Nitrogen Modeling to Support Watershed Management: Comparison of Approaches and Sensitivity Analysis

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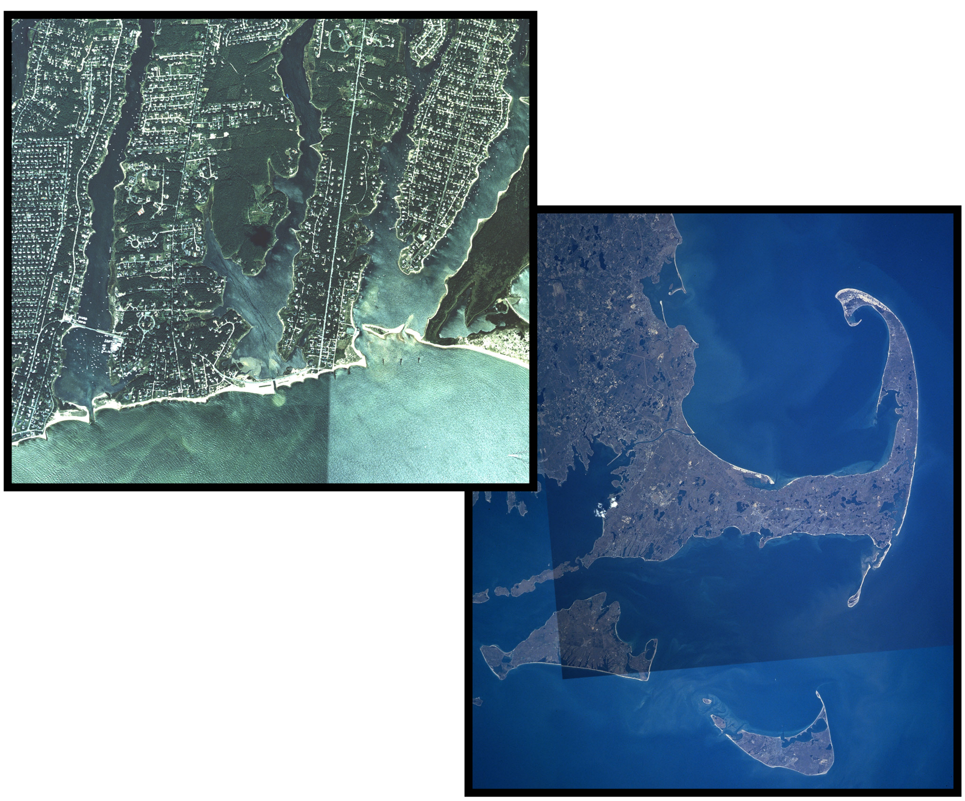
Prepared For:

Massachusetts Department of Environmental Protection
Bureau of Resource Protection

And

U.S. Environmental Protection Agency
Region I



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EXECUTIVE SUMMARY

Overview: Coastal embayments throughout the State of Massachusetts (and along the U.S. eastern seaboard) are becoming nutrient enriched. The nutrients are primarily related to changes in watershed land-use associated with increasing population within the coastal zone over the past half century. Many of Massachusetts' embayments have nutrient levels that are approaching or are currently over their 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 use 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. DEP, SMAST, and others have partnered under the Massachusetts Estuary Project to address these embayment management and planning issues.

A central component of the Massachusetts Estuary Project is the task of providing a quantitative tool for watershed-embayment management throughout Southeastern Massachusetts. This Massachusetts Estuary Project is founded upon science-based management. The Project will use a consistent, state-of-the-art approach throughout the region's coastal waters and provide technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the project is to provide technical guidance to DEP in support of policies regarding nitrogen loading to embayments and to conduct nitrogen TMDL's. In appropriate estuaries, TMDL's for bacterial contamination will also be conducted in concert with the nutrient effort (particularly if there is a 303d listing). However, the goal of the bacterial program is to provide information to guide targeted sampling for specific source identification and remediation. As part of the overall effort, the evaluation 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 Massachusetts,
- provide necessary data collection and analysis required for quantitative modeling,
- conduct quantitative TMDL analysis, outreach, and planning, and
- keep each embayment's model "alive" to address future regulatory needs.

The Massachusetts Estuary Project is comprised of four phases relating to project design, project development, implementation of approach, and application of management models to on-going management issues. The project phases are:

- *Phase I:* Assemble a working group, design the project organizational framework, evaluate existing management models and select appropriate approach for regional implementation, and survey existing data sources with regard to potential to support selected approach.
- *Phase II:* Determine the prioritization procedure and select initial embayments, promote water quality data collection in embayments with insufficient baseline data, educate local

stakeholders as to Project goals, approach, results and data needs and complete the assessment of existing data and data gaps. Also, establish necessary regulatory and stakeholder committees and increase the analytical capability of the Project Team relative to collection of field data needed to support the management approach.

- *Phase III:* Implement embayment management approach on a 2-year cycle, which includes field data collection, modeling, reporting, and a significant level of public outreach. Year 1 focuses on site-specific data collection to fill data gaps, Year 2 focuses on modeling, synthesis, and evaluation of management options.
- *Phase IV:* Keep quantitative models and embayment specific management approaches "alive" for future DEP and other management/planning needs. Also, provide a platform (upon request) for tracking embayment changes.

This Report is part of Phase I of the Massachusetts Estuary Project. Before implementing a specific approach to support nitrogen management, it is necessary to evaluate current watershed and embayment nitrogen management models as to their accuracy, data needs, comparability, and applicability across embayment types observed in southeastern Massachusetts. At present, three approaches have been relatively widely applied: the Buzzards Bay Project Nitrogen Loading Methodology (BBP), the Cape Cod Commission Nitrogen Loading/Critical Loads Methodology (CCC), and the Linked Watershed-Embayment Modeling Approach (Linked).

A Case Study approach was used to evaluate and compare the available nitrogen management models. The Case Studies were selected from among the ca. 15 embayments in Southeastern Massachusetts for which we have developed the data to support the higher level Linked Modeling Approach. The Case Study embayments selected were within the Towns of Falmouth and Chatham on Cape Cod. These embayments were selected for the following reasons:

- the embayments had sufficient data to parameterize each of the management models;
- the embayments used are similar in structure, nitrogen loading, and hydrodynamics to those throughout Southeastern Massachusetts;
- the results from evaluation of these embayments are immediately transferable to other embayments throughout the region.

The overall assessment of the management models included:

- comparison of watershed nitrogen loading results from each model and the resultant embayment nitrogen distribution based upon the Linked Model;
- evaluation of predicted critical nitrogen loading thresholds (BBP) relative to the resultant embayment nitrogen distribution based upon the Linked Model;
- sensitivity analysis for the Linked Watershed-Embayment Approach.

The Linked Model was used to portray the nitrogen distribution within each embayment resulting from the BBP and CCC methodologies, because only the Linked Model has this capability. In addition the Waquoit Model for watershed nitrogen loading was utilized to portray embayment nitrogen distribution for the one case study available. In general construct, the Linked Model uses a watershed land-use loading approach similar to the other models, but it also is coupled to a numerical hydrodynamic/water quality model, which encompasses the circulation and dispersion of nitrogen within the receiving waters. This linkage of watershed and embayment not only provides for assessment of specific areas within embayments, but also allows for calibration and validation approaches not available for the other methodologies.

The comparative application of the various methodologies to the case study embayments also provided an analysis of the consistency of model results between systems. This latter point is critical in evaluating a model for use by the Massachusetts Estuary Project, that will cover all 89 embayments of Southeastern Massachusetts. Given the specific regional nature of the project (all embayments in Massachusetts from Duxbury to Mt. Hope Bay, including Cape Cod, Nantucket, and Martha's Vineyard), the evaluation and selection of an appropriate model must focus on its utility in these specific systems. The models, in this evaluation are directly applicable to shallow (generally <5m), primarily vertically mixed (only supporting periodic short term stratification), enclosed or semi-enclosed embayments, surrounded by permeable watersheds with significant groundwater discharges. The approaches can also be used in Mt. Hope Bay, a deeper estuary, which supports periodic strong salinity stratification. However, this analysis will require additional parameterization and complexity of the underlying hydrodynamic model component, as well as development of a separate system-specific uncertainty analysis. Therefore, the results of the various model evaluations are directly applicable to 88 of the 89 embayments within the project area.

In addition to the comparison of the various management models and sensitivity analyses, specific management applications of the Linked Watershed-Embayment Model are also presented which address the utility of the Linked Model for "real-world" embayment management issues. Four examples are presented of specific management scenarios related to the Case Studies. These management alternatives included the determination of estuarine nitrogen levels resulting from: removal of Title 5 septic loads, increased watershed nitrogen loading at build-out, and modifications of tidal inlets (e.g. improvements to tidal flushing).

Conclusions Based Upon Model Comparisons and Sensitivity Analysis: The specific results of the comparative analysis of the models, the sensitivity analysis of the Linked Model, and the management application Case Studies are summarized below. *The overall conclusion from the evaluation was that the Linked Watershed-Embayment Model was the best available model for use in the 89 embayments within the Massachusetts Estuary Project.* The Linked model outperformed the various other management models in predicting estuarine nitrogen levels, was the only model to include a quantitative (both calibrated and validated) embayment component, was robust relative to the watershed model component, and included both nitrogen attenuation during transport, nitrogen regenerated within the estuary, and considers groundwater transport time. In addition, the Linked Model Approach provides for independent calibration and validation at each level of assessment, thus increasing the certainty of the results and the confidence needed to guide management.

1. Watershed-embayment nitrogen management requires an approach that can accurately portray nitrogen levels within receiving waters and relate them to habitat quality. The approach needs to be holistic and allow evaluation of the effects of spatially altering nitrogen loads, determine the effects associated with changes in key determinants (e.g., tidal exchange, source waters, freshwater flows), and allow evaluations of all spatial scales of the embayment (tributary, upper, lower, coves, etc.). The Linked Watershed-Embayment Model is consistent with these management needs.

2. Of the Models evaluated, only the Linked Model provides output as to the nitrogen distribution throughout an embayment resulting from determined watershed load. In general construct, the Linked Model uses a watershed land-use loading approach similar to the BBP and CCC models, but also is coupled to a numerical hydrodynamic/water quality model which encompasses the circulation and dispersion of nitrogen within the receiving waters.

This linkage of watershed and embayment not only provides for assessment of specific areas within embayments, but also allows for calibration and validation approaches not open to the other models. The BBP and CCC Models typically distribute bulk nitrogen loads to sub-embayments or to entire embayment systems, since they do not have a spatially dependent embayment component.

3. The Linked Watershed-Embayment Nitrogen Management Model requires additional data, not needed by the BBP and CCC Models, to support its embayment component that includes both hydrodynamic and ecological parameters. However, the BBP and CCC do require measurement of embayment flushing rates for determination of critical nitrogen loads.

4. In almost all cases, the standard nitrogen loading terms are consistent among the BBP, CCC, and Linked Models. This is not surprising, since they are based upon the same studies and base data. However, the septic loading term is about 25% lower in the Linked Model than the BBP or CCC Models. This results from the use of Title 5 design flows for the BBP and CCC methodologies (with 35 mgN/L), while the Linked Model is based upon regional septic system discharge and transport studies. In addition, while all methods correct for occupancy, this is deemed a major error in some applications that have not properly evaluated occupancy rates in seasonal communities. Errors in occupancy create a proportional error in residential wastewater loading. Although the error in final nitrogen load due to incorrect occupancy data is “project specific” (and cannot be evaluated here), the resulting error that it generates in the final watershed loading is relatively significant, due to the preponderance of on-site wastewater in most Southeastern Massachusetts coastal watersheds.

5. In contrast to the land-use nitrogen loading terms, there is not consistency among the various methodologies as to the extent of nitrogen attenuation within the watershed as nitrogen moves via groundwater or surface water from the source to the receiving waters. The Linked Model includes attenuation, based upon site-specific measurements. The BBP Model did not use attenuation prior to 2000 and now uses a generic attenuation of 30% for surface and groundwater transport (>1 km). The CCC Model does not include attenuation during transport.

6. The similarity in construct of the BBP and CCC watershed nitrogen loading models and the watershed portion of the Linked Model suggests that previous watershed loading databases might be easily modified to support the Linked Watershed-Embayment Approach.

7. The overall calibration process for the hydrodynamic modeling generally produces errors in tidal elevation and phase of less than 3%. For the more complex embayments, current measurements provide additional model validation data. A comparison of the measured and computed volume flow rates at the Stage Harbor Inlet based upon the hydrodynamic component of the Linked Model showed remarkably good agreement. The calibrated model accurately describes both the general conditions and the irregularities of the discharge through the inlet.

8. Water quality models of estuaries are typically calibrated using salinity data, though the ultimate purpose of the model is to model total nitrogen concentrations. Since salinity and total nitrogen are dissolved constituents, they both will exhibit similar dispersion characteristics. Salinity measurements are commonly used to determine the dispersion coefficients of estuaries (e.g., as in the general method and examples provided by Fischer,

et al, 1979). This is a valid assumption because the modeled systems do not have strong gradients in salinity or nitrogen concentrations, which makes turbulent mixing the dominant dispersive phenomenon in the modeled estuaries. Therefore, dispersion coefficients determined for salinity are appropriate for total nitrogen.

9. The Linked Model Approach (standard protocol with attenuation) was able to predict observed embayment nitrogen levels with percent errors less than 10% in 13 of 15 cases. Similarly, for Great and Green Ponds, the BBP and CCC also yielded good fits to the measured nitrogen levels with percent errors generally less than 10%, but had difficulties in Bournes Pond, likely a result of not accounting for benthic regeneration. The Waquoit Model yielded an exceedingly poor fit to observed nitrogen levels. The Waquoit Model underestimated nitrogen levels by an average of 35% (range: 27%-57%). Based upon these results the Waquoit Model is not recommended for use by the Massachusetts Estuary Project. Based upon the ability to predict the actual nitrogen levels in a consistent fashion across all of the embayment the Models rank as follows (best to worst): Linked>BBP>CCC>>>Waquoit. The fit appears to be improved if benthic regeneration is added to the BBP and CCC Models.

10. The BBP Critical Nitrogen Loads were found to vary in near direct proportion with alteration of the residence time (r) employed (0-10 days). In addition, since the upper third of an estuary has no volumetric or functional significance, the focus on its flushing rate may not always yield protective or meaningful results. More significant is that almost all embayments are sub-embayments to a larger bay and some embayments have multiple upper sub-embayments. This approach is open to manipulation of the critical loading limit through selection of flushing rate (e.g., use of the flushing rate for a sub-embayment versus the flushing rate for the system).

11. The critical nitrogen limits based upon the BBP Approach do not consistently approximate measured habitat health conditions. Generally, the generated limits tend to over-estimate the loads which a system can tolerate. It is likely that the poor fit is due to the non-inclusion of benthic regeneration and the lack of the nitrogen load spatial distribution along the estuary.

12. The overall result of the sensitivity analysis is that the Linked Model predictions of embayment nitrogen level and distribution are relatively robust. The Model is most sensitive to (in the following order of most to least sensitive): watercolumn dispersion > source water nitrogen concentration > benthic regeneration, septic load > attenuation, fertilizer, impermeable surfaces. The effect of varying the watershed nitrogen loading or attenuation terms was largest in the upper reaches of the embayment and diminished toward the inlet. The effect is seen both as a reduction in the percent change and the nitrogen concentration change. Dispersion was also most sensitive to upper estuary processes. This pattern is due to the increasing dominance of inflowing tidal source waters near the inlet versus the dominance of watershed processes in the upper reaches of embayments. This latter effect is demonstrated by the results of varying the source water concentration, which results in large (20%) changes in nitrogen levels near the inlet and diminishing effects in the upper estuary. Benthic regeneration tended to show the largest changes at mid-estuary.

13. Once the Linked Model has been calibrated and validated to existing estuarine conditions, it provides a powerful management tool to evaluate various nitrogen loading scenarios. Example case studies indicate the expected nitrogen concentration changes, as well as the associated shifts in ecological health, for alterations to septic loading in Great, Green, and

Bournes Ponds. The Linked Model can be run under user selected nitrogen management scenarios, to evaluate the most cost effective watershed management alternative for estuarine protection/restoration. These “what if scenarios” play a central role in both local decision making and the larger TMDL process. In addition, projected nitrogen concentration shifts associated with modifications to inlet channels can be used to assess potential impacts resulting from either dredging or structural modifications (e.g. jetty configuration or culvert redesign) to an estuary. This latter model application supports engineering design and feasibility analysis for both embayment and wetland restoration.

Next Steps:

1. A Quality Assurance Project Plan (QAPP) is being developed which will include the parameters required for the implementation, calibration, and validation of the model for an embayment system. As part of the QAPP, there will be the site-specific information required for the field data to be collected to fill data gaps for each of 89 embayments within the project area.

2. Embayments will be prioritized for model implementation generally based upon: the existence of appropriate water quality monitoring data, existing data on model parameters, current and future nutrient related health, local support (municipal, NGO, citizens), and regulatory needs (e.g. permit development, 303d listing).

3. Outreach to regional and local stakeholder groups and organization of stakeholder committees is being formalized and expanded.

4. The Massachusetts Estuary Project is currently in its implementation phase (Phase III) and is initiating modeling of the first series of embayment systems.

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I. INTRODUCTION

Coastal embayments throughout the State 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 are grappling with Comprehensive Wastewater Planning and/or environmental management issues related to the declining health of their estuaries. The 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. For example on Cape Cod, several towns (e.g., Chatham, Falmouth, and Mashpee) are in the midst of determining the nitrogen sensitivity of their embayments as part of wastewater facilities planning. 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.

DEP, SMAST, and others have partnered to address these planning and methodological issues, as well as to assist coastal communities in the goal of watershed-embayment management. This report is an evaluation of existing approaches and is aimed at ensuring that the proper assessment, modeling and management approach is applied to these embayment systems throughout the on-going multi-year effort.

Goals of Current Model Evaluation and Report: The Massachusetts DEP and SMAST with collaborating agencies have undertaken the task of providing a quantitative tool for watershed-embayment management for communities throughout Southeastern Massachusetts. This Massachusetts Estuary Project is founded upon science-based management. The Project will use a consistent, state-of-the-art approach throughout the region's coastal waters and provide technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the project is to provide technical guidance to DEP in support of policies on nitrogen loading to embayments and to conduct nitrogen TMDL's. In appropriate estuaries, TMDL's for bacterial contamination will also be conducted in concert with the nutrient effort (particularly if there is a 303d listing). However, the goal of the bacterial program is to provide information to guide targeted sampling for specific source identification and remediation. As part of the overall effort, the evaluation and modeling approach will be used to assess available options for meeting selected nitrogen goals, protective of embayment health. The major Project goals are to:

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

The Massachusetts Estuary Project is comprised of four phases relating to project design, project development, implementation of approach and application of management models to on-going management issues. The project phases are:

- Phase I: Assemble a working group, design the project organizational framework, evaluate existing management models and select appropriate approach for regional implementation, and survey existing data sources as to potential to support selected approach.
- Phase II: Determine the prioritization procedure and select initial embayments, promote water quality data collection in embayments with insufficient baseline data, educate local stakeholders as to Project goals, approach, results and data needs and complete the assessment of existing data and data gaps. Also, establish necessary regulatory and stakeholder committees and increase the analytical capability of the Project Team relative to collection of field data needed to support the management approach.
- Phase III: Implement embayment management approach on a 2 year cycle, which includes field data collection, modeling, reporting and a significant level of outreach. Year 1 focuses on site-specific data collection to fill data gaps, Year 2 focuses on modeling, synthesis and evaluation of management options.
- Phase IV: Keep quantitative models and embayment specific management approaches "alive" for future DEP and other management and planning needs. Also, provide a platform (upon request) for tracking embayment changes.

This Report is a fundamental part of Phase I of the Massachusetts Estuary Project. Before implementing a specific approach to support nitrogen management, it is necessary to evaluate current watershed and embayment nitrogen management models as to their accuracy, data needs, comparability, and applicability across embayment types. The results of such an evaluation both guide the selection of an approach and help to gauge the level of certainty associated with its application. At present, three approaches have been applied to serve planning or regulatory needs relating the shallow enclosed embayments typical of Southeastern Massachusetts:

- Buzzards Bay Project Nitrogen Loading Methodology (BBP),
- Cape Cod Commission Nitrogen Loading/Critical Loads Methodology (CCC),
- Linked Watershed-Embayment Modeling Approach (Linked).

In addition, the Waquoit Bay Model, a theoretical nitrogen flow approach, was compared for the one case study available.

A Case Study approach was used to evaluate and compare the available nitrogen management models. The Case Studies were selected from among the ca. 15 embayments in Southeastern Massachusetts for which we have developed the data to support the higher level Linked Modeling Approach. The overall assessment included:

- comparison of watershed nitrogen loading results from each model and resultant embayment nitrogen distribution based upon the Linked Model;
- evaluation of predicted critical nitrogen loading thresholds (BBP) relative to resultant embayment nitrogen distribution based upon the Linked Model;
- sensitivity analysis for the Linked Watershed-Embayment Approach.

It should be noted that the use of the Linked Model in each case to portray the nitrogen distribution within each embayment resulting from the BBP and CCC models (and Waquoit Model) is necessary, because among the available approaches, only the Linked Model has this capability. In general construct, the Linked Model uses a watershed land-use loading approach similar to the other models, but it also is coupled to a numerical hydrodynamic model, which encompasses the circulation and dispersion of nitrogen within the receiving waters. This linkage of watershed and embayment not only provides for assessment of specific areas within embayments, but also allows for calibration and validation approaches not available to the other models.

II. METHODOLOGY

The various nitrogen loading approaches were compared using Case Studies focusing on specific embayments within the Towns of Falmouth and Chatham on Cape Cod. These embayments were selected because:

- the embayments had sufficient data to parameterize each of management models;
- the embayments used are similar in structure, nitrogen loading, and hydrodynamics to those throughout Southeastern Massachusetts;
- the results from evaluation of these embayments are immediately transferable to other embayments throughout the region.

The nitrogen management model evaluation and analysis is described in the sections below. Specifically, the description includes:

- major Nitrogen Modeling Approaches in use for watershed management,
- case study embayments where these approaches were compared,
- construct of the sensitivity analysis of the Linked Watershed-Embayment Model.

The comparative application of the various models to the case study embayments also provides an analysis of the consistency of model results between systems. This latter point is critical in evaluating a model for use by the Massachusetts Estuary Project, which will cover all 89 embayments of Southeastern Massachusetts. Given the specific regional nature of the project (all embayments in Massachusetts from Duxbury to Mt. Hope Bay, including Cape Cod, Nantucket, and Martha's Vineyard), the evaluation and selection of an appropriate model must focus on its utility in these specific systems. The models, in this evaluation are directly applicable to shallow (generally <5m), primarily vertically mixed (only supporting periodic short term stratification), enclosed or semi-enclosed embayments, surrounded by permeable watersheds with significant groundwater discharges. The approaches can also be used in Mt. Hope Bay, a deeper estuary which supports periodic strong salinity stratification, but this will require additional parameterization and complexity of the underlying hydrodynamic model component. The results of the various model evaluations, presented in Section III below, are directly applicable to 88 of the 89 embayments within the project area. Application of these models to Mt. Hope Bay will require a separate specific uncertainty analysis (note that SMAST has a 5 year eutrophication modeling project underway in this bay).

A. Description of Nitrogen Modeling Approaches

The focus of the model evaluation is to use comparative analysis to indicate the similarities and differences in the existing modeling approaches, as well as to demonstrate their various strengths and weaknesses. The primary nitrogen management models which were evaluated (BBP, CCC, Linked) have been applied in a variety of embayment watersheds over the past decade.

Watershed nitrogen management "models" have been in use for several decades. Many of these models have developed in regions that have a significant groundwater component to their freshwater hydrology. Nitrogen transport within these "groundwater" watersheds is more difficult to quantify than through surface water dominated systems. Surface water systems are generally managed by quantifying conditions within or at the discharge of rivers. Surface waters are exposed and allow for relatively simple approaches to direct measurement. In contrast, groundwater systems have more diffuse transport and discharge pathways; therefore, require different approaches.

The watersheds to embayments within Southeastern Massachusetts generally have significant groundwater components. In addition, it is clear to most regulators, citizens, and scientists that the embayments into which the groundwater flows have declining water quality. The ongoing ecological decline has prompted an active regional effort to begin to develop and implement watershed management approaches. The current management approaches have been developed, based upon established need, over the past 15 years. While these management approaches can be applied to any system, they tend to be based upon regional or site specific waters quality issues that increase their accuracy in our applications.

The initial implementation of the BBP and CCC watershed management models was based primarily upon watershed land-use assessments and embayment hydraulic residence time (a simple flushing parameter). In the early 1990's, there was a real and immediate planning and regulatory need for methodologies which could assist in watershed nitrogen management. These approaches were important in fulfilling this need. To support the development of these (and other) approaches, there was a large effort by our group and others to develop an improved understanding of nitrogen loading for a variety of land-uses and the transport characteristics which dominate nitrogen inputs to Massachusetts estuaries. While our research team continues to refine these parameters, the basic components have been well tested and evaluated.

The Linked Watershed-Embayment Management Modeling Approach represents the "next generation" of nitrogen management approaches. It fully links watershed inputs with embayment circulation and nitrogen characteristics. The Linked Model builds on the same basic watershed nitrogen loading approach used in the BBP, CCC models, and other relevant models. The Linked Model differs from the BBP and CCC nitrogen 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.

1. Buzzards Bay Project Nitrogen Loading Approach.

The BBP Nitrogen Loading Approach is based upon using GIS to determine the number and/or area of each major land-use type within a coastal watershed and then applying a standard nitrogen load per land-use unit (Costa et al. 1994). The approach multiplies each land-use by its standard nitrogen load and adds them cumulatively across the watershed (Costa 2000). This approach yields a total nitrogen load from the watershed to the embayment if (1) the spatial data is correct (i.e. the number of houses, occupancy, lawn sizes etc.), (2) the nitrogen loadings from each land-use are appropriate for the region, (3) there is no attenuation of the nitrogen during transport to the embayment. Prior to 2000, the BBP model did not provide for attenuation of nitrogen during transport; however, more recently (based upon various nitrogen attenuation studies by our group and others) a standard 30% attenuation factor is applied for sources more than 1 km groundwater transport distance to an embayment (Waquoit Model) and for cases where nitrogen passes through a pond, stream, or wetland prior to discharge to the embayment system (simplified from Linked Model).

The BBP Approach also provides a method for evaluating whether a watershed load will have negative effects upon a receiving water body due to an exceedence of the system's nitrogen assimilative capacity (Costa et al. 1999). The BBP has tied critical nitrogen loading levels to the Commonwealth's SA/SB/ORW classification (310 CMR 4.04, 4.05(4)). The concept is that watershed nitrogen loading to an embayment affects water quality, if the enrichment is to a level that causes eutrophication and habitat degradation. Ecological habitat degradation is then tied to the SA/SB/OWR classification. However, the Massachusetts regulations, currently in place, focus on water quality from the standpoint of economic and recreational uses (fish, shellfish, swimming) and point source discharges, rather than habitat quality or nutrient standards. Because the linkage of nitrogen thresholds and the regulatory classifications are only now being developed and since the refinement of critical nitrogen thresholds is not within the scope of the present project, we will only address the BBP classifications as it relates to the BBP Models. In Phase II and III of the Massachusetts Estuary Project, we will be assessing the habitat health response of embayments to different nitrogen levels. This assessment is aimed at defining appropriate embayment specific nitrogen targets relating to ecological response, rather than regulatory levels of SA, SB and ORW. This effort will support community decision -making relative to potential nitrogen management options.

The BBP uses an estimate of embayment flushing in determining management loading levels. The flushing term selected is "residence time". However, there are a variety of residence times for a given embayment which have been used in the BBP Approach: whole system residence time (average of whole system), upper 1/3 of embayment-system residence time (average time for water from upper embayment to exit to adjacent coastal source waterbody), and upper 1/3 of embayment local residence time (average time for water from upper embayment to exit to the down-gradient region of the greater embayment). While there appears to be some lack of consistency as to the residence time term used, the BBP tends to use upper estuary residence time or "flushing rates". The BBP is clear in its guidance (at least for embayment region), "turnover times for the upper third of the estuary were employed for our calculations". However, recently there has been some question as to whether the upper third of the embayment based upon volume or area is the better indicator. In addition, the BBP has recently (Costa 2000) reduced its threshold by 14%, 25%, and 50% for SB, SA, and OWR waters shallower than 2 m mean depth and 20%, 23%, 42% for SB, SA, and OWR waters greater than 2 m mean depth. These changes result from field studies which found that BBP Model predictions significantly overestimated the ability of embayments to tolerate watershed nitrogen loads. The changes result in proportional reductions in the calculated allowable nitrogen loading rates.

The upper estuary residence time is related to the whole system nitrogen load by a simple equation to yield the nitrogen management threshold. Some of the major shortcomings with this approach include:

- the possibility of obtaining highly variable results through selection of different spatial regions for residence time;
- the use of residence time which does not take into account dispersion;
- combination of whole system nitrogen input with residence time of only upper system;
- no accounting for internal nitrogen cycling within embayment;

In addition, there is currently no site-specific calibration or validation procedure required for the use of this methodology.

2. Cape Cod Commission Nitrogen Loading Approach.

The Cape Cod Commission Approach differs little from other watershed nitrogen loading calculations, although it was developed somewhat independently. In addition, the Commission has conducted some of the site specific studies needed to determine nitrogen loading rates from various regional land-uses. The CCC approach also has adopted the BBP critical loading approach (nitrogen mass loading) and used it to guide watershed management decisions. In addition, the CCC has used the nitrogen thresholds approach of the Linked Model, which is based upon nitrogen concentrations produced by the interaction of watershed nitrogen loads and embayment circulation and mixing. The shift to embayment concentration standards was likely associated with the CCC's finding that 'the BBP designations and recommended limits appear to be associated with greater impacts than one would associate with state regulatory descriptions of coastal water quality' (Eichner & Cambareri 1998, p.13). The CCC also uses a variety of flushing approaches to determine residence time for its threshold determinations.

Both the BBP and CCC recommend more sophisticated and comprehensive models for facilities planning and to guide watershed management decisions. The value of their current land-use approaches is that they can generate loading guidance with little field effort (and at low cost) and can be used when situations do not allow the time to conduct a quantitative modeling approach. The Linked Model can be employed under the same circumstances, but requires a field program and the time to accomplish it. However, once data needs are met, the Linked Model can be used at any future time with little additional effort and at low cost. The Massachusetts Estuary Project's focus is to fulfill the need for the site-specific data collection necessary to allow the use of this more sophisticated, quantitative approach for watershed-embayment management.

3. Linked Watershed-Embayment Nitrogen Model Approach**.**

Scientists comprising the SMAST Coastal Systems Program and their colleagues have been developing watershed nitrogen loading models and embayment water quality models for shallow embayments, since the mid 1980's. The theoretical aspects of this work have been funded by the National Science Foundation, NOAA, and EPA, while the adaptation, parameterization and practical application to nitrogen management issues has been principally through DEP, DOD, Foundations and municipalities. The numerical hydrodynamic and water quality models used for the approach were developed by U.S. Army Corps of Engineers. Over the past decade we have applied the approach in a variety of sites throughout Southeastern Massachusetts to address site specific nitrogen management problems (wastewater discharges, agricultural impacts, fisheries, tidal restrictions, etc.). The Linked Watershed-Embayment Model in its basic form has been applied in about 15 embayments, primarily to guide nitrogen management and wastewater planning.

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. This methodology integrates a variety of field data and models, specifically:

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

a. Monitoring

Over the past decade, in which we have been refining and applying this quantitative Linked Model, issues regarding nitrogen modeling have become clear. Early in the process it became clear that the need for multi-year nutrient-water quality data collection on a wide variety of embayments presented a major obstacle. To address this issue, SMAST helped to establish a large embayment monitoring effort in collaboration with local stakeholders, municipalities, regional and state agencies. Currently, almost all of the embayments in Southeastern Massachusetts have ongoing monitoring or are planning programs for start-up in 2001. SMAST provides the technical guidance, analytical facility and synthesis for more than 95% of these monitoring programs. In addition, SMAST with its collaborators will have completed quantitative nitrogen loading assessments for almost 15 embayments by March 2002 and has assembled much of the available data for many of the larger remaining systems.

In general, calibration of the total nitrogen embayment model requires both spatial and temporal (during the critical summer months) measurements of water column nitrogen concentrations. For some systems within Southeastern Massachusetts, more than 10 years of high quality data exists. However, other systems have relatively little data or only sporadic data sets that do not represent long-term summertime conditions. Therefore, a multi-year nutrient monitoring effort needs to be performed in conjunction with the other tasks in the linked approach to ensure a high-quality embayment nitrogen data set.

b. Hydrodynamic Modeling

A number of different modeling approaches are available to evaluate flushing characteristics within tidal estuaries. Models range from simple box models based on dye or salinity data, to complex three-dimensional hydrodynamic models. Complex models (two- and three-dimensional hydrodynamic models) yield a detailed multi-dimensional representation of time-varying estuarine currents and tides. Calibrated properly with field data, complex hydrodynamic models provide a more accurate representation of estuarine flushing than simple box models. A hydrodynamic model also provides a useful tool for additional applications, such as evaluating the effects of dredging, culvert redesign and modeling water quality. Since an assessment of water quality is an integral part of the Linked approach for assessing the existing ecological health of estuaries and evaluating potential methodologies for improving estuarine water quality, use of an accurate numerical hydrodynamic model is necessary.

The technical approach for evaluating estuarine hydrodynamics includes model development and calibration. The Surface Water Modeling System (SMS) will be utilized to model both hydrodynamics and water quality (an overview of the SMS package will be provided upon request). The SMS package provides a user-friendly grid generation program and post-processing software for the RMA-2 model developed by Resource Management Associates (King, 1990). The RMA-2 model is a two-dimensional, depth-averaged finite element model, capable of simulating hydrodynamics in complex river and estuary systems. The model has been standardized by the U.S. Army Corps of Engineers and is widely accepted for a broad range of estuarine applications. Members of the team have used RMA-2 for numerous hydrodynamic studies on Cape Cod in a range of embayment types from simple to complex.

The model allows for variable computation cell sizes to take advantage of limited bathymetric information. This type of model was developed to handle complex flow patterns associated with systems like those along the southeast coast of Massachusetts. Output from the model provides complete depth-averaged, two-dimensional current velocities for all computational nodes as a function of time within the estuary system. Our modeling approach is a three-part process:

- Development: Set-up the model for each embayment system. Input shoreline positions, bathymetry data, and specify boundary conditions (i.e., tides measured in the Atlantic Ocean, Nantucket Sound, Cape Cod Bay, or Vineyard Sound).
- Calibration: Ensure model predictions are consistent with natural processes using a thorough comparison to measured field data (ADCP, dispersion, etc).
- Application: Use the calibrated model to evaluate hydrodynamic characteristics and to determine potential improvements to flushing associated with engineering alternatives.

Flushing rates and residence times are computed easily from the wealth of detailed site-specific data this model will provide. Both *local* and *system* residence times can be computed. Local residence times represent the average time required for a parcel of water in a sub-embayment to be flushed out of the sub-embayment, and will be computed as the volume of water in each sub-embayment divided by the volume of water entering the sub-embayment over an average tidal cycle (tidal prism). System residence times represent the average time required for a parcel of water to be flushed out of the estuary from the sub-embayment, and are computed as the volume of water in the estuary divided by the tidal prism of the sub-embayment. The Linked Approach allows calculations based upon the BBP approach based upon residence time and therefore can be used to refine previous critical loading decisions.

Since the flushing analysis uses a numerical hydrodynamic model, results can be directly incorporated into a water quality model to indicate nutrient or other constituents (pollutants, bacteria) conditions within the estuarine system. The water quality model (RMA-4) is implemented by incorporating the nutrient loading data and other data collected as part of the Linked Approach with the two-dimensional hydrodynamic model. The water quality model, therefore, integrates estuarine hydrodynamics, point and non-point source nutrient (bacterial) inputs, and constituent transformations within the estuary. In addition, the hydrodynamic model is used to evaluate potential improvement of flushing rates as a result of channel dredging and/or other engineering modifications (e.g. alterations to culverts). This latter application when integrated with the water quality model, further provides the ability to assess the effects of enhanced tidal flushing to lower the effective nitrogen load to an embayment (at no additional cost). In addition, the Linked Model can then be used for inlet design and engineering with no additional data collection, if flushing alteration is a selected option, another benefit to employing a holistic modeling approach.

To ensure a calibrated hydrodynamic modeling effort, field data in the form of local bathymetric mapping, tidal measurements, and sometimes current velocities, are required. First, measurements of tidal elevation are made over a 28 day lunar cycle at 6-8 locations within the complex embayments and 2-5 locations within the more typical and simpler estuaries. Precise knowledge of tidal elevation time histories at strategic points is required to understand the flushing characteristics throughout the system. For example, as the tide rises in Nantucket Sound, water floods into Stage Harbor (Chatham) and is distributed through to the upper portions of each sub-embayment. Characteristics of the embayments, such as inlet size, depth, and bottom roughness (friction), distort the tidal wave as it travels through the system from Nantucket Sound. Tidal distortion may include a reduction in the range of the tide (damping) and/or a delay in the time of occurrence of low and high tides. The extent of tidal damping through the system affects the volume exchange, and hence the flushing characteristics, between the Sound and the remote portions of the system. Measurement of the tidal elevation time histories at key locations provides information to calibrate and verify the two-dimensional numerical model, which is used to evaluate flushing characteristics and compute residence times (and is the underpinning of the water quality modeling component).

Measurement of the tidal response in portions of each embayment quantifies the extent of tidal damping through the connecting channels and coves. Spatially distributed placement of gauges within each system allows for detailed analysis of tidal damping throughout the system. Each gage is installed to an existing pier piling or other stable structure such that the recording sensor remains submerged at all stages of the tidal cycle. Tide gages are located at key locations within an embayment system to determine the degree of tidal attenuation through various channels.

The bathymetric data is acquired utilizing a portable bathymetric data acquisition system. This system is currently in-place due to capacity building under Phase II of the Massachusetts Estuary Project. The system links a fathometer with survey-quality depth resolution to a differential geographic positioning (DGPS) system. This system records depth as a function of survey location. DGPS offers unparalleled positioning precision for the cost, typically yielding absolute horizontal accuracy to within 2 meters over 95 percent of the time. An integrated survey software package is used to record depth and DGPS data simultaneously to a laptop computer.

In addition to bathymetric and tidal data collection, the level of numerical modeling needed for some of the complex embayment systems (e.g. Stage Harbor, Chatham) warrants the collection of tidal current data. Tidal cycle measurements typically are performed utilizing a boat-mounted Acoustic Doppler Current Profiler (ADCP). This instrument is capable of measuring the current velocity distribution (direction and speed) throughout the water column. Cross-sectional transects are run continuously over a 12.5 hour tide cycle to measure tidal prism. These tidal current measurements are then used as an independent validation of the hydrodynamic model.

c. Watershed Nitrogen Loading Approach

The total nitrogen input to each embayment is determined using a land-use loading model. This model is similar in principle to the land-use models used by the Cape Cod Commission and the Buzzards Bay Project and has been accepted by the general estuarine community for over a decade. The loading model uses land-use distributions and loading rates specific to each land-use to determine total system loading. The land-use loading terms that we employ are all from regional studies conducted in similar aquifer materials. Attenuation of nitrogen during transport through the aquifer and surface freshwater systems of the watershed is generally based upon site specific data collection. This attenuation of nitrogen is a critical part of the land-use model used in the Linked Model. Omitting attenuation of nitrogen can result in significant errors in the determined nitrogen loads, resulting in incorrect management decisions.

The watershed for each embayment is divided into regions a) contributing directly via groundwater to the estuaries, b) contributing to freshwater lakes and ponds, c) contributing to fresh and saltwater wetlands (and the size of the wetlands), and d) contributing areas to streams and rivers. A full land-use loading model is applied to each of these sub-watersheds (where they exist) to determine the spatial distribution and amount of nitrogen loading to the estuarine portions each watershed-embayment system.

In support of the loading models is data collection on land-use and its validation, physical and biogeochemical data collection on the freshwater lakes and ponds, and the spatial and areal distribution of wetlands. The land-use modeling relies on GIS approaches, but field data collection is also generally required. Lake and pond field data collection of stratification, nutrient and chlorophyll levels is important for determining attenuation rates. During summer 2000, the CCC and SMAST conducted a “snapshot” survey of most of the lakes and ponds on Cape Cod to determine their trophic status. These data will be coupled to physical data to form a freshwater reference for use by the Massachusetts Estuary Project.

In addition to freshwater systems, salt marshes play an important role in the estuary as sites of denitrification, the transformation of plant available nitrate to unavailable nitrogen gas. Since groundwater and surface freshwater typically flow through salt marshes before entering the estuarine systems, this process can significantly reduce the nitrogen load from the watershed to the estuary ((Howes et al. 1996 and West Falmouth Harbor). Where appropriate (based upon system structure), a salt marsh survey is conducted to determine the extent of this nitrogen interception for each embayment. Measurements include salt marsh distribution and nitrate attenuation during low tide flow in creek bottoms. Data collection is focused on warmer months, when embayments are most sensitive to nitrogen loading and habitat quality parameters typically exhibit their annual minima.

Attenuation of watershed nitrogen by wetlands, through the activity of their natural denitrifying communities, is also evaluated as a potential management option for lowering nitrogen discharges to receiving embayment waters. In other nutrient management work, we have recommended restoration of riparian wetlands or salt marsh restoration as “soft” solutions for embayment restoration. This approach provides another management tool, based upon ecological engineering, to be evaluated in concert with the variety of standard nitrogen management tools used for embayment nutrient management planning.

In most estuarine systems in Southeastern Massachusetts, attenuation of watershed nitrogen during surface water transport can significantly reduce the amount of nitrogen entering an estuary (e.g. Wareham River Estuary, Eel River, Falmouth Ponds, Bassing Harbor System, Three Bays, etc.). This attenuation of nitrogen in surface water results from natural biological processes, which denitrify or sediment nitrogen within wetlands, streams, rivers, lakes and ponds. Nitrogen transported via groundwater is also attenuated after entering the biologically active regions of these surface water aquatic systems. To account for this attenuation, nitrogen and phosphorus inputs from each major stream/river are measured. These direct measurements of watershed freshwater and nutrient inputs a) serve to validate the watershed delineations, b) validate the land-use models, c) help to calibrate the hydrodynamic models applied to estuarine circulation. These direct measurements are also applied to the system nitrogen loading models. The use of the surface water inflow data to validate the loading and hydrodynamic models differs from studies relying on land-use data alone. Unfortunately the land-use only studies have often resulted in overestimates on nitrogen loading and distribution and lead to improper or ineffective management options.

Surface water inflow volume and mass transport of nitrogen and phosphorus is determined using site-specific flow discharge relationships and continuous records of water levels (CDM and Howes 2000). Nutrient samples are collected for inorganic and organic nitrogen and phosphorus concentrations using automated water samplers deployed over a 12-month period. Total nitrogen (not just inorganic forms) is required to accurately determine the input through this pathway.

In addition to "new" nitrogen entering the estuary from the surrounding watershed, nitrogen is recycled within the sediments and watercolumn. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that we have investigated, recycled nitrogen can account for about half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. The measurement of summer nitrogen regeneration incorporates the effects of nitrogen deposited during other periods of the year. This non-summer nitrogen deposition, typically as organic nitrogen, is critical to determining the summer loading through regenerative processes. In essence the sediments serve as a "nitrogen battery", accumulating nitrogen during the cooler months for release during summer when decomposition and bioirrigation are highest. Failure to account for this recycled nitrogen generally results in significant errors in determination of threshold nitrogen loadings. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data is needed to determine the best approaches for nitrogen mitigation.

The organic rich nature and relatively shallow waters of coastal systems, like many of those in Southeastern Massachusetts, result in sediments playing a significant role in system biogeochemical cycles (Ramsey et al. 2000). Organic matter deposition to sediments, hence benthic respiration, tends to decrease with increasing depth of overlying waters due to interception by watercolumn heterotrophic processes. The result is that Harbor respiration rates are typically many fold higher than in the adjacent offshore waters. With stratification of harbor waters, sediment metabolism plays a major role in bottom water oxygen declines (an ecosystem structuring parameter).

In order to determine the contribution of sediment regeneration to nutrient levels within embayments during the most sensitive summer interval (June-August), sediment samples are collected and incubated under *in situ* conditions (following Quality Assurance Project Plan procedures of Cibik and Howes 1995). Cores are collected by SCUBA divers to ensure their integrity and are incubated in a portable field laboratory to prevent disturbance from transport. Time series measurements of total dissolved nitrogen, nitrate+nitrite, ammonium and ortho-phosphate are made on each incubated core sample. Rates are determined from linear regression of analyte concentrations through time. The rates of oxygen uptake by the sediments and watercolumn are also be determined in order to: (1) allow evaluation of the sensitivity to oxygen depletion of each embayment area under periodic stratification, (2) allow ranking of sediments as to organic matter deposition rates (not possible using organic content), and (3) to parameterize nitrogen models. Up to 16 sites are measured in the more complex systems and between 4-12 in the more typical and simple systems.

The results from the regeneration studies allow the spatial pattern and rate of nutrient inputs from the sediments to the water column to be entered into the nutrient model. From our experience, sediment regeneration during the summer is a large and important source of nutrients supporting both phytoplankton and macroalgal blooms in embayments throughout Southeastern Massachusetts.

The incorporation of recycled nitrogen in the determination of total nitrogen loading to embayments during the critical summer interval is another significant component incorporated in the Linked approach. Recycled nitrogen is not accounted for in the BBP and CCC approaches, which include only watershed loads (and sometimes rain). However, all established coastal and estuarine water quality models (as well as river transport models like Qual 2E) recognize the importance of and incorporate this term. In addition, assessment of benthic regeneration also allows for an independent evaluation of the potential for the development of bottom water hypoxia (before it develops) and therefore provides key habitat sensitivity information as well. Given its importance, we have developed a rapid and simplified approach to accurately measuring this primary parameter.

d. Embayment TMDL – Synthesis

The flushing model provides an indicator of estuarine health; however, it does not provide a direct quantifiable measure of water quality. To quantify the existing ecological health of estuarine systems, the anticipated magnitude of deterioration if no corrective measures are implemented, and the anticipated improvement from various corrective measures, a water quality analysis must be added to the hydrodynamic model. With proper evaluations of both sources and sinks of nutrients, a water quality model can be implemented to accurately quantify existing conditions and future scenarios.

The RMA-2 flushing model provides detailed flow and water elevation estimates throughout an estuarine system. Combining this hydrodynamic information with nitrogen loading data and reasonable predictions of nutrient sinks/sources (e.g. benthic flux of dissolved nitrogen in various embayments) will allow numerical estimates of nitrogen levels throughout the three estuaries. A water quality model, RMA-4, is utilized to link the hydrodynamic information obtained from RMA-2 directly to estuarine water quality. The RMA model is currently used by our team to model nitrogen levels within embayments. Modeling of dissolved oxygen and chlorophyll a are supported by the model, but these have not yet received sufficient application to encourage use of these additional modeling layers at this time. A survey of models for prediction of these parameters yields the conclusion that they typically predict the mean values well, but are poor at predicting the maxima and minima. Unfortunately, it is these “extremes” which modify the habitat quality of embayment systems. At present, the Linked Model approach for assessing chlorophyll and oxygen levels rely primarily upon water quality monitoring and long-term deployments of autonomous instrumentation.

To model water quality within an estuarine system, salinity variation typically is used to validate the necessary dispersion coefficients. Although salinity is not an effective tracer for evaluating flushing rates within an estuary, measurement of salinity variations is used to provide required information regarding the dispersion characteristics of each estuary. Dispersion is an estimate of a pollutant’s ability to “spread” within the embayment waters independent of advective processes. Using a conductivity-temperature-depth (CTD) instrument, salinity profiles are collected along the central axes an embayment system and offshore of each entrance channel. The CTD converts conductivity, temperature, and pressure internally to salinity using the Practical Salinity Scale of 1978 equations and constants. Once analyzed, the CTD data are used to evaluate mixing or dispersion of freshwater entering the system with the saline waters within an estuary. Based on the observed salinity values, parameters within the RMA-4 water quality model can be adjusted to calibrate the model to existing conditions (for more details on determination of dispersion based upon salinity see Section III.C., below).

To properly evaluate mixing of fresh and saltwater, freshwater inflow rates are quantified in the major tributaries of the system. For a typical embayment system, freshwater inflow is measured upstream of the brackish portion of the estuary and average annual groundwater inflow is determined from the water balance. All flow measurements take place coincident with the CTD survey described above. In this manner, the salt/freshwater balance and mixing coefficients for the water quality model can be determined.

A two-dimensional water quality model, RMA-4, is employed to evaluate existing nitrogen loading on an estuary, as well as to evaluate the potential future deterioration of water quality if no corrective measures are implemented. In addition, the RMA-4 water quality model is used to quantify anticipated improvements associated with nitrogen reduction strategies and/or alterations to flushing. This model utilizes the hydrodynamic information generated by the RMA-2 model, estuarine mixing data from the CTD surveys and freshwater inflow measurements, available nitrogen loading and groundwater flow rates, and nutrient source/sink information to quantify total nitrogen concentrations throughout the estuary. Once calibrated to existing conditions, model boundary conditions can be altered to indicate increases or decreases of nitrogen loading associated with no action or effective corrective measures, respectively. Prior to modeling of nitrogen concentrations, the water quality model is first utilized to evaluate the salinity regime within an estuary. Since salt is a conservative constituent (i.e. the only source is the Ocean), it is a simple process to calibrate the model dispersion coefficients utilizing the freshwater inflow data and the CTD survey information. Once calibrated using salinity, the dispersion coefficients remain constant for other constituents such as total nitrogen.

To properly model nitrogen concentrations, accurate boundary conditions are required to represent the various nutrient sources. First, the watershed nitrogen loading model data is utilized to establish the level and distribution of nitrogen input along the embayment shoreline for each tributary. Non-point groundwater sources are distributed along the shorelines within the appropriate sub-watershed. In addition to nutrient loading from upland sources, benthic flux of inorganic dissolved nitrogen levels are evaluated within the estuary. This benthic flux term is estimated for the critical management period, the summer months. Finally, a ‘background’ nitrogen concentration is required offshore to establish the amount of nitrogen entering each estuary during a tidal cycle. To meet this objective, we have several reference sites within Nantucket and Vineyard Sounds, Buzzards Bay, Outer Cape-Atlantic Ocean, and Cape Cod Bay.

Once the boundary conditions and the dispersion coefficients are established, the RMA-4 model is run and the results compared to available water column nitrogen data. The water quality model then can be fine-tuned to existing conditions. The calibrated water quality model forms the basis for evaluating the deterioration of estuarine water quality associated with the no-action alternative, as well as evaluating the various corrective measures.

Using the calibrated water quality model, a maximum acceptable nitrogen concentration can be determined for each sub-embayment of interest. This “acceptable” total nitrogen level is developed from the field evaluation of estuarine health (based upon habitat indicators). For scenario testing, upland nitrogen loads (both tributary and groundwater derived) can be lowered in the model until the target nitrogen concentrations within the embayment zones are achieved. Initial evaluations of corrective measures utilize this reduced loading estimate to judge applicability of each measure. Analysis of structural improvements to coastal entrances or dredging of channels use the predicted changes in hydrodynamics from the RMA-2 hydrodynamic model as input to the water quality model. Once the hydrodynamic model has been re-run, the water quality model can be used to quantify the anticipated reduction in nitrogen concentrations resulting from structural improvements. An analysis of tidal inlet restriction is one of the basic evaluations conducted in the Linked Model Approach for each embayment.

The nutrient related ecological health of each estuary is 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. These data are synthesized into an assessment of the systems present health and temporal trends (frequently based upon DEP Eelgrass Mapping Program). It should be noted that temporal trends in eelgrass are evaluated relative to the spectrum of other parameters, since eelgrass declines can result from a variety of causes. However, within almost all of the embayments of Southeastern Massachusetts the major cause of decline appears to be related to degradation of nutrient related water quality. The assessment also is used for projections of future conditions based upon the water quality modeling effort. Generally, sampling is conducted at water quality monitoring stations and the macrophyte surveys along transects throughout the embayment systems. Sampling is most intense during the critical warmer months. Depth profiles for nutrients are collected where the water is >0.8 meters at low tide. Since watercolumn monitoring is on going in each system, only periodic field samples are collected for confirmation and intercalibration with the long-term record.

Mapping of eelgrass and macroalgal communities is conducted during the summer months. SMAST has refined these survey and mapping techniques for recent studies in the Town of Chatham and for MA DEP in West Falmouth Harbor. Macroalgal mapping includes distribution and density. Macroalgae are a natural part of estuarine systems, but become "excessive" under nutrient enrichment. Existing data on the system collected over the past decades is used to assess changes in distribution or community structure. SMAST scientists have collaborated with the DEP Eelgrass Mapping Program since its inception and this collaboration will be further developed within the Massachusetts Estuary Project.

Benthic animals are a key indicator of the health of estuarine systems. Benthic sampling and analysis is performed primarily by SMAST staff, who have more than 30 years of experience with Cape Cod embayments, using techniques which are well established in the research community. Linked to the benthic analysis is the production of a sediment quality map (organic content, nitrogen, etc). The map is based upon samples collected throughout the estuary. These water quality and habitat data sets are synthesized and integrated with the modeling efforts, to support both hind and forecasting of habitat quality changes linked to watershed management.

Summary: The Linked Model uses the total nitrogen input to each embayment, determined using a land-use loading model. This model is similar in principle to the land-use models used by the Cape Cod Commission and the Buzzards Bay Project and has been accepted by the general estuarine community for over a decade. The loading model uses land-use distributions and loading rates specific to each land-use to determine total system loading. The land-use loading terms are all from regional studies conducted in similar aquifer materials. Attenuation of nitrogen during transport through the aquifer and surface freshwater systems of the watershed is generally based upon site specific data collection. This attenuation of nitrogen is a critical part of the land-use model used in the Linked Model. Attenuation was not included in the BBP Model prior to Yr 2000 and is not included in the CCC Model. The Linked Model uses measured site-specific attenuation rates, while the BBP uses a set attenuation rate. Omitting attenuation of nitrogen can result in significant errors in the determined nitrogen loads, resulting in incorrect management decisions.

The Linked Model approach incorporates recycled nitrogen in the determination of total nitrogen loading to embayments during the critical summer interval. Recycled nitrogen is not accounted for in the BBP and CCC approaches, which include only watershed loads (and sometimes rain). However, all established coastal and estuarine water quality models (as well as river transport models) recognize the importance of and incorporate this term. In addition, assessment of benthic regeneration also provides an independent evaluation of the potential for the development of bottom water hypoxia (before it develops) and therefore provides key habitat sensitivity information as well.

The Linked Model approach incorporates multiple embayment parameters in its model and assessment including: hydrodynamics, recycled nitrogen, eelgrass/macroalgal mapping, and benthic animal indicators. The Linked Model also uses site-specific salinity and nitrogen levels for calibration and validation. The significant focus on site-specific embayment parameters in the Linked Model contrasts with the other management models, which at most incorporate embayment residence time.

The Linked Model employs a two-dimensional water quality model, RMA-4 to evaluate existing nitrogen loading on an estuary, as well as to evaluate the potential future deterioration of water quality if no corrective measures are implemented. The RMA-4 water quality model is also used to quantify anticipated improvements associated with nitrogen reduction strategies and/or alterations to flushing. This model is incorporates the hydrodynamic model output, estuarine mixing data, nitrogen loading and regeneration data, and nutrient attenuation. The water quality model output is a quantitative depiction of nitrogen concentrations throughout the estuary. Once calibrated to existing conditions, model boundary conditions can be altered to indicate increases or decreases of nitrogen loading associated with no action or effective corrective measures, respectively. Prior to modeling of nitrogen concentrations, the water quality model is first utilized to evaluate the salinity regime within an estuary. Since salt is a conservative constituent (i.e. the only source is the Ocean), it is a simple process to calibrate the model dispersion coefficients utilizing the freshwater inflow data and the CTD survey information. Once calibrated using salinity, the dispersion coefficients remain constant for other constituents, such as total nitrogen.

The Linked Watershed-Embayment Nitrogen Model Approach presents a holistic approach to managing key aspects of embayment habitat quality and health. In addition to its value as a watershed-embayment management tool for assessing and evaluating management options, the approach includes a determination of the level of health or impairment of the major portions of each estuary, their sensitivity to nitrogen loading, and future conditions at build-out. The approach supports the further development and implementation of a wide-variety of management options that fits well into the TMDL process. In addition, once calibrated and validated for an embayment the model can be used to evaluate unanticipated future specific watershed or embayment alterations, at very low cost (comparable to the cost of re-runs of the BBP or CCC modeling efforts).

B. Description of Embayment Case Studies Used In Model Evaluations.

Five embayment systems were selected as case studies for evaluation of the various nitrogen management models currently in use. Since the Linked Model Approach requires quantitative data on both the watershed and the embayment beyond what is required for the BBP or CCC Models, we selected the Case Studies, from among the fifteen for which we have developed this information.

Selection of specific embayments for case studies depended on several variables including 1) the level of existing data and previously completed analyses, 2) the hydrodynamic complexity of the system, and 3) the observed "estuarine health" of the system. The estuaries selected generally cover the full range of hydrodynamic and water quality conditions observed in Southeastern Massachusetts. The following five (5) estuarine systems were chosen as case studies:

- Great Pond, Falmouth
- Green Pond, Falmouth
- Bournes Pond, Falmouth
- Stage Harbor, Chatham
- Frost Fish Creek, Chatham

These embayments cover the range of physical structures from simple (Green Pond, Bournes Pond, Frost Fish Creek), to typical (Great/Perch Pond) to complex (Stage Harbor).

1. Falmouth Salt Ponds (Great, Green, and Bournes Ponds)

The three salt ponds in Falmouth being assessed as case studies for this evaluation include Great, Green and Bournes Ponds (Figure II-1). Figure II-1 illustrates the location of water quality monitoring stations used to assess estuarine health for the past 13 years. These salt ponds are estuaries with focused freshwater inputs at the headwaters and tidal exchange of marine waters from Vineyard Sound (tide range of approximately 0.5 m) at their southern inlets. Perch Pond is a tributary to Great Pond and is predominantly influenced by the water quality of the much larger Great Pond through tidal exchange. The three main ponds are similar in length, but show a range of widths that result in their differing surface areas. Great/Perch Pond is the largest at 109 hectares (1 ha = 2.47 acres) with Bournes (62 ha) and Green (53 ha) being about half as large.

Great, Green, and Bournes Ponds are shallow mesotrophic (moderately nutrient impacted) to eutrophic (nutrient-rich) coastal ponds on the southern coast of Falmouth. These ponds are situated on the southern margin of the Mashpee Outwash Plain, consisting of deposits about 50 to 60 ft thick in the study area. The outwash material is highly permeable, varying in composition from well-sorted medium sands to coarse sands and gravels (Millham and Howes, 1994). Due to the soil permeability, rainwater runoff is low and more than half of the freshwater enters the ponds via groundwater flow. Each of the three ponds is a true estuary, acting as the mixing zone of terrestrial freshwater inflow and saline tidal waters from Vineyard Sound. Salinity in the three ponds ranges from approximately 30 ppt at the Vineyard Sound inlets to less than 10 ppt at the northern ends.

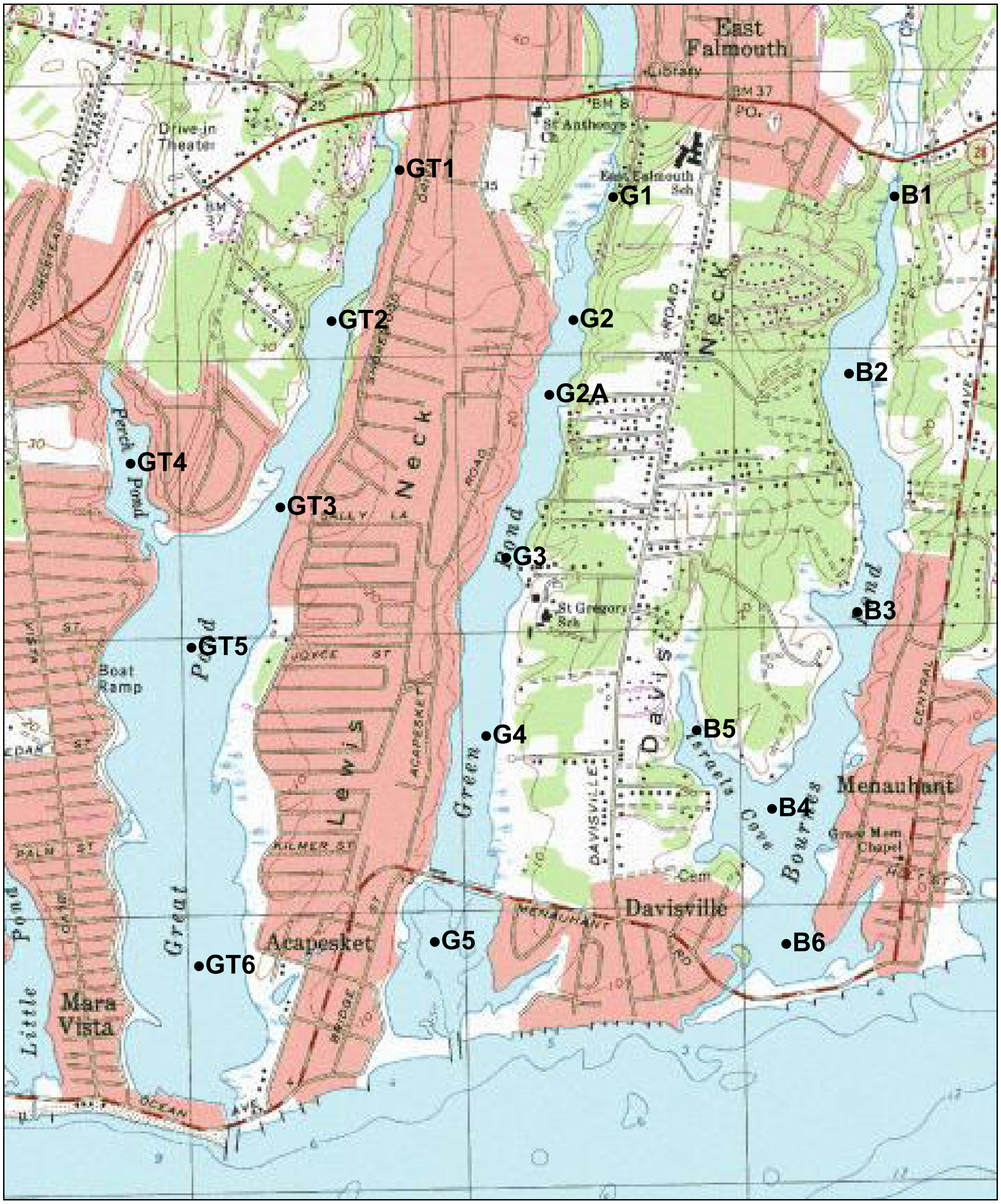

Figure II-1. Map showing location of monitoring stations used in Falmouth Pondwatch program, for Great, Green, and Bournes Ponds. All water quality parameters were sampled and assayed following the Quality Assurance Project Plan methods of Williams and Howes 2001.

To aid in the presentation of the nutrient related health of Great, Green and Bournes Ponds, various key parameters of eutrophication were integrated into a single term or Index. This Eutrophication Index was developed for Buzzards Bay (Costa *et al.*, 1992 and in press) and we have modified it slightly using recent data (Howes *et al.*, 1999). The Index is based upon transparency (measured by secchi), nitrogen concentration, chlorophyll *a* pigments, and oxygen levels (lowest 20% of samples). Best and worst average conditions for each parameter yield scores of 100 and 0, respectively. The range for each parameter utilized to develop the Eutrophication Index is as follows:

- oxygen between 90% and 40% of air equilibration
- transparency between 3 m and 0.6 m
- total nitrogen between 0.28 mg N/L and 0.61 mg N/L, and
- chlorophyll *a* pigments between 3 :g/L and 10 :g/L

a. Great Pond

There is a strong gradient in nitrogen level and health in Great Pond, with highest nitrogen and lowest environmental health in the headwaters of Great and Perch Ponds and lowest nitrogen and greatest health near the inlet to Vineyard Sound (Table II-1). Both of these upper arms to Great Pond are presently showing "Severely Degraded" water quality and "Hyper-Eutrophic" conditions. Eelgrass is absent from these regions and periodic fish kills have been reported, resulting from oxygen depletion. Upper Great Pond periodically shows macro-algal accumulations.

Table II-1. Primary indicators of eutrophication (nitrogen, chlorophyll *a*, transparency, and bottom dissolved oxygen (D.O.)) in Great, Green, and Bournes Ponds. Pond stations relate to locations in Figure II-1. Data from Falmouth PondWatch Program, 1990-98, School for Marine Science and Technology, UMass Dartmouth.

Station	Total nitrogen		Chlorophyll a Pigments		Bottom D.O./ Lowest 20%		Secchi Depth
	mean mg N/L	std. error mg N/L	mean :g N/L	std. error :g N/L	mean mg N/L	std. error mg N/L	m
Great Pond							
GT2	0.90	0.033	22.5	2.9	1.8	0.2	0.98
GT3	0.77	0.039	17.5	2.1	2.3	0.4	1.15
GT4 Perch Pond	0.88	0.026	69.8	34.3	3.9	0.6	1.17
GT5	0.61	0.034	18.4	3.8	4.9	0.2	1.34
GT6	0.52	0.030	8.3	1.8	5.3	0.2	1.24
Green Pond							
G2	1.07	0.075	65.7	13.3	2.2	0.3	0.87
G2A	0.99	0.047	38.0	4.0	3.1	0.2	0.95
G3	0.81	0.034	21.3	1.7	3.6	0.2	0.97
G4	0.60	0.015	10.3	1.0	4.8	0.1	1.15
G5	0.50	0.029	7.6	0.9	5.6	0.2	1.53
Bournes Pond							
B2	0.79	0.034	24.0	2.8	3.9	0.4	0.92
B3	0.65	0.063	9.4	2.2	4.6	0.2	1.56
B4	0.54	0.042	4.7	0.3	5.1	0.3	1.26
B5 Israels Cove	0.65	0.023	13.4	3.2	3.6	0.2	1.01
B6	0.39	0.020	3.3	0.3	6.1	0.2	0.97
Vineyard Sound	0.29	0.012			6.7	0.1	3.23

Perch Pond exhibits higher nitrogen levels than its adjacent source waters primarily due to the shoaling of its short inlet to central Great Pond. Perch Pond has experienced increased nitrogen levels, 1994 through 1997 compared to 1990 through 1993. Maintenance of the Perch Pond to Great Pond tidal flow has historically been a problem (see Case Study Section IV.D. below). Past monitoring of benthic animal populations in Perch Pond have indicated a near extinction occurring with the onset of summer low oxygen conditions. This effect is related, in part, to the periodic shoaling of the tidal channel, but primarily results from nutrient enrichment of this system.

The physical structure of Great Pond results in the main basin being in the "Significantly Impaired" class of waters based upon nitrogen and in the moderate level based upon the Eutrophication Index (Figure 2). This indicates that the high nitrogen levels within the main basin have not degraded other portions of the system to the levels observed in neighboring Green Pond. The major difference is in oxygen levels. The greater surface area of the lower Great Pond basin appears to allow wind mixing of the waters, thereby increasing aeration. Due to this aeration, the high chlorophyll pigment concentrations found in the upper main basin (>15 :g/L) are not matched by low oxygen levels. Due to the oxygen conditions, Great Pond still supports highly productive shellfish beds and small areas of eelgrass. Scallops were observed within the pond in 1998.

b. Green Pond

There is a strong gradient in nitrogen levels and environmental health within Green Pond, with highest nitrogen at the estuarine headwaters (approximately 1 mg N/L) and decreasing concentrations to a low in the boat basin between the bridge and the inlet (approximately 0.5 mg N/L). As a result of high watershed nitrogen loading, Green Pond is currently showing Severe Degradation (N>0.7mg/L) over the entire upper two-thirds of its length (Table II-1). All eelgrass is absent in this region. There are macro-algal accumulations that smother shellfish and other bottom-dwelling animals. Large phytoplankton blooms (>20:g/L Chlorophyll a) are typical summer occurrences. These blooms result from the high nutrient availability and cause low watercolumn transparency (secchi depth <1 meter) and oxygen depletions to stressful levels (<4 mg/L, Figure 22). Fish kills related to periodic hypoxia occur almost every year.

The lower third of Green Pond supports "Significantly Impaired" waters just north of the bridge, grading to "Moderately Impaired" waters in the lower boat basin. Above the bridge eelgrass is absent, while below the bridge some small patches remain. This region of the Pond (near the bridge) supports large concentrations of quahogs and soft-shell clams. These shellfish are likely benefiting from the high phytoplankton production from the upper pond. Chlorophyll a levels within this region are typically ≤10 :g/L, half (or less) than the upper pond.

Summarizing the nutrient related health of Green Pond using the Eutrophication Index (described above), indicates a continuous gradient of declining pond health moving from Vineyard Sound to the Pond's headwaters (Figure 2). All of the key eutrophication indicators support this conclusion. The levels from Stations G2, G2A and G3 would rank the health of these areas in the lowest 20% of embayments monitored in Buzzards Bay. The lower Stations are still capable of supporting benthic animal populations and shellfish, but not eelgrass.

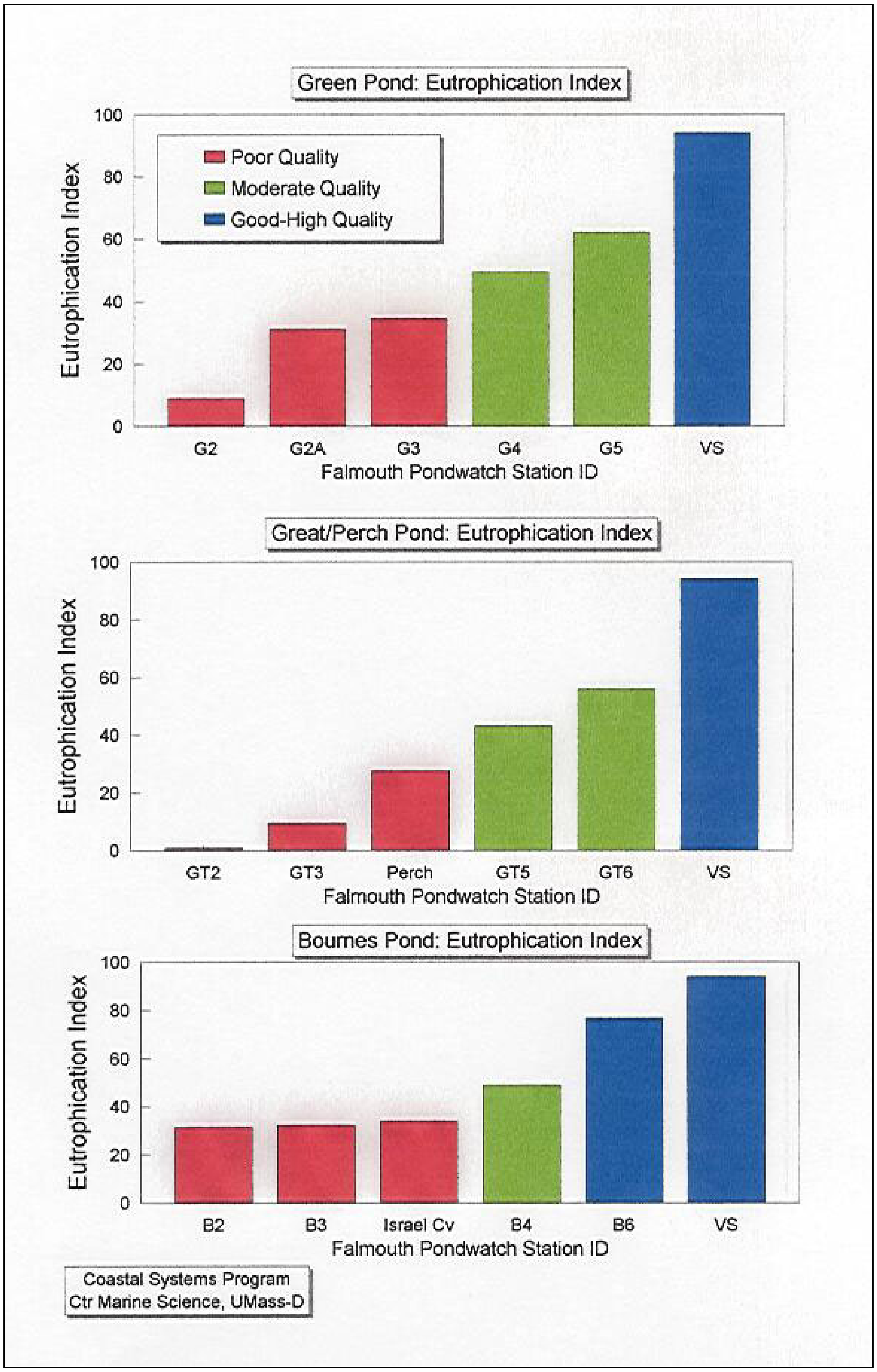

Figure II-2. Eutrophication Index results for each sampling location within (top) Green Pond, (mid) Great/Perch Pond, and (btm) Bournes Pond. The Index is based upon sampling results on transparency, nitrogen, chlorophyll a pigments, and bottom water dissolved oxygen measured from 1990-1998 by the Falmouth Pondwatch Program, Center for Marine Science and Technology, UMass-D.

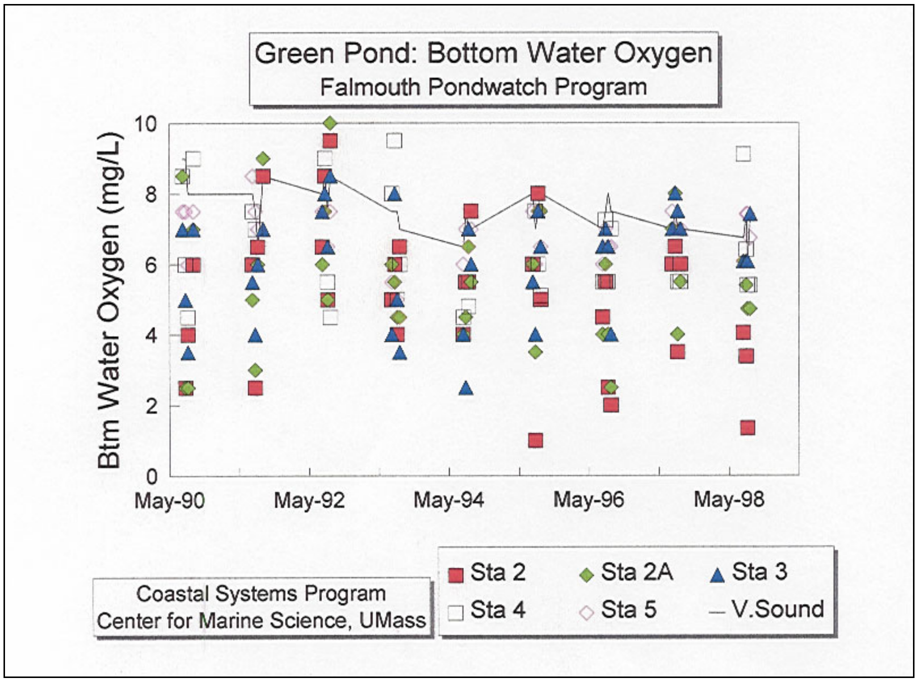

Figure II-3. Green Pond bottom water oxygen levels during July and August 1990-1998. Data are from the Falmouth Pondwatch Program and are assayed by Winkler Titration. Oxygen levels below 4 mg/L are considered ecologically "stressful". Line represents oxygen level in Vineyard Sound. Locations of sampling stations are shown in Figure II-1.

c. Bournes Pond

Bournes Pond is the least nitrogen loaded of the three Falmouth ponds used as case studies. As a result, it currently supports sections with the highest water quality within the three systems. In addition, the nitrogen plume from the former Massachusetts Military Reservation Wastewater Treatment Facility (now closed) is not projected to enter this system. The Town owns significant acreage of conservation land on the western side of the pond. Bournes Pond is the only pond that has restricted tidal exchange with Vineyard Sound, primarily due to periodic sedimentation of the inlet and the narrow/shallow inlet channel.

Bournes Pond, like Green and Great Ponds, shows a strong horizontal gradient in nitrogen and health resulting from the distribution of its watershed nitrogen inputs and the exchanges with the high quality waters of Vineyard Sound. The headwaters of the pond, including the northerly third of the pond (marine reach of Bournes Brook), are currently showing "Severely Degraded" water quality and "Hyper-Eutrophic" conditions, similar to upper Green and Great Ponds. The high nitrogen levels are associated with moderate to high chlorophyll *a* concentrations and moderate to high oxygen depletions. Water transparency within all but the most northerly reaches is sufficient to support benthic plant production. Although conditions within the upper tributary fall within the "Hyper-Eutrophic" classification, most of the area is near the threshold to "Eutrophic" or "Significant Impairment" status. For this reason, water quality conditions are less degraded than the upper reaches of Green and Great Ponds, where nitrogen levels are higher.

Most of the lower portion of Bournes Pond (more than half of the surface area) can be considered between "Significant Impairment" and "Moderately Impaired" (or Mesotrophic and

Eutrophic). This can be seen in the modest chlorophyll a levels (<5:g/L), moderate oxygen depletions and moderate to poor water column transparencies. This region of the pond supports shellfish beds and does not appear to accumulate significant amounts of macro-algae. The southern portion of the lower basin might support eelgrass. The Eutrophication Index indicates the higher general quality of the Bournes Pond southern basin (Figure II-2). Based upon this tool, this portion of Bournes Pond would rank at the middle to upper tier of Buzzards Bay embayments.

2. Stage Harbor, Chatham

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

The south shore of Chatham exhibits a moderate tide range, with a mean range of about 4.5 ft (1.5 m). Since the water elevation difference between Nantucket Sound and each of the estuarine systems is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle. Tidal damping (reduction in tidal amplitude) through the Stage Harbor system is negligible indicating “well-flushed” systems. However, water quality evaluations indicate that estuarine health within the Stage Harbor System is dominated by watershed nutrient loading rather than tidal characteristics.

As discussed above, the nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For Stage Harbor, an assessment was based upon data from the water quality monitoring database and our surveys of eelgrass distribution, benthic animal communities, and sediment characteristics conducted during the summer and fall of 2000. These data form the basis of our assessment of Stage Harbor’s present health.

There are several major conclusions relative to nutrient related habitat quality that can be derived from an examination and comparison of eelgrass coverage in the Year 2000 (Figure II-4) with MA DEP Eelgrass Mapping Program data from 1994 (Figure II-5). Eelgrass coverage is declining within the Stage Harbor System. Oyster Pond and Oyster Pond River appear to have had bed loss between 1994 and 2000. It is likely that the eelgrass beds within Oyster Pond were relatively extensive in recent times (1970’s or 1980’s) based upon the apparent rapid rate of loss in other parts of the system. Similar to Oyster Pond, the Mill Pond tributary to Stage Harbor also appears to be losing eelgrass. The pattern of loss is also similar, with loss beginning in the innermost reaches with migration toward the lower parts of the System. The loss of eelgrass from 1994 to 2000 from Mill Pond, Mitchell River, and upper Stage Harbor

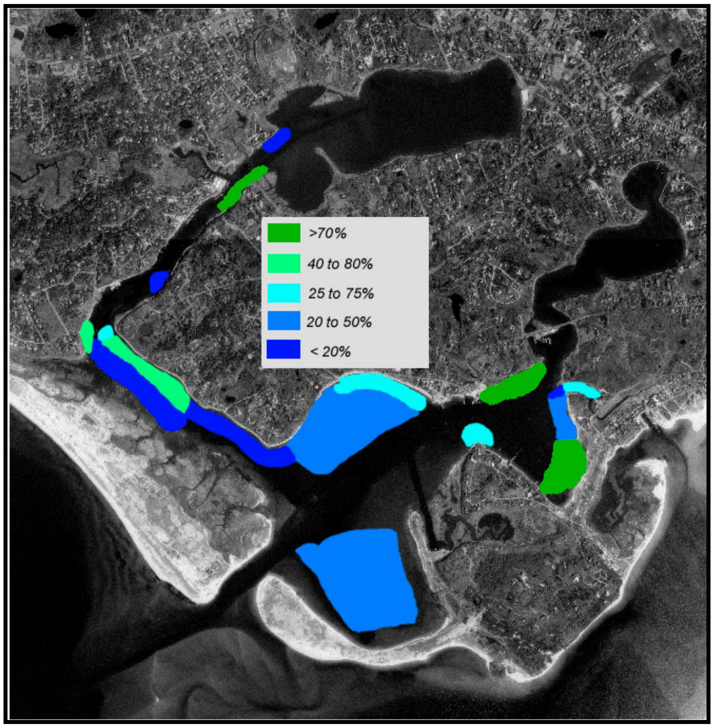

Figure II-4 Map of Stage Harbor eelgrass distribution as observed in 2000, School for Marine Science and Technology, UMass-D. No mapping was conducted in offshore waters beyond the tidal inlet.

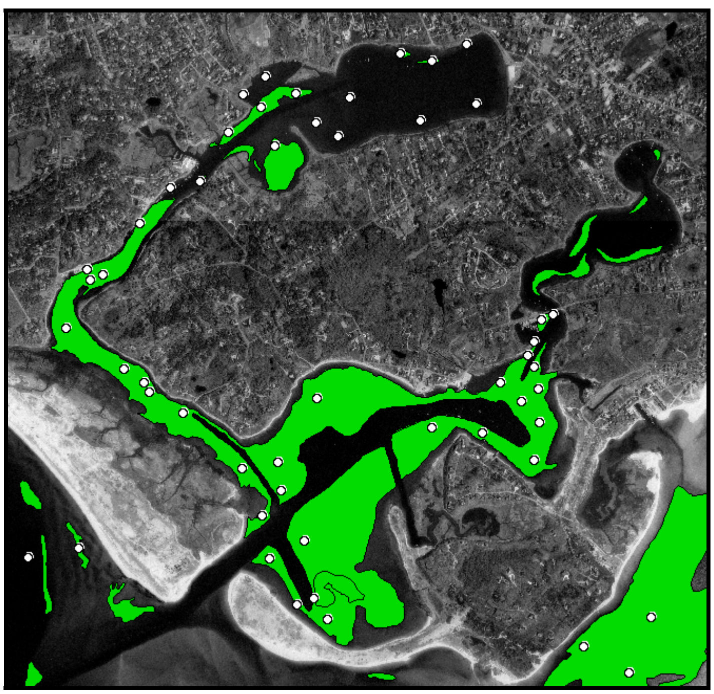

Figure II-5. Map of eelgrass distribution (green shaded areas) within Stage Harbor and nearby shallow offshore regions as determined by Massachusetts DEP in 1994 by analysis of aerial photographs. White circles indicate sites where eelgrass coverage was field-confirmed.

mirrors the loss from Oyster Pond and Oyster River over the same period. This short-term loss of eelgrass indicates a system in transition, with rapidly declining water quality.

Mill Pond and Little Mill Pond within the Stage Harbor system show poor habitat quality. These embayments have lost their eelgrass beds and have benthic animal populations dominated by small oligochaete worms and nematodes, indicative of eutrophic conditions. Similarly, the total nitrogen levels in these sub-embayments are 0.51 to 0.77 mg N/L and their bottom waters show periodic oxygen depletion. Oyster Pond supports a higher habitat quality as seen in the infauna populations and small remaining pockets of eelgrass. However, the water quality database indicates total nitrogen levels averaging 0.78 mg N/L which appear to be high relative to similar embayments of comparable health.

The larger sub-embayments to Stage Harbor are currently supporting moderate habitat quality, although it is apparently declining based upon trends in eelgrass distribution. The observed level of habitat quality appears to be better than would be indicated by the total nitrogen levels alone (0.53 - 0.66 mg N/L). However, the system is clearly not in steady state and appears to be reaching an equilibrium with nitrogen levels, likely below the present level of habitat quality. Based upon other embayment systems, the present nitrogen levels (to the extent that they are representative) would not support eelgrass within the middle and upper regions of Stage Harbor. However, further data collection and analysis of watercolumn nitrogen levels are needed to support this conclusion.

Overall, the habitat quality within the Stage Harbor System can be classified as poor (impaired, eutrophic) in the upper reach within Little Mill Pond and Mill Pond; moderate or mesotrophic within the Stage Harbor Basin and Oyster River; and Meso-Eutrophic in Oyster Pond. There appears to be a system-wide on-going decline in nutrient related water quality that is most pronounced in the upper reaches of the 2 major tributary systems. Eelgrass has disappeared in these upper reaches and appears to be gradually retreating toward the inlet. However, the system still supports significant eelgrass habitat, fish and shellfish and regions of healthy benthic habitat.

3. Frost Fish Creek, Chatham

Frost Fish Creek is an upper tributary to the Bassing Harbor System within Pleasant Bay. This estuary is a composite of both emergent salt marsh and embayment. The central permanently flooded tidal basin largely results from the construction of a weir at the tidal outlet to Ryder Cove. The system is hyper-eutrophic by embayment standards.

The tide propagating through New Inlet and Chatham Harbor is significantly attenuated by the series of flood tidal shoals within the inlet throat. The mean tide range drops from just under 8 feet in the Atlantic Ocean to around 5 feet at the Chatham Fish Pier. Only minor attenuation occurs between the Fish Pier and Pleasant Bay; however, smaller sub-embayments separated from the main system by culverts exhibit significant additional tidal attenuation. Frost Fish Creek is one of these smaller sub-embayments with diminished tidal amplitude, mean tide range of less than 1 ft, due in significant part to the tidal restriction caused by its inlet culverts. Within the Bassing Harbor system, nitrogen loading is primarily focused in the Frost Fish Creek and Ryder Cove sub-embayments. Upper Frost Fish Creek has the second highest annual watershed nitrogen load of all of Chatham's embayments and the highest per unit area load.

When existing water quality and biological indicator data are combined it is clear that Frost Fish Creek is currently showing impaired nutrient related water quality, due to its high watershed nitrogen load. The assessment of Frost Fish Creek's poor habitat quality is based upon direct measurement of key water quality and ecological parameters (nutrients, oxygen, chlorophyll a, Secchi depth, eelgrass, benthic animal indicators, sediment quality, macroalgae). However, the portions of Frost Fish Creek that contain emergent salt marsh are functional and appear to be of moderate to high quality (for wetlands). Relative to embayment criteria (as opposed to salt marsh criteria) the system is of poor quality due to its high phytoplankton production, low transparency (periodically <0.5 m) and indications of oxygen depletion. In addition, Frost Fish Creek waters are sufficiently eutrophic to be negatively affecting infauna populations and eelgrass. Small oligochaete worms, indicative of very poor habitat quality, dominate what infauna community exists in these systems. The nitrogen levels in these systems are generally very high, 0.89 - 1.20 mg N/L (total N).

Overall, Frost Fish Creek is functioning like a semi-impounded salt marsh. As a result of its structure and algal production from watershed nutrient inflows, this system supports highly organic fluid muds that are not good habitat for benthic animals. The benthic community is depauperate and highly stressed. Eelgrass is absent, which is typical of most salt marsh creeks. Phytoplankton blooms appear to be common during the summer and some oxygen depletion has been observed. Frost Fish Creek is currently eutrophic and of poor water quality by estuarine standards. However, when the emergent marsh is considered, the system continues to provide many of the ecological "services" of healthy salt marsh systems.

III. EVALUATION OF NITROGEN MANAGEMENT APPROACHES

The current nitrogen management models were compared as to their underlying assumptions as well as specific loading coefficients and rates. We have relied upon available publications and reports for the parameterization of each of the models and have attempted to include all updates to approaches made to date.

The watershed and embayment nitrogen management models were evaluated as to their accuracy, data needs, comparability and applicability across embayment types. Based upon detailed discussions with MA DEP, it was decided to focus on the three approaches that have been relatively widely applied in Massachusetts:

- Buzzards Bay Project Nitrogen Loading Methodology (BBP),
- Cape Cod Commission Nitrogen Loading/Critical Loads Methodology (CCC),
- Linked Watershed-Embayment Modeling Approach (Linked).

In addition, we have included a brief evaluation of the Waquoit Bay Model, a theoretical nitrogen flow approach, for the 1 case study available. Since each of the three major models (BBP, CCC, Linked) has been widely applied, they are the models that have been "selected" over the past decade, by the management, regulatory and scientific communities for use in these systems.

We have used a Case Study approach to evaluate and compare the available nitrogen management models. The Case Studies were selected from among the ca. 15 embayments in Southeastern Massachusetts for which we have developed the data to support the more sophisticated Linked Modeling Approach. The overall assessment described in more detail below included:

- comparison of watershed nitrogen loading results from each model and resultant embayment nitrogen distribution based upon the Linked Model;
- evaluation of predicted critical nitrogen loading thresholds (BBP) relative to resultant embayment nitrogen distribution based upon the Linked Model;
- sensitivity analysis for Linked Watershed-Embayment Approach.

Since only the Linked Model yields spatially distributed nitrogen levels within an embayment, it was used to portray the nitrogen distribution resulting from the BBP and CCC models (and Waquoit Model).

A. Evaluation of Nitrogen Loading and Modeling Approaches

Comparison and assessment of the available Nitrogen Loading Approaches focuses on their ability to produce proper loading rates and to accurately portray an embayment's nutrient related ecological health. The BBP and CCC Nitrogen Loading Models are in construct the same as the watershed component of the Linked Model (and all other extant watershed models). The methodology of the watershed nitrogen loading models can be summarized as: sum all of the nitrogen sources and sinks within the watershed and the remainder is the nitrogen load to the adjacent embayment waters (assuming steady-state conditions). In order to evaluate the models, we first compared their scope and loading terms and then the standard nitrogen values which they employ, and finally any field data that they require. The resulting nitrogen loads were then used as input into the Linked Model to yield the spatial distribution of nitrogen within the receiving waters. Similarly, we tested the BBP methodology for developing critical nitrogen loading limits. This approach has been used in a variety of embayments and is based solely upon an embayment's volume and flushing rate relative to a prescribed fixed management nitrogen term. This approach was tested in the Case Studies where the estuarine health has been fully documented. The evaluation required conducting a series of comparable modeling runs using each approach using the following format:

- Comparison of Watershed (Land-Use) Nitrogen Models:
    - ⇒ Linked Model with Attenuation of N Load during transport (and no Attenuation-for comparison purposes only).
    - ⇒ BBP Model with no Attenuation (pre-2000 method) and Attenuation (post-2000 method) of N Load during transport.
    - ⇒ CCC Model without Attenuation (consistent method from 1992).
    - ⇒ Waquoit Model with Attenuation (comparison for Green Pond only)
- Watershed Model Comparison with output entered into the Linked Model's Estuarine Component Model.
- Comparison of BBP Critical N Load Method with output entered into the Linked Model's Estuarine Component Model. Variables tested included:
    - ⇒ Whole Embayment System Residence Time
    - ⇒ Upper Embayment System Residence Time

Comparison of Watershed (Land-Use) Nitrogen Models: The BBP, CCC and Linked Models all use a similar watershed land-use approach. While it is clear that the quality of the land-use data varies in different applications, the Linked Model is typically based upon local assessor's data and is ground-truthed prior to use. The Linked Model also directly surveys the larger nitrogen sources as part of its routine application. All of the Models include standard loading terms as well as "special source" loads. The standard loading terms have been developed because they are common to almost all watersheds. The special sources loads are less common and have a system specific component, these include landfills, centralized wastewater discharges, and other large sources.

All of the watershed loading models depend upon "steady state" conditions, i.e. an equal amount of nitrogen is discharged to the embayment as is added to the watershed each year from each land-use (after correction for any attenuation). The practical underlying assumption is that current land-uses have been in place for sufficient time, so that resultant nitrogen additions to groundwater have reached the receiving embayment (i.e. time since initiation of a land-use change is greater than the groundwater transport time). Generally, steady state is assumed, but the Linked Model evaluates this assumption based upon the system specific groundwater transport times within each sub-watershed to an embayment (based upon USGS groundwater transport modeling). Generally, in Southeastern Massachusetts embayments the assumption of steady state is valid, due to the preponderance of new development within the zone of 10 year groundwater travel time. In specific instances where groundwater travel time is causing a violation of steady-state conditions, loading rates are adjusted to reflect current discharges to the embayment and the nitrogen loading "in transit" is used as a "future" load.

We have assembled the standard values employed by the models in Table III-1. **In almost all cases the standard loading terms are consistent among the BBP, CCC and Linked Models.** This is not surprising, since they are based upon the same regional studies and literature data. However, the septic loading term is about 25% lower in the Linked Model than the BBP or CCC Models. This results from the use of Title 5 design flows (with 35 mgN/L) in the CCC and BBP Models, while the Linked Model is based upon regional septic system discharge and transport studies. In addition, while all methods correct for occupancy, this is deemed a major error in some applications, which have not properly evaluated occupancy rates in seasonal communities. Errors in occupancy rates create proportional errors in residential wastewater loading, since it is the product of the occupancy rate times the per capita nitrogen input rate. In actual applications, errors resulting from incorrect occupancy data have been found to generate a relatively large error in the final watershed loading value, due to the preponderance of on-site wastewater to the total nitrogen load within most Southeastern Massachusetts coastal watersheds (Weiskel and Howes 1991).

In contrast to the land-use nitrogen loading terms, there is not consistency among the Models as to the extent of nitrogen attenuation within the watershed as nitrogen moves via freshwater flows from the source to the receiving waters. The Linked Model includes attenuation, the BBP Model did not use attenuation prior to 2000, and the CCC Model does not include attenuation during transport.

The Linked Model uses attenuation based primarily upon direct measurements within the watershed of freshwater systems and surface water inflows. When this data is not fully available, standard attenuation terms are used which vary from 30%-60% depending upon the aquatic system through which the nitrogen passes and its structure. However, it must be stressed that since attenuation of watershed nitrogen can provide a large degree of nitrogen removal, failure to accurately quantify it for a watershed can have significant management consequences. For this reason the Linked Model always includes at a minimum, verification of attenuation through direct measure if a full field evaluation is not available. The Linked Model relies on the series of attenuation studies conducted by SMAST and others. The SMAST studies have been conducted region-wide on lakes/ponds (e.g. Ashumet Pond, Howes 2000), streams/rivers (CDM & Howes 2000), and wetlands positioned to intercept nitrogen transport to estuaries (Howes et al. 1996). SMAST has also conducted a series of studies of attenuation of nitrogen during groundwater transport, several in concert with the USGS, which have indicated that groundwater attenuation is not occurring (notably at the USGS site at MMR and Namskaket Marsh, and Buttermilk Bay). This finding is consistent with both small-scale and large-scale groundwater transport studies in other similar aquifer settings and studies conducted by the CCC relative to drinking water protection.

The CCC Model does not include nitrogen attenuation within groundwater or surface waters. In this sense the CCC Approach is conservative for regulatory purposes. However, it is clear that attenuation is occurring in surface water systems and CCC staff have stated the need for site-specific data to support attenuation factors relative to decision making on specific embayments.

The BBP Watershed Model currently (since Yr 2000) includes a fixed attenuation factor. The model was updated to include a 30% reduction in watershed loads if the nitrogen passed through a wetland, pond, or stream on its way to the estuary or if the source was located more than 1 km from the estuary or a direct surface water discharge. The Approach requires no field verification for these terms (or for any other nitrogen parameter). The Approach does not include multiple reductions for sequential passage through different aquatic systems. The BBP Approach to attenuation is neither conservative (as CCC Model), nor necessarily tied to the individual systems (except in a general land-use sense). The underpinnings of the attenuation factors have not yet been defined.

Table III-1. Comparison of key model components used by the various Nitrogen Management Models linking watershed land-use and embayment habitat quality. The Coefficients represent values employed from technical or literature surveys, while Direct Measurements indicate the incorporation of embayment-specific field data in model generation, calibration or validation.

Parameter	Linked Model Watershed-Embayment		BBP Model: Watershed Load-Residence Time		CCC Model: Watershed Load-Residence Time	
	Coefficient	Direct Measure	Coefficient	Direct Measure	Coefficient	Direct Measure
Watershed Land-Use N Loading Parameters:						
Septic Systems	1.80 kg/person	Occupancy	2.67 kg/person Adjusted	Occupancy Water-Use	2.67 kg/person Adjusted	Occupancy Water-Use
Lawns$^{\alpha}$	1.36 kg N	Survey	1.70 kg N	No	1.70 kg N	No
Impervious$^{\beta}$	0.75 mg/L	No	0.75 mg/L	No	0.75 mg/L	No
Roadway$^{\delta}$	1.5 mg/L	Survey	1.5 mg/L	No	1.5 mg/L	No
Natural Areas	Yes	Some	Since 2000	--	Yes	--
Other	Regional	No	Regional	No	Regional	No
Attenuation * Groundwater	0%	Yes	0% pre-2000; 30% post-2000	No	0%	No
Attenuation Surface Aquatic	30%-50%	Yes	0% pre-2000** 30% post-2000	No	0%***	No
Watershed Input	--	Spatially Distributed		Bulk Load		Bulk Load
N Loading Validation$^{\varepsilon}$	--	Yes	--	No	--	No
Embayment Model Parameters:						
Numerical Circulation Model	--	Whole System	--	No	--	No
Flushing	--	Whole System	--	Upper 1/3 Res. Time	--	Upper 1/3 Res.Time
Validation Hydrodynamics	--	Key Basins	--	No	--	No
N Recycling /Regeneration	--	Whole System	No	No	No	No
Loading Calibration	--	System Salinity	--	No	--	No
Validation Embayment Nitrogen Level	--	Whole System	--	No	--	No
Embayment Ecological Parameters:						
Eelgrass/Algae Distribution	--	Temporal Trend	--	No	--	No
Benthic Comm.	--	Indicators	--	No	--	No
Oxygen Levels	--	July-Aug.	--	No	--	No

$^{\alpha}$ Nitrogen added to residential lawn (3lb/1000 sq ft, 5000 sq ft lawn average). Linked model uses 20% leaching, BBP uses 25% leaching, CCC uses 25% leaching.

$^{\beta}$ Nitrogen added which enters groundwater.

$^{\beta\ \delta}$ Only 90% of precipitation to surface reaches groundwater.

$^{\varepsilon}$ Load Validation is through measurements of groundwater or groundwater seepage

* The Linked Model accounts for groundwater attenuation in either its direct measurements or in the way its coefficients were derived. Further attenuation is not supported by a wide-variety of direct measurements in SE Massachusetts watersheds. BBP groundwater attenuation is only for watershed areas more than 1 km from embayment.

** The BBP Model had 0% Attenuation prior to January 2000, after a CDM/SMAST study of Wareham demonstrated a 50% Attenuation in the Wareham River Estuary.

*** The CCC Model has allowed for attenuation, when site-specific data were available.

The Linked Watershed-Embayment Model also differs from the BBP and CCC Models by spatially distributing the watershed nitrogen load to the receiving waters based upon the distribution of groundwater and surface water inflow and up-gradient land-use. The BBP and CCC Models typically distribute bulk nitrogen loads to sub-embayments or to whole embayments. The reason for this difference is that the BBP and CCC Models do not have a spatially dependent embayment component, whereas the Linked Model's marine component yields fine-scale nitrogen distributions throughout the estuary (which in part depends on where the nitrogen load enters).

Similarly, the Linked Model requires additional data, not needed by the BBP and CCC Models, to support its embayment component which include both modeling and ecological parameters (Table III-1). However, the BBP and CCC do require measurement of embayment flushing rates for determination of critical nitrogen loads that will be discussed below.

We can evaluate differences in calculated watershed nitrogen loads based on the BBP, CCC and Linked Models using three case study embayments, Great Pond, Green Pond and Bournes Pond (detailed in Section IIB above, cf. Ramsey et al 2000, Howes & Goehringer 1993, 1996, Wood et al. 1995). The nitrogen loading factors in Table III-1 were used to generate the watershed loads for each embayment following the prescribed protocols. The BBP Model was run with and without attenuation, since most existing applications of this model have not included this process. We also included available output from the Waquoit Model from Green Pond (Kroger et al. 1999). The Linked Model was run with and without attenuation to allow better comparison to the other approaches (Table III-2). The other nitrogen sources, atmospheric deposition and benthic recycling, were also included in the analysis.

Table III-2. Comparison of nitrogen loading results for Great, Green and Bournes Ponds using various watershed nitrogen models. The "Linked" model includes nitrogen attenuation by surface aquatic systems; however, results without attenuation are provided to allow comparison to the Cape Cod Commission (CCC) Model and the Buzzards Bay Project (BBP) Model (pre-2000). The BBP Model now (post 2000) includes a set 30% attenuation term, and the Waquoit model also includes various attenuation factors. Note that when attenuation is removed, the Linked Model is generally about 25% lower than the BBP and CCC models primarily due to the lower septic system value, the Waquoit Model differs from the BBP and Linked Models (when attenuation is included to allow comparison), by more the 50%. Only the Linked Model includes nitrogen from regeneration of nitrogen within the embayments in the loading determinations.

Case:	A	B		C			D		E			F	
Land -Use Model:	Linked			BBP			BBP		CCC			Waquoit	
	With Atten kg/yr	No Atten kg/yr	Ratio: B/A	No Atten kg/yr	Ratio: C/A	Ratio: C/B	With Atten kg/yr	Ratio: D/A	No Atten kg/yr	Ratio: E/A	Ratio: E/B	With Atten kg/yr	Ratio F/A
Great/Perch Pond													
Watershed Total =	20004	26096	1.30	31824	1.59	1.22	26897	1.34	32772	1.64	1.26		
Direct Rain=	1212	1212		797			797		797				
Benthic Load=	9444	9444		0			0		0				
Great Pond TOTAL=	30660	36752		32621			27694		33569				
Bournes Pond													
Watershed Total =	4161	4537	1.09	5715	1.37	1.26	5305	1.28	5870	1.41	1.29		
Direct Rain=	687	687		452			452		452				
Benthic Load=	5796	5796		0			0		0				
Bournes Pond TOTAL =	10644	11020		6167			5757		6322				
Green Pond													
Watershed Total =	12104	13658	1.13	17288	1.43	1.27	15498	1.28	17586	1.45	1.29	6445	0.53
Direct Rain=	589	589		388			388		388			779	
Benthic Load=	4487	4487		0			0		0			0	
Green Pond TOTAL =	17180	18735		17675			15886		17974			7224	

The BBP, CCC and Linked Models all yield consistent results as expected from their commonality of loading terms when comparable approaches are assessed, attenuated versus attenuated and unattenuated versus unattenuated loads (Table III-1). However there was generally a 25%-30% higher watershed nitrogen loading in the BBP and CCC Models compared to the Linked Model output. This results, almost entirely, from the lower (25%) residential wastewater nitrogen loading factor in the Linked Model. The unattenuated BBP and CCC loads were virtually the same.

Comparing the "best-estimate" of nitrogen loads yields larger differences with the attenuated Linked Model being 28%-34% lower than the attenuated BBP Model and 41%-64% lower than the unattenuated CCC Model. However, the Waquoit Model was striking in its divergence from the BBP, CCC and Linked Models. The Waquoit Model uses different land-use terms and is built from a different basis than the other models. The Waquoit Model is a nitrogen cycle model which seeks to track all of the transfers and transformations of nitrogen within the watershed (e.g. plant uptake, sorption, denitrification etc.) with the residual being released to the estuary. However, the Waquoit Model does not use site-specific field measurements for its parameterization. The model clearly yields much lower nitrogen loads than the other approaches: 58% lower than BBP Model, 63% lower than CCC Model, and 47% lower than the Linked Model. As discussed below, the Waquoit Model does not yield embayment nitrogen levels comparable to the measured levels in Green Pond, indicating that it is yielding unrealistic loads. Equally important, the Waquoit Approach suggested that a large nitrogen load from the Ashumet Valley Wastewater Plume was entering Green Pond. However, this has been demonstrated by USGS and others to not be the case, the plume discharges to Great and Bournes Ponds not Green Pond (Masterson data in Ramsey et al. 2000, Masterson et al 1997).

<u>Nitrogen Recycling</u>: As a data driven approach, the Linked Model includes benthic nitrogen regeneration as a source term, which the other models do not. 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.

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

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

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

When benthic nitrogen loading (recycling) is added into the Linked Model Approach the values for total nitrogen load more closely approximate that of the BBP and CCC Models for Great and Green Ponds but diverge more widely for Bournes Pond (Table III-2). However, it is certain that that even wider divergences would be found in systems like Nantucket Harbor or Stage Harbor in Chatham where benthic recycling is even a more important part of the nitrogen input than in these three Falmouth embayments. The nitrogen inputs from the sediments are relatively large in these embayments accounting for 25%-33% of the total summer nitrogen loading. It is also important to note that the nitrogen loading from sediments is generally distributed differently than input from the watershed. The result is that while benthic loading needs to be included, it requires an approach which will spatially distribute the load for proper application.

Overall the BBP, CCC and Linked Models are comparable, but with lower loads due to on-site wastewater values. In contrast the Waquoit Model is widely divergent in its nitrogen loading estimates, and has demonstrated predictive failure. The next step in the model evaluation was to determine the effect of these differences in determined watershed nitrogen loading rates as they relate to the embayment nitrogen levels.

Watershed Model Comparison with output entered into the Linked Model's Estuarine Component Model: The goal of this comparative evaluation was to produce a verifiable result of the watershed nitrogen loads produced by the various Models. Since only the Linked Model yields an independently verifiable result (i.e. nitrogen concentrations within the estuary), the results from each watershed model (BBP, CCC, Linked, Waquoit) was used as the watershed input component in the Linked Model. The Linked Model was run and the resultant nitrogen distribution throughout each embayment was predicted. The nitrogen inputs were distributed spatially. The predicted nitrogen concentrations can then be compared (verified) to the measured concentrations from the Falmouth PondWatch Program (run by SMAST) made each summer over the past decade. The source water nitrogen levels are from the Vineyard Sound reference station monitored since 1987.

Since the BBP and CCC models (and Waquoit Model) do not include the nitrogen input to the embayment waters from the sediment regeneration of nitrogen, this term was not included in the present evaluation. This makes any differences conservative, since inclusion of benthic regeneration in the Linked Model Approach tends to bring the results closer together for the three case study estuaries. The absence of benthic regeneration rates is a likely explanation of the disagreement between predicted and measured nitrogen levels in certain CCC and BBP model applications. It is important to note that benthic regeneration is a fundamental part of estuarine nitrogen cycling and is part of freshwater TMDL determinations (for example, Assabet River MA) and coastal nitrogen modeling (for example, MWRA's HOM Program). The near agreement of the CCC and BBP models in 2 of the 3 embayments, when benthic regeneration was not included results from the coincidence that the over-estimation of the watershed loading, due to use of Title 5 wastewater loads, offset the absence of the nitrogen loading from benthic regeneration. This was not the case in the third embayment, nor can it be predicted a priori, in which estuaries such coincidences will occur. A more stringent evaluation would include benthic regeneration in both the BBP and CCC models and would have resulted in a much poorer performance of these models in predicting nitrogen levels in all of the Case Study embayments

The results of the evaluation indicate that although modeled nitrogen concentrations approximated measured levels (within 0.5 mgN/L), there were significant differences between the Models. In addition, differences between predicted and observed levels were not consistent along the nutrient gradient within the estuary (Table III-3). All of the models had the most difficulty reproducing the nitrogen levels within the headwaters of each estuary. Although the Linked Model showed a relatively large error in this region (14%-33%), it out performed the BBP (26%-51%), CCC (20%-73%) and Waquoit (57%) models. The headwaters of these estuaries are particularly difficult to evaluate, since they are very shallow (<0.5m) and represent the initial mixing zone for the discharging fresh streamwaters and tidal marine waters. These regions show the highest nitrogen levels, but fortunately generally represent only a very small portion of the embayment area. In the Falmouth Pond Case Studies, stations 2 or higher are more typical of embayment conditions.

Differences between the models and comparison of each model with the observed levels is enhanced if the percent error (percent difference) is examined at each location within the estuary rather than the absolute nitrogen value. The percent error is independent of nitrogen level and just focuses on the difference between the measured and predicted nitrogen level at each estuarine location (Table III-4, Figure III-1).

Table III-3. Modeled nitrogen concentrations in Great, Green, and Bournes Ponds resulting from watershed loads determined by Linked Model approach, Cape Cod Commission approach, and the Buzzards Bay Project approach.

Method	nitrogen concentration (mg/L)					
GREAT POND	GT1	GT2	GT3	GT4 - PP	GT5	GT6
Measured	0.94	0.90	0.77	0.88	0.61	0.52
Linked w/attn	1.22	0.91	0.70	0.85	0.64	0.57
Linked no attn	1.47	1.05	0.79	0.97	0.71	0.60
CCC no attn	1.62	1.12	0.79	1.02	0.67	0.56
BBP w/attn	1.42	0.97	0.71	0.89	0.61	0.51
GREEN POND	G1	G2	G2A	G3	G4	G5
Measured	1.67	1.07	0.99	0.81	0.60	0.50
Linked w/attn	1.43	1.15	1.06	0.86	0.67	0.49
Linked no attn	1.49	1.22	1.11	0.91	0.68	0.50
CCC no attn	1.34	1.12	1.02	0.86	0.66	0.48
BBP w/attn	1.23	1.01	0.94	0.79	0.62	0.46
Waquoit	0.72	0.61	0.58	0.51	0.44	0.36
BOURNES POND	B1	B2	B3	B4	B5 - IC	B6
Measured	0.80	0.79	0.65	0.54	0.65	0.39
Linked w/attn	1.06	0.78	0.62	0.51	0.64	0.49
Linked no attn	1.10	0.81	0.64	0.52	0.65	0.50
CCC no attn	1.29	0.85	0.63	0.45	0.47	0.43
BBP w/attn	1.09	0.80	0.59	0.44	0.45	0.42

Table III-4. Percent error from observed long-term averaged nitrogen concentrations and model output (Table III-3) for Great, Green, and Bournes Ponds resulting from watershed loads determined by Linked Model approach, Cape Cod Commission approach, and the Buzzards Bay Project approach. Mean error and standard deviation is given for data exclusive of the first station in each pond.								
Method	percent error							
GREAT POND	GT1	GT2	GT3	GT4 - PP	GT5	GT6	mean	std
Linked w/attn	30.1	1.2	-8.4	-4.1	4.7	8.4	0.4	6.72
Linked no attn	56.7	16.7	2.9	10.1	15.0	15.7	12.1	5.73
CCC no attn	72.7	24.4	2.6	15.5	9.8	6.9	11.8	8.46
BBP w/attn	51.4	8.1	-8.0	0.6	-1.0	-2.9	-0.6	5.85
GREEN POND	G1	G2	G2A	G3	G4	G5	mean	std
Linked w/attn	-14.2	7.6	7.4	6.4	12.6	-2.2	6.4	5.35
Linked no attn	-10.6	14.1	12.5	12.7	14.8	0.0	10.8	6.12
CCC no attn	-19.6	4.8	3.3	6.7	11.4	-3.6	4.5	5.47
BBP w/attn	-26.2	-5.5	-5.1	-2.3	4.0	-7.8	-3.3	4.56
Waquoit	-57.1	-43.1	-41.6	-36.5	-26.8	-27.5	-35.1	7.68
BOURNES POND	B1	B2	B3	B4	B5 - IC	B6	mean	std
Linked w/attn	32.8	-1.9	-4.3	-6.3	-2.5	26.0	2.2	13.39
Linked no attn	37.8	2.0	-1.4	-4.6	-0.5	28.0	4.7	13.24
CCC no attn	61.7	6.9	-4.1	-16.5	-28.5	11.3	-6.2	16.47
BBP w/attn	36.6	1.4	-9.2	-19.3	-31.1	7.7	-10.1	15.61

The Linked Model Approach (standard protocol with attenuation) yielded the best-fit to the observed nitrogen levels overall, with percent errors less than 10% in 13 of 15 cases. Similarly, for Great and Green Ponds, the BBP and CCC also yielded good fits to the measured nitrogen levels with percent errors generally less than 10%. However, both the CCC and BBP Models had difficulties in Bournes Pond, likely a result of not accounting for benthic regeneration. The Waquoit Model yielded an exceedingly poor fit with observed nitrogen levels. The Waquoit Model underestimated nitrogen levels by an average of 35% (range: 27%-57%). Based upon these results the Waquoit Model is not recommended for use by the Massachusetts Estuary Project. Based upon the ability to predict the actual nitrogen levels in a consistent fashion across all of the embayment the Models rank as follows (best to worst): Linked>BBP>CCC>>>Waquoit. The fit appears to be improved if benthic regeneration is added to the BBP and CCC Models.

Comparison of BBP Critical N Load Method with Output to the Linked Model's Estuarine Component Model: The BBP Approach has been used to estimate Critical Nitrogen Loads for embayments and sub-embayments in Southeastern Massachusetts. While the Approach has value, it has not always been proven in its application. The general issues with the approach are its lack of calibration and validation and lack of requirement for an assessment of existing conditions of habitat health. In a Wastewater Facilities Planning Study of Stage Harbor (Chatham MA), the BBP Approach yielded allowable nitrogen loads much higher than the existing nitrogen loads. Yet direct assessment of the embayment indicated that it was already

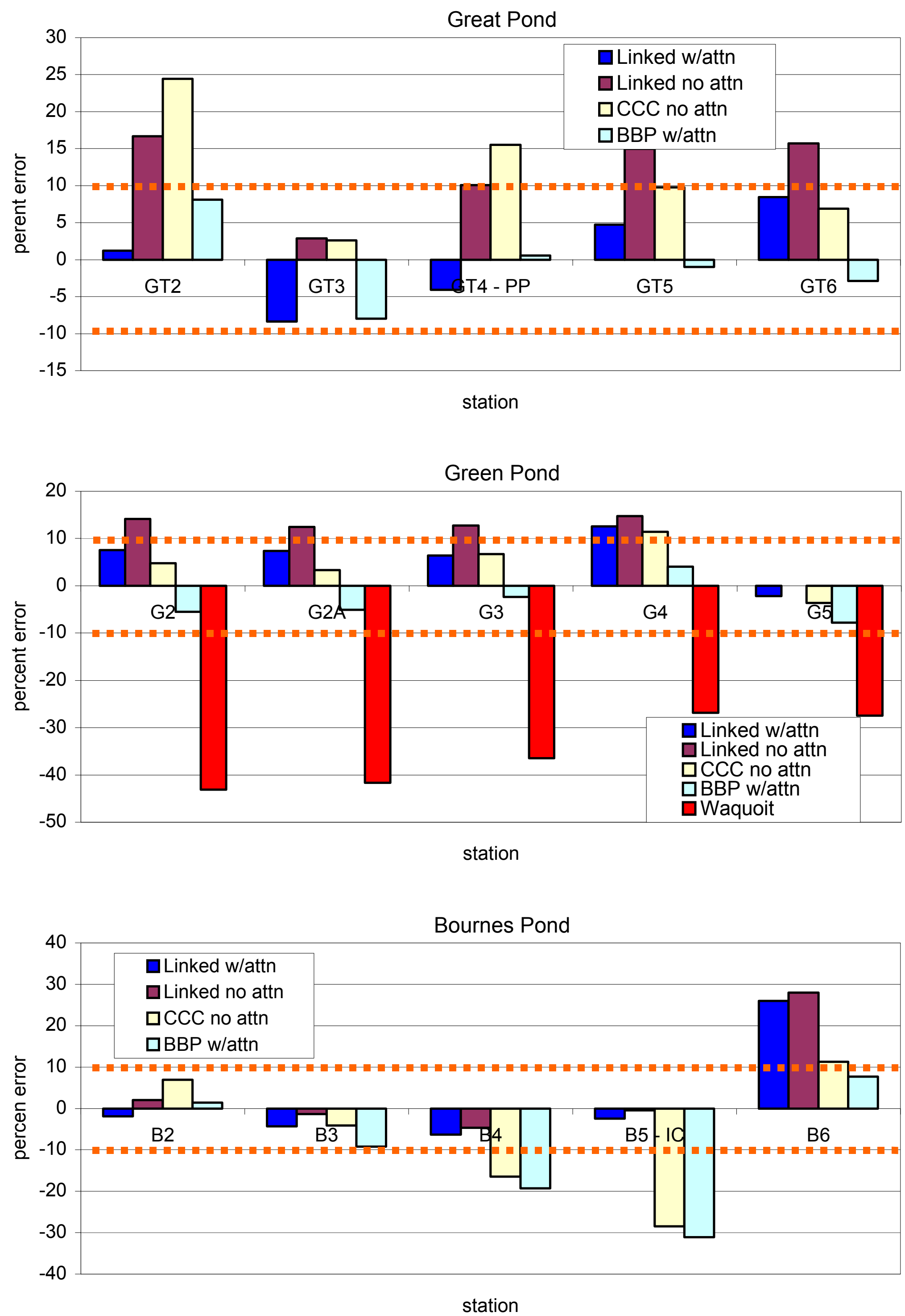

Figure III-1. Percent error between mean of long-term measured total N concentrations and computed concentrations resulting from different upland loading models: 1) Linked model with attenuation, 2) Linked model with no attenuation, 3) Cape Cod Commission model with no attenuation, 4) Buzzards Bay Project model with attenuation, and for Green pond only, 5) the Waquoit model. Ten percent error envelope is indicated by orange-dashed lines.

well beyond its nitrogen threshold level. There were several factors which were found to cause this error, but a major one is outlined in Case Study D below, the effect of lower estuary inputs on upper estuary water quality. These issues will not be further discussed in this section. Instead, the present section focuses on using the three Falmouth Case Study embayments to evaluate this approach relative to the documented water quality and habitat health conditions.

The BBP Critical Nitrogen Load Approach is used by the BBP and the CCC and some individuals and municipalities. The Approach is described in Section II above and has also been reviewed by Eichner and Cambareri 1998. The basic construct of the method is to use a standard loading term associated with different water qualities (SA, SB, ORW) and combine this term with embayment specific flushing. The flushing terms employed are the residence time and the embayment volume. The major issues with the approach relate to the region of the embayment used for the residence time (the BBP indicates the upper third, but does not indicate by volume or area) and the type of residence time (local or system).

For this evaluation both the whole embayment and sub-embayment system residence times were used. After determining the critical loads (Table III-5) to protect embayment water quality, the loads were run as watershed inputs through the Linked Model embayment component to generate predicted nitrogen levels throughout the three case study embayments j(Figure III-2, III-3).

Table III-5. Determination of Critical Nitrogen Loads using the BBP and CCC Approaches (following Eichner & Cambareri 1998, p12). Higher allowable loadings are allowed for lower quality waters (SB>SA>OWR). Residence time, r, was determined using the Linked Model, upper system r is the system residence time for the upper estuary, Whole System r is the system residence time for the entire estuary. In the three case studies, changes in the critical N loads are nearly equivalent to the changes in the selected residence time.

		Water Quality Classification - BBP		
Parameter	**Units**	**SB**	**SA**	**ORW**
Great/Perch Pond (Volume= 1,278,000 m^3)				
Upper System r*	day	8.40	8.40	8.40
N Limit	kg N yr^{-1}	19,200	9,600	3,200
Whole System r	day	0.99	0.99	0.99
N Limit	kg N yr^{-1}	148,700	74,300	24,800
Ratio: Critical Nitrogen Load Whole/Upper = 7.7				
Green Pond (Volume= 513,000 m^3)				
Upper System r	day	2.19	2.19	2.19
N Limit	kg N yr^{-1}	27,650	13,800	4,610
Whole System r	day	1.38	1.38	1.38
N Limit	kg N yr^{-1}	43,200	21,600	7,200
Ratio: Critical Nitrogen Load Whole/Upper = 1.6				
Bournes Pond (Volume= 471,000 m^3)				
Upper System r*	day	1.58	1.58	1.58
N Limit	kg N yr^{-1}	34,800	17,400	5,800
Whole System r	day	0.64	0.64	0.64
N Limit	kg N yr^{-1}	84,000	42,000	14,000
Ratio: Critical Nitrogen Load Whole/Upper = 2.4				

The BBP Critical Nitrogen Loads were found to vary in near direct proportion with alteration of the residence time (r) employed. This is clearly seen in the Critical Loads based upon the Upper system versus whole system r. Since the upper third of an estuary has no volumetric or functional significance, the focus on its flushing rate may not always yield protective results. More significant is that almost all embayments are sub-embayments to a larger bay and some embayments have multiple upper sub-embayments.

The nitrogen levels resulting from the BBP limits using both the upper system and whole system residence time can be compared to the measured values for each of the three embayment systems (Figure III-2, III-3). The present "state of the embayment" can be assessed based upon which critical nitrogen level each embayment is currently over or under (note that use of the BBP's terminology of SA, SB and ORW is for discussion only, as these thresholds have not been formalized). Green Pond current nitrogen levels are close to BBP SA waters and well under the threshold for SB waters. Bournes Pond is generally ORW while Great Pond currently is of lower quality than SB. However, these conclusions do not match the current health of these systems. Green Pond is one of the most eutrophic embayments on Cape Cod. It has lost almost all of its eelgrass, routinely experiences hypoxic conditions, fish kills and floating mats of algae. This does not match with the SA designation. Similarly, Bournes Pond is nutrient enriched and has degraded habitat quality in its tributary systems. It is of the highest quality of the three systems, but does not rise to ORW conditions. In contrast, the BBP Approach appears to work well for Great Pond, the predicted conditions approximating the actual habitat health.

Overall, the critical nitrogen limits based upon the BBP Approach do not consistently approximate measured habitat health conditions. Generally the generated limits tend to over-estimate the loads which a system can tolerate. It is likely that the non-inclusion of benthic regeneration and the lack of accounting for the spatial distribution of nitrogen loads to the entire embayment contribute to the lack of fit.

Summary: The goal of the comparative evaluation was to produce a verifiable result of the watershed nitrogen loads produced by the various Models. Since only the Linked Model yields an independently verifiable result (i.e. nitrogen concentrations within the estuary), the results from each watershed model (BBP, CCC, Linked, Waquoit) was used as the watershed input component in the Linked Model. The Linked Model was run and the resultant nitrogen distribution throughout each embayment was predicted. The nitrogen inputs were distributed spatially and compared (verified) to measured concentrations

The Linked Model Approach (standard protocol with attenuation) yielded the best-fit to the observed nitrogen levels overall, with percent errors less than 10% in 13 of 15 cases. Similarly, for Great and Green Ponds, the BBP and CCC also yielded good fits to the measured nitrogen levels with percent errors generally less than 10%. However, both the CCC and BBP Models had difficulties in Bournes Pond, likely a result of not accounting for benthic regeneration. The Waquoit Model yielded an exceedingly poor fit with observed nitrogen levels. The Waquoit Model underestimated nitrogen levels by an average of 35% (range: 27%-57%). Based upon these results the Waquoit Model is not recommended for use by the Massachusetts Estuary Project. Based upon the ability to predict the actual nitrogen levels in a consistent fashion across all of the embayment the Models rank as follows (best to worst): Linked>BBP>CCC>>>Waquoit. The fit appears to be improved if benthic regeneration is added to the BBP and CCC Models.

Assessment of the critical nitrogen limits based upon the BBP Approach indicated that they do not consistently approximate measured habitat health conditions. Generally the BBP Approach generated limits which tended to over-estimate the loads which a system can tolerate. It is likely that the non-inclusion of benthic regeneration and the lack of accounting for the spatial distribution of nitrogen loads to the entire embayment contributed to the lack of fit.

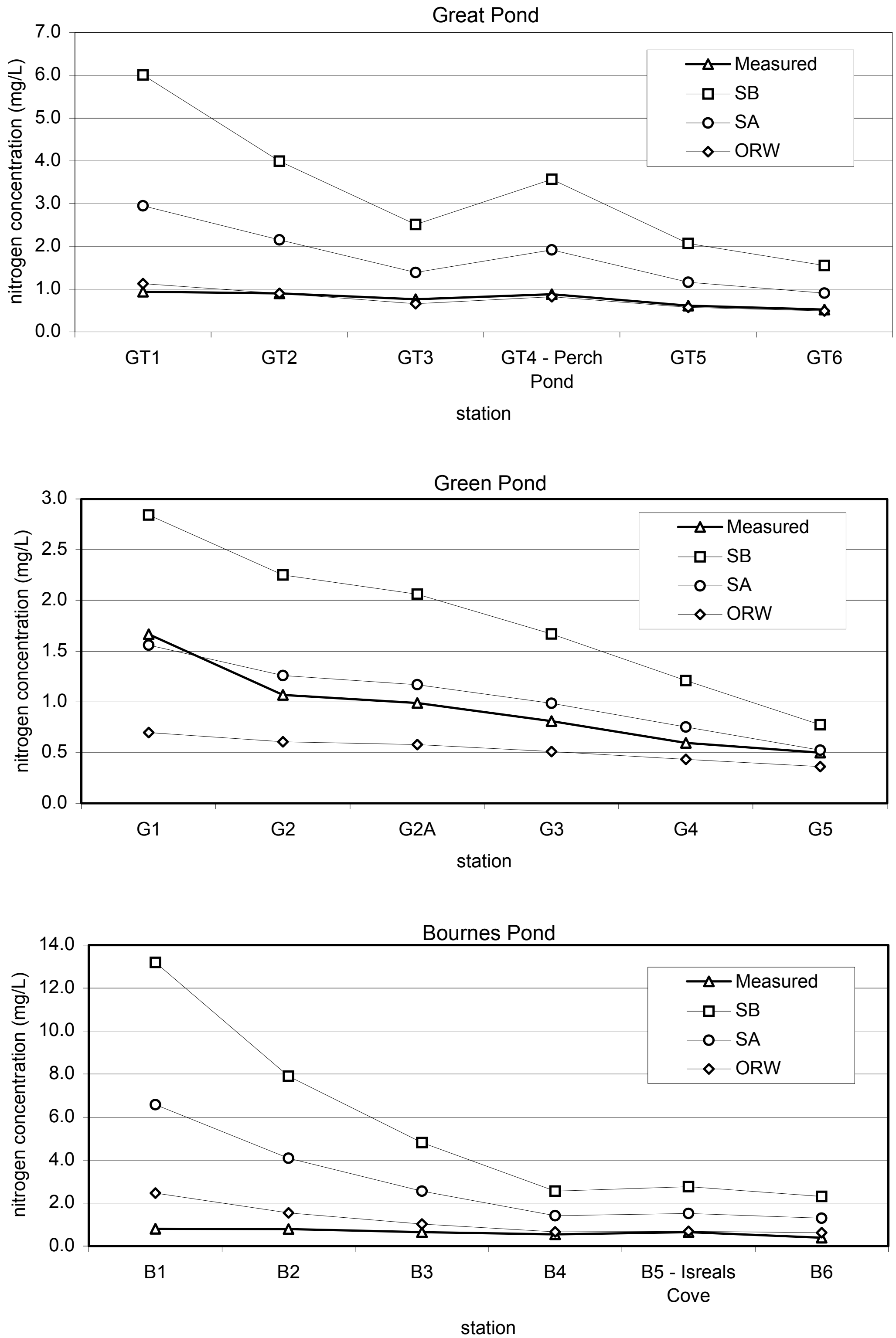

Figure III-2. Total nitrogen concentrations in Great, Green, and Bournes Pond s resulting from watershed load determined using *total system* residence times and BBP methodology for allowable loads for SB, SA, and OWR resource criteria. Long-term means of measured data are also shown.

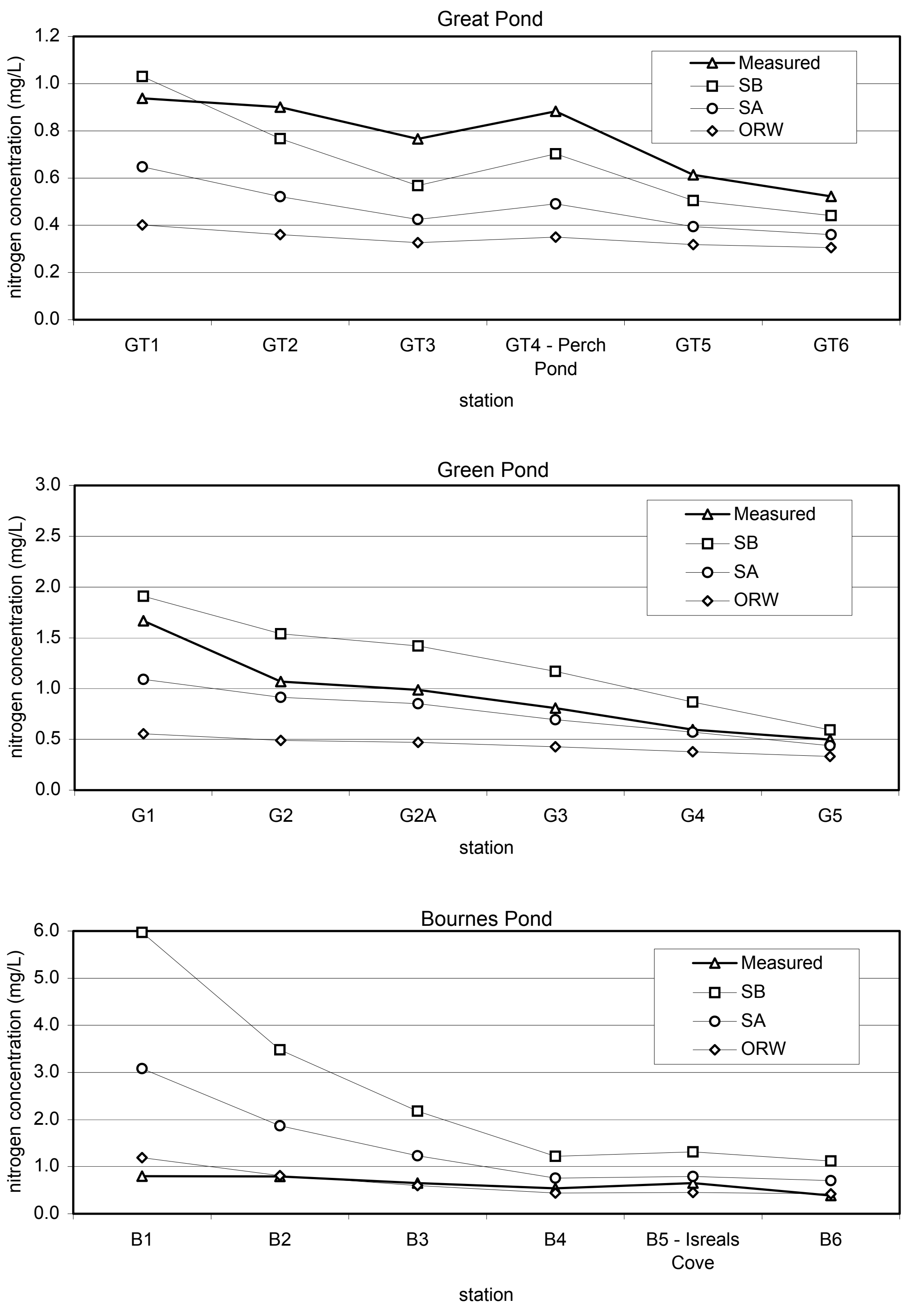

Figure III-3. Total nitrogen concentrations in Great, Green, and Bournes Pond s resulting from watershed load determined using ***system*** residence times of the upper embayment and BBP methodology for allowable loads for SB, SA, and OWR resource criteria. Long-term means of measured data are also shown.

B. Linked Model Hydrodynamic Calibration and Validation

In addition to the evaluation of nitrogen sources and sinks, development of a calibrated and validated hydrodynamic model is essential as the basis for estuarine water quality analysis. A state-of-the-art computer model is utilized to evaluate tidal circulation and flushing in the systems investigated using the Linked Model approach. The particular model employed is the RMA-2 model developed by Resource Management Associates (King, 1990). It is a two-dimensional, depth-averaged finite element model, capable of simulating transient hydrodynamics. The model is widely accepted and tested for analyses of estuaries or rivers. Team members have utilized RMA-2 for numerous flushing studies in Southeastern Massachusetts, including West Falmouth Harbor, Popponesset Bay, Pleasant Bay, Falmouth "finger" Ponds, and Barnstable Harbor. Since the evaluation of hydrodynamics typically calibrates to within 1% or 2% of the observed flow patterns throughout an estuarine system, evaluation of model sensitivity was deemed unnecessary. However, care should be exercised when parameterizing the model boundary conditions in a manner that allows accurate calibration. Instead of presenting model sensitivity, the following section describes the methodology for calibrating and validating hydrodynamic estuarine models.

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

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

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

- Grid generation
- Boundary condition specification
- Calibration
- Validation

The extent of each finite element grid can be generated using contour data developed from Geographic Information System (GIS) data, digital aerial photographs, or appropriate reo-referenced maps. A time-varying water surface elevation boundary condition (measured tide) is specified at the entrance of each system based on actual tide gauge data collected in the source waters of an embayment (e.g., Nantucket Sound for Stage Harbor in Chatham). Freshwater recharge boundary conditions for significant surface water flows into an estuary (e.g., streams or rivers) can be specified to approximate average fresh water inputs to the systems. Once the grid and boundary conditions are set, the model is calibrated to ensure the accuracy of the hydrodynamics, which are used later for the runs of the water quality model. Various friction and eddy viscosity coefficients are adjusted, through several model calibration simulations for a system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

The finite element grid (Figure III-4) for each modeled system provides the detail necessary to evaluate accurately the variation in the hydrodynamic properties of each estuary. Fine resolution is required to simulate the numerous channel constrictions that significantly impact the estuarine hydrodynamics. The grid generation program is used to develop quadrilateral and triangular two-dimensional elements throughout the estuary. Reference water depths at each node of the model grid of the Stage Harbor example were interpreted from bathymetry data obtained from a combination of sources, including recent bathymetry surveys and NOAA bathymetry data. A plot of the bathymetry of the finished Stage Harbor grid is shown in Figure III-5.

Grid resolution is governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability in each system. Relatively fine grid resolution was employed where complex flow patterns were expected. For example, smaller node spacing in marsh creeks and channels will provide a more detailed analysis in these regions of rapidly varying flow. Also, elements through deep channels (e.g., Stage Harbor Inlet channel) were designed to account for rapid changes in bathymetry caused by inlet shoaling and scour processes. Widely spaced nodes were often employed in areas where flow patterns are not likely to change dramatically, such as in the main basin of Stage Harbor and in Oyster Pond. Appropriate implementation of wider node spacing and larger elements reduced computer run time without sacrificing accuracy.

Three types of boundary conditions generally are employed for the RMA-2 models of the estuarine systems in Southeastern Massachusetts: 1) "slip" boundaries, 2) freshwater inflow, and 3) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generates all internal boundary conditions from the governing conservation equations. A tidal boundary condition is specified seaward of the inlet to each system. To accurately define this boundary, tide measurements provided the required data. Dynamic (time-varying) model simulations are used to specify a new water surface elevation in every model time step (12 minutes for Stage Harbor).

1. Model Calibration

After developing the finite element grids, and specifying boundary conditions, the estuarine hydrodynamic model is calibrated. The calibration procedure ensures that the model predicts accurately what was observed in nature during the field measurement program. Calibrated models provide a diagnostic tool to evaluate other scenarios (e.g., the effects of increasing the size of an inlet channel to improve flushing). Calibration of the flushing model requires a close match between the modeled and measured tides in each of the sub-embayments where tides were measured. Initially, the model is calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement is achieved, a model is calibrated based on dominant tidal constituents determine by processing the measured tide data.

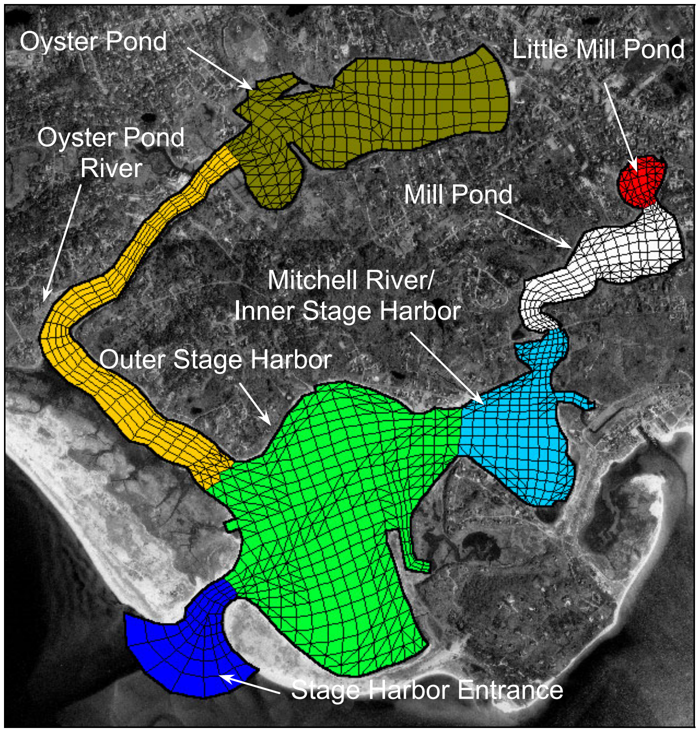

Figure III-4. Plot of numerical grid used for hydrodynamic modeling of Stage Harbor system. Colored divisions indicate boundaries of different grid material types, as well as volumes used to compute flushing rates for individual embayments.

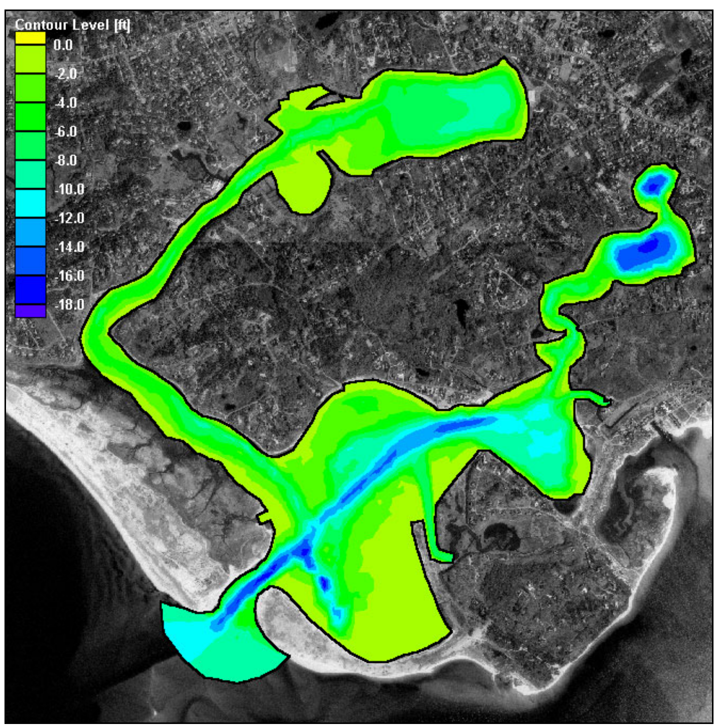

Figure III-5. Depth contour plots of the numerical grid for the Stage Harbor system at 2-foot contour intervals relative to NGVD29.

For the Stage Harbor model example, the calibration was performed for a seven-day period beginning July 25, 2000 at 1600 EDT. This representative time period includes the spring tide conditions, when the tidal range is largest, and resulting tidal currents are greater as well. To provide average tidal forcing conditions for the water quality modeling of Stage Harbor, a separate time period was chosen that spanned the transition between spring and neap tide ranges (bi-weekly maximum and minimum tidal ranges, respectively). For the water quality modeling effort, the 7.25-day period (14 tide cycles) beginning July 31 2000, at 1300 EDT was used.

To improve model accuracy, friction coefficients are varied throughout the model domain. First, the Manning's coefficients are matched to bottom type. For example, lower friction coefficients were specified for the smooth sandy channels in the entrance channel of each Pond, versus the silty bottom of the shallow regions in the upper portions of each Pond, which provided greater flow resistance. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within each estuary. Manning's values for different bottom types were initially selected based on ranges provided by a source such as the Civil Engineering Reference Manual (Lindeburg, 1992), these values are then changed incrementally when necessary to obtain a close match between measured and modeled tides.

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy loss increases where flow is swifter, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). Often the modeled systems are relatively insensitive to turbulent exchange coefficients because they have no regions of strong turbulent flow. Typically, model turbulence coefficients were set between 50 and 100 lb-sec/ft^2. Higher values (up to 300 lb-sec/ft^2) are typically used for marsh plains and in culverts.

For the Stage Harbor example, an element wetting and drying was employed to simulate the periodic inundation and drying of tidal flats in these systems. Nodal wetting and drying is a feature of RMA-2 that allows grid elements to be removed and re-inserted during the course of the model run. Figure III-6 presents an example of how the computational grid is modified by element elimination. This figure shows the Stage Harbor model at a point just after low tide. White areas within the boundary of the mesh are elements that have gone dry, and as a result, have been removed temporarily from the model solution. The wetting and drying feature has two keys benefits for the simulation, 1) it enhances the stability of the model by eliminating nodes that have bottom elevations that are higher than the water surface elevation at that time, and 2) it reduces total model run time because node elimination can reduce the size of the computational grid significantly during periods of a model run. Wetting and drying is employed for estuarine systems with relatively shallow borders and/or tidal flats.

A best-fit of model predictions for a tide gage deployment period was achieved using the aforementioned general method of hydrodynamic model calibration. Figures III-7 illustrates the seven-day calibration simulation along with a two-day sub-section for Mill Pond in the Stage Harbor system. Modeled (solid line) and measured (dotted line) tides are illustrated at the model location that corresponds to the location of a deployed tide gage.

Although visual calibration achieved reasonable modeled tidal hydrodynamics, further tidal constituent calibration is required to quantify the accuracy of the models. Calibration of M_2

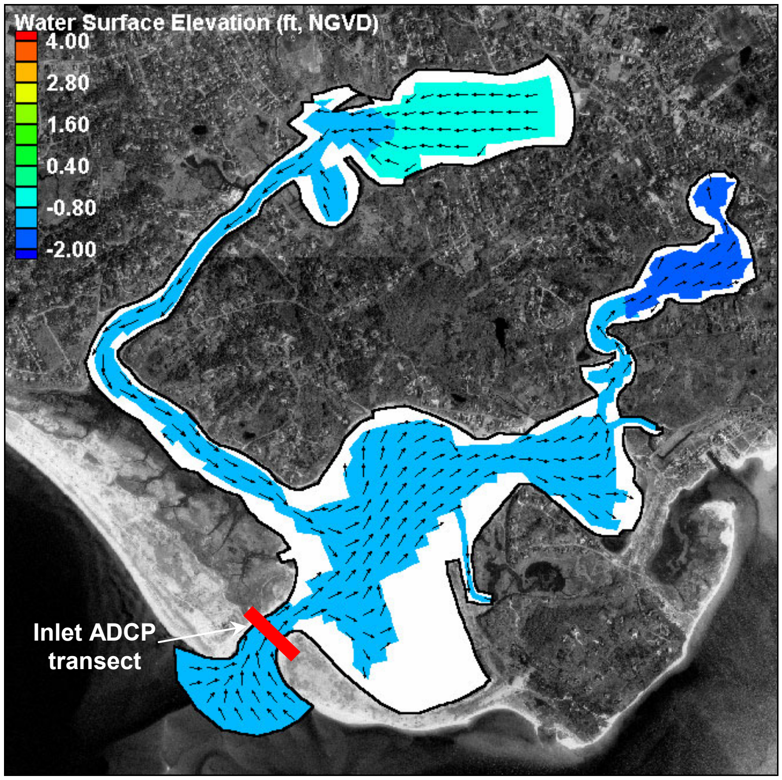

Figure III-6. Stage Harbor model at the inception of a flood tide, with white areas indicating dry elements.

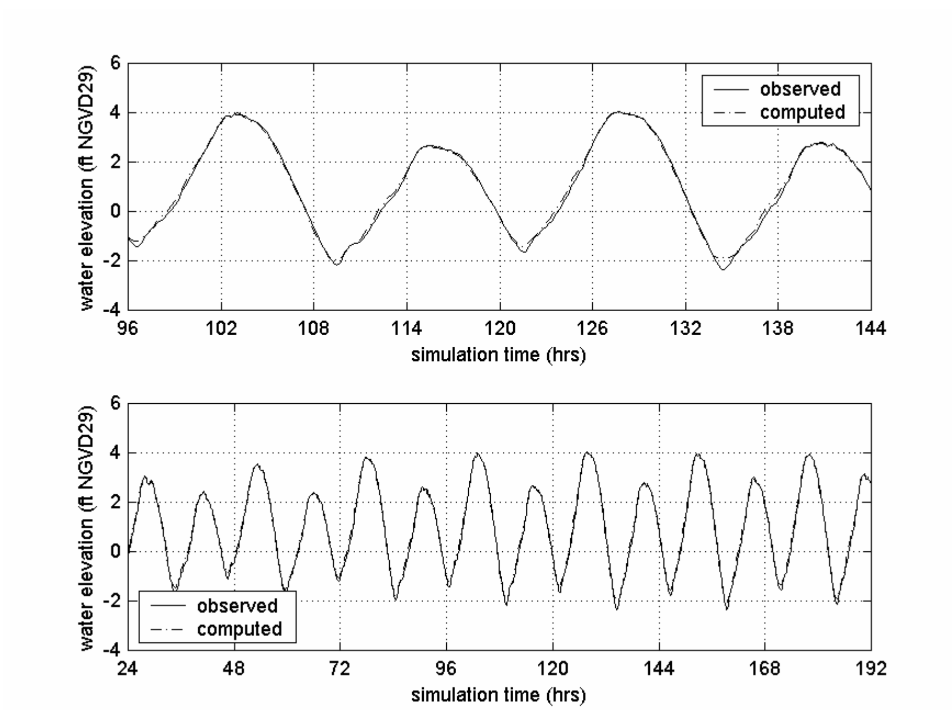

Figure III-7. Observed and computed water surface elevations during calibration time period, for Mill Pond.

was the highest priority since M_2 accounted for a majority of the forcing tide energy in the modeled systems. The tidal constituent calibration of a hydrodynamic model typically results in excellent agreement between modeled and measured tides. The largest errors usually associated with tidal constituent amplitude between the model and data are typically on the order of 0.1 ft, which is only slightly larger than the accuracy of the tide gages.

2. Model Verification Using ADCP Measurements

As discussed previously, the *calibration* procedure used in the development of a finite-element hydrodynamic model requires a match between measured and modeled tides. Model *verification* can be performed to ensure the ability of a model to accurately simulate the physical processes of the real system. In most modeling efforts, verification is performed by running the model for a time period not covered in the initial calibration runs. Similar to the calibration procedure, tidal constituents determined from the model verification run are compared to tidal constituents computed from tide data from the same period, the difference is that only one verification run is made.

For some systems, an additional effort is made to collect current data over the duration of a single tide cycle, using an Acoustic Doppler Current Profiler (ADCP). This current data is very useful for verification purposes, because it can be used to compute flowrates for an independent comparison of model performance. As an example, to verify the performance of the Stage Harbor models, computed flow rates were compared to flow rates measured using an ADCP. For model verification, the model was run for the same period covered during the ADCP survey, during the day of August 16, 2000. Model flow rates were computed in RMA-2 at continuity lines (channel cross-sections) that correspond to the actual ADCP transects followed in each survey. The Transect at the Stage Harbor inlet is shown in Figure III-6.

A comparison of the measured and computed volume flow rates at the Stage Harbor Inlet is shown in Figure III-8 in the top plot, and the tide curve for the same time period is shown in the lower plot. Each ADCP point is a summation of flow measured along the ADCP transect. The ‘bumps’ and ‘skips’ of the flow rate curve can be attributed to the effects of winds (i.e., atmospheric effects) on the water surface and friction across the seabed periodically retarding or accelerating the flow through the inlet, and in the harbor. If water surface elevations changed smoothly as a sinusoid, the volume flow rate would also appear as a smooth curve. However, since the rate at which water surface elevations change does not vary smoothly, the flow rate curve is expected to show short-period fluctuations. Figure III-8 for the Stage Harbor inlet shows a remarkably good agreement with the model predictions. The calibrated model accurately describes the general conditions and the irregularities of the discharge through the Stage Harbor inlet.

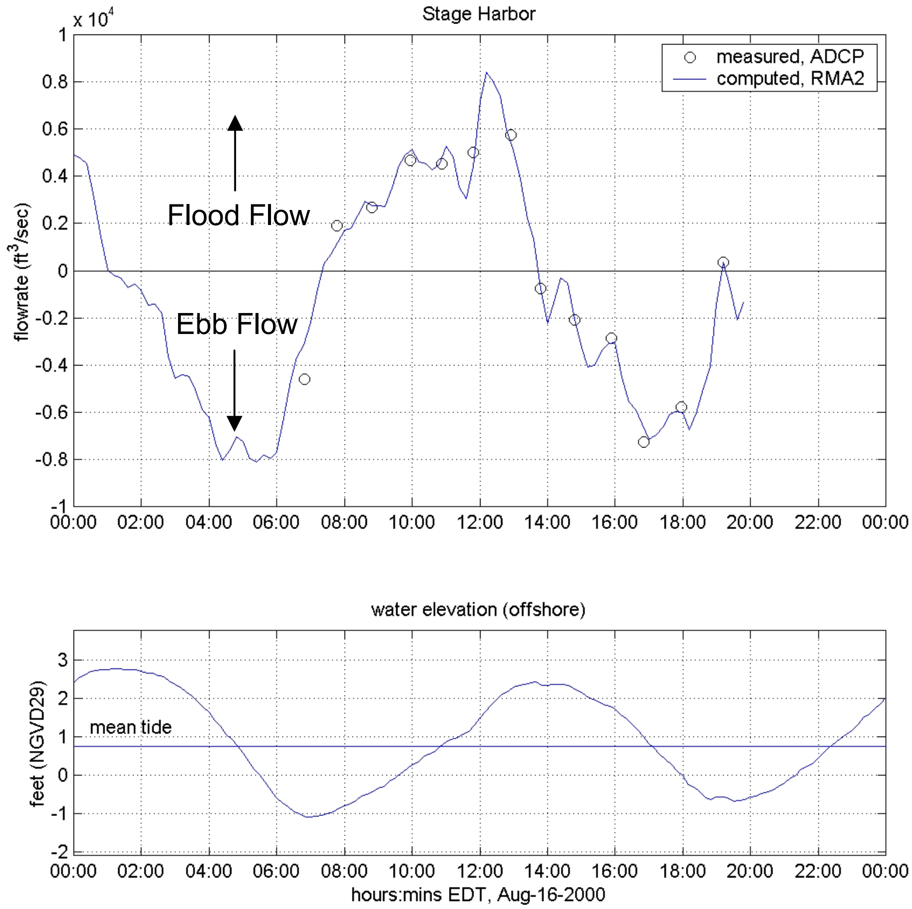

Figure III-8. Comparison of measured volume flow rates versus modeled flow rates through the Stage Harbor Inlet over a tidal cycle on August 16, 2000. Flood flows into the harbor are positive (+), and ebb flows out of the harbor are negative (-). Tidal elevation in the adjacent offshore region is presented for reference.

3. Tidal Flushing and Residence Times

For Stage Harbor, a rising tide offshore in Nantucket Sound creates a slope in water surface from the ocean into the modeled systems. Consequently, water flows into (floods) the system. Similarly, each estuary drains into the open waters of Nantucket Sound on an ebbing tide. This exchange of water between each system and the ocean is defined as tidal flushing. The calibrated hydrodynamic model is a tool to evaluate quantitatively tidal flushing of each system, and was used to compute flushing rates (residence times) and tidal circulation patterns.

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

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

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

In addition to system residence times, a second residence, the **local residence time**, was defined as the average time required for a water parcel to migrate from a location within a sub-embayment to a point outside the sub-embayment. Using Little Mill Pond (Chatham) as an example, the **system residence time** is the average time required for water to migrate from Little Mill Pond, through the Stage Harbor system, and into Nantucket Sound, where the **local residence time** is the average time required for water to migrate from Little Mill Pond to Mill Pond. Local residence times for each sub-embayment are computed as:

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

where T_{local} denotes the residence time for the local sub-embayment, V_{local} represents the volume of the sub-embayment at mean tide level, P equals the tidal prism (or volume entering the local sub-embayment through a single tidal cycle), and t_{cycle} the period of the tidal cycle (again, 0.52 days).

Residence times are provided as a first order evaluation of estuarine water quality. Lower residence times generally correspond to higher water quality; however, residence times may be misleading depending upon pollutant/nutrient loading rates and the overall quality of the receiving waters. As a qualitative guide, **system residence times** are applicable for systems where the water quality within the entire estuary is degraded and higher quality waters provide the only means of reducing the high nutrient levels. For the Stage Harbor Region estuaries this approach is applicable, since it assumes the main system has relatively low quality water relative to Nantucket Sound.

The rate of pollutant/nutrient loading and the quality of water outside the estuary both must be evaluated in conjunction with residence times to obtain a clear picture of water quality. Efficient tidal flushing (low residence time) is not an indication of high water quality if pollutants and nutrients are loaded into the estuary faster than the tidal circulation can flush the system. Neither are low residence times an indicator of high water quality if the water flushed into the estuary is of poor quality. Advanced understanding of water quality will be obtained from the calibrated hydrodynamic model by extending the model to include total nitrogen loading and dispersion (the RMA-4 modeling described in Section II.A.3.

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in each estuary, model results were used to compute residence times. Residence times are computed for the entire estuary, as well as several sub-embayments within the estuary. In addition, **system** and **local residence times** are computed to indicate the range of conditions possible for each of the estuarine systems. Residence times are calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism). Although residence times are still calculated as part of the linked modeling approach, they only are used to provide a general indication of embayment water quality.

Quantitative analysis of water quality is established through the total nitrogen modeling (i.e. the determination of water column nitrogen concentrations throughout an estuarine system).

C. Linked Model Water Quality Constituent Transport Calibration

The movement or transport of water quality constituents within the waters of an estuary is the result of two primary hydrodynamic mechanisms, advection and dispersion. Advection is the transport of a constituent contained within a fluid flow, as in the case of a tidal estuary, where the flow of sea water in and out over the course of a tidal cycle carries along with it dissolved and suspended matter. Dispersion is a small-scale process that results from the random scattering of particles by the combined effects of shear within the flow (velocity gradients and eddies) and diffusion (concentration gradients and molecular motion). In the RMA-4 water quality model, the advective mechanism of constituent transport results from the hydrodynamic output of the RMA-2 model. Dispersion is included in RMA-4 through the use of coefficients that are set by the user.

The RMA-4 model is calibrated to a specific estuary by making adjustments to the dispersion coefficients applied to the model. For example, at the initiation of the calibration process for the Ashumet Valley ponds, dispersion coefficients were applied based upon previous experience in similar systems and based on observed values reported for similar flows. The RMA-4 model allows the division of each embayment into separate model regions, which allows the dispersion coefficients to be varied throughout the pond, as appropriate. The ability to vary dispersion is critical to proper representation of estuarine characteristics, as embayments typically contain both quiescent and turbulent regions. Typically, values between 10 to 50 m^2/sec are observed in relatively quiescent waters (e.g., the upper reaches). Values up to 300 m^2/sec are observed in moderately sized river flows (Fischer, *et al.*, 1979), and are typical of tidal inlets.

Through evaluation of the model output based upon the initial selection of dispersion coefficients, the calibration process proceeds by adjusting the longitudinal (headwaters to inlet) dispersion coefficients in each pond section until the model predictions match the field observations. By this method, estuary-specific dispersion characteristics are represented. During calibration, model dispersion coefficients may be varied by as much as a factor of three or four from the initial estimate, but the coefficients of the calibrated model should still be within ranges that are observed in estuaries (e.g., as in Fischer, *et al.*, 1979). Table III-8 provides final longitudinal dispersion coefficients utilized in each of the three coastal ponds of the Ashumet Valley example. Figure III-10 illustrates the location of different dispersion coefficient regions defined in Table III-8.

For the Ashumet Valley ponds (and in our general protocol), the water quality model of each embayment was calibrated using salinity data, though the ultimate goal was to model total nitrogen concentrations. Since dispersion in shallow estuaries is dominated by turbulent flow versus diffusion and since the bulk of watercolumn nitrogen is dissolved, salinity and total nitrogen both exhibit similar dispersion characteristics.

The use of salinity measurements to determine the dispersion coefficients of estuaries is common practice in estuarine modeling, e.g., as in the general method and examples provided by Fischer, et al. (1979). This is a valid assumption because the modeled systems do not have strong gradients in salinity or nitrogen concentrations. The result in these shallow systems is a dominance of dispersion by turbulent mixing. This dominance also means that suspended

Table III-8. Values of longitudinal dispersion coefficient, E, used in calibrated RMA-4 model runs of salinity and nitrogen concentration for Great, Green, and Bournes Ponds.

Pond Division	E m^2/sec
Great Pond Entrance	100
Lower Great Pond	100
Perch Pond	30
Upper Great Pond	50
Green Pond Entrance	100
Lower Green Pond	60
Upper Green Pond	50
Highest Reach of Upper Green Pond	10
Bournes Pond Entrance	100
Lower Bournes Pond	200
Israels Cove	10
Upper Bournes Pond	30

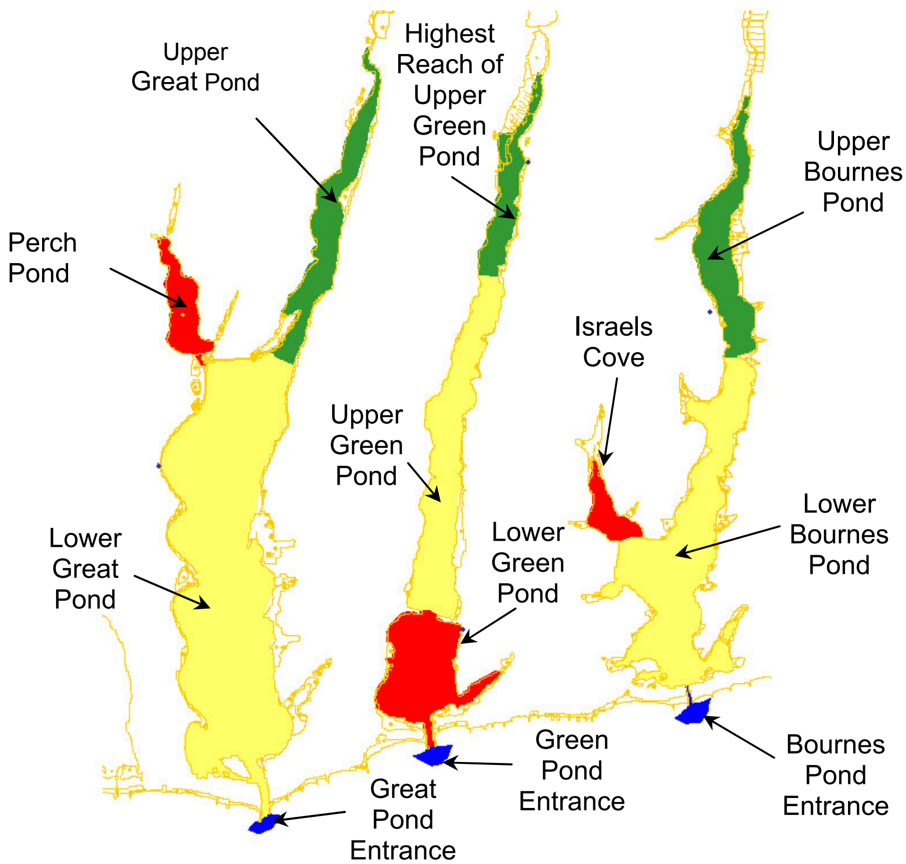

Figure III-10. Map of pond divisions used to vary model dispersion coefficient, E, as in Table 3. Shoreline is indicated by heavy yellow line.

material will follow the same dispersion coefficients as dissolved matter. Therefore, dispersion coefficients determined for salinity are appropriate for total nitrogen.

D. Sensitivity Analysis of Linked Watershed-Embayment Model

Based upon the results of the comparisons of the BBP, CCC and Linked Management Models, we decided to focus the sensitivity analysis on the Linked Model. The Linked Model uses a similar construct and similar loading terms for its watershed model component and therefore the sensitivity analysis will apply in general form to all three approaches. However, as the Linked Model resulted in the "best estimates" relative to measured conditions, it should be the focus of continued evaluation.

The sensitivity analysis was conducted using only the Great/Perch Pond Case Study. This embayment was selected as it had the most complex watershed (ponds, streams and groundwater) and embayment structure of the three Falmouth embayments. Changes in the model output, due to the parameter adjustments in the sensitivity analysis, are expected to be greatest for Great/Perch Pond compared to the other Case Study embayments. The primary model output for evaluation is the distribution of watercolumn nitrogen within the embayment.

Parameter Adjustments for Sensitivity Analysis for Linked Watershed-Embayment Approach: It is critical that before extensive application of any management model, a full analysis of the sensitivity of the model to variations in key input terms be conducted. To date it appears that a sensitivity analysis of the watershed Models has yet to be conducted, although extensive reviews of loading terms were conducted in their development. Limited sensitivity analyses of the Linked Model have been previously performed relative to specific embayments, therefore this represents the first wide-spectrum analysis.

The key parameters which were varied in the sensitivity analysis fall into 2 categories, those related to nitrogen loading and those related to the physics of nitrogen movement within the embayment. The parameters and their variations were as follows:

<u>Nitrogen Loading and Transport Model Parameters:</u>

- Watershed Nitrogen Loading Source Strengths
  - ⇒ Occupancy Rates & Septic N Load -- 1.5x, 1x, 0.5x determined level
  - ⇒ Lawn Fertilizer N Load to Groundwater -- 1x, 0.5x, 0.25x measured level
  - ⇒ Impermeable Surfaces (roof + roadways) -- 1.5x, 1x, 0.5x determined levels
  - ⇒ Direct Rainfall to water surfaces -- 1.5x, 1x, 0.5x determined rates
- N Attenuation in watershed surface water transport -- 1.5x, 1x, 0.5x, 0x measured level;
- Benthic Nitrogen Recycling (summer) -- 1.5x, 1x, 0.5x the measured rates;
- Nitrogen Levels in Marine Source Waters -- 0.5x, 1.25x, 1x the measured value;

<u>Embayment Modeling Parameters:</u>

- Dispersion Coefficients for nitrogen in embayments -- 2x, 1x, 0.5x determined rates;

The approach used in the analysis was to individually vary each of the above key parameters and run the Linked Watershed-Embayment Model to generate the resultant nitrogen distribution within the Great/Perch Pond System. The level of variation in a parameter was selected to reflect the potential maximum range that has been encountered in our regional studies or reported by others.

Nitrogen Loading and Transport Model Parameters: The parameters evaluated under nitrogen loading and transport represent the major watershed nitrogen sources (septic systems, fertilizer, impermeable surfaces, rainfall), embayment nitrogen sources (benthic regeneration and source water concentrations) and transport determinants (attenuation by surface aquatic systems during watershed transport).

Of the major watershed nitrogen sources, the septic system nitrogen load (noted as "Septic Load" in Table III-6, III-7 and Figure III-8) had the largest effect on the resultant embayment nitrogen levels. However, the change was greatest within the uppermost portion of the estuary (about 15%) and diminished to only 5% by the tidal inlet to Vineyard Sound. The model appears to be quite robust relative to watershed nitrogen source terms, as this variation in septic load resulted from a 50% variation over our "best estimate" source term. The other major source terms resulted in less than a 10% variation in embayment nitrogen levels and their effect also generally diminished towards the inlet. Some of this relative "insensitivity" results from the fact that parameters are being varied individually, but the greater reason is that a large fraction of the nitrogen enters in the tidal water.

Table III-6. Modeled nitrogen concentrations at each station (1-6) in Great Pond for each change in a key model parameter in the sensitivity analysis of the Linked Model approach.

	nitrogen concentration (mg/L)					
Great Pond	GT1	GT2	GT3	GT4 - Perch Pond	GT5	GT6
Linked w/attn	1.22	0.91	0.70	0.85	0.64	0.57
1.5x Attenuation	1.09	0.83	0.65	0.77	0.60	0.53
0.5x Attenuation	1.33	0.98	0.75	0.91	0.67	0.58
0.5x Fertilizer	1.10	0.84	0.67	0.78	0.61	0.54
0.25x Fertilizer	1.06	0.81	0.65	0.75	0.59	0.52
1.5x Septic Load	1.41	1.03	0.78	0.95	0.69	0.59
0.5x Septic Load	1.02	0.78	0.63	0.73	0.58	0.51
1.5x Rainfall	1.27	0.94	0.72	0.86	0.65	0.57
0.5x Rainfall	1.16	0.87	0.68	0.81	0.62	0.55
1.5x Impermeable Surface	1.24	0.92	0.72	0.85	0.64	0.56
0.5x Impermeable Surface	1.19	0.89	0.69	0.82	0.63	0.55
Source 0.35	1.29	0.98	0.77	0.91	0.71	0.69
Source 0.42	1.35	1.05	0.84	0.98	0.78	0.70
1.5x Benthic Flux	1.22	1.04	0.81	0.96	0.76	0.67
0.5x Benthic Flux	1.06	0.86	0.64	0.78	0.58	0.51
2x Dispersion	0.74	0.64	0.53	0.61	0.50	0.46
0.5x Dispersion	2.36	1.69	1.11	1.35	0.93	0.72

Table III-7. Percent change in nitrogen concentrations from calibrated Linked Model with Attenuation at stations (1-6) in Great Pond for each parameter manipulated for sensitivity analysis of Linked Model approach, with mean and standard deviation of model change

Great Pond	percent change GT1	GT2	GT3	GT4 - Perch Pond	GT5	GT6	mean	std
1.5x Attenuation	-10.7	-9.4	-7.1	-8.7	-6.5	-6.4	-8.1	1.6
0.5x Attenuation	9.0	7.6	7.0	6.8	4.5	2.7	6.3	2.1
0.5x Fertilizer	-9.8	-7.4	-5.3	-7.7	-5.8	-5.5	-6.9	1.6
0.25x Fertilizer	-13.1	-10.9	-8.1	-11.0	-7.9	-7.8	-9.8	2.0
1.5x Septic Load	15.6	13.1	11.3	11.7	7.8	4.9	10.7	3.5
0.5x Septic Load	-16.4	-14.5	-10.8	-14.3	-10.6	-9.9	-12.7	2.4
1.5x Rainfall	4.1	2.7	3.0	1.9	1.1	0.9	2.3	1.1
0.5x Rainfall	-4.9	-4.1	-2.7	-4.4	-3.3	-3.5	-3.8	0.7
1.5x Impermeable Surface	1.6	1.2	2.0	0.8	0.2	-0.4	0.9	0.8
0.5x Impermeable Surface	-2.5	-2.5	-1.7	-3.2	-2.6	-3.5	-2.7	0.6
Source 0.35	5.7	7.7	10.1	7.6	10.0	22.3	10.6	5.4
Source 0.42	10.7	15.3	20.2	15.9	20.8	23.1	17.7	4.2
1.5x Benthic Flux	0.0	14.2	15.2	13.7	18.0	18.9	13.3	6.3
0.5x Benthic Flux	-13.1	-6.1	-8.8	-7.4	-9.8	-10.6	-9.3	2.2
2x Dispersion	-39.0	-29.9	-24.4	-28.1	-22.4	-19.6	-27.2	6.3
0.5x Dispersion	93.4	85.5	58.1	59.4	44.6	26.9	61.3	22.7

The Linked Model results appear to be relatively insensitive to atmospheric deposition with variations of 50% in loading rates causing shifts of less than 5%. Watershed attenuation was a more significant term, second only to septic load for its impact on Model output. Attenuation shifts of 50% resulted in 5%-10% shifts in Model output in the upper reaches of Great Pond and in Perch Pond. In all cases the effect of varying the nitrogen terms was largest in the upper reaches of the embayment and diminished toward the inlet. The effect is both a reduction in the percent change and the nitrogen mass change relative to the manipulation of model input terms. This is due to the increasing dominance of inflowing tidal waters near the inlet versus the dominance of watershed processes in the upper reaches of embayments. This latter effect is demonstrated by the results of varying the source water concentration which results in large (20%) changes in nitrogen levels near the inlet and diminishing effects in the upper estuary.

Benthic regeneration showed a similar degree of response as varying septic loading, averaging 9%-13% over the embayment with a range of 0%-18%. Benthic regeneration is a relatively large fraction of the total system nitrogen load. The results support the concept that regeneration plays a critical role in determining embayment nitrogen levels and the requirement that it be directly measured for accurate model predictions. The effect of changing regeneration rates was seen to be greatest in the mid region of the embayment, consistent with the field observation that deposition of nitrogen is not related to the sites of watershed nitrogen input.

The predicted embayment nitrogen levels from the Linked Model were most sensitive to varying the dispersion coefficients for nitrogen within the embayment waters. This is discussed in the following section.

The overall result of the sensitivity analysis is that the Linked Model predictions of embayment nitrogen level and distribution are relatively robust. The Model is most sensitive to (in the following order of most to least sensitive): watercolumn dispersion > source water nitrogen concentration > benthic regeneration, septic load > attenuation, fertilizer, impermeable surfaces. The effect of varying the watershed nitrogen loading or attenuation terms was largest in the upper reaches of the embayment and diminished toward the inlet. The effect is seen both as a reduction in the percent change and the relative nitrogen mass change. Dispersion was also most sensitive to upper estuary processes. This pattern is due to the increasing dominance of inflowing tidal source waters near the inlet versus the dominance of watershed processes in the upper reaches of embayments. This latter effect is demonstrated by the results of varying the source water concentration that results in large (20%) changes in nitrogen levels near the inlet and diminishing effects in the upper estuary. Benthic regeneration tended to show the largest changes at mid-estuary.

Embayment Modeling Parameters: As part of the Linked Model sensitivity analysis, dispersion coefficients were adjusted for the Great Pond water quality model. Two additional model runs were made using values 2 times and ½ times the dispersion values shown in Table III-8. While this is a greater range of variation than used in the nitrogen loading and transport modeling parameters (2-0.5 times versus 1.5-0.5 times), this range of dispersion better encompasses the potential range of dispersion coefficients within the 89 embayment systems encompassed by the Massachusetts Estuary Project.

Dispersion relates to the rate at which a constituent (e.g. nitrogen) is distributed within an estuary. When dispersion values are increased in the model, the effect is to increase mixing and speed up the diffusion of water quality constituents within a modeled embayment. Intuitively, the opposite is true for the case where model dispersion values are reduced, the mixing process is slowed down, allowing steeper gradients and higher concentrations of a constituent within an embayment. Since higher dispersion results in faster mixing, the result is a lower observed nitrogen concentration (the opposite is also true). Therefore, when dispersion was increased to 2 times the calibrated model values, the projected nitrogen concentrations decreased. The maximum decrease was 39%, at the upper portion of the pond. The minimum decrease was 19%, and the mean change in the projected pond nitrogen levels was a decrease of 27%. When dispersion values were reduced to ½ the calibrated values, there was an increase in modeled nitrogen concentrations. Again the maximum change was within the upper pond, 93%, while the smallest increase was 27% near the inlet. The mean increase over the entire pond was 61%.

These model runs of Great Pond show that modeled nitrogen concentrations are sensitive to selected values of the dispersion coefficient. By changing the dispersion values used in the calibrated model by +100% and –50% the observed mean change in model output was –27% and +61%. It is important to note that though the water quality model is very sensitive to changes in the applied dispersion coefficients, there is a high degree of confidence in the values used in a calibrated model. Values are determined in a process that typically uses salinity data to calibrate the water quality model, and then observed nitrogen concentration data for verification of the overall Linked Model performance. This process of calibration and validation uses independent data sets, dispersion coefficients are not separately adjusted using nitrogen levels.

Summary: The overall result of the sensitivity analysis is that the Linked Model predictions of embayment nitrogen level and distribution are relatively robust. The Model is most sensitive to (in the following order of most to least sensitive): watercolumn dispersion > source water nitrogen concentration > benthic regeneration, septic load > attenuation, fertilizer, impermeable surfaces. The effect of varying the watershed nitrogen loading or attenuation terms was largest in the upper reaches of the embayment and diminished toward the inlet. The effect is seen both as a reduction in the percent change and the relative nitrogen mass change. Dispersion was also most sensitive to upper estuary processes. This pattern is due to the increasing dominance of inflowing tidal source waters near the inlet versus the dominance of watershed processes in the upper reaches of embayments. This latter effect is demonstrated by the results of varying the source water concentration that results in large (20%) changes in nitrogen levels near the inlet and diminishing effects in the upper estuary. Benthic regeneration tended to show the largest changes at mid-estuary.

It is important to note that though the water quality model is very sensitive to changes in the applied dispersion coefficients, there is a high degree of confidence in the values used in a calibrated model. Values are determined in a process that typically uses salinity data to calibrate the water quality model, and then observed nitrogen concentration data for verification of the overall Linked Model performance. This process of calibration and validation uses independent data sets, dispersion coefficients are not separately adjusted using nitrogen levels.

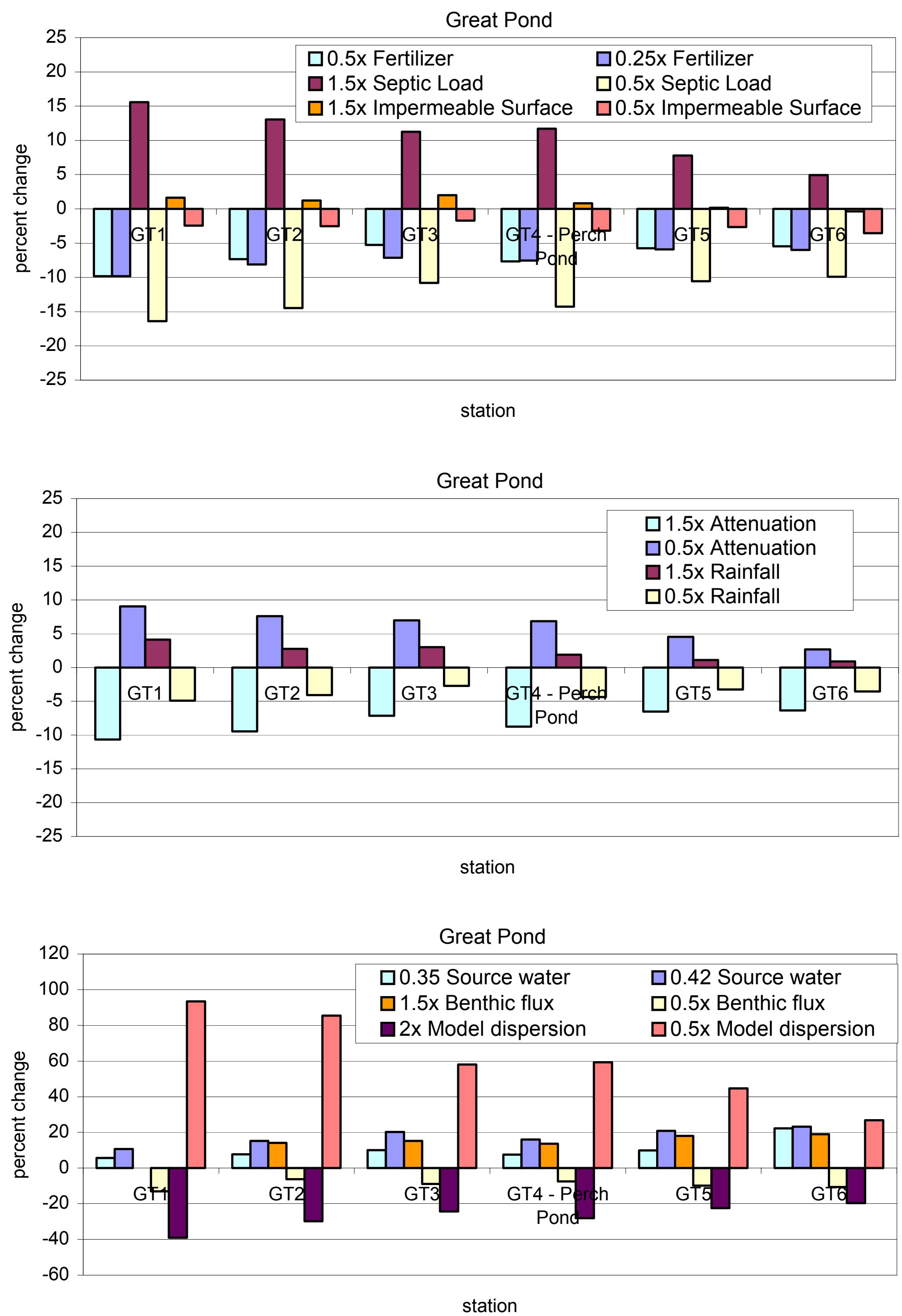

Figure III-9. Percent change in modeled Great Pond total N concentration at each station (1-6) resulting from step adjustments to model input factors.

IV. LINKED MODEL MANAGEMENT APPLICATIONS

Despite the difficulties, the protection and restoration of coastal embayments from nitrogen overloading has required the development of approaches for determining nitrogen thresholds. While this effort is ongoing, 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). While each approach may be different, they all focus on matching changes in nitrogen loading from watersheds to embayments with the goal of projecting the level of increase in nitrogen concentration within the embayment waters. Each approach depends upon estimates of circulation with the embayment; however, few directly link the watershed and hydrodynamic models and virtually none include internal recycling of nitrogen. Therefore, determination of the “allowable N concentration increase” or “threshold nitrogen concentration” remains somewhat subjective.

Since the nitrogen levels in receiving waterbodies increase gradually during the incremental development of coastal watersheds, their health undergoes gradual decline as a result of cultural eutrophication. The gradual ecological changes within estuarine systems, take the form of increasing phytoplankton production and epiphyte production and reducing light penetration. These processes reduce the habitat quality for eelgrass, but during initial stages of these processes or in “borderline” cases, eelgrass beds persist. At higher nitrogen levels, eelgrass beds will become less dense and will begin to disappear from the deeper areas. At even higher nitrogen levels, the beds will disappear. Since the presence of eelgrass beds in coastal environments is a generally accepted criterion of high quality conditions, the level of nitrogen at which eelgrass beds become impacted can be considered one type of “threshold”.

The difficulty is to link nitrogen concentrations to these general conditions of ecological health. The results for the Falmouth case studies using three attempts at nitrogen thresholds for embayments are shown in Table IV-1. The results represent site-specific analyses for the south coast of Falmouth; however, refinement of these “limits” will be performed as part on-going investigations of estuaries in Southeastern Massachusetts. The values that are used in the present analysis (SMAST/Applied Coastal, column 3) are the synthesis of data from the Cape Cod Commission (Eichner *et al.*, 1998) and Buzzards Bay Project/MCZM (Costa *et al.*, 1992 and in press), as well as comparative results from the Coalition for Buzzards Bay Monitoring Program (Howes *et al.*, 1999). In addition, these data were augmented with site-specific information from the Falmouth Pondwatch Program on Great, Green and Bournes Ponds (Howes and Goehringer, 1996). Information on eelgrass distribution and fish kills was developed from Pondwatch data. The rationale for each of the classifications of nitrogen based water quality thresholds is as follows:

Excellent Health: Nitrogen levels below 0.30 mgN/L are typical of nearshore Buzzards Bay (Howes *et al.*, 1999, Costa *et al.*, 1992 and in press), Vineyard Sound (Howes and Goehringer, 1996) and the scallop producing areas of Nantucket (Howes *et al.*, 1997). Waters with these nitrogen levels typically have only small oxygen depletions, generally not to less than 90% of air equilibration. Chlorophyll a pigment levels are typically less than 3 µg/L and transparency (secchi depth) greater than 3 meters. These coastal waters all support dense eelgrass beds and may have scallops. The conditions represent the “best” quality waters that Great, Green, and Bournes Ponds could support.

Table IV-1. Nitrogen thresholds and coastal water classifications used in the present

	study. Threshold values are site-specific. Abbreviations: CCC – Cape Cod Commission, BBP/MCZM – Buzzards Bay Project/ Massachusetts Coastal Zone Management, ND – not determined. Values are concentrations of total nitrogen (mg/L) within the water column of Great, Green, or Bournes Ponds.				
Classification of N based water quality	Trophic classification	CMAST/ Applied Coastal	CCC	BBP/MCZM	Mass Classification (310 CMR 4.05(4))
Excellent	Oligo to Mesotrophic	< 0.30	ND	ND	
Good	Oligo to Mesotrophic	0.30 – 0.39	< 0.34	< 0.39	SA
Moderate Quality	Mesotrophic	0.39 – 0.50	0.34 – 0.39	0.39 – 0.44	SB
Significant Impairment	Eutrophic	0.50 – 0.70	ND	ND	
Severe Degradation	Hyper-Eutrophic	> 0.70	ND	ND	
SA waters: (a) suitable for shellfish harvesting without depuration, (b) excellent habitat for fish, other aquatic life and wildlife and for primary and secondary contact recreation, (c) excellent aesthetic value. SB waters: (a) suitable for shellfish harvesting with depuration, (b) habitat for fish, other aquatic life and wildlife and for primary and secondary contact recreation, (c) consistently good aesthetic value. Note: these thresholds are currently being refined and are presented for discussion purposes only.					

Good Health: Good nitrogen related water quality conditions show enrichment over offshore source waters of Vineyard Sound, with some possible (but hard to quantify) decline in quality. Eelgrass beds are still present, benthic animal diversity and shellfish productivity are high, oxygen depletions to <4 mg/L are rare (if at all), chlorophyll a levels are in the 3 to 5 µg/L range. The Cape Cod Commission concluded that the threshold of nitrogen enrichment, which is protective of embayment habitat quality, is “background” plus 0.05 mg N/L, the Buzzards Bay Project using a similar approach determined “background” plus 0.10 mg N/L. Existing data indicates that there are embayments where each criterion (+0.05 or +0.10 mg N/L) is most appropriate. It is equally clear that +0.05 mg N/L is more protective of the embayment health. The CCC and BBP thresholds are <0.34 mg N/L and <0.39 mgN/L, respectively.

Moderately Impaired Health: Similar to the threshold for Good Quality areas, the upper limit where “moderate impairment” becomes “significant impairment” is somewhat broad. This is clearly a subjective point, as there is no clear ecological principal that can be used for reference. We can then define the threshold to “Significant Impairment” used for this evaluation as the nitrogen level where there is loss of diverse animal communities and replacement by smaller, shorter-lived animals of intermediate burrowing capabilities. Shellfisheries may shift to more resistant species. Oxygen levels do not generally fall below 4 or 5 mg/L, although phytoplankton blooms raise chlorophyll a levels to around 10 µg/L. Macro-algae may be present.

Significantly Impaired Health: As a result of the above discussion, the lower end of this nutrient related water quality classification, also referred to as “Eutrophic Conditions”, was set at 0.50

mg N/L. The upper end was determined from the threshold for “Severe Degradation” or “Hyper-Eutrophic” conditions. This upper end can be seen in the Buzzards Bay Monitoring Program results as 0.70 mg N/L. This level of nitrogen is associated with large phytoplankton blooms (chlorophyll a of approximately 20 mg/L) and such impacted environments as Eel Pond in Mattapoisett, Slocums River, and Little River. Within Great, Green, and Bournes Ponds, concentrations of approximately 0.7 mg N/L are associated with stressful oxygen conditions, major phytoplankton blooms, and complete loss of eelgrass. At higher levels, periodic fish kills, macro-algal accumulations, and aesthetic (odor) problems are observed. The range of 0.50-0.70 mg N/L is indicative of conditions where stress tolerant species persist in these ponds.

Severely Degraded: This classification is the analog to Hyper-Eutrophic, where periodic complete or near complete loss of oxygen occurs periodically in bottom waters. Macro-algal accumulations and fish kills are observed periodically. Drift algae, lift-off mats, and near complete loss of benthic animal communities occurs during a portion of the summer. The levels consistent with this definition are total nitrogen values of >0.70 mg N/L.

Case Study - Linked Model Approach, Management Application:

A. Ashumet Valley Finger Ponds, Falmouth, MA

In the study of the finger ponds of Ashumet Valley (Ramsey, *et al.*, 2000b), the site-specific data (specifically, the gradient in N concentration) and ecological health within the embayments monitored by Falmouth Pondwatch was used to “tune” general thresholds used by the Cape Cod Commission, Buzzards Bay Project and Massachusetts State Regulatory Agencies.

It appears from the nitrogen thresholds that the entirety of Great and Perch Ponds, Green Pond, and all but the most southerly section of Bournes Pond are currently showing “Significantly Impaired” nutrient related water quality conditions. Therefore, nitrogen management of these systems must be aimed at restoration, not protection or maintenance of existing conditions. The ultimate cause of the nitrogen overload to these systems is the increase in nitrogen inputs to the upper and lower watershed from changing land-use over the past century. The shift from pasture-land or forest to residential development with on-site disposal of wastewater, represents nearly a 10-fold increase in nitrogen loading on a per area basis. The year-round population of Falmouth has increased more than 5-fold from the 1930 census to the present time, with most of this increase being in the watersheds of the Town’s coastal ponds. The population of Cape Cod has increased 6-fold since 1920. In 1920, all of Cape Cod supported the same population that lives year-round in Falmouth today.

The water quality evaluation examined the potential impacts of nitrogen loading into three “finger ponds” along Falmouth’s south shore (Great, Green, and Bournes Ponds) and that resulting from existing and future natural, as well as anthropogenic sources. Once the hydrodynamic properties of each pond were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current and future loading rates. The water quality model, developed by King (1990b), has the capability for the simulation of the advection-diffusion process in the aquatic environment. The formulation of the model is for depth-averaged simulations in which concentration in the vertical direction is assumed to be uniform. For the “finger ponds” along the south shore of Falmouth, the depth-averaged assumption is justified since vertical mixing by wind and tidal processes prevent significant stratification in these shallow systems. Additional information regarding the water quality modeling effort can be found in Ramsey *et al.* (2000b).

Estimates for the dispersion coefficients required by the water quality model were determined by evaluating natural dispersion of salt within each pond. Salinity measurements during slack and/or ebbing tide conditions along the central axes of Great, Green, and Bournes Ponds provided the basis for dispersion estimates. Since salt is a conservative constituent (the only source is Vineyard Sound and there are no sinks); therefore, measurement of salt in combination with an analysis of freshwater inflow allows determination of dispersion/diffusion along the longitudinal axis of each pond.

Using dispersion relationships from the salinity study, the water quality and hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic processes. The distributions, strengths and discharge locations of nitrogen loads from watershed sources were determined from land-use analysis (coupled with direct measures) and the volume and locations of freshwater inflows. Almost all nitrogen entering the three ponds is transported by freshwater, predominantly groundwater. Estimates of the volume discharge and mean nitrogen concentration of entering waters to the various planning areas were used as the watershed sources in the water quality model. Concentrations in Vineyard Sound source waters were taken from Falmouth Pondwatch data. Measurements of current nitrogen distributions throughout harbor waters (from Pondwatch) were used to validate the water quality model (under existing loading conditions). In addition, the effects of increased nitrogen loading associated with projected build-out under current zoning conditions were evaluated within the context of increased pond nitrogen levels.

Nitrogen loading rates for the upper watersheds of each pond depended on the total nitrogen load generated from anthropogenic and natural sources. The load was distributed as a nitrogen concentration entering the streams at the upper end of each pond and a series of mass loads representing groundwater loading along the shoreline. Stream concentration represented the total load from the upper watersheds. Mass loads represented each “planning area” (shown in Figure IV-1) in the lower watershed of each pond. In this manner, simulation of various engineering methodologies for reducing nitrogen within specific planning areas could be performed efficiently. Although the mass loads of nitrogen were entered along the shoreline model elements, the exact location of where nitrogen enters the system is less important, since the model representation of the diffusion process rapidly disperses the nitrogen. Upland nitrogen loads for existing conditions are presented in Table IV-2.

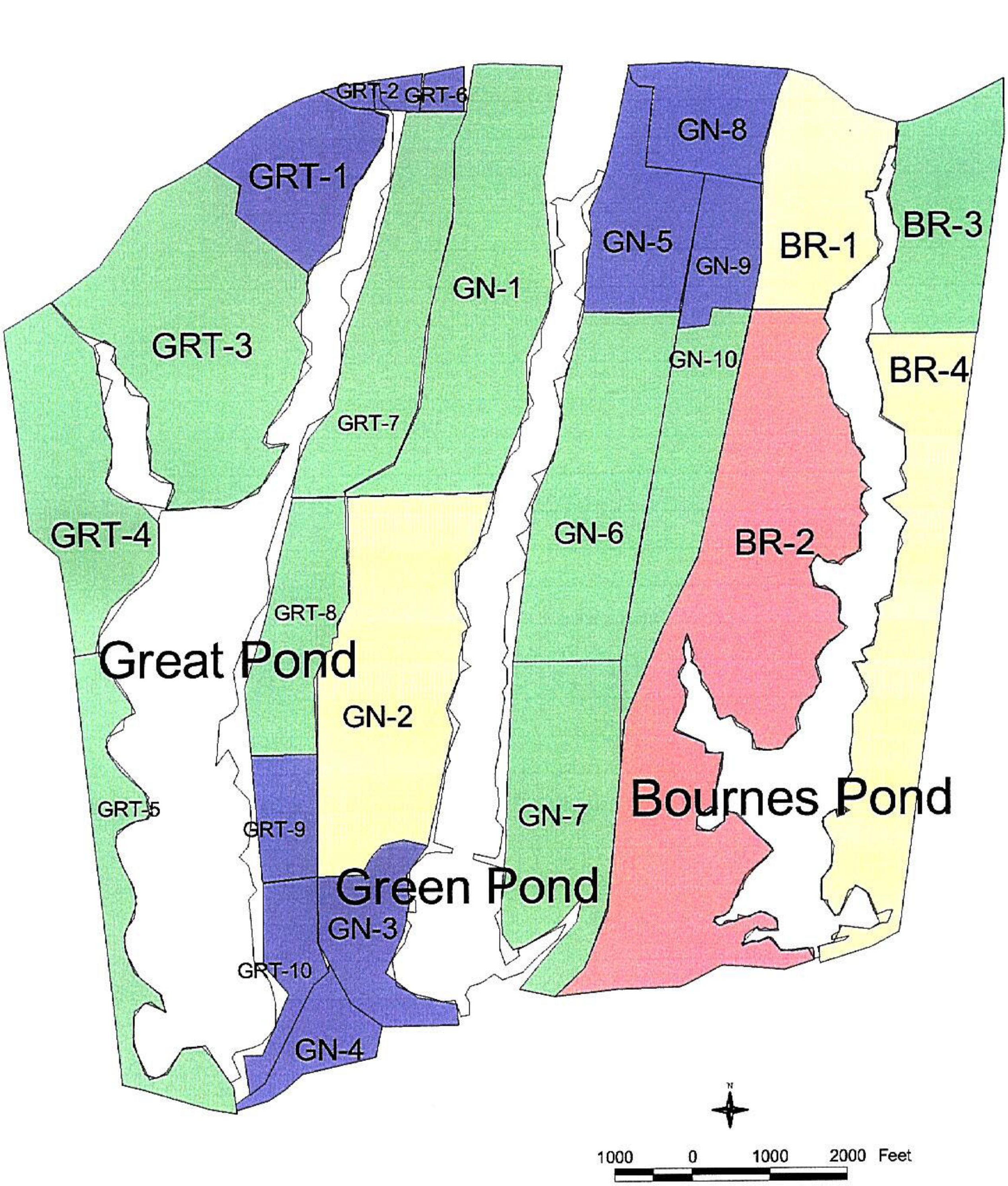

Figure IV-1. Planning area divisions for Great, Green, and Bournes Ponds used to determine nitrogen loading of ponds for water quality modeling.

TableIV-2. Present conditions for ground water nitrogen loading in the upper and lower watersheds of Great, Green, and Bournes Ponds. Loads are distributed by planning area for lower watershed, and by stream for upper watershed.					
Great Pond		Green Pond		Bournes Pond	
Planning Area	kg N/yr	Planning Area	kg N/yr	Planning Area	kg N/yr
GRT-1	2216	GN-1	1933	BR-1	890
GRT-2	24	GN-2	1966	BR-2	1160
GRT-3	3789	GN-3	306	BR-3	537
GRT-4	2174	GN-4	248	BR-4	682
GRT-5	1006	GN-5	656	Bournes Brook	877
GRT-6	54	GN-6	1043		
GRT-7	1264	GN-7	795		
GRT-8	787	GN-8	666		
GRT-9	309	GN-9	365		
GRT-10	154	GN-10	466		
Coonamessett River	8155	Backus River	3618		

The water quality modeling procedure consisted of running time simulations until steady state conditions were achieved. Three model runs were performed for each coastal pond: the first with the existing nitrogen loading conditions, the second with full build-out conditions, and the third with the removal of all septic loads. The second scenario represents the realistic "worst-case" scenario, where all land zoned for housing has been developed. The increase in total nitrogen load to the ponds associated with build-out conditions is shown in Table IV-3. Due to the extensive amount of development that exists around all three ponds, the effect of build-out conditions is relatively minor. The third scenario represents “base line” conditions that likely existed prior to degradation of estuarine water quality due to nitrogen loading from outside wastewater disposal associated primarily with residential development. The total nitrogen values computed for these conditions represent the “best-case” scenario for the future of the three ponds. The decrease in total nitrogen load to the ponds associated with septic load removal is shown in Table IV-4.

Table IV-3. Projected total increase of nitrogen loads for build-out conditions in lower watershed groundwater and upper watershed, stream inflows, for Great, Green, and Bournes Ponds.						
Watershed	Great Pond		Green Pond		Bournes Pond	
	kg N/yr	change	kg N/yr	change	kg N/yr	change
Stream Inflow – Upper watershed	2063	+25%	729	+20%	262	+30%
Groundwater Load - Lower watershed	1200	+10%	865	+11%	618	+19%

Table IV-4. Projected total decrease of nitrogen loads resulting from removal of on-site wastewater disposal from lower watershed groundwater and upper watershed, stream inflows, for Great, Green, and Bournes Ponds.						
Watershed	Great Pond		Green Pond		Bournes Pond	
	kg N/yr	change	kg N/yr	change	kg N/yr	change
Stream Inflow – Upper watershed	3515	-43%	1109	-31%	183	-21%
Groundwater Load - Lower watershed	7577	-64%	5372	-64%	1974	-60%

To illustrate the results of the water quality model, Figure IV-2 shows the modeled total nitrogen concentrations within Great, Green, and Bournes Ponds. The model results shown in Figure IV-2 represent total nitrogen levels during the critical summer months for existing conditions. Modeled values represent steady-state nitrogen concentrations averaged over the tidal cycle (tidally-averaged). Variability in the measured nitrogen concentrations is the result of a combination of environmental conditions (temperature, freshwater inflow, etc.), as well as tide phase. Although natural variability in total nitrogen concentrations exists, the model accurately predicts average total nitrogen concentrations within each of the coastal ponds. Within Great Pond, the nitrogen concentrations range from 0.75 mg/liter near the inlet to almost 1.0 mg/liter within the upper pond and throughout Perch Pond. Due to the narrow channel connecting Perch Pond to Great Pond, as well as the large sub-watershed associated with Perch Pond, total nitrogen concentrations in Perch Pond remain high. In the upper portions of Green Pond, the total nitrogen concentrations exceed 1.5 mg/liter. This value represents the highest nitrogen concentrations modeled in these three systems. Of the three ponds studied, Bournes Pond generally experiences the lowest concentrations of total nitrogen. Concentrations range

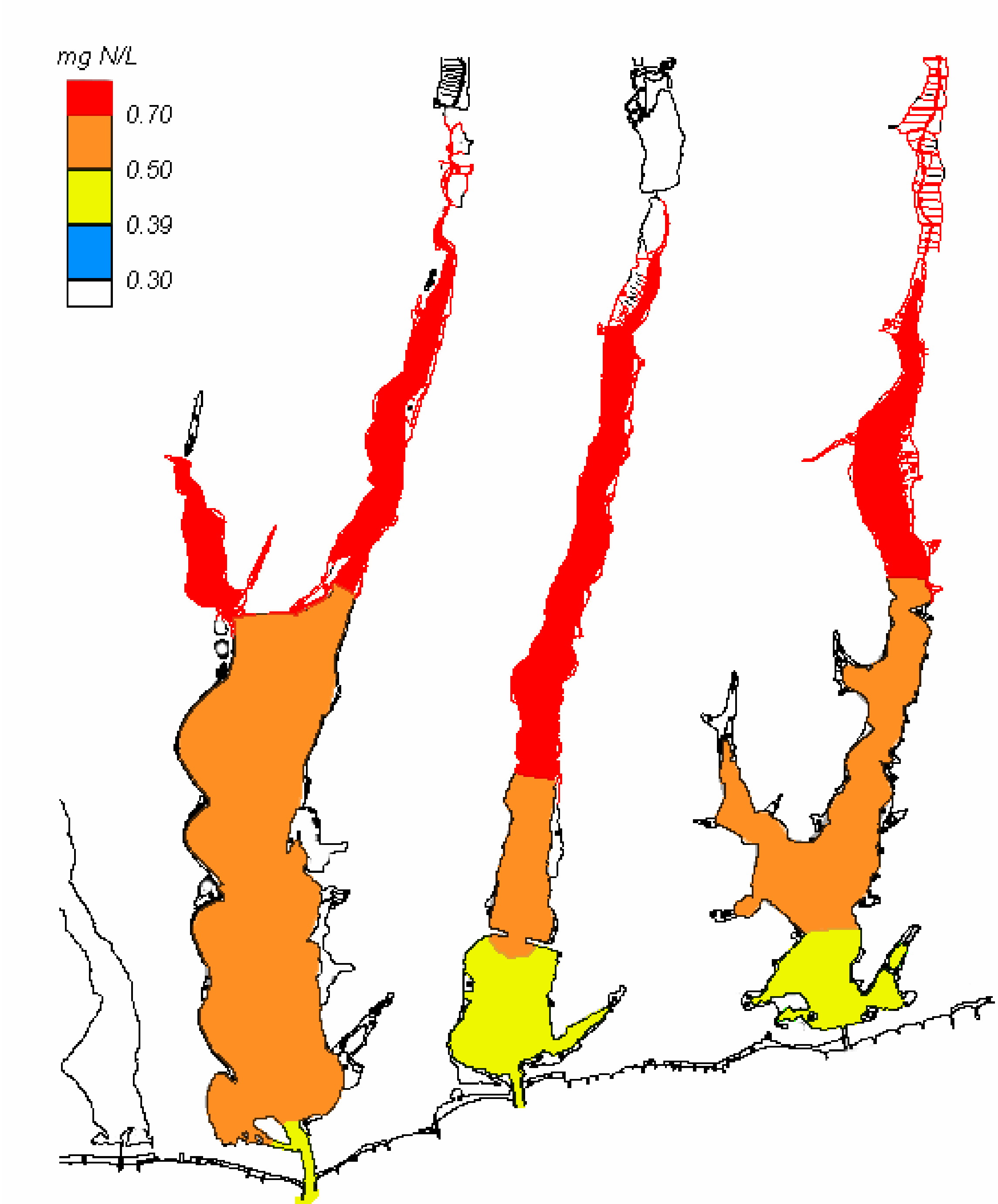

Figure IV-2. Modeled long-term nitrogen concentrations (mg N/L) in Great, Green, and Bournes Ponds for present conditions.

from approximately 0.5 mg/liter (near Vineyard Sound) to slightly greater than 1.0 mg/liter (at the north end of the pond).

For build-out conditions in the three ponds, Figure IV-3 illustrates the modeled total nitrogen concentrations at sites corresponding to the Pondwatch monitoring stations (shown in Figure I-1). Since relatively little developable land exists within the lower watersheds of Great and Green Ponds, most future development will occur in the upper watersheds. Based on

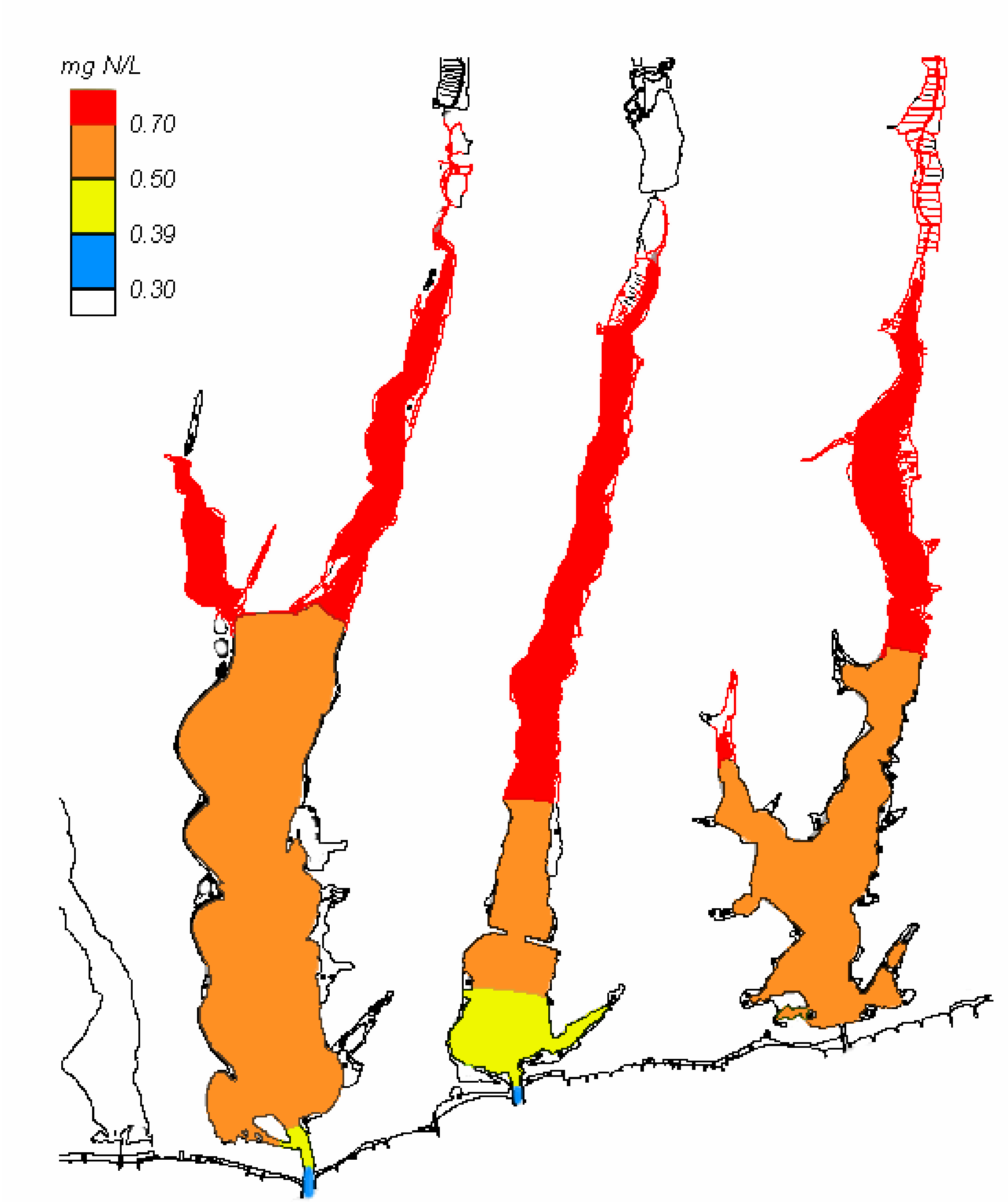

Figure IV-3. Modeled long-term nitrogen concentrations (mg N/L) in Great, Green, and Bournes Ponds for projected build-out conditions.

information from the nitrogen loading analysis, attenuation of total nitrogen in the upper watersheds is much higher than in the groundwater-dominated lower watersheds. Increasing the nitrogen load by build-out in the upper watersheds would only provide a fraction of the nutrients to the ponds as similar development in the lower watersheds. Since most of the lower watersheds of Great and Green Ponds have been developed, the impact of build-out from the lower watersheds is relatively small. In Bournes Pond, a significant portion of developable land remains in the lower watershed and approximately two-thirds of total nitrogen increase from build-out is derived from the lower watershed. The model indicated that approximately 1.0, 14, and 29 acres of pond area would degrade from “Moderate Quality” to “Significantly Impaired” within Great, Green, and Bournes Ponds, respectively. In addition, the “Severely Degraded” area would increase by 2.2 acres in Great Pond, by 3.2 acres in Green Pond, and by 9.4 acres in Bournes Pond.

Anticipated changes to total nitrogen concentrations at each Pondwatch monitoring station (Figure I-1) within the three ponds is shown in Table IV-5. In general, the greatest changes will occur in the upper portions of each pond. In the case of Green and Great Ponds, the development of the upper and lower watersheds is nearly complete; therefore, the changes in total nitrogen concentration associated with build-out are relatively minor. In Bournes Pond, the increased nitrogen levels are significant, especially in the northern portions of the pond. According to the model results, the total nitrogen concentration will increase by more than 15% at Station B1 as a result of build-out. Other portions of this pond show significant increases (more than 8%) of nitrogen relative to existing conditions.

Table IV-5. Values of percent change of nitrogen concentrations between modeled present and projected build-out conditions for each water quality sampling station within Great, Green, and Bournes Ponds.

Great Pond		Green Pond		Bournes Pond	
Station	% change	Station	% change	Station	% change
GT1	8.2	G1	5.6	B1	15.1
GT2	7.6	G2	5.2	B2	10.0
GT3	5.6	G2A	4.7	B3	8.3
GT4 Perch Pond	5.5	G3	5.9	B4	5.3
GT5	4.8	G4	5.2	B5 Israels Cove	5.5
GT6	3.4	G5	3.1	B6	4.9

In addition to the build-out scenario, total nitrogen loads based on groundwater and surface water sources without septic loads or the MMR wastewater treatment plume also were modeled. Due to the substantial land area within the upper watersheds, as well as the natural attenuation of nitrogen load within the freshwater ponds/streams in these regions, the decrease in nitrogen loads due to removal of septic systems is significantly lower in the upper watersheds in comparison to the lower watersheds. Septic loads within the lower watersheds are not attenuated within the sandy aquifer; therefore, removal of these loads from the analysis causes a significant reduction in total nitrogen loads (equal or greater than 60%) from these regions. Again, this nitrogen loading scenario represents “best-case” conditions, where the reduction in nitrogen assumes that all loading associated with septic systems does not exist. However, nitrogen loads associated with other anthropogenic sources (e.g. road runoff, golf courses, etc.) were not altered in this modeling effort. Although this simplification does not represent pre-development conditions, it does represent potential improvement to each pond assuming septic load can be treated in a manner that removes nitrogen contributions from the local watersheds.

Within the three ponds, Figure IV-4 indicates that total nitrogen levels will decrease significantly if loads associated with septic systems are removed. Similar to the build-out analysis, the effect of altering nitrogen loads on nitrogen concentrations is greatest at the northern end of each pond, but the effects on habitat quality are greater in the southern end of each pond. Due to the level of nitrogen loading from septic systems, both Great and Green

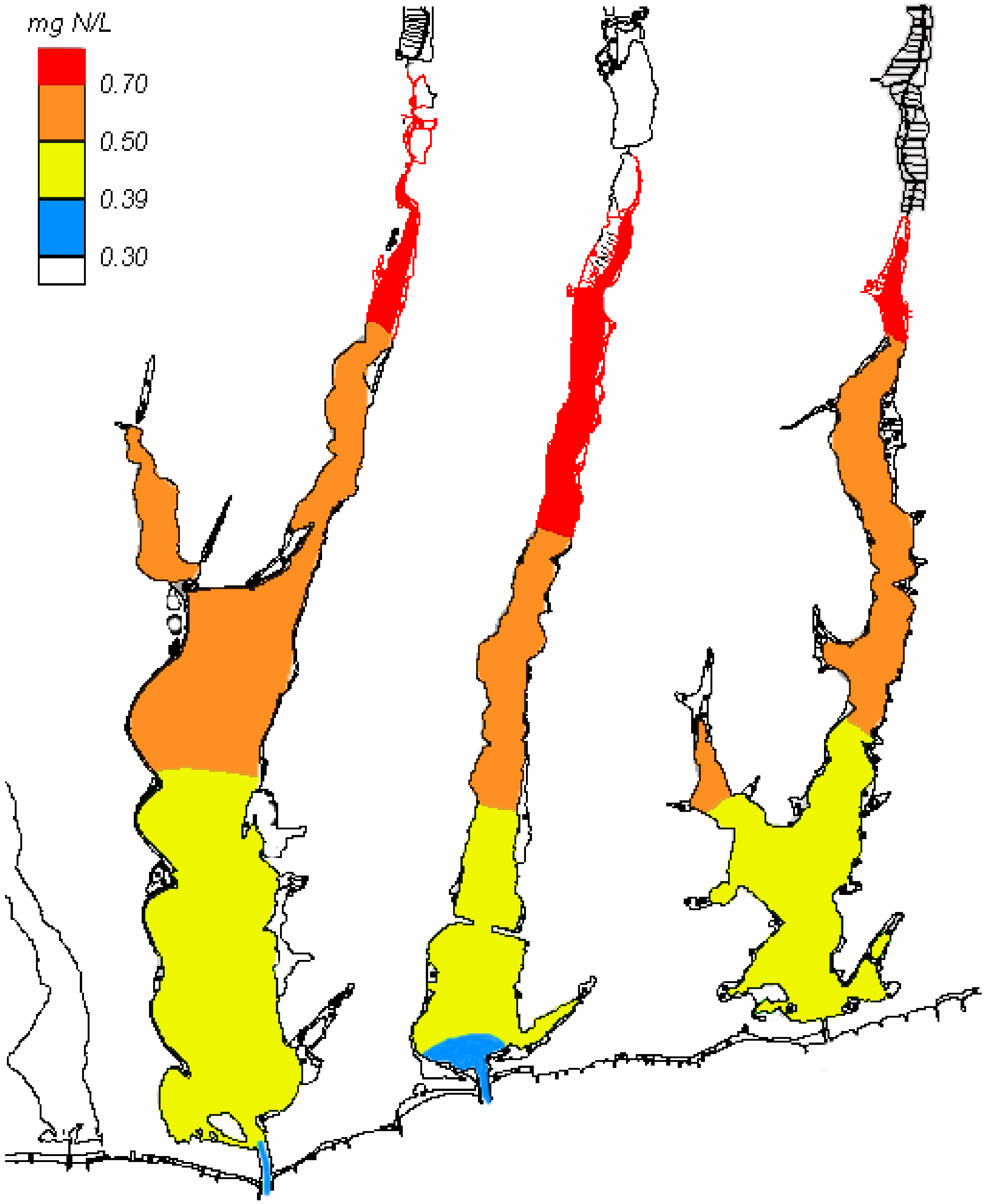

Figure IV-4. Modeled long-term nitrogen concentrations (mg N/L) in Great, Green, and Bournes Ponds resulting from removal of septic loads in upper and lower watersheds.

Ponds indicate significantly larger reductions in total nitrogen concentrations in comparison to Bournes Pond. The model indicated that approximately 132, 18, and 56 acres of pond area would improve from "Significantly Impaired" to "Moderate Quality" within Great, Green, and Bournes Ponds, respectively. In addition, the "Severely Degraded" area would decrease by 41 acres in Great Pond, by 32 acres in Green Pond, and by 30 acres in Bournes Pond.

Table IV-6 shows the percent reduction of modeled total nitrogen concentrations resulting from removal of septic loads in both the upper and lower watersheds for each pond. Although reduction of nitrogen levels is significant, especially within Great and Green Ponds, nitrogen concentrations without septic loading remain relatively high in the northern portions of each pond. For example, nitrogen concentrations north of the causeway in Green Pond will remain above 0.5 mg/liter, even with all septic loads removed.

Table IV-6. Values of percent change of nitrogen concentrations between modeled present conditions and projected conditions resulting from removal of septic loads in upper and lower watersheds for each water quality sampling station within Great, Green, and Bournes Ponds.

Great Pond		Green Pond		Bournes Pond	
Station	% change	Station	% change	Station	% change
GT1	-32.5	G1	-28.0	B1	-17.3
GT2	-27.4	G2	-27.4	B2	-17.4
GT3	-23.1	G2A	-27.4	B3	-15.1
GT4 Perch Pond	-30.7	G3	-26.0	B4	-10.3
GT5	-21.5	G4	-22.8	B5 Israels Cove	-11.5
GT6	-17.3	G5	-17.0	B6	-9.2

***Summary and Management Conclusions*:** The assessment of current and future water quality of Great, Green, and Bournes Ponds was achieved by assembling existing water quality information (Falmouth Pondwatch and SMAST monitoring data) on nutrients, and predictions of future changes in nitrogen levels estimated from the water quality model for each pond. In addition, the existing health of each pond has been assessed and appropriate nitrogen loading “thresholds” have been established.

The best mechanism to integrate the effects of nitrogen inputs from land with outputs via tidal exchanges is through numerical modeling of flows coupled to quantitative nitrogen input data. The water quality model was calibrated based on existing information and then used to predict levels of total nitrogen within the various sections of Great, Green, and Bournes Ponds. Predicting future nitrogen distributions and concentrations is a useful process for assessing potential changes to the ponds as a result of development. In addition, the calibrated hydrodynamic and water quality models can provide the basis for evaluating various nitrogen management strategies. For example, installation of sewers within various portions of the Ashumet Valley can be evaluated quantitatively by adjusting the nitrogen loading model to reflect the reduction in nitrogen entering the coastal ponds. The water quality model is then run to determine the nitrogen concentrations resulting from this reduction in loading. Future predictions need to be interpreted within the constraints of the model, since any forecasting is based on the existing conditions within Great, Green, and Bournes Ponds. Major changes in the biological systems which control nitrogen processing (e.g. a change from phytoplankton to macroalgal production), or physical features which control hydrodynamics (e.g. channel configurations) will invalidate the results.

The water quality modeling and the evaluation of ecological health revealed five major findings relating to the N-Offset Program goals of nitrogen management for restoration of Great, Green, and Bournes Ponds:

1) The upper reaches of each of the Great, Green and Bournes Ponds are currently showing poor nutrient related water quality as a result of nitrogen loading from the upper and lower watersheds. While the lower portions of each pond support at least moderate quality waters, only lower Bournes Pond exhibits a good level of environmental quality. The severely degraded environmental health of the upper portions of each of the Ponds is manifested in high chlorophyll a levels (>10 μg/L and typically >20 μg/L), periodic oxygen depletions to less than 4 mg/L, low water column transparency, and high nitrogen concentrations (>0.7 mg N/L). The nutrient overloaded nature of these systems is consistent with (a) the loss of eelgrass, (b) periodic fish kills due to oxygen depletion, and (c) periodic appearance of macro-algae. Each of the three ponds has total nitrogen concentrations above the levels set by the Falmouth Nutrient Overlay By-law. Since each of these coastal ponds shows signs of degraded water quality, steps should be taken to limit additional nitrogen loading.

2) The effects of nitrogen loading to each of the Ponds were based upon threshold nitrogen concentrations for the water column. These levels ranged from “Excellent Quality” where nitrogen levels average less than 0.30 mgN/L, “Good Quality” environments were associated with nitrogen levels between 0.30-0.39 mg N/L, “Moderately Impaired” was associated with levels from 0.39-0.50 mg N/L, systems were deemed “Significantly Impaired” at levels from 0.50-0.70 mg N/L, and “Severely Degraded” (hyper-eutrophic) above 0.70 mg N/L. These levels were determined from site specific data and thresholds developed by the Cape Cod Commission and Buzzards Bay Project.

3) Removing all nitrogen from on-site septic disposal of wastewater from the watersheds of each Salt Pond results in a significant lowering of nitrogen levels in both the upper reaches of each pond (Great: 23%-31%, Green: 27%, Bournes: 15%-17%) and the lower reaches of each pond (Great: 17%-22%, Green: 17%-23%, Bournes: 10%). However, even the removal of all wastewater nitrogen is insufficient to fully restore the Ponds to a high level of nutrient related health. Approximately the upper (northern) half of Green, Great, and Bournes Ponds would remain moderately to highly degraded by nutrient over-fertilization due to the low flushing rates associated with the small tide range, and nitrogen loading associated with non-wastewater sources and natural inputs. However, it appears that there would be a notable reduction in the present significantly impaired areas in each of the three Ponds. The effect would be that the lower basins of each Pond would be support nitrogen levels of less than 0.5 mg N/L, which would be a significant improvement in ecological health.

4) Under current zoning and on-site wastewater disposal practices, nitrogen loading to the Ponds at full build-out of the watersheds will increase by only 13%-16%. However, this modest increase in nitrogen loading is projected to reduce the remaining better water quality regions within the lower reaches of each pond. Within Great Pond the <0.6 mg N/L zone will be reduced by about half, currently the entire pond is >0.5 mg N/L. The present moderately impaired lower basin (below the bridge/causeway) in Green Pond is project to see a rise in nitrogen to >0.5 mg N/L over about 40% of the lower basin area. Bournes Pond, which currently has the lowest level of relative nitrogen loading and concurrently the highest water quality (primarily in the lower reaches), will see a relatively small shift primarily near the inlet. This small concentration shift is sufficient to move this region from moderately impaired to significantly impaired, and therefore may represent an important ecological shift.

5) Pondwatch data has indicated that both nitrogen and chlorophyll a periodically increase to excessive levels in Perch Pond. This increase likely is related to poor exchange of Perch Pond waters with Vineyard Sound waters (poor tidal flushing); therefore, channel dredging should be evaluated as a method for improving tidal flushing in Perch Pond. Dredged sediment could be utilized to (a) fill the deep portion of Perch Pond to reduce the volume and improve tidal flushing, and (b) cover the red tide cysts that exist within the bottom sediments.

Case Study - Linked Model Approach, Management Application:

B. Frost Fish Creek, Chatham, MA

Frost Fish Creek is connected to Pleasant Bay on Cape Cod through the Bassing Harbor System (Figure IV-5). Currently, the creek exhibits relatively poor tidal flushing due to the poor condition of the culvert that connects the upper portion of the creek with Bassing Harbor. Based on the previous hydrodynamic modeling (Kelley *et al*., 2001), it was anticipated that water quality improvements to these systems likely can be achieved through either resizing of culverts or turning upper portions of the coastal embayments into freshwater ponds. Evaluation of potential alternatives is critical to achieve water quality goals, as well as to avoid adverse environmental impacts. The hydrodynamic model utilized to evaluate tidal flushing provide the basis for quantitatively analyzing the effects of various alternatives on tidal exchange. Using the calibrated model for the system, the model grid was modified to reflect alterations in culvert dimensions and/or bathymetry. Once the hydrodynamic simulations were completed, total nitrogen modeling of each scenario was performed to indicate changes in water column nitrogen concentrations. Similar to total nitrogen modeling presented for the Ashumet Valley finger ponds, boundary conditions for the RMA-4 water quality simulation utilized watershed nitrogen loading, benthic flux of nitrogen from bottom, and offshore nitrogen concentrations in Pleasant Bay.

In its present configuration, water levels in Frost Fish Creek are controlled by the combination of three culverts under the Route 28 roadway embankment, and a dilapidated weir that is located upstream of the culverts. The 1.5 ft diameter roadway culverts are partially blocked, which results in poor tidal exchange (> 0.5 ft tidal range) with the lower portion of Frost Fish Creek and Bassing Harbor. To determine the effects of culvert improvements on water quality on the upper portion of the Creek, two alternatives were studied: Alternative F1 was designed to increase the tide range in the creek to 1 ft by installing a new box culvert with a width of 5 ft; Alternative F2 was designed to increase the tide range of the Creek to 1.5 ft by installing a 7 ft wide box culvert. Hydrodynamic modeling of the two alternatives showed that the local residence times of would decrease from 3.0 days for the present conditions, to 1.3 days, and 0.9 days for Alternative F1 and F2 respectively. Total nitrogen modeling using the RMA-4 constituent transport model was performed to better quantify the actual impact of these two proposed culverts on water quality in the Creek.

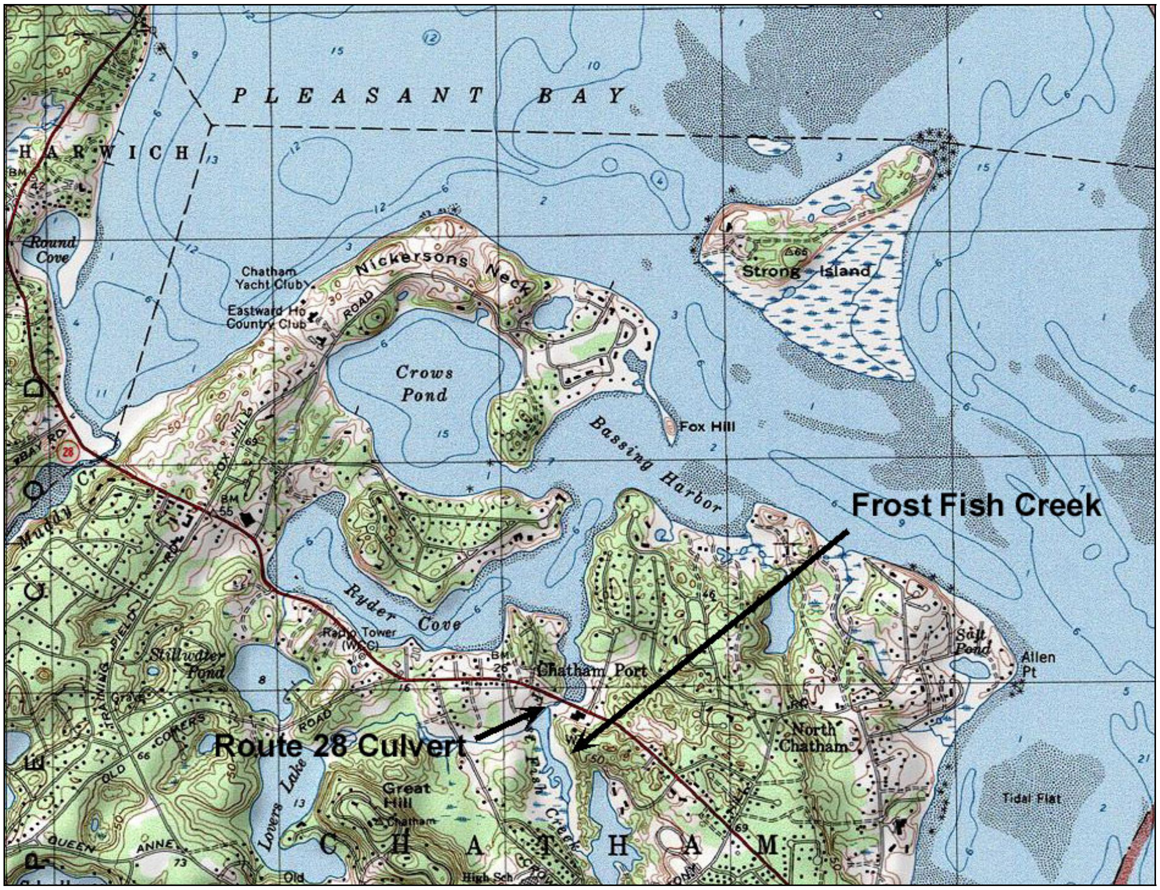

Figure IV-5. Detail of Chatham, MA topographic map showing Bassing Harbor system and Frost Fish Creek

Based on the hydrodynamic model results for Frost Fish Creek, Table IV-7 illustrates the change in tidal flushing associated with the two culvert alternatives. The smaller culvert alternative (Alternative F1) provided a tide range of about 1.0 feet, with a significantly reduced local residence time of 1.3 days. The larger culvert alternative (Alternative F2) provided approximately a 1.5 ft tide range, as well as a lower residence time than Alternative F1. The tidal curves for Alternatives F1 and F2 relative to existing conditions are shown in Figure IV-6. Due to the substantial tidal attenuation caused by the existing (partially blocked) culverts, the model indicated installation of larger culverts would significantly reduce the mean tide level with a negligible increase in the high tide elevation.

Table IV-7. Comparison of system volume, tide prism, and residence tides for Frost Fish Creek for alternatives F1 and F2.

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

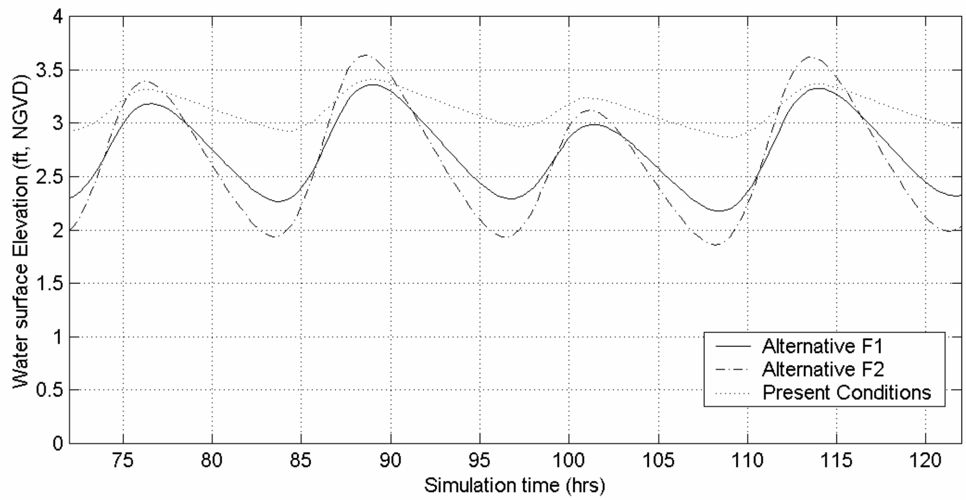

Figure IV-6. Modeled tide range in Frost Fish Creek for alternative culvert designs, F1 and F2, compared with present conditions.

Though the flushing rates were significantly improved for each of the culvert alternatives, total nitrogen modeling reveals that there would be actually no improvement to water quality in the upper portion of Frost Fish Creek. Figures IV-7 through IV-9 show contour plots of model total nitrogen for present conditions and the two model alternatives. As can be seen in these figures, the total N concentrations actually increase in Alternative F1, and again for F2. The reason for the increased concentrations is related to how the system volumes change for the alternatives. For example, in Alternative F1 the mean system volume is reduced by approximately 15%, and the mean low tide volume is reduced by approximately 30%. Therefore, while the N load to the Creek remains the same, with a reduction in system volume, there will be a resulting increase in mean concentrations. Therefore, total nitrogen modeling shows that the culvert alternatives as configured will not improve water quality, even though flushing in the upper portion of the creek is improved, hence these alternatives should not be considered further in the future.

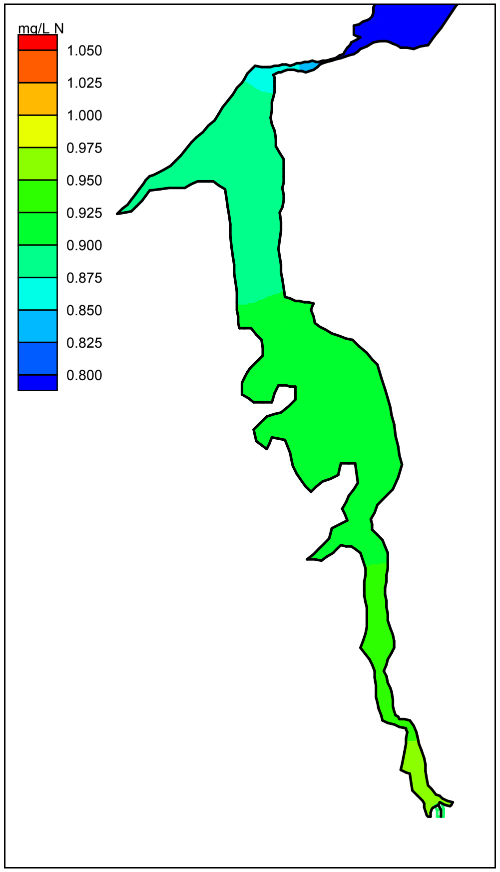

Figure IV-7. Modeled Frost Fish Creek total nitrogen concentrations for current culvert and weir configuration, with an approximate 0.5 ft tide range

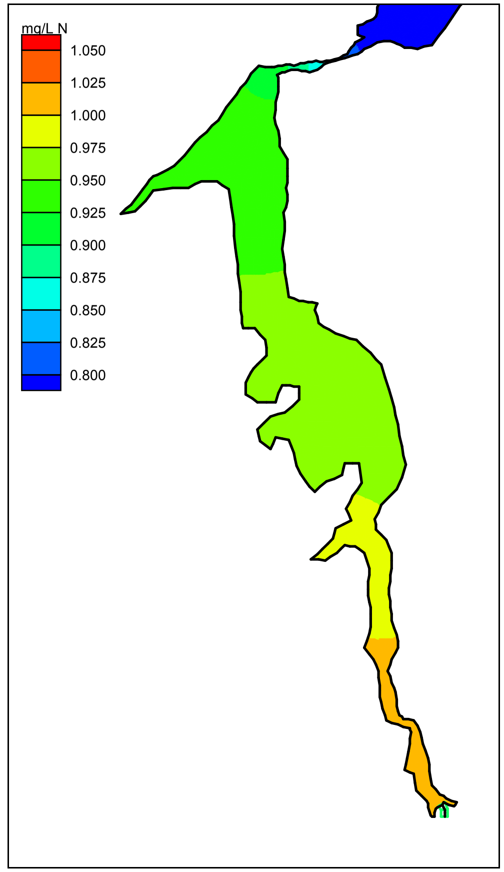

Figure IV-8. Modeled Frost Fish Creek total nitrogen concentrations for culvert design alternative F1, a 5 ft wide box culvert, which increases the tide range in the upper portion of the creek to approximately 1 ft.

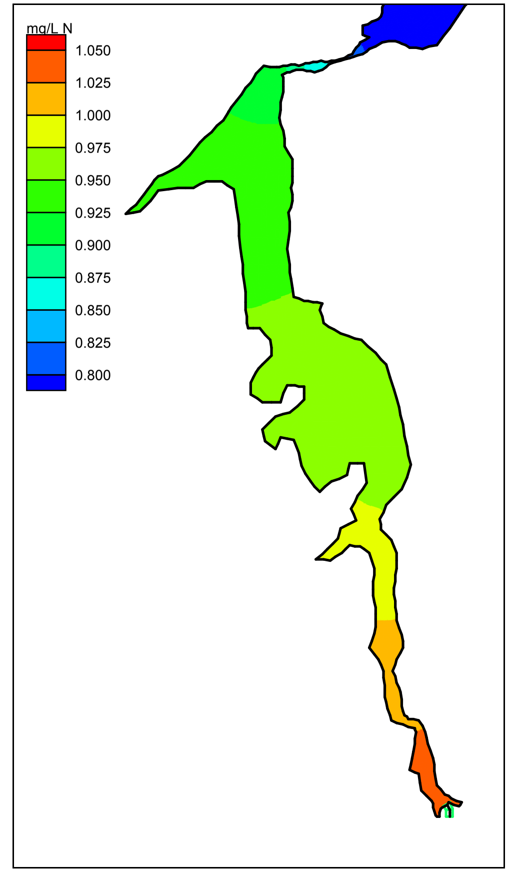

Figure IV-9. Modeled Frost Fish Creek total nitrogen concentrations for culvert design Alternative F2, a 7 ft wide box culvert, which increases the tide range in the upper portion of the creek to approximately 1.5 ft.

As described above, improving tidal exchange by increasing culvert size alone will not improve total nitrogen concentrations within Frost Fish Creek. Due to the significant flow restriction caused by the existing culvert systems, improvements in tidal exchange tend to lower the mean sub-embayment volume. Therefore, this lower system volume causes the same nitrogen load to have a higher concentration. A balance between tidal exchange and mean sub-embayment volume governs the overall estuarine nitrogen concentration. If the tidal source waters are low in nitrogen and a culvert alternative generates a tidal exchange that is large relative to the sub-embayment volume, a significant improvement in water quality can be anticipated. However, the culvert alternatives modeled only allow moderate improvements to tidal exchange, but each alternative causes a significant reduction in sub-embayment volume. Therefore, the culvert alternatives evaluated actually caused an increase in nitrogen concentration (potentially a decrease in ecological health of the sub-embayment). Generally, it is anticipated that improvements to tidal exchange improve a sub-embayment's water quality. However, the total nitrogen modeling indicates that this is not the case.

Figure M-10 illustrates the modeled existing conditions at the water quality monitoring station in Frost Fish Creek. In addition, the Figure shows the range and mean modeled nitrogen concentrations created by each of the alternatives. For Frost Fish Creek (Figure M-10), none of the alternative culvert designs lower nitrogen concentrations due to the greatly reduced mean volume relative to the tidal exchange.

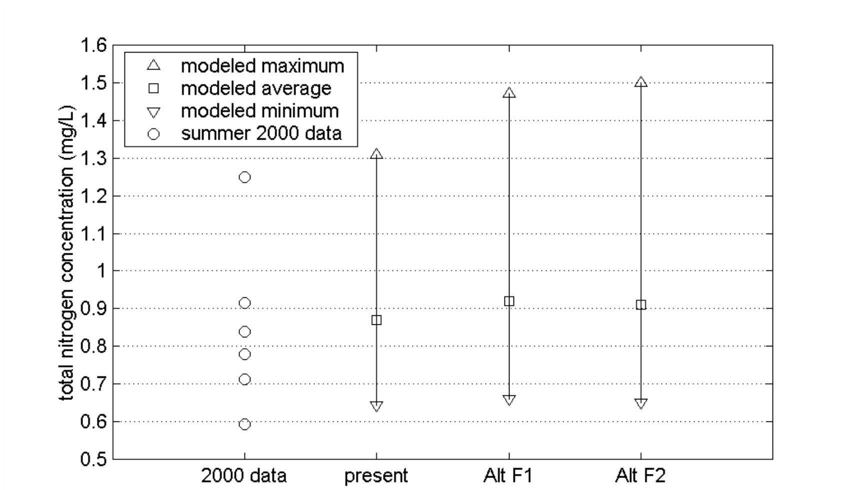

Figure IV-10. Maximum, mean, and minimum total nitrogen concentrations at the Frost Fish Creek water quality sampling station for present conditions and for the culvert alternative F1 and F2 determined from the model versus measured total nitrogen concentrations collected during summer 2000

Although the culvert options modeled for Frost Fish Creek did not improve nitrogen concentrations and the associated water quality, it is possible to develop culvert alternatives that will lower nitrogen concentrations. In general, flow control structures can be installed to prevent a reduction in mean sub-embayment volume, while allowing improved tidal exchange. These structures typically would be designed as weirs that do not allow the water elevation to drop below the existing low tide level. For Frost Fish Creek, modification of the existing weir structure would provide an appropriate solution. Since larger culverts will increase the high tide elevation within Frost Fish Creek, more tidal exchange will occur and the mean system volume will remain the same (or increase slightly). Therefore, the increased tidal exchange with a similar sub-embayment volume will reduce nitrogen concentrations. Optimization of the culvert and weir design is beyond the scope of the existing project, but existing model results indicate that a system could be designed to significantly improve nitrogen concentrations within Frost Fish Creek.

Case Study - Linked Model Approach, Management Application:

C. Little Mill Pond, Stage Harbor, Chatham, MA

Water quality in a sub-embayment of a larger system depends greatly on the position of the embayment relative to the system inlet, where embayment waters are exchanged with cleaner water from outside the system. Little Mill Pond is such an embayment, where water quality in Little Mill Pond is impacted more by the quality of waters immediately adjacent to it (Mill Pond), than by the watershed load that it receives. Mill Pond is located at the end of the Mitchell River branch of the Stage Harbor System. The approximate 4 ft tide range at Little Mill Pond is indicative of good tidal exchange between Little Mill Pond and Mill Pond, however because of the geometry of the system, a substantial portion of the water that exits the Pond during an ebb tide returns on the following flooding tide, and the result is fairly poor exchange with cleaner water from Nantucket Sound. The poor flushing characteristics of the Pond compound the impact of nutrient loading from groundwater, and result in the current poor water quality experienced at the Pond.

To demonstrate how the adjacent waters to Little Mill Pond impact the water quality in the pond, RMA-4 model runs were performed using a calibrated RMA-2 hydrodynamic model of the Stage Harbor system. For the modeled scenario, all wastewater nitrogen loads in the Little Mill Pond watershed where removed, as would be the case if the Pond's watershed was to be fully sewered. This would reduce the watershed load to the pond by 78%, from 1399 kg/yr (S&W, 2000) to just 303 kg/yr, which is generated by sources other than wastewater. All remaining watershed loads to the Stage Harbor system were not changed in the modeled scenario.

The resulting change in modeled nitrogen concentrations in the Stage Harbor System resulting from the sewering of the Little Mill Pond watershed is presented graphically in Figure IV-11. A maximum reduction of 0.08 mg/L occurs in the Pond, which represents a change of only 10%, from 0.77 mg/L to 0.69 mg/L (see Figure IV-12 for existing nitrogen distribution in the Stage Harbor system). Therefore, the 78% reduction in N loading to the Pond's watershed would have a negligible effect on the nitrogen concentrations within the Pond. In this case, substantial improvement of sub-embayment health would require management of tidal source water nitrogen. Additional reductions in nitrogen loads to the remaining watershed areas of the Mitchell River branch of Stage Harbor potentially could improve water quality.

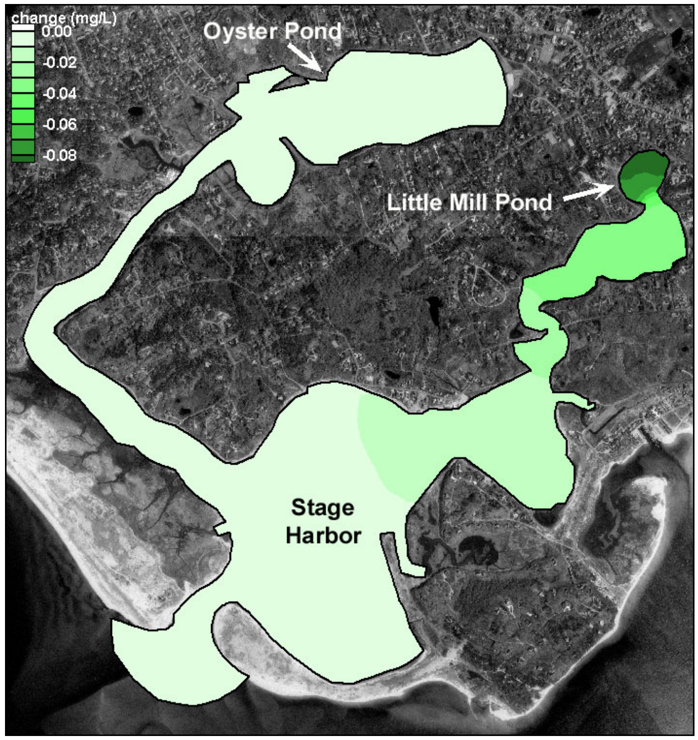

Figure IV-11. Plot of change in modeled nitrogen concentrations (mgN/L) in Stage Harbor, Chatham, MA resulting from removal of all watershed wastewater nitrogen loading to Little Mill Pond versus present conditions.

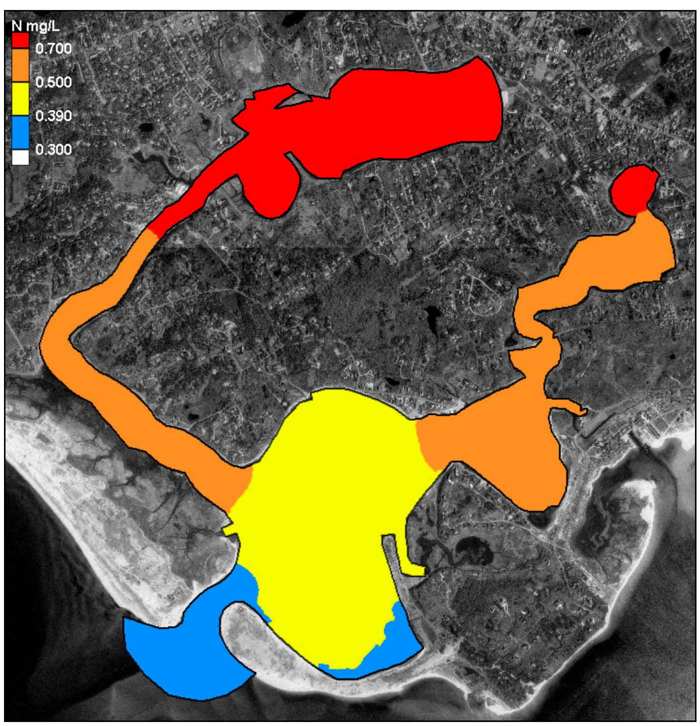

Figure IV-12. Plot of modeled nitrogen concentrations in Stage Harbor, Chatham, MA, under the present condition of watershed nitrogen loading.

Case Study – Linked Model Approach, Management Application:

D. Perch Pond Tidal Restriction, Falmouth, MA

Utilizing the validated hydrodynamic and water quality models derived from our diverse studies of the Perch/Great Pond System (discussed above), we conducted an analysis of tidal exchanges and nitrogen conditions within Perch Pond. This analysis focused on determining the (1) level of tidal restriction resulting from sedimentation of the channel between Perch Pond and Great Pond, (2) the extent to which tidal flushing will be increased by dredging, and (3) the level of nitrogen related water quality improvement in Perch Pond and decline in Great Pond resulting from increasing tidal flows.

The use of our field validated Linked Watershed-Embayment Model is appropriate to the dredging issue, since it was based upon site specific measurements of hydrodynamics (1999) and habitat quality (1985-1999) made within Perch Pond, Great Pond, and Vineyard Sound. However, it was necessary to perform additional modeling to better match the Town's proposal to widen the dredged channel into Perch Pond to 40 feet. The refined model should accurately predict the impact of dredging on tide range, tidal flushing, and total nitrogen concentrations for Perch and Great Ponds. These flushing and nitrogen results are the basis for predicting changes in environmental health of Perch and Great Ponds relating to the dredge project. This modeling represents the state-of-the-art approach for predicting changes in nutrient related environmental health in coastal waters.

Tidal Flow Analysis: One of the major questions to be addressed by the project is whether lowering the controlling channel depth of tidal flushing by 2-to-4 feet will decrease the level of low tide and potentially expose more pond bottom; therefore, potentially increasing odor problems. Although minor tidal attenuation (reduction in tide height) occurs through the existing Perch Pond entrance, deepening the channel and widening to a 40 ft width will result in only a very small lowering of the low tide level. As shown in Figure IV-13, the low tide elevation following the proposed dredging will be a maximum of one inch lower than the existing low tide elevation. Therefore, the proposed dredging will have a negligible impact on the tide range. Any increase in area of pond bottom exposed at low tide will also be small given the bathymetry of this kettle hole pond. In any case, it is more likely that odor problems (to the extent that they currently exist) are more the result of low oxygen conditions or accumulations of decaying macro-algae or even the salt marshes around the shoreline, rather than from exposed bottom sediments.

Generally, only small improvements to tidal flushing are achieved in estuarine systems where the tide indicates relatively little attenuation. However, the existing narrow, shallow channel that connects Perch Pond to Great Pond inhibits the exchange of tidal flow between the two ponds. Based on the hydrodynamic model, changes in tidal flushing associated with Perch Pond dredging are shown in Table IV-8. The model indicates that the Perch Pond flushing is improved by more than 25% as a result of dredging; however, there will be a minor (1%) decrease in flushing of the entire Great Pond system.

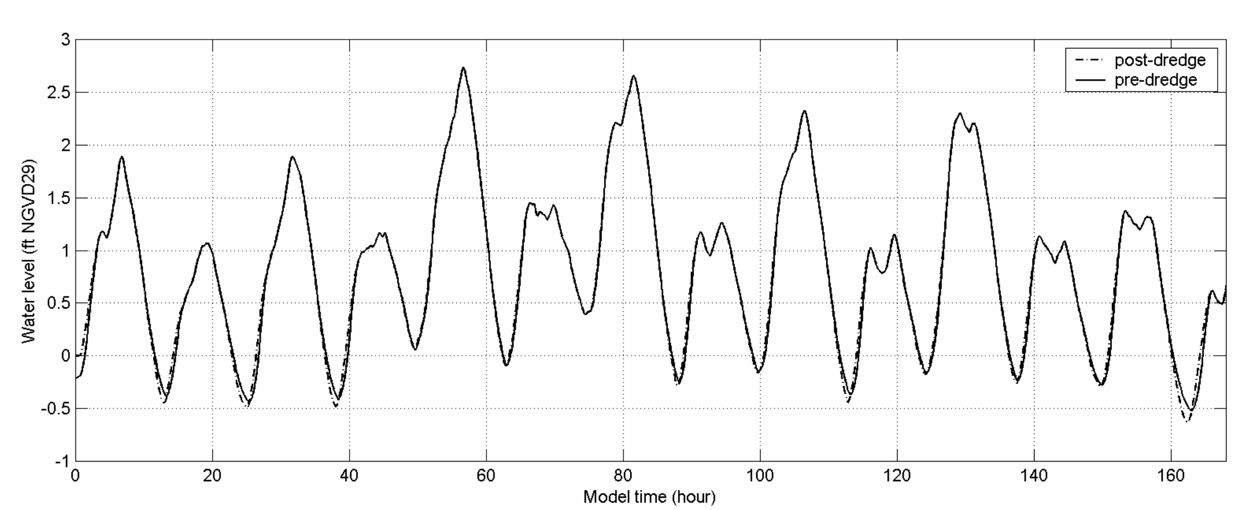

Figure IV-13. Comparison of modeled time series of water level in Perch Pond, under current conditions and after enhancing flow by dredging the tidal inlet.

Table IV-8. Modeled flushing rates for pre- and post-dredge conditions at Perch Pond tidal inlet.

Sub-Embayment	Pre-dredge Local residence time (day)	Post-dredge Local residence time (day)	Percent change
Perch Pond	1.53	1.14	-25.5
Great Pond (upper)	0.74	0.74	0.0
Great Pond (total)	0.99	1.00	1.0

Although the tidal range within the Perch Pond will not be significantly altered by the proposed project, a significant improvement should be observed in water quality. Often tidal flushing is used as an indicator of estuarine health, but habitat quality is better gauged by total nitrogen concentrations. Total nitrogen concentrations in the Perch/Great Pond System are the integration of tidal flushing, watershed nitrogen inputs and nitrogen recycling within the embayments. Total nitrogen concentrations relate directly to key habitat parameters like water clarity, algal blooms, and oxygen levels.

The validated site-specific water quality model provides the predictive capability for evaluating the nitrogen concentrations throughout the Great Pond system associated with the proposed dredging. It should be noted that a detailed evaluation/quantification of upland nitrogen loading to the Perch and Great Pond watersheds was included as part of the water quality analysis. Based on the model results, Table IV-9 indicates that a reduction of 0.10 mg/L of total nitrogen (about 11%) would be achieved by dredging the Perch Pond entrance channel. This value is more than 35% of the reduction that could be achieved through removal of all septic loads to the system (see Table 6 Ramsey et al. 2000 for a comparison). While Perch Pond will still show degraded water quality as a result of continued watershed nitrogen inputs, the dredging will produce a significant reduction in total nitrogen levels within Perch Pond waters and a concomitant improvement in water quality. Increasing tidal exchange through the Perch Pond inlet is a viable option for habitat quality improvement/management. However, maximizing water quality conditions within Perch Pond also requires watershed nitrogen management, which is ongoing through the Nitrogen Offset Program.

Although there will be a significant improvement in the health of Perch Pond, the model does not predict a measurable increase in nitrogen concentrations within Great Pond as a result of the proposed dredging. The reason for this non-effect is that almost all of the nitrogen entering Perch Pond is currently being flushed to Great Pond. The proposed project would decrease the time a specific nitrogen mass stays within Perch Pond, but not the amount that moves through the system. Numerically, these results are reflected in Table IV-9. Visually, the changes in average total nitrogen concentrations are shown in Figure IV-14, where the blue areas indicate virtually no change. In addition, the overall nitrogen concentrations for existing and post-dredging conditions are shown in Figures IV-15 and IV-16.

Our evaluation of the data indicates that a significant improvement to the Perch Pond system will result with undetectable negative impacts on Great Pond health. Issues involving sediment transport should be evaluated by considering by-passing sand from the up-drift to down-drift side of the inlet (south-to-north).

Table IV-9. Comparison of measured and modeled total nitrogen concentrations (mg/L). Measured values were compiled from data collected as part of the Falmouth Pond Watch program. Measurements are from the summer seasons of nine years (1990-1998). Model results are time-averaged concentrations, for pre- and post-dredge conditions.

Pond Watch Station	Measured Mean	Measured Max	Measured Min	Modeled Pre-dredge	Modeled Post-dredge	Percent change (modeled)
GT1 (Uppermost)	0.94	1.35	0.71	1.12	1.12	0
GT2	0.90	1.12	0.78	0.91	0.91	0
GT3	0.77	0.91	0.51	0.71	0.71	0
GT4 (Perch Pond)	0.88	0.98	0.75	0.85	0.75	-11
GT5	0.61	0.80	0.46	0.64	0.64	0
GT6 (Lowermost)	0.52	0.66	0.41	0.57	0.57	0

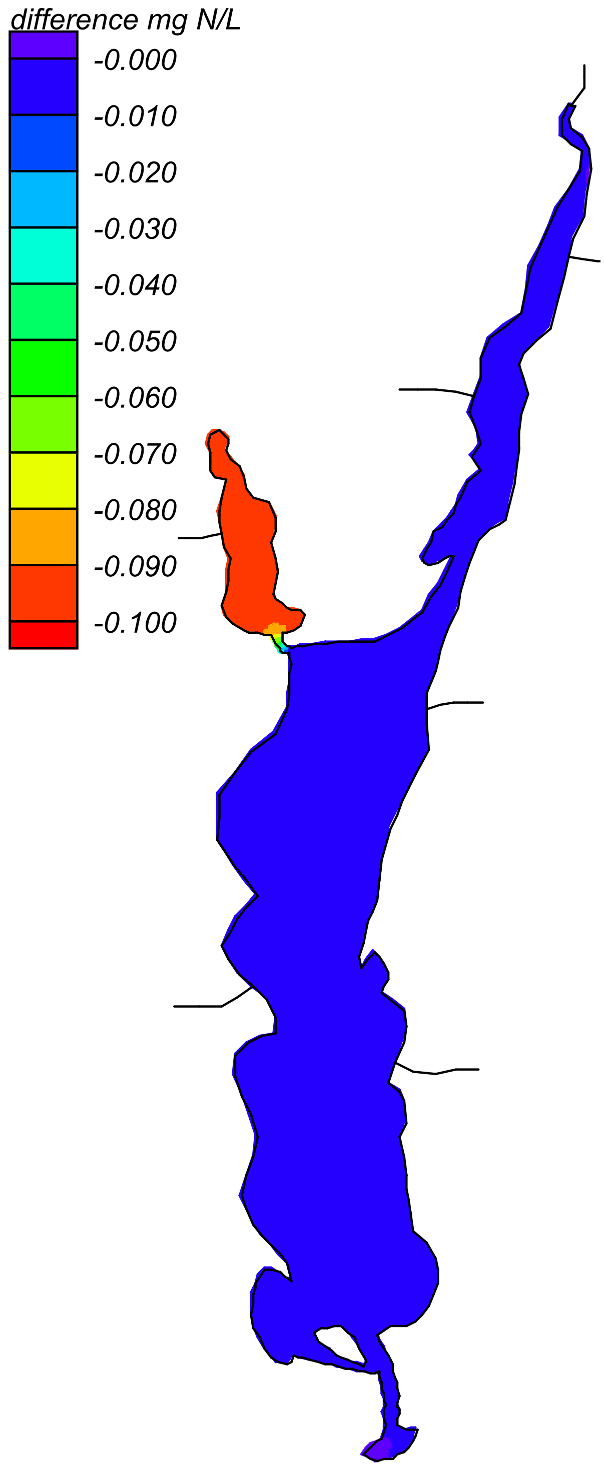

Figure IV-14. Difference in modeled time-averaged total nitrogen levels (mg/L) for Great Pond, showing changes resulting from dredging of the Perch Pond channel to –5 ft NGVD29. Pre-dredge model results were subtracted from Post-dredge results. Negative difference indicates a decrease in total nitrogen levels.

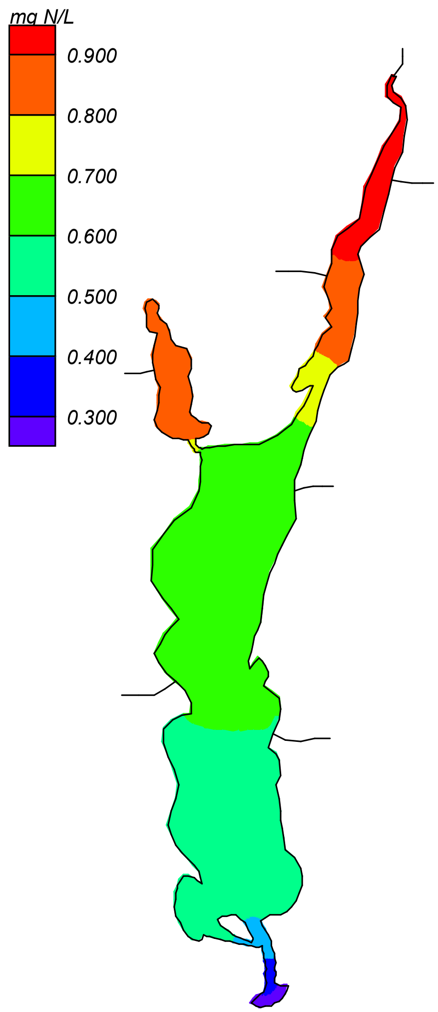

Figure IV-15. Time-averaged model results of total nitrogen concentration (mg/L) in Great Pond for existing pre-dredge conditions.

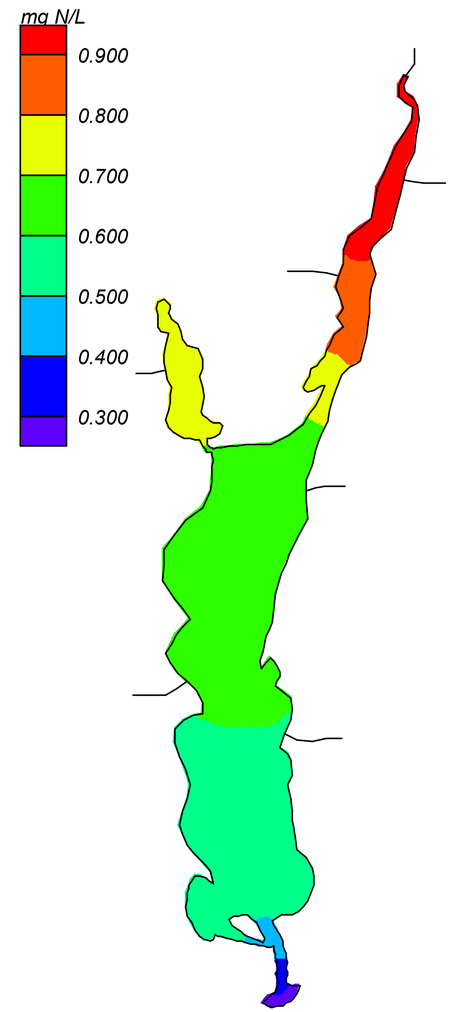

Figure IV-16. Time-averaged model results of total nitrogen concentration (mg/L) in Great Pond for projected post-dredge conditions

V. CONCLUSIONS

Overview: This Report is part of Phase I of the Massachusetts Estuary Project. Before implementing a specific approach to support nitrogen management, it is necessary to evaluate current watershed and embayment nitrogen management models as to their accuracy, data needs, comparability, and applicability across embayment types. At present, three approaches have been relatively widely applied: Buzzards Bay Project Nitrogen Loading Methodology (BBP); Cape Cod Commission Nitrogen Loading/Critical Loads Methodology (CCC); Linked Watershed-Embayment Modeling Approach (Linked).

The nitrogen management model evaluation and analysis as detailed in the report includes descriptions of:

- major nitrogen modeling approaches in use for watershed management,
- case study embayments where these approaches were compared, and
- the construct of the sensitivity analysis for the Linked Watershed-Embayment Model.

The comparative application of the various models to the case study embayments also provides an analysis of the consistency of model results between systems. This latter point is critical in evaluating a model for use by the Massachusetts Estuary Project, which will cover all 89 embayments of Southeastern Massachusetts. Given the specific regional nature of the project (all embayments in Massachusetts from Duxbury to Mt. Hope Bay, including Cape Cod, Nantucket, and Martha's Vineyard), the evaluation and selection of an appropriate model must focus on its utility in these specific systems. The models, in this evaluation are directly applicable to shallow (generally <5m), primarily vertically mixed (only supporting periodic short term stratification), enclosed or semi-enclosed embayments, surrounded by permeable watersheds with significant groundwater discharges. The approach can also be used in Mt. Hope Bay, a deeper estuary which supports periodic strong salinity stratification. However, this requires additional parameterization and complexity of the underlying hydrodynamic model component, as well as development of a separate system-specific uncertainty analysis. Therefore, results of the various model evaluations, presented in Section III below, are directly applicable to 88 of the 89 embayments within the project area.

The overall assessment of the management models included:
- comparison of watershed nitrogen loading results from each model and resultant embayment nitrogen distribution based upon the Linked Model;
- evaluation of predicted critical nitrogen loading thresholds (BBP) relative to resultant embayment nitrogen distribution based upon the Linked Model;
- sensitivity analysis for the Linked Watershed-Embayment Approach.

To address the utility of the Linked Watershed-Embayment Model for embayment management, we developed four additional examples of specific management scenarios related to the case studies. These management alternatives included the determination of estuarine nitrogen levels resulting from: removal of Title 5 septic loads, increased watershed nitrogen loading at build-out, and modifications of tidal inlets (e.g. improvements to tidal flushing).

Conclusions Based Upon Model Comparisons and Sensitivity Analysis: The results of the comparative analysis of the models, the sensitivity analysis of the Linked Model, and the management application Case Studies are summarized below. The overall conclusion of the evaluation was that the Linked Watershed-Embayment Model provides the best available model for use in the 89 embayments within the Massachusetts Estuary Project. The Linked model outperformed the various other management models in predicting estuarine nitrogen levels, was the only model to include a quantitative and validated embayment component, and was sufficient robust relative to the watershed model component. In addition, the Linked Model Approach provides for independent calibration and validation at each level of assessment, thus increasing the certainty of the results and the confidence needed to guide management.

1. Watershed-embayment nitrogen management requires an approach that can accurately portray nitrogen levels within receiving waters and relate them to habitat quality. The approach needs to be holistic and allow evaluation of the effects of spatially altering nitrogen loads, determine the effects associated with changes in key determinants (e.g., tidal exchange, source waters, freshwater flows), and allow evaluations of all spatial scales of the embayment (tributary, upper, lower, coves, etc.). The Linked Watershed-Embayment Model is consistent with these management needs.

2. Of the Models evaluated, only the Linked Model provides output as to the nitrogen distribution throughout an embayment resulting from determined watershed load. In general construct, the Linked Model uses a watershed land-use loading approach similar to the BBP and CCC models, but also is coupled to a numerical hydrodynamic/water quality model which encompasses the circulation and dispersion of nitrogen within the receiving waters. This linkage of watershed and embayment not only provides for assessment of specific areas within embayments, but also allows for calibration and validation approaches not open to the other models. The BBP and CCC Models typically distribute bulk nitrogen loads to sub-embayments or to entire embayment systems, since they do not have a spatially dependent embayment component.

3. The Linked Watershed-Embayment Nitrogen Management Model requires additional data, not needed by the BBP and CCC Models, to support its embayment component that includes both modeling and ecological parameters. However, the BBP and CCC do require measurement of embayment flushing rates for determination of critical nitrogen loads.

4. In almost all cases, the standard nitrogen loading terms are consistent among the BBP, CCC, and Linked Models. This is not surprising, since they are based upon the same studies and base data. However, the septic loading term is about 25% lower in the Linked Model than the BBP or CCC Models. This results from the use of Title 5 flows for the BBP and CCC methodologies (with 35 mgN/L) while the Linked Model is based upon regional septic system discharge and transport studies. In addition, while all methods correct for occupancy, this is deemed a major error in some applications that have not properly evaluated occupancy rates in seasonal communities. Errors in occupancy create a proportional error in residential wastewater loading. Although the error in final nitrogen load due to incorrect occupancy data is “project specific” (and cannot be evaluated here), the resulting error that it generates in the final watershed loading is relatively significant, due to the preponderance of on-site wastewater in most Southeastern Massachusetts coastal watersheds.

5. In contrast to the land-use nitrogen loading terms, there is not consistency among the various methodologies as to the extent of nitrogen attenuation within the watershed as nitrogen moves via groundwater or surface water from the source to the receiving waters. The Linked Model includes attenuation, based upon site-specific measurements. The BBP Model did not use attenuation prior to 2000 and now uses a

generic attenuation of 30% for surface and groundwater transport (>1 km). The CCC Model does not include attenuation during transport.

6. The similarity in construct of the BBP and CCC watershed nitrogen loading models and the watershed portion of the Linked Model suggests that previous watershed loading databases might be easily modified to support the Linked Watershed-Embayment Approach.

7. The overall calibration process for the hydrodynamic modeling generally produces errors in tidal elevation and phase of less than 3%. For the more complex embayments, current measurements provide additional model validation data. A comparison of the measured and computed volume flow rates at the Stage Harbor Inlet based upon the hydrodynamic component of the Linked Model showed remarkably good agreement. The calibrated model accurately describes both the general conditions and the irregularities of the discharge through the inlet.

8. Water quality models of estuaries are typically calibrated using salinity data, though the ultimate purpose of the model is to model total nitrogen concentrations. Since salinity and total nitrogen are dissolved constituents, they both will exhibit similar dispersion characteristics. Salinity measurements are commonly used to determine the dispersion coefficients of estuaries (e.g., as in the general method and examples provided by Fischer, et al, 1979). This is a valid assumption because the modeled systems do not have strong gradients in salinity or nitrogen concentrations, which makes turbulent mixing the dominant dispersive phenomenon in the modeled estuaries. Therefore, dispersion coefficients determined for salinity are appropriate for total nitrogen.

9. The Linked Model Approach (standard protocol with attenuation) was able to predict observed embayment nitrogen levels with percent errors less than 10% in 13 of 15 cases. Similarly, for Great and Green Ponds, the BBP and CCC also yielded good fits to the measured nitrogen levels with percent errors generally less than 10%, but had difficulties in Bournes Pond, likely a result of not accounting for benthic regeneration. The Waquoit Model yielded an exceedingly poor fit to observed nitrogen levels. The Waquoit Model underestimated nitrogen levels by an average of 35% (range: 27%-57%). Based upon these results the Waquoit Model is not recommended for use by the Massachusetts Estuary Project. Based upon the ability to predict the actual nitrogen levels in a consistent fashion across all of the embayment the Models rank as follows (best to worst): Linked>BBP>CCC>>>Waquoit. The fit appears to be improved if benthic regeneration is added to the BBP and CCC Models.

10. The BBP Critical Nitrogen Loads were found to vary in near direct proportion with alteration of the residence time (r) employed (0-10 days). In addition, since the upper third of an estuary has no volumetric or functional significance, the focus on its flushing rate may not always yield protective or meaningful results. More significant is that almost all embayments are sub-embayments to a larger bay and some embayments have multiple upper sub-embayments. This approach is open to manipulation of the critical loading limit through selection of flushing rate (e.g. use of the flushing rate for a sub-embayment versus the flushing rate for the system).

11. The critical nitrogen limits based upon the BBP Approach do not consistently approximate measured habitat health conditions. Generally, the generated limits tend to over-estimate the loads which a system can tolerate. It is likely that the poor fit is due to

the non-inclusion of benthic regeneration and the lack of the nitrogen load spatial distribution along the estuary.

12. The overall result of the sensitivity analysis is that the Linked Model predictions of embayment nitrogen level and distribution are relatively robust. The Model is most sensitive to (in the following order of most to least sensitive): watercolumn dispersion > source water nitrogen concentration > benthic regeneration, septic load > attenuation, fertilizer, impermeable surfaces. The effect of varying the watershed nitrogen loading or attenuation terms was largest in the upper reaches of the embayment and diminished toward the inlet. The effect is seen both as a reduction in the percent change and the nitrogen concentration change. Dispersion was also most sensitive to upper estuary processes. This pattern is due to the increasing dominance of inflowing tidal source waters near the inlet versus the dominance of watershed processes in the upper reaches of embayments. This latter effect is demonstrated by the results of varying the source water concentration, which results in large (20%) changes in nitrogen levels near the inlet and diminishing effects in the upper estuary. Benthic regeneration tended to show the largest changes at mid-estuary.

13. Once the Linked Model has been calibrated and validated to existing estuarine conditions, it provides a powerful management tool to evaluate various nitrogen loading scenarios. Example case studies indicate the expected nitrogen concentration changes, as well as the associated shifts in ecological health, for alterations to septic loading in Great, Green, and Bournes Ponds. The Linked Model can be run under user selected nitrogen management scenarios, to evaluate the most cost effective watershed management alternative for estuarine protection/restoration. These “what if scenarios” play a central role in both local decision making and the larger TMDL process. In addition, projected nitrogen concentration shifts associated with modifications to inlet channels can be used to assess potential impacts resulting from either dredging or structural modifications (e.g. jetty configuration or culvert redesign) to an estuary. This latter model application supports engineering design and feasibility analysis for both embayment and wetland restoration.

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