United States
Environmental Protection
Agency

Waquoit Bay Watershed Ecological Risk Assessment: The effect of land-derived nitrogen loads on estuarine eutrophication

EPA/600/R-02/079
October 2002

Waquoit Bay Watershed Ecological Risk Assessment: The effect of land-derived nitrogen loads on estuarine eutrophication

U.S. Environmental Protection Agency
National Center for Environmental Assessment–Washington Office
Office of Research and Development
Washington, DC

DISCLAIMER

ABSTRACT

A watershed ecological risk assessment of Waquoit Bay, located on the south coast of Cape Cod, MA, was performed for managers to better understand the environmental impacts of human activities. An interdisciplinary and interagency workgroup identified all the stressors of concern (chemical pollution, pathogens, altered freshwater flow, nutrient enrichment/eutrophication, physical alteration of habitat, and fishing/shellfishing) and selected assessment endpoints (estuarine percent eelgrass cover, finfish diversity and abundance, scallop abundance, anadromous fish reproduction, wetland bird and piping plover habitat distribution and abundance, and tissue contamination of fish and shellfish). The workgroup later decided to focus on nutrient enrichment and its impacts on percent eelgrass cover and scallop abundance.

A nitrogen loading model (NLM) was used to estimate the amount and sources of nitrogen entering the watershed, and an estuarine loading model (ELM) was used to estimate the nitrogen available to producers in shallow estuaries. The NLM indicates that atmospheric deposition is the largest source of nitrogen, but because more atmospheric nitrogen than wastewater nitrogen is intercepted in the watershed, wastewater becomes the largest contributor of nitrogen reaching the bay. By comparing increases in nitrogen loads to losses in the area of eelgrass cover over the last 60 years, it appears that eelgrass disappears once nitrogen loads reach 20 kg/ha/yr. Both the increase in nitrogen load and the decrease in eelgrass can be correlated to decreases in the annual harvest of scallops. The models provide the opportunity for managers to assess a variety of options to reduce nitrogen loads to their estuaries and to achieve the loads that could allow the return of eelgrass to the target area.

Preferred citation:
U.S. EPA (Environmental Protection Agency). (2002) Waquoit Bay watershed ecological risk assessment. National Center for Environmental Assessment, Washington, DC; EPA/600/R-02/079. Available from: National Technical Information Service, Springfield, VA; PB2003-102013 and <http://www.epa.gov/ncea>.

CONTENTS

LIST OF TABLES . . . v
LIST OF FIGURES . . . vi
LIST OF ACRONYMS . . . viii
FOREWORD . . . x
PREFACE . . . xi
AUTHORS, CONTRIBUTORS, AND REVIEWERS . . . xii

1. EXECUTIVE SUMMARY . . . 1-1
   1.1. ECOLOGICAL RISK ASSESSMENT . . . 1-1
   1.2. PLANNING AND PROBLEM FORMULATION . . . 1-2
      1.2.1. Management Goal and Objectives . . . 1-2
      1.2.2. Stressors . . . 1-3
      1.2.3. Assessment Endpoints . . . 1-3
      1.2.4. Conceptual Model . . . 1-3
      1.2.5. Analysis Plan . . . 1-4
   1.3. RISK ANALYSIS . . . 1-4
   1.4. RISK CHARACTERIZATION . . . 1-5
   1.5. MANAGEMENT IMPLICATIONS . . . 1-6

2. INTRODUCTION . . . 2-1
   2.1. THE WATERSHED . . . 2-1
   2.2. THE WATERSHED ECOLOGICAL RISK ASSESSMENT PROCESS . . . 2-8

3. PLANNING FOR THE ECOLOGICAL RISK ASSESSMENT . . . 3-1

4. PROBLEM FORMULATION . . . 4-1
   4.1. CONCEPTUAL MODEL . . . 4-1
      4.1.1. Sources of Stressors . . . 4-1
      4.1.2. Selected Stressors . . . 4-3
   4.2. ASSESSMENT ENDPOINTS . . . 4-4
   4.3. ANALYSIS PLAN . . . 4-6
      4.3.1. Comparative Risk Ranking . . . 4-6
      4.3.2. Relationships Between Stressors and Ecological Responses . . . 4-8
         4.3.2.1. The Importance of Stressors in Waquoit Bay . . . 4-9
         4.3.2.2. Focus on Nutrient Enrichment as the Dominant Stressor . . . 4-13
      4.3.3. Summary of the Analysis Plan . . . 4-13

5. RISK ANALYSIS . . . 5-1
   5.1. EXPOSURE ANALYSIS . . . 5-1
      5.1.1. Nitrogen Loads . . . 5-1
         5.1.1.1. Estimates Using the Nitrogen Loading Model (NLM) . . . 5-1
         5.1.1.2. Measurements . . . 5-7
         5.1.1.3. NLM Validation . . . 5-7
         5.1.1.4. NLM Uncertainty . . . 5-10
   5.2. EFFECTS ANALYSIS . . . 5-12
      5.2.1. Cascade of Effects on Ecosystem Components . . . 5-13
         5.2.1.1. Nitrogen Concentrations . . . 5-13
         5.2.1.2. Phytoplankton Biomass and Production . . . 5-17
         5.2.1.3. Macroalgae Biomass and Production . . . 5-17

CONTENTS (continued)

5.2.1.4. Eelgrass Biomass and Production 5-17
5.2.1.5. Combined Effect of Nitrogen Loading on Primary Producers 5-19
5.2.1.6. Zooplankton Egg Production 5-20
5.2.1.7. Shellfish Growth Rates 5-20
5.2.1.8. Finfish Abundance 5-20
5.2.1.9. Summary of Effects on Ecosystem Components 5-20
5.2.2. Effects on Assessment Endpoints 5-23
5.2.2.1. Percent Eelgrass Cover 5-23
5.2.2.2. Scallop Harvest 5-23
5.2.2.3. Summary of Effects on Assessment Endpoints 5-23

6. RISK CHARACTERIZATION 6-1
6.1. TEMPORAL CHANGES IN EXPOSURE AND EFFECTS 6-1
6.1.1. Changes in Exposure: Back-Casting Nitrogen Loads 6-1
6.1.2. Temporal Changes in Effects: Impact on Percent Eelgrass Cover and Scallop Harvest 6-5
6.1.3. Effects of Other Stressors on Eelgrass 6-7

7. MANAGEMENT IMPLICATIONS 7-1
7.1. ASSESSMENT OF RESTORATION MEASURES 7-1
7.1.1. Reducing Fertilizer Application Rates 7-4
7.1.2. Managing Wastewater 7-4
7.2. USING THE ECOLOGICAL RISK ASSESSMENT TO CONVERT SCIENCE TO MANAGEMENT ADVICE: CAVEATS AND LESSONS LEARNED 7-9

APPENDIX A. Supplemental Information on the Waquoit Bay Estuarine Complex A-1
A.1 Geological and Hydrological Characteristics A-1
A.2 Biological Characteristics A-2

APPENDIX B. Organizations Concerned About the Waquoit Bay B-1

APPENDIX C. Public Concerns and Waquoit Bay Stressors C-1
C.1 Public Concerns C-1
C.2 Sources and Stressors in the Waquoit Bay Watershed C-2

APPENDIX D. Attendees at the Waquoit Bay Management Goals Meeting D-1

APPENDIX E. Information on Contamination from the Massachusetts Military Reservation E-1
E.1 Phosphorus Loading to Ashumet Pond E-1
E.2 Volatile Organic Compound (VOC) Contamination in Plumes E-3

REFERENCES R-1

LIST OF TABLES

3-1. The Waquoit Bay watershed management objectives 3-2

4-1. Relationship between assessment endpoints and management objectives 4-5

4-2. Effects matrix for the Waquoit Bay watershed 4-7

4-3. Relative importance of identified stressors to the Waquoit Bay ecosystem 4-8

5-1. Inputs, losses, and default terms used by the nitrogen loading model (NLM) 5-3

5-2. Measured nitrogen loads to the subestuaries of Waquoit Bay 5-8

5-3. Sources and $\delta^{15}N$ values of nitrate in groundwater 5-10

5-4. Error analysis of NLM variables 5-12

5-5. Characteristics of subestuaries of the Waquoit Bay watershed 5-14

6-1. Relative contribution of each of the major sources of nitrogen to the Waquoit Bay estuary in 1938 and 1990 6-5

7-1. Onsite septic system retention efficiencies reported for various alternative systems . . . 7-8

7-2. Changes in water residence time predicted by dredging simulations 7-9

LIST OF FIGURES

2-1. Delineation of the watershed of Waquoit Bay, MA 2-2

2-2. Aerial photographs of Waquoit Bay in 1938 (top) and 1990 (bottom) 2-4

2-3. Number of parcels that are built upon (black) and remain to be built (white) as of 1990 in the Childs River watershed of Waquoit Bay, MA 2-5

2-4. Area of eelgrass in Waquoit Bay between 1951 and 1992 2-6

2-5. Framework for the ecological risk assessment 2-9

4-1. Conceptual model of the Waquoit Bay watershed ecological risk assessment 4-2

5-1. Schematic of the nitrogen loading model 5-2

5-2. Measured versus modeled nitrogen (N) loads to Waquoit Bay 5-9

5-3. Values of $\delta^{15}N$ of nitrate in groundwater versus percent of nitrogen (N) from wastewater 5-11

5-4. Dissolved inorganic nitrogen (DIN) concentrations in the three subestuaries of Waquoit Bay subject to land-derived nitrogen (N) loads 5-14

5-5. Schematic of inputs and exports of the ELM and the NLM 5-15

5-6. Comparison of dissolved inorganic nitrogen (DIN) concentrations predicted by the estuarine loading model (ELM) with measured concentrations in the water column of several Waquoit Bay estuaries (p<0.01) 5-16

5-7. Effects of nitrogen (N) loading on biomass and primary production of phytoplankton, macroalgae, and eelgrass in Sage Lot Pond, Quashnet River, and Childs River (top to bottom) 5-18

5-8. Partition of total primary production in shallow estuaries into contributions by phytoplankton, macroalgae, and seagrasses, all plotted against measured annual nitrogen (N) load 5-19

5-9. Characterization of *Acartia tonsa* in Waquoit Bay estuaries 5-21

5-10. The effects of differences in nitrogen (N) concentration on the growth rates of softshell clams (top) and quahogs (bottom) 5-22

5-11. Percent seagrass cover lost as nitrogen (N) load increases for a multitude of temperate and tropical ecosystems 5-24

5-12. Volume of bay scallop harvest in Waquoit Bay from 1965-1995 5-24

6-1. Changes in land uses in the Waquoit Bay watershed from 1938 to 1990 6-2

LIST OF FIGURES (continued)

6-2. Historical changes in nitrogen (N) loading to the watershed and estuary of Waquoit Bay 6-4

6-3. Decreases in area of eelgrass and volume of scallops harvested over time as a function of increasing nitrogen (N) loads 6-6

7-1. Relationship between nitrogen (N) loads, dissolved inorganic nitrogen (DIN) concentration, and percent eelgrass cover 7-3

7-2. Historical changes in nitrogen (N) loading predicted by the nitrogen loading model 7-4

7-3. Reduction in the total nitrogen (N) load that would result from varying the amount of fertilizers used in the Waquoit Bay watershed 7-5

7-4. Reduction in total nitrogen (N) load in the Waquoit Bay watershed (and corresponding year) that would result from implementing various wastewater treatment systems . . . 7-7

E-1. Map of the plumes emanating from the Massachusetts military reservation E-4

LIST OF ACRONYMS

AFCEE	Air Force Center for Environmental Excellence
DIN	Dissolved Inorganic Nitrogen
DON	Dissolved Organic Nitrogen
ELM	Estuarine Loading Model
EPA	U. S. Environmental Protection Agency
GIS	Geographic Information System(s)
gpm	gallons per minute
IRP	Installation Restoration Program
L	Landings
MMR	Massachusetts Military Reservation
N	Nitrogen
N_2	Atmospheric Nitrogen
NCEA	National Center for Environmental Assessment
NGOs	Nongovernmental Organizations
NH_4	Ammonium
NLM	Nitrogen Loading Model
NMFS	National Marine Fisheries Service
NO_3	Nitrate
NOAA	National Oceanic and Atmospheric Administration
ppb	parts per billion
RDX	Royal Dutch Explosive
T_r	Residence time
TCE	Trichloroethylene
TDN	Total Dissolved Nitrogen
USGS	U.S. Geological Survey

LIST OF ACRONYMS (continued)

VOC	Volatile Organic Compound
WBLMER	Waquoit Bay Land Margin Ecosystems Research Project
WBNERR	Waquoit Bay National Estuarine Research Reserve

FOREWORD

Risk assessment plays an increasingly important role in determining environmental policies and decisions at the U.S. Environmental Protection Agency (EPA). In 1998, EPA published *Guidelines for Ecological Risk Assessment* to provide a broad framework applicable to a range of environmental problems associated with chemical, physical, and biological stressors. As ecological risk assessment evolves, it is moving beyond a focus on assessing effects of simple chemical toxicity on single species to the cumulative impacts of multiple interacting chemical, physical, and biological stressors on populations, communities, and ecosystems. Although EPA has considerable experience in applying the ecological risk assessment paradigm in source-based approaches, such as those focused on particular chemicals, specific guidance on place-based approaches (e.g., watersheds and regions) is still limited. This assessment of the Waquoit Bay watershed was completed to address a specific environmental problem through application of the risk assessment methods represented in the Guidelines. Through this assessment, and other watershed scale assessments like it, the Office of Research and Development is learning how to develop new tools and approaches to support local environmental decision makers. An important component of these approaches is active participation by local stakeholders. The Waquoit Bay watershed assessment provides a good example of partnering between government, environmental organizations, and others to support environmental decision making with strong science.

Waquoit Bay was selected because its watershed contains valued and threatened ecological resources; there was an abundance of previously collected stressor and effects data; it is subjected to multiple physical, chemical, and biological stressors; and it has a number of organizations working to protect the ecological resources. This assessment is intended to address such concerns by analyzing stressors and resulting ecological effects and by stimulating broader public awareness and participation in decision making for reducing ecological risks. This watershed assessment report serves as an example for others to follow on how to use ecological risk assessment principles in a watershed-scale assessment to improve the use of science in decision making.

Michael Slimak
Associate Director of Ecology
National Center for Environmental Assessment
U.S. EPA, Office of Research and Development

PREFACE

The National Center for Environmental Assessment–Washington Office (NCEA–W), National Oceanic and Atmospheric Administration–National Marine Fisheries Service, EPA Region I, Boston University Marine Program, and other organizations developed this watershed ecological risk assessment to help protect the Waquoit Bay watershed. The document has three purposes: (1) to provide information to help make more informed decisions on how to protect the valued ecological resources of the watershed; (2) to provide data and references for future research in the watershed; and (3) to demonstrate the benefits of applying ecological risk assessment at the watershed scale. The report is based on the *Guidelines for Ecological Risk Assessment* and advice and support from NCEA, while exercising the necessary flexibility to implement the risk assessment approach at the watershed scale. To serve as an example for others seeking to increase the use of science in place-based decision making, the document includes brief descriptions of the process the workgroup followed along with the major analyses performed. The literature search supporting the document was completed in May 2000.

A more concise report of the assessment's findings and methods can be found in Serveiss et al. (submitted). Lessons learned about applying ecological risk assessment to the watershed scale, including those acquired from this assessment, are described in Serveiss et al. (2000) and Serveiss (2002). Diamond and Serveiss (2001) provides another example of an EPA-sponsored ecological risk assessment. Discussion on how ecological risk assessment principles can be applied at an even larger spatial scale (e.g., a region) can be found in Landis and Wiegers (1997) and Wiegers et al. (1998).

AUTHORS, CONTRIBUTORS, AND REVIEWERS

The National Center for Environmental Assessment–Washington Office (NCEA–W) within the EPA's Office of Research and Development was responsible for preparing this document. Draft reports were prepared by Boston University's Marine Program (BUMP) at the Marine Biological Laboratory (MBL) under cooperative agreement No. CR 825851-01-0 and Purchase Order No. 1W-0332-NAEX with input from the other authors and workgroup members. The Waquoit Bay risk assessment was prepared by a diverse group of people representing organizations and agencies interested in the management and protection of the biota of the Waquoit Bay watershed. Much of the data used in the risk analysis were collected and analyzed as part of the Waquoit Bay Land Margin Ecosystems Research project, which was funded by a grant from the National Science Foundation's Land Margin Ecosystems Research initiative, by EPA's Region I, and by the Northeast Fisheries Science Center (NEFSC) of the National Oceanic and Atmospheric Administration's (NOAA's) Sanctuaries and Reserves Division. The analysis itself was performed at the Boston University Marine Program with input from the NEFSC of NOAA's National Marine Fisheries Service (NMFS).

In a project that has lasted almost a decade, it is difficult to acknowledge all of the key players, but the following individuals helped bring this endeavor to fruition: EPA project managers through problem formulation (Suzanne Marcy and John Miller); Workgroup Chairs (Maggie Geist and Patti Tyler); MBL/BUMP scientists (Jennifer L. Bowen and Ivan Valiela); Terri Konoza (EPA/NCEA–W) who managed the document production activities and provided editing and word processing support; Leela Rao (EPA/NCEA–W) who helped revise the final report; and mostly David Dow (NOAA/NMFS), who provided continuity and historical perspective between the Workgroup Problem Formulation report and the Risk Analysis report.

EPA Project Officer:
Victor B. Serveiss, U.S. EPA, NCEA–W, Washington, DC

Authors:
Jennifer L. Bowen, Boston University Marine Program, Marine Biological Laboratory, Woods Hole, MA
David Dow, Northeast Fisheries Science Center, National Marine Fisheries Service, Woods Hole, MA
Victor B. Serveiss, U.S. EPA, NCEA–W, Washington, DC
Ivan Valiela, Boston University Marine Program, Marine Biological Laboratory, Woods Hole, MA
Leela Rao, U.S. EPA, NCEA–W, Washington, DC

Contributors:
Maggie Geist, Association for the Preservation of Cape Cod, Orleans, MA, formerly with the

AUTHORS, CONTRIBUTORS, AND REVIEWERS (continued)

Waquoit Bay National Estuarine Research Reserve, Waquoit, MA
Patti Tyler, EPA Region VIII, Denver, CO, formerly with EPA Region I, Lexington, MA
Suzanne Marcy, U.S. EPA, NCEA-IO, Anchorage, AK

EPA's Ecological Risk Assessment Co-chairs:
Maggie Geist, Association for the Preservation of Cape Cod, Orleans, MA, formerly with the Waquoit Bay National Estuarine Research Reserve, Waquoit, MA
Patti Tyler, EPA Region VIII, Denver, CO, formerly with EPA Region I, Lexington, MA

Other Former Workgroup Members:
Vicki Atwell, formerly with U.S. EPA, Office of Research and Development, Washington, DC
Jeroen Gerritsen, Tetra Tech, Inc., Owings Mills, MD
John Miller, U.S. EPA, Office of Water, Washington, DC
Conchi Rodriguez, formerly with EPA, Office of Prevention, Pesticides, and Toxic Substances, Washington, DC
Chuck Spooner, U.S. EPA, Office of Water, Washington, DC

EPA Reviewers:
James Andreasen, U.S. EPA, NCEA–W, Washington, DC
Patricia Cirone, U.S. EPA, Region X, Seattle, WA
Brian Hill, U.S. EPA, Office of Research and Development, Duluth, MN
Lester Yuan, U.S. EPA, NCEA–W, Washington, DC
Susan Norton, U.S. EPA, NCEA–W, Washington, DC

Other Reviewers:
Peter DeFur, Virginia Commonwealth University, Richmond, VA
Wayne Landis, Western Washington University, Bellingham, WA
Kenneth Foreman, The Ecosystems Center, Marine Biological Laboratory, Woods Hole, MA

1. EXECUTIVE SUMMARY

Waquoit Bay is a small estuary on the south coast of Cape Cod, Massachussetts. It is prized by residents and visitors for its aesthetic beauty and recreational opportunities. The watershed and bay and the adjoining marshes, tidal rivers, and barrier beaches provide ideal habitat for plant and animal life, including piping plovers and least terns (endangered birds), the sandplain gerardia (endangered plant), winter flounder, blue crabs, scallops, and clams and anadromous (spawn in fresh water) and catadromous (spawn in salt water) fish.

Human encroachment is changing the landscape and contributing nutrients and contaminants to the bay. In the Waquoit Bay watershed more than 85% of homes use onsite septic systems. The permeable sandy soils allow much of the nitrogen input from the septic tank leachate to reach groundwater. The nitrogen in groundwater travels to coastal waters, where it stimulates primary production, resulting in thick mats of macroalgae that have now replaced once-abundant eelgrass meadows, the preferred habitat for many organisms in the bay. The EPA-sponsored ecological risk assessment created a mechanism to bring organizations together to integrate the results of various research and planning efforts. Documenting the process, the analyses, and the modeling approach provides a means by which resource managers can assess the risk of land-derived nitrogen to their estuaries and evaluate the magnitude of the problem caused by nitrogen in comparison to other stressors. This information can be used to make wiser remediation decisions related to providing funds for eelgrass planting, or requiring denitrifying septic systems, or making land use changes.

1.1. ECOLOGICAL RISK ASSESSMENT

Ecological risk assessment is a process for collecting, organizing, analyzing, and presenting scientific information to make it more useful for decision making. It is a unique form of ecological assessment and includes the term "risk" because it presumes that a cause and effect relationship exists and that the relationship can be expressed as a stressor-response curve. Although extensively used to predict the impacts of single stressors (e.g., a particular pesticide) on a particular species, EPA seeks to demonstrate the use of ecological risk assessment in evaluating environmental problems addressed through the watershed approach. The watershed approach is based on using partnerships, sound science, and environmental management in decision making. EPA seeks to demonstrate that integrating the watershed approach with ecological risk assessment will increase the likelihood that environmental monitoring and assessment data will be used to inform decisions.

The Waquoit Bay watershed was selected as the site of an EPA-sponsored watershed ecological risk assessment because local, state, and federal organizations were interested in cooperating in a risk assessment; it has multiple stressors (e.g., nutrients, toxic chemicals, and altered freshwater flow); there is abundant data; and the Waquoit Bay National Estuarine Research Reserve (WBNERR) and EPA Region I were willing to lead the risk assessment workgroup. Other members of the workgroup included NOAA's Northeast Fisheries Science

Center, National Marine Fisheries Service; Boston University Marine Program, Marine Biological Laboratory; and EPA's Offices of Research and Development, Water, and Pollution Prevention and Toxic Substances.

The assessment process began with planning and problem formulation, proceeds through analysis of exposure and effects, and ends with risk characterization. During planning, the management goal for the watershed and the purpose, scope, and complexity of the assessment was established. In problem formulation, ecologically and socioeconomically relevant assessment endpoints were defined, and conceptual models and plans for the analysis were developed. In this assessment, a nitrogen loading model (NLM) predicted the amount of nitrogen reaching the bay from various sources within the watershed. During risk characterization, the results of the model were combined with historical data on eelgrass area cover and scallop harvests to provide insights on how these valued resources can be reestablished to historical levels.

1.2. PLANNING AND PROBLEM FORMULATION

1.2.1. Management Goal and Objectives

The management goal was defined by an interdisciplinary and interagency workgroup of scientists and managers on the basis of a public meeting, a meeting of workgroup members to develop the goal and objectives, and a meeting with local resource managers to refine the goal and objectives. The agreed-upon management goal was:

> *Reestablish and maintain water quality and habitat conditions in Waquoit Bay and associated wetlands, freshwater rivers, and ponds to (a) support diverse, self-sustaining commercial, recreational, and native fish and shellfish populations and (b) reverse ongoing degradation of ecological resources in the watershed.*

The 10 management objectives that more explicitly state the kinds of management results that were implied in the general goal statement include:

1. Reduce or eliminate hypoxic or anoxic events
2. Prevent toxic levels of contamination in water, sediments, and biota
3. Restore and maintain self-sustaining native fish populations and their habitat
4. Reestablish viable eelgrass meadows and associated aquatic communities in the bay
5. Reestablish a self-sustaining scallop population that can support a viable fishery
6. Protect shellfish beds from bacterial contamination that results in bed closures
7. Reduce or eliminate nuisance macroalgal growth
8. Prevent eutrophication of rivers and ponds
9. Maintain diversity of native biotic communities
10. Maintain diversity of wetlands habitat

The workgroup developed a list of stressors and endpoints, built a conceptual model that tied the stressors to their endpoints, and decided on an analysis plan.

1.2.2. Stresssors

Six stressors from human activities that could impact resources in Waquoit Bay and the potential sources of those stressors within the watershed were selected for analysis. These included:

1. Chemical pollution from pesticides and herbicides, emissions, industrial point sources, and boating activities
2. Altered freshwater flow from new construction and waste treatment
3. Nutrient enrichment/eutrophication from agriculture, lawn and garden fertilization, waste treatment, industrial point sources, and atmospheric deposition of emissions
4. Physical alteration of habitat from dredging and boating activities
5. Fishing and shellfishing resulting from harvest pressure from commercial and recreational fishing
6. Pathogens from industrial point sources, runoff from impervious surfaces, and waste treatment

1.2.3. Assessment Endpoints

Assessment endpoints are the link between scientifically measurable endpoints and the objectives of the stakeholders and resource managers (Suter 1989, 1993). Endpoints should be ecologically relevant, related to the previously defined management objectives, and susceptible to stressors (U.S. EPA, 1998). The workgroup originally selected seven assessment endpoints. The endpoints were:

1. Estuarine percent eelgrass cover
2. Finfish diversity and abundance
3. Scallop abundance
4. Anadromous fish reproduction
5. Wetland bird habitat distribution and abundance
6. Piping plover habitat distribution and abundance
7. Tissue contamination of fish and shellfish

1.2.4. Conceptual Model

The conceptual model is a broad representation of relationships among human activities in the watershed (sources) are the stressors believed to result from those sources (as described in Section 1.2.2), the exposure pathways linking stressors to effects, and the assessment endpoints. Each of the pathways in the conceptual model was derived from information about the Waquoit Bay watershed and estuary in the peer-reviewed scientific literature, from ecological theory on how systems function, and from similar relationships established in other watersheds. The conceptual model illustrates relationships such as how excess nutrients increase algal growth, shading and reducing eelgrass habitat, and reducing scallop abundance. This model provides the foundation for the analysis plan.

1.2.5. Analysis Plan

The workgroup conducted a comparative risk analysis to help define which stressors, assessment endpoints, and relationships should be further examined. Stressors were ranked in terms of potential risk to all resources in the watershed on the basis of best professional judgment of the workgroup. The results of the comparative analysis ranked nutrients first. To verify that nutrients were indeed the largest stressor in Waquoit Bay, the authors later examined each of the stressors for the intensity of its impact, extensiveness within the watershed, and the likelihood to increase over time. This analysis also indicated that nutrient loading is the dominant agent of change in the Waquoit Bay watershed.

Later, the authors agreed to focus on nitrogen loading because phosphorus input, although important in eutrophication of freshwater ponds, is being analyzed and mitigated by the Air Force Center for Environmental Excellence. In addition, the authors agreed that there were sufficient data available to develop models to evaluate the quantity of nitrogen entering the bay and draw correlations with critical ecosystem components.

To evaluate risk from nitrogen loading to assessment endpoints it was necessary to quantify the loading of nitrogen into the watershed and estuary and to evaluate how a given load of nitrogen impacts the estuarine ecosystem. The area of eelgrass cover and scallop abundance were chosen as the assessment endpoints because they were most susceptible to the identified stressor of concern, nitrogen loading, and were not so variable that they easily evaded characterization. Additionally, several other characteristics that are necessary for ecosystem functioning, including nitrogen concentrations in the estuary, phytoplankton and macroalgae biomass and production, zooplankton egg production, shellfish growth rates, and finfish abundance were evaluated in less detail. Although it is possible to evaluate the risk to these other ecosystem components, their complexity makes risk modeling extremely difficult.

1.3. RISK ANALYSIS

The NLM sums the nitrogen loads to the watershed from three major sources: atmospheric deposition, septic-derived wastewater, and fertilizer application. It then subtracts losses during transport to yield a value for the quantity of nitrogen arriving at the edge of the estuary (or salt marsh). The estuarine loading model (ELM) estimates the concentrations of dissolved inorganic nitrogen available to producers in shallow estuaries. These estimates account for inputs and losses of nitrogen after it reaches the bay through processes such as denitrification, burial of nitrogen in sediments, speed of water movement out of the bay (turnover rate), regeneration of nitrogen from the sediments, nitrogen fixation, and direct atmospheric deposition of nitrogen onto the surface of the bay.

The loading rates and effects of nitrogen input on various components of ecosystem function are variable. Nitrogen loads entering Waquoit Bay were measured by sampling groundwater around the periphery of the bay and the subestuaries. Nitrogen loads ranged from 433 kg N/yr in the pristine Sage Lot Pond to 9879 kg N/yr in the Quashnet River. The total amount of nitrogen reaching all of Waquoit Bay was just over 26,500 kg N/yr.

Nitrogen loading leads to an increase in the concentration of nitrogen that is available in the estuary. As a result, the biomass and production of both phytoplankton and macroalgae, two of the dominant primary producers in Waquoit Bay, have increased substantially. The increases in these two producers worked together to prevent light from penetrating the bottom of the estuaries, resulting in a decrease in the biomass and production of the other dominant primary producer—eelgrass *(Zostera marina)*. Eelgrass is especially important because it provides habitat and a refugia for juvenile fish and shellfish. The effect of nutrient input on higher trophic levels also was variable. Evidence suggests that copepod egg production increases as a result of the greater production of phytoplankton, a preferred food choice. However, there was no related increase in the number of copepods (free-floating tiny aquatic arthropods) in the Waquoit Bay estuaries. This suggests that there was a lack of response of copepods to increased food supply, probably as a result of the short residence time of the water within these estuaries. Growth rates of two commercially important shellfish species, the softshell clam *(Mya arenaria)* and hardshell clam (*Mercenaria mercenaria*), increased as nitrogen load increased, but populations of the bay scallop *(Argopecten irradians)* decreased drastically as nitrogen load increased. Finally, the two most common estuarine finfish species, Atlantic silverside *(Menidia menidia)* and mummichog *(Fundulus heteroclitus)*, showed no response to nutrient loading in any of the estuaries of Waquoit Bay.

1.4. RISK CHARACTERIZATION

In risk characterization, the exposure and effects analyses are integrated to estimate risks to the assessment endpoints. Risk characterization also serves to summarize and describe the results of the risk analysis in such a way that results can be readily translated to managers and other stakeholders. In the risk characterization for the Waquoit Bay watershed, the authors integrated the findings from the series of models that were created to assess the eutrophication exposure and effects and calculated the risk of eutrophication.

The models were used to characterize changes in nitrogen loading. Land-use changes over the last 60 years were assessed from aerial photographs, and this information was integrated into the model structure to back-cast nitrogen loads to the late 1930s. As the Waquoit Bay watershed became more urbanized, the modeled nitrogen load to the watershed increased from slightly more than 10,000 kg N/yr to more than 24,000 kg N/yr in 1990. This increase in nitrogen load can be correlated to decreases in both eelgrass area and scallop harvest with time. Eelgrass area declined from 60% coverage in 1955 to less than 10% coverage in 1990. This decrease in eelgrass area occurred as nitrogen loads exceeded 20 kg N/ha/yr. Additionally, the harvest of scallops decreased from over 200,000 landings (L)/yr to well under 20,000 L/yr during the same time period.

1.5. MANAGEMENT IMPLICATIONS

The historical reconstruction of land-derived nitrogen loads, plus the linkage of these loads to assessment endpoints such as percent eelgrass cover and bay scallop harvest, provided some means to identify management priorities and define potential restoration measures. If management goals included reduction of nitrogen loads, then resource managers could use the changes that occurred in assessment endpoints to establish remediation targets. For example, if stakeholders wanted to restore eelgrass to 30% coverage in the estuary, then this modeling characterization indicates that managers would need to reduce nitrogen loads to approximately 18,000 kg N/yr to generate conditions that could potentially allow eelgrass to survive. As a further step, these models also could be used to run simulations identifying management options that could produce the desired targets. The models could be used to assess, for example, the reduction in nitrogen loads that would occur if fertilizer application rates were lowered or septic tank retention efficiency were increased. Thus the risk assessment has provided not only an analysis of the impact of major stressors on the ecological components of Waquoit Bay, but also provided insights into how the Waquoit Bay community can reestablish and maintain the Bay's habitat and water quality.

2. INTRODUCTION

The Waquoit Bay watershed in Cape Cod, Massachusetts, has many valuable natural resources that are being jeopardized by urbanization and other human activities. Nitrogen loading was suspected to be the principal cause of environmental problems, such as loss of eelgrass and fish populations (including scallops), which are dependent on eelgrass habitat. This risk assessment describes how excess nitrogen input is the stressor of concern in the Waquoit Bay watershed and then evaluates and models its effects on valued ecological resources. Following this introduction, the report describes the process by which the assessment was performed and presents the analytical results.

An interdisciplinary and interagency workgroup used watershed ecological risk assessment principles to analyze available information in a manner that would increase the likelihood that it would be useful for decision making. The watershed approach (U.S. EPA, 1996) is based on using partnerships, a hydrologically defined geographic boundary, sound science, and sound environmental management in decision making. Ecological risk assessment (U.S. EPA, 1998) is a process to collect, organize, analyze, and present scientific information. Watershed ecological risk assessment (Serveiss et al., 2000; Serveiss, 2002) integrates the watershed approach with ecological risk assessment to increase the likelihood that environmental monitoring and assessment data will be used in watershed-scale decision making. Documenting the process and performing the modeling and other analyses provides scientific information to help managers justify taking actions to address problems.

2.1. THE WATERSHED

Waquoit Bay is a small estuary on the south coast of Cape Cod, Massachusetts (Figure 2-1). Its watershed covers about 53 km^2 (21 mi^2) and includes freshwater streams and ponds, salt ponds and marshes, pine and oak forests, barrier beaches, and open estuarine waters. The land and water are home, spawning ground, and nursery for a diversity of plant and animal life, including threatened and endangered species, such as piping plovers (*Charadrius melodius*), least terns (*Sterna albifrons*), and the sandplain gerardia (*Agalinis acuta*), as well as commercially important shellfish species such as blue crabs (*Callinectes sapidus*), bay scallops (*Argopecten irradians*), and hardshell clams (*Mercenaria mercenaria*), and a multitude of recreational and commercial fish species, including alewife (*Alosa pseudoharengus*), and winter flounder (*Pseudopleuronectes americanus*). For more information on the geological, hydrological, and biological characteristics of the Waquoit Bay estuary see Appendix A.

The Waquoit Bay region was part of the Wampanoag tribal lands when European settlers arrived in the early 1600s on what is now Cape Cod (Gallagher, 1983). For more than 200 years, the Waquoit watershed was used primarily for hunting, farming (strawberries and potatoes were important crops), and maritime industries such as fishing, whaling, and shipbuilding (Faught,

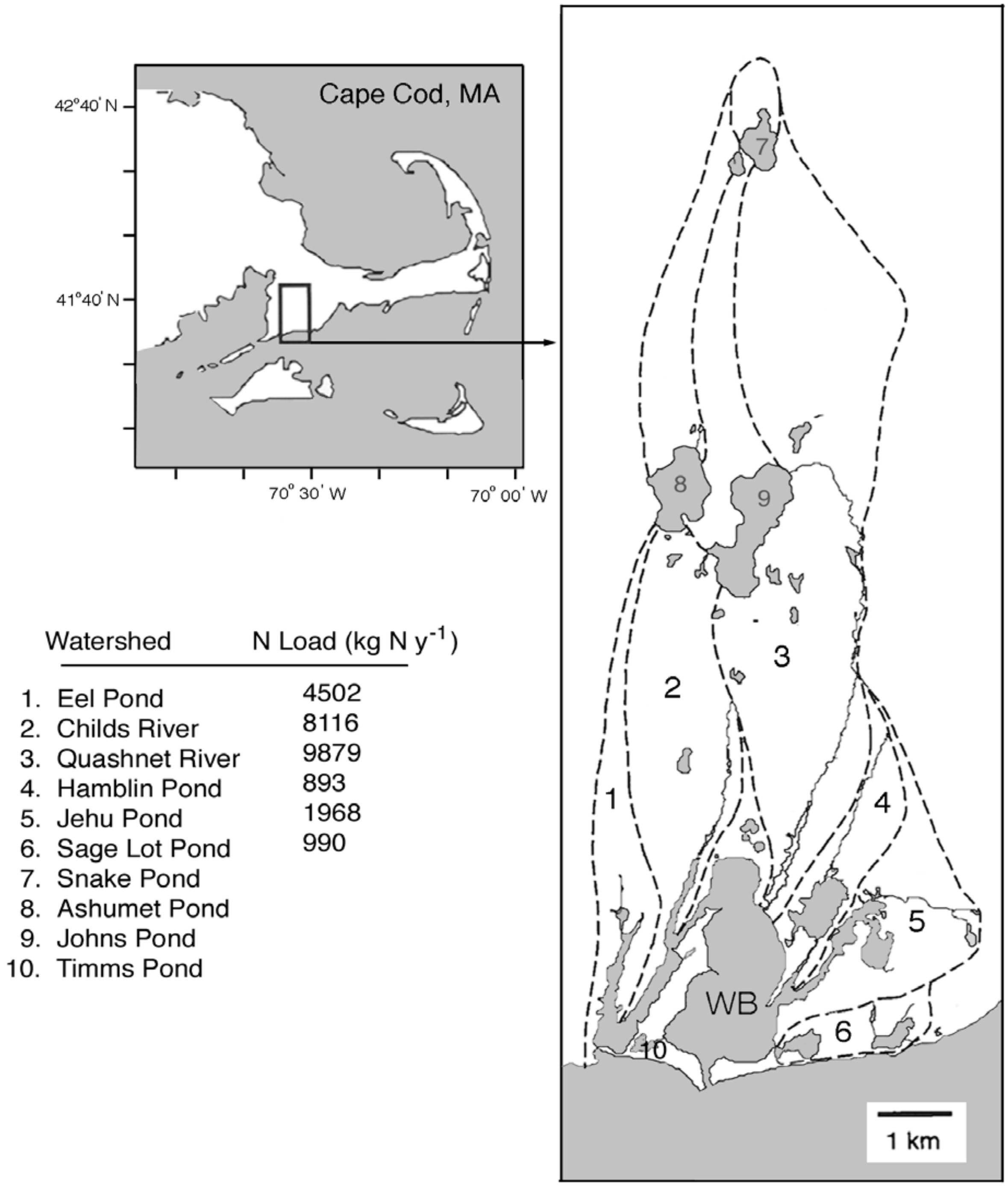


Figure 2-1. Delineation of the watershed of Waquoit Bay, MA. Subwatersheds draining into the six estuaries of Waquoit Bay are noted with dashed lines. The inset shows the location of Waquoit Bay on Cape Cod, MA. Annual measured nitrogen loads to the subestuaries are provided. The nitrogen loads entering the three upper ponds are included as a portion of the loads to the receiving estuaries.
Source: Modified from Bowen and Valiela (2001a).

1945). Since the late 1800s, and with the advent of rail service from Boston to Cape Cod, the natural beauty of the area has attracted more people. Today, more than 8,000 people live within the Waquoit watershed. The population of the watershed swells in summer months, largely as a result of visitors from the greater metropolitan areas of Boston, Providence, and New York City. Cape Cod's economic viability depends largely on tourists who are drawn to the abundant sandy beaches, seafood restaurants, boating opportunities, and water recreation activities. Thus, the economy and the environment of Cape Cod depend on one another.

These once-rural surroundings have become increasingly suburbanized with the development of bedroom and retirement communities. Aerial photographs of a portion of the Waquoit Bay watershed taken in 1938 and in 1990 illustrate the population expansion that has occurred over the years (Figure 2-2). According to the U.S. Census Bureau, the population of Barnstable county, within which the Waquoit Bay watershed is located, increased by 13.9% between 1990 and 1999. In Massachusetts this was surpassed only by Dukes County (Martha's Vineyard), which increased by 20.7%, and Nantucket County, which increased by 36.5%. By comparison, the increase in population of the entire state of Massachusetts was 2.6% during the same period (U.S. Census Bureau, 2000).

As the population increases, so does the pressure on the valuable natural resources that attract people to the region. These resources include clean beaches, healthy eelgrass meadows, and viable fisheries. People want to live as near to the water as possible; the result is that the near-shore land parcels are the most frequently developed (Figure 2-3). The number of buildable lots that remain is still substantial, but most future building will take place farther away from shore.

Resource managers have expressed concern about the condition of the bay's natural resources, specifically, the significant decreases in the area of eelgrass meadows in Waquoit Bay and its subestuaries, increases in the frequency of anoxic events, groundwater contamination from a military reservation located in the far reaches of the watershed, changes in the abundance of recreationally important species, diminished aesthetics, and the ecological stress caused by greater recreational use of water resources. The next few paragraphs provide more detail on these concerns.

Eelgrass (*Zostera marina*) is a flowering plant that inhabits the sediments of many shallow embayments of the northwestern Atlantic Ocean. Numerous studies have shown that eelgrass meadows provide an important nursery habitat for the juvenile stages of commercially and recreationally important fish and shellfish (Valiela et al., 1992; Duarte, 1995). Eelgrass, macroalgae, and phytoplankton are the three dominant primary producers in shallow coastal systems like Waquoit Bay. Unlike the other two producers, eelgrass can acquire many of its required nutrients through roots, but because it is rooted to the bottom of the estuary, it requires sufficient light penetration through the water for photosynthesis. In Waquoit Bay, as in many shallow estuaries, increased phytoplankton and macroalgae, fueled by the addition of nitrogen from human sources, have reduced the amount of light that reaches the eelgrass. As shown in

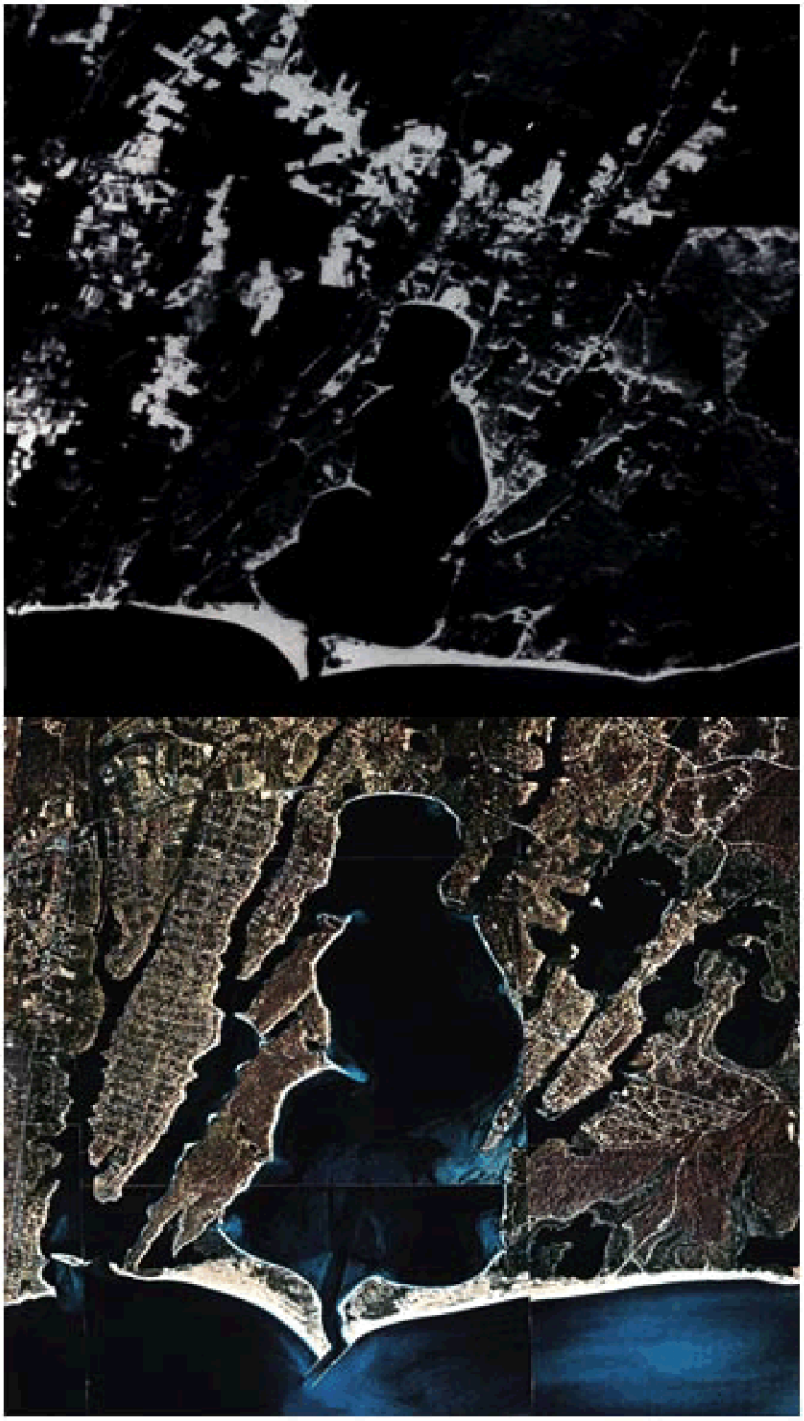
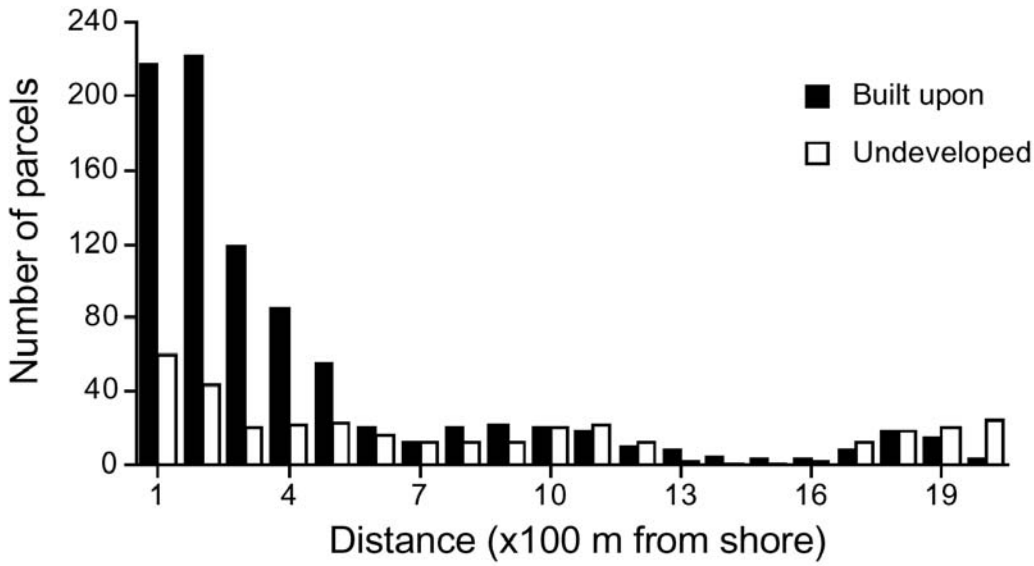


Figure 2-2. Aerial photographs of Waquoit Bay in 1938 (top) and 1990 (bottom). White areas in the 1938 figure are agricultural lands and beach. The 1990 photo shows the disappearance of agricultural land and the preponderance of suburban sprawl, particularly near the coast.
Source: Massachusetts Department of Environmental Protection.



Figue 2-3. Number of parcels that are built upon (black) and remain to be built (white) as of 1990 in the Childs River watershed of Waquoit Bay, MA.

Source: Modified from Valiela et al. (1992).

Figure 2-4, in 1951, eelgrass meadows covered most of Waquoit Bay proper and its adjoining coastal ponds and rivers (Costa, 1988); today, eelgrass is absent from the bay proper (Short and Burdick, 1996; Hauxwell et al., 2001). It still exists in some of the subestuaries of the bay, although it has declined significantly in the Quashnet River, Hamblin Pond, and Jehu Pond. Sage Lot Pond and Timms Pond still have healthy eelgrass populations (Short and Burdick, 1996; Hauxwell et al., 2001). Species that are dependent on eelgrass for habitat, feeding, or spawning grounds also have suffered population declines. In particular bay scallops, which rely on eelgrass for larvae settlement and predator refuge, showed a decline in abundance, as evidenced by scallop harvest, from nearly 200,000 L/yr in the early 1960s to less than 20,000 L/yr in the mid-1990s (Shumway, 1991; Bowen and Valiela, 2001a).

Oxygen depletion of the water is another effect of the increased abundance of phytoplankton and macroalgae in bays. The decomposition of organic matter from decaying plankton and algae consumes oxygen, and if there is not enough sunlight to produce oxygen through photosynthesis, a condition known as hypoxia (low oxygen) or anoxia (no oxygen) can occur. Very low oxygen concentrations can lead to the death of organisms. In 1988 and 1990, fish kills occurred in Waquoit Bay; the northern beach was covered with thousands of dead winter flounder, shrimp, blue crabs, and other estuarine species (D’Avanzo and Kremer, 1994).

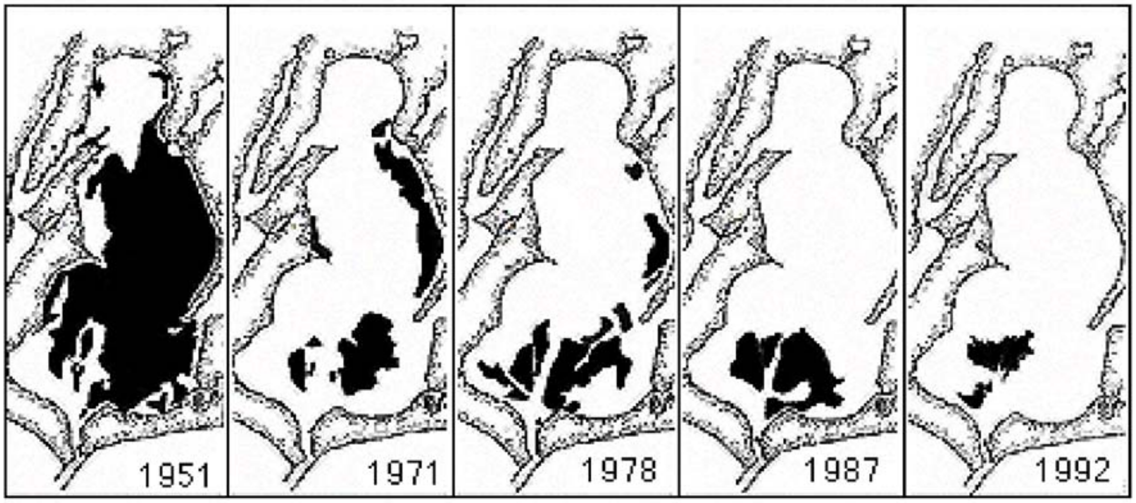


Figure 2-4. Area of eelgrass in Waquoit Bay between 1951 and 1992. Black areas represent the extent of eelgrass beds.

Source: Figures for 1951–1987 by J. Costa et al. (1992) and data from Short and Burdick (1996), published in Valiela et al. (1992).

Smaller-scale fish kills also occurred in the summer of 2001. Anoxia can occur in freshwater ponds as well. In Ashumet Pond and Johns Pond, both located in the upper reaches of the Waquoit Bay watershed, blooms of phytoplankton have changed the color of the water and depleted oxygen levels in the bottom waters of the pond. In Ashumet Pond, the deep bottom waters (>12 m) are consistently anoxic during the summer months. Fish kills occurred in Ashumet Pond in 1985 and 1986. Several expert panels have concluded that nutrient enrichment—and the resulting anoxia—is the major man-made pollution problem affecting coastal waters (GESAMP, 1990; NRC, 2000).

The Massachusetts Military Reservation (MMR), which is located in the northern portion of the Waquoit Bay watershed, was designated a Superfund site in 1989 due to extensive groundwater contamination from a number of sources. Nutrients from a wastewater treatment plant, chemicals from fuel spills, and chlorinated solvents used at MMR in the past have contaminated 55 to 60 billion gallons of Cape Cod's sole-source aquifer for drinking water. These toxic chemicals include benzene, toluene, ethylbenzene, xylene, trichloroethylene, tetrachloroethylene, and ethylene dibromide. A human health risk assessment was conducted by MMR's Installation Restoration Program (IRP) to evaluate the potential risks of these contaminants to the public drinking water source.

Because the Air Force Center for Environmental Excellence (AFCEE) has provided money to the surrounding towns of Falmouth and Mashpee to hook up residents with private wells to public water supplies when their water wells were threatened by MMR plumes, the plumes pose no threat to public health. The goal of the cleanup is restoration of the sole-source aquifer for drinking water with the target cleanup level being the established maximum contaminant level for each contaminant. The active extraction, treatment, and reinjection systems have reduced the contaminant mass in the plumes, as have some of the plumes for which monitored natural attenuation was chosen as the remedy.

The IRP evaluates ecological impacts on aquatic components on the basis of alteration of the hydrologic flow to rivers and ponds that accompanies operation of groundwater extraction and treatment systems. Based on the IRP, potential impacts of changed hydrologic flow are minimized during design of these groundwater treatment systems. Given that the focus of the IRP is the protection of the drinking water source, contaminant levels are being reduced based on human health drinking water criteria standards. However, the human health maximum contaminant levels for toxic contaminants in potential sole-source aquifers for drinking water are two to three orders of magnitude lower than the acute/chronic toxicity thresholds for aquatic biota. Thus, while there are no known ecological impacts from the toxic chemical plume discharges, there is an impact on the freshwater pond from phosphorus discharge (Howarth, 1988).

The Ashumet Valley wastewater plume, which originates at MMR, has contributed to seasonal anoxia through increases in phosphorus and nitrogen loading in Ashumet Pond. The pond is the subject of a mitigation plan by the AFCEE that uses an alum treatment to reduce the amount of phosphorus released from the sediments in the deep-water portion of the pond.

With the increase in population in the Waquoit Bay watershed, the number of people who use the bay for recreational boating also has increased. Resuspended sediments from boating activities, toxic chemicals from pressure-treated wood in docks, oil and gas leaks from boat motors, propeller scarring, shading of eelgrass beds by docks, and erosion of shores along Vineyard Sound have been cited by residents as additional potential sources of stress to valuable marine resources. Sewage discharge from small boats was deemed to be of little concern since Waquoit Bay was designated as a "No Discharge Area" in 1994, and any inputs from wastewater discharge still occurring would be small relative to the inputs from land-derived sources. Concern about the effects of population growth on Cape Cod has led to several initiatives and activities, including the following, that predate the risk assessment:

- Creation of the Cape Cod Commission, a regional planning agency that has authority over all construction that could have regional impact on Cape Cod resources

- Work by the Association for the Preservation of Cape Cod, which has been instrumental in fighting for the protection of the Cape's drinking water supply and determining the impact of watershed nitrogen loading on coastal water quality

- Efforts by the Waquoit Bay Land-Margin Ecosystem Research Project, a multi-institutional, interdisciplinary research program that has contributed to the knowledge of the impact of land-derived nitrogen loading on coastal waters and on the ecological functioning of the bay

- Designation of the Waquoit Bay National Estuarine Research Reserve (WBNERR), which also serves to protect the resources of the bay and its adjacent lands and to educate the public about their precious natural resources

- Designation of part of the Waquoit Bay watershed as a U.S. Fish and Wildlife Refuge, which has resulted in land acquisition and a multi-agency memorandum of understanding to develop a coordinated management plan for the reserve

- Designation of the Waquoit Bay area as an Area of Critical Environmental Concern, a Massachusetts designation that provides additional regulation for development that might impact natural resources

- Designation of the Waquoit Bay watershed as one of the EPA's sponsored ecological risk assessments

2.2. THE WATERSHED ECOLOGICAL RISK ASSESSMENT PROCESS

The EPA-sponsored ecological risk assessment of Waquoit Bay builds on local and regional work by creating a mechanism to integrate the results of various research and planning efforts into management options for local coastal decision makers (Figure 2-5). The purpose of an ecological risk assessment is to collect, organize, analyze, and present scientific information to help decision makers achieve management goals. It is a unique form of ecological assessment and includes the term "risk" because it presumes that a cause and effect relationship exists and that the relationship can be expressed as a stressor-response curve. The Waquoit Bay watershed ecological risk assessment was initiated because of interest by local, state, and federal organizations in the watershed, the type of watershed (estuarine), the diversity of stressors (e.g., nutrients, toxic chemicals, altered hydrology), an abundance of previously collected data, and willingness by WBNERR and EPA New England to lead the risk assessment workgroup.

There is much literature on the ecological risk assessment process (e.g., U.S. EPA, 1992, 1998; Suter, 1993) and limited literature on how it can be applied to watershed and regional assessments (Landis and Wiegers, 1997; Wiegers et al., 1998; Norton et al., 2000; Serveiss et al., 2000; Serveiss, 2002). In watershed ecological risk assessments, the planning effort often is extensive because there are many interested parties with diverse interests and numerous stressors impacting many valued ecological resources through numerous pathways. Unlike other types of smaller-scale assessments, all of these pathways cannot be precisely quantitatively analyzed, and innovative methods need to be used to analyze information and produce scientifically credible and useful conclusions for resource managers.

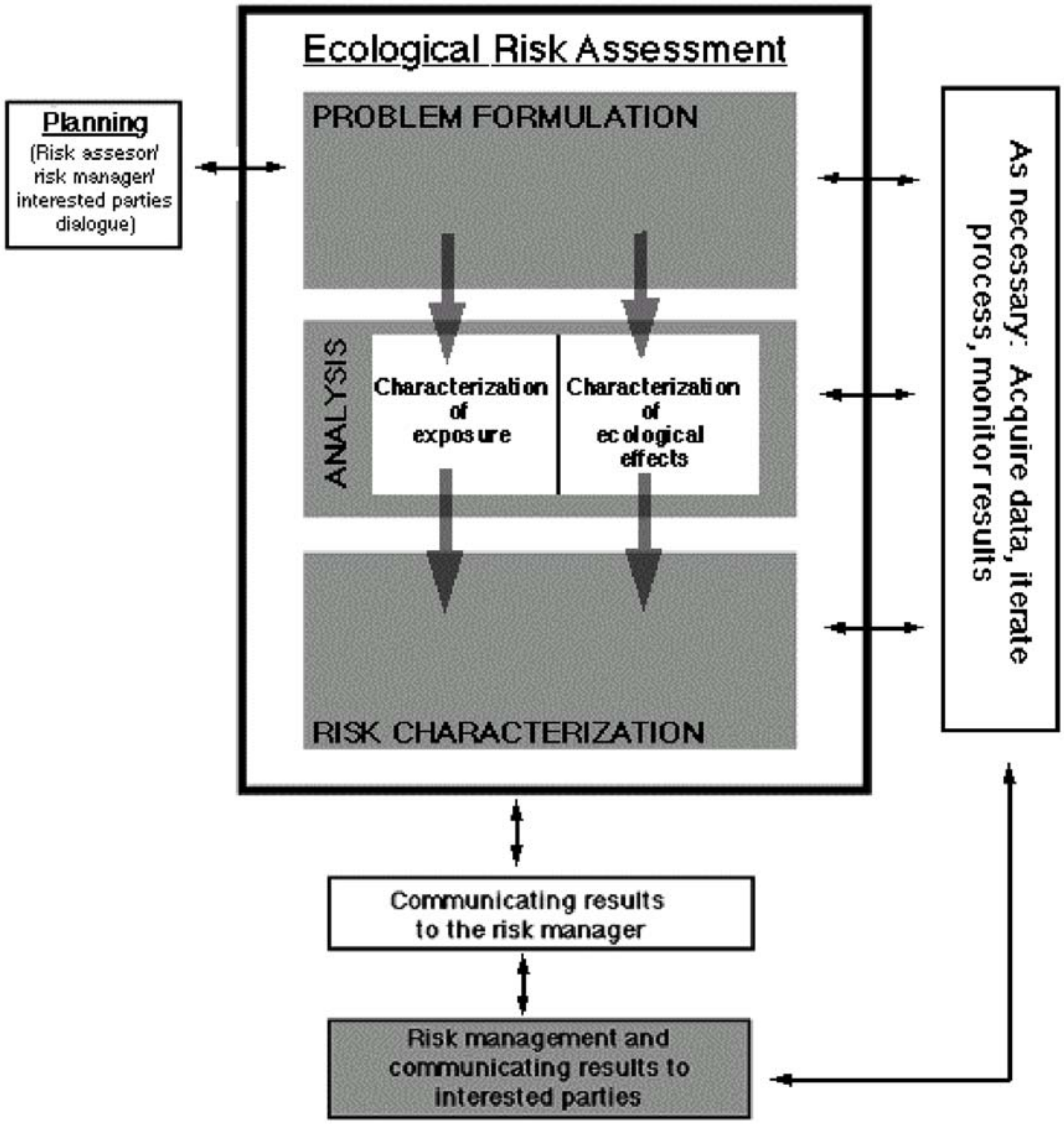


Figure 2-5. Framework for the ecological risk assessment.

Source: U.S. EPA (1998).

3. PLANNING FOR THE ECOLOGICAL RISK ASSESSMENT

The Waquoit Bay watershed ecological risk assessment was based on a proposal by managers at Waquoit Bay National Estuarine Research Reserve (WBNERR) and EPA Region I, who were concerned about the deteriorating ecological quality of the bay. Based on this interest, a risk assessment workgroup was established (see page xii). The purpose of the risk assessment planning was to establish clear and agreed-upon environmental management goals and objectives. These objectives provided the context for the assessment and were supported by the diverse members of the Waquoit Bay community. To achieve this support, the workgroup held a public meeting, reviewed the goals of environmental organizations, and interacted with nongovernmental organizations (NGOs), scientists, and managers at the state and local level.

Public meeting. EPA, in conjunction with WBNERR, held a public forum on September 21, 1993. Concerned citizens and representatives from some of the organizations that have interests in Waquoit Bay attended (Appendix B). At this meeting WBNERR/EPA representatives asked participants to provide answers to two questions:

- What environmental resources do you value in Waquoit Bay?
- What is placing those values at risk?

The attendees also were given a list of biological, chemical, and physical stressors and were asked to evaluate them with respect to the valued resources within the watershed. The results of the analyses are presented in Appendix C. From the information gathered at this public meeting, the workgroup developed a preliminary management goal.

Goals of management organizations. To further refine the preliminary management goal into more specific management objectives, the workgroup reviewed the goals of local, regional, and national resource management organizations and NGOs with responsibilities in the watershed (Appendix B). This review was based on either written documentation published by the organizations or on statutes.

Risk assessment workgroup consensus. Members of local resource management organizations were invited to a meeting sponsored by WBNERR on February 24, 1995 (Appendix D). The management goal and objectives, along with the summary of the goals of various organizations, were presented to meeting participants. Ultimately, the management goal and objectives for the risk assessment were modified by the workgroup from comments from the managers and regulators present at that meeting. The agreed-upon management goal became:

> *Reestablish and maintain water quality and habitat conditions in Waquoit Bay and associated wetlands, freshwater rivers, and ponds to (a) support diverse, self-sustaining commercial, recreational, and native fish and shellfish populations and (b) reverse ongoing degradation of ecological resources in the watershed.*

Ten management objectives were developed to characterize the management goals that were implied in the more general statement. The management objectives were partitioned into three categories (Table 3-1). The estuarine and freshwater category included three objectives that were common to both surface water types. Four objectives under the estuarine category and three objectives under the freshwater category were unique to those waters. The management objectives reflected generic concerns of the NGOs and local/state/federal managers with interests in the watershed. This systematic process enabled the workgroup to identify assessment endpoints and develop an analysis plan for the assessment.

Table 3-1. The Waquoit Bay watershed management objectives

Affected area	Management Objective	
Estuarine and freshwater	1	Reduce or eliminate hypoxic or anoxic events
	2	Prevent toxic levels of contamination in water, sediments, and biota
	3	Restore and maintain self-sustaining native fish populations and their habitat
Estuarine	4	Reestablish viable eelgrass beds and associated aquatic communities in the bay
	5	Reestablish a self-sustaining scallop population in the bay that can support a viable sport fishery
	6	Protect shellfish beds from bacterial contamination that results in bed closures
	7	Reduce or eliminate nuisance macroalgal growth
Freshwater	8	Prevent eutrophication of rivers and ponds
	9	Maintain diversity of native biotic communities
	10	Maintain diversity of water-dependent wildlife

4. PROBLEM FORMULATION

During problem formulation, conceptual models, assessment endpoints, and an analysis plan are developed. Problem formulation uses available information on sources, stressors, ecological resources potentially at risk and ecological effects to 1) identify the ecological resources that will be the focus of the risk assessment; 2) develop conceptual models of how these resources may be impacted by human activities; and 3) develop a plan for the analysis phase.

4.1. CONCEPTUAL MODEL

The watershed conceptual model is a broad representation of relationships among human activities in the watershed; the physical, chemical, and biological stressors believed to occur as a result of those activities; and the ecological effects likely to occur to each of the assessment endpoints. An earlier conceptual model developed on the basis of input from meeting participants, was found to help resource managers—who typically focus on a resource or a problem—to better consider the big picture. The earlier model, which is on display at the WBNERR visitor center, has been found to be a useful tool for communicating with the public. Figure 4-1 shows a revised model that reflects further insight by authors, additional input provided by reviewers, and modifications to further improve clarity.

The pathways Figure 4-1 were organized around stressors. Each stressor has a colored line that illustrates a pathway connecting its sources to possible effects and selected endpoints. Each of the components of the model is represented by a different geometrical shape to aid interpretation. This is a broad-based model that provides a framework for the risk assessment and an overview of ecosystem processes. The diagram shows only stressors and effects thought to be potentially important in the Waquoit Bay watershed. The exposure pathways, from the source of stressors to valued resources, are the possible risk hypotheses to consider for analysis as part of the risk assessment. In the following sections of problem formulation the sources of stressors, the nature of those stressors, and the assessment endpoints that they affect are described.

4.1.1. Sources of Stressors

The changing land and water use patterns in coastal and upland areas of the Waquoit Bay watershed are largely responsible for increasing the intensity of stressors in the watershed. The main sources of stress include agriculture, urbanization and industrial development, and marine activities.

Agriculture. Fertilizers, pesticides, and herbicides from agricultural applications can leach into groundwater and eventually enter estuaries. In general, some portion of the fertilizers that are applied to agricultural crops can be volatilized to ammonium and then deposited in the form of nitrogen farther downwind of the source, but for the Waquoit Bay area, the atmospheric sources are likely derived from far outside the watershed (Bowen and Valiela, 2001a).

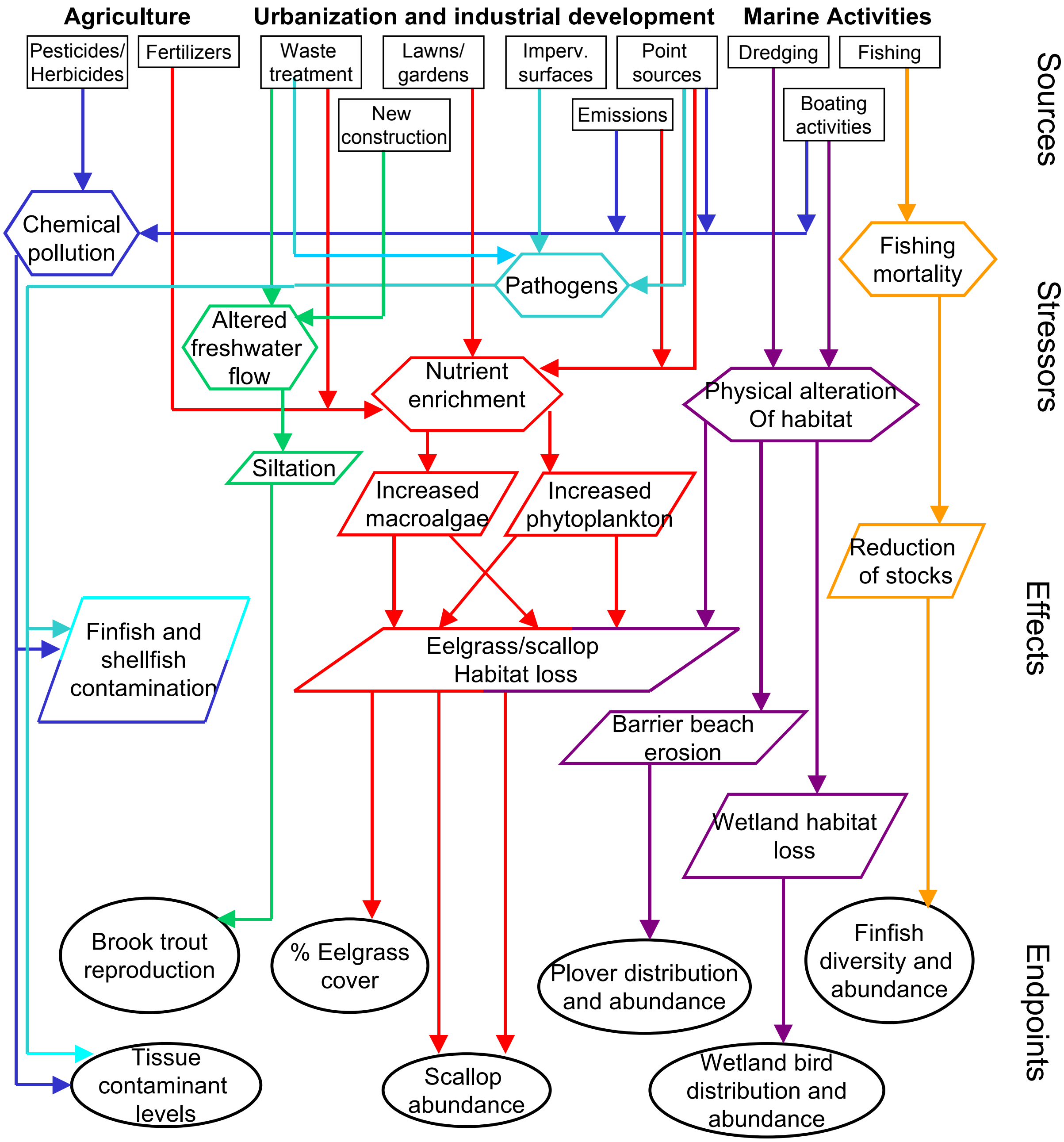


Figure 4-1. Conceptual model of the Waquoit Bay watershed ecological risk assessment. Rectangles represent sources of stressors, hexagons are the specific stressors to the system, trapezoids represent the effects of those stressors, and the ellipses indicate specific endpoints that are affected.

Conversion of naturally vegetated land to agriculture can also result in erosion of upland soils and siltation in adjacent streams and estuaries, but for the coarse soils of Cape Cod, runoff and siltation are, at most, of moderate importance. Agricultural use of land in Waquoit Bay was prevalent early in the past century, but in recent years the area of land dedicated to agriculture has decreased dramatically (Bowen and Valiela, 2001a). Cranberry cultivation remains as the last notable commercial agricultural land use in the Waquoit Bay watershed.

Urbanization and industrial development. Urbanization has resulted in numerous sources of ecological stress in the Waquoit Bay system. As human populations on watersheds increase, so does the waste generated. Thus, wastewater disposal is one of the major sources of stress resulting from population expansion. An additional stressor associated with urbanization is the fertilizer applied to lawns, gardens, and golf courses. New construction can result in increased areas of impervious surfaces, which can alter hydrological flow patterns. The use of aquifer water for both drinking and waste disposal can alter hydrologic flow patterns, providing an additional source of stress to the groundwater system. Finally, increased emissions from automobiles can provide an additional input of nutrients and contaminants in downwind locations, but on Cape Cod these local atmospheric sources are small compared to upwind sources (Bowen and Valiela, 2001a).

Industrial sources of stress in watersheds can stem from point-source industrial waste disposal, runoff from areas of impervious land, and from industrial emissions generated both within and outside the watershed. On Cape Cod, there is minimal industrial land use, and the industries that are present are not of the type that lead to severe ecological effects or contamination. Disposal of hazardous materials and sewage treatment plant waste at MMR have resulted in 14 groundwater plumes that are contaminated with chlorinated solvents, fuel constituents, and high concentrations of nutrients.

Marine activities. Stresses associated with marine activities include fuel leaks and chemical contamination from boating activities. Physical alteration of habitat could result from anchors and boat propellers. Stressors also are associated with the construction and operation of marinas, docks, and piers, all of which disturb sediments, alter habitats, and contribute chemical contaminants to the water. Dredging and shoreline modification in the form of jetty and breakwater construction can also disturb sediments and alter estuarine circulation patterns. Finally, over-harvesting of stocks can result in the depletion of fishery resources.

4.1.2. Selected Stressors

From the descriptions of the potential sources of stress reviewed above, the workgroup selected six stressor categories that potentially affect the Waquoit Bay watershed to consider for further study:

1. *Chemical pollution* (from pesticide and herbicide application, emissions, industrial point sources, and boating activities)

2. *Pathogens* (from industrial point sources, runoff from impervious surfaces, and leaching of wastewater from septic systems)
3. *Altered freshwater flow* (from new construction and potential increased use of centralized wastewater treatment)
4. *Nutrient enrichment/eutrophication* (from agricultural and lawn and garden fertilization, leaching of wastewater from septic systems, industrial point sources, and atmospheric deposition of emissions)
5. *Physical alteration of habitat* (from dredging and boating activities)
6. *Fishing/shellfishing* (from harvest pressure from commercial and recreational fishing)

4.2. ASSESSMENT ENDPOINTS

Once the six potential stressor categories were identified, the next step was to select assessment endpoints to establish the link between scientifically measurable endpoints and the objectives identified by the resource managers (Suter, 1989; Suter, 1993). Assessment endpoints are defined by a valued ecological entity (e.g., finfish) and the specific attributes of interest (e.g., diversity and abundance). Specifically, endpoints should be ecologically relevant, they should be related to the previously defined management objectives, and they should be susceptible to stressors. The workgroup selected several assessment endpoints that could be potentially affected by stressors in Waquoit Bay. Each endpoint is related to several of the original management objectives (Table 4-1). The endpoints and the reason for their selection are detailed below.

Estuarine percent eelgrass coverage. Eelgrass is a rooted vascular plant that is a critical habitat to a variety of finfish and shellfish (Heck et al., 1989; Thayer et al., 1984) and is dependent on sufficient light penetration for its survival. Increases in nitrogen stimulates the growth of phytoplankton and macroalgae, thereby limiting the amount of light exposure to eelgrass beds. Because the Waquoit Bay system is groundwater dominated, suspended sediments do not contribute significantly to water turbidity to cause an effect on eelgrass productivity. Macroalgae are a major component of shallow estuaries and they respond impressively to nutrient enrichment. Biomass and composition of macroalgal canopies may become dominant features that restructure the benthos of nutrient enriched estuaries (Valiela et al., 1997a; Hauxwell et al., 1998, 2001) and block light penetration, effectively shading out the eelgrass. Thus, eelgrass is highly sensitive to the decrease in light availability that results from nutrient enrichment and it is an effective indicator of eutrophication.

Finfish diversity and abundance. Finfish are susceptible to the dissolved oxygen concenration (DO) in the water, with the DO levels reflecting the balance between daytime photosynthesis and the respiration of decomposing organic matter. The minimal DO levels which stress fish often occur just before dawn as the result of nighttime respiration and limited reaeration of the bottom waters by physical mixing processes.

Table 4-1. Relationship between assessment endpoints and management objectives

Assessment endpoint	Management objective number									
	1	2	3	4	5	6	7	8	9	10
Estuarine percent eelgrass coverage		X	X	X	X		X	X		
Finfish species diversity and abundance	X	X	X	X			X	X		
Scallop abundance	X	X	X	X	X		X	X		
Anadromous fish reproduction		X	X						X	
Piping plovers and wetland bird distribution and abundance (both relate to migratory bird habitat)[a]		X								X
Fish and shellfish tissue contamination		X				X				

[a]The migratory bird assessment endpoint includes both piping plovers and wetland bird species. These endpoints address the same management objectives, and so are combined here, but they have different stressors, and so are considered independently in the stressor and endpoint matrix.

Scallop abundance. Bay scallops were once a commercially important invertebrate in Waquoit Bay. However, because of their dependence on eelgrass beds, the decline in scallop populations has reduced their commercial importance (Thayer et al. 1984). Bay scallop larvae attach to seagrass blades because they prefer the slower currents induced by the eelgrass meadows and it provides protection from predators (Shumway, 1991). Thus, bay scallops are susceptible to losses of eelgrass habitat. They also are susceptible to the periodic anoxic events that occur as a result of eutrophication. In past anoxic events in Waquoit Bay, there has been significant scallop mortality (D'Avanzo and Kremer, 1994).

No surveys have been conducted in Waquoit Bay to measure the assessment endpoint of scallop abundance directly, so scallop harvest was used as a proxy variable to estimate scallop abundance. This is based upon the relationship of $C=qfN$ where C is the catch, f is the effort, q is the catchability (assumed to be a constant), and N is the abundance. This equation simplifies to $N=C/f$ or the relative abundance is estimated from the catch per unit effort. In our case, the only available information on effort was the number of permits issued, which has remained relatively constant over time. Data were available on landings (in bushels) but not the catch rate (landings plus discarded, undersized shellfish). It would have been preferable to measure effort in days fished, but such information was not available. Thus, an approximation of catch per unit of scallop harvest effort was used as a surrogate measure of effect for the assessment endpoint, scallop abundance.

Anadromous fish reproduction. Anadromous brook trout and alewife (river herring) are commercially important species that use the Waquoit watershed as a breeding ground. These species depend on swiftly flowing water that is high in dissolved oxygen.

Piping plover and wetland bird habitat distribution and abundance. Many avian species nest or forage on the beaches and wetlands of Waquoit Bay. As urbanization pressure increases, these species and other coastal wildlife will be threatened because of habitat loss. In the Waquoit watershed two barrier beach birds, the piping plover (endangered) and the least tern (threatened), are listed under the Endangered Species Act, although no species of wetland birds are currently listed. Because the habitats of barrier beach birds and wetland birds vary, both were considered as assessment endpoints.

Tissue contamination of fish and shellfish. Tissue contamination of fish and shellfish was a concern voiced by many stakeholders in the watershed. Toxic plumes emanating from MMR carry trace quantities of carcinogenic substances. Although there is little evidence that these contaminants have been incorporated into the food web of aquatic organisms, the possibility of contamination was sufficient for the working group to list tissue contaminant concentration as an assessment endpoint. Additionally, bacterial contamination of estuarine water has led to some shellfish bed closures, resulting in further concerns about water quality.

These endpoints overlap broadly and may not be mutually exclusive, but they served to start the process of evaluation of risk. In addition to the endpoints just defined, the workgroup initially considered other assessment endpoints, including freshwater pond trophic status and freshwater stream benthic invertebrate diversity and abundance. Freshwater pond trophic status is addressed in more detail by AFCEE (2001). The authors later combined stream benthic invertebrate diversity and abundance with anadromous fish reproduction. The authors eventually opted to focus on the latter because of the ecological and economic importance of those fish species.

4.3. ANALYSIS PLAN

To formulate an analysis plan, the authors began by ranking the comparative risk of each stressor on each endpoint. To further analyze these interactions, each stressor was ranked based on its intensity, extensiveness, and likelihood of increase over time.

4.3.1. Comparative Risk Ranking

The workgroup performed a comparative risk ranking that reviewed the stressors in terms of their potential risk to all resources in the watershed, based on best professional judgment and a fuzzy set decision-making procedure (Harris et al., 1994). The analysis involved ranking the stressors by their impact on the freshwater and estuarine assessment endpoints (Table 4-2). Through consensus of the workgroup, each stressor endpoint pair was given a score ranging from minimal effect (1) to severe effect (5). Scores were summed across endpoints to develop a cumulative ranking for each stressor. The comparative analysis of the cumulative ranking suggests that nutrient enrichment, with a total score of 22, is likely to have the largest potential aggregate effect.

Table 4-2. Effects matrix for the Waquoit Bay watershed[a]

Stressor	Assessment endpoints ranking							
	Percent eelgrass coverage	Finfish diversity abundance	Scallop harvest	Anadromous fish reproduction	Wetland birds	Piping plovers	Fish/ shellfish contamination	Totals
Chemical pollution	1	1	1	1	1	1	3	**9**
Altered freshwater flow	1	1	1	2	3	1	1	**10**
Nutrient enrichment	5	5	5	3	2	1	1	**22**
Physical alteration of habitat	2	1	2	1	2	3	1	**12**
Fishing pressure	1	1	2	3	1	1	1	**10**
Pathogens	2	1	1	1	1	1	3	**10**

[a]Each cell represents the relative effect of a stressor on an endpoint. The ranking (1=minor, 5=severe) reflects experience with the likely effects specifically for the Waquoit Bay watershed.

The ordinal scale used here should be taken with a degree of skepticism. First, some of the endpoints are shown as aggregates (e.g. wetland birds, fish/shellfish contamination), but subcomponents of each aggregate may have quite different responses. For example, shellfish do not respond like finfish to contamination because shellfish are full-time residents of the bay, whereas many commercially important finfish are transient, so they are less likely to be affected by contaminants within the watershed. Additionally, other contaminants come into the estuary from outside the watershed boundaries. The transfer of stressors across watershed boundaries poses a limitation of the ecological risk assessment process, and has an impact on mitigation efforts. For example, no method exists to locally mitigate the effects of atmospheric deposition of nitrogen and mercury, both of which are derived far outside the watershed.

Because there may be differential responses that are lost in the aggregation of endpoints, the authors felt that the comparative analysis in Table 4-2 was not robust enough to form the basis for the risk analysis and risk characterization portions of the risk assessment. To reinforce the conclusion that nutrient enrichment was the dominant stressor, the authors considered a few more features of stressor effects, including intensity and extensiveness, and the likelihood that the source of stress will change across time. These considerations sharpen the focus of the effects of stressors.

Table 4-3. Relative importance of identified stressors to the Waquoit Bay ecosystem[a]

Stressor	Intensity	Extensiveness	Likely increase overtime	Risk ranking
Chemical pollution				
Heavy metals	1	1	1	3
Chlorinated solvents	1	2	1	4
Altered freshwater flow	1	2	2	5
Nutrient enrichment				
Nitrogen	5[b]	5	5	15
Phosphorus	4[c]	2	2	8
Physical alteration of habitat				
Propeller Scarring (Boats)	2	1	3	6
Shading of Benthos (Docks and marinas)	1	2	2	5
Benthic Disturbance (Dredging)	3	1	1	5
Fishing pressure				
Shellfish	3	2	3	8
Finfish	1	2	2	5
Pathogens				
Public health	1	1	1	3
Marine plant disease	2	1	1	4

[a]Ranking is based on available evidence and local experience (1=minor effect, 5=severe effect).
[b] Intensity in estuaries.
[c] Intensity in fresh-water bodies.

4.3.2. Relationships Between Stressors and Ecological Responses

Next, each stressor in the context of the intensity of the effect on the assessment endpoints, the extensiveness of the stressor in time and area throughout the Waquoit Bay ecosystem, and the likelihood that the source of stress will change over time is evaluated. The extensiveness of a stressor was only considered if the intensity was strong enough to have some noticeable impact. Thus, if the intensity of a stressor did not surpass a certain threshold as defined by evidence of impact, it was not deemed significant enough to warrant consideration under extensiveness. The results of the evaluation and a risk ranking of the identified stressors are presented in Table 4-3.

4.3.2.1. *The Importance of Stressors in Waquoit Bay*

Chemical pollution includes two categories of potential stressors to the Waquoit Bay watershed: heavy metals and chlorinated and other hydrocarbons. Some heavy metals (methylated forms) can accumulate in the tissues of organisms and can be passed on to predators (Riisgard and Hansen, 1990). Potential sources of heavy metal contamination in Waquoit Bay include atmospheric deposition of mercury from industrial processes (Golomb et al., 1997), lead residue from paints and from past use of leaded gasoline (Legra et al., 1998), and tributyltin, which is used in antifouling paints for boats, docks, and marinas (Wuertz et al., 1991). Land use in the Waquoit Bay watershed is predominantly residential, with very little industry. There is no available evidence to suggest that concentrations of heavy metals have increased to toxic levels in these waters, thus heavy metals receive the lowest ranking for intensity. Since there is no evidence of an impact from heavy metal contamination, the threshold for consideration of extensiveness was not reached. Additionally, there is no point-source effluent entering Waquoit Bay resulting in a small local contribution of these contaminants and a low extensiveness ranking. Because heavy metals are now regulated, there is no reason to believe that there will be an increase in their concentrations with time.

Chlorinated solvents are present at rather low concentrations in some of the plumes emanating from MMR, and some of the plumes are moving under the Waquoit Bay watershed. These solvents are being treated under the direction of AFCEE (Appendix E). The goal of the mitigation effort is to lower the concentrations of these solvents to below the maximum allowable contaminant levels (0.002 ppb for ethylene dibromide, one of the fuel-related contaminants). Volatile organic carbons (VOCs) are present only at rather low concentrations, and they are highly volatile, so even if they reach Waquoit Bay they are not expected to enter the food webs. The authors therefore suspect that the consequent effects of these compounds will be slight, if measurable at all. As with heavy metals, other sources of organic contaminants are heavily regulated, so there is no reason to believe concentrations will increase in the future.

Altered freshwater flow can occur through the construction of flow-control structures in streams, through removal of groundwater from the aquifer, or through an increase in the proportion of impervious surfaces in the watershed. Although flow-control structures and domestic and public water supply wells could have an intense effect if they were constructed, they are so closely regulated as to not warrant consideration as a potential stressor to Waquoit Bay. The removal of aquifer water via domestic water wells is a somewhat more extensive occurrence, as some of the houses in the watershed have drinking wells. Currently, most of the wastewater released from buildings (including houses) on the watershed is treated with the use of on-site septic systems. With septic systems, the wastewater recharges the aquifer through direct leaching of the wastewater into the ground. There is one small waste treatment plant at MMR that discharges sewage waste outside the watershed. If centralized waste treatment becomes the dominant form of wastewater removal instead of septic use, the aquifer will no longer be recharged with the leachate, and freshwater flow would be impacted. Additionally, with increased urban development, the proportion of impervious surfaces in the watershed would rise, resulting in decreased groundwater recharge and increased overland runoff. Unless artificial recharge of groundwater with treated wastewater or excess surface water is implemented to replenish overdrafted aquifers, the alteration of freshwater flow has a moderate likelihood of increasing across time.

Nutrient enrichment, which is the addition of nutrients to aquatic systems that results in eutrophication, takes into consideration the effects of both nitrogen and phosphorus enrichment. Because the supply of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus determines rates of primary production in estuaries and ponds, these nutrient concentrations may be a telling indicator of eutrophication (Valiela et al. 1992, 2000b; Tomasky et al., 1999; Foreman et al., submitted). There are two primary sources of nutrients in the water columns of ponds and estuaries. The nutrients can be either new (or allochthonous), meaning that they are being discharged from land into the receiving water, or recycled (or autochthonous), meaning they are recycled within the water column from decaying plants and algae and from waste excreted by water column organisms, or they can be regenerated from the sediments. Our focus is on allochthonous nutrients since in the Waquoit system, neither dissolved oxygen in the water nor loss of subtidal habitat have significant impacts on nutrient concentrations in the water when compared to the effect of land-derived nutrient loading (LaMontagne and Valiela, 1995). Because nitrogen and phosphorus are the nutrients that limit primary production in salt water and fresh water, respectively, the relative risk posed by these stressors to the ecosystem are ranked accordingly.

Nitrogen is limiting to primary producers in most coastal waters (Howarth, 1988). The intensity of nitrogen overenrichment results in increases in primary production and the eutrophication of lakes and streams, an effect that has been well documented in reports on Waquoit Bay (Valiela et al., 1992, 1997b) and in other scientific literature (Duarte, 1995; Nixon, 1995; NRC, 2000). Previous research in Waquoit Bay has shown that nutrient overenrichment has resulted in the eutrophication of some subestuaries of the Bay (Valiela et al., 1992, 1997b), thus the highest ranking possible is assigned for both intensity and extensiveness. Finally, regulation of nonpoint-source nitrogen is underdeveloped, so the potential for increasing rates of supply of nitrogen to coastal waters is high.

Phosphorus tends to be limiting to primary production in freshwater systems (Howarth, 1988). The effects of phosphorus enrichment in freshwater systems are similar to those of nitrogen in estuaries: they both result in increased primary production and eutrophication. In the Waquoit watershed, the intensity of phosphorus enrichment is similar to that of nitrogen. Phosphorus, however, is unavailable under aerobic conditions because it is immobilized in sediments (Stumm and Morgan, 1981). When sediments become anaerobic, phosphorus may be released and move downstream (Krom and Berner, 1980). Because the sediments in the Waquoit system are very sandy, there is virtually no overland flow of water. As a result, the sources of phosphorus enrichment are not fertilizers from lawns and crops, but wastewater systems that allow phosphorus to seep into the groundwater. Most groundwater, however, is aerobic; therefore the extensiveness of phosphorus enrichment is not as great as that of nitrogen. The largest exogenous source of phosphorus was the use of detergents, much of which is now regulated, so the likelihood of increases in phosphorus concentrations is small.

Physical alteration of aquatic habitat can result from three major pathways: boating activities, docks and marinas, and dredging. Boating activities produce a moderately intense—but extremely localized—impact on the Waquoit Bay watershed. Propeller scarring can significantly disturb the sediments of shallow coastal waters, but this occurrence is infrequent and is predominantly seasonal. Likewise, fuel spills from boat motors also are seasonal. Thus, boating activities receive a moderate ranking for intensity and the lowest ranking for extensiveness. With increasing human populations on coasts, the number of boaters and the frequency of occurrence of boating incidents likely will increase.

Docks and marinas impact coastal waters through shading of benthic communities and through the introduction of toxins from antifouling agents, especially copper, chromium, and arsenic. Docks have been shown to have a minor impact on benthic fauna (Butler and Connolly, 1996), but they can have some impact on eelgrass beds (Short and Burdick, 1996). Although the use of private docks is moderately extensive, they are now closely regulated and are not likely to increase in the future.

Dredging results in intense physical disturbance of benthic communities and it can alter estuarine circulation. In Waquoit Bay, however, there are very few areas that could be dredged, so the authors assigned a low rating for extensiveness. As with the construction of docks and marinas, dredging is highly regulated and is not likely to increase in the future.

Fishing pressure in Waquoit Bay can potentially affect stocks of shellfish and finfish. Shellfishing is the dominant of the two fisheries in the bay. The shallow waters make it an ideal location for commercial/recreational and subsistence fishing for quahogs, scallops, and softshell clams, all of which are extensively harvested. Shellfishing can result in substantial disturbance to the benthic community of estuaries, so the authors consider it a moderately intensive activity. Continued growth in the coastal zone likely will cause increases in harvest pressure in the near future.

In Ashumet Pond many of the native fish are planktivores or detritivores, and the fish community has changed over time for a variety of reasons (e-mail dated 7/9/02 from S. Hurley, MA Fisheries and Game Division to David Dow, Woods Hole Laboratory). Several of the piscivores targeted by recreational fishers (brook, rainbow and brown trout, and smallmouth bass) are stocked by the state, with the predominant native species being yellow perch, brown bullhead, chain pickerel, and banded killifish and the introduced species being represented by alewife and green sunfish. Possible evidence for impacts of phosphorus enrichment can be seen in the loss of the original native brook trout and the increases in brown bullhead in recent years, but stocking and introductions make it difficult to attribute changes in the fish community to one factor. The proposed mechanism is that bottom water anoxia/hypoxia prevents cold-water fish, such as the stocked brown trout, from surviving over the summer, but many of the native warm-water fish can move up from the bottom waters during the summer and escape the anoxia.

Changes in the fish species community composition over a 20-year time period (1967 to 1987) showed that 20 species exhibited decreased abundance while only 7 species increased in abundance (Deegan and Buchsbaum, in press). In general, the piscivores with an offshore distribution (cunner, pollock, white hake, winter flounder) exhibited the greatest declines, while the inshore resident, planktivorous/benthivorous forage fish species (rainbow killifish, blueback herring, three-spined and four spined stickleback) showed increases. The offshore species use the estuary as a nursery area and since many of the piscivores are targeted by recreational users and commercial fishing operations, these species can be impacted by both offshore harvesting and inshore habitat degradation in their nursery areas. Deegan and Buchsbaum (in press) attribute a greater relative role for these declines in abundance to habitat degradation, loss of wetlands and submerged aquatic vegetation, low dissolved oxygen concentrations, toxic pollutants; or nutrient enrichment, while Link and Brodziak (2002) attribute a greater role to regional commercial and recreational fisheries harvesting. Since offshore and estuarine systems frequently mix, it is not easy to delineate the relative importance of fisheries harvesting and habitat degradation on these declines in the abundance of piscivores. The causes for increases in the abundance of resident fish species are not well understood, but probably reflect changes in the abundance of piscivores inshore, alteration in the predation pressure, and changes in competition from other resident forage fish species that are negatively impacted by inshore habitat degradation.

Habitat loss could remove valuable nursery grounds for the many species of fish that depend on macrophytes for their survival. It also has been theorized that replacement of native marsh species with the invasive *Phragmites* may reduce the habitat quality for finfish that feed on invertebrates in marsh habitats (Weinstein and Balletto, 1999). The impact of *Phragmites* invasion in Waquoit Bay ponds and estuaries has not been evaluated.

Finfishing in Waquoit Bay is not as intensive as shellfishing, and hence the authors assigned it a lower ranking. At the 1993 meeting, much concern was expressed about the abundance of key recreational species, such as bluefish and striped bass; however, most of the species are over-fished at the regional level by commercial and recreational fishers outside the watershed. As with shellfishing, finfishing is likely to increase in the future.

Pathogens in the environment can be divided into two categories: public health and plant. Public health pathogens have a variety of forms, all of which are estimated by using fecal coliform concentration as a proxy for the extent of contamination caused by wastewater input. Although fishing beds are occasionally closed in some of the estuaries of Waquoit Bay because fecal coliform loadings exceed the limits for seafood consumption standards for shellfish, none of the area beaches have been closed to swimmers, indicating that the contamination is not intense enough to cause human heath concerns. Because of the regular monitoring of shellfish beds, it does not appear that shellfish exposure to public health pathogens in estuaries will increase over the next several decades. Additionally, many pathogens that are considered to be threatening to public health have no effect on the invertebrates that carry them; thus, they are outside the purview of an ecological risk assessment, but they should be considered in a human health risk assessment.

Plant pathogens have appeared sporadically in the past. Specifically, slime mold *Labyrinthula* (Short et al., 1988), which infects eelgrass beds, destroyed much of the eelgrass in Waquoit Bay and elsewhere in the 1930s. Continued spread of this fungus could destroy existing eelgrass meadows, but evidence of the disease does not presently extend beyond a few locations in the Northeast (Short et al., 1986).

4.3.2.2. *Focus on Nutrient Enrichment as the Dominant Stressor*

The aggregate score for nutrient enrichment on the stressor/endpoint matrix in Table 4-2 is 22. None of the other stressors rank as high; in fact all rank substantially lower. The risk ranking in Table 4-3 also identifies nutrient enrichment as the most important stressor in the Waquoit Bay ecosystem, as it has a maximum rating for all three contexts. These results were derived from local experience with the actual effects as well as from an abundance of previously collected data. Because of the overwhelming importance of nutrient enrichment as a stressor to the Waquoit Bay watershed, the authors focused the risk analysis on the various effects of nutrient enrichment on the two endpoints that are linked to the nutrient enrichment stressor: percent eelgrass cover and scallop abundance (as represented by the surrogate measure of the effect of scallop harvests) (Figure 4-1). The other assessment endpoints selected by the workgroup are not directly susceptible to the nutrient enrichment stressor, so the decision was made to focus on these two endpoints in the risk analysis and risk characterization.

4.3.3. Summary of the Analysis Plan

The results of the previous section indicate that the effects of eutrophication on components of the Waquoit Bay ecosystem are of critical concern. In the risk analysis (Chapter 5), quantitative information on the impact of nitrogen loading on each of the critical components is provided. A modeling approach to assess the causes and effects of nitrogen loading in Waquoit Bay was used through a nitrogen loading model (NLM) and an estuarine loading model (ELM). The focus was on nitrogen loading because phosphorus input, although important in the eutrophication of freshwater ponds, is being analyzed and mitigated by AFCEE. AFCEE has a research program in place that aims to mitigate the phosphorus that is being carried in the Ashumet Valley plume (Appendix E).

In Chapter 5, the risk to percent eelgrass cover and scallop harvest is characterized. These endpoints were selected for further analysis because they are the most susceptible to the effects of eutrophication and because sufficient historical information exists that correlative models can be developed to predict their response to increases in nitrogen loading. Many of the other components of the ecosystem, such as phytoplankton biomass and zooplankton dynamics, are too temporally variable to be modeled effectively.

5. RISK ANALYSIS

In the analysis phase, this assessment uses a nitrogen loading model (NLM) to estimate exposure to the primary stressor and an estuarine loading model (ELM) to predict impacts of nitrogen loading on two assessment endpoints. Ecological responses were characterized by comparing effects in three subwatersheds subjected to different nitrogen loads. This Chapter describes the NLM and the ELM and discusses how they were validated and the level of uncertainty in these estimates. The predictions from the NLM and the ELM also are included in this Chapter.

5.1. EXPOSURE ANALYSIS

Measures of exposure take on a new definition when considering risk assessment for a watershed. When conducting a large-scale assessment like this one, it is helpful to consider the exposure of the entire system as a unit (Suter, 1993). For this Waquoit watershed assessment, the measure of exposure for coastal eutrophication is land-derived nitrogen loading to the bay's subwatersheds.

5.1.1. Nitrogen Loads

The next few sections, describe a model that was created and validated against measured data to assess eutrophication exposure. These results may be relevant for estuarine systems that lack the resources to undergo the intensive study that the Waquoit Bay system has received.

5.1.1.1. *Estimates Using the Nitrogen Loading Model*

Researchers working on the WBLMER project developed a method for estimating the magnitude of exposure to nitrogen loading in the form of an NLM that estimates the amount of nitrogen delivered to the estuary on the basis of the land use within the watershed (Figure 5-1, Table 5-1). The model sums the nitrogen from three major sources—atmospheric deposition, septic-derived wastewater, and fertilizer use—and subtracts all of the losses that occur during transport through the various watershed tessera (an individual part of a larger mosaic). Each tesserae is considered as a component along a depth transect from surface to aquifer and is given a loss coefficient for each source of nitrogen in the model. After tracking losses and transformations of the nitrogen, the model yields an estimate of total nitrogen entering the estuary at the seepage face.

The input terms that are required by this NLM are basic land-use data, which can be derived using either geographic information systems (GIS) or aerial photographs. The most essential piece of information needed to accurately determine the extent of each land-use type is accurate watershed delineations. For this watershed the authors used local information on water table contours, hydraulic conductivity, and effective porosity to simulate the tracks of water flow

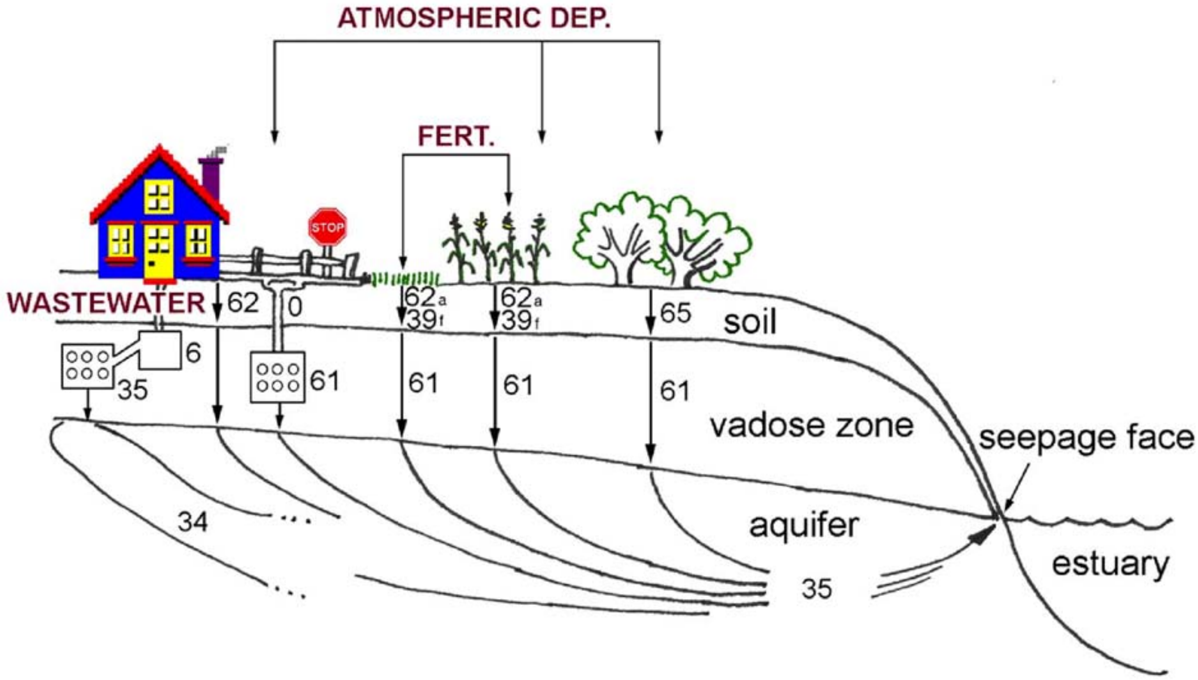


Figure 5-1. Schematic of the nitrogen loading model. Inputs of wastewater-, fertilizer-, and atmospherically derived nitrogen to the watershed and the percent losses (shown as numbers) as the nitrogen from the three main sources percolates through the watershed and enters the estuary at the seepage face.

Source: Valiela et al. (2000a) with permission of Kluwer Academic Publishers.

with a three-dimensional, finite-difference hydrologic flow model called MODFLOW (McDonald and Harbaugh, 1988). Once the watersheds were delineated, the land use within each watershed was estimated from aerial photographs (Brawley et al., 2000). GIS technology has improved markedly since the start of this modeling effort, and watersheds can be delinated and land use tabulated with the latest GIS software.

The specific information on land use required by the model is described in Table 5-1. It is important to note that these land-use categories can, to some degree, be modified for use in other watersheds. For example, cranberry bogs are a dominant agricultural crop on Cape Cod, but the fertilizer inputs from cranberry bogs could easily be replaced by the fertilizer inputs from corn, if corn was the dominant agriculture type in the watershed being modeled. The loss terms used in an NLM are related to the nitrogen that is delivered below the watershed surface via two main routes (Valiela et al., 1997b). Nitrogen derived from atmospheric deposition and fertilizer use percolates to the unsaturated vadose zone after losses in the turf and vegetation (Lajtha et al.,

Table 5-1. Input, loss, and default terms used by the nitrogen loading model (NLM)

Inputs needed to apply the NLM	Losses considered by the NLM	Defaults considered by the NLM
Drainage area	Uptake by forests (65%)	Evapotranspiration (45% of precipitation)
Number of houses	Uptake by the vadose zone (61%)	Nitrogen in deposition (15 kg N/ha/yr)
Area of receiving estuary	Uptake by turf (62%)	Area of lawns (0.05 ha/house)
Area of freshwater ponds	Uptake by freshwater ponds (66%)	Area of roofs and driveways (185.8 and 46.45 m^2/house)
Area of wetlands	Uptake by wetlands (78%)	Area of roads (3% of watershed area)
Area of natural vegetation	Loss in the septic tank (6%)	Occupancy rate (1.79 people/ house)
Area of cranberry bogs	Loss in the leaching field (35%)	Per capita biological nitrogen release (4.82 kg/person/yr)
Area of other agriculture	Loss in the septic plume (44%)	Fertilizer application to lawns (122.33 kg N/ha/yr)
Area of golf courses	Volatilization of fertilizer (38.5%)	Fertilizer application to golf courses (171 kg N/ha/yr)
Area of other types of turf	Loss in the aquifer (35%)	Fertilizer application to cranberry bogs (28 kg N/ha/ yr)
Area of other impervious surfaces		Fertilizer application to other agricultural land (136 kg N/ha/yr)

Source: Valiela et al. (1997b).

1995). Nitrogen from wastewater undergoes losses and transformations as it moves through septic tanks and leaching fields and as it moves through the soil layers in a wastewater plume (Valiela et al., 1997b). Nitrogen from all three sources undergoes losses in the vadose zone and in the aquifer (Valiela et al., 1997b). The specific loss terms used in the Waquoit Bay NLM are summarized next and described in more detail in Valiela et al. (1997b).

<u>Losses considered by the NLM</u>

Uptake by forests — The retention of atmospheric nitrogen delivered to coastal forests in the watershed of Waquoit Bay was determined to be 65%. This estimate was found by comparing concentrations of total dissolved nitrogen (TDN) in atmospheric deposition to those in water collected below the root zone (Valiela et al., 1997b). The estimate of 65% retention of atmospheric nitrogen by coastal forests is lower than the 80 to 90% retention reported for upland forests of New England (Aber et al., 1993), but similar to the 68% retained in a riparian forest in Georgia (Lowrance et al., 1984).

Uptake in the vadose zone — The fate of nitrogen during travel through the vadose zone is not well known (Keeney, 1986; Korom, 1992), although losses of nitrogen during transport through unsaturated vadose layers of sands have been reported in agricultural fields (Cameron and Wild, 1982; Starr and Gillham, 1993). From these data, it was estimated that 61% of the nitrogen that reached the vadose zone of forested or cultivated parcels in the Waquoit Bay watershed was lost in the unsaturated sediments (Valiela et al., 1997b). The NLM therefore calculated 39% of atmospheric nitrogen percolated from forested or cultivated areas of the watershed through the vadose zone. It was assumed that these values were applicable to the other land-cover types, and the calculation was repeated for the remaining land uses to obtain an estimated total loss in the vadose zone.

Uptake by turf — By comparing nitrogen concentrations from the drainage through soil under lawns and a grass field (79.6 mM TDN, from Lajtha et al., 1995), with concentrations in atmospheric deposition (209 mM TDN, Valiela et al., 1997b) onto the Waquoit Bay watershed, losses of 62% of atmospheric-derived nitrogen in turf plants and soils were calculated.

Uptake by freshwater ponds — Freshwater bodies located within coastal watersheds capture groundwater flow and any nitrogen dissolved in that water. The amount of nitrogen that traverses the freshwater bodies and moves downgradient is less than the nitrogen that enters the ponds, lakes, and wetlands. Mass balance studies in lakes, ponds, and various types of freshwater wetlands show that retention of nitrogen entering the waterbodies ranges from 14 to 100% (Valiela et al., 1997b). The median retention of nitrogen in ponds and lakes is 56% (Valiela et al., 1997b). In these studies, retention was underestimated because inputs by dry precipitation were not included.

Uptake by wetlands — Similar to mass balance studies of lakes and ponds, wetlands have been shown to retain a median of 77% of the nitrogen that they receive (Billen et al., 1985; Billen et al., 1991; Johnston et al., 1990; Johnston, 1991; Molot and Dillon, 1993).

Loss in the aquifer — Because there is insufficient published information with which to estimate the magnitude of nitrogen loss in aquifers (Korom, 1992), the loss for Waquoit Bay was estimated using a steady-state approach. The WBLMER data were used to calculate that 35% of diffuse nitrogen entering the aquifer under the Waquoit Bay watershed was lost within the aquifer (Valiela et al., 1997b). This estimate of losses in aquifers was obtained by calculating the percent difference between nitrogen concentrations in groundwater near the water table and groundwater about to enter the receiving estuaries. These data were compiled from samples of groundwater originating under land parcels covered by natural vegetation.

Loss in the septic tank — To ascertain if losses of nitrogen occurred within septic tanks, published nitrogen concentrations in wastewater entering septic tanks were compared to the concentrations in effluent leaving the tanks (Valiela et al., 1997b). Concentrations of total dissolved nitrogen in water entering septic tanks were about 5% higher than those in effluent leaving septic tanks. These results were smaller than the 10 to 20% reductions suggested by others (Andreoli et al., 1979). Even when the septic holding tanks were pumped out, removal averages only 4% of nitrogen entering septic systems (Kaplan, 1991), and this septage is often kept within the watershed through release to septage lagoons.

Loss in the leaching field — The leaching field disperses the water that has drained from the septic tank, allowing it to percolate slowly into the soil. To calculate the loss of nitrogen in the leaching field, two values of estimated nitrogen retention were averaged. One estimate was based on all retention data available to us on the nitrogen content of septic-tank effluent and on effluent from leaching fields. The second estimate used data from a group of 12 papers (e.g. Walker et al., 1973; Reneau, 1979; Starr and Sawhney, 1980) that provided concentrations of nitrogen in both inputs to holding tanks and outputs from leaching fields for a given septic system. The loss in leaching fields obtained from the aggregate data (35%) and the system-specific nitrogen retention (46%), were then averaged to obtain 40% as an estimate of nitrogen losses in septic systems of conventional design. Hence, about 60% of wastewater nitrogen is likely to travel beyond leaching fields and into the septic plume (Valiela et al., 1997b). This could be a true loss caused by denitrification, or merely a dilution effect. However, studies have shown there is little evidence of dilution within the leaching fields of this watershed (Valiela et al., 1997b).

Loss in the septic plume — The water from the leaching field percolates through the vadose zone and into the aquifer. The water moves for a distance in a confined plume of higher concentration known as the septic plume. Concentrations of dissolved inorganic nitrogen in the septic plumes have been found to decrease with distance away from leaching fields (Valiela et al., 1997b). As with losses in the leaching field, such decreases could be a true loss caused by denitrification, or an apparent loss caused by dispersion. Each possibility was examined by comparing decreases of DIN with decreases in concentrations of chloride or sodium when these tracers were also reported. The results showed that chloride concentrations decreased over distance (Valiela et al., 1997b), so dispersive mixing did take place, and actual losses of nitrogen also took place. The mean value of septic nitrogen that may be lost during travel in plumes was 34%.

Volatilization of fertilizer — Gaseous losses by denitrification and volatilization from turf account for about 39% of fertilizer nitrogen (Petrovic, 1990).

Defaults considered by the NLM

The defaults that are used by the NLM are specific for the Cape Cod region where the model was developed. The model is designed in such a way that these defaults can be changed to model another watershed. The derivation of the defaults for the Waquoit Bay watershed are described next.

Evapotranspiration — Regional estimates of evapotranspiration of 45% were derived from Thornthwaite and Mather (1957), Running et al. (1988), and Eichner and Cambareri (1992).

Nitrogen in deposition — Watersheds receive significant amounts of nitrogen from the atmosphere. Deposition of atmospheric nitrogen has varied over recent decades and wide regional differences in chemistry of precipitation have been reported (U.S. EPA, 1982; Shannon and Sisterson, 1992; Davies et al., 1992; Fricke and Beilke, 1992; Pack, 1980; Stensland et al., 1986; Ollinger et al., 1993). Due to these differences, it was necessary to calculate nitrogen input via atmospheric deposition using local deposition data.

Local data on wet deposition (Lajtha et al., 1995) and a calculation of dry deposition were used to estimate total atmospheric deposition to forests. Because local dry deposition data were unavailable, estimates for the ratios of wet-to-dry deposition from published data were calculated, and a mean ratio of 0.9 was obtained (Valiela et al., 1997b). This agrees with Hinga et al. (1991), who concluded that wet and dry deposition were of the same magnitude. Thus, atmospheric deposition was estimated to be twice as high as wet deposition.

Area of lawns — Lawns in suburban to semirural coastal areas such as Long Island and Cape Cod average 0.05 ha in area (Koppelman, 1978; Frimpter et al., 1990; Interdisciplinary Environmental Planning (IEP) 1986; Eichner and Cambareri, 1992).

Area of roofs and driveways — Roofs in suburban to semirural coastal areas such as Long Island and Cape Cod average 185.8 ha in area, and driveways average 46.45 ha in area (Koppelman, 1978; Frimpter et al., 1990; IEP, 1986; Eichner and Cambareri, 1992).

Area of roads — Area roads are estimated as 3 percent of the watershed area (Koppleman, 1978).

Occupancy rate — The total number of residences or buildings within a watershed can be estimated from either aerial photos or GIS data (Lindhult and Godfrey, 1988). The number of people per house can be obtained from municipal records, census data, or by using data from other areas of similar ecological setting and socioeconomic background (Valiela and Costa, 1988; Hayes et al., 1990). In some coastal areas, many houses are occupied only during summer, vacations, or on weekends. In addition, coastal areas such as Cape Cod are home to many retired people who spend the winter in warmer climates. In such cases, it is necessary to adjust occupancy rates. Census data from Cape Cod in 1990 were used to determine the duration of occupancy of individual houses in the Waquoit Bay watershed. Using these data along with GIS-based land parcel data and aerial photos, a weighted mean (± sd) was calculated as 1.8 ± 0.6 people/house/yr. This value is similar to the seasonally adjusted value of 1.9 people/house/yr suggested for an area where 23% of the houses were used only during the summer (Weiskel and Howes, 1991).

Per capita biological nitrogen release — On average, 4.8 kg N/person/yr are released, with a range of 1.8 to 5.4 (Valiela et al., 1997b). These values are similar to those found by others (1.8 to 7.4 kg N/person/yr) as observed by Vollenweider (1987), Koppelman (1978), and U.S. EPA (1980).

Fertilizer application to lawns — Mean fertilizer use applied to lawns in suburban areas such as Cape Cod average 122.33 kg N/ha/yr (Koppleman, 1978; Nelson et al., 1988).

Fertilizer application to golf courses — The average nitrogen fertilizer content applied to golf courses is 171 kg N/ha/yr (Frimpter et al., 1990; Eichner and Cambareri, 1992; Petrovic, 1990).

Fertilizer application to cranberry bogs — Fertilizer use on cranberry bogs was reported in Valiela et al. (1978).

Fertilizer application to other agricultural land — Calculations of nitrogen loading from agricultural fertilizers should use local application rates (Loehr, 1974; Stanley, 1988; Frimpter et al., 1990; Hayes et al., 1990; Correll et al., 1992) as much as possible. In suburban areas, where agricultural land use is at a small, local scale and crops are mixed, an average of about 136 kg N/ha/yr may be appropriate (average obtained from data in Loehr, 1974; Stanley, 1988; Hayes et al., 1990; Correll et al., 1992).

5.1.1.2. *Measurements*

Nitrogen loading rates to the Waquoit Bay estuarine complex were measured by obtaining groundwater samples roughly every 50 meters from the periphery of each of the subestuaries in the bay (Valiela et al., 2000a). The timing and frequency of groundwater sampling varied by subestuary, but the differences did not affect the analyses since there appears to be no seasonal variation in groundwater concentrations (Table 5-2). Using standard methods, the samples were analyzed for concentrations of NO_3 (QuikChem® method 31-107-06-1-C), NH_4 (QuikChem® method 31-107-04-1-C), and DON (modified from D'Elia et al., 1977) (Valiela et al., 2000a). The subwatersheds were divided into smaller recharge zones to prevent differences in groundwater flow or the number of samples taken in each recharge zone from biasing the estimates of nitrogen loads. These zones were delineated from land surface features and hydrological flow lines using the MODFLOW groundwater transport model (McDonald and Harbaugh, 1988). The authors used the area of each recharge zone, the average annual precipitation rate, and regional estimates of evapotranspiration to estimate the volume of water that flows through the aquifer and into the estuary. The volume of water and the average concentration of nitrogen in the groundwater samples taken within each recharge zone were multiplied to determine the total nitrogen load (Valiela et al., 2000a).

As shown in Table 5-2, the measured nitrogen loads to the subestuaries of Waquoit Bay are extremely variable, ranging from 433 to 9879 kg N/yr (Valiela et al., 2000a) and are determined in part by the size on the land parcel. The nitrogen load to the entire bay was more than 26,500 kg N/yr.

5.1.1.3. *NLM Validation*

Modeled predictions of the quantity of nitrogen entering the estuary were validated in two independent ways. First, the NLM predictions were verified against actual measurements of nitrogen in groundwater about to enter estuaries (Figure 5-2) (Valiela et al., 2000a). The authors quantified the performance of the NLM by comparing a suite of statistical features that were derived from the verification plot. Four statistical inferences were used to quantify performance.

Table 5-2. Measured nitrogen loads to the subestuaries of Waquoit Bay

Land parcel	Land area (ha)	No. of groundwater samples	Mean total dissolved nitrogen (µM)	Nitrogen load (kg N/yr) ± SE
Childs River	866	386	313	8116 ± 22.4
Quashnet River	2055	232	60	9879 ± 11.0
Eel Pond	354	89	164	4502 ± 20.4
Hamblin Pond	260	46	44	893 ± 5.6
Jehu Pond	422	78	62	1968 ± 9.4
Sage Lot Pond	119	191	90	990 ± 8.8
Head of the Bay	92	37	60	433 ± 6.7
Waquoit Bay watershed	3788	1059	85	26,781 ± 9.6

Source: Modified from Valiela et al. (2000a).

Second, *r* was calculated, which evaluated the precision of the model by assessing the degree of scatter around the regression line between measured and modeled data. Third, the accuracy of the model was tested by using a t-test between the slope of the regression line and the 1:1 line of perfect fit. Finally, the authors examined the R^2 term that, based on Prairie (1996), gives a measure of the predictive ability of the model.

The NLM was extremely responsive to measured data, based on highly significant F_{reg} values (116.6, p=0.01). In addition, the NLM precisely predicted measured loads, with a very small degree of scatter around the regression line, as demonstrated by highly significant *r* values (0.97, p=0.01). The slope of the regression line comparing model estimates to those that were measured in the field could not be statistically distinguished from the 1:1 line of perfect fit, indicating that the model is highly accurate (t=1.79, not significant). Finally, the R^2 value was substantially higher than the 0.65 proposed by Prairie (1996) as the threshold for the predictive ability of regression models. The results of these four statistical features indicate that despite the associated uncertainty, the NLM accurately represents the processes involved in land-derived nitrogen loading and can thus be used effectively as a measuring tool (Valiela et al., 2002).

The second verification of the NLM assessed the ability of the model to accurately predict the dominant source of nitrogen to the estuary. The NLM was used to predict the percentage of nitrogen derived from wastewater and compared this prediction to the $\delta^{15}N$ signature of the groundwater measured in the subestuaries of Waquoit Bay (McClelland and Valiela, 1998). The percent of nitrate as $\delta^{15}N$ derived from wastewater is higher than the percentage derived from

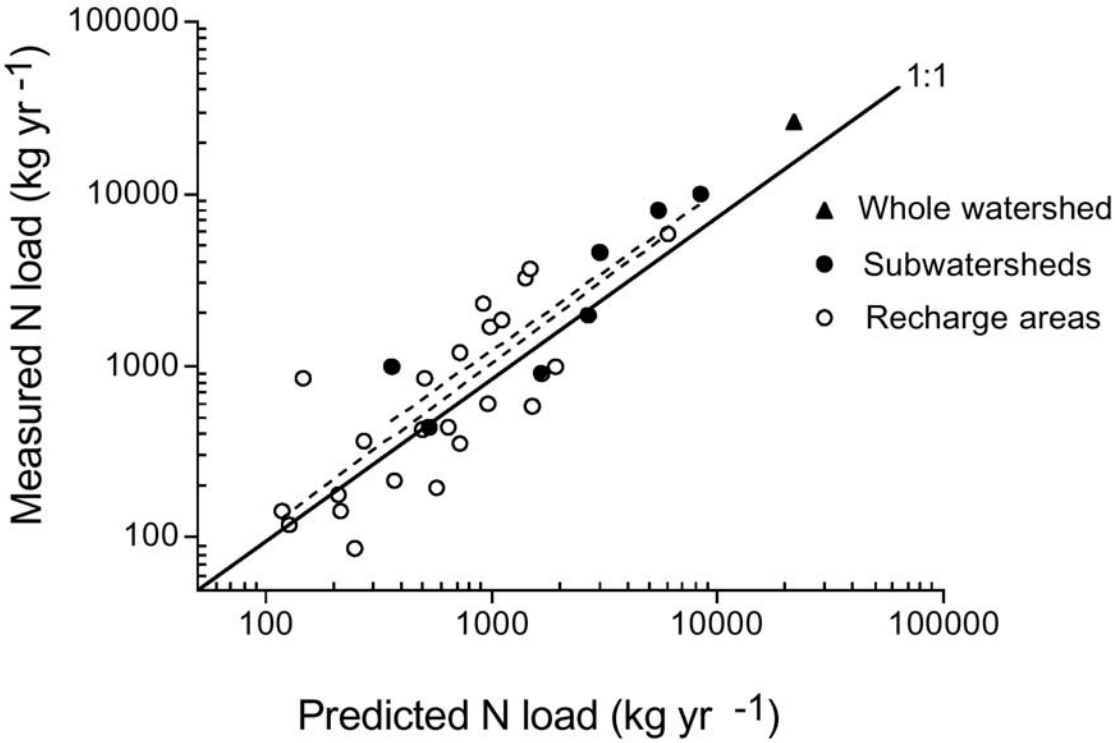


Figure 5-2. Measured versus modeled nitrogen (N) loads to Waquoit Bay. Plotted against a one-to-one line of perfect fit.

Source: Valiela et al. (2000a) with kind permission of Kluwer Academic Publishers.

fertilizers and from atmospheric deposition (Table 5-3), making it possible to verify the percentage of nitrogen derived from wastewater predicted by the NLM. Each watershed has land uses that contribute different proportions of nitrogen derived from the three main sources; these mixes are sufficiently distinct and can be linked to nitrogen loads entering the estuaries. The high correlation between the $\delta^{15}N$ signature of nitrate measured in groundwater and the proportion of groundwater nitrogen derived from wastewater as calculated by the nitrogen loading model, helps verify that the NLM can describe the proportion of nitrogen coming from a given source (Figure 5-3).

The reported results on $\delta^{15}N$ as tracers of land-derived nitrogen are not isolated examples. Hansson et al. (1997) showed that isotopic signatures in organisms in different areas of the Baltic Sea reflected the nitrogen contributions from sewage, and the signatures were pervasive in the food web up into the top predators. Udy and Dennison (1997, 1998) showed that isotopic signatures in macrophytes identified sources and loading of nitrogen to Australian estuaries.

Table 5-3. Sources and δ^{15}N values of nitrate in groundwater

Source	δ^{15}N (%) of nitrate	References
Wastewater[a]	10 to 20	Kreitler (1975), Kreitler and Jones (1975), Gormly and Spalding (1979), Aravena et al. (1993)
Atmospheric deposition	2 to 8	Kreitler (1975), Kreitler and Jones (1975), Gormly and Spalding (1979)
Fertilizer	-3 to 3	Kohl et al. (1973), Freyer and Aly (1974), Mariotti and Lètolle (1977), Macko and Ostrom (1994)

[a]Includes nitrate derived from both human and animal waste.

Unpublished data from Brazilian tropical coastal lagoons show that 2 to 6% heavier isotopic signatures in macrophytes were collected from a lagoon with densely populated villages on its shore, relative to macrophytes collected far from human populations. Water samples collected in coastal lagoons on the coast of Sinaloa, Mexico, revealed that δ^{15}N values of ammonium were about 2 to 3% in areas remote from people but increased to 21 to 29% in water near wastewater sources. Similar trends were found in water as well as macrophytes collected from freshwater ponds and coastal lagoons in other Cape Cod regions. Thus, the isotopic approach furnishes both an independent verification that an NLM can accurately describe the proportion of stressor coming from wastewater, and that it can provide a means to estimate the incorporation of the stressor on the biota in a variety of estuarine systems.

5.1.1.4. ***NLM Uncertainty***

Uncertainty is a necessary component of every model. Traditional means of addressing issues of uncertainty involve estimating the variation in replicate samples. Since it is impossible to have replicate watersheds, the uncertainty of the NLM was estimated using a bootstrapping method in which the loading calculation was obtained using 2,000 estimates of loads, each of which was based on a different set of means. The calculations were based on the set of observed values used for each algorithm variable. Then, the 2,000 bootstrapped means were used to calculate measures of variation (Table 5-4). The standard error of the mean for the NLM, as calculated by the above bootstrapping method, was 12%, and the standard deviation of the mean was 37% (Collins et al., 2000). The error was also propagated and similar results were found.

Uncertainty in model estimates can be introduced in three ways: statistical uncertainty, inherent uncertainty, and model uncertainty (Collins et al., 2000). Additional data reduce the level of uncertainty in any model, but information is frequently difficult or expensive to acquire. One form of sensitivity analysis is a technique based on Monte Carlo simulations that gives an indication of the parameters of the model that are the most uncertain, thereby providing a means

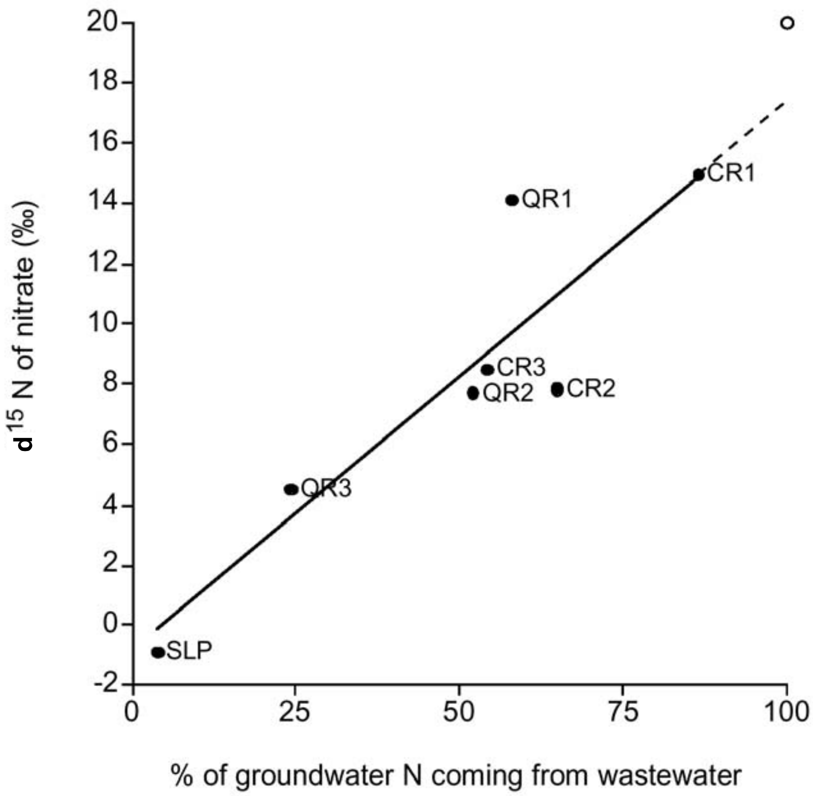


Figure 5-3. Values of δ^{15}N of nitrate in groundwater versus percent of nitrogen (N) from wastewater. The relationship of the δ^{15}N signature of nitrate measured in groundwater of the Waquoit Bay estuarine system to the proportion of groundwater nitrogen (N) derived from wastewater was calculated by the nitrogen loading model. Each point represents a recharge zone of Waquoit Bay. The open circle is upper value provided by the literature for δ^{15}N of wastewater-derived N; the dashed line shows the extrapolation for data from Waquoit Bay. F=22.6, *p*=0.01, r=0.91.

Source: Valiela et al. (2000a) with kind permission of Kluwer Academic Publishers.

for focusing research on the terms that most strongly influence the model output. Such analysis indicated that eliminating the uncertainty in any one variable would result in small changes (Collins et al., 2000). The analysis demonstrated that the error associated with the amount of nitrogen released per person per day accounted for the majority of the uncertainty in the final loading estimate. However, lowering the uncertainty in load estimates required lowering the uncertainty in several other model variables.

The effects of temporal variability also could introduce uncertainty into the model results. It takes longer to feel the impact of land-use changes that occur farther back in the watershed

Table 5-4. Error analysis of NLM variables

Method	Specific measures of uncertainty (%)			
	All measures of uncertainty	Septic efficiency	Attenuation in plumes	Nitrogen release by humans
Propagation of standard error				
Standard error	14	13	12	11
95% CI	73–127	75–125	77–123	78–122
Bootstrapping				
Standard error	13	11	10	10
95% CI	78–136	80–125	81–122	82–122

Source: Data from Collins et al. (2000).

than it takes for those occurring on the banks of the estuary, because of the length of time it takes the groundwater to travel. To address this uncertainty, a dynamic version of the NLM was developed (Brawley et al., 2000). This dynamic model incorporates spatial and temporal trends in land use into the NLM framework. The results of this exercise indicate that the variability associated with the lag due to groundwater travel was less than the uncertainty in the model, therefore, it is concluded that temporal variability is relatively small.

The problem of uncertainty associated with model development is not unique to the NLM. In fact, every model designed to approximate the impact of land use on the nitrogen loads of estuarine systems is plagued by the same dearth of data. The difference between the NLM and
other models designed to predict nitrogen loads and concentrations is that the NLM is verified against actual measured data. It seems plausible that, despite the associated uncertainty, if a model can reasonably predict nitrogen loads that are occurring in the environment, then it has served a valuable purpose.

5.2. EFFECTS ANALYSIS

The effects of nitrogen loading on the ecosystem components outlined in Table 4-4 was measured to look at changes occurring throughout the Waquoit Bay watershed. For our purposes, nitrogen load is treated as the exposure, and nitrogen concentration is an effect of that load. Thus, a discussion of the cascading effects from nitrogen loading begins with a description of the nitrogen concentration in the estuary. The analysis of the cascading effects is followed by a characterization of the effects on the two assessment endpoints upon which the risk assessment focuses (Section 5.2.2).

A space-for-time substitution (Pickett, 1989) was used to compare estuaries that differ spatially in the degree of exposure to nitrogen loading as a proxy for the loading change that will occur in the ecosystem over time. Because different subestuaries within the Waquoit Bay watershed have been urbanized at different rates, the situation was ideal for establishing ecosystem changes that could occur over time without having to conduct long-term ecosystem studies. The three estuaries that are most frequently used in this comparison are the nearly pristine Sage Lot Pond, the moderately urbanized Quashnet River, and the highly urbanized Childs River watershed (Figure 2-1, Table 5-5). Next, empirical data from these subestuaries were used to show the responses of the ecological endpoints to land-derived nitrogen loads.

5.2.1. Cascade of Effects on Ecosystem Components

5.2.1.1. *Nitrogen Concentrations*

In this assessment of Waquoit Bay, land-derived nitrogen load is the exposure that is studied. This section describes the first effect of that loading as the measured nitrogen concentration. The use of the measured nitrogen to validate the Estuarine Loading Model (ELM), which predicts the concentrations of nitrogen available to primary producers, is also demonstrated.

Measurements – Nitrogen concentrations in the estuary differ markedly from groundwater concentrations. Once groundwater traverses the seepage face, biotic and abiotic processes occur in the estuary that influence the concentration of nitrogen available to primary producers. We wanted to assess whether increasing the nitrogen load to estuaries would result in increasing concentrations of nitrogen in the water. Indeed, water samples collected monthly from the surface and the bottom of each estuary from 1991 to 1995 indicate that as nitrogen loads increase, the mean concentration of nitrogen in estuarine water also increases (Foreman et al., submitted) (Figure 5-4).

Estimates using the Estuarine Loading Model (ELM) – The NLM calculated the nitrogen load that enters the estuary at the seepage face. To then address the concentration of nitrogen in the estuary over an annual time scale, an additional estuarine loading model (ELM) was developed. The ELM calculates mean annual concentrations of DIN available to producers in shallow estuaries (Valiela et al., in press). The ELM requires inputs of land-derived nitrogen to the estuary obtained by the NLM. It then accounts for direct atmospheric deposition to the estuary surface as determined for the NLM, N_2 fixation, denitrification and burial of nitrogen in wetland and subtidal sediments, regeneration from sediments, and water residence time to estimate DIN concentrations in the estuary (Figure 5-5).

ELM Validation – Estimates of mean annual DIN concentrations calculated by the ELM were verified against values measured in Waquoit Bay estuaries (Figure 5-6). Like the NLM, the ELM is extremely responsive to measured data ($F=75.6$, $p=0.01$). It should be noted that although there are only five data points in this regression, each represents a subestuary of Waquoit Bay, and each data point, in turn, represents the mean of surface and bottom samples

Table 5-5. Characteristics of subestuaries of the Waquoit Bay watershed[a]

Characteristic	Childs River	Quashnet River	Sage Lot Pond
Area of the watershed (ha)	875	2084	134
Number of houses	1232	740	14
% of land that is agriculture	2	4	0
Measured nitrogen load (kg N/ha/yr)	14	350	601

[a]See Figure 2-1 for watershed locations.

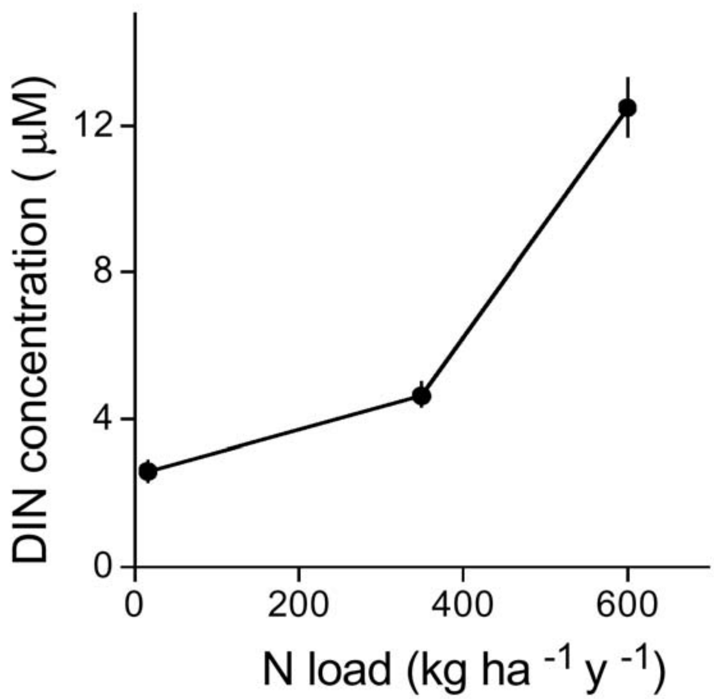


Figure 5-4. Dissolved inorganic nitrogen (DIN) concentrations in the three subestuaries of Waquoit Bay subject to land-derived nitrogen (N) loads.

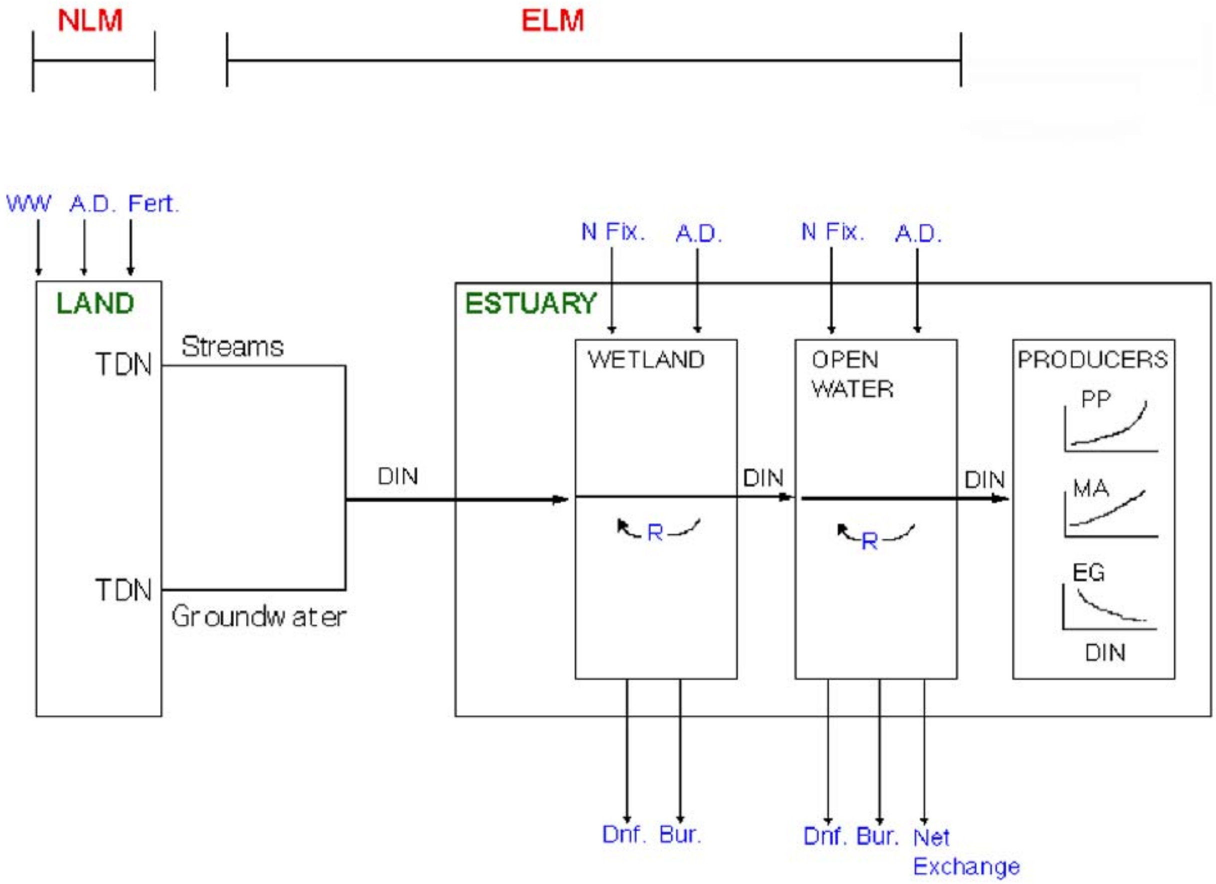


Figure 5-5. Schematic of inputs and exports of the ELM and NLM. The ELM combines the NLM estimated inputs from land, nitrogen fixation, and direct deposition and subtracts losses due to denitrification and burial. It then predicts the amount of dissolved inorganic nitrogen (DIN) that is available in the estuary. The DIN can then be linked to the dominant primary producers if there is sufficient data available.

TDN = total dissolved nitrogen
PP = phytoplankton
MA = macroalgae
EG = eelgrasses

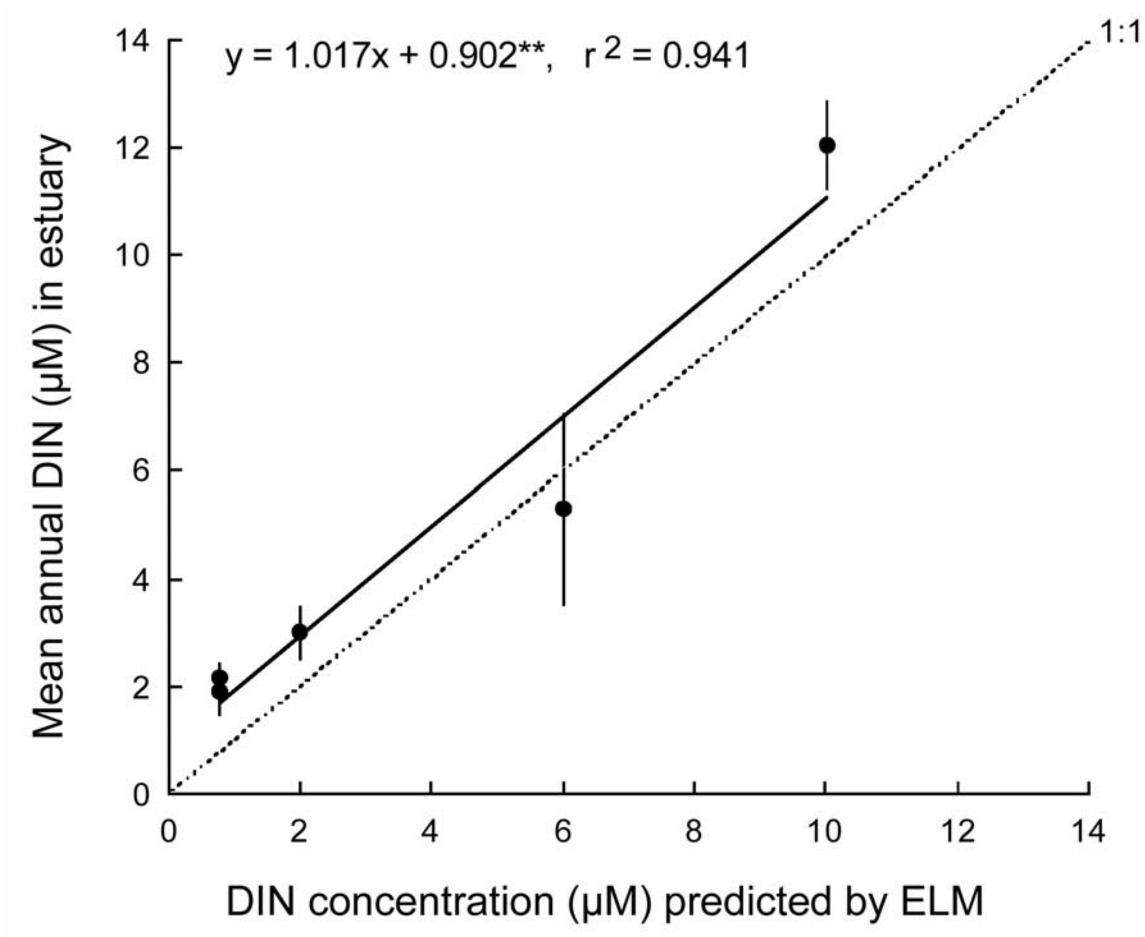


Figure 5-6. Comparison of dissolved inorganic nitrogen (DIN) concentrations predicted by the estuarine loading model (ELM) with measured concentrations in the water column of several Waquoit Bay estuaries (*p*<0.01).

taken monthly from 10 locations within each estuary from 1991 to 1995. The precision of the ELM is demonstrated by the highly significant *r* values (0.997, *p*=0.01). Finally, if the ELM predictions of average annual DIN concentrations were the same as those measured in the estuary, all the data points would lie on the 1:1 line of perfect fit. A linear regression fitted to these data is
statistically indistinguishable from the 1:1 line (Figure 5-6), as demonstrated by a nonsignificant result from a t-test between the slope of the regression and a slope of 1 (t=0.97, not significant). Finally, the predictive ability of the ELM is high, based on Prairie (1996), with an R^2 of 0.94 (Valiela et al., 2002).

The ELM, therefore, furnishes a reasonable fit to actual measurements, even though it contains many terms that are associated with uncertainties—and many untested assumptions regarding various inputs, losses, and pools of nitrogen—and how these measurements interact in estuaries. Uncertainties characterized the steps taken to construct the ELM; at every stage we made simplifications, best guesses, and reasonable assumptions. In fact, all models suffer from similar uncertainties, but one contribution of such models is that they, despite the simplifications and guesses, help predict natural responses. If model predictions parallel observations, there is some reassurance that we have captured a sufficiently complex representation of the workings of the systems under study and that the model may be useful for further applications. Thus, these comparisons furnish some confidence that the ELM is sufficiently competent to be useful, even though many assumptions were involved in its development.

5.2.1.2. ***Phytoplankton Biomass and Production***

Mean annual phytoplankton biomass and mean annual production both increased as the nitrogen load to the estuary increased (Figure 5-7, top). It is concluded that there is a link between land-derived nitrogen and the response of phytoplankton in the receiving estuaries and that increases in urbanization on coastal watersheds result in increased phytoplankton biomass and production. The increase in phytoplankton stock is not surprising in view of the widespread nitrogen limitation of such producers in shallow coastal waters (Boynton et al., 1982; Vitousek and Howarth, 1991; Valiela, 1995; Aguilar et al., 1999; Downing et al., 1999; Tomasky et al., 1999). The range of land-derived loads observed in the Waquoit Bay estuaries spans about two-thirds of the measured loads to various estuaries worldwide (Nixon, 1992). Hence, across most of the range of loads to be expected in shallow estuaries, we would predict increases in phytoplankton biomass and production as nitrogen loads increase were predicted.

5.2.1.3. ***Macroalgae Biomass and Production***

Biomass and production by macroalgae also increased as nitrogen loads from land increased (Figure 5-7, middle). Many references cite widespread nitrogen limitation of macroalgal growth in shallow estuaries and lagoons (Harlin and Thorne-Miller, 1981; Valiela et al., 1992; Peckol et al., 1994; Duarte, 1995; Valiela et al., 1997a). As with phytoplankton, we predicted that increases in nitrogen supply to estuaries would increase the standing crop of macroalgae.

5.2.1.4. ***Eelgrass Biomass and Production***

In contrast to the response of phytoplankton and macroalgae, both biomass and production by eelgrass sharply decreased as total nitrogen load increased beyond 350 kg N/ha/yr (Figure 5-7, bottom). It is important to note that eelgrass may be more sensitive to nitrogen loading than would be predicted from these data. Additional features of eelgrass response to nitrogen loading are considered in Section 5.2.2, where the effects of exposure on the assessment endpoints are characterized.

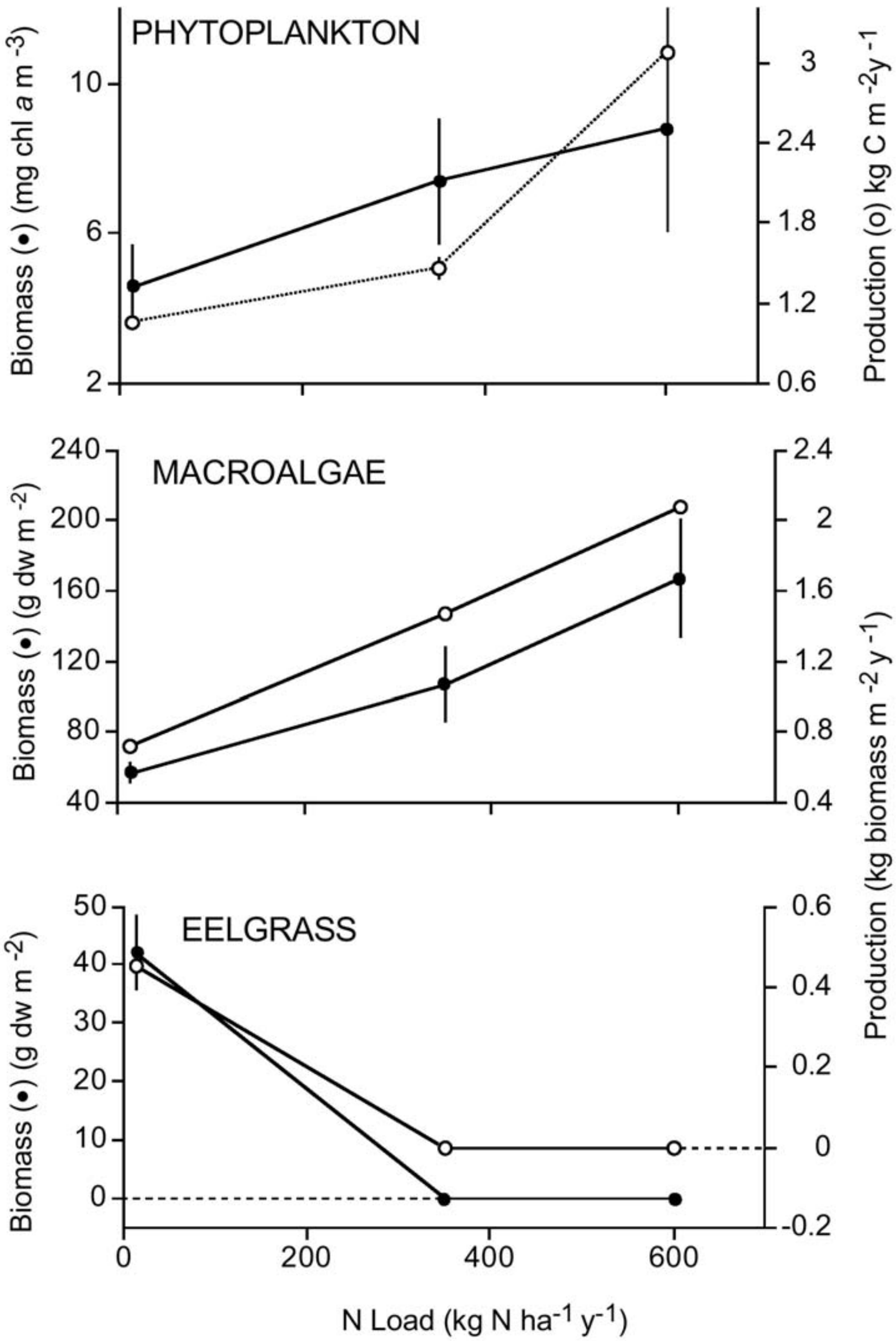


Figure 5-7. Effects of nitrogen (N) loading on biomass and primary production of phytoplankton, macroalgae, and eelgrass in Sage Lot Pond, Quashnet River, and Childs River (top to bottom).

Source: Modified from Valiela et al. (2000b) using N-load data from Valiela et al. (2000a).

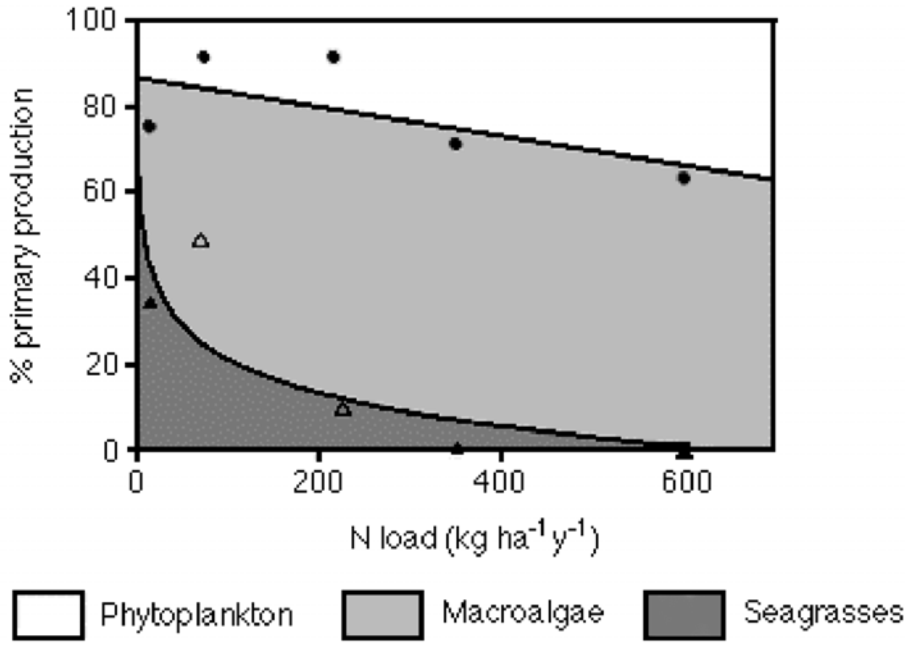


Figure 5-8. Partition of total primary production in shallow estuaries into contributions by phytoplankton, macroalgae, and seagrasses, all plotted against measured annual nitrogen (N) load. Data are from Waquoit Bay, and from Buttermilk Bay and Bass Harbor, two other Cape Cod estuaries.

Source: Adapted from Valiela et al. (2000b).

5.2.1.5. *Combined Effect of Nitrogen Loading on Primary Producers*

These data are presented in synthetic form in Figure 5-8. As nitrogen loads increased, it was predicted that the proportion of primary production carried out by the three selected producers would shift, with eelgrass quickly disappearing and production becoming dominated first by macroalgae, then by phytoplankton.

The impact of land-derived nitrogen loads, however, is mediated by other factors, in particular by water-residence times (Ketchum, 1951; Pace et al., 1992). Residence time is a measure of how quickly water is flushed from the estuary, and it is inversely related to the water turnover rate. In systems with short residence times (T_r), such as Waquoit Bay, the effects of nitrogen loading are reduced in two ways. First, a short T_r dilutes the concentration of available nitrogen when there is a rapid exchange with seawater (e.g., Vineyard Sound). Second, if the T_r is short enough, then phytoplankton that would typically increase in biomass and production in response to the added nitrogen source are unable to significantly multiply before they are swept out of the estuary. Thus, our predictions in Figures 5-7 and 5-8 refer to shallow estuaries and lagoons with residence times of less than 5 days. For waters with a longer T_r, the responses of the phytoplankton to land-derived nitrogen loads would be considerably more intense, with larger biomass and higher production rates.

5.2.1.6. *Zooplankton Egg Production*

Short T_rs, such as those found in the Waquoit subestuaries, may have an even more pronounced effect on organisms that have longer generation times, such as zooplankton. Egg production of the female Acartia tonsa, the dominant copepod, clearly responds to the food abundance provided by the phytoplankton, which increase because of larger nitrogen loads (Figure 5-9, top). Thus, the copepods in each estuary are coupled to the ambient food supply conditions in that estuary. This response did not translate into a parallel response of abundance, as there were no significant differences in the number of copepods among estuaries (Figure 5-9, bottom). It is inferred from these responses that the short T_rs of the Waquoit subestuaries means that Acartia tonsa population abundance is not directly controlled by food supply, although clearly that is what controls egg production.

5.2.1.7. *Shellfish Growth Rates*

The response of shellfish to increases in nitrogen loading is variable and depends largely on the species being considered. Research performed in subestuaries of Waquoit Bay and in other nearby estuaries indicated that the growth rates of softshell clams (*Mya arenaria*) and in quahogs (*Mercenaria mercenaria*) increase as nitrogen concentrations increase (Figure 5-10) (Weiss et al., 2002; Evgenidou and Valiela, in press), although there has been no trend in reported harvest of these species to coincide with the increases in growth rates. Additional research on the ribbed mussel (*Geukensia demissa*) shows similar patterns. Ongoing research in several Cape Cod estuaries indicates that bay scallop populations decline when they are transplanted to estuaries that have high nitrogen-loading rates (Shriver et al., 2002). More features of the dynamics of bay scallops are considered in Section 5.2.2.2.

5.2.1.8. *Finfish Abundance*

Recent research on finfish in Waquoit Bay indicates no clear relationship between nitrogen load and the abundance or growth rates of the two most common estuarine finfish species, Atlantic silverside (*Menidia menidia*) and mummichog (*Fundulus heteroclitus*) (Tober et al., 2000; Griffin and Valiela, 2001). This is probably due, in part, to the lack of a direct relationship between food supply and growth rates for copepods, which are a dominant food source for these estuarine fish. These species are not commercially harvested so that fishing pressure is not responsible for the lack of a relationship.

5.2.1.9. *Summary of Effects on Ecosystem Components*

The previous analysis suggests that in the Waquoit Bay system, the likely responses to nitrogen loading include a high risk of increases in phytoplankton and macroalgae biomass, a

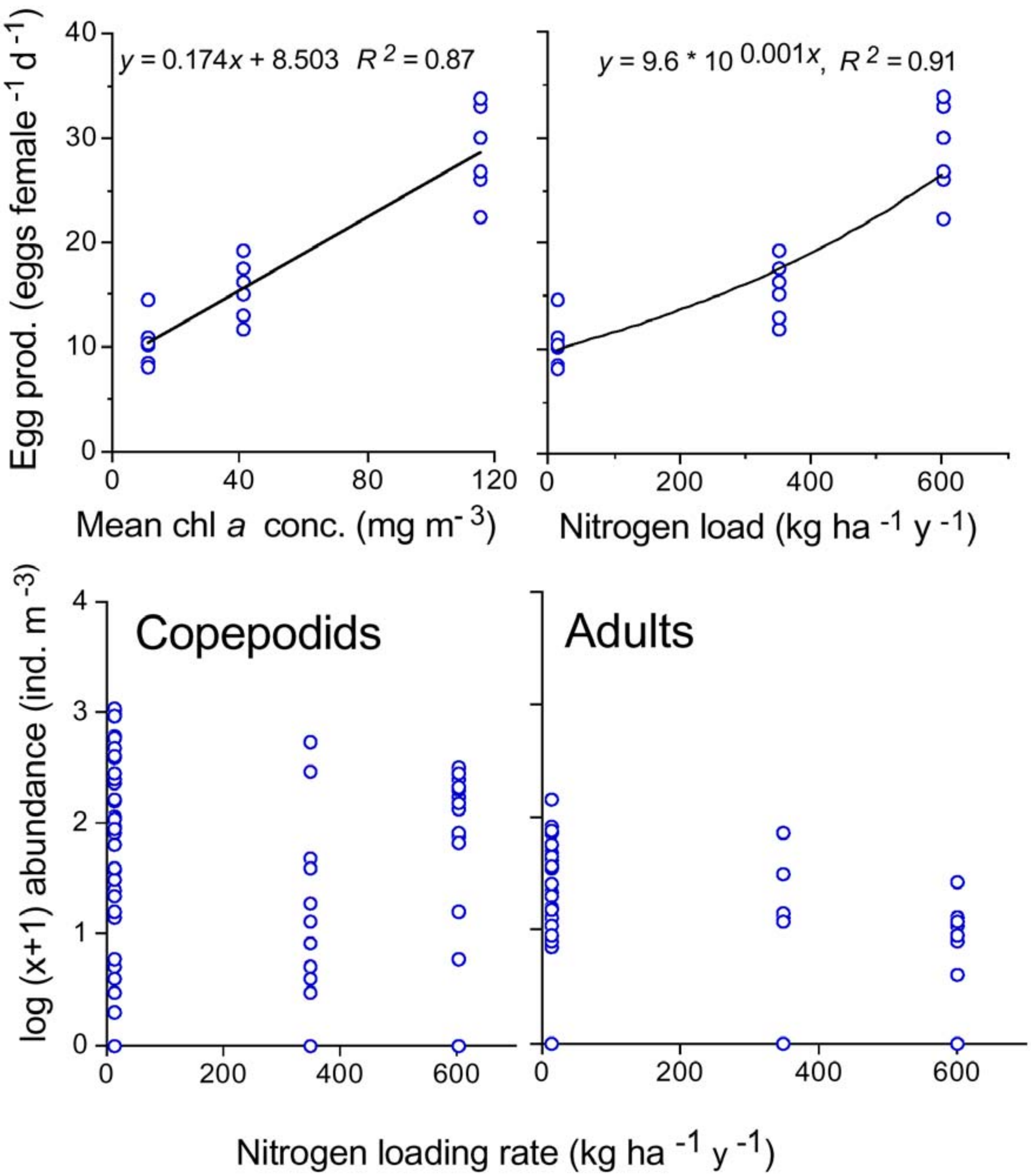


Figure 5-9. Characterization of *Acartia tonsa* in Waquoit Bay estuaries.
Top: egg production in estuaries exposed to different nitrogen rates (right) and chlorophyll concentrations (left) (Cubbage et al., 1999). Bottom: abundance of copepodids (left) and adults (right) relative to land-derived nitrogen load in Waquoit estuaries (Lawrence 2000).

Source: Valiela et al. (2001).

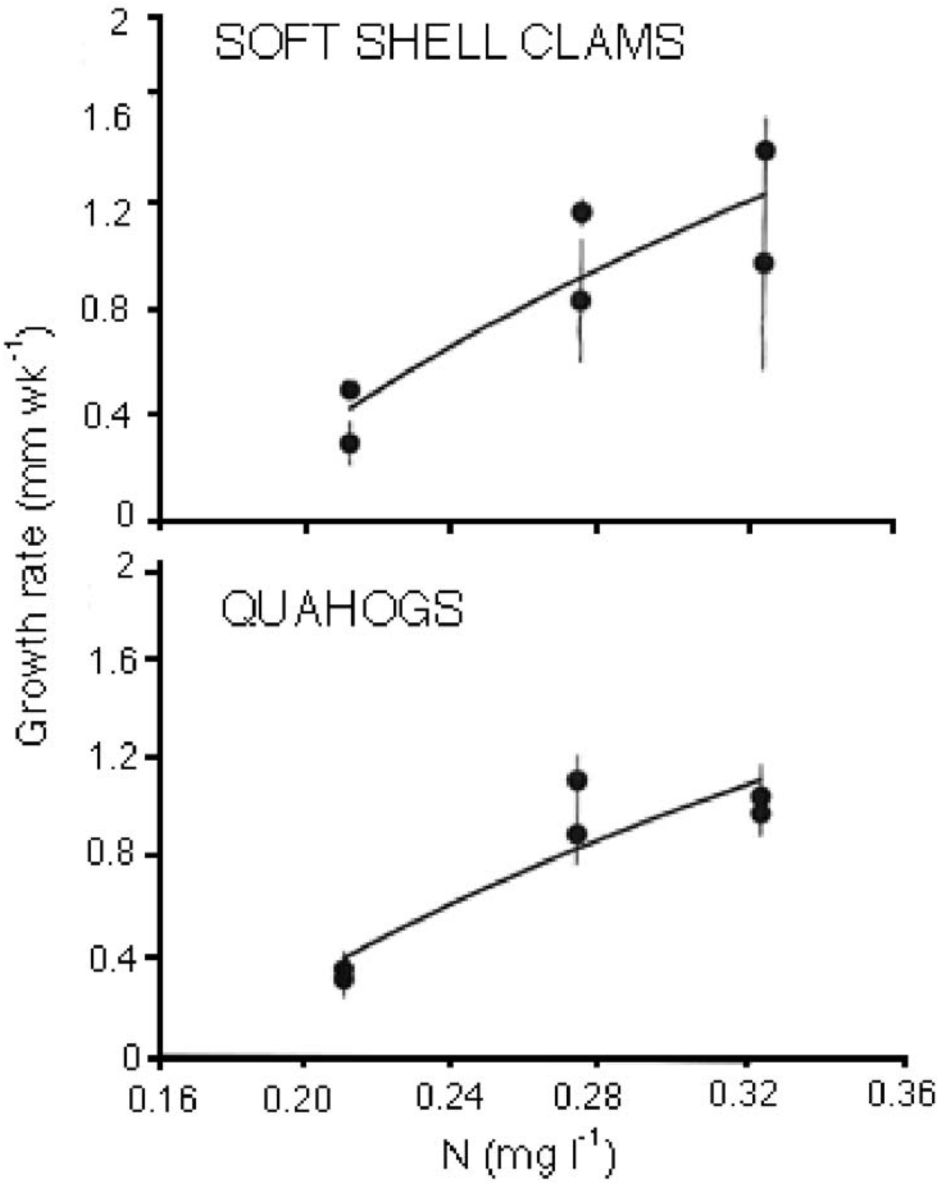


Figure 5-10. The effects of differences in nitrogen (N) concentration on the growth rates of softshell clams (top) and quahogs (bottom).

Source: Data from Weiss et al. (2002).

loss of seagrass habitat, a mixed risk of effects on shellfish, and a smaller risk of effects on fish and zooplankton.

5.2.2. Effects on Assessment Endpoints

The assessment focuses on two measures, percent eelgrass cover and scallop harvest, which are discussed in the sections below.

5.2.2.1. *Percent Eelgrass Cover*

Previously, it was reported that eelgrass biomass and production decrease as nitrogen loads increase, and in the space-for-time substitution presented at the bottom of Figure 5-7, this decrease occurred before nitrogen loads reached 350 kg/ha/yr. To more completely assess the critical load at which eelgrass begins to decline, data from various estuaries were compiled that extend the Waquoit Bay results (Valiela and Cole, 2002). It is evident that for a wide variety of estuaries, eelgrass is highly sensitive to increases in nitrogen loads (Figure 5-11) and that the aerial extent of eelgrass was sharply reduced at loads greater than 20 kg/ha/yr. The meadows disappeared completely by the time nitrogen loads exceeded 100 kg/ha/yr.

5.2.2.2. *Scallop Harvest*

The bay scallop was once one of the most commercially valuable species in the Waquoit Bay estuarine system. Scallops are a benthic invertebrate whose preferred habitat is eelgrass. Records have been maintained by the Town of Falmouth Shellfish Warden on the volume of scallops harvested (the surrogate measure of effect for this assessment endpoint) over the last several decades (Figure 5-12). The quantity of scallops harvested has decreased drastically over the last several years, from almost 200,000 L/yr in 1965 to less than 20,000 L/yr in more recent years (Bowen and Valiela, 2001a).

5.2.2.3. *Summary of Effects on Assessment Endpoints*

As exposure to nitrogen loading increases, there appears to be a loss of eelgrass area, especially as nitrogen loads exceed 20 kg/ha/yr. In addition to the loss of eelgrass area, there also has been a reduction in scallop harvest. Scallop larvae attach to seagrass blades and prefer the slower currents induced by the presence of eelgrass. Additionally, eelgrass beds provide protection from predators (Shumway, 1991). There is evidence that as the area of eelgrass decreases so does the yield of the bay scallop (Bowen and Valiela, 2001a).

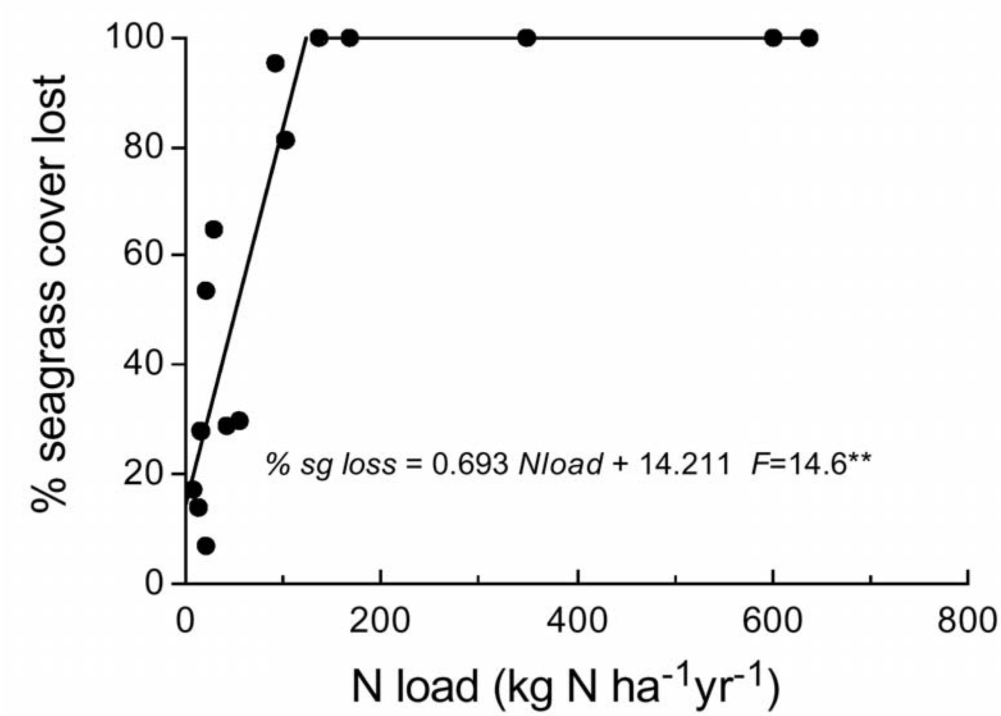


Figure 5-11. Percent seagrass cover lost as nitrogen (N) load increases for a multitude of temperate and tropical ecosystems.

Source: Adapted from Valiela and Cole (2002).

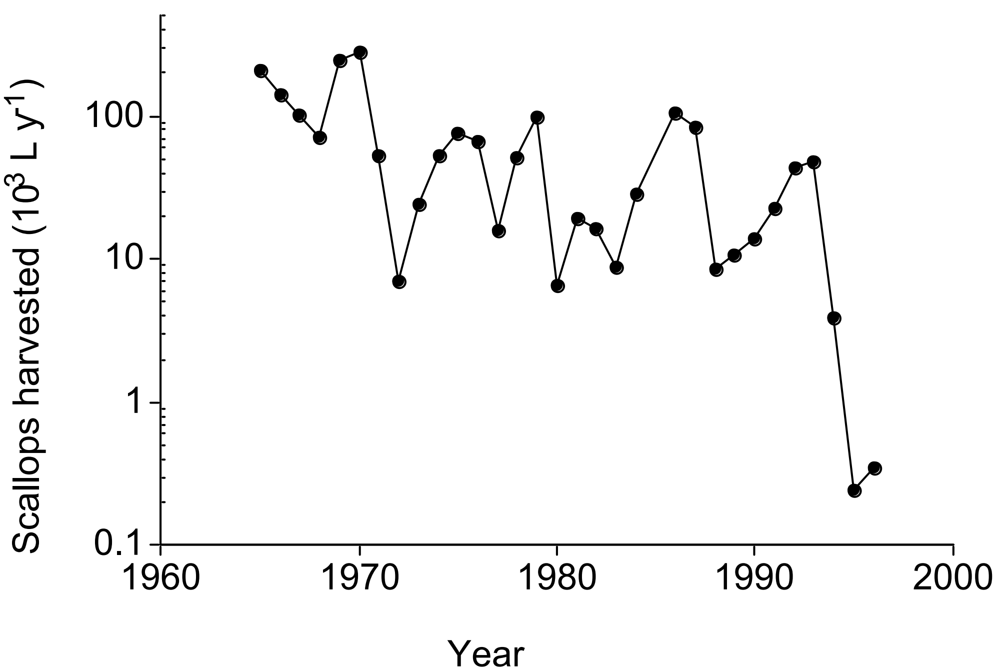


Figure 5-12. Volume of bay scallop harvest in Waquoit Bay from 1965-1995.

Source: Bowen and Valiela (2001a).

6. RISK CHARACTERIZATION

Risk characterization integrates the measures of exposure and the measures of effect to estimate risk to the assessment endpoints. It also serves to summarize and describe the results of the risk analysis in such a way that it can be readily translated to managers and other stakeholders. In the risk characterization for the Waquoit Bay watershed, the NLM and ELM models were used to correlate the temporal changes in exposure to the effects of that exposure on the assessment endpoints. Finally, the models were used to assess a variety of options that can be employed to reduce exposure within the watershed.

6.1. TEMPORAL CHANGES IN EXPOSURE AND EFFECTS

The issue of scale, both spatial and temporal, is critical to understanding and predicting the dynamics of ecosystem perturbation. To understand the effects of eutrophication on coastal watersheds, we must consider how these changes have occurred over time. To this end, we applied the previously described models to explore historical changes in nitrogen loading and the effects of that loading on the two assessment endpoints.

6.1.1. Changes in Exposure: Back-casting Nitrogen Loads

The relative contribution of nitrogen to an estuary from atmospheric deposition, fertilizer use, and wastewater disposal depends on the specific mosaic of land covers present on the watershed surface. There has been a worldwide transition within watershed mosaics from naturally vegetated landscapes to human land uses, both agricultural and urbanized. In many inland areas, there has been a similar transition to agricultural land covers, and hence, fertilizer inputs become a major feature of the loading to watersheds and their receiving waters (Correll and Ford, 1982; Jordan and Weller, 1996; Jordan et al., 1997). In coastal areas, urban sprawl has become an increasingly dominant feature of the land-cover mosaic, and wastewater is becoming a major nitrogen source. Near the end of the 20th century, as much as 37% of the world population resided within 100 km of a shoreline (Cohen et al., 1997), and human populations increased markedly in the near shore of every estuary around the world (Nixon, 1986; Valiela et al., 1992). It necessarily follows that the wastewater released from these increasingly urbanized coastal zones has and will increase. The net results of the geographic land-cover transitions, both toward agricultural or urbanized landscapes, have been that, first, nitrogen loads have increased, and second, the relative contribution from each of the three major sources—atmospheric deposition, fertilizer use, and wastewater disposal—has changed.

As shown in Figure 6-1, the Waquoit Bay watershed has undergone such a land-cover transition over the last 60 years. The area of natural vegetation on the watershed diminished about fourfold from 1938 to 1990. This change was the result of conversion of vegetated land to human uses. In this case, the geographic transition was not just an urbanization of the watershed, but a more complex rearrangement of land-use categories. Turf associated with lawns, parks,

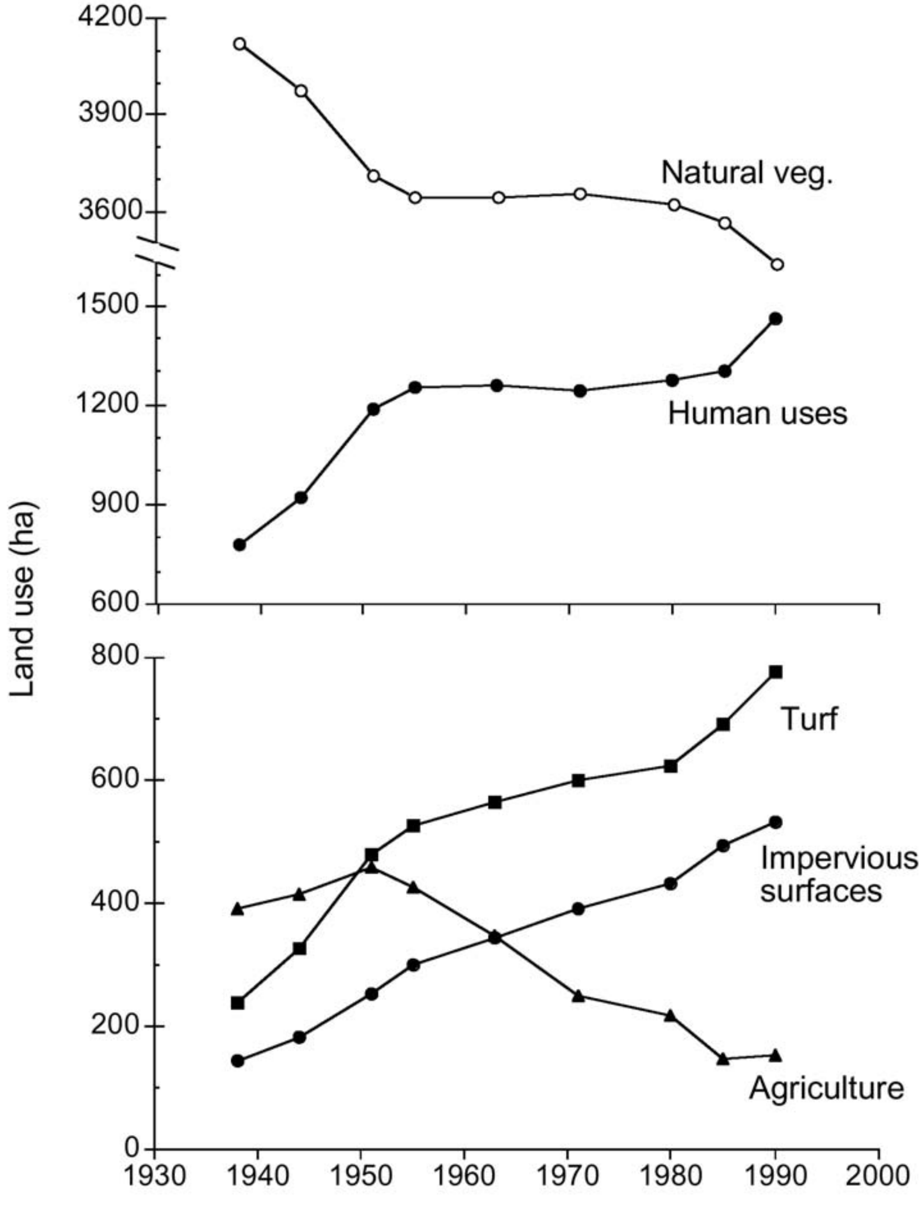


Figure 6-1. Changes in land uses in the Waquoit Bay watershed from 1938 to 1990.
Top: changes in the area of natural vegetation. Bottom: breakdown of human land uses into three major components: turf (including lawns, parks, and golf courses), impervious surfaces (including roads, roofs, driveways, runways, and parking lots), and agriculture (predominantly horticultural crops and cranberries).

Source: Valiela and Bowen, 2002 (Reprinted from Environmental Pollution with permission from Elsevier Science).

and golf courses increased as the number of inhabitants in the area increased. Land devoted to agriculture decreased while land covered by impervious surfaces (roofs, driveways, roads) increased. The NLM and information gathered from aerial photographs was used to estimate the change in exposure that has occurred in the last 60 years. Information on the atmospheric deposition of nitrogen was used to perform the historical analysis needed for the risk characterization. These were based on a reconstruction of measured nitrogen deposition in the region from the turn of the century to the present, as well as a reconstruction of changes in fertilizer use during the same time period (Bowen and Valiela, 2001b). Specifically, atmospheric deposition was reconstructed using compiled historical data that were adjusted to account for regional variability and extrapolated over the time course to include dry and organic depositions of nitrogen. Fertilizer use was reconstructed by estimating the changes in fertilizer application rates for the northeastern United States as a proxy for those on Cape Cod. As a result of the changes in land use, the total nitrogen load *to the watershed* of Waquoit Bay has increased (Figure 6-2, Table 6-1).

Throughout the 50-year period, atmospheric deposition consistently contributed the largest portion of the nitrogen load. The atmospheric supply of nitrogen after 1970 appears to have stabilized, at least in this region. In 1938 nearly all nitrogen delivered to the watershed (95%) was by precipitation, but by 1990, atmospheric sources had dwindled to 59% (Table 6-1). Although the implementation of the Clean Air Act in the early 1970s did not directly address emissions of nitrogen, the indirect effects of increased fuel efficiency in automobiles and reductions in other pollutants emitted from industrial processes may have prevented further increases in the atmospheric nitrogen delivered to the land surface in this area (Figure 6-2, top). Additionally, the reconstruction indicates that there was a decrease in the amount of ammonium in rainwater and an increase in the amount of nitrate, at least for the Cape Cod area. This likely represents a shift from an agricultural base (where ammonium dominates) to an industrial base (where nitrate dominates). Thus, over the whole century, the two forms of nitrogen are in balance and the total increase in nitrogen deposition is not as great as expected if one were looking at nitrogen oxides alone.

Fertilizer- and wastewater-derived nitrogen clearly increased over the 50-year period. Wastewater nitrogen contributions reached 22% of inputs by 1990, and 19% of inputs were contributed by fertilizer use (Figure 6-2, Table 6-1).

Nitrogen loads *to estuaries* differ markedly from nitrogen loads to watersheds, owing to significant interception of nitrogen within the watersheds. Nitrogen losses within watersheds and during passage through the various land-cover types were considerable. Almost 90% of atmospheric nitrogen was intercepted within the watershed, compared with 79% of fertilizer nitrogen and only 65% of wastewater nitrogen (Valiela et al., 1997b). This differential through-put means that, as the nitrogen in groundwater is about to seep into receiving estuarine waters, the relative proportions of nitrogen from the different sources and land-cover types differ markedly from the proportions that entered the watershed.

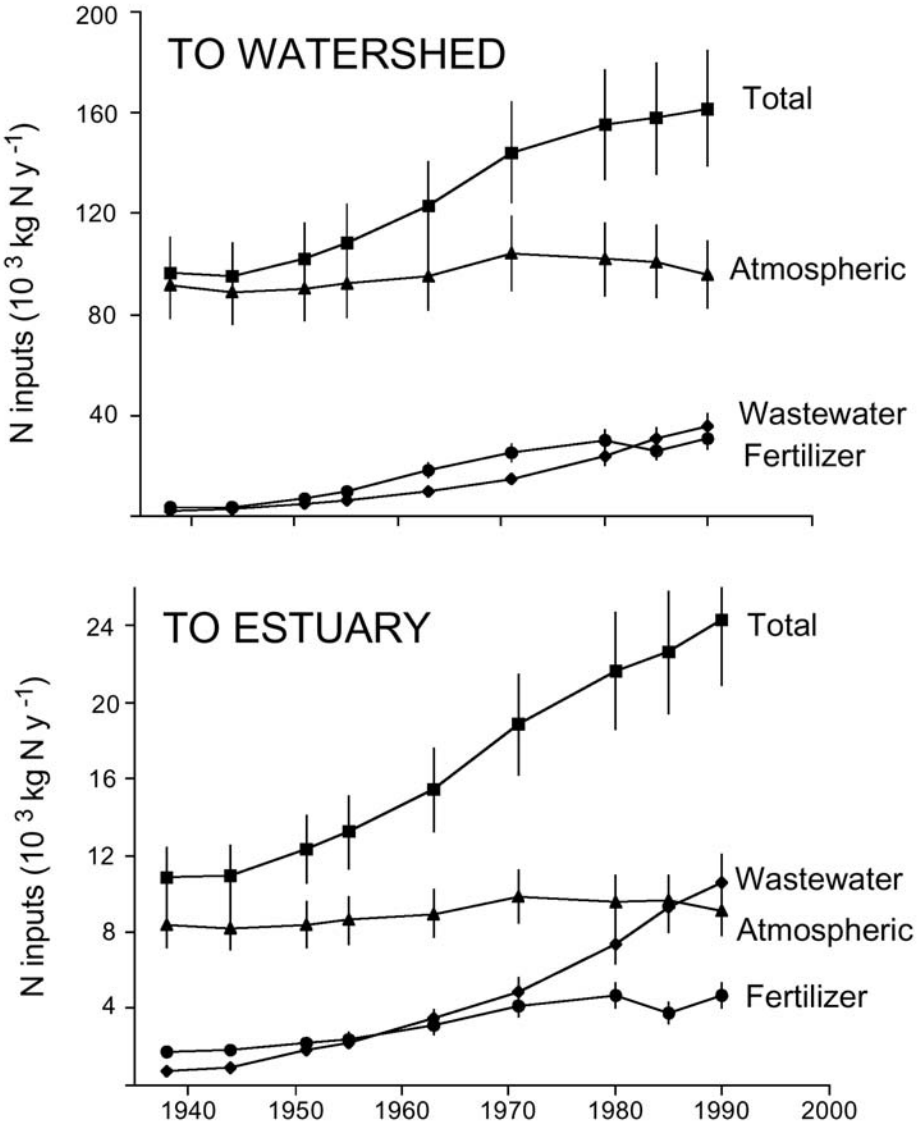


Figure 6-2. Historical changes in nitrogen (N) loading to the watershed and estuary of Waquoit Bay. Modeled loads to the watershed (top) and the estuary (bottom). The loads are broken down into the three major sources of N: atmospheric deposition, wastewater, and fertilizer application. Note the order-of-magnitude difference between N inputs to the watershed (top) and to the estuary (bottom). Data are plotted with the 12% standard error that is associated with the nitrogen loading model.

Source: Bowen and Valiela (2001a).

Table 6-1. Relative contributions of each of the major sources of nitrogen to the Waquoit Bay estuary in 1938 and 1990[a]

Source of nitrogen	1938 Nitrogen load		1990 Nitrogen load	
	10^3 kg y $^{-1}$	%	10^3 kg y $^{-1}$	%
To the watershed				
Atmospheric deposition	91.3	95	95.5	59
Wastewater disposal	2.1	2	35.7	22
Fertilizer use	3.2	3	30.5	19
Total	96.6	100	161.7	100
To the estuary				
Atmospheric deposition	8.4	77	9.1	38
Wastewater disposal	0.7	7	10.5	43
Fertilizer use	1.7	16	4.7	19
Total	10.9	100	24.3	100

[a]The propagated error of the modeled N load is 14% (Valiela et al., 1997b). The percent contribution from each source of nitrogen is slightly different from those published in Valiela et al. 1997b and those used in Figure 2-5. The differences result from the need to use regional trends to incorporate historical changes. In some instances these regional trends were slightly different from the Waquoit Bay specific information used in the original publication. The difference between the regional approach, and the Valiela et al. (1997b) results fall within the standard error of the model.

Source: Bowen and Valiela (2001a).

Wastewater contributions to the receiving estuaries were a minor part of the nitrogen loads early on, but they had become the main source by the 1990s (Figure 6-2, bottom), surpassing both fertilizer and atmospheric deposition as contributors to the load. The total load of nitrogen to the Waquoit Bay estuary increased from slightly more than 10,000 kg N/yr (11.6 kg N/ha/yr) in 1938 to more than 24,000 kg N/yr (28.2 kg/ha/yr) in 1990. This more than doubling of nitrogen loads can be largely attributed to an increase in the wastewater-derived nitrogen, which accounted for only 7% of the total load to the estuary in 1938 but had jumped to 43% by 1990.

6.1.2. Temporal Changes in Effects: Impact on Percent Eelgrass Cover and Scallop Harvest

Temporal changes in eelgrass areas during the last 60 years in Waquoit Bay demonstrate a sharp decline (Figure 6-3, top). Before the 1950s, eelgrass was still recovering from the near-complete loss caused by the wasting disease of the early 1930s (Cottam, 1933; Cotton, 1933; Renn, 1935). Note also that during the 1930–1950 period, nitrogen loads were lower than the 20 kg N/ha/yr that was suggested in Section 5.3.1 as the upper limit of eelgrass survival. The building boom on Cape Cod during the 1960s resulted in an increase in nitrogen loads. By the

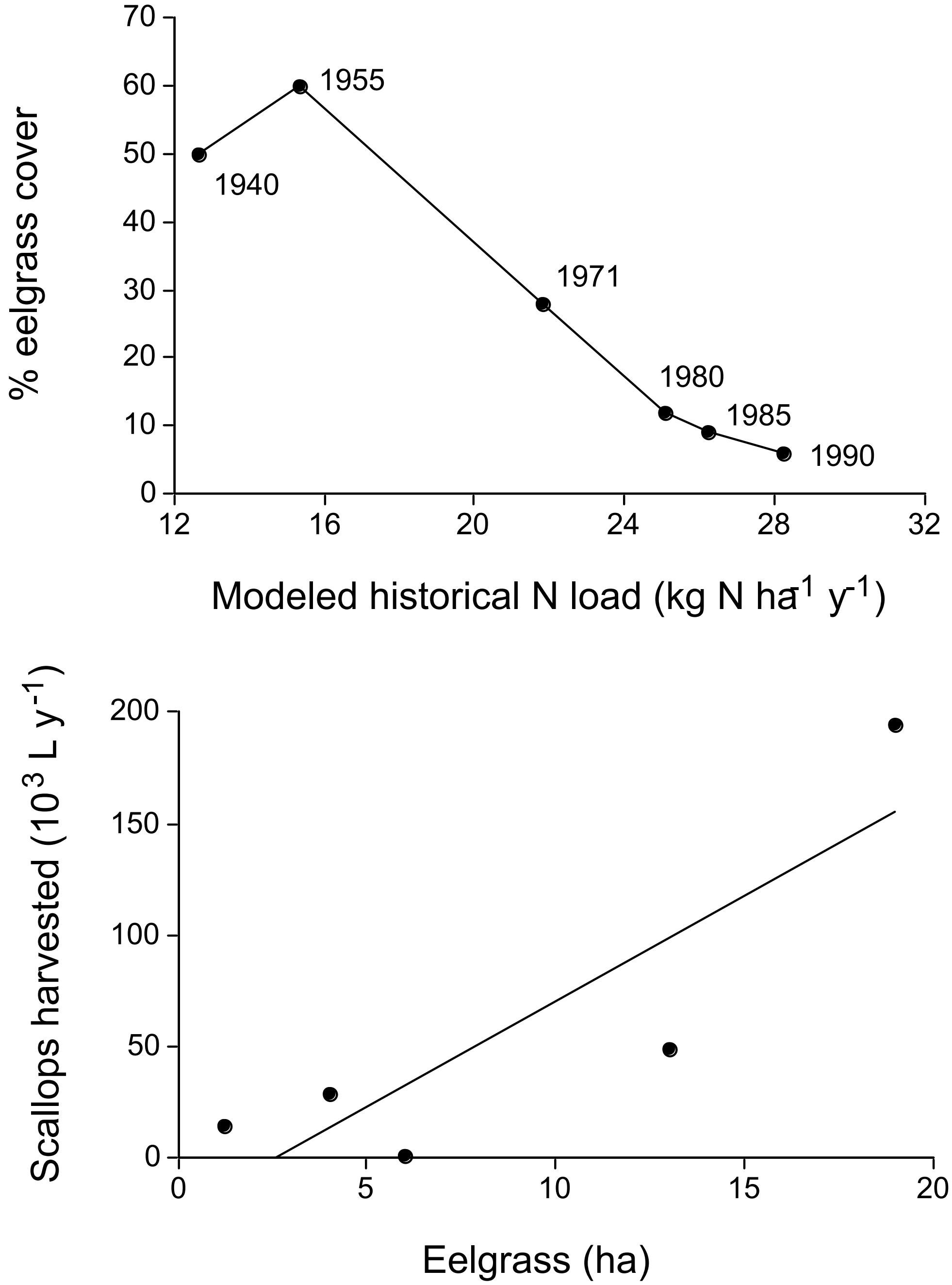


Figure 6-3. Decreases in area of eelgrass and volume of scallops harvested over time as a function of increasing nitrogen (N) loads. Top: percent of area covered with eelgrass in Waquoit Bay estimated by aerial photographs between 1940 and 1990 (Costa 1988, Short and Burdick 1996) plotted against modeled historical N loads. Bottom: the number of reported scallops harvested in Waquoit Bay as a function of eelgrass area. Scallop data are from the Shellfish Warden's annual reports, in the Town of Falmouth Annual Report.

Source: Adapted from Bowen and Valiela (2001a).

early 1970s, the nitrogen load exceeded 20 kg N/ha/yr, and eelgrass meadows were notably smaller in area. The loss of eelgrass habitat continued through 1990. The historical reconstruction indicates that the nitrogen loads corresponding to the near-complete destruction of eelgrass meadows ranged only between 15 and 30 kg N/ha/year (Figure 6-3, top).

The extrapolation was carried one step further to look at the secondary effects of eelgrass decline on the decrease in scallop harvest. Because the presence of seagrass is required for the maintenance of many taxa, including commercial shellfish and finfish species, a change in eelgrass cover implies drastic changes in the rest of the estuarine food webs in affected estuaries. During the time span when eelgrass meadow area decreased, the annual harvest of bay scallops in Waquoit Bay decreased (Figure 6-3). We can therefore claim that urban development can be demonstrably linked to drastic restructuring of estuarine ecosystems.

6.1.3. Effects of Other Stressors on Eelgrass

Other stressors are potentially damaging both to existing eelgrass beds and to efforts at reintroducing eelgrass to estuaries, although these stressors are minor in comparison to nitrogen loading because they are restricted to very small regions of the bay or they occur only sporadically. For example, remaining beds of eelgrass may be further impacted by natural events. In 1991, Hurricane Bob, a category 3 hurricane, made landfall on Cape Cod. The storm surge washed over a spit on Washburn Island in Waquoit Bay, resulting in the burial of an eelgrass bed on the inside of Eel Pond (Valiela et al., 1996, 1998). Although this was an extremely severe effect on that eelgrass meadow, the impact was limited spatially, and the meadow recovered fully during the next growing season.

The eelgrass population of Waquoit Bay was nearly destroyed during the early 1930s as a result of an infection of eelgrass wasting disease. The epidemic struck many northeastern U.S. estuaries earlier last century and is thought to have been caused by the slime mold *Labyrinthula* (Short et al., 1988). Evidence of a pathogenic strain of *Labyrinthula* has been seen recently in scattered locations. Continued spread of this fungus could destroy existing eelgrass meadows, but evidence of the disease does not presently extend beyond a few locations in the Northeast (Short et al., 1986), and it does not exist presently in Waquoit Bay.

Human activities on estuaries can impact eelgrass meadows through physical disturbances. Docks and marinas built over eelgrass beds limit light through shading, resulting in the eventual destruction of the eelgrass meadow. Dredging activities result in a direct loss of eelgrass through removal of shoots. Dredging also has an indirect effect because of the resuspension of sediment particles, which increases water turbidity and decreases the amount of available light for eelgrass growth. Propeller scour from passing boats and mooring scars are also localized stressors. The impact of shellfishing on eelgrass meadows has not been examined extensively, although it could have a significant, but local, impact (verbal communication from J. Hauxwell, Wisconsin Department of Natural Resources, to I. Valiela, Boston University Marine Program).

It is important to note here that ecosystems are highly complex and variable systems that can and do change in species composition, distribution, and abundance. It is an established fact that nitrogen loading is the major stressor on eelgrass and that decreasing the load of nitrogen to Waquoit Bay may result in water quality conditions that could support eelgrass. This does not, however, guarantee that eelgrass will reestablish itself or maintain itself if replanted. Many efforts have been made recently to restore eelgrass beds in estuaries where meadows no longer exist.

Research indicates that one of the major impediments to reconstruction of eelgrass habitat is the lack of genetic diversity in the colonized plants. Williams and Davis (1996) used several measures of genetic diversity (percentage of polymorphic loci, allele richness, expected and observed heterozygosities, and proportion of genetically unique individuals). They found that overall genetic diversity was significantly reduced in transplanted eelgrass beds. In addition, Smith et al. (1989) compared the habitat value of natural meadows and recently transplanted eelgrass meadows for the commercially important bay scallop. Stocking adult scallops in recently transplanted meadows was not successful, indicating a lag time may be needed between transplantation and functional habitat use. Thus, the management options outlined in the next chapter do not guarantee the return of eelgrass with the lowering of nitrogen loads; they simply reduce loads to levels where eelgrass might be able to survive.

7. MANAGEMENT IMPLICATIONS

At the outset of the risk assessment process, the workgroup proposed a management goal that aimed to reestablish and maintain water quality and habitat conditions in Waquoit Bay, to support diverse, self-sustaining commercial and recreational fisheries and native fish and shellfish populations, and to reverse ongoing degradation of ecological resources of the watershed. To achieve this goal, the workgroup initially evaluated the various sources of stress and the effects of that stress on the ecology of Waquoit Bay. The results of this analysis indicate that nitrogen loading is the biggest threat to the ecology of Waquoit Bay.

The results of the risk assessment indicate that the bulk of excess nitrogen enrichment stems from nonpoint pollution sources, namely, land-use activities within the watershed. Because 35% of the U.S. population lives within 100 km of the coast, this problem has ramifications nationwide. It poses a challenge for local and state managers who are responsible for land-use decisions in the coastal zone and for federal and state agencies that are responsible for natural resources (NOAA's NMFS and the U.S. Fish and Wildlife Service) or for pollution control (U.S. EPA, NOAA, and Massachusetts Coastal Zone Management).

The Waquoit Bay watershed ecological risk assessment provides a tool for helping local managers recognize the consequences of decisions about land use within the watershed on the water quality of coastal embayments. This document should aid managers in resource planning and provide useful information supporting zoning decisions. The models developed here are applicable to watersheds that are dominated by land use in rural and suburban areas. These models can analyze watersheds that lay above sandy soil in which groundwater flow dominates the hydrology and in which the receiving water has a relatively short residence time (less than a week).

The tools provided by this risk assessment allow local managers to anticipate the impact of future land-use changes within the watershed and explore various treatment options to minimize future impact as well as reduce present levels of nitrogen loading to Waquoit Bay. Through a series of possible scenarios, managers can use the combination of models to generate solutions that will restore water quality to levels that existed in the 1960s.

7.1. ASSESSMENT OF RESTORATION MEASURES

The historical reconstruction of land-derived nitrogen loads, plus the linking of these loads to assessment endpoints, such as percent eelgrass cover and bay scallop harvest, provides some means for identifying management priorities and defining potential restoration measures. If management goals include reduction of nitrogen loads, even a cursory look at the results of this work suggest some general approaches that could be used to remediate the increased nitrogen loads. A first step in evaluating the potential effects of remediation options is to examine the relative nitrogen loads from wastewater disposal, fertilizer use, and atmospheric deposition (Table 6-1). Model estimates using 1990 data indicate that wastewater contributed 43% of the total nitrogen in Waquoit Bay. Wastewater inputs would thus seem to have a higher priority than fertilizer use; at most, management of the fertilizer inputs could be expected to lower nitrogen loads by 19%.

There are many restoration measures that could reduce the loads of nitrogen from wastewater and fertilizer use. Management of residential wastewater and fertilizer use is the most practical measure available locally. Zoning restrictions, improvement of *in situ* septic systems, and installation of small sewage treatment plants could all lower the inputs of wastewater nitrogen. Restrictions on the use of fertilizer and turf area would also, to a smaller degree, lower nitrogen inputs to Waquoit Bay, but this management option would be especially effective in more agriculturally intensive areas. The choice of the appropriate management option depends on the target endpoint a community wants to achieve.

Although direct management of atmospheric deposition is impossible on a local scale, preservation of green, open space can make a contribution to management of atmospheric nitrogen loads. Atmospheric nitrogen deposition is still the largest nitrogen source to the Waquoit Bay watershed. Interception of atmospheric nitrogen is highest where natural vegetation covers land parcels, and this measure should be a part of the managers' tool kit when thinking about the coastal zone. Preservation of green space, especially vegetated buffers along streams, may slow the increase in nitrogen enrichment to receiving waters by intercepting atmospheric deposition and by limiting the supply of nitrogen that would result from conversion to other land uses, such as agricultural or residential. Other means of remediating nitrogen inputs from atmospheric deposition require legislative actions that affect a multistate area, because the airshed that contributes nitrate pollution to a region is significantly larger than the area that is under local management control. However, if actions that could mitigate sources within the watershed prove insufficient, citizens in the Waquoit watershed will have no choice but to consider legislative or cooperative actions to mitigate sources outside the watershed.

In the initial part of the risk characterization, linkages between nitrogen loading and the defined assessment endpoints were developed. Here we develop an approach to using these relationships to help managers make informed decisions. Through the historical reconstruction we have established that nitrogen loads and concentrations have increased over time. Managers and stakeholders can examine the historical record and select, using Figure 7-1 as an example, the percent eelgrass bed coverage they want to adopt as a management target. Once the target status is selected, we can use Figure 7-2 to find the year that the nitrogen load corresponded to the selected eelgrass target. Then we can investigate what management practices could be deployed to lower nitrogen loads to the target levels. For example, suppose the restoration target is to return to conditions that could support the growth of eelgrass on 30% of the main estuary (Figure 7-1). The corresponding load would be about 18,000 kg N/yr. To achieve the restoration target, the nitrogen loads from the watershed to the estuary need to resemble those existing around 1970 (Figure 7-2) . The management goal in that scenario would be to reduce the nitrogen loads by approximately 6,500 kg N/yr. The task then becomes to explore, using simulations with the NLM/ELM, the management practices that might be put in place to achieve those restoration measures.

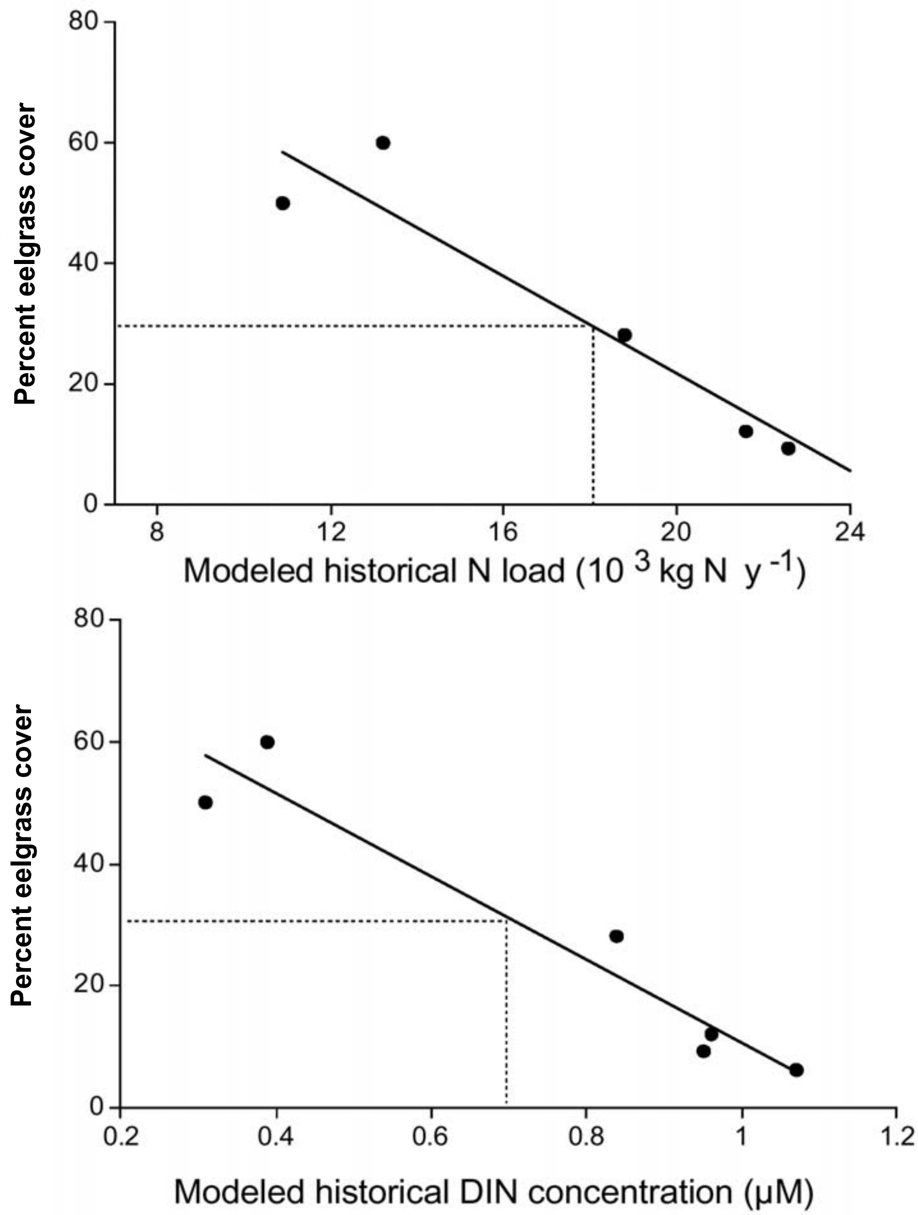


Figure 7-1. Relationship between nitrogen (N) loads, dissolved inorganic nitrogen (DIN) concentration, and percent eelgrass cover. Top: nitrogen loading model estimates of N loads from 1938 to the present plotted against percent eelgrass cover in Waquoit Bay in the corresponding years. To restore eelgrass in Waquoit Bay to 30% of the original area it is necessary to reduce the N Load to 18,000 kg N/yr. Bottom: estuarine loading model-predicted N concentrations in Waquoit Bay since 1938 plotted against eelgrass area. To restore eelgrass in Waquoit Bay to 30% of the original area would require attaining a dissolved inorganic concentration of about 0.75 µM.

Source: Eelgrass data from Costa (1988).

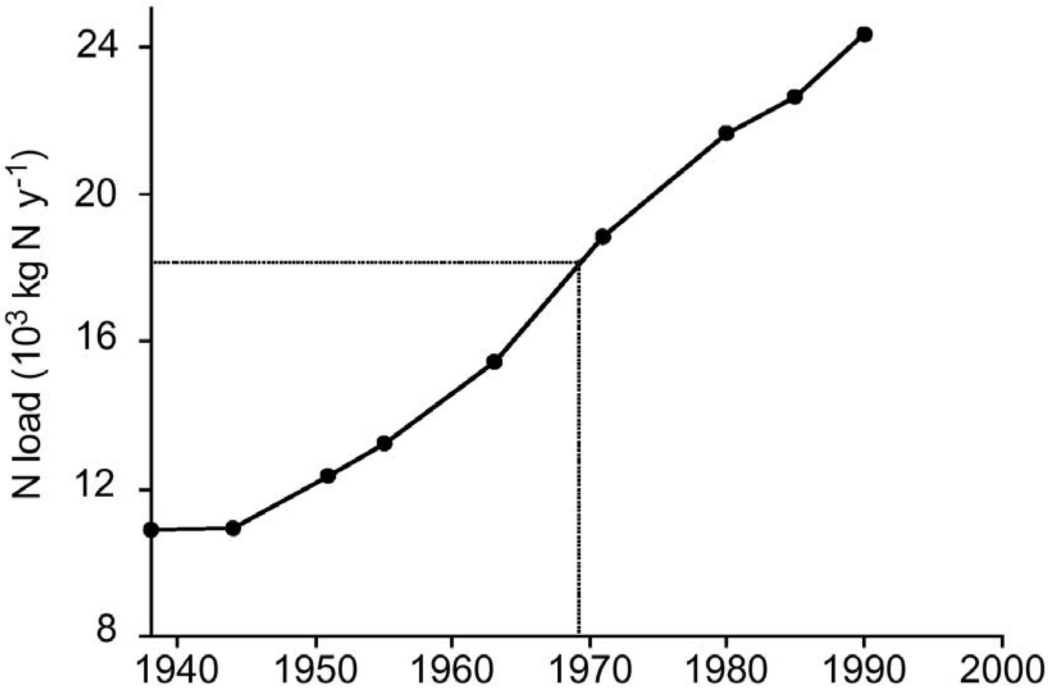


Figure 7-2. Historical changes in nitrogen (N) loading predicted by the nitrogen loading model. If the management target is a return to 30% eelgrass cover, then managers must reduce N loads to 18,000 kg N/yr, loads that are comparable to those of around 1970.

7.1.1. Reducing Fertilizer Application Rates

Reducing the application rates of fertilizer, or the amount of land that is fertilized, will reduce the nitrogen loading to Waquoit Bay, although at most the reduction will be only 17% of the total nitrogen load (Figure 7-3). Notice that fertilizers from lawns and golf courses combined contribute almost the same amount of nitrogen as that derived from agriculture, and that even the complete abatement of all fertilizer use in the Waquoit Bay watershed would reduce the nitrogen load only to the level of the early 1970s. The benefits of this option must, of course, be weighed against the economic costs of decreased agricultural output and the potential loss of tourism resulting from less attractive greens on local golf courses.

7.1.2. Managing Wastewater

About 25% of the population of the United States disposes of its wastewater via *in situ* septic systems of conventional design. In many coastal areas, the use of septic systems is widespread. For example, in Cape Cod only 2% of the land surface is serviced by wastewater plants (Mitchell, 1999); septic systems are used in the remaining areas (Heufelder and Rask, 1996, 1997). Wastewater disposal through septic systems has in recent decades become the major source of nitrogen entering the estuaries of Waquoit Bay (Table 6-1). The most likely

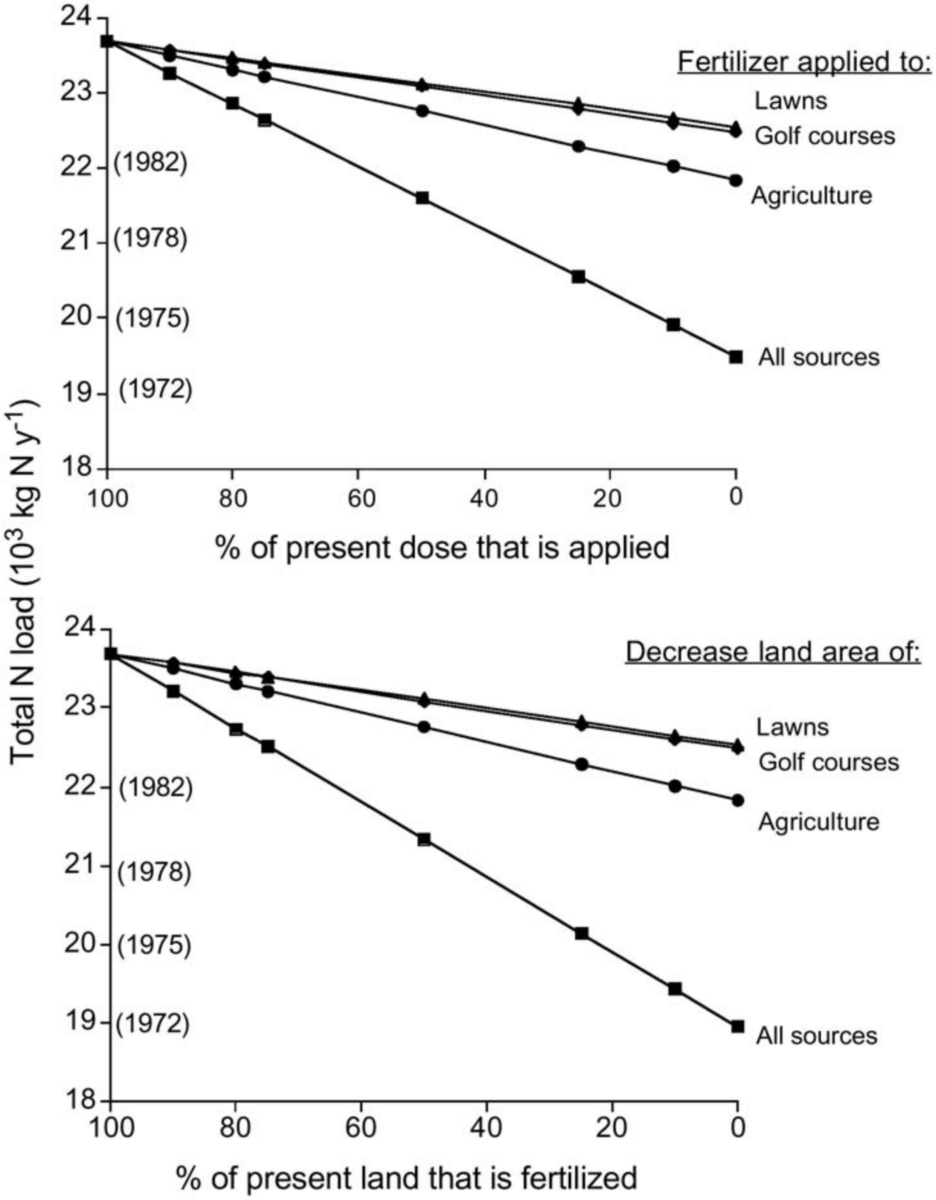


Figure 7-3. Reduction in the total nitrogen (N) load that would result from varying the amount of fertilizers used in the Waquoit Bay watershed. The reduction in total nitrogen load in the Waquoit Bay watershed that would occur if varying percentages of fertilizer were withheld (top) or the area of fertilized land were reduced (bottom).

Source: Data from Bowen and Valiela (to be submitted).

option for managing wastewater nitrogen is altering the nitrogen retention in septic systems.

Potential changes in nitrogen loads that would result by assuming different degrees of nitrogen retention in septic systems were assessed. The NLM was used to run simulations on various nitrogen retention values ranging from the 39% nitrogen removal achieved by septic systems of conventional design presently being allowed by the Commonwealth of Massachusetts up to complete retention (Figure 7-4).

The first set of simulations took into consideration all of the buildings on the watershed. The next set of simulations considered only those buildings up to 1,000 m, 500 m, and 200 m from shore. The purpose of these simulations was to ascertain whether buildings closer to the shore disproportionately contributed to the nitrogen load. Replacing or retrofitting septic systems in the entire watershed might be an expensive and politically daunting alternative. Management of wastewater nitrogen loads from smaller areas closer to shore might be a more feasible alternative.

The simulations suggest that if retention of wastewater nitrogen in all septic systems were to increase above values characteristic of current systems (39%) (Valiela et al., 1997b), the nitrogen loads to Waquoit Bay would decrease from values extant in 1990 to those of 1958, roughly by 42% (Figure 7-4). If only those septic systems located within 200 m of shore (roughly 2 years' travel time within this aquifer) were considered, the reduction in nitrogen load would amount to values seen in 1975, a reduction of about 17%. The simulations concerning other distances yielded intermediate values.

The results of the simulations, shown in Figure 7-4, suggest that areas of watersheds closer to shore contribute proportionately more to estuarine nitrogen loads than areas farther from shore. This result may derive from two sources. First, buildings in this watershed (and elsewhere) are more numerous closer to water (Valiela et al., 1992). Second, groundwater plumes bearing nitrogen from septic systems may reach the shore only in those areas within 200 m of shore; farther away, the plumes disperse into bulk groundwater and may lose more nitrogen during travel. Regardless of the mechanism, this result points to strategies that target near-shore septic systems as being potentially more effective than efforts to manage all of the septic systems in a watershed.

So far, we have discussed the benefits of complete retention of wastewater nitrogen within *in situ* systems. In reality, it is hard to find a practical way for *in situ* systems to be so effective. Some alternative designs have lower nitrogen retention percentages (Table 7-1), and the corresponding nitrogen loads are lower (Figure 7-4). Given the measures of uncertainty associated with the various alternative septic-system designs (note the ranges given in Table 7-1), it is not altogether evident that there are significant differences in nitrogen retention among the alternative designs.

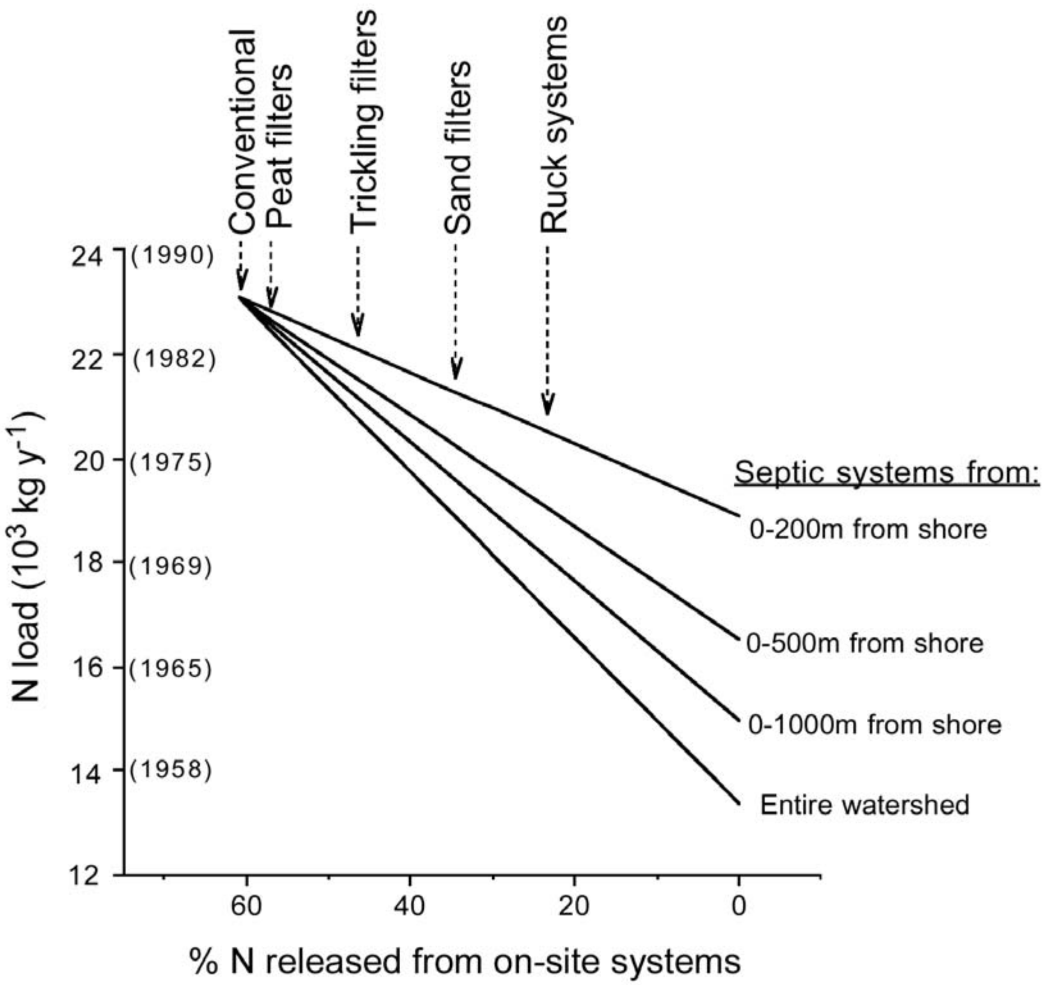


Figure 7-4. Reduction in total nitrogen (N) load in the Waquoit Bay watershed (and corresponding year) that would result from implementing various wastewater treatment systems.

Source: Bowen and Valiela (to be submitted).

These scenarios are by no means the only options. They merely suggest that management of certain design features may achieve selected targets and that the models we developed are flexible and useful tools for addressing the issues raised by managers. Further, our approach aims to ensure that stakeholders make the selections for scenarios and options and that we use the models and science to provide evaluations for stakeholders to consider. Clearly, stakeholders must decide, and they will surely need to include criteria other than the loadings produced. Our

Table 7-1. Onsite septic system retention efficiencies reported for various alternative systems

Septic System	Retention Efficiency		Source	Reference
	Mean	Range		
Conventional Massachusetts Title 5 septic system	39	10–90	Various published estimates	Valiela et al. (1997b)
Peat filters	43	30–65	Six *in situ* systems on East Coast	Heufelder and Rask (1996)
Trickling filters	54	22–86	Various systems	Stokes (2000)
Recirculating Sand Filters	64	59–70	Multiple samples from four Maryland systems	Piluk and Peters (1995)
Ruck®	88	66–99	Mean from six *in situ* systems in Massachusetts	Rask and Heufelder (1998)

Source: Bowen and Valiela (2001a).

view is that results from the assessment of restoration should enhance optimal decisions, which must be made by stakeholders.

Stakeholders frequently suggest that one option for improving water quality in coastal embayments is to shorten the water residence time through either dredging or altering the inlet morphology and/or depth. This approach is based on the inverse relationship between the nitrogen loading rate and water residence time in determining the nitrogen concentrations within the water column. Hydrodynamic model simulations by T. Asaji at Applied Science Associates, Inc. indicate that increasing the cross-sectional area of both the Eel Pond and the Waquoit Bay inlets by 100% would reduce the residence time of the system from 45.42 hours to 40.46 hours (Table 7-2). This would lower the residence time by only 11%, with a 100% increase in the area of both inlets. If the inverse relationship between residence time and nitrogen loading is accurate, then the nitrogen loads would be decreased by only 11%—within the margin of error of the model. Dredging is extremely expensive, causes adverse environmental impacts, and requires much permitting. In addition, to keep inlets open, dredging needs to be repeated at regular intervals, requiring more permits and more money. Dredging, therefore, does not seem like a desirable approach to resolving water-quality issues in Waquoit Bay.

Table 7-2. Changes in water residence time predicted by dredging simulations

Dredging scenario	Tidal volume increase (%)	Residence time (hours)
Present condition	0.00	45.42
Increase Eel Pond by 50%	0.28	45.29
Increase Eel Pond by 100%	0.36	45.26
Increase Waquoit Bay by 10%	4.70	43.29
Increase Waquoit Bay by 25%	7.66	41.94
Increase Waquoit Bay by 50%	9.91	40.92
Increase Waquoit Bay by 100%	10.93	40.46
Increase both by 50%	9.97	40.89
Increase both by 100%	10.91	40.46

Source: Bowen and Valiela (to be submitted).

7.2. USING THE ECOLOGICAL RISK ASSESSMENT TO CONVERT SCIENCE TO MANAGEMENT ADVICE: CAVEATS AND LESSONS LEARNED

When public meetings were held, the workgroup noted that the terminology used in the risk assessment created confusion, because neither the public nor the attending managers or scientists were familiar with concepts such as stressors, ecological effects, assessment endpoints, measures of effect, or exposure pathways. Even with the use of the conceptual model at the February 1995 meeting with managers, it was difficult to communicate these concepts. Difficulties with terminology were exacerbated by issues of scale, since most of the legislative mandates, regulations, and jurisdictional boundaries for local/state/federal managers are not set at the watershed level.

The concerns of the public are more localized, and some people had a hard time with the concept of evaluating stressors and their impacts on valued resources at the watershed level. For example, the focus of stakeholders who are involved in the Superfund cleanup at MMR are human health risks (toxic contamination of their sole-source aquifer for drinking water and the risks of swimming and eating fish from local ponds), whereas constituents near Waquoit Bay are concerned about ecological risks (anoxia/fish kills, loss of eelgrass and bay scallops, changes in the size and abundance of recreational finfish species, diminished aesthetics, etc.).

Given the limited public outreach program permitted by the resource limitations of this assessment, getting a consensus on the conceptual model and the resulting risk analysis from constituents who have diverse localized concerns, scientists who study separate components of the system, and local/state/federal managers who have differing mandates and jurisdictional boundaries was difficult.

The risk analysis report is lengthy and complex, so some type of translation endeavor will have to be undertaken to make the project's findings accessible to local citizens and local/state/ federal managers. Uncertainty is a key component of the risk assessment process, and given the technical nature of the analysis, it is a challenge to convert these data into information that will be useful to managers. Because the costs that local citizens will bear to diminish nitrogen input from septic systems and fertilizer use are likely to be significant, it will be critical in the risk management phase to clearly articulate the benefits and uncertainties associated with the NLM/ELM models and to provide socioeconomic information on how the costs will be distributed among different stakeholders. A sociopolitical process will need to be established to engage citizens and their elected representatives in resolving these cost/benefit issues.

The ecological risk assessment is an information-intensive process for evaluating risks to environmental receptors. The unevenness of the information available for the Waquoit Bay watershed limits the ability to conduct a comparative analysis of the risks from multiple man-made stressors. In our case, we had a lot of information on nutrient enrichment and its chemical and biological effects in ponds and the estuary, but we lacked good information on, for example, the reproductive status of brook trout and the quality of their breeding habitat, the threats to wetland bird breeding habitat, and the impacts of volatile organic contaminants on fish health in Johns Pond.

In addition, some of the stressors that impact the ecological resources in the watershed have a regional source rather than a source within the watershed. These would include regional commercial/recreational fishing pressure and recruitment, which influence the estuarine finfish assessment endpoint, and atmospheric mercury and nitrogen input from the regional airshed, which contributes to the eelgrass and tissue contaminant assessment endpoints. To deal with these regional stressors, an entirely different level of management would be needed to augment the risk management activities of local/state/federal managers. The public desires a seamless process to address these issues, but in practice this is rarely possible, and local actions may result in limited benefits for the costs incurred.

Before the Waquoit Bay ecological risk assessment was begun, nitrogen loading from the watershed led to water quality problems and diminished populations of valued biotic resources. By developing the coupled NLM/ELM models, this study provided a methodology for local/state/ federal managers to better understand nitrogen impacts so they can make wiser decisions in addressing the problem and engaging local citizens in the dialogue on mitigation strategies (including validating model predictions, long-term monitoring, and adaptive management). The model will be available on the WBNERR web site (Geist, 1996). It will be important for managers to communicate to the public that these model predictions for nitrogen loading will lead to improved water quality, but they may not lead to recovery of the eelgrass beds or bay scallop populations, as discussed in Section 6.1.2 and 6.1.3.

Because the temporal scale for local management actions is on an individual project basis, and the spatial scale is on the parcel level, it is difficult to address issues over a long time-scale on a watershed basis. Thus, new management regimes, such as wastewater management districts, may need to be developed. For Ashumet Pond, the Superfund program's focus has historically been on the VOCs being discharged into the pond and the safety of swimming and consuming fish caught in these areas. It was known for many years that the former MMR sewage treatment plant discharged high levels of phosphorus, which remained in the sediments in the saturated zone upgradient of Ashumet Pond. AFCEE has studied the phosphorus discharge into the pond and its ecological impacts in more recent times, since it is the major ecological stressor in the pond.

One of the most valuable products of watershed ecological risk assessment is the conceptual model, which allows citizens and local managers to view the watershed stressors and their associated ecological effects from a more holistic perspective. It emphasizes both point and nonpoint sources of pollution and shows how valued ecological resources are impacted by multiple stressors. More conventional risk assessments and fisheries management efforts focus on describing a dose-response relationship between a single stressor on a single species or on managing fisheries on stock-by-stock basis. By having such a narrow focus, assessment and management efforts would not specifically address the effects of multiple pollutants and habitat effects on the fish community. The model should aid in the development of strategies to deal with these problems in a more comprehensive, innovative, and successful fashion than in the past.

APPENDIX A: Supplemental Information on the Waquoit Bay Estuarine Complex

The Waquoit Bay watershed covers approximately 53 km^2 (21 mi^2) and spans parts of the towns of Falmouth, Mashpee, and Sandwich on the south coast of Cape Cod, MA. The watershed was first delineated by Babione (1990) and further refined by Cambareri et al. (1992) and Sham et al. (1995). The watershed covers 8 km (5 mi) from the head of the bay to the regional groundwater divide. The bay and its tributaries encompass a total surface water area of 3.9 km^2 (389 ha or 1.5 mi^2). The major surface water components of the watershed include Waquoit Bay, two major rivers and several smaller streams, freshwater ponds, and freshwater wetlands. Within the Waquoit Bay watershed are seven subwatersheds (Childs River, Sage Lot Pond, Quashnet River, Eel Pond, Head of the Bay, Hamblin Pond, and Jehu Pond) and three freshwater ponds (Ashumet, Johns, and Snake). These subwatersheds and adjacent ponds provide diverse habitats, including barrier beaches, eelgrass beds, saltwater and freshwater marshes, coastal sand dunes, and brackish and freshwater ponds.

A.1. Geological and hydrological characteristics

The Waquoit Bay watershed lies entirely within the Mashpee pitted outwash plain (LeBlanc et al., 1986), a geologically young landform composed of glacial materials deposited on top of bedrock toward the end of the Wisconsinian Glacial Stage, about 12,000 years ago (Oldale, 1992). Outwash plains were created by broad meltwater streams that size-sorted the drift materials, depositing the heavier boulders and pebbles near the glacial margin and gravel and sands farther away. Because Cape Cod is geologically young, the glacial materials have not been altered significantly, resulting in a generally sandy, porous soil throughout the area. Clay and silt lenses are also found in deeper sediments to the south.

The outwash plain is “pitted” as a result of the numerous kettle ponds dotting the landscape. Kettle ponds mark the sites where blocks of ice were buried by sediment-laden meltwater streams beyond the glacial margin. Johns Pond and Ashumet Pond are two examples of freshwater kettle ponds in the watershed. Waquoit Bay itself may have originated as a kettle pond. The southern margin of the bay was flooded by sea-level rise at the close of the Wisconsinian Glacial Stage. At this point the ice sheet retreated, the low-lying coastal areas were inundated, and the water table inland was raised due to hydrostatic pressure at the saltwater-freshwater interface. The action of winds, waves, and currents continually eroded and displaced the loose glacial sand and gravel, contributing to the formation of coastal sand dunes, sea cliffs, barrier beaches, and salt marshes. These processes continue to alter the dynamic shore (Oldale, 1992).

The geology of Waquoit Bay controls the region's hydrology, which is typical of a glacial outwash plain. The bay, 1.2 km (4,000 ft) wide and 3.4 km (11,000 ft) long, is a shallow-water system (average depth of 1.5 m). It receives input from freshwater streams and ground-water flow, with tidal exchange to Vineyard Sound through two dredged and maintained channels and a recent breach caused by overwash during a hurricane in August 1991 (Valiela et al., 1996). Fifty percent of the water entering Waquoit Bay comes from the Quashnet and Childs rivers, 23% comes from direct precipitation, and 27% comes from ground-water seepage (Cambareri et al., 1992). The rivers derive most of their water from ground-water discharge, draining the shallow surface aquifer. Groundwater is forced to the surface as the permeable aquifer thins from north to south in the watershed.

The unconsolidated sediments of Cape Cod make ideal aquifers. The permeable aquifer is about 46 m (150 feet) thick near Snake Pond and thins to 9 m (30 feet) near Waquoit Bay (Garabedian et al., 1991, Cambareri et al., 1992). The porous soils support rapid percolation of rain, nutrients, and contaminants into the groundwater. In recognition of the unique ground-water characteristics of Cape Cod, the U.S. Environmental Protection Agency declared this region a sole-source aquifer in 1982, a designation that facilitated protection of the water supply. The Cape Cod aquifer can be subdivided into six groundwater lenses, and generally, groundwater does not flow between lenses. The Waquoit Bay watershed lies within the Sagamore or western Cape lens of the Cape Cod Aquifer (Guswa and LeBlanc, 1981).

A.2 Biological characteristics

The waters of the southward-flowing, cold Gulf of Maine and the northward-flowing, warm Gulf Stream mix off of the coast of Cape Cod to form a biological transition zone between the Virginian (temperate) and Acadian (boreal) biogeographic provinces (Ayvazian et al., 1992). This overlap produces more diverse communities than occur in either province. The Waquoit Bay estuarine complex benefits from this increased diversity. The watershed also lies near the Atlantic coast flyway, an important migratory corridor for many coastal and arctic-nesting birds, particularly shorebirds.

The flora of the watershed include scrub oak and pitch pine forests (Bailey, 1995). Forests covered 2,650 ha (6,548 acres) of the watershed in 1990. Among the state-protected plant species found in the watershed are the endangered sandplain gerardia *(Agalinis acuta)*; the threatened bushy rockrose *(Helianthuemum dumosum)*; the knotroot foxtail *(Setaria geniculata)*, which is of special concern; and the butterfly weed *(Asclepias tuberosa),* little ladies' tresses *(Spiranthes tuberosa)*, eastern lilaeopsis *(Lilaeopsis chinensis)*, New England blazing star *(Liatris borealis)*, thread-leaved sundew *(Drosera filiformis)*, vetchling *(Lathyrus palustris)*, and wild rice *(Zizania aquatica)*, which are on the watch list (Geist, 1996) to attempt to keep them from becoming threatened or endangered species.

The fish in Waquoit Bay include freshwater, estuarine, and marine species. The part-time residents represent a composite of estuarine spawners, such as winter flounder (*Pseudopleuronectes americanus*), longhorn sculpin (*Myoxocephalus octodecemspinosus*), scup (*Stenotomus chrysops*), and tautog (*Tautoga onitis*); marine species that are estuarine visitors, such as the sand lance (*Amodytes americanus*), summer flounder (*Paralichthys dentatus*), and American pollack (*Pollachius virens*); nursery species or young-of-the-year, such as winter flounder juveniles, mullets (*Mugil cephalus*), juvenile tautogs, menhaden (*Brevoortia tyrannys*), Atlantic silversides (*Menidia menidia*), bluefish (*Pomatomus saltatrix*), and bay anchovy (*Anchoa mitchilli*); and adventitious species that have a more southern distributions but that lack an apparent estuarine dependence, such as ladyfish (*Elops saurus)*, halfbeak (*Hemiramphus brasiliensis*), and crevalle jack (*Caranx hippos)*. Alewives (*Alosa pseudoharengus*) and blueback herring (*Alosa aestivalis*) cross Waquoit Bay on their annual spawning migrations to fresh water, and larger fish such as bluefish and striped bass (*Morone saxitali*s) enter in pursuit of smaller prey fish. Many primarily marine fishes use the estuary in the winter as a spawning and nursery ground. Bluefish, tomcod (*Microgadus tomcod*), white hake *(Urophycis tenuis)*, and pollack inhabit the bay as juveniles but are rarely present as adults.

Shellfish species harvested in the estuary include bay scallops (*Argopecten irradians*), which are found in the eelgrass habitat, and hardshell (*Mercenaria mercenaria*) and softshell (*Mya arenaria*) clams, generally found in the sand and mud habitats, respectively. The biota of the estuary also include a variety of temperate and boreal species of planktonic and benthic algae and invertebrates, which provide food resources for the finfish as well as the terrestrial and avian wildlife in the watershed.

Numerous shorebirds use the barrier beach and coastal salt marsh as an important stopover on their spring journey north to breeding grounds in Canada and on their fall journey south to the southern United States and Central and South America. Shorebirds that appear in abundance in the spring and fall on Waquoit Bay's barrier beaches include black-bellied *(Squatarola squatarola)* and semipalmated *(Charadrius semipalmatus)* plovers; sanderlings *(Crocethia alba)*; dunlin *(Calidris alpina)*; semipalmated *(Ereunetes pusillus)*, least *(Pisobia fusicollis)*, and western sandpipers *(Pisobia minutilla)*; ruddy turnstones *(Arenaria interpres)*;

willets *(Catoptrophorus semipalmatus)*; lesser *(Totanus flavipes)* and greater *(Tringa melanoleuca)* yellowlegs; and short-billed dowitchers *(Limnodromus griseus)*. Sharp-tailed sparrows *(Ammodramus cudacutus)*, black-crowned night-herons (*Nycticorax nycticorax*), snowy egrets *(Leucophoyx thula)*, and mute swans *(Cygnus olor)* are found in the saltmarshes. Several species of birds that use the waters as nesting or feeding grounds are state- and federally protected species. The piping plover *(Charadrius melodus)*, listed as threatened, and the least tern *(Sterna antillarum)*, listed as being of special concern, nest on South Cape Beach and Washburn Island. The roseate tern *(Sterna dougalli)*, an endangered species, forages in the water and rests on the beach proper (Geist, 1996).

The Childs and Quashnet rivers provide a relatively rare and shrinking habitat for several anadromous and catadromous finfish species (Baevsky, 1991). Brown trout *(Salmo trutta)*, brook trout *(Salvelinus fontinalis)*, alewife *(Alosa aestivalis)*, and white perch *(Morone americana)* use these rivers as spawning grounds either within the rivers themselves or within John's Pond (McLarney, 1988; Hurley, 1990). American eels *(Anguilla rostrata)* use these rivers to reach spawning grounds in the open sea. These species require very specific ranges of water-quality parameters (temperature, pH, dissolved oxygen, salinity) for development (Hunter, 1991). Under the care of the northeast chapter of Trout Unlimited, the ecological integrity and stability of the Quashnet River have improved significantly.

The good-quality water of the Quashnet River also provides habitat for a variety of macroinvertebrates that serve as a food source for the finfish communities (Pennak, 1989). As part of Trout Unlimited’s restoration project, macroinvertebrate species were reintroduced to the Quashnet from other freshwater streams. A survey done in 1982–1983 found species representing the Trichoptera (caddisfly), Diptera (true flies), Lepidoptera (butterflies and moths), Ephemeroptera (mayflies), and Plecoptera (stoneflies) orders. Stoneflies, and to some extent mayflies and caddisflies, are good indicators of healthy water quality, as they require fairly high levels of dissolved oxygen.

APPENDIX B: Organizations Concerned About Waquoit Bay

Ashumet – John's Pond Association
Ashumet Valley Property Owners Association
Association for the Preservation of Cape Cod
Atlantic States Marine Fisheries Commission
Barnstable County Department of Health and the Environment
Buzzards Bay National Estuary Program
Cape and Islands Coastal Waters Steering Committee
Cape Cod Beagle Club
Cape Cod Commission
Cape Cod Cooperative Extension Service
Cape and Islands Self Reliance Corporation
Citizens for the Protection of Waquoit Bay
Davisville Association
Falmouth Condo Trust
Falmouth Conservation Commission
Falmouth Rod and Gun Club
Green Briar Nature Center
League of Women Voters, Falmouth
Mashpee Briarwood Association, Inc.
Mashpee Conservation Commission
Mashpee Harbor Master
Mashpee Shellfish Department
Massachusetts Coastal Zone Management
Massachusetts Department of Fisheries, Wildlife, and Environmental Law Enforcement
Massachusetts Audubon Society
Massachusetts Department of Environmental Protection
Massachusetts Military Reservation
Massachusetts Heritage Society
Menauhant Harbor Association
Monomoscoy Improvement Trust
National Science Foundation's Land-Margin Ecosystems Research Program
NOAA's National Marine Fisheries Service
NOAA's National Estuarine Research Division
Otis Installation Restoration Program
Seacoast Shores Owners Association
Shorewood Beach Owners
Sierra Club – Cape Cod Group
South Cape Beach Advocates
The Nature Conservancy
The 300 Committee
Town of Mashpee

Town of Falmouth
Town of Sandwich
Trout Unlimited
U.S. Fish and Wildlife Service
U.S. Geological Survey
U.S. Army Corps of Engineers
U.S. Department of Agriculture Soil Conservation Service
U.S. EPA, Region 1
Wampanoag Tribal Council
Waquoit Bay Watershed Citizens Action Committee
Waquoit Bay Watershed Inter-municipal Committee
Waquoit Bay National Estuarine Research Reserve

APPENDIX C: Public Concerns and Waquoit Bay Stressors

At a public forum on September 21, 1993, of local environmental officials, scientists, and stakeholders, the following information was generated on environmental concerns and resources that should be protected and the stressors and ecological effects of those stressors on Waquoit Bay.

C.1. Public Concerns

Open space
Indigenous wildlife
Scenic views
Flyway integrity (migrating waterfowl)
Recreation (swimming)
Noneconomic values
Traditional life styles
Historical/political perspective
"Historical" bay ecosystem structure
Food resource safety
Tourists
Shellfishery
Shoreline
Wildlife
Vegetation
Clean water
Clean air
Marshland
Upland-marsh ecotone
Habitat
Groundwater quality
Flushing rates
Air quality
Washburn Island
Human health and domestic animal health
Recreational "atmosphere"
Water quality
Eelgrass
Marine organisms
Finfishing
River herring

"Quality of life" including:

Access to natural beauty
Freedom to enjoy
Human serenity
Natural noise
Night sky/darkness
Visual beauty
Pleasant sensory experiences
Sights
Smells

C.2 Sources and Stressors in the Waquoit Bay Watershed

Source	Stressor	Type	Ecological effects
Septic systems, fertilizers, atmospheric deposition	Nitrogen	C	Increase in macroalgae and phytoplankton growth
Septic systems	Pathogens	B	Introduction of pathogens and fecal coliforms to surface water
	Fecal coliforms	B	Shellfish bed contamination
Nutrient input	Shading by macroalgae	P	Alteration of substrate, light attenuation
	Shading by macroalgae	B	Major faunal alterations in benthic and fish communities
	Increase in macroalgal growth	B	Alteration of macroalgal species composition, loss of habitat for submerged aquatic vegetation, loss of spawning sites for fish, loss of hiding places and protection for fish, loss of scallop larvae settling habitat
	Increase in macroalgal growth	P	Change in water coloration
Macroalgal growth	Increased respiration of macroalgae	C	Decrease in dissolved oxygen within water column
	Increased respiration of macroalgae	B	Mortality of benthic invertebrates and fish
	Competition by macroalgae	B	Loss of eelgrass habitat
Introduced predators	Predation	P, B	Disturbance of habitat for endangered and threatened wildlife
Introduction of exotic species	Mute swan	B	Displacement of native waterfowl species
Marinas and piers	Antifouling chemical leachate	C	Negative biological effects on organisms

C=chemical
P=physical
B=biological

C.2 Sources and Stressors in the Waquoit Bay Watershed (continued)

Source	Stressor	Type	Ecological effects
Gasoline and motor oil from automobile and boat engines	Organic compounds	C	Unknown
Massachusetts Military Reserve, Otis Air Force Base	Organic compounds	C	Unknown
Landfill leachates	Organic compounds	C	Unknown
Construction	Seawalls and jetties	P	Major alteration of shoreline dynamics, sediment resuspension, coastal erosion, sediment build-up, change in flushing rates
Boating	Boat propellers	P	Destruction of vegetation, sediment resuspension, increased turbulence and mixing in water column
Commercial shellfishing	Raking and plunging for scallops	P	Disturbance of sediment, resuspension of nutrients, increasing turbidity
Development	Filling wetlands	P	Loss of marsh-uplands ecotone, increase in surface water runoff, increase in sediment loading, altered ground-water flow
	Dredging channels	P	Sediment disturbance and increase in turbidity

C=chemical
P=physical
B=biological

APPENDIX D: Attendees at the Waquoit Bay Management Goals Meeting

Attendee	Affiliation
Tom Cambareri	Cape Cod Commission
Bruce Carlisle	Massachusetts Coastal Zone Management
Joe Costa	Buzzards Bay National Estuary Program
David Dow	National Marine Fisheries Service
Perry Ellis	Mashpee Harbor Master
Tom Fudula	Mashpee Planning Department
Jeroen Gerritsen	Tetra Tech Inc.
Steve Hurley	Massachusetts Division of Fisheries and Wildlife
Chuck Lawrence	Cape Cod Commission
Sandy McClean	Citizens for the Protection of Waquoit Bay
Carl Melberg	U.S. Fish and Wildlife Service
JoAnne Muramoto	Falmouth Conservation Commission
Mark Patton	Otis Installation Restoration Program
Pam Polloni	League of Women Voters, Falmouth
Bob Sherman	Mashpee Conservation Commission
Jan Smith	Massachusetts Coastal Zone Management
Patti Tyler	U.S. EPA, Region 1
Mary Varteresian	U.S. Fish and Wildlife Service
Brooks Wood	Monomoscoy Improvement Trust
Rick York	Mashpee Shellfish Department

APPENDIX E: Information on Contamination from the Massachusetts Military Reservation

E.1 Phosphorus Loading to Ashumet Pond

Between 1936 and 1995 the sewage treatment plant at the Massachusetts Military Reserve (MMR) discharged treated sewage into infiltration ponds on the southern edge of the facility that contributed both toxic contaminants and high phosphate levels to the Ashumet Valley Plume. The U.S. Geological Survey (USGS) established a study to examine the transport of the dissolved phosphorus contained in the Ashumet Valley Plume. This study showed that the bulk of the phosphorus is trapped on sediment particles. The Air Force Center for Environmental Excellence (AFCEE) agreed to mitigate the high phosphorus levels in the Ashumet Valley Plume as part of the Installation Restoration Program (IRP), even though this is not a Superfund issue. Because the bulk of the phosphorus is trapped in the sediments in the saturated zone upgradient of Ashumet Pond, AFCEE decided to fund studies that would develop a phosphorus budget for Ashumet Pond, determine the existing water quality and the limiting nutrients for phytoplankton growth, and develop in-pond mitigation strategies for phosphorus. In September 2001, AFCEE and its contractors completed a successful alum addition to the deep hole in Ashumet Pond in order to trap the phosphorus in the sediments and thus reduce the dissolved phosphorus loading to the hypolimnion (bottom waters) during summer stratification. The buffered dose of alum and sodium aluminate did not change the pH or alkalinity in the receiving water, and the dissolved aluminum concentration in the water was below the detection limit (1 ppm). No fish kills accompanied this treatment process.

The results of the AFCEE supported other investigations showing that phosphorus discharges from the Ashumet Valley Plume into Ashumet Pond in the Fisherman's Cove region reached at an annual loading rate of 180 kg P/yr, with maximum dissolved phosphorus concentrations of roughly 2 mg/L. Other external loading rates were 51–63 kg P/yr for other groundwater and 13–22 kg P/yr for surface water inflows. This loading represented new phosphorus into the system, which could be used in a Vollenweider Loading Model to estimate the resulting chlorophyll levels and dissolved phosphorus concentrations in the water column. The chlorophyll concentrations measured in the summer of 1999 were less than 1 µg/L, and the total dissolved phosphorus levels were less than 15 ppb, suggesting that the system is mesotrophic.

Experiments in nutrient uptake (measured by alkaline phosphatase induction, which estimates the potential for converting organic phosphorus to inorganic phosphorus) were conducted in the phytoplankton after the spring bloom. It is unknown whether silicon or phosphorus is limiting to the spring diatom bloom. Cyanobacteria dominate the phytoplankton in the surface waters during the summer, with a seasonal succession from phosphorus limitation to nitrogen fixers and back to phosphorus limitation, based on the available dissolved inorganic nitrogen and phosphorus concentrations in the water. Thus a dynamic interaction apparently exists between the dissolved nutrient levels and the species that dominate the phytoplankton in the epilimnion (surface waters). Because zooplankton cannot graze many cyanobacteria effectively, this was considered an ecological stressor that might not be eliminated by reducing the phosphorus loading in the bottom waters. Cyanobacteria blooms can also cause taste and odor problems in surface waters.

Anoxia develops in the deeper, stratified portion of the pond between May and October, resulting in recycling of phosphorus from the sediments back into the water column at an annual rate of 194 kg P/yr. For the shallow (less than 7.6 m) sediments in the pond that remain oxic throughout the summer, the recycling rate is 195 kg P/yr. Within the epilimnion, the annual recycling rate is 3,370 kg P/yr. Thus the recycled phosphorus greatly exceeds the annual external loading rate of new phosphorus. The recycled phosphorus is not included in the Vollenweider model. Presumably the particulate phosphorus that settles into the bottom waters from the surface is balanced by the dissolved phosphorus recycled from the sediments.

Seasonal anoxia resulted in fish kills in Ashumet Pond in July 1985 and May 1996. Seasonal anoxia is a result of bacterially mediated degradation of particulate organic matter in the hypoliminion. This degradation uses up dissolved oxygen in the bottom water and, because of water column stratification, it cannot be replenished. From an ecological risk assessment perspective, this anoxia/hydroxia is an ecological stressor, even though the surface water is classified as mesotrophic. The amount of anoxia/hypoxia resulting from the morphology of the kettle hole ponds and the amount due to the nutrient loading and phytoplankton production in the surface water are unknown. The USGS has prepared a rough phosphorus budget for Ashument Pond (oral communication between D. LeBlanc, USGS, Marlboro, MA, and David Dow NOAA/NMFS/ NEFSC, Woods Hole Laboratory, Woods Hole, MA regarding poster presented at the Ashmut Pond phosphorus project update meeting, Barnstable County Fairgrounds, MA, August 2, 2001, by D. LeBlanc) suggesting that the new phosphorus loading sources are distributed as follows: precipitation, 10%; surface runoff, 10%; MMR Ashumet Valley Plume, 50%; and other groundwater inputs, 30%.

E.2 Volatile Organic Compound (VOC) Contamination in Plumes

MMR was designated as a Federal Superfund site in 1989, and since that time 14 ground-water plumes have been identified that emanate from the base and extend into the surrounding communities (Bourne, Falmouth, Mashpee, and Sandwich). The VOCs at MMR mainly come from fuel spills, which produce benzene, toluene, ethylbenzene, xylene and ethylene dibromide contaminants, or from chemical spills, which produce chlorinated solvents, such as trichloroethylene (TCE), tetrachloroethylene, carbon tetrachloride, etc. The benzene, toluene, ethylbenzene, and xylene are often biodegraded by aerobic metabolism near the source areas, whereas the ethylene dibromide in aviation fuel is left behind in detached plumes downgradient from the source area. The chlorinated solvents are subject to slow anaerobic metabolism via reductive dechlorination in the groundwater, but they tend to be persistent, like the ethylene dibromide in the fuel spill plumes. The chemical spill-10 plume discharges into Ashumet Pond, with a portion passing through the isthmus connecting Johns and Ashumet ponds and discharging into Johns Pond, where it is referred to as the TCE Plume. The storm drain-5 plume discharges small quantities of trichloroethylene into Johns Pond as well. Fuel spill-1 plume is a detached plume that discharges ethylene dibromide into the Quashnet River downstream from its Johns Pond source. The elliptical shapes in Figure E-1 illustrate the locations and extents of these plumes.

The old fire-training-area site (site 1) is the source area for the Ashumet Valley Plume, and the primary constituents within this ground-water plume have been identified as VOCs and inorganics. Fire-training-area-1 is a 3-acre parcel of land where fire-training exercises were conducted from 1958 to 1985 and materials burned here consist of different fuel types, waste oils, solvents, thinners, transformer oils, and spent hydraulic fluids. Southeast Regional Groundwater Operable Unit plumes originate from fire-training-area-2, landfill-2, petroleum fuel storage area, and storm drain-5. Fire-training-area-2 and landfill-2 comprise a 20-acre area of land located on top of a former industrial/ municipal landfill that was used for fire-training exercises. Compounds disposed of in the landfill or burned on the fire-training area consist of fuel, waste oils, waste petroleum distillate solvents, and domestic refuse. The petroleum fuel storage area is an active facility that is involved in the delivery of various types of fuel. It was the site of a 2,000 gallon fuel spill in the 1960s.

AFCEE leads the mitigation of these underground pollution plumes and the associated source areas under the Superfund program. The following is a brief description of the mitigation programs that are either underway or planned for both the plumes and their source areas, which are often multiple for a given plume. The plume mitigation goal is to reduce the contamination

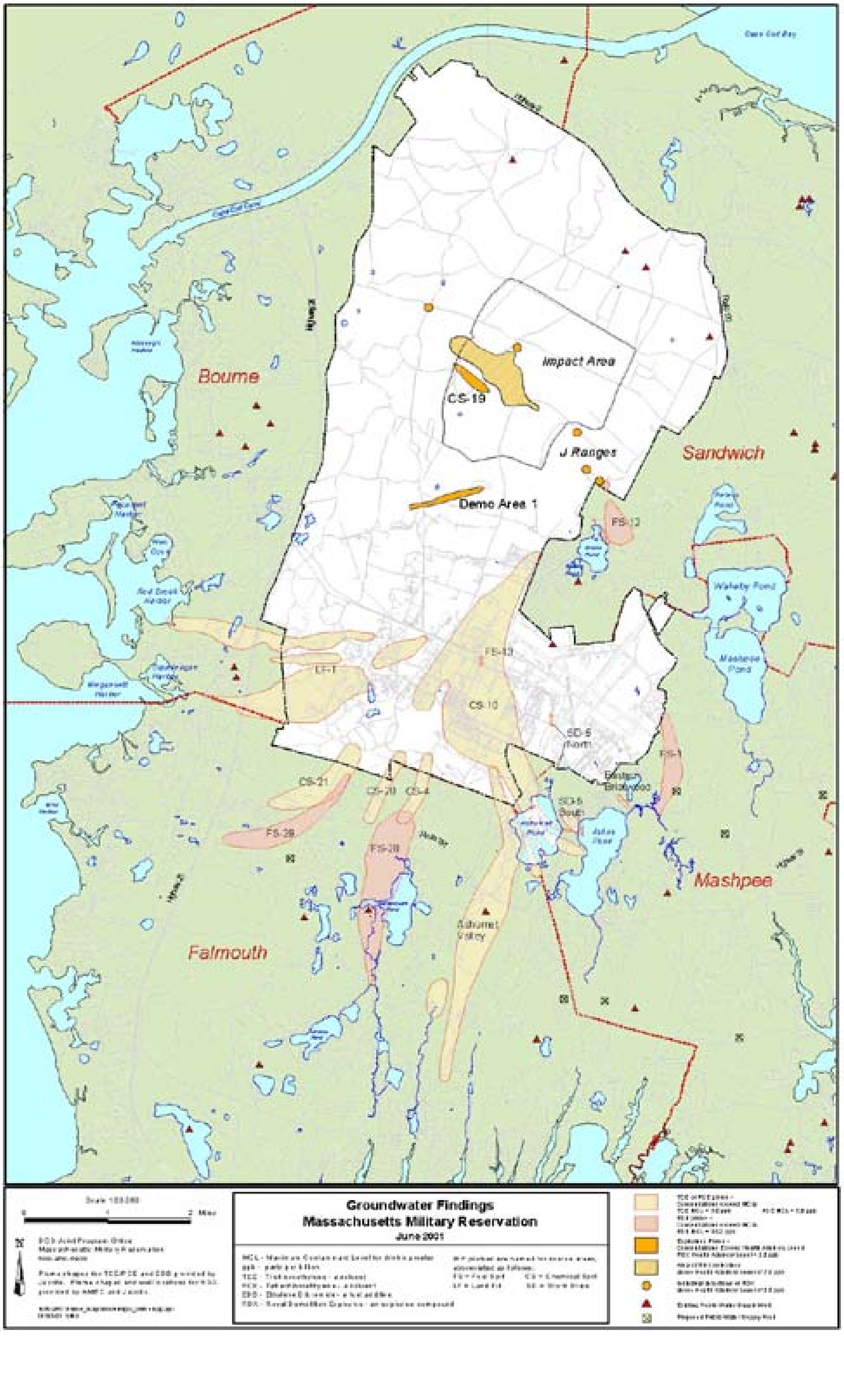

Bourne
Sandwich
Falmouth
Mashpee
Impact Area
CS-19
J Ranges
Demo Area 1
FS-12
FS-13
LF-1
CS-10
SD-5
North
FS-1
CS-21
CS-20
CS-4
SD-5
South
FS-29
FS-28
Ashumet
Valley
Wakeby Pond
Mashpee
Pond
Groundwater Findings
Massachusetts Military Reservation
June 2001

Figure E-1. Map of the plumes emanating from the Massachusetts Military Reservation.

to below maximum contaminant levels (5 ppb for VOCs and 0.020 ppb for ethylene dibromide), which are much lower than the acute/chronic toxicity levels for biota, and then to conduct a risk assessment and feasibility study to see whether cleanup should proceed to nondetect levels. The source removal program is on a separate, less-advanced timeline than the plume cleanup.

The chemical spill-10 plume is the largest, most contaminated plume at the MMR Superfund site. For the chemical spill-10 plume plume, the proposed treatment system will feature a combination of shallow well points and four deep-extraction wells that withdraw 750 gpm, followed by reinjection of the treated water into an infiltration trench. The extracted water is treated with granulated activated carbon treatment and is reinjected back into the groundwater. The portion of the chemical spill-10 plume downgradient of the Sandwich Road extraction, treatment, and reinjection discharges into the western side of Ashumet Pond, while a portion flows under the pond, emerging as the TCE Plume on the isthmus connecting Ashumet Pond to Johns Pond. It is unknown whether the Ashumet Pond trichloroethylene discharge has any toxic effects on the benthos, but risk assessments suggest that it does not pose a threat to human health. The TCE Plume has concentrations of 263 ppb on the eastern side of Ashumet Pond, with maximum concentrations near the Johns Pond upwelling point of 1502 ppb trichloroethylene and 6 ppb tetrachloroethylene. This plume is only 75 feet wide and has a maximum trichloroethylene level of 2,200 ppb, so it is very narrow, but highly contaminated. The maximum concentration in the pond water above the discharge point is 3.5 ppb trichloroethylene.

The storm drain-5 south plume also discharges into Johns Pond, but its maximum trichloroethylene level is only 60 ppb, with the average near 10 ppb trichloroethylene. Even though it has a larger discharge footprint within Johns Pond than does the TCE Plume, trichloroethylene is not detectable in the pond water above the bottom. Storm drain-5 south has two recirculating wells within the plume that operate at 120 gpm. A proposed extraction, treatment and reinjection well along Hooppole Road would connect to the Sandwich Road Treatment System. A portion of storm drain-5 south will discharge into Johns Pond over the next 10–12 years, as the combined recirculating/ extraction, treatment and reinjection system can only capture 55% of the contaminant mass. There is also a system of 10 extraction wells and eight reinjection wells that captures the northern portion of the storm drain-5 north plume.

In Johns Pond, the brown bullheads have exhibited papillomas around their mouths/ jaws and adenocarcinomas in the liver, which has created controversy about whether there is a link between the trichloroethylene discharges from storm drain-5 south and the TCE Plume or whether these tumors stem from viruses or genetic factors. Even though trichloroethylene is not bioaccumulated in fish, it can bioconcentrate 15- to 30-fold over the concentrations found in the water. Past studies of these fish detected polycyclic aromatic hydrocarbons in the fish flesh, and induction of the mixed-function oxidase enzyme system that metabolizes these contaminants. No tests were conducted on the induction of trichloroethylene-metabolizing enzymes in these fish, so there is no direct evidence for an impact of the trichloroethylene discharged into Johns Pond on the biota. Fuel spill-1 plume discharges into springs that feed the upper portions of the Quashnet River, and ethylene dibromide has been detected in the river water at 1.4 ppb, with a maximum concentration within the groundwater of 10 ppb. The K1 cranberry bog area on the Quashnet River is an important breeding habitat for eastern brook trout. Because of the concern about ethylene dibromide contamination of cranberry bogs in the discharge region, an elaborate bog separation program has been put in place, along with an a pilot treatment system. This system has 138 shallow wells points that extract 400 gpm and one deep-extraction well that withdraws 200 gpm. Water from these wells will undergo onsite granulated activated carbon treatment and then be reinjected. The surface discharge has a diffuser in order to maintain the dissolved oxygen at levels (above 8 ppm) necessary for brook trout breeding. A plan has been developed by AFCEE and approved by the U.S. Environmental Protection Agency and the Massachusetts Department of Environmental Protection to place three additional extraction wells in the southern toe of the plume and use the existing treatment plant.

Snake Pond is the farthest upgradient freshwater pond in the Waquoit Bay watershed, and it has a fish advisory in effect because of mercury levels. The advisory was issued by the Massachusetts Department of Public Health on the basis of human health concerns and recommends no fish consumption by children younger than 12 years or by pregnant/nursing women. The presumed source of the mercury is the atmosphere. It is unknown whether these higher mercury levels have an impact on the fish themselves or on other biota, even though the scientific literature suggests possible effects on growth for some freshwater species (Stafford and Haines, 2001). Groundwater underneath Snake Pond has elevated levels of ethylene dibromide (above the maximum contaminant level of 0.020 ppb). The ethylene dibromide contaminants are associated with the fuel spill-12 plume and explosives, such as Royal Dutch Eplosive (RDX), from contamination emanating from the J-3 Range in the Impact Area at Camp Edwards. However, these contaminants have not been detected in either the water or the sediments of Snake Pond, even though models predict that the RDX will upwell into the pond. Thus, no well-defined ecological stressors from human activities have been identified in Snake Pond.

REFERENCES

Aber, JD; Magill, R; Boone, JM; et al. (1993) Plant and soil responses to chronic nitrogen additions at the Harvard Forest, Massachusetts. Ecological Applications 3:156–166.

AFCEE (Air Force Center For Environmental Excellence). (2001) Final Ashumet Pond Phosphorus Management Plan. Brooks Air Force Base, San Antonio, TX.

Aguilar, C; Fogel, ML; Paerl, HW. (1999) Dynamics of atmospheric combined inorganic nitrogen utilization in the coastal waters off North Carolina. Marine Ecology Progress Series 180:65–79.

Andreoli, A; Bartilucci, N; Forgione, R; et al. (1979) Nitrogen removal in a subsurface removal system. Journal of the Water Pollution Control Federation 51:841-855.

Aravena, R; Evans, ML; Cherry, JA. (1993) Stable isotopes of oxygen and nitrogen in source identification of nitrate from septic systems. Ground Water 31:180-186.

Ayvazian, SG; Deegan, LA; Finn, JT. (1992) Comparison of habitat use by estuarine fish assemblages in the Acadian and Virginian zoogeographic provinces. Estuaries 15:368-383.

Babione, M. (1990) Land use change in the watershed of Waquoit Bay, Massachusetts. Division III Exam, School of Natural Science, Hampshire College, MA.

Baevsky, YH. (1991) Physical and water-quality characteristics affecting trout-spawning habitat in the Quashnet River, Cape Cod, Massachusetts. USGS Water Resources Investigations Report 91-4045. USGS, Marlborough, MA.

Bailey, RG. (1995) Eastern broadleaf forest (oceanic) province. In: Description of the ecoregions of the United States. U.S. Department of Agriculture, Forest Service, Miscellaneous Publication 1391, 2nd Edition. Washington, DC.

Billen, G; Somville, M; De Becker, E; et al. (1985) A nitrogen budget of the Scheldt hydrographical basin. Netherlands Journal of Sea Research 19:223-230.

Billen, G; Lancelot, C; Meybeck, M. (1991) N, P, and Si retention along the aquatic continuum from land to ocean. Pp. 19-44 in R. F. C. Mantoura, J.-M. Martin, and R. Wollast, editors. Ocean Margin Processes in Global Change. New York: John Wiley & Sons.

Bowen, JL; Valiela, I. (2001a) The ecological effects of urbanization of coastal watersheds: Historical increases in nitrogen loads and eutrophication of Waquoit Bay estuaries. Canadian Journal of Fisheries and Aquatic Sciences 58:1489–1500.

Bowen, JL; Valiela, I. (2001b) Historical changes in atmospheric nitrogen deposition to Cape Cod, Massachusetts, USA. Atmospheric Environment 35:1039–1051.

Bowen, JL; Valiela, I. Nitrogen loads to estuaries: Using loading models to assess the effectiveness of management options to restore estuarine water quality. Estuaries: to be submitted.

Boynton, WR; Kemp, WM; Keefe, CW. (1982) A comparative analysis of nutrients and other factors influencing primary production. p. 69–80. In: Estuarine comparisons, Kennedy, VS (ed). New York: Academic Press.

Brawley, JW; Collins, G; Kremer, JN; et al. (2000) A time-dependent model of nitrogen loading to estuaries from coastal watersheds. Journal of Environmental Quality 29:1448–1461.

Butler, AJ; Connolly, RM. (1996) Development and long-term dynamics of a fouling assemblage of sessile marine invertebrates. Biofouling 9:187–209.

Cambareri, TC; Eichner, EM; Griffith, CA. (1992) Sub-marine groundwater discharge and nitrate loading to shallow coastal embayments. In: Proceedings, Eastern Regional Groundwater Conference, Newton, MA. Oct. 13–15, 1992. National Groundwater Association, Westerville, OH.

Cameron. KC; Wild, A. (1982) Comparative rates of leaching of chloride, nutrients, and tritiated water under field conditions. Journal of Soil Science 33:649-657.

Cohen, JE; Small, C; Mellinger, A; et al. (1997) Estimates of coastal populations. Science 278:1211.

Collins, G; Kremer, JN; Valiela, I. (2000) Assessing uncertainty in estimates of nitrogen loading to estuaries for research, planning and risk assessment. Environmental Management 25:635–645.

Correll, DL; Ford, D. (1982) Comparison of precipitation and land runoff as sources of estuarine nitrogen. Estuarine, Coastal, and Shelf Science 15:45–56.

Correll, DL; Jordan, TE; Weller, DE. (1992) Nutrient flux in a landscape: effects of coastal land use and terrestrial community mosaic on nutrient transport to coastal waters. Estuaries 15:431-442.

Costa, JE. (1988) Distribution, production, and historical changes in abundance of eelgrass (*Zostera marina*) in Southeastern Massachusetts. Dissertation. Boston University, Boston, MA.

Costa, JE; Howes, BL; Giblin, AE; et al. (1992) Monitoring nitrogen and indicators of nitrogen to suggest management action in Buzzards Bay. In: Mckenzie, DH; Hyatt, DE; McDonald, VJ; eds. Ecological Indicators 1:499-534.

Cottam, C. (1933) Eel grass dying of mysterious disease. Science News Letters 24:73.

Cotton, AD. (1933) Disappearance of *Zostera marina*. Nature 132:277.

Cubbage, A; Lawerence, D; Tomasky, G; et al. (1999) Relationship of reproductive output in *Acartia tonsa*, chlorophyll concentration, and land-derived nitrogen loads in estuaries in Waquoit Bay, MA. Biol Bull 197:294–295.

D'Avanzo, C; Kremer, J. (1994) Diel oxygen dynamics and anoxic events in an eutrophic estuary of Waquoit Bay Massachusetts. Estuaries 17:131–139.

D'Elia, CF; Steudler, PA; Corwin, N. (1977) Determination of total nitrogen in aqueous samples using persulfate digestion. Limnology and Oceanography 22:760–764.

Davies, TD; Pierce, CE; Robinson, HJ; et al. (1992) Towards an assessment of the influence of climate on wet acidic deposition in Europe. Environmental Pollution 75:111-120.

Deegan, LA; Buchsbaum, RN. (In press) The effect of habitat loss and degradation on fisheries. In: The Decline of Fisheries Resources in New England: Evaluating the Impact of Overfishing, Contamination, and Habitat Degradation. Buchsbaum, RN; Robinson, WE; Pederson, J (eds). Massachusetts Bays Program, University of Massachusetts Press, Amherst, MA.

Diamond, JM; Serveiss, VB. (2001) Identifying sources of stress to native aquatic species using a watershed ecological risk assessment framework. Environmental Science and Technology (35)24:4711–4718.

Downing, JA; Osenberg, CW; Sarnelle, O. (1999) Meta-analysis of marine nutrient enrichment experiments: Variation in the magnitude of nutrient limitation. Ecology 80:1157–1167.

Duarte, CM. (1995) Submerged aquatic vegetation in relation to different nutrient regimes. Ophelia 41:87–112.

Eichner, EM; Cambareri, TC. (1992) Nitrogen Loading. Cape Cod Commission Technical Bulletin 91-001. Barnstable, MA.

Evgenidou A; Valiela, I. (In press) Response of growth and density of a population of Geukensia demissa to land-derived nitrogen loading to Waquoit Bay, Massachusetts. Estuarine, Coastal, and Shelf Science.

Faught, MC. (1945) Falmouth, Massachusetts: Problems of a Resort Community. New York: Columbia University Press.

Foreman, K; Tomasky, G; Soucy, LA; et al. (Submitted) Responses of phytoplankton to land-derived nitrogen loads in Waquoit Bay, MA. Aquatic Ecology.

Freyer, HD; Aly, AIM. (1974) ^{15}N variations in fertilizer nitrogen. Journal of Environmental Quality 3:405–406.

Fricke, W; Beilke, S. (1992) Indication for changing deposition patterns in central Europe. Environmental Pollution 75:121-127.

Frimpter, MH; Donahue, JJ; Rapacz, M. (1990) A mass-balance model for predicting nitrate in groundwater. New England Water Works Association 104:219-232.

Gallagher, J. (1983) The South Cape Beach, Mashpee, Massachusetts: An Intensive Survey of the Balance of the Park, Phase 1 (B). Massachusetts Department of Environmental Management, Boston, MA.

Garabedian, SP; LeBlanc, DR; Gelhar, LW; et al. (1991) Large-scale natural gradient tracer test in sand and gravel, Cape Cod, Massachusetts, 2, analysis of spatial moments for a nonreactive tracer. Water Resources Research 27:911–924.

Geist, MA. (1996) The Ecology of the Waquoit Bay National Estuarine Research Reserve. Waquoit Bay National Estuarine Research Reserve, Waquoit Bay, Massachusetts.

GESAMP (Group of Experts on the Scientific Aspects of Marine Pollution). (1990) The State of the Marine Environment. IMO/FAO/UNESCO/WMO/WHO/IAEA/UN/UNEP. Rep. and Stud. 39. Blackwell, London, UK.

Golomb, D; Ryan, D; Eby, N; et al. (1997) Atmospheric deposition of toxics onto Massachusetts Bay: metals. Atmospheric Environment 31:1349–1359.

Gormly, JR; Spaulding, RF. (1979) Sources and concentrations of nitrate nitrogen in the groundwater of the central Platte region, Nebraska. Ground Water 17:291–201.

Griffin, MPA; Valiela, I. (2001) $d^{15}N$ isotope studies of life history and trophic position of Fundulus heteroclitus and Menidia menidia. Marine Ecology Progress Series 214:299–305.

Guswa, JH; LeBlanc, DR. (1981) Digital models of groundwater flow in the Cape Cod aquifer system, Cape Cod, MA. USGS Water Resources Investigations Open File Report 80–67.

Hansson, S; Hobbie, JE; Elmgren, R; et al. (1997) The stable nitrogen isotope ratio as a marker of food-web interactions and fish migration. Ecology 78:2249–2257.

Harlin, MM; Thorne-Miller, B. (1981) Nutrient enrichment of seagrass beds in a Rhode Island coastal lagoon. Marine Biology 65:221–229.

Harris, HJ; Wegner, RB; Harris, VA; et al. (1994) A method for assessing environmental risk: A case study of Green Bay, Lake Michigan, USA. Environmental Management 18:295–306.

Hauxwell, J; Behr, P; McClelland, J; et al. (1998) Grazing and nutrient control on macroalgal biomass in Waquoit Bay, MA. Estuaries 21:347–360.

Hauxwell, J; Cebrian, J; Furlong, C; et al. (2001) Macroalgal canopies contribute to eelgrass (*Zostera marina*) decline in temperate estuarine ecosystems. Ecology 82:1007–1022.

Hayes, S; Newland, L; Morgan, K; et al. (1990) Septic tank and agricultural non-point source pollution within a rural watershed. Toxicological and Environmental Chemistry 26:137-155.

Heck, KL; Able, KW; Fahey, MP; et al. (1989) Fishes and decapod crustaceans of Cape Cod eelgrass meadows: species composition, seasonal abundance patterns, and comparisons with unvegetated substrates. Estuaries 12:59–65.

Heufelder, G; Rask, S. (1996) Alternative Septic System Update #9. Barnstable County Department of Health and the Environment, Barnstable, MA.

Heufelder, G; Rask, S. (1997) Alternative Septic System Update #10. Barnstable County Department of Health and the Environment, Barnstable, MA.

Hinga, KR; Keller, AA; Oviatt, CA. (1991) Atmospheric deposition and nitrogen inputs to coastal waters. Ambio 20:256-260.

Howarth, R. (1988) Nutrient limitation of net primary production in marine ecosystems. Annual Review of Ecology and Systematics 19:89–110.

Hunter, CJ. (1991) Better trout habitat: A guide to stream restoration and management. Washington, DC: Island Press.

Hurley, ST. (1990) Fisheries sampling report: Quashnet River, Falmouth, Mashpee. Unpublished report, Massachusetts Division of Fisheries and Wildlife, Buzzards Bay, MA.

Interdisciplinary Environmental Planning (IEP). (1986) Hydrological analisis and impact assessment– proposed Yarmouth public golf course. IEP, Inc., Sandwich, Mass.

Johnston, CA. (1991) Sediment and nutrient retention by freshwater wetlands: Effects of surface water quality. Critical Reviews in Environmental Control 21:491-565.

Johnston, CA; Detenbeck, NE; Niemi, GJ. (1990) The cumulative effect of wetlands on stream water quality and quantity: a landscape approach. Biogeochemistry 10:105-141.

Jordan, TE; Weller, DE. (1996) Human contributions to total nitrogen flux. BioScience 46:655–664.

Jordan TE; Correll, DL; Weller, DE. (1997) Relating nutrient discharges from watersheds to land use and streamflow variability. Water Resources Research 33:2579–2590.

Kaplan, OB. (1991) Septic systems handbook. Boca Raton, FL: Lewis Publishers.

Keeney, D. (1986) Sources of nitrate to ground water. Critical Reviews in Environmental Control 16:257–304.

Ketchum, BH. (1951) The exchanges of fresh and salt water in tidal estuaries. Journal of Marine Research 10:18–38.

Kohl, DH; Shearer, GB; Commoner, B. (1973) Variation of d^{15}N in corn and soil following application of fertilizer nitrogen. Soil Science Society of America Proceedings 37:888–892.

Koppelman, LE. (1978) The Long Island comprehensive waste treatment management plan, vol. II: summary documentation. Nassau-Suffolk Regional Planning Board, Haupauge, NY.

Korom, SF. (1992) Natural denitrification in the saturated zone: a review. Water Resources Research 28:1657-1668.

Kreitler, CW. (1975) Nitrogen-isotope ratio study of soils and groundwater nitrate from alluvial fan aquifers in Texas. Journal of Hydrology 42:147–170.

Kreitler, CW; Jones, DC. (1975) Natural soil nitrate: the cause of the nitrate contamination of groundwater in Runnels County, TX. Ground Water 13:53–61.

Krom, MD; Berner, RA. (1980) Adsorption of phosphate in anoxic marine sediments. Limnology and Oceanography 25:327–337.

Lajtha, K; Seely, B; Valiela, I. (1995) Retention and leaching losses of atmospherically derived nitrogen in the aggrading coastal watershed of Waquoit Bay, MA. Biogeochemistry 28:33–54.

LaMontagne, MG; Valiela, I. (1995) Denitrification measured by a direct N_2 flux method in sediments of Waquoit Bay, MA. Biogeochemistry 31:63–83.

Landis, WG; Wiegers, JA. (1997) Design considerations and a suggested approach for regional and comparative ecological risk assessment. Human and Ecological Risk Assessment 3:287–297.

Lawrence, DL. (2000) Estuarine calanoid copepod abundance in Waquoit Bay MA: Effects of season, salinity, and land-derived nitrogen loads. Master's Thesis. Boston University.

LeBlanc, DR; Guswa, JH; Frimpter, MH; et al. (1986) Groundwater resources of Cape Cod, Massachusetts. USGS Hydrologic Investigations Atlas, Volume 692.

Legra, JC; Safran, RE; Valiela, I. (1998) Lead concentration as an indicator of contamination history in estuarine sediments. Biological Bulletin 195:243–244.

Lindhult, MS; Godfrey, PJ. (1988) Using Geographic Information Systems to predict the trophic states of lakes. In: Proceedings from the Symposium on Coastal Water Resources Wilmington, NC. American Water Resources Association, Bethesda, MD.

Link, J; Brodziak, J. (eds). (2002) Report on the Status of the NE US Continental Shelf Ecosystem. NEFSC Ecosystem Status Working Group. Document # 02-11, Northeast Fisheries Science Center, Woods Hole, MA.

Loehr, RC. (1974) Characteristics and comparative magnitude of nonpoint sources. J Water Pollut Control Feder 46:1849–1872.

Lowrance, R; Todd, RL; Asmussen, LE. (1984) Riparian forests as nutrient filters in agricultural watersheds. BioScience 34:374-377.

Macko, SA; Ostrom, NE. (1994) Pollution studies using stable isotopes. Pp. 45–62. In: Stable Isotopes in Ecology. Lajtha, K; Michener, RS (eds). London: Blackwell Scientific.

Mariotti, A; LèTolle, R. (1977) Application of nitrogen isotopes to hydrology and hydrogeology: Analysis of the peculiar case of the Melarchez Basin (Siene-et-Marne, France). Journal of Hydrology 33:157–172.

McClelland, JW; Valiela, I. (1998) Linking nitrogen in estuarine producers to land-derived sources. Limnology and Oceanography 43:577–585.

McDonald, MG; Harbaugh, AW. (1988) A Modular Three-Dimensional Finite Difference Groundwater Flow Model. US Geological Survey, Techniques of Water Resources Investigations, Book 6, U.S. Government Printing Office, Washington, DC.

McLarney, WO. (1988) Who says they don't make trout streams anymore? Trout, Autumn: 22–31.

Mitchell, S. (1999) Homeowner's Guide to Title 5. Association for the Preservation of Cape Cod, Orleans, MA.

Molot, LA; Dillon, PJ. (1993) Nitrogen mass balances and denitrification rates in central Ontario lakes. Biogeochemistry 20:195-212.

Nelson, ME; Horsley, SW; Cambareri, TC; et al. (1988) Predicting nitrogen concentrations in groundwater: an analytical model. In: Proceedings from the Conference on Eastern Regional Groundwater Issues. September 27-29, 1988, Stamford, CT. National Well Water Association. Dublin, OH.

Nixon, SW. (1986) Nutrients and the productivity of estuarine and coastal marine ecosystems. Journal of the Limnological Society of South Africa 12:43–71.

Nixon, SW. (1992) Quantifying the relationship between nitrogen input and the productivity of marine ecosystems. Proceedings of the Advanced Marine Technology Conference, Tokyo, Japan. 5:57–83.

Nixon, SW. (1995) Coastal marine eutrophication: A definition, social causes, and future concerns. Ophelia 41:199–219.

Norton, SB; Cormier, S; Smith, M; et al. (2000) Can biological assessments discriminate among types of stress? A case study from the eastern corn belts plains ecoregion. Environmental Toxicology and Chemistry 19(4):1113–1119.

NRC (National Research Council). (2000) Clean coastal waters. Committee on the causes and management of coastal eutrophication, ocean studies board, and water science and technology board. Washington, DC: National Academy Press.

Oldale, RN. (1992) Cape Cod and the Islands: The Geologic Story. East Orleans, MA: Parnassus Imprints.

Ollinger, SV; Aber, JD; Lovett, GM; et al. (1993) A spatial model of atmospheric deposition for the northeastern U.S. Ecological Applications 3:459-472.

Pace, ML; Findlay, SEF; Lints, D. (1992) Zooplankton in advective environments: The Hudson River community and a comparative analysis. Canadian Journal of Fisheries and Aquatic Sciences 49:1060–1069.

Pack, DH. (1980) Precipitation chemistry patterns: a two-network data set. Science 208:1143-1145.

Peckol, P; DeMeo–Anderson, B; Rivers, J; et al. (1994) Growth, nutrient uptake capabilities, and tissue constituents of the macroalgae, *Cladophora vagabunda*, and *Gracilaria tikvahiae,* related to site specific nutrient loading rates. Marine Biology 121:175–185.

Pennak, RW. (1989) Freshwater Invertebrates of the United States, 3rd ed.: Protozoa to mollusca. New York: Wiley.

Petrovic, AM. (1990) The fate of nitrogenous fertilizer applied to turfgrass. Journal of Environmental Quality 19:1-14.

Pickett, STA. (1989) Space-for-time substitutions as an alternative to long-term studies. In: Long-term Studies in Ecology: Approaches and Alternatives. Likens, GE (ed.), New York: Springer-Verlag; pp. 110–136.

Piluk, RJ; Peters, EC. (1995) Small recirculating sand filters for individual homes. Anne Arundel County Health Department. Annapolis, MD. Available at: http://plymouth.ces/state.nc.us/septic/95piluk.html.

Prairie, YT. (1996) Evaluating the predictive power of regression models. Canadian Journal of Fisheries and Aquatic Sciences 53:490–492.

Rask, S; Heufelder, G. (1998) Alternative Septic System Update #5. Barnstable County Department of Health and the Environment. Barnstable, MA.

Reneau, RB. (1979) Changes in concentrations of selected chemical pollutants in wet, tile-drained soil systems as influenced by disposal of septic tank effluents. Journal of Environmental Quality 8:189-196.

Renn, CE. (1935) A mycetozoan parasite of *Zostera marina.* Nature 135:544–545.

Riisgard, HU; Hansen, S. (1990) Biomagnification of mercury in a marine grazing food chain: algal cells (*Phaeodactylum tricornutum*) musselys (*Mytilus edulis*) and flounder (*Platichtys flesus*) studied by means of a stepwise reduction CVAA method. Marine Ecology Progress Series 62:259–270.

Running, SW; Nemani, RR; Hungerford, RD. (1988) Extrapolation of synoptic meteorological data in mountainous terrain and its use for simulating forest evapotranspiration and photosynthesis. Canadian Journal of Forestry Research 17:472-483.

Serveiss, VB. (2002) Applying ecological risk principles to watershed assessment and management. Environmental Management 29(2):145–154.

Serveiss, VB; Norton, DJ; Norton, SB. (2000) Watershed ecological risk assessment: the watershed academy. U.S. Environmental Protection Agency on-line training module at: http://www.epa.gov/owow/watershed/wacademy/acad2000/ecorisk

Serveiss, VB; Bowen, JL; Dow, D; et al. Waquoit Bay watershed ecological risk assessment. Environmental Management: submitted.

Sham, C-H; Brawley, J; Moritz, MA. (1995) Quantifying nitrogen loading from residential septic sources to a shallow coastal embayment. International Journal of Geographic Information Systems 9:463–473.

Shannon, JD; Sisterson. (1992) Estimation of S and NOx-N deposition budgets for the United States and Canada. Water, Air, and Soil Pollution 63:211-235.

Shriver, AC; Carmichael, RH; Valiela, I. (2002) Growth, condition, reproductive potential, and mortality of bay scallops, Argopecten irradians, in response to eutrophi-driven changes in food resources. Journal of Experimental Marine Biology and Ecology 279:21-40.

Short, FT; Burdock, DM. (1996) Quantifying eelgrass habitat loss in relation to housing development and nitrogen loading in Waquoit Bay, Massachusetts. Estuaries 19:730–739.

Short, FT; Mathissen, AC; Nelson, JJ. (1986) Recurrence of the eelgrass wasting disease at the border of New Hampshire and Maine, USA. Marine Ecology Progress Series 29:89–92.

Short, FT; Ibelings, BW; DenHartog, C. (1988) Comparison of a current outbreak of eelgrass disease to the wasting disease in the 1930s. Aquatic Botany 30:295–304.

Shumway, SE. (1991) Scallops: Biology, Ecology and Aquaculture. Amsterdam: Elsevier Press.

Smith, I; Fonseca, MS; Rivera, JA; et al. (1989) Habitat value of natural versus recently transplanted eelgrass for the bay scallop. Fishery Bulletin 87:189–196.

Sogard, SM; Able, KW. (1991) A comparison of eelgrass, sea lettuce, macroalgae, and marsh creeks as habitats for epibenthic fishes and copepods. Estuarine, Coastal and Shelf Science 33:501–519.

Stafford, CP; Haines, TA. (2001) Mercury contamination and growth rate in two piscivore populations. Environmental Toxicology and Chemistry 20:2099–2101.

Stanley, DW. (1988) Historical trends in nutrient loading to the Neuse River estuary, NC. In: Proceedings from the Symposium on Coastal Water Resources, Wilmington, NC. American Water Resources Association, Bethesda, MD.

Starr, RD; Sawhney, BL. (1980) Movement of nitrogen and carbon from a septic system drainfield. Water, Air and Soil Pollution 13:113-123.

Starr, RC; Gillham, RW. (1993) Denitrification and organic carbon availability in two aquifers. Ground Water 31:934-947.

Stensland, GJ; Whelpdale, DM; Oehlert, G. (1986) Precipitation Chemistry. In: Acid Deposition: Long-Term Trends. Committee on Monitoring and Assessment of Trends in Acid Deposition, National Research Council. Washington, DC: National Academy Press; pp. 128-199.

Stokes, JC. (2000) General information on wastewater and alternative onsite sewage systems. Ad Hoc Committee on Alternative Septic Systems. New Jersey Pinelands Commission. New Lisbon, New Jersey. Available at: http://www.state.nj.us/pinelands/428rept.htm.

Stumm W; Morgan, JJ. (1981) Aquatic Chemistry, 2nd ed. New York: Wiley.

Suter, GW. (1989) Ecological endoints. In: Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference Document. Warren-Hicks, W; Parkhurst, BR; Baker Jr, SS. (eds.), EPA 600/3-89/013. Corvallis Environmental Research Laboratory, OR.

Suter, GW. (1993) Ecological Risk Assessment. Boca Raton, FL: Lewis Publishers.

Thayer, GW; Stuart, H. (1974) The bay scallop makes its bed of seagrass. Marine Fisheries Review 36(7):27-30.

Thayer, GW; Kenworthy, WJ; Fonseca, MS. (1984) The ecology of eelgrass on the Atlantic coast: A community profile. FWS/OBS-84-2. US Fish and Wildlife Service, Washington, DC.

Thornthwaite, CW; Mather, JR. (1957) Instructions and tables for computing potential evapotranspiration and the water balance. In: Publications in Climatology. Drexel Institute of Technology, Philidelphia, PA; 10:185-311.

Tober, JD; Griffin, MPA; Valiela, I. (2000) Growth and abundance of Fundulus heteroclitus and Menidia menidia in estuaries of Waquoit Bay, MA exposed to different rates of nitrogen loading. Aquatic Ecology 34:299–306.

Tomasky, G; Barak, J; Valiela, I; et al. (1999) Nutrient limitation of phytoplankton growth in Waquoit Bay, Massachusetts, USA: a nutrient enrichment study. Aquatic Ecology 33:137–155.

Udy, JW; Dennison, WC. (1997) Growth and physiological responses of three seagrass species to elevated sediment nutrients in Moreton Bay, Australia. Journal of Experimental Marine Biology and Ecology 217:253–277.

Udy, JW; Dennison, WC. (1998) The use of the seagrass *Zostera capricorni* to identify anthropogenic nutrient sources in Moreton Bay. In: Moreton Bay and Catchment. Tibbetts, IR; Hall, NJ; Dennison, WC (eds.). The University of Queensland, Brisbane, Australia.

U.S. Census Bureau. (2000) Population Estimates Program, Population Division, Washington, DC. Data obtained from http://www.census.gov/population/estimates/.

U.S. EPA (Environmental Protection Agency). (1980) Design manual, onsite waste treatment and disposal systems. EPA/625/1-80/012. Washington, DC.

U.S. EPA. (1982) National air pollution emission estimates 1940-1980. EPA/450/4-82/001. Washington, DC.

U.S. EPA. (1992) Framework for ecological risk assessment. EPA/630/R-92/001. Office of Water, Washington, DC.

U.S. EPA. (1996) Watershed approach framework. EPA/840/S-96/001. Office of Water. Washington, DC.

U.S. EPA. (1998) Guidelines for ecological risk assessment. EPA/630/R-95/002Fa. Washington, DC.

Valiela, I. (1995) Marine ecological processes. 2nd ed. New York: Springer-Verlag.

Valiela, I; Bowen, JL. (2002) Nitrogen sources to watersheds and estuaries: role of land cover mosaics and losses within watersheds. Environmental Pollution 118:239-248.

Valiela, I; Cole, ML. (2002) Comparing evidence that salt marshes and mangroves protect seagrass meadows from land-derived nitrogen loads. Ecosystems 5:92-102.

Valiela, I; Costa, JE. (1988) Eutrophication of Buttermilk Bay, a Cape Cod coastal embayment: concentrations of nutrients and watershed nutrient budgets. Environmental Management 12: 539–553.

Valiela I; Teal, JM; Volkmann, S; et al. (1978) Nutrient and particulate fluxes in a salt marsh ecosystem: tidal exchanges and inputs by precipitation and groundwater. Limnology and Oceanography 23:708–812.

Valiela, I; Foreman, K; LaMontagne, M; et al. (1992) Couplings of watersheds and coastal waters: sources and consequences of nutrient enrichment in Waquoit Bay, Massachusetts. Estuaries 15:443–457.

Valiela, I; Peckol, P; D'Avanzo, C: et al. (1996) Hurricane Bob on Cape Cod. American Scientist 84:154–165.

Valiela, I; McClelland, J; Hauxwell, J; et al. (1997a) Macroalgal blooms in shallow estuaries: controls and ecophysiological and ecosystem consequences. Limnol Oceanography 42:1105–1118.

Valiela I; Collins, G; Kremer, J; et al. (1997b) Nitrogen loading from coastal waters to receiving estuaries: new method and application. Ecological Applications 7:358–380. Model available at <http://www.wa

Valiela, I; Peckol, P; D'Avanzo, C; et al. (1998) Ecological effects of major storms on coastal watersheds and coastal watersheds: Hurricane Bob on Cape Cod. Journal of Coastal Research 14:218–238.

Valiela, I; Geist, M; McClelland, J; et al. (2000a) Nitrogen loading from watersheds to estuaries: verification of the Waquoit Bay nitrogen loading model. Biogeochemistry 49:277–293.

Valiela, I; Tomasky, G; Hauxwell, J; et al. (2000b) Operationalizing sustainability: management and risk assessment of land-derived nitrogen loads to estuaries. Ecological Applications 10:1006–1023.

Valiela, I; Bowen, JL; Cole, ML; et al. (2001) Following up on a Margalevian concept: interactions and exchanges among adjacent parcels of coastal landscapes. Scientia Marina 65:271–231.

Valiela, I; Bowen, JL; Kroeger, KD. (2002) Assessment of different models for estimation of land-derived nitrogen loads to shallow estuaries. Applied Geochemistry 17: 935–953.

Valiela, I; Bowen, JL; Stieve, EL; et al. ELM, an estuarine nitrogen loading model: formulation, verification of predicted concentrations, and links to producers. Ecological Modelling: in press.

Vitousek, PM; Howarth, RW. (1991) Nitrogen limitation on land and in the sea: how can it occur? Biogeochemistry 13:87–115.

Vollenweider, RA. (1987) Scientific concepts and methodologies pertinent to lake research and lake restoration. Schweizerische Zeitschrift F¸r Hydrologie 49:129–147.

Walker, WG; Bouma, GJ; Kenney, DR; et al. (1973) Nitrogen transformations during subsurface disposal of septic tank effluent in sands: I. Soil transformations. Journal of Environmental Quality 2:475–480.

Weinstein, M; Balletto, J. (1999) Does the common reed *Phragmities australis* affect essential fish habitat? Estuaries 22:793–802.

Weiskel, PK; Howes, BL. (1991) Quantifying dissolved nitrogen flux through a coastal watershed. Water Resources Research 27: 2929–2939.

Weiss, E; Carmichael, RH; Valiela, I. (2002) The effects of N loading on the growth rates of quahogs and softshell clams through changes in food supply. Aquaculture 211:275–289.

Wiegers, JK; Feder, HM; Mortensen, LS; et al. (1998) A regional multiple stressor rank-based ecological risk assessment for the fjord of Port Valdez, AK. Human and Ecological Risk Assessment 4(5):1125–1173.

Williams, SL; Davis, CA. (1996) Population genetic analyses of transplanted eelgrass beds reveal reduced genetic diversity in Southern California. Restoration Ecology 4:163–180.

Wuertz, S; Miller, ME; Doolittle, MM; et al. (1991) Butyltins in estuarine sediments two years after tributyltin was restricted. Chemosphere 22:1113–1120.

United States
Environmental Protection Agency/ORD
National Center for
Environmental Assessment
Washington, DC 20460

Official Business
Penalty for Private Use
$300

EPA/600/R-02/079
October 2002

Please make all necessary changes on the below label, detach or copy, and return to the address in the upper left-hand corner.

If you do not wish to receive these reports CHECK HERE ☐; detach, or copy this cover, and return to the address in the upper left-hand corner.

PRESORTED STANDARD
POSTAGE & FEES PAID
EPA
PERMIT No. G-35