Pond System – Taylors Pond represents the inland-most sub-embayment and is a drowned kettle pond. The lower portion of this system is comprised of a tidal salt marsh, Mill Creek. Like the Sulphur Springs System, the inner basin functions as an embayment and the tidal creek as a salt marsh with low sensitivity to nitrogen inputs. Taylors Pond is currently showing poor habitat quality. There is currently no eelgrass community and no record of eelgrass for over a decade. Watercolumn nitrogen levels are enriched over incoming tidal waters (Section VI) and dissolved oxygen depletion to ~4 mg L^{-1} is common. Chlorophyll a levels of 10-15 ug L^{-1} are common during summer. The benthic infaunal community is impoverished, with only a mean of 43 individuals collected in the grab samples, compared to several hundred in the high quality sub-embayments.

Bassing Harbor System – The inner-most sub-embayments to this system contain high quality habitat that is currently becoming impaired by nitrogen enrichment. Ryder Cove receives the greatest watershed nitrogen load of the Bassing Harbor sub-systems. This sub-embayment has been losing its eelgrass over at least the last decade. In 1951 the full basin appears to have supported eelgrass beds many of which do not exist today. Infaunal communities indicate a moderate quality system with relatively low diversity and evenness. This is consistent with a system whose habitat is in transition from high to moderate level of quality. Upper Ryder Cove is currently showing bottom water oxygen depletion, frequently to <4 mg L^{-1} and occasionally to < 3 mg L^{-1}. The periodic oxygen declines, loss of eelgrass, and watershed nitrogen loading is consistent with the observed phytoplankton blooms, which generally (>40% of time) are >15 ug L^{-1} and frequently >20 ug L^{-1}. In contrast, the outer reach of Ryder Cove still supports relatively high habitat quality with dissolved oxygen levels almost always above 5 mg L^{-1} (99%) and moderate chlorophyll a levels (<15 ug L^{-1}). These watercolumn parameters are consistent with the high eelgrass coverage. Crows Pond is the other inland-most sub-embayment in this bifurcated estuary. However, Crows Pond has a significantly lower watershed nitrogen load than that to Ryder Cove. Crows Pond currently supports a high level of habitat quality, with eelgrass beds surrounding the central basin and sparse coverage throughout. Infaunal diversity and evenness is consistent with a high quality habitat. Oxygen levels are consistently above 5 mg L^{-1} and chlorophyll a levels also are moderate (generally 10-15 ug L^{-1}). However, it appears that habitat quality is currently declining. Eelgrass coverage is less than in the 1951 and 1995 records. At present it appears the Crows Pond is slightly beyond its threshold nitrogen level and is beginning to decline in habitat quality. In addition, Frost Fish Creek is a tributary system to outer Ryder Cove which functions primarily as a salt marsh with a central basin (Section IV, Section VI). The outer-most basin is Bassing Harbor which receives tidal exchanges with Pleasant Bay. Bassing Harbor currently supports high habitat quality and based upon the eelgrass records has been relatively constant since 1951. The infaunal community is consistent with high habitat quality as is the maintenance of oxygen levels and moderate to low chlorophyll a levels (typically 5-10 ug L^{-1}. The Bassing Harbor sub-embayments appears to be a relatively stable high habitat quality system, with demonstrated good eelgrass and infaunal communities.

Muddy Creek – Muddy Creek like Bassing Harbor exchanges tidal waters with the greater Pleasant Bay System. However, unlike Bassing Harbor, Muddy Creek is a highly eutrophic embayment. Muddy Creek does not support significant eelgrass beds; however, a small sparse bed has persisted adjacent to the inlet. Muddy Creek is divided into an upper and lower portion by a dike whose weir has been removed or washed away. Both portions are highly eutrophic with frequent bottomwater anoxia and large algal blooms (chlorophyll a frequently >50 ug L^{-1}). The upper portion has a lower habitat quality than the lower portion, most likely as a result of access to the higher quality waters entering from Pleasant Bay. An infaunal community persists but it is dominated by species tolerant of organic enrichment. Species diversity and evenness are low. The whole of Muddy Creek currently supports nitrogen impaired habitat of poor quality.

VIII-2. THRESHOLD NITROGEN CONCENTRATIONS

The threshold nitrogen level for an embayment represents the average watercolumn concentration of nitrogen that will support the habitat quality being sought. The watercolumn nitrogen level is ultimately controlled by the watershed nitrogen load and the nitrogen concentration in the inflowing tidal waters (boundary condition). The watercolumn nitrogen concentration is modified by the extent of sediment regeneration.

Threshold nitrogen levels for each of the five embayment systems in this study were developed to restore or maintain SA waters or high habitat quality. In these five systems, high habitat quality was defined as supportive of eelgrass and infaunal communities. Dissolved oxygen and chlorophyll a were considered in the assessment.

The approach developed by the MEP has been to select a sentinel sub-embayment within each embayment system. First, a sentinel sub-embayment is selected based upon its location within the system. The sentinel should be close to the inland-most reach as this is typically where water quality is lowest in an embayment system. Therefore, restoration or protection of the sentinel sub-embayment will necessarily create high quality habitat throughout the estuary. Second, a sentinel sub-embayment should be sufficiently large to prevent steep horizontal water quality gradients, such as would be found in the region of entry of a stream or river or in the upper most region of a narrow, shallow estuary. This second criteria relates to the ability to accurately determine the baseline nitrogen level and to conduct the predictive modeling runs. Finally, the sentinel system should be able to obtain the minimum level of habitat quality acceptable for the greater system (unless a multiple classification is to be used).

After the sentinel sub-system (or systems) is selected, the nitrogen level associated with high and stable habitat quality typically derived from a lower reach of the same system or an adjacent embayment is used as the nitrogen concentration target. Finally, the watershed nitrogen loading rate is manipulated in the calibrated water quality model to determine the watershed nitrogen load which will produce the target nitrogen level within the sentinel system. Differences between the required modeled nitrogen load to achieve the target nitrogen level and the present watershed nitrogen load represent nitrogen management goals for restoration or protection of the embayment system as a whole.

The threshold nitrogen levels for the each embayment system was determined as follows:

Stage Harbor System – This embayment system has two upper reaches. Therefore, two sentinel sub-embayments were selected, mid-Oyster Pond and Mill Pond. Little Mill Pond could not be used because it is small and has steep horizontal nitrogen gradients (see Section VI). Within the Stage Harbor System, the uppermost sub-embayment supportive of high quality habitat was upper Stage Harbor (Section VII, VIII-1). Watercolumn total nitrogen levels within this embayment region vary with the tidal stage due to high nitrogen outflowing waters and low nitrogen inflowing waters (Section VI). The calibrated water quality model for this system indicates an average total nitrogen level in the upper Stage Harbor of about 0.40 mg N L^{-1} is most representative of the conditions within this sub-embayment. However, upper Stage Harbor does not appear to be stable based upon changes in eelgrass distribution. Therefore, a nitrogen level reflective of conditions closer to the inlet should achieve the stability required. The lower nitrogen level is equivalent to the tidally averaged total nitrogen concentration mid-way between upper Stage Harbor and Stage Harbor or 0.38 mg N L^{-1}. This threshold selection is supported by the fact that the high quality and stable habitat near the mouth of the Oyster River is also at a tidally averaged total nitrogen concentration of 0.37 mg N L^{-1}. The 0.38 mg N

L^{-1} was used to develop watershed nitrogen loads required to reduce the average nitrogen concentrations in each sentinel system to this level. Tidal waters inflowing from Nantucket Sound have an average concentration of total nitrogen of 0.285 mg N L^{-1}.

Sulphur Springs System – The Sulphur Springs basin is both the inland-most sub-embayment and also represents the largest component of the Sulphur Springs System (which also includes Mill Creek and Bucks Creek). Since this System exchanges tidal waters with Nantucket Sound (0.285 mg N L^{-1}), as does Stage Harbor, and since there is currently no high quality habitat within this system, Stage Harbor habitat quality information was used to support the Sulphur Springs thresholds analysis. The tidally averaged nitrogen threshold concentration for this system was determined to be the same as for the sentinel sub-embayments to the Stage Harbor System or 0.38 mg N L^{-1}. The 0.38 mg N L^{-1} was used to develop watershed nitrogen loads required to reduce the average nitrogen concentrations in the Sulphur Springs sentinel system to this level.

Taylors Pond System – This system was approached in a similar manner to the Sulphur Springs System and for the same reasons. Taylors Pond represents the innermost and functional embayment within this system. This system also exchanges tidal waters with Nantucket Sound (0.285 mg N L^{-1}), as does the Stage Harbor System and there is no high quality stable embayment habitat within this system. Therefore, the tidally averaged nitrogen threshold concentration for this system was determined to be the same as for the sentinel sub-embayments to the Stage Harbor System or 0.38 mg N L^{-1}. The 0.38 mg N L^{-1} was used to develop watershed nitrogen loads required to reduce the average nitrogen concentrations in Taylors Pond to this level.

Bassing Harbor System – Although this system has two inland-most sub-embayments, Ryder Cove and Crows Pond, only Ryder Cove was selected as the sentinel system. This resulted from the fact that Crows Pond has a relatively low nitrogen load from its watershed and appears to currently support higher quality habitat than Ryder Cove. Ryder Cove currently shows a gradient in habitat quality with lower quality habitat in the upper reach and higher quality in the lower reach. Ryder Cove represents a system capable of fully supporting eelgrass beds and stable high quality habitat. At present, this basin is transitioning from high to low habitat quality in response to increased nitrogen loading. Restoration of nitrogen levels in upper Ryder Cove to levels supportive of high quality habitat should also result in the restoration and protection of the whole of the Bassing Harbor System.

Following the approach used for the Stage Harbor System, a region of stable high quality habitat was selected within the Bassing Harbor System. The region selected was Bassing Harbor which has both high quality eelgrass and benthic animal communities, which appear to be stable. Unfortunately, total nitrogen within this system appears to be very high. In fact, the whole of lower Pleasant Bay appears to contain very high levels of total nitrogen. Analysis of the composition of the watercolumn nitrogen pool within these embayments revealed that the concentrations of dissolved inorganic nitrogen (DIN) and particulate organic nitrogen (PON) were the same as for the Stage Harbor System. In fact, the level of these combined pools (DIN+PON) was lower in Bassing Harbor (0.133 mg N L^{-1}) than in the Stage Harbor (0.158 mg N L^{-1}) and the mouth of Oyster River (0.160 mg N L^{-1}). It appears that the reason for the higher total nitrogen levels in the Pleasant Bay waters results from the accumulation of dissolved organic nitrogen. The bulk of dissolved organic nitrogen (DON) is relatively non-supportive of phytoplankton production in shallow estuaries, although some fraction is actively cycling. It is likely that the high background DON results from the relatively long residence time of Pleasant Bay waters relative to the smaller systems. This allows the accumulation of the less biologically active nitrogen forms, hence the higher background. Decomposition of phytoplankton, macroalgae and eelgrass release DON to estuarine waters as do salt marshes and surface freshwater inflows.

Based upon these site-specific observations, an adjusted nitrogen threshold could be developed for the Bassing Harbor System. The approach was to determine the baseline dissolved organic nitrogen level for the region (average of inner and outer Ryder Cove, Bassing Harbor, Frost Fish Creek, Tern Island, and Pleasant Bay), which was determined to be 0.394 mg N L^{-1}. A threshold range was then developed using a conservative DIN+PON level from the Bassing Harbor sub-embayment plus the dissolved organic nitrogen background and an upper threshold based upon the Stage Harbor DIN and PON values discussed above. The threshold range for this system was set as 0.527 mg N L^{-1} to 0.552 mg N L^{-1} and the higher threshold was used to develop watershed nitrogen loads required to reduce the average nitrogen concentrations in upper Ryder Cove to this level. The nitrogen boundary condition (the concentration of nitrogen in inflowing tidal waters from Pleasant Bay) for the Bassing Harbor System is 0.48 mg N L^{-1}.

Muddy Creek System – This system is highly eutrophic. Given the long narrow basin and the hydrodynamic evaluation (Section V), it was decided to make lower Muddy Creek the sentinel system. This is also based upon the fact that the upper portion was historically a freshwater system. Following the approach for the Bassing Harbor System, the MEP Team considered the Ryder Cove Threshold appropriate for application to lower Muddy Creek. Note that lower Muddy Creek recently supported a sparse eelgrass bed. The threshold was used to develop watershed nitrogen loads required to reduce the average nitrogen concentrations in lower Muddy Creek to this level. However, threshold relates to The nitrogen boundary condition (the concentration of nitrogen in inflowing tidal waters from Pleasant Bay) for the Muddy Creek System is 0.50 mg N L^{-1}.

VIII-3. DEVELOPMENT OF TARGET NITROGEN LOADS

The tidally averaged total nitrogen thresholds derived in Section VIII-2 were used to adjust the calibrated constituent transport model developed in Section V. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold levels in each sentinel system.

As shown in Table VIII-2, the nitrogen load reductions within the Stage Harbor system necessary to achieve the threshold nitrogen concentrations were relatively high, with more than 90% removal of septic load required within three sub-embayments (Oyster Pond, Oyster River, and Stage Harbor). For the other south coastal embayments (Sulphur Springs and Taylors Pond systems), between 50% and 60% of the septic load would need to be removed to achieve the nitrogen concentration targets. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis are shown in Figures VIII-1 through VIII-6.

As shown in Table VIII-3, the nitrogen load reductions within the Bassing Harbor system necessary to achieve the threshold nitrogen concentrations were relatively low, with between 30% and 50% removal of septic load required within the sub-embayments. For Muddy Creek, between 50% and 60% of the septic load would need to be removed to achieve the nitrogen concentration targets for Lower Muddy Creek. Modeling to attain this target for upper Muddy Creek indicated that most of the load would have to be removed. This resulted in a variety of modeling scenarios, which are presented in Chapter IX, and the development of a possible dike scenario (which would require additional modeling for full consideration). The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis are shown in Figures VIII-7 through VIII-10.

Tables VIII-4 and VIII-5, show the total nitrogen load associated with the threshold scenarios for the south coastal and Pleasant Bay embayments, respectively. Due to the high fraction of septic load relative to the total nitrogen load to each sub-embayment, the percent of total load that needs to be removed to achieve the threshold targets is only slightly lower than the percent of septic load that needs to be removed. A more complete breakdown of the nitrogen loads for each of the threshold scenarios modeled is shown in Tables VIII-6 and VIII-7.

Although the above modeling results provide one manner of achieving the selected threshold levels for the sentinel sub-embayments within each estuarine system, the specific examples do not represent the only method for achieving this goal. However, the thresholds analysis provides general guidelines needed for the nitrogen management of these systems. Future water quality modeling scenarios can be run based on other nitrogen removal strategies.

Table VIII-2. Comparison of sub-embayment watershed septic loads used for modeling of present and threshold loading scenarios of the South Coastal embayments and Stage Harbor systems. These loads represent groundwater load contribution from septic systems only, and do not include runoff, fertilizer, atmospheric deposition and benthic flux loading terms.

Sub-embayment	Present Septic Load g/day)	New Septic Load (kg/day)	Threshold % Change
Stage Harbor			
Oyster Pond	11.16	0.11	-99%
Oyster River	9.69	0.79	-92%
Stage Harbor	2.32	0.00	-100%
Mitchell River	5.57	2.66	-52%
Mill Pond	1.55	0.59	-62%
Little Mill Pond	1.35	0.65	-52%
Sulphur Springs			
Sulphur Springs	13.74	6.67	-52%
Bucks Creek	3.51	1.62	-54%
Cockle Cove Creek	2.72	2.72	0%
Waste Water TF	3.03	3.03	0%
Taylors Pond			
Mill Creek	5.33	2.14	-60%
Taylors Pond	7.11	2.91	-59%

Table VIII-3. Comparison of sub-embayment watershed septic loads used for modeling of present and threshold loading scenarios of the Pleasant Bay embayment systems. These loads represent groundwater load contribution from septic systems only, and do not include runoff, fertilizer, atmospheric deposition and benthic flux loading terms.

Sub-embayment	Present Septic Load (kg/day)	New Septic Load (kg/day)	Threshold % Change
Bassing Harbor			
Crows Pond	5.12	3.32	-35%
Ryder Cove	11.14	5.71	-49%
Frost Fish Creek	3.09	2.17	-30%
Bassing Harbor	2.41	1.48	-39%
Muddy Creek			
Muddy Creek -lower	11.49	4.71	-59%
Muddy Creek - upper	16.69	7.07	-58%

Table VIII-4. Comparison of sub-embayment watershed loads (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the South Coastal embayments and Stage Harbor systems. These loads do not include atmospheric deposition and benthic flux loading terms. Note that this is but one of many approaches for reaching the "target" N value.

Sub-embayment	Present Total Load (kg/day)	Threshold Total Load (kg/day)	Threshold % Change
Stage Harbor			
Oyster Pond	13.03	1.98	-85%
Oyster River	11.47	2.76	-76%
Stage Harbor	2.76	0.44	-84%
Mitchell River	6.38	3.47	-46%
Mill Pond	1.78	0.81	-54%
Little Mill Pond	1.64	0.93	-43%
Sulphur Springs			
Sulphur Springs	15.33	8.26	-46%
Bucks Creek	4.08	2.18	-46%
Cockle Cove Creek	6.66	6.66	0%
Waste Water TF	3.03	3.03	0%
Taylors Pond			
Mill Creek	6.22	3.03	-51%
Taylors Pond	8.21	4.01	-51%

Table VIII-5. Comparison of sub-embayment watershed loads (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Pleasant Bay embayment systems. These loads do not include atmospheric deposition and benthic flux loading terms.

Sub-embayment	Present Total Load (kg/day)	Threshold Total Load (kg/day)	Threshold % Change
Bassing Harbor			
Crows Pond	5.79	4.01	-30.6%
Ryder Cove	12.35	6.92	-44.0%
Frost Fish Creek	3.59	2.67	-25.7%
Bassing Harbor	2.66	1.73	-35.1%
Muddy Creek			
Muddy Creek -lower	13.36	6.58	-50.8%
Muddy Creek - upper	19.05	9.43	-50.5%

Table VIII-6. Sub-embayment loads used for nitrogen threshold scenarios run for the Stage Harbor and South Coastal embayment systems, with total watershed N loads, atmospheric N loads, and benthic flux.

Sub-embayment	Watershed Load (kg/day)	Atmospheric Deposition (kg/day)	Benthic Flux (kg/day)
Stage Harbor			
Oyster Pond	1.98	0.29	10.2
Oyster River	2.76	1.05	0.3
Stage Harbor	0.44	3.25	4.9
Mitchell River	3.47	0.88	-1.3
Mill Pond	0.81	0.63	1.4
Little Mill Pond	0.93	0.12	0.8
Sulphur Springs			
Sulphur Springs	8.26	0.38	-2.3
Bucks Creek	2.18	0.13	1.9
Cockle Cove Creek	6.66	0.06	-0.6
Waste Water TF	3.03	-	
Taylors Pond			
Mill Creek	3.03	0.17	-0.2
Taylors Pond	4.01	0.19	-0.9

Table VIII-7. Sub-embayment loads used for nitrogen threshold scenarios run for the Bassing Harbor and Muddy Creek systems of Pleasant Bay, with total watershed N loads, atmospheric N loads, and benthic flux.

Sub-embayment	Watershed Load (kg/day)	Atmospheric Deposition (kg/day)	Benthic Flux (kg/day)
Bassing Harbor			
Crows Pond	4.01	1.39	2.6
Ryder Cove	6.92	1.30	5.6
Frost Fish Creek	2.67	0.10	-0.1
Bassing Harbor	1.73	1.08	-0.1
Muddy Creek			
Muddy Creek –lower	6.58	0.21	-0.9
Muddy Creek - upper	9.43	0.20	2.3

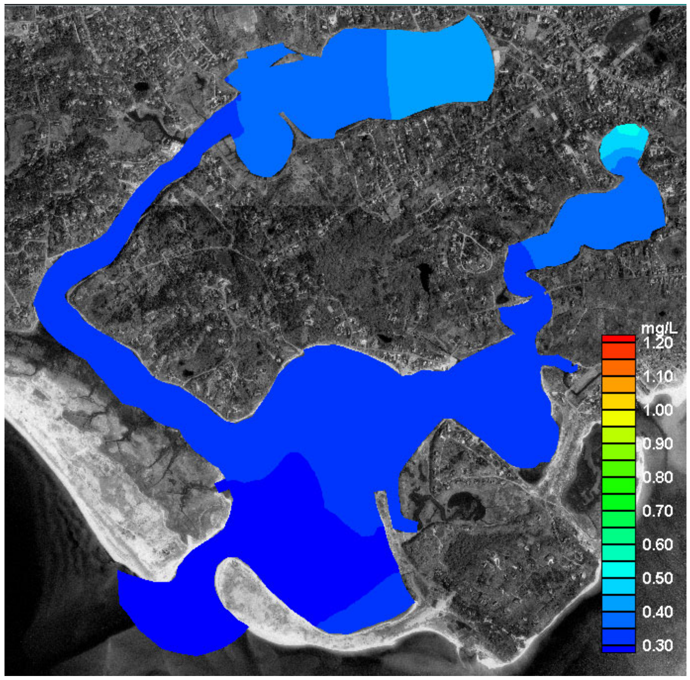


Figure VIII-1. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Stage Harbor system, for threshold loading conditions (0.38 mg/L in both Mill Pond and Oyster Pond).

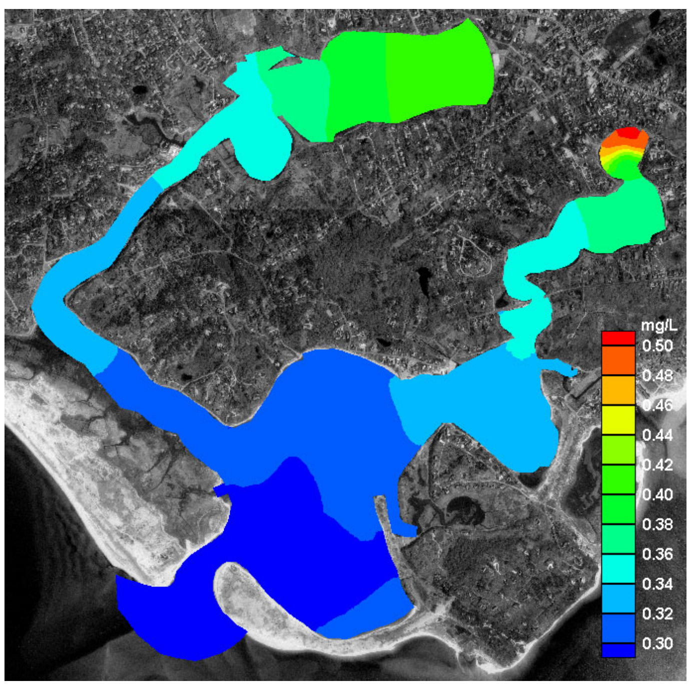


Figure VIII-2. Same results as for Figure VII-25, but shown with finer contour increments for emphasis. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Stage Harbor system, for threshold loading conditions (0.38 mg/L in both Mill Pond and Oyster Pond).

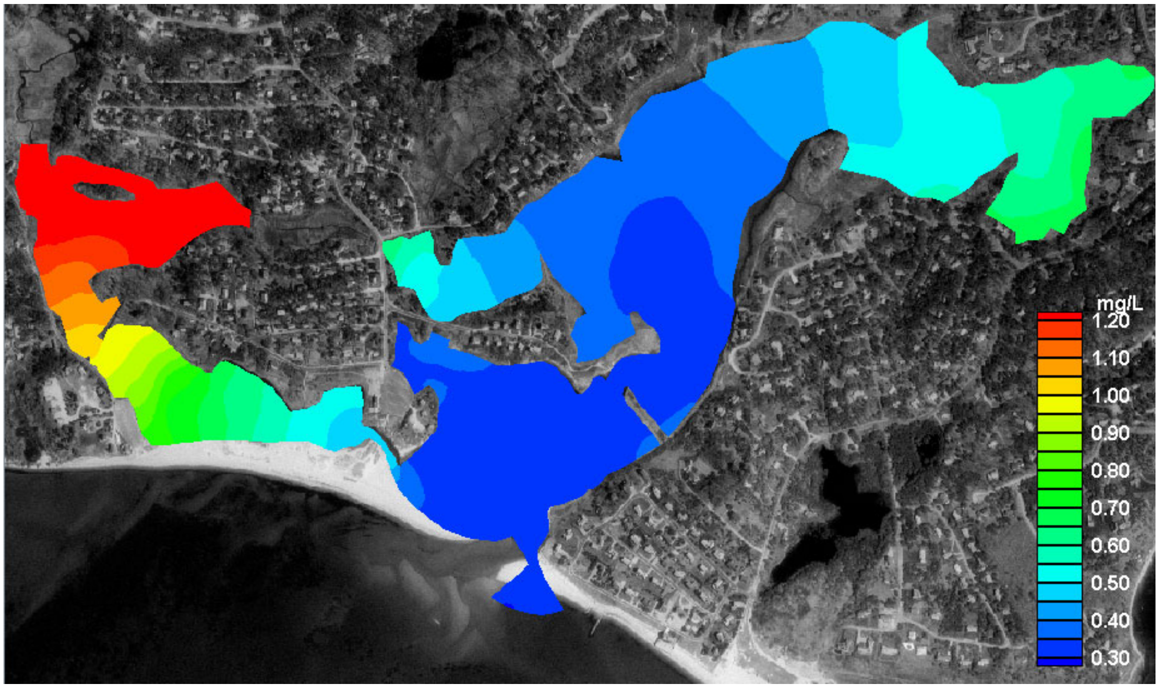


Figure VIII-3. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Sulphur Springs/Cockle Cove Creek system, for threshold loading conditions (0.38 mg/L in Sulphur Springs).

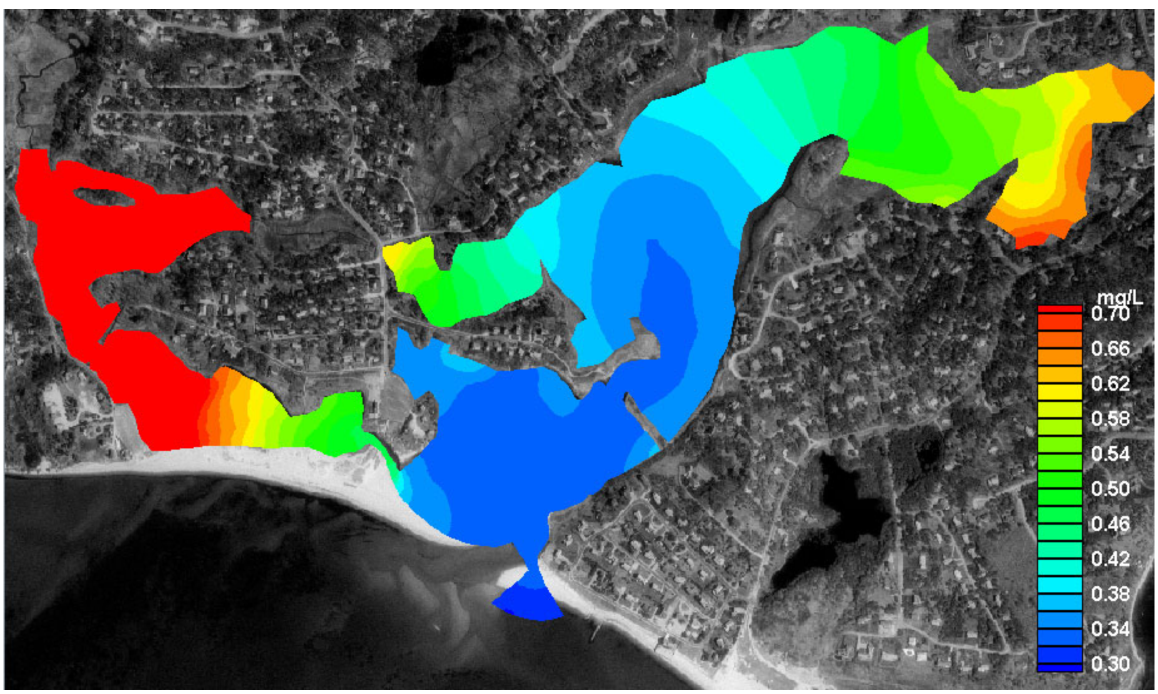


Figure VIII-4. Same results as for Figure VII-29, but shown with finer contour increments for emphasis. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Sulphur Springs/Cockle Cove Creek system, for threshold loading conditions (0.38 mg/L in Sulphur Springs).

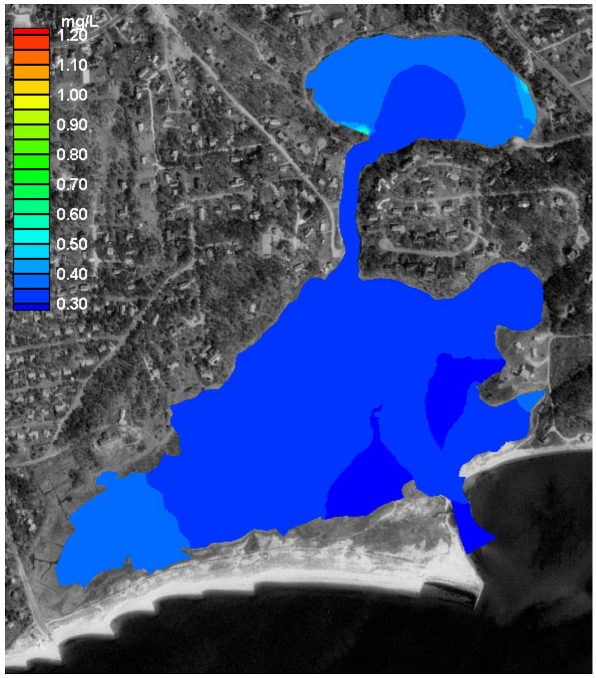


Figure VIII-5. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Taylors Pond/Mill Creek system, for threshold loading conditions (0.38 mg/L in Taylors Pond).

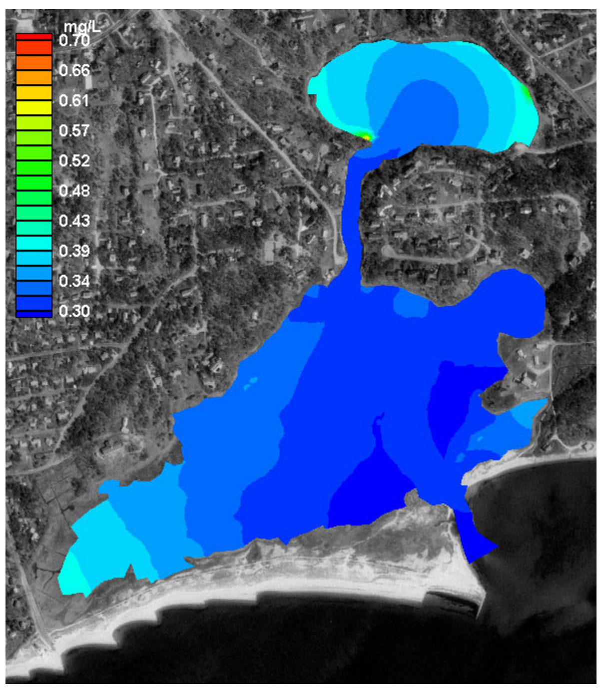


Figure VIII-6. Same results as for Figure VII-34, but shown with finer contour increments for emphasis. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Taylors Pond/Mill Creek system, for threshold loading conditions (0.38 mg/L in Taylors Pond).

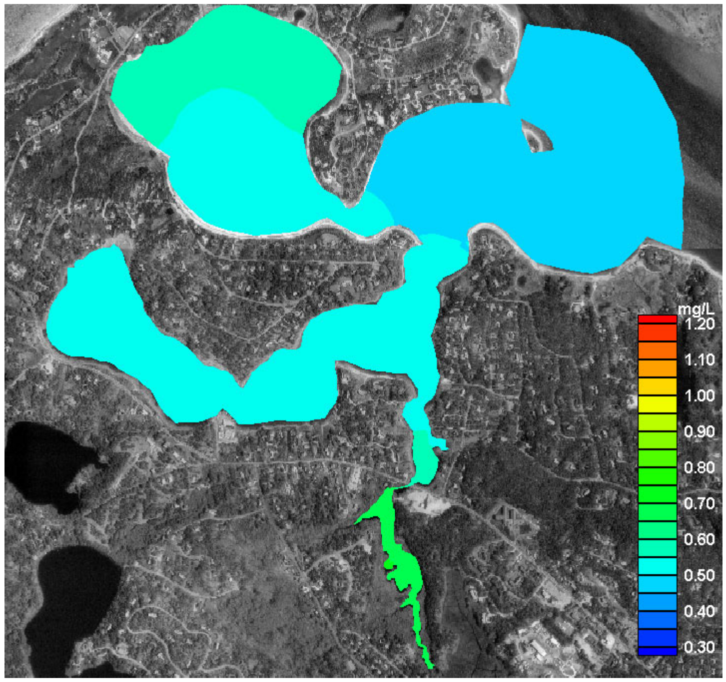


Figure VIII-7. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Bassing Harbor system, for threshold loading conditions (0.55 mg/L in Ryder Cove), and present background N concentration at the entrance to Pleasant Bay (0.48 mg/L).

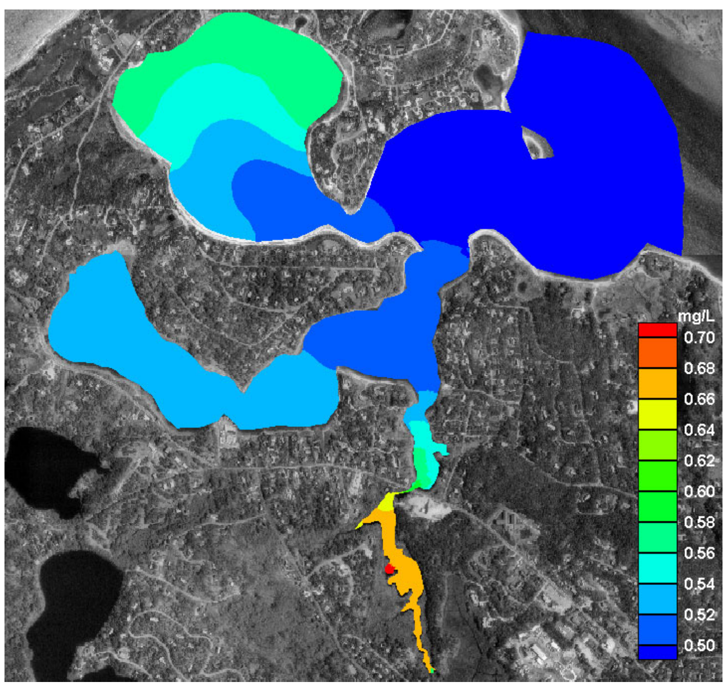


Figure VIII-8. Same results as for Figure VII-15, but shown with finer contour increments for emphasis. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Bassing Harbor system, for threshold loading conditions (0.55 mg/L in Ryder Cove), and present background N concentration at the entrance to Pleasant Bay (0.48 mg/L).

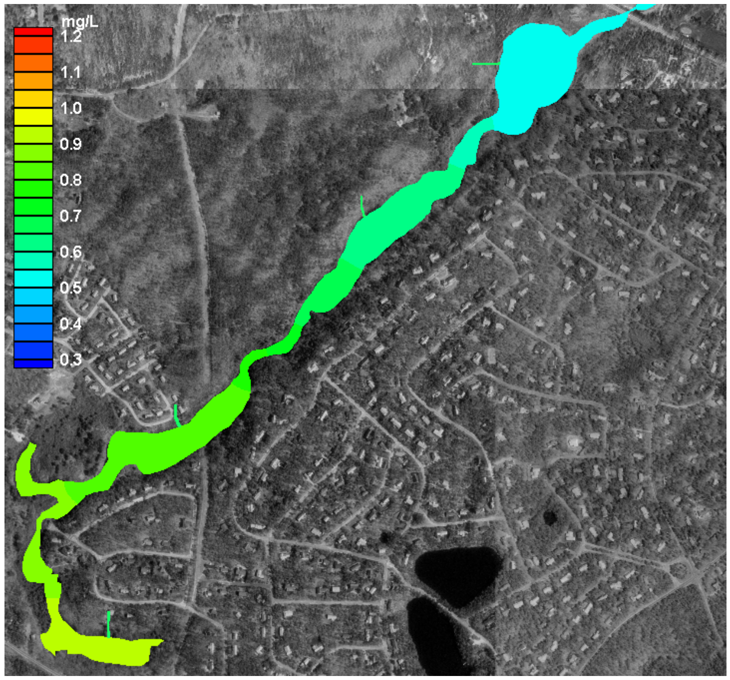


Figure VIII-9. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Muddy Creek system, for threshold loading conditions (0.55 mg/L in lower Muddy Creek), and present background N concentration at the entrance to Pleasant Bay (0.50 mg/L).

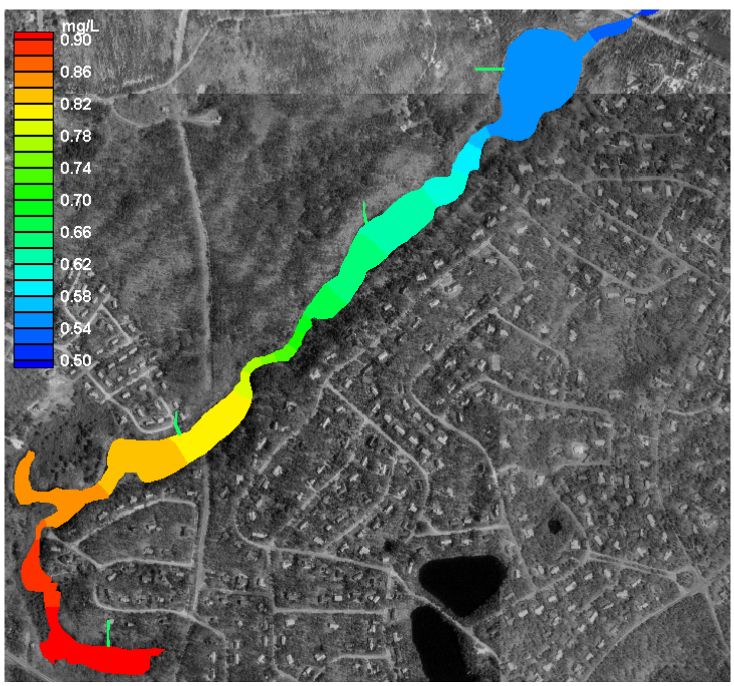


Figure VIII-10. Same results as for Figure VII-20, but shown with finer contour increments for emphasis. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Muddy Creek system, for threshold loading conditions (0.55 mg/L in lower Muddy Creek), and present background N concentration at the entrance to Pleasant Bay (0.50 mg/L).

IX. ALTERNATIVES TO IMPROVE TIDAL FLUSHING AND WATER QUALITY

The two sub-embayments linked to the Pleasant Bay estuary by culverts (Muddy Creek and Frost Fish Creek) exhibit relatively poor tidal flushing. Based on the previous hydrodynamic modeling (Kelley *et al*., 2001), it was anticipated that water quality improvements to these systems likely can be achieved through either resizing of culverts or turning upper portions of the coastal embayments into freshwater ponds. Evaluation of potential alternatives is critical to achieve water quality goals, as well as to avoid adverse environmental impacts. The hydrodynamic models utilized to evaluate tidal flushing provide the basis for quantitatively analyzing the effects of various alternatives on tidal exchange. Using the calibrated models for each system, the model grids were modified to reflect alterations in culvert dimensions and/or bathymetry. Once the hydrodynamic simulations were completed, total nitrogen modeling of each scenario was performed to indicate changes in water column nitrogen concentrations.

The following section describes results of the water quality (nitrogen) analysis performed for the Muddy Creek system, and discusses the implications for each alternative for possible improvements to water quality. This alternatives analysis utilized watershed nitrogen loading and benthic flux loads based on values presented previously. In general, offshore nitrogen concentrations in Pleasant Bay of 0.50 mg/L were used for all alternatives modeling of this analysis; however, an evaluation of an alternative boundary condition was evaluated assuming that potential long-term nitrogen load reductions could lower the total nitrogen concentration in Pleasant Bay (the receiving waters).

The alternatives discussed in this Section do not represent recommendations of the Massachusetts DEP or the MEP. They merely represent how the water quality modeling tool can be utilized to assess potential management alternatives. Prior to implementation of any alternative that alters the system hydrodynamics, a complete environmental assessment of potential adverse impacts will be required.

IX.1 MUDDY CREEK HYDRODYNAMIC ALTERNATIVES

The two culverts running under Route 28 at Muddy Creek each have a height of approximately 2.6 feet and a width of 3.7 feet. Since the surface area of Muddy Creek is relatively large, these culverts are not of sufficient size to allow complete tidal exchange between Pleasant Bay into Muddy Creek. This poor tidal exchange contributes to the water quality concerns for the Muddy Creek system, together with the very high watershed nutrient loading to the Creek (>10,000 Kg/yr). In addition, replacement of these culverts will likely be an expensive alternative due to the large roadway embankment overlying the flow control structures.

Due to the elevation of Route 28 in this region, the roadway embankment prevents storm surge from overtopping the road and “shocking” the ecosystem in Muddy Creek with a pulse of higher salinity Pleasant Bay water. Therefore, turning Muddy Creek into a completely freshwater system is a viable alternative. Other alternatives considered include turning a portion of the system to freshwater and enlarging the culverts to improve tidal exchange.

IX.1.1 Alternative 1 – Muddy Creek as a Freshwater System

Gates could be installed on the Pleasant Bay end of the existing culverts to convert the estuarine system to completely freshwater. As mentioned above, the Route 28 embankment prevents floodwaters from overtopping the road; therefore, the freshwater ecosystem would remain stable during severe conditions. The gates only would allow unidirectional flow from Muddy Creek into Pleasant Bay. Periodic maintenance of the culvert gates would be required, due to their open exposure within Pleasant Bay. A potential environmental drawback to this alternative is the loss of salt marsh that exists within approximately the northern third of the estuary. In addition, benthic analysis indicated that the region immediately upstream of the culverts contains softshell clam resources. Due to potential damage to benthic and wetland resources, it is anticipated that this alternative is not a viable option.

IX.1.2 Alternative 2 – Muddy Creek as a Partial Freshwater System

To preserve the salt marsh and softshell clam resources in the lower portion of Muddy Creek and improve tidal flushing characteristics without altering the culvert configuration, a dike could be placed approximately ½ mile upstream from the roadway embankment (see Figure IX-1). The region upstream of the dike would be maintained as a freshwater pond, again with a gate that only allowed unidirectional flow from the upper portion of Muddy Creek to the lower estuarine portion. Since the poor tidal exchange through the existing culverts is caused by the small cross-sectional area of the culverts relative to the surface area of Muddy Creek estuary, reducing the estuarine surface area will improve flushing characteristics. For example, hydrodynamic model simulations of dike placement as shown in Figure IX-1, reduces the mean-tide estuarine volume by 55%; however, it causes very little reduction in tidal prism (Kelley *et al.*, 2001).

Total nitrogen modeling of the split system required assumptions regarding potential attenuation of nitrogen within the upstream freshwater section. Due to the relatively short retention time of water (~11 days) in this upper portion, resulting from the large volume of groundwater flow entering this portion of the system, it was anticipated that a moderate attenuation of nitrogen would occur in the freshwater portion. Using modest estimates of a 40% reduction in the watershed and sediment loading in the freshwater portion, the modeled reduction in nitrogen concentration for both the existing and Alternative 2 conditions is shown Figures IX-2 and IX-3, respectively. Based on these results, a significant reduction in total nitrogen would occur in the lower portion of Muddy Creek as a result of this alternative.

As described in Kelley *et al*. (2001), design considerations for the dike should include sufficient elevation to minimize potential overtopping during storm conditions. In addition, the freshwater pond level should be set at least 1 ft above the anticipated mean tide level in the estuarine section (about 3.5 feet NGVD according to the hydrodynamic modeling) to ensure flow exits the freshwater section during all phases of the tide. A simple adjustable weir could be designed to fine-tune the water elevation in the freshwater section.

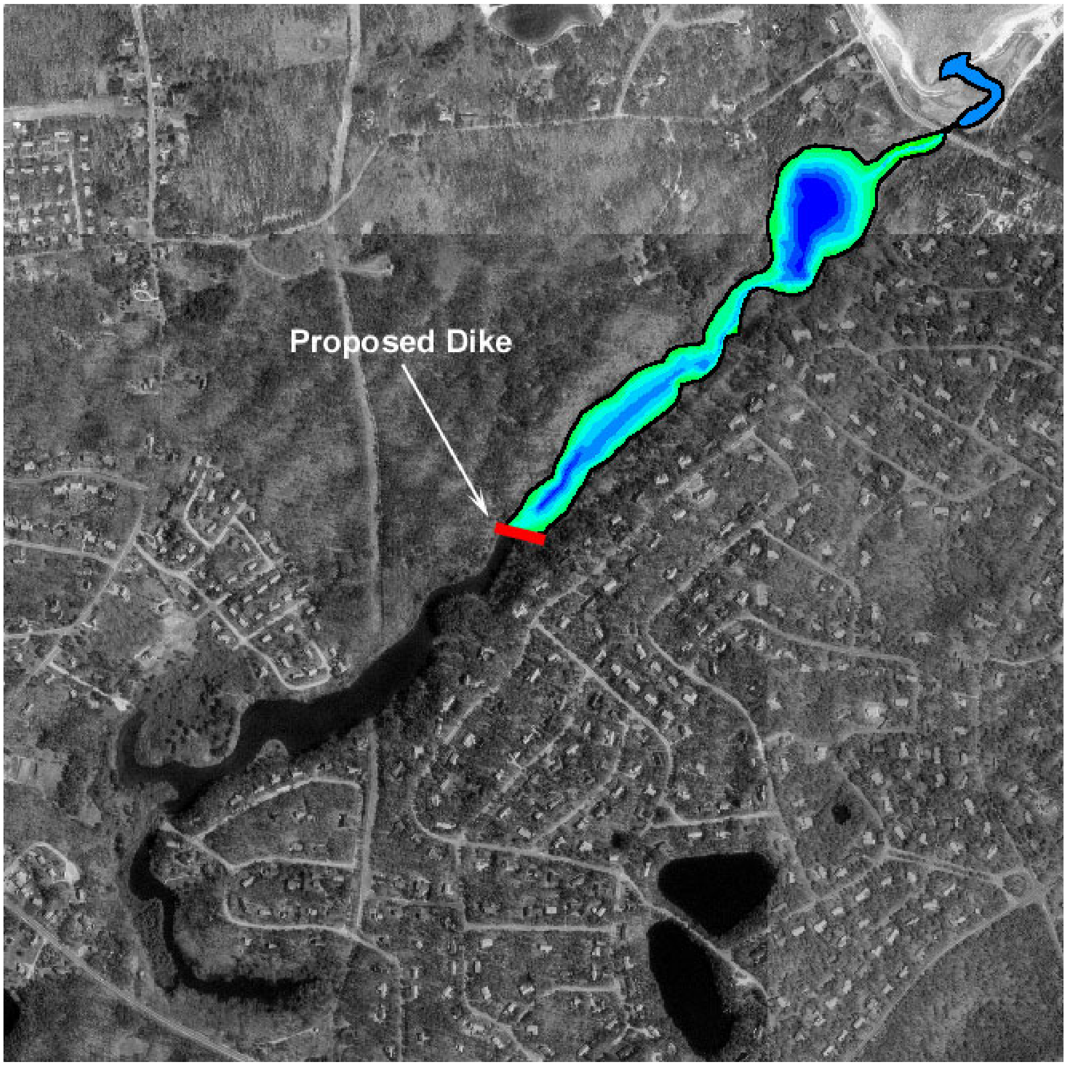


Figure IX-1. Muddy Creek Alternative 2 illustrating the approximate position of the dike separating the freshwater and brackish regions.

IX.1.3 Alternatives 3 and 4 – Increase Size of Route 28 Culverts

Although the Muddy Creek culverts are in good structural shape, it is possible that the Massachusetts Highway Department would consider culvert upgrading as part of the planned Route 28 improvements, if it clearly can be demonstrated that larger culverts are necessary to improve water quality. To assess tidal flushing improvements associated with larger culverts, two alternative culvert sizes were considered: a width of 8 feet and a width of 16 feet. Unlike the existing culverts, the culverts would be designed with a height similar to the tide range in Pleasant Bay (approximately 4.5 feet) to prevent the additional frictional drag associated with totally submerged culverts.

Based on the hydrodynamic modeling results (Kelley *et al.*, 2001), the residence time for Alternative 3 is similar to existing conditions, since the tidal prism increases by only about 20% and the mean-tide volume remains similar. Therefore, the decrease in residence time resulting from this Alternative was about 17%. The larger culvert alternative (Alternative 4) provided a significantly larger tide range, but a similar residence time to Alternative 2.

Although both alternatives 3 and 4 provide significantly better water exchange between Pleasant Bay and Muddy Creek, improvements to average total nitrogen concentrations resulting from the larger culverts are negligible. Figures IX-4, IX-5, and IX-6 illustrate the relative changes in average nitrogen concentrations for existing, 8 ft wide culvert, and 16 ft wide culvert, respectively. Due to a net decrease in the mean volume of Muddy Creek resulting from better flushing characteristics, the nitrogen load potentially could become more concentrated in much of the embayment. A balance between improved flushing and decreased sub-embayment volume governs the mean total nitrogen concentrations. As a result, for both 3 and 4, N concentrations in the lower pond do not change from present conditions. Only for alternative 4 is there a change in the N concentrations of upper portion of the pond of approximately 0.1 mg/L. Therefore, total nitrogen modeling shows that the culvert alternatives as configured will not significantly improve water quality, even though flushing in the upper portion of the creek is improved, hence these alternatives should not be considered further in the future.

IX.2 MUDDY CREEK NITROGEN LOADING ALTERNATIVES

Due to the hydrodynamic simplicity of Muddy Creek, this system allowed rapid analysis of several nitrogen loading alternatives. The sensitivity of the model results to a range of different nitrogen loading scenarios, as well as alternate boundary conditions, were evaluated in the context of the water quality model. Similar to all previous modeling scenarios described, benthic flux was dependent on the overall sub-embayment nitrogen load, where a linear relationship exists between the nitrogen load derived from external sources and the benthic regeneration.

Including the three original modeling scenarios (existing conditions, build out, and no anthropogenic load), a total of 14 water quality modeling scenarios were evaluated. A summary of the nitrogen loading and water quality modeling results from these scenarios is shown in Tables IX-1 and IX-2. Based on the results of this analysis, several alternatives show promise with regards to nitrogen load reduction including Alternative E (3,000 kg/year reduction in upper watershed) and Alternative N (50% reduction in watershed load). Figures IX-7 through IX-10 illustrate the results of selected alternatives for the Muddy Creek system. Based on the results of the modeling, both reducing the load to the upper watershed and bifurcating the estuarine system (making the upper portion freshwater) will improve the overall water quality.

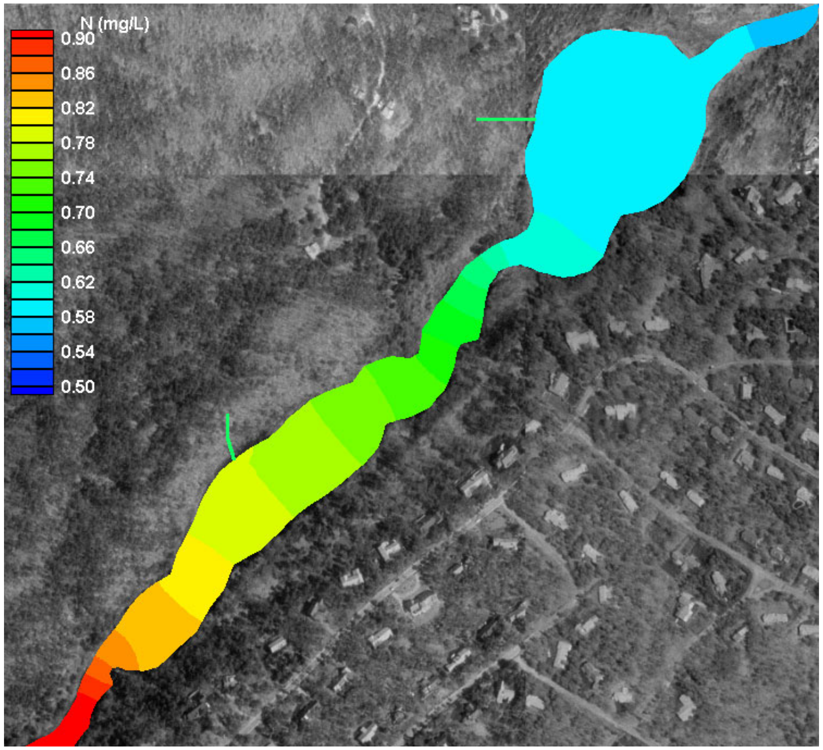


Figure IX-2. Close up of the lower portion of Muddy Creek showing total nitrogen concentration contours for modeled present conditions.

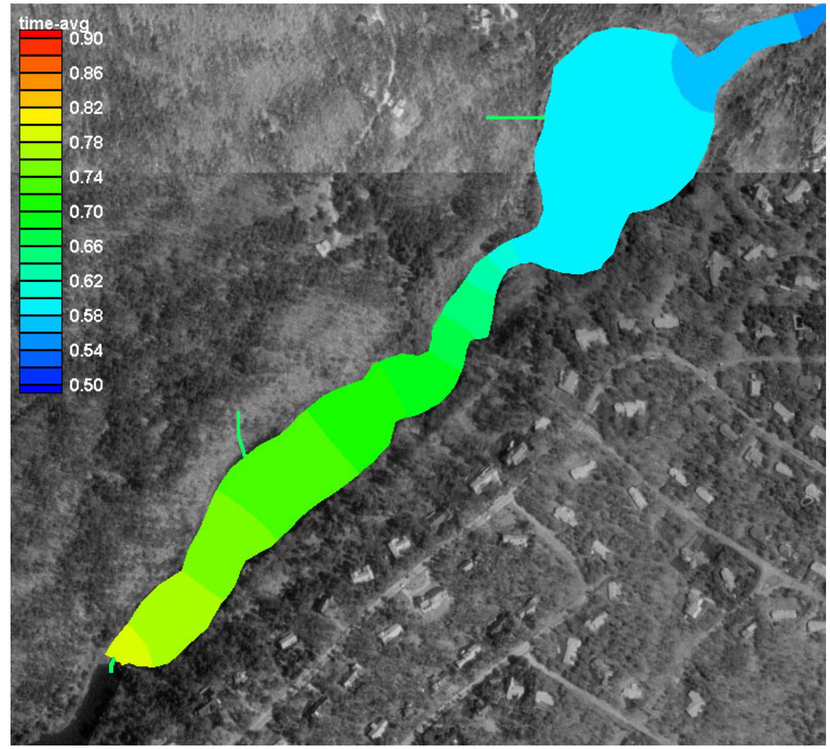


Figure IX-3. Muddy Creek total nitrogen concentration contours for Flushing Alternative 2, where the upper portion of the creek is turned into a freshwater system by the construction of a dike approximately ½ mile upstream of the route 28 roadway embankment.

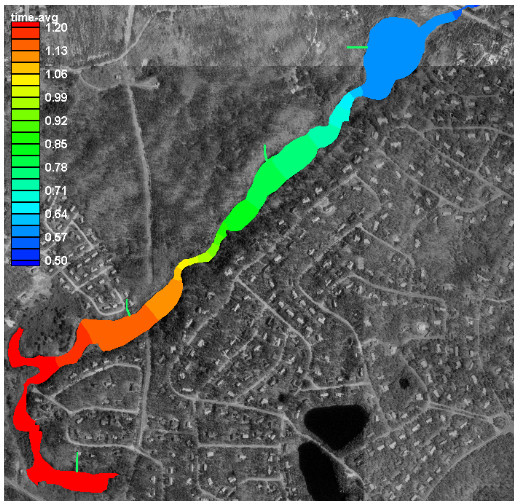


Figure IX-4. Contour plot of modeled present conditions for Muddy Creek, showing total nitrogen concentrations.

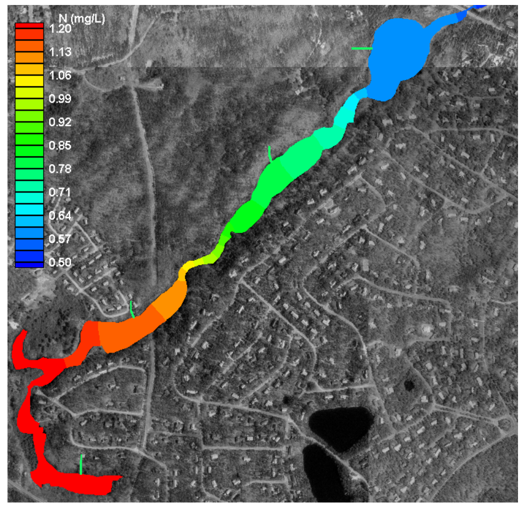


Figure IX-5. Contour plot of total nitrogen concentrations for Muddy Creek flushing Alternative 3, an 8 ft wide box culvert replacement for the existing Route 28 culverts.

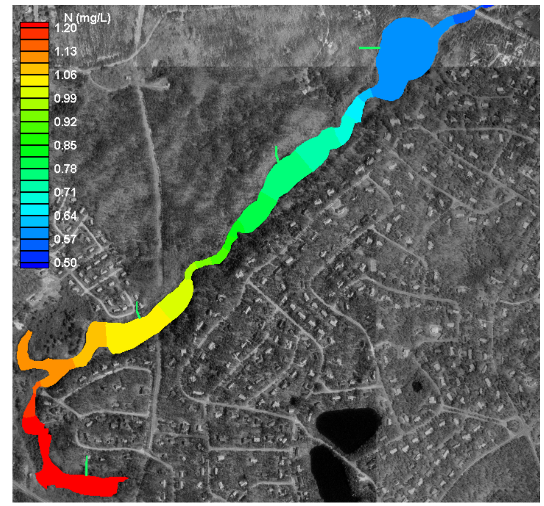


Figure IX-6. Contour plot of total nitrogen concentrations for Muddy Creek flushing Alternative 4, a 16 ft wide box culvert replacement for the existing Route 28 culverts.

Table IX-1. Alternative water quality scenarios run for the Muddy Creek system, including scenarios which modify the hydrodynamics of the system (g, h, I) for present loading conditions, and others that demonstrate the relative impact of load reductions in different areas of the system (i.e., lower creek vs. upper creek, as in e and f).

sub-embayment	watershed load (kg/day)	atmos. deposition (kg/day)	benthic flux (kg/day)	sub-embayment	watershed load (kg/day)	atmos. deposition (kg/day)	benthic flux (kg/day)
a) Present				h) Alt3			
MC-lower	13.36	0.21	-1.88	MC-lower	13.36	0.21	-1.88
MC-upper	19.05	0.20	4.69	MC-upper	19.05	0.20	4.69
b) Build out				i) Alt4			
MC-lower	14.24	0.21	-2.14	MC-lower	13.36	0.21	-1.88
MC-upper	22.69	0.20	5.34	MC-upper	19.05	0.20	4.69
c) No Anthropogenic Load				j) Alt2: no anthropogenic load			
MC-lower	0.50	0.21	-0.10	MC-lower	0.50	0.21	-0.08
MC-upper	0.87	0.20	0.25	MC-upper	0.44	0.20	0.09
d) Present Loading -alternate boundary condition (0.4 mg/L)				k) Alt2: no load w/alt boundary condition (0.40 mg/L)			
MC-lower	13.36	0.21	-1.88	MC-lower	0.50	0.21	-0.08
MC-upper	19.05	0.20	4.69	MC-upper	0.44	0.20	0.09
e) 3000 kg/yr reduction in upper watershed				l) Alt2: 3000 kg/yr reduction in upper watershed			
MC-lower	13.36	0.21	-1.41	MC-lower	13.36	0.21	-1.16
MC-upper	10.83	0.20	3.52	MC-upper	6.42	0.20	0.97
f) 3000 kg/yr reduction in lower watershed				m) Alt2: 3000 kg/yr reduction in lower watershed			
MC-lower	5.15	0.21	-1.41	MC-lower	5.15	0.21	-0.97
MC-upper	19.05	0.20	3.52	MC-upper	11.35	0.20	2.82
g) Alt2: (40% attenuation of upper ws)				n) 0.55 mg/L threshold with 0.50 mg/L BC 50% ws load reduction)			
MC-lower	13.36	0.21	-1.44	MC-lower	6.58	0.21	-0.94
MC-upper	11.35	0.20	2.82	MC-upper	9.43	0.20	2.35

Table IX-2. Comparison of model average total N concentrations from present loading and build out scenario, with percent change, for Muddy Creek water quality alternative scenarios shown in Table IX-1.

sub-embayment	present N conc. (mg/l)	alternative N conc. (mg/l)	percent change (mg/l)	sub-embayment	present N conc. (mg/l)	alternative N conc. (mg/l)	percent change (mg/l)
b) Build out				h) Alt3			
MC-lower	0.60	0.61	2.4%	MC-lower	0.60	0.60	-0.1%
MC-upper	1.21	1.32	9.9%	MC-upper	1.21	1.21	0.3%
c) No Anthropogenic Load				i) Alt4			
MC-lower	0.60	0.50	-16.2%	MC-lower	0.60	0.59	-1.4%
MC-upper	1.21	0.53	-55.7%	MC-upper	1.21	1.11	-8.0%
d) Present Loading -alternate boundary condition (0.4 mg/L)				j) Alt2: no anthropogenic load			
MC-lower	0.60	0.50	-15.9%	MC-lower	0.60	0.50	-16.4%
MC-upper	1.21	1.11	-7.9%	MC-upper	1.21	-	-
e) 3000 kg/yr reduction in upper watershed				k) Alt2: no load w/alt boundary condition (0.40 mg/L)			
MC-lower	0.60	0.55	-7.4%	MC-lower	0.60	0.40	-32.3%
MC-upper	1.21	0.67	-44.0%	MC-upper	1.21	-	-
f) 3000 kg/yr reduction in lower watershed				l) Alt2: 3000 kg/yr reduction in upper watershed			
MC-lower	0.60	0.57	-3.9%	MC-lower	0.60	0.55	-7.2%
MC-upper	1.21	1.13	-6.1%	MC-upper	1.21	-	-
g) Alt2: (40% attenuation of upper ws)				m) Alt2: 3000 kg/yr reduction in lower watershed			
MC-lower	0.60	0.58	-2.5%	MC-lower	0.60	0.56	-5.9%
MC-upper	1.21	-	-	MC-upper	1.21	-	-
				n) 0.55 mg/L threshold with 0.50 mg/L BC 50% ws load reduction)			
				MC-lower	0.60	0.49	-18.4%
				MC-upper	1.21	-	-

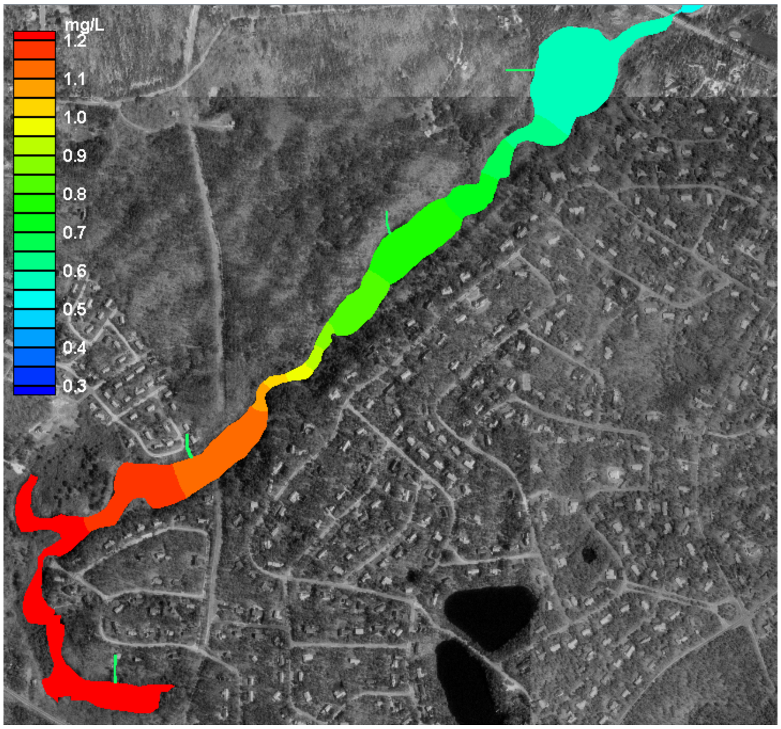


Figure IX-7. Scenario A: Contour plot of modeled total nitrogen concentrations in Muddy Creek, for present loading conditions, and present total nitrogen concentration in Pleasant Bay (0.50 mg/L).

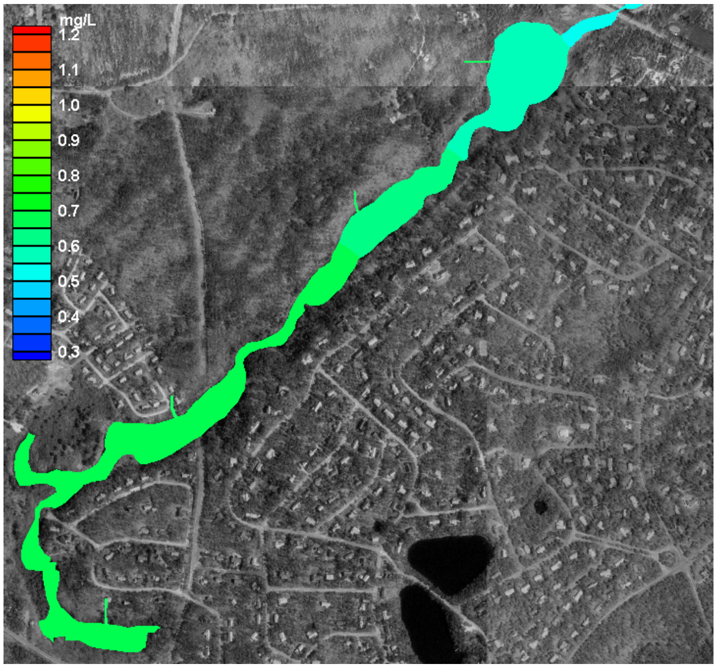


Figure IX-8. Scenario E: Contour plot of modeled total nitrogen concentrations in Muddy Creek, for present loading conditions, with 3000 kg/yr reduction in the load to the upper creek watershed, and present total nitrogen concentration in Pleasant Bay (0.50 mg/L).

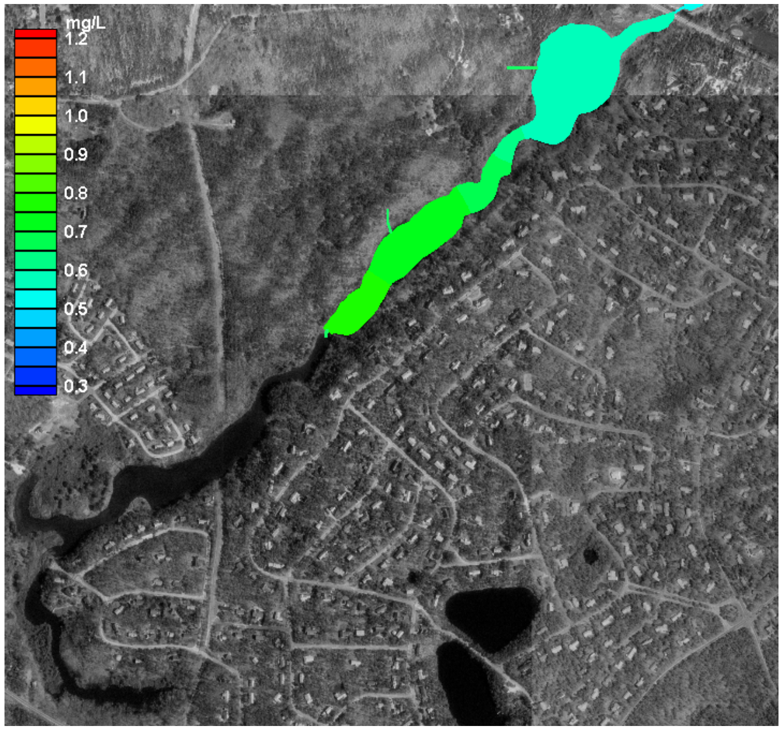


Figure IX-9. Scenario H: Contour plot of modeled total nitrogen concentrations in Muddy Creek, for present loading conditions, and alternate fresh water configuration of the upper creek (alternative 2), with present total nitrogen concentration in Pleasant Bay (0.50 mg/L).

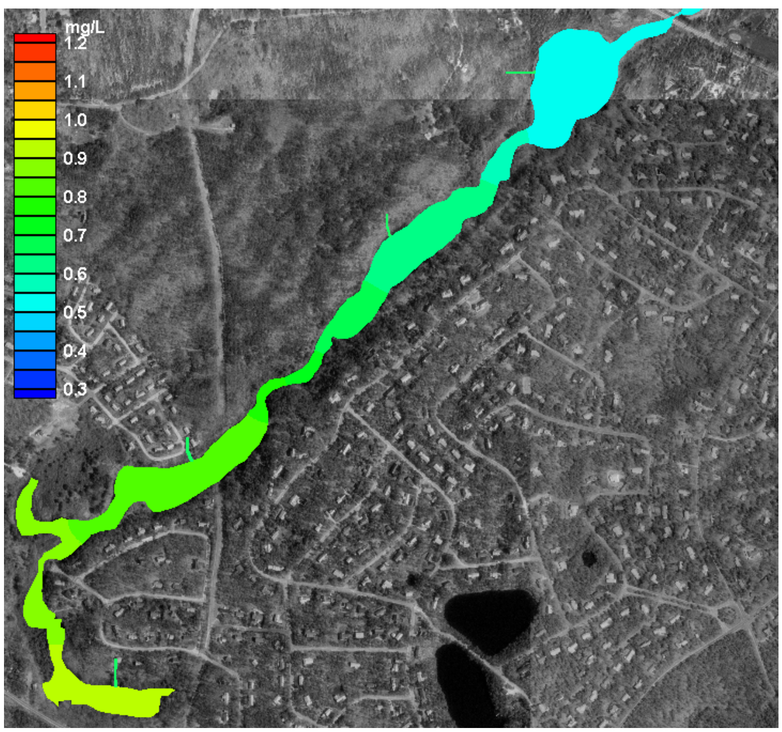


Figure IX-10. Scenario N: Contour Plot of modeled total nitrogen concentrations (mg/L) in the Muddy Creek system, for threshold loading conditions (0.55 mg/L in lower Muddy Creek), and present background N concentration at the entrance to Pleasant Bay (0.50 mg/L). 50% watershed load reduction is required to achieve target N concentration in lower Muddy Creek.

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