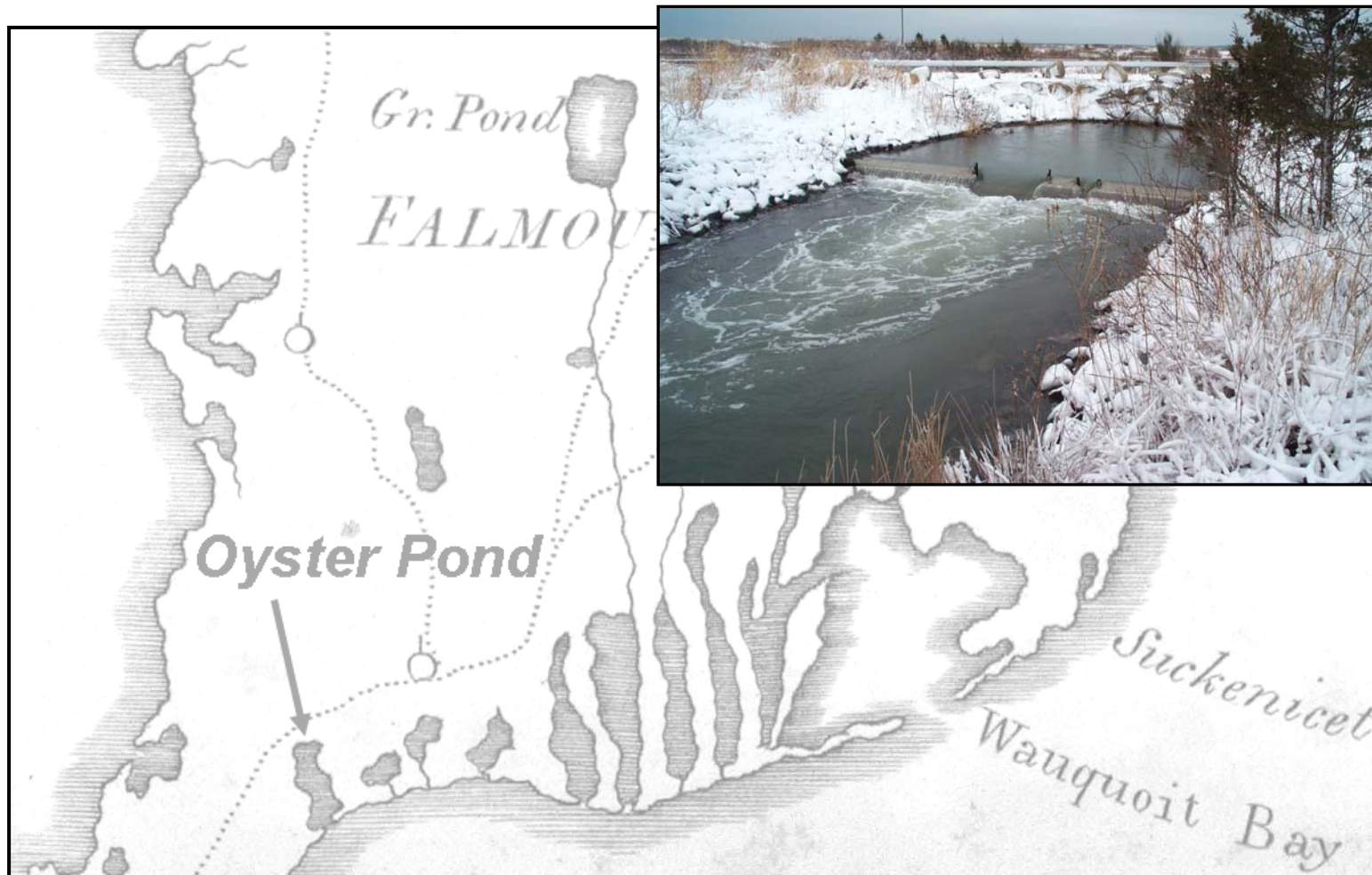


MASSACHUSETTS ESTUARIES PROJECT

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Oyster Pond System, Falmouth, Massachusetts



University of Massachusetts Dartmouth
School of Marine Science and Technology



Massachusetts Department of
Environmental Protection

FINAL REPORT – JANUARY 2006

Massachusetts Estuaries Project

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TABLE OF CONTENTS

I. INTRODUCTION	1
I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH	4
I.2 SITE DESCRIPTION	9
I.3 NITROGEN LOADING	13
I.4 WATER QUALITY MODELING	15
I.5 REPORT DESCRIPTION	16
II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT	17
III. DELINEATION OF WATERSHEDS	20
III.1 BACKGROUND	20
III.2 MODEL DESCRIPTION	20
III.3 OYSTER POND CONTRIBUTORY AREAS	21
IV. WATERSHED NITROGEN LOADING TO EMBAYMENT: LAND USE, STREAM INPUTS, AND SEDIMENT NITROGEN RECYCLING	25
IV.1 WATERSHED LAND USE BASED NITROGEN LOADING ANALYSIS	25
IV.1.1 Land Use and Database Preparation	26
IV.1.2 Nitrogen Loading Input Factors	28
IV.1.3 Calculating Nitrogen Loads	35
IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT	38
IV.2.1 Background and Purpose	38
IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Mosquito/Quivett Creek Discharge to Oyster Pond (upper)	42
IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS	49
IV.3.1 Sediment-Watercolumn Exchange of Nitrogen	49
IV.3.2 Method for determining sediment-watercolumn nitrogen exchange	50
IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments	52
V. HYDRODYNAMIC MODELING	56
V.1 INTRODUCTION	56
V.1.1 System Physical Setting	56
V.1.2 System Hydrodynamic Setting	57
V.2 GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE SYSTEM	59
V.2.1 Coastal Processes and Inlet Stability	60
V.2.2 Shoreline Change Analysis	61
V.2.3 Inlet Management Implications	66
V.3 HYDRODYNAMIC FIELD DATA COLLECTION AND ANALYSIS	66
V.3.1. Bathymetry	67
V.3.2 Tide Data Collection and Analysis	67
V.4 HYDRODYNAMIC MODEL DEVELOPMENT	74
V.4.1 Weir model development	74
V.4.2 Application of the weir model	76
V.4.2 RMA2 model development	79
VI. WATER QUALITY MODELING	81
VI.1 DATA SOURCES FOR THE MODEL	81

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayments	81
VI.1.2 Nitrogen Loading to the Embayments	81
VI.1.3 Measured Nitrogen Concentrations in the Embayments.....	81
VI.2 MODEL DESCRIPTION AND APPLICATION	82
VI.2.1 Model Formulation.....	83
VI.2.2 Water Quality Model Setup	84
VI.2.3 Boundary Condition Specification	84
VI.2.4 Model Calibration	85
VI.2.5 Model Salinity Verification	87
VI.2.6 Build-Out and No Anthropogenic Load Scenarios.....	89
VI.2.6.1 Build-Out.....	89
VI.2.6.2 No Anthropogenic Load	91
VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH	93
VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS.....	94
VII.2 BOTTOM WATER DISSOLVED OXYGEN.....	94
VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS	98
VII.4 BENTHIC INFAUNA ANALYSIS	98
VIII. CRITICAL NUTRIENT THRESHOLD DETERMINATION AND DEVELOPMENT OF WATER QUALITY TARGETS	101
VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY	101
VIII.2. THRESHOLD NITROGEN CONCENTRATIONS	102
VIII.3. DEVELOPMENT OF TARGET NITROGEN LOADS	104
IX. LIST OF REFERENCES	111

LIST OF FIGURES

Figure I-1.	Study region proximal to the Oyster Pond embayment system for the Massachusetts Estuaries Project nutrient analysis. Tidal waters periodically enter the salt pond through one inlet to Vineyard Sound. Freshwaters enter from the watershed primarily through 1 surface water discharge (stream to the head of Oyster Pond) and direct groundwater discharge.....	3
Figure I-2.	Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach	8
Figure I-3.	Partitioning of the Oyster Pond embayment system relative to the three main basins, used for volume calculations.....	10
Figure I-4.	Oyster Pond embayment bathymetry (from K.O. Emery 1997).....	11
Figure I-5.	Physiographic units of the land adjacent the Oyster Pond embayment system (from K.O. Emery 1997).....	12
Figure III-1.	Watershed and sub-watershed delineations for the Oyster Pond estuary system. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a “10” in the watershed names (above). Sub-watersheds to embayments were selected based upon the functional estuarine sub-units in the water quality model (see section VI).	22
Figure III-2.	Comparison of previous and current Oyster Pond watershed and subwatershed delineations.....	24
Figure IV-1.	Land-use coverage in the Oyster Pond watershed. Watershed data encompasses portions of the Town of Falmouth and land use classifications are based on Town of Falmouth Planning Department records.	27
Figure IV-2.	Distribution of land-uses within the major subwatersheds and whole watershed to Oyster Pond.....	29
Figure IV-3.	Parcels, Parcelized Watersheds, and Developable Parcels in the Oyster Pond watersheds. The Developable Commercial Parcel is owned by WHOI.....	33
Figure IV-4 (a-c).	Land use-specific unattenuated nitrogen load (by percent) to the (a) Oyster Pond Main subwatershed, (b) Oyster Pond South subwatershed, and (c) Mosquito Creek subwatershed. “Overall Load” is the total nitrogen input within the watershed, while the “Local Control Load” represents only those nitrogen sources that could potentially be under local regulatory control.....	37
Figure IV-5.	Location of stream gauge on Mosquito/Quivett Creek (red triangle), which discharges to the brackish waters of Oyster Pond.	41
Figure IV-6.	Mosquito Creek (Quivett Creek) discharge (solid pink line) and nitrate+nitrite (blue boxes) concentrations for determination of annual volumetric discharge and nitrogen load from the Mosquito Creek sub-watershed to the Oyster Pond system (Table IV-5).....	45
Figure IV-7.	Mosquito Creek (Quivett Creek) discharge (solid pink line) and total nitrogen (yellow triangles) concentrations for determination of annual volumetric discharge and nitrogen load from the Mosquito Creek sub-watershed to the Oyster Pond System (Table IV-5).	46
Figure IV-8.	Mosquito Creek (Quivett Creek) discharge (solid pink line), Particulate Organic Nitrogen (green diamonds) and Dissolved Organic Nitrogen (blue	

	triangles) concentrations for determination of annual volumetric discharge and nitrogen load from the Mosquito Creek sub-watershed to the Oyster Pond System.	47
Figure IV-9.	Mosquito Creek (Quivett Creek) discharge (solid pink line), Particulate Organic Nitrogen (green diamonds) and Dissolved Organic Nitrogen (blue triangles) loads for determination of annual volumetric discharge and nitrogen load from the Mosquito Creek sub-watershed to the Oyster Pond System.	48
Figure IV-10.	Oyster Pond embayment system sediment sampling sites (red triangles) for determination of nitrogen regeneration rates. Blue diamond represents stream gauge location on Mosquito/Quivett Creek. Numbers are for reference in Table IV-7.	51
Figure IV-11.	Total nitrogen concentrations in the Oyster Pond mixed layer (0-4m) courtesy of Falmouth PondWatch. Day 0 is in mid July (~15-July) in each year, 1997-2004.	55
Figure V-1.	Topographic map detail of Oyster Pond, in Falmouth, Massachusetts, with annotations indicating the location of the salinity control structure (weir), Lagoon, and Truck River inlet.	57
Figure V-2.	Schematic cross-sectional view of the Oyster Pond salinity control/fish weir (facing into the pond), with dimensions and elevations (not drawn to scale). Weir crest elevations are as observed in March 2001 and August 2003.	58
Figure V-3.	View (circa 1999) looking offshore into Vineyard Sound from the Trunk River inlet prior to reconstruction of the jetties and inlet channel (photograph credit: John Ramsey).	59
Figure V-4.	A portion of an 1800 map showing the south coast of Falmouth and Mashpee. This map depicts the condition of these inlets prior to the installation of jetties.	61
Figure V-5.	The 2001 aerial photograph showing scaled transects that indicate computed shoreline change rates between 1938 and 2004.	65
Figure V-6.	Shoreline change data distribution calculated along the south coast of Falmouth (from 2500 ft southwest of the Trunk River jetties to Falmouth Harbor) between 1938 and 2004.	66
Figure V-7.	The Oyster Pond weir and canoe set-up for the bathymetry survey of the Pond in August 2003 (photograph credit: Sean Kelley).	68
Figure V-8.	Bathymetry survey lines (yellow) followed by canoe in Oyster Pond. Tide gauge stations in the Pond and Lagoon are indicated by yellow dots T2 and T3. The final RMA2 grid used for hydrodynamic and water quality modeling is also shown as the white triangular mesh.	69
Figure V-9.	Tide records from March 2001 deployment period for Oyster Pond. Captured in the data are four storm events, at March 7th, 14th, 23rd, and 30th.	70
Figure V-10.	Oyster Pond tides collected during a quiescent summer period in 2003. The bathymetry survey of Oyster Pond was performed August 25.	70
Figure V-11.	Example of observed astronomical tide as the sum of its primary constituents. In this example the observed tide signal is the sum of individual constituents (M2, M4, K1, N2), with varying amplitude and frequency.	71
Figure V-12.	Results of the harmonic analysis and the separation of the tidal from the non-tidal, or residual, signal measured offshore Oyster Pond in Vineyard Sound during the 2001 gauge deployment.	72

Figure V-13.	Results of the harmonic analysis and the separation of the tidal from the non-tidal, or residual, signal measured offshore Oyster Pond in Vineyard Sound during the 2003 gauge deployment.....	73
Figure V-14.	Oyster Pond weir overtopped from the Lagoon side during the early March 2001 northeast storm. At the time of the photograph, the culvert under Oyster Pond Road is completely submerged (photograph credit: Sean Kelley).	77
Figure V-15.	Close up view of the Oyster Pond weir as it is overtopped from the Lagoon side during the northeast storm of March 6 and 7, 2001. For this photograph, left is toward the pond (photograph credit: Sean Kelley).....	77
Figure V-16.	Detail of the March 2001 tide record from Oyster Pond.	78
Figure V-17.	Comparison of measured tide data and weir model output for a northeast storm event (R^2 correlation coefficient of 0.94, rms error of 0.04 ft). Also plotted are water level variations measured in the Trunk River lagoon, used as the boundary condition for the weir model. This simulation begins at 1200h EST March 5, 2001.....	79
Figure V-18.	Weir model output flow rate over the Oyster Pond weir during the March 2001 northeast storm event.....	79
Figure V-19.	Interpolated bathymetric contours of the final RMA2 computational mesh of Oyster Pond. Depth contours are relative to the NGVD 29 vertical datum, and are at 3-foot intervals.....	80
Figure VI-1.	Estuarine water quality monitoring station locations in Oyster Pond. Station labels correspond to those provided in Table VI-1.....	82
Figure VI-2.	Comparison of measured total nitrogen concentrations and calibrated model output at stations in the Oyster Pond system. For the left plots, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate \pm one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line.	86
Figure VI-3.	Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for Oyster Pond.....	87
Figure VI-4.	Comparison of measured and calibrated model output at stations in the Great Pond. For the left plots, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed salinity for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate \pm one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line.	88
Figure VI-5.	Contour Plot of modeled salinity (ppt) in Oyster Pond.....	88
Figure VI-6.	Contour plot of modeled total nitrogen concentrations (mg/L) in Oyster Pond, for projected build-out loading conditions.....	90
Figure VI-7.	Contour plot of modeled total nitrogen concentrations (mg/L) in Oyster Pond, for no anthropogenic loading conditions.	92

Figure VII-1.	Salinity of Oyster Pond surface waters over the past 60 years. The salinity increase over 4 ppt from 1987-1995 resulted from the installation of a new culvert in the tidal channel between the Lagoon and the main basin. The recent lowering and stabilization of the salinity has resulted from the installation of a salinity control/fish weir upgradient of the culvert. Salinity reconstruction courtesy of B.L. Howes.	93
Figure VII-2.	Average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Howes et al., 2004). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.....	95
Figure VII-3.	Bottom water dissolved oxygen levels measured by Falmouth PondWatch during July and August, determined from grab samples and Winkler Titrations (to 0.5 mg/L). Note that OP 1, 4 m (within the enclosed northern basin) and OP 3, 6 meters (within the deep southern basin) periodically go anoxic. The benchmark for the threshold analysis is OP-3 at 4 meters, this site had D.O. > 3 mg/L on 96% of sampling dates.	97
Figure VII-4.	Aerial photograph of Oyster Pond showing location of benthic infaunal sampling stations (red symbol).....	99
Figure VIII-1.	Contour plot of modeled average total nitrogen concentrations (mg/L) in the Oyster Pond system, for threshold conditions (minimum 3.8 mg/L Dissolved Oxygen pond-wide).....	108
Figure VIII-2.	Contour plot of modeled average total nitrogen concentrations (mg/L) in the Oyster Pond system, for threshold conditions (minimum 5.0 mg/L Dissolved Oxygen pond-wide).....	109
Figure VIII-3.	Contour plot of modeled average total nitrogen concentrations (mg/L) in the Oyster Pond system, for threshold conditions (minimum 6.0 mg/L Dissolved Oxygen pond-wide).....	110

LIST OF TABLES

Table III-1.	Daily groundwater discharge to each of the sub-embayments in the Oyster Pond system, as determined from the USGS groundwater model.	23
Table IV-1.	Percentage of unattenuated nitrogen loads in less than 10 year time of travel subwatersheds to Oyster Pond. Note that Oyster Pond South is to the salt marsh Lagoon which down-gradient from Oyster Pond proper.	26
Table IV-2.	Average Water Use in Oyster Pond Watershed.	32
Table IV-3.	Primary Nitrogen Loading Factors used in Oyster Pond MEP analysis. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Falmouth data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.	34
Table IV-4.	Oyster Pond Nitrogen Loads. Attenuation of system nitrogen loads occurs as nitrogen moves through the Mosquito Creek stream system during transport to the estuary. Atmospheric deposition of nitrogen to the estuarine surface was determined (main basin = 282 kg yr ⁻¹ , Lagoon = 10 kg yr ⁻¹) and used in the modeling but was not included in this table.	36
Table IV-5.	Comparison of water flow and nitrogen discharges from Mosquito Creek (Quivett Creek) discharging to Oyster Pond. The "Stream" data is from the MEP stream gauging effort. Watershed data is based upon the MEP watershed modeling effort by USGS and CCC.	43
Table IV-6.	Summary of annual volumetric discharge and nitrogen load from Mosquito Creek (Quivett Creek) to the head of Oyster Pond based upon the data presented in Figures IV-6, 7, 8, 9 and Table IV-5.	44
Table IV-7.	Rates of net nitrogen return from sediments to the overlying waters of the Oyster Pond embayment system. These values are combined with the basin area to determine total nitrogen mass for the water quality model (see Section VI). Measurements represent July -August rates. The negative nitrogen return is consistent with Oyster Pond's physical structure as a brackish kettle pond.	52
Table V-1.	Estimates of potential error associated with shoreline position surveys.	65
Table V-2.	Tidal Constituents for stations in the Lagoon Pond System, from data collected July 25, 2003 14 to September 11, 2003.	71
Table V-3.	Tide datums computed from tide record collected in Vineyard Sound July 25, 2003 14 to August 25, 2003. Datum elevations are given relative to NGVD 27.	74
Table V-4.	Model input parameters used for simulations of the Oyster Pond weir.	78
Table VI-1.	Falmouth PondWatch measured data, and modeled Nitrogen concentrations for the Oyster Pond system used in the model calibration plots of Figure VI-2. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of the separate yearly means.	82
Table VI-2.	Sub-embayment and surface water loads used for total nitrogen modeling of the Oyster Pond system, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent present loading conditions for the listed sub-embayments.	85
Table VI-3.	Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic ("no-load") loading scenarios of Oyster Pond. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.	89

Table VI-4.	Build-out sub-embayment and surface water loads used for total nitrogen modeling of Oyster Pond, with total watershed N loads, atmospheric N loads, and benthic flux.....	90
Table VI-5.	Comparison of model average total N concentrations from present loading and the build-out scenario, with percent change, Oyster Pond.	90
Table VI-6.	“No anthropogenic loading” (“no load”) sub-embayment and surface water loads used for total nitrogen modeling Oyster Pond, with total watershed N loads, atmospheric N loads, and benthic flux	91
Table VI-7.	Comparison of model average total N concentrations from present loading and the no anthropogenic (“no load”) scenario, with percent change, for Oyster Pond. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).	91
Table VII-1.	Frequency of grab samples for summer chlorophyll a levels above various benchmark levels within the surface waters (0-3.5m) of the Oyster Pond System. Data courtesy of the Falmouth PondWatch Water Quality Monitoring Program and Coastal Systems Program, SMAST. Geometric averages were used to estimate “average” conditions, given the periodic phytoplankton blooms.	97
Table VII-2.	Benthic infaunal community data for the Oyster Pond embayment system. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.0625 m ²).	100
Table VIII-1.	Average total N concentrations to achieve target bottom water oxygen minima. Based upon modeled nitrogen reductions for the Oyster Pond system. Selected sentinel station is OP-3	103
Table VIII-2.	Comparison of sub-embayment watershed septic loads (attenuated) used for modeling of present and threshold loading scenarios of the Oyster Pond system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.	105
Table VIII-3.	Comparison of sub-embayment total attenuated watershed loads (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Oyster Pond system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.....	106
Table VIII-4.	Threshold sub-embayment and surface water loads used for total nitrogen modeling of the Oyster Pond system, with total watershed N loads, atmospheric N loads, and benthic flux.....	106
Table VIII-5.	Comparison of model average total N concentrations from present loading and the modeled threshold scenario, with percent change, for the Oyster Pond system. Sentinel threshold stations are in bold print.	107

I. INTRODUCTION

The Oyster Pond embayment system is located within the Town of Falmouth, on Cape Cod Massachusetts. The system has a southern shore bounded by water from Vineyard Sound (Figure I-1). The open waters and watershed of this salt pond system are fully within the Town of Falmouth. The present configuration of the Oyster Pond embayment results from tidal flooding of a multi-basin coastal kettle pond, separated from the ocean by a barrier beach and with a small creek discharging into the inner- most basin. Flooding of the kettle pond resulted from rising sea level at the end of the last glacial period. Oyster Pond is a relatively young embayment, cores indicate that salinities of present day Vineyard Sound waters were reached throughout Pond basins ~1300 years B.P. (Emery, 1997).

As is typical with other Falmouth embayments (Great, Green, and Bournes Pond) Oyster Pond is separated from Vineyard Sound by a barrier beach and a small saline lagoon. The Oyster Pond embayment exchanges tidal water with Vineyard Sound. Tidal flow is through the Trunk River, whose west branch flows to Oyster Pond through the Lagoon. The beach and the opening to the lagoon are very dynamic geomorphic features due to the influence of littoral transport processes. As a result of periodic sedimentation, generally associated with storms, the Trunk River flows must be maintained to sustain the salt marshes within the Lagoon and the brackish nature of Oyster Pond. At present, the Lagoon is separated from Oyster Pond by Oyster Pond Road/Surf Drive. There is a culvert passing under the road that connects the Lagoon to Oyster Pond, however, a control structure was designed and installed just up-gradient of the culvert at the entrance to Oyster Pond to control and stabilize the salinity of Oyster Pond waters as well as to allow passage of anadromous fish. The control structure was originally designed such that only the highest spring tides would interact with Oyster Pond in order that the salinity of Oyster Pond could be maintained between 2 and 4 ppt. However, to become fully functional, the tidal channel between the Trunk River and the Lagoon needs to be sufficiently open to allow free tidal exchange between the Lagoon and Vineyard Sound. Occlusion of this channel has periodically resulted in a slight freshening of Oyster Pond waters beyond their management limits. The need for periodic maintenance of tidal inlets to sustain tidal flows is typical of the flood dominated embayments on Cape Cod. For example, Bournes Pond (Falmouth) became very restricted and finally completely isolated from Vineyard Sound waters in the late 1970's/early 1980's and was re-opened with a fixed inlet in mid 1980's. Protection of the natural resources of Oyster Pond (and the Lagoon) requires maintenance of the tidal flows. The tidal control structure and pond salinities will be discussed further in Sections V and VII. .

Similar to the Great, Green, Bournes and Little Pond embayment systems, Oyster Pond is a shallow coastal salt pond located within a glacial outwash plain, the Mashpee Pitted Plain, consisting of material deposited after the retreat of the Cape Cod Lobe of the Laurentide Ice sheet ~18,000 years ago. The outwash material is highly permeable and varies in composition from well-sorted medium sands to coarse pebble sands and gravels (Millham and Howes, 1994). As such, direct rainwater run-off to the adjacent estuaries is generally small with most freshwater inflow via groundwater discharge or groundwater fed surface water flow (e.g. stream to the head of Little Pond, Coonamessett River to Great Pond, Backus River to Green Pond etc.). Oyster Pond currently functions as a brackish great salt pond receiving minimal terrestrial fresh, surface water inflow (called alternatively Mosquito Creek or Quivett Creek), being mainly supplied by groundwater discharges and with very restricted tidal inflows of saline Vineyard Sound waters. The salinity characteristic of the "salt" pond varies with the volume of freshwater

inflow as well as the effectiveness of tidal exchange with Vineyard Sound. At present the Pond waters are slightly above 2 ppt, within the management target range of 2-4 ppt.

Oyster Pond, along with the other salt pond embayments along Falmouth's south coast, constitutes an important component of the Town's natural and cultural resources. In addition, the pond's location in a heavily residential area of Falmouth greatly increases the potential for direct discharges from homes situated on the shore and decreases the travel time of groundwater from the watershed recharge areas to the salt pond. Moreover, given that the embayment has limited tidal exchange (flushing) and is relatively deep (mean depth 3m, maximum depth ~6.5m) Oyster Pond is relatively sensitive to the effects of nutrient enrichment from watershed based sources. In general, the nature of enclosed embayments in populous regions brings two opposing elements to bear: as protected marine shoreline they are popular regions for boating, recreation, and land development; as enclosed bodies of water, they may not be readily flushed of the pollutants that they receive due to the proximity and density of development near and along their shores. In particular, Oyster Pond as well as the Great, Green, Bournes and Little Pond embayment systems along the southern shore of Falmouth are at risk of eutrophication from high nitrogen loads in the groundwater and runoff from their watersheds.

The primary ecological threat to Oyster Pond resources is degradation resulting from nutrient enrichment. Loading of the critical eutrophying nutrient, nitrogen, to the embayment waters has been greatly increased over the past several decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to this and other Falmouth salt ponds, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater. The Town of Falmouth has been among the fastest growing towns in the Commonwealth over the past three decades and does not have centralized wastewater treatment throughout the entire Town. As existing and probable increasing levels of nutrients impact Falmouth's coastal embayments, water quality degradation will accelerate, with further harm to invaluable environmental resources.

The Town of Falmouth (via the Planning Office) was one of the first communities in Massachusetts to become concerned over perceived degradation of its embayment systems due to nutrient overloading. In the mid-1980's the Town enacted an innovative Nutrient Overlay By-law that tied watershed development to water quality within the adjacent embayment. Nutrient limits were set for nitrogen in each of the Town's embayments. The goal was to keep nitrogen concentrations in the receiving systems below thresholds that were projected to cause water quality shifts. To acquire baseline water quality data necessary for ecological management of Falmouth's coastal salt ponds and harbors, a citizen-based water quality monitoring program was initiated by the Town of Falmouth. Falmouth Pondwatch, was established to provide on-going nutrient related embayment health information in support of the By-law. The water quality monitoring program was based on a collaborative effort between scientists, citizens and representatives of the Town of Falmouth. As originally conceived, the monitoring program focused on data collection in three original ponds, Oyster Pond, Little Pond and Green Pond. By 1990, the scope of water quality data collection expanded to include two additional ponds, Great/Perch Pond and Bournes Pond. In 1992, the scope of data collection was once again expanded to include West Falmouth Harbor in order to evaluate the effects from a nutrient enriched wastewater plume generated by the Falmouth Wastewater Treatment Facility. Since 1997, technical aspects of the Falmouth PondWatch Program have been coordinated through the Coastal Systems Program at SMAST-UMassD.

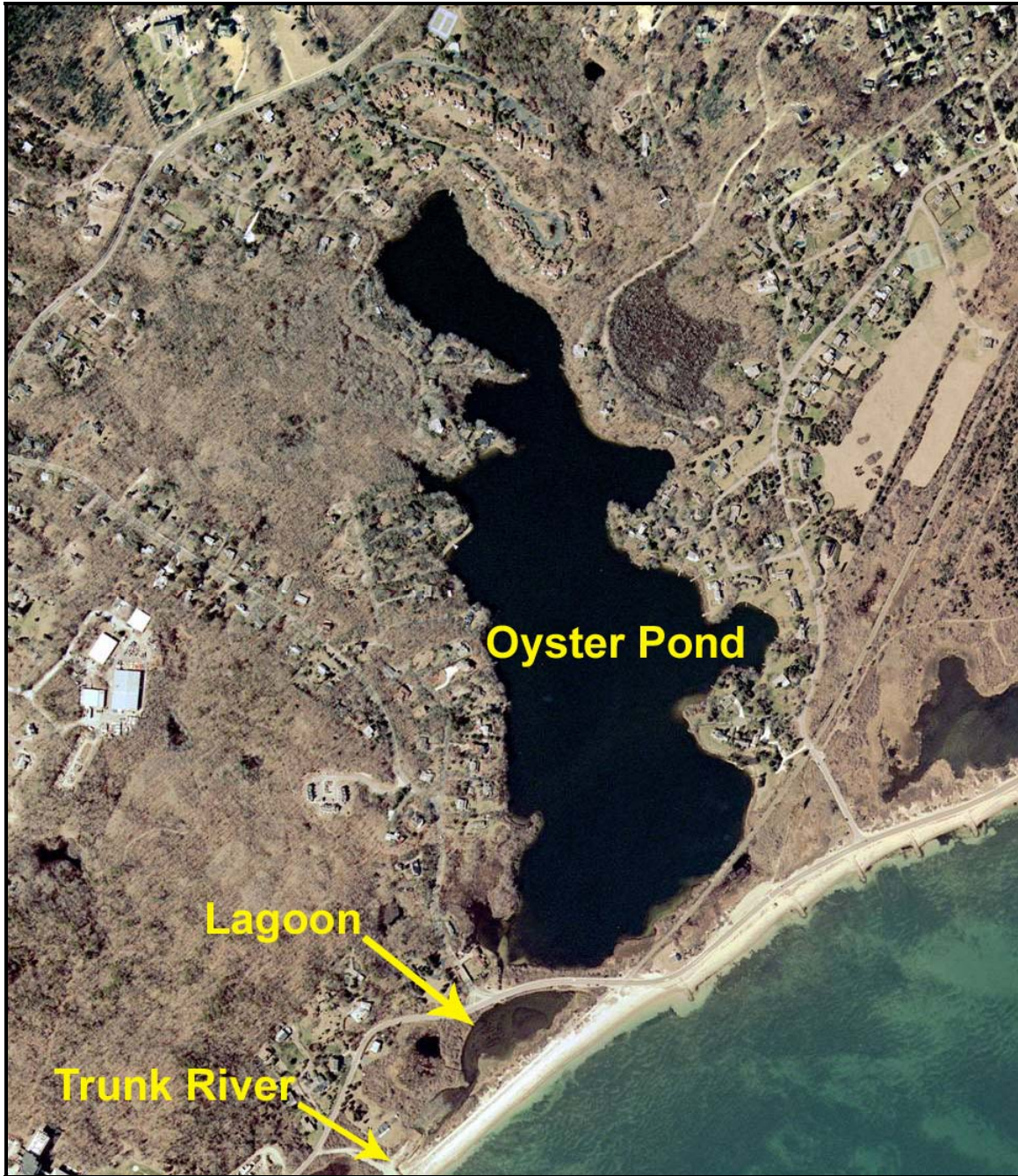


Figure I-1. Study region proximal to the Oyster Pond embayment system for the Massachusetts Estuaries Project nutrient analysis. Tidal waters periodically enter the salt pond through one inlet to Vineyard Sound. Freshwaters enter from the watershed primarily through 1 surface water discharge (stream to the head of Oyster Pond) and direct groundwater discharge.

The Falmouth PondWatch Program, as the water quality monitoring effort came to be known, continues to play an active role in the collection of baseline water quality data to this day though it has evolved beyond its original mandate of providing basic environmental data relative to the Coastal Pond Overlay Bylaw (Nutrient Bylaw). The Pond Watch Program brings together, as requested by Town boards, ecological information relative to specific water quality issues.

Additionally, as remediation plans for various systems are implemented, the continued monitoring satisfies demands by State regulatory agencies and provides quantitative information to the Town relative to the efficacy of remediation efforts. Lastly, the PondWatch Program has grown into being a repository of environmental data on Falmouth's coastal ponds. The database includes basic water quality monitoring data in addition to special project data on watershed nutrient loading and watershed delineation, circulation characteristics of the ponds, wetland delineations and plant and animal distributions.

Falmouth's Planning Office continues to enhance its tools for gauging future nutrient effects from changing land-uses. The GIS database used in the present evaluation is part of that continuing effort. Unfortunately, monitoring has documented that most regions within the Town's coastal ponds, including Oyster Pond, are currently showing water quality declines and are beyond the limits set by the By-law. Based on the wealth of information obtained over the many years of study of these coastal ponds, in addition to the nutrient analyses undertaken as a precursor to the Massachusetts Estuaries Project, the Oyster Pond embayment system was included in the first round prioritization of the Massachusetts Estuaries Project to provide state-of-the-art analysis and modeling. However, given that the MEP was able to fully integrate the Towns' on-going data collection and modeling effort, minimal additional municipal funds were required for MEP tasks.

The common focus of the Falmouth Pond Watch Program effort has been to gather site-specific data on the current nitrogen related water quality throughout Falmouth's coastal embayments, such as Little Pond, Great Pond, Green Pond, and Bournes Pond embayment systems, and determine the relationship between observed water quality and watershed nitrogen loads. This multi-year effort has provided the baseline information required for determining the link between upland loading, tidal flushing, and water quality in Oyster Pond. The Pond Watch Program in Oyster Pond developed a data set that elucidated the long-term trend of declining water quality and its relation to watershed based nutrient loading. The MEP effort builds upon the Falmouth water quality monitoring program, and previous hydrodynamic and water quality analyses, and includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Oyster Pond embayment system. The critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater master planning and nitrogen management alternatives development needed by the Town of Falmouth. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the SMAST/DEP Massachusetts Estuaries Project, the results stem directly from the efforts of a large number of Town staff and volunteers over many years. The modeling tools developed as part of this program provide the quantitative information necessary for the Town of Falmouth to develop and evaluate the most cost effective nitrogen management alternatives to restore these valuable coastal resource which are currently being degraded by nitrogen overloading.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

Coastal embayments throughout the Commonwealth of Massachusetts (and along the U.S. eastern seaboard) are becoming nutrient enriched. The nutrients are primarily related to changes in watershed land-use associated with increasing population within the coastal zone over the past half century. Many of Massachusetts' embayments have nutrient levels that are approaching or are currently over this assimilative capacity, which begins to cause declines in their ecological health. The result is the loss of fisheries habitat, eelgrass beds, and a general disruption of benthic communities. At its higher levels, enhanced loading from

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

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

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

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

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

In appropriate estuaries, TMDLs for bacterial contamination will also be conducted in concert with the nutrient effort (particularly if there is a 303d listing). However, the goal of the bacterial program is to provide information to guide targeted sampling for specific source identification and remediation. As part of the overall effort, the evaluation and modeling approach will be used to assess available options for meeting selected nitrogen goals, protective of embayment health.

The major Project goals are to:

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

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

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

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

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

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

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

Nitrogen Thresholds Analysis

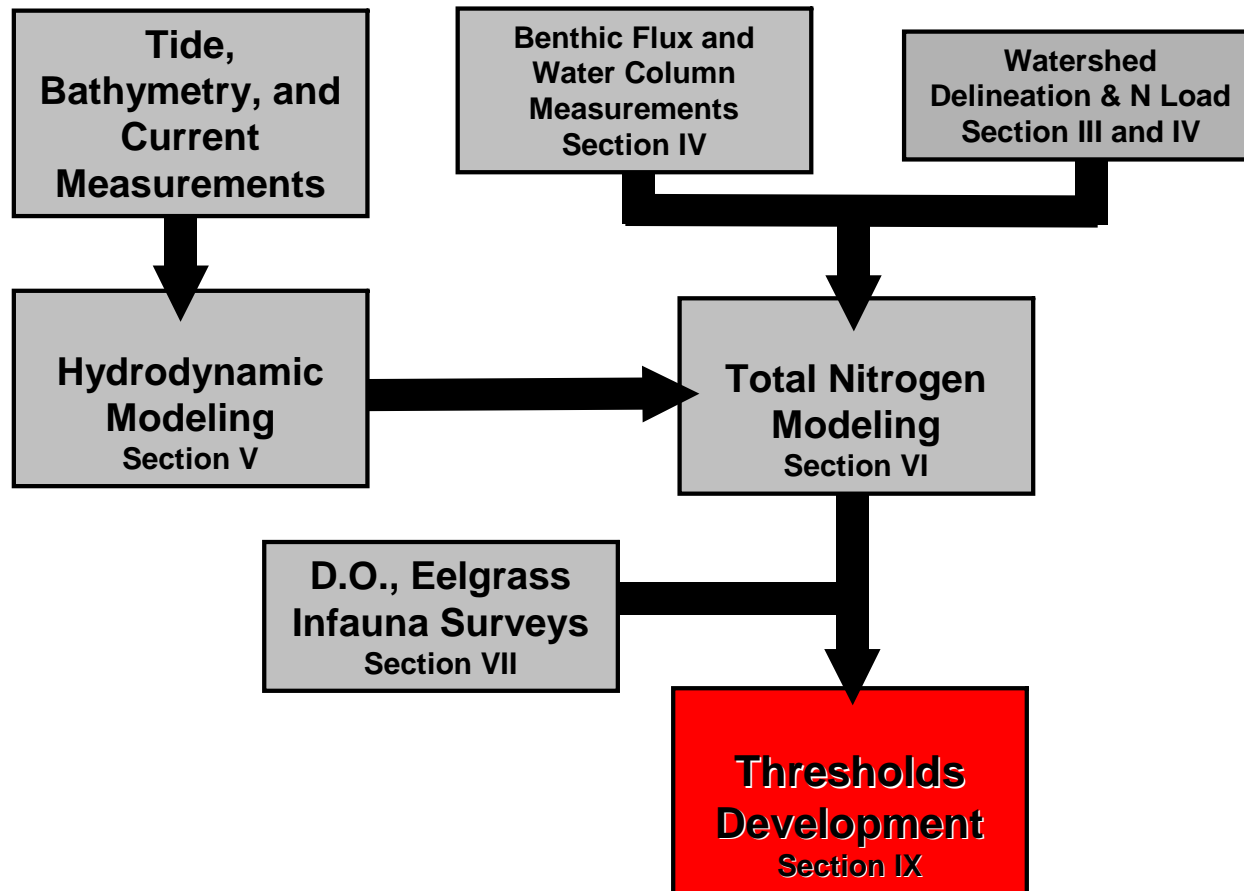


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach

I.2 SITE DESCRIPTION

The coastal salt ponds of Falmouth including Oyster Pond are generally oriented north-south, and open to varying degrees to Vineyard Sound via inlets. These inlets are affected significantly by longshore sand transport (west to east) as is the case with Oyster Pond, where shoaling can impede hydrodynamic exchange at each mouth. As depicted in Figure I-3 Oyster Pond is somewhat cut off from Vineyard Sound by a baymouth bar. An occluded inlet is present forming what is commonly known as the Trunk River that leads into a saline lagoon adjacent to Oyster Pond and connected to the pond system via a culvert under Oyster Pond Road.

Oyster Pond is approximately 1050 meters long and is oriented to the north-northwest at nearly right angles to the shore of Vineyard Sound. The maximum width of the pond is 400 meters but because the shores are irregular the average width is about half that figure. The total surface area of Oyster Pond is 63 acres with the surface areas of each of its three main basins (upper, middle, lower) being 14.46, 26.74, and 21.80 acres respectively. The embayment is separated from Vineyard Sound to the south, by a barrier beach, composed of sand and gravel. However, a railroad embankment and roadway (built along the north side of the bar and atop a marsh) constructed in 1872 gave the southern boundary its present form. Between the former railroad bed and the roadway is a small and shallow (0.5-meter deep) salt marsh lagoon, which receives tidal inflow from Vineyard Sound and discharge from Oyster Pond. Oyster Pond drains into the Lagoon through a culvert under the roadway (Oyster Pond Road) located in the southwestern corner of the pond. The marsh lagoon in turn connects with open Vineyard Sound through the Trunk River, which is fixed by jetties 200 meters southwest of the pond itself.

For the MEP analysis, the Oyster Pond system was analyzed individually as a stand alone system. The Oyster Pond estuarine system was partitioned into three general basin groups according to bathymetric contours: an 1) upper basin also considered the head of the estuary and 2) middle portion that averages three meters in depth and 3) a lower basin that reaches a maximum depth of slightly over 6 meters (see Figure I-3 and I-4). Unlike other salt ponds in Falmouth (e.g. Great/Perch Pond, Green Pond, Bournes Pond and Little Pond) Oyster Pond is a kettle pond that was likely both a bay and an estuary in the geologic past. Currently, it functions as a salt pond with controlled introduction of sufficient saltwater from Vineyard Sound to maintain a mixed layer salinity of 2-4 ppt. The head of Oyster Pond supports a very small yet focused freshwater input (Mosquito/Quivett Creek).

Oyster Pond is a shallow mesotrophic (moderately nutrient impacted) to eutrophic (nutrient-rich) coastal pond on the southern coast of Falmouth. The shores of the pond can be characterized as: 1) steeply sloping to a height of 5 to 10 meters above the pond with boulders at the waters edge (70 percent of perimeter), 2) slightly sloping to a height above the pond of less than 5 meters on the southeastern side of the pond with gravel and few boulders (12 percent of perimeter) and 3) marshes along the south and southwestern sides of the pond (18 percent of the perimeter). The steeply sloping shores of the pond are backed by glacial till consisting of large boulders embedded in gravel and coarse sand. The till belongs to the Buzzards Bay recessional moraine which extends southwesterly through Woods Hole and along the Elizabeth Islands (Shaler, 1897; Woodsworth and Wigglesworth, 1934; Mather, Goldthwait, and Theismeyer, 1942). The slightly sloping shores of the pond are generally backed by relatively flat outwash plain composed of sands and gravels deposited by glacial meltwater streams (Figure I-5).

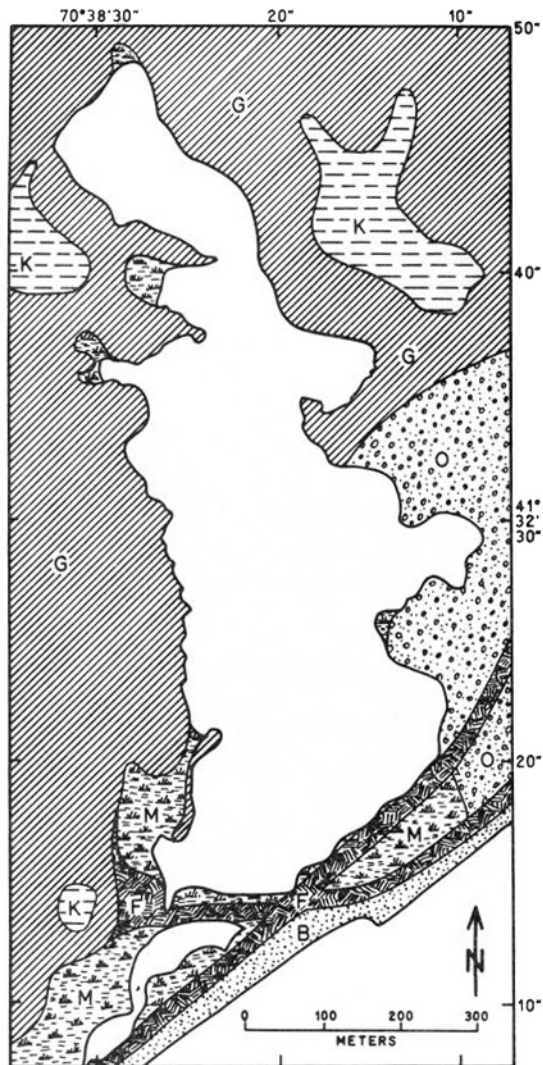
The north end of Oyster Pond is a kettle left by the melting of a residual mass of glacial ice within the Buzzards Bay moraine. The southward extension of the pond consists of two other kettles that mark the former positions of ice masses within the outwash plain. The total thickness of the Pleistocene deposits in the area of Oyster Pond is not well known but likely to be approximately 80 meters. A foundation test boring made during August 1965 for a pier at the Woods Hole Oceanographic Institution nearly 4 km southwest of Oyster Pond encountered granodiorite at 83 meters below mean low water. This was found to be overlain by 81 meters of clean sand, most likely glacial outwash, and two meters of seawater (Emery, 1997).



Figure I-3. Partitioning of the Oyster Pond embayment system relative to the three main basins, used for volume calculations.



Figure I-4. Oyster Pond embayment bathymetry (from K.O. Emery 1997).



**Symbols: G – glacial till, K – kettles, O – outwash plain, M – marsh
B – beach, F – artificial fill**

Figure I-5. Physiographic units of the land adjacent the Oyster Pond embayment system (from K.O. Emery 1997).

Although the salt pond embayment systems of Falmouth bounding Vineyard Sound exhibit slightly different hydrologic characteristics (river dominated versus tidally dominated), the tidal forcing for these systems is generated from Vineyard Sound. Vineyard Sound, adjacent the barrier beach separating the Oyster Pond embayment system from the ocean, exhibits a moderate to low tide range, with a mean range of about 0.5 m at the southern inlet of the salt marsh lagoon down-gradient of the pond. Tidal influence on Oyster Pond by Vineyard Sound is

further limited by a control structure that was implemented to manage salinity in Oyster Pond and the fact that the inlet to the salt marsh lagoon has gradually become occluded thereby further diminishing the flow of Vineyard Sound water into the system. Since the water elevation difference between Vineyard Sound and Oyster Pond is the primary driving force for tidal exchange (relative to the specifications of the control structure in Oyster Pond), the local tide range naturally limits the volume of water flushed during a tidal cycle (note the tide range off Stage Harbor Chatham is ~1.5 m, Wellfleet Harbor is ~3 m).

Tidal damping (reduction in tidal amplitude) through an embayment can be negligible indicating “well-flushed” conditions or show tidal attenuation caused by constricted channels and marsh plains indicating a “restrictive” system, where tidal flow and the associated flushing are inhibited. Oyster Pond differs from most embayments in that it has been intentionally tidally restricted as part of a management plan to provide ecological stability and support an anadromous fish run (see Section V.1). The result of the tidal control structure in the channel between the Pond and the Lagoon is that “tidal” inflows are present during only the highest tides and associated storm tides. Given that maintenance of the resources of Oyster Pond requires a salinity in the range of 2-4 ppt, which is based upon the present hydrodynamic characteristics, it appears that improvements in estuarine habitat quality is currently dependent on nutrient loading. The current watershed nitrogen loads and effects on pond resources are the focus of the present MEP analysis described in this report.

Nitrogen loading to the Oyster Pond embayment system was determined relative to the upper, middle and lower portions of the salt pond as depicted in Figure I-3. Based upon the watershed and Pond being fully within the Town of Falmouth, it appears that nitrogen management for Pond restoration depends upon the efforts by Town Departments and citizens.

As management alternatives are being developed and evaluated, it is important to note that Oyster Pond is brackish and only very weakly tidal. Its basin configuration results in pulses of salt water entering the deep basin (~6m) during extreme storm events. As a result the deep basins stratify and become anoxic below 4 meters depth. This physical characteristic of the pond and resultant anoxic bottom waters and sediments in the affected areas, was more prevalent when pond salinities were higher (10-16 ppt) and extended over pond regions covered by waters 3 meters in depth (compared to 4 meters in depth currently). Since it is not possible to prevent the entry of salt water during hurricanes, when the barrier beach is overwashed, it is not realistic to attempt to convert the pond to fully freshwater. Similarly, it is not possible to attempt to target these deep basins for “restoration”, as they naturally become anoxic and evidence suggests that they were periodically anoxic over the past ~100 years. This includes the period prior to 1960, when there were fewer than 50 residences within the watershed. However, the pond waters and sediments at <4 meters depth account for most of the Pond resources and they are currently showing nutrient overloading and meso- to eutrophic conditions (see Section VII).

I.3 NITROGEN LOADING

Surface and groundwater flows are pathways for the transfer of land-sourced nutrients to coastal waters. Fluxes of primary ecosystem structuring nutrients, nitrogen and phosphorus, differ significantly as a result of their hydrologic transport pathway (i.e. streams versus groundwater). In sandy glacial outwash aquifers, such as in the watershed to the Oyster Pond embayment system, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer mineral (Weiskel and Howes 1992). Since even Cape Cod “rivers” are primarily groundwater fed, watersheds tend to release little phosphorus to coastal waters. In

contrast, nitrogen, primarily as plant available nitrate, is readily transported through oxygenated groundwater systems on Cape Cod (DeSimone and Howes 1998, Weiskel and Howes 1992, Smith *et al.* 1991). The result is that terrestrial inputs to coastal waters tend to be higher in plant available nitrogen than phosphorus (relative to plant growth requirements). However, coastal estuaries tend to have algal growth limited by nitrogen availability, due to their flooding with low nitrogen coastal waters (Ryther and Dunstan 1971). Tidally influenced areas within Oyster Pond follow this general pattern, where the primary nutrient of eutrophication in the system is nitrogen, however, as Oyster Pond is generally more dominated by freshwater rather than saltwater (particularly in the upper regions of the pond), phosphorous has been investigated. A recent study using a bio-assay for nutrient limitation in Pond phytoplankton, demonstrated that algae within Oyster Pond are presently nitrogen limited. Incubations only yielded growth under nitrogen additions and not under phosphorus addition (Barron et al. 2002). As a result of this study and supporting PondWatch reports, N/P ratios also indicated that Oyster Pond is nitrogen limited, phosphorus was not a focus nutrient for the MEP analysis.

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

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

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

Unfortunately, almost all of the Oyster Pond system is beyond its ability to assimilate additional nutrients without impacting ecological health. Nitrogen and phosphorous levels are elevated throughout the system and infaunal animal communities are diminished and lack

diversity in many areas. It should be noted that eelgrass was structurally eliminated from the system when the decision was made to maintain the salinity regime of the pond at between 2 and 4 ppt. The result is that nitrogen management of the Oyster Pond system is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed “eutrophication” and when the nutrient loading is primarily from human activities, “cultural eutrophication”. Although the influence of human-induced changes has increased nitrogen loading to the system and contributed to the degradation in ecological health, it is also possible that some of the eutrophication within Oyster Pond may occur without man’s influence and must be considered in the nutrient threshold analysis. While this finding would not change the need for restoration, it would change the approach and potential targets for management. As part of future restoration efforts, it is important to understand that it may not be possible to turn each embayment into a “pristine” system.

I.4 WATER QUALITY MODELING

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

The MEP water quality evaluation examined the potential impacts of nitrogen loading into Oyster Pond. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for the system. Once the hydrodynamic properties of the estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates within the mixed layer of the pond. Exchanges between the deep basin hypolimnetic waters and the mixed layer were evaluated by an examination of rates of exchange via eddy diffusion.

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape model for sub-watershed areas designated by MEP. Almost all nitrogen entering Falmouth’s salt ponds is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Vineyard Sound source waters and throughout the Oyster Pond system was taken from the Falmouth PondWatcher Monitoring Program (supported by the Town of Falmouth and associated with the Coastal Systems Program at SMAST). Measurements of current salinity and nitrogen and salinity distributions throughout the system were used to calibrate and validate the water quality model (under existing loading conditions).

I.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Oyster Pond system for the Town of Falmouth. A review of existing water quality studies is provided (Section II). The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Sections III and IV. In addition, nitrogen input parameters to the water quality model are described. Since benthic flux of nitrogen from bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Section IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Town of Falmouth Planning Department supplied data and water-use data. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling in Section VI. Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data from within the embayment and offshore waters (conducted by municipalities). These data and its relevance to the water quality modeling of present conditions are discussed in Section VI. Predictions based upon the water quality model to evaluate nitrogen levels at watershed build-out, and with removal of anthropogenic nitrogen sources are also presented in Section VI. In addition, an ecological assessment of the embayment was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of each embayment in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined threshold for restoration of this salt pond. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for the Oyster Pond system.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

In most marine and estuarine systems, such as Great Pond, Green Pond, Bournes Pond and Little Pond in Falmouth, the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if nitrogen addition is controlled, then eutrophication is controlled. This approach has been formalized through the development of tools for predicting nitrogen loads from watersheds and the concentrations of water column nitrogen that may result. Additional development of the eutrophication management approach via the reduction of nitrogen loads 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. 2003). The Oyster Pond system is structurally different than the more typical Falmouth salt pond estuarine systems mentioned above and has also been managed as a brackish water pond with highly restricted tidal exchange. As such it is necessary for the purpose of the MEP to adjust the eutrophication management approach in order to capture the unique characteristics of the Oyster Pond system

Until recently, the tools for predicting loads and concentrations tended to be generic in nature, and overlooked 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 Oyster Pond System.

The Town of Falmouth, Massachusetts, has long recognized the potential threat of nutrient over-enrichment of its coastal salt ponds and embayments. In the mid-1980's the Town enacted an innovative Nutrient Overlay By-law that tied watershed development to water quality within the adjacent embayment. The goal was to keep nitrogen concentrations in the receiving systems below thresholds that were projected to cause water quality shifts. A water quality monitoring program, Falmouth PondWatch, was established to provide on-going nutrient related embayment health information in support of the By-law. Oyster Pond was among the first three Ponds (Oyster Pond, Little Pond, Green Pond) to undergo water quality monitoring in the Town of Falmouth. These approaches were primarily initiated for planning as development within coastal watersheds progressed. Falmouth's Planning Department has continued to enhance its tools for gauging future nutrient effects from changing land-uses. The GIS database used in the present study is part of that continuing effort. Unfortunately, monitoring has documented that most regions within the Town's coastal ponds are currently showing water quality declines and are beyond the limits set by the By-law.

Given the brackish waters of Oyster Pond and the management need for targeting the nutrient which limits algal production, the MEP sought site-specific evidence of either nitrogen or phosphorus limitation in this system. Two lines of evidence were found, both of which indicate that Oyster Pond, like the other embayments of Cape Cod is nitrogen limited. A study conducted in June and July 2002, added nitrogen or phosphorus to surface water samples from Oyster Pond and followed chlorophyll *a* levels (as an indicator of phytoplankton growth) over 4 days. In the two experiments conducted, only the nitrogen enriched waters showed stimulation of phytoplankton (Weber et al. 2002). The second line of evidence is based upon the elemental ratio method (Redfield Ratio), where molar ratios of inorganic N/P less than 16 suggest that as nutrient uptake occurs to support algal growth (phytoplankton and macro-algae) nitrogen will

become depleted before phosphorus. In other words, addition of phosphorus would not stimulate algal growth, but nitrogen would. The Falmouth PondWatch Program has found that during summer samplings, 2000-2004, the geometric mean ratio of inorganic nitrogen to inorganic phosphorus (N/P) in samples from 0-3.25 m was 10 (s.d. 4, N=102). During this period the salinity of the surface waters averaged 2.1 ppt, range 1.6-2.8 ppt (Howes and Goehring unpublished data). A different study found a similarly low N/P ratio (~5) in September 2002 (Dixon et al. 2002). While this is only an approximate method, it is consistent with other studies concluding nitrogen limitation in estuaries throughout the region. These studies support the targeting of nitrogen as the key nutrient for management of the habitat quality of this estuarine system.

Data generated by the Falmouth PondWatch Program has also yielded clear indications of nutrient related impairment to the Oyster Pond system and has assisted the Town in the development of initial management options for improving the ecological health of the system. Specifically, the PondWatch Program water quality database for Oyster Pond assisted the Town of Falmouth in the design and permitting of the salinity control/fish ladder which separates Oyster Pond from the down gradient salt marsh lagoon. This control structure is what currently maintains the surface mixed waters between 2 – 4 ppt. The concept of salinity control as phase I of the management of the Oyster Pond System was to achieve a variety of resource management goals. The phase I management plan was devised by PondWatch working closely with the Oyster Pond Environmental Trust (OPET) and the Town of Falmouth (planning, engineering, conservation, natural resources, Herring Warden) and the Town's consulting engineers at Applied Coastal Research and Engineering. The management goals were to create a "stable" salinity environment for Oyster Pond and to improve the oxygen status of the bottom waters, which were typically hypoxic-anoxic below 3 meters during summer (Howes and Hart 1997). The option to increase the tidal flushing to create a high salinity environment was rejected due to (a) the difficulty in maintaining an open inlet, (b) the need for additional hard-structures to fix a new tidal inlet to Vineyard Sound, (c) the salinity history of the Pond which indicated that for most of the 1900's the surface waters were <5 ppt, and (d) the cost of creating and maintaining a 100's of meters long tidal channel from the Sound. Creating a freshwater pond by closing off of the tides was rejected due to the periodic overwash of salt waters during major storms (e.g. Hurricane Bob 1991). The phase I plan was also consistent with the uses of the Pond by local residents as determined from surveys. The phase I plan was implemented, initially by allowing sedimentation to restrict tidal flow in the Trunk River and between the Trunk River and Oyster Pond. By 1996 Oyster Pond salinity levels had dropped below 4 ppt and by 2000 the salinity levels were stabilized at ~2 ppt. The phase I results have been to restore the salinity of the pond to the historic levels, to maintain relatively stable salinity conditions, to restore the fish community, including the herring run, and to improve the oxygen conditions. The design and role in tidal hydrodynamics of the salinity control/fish weir are presented in detail in Section 5 and the salinity history, salinity levels and implications for Oyster Pond management are presented in Section 7, below. At present, phase I is nearly completed, needing only a channel maintenance program to keep the short channel between the Trunk River and the salt marsh Lagoon open. Phase II was slated to focus on watershed nitrogen management, which is also the purpose of the SMAST/DEP Massachusetts Estuaries Project Linked Watershed-Embayment Approach.

Oyster Pond has been the focus of a variety of scientific studies of which the most notable was by K.O. Emery, who provided a full analysis of the geomorphology, geology, chemistry and biology of the Pond in the early 1960's. This study was updated (and republished) in 1997 to include a nitrogen balance and to outline the anthropogenic alterations which affect the Pond ecosystem and set forward the path for Pond management and restoration (Howes and Hart

1997). More recently, Oyster Pond has been used by Boston University students for class projects (2001 and 2002). These data have been evaluated by the MEP Technical Team. Unfortunately, use of results from the series of studies based on watershed nitrogen loading rates is problematic, as the delineated watershed is very different from the present MEP watershed and earlier watersheds derived for this system (see Section 3). Similarly, the series of studies showing shallow (0.25-1.5 m below land surface) “groundwater” nitrogen concentrations collected 0-2 m from the pond edge could not be validated, as groundwater was defined as having a salinity of <2ppt and the pondwater itself in the upper basin (0.15m) was 1.7-2.3 in 2001-2002. However, some of the results of these studies have been incorporated as supporting information into the present MEP analysis, specifically as relates to nitrogen as the critical nutrient for management (as described above) and mapping of the various aquatic resources.

For the MEP modeling analysis, the data from the previous studies were evaluated relative to the needs of the Linked Watershed-Embayment Model. The PondWatch watercolumn nutrient data was deemed acceptable as these data met MEP protocols and the assays were conducted by the Coastal Systems Analytical Facility at SMAST, whose protocols have been reviewed by DEP and EPA. In addition, the prior hydrodynamic analysis was also included as it met with MEP methods and quality assurance procedures. In addition, K.O. Emery’s study was a valuable source of information related to the geology, geomorphology and history of Oyster Pond.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

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

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watershed or contributing area to the Oyster Pond system under evaluation by the Project Team. The Oyster Pond estuarine system is composed of a simple estuary, originating from sea level flooding of a coastal kettle basin. The system is naturally tidally restricted due to the formation of a barrier beach to Vineyard Sound. In 1872 the construction of a railroad on the barrier beach complex further restricted the tidal exchange. The present tidal channel and restricted tidal exchange and approximate water levels have likely been in place for ~100 years, with only periodic short-term disruptions. Watershed modeling was undertaken to sub-divide the overall watershed to the Oyster Pond system into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system, (b) defining contributing areas to major freshwater aquatic systems which generally attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining 10 year time-of-travel distributions within each sub-watershed as a procedural check to gauge the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in the Sagamore flow cell on Cape Cod. Model assumptions for calibration were matched to surface water inputs and flows from current stream gage information (2002-2003).

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

III.2 MODEL DESCRIPTION

Contributing areas to the Oyster Pond system and local freshwater bodies were delineated using a regional model of the Sagamore Lens (Walter and Whealan, in press). The

USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, et al., 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to the main basins of the Oyster Pond system and also to determine portions of recharged water that may flow through freshwater ponds and streams prior to discharging into coastal water bodies.

The Sagamore Flow Model grid consists of 246 rows, 365 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below sea level and have a uniform thickness of 10 ft. The top of layer 8 resides at sea level with layers 1-7 stacked above sea level to a maximum elevation of +70 feet. In the portion of the Sagamore Lens in which the Oyster Pond system resides, water elevations are generally less than +40 ft and, therefore, over much of the study area the uppermost layers are inactive. Layer 18 has a thickness of 40 feet and layer 19 extends to 240 feet below sea level. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics.

The glacial sediments that comprise the aquifer of the Sagamore flow cell consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward with sand and gravel deposits deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. Oyster Pond is situated on the boundary between the Buzzards Bay Moraine and very-coarse grained Mashpee Pitted Plain deposits (Masterson, et al., 1996). In fact the upper basins of Oyster Pond and watershed are situated in glacial till and only the lower deep basin and adjacent watershed are in outwash (Emery 1997). Lithologic data used to determine hydraulic conductivities used in the groundwater model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and water level and streamflow data collected in May 2002.

The regional model simulates steady state, or long-term average, hydrologic conditions including a long-term average recharge rate of 27.25 inches/year and the pumping of public-supply wells at average annual withdrawal rates for the period 1995-2000 with a 15% consumptive loss. This recharge rate is based on the most recent USGS information. Large withdrawals of groundwater from pumping wells may have a significant influence on water tables and watershed boundaries and therefore the flow and distribution of nitrogen within the aquifer. Since all of the watersheds to Oyster Pond are unsewered, 85% of the water pumped from wells was modeled as being returned to the ground via on-site septic systems.

III.3 OYSTER POND CONTRIBUTORY AREAS

Revised watershed and sub-watershed boundaries were determined by the United States Geological Survey (USGS) for the Oyster Pond embayment system (Figure III-1). Model outputs of MEP watershed boundaries were “smoothed” to (a) account for the grid spacing, (b) to enhance the accuracy of the characterization of the shoreline, and (c) to more closely match

the sub-embayment segmentation of the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. Overall, five sub-watershed areas were delineated within the watershed to the Oyster Pond embayment system.

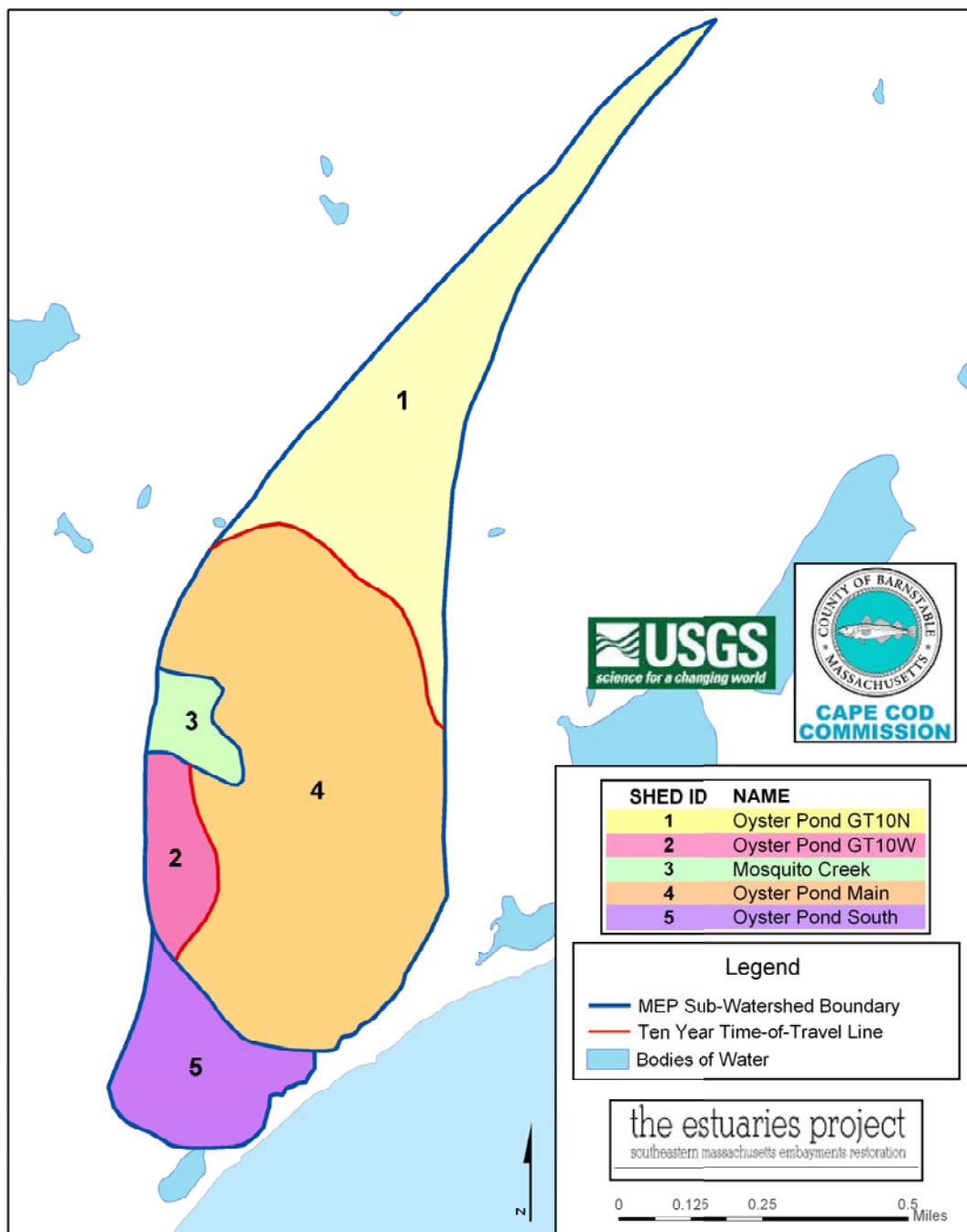


Figure III-1. Watershed and sub-watershed delineations for the Oyster Pond estuary system. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a “10” in the watershed names (above). Sub-watersheds to embayments were selected based upon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

Table III-1 provides the daily discharge volumes for various watersheds as calculated by the groundwater model; these volumes were used to assist in the salinity calibration of the tidal hydrodynamic models and for comparison to measured surface water discharges. The MEP delineation includes 10 yr time of travel boundaries. The overall estimated groundwater flow into Oyster Pond from the MEP watershed is 2,587 m³/d. This flow compares favorably with the 2,600 m³/d average calculated from estimates by K.O. Emery (1969) from a variety of measurement methods. Note that the values re-calculated from Emery, are based upon Pond level changes and freshwater discharge to Vineyard Sound. These results provide an independent validation of the regional model, which is founded on watershed hydrologic information.

Table III-1. Daily groundwater discharge to each of the sub-embayments in the Oyster Pond system, as determined from the USGS groundwater model.		
Watershed	Discharge	
	ft ³ /day	m ³ /day
Oyster Pond GT10N	31,594	895
Oyster Pond GT10W	5,794	164
Mosquito Creek_Oyster Pond	3,571	101
Oyster Pond_Main	39,283	1,113
Oyster Pond_South	11,100	314
Whole System	91,343	2,587

The delineations completed for the MEP project are the third watershed delineation completed in recent years for the Oyster Pond estuary. Figure III-2 compares the delineation completed under the current effort with the delineation completed by the Cape Cod Commission in 1995 (Eichner, et al., 1998). The delineation completed in 1995 was defined based on regional water table measurements collected over a number of years and normalized to average conditions; delineations based on this effort were incorporated into the Commission's regulations through the Regional Policy Plan (CCC, 1996 & 2001). The third watershed delineation was based upon the 1995 Commission effort with additional groundwater elevation information and topographic interpretation (Howes and Hart 1997). As this latter watershed was not a regulatory effort and was in between the MEP and Commission delineations, it will not be discussed further here.

The MEP watershed area for the Oyster Pond as a whole is 22% larger (74 acres) than the 1995 Cape Cod Commission (CCC) delineation. Most of this change is attributable to the larger northern section of the watershed in the MEP watershed. The expansion of this area is due to the USGS model assigning a more northern location to the groundwater divide between Vineyard Sound and Buzzards Bay. The more northern location for the divide also results in a slight expansion to the west for the MEP watershed as compared to the CCC watershed (see Figure III-2). It should also be noted that the CCC watershed includes an area surrounding a small pond to the southwest of the Oyster Pond estuary; this area is not included in the MEP watershed. Sub-watersheds were not delineated in the 1995 CCC watershed.

The evolution of the watershed delineations for the Oyster Pond System have built one on another as the underlying hydrologic data supporting the modeling has increased, thereby yielding higher accuracy. This is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model used for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are

included/excluded within the contributing areas. Small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the down-gradient estuary. The present watershed delineation to Oyster Pond appears to be relatively robust given its general similarity to the previous 2 delineations and the good agreement with the estimates of freshwater exiting to Vineyard Sound through the salinity control/fish weir at the south end of the Pond.

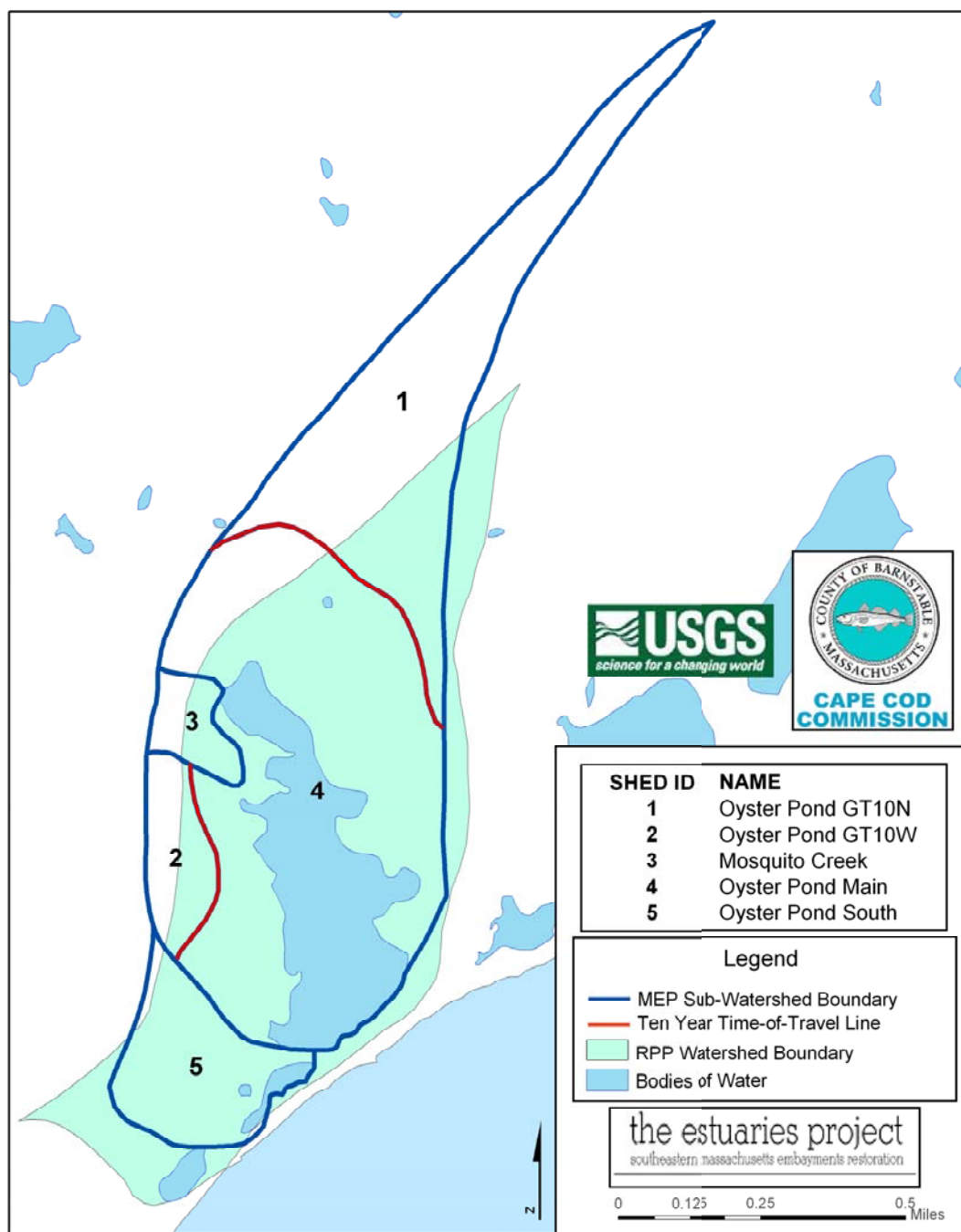


Figure III-2. Comparison of previous and current Oyster Pond watershed and subwatershed delineations.

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

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 Oyster Pond system. Determination of watershed nitrogen inputs to these embayment systems requires the (a) identification and quantification of the nutrient sources and their loading rates to the land or aquifer (Section IV.1), (b) confirmation that a groundwater transported load has reached the embayment at the time of analysis (Section IV.1), and (c) quantification of nitrogen attenuation that can occur during travel through lakes, ponds, streams and marshes (Section IV.2). 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 (Section IV.3) results primarily from the settling and decay of phytoplankton and macroalgae (and eelgrass when present). During decay, organic nitrogen is transformed to inorganic forms, which may be released to the overlying waters or lost to denitrification within the sediments. Burial of nitrogen is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters and leads to errors in predicting water quality if not included in determination of summertime nitrogen load.

IV.1 WATERSHED LAND USE BASED NITROGEN LOADING ANALYSIS

The MEP Technical Team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP technical team staff, CCC staff developed nitrogen loading rates (Section IV.1) within the watershed to the Oyster Pond embayment system (Section III). The watershed was sub-divided to define a contributing area to Mosquito Creek and further sub-divided into regions greater and less than 10 year groundwater travel time from the receiving estuary, a total of 5 sub-watersheds in all. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to the embayment.

The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges to the watershed have reached the embayment. This involves a temporal review of land use changes and the time of groundwater travel provided by the USGS watershed model. After reviewing the percentage of nitrogen loading in the less than 10 year time of travel (LT10) and greater than 10 year time of travel (GT10) watersheds (Table IV-1), reviewing Falmouth land use development records and 2001 land use in the time of travel watersheds, as well as reviewing water quality modeling, it was determined that Oyster Pond is currently in balance with its watershed load. The bulk (66%) of the watershed nitrogen load is within 10 years flow to Oyster Pond. In addition, the average year built of residential properties in the greater than 10 year watersheds is 1970 or 35 years ago; this suggests that the nitrogen from the majority of development in the greater than 10 year watersheds should currently be observed in Oyster Pond. Therefore, the distinction of less than 10 year and greater than 10 year time of travel regions within a subwatershed (Figure III-1) was eliminated and the number of subwatersheds was reduced to three. The overall result of the timing of development relative to groundwater

travel times is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuaries (after accounting for natural attenuation, see below).

Table IV-1. Percentage of unattenuated nitrogen loads in less than 10 year time of travel subwatersheds to Oyster Pond. Note that Oyster Pond South is to the salt marsh Lagoon which down-gradient from Oyster Pond proper.				
	LT10	GT10	TOTAL	%LT10
WATERSHED	kg/yr	kg/yr	kg/yr	
Oyster Pond Proper	973	522	1495	65%
Oyster Pond South	115	0	115	100%
Mosquito Creek	22	0	22	100%
OYSTER POND TOTAL	1110	522	1632	68%

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

Natural attenuation of nitrogen during transport from land-to-sea (Section IV.2) was determined based upon a site-specific study within the freshwater portion of Mosquito Creek. Internal nitrogen recycling was also determined throughout the tidal reaches of the Oyster Pond embayment; measurements were made to determine the rate of sediment nitrogen regeneration from the sediments to the overlying watercolumn. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

IV.1.1 Land Use and Database Preparation

Estuaries Project staff obtained digital parcel and tax assessors data from the Town of Falmouth. Digital parcels and land use data are from 2001 and 2003, respectively, and were obtained from the Town of Falmouth Planning Department. The land use database contains traditional information regarding land use classifications (MADOR, 2002) plus additional information developed by the Town about impervious surfaces (building area, driveways, and parking area) on individual lots. The parcel coverages and assessors database were combined for MEP analysis by using the Cape Cod Commission Geographic Information System (GIS).

Figure IV-1 shows the land uses within the study area. Land use in the study area is one of three land use types: 1) residential, 2) undeveloped, or 3) public service/government. Massachusetts Assessors land uses classifications (MADOR, 2002) in the watershed are aggregated into these three land use categories.

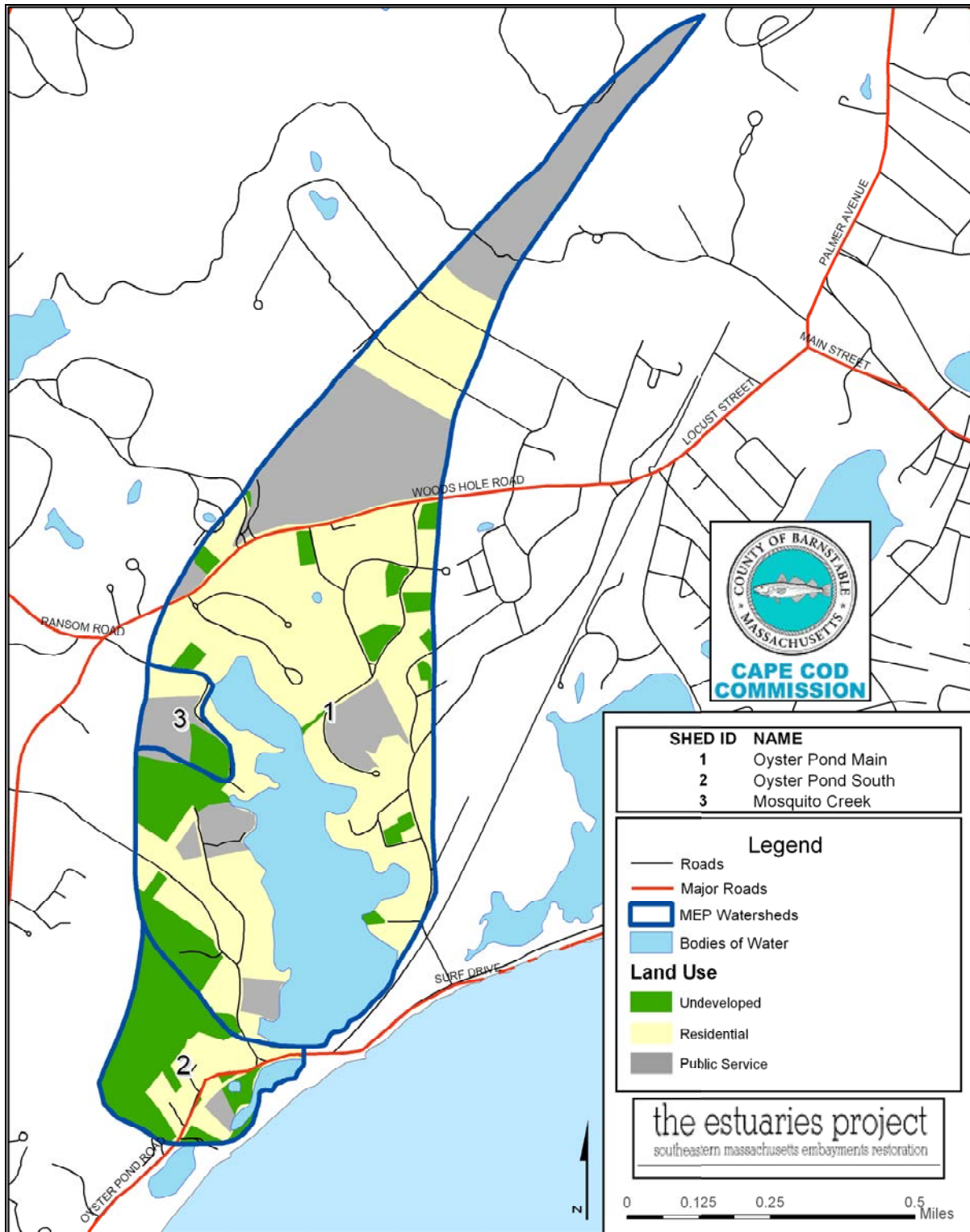


Figure IV-1. Land-use coverage in the Oyster Pond watershed. Watershed data encompasses portions of the Town of Falmouth and land use classifications are based on Town of Falmouth Planning Department records.

In the Oyster Pond watershed, the predominant land use based on area is residential, accounting for more than half (56%) of the watershed area. In addition, 75% of the residential land uses are single family residences (land use code = 101). In contrast, the small Oyster Pond South watershed (to the salt marsh Lagoon) is mostly undeveloped, 68% of land area (Figure IV-2). However, half (52% or 13 acres) of the undeveloped land in Oyster Pond South is classified as residentially developable. There are no developed commercial properties in the Oyster Pond watershed.

In order to estimate wastewater flows within the study area, MEP staff also obtained parcel by parcel water use information from the Town of Falmouth Water Department. This information included two years of water use information with the final reading in May 2003. Water use information was linked to the parcel and assessors data using GIS techniques. Water use for each parcel was converted to an annual volume for purposes of the nitrogen loading calculations. No wastewater treatment facilities (WWTFs) currently exist in the watershed and none of the properties in the watershed are connected to the existing municipal wastewater treatment facility.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

Similar to many other watershed nitrogen loading analyses, 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 directly measured septic system and per capita loads determined 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 in many of the watersheds throughout southeastern Massachusetts, census data yields accurate estimates of total population only in specific watersheds (see below). To correct for this uncertainty, the MEP employs a water-use approach. The water-use approach (Weiskel and Howes 1992) is applied on a parcel-by-parcel basis within a watershed, where usually an average of multiple years annual water meter data is linked to assessors parcel information using GIS techniques. The parcel specific water use data is converted to septic system nitrogen discharges (to the receiving aquatic systems) by adjusting for consumptive use (e.g. irrigation) and applying a wastewater nitrogen concentration. The water use approach focuses on the nitrogen load, which reaches the aquatic receptors down-gradient in the aquifer. All losses within the septic system are incorporated. For example, information developed at the DEP 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). Aquifer studies indicate that further nitrogen loss during aquifer transport is negligible (Robertson et al. 1991, DeSimone and Howes 1996).

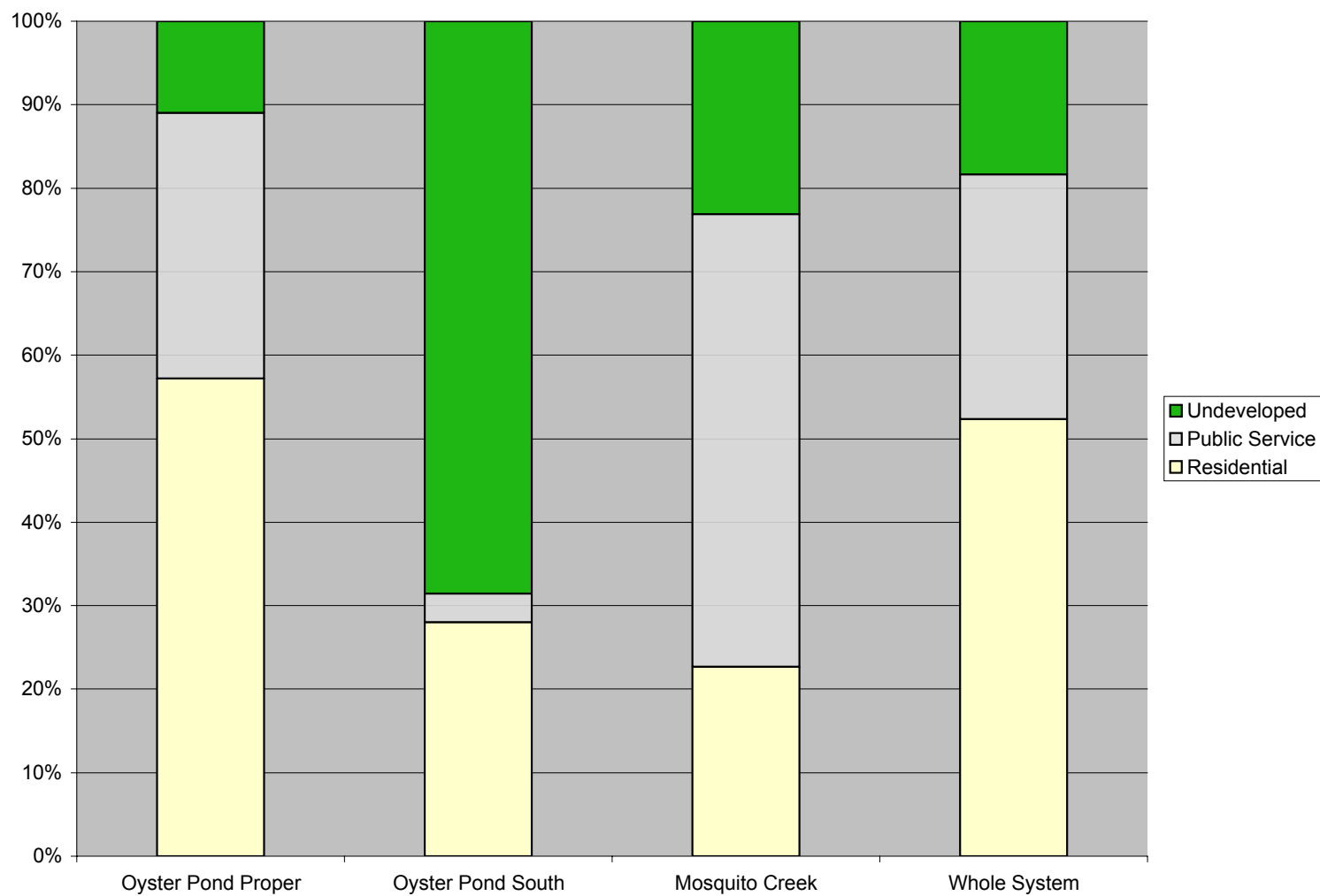


Figure IV-2. Distribution of land-uses within the major subwatersheds and whole watershed to Oyster Pond.

In its application of the water-use approach to septic system nitrogen loads, the MEP has ascertained for the Estuaries Project region that while the per capita septic load is well constrained by direct studies, the consumptive use and nitrogen concentration data are less certain. As a result, the MEP has derived a combined term the effective N Loading Coefficient (consumptive use times N concentration) of 23.63, to convert water (per cubic meter) to nitrogen load (N grams). This term 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, etc.).

The resulting nitrogen loads, based upon the above 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. For example, 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). The selected “effective N loading coefficient” also agrees with available watershed nitrogen loading analyses conducted on other Cape Cod estuaries. Aside from the concurrence observed between modeled and observed nitrogen concentrations in the estuary analyses completed under the MEP, analyses of other estuaries completed using this effective septic system nitrogen loading coefficient, the modeled loads also match observed concentrations in streams in the MEP region. Modeled and measured nitrogen loads were determined for a small sub-watershed to West Falmouth Harbor (Smith and Howes 2006) where a small stream drained the aquifer from a residential neighborhood. In this effort, the measured nitrogen discharge from the aquifer was within 5% of the modeled N load. A second 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 measured and observed loads showed a difference of less than 8%, easily attributable to the low rate of attenuation expected at that time of year and under the ecological situation (Samimy and Howes unpublished data).

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

Overall, the MEP water use approach for determining septic system nitrogen loads has been both calibrated and validated in a variety of watershed settings. The approach: (a) is consistent with a suite of studies on per capita nitrogen loads from septic systems in sandy outwash aquifers; (b) has been validated in studies of the MEP Watershed “Module”, where there is been excellent agreement between the nitrogen load predicted and that observed in direct field measurements corrected to other MEP Coefficients for stormwater, lawn fertilization,

etc; (c) the MEP septic nitrogen loading coefficient agrees in specific studies of consumptive water use and N 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 worked out the septic system nitrogen load over many years, and the general agreement among the number of supporting studies has greatly enhanced the certainty of this critical watershed nitrogen loading term.

The independent validation of the water quality model (Section VI) and the reasonableness of the freshwater attenuation (Section IV.2) add additional weight to the nitrogen loading coefficients used in the MEP analyses and a variety of other MEP embayments. While the MEP septic system nitrogen load is the best estimate possible, to the extent that it may underestimate the nitrogen load from this source reaching receiving waters, the MEP approach provides a safety factor relative to other higher loads that are generally used in regulatory situations to add a safety factor for the protection of impacted resources. 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 directly proportional to the septic system nitrogen level, but rather is related to how the nitrogen load from septic systems compares to the nitrogen loads from all other sources that reach the estuary (i.e. attenuated loads).

In order to provide an independent validation of the residential water use average within the study area, MEP staff reviewed US Census population values. The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so each person generates 55 gpd of wastewater. Average occupancy within the Town of Falmouth during the 2000 US Census was 2.36 people per household. If 2.36 is multiplied by 55 gpd, 130 gpd would be calculated as the average residential wastewater flow in Falmouth. In the Oyster Pond watershed, however, the average residential flow is 209 gpd,

In previously completed MEP studies, average population and average water use have generally agreed fairly well. Since they did not in this watershed, MEP staff reviewed more refined US Census information and water use information for each parcel within the watershed. Besides reviewing data on town and state levels, the US Census also develops information for smaller areas (*i.e.*, tracts and blocks). MEP staff review of this information in the watershed area did not show significantly different average population from the town average. MEP staff then reviewed individual water uses for each residential parcel. This review found that although the range of residential flows (minimum and maximum) is smaller in this watershed than for the whole town, there is a greater concentration of high water uses than for the rest of Falmouth. Since no information was available to suggest that the water uses in this watershed are inappropriate, MEP staff decided to continue to use the watershed-specific average water use for the four developed parcels without water use and for the 23 additional parcels included in the buildout analysis.

While almost all of the 165 developed parcels within the Oyster Pond watershed have corresponding water use accounts six (4% of all parcels) did not. Four of the six parcels are residential and two are public service/government; all six are assumed to utilize private wells for drinking water. In order to complete the nitrogen loading, the average water use from parcels with water use accounts (Table IV-2) was applied to the parcels assumed to be on private wells.

Average water use was also used for determining nitrogen loads from new development determined in the buildout analyses.

Table IV-2. Average Water Use in Oyster Pond Watershed.					
Land Use	State Class Codes	# of Parcels in Study Area	Water Use (gallons per day)		
			Study Area Avg.	Town Avg.	Town Range
Residential	101	149	209	153	0 to 79,618
Commercial	300 to 389	0	0	911	0 to 29,115
Industrial	400 to 439	0	0	1,229	0 to 13,583
Note: No commercial or industrial land uses exist in the Oyster Pond watershed. All data for analysis supplied by the Town of Falmouth.					

It should be noted that the higher water use in the Oyster Pond watershed may result from a higher actual occupancy than revealed in the census figures. The difficulty in determining seasonal shifts in occupancy is one of the reasons for using actual water use, rather than census figures. The relatively small number of residences in this watershed may also skew the water use average, but none-the-less the measured flows still represent the actual usage. Therefore, since there is no evidence of significantly different non-wastewater water uses in the Oyster Pond watershed, relative to the rest of Falmouth, the water use was deemed the best approach for determination of septic system flows.

There are no existing or future (buildout) water use for commercial or industrial-classified lands in the Oyster Pond watershed. There is no land classified for industrial use in the watershed and no existing commercial uses. The land shown in Figure IV-3 that is classified as commercially developable land is owned by the Woods Hole Oceanographic Institution (WHOI). As a result of planned expansion on this site, WHOI recently (2/5/04) received a Cape Cod Commission regulatory decision and the proposed wastewater discharge from this expansion will be located outside of the Oyster Pond watershed. This decision was taken in part based upon the present MEP Oyster Pond analysis. Thus, there are no buildout commercial, or industrial, water use additions in the watershed.

Nitrogen Loading Input Factors: Residential Lawns

In most southeastern Massachusetts watersheds, nitrogen applied to the land to fertilize residential lawns is the second major source of nitrogen to receiving coastal waters after wastewater associated nitrogen discharges. However, residential lawn fertilizer use has rarely been directly measured in previous watershed-based nitrogen loading investigations. Instead, lawn fertilizer nitrogen loads have been estimated based upon a number of assumptions: a) each household applies fertilizer, b) cumulative annual applications are 3 pounds per 1,000 sq. ft., c) each lawn is 5000 sq. ft., and d) only 25% of the nitrogen applied reaches the groundwater

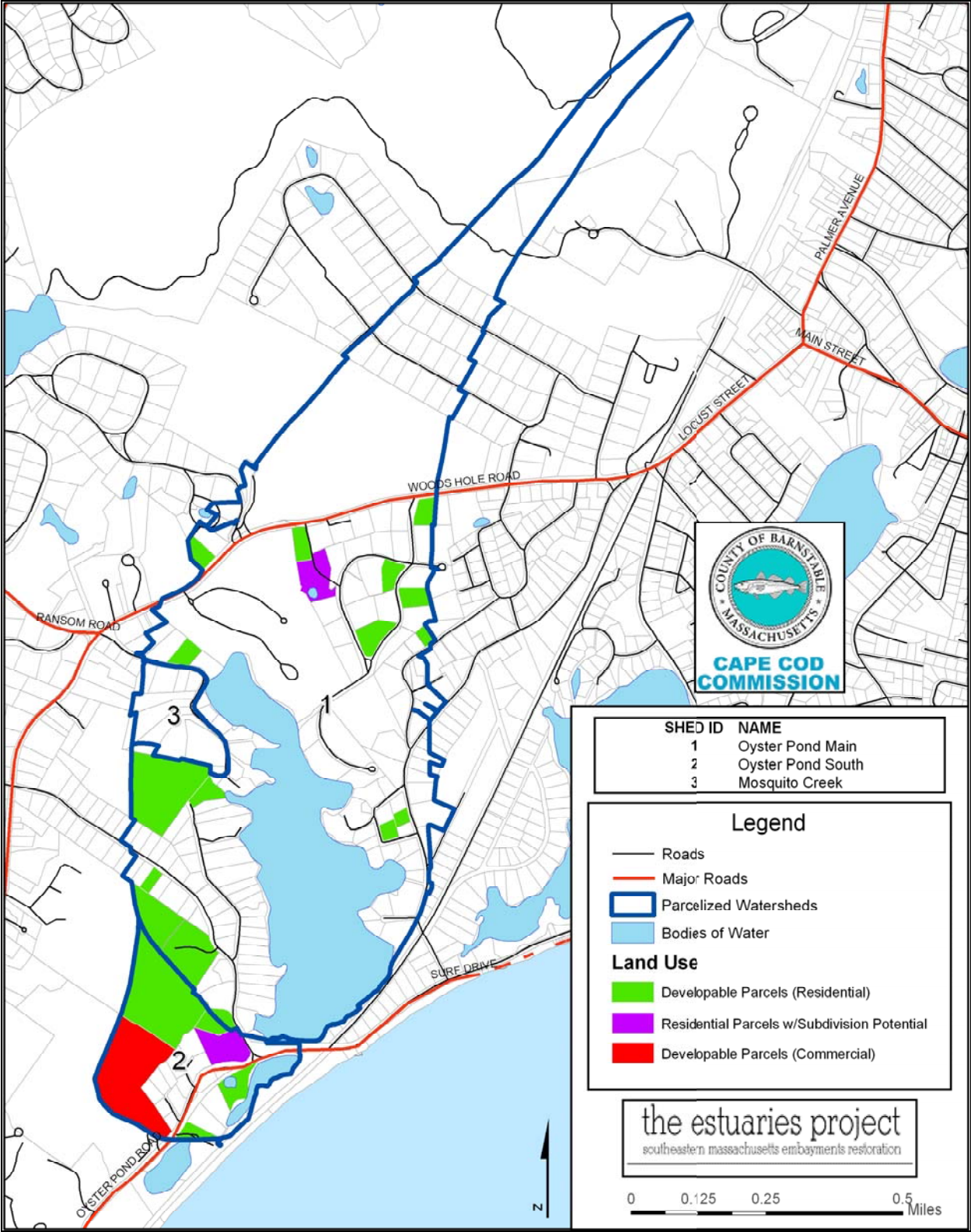


Figure IV-3. Parcels, Parcelized Watersheds, and Developable Parcels in the Oyster Pond watersheds. The Developable Commercial Parcel is owned by WHOI.

(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 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 for use in the nitrogen loading calculations. It is likely that this still represents a conservative estimate of nitrogen load from residential lawns. It should be noted that professionally maintained lawns were found to have the higher rate of fertilization application and hence higher estimated loss to groundwater of 3 lb/lawn/yr.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2000). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and Massachusetts DEP's Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). Factors used in the nitrogen loading analysis for the Oyster Pond watershed are listed in Table IV-3.

Table IV-3. Primary Nitrogen Loading Factors used in Oyster Pond MEP analysis. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Falmouth data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.			
Nitrogen Concentrations:	mg/l	Recharge Rates:	in/yr
Road Run-off	1.5	Impervious Surfaces	40
Roof Run-off	0.75	Natural and Lawn Areas	27.25
Direct Precipitation on Embayments and Ponds	1.09	Water Use/Wastewater:	
Natural Area Recharge	0.072	For Single Family Residence Parcels wo/water accounts and Buildout additions:	209 gpd
Wastewater Coefficient	23.63		
Fertilizers:		For Parcels w/water accounts:	Measured annual water use
Average Residential Lawn Size (ft ²)*	5,000		
Residential Watershed Nitrogen Rate (lbs/lawn)*	1.08		
Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined by site-specific information			

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information was linked to the parcel coverages, parcels were assigned to various watersheds based initially on whether at least 50% or more of the land area of each parcel was located within a respective watershed. Following the assigning of boundary parcels, all large parcels were examined separately and were split (as appropriate) in order to obtain less than a 2% difference between the total land area of each watershed and the sum of the area of the parcels within each watershed. The resulting “parcelized” watersheds are shown in Figure IV-3. 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 (condominiums, etc.) were also assigned at this stage. It should be noted that small shifts in nitrogen loading due to the above assignment procedure, has a negligible effect on the total nitrogen loading to the Oyster Pond estuary. The effort was undertaken to better define the sub-embayment loads to enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

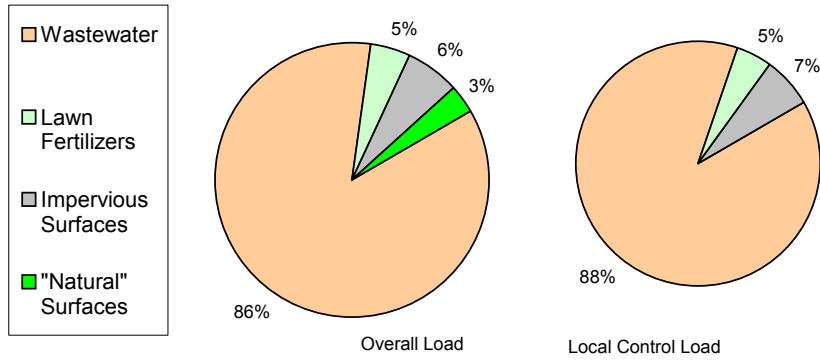
Following the assignment of all parcels to individual watersheds, tables were generated for each of the five sub-watersheds summarizing water use, parcel area, frequency, sewer connections, private wells, and road area. As mentioned above, these tables were then condensed to three subwatersheds based upon the time of travel analysis (<10 yr vs. > 10 yr) discussed above.

The three individual sub-watershed assessments were then integrated to generate nitrogen loading tables relating to the each of the individual estuaries and their major components: Oyster Pond Main, Oyster Pond South, and Mosquito Creek. The sub-embayments represent the functional embayment units for the Linked Watershed-Embayment Model’s water quality component.

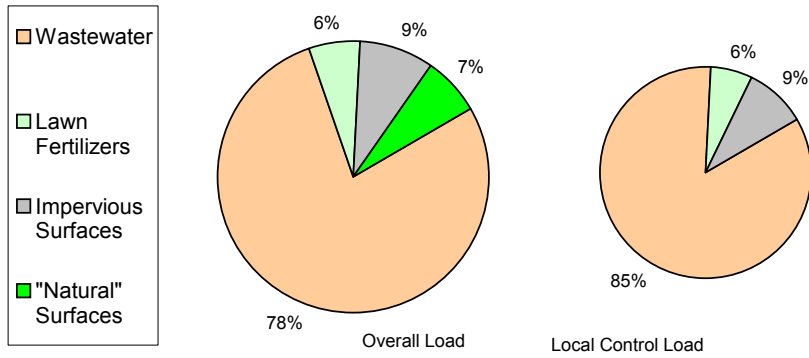
For management purposes, the aggregated sub-embayment watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Oyster Pond system the major types of nitrogen loads are: wastewater, fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-4). The output of the watershed nitrogen loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each land use category (Figures IV-4 a-c). This annual watershed nitrogen input is then adjusted for natural nitrogen attenuation during transport to the estuarine system before use in the embayment water quality sub-model. Natural attenuation within Mosquito Creek is also directly measured (Section IV.2) and compared to the attenuated annual watershed nitrogen load from the land-use sub-model.

Table IV-4. Oyster Pond Nitrogen Loads. Attenuation of system nitrogen loads occurs as nitrogen moves through the Mosquito Creek stream system during transport to the estuary. Atmospheric deposition of nitrogen to the estuarine surface was determined (main basin = 282 kg yr⁻¹, Lagoon = 10 kg yr⁻¹) and used in the modeling but was not included in this table.

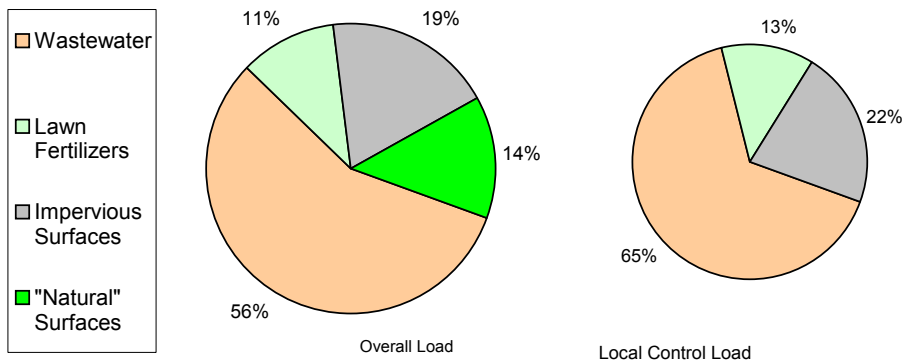
		<i>Oyster Pond N Loads by Input:</i>					<i>Present N Loads</i>			<i>Buildout N Loads</i>		
Name	Watershed ID#	SEPT	Lawn Fertilizers	Impervious Surfaces	"Natural" Surfaces	Buildout	UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Oyster Pond System		1364	78	107	60	174	1609		1603	1783		1778
Oyster Pond Main	1	1270	70	94	50	174	1484		1484	1658		1658
Oyster Pond_South	2	84	6	9	8	0	107		107	107		107
Mosquito Creek	3	10	2	3	2	0	18	30%	12	18	30%	12



a. Oyster Pond Main



b. Oyster Pond South



c. Mosquito Creek

Figure IV-4 (a-c). Land use-specific unattenuated nitrogen load (by percent) to the (a) Oyster Pond Main subwatershed, (b) Oyster Pond South subwatershed, and (c) Mosquito Creek subwatershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.

Buildout

In order to gauge potential future nitrogen loads resulting from continuing development, the potential number of residential, commercial, and industrial lots within each subwatershed was determined from the GIS database (Figure IV-3). Buildout of parcels were determined in consultation with the Falmouth Planning Department, including commercial parcel estimates. All municipal overlay districts (e.g., water resource protection districts) and existing zoning were considered in the determination of minimum lot sizes. A nitrogen load for each parcel was determined for the existing development using the factors presented in Table IV-3 and discussed above. A summary of potential additional nitrogen loading from build-out is presented as unattenuated and attenuated loads in Table IV-4. However, only the attenuated nitrogen loads were used for the water quality modeling, as the unattenuated rates of nitrogen loading would not permit model validation to conditions within embayment waters under any realistic physical conditions.

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, sewerage analysis, enhanced flushing, pond/wetland restoration for natural attenuation, etc.) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the sub-watersheds of the Oyster Pond system being investigated under this nutrient threshold analysis was based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1). If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watersheds where nitrogen transport from source to estuarine waters is through groundwater flow in sandy outwash aquifers. The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, nitrogen passes through a surface water ecosystem (pond, wetland, stream) on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) is diminished by natural biological processes that represent removal (not just temporary storage). However, this natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes. In the case of the Oyster Pond System watershed, only a small fraction of the freshwater flow (<5%) and transported nitrogen passes through a surface water system (Mosquito/Quivett Creek) prior to entering the head of the pond. Therefore, there is only a limited potential for nitrogen attenuation in this system. Stream gauging and water quality sampling on the creek was undertaken to determine the degree of natural attenuation.

Failure to determine the attenuation of watershed derived nitrogen generally results in overestimates of the nitrogen load to receiving estuarine waters. If nitrogen attenuation is significant in one portion of a watershed and insignificant in another the result is that nitrogen management would likely be more effective in achieving water quality improvements if focused on the watershed region having unattenuated nitrogen transport (other factors being equal). In addition to attenuation by freshwater ponds (see Section IV.1.3, above), attenuation in surface

water flows is also important. An example of the significance of surface water nitrogen attenuation relating to embayment nitrogen management was seen in the Agawam River, where >50% of nitrogen originating within the upper watershed was attenuated prior to discharge to the Wareham River Estuary (CDM 2001). Similarly, MEP analysis of the Quashnet River indicates that in the upland watershed, which has natural attenuation predominantly associated with riverine processes, the integrated attenuation was 39% (Howes et al. 2004). In addition, a preliminary study of Great, Green and Bournes Ponds in Falmouth, measurements indicated a 30% attenuation of nitrogen during stream transport (Howes and Ramsey 2001). An example where natural attenuation played a significant role in nitrogen management can be seen relative to West Falmouth Harbor (Falmouth, MA), where ~40% of the nitrogen discharge to the Harbor originating from the groundwater effluent plume emanating from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Similarly, the small tidal basin of Frost Fish Creek in the Town of Chatham showed ~20% nitrogen attenuation or watershed nitrogen load prior to discharge to Ryders Cove. Clearly, proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements of upper watershed attenuation were undertaken as part of the MEP Approach. MEP conducted long-term measurements of natural attenuation relating to surface water discharge to the head of the Oyster Pond embayment system. There are no natural attenuation measures by fresh kettle ponds, as there are no significant ponds within the Oyster Pond watershed. Therefore, in this system the only site-specific natural attenuation study conducted was in the 1 minor surface water flow system (e.g. Mosquito/Quivett Creek discharging to the head of Oyster Pond).

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 Mosquito/Quivett Creek (at Ransom Road) provides a direct integrated measure of all of the processes presently transforming and attenuating nitrogen in the sub-watershed upgradient from the gauging site. Flow and nitrogen load were measured at the site for 20 months of record (Figure IV-5). During the study period, velocity profiles were conducted on the creek at one to two month intervals. The summation of the products of stream subsection areas of the stream cross-section and the respective measured velocities represent the computation of instantaneous stream flow (Q).

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

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

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

where by:

Q = Stream discharge (m^3/s)

A = Stream subsection cross sectional area (m^2)

V = Stream subsection velocity (m/s)

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

Periodic measurement of flows over the entire stream gauge deployment period allowed for the development of a stage-discharge 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 cases of tidal influence on stream stage (which was not the case in Mosquito/Quivett Creek), the diurnal low tide stage value is extracted on a day by day basis in order to resolve the stage value indicative of strictly freshwater flow. The two low tide stage values for any given day were averaged and the average stage value for a given day was then entered into the stage – discharge relation in order to compute daily flow. A complete annual record of stream flow (365 days) was generated for the surface water discharge flowing into the head of the Oyster Pond system.

The annual flow record for the Mosquito / Quivett Creek surface water flow was merged with the nutrient data set generated through the weekly water quality sampling to determine nitrogen loading rates to the head of Oyster Pond. Nitrogen discharge from the stream was calculated using the paired daily discharge and nitrogen concentration data to determine the mass flux of nitrogen through the gauging site. 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 the pond currently reduces (percent attenuation) nitrogen loading to the Oyster Pond embayment system.



Figure IV-5. Location of stream gauge on Mosquito/Quivett Creek (red triangle), which discharges to the brackish waters of Oyster Pond.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Mosquito/Quivett Creek Discharge to Oyster Pond (upper)

Unlike many of the embayment systems on Cape Cod, the Oyster Pond system does not receive significant stream discharges originating from a freshwater pond or groundwater inflows. Instead, Oyster Pond receives almost all of the watershed freshwater inflow thorough direct groundwater discharge. However, like most embayments in southeastern Massachusetts, the Oyster Pond system does receive most of its freshwater inflow to its more inland portions and does support a small stream (Mosquito/Quivett Creek) discharging to its headwaters. Mosquito/Quivett Creek is essentially a groundwater fed stream that originates from a relatively undeveloped upland area and small wetland. The stream may serve to enhance attenuation of nitrogen that it transports within the small streambed associated with the Creek. 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 Mosquito / Quivett Creek above the gauge site and the measured annual discharge of nitrogen to the head of Oyster Pond, Figure IV-5. However, given its small flows, limited watershed nitrogen load and configuration, significant nitrogen attenuation was not anticipated for this stream.

At the Mosquito Creek (Quivett Creek) gauge site, a continuously recording vented calibrated water level gauge was installed to yield the level of water in the creek that carries the flows and associated nitrogen load to the head of Oyster Pond. Calibration of the gauge was checked monthly. The gauge on Mosquito Creek was installed on July 1, 2002 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection was conducted until March 11, 2004 for a total deployment of 21 months. The 12-month uninterrupted record used in this analysis encompasses the summer 2003 field season.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Mosquito Creek site based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allowed for the determination of nitrogen mass discharge to the head of Oyster Pond (Figures IV-6, 7, 8, 9 and Table IV-5). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gauge site.

The annual freshwater flow record for Mosquito Creek measured by the MEP, was compared to the long-term average flows determined by the USGS/CCC modeling effort (Table III-1). The measured freshwater discharge from Mosquito Creek ($35,521 \text{ m}^3 \text{ yr}^{-1}$) was 96% of the long-term average modeled flows ($36,917 \text{ m}^3 \text{ yr}^{-1}$) based on a 27.25 inch recharge rate. As such, the watershed and river datasets appear to be in balance.

Total nitrogen concentrations within the Mosquito Creek outflow were relatively high, $1.182 \text{ mg N L}^{-1}$, yielding an average daily total nitrogen discharge to the estuary of 0.12 kg/day and a measured total annual TN load of 42 kg/yr . This measured nitrogen load is higher than the watershed nitrogen loading (Table IV-6) and therefore it appears that the small stream has processes that provide additional nitrogen to its waters and that these additional sources exceed any nitrogen removal processes that may be occurring within its channel. Note that the higher than expected nitrogen discharge (42 kg N yr^{-1}) is unlikely the result of an overestimate of

stream flow volume, since the USGS modeled and directly measured annual stream volumes agreed to within 5 %.

Table IV-5. Comparison of water flow and nitrogen discharges from Mosquito Creek (Quivett Creek) discharging to Oyster Pond. The "Stream" data is from the MEP stream gauging effort. Watershed data is based upon the MEP watershed modeling effort by USGS and CCC.

Stream Discharge Parameter	Stream Discharge to Oyster Pond ^(a)	Data Source
Total Days of Record	365 ^(b)	(1)
Flow Characteristics		
Stream Average Discharge (m3/day)	97	(1)
Contributing Area Average Discharge (m3/day)		(2)
Discharge Stream 2002-03 vs. Long-term Discharge		
Nitrogen Characteristics		
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.197	(1)
Stream Average Total N Concentration (mg N/L)	1.182	(1)
Nitrate + Nitrite as Percent of Total N (%)	17%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	0.115	(1)
TN Average Contributing UN-attenuated Load (kg/day)	0.049	(2)
Attenuation of Nitrogen in Pond/Stream (%)	-57% **	(3)
<p>(a) Flow and N load to stream discharging to Oyster Pond is from the Mosquito Creek Sub-watershed #3 in Figure IV-3.</p> <p>(b) September 10, 2002 to September 10, 2003.</p> <p>(1) MEP gage site data</p> <p>(2) Calculated from MEP watershed delineations to Gage location on Mosquito Creek (Quivett Creek) for flow to head of Oyster Pond; and the annual recharge rate.</p> <p>(3) Calculated based upon the measured TN discharge from the stream vs. the unattenuated watershed load.</p> <p>** The negative attenuation indicates that nitrogen is <u>added</u> to creek waters during passage to Oyster Pond.</p>		

A potential explanation of the higher nitrogen loads in Mosquito Creek can be found in the forms of nitrogen that the stream is discharging. In typical groundwater fed streams on Cape Cod, the predominant form of nitrogen is nitrate. This reflects the fact that the predominant form of nitrogen in groundwater is also nitrate. As the nitrogen moves through surface waters, the nitrate is either denitrified (to dinitrogen gas) or taken up by plants and converted to organic forms. These may be later released during plant decay and some may be subsequently denitrified. In Mosquito Creek nitrate was only a minor portion of the total nitrogen pool (17%), rather, organic forms predominated (particulate 9% and dissolved organic nitrogen 66%) and ammonium, an indicator of plant decay, accounted for 8%. At present the most likely cause of the high nitrogen discharge from Mosquito Creek appears to be related to organic matter which is observed to be deposited within the stream channel from the riparian zone. Accumulations of leaves and vascular plants are common. To the extent that these represent non-creek plants, this provides a potential pathway for transport of nitrogen that would not generally reach the Pond. The fact that three quarters of the total nitrogen pool is in organic forms and that one third of the inorganic nitrogen pool is ammonium (an indicator of organic decay) supports this hypothesis. The high dissolved organic nitrogen levels during the period of plant senescence

and leaf fall is also indicative (Figure IV-8). It should be noted that the MEP Technical Team examined the possibility of missing nitrogen sources within the watershed to Mosquito Creek, but could not account for the observations. While Mosquito/Quivett Creek did not show nitrogen attenuation, it should also be noted that the measured nitrogen loads are small, only ~2% of the total load.

The directly measured nitrogen loads discharging from Mosquito/Quivett Creek to Oyster Pond waters were used in the nitrogen modeling in Section VI.

Table IV-6. Summary of annual volumetric discharge and nitrogen load from Mosquito Creek (Quivett Creek) to the head of Oyster Pond based upon the data presented in Figures IV-6, 7, 8, 9 and Table IV-5.				
EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (M ³ /YEAR)	ATTENUATED LOAD (KG/YEAR)	
			NOX	TN
Mosquito Creek to Oyster Creek	September 10, 2002 to September 10, 2003	35521	7	42
Mosquito Creek to Oyster Pond (CCC)	Based on Watershed Area and Recharge and Nitrogen Removal	36865	--	12

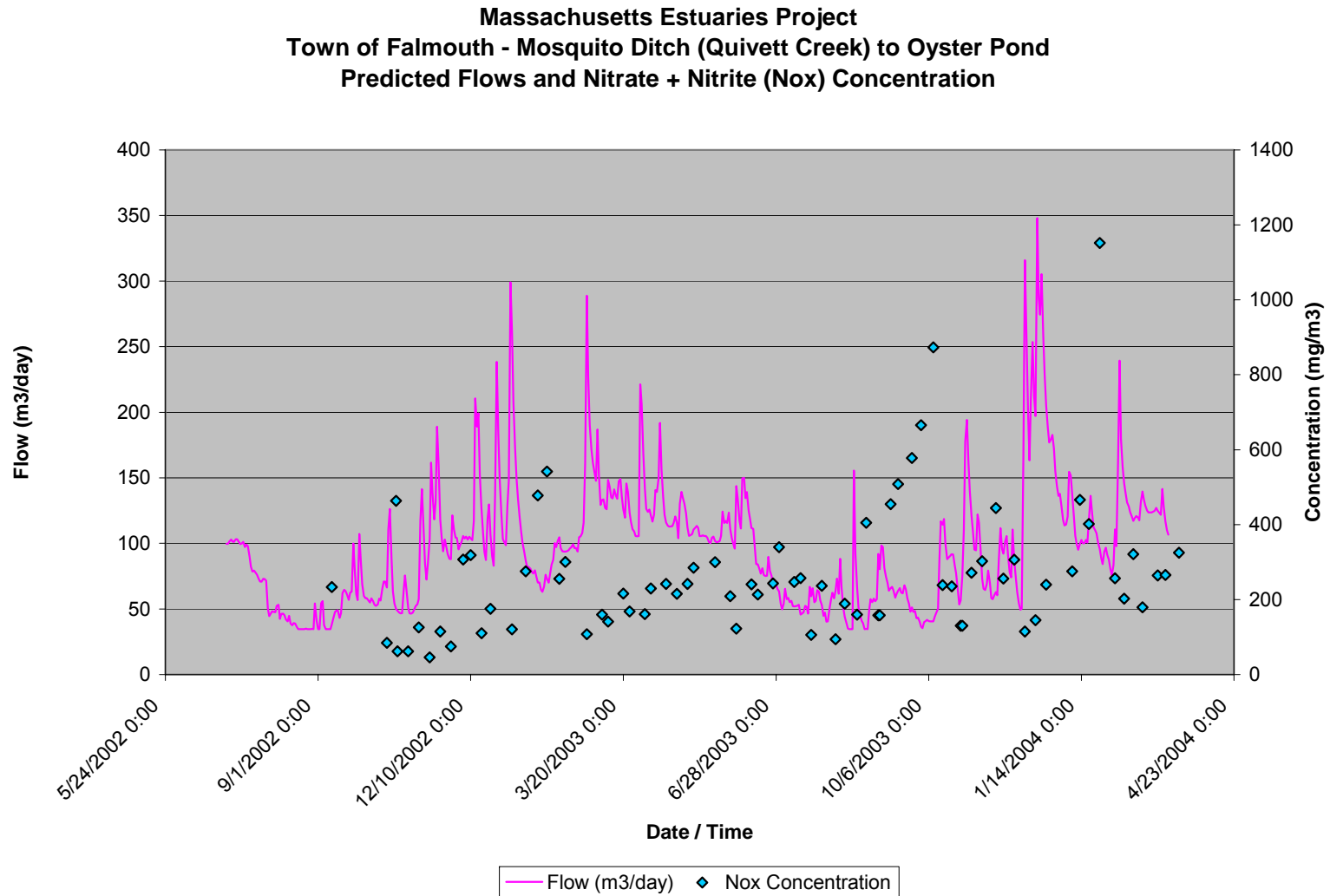


Figure IV-6. Mosquito Creek (Quivett Creek) discharge (solid pink line) and nitrate+nitrite (blue boxes) concentrations for determination of annual volumetric discharge and nitrogen load from the Mosquito Creek sub-watershed to the Oyster Pond system (Table IV-5).

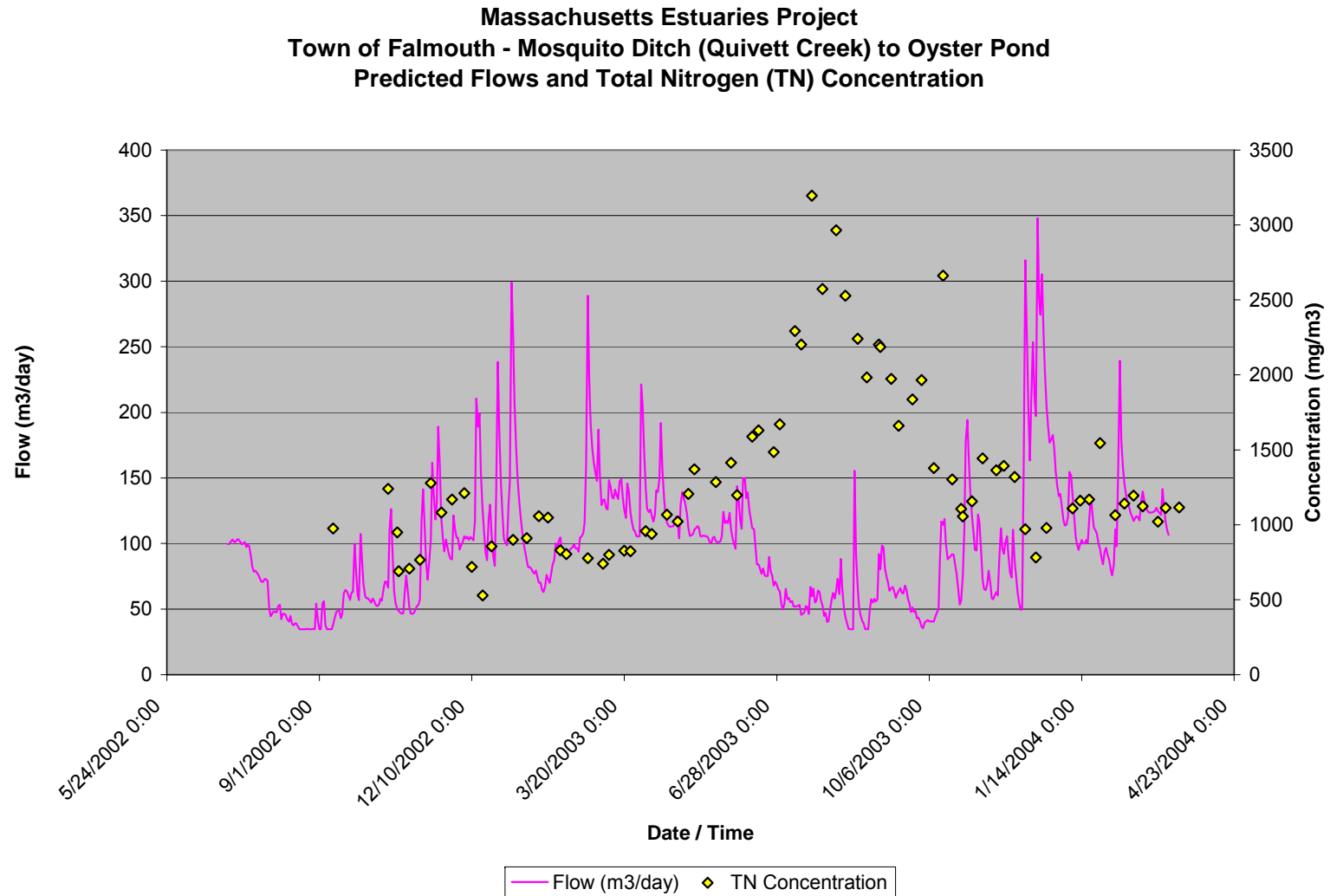


Figure IV-7. Mosquito Creek (Quivett Creek) discharge (solid pink line) and total nitrogen (yellow triangles) concentrations for determination of annual volumetric discharge and nitrogen load from the Mosquito Creek sub-watershed to the Oyster Pond System (Table IV-5).

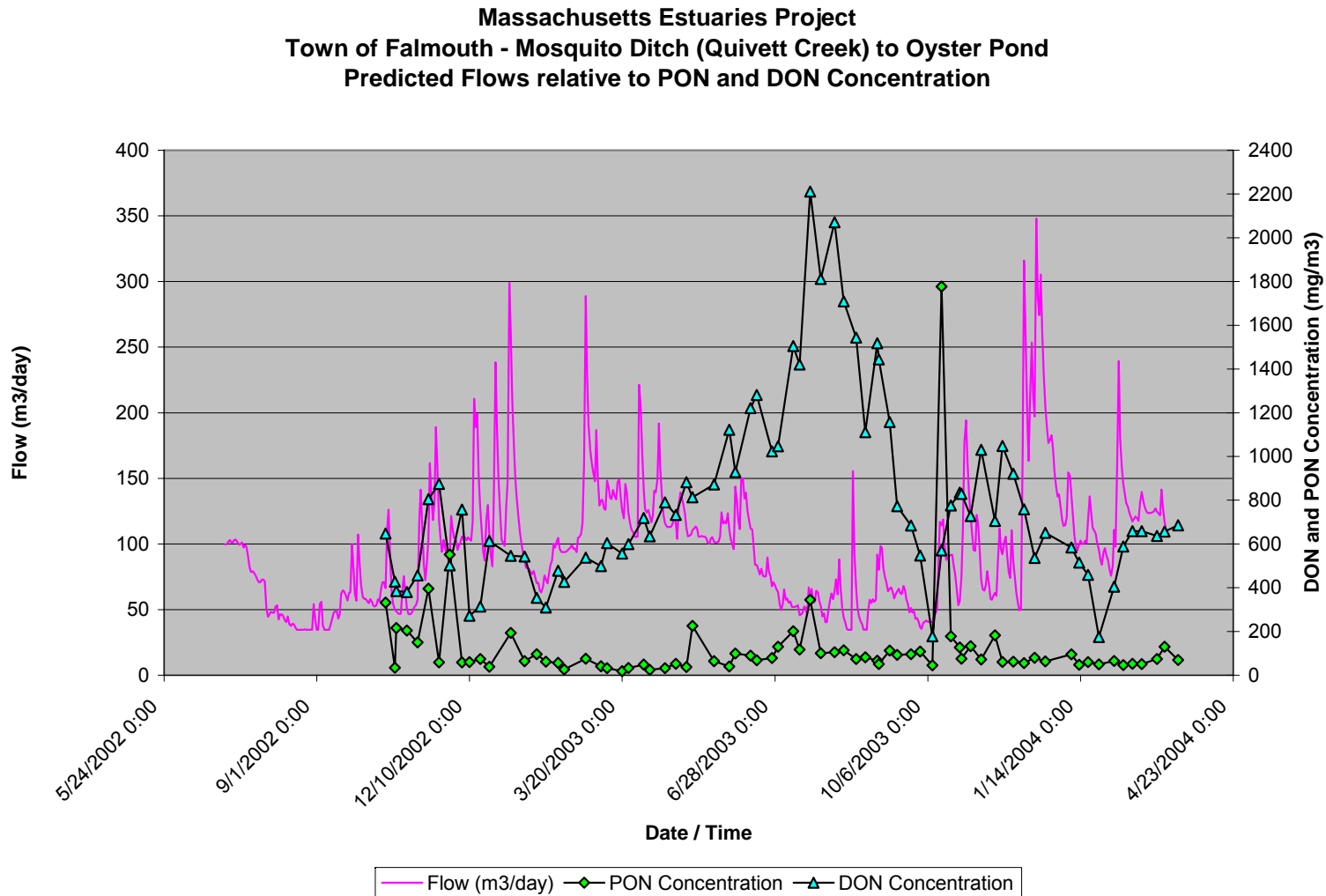


Figure IV-8. Mosquito Creek (Quivett Creek) discharge (solid pink line), Particulate Organic Nitrogen (green diamonds) and Dissolved Organic Nitrogen (blue triangles) concentrations for determination of annual volumetric discharge and nitrogen load from the Mosquito Creek sub-watershed to the Oyster Pond System.

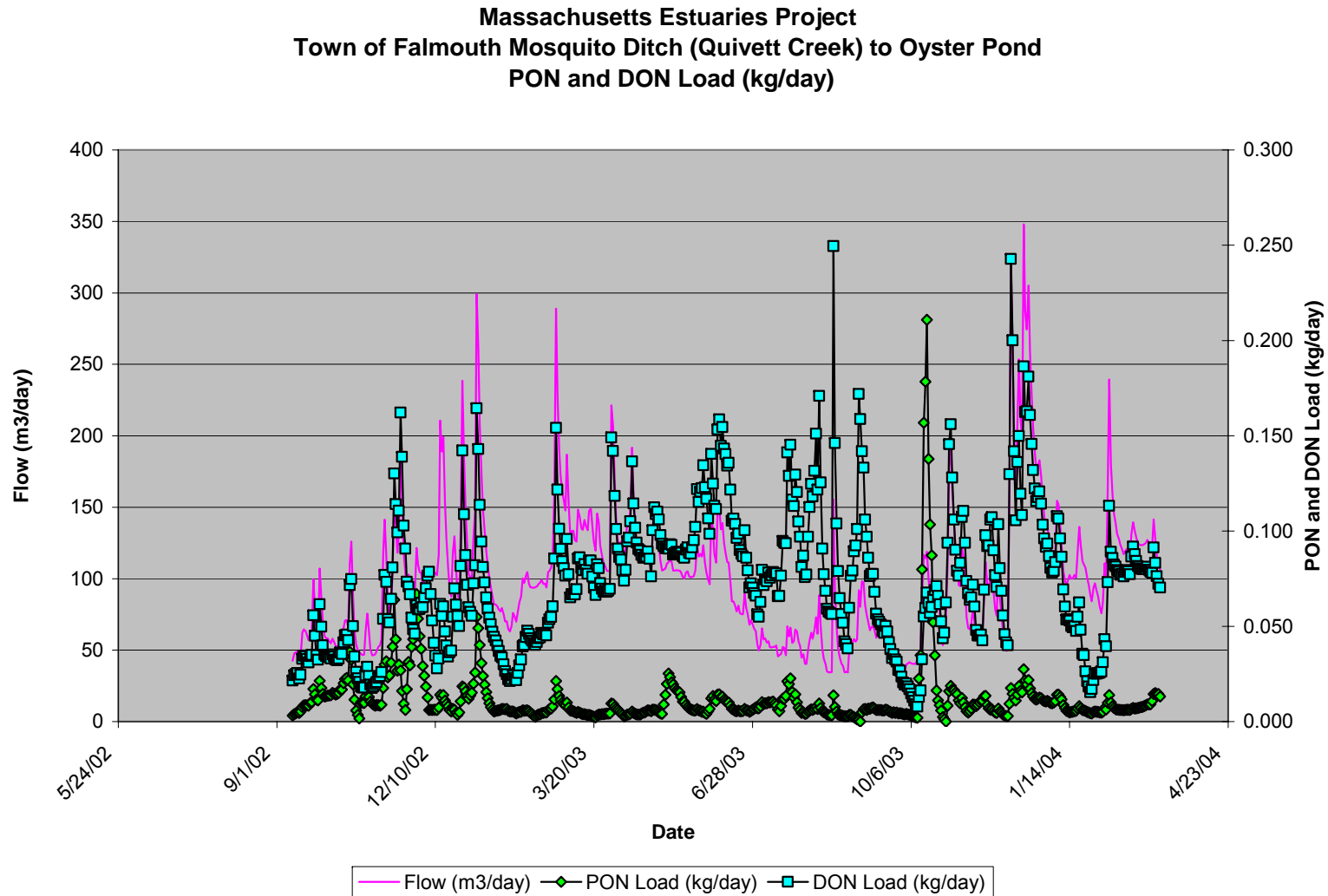


Figure IV-9. Mosquito Creek (Quivett Creek) discharge (solid pink line), Particulate Organic Nitrogen (green diamonds) and Dissolved Organic Nitrogen (blue triangles) loads for determination of annual volumetric discharge and nitrogen load from the Mosquito Creek sub-watershed to the Oyster Pond System.

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 within each major basin area of the Oyster Pond embayment system. The mass exchange of nitrogen between watercolumn and sediments is a fundamental factor in controlling nitrogen levels within coastal waters. These fluxes and their associated biogeochemical pools relate directly to carbon, nutrient and oxygen dynamics and the nutrient related ecological health of these shallow marine ecosystems. In addition, these data are required for the proper modeling of nitrogen in shallow aquatic systems, both fresh and salt water.

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

As stated in above sections, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the Oyster Pond embayment 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 watercolumn (once it entered), then predicting watercolumn nitrogen levels would be simply a matter of determining the watershed loads, dispersion, and hydrodynamic flushing. However, as nitrogen enters the 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”. In well flushed estuaries, most of these “particles” remain in the watercolumn for sufficient time to be flushed out to a downgradient larger waterbody (like Vineyard Sound). However in systems with very low flushing rates, like Oyster Pond, most of these phytoplankton particles are deposited to the bottom through grazing by zooplankton or filtration from the water by shellfish and other benthic animals or merely by senescence and death. With these processes of deposition both remove nitrogen from the watercolumn and support its return as inorganic nitrogen after decay and regeneration.

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. Oyster Pond is tidally restricted in the extreme and operates essentially as a brackish kettle pond, with a very small outflow, exchanging <0.5% of its volume per day.

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 watercolumn for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that 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. However, in some enclosed depositional basins the sediments may provide a net removal of nitrogen in summer due to denitrification losses and burial in the sediments. It is during these warmer months that estuarine waters are most sensitive to nitrogen removals and loadings. Failure to account for these biogeochemical processes in the sediments generally

results in significant errors in determination of threshold nitrogen loadings. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

IV.3.2 Method for determining sediment-watercolumn nitrogen exchange

For the Oyster Pond system, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (June-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected from 7 sites in Oyster Pond system (Figure IV-10) in waters less than 3.5 meters depth (i.e. above the depth of stratification). Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample.

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

Oyster Pond Benthic Nutrient Regeneration Cores

• Station OP-7	1 core	(Upper Region of Pond)
• Station OP-6	1 core	(Upper Region of Pond)
• Station OP-5	1 core	(Middle Region of Pond)
• Station OP-4	1 core	(Middle Region of Pond)
• Station OP-3	1 core	(Middle Region of Pond)
• Station OP-2	1 core	(Middle Region of Pond)
• Station OP-1	1 core	(Middle Region of Pond)

The results for each core site were combined for calculating the net nitrogen regeneration rates for the water quality modeling effort.

Sediment-watercolumn exchange follows the methods of Jorgensen (1977), Klump and Martens (1983), and Howes *et al.* (1995) for nutrients and metabolism. Upon return to the field laboratory (private residence located nearby to Oyster 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°C) for assay of nitrate + nitrite (Cd reduction: Lachat Autoanalysis), and DON (D'Elia *et al.* 1977). Rates were determined from linear regression of analyte concentrations through time.

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



Figure IV-10. Oyster Pond embayment system sediment sampling sites (red triangles) for determination of nitrogen regeneration rates. Blue diamond represents stream gauge location on Mosquito/Quivett Creek. Numbers are for reference in Table IV-7.

Table IV-7. Rates of net nitrogen return from sediments to the overlying waters of the Oyster Pond embayment system. These values are combined with the basin area to determine total nitrogen mass for the water quality model (see Section VI). Measurements represent July -August rates. The negative nitrogen return is consistent with Oyster Pond's physical structure as a brackish kettle pond.

Location	Sediment Nitrogen Flux (mg N m ⁻² d ⁻¹)			
	Mean	S.E.	N	Sta
Oyster Pond Estuary				
North Basin	-7.1	4.5	2	6,7
Main Basin	-11.5	3.2	5	1-5
Whole System	-10.3	4.8	7	1-7
Station numbers refer to Figure IV-10.				

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

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

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

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

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. Sediments within deep depositional basins or those in systems with low flushing rates, tend to show a low or negative net release of 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.

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

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

Sediment sampling was conducted in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model (Figure IV-10). 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 and the average summer particulate carbon and nitrogen concentration within the overlying water. Oyster Pond is operating as a brackish kettle pond with very low tidal exchange rates and relatively deep water compared to other estuaries in southeastern Massachusetts. The low tide exchange results in very low velocities and the deep water increases the time required to settle to the bottom. The average water depth in Oyster Pond is 3m, compared to <1 meter in other estuaries along Falmouth's south coast. Settling rates of two thirds of the rates in other large, but shallower, basins were used. Adjusting the measured sediment releases is essential in order not to over-estimate the sediment nitrogen source and to account for those sediment areas that are net nitrogen sinks for the aquatic system. This approach was validated in outer Cape Cod embayments (Town of Chatham) by examining the relative fraction of the sediment

carbon turnover (total sediment metabolism) that 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.

Sampling was distributed throughout the embayment system, except in the deep waters which stratify and show periodic hypoxia/anoxia during summer. Nitrogen transformations in these deep sediments was determined not to have impact on surface water nitrogen levels during stratification. Vertical mixing in Oyster Pond during summer is predominantly driven by wind. Mixing by tides, so important to many estuarine systems is relatively unimportant due to the restricted nature of the Oyster Pond exchange with Vineyard Sound. Nitrogen settling below the mixed surface layer of Oyster Pond enters sediments overlain by unmixed bottomwaters. Nitrogen release from these sediments must exit the sediment surface through diffusion, as there are typically not animals to enhance release through their activities (bioturbation). Once in the bottom waters, the released nutrients move upward (and horizontally) by diffusion. Evidence of this diffusion cell in bottom waters can be seen in dissolved sulfide profiles (Howes and Hart 1997). The result is nutrients build up in bottom waters during summer, with release in fall/winter storm driven mixing. The effect of this “capping” of about 20% of the bottom sediments of the pond is that nutrient release in these regions does not support mixed layer nitrogen balance during summer. Therefore, for the modeling of summer water quality (Section VI) the measured sediment nutrient release was distributed over the >80% of the pond bottom that is overlain by a mixed watercolumn. The rates in the Oyster Pond system were similar to other enclosed depositional basins, for example in the Town of Falmouth, Great Pond and Perch Pond (-16.4 and -20.2 mg N m⁻² d⁻¹). Similarly, in the nearby Vineyard Sound estuary, Popponesset Bay, sediment regeneration ranged from 85 to - 17 mg N m⁻² d⁻¹.

The physical structure of Oyster Pond provides for a whole system approach for determining sediment net nutrient release. Since Oyster Pond has low tidal exchange relative to its daily inputs, it is possible to determine the sediment nutrient release from changes in the mixed layer nitrogen mass. Over short periods (July 15 – August 30) Oyster Pond's nitrogen inputs from its watershed and atmosphere and its outputs to Vineyard Sound are constant. During this period losses or gains of nitrogen from the mixed layer can be attributed to net release from the sediments or net deposition. This approach provides an estimate of nitrogen release for the whole of Oyster Pond for comparison to the sediment core measurements. Based upon average total nitrogen information from the Falmouth PondWatch, for July and August collected over 1997-2004, there appears to be a decline in nitrogen in pond waters (Figure IV-11), which equates to a loss from the mixed layer of 0.0028 mg/L d⁻¹ or 1.7 kg N per pond d⁻¹ (based upon a mixed layer volume of 604,500 m³). This compares well with the measurements from the sediments, which yielded a net uptake from the mixed layer of 2.3 kg N per pond d⁻¹. Given that 1.7 kg d⁻¹ results in a more conservative estimate for pond restoration, it was the primary focus for the water quality modeling (Section VI).

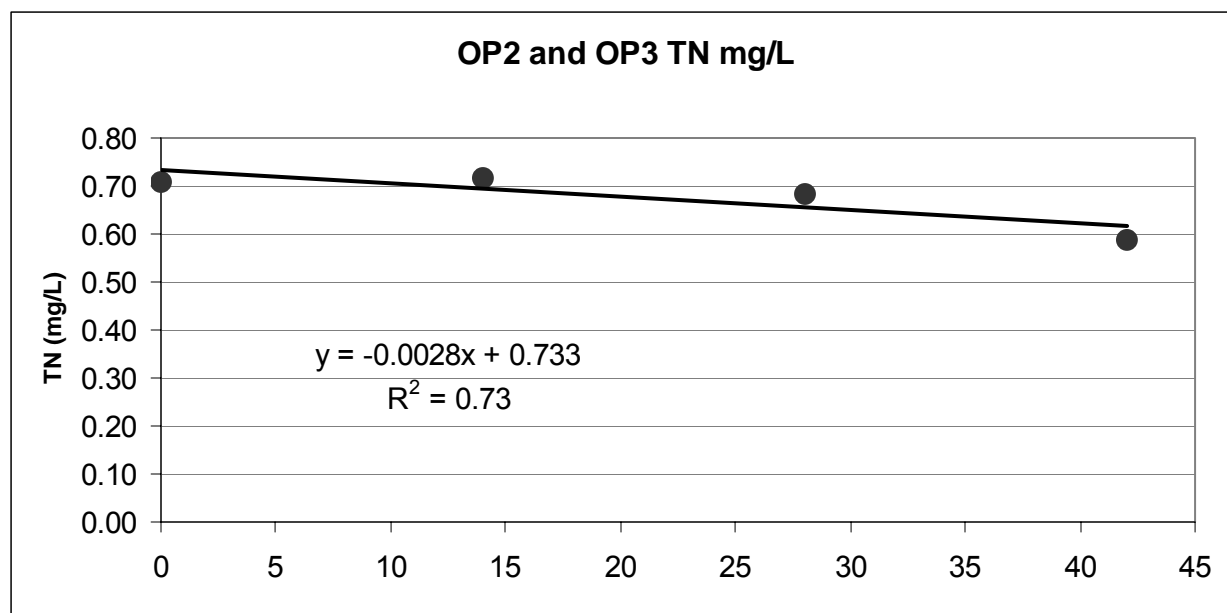


Figure IV-11. Total nitrogen concentrations in the Oyster Pond mixed layer (0-4m) courtesy of Falmouth PondWatch. Day 0 is in mid July (~15-July) in each year, 1997-2004.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This section summarizes field data collection effort and the development of hydrodynamic models for the Oyster Pond estuary system (Figure V-1). For this system, the final calibrated model offers an understanding of water movement through the estuary, and provides the first step towards evaluating water quality, as well as a tool for later determining nitrogen loading “thresholds”. Nutrient loading data combined with measured environmental parameters within the system become the basis for an advanced water quality model based on total nitrogen concentrations. This type of model provides a tool for evaluating existing estuarine water quality parameters, as well as determining the likely positive impacts of various alternatives for improving overall estuarine health, facilitating the understanding how pollutant loadings into the estuary will affect the biochemical environment and its ability to sustain a healthy marine habitat.

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

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

V.1.1 System Physical Setting

Oyster Pond is set in the southern shoreline of Falmouth. The layout of the Oyster Pond system is shown in the topographic map detail of Figure V-1. The main basin of the pond has a surface area of approximately 64 acres, and is open to Vineyard Sound via Trunk River, a small 800-foot-long tidal creek. Trunk River and Oyster Pond are separated by a small salt pond called the Lagoon, which has a surface area of less than 2 acres. Additionally, a weir is in place between the Pond and Lagoon. A barrier beach (which is typically 300 feet/100 m wide) forms the southeastern boundary of the Pond. Across this low lying coastal barrier beach run an abandoned railroad right of way (now a bike path) and Surf Drive. During extreme storm surge events (e.g., Hurricane Bob in August 1991) this barrier is overtopped, resulting in infrequent spikes in the salinity of this mainly brackish Pond.

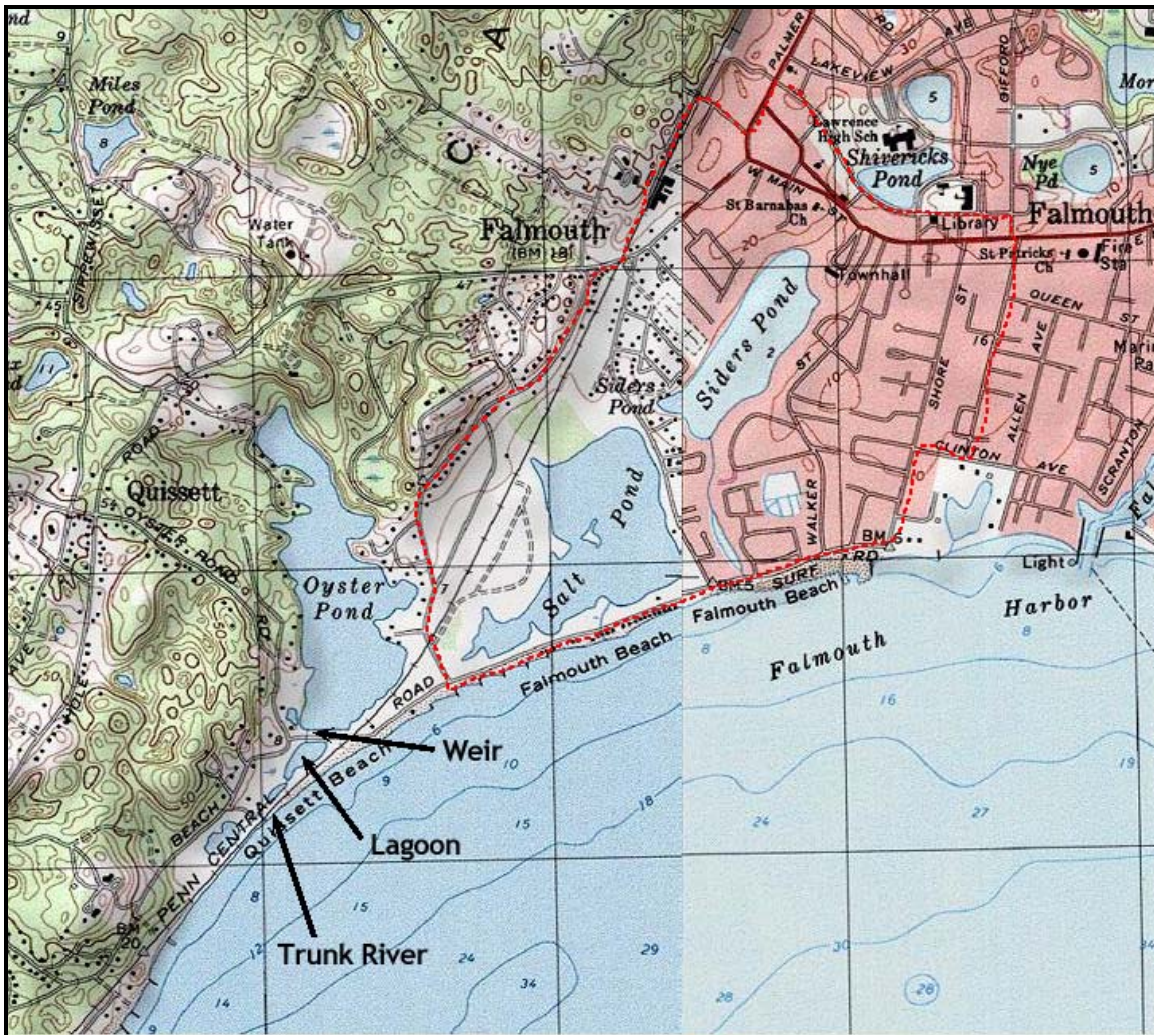


Figure V-1. Topographic map detail of Oyster Pond, in Falmouth, Massachusetts, with annotations indicating the location of the salinity control structure (weir), Lagoon, and Truck River inlet.

V.1.2 System Hydrodynamic Setting

In Oyster 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. Though tides in Vineyard Sound are only occasionally high enough to cause seawater flows into the pond, tidal flushing is still important to the stability and health of this estuary, mostly by its effect on salinity in the pond.

Flow control structures in the pond's inlet channel are used to maintain salinities between 2 and 4 ppt, and therefore foster a brackish ecosystem that is less vulnerable to salinity shocks from infrequent storm surges (Ramsey and Howes, 1996). Without periodic input of seawater (from normal tidal flow over the salinity control structure) to maintain the pond's salinity, the system over time would become fresher, and as a result more susceptible to habitat impairment due to large variations in salinity caused by infrequent storm surges from hurricanes and northeast storms (see Section VII).

The hydrodynamic modeling effort for Oyster Pond was similar to other estuarine systems modeled as part of the MEP, though the tidally restricted nature of this system required modifications to the modeling and analysis techniques that have been applied to simpler embayments. From the perspective of hydrodynamics, the most important difference between the Oyster Pond system and other estuaries in Falmouth is the adjustable salinity control/fish weir in the inlet channel to the Pond. A schematic of the weir is shown in Figure V-2. The weir has been in place since March 1998, and its primary utility is to control salinity in the main basin of the pond. Prior to the installation of the weir and improvements to the inlet channel, Oyster Pond was subject to widely varying salinities (Ramsey and Howes, 1996, see Section VII) and therefore provided variable quality habitat. Another important function of the salinity control/fish weir is to allow passage by anadromous fish (e.g., herring). For this purpose, the weir was constructed with a foot-wide adjustable fish weir section. The weir crest of this narrow passage is typically set at a lower elevation than the main weir section to help herring pass through the structure.

At the time that the weir was installed, the inlet channel (i.e., Trunk River) was essentially blocked by a sandy sill, which was high enough to prevent even minor storm surges from entering the pond. The result was the lowering of the pond salinity to ~2.5 ppt 12 months before the weir was installed. During this period water levels in the pond and lagoon were controlled by the sill in the inlet channel, and not by the weir. So, though the weir had been designed to maintain pond levels as +1.5 feet NGVD, the water level in both the pond and lagoon were typically +2.2 feet NGVD or greater prior to 2000.

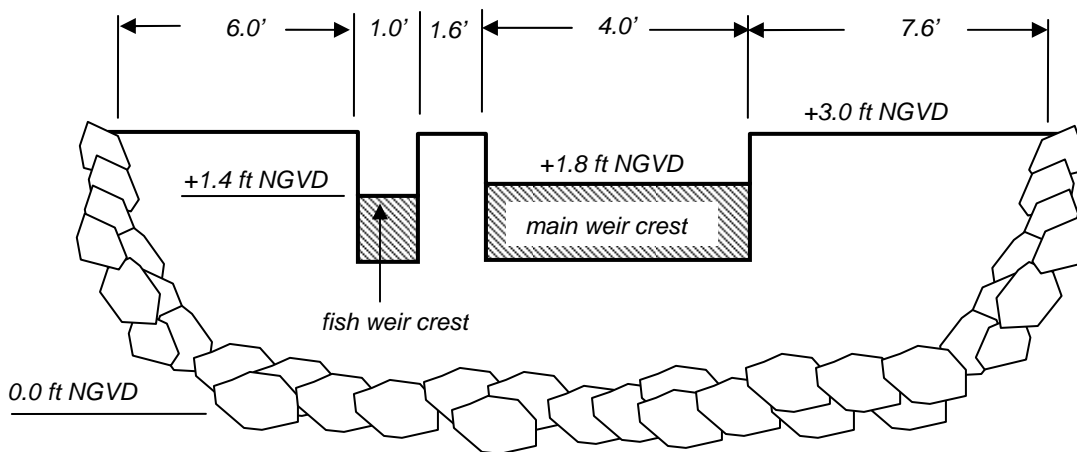


Figure V-2. Schematic cross-sectional view of the Oyster Pond salinity control/fish weir (facing into the pond), with dimensions and elevations (not drawn to scale). Weir crest elevations are as observed in March 2001 and August 2003.

As part of the original management plan for Oyster Pond, Trunk River was subsequently dredged to a depth of +1.2 feet NGVD, and the inlet jetties were reconstructed to limit shoaling in the inlet channel. The conditions at the Trunk River inlet are shown in a 1999 photograph presented in Figure V-3. The improvement work at the inlet was completed in 2000. Water elevation data collected in March of 2001 shows that the inlet improvements allowed the weir to control levels in the pond, rather than the sill in the inlet channel. However, tide data collected in August 2003, in support of the bathymetry survey of the Pond, showed that sill in Trunk River had reformed, and was again impairing tidal exchange between the lagoon and the sound. However, during the summers of 2001-2004 the range of surface water salinities throughout Oyster Pond was relatively constant, ranging from 1.7-2.4 ppt (Falmouth PondWatch).



Figure V-3. View (circa 1999) looking offshore into Vineyard Sound from the Trunk River inlet prior to reconstruction of the jetties and inlet channel (photograph credit: John Ramsey).

V.2 GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE SYSTEM

The southern coast of Cape Cod in the vicinity of Oyster and Salt Ponds is a relatively quiescent region. Although natural wave and tidal forces continue to reshape the shoreline, day-to-day conditions have limited impact on the shoreline migration and/or inlet stability. For typical wave conditions, longshore transport of sand is from west-to-east along the south coast of Falmouth, due primarily to the predominant local wind-driven waves. In contrast to the mild day-to-day conditions, infrequent hurricane events such as the hurricanes of 1938, 1944, and 1954, as well as Hurricane Bob in 1991, all caused significant overwash and transport of beach sediments. In addition, northeast storm events (causing waves to approach the Falmouth shoreline from the east and southeast) create a sediment transport reversal from typical conditions, where the longshore sediment transport is generally from east-to-west. The effect of this sediment transport reversal can often be seen by observing the sand impounded by the groins found along the shoreline. Under typical summer conditions sand is impounded along the west or southwest side of the groins and during winter easterly storm events sand is impounded along the east or northeast side of the groins. In years with a significant number of easterly storm events, the net alongshore sand transport direction becomes unclear. However, it appears that the overall long-term trend is a west-to-east transport of sand on the order of 4,000 cubic yards per year (based on an analysis of observed volumetric accretion at the Waquoit Bay west jetty between 1938 and 1961, performed by the U.S. Army Corps of Engineers, 1964). More recent dredging volumes at Great, Green, and Bournes Ponds entrances indicate that longshore transport may be as low as 1,000 cubic yards per year (personal communication with Barnstable County Dredge personnel, 2004). For comparison, longshore sediment transport rates along U.S. beaches exposed to Atlantic Ocean waves are

orders of magnitude higher (e.g. the Cape Cod National Seashore, the New Jersey Coast, and the east coast of Florida), typically 100,000 to 500,000 cubic yards per year.

Due to the quiescent wave environment and small tide range in the vicinity of the Oyster Pond and Salt Pond inlets, inlet migration is less of a concern than other areas of Cape Cod. According to Fitzgerald (1993), jetties along this shoreline were required to keep inlets open. Due to the relatively small surface area of Oyster Pond (compared to Great, Green, and Bournes Ponds), tidal flushing likely is inadequate to maintain a stable tidal inlet. The ephemeral nature of the inlet prior to placement of the railroad tracks in the late 1800's indicates this trend. Figure V-4 shows the location of the inlet in 1800, where a direct connection existed between Vineyard Sound and Oyster Pond. The more detailed survey by the U.S. Coast and Geodetic Survey in 1845 also shows a small tidal inlet in the southeastern corner of the Pond. Emory (1999) indicated that the inlet to Oyster Pond had been modified through natural coastal processes and installation of culverts following construction of the railroad bed in 1872. Following the collapse of a culvert directly connecting Oyster Pond to Vineyard Sound, the system was modified to the general condition shown in Figure V-1 (minus the weir). Small jetties were constructed at the inlet to inhibit infrequent inlet closures that could cause large ecological shifts to estuarine plant and animal communities.

V.2.1 Coastal Processes and Inlet Stability

Since inlet stability is partially governed by longshore coastal sediment transport, understanding the regional long-term shoreline change and littoral movement of sand is critical for evaluating stability of the tidal inlets along Falmouth's south shore. As discussed above, the observed longshore transport rates are relatively low, primarily as a result of the quiescent wave environment of Nantucket and Vineyard Sounds from Oyster Pond to Waquoit Bay. Although the amount of sand moving along the coast is small, the tidal prism through even the fully tidal inlets to Great, Green and Bournes Ponds also is relatively small. Since the construction of the jetty systems at each of the entrances, the inlets have generally reached equilibrium, where the tidal velocities through the main channels are sufficient to prevent significant shoaling. Recent annual dredging of these inlets has only been on the order of 1,000 cubic yards per year (personal communication with Barnstable County Dredge personnel, 2004).

In addition, it appears that the south coast of Falmouth (between Falmouth Harbor and the west entrance to Waquoit Bay) has generally equilibrated to changes in local coastal sediment transport caused by the construction of shoreline armoring. Extensive armoring of the Falmouth shoreline began in the late 1800s and early 1900s with construction of the railroad to Woods Hole, the old stone dock, the Falmouth Harbor jetties, and the Waquoit Bay east jetties. This shoreline armoring continued through the mid-1900s with the construction of stone groin fields, which often replaced existing wooden structures. In the Great, Green, and Bournes Ponds region, these wooden and stone structures were constructed to protect Menauhant Road and waterfront dwellings (Figures V-3 and V-4). The remnants of wooden groins and bulkheads can be found along much of Falmouth's south coast (Figure V-5).



Figure V-4. A portion of an 1800 map showing the south coast of Falmouth and Mashpee. This map depicts the condition of these inlets prior to the installation of jetties.

V.2.2 Shoreline Change Analysis

Shoreline change maps can effectively be used to evaluate the effects of long-term coastal processes. In addition, shoreline change maps also can indicate the effects of short-term changes that often occur as the result of anthropogenic (e.g. development of extensive shore protection structures) or natural (e.g. inlet migration) processes. Prior to developing conclusions and/or management recommendations that depend on shoreline change estimates, it is critical to understand potential errors and uncertainties associated with this type of analysis. Understanding the limitations of shoreline change data is critical for developing appropriate management strategies for shorelines and inlets in areas with relatively low shoreline migration rates, such as Falmouth's south coast.

The Massachusetts Coastal Zone Management Office (MCZM) recently updated their shoreline change analysis (Thieler et al., 2001) to incorporate more recent shoreline information. Specifically, the updated Massachusetts Shoreline Change Project included a 1994 shoreline developed from orthographic aerial photographs. Along much of the south coast of Falmouth, the three most recent shorelines available from the MCZM dataset are 1938, 1975, and 1994. Based on the maps, the long-term shoreline change rates for this stretch of the Falmouth coast (from the mid-1800s to 1994) were less than 0.5 ± 0.4 feet per year of erosion, indicative of a stable shoreline.

A recent report published by the Falmouth Coastal Resources Working Group (CRWG, 2003), a citizens group focused on long-term management of the Falmouth shoreline, used the updated MCZM shoreline data set to analyze the shoreline in the region of Oyster Pond, i.e. from Nobska Point to the Waquoit Bay jetties. They determined that recent shoreline change in this region averaged about 2.4 feet of erosion annually from 1975 to 1994 (or about 46 feet of landward movement over this time period). An erosion rate of this magnitude would suggest

significant coastal erosion and the associated longshore transport of beach-derived sediments. For the inlet to Oyster Pond, the large littoral drift indicated by the shoreline retreat rate would be expected to cause severe shoaling problems. It was therefore necessary to gain clarity as to the rate of shoreline retreat in this area.

Analysis of the rate of shoreline retreat in the region of Oyster Pond in the CRWG report appears to contradict much of the available historical data and is uncharacteristic of south-facing shorelines in the quiescent wave environments of Vineyard Sound. For example, the estimated recent erosion rate for Falmouth from the MCZM data set (1975 to 1994, 2.4 feet yr^{-1}) is nearly identical to the long-term erosion rate reported for the bluffs along the Cape Cod National Seashore of 2.54 feet yr^{-1} (Geise and Aubrey 1990). Unlike the east facing bluffs along the Cape Cod National Seashore, Falmouth's south coast is not exposed to open Atlantic Ocean wave conditions and the erosional forces associated with that environment. In addition, many of the groins and jetties constructed between the early 1900s and the mid-1950s do not extend 50 feet beyond the existing high water line; therefore, these groins would have been completely buried in the beach in the mid-1970's according to the shoreline change data set utilized by the CRWG (the two most recent shoreline available from the Massachusetts Coastal Zone Management shoreline change information). Since this is unlikely it was necessary to examine the shoreline data itself. A review of shoreline data indicated that indeed, the groins were not buried during this period. Therefore, the statewide MCZM data set appears to have insufficient accuracy for evaluating recent shoreline change along Falmouth's south coast. Additional information is required before a long-term coastal management plan can be developed.

To improve the Town's ability to manage their coastal resources, some of the other shortcomings of this recent coastline analysis are presented below. This is provided for clarification and to help prevent potential misinterpretation of regional coastal processes, which might result in improper decisions regarding long-term coastal management:

- The Falmouth CRWG report might be interpreted as suggesting that the analytical error of the shoreline change analysis for all time periods is 0.4 feet per year; however, this is in disagreement with the technical report for the shoreline change analysis (Thieler, et al., 2001). Specifically, the technical report indicates shoreline position errors of ± 8.5 meters (± 28 feet) exist for each data set. For the 1975 to 1994 shoreline change predictions, the root-mean-square error (RMS error) would be approximately ± 2.1 feet per year, not ± 0.4 feet per year (the ± 0.4 feet per year error is only appropriate for the entire time period from the mid 1800s to 1994). This misinterpretation of the errors associated with shoreline change predictions would incorrectly indicate that much of the measured shoreline change between 1975 and 1994 was actual shoreline migration, rather than error associated with the limits of the analysis technique. In general, the error in shoreline change rate predictions is higher for short time periods. Therefore, if the shorelines were properly evaluated, the recent 1975 to 1994 shoreline change would be correctly presented as averaging -2.4 feet ± 2.1 feet along the south coast of Falmouth. The potential error in this short-term analysis is nearly identical to the observed shoreline change.
- Construction of shore protection structures along the Falmouth shoreline was not limited to the time period of the 1930s to 1960s. Structures that existed at the time of the 1938/1948 shoreline included numerous groins between Nobska Point and Trunk River, the Old Stone Dock groins along Shore Drive, the

Falmouth Harbor jetties, numerous groins and seawalls between Great and Bournes Ponds, and the Waquoit Bay jetties. Once constructed, these structures immediately altered the longshore transport of sediments along the south coast of Falmouth. To evaluate how the Falmouth shoreline has responded to the existence of coastal engineering structures, a more appropriate time period to evaluate is from 1938/1948 to the most recent shoreline available, not the 1975 to 1994 time period selected for the analysis in the CRWG report.

As the Town of Falmouth is moving forward on coast issues based upon shoreline information (and this effects estuarine management options), a review of the existing shoreline data sets was performed to address the above concerns. As part of the review process, recent imagery was downloaded from the MassGIS website and these readily available aerial photographs were compared to assess the horizontal control. In addition, the interpreted shoreline data was provided by Massachusetts Coastal Zone Management (MCZM). The 2001 aerial photography was flown in April, 2001; the 1994 aerial photos were flown in September/October 1994. To evaluate the horizontal control of the two aerial photograph sets, a differential GPS was utilized to locate a series of common features visible on both orthophoto sets. This analysis indicated that horizontal control issues exist for both the 1994 and 2001 orthophotos; however, the errors appear to fall within the acceptable range of ± 3 meters (± 10 feet) for control points (see Anders and Byrnes, 1991 or Crowell, et al., 1991 for more information).

In addition to horizontal control, interpretation of the shoreline from aerial photographs also can lead to non-random errors regarding mapped shoreline positions. Due to the poor quality of the 1994 Falmouth orthophotos, interpretation of the shoreline from these images appears to be a problem. For Falmouth, it appears very difficult to select a high water shoreline from the 1994 imagery, primarily because the orthophotos appear overexposed. The 2001 aerial photography is of much higher quality, where the high water shoreline is typically discernable on the beach. Based on a cursory review of the 1994 shoreline overlaid on both the 1994 and 2001 orthophotos, it appears that incorrect identification of the high water line from the 1994 photographs caused an over-prediction of recent shoreline erosion rates. This conclusion is further supported by a comparison of the 1994 shoreline interpreted from the orthophotos and a 2004 shoreline determined using a differential GPS (survey described below). A cursory analysis of the 1994 and 2004 shorelines indicated an apparent shoreline *accretion* in the region of Oyster Pond, between the shoreline 2500 ft southwest of the Trunk River jetties and Falmouth Harbor. Since there does not appear to be a recent large-scale sediment source that would be responsible for an accreting shoreline, the most likely explanation is a misinterpretation of the 1994 shoreline, due to the poor quality of the orthophotos. Whatever the cause, the 1994 shoreline is suspect and should not be used for shoreline change analyses.

Excluding the 1994 shoreline, three outer coast shoreline surveys were available for quantifying historical shoreline change between the area southwest of the Trunk River jetties and Falmouth Harbor during the time period from 1938 to 2004. Available data layers for this time period include 1938, 1975, and 2004. The 1938 shoreline survey was interpreted from aerial photography by the U.S. Coast and Geodetic Survey (USC&GS; predecessor to NOS) and vector data were provided online at the Shoreline Data Explorer website (http://www.ngs.noaa.gov/newsys_ims/shoreline/index.cfm). This 1938 shoreline was used in favor of the 1938 shoreline currently in the MCZM database, since the horizontal control for the NOAA digitized shoreline appears to be more accurate. It should be noted that both 1938 shorelines (MCZM and NOAA) were digitized from the same source. The 1975 shoreline was provided in digital format by the Massachusetts Coastal Zone Management (MCZM) Office, as

part of the Massachusetts Shoreline Change project. Digital shoreline data for 1975 were digitized and assembled from aerial photographs by previous investigators (Theiler et al., 2001). The 2004 shoreline survey was developed using a Trimble Pro/XR differential GPS (Applied Coastal personnel). This shoreline was added to the data set because of the concerns associated with the existing 1994 shoreline.

Digital data were reviewed for accuracy and shoreline structure consistency. A review of metadata provided by MCZM regarding the quality of the 1975 data set indicated that the accuracy of the data was relatively low. This information, in combination with a review of the digital data set required that the data be excluded from the analysis. As such, the overall time period (1938 to 2004) was used to represent shoreline change conditions for this study. This 66-year span effectively represents the period of time that the south coast of Falmouth has been influenced and/or governed by coastal engineering structures.

When determining shoreline position change, all data contain inherent errors associated with field and laboratory compilation procedures. These errors should be quantified to gauge the significance of measurements used for research/engineering applications and management decisions. Table V-1 summarizes estimates of potential error associated with shoreline data sets used for this study. Because individual errors are considered to represent standard deviations, root-mean square error estimates are calculated as a realistic assessment of combined potential error. Using these estimates, the total root mean square (RMS) estimate for the 1938 to 2004 time period is ± 29 feet, or approximately 0.4 ft/yr.

Shoreline change calculations were made at 30-meter intervals along the outer coast between Falmouth Harbor and Menauhant Beach, MA using the Automated Shoreline Analysis Program (ASAP) for ArcGIS 8.3, using the 1938 to 2004 surveys. Shore-normal transects were developed using average shoreline angles determined at each analysis point. All transects used for determining change rates were visually inspected to ensure suitability for analysis and shoreline structure avoidance.

Shoreline change calculated between 1938 and 2004 showed a relatively stable shoreline for the majority of the southern coast of Falmouth. During this time interval, change rates ranged from about -1.37 ft/yr to +0.41 ft/yr, with an average rate over the study area of about -0.40 feet/yr, where change denoted with a minus represents erosion and change denoted with a plus represents accretion. Maximum erosion rates for the study area were recorded immediately west of the public facilities at Surf Drive Beach, while the most stable and/or accreting portion of the beach for this time interval was observed adjacent to the Old Stone Dock at Surf Drive Beach, as well as immediately west of the Falmouth Harbor entrance. The change transects and data distribution for this time interval are shown in figures V-5 and V-6, respectively. Overall, 96% of shoreline change calculated within the study area during this time period ranged between -1.0 and 1.0 ft/yr. The average shoreline change between Trunk River and Falmouth Harbor appears to be slightly erosional; however, the magnitude of shoreline recession is actually smaller than the error estimates associated with the shoreline datasets (± 0.44 ft/yr).

Table V-1. Estimates of potential error associated with shoreline position surveys.	
Cartographic / Interpretation Errors (1938 Shoreline Survey)	
Inaccurate location of control points on map relative to true field location	up to ± 10 ft
Placement of shoreline on map	± 16 ft
Line width for representing shoreline	± 10 ft
Digitizer error	± 3 ft
Operator error	± 3 ft
Delineating high-water shoreline position	± 16 ft
GPS Survey Errors (2004 shoreline survey)	
Delineating high-water shoreline	± 10 ft
Total Potential RMS Error Between 1938 and 2004	± 28.8 ft (± 0.44 ft/yr)
Sources: Shalowitz, 1964; Ellis 1978; Anders and Byrnes, 1991; Crowell et al., 1991.	

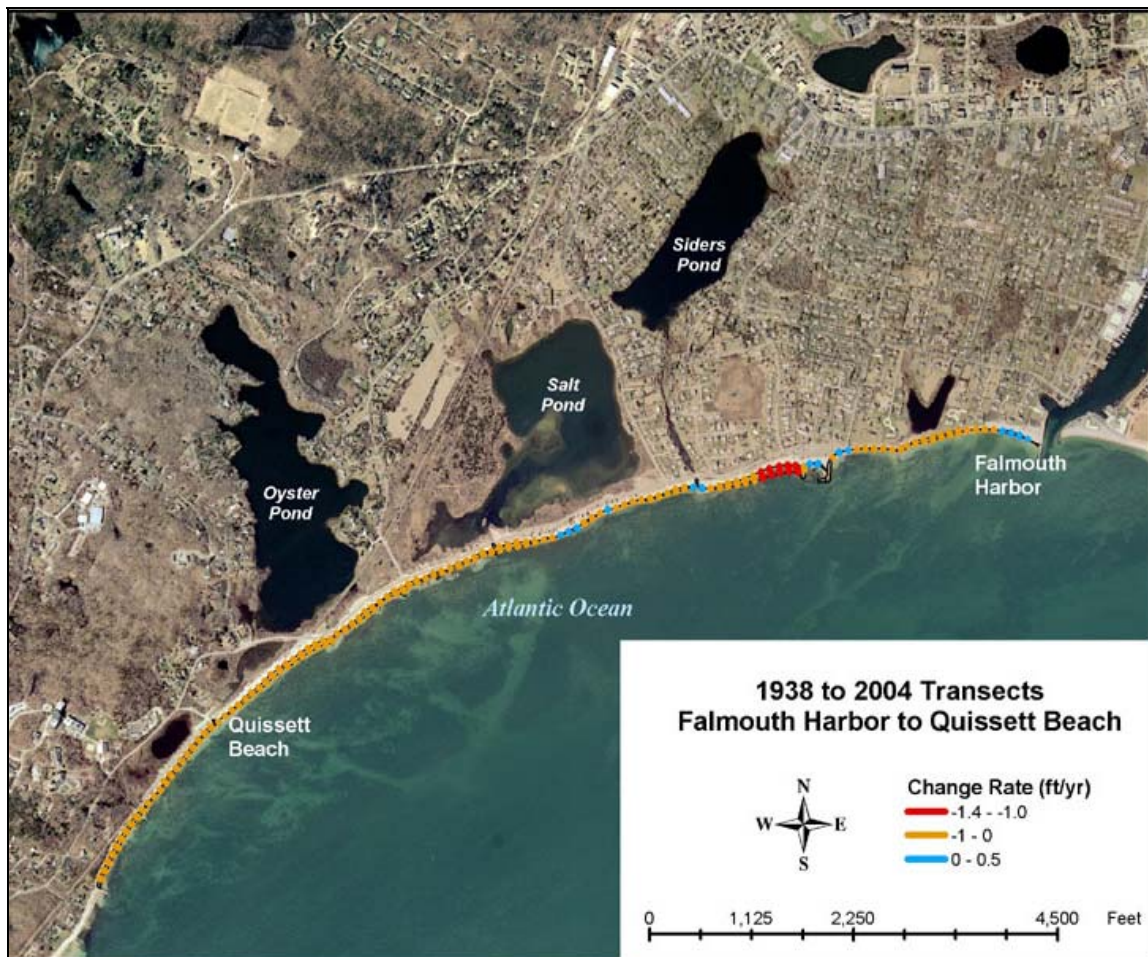


Figure V-5. The 2001 aerial photograph showing scaled transects that indicate computed shoreline change rates between 1938 and 2004.

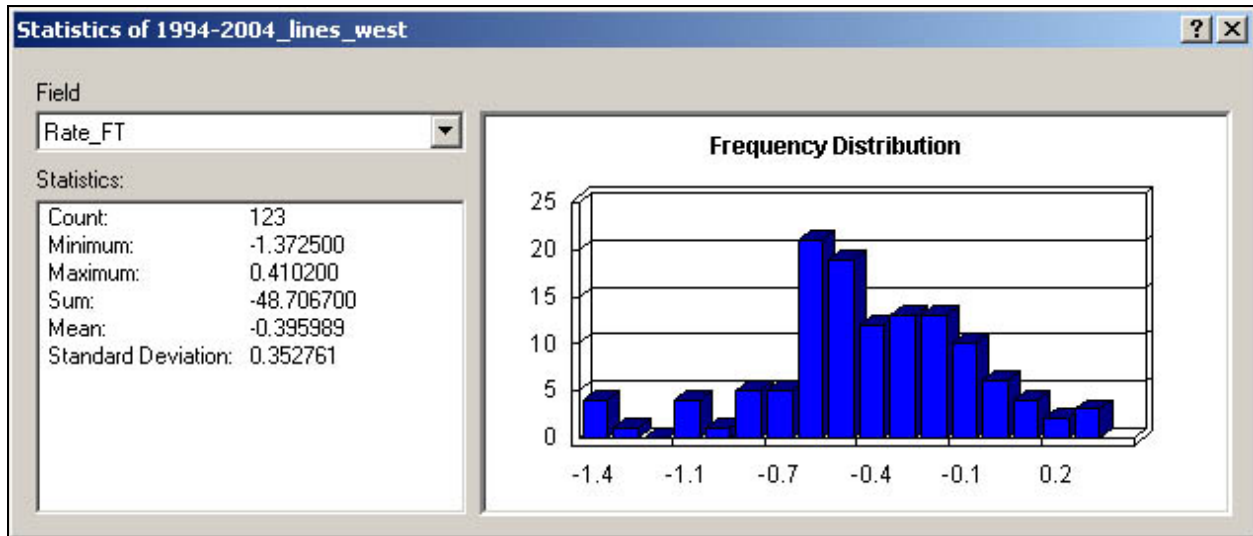


Figure V-6. Shoreline change data distribution calculated along the south coast of Falmouth (from 2500 ft southwest of the Trunk River jetties to Falmouth Harbor) between 1938 and 2004.

V.2.3 Inlet Management Implications

For the relatively small tidal inlet to Oyster Pond (Trunk River), the influence of shoreline change and the related longshore sediment transport rates directly influence long-term stability. Since observations did not suggest a consistent longshore sediment transport direction, the southwest jetty was shortened in 2000 to match the length of the northeast jetty. Historic “plugging” of the inlet by littoral drift historically occurred during severe easterly storm events. After the 2000 reconstruction of the jetties, maintenance dredging of the main inlet channel has been reduced. The lack of inlet maintenance requirements is consistent with the other observations regarding local coastal processes: low natural littoral transport and minor historic shoreline change. However, for the Oyster Pond embayment, it is critical to maintain control of tidal inflows at the salinity control structure/fish weir. Periodic accumulations in upper reach of the Trunk River or at the mouth of the Lagoon need to be removed to maintain flows.

V.3 HYDRODYNAMIC FIELD DATA COLLECTION AND ANALYSIS

Accurate modeling of system hydrodynamics is dependent upon measured conditions within the estuary for two important reasons:

- To define accurately the system geometry and boundary conditions for the numerical model
- To provide ‘real’ observations of hydrodynamic behavior to calibrate the model results

System geometry is defined by the shoreline of the system, including all coves, creeks, and marshes, as well as accompanying depth (or bathymetric) information. The three-dimensional surface of an estuary must be accurately mapped, since the resulting hydrodynamic behavior is strongly dependent upon features, such as channel widths and depths, sills, marsh elevations, and inter-tidal flats. Hence, this study included an effort to collect bathymetric information in the field.

Pressure sensors were installed at selected interior locations to measure variations of water surface elevation along the length of the system (gauging locations are shown in Figure V-3). These measurements are used to calibrate and verify the model results, and to assure

that the dynamics of the physical system are properly simulated in the model. The pressure sensor data record from the Lagoon section of Trunk River was utilized as the source of the boundary condition for the numerical model.

V.3.1. Bathymetry

Bathymetry data (i.e., depth measurements) for the hydrodynamic model of the Oyster Pond system was assembled from a recent boat based hydrographic survey supplemented with land based spot measurements. This survey was executed specifically as part of the Massachusetts Estuaries Project analysis..

The hydrographic survey of August 2003 was designed to cover the entire main basin of Oyster Pond. The survey was conducted from a canoe with an installed 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. This canoe-based bathymetry set-up is shown in Figure V-7. As the canoe was paddled 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 (latitude/longitude).

The raw measured water depths were merged with water surface elevation measurements to determine bathymetric elevations relative to the NGVD 1927 vertical datum. Once rectified, the finished processed data were archived as 'xyz' files containing x-y horizontal position (in Massachusetts State Plan 1983 coordinates) and vertical elevation of the bottom (z). These xyz files were then interpolated into the finite element mesh used for the hydrodynamic simulations. The tracks followed by the canoe rig during the bathymetry survey are presented in Figure V-7.

V.3.2 Tide Data Collection and Analysis

The hydrodynamic analysis required for this study of Oyster Pond utilizes tide data collected during periods that represent well the range of hydrologic conditions that exist in this system. Data are available from a stormy winter period in March 2001, during which four storm events occurred. Significant storm surge levels were observed during this first measurement period. The other data collection period associated with the August 2003 bathymetry survey, represents a quiescent period, where water levels were much less affected by meteorological events.

Variations in water surface elevation were measured at two stations within the Oyster Pond system (see Figure V-8), and at a station in Vineyard Sound. Stations were located in the Lagoon (T2) and in the Oyster Pond main basin. For the first winter/sprint 2001 deployment, the gauge in Oyster Pond was affixed to a screw anchor placed at the southern end of the pond. The offshore gauge was attached to a small pier fronting Salt Pond (northeast of the Trunk River inlet). For the second deployment in summer 2003, the gauge was affixed to the dock at Spore Gardens, which is approximately the north-south midpoint of the pond. During this second deployment the offshore gauge was deployed from the WHOI pier 1700 ft southwest of the Trunk River inlet.



Figure V-7. The Oyster Pond weir and canoe set-up for the bathymetry survey of the Pond in August 2003 (photograph credit: Sean Kelley).

For the first deployment in 2001, Temperature/Depth Recorders (TDRs) were deployed at each gauging station in early March, and recovered in early April. The purpose of this initial gauge deployment was to monitor the performance of the new weir during a northeast storm. The second data collection period was from the end of July to mid September 2003. The purpose of the second deployment period was to gather water level data to rectify bathymetry data collected that summer and also monitor the system during a period that was more representative of the design conditions of the weir.

The tide records from both deployment periods were corrected for atmospheric pressure variations and then rectified to the NGVD 27 vertical datum. Atmospheric pressure data, available in one-hour intervals from the NDBC Buzzards Bay C-MAN platform, were used to pressure correct the raw tide pressure data record. Final processed tide data from stations used for this study are presented in Figure V-9 for the complete 2001 deployment period and in Figure V-10 for the complete 2003 deployment.

The typical tide data analysis that is often performed on time series water level measurements was performed only for the data records collected offshore in Vineyard sound. This is due to the obvious fact the water levels in the Lagoon and Oyster Pond do not respond in a strictly tidal manor, and therefore a harmonic analysis of this gauge records would not be appropriate.

For the offshore records, a tidal harmonic analysis was performed mostly to determine the relative influence of non-tidal effects during the two gauging periods. It can be clearly seen in the plot of the spring 2001 data (Figure V-9) that storms passing through the area of the Pond had a great influence on offshore water levels. With the harmonic analysis of these tide records, it is possible to quantify precisely the magnitude of the storm surge. Tide records

longer than 29 days are necessary to complete a harmonic analysis of a tide record using more than just a few constituents. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The amplitudes and phase of 23 tidal constituents, with periods between 4 hours and 2 weeks, result from this procedure. The observed tide is therefore the sum of an astronomical tide component and a residual atmospheric component. Astronomical tide in turn is the sum of several individual tidal constituents, each with a particular amplitude and frequency. For demonstration purposes a graphical example of how these constituents add together is shown in Figure V-11.

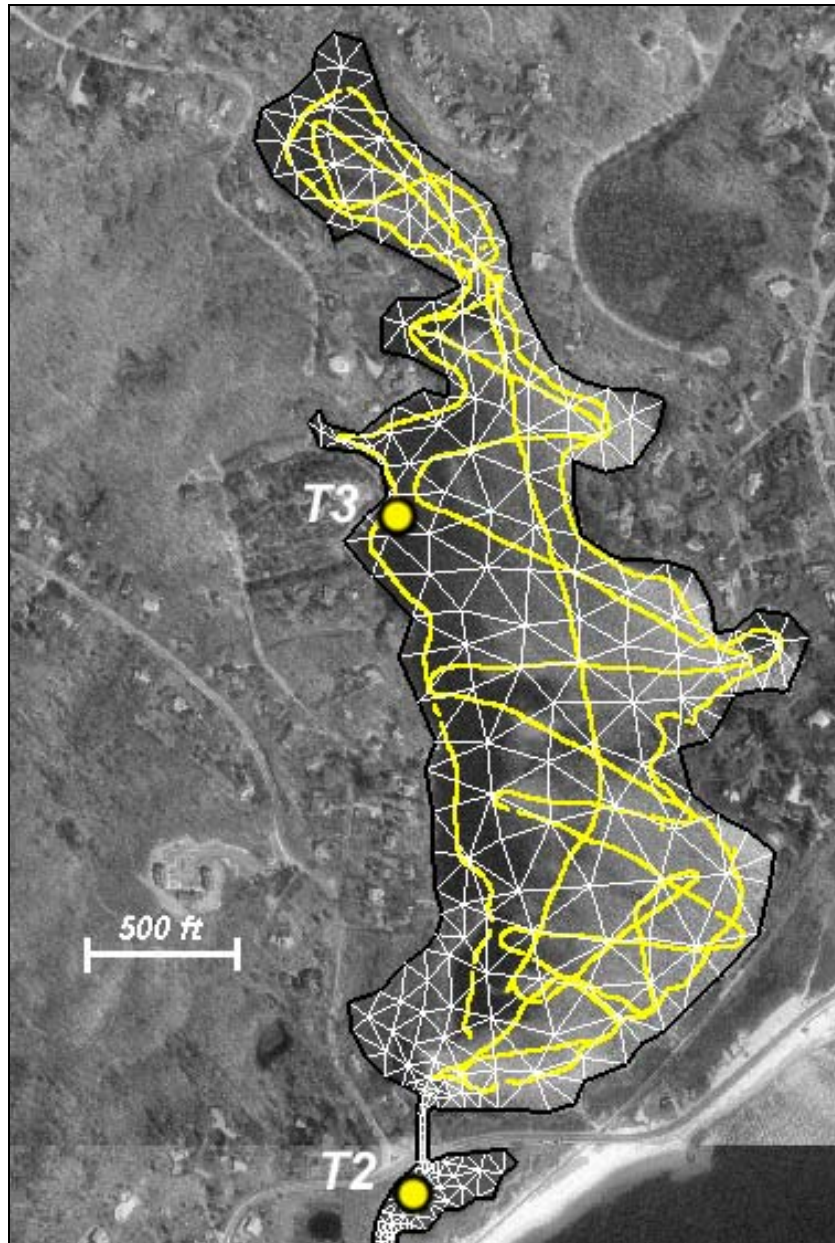


Figure V-8. Bathymetry survey lines (yellow) followed by canoe in Oyster Pond. Tide gauge stations in the Pond and Lagoon are indicated by yellow dots T2 and T3. The final RMA2 grid used for hydrodynamic and water quality modeling is also shown as the white triangular mesh.

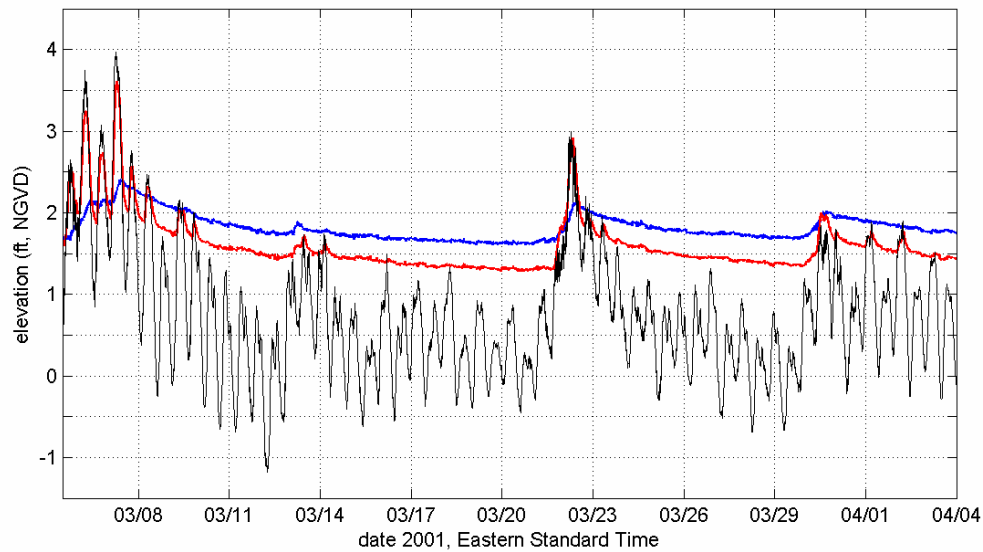


Figure V-9. Tide records from March 2001 deployment period for Oyster Pond. Captured in the data are four storm events, at March 7th, 14th, 23rd, and 30th.

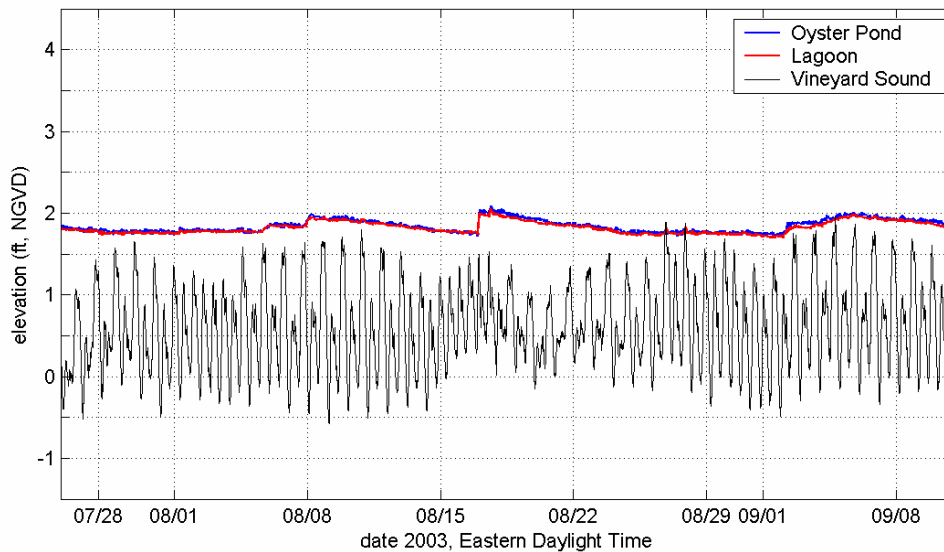


Figure V-10. Oyster Pond tides collected during a quiescent summer period in 2003. The bathymetry survey of Oyster Pond was performed August 25.

Table V-2 presents the amplitudes of eight significant tidal constituents from the summer 2003 gauging period. The M_2 , or the familiar twice-a-day lunar semi-diurnal, tide is the strongest contributor to the signal with an amplitude of 0.5 feet in Vineyard Sound. The range of the M_2 tide is twice the amplitude, or about 1.0 feet. The diurnal (once daily) tide constituents, K_1 (solar) and O_1 (lunar), possess amplitudes of approximately 0.24 and 0.23 feet respectively. The N_2 tide, a lunar constituent with a semi-diurnal period, is only slightly small than the main diurnal constituents with an amplitude of 0.17 feet. The M_4 tide, a higher frequency harmonic of the M_2 lunar tide (twice the frequency of the M_2), results from frictional dissipation of the M_2 tide in shallow water. Typically the M_4 represents a small fraction of the total tide amplitude.

However, in Vineyard Sound, the M_4 and M_6 amplitudes are large compared to the M_2 (i.e., 36% and 12% of the M_2 amplitude, respectively). As a result, they have more influence on the shape of the tide signal than is observed in most other corners of the world's oceans. This leads to the characteristic double high tide observed in Vineyard Sound, which is sometimes referred to as an agger (NOS, 1989).

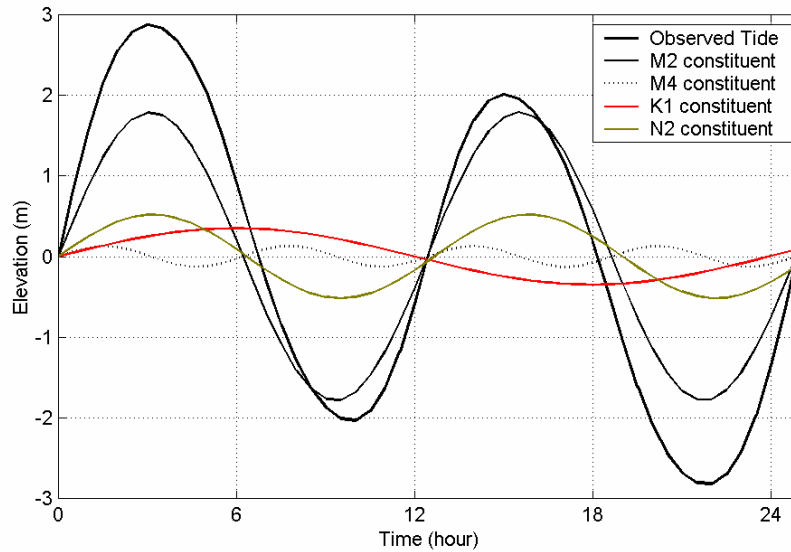


Figure V-11. Example of observed astronomical tide as the sum of its primary constituents. In this example the observed tide signal is the sum of individual constituents (M_2 , M_4 , K_1 , N_2), with varying amplitude and frequency.

Table V-2. Tidal Constituents for stations in the Lagoon Pond System, from data collected July 25, 2003 14 to September 11, 2003.

	AMPLITUDE (feet)							
	M_2	M_4	M_6	S_2	N_2	K_1	O_1	M_{sf}
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Vineyard Sound 2003	0.50	0.18	0.06	0.10	0.17	0.24	0.23	0.05

With the 23 harmonic constituents computed, it is possible to determine the tidal residual. The residual signal is computed as simply the difference between the measured tide and the astronomical component of the observed tide, where the astronomical tide is reconstituted using the individual tidal constituents. Figures V-12 and V-11 show comparisons between observed and astronomical tides and the computed residual. In Figure V-12, the residual analysis shows four distinct storm events in the record from the spring of 2001. During both the storms of March 6-7 and March 23 the maximum storm surge was more than two feet above the astronomical tide. The first storm occurred during a spring tide cycle, and caused the ocean surface to ride to over 4 feet NGVD. A quantitative indicator that this period in 2001 was stormy is the relative energy content of the residual signal compared to the total observed signal, determined by computing the variance of the tide signal components (astronomical *versus* residual). The variance of the residual signal was computed to be 66% of the total measured tide signal, which indicates that atmospheric effects had an overwhelming influence on the Vineyard Sound tides during this time period.

During the summer 2003 deployment period, the tidal residual was a much smaller contributor to the total observed tide. For this tranquil period, the residual signal contributed on 13% of the variance observed in the measured tide signal. This can be seen directly in the close correspondence of the observed and astronomical tides plotted in Figure V-13 for this deployment period. The computed residual is typically less than 0.5 ft for this data set.

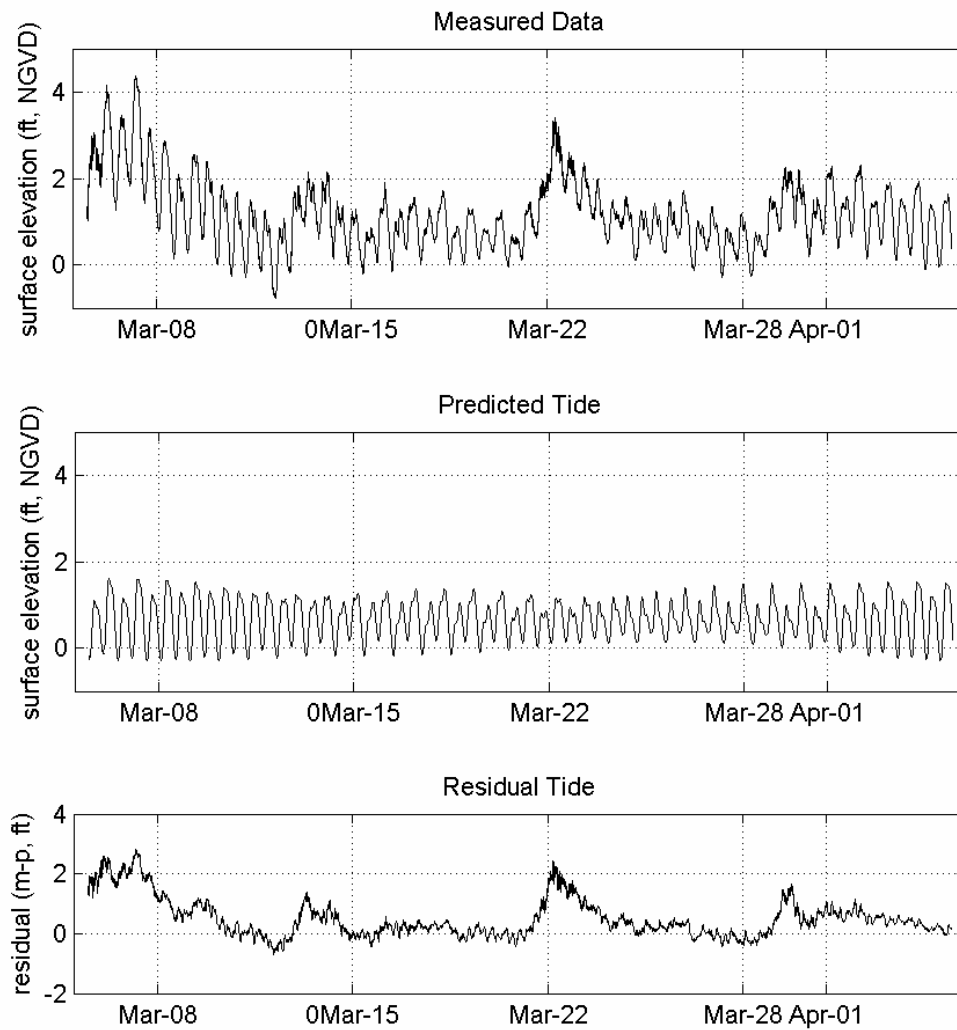


Figure V-12. Results of the harmonic analysis and the separation of the tidal from the non-tidal, or residual, signal measured offshore Oyster Pond in Vineyard Sound during the 2001 gauge deployment.

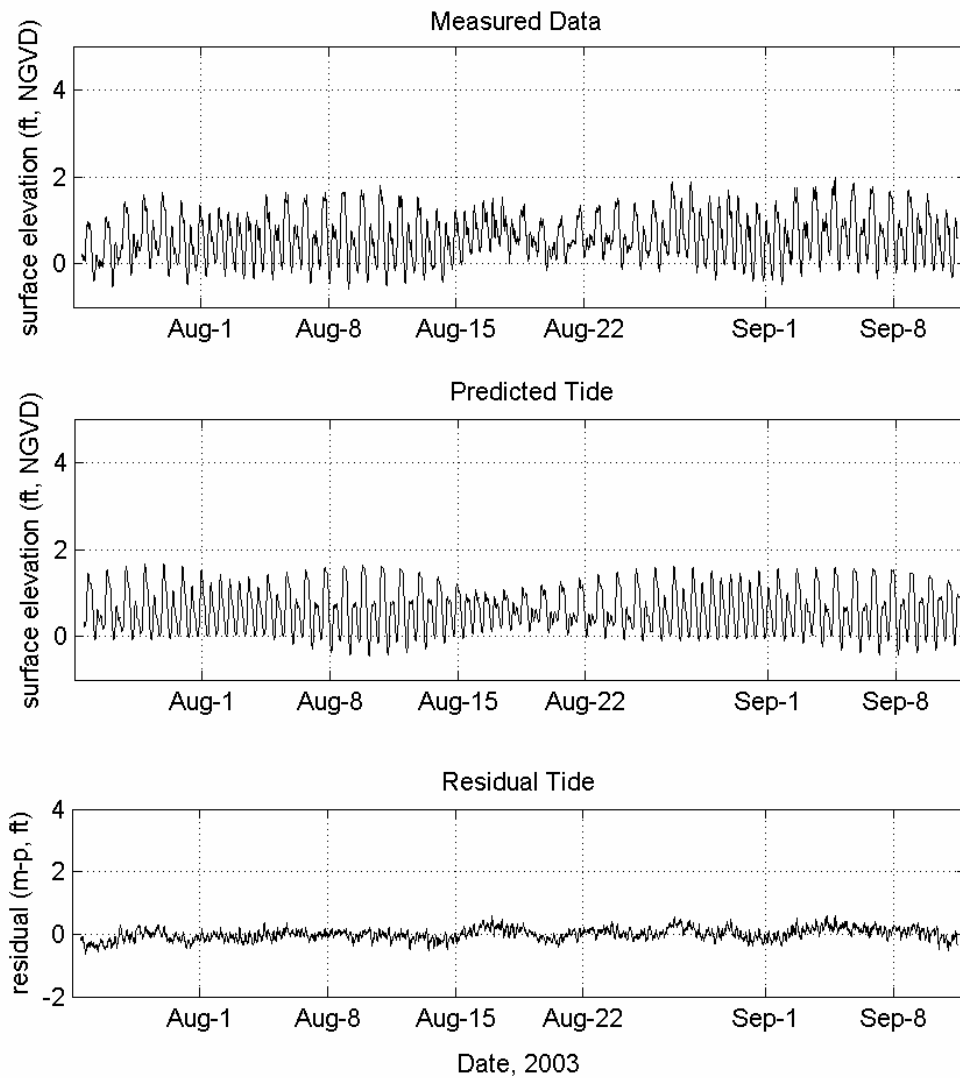


Figure V-13. Results of the harmonic analysis and the separation of the tidal from the non-tidal, or residual, signal measured offshore Oyster Pond in Vineyard Sound during the 2003 gauge deployment.

In addition to the in-depth harmonic analysis of the offshore data records, a simpler analysis of the 2003 offshore tide record was undertaken to determine the elevation of standard tide datums. These computed datums are presented in Table V-3. This analysis was not performed using the 2001 offshore data due to the prevalent storm bias in that data set. The Mean Higher High Water (MHHW) and Mean Lower Low Water (MLLW) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. The Mean Tide Level (MTL) is simply the mean of MHW and MLW. The tides in Buzzards Bay are semi-diurnal, meaning that there are typically two tide cycles in a day. There is usually a small variation in the level of the two daily tides. This variation can be seen in the differences between the MHHW and MHW, as well as the MLLW and MLW levels. The computed datums for Lagoon Pond and Vineyard Sound compare well to similar datums

computed for the south shore of Falmouth using a 38-day record from 1999 (MTL 0.8 ft, MHW 1.7 ft, MLW 0.0 ft NGVD). Difference can be attributed to survey errors (absolute elevation errors caused by survey and/or benchmarks) and also to variations in the tide range that are occur in cycles greater than the bi-monthly spring to neap cycle.

Table V-3. Tide datums computed from tide record collected in Vineyard Sound July 25, 2003 14 to August 25, 2003. Datum elevations are given relative to NGVD 27.	
Tide Datum	Offshore (feet)
Maximum Tide	1.8
MHHW	1.4
MHW	1.2
MTL	0.5
MLW	-0.1
MLLW	-0.2
Minimum Tide	-0.6

V.4 HYDRODYNAMIC MODEL DEVELOPMENT

Since it is not possible with the RMA suite of models to directly simulate the culvert/weir hydraulic control system located in Oyster Pond, an alternative modeling approach was developed. Though the RMA-2 model is able to simulate simple weirs and culverts, the Oyster Pond weir and culvert system presents modeling challenges that are beyond the capabilities of the intrinsic RMA-2 representation of these types of flow control structures. Therefore, a computer model independent of RMA-2 was used to simulate the culvert/weir system. Using this weir model, time varying boundary conditions were developed for RMA-2 model runs of the main basin of Oyster Pond, up to the weir.

V.4.1 Weir model development

The weir model was coded to include both a culvert section and the weir. The particular dimensions and elevations of the Oyster Pond culvert and weir sections are inputs to the model. Once the weir model is calibrated, it can be used to simulate alternate weir scenarios. Specifically, it can be used to evaluate how the elevation of the adjustable weir section affects salinity or nitrogen concentrations in the Pond.

Widely available formulations (e.g., CERM, 2001 and Henderson, 1966) for weir flow serve as the basis for the weir model code. The *Frances weir equation*, based on the Bernoulli equation (energy conservation in non-compressible fluid flows), relates the flowrate of water over a weir to the elevation of the upstream water surface above the weir crest as

$$Q = \frac{2}{3} C_1 b \sqrt{2gH}^{\frac{2}{3}}$$

where Q is the flowrate across the weir, b is the cross-stream width of the weir crest, g is the gravitational constant, H is the upstream water surface elevation above the weir crest, and C_1 is an empirical discharge coefficient computed using the equation

$$C_1 = \left[0.6035 + 0.0813 \left(\frac{H}{Y} \right) + \frac{0.000295}{Y} \right] \times \left(1 + \frac{0.00361}{H} \right)^{\frac{2}{3}}$$

where Y is the elevation of the weir crest above the channel bottom. The flow across the weir can be modified to account for contracted flow and also for instances where the weir crest is lower than the down stream water elevation (i.e., a submerged weir). Contracted flow occurs when the weir sides do not extend completely to the channel sides. The effect of the contraction is to reduce the flow volume through weir, and is accounted for by determining an effective width ($b_{\text{effective}}$) of the weir using

$$b_{\text{effective}} = b - 0.2H.$$

For a submerged weir, flows are computed using the rate determined using the *Frances weir equation*, but modified as

$$Q_{\text{submerged}} = Q \left[1 - \left(\frac{H_{\text{downstream}}}{H_{\text{upstream}}} \right)^{\frac{2}{3}} \right].$$

For the culvert section included in the weir model code, two different flow conditions are considered. First is the case where there is tranquil flow throughout the length of the culvert, and neither end is completely submerged. This is the prevailing hydrologic condition for the culvert between the Lagoon and Oyster Pond. The second condition considered in the weir model is for a submerged inlet and outlet. This condition occurs only infrequently, during storm events that bring about a surge in the water level of Vineyard Sound. This was the condition of the culvert at the time of the photograph shown in Figure V-10. Both flow types are described in the CERM (2001).

When the downstream water surface elevation is the factor that controls flow through the culvert, and the exit is not completely submerged, the flowrate through the culvert can be determined by the equation

$$Q = C_d A \sqrt{2g \left(h_1 + \frac{\alpha v^2}{2g} - h_3 - h_{f,1-2} - h_{f,2-3} \right)}$$

where C_d is the discharge coefficient of the culvert opening, A is the flow area at the discharge end of the culvert (some fraction of the total culvert cross-sectional area, dependent on the water depth in the culvert), h_1 is the water surface elevation at the inlet above the culvert invert (i.e., bottom), v is the flow velocity through the culvert, h_3 is the water surface elevation at the outlet above the invert, $h_{f,1-2}$ is the head loss at the inlet, and $h_{f,2-3}$ is the head loss through the culvert. For the purposes of the weir model, the velocity head term of this equation was considered to be negligible, a valid assumption considering that normal discharge through this wide culvert is of the order 1 ft³/sec, and velocities through the culvert are small (of order 0.1 ft²/sec).

The alternate culvert flow type considered in the weir model is the case where both the inlet and discharge are submerged. For this case, flow through the culvert is determined using the equation

$$Q = C_d A \sqrt{2g \left(\frac{h_1 - h_4}{1 + \frac{29C_d^2 n^2 L}{R^{4/3}}} \right)}$$

where h_4 is the water surface elevation above the culvert invert, n is the Manning's coefficient of the culvert ($n=0.024$ for large diameter culverts), L is the length of the culvert, and R is the hydraulic radius (based on the culvert diameter).

The weir model uses the equations that describe flow through both the culvert and weir to determine a time varying water surface elevation in Oyster Pond and also the flowrate into or out of the pond. Inputs include weir and culvert characteristic dimensions, the freshwater recharge rate into the pond, and surface area of the pond, and a time-varying water surface elevation boundary condition in the Lagoon. Flow rates are computed at every time step using a simple iterative technique base on mass conservation through culvert and weir. Five separate weir sub-sections are represented in the model, including the main weir opening and the smaller fish chute. Sections of the weir structure that are normally dry are also represented in the model so that their contribution to the total flow is included during infrequent times when the weir overtopped completely.

V.4.2 Application of the weir model

The weir model was calibrated using tide data collected during the northeast storm of early March 2001. This event is very useful for this purpose because water levels changes during this event are likely representative of a maximum range through which the weir and culvert would operate. Photographs presented in Figure V-14 and V-15 as well as the tide plot of Figure V-16 show examples of the extreme conditions captured in this data record. The two photographs visually document how the entire weir structure was at times completely submerged as water from Vineyard Sound flooded the Lagoon. The water surface elevation was approximately 0.1 ft higher than the crest of the weir structure (+ 3.0 feet NGVD). These pictures were taken during a period of falling offshore tide levels. The snow line around the weir channel perimeter indicated that the water level behind the weir (i.e., between the culvert and the weir) had not been higher, at least since the point in time where it had stopped snowing. Water levels in the Lagoon were at times more than a foot higher than at the weir because of the culvert between the Lagoon the Pond acted to restrict flow.

For this particular storm, most of the resulting precipitation was snow, with a total accumulation of a few inches, which means that the resulting water level changes measured in the pond are caused by seawater entering from Trunk River and the Lagoon, rather than due to rain and runoff. Also, at no point during this storm were offshore surge levels high enough to overtop Surf Drive, therefore water level changes in Oyster Pond were due mostly to seawater input over the weir.



Figure V-14. Oyster Pond weir overtopped from the Lagoon side during the early March 2001 northeast storm. At the time of the photograph, the culvert under Oyster Pond Road is completely submerged (photograph credit: Sean Kelley).



Figure V-15. Close up view of the Oyster Pond weir as it is overtopped from the Lagoon side during the northeast storm of March 6 and 7, 2001. For this photograph, left is toward the pond (photograph credit: Sean Kelley).

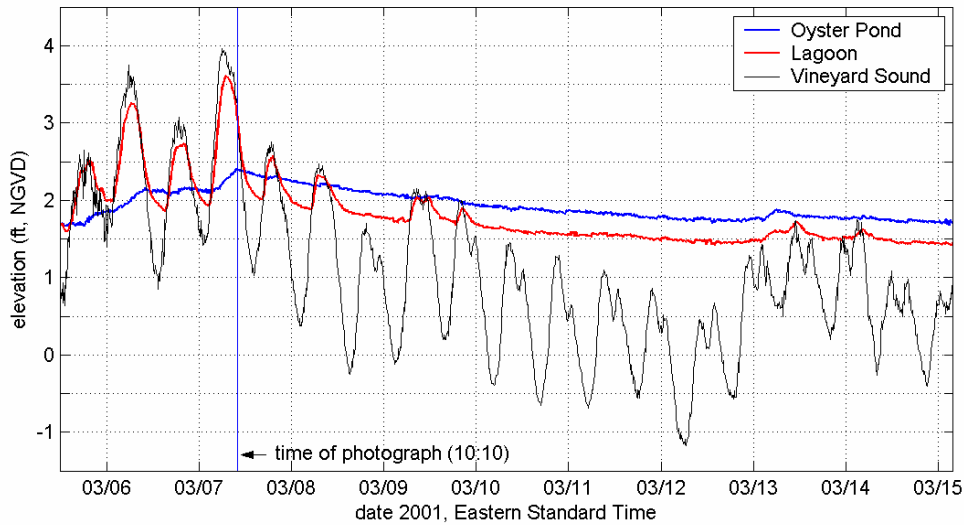


Figure V-16. Detail of the March 2001 tide record from Oyster Pond.

A comparison of model output and measured tides are presented in Figure V-17. Model input parameters specific to the Oyster Pond system are listed in Table V-4. For this 240-hour simulation, the model has very good correlation with the measured pond elevation data, with a computed R^2 correlation coefficient of 0.94, and an rms error of 0.04 ft.

Table V-4. Model input parameters used for simulations of the Oyster Pond weir.	
culvert diameter	3.2 feet
culvert length	60 feet
culvert invert elevation	-0.5 feet NGVD
Oyster Pond typical surface area	64.4 acres
culvert Manning coefficient	0.024
culvert entrance C_d	0.90
average fresh water recharge rate	1.06 ft ³ /sec

In addition to water surface elevation outputs, the weir model outputs include total flow over the weir. Model flow rate outputs for the storm simulation period are shown in Figure V-18. The maximum volume flux into the pond, at the height of the storm, was computed to be 29.9 ft³/sec. Usually, the weir experiences flow rates equal to the total freshwater recharge rate (i.e., 1.06 ft³/sec), so this storm event caused flow rates that were nearly 30 times the normal background rate. By integrating the flow rate curve, the total seawater input into Oyster Pond during this storm was computed to be 2,008,700 ft³. This input represents 8% of the normal 24,3004,000 ft³ operating volume of the Pond, (with the weir set, as it was at this time, to +1.8 ft NGVD).

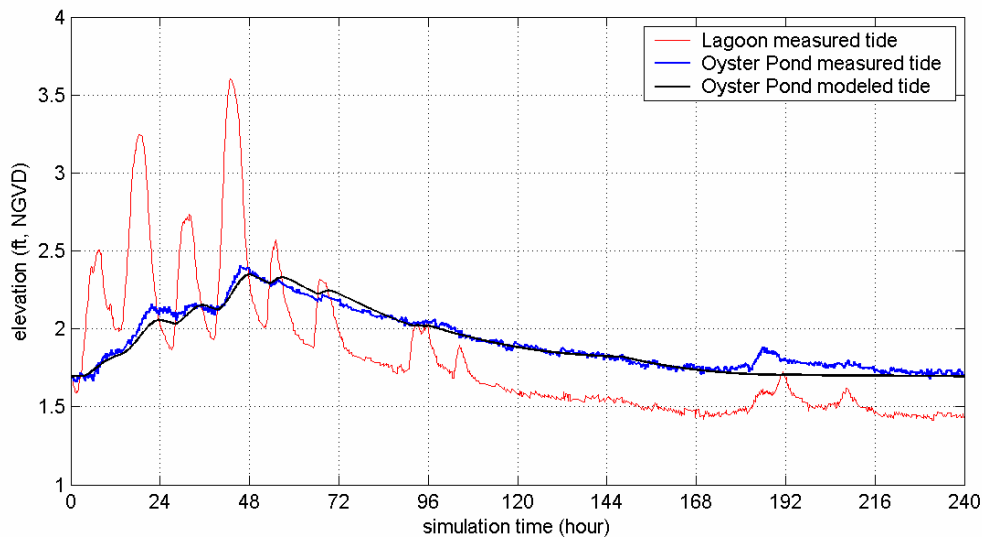


Figure V-17. Comparison of measured tide data and weir model output for a northeast storm event (R^2 correlation coefficient of 0.94, rms error of 0.04 ft). Also plotted are water level variations measured in the Trunk River lagoon, used as the boundary condition for the weir model. This simulation begins at 1200h EST March 5, 2001.

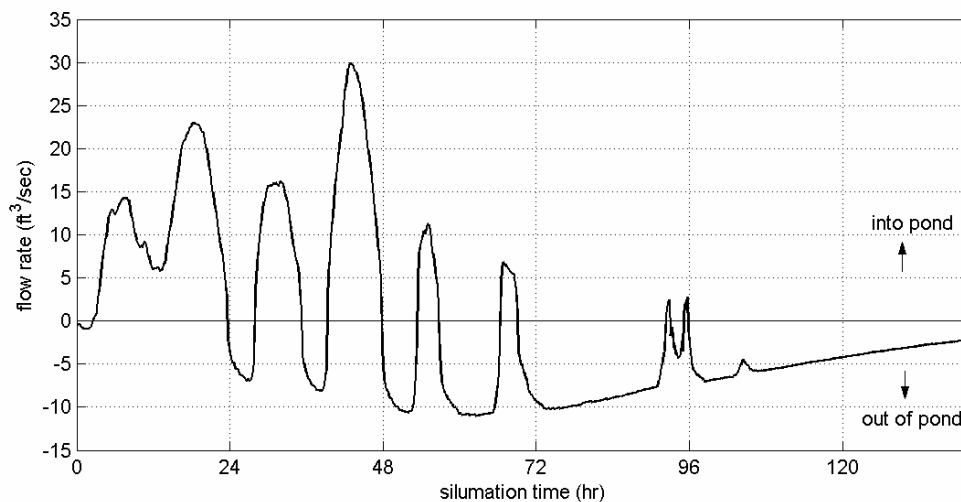


Figure V-18. Weir model output flow rate over the Oyster Pond weir during the March 2001 northeast storm event.

V.4.2 RMA2 model development

A 2-dimensional hydrodynamic model of Oyster Pond was developed using inputs of bathymetry and modeled water surface elevations determined using the weir model. This hydrodynamic model in turn is used as input into the final 2-dimensional water quality of the Pond.

The finite element mesh created for the Pond is shown in Figure V-8, and the final grid with interpolated bathymetry is shown in Figure V-19. The grid is composed of 479 quadratic finite elements (both triangular and quadrilateral elements) and 1142 computational nodes. The grid has a maximum depth of -22.9 ft NGVD, which is located in the deep area in the

southeastern region of the pond. Another deep area exists in the northernmost reach of the pond, where depths are greater than -12 ft NGVD.

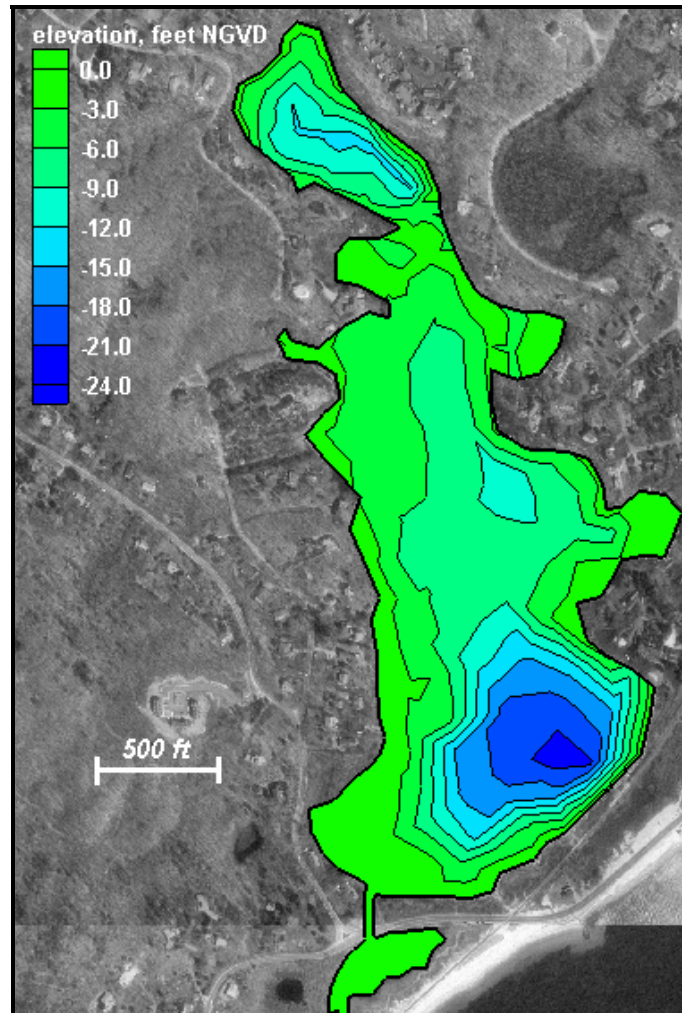


Figure V-19. Interpolated bathymetric contours of the final RMA2 computational mesh of Oyster Pond. Depth contours are relative to the NGVD 29 vertical datum, and are at 3-foot intervals.

VI. WATER QUALITY MODELING

VI.1 DATA SOURCES FOR THE MODEL

Information from several different sources were required to support the water quality modeling effort for the Oyster Pond estuary system. These sources include output from the hydrodynamics model, calculations of external nitrogen loads from the watersheds, measurements of internal nitrogen loads from the sediment (benthic flux), and measurements of nitrogen in the water column.

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayments

Extensive field measurements and hydrodynamic modeling of the Oyster Pond system were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was calibrated model output representing the transport of water within Oyster Pond. Files of node locations and node connectivity for the RMA-2V model grids were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic output for the water quality model calibration was a 45-day period in summer 2002. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 45-day spin-up period, to allow the model had reached a dynamic “steady state”, and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayments

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

VI.1.3 Measured Nitrogen Concentrations in the Embayments

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in Figure VI-1. The multi-year averages present the “best” comparison to the water quality model output, since factors of tide, temperature, rainfall, and hydraulic conditions at the inlet and weir may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data is the minimum required to provide a baseline for MEP analysis. Typically, eight years of data (collected between 1997 and 2004 with an average salinity of ~2 ppt) were available for stations monitored by SMAST in Oyster Pond.

Table VI-1. Falmouth PondWatch measured data, and modeled Nitrogen concentrations for the Oyster Pond system used in the model calibration plots of Figure VI-2. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of the separate yearly means.

Sub-Embayment	monitoring station	data mean	s.d. all data	N	model min	Model max	model average
Oyster Pond - Head	OP1	0.695	0.026	50	0.646	0.747	0.696
Oyster Pond - Mid	OP2	0.669	0.018	81	0.644	0.746	0.694
Oyster Pond - Lower, deep basin	OP3	0.705	0.157	84	0.641	0.746	0.693
Vineyard Sound	VS	0.280	0.065	196	-	-	0.280

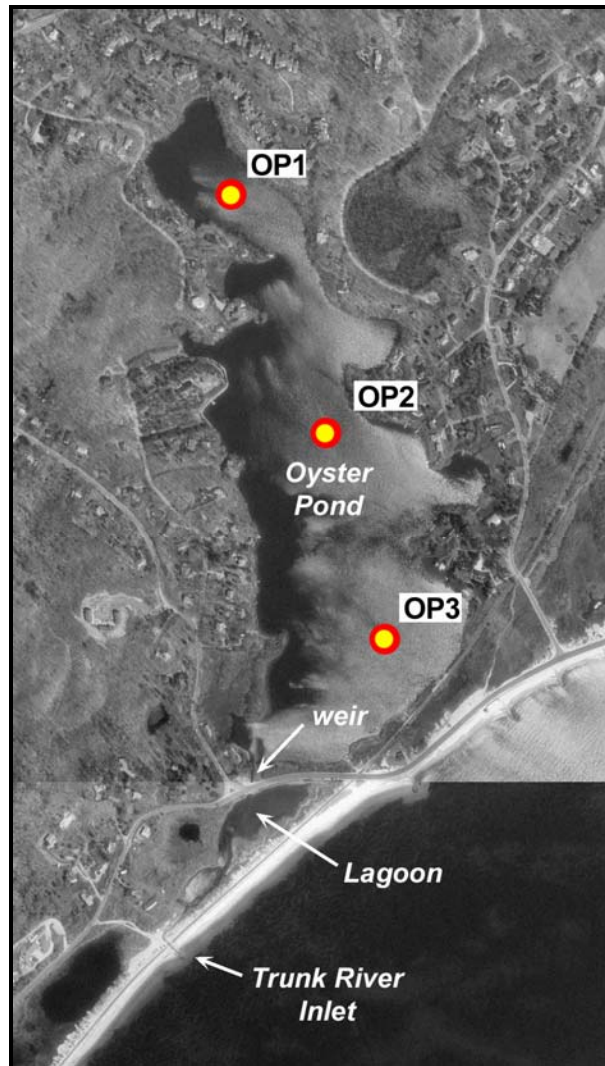


Figure VI-1. Estuarine water quality monitoring station locations in Oyster Pond. Station labels correspond to those provided in Table VI-1.

VI.2 MODEL DESCRIPTION AND APPLICATION

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of nitrogen loading in Oyster Pond. 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 in Falmouth (Howes *et al.*, 2005); Mashpee, MA (Howes *et al.*, 2004) and Chatham, MA (Howes *et al.*, 2003).

The overall approach involves modeling total nitrogen as a non-conservative constituent, where bottom sediments act as a source or sink of nitrogen, based on local biochemical characteristics. This modeling represents summertime conditions, when algal growth is at its maximum. Total nitrogen modeling is based upon various data collection efforts and analyses presented in previous sections of this report. Nitrogen loading information was derived from the watershed loading analysis (Section IV.1), the measured stream discharge (Section IV.2), as well as the measured bottom sediment nitrogen fluxes (Section IV.3). Water column nitrogen measurements were utilized as model boundaries and as calibration data. Hydrodynamic model output (discussed in Section V) provided the remaining information (tides, currents, and bathymetry) needed to parameterize the water quality model of Oyster Pond.

VI.2.1 Model Formulation

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

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

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

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

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

application, the RMA-4 model was used to predict tidally averaged total nitrogen concentrations throughout Oyster Pond.

VI.2.2 Water Quality Model Setup

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

Based on measured flow rates from SMAST and groundwater recharge rates from the USGS groundwater model, the hydrodynamic model was set-up to include the latest estimates of surface water flows from the Mosquito Creek, which has a measure flow rate of 0.044 ft³/sec (108 m³/day), which is 4.2% of the total freshwater input into the Pond (including rainfall and groundwater recharge).

For the model, an initial total N concentration equal to the average measured concentration of the pond was applied to the entire model domain. The model was then run for a simulated month-long (45-day) spin-up period. Model results were recorded only after the initial spin-up period. The time step used for the water quality computations was 10 minutes, which corresponds to the time step of the hydrodynamics input for the Pond

At the end of the spin-up period, the model was run for an additional 45-day (1080 hour) period. This long simulation period was required due to the nature of tidal exchange between the Pond and Vineyard Sound. Because the weir functions to limit tidal exchange with Vineyard Sound (in order to control salinity, primarily), a long simulation (> one week) must be performed in order to accurately determine average water quality conditions.

VI.2.3 Boundary Condition Specification

Mass loading of nitrogen into each model included 1) sources developed from the results of the watershed analysis, 2) estimates of direct atmospheric deposition, 3) summer benthic regeneration, 4) the “point source” input from Mosquito Creek. Nitrogen loads from the Oyster Pond watersheds were distributed across the sub-embayment. The combined watershed direct atmospheric deposition loads for the pond were evenly spread among grid cells in the model that formed the perimeter of the embayment. Benthic regeneration loads were distributed among another sub-set of grid cells, which are in the interior portion of the pond basin.

The loadings used to model present conditions in the Oyster 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 and from the whole system nitrogen balance in Section IV.3. The area rate (g/sec/m²) of nitrogen flux from that analysis was applied to the surface area coverage computed for each sub-embayment (excluding marsh coverages, when present), resulting in a total flux for each embayment (as listed in Table VI-2). Due to the highly variable nature of bottom sediments and other estuarine characteristics of coastal embayments in general, the measured benthic flux for existing conditions also is variable. For present conditions, some sub-embayments (e.g., Green Pond) have almost twice the loading rate from benthic regeneration as from watershed loads. In other sub-embayments with basins similar to Oyster Pond (e.g., Perch Pond), the benthic flux is relatively low or negative indicating a net uptake of nitrogen in the bottom sediments.

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. TN concentrations of the incoming water are set at the value designated for the open boundary. The boundary concentration in Vineyard Sound was set at 0.280 mg/L, based on SMAST data from the Sound (station VS). The open boundary total nitrogen concentration represents long-term average summer concentrations found within Vineyard Sound.

Table VI-2. Sub-embayment and surface water loads used for total nitrogen modeling of the Oyster Pond system, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent present loading conditions for the listed sub-embayments.			
sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Oyster Pond	4.066	0.773	-1.733
Oyster Pond Lagoon	0.293	0.027	-0.048
Mosquito Creek (surface water)	0.115	-	-

VI.2.4 Model Calibration

Calibration of the Oyster Pond total nitrogen model proceeded by changing model dispersion coefficients so that model output of nitrogen concentrations matched measured data. Generally, several model runs of each system were required to match the water column measurements. Model hydrodynamics used as an input into the water quality model represent conditions that were measured in August 2002, with the main weir section set to an elevation of 1.8 feet NGVD. Dispersion coefficient (E) values were varied through the modeled systems by setting different values of E for the main basin of Oyster Pond, for the Lagoon, and for fresh water inputs (i.e., Mosquito Creek). Observed values of E (Fischer, *et al.*, 1979) vary between order 10 and order 1000 m²/sec for riverine estuary systems characterized by relatively wide channels (compared to channel depth) with moderate currents (from tides or atmospheric forcing). Observed values of E in calmer areas typically range between order 10 and order 0.001 m²/sec (USACE, 2001). Generally, the relatively quiescent Oyster Pond system require values of E that in the lower range relative to the riverine estuary systems evaluated by Fischer, *et al.*, (1979). The final values of E used in each sub-embayment of the modeled systems were 17.0 m²/sec for the main basin of Oyster Pond and the Lagoon, and 0.5 m²/sec in the vicinity of Mosquito Creek. These values were used to develop the “best-fit” total nitrogen model calibration. For the case of TN modeling, “best fit” can be defined as minimizing the error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within each sub-embayment.

Comparisons between model output and measured nitrogen concentrations are shown in plots presented in Figure VI-2. In this plot, means of the water column data and a range of two standard deviations of the annual means at each individual station are plotted against the modeled maximum, mean, and minimum concentrations output from the model at locations which corresponds to the SMAST monitoring stations. The rms error of the model output is 0.016 mg/L compared to the measured data average concentrations.

For model calibration, the average modeled TN was compared to mean measured TN data values, at each PondWatcher water-quality monitoring station. The calibration target would fall at the modeled mean TN because the Pond experiences hardly any variation in N concentrations due to tidal exchange. This procedure is different than what is used for other estuarine systems (e.g., Great, Green and Bournes Ponds) where tidal exchange can cause wide variations in N concentrations during the course of a single tide cycle. For these well-flushing estuarine systems, the calibration target is chosen as the mid-point between the maximum modeled and averaged modeled concentration, to represent the mid-ebb concentration at the monitoring station. In addition to having little tidal variation in total nitrogen levels, Oyster Pond waters showed little horizontal variation, which contrasts with most estuaries. This is also due to the lack of significant tidal exchange in this system. In essence Oyster Pond operates as a brackish kettle pond, where wind driven circulation creates generally homogeneous nutrient distributions in surface waters.

Contour plots of calibrated model output are shown in Figure VI-3. In this figure, color contours indicate nitrogen concentrations throughout the model domain. The output in this figure shows average total nitrogen concentrations, computed using the full 45-day model simulation output period.

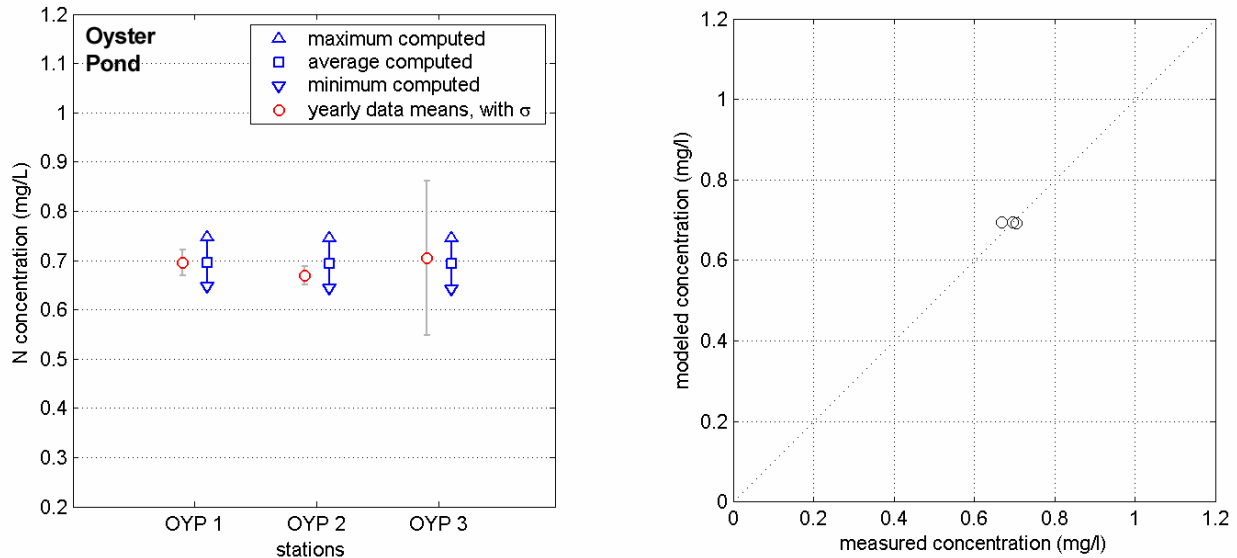


Figure VI-2. Comparison of measured total nitrogen concentrations and calibrated model output at stations in the Oyster Pond system. For the left plots, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate \pm one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line.

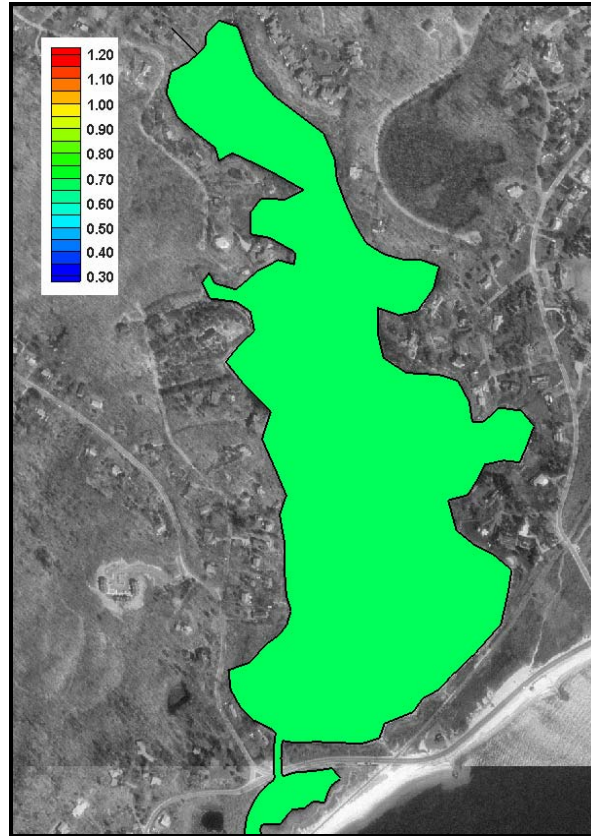


Figure VI-3. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for Oyster Pond.

VI.2.5 Model Salinity Verification

In addition to the model calibration based on nitrogen loading and water column measurements, numerical water quality model performance is typically verified by modeling salinity. This step was performed for the Oyster Pond model using salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of the Pond, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary, at the freshwater stream discharges, groundwater inputs, and net rainfall. The open boundary salinity was set at 29.6 ppt. For surface water streams and groundwater inputs salinities were set at 0 ppt. Surface water stream flow rates for the streams were the same as those used for the total nitrogen model, as presented earlier in this section. Freshwater inputs ($1.01 \text{ ft}^3/\text{sec}$) from groundwater and net rainfall (minus evaporation) were distributed evenly in each model through the use of several 1-D element input points positioned along each model's land boundary. The four groundwater inputs had dispersion coefficients set to $6.5 \text{ m}^2/\text{sec}$.

Comparisons of modeled and measured salinities are presented in Figure VI-4, with contour plots of model output shown in Figure VI-5. The rms error of the salinity models is less than 0.08 ppt. The salinity verification provides a further independent confirmation that model dispersion coefficients and represented freshwater inputs to the model correctly simulate the real physical systems.

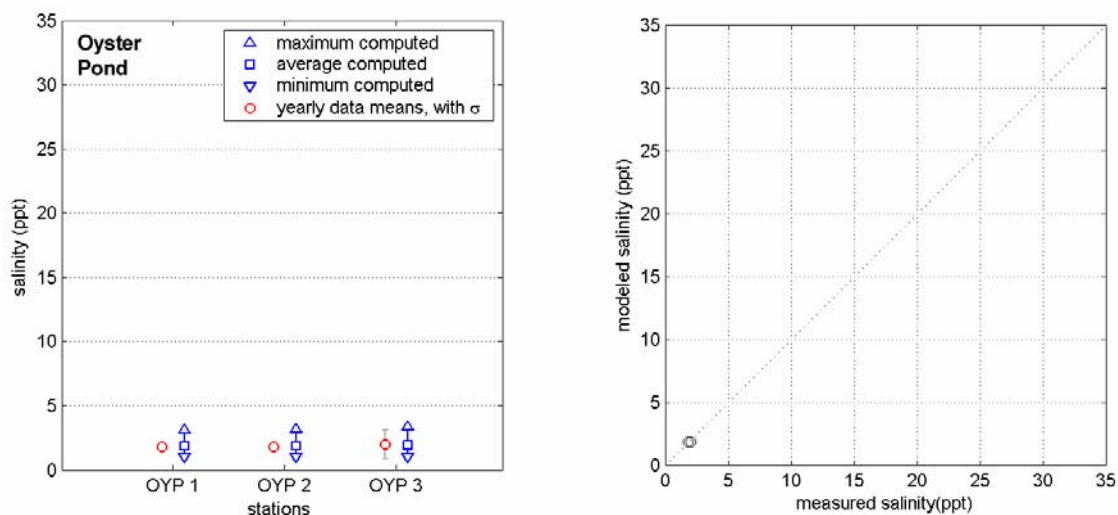


Figure VI-4. Comparison of measured and calibrated model output at stations in the Great Pond. For the left plots, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed salinity for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate \pm one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line.

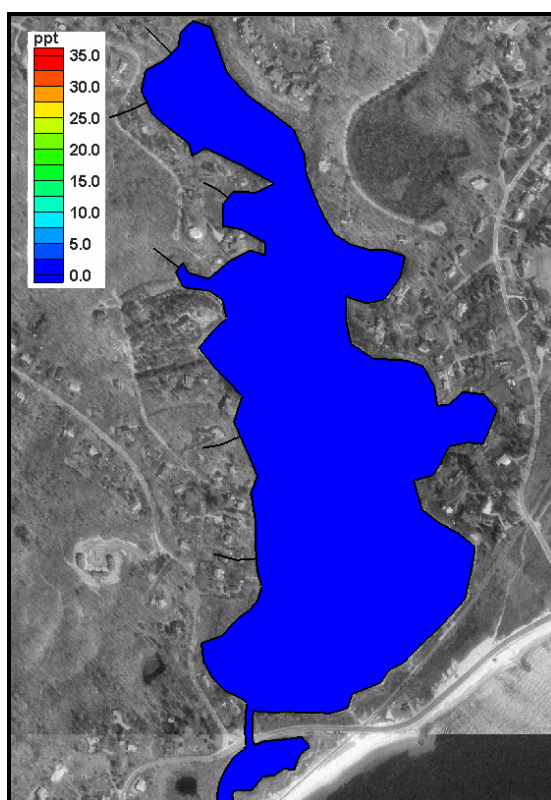


Figure VI-5. Contour Plot of modeled salinity (ppt) in Oyster Pond.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

To assess the influence of nitrogen loading on total nitrogen concentrations within each of the embayment systems, two standard water quality modeling scenarios were run: a “build-out” scenario based on potential development (described in more detail in Section IV) and a “no anthropogenic load” or “no load” scenario assuming only atmospheric deposition on the watershed and sub-embayment, as well as a natural forest within each watershed. Comparisons of the alternate watershed loading analyses are shown in Table VI-3. Loads are presented in kilograms per day (kg/day) in this Section, since it is inappropriate to show benthic flux loads in kilograms per year due to seasonal variability.

Table VI-3. Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic (“no-load”) loading scenarios of Oyster 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
Oyster Pond	4.066	4.542	+11.7%	0.397	-90.2%
Oyster Pond Lagoon	0.293	0.293	0.0%	0.047	-84.1%
Mosquito Creek (surface water)	0.115	0.115	0.0%	0.093	-19.0%

VI.2.6.1 Build-Out

In general, certain sub-embayments would be impacted more than others. The build-out scenario indicates that there would be less than a 12% increase in watershed nitrogen load to Oyster Pond as a result of potential future development. There are no build-out load increases for the Lagoon and Mosquito Creek watersheds (i.e. these sub-watersheds are currently “built out”). For the no load scenarios, almost all of the load entering the watershed is removed; therefore, the load to the main basin of the Oyster Pond system lower than existing conditions by over 90%.

For the build-out scenario, a breakdown of the total nitrogen load entering Oyster Pond is shown in Table VI-4. The benthic flux for the build-out scenarios is assumed to vary proportional to the watershed load, where an increase in watershed load will result in an increase in benthic flux (i.e., a positive change in the absolute value of the flux), and *visé 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-4. Build-out sub-embayment and surface water loads used for total nitrogen modeling of Oyster Pond, 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)
Oyster Pond	4.542	0.773	-1.874
Oyster Pond Lagoon	0.293	0.027	-0.052
Mosquito Creek (surface water)	0.115	-	-

Following development of the nitrogen loading estimates for the build-out scenario, the water quality models of each system were run to determine nitrogen concentrations within each sub-embayment (Table VI-5). Total nitrogen concentrations in the receiving waters (i.e., Vineyard Sound) remained identical to the existing conditions modeling scenarios. Total N concentrations increased in the Pond increased by 3.0 % throughout the main basin. A color contour plot of build-out N concentrations is shown in Figure VI-6.

Table VI-5. Comparison of model average total N concentrations from present loading and the build-out scenario, with percent change, Oyster Pond.

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Oyster Pond	OP1	0.696	0.717	+3.0%
Oyster Pond Lagoon	OP2	0.694	0.715	+3.0%
Mosquito Creek (surface water)	OP3	0.693	0.714	+3.0%

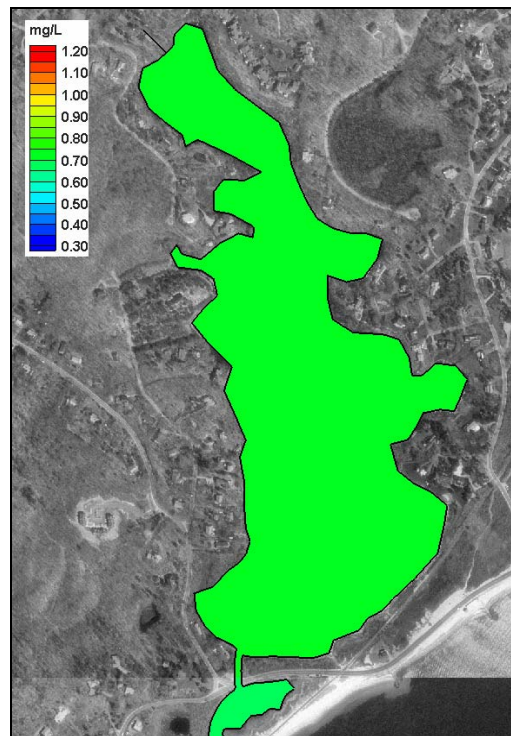


Figure VI-6. Contour plot of modeled total nitrogen concentrations (mg/L) in Oyster Pond, for projected build-out loading conditions.

VI.2.6.2 No Anthropogenic Load

A breakdown of the total nitrogen load entering each sub-embayment for the no anthropogenic load (“no load”) scenarios is shown in Table VI-6. The benthic flux input to each embayment was reduced (toward zero) based on the reduction in the watershed load (as discussed in §VI.2.6.1). The sediments become less a source of nitrogen removal with decreased loading, due to the lower amount of phytoplankton production, hence phytoplankton deposition to the sediments. 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 Oyster Pond, 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)
Oyster Pond	0.397	0.773	-0.644
Oyster Pond Lagoon	0.047	0.027	-0.018
Mosquito Creek (surface water)	0.093	-	-

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Again, total nitrogen concentrations in the receiving waters (i.e., Vineyard Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from “no load” was significant as shown in Table VI-7, with reductions greater than 44% occurring throughout the main basin of the system. Results for the system are shown in Figure VI-7.

Table VI-7. Comparison of model average total N concentrations from present loading and the no anthropogenic (“no load”) scenario, with percent change, for Oyster Pond. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).				
Sub-Embayment	monitoring station	present (mg/L)	no-load (mg/L)	% change
Oyster Pond	OP1	0.696	0.385	-44.6%
Oyster Pond Lagoon	OP2	0.694	0.385	-44.6%
Mosquito Creek (surface water)	OP3	0.693	0.384	-44.6%

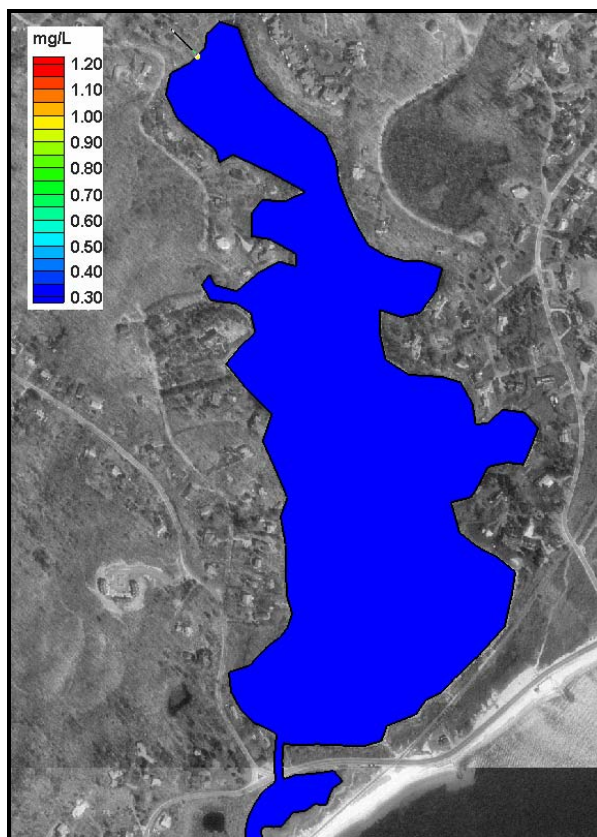


Figure VI-7. Contour plot of modeled total nitrogen concentrations (mg/L) in Oyster Pond, for no anthropogenic loading conditions.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. The MEP assessment of the Oyster Pond embayment system in the Town of Falmouth, Cape Cod, MA, has been adjusted to accommodate the brackish water nature of this tidal system. For example, the absence of eelgrass (*Zostera marina*), which generally is associated with nutrient enrichment in most southeastern Massachusetts estuaries cannot be employed in the assessment of Oyster Pond. Oyster Pond has generally supported a salinity of 2-4 ppt over the past 60 years, except for 9 years, 1987-1995, (B.L. Howes personal communication, Figure VII-1). While eelgrass does form beds over a wide salinity range, it is generally considered to be distributed in areas of 10-39 ppt (Davison and Hughes 1998), although it does germinate well at 10 ppt. Surveys in the United Kingdom generally indicate eelgrass in areas of 18-40 ppt (Tyler-Walters 2004). Therefore, there is little physiological support for targeting the establishment of eelgrass as the restoration goal for Oyster Pond. Similarly, there is no support for the occurrence of eelgrass in Oyster Pond over the past century (since it freshened). As a result, habitat assessment of this system is based upon data from the water quality monitoring database (chlorophyll and dissolved oxygen) and MEP surveys of benthic animal communities and sediment characteristics and previous studies undertaken to reorient the salinity regime of the Oyster Pond system. These data form the basis of an assessment of this system's present health, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for this system (Section VIII). Note that based upon site-specific evidence from bioassays and the elemental ratio method (Redfield Ratio), the MEP Technical Team concluded that nitrogen should be targeted as the key nutrient for management of the habitat quality of this estuarine system (see Section II).

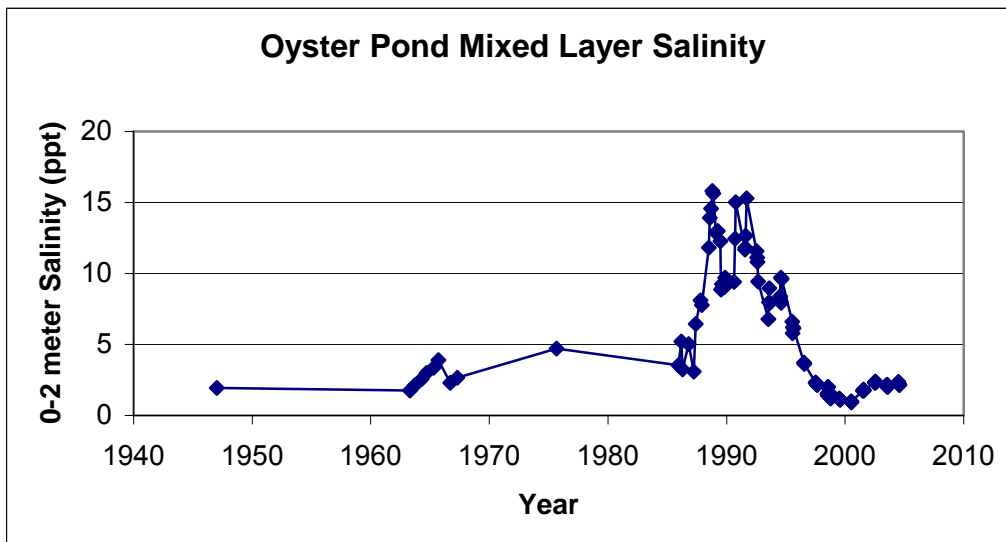


Figure VII-1. Salinity of Oyster Pond surface waters over the past 60 years. The salinity increase over 4 ppt from 1987-1995 resulted from the installation of a new culvert in the tidal channel between the Lagoon and the main basin. The recent lowering and stabilization of the salinity has resulted from the installation of a salinity control/fish weir upgradient of the culvert. Salinity reconstruction courtesy of B.L. Howes.

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

As a basis for nitrogen thresholds determination, MEP focuses 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. In the micro-tidal system of Oyster Pond oxygen depletion tends to be associated with periodic stratification by temperature (above 4 meters) and by salinity (below 4 meters in the southern basin only). The MEP Technical Team used the July and August dissolved oxygen levels (determined from grab samples) as the record of the frequency and extent of low oxygen conditions during the critical summer period. The MEP habitat analysis normally 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 throughout southeastern Massachusetts is conducted for comparison to historic records (DEP Eelgrass Mapping Program, C. Costello). Temporal trends in the distribution of eelgrass beds are used by the MEP to assess the stability of the habitat and to determine trends potentially related to 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. However, for the reasons stated above, the absence of eelgrass in Oyster Pond is the result of low salinity and linkage to nitrogen levels is therefore not possible in this system.

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

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the

Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 3.8 mg L^{-1} . Massachusetts State Water Quality Classification indicates that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1} . The tidal waters of the Oyster 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-2). 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 require several years of sampling in order to accurately capture the minimum oxygen conditions within shallow embayments (Taylor and Howes, 1994). Given the oxygen profiling throughout Oyster Pond at relatively short intervals (2 week) during the period of lowest dissolved oxygen (mid-July to late August) and given that Oyster Pond operates primarily as a brackish water lake, the multi-year grab sample data should yield an excellent estimate of minimum oxygen levels. Oyster Pond, unlike more tidally dominated systems, tends to have oxygen minima proximately caused by periodic stratification. Without the tidal currents, this stratification is only broken down by wind-driven mixing and therefore low oxygen events appear to persist for longer periods (to the extent that they occur) than in better flushed estuaries (Howes and Hart, 1997). This lends further support to the use of the high frequency grab sampling approach over several years for finding the oxygen minima for this system. Note that all oxygen data were collected ~7-8 AM, when oxygen level levels are generally at their lowest for a day in these estuaries (Taylor and Howes, 1994).

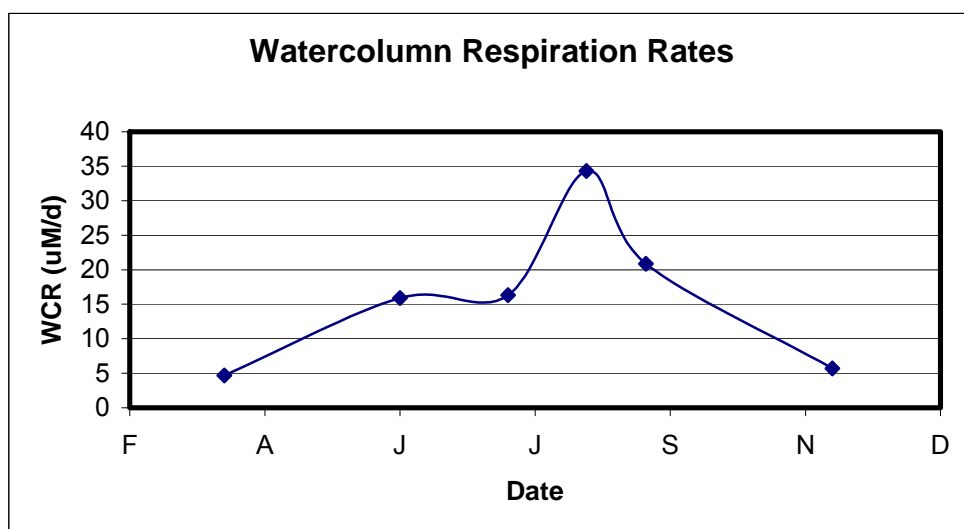


Figure VII-2. Average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Howes et al., 2004). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

Unlike many of the other embayments in southeastern Massachusetts, Oyster Pond showed a relatively consistent pattern of low oxygen in its bottom waters throughout its basins (Figure VII-3). The deep, southern basin (6 meters) is consistently anoxic during summer months due to its salinity stratification which persists for months to years. However, this represents only ~10% of the pond bottom. The remaining areas, ≤ 4 meters depth are only periodically anoxic or hypoxic. The northern basin was periodically anoxic between 1998-2004. However, this basin is enclosed and this anoxia is driven mainly by stratification. The majority of the sediments in the pond (~80%) are represented by the oxygen levels observed in the upper and lower main basin (OP-2 3.25 m, OP-3 4 m). These regions are more open to wind-driven mixing and showed oxygen levels 3 mg/L or above in 96% of samplings and 2 mg/L as a minimum level. Restoration of this system will require an improvement of oxygen levels in this lower basin, which represents most of the benthic habitat and which does not appear to support long periods of stratification shallower than 4 meters depth (as opposed to the northern basin).

Nitrogen enrichment of embayment waters generally manifests itself in the dissolved oxygen record. However, the effect of nitrogen enrichment is magnified in bottomwaters which periodically stratify, like Oyster Pond. Nitrogen enrichment affects oxygen levels by increasing the amount of phytoplankton and algae within estuarine waters. This increased organic matter results in increased rates of oxygen uptake from the water either when the plants die and decay, or at night when they do not photosynthesize. Stratification affects oxygen levels by preventing significant oxygen transport to bottom waters from the generally well oxygenated surface waters. In Oyster Pond the level of nitrogen enrichment watershed inputs is relatively low, as reflected in the moderate total chlorophyll a levels in summer (Table VII-1). In estuaries with vertically well mixed waters, these chlorophyll levels would likely not produce the consistently low summer bottom water oxygen levels. However, the depth of Oyster Pond supports periodic stratification during summer months and the resulting loss of oxygen transport from the surface to bottom waters results in the observed oxygen depletions at even these biomass levels. Note that the ultimate cause of the oxygen depletion is still nitrogen enrichment, stratification only makes the system more sensitive to its effects on oxygen levels.

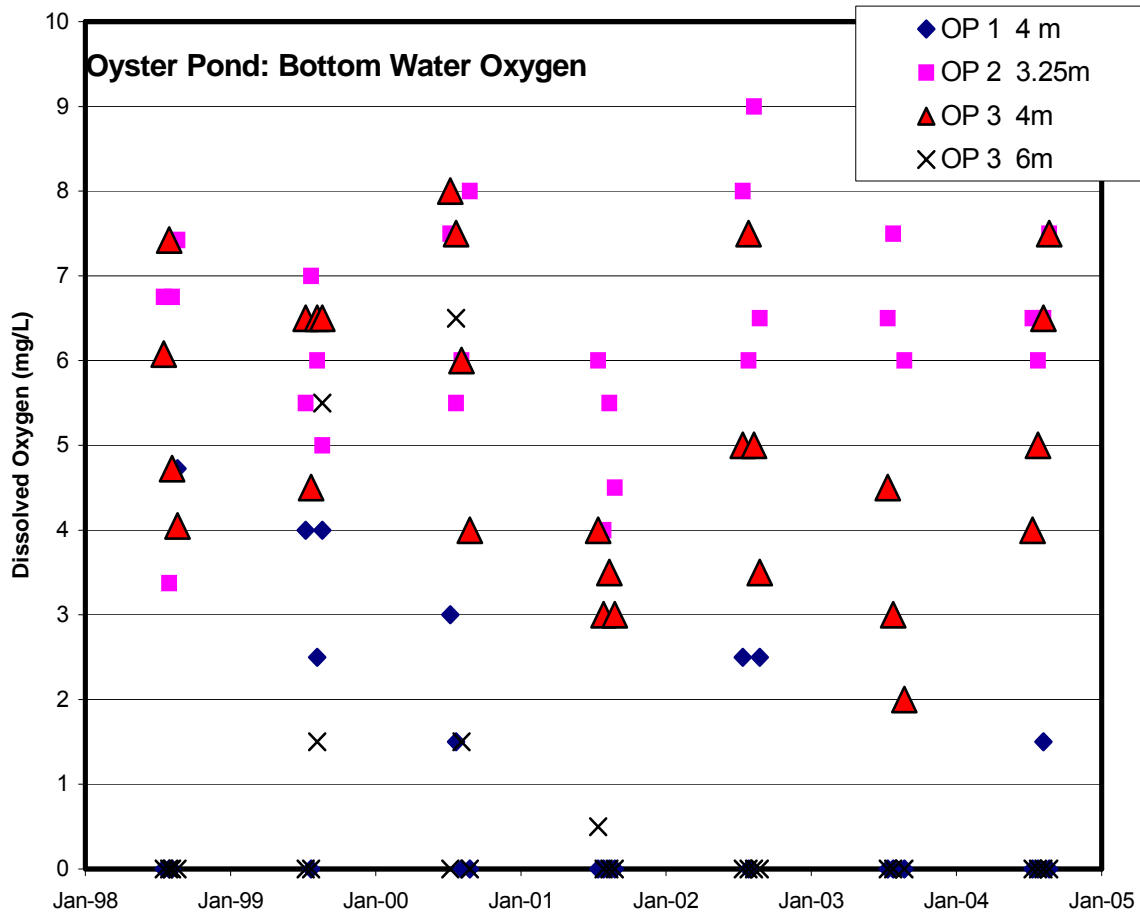


Figure VII-3. Bottom water dissolved oxygen levels measured by Falmouth PondWatch during July and August, determined from grab samples and Winkler Titrations (to 0.5 mg/L). Note that OP 1, 4 m (within the enclosed northern basin) and OP 3, 6 meters (within the deep southern basin) periodically go anoxic. The benchmark for the threshold analysis is OP-3 at 4 meters, this site had D.O. > 3 mg/L on 96% of sampling dates.

Table VII-1. Frequency of grab samples for summer chlorophyll a levels above various benchmark levels within the surface waters (0-3.5m) of the Oyster Pond System. Data courtesy of the Falmouth PondWatch Water Quality Monitoring Program and Coastal Systems Program, SMAST. Geometric averages were used to estimate "average" conditions, given the periodic phytoplankton blooms.

Sub-Embayment	Sta ID	Frequency						Statistics		
		<5 ug/L	5-10 ug/L	10-15 ug/L	15-20 ug/L	20-25 ug/L	>25 ug/L	Geo Mean ug/l	Geo s.d. ug/L	N
		%	%	%	%	%	%			
Oyster Pond System: 2000 – 2004										
Main Basin	OP1-3	57	37	1	1	1	1	4.9	1.9	134

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

As stated above, the low salinity waters of Oyster Pond are not supportive of eelgrass bed formation. The DEP Eelgrass Mapping Program has conducted no surveys in Oyster Pond. However, observations have been made by PondWatch from 1987 to present which support the lack of eelgrass in this system. Similarly, a complete system data collection and analysis effort conducted in the 1960's throughout the main basin of Oyster Pond did not indicate the presence of eelgrass (Emery, 1997). This latter effort included a census of submerged aquatic vegetation, which did not indicate eelgrass, but did indicate that the dominant SAV in 2004, *Ceratophyllum demersum*, was also dominant in the 1960's. Therefore, the most likely reason for the absence of eelgrass in the main basin of Oyster Pond is the low salinity. This indicates that eelgrass cannot be used as a habitat restoration indicator for this system.

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling to determine infaunal animal communities was conducted at 6 locations throughout the Oyster Pond System (Figure VII-4). Sites were selected at 1, 2, and 4 meter depths and in 5 of 6 cases duplicate samples were collected. Since Oyster Pond does not support the use of eelgrass as a habitat quality indicator (or as a restoration target), benthic animals were the primary biological indicator employed in the habitat quality assessment. 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 low oxygen levels in the bottom waters (4 meters) of Oyster Pond, the system is clearly impaired by nutrient overloading. However, to the extent that it can still support healthy infaunal communities, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

Analysis of the evenness and diversity of the benthic animal communities was also used to support the density data and the natural history information. The evenness statistic can range from 0-1 (one being most even), while the diversity index does not have a theoretical upper limit. In general, the highest quality habitat areas, as shown by the oxygen and chlorophyll records and eelgrass coverage (in other estuaries), 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.

The Benthic Infaunal Study (Table VII-2) indicated that Oyster Pond is not presently supportive of either diverse (H' 0-1.12, mean 0.65) or evenly distributed (mean $E = 0.46$) benthic infauna. More telling is the low number of species (0-6, mean=3) compared to nearby healthy estuarine areas (~30 species per sample). Due to its brackish waters, Oyster Pond sediments supported both freshwater and estuarine invertebrate populations. The freshwater species were generally insect larvae and these tended to dominate the community. Also notable was that almost half of the samples (5 of 11) had only 0-84 individuals, indicative of an impoverished community. Although the remaining samples had dense populations, they were distributed

among a very few species, 6 or less, indicating a stressed community. There was also a potential pattern of more stressed communities in the shallow (1 meter) and deep (4 meter) depths. Overall, the infauna community was consistent with the low dissolved oxygen and organic matter deposition observed in this relatively closed estuarine basin. The lack of a clear spatial pattern is consistent with the generally horizontally well mixed nature of this system (Section 6).

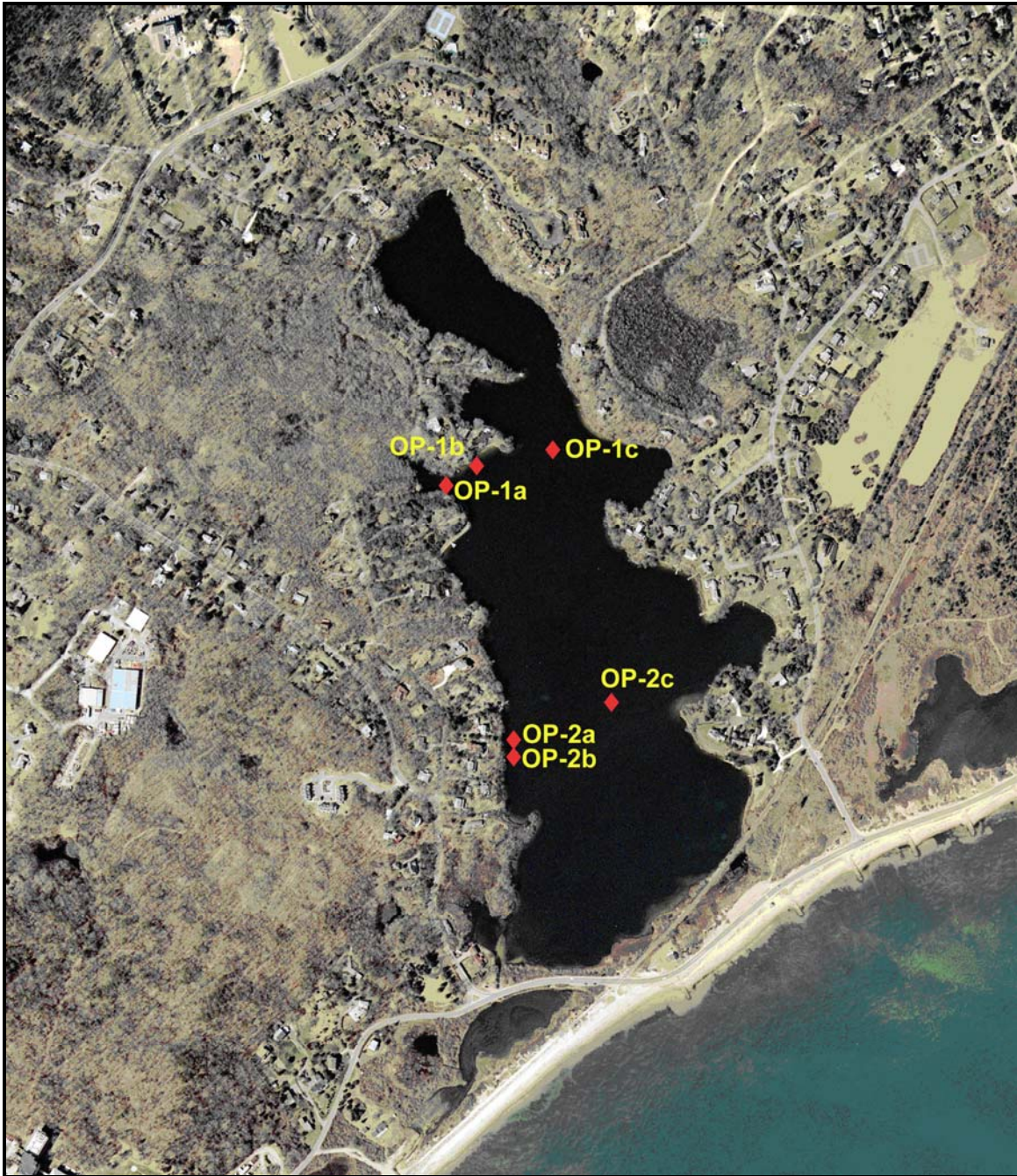


Figure VII-4. Aerial photograph of Oyster Pond showing location of benthic infaunal sampling stations (red symbol).

Table VII-2. Benthic infaunal community data for the Oyster Pond embayment system. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.0625 m²).

Marine Adults+All Freshwater Organisms					
Location	Total Actual* Species	Total Actual* Individuals	Species Calculated @75 individuals	Weiner Diversity (H')	Evenness (E)
Oyster Pond System					
OP-1-a 1m	6 (2)	1165 (241)	3.1	0.87	0.34
OP-2-a 1m	1 (0)	4 (0)	N/A	0.00	N/A
OP-2-a 1m	3 (0)	24 (0)	N/A	0.50	0.31
OP-1-b 2m	5 (1)	1200 (510)	3.0	1.12	0.48
OP-2-b 2m	5 (2)	1368 (441)	2.2	0.94	0.40
OP-1-b 2m	4 (0)	84 (0)	4.0	1.04	0.52
OP-2-b 2m	2 (1)	1271 (412)	2.0	0.91	0.91
OP-1-c 4m	4 (0)	676 (0)	1.7	0.10	0.05
OP-2-c 4m	0	0	N/A	N/A	N/A
OP-1-c 4m	1 (0)	5 (0)	N/A	0.00	N/A
OP-2-c 4m	3 (1)	1613 (12)	2.4	1.05	0.66
Overall:	3	674	2.6	0.65	0.46
* values are totals of marine and freshwater organisms, values in (#) are for marine organisms only.					

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

VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires the integration of key habitat parameters (infauna and eelgrass), sediment characteristics, and nutrient related water quality information (particularly dissolved oxygen and chlorophyll a). Additional information on temporal changes within each sub-embayment and its watershed further strengthen the analysis. These data were collected to support threshold development for the Oyster Pond System by the MEP Team and were discussed in Chapter VII. Nitrogen threshold development builds on this data and links habitat quality to summer water column nitrogen levels from the long-term baseline water quality monitoring program. At present the bulk of the Oyster Pond is showing moderately to significantly impaired habitat quality (Chapter VII).

Eelgrass: There is no evidence that the Pond has supported eelgrass from 1947 to present. If the pond supported eelgrass prior to the construction of the railroad embankment (1872), then as the tidal exchange became restricted due to the embankment and the subsequent failure of the tidal culvert and the pond freshened, eelgrass was lost. A likely estimate is that the Pond lost eelgrass more than 100 years ago as a result of freshening (not nutrients) and can no longer support eelgrass at “present” pond salinities (Section VII). Since Oyster Pond cannot support eelgrass at or near its present salinity status, eelgrass cannot be used as either an indicator of habitat health or as a target for habitat restoration. Therefore, in keeping with the MEP thresholds approach, restoration of infaunal habitat will be used as the restoration target.

Water Quality: At present, the waters of Oyster Pond generally support a moderate level of water quality as evidenced by the moderate levels of phytoplankton biomass (indicated by chlorophyll a) in summer (mean 4.9 ug/L). However, given the limited tidal flushing of the system most of the primary production remains in the Pond, to decompose and consume oxygen and regenerate nitrogen. The keystone water quality issue in Oyster Pond involves the interaction between nitrogen enrichment (through its produced organic enrichment) and dissolved oxygen in bottomwaters as also influenced by periodic vertical stratification. The ecologically significant result of this interaction is the periodic depletion of oxygen in bottom waters (4 meters) overlying significant areas of pond bottom. In this system, this periodic hypoxia is most likely the cause of the poor infaunal habitat, as evidenced by the present infaunal communities structure.

Infaunal Communities: The Infaunal Study showed communities consistent with it oligohaline condition, namely a mixture of estuarine species and species more typical of “fresh” water. Under present conditions, the community within the pond shows signs of nutrient related stress, in that while in some areas (typically at 2 meters) there are relatively large populations, the species numbers are very low, hence diversity (H') and evenness (E) are low. The loss of benthic communities at 4 meters appears to be due to low oxygen and at 1 meter potentially due to accumulations of macrophytes. However, the macrophytes themselves provide habitat for invertebrates, which was not assessed. This suggests that the 1 meter samples may not represent the “resident” invertebrate community at this depth. The 2 meter and 4 meter samples are not compromised by macrophyte effects and therefore will be used in the development of the nitrogen threshold for restoration. The overall results indicate a system

almost certainly capable of supporting diverse healthy communities but which currently has infaunal habitat that is significantly impaired under present nitrogen loading conditions.

Overall, all of the indicators show a consistent pattern of moderate to significant impairment throughout the basins of Oyster Pond. While the Pond does not show strong gradients in salinity or water quality parameters, the enclosed nature of the northern basin appears to increase the duration of stratification and subsequent hypoxia. The deep southern basin (~6 m) is salinity stratified for months to years at a time and is generally anoxic as a result of this natural process. Based primarily on the infaunal communities and the bottomwater hypoxia, it was concluded that Oyster Pond habitat is presently moderately to significantly impaired. Since the ultimate cause of the low dissolved oxygen (≤ 4 m) results from nitrogen enrichment, it can also be concluded that the system is nitrogen overloaded at present.

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.

As stated above, Oyster Pond differs from most other estuaries in its lack of horizontal gradients in salinity, nitrogen, and nitrogen related parameters (chlorophyll a, D.O., transparency, etc.). Therefore, selection of the sentinel station was not based on horizontal gradients and their response to changing nitrogen loads. Instead, the sentinel station was selected to best capture the overall conditions of the Pond waters. Stations OP-1, OP-2, OP-3 could all be used as the sentinel station, but if a single point needs to be monitored, then OP-3 appears to be the most representative of the pond waters. In addition, since Oyster Pond is vertically stratified, the surface mixed layer (0-4 m) is the target for setting the nitrogen threshold level, as this is the zone in the pond which impinges on potentially usable benthic habitat.

The nitrogen threshold for Oyster Pond is based upon restoring benthic habitat for infaunal animals. Given the natural stratification of Pond waters, sediments < 4 meters depth representing ~80% of the pond bottom were targeted. This depth is based upon the depth distribution of the bottom and the depth of the mixed layer. Since the present nitrogen levels result in periodic hypoxia at 4 meters depth, the nitrogen threshold was set to improve and maintain oxygen levels ≥ 6 mg/L at 4 meters depth in the main basin (OP-3). At present, the minimum dissolved oxygen at this station is most likely 3 mg/L, although a single reading of 2 mg/L was recorded. Given the uncertainties in determining minimum D.O. in any estuary, the nitrogen threshold was set using 2 mg/L as the current minimum D.O. level. This adds a level of safety to the analysis.

The concept underpinning the linkage of nitrogen levels (threshold) to bottom water dissolved oxygen minimum is based upon the fact that during brief stratification events (as opposed to the prolonged salinity stratification), oxygen is taken up in respiration and oxygen resupply by ventilation or photosynthesis is trivial. Since the rate of oxygen uptake is directly proportional to the amount of organic matter in the system and the amount of organic matter in this tidally restricted brackish kettle pond is proportional to the amount of nitrogen input, then

decreasing nitrogen inputs should result in proportional decreases in the rate of oxygen uptake in the hypolimnion during stratification. In other words, since the pond and its watershed are operating nearly completely as a biogeochemically “closed” system, reductions in nitrogen inputs should result in proportional reductions in oxygen uptake in bottom waters.

Since at summer temperatures (25°C) and salinities (2 ppt), dissolved oxygen saturation is 8.2 mg/L and current oxygen minimum is 2 mg/L then raising the minimum oxygen level to 6 mg/L would require 4/6.2 or 65% reduction in the rate of oxygen uptake during stratification. This assumes that the present duration and frequency of stratification of waters overlying sediments 4 meters or less deep will remain as at present. This is a safe assumption as long as the management plan does not allow the pond salinity levels to climb above target 2-4 ppt range. Given the link between nitrogen load and oxygen uptake rate, this 65% reduction in oxygen uptake would require a 65% reduction in nitrogen loading to Oyster Pond. Using a similar analysis, raising the periodic minimum dissolved oxygen to 3.8 mg/L (Chesapeake Bay value) or the SB criteria of 5 mg/L would require reductions in nitrogen loading of 29% and 48%, respectively. These translate into the total nitrogen threshold levels in the mixed layer of Oyster Pond shown in Table VIII-5.

Table VIII-1. Average total N concentrations to achieve target bottom water oxygen minima. Based upon modeled nitrogen reductions for the Oyster Pond system. Selected sentinel station is OP-3 .		
Sub-Embayment	monitoring station	threshold (mg/L)
3.8 mg/L Minimum D.O. Threshold		
Oyster Pond – upper	OP1	0.635
Oyster Pond – mid	OP2	0.634
Oyster Pond – lower	OP3	0.633
5.0 mg/L Minimum D.O. Threshold		
Oyster Pond – upper	OP1	0.589
Oyster Pond – mid	OP2	0.588
Oyster Pond – lower	OP3	0.588
6.0 mg/L Minimum D.O. Threshold		
Oyster Pond – upper	OP1	0.549
Oyster Pond – mid	OP2	0.548
Oyster Pond – lower	OP3	0.548

Stratification and Oxygen Transport to Bottom waters: Given the importance to the thresholds development of negligible oxygen input from the surface mixed layer to the bottom waters (hypolimnion), the MEP examined the potential rate of diffusion (eddy diffusion) to supply oxygen to bottom waters during stratification. This analysis was conducted in collaboration with SMAST scientists (Dr. M. Sundermeyer). The analysis revealed that during periods of light winds, enhanced stratification in the upper-most few meters of the water column can significantly reduce, or even halt ventilation of deeper waters. The duration of such events can have a significant influence on subsurface D.O. levels. Consider an event in which the upper few meters of the water column go from a uniform, well-mixed layer of temperature 20° C, to the upper 2 m having a temperature of 22° C, capping a 20° C layer below. Assuming a surface heat flux into the water during daylight hours of order 400 W/m², the time required to heat the upper 2 m of water by 2° C is: $\Delta t = \Delta T \cdot \rho \cdot c_p \cdot \Delta z / Q_{net}$, where Δt is time, ΔT is change in temperature, $\rho \approx 1020 \text{ kg/m}^3$, is the density of seawater, $c_p \approx 0.95 \text{ cal/g/C}$ is the specific heat of water, Δz is 2 m, and Q_{net} is the mean heat flux over the time of interest. This suggests that it would take approximately 12 hours, or about 1 day (sunrise to sunset) to heat the upper 2 m by

this amount. Such stratification events can thus occur as the result of only a single day of such conditions. Assuming a “worst-case” scenario, in which vertical eddy diffusivity is completely suppressed by such stratification events, so that the vertical diffusivity of dissolved oxygen is reduced to its molecular value of $\kappa = 2 \times 10^{-9} \text{ m}^2/\text{s}$, the characteristic time to ventilate the deeper waters by molecular diffusion alone can be estimated. Assuming a D.O. concentration of 8 mg/l at a depth of 3.25 m, decreasing to 2 mg/l at 4 m, the characteristic diffusion time scale to bring the 4 m D.O. level up to the 3.25 m value is $T_{\text{diffusion}} = \Delta z^2 / \kappa = (0.75 \text{ m})^2 / 2 \times 10^{-9} \text{ m}^2/\text{s} \approx 3,255 \text{ days} \approx 9 \text{ yrs}$; i.e., much too long to be relevant on the timescales of these events. Put another way, considering the same D.O. values from the perspective of vertical fluxes, the downward flux of D.O. by molecular diffusion under such conditions would be of order, $\text{Vert. D.O. Flux} = \kappa \cdot \Delta \text{D.O.} / \Delta z = 2 \times 10^{-9} \text{ m}^2/\text{s} \times 6 \text{ mg/l} / 0.75 \text{ m} = 1.6 \times 10^{-5} \text{ mg/m}^2/\text{s} = 1.4 \text{ mg/m}^2/\text{day}$. Put another way, this would amount to a rate of increase of DO of $\kappa \cdot \Delta \text{DO} / \Delta z^2 = 2 \times 10^{-9} \text{ m}^2/\text{s} \times 6 \text{ mg/l} / (0.75 \text{ m})^2 = 2.1 \times 10^{-8} \text{ mg/l/s} = 1.8 \times 10^{-3} \text{ mg/l/day}$; i.e. much too small to be relevant compared to respiration and/or generation by photosynthesis.

VIII.3. DEVELOPMENT OF TARGET NITROGEN LOADS

The nitrogen thresholds developed in the previous section were used to determine the amount of total nitrogen mass loading reduction required for restoration of Oyster Pond to a series of dissolved oxygen values. Due to the existing salinity levels in the Pond (historically between 0 and 4 ppt), eelgrass cannot be established within this brackish water body. Instead, development of an appropriate threshold to restore infaunal habitat was based on minimum dissolved oxygen within the lower basin of Oyster Pond. It was determined that a linear relationship was appropriate to assess the expected changes in dissolved oxygen relative to total nitrogen for the site-specific conditions within the main basin of the Pond. Minimum dissolved oxygen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were lowered by the percentage derived by the following equation:

$$\% \text{ N Reduction} = \frac{(\text{Target D.O.} - \text{Min Observed DO})}{(\text{Max Saturation} - \text{Min Observed DO})} * 100\%$$

It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the communities.

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold dissolved oxygen concentrations were higher for higher minimum dissolved oxygen levels. Since the nitrogen concentrations are generally uniform across the entire surface of Oyster Pond (i.e. there is virtually no spatial gradient in nitrogen concentration), the nitrogen load was removed uniformly. Distributions of tidally-averaged nitrogen concentrations associated with the above thresholds analysis are shown in Figures VIII-1 through VIII-3 for each pond separately.

Tables VIII-3 and VIII-4 provide additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. Table VIII-4 shows the breakdown of threshold sub-embayment and surface water loads used for total nitrogen modeling. In Table VIII-4, loading rates are shown in kilograms per day, since benthic loading varies throughout the

year and the values shown represent 'worst-case' summertime conditions. The benthic flux for this modeling effort is reduced from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Vineyard Sound.

Comparison of model results between existing loading conditions and the selected loading scenarios to achieve the target D.O. concentrations within Oyster Pond are shown in Table VIII-5. To achieve the threshold dissolved oxygen concentrations at the sentinel stations, a reduction in TN concentration of approximately 9%, 15%, and 21% is required for dissolved oxygen concentrations of 3.8 mg/l (based on the EPA's Chesapeake Bay limit), 5.0 mg/l (Massachusetts SB waters), and 6.0 mg/l (Massachusetts SA waters), respectively.

Although the above modeling results provide one manner of achieving the selected threshold levels within the Oyster Pond system, the specific examples do not represent the only method for achieving this goal. However, the thresholds analysis provides general guidelines needed for the nitrogen management of this embayment.

Table VIII-2. Comparison of sub-embayment watershed septic loads (attenuated) used for modeling of present and threshold loading scenarios of the Oyster Pond system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.			
sub-embayment	present septic load (kg/day)	threshold septic load (kg/day)	threshold septic load % change
3.8 D.O. threshold			
Oyster Pond	3.477	2.192	-37.0%
Oyster Pond Lagoon	0.230	0.230	+0.0%
Mosquito Creek (surface water)	0.110	0.068	-37.5%
5.0 D.O. Threshold			
Oyster Pond	3.477	1.332	-61.7%
Oyster Pond Lagoon	0.230	0.230	+0.0%
Mosquito Creek (surface water)	0.110	0.041	-62.5%
6.0 D.O. threshold			
Oyster Pond	3.477	0.619	-82.2%
Oyster Pond Lagoon	0.230	0.230	+0.0%
Mosquito Creek (surface water)	0.110	0.019	-82.5%

Table VIII-3. Comparison of sub-embayment total attenuated watershed loads (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Oyster Pond system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.			
sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
3.8 D.O. threshold			
Oyster Pond	4.066	2.781	-31.6%
Oyster Pond Lagoon	0.293	0.293	+0.0%
Mosquito Creek (surface water)	0.115	0.074	-35.7%
5.0 D.O. Threshold			
Oyster Pond	4.066	1.921	-52.8%
Oyster Pond Lagoon	0.293	0.293	+0.0%
Mosquito Creek (surface water)	0.115	0.047	-59.5%
6.0 D.O. threshold			
Oyster Pond	4.066	1.208	-70.3%
Oyster Pond Lagoon	0.293	0.293	+0.0%
Mosquito Creek (surface water)	0.115	0.025	-78.6%

Table VIII-4. Threshold sub-embayment and surface water loads used for total nitrogen modeling of the Oyster 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)
3.8 D.O. threshold			
Oyster Pond	2.781	0.773	-1.342
Oyster Pond Lagoon	0.293	0.027	-0.037
Mosquito Creek (surface water)	0.074	-	-
5.0 D.O. Threshold			
Oyster Pond	1.921	0.773	-1.080
Oyster Pond Lagoon	0.293	0.027	-0.030
Mosquito Creek (surface water)	0.047	-	-
6.0 D.O. threshold			
Oyster Pond	1.208	0.773	-0.863
Oyster Pond Lagoon	0.293	0.027	-0.024
Mosquito Creek (surface water)	0.025	-	-

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

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
3.8 D.O. threshold				
Oyster Pond – upper	OP1	0.696	0.635	-8.7%
Oyster Pond – mid	OP2	0.694	0.634	-8.7%
Oyster Pond – lower	OP3	0.693	0.633	-8.7%
5.0 D.O. Threshold				
Oyster Pond – upper	OP1	0.696	0.589	-15.4%
Oyster Pond – mid	OP2	0.694	0.588	-15.3%
Oyster Pond – lower	OP3	0.693	0.588	-15.2%
6.0 D.O. threshold				
Oyster Pond – upper	OP1	0.696	0.549	-21.1%
Oyster Pond – mid	OP2	0.694	0.548	-21.0%
Oyster Pond – lower	OP3	0.693	0.548	-21.0%

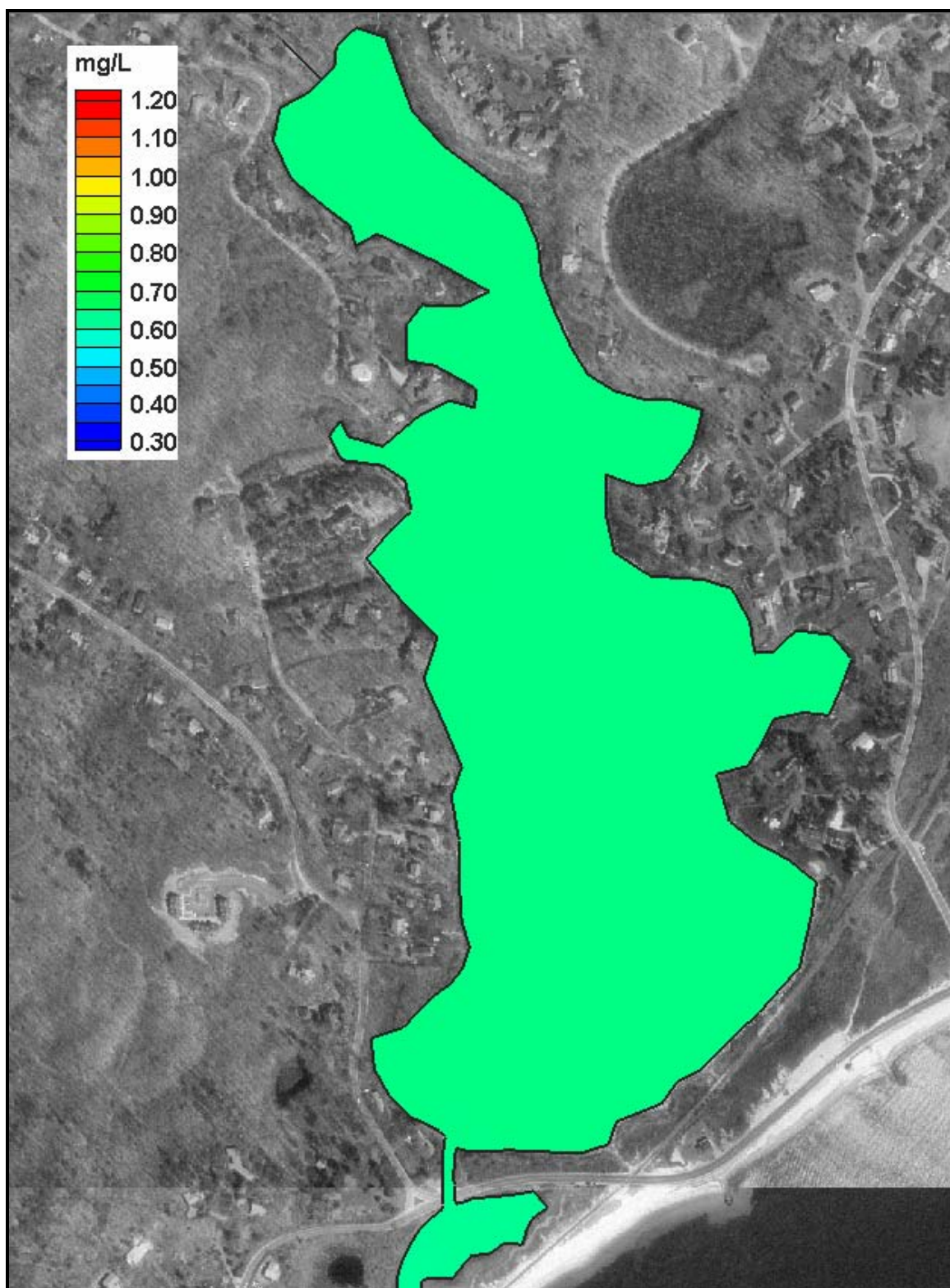


Figure VIII-1. Contour plot of modeled average total nitrogen concentrations (mg/L) in the Oyster Pond system, for threshold conditions (minimum 3.8 mg/L Dissolved Oxygen pond-wide).

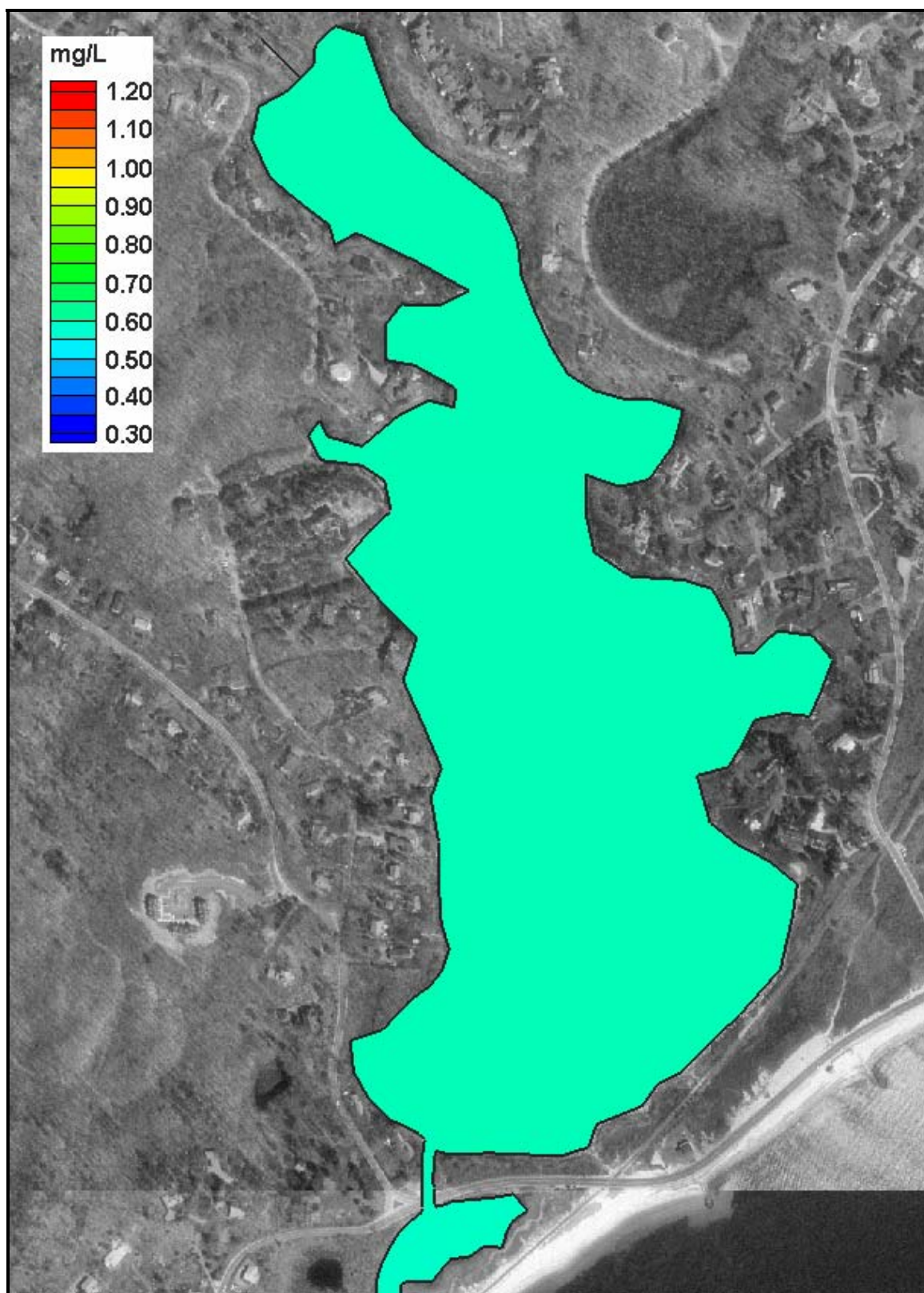


Figure VIII-2. Contour plot of modeled average total nitrogen concentrations (mg/L) in the Oyster Pond system, for threshold conditions (minimum 5.0 mg/L Dissolved Oxygen pond-wide).

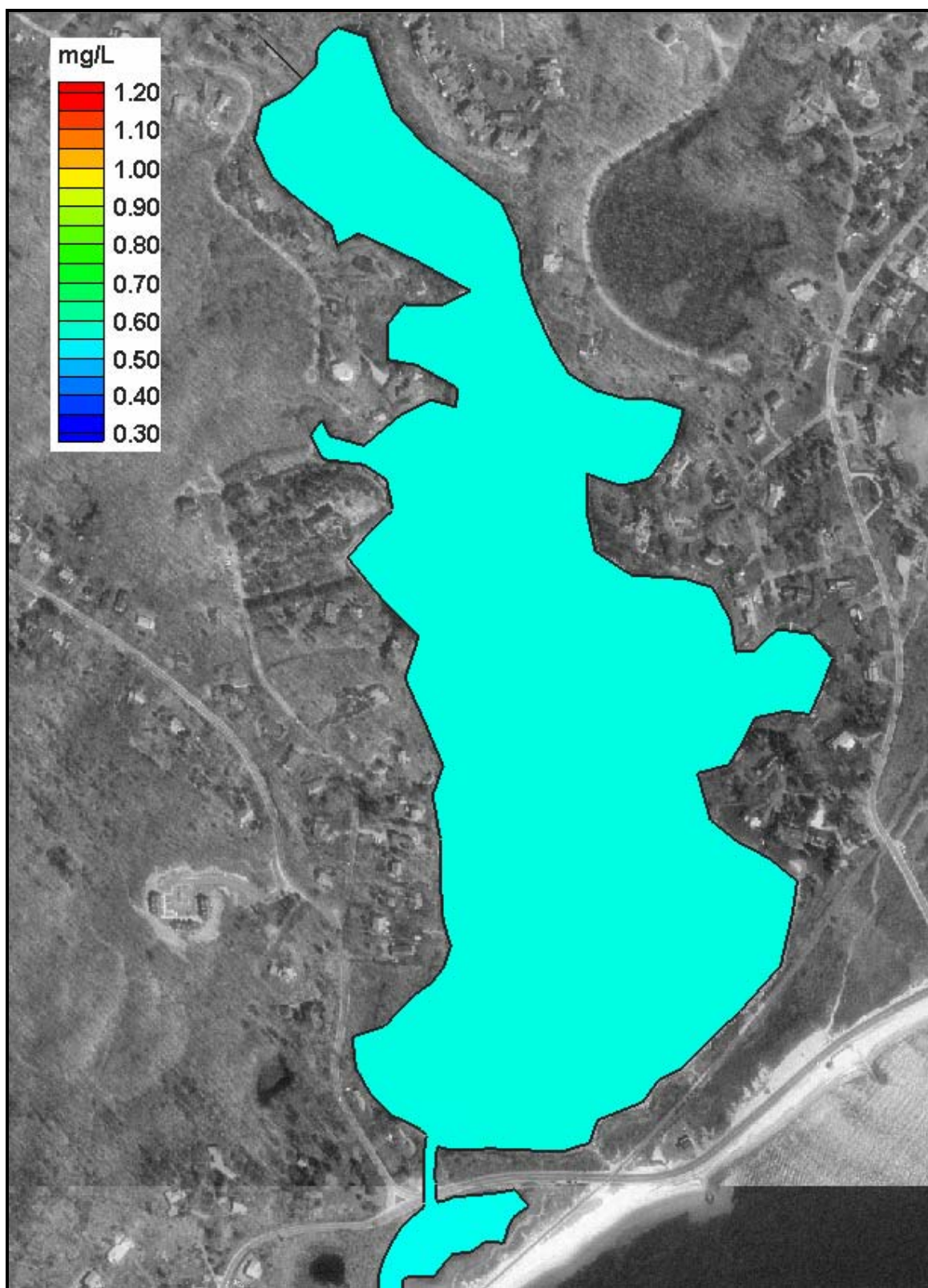


Figure VIII-3. Contour plot of modeled average total nitrogen concentrations (mg/L) in the Oyster Pond system, for threshold conditions (minimum 6.0 mg/L Dissolved Oxygen pond-wide).

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