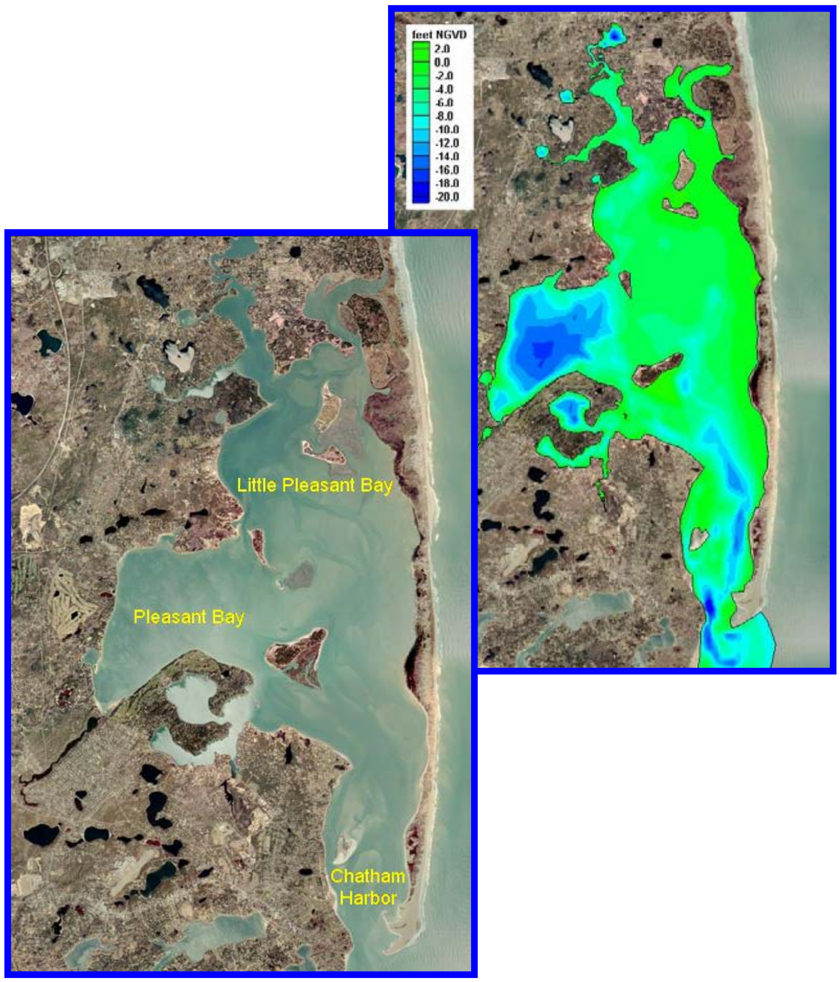
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Pleasant Bay System, Orleans, Chatham, Brewster and Harwich, Massachusetts



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Massachusetts Department of
Environmental Protection

FINAL REPORT –MAY 2006

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I. INTRODUCTION

The Pleasant Bay embayment system is located within the Towns of Chatham, Harwich, Orleans, and Brewster on Cape Cod Massachusetts. The system has an eastern shore bounded by a narrow barrier beach, Nauset Spit, separating the Bay from the Atlantic Ocean, with which it exchanges tidal waters. The Pleasant Bay Estuary is the largest embayment on Cape Cod and is comprised of large open water areas (namely Little Pleasant Bay, Pleasant Bay and Chatham Harbor) as well as small tributary sub-embayments such as Meetinghouse Pond, Areys Pond, Lonnies Pond, Paw Wah Pond, Quanset Pond, Pochet, Round Cove, Muddy Creek, and the moderately sized Bassing Harbor sub-system (e.g. Crows Pond, Ryders Cove, and Bassing Harbor; Figure I-1). The watershed contributing nitrogen to the waters of the Pleasant Bay Estuary is distributed among the Towns of Orleans, Harwich, Brewster and Chatham. Restoration of degraded habitats within the estuary will depend upon the coordinated efforts of these municipalities and their citizens.

The present configuration of the Pleasant Bay embayment system results from a combination of glacially dominated geologic processes including the deposition of glacial outwash deposits and tidal flooding of drowned river valleys formed primarily by post-glacial rivers and enhancements to support human uses (e.g. tidal channel to Lonnies Pond). The major drowned-river valley components are found in The River with its associated tributaries. Pochet in its present configuration appears to be formed as a marsh behind the barrier beach. In the lower basin, Muddy Creek represents the major drowned river valley estuary. Overall, the Pleasant Bay System is a composite or complex estuary comprised of the aforementioned drowned river valley estuaries exchanging tidal waters with a large lagoonal estuary, represented by the large central basins and whose axis runs parallel to the shore line. The lagoon represents more than ¾ of the estuarine area and habitat and includes Little Pleasant Bay, Pleasant Bay and Chatham Harbor. The Pleasant Bay System is a relatively “young” estuary and coastal feature that required significant post glaciation sea-level rise and the formation of the barrier beach, occurring on the order of 2500-4000 years b.p.

Although erosional processes associated with post-glacial streams and rivers were fundamental to the formation of this system, at present streams are relatively small and discharge only a small portion of the aquifer recharge to the estuary. Small freshwater streams discharge to the uppermost reaches of the system such as Meetinghouse Pond, the Namequoit River, Areys Pond, and Lonnies Pond and in the lower Bay to Frost Fish Creek, Muddy Creek, Tar Kiln, and a small herring ladder to Ryders Cove from Stillwater Pond. Most freshwater from the watershed enters the Bay through direct groundwater seepage along the western shore.

As is typical of many other Cape Cod embayments (Nauset System, Popponesset Bay, Three Bays), Pleasant Bay is separated from the Atlantic Ocean by a barrier beach, which is heavily influenced by coastal storms and was recently breached forming the new tidal inlet. 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.

The beach and the inlet are very dynamic geomorphic features, due to the influence of littoral transport processes. The Pleasant Bay embayment system presently exchanges tidal water with the Atlantic Ocean through a single inlet to Chatham Harbor at the southern end of the overall Pleasant Bay system. While the formation of the Pleasant Bay system was dependent upon coastal processes which formed the barrier beach to form the lagoon, the estuary continues to be affected by these same coastal processes as they alter both the length of the spit and the location of the tidal inlet. The effect of these processes is no longer to significantly affect the geomorphology of the estuary and its basins, but to partially control the quality of the habitats within the estuary. Changes in hydrodynamics wrought by inlet dynamics is a key factor in determining the effects on watershed nitrogen loading on estuarine health (see Chapters V & IX). To the extent that the inlet becomes restricted or migrates south and tidal flushing is reduced, nitrogen loading impacts will be magnified over present conditions. Any long term habitat management plan for the Pleasant Bay System must recognize the importance of inlet dynamics and include options to maintain the present (or other suitable) hydrodynamic conditions (see Chapter IX).

Similar to the Nauset and Barnstable Harbor embayment systems, Pleasant Bay is a shallow coastal estuary dominated by salt marsh and tidal flats, as well as being located within a watershed that includes glacial outwash plain (Harwich Outwash Plain) and ice contact deposits (Nauset Height ice-contact deposits) consisting of material deposited after the retreat of the South Channel Lobe of the Laurentide Ice sheet ~15,000 years ago. In fact, Pleasant Bay is situated in the location of 2 sub-lobes of the South Channel lobe, from which these deposits were generated (Oldale, 1992). The material is highly permeable and varies in composition from well sorted medium sands to course pebble sands and gravels. As such, direct rainwater run-off is typically rather low for these coastal systems and therefore, most freshwater inflow to these estuarine systems is via groundwater discharge or groundwater fed surface water flow (e.g. stream to the head of Paw Wah Pond and Lonnies Pond). Pleasant Bay acts as a large mixing zone for terrestrial freshwater inflow and saline tidal flow from the Atlantic Ocean, however, the salinity characteristics of the embayment system varies with the volume of freshwater inflow as well as the effectiveness of tidal exchange with the Atlantic Ocean. Given the large tidal flows and volumetric exchange, there is presently only minor dilution of salinity throughout most of the estuary, with the exception of a few of the tidally restricted sub-embayments (e.g. upper Muddy Creek, upper Frost Fish Creek).

Pleasant Bay, along with its associated terminal sub-embayments, constitutes an important component of the natural and cultural resources of Cape Cod and the Towns of Orleans, Harwich, Chatham and Brewster (though Brewster occupies large parts of the upper watershed to portions of Pleasant Bay, it has relatively limited frontage on the Bay compared to the other Towns). As such the Towns of Orleans, Harwich, and Chatham have worked steadily over many years to have the Pleasant Bay embayment system designated in 1987 as an Area of Critical Environmental Concern (ACEC). In addition, a cooperative agreement was developed between the Towns enabling the development of a resource management plan for Pleasant Bay and in 1998 the Towns formed the Pleasant Bay Alliance to implement the recommendations of the resource management plan.

The primary ecological threat to Pleasant Bay resources is degradation resulting from nutrient enrichment. Loading of the critical eutrophying nutrient, nitrogen, to the embayment waters has been greatly increased over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to this and other outer Cape embayment systems such as Nauset in the Town of Orleans, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater. The Towns of Orleans and Chatham have been among the fastest growing towns in the Commonwealth over the past two decades and do not have centralized wastewater treatment throughout all

Town areas. As existing and probable increasing levels of nutrients impact the coastal embayments of Orleans, Harwich and Chatham, water quality degradation will accelerate, with further harm to invaluable environmental resources.

The large shoreline and numerous terminal sub-embayments greatly increases the potential for direct discharges from homes situated on the shore and decreases the travel time of groundwater from the watershed recharge areas to bay regions of discharge. The nature of enclosed embayments in populous regions brings two opposing elements to bear: as protected marine shoreline they are popular regions for boating, recreation, and land development; as enclosed bodies of water, they may not be readily flushed of the pollutants that they receive due to the proximity and density of development near and along their shores. In particular, the more enclosed basins within the upper reaches of the Bay, as well as terminal sub-embayments such as Quanset Pond, Paw Wah Pond, and Round Cove along the Pleasant Bay shoreline, are at risk of eutrophication from high nitrogen loads entering via direct groundwater seepage in addition to surface water inflows from adjacent sub-watersheds.

Figure I-1. Study region proximal to the Pleasant Bay embayment system for the Massachusetts Estuaries Project nitrogen thresholds analysis. Tidal waters enter the system through one inlet to the Atlantic Ocean. Freshwaters enter from the watershed primarily through 3 surface water discharges to Paw Wah Pond, Lonnies Pond and Tar Kiln Marsh, as well as direct groundwater discharge. The main basins forming most of the estuarine area are Little Pleasant Bay, Pleasant Bay and Chatham Harbor.

As the primary stakeholders to the Pleasant Bay embayment system, the Towns of Orleans, Harwich and Chatham were among the first communities to become concerned over perceived degradation of embayment health. The Town of Orleans (via the Planning Office) and the Town of Chatham (via the Chatham Water Watchers / Water Quality Laboratory) and the Town of Harwich (via the Natural Resources Office) have long recognized the potential threat of nutrient over-enrichment of the Town's coastal embayments. As such, a comprehensive water quality monitoring program was developed as a coordinated effort among the three Towns as well as the Pleasant Bay Resource Management Alliance. Each Town as well as the Pleasant Bay Alliance became responsible for collection of water samples from specific monitoring stations situated throughout Pleasant Bay. These water quality programs coordinated in order to collect consistent comparable data system-wide, essential to the application of the MEP Linked Watershed-Embayment Management Modeling Approach.

The common focus of the water quality monitoring efforts undertaken by the Towns of Orleans, Chatham, Harwich and the Pleasant Bay Alliance has been to gather site-specific data on the current nitrogen related water quality throughout the Pleasant Bay system, such as Meetinghouse Pond, Pochet, Areys Pond, Quanset Pond etc., and determine the relationship between observed water quality and watershed nitrogen loads. This multi-year effort has provided the baseline information required for determining the link between upland loading, tidal flushing, and estuarine water quality. The combined water quality data sets from each of the above mentioned water quality monitoring programs in Pleasant Bay form a baseline from which to gauge long-term changes as watershed nitrogen management moves forward. The quality of these data allowed the MEP to prioritize the Pleasant Bay System for this next step in the Bay's restoration and management.

The MEP effort builds upon the efforts of the water quality monitoring programs, and previous hydrodynamic and water quality analyses, and includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Pleasant Bay embayment system, including all sub-embayments such as Bassing Harbor, Namequoit River and others.

The critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater master planning and nitrogen management alternatives development needed by the Towns of Orleans, Harwich, Chatham and Brewster, for restoration of the impaired habitats within the Pleasant Bay System. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff and volunteers over many years. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns to develop and evaluate the most cost effective nitrogen management alternatives to restore these valuable coastal resources which are currently being degraded by nitrogen overloading.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

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

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

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

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

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

In appropriate estuaries, TMDLs for bacterial contamination will also be conducted in concert with the nutrient effort (particularly if there is a 303d listing). Within the Pleasant Bay System, the MEP has already completed the Technical Reports and MASSDEP the TMDLs related to bacterial contamination in Muddy Creek and Frost Fish Creek sub-embayments. 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 model available to address future regulatory needs.

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

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

The Linked Model has been applied for watershed nitrogen management in ca. 20 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

I.2 SITE DESCRIPTION

The coastal embayment system of Pleasant Bay is the largest estuarine system on Cape Cod and is comprised of approximately 7,000 acres of barrier beaches and islands, salt marsh, tidal flats, as well as both fresh and saltwater ponds. The System contains more than 1000 acres of salt marsh, more than most other estuaries in southeastern Massachusetts. The system is situated on the eastern shore of Cape Cod with the main basins forming a lagoonal estuary oriented in a north – south manner with one large inlet at the southern end nearest Chatham Harbor lighthouse and tributary drowned river valley estuaries entering along the western shore of the lagoon. The inlet provides Atlantic Ocean source water to the overall Pleasant Bay system. The inlet can be significantly affected by longshore sand transport (north to south), where shoaling can impede hydrodynamic exchange at the mouth and, in the case of extreme events, close an existing inlet and open a new one, as was the case in 1987 when the barrier beach was breached and the New inlet opened up. The existing inlet to the Pleasant Bay system is a natural inlet and is not armored in any way. A navigational channel is maintained, however, shoals are abundant in the vicinity of the inlet and depths vary significantly. Depths throughout Pleasant Bay vary due to the tidal salt marsh characteristics of the system in combination with the open water areas that can be as deep as 10 to 14 feet. At low tide large areas of Pleasant Bay are exposed tidal flats with little to no water.

Similar to the Nauset embayment system just to the north of Pleasant Bay, Pleasant Bay exchanges tidal water with the Atlantic Ocean through a single natural inlet crossing the barrier beach that separates this estuarine system from the ocean. The inlet to Pleasant Bay has not been stabilized with riprap and is greatly influenced by shifting sands. The inlet to Pleasant Bay has gone through significant transformations over the last century as a result of intense coastal processes (refer to Chapter V on hydrodynamics). Most recently, Nauset Spit was breached during a northeast storm that occurred on January 2, 1987 thus forming what is commonly referred to as New Inlet. The tidal exchange of waters from Pleasant Bay with the Atlantic Ocean water is driven by a moderate tidal difference between the estuary and the ocean of approximately 5 ft (Chapter V).

For the MEP analysis, the Pleasant Bay system was analyzed in totality with all the associated sub-embayments contributing to the estuarine dynamics of the overall system. This required the integration of previous MEP modeling efforts undertaken for the Town of Chatham specific to the Muddy Creek system, Frost Fish Creek, Ryders Cove, Crows Pond and Bassing Harbor. It was not reasonable to model the Pleasant Bay system without reconsidering the role that the Chatham sub-embayments play relative to the Pleasant Bay nutrient regime and vice versa. The Pleasant Bay estuarine system was partitioned into four general embayment groups: 1) the upper tributary estuaries of The River and Pochet, 2) the coves and drowned kettles along the western shore, 3) the mid and lower tributary estuaries of Bassing Harbor and Muddy Creek, and 4) the main lagoonal basins of Little Pleasant Bay, Pleasant Bay and Chatham Harbor (see Figure I-1). Similar to other embayment systems throughout the MEP study area (e.g. Nauset system, Popponesset Bay, Three Bays) Pleasant Bay is an estuary with focused freshwater input at the headwaters and tidal exchange of marine waters from the Atlantic Ocean (tide range of approximately 5 ft) at its southern inlet. Though the system does receive freshwater discharges to a limited number of terminal sub-embayments, these stream discharges are relatively small and groundwater seepage is the predominant pathway for freshwater recharge from the watershed to enter the estuary. The high rate of tidal exchange and the entry of freshwater all along the western shore (perpendicular to the long axis of the estuary) combine to minimize the salinity gradients in the open basins.

Overall, the Pleasant Bay system is a shallow mesotrophic, moderately nutrient impacted, (with some eutrophic sub-embayments) coastal embayment system on the eastern coast of Cape Cod. The estuary is situated on the southern margin of the Harwich Outwash Plain and Nauset Ice Contact deposits are the primary sediments in the study area. Pleasant Bay is a true composite estuary with a large lagoon formed behind the barrier beach and smaller tributary drowned river valley estuaries entering perpendicular to the lagoon. The System acts as the mixing zone of terrestrial freshwater inflow and saline tidal waters from the Atlantic Ocean. Salinity in the system ranges from approximately 32 ppt at the New inlet to generally not less than 28 ppt in the headwaters of its sub-embayments.

The beach and the inlet are very dynamic geomorphic features, due to the influence of littoral transport processes. The Pleasant Bay embayment system presently exchanges tidal water with the Atlantic Ocean through a single inlet to Chatham Harbor at the southern end of the overall Pleasant Bay system. The tide is the main driver of circulation throughout the Pleasant Bay System and tidal forcing for the system is generated from the Atlantic Ocean. The Atlantic Ocean, adjacent the Nauset Spit barrier beach separating the Pleasant Bay embayment system from the ocean, exhibits a moderate tide range, with a mean range of about 5 ft at the southern inlet of the system. Since the water elevation difference between the Atlantic Ocean and Pleasant Bay is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle (note the tide range off Stage Harbor Chatham is ~4.5 ft, Wellfleet Harbor is ~10 ft).

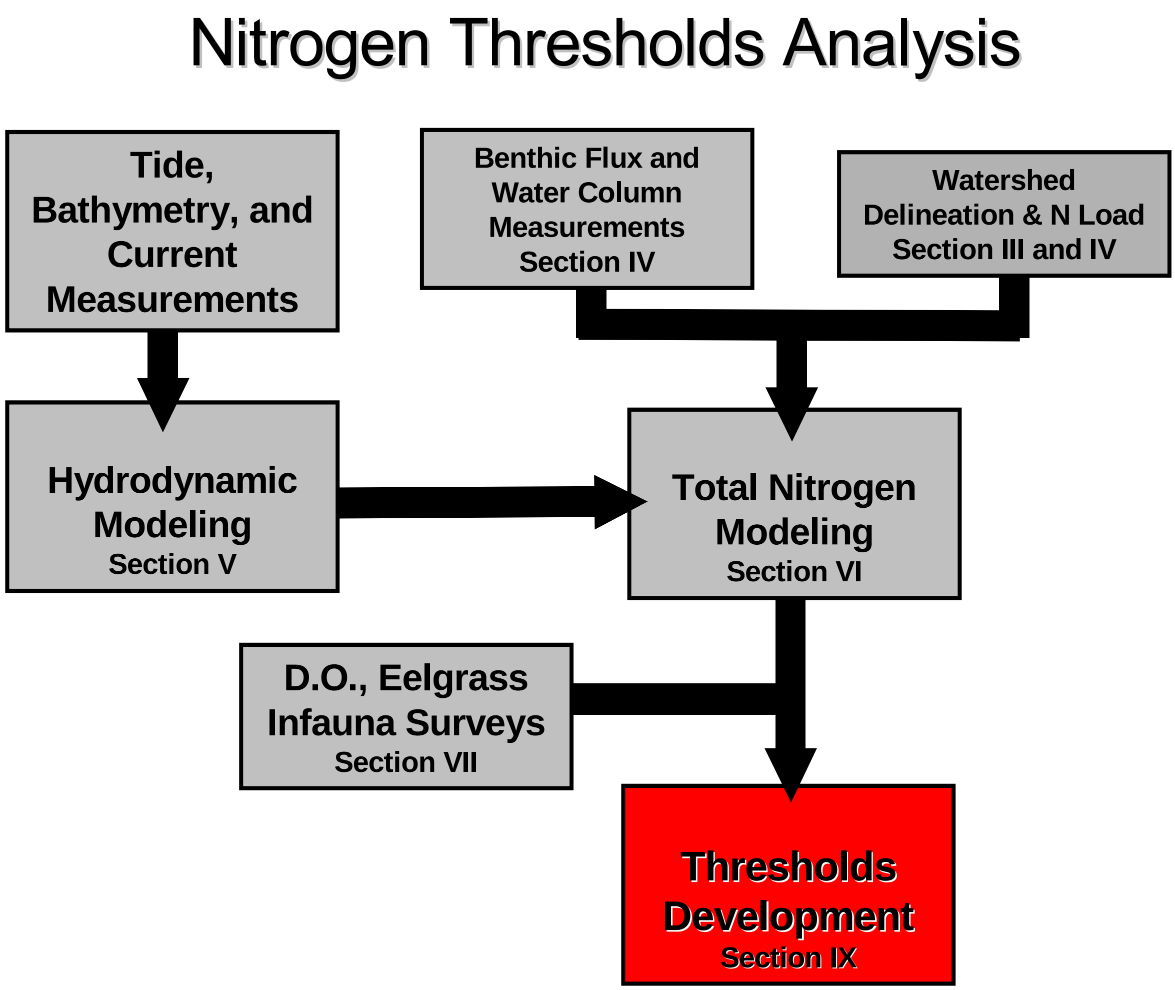


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach

As management alternatives are being developed and evaluated, it is important to note that nitrogen loading and tidal exchange within each sub-embayment is the primary factor controlling habitat health in that sub-basin. The quality of the inflowing waters from Pleasant Bay is the other, although a slightly less critical controlling factor. In addition the nitrogen loading to each sub-embayment affects the health of the receiving main basin of the System. Most of the nitrogen entering the lagoonal component, first passes through a sub-embayment. The result is that the restoration of nitrogen impaired sub-embayments to the Pleasant Bay System require both “local” or contributing area specific nitrogen management, as well as management of nitrogen levels within the watershed of the larger “regional” main basins.

Unlike many smaller estuarine systems, the main nutrient gradients are found in the sub-embayments rather than in the larger lagoon, which accounts for about ¾ of the estuarine areas. For example, there is a large steep gradient in nitrogen from the mouth of Muddy Creek to its headwaters which is many time the gradient found from Chatham Harbor inlet to the upper reach of Little Pleasant Bay. It is in these small tributary estuaries that the greatest nitrogen related impairment of habitat quality is found within the Pleasant Bay System.

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 watershed to the Pleasant Bay embayment system, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer mineral (Weiskel and Howes 1992). Since even Cape Cod "rivers" are primarily groundwater fed, watersheds tend to release little phosphorus to coastal waters. In contrast, nitrogen, primarily as plant available nitrate, is readily transported through oxygenated groundwater systems on Cape Cod (DeSimone and Howes 1998, Weiskel and Howes 1992, Smith *et al.* 1991). The result is that terrestrial inputs to coastal waters tend to be higher in plant available nitrogen than phosphorus (relative to plant growth requirements). However, coastal estuaries tend to have algal growth limited by nitrogen availability, due to their flooding with low nitrogen coastal waters (Ryther and Dunstan 1971). Tidal reaches within Pleasant Bay system follow this general pattern, where the primary nutrient of eutrophication in these systems is nitrogen.

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

Each embayment system maintains a capacity to assimilate watershed nitrogen inputs without degradation. However, as loading increases a point is reached at which the capacity (termed assimilative capacity) is exceeded and nutrient related water quality degradation occurs. 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, the Falmouth Coastal Overlay Bylaw). While each approach may be different, they all focus on changes in nitrogen loading from watershed to embayment, and aim at projecting the level of increase in nitrogen concentration within the receiving waters. Each approach depends upon estimates of circulation within the embayment; however, few directly link the watershed and hydrodynamic models, and virtually none include internal recycling of nitrogen (as was done in the present effort). However, determination of the "allowable N concentration increase" or "threshold nitrogen concentration" used in previous studies had a significant uncertainty due to the need for direct linkage of watershed and embayment models and site-specific data. In the present effort we have integrated site-specific data on nitrogen levels and the gradient in N concentration throughout the Pleasant Bay system monitored by the Chatham, Orleans, and Pleasant Bay Alliance Water Quality Monitoring Programs with site-specific habitat quality data (D.O., eelgrass, phytoplankton blooms, benthic animals) to “tune” general nitrogen thresholds typically used by the Cape Cod Commission, Buzzards Bay Project, and Massachusetts State Regulatory Agencies.

Unfortunately, almost all smaller sub-embayments to Pleasant Bay (Meetinghouse Pond, Lonnies Pond, Areys Pond, The River, Muddy Creek, Round Cove, Quanset Pond, Paw Wah Pond, Ryders Cove in the Bassing Harbor sub-system) are near or beyond their ability to assimilate additional nutrients without impacting ecological health. Nitrogen levels are elevated throughout the upper portions of the system and eelgrass is showing a downward trend. The result is that nitrogen management of the primary sub-embayments tributary to the main basins of Pleasant Bay is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed “eutrophication” and when the nutrient loading is primarily from human activities, “cultural eutrophication”. Although the influence of human-induced changes has increased nitrogen loading to the system and contributed to the degradation in ecological health, it is sometimes possible that eutrophication within the Pleasant Bay system could potentially occur without anthropogenic influence and must be considered in the nutrient threshold analysis. While this finding would not change the need for restoration, it would change the approach and potential targets for management. As part of future restoration efforts, it is important to understand that it may not be possible to turn each embayment into a “pristine” system.

I.4 WATER QUALITY MODELING

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

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

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the Monomoy model for sub-watershed areas designated by MEP. Almost all nitrogen entering the Pleasant Bay system is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Atlantic Ocean source waters and throughout the Pleasant Bay system was taken from the water quality monitoring programs run by the Towns of Orleans and Chatham as well as the Pleasant Bay Alliance (associated with the Coastal Systems Program at SMAST). Measurements of current salinity and nitrogen and salinity distributions throughout estuarine waters of the system were used to calibrate and validate the water quality model (under existing loading conditions).

I.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Pleasant Bay System for the Towns of Chatham, Harwich, Orleans and Brewster. A review of existing studies related to habitat health or nutrient related water quality is provided in Chapter II with a more detailed review of prior hydrodynamic investigations in Chapter V. The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Chapters III and IV. In addition, nitrogen input parameters to the water quality model are described. Since nitrogen recycling associated with the bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Chapter IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Cape Cod Commission data and offshore water column nitrogen values were derived from an analysis of monitoring station data on the flooding tide just inside the inlet to the Pleasant Bay system (Chapter 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 Chapter IV. Results of hydrodynamic modeling of embayment circulation are discussed in Chapter 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 Chapter VI. This analysis includes modeling of current conditions, conditions at watershed build-out, and with removal of anthropogenic nitrogen sources. In addition, an ecological assessment of each embayment was performed that included a review of existing water quality information and the results of a benthic analysis (Chapter VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of each embayment in Chapter VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined threshold for restoration in a given salt pond. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for the Pleasant Bay system. Finally, analyses of the Pleasant Bay system was relative to potential alterations of circulation and flushing, including an analysis to identify hydrodynamic restrictions and an examination of dredging options to improve nitrogen related water quality. In the case of the Pleasant Bay System, this included an evaluation of potential habitat quality shifts that might occur should the present inlet shift causing a lower rate of tidal exchange or a different offshore source water (i.e. Chatham Roads rather than the Atlantic Ocean) The results of the nitrogen modeling for each scenario have been presented (Chapter IX).

II. PREVIOUS NITROGEN MANAGEMENT STUDIES

Nutrient additions to aquatic systems cause shifts in a series of biological processes that can result in impaired nutrient related habitat quality. Effects include: 1) excessive plankton and macrophyte growth, which in turn lead to reduced water clarity, 2) organic matter enrichment of waters and sediments with the concomitant increased rates of oxygen consumption and periodic depletion of dissolved oxygen (especially in bottom waters), and 3) limitation of the growth of desirable species such as eel grass. Even without changes to water clarity and bottom water dissolved oxygen, the increased organic matter deposition to the sediments generally results in a decline in habitat quality for benthic infaunal communities (animals living in the sediments). This habitat change causes a shift in infaunal communities from high diversity deep burrowing forms (which include economically important species), to low diversity shallow dwelling organisms. This shift alone causes significant degradation of the resource and a loss of productivity to both the local shellfishermen and to the sport-fishery and offshore finfishery, all of which are dependent upon these highly productive estuarine systems as a habitat and food resource during migration or different life cycle phases. This process of degradation is generally termed “eutrophication” and in embayment systems, unlike in shallow lakes and ponds, it is not necessarily a part of the natural evolution of a system.

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

These tools for predicting loads and concentrations tend to be generic in nature, and overlook some of the specifics for any given water body. The present Massachusetts Estuaries Project (MEP) study focuses on linking water quality model predictions, based upon watershed nitrogen loading and embayment recycling and system hydrodynamics, to actual measured values for specific nutrient species. The linked watershed-embayment model is built using embayment specific measurements, thus enabling calibration of the prediction process for specific conditions in each of the coastal embayments of southeastern Massachusetts, including the Pleasant Bay System. As the MEP approach requires substantial amounts of site specific data collection, part of the program is to review previous data collection and modeling efforts. These reviews are both for purposes of “data mining” and to gather additional information on an estuary’s habitat quality or unique features.

Numerous studies relating to nitrogen loading, hydrodynamics and habitat health have been conducted within the Pleasant Bay System over the past 10 years. In the late 1990’s local concern over the health of the sub-embayments to Pleasant Bay, particularly in the main tributary embayments of Bassing Harbor and in the upper reaches of The River sub-system, as well as the smaller coves and ponds (Round Cove, Quanset Pond, Paw Wah Pond and Muddy Creek), focused on assessing the water quality related to bacterial contamination and nitrogen inputs. This concern about nutrient related habitat declines resulted in a nitrogen loading and flushing analysis. A detailed watershed loading analysis for the sub-embayment of Round Cove was conducted by the Cape Cod Commission under the Cape Cod Coastal Embayment Project (Eichner et al. 1998). The first hydrodynamic model of Pleasant Bay was conducted at about the same time (Ramsey 1997). Over the past 5-6 years, water quality monitoring programs were established for "local" waters by the Towns of Chatham (Chatham Water Quality Laboratory 2005) and Orleans (Howes and Ramsey 2002, Wineman 1997) as well as on a regional basis through the Pleasant Bay Alliance (PBA 2001). The initial results of the coordinated monitoring efforts indicated that the upper reaches of the Pleasant Bay System and the drowned kettle ponds and small enclosed coves were experiencing habitat degradation as a result of increasing watershed nitrogen inputs. Specifically, the monitoring results indicated that Muddy Creek, Kescayo Gansett Pond (i.e. Lonnies Pond), Meeting House Pond, Paw Wah Pond and Quanset Pond were already showing signs of nutrient related water quality declines, suggesting that they are beyond their critical loading limit (note only Muddy Creek and Arey's Pond were indicated by the land-use studies). While these assessments were based primarily on water quality data, many of the conclusions as to degradation were clearly supported by unequivocal datasets and have been corroborated by the more detailed MEP assessments (Chapter VII). All of these studies provided useful information and quantitative data which have been integrated into the present MEP analysis.

The Cape Cod Commission (CCC) conducted a nitrogen loading study for the Pleasant Bay System to determine the maximum allowable loads that 16 sub-embayments 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 sub-embayments and those delineations enabled the development of 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). 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. A major part of the analysis was to examine the effects of the pre- and post- breach inlet conditions on the flushing times and nitrogen related habitat quality within the Pleasant Bay System.

The resulting nitrogen conditions were compared to critical levels as defined by the Buzzards Bay Project Outstanding Resource Waters (BBP ORW) and Outstanding Resource Waters – Nitrogen (ORW-N) limits. The results indicated 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.

A more recent watershed loading analysis was undertaken using nitrogen coefficients that differed from the original Cape Cod Commission (CCC) study (Carmichael et al. 2004). The study followed the basic Buzzards Bay Project approach, based upon residence times rather than actual circulation and volumetric exchange rates. The watershed analysis used the previous CCC watershed based upon water table elevations, which differs from the groundwater watershed mapped by the USGS for MEP (Chapter 3). The model was not calibrated and the "validation" used only dissolved inorganic nitrogen from 7 sites collected in 2000 and 2001. The model accounted for less than 50% of the observed nitrogen variation. However, the study did confirm the importance of atmospheric deposition versus watershed derived nitrogen to the nitrogen balance of Pleasant Bay.

As a key process controlling the habitat quality within the whole of the Pleasant Bay System, tidal exchange with high quality Atlantic Ocean waters must be considered and accurately quantified. The MEP has re-evaluated the various studies of the migration and breaching of the Nauset Spit as it affects tidal flushing of Pleasant Bay. The results of these previous studies are fully discussed in Chapter V as part of the MEP hydrodynamic evaluation and modeling of the Pleasant Bay System. The potential inlet size and/or migration of the tidal inlet to Pleasant Bay is critical to the flushing, and as such the nitrogen related habitat quality of the Pleasant Bay sub-embayments. Flushing provides the primary mechanism for lowering nitrogen levels within the estuary once nitrogen has entered bay waters. In Chapter IX, the MEP Technical Team has used the calibrated and validated Linked Watershed-Embayment Model to evaluate potential shifts in habitat health as a result of inlet dynamics.

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.

Also over the past decade there were significant efforts at habitat protection/restoration related to Comprehensive Wastewater Management and Planning efforts, particularly within the Town of Chatham. As part of the initial wastewater management planning study a nitrogen loading analysis to Bassing Harbor and Muddy Creek sub-embayments was performed by Stearns & Wheler. This initial wastewater management planning study was part of a needs assessment for the Town of Chatham (Stearns & Wheler, 1999). The study divided Chatham into three groups that were analyzed separately: Pleasant Bay Region, Stage Harbor System, and the South Coast Embayments. The study followed a similar protocol as the earlier studies: 1) use of existing subwatersheds information, 2) calculation of existing and future nitrogen loading to each water body based on land use in respective subwatersheds, 3) calculation of steady-state nitrogen concentration to be expected based on flushing rate estimates, and 4) 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. The analysis of existing loading to the Pleasant Bay systems embayments was integrated into the previous Pleasant Bay study conducted by the CCC. 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.

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) and resulting a multi-year water quality monitoring effort that continues to this day under the direction of the Chatham Water Quality Laboratory. Based upon the initial land-use analysis and the results of the water quality monitoring efforts, additional levels of analysis were undertaken to increase the accuracy of the assessments and predictions. These included embayment specific hydrodynamic modeling, water quality modeling, and habitat assessment (Kelley *et al*., 2001 and Applied Coastal *et al*., 2001). Based on site-specific nutrient analysis for the coastal systems of Chatham and developed to support embayment nutrient threshold development, it appeared that Muddy Creek and the Bassing Harbor systems 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. To address some of the shortcomings inherent in the 2001 study, the Town of Chatham continued its water column nitrogen monitoring program and updated measurements of benthic nitrogen recycling flux within the Bassing Harbor system.

These efforts by the Chatham Wastewater Planning Committee, provided information and data that was seamlessly incorporated into a recent full application of the assessment and modeling effort by the MEP for the Bassing Harbor and Muddy Creek sub-embayments to Pleasant Bay (Howes et al. 2003). These full applications of the MEP approach developed nitrogen thresholds for these component systems, but indicated their dependence on the state of the greater Pleasant Bay system and the need to incorporate the Chatham embayments tributary to Pleasant Bay into a full Pleasant Bay analysis. This linkage has been fully carried out in the present report. The earlier results remain substantively intact, but have been refined by new datasets. The results of the early MEP analysis are discussed in the context of the new and broader analysis of Pleasant Bay in the chapters that follow. The present refinements include integration and updating of all the watershed analysis (and wateruse) system-wide into a consistent database and incorporation of all of the water quality data produced in intervening years from all water quality monitoring efforts (Chatham, Orleans, Pleasant Bay Alliance). This new data creates a sound baseline and significantly increases the certainty of the analysis for the whole of Pleasant Bay.

In addition to the large scale studies investigating the whole of Pleasant Bay or its major sub-embayments as discussed above, there have been other efforts aimed at specific aspects of the nutrient issue as it pertains to Pleasant Bay. As part of the MEP effort the Town of Orleans, through its Wastewater Management Steering Committee, compiled more than 25 studies relating to the marine systems of Orleans. Of these, 5 studies were selected as likely to contribute information or quantitative data to the Linked Watershed-Embayment Management Approach for the Pleasant Bay System. These studies were reviewed by MEP technical experts for (a) information or quantitative data to support the Linked Management Approach, (b) acceptability of results based upon quality assurance or comparability, and (c) data gaps seen in the integrated data set of the existing studies and present Program. The results of the evaluations of these studies are presented in detail in an SMAST Technical Report (Howes and Ramsey 2002) and briefly discussed above. Moreover, as regards fertilizer application rates, the MEP Technical Team working with the Orleans Wastewater Planning Committee, conducted a survey of 340 homes throughout the Town of Orleans. The results of this survey indicated that the number of fertilizations per lawn in Orleans was similar to that in an upper Cape survey involving the Towns of Falmouth, Mashpee and Barnstable, 1.76 versus 1.44. However, within the survey there was a very high number of homes serviced by commercial lawn companies (over 1/3). These lawns were fertilized at a high rate relative to home-owner serviced properties. The overall results indicated a potentially higher nitrogen loading per lawn in Orleans of 1.51 lb/lawn/yr (weighted average). However, this is due to the high fraction of homes with professionally maintained lawns in the survey. Given the large areas of the watershed within Harwich, Brewster and Chatham, the uncertainty in the regional percentage of lawns maintained professionally; and the fact that if the Orleans rates are applied to the entire Pleasant Bay watershed the change in loading is <2%, however, it cautions of the potential for behavioral changes to greatly increase the nitrogen loading from this source.

The marine systems of the Pleasant Bay estuary have been the subject of a variety of studies ranging from investigations of physical processes to watershed nitrogen loading surveys and site specific investigations of nitrogen transformations. In addition, nutrient related water quality monitoring was undertaken by the Towns of Chatham and Orleans as well as the Pleasant Bay Alliance (partnership between the Towns of Chatham, Harwich and Orleans). Given the need for diverse data sets to implement the MEP Linked Watershed-Embayment Nitrogen Management Approach, all relevant sources of information were evaluated for inclusion. The MEP as incorporated all appropriate data from all previous studies to enhance the determination of the nitrogen thresholds for the Pleasant Bay system and to reduce costs to the Towns of Chatham, Harwich, Orleans and Brewster of watershed based nitrogen management.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Massachusetts Estuaries Project team includes technical staff from the United States Geological Survey (USGS). The USGS groundwater modelers were central to the development of the groundwater modeling approach used by the Estuaries Project. The USGS has a long history of developing regional models for the six-groundwater flow cells on Cape Cod. Through the years, advances in computing, lithologic information from well installations, water level monitoring, stream flow measurements, and reconstruction of glacial history have allowed the USGS to update and refine the groundwater models. The MODFLOW and MODPATH models utilized by the USGS to organize and analyze the available data use up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation, surface water/groundwater interaction, groundwater travel time, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including the Pleasant Bay embayment system. The Pleasant Bay System and its watershed are located within the Towns of Orleans, Brewster, Harwich, and Chatham. Pleasant Bay is the largest estuarine system on Cape Cod, situated along its southeastern edge. The Pleasant Bay System currently exchanges tidal water with the Atlantic Ocean, but prior to the formation of the New inlet through Nauset Spit, it exchanged water off south Chatham, i.e. Chatham Roads.

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watershed or contributing area to the Pleasant Bay system under evaluation by the Project Team. Further modeling of the Pleasant Bay system was undertaken to sub-divide the overall watershed into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system, (b) defining contributing areas to major freshwater aquatic systems which generally attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining 10 year time-of-travel distributions within each sub-watershed as a procedural check to gauge the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in the overall Monomoy groundwater flow cell. Model assumptions for calibration were matched to surface water inputs and flows from current (2002 to 2003) stream gauge information. Given the recent alteration of the hydrodynamics of the Pleasant Bay System, resulting from the new inlet formation, the USGS used the present mean tidal levels in Pleasant Bay as the boundary condition in its watershed delineation effort.

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

III.2 MODEL DESCRIPTION

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

The Monomoy Flow Model grid consists of 164 rows, 220 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft. The top of layer 8 resides at National Geodetic Vertical Datum (NGVD) 29 with layers 1-7 stacked above and layers 8-20 below. Layer 18 has a thickness of 40 feet and extends to 140 feet below NGVD 29, while layer 19 extends to 240 feet below NGVD 29. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics (up to 525 feet below NGVD 29). The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, was used to simulate the top of the water table, which varies in elevation depending on the location in the Lens. Since water elevations are less than +40 ft in the portion of the Monomoy Lens in which the Pleasant Bay system resides, the three uppermost layers of the model are inactive.

The glacial sediments that comprise the aquifer of the Monomoy Lens consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward sequence with sand and gravel deposits deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. The Pleasant Bay watershed is located in the Harwich Plains, which were deposited as glacial ice lobes were retreating to positions near the current Cape Cod Bay shoreline and the barrier beach along the eastern edge of Pleasant Bay (Walter and Whealan, 2005). Lithologic data used to determine hydraulic conductivities used in the model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and water-level and streamflow data collected in May 2002.

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

III.3 PLEASANT BAY CONTRIBUTORY AREA

Newly revised watershed and sub-watershed boundaries were determined by the United States Geological Survey (USGS) for the Pleasant Bay estuary system (Figure III-1). Model outputs of MEP watershed boundaries were “smoothed” to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, and (c) to more closely match the sub-embayment segmentation of the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. The MEP sub-watershed delineations also include 10 yr time of travel boundaries. Overall, 95 sub-watershed areas, including 25 freshwater ponds and 7 public water supply wellfields, were delineated within the watershed to the Pleasant Bay estuary system.

Table III-1 provides the daily freshwater discharge volumes for each of the subwatersheds as calculated by the groundwater model and these volumes were used to assist in the salinity calibration of the tidal hydrodynamic models and to determine hydrologic turnover in the lakes/ponds, as well as for comparison to measured surface water discharges. The MEP delineation includes 10 yr time of travel boundaries. The overall estimated freshwater inflow to the estuarine waters of Pleasant Bay from the MEP watershed is approximately 107,000 m3/d.

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

Overall, the MEP contributing area to Pleasant Bay based upon the groundwater modeling effort is very similar to the previous delineation based upon available well data, the MEP area is only 1% or 164 acres larger. However, some of the interior subwatersheds areas are different; for example more refined delineation of the pond subwatersheds in the MEP delineations causes the Arey’s Pond/Namequoit River subwatershed to shift more to the north and reduces the watershed area from 2,737 acres to 921 acres. On the other hand, the subwatershed to Kescayo Gansett Pond only changes by 7 acres or 2%.

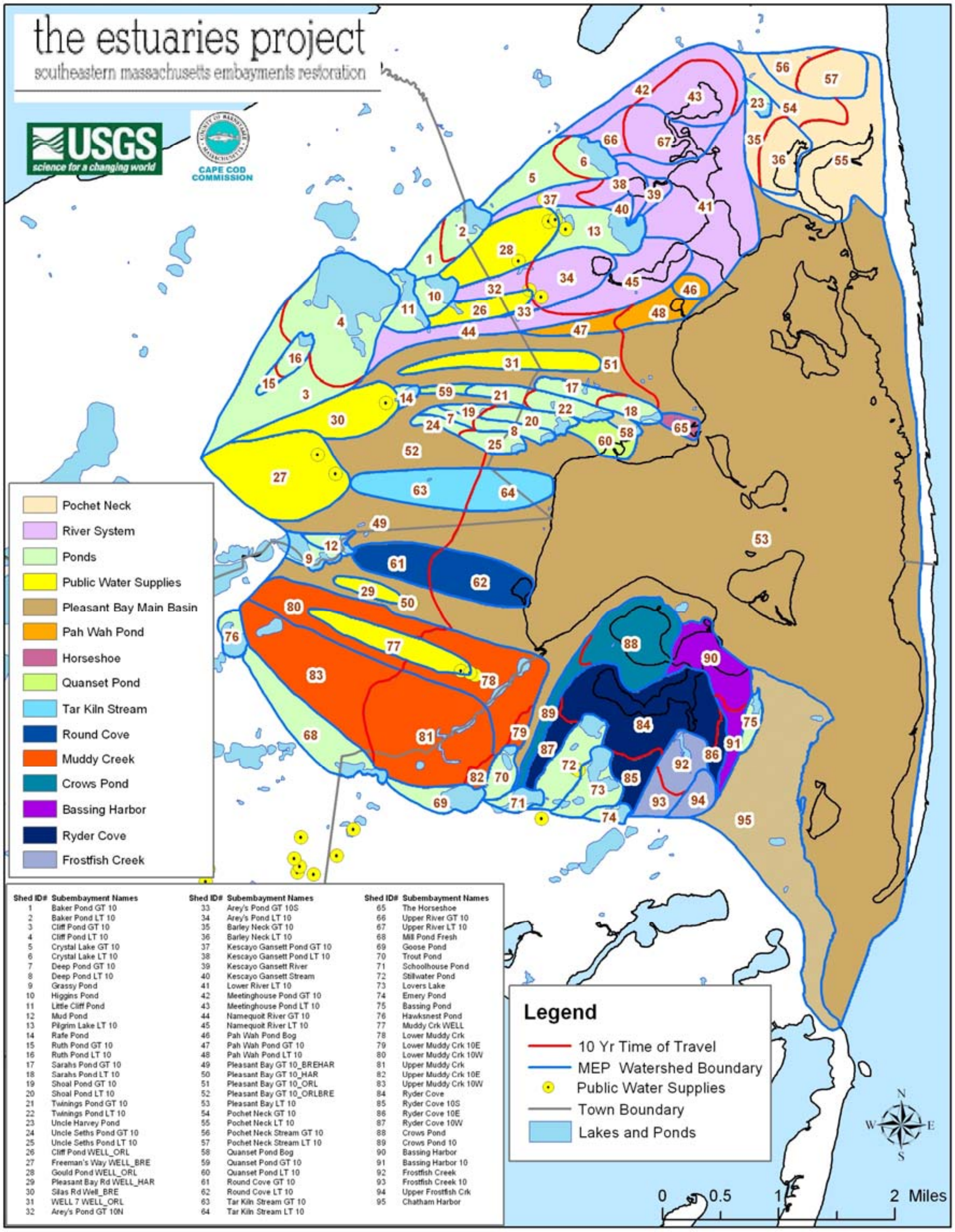


Figure III-1. Watershed delineation for the Pleasant Bay Embayment System. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the watershed names (above). Sub-watersheds to great ponds were developed for determining nitrogen loss in transport. Sub-watersheds to embayments were selected based upon the functional estuarine sub-units in the water quality model (see section VI).

Table III-1. Daily groundwater discharge to each of the sub-embayments in the Pleasant Bay system, as determined from the USGS groundwater model.		
Watershed	Discharge	
	m3/day	ft3/day
Pochet Neck	6,757	238,638
Meetinghouse Pond	2,510	88,641
Upper River	2,980	105,224
Kescayo Gansett Pond	3,235	114,239
Kescayo Gansett Stream	1,066	37,649
Kescayo Gansett River	1,240	43,782
Areys Pond	3,272	115,563
Namequoit River	3,798	134,113
Lower River	4,409	155,718
Pah Wah Pond	1,382	48,808
Quanset Pond	1,531	54,076
Tar Kiln Stream	2,500	88,277
Round Cove	2,503	88,394
The Horseshoe	1,035	36,546
Upper Muddy Creek	8,648	305,416
Lower Muddy Creek	6,626	233,980
Ryders Cove	7,465	263,613
Crows Pond	2,680	94,660
Bassing Harbor	1,859	65,638
Frostfish Creek	1,765	62,342
Pleasant Bay Proper	33,876	1,196,337
Chatham Harbor	6,494	229,343
TOTAL	107,632	3,800,998
NOTE: Discharge rates are based on 27.25 inches per year of recharge (Walter and Whealan, 2005).		

The evolution of the watershed delineations for the Pleasant Bay Estuarine System has allowed increasing accuracy as each new version adds new hydrologic data to that previously collected; the model allows all this data to be organized and to be brought into congruence with data from adjacent watersheds. The evaluation of older data and incorporation of new data during the development of the model is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model used for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are included/excluded within the contributing areas. Small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the down gradient estuary. In addition, most of these “errors” are between adjacent sub-watersheds, within the overall Pleasant Bay System watershed. Therefore they do not result in a difference in total loading to the Pleasant Bay Estuarine System. For example, a shift in watershed load between the Areys Pond watershed and the adjacent Namequoit River sub-watershed has little effect even on the results for Areys Pond and no effect on the results for The River sub-embayment and to greater Pleasant Bay.

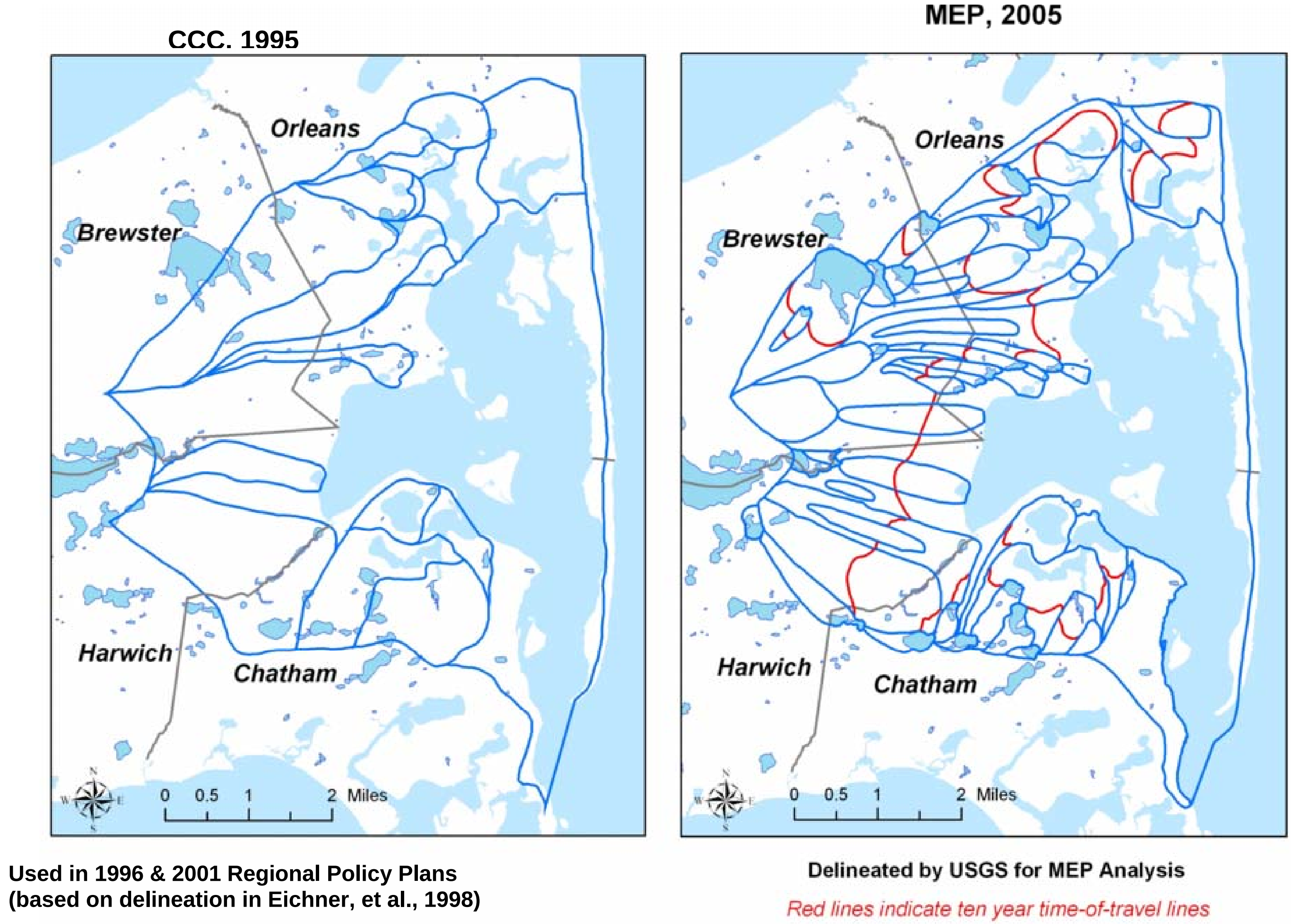


Figure III-2. Comparison of previous CCC (left) and MEP (right) Pleasant Bay watershed and subwatershed delineations. The MEP system watershed area is 1% or 164 acres larger. The MEP sub-watersheds to Bassing Harbor and Muddy Creek are unchanged from the previous MEP analysis for these systems.

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

IV.1 WATERSHED LAND USE BASED NITROGEN LOADING ANALYSIS

Management of nutrient related water quality and habitat health in coastal waters requires determination of the amount of nitrogen transported by freshwaters (surface water flow, groundwater flow) from the surrounding watershed to the receiving embayment of interest. In southeastern Massachusetts, the nutrient of management concern for estuarine systems is nitrogen and this is true for the Pleasant Bay system. Determination of watershed nitrogen inputs to these embayment systems requires the (a) identification and quantification of the nutrient sources and their loading rates to the land or aquifer, (b) confirmation that a groundwater transported load has reached the embayment at the time of analysis, and (c) quantification of nitrogen attenuation that can occur during travel through lakes, ponds, streams and marshes. This latter natural attenuation process results from biological processes that naturally occur within ecosystems. Failure to account for attenuation of nitrogen during transport results in an over-estimate of nitrogen inputs to an estuary and an underestimate of the sensitivity of a system to new inputs (or removals). In addition to the nitrogen transport from land to sea, the amount of direct atmospheric deposition on each embayment surface must be determined as well as the amount of nitrogen recycling within the embayment, specifically nitrogen regeneration from sediments. Sediment nitrogen recycling results primarily from the settling and decay of phytoplankton and macroalgae (and eelgrass when present). During decay, organic nitrogen is transformed to inorganic forms, which may be released to the overlying waters or lost to denitrification within the sediments. Burial of nitrogen is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters and leads to errors in predicting water quality if it is not included in determination of summertime nitrogen load.

The MEP Technical 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) to the Pleasant Bay embayment system by sub-watershed. The Pleasant Bay watershed was sub-divided to define contributing areas each of the major inland freshwater systems and to each major sub-embayment to Pleasant Bay and further sub-divided into regions greater and less than 10 year groundwater travel time from the receiving estuary, a total of 95 sub-watersheds in all. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds and each embayment (see Chapter III).

The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges to the watershed have reached the embayment. This involves a temporal review of land use changes and the time of groundwater travel provided by the USGS watershed model. After reviewing the percentage of nitrogen loading in the less than 10 year time of travel (LT10) and greater than 10 year time of travel (GT10) watersheds (Table IV-1), previous nitrogen loading assessments (Eichner, et al., 1998), land use development records, and water quality modeling, it was determined that Pleasant Bay is currently in balance with its watershed load. The bulk (74%) of the watershed nitrogen load is within 10 years flow to Pleasant Bay and its subestuaries. Therefore, the distinction of less than 10 year and greater than 10 year time of travel regions within a subwatershed (Figure III-1) was eliminated and the number of subwatersheds was reduced to 59 (Figure IV-1). The overall result of the timing of development relative to groundwater travel times is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuaries (after accounting for natural attenuation, see below).

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

WATERSHED		LT10	GT10	TOTAL	%LT10
Name	#	kg/yr	kg/yr	kg/yr	
Baker Pond	1	208	35	243	86%
Cliff Pond	2	960	1262	2222	43%
Crystal Lake	3	388	128	517	75%
Deep Pond	4	170	5	174	97%
Grassy Pond	5	183	0	183	100%
Higgins Pond	6	134	0	134	100%
Little Cliff Pond	7	163	0	163	100%
Mud Pond	8	54	0	54	100%
Pilgrim Lake	9	562	0	562	100%
Rafe Pond	10	42	0	42	100%
Ruth Pond	11	41	226	267	15%
Sarahs Pond	12	85	311	397	22%
Shoal Pond	13	405	4	409	99%
Twinings Pond	14	298	80	378	79%
Uncle Harveys Pond	15	123	0	123	100%
Uncle Seths Pond	16	196	6	202	97%
Cliff Pond WELL_ORL	17	27	0	27	100%
Freeman's Way WELL_BRE	18	1063	0	1063	100%
Gould Pond WELL_ORL	19	276	0	276	100%
Pleasant Bay Rd WELL_HAR	20	331	0	331	100%
Silas Rd WELL_BRE	21	240	0	240	100%
Well 7 WELL_ORL	22	407	0	407	100%
Areys Pond	23	417	20	436	95%
Barley Neck	24	633	433	1066	59%
Kescayo Gansett Pond	25	341	103	444	77%
Kescayo Gansett River	26	132	0	132	100%
Kescayo Gansett Stream	27	45	0	45	100%
Lower River	28	2055	0	2055	100%
Meetinghouse Pond	29	1516	953	2470	61%
Namequoit River	30	932	143	1074	87%
Pah Wah Pond Bog	31	48	0	48	100%
Pah Wah Pond	32	276	386	661	42%
Pleasant Bay Proper	33	32310	3801	36111	89%
Pochet Neck	34	933	787	1720	54%

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

WATERSHED		LT10	GT10	TOTAL	%LT10
Name	#	kg/yr	kg/yr	kg/yr	
Pochet Neck Stream	35	493	377	870	57%
Quanset Pond Bog	36	21	0	21	100%
Quanset Pond	37	489	6	495	99%
Round Cove	38	827	772	1599	52%
Tar Kiln Stream	39	846	1413	2259	37%
The Horseshoe	40	58	0	58	100%
Upper River	41	706	238	943	75%
Mill Pond Fresh	42	650	0	650	100%
Goose Pond	43	343	0	343	100%
Trout Pond	44	320	0	320	100%
Schoolhouse Pond	45	195	0	195	100%
Stillwater Pond	46	387	0	387	100%
Lovers Lake	47	559	0	559	100%
Emery Pond	48	114	0	114	100%
Bassing Pond	49	160	0	160	100%
Hawksnest Pond	50	133	0	133	100%
Muddy Creek WELL_HAR	51	589	0	589	100%
Lower Muddy Creek	52	1408	992	2400	59%
Upper Muddy Creek	53	2710	1004	3714	73%
Ryder Cove	54	2306	1019	3325	69%
Crows Pond	55	1388	651	2039	68%
Bassing Harbor	56	753	234	987	76%
Frostfish Creek	57	467	363	830	56%
Upper Frostfish Creek	58	264	0	264	100%
Chatham Harbor	59	6242	0	6242	100%
TOTAL		68747	15751	84174	81%

NOTE: Less than 10 year time of travel loads are for individual resources. If total loads for resources beyond the ten year time of travel line (e.g., Cliff Pond) are moved into the GT10 column for the purposes of considering travel time to Pleasant Bay proper, 74% of the overall load to Pleasant Bay is within less than 10 years. Totals may not add due to rounding.

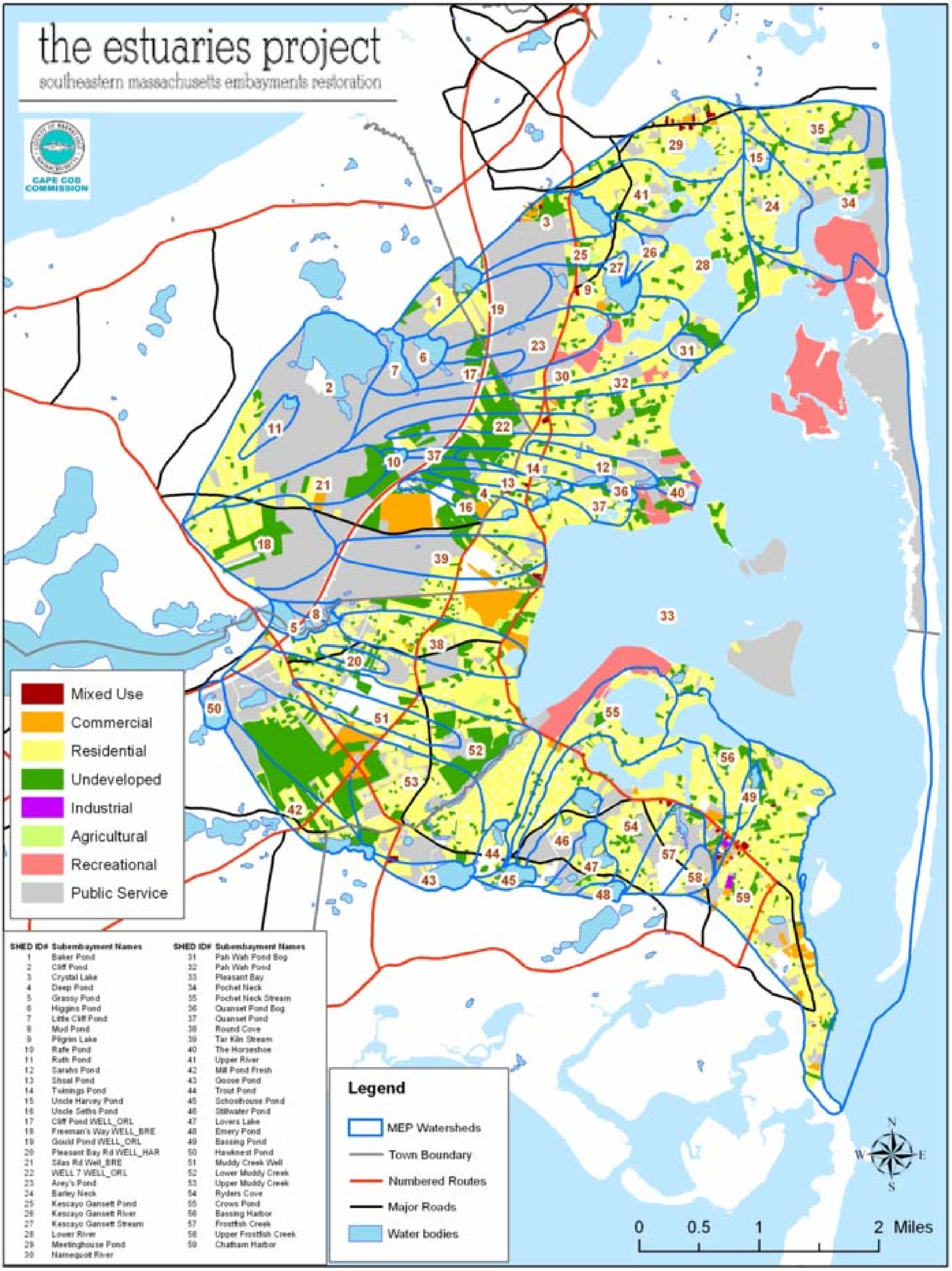


Figure IV-1. Land-use in the Pleasant Bay watershed. The watershed encompasses portions of the Towns of Orleans, Brewster, Harwich, and Chatham. Land use classifications are based on assessors' records provided by each of the towns.

In order to determine nitrogen loads from the watersheds, detailed individual lot-by-lot data is used for some portion of the loads, while information developed from other detailed studies is applied to other portions. The Linked Watershed-Embayment Management Model (Howe's & Ramsey 2001) uses a land-use Nitrogen Loading Sub-Model based upon subwatershed-specific land-uses and pre-determined nitrogen loading rates. For the Pleasant Bay embayment system, the model used Orleans, Brewster, Harwich, and Chatham-specific land-use data transformed to nitrogen loads using both regional nitrogen load factors and local watershed-specific data (such as parcel by parcel water use). Determination of the nitrogen loads required obtaining watershed-specific information regarding wastewater, fertilizers, runoff from impervious surfaces and atmospheric deposition. The primary regional factors were derived for southeastern Massachusetts from direct measurements. The resulting nitrogen loads represent the "potential" or unattenuated nitrogen load to each receiving embayment, since attenuation during transport has not yet been included.

Natural attenuation within the Pleasant Bay watershed of nitrogen during transport from land-to-sea (Section IV.2) was determined based upon a site-specific study within the freshwater portions of Tar Kiln Stream and Kescayo Gansett Stream and through the 25 freshwater ponds within the watershed. Attenuation during transport through each of the major fresh ponds was determined through (a) comparison with other Cape Cod lake studies and (b) data collected on each pond. Attenuation during transport through these fresh ponds was conservatively assumed to equal 50% based on available monitoring of selected Cape Cod lakes. Available historic data collected from individual fresh ponds in the Pleasant Bay watershed confirmed the appropriateness of this general assumption.

Natural attenuation during stream transport or in passage through fresh ponds of sufficient size to effect groundwater flow patterns (area and depth) is a standard part of the data collection effort of the MEP. However, if additional attenuation of nitrogen were occurring during transport, given the distribution of the nitrogen sources, nitrogen loading to the estuary would only be slightly (~10%) overestimated. Based upon these considerations, the MEP Technical Team used the conservative estimate of nitrogen loading based upon direct groundwater discharge. Internal nitrogen recycling was also determined throughout the tidal reaches of the Pleasant Bay embayment; measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying water-column. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

IV.1.1 Land Use and Water Use Database Preparation

Estuaries Project staff obtained digital parcel and tax assessors data from the Towns of Orleans, Brewster, Harwich, and Chatham. Digital parcels and land use data are from 2004 for Orleans, 2004 for Brewster, 1999 and 2005, respectively, for Harwich, and 2004 for Chatham and were obtained from the respective town planning departments or Cape Cod Commission files. These land use databases contain traditional information regarding land use classifications (MADOR, 2002) plus additional information developed by each of the towns. The parcel data and assessors' databases for all the towns were combined for the MEP analysis by using the Cape Cod Commission Geographic Information System (GIS).

Figure IV-1 shows the land uses within the Pleasant Bay study area. Land use in the study area is one of nine land use categories: 1) residential, 2) commercial, 3) industrial, 4) undeveloped, 5) agricultural, 6) mixed use, 7) golf course and recreational land, 8) public service/government, including road rights-of-way, and 9) freshwater ponds. These land use categories, except the ponds, are aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MADOR, 2002). These categories are common to each town in the watershed. "Public service" in the MADOR system is tax exempt properties, including lands owned by government (e.g., wellfields, schools, open space, roads) and private groups like churches and colleges.

In the overall Pleasant Bay watershed, the predominant land use based on area is residential, which accounts for 38% of the watershed area; public service (government owned lands, roads, and rights-of-way) is the second highest percentage of the watershed (37%) (Figure IV-2). In addition, 74% of the parcels in the system watershed are classified as residential. Single family residences (MADOR land use code 101) are 80% of the residential parcels and single family residences are 90% of the residential land area. In the individual subwatersheds, residential land uses vary between 23 and 69% of the subwatershed areas. Public service land uses are the dominant category in subwatersheds where residential land uses are the second highest percentage and are usually the second highest percentage use in subwatersheds where residential uses are the highest. Recreational (e.g. golf courses) or undeveloped land uses are usually either the third or the fourth highest percentage land uses. Overall, undeveloped land uses account for 12% of the whole Pleasant Bay watershed, while commercial properties account for approximately 2% of the watershed area.

In order to estimate wastewater flows within the Pleasant Bay study area, MEP staff also obtained parcel by parcel water use information for the Orleans Water Department via the Planning Department, the Brewster Water Department, the Harwich Water Department, and the Chatham Department of Health and Environment. Orleans water data is twelve months of water use between 2002 and 2003, Brewster water data is three years between 2002 and 2004, Harwich data is 2004 water use, and Chatham data is annualized water consumption between 2002 and 2003. Water use information was linked to the parcel and assessors data using GIS techniques. Water use for each parcel was converted to an annual volume for purposes of the nitrogen loading calculations; multiple year data was averaged. There are no municipal WWTFs in the Pleasant Bay watershed. Wastewater-based nitrogen loading from the individual parcels using on-site septic systems is based upon the measured water-use, nitrogen concentration, and an assumed consumptive loss of water before the remainder is treated in a septic system.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

The Massachusetts Estuaries Project septic system nitrogen loading rate is fundamentally based upon a per Capita Nitrogen load to the receiving aquatic system. Specifically, the MEP septic system wastewater nitrogen loading is based upon a number of studies and additional information that directly measured septic system and per capita loads on Cape Cod or in similar geologic settings (Nelson et al. 1990, Weiskel & Howes 1991, 1992, Koppelman 1978, Frimpter et al. 1990, Brawley et al. 2000, Howes and Ramsey 2000, Costa et al. 2001). Variation in per capita nitrogen load has been found to be relatively small, with average annual per capita nitrogen loads generally between 1.9 to 2.3 kg person-yr^{-1}.

However, given the seasonal shifts in occupancy and rapid population growth throughout southeastern Massachusetts, decennial census data yields accurate estimates of total population only in selected watersheds. To correct for this uncertainty and more accurately assess current nitrogen loads, the MEP employs a water-use approach. The water-use

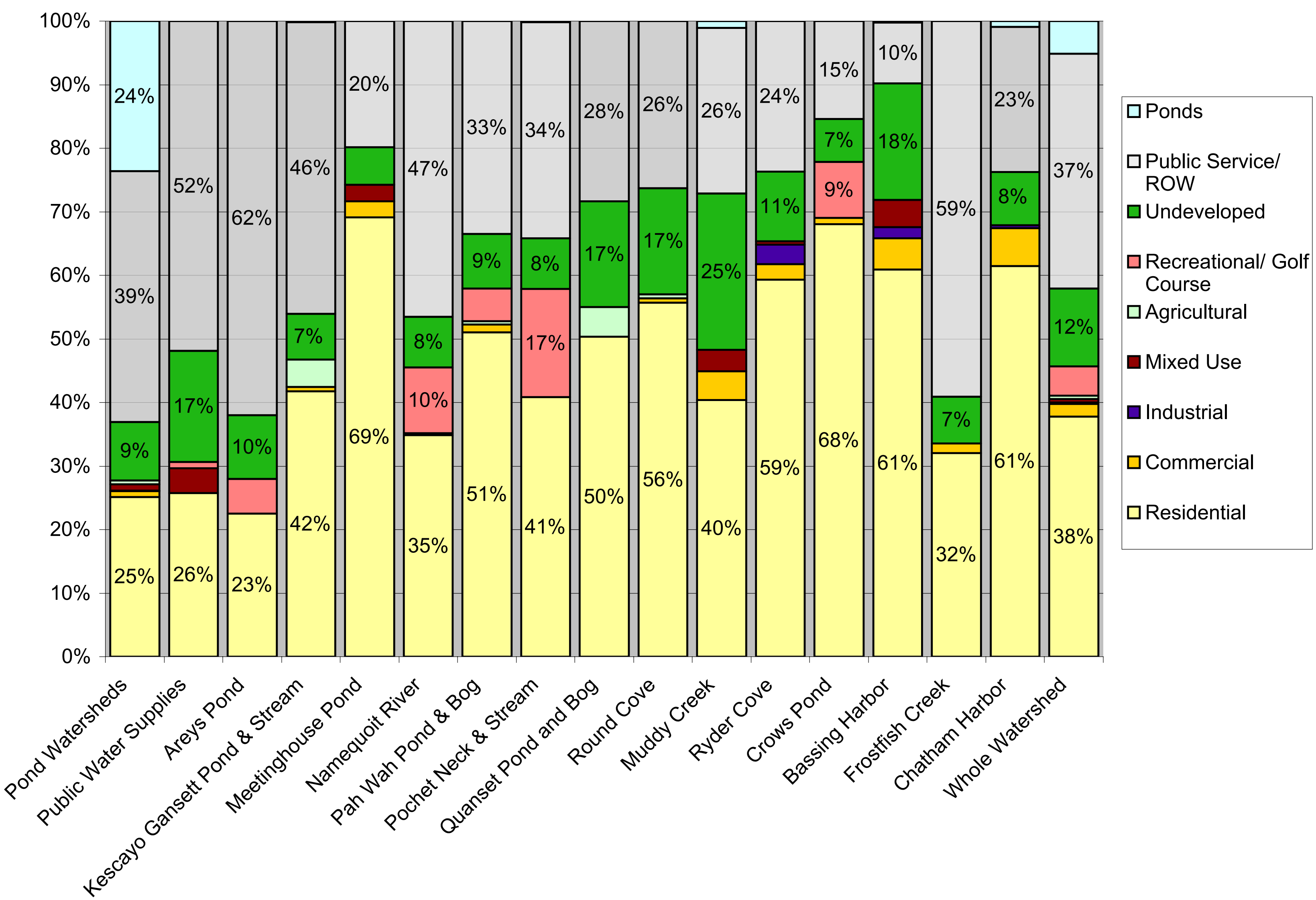


Figure IV-2. Distribution of land-uses within the major subwatersheds and whole watershed to Pleasant Bay. Only percentages greater than or equal to 7% are shown.

approach is applied on a parcel-by-parcel basis within a watershed, where annual water meter data is linked to assessors parcel information using GIS techniques. The parcel specific water use data is converted to septic system nitrogen discharges (to the receiving aquatic systems) by adjusting for consumptive use (e.g. irrigation) and applying a wastewater nitrogen concentration. The water use approach focuses on the nitrogen load, which reaches the aquatic receptors downgradient in the aquifer.

All nitrogen losses within the septic system are incorporated. For example, information developed at the MASSDEP Alternative Septic System Test Center at the Massachusetts Military Reservation on Title 5 septic systems have shown nitrogen removals between 21% and 25%. Multi-year monitoring from the Test Center has revealed that nitrogen removal within the septic tank was small (1% to 3%), with most (20 to 22%) of the removal occurring within five feet of the soil adsorption system (Costa et al. 2001). Downgradient studies of septic system plumes indicate that further nitrogen loss during aquifer transport is negligible (Robertson et al. 1991, DeSimone and Howes 1996).

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

The nitrogen loads developed using this approach have been validated in a number of long and short term field studies where integrated measurements of nitrogen discharge from watersheds could be directly measured. Weiskel and Howes (1991, 1992) conducted a detailed watershed/stream tube study that monitored septic systems, leaching fields and the transport of the nitrogen in groundwater to adjacent Buttermilk Bay. This monitoring resulted in estimated annual per capita nitrogen loads of 2.17 kg (as published) to 2.04 kg (if new attenuation information is included). Modeled and measured nitrogen loads were determined for a small sub-watershed to Mashapaquit Creek in West Falmouth Harbor (Smith and Howes 2006) where measured nitrogen discharge from the aquifer was within 5% of the modeled N load. Another evaluation was conducted by surveying nitrogen discharge to the Mashpee River in reaches with swept sand channels and in winter when nitrogen attenuation is minimal. The modeled and observed loads showed a difference of less than 8%, easily attributable to the low rate of attenuation expected at that time of year and under the ecological situation (Samimy and Howes, unpublished data).

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

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

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

In order to provide an independent validation of the residential water use average within the Pleasant Bay study area, MEP staff reviewed US Census population values in the towns in the watershed. The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so each person generates 55 gpd of wastewater. Based on data collected during the 2000 US Census, average occupancy within the four towns in the watershed varies between 2.05 (Orleans) and 2.45 (Brewster), while year-round occupancy of available housing units varies between 47% (Chatham) and 61% (Orleans). Average water use for single family residences with municipal water accounts in the Pleasant Bay watershed is 148 gpd. If this flow is multiplied by 0.9 (a reasonable factor for consumptive loss), the watershed average is 133 gpd. If this flow is then divided by 55 gpd, the average estimated occupancy in the watershed is 2.4 people per household. This simple comparison between population and water use shows a good match and provides further validation for the use of water use data for calculating wastewater nitrogen loads.

Although water use information exists for 86% of the over 6,600 developed parcels in the Pleasant Bay watershed, there are 924 parcels that are assumed to utilize private wells for drinking water. These are properties that classified with land use codes that should be developed (e.g., 101 or 325), have been confirmed as having buildings on them through a review of aerial photographs, and do not have a listed account in the water use databases. Of the 924 parcels, 66% of them (614) are classified as single family residences (land use code 101) and another 30% are classified as other types of residential development (e.g. 109 (multiple houses on a single property)). The remaining 4% are either commercial, industrial, or tax-exempt (e.g., 900's in state class code). MEP staff used current water use to develop a watershed-specific water use estimate for residential uses assumed to utilize private wells (Table IV-2). This flow was also used for the 31 existing commercial, industrial, and government properties without water use located within the watershed.

Table IV-2. Average Water Use in Pleasant Bay Watershed.

Land Use	State Class Codes	# of Parcels with Water Use in Watershed	Water Use (gallons per day)	
			Watershed Average	Subwatershed Average Range
Residential	101	5,395	148	0 to 263
Commercial	300 to 389	43	466	184 to 1,256
Industrial	400 to 439	1	77	-

Note: All data for analysis supplied by towns.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of estuary watershed nitrogen loading is usually fertilized lawns and golf courses, with lawns being the predominant source within this category. In order to add this source to the nitrogen loading model for the Pleasant Bay system, MEP staff reviewed available information about residential lawn fertilizing practices and incorporated site-specific fertilizer application rates for large tracts of turf, such as golf courses and ballfields, by contacting turf managers or town/school staff and reviewing applicable regulatory materials submitted to the Cape Cod Commission for individual golf courses within the watershed.

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

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

Technical Team working with the Orleans Wastewater Planning Committee, conducted a smaller survey of 340 homes throughout the Town of Orleans. The results of this survey indicated that the number of fertilizations per lawn in Orleans was similar to that in the upper Cape survey, 1.76 versus 1.44. However, within the survey there was a very high number of homes serviced by commercial lawn companies (over 1/3). These lawns were fertilized at a high rate relative to home-owner serviced properties. The home-owner service properties were fertilized only at a rate of 0.85 applications per year, with an average contribution to ground water of 0.64 lb N/lawn/yr, compared to 3.29 lb N/lawn/yr to groundwater for the professionally maintained lawns. The overall results indicated a potentially higher nitrogen loading per lawn in Orleans of 1.51 lb/lawn/yr (weighted average). However, this is due to the high fraction of homes with professionally maintained lawns in the survey. Given the large areas of the watershed within Harwich, Brewster and Chatham, the uncertainty in the regional percentage of lawns maintained professionally; and the fact that if the Orleans rates are applied to the entire Pleasant Bay watershed the change in loading is <2%, the estimate of 1.08 lb N/lawn/yr was used in the water quality model. However, it cautions of the potential for behavioral changes to greatly increase the nitrogen loading from this source..

There are four golf courses in the Pleasant Bay watershed: Captains Golf Course, Eastward Ho in Chatham, Chatham Seaside Links, and Cape Cod National Golf Club on the border between Brewster and Harwich. Golf courses usually have different fertilizer application rates for different turf areas, usually higher annual application rates for tees and greens (~3-4 pounds per 1,000 square feet) and lower rates for fairways and roughs (~2-3.5 pounds per 1,000 square feet). MEP staff reviewed available information and contacted turf managers for golf courses in the watershed in order to develop watershed-specific nitrogen application rates. Table IV-3 summarizes the application rates used in the Pleasant Bay watershed nitrogen loading model.

Table IV-3. Nitrogen Application Rates for Golf Courses in the Pleasant Bay Watershed.

	Turf Area Application Rate (lb/1,000 ft2/yr)				Source:
Course	Green	Tee	Fairway	Rough	
Captains Golf Course	6	5.5	4.5	4.5	CCC files
Eastward Ho CC	3	3	3	2	Chatham MEP Report
Chatham Seaside Links	3	3	3	2	Chatham MEP Report
Cape Cod National Golf Club	4.5	2.25	1.75	2	CCC files

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and MASSDEP'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). Cranberry bog fertilizer application rate and percent nitrogen attenuation in the bogs is based on the only annual study of nutrient cycling and loss from cranberry agriculture (Howes and Teal, 1995). Only the bog loses measurable nitrogen, the forested upland release only very low amounts. For the watershed N loading analysis, the areas of active bog surface are based on 85% of the total area for properties classified as cranberry bogs in the town-supplied land use classifications. Factors used in the MEP nitrogen loading analysis for the Pleasant watershed are summarized in Table IV-4.

Table IV-4. Primary Nitrogen Loading Factors used in the Pleasant Bay MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from the Orleans, Brewster, Harwich, and Chatham data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.

Nitrogen Concentrations:	mg/l	Recharge Rates:	in/yr
Road Run-off	1.5	Impervious Surfaces	40
Roof Run-off	0.75	Natural and Lawn Areas	27.25
Direct Precipitation on Embayments and Ponds	1.09	Water Use/Wastewater:	
Natural Area Recharge	0.072	Existing developed parcels wo/water accounts:	148 gpd
Wastewater Coefficient	23.63		
Fertilizers:			
Average Residential Lawn Size (ft^2)*	5,000		
Residential Watershed Nitrogen Rate (lbs/lawn)*	1.08	Existing developed parcels w/water accounts:	Measured annual water use
Cranberry Bogs nitrogen application (lbs/ac)	31	Buildout Parcels Assumptions:	
Cranberry Bogs nitrogen attenuation	34%	Residential parcels:	148 gpd
Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined from site-specific information		Commercial and industrial parcels:	95 gpd/1,000 ft2 of building
		Commercial and industrial building coverage	12%

IV.1.3 Calculating Nitrogen Loads

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

The review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Individualized information for parcels with atypical nitrogen loading (condominiums, golf courses, etc.) were also assigned at this stage. It should be noted that small shifts in nitrogen loading due to the above assignment procedure generally have a negligible effect on the total nitrogen loading to the Pleasant Bay estuary. The assignment effort was undertaken to better define the sub-embayment loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

Following the assignment of all parcels, subwatershed modules were generated for each of the 95 sub-watersheds summarizing water use, parcel area, frequency, sewer connections, private wells, and road area. As mentioned above, these results were then condensed to 59 subwatersheds based upon the time of travel analysis (less than 10 years vs. greater than 10 years) discussed above. The individual sub-watershed modules were then integrated to create the Pleasant Bay Watershed Nitrogen Loading module with summaries for each of the individual sub-embayments. The sub-embayments represent the functional embayment units for the Linked Watershed-Embayment Model's water quality component.

For management purposes, the aggregated embayment watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Pleasant Bay System, the major types of nitrogen loads are: wastewater (e.g., septic systems), fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-5). The output of the watershed nitrogen loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-4 a-f). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation during transport to the estuarine system before use in the embayment water quality sub-model.

It should be noted that 2 of the tributary sub-embayments to the Pleasant Bay System (Bassing Harbor, Muddy Creek) were previously analyzed by the MEP (2003). Total nitrogen loads determined in that analysis were higher than those in the present study. At the time of the earlier effort, it was clear that these systems would have to be revisited as part of the Pleasant Bay effort (i.e. this report), not just to refine the nitrogen loading estimates, but also because the conditions in Pleasant Bay play an important role in the ability of these 2 sub-embayments to tolerate nitrogen inputs from their associated sub-watersheds. Since the previous models of Bassing Harbor and Muddy Creek were integrated into the Pleasant Bay system-wide models, they were also refined for the present report. The critical issue is whether or not the change in nitrogen loading to these systems has any effect on the amount of wastewater than needs to be removed from their associated sub-watersheds, it does not. However, due to the significantly greater amount of water quality data (both number of samplings and improved spatial distribution), the availability of boundary condition data (both the Atlantic Ocean and in the Bay adjacent the tidal inlets to Bassing Harbor and Muddy Creek, and refined nitrogen threshold data (system-wide) an increase in the amount of wastewater management will be needed for habitat restoration. A more complete explanation of the nitrogen loading and the relationship to the threshold nitrogen levels and threshold nitrogen loads is presented in Section VIII-3.

Since groundwater outflow from a pond can enter more than one downgradient sub-watershed, the length of shoreline on the downgradient side of the pond was used to apportion the pond-attenuated nitrogen load to respective downgradient watersheds. The apportionment was based on the percentage of discharging shoreline bordering each downgradient sub-watershed. So for example, Little Cliff Pond has a downgradient shoreline of 3,701 feet; 16% of that shoreline discharges out of the Pleasant Bay watershed, 21% goes to Baker Pond (watershed 1 in Figure IV-1), 49% goes to Higgins Pond (watershed 6 in Figure IV-1) and 13% goes to the Cliff Pond Well in Orleans (watershed 17 in Figure IV-1). The attenuated nitrogen load discharging from Little Cliff Pond is divided among these subwatersheds based on these percentages of the downgradient shoreline.

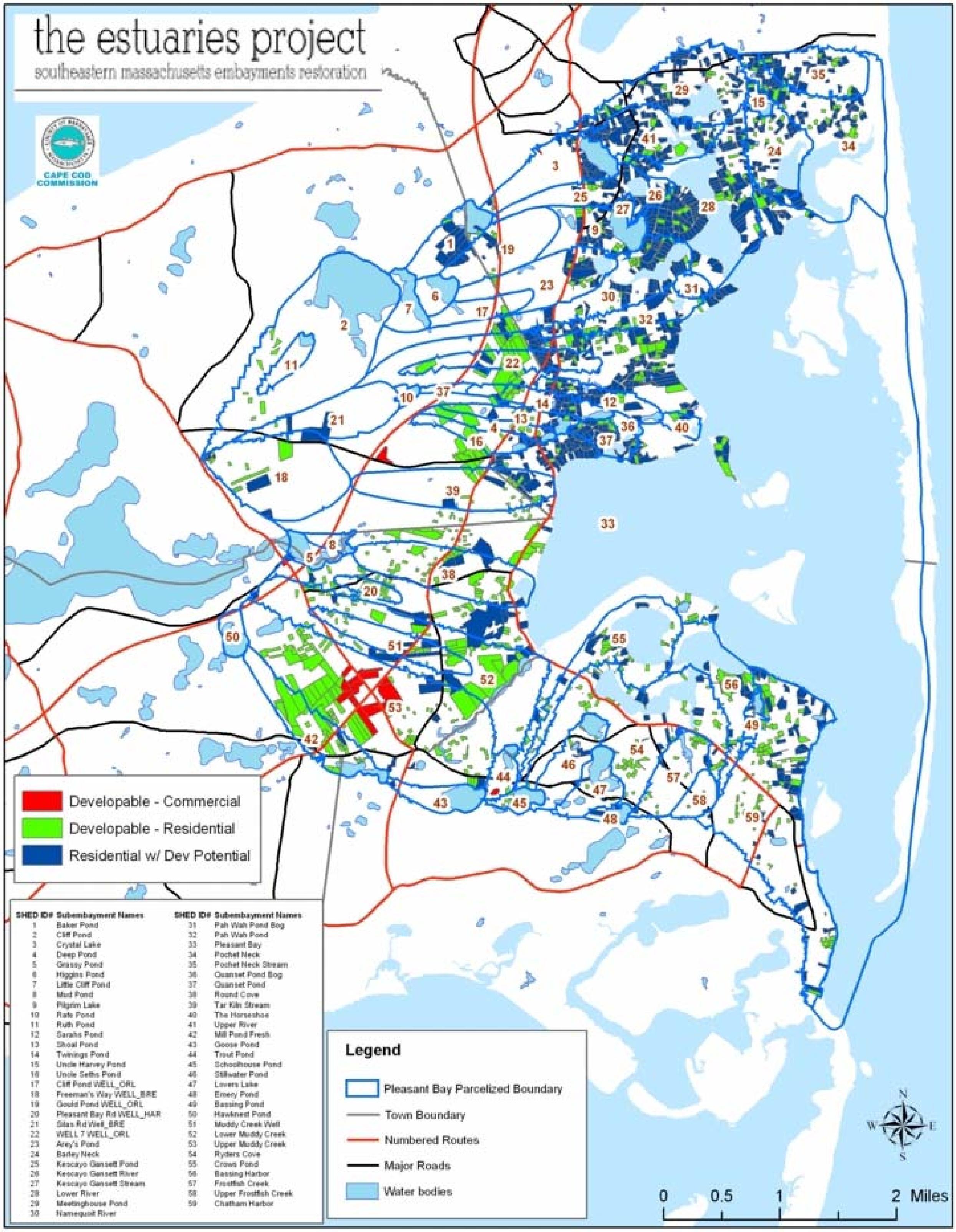
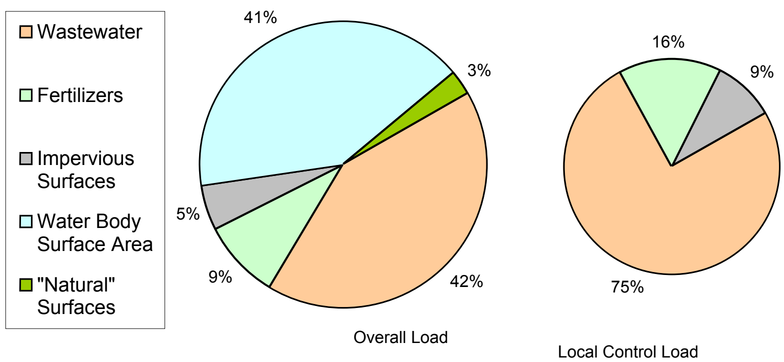


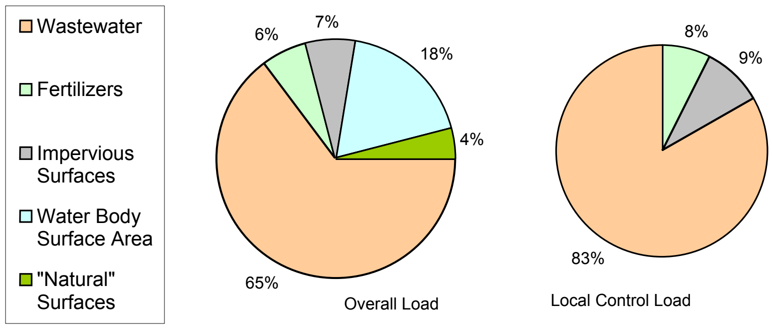
Figure IV-3. Parcels, Parcelized Watersheds, and Developable Parcels in the Pleasant Bay watersheds.

Table IV-5. Pleasant Bay Nitrogen Loads. Attenuation of Pleasant Bay system nitrogen loads occurs as nitrogen moves through upgradient ponds and streams during transport to the estuary.

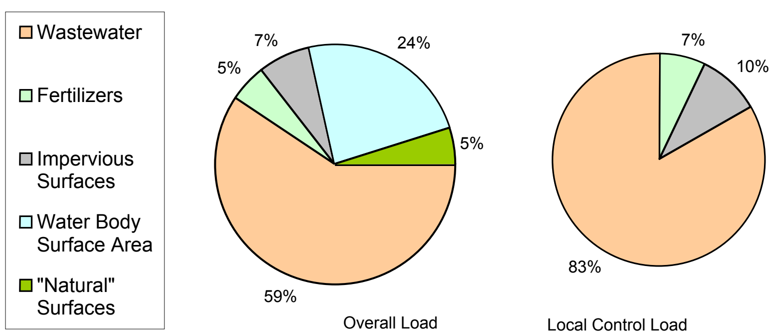
		Pleasant Bay N Loads by Input:							Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout	% of Pond Outflow	UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Pleasant Bay Whole System		**34290**	**7117**	**4074**	**33403**	**2283**	**14636**		**81167**		**78001**	**95803**		**91924**
Pleasant Bay Main Basin	**18, 20, 22, 33, 59 + Ponds**	**12043**	**3443**	**1344**	**27829**	**882**	**5335**		**45542**		**44955**	**50877**		**50091**
Chatham Harbor	95	5182	408	481	34	137	711		6242		6242	6953		6953
Pleasant Bay Main Basin Estuary surface deposition					27417				27417		27417	27417		27417
Pochet Neck	**24, 24, 35 + UHP**	**2449**	**223**	**269**	**681**	**157**	**1276**		**3779**		**3718**	**5056**		**4980**
Barley Neck	24	713	73	75	0	28	210		889		889	1099		1099
Pochet Neck	34	980	80	103	0	90	417		1253		1253	1670		1670
Pochet Neck Stream	35	685	64	82	4	35	621		870		870	1492		1492
Barley Neck Estuary surface deposition					177				177		177	177		177
Pochet Neck Estuary surface deposition					467				467		467	467		467
River System	**23, 25, 26, 27, 28, 29, 30, 41 + BP, PL, CL, HP, LCP + CPW**	**5844**	**486**	**667**	**2322**	**484**	**3648**		**9802**		**8536**	**13449**		**11887**
Meetinghouse Pond	29	1871	139	193	0	53	752		2256		2256	3008		3008
Kescayo Gansett Pond	25,27 + BP,PL,CL	567	49	63	214	71	417		965		649	1382		983
Kescayo Gansett River	26 + PL	244	19	24	98	27	106		411		247	516		319
Arey's Pond	23 + HP + CPW	343	28	58	144	77	273		650		475	922		745
Namequoit River	30 + PL	807	69	97	92	89	511		1155		1001	1666		1479
Upper River	41 + CL	854	75	87	152	66	526		1234		1013	1759		1452
Lower River	28 + CL,PL	1158	106	144	146	101	1064		1655		1418	2719		2424
Meetinghouse Pond Estuary surface deposition					213				213		213	213		213
Kescayo Gansett Pond Estuary surface deposition					73				73		73	73		73
Kescayo Gansett River Estuary surface deposition					9				9		9	9		9
Arey's Pond Estuary surface deposition					66				66		66	66		66
Namequoit River Estuary surface deposition					191				191		191	191		191
Upper River Estuary surface deposition					105				105		105	105		105
Lower River Estuary surface deposition					818				818		818	818		818
Pah Wah Pond	**31, 32**	**551**	**45**	**50**	**30**	**33**	**344**		**709**		**709**	**1053**		**1053**
Pah Wah Pond Estuary surface deposition					30				30		30	30		30
Quanset Pond	**37 + RFP, SHP, TP, QPB**	**652**	**46**	**71**	**121**	**36**	**276**		**927**		**713**	**1203**		**936**
Quanset Pond Estuary surface deposition					62				62		62	62		62
Tar Kiln Stream	**39**	**655**	**1485**	**52**	**24**	**43**	**316**		**2259**		**2259**	**2575**		**2575**
Tar Kiln Estuary surface deposition					24				24		24	24		24
Round Cove	**38 + MP**	**1157**	**175**	**154**	**77**	**54**	**347**		**1616**		**1607**	**1963**		**1954**
Round Cove Estuary surface deposition					62				62		62	62		62
The Horseshoe	**40 + SP**	**322**	**24**	**33**	**52**	**24**	**224**		**454**		**256**	**678**		**385**
The Horseshoe Estuary surface deposition					23				23		23	23		23
Muddy Creek	**51, 52, 53 + MPF, GOP, HWP, TTP**	**5275**	**612**	**776**	**400**	**332**	**1946**		**7395**		**7027**	**9341**		**8946**
Upper Muddy Creek	53	2839	344	395	247	189	1322		4014		3860	5336		5156
Upper Muddy Creek Estuary surface deposition					59				59		59	59		59
Lower Muddy Creek	51, 52 + TTP	2436	268	381	153	143	624		3381		3167	4006		3789
Lower Muddy Creek Estuary surface deposition					75				75		75	75		75
Ryder Cove System	**54, 55, 56, 57, 58 + SCP, LL, EP, SWP, BSP**	**5340**	**577**	**659**	**1868**	**239**	**923**		**8683**		**8221**	**9606**		**9117**
Ryder Cove	54 + SCP, LL, EP, SWP	**2808**	**308**	**357**	**922**	**133**	**505**		**4527**		**4083**	**5032**		**4562**
Ryder Cove Estuary surface deposition					473				473		473	473		473
Crows Pond	55 + SCP	**1212**	**138**	**147**	**512**	**39**	**157**		**2049**		**2044**	**2206**		**2201**
Crows Pond Estuary surface deposition					507				507		507	507		507
Bassing Harbor	56 + BSP	**518**	**14**	**51**	**399**	**30**	**110**		**1012**		**999**	**1123**		**1108**
Bassing Harbor Estuary surface deposition					391				391		391	391		391
Frostfish Creek	57,58	**802**	**117**	**104**	**35**	**37**	**151**		**1095**		**1095**	**1246**		**1246**
Frostfish Creek Estuary surface deposition					35				35		35	35		35



a. Pleasant Bay System Overall

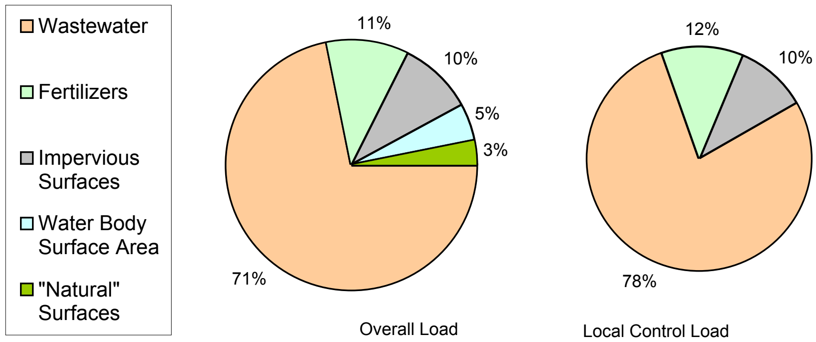


b. Pochet Neck

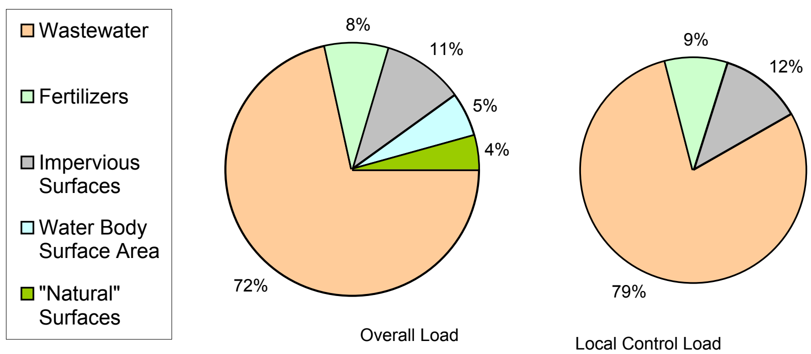


c. The River System

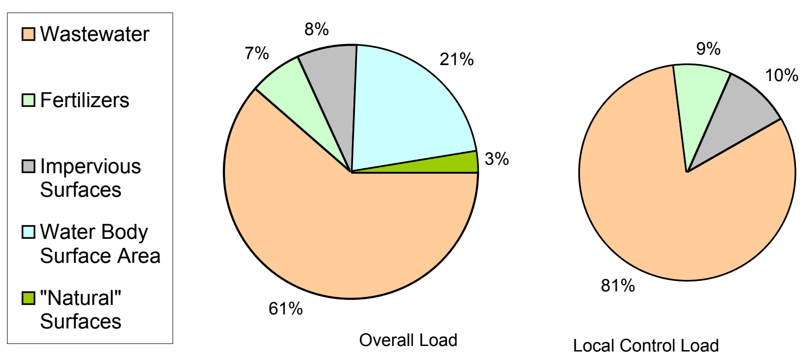
Figure IV-4 (a-c). Land use-specific unattenuated nitrogen load (by percent) for select sub-embayments to reflect a variety of load distributions: (a) overall Pleasant Bay System watershed, (b) Pochet Neck subwatershed, and (c) River System subwatershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.



d. Round Cove



e. Muddy Creek



f. Ryder Cove System

Figure IV-4 (d-f). Land use-specific unattenuated nitrogen load (by percent) for select sub-embayments to reflect a variety of load distributions: (d) Round Cove subwatershed, (e) Muddy Creek subwatershed, and (f) Ryder Cove subwatershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally kettle hole depressions that intercept the surrounding groundwater table revealing what some call “windows on the aquifer.” Groundwater typically flows into the pond along the upgradient shoreline, then lake water flows back into the groundwater system along the downgradient shoreline. Occasionally, a Cape Cod pond will have a stream outlet or herring run as well. Since the nitrogen loads flow into the pond with the groundwater, the relatively more productive ecosystems in the ponds incorporate some of the nitrogen, retain some of it in the sediments, and change it among its various oxidized and reduced forms. As a result of these interactions, some of the nitrogen is removed from the watershed system, mostly through burial in the sediments and denitrification that returns it to the atmosphere. Following these reductions, the remaining, reduced or attenuated loads flow back into the groundwater system along the downgradient side of the pond or leave the pond through a stream outlet with eventual discharge into the downgradient embayment. The nitrogen load summary in Table IV-5 includes both the unattenuated (nitrogen load to each subwatershed) and attenuated nitrogen loads.

Pond nitrogen attenuation in freshwater ponds is generally assumed to be 50% in MEP analyses; in some cases, if sufficient monitoring information is available, an alternative attenuation rate is incorporated into the watershed nitrogen loading modeling (Three Bays MEP Report, 2005). Detailed studies of other southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE, 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM, 2001) have supported a 50% attenuation factor. In order to estimate nitrogen attenuation in the ponds physical and chemical data for each pond is reviewed. Available bathymetric information is reviewed relative to measured pond temperature profiles to determine whether an epilimnion (*i.e.*, well mixed, homothermic, upper portion of the water column) exists in each pond. Bathymetric information is necessary to develop a residence or turnover time and complete a estimate of nitrogen attenuation. Of the ponds in the Pleasant Bay study area, bathymetric information is available for 17 of the ponds with delineated watersheds, but unavailable for Ruth, Rafe, Grassy, Hawksnest, Shoal, Uncle Seths, Bassing, and Mud. Of the ponds with bathymetric information, Goose, Schoolhouse, Cliff, Higgins, Crystal, and Pilgrim are deep enough to develop strong temperature stratification and a separate epilimnion. Generally, if a stable epilimnion develops, it is the appropriate volume for gauging nitrogen attenuation in a pond, since it is separate from the lower thermal layers, which are, in turn, usually impacted by sediment regeneration of nitrogen.

In MEP analyses, available nitrogen concentrations from individual ponds are reviewed to establish whether sediment regeneration is a significant factor in a pond and, if not, the entire volume of the pond is used to determine a turnover time. Turnover time is how long it takes the recharge from the upgradient watershed to completely exchange the water in the pond or, in the case of a thermally stratified pond, exchange just the epilimnion. The total mass of nitrogen in the pond or epilimnion is adjusted using the pond turnover time to determine the annual nitrogen load returned to the aquifer through the downgradient shoreline. This mass is then compared to the nitrogen load coming from the pond’s watershed to determine the nitrogen attenuation factor for the pond. Generally, monitoring is insufficient to support use of a factor different than the standard 50% attenuation. Table IV-6 presents available turnover times and attenuation factors for the 25 ponds with subwatersheds within the overall Pleasant Bay watershed.

Table IV-6. Nitrogen attenuation by Freshwater Ponds in the Pleasant Bay watershed based upon 2001 through 2004 Cape Cod Pond and Lakes Stewardship (PALS) program sampling and National Park Service-supported sampling in Orleans and Brewster. These data were collected to provide a site specific check on nitrogen attenuation by these systems. The MEP Linked N Model for Pleasant Bay uses a standard value of 50% for the pond systems.

Pond	PALS ID	Area acres	Maximum Depth m	Overall turnover time yrs	# of TN samples	N Load Attenuation %
Emery	CH-491	14.11	6.2	1.8	9	11%
Goose	CH-458	41.25	11.0	4.0	14	81%
Lovers	CH-428	37.73	9.6	2.0	8	62%
Mill	CH-440	23.45	2.8	0.2	8	47%
Schoolhouse	CH-463	22.78	13.2	4.2	12	81%
Stillwater	CH-396	18.71	13.7	0.7	8	60%
Cliff	BR-1028	201.86	26.8	3.4	27	73%
Higgins	BR-194	28.51	20.4	1.8	48	72%
Little Cliff	BR-192	34.49	13.8	0.6	24	77%
Crystal	OR-153	38.25	13.5	1.2	41	68%
Baker	OR-167	29.34	18.0	0.9	53	65%
Pilgrim	OR-176	44.73	8.7	0.4	21	37%
Uncle Harvey	OR-142	6.95		1.1	11	73%
Twinings	OR-247	9.05		0.2	23	71%
Sarahs	OR-249	5.56		0.2	11	60%
Deep	OR-262	4.72		0.3	12	44%
Trout	CH-425	4.88				
Ruth	BR-209	7.52				
Rafe	BR-232	9.14				
Grassy	BR-319	13.08				
Hawksnest	HA-354	27.32				
Shoal	OR-253	8.62				
Uncle Seths	OR-264	5.37				
Bassing		9.06				
Mud		10.47				
				Mean		61%
				std dev		19%

Data sources: all areas from CCC GIS; Max Depth from MADFW or Cape Cod PALS monitoring; Volume for turnover time calculations from MADFW bathymetric maps (www.mass.gov/dfwele/dfw/dfw_pond.htm) and data developed by the Town of Orleans Planning Department ; TN concentrations for attenuation calculation from PALS monitoring and NPS-supported volunteer monitoring

The standard attenuation assumption for the ponds in the Pleasant Bay watershed was checked through the use of pond water quality information collected from the annual Cape Cod Pond and Lake Stewardship (PALS) water quality snapshot, as well as data collected by volunteers in Orleans and Brewster with the assistance of the National Park Service/Cape Cod National Seashore. The PALS Snapshot is a collaborative Cape Cod Commission/SMAST Program that allows trained, citizen volunteers in each of the 15 Cape Cod towns to collect pond samples in August and September using a standard protocol. Snapshot samples have been collected every year between 2001 and 2005. The standard protocol for the Snapshot includes field collection of dissolved oxygen and temperature profiles, Secchi disk depth readings and water samples at various depths depending on the total depth of the pond. Water samples were analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH. PALS Snapshot data is available for all but nine of the 25 ponds in the Pleasant Bay watershed. Citizens in Orleans and Brewster also collected samples during June through October, in some cases, using the same PALS protocol, but water samples were analyzed at the NPS laboratory. Table IV-6 summarizes the cumulative number of nitrogen samples available for review from both the PALS Snapshot and the NPS-supported sampling. Nitrogen attenuation estimates for the ponds reviewed in the Pleasant Bay watershed vary between 11 and 81%.

The attenuated nitrogen loads in Table IV-5 include pond attenuation based on the 50% assumption. Since each pond has this assigned attenuation factor, nitrogen loads in the Watershed Nitrogen Loading model can be subject to a number of attenuation steps as loads flow into the downgradient aquifer from one pond and then into another pond.

Buildout

Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment of potential development within the study area watershed. For the Pleasant Bay modeling, MEP staff consulted with respective town planners to determine the information that would be used in the assessment. Buildout information was submitted to MEP staff by the towns of Orleans, Harwich, and Chatham, including assistance from the Pleasant Bay Alliance. In the case of Brewster, MEP staff developed the buildout by reviewing the development potential of each property in the town's portion of the watershed. There are some differences in the assumptions used in each of these towns' buildouts (e.g., Orleans assumes that properties can have half of an additional dwelling), but they are relatively consistent for the purposes of this overall assessment. Overall, buildout additions within the Pleasant Bay watershed will increase the unattenuated loading rate by 18%.

A standard buildout assessment is to evaluate town zoning to determine minimum lot sizes in each of the zoning districts, including overlay districts (*e.g.*, water resource protection districts). Larger lots are subdivided by the minimum lot size to determine the total number of new lots and existing developed properties are reviewed for additional development potential; for example, residential lots that are twice the minimum lot size, but have only one residence. In the Brewster buildout completed by MEP staff, parcels that are classified as developable residential (state class land use codes 130 and 131) but are less than the minimum lot size and are greater than 5,000 square feet are assigned one residence in the buildout; 5,000 square feet is a common minimum buildable lot size in Cape Cod town regulations. Properties classified by the Brewster assessor as "undevelopable" (e.g., codes 132, 392, and 442) were not assigned any development at buildout; this is different than the assumption used in the Orleans buildout, for example. Commercially developable properties were not subdivided in any of the towns; the area of each parcel and the factors in Table IV-4 were used to determine a wastewater flow for these properties. All the parcels included in the buildout assessment of the Pleasant Bay watershed are shown in Figure IV-3. A nitrogen load for each additional parcel included in the buildout was developed using the factors in Table IV-4 and the cumulative unattenuated load from all additional properties at buildout is indicated in a separate column in Table IV-5.

IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT

IV.2.1 Background and Purpose

Modeling and predicting changes in coastal embayment nitrogen related water quality is based, in part, on determination of the inputs of nitrogen from the surrounding contributing land or watershed. This watershed nitrogen input parameter is the primary term used to relate present and future loads (build-out, sewering analysis, enhanced flushing, pond/wetland restoration for natural attenuation, etc.) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the sub-watersheds of the Pleasant Bay system (inclusive of Bassing Harbor, Ryders Cove and Chatham Harbors) being investigated under this nutrient threshold analysis were based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1). If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watersheds where nitrogen transport from source to estuarine waters is through groundwater flow in sandy outwash aquifers. The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, nitrogen passes through a surface water ecosystem (pond, wetland, stream) on its way to the adjoining embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) is diminished by natural biological processes that represent removal (not just temporary storage). However, this natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes. In the case of the Pleasant Bay embayment system watersheds, a portion of the freshwater flow and transported nitrogen passes through several small surface water systems (stream from Pilgrim Lake to Kescayo Gansett (Lonnies) Pond, stream into head of Paw Wah Pond and stream discharge from Tar Kiln Marsh) prior to entering the estuaries, producing the opportunity for significant nitrogen attenuation. Additionally, two small freshwater discharges (stream from Stillwater Pond to Ryders Cove and Frost Fish Creek discharging to Bassing Harbor) generate natural attenuation of nitrogen load into Ryders Cove and Bassing Harbor in the Chatham portion of Pleasant Bay.

Failure to determine the attenuation of watershed derived nitrogen overestimates the nitrogen load to receiving estuarine waters. If nitrogen attenuation is significant in one portion of a watershed and insignificant in another the result is that nitrogen management would likely be more effective in achieving water quality improvements if focused on the watershed region having unattenuated nitrogen transport (other factors being equal). In addition to attenuation by freshwater ponds (see Section IV.1.3, above), attenuation in surface water flows is also important. An example of the significance of surface water nitrogen attenuation relating to embayment nitrogen management was seen in the Agawam River, where >50% of nitrogen originating within the upper watershed was attenuated prior to discharge to the Wareham River Estuary (CDM 2001). Similarly, MEP analysis of the Quashnet River indicates that in the upland watershed, which has natural attenuation predominantly associated with riverine processes, the integrated attenuation was 39% (Howes et al. 2004). In addition, a preliminary study of Great, Green and Bournes Ponds in Falmouth, measurements indicated a 30% attenuation of nitrogen during stream transport (Howes and Ramsey 2001). An example where natural attenuation played a significant role in nitrogen management can be seen relative to West Falmouth Harbor (Falmouth, MA), where ~40% of the nitrogen discharge to the Harbor originating from the groundwater effluent plume emanating from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Similarly, the small tidal basin of Frost Fish Creek in the Town of Chatham showed ~20% nitrogen attenuation or watershed nitrogen load prior to discharge to Bassing Harbor. Clearly, proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements of upper watershed attenuation were undertaken as part of the MEP Approach. MEP conducted long-term measurements of natural attenuation relating to surface water discharges to the head of specific terminal sub-embayments in the Pleasant Bay system in addition to the natural attenuation measures by fresh kettle ponds, addressed above in Section IV.2. These additional site-specific studies were conducted in the 5 major surface water flow systems contributing nitrogen load to Pleasant Bay.

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the surfacewater flows into Pleasant Bay provides a direct integrated measure of all of the processes presently attenuating nitrogen in the sub-watersheds upgradient from the gauging sites. Flow and nitrogen load were measured at the stream gaging sites for 16 to 24 months of record (Figures IV-5, 6, 7). During the study period, velocity profiles were completed on each stream every month to two months. The summation of the products of stream subsection areas of the stream cross-section and the respective measured velocities represent the computation of instantaneous stream flow (Q).

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

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

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

where by:

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

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

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

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

Figure IV-5. Location of Stream gauge (yellow triangle) in the upper portions of the Pleasant Bay system embayment system.

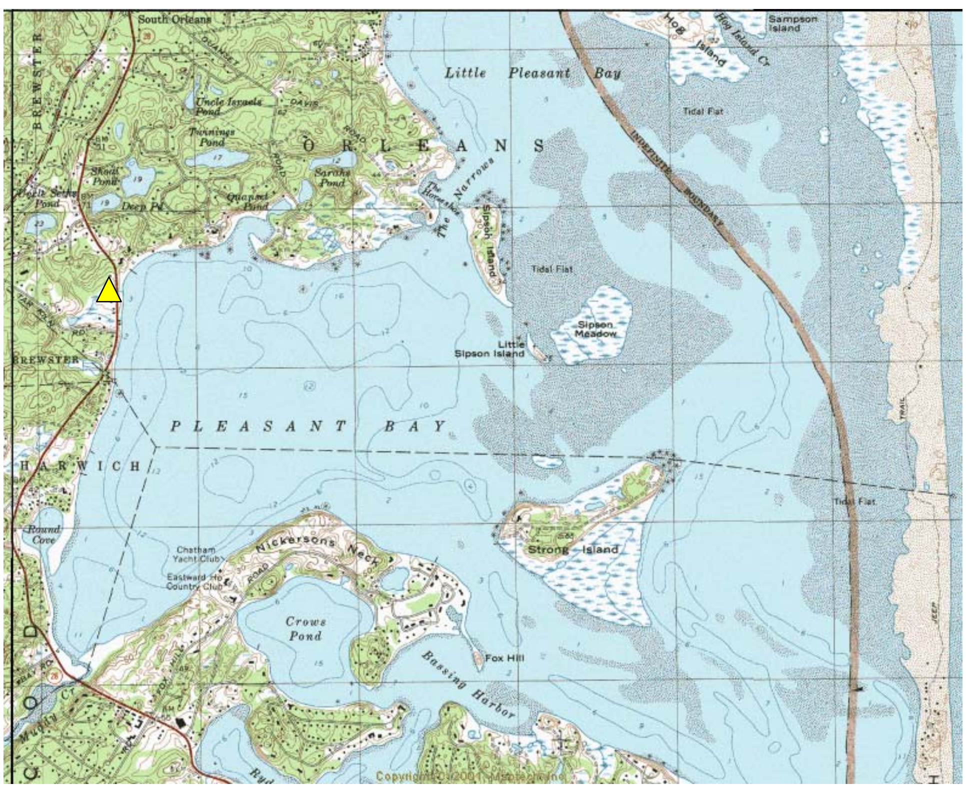


Figure IV-6. Location of Stream gauge (yellow triangle) discharging from Tar Kiln Marsh to Pleasant Bay.

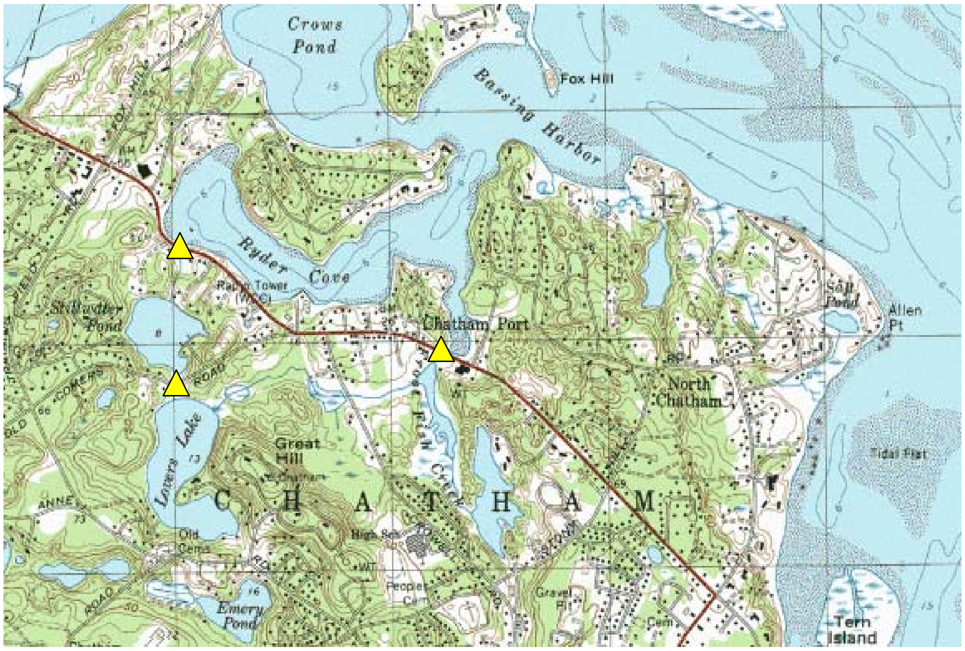


Figure IV-7. Location of Stream gauges (yellow triangle) discharging from Lovers Lake to Stillwater Pond, Stillwater Pond to Ryder Cove and Frost Fish Creek to Bassing Harbor, tributary sub-embayments to Pleasant Bay.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Stream Discharge from Pilgrim Lake to Kescayo Gansett (Lonnies) Pond – Upper Pleasant Bay

Pilgrim Lake located upgradient of the stream gauge site is a small pond on Cape Cod and unlike many of the freshwater ponds, this pond has stream outflow rather than discharging solely to the aquifer along its down-gradient shore. This stream outflow may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation. In addition, nitrogen attenuation also occurs within the wetlands and streambed, however this is limited in the case of this surfacewater feature as large parts of it are a manmade herring run. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to Pilgrim Lake above the gauge site (Figure IV-5) and the measured annual discharge of nitrogen to Kescayo Gansett Pond at the gauge site, (Figure IV-8).

At the Pilgrim Lake gauge site, a continuously recording vented calibrated water level gauge was installed to yield the level of water in the herring run that carries the flows and associated nitrogen load to the head of Kescayo Gansett Pond. The gauge was placed at the down gradient end of the herring run but immediately upgradient of the culvert that passes underneath Herring Brook Way prior to discharging to Kescayo Gansett Pond. As the herring brook is tidally influenced the gauge was located above the saltwater reach such that freshwater flow could be measured without tidal influence. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average low tide salinity was determined to be 0.3 ppt therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked monthly. The gauge on the herring brook was installed on June 28, 2002 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until March 14, 2004 for a total deployment of 21 months. The 12-month uninterrupted record (October 11, 2002 to October 10, 2003) used in this analysis encompasses the summer 2003 field season.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the herring brook gauge site based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allowed for the determination of nitrogen mass discharge to Kescayo Gansett Pond (Figure IV-8 and Table IV-7). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gauge site.

The annual freshwater flow record for the herring brook measured by the MEP (Table IV-8), was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from the herring brook was 8% of the long-term average modeled flows. Therefore, the watershed and river datasets appear to be in balance.

Total nitrogen concentrations within the herring brook outflow were relatively high, 0.796 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 0.78 kg/day and a measured total annual TN load of 285 kg/yr. In the herring brook, nitrate was not the predominant form of nitrogen (24%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was well taken up by plants within the pond or stream ecosystems. The relatively low concentration of inorganic nitrogen (compared to other streams on Cape Cod) in the outflowing stream waters also suggests biological cycling of nitrogen within the upgradient freshwater ecosystems is converting inorganic to organic forms, with the associated effect that nitrogen removal by the pond system is being enhanced.

From the measured nitrogen load discharged by the herring brook to Kescayo Gansett (Lonnies) Pond and the nitrogen load determined from the watershed based land use analysis, it appears that there is nitrogen attenuation of upper watershed derived nitrogen during transport to the Pond and ultimately the upper portion of Pleasant Bay. Based upon lower nitrogen load (285 kg yr^{-1}, 0.78 kg d^{-1}) discharged from the freshwater herring brook compared to that added by the various land-uses to the associated watershed (2.62 kg d^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands is 70% (i.e. 70% of nitrogen input to watershed does not reach the estuary). This level of attenuation is also greater than the integrated attenuation rate determined from the watershed nitrogen model of 48% (Table IV-5). This is expected given the conservative assumptions of nitrogen attenuation used in the model. The directly measured nitrogen loads from the river was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

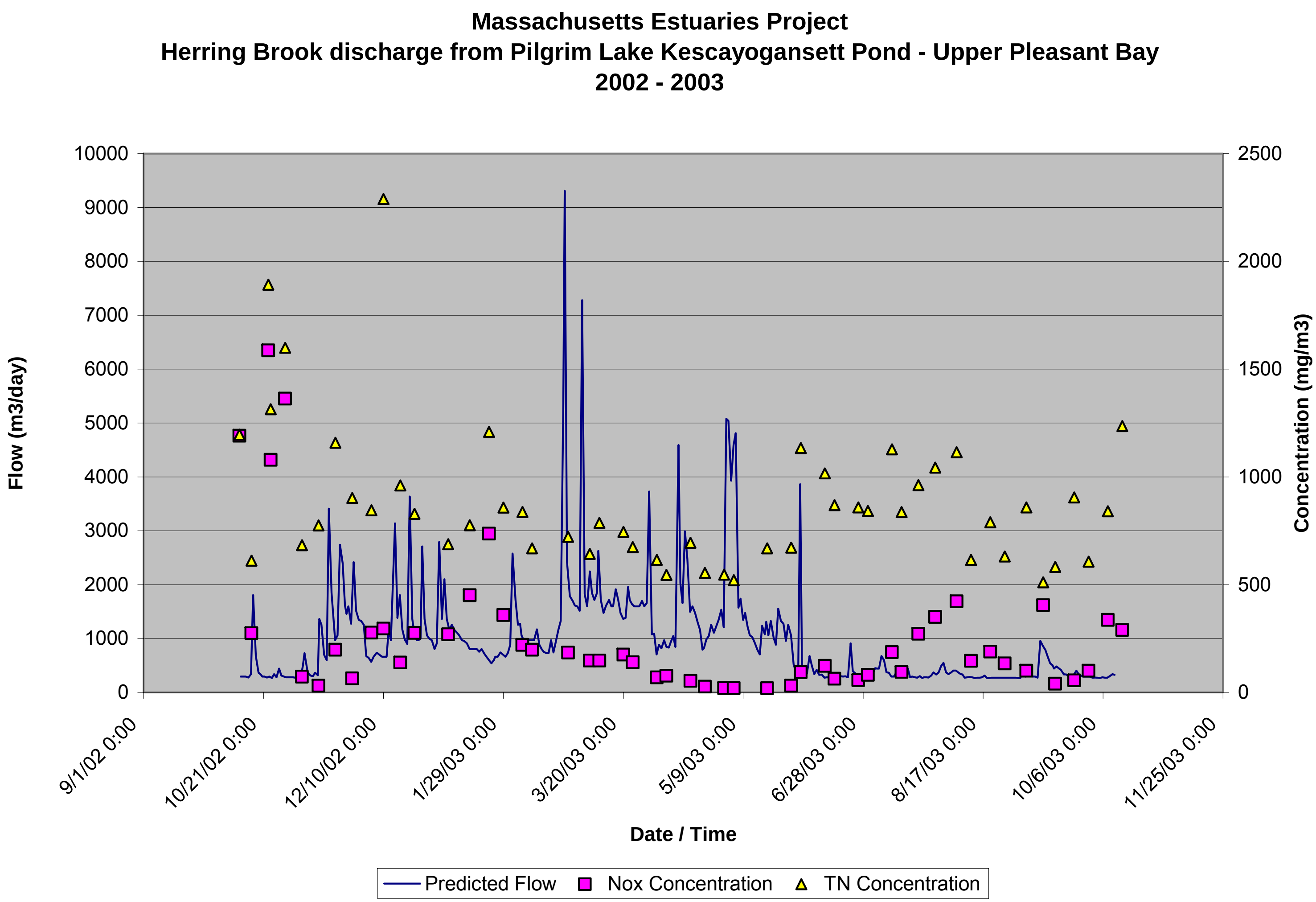


Figure IV-8. Stream discharge from Pilgrim Lake to Kescayo Gansett Pond (solid blue line), total nitrogen (yellow triangle) and nitrate+nitrite (pink box) concentrations for determination of annual volumetric discharge and nitrogen load from the subwatershed to upper Pleasant Bay system (Table IV-7).

Table IV-7. Comparison of water flow and nitrogen discharges from streams (freshwater) discharging to Pleasant Bay. The "Stream" data is from the MEP stream gauging effort. Watershed data is based upon the MEP watershed modeling effort by USGS.

Stream Discharge Parameter	Stream Discharge to Kescayo Gansett Pond[d]	Stream Discharge to Paw Wah Pond[e]	Tar Kiln Marsh Discharge Pleasant Bay[f]	Data Source
Total Days of Record	365[a]	365[b]	365[c]	(1)
Flow Characteristics				
Stream Average Discharge (m3/day)	981	388	2763	(1)
Contributing Area Average Discharge (m3/day)	1066	271	2525	(2)
Stream 2002-03 vs. Long-term Discharge (% difference)	8%	43%	9%	
Nitrogen Characteristics				
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.19	0.191	0.352	(1)
Stream Average Total N Concentration (mg N/L)	0.796	1.618	0.687	(1)
Nitrate + Nitrite as Percent of Total N (%)	24%	12%	51%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	0.78	0.63	1.90	(1)
TN Average Contributing Area Attenuated Load (kg/day)	1.77	--	--	(2)
TN Average Contributing UN-attenuated Load (kg/day)	2.62	1.94	6.19	(3)
Attenuation of Nitrogen in Pond/Stream (%)	70%	60%	69%	(4)

(a) from October 11, 2002 to October 10, 2003
(b) from September 5, 2002 to September 4, 2003
(c) from May 25, 2004 to May 24, 2005
(d) Flow and N load to creek discharging into Kescayogansett Pond includes Pilgrim Lake contributing area.
(e) Flow and N load to stream discharging to Paw Wah Pond includes cranberry bog contributing area.
(f) Flow and N load to stream discharging from Tar Kiln Marsh to Pleasant Bay includes the Tar Kiln Marsh contributing areas.

(1) MEP gage site data
(2) Contributing area discharge for each of the the three surface water features is based on a recharge rate of 27.25 inches over the sub-watershed are to each stream
(3) As in footnote (2), with the addition of pond and stream conservative attentuation rates.
(4) Calculated based upon the measured TN discharge from the rivers vs. the unattenuated watershed load.

Table IV-8. Summary of annual volumetric discharge and nitrogen load from streams (freshwater) discharging to the head of Kescayo Gansett Pond, Cranberry Bog to Paw Wah Pond, and Tar Kiln Marsh discharging to Pleasant Bay based upon the data presented in Figures IV-9, 10, and 11 and Table IV-7.

SYSTEM	PERIOD	DISCHARGE (m3/yr)	LOAD (Kg/yr)	
			Nox	TN
Herring Brook from Pilgrim Lake	October 11, 2002 to October 10, 2003	**358169**	**68**	**285**
Stream from Bog to Paw Wah Pond	September 5, 2002 to September 4, 2003	**141522**	**27**	**229**
Stream from Tar Kiln Marsh	May 25, 2004 to May 24, 2005	**1008446**	**355**	**693**

IV.2.3 Surface water Discharge and Attenuation of Watershed Nitrogen: Stream Discharge from Cranberry Bog to Paw Wah Pond – Upper Pleasant Bay

The cranberry bog located immediately upgradient of the Paw Wah Pond gauge site is a small bog and like many of the bogs on the Cape, this bog has measurable stream outflow. This stream outflow, creek to Paw Wah Pond, may serve to decrease natural attenuation of nitrogen as would be seen in a stream coming from a natural pond, however it still provides for a direct measurement of the nitrogen attenuation occurring in the small sub-watershed to Paw Wah Pond. In addition, nitrogen attenuation also occurs within the wetlands and streambed associated with the creek though these attenuation process may be small in the case of this creek due to its small size. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the creek above the gauge site (Figure IV-5) and the measured annual discharge of nitrogen to the tidal portion of Paw Wah Pond, (Figure IV-9).

At the Paw Wah Pond gauge site, a continuously recording vented calibrated water level gauge was installed to yield the level of freshwater in the creek (at low tide) that carries the flows and associated nitrogen load to Paw Wah Pond and ultimately upper Pleasant Bay. As the creek flowing out of the cranberry bog is tidally influenced the gauge was located above the saltwater reach such that freshwater flow could be measured at low tide without tidal influence. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average low tide salinity was determined to be 3.7 ppt therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked monthly. The gauge on the creek to Paw Wah Pond was installed on June 29, 2002 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until April 6, 2004 for a total deployment of 22 months. The 12-month uninterrupted record used in this analysis encompasses the summer 2003 field season.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Paw Wah Pond creek site based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly at low tide for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allowed for the determination of nitrogen mass discharge to Paw Wah (Figure IV-9 and Table IV-7). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gauge site.

The annual freshwater flow record for the creek into Paw Wah Pond measured by the MEP, was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from the creek was 43% larger than the long-term average modeled flows. However, the projected and measured flows are very small in this creek, 3 to 9 fold lower than the other streams measured. In addition, alterations in flow may have occurred due to manipulation of control structures upgradient.

Total nitrogen concentrations within the creek outflow to Paw Wah Pond were relatively high, 1.62 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 0.63 kg/day and a measured total annual TN load of 229 kg/yr. In the creek discharge to Paw Wah

Pond, nitrate was not the predominant form of nitrogen (12%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the upgradient cranberry bog and to the river was almost completely taken up by plants within the bog or creek ecosystems. As for Lonnies Pond stream, the low concentration of inorganic nitrogen and its small proportion relative to the total nitrogen pool suggests that nitrogen cycling in the upgradient systems is converting inorganic nitrogen to organic forms increasing the potential for removal prior to discharge.

From the measured nitrogen load discharged by the creek to Paw Wah Pond and the nitrogen load determined from the watershed based land use analysis, it appears that there is nitrogen attenuation of upper watershed derived nitrogen during transport to the Bay. Based upon lower nitrogen load (229 kg yr^{-1}, 0.63 kg d^{-1}) discharged from the freshwater creek to Paw Wah Pond compared to that added by the various land-uses to the associated watershed (710 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands is 60% (i.e. 60% of nitrogen input to watershed does not reach the estuary; Table IV-5). The directly measured nitrogen loads from the creek was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

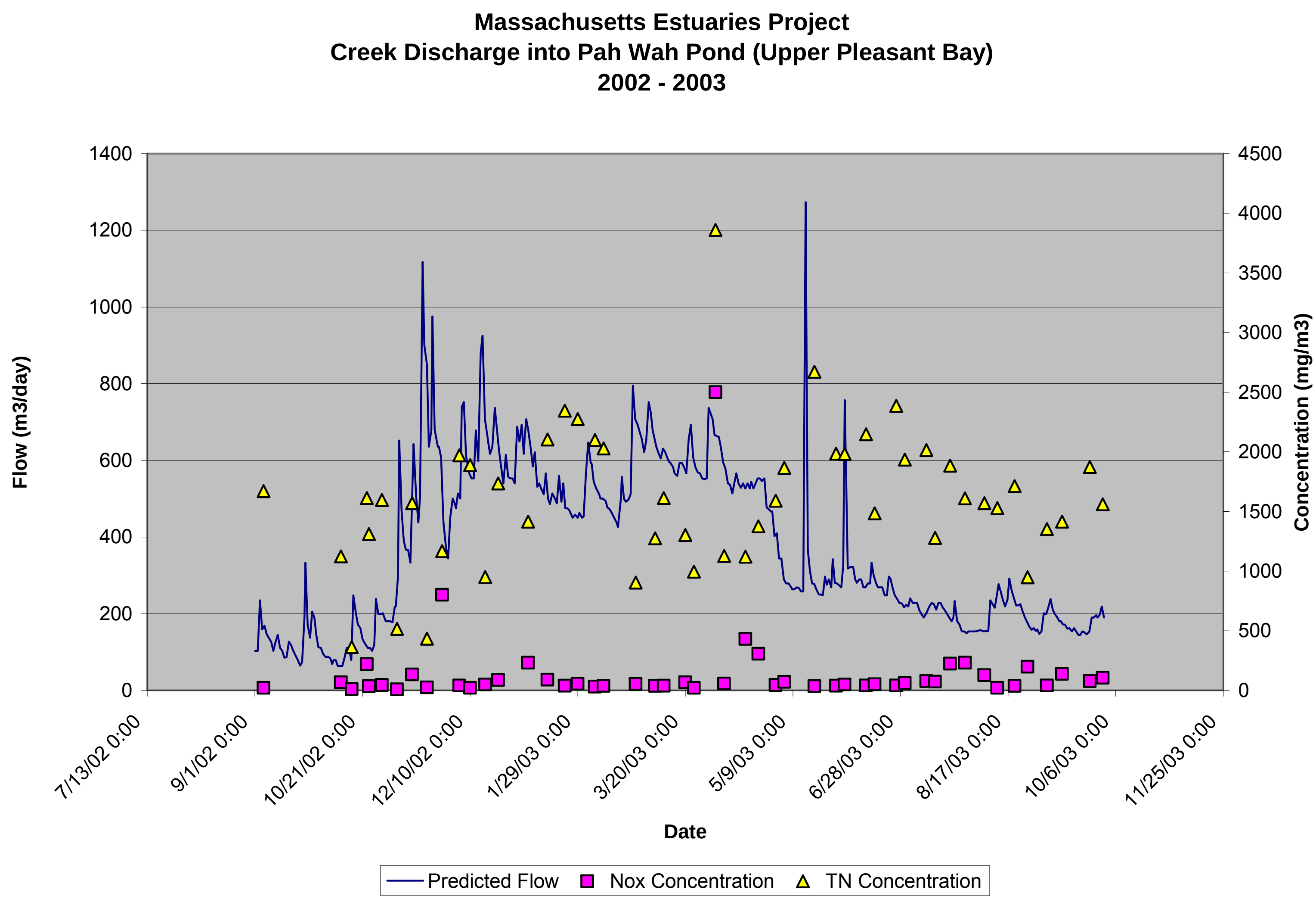


Figure IV-9. Stream discharge from Cranberry Bog to Paw Wah Pond (solid blue line), total nitrogen (yellow triangle) and nitrate+nitrite (pink box) concentrations for determination of annual volumetric discharge and nitrogen load from the subwatershed to Paw Wah Pond discharging to upper Pleasant Bay system (Table IV-7).

IV.2.4 Surface water Discharge and Attenuation of Watershed Nitrogen: Stream Discharge from Tar Kiln Marsh to Pleasant Bay

Tar Kiln Marsh located upgradient of the gauge site is a small salt marsh on Cape Cod and like many of the salt marsh systems on the Cape, this marsh has a stream bed that is at a high enough elevation relative to the Bay that freshwater outflow can be measured at low tide. This stream outflow may serve to decrease the marsh attenuation of nitrogen due the quicker transit time of water through the marsh, however it provides for a direct measurement of the nitrogen attenuation occurring in the marsh systems and upgradient subwatershed. In addition, nitrogen attenuation also occurs within the streambed associated with the stream draining the salt marsh. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the salt marsh creek above the gauge site (Figure IV-6) and the measured annual discharge of nitrogen to the gauge, (Figure IV-10).

At the Tar Kiln Marsh gauge site, a continuously recording vented calibrated water level gauge was installed to yield the level of freshwater in the tidal creek (at low tide) that carries the flows and associated nitrogen load to Pleasant Bay. As the tidal creek flowing out of Tar Kiln Marsh is tidally influenced the gauge was located above the saltwater reach such that freshwater flow could be measured at low tide without tidal influence. To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average low tide salinity was determined to be 5.1 ppt therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked monthly. The gauge on the Tar Kiln Marsh tidal creek was installed on July 2, 2003 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until May 30, 2005 for a total deployment of 22 months. The 12-month uninterrupted record (May 25, 2004 to May 24, 2005) used in this analysis encompasses the summer 2004 field season.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the tidal creek gauge site based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allowed for the determination of nitrogen mass discharge to Pleasant Bay from Tar Kiln Marsh and its associated tidal creek (Figure IV-10 and Table IV-7). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gauge site.

The annual freshwater flow record for the Tar Kiln Marsh tidal creek measured by the MEP, was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from the tidal creek was within 9% of the long-term average modeled flows. Therefore, the watershed and river datasets appear to be in balance.

Total nitrogen concentrations within the Tar Kiln Marsh tidal creek outflow were relatively high, 0.687 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 1.90 kg/day and a measured total annual TN load of 693 kg/yr. In the tidal creek, nitrate was the predominant form of nitrogen (51%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the marsh and to the tidal creek was not completely taken up by plants within the marsh or creek ecosystems. The high concentration of inorganic nitrogen in the outflowing creek waters also suggests that plant production within the upgradient freshwater ecosystems is not nitrogen limited.

From the measured nitrogen load discharged by the Tar Kiln Marsh tidal creek to Pleasant Bay and the nitrogen load determined from the watershed based land use analysis, it appears that there is nitrogen attenuation of upper watershed derived nitrogen during transport to the Bay. Based upon lower nitrogen load (764 kg yr^{-1}) discharged from the freshwater flow in the tidal creek compared to that added by the various land-uses to the associated watershed (2258 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands is 69% (i.e. 69% of nitrogen input to watershed does not reach the estuary; Table IV-5). The directly measured nitrogen loads from the river was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

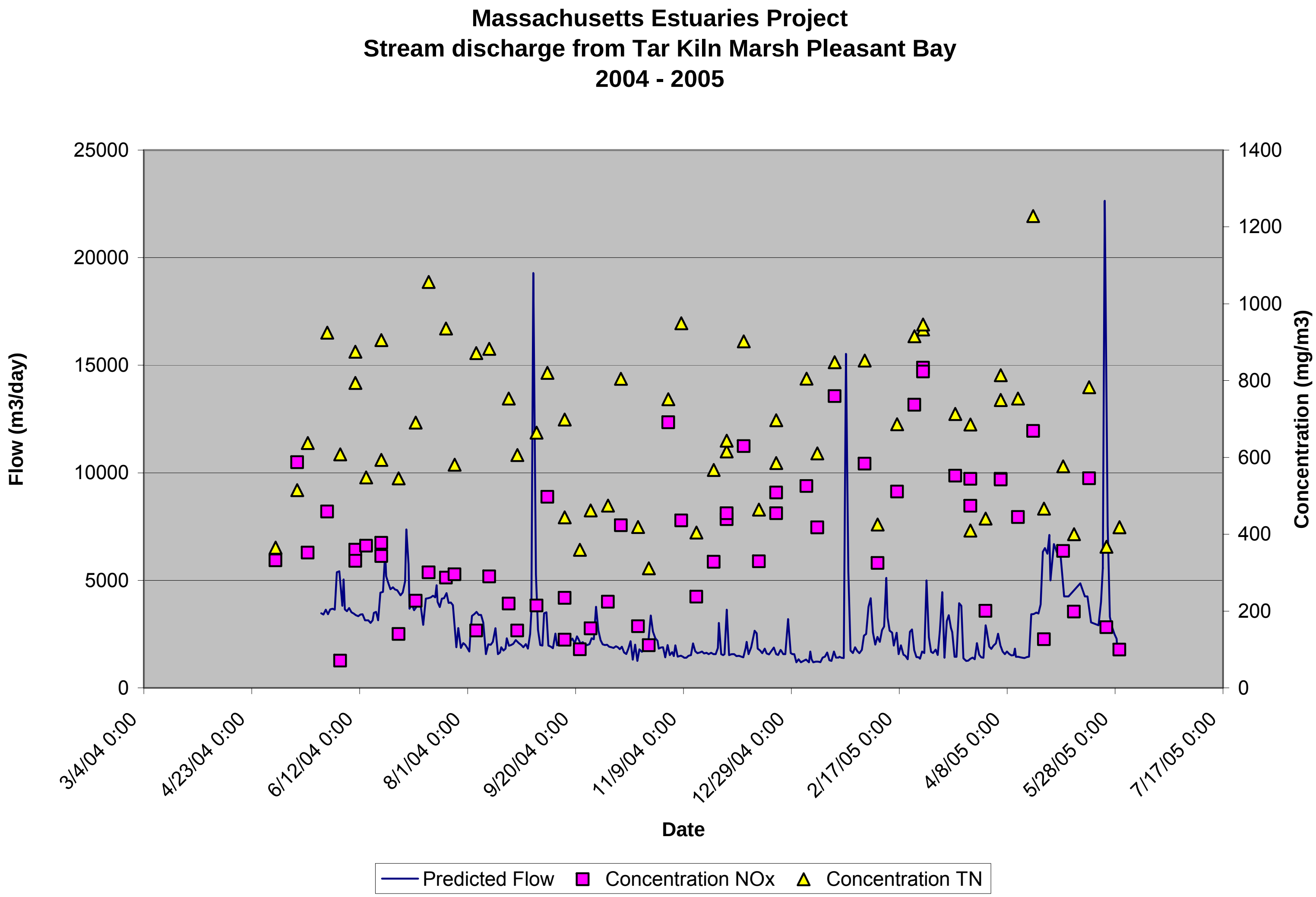


Figure IV-10. Stream discharge from Tar Kiln Marsh to Pleasant Bay (solid blue line), total nitrogen (yellow triangle) and nitrate+nitrite (pink box) concentrations for determination of annual volumetric discharge and nitrogen load from the subwatershed to Tar Kiln Marsh discharging to the Pleasant Bay system (Table IV-7).

IV.2.5 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-7). 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-12, IV-13, Table IV-8). 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 (2058 g N d^{-1}) compared to Lovers Lake (1532 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-8). The slightly lower measured discharge likely results from the lower than average groundwater levels during the study period (Figure IV-11). 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 (1532 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 range determined from the pond survey method (see above) and consistent with use of a 50% 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.

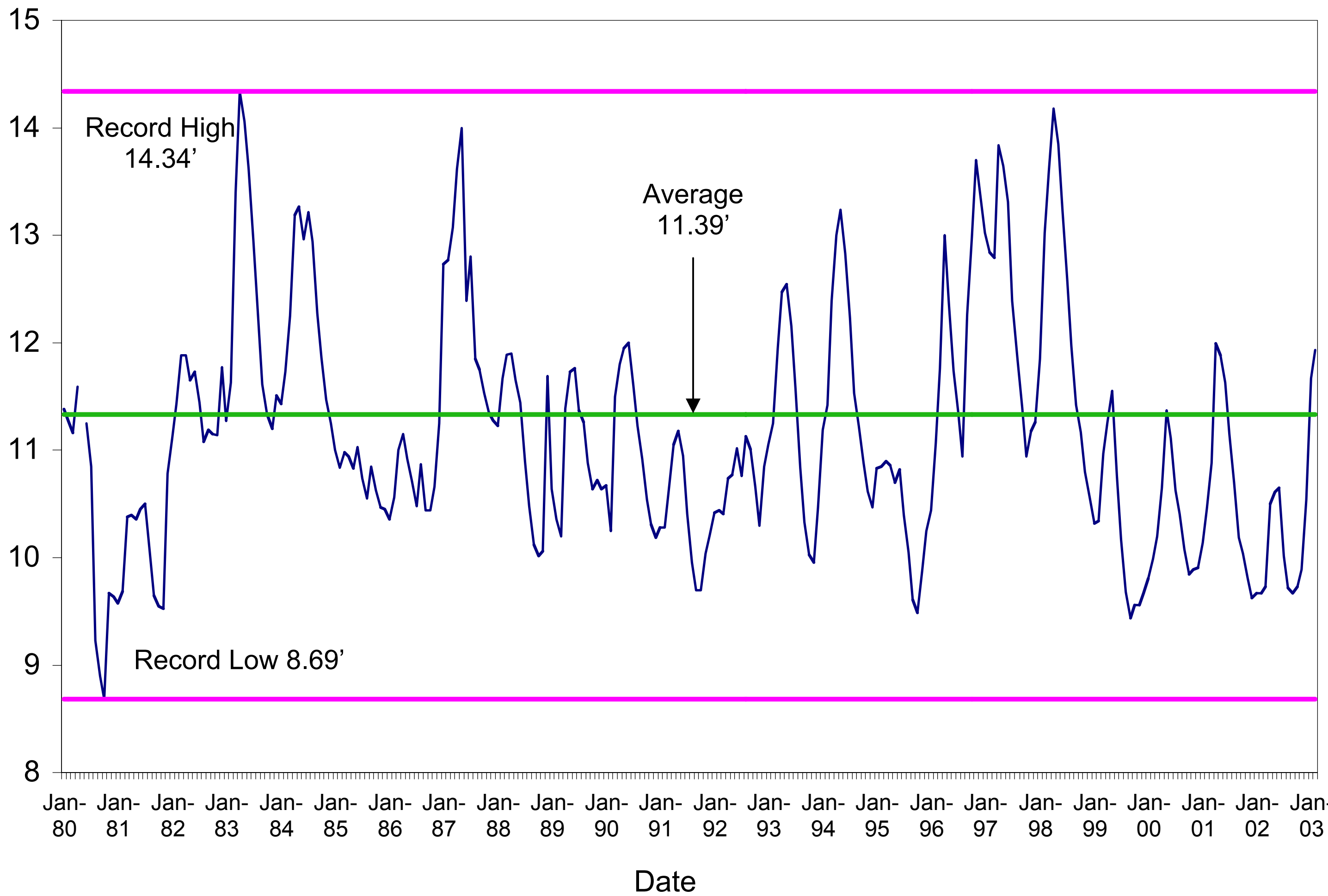


Figure IV-11. CGW138 Hydrograph. Trace indicates the water table elevation at the well site from 1980-2002.

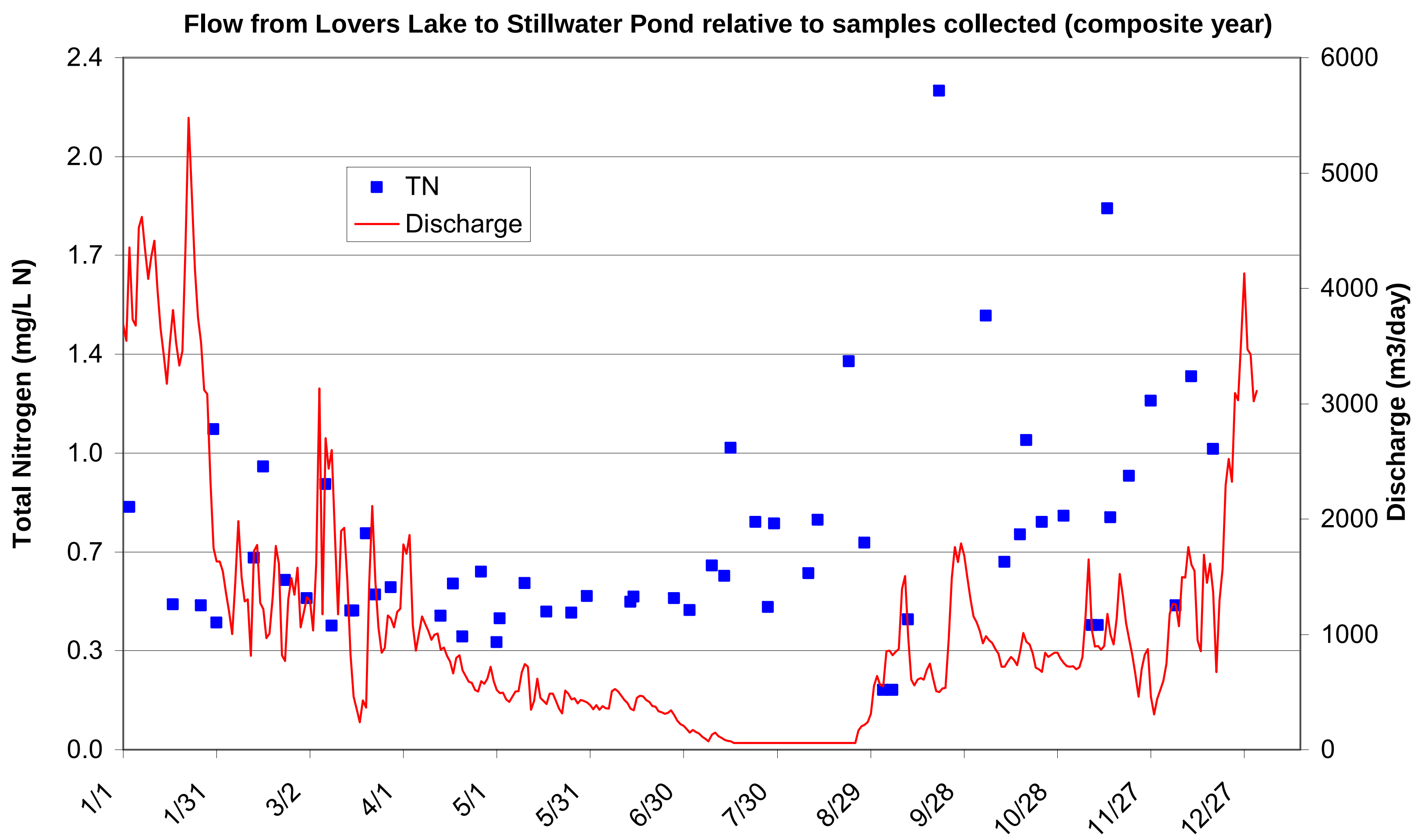


Figure IV-12. Annual composite developed from a stream gauge maintained in the outflow stream from Lovers Lake discharging to Stillwater Pond. Nutrient samples were collected approximately weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow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

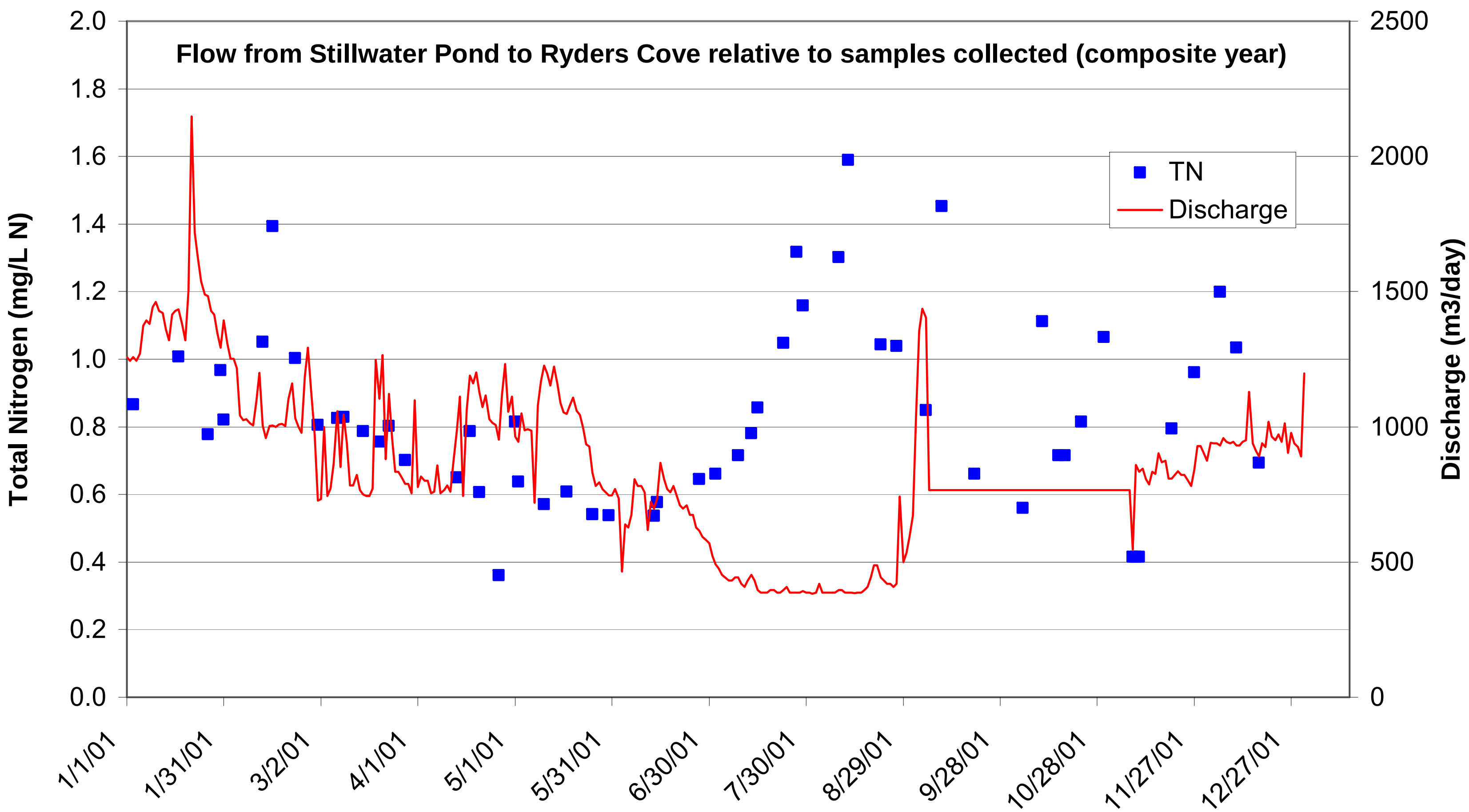


Figure IV-13. Annual composite developed from a stream gauge maintained in the outflow stream from Stillwater Pond discharging to Ryders Cove. Nutrient samples were collected approximately weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow and total nitrogen transport for determining nitrogen attenuation (see Table IV-5).

Table IV-9. 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)	2058*	1532	(2)
Proportion Total Nitrogen Stream vs. Contributing Area (%)	N/A	47%	
Attenuation (Total) of Nitrogen in Pond/Stream (%)	7%**	52%	

[a] from 11/8/00 to September 2002 (Ryder gauge) and December 2002 (Stillwater Pond gauge)
[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
* using watershed model, measured load from Lovers Lake, and correcting to 91% watershed discharge
** 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.

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 7% for Stillwater Pond. Note that it 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. 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.2.6 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-7) 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.

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-14, 15, 16, 17). 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-10). 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 initially calculated during the MEP nutrient threshold analysis for Chatham embayments including Frost Fish Creek in order to make the comparison with watershed nitrogen load for determination of natural attenuation potential in Frost Fish Creek. Nitrogen mass on inflowing and outgoing tides 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-10). 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 and the observed net loss of nitrogen to lower Ryder Cove. At the time of the initial analysis conducted during the development of the MEP Chatham threshold report a comparison of 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 $tide^{-1}$ was undertaken. As a result, natural attenuation of watershed derived nitrogen within Frost Fish Creek was determined to be 19%. Further refinement of the water use data set for Chatham resulted in a slightly lower watershed nitrogen load to Frost Fish Creek indicating that the watershed nitrogen load to the Frost Fish Creek system equaled what was measured to be leaving during the tidal flux studies performed in the summer of 2002. As such, though Frost Fish Creek was previously assigned a 19 percent attenuation value for the initial MEP threshold analysis of Bassing Harbor, for the purpose of the analysis for Pleasant Bay the MEP technical Team thought it more prudent and conservative to assign zero natural attenuation to Frost Fish Creek.

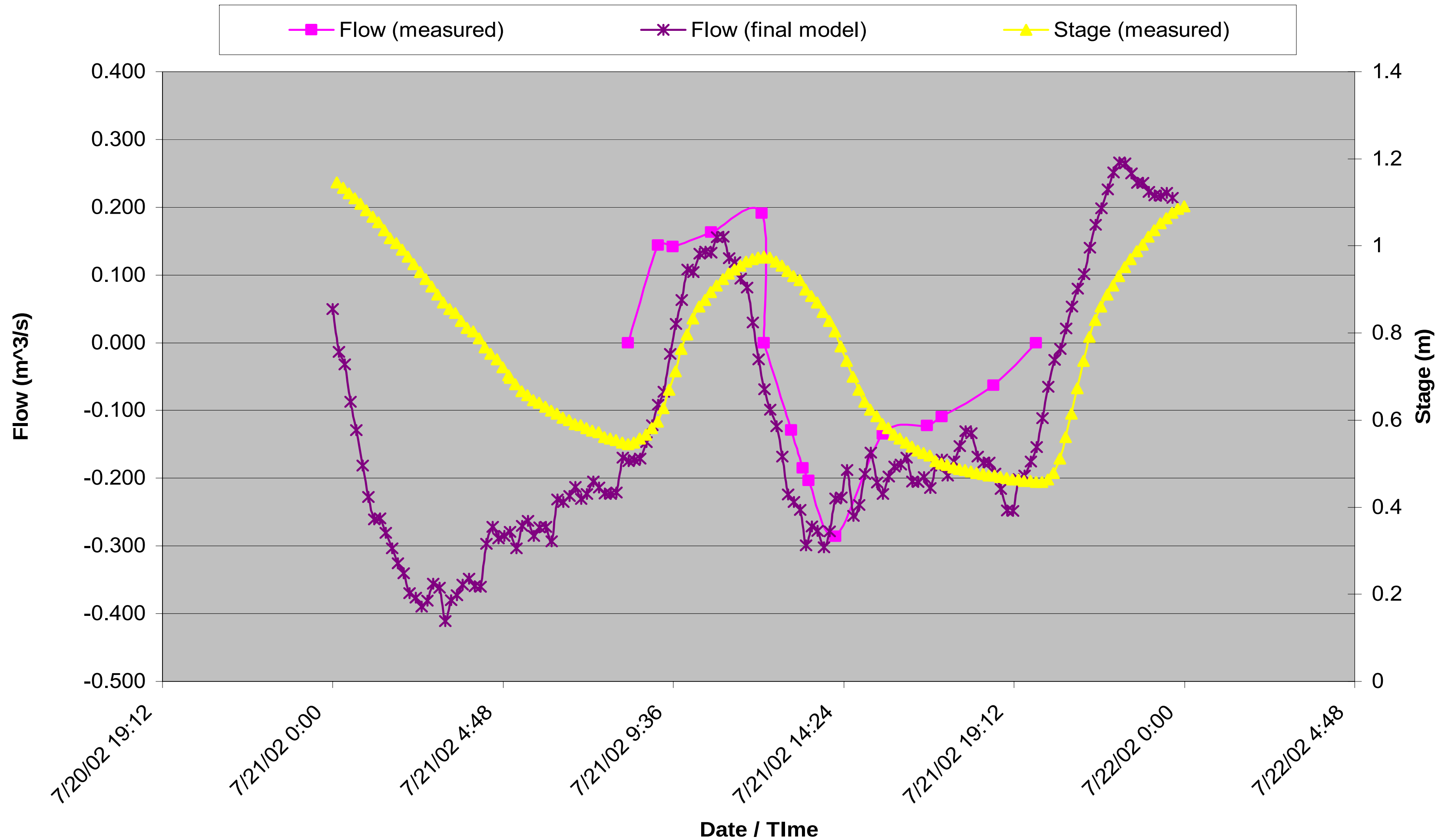


Figure IV-14. Frost Fish Creek Tidal Study 1 (July 21, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

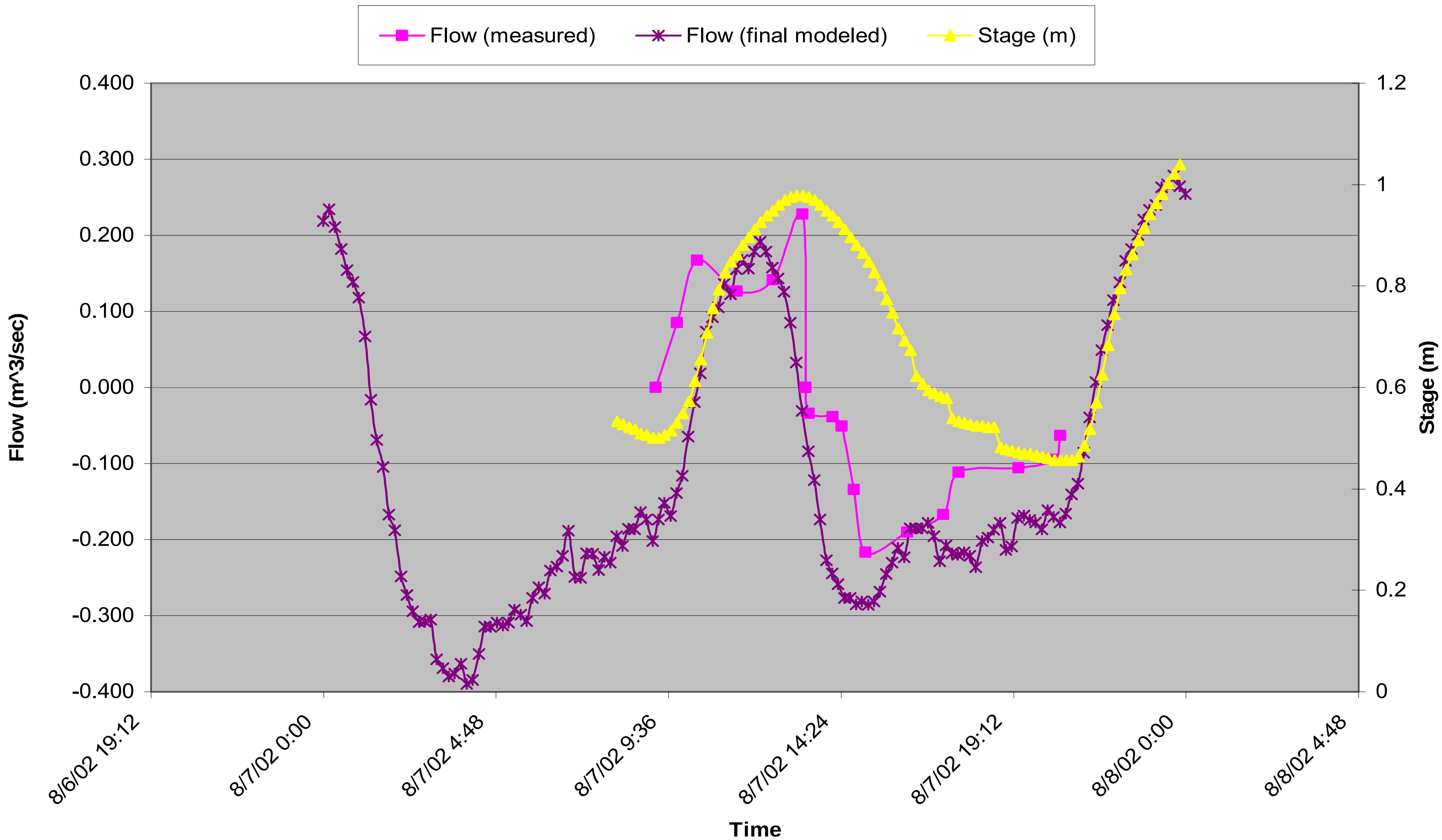


Figure IV-15. Frost Fish Creek Tidal Study 2 (August 8, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

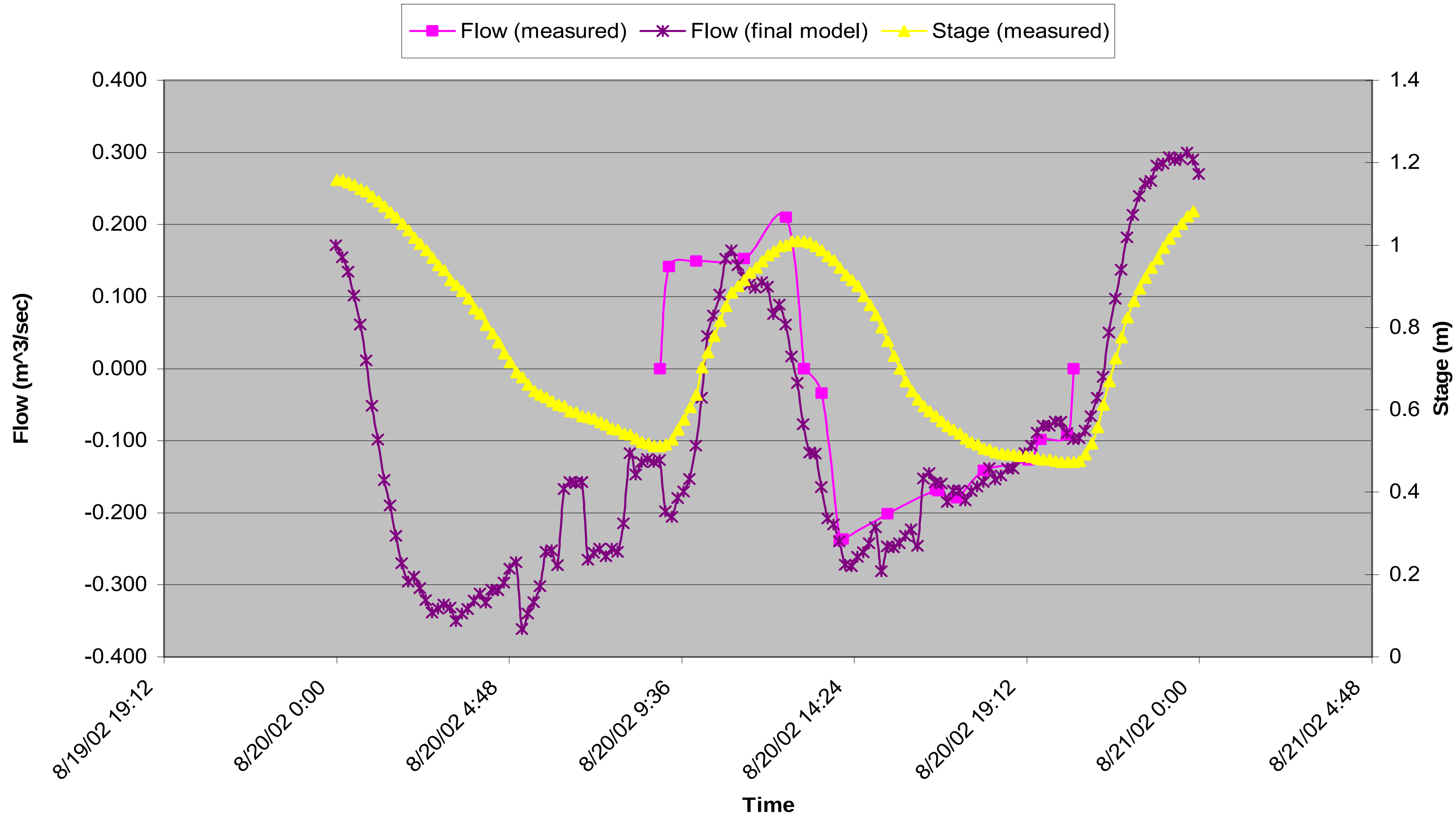


Figure IV-16. Frost Fish Creek Tidal Study 3 (August 20, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

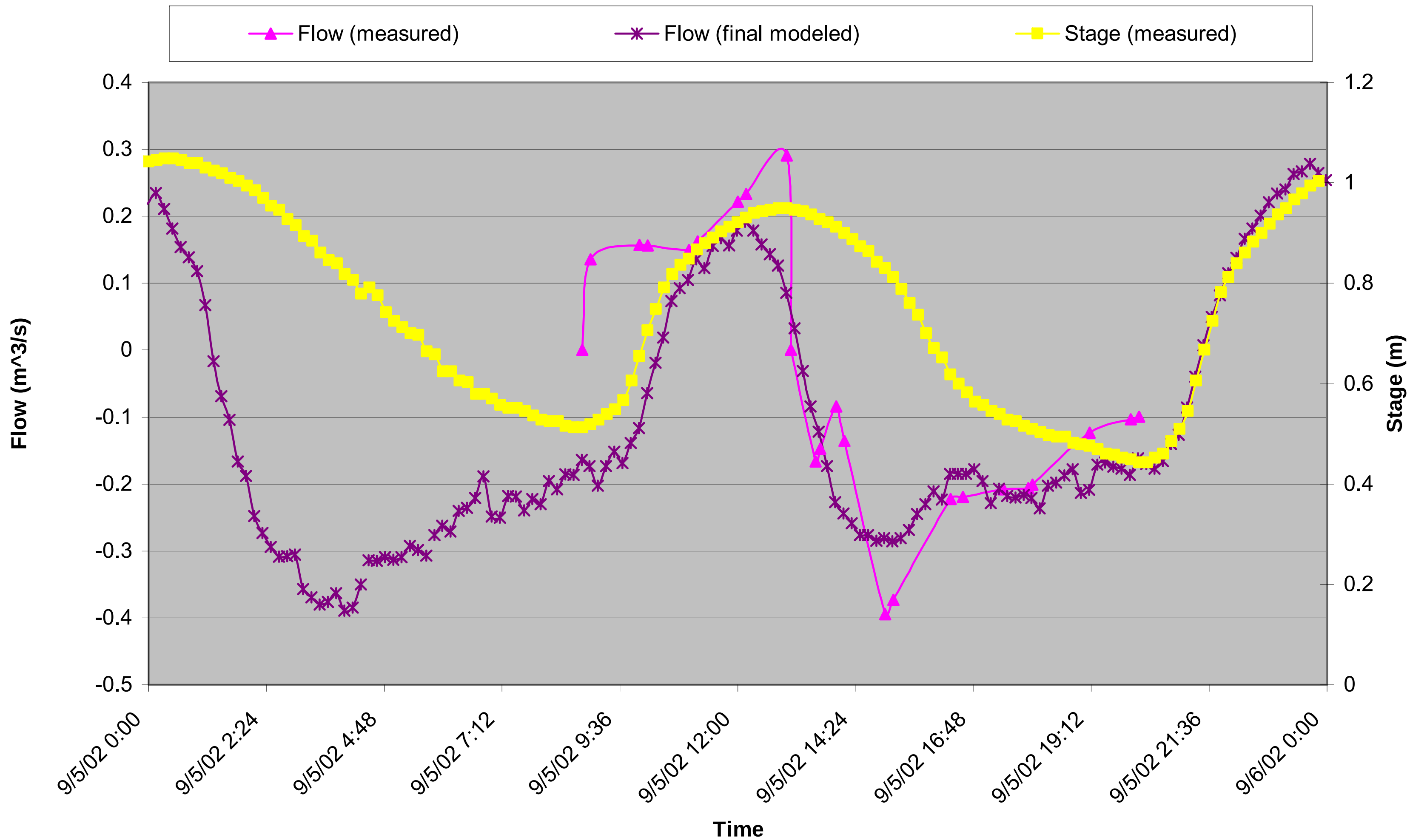


Figure IV-17. Frost Fish Creek Tidal Study 4 (September 5, 2002). Comparison of measured and modeled tidal flow and measured tidal elevation.

Table IV-10. 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 (Chatham MEP Report)					**2.24**				
Attenuation (calculated)					**19%**				
Total Land + Atmos. Inputs N kg/tide (Pleasant Bay MEP Report)					**1.50**				
Attenuation (calculated)					**0 %**				

This finding does differ from fully tidal (i.e. creeks empty at low tide) salt marsh creeks, for example 40% attenuation observed in the Mashapaquit Creek Marsh in the West Falmouth Harbor System (Howes and Smith, 1999). However, the Frost Fish Creek basin appears to act more as a salt marsh pond, than a tidal creek. The impounding of water 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. In summary, the mass of nitrogen entering lower Ryder Cove from Frost Fish Creek is approximately equal to the nitrogen load calculated from the sub-watershed land use analysis conducted for the Pleasant Bay analysis based on refined water use data.

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

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

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

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

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

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bioavailable nitrogen is returned to the embayment water column for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that have been investigated by SMAST and the MEP, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. 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 Pleasant Bay system, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected from 62 sites in Upper Pleasant Bay, Pleasant Bay (inclusive of the Bassing Harbor sub-embayment and Muddy Creek, previously sampled in 2003) and in Chatham Harbor. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample.

Rates of nitrogen release were determined using undisturbed sediment cores incubated for 24 hours in temperature-controlled baths. Sediment cores (15 cm inside diameter) were collected by SCUBA divers and cores transported by small boat to a shore side field lab. Cores were maintained from collection through incubation at *in situ* temperatures. Bottom water was collected and filtered from each core site to replace the headspace water of the flux cores prior to incubation. The number of core samples from each site (see Figures IV-18 and 19) per incubation were as follows:

Upper Pleasant Bay Benthic Nutrient Regeneration Cores

- Station PB-22/23 2 cores (Upper Region)
- Station PB-24 1 core (Upper Region)
- Station PB-25 1 core (Upper Region)
- Station PB-26 1 core (Upper Region)
- Station PB-27 1 core (Upper Region)
- Station PB-28 1 core (Upper Region)
- Station PB-29 1 core (Upper Region)
- Station PB-36 1 core (Upper Region)
- Station PB-38 1 core (Upper Region)
- Station PB-39 1 core (Upper Region)
- Station PB-40 1 core (Upper Region)
- Station PB-41 1 core (Upper Region)
- Station PB-42 1 core (Upper Region)
- Station PB-43 1 core (Upper Region)
- Station PB-44 1 core (Upper Region)
- Station PB-45 1 core (Upper Region)
- Station PB-46 1 core (Upper Region)
- Station PB-47 1 core (Upper Region)
- Station PB-48 1 core (Upper Region)

- Station PB-49 1 core (Upper Region)
- Station PB-50 1 core (Upper Region)
- Station PB-51 1 core (Upper Region)
- Station PB-52 1 core (Upper Region)
- Station PB-53 1 core (Upper Region)
- Station PB-54 1 core (Upper Region)
- Station PB-56/57 2 cores (Upper Region)
- Station PB-58 1 core (Upper Region)
- Station PB-59 1 core (Upper Region)
- Station PB-60 1 core (Upper Region)
- Station PB-61 1 core (Upper Region)
- Station PB-62 1 core (Upper Region)

Pleasant Bay Benthic Nutrient Regeneration Cores

- Station PB-1 1 core (Middle Region)
- Station PB-2 1 core (Middle Region)
- Station PB-3 1 core (Middle Region)
- Station PB-4 1 core (Middle Region)
- Station PB-5 1 core (Middle Region)
- Station PB-6 1 core (Middle Region)
- Station PB-7 1 core (Middle Region)
- Station PB-8 1 core (Middle Region)
- Station PB-13 1 core (Middle Region)
- Station PB-14 1 core (Middle Region)
- Station PB-15 1 core (Middle Region)
- Station PB-16 1 core (Middle Region)
- Station PB-17 1 core (Middle Region)
- Station PB-18 1 core (Middle Region)
- Station PB-19 1 core (Middle Region)
- Station PB-20/21 2 core (Middle Region)
- Station PB-30 1 core (Middle Region)
- Station PB-31 1 core (Middle Region)
- Station PB-32 1 core (Middle Region)
- Station PB-33 1 core (Middle Region)
- Station PB-34 1 core (Middle Region)
- Station PB-35 1 core (Middle Region)
- Station PB-37 1 core (Middle Region)
- Station PB-55 1 core (Middle Region)

Chatham Harbor Benthic Nutrient Regeneration Cores

- Station PB-9 1 core (Lower Region)
- Station PB-10 1 core (Lower Region)
- Station PB-11 1 core (Lower Region)
- Station PB-12 1 core (Lower Region)

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

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

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

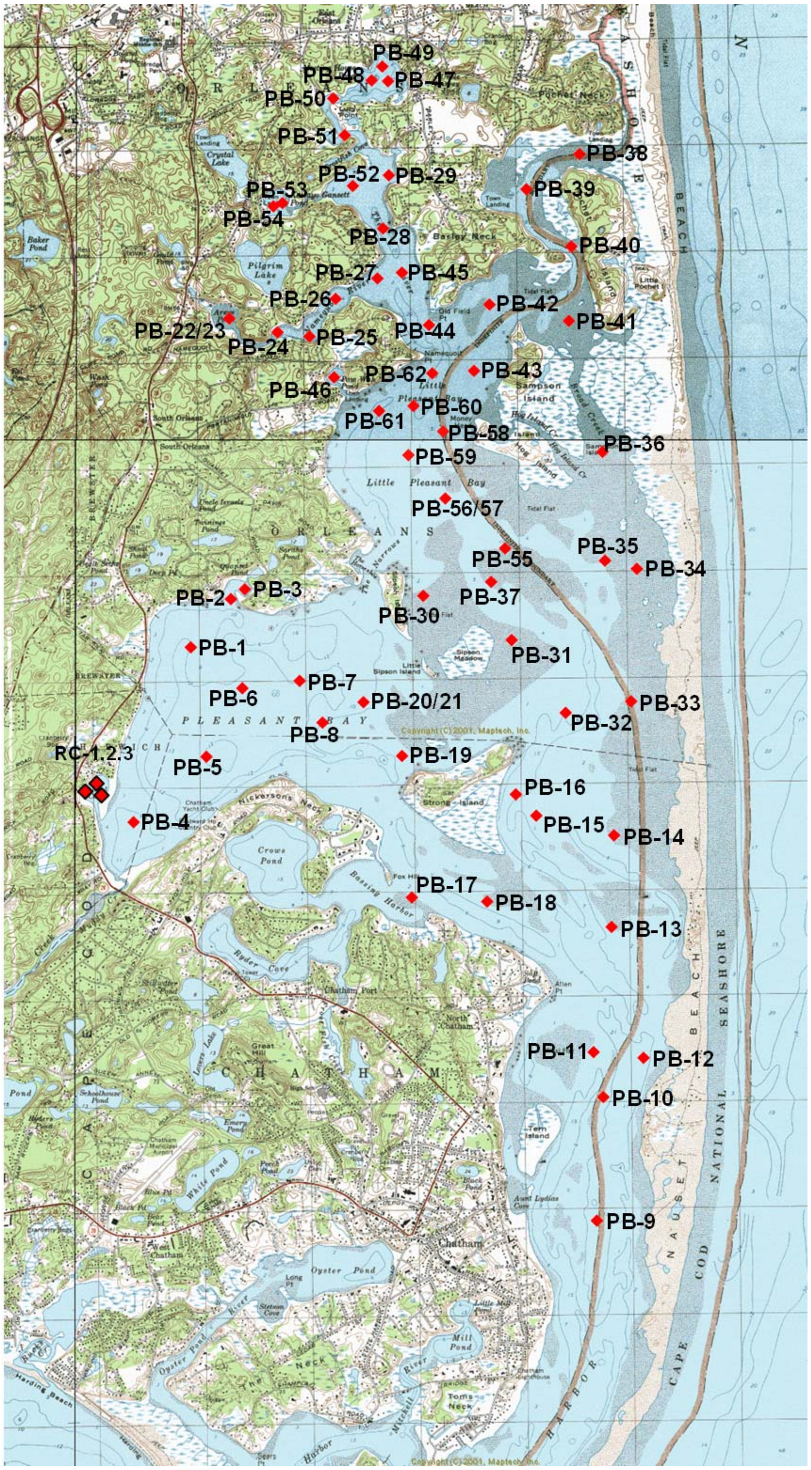


Figure IV-18. Pleasant Bay embayment system sediment sampling sites (red diamonds) for determination of nitrogen regeneration rates. Numbers are for reference to list of core stations above. See also Figure IV-19 for Muddy Creek and Bassing Harbor sites.

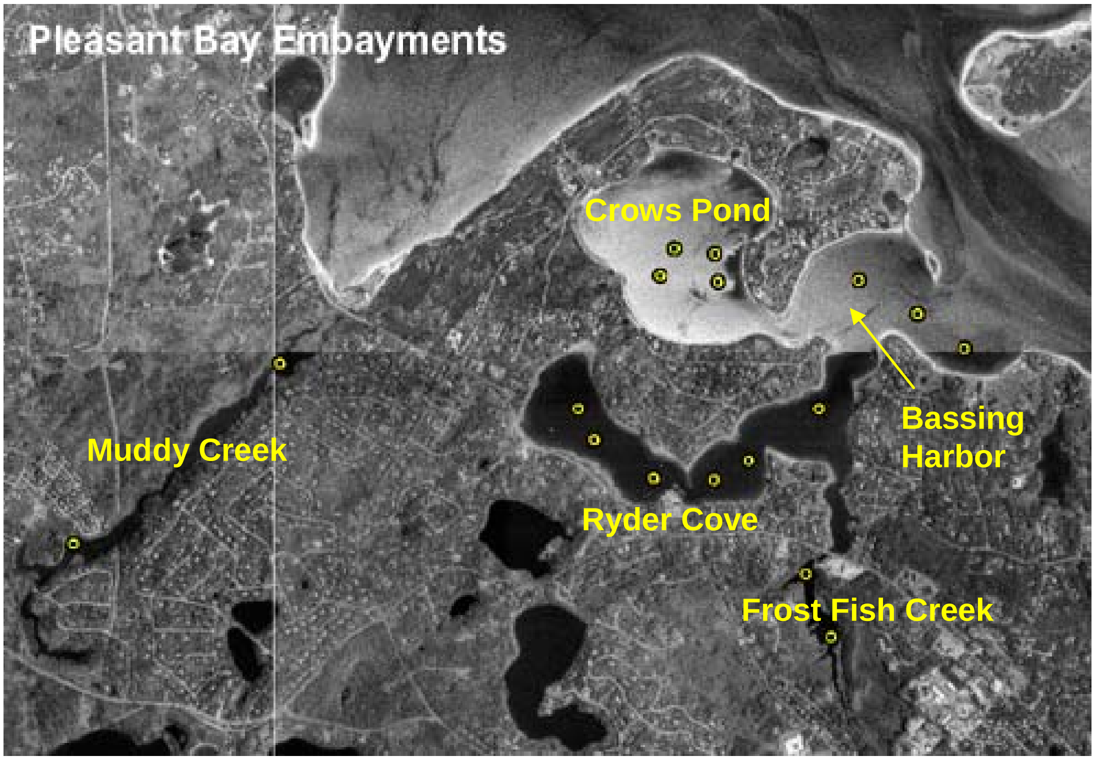


Figure IV-19. Pleasant Bay embayment system sediment sampling sites (yellow circles) for determination of nitrogen regeneration rates. These sites were sampled previously as part of the Chatham Wastewater Planning Study and reported in the MEP Technical Report for Chatham's embayments (Howes et al. 2003). Numbers reference list of core stations above.

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

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

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

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

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

Overall, coastal sediments are not overlain by nitrate rich waters and the major nitrogen input is via phytoplankton grazing or direct settling. In these systems, on an annual basis, the amount of nitrogen input to sediments is generally higher than the amount of nitrogen release. This net sink results from the burial of reworked refractory organic compounds, sorption of inorganic nitrogen and some denitrification of produced inorganic nitrogen before it can "escape" to the overlying waters. However, this net sink evaluation of coastal sediments is based upon annual fluxes. If seasonality is taken into account, it is clear that sediments undergo periods of net input and net output. The net output is generally during warmer periods and the net input is during colder periods. The result can be an accumulation of nitrogen within late fall, winter, and early spring and a net release during summer. The conceptual model of this seasonality has the sediments acting as a battery with the flux balance controlled by temperature (Figure IV-20).

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.

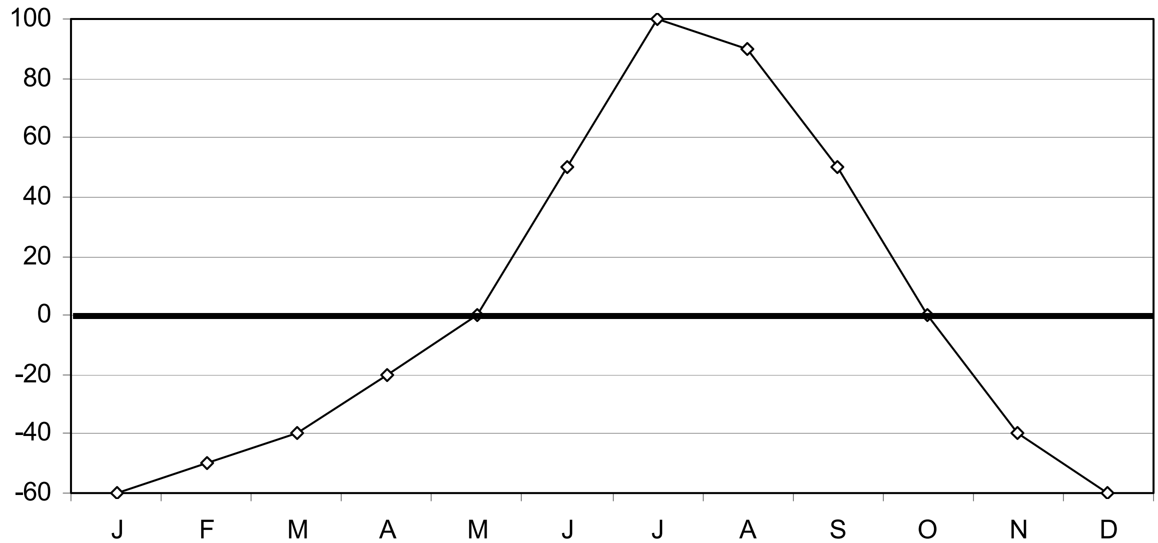


Figure IV-20. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

Sediment sampling was conducted within each of the sub-embayments of the Pleasant Bay System in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model (Figure IV-18**)**. The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content and bulk density and an analysis of each site's tidal flow velocities. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within each sub-embayment.

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

Net nitrogen release or uptake from the sediments within the Pleasant Bay System Embayment for use in the water quality modeling effort (Chapter VI) are presented in Table IV-11. Net nitrogen release from the sediments of the Pleasant Bay sub-embayments shows significant spatial variation, but is typical of other embayment within the MEP region. Although there are a large number of sub-embayments to the Pleasant Bay System, the rates of sediment nitrogen regeneration fell into only 4 (relatively tight) groups based upon the basin type:

(A) small enclosed basin (Meetinghouse Pond, Lonnies Pond, Areys Pond, Round Cove, Quanset Pond, Paw Wah Pond, Upper Muddy Creek),
(B) moderate sized tributary sub-embayment (The River, Bassing Harbor, Muddy Creek),
(C) salt marsh dominated tidal sub-estuary (Pochet),
(D) large lagoonal estuarine basin (Little Pleasant Bay, Pleasant Bay, Chatham Harbor).

The general pattern is for higher release from the small enclosed basins (group A) which tend to have higher nitrogen levels due to their circulation and focus of watershed nitrogen loads. In contrast the larger tributary sub-embayments (group B) tend to have better circulation relative to the watershed inputs and only moderate nitrogen regeneration rates. The large main basins of the lagoonal estuarine component (group D) exhibited low to negative regeneration rates consistent with their deep waters, depositional nature (Little Pleasant Bay, Pleasant Bay) or high tidal velocities (Chatham Harbor and eastern channel form Chatham Harbor to Little Pleasant Bay, channel between Strong Island and Bassing Harbor). The net nitrogen uptake by the predominantly salt marsh basin of Pochet is consistent with many observations of salt marsh nitrogen cycling (e.g. West Falmouth Harbor). This overall pattern generally reflects the particle distribution within Pleasant Bay, due to phytoplankton production and deposition. This pattern, on a smaller scale, was also observed within upper Cape embayments of Popponesset Bay and Three Bays, which have similar patterns of loading and multiple large sub-embayments. Lowering the nitrogen inputs to the inner basins will result in lower net nitrogen release rates over relatively short time scales.

Higher nitrogen net fluxes from sediments of the upper more nitrogen enriched basins also may result from differences in sediment nitrogen cycling. There is an indication that the very reducing (anoxic) nature of the Paw Wah Pond, Upper Muddy Creek and other group A basins may be increasing the percentage of nitrogen which is released from the sediments versus the amount of nitrogen being lost to denitrification via the pathway of mineralization→ nitrification → denitrification. The coupled nitrification-denitrification step in the pathway is significantly influenced by the availability of oxygen within the surficial sediments for nitrifying bacteria. That the anoxic/sulfidic nature of the sediment of these basins may be affecting enhancement of nitrogen release is supported by comparisons of measured release with estimates of total nitrogen regeneration (i.e. maximum potentially releasable). Using this rough approximation, a greater proportion of the potential release rates of nitrogen is achieved in the upper basins than from the other sites. Note that this approach yields general patterns and cannot be used to determine accurate nitrogen removal rates. Lowering nitrogen loading to these upper systems should improve sediment oxidation and improve nitrogen removal rates by these sediments, although quantifying this enhancement is highly site specific. However, based upon this information a linear model for the lowering of nitrogen release with lowered watershed nitrogen loading is conservative.

Table IV-11. Rates of net nitrogen return from sediments to the overlying waters of the Pleasant Bay embayment system. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (see Chapter VI). Measurements represent July -August rates. N represents sample size.

Sub-embayment	Site ID	Sediment N Regeneration mgN/m2/d		
		Mean	s.d.	N
Meetinghouse Pond				
Pond Basin	47,48,49	79.5	12.7	3
Lonnies Pond				
Pond Basin	53,54	22.7	3.9	2
Areys Pond				
Pond Basin	22,23	107.3	13.9	2
Namequoit River	24,25,26	107.3	2.1	3
The River				
Mtghouse Channel	50,51	113.0	13.5	2
Upper River	52,29	14.3	5.6	2
Mid River	28	12.0	1.8	1
Lower River	27,45	34.2	6.4	2
Mouth River	44	-10.9	11.3	1
Paw Wah Pond				
Pond Basin	46	120.7	13.9	2
Quanset Pond				
Pond Basin	2,3	98.0	5.9	2
Round Cove				
Cove Basin	1,2,3,4	138.9	10.4	4
Muddy Creek				
Upper	A:1,2	81.8	1.7	2
Lower	A:3,4	-16.0	5.0	2
Bassing Harbor Sub-System				
Ryders Cove	A:5,4AB,3/4,3,2,1	19.7	1.6	7
Frost Fish Creek Upper	A;1,2	-5.1	0.0	2
Crows Pond	A:1,2,3,3A,3B,4	12.3	1.3	6
Bassing Harbor Basin	A: 1,2,3	-8.9	1.8	3
Pochet				
Upper-Mid	38,39,40	-1.2	1.5	3
Lower Basin	41,42	-1.7	2.5	2
Little Pleasant Bay				
Upper	43,60,61,62	16.0	1.1	4
Mid	56,57,58,59	0.2	1.3	4
Broad Creek	36	4.1	2.3	1
Lower	30,31,34,35,37,55	-1.1	1.9	6
Pleasant Bay				
Main Basin	1,4,5,6,7,8,19,20,21	24.1	2.2	9
Little PB-ChatHbr	33,32,16,15,14,13	-7.0	1.4	6
Strong Isl-Bassing Hbr	17,18	-18.1	1.1	2
Chatham Harbor				
Basin	12,10,9	-8.8	0.7	3

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This section summarizes field data collection effort and the development of hydrodynamic models for the Pleasant Bay estuary systems (Figure V-1). For this system, the final calibrated model offers an understanding of water movement through the estuary, and provides the first step towards evaluating the water quality of these estuarine systems, as well as understanding nitrogen loading "thresholds" for each system. Tidal flushing information will be utilized as the basis for a quantitative evaluation of water quality. Nutrient loading data combined with measured environmental parameters within the various sub-embayments become the basis for an advanced water quality model based on total nitrogen concentrations. This type of model provides a tool for evaluating existing estuarine water quality, as well as determining the likely positive impacts of various alternatives for improving overall estuarine health, enabling the bordering towns to understand how pollutant loadings into the estuary will affect the biochemical environment and its ability to sustain a healthy marine habitat.

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.

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. the Atlantic Ocean). Several embayment-specific parameters influence tidal flushing and the associated residence time of water within an estuary. For the Pleasant Bay system, the most important parameters are the tide attenuation along with the shape, length and depth of the estuary and its attached sub-systems.

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

Figure V-1. Topographic map detail of the Pleasant Bay System, on Cape Cod, Massachusetts.

This hydrodynamic study was performed for the Pleasant Bay system, which is shared among the Towns of Chatham, Orleans and Harwich, Massachusetts, and located at the southeastern tip of Cape Cod. A map the general study area, assembled from topographic maps of the Pleasant Bay region, is shown in Figure V-1. The main basin of Pleasant Bay has two major subdivisions, a western portion Pleasant Bay, toward Harwich, and Little Pleasant Bay toward Orleans to the north. There are several smaller embayment systems that are part of the larger Pleasant Bay estuary including Bassing Harbor (Howes et al., 2003), The River (Kelley et al., 2003), and Muddy Creek (Howes et al., 2003).

The entire Pleasant Bay system, inclusive of associated sub-embayments, has a surface coverage of approximately 7,000 acres. The average depth of the entire Pleasant Bay is 6.0 ft, mean tide level (MTL). Bassing Harbor has a coverage of 340 acres, with an average depth of 4.3 ft MTL. The River, located in Orleans at the northern limit of Pleasant Bay, has an area coverage of 336 acres and an average depth of 6.3 feet MTL. Muddy Creek is situated along a portion of the municipal boundary between Chatham and Harwich, and has an area coverage of 34 acres, with an average depth of approximately 3.1 ft MTL.

Circulation throughout Pleasant Bay is dominated by tidal exchange directly with the Atlantic Ocean. The tide range in the main basin of Pleasant Bay is less than 45% of tide range offshore of the inlet to the system (55% tidal attenuation). The main lunar tidal constituent (i.e., the M_2) is reduced from an offshore range of 6.2 feet to 3.4 feet inside the Bay. The great degree of tidal attenuation between the open ocean and inside the bay is caused by flow restrictions at the inlet to the system. Hydraulic conditions at the inlet to Pleasant Bay are similar to the inlet of Nauset Harbor, the only other inlet on the open ocean side of the outer Cape. Tidal attenuation in Nauset Harbor is 52% of the open ocean tide. In addition to significant tide attenuation, both systems have large tidal prisms; 267 million cubic feet (MCF) for Nauset Harbor, and 1,191 MCF for Pleasant Bay. Also, both inlets exist in a very dynamic littoral environment (i.e., sediment transport) and are unstructured and self-maintaining.

This hydrodynamic portion of this study proceeded as two component efforts. In the first portion of the study, bathymetry, tide, and circulation velocity data were collected in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the study. The bathymetry survey executed for this project was performed to determine the variation of embayment and channel depths specifically at the inlet channel. The survey at the inlet was used to supplement other recent bathymetric data that were available from previous studies in the Pleasant Bay area.

In addition to the bathymetry survey, tides were recorded at seven locations within Pleasant Bay for 43 days. These tide data were necessary to run and calibrate the hydrodynamic model of the system. Finally, an Acoustic Doppler Current Profiler (ADCP) survey was completed during a single tide cycle to measure ebb and flood velocities across two channel transects. The ADCP data were used to compute system flow rates and to provide an independent means of verifying the performance of the hydrodynamic model.

A numerical hydrodynamic model of the Pleasant Bay system was developed in the second portion of the hydrodynamic component of this study. Using the bathymetry survey data, a model grid mesh was generated for use with the RMA-2 hydrodynamic code. The tide data from offshore Nauset Beach were used to define the open boundary conditions that drive the circulation of the model at the two system inlets, and data from the five TDR stations within the system were used to calibrate and verify model performance to ensure that it accurately represents the dynamics of the real, physical system.

The calibrated computer model of Pleasant Bay was used to compute the flushing rates of selected sub-embayments. Though water quality in an embayment cannot be directly inferred by use of computed flushing rates alone, they can serve as useful indicators of embayment flushing performance relative to other areas in the same system. The ultimate utility of this hydrodynamic model is as input into a constituent transport model, where water quality constituents like nitrogen are modeled to determine the real water quality dynamics of a system. The water quality modeling portion of this study is presented in Chapter VI.

V.2 GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE SYSTEM

The Atlantic Coast of the Pleasant Bay region is a highly dynamic region, where natural forces continue to have dramatic impacts on 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 Pleasant Bay estuary. Storm-driven inlet formation often leads to hydraulically efficient estuarine systems, where seawater exchanges more rapidly with water inside the estuary. However, this episodic inlet formation is balanced by the gradual wave-driven migration of the barrier beach separating the estuary 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).

Although man has modified much of the Pleasant Bay 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 resulted in improved water quality in the upper portions of the estuary. Most of the manmade modifications to Pleasant Bay have caused relatively small changes to overall estuarine health; however, water quality in some sub-embayments has been impacted by infrastructure improvements. Notably, the culverts restricting tidal flow under Route 28 have had a negative influence on water quality within Muddy and Frost Fish Creeks.

V.2.1 Natural Inlet Processes

Many of the regional barrier beach systems in the Pleasant Bay region 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, with an approximate 1 ft increase in tide range and a corresponding improvement to tidal flushing within the northern portions of the estuary. The inlet continues migrating south and Nauset Beach likely 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 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.

Following the 1987 breach, the beach system returned to a form similar to the 1846 condition 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 the barrier spit has reached a point where its hydraulic efficiency has reduced significantly, storm overwash conditions can 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-3 illustrates this initial widening of the inlet following the most recent breach. By the early 1990s, South Beach (the portion of Nauset Beach south of New Inlet) had attached to the mainland of Cape Cod, closing navigation access to the Chatham Bars channel.

Once the initial breach widened to form New Inlet, the strong tidal currents inhibited and/or prevented migration of natural littoral sediments along the beach. Instead, much of the wave-driven sand traveling along the Nauset Beach face has been deposited either in the flood or ebb shoals. The flood shoal, which forms during the flooding portion of the tide, consists of the series of shoals within Chatham Harbor that extend north from the inlet (Figure V-4). The ebb shoal, which forms during the ebbing portion of the tide, consists of the shoal immediately offshore and to the south of New Inlet (Figure V-4). Since much of the ebb shoal is in relatively deep water, the form of this shoal is not as obvious as the flood shoal; however, a substantial volume of littoral sediments is contained within this deposit. Since the net north-to-south littoral drift has been interrupted by New Inlet, the sediment supply to South Beach has been greatly reduced. Reduction to the South Beach regional sediment supply will allow more frequent storm overwash and barrier beach migration in this area. Once the form of South Beach has been destabilized, relatively rapid southerly migration of the inlet is anticipated. However, the destabilization/beach migration process for South Beach will require several decades (likely another 40-to-60 years before large-scale inlet migration, based upon inlet dynamics following the 1846 breach).

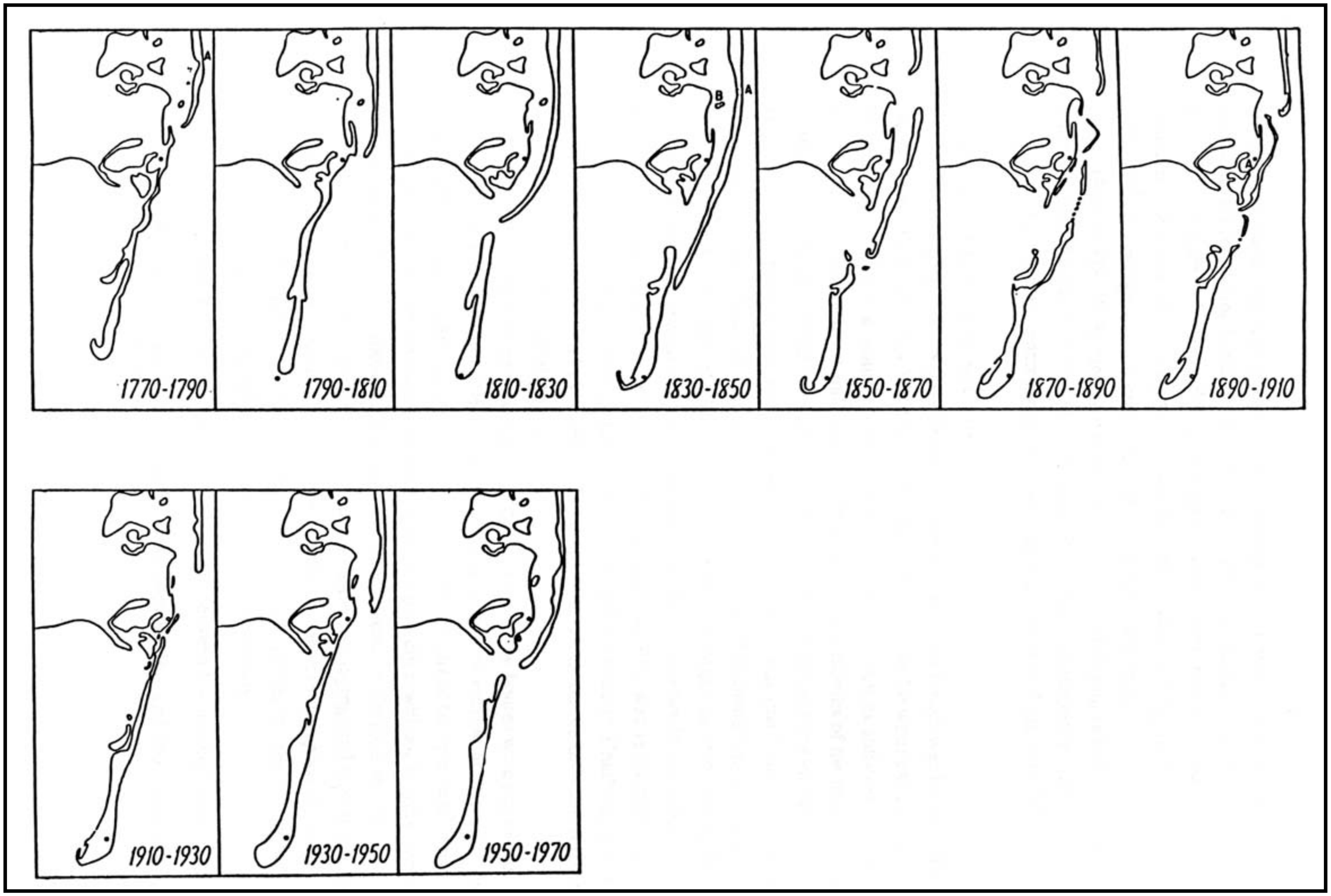


Figure V-2. Historical changes in the Nauset Beach-Monomoy barrier system illustrated by generalized 20-year diagrams from 1770-1790 to 1950-1970 (from Geise, 1988).

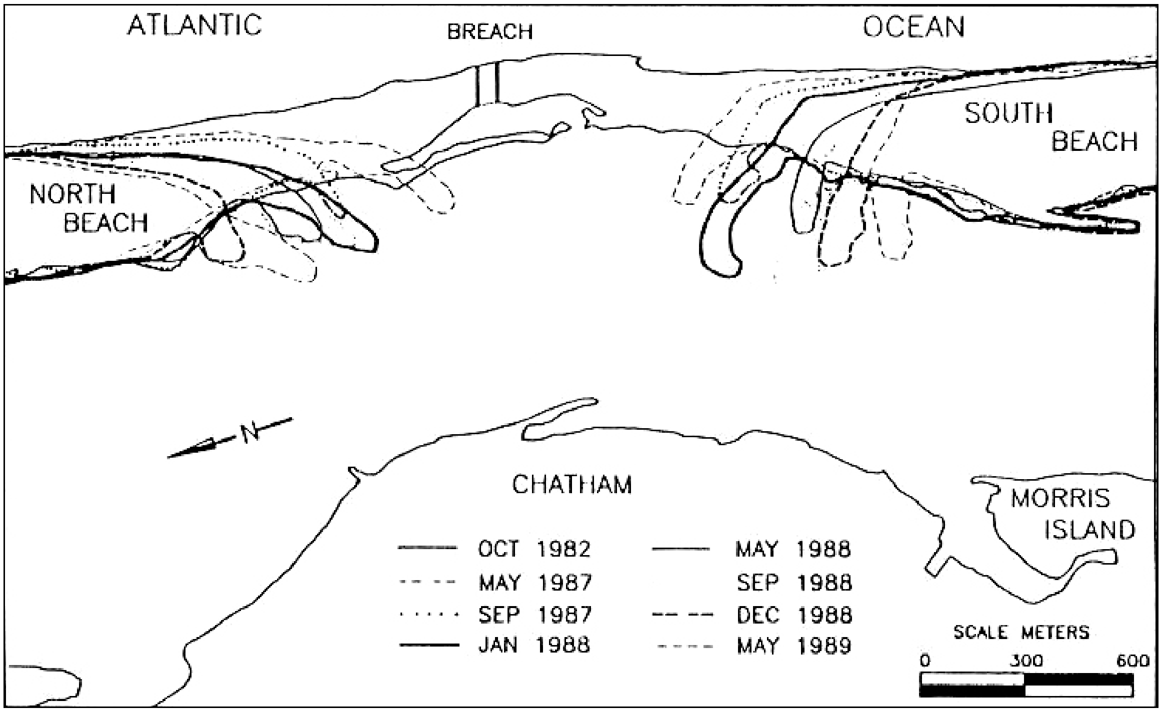


Figure V-3. Sequential shoreline changes of the two spits flanking New Inlet (from May 1987 to May 1989). The pre-breach shoreline of October 1982 also is shown for comparison (Liu, *et al*., 1993).

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 towns surrounding Pleasant Bay 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 (Ramsey, 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.

A comparison of Figures V-4, V-5, and V-6 indicates only minor changes in the overall inlet morphology for 1990, 1994, and 2001, respectively. The August 1990 aerial photograph (Figure V-4) shows early development of the flood shoal and swash platform. Prior to full development of the swash platform and attachment of South Beach to the Chatham shoreline, the main channel was relatively unconstrained with depths in excess of 3 meters (10 feet) through the flood shoal (FitzGerald and Montello, 1993). As depicted in the 1994 photograph (Figure V-5), the main tidal channel servicing the Chatham Harbor/Pleasant Bay system had migrated to the south following attachment of South Beach to the mainland. Following initial attachment of South Beach to the Chatham shoreline, rapid beach accretion occurred from Outermost Harbor Marine and Holway Street. Prior to beach attachment, the Chatham coast between Bearses Lane and Watch Hill had experienced significant erosion. Although this region remains exposed to open Atlantic Ocean wave conditions, accretion of the fronting beach, as well as development of the ebb tidal shoal, have reduced storm wave heights along the Chatham mainland shoreline.

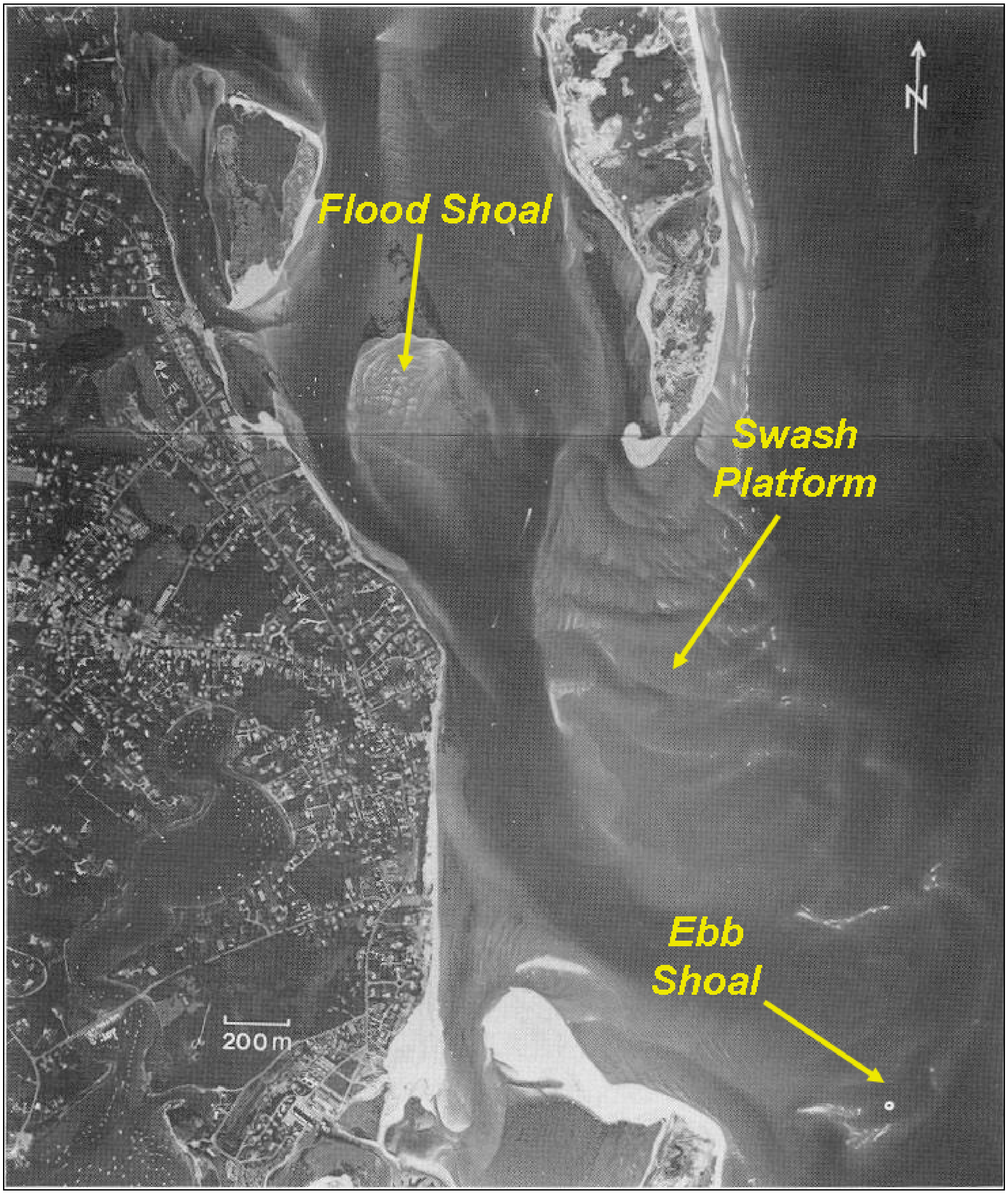


Figure V-4. August 1990 aerial photograph of the New Inlet region showing the flood shoal, ebb shoal, and swash platform (modified from aerial photograph in FitzGerald and Montello, 1993).

Figure V-6 indicates that the southern tip of Nauset Beach continues to migrate in a southerly direction. However, discussions with Ted Keon (Chatham Coastal Resources Director) during January 2006 indicated that the southern tip of Nauset Beach again had retreated, at least temporarily, to the north. Comparison of Figures V-5 and V-6 indicates that the overall beach width of South Beach continues to decrease, likely due to loss in the littoral sediment supply. This observation is supported by the loss of dune vegetation on South Beach along the area directly east of Morris Island. Northeast storms continue to overwash areas of South Beach indicating loss in overall beach volume, as well as landward migration of this feature.

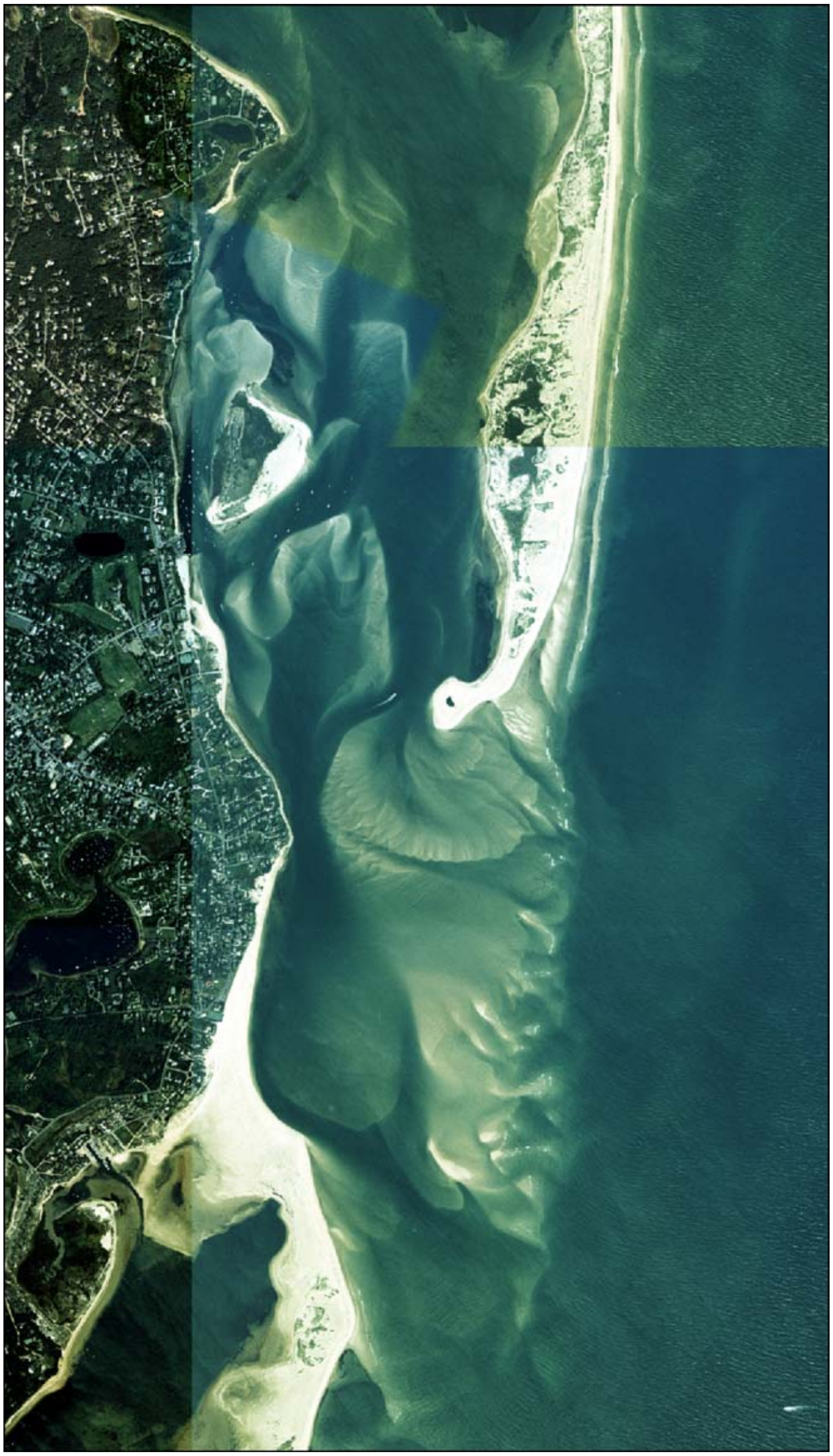


Figure V-5. Aerial photograph from September/October 1994 illustrating changes in the inlet geomorphology (source: MassGIS).

Figure V-6. Aerial photograph from April 2001 illustrating changes in the inlet geomorphology (source: MassGIS).

Figure V-7 provides an overview of sedimentation processes within New Inlet for the two-year period following the 1987 breach. Shoal formation during this initial inlet equilibration period provides valuable insight into how the inlet morphology will evolve. In addition, the more recent aerial photography (Figure V-5 and V-6) allows at least qualitative evaluation of inlet evolution hypotheses initially developed following the formation of New Inlet.

Initial shoal formation after development of the 1987 breach is shown in Figure V-7. Initially, sediments eroded from the inlet channel were deposited on both the ebb and flood tidal shoals. Within approximately one year, the spatial extent of the ebb flood shoal had equilibrated; however, both the flood shoal and swash platform continued to grow. Due to the net north-to-south wave-driven sediment transport along Nauset Beach, continued growth of the swash platform is anticipated. Sediment derived from Nauset Beach continues to expand the limits of the flood shoal, where the comparison of Figures V-5 and V-6 illustrates northerly migration of the flood shoal front. Increase in volume of this feature continues to require frequent maintenance dredging of the channel servicing Aunt Lydia's Cove, where dredge volumes necessary to maintain a navigable channel have increased since the late 1990s (personal communication, Ted Keon, January, 2006). Although shoal features continue to evolve within the Chatham Harbor region, the relative stability of South Beach has prevented rapid southerly growth of Nauset Beach. Since the sediment supply to South Beach is limited, natural barrier beach processes eventually will cause this feature to migrate toward the west. This process likely will not be gradual, but rather occur as a result of storm-driven barrier beach overwash and breaching of the narrowed barrier system. Once the continuity of South Beach is compromised, rapid land ward migration of this feature likely will occur, with resulting southerly migration of New Inlet.

The tidal regime (tide range and phase) within the Pleasant Bay system has been relatively stable for approximately 15 years. When inlet geomorphology returns to the general conditions for 1910-1930 shown in Figure V-2, Nauset Beach again will rapidly migrate to the south. This southerly migration will make the inlet less efficient, with a reduction in tide range and an increase in the delay of the tidal phase between the Atlantic Ocean and Pleasant Bay. In addition, the distance that higher quality ocean water needs to travel to reach Pleasant Bay increases as well, causing an overall reduction in Pleasant Bay water quality. As a "worst case" scenario, hydrodynamic and water quality conditions associated with the pre-1987 breach morphology were evaluated (see Section IX of this report) to determine the impacts of natural inlet migration on overall estuarine health.

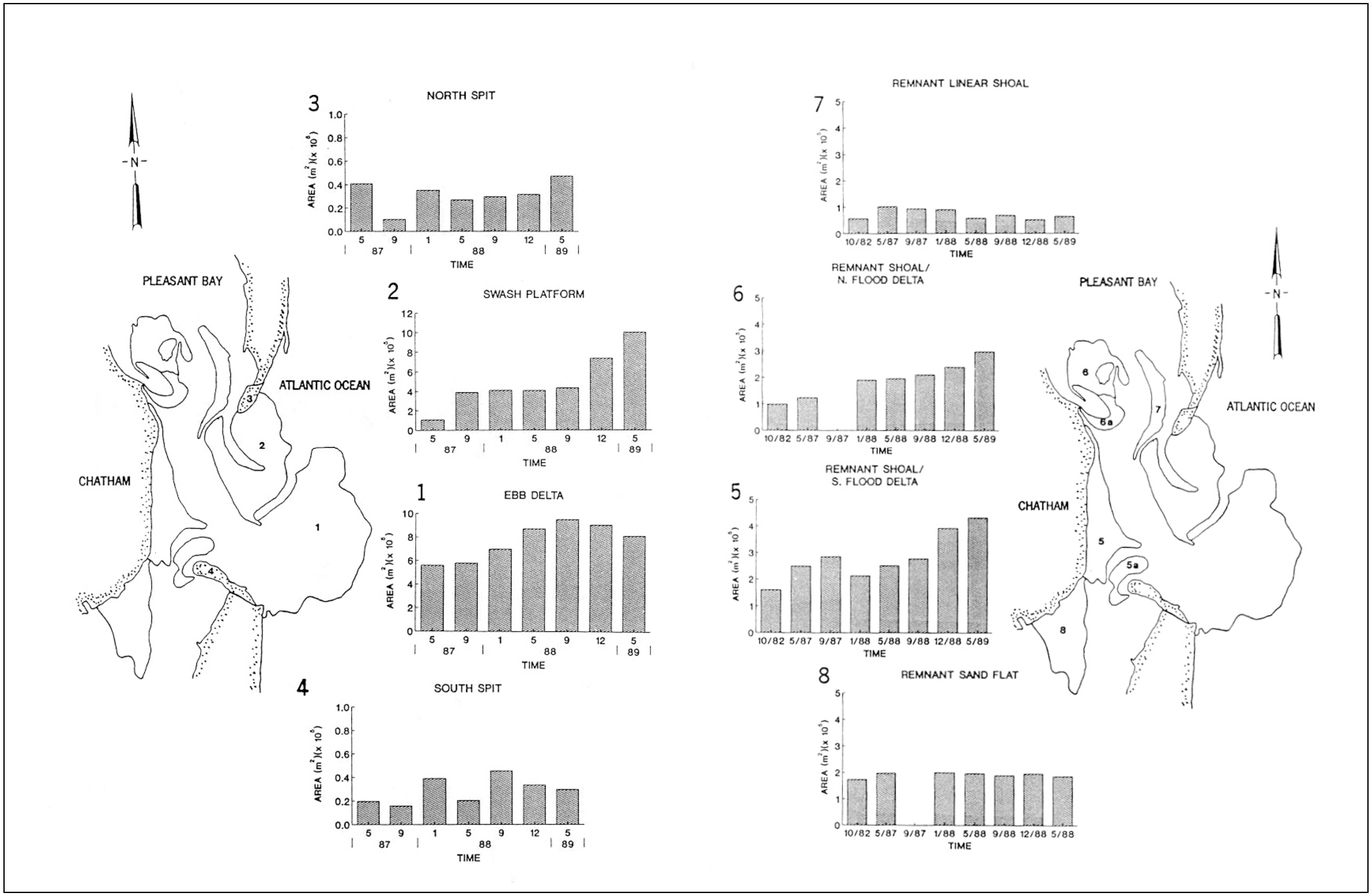


Figure V-7. Post-breach shoal and spit area changes between mid 1997 and early 1989 (Liu, *et al*., 1993).

V.2.2 Anthropogenic Changes

Although the formation of New Inlet in 1987 has dominated tidal circulation dynamics within the Pleasant Bay estuarine system, anthropogenic changes within specific sub-embayments have influenced both hydrodynamics and the associated water quality. Specifically, roadway construction required construction of culverts across two Pleasant Bay sub-embayments: 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 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 were assessed as part of the Massachusetts Estuaries Project report regarding Chatham's embayments (Howes, *et al*., 2003).

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 more than 50% 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. Similar to Muddy Creek, alternatives to improve tidal flushing in Frost Fish Creek were assessed as part of the Massachusetts Estuaries Project report regarding Chatham's embayments (Howes, *et al*., 2003).

V.3 DATA COLLECTION AND ANALYSIS

The field data collection portion of this study was performed to characterize the physical properties of the Pleasant Bay estuary. Bathymetry were collected throughout the system so that it could be accurately represented as a computer hydrodynamic model, and so that flushing rates could be determined for the system sub-embayments. In addition to the bathymetry, tide data were also collected at seven locations, to run the circulation model with real tides, and also to calibrate and verify its performance. Data from the NOAA GEODAS bathymetry database were used as a supplemental source in areas not covered by the recent surveys.

V.3.1 Bathymetry Data Collection

Bathymetry data in the inlet channel to Pleasant Bay were collected during November 2004. Supplemental bathymetry were available from past studies in the Pleasant Bay region, including for Bassing Harbor (Howes et al., 2003) and the main Basin of Pleasant Bay and "The River" (Ramsey, 1997). Positioning data were collected using a differential GPS. The 2004 survey design included gridded cross-channel transects between 660 ft (200 meter) to 330 ft (100 meter) spacings in inlet throat. Closer transect spacings were followed in areas were greater variation in bottom bathymetry was expected. Survey paths for the various surveys are shown in Figure V-8. The resulting bathymetric surface created by interpolating the data to a finite element mesh is shown in Figure V-9. All bathymetry was tide corrected, and referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29), using survey benchmarks located in the project area.

Results from the surveys show that the deepest point in the Pleasant Bay system is located in the inlet channel, and is –51.8 ft NGVD. Outside the inlet channel, the greatest depths (approximately 16 ft) in the main basin of the system are located in West Pleasant Bay. Meetinghouse Pond, at the northern extent of the system, is deep kettle pond, with a maximum depth of about 22 feet.

V.3.2 Tide Data Collection and Analysis

Tide data records were collected at seven stations in the Pleasant Bay estuary: 1) offshore Nauset Beach, 2) Chatham Harbor at the Chatham Fish Pier, 3) Ryder Cove in Bassing Harbor 4) West Pleasant Bay, 5) Round Cove, 6) Meetinghouse Pond, and 7) Pochet Neck. The locations of the stations are shown in Figure V-8. The Temperature Depth Recorders (TDR) used to record the tide data were deployed for a 43-day period between October 19 and November 30, 2004. The elevation of each gauge was leveled relative to NGVD 29. Two gauges were deployed together offshore Nauset Beach by SCUBA divers using a screw anchor. Duplicate offshore gauges were deployed to ensure data recovery, since the offshore tide record is crucial for developing the open boundary condition of the hydrodynamic model of the Pleasant Bay system. Data from the other six locations were used to calibrate the model, and also to tide correct raw bathymetric data collected during the time span of the deployment.

Plots of the tide data from three representative gauges are shown in Figure V-10, for the entire 43-day deployment. The spring-to-neap variation in tide can be seen in these plots, particularly in the offshore record. From the plot of the data from offshore Nauset Beach, the tide reaches its minimum neap tide range of 3.7 feet around November 5, and about seven days later the maximum spring tide range is 9.8 feet.

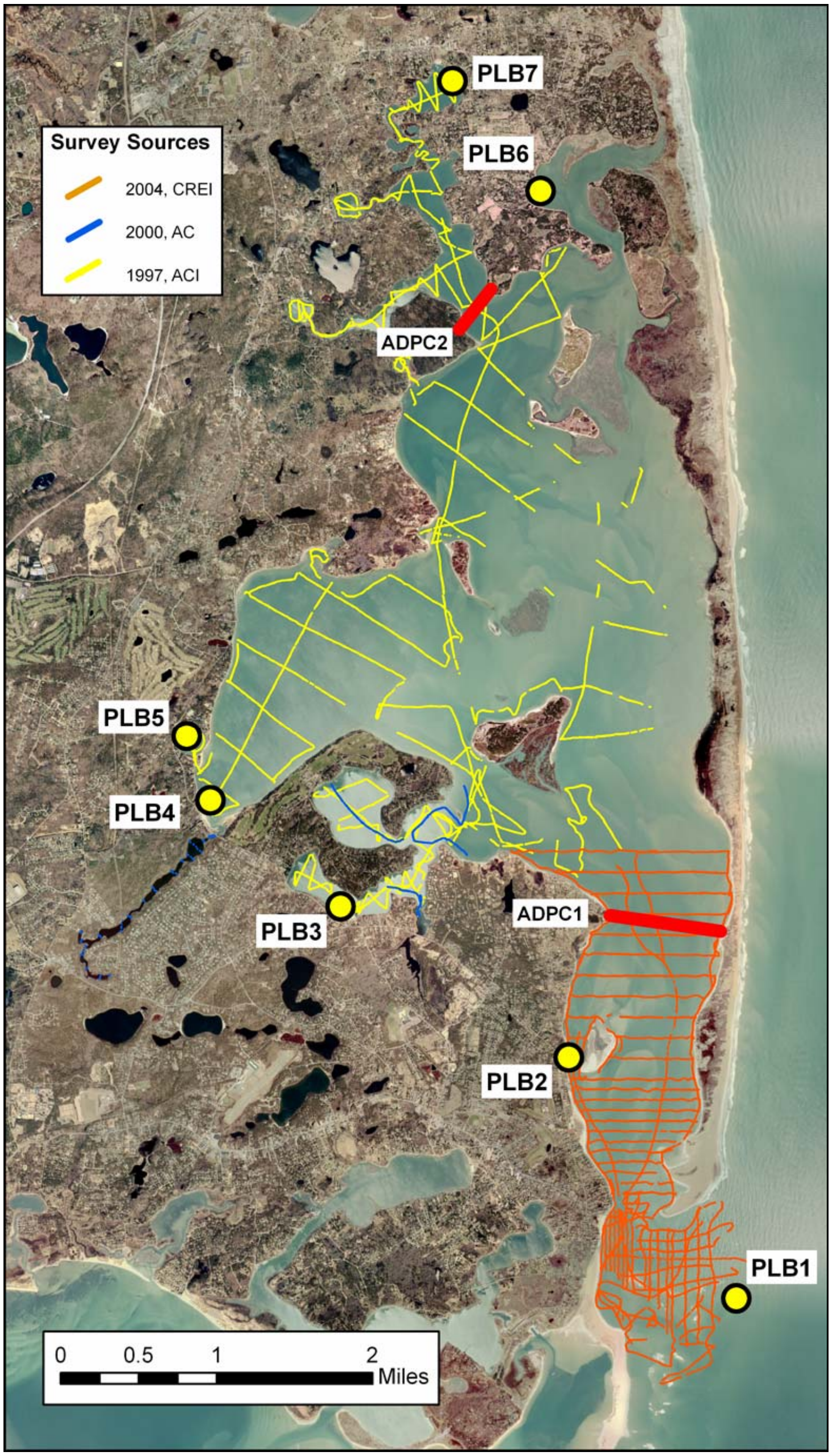


Figure V-8. Transects from recent bathymetry surveys of the Pleasant Bay system. Three different sources for the bathymetry data are indicated by different colors: orange for the 2004 MEP survey of the inlet, blue for the 2000 Applied Coastal surveys (for the Town of Chatham) and yellow for the 1997 survey performed by Aubrey Consulting (Ramsey 1997). Yellow markers show the locations of tide recorders deployed for this study. The two ADCP transects followed across the Pleasant Bay inlet and the mouth to "The River" are indicated by the thick solid red lines.

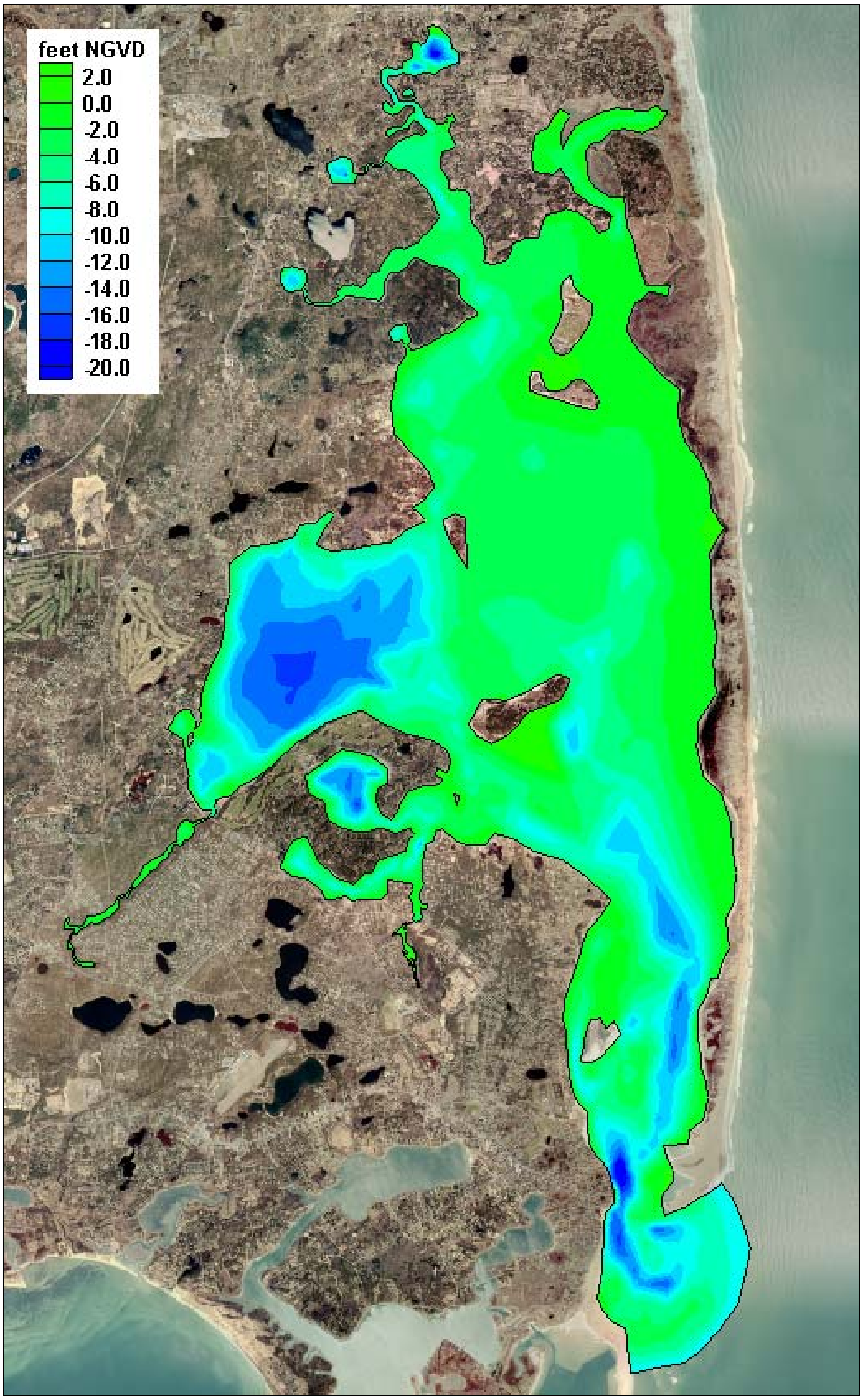


Figure V-9. Plot of interpolated finite-element grid bathymetry of the Pleasant Bay system, shown superimposed on 2001 aerial photos of the system locale. Bathymetric contours are shown in color at two-foot intervals, and also as lines at four-foot intervals.

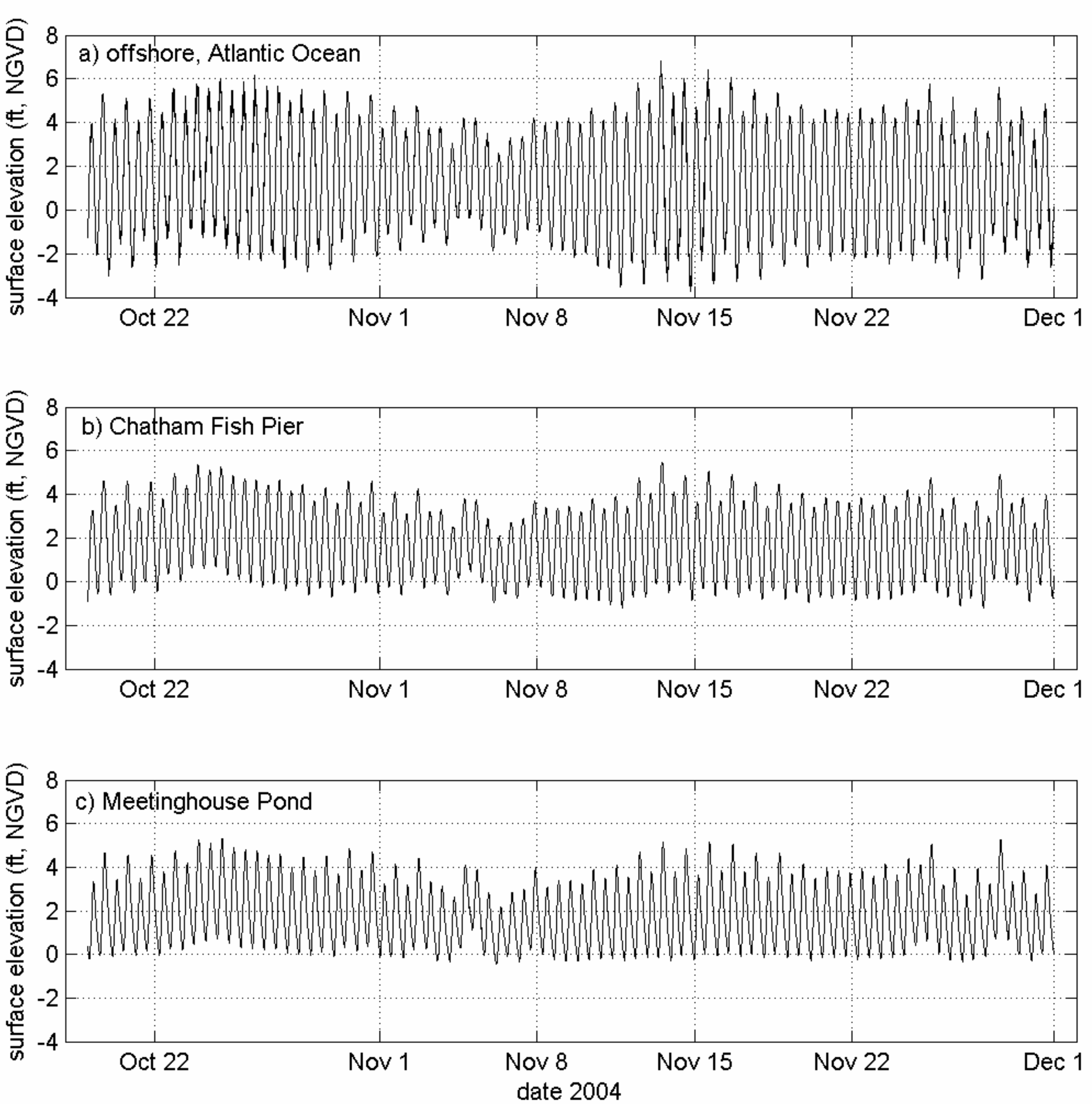


Figure V-10. Plots of observed tides for the Pleasant Bay system, for the 43-day period between October 19 and November 30, 2004. The top plot shows tides offshore Nauset Beach, in the vicinity of the inlet. The middle plot shows tides recorded in the inlet channel at the Chatham Fish Pier, and the bottom plot shows tides recorded at Meetinghouse Pond, at the northernmost reach of Pleasant Bay. All water levels are referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29).

A visual comparison in Figure V-11 between tide elevations at five of the Pleasant Bay stations shows that there is a significant reduction in the tide range, even over the relatively short distance between the open ocean and the fish pier. The loss of amplitude with distance from the inlet is described as tidal attenuation. Frictional mechanisms dissipate tidal flow energy, resulting in a reduction of the height of the tide. Tide attenuation is accompanied by a time delay (or phase lag) in the time of high and low tide (relative to the offshore tide), which becomes more pronounced farther into an estuary. Both effects are plainly evident in plot of

Figure V-11. Tide attenuation between the inlet and Meetinghouse Pond is nearly 50%. For the data plotted from November 11, the offshore tide range is 8.4 feet while the corresponding tide range in Meetinghouse Pond that day was 4.4 feet. The tide lag is greatest in Meetinghouse Pond, as seen in Figure V-11, where low tide in this sub-embayment occurs approximately four hours after low tide offshore Nauset Beach. The great degree of tidal energy attenuation across the inlet to Pleasant Bay gives an indication of the energy required to balance the pressure applied to the inlet from the large amount of littoral drift (sediment transport) along the shoreline of the outer Cape and Nauset Beach. The attenuated tidal energy is spent to keep the inlet open against the constant input of sand from Nauset Beach. However, the tidal and littoral dynamics are not in perfect balance, as is evident by the cyclical migration of the inlet.

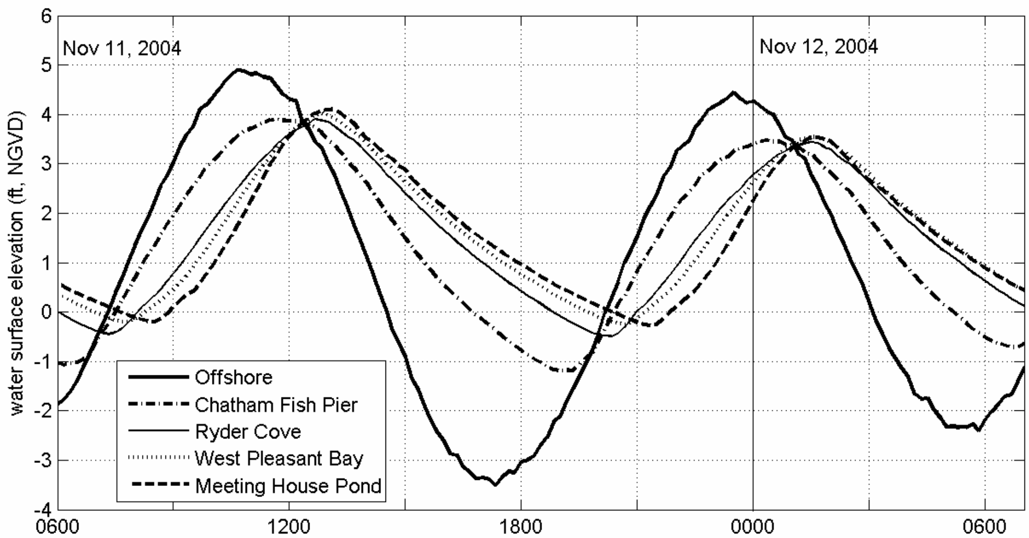


Figure V-11. Plot showing two tide cycles tides at five stations in Pleasant Bay plotted together. Demonstrated in this plot is the significant frictional damping effect caused by flow restrictions at the inlet channel. The damping effects are seen as a reduction in the range of the tide and a lag in time of high and low tides from the Atlantic Ocean. The time lag of low tide between the ocean and Meetinghouse Pond in this plot is four hours.

Standard tide datums were computed from the 43-day records from Pleasant Bay. These datums are presented in Table V-1. For most NOAA tide stations, these datums are computed using 19 years of tide data, the definition of a tidal epoch. For this study, a significantly shorter time span of data was available; however, these datums still provide a useful comparison of tidal dynamics within the system. The Mean Higher High (MHH) and Mean Lower Low (MLL) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. The Mean Tide Level (MTL) is simply the mean of MHW and MLW. Tide attenuation through the Pleasant Bay estuary is apparent again by how the MHW and MLW elevations change between the offshore gauges and those in the Bay.

Table V-1. Tide datums computed from 43-day records collected at stations in the Pleasant Bay system. Datum elevations are given relative to NGVD 29.

Tide Datum	Offshore	Fish Pier	Ryder Cove	West Pleasant Bay	Pochet Neck	Meeting House Pond
Maximum Tide	6.8	5.5	5.4	5.4	5.4	5.4
MHHW	5.2	4.4	4.4	4.4	4.5	4.5
MHW	4.8	4.0	4.0	4.0	4.1	4.1
MTL	1.4	1.8	2.1	2.1	2.1	2.1
MLW	-1.9	-0.3	0.1	0.1	0.1	0.1
MLLW	-2.4	-0.5	0.0	0.0	0.0	-0.0
Minimum Tide	-3.7	-1.2	-0.6	-0.5	-0.4	-0.4

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

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

A harmonic analysis was performed on the time series from each gauge location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The observed astronomical tide is 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-12. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-2 presents the amplitudes of eight tidal constituents in the Pleasant Bays system.

The M_2, or the familiar twice-a-day lunar semi-diurnal tide, is the strongest contributor to the signal with an offshore amplitude of 3.1 ft. The total range of the offshore M_2 tide is twice the amplitude, or 6.2 ft. Through the inlet channel, the M_2 amplitude is reduced through hydraulic resistance. At the Chatham Fish Pier the M_2 amplitude is already a foot less than offshore. Within the main basin of Pleasant Bay, the M_2 is consistently 1.7 ft.

The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 period for the M_6), results from frictional attenuation of the M_2 tide in shallow water. The M_4 has nearly a zero amplitude offshore, but grows to 0.4 ft in the northern end of the Bay. The M_6 has a very small amplitude throughout the system (less than 0.1 feet).

The other major tide constituents also show similar variation between the offshore station and within the Bay. The diurnal tides (once daily), K_1 and O_1, show similar amplitude reductions of approximately 0.15 feet between the offshore and Bay gauges. Other semi-diurnal tides, the S_2 (12.00 hour period) and N_2 (12.66-hour period) tides (with offshore amplitudes of 0.5 and 0.7 ft respectively), also are attenuated though the inlet, resulting in amplitude reductions of approximately 50 % from offshore. The M_{sf} is a lunarsolar fortnightly constituent with a period of approximately 14 days, and is the result of the periodic conjunction of the sun and moon, and has an amplitude less than 0.1 ft.

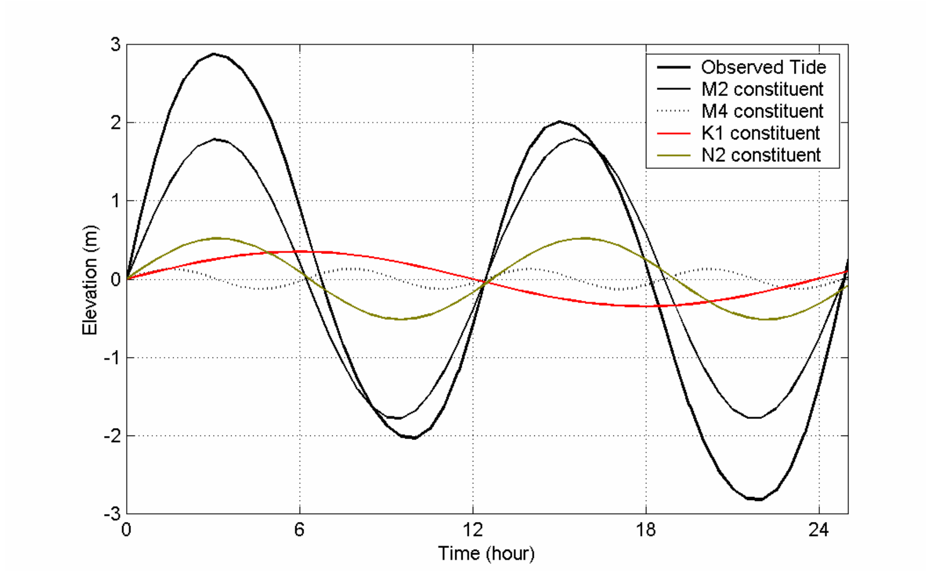


Figure V-12. Example of an observed astronomical tide as the sum of its primary constituents.

Table V-2. Major tidal constituents determined for gauge locations in Pleasant Bay, October 19 through November 30, 2004.

	Amplitude (feet)							
Constituent	M_2	M_4	M_6	S_2	N_2	K_1	O_1	M_{sf}
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Offshore	3.12	0.02	0.01	0.52	0.67	0.45	0.42	0.05
Chatham Fish Pier	2.03	0.08	0.06	0.27	0.39	0.33	0.29	0.06
Ryder Cove	1.72	0.14	0.06	0.22	0.32	0.31	0.28	0.07
West Pleasant Bay	1.69	0.23	0.04	0.22	0.31	0.30	0.28	0.07
Round Cove	1.69	0.23	0.04	0.22	0.31	0.30	0.28	0.07
Pochet Neck	1.69	0.35	0.03	0.23	0.30	0.32	0.28	0.06
Meeting House Pond	1.69	0.33	0.03	0.22	0.30	0.31	0.28	0.08

Together with the change in constituent amplitudes, the phase change of the tide is easily seen from the results of the harmonic analysis. Table V-3 shows the delay of the M_2 at different points in the Pleasant Bay system, relative to the timing of the M_2 constituent offshore Nauset Beach. The greatest delay is at the Pochet Neck TDR station. Though the Pochet Neck station has the greatest phase lag, it has the same M_2 amplitude as the other stations situated in the main basin of the Bay. This indicates that within the Bay, the phase delay of the tide is more a function of the distance between the stations, rather than additional fictional attenuation. The tide and its individual constituents move through the bay as very long period waves, and therefore, with greater distances, more time (i.e., greater phase difference) is required for the tide to propagate between stations. This is further demonstrated by utilizing the shallow wave dispersion relationship to estimate the propagation velocity of the tide. For waves propagating across shallow depths (relative to their wavelength), the wave celerity, C (propagation velocity) , can be determined using the simple equation $C = \sqrt{gh}$, where g is the gravitational constant and *h* is the average depth of the wave fetch. For the reach between the Chatham Fish pier and Pochet Neck gauging stations, the average depth and distance are approximately 4 ft MTL and 30,000 ft, respectively. Using these parameters the estimated time require for the tide wave to propagate from the fish pier to Pochet Neck is 44 minutes, which compares well with the 46 minute time delay calculated using the times given in Table V-3.

Table V-3. M_2 tidal constituent phase delay (relative to the Atlantic Ocean offshore of Nauset Beach) for gauge locations in the Pleasant Bay system, determined from measured tide data.

Station	Delay (minutes)
Chatham Fish Pier	27.3
Ryder Cove	55.4
West Pleasant Bay	63.5
Round Cove	63.8
Pochet Neck	72.9
Meeting House Pond	71.9

In addition to the tidal analysis, the data were further evaluated to determine the importance of tidal versus non-tidal processes to changes in water surface elevation. These other processes include wind forcing (set-up or set-down) within the estuary, as well as sub-tidal oscillations of the sea surface. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large compared to tidal flow. The results of an analysis to determine the energy distribution (or variance) of the original water elevation time series for the Pleasant Bay system is presented in Table V-4 compared to the energy content of the astronomical tidal signal (re-created by summing the contributions from the 23 constituents determined by the harmonic analysis). Subtracting the tidal signal from the original elevation time series resulted with the non-tidal, or residual, portion of the water elevation changes. The energy of this non-tidal signal is compared to the tidal signal, and yields a quantitative measure of how important these non-tidal physical processes can be to hydrodynamic circulation within the estuary. Figure V-13 shows the comparison of the measured tide from the Chatham Fish Pier, with the computed astronomical tide resulting from the harmonic analysis, and the resulting non-tidal residual.

Table V-4. Percentages of Tidal versus Non-Tidal Energy for tide gauge records from the Pleasant Bay system, October through November, 2004.

TDR LOCATION	Total Variance ($ft^2 \cdot sec$)	Tidal (%)	Non-tidal (%)
Offshore	5.665	97.5	2.5
Chatham Fish Pier	2.336	92.6	7.4
Ryder Cove	1.728	90.1	9.9
West Pleasant Bay	1.675	90.1	9.9
Round Cove	1.675	90.1	9.9
Pochet Neck	1.718	90.5	9.5
Meeting House Pond	1.717	91.1	8.9

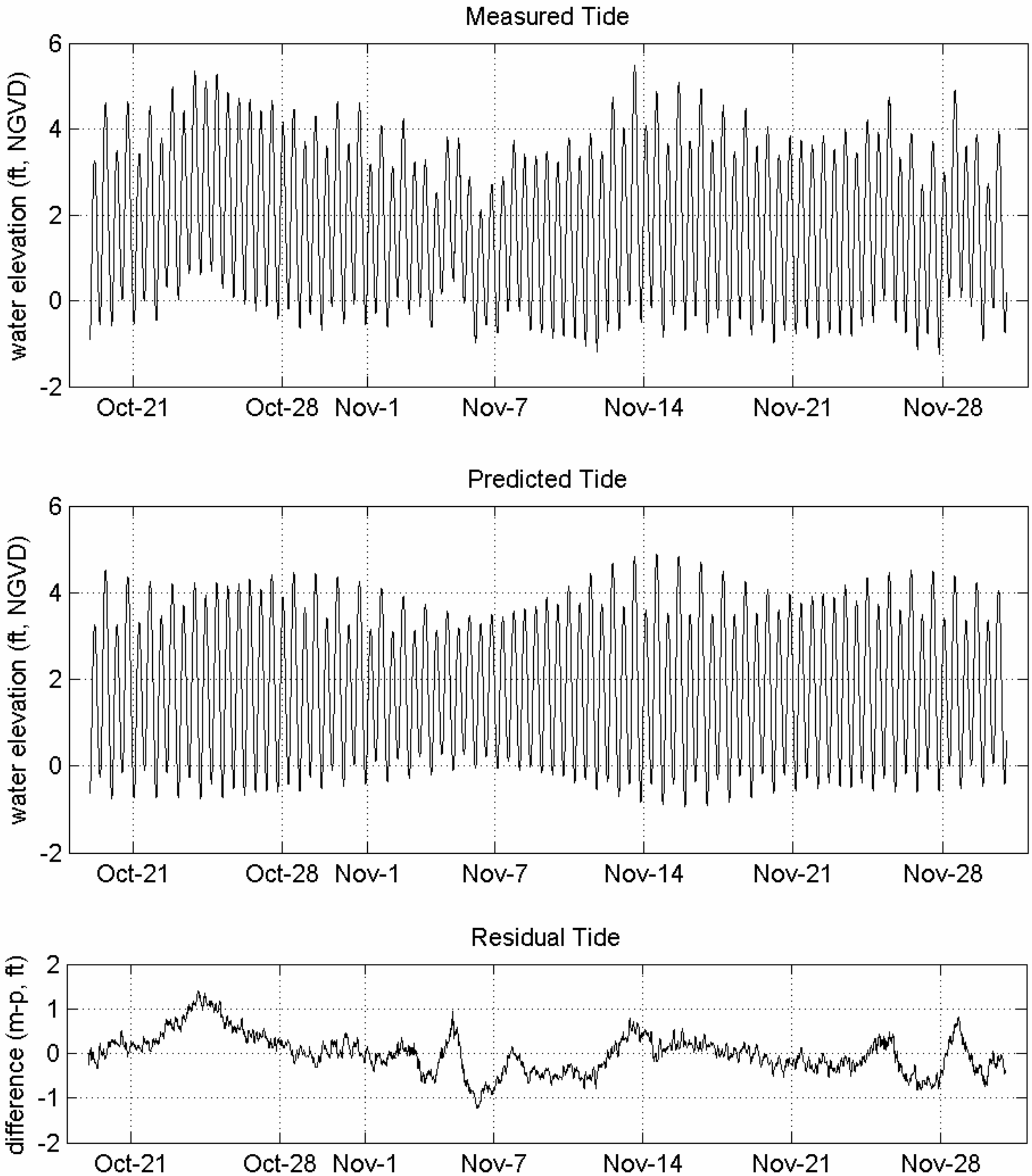


Figure V-13. Plot showing the comparison between the measured tide time series (top plot), and the predicted astronomical tide (middle plot) computed using the 23 individual tide constituents determine in the harmonic analysis of the Chatham Fish Pier gauge data. The residual tide shown in the bottom plot is computed as the difference between the measured and predicted time series (r=m-p).

Table V-4 shows that the variance of tidal energy was essentially equal in all parts of the main basin of Pleasant Bay; as should be expected given the minimal tidal attenuation across the Bay (apart from the inlet). The analysis also shows that tides are responsible for more than 90% of the water level changes in the Pleasant Bay system. The remaining 10% was the result of atmospheric forcing, due to winds, or barometric pressure gradients. The largest tide residuals occurred at during October 24. Atmospheric pressure data from archived regional meteorological records indicate that this was due to the passage of a low pressure system through the area.

V.3.3 ADCP Data Analysis

Cross-channel current measurements were surveyed in the inlet channel and at the mouth of “The River” in northern Pleasant Bay through a complete tidal cycle to resolve spatial and temporal variations in tidal current patterns. The surveys at the inlet and “The River” were executed November 16 and 23, 2004, respectively. These surveys were designed to observe tidal flow across two transects in the system at hourly intervals (with each transect surveyed twice each hour. These transects are indicated in Figure V-8. The data collected during this survey provided information that was necessary to model properly the hydrodynamics of the Pleasant Bay system.

Figures V-14 through V-17 show color contours of the current measurements observed during the flood and ebb tides at each of the 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 inlet to Pleasant Bay, positive along-channel flow is to the north, and positive cross-channel flow is moving to east. In Figure V-14, the lower left panel shows depth-averaged currents across the channel projected onto a 1994 aerial photograph of the inlet. The lower right panel of each figure indicates the stage of the tide that the survey transect was taken by a vertical line through the water elevation curve.

At the inlet, maximum measured currents in the water column were between 2.3 and 4.1 ft/sec (1.4 and 2.4 knots), with the larger velocities occurring during the flooding tide. Maximum flood flows in the near noon of November 16 was 79,000 ft^3/sec. Later that afternoon, the maximum ebb discharge was 59,000 ft^3/sec. For “The River”, maximum measured currents in the water column were between 0.8 and 1.3 ft/sec (0.5 and 0.8 knots), with greater velocities again during the flooding tide. Maximum flow rates computed using the ADCP velocity data were 4910 ft^3/sec during the flooding tide, and 3330 ft^3/sec during the ebbing tide.

The plots of Figures V-14 and V-15 show that velocities and flows through the inlet channel are evenly distributed across the width of the channel, with a slight peak in velocities toward the eastern end of the transect, in the main channel. For “The River”, the contour plots in Figures V-16 and V-17 show that velocities are greater between the channel mid-point and the northeastern shore, with maximum velocities occurring in the deepest part of the channel.

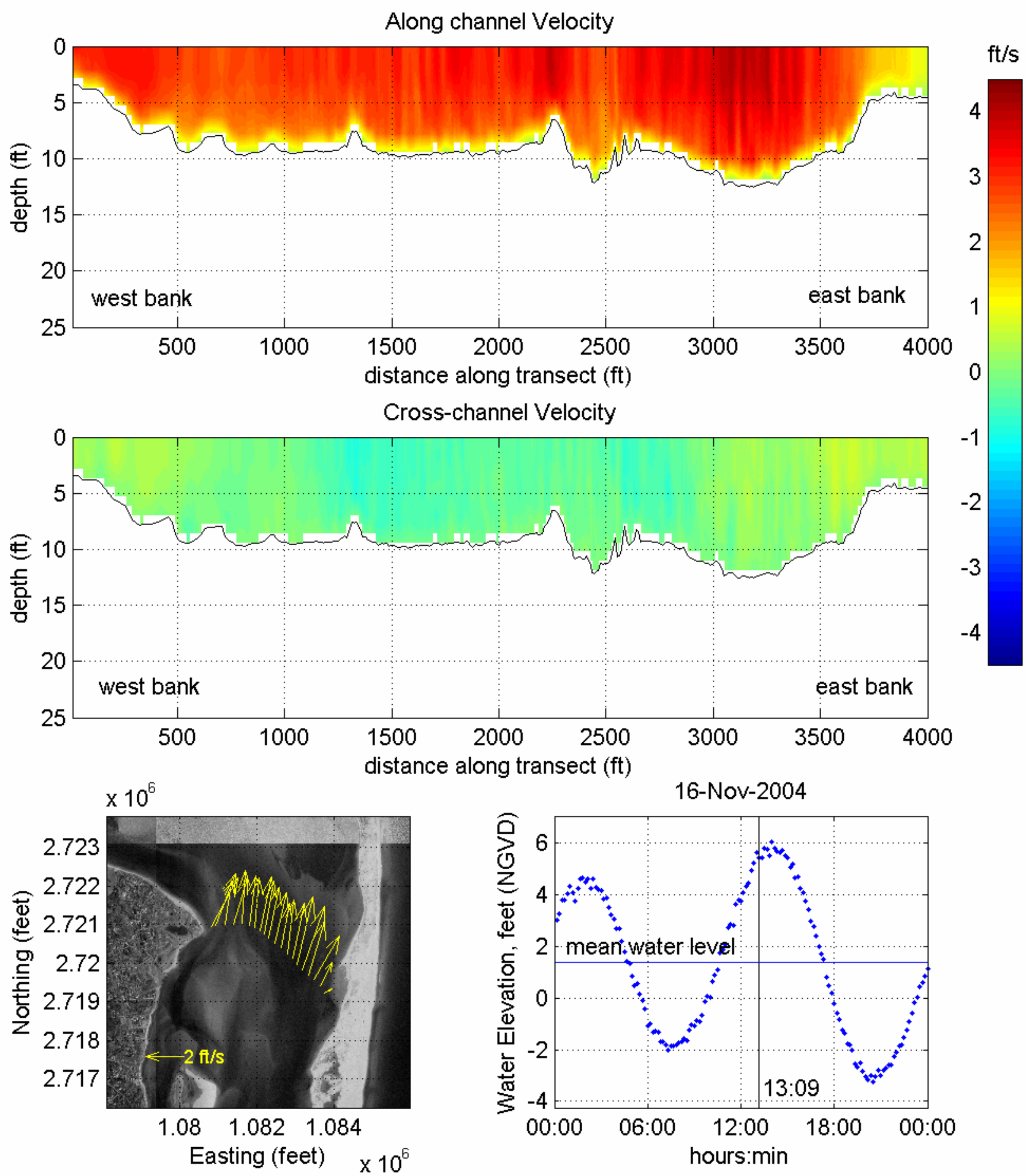


Figure V-14. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across West Bay inlet measured at 9:57 on October 24, 2001 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot of data from the offshore gauge for the survey day is also given.

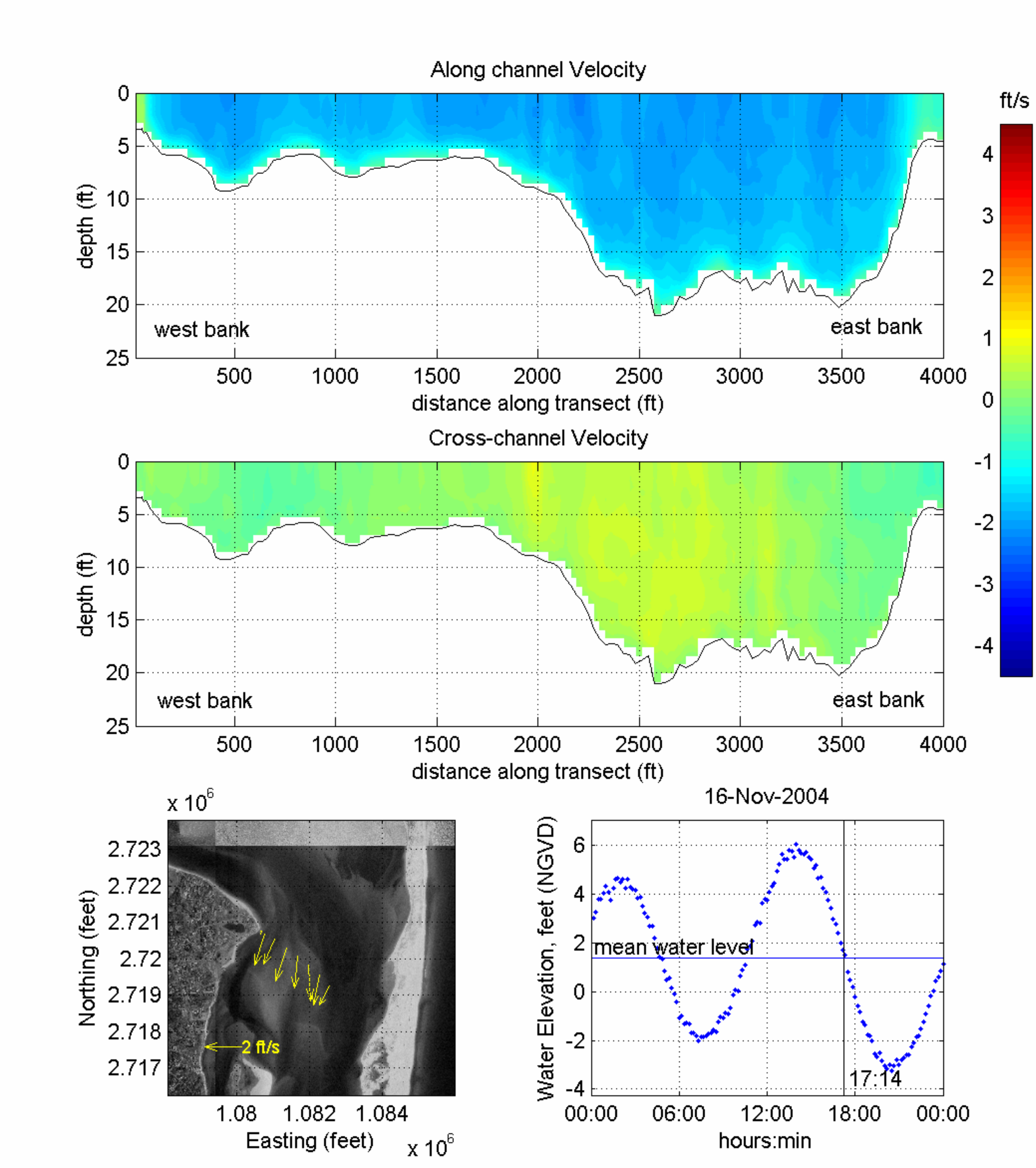


Figure V-15. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across West Bay inlet measured at 18:30 on October 24, 2001 during the period of maximum ebb tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot of data from the offshore gauge for the survey day is also given.

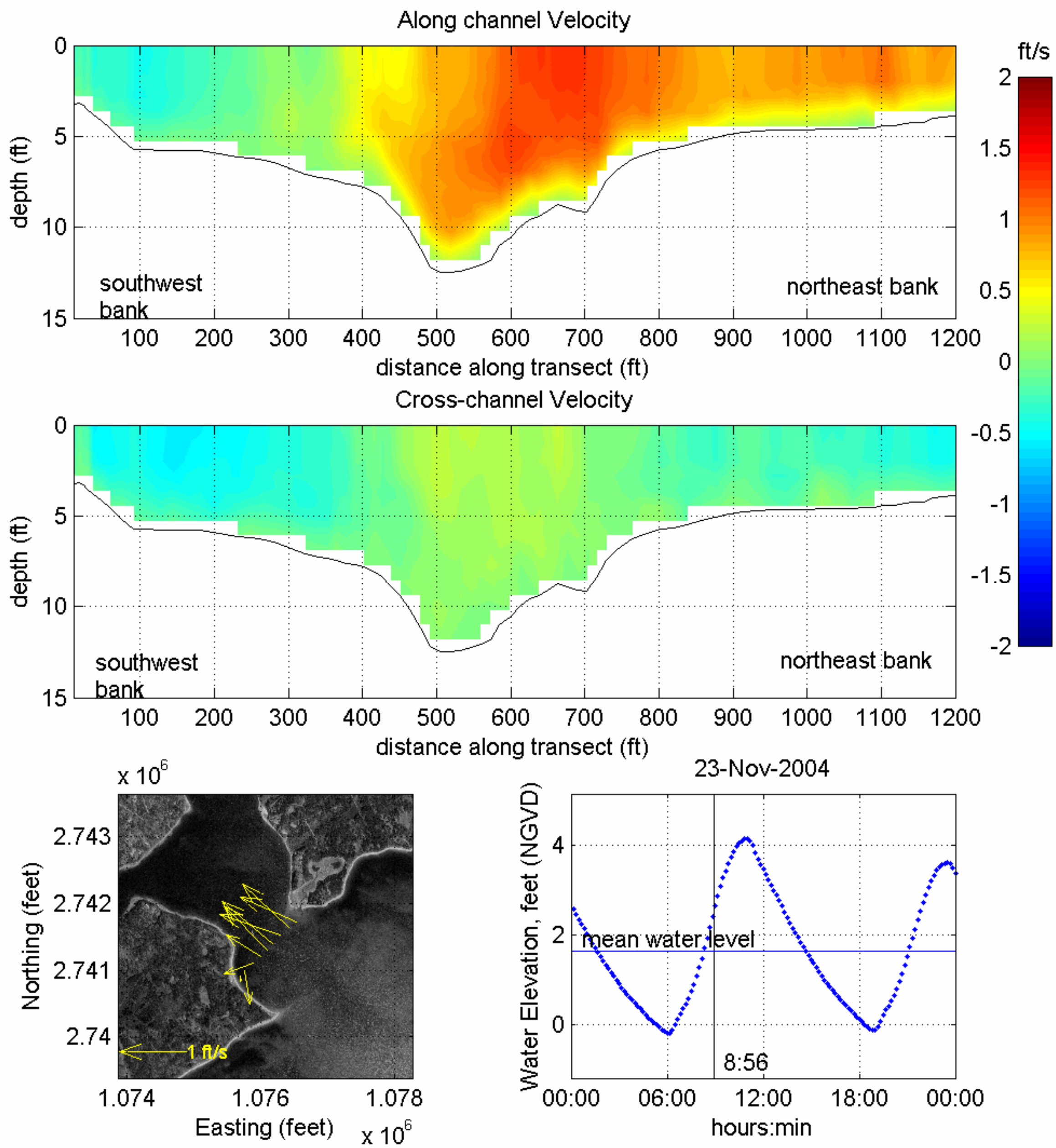


Figure V-16. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across the entrance to West Bay, measured at 10:02 on October 24, 2001 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot of data from the gauge in Meetinghouse Pond for the survey day is also given.

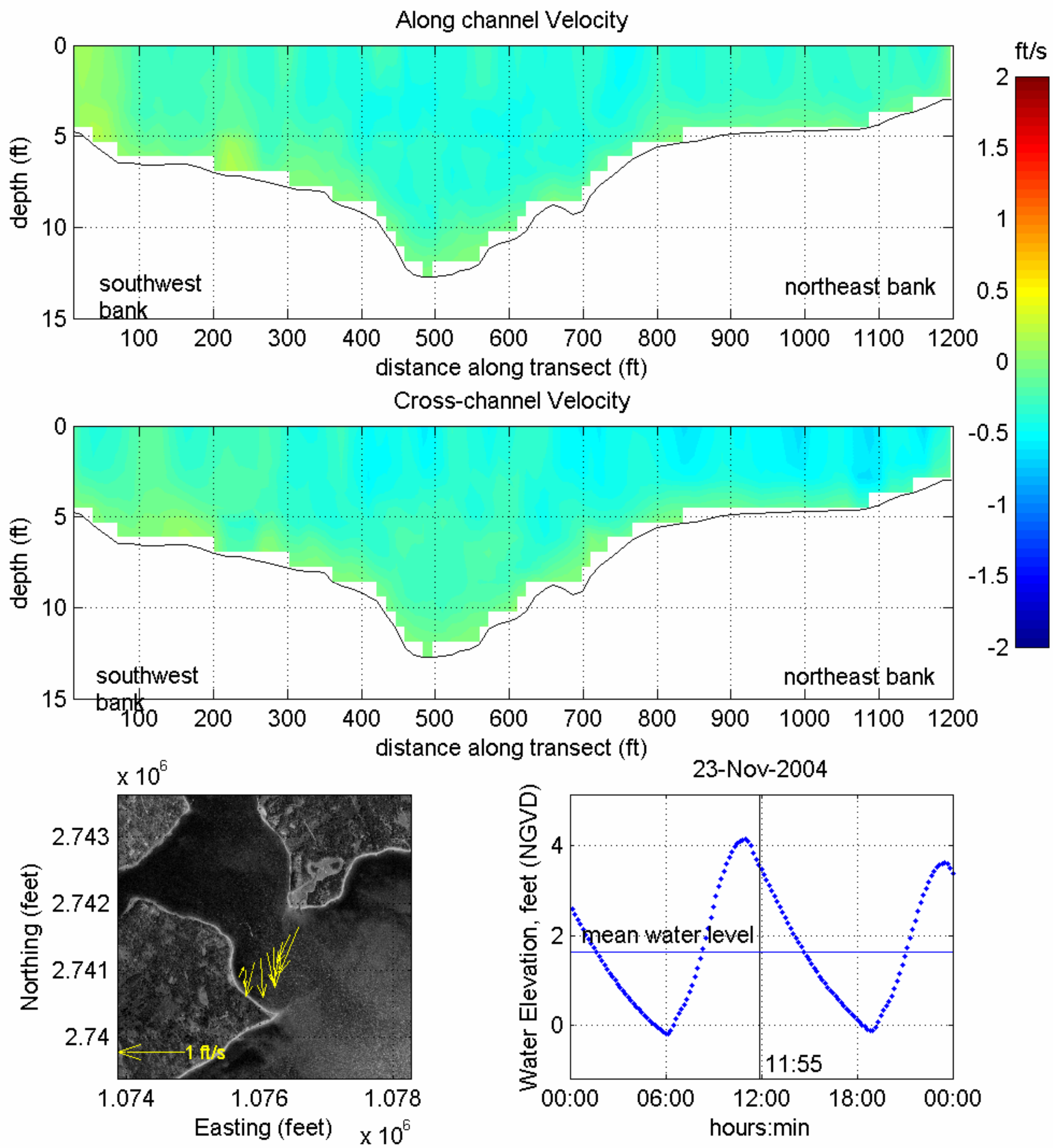


Figure V-17. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across the entrance to West Bay, measured at 18:34 on October 24, 2001 during the period of maximum ebb tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot of data from the gauge in Meetinghouse Pond for the survey day is also given.

V.4 HYDRODYNAMIC MODELING

For the modeling of the Pleasant Bay system, Applied Coastal utilized a state-of-the-art computer model to evaluate tidal circulation and flushing in these systems. The particular model employed was the RMA-2 model developed by Resource Management Associates (King, 1990). It is a two-dimensional, depth-averaged finite element model, capable of simulating transient hydrodynamics. The model is widely accepted and tested for analyses of estuaries or rivers. Applied Coastal staff members have utilized RMA-2 for numerous flushing studies on Cape Cod, including West Falmouth Harbor, Popponesset Bay, Chatham embayments (Kelley, *et al*, 2001), Falmouth "finger" Ponds (Howes *et al*, 2005), Three Bays (Kelley *et al*, 2003) and Barnstable Harbor (Wood, *et al*, 1999).

V.4.1 Model Theory

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

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

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

V.4.2 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using 2001 color digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of the Pleasant Bay system based on the tide gauge data collected offshore Nauset Beach, in the Atlantic Ocean. 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 (10) 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 aided by the use of the SMS package. 2001 digital aerial orthophotos and recent bathymetry survey data were imported to SMS, and a finite element grid was created to represent the estuary. The aerial photographs were used to determine the land boundary of the system. Bathymetry data were interpolated to the developed finite element mesh used to numerically represent the Pleasant Bay system. The completed grid consists of 6753 nodes, which describe 2339 total 2-dimensional (depth averaged) quadratic elements, and covers 7300 acres. The maximum nodal depth is –25.5 ft (NGVD 29), in the main channel of the system inlet. The completed grid mesh of Pleasant Bay is shown in Figure V-18, and grid bathymetry was shown previously in Figure V-9.

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

V.4.2.2 Boundary condition specification

Two types of boundary conditions were employed for the RMA-2 model of the Pleasant Bay system: 1) "slip" boundaries, and 2) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. A tidal boundary condition was specified at the inlet to the Bay. TDR measurements from a gauge deployed offshore Nauset Beach provided the required data.

The rise and fall of the tide in the Atlantic Ocean is the primary driving force for estuarine circulation in this system. Dynamic (time-varying) model simulations specified a new water surface elevation at the Pleasant Bay inlet every model time step of 10 minutes, which corresponds to the time step of the TDR data measurements.

V.4.2.3 Calibration

After developing the finite element grid, and specifying boundary conditions, the model for the Pleasant Bay system was calibrated. The calibration procedure ensures that the model predicts accurately what was observed in nature during the field measurement program. Numerous model simulations are required (typically 20+) for an estuary model, specifying a range of friction and eddy viscosity coefficients, to calibrate the model.

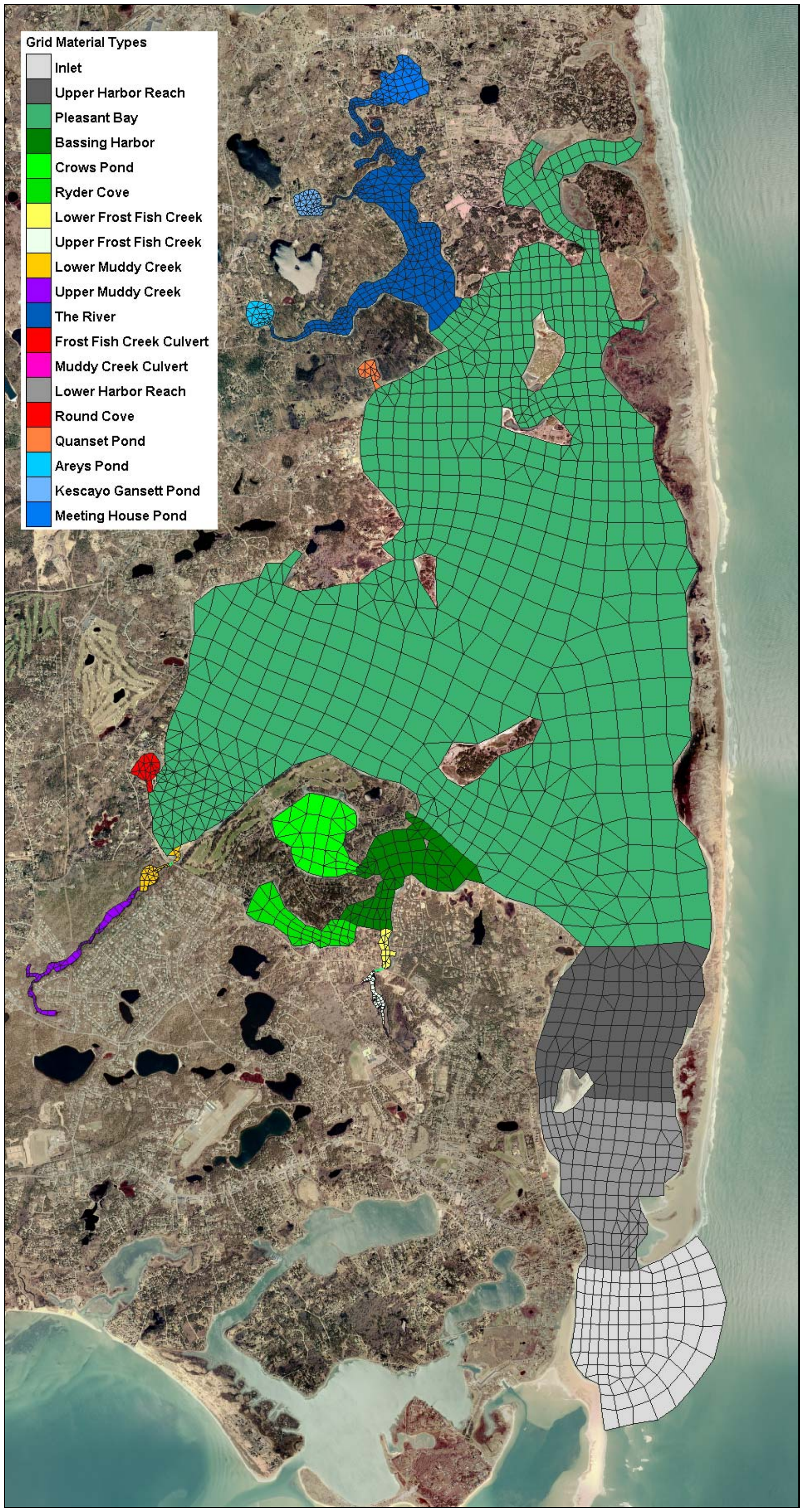


Figure V-18. Plot of hydrodynamic model grid mesh for the Pleasant Bay system of Chatham, Harwich and Orleans, MA. Color patterns designate the different model material types used to vary model calibration parameters and compute flushing rates.

Calibration of the hydrodynamic model required a close match between the modeled and measured tides in each of the sub-embayments where tides were measured (i.e., from the TDR deployments). Initially, the model was calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement was achieved, a five lunar-day period (10 tide cycles) was chosen to calibrate the model based on dominant tidal constituents. The five-day period was extracted from a longer simulation to avoid effects of model spin-up. Modeled tides for the calibration time period were evaluated for time (phase) lag and height damping of dominant tidal constituents.

The calibration was performed for a five-day period beginning November 13, 2004 1345 EST. This representative time period included the spring tide range of conditions, where the tide range and tidal currents are greatest, and model numerical stability is often most sensitive. To provide average tidal forcing conditions for model verification and the flushing analysis, a separate time period was chosen that spanned the transition between spring and neap tide ranges (bi-weekly maximum and minimum tidal ranges, respectively).

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

V.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-2 as a Manning coefficient, and is applied to grid areas by user specified material types. Initially, Manning's friction coefficient values of 0.03 were specified for all element material types. This values corresponds to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) (Henderson, 1966).

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

V.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). Typically, model turbulence coefficients were set between 80 and 500 lb-sec/ft^2. In most cases, the Pleasant Bay model was relatively insensitive to turbulent exchange coefficients. The exception was at the inlet, where higher exchange coefficient values (500 lb-sec/ft^2) were used to ensure numerical stability in this area characterized by strong turbulent flows and large velocity magnitudes.

Table V-5. Manning's Roughness coefficients used in simulations of modeled embayments. These embayment delineations correspond to the material type areas shown in Figure V-18.

System Embayment	Bottom Friction
Inlet	0.027
Upper Harbor Reach	0.028
Pleasant Bay	0.035
Bassing Harbor	0.025
Crows Pond	0.025
Ryder Cove	0.027
Lower Frost Fish Creek	0.027
Upper Frost Fish Creek	0.025
Lower Muddy Creek	0.027
Upper Muddy Creek	0.027
The River	0.027
Frost Fish Creek Culvert	0.500
Muddy Creek Culvert	0.500
Lower Harbor Reach	0.027
Round Cove	0.027
Quanset Pond	0.035
Areys Pond	0.035
Kescayo Gansett Pond	0.027
Meeting House Pond	0.027

V.4.2.3.3 Wetting and Drying

Modeled hydrodynamics were complicated by wetting/drying cycles in shallow flats and marsh areas included in the model of Pleasant Bay. A method was employed to simulate the periodic inundation and drying of tidal flats in the system. Nodal wetting and drying is a feature of RMA-2 that allows grid elements to be removed and re-inserted during the course of the model run. The wetting and drying feature has two keys 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.

V.4.2.3.4 Comparison of modeled tides and measured tide data

A best-fit of model predictions for the TDR deployment was achieved using the aforementioned values for friction and turbulent exchange. Figures V-19 through V-25 illustrate the five-day calibration simulation along with a 50-hour 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 (principle lunar semidiurnal constituent) was the highest priority since M_2 accounted for a majority of the forcing tide energy in the modeled systems. Due to the duration of the model runs, four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Table V-6 for the calibration period differ from those in Table V-2 because constituents were computed for only the five-day section of the 43-days represented in Table V-2. Table V-6 compares tidal constituent amplitude (height) and relative phase (time) for modeled and measured tides at the TDR locations. The constituent phase shows the relative timing of each separate constituent at a particular location, and also the change (or phase lag) in timing of a single constituent at different locations in an estuary.

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 0.01 ft, which is better than the order of accuracy of the tide gauges (±0.12 ft). Time lag errors were typically less than the time increment resolved by the model (1/6 hours or 10 minutes), indicating good agreement between the model and data.

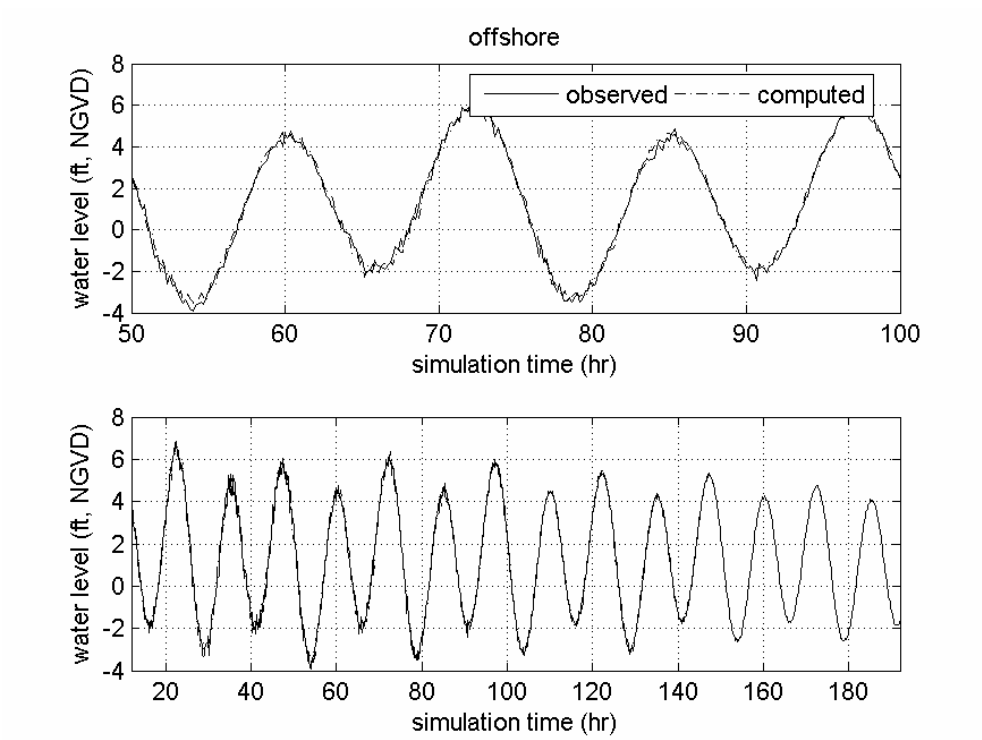


Figure V-19. Comparison of model output and measured tides for the TDR location offshore New Inlet, in the Atlantic Ocean. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

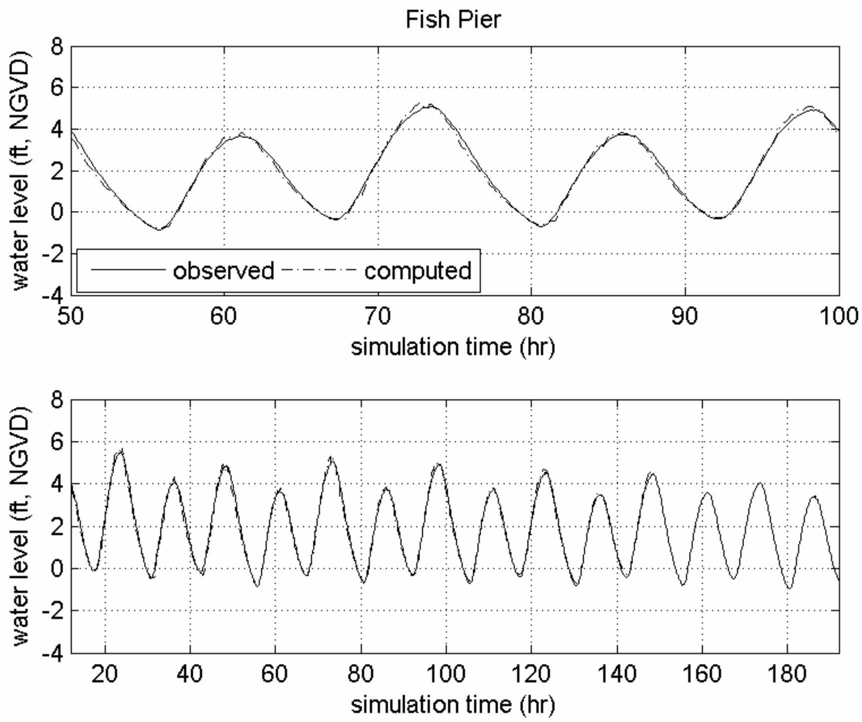


Figure V-20. Comparison of model output and measured tides for the TDR location at the Fish Pier. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

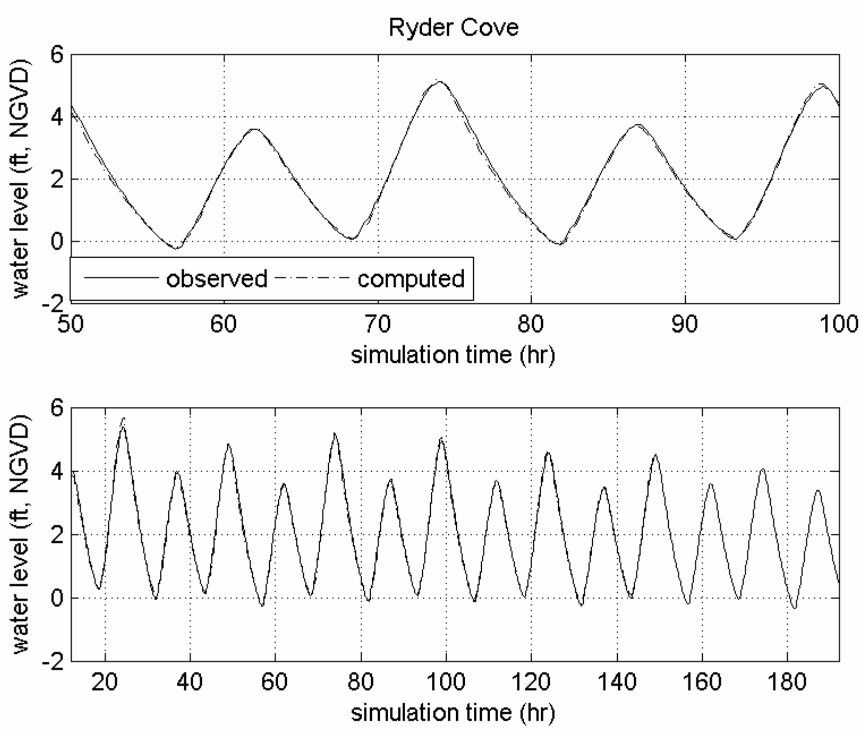


Figure V-21. Comparison of model output and measured tides for the TDR location in Ryder Cove. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

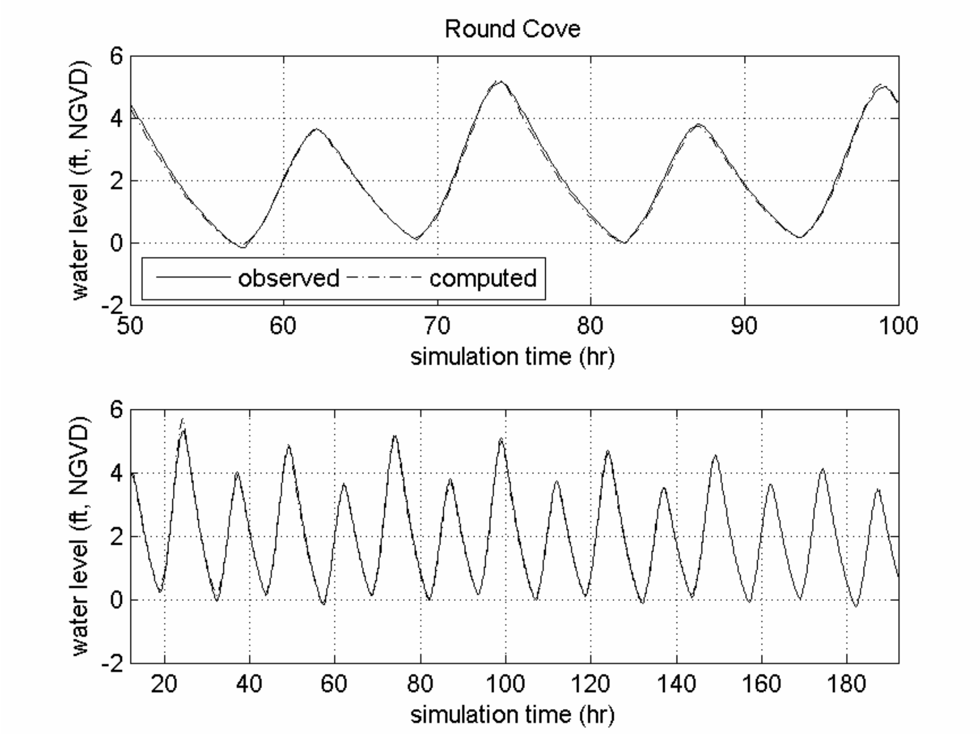


Figure V-22. Comparison of model output and measured tides for the TDR location in Round Cove. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

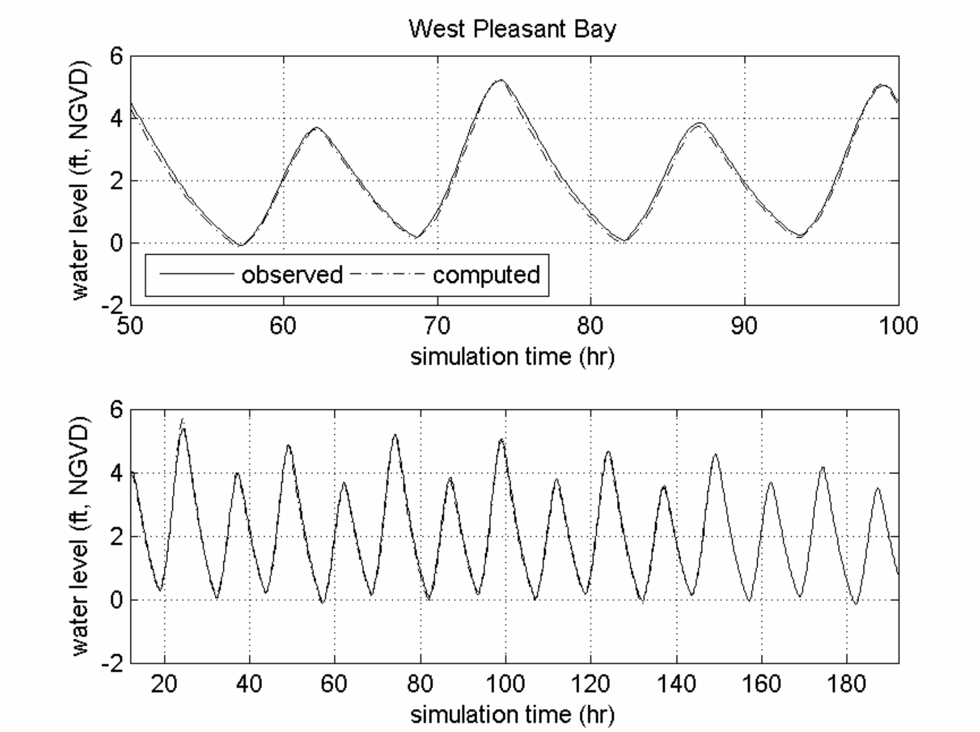


Figure V-23. Comparison of model output and measured tides for the TDR location in the western portion of Pleasant Bay. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

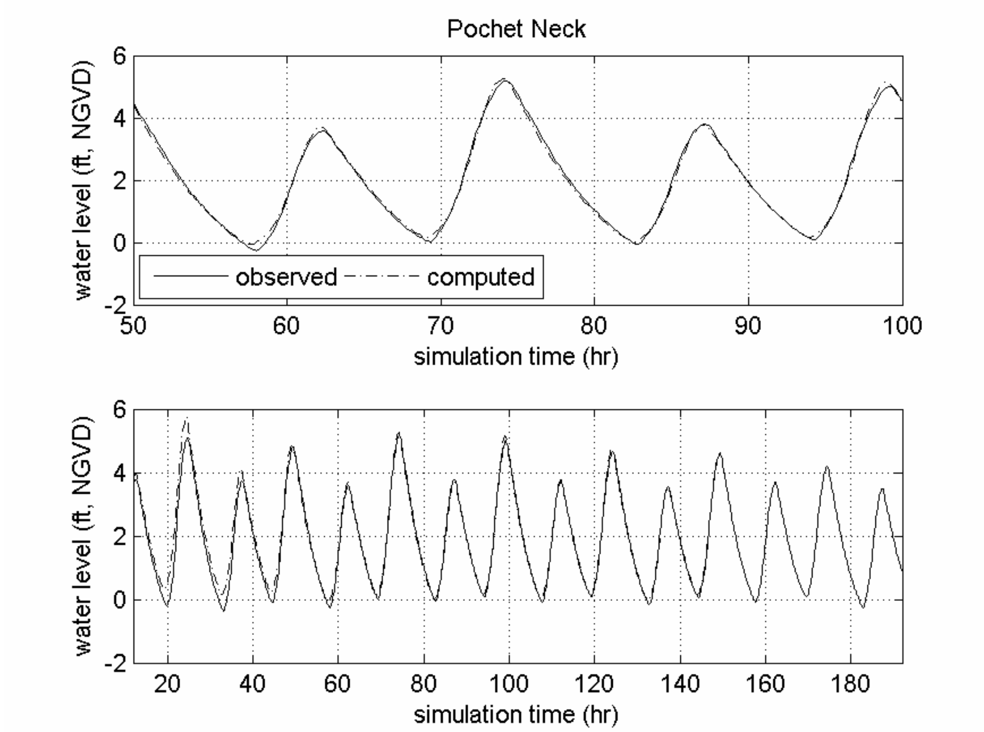


Figure V-24. Comparison of model output and measured tides for the TDR location in Pochet Neck. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

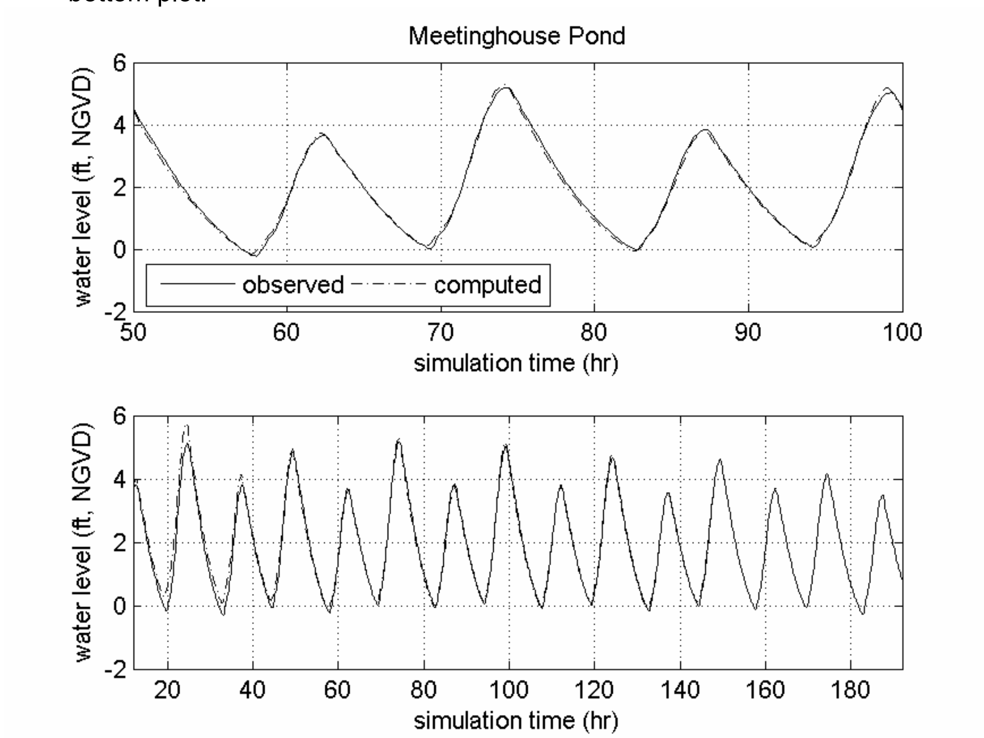


Figure V-25. Comparison of model output and measured tides for the TDR location in Meetinghouse Pond. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

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

Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Model calibration run						
Offshore	3.80	0.05	0.02	0.84	-29.9	5.6
Chatham Fish Pier	2.33	0.22	0.08	0.58	-3.5	-74.4
Ryder Cove	1.96	0.27	0.07	0.56	25.7	-6.6
West Pleasant Bay	1.92	0.34	0.08	0.56	34.1	12.3
Round Cove	1.92	0.34	0.08	0.56	34.3	12.5
Pochet Neck	1.90	0.45	0.10	0.56	41.8	25.1
Meetinghouse Pond	1.93	0.43	0.09	0.56	40.8	22.5
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Offshore	3.81	0.05	0.02	0.85	-28.0	18.2
Chatham Fish Pier	2.32	0.11	0.10	0.57	1.1	-100.3
Ryder Cove	1.98	0.21	0.08	0.56	28.7	-5.1
West Pleasant Bay	1.94	0.31	0.05	0.56	36.6	12.7
Round Cove	1.94	0.31	0.05	0.56	36.9	12.8
Pochet Neck	1.94	0.44	0.06	0.58	45.3	28.8
Meetinghouse Pond	1.95	0.43	0.05	0.56	44.7	27.4
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Offshore	-0.01	0.00	0.00	-0.01	-3.9	-13.0
Chatham Fish Pier	0.01	0.11	-0.02	0.01	-9.5	26.8
Ryder Cove	-0.02	0.06	-0.01	0.00	-6.2	-1.6
West Pleasant Bay	-0.02	0.03	0.03	0.00	-5.2	-0.4
Round Cove	-0.02	0.03	0.03	0.00	-5.4	-0.3
Pochet Neck	-0.04	0.01	0.04	-0.02	-7.2	-3.8
Meetinghouse Pond	-0.02	0.00	0.04	0.00	-8.1	-5.1

V.4.2.3.5 ADCP Verification of the Pleasant Bay System

A model verification check was possible by using collected ADCP velocity data to verify the performance of the Pleasant Bay system model. Computed flow rates from the model were compared to flow rates determined using the measured velocity data. The ADCP data survey efforts are described previously in this chapter. For the model ADCP verification, the Pleasant Bay model was run for the two separate periods covered during the ADCP surveys on November 16 and 23, 2004. Model flow rates were computed in RMA-2 at continuity lines (channel cross-sections) that correspond to the actual ADCP transects followed in each survey (i.e., across the inlet channel and the mouth to "The River").

Comparisons of the measured and modeled volume flow rates in the Pleasant Bay system are shown in Figures V-26 and V-27. For each figure, the top plot shows the flow comparison, and the lower plot shows the time series of tide elevation for the same period.

Each ADCP point (blue triangles shown on the plots) is a summation of flow measured along the ADCP transect. The 'bumps' and 'skips' of the flow rate curve (more evident in the model output) 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 inside system channels. 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.

Data comparisons at all five ADCP transect show exceptionally good agreement with the model predictions. The calibrated model accurately describes the discharge magnitude at both inlets. For both transects except R^2 correlation coefficients between data and model results are greater than 0.97. Computed RMS error for both simulation periods was less than 10%, with exceptional agreement at the inlet, where the error was computed to be less than 4%.

V.4.2.3.6 Model Circulation Characteristics

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

From the model run of the Pleasant Bay system, maximum ebb velocities in the inlet channels are slightly larger than velocities during maximum flood. Maximum depth-averaged flood velocities in the model are approximately 8.9 feet/sec (5.3 knots) at the inlet, while maximum ebb velocities are about 10.0 feet/sec (5.9 knots). Close-up views of model output are presented in Figure V-28 and V-29, which show contours of velocity magnitude along with velocity vectors that indicate flow direction, each for a single model time-step, at the portion of the tide where maximum ebb velocities occur (in Figure V-28), and for maximum flood velocities in Figure V-29.

In addition to depth-averaged velocities, the total flow rate of water flowing through a channel can be computed with the hydrodynamic model. For the flushing analysis in the next section, flow rates where computed across 11 separate transects in the Pleasant Bay system. The variation of flow as the tide floods and ebbs at the inlets is shown in the plot of flow rates in Figure V-30. Maximum flow rates occur during flooding tides in this system. During spring tides at the inlet the maximum flood flow rate reaches 111,700 ft^3/sec (3160 m^3/sec). Maximum ebb flow rates during spring tides are 75,800 ft^3/sec (2150 m^3/sec). During neap tides, the maximum tidal flow rates during flood and ebb tides are nearly half of the maximum rates during spring tide periods. The maximum neap flood discharge rate through the inlet is 67,700 ft^3/sec (1900 m^3/sec), while the maximum neap ebb flow rate is 48,100 ft^3/sec (1400 m^3/sec).

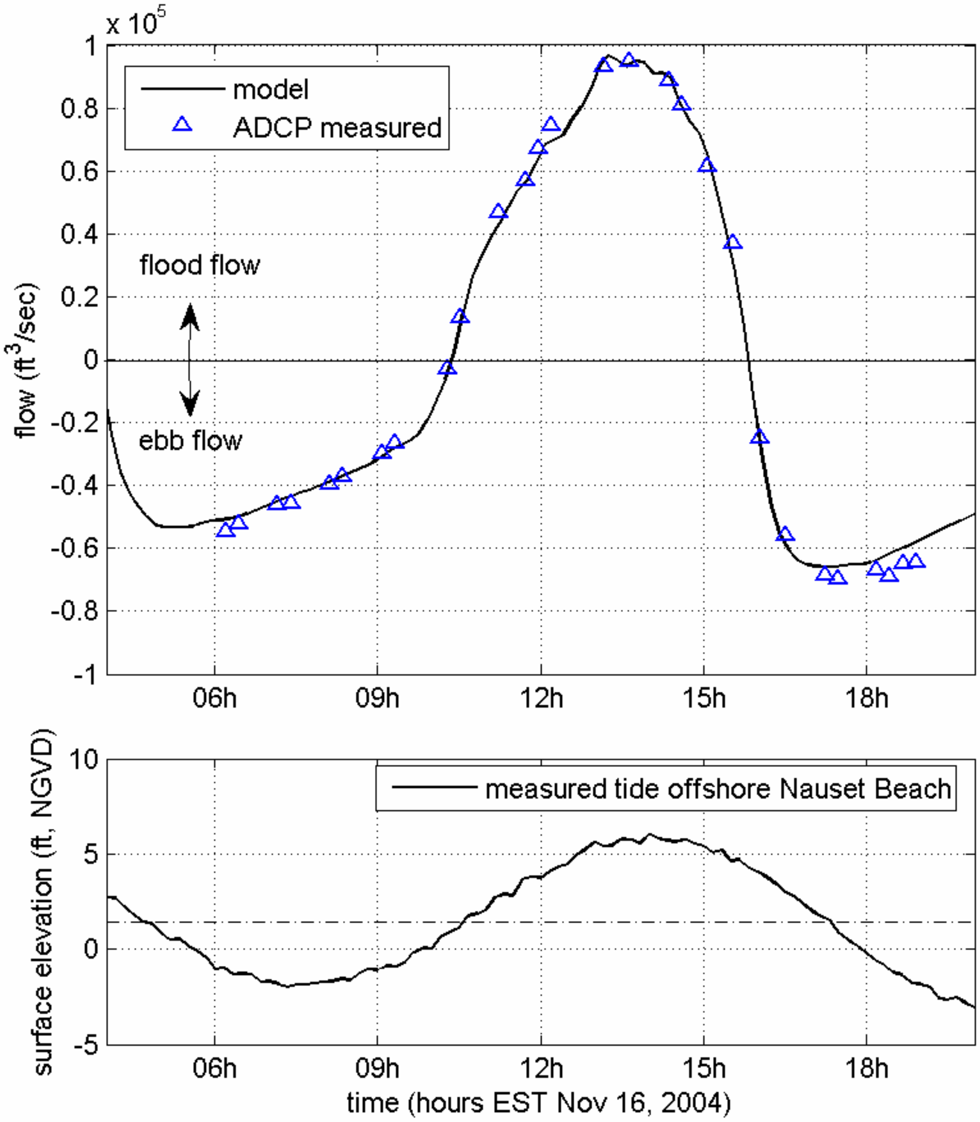


Figure V-26. Comparison of measured volume flow rates versus modeled flow rates (top plot) through the Cotuit Bay Inlet over a tidal cycle November 16, 2004. The computed RMS error for this model run was 3.8% of maximum measured flow, with a R^2 correlation coefficient of 0.99. Flood flows into the inlet are positive (+), and ebb flows out of the inlet are negative (-). The bottom plot shows the tide elevation offshore Nauset Beach.

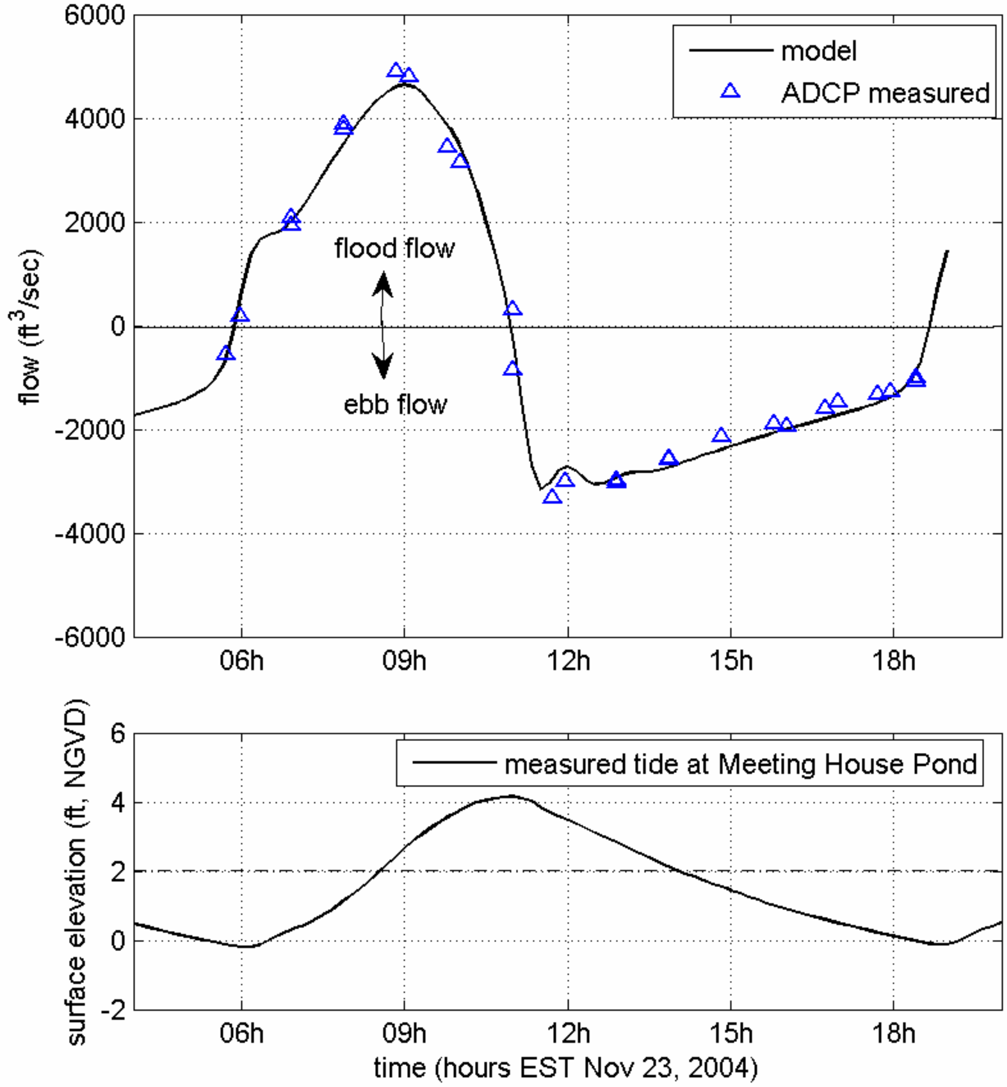


Figure V-27. Comparison of measured volume flow rates versus modeled flow rates (top plot) through the mouth of "The River", in northern Pleasant Bay, over a tidal cycle on November 23, 2004. The computed RMS error for this model run was 9.8% of maximum measured flow, with a R^2 correlation coefficient of 0.97. Flood flows into the inlet are positive (+), and ebb flows out of the inlet are negative (-). The bottom plot shows the tide elevation in Meetinghouse Pond.

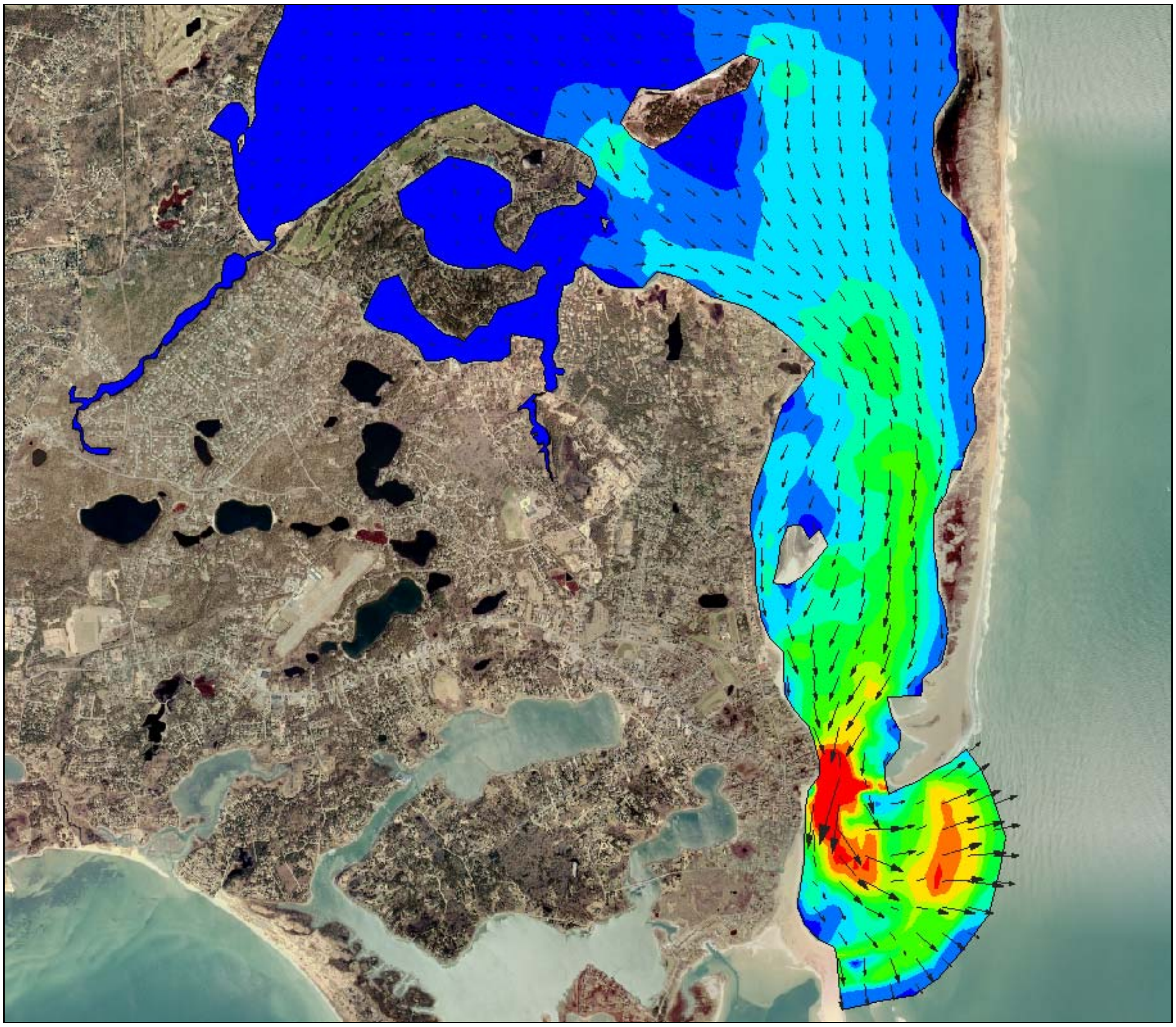


Figure V-28. Example of hydrodynamic model output for a single time step where maximum ebb velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

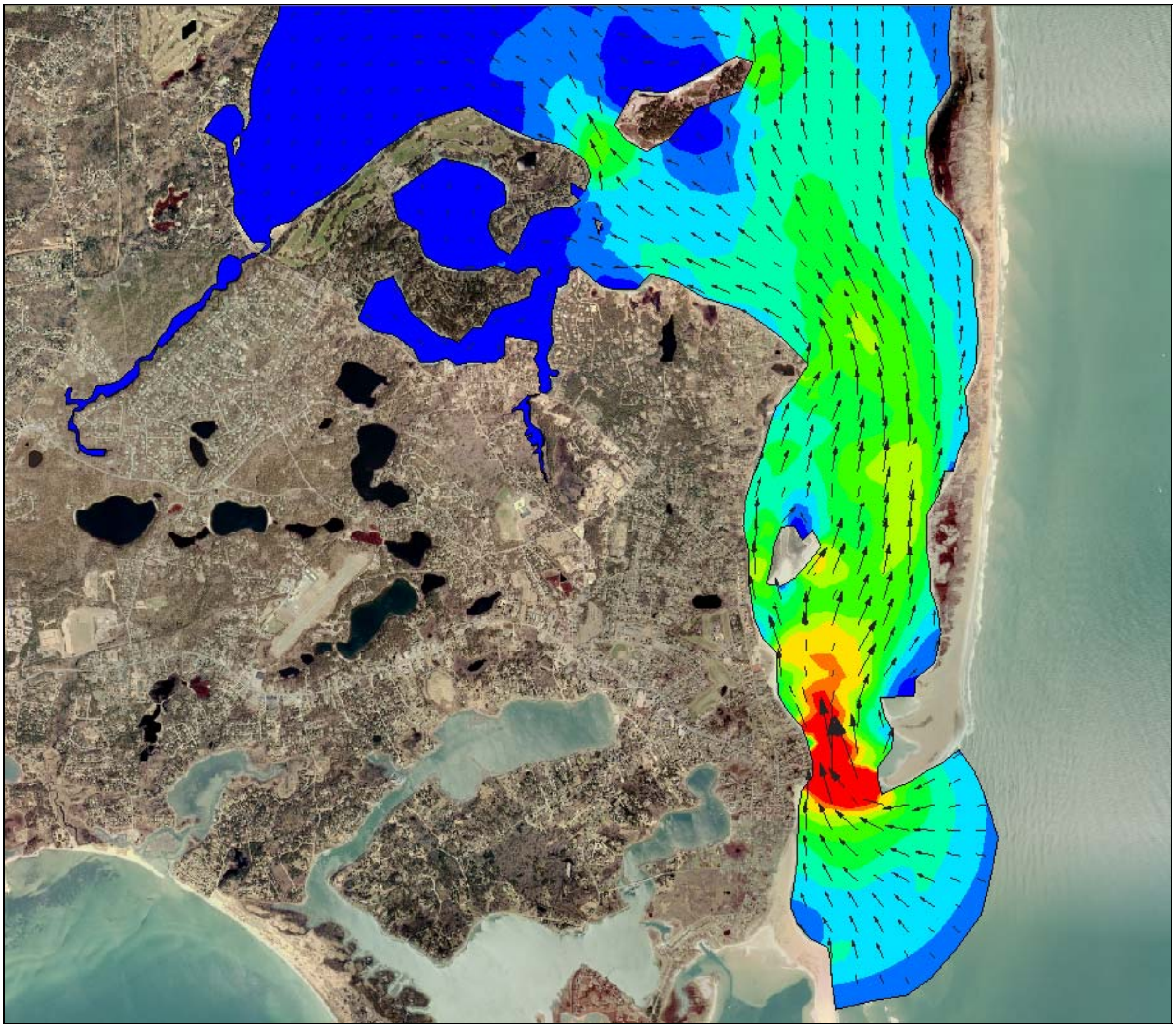


Figure V-29. Example of hydrodynamic model output for a single time step where maximum ebb velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

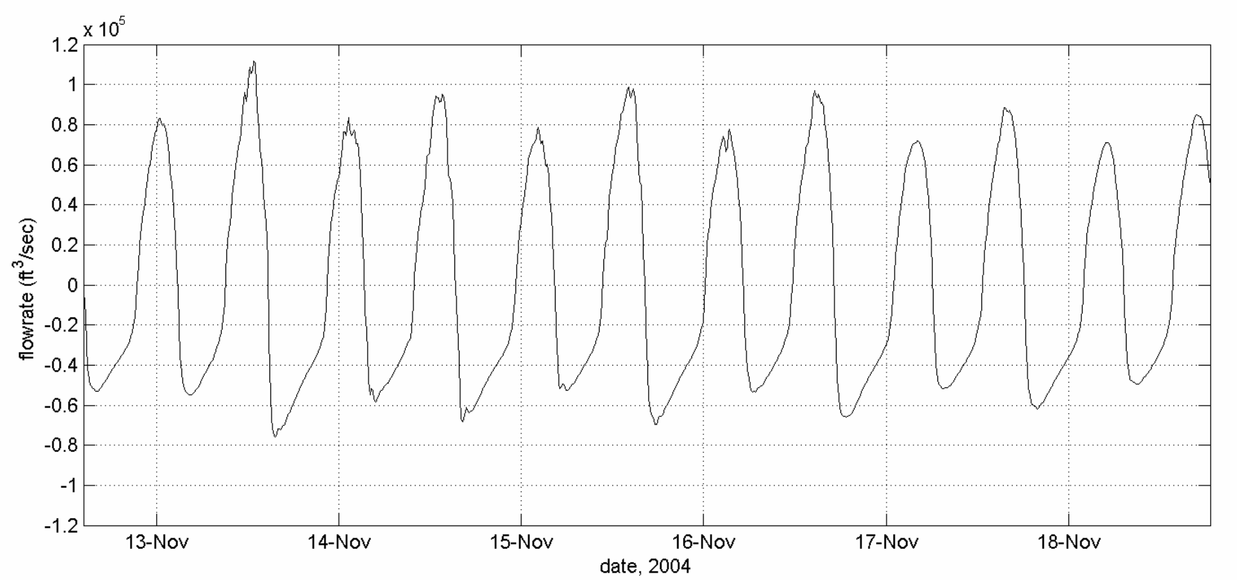


Figure V-30. Pleasant Bay model computed flowrate for transect across the inlet channel, showing the variation of flood and ebb discharges of the Pleasant Bay system through 12 tide cycles.

V.5 FLUSHING CHARACTERISTICS

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

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

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

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

In addition to system residence times, a second residence, the **local residence time**, was defined as the average time required for a water parcel to migrate from a location within a sub-embayment to a point outside the sub-embayment. Using Meetinghouse Pond as an example, the **system residence time** is the average time required for water to migrate from Meetinghouse Pond, through “The River”, through the main basin of Pleasant Bay, and into the Atlantic Ocean, where the **local residence time** is the average time required for water to migrate from Meetinghouse Pond to just “The River” (not all the way to the ocean). 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 Pleasant Bay system this approach is applicable, since it assumes the main basin of the system has relatively decent quality water relative to its sub-systems and sub-embayments.

The rate of pollutant/nutrient loading and the quality of water outside the estuary both must be evaluated in conjunction with residence times to obtain a clear picture of water quality. It is impossible to evaluate an estuary’s health based solely on flushing rates. Efficient tidal flushing (low residence time) is not an indication of high water quality if pollutants and nutrients are loaded into the estuary faster than the tidal circulation can flush the system. Neither are low residence times an indicator of high water quality if the water flushed into the estuary is of poor quality. Advanced understanding of water quality will be obtained from the calibrated hydrodynamic model by extending the model to include pollutant/nutrient dispersion. The water quality model will provide a valuable tool to evaluate the complex mechanisms governing estuarine water quality in the Pleasant Bay system.

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in the system, model results were used to compute residence times. Residence times were computed for the entire estuary, as well 11 sub-embayments within the system. In addition, **system** and **local residence times** were computed to indicate the range of conditions possible for the system.

Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism). Units then were converted to days. The volume of the entire estuary was computed as cubic feet. Model divisions used to define the system sub-embayments include 1) the entire Pleasant Bay system; 2) Bassing Harbor, including Crows Pond, Ryder Cove and Frost Fish Creek ; 3) Muddy Creek; 4) The River, including Areys Pond, Kescayo Gansett Pond and Meetinghouse Pond; 5) Round Cove; 6) Pah Wah Pond; and separately 7) Areys Pond; 8) Kescayo Gansett Pond and 9) Meetinghouse Pond. These system divisions follow the model material type areas designated in Figure V-18. Sub-embayment mean volumes and tide prisms are presented in Table V-7.

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

The computed flushing rates for the Pleasant Bay system show that as a whole, the system flushes well. A flushing time of 0.9 days for the entire estuary shows that on average, water is resident in the system less than one days. All system sub-embayments utilized in this analysis, except for Muddy Creek, have local flushing times that are equal to or less than 2 days. Round Cove has the shortest local flushing time, because of its small mean sub-embayment volume, relative to its tide prism.

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

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Pleasant Bay	2,076,848,000	1,190,817,000
Bassing Harbor	109,139,000	66,133,000
Crows Pond	50,208,000	21,898,000
Ryder Cove	18,070,000	12,534,000
Muddy Creek	5,541,000	806,000
The River	96,032,000	60,199,000
Round Cove	2,913,000	2,738,000
Pah Wah Pond	2,341,000	1,538,000
Areys Pond	5,474,000	2,623,000
Kescayo Gansett Pond	6,330,000	2,864,000
Meetinghouse Pond	19,406,000	8,167,000

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

Embayment	System Residence Time (days)	Local Residence Time (days)
Pleasant Bay	0.9	0.9
Bassing Harbor	16.3	0.9
Crows Pond	49.1	1.2
Ryder Cove	85.7	0.7
Muddy Creek	1333.5	3.6
The River	17.9	0.8
Round Cove	392.5	0.6
Pah Wah Pond	698.8	0.8
Areys Pond	409.7	1.1
Kescayo Gansett Pond	375.3	1.1
Meetinghouse Pond	131.6	1.2

The low local residence times in all areas of the Pleasant Bay system show that they would likely have good water quality if the system water with which it exchanges also has good water quality. For example, the water quality of Pah Wah Pond would likely be good as long as the water quality of the main basin of Pleasant Bay was also good. Actual water quality would still also depend upon the total nutrient load to each embayment.

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

Based on our knowledge of estuarine processes, we estimate that the combined errors associated with the method applied to compute residence times are within 10% to 15% of "true" residence times, for the Pleasant Bay system. Possible errors in computed residence times can be linked to two sources: the bathymetry information and simplifications employed to calculate residence time. In this study, the most significant errors associated with the bathymetry data result from the process of interpolating the data to the finite element mesh, which was the basis for all the flushing volumes used in the analysis. In addition, limited topographic measurements were available in some of the smaller sub-embayments of the system.

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

VI. WATER QUALITY MODELING

VI.1 DATA SOURCES FOR THE MODEL

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

VI.1.1 Hydrodynamics and Tidal Flushing in the 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 calibrated hydrodynamic model representing the transport of water within the Pleasant Bay system. Files of node locations and node connectivity for the RMA-2 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 the 7 day period beginning November 13, 2004 1345 EST. This period corresponds to that used in the flushing analysis presented in Chapter V. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period thereby allowing the model to reach a dynamic “steady state” and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayments

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

VI.1.3 Measured Nitrogen Concentrations in the Embayments

In order to create a model that realistically simulates bioactive nitrogen (DIN+PON) concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in the area map presented in Figure VI-1. The multi-year averages present the “best” comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data are the minimum required to provide a baseline for MEP analysis. Typically, ten years of data (collected between 1995 and 2004) were available for stations monitored by in Pleasant Bay.

Table VI-1. Measured total (DIN+PON+DON) and bioactive nitrogen (DIN+PON) data and modeled bioactive nitrogen concentrations for the Pleasant Bay estuarine system used in the model calibration plots of Figures VI-2 and VI-3. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of the separate yearly means. Data represented in this table were collected in the summers of 2000 through 2005. N represents sample size. The sentinel threshold stations are in bold print and depicted in Figure VI-1.

Bioactive Nitrogen	monitoring station	Total Nitrogen			Bioactive Nitrogen			model min (mg/L)	model max (mg/L)	model average (mg/L)
		data mean (mg/L)	s.d. all data (mg/L)	N	data mean (mg/L)	s.d. all data (mg/L)	N			
Meetinghouse Pond	PBA-16	0.724	0.218	83	0.407	0.351	90	0.351	0.401	0.380
Meetinghouse Pond	WMO-10	0.979	0.290	28	0.279	0.098	30	0.210	0.322	0.261
The River - upper	WMO-09	0.862	0.235	29	0.252	0.072	29	0.203	0.286	0.239
The River – mid	WMO-08	0.846	0.248	23	0.222	0.060	23	0.187	0.235	0.211
Lonnies Pond (Kescayo Ganset Pond)	PBA-15	0.777	0.188	80	0.281	0.103	86	0.241	0.260	0.250
Areys Pond	PBA-14	0.731	0.109	83	0.304	0.092	91	0.282	0.314	0.297
Namequoit River - upper	WMO-6	0.829	0.206	21	0.300	0.101	23	0.203	0.272	0.239
Namequoit River - lower	WMO-7	0.728	0.168	20	0.241	0.087	22	0.185	0.245	0.216
The River - lower	PBA-13	0.561	0.102	72	0.175	0.060	78	0.166	0.220	0.195
Pochet – upper	WMO-05	0.838	0.266	27	0.283	0.106	28	0.211	0.309	0.269
Pochet - lower	WMO-04	0.777	0.210	24	0.241	0.076	24	0.175	0.257	0.209
Pochet – mouth	WMO-03	0.716	0.239	39	0.180	0.063	39	0.163	0.202	0.183
Little Pleasant Bay - head	**PBA-12**	0.773	0.280	83	0.183	0.093	84	0.145	0.203	0.178
Little Pleasant Bay - main basin	PBA-21	0.565	0.174	51	0.135	0.038	52	0.133	0.187	0.162
Paw Wah Pond	PBA-11	0.707	0.216	75	0.268	0.160	79	0.231	0.286	0.257
Little Quanset Pond	WMO-12	0.599	0.116	22	0.205	0.071	24	0.220	0.240	0.229
Quanset Pond	WMO-01	0.562	0.149	79	0.189	0.063	87	0.176	0.208	0.191
Round Cove	PBA-09	0.707	0.230	83	0.246	0.097	84	0.222	0.266	0.241
Muddy Creek - upper	PBA-05a	1.257	0.368	25	0.700	0.411	27	0.660	0.690	0.674
Muddy Creek - lower	PBA-05	0.574	0.097	40	0.243	0.094	46	0.260	0.308	0.286
Pleasant Bay - head	PBA-08	0.439	0.099	83	0.162	0.063	86	0.132	0.162	0.149
Pleasant Bay - off Quanset Pond	WMO-02	0.555	0.144	34	0.174	0.049	38	0.153	0.166	0.160
Pleasant Bay- upper Strong Island	PBA-19	0.728	0.237	39	0.169	0.113	42	0.094	0.148	0.117
Pleasant Bay - mid west basin	PBA-07	0.434	0.118	79	0.161	0.054	84	0.163	0.174	0.168
Pleasant Bay - off Muddy Creek	PBA-06	0.489	0.117	67	0.188	0.057	70	0.187	0.199	0.192
Pleasant Bay - Strong Island channel	PBA-20	0.566	0.222	44	0.141	0.044	47	0.094	0.155	0.124
Ryders Cove - upper	**PBA-03**	0.718	0.255	97	0.254	0.114	100	0.234	0.260	0.250
Ryders Cove - lower	**CM-13**	0.417	0.071	86	0.159	0.044	92	0.117	0.196	0.158
Frost Fish - lower	CM-14	1.158	0.395	44	0.349	0.296	45	0.155	0.434	0.243
Crows Pond	PBA-04	0.838	0.325	96	0.208	0.093	97	0.158	0.165	0.162
Bassing Harbor	PBA-02	0.489	0.161	37	0.121	0.035	38	0.097	0.158	0.127
Pleasant Bay - lower	PBA-18	0.463	0.168	47	0.123	0.040	47	0.094	0.148	0.116
Chatham Harbor - upper	PBA-01	0.433	0.198	87	0.105	0.036	90	0.094	0.132	0.104
Chatham Harbor – lower (by CH buoy)	PBA-17	0.349	0.134	2	0.100	0.010	2	0.094	0.121	0.099
Chatham Harbor – lower (Flood Tide)	PBA-17a	0.232	0.044	17	0.094	0.020	18	-	-	-

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

VI.2 MODEL DESCRIPTION AND APPLICATION

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of nitrogen loading in the Pleasant Bay estuarine system. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to simulate the fluid dynamics of Pleasant Bay. 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 systems in Falmouth (Howes *et al*., 2005); Mashpee, MA (Howes *et al*., 2004), Barnstable (Howes *et al*., 2005) and Chatham, MA (Howes *et al*., 2003).

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

VI.2.1 Model Formulation

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

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

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

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

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

VI.2.2 Water Quality Model Setup

Required inputs to the RMA-4 model include a computational mesh, computed water elevations and velocities at all nodes of the mesh, constituent mass loading, and spatially varying values of the dispersion coefficient. Because the RMA-4 model is part of a suite of integrated computer models, the finite-element meshes and the resulting hydrodynamic simulations previously developed for Pleasant Bay also were used for the water quality constituent modeling portion of this study.

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

VI.2.3 Boundary Condition Specification

Mass loading of nitrogen into each model included 1) sources developed from the results of the watershed analysis, 2) estimates of direct atmospheric deposition, and 3) summer benthic regeneration. Nitrogen loads from each separate sub-embayment watershed were distributed across the sub-embayment. For example, the watershed loads for Meetinghouse Pond were evenly distributed at grid cells that formed the perimeter of the sub-embayment. Combined benthic regeneration and direct atmospheric deposition loads were evenly distributed among another sub-set of grid cells which form in the interior portion of each basin.

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

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. TN concentrations of the incoming water are set at the value designated for the open boundary. The boundary concentration Atlantic Ocean region offshore Pleasant Bay was set at 0.094 mg/L (bioactive N), based on Chatham data collected in the summer of 2005 and analyzed by SMAST.

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Meetinghouse Pond	6.197	0.584	14.365
The River – upper	2.773	0.288	6.263
The River – lower	3.879	2.241	10.480
Lonnies Pond	2.441	0.225	1.591
Areys Pond	1.304	0.181	5.996
Namequoit River	2.737	0.523	14.570
Paw Wah Pond	1.860	0.082	3.630
Pochet Neck	8.422	1.767	-0.791
Little Pleasant Bay	7.496	24.023	37.226
Quanset Pond	1.781	0.170	5.988
Tar Kiln Stream	6.123	0.066	-
Round Cove	4.225	0.170	8.416
The Horseshoe	0.638	0.063	-
Muddy Creek - upper	9.981	0.162	4.560
Muddy Creek - lower	8.477	0.205	-1.226
Pleasant Bay	23.159	19.153	149.013
Pleasant Bay/Chatham Harbor Channel	-	17.786	-40.192
Bassing Harbor - Ryder Cove	9.819	1.296	9.356
Bassing Harbor - Frost Fish Creek	2.904	0.096	-0.154
Bassing Harbor - Crows Pond	4.219	1.389	0.612
Bassing Harbor	1.668	1.071	-4.976
Chatham Harbor	17.099	14.153	-40.208
TOTAL - Pleasant Bay System	**127.203**	**85.693**	**184.519**

VI.2.4 Model Calibration

Calibration of the bioactive nitrogen model of Pleasant Bay proceeded by changing model dispersion coefficients so that model output of nitrogen concentrations matched measured data. Generally, several model runs of each system were required to match the water column measurements. Dispersion coefficient (*E*) values were varied through the modeled system by setting different values of *E* for each grid material type, as designated in 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 (from tides or atmospheric forcing). Generally, the relatively quiescent main basin of Pleasant Bay and its tributary sub-embayment systems required values of *E* that are lower compared to the riverine estuary systems evaluated by Fischer, *et al.*, (1979). Observed values of *E* in these calmer areas typically range between order 10 and order 0.001 m^2/sec (USACE, 2001). The final values of *E* used in each sub-embayment of the modeled system are presented in Table VI-3. These values were used to develop the “best-fit” total nitrogen model calibration. For the case of bioactive N 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.

A contour plot of calibrated model output is shown in Figure VI-2. In this figure, color contours indicate nitrogen concentrations throughout the model domain. The output in these figures show average bioactive nitrogen concentrations, computed using the full 5-tidal-day model simulation output period.

Comparisons between calibrated model output and measured nitrogen concentrations are shown in plots presented in Figures VI-3 and VI-4. In these plots, means of the water column data and a range of two standard deviations of the annual means at each individual station are plotted against the modeled maximum, mean, and minimum concentrations output from the model at locations which corresponds to the water quality monitoring stations shown in Figure VI-1.

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

Also presented in this figure are unity plot comparisons of measured data verses modeled target values for each system. The R2 correlation between the model output and measured data is greater than 0.96 and the computed root mean squared (rms) error is 0.021 mg/L, which demonstrates the exceptional fit of the model for this system.

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

Embayment Division	E m^2/sec
Pleasant Bay Inlet	100.0
Lower Chatham Harbor	100.0
upper Chatham Harbor	100.0
Pleasant Bay - east basin	70.0
Pleasant Bay - West Basin	10.0
Little Pleasant Bay	20.0
Bassing Harbor - Main Basin	15.0
Crows Pond	0.5
Ryder Cove	0.8
Lower Frost Fish Creek	1.5
Upper Frost Fish Creek	5.0
Frost Fish Creek Culvert	10.0
Lower Muddy Creek	50.0
Upper Muddy Creek	10.0
Muddy Creek Culvert	50.0
Round Cove	2.5
Quonset Pond	0.5
Paw Wah Pond	1.0
The River -lower	60.0
The River - upper	30.0
Namequoit River	20.0
Areys Pond	10.0
Lonnies Pond (Kescayo Ganset) Creek	0.5
Kescayo Ganset (Lonnies) Pond	0.5
Meetinghouse Pond	0.5
Pochet Neck	1.0

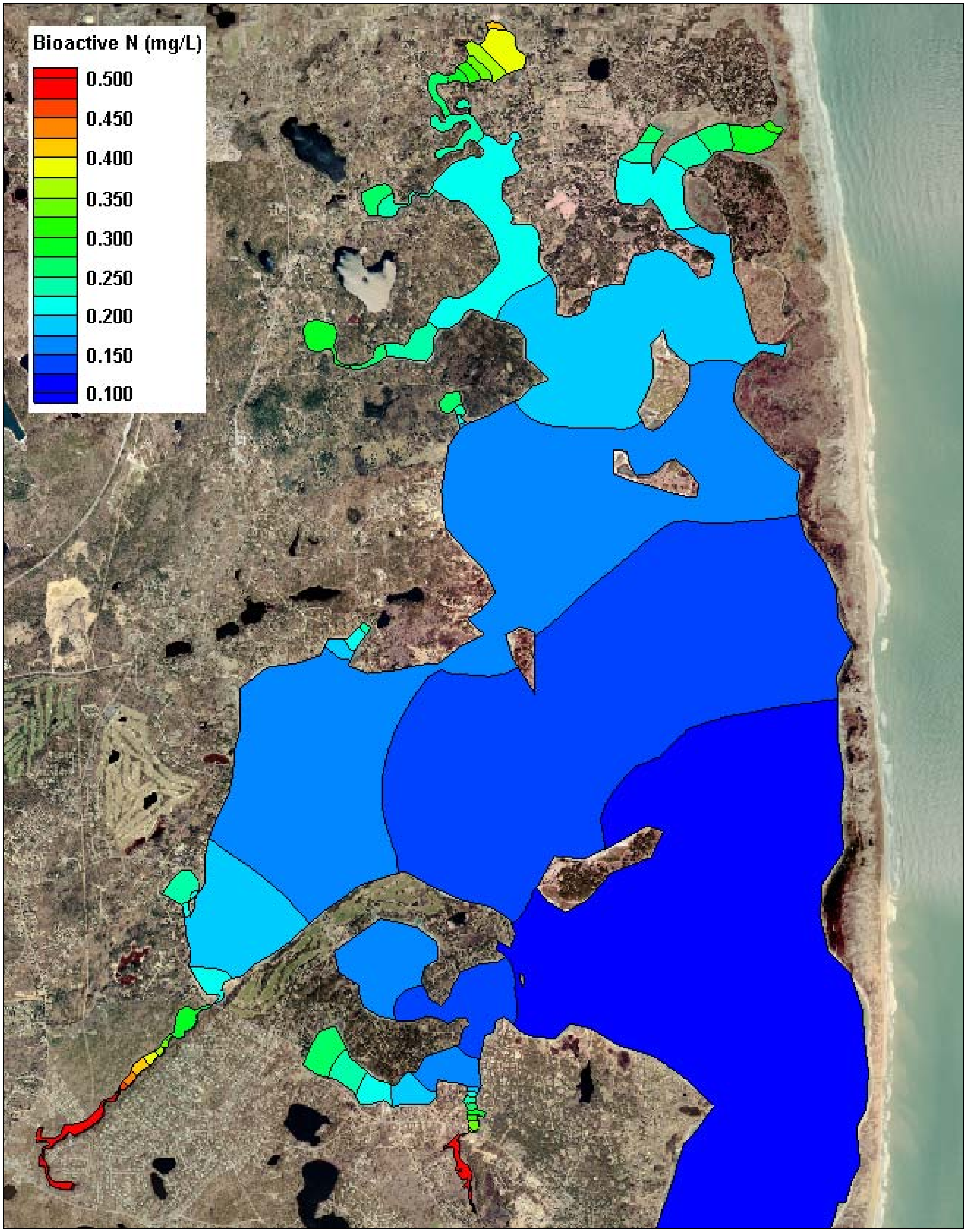


Figure VI-2. Contour plot of average bioactive (DIN+PON) nitrogen concentrations from results of the present conditions loading scenario, for the Pleasant Bay system.

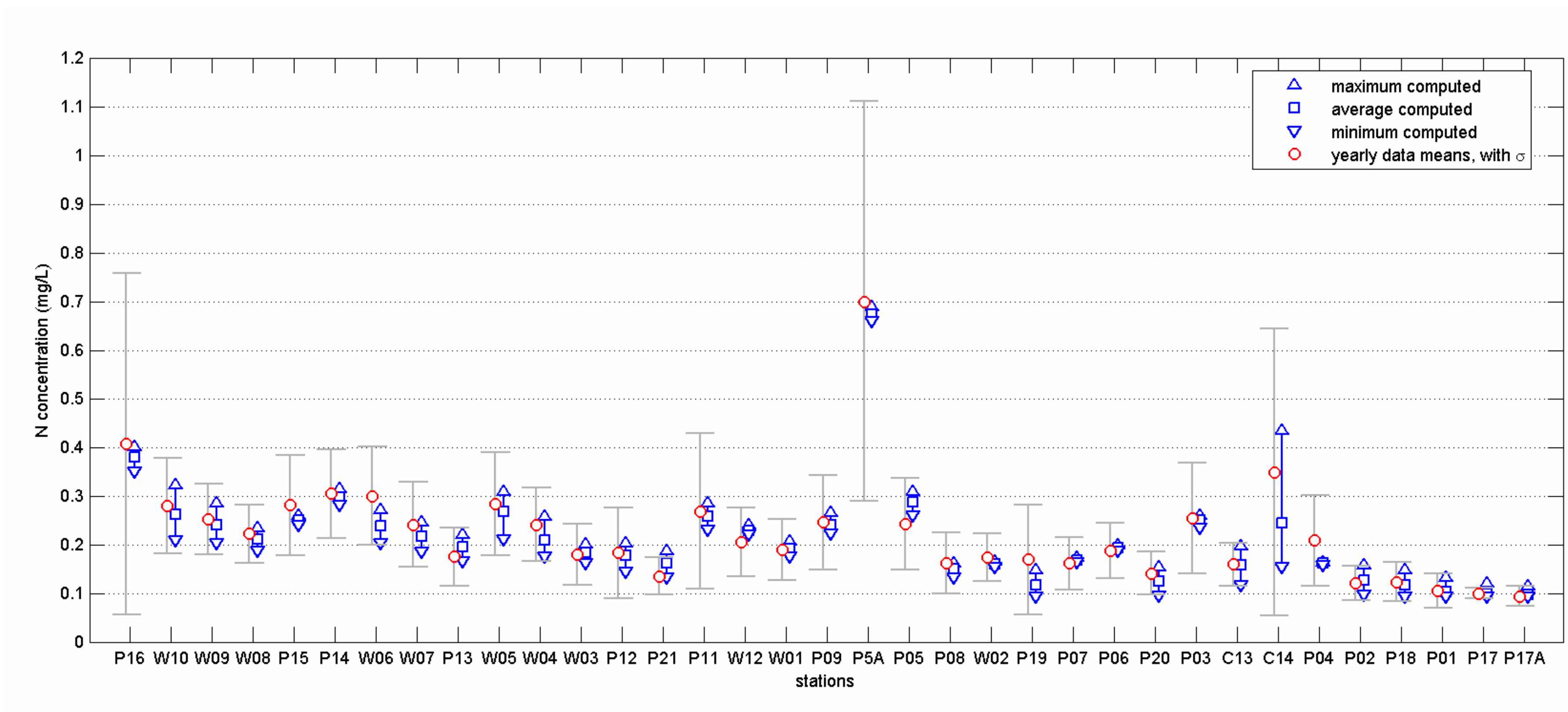


Figure VI-3. Comparison of measured bioactive (DIN+PON) nitrogen concentrations and calibrated model output at stations in the Pleasant Bay system. Station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset

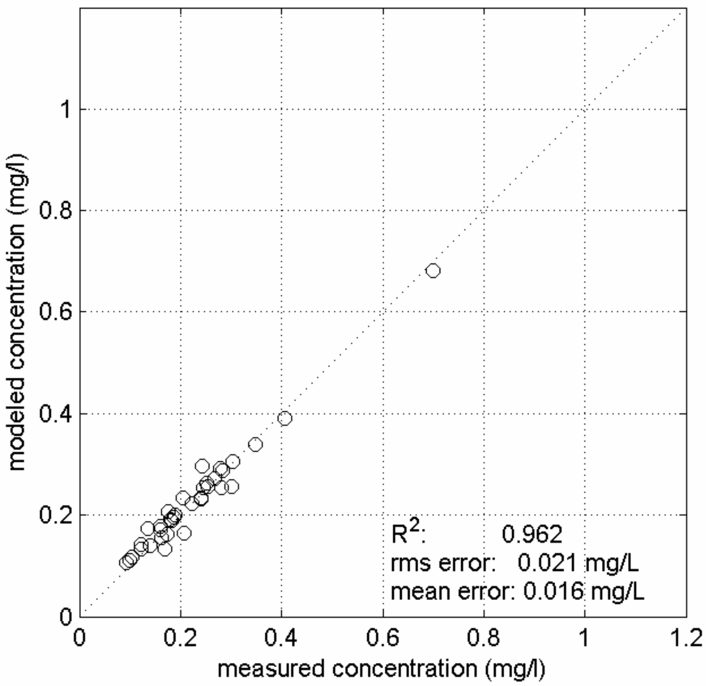


Figure VI-4. Model total nitrogen calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for the model are also presented.

VI.2.5 Model Salinity Verification

In addition to the model calibration based on nitrogen loading and water column measurements, numerical water quality model performance is typically verified by modeling salinity. This step was performed for the Pleasant Bay system using salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of each system, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary, at the freshwater stream discharges, and groundwater inputs. The open boundary salinity was set at 32.3 ppt, based on measurements taken at the inlet during flooding tides. For surface water steams and groundwater inputs salinities were set at 0 ppt. Surface water and groundwater inputs used for the model are listed in Table VI-4. Groundwater flows were distributed evenly in the model through the use of several 1-D element input points positioned along each model's land boundary. Rainfall to the estuary surface was also included as an additional freshwater input into the model. Based on an annual average rainfall of 27.25 inches, the freshwater input into the model from rain was computed to be 26.47 ft^3/sec.

Table VI-4. Freshwater inputs (groundwater and surface water) used as inputs to the salinity model of the Pleasant Bay estuary system.

System Division	flow ft^3/sec
Pochet Neck	2.76
Meetinghouse Pond	1.03
The River - upper	1.22
Lonnies Pond	1.32
Kescayo Ganset Stream	0.44
Lonnies Pond River	0.51
Areys Pond	1.34
Namequoit River	1.55
The River - lower	1.80
Paw Wah Pond	0.56
Quanset Pond	0.63
Tar Kiln Stream	1.02
Round Cove	1.02
The Horseshoe	0.42
Muddy Creek - upper	3.53
Muddy Creek - lower	2.71
Ryders Cove	3.05
Crows Pond	1.10
Bassing Harbor	0.76
Frost Fish Creek	0.72
Pleasant Bay/Little Pleasant Bay	13.85
Chatham Harbor	2.65

A contour plot of model output is shown in Figure VI-7, with comparisons of modeled and measured salinities presented in Figures VI-5 and VI-6. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model skillfully represents salinity gradients throughout the Pleasant Bay estuary system. The rms error of the three models is 1.21 ppt, and correlation coefficient between the model and measured salinity data is 0.85. The salinity verification provides a further independent confirmation that model dispersion coefficients and represented freshwater inputs to the model correctly simulate the real physical system.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

To assess the influence of nitrogen loading on bioactive nitrogen concentrations within Pleasant Bay, two standard water quality modeling scenarios were run: a “build-out” scenario based on potential development (described in more detail in Section IV) and a “no anthropogenic load” or “no load” scenario assuming only atmospheric deposition on the watershed and sub-embayment, as well as a natural forest within each watershed. Comparisons of the alternate watershed loading analyses are shown in Table VI-5. Loads are presented in kilograms per day (kg/day) in this Section, since it is inappropriate to show benthic flux loads in kilograms per year due to seasonal variability.

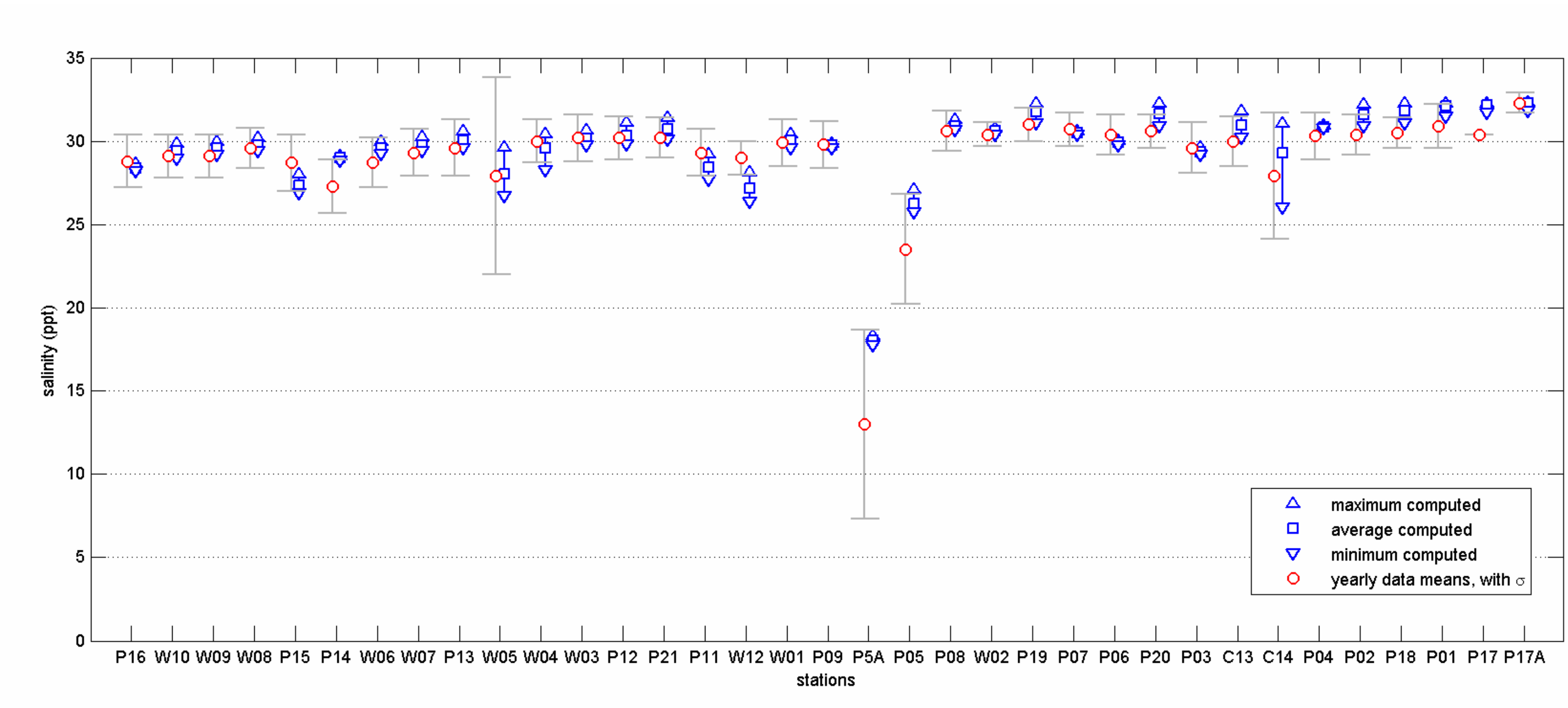


Figure VI-5. Comparison of measured and calibrated model output at stations in Pleasant Bay. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with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

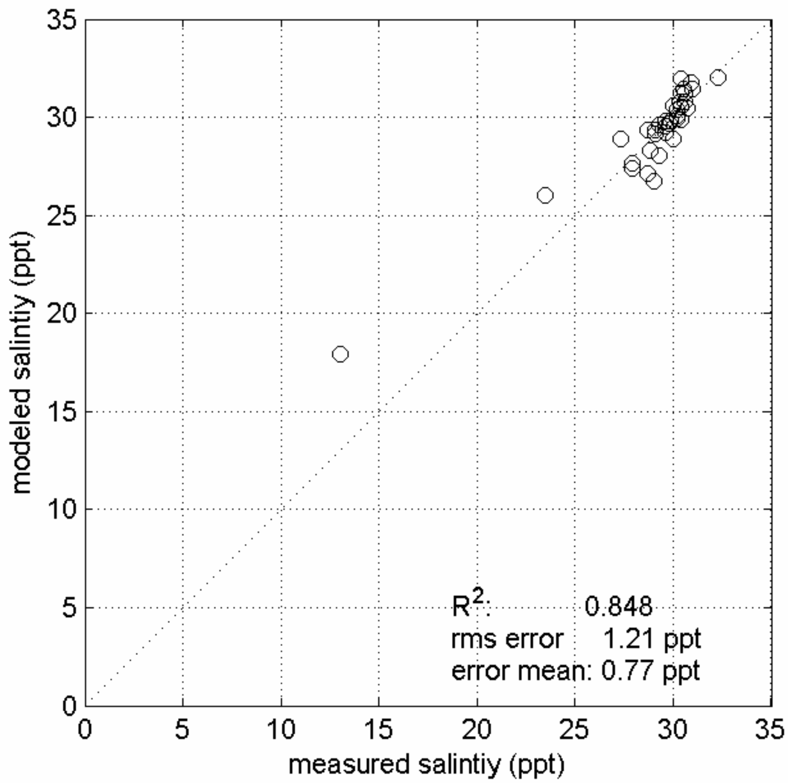


Figure VI-6. Model salinity target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for each model are also presented.

VI.2.6.1 Build-Out

In general, certain sub-embayments would be impacted more than others. For the build-out scenario, for the entire Pleasant Bay system the nitrogen load increases by more than 30%, or 38.273 kg/day. The smallest increase occurs in the Crows Pond watershed, where the nitrogen increases 10% due to potential future development. Other watershed areas would experience much greater load increases, for example the loads to the Little Pleasant Bay watershed would increase 61% from the present day loading levels. A maximum increase in watershed loading resulting from future development would occur in the lower watershed of The River, where the increase would be 2.768 kg/day, or 71% more than present conditions. 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 80%.

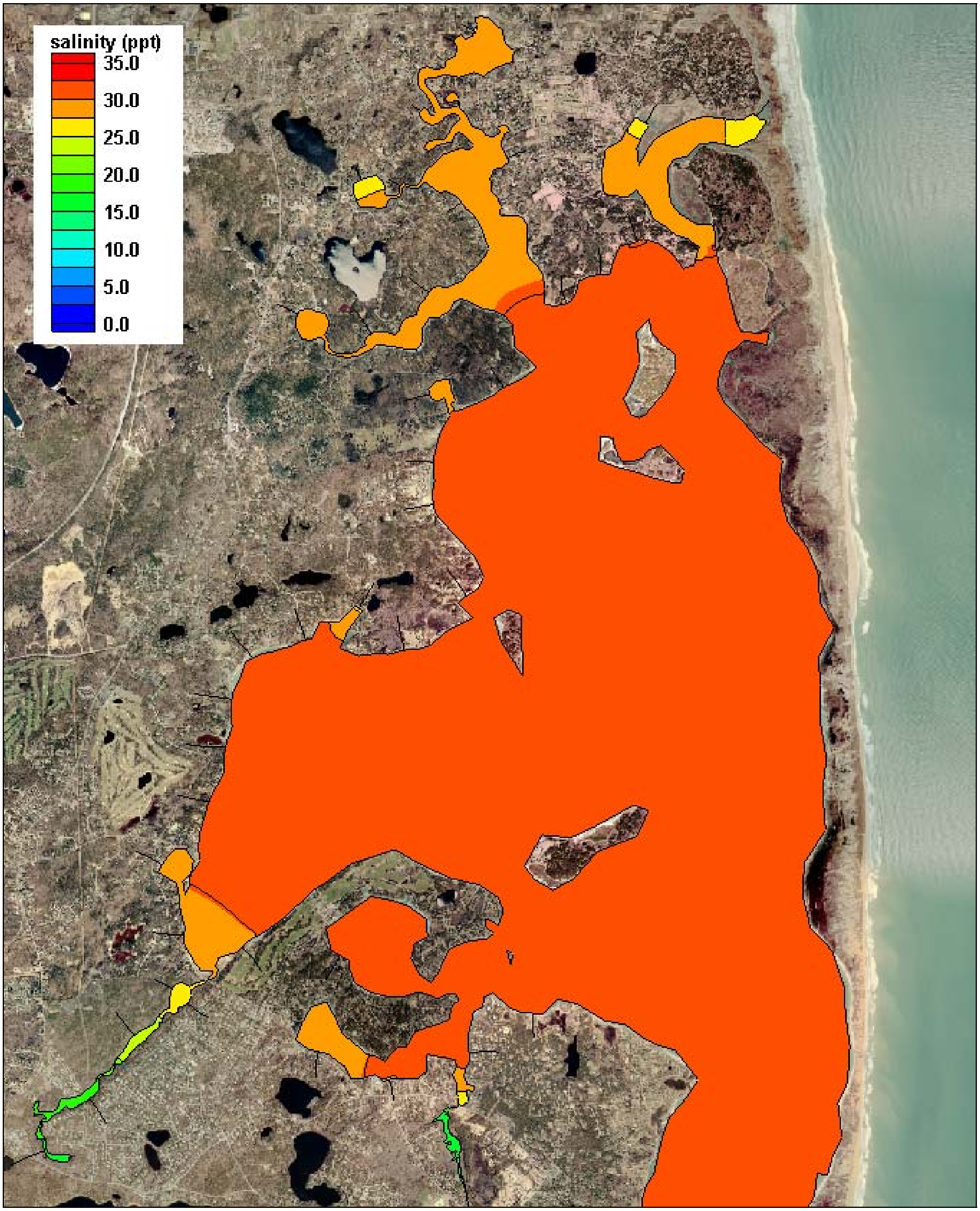


Figure VI-7. Contour Plot of modeled salinity (ppt) in the Pleasant Bay system.

Table VI-5. Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic ("no-load") loading scenarios of the Pleasant Bay system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	build out (kg/day)	build-out % change	no load (kg/day)	no load % change
Meetinghouse Pond	6.197	8.263	+33.3%	0.69	-88.8%
The River – upper	2.773	3.978	+43.5%	0.53	-81.0%
The River – lower	3.879	6.647	+71.3%	0.76	-80.5%
Lonnies Pond	2.441	3.556	+45.7%	0.68	-72.1%
Areys Pond	1.304	2.044	+56.7%	0.47	-64.1%
Namequoit River	2.737	4.052	+48.0%	0.56	-79.5%
Paw Wah Pond	1.860	2.808	+51.0%	0.23	-87.5%
Pochet Neck	8.422	11.893	+41.2%	1.23	-85.4%
Little Pleasant Bay	7.496	12.036	+60.6%	1.53	-79.6%
Quanset Pond	1.781	2.395	+34.5%	0.30	-83.4%
Tar Kiln Stream	6.123	6.992	+14.2%	0.32	-94.9%
Round Cove	4.225	5.178	+22.6%	0.60	-85.7%
The Horseshoe	0.638	0.992	+55.4%	0.13	-79.4%
Muddy Creek - upper	9.981	13.540	+35.7%	1.95	-80.5%
Muddy Creek - lower	8.477	10.189	+20.2%	1.47	-82.6%
Pleasant Bay	23.159	30.792	+33.0%	3.49	-84.9%
Pleasant Bay/Chatham Harbor Channel	-	-	-	-	-
Bassing Harbor - Ryder Cove	9.819	11.137	+13.4%	2.00	-79.6%
Bassing Harbor - Frost Fish Creek	2.904	3.318	+14.2%	0.40	-86.2%
Bassing Harbor - Crows Pond	4.219	4.647	+10.1%	0.53	-87.3%
Bassing Harbor	1.668	1.967	+17.9%	0.23	-86.0%
Chatham Harbor	17.099	19.055	+11.4%	1.84	-89.2%
TOTAL - Pleasant Bay System	**127.203**	**165.477**	**+30.1%**	**19.951**	**-84.3%**

For the build-out scenario, a breakdown of the nitrogen load entering each sub-embayment is shown in Table VI-6. The benthic flux for the build-out scenarios is assumed to vary proportional to the watershed load, where an increase in watershed load will result in an increase in benthic flux (i.e., a positive change in the absolute value of the flux), and *vise versa*.

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-6. **Build-out** sub-embayment and surface water loads used for total nitrogen modeling of the Pleasant Bay system, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Meetinghouse Pond	8.263	0.584	16.976
The River – upper	3.978	0.288	7.408
The River – lower	6.647	2.241	12.086
Lonnies Pond	3.556	0.225	1.982
Areys Pond	2.044	0.181	8.036
Namequoit River	4.052	0.523	16.652
Paw Wah Pond	2.808	0.082	4.865
Pochet Neck	11.893	1.767	-0.904
Little Pleasant Bay	12.036	24.023	39.552
Quanset Pond	2.395	0.170	7.040
Tar Kiln Stream	6.992	0.066	-
Round Cove	5.178	0.170	9.680
The Horseshoe	0.992	0.063	-
Muddy Creek - upper	13.540	0.162	5.793
Muddy Creek - lower	10.189	0.205	-1.383
Pleasant Bay	30.792	19.153	163.977
Pleasant Bay/Chatham Harbor Channel	-	17.786	-42.317
Bassing Harbor - Ryder Cove	11.137	1.296	10.334
Bassing Harbor - Frost Fish Creek	3.318	0.096	-0.166
Bassing Harbor - Crows Pond	4.647	1.389	0.636
Bassing Harbor	1.967	1.071	-5.178
Chatham Harbor	19.055	14.153	-42.173
TOTAL - Pleasant Bay System	**165.477**	**85.693**	**212.895**

Following development of the nitrogen loading estimates for the build-out scenario, the water quality models of each system were run to determine nitrogen concentrations within each sub-embayment (Table VI-7). Total nitrogen concentrations in the receiving waters (i.e., the Atlantic Ocean) remained identical to the existing conditions modeling scenarios. Total N concentrations increased the most in the upper portions of the system, with the largest change at a station in upper muddy Creek (+25.4% at PBA-05a), with the least change occurring in Chatham Harbor (+0.8% at PBA-17) near the system's inlet to the open ocean. Color contours of model output for the build-out scenario are present in Figure VI-8. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figure VI-2, which allows direct comparison of nitrogen concentrations between loading scenarios.

Table VI-7. Comparison of model average bioactive N (DIN+PON) concentrations from present loading and the build-out scenario, with percent change, for the Pleasant Bay system. The sentinel threshold stations are in bold print and depicted in Figure VI-1.

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Meetinghouse Pond	PBA-16	0.380	0.441	+16.1%
Meetinghouse Pond	WMO-10	0.261	0.296	+13.4%
The River - upper	WMO-09	0.239	0.269	+12.5%
The River – mid	WMO-08	0.211	0.235	+11.1%
Lonnies Pond (Kescayo Ganset Pond)	PBA-15	0.250	0.286	+14.5%
Areys Pond	PBA-14	0.297	0.345	+16.1%
Namequoit River - upper	WMO-6	0.239	0.269	+12.7%
Namequoit River - lower	WMO-7	0.216	0.240	+11.3%
The River - lower	PBA-13	0.195	0.214	+10.0%
Pochet – upper	WMO-05	0.269	0.316	+17.8%
Pochet - lower	WMO-04	0.209	0.234	+11.9%
Pochet – mouth	WMO-03	0.183	0.199	+8.8%
Little Pleasant Bay - head	**PBA-12**	**0.178**	**0.193**	**+8.6%**
Little Pleasant Bay - main basin	PBA-21	0.162	0.173	+7.4%
Paw Wah Pond	PBA-11	0.257	0.304	+18.4%
Little Quanset Pond	WMO-12	0.229	0.260	+13.3%
Quanset Pond	WMO-01	0.191	0.209	+9.3%
Round Cove	PBA-09	0.241	0.267	+10.8%
Muddy Creek - upper	PBA-05a	0.674	0.845	+25.4%
Muddy Creek - lower	PBA-05	0.286	0.331	+15.4%
Pleasant Bay - head	PBA-08	0.149	0.158	+6.0%
Pleasant Bay - off Quanset Pond	WMO-02	0.160	0.171	+6.8%
Pleasant Bay- upper Strong Island	PBA-19	0.117	0.121	+3.2%
Pleasant Bay - mid west basin	PBA-07	0.168	0.181	+7.4%
Pleasant Bay - off Muddy Creek	PBA-06	0.192	0.210	+9.0%
Pleasant Bay - Strong Island channel	PBA-20	0.124	0.129	+3.9%
Ryders Cove - upper	**PBA-03**	**0.250**	**0.270**	**+8.0%**
Ryders Cove - lower	**CM-13**	**0.158**	**0.168**	**+5.7%**
Frost Fish - lower	CM-14	0.243	0.265	+8.8%
Crows Pond	PBA-04	0.162	0.171	+5.5%
Bassing Harbor	PBA-02	0.127	0.132	+4.0%
Pleasant Bay - lower	PBA-18	0.116	0.120	+3.0%
Chatham Harbor - upper	PBA-01	0.104	0.105	+1.4%
Chatham Harbor - lower	PBA-17	0.099	0.100	+0.8%

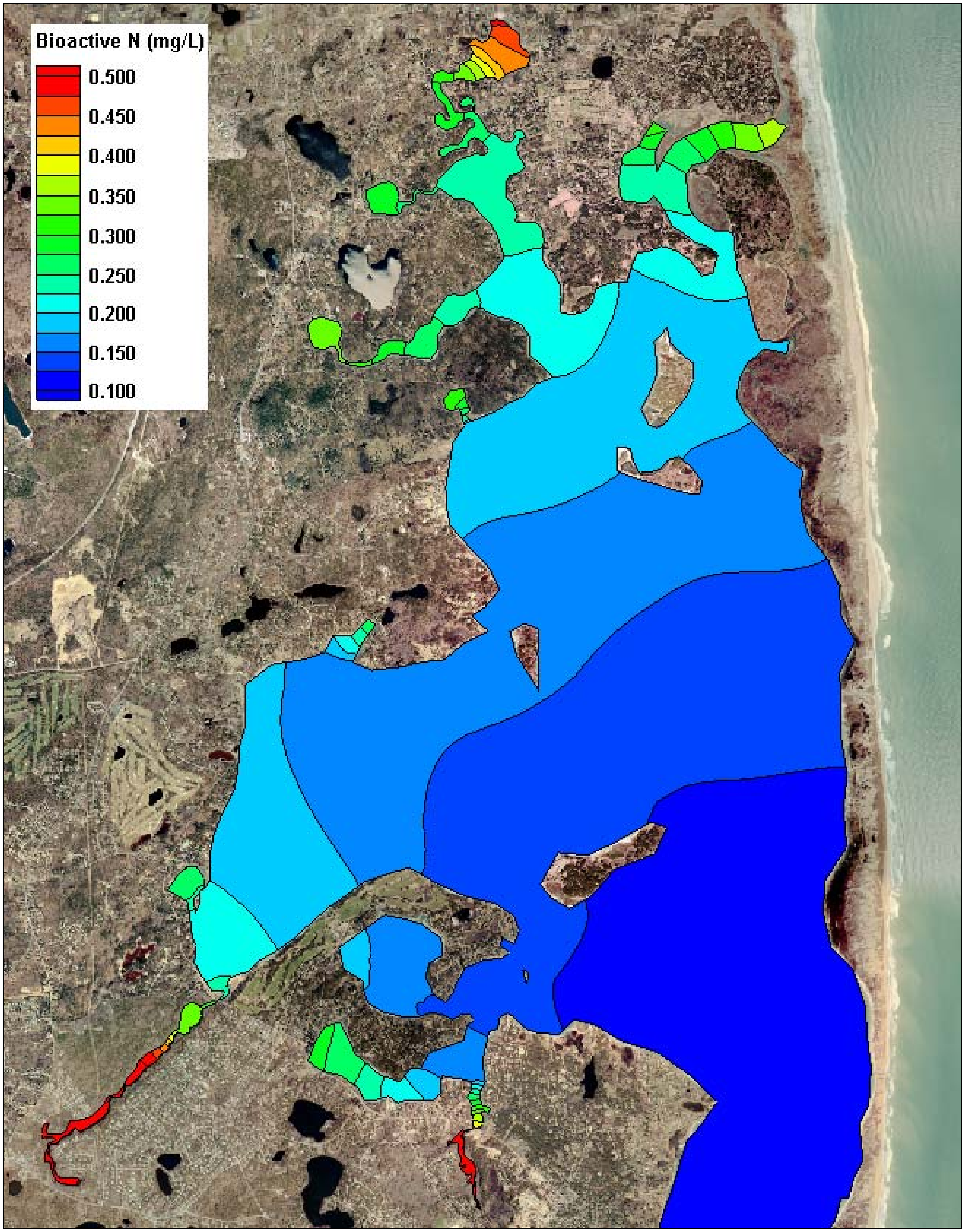


Figure VI-8. Contour plot of modeled total nitrogen concentrations (mg/L) in the Pleasant Bay system, for projected build-out loading conditions.

VI.2.6.2 No Anthropogenic Load

A breakdown of the total nitrogen load entering each sub-embayment for the no anthropogenic load (“no load”) scenarios is shown in Table VI-8. The benthic flux input to each embayment was reduced (toward zero) based on the reduction in the watershed load (as discussed in §VI.2.6.1). Compared to the modeled present conditions and build-out scenario, atmospheric deposition directly to each sub-embayment becomes a greater percentage of the total nitrogen load as the watershed load and related benthic flux decrease.

Table VI-8. “**No anthropogenic loading**” (“no load”) sub-embayment and surface water loads used for total nitrogen modeling of the Pleasant Bay system, with watershed N loads, atmospheric N loads, and benthic flux

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Meetinghouse Pond	0.69	0.58	7.40
The River – upper	0.53	0.29	3.55
The River – lower	0.76	2.24	7.35
Lonnies Pond	0.68	0.22	0.97
Areys Pond	0.47	0.18	3.70
Namequoit River	0.56	0.52	9.52
Paw Wah Pond	0.23	0.08	2.24
Pochet Neck	1.23	1.77	-0.51
Little Pleasant Bay	1.53	24.02	32.57
Quanset Pond	0.30	0.17	3.45
Tar Kiln Stream	0.32	0.07	-
Round Cove	0.60	0.17	3.61
The Horseshoe	0.13	0.06	-
Muddy Creek - upper	1.95	0.16	1.78
Muddy Creek - lower	1.47	0.21	-0.56
Pleasant Bay	3.49	19.15	114.57
Pleasant Bay/Chatham Harbor Channel	-	17.79	-35.14
Bassing Harbor - Ryder Cove	2.00	1.30	5.49
Bassing Harbor - Frost Fish Creek	0.40	0.10	-0.08
Bassing Harbor - Crows Pond	0.53	1.39	0.43
Bassing Harbor	0.23	1.07	-3.92
Chatham Harbor	1.84	14.15	-36.00
TOTAL - Pleasant Bay System	**19.951**	**85.693**	**120.417**

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Again, total nitrogen concentrations in the receiving waters (i.e., the Atlantic Ocean) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from “no load” was significant as shown in Table VI-9, with reductions greater than 35% (at PBA-16 and PBA-05a) occurring the upper portions of the system. Results for each system are shown pictorially in Figure VI-9.

Table VI-9. Comparison of model average bioactive N (DIN+PON) concentrations from present loading and the no anthropogenic ("no load") scenario, with percent change, for the Pleasant Bay system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). The sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	no load (mg/L)	% change
Meetinghouse Pond	PBA-16	0.380	0.233	-38.7%
Meetinghouse Pond	WMO-10	0.261	0.184	-29.7%
The River - upper	WMO-09	0.239	0.174	-27.2%
The River – mid	WMO-08	0.211	0.162	-23.3%
Lonnies Pond (Kescayo Ganset Pond)	PBA-15	0.250	0.179	-28.4%
Areys Pond	PBA-14	0.297	0.214	-27.9%
Namequoit River - upper	WMO-6	0.239	0.180	-24.7%
Namequoit River - lower	WMO-7	0.216	0.166	-22.9%
The River - lower	PBA-13	0.195	0.154	-20.7%
Pochet – upper	WMO-05	0.269	0.170	-36.6%
Pochet - lower	WMO-04	0.209	0.157	-24.8%
Pochet – mouth	WMO-03	0.183	0.149	-18.7%
Little Pleasant Bay - head	**PBA-12**	**0.178**	**0.145**	**-18.3%**
Little Pleasant Bay - main basin	PBA-21	0.162	0.136	-16.1%
Paw Wah Pond	PBA-11	0.257	0.181	-29.6%
Little Quanset Pond	WMO-12	0.229	0.155	-32.5%
Quanset Pond	WMO-01	0.191	0.147	-23.0%
Round Cove	PBA-09	0.241	0.163	-32.3%
Muddy Creek - upper	PBA-05a	0.674	0.273	-59.5%
Muddy Creek - lower	PBA-05	0.286	0.169	-41.0%
Pleasant Bay - head	PBA-08	0.149	0.128	-14.5%
Pleasant Bay - off Quanset Pond	WMO-02	0.160	0.133	-16.9%
Pleasant Bay- upper Strong Island	PBA-19	0.117	0.108	-7.8%
Pleasant Bay - mid west basin	PBA-07	0.168	0.137	-18.8%
Pleasant Bay - off Muddy Creek	PBA-06	0.192	0.147	-23.5%
Pleasant Bay - Strong Island channel	PBA-20	0.124	0.112	-9.8%
Ryders Cove - upper	**PBA-03**	**0.250**	**0.159**	**-36.7%**
Ryders Cove - lower	**CM-13**	**0.158**	**0.125**	**-21.1%**
Frost Fish - lower	CM-14	0.243	0.148	-39.2%
Crows Pond	PBA-04	0.162	0.128	-21.1%
Bassing Harbor	PBA-02	0.127	0.112	-11.9%
Pleasant Bay - lower	PBA-18	0.116	0.107	-7.9%
Chatham Harbor - upper	PBA-01	0.104	0.100	-3.9%
Chatham Harbor - lower	PBA-17	0.099	0.097	-2.2%

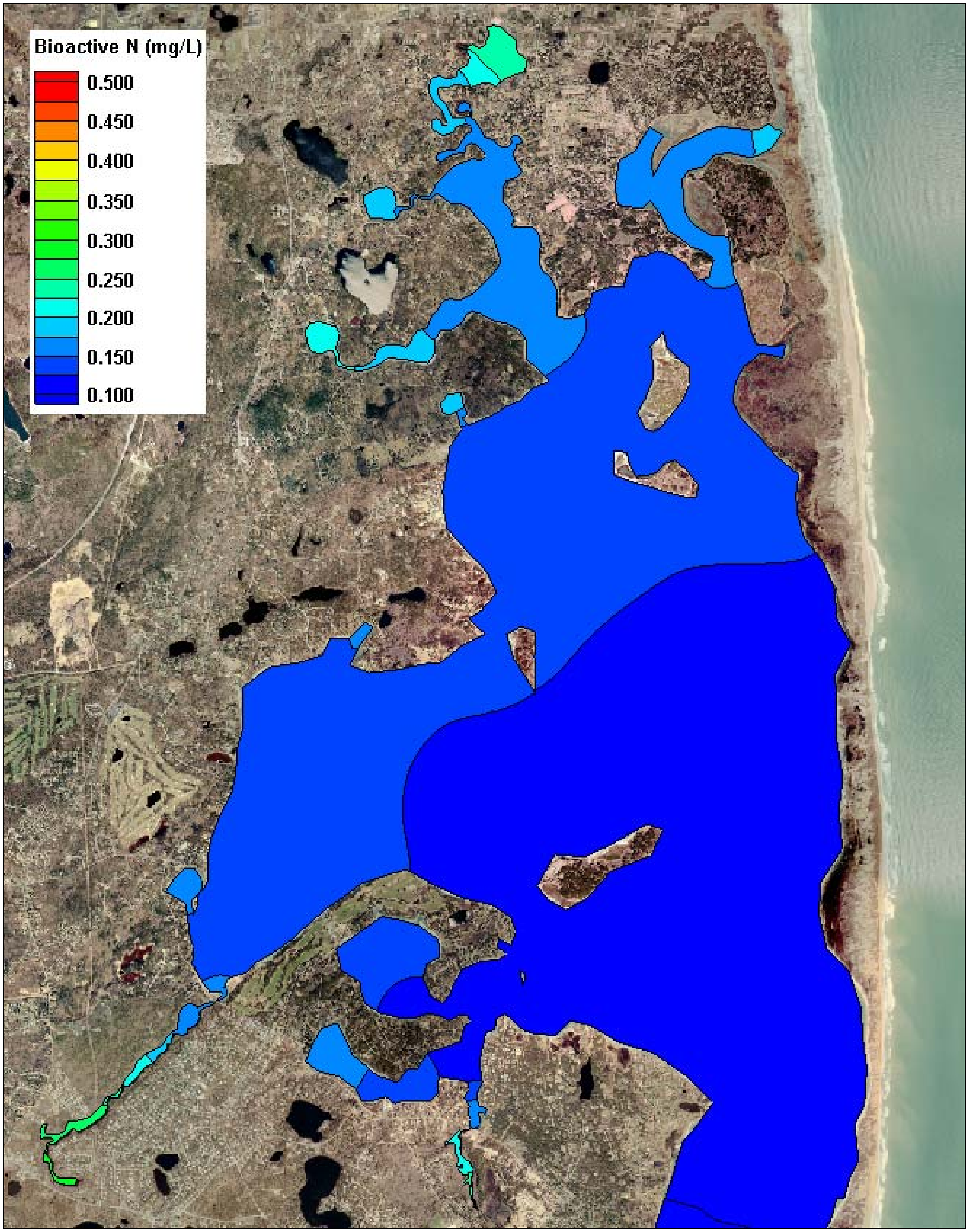


Figure VI-9. Contour plot of modeled total nitrogen concentrations (mg/L) in Pleasant Bay, for no anthropogenic loading conditions.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Pleasant Bay embayment system our assessment is based upon data from the water quality monitoring database (2000-2005) and our surveys of eelgrass distribution (1951, 1995, 2001), benthic animal communities and sediment characteristics, and dissolved oxygen records conducted during the summer and fall of 2003. These data form the basis of an assessment of this system's present health, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for these systems (Chapter VIII).

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

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

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

VII.2 BOTTOM WATER DISSOLVED OXYGEN

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

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (Figure VII-1). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the embayment bottom within key regions of the Pleasant Bay System (Figure VII-2). The sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployment. In addition periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a minimum deployment of 30 days within the interval from July through mid-September. All of the mooring data from the Pleasant Bay embayment system was collected during the summer of 2003 with the exception of two moorings (Meetinghouse Pond and Pochet) that failed in the summer of 2003 and were therefore redeployed in the summer of 2004 as well as several moorings in the Bassing Harbor sub-embayment system that where deployed by the MEP in 2002 for the development of the Nutrient Threshold Reports that covered the Town of Chatham embayments.

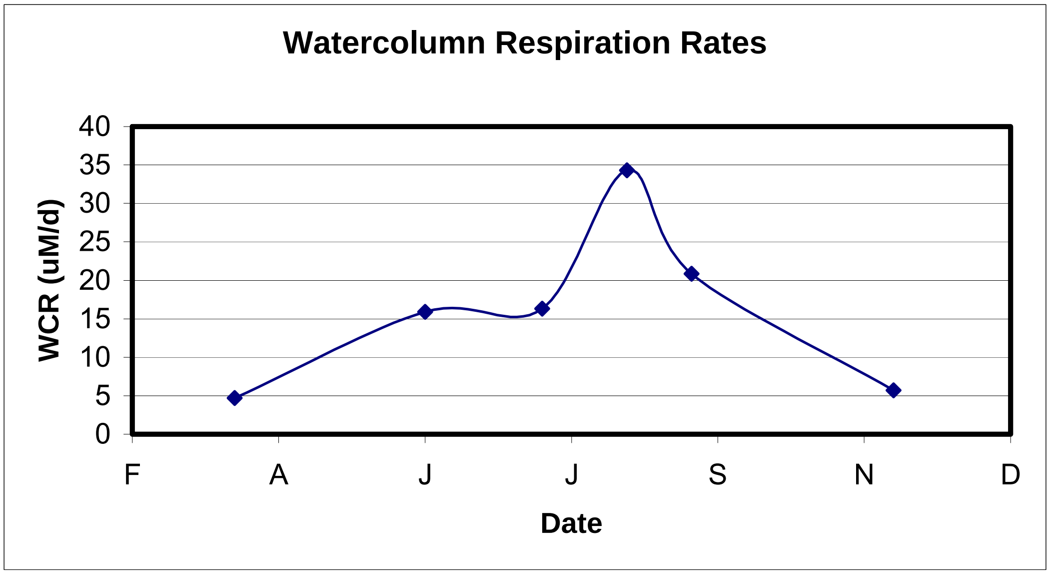


Figure VII-1. Average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

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

Dissolved oxygen and chlorophyll a records were examined both for temporal trends and to determine the percent of the 29-51 day deployment period (depending on the mooring) that these parameters were below/above various benchmark concentrations (Tables VII-1, VII-2, VII-3). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions. The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate highly nutrient enriched waters and impaired habitat quality at all mooring sites within each estuary (Figures VII-3 through VII-41). The oxygen data is consistent with high organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment of these estuarine systems.

As for the discussion of sediment nitrogen regeneration (Section IV.3, above) the extent of oxygen related stress among the Pleasant Bay sub-embayments showed significant spatial variation, typical of other embayments within the MEP region. Although there are a large number of sub-embayments to the Pleasant Bay System, the habitat impairment associated with oxygen depletion tended to follow the 4 groups based upon the basin type:

(E) small enclosed basin (Meetinghouse Pond, Lonnies Pond, Areys Pond, Round Cove, Quanset Pond, Paw Wah Pond, Upper Muddy Creek),
(F) moderate sized tributary sub-embayment (The River, Muddy Creek),
(G) salt marsh dominated tidal sub-estuary (Pochet),
(H) large lagoonal estuarine basin (Little Pleasant Bay, Pleasant Bay, Chatham Harbor).

The general pattern is for a high level of oxygen stress (frequent hypoxia or anoxia) in the bottomwaters of the small enclosed basins (group A) which tend to have higher nitrogen levels and high rates of sediment metabolism, associated with their circulation and focus of watershed nitrogen loads. The Meetinghouse Pond basin and outlet channel, Lonnies Pond and its outlet channel, the Areys Pond outlet channel (Namequoit River), Quanset Pond all showed significant levels of oxygen depletion were routinely hypoxic and except for Quanset Pond levels were frequently <2 mg/L. In the same group of enclosed basins, Areys Pond, Paw Wah Pond and upper Muddy Creek showed frequent anoxia (absence of oxygen). Among the enclosed basins only Round Cove showed only mild hypoxia with levels above 4 mg/L and generally above 5 mg/L during the full deployment.

In contrast, the salt marsh dominated tidal creek of Pochet showed frequent oxygen depletions to 3-4 mg/L, but was generally above 4 mg/L. The oxygen conditions in Pochet creek are consistent with the biogeochemistry of salt marshes. Salt marsh creeks (that do not empty at low tide) frequently become hypoxic in summer as a result of the high organic matter loading associated with marshes. Even pristine salt marshes can exhibit this behavior.

The large main basins of the lagoonal estuarine component showed oxygen conditions consistent with their rates of sediment metabolism associated with their deep waters and depositional nature (Little Pleasant Bay, Pleasant Bay) or their high tidal velocities (Chatham Harbor and eastern channel form Chatham Harbor to Little Pleasant Bay, channel between Strong Island and Bassing Harbor). The Upper Pleasant Bay at Namequoit Point showed oxygen levels frequently declining to 4-5 mg/L and the western most basin of Pleasant Bay (between Round Cove and Muddy Creek) had a single event to 2-4 mg/L, although was generally >5 mg/L. Approaching Chatham Harbor oxygen conditions improved (see Strong Island results), with oxygen conditions generally >6 mg/L with short declines to 5 mg/L associated with the outflow of lower oxygen waters from Pleasant Bay.

The oxygen records further indicate that the systems with the lower minimum oxygen depletions were also those with the largest daily oxygen excursions, which further supports the assessment of a high degree of nutrient enrichment. The use of only the duration of oxygen below, for example 4 mg L^{-1}, can underestimate the level of habitat impairment in these locations. The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production, oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites).

This latter effect of nitrogen enrichment on the daily oxygen excursion in embayment waters can be seen in the Muddy Creek, Areys Pond, Paw Wah Pond records, 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-14). All of the enclosed basins show this pattern in some form. 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 relationship was seen in the Upper and Lower Muddy Creek (Figure VII-14 and

VII-15). 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-19, 20, 21, 22) to Muddy Creek sub-embayment (Figures VII-14 and 15). This characteristic is also seen within the Bassing Harbor System, which shows an inverse gradient in oxygen minima to chlorophyll levels moving from Ryder Cove to Crows Pond to Bassing Harbor.

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}.

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.

Overall, the oxygen and chlorophyll records show a consistent pattern of higher organic matter production (chlorophyll) in embayments with greater oxygen depletions. The pattern is one of a sub-embayments that are enclosed (group A) having habitat impairment by frequent low oxygen events, with the larger lagoonal basins (group B) showing less frequent and extreme levels of oxygen depletion and moderate impairment, grading to good oxygen conditions near (and presumably in) Chatham Harbor. This pattern follows the nitrogen gradients in the System (Chapter VI), the eelgrass distribution (below) and the infaunal habitat quality.

Figure VII-2. Aerial Photograph of the Pleasant Bay system in the Towns of Chatham, Orleans and Harwich showing locations of Dissolved Oxygen mooring deployments conducted in the Summer of 2003 and 2004.

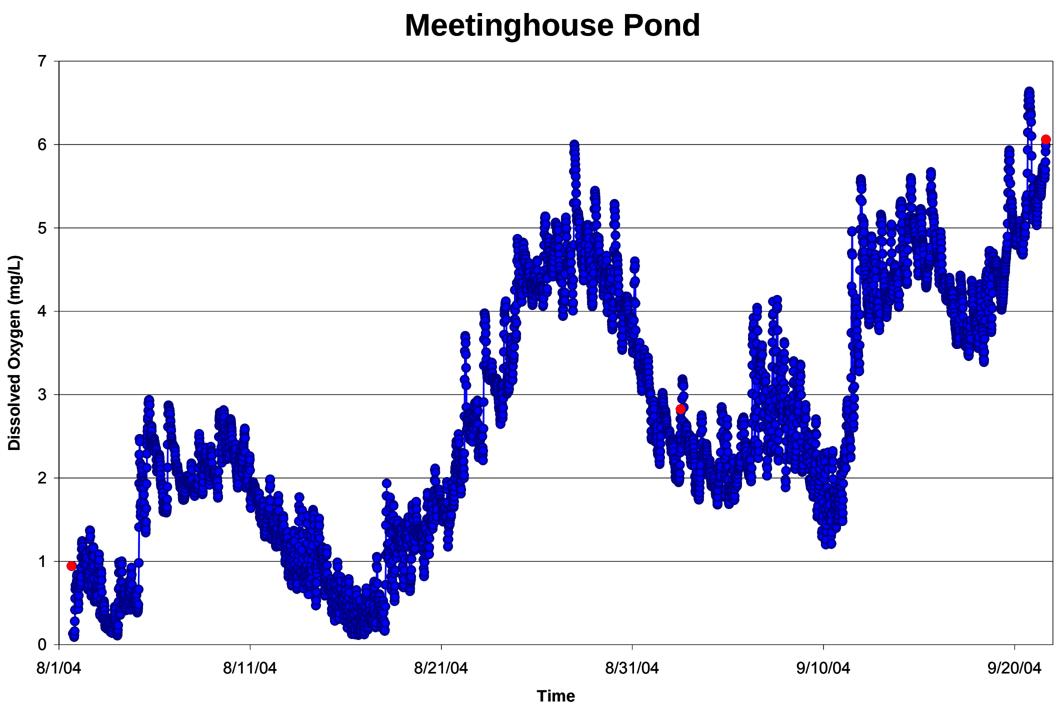


Figure VII-3. Bottom water record of dissolved oxygen at Meetinghouse Pond station, Summer 2004. Calibration samples represented as red dots.

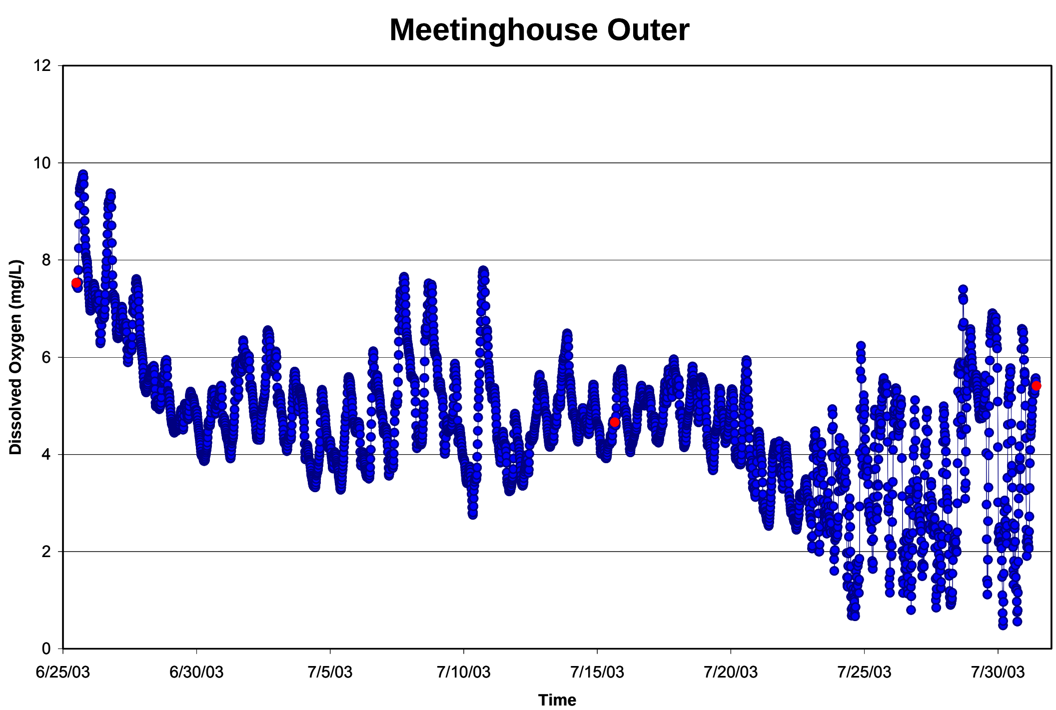


Figure VII-4. Bottom water record of dissolved oxygen in Meetinghouse Outer station, Summer 2003. Calibration samples represented as red dots.

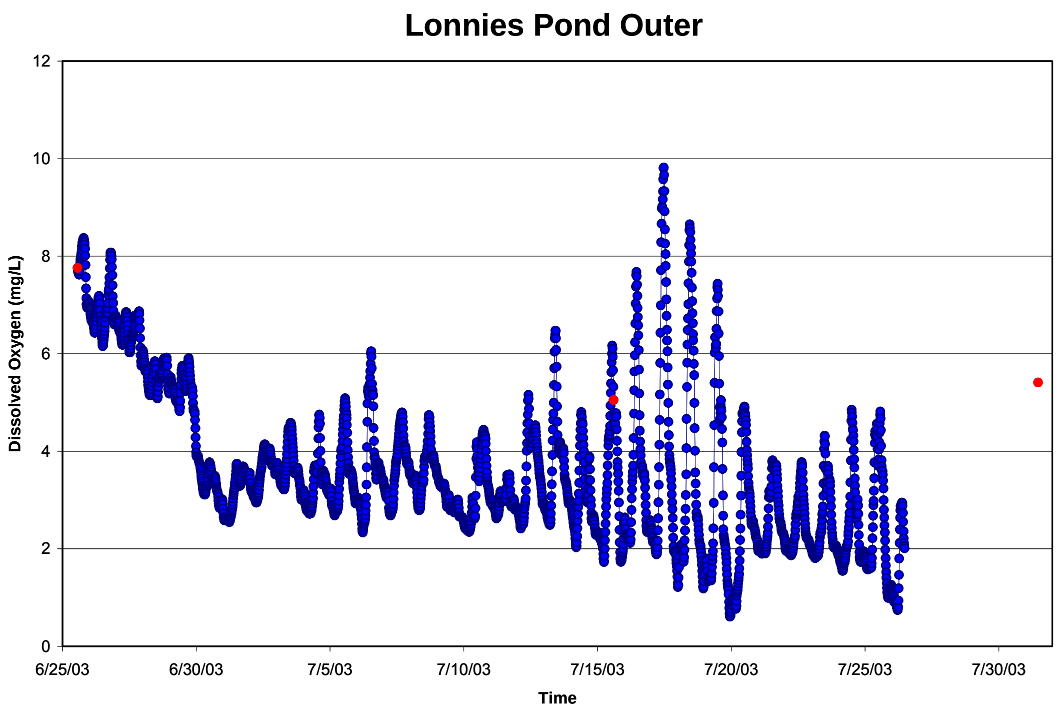


Figure VII-5. Bottom water record of dissolved oxygen at Lonnies Pond station, Summer 2003. Calibration samples represented as red dots.

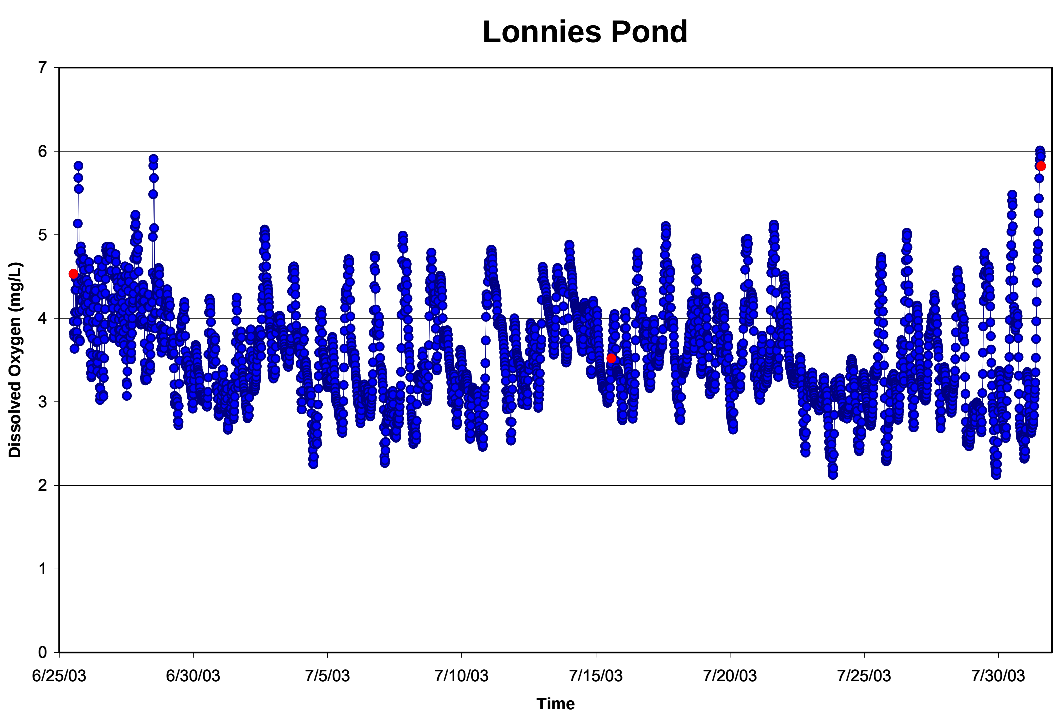


Figure VII-6. Bottom water record of dissolved oxygen in Lonnies Pond Outer station, Summer 2003. Calibration samples represented as red dots.

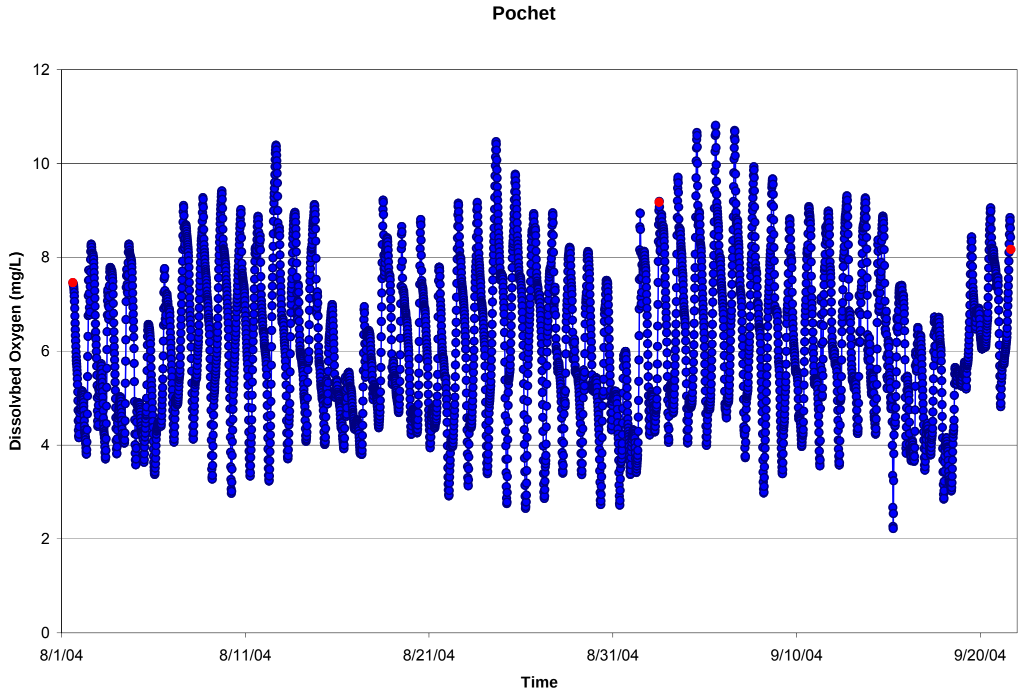


Figure VII-7. Bottom water record of dissolved oxygen in Pochet station, Summer 2004. Calibration samples represented as red dots.

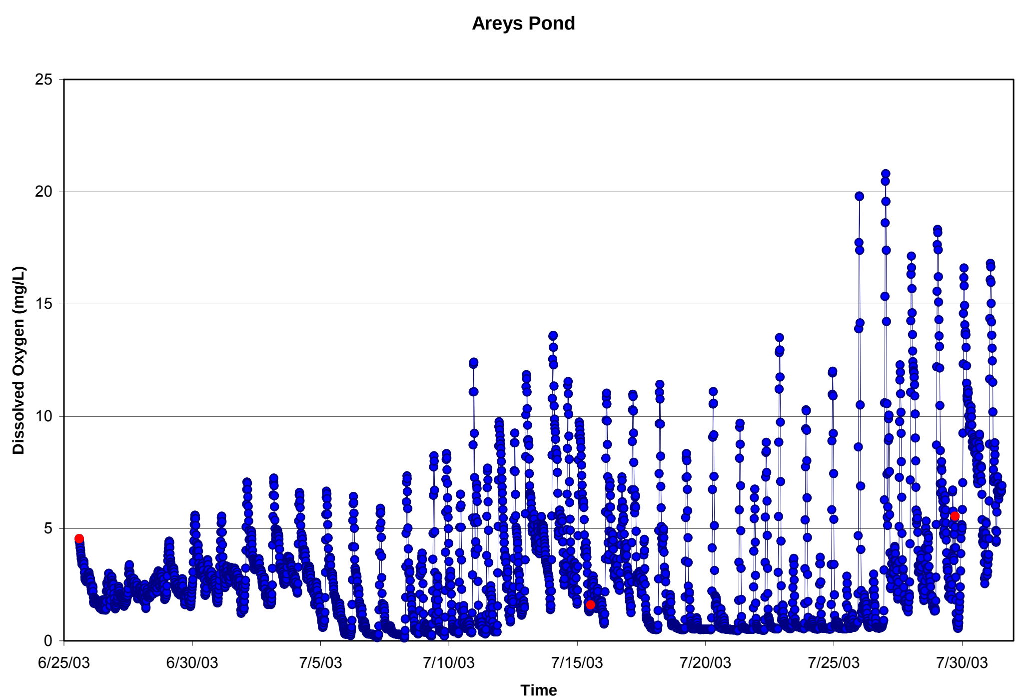


Figure VII-8. Bottom water record of dissolved oxygen in Areys Pond station, Summer 2003. Calibration samples represented as red dots.

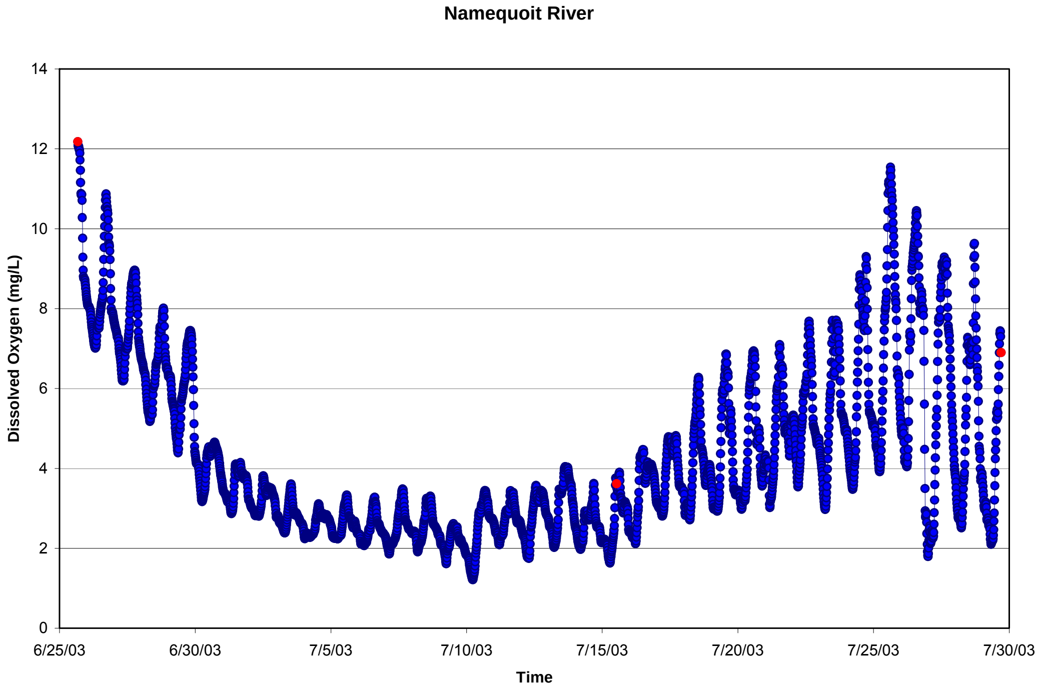


Figure VII-9. Bottom water record of dissolved oxygen in Namequoit River station, Summer 2003. Calibration samples represented as red dots.

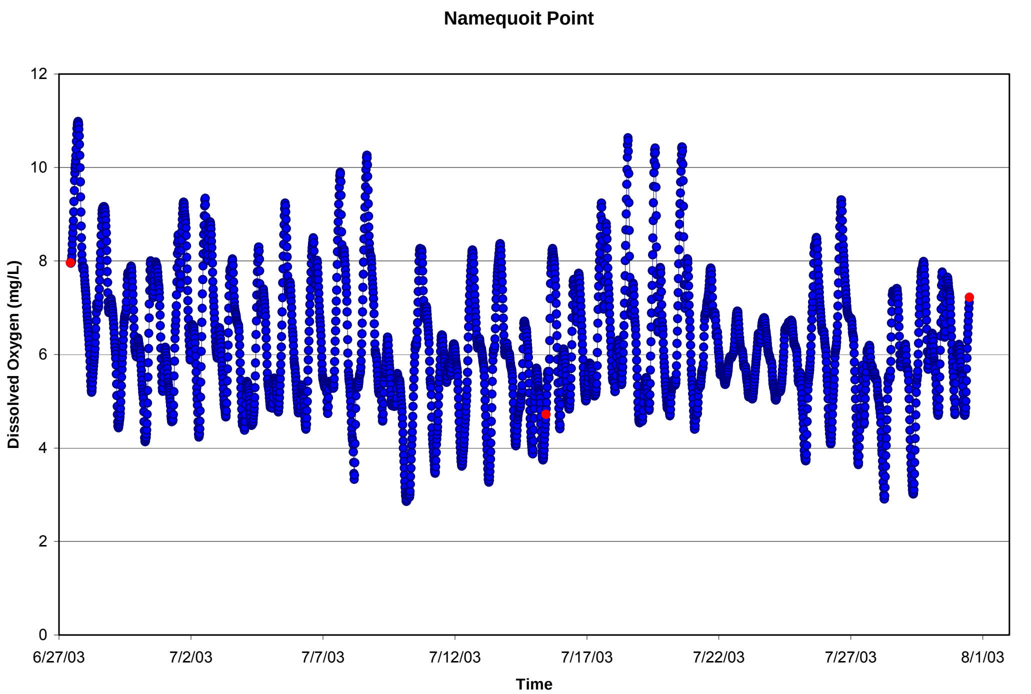


Figure VII-10. Bottom water record of dissolved oxygen in Namequoit Point station, Summer 2003. Calibration samples represented as red dots

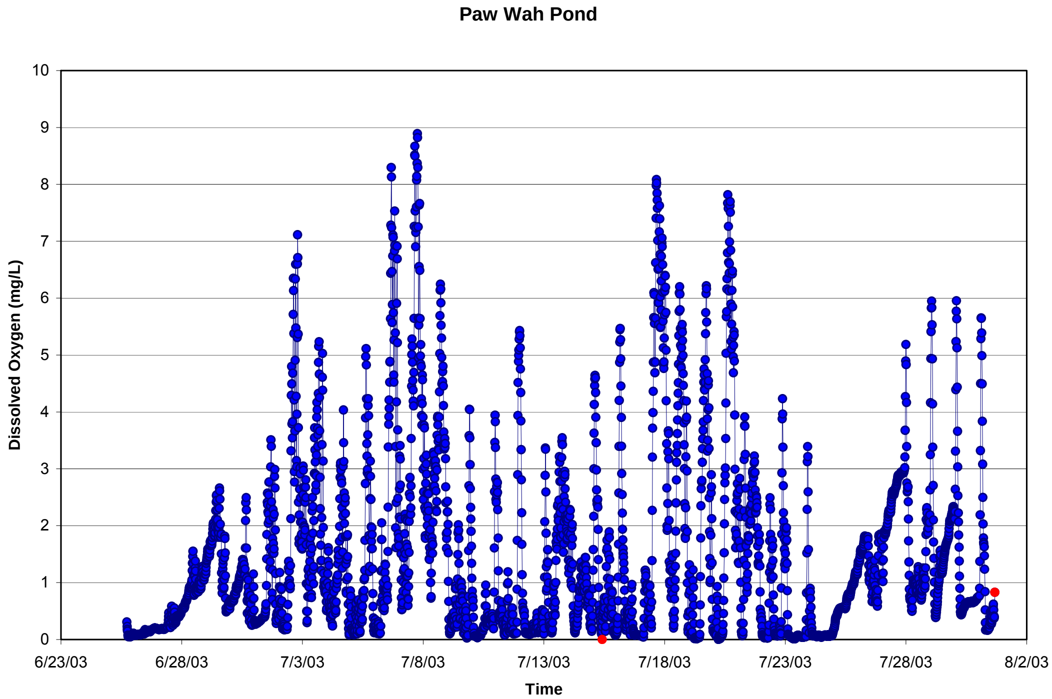


Figure VII-11. Bottom water record of dissolved oxygen in Paw Wah Pond station, Summer 2003. Calibration samples represented as red dots.

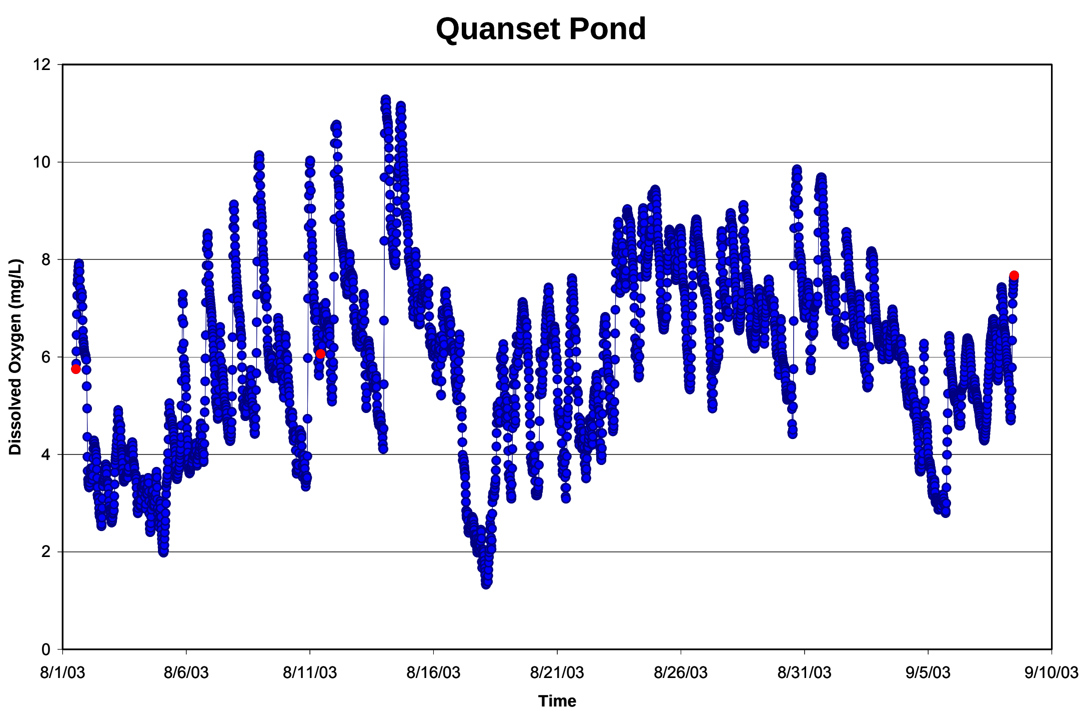


Figure VII-12. Bottom water record of dissolved oxygen in Quanset Pond station, Summer 2003. Calibration samples represented as red dots

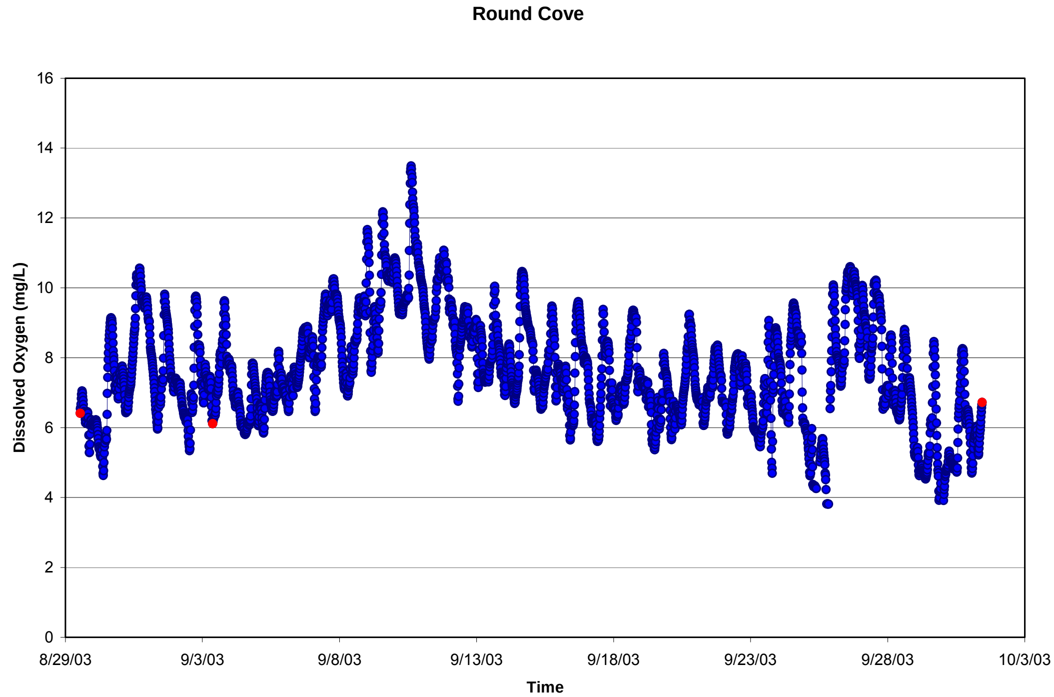


Figure VII-13. Bottom water record of dissolved oxygen in Round Cove station, Summer 2003. Calibration samples represented as red dots.

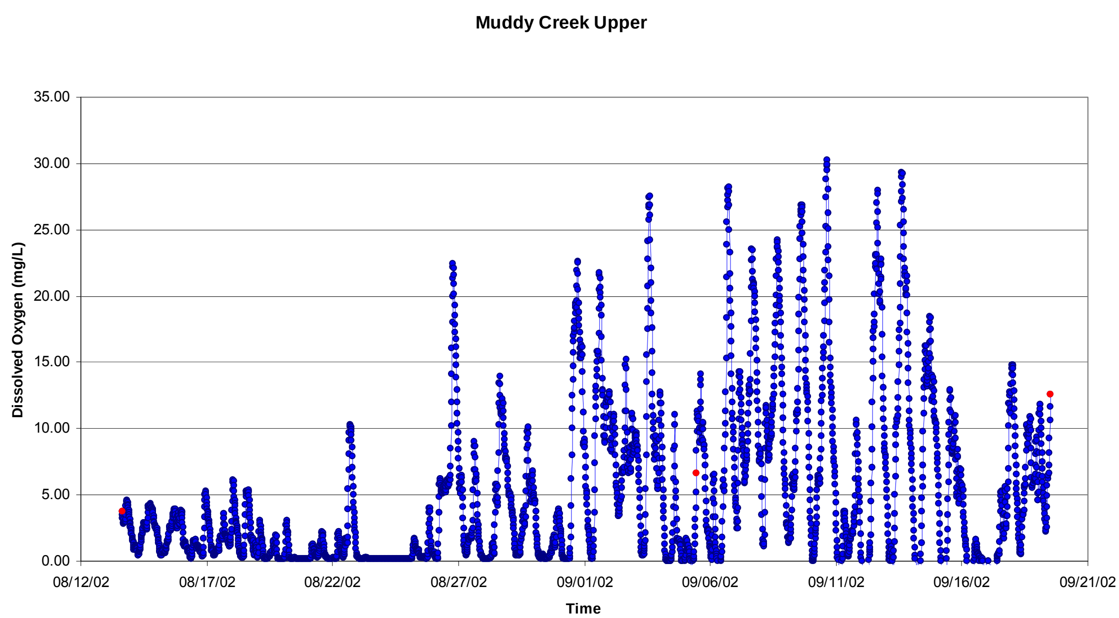


Figure VII-14. Bottom water record of dissolved oxygen in Muddy Creek Upper station, Summer 2002. Calibration samples represented as red dots.

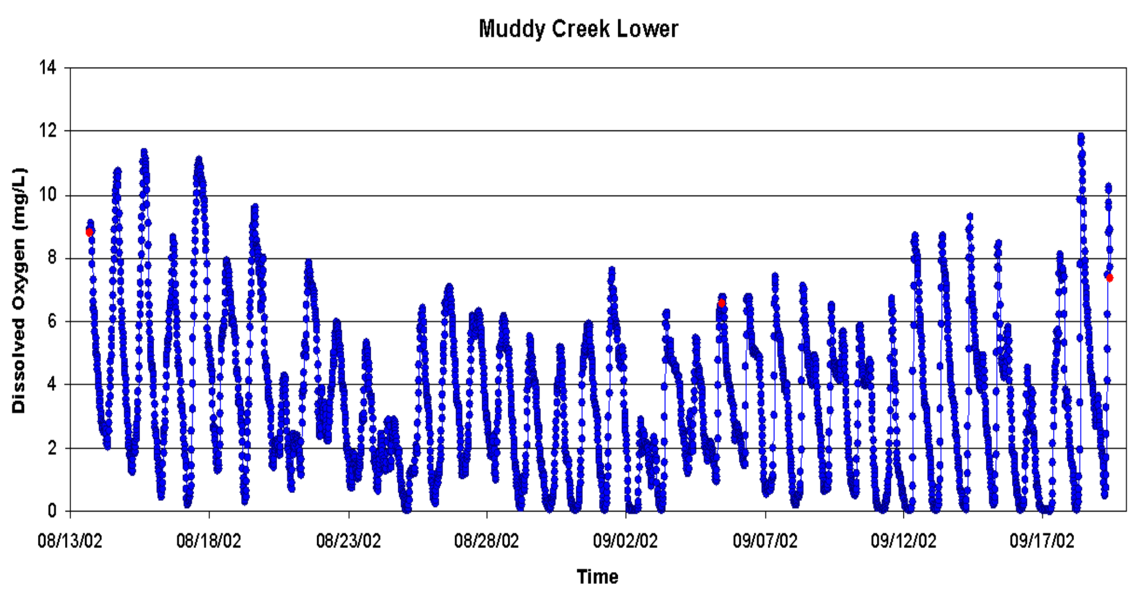


Figure VII-15. Bottom water record of dissolved oxygen in Muddy Creek Lower station, Summer 2002. Calibration samples represented as red dots.

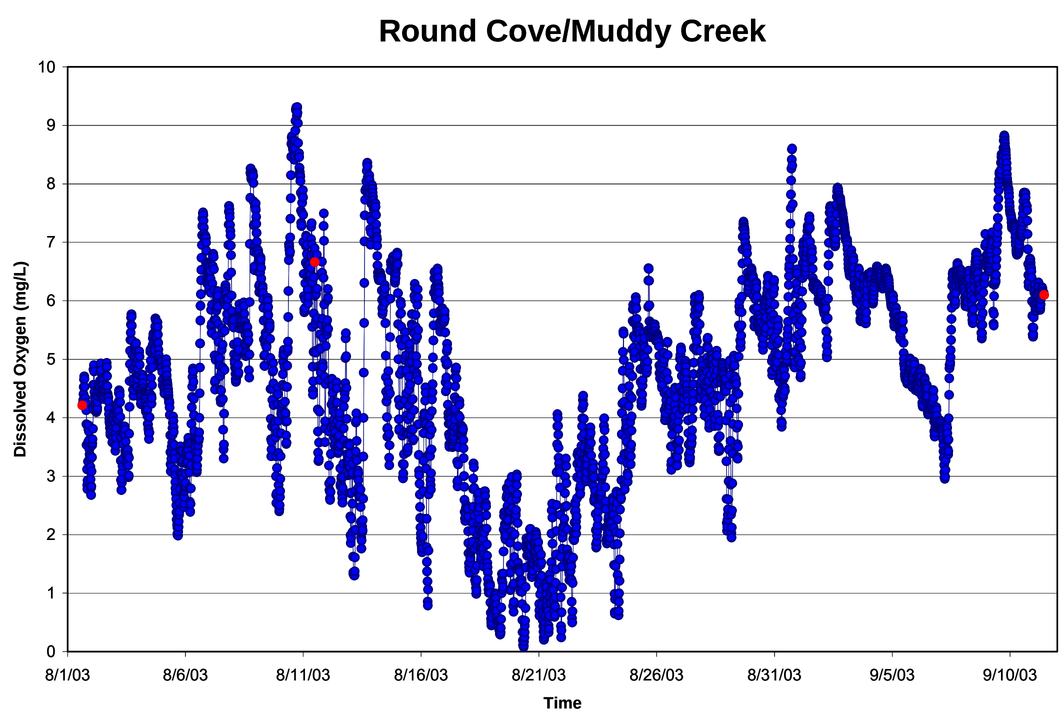


Figure VII-16. Bottom water record of dissolved oxygen in Round Cove / Muddy Creek station located between inlet to Round Cove and the mouth of Muddy Creek, Summer 2003. Calibration samples represented as red dots.

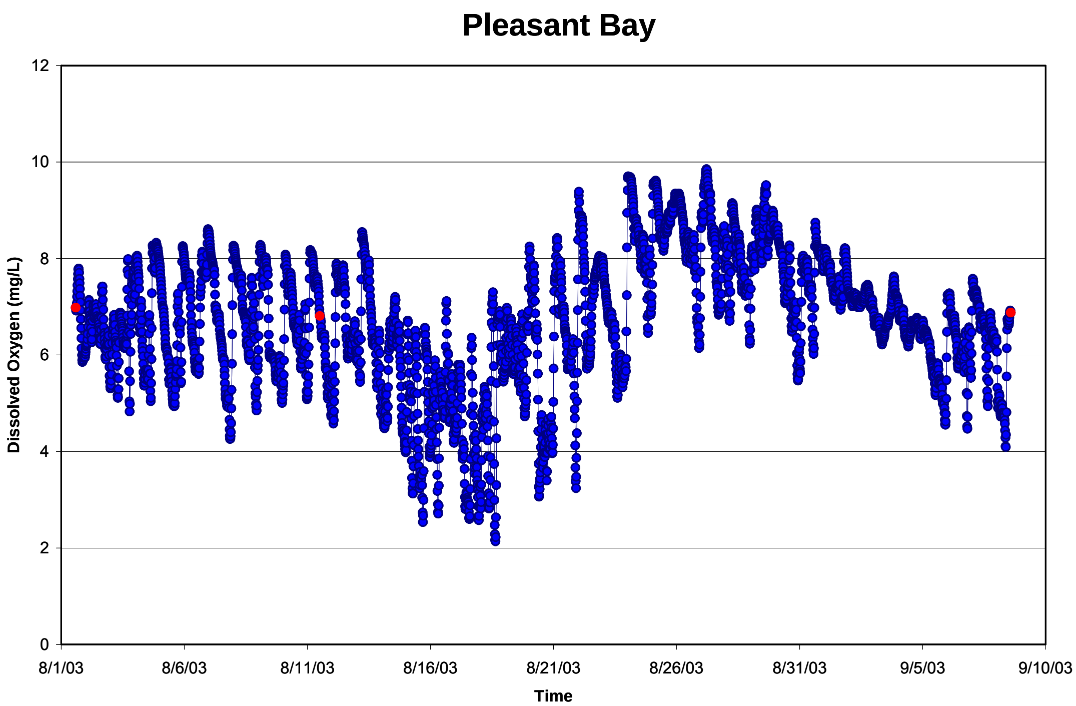


Figure VII-17. Bottom water record of dissolved oxygen in Pleasant Bay station, Summer 2003. Calibration samples represented as red dots.

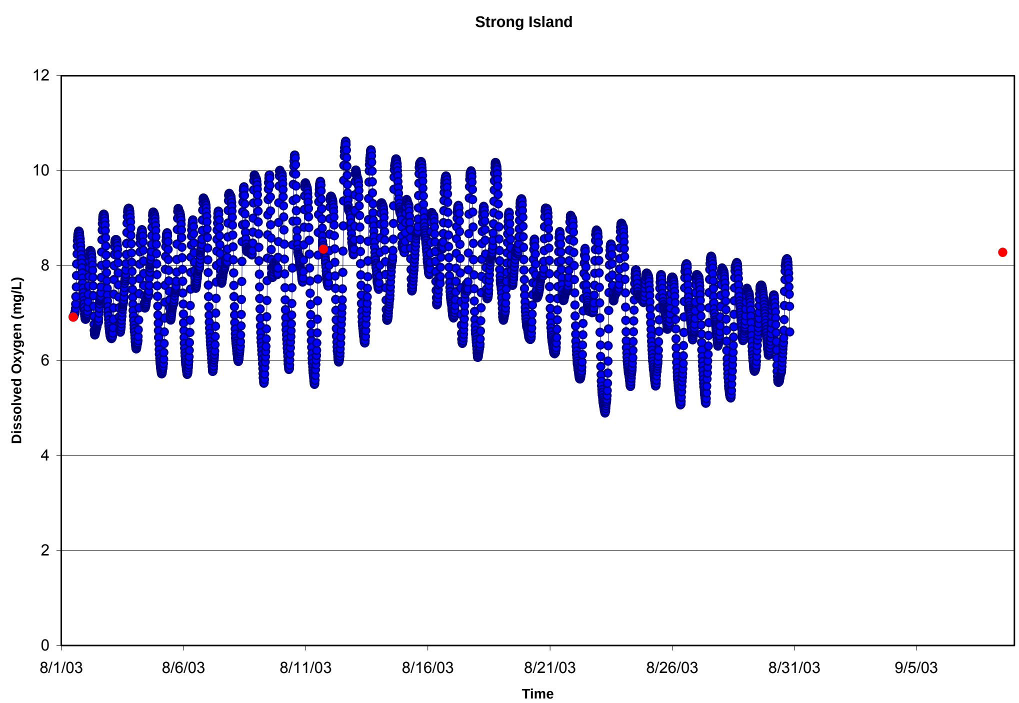


Figure VII-18. Bottom water record of dissolved oxygen in Strong Island station, Summer 2003. Calibration samples represented as red dots.

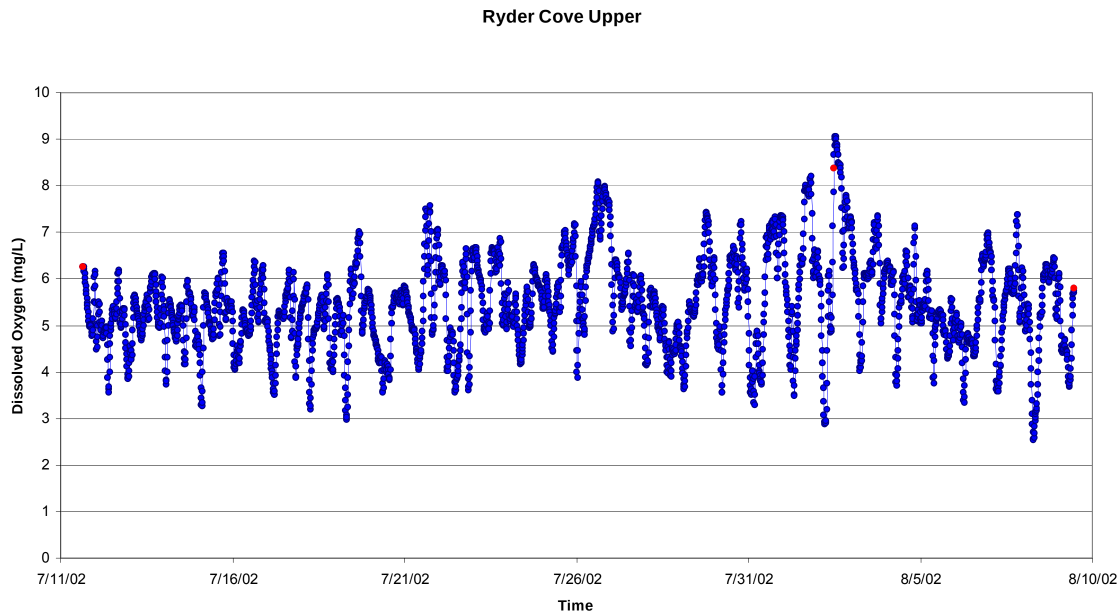


Figure VII-19. Bottom water record of dissolved oxygen in Ryder Cove Upper station, Summer 2002. Calibration samples represented as red dots.

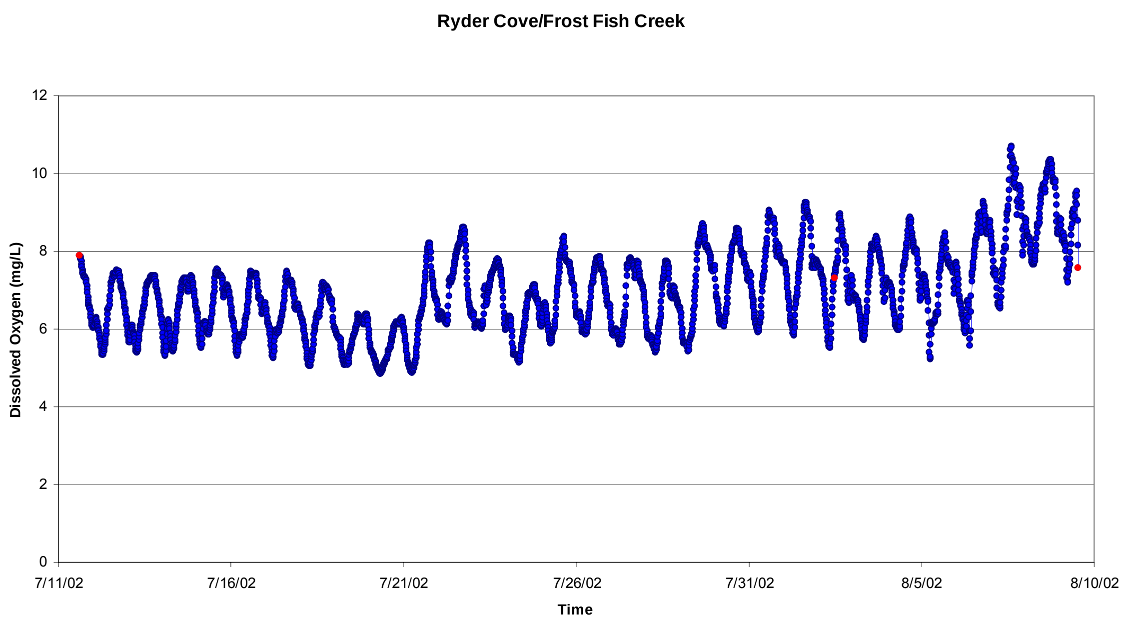


Figure VII-20. Bottom water record of dissolved oxygen in Ryder Cove/Frost Fish Creek station, Summer 2002. Calibration samples represented as red dots.

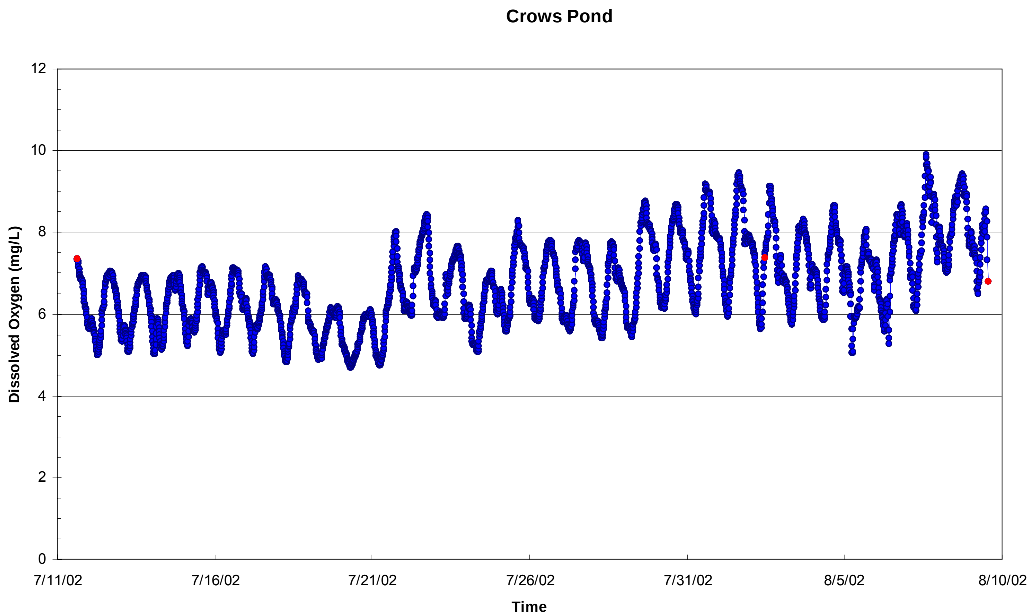


Figure VII-21. Bottom water record of dissolved oxygen in Crows Pond station, Summer 2002. Calibration samples represented as red dots.

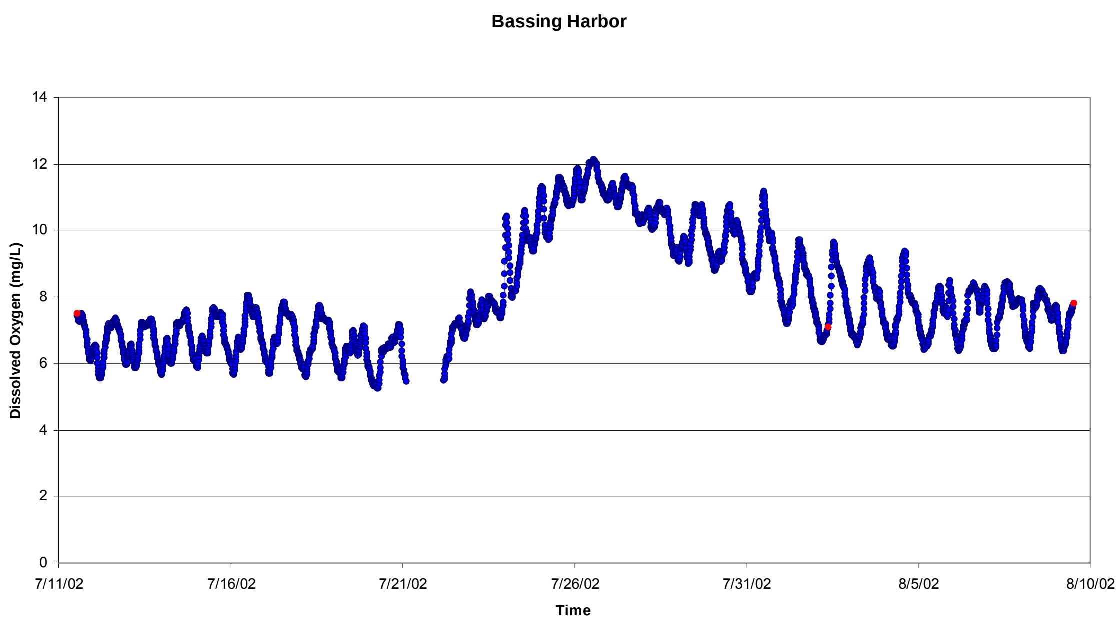


Figure VII-22. Bottom water record of dissolved oxygen in Bassing Harbor station, Summer 2002. Calibration samples represented as red dots

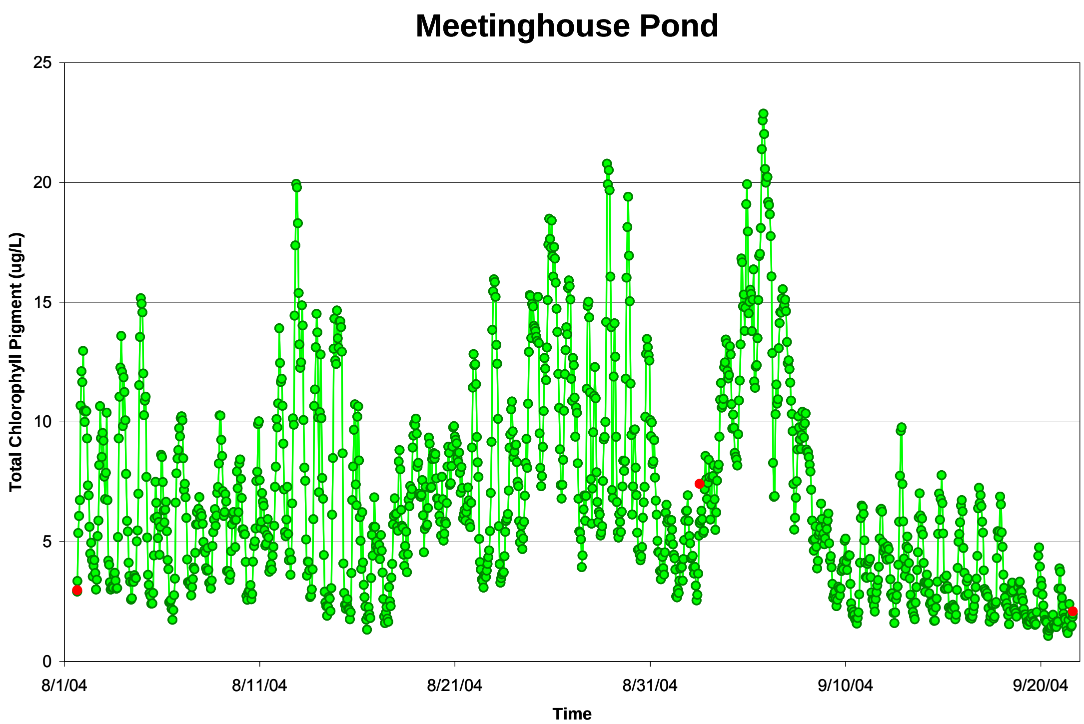


Figure VII-23. Bottom water record of Chlorophyll-*a* in Meetinghouse Pond station, Summer 2004. Calibration samples represented as red dots.

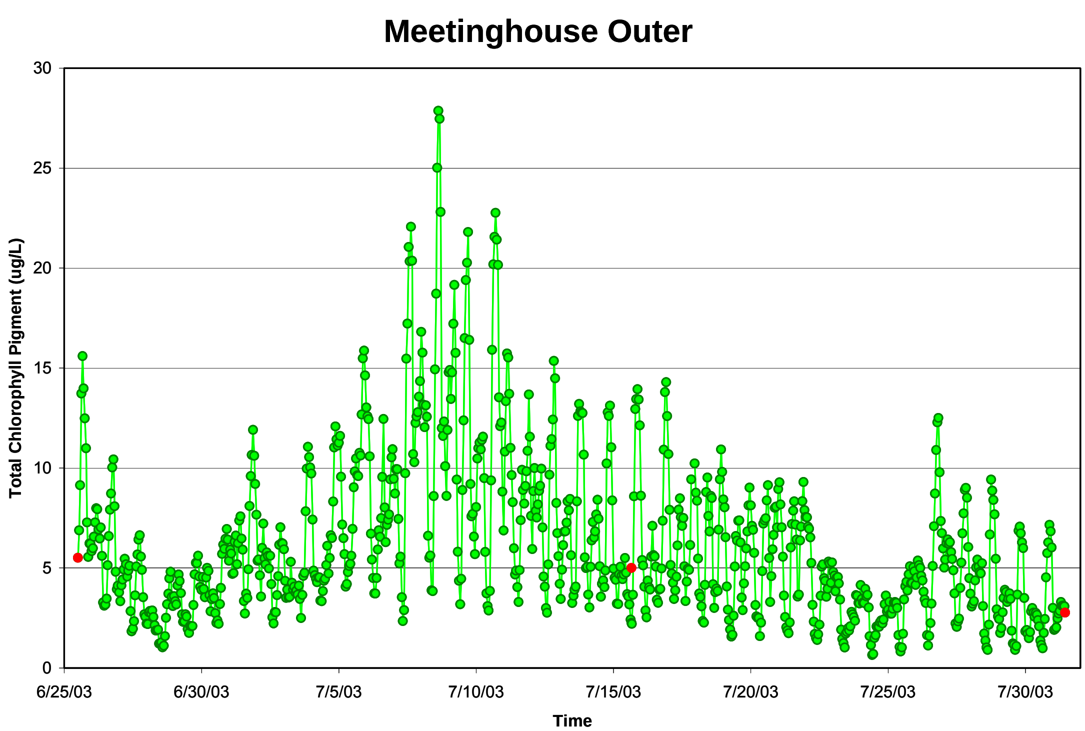


Figure VII-24. Bottom water record of Chlorophyll-*a* in Meetinghouse Outer station, Summer 2003. Calibration samples represented as red dots.

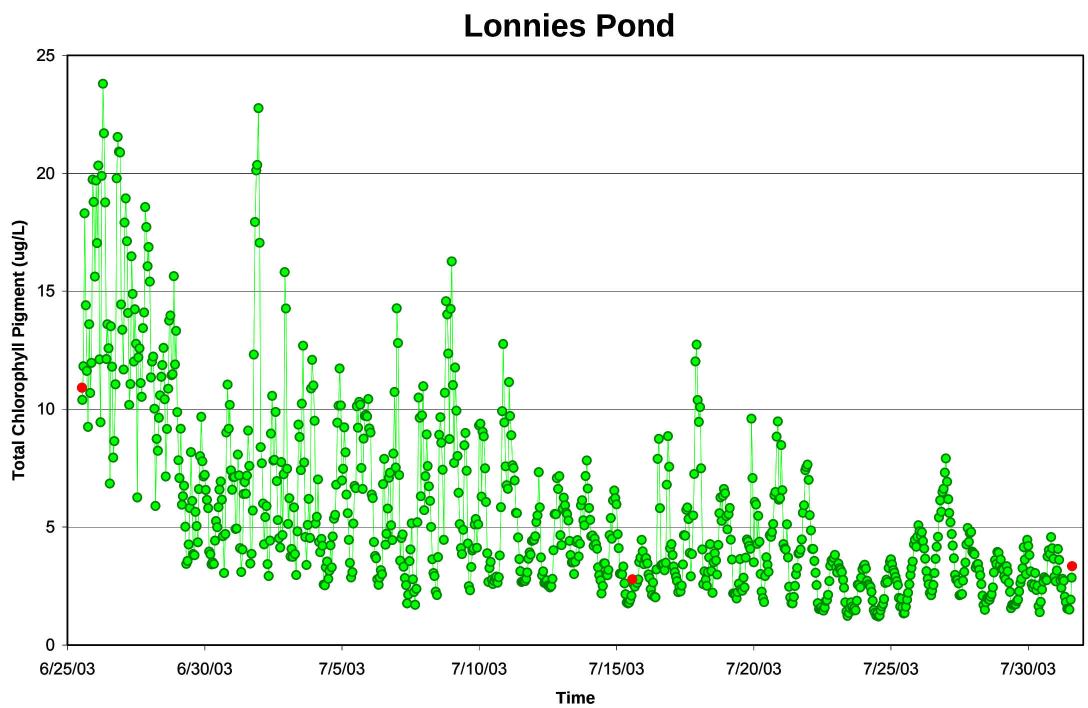


Figure VII-25. Bottom water record of Chlorophyll-*a* in Lonnies Pond station, Summer 2003. Calibration samples represented as red dots.

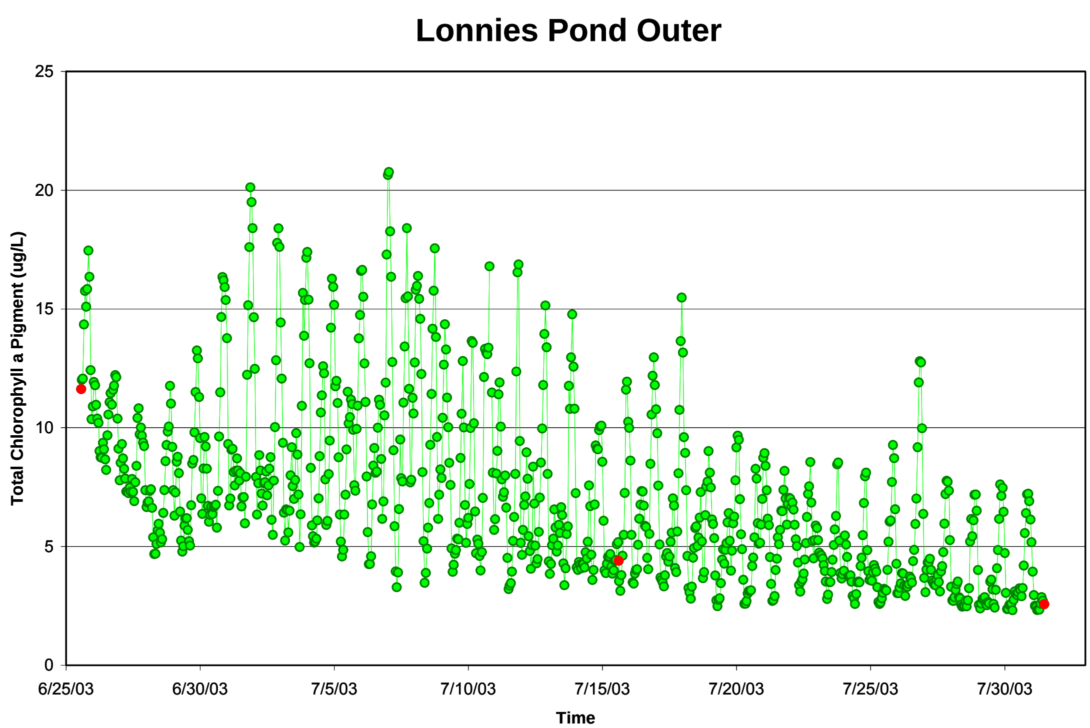


Figure VII-26. Bottom water record of Chlorophyll-*a* in Lonnies Pond Outer station, Summer 2003. Calibration samples represented as red dots.

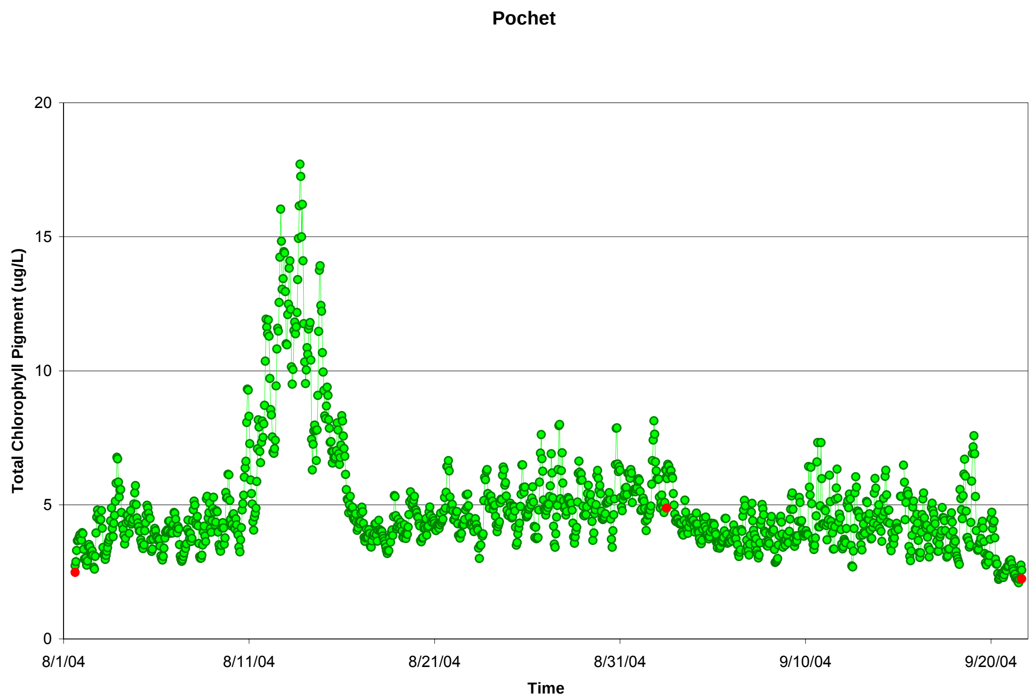


Figure VII-27. Bottom water record of Chlorophyll-*a* in Pochet station, Summer 2004. Calibration samples represented as red dots.

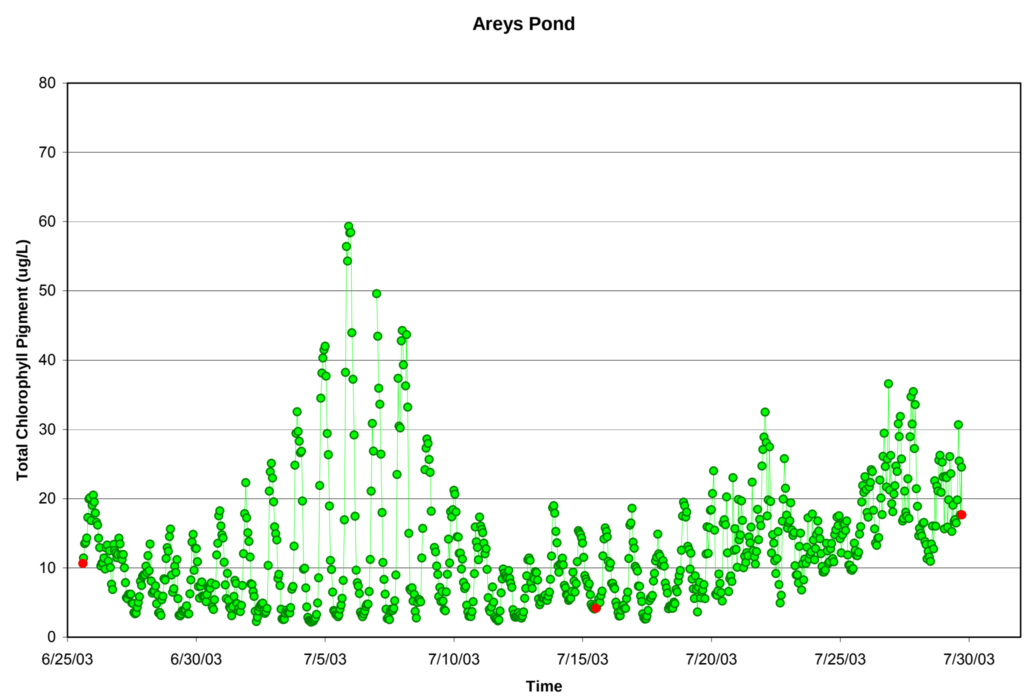


Figure VII-28. Bottom water record of Chlorophyll-*a* in Areys Pond station, Summer 2003. Calibration samples represented as red dots.

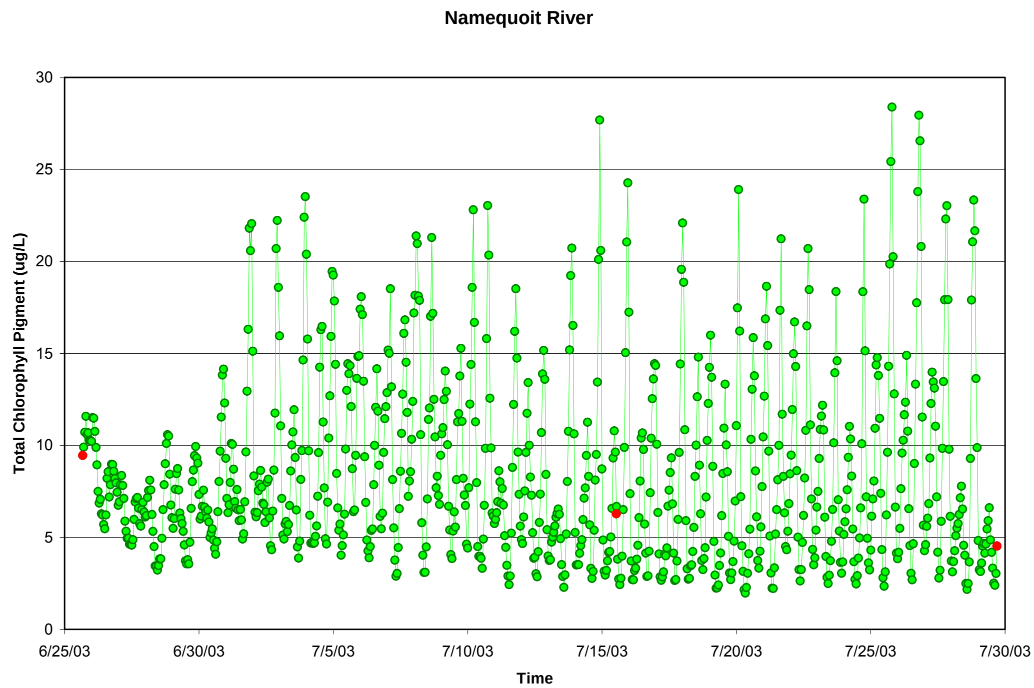


Figure VII-29. Bottom water record of Chlorophyll-*a* in Namequoit River station, Summer 2003. Calibration samples represented as red dots.

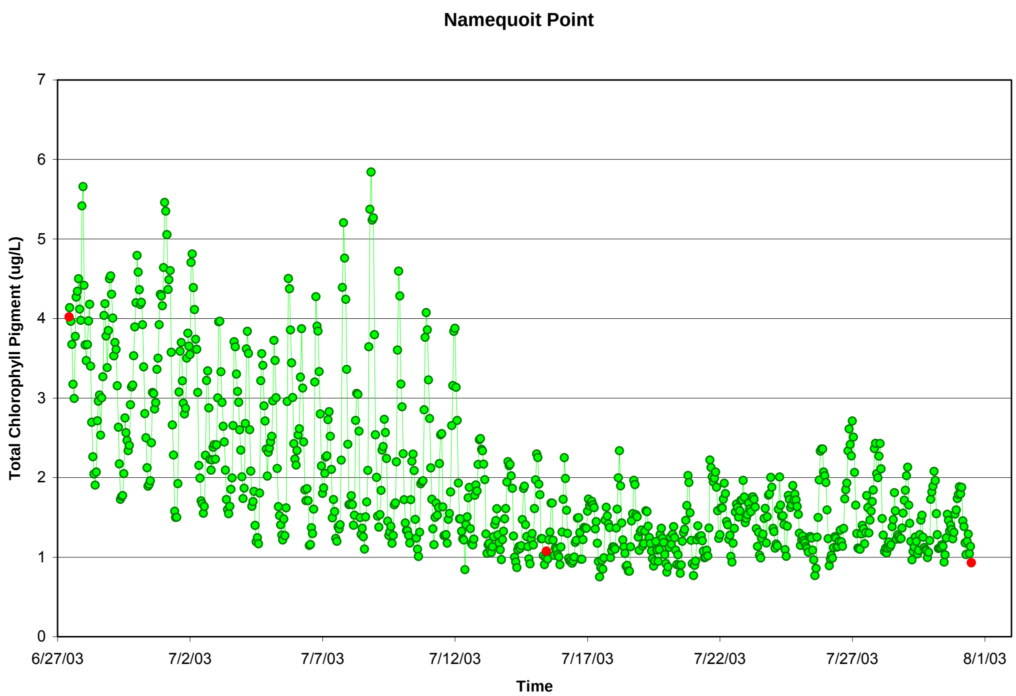


Figure VII-30. Bottom water record of Chlorophyll-*a* in Namequoit Point station, Summer 2003. Calibration samples represented as red dots

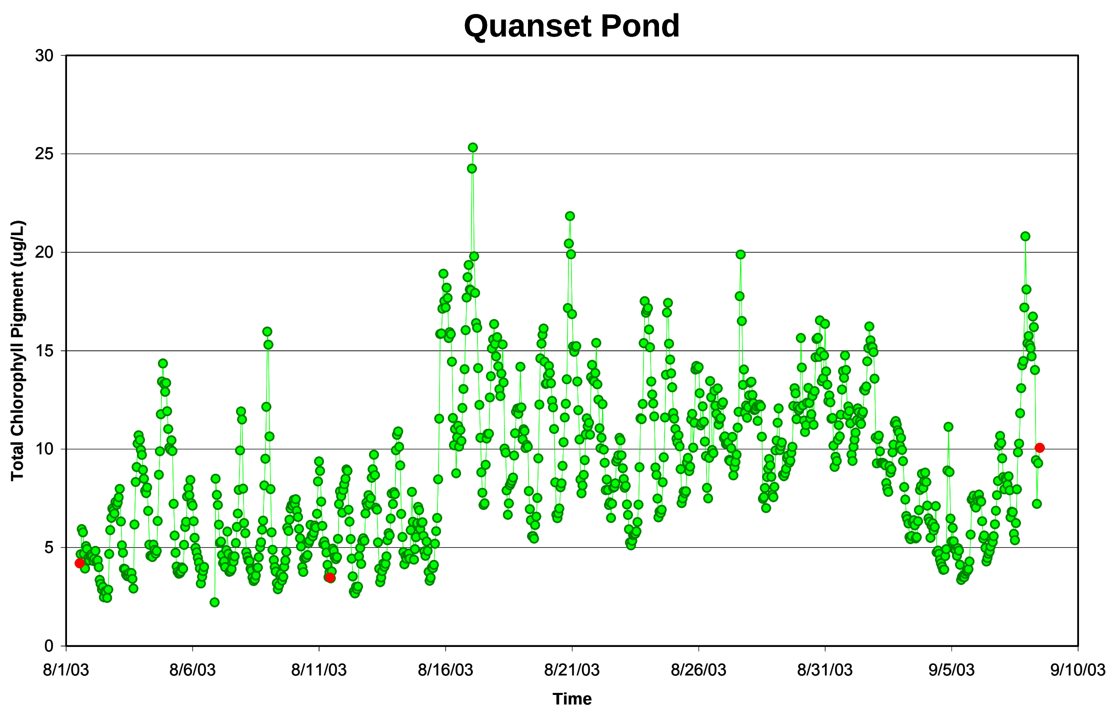


Figure VII-31. Bottom water record of Chlorophyll-*a* in Quanset Pond station, Summer 2003. Calibration samples represented as red dots.

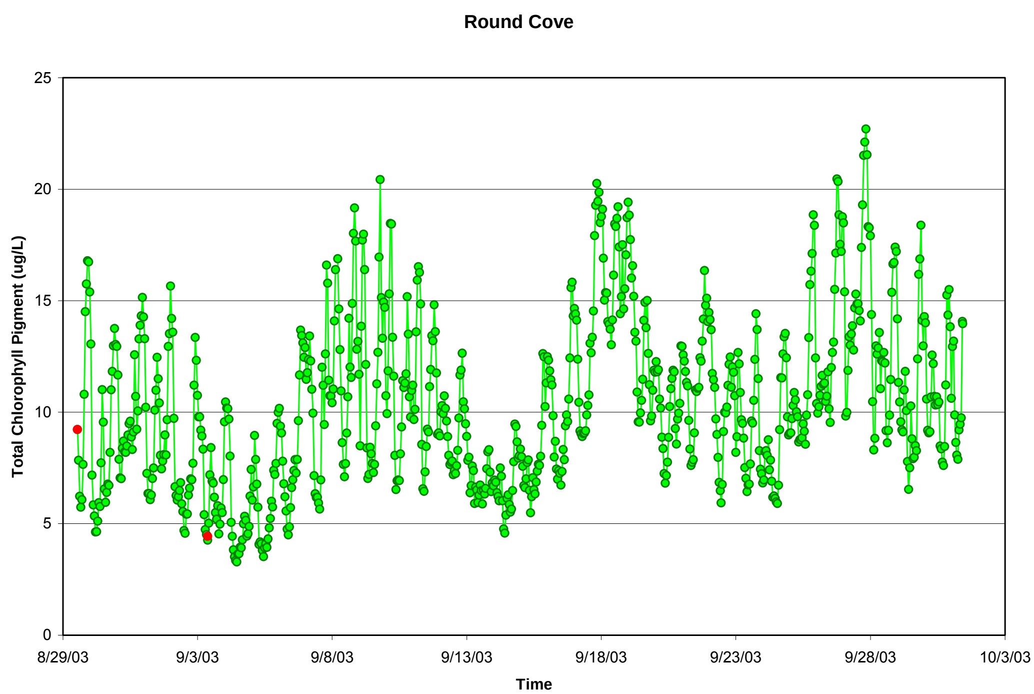


Figure VII-32. Bottom water record of Chlorophyll-*a* in Round Cove station, Summer 2003. Calibration samples represented as red dots.

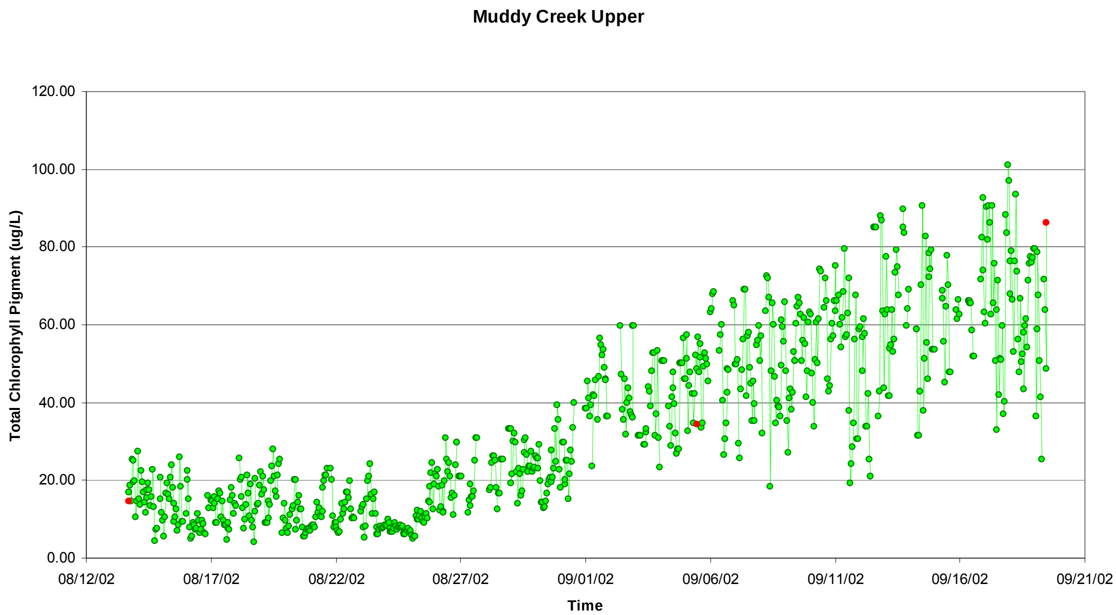


Figure VII-33. Bottom water record of Chlorophyll-*a* in Muddy Creek Upper station, Summer 2002. Calibration samples represented as red dots.

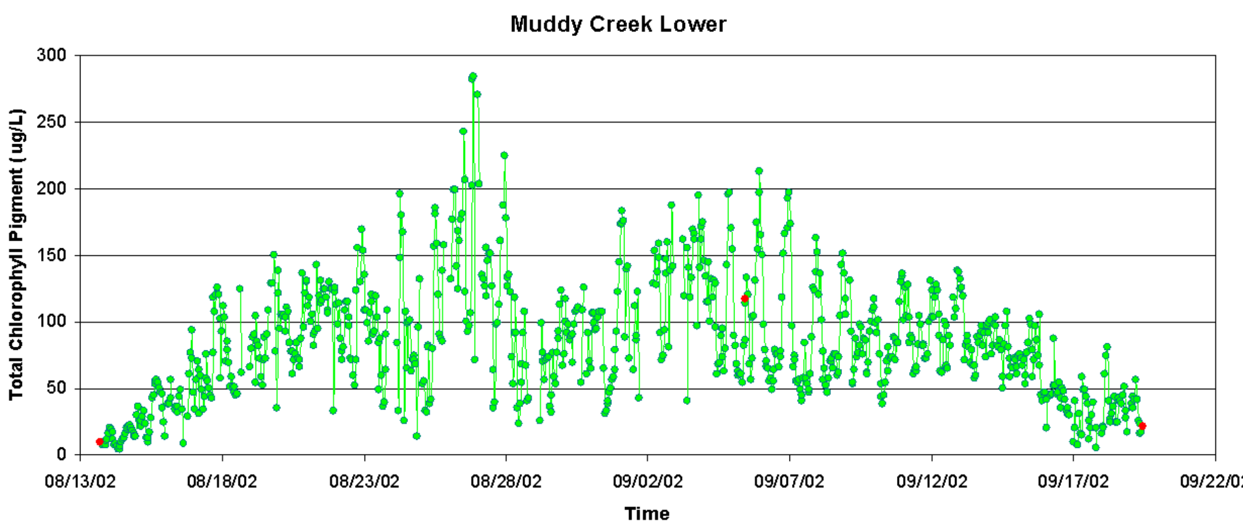


Figure VII-34. Bottom water record of Chlorophyll-*a* in Muddy Creek Lower station, Summer 2002. Calibration samples represented as red dots.

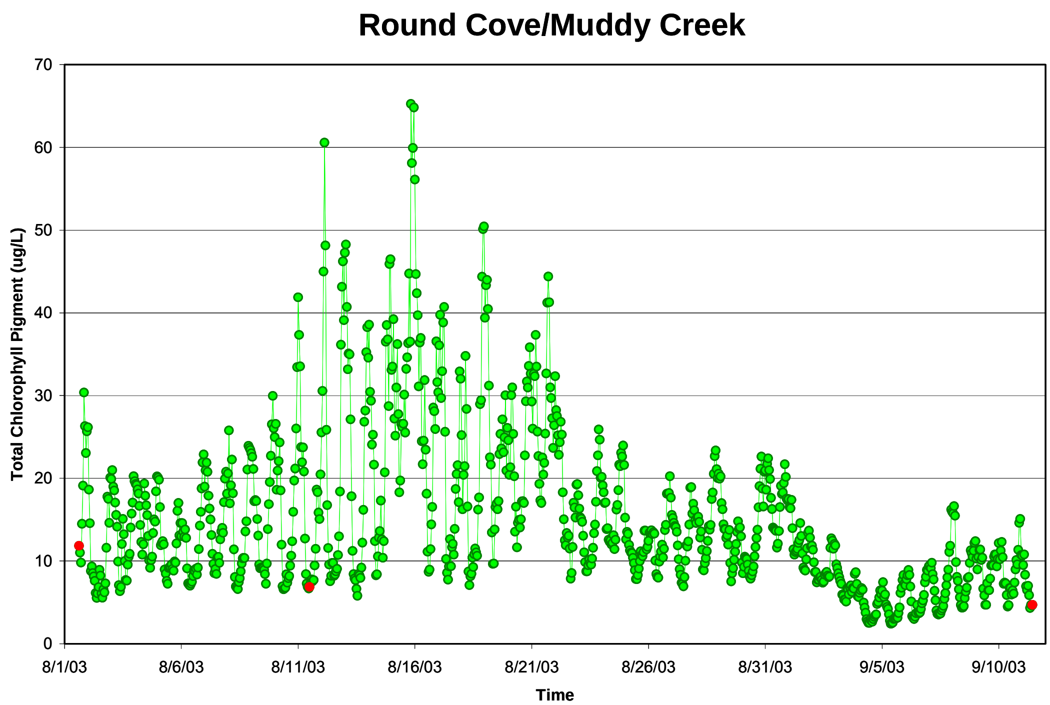


Figure VII-35. Bottom water record of Chlorophyll-*a* in Round Cove / Muddy Creek station located between the inlet to Round Cove and the mouth of Muddy Creek, Summer 2003. Calibration samples represented as red dots.

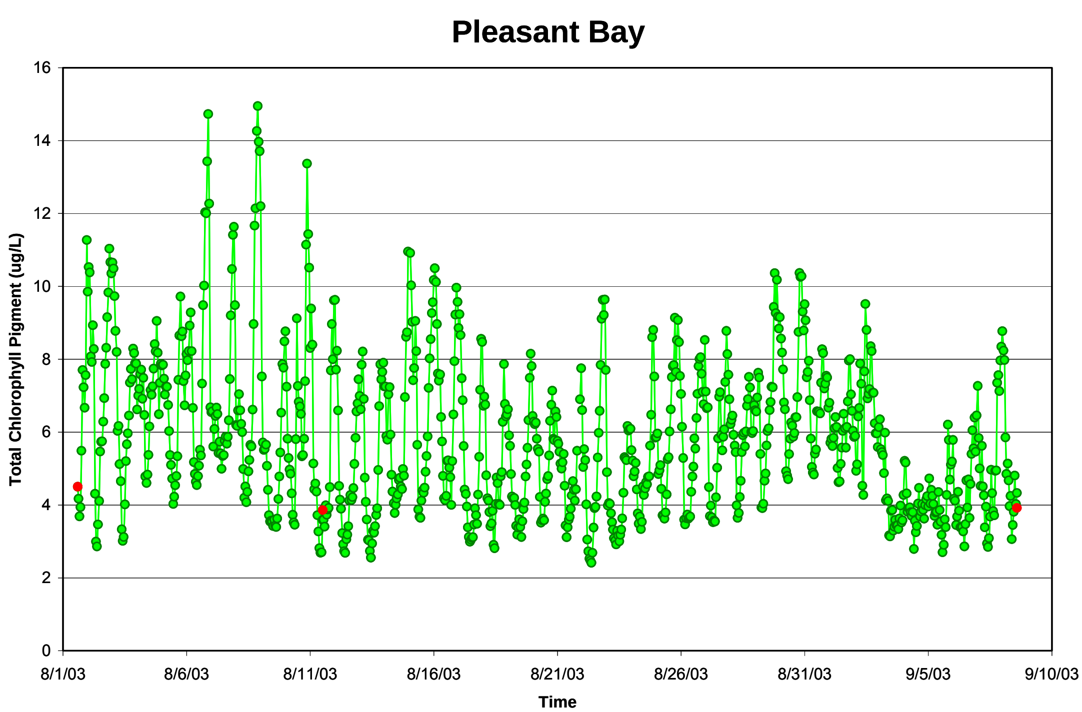


Figure VII-36. Bottom water record of Chlorophyll-*a* in Pleasant Bay station, Summer 2003. Calibration samples represented as red dots.

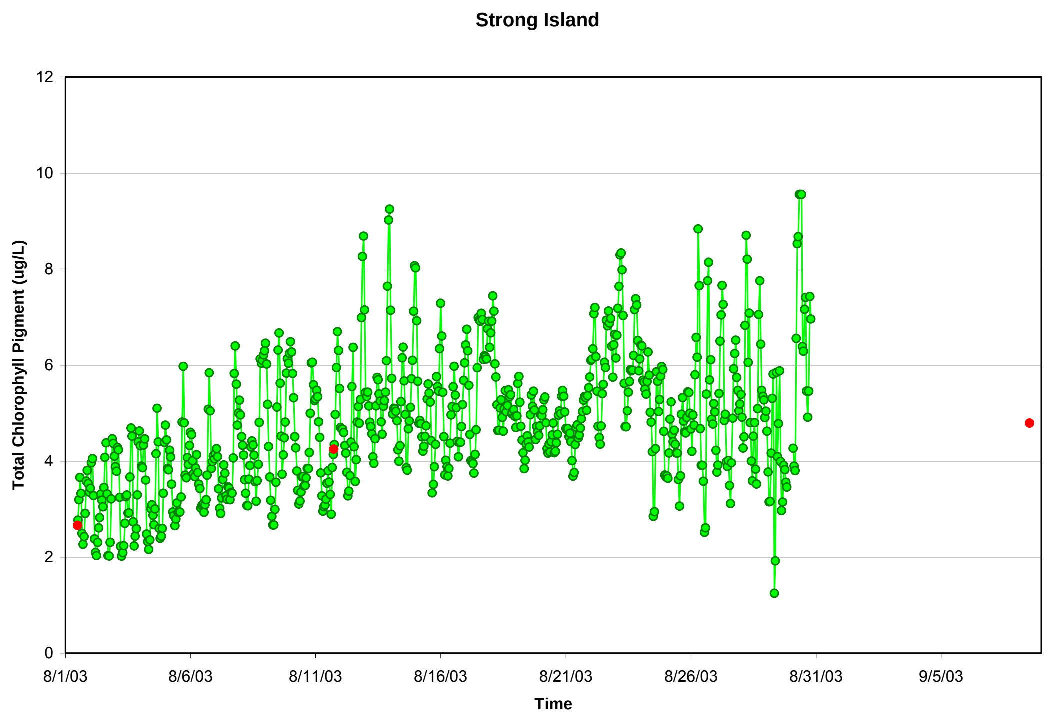


Figure VII-37. Bottom water record of Chlorophyll-*a* in Strong Island station, Summer 2003. Calibration samples represented as red dots.

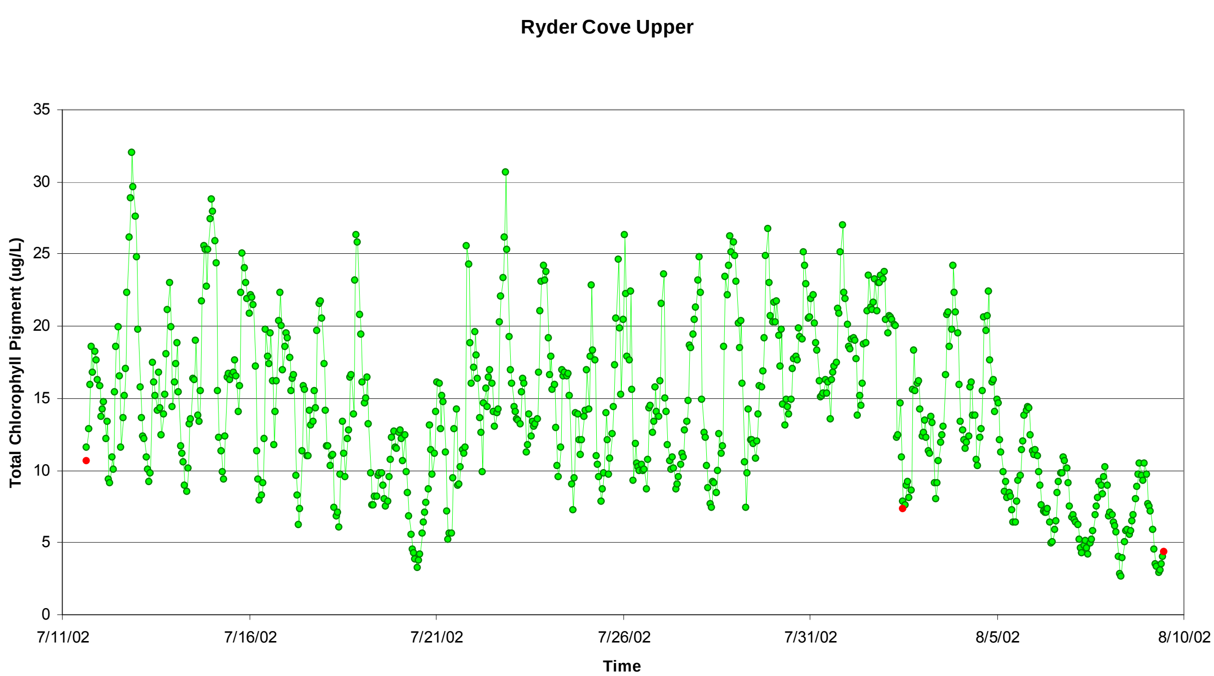


Figure VII-38. Bottom water record of Chlorophyll-*a* in Ryder Cove Upper station, Summer 2002. Calibration samples represented as red dots

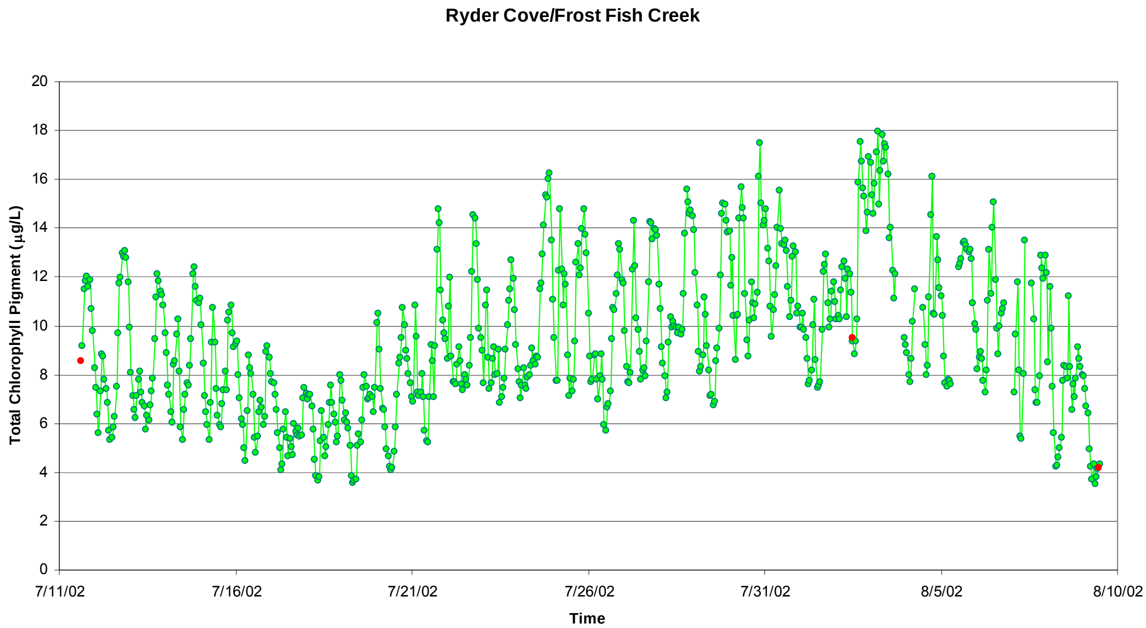


Figure VII-39. Bottom water record of Chlorophyll-*a* in Ryder Cove/Frost Fish Creek station, Summer 2002. Calibration samples represented as red dots.

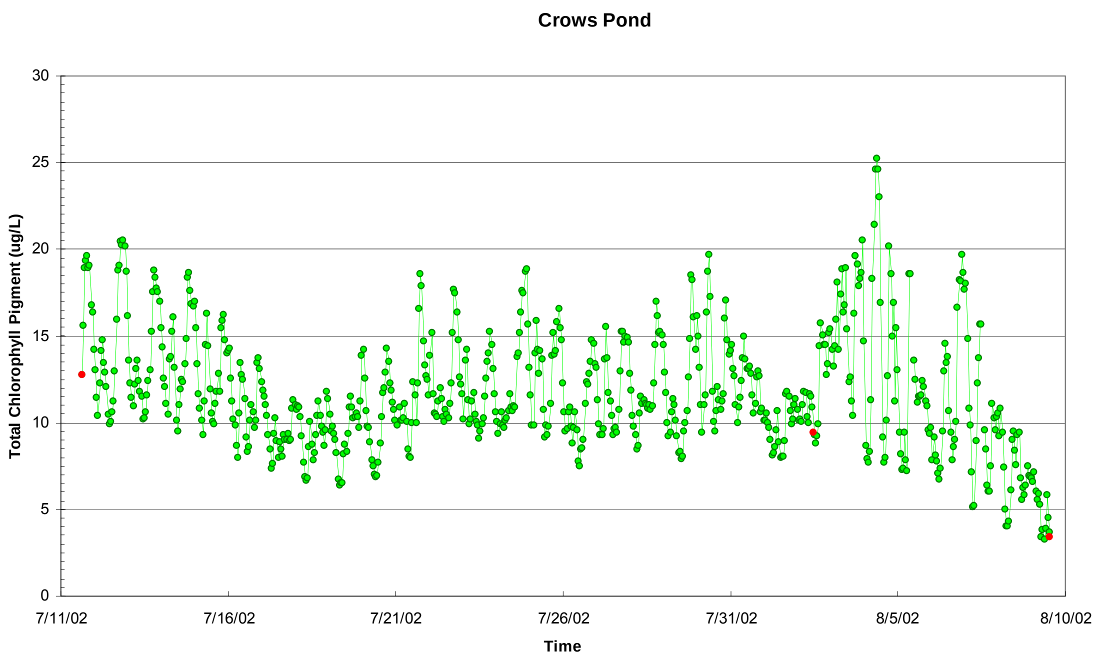


Figure VII-40. Bottom water record of Chlorophyll-*a* in Crows Pond station, Summer 2002. Calibration samples represented as red dots.

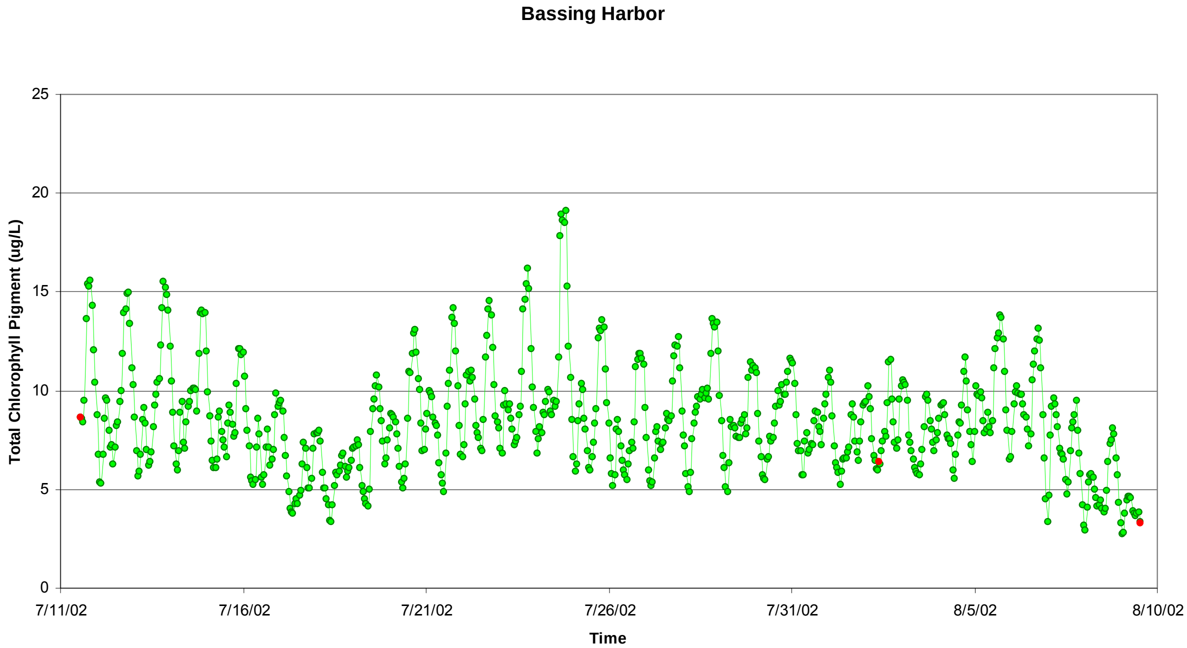


Figure VII-41. Bottom water record of Chlorophyll-*a* in Bassing Harbor station, Summer 2002. Calibration samples represented as red dots.

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

Mooring Location	Start Date	End Date	Total Deployment (Days)	<6 mg/L Duration (Days)	<5 mg/L Duration (Days)	<4 mg/L Duration (Days)	<3 mg/L Duration (Days)
Strong Island	8/1/2003	9/8/2003	38.0	2.80	0.07	0.00	0.00
			Mean	0.16	0.07	N/A	N/A
			Min	0.03	0.07	0.00	0.00
			Max	0.30	0.07	0.00	0.00
			S.D.	0.08	N/A	N/A	N/A
Namequoit River	6/25/2003	7/29/2003	34.0	26.60	24.13	19.84	12.14
			Mean	1.90	1.27	0.94	0.45
			Min	0.27	0.02	0.02	0.01
			Max	18.58	18.44	11.92	1.81
			S.D.	4.81	4.16	2.57	0.42
Lonnies Pond	6/25/2003	7/31/2003	36.1	36.02	34.69	26.53	6.55
			Mean	36.02	4.34	0.41	0.12
			Min	36.02	0.17	0.01	0.01
			Max	36.02	14.92	3.42	0.63
			S.D.	NA	4.61	0.54	0.11
Lonnies Pond Outer	6/25/2003	7/31/2003	35.9	27.60	25.19	21.71	12.23
			Mean	3.07	2.10	0.87	0.33
			Min	0.07	0.05	0.04	0.01
			Max	8.50	6.86	2.88	0.79
			S.D.	3.35	2.48	0.67	0.26
Pochet	8/1/2004	9/22/2004	51.8	25.82	14.78	4.07	0.29
			Mean	0.47	0.24	0.10	0.04
			Min	0.03	0.01	0.01	0.01
			Max	1.61	0.68	0.31	0.06
			S.D.	0.33	0.16	0.06	0.02
Namequoit Point	6/27/2003	7/31/2003	34.1	16.75	6.16	1.35	0.16
			Mean	0.36	0.16	0.12	0.05
			Min	0.01	0.04	0.04	0.04
			Max	0.99	0.45	0.31	0.07
			S.D.	0.23	0.09	0.08	0.02
Areys Pond	6/25/2003	7/31/2003	36.0	31.28	29.71	27.42	23.15
			Mean	0.73	0.57	0.50	0.32
			Min	0.02	0.03	0.01	0.01
			Max	6.51	4.48	3.42	1.09
			S.D.	0.99	0.66	0.53	0.30
Quanset Pond	8/1/2003	9/8/2003	37.9	18.16	11.75	6.77	2.23
			Mean	0.41	0.38	0.34	0.17
			Min	0.01	0.01	0.04	0.03
			Max	3.89	3.32	1.42	1.06
			S.D.	0.65	0.64	0.41	0.27
Round Cove / Muddy Creek	8/1/2003	9/11/2003	40.8	29.05	21.94	13.57	7.65
			Mean	0.56	0.41	0.25	0.21
			Min	0.01	0.01	0.01	0.01
			Max	8.33	7.45	4.14	1.81
			S.D.	1.38	1.07	0.60	0.41
Pleasant Bay	8/1/2003	9/8/2003	38.0	10.93	4.23	1.53	0.46
			Mean	0.20	0.13	0.09	0.06
			Min	0.01	0.02	0.01	0.03
			Max	0.93	0.61	0.27	0.13
			S.D.	0.22	0.14	0.08	0.03
Round Cove	8/29/2003	10/1/2003	32.9	4.28	1.57	0.39	0.27
			Mean	0.19	0.16	0.10	0.07
			Min	0.01	0.02	0.03	0.02
			Max	0.79	0.39	0.14	0.11
			S.D.	0.22	0.12	0.05	0.04
Paw Wah Pond	6/25/2003	7/31/2003	36.0	34.93	33.94	32.67	31.00
			Mean	1.75	1.17	1.05	0.69
			Min	0.02	0.01	0.02	0.02
			Max	10.86	7.10	6.80	5.95
			S.D.	3.25	1.87	1.49	1.04
Meeting House Pond Outer	6/25/2003	7/31/2003	35.9	31.26	22.17	10.16	4.40
			Mean	1.74	0.52	0.25	0.18
			Min	0.01	0.02	0.03	0.05
			Max	10.93	4.20	1.01	0.41
			S.D.	2.66	0.67	0.20	0.11
Meeting House Pond	8/1/2004	9/21/2004	51.0	49.92	47.51	36.72	30.93
			Mean	49.92	2.16	1.53	1.55
			Min	49.92	0.01	0.02	0.01
			Max	49.92	24.68	22.58	20.49
			S.D.	NA	5.73	4.71	4.56

Table VII-2. Pecent 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%
Bassing 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-3. Duration (% of deployment time) that chlorophyll a levels exceed various benchmark levels within the embayment system. “Mean” represents the average duration of each event over the benchmark level and “S.D.” its standard deviation. Data collected by the Coastal Systems Program, SMAST.

Mooring Location	Start Date	End Date	Total Deployment (Days)	>5 ug/L Duration (Days)	>10 ug/L Duration (Days)	>15 ug/L Duration (Days)	>20 ug/L Duration (Days)	>25 ug/L Duration (Days)
Strong Island	8/1/2003	9/8/2003	38.0	11.54	0.00	0.00	0.00	0.00
Mean Chl Value = 4.77 ug/L			Mean	0.23	N/A	N/A	N/A	N/A
			Min	0.04	0.00	0.00	0.00	0.00
			Max	0.96	0.00	0.00	0.00	0.00
			S.D.	0.21	N/A	N/A	N/A	N/A
Namequoit River	6/25/2003	7/29/2003	34.0	24.46	10.17	3.96	1.54	0.21
Mean Chl Value = 8.49 ug/L			Mean	0.45	0.20	0.12	0.08	0.07
			Min	0.08	0.04	0.04	0.04	0.04
			Max	1.67	0.42	0.25	0.17	0.08
			S.D.	0.27	0.07	0.05	0.04	0.02
Lonnies Pond	6/25/2003	7/31/2003	36.1	15.04	4.75	1.33	0.38	0.00
Mean Chl Value = 5.65 ug/L			Mean	0.29	0.16	0.12	0.09	N/A
			Min	0.04	0.04	0.04	0.04	0.00
			Max	3.83	0.79	0.25	0.13	0.00
			S.D.	0.53	0.18	0.08	0.04	N/A
Lonnies Pond Outer	6/25/2003	7/31/2003	35.9	24.08	7.21	1.96	0.13	0.00
Mean Chl Value = 7.18 ug/L			Mean	0.55	0.22	0.12	0.06	N/A
			Min	0.08	0.04	0.04	0.04	0.00
			Max	4.33	0.71	0.21	0.08	0.00
			S.D.	0.75	0.13	0.06	0.03	N/A
Namequoit Point	6/27/2003	7/31/2003	34.1	0.42	0.00	0.00	0.00	0.00
Mean Chl Value = 1.97 ug/L			Mean	0.10	N/A	N/A	N/A	N/A
			Min	0.04	0.00	0.00	0.00	0.00
			Max	0.17	0.00	0.00	0.00	0.00
			S.D.	0.05	N/A	N/A	N/A	N/A
Areys Pond	6/25/2003	7/31/2003	36.0	27.75	18.04	10.17	5.04	2.96
Mean Chl Value = 12.49 ug/L			Mean	0.87	0.39	0.22	0.15	0.14
			Min	0.04	0.04	0.04	0.04	0.04
			Max	7.08	4.25	1.50	0.38	0.38
			S.D.	1.29	0.65	0.27	0.11	0.10
Quanset Pond	8/1/2003	9/8/2003	37.9	29.96	14.08	3.13	0.21	0.04
Mean Chl Value = 8.88 ug/L			Mean	1.30	0.44	0.16	0.07	0.04
			Min	0.04	0.04	0.04	0.04	0.04
			Max	19.83	1.67	0.50	0.08	0.04
			S.D.	4.06	0.38	0.14	0.02	N/A
Round Cove / Muddy Creek	8/1/2003	9/11/2003	40.8	38.38	26.17	15.58	9.58	5.83
Mean Chl Value = 15.23 ug/L			Mean	4.80	0.61	0.43	0.27	0.22
			Min	0.33	0.04	0.04	0.04	0.04
			Max	33.63	3.25	1.88	1.08	0.88
			S.D.	11.65	0.64	0.41	0.26	0.22
Pleasant Bay	8/1/2003	9/8/2003	38.0	22.08	1.63	0.00	0.00	0.00
Mean Chl Value = 5.84 ug/L			Mean	0.46	0.13	N/A	N/A	N/A
			Min	0.04	0.04	0.00	0.00	0.00
			Max	0.96	0.29	0.00	0.00	0.00
			S.D.	0.30	0.08	N/A	N/A	N/A
Round Cove	8/29/2003	10/1/2003	32.9	31.33	15.63	3.96	0.33	0.00
Mean Chl Value = 10.31 ug/L			Mean	2.85	0.31	0.14	0.08	N/A
			Min	0.08	0.04	0.04	0.04	0.00
			Max	17.04	1.88	0.50	0.17	0.00
			S.D.	5.04	0.29	0.13	0.06	N/A
Paw Wah Pond	6/25/2003	7/31/2003	36.0					
Macro Algae Interference			Mean					
Data not reliable			Min					
			Max					
			S.D.					
Meeting House Pond Outer	6/25/2003	7/31/2003	35.9	18.63	5.46	1.38	0.63	0.13
Mean Chl Value = 6.22 ug/L			Mean	0.32	0.22	0.14	0.16	0.13
			Min	0.04	0.04	0.04	0.08	0.13
			Max	0.96	0.79	0.25	0.21	0.13
			S.D.	0.27	0.16	0.08	0.05	N/A
Pochet			51.8	15.96	2.29	0.21	0.00	0.00
Mean Chl Value = 4.95 ug/L	8/1/2004	9/22/2004	Mean	0.23	0.46	0.07	N/A	N/A
			Min	0.04	0.25	0.04	0.00	0.00
			Max	4.96	0.83	0.13	0.00	0.00
			S.D.	0.60	0.27	0.05	N/A	N/A
Meeting House Pond	8/1/2004	9/21/2004	51.0	30.63	10.67	2.92	0.38	0.00
Mean Chl Value = 6.82 ug/L			Mean	0.63	0.27	0.16	0.09	N/A
			Min	0.04	0.04	0.04	0.04	0.00
			Max	5.92	1.75	0.71	0.21	0.00
			S.D.	1.01	0.33	0.17	0.08	N/A

Table VII-4. 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)					Frequency (# events)				
				>5 ug/L (Days)	>10 ug/L (Days)	>15 ug/L (Days)	>20 ug/L (Days)	>25 ug/L (Days)	>5 ug/L (#)	>10 ug/L (#)	>15 ug/L (#)	>20 ug/L (#)	>25 ug/L (#)
Basssing 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-4. (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 Lower	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 Upper	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					

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data was conducted for the Pleasant Bay System by the MassDEP Eelgrass Mapping Program as part of the MEP Technical Team. Surveys were conducted in 1995 and 2001, as part of this program. Additional analysis of available aerial photographs from 1951 was used to reconstruct the eelgrass distribution prior to any substantial development of the watershed. In the case of Round Cove, the 1951 aerial photograph was difficult to interpret with certainty due to conditions at the time the photograph was taken. Therefore, a later aerial photograph (1960) was identified for interpretation, however, that photograph was of poor quality as well. Presence or absence of eelgrass in 1951 or 1960 could not be determined with certainty. The 1951 data were only anecdotally validated, while the 1995 and 2001 maps were field validated. The primary use of the data is to indicate (a) if eelgrass once or currently colonizes a basin and (b) if large-scale system-wide shifts have occurred. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 1995 to 2001 (Figure VII-42 through VII-46); the period in which watershed nitrogen loading significantly increased to its present level. This temporal information can be used to determine the stability of the eelgrass community.

Pleasant Bay Eelgrass Presence

At present, eelgrass is present within large portions of the Pleasant Bay System, indicative of a system with high habitat quality areas. These eelgrass beds are generally restricted to the larger lagoonal basins, Little Pleasant Bay, Pleasant Bay and Chatham Harbor. There are also smaller eelgrass areas in Pochet and fringing shallow areas in The River and Meetinghouse Pond. The only tributary embayment to Pleasant Bay with significant eelgrass habitat is Bassing Harbor (see below). The basins presently supporting eelgrass habitat also supported habitat in the 1951 historical analysis. However, it is clear from the 1951, 1995 and 2001 temporal sequence that the eelgrass areas in each basin, except Chatham Harbor, are declining in coverage. In The River and Pochet the eelgrass areas were always patchy and in the shallows. By the 2001 survey this pattern continues, but the beds appear to be declining, although they persist.

Virtually all of the small enclosed basins (group A, above) did not appear to support eelgrass historically and do not support it today, with the exception of the small patch in the shallows of Meetinghouse Pond and in lower Muddy Creek (see below). The general pattern is consistent with the deeper waters of these basins and their location and structure which tends to result in nitrogen enrichment.

The overall pattern of eelgrass distribution and temporal decline in coverage is fully consistent with the spatial pattern of nitrogen enrichment (Chapter VI) and oxygen and chlorophyll levels in the various basins (see above). The pattern of decline is typical of environmental changes wrought by nutrient enrichment. Nutrient enrichment tends to result in loss of eelgrass habitat in the more tidally restricted basins which also tend to be the focus areas for watershed nitrogen inputs. Loss is first in the deeper waters (like in kettle basins) where increases in turbidity from increased phytoplankton production cause shading of the bottom. The pattern of loss from the tidal reaches furthest from the inlet can also be seen in the Pleasant Bay System, where healthy beds remain within the region of the Chatham Harbor basin.

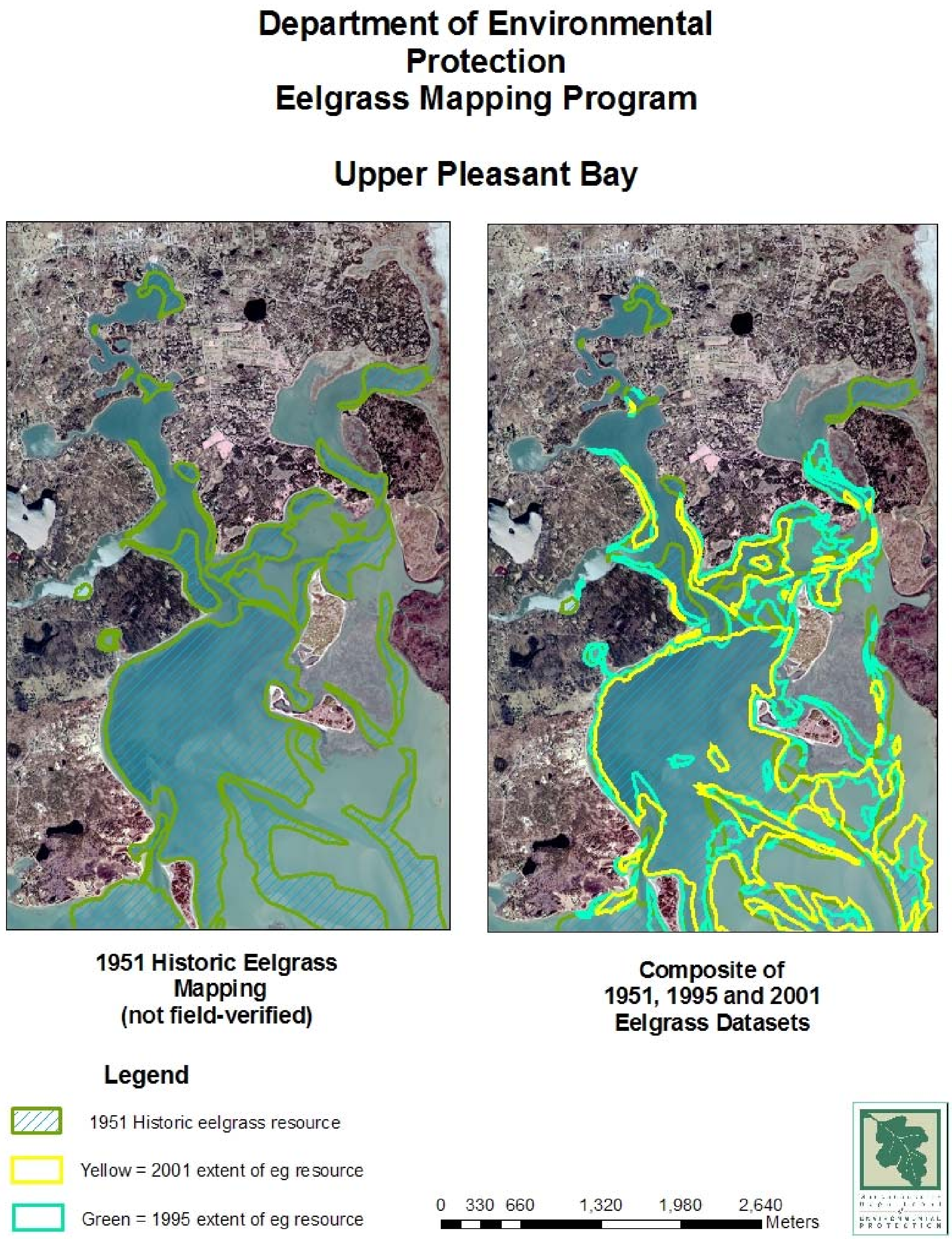


Figure VII-42. Eelgrass bed distribution within the upper portion of the Pleasant Bay System. The 1995 coverage is depicted by the green outline inside of which circumscribes the eelgrass beds. The yellow (2001) areas were mapped by MassDEP. All data was provided by the MassDEP Eelgrass Mapping Program.

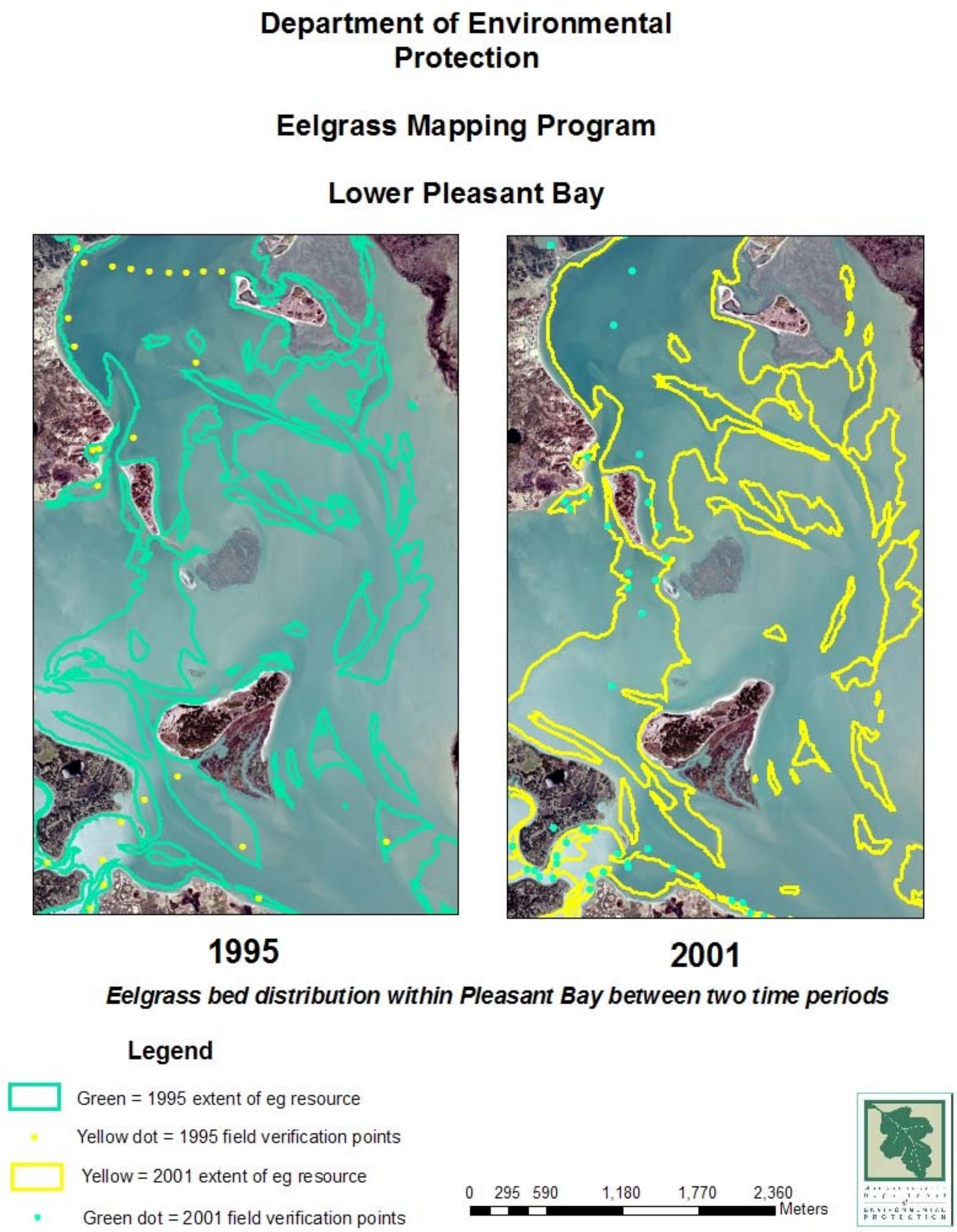


Figure VII-43. Eelgrass bed distribution within the lower portion of the Pleasant Bay System. The 1995 coverage is depicted by the green outline inside of which circumscribes the eelgrass beds. The yellow (2001) areas were mapped by MASSDEP. All data was provided by the MASSDEP Eelgrass Mapping Program

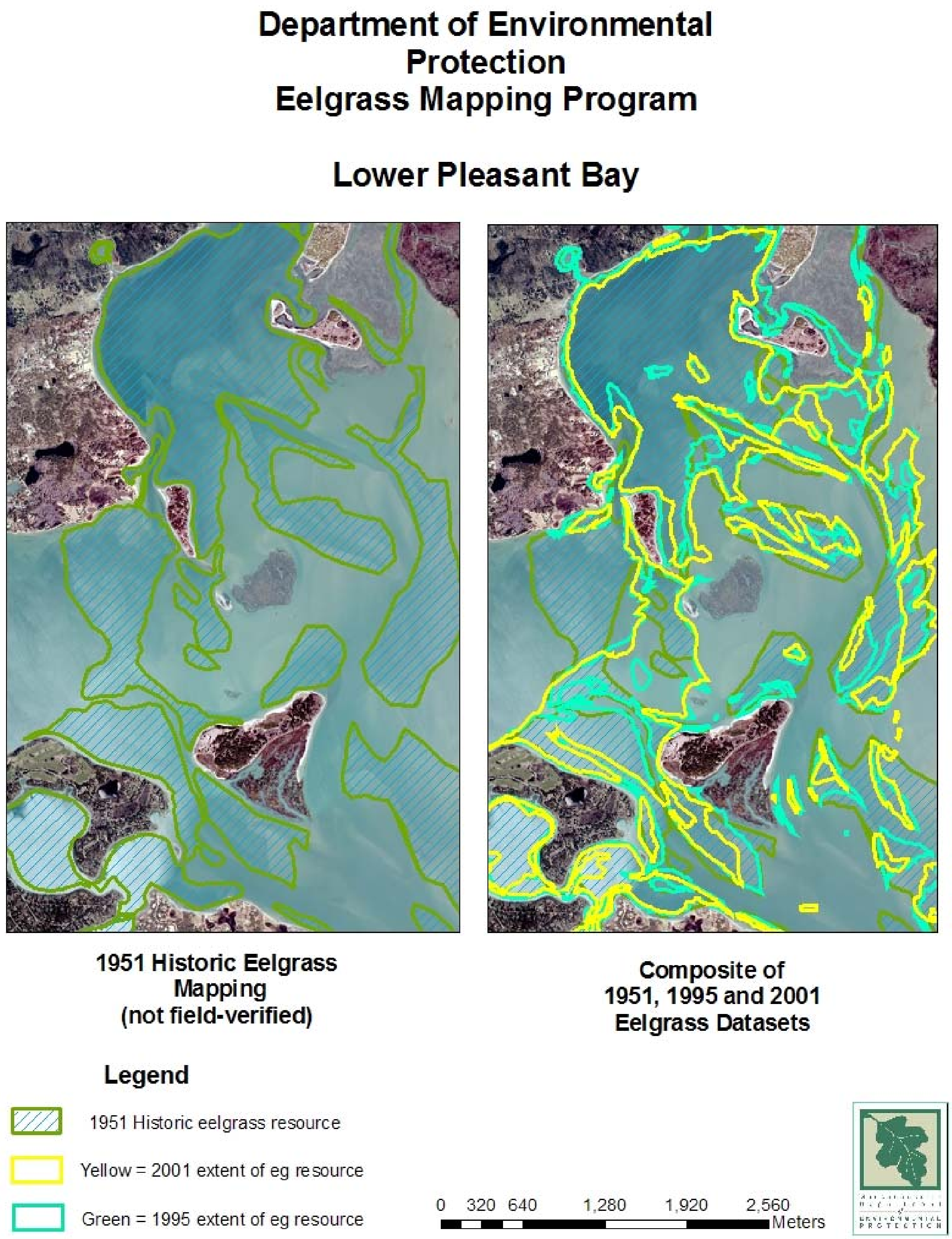


Figure VII-44. Eelgrass bed distribution within the lower portion of the Pleasant Bay System. The 1995 coverage is depicted by the green outline inside of which circumscribes the eelgrass beds. The yellow (2001) areas were mapped by MASSDEP. Aerial photography for Round Cove (1951, 1960) was inclusive. All data was provided by the MASSDEP Eelgrass Mapping Program.

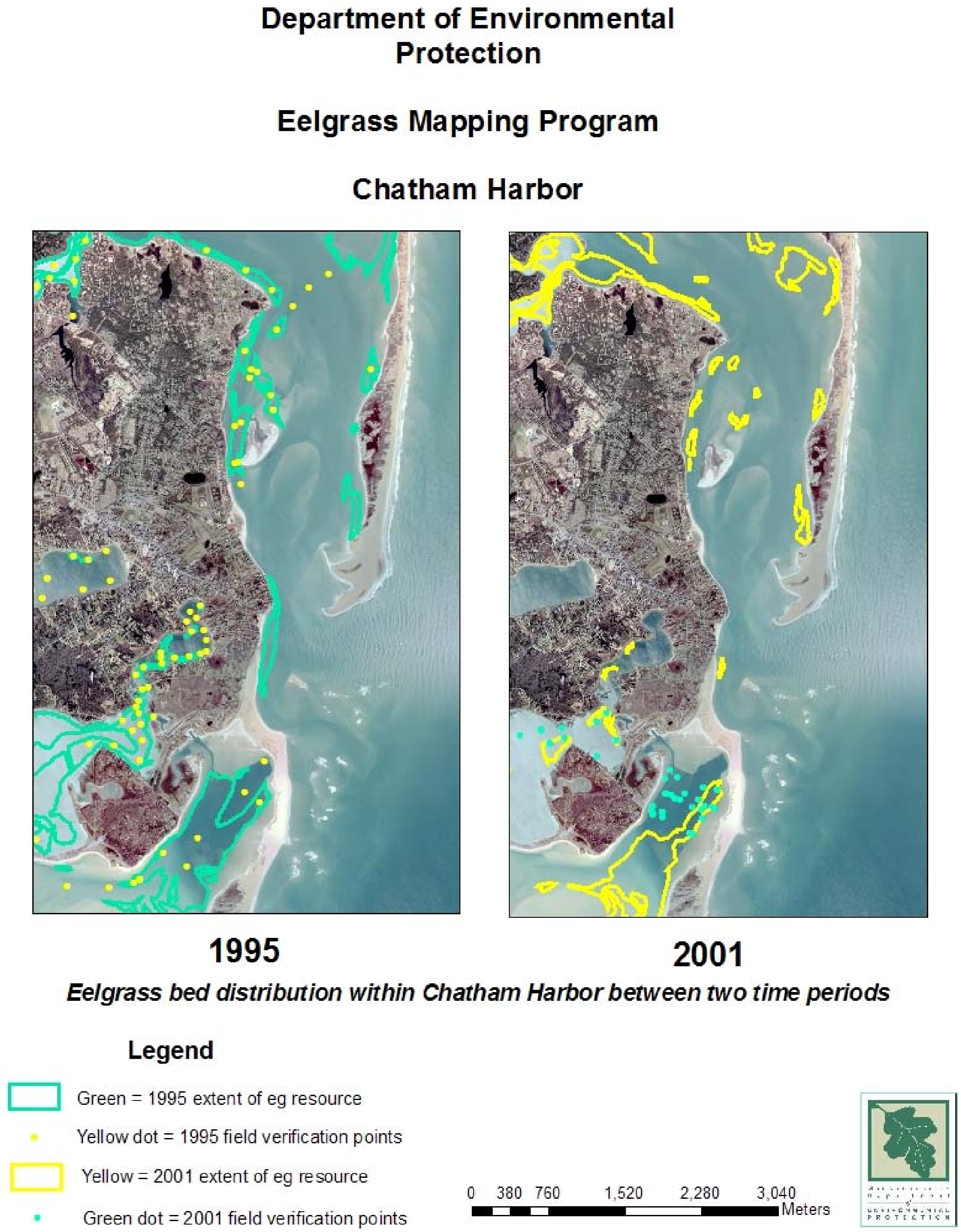


Figure VII-45. Eelgrass bed distribution within the Chatham Harbor portion of the Pleasant Bay System. The 1995 coverage is depicted by the green outline inside of which circumscribes the eelgrass beds. The yellow (2001) areas were mapped by MASSDEP. All data was provided by the MASSDEP Eelgrass Mapping Program.

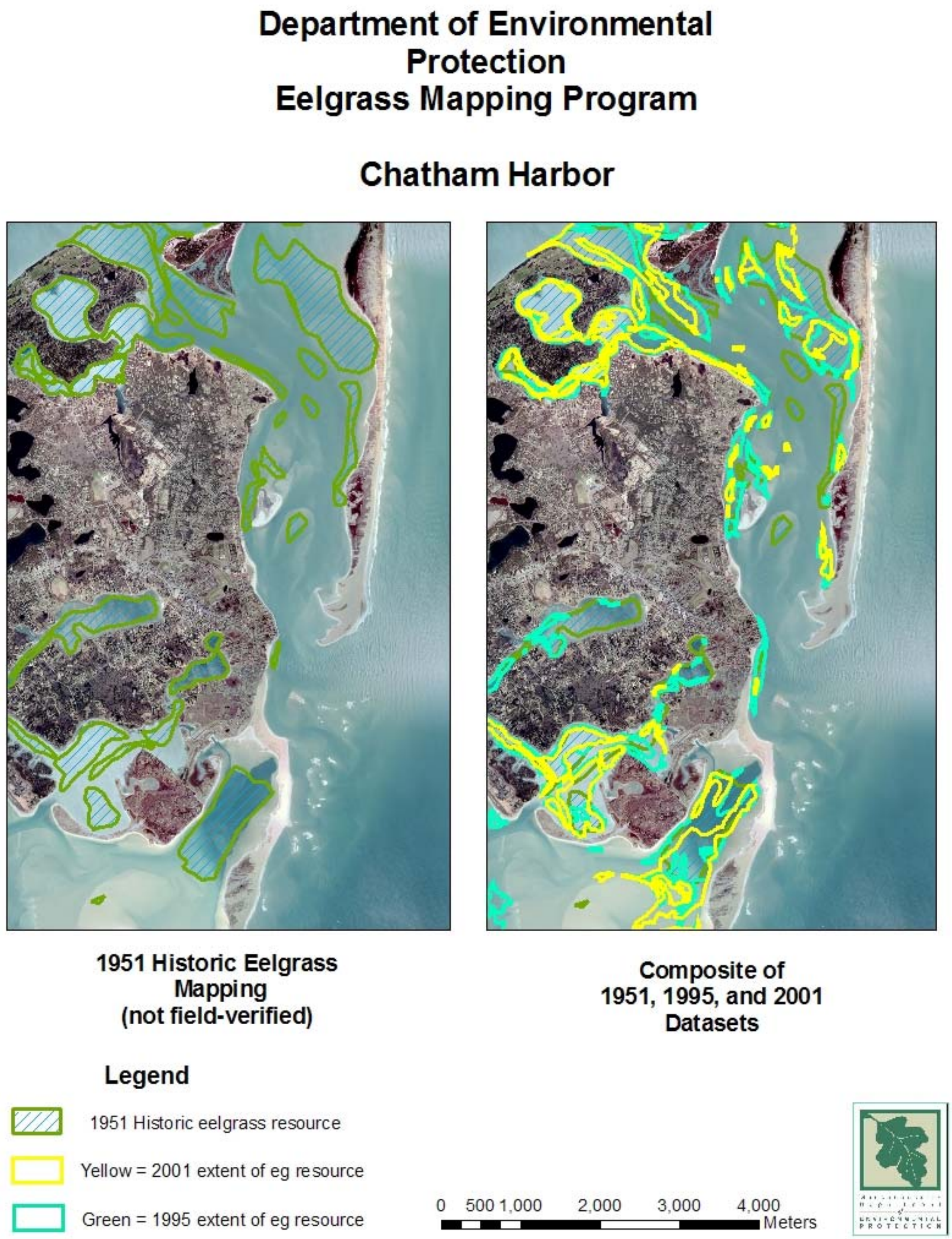


Figure VII-46. Eelgrass bed distribution within the Chatham Harbor portion of the Pleasant Bay System. The 1995 coverage is depicted by the green outline inside of which circumscribes the eelgrass beds. The yellow (2001) areas were mapped by MASSDEP. All data was provided by the MASSDEP Eelgrass Mapping Program.

Other factors which influence eelgrass bed loss in embayments may also be at play in the Pleasant Bay system, though the loss seems completely in-line with nitrogen enrichment. However, a brief listing of non-nitrogen related factors is useful. Eelgrass bed loss does not seem to be directly related to mooring density, as the system main basins where eelgrass areal loss is greatest have relatively few boat moorings. Similarly, pier construction and boating pressure may be adding additional stress in nutrient enriched areas, but do not seem to be the overarching factor. It is not possible at this time to determine the potential effect of shellfishing on eelgrass bed distribution.

It is possible to determine a general idea of short- and long-term rates of change in eelgrass coverage from the mapping data, although there are only 3 surveys. Over the 50 year period1951-2001 the Pleasant Bay System has lost ~583 acres of eelgrass habitat. Interestingly, the rate of loss has been relatively constant at ~11 acres per year. This loss has occurred as watershed nitrogen loading rates gradually increased several fold due to changes in land use within the Pleasant Bay watershed.

Based upon the 1951 coverage it appears that nitrogen management to restore eelgrass habitat has the potential to recover a significant resource to this System and to the lower Cape (Table VII-5). However, for the reasons discussed above creation of eelgrass habitat within the enclosed basins is unlikely and not supported by the historical analysis. Since most of the eelgrass habitat has been lost in the larger basins, it is likely that the smaller enclosed basins will see improved habitat quality, as most of the nitrogen entering from the watershed enters first into these systems before being flushed to Little Pleasant Bay and Pleasant Bay.

Table VII-5. Changes in eelgrass coverage in the Pleasant Bay Embayment System within the Towns of Chatham, Orleans, Brewster and Harwich over the past half century (C. Costello).

Pleasant Bay System			
Eelgrass Coverage (acres)			% Change from 1951
1951	1995	2001	
2390	1899	1807	-24%

Coverages by DEP Eelgrass Mapping Program

Bassing Harbor Eelgrass Presence

Included as part of the MEP Nutrient Threshold Report for the Town of Chatham, a detailed eelgrass survey was conducted in the Fall of 2000 for the Bassing Harbor sub-embayment to Pleasant Bay. 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 was assessed in order to determine the degree to which the eelgrass resources affects system function. Density relates to the amount of bottom covered with eelgrass within the 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 MASSDEP in support of MEP (C. Costello), provided a view of temporal trends in eelgrass distribution from 1951 to 1994/5 to 2000. This temporal information was 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 could be determined. However, the primary use of the data indicated (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.

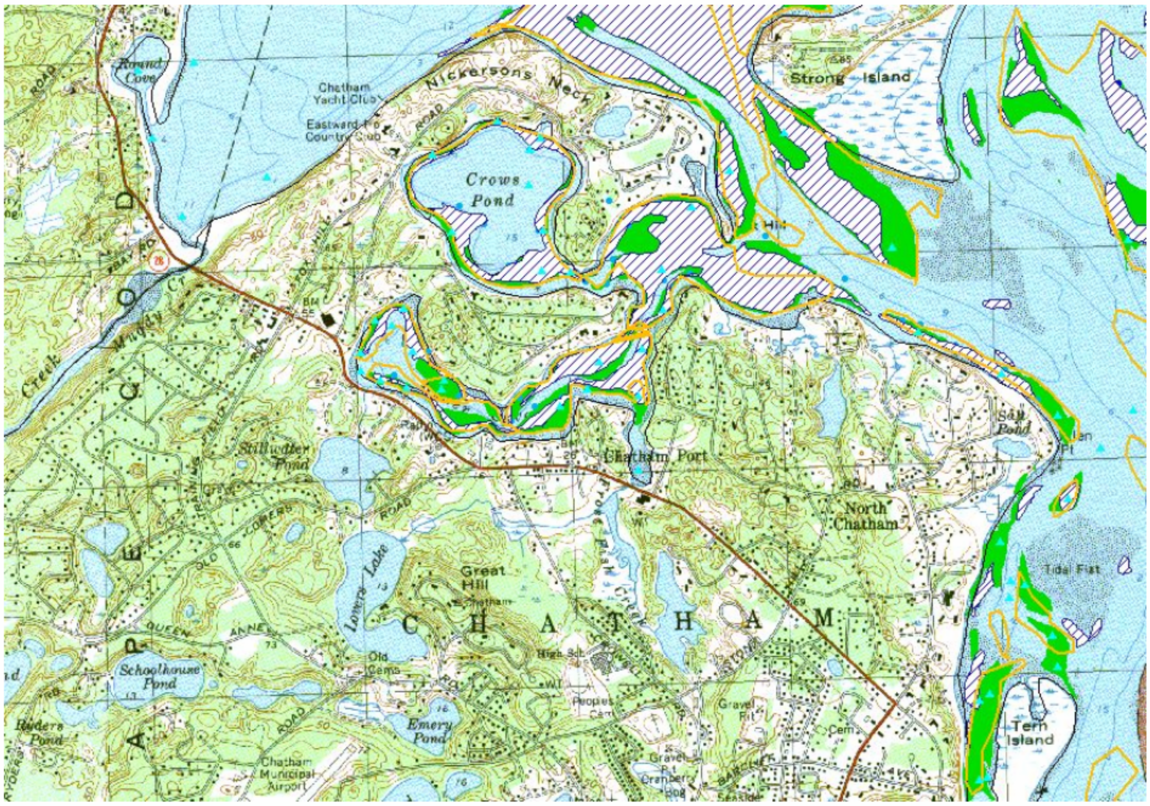
In 2000 only the larger embayment systems in Chatham contained notable eelgrass coverage. Muddy Creek was devoid of eelgrass except for a small patch (about 10% density) adjacent the inlet. The eelgrass survey data from the Bassing Harbor Systems was used to produce the eelgrass coverage maps shown in Figure VII-48. Within the Bassing Harbor system, eelgrass was not observed within the Frost Fish Creek sub-embayment in the Bassing Harbor System.

Due to our concern over potential recent changes in nutrient conditions within the Bassing Harbor sub-embayment system to Pleasant Bay 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 Bassing Harbor system. Figure VII-47 shows the distribution of eelgrass coverage in 1994/5.

The 1951 eelgrass distribution maps for the Bassing Harbor System (Figure VII-47) suggests that eelgrass coverage was significantly greater in some of the sub-embayments when compared to present conditions. In 1994, the Bassing Harbor system still appeared to have relatively good coverage, however, significant loss becomes apparent by 2000. In reviewing the series of aerials, it appears that most of the Bassing Harbor sub-embayment systems was capable of supporting relatively dense eelgrass stands in 1951.

Based on the 1951, 1995 and 2001 MASSDEP mapping data, it was possible to determine a general idea of short and long term rates of change in eelgrass coverage. However, as the 2000 mapping program (completed independent of the MASSDEP eelgrass mapping effort) was done fully by on-site transect surveys, sparse eelgrass beds could be detected that typically could not be resolved via aerial mapping techniques (Table VII-6). Therefore, while the 2000 study may represent more fully the eelgrass situation, it was not directly comparable to the historical data. Therefore, to determine historical changes we used the distribution shown in Figure VII-48, which were all generally collected using a similar approach (Table VII-6). The latter data represent relatively established beds and therefore the spatial coverage's are less than observed in the transect study. Nonetheless, it is clear that each of the sub-embayments to the Bassing Harbor (Figure VII-46) System 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 the Bassing Harbor system (Table VII-7). That this change is still occurring is seen in the aerial mapping data (Table VII-7) and by comparing the 1994/5 and 2000 maps for each system. Since the 2000 maps (Figures VII-48) use a more sensitive technique than the 1994/5 maps (Figure VII-47), the lower coverage in 2000 suggests a "true" loss of bed area.

CHATHAM – RYDERS COVE



Historic 1951 Photos	Orange
1995 Photos and extensive fieldwk.	Green
2000 Photos and extensive fieldwk.	Blue hatched

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-47. 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.

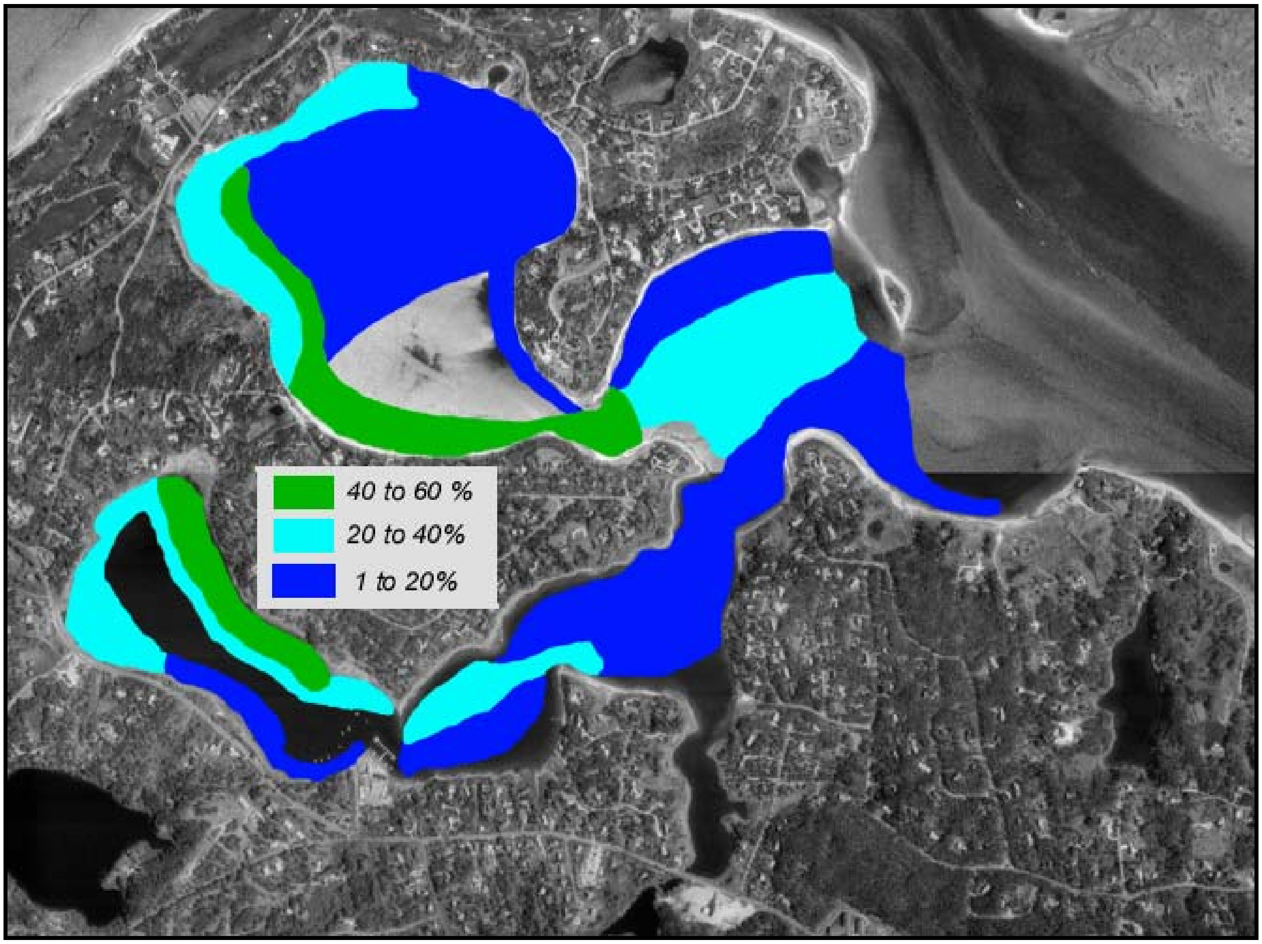


Figure VII-48. Map of Bassing Harbor eelgrass distribution and density (percent of cover) as observed in 2000.

Table VII-6. 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)	Eelgrass Density	Area (ac)	Coverage Area percentage of total embayment area
Bassing Harbor System			
Crows Pond (115.7 ac)	40 to 60%	17.2	14.8
	20 to 40%	17.3	14.9
	1 to 20%	65.4	56.5
Ryder Cove (46.9 ac)	40 to 60%	9.5	20.3
	20 to 40%	15.1	32.1
	1 to 20%	5.1	10.9
Outer Ryder Cover (54.2 ac)	20 to 40%	6.9	12.8
	1 to 20%	34.1	62.9
Bassing Harbor (86.5 ac)	40 to 60%	3.7	4.3
	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 Eelgrass coverage: 231 ac Percent coverage total system: 72.2%			

Table VII-7. Changes in eelgrass coverage in the Bassing Harbor embayment system within the Town of Chatham over the past half century (C. Costello). Note: data from Table VII-6 collected by different approach not included.

Embayment*	1951 (acres)	1995 (acres)	2000 (acres)	% Difference (1951 to 2000)
Bassing Harbor System	**246**	**153**	**114**	**46%**

*No Eelgrass in the Following Embayment Areas: Muddy Creek and Frost Fish Creek.

The pattern of eelgrass loss in the Bassing Harbor system 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 Bassing Harbor, 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 Bassing Harbor system in the Town of Chatham.

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 (Table VII-6 and VII-7 show changes to eelgrass coverage). They can be summarized as follows:

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

- 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 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 34 locations throughout the Pleasant Bay System (Figure VII-49). In some cases multiple samples and assays were conducted at a station. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that, given the loss of eelgrass beds, certain portions of the

Pleasant Bay System is showing indications of impairment due to nutrient overloading. However, to the extent that it can still support healthy infaunal communities, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

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

Pleasant Bay Infaunal Characterization

The Infauna Study indicated that as for the oxygen and chlorophyll indicators and the distribution of sediment metabolism, the enclosed basins (group A, above) are generally significantly to severely impaired relative to benthic infaunal habitat quality. Among the enclosed basins, all were at least significantly impaired. Paw Wah Pond is virtually devoid of benthic animals (only 1-4 individuals per sample) as would be expected from its high level of oxygen stress. Similarly, Areys Pond, Quanset Pond, Upper Muddy Creek supported significantly depleted benthic animal populations, consistent with their nitrogen related oxygen stress. The other enclosed basins were able to support benthic infauna, but the community was dominated by opportunistic species (*Capitella*, *Streblospio*) indicative of very high organic matter loading (Lonnies Pond, Meetinghouse Pond outlet channel) or by intermediate stress indicators (*Gemma*, Amphipods). The dominance of these intermediate indicators in The River, Round Cove, Meetinghouse Pond suggests that these systems, which also showed only moderate oxygen stress, are only moderately beyond their nitrogen loading limits (Table VII-8).

The larger lagoonal basins of Little Pleasant Bay generally supported infaunal communities indicative of a moderate level of stress from organic matter loading and oxygen depletion. However, the pattern was for a decrease in habitat quality moving from the margins to depths. This pattern is typical of a system near, but beyond its nitrogen loading limit, where organic matter deposition in the deep basin areas is the proximate cause of the impairment of benthic habitat quality. Chatham Harbor habitat supported only moderate numbers of individuals and species, but this appeared to result from the dynamic nature of the bottom sediments (unstable bottom), due to the high tidal velocities, rather than nutrient related impairment.

Bassing Harbor Infaunal Characterization

A separate analysis of the habitat quality within the Bassing Harbor sub-system was conducted in 2001. Quantitative sediment sampling was conducted at 7 locations within the Bassing Harbor sub-embayment system to Pleasant Bay (Figure VII-50). Tidal salt marsh creeks and shallow pools were excluded. Samples were collected from: Ryder Cove, Bassing Harbor, Frost Fish Creek, Crows Pond, and Muddy Creek. 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.). As previously mentioned above, certain species or species assemblages reflect the quality of the habitat in which they live. 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 Bassing Harbor embayment are currently supporting habitats under either intermediate or high stress (Table VII-8). The lower regions (those nearest the inlet to Bassing Harbor) 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 from Pleasant Bay.

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 basin within the Bassing Harbor System generally registered as intermediate habitat quality. Only 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 Bassing Harbor System (for example Crows Pond and Bassing Harbor). The converse is also true, with poorest habitat quality found in upper Muddy Creek (H'=1.35, E=0.52) and Frost Fish Creek (H'=1.53, E=0.66).

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 the Town of Chatham embayments. The majority of the area within the Bassing Harbor system appeared 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.

Given the present ecological status of the small enclosed basins (significantly to severely degraded) and the large upper lagoonal basins (moderately impaired), nitrogen management is likely to restore large areas of benthic habitat throughout much of the Pleasant Bay System.

Figure VII-49. Aerial photograph of the Pleasant Bay system showing location of benthic infaunal sampling stations (red symbols).

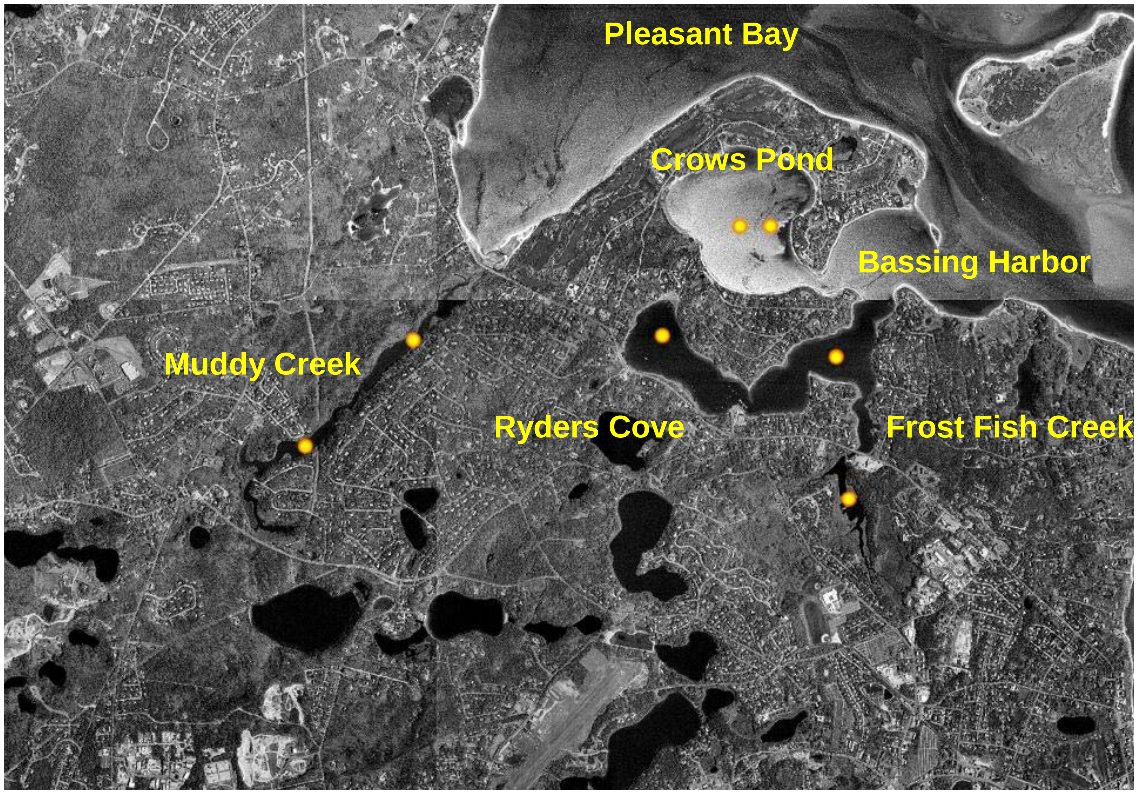


Figure VII-50. Aerial photograph of the Bassing Harbor system showing location of benthic infaunal sampling stations (orange symbols).

Figure VII-51. Aerial photograph of the Round Cove sub-embayment (Town of Harwich) showing location of benthic infaunal sampling stations (red symbols).

Table VII-8. Benthic infaunal community data for the Pleasant Bay System. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.0625 m2).

Location	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Meetinghouse Pond					
Sta. 47	6	672	6	1.60	0.62
Sta. 48	6	752	6	1.90	0.73
Sta. 49	7	800	6	1.65	0.59
Lonnies Pond					
Sta. 53a	12	801	10	1.93	0.54
Sta. 53b	7	993	6	1.52	0.54
Areys Pond					
Sta.22,23A	4	128	5	1.67	0.84
Sta. 22,23B	4	57	N/A	1.48	0.74
The River					
Sta. 50a	7	256	7	2.41	0.86
Sta. 50b	11	1411	8	2.12	0.61
Sta. 52a	8	2816	8	2.12	0.71
Sta. 52b	10	2080	10	2.74	0.83
Sta. 26a	10	2000	8	1.57	0.47
Sta. 26b	8	1121	6	1.49	0.50
Sta. 45a	2	96	N/A	0.65	0.65
Sta. 45b	5	113	5	1.90	0.82
Paw Wah Pond					
Sta. 46A	1	1	N/A	0.00	N/A
Sta. 46B	2	4	N/A	0.81	0.81
Quanset Pond					
Sta. 3a	2	64	N/A	1.00	1.00
Sta. 3a	1	32	N/A	0.00	N/A
Round Cove					
RCV-1	5	397	4	1.21	0.52
RCV-2	8	227	7	1.52	0.51
RCV-3	5	296	4	1.21	0.52
RCV-4	10	551	6	1.64	0.49
Muddy Creek					
Upper	6	77	6	1.35	0.523
Lower	8	200	7	2.02	0.670
Bassing Harbor Sub-System					
Ryder's Cove	18	633	11	1.81	0.43
Bassing Is.	16	136	13	3.06	0.77
Crow's Pond Inner	29	287	18	3.76	0.77
Crow's Pond Outer	30	374	18	3.63	0.74
Frost Fish Creek	5	125	5	1.53	0.66
Pochet					
Sta. 39	9	480	9	2.98	0.94
Sta. 41A	10	448	10	2.53	0.76
Sta. 41B	13	752	12	3.03	0.68
Little Pleasant Bay					
Sta. 43A	5	296	5	1.78	0.77
Sta. 43B	12	1152	11	3.01	0.84
Sta. LPBa	7	480	7	2.01	0.71
Sta. LPBb	7	192	7	2.58	0.92
Sta. 36	12	992	8	2.04	0.57
Sta. 35a	9	1496	7	1.96	0.62
Sta. 35b	9	808	8	2.58	0.81
Sta. 37	10	944	9	2.26	0.68
Pleasant Bay					
Sta. 6a	10	3328	8	2.08	0.63
Sta. 6b	7	1952	6	1.73	0.62
Sta. 20	4	208	N/A	1.35	0.68
Sta. 32a	4	312	4	1.10	0.68
Sta. 32b	3	80	3	1.49	0.94
Sta. 16a	4	112	5	1.92	0.96
Sta. 16b	6	32	N/A	2.50	0.97
Sta. 14	6	116	7	2.00	0.77
Chatham Harbor					
Sta. 12a	8	188	9	2.50	0.83
Sta. 12b	6	260	5	1.70	0.66

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 associated watershed further strengthens the analysis. These data were all collected by the MEP Technical Team to support threshold development within the component sub-embayments comprising the Pleasant Bay System and are discussed in Section VII. Nitrogen threshold development builds on these data and links habitat quality to summer water column nitrogen levels obtained from long-term baseline water quality monitoring (Towns of Chatham and Orleans and Pleasant Bay Alliance Water Quality Monitoring Programs, and MEP Technical Team).

The Pleasant Bay System is comprised of a variety of basins showing a range of habitat health from “Healthy” (supportive of eelgrass, infaunal communities and with little oxygen stress) to “Degraded” (absence of eelgrass and benthic animals and periodic hypoxia/anoxia). There appears to be a clear relationship between habitat quality and the level of nitrogen enrichment. The less well flushed enclosed basins tend to be focal points for watershed nitrogen inputs and have relatively lower tidal flushing rates. In contrast, the larger basins and areas near the tidal inlet have a range in habitat quality, Moderately Impaired to Healthy, related to their flushing rate and depth.

The spatial distribution of habitat quality among the Pleasant Bay sub-embayments shows significant spatial variation, typical of other embayments within the MEP region. Although there are a large number of sub-embayments to the Pleasant Bay System, the habitat health or impairment associated with each of the key indicators (oxygen/chlorophyll a, eelgrass, infauna communities) tends to follow the 4 classifications listed below based upon the basin type:

(A) small enclosed basin (Meetinghouse Pond, Lonnies Pond, Areys Pond, Round Cove, Quanset Pond, Paw Wah Pond, Upper Muddy Creek),
(B) moderate sized tributary sub-embayment (The River, Muddy Creek),
(C) salt marsh dominated tidal sub-estuary (Pochet),
(D) large lagoonal estuarine basin (Little Pleasant Bay, Pleasant Bay, Chatham Harbor).

Dissolved Oxygen. The general pattern is for a high level of oxygen stress (frequent hypoxia or anoxia) in the bottom waters of the small enclosed basins (group A). These small enclosed basins tend to have higher nitrogen levels and high rates of sediment metabolism associated with their circulation and focus of watershed nitrogen loads. The Meetinghouse Pond basin and outlet channel, Lonnies Pond and its outlet channel, the Areys Pond outlet channel (Namequoit River), and Quanset Pond all showed significant levels of oxygen depletion, were routinely hypoxic and, except for Quanset Pond, D.O. levels were frequently <2 mg/L. In the same group of enclosed basins, Areys Pond, Paw Wah Pond and upper Muddy Creek showed frequent anoxia (absence of oxygen). Among the enclosed basins only Round Cove showed mild hypoxia with levels above 4 mg/L and generally above 5 mg/L during the full deployment.

In contrast, the salt marsh dominated tidal creek of Pochet showed frequent oxygen depletions to 3-4 mg/L, but was generally above 4 mg/L. The oxygen conditions in Pochet creek are consistent with the biogeochemistry of salt marshes. Salt marsh creeks (that do not empty at low tide) frequently become hypoxic in summer as a result of the high organic matter loading associated with marshes. Even pristine salt marshes can exhibit this behavior.

The large main basins of the lagoonal estuarine component showed oxygen conditions consistent with rates of sediment metabolism associated with deep waters and a depositional environment (Little Pleasant Bay, Pleasant Bay) or high tidal velocities (Chatham Harbor and eastern channel form Chatham Harbor to Little Pleasant Bay, channel between Strong Island and Bassing Harbor). Upper Pleasant Bay at Namequoit Point showed oxygen levels frequently declining to 4-5 mg/L and the western most basin of Pleasant Bay (between Round Cove and Muddy Creek) had a single event to 2-4 mg/L, although it was generally >5 mg/L. Approaching Chatham Harbor, oxygen conditions improve (see Strong Island results), with oxygen conditions generally >6 mg/L with short declines to 5 mg/L and associated with the outflow of lower oxygen waters from Pleasant Bay.

Overall, the oxygen and chlorophyll records show a consistent pattern of higher organic matter production (chlorophyll) in embayments with greater oxygen depletions. The pattern is one of sub-embayments that are enclosed (group A) having habitat impairment by frequent low oxygen events, with the larger lagoonal basins (group B) showing less frequent and extreme levels of oxygen depletion and moderate impairment, grading to good oxygen conditions near (and presumably in) Chatham Harbor. This pattern follows the nitrogen gradients in the System (Chapter VI), the eelgrass distribution (below) and the infaunal habitat quality.

Eelgrass. At present, eelgrass is present within large portions of the Pleasant Bay System, indicative of a system with areas of high habitat quality. These eelgrass beds are generally restricted to the larger lagoonal basins, such as Little Pleasant Bay, Pleasant Bay and Chatham Harbor. There are also smaller eelgrass areas in Pochet and fringing shallow areas in The River and Meetinghouse Pond. The only tributary embayment to Pleasant Bay with significant eelgrass habitat is Bassing Harbor (see below). The basins presently supporting eelgrass habitat also supported habitat in the 1951 historical analysis. However, it is clear from the 1951, 1995 and 2001 temporal sequence that the eelgrass areas in each basin, except Chatham Harbor, are declining in coverage. In The River and Pochet the eelgrass areas were always patchy and present in the shallows. In the 2001 survey this pattern persists, but the beds appear to be on the decline. The overall pattern of eelgrass distribution and temporal decline in coverage is fully consistent with the spatial pattern of nitrogen enrichment (Chapter VI) and oxygen and chlorophyll levels in the various basins (see above). The present rate of eelgrass habitat throughout the Pleasant Bay System is ~11 acres per year.

Virtually all of the small enclosed basins (group A, above) did not appear to support eelgrass historically and do not support it today, with the exception of the small patch in the shallows of Meetinghouse Pond and in lower Muddy Creek (see below). This general pattern is consistent with the deeper waters of these basins and their location and structure which tends to result in nitrogen enrichment.

Based upon the 1951 coverage it appears that nitrogen management to restore eelgrass habitat has the potential to recover a significant resource to this System and to the lower Cape (Table VII-5), on the order of 500-600 acres system-wide. However, for the reasons discussed above creation of eelgrass habitat within the enclosed basins is unlikely and not supported by the historical analysis.

Infaunal Animal Communities. As for the oxygen and chlorophyll indicators and the distribution of sediment metabolism, the enclosed basins (group A, above) are generally significantly to severely impaired relative to benthic infaunal habitat quality. Among the enclosed basins, all were at least significantly impaired. Paw Wah Pond is virtually devoid of benthic animals (only 1-4 individuals per sample) as would be expected from its high level of oxygen stress. Similarly, Areys Pond, Quanset Pond, Upper Muddy Creek supported significantly depleted benthic animal populations, consistent with their nitrogen related oxygen stress. The other enclosed basins were able to support benthic infauna, but the community was dominated by opportunistic species (*Capitella*, *Streblospio*) indicative of very high organic matter loading (Lonnies Pond, Meetinghouse Pond outlet channel) or by intermediate stress indicators (*Gemma*, Amphipods). The dominance of these intermediate indicators in The River, Round Cove, Meetinghouse Pond suggests that these systems, which also showed only moderate oxygen stress, are only moderately beyond their nitrogen loading limits (Table VII-8).

The larger lagoonal basins of Little Pleasant Bay generally supported infaunal communities indicative of a moderate level of stress from organic matter loading and oxygen depletion. However, the pattern was for a decrease in habitat quality moving from the marginal to depths. This pattern is typical of a system near, but beyond its nitrogen loading limit, where organic matter deposition in the deep basin areas is the proximate cause of the impairment of benthic habitat quality. Chatham Harbor habitat supported only moderate numbers of individuals and species, but this appeared to result from the dynamic nature of the bottom sediments (unstable bottom), due to the high tidal velocities, rather than nutrient related impairment.

The results of the evaluations of the 3 key habitat indicators (infaunal animals, eelgrass. dissolved oxygen/chlorophyll a) coupled with macroalgal survey data were used to assess the overall habitat quality of each component sub-embayment to the Pleasant Bay System (Table VIII-1). The results of the habitat assessment show consistent assessments between indicators and follow the pattern of nitrogen enrichment (see Section VIII.3, below). All of these data were integrated in the development of the nitrogen thresholds for the restoration of eelgrass and infaunal habitats throughout the Pleasant Bay System (Section VIII.2).

Table VIII-1. Summary of Nutrient Related Habitat Health within the Pleasant Bay Estuarine System Cape Cod, MA., based upon assessment data presented in Chapter VII. D.O. and Chl a are dissolved oxygen and chlorophyll a from the mooring data (VII.2).

Sub-Embayment	Nutrient related Health Indicator					
	D.O.	**Chl a**	**Macro-algae**	**Eelgrass**	**Infaunal Animals**	**Overall**
Meetinghouse Pond & Outlet	SI/SD[1,2]	SI/MI	MI[3]	--	SI	**SI**
Lonnies Pond	SI[2]	MI	MI	--	SI	**SI/MI**
Areys Pond & Outlet	SD[1]	SI	SI/SD[13]	--	SD[9]	**SD/SI**
The River	MI/SI[3]	MI	MI/SI	MI	SI	**MI**
Paw Wah Pond	SD[1]	SI	SI	--	SD[10]	**SD**
Quanset Pond	SI	SI	--[c]	--	SD[9]	**SI**
Round Cove	MI[4]	SI	--[c]	--	SI	**SI/MI**
Muddy Creek Upper	SD[1]	SI/SD	--[c]	--	SD[9]	**SD**
Muddy Creek Lower	SI/SD[1]	SI/MI	--[c]	SI	SI	**SI**
Bassing Hbr: Ryders Cove		SI	MI	MI[7]	MI	**MI**
Bassing Hbr: Crows Pond		MI	MI	MI[7]	H/MI	**MI**
Bassing Hbr: Lower Basin		MI/H	--[c]	H/MI	MI	**H/MI**
Bassing Hbr: Frost Fish Crk		SI	SI	--	SI	**SI**
Pochet	H/MI[3a]	H	--[c]	--	H/MI	**H**
Little Pleasant Bay	MI[3]	H	--[c]	MI[7]	MI	**MI**
Pleasant Bay	MI/SI	MI	--[c]	MI/SI	MI-SI	**MI**
Chatham Harbor	H[b]	H[b]	--[c]	H	H	**H**

1 – frequent oxygen depletions to 0-2 mg/L (i.e periodic anoxia)
2 – periodic oxygen depletions to <2 mg/L and frequently <4 mg/L
3 – infrequent oxygen depletions to 3-4 mg/L, periodic 4-5 mg/L., generally >5 mg/L.
4– generally >5 mg/L..
5 – high macroalgal accumulations during summer
6 – moderate macroalgal accumulations or patches on bottom.
7 – eelgrass present but beds appear to be thining or declining in areal coverage
8 – modest numbers of individuals dominated by stress indicator species.
9 – depleted infaunal community (<100 individuals/grab).
10– absence of infaunal community (<15 individuals/grab).
11 – no evidence this basin is supportive of eelgrass.
12 – infaunal community dominated by opportunistic species.
13 – dense macroalgal accumulation in the Namequoit River
a – periodic oxygen depletion is typical of salt marsh creeks.
b – based upon Strong Island Channel data
c – no accumulations observed during MEP field surveys
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment;
SD = Severe Degradation; -- = not applicable to this estuarine reach

VIII-2. THRESHOLD NITROGEN CONCENTRATIONS

The approach for determining nitrogen loading rates, which will maintain acceptable habitat quality throughout an embayment system, is to first identify a sentinel location within the embayment and second to determine the nitrogen concentration within the water column which will restore that location to the desired habitat quality (threshold nitrogen level). The sentinel location is selected such that the restoration of that one site will necessarily bring the other regions of the system to acceptable habitat quality levels. Once the sentinel site and its target nitrogen level are determined, the Linked Watershed-Embayment Model is used to adjust nitrogen loads sequentially until the targeted nitrogen concentration is achieved. For the Pleasant Bay System, the restoration target should reflect both recent pre-degradation habitat quality and be reasonably achievable.

The threshold nitrogen level for an embayment represents the tidally averaged 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.

The threshold nitrogen level for the Pleasant Bay System was developed to restore or maintain SA waters or high habitat quality. High habitat quality was defined as supportive of eelgrass and infaunal communities. Dissolved oxygen and chlorophyll-*a* were considered in the assessment. While there is a single sentinel station, given the number of semi-enclosed sub-embayments, several secondary “check” stations were also selected.

The approach developed by the MEP has been to select a sentinel sub-embayment within an embayment system. First, a sentinel sub-embayment is selected based upon its location within the system. The sentinel sub-embayment 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 tidally averaged target nitrogen level at the sentinel location. 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.

Based upon the significant historical and present eelgrass habitat within the Pleasant Bay System, 2400 acres and 1800 acres respectively (Chapter VII), eelgrass was selected as the target for the development of the site-specific nitrogen threshold. In addition, a secondary threshold supportive of benthic animal communities (infauna) was developed in areas that do not have documented eelgrass habitat. The eelgrass threshold applies to the sentinel station (and secondary eelgrass station in Ryders Cove) and the secondary "check" thresholds for infauna habitat is for the smaller sub-basins not naturally supportive of eelgrass based on historical records.

The MEP's previous analysis of Bassing Harbor found very high levels of dissolved organic nitrogen within the embayment's waters (based upon data from the Chatham and Pleasant Bay Alliance Water Quality Monitoring Programs). While some portion of the dissolved organic nitrogen is actively cycling, the vast majority is refractory (non-biologically active) within the timeframe of the flushing of the Pleasant Bay System. The result is that the dissolved organic nitrogen presents a large non-active pool generally separate from the nitrogen fractions active in eutrophication (i.e. ammonium and nitrate+nitrite, particulate organic nitrogen). The biologically active nitrogen pools are represented by the species directly available to phytoplankton and algae (plant available nitrogen), ammonium and nitrate+nitrite, and the particulate organic nitrogen comprised primarily of phytoplankton (live and dead). Together this nitrogen group is termed bioactive nitrogen. Given the large dissolved organic nitrogen pool within Pleasant Bay the MEP Technical Team adopted the same approach used previously for the TMDL analysis of Bassing Harbor. In this previous analysis, the threshold was developed based upon the bioactive nitrogen pool, which appears to be relatively consistent between embayments both within and outside of Pleasant Bay, and then the bioactive threshold was transformed to the total nitrogen level by adding back in the dissolved organic nitrogen concentration derived for the site from direct measurements. In meeting the threshold value and achieving restoration, the bioactive nitrogen threshold has slightly less uncertainty than the total nitrogen threshold given the biogeochemistry of this system. Therefore, while both values form the basis for guiding nitrogen reductions to achieve ecological restoration, the total nitrogen value should only be evaluated in light of the bioactive nitrogen threshold. Critical nitrogen threshold levels were developed to support both healthy eelgrass and healthy infaunal habitat as described below.

While there is significant variation in the dissolved organic nitrogen levels, hence total nitrogen levels supportive of healthy eelgrass habitat, the level of bioactive nitrogen supportive of this habitat appears to be relatively constant. Therefore, the MEP Technical Team set a single eelgrass threshold based upon stable eelgrass beds, tidally averaged bioactive N levels and the stability of eelgrass as depicted in coverages from 1951-2001. The eelgrass threshold was set at 0.16 mg bioactive N/L based upon the Chatham (Dec 2003 MEP report) analysis for Bassing Harbor. That report for Bassing Harbor indicated a bioactive level for high quality eelgrass habitat of 0.160 mg bioactive N/L based upon Healthy eelgrass community in both Bassing Harbor at 0.135 bioactive N/L and in Stage Harbor at 0.160 bioactive N/L (Oyster River Mouth). The higher value was used as the eelgrass habitat in Bassing Harbor was below its nitrogen loading limit at that time.

Although the Bassing Harbor System (comprised of Ryder Cove, Crows Pond, Frost Fish Creek and Bassing Harbor) has two inland-most sub-embayments, Ryder Cove and Crows Pond, only Ryder Cove was selected as the sentinel system for this sub-embayment. 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 based upon the 1951-2001 surveys. 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 in the Town of Chatham, 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}). Note that the mouth of the Oyster River exhibits a documented stable healthy eelgrass habitat (MEP 2003). 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 was 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), is 0.363 mg N L^{-1} based upon the long-term monitoring data, 2000-05. A site specific threshold level was then developed using the conservative DIN+PON level from the Stage Harbor System plus the new analysis of the Pleasant Bay System (see below) of 0.160 mg N L^{-1}. This yields an equivalent Total Nitrogen Threshold for the Bassing Harbor Sub-embayment (average upper and lower Ryders Cove stations) of 0.523 mg N L^{-1}. This value is very close to the previous Bassing Harbor specific threshold range of 0.527-0.552 mg N L^{-1}. The slight shift in threshold level results from the greatly expanded water quality database for the present versus previous analysis. The nitrogen boundary condition (the concentration of nitrogen in inflowing tidal waters from Pleasant Bay) for the Bassing Harbor System is 0.45 mg N L^{-1}.

The above analysis was expanded into a full Pleasant Bay analysis, which was based upon examining eelgrass beds which appear in all three surveys between 1951-2001 and using MEP field observations made in 2003. This detailed analysis strongly supported the use of a 0.16 mg Bioactive N/L threshold for all of Pleasant Bay. These additional lines of evidence (PBA#, WMO# refer to water quality sampling stations Chapter VI) are as follows:

a) The upper most reach of the contiguous eelgrass beds in Little Pleasant Bay (PBA-12) have been extant from 1951-2001. The mapping indicates a large contiguous areal coverage within Little Pleasant Bay with PBA-12 approximately at the uppermost point. Above these beds moving into the mouth of The River (PBA-13) and the lowermost basin of Pochet (WMO-03) eelgrass coverage appears to have declined since 1951, although eelgrass is still present. This loss of beds indicates that the habitat quality has become impaired, but since eelgrass remains, the impairment is judged to be "moderate". Under existing conditions the tidally averaged bioactive nitrogen levels at each of these sites are 0.178 for upper Little Pleasant Bay (PBA-12), 0.195 for the mouth of The River (PBA-13), and 0.183 for the lowermost basin in Pochet (WOM-03). It appears from the bathymetry that the eelgrass in Pochet and the patches in The River are restricted to the shallows, generally <1 meters depth. This is consistent with the persistence of these beds at a higher nitrogen level since the effect of eutrophication on eelgrass (through shading effects) is directly dependent on depth (i.e. deeper beds are lost first). Based upon these data the conditions of the eelgrass beds in upper Little Pleasant Bay were examined. Visual surveys by MEP staff indicated that the eelgrass beds in deeper waters of upper Little Pleasant Bay indicated the presence of filamentous green algae in moderate amounts. In addition some of the upper beds had coverages of 30%-50%, suggesting a decline in habitat quality, although healthy beds were also observed. Equally important was an absence of some beds in the deeper waters along the western shore of Little Pleasant Bay (Paw Wah Pond shoreline) which showed eelgrass in the 1951-2001 surveys. This suggests that this basin has recently exceeded its nitrogen loading threshold (i.e. the 0.178 mg bioactive N/L is too high). However, the data from the lower reach of Upper Pleasant indicates a healthy eelgrass habitat at tidally averaged bioactive N levels of 0.161 mgN/L. In addition, the decline in eelgrass coverage at the mouth of The River and in Pochet is consistent with a recent initial (gradual) decline in the deeper areas of Upper Little Pleasant Bay. As a result the eelgrass threshold for the Pleasant Bay system appears to be between 0.160 and 0.178 mg Bioactive N/L.

b) Eelgrass beds are no longer present in the Pleasant Bay basin bounded by Round Cove and Muddy Creek on the West and Strong Island on the east. The major proximate cause appears to be the much greater depth of this basin than the depth of Little Pleasant Bay and the basin between Strong Island and the western barrier beach boundary. However, even when comparing similar depths from these 3 basins, it is clear that the western Pleasant Bay basin does not support eelgrass habitat, while the others do. The western Pleasant Bay basin's tidally averaged bioactive N levels are between 0.168 (PBA-07) and 0.192 (PBA-06. Furthermore the uppermost station in this basin, off Simpson Island has a small remaining eelgrass bed near water quality station, PBA-08, which had a tidally averaged bioactive N level of 0.149 mg N/L. and a measured ebb tide average of 0.162 mg N/L. Supportive of an eelgrass threshold of 0.160 mg N/L tidally averaged bioactive N level.

c) Crows Pond in the Bassing Harbor sub-system currently supports a high level of habitat quality, with eelgrass beds surrounding the central deep basin and sparse coverage throughout. Note that the deep basin in Crows Pond is similar to the deep basin in Pleasant Bay and the terminal kettle ponds in the upper reaches of the Pleasant Bay System. Crows Pond supports healthy habitat in its shallower waters (similar depths to Little Pleasant Bay) at a tidally averaged bioactive N level of 0.162 mg N/L and measured ebb tidal average of 0.208 mg N/L. 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}). The apparent slight decline in habitat quality stems from the observed very sparse coverage in deep central basin (Chatham mapping 2000), although the MASSDEP mapping programs 1951 and 2001 analysis show similar overall coverages. At present it appears that Crows Pond approaching and possibly at its threshold nitrogen level. However, the Crows Pond data supports an eelgrass threshold of 0.160 mg N L^{-1} tidally averaged bioactive N level.

The sentinel station for the Pleasant Bay System based on a nitrogen threshold targeting restoration of eelgrass was placed within the uppermost reach of Little Pleasant Bay (PBA-12) near the inlets to The River and Pochet. The threshold bioactive nitrogen level at this site (as for Ryders Cove) is 0.160 mg bioactive N L^{-1}. Based upon the background dissolved organic nitrogen average of upper Little Pleasant Bay and Lower Pochet 0.563 mg N L^{-1} and the bioactive threshold value, the total nitrogen level at the sentinel station (PBA-12) is 0.723 mg N L^{-1}. The restoration goal is to improve the eelgrass habitat throughout Little Pleasant Bay and the historic distribution in Pleasant Bay, which will see lower nitrogen levels when the threshold is reached. In addition, the fringing eelgrass beds within The River and within Pochet should also be restored, as they are in shallower water than the nearby sentinel site and therefore are able to tolerate slightly higher watercolumn nitrogen levels. Moreover, the same threshold bioactive nitrogen level should be met for the previous sentinel station (upper Ryders Cove) in Bassing Harbor System when levels are achieved at the sentinel station in upper Little Pleasant Bay. However, given the partial independence of the Bassing Harbor sub-embayment system relative to the greater Pleasant Bay System (i.e. its own local watershed nitrogen load plays a critical role in its health), the upper Ryders Cove sentinel station should be maintained as the guide for this sub-embayment to Pleasant Bay. It should also be noted that while the bioactive threshold is the same at both sites, the Total Nitrogen level in Ryders Cove is 0.523 mg N L^{-1}, due to the lower dissolved organic nitrogen levels in the lower Bay.

While eelgrass restoration is primary nitrogen management goal within the Pleasant Bay System, there are small basins which do not appear to have historically (1951) supported eelgrass habitat. For these sub-embayments, restoration and maintenance of healthy animal communities is the management goal. It should be noted that restoration of eelgrass is not the only criteria for restoration of habitat health throughout the Pleasant Bay System. Based upon the 1951 eelgrass analysis there are eight (8) sub-embayments to Pleasant Bay that are not likely to support eelgrass habitat for structural reasons. These are all drowned kettle ponds or coves that have been enclosed by a barrier beach. The typical structure of each of these sub-embayments is that they have a relatively narrow tidal channel from the Bay into a relative deep basin. While these systems may not be supportive of eelgrass habitat, they are generally capable of supporting healthy benthic animal habitat. Infaunal animals are sensitive to the organic matter loading and resulting periodic oxygen depletions associated with nitrogen overloading. Since these conditions typically occur at higher nitrogen loads than does the shading of the bottom by increased phytoplankton production (principal cause of eelgrass loss), the nitrogen threshold level for healthy benthic animal habitat is higher than for healthy eelgrass habitat. This has been found to be the case throughout the MEP study area.

The infaunal habitat threshold was derived in a similar manner to the site-specific eelgrass threshold for Pleasant Bay as described above. The threshold depends heavily upon the present distribution of infaunal communities relative to watercolumn nitrogen levels and measured oxygen depletions. The presence of some eelgrass is also noted. At present, moderately impaired infaunal communities are present in Ryders Cove (PBA-03) at tidally averaged bioactive nitrogen levels of 0.244 mg N L^{-1}. Similarly, there are moderately impaired infaunal communities, designated primarily by the dominance of amphipods (amphipod mats) in most of the 8 sub-embayments of focus. These communities are present adjacent the inlet to Lonnies Pond (in The River Upper) at bioactive nitrogen levels of 0.217 mg N L^{-1}, in the Namequoit River at 0.216-0.239 mg N L^{-1} and in Round Cove at mg N L^{-1} at 0.239 mg N L^{-1}. These communities can be found at even higher levels in the fringing shallow areas of deep basins like Areys Pond (0.299 mg N L^{-1}) and Meetinghouse Pond (0.411 mg N L^{-1}). Very shallow waters tend to minimize oxygen depletion that severely stress infaunal communities in deeper basins. Paw Wah Pond is periodically hypoxic and as a result does not presently support infaunal habitat. These data are at higher bioactive nitrogen levels than the healthy infaunal habitat in the lower Pochet Basin (WMO-03) at 0.178 mg N L^{-1}. It appears that the infaunal threshold lies between 0.18 and 0.22 mg N L^{-1} tidally averaged bioactive nitrogen. Note that within the shallow margins of the river eelgrass is present at 0.191 mg N l^{-1}, suggesting that healthy infaunal habitat is likely at this level.

Based upon the animal community and nitrogen analysis mentioned above the restoration goal for these 8 systems is to restore a healthy habitat to the full basin in the shallower or more open waters and to the margins in the deep drowned kettles that periodically stratify. This would argue for a bioactive nitrogen threshold of 0.21 mg N L-1, lower than the lowest station with significant amphipod presence. Note that achieving the infaunal threshold in all of the sub-embayments and the eelgrass threshold in Upper Pleasant Bay (and Ryders Cove) will generate high quality habitats throughout the Pleasant Bay system. To achieve these goals, infaunal check stations were placed (where appropriate) in the inlet to kettle ponds (for deeper ponds) or in the center of the ponds for shallower ponds.

At present all eight sub-embayments are above the level required for healthy infaunal habitat. The tidally averaged bioactive nitrogen levels and associated total nitrogen levels and threshold levels are shown in Table VIII-2, along with current water quality station identification numbers.

Table VIII-2. Bioactive nitrogen thresholds and associated Total Nitrogen (TN) levels in sub-embayments to Pleasant Bay targeting restoration of benthic animal habitat under one possible restoration scenario. Note that the range in TN levels stems from the varying levels of dissolved organic nitrogen within the Pleasant Bay System. The site-specific DON level was used to adjust the bioactive nitrogen threshold to total nitrogen.

Location	WQ Station ID	Bioactive N Threshold mg/L	DON mg/L	TN Threshold mgN/L
Meetinghouse @Rattles Dock	WMO-10	0.210	0.700	**0.910**
Lonnies Pond	PBA-15	0.210	0.496	**0.706**
Namequoit River Upper (Areys Pond)	WMO-6	0.210	0.529	**0.739**
Pochet - Upper off Town Landing	WMO-05	0.210	0.555	**0.765**
Paw Wah Pond	PBA-11	0.210	0.439	**0.649**
Little Quanset Pond	WMO-12	0.210	0.394	**0.604**
Round Cove	PBA-09	0.210	0.461	**0.671**
Muddy Creek – Lower	PBA-05	0.210	0.331	**0.541**

The secondary "infaunal" thresholds for each of these sub-embayments must be reached in order to restore their habitat quality. Depending upon the specific strategy for lowering watershed nitrogen loading to the entire Pleasant Bay System to achieve the threshold at the sentinel station in upper Pleasant Bay, it may be possible that a specific sub-embayment may or may not achieve its secondary threshold, even though the eelgrass threshold at the sentinel station for the System is reached. This results from the size of the Pleasant Bay System and the relatively isolated nature of some of the small sub-embayments. Even though these sub-embayments receive water from the main System, their localized watershed load predominates in some cases. Therefore, restoration success will be gauged by reaching the target at the sentinel station and at the secondary stations for eelgrass (Ryders Cove) and infauna. Overall, there are 3 primary (PBA-12, PBA-03 and CM-13) and 8 secondary target stations within this System, the largest embayment on Cape Cod.

VIII.3 DEVELOPMENT OF TARGET NITROGEN LOADS

It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the communities that impact Pleasant Bay waters. The purpose of the load reduction scenario presented is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

The develop the scenario presented, nitrogen thresholds determined in the previous section were used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Pleasant Bay system. Tidally averaged total nitrogen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold levels at the sentinel stations selected for eelgrass and infaunal habitat restoration within the Pleasant Bay system.

Development of nitrogen load reductions needed to meet the threshold concentration of 0.16 mg/l bioactive nitrogen (DIN+PON) in Ryders Cove (the average of PBA-03 and CM-13) and Upper Little Pleasant Bay (PBA-13) focused primarily on septic load removal within the River and Bassing Harbor systems. Due to the relatively large size of the Pleasant Bay system, achieving the primary threshold concentration for the restoration of eelgrass at the sentinel stations alone did not achieve the secondary threshold at the series of small embayments surrounding Pleasant and Little Pleasant Bays. The secondary threshold concentration of 0.21 mg/l bioactive nitrogen (DIN+PON) in Meetinghouse Pond (Outer), Lonnies Pond, Upper Namequoit River, Upper Pochet, Paw Wah Pond, Little Quanset Pound, Round Cove and Lower Muddy Creek required site-specific removal of septic nitrogen from the watersheds directly impacting these sub-embayments. Table VIII-3 shows the percent of septic load removed from the various watersheds to achieve both the primary and secondary threshold concentrations of bioactive nitrogen at the sentinel stations.

Tables VIII-4 and VIII-5 provide additional loading information associated with the threshold scenario developed for this Report. Table VIII-4 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-3. For Example, removal of 100% of the septic load from the Meetinghouse Pond sub-watershed results in an 83% reduction in total nitrogen load to that sub-embayment. Table VIII-4 shows the breakdown of threshold sub-embayment and surface water loads used for total nitrogen modeling. In Table VIII-5, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent ‘worst-case’ summertime conditions. Table VIII-5 illustrates the significant role of atmospheric deposition relative to the total nitrogen load to the Pleasant Bay system. Unlike most estuarine systems in southeastern Massachusetts, the water surface area of the estuarine system is large relative to the overall watershed area. For the case of Pleasant Bay, the atmospheric load actually is larger than the watershed nitrogen load under the selected conditions necessary to meet the various thresholds. In addition, benthic flux within the main body of Pleasant Bay is larger than either the threshold watershed load or the atmospheric deposition. Again, the significant magnitude of the load associated with benthic regeneration within Pleasant Bay is caused by the substantial surface area of the water body.

Model results for the septic load removal scenario described in Table VIII-3 achieve the target bioactive nitrogen concentrations at the primary and secondary sentinel stations, as shown in Table VIII-6 and Figure VIII-1. To achieve the threshold nitrogen concentrations at the sentinel stations, a reduction in bioactive nitrogen concentration of between 15% and 40% is required in the upper regions of the Pleasant Bay system, with bioactive nitrogen reduction levels decreasing toward Chatham Harbor and New Inlet. The maximum reduction in bioactive nitrogen levels occurs in Upper Muddy Creek, followed by Meetinghouse Pond, Lower Frost Fish Creek, Ryders Cove, and Upper Pochet, respectively.

The results from the 2 tributary sub-embayments, Muddy Creek and Bassing Harbor requires a higher proportional amount of nitrogen removal to achieve the threshold nitrogen level than was previously determined by the MEP analysis in 2003. Note that for watershed planning it is the proportion of septic systems being removed that is the critical consideration. The reason for the increased wastewater nitrogen management within the sub-watersheds associated with these sub-embayments stems from the significantly larger data base on water quality and habitat health and the ability (for the first time) to accurately determine the water quality of the Pleasant Bay waters which flow into these sub-embayments.

In the earlier analysis (2003), it was noted that a refinement would be needed, since restoration of these 2 sub-embayments is effected by the nitrogen levels in Pleasant Bay waters as the boundary condition. The integration of the 2 previous models (Muddy Creek and Bassing Harbor) into the Pleasant Bay System-wide model decreased uncertainty of model parameters and allowed for the necessary evaluation of these sub-embayments in the context of the Pleasant Bay System as a whole. However, this integration required that the previous models for Muddy Creek and Bassing Harbor be recalibrated and revalidated as part of being joined to the large system-wide model.

The specific reasons for the greater level of wastewater management (i.e. proportional reduction in septic system loadings) for these 2 sub-systems to Pleasant Bay are: (1) the near doubling of the water quality database yielding a better assessment of the nitrogen levels relative to the habitat indicators and (2) the sub-system habitats relative to nitrogen levels could also be compared to other similar areas within Pleasant Bay. Both of these factors resulted in the selection of a nitrogen threshold for these sub-systems which was at the low end of the stated acceptable range presented in the 2003 analysis (0.523 mgTN-N/L versus 0.527-0.553 mgTN-N/L). It is the shift in threshold that required that more title 5 septic system be taken off-line in the threshold loading analysis (Table VIII-3). In addition, in the earlier analysis it was not possible to develop an accurate boundary condition for Muddy Creek, as the system-wide model for Pleasant Bay was not available. It was clear in the earlier effort that nitrogen management in the Muddy Creek sub-embayment was linked to the adjacent Pleasant Bay waters. It should be noted that due to the proportional nature of the shift in sub-watershed nitrogen loads as determined in the 2003 report the shift does not effect the level of wastewater management required to meet the threshold. If the threshold and boundary condition parameters had not been refined for the present effort, the number of homes requiring wastewater nitrogen management (for example, number of residences to be hooked to a WWTF), would not have changed. Taking that into consideration, it is useful to indicate the reasons for the watershed loading shift for these 2 sub-embayments. First, the Town of Chatham requested that the MEP move forward with 3 quarters of water-use data, as that was all that was available. The MassDEP decided that the MEP should move forward, partially because these 2 sub-embayments would be refined in the Pleasant Bay analysis. This led to an overestimate of the extent of the wastewater load. While the water use data was inflated (as subsequent analysis of years of data collected by the CCC, the Chatham TAC and CAC has demonstrated), it also had secondary effects on estimates of population that were based on water use. A second issue resulted from use of the wastewater effluent and consumptive use terms that inflated the per capita load contribution. These issues were discovered very early on and resolved. Most importantly, wastewater planning is not effected by these input data issues. In sensitivity analyses conducted by the Technical Team in 2004, changes in wastewater coefficients of 33% resulted in only a 1%-2% change in the proportion of dwellings that needed to be hooked to a WWTF to accomplish habitat restoration. This results from the robustness of the models. However, it should be noted that it is not possible to set a threshold under one set of conditions and then compare the load reductions required using another set of conditions. It only works when a consistent set of input data are used throughout the analysis.

Table VIII-3. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading under one possible restoration scenario of the Pleasant Bay system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.

sub-embayment	present septic load (kg/day)	threshold septic load (kg/day)	threshold septic load % change
Meetinghouse Pond	5.140	0.000	-100.0%
The River – upper	2.071	1.036	-50.0%
The River – lower	2.871	1.436	-50.0%
Lonnies Pond	1.630	0.815	-50.0%
Areys Pond	0.778	0.389	-50.0%
Namequoit River	2.011	1.005	-50.0%
Paw Wah Pond	1.510	0.377	-75.0%
Pochet Neck	6.614	2.315	-65.0%
Little Pleasant Bay	4.512	2.256	-50.0%
Quanset Pond	1.403	0.701	-50.0%
Tar Kiln Stream	1.797	0.899	-50.0%
Round Cove	3.162	1.897	-40.0%
The Horseshoe	0.474	0.474	0.0%
Muddy Creek - upper	7.156	1.789	-75.0%
Muddy Creek - lower	6.340	0.000	-100.0%
Pleasant Bay	13.077	6.538	-50.0%
Pleasant Bay/Chatham Harbor Channel	-	-	-
Bassing Harbor - Ryder Cove	7.137	1.784	-75.0%
Bassing Harbor - Frost Fish Creek	2.200	0.000	-100.0%
Bassing Harbor - Crows Pond	3.326	3.326	0.0%
Bassing Harbor	1.400	1.400	0.0%
Chatham Harbor	14.195	14.195	0.0%
TOTAL - Pleasant Bay System	88.803	42.632	-52.0%

Table VIII-4. Comparison of sub-embayment ***total watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading under one possible restoration scenario of the Pleasant Bay system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Meetinghouse Pond	6.197	1.058	-82.9%
The River – upper	2.773	1.737	-37.4%
The River – lower	3.879	2.444	-37.0%
Lonnies Pond	2.441	1.626	-33.4%
Areys Pond	1.304	0.915	-29.8%
Namequoit River	2.737	1.732	-36.7%
Paw Wah Pond	1.860	0.728	-60.9%
Pochet Neck	8.422	4.123	-51.0%
Little Pleasant Bay	7.496	5.240	-30.1%
Quanset Pond	1.781	1.079	-39.4%
Tar Kiln Stream	6.123	5.225	-14.7%
Round Cove	4.225	2.960	-29.9%
The Horseshoe	0.638	0.638	0.0%
Muddy Creek - upper	9.981	4.614	-53.8%
Muddy Creek - lower	8.477	2.137	-74.8%
Pleasant Bay	23.159	16.621	-28.2%
Pleasant Bay/Chatham Harbor Channel	-	-	-
Bassing Harbor - Ryder Cove	9.819	4.466	-54.5%
Bassing Harbor - Frost Fish Creek	2.904	0.704	-75.8%
Bassing Harbor - Crows Pond	4.219	4.219	0.0%
Bassing Harbor	1.668	1.668	0.0%
Chatham Harbor	17.099	17.099	0.0%
TOTAL - Pleasant Bay System	127.203	81.032	-36.3%

Table VIII-5. Threshold sub-embayment loads used for bioactive nitrogen (DIN+PON) modeling of the Pleasant Bay system under one possible restoration scenario, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Meetinghouse Pond	1.058	0.584	7.857
The River – upper	1.737	0.288	4.102
The River – lower	2.444	2.241	8.517
Lonnies Pond	1.626	0.225	1.304
Areys Pond	0.915	0.181	4.929
Namequoit River	1.732	0.523	12.232
Paw Wah Pond	0.728	0.082	2.665
Pochet Neck	4.123	1.767	-0.622
Little Pleasant Bay	5.240	24.023	35.222
Quanset Pond	1.079	0.170	4.787
Tar Kiln Stream	5.225	0.066	-
Round Cove	2.960	0.170	6.739
The Horseshoe	0.638	0.063	-
Muddy Creek - upper	4.614	0.162	2.700
Muddy Creek - lower	2.137	0.205	-0.710
Pleasant Bay	16.621	19.153	134.187
Pleasant Bay/Chatham Harbor Channel	-	17.786	-38.017
Bassing Harbor - Ryder Cove	4.466	1.296	6.705
Bassing Harbor - Frost Fish Creek	0.704	0.096	-0.087
Bassing Harbor - Crows Pond	4.219	1.389	0.612
Bassing Harbor	1.668	1.071	-4.460
Chatham Harbor	17.099	14.153	-38.398
TOTAL - Pleasant Bay System	**81.032**	**85.693**	**150.264**

Table VIII-6. Comparison of model average bioactive N (DIN+PON) concentrations from present loading and the threshold scenario, with percent change, under one possible restoration scenario for the Pleasant Bay system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). The threshold stations for eelgrass restoration are shown in bold print (0.16 mg/L at PBA-12 and the average of PBA-03 and CM-13) and for benthic infauna restoration are shown in italics (0.21 mg/L at WMO-10, PBA-15, WMO-6, WMO-5, PBA-11, WMO-12, PBA-09 and PBA-05).

Sub-Embayment	monitoring station	present (mg/L)	Threshold (mg/L)	% change
Meetinghouse Pond	PBA-16	0.380	0.262	-31.1%
Meetinghouse Pond (Outer)	*WMO-10*	*0.261*	*0.207*	*-20.7%*
The River - upper	WMO-09	0.239	0.196	-18.0%
The River – mid	WMO-08	0.211	0.182	-14.0%
Lonnies Pond (Kescayo Ganset Pond)	*PBA-15*	*0.250*	*0.208*	*-16.7%*
Areys Pond	PBA-14	0.297	0.253	-14.9%
Namequoit River - upper	*WMO-6*	*0.239*	*0.206*	*-13.6%*
Namequoit River - lower	WMO-7	0.216	0.188	-13.0%
The River - lower	PBA-13	0.195	0.172	-11.9%
Pochet – upper	*WMO-05*	*0.269*	*0.211*	*-21.3%*
Pochet - lower	WMO-04	0.209	0.179	-14.1%
Pochet – mouth	WMO-03	0.183	0.164	-10.4%
Little Pleasant Bay - head	**PBA-12**	**0.178**	**0.160**	**-10.1%**
Little Pleasant Bay - main basin	PBA-21	0.162	0.148	-8.5%
Paw Wah Pond	*PBA-11*	*0.257*	*0.209*	*-18.8%*
Little Quanset Pond	*WMO-12*	*0.229*	*0.194*	*-15.3%*
Quanset Pond	WMO-01	0.191	0.171	-10.8%
Round Cove	*PBA-09*	*0.241*	*0.207*	*-13.9%*
Muddy Creek - upper	PBA-05a	0.674	0.405	-40.0%
Muddy Creek - lower	*PBA-05*	*0.286*	*0.208*	*-27.3%*
Pleasant Bay - head	PBA-08	0.149	0.139	-7.1%
Pleasant Bay - off Quanset Pond	WMO-02	0.160	0.147	-8.0%
Pleasant Bay- upper Strong Island	PBA-19	0.117	0.113	-3.8%
Pleasant Bay - mid west basin	PBA-07	0.168	0.153	-8.9%
Pleasant Bay - off Muddy Creek	PBA-06	0.192	0.169	-12.0%
Pleasant Bay - Strong Island channel	PBA-20	0.124	0.118	-4.8%
Ryders Cove - upper	**PBA-03**	**0.250**	**0.190**	**-24.0%**
Ryders Cove - lower	**CM-13**	**0.158**	**0.138**	**-12.7%**
Frost Fish - lower	CM-14	0.243	0.173	-29.1%
Crows Pond	PBA-04	0.162	0.149	-8.0%
Bassing Harbor	PBA-02	0.127	0.120	-6.0%
Pleasant Bay - lower	PBA-18	0.116	0.112	-3.9%
Chatham Harbor - upper	PBA-01	0.104	0.102	-1.9%
Chatham Harbor - lower	PBA-17a	0.099	0.098	-1.0%

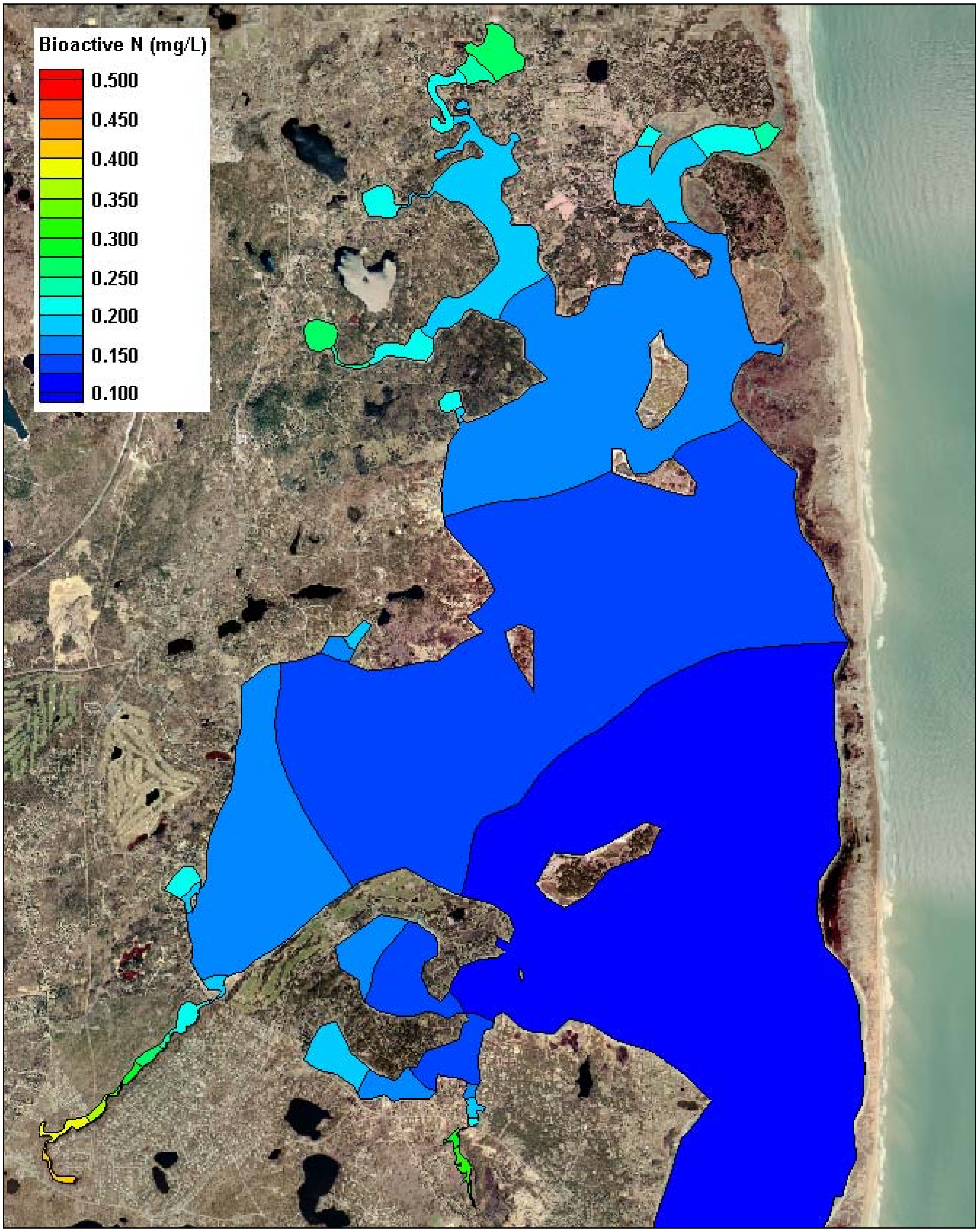


Figure VIII-1. Contour plot of modeled bioactive nitrogen (DIN+PON) concentrations (mg/L) in the Pleasant Bay system, for threshold conditions (0.16 mg/L at Upper Little Pleasant Bay and Ryder Cove).

IX. IMPACTS TO WATER QUALITY DUE TO INLET MIGRATION

IX.1 HYDRODYNAMIC EFFECTS FROM ALTERNATE INLET CONFIGURATION

As discussed in Chapter V, Pleasant Bay has a migrating inlet that over time can vary greatly the tidal conditions throughout the estuary. The present inlet configuration is nearly optimal with regard to tidal exchange for Pleasant Bay. In past years, when the inlet was positioned farther south (as it was prior to the 1987 breach) tidal conditions were less than optimal due to the additional hydraulic resistance caused by the longer inlet channel, and also because the tide range at the inlet was less. The tide range decreases as the inlet moves south due to the difference in tide ranges between the Atlantic Ocean offshore Nauset Beach and Nantucket Sound offshore Stage Harbor (southern Chatham). Therefore, as the inlet migrates, the average tide range that drives circulation in the Pleasant Bay system could vary potentially about 4 feet.

An analysis was performed to evaluate water quality conditions for the worst-case flushing scenario for Pleasant Bay. The hydrodynamic model grid of Pleasant Bay was modified to include the inlet as it existed pre-breach. The tidal open boundary condition used to drive the model was developed from a tide record measured offshore Stage Harbor in Nantucket Sound in the summer of 2000. The tide in Nantucket Sound represents the smallest tide that the inlet to Pleasant Bay could be exposed to, which is why it was selected for this worst-cases analysis.

A comparison of present a worst case tidal conditions in Pleasant Bay is presented in Figure IX-1. In this figure, hydrodynamic model output from the simulations of present and worst-case conditions are shown for stations at the inlet, at the fish pier at Chatham Harbor and in Meetinghouse Pond. From the data, the maximum tide range at the inlet is reduced from approximately 10 feet to 6 feet. At the fish pier, the range is reduced from 7 feet to 4 feet, and in Meetinghouse Pond the tide range for the worst case scenario is 2 feet smaller than the 5.5 foot range from present conditions.

Flushing rates for the old inlet scenario were computed based on the mean system volumes and prisms computed from the hydrodynamic model output. The comparison between present flushing conditions and those for the old inlet are presented in Tables IX-1 and XI-2. Generally, the mean volume of all the system sub-embayments changes less than 12%. The mean tide prisms computed for all the sub-embayments decrease more than 34%. The large decrease in tide prism results in greatly impaired flushing conditions for the whole of the Pleasant Bay system as indicated by the residence times shown in Table IX-2, where it can be seen that local flushing times increase between 22% and 44% in the system sub-embayments. By the flushing analysis alone, it is apparent that water quality conditions in Pleasant Bay could be severely impacted by a less than optimal arrangement of the system inlet.

IX.2 WATER QUALITY COMPARISON OF INLET SCENARIOS

Water quality impacts resulting from the worst-case inlet configuration are further investigated through the use of the RMA-4 water quality model created for Pleasant Bay. Using the hydrodynamic model output developed for the old inlet scenario and present nitrogen loading conditions (Table V-2), the RMA-4 model was re-run to quantify how N concentrations in the system would change as a result of impaired tidal flushing.

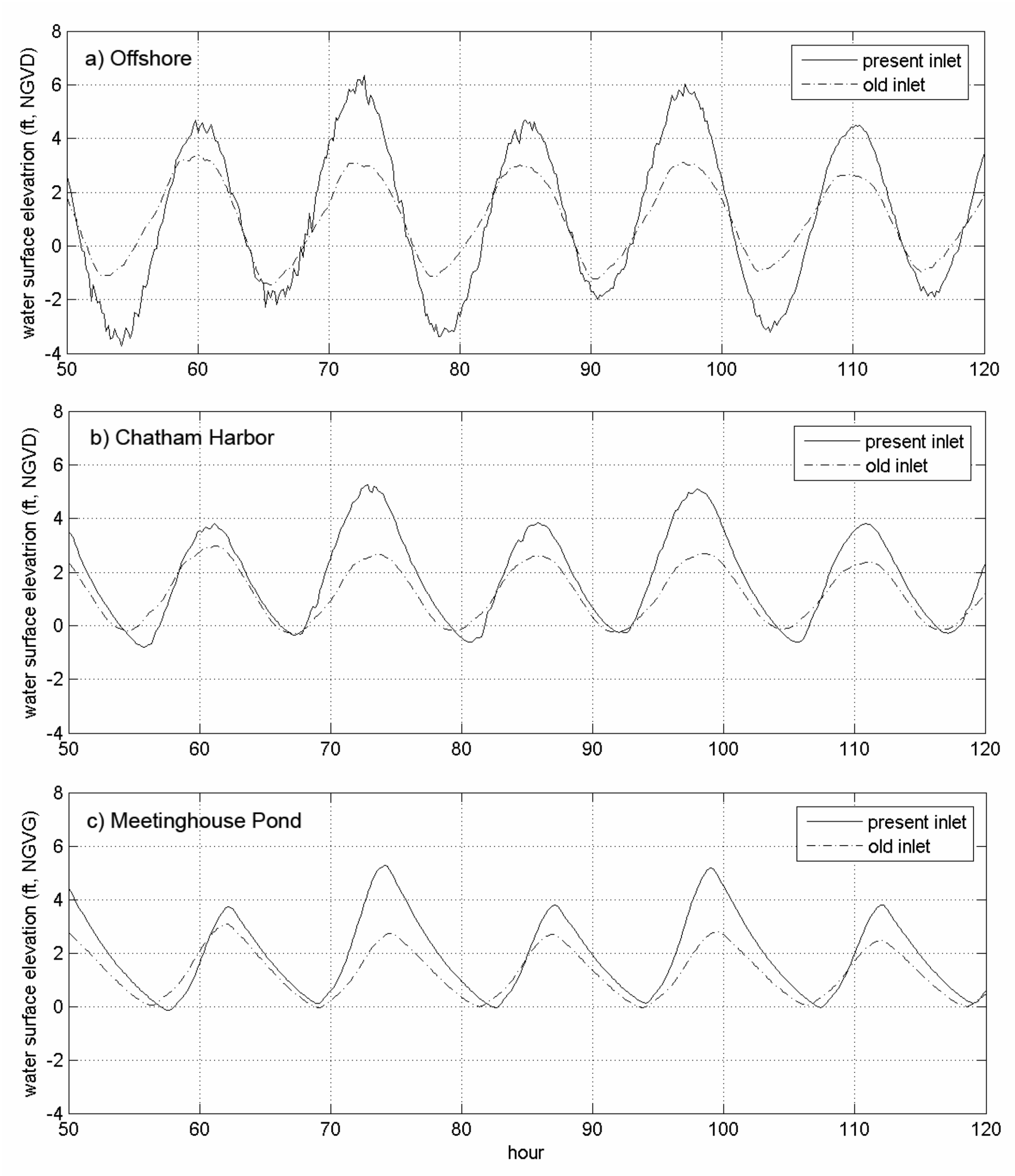


Figure IX-1. Comparison of hydrodynamic model output from simulations of present and historical ("old inlet") configurations of the inlet to Pleasant Bay. The old inlet simulation included a tidal boundary condition developed from a data record measured offshore Stage Harbor in Nantucket Sound, which is considered to be the worst-case tidal condition for Pleasant Bay.

Table IX-1. Embayment mean volumes and average tidal prism during simulation period, for present condition and historical pre-breach inlet configuration with Nantucket Sound tides ("Old Inlet").

Embayment	Present Mean System Volume (ft^3)	Old Inlet Mean System Volume (ft^3)	% change
Pleasant Bay	2,113,621,000	1,941,501,000	-8.1%
Bassing Harbor	109,139,000	97,626,000	-10.5%
Crows Pond	50,208,000	46,482,000	-7.4%
Ryder Cove	18,070,000	15,941,000	-11.8%
Muddy Creek	5,541,000	4,309,000	-22.2%
The River	96,032,000	85,417,000	-11.1%
Round Cove	2,913,000	2,428,000	-16.6%
Paw Wah Pond	2,341,000	2,067,000	-11.7%
Areys Pond	5,474,000	5,013,000	-8.4%
Kescayo Gansett Pond	6,330,000	5,827,000	-7.9%
Meetinghouse Pond	19,406,000	17,974,000	-7.4%
Embayment	**Present Tide Prism Volume (ft^3)**	**Old Inlet Tide Prism Volume (ft^3)**	**% change**
Pleasant Bay	1,207,917,000	789,266,000	-34.7%
Bassing Harbor	66,133,000	42,656,000	-35.5%
Crows Pond	21,898,000	14,124,000	-35.5%
Ryder Cove	12,534,000	8,086,000	-35.5%
Muddy Creek	806,000	515,000	-36.1%
The River	60,199,000	39,384,000	-34.6%
Round Cove	2,738,000	1,777,000	-35.1%
Paw Wah Pond	1,538,000	1,005,000	-34.7%
Areys Pond	2,623,000	1,715,000	-34.6%
Kescayo Gansett Pond	2,864,000	1,874,000	-34.6%
Meetinghouse Pond	8,167,000	5,341,000	-34.6%

A side-by-side comparison of bioactive nitrogen model output from the simulations of present and worst-case inlet conditions is presented in Figure IX-2. The color contour plots emphasize dramatically that there would be a serious degradation in water quality in the whole of the Pleasant Bay system as a result of worst-case flushing conditions at the inlet. The average bioactive N concentration in the main basin of Pleasant Bay increases 50%, from 0.157 mg/L to 0.235 mg/L. The range of concentrations in the main basin for the worst-case would be from 0.146 mg/L at the entrance to Chatham Harbor to 0.279 mg/L at the northernmost reach of Little Pleasant Bay, compared to 0.107 mg/L to 0.184 mg/L for present conditions.

Table IX-2. Computed System and Local residence times for embayments in the Pleasant Bay system, for present conditions and the historical pre-breach inlet configuration with Nantucket Sound tides ("Old Inlet").

Embayment	Present System residence time (days)	Old Inlet System residence time (days)	% change
Pleasant Bay	0.9	1.3	+40.6%
Bassing Harbor	16.5	23.6	+42.4%
Crows Pond	49.9	71.1	+42.4%
Ryder Cove	87.3	124.3	+42.4%
Muddy Creek	1357.1	1950.9	+43.8%
The River	18.2	25.5	+40.4%
Round Cove	399.5	565.4	+41.5%
Paw Wah Pond	711.2	999.7	+40.6%
Areys Pond	417.0	585.8	+40.5%
Kescayo Gansett Pond	381.9	536.1	+40.4%
Meetinghouse Pond	133.9	188.1	+40.5%
Embayment	Present Local residence time (days)	Old Inlet Local residence time (days)	% change
Pleasant Bay	0.9	1.3	+40.6%
Bassing Harbor	0.9	1.2	+38.7%
Crows Pond	1.2	1.7	+43.5%
Ryder Cove	0.7	1.0	+36.7%
Muddy Creek	3.6	4.3	+21.7%
The River	0.8	1.1	+36.0%
Round Cove	0.6	0.7	+28.4%
Paw Wah Pond	0.8	1.1	+35.1%
Areys Pond	1.1	1.5	+40.1%
Kescayo Gansett Pond	1.1	1.6	+40.7%
Meetinghouse Pond	1.2	1.7	+41.6%

An additional comparison of bioactive N model output for the two scenarios is presented in Table IX-3, which shows the difference in N concentrations at each of the water quality monitoring stations (Figure V-1). Increases in bioactive N concentrations range from 30% in Chatham Harbor to 62% at Pochet Neck. With the old inlet hydrodynamics and present loading conditions, the eelgrass threshold sentinel stations at the head of Pleasant Bay (PBA-12) and in Ryder Cove (PBA-03 and CH-13) would not be supportive of even quality benthic infaunal habitat.

The widespread loss of quality eelgrass habitat results from the poor tide flushing of the old inlet configuration run with Nantucket Sound tides. The area coverage of the main basin of Pleasant Bay with a bioactive N concentration less than the eelgrass threshold, discussed in Chapter VIII (0.16 mg/L), would decrease approximately 89% from present conditions.

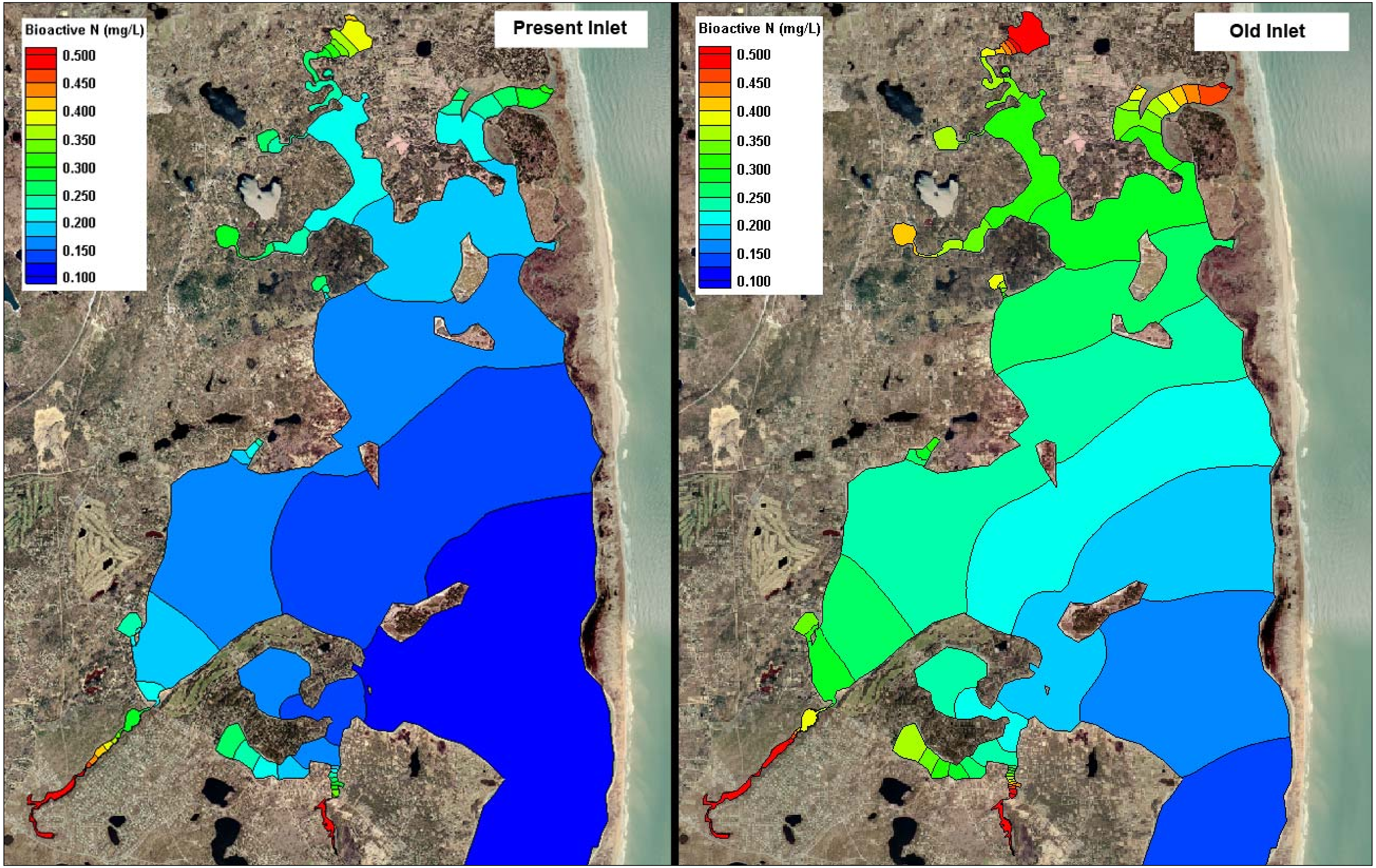


Figure IX-2. Comparison of bioactive N (DIN+PON) model runs for present inlet conditions and historical inlet (pre-breach) configuration for the Pleasant Bay system. Color contours indicate average bioactive nitrogen concentrations resulting from the present conditions loading scenario (Table VI-2).

Table IX-3. Comparison of model average bioactive N (DIN+PON) concentrations from present loading and the historical inlet configuration scenario ("old inlet") driven with Nantucket Sound Tides, with percent change, for the Pleasant Bay system. Loads for both present and "old inlet" bioactive N model runs are based on the present loading scenario (Table VI-2) The threshold stations are shown in bold print.

Sub-Embayment	monitoring station	present (mg/L)	old inlet (mg/L)	% change
Meetinghouse Pond	PBA-16	0.380	0.551	+45.0%
Meetinghouse Pond	WMO-10	0.261	0.372	+42.3%
The River - upper	WMO-09	0.239	0.345	+44.2%
The River – mid	WMO-08	0.211	0.313	+48.2%
Lonnies Pond (Kescayo Ganset Pond)	PBA-15	0.250	0.365	+46.0%
Areys Pond	PBA-14	0.297	0.417	+40.5%
Namequoit River - upper	WMO-6	0.239	0.346	+44.7%
Namequoit River - lower	WMO-7	0.216	0.319	+47.5%
The River - lower	PBA-13	0.195	0.293	+50.6%
Pochet – upper	WMO-05	0.269	0.434	+61.8%
Pochet - lower	WMO-04	0.209	0.322	+54.1%
Pochet – mouth	WMO-03	0.183	0.278	+52.1%
Little Pleasant Bay - head	**PBA-12**	**0.178**	**0.270**	**+51.9%**
Little Pleasant Bay - main basin	PBA-21	0.162	0.247	+53.1%
Paw Wah Pond	PBA-11	0.257	0.380	+47.9%
Little Quanset Pond	WMO-12	0.229	0.320	+39.7%
Quanset Pond	WMO-01	0.191	0.277	+44.9%
Round Cove	PBA-09	0.241	0.337	+40.1%
Muddy Creek - upper	PBA-05a	0.674	0.906	+34.4%
Muddy Creek - lower	PBA-05	0.286	0.387	+35.2%
Pleasant Bay - head	PBA-08	0.149	0.230	+54.3%
Pleasant Bay - off Quanset Pond	WMO-02	0.160	0.242	+51.2%
Pleasant Bay- upper Strong Island	PBA-19	0.117	0.175	+48.6%
Pleasant Bay - mid west basin	PBA-07	0.168	0.251	+48.8%
Pleasant Bay - off Muddy Creek	PBA-06	0.192	0.276	+43.7%
Pleasant Bay - Strong Island channel	PBA-20	0.124	0.186	+49.6%
Ryders Cove - upper	**PBA-03**	**0.250**	**0.360**	**+43.7%**
Ryders Cove - lower	**CM-13**	**0.158**	**0.231**	**+45.8%**
Frost Fish - lower	CM-14	0.243	0.351	+44.1%
Crows Pond	PBA-04	0.162	0.230	+41.7%
Bassing Harbor	PBA-02	0.127	0.191	+50.0%
Pleasant Bay - lower	PBA-18	0.116	0.169	+45.3%
Chatham Harbor - upper	PBA-01	0.104	0.135	+30.2%
Chatham Harbor - lower	PBA-17a	0.099	0.140	+41.3%

The results of this analysis indicate that the natural range of hydraulic conditions at the inlet to the Pleasant Bay system has a much greater potential influence on water quality conditions than anthropomorphic effects, such as those from the projected build-out nitrogen loading scenario. As a suggestion, an inlet management plan should be developed to address possible future water quality problems that could occur as a result of less-than-optimal configurations of the Pleasant Bay inlet.

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