Massachusetts Estuary Project (MEP) Linked Watershed Embayment Model Peer Review

Scientific Peer Review Panel Report

Prepared for:
Barnstable County, Massachusetts
Cape Cod Water Protection Collaborative

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1. PEER REVIEW PANEL MEMBERS

Estuarine Water Quality Modeling

Victor J. Bierman, Jr., Ph.D., BCEEM (Chair)

Dr. Bierman is currently a Senior Scientist at LimnoTech, an environmental science and engineering consulting company with headquarters in Ann Arbor, Michigan. He conducts research and development on projects for federal, state and regional clients, and provides scientific peer review, litigation support and expert testimony on a variety of environmental issues. Dr. Bierman has 38 years of experience in the development and application of water quality models. He is a former U.S. EPA National Expert in Environmental Exposure Assessment and a former Associate Professor in the Department of Civil Engineering at the University of Notre Dame. Dr. Bierman holds an A.B. in Science from Villanova University, and an M.S. in Physics and a Ph.D. in Environmental Engineering from the University of Notre Dame, and is a Board Certified Environmental Engineering Member of the American Academy of Environmental Engineers.

Dr. Bierman has developed and applied water quality models for the Great Lakes to support phosphorus management strategies, for the “dead zone” in the Gulf of Mexico to support the Gulf Hypoxia Action Plan for nitrogen and phosphorus loads, and for nitrogen TMDLs in the St. Johns and Caloosahatchee River Estuaries in Florida. He is a member of the EPA/USACE Chesapeake Bay Modeling Team for development of an advanced eutrophication model for multiple algal functional groups in the Potomac River Estuary. Dr. Bierman also served as an expert consultant to the U.S. EPA Science Advisory Board for peer review of technical guidance on development of numeric nutrient criteria for the protection of aquatic life. He is currently serving on an Independent External Peer Review Panel for the St. Johns Bayou and New Madrid Floodway (MO) Project to review a National Environmental Policy Act (NEPA) document prepared by the U.S. Army Corps of Engineers.

Groundwater Modeling

Peter Shanahan, Ph.D., P.E.

Peter Shanahan is a Senior Lecturer in the Department of Civil and Environmental Engineering at MIT where he teaches classes and supervises graduate students in environmental engineering. He is also the President of HydroAnalysis, Inc., a three-person consulting practice in Acton, Massachusetts that specializes in hydrology, water quality, and computer modeling. He holds academic degrees from MIT and Stanford University in environmental engineering and earth sciences, including a PhD in environmental engineering from MIT. He is also a Professional Engineer in Massachusetts.

Dr. Shanahan has practiced in the areas of hydrology and water quality for 37 years. His professional practice in groundwater hydrology has included both clean ground water in the context of groundwater supply development, and contaminated ground water in the context of investigation and remedial design for hazardous waste sites. His consulting work on Cape Cod has included a study of the effects of the MMR sewage plume on Ashumet Pond on behalf of a neighborhood citizen’s group. His work at MIT has included the supervision of numerous Master of Engineering theses that examined various aspects of groundwater contamination at the MMR Superfund Site.

Soils Science & Nitrogen Transport

Lawrence E. Band, Ph.D.

Dr. Band is the Voit Gilmore Distinguished Professor of Geography and the Director of the Institute for the Environment at the University of North Carolina, Chapel Hill. His research is in the ecohydrology of watersheds, including the cycling of water, carbon and nutrients, the development and impacts of droughts and floods, and human/environment interactions. Dr. Band's current research focuses in two National Science Foundation funded Long Term Ecological Research (LTER) sites: the Coweeta LTER in western North Carolina, and the Baltimore Ecosystem Study, and the NSF funded Triangle ULTRA-EX (Urban Long Term Research Area - Exploratory) project in the NC Triangle. Past research has included projects in the Pacific Northwest, Rocky Mountains, China, Canada and Australia. In 2010 he was Board Chair for the Consortium of Universities for the Advancement of Hydrologic Sciences, a consortium of ~130 US and foreign universities, non-profit institutes, and domestic and foreign water science and management agencies.

Dr. Band has published more than 120 papers, book chapters and technical reports and has consulted with the EPA Chesapeake Bay Program, the South Florida Water Management District and State of California on watershed protection, stormwater and ecosystem restoration. He was a committee member on two recent National Academy of Science (NAS) panels on stormwater management (NRC, 2008a) and on integrated hydrological/biogeochemical measurement (NRC, 2008b) and is currently a member of a (NAS) committee on land use change impacts. Dr. Band also participated in two reviews of the Chesapeake Bay Watershed Model (Band et al. 2005, 2008).

Estuarine Hydrodynamic Modeling

Billy H. Johnson, Ph.D., P.E., D.WRE

Billy H. Johnson retired as a research hydraulic engineer in 2001 from the US Army Corps of Engineers Engineering Research and Development Center (ERDC) located in Vicksburg, MS. He is currently the Managing Partner of a small numerical modeling and engineering consulting firm known as Computational Hydraulics and Transport.

He obtained a PhD in engineering from Mississippi State University and is a registered Professional Engineer in Mississippi. He is also a Diplomate in the Institute of Water Resources Engineering in the American Society of Civil Engineers. He was the ASCE Hydraulic Engineer of the Year in Mississippi in 1990 and was inducted into the ERDC Gallery of Distinguished Employees in 2005.

He has 40 years of experience in developing and applying 1D, 2D, and 3D numerical hydrodynamic models. Areas in which he has been involved in modeling studies include Chesapeake Bay, Delaware Bay, Appalachicola Bay, Atchafalaya Bay, etc. He led the development of the 3D Chesapeake Bay hydrodynamic model, which is still employed in TMDL studies in the Bay. He introduced the concept of non-orthogonal boundary fitted coordinates in the area of numerical hydrodynamic modeling. Several researchers, including Michael Spaulding and Peter Sheng have built upon that concept in the development of 3D hydrodynamic models. He has served on several technical review panels, including Florida Bay, Indian River, and several minimum flow studies on rivers in the Southwest Florida Water Management District.

Estuarine Biology

W. Judson Kenworthy, Ph.D.

Dr. Kenwothy holds a BSc from the University of Rhode Island, a M.S. in Environmental Sciences from the University of Virginia and a PhD in Zoology at N.C. State University.

Dr. Kenworthy is recently retired from the Center for Coastal Fisheries and Habitat Research, NCCOS, NOS, NOAA after 33 years of federal service. Currently Dr. Kenworthy is working as a sub-contractor with NOAA's Submerged Aquatic Vegetation Technical Working Group assessing the impacts of the Gulf of Mexico oil spill on seagrass ecosystems. As a student and NOAA research scientist Dr. Kenworthy has over 35 years of experience in coastal ecology with particular emphasis on seagrasses and the effects of natural and anthropogenic disturbance on coastal environments. Dr. Kenworthy's areas of expertise in applied science include research on water quality impacts on seagrasses, seagrass restoration, designing and implementing environmental assessments and resource monitoring programs and assisting State, Federal and International Resource Management Agencies in planning and implementing conservation and restoration programs.

TMDL Policy & Regulatory Issues

Paul E. Stacey, M.S.

Paul E. Stacey received a Bachelor of Arts Degree from the College of the Holy Cross in Worcester, MA; a Bachelor of Science Degree from Utah State University; and a Master of Science Degree from Colorado State University.

He was hired by New Hampshire Fish and Game in February 2011 as the Research Coordinator for the Great Bay National Estuarine Research Reserve. From 2006--2011, he served as Director of Planning and Standards with the Connecticut Department of Environmental Protection's Bureau of Water Protection and Land Reuse where he led Connecticut's involvement in all surface water quality matters. From 1985-2006 he had served as state coordinator for the National Estuary Program Long Island Sound Study, which included development of the TMDL for Long Island Sound and Connecticut's Nitrogen Credit Exchange. His areas of expertise include nutrient dynamics and impacts on freshwater and estuarine ecosystems, climate change effects and adaptation, and multi-media and ecosystem-based approaches for management of pollutant sources from air, land and water. He was previously employed by the Academy of Natural Sciences in Philadelphia in the Applied Ecology program from 1977-1985.

2. EXECUTIVE SUMMARY

Nitrogen enrichment is a widespread problem, leading to cultural eutrophication of coastal ecosystems worldwide. Cultural eutrophication is a condition caused by excess nutrient loads from human sources, leading to increased algal growth, reduced dissolved oxygen and adverse ecological impacts. Many estuaries are identified as being “at risk” or already affected by excess nitrogen loading, which may affect as many as 89 estuaries on Cape Cod.

The Massachusetts Estuaries Project (MEP) partnership was organized to provide a technical underpinning for development of total maximum daily loads (TMDLs), especially the establishment of water quality goals, source assessments and recommendations for source reductions. Nitrogen delivery to Cape Cod estuaries from human sources is dominated by septic inputs delivered to local waters through groundwater transport. This presents a unique challenge to local stakeholders who desire to protect and restore these sensitive ecosystems for their important contribution to the local lifestyle and economy.

This scientific peer review was sponsored by the Cape Cod Water Protection Collaborative (Collaborative), an agency of Barnstable County. The purpose was to conduct an independent scientific peer review of the MEP methodology for developing appropriate TMDLs for the estuaries and embayments of Cape Cod, and the use of that methodology as a basis for wastewater and nutrient management planning and implementation on Cape Cod. This scientific peer review process was independent and objective, and operated externally from the Collaborative and from any other Cape Cod stakeholders.

The Panel finds that the MEP modeling approach is scientifically credible. It is consistent with current understanding of existing conditions for Cape Cod estuaries, based on available data. The components in the approach are well-known and documented. Computation of watershed nitrogen loads is strongly data-driven and quantitatively linked to estuarine nitrogen concentrations. A fundamental principle in the development and application of environmental models to inform management decisions is that there should be compatibility among the study questions and objectives, available data and resources, and level of model complexity. The Panel finds that the level of complexity in the components and linkages of the MEP modeling approach is simple, parsimonious and well balanced within this context.

The Panel also finds that the MEP modeling approach is functionally adequate. This approach is specifically designed for groundwater dominated systems and explicitly considers nitrogen loads from septic systems, the dominant controllable watershed source of nitrogen for Cape Cod estuaries. The MEP modeling approach is appropriate and useful for evaluating alternative scenarios and informing nutrient management plans, and is consistent with existing nationwide TMDL practices.

The Panel recommends that the MEP modeling approach be considered within the larger context of the overall decision support system and not be limited to just the linked watershed-embayment model. The Panel further recommends that an adaptive management framework be used for this decision support system, which integrates the watershed-embayment model. This integration should include continued monitoring, data analysis and modeling to improve scientific understanding and reduce uncertainties in the physical, chemical and biological processes in the watersheds and estuaries.

The Panel recommends that the towns proceed within this adaptive management framework to develop and implement wastewater and nutrient management plans, and make improvements along the way to reduce management uncertainties. This will ensure that TMDL implementation is not compromised due to a lack of information, and that progress will be made in the most cost-effective manner while gathering new information to improve upon the scientific analysis, and the initial wastewater and nutrient management plans.

3. INTRODUCTION

3.1. Background

Nitrogen enrichment is a widespread problem, leading to cultural eutrophication of coastal ecosystems worldwide. Cultural eutrophication is a condition caused by excess nutrient loads from human sources, leading to increased algal growth, reduced dissolved oxygen and adverse ecological impacts. Many estuaries are identified as being "at risk" or already affected by excess nitrogen loading, which may affect as many as 89 estuaries on Cape Cod (Massachusetts Department of Environmental Protection, 2003). The Massachusetts Estuaries Project (MEP) partnership was organized to provide a technical underpinning for development of total maximum daily loads (TMDLs), especially the establishment of water quality goals, source assessments and recommendations for source reductions. Nitrogen delivery to Cape Cod estuaries from human sources is dominated by septic inputs delivered to local waters through groundwater transport. This presents a unique challenge to local stakeholders who desire to protect and restore these sensitive ecosystems for their important contribution to the local lifestyle and economy.

3.2. Purpose

This scientific peer review was sponsored by the Cape Cod Water Protection Collaborative (Collaborative), an agency of Barnstable County. The purpose was to conduct an independent scientific peer review of the MEP methodology for developing appropriate TMDLs for the estuaries and embayments of Cape Cod, and the use of that methodology as a basis for wastewater and nutrient management planning and implementation on Cape Cod.

3.3. Charge to Panel

The scientific peer review panel (Panel) was presented with two charge questions:

1. Is the MEP modeling approach scientifically defensible and functionally adequate to be relied upon in the development and implementation of appropriate nitrogen TMDLs for the estuaries and embayments of Cape Cod in support of the state's Comprehensive Wastewater Management Planning and EPA Clean Water Act requirements and in developing overall wastewater and nutrient management plans for Cape Cod to meet the TMDLs?
2. To what level of accuracy will the MEP linked model predict the effect of alternative nitrogen load planning scenarios and/or the prospective water quality in the affected estuaries and embayments and what is the degree of uncertainty in those predictions relative to alternative planning methodologies available in the industry?

These questions provided focus and context for the review, and guided all of the work by the Panel to review relevant documents, identify key issues, conduct deliberations, and develop findings, conclusions and recommendations.

3.4. Scope

This review was conducted over a three-month period beginning in October 2011, and included a four-day technical workshop on November 13-16, 2011. Panel members reviewed MEP documents, reports from previous peer reviews, reviews and comments by the Town of Orleans Wastewater Management Validation and Design Committee (WMVDC), comments from other stakeholders, and numerous papers and reports from government agencies and the published scientific literature. The Panel was asked to focus on the Pleasant Bay and Bournes Pond systems as case studies.

The Panel relied upon published reports, presentations by the project team, led by the University of Massachusetts Dartmouth, School of Marine Science and Technology (SMAST), and a question and answer session with the team following their presentations. The Panel identified what it considered to be key technical issues and discussed each of them in this report. Many other issues had been identified in previous studies and reviews. Issues were not specifically discussed in this report if the Panel did not consider them to be of strategic concern, they were not part of the MEP approach, they were not part of the charge to the Panel, or if the Panel did not consider them to be significant.

3.5. Process

This scientific peer review process was independent and objective, and operated externally from the Collaborative and from any other Cape Cod stakeholders.

The first meeting of the Panel was a kick-off teleconference on October 10. Shortly thereafter, CH2M Hill, the coordinating organization for the review, set up a Project Insight Website. Project Insight is a web tool that allows project teams to exchange information and documents, and to collaborate during the delivery of a project. Panel members reviewed MEP documents, reports from previous peer reviews, WMVDC reviews and comments, comments from other stakeholders, and numerous papers and reports from government agencies and the published scientific literature.

The Panel produced a memorandum on October 21 that provided a preliminary summary of key technical issues and included requests for additional information from the SMAST Team. Panel members met again via teleconference on October 31 to discuss these preliminary technical issues. The SMAST Team produced a Memorandum on November 7 that provided responses to the October 21 Memorandum by the Panel. Panel members then participated in a coordination meeting with the SMAST Team via teleconference on November 9.

Panel members participated in a workshop in Sandwich, MA, November 13-16. The first day included a group tour by boat of the Pleasant Bay system. Individual panel members also visited Bournes Pond. The second day included presentations by SMAST, Applied Coastal Research and Engineering, Massachusetts Department of Environmental Protection (MDEP), the Comprehensive Wastewater Management Plan (CWMP) and the U.S. Geological Survey (USGS). These presentations were followed by a question and answer session between the Panel and all of the presenters. The Panel then deliberated in closed sessions for the next day and a half.

The Panel presented its preliminary findings to the Collaborative Board of Directors at a public meeting in Hyannis, MA, on the afternoon of November 16. The presentation was followed by discussions and questions from the Board, and then questions from the public and other attendees.

This written report documents all of the findings, conclusions and recommendations of the MEP Scientific Peer Review Panel.

4. MEP MODELING APPROACH

4.1. Decision Support and Adaptive Management

The MEP modeling approach embodies a watershed-embayment model that links watershed inputs with water circulation and nitrogen characteristics in the estuaries and bays. The Panel recommends that the MEP modeling approach be considered within the larger context of the overall decision support system (DSS) and not be limited just to the linked watershed-embayment model. The Panel further recommends that an adaptive management framework (Figure 1) be used for the DSS, which integrates the MEP watershed-embayment model. This model serves important problem assessment, design and implementation components of the DSS, but it only links nitrogen loads to nitrogen concentrations in the estuaries. The development of threshold nitrogen concentration targets is based on observed data that link nitrogen concentrations to water quality and ecological responses. In turn, to develop TMDLs, which drive the implementation component of the DSS, these nitrogen concentration targets are linked to nitrogen concentrations computed by the watershed-embayment model, and then linked back to nitrogen mass loads to the estuaries. To complete the adaptive management framework as a DSS, implementation must be followed by an integrated monitoring, evaluation (supported by the MEP model) and adjustment process to improve understanding and reduce uncertainties in the overall DSS cycle.

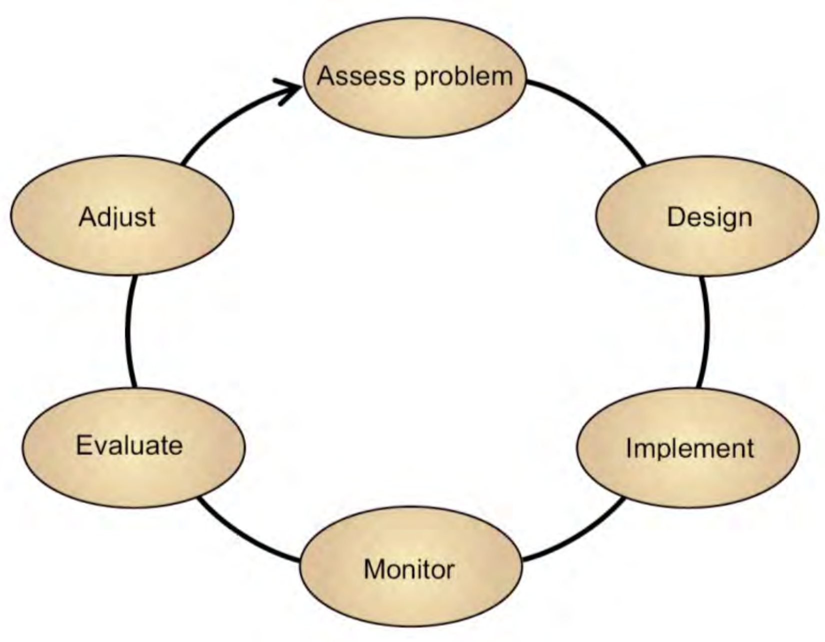

Figure 1. Adaptive Management Process in a Decision Support System Framework (Williams et al. 2009)

4.2. Components and Linkages

The role of the MEP modeling approach within the overall DSS involves two distinctly different types of components, those that are model-based and those that are data-based (Figure 2). Model-based components (colored blue in Figure 2) relate computed nitrogen loads from the watershed to computed nitrogen concentrations in the estuaries. The watershed loading model is not a single entity but is formed by the integration of loads from septic, fertilizer, stormwater and nonpoint sources with flow pathways through groundwater, ponds, wetlands and streams to the estuaries. Data-based components (colored green in Figure 2) relate observed water quality and ecological responses to observed nitrogen concentrations in the estuaries to derive threshold nitrogen concentration targets. Figure 2 embodies the Assess, Design and Implement steps in the DSS illustrated in Figure 1.

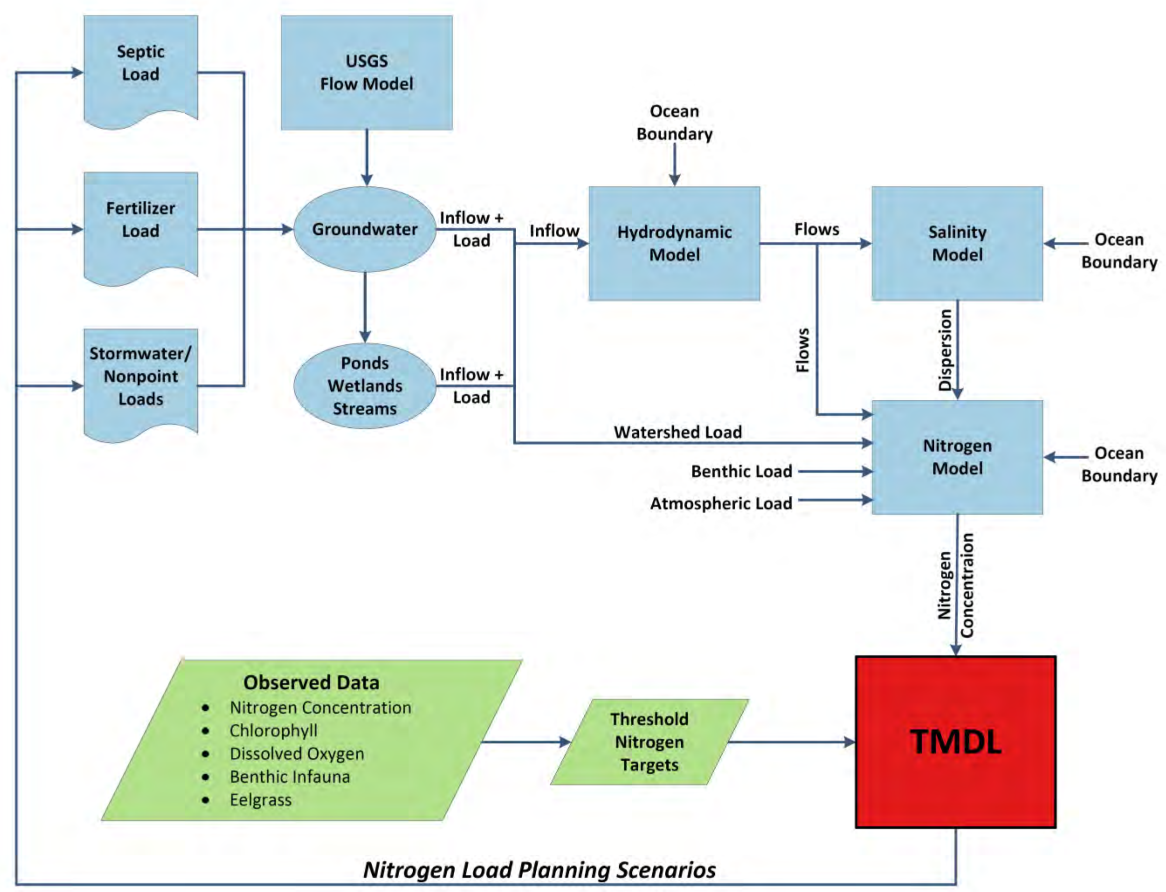

Figure 2. MEP Modeling Approach within the Overall Decision Support System

The USGS groundwater flow model is used to compute subsurface inflow discharge within distinct subsurface flowpath patterns which drain the watershed directly to the estuaries or indirectly through ponds, streams and marshes. Surface water inflows through a subset of streams are also determined using observed flow measurements.

The watershed model is used to compute nitrogen loads from septic sources, fertilizers, stormwater and nonpoint sources into the flowpath system developed by the groundwater model. Given the high infiltration capacities of soils and depression-dominated glacial topography on Cape Cod, all watershed sources were assumed to enter the groundwater system, with negligible direct surface water delivery to ponds, streams or estuaries. The inputs to these watershed models are the following:

- Septic
  - Parcel-level water use
  - Applied nitrogen concentrations
- Fertilizer
  - Application rates and lawn size
- Stormwater and nonpoint sources
  - Nitrogen load factors based on event mean concentrations

The hydrodynamic models compute flow velocities and tidal heights in the estuaries. The inputs to these models are inflows from the watershed, bathymetry and ocean boundary conditions for tidal heights.

The salinity models compute dispersion coefficients in the estuaries. The inputs to these models are the flows and tidal heights from the hydrodynamic models, and ocean boundary concentrations for salinity.

The nitrogen water quality models compute nitrogen concentrations in the estuaries. The inputs to these models are the flows and tidal heights from the hydrodynamic models, dispersion coefficients from the salinity models, nitrogen mass loads from the watershed, atmosphere and benthic regeneration, and ocean boundary conditions for nitrogen concentration.

Threshold nitrogen concentration targets are developed by MEP for each specific estuary solely on the basis of observed monitoring data. These data are used to link nitrogen concentrations in the estuaries to water quality (chlorophyll and dissolved oxygen) and ecological (benthic communities and eelgrass) responses. This is considered a “reference” approach for developing site-specific numeric nutrient criteria (Howes et al. 2003, U.S. EPA 2001).

Finally, to conduct nitrogen load planning scenarios and develop TMDLs, these nitrogen concentration targets are linked to nitrogen concentrations computed by the nitrogen water quality model, and then back to nitrogen mass loads computed by the watershed model (Figure 2). However, because of uncertainties incumbent in management and response of nitrogen-enriched estuaries, and consistent with the recommended adaptive management framework, the DSS should continue to inform and reinforce the process beyond TMDL development. Continued monitoring, evaluation and adjustment of the modeling tools (Figure 1), as well as the management actions, are essential to cost effective outcomes.

4.3. Scientific Credibility and Functional Adequacy

The Panel finds that the MEP modeling approach is scientifically credible. It is consistent with current understanding of existing conditions for Cape Cod estuaries, based on available data. The components in the approach are well-known and documented. Computation of watershed nitrogen loads is strongly data-driven and quantitatively linked to estuarine nitrogen concentrations. A fundamental principle in the development and application of environmental models to inform management decisions is that there should be compatibility among the study questions and objectives, available data and resources, and level of model complexity. The Panel finds that the level of complexity in the components and linkages of the MEP modeling approach is simple, parsimonious and well balanced within this context.

The Panel also finds that the MEP modeling approach is functionally adequate. This approach is specifically designed for groundwater dominated systems and explicitly considers nitrogen loads from septic systems, the dominant controllable watershed source of nitrogen for Cape Cod estuaries. The MEP modeling approach is appropriate and useful, within the overall DSS, for evaluating alternative scenarios and informing nutrient management plans, and is consistent with existing nationwide TMDL practices.

5. KEY ISSUES IDENTIFIED BY PANEL

5.1. Introduction

The Panel identified key issues with the MEP modeling approach, and its application to the Pleasant Bay and Bournes Pond case studies, and organized these issues by topic area. Presentation of each key issue includes discussion, conclusions and recommendations.

Identification of these key issues by the Panel does not imply that the MEP modeling approach is flawed and in need of repairs. Rather, the opinion of the Panel is that the MEP modeling approach is scientifically defensible and functionally adequate, and can be improved going forward within the recommended adaptive management framework. This is a normal process for management of complex water resource systems like the estuaries of Cape Cod. The towns should proceed within this framework to develop and implement wastewater and nutrient management plans, and make improvements along the way to increase scientific understanding and reduce management uncertainties.

5.2. Model-Based Issues

5.2.1. Groundwater and Watershed Nitrogen Loads

Key Issue 1 - Groundwater Modeling

Cape Cod differs from most other watersheds in that drainage of the watershed to receiving water bodies occurs largely via groundwater rather than via surface water flow over the land surface. Groundwater discharge to the estuaries occurs either indirectly through base flow in small streams and rivers, or directly through seepage at the shoreline, in the intertidal zone and more limited submarine discharge. In most surface-water-drained watersheds, delineation of the watershed boundary is simply a matter of identifying the ridge lines that separate watersheds from topographic maps, and delineating the watershed by connecting ridge lines appropriately, or automatically from digital elevation models. This is not possible for Cape Cod. Thus, in order to delineate watersheds, the SMAST Team depended upon maps of aquifer areas contributing groundwater flow to the various bays and estuaries. These maps were developed using groundwater flow models developed by the USGS based on the widely used USGS MODFLOW computer code. The USGS has been working with and refining their groundwater models of Cape Cod for decades, the work is very well done, and the personnel highly qualified. The Panel is therefore confident that this aspect of the MEP studies has been done using appropriate tools and with sufficient accuracy.

Key Issue 2 – Nitrogen Load Estimation

A key aspect of the MEP modeling approach is the estimation of nitrogen loads generated within the watershed and the subsequent partial attenuation of those loads during transport to the receiving bays and estuaries. This requires consideration of all significant sources of nitrogen inputs to the contributing watershed and directly to the estuaries. For all sources delivered to groundwater and surface water, the nitrogen concentrations and the volumes (effluent volumes, effective precipitation) need to be estimated. Major sources are currently atmospheric deposition, stormwater, fertilizer for agriculture, urban lawns and recreational areas; and, wastewater loading from on-site disposal (septic systems) and outfalls from sewage treatment plants. Controllable loads include wastewater, fertilizer and stormwater, while atmospheric loads cannot be controlled at the local level. Cape Cod residences, and commercial, institutional and industrial parcels dominantly use on-site disposal, although some areas have installed sanitary sewers and central treatment. The geology, sanitary infrastructure, parcel density and topography indicate on-site septic systems are a major source of nitrogen load to the watershed. Dominance of septic systems as nitrogen sources in receiving water bodies is by no means unique to the Cape, and occurs in other areas of the country that are septic served, even with lower population densities and less conductive soils and groundwater (e.g. Kaushal et al. 2011). An excellent review of on-site septic systems risk in terms of nitrogen contamination is given by Gold and Sims (2000), including systems on glacial outwash systems similar to Cape Cod. Specific research on seasonally used septic systems in similar geologic settings in Rhode Island has shown specific vulnerability to nitrogen leaching (Postma et al. 1992) due to incomplete formation of biological clogging mats.

Estimating loading from individual and institutional activity at the parcel level is necessarily imprecise as details of every residential and non-residential unit cannot be measured, and can only be inferred using limited measurements or surveys and statistical methods. However, the methods used by the SMAST Team are reasonable and consistent with approaches used by others working locally, including the MDEP (Horsley & Witten, 1996), the Cape Cod Commission (Eichner and Cambareri, 1992), and the Woods Hole Marine Biological Laboratory (Valiela et al. 1997a and Latimer and Charpentier, 2010) as well as other areas of the country (e.g. Law et al. 2004, Osmond et al. 2004, Groffman et al. 2004). Importantly, rather than relying on literature values for key information, such as the nitrogen concentration of septic effluent, the volume of wastewater per parcel or per capita, and lawn fertilization rates, care has been taken to use local measurements and inference using data collected in the area:

1. Septic effluent nitrogen sources: Detailed Geographic Information System (GIS) analysis of parcel-scale water use was used to estimate total septic and stormwater input, and mapped into detailed groundwater flowpaths derived from USGS MODFLOW simulations. Septic wastewater volume is typically difficult to reliably estimate as water supply for septic-served parcels often comes from individual groundwater wells. However, the Cape is somewhat unique with water supply provided by local utilities, allowing parcel-scale billing to be used to estimate septic recharge, after allowing for outdoor water and other consumptive use. This obviates some of the problems of estimating occupancy and per capita water use to estimate effluent volumes as water use is directly known. Nitrogen concentrations of wastewater varies with water use patterns, and local measurements have been used to constrain these estimates.
2. Lawn fertilizer sources: Parcel scale surveys (households, golf courses, etc.) of fertilizer use estimated fertilization rates similar to those estimated elsewhere in the eastern US (e.g. Law et al. 2004, Osmond et al. 2004). While these estimates do not have the spatial precision that septic volume estimates have (which have parcel-scale billing information as discussed above), it provides reasonable estimates of total lawn fertilizer nitrogen applied within groundwater catchment areas. Estimates of lawn nitrogen attenuation and leaching rates are inferred from measurements in similar soils (Cape Cod and Long Island) and remain an additional source of uncertainty.
3. Stormwater sources: Given the high infiltration capacities of Cape Cod glacial moraine and outwash derived soils, the flat topography with numerous small wetlands and depressions, and road construction patterns, stormwater largely enters groundwater systems through run-on infiltration (impervious runoff routing to pervious areas) or into local wetlands. Local high density commercial development may provide limited areas of higher stormwater input, but these comprise small areas of the Cape. Detailed GIS analysis of groundwater catchment area land covers,

including stormwater generation from transportation (roads, parking lots), roof and other runoff-generating areas was completed to estimate stormwater volumes, with concentrations estimated from standard sources.

While individual parcel, road and other land cover estimates of nitrogen input to the watershed and recharge to groundwater will be uncertain, averaging over the number of parcels and total impervious areas in each groundwater catchment area provides the total and mean application and recharge rates required for this analysis. These means and totals are more stable and have less uncertainty than individual parcels (based on the law of large numbers) and are the required information for loading magnitude and delivery patterns to the estuaries. Moreover, independent measurements of groundwater loads by Kroeger et al. (1999, 2006) and Valiela et al. (2000) for Green Pond approximately match the MEP estimates, building confidence that the MEP results are reasonable.

Key Issue 3 – Uncertainty and Sensitivity Analysis

Sensitivity analysis evaluates the effects of changes in input values, parameter estimates, assumptions and algorithms on a model's results. Uncertainty analysis investigates the effects of lack of knowledge and other potential sources of error in the model, and when conducted in combination with sensitivity analysis, allows a model user to be adequately informed about the confidence that can be placed in the model results. U.S. EPA (2009) recommends sensitivity analysis as the principal evaluation tool for characterizing the most and least important sources of uncertainty in environmental models.

There are numerous and varied inputs to the load estimates as discussed above (e.g. septic nitrogen concentrations, fertilizer leaching rates, groundwater recharge rates, etc.). Further, some important contributing processes are captured with only one- or two-digit accuracy—for example, 50% of the nitrogen transported through each of the major freshwater ponds is assumed to be attenuated. These low-accuracy inputs imply that nitrogen load estimates with five-digit accuracy, as included in the MEP reports, are impossible. The uncertainty in these inputs also implies that there are margins of uncertainty around the resulting load estimates. While the Panel does not recommend any wholesale changes in the approach to estimating nitrogen loads to the bays and estuaries at this stage, we recommend changes in the presentation of MEP results so as to acknowledge explicitly that there is uncertainty in the load estimates and to provide some estimate of the degree of uncertainty.

The Panel further recommends that model sensitivity analyses be conducted for nitrogen mass loads for each specific estuary. A healthy recognition that there is uncertainty would encourage planning bodies to pursue an adaptive monitoring and management strategy as they move forward to understand and remedy the impacts of nitrogen on bays and estuaries. Such an adaptive strategy is wise in light of the uncertainties in predicting the response of bays and estuaries to future load reductions.

Key Issue 4 – Nitrogen Attenuation in Groundwater

The SMAST Team assumes there is no attenuation of nitrogen in groundwater, which they appropriately indicate to be a conservative approach (Howes et al. 2002). While this is a reasonable and conservative assumption, there is ample information in the technical literature to show that nitrate (NO_3) from septic systems is almost certainly converted to nitrogen gas (N_2) by bacterially-mediated denitrification in the subsurface, but the extent varies widely. Denitrification occurs by a microbially mediated stepwise chemical reaction as follows:

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$$

Anaerobic conditions and a substrate to support microbial populations (typically carbon) are needed for denitrification and most denitrification occurs in the vadose zone in the near vicinity of septic system leaching fields and subsurface wastewater discharges. In saturated groundwater, the extent of denitrification is highly variable and depends primarily on the availability of dissolved organic carbon (DOC) (Pabich et al. 2001). Kinetic models for denitrification indicate that the reaction proceeds quickly while DOC is present but essentially stops once DOC concentrations drop to low levels. Thus, whatever DOC is introduced at wastewater sources is rapidly consumed near the source. Further from the source there is little remaining DOC and denitrification is typically minimal. Recent work suggests iron from pyrite weathering or other materials may substitute in the absence of organic carbon, but this is not thought to be significant in this area. In some localities, more DOC may be added from natural sources, particularly where a thin vadose zone (shallow water table) allows more transport of DOC from shallow soils to the ground water or potentially from buried organic horizons (e.g. Hill, 2011). The Panel's rough estimate from the literature is that if nitrogen attenuation were incorporated into the analysis it would reduce the groundwater loads on the order of only a few percent. Given the other uncertainties in the nitrogen loads generally, and the variability in denitrification, this potential error is relatively minor and would probably be best addressed in sensitivity analyses recommended above.

The fact that DOC limits denitrification suggests some possibilities for local in-situ removal of nitrate via permeable reactive barriers or DOC injection of some sort. Several researchers have tested these approaches and they may constitute a cost-effective alternative to sewering. For example, trenches filled with sawdust have been found to provide DOC and enhance denitrification (e.g. Schipper et al. 2010).

Finally, one fact is clear: natural groundwater denitrification is not the solution to the Cape's nitrate problem. Although inclusion or exclusion of denitrification from the mass balance adds some uncertainty to the exact loads, discharge of nitrogen to the estuaries via groundwater is a significant load whether or not there is an accounting for denitrification in groundwater.

As mentioned above, significant attenuation (50%) of nitrogen load is assumed to take place in flowpaths passing through freshwater ponds, streams and wetlands, as mapped by the MODFLOW simulation. Estimates of attenuation were based on limited measurements of stream nitrogen concentrations and flows to estimate annual loads, compared to the linked watershed nitrogen loading and groundwater delivery models, as well as surveys of pond physical and hydrodynamic conditions. These losses can be considered to occur at the interface between groundwater and surface water systems, and represent an important transition where water is passing through organic material, which can act as substrate for microbial denitrification or immobilization. Plant uptake and burial, or incorporation into long term storage, are additional attenuation processes that may be active in these regions. However, no details of specific processes are incorporated into these estimates of attenuation, which are determined by simple mass balance estimates. This is consistent with the level of process specification in other model components and should be treated as an additional source of uncertainty.

5.2.2. Estuarine Hydrodynamics and Water Quality

As noted above in Section 4.2, the MEP modeling approach includes the application of a numerical groundwater model, a numerical hydrodynamic model and a numerical water quality model. Once these models are corroborated for a particular water body (e.g. Pleasant Bay), they offer highly effective tools for supporting the determination of TMDLs. With this approach, many scenarios can be modeled to assess the impact of different levels of nitrogen loading and/or physical changes on water column concentrations of nitrogen.

The particular numerical models used by the SMAST Team for the computation of hydrodynamics and water quality of the Cape Cod estuaries are known as RMA2 (Donnell et al. 2011) and RMA4 (Letter et al. 2011), respectively. The RMA2 hydrodynamic model serves as the hydraulic "chassis" for the RMA4 water quality model. These two models share the same spatial segmentation grid and numerical solution method. Various issues raised by the Panel concerning the numerical hydrodynamic and water quality models are discussed below.

5.2.2.1. Estuarine Hydrodynamic Model

Key Issue 1 - Are the RMA2 and RMA4 Numerical Models Appropriate/Adequate for Computing TMDLs for Cape Cod Estuaries and Bays?

In the analysis of the adequacy of these models there are four sub-issues; namely, *1. Is the vertically-averaged assumption appropriate? 2. Are the numerical grids adequate? 3. Is mass conserved? 4. Are computational times excessive?*

Is the vertically-averaged assumption appropriate? There are numerous fully three-dimensional (3D) models that could have been selected for application by the SMAST Team. The RMA2 and RMA4 models are based on the assumption that vertical variations in the salinity and flow fields are negligible. The SMAST Team has presented data demonstrating that this assumption is valid for all but one (i.e. Pleasant Bay) of the estuaries and bays for which TMDLs will be developed. There are some kettle ponds in Pleasant Bay where vertical stratification can occur at times. However, these ponds generally have a layer of heavy saline water very near the bottom that perhaps acts as a barrier to the exchange of nitrogen from the bottom sediments with the water column but do not significantly result in the generation of residual currents due to gravitational circulation.

A full 3D hydrodynamic and mass transport model would theoretically give more accurate computations of flow and mass concentration fields. However, the Panel concludes that the two-dimensional (2D) approach is adequate and more preferable because it is more cost-effective for scenario analyses of nitrogen loads needed by policy and management planning.

Are the numerical grids adequate? The RMA2 and RMA4 models are based on the finite element method for solving the governing equations of motion in RMA2 and the conservation of mass equation for a constituent such as nitrogen in RMA4. The other commonly employed solution method is referred to as the finite difference method. With the finite element method, one assumes the solution and then attempts to minimize the error when the assumed solution is inserted into the governing equations. The most common form of the assumed solution is a polynomial. In the case of RMA2 and RMA4, a quadratic polynomial is assumed. With the finite difference solution method, the derivatives in the governing equations are approximated by finite differences.

There are two major differences in the two most common solution methods. The first is that finite element models generally utilize unstructured grids, whereas the finite difference method utilizes structured grids. With structured grids, there is an order to the labeling of the computational cells. In other words, each cell knows its neighbor by an (I,J,K) accounting. With an unstructured grid, a connectivity table must be constructed so that each computational element knows its neighbor elements.

A major advantage of the finite element method of solution and the resulting unstructured numerical grid is that physical features such as the geometry of the water body, interior channels, etc. can be more accurately resolved than with a structured grid representing the water body. The Panel considers this a major strength of using the RMA2 and RMA4 models.

Is mass conserved? The second major difference between the two commonly employed solution methods concerns the conservation of mass. If a staggered structured numerical grid is employed with the finite difference method such that the value of a constituent such as nitrogen is computed in the center of a computational cell and transport (water velocity) is specified on the cell boundaries, mass is absolutely conserved. With the finite element solution method, mass over the entire grid is also absolutely conserved. However, mass in the interior of the grid is not constrained to be absolutely conserved (Galland et al. 1991). When modeling water quality parameters, this is often a concern. However, the problem can be minimized through the construction of a grid that accurately resolves the bathymetry of the water body and gradients in the variable being computed, e.g., nitrogen. In the November 14, 2011 meeting with the SMAST Team, including Applied Coastal and Research Engineering, the concern about mass conservation was discussed. One way to quantitatively address this concern is to set the value of the computed water quality variable to 1.0 at every node of the grid and to attach that concentration to all inflows and loads. If mass is not conserved there will be deviations during the computations from the specified value of 1.0. The Panel recommended to the SMAST and Applied Coastal Research and Engineering modelers that this computation be made for the Pleasant Bay study. Results from that recommendation were provided to the Panel. For the 7-day simulation period the maximum error in the interior of the grid was 0.54 parts per trillion. This substantiates that the Pleasant Bay numerical grid is a very good representation of the estuary.

In the review of the SMAST approach, the Panel was instructed to focus on the Pleasant Bay and Bourne Pond studies. The numerical grid covering Bournes Pond was constructed in 1999 and visually appears to perhaps need additional grid elements. The coarseness of the existing grid is likely related to the need to minimize computing time with the computer resources that were available in 1999. Since the computation of TMDLs in the Cape Cod estuaries and bays is an ongoing process, the Panel recommends that as the process proceeds for Bournes Pond, the mass conservation check outlined above be applied to help guide additional grid refinement.

Are computational times excessive? Generally speaking, numerical models based on the finite element solution method can be computer intensive. However, the simulation periods for the Cape Cod estuaries and bays are relatively short, i.e., days rather than months or years, so the computational time is not considered a detriment. Thus, this is not considered a weakness of employing the RMA2 and RMA4 models.

Key Issue 2 - Should the RMA2 Simulations Include Sub-Tidal Events as well as Purely Tidal Conditions?

The simulations for Pleasant Bay and Bournes Pond were for 7 and 5 days, respectively. The water surface elevation boundary condition for each contains only tidal fluctuations. If sub-tidal events, such as set ups and set downs of the water surface due to the passage of fronts, were also contained in the boundary conditions, there would be increased tidal flushing. However, as noted by Mr. Paul Stacey of the Panel, in the determination of TMDLs the conservative approach is taken. Thus, simulations that only contain tidal fluctuations are the appropriate approach.

Key Issue 3 - How are Culverts in Pleasant Bay Handled in RMA2?

Applied Coastal Research and Engineering responded that the culverts at Frost Fish Creek are crushed and partly blocked. They are modeled in RMA2 through the specification of an extremely high friction coefficient. There are also culverts at Muddy Creek. In the existing model they are also represented by a high friction coefficient. However, since Muddy Creek is part of an estuarine and salt marsh restoration project, plans call for replacing the existing culverts with a much larger culvert or perhaps a bridge. As part of the restoration study, the grid resolution for Muddy Creek was increased, resulting in the use of more normal friction coefficients.

Key Issue 4 - Is the RMA2 Model Well Corroborated?

As noted above, the Panel was instructed to focus on the Pleasant Bay and Bournes Pond applications. In these applications, measured water surface elevations on the ocean boundary were applied. After varying the bottom friction and eddy viscosities, the computed water surface elevations in the interior were then compared to collected field data in two ways. First there was a visual comparison of the computed and observed elevations. Next, a harmonic analysis of the measured and computed elevations was conducted to yield the major harmonic constituents of the tide. Both the amplitude and phase of each measured and computed constituent were then compared. The Panel considers the resulting comparisons for both Pleasant Bay and Bournes Pond to be excellent.

In both the Pleasant Bay and Bournes Pond applications, Acoustic Doppler Current Profiler (ADCP) velocity measurements were available across the inlets and for several other transects. These measurements were used to compute the fluxes of water at these locations, thus providing data for an independent corroboration of the RMA2 model. The measured water fluxes closely match those computed by RMA2. Thus, the Panel concludes that the RMA2 models for Pleasant Bay and Bournes Pond are well corroborated.

Key Issue 5 - Will the RMA2 and RMA4 Models be Updated to Include the 2007 Breach in Pleasant Bay?

After this issue was raised, Applied Coastal Research and Engineering provided a report describing a study for the New England District of the Corps of Engineers in which the 2007 breach was included in the Pleasant Bay model. The impact of the breach is to increase tidal flushing by about 15%.

Given that implementation of TMDLs for the Cape Cod estuaries and bays is projected to be an ongoing process for the next 20-30 years, and the uncertainty of future conditions of the breach, the Panel anticipates that the numerical models will continue to evolve. As part of this process, the Panel recommends that the bathymetry and geometry in the RMA2 and RMA4 models continue to be updated as new data become available.

5.2.2.2. Estuarine Water Quality Model

Key Issue 1 – RMA4 Model Conceptual Framework

Total nitrogen (TN) is the sum of dissolved inorganic nitrogen (DIN), dissolved organic nitrogen (DON) and particulate organic nitrogen (PON). The RMA4 water quality model is calibrated to TN in Bournes Pond (Howes et al. 2005) and bioactive nitrogen (DIN + PON) in Pleasant Bay (Howes et al. 2006). In the actual estuaries, PON is a component of TN and bioactive nitrogen that settles from the water column into the sediment bed. To model TN or bioactive nitrogen in the water column, this PON settling flux must be taken into account. One way to do this is to include a net apparent settling velocity in the model, and another way is to adjust a nitrogen input load to compensate for lack of inclusion of an explicit apparent net settling velocity.

In the SMAST RMA4 water quality model, PON settling flux is represented implicitly as an adjustment to observed benthic regeneration fluxes. The observed benthic fluxes are discounted by an assumed PON settling flux so that when they are specified as a sediment-water input load to the model, the PON which has remained in the water column does not get double-counted. Basically, this approach “pre-calibrates” the water quality model with a fixed value for PON apparent net settling velocity.

A strength of this approach is that it is based on observed data for benthic fluxes, but a concern is that it is conducted outside the mass balance equation in the model. Consequently, there is no direct constraint to ensure that the correct adjustment has been made. Furthermore, because this approach implicitly “pre-calibrates” PON net settling velocities, it limits the flexibility to adjust PON net settling fluxes as part of the model calibration.

As the models continue to evolve and improve, the Panel recommends that the SMAST Team consider representing PON settling flux using a net apparent settling velocity in the RMA4 water quality model. A benefit of this approach would be closer alignment between the conceptual framework of the model and net settling fluxes for PON in the estuaries. Additional benefits are that observed benthic fluxes could be directly specified as model input, thus enhancing transparency, and that the model calibration, using apparent net settling velocities, would be conducted within the mass balance equation of the model. These improvements to the RMA4 water quality model would strengthen the calibrations to the estuaries and reduce uncertainties in computed nitrogen concentrations.

The Panel notes that the SMAST RMA4 model assumes a linear relationship between reduction in nitrogen load from the watershed and net benthic nitrogen load from the sediments. This is a simplification that ignores potential "memory effects" in the sediments and it adds additional uncertainty to model results for load reduction scenarios.

Key Issue 2 – RMA4 Model Calibration and Validation

Model calibration is the process of adjusting model parameters within physically defensible ranges until the resulting predictions give the best possible fit to the observed data. The traditional paradigm in environmental modeling has been calibration to one set of data and validation of the calibrated model to a second, independent set of data that was not used in the calibration. U.S. EPA (2009) recommends best practices for evaluation of environmental models to help determine when a model, despite its uncertainties, can be appropriately used to inform a decision. The proposed "tools" or practices emphasized by EPA include model corroboration, and sensitivity and uncertainty analysis. Model corroboration is the use of quantitative and qualitative methods to evaluate the degree to which a model corresponds to reality. In practical terms, it is the process of "confronting models with data." In some disciplines, this process has been referred to as validation. The EPA guidance states that in general, the term "corroboration" is preferred because it implies a claim of usefulness and not truth. Wells (2005) has actually argued that validation is a fallacy and that it is all calibration.

The characterizations of model calibration, validation and verification of the RMA4 water quality model appear to differ among the SMAST modeling applications. In Howes et al. (2001), it is stated that calibration of dispersion coefficients in the RMA4 model is typically conducted using salinity, and then verification (validation) is typically conducted using nitrogen concentrations. In the Pleasant Bay (Howes et al. 2006) and Bournes Pond (Howes et al. 2005) applications, the RMA4 model is calibrated to nitrogen concentrations and then verified (validated) to salinity.

The Panel recommends that the SMAST Team adopt terminology that is consistent with the U.S. EPA (2009) guidance and avoid characterizing their models as "validated." A claim of model validation tends to confer a model with legitimacy even though the use of models to develop TMDLs involves conducting forecast simulations for nutrient loadings and/or environmental conditions outside the range of those in the model calibration datasets. Environmental processes are extremely complex and inherently uncertain, and even the best and most sophisticated models are only simplistic representations. If the SMAST Team is compelled to state whether their models have been validated, the Panel recommends that they characterize validation as a process, not an end result, and point out that model validation cannot ensure acceptable predictions (Hassan, 2004).

The Panel further recommends that the SMAST Team clarify their use of salinity and nitrogen concentrations to determine dispersion coefficients in the RMA4 model. Salinity can be used to calibrate the dispersion coefficients in a water quality transport model because it is a conserved constituent. Once physical processes are calibrated, they should not be modified during the water quality calibration (Chapra 2003). However, the use of salinity to determine dispersion coefficients can become problematic in cases where high rates of tidal exchange and small freshwater inflows tend to minimize salinity gradients, especially in the open bays.

One possible alternative is to use the dispersion of momentum coefficients (eddy viscosities) that can be computed in RMA2 to specify the dispersion of mass coefficients in RMA4. The Schmidt Number is defined as the ratio of the eddy viscosity coefficient to the mass dispersion coefficient. Thus, if an estimate of the Schmidt Number can be made, the mass dispersion coefficients in RMA4 can be determined from the computed eddy viscosities in RMA2. As discussed by Duan (2004), various studies have been conducted to determine the Schmidt Number as it relates to vertical coefficients for eddy viscosity and mass dispersion (Rodi, 1984, Demuren and Rodi, 1986). These authors recommend a value of 0.5 in a fully 3D model. However, Ye and McCorquodale (1997) found the Schmidt Number should be reduced to 0.15 in a depth-averaged model. The Panel has recommended to the SMAST Team that this approach be explored for determination of dispersion coefficients in the RMA4 model.

Key Issue 3 - RMA4 Application to Bioactive Nitrogen in Pleasant Bay

Total nitrogen concentration in the water column is the state variable in the RMA4 model applications for all of the MEP estuaries to date, with the exception of Pleasant Bay. Bioactive nitrogen concentration is the state variable for the Pleasant Bay application. Total nitrogen is the sum of DIN, PON and DON, and bioactive nitrogen is the sum of only DIN and PON. The SMAST Team considered DON to be a large non-active pool generally separate from the nitrogen fractions active in eutrophication (Howes et al. 2006).

In discussions with the SMAST Team, they stated that the reason for applying the RMA4 model to bioactive nitrogen in Pleasant Bay was a large, unexplained difference in observed total nitrogen concentrations between the ocean boundary monitoring station (PBA-17a) and the monitoring stations in the lower portion of Pleasant Bay (PBA-18, PBA-19 and PBA-01). The SMAST Team judged that this difference was inconsistent with the highly dispersive mixing in lower Pleasant Bay, and that bioactive nitrogen was a more appropriate state variable because it appeared to be relatively constant within and outside of Pleasant Bay.

The results for observed total nitrogen and bioactive nitrogen concentrations in Table VI-I of Howes et al. (2006) appear to support this reasoning. Mean total nitrogen concentrations at PBA-01, the station in the lower-most portion of Pleasant Bay, and PBA-17a, the station used to specify the ocean boundary concentration in the RMA4 model, are 0.433 and 0.232 mg/L, respectively. Mean bioactive nitrogen concentrations at these same stations are 0.105 and 0.094 mg/L, respectively. There appears to be a large spatial gradient in total nitrogen concentration, but a small or no gradient in bioactive nitrogen concentration. However, the observations for the ocean boundary station (PBA-17a) represent only summer 2005 conditions because data were unavailable for other years, while the observations for all of the stations within Pleasant Bay, including PBA-01, represent summer average conditions for 2000-2005.

The SMAST Team provided the Panel with mean values for observed total and bioactive nitrogen concentrations for individual summers for selected stations in the lower portion of Pleasant Bay. Comparisons using data for only 2005 appear to show small or no spatial gradients for either total nitrogen or bioactive nitrogen concentrations between stations PBA-01 and PBA-17a. Total nitrogen concentrations at PBA-01 (lower bay) and PBA-17a (ocean boundary) for 2005 are 0.290 and 0.232 mg/L, respectively. Bioactive nitrogen concentrations for 2005 at these same stations are 0.086 and 0.094 mg/L, respectively.

The Panel recommends that the SMAST Team conduct a detailed review of the primary data for total and bioactive nitrogen concentrations for monitoring stations in lower Pleasant Bay and the vicinity of the ocean boundary. Depending on the results of such a review, the SMAST Team might re-consider their decision to apply the RMA4 model to bioactive nitrogen instead of total nitrogen. If the RMA4 model could be applied to total nitrogen in Pleasant Bay, this would result in consistency among the model applications across all of the MEP estuaries, and would eliminate potential ambiguity in the use of bioactive nitrogen for only Pleasant Bay.

Key Issue 4 - Sensitivity Analysis

As discussed above in Section 5.2.1, U.S. EPA (2009) recommends sensitivity analysis as the principal evaluation tool for characterizing the most and least important sources of uncertainty in environmental models. The SMAST Team conducted sensitivity analyses for the Great Pond Case Study (Howes et al. 2001) that included watershed nitrogen loads, atmospheric deposition, benthic fluxes and dispersion coefficients. These analyses did not include nitrogen concentration at the ocean boundary. Furthermore, these sensitivity analyses were not extended to include either the Pleasant Bay or Bournes Pond applications.

There are important differences among these three systems and the Panel recommends that the SMAST Team conduct sensitivity analyses for each specific estuary. For example, the benthic flux loads for Great Pond, Pleasant Bay and Bournes Pond are responsible for 1, 46 and 72 percent, respectively, of the total nitrogen loads to the water column. The benthic flux load for Great Pond is actually slightly negative, indicating that the sediments are a net sink and not a net source for nitrogen. The atmospheric deposition loads for Great Pond, Pleasant Bay and Bournes Pond are responsible for 12, 22 and 4 percent, respectively, of the total nitrogen loads to the water column. The sensitivity analyses for benthic fluxes should be designed to encompass not only calibration uncertainties, but also uncertainties due to the simplifying assumption of a linear relationship between reductions in nitrogen loads from the watershed and net benthic nitrogen loads from the sediments.

Key Issue 5 - Mass Balance Analyses

In addition to the mass conservation checks discussed above in Section 5.2.2.1, the Panel recommends that the SMAST Team consider using output from the calibrated water quality models to conduct mass balance analyses for the whole Pleasant Bay and Bournes Pond systems, and the principal embayments in each system. These analyses should include all of the individual nitrogen mass flux components into and out of the water column. Results from such analyses would provide useful diagnostic information on the relative contributions of individual nitrogen sources to the most impacted embayments in each system, and would better inform management decisions on how to phase wastewater and nutrient management plans.

5.3. Data-Based Issues

Key Issue 1 - Use of Eelgrass as a Bio-Indicator of Embayment Health

Eelgrass (*Zostera marina*) is a critical and vital bio-physical resource in the coastal embayments and near shore waters of Massachusetts (Costello and Kenworthy, 2011). The high rates of primary productivity and widespread distribution of eelgrass provide essential habitat and food for many species of water fowl and commercially and recreationally important fish and shellfish resources, as well as many valuable bio-physical ecosystem services including sediment stabilization, shoreline erosion, nutrient cycling and nutrient storage (Moore and Short, 2006). There are limits to the degree to which eelgrass can act as an effective nutrient reservoir. These limits have been reached in many coastal ecosystems worldwide where nutrient enrichment and eutrophication have led to significant impairment and eelgrass declines (National Research Council, 2000, Orth et al. 2006a, Waycott et al. 2009, van der Heide et al. 2011). These and other studies have led to a general scientific consensus that eelgrass is a sensitive bio-indicator of ecosystem impairment and can be used to identify thresholds of nutrient concentrations for establishing water quality criteria that will support eelgrass (Dennison et al. 1993, Lee et al. 2004, Kemp et al. 2004, Biber et al. 2008). In Massachusetts estuaries nitrogen is documented to be the primary driving factor for eutrophication; this has been confirmed by many peer reviewed scientific studies and in many different eelgrass systems (e.g., Short et al. 1995, Valiela et al. 1997a, 1997b, Havens et al. 2001, Orth et al. 2006b, Krause-Jensen et al. 2008, Waycott et al. 2009).

The strength in the MEP approach is that it uses using eelgrass as one of the key response indicators. This approach assumes that eelgrass declines result from nitrogen enrichment, an assumption that is well supported by empirical studies. MEP further assumes that nitrogen remediation will lead to environmental conditions that support eelgrass recovery in embayments where groundwater nitrogen sources are reduced so that the concentration in the embayment achieves a threshold value that will support eelgrass recovery. The expectation that eelgrass will recover following nitrogen remediation is supported by observations in similar coastal embayment systems (Vaudry et al. 2010). It is very important to understand that the MEP models do not directly predict eelgrass recovery or any measure of uncertainty in eelgrass recovery. The models only predict the concentrations of nitrogen in the water column. Application of the models to eelgrass recovery assumes that the thresholds determined by the MEP process are supportive for eelgrass. Below, the Panel discusses both the strengths and weaknesses associated with this process of model application and nitrogen thresholds.

Key Issue 2 - Sentinel Station Protocol by which the Absolute Nitrogen Threshold is Determined

The protocol MEP uses to establish the nitrogen threshold for eelgrass in an embayment is based on developing a direct correspondence between the nitrogen concentrations (either total or bioactive, depending on the embayment) and eelgrass condition at one or more sentinel stations. Using this approach it is assumed that: 1) the status of eelgrass at the sentinel station represents a “healthy” functioning eelgrass condition, 2) eelgrass is in an equilibrium state at the sentinel station, and 3) the nitrogen concentration observed at that station is the most significant factor affecting eelgrass status. The underlying strength to this approach is that it is based on data; there are direct measurements of nitrogen concentrations and observations of the correspondence of nitrogen with eelgrass condition. Additional confidence in this approach is gained by studies in eelgrass systems that corroborate a similar relationship in the correspondence between nitrogen and eelgrass condition. Further support for this approach is attained by using more than one data set to evaluate the status of eelgrass condition, including MEP’s evaluation and interpretation of eelgrass mapping data provided by the MDEP and field surveys conducted by the MEP team.

Uncertainties arise from potential weaknesses in the basic assumptions of the correspondence approach. Scientific consensus indicates that a healthy eelgrass ecosystem has a very significant influence on the local environment with many positive feedbacks that affect biophysical processes and support eelgrass growth and survival, including parameters such as nutrient cycling and storage, sediment stabilization, optical water quality and biological oxygen demand (Moore, 2004, Moore et al. 1996, Havens et al. 2001, Homer and Bondgaard, 2001, Carr et al. 2010, van der Heide 2011). If it is assumed that the sentinel station is a healthy functioning eelgrass system, the environmental conditions at that station would be affected by the presence of eelgrass and the nitrogen concentrations at the sentinel site will reflect this. In the transition from impaired to unimpaired state where eelgrass is sparse or absent, the positive feedbacks will not be functioning and nitrogen delivered to the system will be available for other primary producers to utilize. Empirical and modeling studies demonstrate that eutrophication in southeastern Massachusetts is derived mostly from a combination of nitrogen enhanced chlorophyll abundance in the water column and excessive macroalgal growth (Valiela et al. 1997b, Hauxwell et al. 2001). The MEP program has not specifically identified nitrogen threshold values for either of these two components, so it is not possible to predict if the threshold concentrations would limit these primary producers from outcompeting eelgrass (Pedersen and Borum, 1996, McGlathry et al. 2007).

The threshold approach assumes that the transition from impaired to unimpaired conditions will operate as a simple switch when nitrogen reaches a specified threshold concentration. This approach oversimplifies the complex process of eutrophication that results in the impairment of several ecosystem functions and changes in the bio-physical properties of the sediment. The positive feedbacks associated with sustaining a healthy eelgrass state, presumably present where the threshold value is determined (e.g, water clarity, nutrient storage, oxygen production), are not present in the impaired condition. Nitrogen is now interacting with other potential stress factors. Many of the most impaired embayments have changed state (Havens et al. 2001, van der Heide et al. 2011, Carr et al. 2010, Fonseca, 2011) and these other potential stress factors, including optical water quality (suspended sediments, chlorophyll), light availability, substrate condition, water depth, and competitors will be important in affecting eelgrass recovery (Goodman et al. 1993, Moore et al. 1997, Kemp et al. 2004, Wazniak et al. 2007, Gallegos et al. 2010) (Figure 3). To achieve desired eelgrass restoration goals the plants will also have to expand across depth gradients into deeper water where light availability will become a critically significant factor.

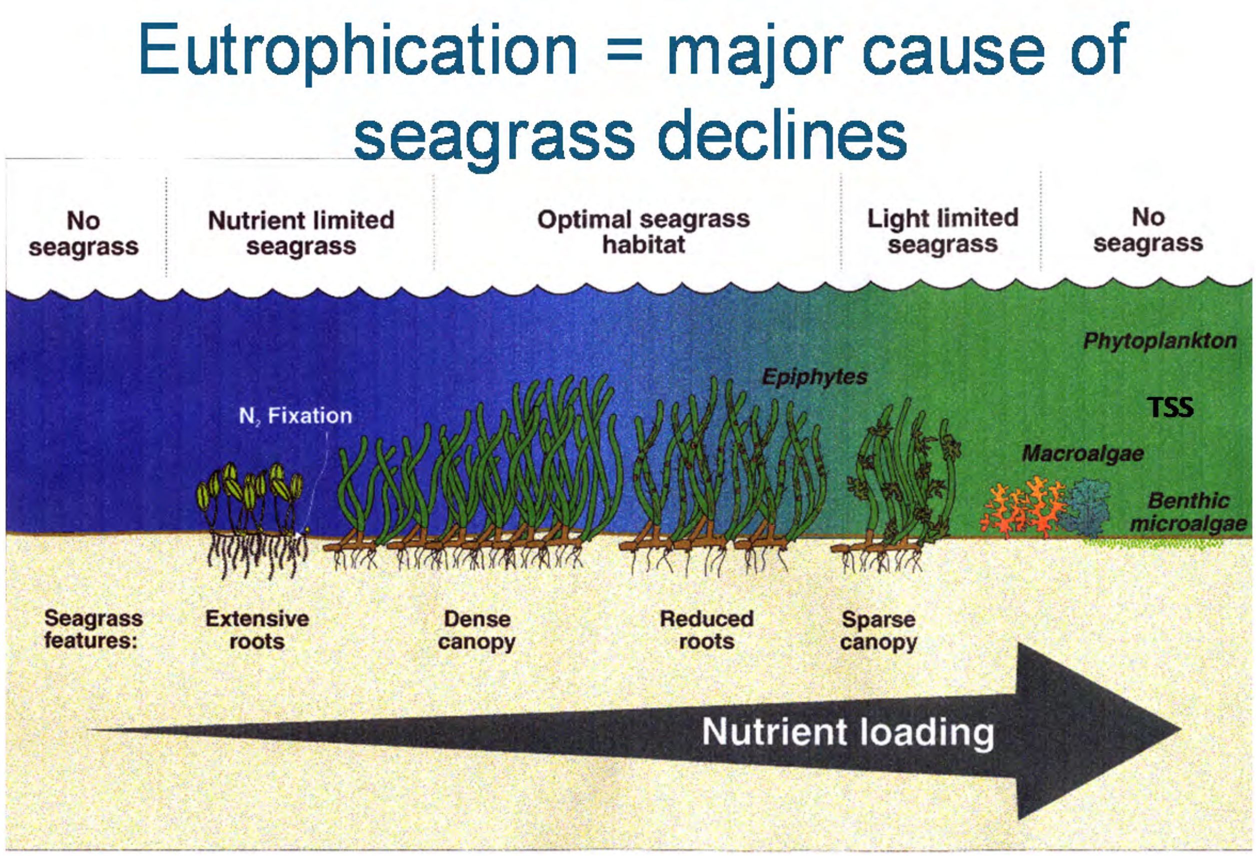

Figure 3. Conceptual Diagram Illustrating How Nutrient Loading and Eutrophication Leads to Bio-physical Changes in the State of an Eelgrass System

(Courtesy of the Integration and Application Network, Center for Environmental Science, University of Maryland)

Further uncertainty is introduced by the assumption that the sentinel station is in a stable equilibrium condition. Recently published studies suggest that many embayments in Massachusetts are in transition with respect to their eelgrass distribution and abundance (Costello and Kenworthy, 2011, Neckles et al. 2011). The majority of Massachusetts embayments are recently displaying declines (Costello and Kenworthy, 2011) and most of the MEP reports state that longer term declines have been occurring in the impaired systems. In the most intensively studied portions of Pleasant Bay there is both evidence for eelgrass increases (Neckles et al. 2011) and decreases (Costello and Kenworthy, 2011), including a recent assessment of Ryders Cove, a sub-embayment of Pleasant Bay and the location of one of the sentinel stations (personal communication, Charles Costello, MDEP). Ryders Cove is now nearly devoid of seagrass. The degree of uncertainty associated with potential transitions to unstable conditions cannot be resolved by inspection of the MEP reports because the description of their field sampling methodologies for assessing eelgrass health and condition is inadequate. There are no detailed descriptions of the survey methods used, the location and number of stations sampled, specific results or methods of data analysis. Results are presented in narrative form and summarized with categorical variables. Furthermore, the MEP program attempts to corroborate their assessment of the sentinel station conditions using the MDEP mapping data, but the mapping data are generalized across the embayments and provide information about the overall condition of the systems, not necessarily any specific sentinel station. It is not possible for an expert to critically evaluate the strength and reliability of the sentinel stations as a quantitative threshold without understanding what specific data are being used and how it was collected and analyzed by MEP (e.g., shoot density, areal coverage, biomass, water depth, etc.).

Determining how to establish nitrogen threshold concentrations in embayments where eelgrass is either completely absent or in very low abundance is another uncertainty (e.g., Buttermilk Bay and Waquoit Bay). In these embayments it is not feasible to designate "in situ" sentinel stations. As each embayment in the MEP program is being treated as a separate entity with unique model parameters and threshold characteristics, it is very difficult to specify a level of confidence in transferring and applying threshold values from other embayment systems. A simple dose-response relationship between nitrogen concentration and eelgrass decline in an individual embayment may be reasonable in explaining long-term declines; however, modeling and empirical studies suggest that the sensitivity of vegetation (eelgrass, macroalgae and phytoplankton) to eutrophication can vary widely across systems (Havens et al. 2001, Krause-Jensen et al. 2008, Latimer and Rego, 2010). In some cases there may be no relationship at all and systems with low nitrogen loading do not support eelgrass for reasons other than nitrogen enrichment (e.g. light limitation, sediment conditions or recruitment limitations).

Moving forward, the Panel recommends that the MEP adopt a more comprehensive approach for assessing the environmental conditions and status of eelgrass at sentinel sites. Predicting the level of certainty and the overall extent for eelgrass expansion into unvegetated and formerly impaired conditions where eelgrass is competing with other primary producers can be improved by incorporating more comprehensive monitoring and assessment of factors which affect eelgrass growth, reproduction and dispersal including optical water quality, substrate condition and water depth. The MDEP has a calibrated optical water quality model for eelgrass in southeastern Massachusetts embayments that could be used by the MEP program to expand the scope of understanding for the status of factors which affect eelgrass distribution, abundance and survival beyond nitrogen (Biber et al. 2008, Gallegos et al. 2010). The MEP also has an extensive bathymetry data set for each of the modeled embayments that can be used with the bio-optical model to conduct a more comprehensive environmental monitoring and assessment of threshold stations as well as making better predictions of the suitability of impaired embayments for eelgrass growth when nitrogen remediation and embayment restoration are implemented.

The Panel further recommends that the MEP consider using standard methods for quantitatively assessing and reporting the health and condition of eelgrass at sentinel stations (see for example, Short and Coles, 2001, Neckles et al. 2011). Where feasible, categorical variables should be avoided and the sampling methods should provide numerical results (metrics) with measures of spatial and temporal variation that can be statistically compared with reference station criteria to ensure that the sentinel sites represent equilibrium conditions for eelgrass.

Key Issue 3 - Establishing Realistic Eelgrass Restoration Goals with Nitrogen Thresholds

The expectation that eelgrass recovery in Massachusetts embayments will occur following nitrogen reductions is supported by empirical studies and observations at locations where point source discharges were modified (e.g., Mumford Cove, CT; New Bedford Harbor, MA; and Boston Harbor, MA) (Vaudry et al. 2010, Leschen et al. 2010, Costello and Kenworthy, 2011). Where viable reproductive populations are present, natural recovery of eelgrass can proceed by dispersal of flowers and seeds to unvegetated areas with suitable substrate and favorable environmental conditions (Greeve et al. 2005, Orth et al. 2006b, Orth et al. 2006c, Vaudry et al. 2010). In contrast to natural recruitment, eelgrass restoration using adult transplants that depend on clonal growth is much slower than seed dispersal. Transplanting adult plants is an unpredictable process because success depends on several biological and physical factors and not just the concentration of nitrogen (Fonseca et al. 1998, Short et al. 2002, Paling et al. 2009, Orth et al. 2010). There is a large variation in the success rate and sustainability of seagrass transplanting (Fonseca et al. 2011). The global success rate of seagrass transplanting is only about 50% and attempts to transplant and restore seagrass at the scale of Massachusetts embayments have not been any more successful than the average rate (Fonseca et al. 1998, Moore and Short, 2006, Paling et al. 2009, Orth et al. 2010, Fonseca et al. 2011, Chesapeake Bay Scientific Technical Advisory Committee, 2011). Recent transplant failures in Massachusetts embayments corroborate these uncertainties and indicate how difficult it is to determine if a small site or embayment is suitable for transplanting and restoration (Leschen et al. 2010, Nature Conservancy, 2011). Short et al. (2002) propose using an eelgrass transplanting suitability index that incorporates at least 7 variables (see Table 1 of Short et al. 2002). Dissolved inorganic nitrogen and total nitrogen, as well as historical eelgrass distribution, are included on the list, but it is clear that several other variables must be assessed to complete a comprehensive evaluation of an embayment's suitability for eelgrass restoration and the extent of coverage that would be expected following nitrogen remediation.

The Panel recommends that uncertainty in restoration goals can be reduced by avoiding use of unreliable records of historical eelgrass coverage (e.g. 1950) and adopting an adaptive management approach that sets restoration targets and adjusts predictions of future extents based on more recent eelgrass coverage data (post 1995) from the MDEP eelgrass mapping program (Costello and Kenworthy, 2011). More realistic and accurate restoration targets can be set by incorporating analyses of the MDEP mapping data with more comprehensive assessments and monitoring of the correspondence between areal coverage and environmental conditions that affect eelgrass distribution and abundance in unimpaired and impaired embayments. As per Short et al. (2002), an embayment restoration suitability index should be developed that incorporates a wider set of predictor variables than just the nitrogen concentrations at the sentinel stations.

5.4. TMDL Issues

Key Issue 1 - Appropriate Translation of Science into Management in TMDLs

Over the last 20 years, TMDL analyses have been the primary mechanism for setting water quality-based management targets for impaired waterbodies that require management beyond levels that technology-based controls have provided. Under Section 303(d) of the Clean Water Act (CWA), the TMDL process requires states or EPA to: set water quality targets, usually based on state water quality standards; determine the degree to which the standard is violated; identify sources of the pollutant by point and nonpoint source category; establish allocations of the pollutant between point and nonpoint sources, including a margin of safety, with consideration given to seasonal variation; and develop an implementation plan and schedule for attaining the TMDL, including "reasonable assurances" that the unregulated nonpoint source target will be attained. In sum, the TMDL itself is defined as a pollutant loading capacity that will meet water quality standards and is the sum of natural background loading + the wasteload allocation assigned to regulated point sources + the load allocation assigned to unregulated nonpoint sources + a margin of safety to account for uncertainty.

The Panel was charged to review the MEP modeling approach to determine its scientific defensibility and functional adequacy for "…the development and implementation of appropriate nitrogen TMDLs for the estuaries and embayments of Cape Cod…" An approach deemed "scientifically defensible" and "functionally adequate" for TMDL development would appropriately meet the requirements for a TMDL and meet with EPA's approval. Other concerns in the Panel's charge relate to the utility of the MEP's models and analyses for comprehensive wastewater nutrient management planning, which would follow if EPA CWA requirements for approval of TMDLs are met. Comprehensive Wastewater Management Plans (CWMP), developed in response to nutrient management requirements, are supported by the same scientific understanding that is required to develop the TMDL (e.g., Stearns & Wheler, LLC, 2009). Simply stated, if the science supports a viable TMDL, there is strong likelihood that the management outcomes will be equally supportable by the science.

The Panel was asked to review two TMDLs and the MEP modeling and analysis behind the development of those TMDLs. The two TMDLs reviewed were for Great, Green and Bournes Pond Embayment Systems (hereafter "Bournes"; Commonwealth of Massachusetts, 2006) and for the Pleasant Bay System (hereafter "Pleasant Bay"; Commonwealth of Massachusetts, 2007), both addressing nitrogen as the pollutant of concern. In Section 4 above, the Panel has determined that the technical underpinning for the TMDLs and management plans that follow provided by the MEP is both scientifically defensible and functionally adequate. So, the science used to identify nutrient-related impairments, establish effects thresholds (or criteria), and develop the TMDL are adequate for that purpose, and for the implementation planning that will follow.

It should be clear that there is no set standard for scientific adequacy under the CWA or established protocols for its determination during EPA review and approval of TMDLs. In 1998, a Federal Advisory Committee (FACA), established in November 1996 by the EPA, reported on its charge of "…recommending ways to improve the effectiveness and efficiency of State, Tribal and EPA programs under Sec. 303(d) of the Clean Water Act" and "…the science and tools needed to support the program" (U.S. EPA 1998). The Committee concluded:

In developing TMDLs, States and EPA must use the highest degree of quantitative analytical rigor available. A reasonable minimum amount of reliable data is always needed. Decisions and assumptions based on best professional judgment must be well-documented. TMDLs for which a high degree of quantitative analytical rigor is not possible in target identification and/or load allocation should contain relatively more rigor or detail in their implementation plans, including provisions for follow-up evaluation and potential revision based on the evaluation.

The EPA FACA stressed rapid progress for restoring impaired waters, which they set at a high priority, further stating, "In cases of uncertainty, an iterative approach to TMDL development and implementation will assure progress toward water quality standards attainment." This report set the stage for developing TMDLs in accordance with the level of scientific certainty, and using phasing or adaptive management to take prudent steps forward, and improve on them later. However, as national TMDLs approved by EPA now exceed the 50,000 mark, with about 8,000 of them addressing nutrient-related impairments, phased or adaptive approaches are seldom realized in practice. Rigorous attainment of numerical standards through permit issuance is the usual outcome of a TMDL, emphasizing regulated point sources subject to the wasteload allocation. Because of the enforceable mechanisms and higher certainty of outcome for regulated sources under the wasteload allocation, EPA generally requires point source reductions be higher to compensate for the uncertainty of unregulated or voluntary nonpoint source management.

In the cases of the Bournes and Pleasant Bay TMDLs, which technically represent 5 and 16 TMDLs, respectively, both have been approved by EPA – Bournes on July 18, 2007 and Pleasant Bay on October 24, 2007. There are 21 TMDLs because each represents an individual impaired waterbody segment, and a single estuary can contain multiple impaired segments. As adopted TMDLs, the target nitrogen reductions that meet the standards described in the TMDL analyses become the regulatory and management targets that define a "threshold" of health for each of the 21 impaired waterbodies within the two estuarine systems. These targets can only be changed with a revised TMDL, supported by scientific analysis that is deemed by EPA to warrant a change.

The EPA approval process occurs at the regional office level, in this case the New England Region (Region 1) office in Boston. EPA staff provided a review memorandum for each TMDL that considered a set suite of 12 elements that include both technical and procedural or administrative requirements. Of special relevance to the Panel's charge on scientific review, both TMDL reports translated Massachusetts narrative standards for nutrients into site specific "threshold nitrogen concentrations" for each waterbody in both estuarine systems. These targets now have the force of adopted numeric criteria. The threshold concentrations were set using a "reference" approach (Howes et al. 2003, U.S. EPA 2001), as described in relevant technical support documents prepared by MEP (Howes et al. 2005 and 2006). Those values are the driving force behind the actions that meet the TMDL load reductions to ensure that the threshold nitrogen concentrations are not exceeded once management practices are in place.

The TMDL load allocation is essentially a pollutant budget that identifies the load reduction, and cap, necessary to attain water quality standards. For the estuaries reviewed, the regulated wasteload allocations (WLA) are minimal, so the focus is necessarily on unregulated nonpoint sources, primarily septic systems (*See* Table 3 in Commonwealth of Massachusetts, 2006 and 2007, for Bournes and Pleasant Bay categorical nitrogen loads, respectively). The Panel's technical analysis has determined that the loading estimates are scientifically defensible and, therefore, appropriate for TMDL development and management planning purposes.

Key Issue 2 - Adaptive Management

As described above, the EPA FACA tacitly endorsed a phased or an adaptive approach to address uncertainty and to ensure immediate management progress consistent with the level of scientific understanding, and the ability to manage pollutant sources. This approach is most relevant to the load allocations for nonpoint sources, which provide pollutant quantification and management challenges that generally exceed those for regulated point sources. Because the Cape Cod sources are almost exclusively nonpoint in nature, especially on-site subsurface disposal systems (OSDS) or "septic" systems, successful attainment of the TMDL cannot rely on emphasizing regulated point sources in the wasteload allocation.

The nitrogen source distribution imbalance towards nonpoint sources will undoubtedly make attainment of the TMDL more challenging, and perhaps more costly. However, it may provide a better framework for phased implementation that could support a structured adaptive management approach. As recommended above in the Panel's review of the science, an adaptive management approach will help reduce scientific uncertainty and help adjust management actions so they may be more cost-effective within the iterative framework of adaptive management (Figures 1 and 2) (Williams et al. 2009). The Cape Cod Commission and the member municipalities have explored adaptive management approaches relative to phased implementation plans (Cape Cod Commission, 2009, GHD, Inc. 2011).

The Bournes TMDL identifies OSDSs as the dominant source of nitrogen loading to each of the five sub-embayments, ranging from 76% to 87% of the Present Watershed Load (Commonwealth of Massachusetts, 2006, Howes et al. 2005). The TMDL reductions of the controllable loads range from 55% to 87% (*See* Table 4 in Commonwealth of Massachusetts, 2006), which means that OSDSs must be aggressively managed if the TMDLs for the Bournes Systems are to be attained. This direction is confirmed in CWMPs (e.g., Stearns & Wheler, 2009) being developed by the municipalities that share the watersheds, which also provide the "reasonable assurance" that action will be taken to implement the TMDL.

For the 16 Pleasant Bay sub-embayments, OSDS nitrogen loads ranged from 51% to 83% of the Present Watershed Load (Commonwealth of Massachusetts, 2007, Howes et al. 2006). TMDL reductions of the controllable load ranged from 25% to 83% (*See* Table 4 in Commonwealth of Massachusetts, 2007). While this broader range reflects a wider variety of conditions in the watersheds of Pleasant Bay compared to Bournes, since OSDSs are again the dominant source, aggressive management will be required and will be detailed in CWMPs developed for the Pleasant Bay System as well.

This Panel's assessment supports the science behind the goal setting, but there is uncertainty as to whether the TMDL will fully restore the estuaries' designated uses, especially for sensitive eelgrass meadows (see Section 5.3 above). However, it seems likely that management will have to *at least* meet the thresholds identified in the TMDLs, and that continued monitoring and assessment are necessary to reduce uncertainty. This sets the stage for an effective adaptive management approach since the towns are allowed flexibility by the MDEP and the EPA to phase implementation, monitor the results, and chart progress with respect to predicted environmental outcomes. In particular, tracking of benchmark nitrogen concentrations, the attainment of the current criteria and the presumptive attainment of eelgrass and other biological indicator endpoints should be a focus of a structured adaptive management program within the CWMPs.

In the EPA approval letter, and in MDEP's construct of the TMDL, there appears to be room for flexibility and a willingness to work with the towns to develop viable implementation plans. Because of the cost and time that will be required to implement infrastructure improvements of this scale, there is a likelihood of phasing that would provide an opportunity for structured adaptive management program. A potential management outcome already identified in CWMPs is constructing sewers and centralized wastewater treatment facilities. Those actions would technically shift the nitrogen load from a nonpoint source load allocation to a point source wasteload allocation within the TMDL. That reallocation requires an update of the adopted TMDL, and reapproval by EPA, which would also provide an opportunity to apply the reassessment and adjustments that are learned in an adaptive management framework.

A watershed/adaptive management approach could serve to more equitably distribute nitrogen reductions and their implementation costs among the contributing watershed sources. Market mechanisms, such as nitrogen trading, may also facilitate management and lower costs. However, the size and character of Cape Cod's watersheds, the high levels of nitrogen removal required to meet TMDLs, and the limited diversity of sources and management options may be obstacles to watershed market mechanisms, like trading. Small watersheds with few sources responsible for most or all of the impairment have low market appeal because options for trading are so limited. Hence, nitrogen credits would likely be a scarce commodity in many Cape Cod watersheds and would become even more limited as attainment of the final TMDL nears.

Nitrogen trading may have a role in accelerating progress in the early phases of management, providing a mechanism for implementing the most cost-effective actions first. Initially, nitrogen management costs on Cape Cod may be low compared to the early, projected water quality benefits, i.e., cost-to-benefit ratios would be lower. Typically, marginal costs will increase with time, especially if attaining the final management target requires intensive application of a high level of costly technology. At the point where the "knee of the curve" for cost-to-benefit ratios is passed, trading may no longer provide an incentive to implement, yielding to a final management structure such as a collective utility with an equitable cost share among residents. More detailed study on market options, utilities and other funding mechanisms is recommended, expanding upon the Cape Cod Commission (2009) initial exploration of trading and other options.

6. CONCLUSIONS

6.1. Response to Charge Question 1

Charge Question 1 to the scientific peer review panel was:

> *Is the MEP modeling approach scientifically defensible and functionally adequate to be relied upon in the development and implementation of appropriate nitrogen TMDLs for the estuaries and embayments of Cape Cod in support of the state's Comprehensive Wastewater Management Planning and EPA Clean Water Act requirements and in developing overall wastewater and nutrient management plans for Cape Cod to meet the TMDLs?*

The Panel finds that the MEP modeling approach is scientifically credible. It is consistent with current understanding of existing conditions for Cape Cod estuaries, based on available data. The components in the approach are well-known and documented. Computation of watershed nitrogen loads is strongly data-driven and quantitatively linked to estuarine nitrogen concentrations.

A fundamental principle in the development and application of environmental models to inform management decisions is that there should be compatibility among the study questions and objectives, available data and resources, and level of model complexity. The Panel finds that the level of complexity in the components and linkages of the MEP modeling approach is simple, parsimonious and well balanced within this context.

The Panel also finds that the MEP modeling approach is functionally adequate. This approach is specifically designed for groundwater dominated systems and explicitly considers nitrogen loads from septic systems, the dominant controllable watershed source of nitrogen for Cape Cod estuaries. The MEP modeling approach is appropriate and useful, within the overall decision support system, for evaluating alternative scenarios and informing nutrient management plans, and is consistent with existing nationwide TMDL practices.

6.2. Response to Charge Question 2

Charge Question 2 to scientific peer review panel was:

> *To what level of accuracy will the MEP linked model predict the effect of alternative nitrogen load planning scenarios and/or the prospective water quality in the affected estuaries and embayments and what is the degree of <u>uncertainty in those predictions relative to alternative planning methodologies available in the industry?</u>*

In order to respond to this question, the Panel assembled a list of the various parameters that serve as inputs to the MEP linked model. For the calculation of nitrogen loads alone, we counted more than twenty. Moreover, each parameter has a greater or lesser effect on the linked model predictions depending on the character of the particular watershed to which it is being applied. For example, for Pleasant Bay with its large surface to watershed area ratio, direct atmospheric deposition to the Bay has a more significant effect than to bays with smaller surface to watershed area ratios. In those bays with smaller surface area, septic and lawn fertilizer loads are correspondingly more significant.

The large number of parameters and their varied impact makes it difficult to provide a quantitative estimate of a single degree of uncertainty. Rather, for complicated and dynamic systems such as these, there will never be a single correct answer and any prediction, no matter how refined, will necessarily carry some degree of uncertainty. Uncertainty arises from the fact that model parameters are not known exactly (parametric uncertainty), that input data is not fully known (input uncertainty) and that processes are inherently more complex than can ever be completely captured by models (i.e., intrinsic or structural uncertainty). In addition, natural processes are variable in time and space, and there can be consistent changes in conditions over time, contributing to non-stationarity in statistical estimates of parameters, forcing conditions and internal adjustments of watershed and responding water bodies (Milly et al. 2008). The issue of non-stationarity reinforces the need for adaptive management, involving continued monitoring and modeling to test restoration expectations and appropriately refine management activities.

As discussed above in Section 5.2.1, U.S. EPA (2009) recommends sensitivity analysis as the principal evaluation tool for characterizing the most and least important sources of uncertainty in environmental models. Perhaps the best place to get a sense for the degree of uncertainty in the MEP linked model is the MEP sensitivity study (Howes et al. 2002). Table III-7 of the sensitivity study shows the extent to which varying selected model parameters changed the predicted nitrogen concentration in Great Pond, the test case for sensitivity evaluation. Relatively drastic variation (± 50%) in the watershed loads from septic sources and lawn fertilizers produced comparatively modest changes in the predicted nitrogen concentrations (on the order of ± 10 to 20%). The Panel's evaluation of the parameter values used in the loading calculations indicates that the loads are uncertain to a degree less than 50% and thus that the predicted nitrogen concentrations are uncertain to a degree less than 20%.

One aspect of the MEP reports that we recommend for improvement is in providing some sense of the uncertainty in the study results. The Panel believes that it would assist those attempting to develop nitrogen control measures if they were provided a realistic sense of the degree to which model results are uncertain and therefore the degree to which flexibility should be included in any action plans. To this end, each of the MEP studies should include a sensitivity analysis of key components and links in the nitrogen loading and transport chain. The goal would be to drive these with realistic estimates of the uncertainty in model inputs and processes in order to determine how those components affect the bottom-line uncertainty in the nitrogen concentrations forecast for the receiving bays and estuaries. The sensitivity analysis included in the MEP sensitivity study (Howes et al. 2002) is a good model, but rather than a one-time exercise, a similar approach should be included in the report for each estuary. Such a sensitivity study is needed for each estuary because sensitivity to the different model parameters varies substantially within the different estuary systems. The Panel also recommends that the list of parameters evaluated in the sensitivity study for each estuary be expanded from that in Howes et al. (2002) to also include the concentration of nitrogen at the ocean boundary since this is known to be a sensitive parameter.

The Panel considered more rigorous and exhaustive alternatives to assessing model uncertainty: for example, Monte Carlo simulation to derive a probability distribution for predicted nitrogen concentrations. In the end, we concluded that such an effort was inconsistent with the scope and purpose of the MEP program. The completion of sensitivity analyses constitutes, we believe, a reasonable compromise between the need for information to understand the level of accuracy in the MEP results and the potentially considerable effort necessary to derive that information. Moreover, sensitivity analyses address the uncertainty in the overall MEP approach to predicting receiving-water nitrogen concentrations. It implicitly considers, for example, model-based linkages using the groundwater, watershed loading, hydrodynamic, salinity and water quality models.

Despite the value of the recommended sensitivity analyses, they fall short of assessing the uncertainty in the response of receiving waters to changes in nitrogen concentrations-for example, the state of eelgrass. Unfortunately, the state of science is such that at the present time, prediction uncertainties for data-based linkages between threshold nitrogen concentrations and water quality/ecological responses can only be based on data analysis and best professional judgment. However, the Panel believes the uncertainties in these data-based linkages are conservative in the sense that the actual nitrogen concentrations required to achieve the eelgrass endpoints are probably lower than the threshold nitrogen concentrations in the TMDLs. The recommended adaptive management approach will result in clarification and appropriate adjustment of these threshold values as the process moves forward.

Overall, results from sensitivity analyses will inform judgments about prediction uncertainties as implementation of nitrogen control measures proceeds. To this end, results from model sensitivity analyses, in combination with knowledge of each of the systems, can be used to identify pilot sites at which temporal changes are expected in key model inputs, linkages, or water quality/ecological response parameters. Focused monitoring at these pilot sites can then provide data for testing predictions, quantifying uncertainties, and improving the components and linkages in the decision support system.

7. RECOMMENDATIONS FOR PATH FORWARD

The preceding sections of this report contain numerous specific recommendations pertaining to the overall MEP modeling approach and the individual topic areas. This section contains a high level categorical summary of the Panel recommendations for a path forward.

The Panel recommends that the MEP modeling approach be considered within the larger context of the overall decision support system and not be limited to just the linked watershed-embayment model. The Panel further recommends that an adaptive management framework be used for this decision support system, which integrates the watershed-embayment model. This integration should include continued monitoring, data analysis and modeling to improve scientific understanding and reduce uncertainties in the physical, chemical and biological processes in the watersheds and estuaries.

The Panel recommends that the towns proceed within this adaptive management framework to develop and implement wastewater and nutrient management plans, and make improvements along the way to reduce management uncertainties. This will ensure that TMDL implementation is not compromised due to a lack of information, and that progress will be made in the most cost effective manner while gathering new information to improve upon the scientific analysis, and the initial wastewater and nutrient management plans.

The Panel recommends that model sensitivity analyses be conducted for the components and linkages in the watershed-embayment model for each specific estuary. Sensitivity analysis is the principal evaluation tool for characterizing the most and least important sources of uncertainty in environmental models. The Panel believes that a healthy recognition of uncertainty would encourage planning bodies to pursue an adaptive science and management strategy as they move forward to understand and remediate the impacts of excessive nitrogen loadings on the estuaries and embayments.

The Panel recommends that the MEP adopt a more comprehensive approach for assessing the environmental conditions and status of eelgrass at sentinel sites. Predictions for the expansion of eelgrass into unvegetated and formerly impaired sites can be improved by incorporating additional factors that affect eelgrass growth, reproduction and dispersal, such as optical water quality, bottom substrate conditions and water depth. Emphasis should also be placed on the use of standard methods for quantitative assessments of eelgrass health and condition that incorporate and report measures of spatial and temporal variation. The Panel also recommends that SMAST and MDEP develop a coordinated effort to utilize more recent data from the MDEP eelgrass mapping program to establish restoration targets.

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Appendix A

Resumes of Panel Members

Victor J. Bierman Jr., Ph.D., BCEEM

Senior Scientist, LimnoTech

Areas of Specialization:

Water Quality Modeling
Toxic Chemicals
Eutrophication
Ecosystems
Environmental Assessment
Regulatory Compliance

Education:

Ph.D., Environmental Engineering, University of Notre Dame, 1974
M.S., Physics, University of Notre Dame, 1971
A.B., Science, Villanova University, 1966

Professional Certifications:

Board Certified Environmental Engineer (by Eminence), American Academy of Environmental Engineers

Career Highlights:

- 38 years' experience in water quality modeling, and publication of over 100 technical papers and reports
- Former U.S. EPA National Expert in Environmental Exposure Assessment
- Conducted transport and fate modeling studies for PCBs as part of the Hudson River Reassessment RI/FS and for Total Maximum Daily Loads (TMDLs) in the Delaware and Potomac River Estuaries
- Conducted eutrophication modeling studies as part of the Gulf of Mexico Hypoxia Assessment, and investigated the influence of nutrient loadings from the Mississippi River Basin
- Developed state-of-the-science mathematical models for nutrients, nuisance algal blooms, nitrogen fixation, exotic species and ecosystem processes in the Great Lakes and Chesapeake Bay

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Dr. Bierman is a Senior Scientist at LimnoTech. He has 38 years of experience in the development and application of water quality models for the transport and fate of toxic chemicals, and eutrophication, leading to his publication of over 100 technical papers and reports. He is a former U.S. Environmental Protection Agency National Expert in Environmental Exposure Assessment, and a former Associate Professor in the Department of Civil Engineering at the University of Notre Dame.

As a LimnoTech Senior Scientist, Dr. Bierman conducts research and development on projects for federal, state and regional government clients. He also provides scientific peer review, litigation support, and expert testimony on a variety of environmental issues for government agencies, and industrial, regulatory and private clients.

Dr. Bierman is a leading expert in toxic chemical transport, fate, partitioning and bioaccumulation. He has conducted studies in major river systems, estuaries, and the Great Lakes, and remedial investigations at U.S. EPA Superfund sites.

Dr. Bierman is also a leading expert in the assessment and solution of problems related to nutrients, nuisance algal blooms, nitrogen fixation, exotic species, and ecosystem processes. He has conducted studies in watersheds, lakes, rivers, estuaries and coastal marine systems.

Key accomplishments by Dr. Bierman include modeling of hypoxia in the Gulf of Mexico to assess the influence of nutrient loadings from the Mississippi River Basin; transport and fate modeling of PCBs as part of the Hudson River Reassessment RI/FS; development of a coupled phytoplankton-exotic species-PCB model of Saginaw Bay, Lake Huron; development of models for PCB TMDLs in the Delaware and Potomac River Estuaries; modeling of eutrophication and sediment diagenesis in Lake Okeechobee; development of models for estuarine phosphorus dynamics and algal speciation in the Potomac River Estuary; modeling of phosphorus transport and fate in the Florida Everglades; and service on independent scientific peer review panels for the U.S. Environmental Protection Agency and U.S. Army Corps of Engineers.

Key Project Experience

Gulf of Mexico Hypoxia Assessment, White House Committee on Environment and Natural Resources. Dr. Bierman developed a water quality model for hypoxia in the northern Gulf of Mexico. As Co-Team Leader for Task Group 4, he used the model to assess hypoxia responses to reductions in nutrient loadings from the Mississippi River Basin to support the 2001 Federal Action Plan. Subsequently, he synthesized results from three different models of Gulf hypoxia to estimate the incremental impacts of produced water discharges from oil and gas platforms.

PCB Transport and Fate in the Hudson River Reassessment RI/FS. Dr. Bierman conducted transport and fate modeling studies for PCB-contaminated sediments to investigate the impacts of continued No Action and various remedial scenarios. The modeling results were reviewed by an Expert Panel of independent scientists and linked to site-specific ecological and human health endpoints. This work supported the EPA Record of Decision to remediate contaminated sediments in the Upper Hudson River.

Peer Review of U.S. Environmental Protection Agency Technical Guidance on Nutrient Criteria. Dr. Bierman served as an expert consultant to the U.S. EPA Science Advisory Board, Ecological Processes and Effects Committee, to provide scientific peer review of a draft technical guidance on development of numeric nutrient criteria for the protection of aquatic life.

PCB TMDL Model for the Delaware River Estuary. Dr. Bierman provided expert assistance to the Delaware River Basin Commission on development of a transport and fate model for PCBs, and use of the model for a Stage 1 TMDL. This work was reviewed by an Expert Panel of independent scientists and included close collaboration with a multi-stakeholder Toxics Advisory Committee and a coalition of industrial and municipal dischargers.

Quantitative Assessment Studies on Lake Okeechobee, FL. Dr. Bierman conducted literature review, data assessment and empirical modeling studies of nitrogen impacts on Lake Okeechobee. He was also a principal architect of the Lake Okeechobee Water Quality Model (LOWQM), results of which were used by the State of Florida to develop a TMDL for phosphorus. He also developed a coupled hydrodynamic-salinity model for the Caloosahatchee Estuary, results of which were used to manage freshwater discharges from Lake Okeechobee.

Ecosystem Modeling Studies on Saginaw Bay, Lake Huron. Dr. Bierman developed a mass balance model for the lower food web that included nutrients, phytoplankton, zooplankton, zebra mussels, and PCBs. Results were used to assess the relative water quality impacts of phosphorus loadings and zebra mussel dynamics on phytoplankton production and PCB transport, fate, and bioavailability.

Expert Testimony for Arkansas Food Processor. Dr. Bierman provided expert testimony on transport and fate of phosphorus from land application of poultry litter in the Illinois River Watershed, Arkansas. He prepared a written expert report, was deposed, and testified at trial in U.S. District Court for the Northern District of Oklahoma.

Chesapeake Bay and Potomac River Estuary Water Quality Models. Dr. Bierman conducted a scientific assessment of the Chesapeake Bay Water Quality Model and its use for developing load allocations for nitrogen, phosphorus and solids in the Chesapeake 2000 Agreement. He also developed state-of-the-science sub-models for estuarine phosphorus dynamics, pH-alkalinity and algal speciation for the Potomac portion of the third-generation Chesapeake Bay Water Quality and Sediment Transport Model.

PCB TMDL Model for the Potomac River Estuary. Dr. Bierman developed a transport and fate model for PCBs in the tidal Potomac and Anacostia Rivers to support development of a TMDL by the District of Columbia, Maryland and Virginia (the "Parties"). This work was conducted in close collaboration with a PCB TMDL Steering Committee consisting of the Parties, the Interstate Commission on the Potomac River Basin, and U.S. EPA Region 3.

Peer Review of National Environmental Policy Act (NEPA) Document for U.S. Army Corps of Engineers. Dr. Bierman served as the water quality expert on an Independent External Peer Review Panel for the St. Johns Bayou and New Madrid Floodway (MO) Project. He reviewed the technical analyses in the NEPA document, recommended additional analyses, and reviewed the Project Work Plan.

Water Quality Model for St. Johns River Estuary. Dr. Bierman applied a state-of-the-science water quality model for nutrients, phytoplankton and dissolved oxygen to the Lower St. Johns River and Lake George, Florida. Results from the model were used to develop TMDLs for nutrients, and to investigate the impacts of water withdrawal scenarios proposed under a Water Supply Impact Study.

Litigation Support for U.S. Department of Justice. Dr. Bierman conducted transport and fate modeling for solids and toxic chemicals discharged from the Hammond (IN) Sanitary District Plant to the Grand Calumet River. He prepared a written expert opinion report, was deposed, and provided technical review of opposing expert reports.

Assessment of Mercury Dynamics and Nutrients in Florida Waters. Dr. Bierman is conducting scientific assessment and review of mercury and sulfur dynamics, and nutrient TMDLs, in the Everglades, and of EPA-proposed numeric nutrient criteria for Florida waters. Results will be used to inform management decisions pertaining to mercury and nutrient TMDLs in south Florida.

Peer Review of Dioxin Issue Paper for San Francisco Bay. Dr. Bierman served on an expert peer review panel convened by the San Francisco Estuary Institute to review a dioxin issue paper that addressed environmental and regulatory problems. The panel provided findings and recommendations to the San Francisco Bay Regional Water Quality Board, the U.S. EPA, and the Bay Area Clean Water Agencies.

Expert Assistance on Fecal Coliform TMDL for Urban Stream. Dr. Bierman provided expert technical assistance to the City of Greensboro on development of a fecal coliform TMDL for North Buffalo Creek. Upon completion, this study was put forth by the U.S. EPA as a national case study for discharger-led TMDLs.

Litigation Support for Industrial Discharger on Ohio River. Dr. Bierman reviewed hydrodynamic, sediment transport and chemical fate models developed by the plaintiffs' experts as part of a Natural Resource Damage Assessment claim. He prepared his own expert and supplemental reports, was deposed by opposing counsel, and supported the deposition of the plaintiffs' expert witness.

Peer Review of a Linked HSPF-AQUATOX Modeling System for U.S. Environmental Protection Agency. Dr. Bierman conducted a scientific peer review of a demonstration application of a linked HSPF-AQUATOX modeling system as an alternate approach for development of numeric nutrient water quality criteria.

PETER SHANAHAN

EDUCATION

1982	Ph.D.	Environmental Engineering	Massachusetts Institute of Technology
1974	M.S.	Environmental Earth Sciences	Stanford University
1973	B.S.	Civil Engineering	Massachusetts Institute of Technology
1973	B.S.	Earth and Planetary Sciences	Massachusetts Institute of Technology

PROFESSIONAL HISTORY

1988-date	HydroAnalysis, Inc.
2004	Tufts University
1996-date	Massachusetts Institute of Technology
1981-1988	ERT, Inc. (now ENSR Corporation)
1980	International Institute for Applied Systems Analysis, Laxenburg, Austria
1978-1981	Massachusetts Institute of Technology
1976-1979	Resource Analysis/Camp Dresser & McKee Inc.
1974-1976	Bechtel, Inc.

AFFILIATIONS

Fellow, American Society of Civil Engineers (Committee on Hydrologic Transport and Dispersion, Chairman 1989-1990)
International Water Association (Task Group on River Water Quality Modeling, 1996-2002; Specialist Group on Systems Analysis and Integrated Assessment, 2000-present)
Water Environment Federation (Committee on Research, 1986-1992)
Association of Ground Water Scientists and Engineers (Editorial Board, Journal of Ground Water, 1990-1992)
American Geophysical Union
American Water Resources Association
Conservation Commission, Acton, Massachusetts, 1990-1996

REGISTRATION

Professional Engineer (Civil), Massachusetts
Professional Engineer, Maine

REPRESENTATIVE EXPERIENCE

Dr. Shanahan has directed or been a major contributor to a wide variety of projects involving analysis and computer modeling of environmental water quality, hydrology, and hydraulics. These studies have included engineering analysis and design of water-pollution controls, hazardous waste site remedial actions, flooding and drainage controls, and water-resources development. Dr. Shanahan is an experienced expert witness and has represented clients in courtroom testimony, administrative hearings, negotiations with regulatory agencies, and public meetings. Dr. Shanahan is currently a Senior Lecturer in the Department of Civil and Environmental Engineering at the Massachusetts Institute of Technology, teaching undergraduate and graduate classes and serving as co-leader of the Master of Engineering Program project course.

Ground-Water Hydrology

Dr. Shanahan's experience includes a wide variety of projects involving the assessment and modeling of ground-water hydrology and quality, as well as using models to design remediation measures for contaminated ground water. Example projects include:

Reilly Tar & Chemical Superfund Site, St. Louis Park, Minnesota	Modeled ground-water flow in multiple aquifers affected by coal-tar compounds; developed model for design of gradient and source control wells.
Massachusetts Military Reservation Cape Cod, Massachusetts	Supervised multiple Master of Engineering theses completed by MIT students addressing various aspects of ground-water contamination and remediation at the MMR.
More than twenty Massachusetts municipalities	Employed ground-water flow models to delineate Massachusetts aquifer protection Zone II

Water Quality

Dr. Shanahan's water-quality experience includes academic research to develop modeling approaches and engineering experience analyzing information and using models in practical applications. Project experience includes a wide range of contaminants in rivers, lakes, and coastal environments. Examples in New England include:

Ashumet Pond, Cape Cod, Massachusetts	Evaluation of lake eutrophication
East Machias River, Maine	Model of fish hatchery discharge
Westfield River, Massachusetts	Model of paper mill discharge
Fort Point Channel, Boston, Massachusetts	Model of cooling water discharge
Spy Pond, Arlington, Massachusetts	Evaluation of lake eutrophication
Worcester, Massachusetts	Model of nonpoint source pollution and runoff

Peer Review

Dr. Shanahan is a past member of the editorial board of the journal of Ground Water and has served as a peer reviewer for numerous journals including Water Research, Water Science & Technology, the Journal of Hydrology, and the ASCE Journals of Environmental Engineering, Pipeline Engineering, and Water Resources Planning and Management. Dr. Shanahan has also participated as a member of peer review panels including the following:

U.S. EPA Superfund Program	Member of an expert panel convened by EPA to review a comprehensive model of PCBs in the Housatonic River.
South Florida Water Management District	Served on peer review panel for SFWMD providing independent evaluation of the South Palm Beach County and South Miami-Dade County ground-water models.
U.S. EPA Office of Research and Development	Served on peer review panel to evaluate proposals for a nonpoint-source pollution project
U.S. EPA Environmental Research Laboratory	Member of peer review panel for the U.S. EPA "Rates Manual" guidance document for water-quality modeling

Lawrence E. Band
Voit Gilmore Distinguished Professor
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Professional Preparation

S.U.N.Y. at Buffalo	Geography	1977 B.A.
University of California, Los Angeles	Geography	1979 M.A.
University of California, Los Angeles	Geography, Advisor: A.R. Orme	1983 Ph.D.

Appointments

2009- : Director, Institute for the Environment, University of North Carolina
2008-2009: Visiting Scientist, Bureau of Meteorology/CSIRO, Canberra, Australia
2002-2007: Chair, Department of Geography, U. North Carolina
1998- : Voit Gilmore Distinguished Prof. Geography, U. North Carolina
1994-1998: Professor, Dept. Geography, University of Toronto
1992-1993: Visiting Scientist, CSIRO, Canberra, Australia
1989-1994: Assoc. Prof., Dept. Geography, University of Toronto
1987-1989: Assist. Prof., Dept. Geography, University of Toronto
1983-1987: Assist. Prof., Dept. Geography and Geology, Hunter College/CUNY

Publications

1. Claessens, L., C. Tague, L. Band, P. Groffman and S. Kenworthy. 2009. Hydro-ecological linkages in urbanizing watersheds: An empirical assessment of in-stream nitrate loss and evidence of saturation kinetics. Journal of Geophysical Research - Biogeosciences. Res., 114, G04016, doi:10.1029/2009JG001017.
2. Shields, C., L.E. Band, N. Law, P. Groffman, S. Kaushal, K. Savvas, G. Fisher, K. Belt, 2008. Streamflow Distribution Of Non-Point Source Nitrogen Export From Urban-Rural Catchments tn the Chesapeake Bay Watershed. Water Resources Research, 44, W09416, doi:10.1029/2007WR006360.
3. S.S. Kaushal, P.M. Groffman, L.E. Band, C.A. Shields, R.P. Morgan, M.A. Palmer, K.N. Eshleman, K.T. Belt, C.M. Swan, S.E.G. Findlay, G.T. Fisher, 2008. Interaction between urbanization and climate variability amplifies watershed nitrate export in Maryland, USA. Env. Sci.&Tech. 42 (16), pp 5872–5878.
4. Smith, M., M. Cadenasso, W. Zhou, M. Grove, L.E. Band 2010. Evaulation of the NLCD for hydrologic applications in urban and suburban Baltimore, Maryland. Journal of the American Water Resources Association (JAWRA) 1-14. DOI: 10.1111⁄j.1752-1688. 2009.00412.x.
5. Kaushal, S., Groffman, P. Band, L, Elliott, E., Shields, C., Kendall, C., 2011. Tracking nonpoint source nitrogen pollution in human-impacted watersheds, Environmental Science and Technology, *45* (19), pp 8225–8232.

Additional 5 relevant publications

1. Hwang, T., L. Band, and T. C. Hales (2009), Ecosystem processes at the watershed scale: Extending optimality theory from plot to catchment, Water Resour. Res., 45, W11425, doi:10.1029/2009WR007775.
2. Pickett, S.T.A., M.L. Cadenasso, J.M. Grove, P.M. Groffman, L.E. Band, C.G. Boone, W.R. Burch Jr., C.S.B. Grimmond, J.Hom, J.C. Jenkins, N.L. Law, C.H. Nilon, R.V. Pouyat, K. Szlavecz, P.S. Warren, M.A. Wilson, 2008. Beyond Urban Legends: An Emerging Framework of Urban Ecology, as Illustrated by the Baltimore Ecosystem Study. Bioscience, v.58. p.139-150.
3. E.S. Bernhardt, L. E. Band, C. J. Walsh, and P.E. Berke, 2008. Understanding, managing, and minimizing urban impacts on surface water nitrogen loading. Annals of the New York Academy of Science, v. 1134, 61-96.

4. Song, C., G. Katul, R. Oren, L. E. Band, C. L. Tague, P. C. Stoy, and H. R. McCarthy (2009), Energy, water, and carbon fluxes in a loblolly pine stand: Results from uniform and gappy canopy models with comparisons to eddy flux data, J. Geophys. Res., 114,G04021, doi:10.1029/2009JG000951.
5. Hwang, T., C. Song, P.V. Bolstad 2011. Downscaling real-time vegetation dynamics by fusing multi-temporal MODIS and Landsat NDVI in topographically complex terrain. Remote Sensing of Environment, doi:10.1016/j/rse.2011.05.010.

Synergistic Activities (last 5 years):

1. Co-PI, Baltimore LTER (through 2011), Coweeta LTER (2008-2013), NC Triangle ULTRA-EX
2. Review Team – Everglades Land Model 2006, Chesapeake Bay Watershed Model 2005 & 2008
3. Consultant, California State Water Resources Control Board, Policy on in-stream flows, 2008 , EPA – Climate Change impacts on watersheds, 2007-2008, EPA - Review of the Environmental Science Division 2009, EPA - Review of Ecosystem Services; Reactive N plan 2009, EPA - Review of Ecosystem Services Modeling 2009-2010
4. CUAHSI. Board of Directors Chair Elect 2009, Chair 2010, Past Chair 2011
5. Member, North Carolina Nutrient Sensitive Waters Scientific Advisory Board 2011
6. Member, National Academy of Science – National Research Council Committee Member: Urban Stormwater Management in the United States, 2008; Integrated Observations for Hydrologic and Related Sciences, 2008; Land Use Change, current

Collaborators (last five years):
E. Bernhardt (Duke University), G. Brush (Johns Hopkins), M. Cadenasso (Yale), G. Characklis (UNC), L. Claessen (VTU), M. Doyle (UNC), M. Emch (UNC), C. Ford (US Forest Service), P. Groffman (Inst. Ecosys. Studies), T. Gragson (UGA), S. Grimmond (Indiana Univ.), M. Grove (US Forest Service), T.C. Hales (Cardiff), G. Katul (Duke University), S. Kaushal (U. Maryland), David Maidment (U. Texas), D. Nowak (US Forest Service), R. Oren (Duke University), S.T. Pickett (UNC), R. Pouyat (US Forest Service), M. Serre (UNC), C. Song (UNC), C..Tague (UCSB.), D. Urban (Duke University), J. Vose (US Forest Service), C. Walsh (U. Melbourne), J. Webster (VTU)

Thesis and post-doctoral advisees: completed – 9 MA, 13 Ph.D.; current –5 Ph.D., 1 PDF
MSc./MA:
University of Toronto - D. Scott Mackay, 1992; R. Lammers, 1992; R. Patterson,1991; David Baldwin, 1997; Anastasia Svirejeva, 1997
UNC Chapel Hill - Sandy Maunz, 2002, Katerina Savvas, 2010; Catherine Shields, 2008; Tamara Mittman, 2009

Phd:
University of Toronto - Axing Zhu, 1994; D. Scott Mackay; 1997; Richard Lammers, 1998; Irena Creed, 1998; Richard Fernandes;1998; Christina Tague, 1999; Tongzhin Zhu; 1998

UNC Chapel Hill - Neely Law, 2004; Laura Jackson; 2005; David Tenenbaum; 2005; Daehyok Shin; 2006; Taehee Hwang, 2010; Monica Smith, 2010

***Current Ph.D.*.:** Antoine Randolph, Yuri Kim, Jon Duncan, Brian Miles, Limei Ran

Post-doctoral students: Soren E. Brun, Christina E. Tague, Stephen Kenworthy, Christopher Kees, Sdhyok Shin, TC Hales, J.R. Rigby, S. Wu, T. Hwang

BILLY H. JOHNSON, PHD, PE, D.WRE

COMPUTATIONAL HYDRAULICS & TRANPORT LLC

Dr. Billy Johnson has over 40 years of experience in developing and applying numerical models to address various types of water resources problems. He retired from the US Army Engineer Waterways Experiment Station (WES) in 2001 after more than 30 years of service. During his 30 years at WES, he was involved in developing models such as FLOWSED (1D unsteady river model), LAEMSED (2D laterally-averaged flow and sediment transport), CH3D (curvilinear hydrodynamics in three-dimensions), STFATE (short-term fate of dredged material disposed in open water), and SSFATE (suspended sediment fate of material re-suspended at the dredging site). He introduced the concept of generalized curvilinear coordinates in numerical hydrodynamic modeling in the late 1970's – early 1980's. His expertise in the development and application of advanced numerical modeling tools is reflected through the many awards he received during his federal career. These include WES Engineer of the Year 1989, Mississippi ASCE Hydraulic Engineer of the Year 1990, Department of the Army Research and Development Award 1991, Associate Editor of ASCE Journal of Hydraulic Engineering 1993-97, and the EPA National Science Award in Water Quality Modeling 1999. In 2005 he was inducted into the WES Gallery of Distinguished Employees and into the American Academy of Water Resource Engineers as a Diplomate He is the author of over 100 technical reports, conference proceedings, book chapters, and journal articles.

EDUCATION

Bachelor of Science, Aerospace Engineering, MS State University, 1967
Master of Science, Aerospace Engineering, MS State University, 1968
Doctor of Philosophy, General Engineering, MS State University, 1971

REGISTRATIONS

Registered Professional Civil Engineer: No. 12078 MS 1992
Diplomate ASCE Water Resources Engineering 2005

PROFESSIONAL ORGANIZATIONS

ASCE

SELECTED PROJECT EXPERIENCE

Technical Review

Technical review of the hydrodynamic component of various studies has been performed. Examples include several minimum flow and levels studies for rivers located in the Southwest Florida Water Management District, water quality modeling of Florida Bay for the South Florida Water Management District, and water quality modeling of Indian River for the St. John's Water Management District.

Development of 3D Hydrodynamic Model of Chesapeake Bay

A 3D hydrodynamic model of Chesapeake Bay to provide flow fields to the 3D CEQUAL-ICM water quality model was developed. The development of the Z-plane version of CH3D was required for the computation of a flow field that remained stratified in the navigation channel. The initial study was completed in 1991, with further enhancements completed in 1997, and 2002.

C&D Deepening Study

A 3D model of the Upper Chesapeake Bay, Chesapeake and Delaware Canal, and the Delaware River and Bay was developed to determine the impact on salinity, water supplies, and circulation due to deepening the C&D Canal and navigation channels in the two bays. The Z-plane version of the model called CH3D was employed. Year long simulations that covered a wide range of hydrologic conditions were made with and without channel deepening.

Site 104 Sedimentation and Water Quality Study

A numerical model study to determine the fate of material proposed for disposal at Site 104 in the Upper Chesapeake Bay, as well as, the impact of the disposed material on the water quality of the surrounding waters was conducted. A 3D numerical hydrodynamic model (CH3D) was employed to compute shear stresses for input to a dredged material mound-building model called MDFATE. In addition, the 3D hydrodynamic model provided flow fields to the 3D CEQUAL-ICM water quality model.

Freshwater Bayou Channel Deepening Study

This was a numerical modeling study for the New Orleans District of the US Army Corps of Engineers to determine the impact of channel deepening on shoaling and bank erosion in the Freshwater Bayou navigation channel located to the west of the Vermilion Bay in Louisiana. With the sediment being primarily fine-grained fluid mud, the CH3DZ-FM model previously developed to investigate sedimentation in the Atchafalaya Bar channel was applied. Plans call for deepening the existing 12 ft channel to 16 ft. An interesting feature is a salinity barrier lock located near the Gulf end of Freshwater Bayou. For the September 2001 – August 2002 time period, the lock was closed about 85% of the time. Thus, the Freshwater Bayou bar channel is effectively disconnected from the GIWW and Vermilion Bay most of the time.

Terrebonne Marsh Salinity Study

This numerical model study was conducted for the Vicksburg District of the US Army Corps of Engineers. The three dimensional numerical model known as CH3DZ was applied to assess the impact on salinity in the Terrebonne Marsh as a result of implementing channel deepening in the Atchafalaya River and Bayous Chene, Boeuf, and Black. Year-long simulations were made for an average flow year and a low flow year with and without channel deepening. In addition, simulations were made with 50-year future bathymetry. Salinity contour plots were generated to illustrate the impact of channel deepening in the marsh and navigation channels.

W. JUDSON KENWORTHY, PhD

EDUCATION: PhD Zoology, North Carolina State University, 1992
MSc Environmental Science, University of Virginia, 1981
BS Resource Development, University of Rhode Island, 1976

ADDRESS: 109 Holly Ln
Beaufort, NC 28516

Phone: 252-646-9174 (mobile)

EMPLOYMENT:

Current; Sub-contractor for Industrial Economics, Incorporated, Cambridge, MA. I serve as an expert consultant representing the National Oceanic and Atmospheric Administration (NOAA) on the Federal and State Trustee Submerged Aquatic Vegetation Technical Working Group (SAVTWG) assessing the impact of the Gulf of Mexico oil spill on seagrasses.

1979-2011; Research Fisheries Biologist, Center for Coastal Fisheries and Habitat Research, National Centers for Coastal Ocean Science, National Ocean Service, National Oceanic and Atmospheric Administration, U.S. Dept. of Commerce, 101 Pivers Island Road, Beaufort, North Carolina, 28516. Served as a research project leader for laboratory, experimental and field ecology research programs including directing basic and applied research on the structure, function, and dynamics of coastal and estuarine ecosystems with special emphasis on marine seagrasses, coral reefs, benthic habitat utilization by marine organisms, environmental factors controlling the distribution, abundance and population dynamics of these communities and restoration. My research directly addressed critical management issues in the coastal zone of the United States and internationally. Daily, I worked on scientific teams dealing with the conservation and restoration of coastal ecosystems providing expert consultation and scientific research results in support of local, state, federal and international resource management programs conserving and restoring marine resources.

2006 –present; Member of the Scientific Technical Advisory Committee for the Albemarle-Pamlico National Estuary Program (APNEP). I provide scientific and technical guidance to the Albemarle-Pamlico National Estuary Program on restoration and protection natural resources in the Albemarle-Pamlico region of North Carolina.

Nov. 1999 – 2011; Adjunct faculty appointment, Department of Biological Sciences, University of North Carolina Wilmington, N.C. I mentor graduate students and serve on graduate student committees.

RELATED ACTIVITIES;

December, 2010; Conducted seagrass restoration workshop for Qatar Ministry of the Environment, Doha, Qatar.

November, 1990; Planned, organized and implement a national workshop to examine the capability of water quality criteria, standards and monitoring programs to protect seagrasses. This workshop led to development of a number of research and management programs intended for the protection of seagrasses throughout the United States.

RELEVENT PUBLICATIONS:

Uhrin, A.V., Kenworthy, W.J, and M.S. Fonseca, M.S. 2011. Understanding uncertainty in seagrass injury recovery: an information-theoretic approach. *Ecological Applications.* 21(4):1365-1379

Costello, C, Kenworthy, W.J. 2011. Twelve-year mapping and change analysis of eelgrass (*Zostera marina*) areal abundance in Massachusetts (USA) identifies Statewide Declines. Estuaries and Coasts. 34:232-242.
Waycott, M., C.M. Duarte, T.J.B. Carruthers, R.G. Orth, W.C. Dennison, S. Olyarnik, A. Calladine, J.W. Fourqurean, K.A. Heck, Jr., A.R. Hughes, G.A. Kendrick, W. J. Kenworthy, F.T. Short, and S.L. Williams. 2009.

Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceedings of the National Academy of Sciences 106:12377-12381.

Biber, Patrick D, W. Judson Kenworthy, and Hans W. Paerl. 2009. Experimental analysis of the response and recovery of *Zostera marina* (L.) and *Halodule wrightii* (Ascher.) to repeated light-limitation stress. 2009. Journal of Experimental Marine Biology and Ecology. 369:110-117.

Biber, P., Gallegos, C, Kenworthy, W. J. 2008. Calibration of a bio-optical model in the North River, North Carolina (Albemarle-Pamlico Sound): A tool to evaluate water quality impact on seagrasses. Estuaries and Coasts 31:177-191.

Orth, Robert J., Tim J.B. Carruthers, William C. Dennison, Carlos M. Duarte, James W. Fourqurean, Kenneth L. Heck, Jr., A Randall Hughes, Gary A. Hendrick, W. Judson Kenworthy, Suzanne Olyarnik, Fred T. Short, Michelle Waycott, and Susan L. Williams. 2006. A global crisis for seagrass ecosystems. Bioscience 56:987-996.

Biber, P.D., H. W. Paerl, C. L. Gallegos, and W. Judson Kenworthy. 2004. Evaluating Indicators of Seagrass Stress to Light. Pages 193-209 in S.A. Bortone (Ed.), Estuarine Indicators. CRC Press, Boca Raton.

Fonseca, Mark S., Whitfield, Paula E., Kenworthy, W. Judson, Colby, David, R., Julius, Brian E. 2004. Use of two spatially explicit models to determine the effect of injury geometry on natural resource recovery. Aquatic Conservation: Marine and Freshwater Ecosystems 14:281-298

Fonseca, M.S., W.J. Kenworthy, B.E. Julius, S. Shutler, and S. Fluke. 2002. Seagrasses, pp. 149-770 In M. R. Perrow and A.J. Davy (eds.), Handbook of Ecological Restoration. University Press, Cambridge.

Fonseca, M.S., B.E. Julius, W. Judson Kenworthy. 2000. Integrating biology and economics in seagrass restoration: How much is enough and why? Ecological Engineering 15:227-237.

Fonseca, Mark S., W. Judson Kenworthy, and Gordon W. Thayer. 1998. Guidelines for the Conservation and Restoration of Seagrasses in the United States and Adjacent Waters. NOAA, Coastal Ocean Program, Decision Analysis Series No. 12. U.S. Department of Commerce, NOAA, Coastal Ocean Office, Silver Spring, MD. 222pp.

Gallegos, C.L. and W.J. Kenworthy. 1996. Seagrass depth limits in the Indian River Lagoon (Florida, U.S.A.): Application of an optical water quality model. Estuarine and Coastal Shelf Science. 42:267-288.

Fonseca, M.S., W.J. Kenworthy, F.X. Courtney, and M.O. Hall. 1994. Seagrass transplanting in the southeastern United States: methods for accelerating habitat development. Restoration Ecology 2:198-212.

Kenworthy, W.J. and D.E. Haunert. 1991. The light requirements of seagrasses. Proceedings of a workshop to examine the capabilities of water quality criteria, standards, and monitoring programs to protect seagrasses. National Oceanic and Atmospheric Administration Technical Memorandum NMFS-SEFC-287.

Thayer, Gordon W., W. Judson Kenworthy, and Mark S. Fonseca. 1984. The ecology of eelgrass meadows of the Atlantic coast: a community profile. U.S. Fish and Wildl. Serv. FWS/OBS-84/02. 147pp.

Kenworthy, W.J., J.C. Zieman, and G.W. Thayer. 1982. Evidence for the influence of seagrasses on the benthic nitrogen cycle in a coastal plain estuary near Beaufort, North Carolina (USA). Oecologia. 54:152-158.

Kenworthy, W.J. and M.S. Fonseca. 1977. Reciprocal transplant of the seagrass, *Zostera marina* L. effect of substrate on growth. Aquaculture 12:197-213.

RESUME: **PAUL E. STACEY**
Great Bay National Estuarine Research Reserve
225 Main Street
Durham, NH 03824
paul.stacey@wildlife.nh.gov

EXPERIENCE:

- Research Coordinator for the Great Bay National Estuarine Research Reserve – Research and Monitoring activities related to understanding and managing Great Bay and its watershed
- Division Director of watershed and estuary research, monitoring and management programs, including: Surface and Groundwater Standards and Criteria; Monitoring and CWA 305(b)/303(d) Assessment; Total Maximum Daily Loads (TMDL); Watershed Management; Aquifer Protection; Low Impact Development (LID); and Municipal Wastewater Facility Regulation and Financing
- Water quality monitoring and management programs with focus on hypoxia and nutrient enrichment, including TMDL development and application
- Development and implementation of CT's award-winning nitrogen trading program
- Nonpoint source and stormwater runoff assessment and management, including collaborative research, implementation of best management practices, LID and project grants
- Multimedia source assessment, criteria development and management of nutrients incorporating watersheds and airsheds
- Oversight of Connecticut's Clean Water Fund for municipal wastewater infrastructure financing
- Climate Change assessment and adaptation planning from an ecosystem-based management perspective

EMPLOYMENT:

- ❖ **2011-Present: New Hampshire Fish & Game, Great Bay National Estuarine Research Reserve**
  Research Coordinator
- ❖ **1985 – 2011: State of Connecticut, Department of Environmental Protection**
  Director of Planning & Standards, Bureau of Water Protection and Land Reuse Since 2006
- ❖ **1977 – 1985: Academy of Natural Sciences, Philadelphia, PA**
  Fisheries Biologist/Aquatic Ecology

EDUCATION:

- ❖ **Colorado State University,** Fort Collins, Colorado
  Master of Science in Fisheries Biology
- ❖ **Utah State University,** Logan, Utah
  Bachelor of Science in Wildlife and Fisheries Biology
- ❖ **College of the Holy Cross,** Worcester Massachusetts
  Bachelor of Arts in Psychology

COMMITTEES AND PROJECTS:

- ❖ **2011-Present** Water Environment Federation Watershed Management Committee
- ❖ **2009-2011** The National Academies National Research Council, Chesapeake Bay Program Nutrient Reduction Evaluation Committee
- ❖ **2009-2011** CT Governor's Committee on Climate Change, Subcommittee on Climate Change Adaptation – Infrastructure Work Group co-chair
- ❖ **2009-2011** Water Quality Standards Forum – Sponsored by EPA through the Water Environment Research Foundation
- ❖ **2006-2011** Association of State and Interstate Water Pollution Control Administrators – Nutrients and Climate Change work groups
- ❖ **2006-2011** New England Interstate Water Pollution Control Commission

- ❖ **2005-2011** EPA's Science Advisory Board Integrated Nitrogen Committee
- ❖ **2005-2006** Project Steering Committee on "Bioassessment: A Tool for Managing Urban Aquatic Life Uses" for the Water Environment Research Foundation
- ❖ **2004-2005** Co-lead, U.S. EPA, Multimedia Assessment Technical Focus Group (ad hoc), Science and Technology Group, Clean Air Act Advisory Committee
- ❖ **2004-2006** Technical Review Committee for Monitoring Plan for Water Quality and Ecology for Portuguese Transitional and Coastal Waters required by the European Union Water Framework Directive
- ❖ **2004-2005** Project Steering Committee on "Comparing Economics of Nitrogen Farming with Traditional Removal" for the Water Environment Research Foundation
- ❖ **2003-2011** Statutory Representative/Commissioner Alternate, Interstate Environmental Commission, CT-NY-NJ
- ❖ **2003-2005** Project Steering Committee on "Watershed-based Trading: A Guide for the Wastewater Community" for the Water Environment Research Foundation
- ❖ **2003-2005** Dissertation Committee for Ph.D. candidate, University of Connecticut, on "Rain Garden Design and Function"
- ❖ **2002-2004** Steering Committee for EPA-Sponsored "Northeast Coastal Indicators Workshop" in Durham, NH, January 2004
- ❖ **2002-2011** EPA Long Island Sound Study Scientific and Technical Advisory Committee
- ❖ **2001-2003** Advisory Committee on "Status and Effects of Nitrogen Pollution in the Northeast United States" for the Hubbard Brook Research Foundation
- ❖ **2000-2005** Project Steering Committee on "Technical Approaches for Setting Site-Specific Nutrient Criteria" for the Water Environment Research Foundation
- ❖ **1999-2006** Acid Rain Steering Committee of the New England Governors and Eastern Canadian Premiers
- ❖ **1999-Present** Regional Technical Advisory Group for EPA Nutrient Criteria Workgroup coordinated by New England Interstate Water Pollution Control Commission

AWARDS:

- ❖ **2011** – Paul Eastman Environmental Statesman Award – Highest Honor of the Association of State and Interstate Water Pollution Control Administrators (now "ACWA")
- ❖ **2010** – Estuarine Research Federation – Recognition for Outstanding Contributions to Manuscript Review
- ❖ **2010** – (Group Award) Outstanding Achievement Award for nitrogen Trading from the New England Water Environment Association
- ❖ **2007** – (Group Award) EPA Blue Ribbon for Excellence in Water Quality Trading
- ❖ **2003** – Curtis and Edith Munson Distinguished Lecturer (Yale University) on Long Island Sound Ecosystems
- ❖ **1998** – Recognition of outstanding participation, National Engineers Week
- ❖ **1994** – Letter of Recognition from Governor Lowell Weicker for efforts to clean up Long Island Sound
- ❖ **1994** – CTDEP Distinguished Service Award

AFFILIATIONS:

- Association Clean Water Administrators (ACWA)
- Coastal and Estuarine Research Federation (CERF)
- Water Environment Federation (WEF)
- New England Water Environment Association (NEWEA)
- New England Estuarine Research Society (NEERS)
- Union of Concerned Scientists (UCS)