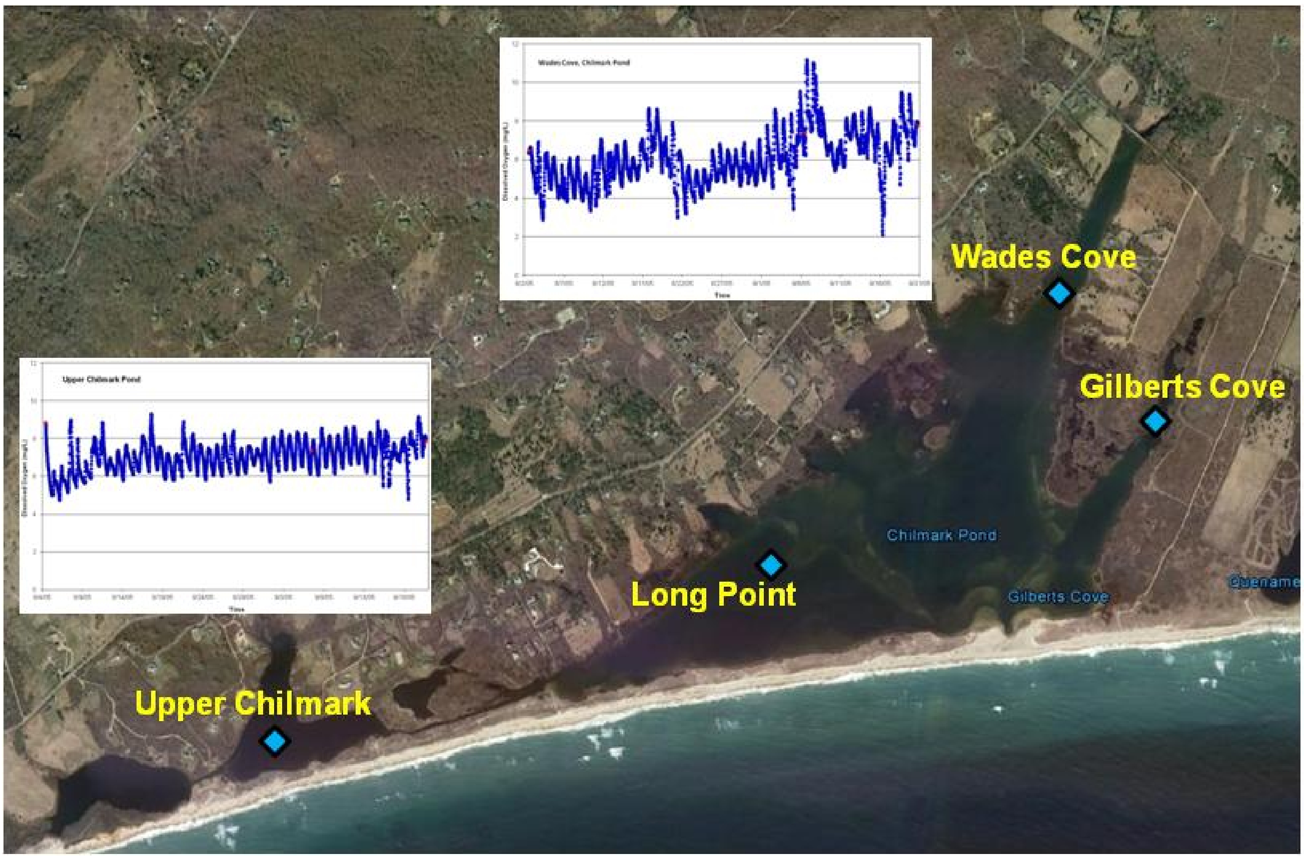
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Threshold for the Chilmark Pond Embayment System, Town of Chilmark, MA



University of Massachusetts Dartmouth
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Massachusetts Department of
Environmental Protection

FINAL REPORT – April 2015

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Executive Summary

1. Background

This report presents the results generated from the implementation of the Massachusetts Estuaries Project's Linked Watershed-Embayment Approach to the Chilmark Pond embayment system, a coastal embayment entirely within the Town of Chilmark, Massachusetts. Analyses of the Chilmark Pond embayment system was performed to assist the Town of Chilmark with up-coming nitrogen management decisions associated with the current and future wastewater planning efforts of the Town, as well as wetland restoration, management of anadromous fish runs and shell fisheries as well as the development of open-space management programs. As part of the MEP approach, habitat assessment was conducted on the embayment based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements, and benthic community structure. Nitrogen loading thresholds for use as goals for watershed nitrogen management are the major product of the MEP effort. In this way, the MEP offers a science-based management approach to support the Town of Chilmark resource planning and decision-making process. The primary products of this effort are: (1) a current quantitative assessment of the nutrient related health of the Chilmark Pond embayment, (2) identification of all nitrogen sources (and respective N loads) to embayment waters, (3) nitrogen threshold levels for maintaining Massachusetts Water Quality Standards within embayment waters, (4) analysis of watershed nitrogen loading reduction to achieve the N threshold concentrations in embayment waters, and (5) a functional calibrated and validated Linked Watershed-Embayment modeling tool that can be readily used for evaluation of nitrogen management alternatives (to be developed by the Town) for the restoration of the Chilmark Pond embayment system.

Wastewater Planning: As increasing numbers of people occupy coastal watersheds, the associated coastal waters receive increasing pollutant loads. Coastal embayments throughout the Commonwealth of Massachusetts (and along the U.S. eastern seaboard) are becoming nutrient enriched. The elevated nutrients levels are primarily related to the land use impacts associated with the increasing population within the coastal zone over the past half-century.

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 our coastal embayments is nitrogen, with its primary sources being wastewater disposal, and nonpoint source runoff that carries nitrogen (e.g. fertilizers) from a range of other sources. Nitrogen related water quality decline represents one of the most serious threats to the ecological health of the nearshore coastal waters. Coastal embayments, because of their shallow nature and large shoreline area, are generally the first coastal systems to show the effect of nutrient pollution from terrestrial sources.

In particular, the Chilmark Pond embayment system within the Town of Chilmark is at risk of eutrophication (over enrichment) from enhanced nitrogen loads entering through groundwater from the increasingly developed watershed to this coastal system. Eutrophication is a process that occurs naturally and gradually over a period of tens or hundreds of years. However, human-related (anthropogenic) sources of nitrogen may be introduced into ecosystems at an accelerated rate that cannot be easily absorbed, resulting in a phenomenon known as cultural eutrophication. In both marine and freshwater systems, cultural eutrophication results in degraded water quality, adverse impacts to ecosystems, and limits on the use of water resources.

The Town of Chilmark has recognized the severity of the problem of eutrophication and the need for watershed nutrient management and is currently engaged in wastewater management at a variety of levels. Moreover, the Town of Chilmark is recognizing the need to work collaboratively regarding the future implementation of the MEP nutrient threshold analysis of the Tisbury Great Pond system (watershed partially in the Town of Chilmark) as well as the upcoming threshold for the Menemsha/Squibnocket system (shared between the Town of Chilmark, Aquinnah and the Wampanoag Tribe). For the Town of Chilmark, this analysis of the Chilmark Pond system should be considered relative to the already completed Tisbury Great Pond nutrient threshold analysis as well as the soon to be completed nutrient threshold analysis of Menemsha/Squibnocket Pond system in order to plan out and implement a unified town-wide approach to nutrient management for Chilmark. The Town of Chilmark with associated working groups (e.g. Chilmark Pond Association, Tisbury Great Pond Riparian Association, Martha's Vineyard Shellfish Group) have recognized that a rigorous scientific approach yielding site-specific nitrogen loading targets was required for decision-making and alternatives analysis. The completion of this multi-step process has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, which is a partnership effort between all MEP collaborators and the Towns in the study region. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns' nutrient management groups to predict the impacts on water quality from a variety of proposed management scenarios.

Nitrogen Loading Thresholds and Watershed Nitrogen Management: Realizing the need for scientifically defensible management tools has resulted in a focus on determining the aquatic system's assimilative capacity for nitrogen. The highest-level approach is to directly link the watershed nitrogen inputs with embayment hydrodynamics to produce water quality results that can be validated by water quality monitoring programs. This approach when linked to state-of-the-art habitat assessments yields accurate determination of the "allowable N concentration increase" or "threshold nitrogen concentration". These determined nitrogen concentrations are then directly relatable to the watershed nitrogen loading, which also accounts for the spatial distribution of the nitrogen sources, not just the total load. As such, changes in nitrogen load from differing parts of the embayment watershed can be evaluated relative to the degree to which those load changes drive embayment water column nitrogen concentrations toward the "threshold" for the embayment system. To increase certainty, the "Linked" Model is independently calibrated and validated for each embayment.

Massachusetts Estuaries Project Approach: The Massachusetts Department of Environmental Protection (DEP), the University of Massachusetts – Dartmouth School of Marine Science and Technology (SMAST), and others including the Cape Cod Commission (CCC) have undertaken the task of providing a quantitative tool to communities throughout southeastern Massachusetts (the Linked Watershed-Embayment Management Model) for nutrient management in their coastal embayment systems. Ultimately, use of the Linked Watershed-Embayment Management Model tool by municipalities in the region results in effective screening of nitrogen reduction approaches and eventual restoration and protection of valuable coastal resources. The MEP provides technical guidance in support of policies on nitrogen loading to embayments, wastewater management decisions, and establishment of nitrogen Total Maximum Daily Loads (TMDLs). A TMDL represents the greatest amount of a pollutant that a waterbody can accept and still meet water quality standards for protecting public health and maintaining the designated beneficial uses of those waters for drinking, swimming, recreation and fishing. The MEP modeling approach assesses available options for meeting selected nitrogen goals that are protective of embayment health and achieve water quality standards.

The core of the Massachusetts Estuaries Project analytical method is the Linked Watershed-Embayment Management Modeling Approach, which links watershed inputs with embayment circulation and nitrogen characteristics.

The Linked Model builds on well-accepted basic watershed nitrogen loading approaches such as those used in the Buzzards Bay Project, the CCC models, and other relevant models. However, the Linked Model differs from other nitrogen management models in that it:

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

The Linked Model 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.

For a comprehensive description of the Linked Model, please refer to the *Full Report: Nitrogen Modeling to Support Watershed Management: Comparison of Approaches and Sensitivity Analysis*, available for download at http://www.mass.gov/dep/water/resources/coastalr.htm . A more basic discussion of the Linked

Model is also provided in Appendix F of the *Massachusetts Estuaries Project Embayment Restoration Guidance for Implementation Strategies*, available for download at http://www.mass.gov/dep/water/resources/coastalr.htm . The Linked Model suggests which management solutions will adequately protect or restore embayment water quality by enabling towns to test specific management scenarios and weigh the resulting water quality impact against the cost of that approach. In addition to the management scenarios modeled for this report, the Linked Model can be used to evaluate additional management scenarios and may be updated to reflect future changes in land-use within an embayment watershed or changing embayment characteristics. 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. Unlike many approaches, the Linked Model accounts for nutrient sources, attenuation, and recycling and variations in tidal hydrodynamics and accommodates the spatial distribution of these processes. For an overview of several management scenarios that may be employed to restore embayment water quality, see *Massachusetts Estuaries Project Embayment Restoration Guidance for Implementation Strategies*, available for download at http://www.mass.gov/dep/water/resources/coastalr.htm.

Application of MEP Approach: The Linked Model was applied to the Chilmark Pond embayment system by using site-specific data collected by the MEP and water quality data from the Water Quality Monitoring Program conducted primarily by the Martha's Vineyard Commission and with field support from the Town of Chilmark. The water quality monitoring program was conducted with technical guidance from the Coastal Systems Program at SMAST (see Section II). Evaluation of upland nitrogen loading was conducted by the MEP and data was provided by the Planning Department in the Town of Chilmark as well as the Martha's Vineyard Commission. The watersheds utilized in the MEP assessment are largely based on delineations created and used by the Martha's Vineyard Commission (MVC). The portions of the watershed within the outwash plain have been delineated based on regional groundwater contours (Delaney, 1980) and with more refined water level readings in selected areas (Wilcox, 1996). In 1994, Whitman and Howard produced a groundwater model with a domain that covered Martha's Vineyard eastern moraine and the outwash plain; this model was based on the publicly available USGS MODFLOW three-dimensional, finite difference groundwater model code. The Wilcox (1996) watershed delineation completed for the MVC utilizes all of the previous characterizations. These watershed delineations and the land-use data were used to determine watershed nitrogen loads within the Chilmark Pond embayment system and each of the systems sub-embayments as appropriate (current and build-out loads are summarized in Section IV). Water quality within a sub-embayment is the integration of nitrogen loads with the site-specific estuarine circulation. Therefore, water quality modeling of this tidally influenced estuary included 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. Once the hydrodynamics of the system was quantified, transport of nitrogen was evaluated from tidal current information developed by the numerical models during breach conditions.

A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents during a breach event and water elevations was employed for the Chilmark Pond embayment system. Once the hydrodynamic properties of the estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates. Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic model was 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. Boundary nutrient concentrations in Atlantic Ocean source waters were taken from water quality monitoring data. Measurements of current salinity distributions throughout the estuarine waters of the Chilmark Pond embayment system was used to calibrate the water quality model, with validation using measured nitrogen concentrations (under existing loading conditions). The underlying hydrodynamic model was calibrated and validated independently using water elevations measured in time series throughout the embayment.

MEP Nitrogen Thresholds Analysis: The threshold nitrogen level for an embayment represents the average water column concentration of nitrogen that will support the habitat quality being sought. The water column nitrogen level is ultimately controlled by the watershed nitrogen load and the nitrogen concentration in the inflowing tidal waters (boundary condition). The water column nitrogen concentration is modified by the extent of sediment regeneration. Threshold nitrogen levels for the embayment systems in this study were 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 also considered in the assessment.

The nitrogen thresholds developed in Section VIII-2 were used to determine the amount of total nitrogen mass loading reduction required for restoration of infaunal habitats (eelgrass has not been documented presently or historically) observed in the Chilmark Pond embayment system. Tidally averaged total nitrogen thresholds derived in Section VIII.1 and VIII.2 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, in concert with modifications to the pond opening schedule until the nitrogen levels reached the threshold level at the sentinel station chosen for the Chilmark Pond system. It is important to note that load reductions can be produced by reduction of any or all sources, increasing flushing of the system with clean open ocean water or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

The Massachusetts Estuaries Project's thresholds analysis, as presented in this technical report, provides the site-specific nitrogen reduction guidelines for nitrogen management of the Chilmark Pond embayment system in the Town of Chilmark. Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in nitrogen loading reduction to the embayment. Hydrodynamic and water quality model runs were performed to investigate quantitatively how flushing and TN concentrations would change in the Chilmark Pond system assuming breaching the pond three times per year could be achieved.

The MEP analysis has initially focused upon nitrogen loads from on-site septic systems as a test of the potential for achieving the level of total nitrogen reduction for restoration of each embayment system. The concept was that since nitrogen loads associated with wastewater generally represent 43% of the system-wide controllable watershed load to the Chilmark Pond embayment system and are more manageable than other of the nitrogen sources (e.g. agriculture @ 45% of the controllable load system-wide), the ability to achieve needed reductions through this source is a good gauge of the feasibility for restoration of this system.

2. Problem Assessment (Current Conditions)

A habitat assessment was conducted throughout the Chilmark Pond embayment system based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements of dissolved oxygen and chlorophyll, and benthic community structure. At present, the Chilmark Pond Estuary is showing nitrogen enrichment and impairment of both eelgrass and infaunal habitats (Section VII), indicating that nitrogen management of this system will be for restoration rather than for protection or maintenance of an unimpaired system.

The Chilmark Pond Embayment System is a complex coastal open water embayment comprised of a large central basin (Lower Chilmark Pond {east}) and multiple sub-embayments (Wades Cove, Gilberts Cove). The western basin, Upper Chilmark Pond, is currently fresh to slightly brackish and has been functionally separated from the estuary by coastal processes. The main basin and its tributary coves are maintained as an estuary by the periodic breaching of the barrier beach with a single temporary inlet. The estuary only occasionally receives tidal waters from the Atlantic Ocean into its main basin based on a schedule of openings set by the Town. Floodwater from the Atlantic Ocean enters the main basin of Lower Chilmark Pond (east) and circulates through channels and across flats making its way up into Wades Cove (the primary tributary basin in this system) as well as into upper Chilmark Pond (west), which is connected to Lower Chilmark Pond via Doctor's Creek, a narrow channel (Figure I-2). Upper Chilmark Pond is really comprised of two basins which are connected by a very small shallow channel locally referred to as Interns Creek. The pond openings follow periods where pond level rises due to groundwater and surface water inflows and precipitation, which creates the hydraulic head needed for the opening process. At present, the number and duration of pond openings plays a fundamental role in the maintenance of nutrient related water quality and habitat health throughout this estuary.

At present, the Chilmark Pond Estuary is beyond its ability to assimilate nitrogen without further impairment. The system is showing a moderate level of nitrogen enrichment, no eelgrass habitat and moderately/significantly impaired benthic animal habitats, regions of periodic moderate oxygen depletion and phytoplankton blooms. All lines of evidence support an assessment of habitat impairment. Since there is no record of eelgrass in this estuary in recent decades, the impairment of concern is that of benthic animal habitat (Table VIII-1). These findings indicate that nitrogen management of this system will be for restoration rather than for protection or maintenance of an unimpaired system.

The measured levels of oxygen depletion and enhanced chlorophyll-*a* levels follows the spatial pattern of total nitrogen levels in this system (Section VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment and only periodic tidal flows. The spatial pattern indicated that the magnitude of oxygen depletion, enhancement of chlorophyll-*a* levels and total nitrogen concentrations were consistent with the absence of eelgrass and the moderate impairment of benthic animal communities.

Given moderate levels of watershed nitrogen loading and limited tidal exchange only periodically occurring during managed breaches of the barrier beach and the nitrogen, chlorophyll and oxygen levels within the pond basins (2000-2012), it can be concluded that Chilmark Pond does not presently support eelgrass habitat. Further, based upon the past decade and analysis of available historic information, the MEP Technical Team concluded that Chilmark Pond Embayment System has not supported eelgrass habitat for at least 50 years. Given that the pond's water quality is controlled in significant part by the amount of induced tidal flushing, it is likely that the Pond has had negligible eelgrass habitat for the past century. As eelgrass habitat could not be documented to exist, either historically or presently, within the Chilmark Pond Embayment System, the threshold analysis for this system is necessarily focused on restoration/protection of infaunal animal habitat.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll-*a* levels indicate highly nutrient enriched waters throughout the Chilmark Pond Estuary, particularly the oxygen depletion and D.O. excursions and phytoplankton biomass in each basin, especially Wades Cove (Section VII). It should be noted that the Water Quality Monitoring Program observed similar levels of chlorophyll and bottom water oxygen depletion in critical areas of the system, although it did not always capture the minimum oxygen or maximum chlorophyll-*a* conditions at each site. The oxygen data is consistent with a high level of organic matter enrichment, primarily from phytoplankton production as seen from the parallel measurements of chlorophyll-*a*. The measured levels of oxygen depletion and enhanced chlorophyll-*a* levels are consistent with the nitrogen levels within the various basins (Section VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment of this estuarine system.

Overall, the infauna survey indicated that most sub-basins comprising the Chilmark Pond Embayment System are presently beyond their ability to tolerate additional nitrogen inputs without further impairment. Consistent with the observed periodic oxygen depletions and large phytoplankton blooms occurring in the main depositional basins, with little drift macroalgal accumulation, the benthic animal communities are showing moderate to significant impairment. The impairment is consistent with organic enrichment resulting from increased nitrogen loading from a combination of watershed inputs and only periodic tidal flushing. The Benthic survey did not reveal any areas of severe degradation (less than 70 animal per grab), as indicated by low numbers of individuals and species or dominance by opportunistic stress indicator species such as Capitellids and Tubificids. In fact, at all locations throughout the estuarine sub-basins of this embayment system, there were high numbers of individuals (400-700 per grab sample), low numbers of opportunistic stress indicator species (Capitellids and Tubificids, generally <10% of community), but the community was composed of few species (7-11) with low diversity (H' = 1.5-2.2). Species numbers of 20-25 and diversity >3.0 generally indicate high quality benthic habitats.

3. Conclusions of the Analysis

The threshold nitrogen level for an embayment represents the average watercolumn concentration of nitrogen that will support the habitat quality being sought. The watercolumn nitrogen level is ultimately controlled by the integration of the watershed nitrogen load, the nitrogen concentration in the inflowing tidal waters (boundary condition) and dilution and flushing via tidal flows. The water column nitrogen concentration is modified by the extent of sediment regeneration and by direct atmospheric deposition.

Threshold nitrogen levels for this embayment system were developed to restore or maintain SA waters or high habitat quality. In this system, high habitat quality was defined as supportive of diverse benthic animal communities. Dissolved oxygen and chlorophyll-*a* were also considered in the assessment.

Watershed nitrogen loads (Tables ES-1 and ES-2) for the Chilmark Pond system in the Town of Chilmark were comprised primarily of wastewater nitrogen and agricultural sources. Land-use and wastewater analysis found that generally about 43% of the controllable watershed nitrogen load (system-wide) to the embayment was from wastewater and 45% system-wide from agricultural activity (Upper Chilmark Pond {west}=55% animal agriculture, Lower Chilmark Pond {east}=19% animal agriculture).

A major finding of the MEP clearly indicates that a single general total nitrogen threshold can not be applied to Massachusetts' estuaries, based upon the results of the Great, Green and Bournes Pond Systems, Popponesset Bay System, the Hamblin / Jehu Pond / Quashnet River analysis in eastern Waquoit Bay and the analysis of the nearby Tisbury Great Pond system and Sengekontacket Pond system as well as Farm Pond, Lagoon Pond and Edgartown Great Pond. This is almost certainly going to continue to be true for the other embayments within the MEP area, as well, inclusive of Chilmark Pond.

The threshold nitrogen levels for the Chilmark Pond embayment system were determined as follows:

Chilmark Pond Threshold Nitrogen Concentrations

- Following the MEP protocol, the restoration target for the Chilmark Pond system should reflect both recent pre-degradation habitat quality and be reasonably achievable. Based upon the assessment data (Section VII), the Chilmark Pond system is presently supportive of habitat in varying states of impairment, depending on the component sub-basins of the overall system (e.g. Upper Chilmark Pond portion of the system which receives the majority of fresh surfacewater inflow compared to shallow areas in the lower portion of the main basin {Lower Chilmark Pond} which is closest the periodic breaches).

- The primary habitat issue within the Chilmark Pond Embayment System relates to the impaired infaunal habitat. While the numbers of individuals remain high throughout the system, the community numbers of species and their Diversity and Evenness are low and indicative of a community under ecological stress. There was little substantive difference between the basins as all are clearly moderately impaired relative to benthic animal habitat. Given the prevalence of species tolerant of moderate organic enrichment, the low numbers of stress indicator organisms, the low numbers of species and the low diversity of Chilmark Pond's benthic communities compared to high quality habitat areas in similarly structured embayments in southeastern Massachusetts, it is clear that the main basin of Chilmark Pond and the major coves (Wades, Gilberts) are currently above their nitrogen threshold and is supporting impaired benthic animal habitat.

- Within the Chilmark Pond Estuary the most appropriate sentinel "station" was to use the average of the 5 long-term monitoring stations (CHP1-5) distributed throughout the main eastern basin, Gilberts Cove and Wades Cove (Figure II-1). This average approach has been used in other open "single basin" estuaries that are only periodically open to tidal flow throughout the MEP region. The average was selected because given the relatively long periods between openings, dispersion and wind driven mixing result in a relatively uniform total nitrogen concentration throughout the estuary. Present TN levels within the Chilmark Pond Estuary during summer are ~0.74 mg TN L^{-1}, consistent with the observed lack of eelgrass beds and impaired benthic animal habitat. Based upon comparisons to other systems, the current TN level within the Chilmark Pond Estuary, the periodic oxygen depletions and phytoplankton blooms, it appears that a water column nitrogen threshold for the Chilmark Pond Estuary of <0.50 mg TN L^{-1} is required for restoration.

- The main goals of the threshold load scenario tested during the threshold analysis were to restore benthic infauna habitat throughout Chilmark Pond and simultaneously attempt to restore a modest level of eelgrass habitat within the main basin which has been non-existent over the past several decades. To restore benthic habitat, load reduction focused on lowering average TN levels of stations with the main basin to 0.50 mg/L during the summer months. This goal was achieved by reducing the watershed loading to the pond and assuming the pond is breached three times a year. Watershed loading was reduced from present conditions until the combined time averaged TN concentration would remain below 0.50 mg/L during a 120-day period during the summer months. The threshold modeling assumptions include a successful spring breach, which remains open for 8 days and lowers the average pond TN concentration to 0.33 mg/L. The Pond is also allowed to be closed for 120 days, which allows the time for the water level in the pond to rise. To achieve the threshold a 30% septic reduction from present conditions was required in the septic load to the pond. This is but one example of a loading reduction that can achieve the threshold assuming the above mentioned breaching criteria can be achieved.

For restoration of the Chilmark Pond Embayment System, the primary nitrogen threshold at the "sentinel station" will need to be achieved. At the point that the threshold level is attained at the sentinel station, water column nutrient concentrations will also be at a level that will be supportive of healthy infaunal communities. The results of the Linked Watershed-Embayment modeling are used to ascertain that when the nitrogen threshold is attained, TN levels in the regions associated with the primary criteria of healthy infauna are also within an acceptable range.

It is important to note that the analysis of future nitrogen loading (build-out) to the Chilmark Pond estuarine system focuses upon additional shifts in land-use from forest/grasslands to residential and commercial development. However, the MEP analysis indicates that significant increases in nitrogen loading can occur under present land-uses, due to shifts in occupancy, shifts from seasonal to year-round usage and increasing use of fertilizers. Therefore, watershed-estuarine nitrogen management must include management approaches to prevent increased nitrogen loading from both shifts in land-uses (new sources) and from loading increases of current land-uses. The overarching conclusion of the MEP analysis of the Chilmark Pond estuarine system is that restoration will necessitate a reduction in the present (Chilmark 2010) nitrogen inputs and management options to negate additional future nitrogen inputs.

Table ES-1. Existing total and sub-embayment nitrogen loads to the estuarine waters of the Chilmark Pond estuary system, observed nitrogen concentrations, and sentinel system threshold nitrogen concentrations.

Sub-embayments	Natural Background Watershed Load [1] (kg/day)	Present Land Use Load [2] (kg/day)	Present Septic System Load (kg/day)	Present WWTF Load [3] (kg/day)	Present Watershed Load [4] (kg/day)	Direct Atmospheric Deposition [5] (kg/day)	Present Net Benthic Flux (kg/day)	Present Total Load [6] (kg/day)	Observed TN Conc. [7] (mg/L)	Threshold TN Conc. (mg/L)
Chilmark East	0.899	2.411	3.074	-	5.485	3.260	-0.273	8.473	0.61	0.50
Chilmark West	3.019	8.545	3.068	-	11.614	0.655	-3.100	9.169	-	-
Combined Total	**3.918**	**10.956**	**6.142**	**-**	**17.099**	**3.915**	**-5.251**	**15.762**	**0.61**	**0.50**

[1] assumes entire watershed is forested (i.e., no anthropogenic sources)
[2] composed of non-wastewater loads, e.g. fertilizer and runoff and natural surfaces and atmospheric deposition to lakes
[3] existing wastewater treatment facility discharges to groundwater
[4] composed of combined natural background, fertilizer, runoff, and septic system loadings.
[5] atmospheric deposition to embayment surface only
[6] composed of natural background, fertilizer, runoff, septic system atmospheric deposition and benthic flux loadings
[7] average of 2004 data, ranges show the upper to lower regions (highest-lowest) of an sub-embayment. Individual yearly means and standard deviations in Table VI-1.
[8] Threshold for sentinel sites are located

Table ES-2. Present Watershed Loads, Thresholds Loads, and the percent reductions necessary to achieve the Thresholds Loads for the Chilmark Pond estuarine system in Chilmark, Massachusetts.

Sub-embayments	Present Watershed Load [1] (kg/day)	Target Threshold Watershed Load [2] (kg/day)	Direct Atmospheric Deposition (kg/day)	Benthic Flux Net [3] (kg/day)	TMDL [4] (kg/day)	Percent watershed change needed to achieve threshold load levels
Chilmark East	5.485	4.255	3.260	-0.297	7.219	22.4%
Chilmark West	11.614	10.540	0.655	-2.924	8.270	9.2%
Combined Total	**17.099**	**14.795**	**3.915**	**-3.221**	**15.489**	**13.5%**

(1) Composed of combined natural background, fertilizer, runoff, and septic system loadings.
(2) Target threshold watershed load is the load from the watershed needed to meet the embayment threshold concentration identified in Table ES-1.
(3) Projected future flux (present rates reduced approximately proportional to watershed load reductions).
(4) Sum of target threshold watershed load, atmospheric deposition load, and benthic flux load.

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

First and foremost we would like to recognize and applaud the significant time and effort in data collection and discussion spent by members of the Martha's Vineyard Commission. These individuals gave of their time to develop a consistent and sound baseline of nutrient related water quality for this system, without which the present analysis would not have been possible. Also, we would like to thank the long standing efforts of the Chilmark Pond Association (specifically Liz Lewenberg and Joan Malkin) who have been steadfast champions for monitoring the state of the pond, educating the public and driving the need to complete the MEP analysis for the Chilmark Pond system.

Staff from the Martha's Vineyard Commission and volunteers from the Town of Chilmark have provided essential insights toward this effort. Of particular note has been the efforts of Joan Malkin, a concerned citizen who lives close to the pond and who was willing to boldly collect samples well into the winter to document effects of one of the pond openings. We also thank Bill Wilcox (former MVC Water Resources Planner), who has spent countless hours reviewing data and information with MEP Technical Team members in support of the MEP analysis of Chilmark Pond. In addition, Sheri Caseau (current MVC Water Resources Planner) provided local insights and worked to formulate the animal database and Chris Seidel, GIS Specialist from the MVC, provided significant support for the MEP land-use analysis, particularly analysis of parcel information and site-specific loading information (e.g. related to wastewater disposal.

In addition to local contributions, technical, policy and regulatory support has been freely and graciously provided by our MassDEP colleagues: Rick Dunn and Dave DeLorenzo. We are also thankful for the long hours in the field and laboratory spent by the technical staff, interns and students within the Coastal Systems Program at SMAST-UMD.

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PROPER CITATION

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I. INTRODUCTION

The Chilmark Pond Embayment System is a complex estuary located entirely within the Town of Chilmark on the island of Martha's Vineyard, Massachusetts with a southern shore bounded by water from the Atlantic Ocean (Figure I -1). The Chilmark Pond watershed is distributed entirely in the Town of Chilmark. Land-uses closest to an embayment generally have greater impact than those in the upper portions of the watershed, which can support attenuation of nitrogen during transport through natural aquatic systems (e.g. ponds, rivers, wetlands etc.) prior to discharge to the embayment. However, effective nutrient management for restoration of the Chilmark Pond Embayment System will require consideration of all sources of nitrogen load throughout the entire watershed. That the open water basins and the entire watershed to the Chilmark Pond system is contained within one town will make development and implementation of a comprehensive nutrient management and restoration plan a little more simple as the challenges are reduced due to the lack of potentially conflicting municipal constraints and regulations.

Figure I-1. Location of the Chilmark Pond Embayment System, Island of Martha's Vineyard, Town of Chilmark, Massachusetts. Chilmark Pond is a great salt pond, maintained by periodic breaching of the barrier beach to lower nitrogen levels and increase salinity via tidal exchange with Atlantic Ocean waters.

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. The multiple coves and sub-embayments to the Chilmark Pond Embayment System greatly increases the shoreline and decreases the travel time of groundwater (and its pollutants) from the watershed recharge areas to bay regions of discharge. As such, the Chilmark Pond estuary is particularly vulnerable to the effects of nutrient enrichment from the watershed, especially considering that circulation is mainly through wind driven mixing in the small tributary sub-embayments, the long shoreline of the pond and the only periodic flushing with "clean" Atlantic Ocean water. In particular, the Chilmark Pond Embayment System and its sub-embayments along the south shore of Martha's Vineyard are at risk of eutrophication (over enrichment) from nitrogen enriched groundwater and surface water flows and runoff from the watershed.

The Chilmark Pond Embayment System is a complex coastal open water embayment comprised of a large eastern central basin (Lower Chilmark) and multiple sub-embayments (Upper Chilmark Pond to the west, Wades Cove, Gilberts Cove). The system is maintained as an estuary by the periodic breaching of the barrier beach with a single temporary inlet. The estuary only occasionally receives tidal waters from the Atlantic Ocean into its main basin based on a schedule of openings set by the Town. Floodwater from the Atlantic Ocean enters the main basin of Lower Chilmark Pond (east) and circulates through channels and across flats making its way up into Wades Cove (the primary tributary basin in this system) as well as into upper Chilmark Pond (west), which is connected to Lower Chilmark Pond via Doctor's Creek, a narrow channel (Figure I-2). Upper Chilmark Pond is really comprised of two basins which are connected by a very small shallow channel locally referred to as Interns Creek. The pond openings follow periods where pond level rises due to groundwater and surface water inflows and precipitation, which creates the hydraulic head needed for the opening process. At present the number and duration of pond openings plays a fundamental role in the maintenance of nutrient related water quality and habitat health throughout this estuary.

The present Chilmark Pond Embayment System results from a complex geologic history dominated by glacial processes occurring during the last glaciation of the southeastern Massachusetts region. The late Wisconsinan Laurentide ice sheet reached its maximum extent and southernmost position about 20,000 years before present (BP), as indicated by the presence of terminal moraines on Martha's Vineyard and Nantucket and the southern limit of abundant gravel on the sea floor of Nantucket Sound and Vineyard Sound (Schlee and Pratt, 1970; Oldale, 1992; Uchupi et al., 1996). The lobate ice front was comprised of the Buzzards Bay lobe that deposited the moraine along the western part of Martha's Vineyard, the Cape Cod Bay lobe that deposited the moraines across eastern Martha's Vineyard and Nantucket, and the South Channel lobe that extended east toward Georges Bank (Oldale and Barlow, 1986; Oldale, 1992). During the retreat of the ice sheet, approximately 18,000 years BP, the main part of Cape Cod was deposited as the Barnstable outwash plain. The watershed to the Chilmark Pond Embayment System is composed of both moraine deposits to the west and sandy outwash plain to the east, with the dividing line running up Wades Cove and inland.

As the ice sheet retreated and a glacial lake occupied Nantucket Sound, the glacial meltwater lake occupying what is now considered Nantucket Sound is likely to have had a profound effect on the geomorphology of Chilmark Pond. The tributary coves (Wades Cove, Gilberts Cove) were likely formed by headward erosion by groundwater seepage fed from the glacial meltwater lake upgradient of present day Chilmark Pond. The process driving the formative headward erosion of the finger tributaries of Chilmark Pond is called spring sapping. This occurs when the water discharging from a spring to a wetland environment carries away loose sand and gravel and causes the spring and associated wetland to erode (and migrate) headward (up-gradient) carving a long straight valley which later fills with seawater with rising sea levels post-glaciation. The terrestrial eroded "valleys" that represent the finger like tributary coves of the Chilmark Pond system are relict, because neither Wades Cove or Gilberts Cove do not presently contain rivers or streams. They remain dry, except where their lower reaches have been drowned by the rise in sea level.

Figure I-2. Study region for the Massachusetts Estuaries Project analysis of the Chilmark Pond Embayment System. Tidal waters enter the Pond through periodic breaching of the barrier beach to allow tidal exchanges with Atlantic Ocean waters. Freshwaters enter from the watershed primarily through direct groundwater discharge as well as surface water inflows via Mill Brook and the Fulling Mill Brook, both of which discharge to the western basin of Chilmark Pond (upper).

The basins of the Chilmark Pond Embayment System, Lower Chilmark Pond (east) and Upper Chilmark Pond (west) were formed by coastal processes forming a barrier beach along the open basin front to the Atlantic Ocean. These basins are properly termed lagoons (e.g. lagoonal estuarine basins) and run parallel to the coast behind the sandy barrier. The formation and structure of the Chilmark Pond Embayment System parallels that of its larger neighbors, Tisbury Great Pond and Edgartown Great Pond.

The formation of the Chilmark Pond System has been and continues to be greatly affected by coastal processes, specifically the role that the barrier beach plays in separating the pond from Atlantic Ocean waters. The ecological and biogeochemical structure of the pond is likely to have changed over time as the barrier beach has migrated land-ward and naturally breached and closed as a function of high pond levels (freshwater inflow) and storm frequency and intensity. It is almost certain that its closed basin is geologically a recent phenomenon, and that the pond was more generally open during lower stands of sea level.

The Chilmark Pond embayment system periodically exchanges tidal water with the Atlantic Ocean through managed "breaching" of the barrier beach (South Beach). This great salt pond is opened to tidal exchange by excavating a trench through the barrier beach seasonally as the water levels in the pond rise sufficiently (by freshwater inflow) to provide sufficient hydraulic head to erode the desired channel to the sea. In addition to insufficient pond level, openings can be delayed due to poor hydrodynamic conditions in the near shore ocean (e.g. wave height and direction that result in rapid in filling of the temporary inlet). Typically, pond water levels of one meter or greater above mean sea level are required, before a breach is attempted. Breaching of the pond is undertaken mainly as a means of controlling salinity levels in the pond and as a flood control measure. If the pond level is not periodically lowered, by breaching, groundwater table levels in the adjacent watershed rise sufficiently to impact basements of houses bordering the pond, and at very high pond levels, parcels may be effected by direct flooding.

Prior to opening the pond to Atlantic Ocean tidal exchange, Pond salinity is typically in the 6-10 ppt. range, due to dilution by groundwater infiltration during closure. After opening of the pond to tidal exchanges, the salinity rises to >20 ppt. A narrow shallow channel was created between the main basin of Lower Chilmark Pond (east) and the smaller basin of Upper Chilmark Pond (west). This channel provides for exchange of water between the basins of Chilmark Pond, flushed with the saline, low nutrient Atlantic Ocean waters during inlet openings. When the inlet is closed, water levels rise in the Chilmark Pond Embayment System above mean sea level and Pond waters that discharge to the ocean by seepage through the barrier beach. The absence of continuous tidal exchanges between the estuary and Atlantic Ocean allows for a greater increase in nitrogen level than in similar sized open estuaries per unit of watershed loading, which results in increased sensitivity of this system to watershed nitrogen loading compared to open tidal systems.

The Chilmark Pond Embayment System is a 178-241 acre (depending on the water level in the pond) coastal salt pond. The eastern sub-watersheds situated in sandy outwash discharge freshwater to the estuary via groundwater flows, while the western sub watersheds formed within the moraine support both surface water flows and groundwater discharge to the Chilmark Pond West basin. The Chilmark Pond West basin receives surfacewater inflow from Mill Brook and the Fulling Mill Brook. Both of these surfacewater inflows are primarily groundwater fed streams. The dry valleys that extend up into the outwash plain deposits contain unique habitat characterized by dry, sandy soils that are exposed to salt spray and frequent frosts in winter. For the MEP analysis, the Chilmark Pond estuarine system was partitioned into two general sub-embayment groups: the 1) the eastern main basin including Wades Cove and Gilberts Cove and 2) the tributary western sub-basin (Chilmark Pond-upper) (see Figure I-2).

The primary ecological threat to the Chilmark Pond Embayment System as a coastal resource is degradation resulting from nutrient enrichment. Nutrient enrichment generally occurs through increases in watershed nitrogen loading resulting from changing land uses (typically conversion of pine/oak forest to residential development) and/or reduced tidal exchanges with offshore waters. Although the watershed and the great pond can have periodic issues relative to bacterial contamination primarily within Wades Cove and Gilberts Cove, fecal coliform contamination does not generally result in ecological impacts, rather it is associated with public health concerns related with consumption of potentially contaminated shellfish. The primary impact of bacterial contamination is the closure of shellfish harvest areas, rather than the destruction of shellfish and other marine habitats. In contrast, increased loading of the critical eutrophying nutrient (nitrogen) to the Chilmark Pond System results in both habitat impairment and loss of the resources themselves. Within the watershed of this great salt pond, nitrogen loading has been increasing as land-uses have changed over the past 60 years. The nitrogen loading to this system, like almost all embayments in southeastern Massachusetts and the Islands, results primarily from on-site disposal of wastewater and fertilizer applications (residential and agricultural), and to a lesser extent stormwater flows. Nitrogen enrichment of all coastal embayments can only be managed through lowering inputs or increasing the rate of loss through tidal flushing. This is discussed in detail in Sections IV.1 and VI.

The Towns of Martha's Vineyard have been among the fastest growing towns in the Commonwealth over the past two decades and unlike the Town of Edgartown, which has a centralized wastewater treatment system with the site of discharge of its tertiary treated effluent being located in the Edgartown Great Pond watershed, the Town of Chilmark does not have such a wastewater system servicing the watersheds of Chilmark Pond or Tisbury Great Pond. Rather, treatment of wastewater within the watershed to the Chilmark Pond Embayment System is by privately maintained on-site septic systems for treatment and disposal of wastewater. As existing and likely increasing levels of nutrients impact the coastal embayments of the Towns of Chilmark, water quality degradation will accelerate, with further harm to valuable aquatic resources of the Town and the Island on the whole.

As the primary stakeholders to the Chilmark Pond Embayment System, the Town of Chilmark in collaboration with the Martha's Vineyard Commission (MVC) were among the first communities on Martha's Vineyard to become concerned over perceived degradation of their coastal embayments. Over the years, this local concern has led to the conduct of several studies (see Chapter II) of nitrogen loading to the system such as the Martha's Vineyard Commission developed Nutrient Loading and Management Plan of Chilmark, Menemsha and Squibnocket Ponds, (MVC, 2001). Key in this effort has been the Water Quality Monitoring Program of Martha's Vineyard's estuaries, spearheaded by the MVC and supported by private, municipal, county and state funds (most recently Massachusetts 604(b) grant program) with technical assistance by the Coastal Systems Program at SMAST-UMD. This effort provides the quantitative watercolumn nitrogen data (2003-2005, 2010, 2012) required for the implementation of the MEP's Linked Watershed-Embayment Approach used in the present study.

Since the initial results of the Water Quality Monitoring Program and the land-use studies indicated that parts of the Chilmark Pond system were presently impaired by land-derived nitrogen inputs, the Towns and Martha's Vineyard Commission (MVC) undertook additional site-specific data collection that has served to support MEP's ecological assessment and modeling effort. The common focus of the Town of Chilmark - MVC work related to the Chilmark Pond System has been to gather site-specific data on the current nitrogen related water quality throughout the estuary and determine its relationship to watershed nitrogen loads (e.g. Martha's Vineyard Commission Nutrient Load to Chilmark Pond, 2001 {updated in 2010}). The multi-year water quality monitoring effort has provided the baseline information required for calibrating and verifying the water quality model linking upland loading, periodic tidal flushing, and estuarine water quality. The MEP effort builds upon the Water Quality Monitoring Program results and includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the restoration of this embayment system. These critical nitrogen threshold levels and the link to specific ecological criteria form the quantitative basis for the nitrogen loading targets necessary for nitrogen management plans and the development of cost-effective alternatives for restoration of habitat impaired by nitrogen enrichment needed by the Town of Chilmark.

While the completion of this complex multi-step process of rigorous site-specific scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff and volunteers over many years and members of the Martha's Vineyard Commission. The modeling tools developed as part of this program provide the quantitative information necessary for the Town of Chilmark to develop and evaluate the most cost effective nitrogen management alternatives to restore this valuable coastal resource which is currently being degraded by nitrogen overloading. It is important to note that the Chilmark Pond System and its associated watershed have been significantly altered by human activities over the past ~100 years. As a result, the present nitrogen "overloading" appears to result partly from alterations to its ecological systems. These alterations subsequently affect nitrogen loading within the watershed and influence the degree to which nitrogen loads impact the estuary. Therefore, restoration of this system should focus on managing nitrogen through both management of nitrogen loading within the watershed, restoration/management of processes which serve to lessen the amount or impact of nitrogen entering the estuary and inlet management to enhance the rate of nitrogen removal from the estuary via tidal flushing.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

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

The primary nutrient causing the increasing impairment of the Commonwealth's coastal embayments is nitrogen and the primary sources of this nitrogen are wastewater disposal, fertilizers, and changes in the freshwater hydrology associated with development. At present there is a critical need for state-of-the-art approaches for evaluating and restoring nitrogen sensitive and impaired embayments. Within Southeastern Massachusetts alone, almost all of the municipalities (as is the case with the Town of Chilmark) 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 Martha's Vineyard Commission (MVC) and the Cape Cod Commission (CCC) have undertaken the task of providing a quantitative tool for watershed-embayment management for communities throughout Southeastern Massachusetts and the Islands.

The Massachusetts Estuary Project is founded upon science-based management. The Project is using a consistent, state-of-the-art approach throughout the region's coastal waters and providing technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the Massachusetts Estuaries Project is to provide the MassDEP and municipalities with technical guidance to support policies on nitrogen loading to embayments. In addition, the technical reports prepared for each embayment system will serve as the basis for the development of Total Maximum Daily Loads (TMDLs) for those estuarine systems that are presently impaired by nitrogen enrichment or which will become impaired as build-out of their watershed continues. Development of TMDLs is required pursuant to Section 303(d) of the Federal Clean Water Act. TMDLs must identify sources of the pollutant of concern (in this case nitrogen) from both point and non-point sources, the allowable load to meet the state water quality standards and then allocate that load to all sources taking into consideration a margin of safety, seasonal variations, and several other factors. In addition, each TMDL must contain an outline of an implementation plan. For this project, the MassDEP recognizes that there are likely to be multiple ways to achieve the desired goals, some of which are more cost effective than others and therefore, it is extremely important for each Town to further evaluate potential options suitable to their community. As such, MassDEP will likely be recommending that specific activities and timelines be further evaluated and developed by the Towns (sometimes jointly) through the Comprehensive Wastewater Management Planning process.

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

The major MEP nitrogen management goals are to:

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

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

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

The Linked Model has been applied for watershed nitrogen management in approximately 60 embayments throughout Southeastern Massachusetts. In these applications it has become clear that the Linked Model Approach’s greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing “what if” scenarios for evaluating watershed nitrogen management options.

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

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

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

- rate of N recycling within embayment
- D.O record
- Macrophyte survey
- Infaunal survey

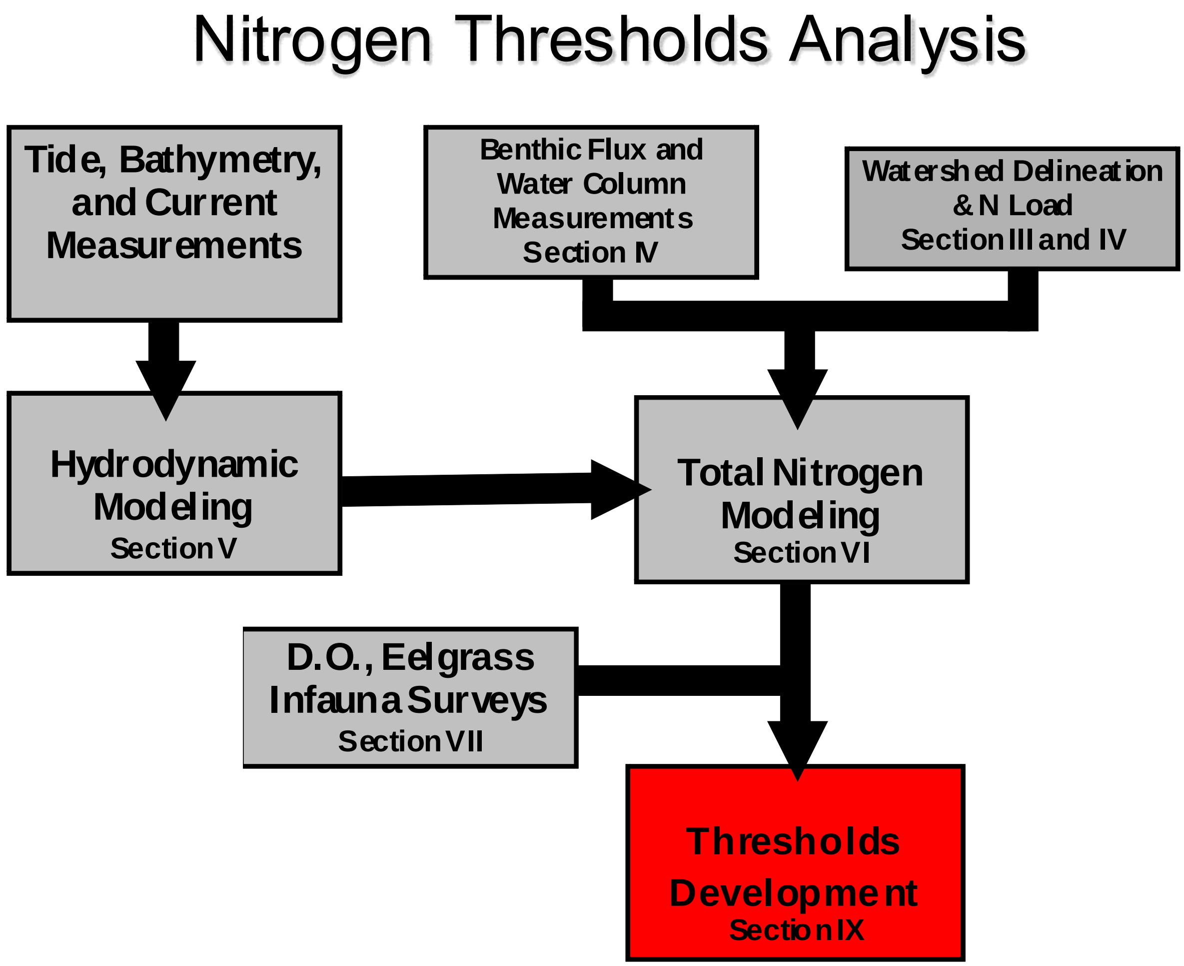


Figure I-3. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach.

I.2 NUTRIENT LOADING

Surface and groundwater flows are pathways for the transfer of land-sourced nutrients to coastal waters. Fluxes of primary ecosystem structuring nutrients, nitrogen and phosphorus, differ significantly as a result of their hydrologic transport pathway (i.e. streams versus groundwater). In sandy glacial outwash aquifers, such as in the watershed to the Chilmark Pond System, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer minerals (Weiskel and Howes 1992). Since even Martha's Vineyard and Cape Cod "rivers" are primarily groundwater fed, watersheds tend to release little phosphorus to coastal waters. In contrast, nitrogen, primarily as plant available nitrate, is readily transported through oxygenated groundwater systems on Cape Cod (DeSimone and Howes 1998, Weiskel and Howes 1992, Smith *et al.* 1991) and Martha's Vineyard. The result is that terrestrial inputs to coastal waters tend to be higher in plant available nitrogen than phosphorus (relative to plant growth requirements). However, coastal estuaries tend to have algal growth limited by nitrogen availability, due to their flooding with low nitrogen coastal waters (Ryther and Dunstan 1971). The estuarine reaches within the Chilmark Pond Embayment System follow this general pattern, with the Redfield Ratio (N/P) averaging <16, but with total dissolved inorganic nitrogen levels quite low (2 uM) indicating that addition of nitrogen would have a stimulatory effect of plant production.

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

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

Protection and restoration of coastal embayments from nitrogen overloading has resulted in a focus on determining the assimilative capacity of these aquatic systems for nitrogen. While this effort is ongoing (e.g. USEPA TMDL studies), southeastern Massachusetts and the Islands has been the site of intensive efforts in this area (Eichner et al., 1998, Costa et al., 1992 and in press, Ramsey et al., 1995, Howes and Taylor, 1990, and the Falmouth Coastal Overlay Bylaw, MVC Water Quality Policy). 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 Chilmark Pond System monitored by the Martha's Vineyard Commission and the Town of Chilmark. The Water Quality Monitoring Program along with site-specific habitat quality data (D.O., eelgrass, phytoplankton blooms, benthic animals) was utilized to refine general nitrogen thresholds typically used by the Cape Cod Commission, Buzzards Bay Project, and Massachusetts State Regulatory Agencies.

Unfortunately, almost all of the estuarine reaches within the Chilmark Pond System are near or slightly beyond their ability to assimilate additional nutrients without impacting their ecological health. Nitrogen levels are elevated throughout this great salt pond and eelgrass beds have been lost over the past ~50 years as indicated by the MassDEP Eelgrass Mapping Program and as confirmed by local officials and citizens and the MEP Technical Team during the summer and fall of 2005. Nitrogen related habitat impairment within the Chilmark Pond system is relatively uniform and consistent with the nitrogen levels. The result is that nitrogen management of the primary sub-embayments to the Chilmark Pond system is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed "eutrophication" and in certain instances can occur naturally over long periods of time. When the nutrient loading is rapid and primarily from human activities leading to changes in a coastal watershed, nutrient enrichment of coastal waters is termed "cultural eutrophication". Although the influence of human-induced changes has increased nitrogen loading to this embayment system and contributed to its decline in ecological health, the Chilmark Pond basins, like those analyzed by the MEP in Tisbury Great Pond and Edgartown Great Pond, are especially sensitive to nitrogen inputs, because of the lack of continuous tidal exchange. The quantitative role of the discontinuous tidal exchange of this system, as a natural process, was also considered in the MEP nutrient threshold analysis. As part of future restoration efforts, it is important to understand that it may not be possible to turn each embayment into a "pristine" system.

I.3 WATER QUALITY MODELING

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

The MEP water quality evaluation examined the potential impacts of nitrogen loading into the Chilmark Pond Embayment System, including the tributary sub-embayments of Wades Cove, Gilberts Cove and Lower Chilmark Pond (east) and Upper Chilmark Pond (west) basins. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents during breaching events and water elevations was employed for the system. Once the hydrodynamic properties of each estuarine basin were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates.

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 MVC/MEP refined watershed and subwatershed delineations are based on 1) water table contours measured in a few locations (*e.g.*, Wilcox, 1996) and modeled throughout the outwash plain and 2) USGS topographic maps in the western moraine. Almost all nitrogen entering the Chilmark Pond System is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Atlantic Ocean source waters and throughout the Chilmark Pond system were taken from the Water Quality Monitoring Program (a coordinated effort between the Town of Chilmark, Martha's Vineyard Commission and the Coastal Systems Program at SMAST). Measurements of salinity and nitrogen and salinity distributions throughout the estuarine waters of the system (2000-2012) were used to calibrate and validate the water quality model (under existing loading conditions).

I.4 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Chilmark Pond Embayment System for the Town of Chilmark. A review of existing water quality studies is provided (Section II). The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Sections III and IV. In addition, nitrogen input parameters to the water quality model are described. Since benthic flux of nitrogen from bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Section IV). Nitrogen loads from the watershed and sub-watersheds surrounding the estuary were derived from the Martha's Vineyard Commission data and offshore water column nitrogen values were derived from an analysis of monitoring stations in the Atlantic Ocean (Section IV and VI). Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data (conducted by municipalities) as discussed in Section VI. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling, as well as an analysis of how the measured nitrogen levels correlate to observed estuarine water quality are described in Section VI. This analysis includes modeling of current conditions, conditions at watershed build-out, and with removal of anthropogenic nitrogen sources. In addition, an ecological assessment of the component sub-embayments was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of the Pond in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined threshold for restoration of the 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 this system. Finally, any additional analyses of the Chilmark Pond System beyond the standard suite offered by the MEP may be undertaken relative to potential alterations of circulation and flushing, including an analysis to identify hydrodynamic restrictions and an examination of dredging/breach options to improve nitrogen related water quality. The results of the nitrogen modeling for any additional scenario, should they be undertaken, are typically presented in Section IX.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

Nutrient additions to aquatic systems cause shifts in a series of biological processes that can result in impaired nutrient related habitat quality. Effects include excessive plankton and macrophyte growth, which in turn lead to reduced water clarity, organic matter enrichment of waters and sediments. This has the concomitant effect of increased rates of oxygen consumption and periodic depletion of dissolved oxygen, especially in bottom waters, as well as limiting the growth of desirable species such as eelgrass. Even without changes to water clarity and bottom water dissolved oxygen, the increased organic matter deposition to the sediments generally results in a decline in habitat quality for benthic infaunal communities (animals living in the sediments). This habitat change causes a shift in infaunal communities from high diversity deep burrowing forms (which include economically important species), to low diversity shallow dwelling organisms. This shift alone causes significant degradation of the resource and a loss of productivity to both the local shell fisherman and to the sport-fishery and offshore fin fishery. Both the sport-fishery and the offshore fin fishery are dependent upon highly productive estuarine systems as a habitat and food resource during migration or during different phases of their life cycles. 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 Chilmark Pond Embayment System (inclusive of Upper Chilmark Pond), the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if nitrogen levels are controlled by source controls or enhanced tidal flushing, then eutrophication is controlled. This approach has been formalized through the development of tools for predicting nitrogen loads from watersheds and the concentrations of water column nitrogen that may result. Additional development of the approach generated specific guidelines as to what is to be considered acceptable water column nitrogen concentrations to achieve desired water quality goals (e.g., see Cape Cod Commission 1991, 1998; Howes et al. 2002).

These tools for predicting loads and concentrations tend to be generic in nature, and overlook some of the specifics for any given water body. The present Massachusetts Estuaries Project (MEP) study focuses on linking water quality model predictions, based upon watershed nitrogen loading and embayment recycling and system hydrodynamics, to actual measured values for specific nutrient species. The linked watershed-embayment model is built using embayment specific measurements, thus enabling calibration of the prediction process for specific conditions in each of the coastal embayments of southeastern Massachusetts, including the estuaries and salt ponds of Martha's Vineyard such as Chilmark Pond presently and Tisbury Great Pond, Edgartown Great Pond, Lagoon Pond, Farm Pond and Sengekontacket, all of which have been previously evaluated by the MEP. As the MEP approach requires substantial amounts of site specific data collection, part of the program is to review previous data collection and modeling efforts. These reviews are both for purposes of "data mining" and to gather additional information on an estuary's habitat quality or unique features.

A number of studies relating to nitrogen loading and water quality have been conducted within the Chilmark Pond System over the past two decades. Among these studies, several contained information of sufficient quality that it could be used to support the MEP modeling and assessment of this estuary and these are described briefly below.

Nutrient Loading to Chilmark Pond (2001, updated 2010): This report was prepared by the Martha's Vineyard Commission and submitted to the Massachusetts Department of Environmental protection (MassDEP) and the US EPA in 2001. The loading study was subsequently up-dated in 2010. Specifically, Mr. William Wilcox (MVC Water Resources Planner at the time of the study) designed the project and served as principal investigator, author, and MVC project quality assurance officer. The study was completed through a contractual arrangement with the University of Massachusetts Cooperative Extension. Additionally, scientists currently from the Coastal Systems Program at the UMASS School for Marine Science and Technology and presently involved with the MEP performed the chemical analyses in support of this 2001 Chilmark Pond loading study when SMAST was originally the Center for Marine Science & Technology. This study was undertaken to assess the potential impact of residential development in the watersheds of three Vineyard ponds inclusive of Chilmark Pond. The components of the study that were used to make the assessment included: the amount of residential development expected in each watershed, the volume of each pond, its tidal circulation and the desired water quality goal.

As summarized in the report, the primary approach to managing water quality in Chilmark Pond is by the number and timing of openings of the temporary tidal inlet. The pond is breached to the Atlantic Ocean by excavating a trench through the barrier beach at intervals of about 4 months. Typically the pond will reach heights of over one meter above mean sea level before it is breached. The breaching is done to maintain salinity in the pond as well as to limit flooding of septic systems and basements in houses bordering the pond. The regular, man-made breaching of the system leads to a somewhat variable but always brackish main basin of the Pond. If the system were not periodically opened to the ocean, the system would have much wider swings in salinity perhaps from nearly fresh to nearly ocean salinity (after a storm produced a breach through the barrier beach) which would probably cause catastrophic loss of fauna.

In the 2001 MVC nitrogen loading analysis for Chilmark Pond, it was determined that the estimated discharge of nitrogen to Chilmark Pond was 3,400 to 3,800 kilograms/year. Moreover, projected nitrogen loading from future development in the Chilmark Pond watershed ranged from 4,946 to 6,551 kilograms of nitrogen per year (13.6 to 17.9 kg/day). According the MVC nitrogen loading analysis, the data collected indicates that in Chilmark Pond nitrogen is generally the limiting nutrient, usually during the growing season. However, based on data from other Great Ponds on Martha's Vineyard and the limited amount of data collected from Chilmark Pond over two growing seasons, according to the MVC, it is highly likely that in Chilmark Pond during the winter or at high pond in spring, phosphorus becomes the nutrient that limits pond productivity (Wilcox, 1999). The report does go on to stress that nutrient management should focus on what happens during the summer when poor water quality can damage the ecosystem and at that time, nitrogen is the limiting nutrient.

In 2001, Chilmark Pond already showed symptoms of excess phytoplankton in the system at the present day (2001) loading rate. According to the 2001 nitrogen loading analysis for Chilmark Pond, the N-loading limit was set at 3,802 kilograms per year and that limit could not be easily reached under the projected loading at build-out. One of the main findings of the 2001 MVC loading analysis was that the projected total nitrogen loading for this system ranged from 4946 to 6551 kilograms per year and that wetland reduction(presumably via natural attenuation of nitrogen) would account for 100 to 200 kilograms reduction per year. As such, the net loading to the pond would be 4846 to 6351 kilograms. According to the MVC analysis, Chilmark Pond was well over its nitrogen loading threshold, e.g. the loading limit for "Good Quality" waters is

2235 kg N yr^{-1}, the "Reduced Quality" at a 25 day flushing interval is 3802 kg N yr^{-1}. In 2001, the Lower Pond demonstrated some symptoms of nitrogen loading today at the current (2001) loading of ~ 3700 kilograms per year, therefore it was recommended that steps be taken to try to improve the circulation in the system by extending the lifetime of openings to the Atlantic.

Water Quality Study of Chilmark Pond (2004): This report was prepared by the Martha's Vineyard Commission and the overall objective of the investigation was to establish existing water quality conditions in the pond as a baseline for future investigations as well as to meet the three year minimum baseline water quality data requirement for inclusion into the Massachusetts Estuaries Project. The 2004 "project" was funded by the Massachusetts 604(b) Grant program continued to build the MVC water quality database for seven coastal ponds, specifically: Chilmark Pond, Sengekontacket Pond, Farm Pond, Lake Tashmoo, Cape Pogue Pond, Pocha Pond and Lagoon Pond.

As described in the 2004 water quality summary report, in general Chilmark Pond does not receive the same daily tidal exchange that the other six systems do as a result of their having permanent inlets allowing for tidal exchange with the ocean. Chilmark Pond is only periodically opened to the ocean, drains and becomes tidal for only a short time before the barrier beach fills the inlet and ends tidal exchange. Interestingly, the opening of Chilmark Pond in the summer period during 2004 was different than in usual years. The spring inlet to the ocean remained functional throughout the sampling period leading to higher salinity and better water quality than seen in 2003. In response and as reported by the MVC, the 2004 salinity values were well above 20 parts per thousand and some strong surface stratification set up at times with much lower salinity at the surface. Chlorophyll showed a clear trend increasing from the eastern half of the Pond toward Doctor's Creek, the input from the western, freshwater pond. Chlorophyll also declined from a high in the north end of Wade's Cove into the main basin of Chilmark Pond. Total organic nitrogen followed a similar pattern with increasing values into the western half of the Pond. According the MVC, this parameter (TON) was elevated well above desirable levels throughout the summer despite the Pond being tidal. Dissolved inorganic nitrogen concentrations were low in 2004. In part, this may have resulted from below normal rainfall for the year (2004) being just over two inches less than average for the June through August period.

MVC/Town of Chilmark Water Quality Monitoring Program (2000-2012): A significant record of baseline water quality throughout the Chilmark Pond System has been developed over the past 15 years, in large part due to the efforts by the Martha's Vineyard Commission. The Martha's Vineyard Commission partnered with SMAST-Coastal Systems Program scientists in 1995 to develop and implement a nutrient related water quality monitoring program of the estuaries of Martha's Vineyard, inclusive of Chilmark Pond in the Town of Chilmark. Sample analysis was conducted by the Coastal Systems Analytical Facility at SMAST-UMD. For the Chilmark Pond system as well as the other estuarine systems of Martha's Vineyard, the focus of the water quality monitoring effort has been to gather site-specific data on the current nitrogen related water quality throughout the estuarine reach of a given system to support assessments of habitat health. This baseline water quality data are a prerequisite to entry into the MEP and the conduct of its Linked Watershed-Embayment Approach. The water quality monitoring program was initiated in 1995 and along the way supported by funds obtained from the Massachusetts 604B Grant Program (1999). It should also be noted that the baseline water quality monitoring program was also supplemented with very specific monitoring supported by volunteers from the Chilmark Pond Association and targeted to the pond openings in order to refine the hydrodynamic modeling of the pond. Throughout the water quality monitoring period, sampling was undertaken between 4 and 6 times per summer between the months of June and

September. The MVC/Town based Water Quality Monitoring Program for Chilmark Pond developed the baseline data from sampling stations distributed throughout the main basin as well as the major tributary coves (Figure II-1). As remediation plans for this and other various systems on Martha's Vineyard are implemented throughout the towns, monitoring will have to be resumed or continued to provide quantitative information to the towns relative to the efficacy of remediation efforts. As Chilmark Pond is only periodically open to tidal exchange, continued monitoring is essential to provide the necessary feedback on the "success" of the openings and the need to refine the approach or schedule for breaching the barrier beach as a means of managing nutrients within this embayment.

Implementation of the MEP Linked Watershed-Embayment Approach incorporates the quantitative water column nitrogen data gathered by the Water Quality Monitoring Program and watershed and embayment data collected by MEP Technical Staff. The MEP effort also builds upon previous watershed delineation and land-use analyses as well as eelgrass surveying by the MassDEP Eelgrass Mapping Program and MEP Technical Staff. This information is integrated with MEP higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Chilmark Pond Embayment System. The MEP has incorporated all appropriate data from previous studies to enhance the determination of nitrogen thresholds for the Chilmark Pond System and to reduce costs of restoration for the Town of Chilmark.

Regulatory Assessments of Chilmark Pond Resources - The Chilmark Pond System (inclusive of Eastern and Western basins) contains a variety of natural resources of value to the citizens of Chilmark and Martha's Vineyard as well as to the Commonwealth. As such, over the years surveys have been conducted to support protection and management of these resources. The MEP gathers the available information on these resources as part of its assessment, and presents them here (Figures II-2 through II-5) for reference by those providing stewardship for this estuary. For the Chilmark Pond Embayment System these include:

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

Figure II-1. MVC/Town of Chilmark Water Quality Monitoring Program. Estuarine water quality monitoring stations sampled by the MVC/SMAST/Town and volunteers from the Chilmark Pond Association.

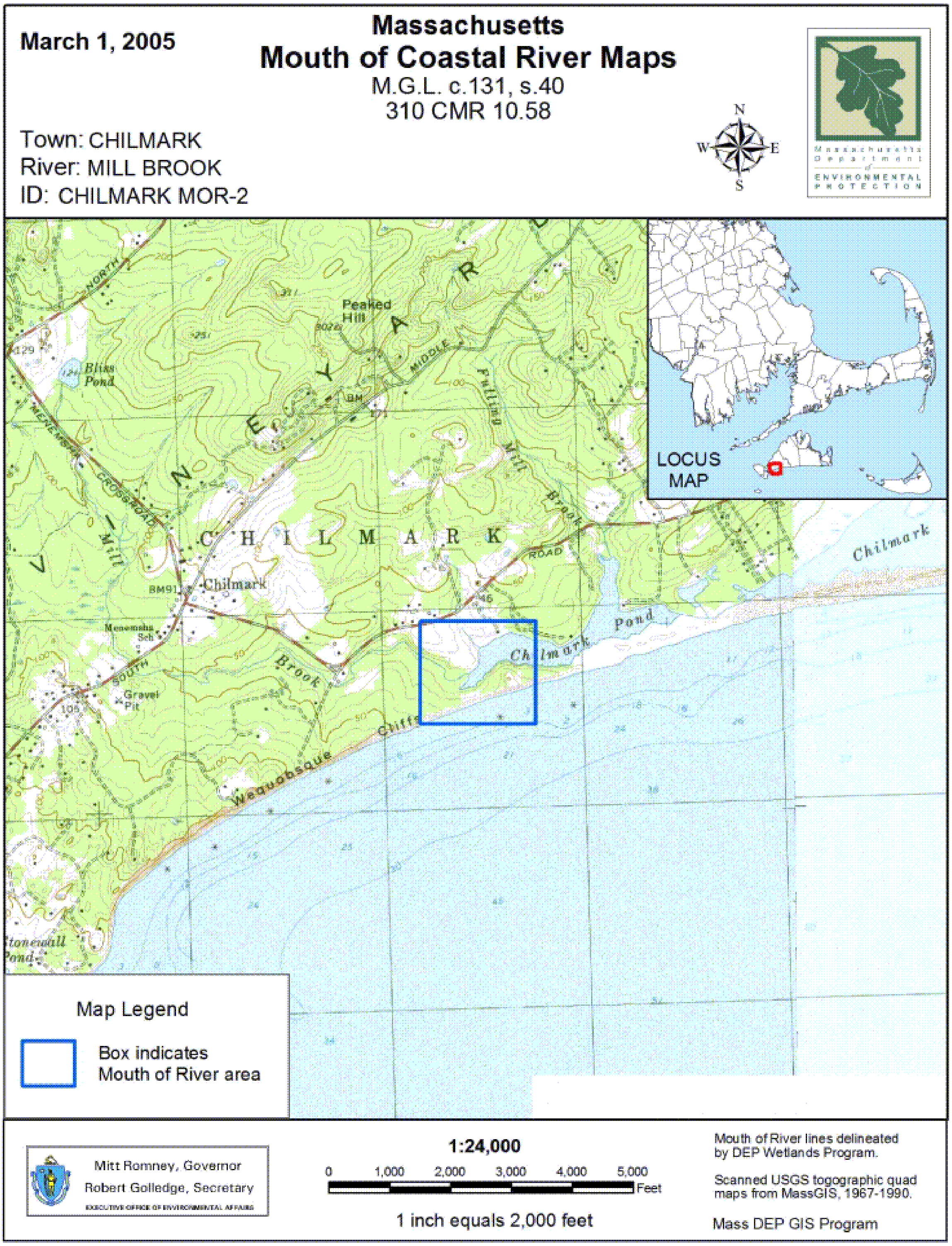


Figure II-2. Regulatory designation for the mouth of “River” under the Massachusetts River Act (MassDEP). Upland adjacent the "river front" inland of the mouth of the river has restrictions specific to the Act.

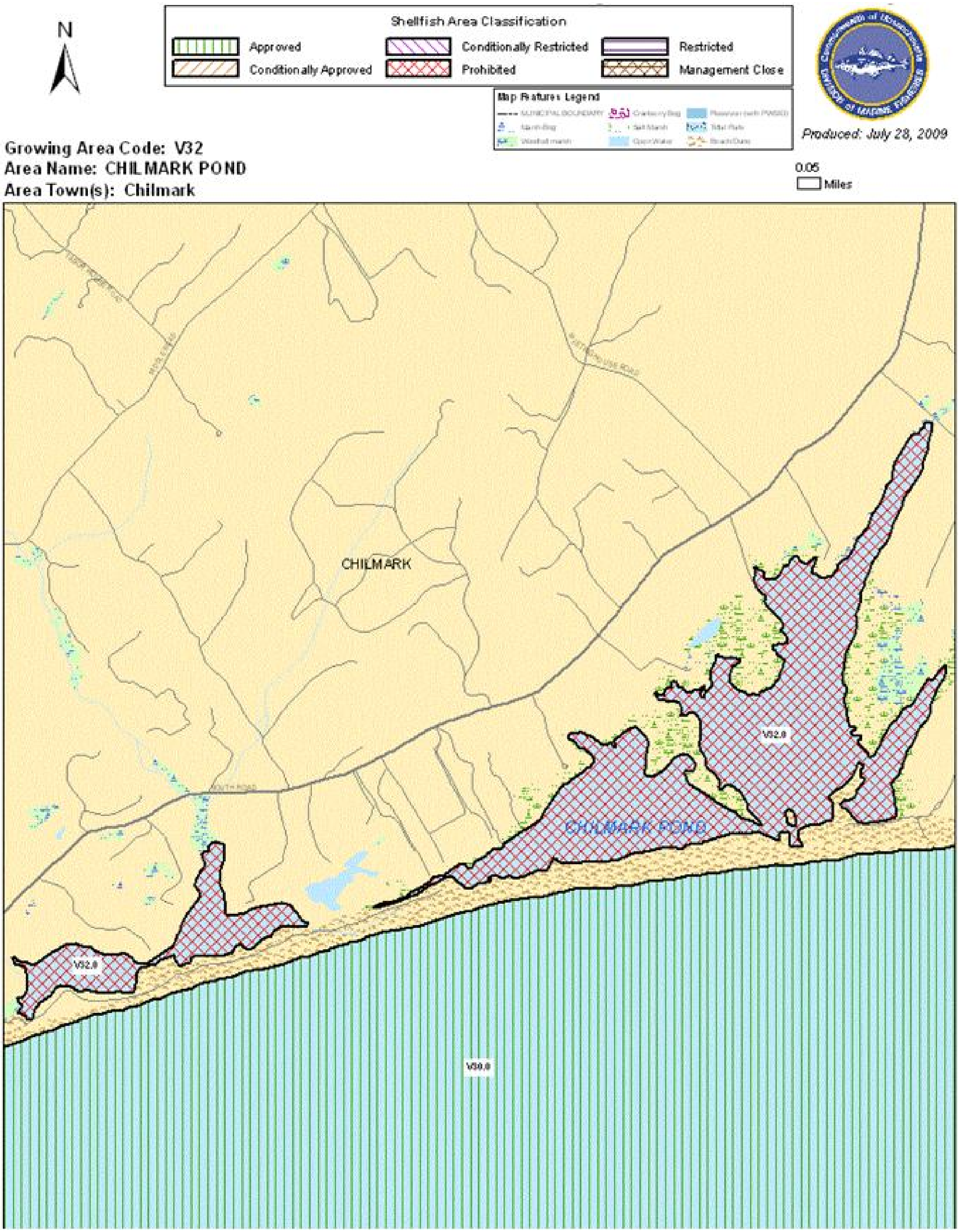


Figure II-3. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas. However, areas dominated by wetlands with persistent fecal coliform levels >14 cfu per 100 mL may be prohibited to shellfishing until the cause of the contamination (frequently wildlife and birds) is documented.

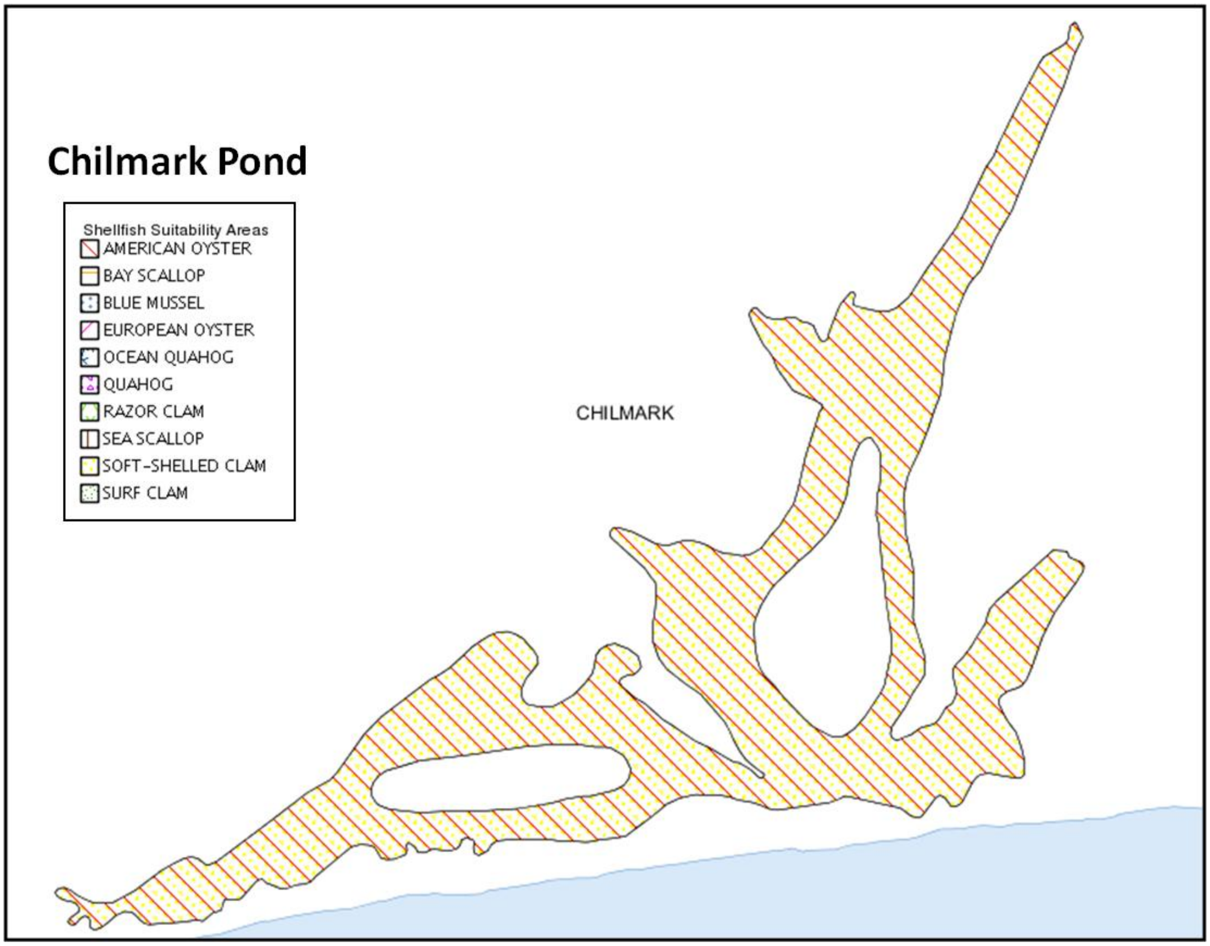


Figure II-4 Location of shellfish suitability areas within the Chilmark Pond Embayment System as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean that a shellfish population is "present" or that harvest is allowed.

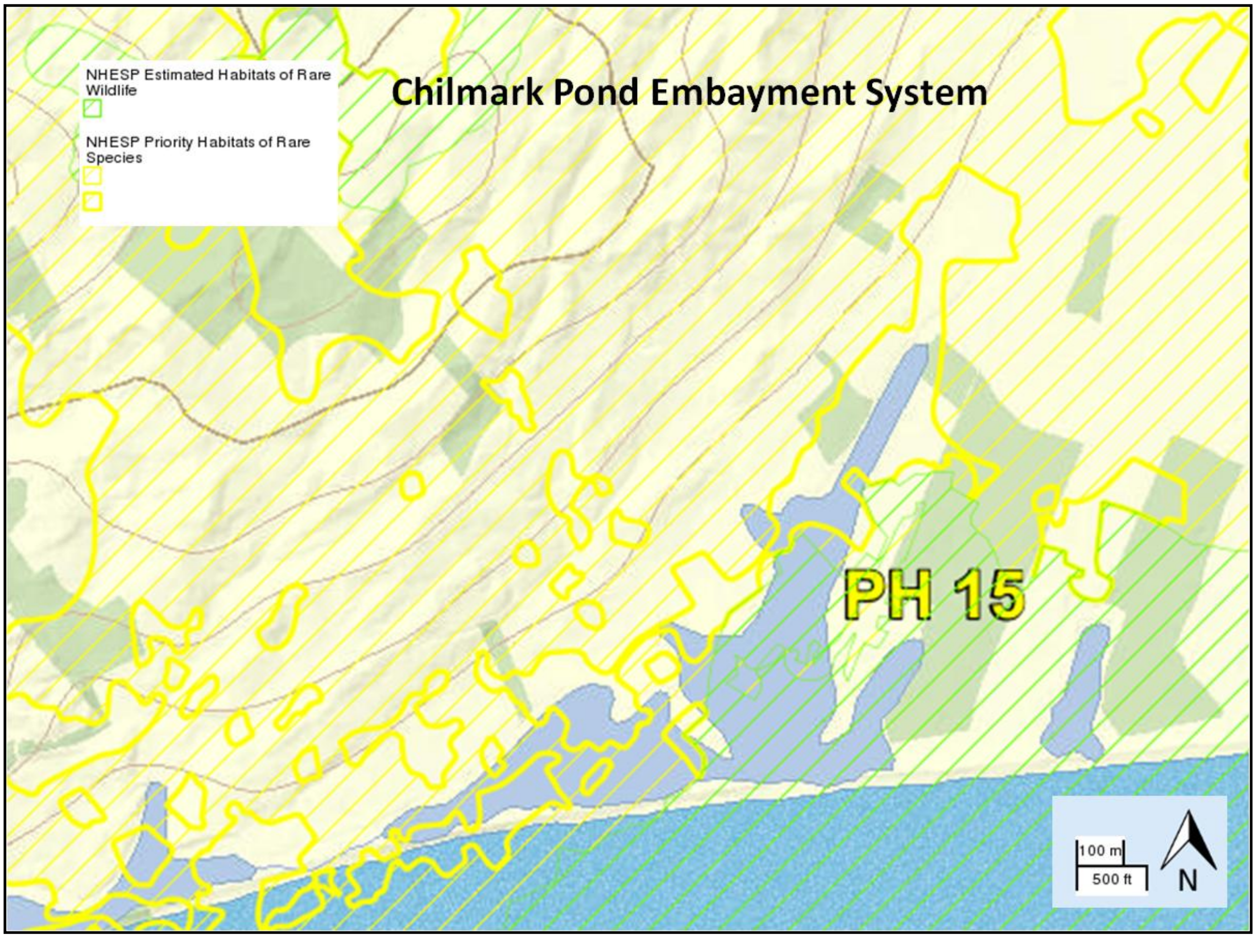


Figure II-5. Estimated Habitats for Rare Wildlife and State Protected Rare Species within the Chilmark Pond Estuary as determined by the Massachusetts Natural Heritage and Endanger Species Program (NHESP).

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The island of Martha's Vineyard is located along the southern edge of late Wisconsinan glaciation (Oldale and Barlow, 1986). As such, the geology of the island is largely composed of glacial outwash plain and moraine with reworking of these deposits by the ocean that has occurred since the retreat of the glaciers. The island was located between the Cape Cod Bay and Buzzards Bay lobes of the Laurentide ice sheet. As such, the areas where the glacial ice lobes moved back and forth with warming and cooling of the climate are dominated by moraine deposits and these moraines are located along the Nantucket Sound/eastern and Vineyard Sound/western sides of the island. These moraines generally consist of unsorted sand, clay, silt, till, and gravel with the western moraine having the more complex geology (*i.e.*, composed of thrust-faulted coastal plain sediments interbedded with clay, till, sand, silt and gravel) and the eastern moraine having more permeable materials overlying poorly sorted clay, silt, and till (Delaney, 1980).

The relatively porous deposits that comprise most of the Vineyard outwash plain and the eastern moraine create a hydrologic environment where watersheds 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). Delaney (1980) and subsequent characterizations have indicated that these characteristics also apply to the eastern moraine. Characterizations of the western moraine are very limited and are likely to be very site-specific given the geologic mix of the moraine. The western portions of the watershed and embayment basins are within the porous sandy outwash plain that extends well to the east of Chilmark Pond and dominates the central region of Martha's Vineyard. The Chilmark Pond system (Lower and Upper) and its watershed are situated within the western Martha's Vineyard moraine to the west and the sandy outwash plain to the east, with the division being along the long axis of Wades Cove and extending inland. Previous watershed delineations within the moraine have been based mainly on surface topography and within the outwash plain based on groundwater elevations (personal communication, Bill Wilcox, MVC). The streams within the watershed are located within the moraine based aquifer areas.

Groundwater modeling of Martha's Vineyard has largely been confined to the outwash plain portions of the island. Regional groundwater contours created for the United States Geological Survey (USGS) regional water table map do not extend into the western moraine (Delaney, 1980). The study grid for the regional MODFLOW groundwater model of the Island originally developed by Whitman and Howard (1994) and updated by EarthTech, Inc. is tilted to avoid the western moraine and includes a no-flow boundary at the western edge of the grid. The Martha's Vineyard Commission (MVC) created watershed delineations to most of the estuaries in the western moraine (*e.g*., Chilmark Pond, Menemsha Pond, Squibnocket Pond) largely based on surface topography (personal communication, Bill Wilcox, MVC). With the collection of MEP streamflow information, project staff has had the chance to re-review these delineations and evaluate/delineate internal stream subwatersheds to these estuaries.

III.2 CHILMARK POND CONTRIBUTORY AREAS

The Chilmark Pond watershed and subwatershed delineations are based on: 1) USGS topographic maps in the western moraine, 2) MEP streamflows, 3) MassDEP wetland characterizations (MassDEP, 2009), 4) groundwater elevations where available in the sandy outwash aquifer areas and 5) best professional judgment. The outer boundary of the Chilmark Pond watershed is based on the MVC delineation, which was created based on topographic inspection. Analysis focuses on determining the pattern of lines of local maximum elevation upon a US Geological Survey 1:25,000 topographic quadrangle map, with watershed divides based upon the tendency of surface water and groundwater to flow downhill perpendicular to the topographic contour lines. Divides drawn upon topographic maps can be confirmed by observing general patterns of groundwater flow and surface water flow during rainfall or snow melt or by measuring the flow of water in streams over a hydrologic cycle as was done by the MEP for this investigation. The northeastern edge of the watershed abuts the Tiasquam River watershed, which is a subwatershed to Tisbury Great Pond. This subwatershed was confirmed through MEP stream monitoring over the 2005-2006 hydrologic year. The eastern edge of the watershed abuts Black Point Pond, which is also associated with Tisbury Great Pond. Both of these subwatersheds are documented in the MEP Tisbury Great Pond nitrogen threshold report (Howes, *et al.*, 2013). The overall Chilmark Pond watershed is situated in the western portion of Martha's Vineyard, is bounded by the Atlantic Ocean to the south and is completely contained within the Town of Chilmark (Figure III-1).

In order to develop the interior stream watersheds, MEP staff initially completed topographic watersheds but found that these were inconsistent with the measured MEP streamflows developed over the 2005-2006 hydrologic year (see Section IV.2). Staff then re-reviewed the USGS topography, incorporated the DEP wetland information, and used best professional judgment regarding groundwater contours. This analysis combined with the island-specific 28.7 inches/year of recharge produced a good match with the measured MEP streamflows. The annual recharge rate is largely based on review of the relationship between recharge and precipitation rates used in Cape Cod groundwater modeling (Walter and Whealan, 2005). The USGS used a recharge rate of 27.25 in/yr for calibration of Cape Cod groundwater models to match measured water levels. The Cape Cod recharge rate is 61% of the estimated average 44.5 in/yr of precipitation on the Cape. Precipitation data collected by the National Weather Service at Edgartown on Martha's Vineyard since 1947 has an average over the last 20 years of 46.9 in/yr (http://www.mass.gov/dcr/waterSupply/rainfall/precipdb.htm). If the Cape Cod relationship between precipitation and recharge is applied to the average Martha's Vineyard precipitation rate, the estimated recharge rate on Martha's Vineyard is 28.7 in/yr. The resulting Chilmark Pond subwatersheds were developed with assistance from the MVC staff.

The overall MEP watershed area to the Chilmark Pond Embayment System is 3,137 acres (Table III-1). Based on available previous reports, this delineation is the second watershed delineation completed for the Chilmark Pond Embayment System. Figure III-2 compares the delineation completed under the current effort with a previous 2001 delineation developed by the Martha's Vineyard Commission (MVC, 2001). The MVC delineation utilized topographic review, but did not have streamflow measurements. The MVC watershed area is only ~1% larger (37 acres) than the MEP watershed area. Given that the same watershed delineation method was used, it is not surprising that the watershed areas are essentially the same.

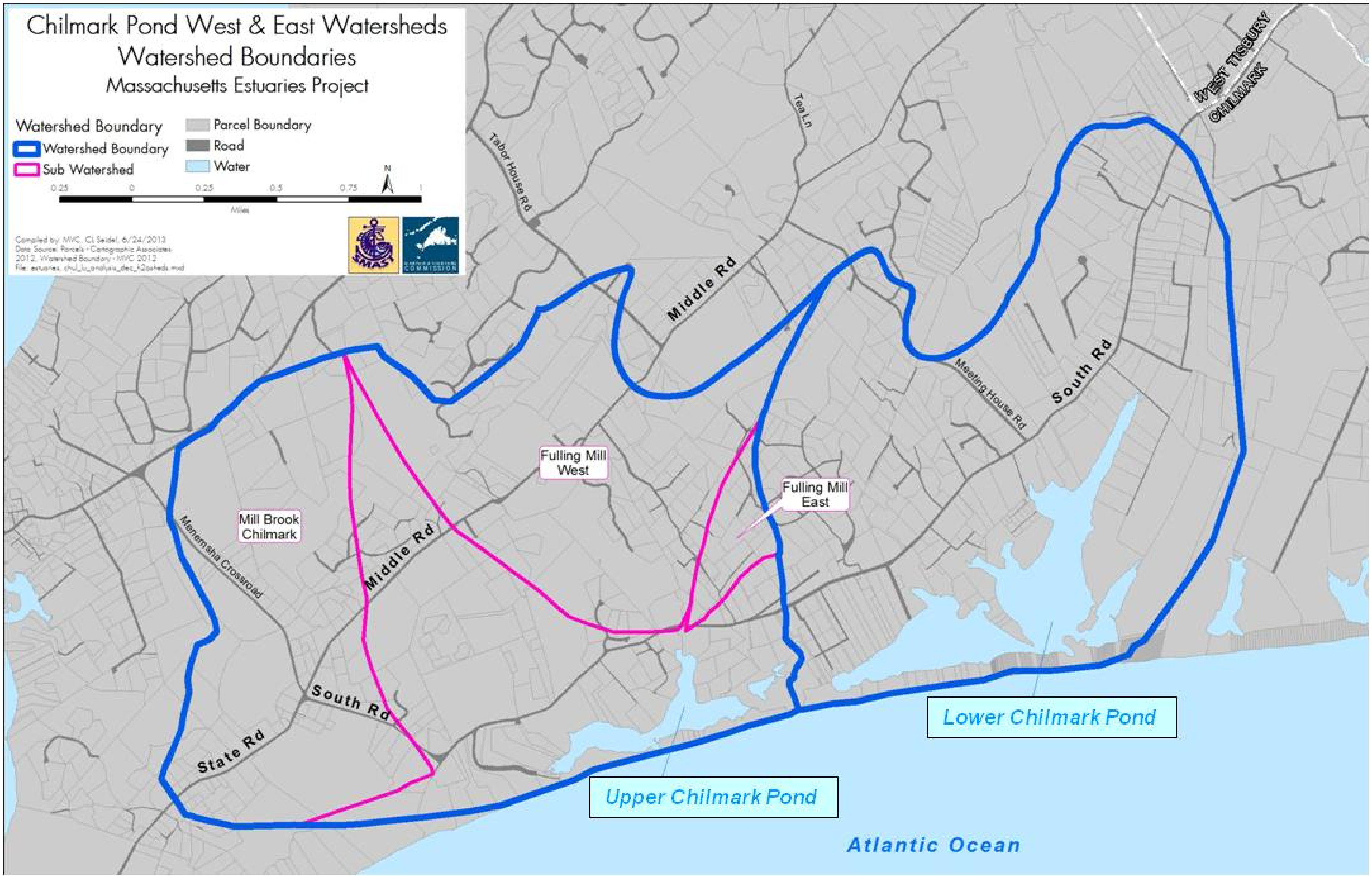


Figure III-1. Watershed and sub-watershed delineations for the Chilmark Pond Embayment System (Upper and Lower). Sub-watersheds are delineated to functional aquatic sub-units in the land-use nitrogen loading and water quality models (see Section VI) and stream gauge locations (see Section IV). The watershed is completely contained within the Town of Chilmark.

Table III-1. Daily groundwater discharge from each of the sub-watersheds to the Chilmark Pond Estuary. Chilmark Pond - East is the embayment basins of Lower Chilmark Pond, Wades Cove and Gilbert's Cove; Chilmark Pond - West (upper) is now an isolated freshwater pond complex with stream inflows.

Watershed	Watershed #	Watershed Area (acres)	Discharge	
			m^3/day	ft^3/day
Estuarine				
Chilmark Pond - lower	1	1,238	10,008	353,430
Freshwater				
Chilmark Pond - upper	2	616	4,982	175,922
Fulling Mill East	3	61	492	17,374
Fulling Mill West	4	605	4,889	172,659
Mill Brook	5	617	4,983	175,970
TOTAL		3,137	25,354	895,355

NOTE: Discharge rates are based on 28.7 inches per year of recharge, which is based on average precipitation recorded at Edgartown over the past 20 years.

Based on the review of the available data, MEP Technical Team staff is confident that the delineation in Figure III-1 is accurate and an appropriate basis for completion of the linked watershed-embayment model for the Chilmark Pond Embayment System. Figure III-1 shows the overall Chilmark Pond MEP watershed and the five subwatersheds, including watersheds to Mill Brook, Fulling Mill East stream and Fulling Mill West Stream. The watershed areas and the island-specific recharge rate were also used to estimate direct groundwater flow to Chilmark Pond (see Table III-1). The subwatershed discharge volumes and measured streamflow volumes were used to assist in the salinity calibration of the hydrodynamic model. The overall estimated groundwater flow into Chilmark Pond from the MEP delineated watershed is 25,354 m3/d. It should be noted that the Chilmark Pond Embayment System as delineated has contributing areas to support groundwater and stream discharge to the freshwater western basin of Upper Chilmark Pond (western-most basin) and groundwater discharge to the estuarine eastern basin, Lower Chilmark Pond, Wades Cove and Gilbert's Cove. The freshwater hydrology is such that much of the freshwater discharge from Upper Chilmark Pond is to the eastern basins through the channel, with the remainder exiting through the barrier beach via seepage.

Review of watershed delineations for the Chilmark Pond Embayment System allows new hydrologic data to be reviewed/incorporated as appropriate and the watershed delineation to be reassessed. The evaluation of older data and incorporation of new data during the development of the MEP watershed model is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model used for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are included/excluded within the contributing areas. Small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the downgradient estuary. The MEP watershed delineation was used to develop the watershed nitrogen loads to each of the aquatic systems and ultimately to the estuarine waters of the Chilmark Pond Embayment System.

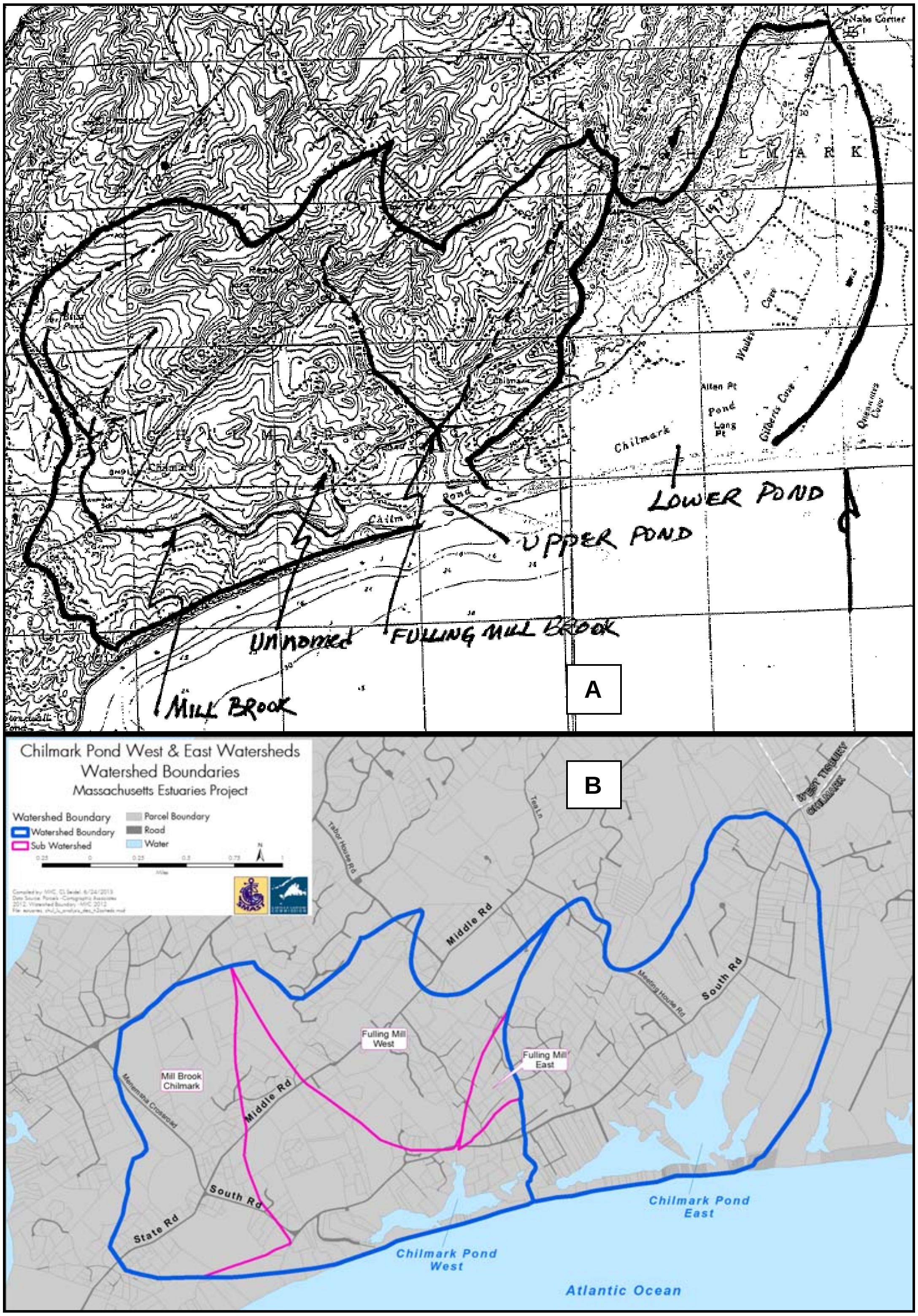


Figure III-2. Comparison of current MEP watershed delineation with historic, previous Chilmark Pond watershed delineation. "A" shows delineation in MVC (1999), while "B" shows current MEP delineation. The MEP watershed delineation, which reflects subsequent data collection, is 1% smaller than the 1999 delineation and includes internal subwatershed delineations.

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

The MEP Technical Team coordinated the development of the watershed nitrogen loading for the Chilmark Pond estuary system with the Martha's Vineyard Commission (MVC) staff. This effort led to the development of nitrogen-loading rates (Section IV.1) to the Chilmark Pond watershed (Section III). The Chilmark Pond watershed was sub-divided into five (5) subwatersheds, including three streams that flow into the western/upper portion of the pond, to define contributing areas to each of the major subwatersheds and basins within the Lower Chilmark Pond.

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 in-depth studies is applied to other portions. The Linked Watershed-Embayment Management Model approach (Howes and Ramsey, 2001) uses a land-use Nitrogen Loading Sub-Model based upon subwatershed-specific land uses and pre-determined nitrogen loading rates. For the Chilmark Pond embayment system, the model used MVC-supplied land-use data transformed to nitrogen loads using both regional nitrogen loading factors and local watershed specific data. 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 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. Attenuation through the ponds is conservatively assumed to equal 50% unless available monitoring and pond physical data is reliable enough to calculate a pond-specific attenuation factor. Attenuation through streams is usually based on site-specific study of streamflow. In the Upper Chilmark Pond watershed, there are delineated watersheds to three streams (Mill Brook, Fulling Mill East, and Fulling Mill West). There are no ponds with delineated watersheds within the overall Chilmark Pond watershed. Surface water attenuation in the streams is discussed in Section IV.2. Other, smaller aquatic features within the watershed to Chilmark Pond do not have separate watersheds delineated and, thus attenuation in these features is not explicitly included in the watershed analysis. If these small features were providing additional attenuation of nitrogen, nitrogen loading to the estuary would only be slightly (~10%) overestimated given the distribution of nitrogen sources, the locations of the gauges, and the locations of these features within the watershed.

Based upon the evaluation of the watershed and the various estimated sources of nitrogen, the MEP Technical Team used the Nitrogen Loading Sub-Model estimate of nitrogen loading for the subwatersheds that directly discharge groundwater to the estuary without flowing through an interim pond or stream measuring point. Reductions in subwatershed nitrogen loads were made to account for natural attenuation in streams. Internal nitrogen recycling was also determined throughout the tidal reaches (when pond is breached) of the Chilmark Pond Estuarine System; measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying water-column. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

IV.1.1 Land Use and Water Use Database Preparation

Martha's Vineyard Commission (MVC) staff, with the guidance of MEP staff, combined Town of Chilmark digital parcel and tax assessors' data from the MVC Geographic Information Systems Department. Digital parcels are from 2011 and land use/assessors data are from January 2012. These land use databases contain traditional information regarding land use classifications (*e.g.*, MADOR, 2012) plus additional information developed by the MVC.

Figure IV-1 shows the land uses within the Chilmark Pond Estuary watershed area. Land uses in the study area are grouped into seven land use categories: 1) residential, 2) commercial, 3) mixed use, 4) industrial, 5) undeveloped (including residential open space), 6) public service/government, including road rights-of-way, and 7) unknown/unclassified. Unknown/unclassified are properties that do not have an assigned land use code in the town assessor's database. These seven land use categories are generally aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MADOR, 2012). "Public service" in the MADOR system is tax-exempt properties, including lands owned by town or state government (*e.g.*, open space, roads, state forest) and private groups like churches and colleges.

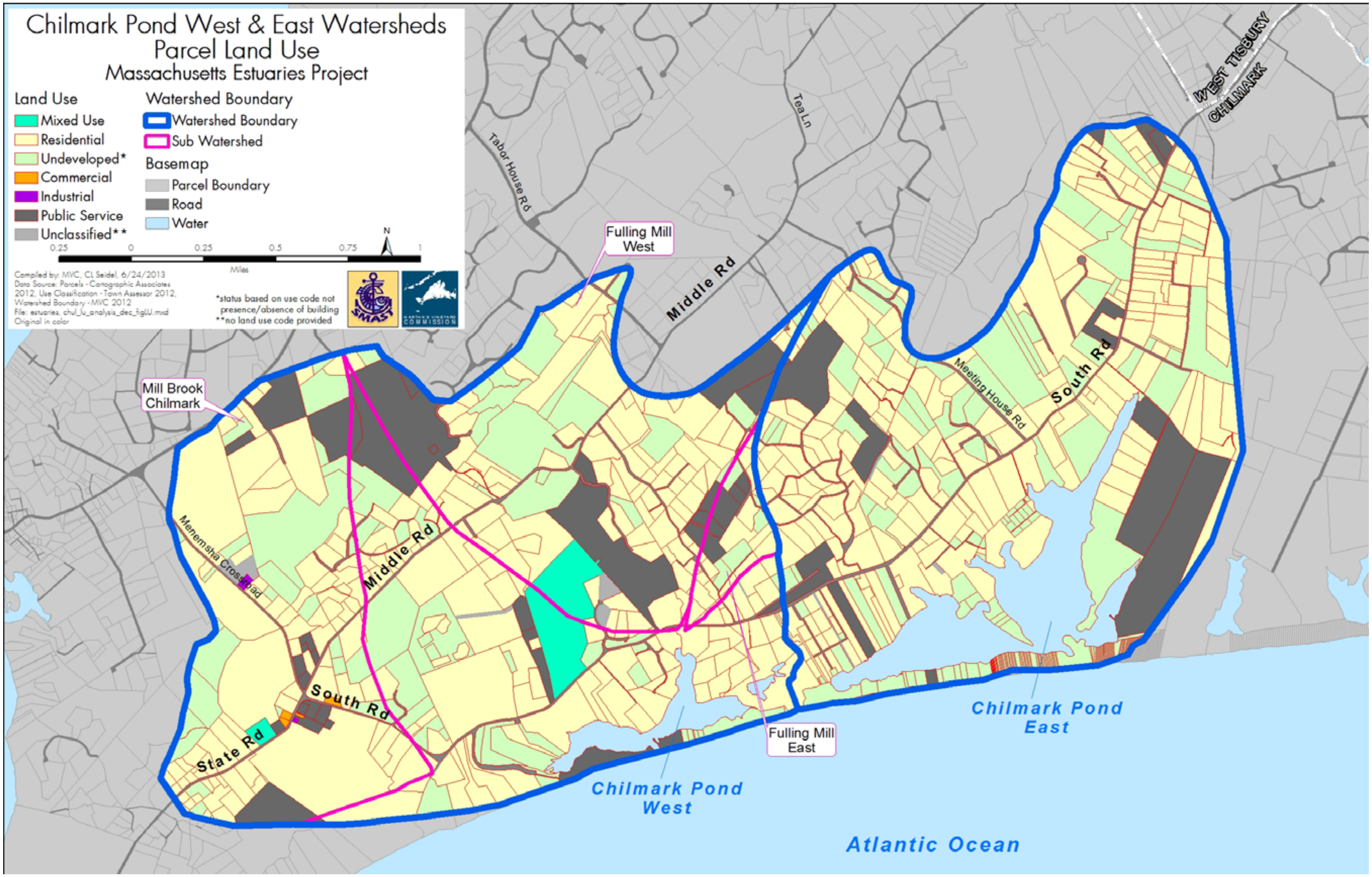


Figure IV-1. Land-use in the Chilmark Pond watershed. Watershed extends is completely within the Town of Chilmark. Land use classifications are based on town assessors' records and general categories in MassDOR (2012).

In the overall Chilmark Pond System watershed, the predominant land use based on area is residential parcels, which accounts for 58% of the overall watershed area; undeveloped lands are the second highest percentage of the system watershed (24%) (Figure IV-2). Single-family residences (MADOR land use code 101) are 65% of the overall system residential land area. Residential land uses are also the dominant land use in the area of all five subwatersheds. Undeveloped land is the second-most predominant land use in three subwatersheds; in Fulling Mill East and Fulling Mill West subwatersheds public service/ROW lands have the next most area after residential parcels. Overall, undeveloped land uses account for 24% of the entire Chilmark Pond watershed area.

In all the subwatershed groupings shown in Figure IV-2, residential parcels are the dominant parcel type in all subwatersheds, generally ranging between 49% and 73% of all parcels in these subwatersheds. Residential parcels are 49% of the parcels in the Chilmark East subwatershed; 47% of the parcels in this subwatershed are classified as undeveloped. Overall, 57% of all parcels in the whole Chilmark Pond system watershed are classified as residential. Single-family residences (MassDOR land use code 101) are 68% to 86% of residential parcels in the individual subwatersheds and 77% of the residential parcels throughout the whole Chilmark Pond system watershed.

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

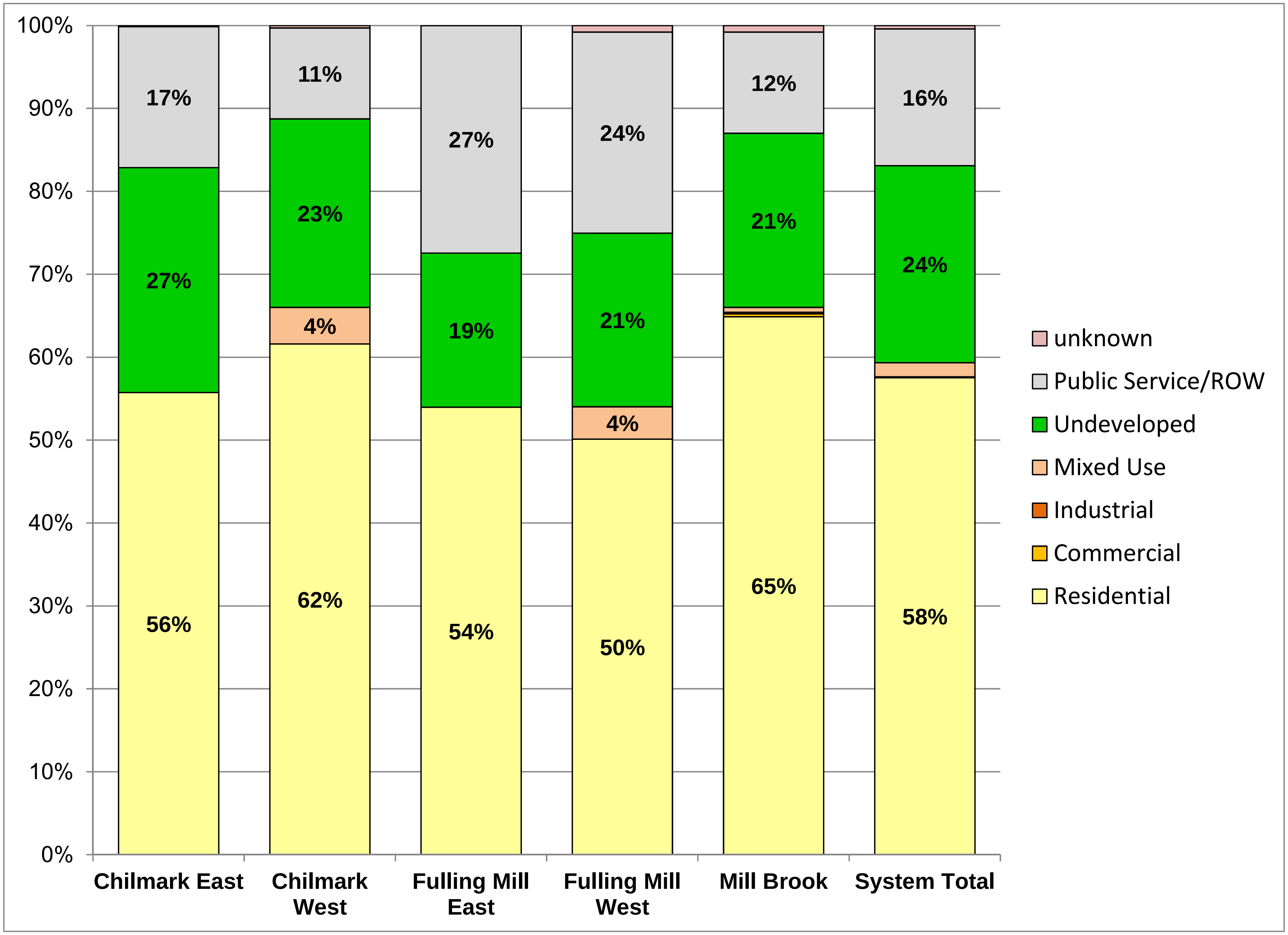


Figure IV-2. Distribution of land-uses by area within the subwatersheds and whole watershed to Chilmark Pond. Only percentages greater than or equal to 4% are shown. Land use categories are based on town and Massachusetts DOR (2012) classifications.

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

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

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

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

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

The independent validation of the water quality model (Section VI) 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 loads that are generally used in regulatory situations. The lower concentration results in slightly higher amounts of nitrogen mitigation (estimated at 1% to 5%) needed to lower embayment nitrogen levels to a nitrogen target (*e.g.* nitrogen threshold, cf. Section VIII). The additional nitrogen removal is not proportional to the septic system nitrogen level, but is related to the how the septic system nitrogen mass compares to the nitrogen loads from all other sources that reach the estuary (*i.e.*, attenuated loads).

In order to estimate wastewater flows, MEP staff generally work with municipal or water supplier partners in the study watershed to obtain parcel-by-parcel water use information. In the Chilmark Pond watershed, this type of water use information was not available. With this in mind, MEP staff reviewed water use information supplied for other MEP studies on Martha's Vineyard and the demographics associated with the towns to find the best match for seasonality and residential occupancy. Among the towns with available town water databases, Oak Bluffs had the best match for Chilmark's seasonality and occupancy. Average water uses for various land use categories were developed from the Oak Bluffs data and assigned to properties classified in the same land use categories in the Chilmark Pond watershed. Review of the water use dataset found that single family residences (MADOR land use code 101) averaged 146 gallons per day (gpd), two family residences (MADOR land use code 104) averaged 250 gpd, and multiple houses on one parcel (MADOR land use code 109) averaged 255 gpd. Average water use was also determined for a variety of other non-residential land uses and site-specific flows were developed based on review of the parcels (*e.g.*, a parcel classified as a 718 pasture land use was assigned the two family water use based on having two houses on the property). These average water uses were used to determine individual parcel water uses in the Chilmark Pond watershed. Water use is used as a proxy for wastewater generation from septic systems on all developed properties in the watershed. Wastewater-based nitrogen loading from the individual parcels using on-site septic systems is based upon the average water-use, nitrogen concentration, and consumptive loss of water before the remainder is treated in a septic system (see Section IV.1.2).

Oak Bluffs was selected as the basis for Chilmark's water use estimates based on a review of US Census information and review of available water use datasets among the towns on Martha's Vineyard. In the 2010 US Census, Chilmark had an average housing occupancy of 2.18 people per occupied housing unit and 74% of the available housing units were classified as seasonal units. These factors changed only slightly from the 2000 US Census were occupancy was 2.21 people per unit and 71% of the units were seasonal. Oak Bluffs has approximately the same occupancy as Chilmark (2.28 people per unit in the 2010 Census), but has a lower seasonal percentage (51%). State on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so for the purposes of Title 5 each person generates 55 gpd of wastewater. Based on the average occupancy within Chilmark and 55 gpd per person, average water use would be 120 gpd, while in Oak Bluffs it would be 125 gpd.

Given that such a high percentage of housing units on Martha's Vineyard are occupied only on a seasonal basis, estimates of water use based on Census data should include an adjustment for the seasonal population increase. Estimates of summer populations on Cape Cod and the Islands derived from a number of approaches (*e.g.*, traffic counts, garbage generation, WWTF flows) generally suggest average summer population increases from two to three times the year-round residential populations measured during the US Census. If it is conservatively assumed that seasonally-classified residential properties in Chilmark are occupied at twice the year-round occupancy for three months, the estimated average town-wide water uses would be 142 gpd, which is approximately the same flow assigned to single-family residences based on the Oak Bluffs water use (146 gpd). This analysis of Census data suggests that use of the Oak Bluffs water use factors is a reasonable basis of estimating Chilmark water uses.

Water use estimates for commercial and industrial properties were treated somewhat the same way. Project staff reviewed the building coverage for each commercial, industrial, and non-profit (land uses classified in the 900 group), seasonality, housing occupancy, and then reviewed previous MEP assessments on Martha's Vineyard for similar percentages. There are four (4) properties with commercial land use classifications in the Chilmark Pond watershed (0.3% of the watershed area), two (2) classified as industrial properties, and ten (10) non-profit parcels with buildings. All of these properties are located in the Mill Brook subwatershed. Among the commercial properties, building coverage averaged 11% of the lot areas; commercial, industrial, and non-profit building coverage collectively averaged 10%. Based on a review of factors from past MEP assessments, project staff determined that Lagoon Pond water use rate of 0.021 gpd/building sq ft was most appropriate estimate for commercial, industrial, and non-profit buildings in the Chilmark Pond watershed.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of estuary watershed nitrogen loading is usually fertilizers, including fertilized lawns, agricultural land uses (including cranberry bogs), and golf courses. Among these, residential lawns are usually the predominant watershed source within this category. In order to add all of these sources to the nitrogen-loading model for the Chilmark Pond system, project staff reviewed available information about residential lawn fertilizing practices within other estuary watersheds on Martha's Vineyard and agricultural fertilizer usage. There are no golf courses or cranberry bogs within the Chilmark Pond 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 of nitrogen per 1,000 sq. ft. of lawn, 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 generally used in the MEP nitrogen loading calculations unless site-specific or watershed-specific data is available.

In order to complete the Chilmark Pond watershed nitrogen loading, project staff utilized lawn areas determined by the MVC in a previous assessment of Tisbury Great Pond (Howes, et al., reviewed past Martha's Vineyard MEP assessments where MVC staff measured hundreds of lawn areas in different subwatersheds. Among the previous MEP reviews, MVC staff found that residential lawn areas averaged approximately 6,100 square feet in the western portion of the Tisbury Great Pond watershed, including the Tiasquam River and Black Point Pond subwatersheds that abut the Chilmark Pond watershed. MEP staff reviewed lawn areas for random parcels within the Chilmark Pond watershed and found that 6,100 square feet seemed to be a reasonable assumption for residences within the Chilmark Pond watershed as well. Other lawn loading factors in the Chilmark Pond model are those generally used in MEP nitrogen loading calculations.

Nitrogen Loading Input Factors: Agricultural Areas

Working with MEP staff, MVC staff also reviewed all parcels classified as agricultural (700s MADOR land use codes), as well as farms on other non-farm coded properties, and determined the area of fertilized crops and obtained counts for farm animals. Nitrogen applications rates and leaching rates are based on standard MEP agricultural crop and farm animal loading factors that have been developed for use in other MEP analyses on Martha's Vineyard. According to this review, the watershed has 12 acres of cropland and these lands add 253 kg/yr of nitrogen to the Chilmark Pond watershed. MVC staff also provided farm animal counts within the watershed (personal communication, Sheri Caseau, MVC, 6/13). This review identified 2,877 animals with cattle (60%) being the most common. MEP nitrogen loading factors have previously been developed for farm animals, including nitrogen leaching rates and species-specific total nitrogen releases, were applied. According to this review, these animals add 2,877 kg/yr of nitrogen to the Chilmark Pond watershed.

Nitrogen Loading Input Factors: Other

One of the other key factors in the nitrogen loading calculations is recharge rates associated with impervious surfaces and natural areas. As discussed in Chapter III, Martha's Vineyard-specific recharge rates were developed and utilized based on comparison to the precipitation data in Edgartown and results of the USGS groundwater modeling effort on Cape Cod. Other nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and Massachusetts DEP's

Nitrogen Loading Computer Model Guidance (1999). Factors used in the MEP nitrogen loading analysis for the Chilmark Pond watershed are summarized in Table IV-1.

Table IV-1. Primary Nitrogen Loading Factors used in the Chilmark Pond MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from watershed-specific data.

Nitrogen Concentrations:	mg/l	Recharge Rates:[2]	in/yr
Road Run-off	1.5	Impervious Surfaces	42.2
Roof Run-off	0.75	Natural and Lawn Areas	28.7
Direct Precipitation on Embayments and Ponds	1.09	Water Use/Wastewater:[3]	
Natural Area Recharge	0.072	Existing and buildout single family residences (land use code 101)	146 gpd
Wastewater Coefficient	23.63		
Fertilizers:			
Average Residential Lawn Size (sq ft)[1]	6,100	Two-family residential (land use code 104)	250 gpd
Residential Watershed Nitrogen Rate (lbs/1,000 sq ft)[1]	1.08	Multiple houses on same lot residential (land use code 109)	255 gpd
Nitrogen leaching rate	20%	Commercial, industrial, and non-profit parcels with buildings (gpd/sq ft of building)	0.021
Building area based on individual building measures		Buildout: no commercial or industrial additions	
Farm Animals[4]	kg/yr /animal	Crops[4]	kg/ha/yr
Horse	32.4	Hay, Pasture, Nursery[5]	5
Cow/Steer	55.8	Field Crop	34
Sheep	7.3	Crop N leaching rate	30%
Hogs/Pigs	14.5		
Chickens	0.4		
Animal N leaching rate	40%		

Notes:
1) MVC staff measured a sample of lawns in western portion of Tisbury Great Pond watershed and found that lawns averaged 6,100 square feet. MEP staff review found this was also appropriate for Chilmark Pond watershed.
2) Based on precipitation rate of 46.9 inches per year (20 year average at long-term Edgartown station); recharge is based on recharge to precipitation relationship used in Cape Cod groundwater modeling (Walter and Whealan, 2005).
3) Water use was unavailable for Chilmark Pond watershed. Water use based on Oak Bluffs water use for parcels with same land use classifications.
4) Crop and farm animal loading rates and leaching rates are standard MEP factors based on available literature and USDA guidance.
5) Hay, pasture, and nursery loading incorporates leaching rate.

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information was linked to the parcel coverages, nitrogen loads from parcels were assigned to various watersheds based initially on whether nitrogen load source areas were located within a respective watershed. This review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Individualized information for parcels with atypical nitrogen loading (farm animals, agricultural fields, etc.) was also assigned at this stage. It should be noted that small shifts in nitrogen loading due to the above assignment procedure generally have a negligible effect on the total nitrogen loading to the Chilmark Pond 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 to subwatersheds, all relevant nitrogen loading data were assigned by subwatershed. This step includes summarizing water use, parcel area, frequency, private wells, and road area. Individual sub-watershed information was then integrated to create the Chilmark Pond Watershed Nitrogen Loading module with summaries for each of the individual subwatersheds. The subwatersheds generally are paired with functional embayment/estuary units for the Linked Watershed-Embayment Model's water quality component.

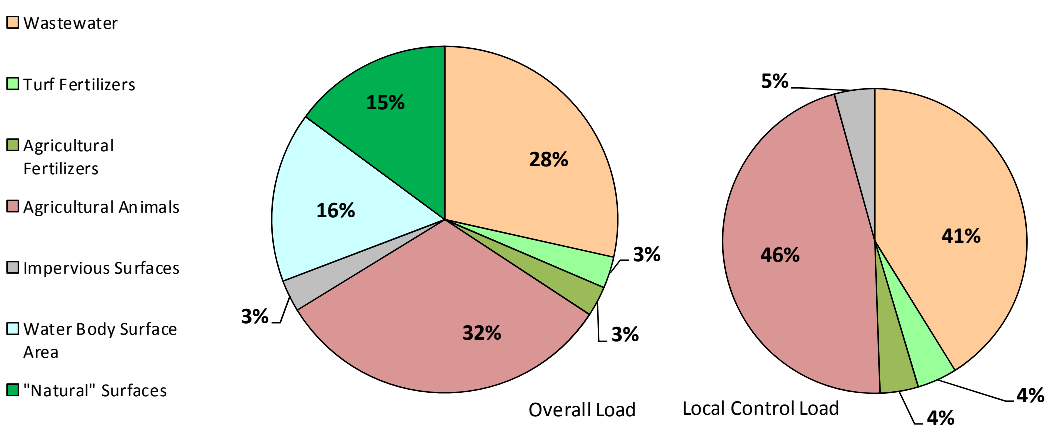
For management purposes, the aggregated embayment watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Chilmark Pond System, the major types of nitrogen loads are: wastewater (*e.g.*, septic systems), fertilizer (including residential lawns and agricultural sources), farm animals, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-2). 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-3). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation during transport through streams or ponds. These attenuated loads reach the estuarine system and are used in the embayment water quality sub-model. Natural nitrogen attenuation in the Chilmark Pond watershed occurs to watershed nitrogen loads that pass through Mill Brook, Fulling Mill Brook East, and the Fulling Mill Brook West (Section IV.2).

Buildout

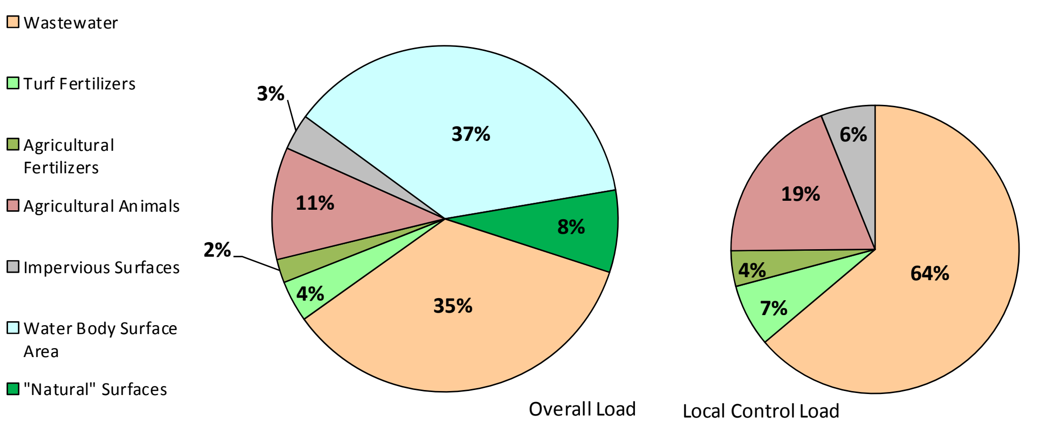
Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment (or scenario) of potential development within the study area watershed. For the Chilmark Pond modeling, MVC staff under the guidance of MEP staff reviewed individual properties for potential additional development. This review included assessment of minimum lot sizes based on current zoning and potential additional development on existing developed lots.

Table IV-2. Chilmark Pond Watershed Nitrogen Loads. Presents nitrogen loads are based on current conditions, including fertilizer loads from golf courses and farms and loads from the West Tisbury and Chilmark landfills. Buildout loads include septic, fertilizer, and impervious surface additions from developable properties. All values are kg N yr^{-1}.

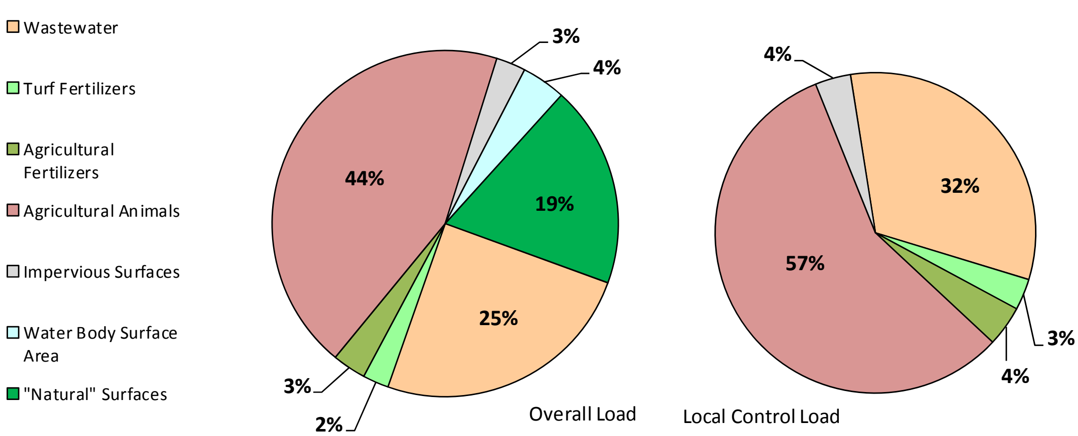
		Chilmark Pond N Loads by Input (kg/yr):								***Present N Loads***			***Buildout N Loads***		
Name	**Watershed ID#**	**Wastewater**	**Turf Fertilizers**	**Agricultural Fertilizers**	**Agricultural Animals**	**Impervious Surfaces**	**Water Body Surface Area**	**"Natural" Surfaces**	**Buildout**	**UnAtten N Load**	**Atten %**	**Atten N Load**	**UnAtten N Load**	**Atten %**	**Atten N Load**
Chilmark Pond System Total		2,560	264	253	2,877	267	1,429	1,337	271	8,985		7,650	9,256		7,805
Chilmark East TOTAL		1,122	123	70	335	107	-	245	15	2,002		2,002	2,017		2,017
Chilmark East	1	1,122	123	70	335	107	-	245	15	2,002	0%	2,002	2,017	0%	2,017
Chilmark West TOTAL		1,438	140	182	2,542	160	239	1,092	256	5,793	8%	4,457	6,049	8%	4,597
Chilmark West	2	419	41	167	1,087	48	-	831	(104)	2,594	0%	2,594	2,490	0%	2,490
Fulling Mill East	3	80	9	-	-	7	-	12	18	108	0%	108	126	0%	126
Fulling Mill West	4	432	47	-	354	44	-	124	172	1,000	20%	800	1,173	20%	938
Mill Brook	5	506	44	15	1,100	61	-	125	170	1,851	41%	1,092	2,021	41%	1,193
Chilmark West Fresh Water Surface Area							239			239		239	239		239
Chilmark East Estuary Surface Area							1,190			1,190		1,190	1,190		1,190



A. Chilmark Pond: Whole System



B. Chilmark Pond: East Total



C. Chilmark Pond: West Total

Figure IV-3. Unattenuated nitrogen load (by percent) for land use categories within the overall Chilmark Pond System watershed and the East and West subwatersheds. "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.

The buildout procedure used in this watershed and generally completed by MEP staff is to evaluate town zoning to determine minimum lot sizes in each of the zoning districts, including overlay districts (*e.g.*, water resource protection districts). Larger lots are subdivided by the minimum lot size to determine the total number of new lots. In addition, existing developed properties are reviewed for any additional development potential; for example, residential lots that are twice the minimum lot size, but have only one residence are assumed to have one additional residence at buildout. Most of the focus of new development is for properties classified as developable by the local assessor (state class land use codes 130 and 131 for residential properties). Properties classified by the town assessors as “undevelopable” (*e.g.*, code 132) were not assigned any development at buildout. Project staff typically reviews these initial results with local experts, who were MVC staff in this case, to produce a final MEP buildout assessment.

Based on the buildout assessment completed for this review, there are 228 potential additional residential dwellings within the Chilmark Pond watershed. There is no potential additional commercial or industrial developable land. All parcels included in the buildout assessment of the Chilmark Pond watershed are shown in Figure IV-4.

Nitrogen loads were developed for these buildout additions based largely on existing development factors within the Chilmark Pond watershed. Additional buildout single-family residential dwellings were assigned a water use flow of 146 gpd, which is the same average water use assigned to developed residences in the watershed. Other factors used in the MEP buildout assessment are listed in Table IV-1.

Many of the parcels assigned additional development at buildout already have agricultural uses (*e.g.*, parcels assigned as developable residential parcels by the town assessor, but currently used as pasture or for hay). Existing agricultural loads were removed if the parcel was identified as having additional residential development in the buildout scenario. It should be noted that this is one example of a buildout scenario; alternative assumptions about future development could be developed to assess the water quality impacts of other buildout scenarios.

Table IV-2 presents a sum of the additional nitrogen loads by subwatershed for the MEP buildout scenario. This sum includes the wastewater, fertilizer, and impervious surface loads from additional residential dwellings added. Overall, MEP buildout additions within the entire Chilmark Pond System watershed will increase the unattenuated loading rate by 3%.

IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT

IV.2.1 Background and Purpose

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

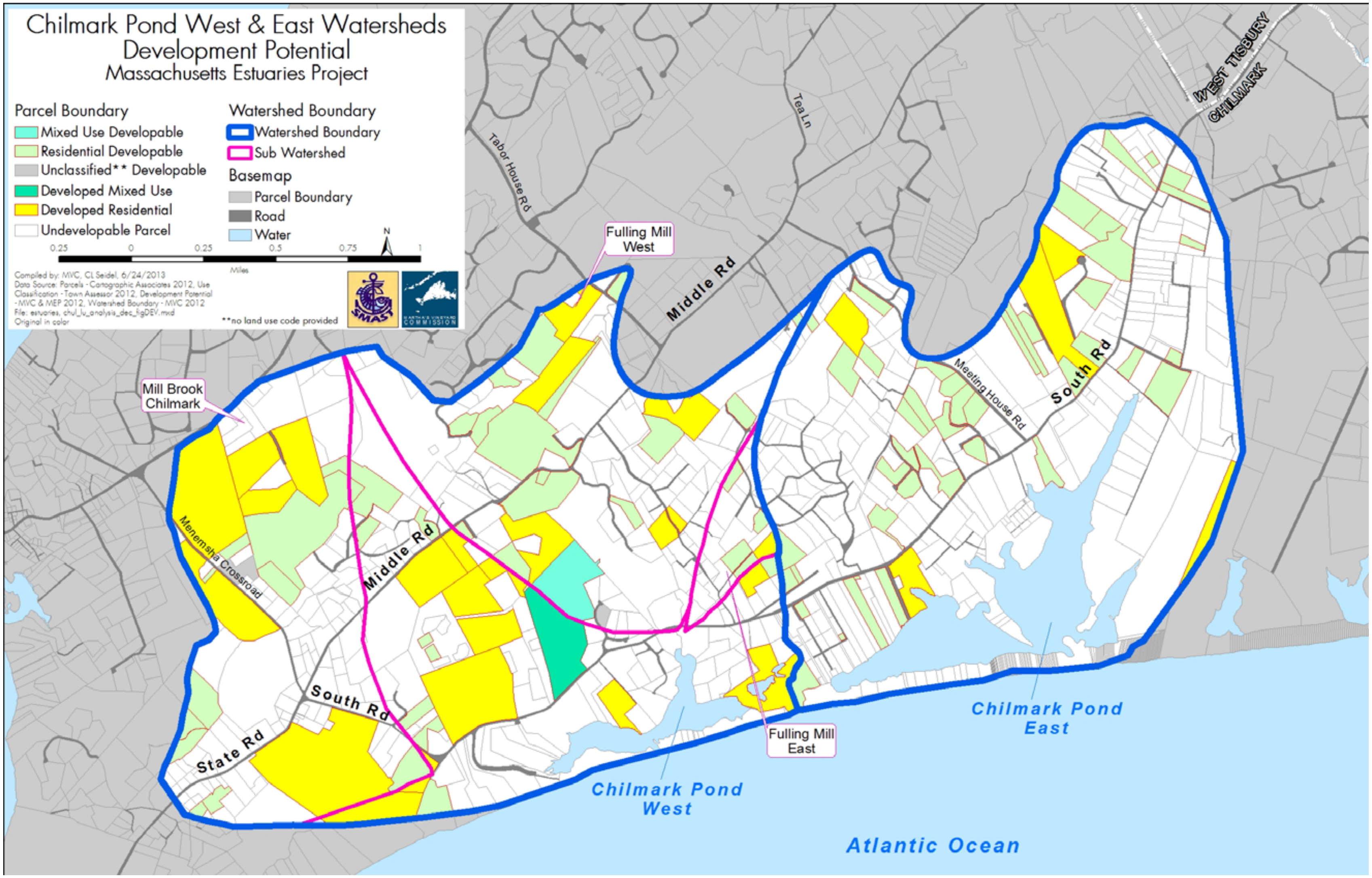


Figure IV-4. Developable Parcels in the Chilmark Pond watershed. Developable parcels and developed parcels with additional development potential are highlighted. The parcels are selected based on town assessors' land use classifications and review of minimum lot sizes in town zoning regulations. Nitrogen loads in the MEP buildout scenario are based on additional development assigned to these parcels.

If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watersheds where nitrogen transport from source to estuarine waters is through groundwater flow in sandy outwash aquifers (such as the developed regions of the Chilmark Pond Embayment System watershed). The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, some portion of the watershed nitrogen passes through a surface water ecosystem (pond, wetland, stream) on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) is diminished by natural biological processes that represent removal (not just temporary storage). However, this natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes. In the watershed for the Chilmark Pond Embayment System, a portion of the freshwater flow and transported nitrogen passes through several surface water systems (e.g. Mill Brook and the east and west branches of Fulling Mill Brook), both of which discharge into Upper Chilmark Pond prior to entering the main estuary basin (Lower Chilmark Pond) and thereby, produce the opportunity for nitrogen attenuation during transport (Figure IV-5).

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

Figure IV-5. Location of Stream gauges (red symbols) in the Upper Chilmark Pond Embayment System watershed. The combined sub-watershed areas contributing to the gauge sites covers ~40% of the entire watershed to the estuary.

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements of upper watershed attenuation were undertaken as part of the MEP Approach in the Chilmark Pond Embayment System. MEP conducted long-term measurements of natural attenuation relating to surface water discharges to the estuary in addition to the natural attenuation measures by fresh kettle ponds in the overall watershed (as data was available), addressed above (Section IV.1). These additional site-specific studies were conducted in the 2 major surface water flow systems in the Chilmark Pond System watershed, 1) Mill Brook discharging to the head of Upper Chilmark Pond system and 2) Fulling Mill Brook (east and west branches), a moderately large stream also discharging to Upper Chilmark Pond. Together these 2 streams "drain" the recharge and transport the recharge from 2 sub-watersheds that combined account for ~40% of the total watershed area to the estuary.

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the freshwater streams discharging to the estuary provides a direct integrated measure of all of the processes presently attenuating nitrogen in the contributing area up-gradient from the gauging sites. Flow and nitrogen load were measured at the gauges in each freshwater stream site for between 16 and 28 months of record depending on the stream gauging location (Figure IV-5). For each time-series period, velocity profiles were completed on each river every month to two months. The summation of the products of stream subsection areas of the stream cross-section and the respective measured velocities represent the computation of instantaneous stream flow (Q).

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

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

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

where by:

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

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

Periodic measurement of flows over the entire stream gauge deployment period allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the detailed record of stage measured by the continuously recording stream gauges. Water level data obtained every 10-minutes was averaged to obtain hourly stages for a given river. These hourly stages values were then entered into the stage-discharge relation to compute hourly flow. Hourly flows were summed over a period of 24 hours to obtain daily flow and further, daily flows summed to obtain annual flow. In the case of tidal influence on stream stage, the diurnal low tide stage value was extracted on a day-by-day basis in order to resolve the stage value indicative of strictly freshwater flow. The lowest tide stage value for a given day was extracted from all the other stage values on a specific day and that lowest stage was then entered into the stage – discharge relation in order to compute daily flow. The lowest stage value in a tidally influenced stream was used as it is most representative of freshwater flow. A complete annual record of stream flow (365 days) was generated for the surface water discharges flowing into the Chilmark Pond Embayment System from Mill Brook and Fulling Mill Brook discharging into Upper Chilmark Pond. In the case of Mill Brook, two years of record were obtained and an average annual flow was obtained based on both years of flow data.

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

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Mill Brook

Bliss Pond, located up gradient of the Mill Brook gauge site (Windy Gates Road crossing) is a small freshwater pond and like many of the freshwater ponds on Martha's Vineyard and Cape Cod, has the potential to attenuate nitrogen from the watershed prior to discharging water to the aquifer along its down-gradient shore. Since Bliss Pond is relatively small and shallow, it has a rapid water turnover time (<1 day) which lowers the level of nitrogen attenuation it can achieve. The Mill Brook outflow, generated via direct groundwater contribution to the stream channel and representing a fraction of the water from Bliss Pond provides for a direct measurement of the sub-watershed nitrogen load to the estuary and the level of nitrogen attenuation occurring in the up-gradient watershed. In addition, nitrogen attenuation also occurs within the wetlands, small impoundments and streambed associated with Mill Brook. The combined rate of nitrogen attenuation by all of these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to Mill Brook above the gauge site and the measured annual discharge of nitrogen to the tidally influenced (when the pond is breached) Upper Chilmark Pond portion of the overall system, Figure IV-5.

At the Mill Brook gauge site, a continuously recording vented calibrated water level gauge was installed to yield the level of water in the lower reach of Mill Brook. As the Upper Chilmark Pond portion of the overall Chilmark Pond Embayment System is tidally influenced when the pond is periodically breached, the gauge was located as far down gradient along Mill Brook such that a complete measure of attenuation in the sub-watershed could be obtained and also such that freshwater flow could be measured at low tide during breach events and at high stand when the pond was closed. To confirm that freshwater was being measured, the stage record was analyzed for any tidal influence during the deployment period and salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average salinity during the entire deployment period was determined to be 0.1 ppt. Therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked monthly. The gauge on Mill Brook was installed on June 20, 2005 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 December 6, 2007 for a total deployment of 29 months.

Surface freshwater flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Mill Brook site based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used to convert the continuously measured stage data to obtain the daily volume of freshwater flow. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the head of the Upper Chilmark Pond portion of the overall system, immediately south of the Windy Gates Road crossing, and reflective of the biological processes occurring in the stream channel and the small up-gradient wetland which contributes to nitrogen attenuation (Figure IV-6 and Table IV-3 and IV-4). In addition, a water balance was constructed based upon the U.S. Geological Survey confirmed groundwater flow model to determine long-term average freshwater discharge expected at each gauge site.

The annual freshwater flow record for Mill Brook measured by the MEP was compared to the long-term average flows determined by the groundwater modeling effort (Table III-1). Two years of flow record (2005-2007) obtained at the gauging station were averaged for the purpose of confirming the watershed delineation. The measured freshwater discharge from Mill Brook was 14% different than the long-term average modeled flows. The average daily flow based on the MEP measured flow data for one hydrologic year beginning September and ending in August (low flow to low flow) was 5,806 m^3/day compared to the long term average flows determined by the USGS modeling effort (4,983 m^3/day). The slight difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Mill Brook is mainly due to inter-annual variation in precipitation. All indications are that the Brook is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within the Mill Brook outflow were moderate, 0.572 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 2.99 kg/day and a measured total annual TN load of 1,093 kg/yr. In Mill Brook, nitrate made up a very small fraction of the total nitrogen pool (16%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the Brook was significantly taken up and converted to organic forms by plants within the pond, wetland and/or stream ecosystems. This is seen in the particulate and dissolved organic nitrogen together accounting for 81 percent of the total nitrogen pool and of that 81 percent, the vast majority (90%) was dissolved organic nitrogen. Given the low level of remaining nitrate in the stream discharge suggests that very little additional uptake of inorganic nitrogen by the natural upgradient freshwater system.

Table IV-3. Comparison of water flow and nitrogen load discharged by Mill Brook and Fulling Mill Brook (east and west branches) to the Upper Chilmark portion of the overall Chilmark Pond Embayment System. The "Stream" data are from the MEP stream gauging effort. Watershed data are based upon the MEP watershed land-use modeling effort (Section IV.1) and the USGS watershed delineation (Section III).

Stream Discharge Parameter	**Mill Brook Discharge**[a] Chilmark Pond	**Fulling Mill Brook East Discharge**[a] Chilmark Pond	**Fulling Mill Brook West Discharge**[a] Chilmark Pond	**Data Source**
Total Days of Record	365[b]	365[c]	365[c]	(1)
Flow Characteristics				
Stream Average Discharge (m3/day)	5,233	470	4,816	(1)
Contributing Area Average Discharge (m3/day)	4,983	492	4889	(2)
Discharge Stream 2004-05 vs. Long-term Discharge	4.78%	-4.68%	-1.52%	
Nitrogen Characteristics				
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.093	0.082	0.085	(1)
Stream Average Total N Concentration (mg N/L)	0.572	0.677	0.448	(1)
Nitrate + Nitrite as Percent of Total N (%)	16%	12%	19%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	2.99	0.31	2.16	(1)
TN Average Contributing UN-attenuated Load (kg/day)	5.07	0.30	2.74	(3)
Attenuation of Nitrogen in Pond/Stream (%)	41%	0%	21%	(4)

(a) Flow and N load to streams discharging to Upper Chilmark Pond includes apportionments of Pond contributing areas as appropriate.
(b) Average September 1, 2005 to August 31, 2007.
(c) September 1, 2005 to August 31, 2006.
(1) MEP gage site data
(2) Calculated from MEP watershed delineations to ponds upgradient of specific gages;
the fractional flow path from each sub-watershed which contribute to the flow in the streams to Chilmark Pond;
and the annual recharge rate.
(3) As in footnote (2), with the addition of pond and stream conservative attentuation rates.
(4) Calculated based upon the measured TN discharge from the rivers vs. the unattenuated watershed load.

Table IV-4. Summary of annual volumetric discharge and nitrogen load from Mill Brook and the Fulling Mill Brook (east and west branches) inflows to Upper Chilmark Pond (head of the Chilmark Pond) estuary based on data presented in Figures IV-6, IV-7, IV-8 and Table IV-3.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Chilmark Pond (Upper) Mill Brook MEP	**September 1, 2005 to August 31, 2007**	**1,909,958**	**178**	**1093**
Chilmark Pond (Upper) Mill Brook MVC	Based on Watershed Area and Recharge	**1,818,795**	--	--
Chilmark Pond (Upper) Fulling Mill (east) MEP	**September 1, 2005 to August 31, 2006**	**171,465**	**14**	**116**
Chilmark Pond (Upper) Fulling Mill (east) MVC	Based on Watershed Area and Recharge	**179,580**	--	--
Chilmark Pond (Upper) Fulling Mill (west) MEP	**September 1, 2005 to August 31, 2006**	**1,758,015**	**150**	**788**
Chilmark Pond (Upper) Fulling Mill (west) MVC	Based on Watershed Area and Recharge	**1,784,485**	--	--

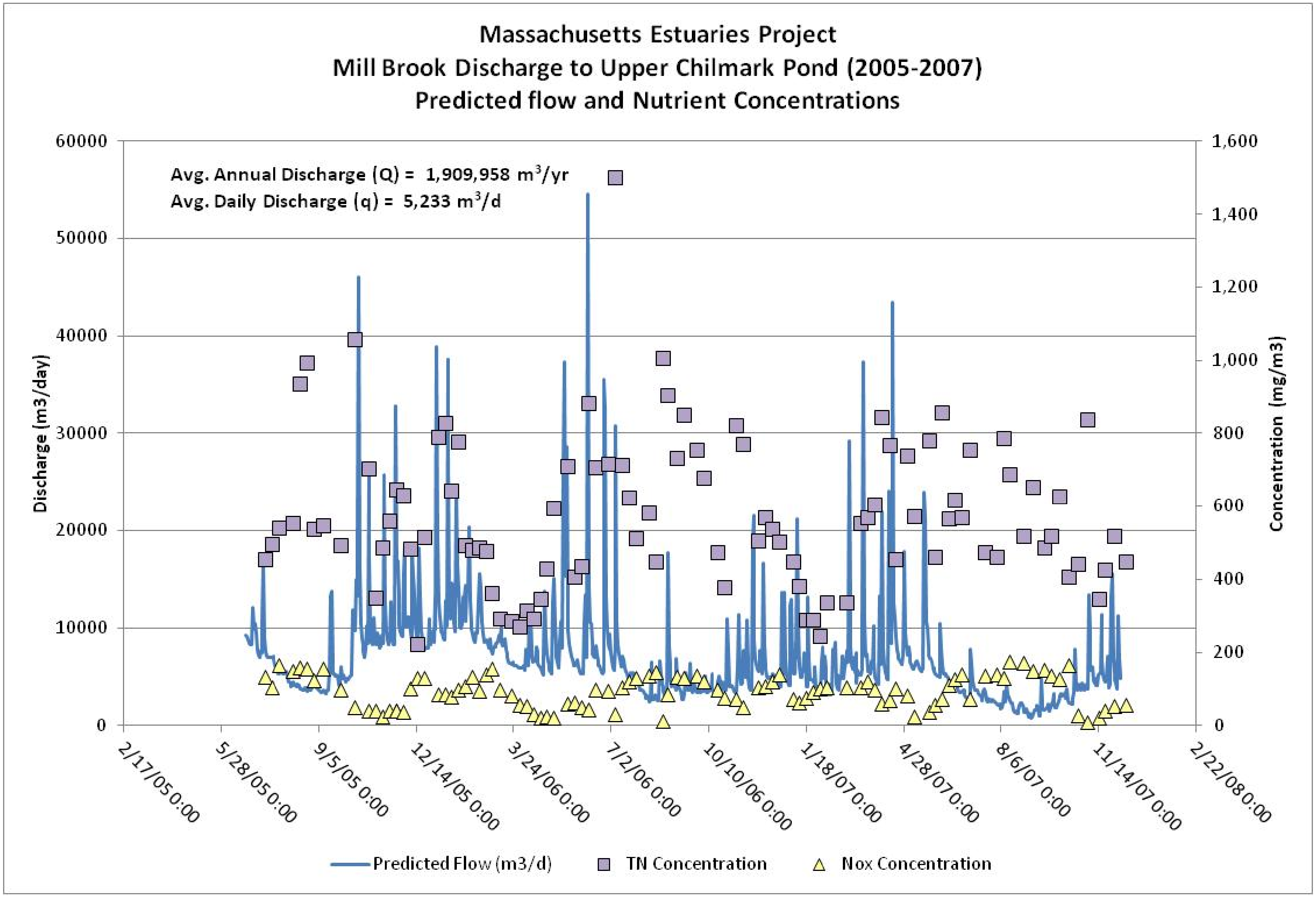


Figure IV-6. Mill Brook discharge (solid blue line), nitrate+nitrite (yellow triangle) and total nitrogen (violet symbol) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Upper Chilmark Pond (Table IV-3).

From the measured nitrogen load discharged by Mill Brook to the estuarine waters of Upper Chilmark Pond and the nitrogen load determined from the watershed based land use analysis, it appears that there is little nitrogen attenuation of watershed derived nitrogen during transport to the estuary. The total nitrogen load (1,093 kg yr^{-1}) discharged from the freshwater Mill Brook to Upper Chilmark Pond compared to that added by the various land-uses to the associated watershed (1,851 kg yr^{-1}), indicates that the integrated attenuation during surface water transport is 41% (i.e. 41% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams evaluated under the MEP is expected given the various aquatic features (small ponds and wetlands) up gradient of the stream gauging location capable of attenuating nitrogen. The directly measured nitrogen load from the brook was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

IV.2.3 Surface water Discharge and Attenuation of Watershed Nitrogen: Fulling Mill Brook (east and west branches) discharge to Upper Chilmark Pond

Similar to Mill Brook, Fulling Mill Brook (inclusive of both the east and west branch) does not have any significant up-gradient pond from which the stream discharges. Rather, this stream system appears to be groundwater fed and emanates from a wooded area up-gradient of South Road, located up gradient of the two (east and west branch) Fulling Mill River gauge sites (South Road crossing). This stream inflow, Fulling Mill Brook, to Upper Chilmark Pond portion of the overall estuarine system provides for a direct measurement of the sub-watershed nitrogen load to the estuary and the level of nitrogen attenuation. In addition, nitrogen attenuation also occurs within any small wetlands, small impoundments and the streambed associated with the two branches of Fulling Mill Brook. 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 east and west branches of Fulling Mill Brook above the gauge sites and the measured annual discharge of nitrogen to the tidal portion (when the pond is breached) of Upper Chilmark Pond, Figure IV-5.

At each gauge site (one on the east branch and one on the west branch), a continuously recording vented calibrated water level gauge was installed to yield the level of water in the lower reach of the freshwater portion of Fulling Mill Brook. As the Upper Chilmark Pond portion of the overall Chilmark Pond system is tidally influenced at times when the pond is breached, the gauge was located as far down gradient along the Fulling Mill Brook reach such that a complete measure of attenuation in the up-gradient sub-watersheds could be obtained and also such that freshwater flow could be measured at low tide during breach events and at high stand when the pond was closed. To confirm that freshwater flow was being measured, the stage record was analyzed for any tidal influence during the deployment period and salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average salinity during the entire deployment period was determined to be 0.1 ppt. Therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked monthly. The gauge on the east and west branches of Fulling Mill Brook were installed on June 20, 2005 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until November 9, 2006 for a total deployment of 17 months.

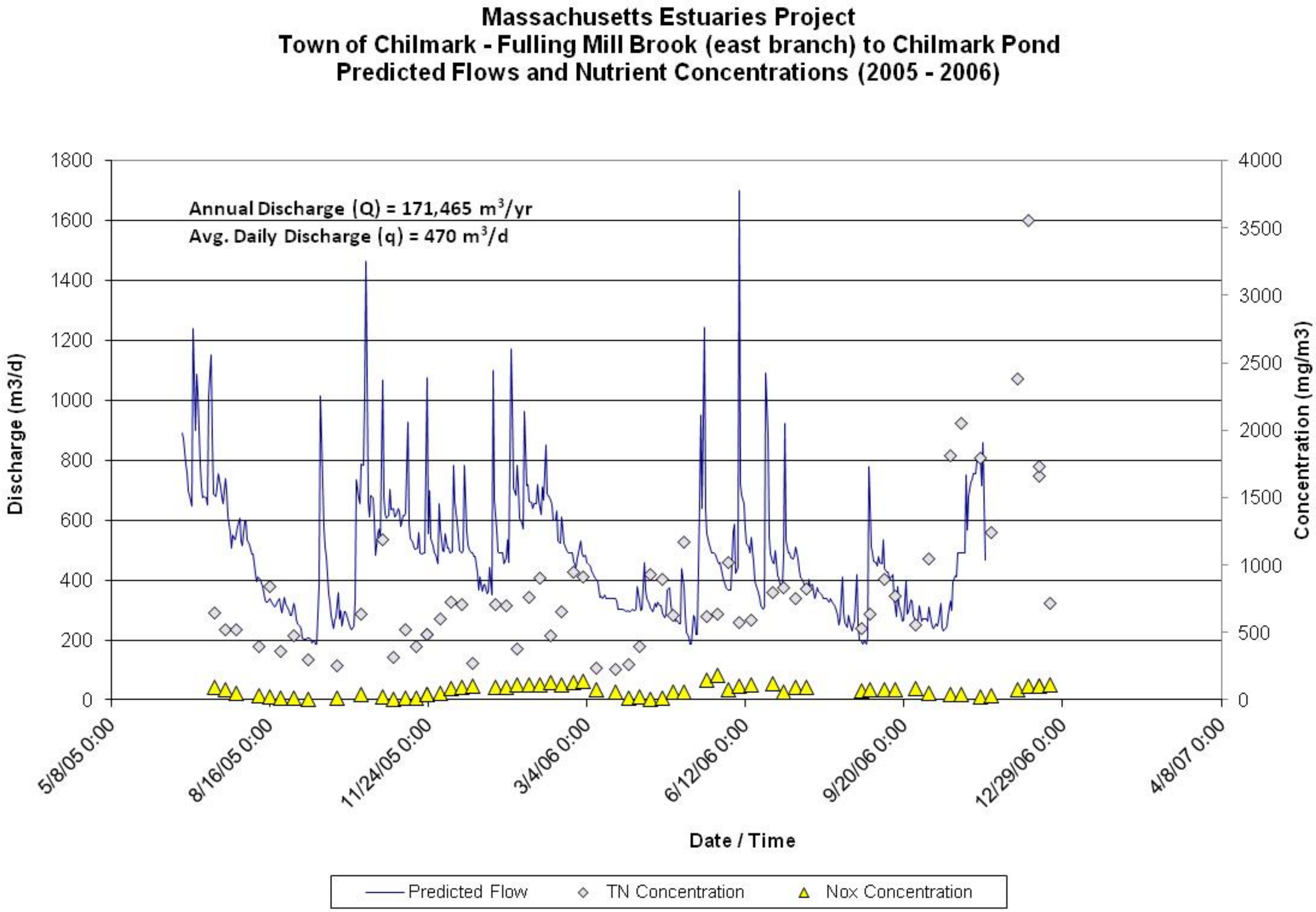


Figure IV-7. Fulling Mill Brook (east) discharge (solid blue line), nitrate+nitrite (yellow symbol) and total nitrogen (violet symbol) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Upper Chilmark Pond portion of the overall Chilmark Pond system (Table IV-3).

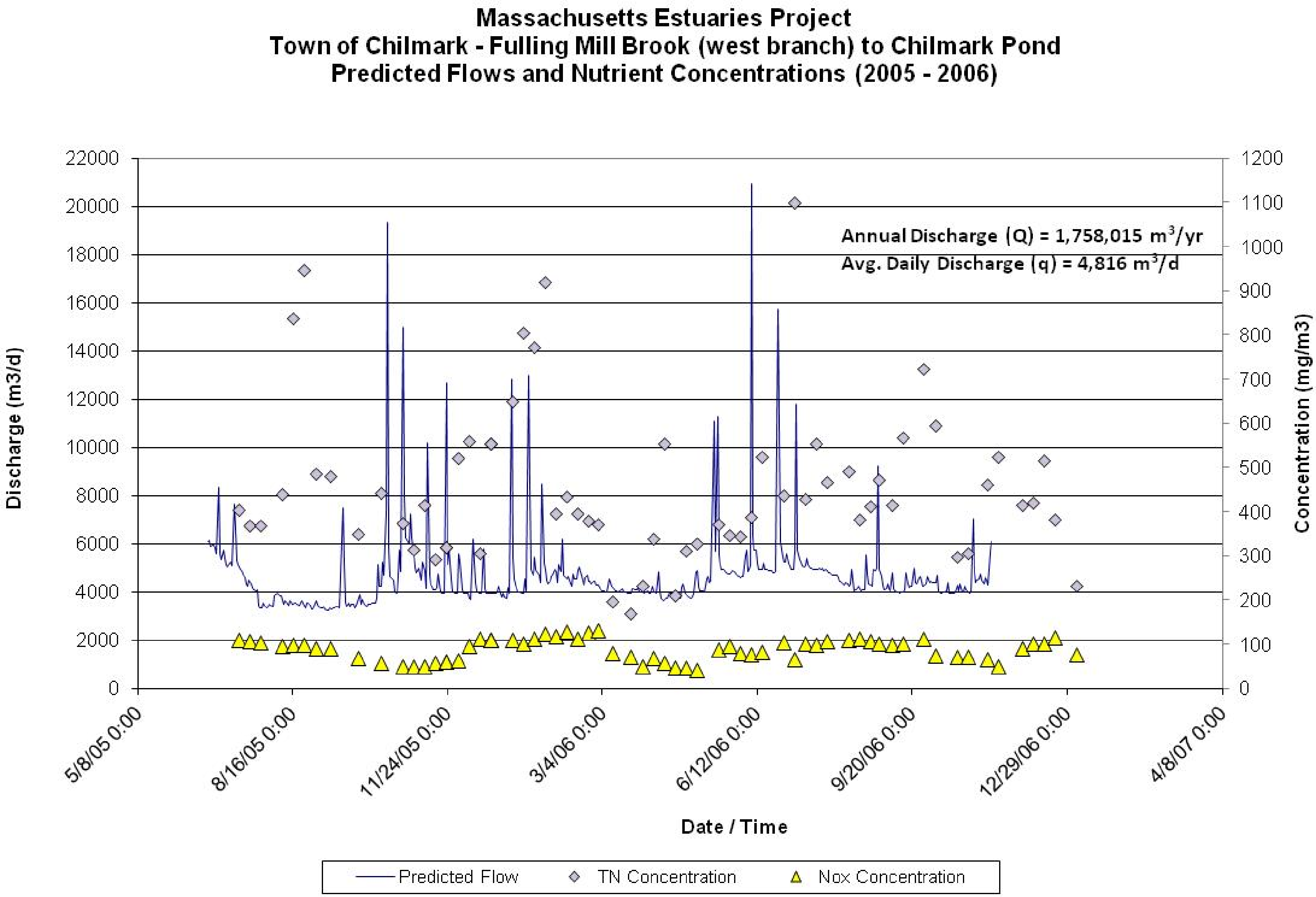


Figure IV-8. Fulling Mill Brook (west) discharge (solid blue line), nitrate+nitrite (yellow symbol) and total nitrogen (violet symbol) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Upper Chilmark Pond portion of the overall Chilmark Pond system (Table IV-3).

Surface freshwater flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for both the Fulling Mill Brook east branch gauge site as well as the Fulling Mill Brook west branch gauge site based upon the flow measurements and measured water levels at both the locations. The rating curves were then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to Upper Chilmark Pond, immediately south of the South Road bridge crossing, and reflective of the biological processes occurring in the stream channel and any up-gradient natural systems that can potentially contribute to nitrogen attenuation (Figure IV-7 and IV-8 and Tables IV-3 and IV-4). In addition, a water balance was constructed based upon the U.S. Geological Survey confirmed groundwater flow model to determine long-term average freshwater discharge expected at each gauge site.

The annual freshwater flow record for the Fulling Mill Brook (east and west branches) as measured by the MEP was compared to the long-term average flow determined by the groundwater modeling effort (Table III-1). The measured freshwater discharge from the Fulling Mill Brook (east + west branch) was the same as the long-term average modeled flows. The average daily flow based on the MEP measured flow data for one hydrologic year beginning September and ending in August (low flow to low flow) was 5,381 m^3/day (470 and 4,816 m^3/d east branch and west branch respectively) compared to the long term average flows from the watershed water balance which indicates a long-term average daily flow of 5,381 m^3/d (492 and 4,889 m^3/d east branch and west branch respectively). The similarity between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in the Fulling Mill Brook watershed is consistent with the observed precipitation for the hydrologic year. Precipitation during the gauge deployment period was very close to long term average precipitation and groundwater levels (see also Section III).

Similar to Mill Creek, the total nitrogen concentrations in the east branch of Fulling Mill Brook was moderate, 0.677 mg N L^{-1}, as was the total nitrogen concentration in the west branch of Fulling Mill Brook, 0.448 mg N L^{-1}. These concentrations, when paired with the respective flows yields an average daily total nitrogen discharge to the estuary of 0.31 kg/day (east branch) and 2.16 kg/day (west branch) and a measured total annual TN load of 116 kg/yr and 788 kg/yr respectively. As with Mill Brook, in the two branches of Fulling Mill Brook, nitrate made up a very small fraction of the total nitrogen pool (12% east and 19% west), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the Brook was significantly taken up and converted to organic forms by plants within the pond and/or stream ecosystems. This is seen in the particulate and dissolved organic nitrogen data which together accounted for 84 percent (east branch) and 77 percent (west branch) of the total nitrogen pool and of those two percentages, the vast majority (68% and 90% respectively) was dissolved organic nitrogen. Given the low level of remaining nitrate in the stream discharge suggests that very little additional uptake of nitrogen by natural up-gradient systems could be achieved to enhance nitrogen attenuation prior to discharge to the estuary.

From the measured nitrogen load discharged by the Fulling Mill Brook (east + west branches) to the Upper Chilmark Pond portion of the overall Chilmark Pond estuary and the nitrogen load determined from the watershed land use analysis, it appears that there is only a slight attenuation of nitrogen during transport from upland sources to the estuary. Based upon nearly equal measured total nitrogen load (116 kg yr^{-1}) discharged from the east branch of Fulling Mill Brook compared to that added by the various land-uses to the associated watershed

(108 kg yr^{-1}), the integrated attenuation during transport from the east branch subwatershed to the estuary is 0% (i.e. nitrogen input to watershed reaches the estuary unattenuated). Similarly, based upon the slightly lower measured total nitrogen load (788 kg yr^{-1}) discharged from the west branch of Fulling Mill Brook compared to that added by the various land-uses to the associated subwatershed (1000 kg yr^{-1}), the integrated attenuation during transport from the west branch subwatershed to the estuary is 21% (i.e. 21% of the nitrogen input to the watershed does not reach the estuary). This level of attenuation compared to other streams evaluated under the MEP is expected given the nature of the network of few up gradient ponds capable of attenuating nitrogen and is also consistent with the limited attenuation observed in the adjacent Tiasquam River system (6% attenuation) discharging to Tisbury Great Pond. The directly measured nitrogen load from the brook was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

The overall objective of the benthic nutrient flux survey was to quantify the summertime exchange of nitrogen, between the sediments and overlying waters throughout the Chilmark Pond (Upper and Lower) 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, brackish and salt water.

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

As stated in above sections, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the Chilmark Pond 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 down gradient larger water body (like the Atlantic Ocean when the pond is breached). However, some of these phytoplankton particles are grazed by zooplankton or filtered from the water by shellfish and other benthic animals and deposited on the bottom sediments. Also, in longer residence time systems (greater than 8 days) these nitrogen rich particles may die and settle to the bottom. In both cases (grazing or senescence), a fraction of the phytoplankton with associated nitrogen “load” become incorporated into the surficial sediments of the system.

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

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bioavailable nitrogen is returned to the embayment water column for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that have been investigated by SMAST and the MEP, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. In contrast in some systems, with salt marsh tidal creeks, the sediments can be a net sink for nitrogen even during summer (e.g. Mashapaquit Creek Salt Marsh, West Falmouth Harbor; Centerville River Salt Marsh). Embayment basins can also be net sinks for nitrogen to the extent that they support relatively oxidized surficial sediments, such as found within nearby Sengekontacket Pond. In contrast, regions of high deposition like Hyannis Inner Harbor on Cape Cod, which is essentially a dredged boat basin, typically support anoxic sediments with elevated rates of nitrogen release during summer months. The consequence of this deposition is that these basin sediments are unconsolidated, organic rich and sulfidic nature (MEP field observations).

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

IV.3.2 Method for determining sediment-watercolumn nitrogen exchange

For the Chilmark Pond embayment 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. Sixteen sediment samples were collected from a total of 15 sites throughout the Chilmark Pond Embayment System. Cores were collected from 9 sites within the main basin of Lower Chilmark Pond, 3 sites from Upper Chilmark Pond, 1 site in Gilberts Cove and 2 sites in Wades Cove (Figure IV-9). All the sediment cores for this system were collected in July-August 2005. 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 shoreside lab. Cores were maintained from collection through incubation at *in situ* temperatures. Bottom water was collected and filtered from core sites to replace the headspace water of each core prior to incubation. The number of core samples from each estuarine component (Figure IV-9) are as follows:

Figure IV-9. Chilmark Pond Embayment System sediment sampling sites (yellow symbols) for determination of sediment-water column exchange rates. Numbers are for reference to station identifications listed above and in Table IV-5. Stations 1-3 are in freshwater.

Chilmark Pond Benthic Nutrient Regeneration Cores

Freshwater

• CHP-1	1 core	(Upper Chilmark Pond - West Basin)
• CHP-2	1 core	(Upper Chilmark Pond - West Basin)
• CHP-3	1 core	(Upper Chilmark Pond - West Basin)

Estuarine

• CHP-4	1 core	(Lower Chilmark Pond - mid basin)
• CHP-5	1 core	(Lower Chilmark Pond - mid basin)
• CHP-6	1 core	(Lower Chilmark Pond - mid basin)
• CHP-7	1 core	(Lower Chilmark Pond - mid basin)
• CHP-8	1 core	(Lower Chilmark Pond - mid basin)
• CHP-9	1 core	(Wades Cove)
• CHP-10	1 core	(Wades Cove)
• CHP-11	1 core	(Wades Cove)
• CHP-12	1 core	(Wades Cove)
• CHP-13/14	2 cores	(Lower Chilmark Pond - East basin)
• CHP-15	1 core	(Lower Chilmark Pond - East basin)
• CHP-16	1 core	(Gilberts Cove)

Sampling was distributed throughout the primary component basins of the Chilmark Pond Embayment System and the results were used for calculating the net nitrogen regeneration rates for the water quality modeling effort.

Sediment-water column exchange follows the methods of Jorgensen (1977), Klump and Martens (1983), and Howes *et al.* (1998) for nutrients and metabolism. Upon return to the field laboratory at a private residence on the shore of Chilmark Pond, 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 (Coastal Systems Analytical Facility, 508-910-6325 or ssampieri@umassd.edu). The laboratory follows standard methods for saltwater analysis and sediment biogeochemistry.

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

Water column nitrogen levels are the balance of inputs from direct sources (land, rain etc), losses (denitrification, burial), regeneration (water column and benthic), and uptake (e.g. photosynthesis). As stated above, during the warmer summer months the sediments of shallow embayments typically act as a net source of nitrogen to the overlying waters and help to stimulate eutrophication in organic rich systems. However, some sediments may be net sinks for nitrogen and some may be in "balance" (organic N particle settling = nitrogen release).

Sediments may also take up dissolved nitrate directly from the water column and convert it to dinitrogen gas (termed “denitrification”), hence effectively removing it from the ecosystem. This process is typically a small component of sediment denitrification in embayment sediments, since the water column nitrogen pool is typically dominated by organic forms of nitrogen, with very low nitrate concentrations. However, this process can be very effective in removing nitrogen loads in some systems, particularly in streams, ponds and salt marshes, where overlying waters support high nitrate levels. In estuarine sediments most denitrification in sediments occurs as settled organic particles decompose and released ammonium is oxidized to nitrate. Some of this nitrate "escapes" to the overlying water and some is denitrified within the sediment column. Both pathways of denitrification are at work within the Chilmark Pond System.

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

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

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

Overall, coastal sediments are not overlain by nitrate rich waters and the major nitrogen input is via phytoplankton grazing or direct settling. In these systems, on an annual basis, the amount of nitrogen input to sediments is generally higher than the amount of nitrogen release. This net sink results from the burial of reworked refractory organic compounds, sorption of inorganic nitrogen and some denitrification of produced inorganic nitrogen before it can “escape” to the overlying waters. However, this net sink evaluation of coastal sediments is based upon annual fluxes. If seasonality is taken into account, it is clear that sediments undergo periods of net input and net output. The net output is generally during warmer periods and the net input is during colder periods. The result can be an accumulation of nitrogen within late fall, winter, and early spring and a net release during summer. The conceptual model of this seasonality has the sediments acting as a battery with the flux balance controlled by temperature (Figure IV-10).

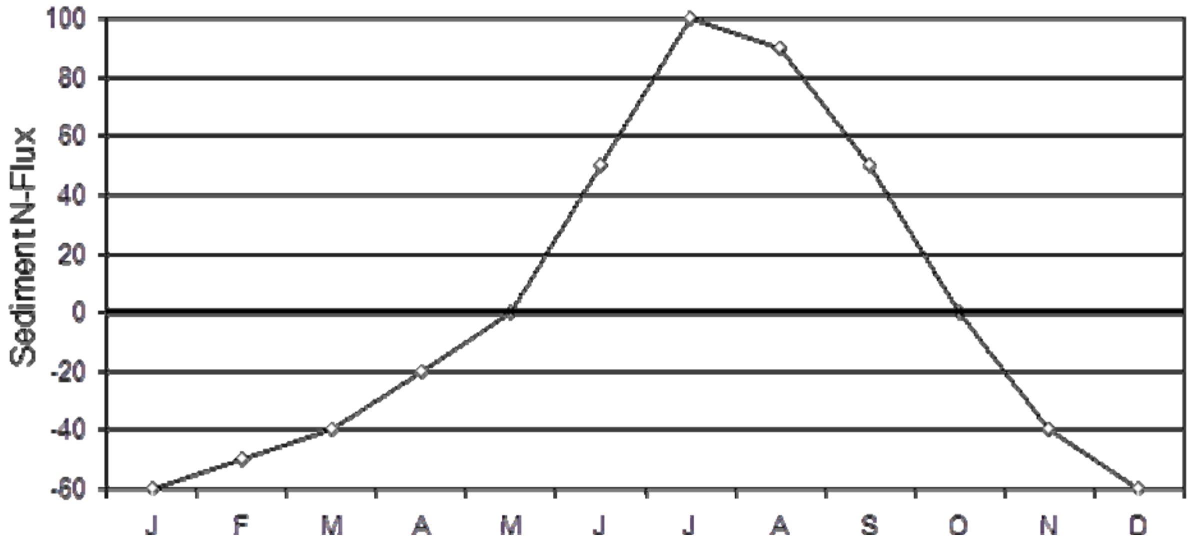


Figure IV-10. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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

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

Sediment sampling was conducted throughout the primary component basins of Chilmark Pond (Upper, Lower, Wades Cove and Gilberts Cove), which comprise the overall Chilmark Pond Embayment System in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model. The distribution of cores in each basin was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content and sediment type and an analysis of each site's tidal flow velocities. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within each sub-embayment.

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

Net nitrogen release or uptake from the sediments within the estuarine portions of the Chilmark Pond Embayment System (227 acres, 92 hectares) were comparable to other similar embayments with similar configuration and flushing rates in southeastern Massachusetts, even though this system is not regularly exposed to tidal flushing. There was a clear pattern of sediment N flux, which followed the phytoplankton biomass in the associated watercolumn. The mid basin of the Lower Chilmark Pond showed a significant net release of nitrogen (23.2 mg N L^{-1} d^{-1}), consistent with the high phytoplankton levels in water discharging from Upper Chilmark Pond and from its own overlying waters (chlorophyll levels of 62 ug L^{-1} and 32 ug L^{-1}, respectively). In contrast the eastern basins of Lower Chilmark Pond, Wades Cove and Gilberts Cove had low-moderate uptake (-15 to 3.5 mg N L^{-1} d^{-1}) and only small levels of release. These levels of sediment nitrogen regeneration are similar to other enclosed basins throughout southeastern Massachusetts. For example, in nearby Tisbury Great Pond, Town Cove supports a low-moderate net release, 23.1 mg N m^{-2} d^{-1}, declining to a relatively consistent rate of 8.8 mg N m^{-2} d^{-1} over the large down-gradient main basin of Tisbury Great Pond. The smaller tributary coves supported low rates of release and even slight uptake: Pear Tree Cove, Tiah Cove and Deep Bottom/Thumb Cove with rates of 0.1 mg N m^{-2} d^{-1}, -1.6 mg N m^{-2} d^{-1}, and 6.5 mg N m^{-2} d^{-1}, respectively. These rates are consistent with the structure of the basins of both Chilmark Pond and Tisbury Great Pond where the predominance of sediments are comprised of soft consolidated mud with an oxidized surface layer generally to ~1 cm depth and do not have microbial mats and accumulations of drift macroalgae. The highest rates of net nitrogen release were measured in Black Point Pond (64 acres), similarly structured to the mid basin of Lower Chilmark Pond (23.2 mg L^{-1} d^{-1}), also with fringing wetlands and showing a moderate net release, 36.9 mg N m^{-2} d^{-1}.

Sediment nitrogen uptake and release rates in Chilmark Pond were also similar to other tidal embayments in the region of similar proportions, particularly those similarly sized and configured. For example, Edgartown Great Pond, located in the same geologic setting and similarly dominated by a large open water lagoon formed behind a barrier beach and only periodically open to tidal exchange with the Atlantic Ocean waters. The large main basin of Edgartown Great Pond (15.2 mg N m^{-2} d^{-1}) showed similar low rates of net release as the mid basin of Chilmark Pond and the five "unrestricted" coves within Edgartown Great Pond generally showed low rates of release and uptake, -16.9 to 7.4 mg N m^{-2} d^{-1} consistent with the eastern basins of the Chilmark Pond Embayment System.

The Chilmark Pond Embayment System supports water column-sediment exchange rates that are also consistent with other embayments within the region that are fully open to tidal exchange. For example in the Lewis Bay System the main basin (also a lagoon) averaged 6.9 mg N M^{-2} d^{-1}. The main basin of Madaket Harbor averaged 6 mg N m^{-2} d^{-1} and the similarly configured West Bay (Three Bays, Barnstable) 4.5 mg N m^{-2} d^{-1}. The few analogous basins in open embayments that are similar to the eastern basins within Chilmark Pond also show similar low rates of net nitrogen uptake/release from their sediments. For example, Eel River and Prince Cove (Three Bays) -6.4 and 10.3 mg N m^{-2} d^{-1}, The Let (Westport River) 20.5 mg N m^{-2} d^{-1}, and Uncle Roberts Cove (Lewis Bay). Based upon the pattern and rate of net nitrogen uptake/release from the sediments in the mid basin and eastern basins of Lower Chilmark Pond and the comparable rates in analogous basins in other estuaries, the measured rates were used in the water quality modeling effort for the component estuarine sub-basins of the Chilmark Pond Embayment System (Section VI). The sediments within the Chilmark Pond Embayment System appear to be in balance with the overlying waters and the nitrogen flux rates consistent with the level of nitrogen loading to this system and periodic exposure to tidal flushing.

System-wide Sediment Nitrogen Release: In a closed basin, such as Chilmark Pond, it is possible to determine the system-wide rate of nitrogen return from the bottom sediments based upon time series water-column total nitrogen data and the rate of external nitrogen loading (watershed + atmosphere). In the case of Chilmark Pond the external loading rate is relatively low for an embayment of this scale in southeastern Massachusetts (21.0 kg N d^{-1}, see Section IV-1), similar to but lower than Tisbury Great Pond and Edgartown Great Pond (58.1 and 41.4 kg N d^{-1}, respectively) but higher than Sesachacha Pond (Nantucket), 4.1 kg N d^{-1}, another periodically opened great salt pond. For comparison, Lewis Bay, Wareham River and Three Bays estuaries have loading rates on the order of 105.8, 130.3 and 146.4 kg N d^{-1} respectively. The low rate of watershed+atmospheric nitrogen input to Chilmark Pond increases the potential sensitivity of using a basin-wide nitrogen mass balance approach to determine the rate of sediment nitrogen flux (Section VI).

Table IV-5. Rates of net nitrogen return from sediments to the overlying waters of the Chilmark Pond Embayment System. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (Section VI). Measurements represent July -August rates. Note that Upper Chilmark Pond (west basin) is freshwater.

Location	Sediment Nitrogen Flux (mg N m^{-2} d^{-1})			Sta. i.d. *
	Mean	S.E.	# sites	
Chilmark Pond Embayment System				
Freshwater				
West or Upper Chilmark Pond-Fresh	-23.2	3.9	3	1,2,3
Estuarine				
East or Lower Chilmark Pond	-15.9	1.9	3	7,13,14,15
East Mid Region Chilmark Pond	25.4	10.4	5	4,5,6,8
Wades Cove	-15.2	5.6	4	9,10,11,12
Gilberts Cove	3.5	0.2	1	16

* Station numbers refer to Figure IV-9.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This section summarizes the field data collection effort and the development of the hydrodynamic model for the Chilmark Pond system (Figure V-1). For this system, the model offers an understanding of water movement from the pond during a breach. It provides the first step towards evaluating water quality, and it is a tool for later determining nitrogen loading "thresholds". Nutrient loading data combined with measured environmental parameters within the system become the basis for an advanced water quality model based on total nitrogen concentrations. This type of model provides a tool for evaluating existing water quality parameters, as well as determining the likely positive impacts of various alternatives for improving health of the pond, facilitating the understanding of how pollutant loading 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 of the hydrodynamic modeling. For example, the spread of pollutants may be analyzed from tidal current information developed by the numerical models.

Coastal ponds like Chilmark Pond 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 within the system 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, rainfall and groundwater flows. Excess nutrients, especially nitrogen, promote phytoplankton blooms, with adverse consequences including low oxygen, shading of submerged aquatic vegetation, and aesthetic problems.

V.1.1 System Physical Setting

Chilmark Pond is set along the southern shoreline of Martha's Vineyard. The layout of the Chilmark Pond system is shown in the topographic map (Figure V-1). The pond has a surface area of approximately 241 acres at high water (Martha's Vineyard Commission, 2001). The pond is fully enclosed, but is periodically opened by means of a trench dug across the beach to drain the pond into the Atlantic Ocean.

Similar systems, sometimes referred to as "blind", "intermittently open", or "seasonally open" estuaries, are also found in Australia, on the west coast of the United States, South America, and India (Stretch and Parkinson, 2006). Perched estuaries are those that have water levels consistently above mean sea level (MSL) and tend to occur on coastlines that have an energetic wave climate with steep beaches and coarse sediments. It is common practice to artificially breach closed ponds/estuaries when water levels become high, typically to prevent flooding of upland properties and to flush the systems from a build-up of contaminants adversely impacting water quality. Other coastal ponds along the south coast of Martha's Vineyard, Nantucket, and the southern shoreline of Massachusetts/Rhode Island are local examples of systems where periodic breaching is a regular facet of pond management.

Figure V-1. Map of the Chilmark Pond estuary system (from United States Geological Survey topographic maps).

V.1.2 System Hydrodynamic Setting

In Chilmark Pond, the hydrodynamic regime is dominated by freshwater inputs to the system from groundwater recharge, surface flow run-off from the watershed, and direct precipitation to the pond's surface. The volume of water in the pond is governed by the balance between additions from freshwater inflow and losses due to evaporation and flow through the southern beach face into the ocean. On average, the inputs are greater than the losses and the pond elevation gradually rises.

When the pond level is deemed high enough, a trench is cut across the southern barrier beach. Because the pond level is higher than the ocean, the pond drains. The initial outflow from the pond causes a relatively small channel to be scoured through the beach and the water level in the pond drops. The ephemeral channel across the beach is a balance between the scouring effect of water flowing through it and the filling effect of sediment transport along the beach. Although Chilmark Pond is large relative to other regional coastal ponds, the wave climate on the southern coast of Martha's Vineyard is one of the most energetic in Massachusetts. As a result, the breach channel typically closes very quickly, sometimes after only minimal tidal exchange has occurred. The result is that these short or failed breaches only remove the top layer of water from the pond. For these failed breaches, there is very little inflow of water from the ocean and little mixing of the nutrient rich water from the pond with low nutrient inflow. As a result, openings that do not allow influx of ocean waters simply lower the water levels and do little to improve the water quality inside the pond.

V.1.3 Pond Management Practices

Water levels in Chilmark Pond are managed by periodic breaching of the barrier beach. Chilmark Pond is breached to the Atlantic Ocean by excavating a trench through the barrier beach at approximately 4 month intervals, similar to other breached ponds on Martha's Vineyard (e.g., Howes, et al., 2007). Typically the pond will reach heights of over one meter above mean sea level before it is breached. Chilmark Pond is similar in form to the other south shore coastal ponds including Tisbury Great Pond, both having elongated coves extending in a northerly direction. Over the two seasons of observations, cuts through the barrier beach closed in less than a week. By example, the June 1999 opening only persisted for about 4 days. In 2000, the opening closed in a matter of a few days as well. The tidal prism as measured during the June 1999 opening event was on the order of 0.45 feet. The inlet was tidal for approximately 4 days (June 6 through June 10). A record of pond breachings between 2011 and present is available from Martha Cottle (Personal communication, 2014). The record (Table V-1) shows that there are typically three/four openings made each year, with an average cumulative total of 30 days open each year. The average duration of all openings in this record is 8 days. Some openings last less than a week, while two in the record lasted for approximately 22 days total.

Table V-1. Annual Chilmark Pond openings between 2011 and 2014, according to Martha Cottle.

Year	2011	2012	2013	2014*
Openings	4	3	4	2
Cumulative days opens	47	23	23	10

*as of May 2014

V.2 HYDRODYNAMIC FIELD DATA COLLECTION AND ANALYSIS

The field data collection portion of this study was performed to characterize the physical properties of Chilmark Pond. Bathymetry data were collected throughout the system so that the structure and volume of the estuary could be accurately represented as a computer hydrodynamic model, and so that flushing rates could be determined for the system. In addition to the bathymetry, tide data were also collected at three locations to run the circulation model with real tides and also to calibrate and verify model performance.

V.2.1. Bathymetry

Bathymetry data (i.e., depth measurements) for the hydrodynamic model of the Chilmark Pond system was assembled from a recent (2011) boat based hydrographic survey. The survey was executed specifically as part of this analysis.

The hydrographic survey of April 2011 was designed to cover the entire main basin of Chilmark Pond, as well as the various coves within the pond. The survey was conducted from a 14' skiff with an installed high precision fathometer (with a depth resolution of approximately 0.1 foot), coupled together with a differential GPS to provide horizontal position measurements accurate to approximately 1-3 feet. As the boat was maneuvered around the pond, digital data output from both the echo sounder fathometer and GPS were logged to a laptop computer which integrated the data to produce a single data set consisting of water depth as a function of geographic position.

The raw measured water depths were merged with water surface elevation measurements to determine bathymetric elevations relative to the North American Vertical Datum 1988 (NAVD88). Once rectified, the finished processed data were archived as 'xyz' files containing x-y horizontal position (in Massachusetts State Plan 1983 coordinates) and vertical elevation of the bottom (z). These xyz files were then interpolated into the finite element mesh used for the hydrodynamic simulations. The tracks followed by the boat during the bathymetry survey are presented in Figure V-2.

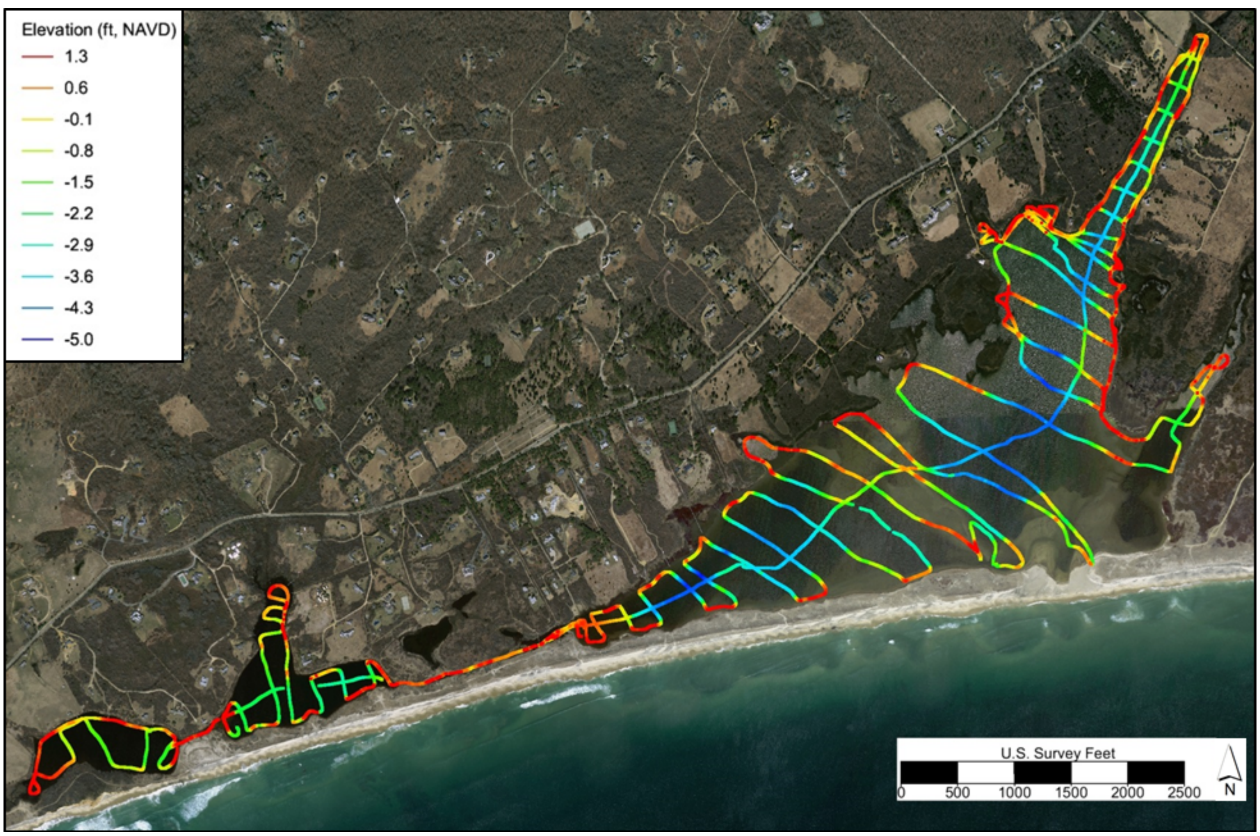


Figure V-2. Bathymetry survey lines and depths (ft., NAVD) for Chilmark Pond.

V.2.2 Tide Data

Tide data records were collected at three stations in the Chilmark Pond system: the north end of Lower Chilmark Pond (main basin), the west end of Lower Chilmark Pond (main basin), and the western sub-embayment commonly referred to as Upper Chilmark Pond. The locations of the stations within the pond are shown in Figure V-3. The Temperature Depth Recorders (TDRs) used to record the tide data were deployed for a 44-day period between May 1 and June 14, 2011, of which approximately 22 days were tidal. The elevation of each gauge was leveled relative to NAVD88. Available data from the Martha's Vineyard Coastal Observatory (MVCO) offshore of Katama Beach was utilized as the offshore boundary condition for the hydrodynamic model. It should be noted for the time period of the tide gauge deployment there were periods when no data was available from the MVCO station. The longest overlapping period of consecutive data when the pond was tidal from the tide gauges and the MVCO station was between May 14, 2011 and May 25, 2011. This period was used for the analyses of the offshore data henceforth, while the Chilmark Pond tide gauge data was analyzed from May 4, 2011 to May 25, 2011.

Once the data were downloaded from each instrument, the water pressure readings were corrected for variations in atmospheric pressure. Hourly atmospheric pressure readings were obtained from the NOAA C-MAN station in Buzzards Bay, interpolated to 10-minute intervals, and subtracted from the pressure readings, resulting in variations in water pressure above the instrument. Further, a (constant) water density value of 1025 kg/m^3 was applied to the readings to convert from pressure units (psi) to head units (for example, feet of water above the tide gauge). Several sensors were surveyed into local benchmarks to provide vertical rectification of the water level; these survey values were used to adjust the water surface to a known vertical datum. The result from each gauge is a time series record representing the variations in water surface elevation relative to the NAVD88 vertical datum. A plot of the observed tide signals is shown below in Figure V-4, where the two stations in the main basin yielded identical tide signals.

To better quantify the tidal changes across the system, from the inlet to inside the system, the standard tide datums were computed from the records. These datums are presented in Table V-2. 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.

Figure V-4 and Table V-2 show a large difference between the tidal range offshore and within the system. In the western and northern portion of the main basin of Chilmark Pond the mean tide range is approximately 2.5 feet less than and only 22% of the mean tide range of the offshore data. In the sub-embayment of Chilmark Pond the mean tide range is only 2% of the offshore mean tide range. The sub-embayment never became tidal as shown by the small tide range. As a result, henceforth the sub-embayment data will not be included in the tidal analyses. For the two stations within the main basin of Chilmark Pond the loss in amplitude are described as tidal attenuation, caused by frictional damping within the system.

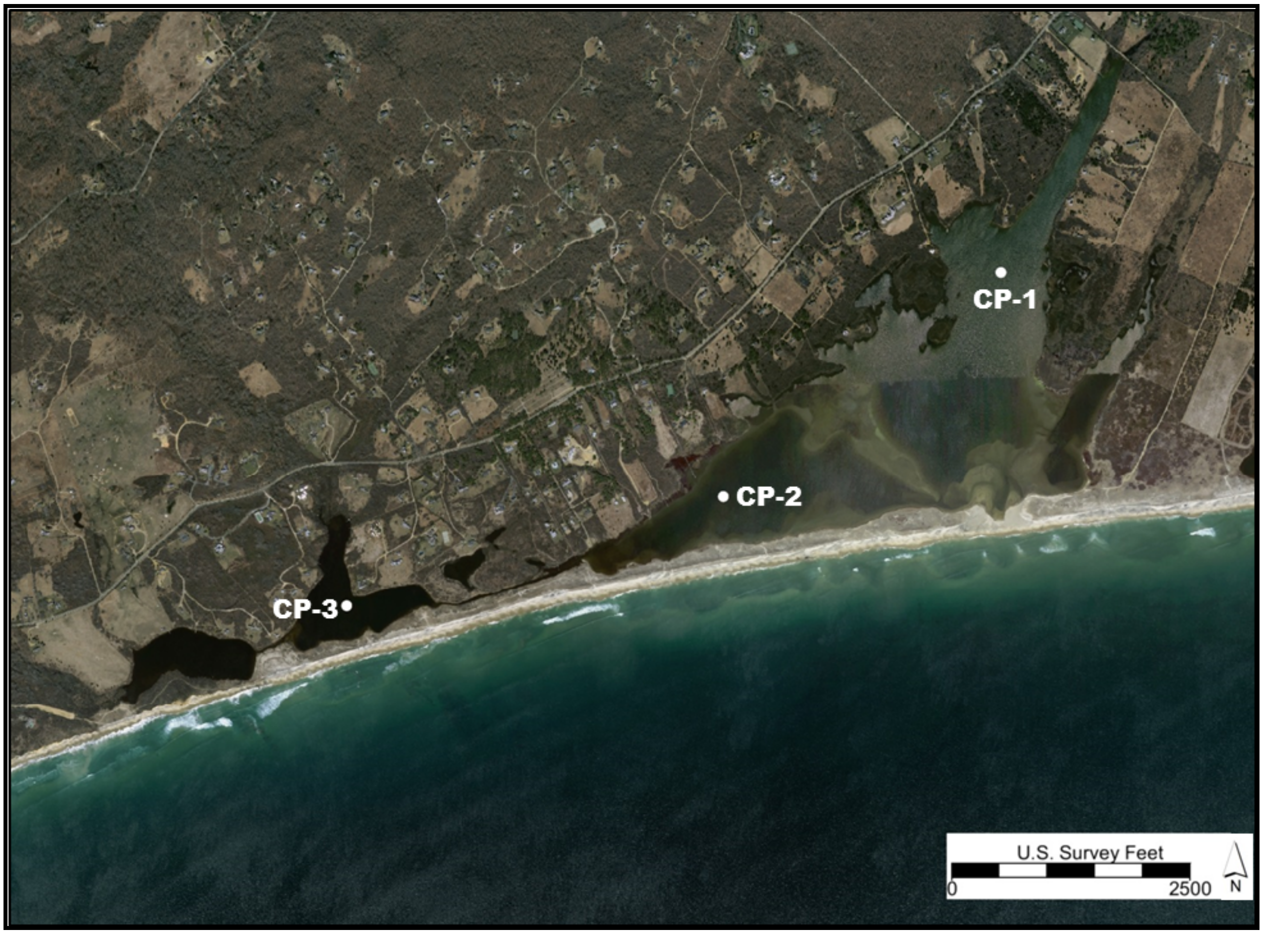


Figure V-3. Aerial photograph of the study region identifying locations of the tide gauges used to measure water level variations throughout the system. The gage locations are shown in white: (CP-1) represents the north end of the main basin, (CP-2) represents the west end of the main basin, and (CP-3) represents the sub-embayments to the west (refer to Figure V-10 for plot of measured tidal stage at each location).

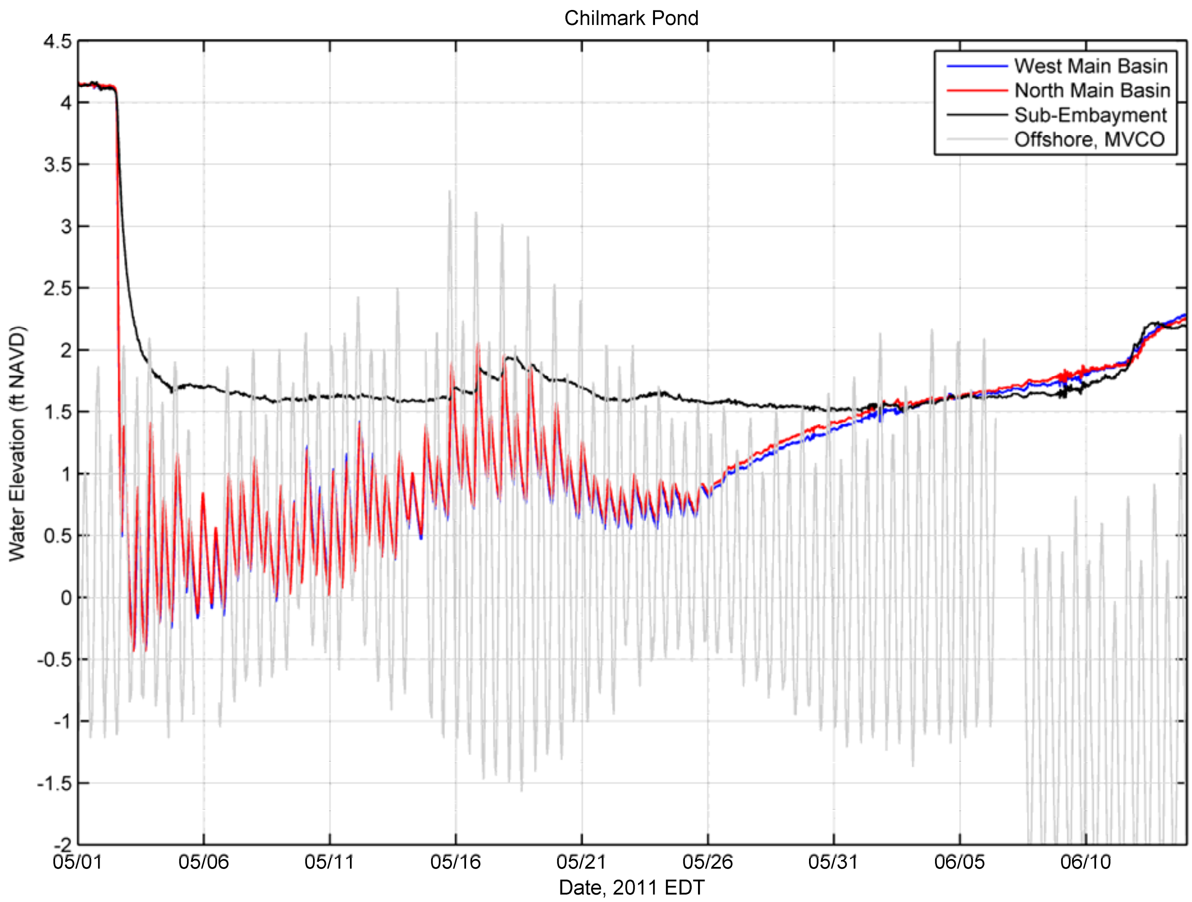


Figure V-4. Tide gage signals measured within Chilmark Pond. The figure represents the entire 44-day record (May 1 to June 14, 2011). The MVCO offshore water levels for the same time period are included (grey). All elevations are referenced to NAVD.

Table V-2. Tide datums computed from the 20 and 11-day records collected in the Chilmark Pond System and offshore, respectively. The record for Chilmark Pond started on May 4, 2011, while the offshore record started May 14, 2011. Datum elevations are given relative to NAVD vertical datum.

Tide Datum	MVCO Offshore Station (feet)	West Main Basin (feet)	North Main Basin (feet)	Sub-Embayment (feet)
Maximum Tide	3.29	2.04	2.07	1.95
MHHW	2.63	1.29	1.33	1.70
MHW	2.17	1.13	1.16	1.69
MTL	0.58	0.78	0.80	1.67
MLW	-1.01	0.43	0.45	1.64
MLLW	-1.13	0.38	0.39	1.62
Minimum Tide	-1.57	-0.25	-0.20	1.57
Mean Range	3.18	0.70	0.71	0.05

A harmonic analysis of the tidal time series was also performed to determine tidal amplitude and phase of the major tidal constituents and provide assessments of hydrodynamic 'efficiency' of the system in terms of tidal attenuation. This analysis also yielded an assessment of the relative influence of non-tidal, or residual, processes (such as wind forcing) on the hydrodynamic characteristics of each system. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The observed astronomical tide is the sum of 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-5. The amplitudes and phase of 21 known tidal constituents result from this procedure. Table V-3 presents the amplitudes of seven tidal constituents computed for the Chilmark Pond station records for the entire time series excluding the period of time immediately after the breach where the tidal influence was minimal.

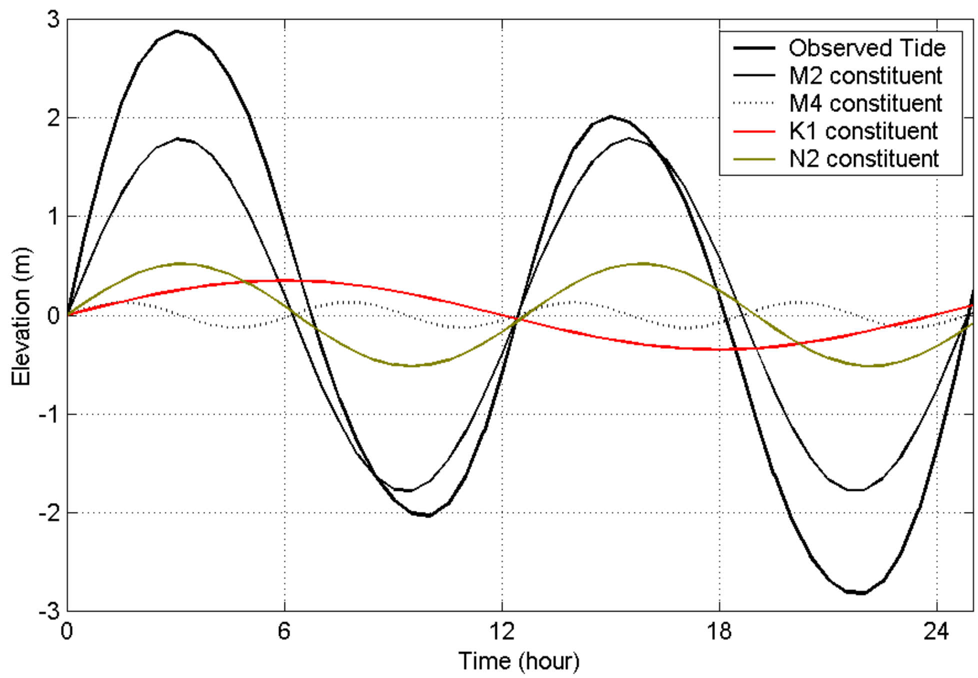


Figure V-5. Example of an observed astronomical tide as the sum of its primary constituents.

Table V-3. Tidal Constituents computed for the tide stations in Chilmark Pond and offshore in Nantucket Sound, from May 4 and May 14 to May 25, 2011, respectively.

	Amplitude (feet)						
Constituent	M_2	M_4	M_6	S_2	N_2	K_1	O_1
Period (hours)	12.42	6.21	4.14	12	12.66	23.93	25.82
MVCO Station	1.29	0.12	0.04	0.23	0.93	0.24	0.23
West Main Basin	0.27	0.05	0.01	0.07	0.09	0.11	0.08
North Main Basin	0.27	0.05	0.01	0.10	0.10	0.10	0.08

An analysis of the MVCO offshore tide data and the records from the tide gauges indicated that 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 1.29 feet. The total range of the M_2 tide is twice the amplitude, or 2.58 feet. This constituent is the largest contributor to the tide throughout the system. The diurnal tides (once daily), K_1 and O_1, possess amplitudes of approximately 0.24 feet and 0.23 feet for the offshore record, respectively. Other semi-diurnal tides, the S_2 (12.00 hour period) and N_2 (12.66 hour period) tides, also contribute to the total tide signal with amplitudes of 0.23 feet and 0.93 feet, respectively.

The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 period for the M_6), resulting from frictional attenuation of the M_2 tide in shallow water. The emergence of these residual tides may be seen within the decay of constituents at the western and northern portion of the main basin of Chilmark Pond stations. While all constituents within the system decay, the M_4 and M_6 constituents decay less, resulting from an energy transfer from the M_2 constituent. Overall, it can be seen that as the total tide range is attenuated through the system there is a corresponding reduction in the amplitude of all of the individual tide constituents.

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

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

As seen in the results presented in Table V-4, the variance of tidal energy was largest in the offshore signal, as should be expected. The analyses also show that tides are responsible for at least 95% of the water level changes at the MVCO station. However, non-tidal influence is quite large within the system in portions of the data. This may be related to effects caused by changes of the recently formed channel.

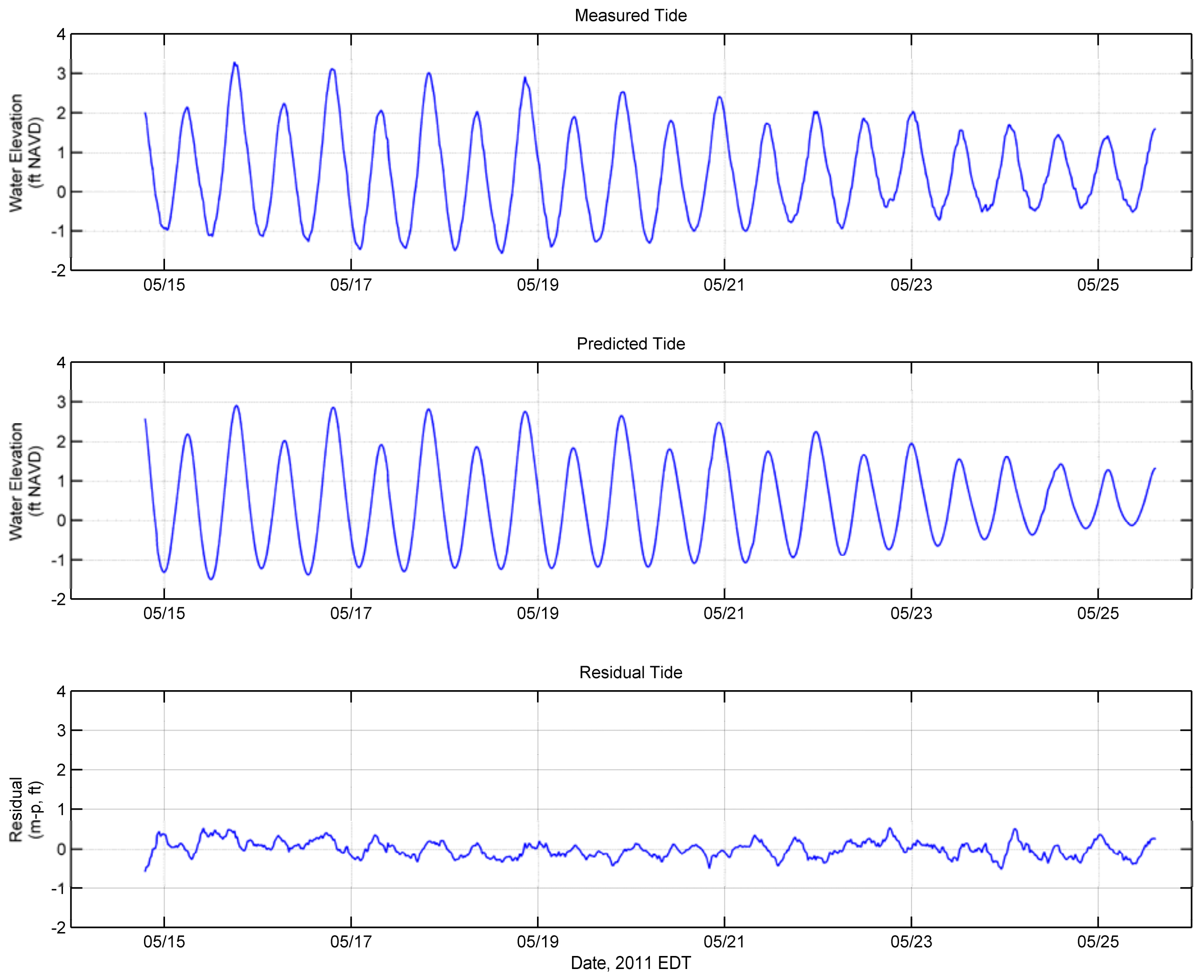


Figure V-6. Measured tide from the MVCO station, with the computed components of astronomical tide resulting from the harmonic analysis, and the resulting non-tidal residual water level.

Table V-4. Percentages of Tidal versus Non-Tidal Energy using a constituent analysis for the record of when Chilmark Pond was tidal.

TDR Location	Total Variance (ft^2)	Tidal (%)	Non-tidal (%)
May 14 through May 25			
MVCO, Offshore	1.29	96.8	3.2
May 4 through May 25			
West Main Basin	0.15	68.1	31.9
North Main Basin	0.15	68.7	31.3

V.3 HYDRODYNAMIC MODEL DEVELOPMENT

The scour of a channel through the beach and the flow of water between the pond and ocean through this channel cannot be directly simulated with the RMA suite of models. Therefore, a computer model independent of RMA-2 was used to simulate the flow through the breach channel. Using this breach model, time varying boundary conditions were developed for RMA-2 model runs of the main portion of Chilmark Pond, up through the inlet channel.

V.3.1 Modeling flow through a breach

When the pond is first opened, the initial trench cut through the beach is scoured out by the rush of water leaving the super-elevated pond. The channel increases in width and depth during this time and over the first few tidal cycles assuming the breach remains open. It would be beyond the scope of this study to model the dynamic growth of the channel during the breach event itself. However, the width and depth of the channel are important variables needed to model the flow between the ocean and Chilmark Pond.

To assist in the determination of the equilibrium size of the Chilmark Pond breach, inlet dimensions from past breaching events were examined using the available historical aerial photographic record. A survey of the aerial record shows that the equilibrated inlet channel width is approximately 80 feet wide, on average. This average width was used to determine the channel scour depth.

To estimate the channel scour depth, the flow rate through the channel is needed. Using the data from the May 2011 breach event and the surface area (241 acres at high water) of the pond, the average maximum flow rate out of the pond was determined to be -380 ft^3/sec.

With the flow rate and channel width established, the channel depth was calculated using an approach described by the U.S. Army Corps of Engineers (USACE) for the analysis of scour depth at tidal inlets (Hughes, 1999). This equation predicts the depth of the channel, given the flow rate, sediment type and channel width as:

$$h = \frac{0.234 q^{8/9}}{\left[g(S-1)\right]^{4/9} d^{1/3}}$$

where h is the elevation of the channel bottom relative to the high water level, q is the flow rate divided by the channel width, S is the specific gravity of the sand and d is the average diameter of the sand. A quartz sand (S = 2.65) of diameter 0.5mm was used to represent the sand in this case.

With the initial pond elevation, offshore tides, channel width, and channel depth established, it is possible to compute water levels in the pond through the draw-down period of the pond after the initial breaching of the inlet and the following period when the pond is open to the ocean and tidal. This computed water level time series can then be compared to the actual measured tide in the pond in order to evaluate whether the channel dimensions determined using the USACE equation has produced a meaningful result that can be used in the development of the RMA-2 hydrodynamic model mesh. To compute a water level time series in the pond, the equation of flow over a broad-crested weir was employed (as described by Hughes, 1999). This equation relates the flow rate through the channel to the channel width and height of water above the channel bottom as:

$$Q = 3.0 b H^{3/2}$$

where Q is the predicted flow rate, b is the channel width and H is the difference in elevation between the high water and the channel bottom.

Using the starting pond level of 4.1 feet NAVD (measured just prior to the May 2011 breach) and the recorded offshore tides, a computer model was created to calculate the time-varying flow through the channel. The pond level and offshore tide height every 10 minutes was input into the model and the flow rate was calculated. Multiplying the flow rate by the time step yields the total volume of water moving through the channel. Knowing the surface area of the pond, the change in pond surface elevation is calculated at each time step.

The comparison between the field data and the broad-crested weir model is shown in Figure V-7. Model R^2 correlation and RMS error were calculated for the first few days when the inlet was open and flushing efficiently enough to maintain the equilibrium dimensions of the inlet. The R^2 correlation between measurements and model output is 0.85 and the RMS error is 0.29 feet. These comparisons show that the dimensions of the equilibrated inlet channel determined from the aerial record and using the USACE scour depth methodology do provide a useful approximation that can be used to develop the inlet included in the RMA-2 hydrodynamic model mesh.

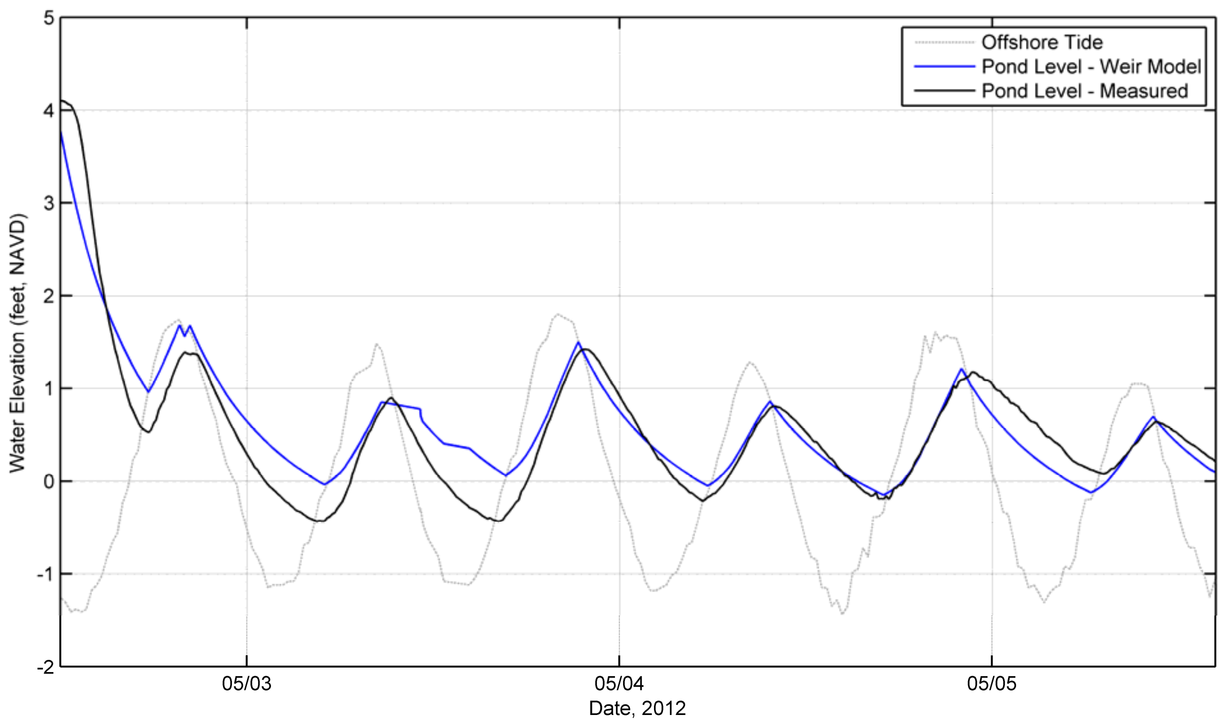


Figure V-7. A comparison of the broad-crested weir model results with the recorded pond elevations during the breach event at Chilmark Pond.

V.3.2 RMA-2 Model Theory

Applied Coastal utilized a state-of-the-art computer model to evaluate tidal flushing during periods when Chilmark Pond is open to the Atlantic Ocean. 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 for estuary systems in southeast Massachusetts, including systems in Chatham, Falmouth's 'finger' ponds, and Popponesset Bay, as well as the Island of Nantucket.

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. Additionally, the graphic pre- and post-processing routines were updated by Brigham Young University through a package called the Surfacewater Modeling System or SMS (BYU, 1998). Graphics generated in support of this report were primarily 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 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 it is unconditionally stable. Once the equations are solved, corrections to the initial estimate of velocity and water elevation are employed, and the equations are re-solved until the convergence criteria is met.

V.3.3 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using 2009 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 Chilmark Pond grid based on the north main basin tide data. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several model calibration simulations for the system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.3.3.1 Grid generation

The grid generation process was assisted through the use of the SMS package. The digital shoreline and bathymetry data were imported to SMS, and a finite element grid was generated to represent the pond with 2282 elements and 7048 nodes (Figure V-8). All regions in the system were represented by two-dimensional (depth-averaged) elements. The finite element grid for the system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties within the estuary. Fine resolution was required to simulate the channel constrictions (e.g., at the creek connecting the main basin and the sub-embayment) that significantly impact the estuarine hydrodynamics. Reference water depths at each node of the model were interpreted from bathymetry data obtained in the 2011 field survey. The maximum nodal depth within the pond is -4.85 ft. NAVD. The bathymetry of the completed model grid mesh of the Chilmark Pond system is shown in Figure V-9. As described previously in this section (V.4.1), the inlet width and depth used in the model are based on the available aerial photographic record and the results of the USACE weir model computations. The model computed water elevation and velocity at each node in the model domain can therefore be determined in order to characterize circulation in the system.

Grid resolution is governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability in each region. Smaller cross channel node spacing in the river channels was designed to provide a more detailed analysis in these regions of rapidly varying velocities and bathymetry. Widely spaced nodes were utilized in areas where velocity gradients were likely to be less acute; for example, on marsh plains and in broad, deep channel sections in the model domain. Appropriate implementation of wider node spacing and larger elements reduced computer run time with no sacrifice of accuracy.

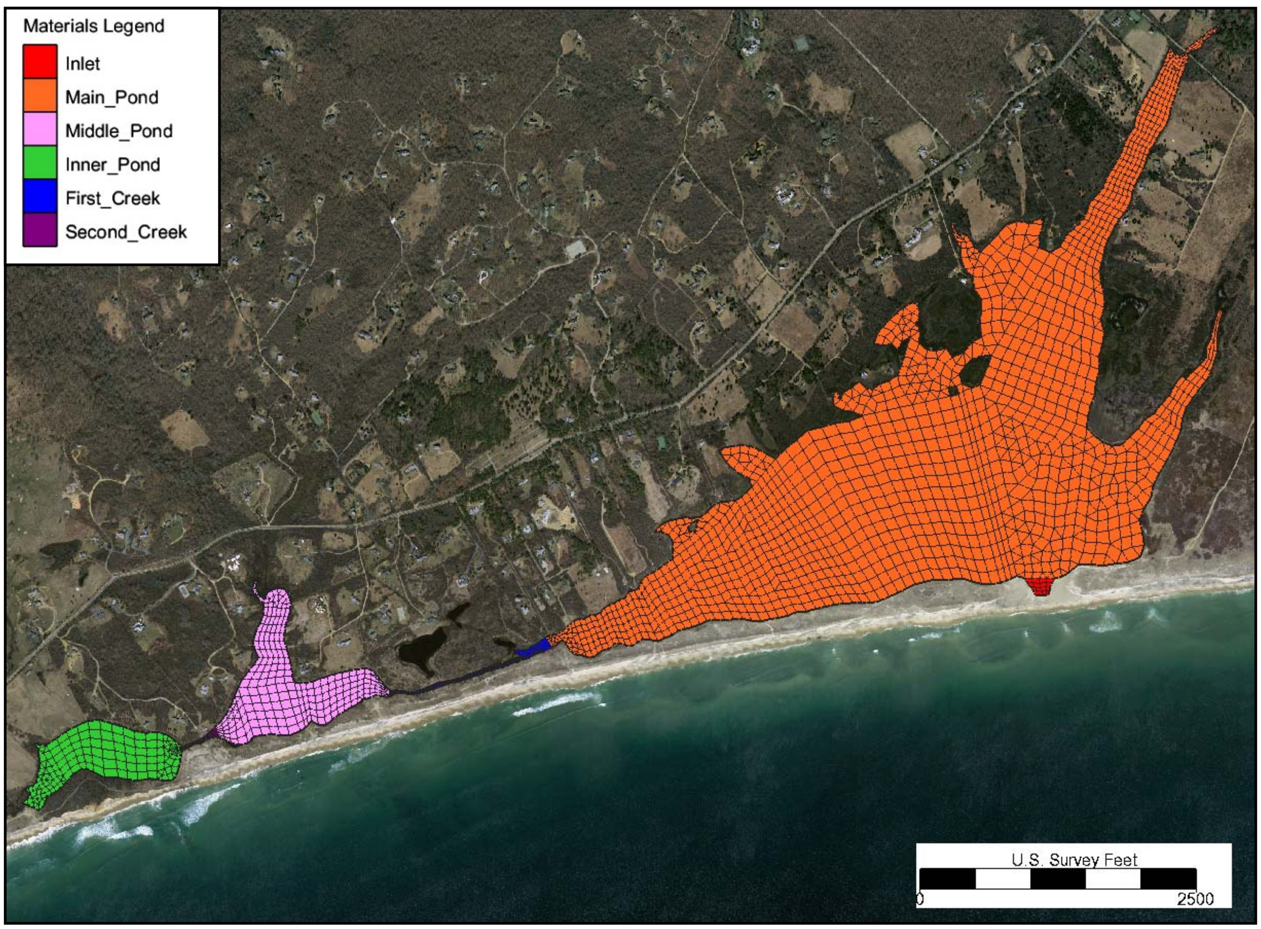


Figure V-8. The model finite element mesh developed for the Chilmark Pond estuary system. The model seaward boundary was specified with a forcing function consisting of water elevation measurements obtained in Chilmark Pond.

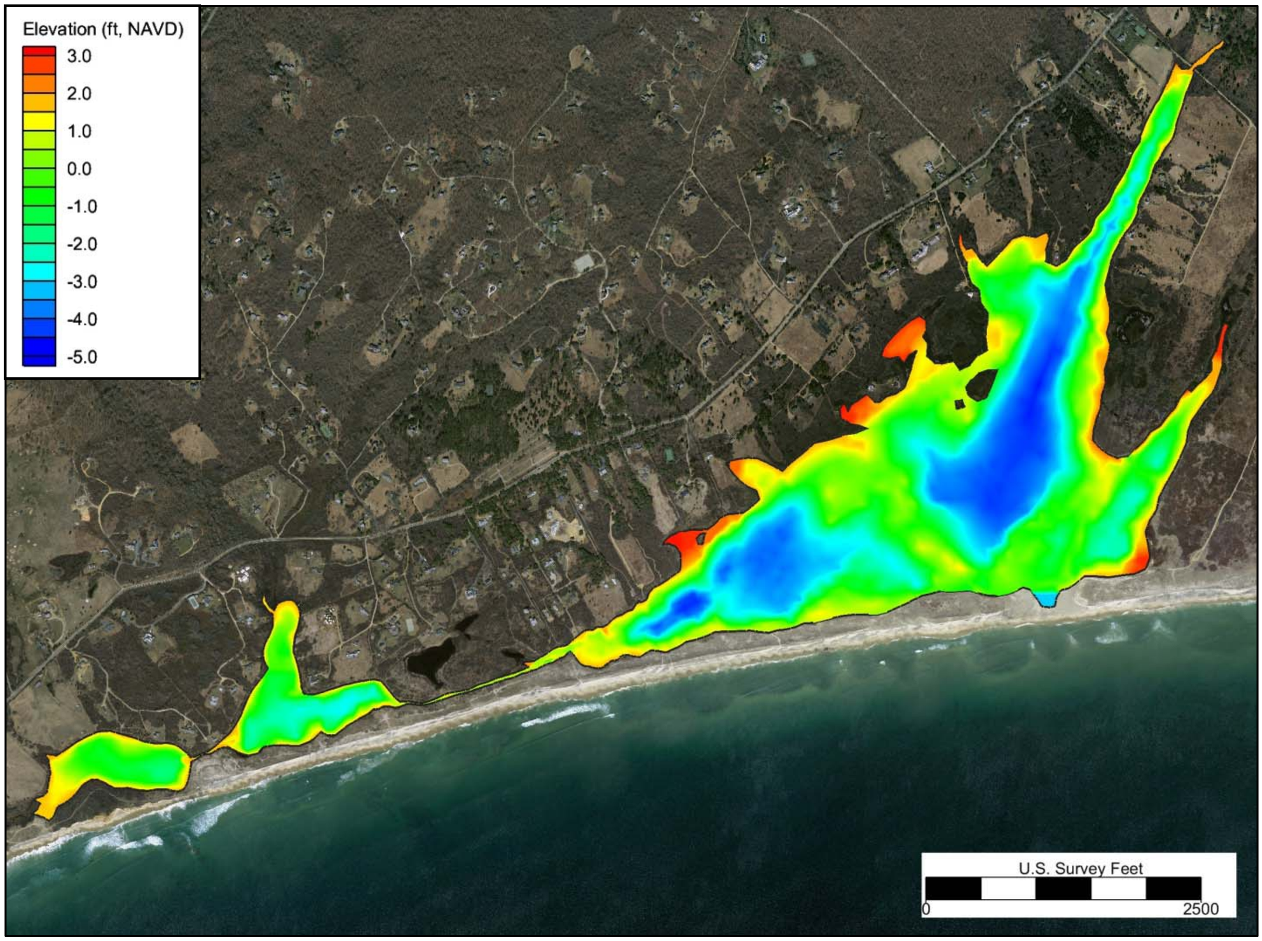


Figure V-9. Bathymetry data interpolated to the finite element mesh used with the RMA-2 hydrodynamic model. The elevations are relative to North American Vertical Datum 1988.

V.3.3.2 Boundary condition specification

Three types of boundary conditions were employed for the RMA-2 model of the Chilmark Pond system: 1) "slip" boundaries, 2) tidal elevation boundaries, and 3) freshwater inflow. 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 freshwater boundary condition was specified at Fulling Mill Brook (East and West) and Mill Brook.

The model was forced at the open boundary using water elevations measurements obtained within the northern portion of Chilmark Pond (described in section V.3.2). This measured time series consists of all physical processes affecting variations of water level: tides, winds, and other non-tidal oscillations of the sea surface. The rise and fall of the tide in the ocean is the primary driving force for estuarine circulation. Dynamic (time-varying) model simulations specified a new water surface elevation at the offshore boundary every 10 minutes. The model specifies the water elevation at the offshore boundary, and uses this value to calculate water elevations at every nodal point within the system, adjusting each value according to solutions of the model equations. Changing water levels in the ocean produce variations in surface slopes within the estuary; these slopes drive water either into the system (if water is higher offshore) or out of the system (if water levels fall in the Pond).

V.3.3.3 Calibration

After developing the finite element grid, and specifying boundary conditions, the model for the Chilmark Pond system was calibrated. Calibration ensured that the model predicts accurately what was observed in nature during the field measurement program. Numerous model simulations were required to calibrate the model, with each run varying specific parameters such as friction and turbulent exchange coefficients to achieve a best fit to the data.

Calibration of the flushing model required a close match between the modeled and measured tides in the sub-embayment where tides were measured. Initially, the model was calibrated by the visual agreement between modeled and measured tides. To refine the calibration procedure, water elevations were output from the model at the same locations in the estuary where tide gauges were installed. These data were processed to calculate standard error as well harmonic constituents (of both measured and modeled data) over the seven-day calibration period. The amplitude and phase of four constituents (M_2, M_4, M_6, and K_1) were compared and the corresponding errors for each were calculated. The intent of the calibration procedure is to minimize the error in amplitude and phase of the individual constituents. In general, minimization of the M_2 amplitude and phase becomes the highest priority, since this is the dominant constituent. Emphasis is also placed on the M_4 constituent, as this constituent has the greatest impact on the degree of tidal distortion within the system, and provides the unique shape of the modified tide wave at various points in the system.

The calibration was performed for an approximate five-day period, beginning 1700 hours EDT May 16, 2011 and ending 1700 EDT May 21, 2011. This time period covers a period when the pond was tidal after the breach in early May 2011. Additionally, this model included a 60 hour model spin-up period that began prior to May 16, 2011. In total, a 9-tide cycle period was used for model calibration. This representative time period was selected because it included tidal conditions where the wind-induced portion of the signals (i.e. the residual) was minimal, hence more typical of tidal circulation within the estuary. The selected time period also spanned the transition from spring (bi-monthly maximum) to neap (bi-monthly minimum) tide ranges, which is representative of average tidal conditions in the embayment system. Throughout the selected 5 day period, the tide ranged approximately 1.5 feet from minimum low to maximum high tides. The ability to model a range of flow conditions is a primary advantage of a numerical tidal flushing model. Modeled tides were evaluated for time (phase) lag and height damping of dominant tidal constituents. The calibrated model was used to analyze existing detailed flow patterns and compute residence times.

V.3.3.3.a Friction Coefficients

Friction inhibits flow along the bottom of estuary channels or other regions where water depths can become shallow and velocities relatively high. Friction is a measure of the channel roughness and can cause both significant amplitude attenuation 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 coefficients between 0.02 and 0.025 were specified for all element material types. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels with pools and shoals with higher friction (Henderson, 1966). To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning’s coefficients were matched to bottom type. Final model calibration runs incorporated various specific values for Manning’s friction coefficients, depending upon flow damping characteristics of separate regions within the estuary. Manning's values for different bottom types were initially selected based on 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 Table V-5 for the different regions of the pond specified by the different grid material types of the numerical grid (Figure V-8).

V.3.3.3.b Turbulent exchange coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swifter, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). In most cases, the modeled systems were relatively insensitive to turbulent exchange coefficients because there were no regions of strong turbulent flow. Typically, model turbulence coefficients were set between 10 and 100 lb.-sec/ft^2 (Table V-5).

Table V-5. Manning's Roughness and turbulence exchange (D) coefficients used in simulations of the Chilmark Pond system. These embayment delineations correspond to the material type areas shown in Figure V-8.

System Embayment	Bottom Friction	D (lb.-sec/ft^2)
Inlet	0.02	100
Main Pond	0.02	10
First Creek	0.1	10
Middle Pond	0.05	10
Second Creek	0.05	10
Inner Pond	0.05	10

V.3.3.3.c Marsh porosity processes

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

V.3.3.3.d Comparison of modeled tides and measured tide data

Several calibration model runs were performed to determine how changes to various parameters (e.g. friction and turbulent exchange coefficients) affected the model results. These trial runs achieved good agreement between the model simulations and the field data. Comparison plots of modeled versus measured water levels at the two gauge locations is presented in Figure V-10. At all gauging stations RMS errors were less than 0.07 ft. (<0.5 inches) and the computed R^2 correlation was better than 0.98 for the two stations in the main basin. Additionally, a tidal constituent comparison was done to further quantify the accuracy of the model. As previously stated, the station within the sub-embayment never became completely tidal and as a result, was not included in the tidal constituent comparison. Errors between the model and observed tide constituents were less than 0.12 inches for the two locations within the main basin, suggesting the model accurately predicts tidal hydrodynamics within Chilmark Pond. Measured tidal constituent amplitudes and time lags (ϕ_{lag}) for the calibration time period are shown in Table V-6. The constituent values for the calibration time period differ from those in Tables V-3 because constituents were computed for only 5 days, rather than the entire 21-day period represented in Tables V-3. Errors associated with tidal constituent height were on the order of hundredths of one foot, which was an order of magnitude better than the accuracy of the tide gauges (±0.12 ft.). Time lag errors of the M_2 were of the order of the tide gauge and model time step and small considering that the period of this tidal component is 12.42 hours long. This small error indicates a good agreement between the model and empirical data, especially considering that the inlet is quasi-stable, unlike most other estuaries that have been modeled as a part of the MEP.

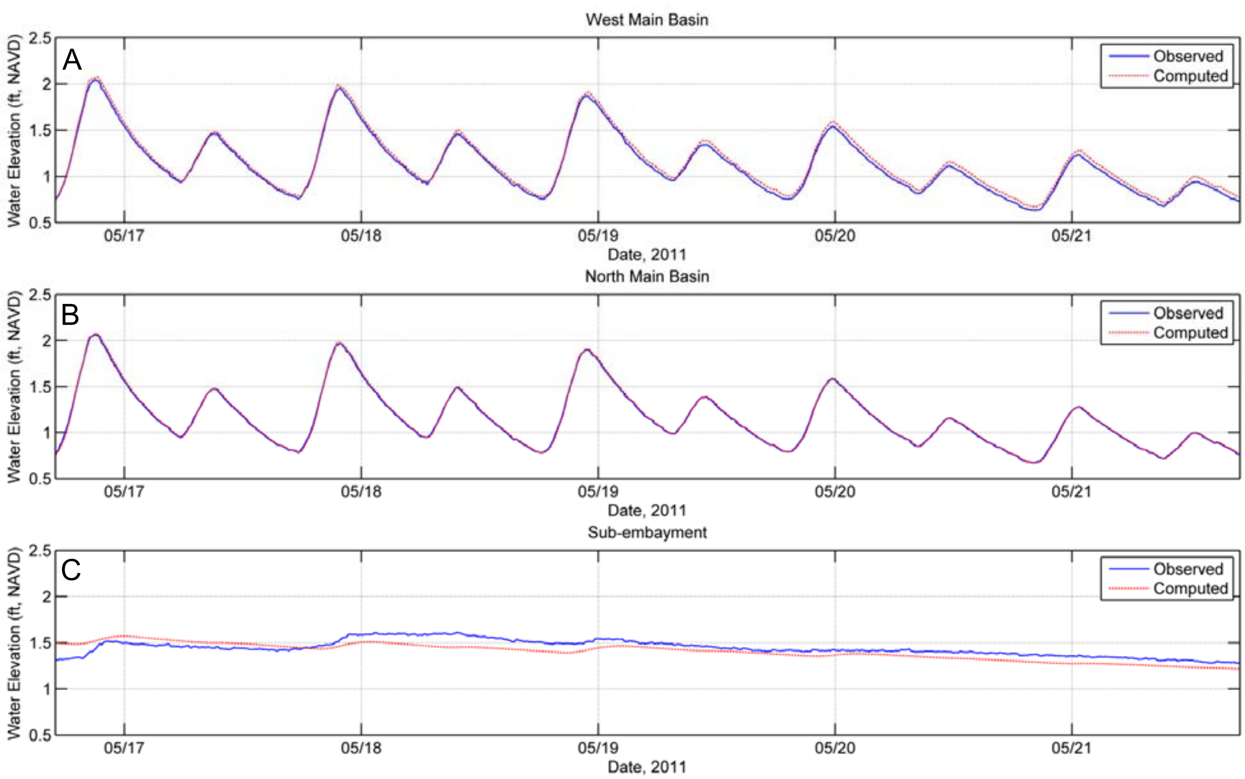


Figure V-10. Comparison of model output and measured tides for the A) West Main Basin station (CP-2) B) North Main Basin station (CP-1) C) and the Sub-embayment station (CP-3) for the final model run (refer to Figure V-3 for gauging station locations).

Table V-6. Tidal constituents for measured water level data and model output, with model error amplitudes, for Chilmark Pond during the model calibration period.

Modeled					
	Constituent Amplitude (ft.)				Constituent Phase (degrees)
Location	M_2	M_4	M_6	K_1	M_2
West Main Basin	0.29	0.09	0.02	0.20	150.98
North Main Basin	0.29	0.09	0.02	0.20	150.89
Measured					
	Constituent Amplitude (ft.)				Constituent Phase (degrees)
Location	M_2	M_4	M_6	K_1	M_2
West Main Basin	0.29	0.09	0.02	0.20	143.74
North Main Basin	0.29	0.10	0.02	0.21	142.93
Error					
	Error Amplitude (ft.)				Error (minutes)
Location	M_2	M_4	M_6	K_1	M_2
West Main Basin	0.00	0.00	0.00	0.00	15.00
North Main Basin	0.00	-0.01	0.00	-0.01	16.48

V.3.4 Flushing Characteristics

During a sustained breach event, the freshwater inflow is negligible in comparison to the tidal exchange through the temporary inlet. A rising tide offshore creates a slope in water surface from the ocean into the upper-most reaches of the modeled system. Consequently, water flows into (floods) the system. Similarly, the pond 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 estuarine 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).

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 Chilmark Pond system this approach is applicable, since it assumes the main system has relatively lower quality water relative to the ocean.

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

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in the system, model results were used to compute a residence time for the entire estuary. The average volume calculated for Chilmark Pond is 30,013,150 ft^3 with a tidal prism of 6,699,878 ft^3 when the inlet is open. This results in a residence time of approximately 2.3 days. This modest residence time provides some confidence that the temporary channel allows enough exchange to significantly improve water quality during a typical breach event.

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

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

VI. WATER QUALITY MODELING

The water quality modeling analysis approach that has been typically used for other systems that have been studied as part of the Massachusetts Estuaries Project was slightly modified for Chilmark Pond. This modified approach has been applied to other estuary systems that are periodically breached, like Edgartown Great Pond, also located on the south shore of the Vineyard, and Sesachacha Pond, on the eastern shore of Nantucket.

This system differs from most other systems modeled as part of the MEP because it does not have an inlet that is open at all times to the ocean. Water quality in the Pond is managed presently by periodically opening an inlet to the ocean. For past breaches, the length of time that the inlet remains open after it is breached varies between less than 1 to 3 weeks, based on observations of openings made from 2011 through 2014. On average, the pond is open 30 days total a year, which means it is closed off from the ocean more than 90% of the time.

Because Chilmark Pond is actively managed in such a fashion, the water quality analysis has to include methods for determining conditions in the Pond at times when it is both open and closed to tidal exchange with the ocean. During times when the Pond inlet is breached, the RMA-4 model was used to model water quality constituent dispersion throughout the Pond's main basin and the series of coves. During the long periods when the breach is closed, a simple mass balance model was developed. As used together in this analysis, these two modeling techniques accurately simulate conditions in the Pond throughout the critical summer months, and provide a method of investigating alternatives to manage pond health.

VI.1 DATA SOURCES FOR THE MODEL

Several different data types and calculations are required to support the water quality modeling effort for the Chilmark Pond 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 salinity and nitrogen in the water column.

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayments

Field measurements and hydrodynamic modeling of the embayment provide essential preparatory input to the water quality model development effort. The pond breach simulation discussed in Chapter V is an important tool for determining the water quality dynamics that are in effect presently, and also for investigating how possibly the pond could be managed differently in the future to further improve water quality conditions. 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. For each of the modeling scenarios presented in this chapter, the breach model was run using tide data record measured in Chilmark Pond due to missing data at the Martha's Vineyard Coastal Observatory (MVCO). These tide data were used as boundary condition used to force the RMA-2 model of Chilmark Pond.

VI.1.2 Nitrogen Loading to the Embayments

Two primary nitrogen loads to Chilmark Pond are included in this modeling study: external loads from the watersheds and internal loads from the sediments. In addition to these two nitrogen loads to the pond, the Atlantic Ocean is a background source of nitrogen that is important to include in the model when simulating periods when the pond inlet is open and flushing. This load is represented as a constant concentration along the seaward boundary of the RMA-4 model grid during the pond breach simulation period.

VI.1.3 Measured Nitrogen Concentrations in the Embayments

In order to create a model that realistically simulates salinity and total nitrogen concentrations in Chilmark in response to the existing flushing conditions and loadings, it was necessary to calibrate the model to actual measurements. The refined and approved data for the monitoring station used in the water quality modeling effort are presented in Table VI-1. The station location is indicated in the area map presented in Figure VI-1. Only one year of data, 2004, of realistic data was available for Chilmark Pond which is less than the typical required three years of baseline field data for the MEP analysis.

Table VI-1. Measured nitrogen concentrations for Chilmark Pond. TN data represented in this table were collected from 2004 in Chilmark Pond. The offshore Atlantic Ocean data (offshore Pleasant Bay Inlet) are from the summer of 2005.

Sampling Station Location	Total Nitrogen		
	Mean (mg/L)	s.d. all data (mg/L)	N
Wades Cove Upper (CHP-1)	0.608	0.119	4
Atlantic Ocean	0.232	0.044	17

VI.2 MODEL DESCRIPTION AND APPLICATION

The overall approach used in the analysis of Chilmark Pond involves first developing a salinity model of the Pond. Salinity is a conservative water quality constituent, meaning that is has no active sources or sinks other than tidal exchange with the ocean. Because salinity data are conservative, they are excellent calibration data for systems such as Chilmark Pond. In such simple systems it is an easy task to compute water recharge and rainfall rates based on the observed salinity record.

The Chilmark Pond analysis requires that both periods when the inlet is open and closed be considered, so a two-part approach was developed. The initial period (when Chilmark Pond inlet is breached in the summer and there is tidal exchange with the ocean) is modeled using the RMA-4 dispersion model. The following period when the inlet is closed, and Chilmark Pond behaves like a simple reservoir, is simulated using a simple mass balance model which considers freshwater inputs and constituent mass flux into the pond, which is zero for the salinity simulation seeing as there was no salt water inflow throughout the simulation period.

With a calibrated salinity model, a verification of the model is performed using total nitrogen, which is as a non-conservative constituent. For TN, bottom sediments act as a source or sink of nitrogen, based on local biochemical characteristics. The TN model considers summertime loading 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 Martha’s Vineyard Commission watershed loading analysis, as well as the measured bottom sediment nitrogen fluxes. Water column nitrogen measurements were utilized as model boundaries and as calibration data.

Figure VI-1. USGS topographic map showing the monitoring station location in Chilmark Pond that was used in the water quality analysis.

VI.2.1 Model Formulation

VI.2.1.1 Dispersion Model

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of water quality constituent dispersion in Chilmark Pond during the periods when it is open. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to simulate the fluid dynamics of the Pond. 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 other Massachusetts estuarine systems such as Pleasant Bay (Howes *et al*., 2006); Falmouth (Howes *et al*., 2005); and Mashpee, MA (Howes *et al*., 2004), and including other periodically breached coastal ponds like Sesachacha Pond on Nantucket Island (Howes *et al*., 2006).

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

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

where c in the water quality constituent concentration; t is time; u and v are the velocities in the x and y directions, respectively; D_o and D_{ee} 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 nitrogen 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 time varying total nitrogen concentrations throughout Pond during an inlet opening. For demonstration purposes, the model was used to simulate a 30 day opening to investigate how salinity and total nitrogen change with opening duration, although openings are on average only open for 8 days.

VI.2.1.2 Mass Balance Model

During the extended periods when Chilmark Pond is closed off from the Ocean, the system is modeled as a simple well mixed reservoir. The concentration c is a function of time t, and can be determined using the relationship

$$c(t)=\frac{m_o+t\,\dfrac{dm}{dt}}{V_o+t\,\dfrac{dV}{dt}},$$

where m is the total mass of the modeled constituent, V is the volume of the pond and the subscript o is used to designate the initial conditions. For the salinity model, the mass flux of salt (dm/dt) into the pond is zero. Using salinity data record from the summer of 2004 and the SMAST measured recharge rate, a mass balance analysis of salt was performed to determine the concentration of salt. This flow is the only possible sink for salinity in the pond system. The one year used for this analysis was selected because it was the only year with adequate salinity data to base the simulation. The breechings of the pond initially raised salinities in the pond, and over the course of the summer, salinities slowly dropped as the pond was diluted by ground water recharge and rainfall. For each simulation, the model was tuned to replicate both the fall in salinity and rise in pond surface elevation.

By this analysis, the freshwater recharge rate was determined to be 10.36 ft^3/sec including the freshwater inflow, rain, and groundwater flow. Given the model's good agreement with this recharge rate, it was assumed that the salinity sink through the barrier beach was to be insignificant. The net flux of salt is therefore zero, and the net volume of flux of water is simply the recharge rate plus direct rainfall minus evaporation. For the TN model, the mass flux of nitrogen is set to the sum of the watershed, atmospheric and benthic loads.

VI.2.2 Boundary Condition Specification

Mass loading of nitrogen into the 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 combined watershed, direct atmospheric deposition and benthic flux loads for the whole Pond were evenly distributed across the cells that make up the RMA computational grid.

The loadings used to model present conditions in Chilmark Pond 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, resulting in a total flux for the system (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. The benthic flux presented in Table VI-2 represents the net flux for the entire embayment. For present conditions, the benthic flux is negative within both of the embayments in Chilmark Pond, which indicates that they are a nitrogen sink.

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified for the dispersion model. The model uses concentrations at the open boundary during the flooding tide periods of the RMA-4 model simulations. TN concentrations of the incoming water are set at the value designated for the open boundary. The TN boundary concentration in the Atlantic Ocean region offshore the Pond was set at 0.232 mg/L, based on SMAST data collected offshore Pleasant Bay in the summer of 2005. As there is no offshore station relative to Chilmark Pond, the offshore station off Pleasant Bay is representative of Atlantic Ocean water that would be flowing into the Chilmark Pond system during a breach event. For the salinity model, the offshore concentration was set at 32.3 ppt.

Table VI-2. Embayment and surface water loads used for total nitrogen modeling of Chilmark Pond, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent the present loading conditions for the listed sub-embayments.

Embayment	Watershed load (kg/day)	Direct atmospheric deposition (kg/day)	Benthic flux net (kg/day)
Chilmark Pond East	5.485	3.260	-0.273
Chilmark Pond West	11.614	0.656	-3.100

VI.2.3 Development of Present Conditions Model

To develop the water quality model of present conditions for Chilmark Pond, the RMA-4 dispersion model and the mass balance model are typically separately developed to simulate salinities in the Pond. However, due to limited salinity data corresponding to known openings only the mass balance model was used to determine the present conditions.

For time periods when the pond was closed off from the ocean, the mass balance model was used as previously stated. This model requires an initial salinity and pond volume, as well as a net fresh water flux. The mass balance model was calibrated using data from summer 2004, which is a period where good-quality contemporaneous TN and salinity data exist. The initial salinity (27.1 ppt) was measured on July 12. The initial Pond volume was determined to be 18,891,000 ft^3, based on results from the hydrodynamic model The net freshwater input to the Pond for the modeled time period was determined to be 8.4 ft^3/sec, which results in the minimum model error.

The comparison of modeled versus measured salinities are presented in Figure VI-2. The comparison shows that the combined mass balance model is able to simulate both salinities with a high degree of skill, with an R^2 correlation of 0.77, and an RMS error of 2.0 ppt. The model is very sensitive to the applied recharge rate. When the recharge rate estimated by the MV Commission is applied to the model (10.4 ft^3/sec), resulting salinities are much lower than the measured data. A tabulation of the salinity calibration and elevation verification data is presented in Table VI-4.

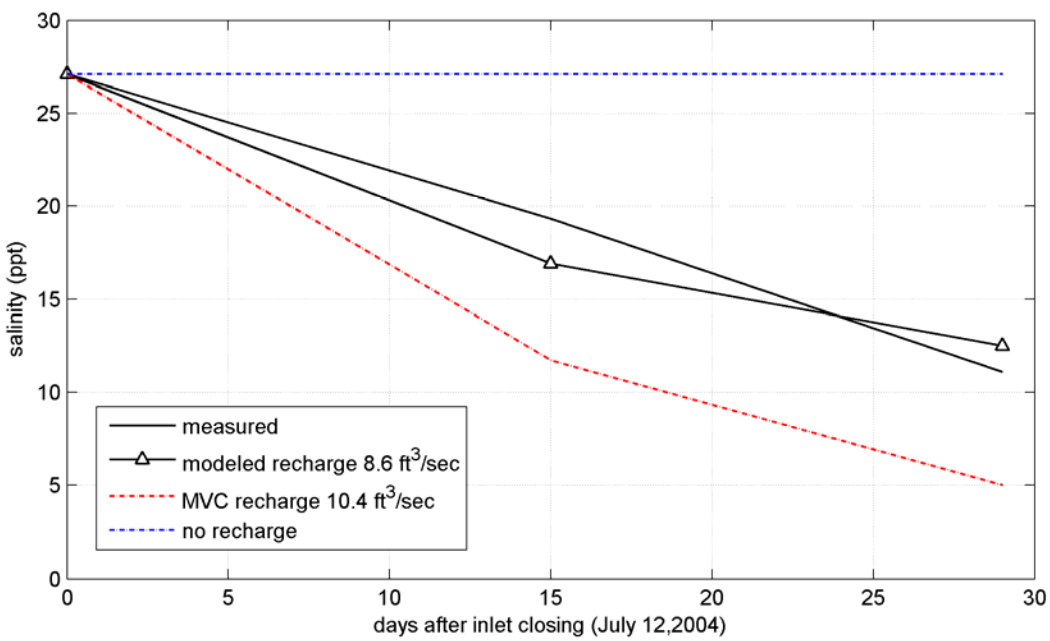


Figure VI-2. Comparison of measured (black line with circles) and modeled (red line with triangles) salinities through the summer of 2004, after the breaching of an inlet to the Atlantic Ocean. This period through the summer was simulated using the mass balance model.

Table VI-3. Comparison of measured data and model output for summer 2004 mass balance model calibration-verification period.

Date, 2004	measured salinity (ppt)	measured TN (mg/L)	modeled salinity (ppt)	modeled TN (mg/L)
July 12	27.1	0.45	27.1	0.45
July 27	19.3	0.61	16.9	0.60
August 10	11.1	0.63	12.5	0.66
September 8	NA	0.74	8.1	0.72

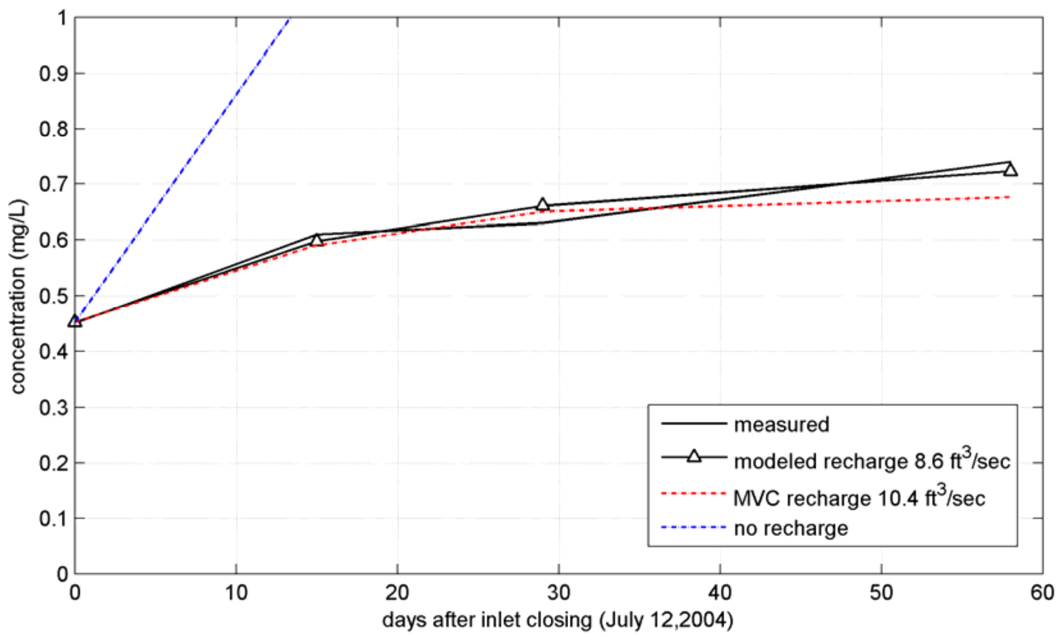


Figure VI-3. Comparison of measured (black line with circles) and modeled (red line with triangles) TN concentrations through the summer of 2004, after the breaching of an inlet to the Atlantic Ocean. This period through the summer was simulated using the mass balance model.

VI.2.4 Total Nitrogen Model Development

With the completion of the salinity model, it was possible to use the components to simulate total nitrogen, which is a water quality constituent that is completely independent of salinity.

The mass balance model was used to simulate the period following the breach closure in July 2004. This model used the same N mass loading rates as the dispersion model and included the same 8.4 ft^3/sec freshwater input used in the calibration of the salinity model.

Model output is compared to measurements for the summer 2004 period in Figure VI-3. Similar to the results of the salinity model, the comparison demonstrates a high degree of modeling skill, with an R^2 correlation of 0.86 and an RMS error of 0.02 mg/L. Like the salinity analysis, the model is very sensitive to the applied recharge rate, and these results indicate that the recharge rate used to simulate this period in 2004 is close to estimated average rate.

The Chilmark Pond RMA-4 model was run to simulate TN concentrations through the average 8-day breach period. An open ocean TN concentration of 0.232 mg/L was used together with the nitrogen mass loading rates presented in Table VI-2 for Chilmark Pond. As the pond tidally flushed through the breach, TN concentrations dropped from the initial 0.78 mg/L to 0.33 mg/L, as can be seen in Figure VI-4.

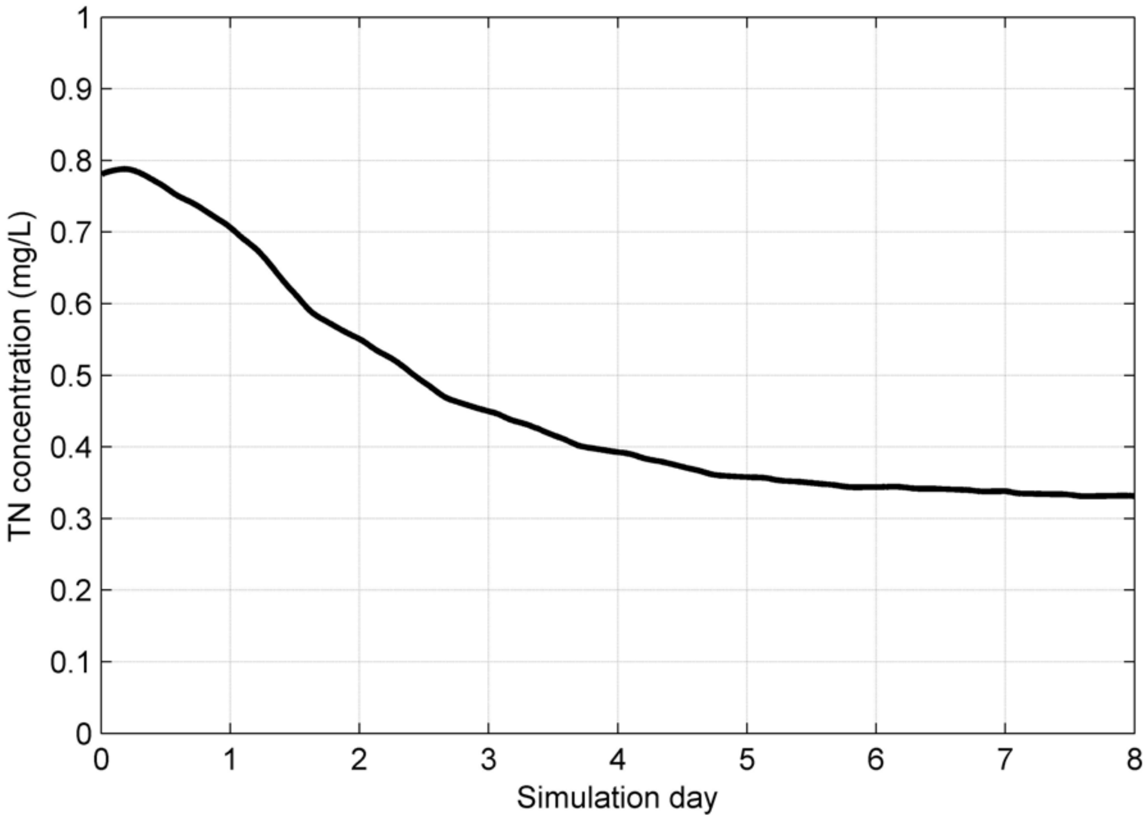


Figure VI-4. RMA-4 model output for Chilmark Pond present (2004) loading conditions (Table VI-1). The average pond opening is 8 days before it closes again. Model results based on minimum recharge rate of 10.36 ft^3/sec. Model results assume a fully open breach for the complete simulation period.

VI.2.5 Build-Out and No Anthropogenic Load Scenarios

To assess the influence of nitrogen loading on total nitrogen concentrations in Chilmark Pond, the standard "build-out" and "no-load" water quality modeling scenarios were run. These runs included two "build-out" scenarios, 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. 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 season variability. The changes in watershed load compared to present conditions are shown in Table VI-4.

Table VI-4. Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic ("no-load") loading scenarios of Chilmark Pond. 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
Chilmark Pond East	5.485	5.526	+0.7%	0.899	-83.6%
Chilmark Pond West	11.614	11.978	+3.1%	3.019	-74.0%

VI.2.5.1 Build Out

A breakdown of the total nitrogen load entering the pond for the model Build-out scenario is shown in Table VI-5. The benthic flux for all scenarios is assumed to vary proportional to the watershed load, where an increase in watershed load will result in an increase in benthic flux (i.e., a positive change in the absolute value of the flux), and *vice versa*.

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-5. Build-out scenario sub-embayment and surface water loads used for total nitrogen modeling of the Chilmark Pond 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)
Chilmark Pond East	5.526	3.260	-0.545
Chilmark Pond West	11.978	0.655	-3.153

For the modeled build-out scenario (given an initial concentration of 0.79 mg/L), modeled TN concentrations drop to 0.33 mg/L at the end of the RMA-4 8-day breach simulation. Using the mass balance model to extend the build-out simulation through the summer, the concentration is computed to be 0.729 mg/L 60 days after the closure of the breach, and 0.776 mg/L 120 days after closure of the breach.

VI.2.5.2 No Anthropogenic Load

A breakdown of the total nitrogen load entering the Pond sub-embayments for the no anthropogenic load ("no load") scenarios is shown in Table VI-6. The benthic flux input to each embayment was reduced (toward zero) based on the reduction in the watershed load (as discussed in Section VI.2.6.1). Compared to the modeled present conditions and build-out scenario, atmospheric deposition directly to each sub-embayment becomes a greater percentage of the total nitrogen load as the watershed load and related benthic flux decrease.

Table VI-6. "No anthropogenic loading" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of the Chilmark Pond 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)
Chilmark Pond East	0.899	3.260	-0.363
Chilmark Pond West	3.019	0.655	-1.697

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations in the Pond. Again, total nitrogen concentrations in the receiving waters (i.e., Atlantic Ocean) remained identical to the existing conditions modeling scenarios.

For the modeled no-anthropogenic scenario (given an initial starting concentration of 0.40 mg/L), modeled TN concentrations decreased to 0.31 mg/L at the end of the RMA-4 8-day breach simulation. Using the mass balance model to extend the no anthropogenic load simulation through the summer, the concentration is computed to be 0.326 mg/L 60 days after the closure of the breach, and 0.305 mg/L 120 days after closure of the breach. It should be noted that the concentration of TN is decreasing during the closure of the breach due to the concentration of the no-anthropogenic scenario being less than the initial concentration of the pond.

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 Chilmark Pond Embayment System (Upper and Lower) in the Town of Chilmark, MA, the MEP assessment is based upon data from the water quality monitoring program developed by the Town and the Martha's Vineyard Commission, with technical assistance from SMAST, as well as field survey and historical data collected under the programmatic umbrella of the Massachusetts Estuaries Project. These data include temporal surveys of eelgrass distribution; surveys of benthic animal communities and sediment characteristics; and time-series measurements of dissolved oxygen and chlorophyll-*a* during the summer and fall of 2005 and summer 2006. These data form the basis of an assessment of the present health of the system, 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 this system (Section VIII). Part of the MEP assessment necessarily includes confirmation that the critical nutrient for management in any embayment is nitrogen and determination that a system is or is not impaired by nitrogen enrichment. Analysis of inorganic N/P molar ratios within the water column of the Chilmark Pond Embayment System support the contention that nitrogen is the nutrient to be managed to control negative effects of nitrogen enrichment. The estuarine reaches within the Chilmark Pond Embayment System follow the general pattern, where the Redfield Ratio (inorganic N/P) averages <16. Redfield ratios >16 generally indicate phosphorus and <16 indicate nitrogen additions will cause eutrophication, respectively. This is also supported by the low levels of total dissolved inorganic nitrogen (2 uM) during summer months. These data indicate that nitrogen additions will increase phytoplankton production, organic matter levels and turbidity within this system. This was also the conclusion of the Martha's Vineyard Commission assessment of 2001 (MVC 2001 updated 2010).

Increased phytoplankton and organic matter levels increase oxygen consumption within the waters and sediments and increase the extent of oxygen depletion and habitat impairment. It should be noted that nitrogen enrichment occurs through two primary mechanisms, high rates of nitrogen entering from the surrounding watershed and/or low rates of flushing due to restriction of tidal exchange with low nitrogen offshore waters. Chilmark Pond has seen increasing nitrogen loading from its watershed from shifting land-uses and due to coastal processes along its barrier beach, which is only periodically opened to tidal exchange. Fundamentally, restrictions of tidal exchange increase the sensitivity of an estuary to nitrogen inputs.

The Chilmark Pond Embayment System is continually being restructured by coastal processes related to inlet dynamics but also fundamental changes in embayment structure due to storm related wash-over events. Wash-over of the barrier beach/dune system during major storms has been dividing the lagoon behind the barrier beach into separate basins. In addition, Long Point, which was a salt marsh, is currently a submerged shoal. But the recent wash-over events have the major potential for causing ecological changes. At present, wash over events beginning in the 1970's have divided the Upper Chilmark Pond basin into 2 separate basins (a western-most basin that is freshwater and a western basin that is brackish) from the main eastern basin of the system that is referred to as Lower Chilmark Pond (Figure VII-1). The western-most upper basin (connected to Upper Chilmark Pond by Interns Creek) has become sufficiently restricted that the small remaining channel to the estuary is sufficiently raised as to prevent entry of water from the estuary when the inlet is periodically open. The result is that the western-most upper basin, once part of the embayment system, is now freshwater (MVC 2001). The second of these basins, between the recently formed freshwater pond and Lower Chilmark Pond (east) currently has a channel (Doctors Creek) that allows outflow to the main basin of Lower Chilmark Pond. This western basin has also become very fresh with only 1 sample from the 2005 showing 2 ppt at the mouth of Doctors Creek. However, a recording of salinity in the mid portion of the eastern basin of Upper Chilmark Pond (see Figure VII-2) showed no detectable salinity over a 2 month deployment. It appears that Upper Chilmark Pond is presently sufficiently isolated to maintain it as a freshwater basin, separate from the estuary. The fresh water status of both basins of Upper Chilmark Pond is also indicated by the benthic community which includes freshwater and riverine species (*Aulodrilus*, Chirinomidae). These organisms typically are found in fresh waters, although they have been observed in sediments overlain by slightly brackish water (1-2 ppt). The MEP assessment and threshold analysis is based upon this basin structure of the Chilmark Pond Embayment System and has not included the fresh to very slightly brackish upper basin as part of the estuary. Similarly, ceasing to open the main basin of the pond to transform it to a freshwater pond was not considered as an option as a freshwater main basin is not sustainable, since storm and wash over events will periodically introduce salt water, causing an unstable impaired resource.

Figure VII-1. Aerial Photograph of the Chilmark Pond system in the Town of Chilmark showing locations of Dissolved Oxygen mooring deployments conducted in the Summer of 2005 and 2006 (Gilberts Cove redeployed due to instrument failure in 2005).

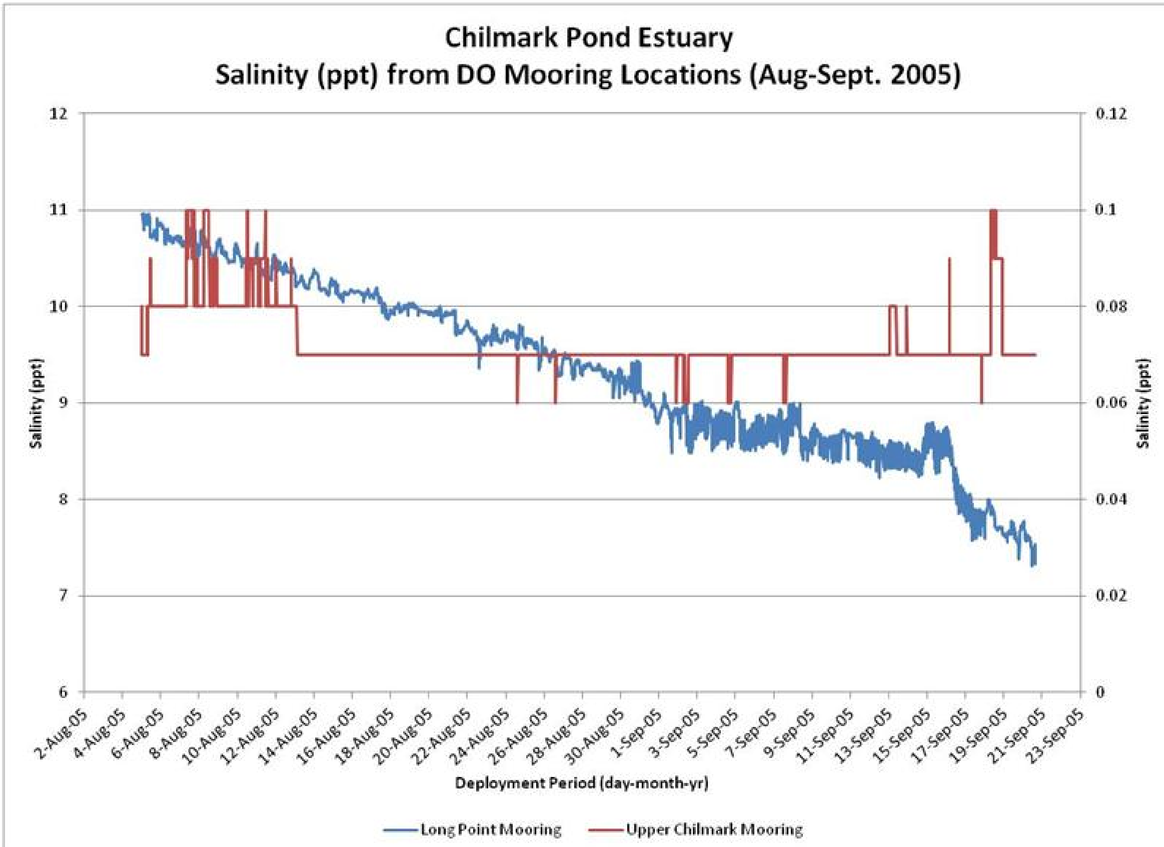


Figure VII-2. Plot of salinity at the Chilmark Pond Upper mooring location as well as the Long Point mooring location. Salinity data in Upper Chilmark Pond indicates a freshwater to periodically brackish aquatic habitat.

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 stable. 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 threshold 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 autonomously recording dissolved oxygen sensors throughout Chilmark Pond at critical points in the system. The sensors were sited such that they would be representative of dissolved oxygen conditions within major sub-basins comprising the Chilmark Pond Estuary, namely the fresh Upper Chilmark Pond, and estuarine Chilmark Pond main Basin (Long Point), Gilberts Cove and Wades Cove. The four dissolved oxygen moorings were deployed 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 Chilmark Pond system was limited as no quantitative information on eelgrass distribution exists from previous studies by the Martha's Vineyard Commission and the MassDEP Eelgrass Mapping Program (C. Costello) due to poor aerial imagery. As a result, the MEP Technical staff did interview various persons knowledgeable about Chilmark Pond and conducted a general survey as part of the mooring program (2005 & 2006) and sediment and infauna surveys in 2005. It should be noted that MEP staff did not observe any eelgrass in the pond as it was completing its data collection tasks. Temporal trends in the distribution of eelgrass beds are typically used by the MEP to assess the stability of the habitat and to determine trends potentially related to nutrient enrichment and water quality. Eelgrass beds can decrease within embayments in response to a variety of causes, but throughout almost all of the embayments within southeastern Massachusetts, the primary cause appears to be related to increases in embayment nitrogen levels. This is consistent with results from the Water Quality Monitoring Program indicating that phytoplankton production (blooms) within the basins of the Chilmark Pond Estuary are prevalent and are enhanced by nitrogen. This is based upon inorganic nitrogen to phosphorus ratios, where system wide the basin summer averages range from 2-7. While this ratio approach (Redfield Ratio) is an approximation, where values <16 indicate nitrogen limitation, >16 phosphorus limitation, the low value of the ratio provides additional site-specific evidence that nitrogen is the appropriate nutrient for management of eutrophication in this system.

While a temporal change in eelgrass distribution typically provides a basis for evaluating increases (nitrogen loading) or decreases (increased flushing- change in breaching schedule) in nutrient enrichment within an embayment system, Chilmark Pond has not historically supported eelgrass. In this case, benthic animal indicators were used to assess the level of habitat health from “healthy” (low organic matter loading, high D.O.) to “highly stressed” (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stressed indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill, benthic population studies in Buzzards Bay (Sanders, H.L. 1960, Sanders, H.L. *et.al.*, 1980, Tian, Y.Q., J.J. Wang, J. A. Duff, B.L. Howes and A. Evgenidou. 2009) and New Bedford (Howes, B.L. and C.T. Taylor, 1990), 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 4 mg L^{-1}, in open water estuarine environments.

Massachusetts State Water Quality Classifications indicate that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1}. The tidally influenced waters of the Chilmark Pond 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-3). 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 Chilmark Pond 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 August through mid-September. The majority of the mooring data from the Chilmark Pond system was collected during the summer of 2005. Only one mooring had to be redeployed in the summer of 2006 (Gilberts Cove) which measured over the same temporal period as in 2005.

Similar to other embayments in southeastern Massachusetts, the Chilmark Pond Embayment System evaluated in this assessment showed high frequency variation in water column oxygen and chlorophyll levels, apparently related to diurnal 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 evaluated both for temporal trends and to determine the percent of the 44 to 49 day deployment period that these parameters were below/above various benchmark concentrations (Tables VII-1, VII-2). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions.

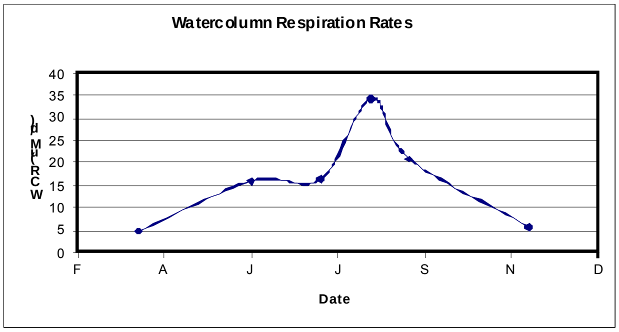


Figure VII-3. Average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll-*a* levels indicate highly nutrient enriched waters throughout the Chilmark Pond system, particularly in the main basin of Upper Chilmark Pond and greater levels of oxygen depletion and excursions and phytoplankton biomass in other locations such as the Long Point mooring location as well as in the coves, such as Wades Cove (Figures VII-4 through VII-11). It should be noted that the Water Quality Monitoring Program observed similar levels of chlorophyll and bottom water oxygen depletion in critical areas of the system, although did not always capture the minimum oxygen or maximum chlorophyll-*a* conditions at site. The oxygen data is consistent with a high level of organic matter enrichment, primarily from phytoplankton production as seen from the parallel measurements of chlorophyll-*a*. The measured levels of oxygen depletion and enhanced chlorophyll-*a* levels are consistent with the nitrogen levels within the various basins (Section VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment of this estuarine system.

The oxygen records show that the inner sub-embayments of Chilmark Pond, specifically Upper Chilmark Pond and Wades Cove, which receive significant watershed nitrogen loads relative to their volumes and turnover rates, have the largest daily oxygen excursions, a nutrient related response. 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). In addition to Upper Chilmark Pond and Wades Cove, the central region of the Long Point basin that receives out-flowing water from Upper Chilmark Pond also shows clear evidence of oxygen levels above atmospheric equilibration providing additional documentation of potential impairment through nitrogen over-enrichment.

Measured dissolved oxygen depletion indicates that sub-basins to the Chilmark Pond Embayment System, such as the Upper Chilmark Pond and Wades Cove, and to a lesser extent, the sub-embayment basin of Gilberts Cove, show high to moderate levels of oxygen stress respectively, as does bottom water oxygen data from the monitoring program (2000-2012). The largest oxygen depletions and excursions were observed in Upper Chilmark Pond, the Long Point portion of the main basin and Wades Cove, which receives large quantities of groundwater and surface water transported nutrient load from the watershed. The observed spatial pattern indicated that the level of oxygen depletion (Table VII-1) and chlorophyll-*a* (Table VII-2) and total nitrogen levels increased with increasing distance from the tidal inlet (when the pond is open to the Atlantic Ocean), periodically created through the barrier beach. This temporary inlet opening serves to lower pond and associated groundwater levels, but also provides the most promising mechanism for restoration of pond habitats presently impaired due to nitrogen enrichment by exchanging nitrogen and organic matter enriched pond waters with high quality waters of the Atlantic Ocean. The Water Quality Monitoring Program, while not yielding insight into the short-term temporal variation in oxygen and chlorophyll, does yield a good baseline for looking at the spatial distribution. The results support the mooring data, also indicating high to moderate levels of nitrogen enrichment depending on the location in the overall pond system and a moderate level of enrichment in the main basin of Chilmark Pond. Measured bottom water oxygen depletion followed this same pattern as did the gradient in chlorophyll.

The pattern of oxygen depletion, elevated chlorophyll-*a* and nitrogen levels are consistent with the present quality of infaunal habitats (Section VII.4) throughout the Chilmark Pond Embayment System. These assessments indicate an estuarine system that is beyond its ability to assimilate nitrogen loads without impairment. The embayment specific results are as follows:

Upper Chilmark Pond – (Figures VII-4 and VII-5):

The Upper Chilmark Pond is a freshwater basin. Data was collected to assist future planning for this freshwater pond. The mooring was centrally located within the main basin of this fresh pond draining surface flow to the main portion of Chilmark Pond (Figure VII-1). Daily excursions (maximum to minimum) in oxygen levels at this location were moderate, generally varying approximately 2 mg L^{-1}. Oxygen levels varied primarily with light (diurnal cycle) as the pond system was closed during the deployment period and therefore could not be affected by tidal exchange. Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). While maximum oxygen levels did not exceed air equilibration (% air saturation), which occurs when nutrient enrichment has stimulated phytoplankton production and oxygen release, an extreme phytoplankton bloom was observed over the last 14 days of the deployment period. Both the moderate oxygen levels (5 to 9 mg L^{-1}), the moderate daily excursion and the extremely high chlorophyll levels suggests that significant organic matter enriched conditions are extant in this region of the basin during the measurement period.

Oxygen levels were generally above 6 mg L^{-1} (90% of record) and only infrequently declined to less than 6 mg L^{-1} for 10% of the 47 day record (Figure VII-4). These values are comparable to the results from the long-term Water Quality Monitoring Program sampling at this location. Oxygen levels at this site in the Upper Chilmark Pond portion of the overall system were always >4 mg L^{-1}, the critical threshold for oxygen stress in an estuarine system (Table VII-1). The infrequent oxygen declines were generally consistent with the moderate to low levels of phytoplankton biomass as measured by chlorophyll-*a* for the first 32 days of the deployment period and then inconsistent with the Chlorophyll record for the last 15 days of the measurement period. Chlorophyll-*a* averaged 62 ug L^{-1} over the record and exceeded 25 ug L^{-1} 40% of the deployment period. The chlorophyll-*a* levels were generally low and constant for the first half of the mooring deployment and then indicated a pronounced bloom during the last 15 days of the 47 day measurement period. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments, a level well surpassed by the average chlorophyll-*a* observed in this basin of Upper Chilmark Pond. These levels of chlorophyll-*a* are indicative of eutrophication in a freshwater pond (Table VII-2, Figure VII-5).

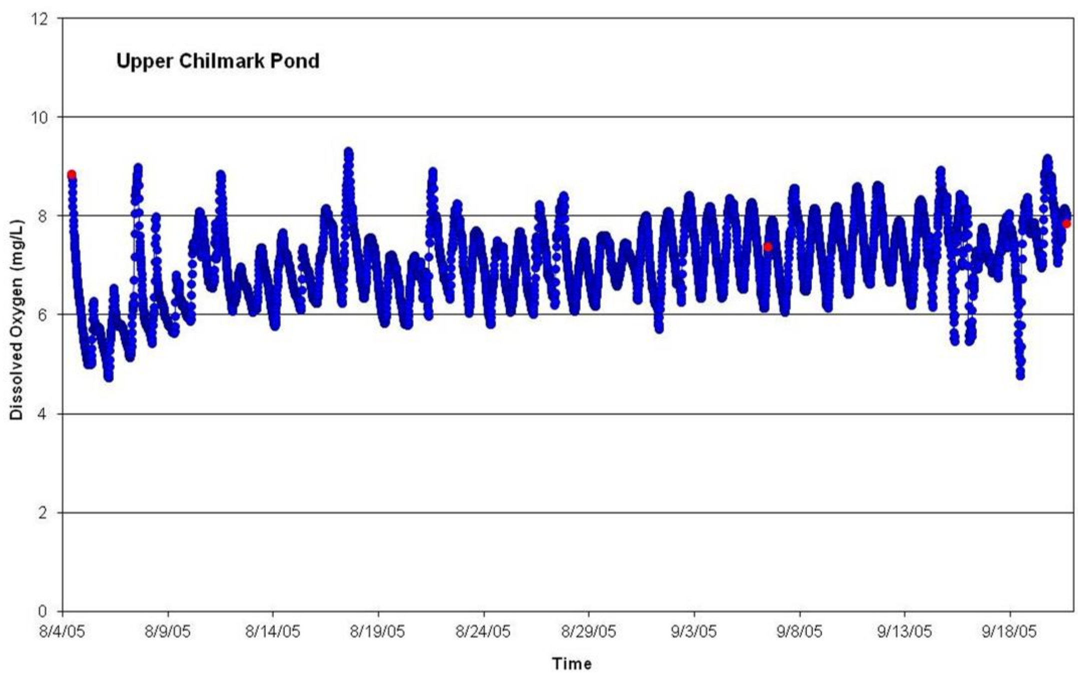


Figure VII-4. Bottom water record of dissolved oxygen at the Upper Chilmark Pond station, Summer 2005 (location in Figure VII-1). Calibration samples represented by red dots.

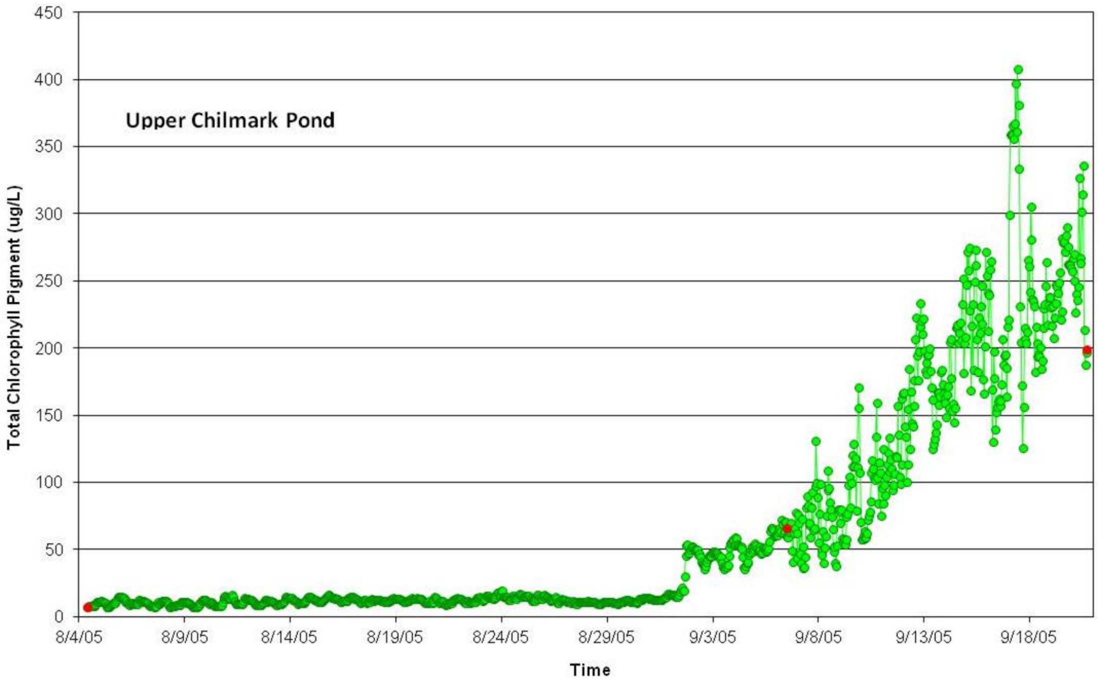


Figure VII-5. Bottom water record of Chlorophyll-*a* in the Upper Chilmark Pond station, Summer 2005 (location in Figure VII-1). Calibration samples represented as red dots.

Long Point (Figures VII-6 and VII-7):

The Long Point mooring was located in the lower portion of the main basin of Lower Chilmark Pond at the southwestern end of the pond (near Long Point) and down gradient of where freshwaters from Upper Chilmark Pond discharge to the estuarine basins of the Chilmark Pond Embayment System (Figure VII-1). Oxygen varied primarily with light (diurnal cycle) as the pond system was closed during the deployment period and therefore could not be affected by the tide. Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). Daily excursions in oxygen levels at this location were generally moderate, <4 mg L^{-1}, indicating organic enrichment and elevated phytoplankton production. Moreover, maximum oxygen levels did exceed air equilibration (% air saturation) reaching as high as 12 mg L^{-1}, which typically occurs at high levels of nitrogen enrichment sufficient to stimulate phytoplankton production (oxygen release). The presence of high oxygen levels (>10 mg L^{-1}) and large daily excursion, but without significant hypoxia is indicative of a system with moderate to high nitrogen and organic matter enrichment.

Oxygen levels occasionally declined below 6 mg L^{-1} and 5 mg L^{-1}, for 12% and 3% of the 47 day record respectively (Figure VII-6). Moreover, oxygen levels periodically dropped below 3 mg L^{-1}, below the oxygen stress threshold of 4 mg/L (Table VII-1). The frequent and significant oxygen declines were consistent with the elevated levels of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 32.8 ug L^{-1} over the record, frequently exceeding 25 ug L^{-1} (53% of record) and showing periodic blooms to >80 ug L^{-1}. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality in temperate embayments. Both the extent of oxygen depletion and the levels of chlorophyll are indicative of a sub-basin with moderate-high nitrogen and organic matter enrichment (Table VII-2, Figure VII-7).

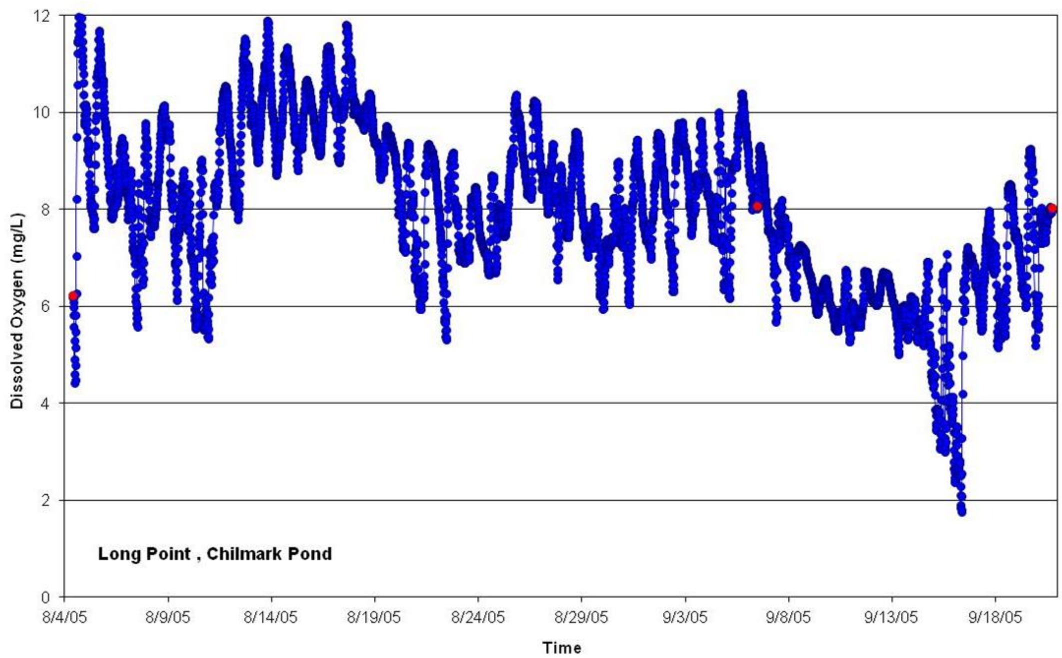


Figure VII-6. Bottom water record of dissolved oxygen recorded within the southern portion of the main basin of Chilmark Pond (Long Point), summer 2005 (location in Figure VII-1). Calibration samples represented as red dots.

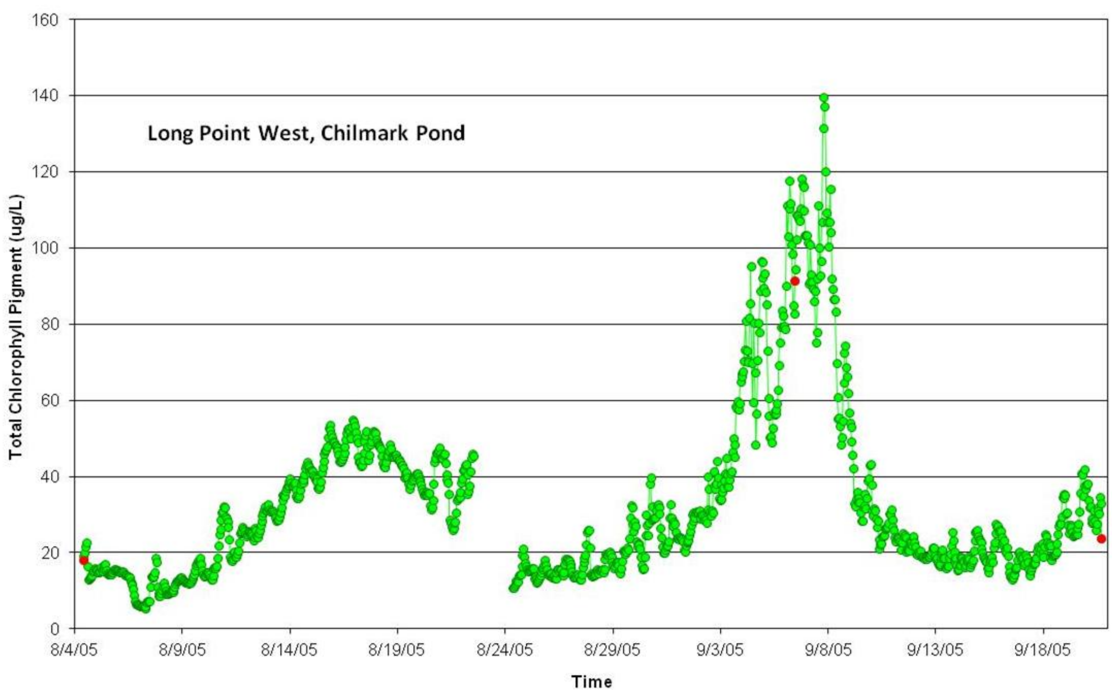


Figure VII-7. Bottom water record of Chlorophyll-*a* recorded within the Long Point portion of the main basin of Chilmark Pond, summer 2005 (location in Figure VII-1). Calibration samples represented as red dots.

Gilberts Cove (Figures VII-8 and VII-9):

The Gilberts Cove mooring site was located within the mid-upper reach of this finger tributary to the main basin that is considered Lower Chilmark Pond (Figure VII-1). Oxygen varied primarily with light (diurnal cycle) as the pond system was closed during the deployment period and therefore could not be affected by the tide. Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). Daily excursions in oxygen levels were generally low, generally ~2 mg L^{-1} from day to night. In addition, maximum oxygen levels did not exceed air equilibration (% air saturation), and were typically between 5 and 8 mg L^{-1} throughout the 45 day record The modest daily excursions and the relatively consistent concentrations is indicative of a system with only moderate nitrogen and organic matter enrichment. The oxygen levels were consistent with the parallel measurements of phytoplankton biomass as assayed by chlorophyll a, which also showed only moderate enhancement by nitrogen.

Oxygen levels frequently declined below 6 mg L^{-1} and 5 mg L^{-1}, for 33% and 8% of the 45 day record respectively (Figure VII-8). Additionally, oxygen was virtually always >4 mg L^{-1} (Table VII-1). The moderate levels of oxygen depletion are indicative of a low-moderate level of organic matter enrichment, consistent with the parallel measures of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 8.3 ug L^{-1} over the 45 day record, was consistently <10 ug L^{-1}, 82% of the record, with only a brief bloom to ~15 ug L^{-1}. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality, a level well below the average chlorophyll-*a* observed in this basin. Both the extent of oxygen depletion and the levels of chlorophyll are indicative of an estuarine reach with only moderate nitrogen and organic matter enrichment at levels associated with moderate habitat impairment in many embayments of southeastern (Table VII-2, Figure VII-9).

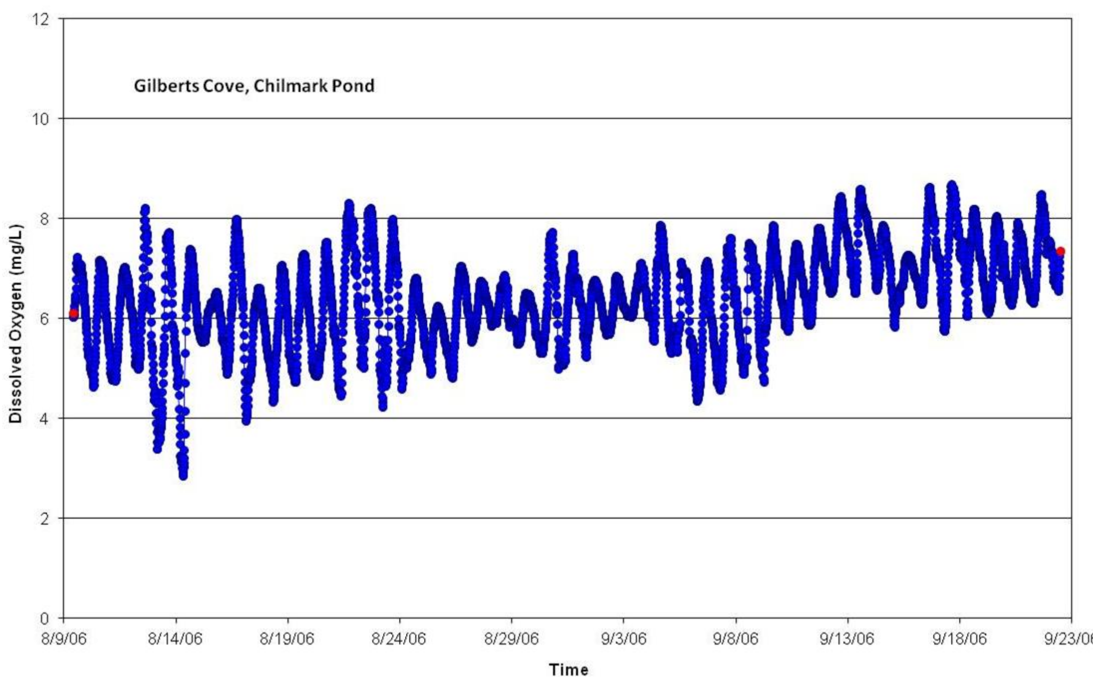


Figure VII-8. Bottom water record of dissolved oxygen within the Gilberts Cove portion of Chilmark Pond, summer 2005 (Figure VII-1). Calibration samples shown as red dots.

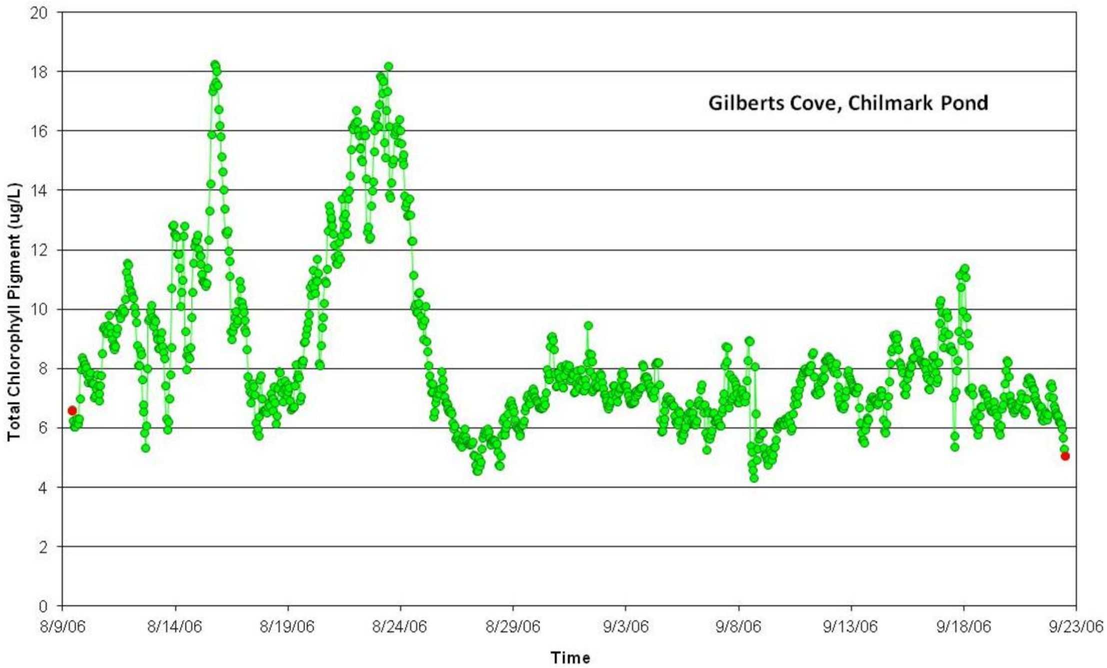


Figure VII-9. Bottom water record of Chlorophyll-*a* within the Gilberts Cove portion of Chilmark Pond, summer 2006 (location in Figure VII-1). Calibration samples shown as red dots.

Wades Cove (Figures VII-10 and VII-11)

The Wades Cove mooring site was located within the middle reach of this finger tributary to the main basin that is considered Lower Chilmark Pond (Figure VII-1). Oxygen varied primarily with light (diurnal cycle) as the pond system was closed during the deployment period and therefore could not be affected by the tide. Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed towards the end of the photocycle (ca. 1500 hrs). Daily excursions in oxygen levels were moderate, generally ~2 to 3 mg L^{-1} from day to night. In addition, maximum oxygen levels did exceed air equilibration (% air saturation), and occasionally exceeded 8 and 10 mg L^{-1}. The modest to high daily excursions and the observed oxygen levels over air equilibration is indicative of a system with moderate nitrogen and organic matter enrichment.

Oxygen levels frequently declined below 6 mg L^{-1} and 5 mg L^{-1}, for 50% and 21% of the 49 day record respectively (Figure VII-10). Additionally, moderate depletions of oxygen were observed, with oxygen levels declined to <4 mg L^{-1}, the oxygen stress threshold of 4 mg L^{-1} (Table VII-1). The moderate levels of oxygen depletion are indicative of a moderate to high level of organic matter enrichment, consistent with the parallel measures of phytoplankton biomass as measured by chlorophyll-*a*. Chlorophyll-*a* averaged 31.4 ug L^{-1} over the 49 day record, was consistently >15 ug L^{-1} and frequently >25 ug L^{-1}, 87% and 48% of time and showed periodic blooms of >80 ug L^{-1}. Average summer chlorophyll levels over 10 ug L^{-1} have been used to indicate impaired nitrogen related water quality, a level well below the average chlorophyll-*a* observed in this basin. Both the extent of oxygen depletion and the levels of chlorophyll are indicative of an estuarine reach with significant nitrogen and organic matter enrichment at levels associated with habitat impairment in many embayments of southeastern (Table VII-2, Figure VII-11).

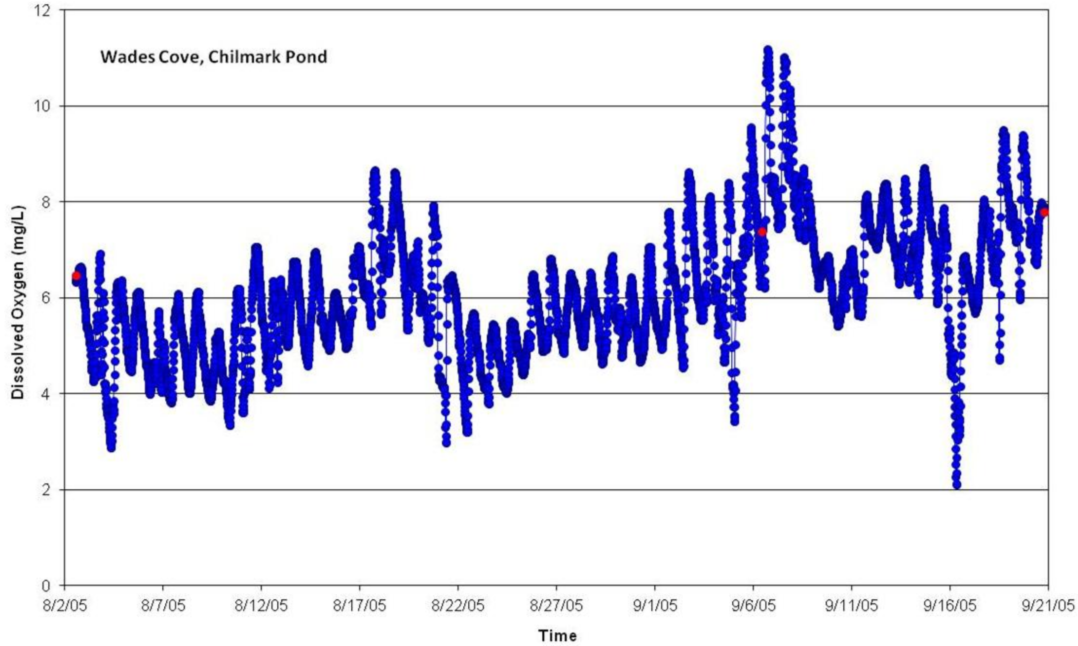


Figure VII-10. Bottom water record of dissolved oxygen within Wades Cove, Chilmark Pond, summer 2005 (Figure VII-1). Calibration samples shown as red dots.

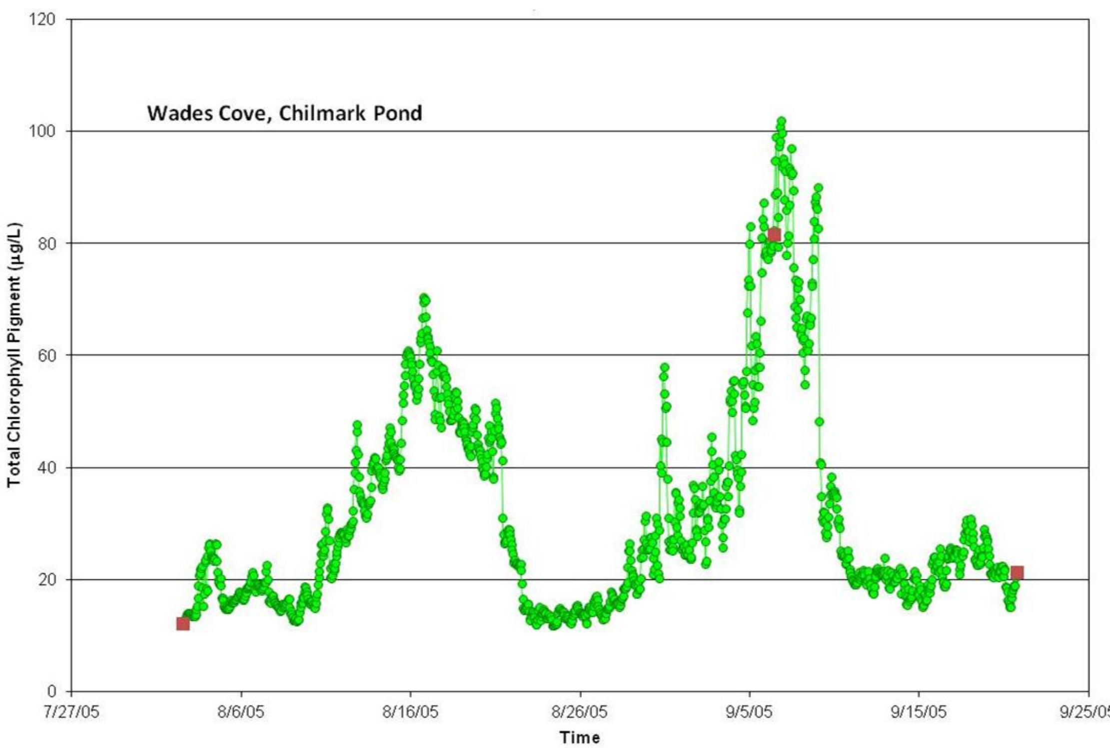


Figure VII-11. Bottom water record of Chlorophyll-*a* within the Wades Cove, Chilmark Pond Estuary, summer 2005 (location in Figure VII-1). Calibration samples shown as red dots.

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

Mooring 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)
Upper Chilmark Pond	**8/4/2005**	**9/20/2005**	**47.3**	**10%**	**0%**	**0%**	**0%**
			Mean	0.29	0.09	N/A	N/A
			Min	0.01	0.04	0.00	0.00
			Max	0.89	0.14	0.00	0.00
			S.D.	0.27	0.07	N/A	N/A
Long Point West, Chilmark	**8/4/2005**	**9/20/2005**	**47.3**	**12%**	**3%**	**2%**	**1%**
			Mean	0.19	0.23	0.23	0.16
			Min	0.01	0.04	0.08	0.09
			Max	0.75	0.64	0.44	0.23
			S.D.	0.21	0.23	0.16	0.10
Wades Cove, Chilmark	**8/2/2005**	**9/20/2005**	**49.23**	**50%**	**21%**	**4%**	**0%**
			Mean	0.52	0.26	0.19	0.05
			Min	0.02	0.01	0.07	0.02
			Max	3.82	0.71	0.42	0.07
			S.D.	0.64	0.23	0.11	0.03
Gilberts Cove, Chilmark	**8/9/2006**	**9/22/2006**	**44.05**	**33%**	**8%**	**1%**	**0%**
			Mean	0.42	0.17	0.11	0.05
			Min	0.05	0.01	0.03	0.05
			Max	0.65	0.46	0.23	0.05
			S.D.	0.17	0.12	0.09	N/A

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

Mooring 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)
Upper Chilmark Pond	**8/4/2005**	**9/20/2005**	**47.42**	**100%**	**89%**	**46%**	**40%**	**40%**
Mean Chl Value = 62.1 ug/L			Mean	47.38	0.44	2.12	0.40	1.21
			Min	47.38	0.04	0.04	0.04	0.04
			Max	47.38	0.83	22.21	0.75	19.38
			S.D.	N/A	0.25	5.04	0.22	4.53
Long Point West, Chilmark	**8/4/2005**	**9/20/2005**	**45.67**	**100%**	**97%**	**84%**	**65%**	**53%**
Mean Chl Value = 32.8 ug/L			Mean	45.63	0.10	11.09	0.44	1.93
			Min	45.63	0.04	0.21	0.04	0.04
			Max	45.63	0.17	41.42	0.83	17.08
			S.D.	N/A	0.05	20.24	0.25	4.46
Wades Cove, Chilmark	**8/2/2005**	**9/20/2005**	**49.33**	**100%**	**100%**	**87%**	**66%**	**48%**
Mean Chl Value = 31.4 ug/L			Mean	49.29	49.29	3.31	1.47	1.12
			Min	49.29	49.29	0.04	0.04	0.04
			Max	49.29	49.29	24.08	12.50	10.33
			S.D.	N/A	N/A	7.07	3.56	2.71
Gilberts Cove, Chilmark	**8/9/2006**	**9/22/2006**	**44.17**	**98%**	**18%**	**5%**	**0%**	**0%**
Mean Chl Value = 8.3 ug/L			Mean	5.43	0.29	0.62	0.10	0.58
			Min	0.04	0.04	0.04	0.04	0.46
			Max	17.92	0.54	4.08	0.17	0.71
			S.D.	7.36	0.16	1.14	0.05	0.12

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Analysis of eelgrass in Chilmark Pond based on historical data was attempted by the MassDEP Eelgrass Mapping Program as part of the MEP. However, developing quantitative results was not possible due to the absence of eelgrass over the long term in this estuary, limited access to the pond as it is closed most of the year and the poor quality historical aerial imagery for photo-interpretation. Analysis of available aerial photos from 1951 are typically used by the MEP to reconstruct the eelgrass distribution prior to any substantial development of the watershed. As historical imagery was of poor quality for use in photo-interpretation of potential eelgrass presence, the 1951 data could only be used qualitatively. As a result, local officials and citizens with long-term firsthand knowledge of the Pond were sought out relative to eelgrass presence/absence in the Pond. In addition, qualitative field observations of eelgrass absence/presence have been made by a variety of scientists ranging from MEP Technical Team in 2005 and 2007 and MVC staff (W. Wilcox) as well as the Town of Chilmark Shellfish Propagation Agent (I. Scheffer). While these latter observations do not lend themselves to mapping of eelgrass coverage, they provide critical information on the absence/presence of eelgrass within this great salt pond and its general locations, depths and density, where present. These data form the basis of the MEP eelgrass assessment for this estuary.

The primary use of the MEP eelgrass assessment for an estuary is to indicate (a) if eelgrass once or currently colonizes a basin and (b) if large-scale system-wide shifts have occurred. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 2012; the period in which watershed nitrogen loading significantly increased to its present level. This temporal information can be used to determine the stability of the eelgrass community and the potential recoverable acreages should it be determined that habitat loss has occurred.

Over the past several decades, eelgrass has not existed within the Chilmark Pond Embayment System. As determined by the Martha's Vineyard Commission in 2001 and presented in its nutrient loading report for Chilmark Pond, Menemsha Pond and Squibnocket Pond, "Eelgrass is not presently found in the Lower pond (Chilmark) and may well not have been present for at least the past 60 years as recalled by pond users (Wakeman, 2000). Other rooted macrophytes are infrequent in the Lower pond resulting in less cover and nursery grounds for fish that might either be found in the pond or potentially be stocked." Note that the MEP observed Ruppia a rooted SAV in its 2005 surveys, but not eelgrass, consistent with the MVC statements. The long time absence of eelgrass in Chilmark Pond was also corroborated by the Chilmark Pond Association (R. Samimy, personal communication) as well as by the Chilmark/West Tisbury Shellfish Propagation Agent (I. Scheffer), who indicated that there has not historically been any eelgrass in Chilmark Pond. Mr. Scheffer also talked to a few old timers who were commercial fisherman on that pond for many years and they also confirmed the historical absence of eelgrass anywhere in Chilmark Pond.

At present, given moderate levels of watershed nitrogen loading and limited tidal exchange only periodically occurring during managed breaches of the barrier beach and the nitrogen, chlorophyll and oxygen levels within the pond basins (2000-2012), it can be concluded that Chilmark Pond does not presently support eelgrass habitat. Further, based upon the past decade and analysis of historic information, the MEP Technical Team concluded that Chilmark Pond Embayment System has not supported eelgrass habitat for at least 50 years. Given that the pond's water quality is controlled in significant part by the amount of induced tidal flushing, it is likely that the Pond has had negligible eelgrass habitat for the past century. As eelgrass habitat could not be documented to exist, either historically or presently, within the Chilmark Pond Embayment System, the threshold analysis for this system is necessarily focused on restoration/protection of infaunal animal habitat.

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling for benthic community characterization was conducted at 9 locations throughout the Chilmark Pond Embayment System (Figure VII-12). Sampling sites were located in the freshwater basin of Upper Chilmark Pond (3), Chilmark Pond (3), Wades Cove (2) and Gilberts Cove (1). At each site multiple assays were conducted. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that, given the complete absence of eelgrass beds, the Chilmark Pond Embayment System nitrogen enrichment is being evaluated relative to the characteristics of the benthic animal community and the other water quality and ecological metrics (see Table VIII-1). The benthic infauna analysis is important for determining the level of impairment (healthy→moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Section VIII).

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

Overall, the infauna survey indicated that most sub-basins comprising the Chilmark Pond Embayment System are presently beyond their ability to tolerate additional nitrogen inputs without impairment. Consistent with the observed periodic oxygen depletions and large phytoplankton blooms occurring in the main depositional basins, with little drift macroalgal accumulation, the benthic animal communities are showing moderate to significant impairment. The impairment is consistent with organic enrichment resulting from nitrogen enrichment, from a combination of watershed inputs and only periodic tidal flushing. The Benthic Survey did not reveal any areas of severe degradation (less than 70 animal per grab), as indicated by low numbers of individuals and species or dominance by opportunistic stress indicator species such as Capitellids and Tubificids. In fact, at all locations throughout the estuarine sub-basins of this embayment system, there were high numbers of individuals (400-700 per grab sample), low numbers of opportunistic stress indicator species (Capitellids and Tubificids, generally <10% of community), but the community was composed of few species (7-11) with low diversity (H' = 1.5-2.2) see Table VII-3. Species numbers of 20-25 and diversity >3.0 generally indicate high quality benthic habitats. While there is little evidence of high levels of nitrogen related impairment of the benthic animal communities, most areas did show clear evidence of moderate to significant impairment associated with nitrogen and organic matter enrichment.

Figure VII-12. Aerial photograph of the Chilmark Pond Embayment System showing location of benthic infaunal sampling stations (yellow symbol). CHP1,2,3 are located in Upper Chilmark Pond.

Table VII-3. Benthic infaunal community data for the Chilmark Pond Embayment System. The western Chilmark Pond basins (Upper Pond) has been separated from the estuary by overwash and is now freshwater. Estimates of the number of species adjusted to the number of individuals and diversity (H') and evenness (E) of the community allow comparison between locations. Samples represent surface area of 0.0625 m^2. Stations refer to map in Figure VII-12, replicate samples were collected at each location.

	Sta. I.D.*	Total Species	Total Individuals	#Species Calc @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Chilmark Pond Embayment System						
Freshwater						
Upper Chilmark-West	1,2,3	8	242	6	1.63	0.56
Estuarine						
Lower Chilmark-East	4,5,15	9	378	7	1.82	0.58
Wades Cove	9,12	11	487	8	2.21	0.66
Gilberts Cove	16	7	684	6	1.45	0.54

* Station i.d.'s refer to Figure VII-11.

Each of the estuarine basins, specifically Lower Chilmark Pond (east), Wades Cove and Gilberts Cove are showing moderate-significant levels of impairment related to their elevated chlorophyll-*a* levels and moderate periodic oxygen depletions. While the numbers of individuals remain high throughout the system, the community numbers of species and their diversity and evenness are low and indicative of a community under ecological stress. In all cases, these basins support communities with low diversity, with the measured index only 1.45 to 2.21. Very similar to impaired benthic communities in the tributary coves to nearby Tisbury Great Pond, 1.44 to 1.82. Evenness (how individuals are distributed among the species) was similarly low, in Chilmark Pond, 0.54-0.66 and indicated that only a few species were accounting for most of the individuals within each basin. There was little substantive difference between the basins as all are clearly moderately impaired relative to benthic animal habitat. For comparison the moderately impaired lower basins of Tisbury Great Pond had similar diversity indices of 2.0 to 2.3 and evenness of 0.49 to 0.54. However, in Chilmark Pond's estuarine basins the dominant species are not opportunistic stress indicators (generally <10%), but are tolerant of moderate levels of organic enrichment (*Streblospio* and *Leptocheirus, an amphipod)*. Streblospio was a dominant species within the coves tributary to Tisbury Great Pond as well. Given the prevalence of species tolerant of moderate organic enrichment, the low numbers of stress indicator organisms, the low numbers of species and the low diversity of Chilmark Pond's benthic communities compared to high quality habitat areas in similarly structured embayments in southeastern Massachusetts, it is clear that the main basin of Chilmark Pond and the major coves (Wades, Gilberts) are currently above their nitrogen threshold and is supporting impaired benthic animal habitat.

The results of the infauna survey and complete absence of eelgrass coverage within the Chilmark Pond Embayment System indicates that the nitrogen management threshold analysis (Section VIII) needs to aim for lowering nitrogen enrichment for restoration of infaunal habitat in the tributary coves showing moderate-high impairment of benthic habitat. Reduction in nitrogen enrichment is required for restoration. It should be emphasized that reducing nitrogen enrichment can be achieved by reducing nitrogen inputs and/or increasing the rate of nitrogen loss through enhanced tidal exchange. Restoring these benthic habitats should be the focus of the nitrogen management threshold analysis (Section VIII).

In addition to benthic infaunal community characterization undertaken as part of the MEP field data collection, other biological resources assessments were integrated into the habitat assessment portion of the MEP nutrient threshold development process as developed by the Commonwealth and available to the MEP Technical Team. The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish areas closed to harvest as well as the suitability of a system for the propagation of shellfish (Figure VII-13). As is the case with some systems on Cape Cod, the enclosed waters of the Chilmark Pond system are closed for the taking of shellfish year round. This general closure of Chilmark Pond is potentially due to bacterial contamination most likely from wildlife and surface water inflows as there are significant wetland surfaces surrounding Chilmark Pond and associated natural fauna living on or around the tributary basins to the overall Pond system. The major shellfish species with potential habitat within the Chilmark Pond Estuary are mainly soft shelled clams (*Mya arenaria*) and the American Oyster (Figure VII-14).

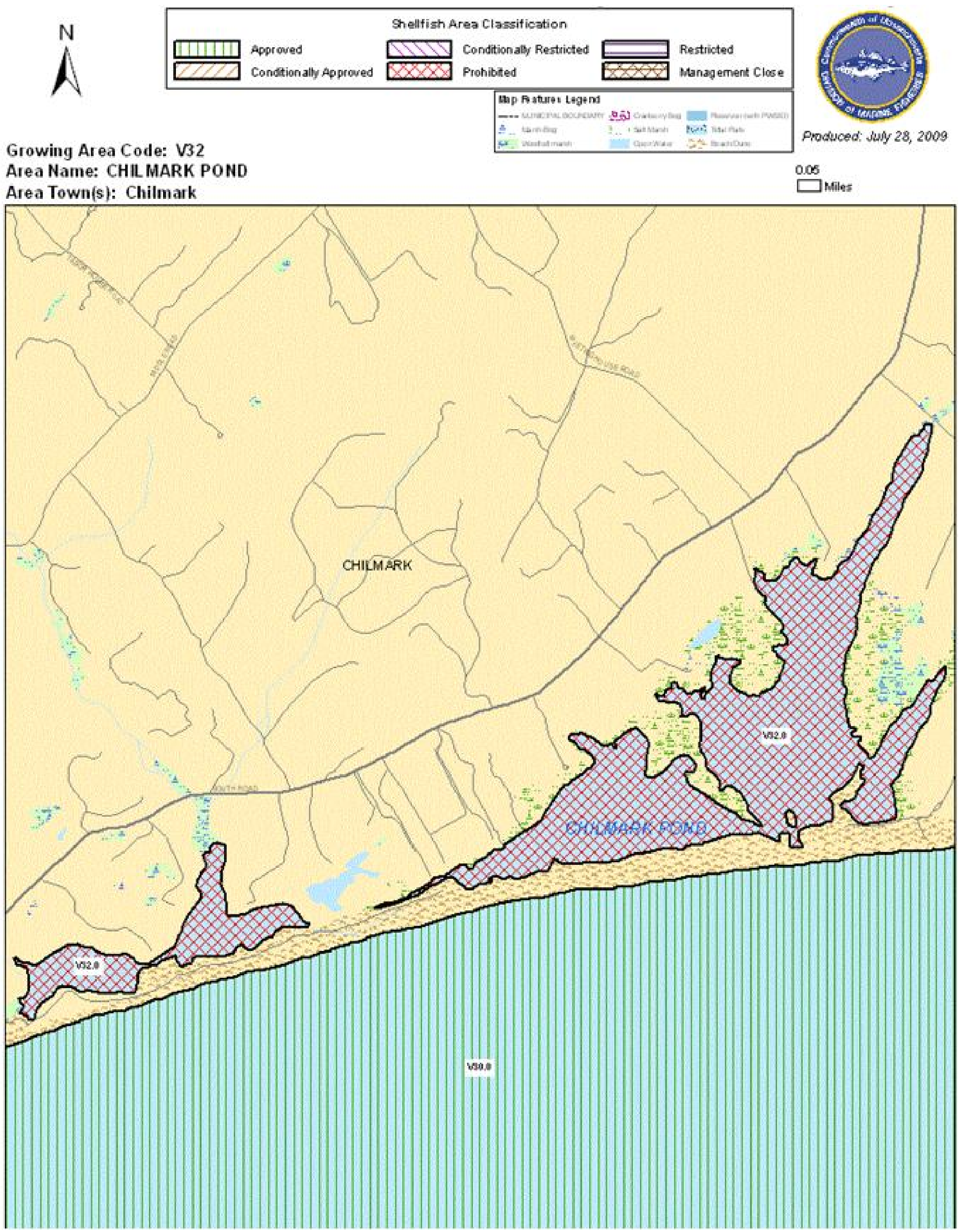


Figure VII-13. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas. However, areas dominated by wetlands with persistent fecal coliform levels >14 cfu per 100 mL may be prohibited to shellfishing until the cause of the contamination (frequently wildlife and birds) is documented.

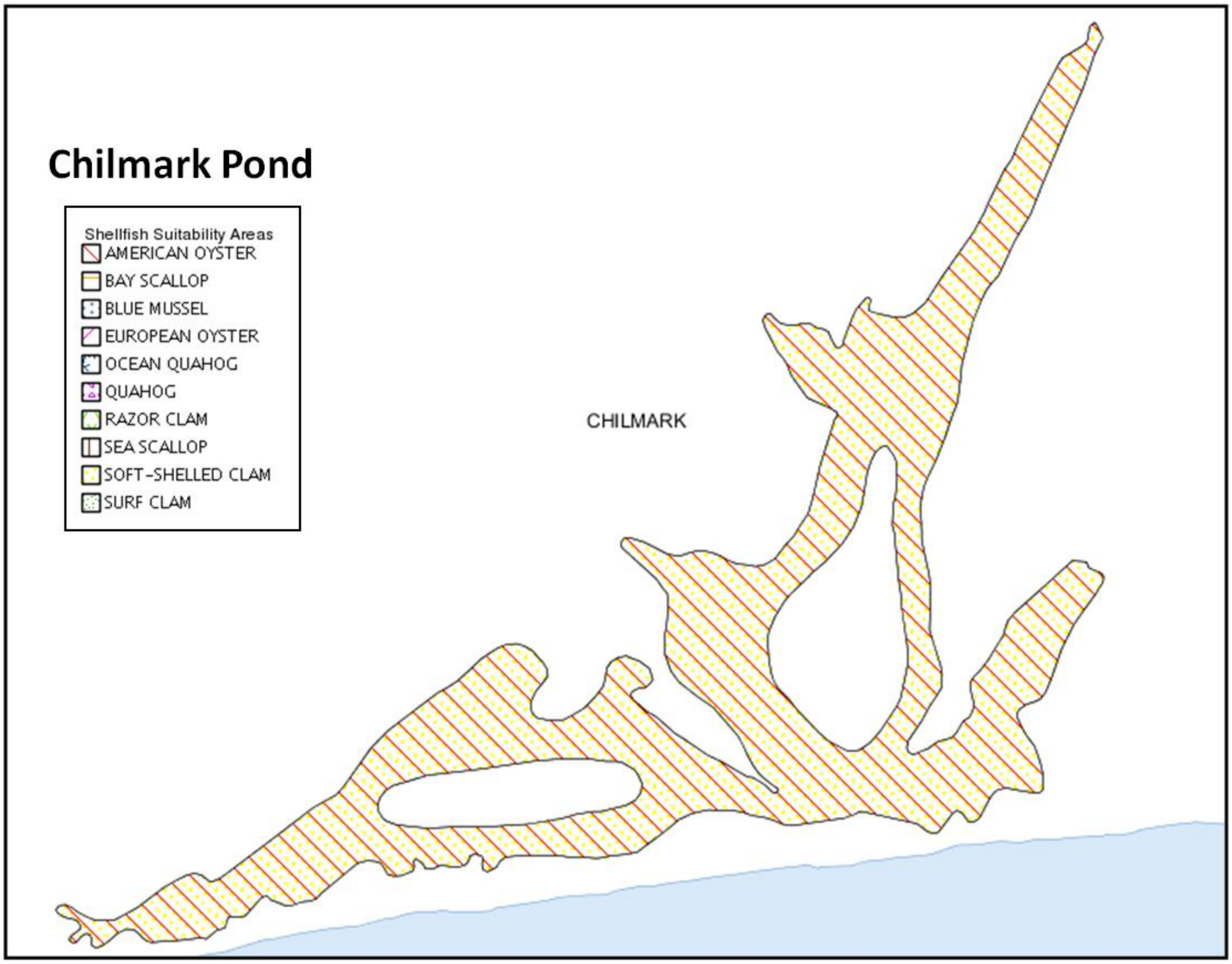


Figure VII-14. Location of shellfish suitability areas within the Chilmark Pond Embayment System as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean that a shellfish population is "present" or that harvest is allowed.

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

VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

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

The Chilmark Pond Embayment System is a complex coastal open water embayment comprised of a large central basin (Lower Chilmark Pond {east}) and multiple sub-embayments (Wades Cove, Gilberts Cove). The western basin, Upper Chilmark Pond, is currently fresh to slightly brackish and has been functionally separated from the estuary by coastal processes. The main basin and its tributary coves are maintained as an estuary by the periodic breaching of the barrier beach with a single temporary inlet. The estuary only occasionally receives tidal waters from the Atlantic Ocean into its main basin based on a schedule of openings set by the Town. Floodwater from the Atlantic Ocean enters the main basin of Lower Chilmark Pond (east) and circulates through channels and across flats making its way up into Wades Cove (the primary tributary basin in this system) as well as into upper Chilmark Pond (west), which is connected to Lower Chilmark Pond via Doctor's Creek, a narrow channel (Figure I-2). Upper Chilmark Pond is really comprised of two basins which are connected by a very small shallow channel locally referred to as Interns Creek. The pond openings follow periods where pond level rises due to groundwater and surface water inflows and precipitation, which creates the hydraulic head needed for the opening process. At present, the number and duration of pond openings plays a fundamental role in the maintenance of nutrient related water quality and habitat health throughout this estuary.

The Chilmark Pond estuary is particularly vulnerable to the effects of nutrient enrichment from the watershed, due to its very limited tidal exchange and that circulation is mainly through wind driven mixing in the small tributary sub-embayments. In particular, the Chilmark Pond Estuary is eutrophying from nitrogen enriched groundwater and surface water flows and runoff from its watershed.

The Chilmark Pond Estuary was formed by rising sea levels drowning a "cove" and formation of a lagoon by coastal processes several thousand years ago. Each type of functional component to an estuary (salt marsh basin, embayment, tidal river, deep basin (sometimes drown kettles), shallow basin, etc.) has a different natural sensitivity to nitrogen enrichment and organic matter loading. Evaluation of eelgrass and infaunal habitat quality must consider the natural structure of the specific basin and its ability to support eelgrass beds and infaunal communities. At present, the Chilmark Pond Estuary is beyond its ability to assimilate nitrogen without further impairment. The system is showing a moderate level of nitrogen enrichment, no eelgrass habitat and moderately/significantly impaired benthic animal habitats, regions of periodic moderate oxygen depletion and phytoplankton blooms. All lines of evidence support an assessment of habitat impairment. Since there is no record of eelgrass in this estuary in recent decades, the impairment of concern is that of benthic animal habitat (Table VIII-1). These findings indicate that nitrogen management of this system will be for restoration rather than for protection or maintenance of an unimpaired system. It should be noted that nitrogen management includes both source reduction and in the case of a tidally restricted embayment, enhanced tidal flushing.

The measured levels of oxygen depletion and enhanced chlorophyll-*a* levels follows the spatial pattern of total nitrogen levels in this system (Chapter VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment and only periodic tidal flows. The spatial pattern indicated that the magnitude of oxygen depletion, enhancement of chlorophyll-*a* levels and total nitrogen concentrations were consistent with the absence of eelgrass and the moderate impairment of benthic animal communities. Given the general lack of tidal action and the dominance of dispersion and wind driven mixing, there is only slight biogeochemical gradients (generally uniform concentrations), except for differences in bottom water oxygen depletion, which are fairly constant except as locally influenced by basin depth. Chilmark Pond is operating as a relatively homogeneous closed pond.

Eelgrass: Analysis of eelgrass in Chilmark Pond based on historical data was attempted by the MassDEP Eelgrass Mapping Program as part of the MEP. However, developing quantitative results was not possible due to the absence of eelgrass over the long term in this estuary, limited access to the pond as it is closed most of the year and the poor quality historical aerial imagery for photo-interpretation. As such, the 1951 data could only be used qualitatively. Local officials and citizens with long-term firsthand knowledge of the Pond were sought out relative to eelgrass presence/absence in the Pond. In addition, qualitative field observations of eelgrass absence/presence have been made by a variety of scientists ranging from MEP Technical Team in 2005 and 2007 and MVC staff (W. Wilcox) as well as the Town of Chilmark Shellfish Propagation Agent (I. Scheffer). While these latter observations do not lend themselves to mapping of eelgrass coverage, they provide critical information on the absence/presence of eelgrass within this great salt pond and its general locations, depths and density, where present. These data form the basis of the MEP eelgrass assessment for this estuary.

Over the past several decades, eelgrass has not existed within the Chilmark Pond Embayment System. As determined by the Martha's Vineyard Commission in 2001 and presented in its nutrient loading report for Chilmark Pond, Menemsha Pond and Squibnocket Pond, "Eelgrass is not presently found in the Lower pond (Chilmark) and may well not have been present for at least the past 60 years as recalled by pond users" (Wakeman, 2000). Other rooted macrophytes are infrequent in the Lower Pond resulting in less cover and nursery grounds for fish that might either be found in the pond or potentially be stocked." Note that the MEP observed Ruppia a rooted SAV in its 2005 surveys, but not eelgrass, consistent with the MVC statements. The long time absence of eelgrass in Chilmark Pond was also corroborated by the Chilmark Pond Association (R. Samimy, personal communication) as well as by the Chilmark/West Tisbury Shellfish Propagation Agent (I. Scheffer), who indicated that there has not historically been any eelgrass in Chilmark Pond. Mr. Scheffer also talked to a few old timers who were commercial fisherman on that pond for many years and they also confirmed the historical absence of eelgrass anywhere in Chilmark Pond.

At present, given moderate levels of watershed nitrogen loading and limited periodic tidal exchange and the nitrogen, chlorophyll and oxygen levels within the pond basins (2000-2012), it can be concluded that Chilmark Pond is not presently supportive of eelgrass beds. Further, based upon the past decade and analysis of historic information, the MEP Technical Team concluded that Chilmark Pond Estuary has not supported eelgrass habitat for at least 50 years. Given that the pond's water quality is controlled in significant part by the amount of induced tidal flushing, it is likely that the Pond has had negligible eelgrass habitat for the past century. As eelgrass habitat could not be documented to exist, either historically or presently, within the Chilmark Pond Estuary, the threshold analysis for this system is necessarily focused on restoration/protection of infaunal animal habitat.

Water Quality: Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 4 mg L^{-1}, in open water estuarine environments. Massachusetts State Water Quality Classifications indicate that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1}. The estuarine basins of Chilmark Pond 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.

Similar to other embayments in southeastern Massachusetts, the Chilmark Pond Estuary has high frequency variation in water column oxygen and chlorophyll levels, apparently related to diurnal 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.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll-*a* levels indicate highly nutrient enriched waters throughout the Chilmark Pond Estuary, particularly the oxygen depletion and D.O. excursions and phytoplankton biomass in each basin, especially Wades Cove (Figures VII-4 through VII-11). It should be noted that the Water Quality Monitoring Program observed similar levels of chlorophyll and bottom water oxygen depletion in critical areas of the system, although it did not always capture the minimum oxygen or maximum chlorophyll-*a* conditions at each site. The oxygen data is consistent with a high level of organic matter enrichment, primarily from phytoplankton production as seen from the parallel measurements of chlorophyll-*a*. The measured levels of oxygen depletion and enhanced chlorophyll-*a* levels are consistent with the nitrogen levels within the various basins (Section VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment of this estuarine system.

The oxygen records show that the sub-embayments of Chilmark Pond, specifically Wades Cove, receive significant watershed nitrogen loads relative to their volumes and turnover rates and consequently have the largest daily oxygen excursions, a nutrient related response. Further, Wades Cove, and to a lesser extent, Gilberts Cove, show high to moderate levels of oxygen depletion and stress to animals. The largest oxygen depletions were observed in the Long Point portion of the main basin and Wades Cove, which receives large quantities of groundwater and surface water transported nutrient load from the watershed. When the Pond is opened by breaching the barrier beach to allow tidal flows, the observed spatial pattern is that the level of oxygen depletion (Table VII-1) and chlorophyll-*a* (Table VII-2) and total nitrogen levels increase with increasing distance from the tidal inlet. This temporary inlet opening serves to lower pond and associated groundwater levels, but also provides the most promising mechanism for restoration of pond habitats, presently impaired due to nitrogen enrichment, by exchanging nitrogen and organic matter enriched pond waters with high quality waters of the Atlantic Ocean.

The pattern of oxygen depletion, elevated chlorophyll-*a* and nitrogen levels are consistent with the present quality of infaunal habitats (Section VII.4) throughout the Chilmark Pond Embayment System. These assessments indicate an estuarine system that is beyond its ability to assimilate nitrogen loads without impairment (Table VIII-1).

Water Quality: While a temporal change in eelgrass distribution typically provides a basis for evaluating increases (nitrogen loading) or decreases (increased flushing - change in breaching schedule) in nutrient enrichment within an embayment system, Chilmark Pond has not historically supported eelgrass. In this case, benthic animal indicators were used to assess the level of habitat health from “healthy” (low organic matter loading, high D.O.) to “highly stressed” (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stressed indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill, benthic population studies in Buzzards Bay (Sanders, H.L. 1960, Sanders, H.L. *et.al.*, 1980, Tian, Y.Q., J.J. Wang, J. A. Duff, B.L. Howes and A. Evgenidou. 2009) and New Bedford (Howes, B.L. and C.T. Taylor, 1990), 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.

It should be noted that, given the complete absence of eelgrass beds, the Chilmark Pond Embayment System nitrogen enrichment is being evaluated relative to the characteristics of the benthic animal community and the other water quality and ecological metrics (see Table VIII-1). The benthic infauna analysis is important for determining the level of impairment (healthy→moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Section VIII-2).

Overall, the infauna survey indicated that most sub-basins comprising the Chilmark Pond Embayment System are presently beyond their ability to tolerate additional nitrogen inputs without further impairment. Consistent with the observed periodic oxygen depletions and large phytoplankton blooms occurring in the main depositional basins, with little drift macroalgal accumulation, the benthic animal communities are showing moderate to significant impairment. The impairment is consistent with organic enrichment resulting from increased nitrogen loading from a combination of watershed inputs and only periodic tidal flushing. The Benthic survey did not reveal any areas of severe degradation (less than 70 animal per grab), as indicated by low numbers of individuals and species or dominance by opportunistic stress indicator species such as Capitellids and Tubificids. In fact, at all locations throughout the estuarine sub-basins of this embayment system, there were high numbers of individuals (400-700 per grab sample), low numbers of opportunistic stress indicator species (Capitellids and Tubificids, generally <10% of community), but the community was composed of few species (7-11) with low diversity (H' = 1.5-2.2). Species numbers of 20-25 and diversity >3.0 generally indicate high quality benthic habitats. While there is little evidence of high levels of nitrogen related impairment of the benthic animal communities, most areas did show clear evidence of moderate to significant impairment associated with nitrogen and organic matter enrichment.

Each of the estuarine basins, specifically Lower Chilmark Pond (east), Wades Cove and Gilberts Cove are showing moderate-significant levels of impairment related to their elevated chlorophyll-*a* levels and moderate periodic oxygen depletions. While the numbers of individuals remain high throughout the system, the community numbers of species and their diversity and Evenness are low and indicative of a community under ecological stress. In all cases, these basins support communities with low diversity, with the measured index only 1.45 to 2.21. Very similar to impaired benthic communities in the tributary coves of nearby Tisbury Great Pond, 1.44 to 1.82. Evenness (how individuals are distributed among the species) was similarly low in Chilmark Pond (0.54-0.66) and indicated that only a few species were accounting for most of the individuals within each basin. There was little substantive difference between the basins as all are clearly moderately impaired relative to benthic animal habitat. For comparison the moderately impaired lower basins of Tisbury Great Pond had similar diversity indices of 2.0 to 2.3 and Evenness of 0.49 to 0.54. However, in Chilmark Pond's estuarine basins the dominant species are not opportunistic stress indicators (generally <10%), but are tolerant of moderate levels of organic enrichment (*Streblospio* and *Leptocheirus,* an amphipod). Streblospio was a dominant species within the coves tributary to Tisbury Great Pond as well. Given the prevalence of species tolerant of moderate organic enrichment, the low numbers of stress indicator organisms, the low numbers of species and the low diversity of Chilmark Pond's benthic communities compared to high quality habitat areas in similarly structured embayments in southeastern Massachusetts, it is clear that the main basin of Chilmark Pond and the major coves (Wades, Gilberts) are currently above their nitrogen threshold and is supporting impaired benthic animal habitat.

The results of the infauna survey and complete absence of eelgrass coverage within the Chilmark Pond Embayment System indicates that the nitrogen management threshold analysis (Section VIII-2) needs to aim for lowering nitrogen enrichment for restoration of infaunal habitat in the tributary coves showing moderate-high impairment of benthic habitat. Reduction in nitrogen enrichment is required for restoration. It should be emphasized that reducing nitrogen enrichment can be achieved by reducing nitrogen inputs and/or increasing the rate of nitrogen loss through enhanced tidal exchange. Restoring these benthic habitats should be the focus of the nitrogen management threshold analysis (Section VIII-2).

Table VIII-1. Summary of nutrient related habitat quality within the Chilmark Pond Embayment System within the Town of Chilmark, MA, based upon assessments in Section VII. The estuarine reaches of Chilmark Pond consist of a main basin (Lower Chilmark Pond) has 2 tributary coves (Wades, Gilberts). The western basin (upper Chilmark Pond) has been separated from the estuary by overwash and is freshwater and not part of the present analysis. WQMP indicates MVC-Town Water Quality Monitoring Project.

Health Indicator	Chilmark Pond Embayment System		
	Lower (East) Chilmark Pond	Wades Cove	Gilberts Cove
Dissolved Oxygen	MI[1]	MI[2]	H/MI[3]
Chlorophyll	MI/SI[4]	MI/SI[4]	MI[5]
Macroalgae	H[6]	H[6]	H[6]
Eelgrass	--[7]	--[7]	--[7]
Infaunal Animals	MI/SI[8]	MI/SI[9]	MI/SI[10]

Overall:	MI/SI[11]	MI/SI[11]	MI[12]

1 – oxygen frequently >6 mg/L, 88% of record and rarely <4 mg/L 2% of record, generally 6-10 mg/L; WQMP >5 mg/L.
2 – oxygen depletion frequently <6 mg/L, 50% , >5 mg/L 21% of record, infrequent brief excursions to <4 mg/L.
3 – oxygen depletion infrequently <5 mg/L 8%, rare very brief excursions to <4 mg/L.
4 – high summer chlorophyll levels generally ~20 ug/L, averaging 33 ug/L at mooring site, average level in summer, large blooms to 80 ug/L during mooring period; WQMP was 8.6-8.9 ug/L, stations CHP-3 & 5 with bloom years >20 ug/L. Large blooms not a consistent summer phenomenon.
5 – moderate summer chlorophyll levels generally <10 ug/L, averaging 8 ug/L at mooring site, mean summer WQMP generally ~5 ug/L, 2005 bloom year ~25 ug/L.
6 -- generally very sparse drift algae and no apparent algal mat, some *Ruppia*.
7 – no evidence that this basin historically supported eelgrass habitat;
8 -- moderate-high numbers of individuals, low # species, low diversity & Evenness, dominated by organic enrichment species (Streblospio, Amphipods), some stress tolerant opportunists (Tubificids, *Capitella*,7% of org)
9 -- high numbers of individuals (~500), low # species (11), low diversity (H'=2.2) & moderate Evenness (0.66), dominated by organic enrichment species, some stress tolerant opportunists (Tubificids, *Capitella*, 13% of organisms)
10 -- high numbers of individuals (>600), low # species (7), low diversity (H'=1.5) & Evenness (0.54), dominated by a few organic tolerant species (Amphipods), but very few opportunists (Tubificids+*Capitella*= 4% of organisms)
11 -- Moderate-Significant Impairment , primarily due to periodic D.O. depletion and large phytoplankton blooms, benthic animal communities contain few species with low diversity and Eveness, with some stress indicator species,, macroalgae absent.
12 -- Moderate Impairment, only moderate oxygen depletion and chlorophyll levels, with summer averages ~8 ug/L (i.e. <10ug/L) few species and low diversity dominated by amphipod mat, but with very few opportunistic stress indicator species, macroalgae absent.

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. The sentinel location is selected such that the restoration of that one site will necessarily bring the other regions of the system to acceptable habitat quality levels. Once the sentinel site and its target nitrogen level are determined, the Linked Watershed-Embayment Model is used to sequentially adjust nitrogen loads until the targeted nitrogen concentration is achieved.

Within the Chilmark Pond Estuary the most appropriate sentinel "station" was to use the average of the 5 long-term monitoring stations (CHP1-5) distributed throughout the main eastern basin, Gilberts Cove and Wades Cove (Figure II-1). This average approach has been used in other open "single basin" estuaries that are only periodically open to tidal flow throughout the MEP region. The average was selected because given the relatively long periods between openings, dispersion and wind driven mixing result in a relatively uniform total nitrogen concentration throughout the estuary. In addition, the benthic animal community is also generally uniform in numbers of organisms and species composition (dominated by an amphipod, *Leptocheirus* and *Streblospio*). It appears that the estuarine reaches of Chilmark Pond are presently functioning as a single basin with of relatively uniform nitrogen levels and habitat quality.

Overall, the infauna survey indicated that the main eastern basin, Wades and Gilberts Coves, which comprise the bulk of the Chilmark Pond Estuary, presently support moderately to significantly impaired benthic infaunal habitat. It appears that organic deposition in these basins is the cause of the stress, consistent with the bottom water oxygen levels and phytoplankton biomass. Animal communities colonizing sediments within throughout the Estuary show low to moderate diversity (7-11) with moderate evenness (0.54-0.66) and high productivity (~400-700 individuals per sample). Equally important, the species dominating the communities were generally representative of moderately stressed environments, with some benthic communities being dominated by an amphipod (*Leptocheirus*) and polychaete (*Streblospio*) common to nitrogen enriched estuaries in southeastern Massachusetts, some areas had some deep burrowers. High numbers of organic enrichment indicators (tubificids, capitellids) were not observed (generally <10% of community).

The benthic animal communities were compared to high quality environments, such as the Outer Basin of Quissett Harbor, indicating a level of impairment throughout the Chilmark Pond Estuary. The Outer Basin of Quissett Harbor supports benthic animal communities with $\geq$28 species, >400 individuals with high diversity (H' $\geq$3.7) and Evenness (E $\geq$0.77). Similarly, outer stations within Lewis Bay in Barnstable currently support similarly high quality benthic habitat as seen in the numbers of individuals (502 per sample), number of species (32), diversity (3.69) and Eveness (0.74). Equally important these communities are not consistent with nutrient enrichment being composed of a variety of polychaete, crustacean and mollusk species, as opposed to stress tolerant small opportunistic oligochaete worms.

Classification of habitat quality necessarily included the structure of the estuarine basin, specifically that it is fully representative of a tidal embayment, as opposed to a tidal river or salt marsh basin. Integration of all of the metrics clearly indicates that the basins of Chilmark Pond Estuary are generally supporting benthic animal habitat that is moderately to significantly impaired. The proximate cause of impairment is organic matter enrichment and oxygen depletion, stemming ultimately from nitrogen enrichment. Nitrogen enrichment of the Chilmark Pond Estuary stems from the combination of watershed nitrogen load and the absence of tidal exchange with offshore waters except during "dredged' openings by the Town. More frequent or prolonged openings has the same effect of lowering nitrogen loads, relative to relieving nitrogen related habitat impairments.

Following the MEP protocol, since eelgrass has not been documented in the Chilmark Pond Estuary, restoration of impaired infaunal habitat is the restoration goal. Infaunal animal habitat is a critical resource to the Chilmark Pond Estuary and estuaries in general. Since there are no unimpaired infaunal animal habitat areas remaining in the Chilmark Pond Estuary, comparisons to the soft bottom basins of other nearby estuarine systems were relied upon for setting the nitrogen threshold for healthy infaunal habitat at a nitrogen level of TN <0.5 mg TN L^{-1}. This level was found for Popponesset Bay where based upon the infaunal analysis coupled with the nitrogen data (measured and modeled), nitrogen levels on the order of 0.4 to 0.5 mg TN L^{-1} were found supportive of high infaunal habitat quality in this system. Similarly, in the Three Bays System, healthy infaunal areas are found at nitrogen levels of TN <0.42 mg TN L^{-1} (Cotuit Bay and West Bay), with impairment in areas where nitrogen levels of TN >0.5 mg TN L-1 (North Bay), and severe degradation at nitrogen levels of TN >0.6 mg TN L-1. Similarly, impaired benthic habitat was quantified within the Fiddlers Cove and within the upper terminal basins of Rands Harbor at TN levels of 0.56 mg TN L^{-1} and 0.57 mg TN L^{-1}, respectively, supporting the contention that levels $\leq$0.50 mg N L^{-1} are needed for restoration of impaired benthic animal habitat in southeastern Massachusetts estuaries.

It should also be noted that in numerous estuaries it has been previously determined that 0.500 mg TN L^{-1} is the upper limit to sustain unimpaired benthic animal habitat (e.g. Eel Pond {Waquoit}, Parkers River, upper Bass River, upper Great Pond, Rands Harbor and Fiddlers Cove). Present TN levels within the Chilmark Pond Estuary during summer are ~0.74 mg TN L^{-1}, consistent with the observed lack of eelgrass beds and impaired benthic animal habitat. Based upon comparisons to other systems, the current TN level within the Chilmark Pond Estuary, the periodic oxygen depletions and phytoplankton blooms, it appears that a water column nitrogen threshold for the Chilmark Pond Estuary of 0.50 mg TN L^{-1} is required for restoration. All habitat metrics indicate a moderate to moderate/significant level of habitat impairment (Table VIII-1). While the TN level is significantly above the threshold (0.74 versus 0.5 mg TN L^{-1}) the system is still supporting a productive if clearly impaired benthic animal community.

Given the relatively low watershed nitrogen load to the Chilmark Pond estuarine basins, it may be difficult to lower TN levels by ~0.2 mg L^{-1} to meet the threshold. This is consistent with the MEP measurements of periodically opened basins and systems with significantly restricted tidal flows, such as Rushy Marsh Pond, Farm Pond. In such cases increases in the amount, duration or frequency of tidal exchange, generally through openings is needed to lower the level of nitrogen enrichment and restore the impaired habitats. This will likely be the case for Chilmark Pond, as well.

The response of the Chilmark Pond Estuary's nitrogen levels to reductions in watershed nitrogen inputs to achieve the TN threshold are developed in the next section (VIII.3).

VIII.3. DEVELOPMENT OF TARGET NITROGEN LOADS

After developing the dispersion-mass balance model of Chilmark Pond to simulate conditions that exist as a result of present management practices, the model was used to simulate a modified management approach that could be followed to improve water quality conditions in the pond year-round.

With a goal of seeking further improvements in water quality conditions in the Pond, an alternate management scheme was modeled using the previously developed dispersion-mass balance model. The main goals of this threshold load scenario were to restore benthic infauna habitat throughout Chilmark Pond and simultaneously attempt to restore a modest level of eelgrass habitat within the main basin which has been non-existent over the past several decades. To restore benthic habitat, load reduction focused on lowering average TN levels of stations with the main basin to 0.50 mg/L during the summer months, when benthic regeneration and algae production is greatest. This goal was achieved by reducing the watershed loading to the pond and assuming the pond is breached three times a year. Watershed loading was reduced from present conditions until the combined time averaged TN concentration would remain below 0.50 mg/L during a 120-day period during the summer months. The threshold modeling assumptions include a successful spring breach, which remains open for 8 days and lowers the average pond TN concentration to 0.33 mg/L. The Pond is also allowed to be closed for 120 days, which allows the time for the water level in the pond to rise using Martha's Vineyard Commission's groundwater/surface water discharge rates as discussed in Section VI.

The resulting threshold septic loading is presented in Table VIII-2. A 37.5% septic (attenuated) reduction from present conditions was required in the septic load to the pond to achieve the threshold requirements. All other watershed sources, including agricultural loads, were not reduced for this scenario. A tabulation of all the loads to the pond is provided in Table VIII-3. The benthic loading term is effected by the change in watershed load.

Table VIII-2. Comparison of embayment attenuated septic loads used for modeling of present and modeled threshold loading scenarios of Chilmark Pond. Septic loads are from existing residential and commercial properties. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	Present Septic N Load (kg/day)	Threshold (kg/day)	Threshold change
Chilmark Pond East	3.074	1.884	40.0%
Chilmark Pond West	3.068	1.995	35.0%

Table VIII-3. Embayment and surface water loads used for total nitrogen modeling of threshold conditions for Chilmark Pond, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	Threshold N Load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Chilmark Pond East	4.255	3.260	-0.297
Chilmark Pond West	10.540	0.655	-2.924

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