Linking Receiving Water Impacts to Sources and
to Water Quality Management Decisions:
Using Nutrients as an Initial Case Study
LINKING RECEIVING WATER IMPACTS TO SOURCES AND TO WATER QUALITY MANAGEMENT DECISIONS: USING NUTRIENTS AS AN INITIAL CASE STUDY

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Hazen and Sawyer, P.C.

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ABSTRACT AND BENEFITS

Abstract:

The Water Environment Research Foundation (WERF) has undertaken a research challenge entitled “Linking Receiving Water Impacts to Sources and to Water Quality Management Decisions”. The research described in this report provides an initial case study towards this larger challenge by focusing on nutrients as a source constituent, primarily nitrogen. The purpose was to provide a state-of-the-knowledge approach to establishing the linkages between sources of nitrogen, the predicted and measured adverse impacts of nitrogen on receiving waters, and the costs and benefits of controls available to address these adverse impacts.

Based on an extensive review of the literature, a framework for an approach to establishing these linkages was developed. The framework included establishing water quality impacts, linking these impacts to nutrients, quantifying major nitrogen sources, evaluating the costs and benefits of available nitrogen controls, estimating receiving water responses to controls, and finally, implementing a nitrogen control strategy and assessing water quality for potential improvements.

Key findings of this effort included: 1) nutrient water quality impacts are typically waterbody specific, and thus waterbody specific assessments are necessary to develop appropriate numeric nutrient criteria; 2) determination of the causative agent for eutrophication impairment is not straightforward, but needs to be determined prior to developing management decisions; 3) nonpoint sources of nitrogen appear to be the largest sources of reactive nitrogen globally, regionally, and locally in many watersheds. Nitrogen controls on these sources would be beneficial in most watersheds; 4) all benefits and costs of available nutrient controls should be evaluated for all stakeholders prior to implementation. The true life-cycle costs of controls can identify those that provide the most cost-effective reduction in nutrient loading; 5) estimating receiving water responses to nutrient controls is difficult but necessary to evaluate options for water quality improvements. This requires adequate data on watershed and waterbody characteristics.

The report identified numerous knowledge gaps limiting the application of the framework for many waterbodies. A targeted research program is needed to fill these knowledge gaps so that nutrient related waterbody impairments can be more easily identified and cost-effectively remediated.

Benefits:

♦ Reviews and provides references for over 450 publications related to nutrient sources, receiving water impacts, and controls.
♦ Provides a framework that can be utilized by subscribers when addressing nitrogen related loading requirements.
♦ Presents evidence supporting the need for quantifiable nutrient controls on nonpoint sources in many watersheds.
♦ Identifies knowledge gaps that need to be addressed to adequately determine the most cost-effective nutrient controls for a given waterbody.

Keywords: Nitrogen loading, nitrogen sources, water quality, nutrient criteria, eutrophication, benefit-cost analysis.
TABLE OF CONTENTS

Acknowledgments .......................................................................................................................... iii
Abstract and Benefits ........................................................................................................................ v
List of Tables ........................................................................................................................................ ix
List of Figures ...................................................................................................................................... x
List of Acronyms .................................................................................................................................. xi
Executive Summary ......................................................................................................................... ES-1

1.0 Introduction................................................................................................................................ 1-1
  1.1 Purpose ...................................................................................................................................... 1-1
  1.2 Scope ....................................................................................................................................... 1-1
  1.3 Florida’s Approach to Achieving Water Quality Standards ................................................. 1-2
  1.4 Project Framework and Approach to Achieving Water Quality Criteria ........................... 1-5

2.0 Water Quality Impacts: Is the Waterbody Impaired? .......................................................... 2-1
  2.1 State-of-the-Knowledge: Linkages between Nutrients and Water Quality Impacts .......... 2-1
  2.2 Symptoms of Water Quality Impairment ............................................................................... 2-4
  2.3 Identification of Impaired Waters: The Florida Impaired Waters Rule ......................... 2-7
  2.4 Identification of Knowledge Gaps ...................................................................................... 2-13

3.0 Nutrient Linkage: Is Nitrogen or Phosphorus Causative? ....................................................... 3-1
  3.1 Methods to Determine Limitation ....................................................................................... 3-2
    3.1.1 Evaluation of Ambient Water Quality Data ......................................................... 3-2
    3.1.2 Experimental Methods ....................................................................................... 3-3
  3.2 Confounding Factors ............................................................................................................ 3-4
  3.3 Nutrient Limitation: Florida Applications ....................................................................... 3-4
  3.4 Identification of Knowledge Gaps ..................................................................................... 3-5

4.0 Nutrient Sources: Can the Major Nitrogen Loads Be Quantified? ........................................ 4-1
  4.1 What Are the Sources of Nitrogen Inputs to the Watershed? ............................................. 4-1
  4.2 Quantify Nitrogen Inputs to Watershed and Linkages to Receiving Waterbody ............... 4-4
    4.2.1 Atmospheric Deposition Sources and Loads ...................................................... 4-5
    4.2.2 Nonpoint Sources and Loads ........................................................................ 4-11
    4.2.3 Wastewater Sources and Loads .................................................................... 4-19
  4.3 Summarize and Rank Sources by Loading to Waterbody ................................................. 4-26
  4.4 Determine Assimilative Capacity of a Receiving Waterbody ........................................... 4-27
  4.5 Determine Need for Nutrient Controls ............................................................................ 4-30
  4.6 Identification of Knowledge Gaps ..................................................................................... 4-31

5.0 Nutrient Controls: ....................................................................................................................... 5-1
  Are There Available Controls to Address Major Source Types? ............................................. 5-1
    5.1 Potential Controls for the Primary Sources ............................................................... 5-1
      5.1.1 Wastewater Treatment Plants (WWTPs) .......................................................... 5-2
### 6.0 Water Quality Management Decisions:

**Are All Benefits and Costs of Nutrient Controls Considered?**

- **6.1** Evaluating the Benefits and Costs of Nutrient Management to Receiving Waters .... 6-1
- **6.2** Previous Uses of Evaluation Models ............................. 6-3
- **6.3** Evaluation Criteria for Initial Screening of Nitrogen Management Technologies .............................................................. 6-5
- **6.4** Criteria that Consider All Benefits and Costs of Each Nitrogen Management Alternative.......................................................... 6-6
  - **6.4.1** Nitrogen Reduction – Average Annual Change .......................... 6-8
  - **6.4.2** Nitrogen Reduction – Nitrogen Concentration .......................... 6-9
  - **6.4.3** Cost-Effectiveness .................................................................. 6-9
  - **6.4.4** Present Value Change in Regional or Statewide Income .......... 6-11
  - **6.4.5** Affordability of Wastewater and Stormwater Bills .................. 6-12
  - **6.4.6** Success in Achieving Other Water Management Objectives .... 6-12
  - **6.4.7** Water Supply Benefits ................................................................ 6-13
  - **6.4.8** Acres of Increased or Improved Wildlife Habitat ....................... 6-13
  - **6.4.9** Change in Recreation Opportunities ......................................... 6-14
  - **6.4.10** Change in Property Values ...................................................... 6-14
  - **6.4.11** Effect on Greenhouse Gas and NOx Emissions .......................... 6-14
  - **6.4.12** Engineering / Technological Track Record .............................. 6-16
  - **6.4.13** Permitting Uncertainty .............................................................. 6-17
- **6.5** Scoring and Weighting the Alternatives and Sensitivity/Uncertainty Analysis ..... 6-18
- **6.6** Summary of Benefit-Cost Evaluation Model ................................. 6-18
- **6.7** Water Quality Trading ..................................................................... 6-19
- **6.8** Develop the Nutrient Control Strategy .............................................. 6-23
- **6.9** Data Gaps and Research Needs ............................................................ 6-24

### 7.0 Water Quality Improvements: Can Receiving Water Responses Be Estimated? .... 7-1

- **7.1** Empirical Models .............................................................. 7-1
  - **7.1.1** Regression Models .............................................................. 7-1
  - **7.1.2** Artificial Neural Networks .................................................. 7-4
- **7.2** Mechanistic Models .............................................................. 7-5
- **7.3** Water Quality Response Modeling in Florida ..................................... 7-9
  - **7.3.1** Caloosahatchee River .............................................................. 7-10
  - **7.3.2** Tampa Bay ........................................................................... 7-11
  - **7.3.3** Lower St. Johns River .............................................................. 7-12
- **7.4** Knowledge Gaps and Uncertainties .................................................. 7-12
8.0 Control Strategy Implementation ................................................................. 8-1

9.0 Water Quality Monitoring and Assessment .................................................. 9-1
  9.1 Monitoring Potential Water Quality Improvements Due to Nutrient Control Strategies ................................................................. 9-1
  9.2 Evaluate Effectiveness of Plan Components ............................................... 9-3

10.0 Conclusions and Recommendations ........................................................... 10-1
  10.1 Summary .................................................................................................. 10-1
  10.2 Conclusions ............................................................................................. 10-1
    10.2.1 Water Quality Impacts ...................................................................... 10-1
    10.2.2 Nitrogen Sources .......................................................................... 10-3
    10.2.3 Nitrogen Controls ......................................................................... 10-4
    10.2.4 Water Quality Management Decisions ............................................. 10-5
    10.2.5 Water Quality Improvements – Estimating Receiving Water Response to Controls ......................................................... 10-6
  10.3 Recommendations for Research Needs ...................................................... 10-7

References........................................................................................................... R-1
<table>
<thead>
<tr>
<th>Table</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>2-1</td>
<td>Number of Nutrient Impaired Waterbodies in Florida Based on the Most Recent Verified List</td>
</tr>
<tr>
<td>2-2</td>
<td>Number of DO Impaired Waterbodies in Florida Based on the Most Recent Verified List</td>
</tr>
<tr>
<td>4-1</td>
<td>2005 NOx Emissions in Florida</td>
</tr>
<tr>
<td>4-2</td>
<td>Relative Contributions by Source to the Nitrogen Loads to Selected Florida Receiving Waterbodies (Percent)</td>
</tr>
<tr>
<td>4-3</td>
<td>Agricultural Facilities in Florida Meeting U.S. EPA Animal Feeding Operations Criteria</td>
</tr>
<tr>
<td>4-4</td>
<td>Livestock and Poultry Manure Generated in Florida, 2007</td>
</tr>
<tr>
<td>5-1</td>
<td>Comparison of Nitrogen Levels Found in Wastewater Effluent from Advanced Treatment and Typical In-Stream Nutrient Criteria</td>
</tr>
<tr>
<td>5-2</td>
<td>Viable Wastewater Treatment Technologies for Effluent Limits of 3 mg/L TN</td>
</tr>
<tr>
<td>5-3</td>
<td>Estimated Pollutant Removal Efficiencies for Common Stormwater Treatment Facilities Used in Florida</td>
</tr>
<tr>
<td>Figure</td>
<td>Title</td>
</tr>
<tr>
<td>--------</td>
<td>----------------------------------------------------------------------</td>
</tr>
<tr>
<td>ES-1</td>
<td>Conceptual Approach to Developing Water Quality Improvement Strategies</td>
</tr>
<tr>
<td>ES-2</td>
<td>Project Framework and Approach</td>
</tr>
<tr>
<td>ES-3</td>
<td>The Nitrogen Cycle</td>
</tr>
<tr>
<td>ES-4</td>
<td>Estimated World-Wide Production of Reactive Nitrogen</td>
</tr>
<tr>
<td>ES-5</td>
<td>Incremental O&amp;M Cost Increase per Additional Pound of N Removed</td>
</tr>
<tr>
<td>ES-6</td>
<td>Incremental GHG Emissions Increase by Treatment Level for N</td>
</tr>
<tr>
<td>1-1</td>
<td>Conceptual Approach to Developing Water Quality Improvement Strategies</td>
</tr>
<tr>
<td>1-2</td>
<td>Project Framework and Approach</td>
</tr>
<tr>
<td>2-1</td>
<td>Bottom DO in the Lower Hillsborough River as a Function of Flow Past the Dam</td>
</tr>
<tr>
<td>2-2</td>
<td>Chlorophyll in the Lower Hillsborough River as a Function of Flow Past the Dam</td>
</tr>
<tr>
<td>4-1</td>
<td>Simple Depiction of the Processes and Transformations in the Nitrogen Cycle</td>
</tr>
<tr>
<td>4-2</td>
<td>Estimated World-Wide Production of Reactive Nitrogen</td>
</tr>
<tr>
<td>4-3</td>
<td>Nitrogen Inputs to a Waterbody and Its Watershed</td>
</tr>
<tr>
<td>4-4</td>
<td>Major Atmospheric Deposition Linkages Within the Watershed and to the Receiving Waterbody</td>
</tr>
<tr>
<td>4-5</td>
<td>Locations of Waterbodies for Which TN Loads are Reported in Table 4-2</td>
</tr>
<tr>
<td>4-6</td>
<td>Major Nonpoint Source Linkages within in the Watershed and to the Waterbody</td>
</tr>
<tr>
<td>4-7</td>
<td>Major Wastewater Pathways in the Watershed</td>
</tr>
<tr>
<td>4-8</td>
<td>Conceptual Basis for Tampa Bay Nitrogen Targets</td>
</tr>
<tr>
<td>4-9</td>
<td>Relationship Between Chlorophyll (a) Concentrations and the Three-Month Cumulative TN Load For Hillsborough Bay, FL</td>
</tr>
<tr>
<td>5-1</td>
<td>Estimated Total Nitrogen Load Baywide Based on Runoff and Compliance Assumptions</td>
</tr>
<tr>
<td>5-2</td>
<td>Removal Efficiency of Total Nitrogen in Wet Detention Ponds as a Function of Residence Time</td>
</tr>
<tr>
<td>6-1</td>
<td>Incremental O&amp;M Cost for Increasing Wastewater Treatment Levels</td>
</tr>
<tr>
<td>6-2</td>
<td>Incremental Increase in GHG Emissions with Increasing Wastewater Treatment Level</td>
</tr>
<tr>
<td>7-1</td>
<td>Linear Regression of Monthly Average TN and Chlorophyll (a) Concentrations in Upper Lemon Bay</td>
</tr>
<tr>
<td>7-2</td>
<td>Goodness of Fit for the Regression of Chlorophyll (a) Concentrations on TN Loads</td>
</tr>
<tr>
<td>7-3</td>
<td>Artificial Neural Network Construct Used to Predict DO</td>
</tr>
<tr>
<td>7-4</td>
<td>Schematic Illustration of Eutrophication Processes Modeled in WASP</td>
</tr>
<tr>
<td>7-5</td>
<td>Sediment Transport Mechanisms</td>
</tr>
<tr>
<td>7-6</td>
<td>BATHTUB Model Construct</td>
</tr>
<tr>
<td>7-7</td>
<td>WASP Model External Linkages</td>
</tr>
<tr>
<td>7-8</td>
<td>EFDC Model Segmentation of the Tidal Caloosahatchee River</td>
</tr>
<tr>
<td>Acronym</td>
<td>Description</td>
</tr>
<tr>
<td>---------</td>
<td>-------------</td>
</tr>
<tr>
<td>ACF</td>
<td>Apalachicola-Chattahoochee-Flint</td>
</tr>
<tr>
<td>AGNPS</td>
<td>Agricultural Nonpoint Source Pollution Model</td>
</tr>
<tr>
<td>ANNs</td>
<td>Artificial Neural Networks</td>
</tr>
<tr>
<td>ARS</td>
<td>Agricultural Research Service</td>
</tr>
<tr>
<td>AWT</td>
<td>Advanced Wastewater Treatment</td>
</tr>
<tr>
<td>BCA</td>
<td>Benefit Cost Analysis</td>
</tr>
<tr>
<td>BMPs</td>
<td>Best Management Practices</td>
</tr>
<tr>
<td>BNF</td>
<td>Biological Nitrogen Fixation</td>
</tr>
<tr>
<td>BOD</td>
<td>Biochemical Oxygen Demand</td>
</tr>
<tr>
<td>CAA</td>
<td>Clean Air Act</td>
</tr>
<tr>
<td>CAFO</td>
<td>Concentrated Animal Feeding Operation</td>
</tr>
<tr>
<td>CAIR</td>
<td>Clean Air Interstate Rule</td>
</tr>
<tr>
<td>CBOD5</td>
<td>Carbonaceous Biochemical Oxygen Demand – 5 day</td>
</tr>
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<td>Criterium Decision Plus</td>
</tr>
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<td>Comprehensive Everglades Restoration Plan</td>
</tr>
<tr>
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<td>Methane</td>
</tr>
<tr>
<td>CNMP</td>
<td>Comprehensive Nutrient Management Plans</td>
</tr>
<tr>
<td>CO2</td>
<td>Carbon Dioxide</td>
</tr>
<tr>
<td>CREAMS</td>
<td>Chemicals, Runoff, and Erosion from Agricultural Management Systems</td>
</tr>
<tr>
<td>CSO</td>
<td>Combined Sewer Overflow</td>
</tr>
<tr>
<td>CV</td>
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</tr>
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<td>Florida Green Industries Manual</td>
</tr>
<tr>
<td>FWRA</td>
<td>Florida Watershed Restoration Act</td>
</tr>
<tr>
<td>GHG</td>
<td>Greenhouse Gas</td>
</tr>
<tr>
<td>GLEAMS</td>
<td>Groundwater Loading Effects on Agricultural Management Systems</td>
</tr>
<tr>
<td>HABs</td>
<td>Harmful Algal Blooms</td>
</tr>
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<td>Hydrologic Soil Group</td>
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<td>MEP</td>
<td>Maximum Extent Practicable</td>
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<tr>
<td>MGD</td>
<td>Million Gallons per Day</td>
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<td>mg/L</td>
<td>Milligrams per Liter</td>
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<td>Operation and Maintenance</td>
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<tr>
<td>OMB</td>
<td>U.S. Office of Management and Budget</td>
</tr>
<tr>
<td>OWTS</td>
<td>Onsite Wastewater Treatment System</td>
</tr>
<tr>
<td>PAN</td>
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<td>Photosynthetically Available Radiation</td>
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<td>Poly-Chlorinated Biphenyl</td>
</tr>
<tr>
<td>PtCo</td>
<td>Platinum Cobalt Units</td>
</tr>
<tr>
<td>PPCP</td>
<td>Pharmaceutical and Personal Care Products</td>
</tr>
<tr>
<td>PPM</td>
<td>Parts Per Million</td>
</tr>
<tr>
<td>RA</td>
<td>Reasonable Assurance</td>
</tr>
<tr>
<td>RO</td>
<td>Reverse Osmosis</td>
</tr>
<tr>
<td>SAV</td>
<td>Submerged Aquatic Vegetation</td>
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<tr>
<td>SCR</td>
<td>Selective Catalytic Reduction</td>
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<td>SERC</td>
<td>Statement of Estimated Regulatory Costs</td>
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<td>SFWMD</td>
<td>South Florida Water Management District</td>
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<td>SJRWMD</td>
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<td>SPAtially Referenced Regressions On Watershed attributes</td>
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<td>Sanitary Sewer Overflow</td>
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<td>SVCSD</td>
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<td>Sustainable Water Industry Asset Resource Decisions</td>
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<td>Soil and Water Assessment Toll</td>
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<td>Stormwater Management Plan</td>
</tr>
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<td>SWRRB</td>
<td>Simulator for Water Resources in Rural Basins</td>
</tr>
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<td>TBADS</td>
<td>Tampa Bay Atmospheric Deposition Study</td>
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<td>Tampa Bay Estuary Program</td>
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<td>TBL</td>
<td>Triple Bottom Line</td>
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<td>Tampa Bay Nitrogen Management Consortium</td>
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<td>TECO</td>
<td>Tampa Electric Company</td>
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<tr>
<td>TKN</td>
<td>Total Kjedahl Nitrogen</td>
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<tr>
<td>TMDL</td>
<td>Total Maximum Daily Load</td>
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<tr>
<td>Abbreviation</td>
<td>Full Form</td>
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<tr>
<td>TN</td>
<td>Total Nitrogen</td>
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<tr>
<td>TP</td>
<td>Total Phosphorus</td>
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<td>Total Suspended Solids</td>
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<td>Vegetated Treatment Area</td>
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<td>Water Cycle Management for New Developments</td>
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Executive Summary

ES.1 Introduction

The Water Environment Research Foundation (WERF) has undertaken a research challenge entitled “Linking Receiving Water Impacts to Sources and to Water Quality Management Decisions”. The research described in this report provides an initial case study towards this larger challenge and focuses on nutrients as a source constituent, primarily nitrogen. The purpose of this effort was to provide a state-of-the-knowledge approach to establishing the linkages between sources of nitrogen, the predicted and measured adverse impacts of nitrogen on receiving waters, and the costs and benefits of controls available to address these adverse impacts. Additionally, it was a goal of the research to identify the knowledge and information gaps that need to be addressed such that the approach can be developed and implemented for a specific waterbody.

Effectively controlling nutrient pollutant loadings to receiving waters has become a major challenge for resource managers. Recent requirements by the U.S. EPA for numeric nutrient criteria in Florida have made this an urgent need, especially since nitrogen is usually the limiting nutrient in Florida’s coastal waters. The linkage of Florida receiving water impacts to nitrogen sources was therefore used as a case study and basis for the approach developed in this project. Specifically, the scope of this project was to:

♦ Develop a technically defensible framework for assessment of the linkage between nitrogen sources and water quality responses;
♦ Understand how the source-response linkage is influenced by nitrogen source controls;
♦ Develop an approach to evaluate the effectiveness of nitrogen source control strategies;
♦ Gather and summarize available information and data relative to the Florida nitrogen situation, and identify any data or knowledge gaps that would need to be filled to implement the recommended approach in Florida.

ES.2 Summary of Approach

Nutrient pollution is one of the leading causes of water quality impairment in the nation, and the quantity of nutrients reaching the nations waters has dramatically escalated over the past 50 years (U.S. EPA, 2009a). Nitrogen and phosphorus loadings to waterbodies impact water quality by stimulating plant and algae growth which subsequently may result in depletion of dissolved oxygen, degradation of habitat, development of harmful algal blooms, impairment of the waterbody’s designated use, and impairment of drinking water sources.

The conceptual approach to linking receiving water impacts to sources and to appropriate water quality management decisions is shown in Figure ES-1. The linkages that must be understood to develop an approach to evaluate nutrient control effectiveness are represented by the arrows between the boxes. Nutrient sources in a watershed can be indentified and quantified, but their linkages to water quality impacts and responses must be understood so that effective nutrient controls can be implemented. This in turn requires knowledge of how the source-response linkage is influenced by various nutrient controls. Finally, each source-response-control strategy...
must be linked to a water quality management decision process that allows assessment of ALL costs and benefits, including social, environmental and economic benefits and costs. This is the only way that nutrient control strategies can be evaluated on like terms, and the most appropriate strategies developed and pursued. This process also includes feedbacks to response-control strategies and allows an adaptive management approach to be used to make progress towards water quality goals. However, the conceptual approach outlined in Figure ES-1 does not lend itself to application to a specific waterbody, and a more detailed and stepwise approach was desired as the outcome of this project.

Based on a state-of-the-knowledge literature review of nitrogen sources, water quality impacts, and nitrogen control strategies, a more detailed approach to linking receiving water impacts to sources and to water quality management decisions was developed. The framework for this approach is illustrated in Figure ES-2. This framework was designed as a decision tree consisting of key steps and questions that lead one through the approach and includes the detailed elements required to develop a process for controlling excess nutrient inputs to surface waterbodies. This report was organized along the lines of this framework and details each key step in this approach by chapter.
Prior to initiating management activities for a waterbody, appropriate water quality criteria should be established. This is arguably the most important step in the entire process. Unless the established criteria are meaningful with respect to meeting desired water quality conditions, subsequent activities may well prove fruitless.

Criteria development should include the use of data as required to establish technically-defensible relationships between levels of contaminants in a waterbody and the manifestations of impairment. Data used to develop criteria, especially numeric nutrient criteria, should be adequate to capture the natural, spatial and temporal variability of the specific waterbody. Characteristics of a waterbody can change in response to site-specific conditions, seasonality, inter-annual meteorological conditions, and confounding factors such as hydrologic alterations and residence time. The criteria should also provide a realistic time frame in which to gauge system response and compliance evaluation (Janicki Environmental, 2010a).

After the appropriate criteria are established, a receiving waterbody can be assessed to determine with relative certainty if it is impaired or not (Chapter 2.0). If there is no impairment, then water quality criteria have been met and no corrective actions are needed. However, the Clean Water Act (CWA) mandates that total maximum daily loads (TMDLs) be set for all waterbodies, not just those exhibiting impairment. The 303(d) list of the CWA sets priorities for TMDL development for individual waterbodies. However, in some cases, for example in an area where rapid urban growth is occurring or where agricultural practices are intensifying, following the process shown in Figure ES-2 would be prudent even for an unimpaired waterbody. Identifying an unimpaired waterbody’s assimilative capacity can help resource managers identify methods of precluding avoidable future water quality degradation.

If the waterbody is shown to be impaired, then the source–response relationship must be demonstrated. First, the causative agent, whether it is nutrients or another factor such as hydrologic conditions, must be identified (Chapter 3.0). If nutrients are determined to be the causative agent, then the limiting nutrient must be determined. Nitrogen is most often the limiting nutrient in estuarine waters, and phosphorus is more often limiting in freshwater systems. This is by no means globally true as discussed in Chapter 3.0, and site-specific conditions must be considered when conducting the assessment.

The assimilative capacity of a waterbody must also be determined if a control strategy is to be effective. The assimilative capacity, or carrying capacity, is the pollutant loading rate that a waterbody can accommodate without showing signs of impairment. For management purposes, the loading rate corresponding to the assimilative capacity should not be considered a target, but rather a threshold for impairment that is to be avoided.
Once the limiting nutrient is identified, significant nutrient loading sources to the subject waterbody must be identified. Additionally, the sources’ relative importance in contributing to impairment must be quantified (Chapter 4.0).

Quantifying individual source loadings is a two-step process. First, the nitrogen, particularly the reactive nitrogen, load entering the watershed must be quantified. Second, in-watershed pathways and processes that can sequester, transform, or remove nitrogen from the watershed must be examined. In-watershed processes represent a critical element of the source-response linkages and are central to this investigation, as discussed in Chapter 4.0. One of the most challenging aspects of nutrient control is quantifying the net effects of these processes on nutrients. The data and information gaps listed at the end of Chapter 4.0 reflect this point.

The process of evaluating and selecting one or more control measures can be initiated when nitrogen loading rates to the waterbody have been characterized (Chapter 5.0). Controls may be structural (stormwater treatment ponds, enhanced wastewater treatment practices, power plant emission scrubbers, etc.), source controls (optimizing fertilizer and animal waste handling...
practices, etc.), public education, or regulatory. Controls should be evaluated with respect to existing and potential future nitrogen loading rates and the specific features of a waterbody.

All the potential benefits and costs of each measure should be assessed following the evaluation of control measures based on technological merit (Chapter 6.0). This assessment should include social, environmental, and economic costs to all parties. Issues included in the evaluation may include public health, aesthetics, supporting diverse or conflicting uses of the waterbody by stakeholders, property values, recreation, and other considerations as discussed in Chapter 6.0.

The potential benefits to water quality resulting from the use of a control should be evaluated prior to implementing a control strategy (Chapter 7.0). This step should of course be a major part of the control strategy development process. Various empirical and computer modeling approaches can be used for this step, as summarized in Chapter 7.0.

Implementing the selected control strategy (Chapter 8.0) can be a challenging process. Elements of implementation include identification and inclusion of stakeholders in the process (starting as early in this entire process as practicable), identifying responsible parties for action items, ensuring that funding sources are sustainable, maintaining the focus of the effect of parties to complete their responsibilities, agreeing on how to measure success or failure of the management activities, and determining a protocol for amending the management strategy if positive results are not observed in an appropriate time frame.

Sustaining a successful management program depends on continuing to monitor and assess the effectiveness of a control strategy (Chapter 9.0). Water quality changes and other relevant metrics should be reviewed on a regular basis so that the control strategy can be adjusted if needed. It is very important to share information obtained from this process with decision-makers. The demonstration of positive results can help ensure that support for the expenditure of funds and effort will continue.

If the waterbody shows signs of improvement based on the results of the monitoring and assessment, the existing plan can be continued or enhanced. If improvements are not observed, or are not adequate to restore water quality, the strategy should be re-evaluated and adjusted.

### ES.3 Conclusions

This section provides a summary of the key conclusions drawn from the state-of-the knowledge review and approach developed for this project.

#### ES.3.1 Water Quality Impacts

There are numerous studies and sources of data providing evidence that waterbodies respond negatively to an excessive supply (loading rate) of the nutrients nitrogen and phosphorus, resulting in the characteristic symptoms of eutrophication. While knowledge of the links between pollutant inputs and waterbody responses may be more advanced for nutrients than for other emerging constituents, significant gaps remain in the understanding of the degree to which waterbodies can assimilate nutrient loading without reflecting these symptoms. The following conclusions were drawn from the literature reviewed in this study.

♦ Since nitrogen and phosphorus are necessary ingredients for life in aquatic ecosystems, developing nutrient criteria is more complicated than developing criteria for toxic substances,
or constituents whose desired quantity is extremely low or zero. While most water quality criteria in the CWA are based on a toxicity threshold where acceptable concentrations can be established below the level found to be harmful, nutrients at some level are necessary to support biological communities, and these natural levels change with environmental conditions. This can lead to considerable difficulty in establishing nutrient concentration limits to meet water quality goals.

- Each waterbody is different, and there are numerous factors which impact how any particular waterbody will respond to excess nutrient loading, including hydraulic residence time, freshwater inflow, clarity and light attenuation, geologic substrate, depth, temperature, and degree of physical alterations, such as channelizing. Additionally, it is often difficult to identify the causative nutrient in eutrophic waterbodies. Some systems vary between nitrogen limitation, phosphorus limitation, or nitrogen-phosphorus co-limitation in response to the relative contribution of each nutrient by season and/or source.

- Determining a causative agent for eutrophication impairment is often not straightforward, therefore, establishing a cause and effect relationship between nutrients and water quality is challenging. Nutrient enrichment and the resulting problems are frequently widespread and manifested at a location remote from the sources, and may not show themselves for some time after significant inputs to the system have occurred. Without the cause and effect, setting numeric concentration criteria is difficult. Clearly, such a relationship is very difficult to develop for a group of waterbodies, and therefore typically requires a site-specific approach based on the characteristics of the individual waterbody.

- The cause of impairment of a waterbody should be determined with relative certainty prior to making any management decisions. The cause may be excess nitrogen, excess phosphorus, or other agent (hydrologic alteration, etc.).

- Development of TMDLs for nutrients without first discerning the dose-response relationships between nutrients and chlorophyll and DO is not justified. Too often both chlorophyll and DO impairments are attributed to excessive nutrients by default. Such ill-conceived conclusions lead to TMDL development that can seriously affect the stakeholders associated with the impaired water by requiring nutrient controls that will have little effect on the water quality.

- The response of an impaired waterbody to control measures may be non-linear. That is, there may not be a direct one-to-one relationship between reduced loadings and improved water quality. At times the aquatic system may need a considerable change in equilibrium to begin the recovery process. This may result in a significant lag between the time of applying controls and observing a response from the waterbody.

- Information obtained from examining N:P ratios can be useful for nutrient management. However, the ratios can vary spatially and most notably temporally. When very costly controls are needed to address nutrient loading, the increased certainty afforded by the application of either laboratory or in situ testing to identify the limiting nutrients should be considered.

- The lack of limitation by either nitrogen or phosphorus can lead to ineffective and unnecessary controls. Nutrient load reductions to a waterbody that is light-limited due to colored dissolved organic matter (CDOM) may result in no change in resultant water quality, for example.
The role of alterations to the hydrologic system on residence time should be considered when investigating the causes of eutrophication. Lower freshwater inflows to a waterbody, or retention of water by a control structure such as a dam, may result in less flushing and longer residence time, while channelization or flood control alterations may increase flow rates and decrease residence times. Either types of alteration could impact water quality and be a cause of impairment.

**ES.3.2 Nitrogen Sources**

Nitrogen is one of the earth’s most abundant elements, and is essential for both plant and animal life within the biosphere, that portion of the earth’s crust and atmosphere where living things reside. Most environmental nitrogen is in the gaseous form, $\text{N}_2$. Nitrogen in this form is essentially unavailable to most life forms. Only after the $\text{N}_2$ is transformed via nitrogen fixation to biologically available nitrogen (called reactive nitrogen) can it be utilized by most plants. In nature, biological fixation of nitrogen (BNF) is carried out by a variety of specialized organisms, including blue-green algae, the soil bacteria *Azobacter*, legume plants (e.g., beans, peas, and clover), and the symbiotic bacteria *Rhizobium*. Additionally, nitrogen can be fixed by some natural inorganic processes such as lightning, solar radiation, and meteorite trails (Sauer et al., 2008). The natural fixation of nitrogen from the atmosphere is balanced by the return of $\text{N}_2$ to the atmosphere by denitrification, another biological process that converts nitrate-nitrogen to nitrogen gas. Figure ES-3 illustrates a simplified version of the nitrogen cycle, which in nature maintains a balance of nitrogen between the $\text{N}_2$ in the atmosphere and the reactive nitrogen in the biosphere. Conclusions regarding nitrogen sources are summarized below.

As a result of man’s activities there has been a tremendous increase in the amount of reactive nitrogen available in the global environment over the past 50 years.
Biological nitrogen fixation (BNF) has been enhanced via increased cultivation of nitrogen fixing crops for agriculture.

The combustion of fossil fuels, previously locked away in the earth’s crust, releases new reactive nitrogen compounds into the atmosphere that then return to earth via atmospheric deposition.

The Haber-Bosch process was developed to synthesize reactive ammonia from gaseous N₂. This industrial process began to be used extensively for synthetic fertilizer production in the 1950s, and its use has grown exponentially since.

Figure ES-4 illustrates the increase in reactive nitrogen in the biosphere over the last century. Globally, the increase in biologically available nitrogen in our biosphere has clearly been influenced by the combustion of fossil fuels and the production of synthetic fertilizers, and these sources are projected to increase further in the future.

![Figure ES-4. Estimated World-Wide Production of Reactive Nitrogen. Reprinted with Permission from Nielsen, 2005.](image)

Nitrogen can enter a receiving waterbody through atmospheric deposition, nonpoint source discharge, or point source discharge pathways. However these linkages are not the ultimate sources of nutrients. The key culturally-derived nitrogen sources are combustion of fossil fuels, fertilizer and animal waste, and human and industrial waste.

In Florida, nitrogen loading estimates have been developed for numerous watersheds. Table 4-2 provided a summary of the relative contributions (%) by source of nitrogen loads to a number of Florida receiving waters. While these relative contributions varied appreciably among waterbodies, nonpoint sources were clearly a major contributor to nitrogen load in every case.
ES.3.3 Nitrogen Controls

There are two ways to control nutrient loadings to receiving waterbodies. The first method of nutrient control is to manage the quantity of nutrients introduced into the environment from the ultimate sources. For example, agricultural and urban fertilization practices can be optimized to reduce the amount of excess nutrient inputs to the environment.

The second method is to remove the nutrients after they have entered the network of pathways and reservoirs between release and delivery to receiving waterbodies. Improvement in wastewater treatment is a good example of this approach that has already been successful. Nutrients in nonpoint sources can also be removed by the use of a variety of stormwater treatment alternatives. Atmospheric sources can be reduced by treatment of power plant and vehicle emissions as they are discharged. Key conclusions developed regarding nitrogen controls follow.

- If WWTP effluent was required to meet the recently proposed Florida in-stream numeric nutrient criteria values (see Table 5-1), it would be difficult to do so consistently with currently accepted limits of wastewater treatment technology. The currently accepted practical limit of treatment technology for nitrogen is approximately 3 mg/L total nitrogen (TN), although emerging technologies have shown that further reductions to 1-2 mg/L are possible with additional research.

- The cost of increasing wastewater treatment to remove nutrients to very low levels is substantial. Figure ES-5 illustrates that the incremental O&M cost increase per additional pound of nitrogen removed as treatment level increases from secondary to a TN effluent limit of 1 mg/L (Reardon et al., 2010) is very high.

![Figure ES-5. Incremental O&M Cost Increase per Additional Pound of N Removed. Reardon et al., 2010.](image-url)

- Conventional onsite wastewater treatment systems (OWTS) are not designed for nitrogen removal, and there are an estimated 2.5 million OWTS in Florida (Briggs et al., 2007). While some level of nitrogen removal occurs naturally in the soil below these systems, they can be a substantial contributor to nitrogen loading in some watersheds. New developments in
OWTS technology have resulted in systems that achieve nitrogen removals of up to 75%, but these systems are expensive, difficult for homeowner operation, and sometimes difficult to retrofit into existing properties.

- The management of OWTS by routine evaluations and maintenance requirements can improve OWTS performance and identify failures that may be impacting water quality. Requirements for such a program have recently been passed into law in Florida, with implementation scheduled for 2011.

- Concentrated animal feeding operations (CAFOs) are estimated to generate an estimated 500 million tons of manure annually, over 3 times the estimated waste generated by the human population of the U.S. This agricultural waste can impact water quality via point source discharges or as a component of nonpoint source inputs to receiving waters, CAFOs and are not required to meet the same effluent quality requirements as domestic waste.

- Fertilizer restriction ordinances and education are controls that can be used to reduce the nitrogen loads in stormwater runoff to receiving waters from urban landscapes. A recent statewide Urban Turf Fertilizer Rule adopted in Florida is expected to reduce the nitrogen fertilizer applied to urban turf and lawns by 20-25% (FDEP, 2010a).

- Florida requires treatment of all stormwater discharges for new development. Current technologies used for stormwater treatment systems in Florida effectively remove 20-30% of TN from stormwater discharges. However, the lack of end-of-pipe monitoring for stormwater discharges makes it extremely difficult to estimate stormwater discharges for nutrients.

- Atmospheric deposition of nitrogen is a major source of nitrogen loading in urban watersheds. The Tampa Bay Estuary Program lists atmospheric deposition as the second leading source of nitrogen loads to Tampa Bay. Nitrogen controls recently completed at the Tampa Electric Company (TECO) Big Bend power plant are expected to reduce the nitrogen load to Tampa Bay by approximately 60 tons per year.

- Water quality trading is a market-based approach to achieving nutrient removal goals at the lowest possible cost by allowing the voluntary exchange of nutrient credits for money. Under the right circumstances, a well-designed and properly implemented water quality trading system can be an effective tool to achieve water quality goals. Consideration of many specific issues is necessary to evaluate, design and implement a successful water quality trading program. Consultations with economists, those who have implemented past programs, and experts at the U.S. EPA and WERF are recommended.

**ES.3.4 Water Quality Management Decisions**

Water quality management decisions required to restore nutrient impaired waterbodies are complex and extend far beyond the technical decisions. The development and implementation of nutrient control strategies must consider the economic, social, and environmental considerations as well as the technical. In the past, water quality improvement projects have been based on a simple cost comparison of the top technically ranked alternatives far too often. In reality, there are many costs and benefits associated with water quality improvements, and the best water quality management decisions are based on consideration of all these costs and benefits. Key conclusions from this review are listed below:

- The economic value of avoiding nutrient enrichment of waterbodies is likely to be substantial and includes income, use, non-use, and property values.
To properly evaluate nutrient control strategies on like terms, all benefits and costs must be considered. The chosen level of nutrient reduction should be based on maximizing the net benefits of nutrient management, where net benefits are the benefits to all persons and entities minus the costs to all persons or entities now and in the future.

When benefits, costs, and public input are not sufficiently considered, the actual design and implementation of nutrient management alternatives can result in choices where the costs are greater than the benefits and/or where the most cost-effective alternatives are not implemented.

An example illustrating this is wastewater treatment improvements. Although it has been demonstrated that wastewater treatment plants can approach carbon neutral operation (Wett et al., 2007) for a total nitrogen limit of 14 mg/L, energy neutrality becomes increasingly unlikely as stricter nitrogen limits are imposed. Figure ES-6 shows the incremental greenhouse gas (GHG) emissions increase associated with the increases in treatment level from secondary to TN effluent limits of 1 mg/L (Reardon et al., 2010). The social and environmental costs of these emissions need to be considered in determining the net environmental benefit of improved treatment.

![Figure ES-6. Incremental GHG Emissions Increase by Treatment Level for N. Reardon et al., 2010.](image)

A benefit-cost evaluation model can be developed to evaluate and rank nitrogen control alternatives. Such models provide a structured approach to:

- Achieving an apples-to-apples comparison of nitrogen control alternatives
- Identifying data and information gaps
- Evaluating the impact of uncertainty on the ranking of each alternative
- Providing useful input into the design of nutrient reduction programs
- Updating the conclusions as new data and information become available
The benefit-cost evaluation model should be designed so that the benefits and costs of nutrient reduction alternatives are better understood; politically sensitive issues can be put into perspective; and stakeholders’ interests are placed on a level playing field.

For water quality improvement alternatives, benefits and costs typically include: value to the economy measured by household income; resident wealth and well-being; use value of recreation activities; non-use value of the ecosystem supported by the waterbody; value as a potable water supply source; avoided costs of disease; replacement cost of ecosystem services; and value of commercial fish harvested.

For some types of benefits and costs, dollar values can be relatively easy to estimate. For other types of benefits and costs, the amount of research necessary to estimate dollar values is relatively high so other measures that are relatively inexpensive to estimate and that reflect the dollar value are often used instead.

Benefits and costs should be assigned monetary values to the extent that the available time and budget will allow. Where insufficient information exists to assign a monetary value, benefits and costs should be described and quantified to the extent practical. In addition, stakeholder involvement in the decision-making process and in the implementation of the nutrient control strategy is critical to success.

It is important to assess nutrient reduction technologies and alternatives well before they need to be implemented because time is needed to estimate the benefits and costs of each alternative, including estimates of changes in income, use, non-use, and property values associated with changes in nitrogen or nutrient levels.

**ES.3.5 Water Quality Improvements – Estimating Receiving Water Response to Controls**

Once impairment of a waterbody has been substantiated and sources and controls identified, an evaluation of the expected waterbody response must be conducted. This can be accomplished using either empirical or mechanistic modeling techniques. Key conclusions drawn from the review of this topic are summarized below:

♦ Both empirical and mechanistic models can be useful in predicting the potential water quality benefits from control measures. Information obtained from the use of these tools can be helpful in establishing water quality targets and determining the assimilative capacity of waterbodies. They may also be useful in determining potential dose-response relationships.

♦ Empirical models depend on relationships between constituents derived solely from observed data, and can be an efficient way to develop usable relationships under certain data domains. However, the assurance of cause and effect linkages can be misleading when using empirical models. Many times the linkages are treated as a black box that can lead to inappropriate assumptions and conclusions regarding causal relationships.

♦ It is important to have adequate data to capture the range of variability in loadings and water quality response. Empirical models should only be used to predict conditions that exist within the model domain.

♦ Mechanistic models utilize mathematical relationships to represent chemical, biological, and physical transformations. When linked with hydrodynamic models, mechanistic water quality models can supply information regarding movement and transformations of constituents of
interest within a system. Mechanistic models are often very data intensive and are based on rate constant values that may not be well-understood.

**ES.4 Recommendations for Research Needs**

Numerous knowledge gaps and the need for additional data were discovered during the literature review and in development of the framework for this project. These identified needs range in scope from simple data collection needs to major research efforts related to cause and effect relationships between water quality and the major constituents of concern. Knowledge gaps were identified at the end of each chapter, and the key research needs formulated during the conduct of this project are summarized below:

♦ Lack of compliance with existing water quality standards or criterion often provides the impetus for the investigation of the linkages between nutrient sources and resulting water quality. Improper criteria, i.e., those that are not defensible or overly-protective, can result in improper water quality management decisions. Therefore, it is absolutely essential that the development and implementation of nutrient criteria be built upon robust, defensible methodologies and data sources. Many states are facing the imposition of numeric nutrient criteria by the U.S. EPA and the lack of appropriate methodologies and data sources remains as a significant gap. Clearly, efforts to fill this gap are warranted if unnecessary nutrient controls are to be avoided. Therefore, it is recommended that a toolbox of potential methodological approaches to the establishment of numeric nutrient criteria be developed. These approaches must build on the specific knowledge of how waterbodies respond to variation in nutrient conditions and most importantly on the choice of appropriate response variables and defensible thresholds. The nature and volume of the available data for both stressor and response variables, as well as for any significant confounding factors, must also be considered.

♦ To better understand the linkage between nutrient sources and receiving water impacts, research should be undertaken to better identify and quantify the transformations of nutrients in the environment, including the confounding factors that affect the behavior of nutrients in aquatic systems.

♦ While it is obvious from the literature that fertilizer use and combustion of fossil fuels are key sources of nitrogen to the environment, few studies have quantified the nutrient load to a given watershed resulting specifically from these sources. Detailed watershed studies documenting the specific contributions from these sources should be conducted to develop a better understanding of the potential role that nutrient controls can play in managing their impacts.

♦ Additional research is needed on the nitrogen removal potential of current and emerging wastewater treatment technologies in order to properly evaluate which processes will perform reliably to meet extremely low proposed nitrogen limits for a waterbody. The impact of increased energy use, GHG emissions, and other social and environmental costs associated with increased treatment also needs to be considered in the evaluation of net benefits.

♦ Onsite wastewater treatment systems serve approximately one-third of the population, yet research expenditures to improve the treatment performance of these systems for nutrient removal has been only a small fraction of that allocated for municipal WWTPs. Further research on nutrient removal technologies for new as well as existing OWTS, is warranted.
♦ Additional monitoring of “end of pipe” stormwater discharges is needed to understand their contribution to a waterbody’s impairment and to properly evaluate and implement control methods.

♦ Research is needed on the economic and environmental benefits of nutrient recovery and recycling, as well natural treatment technologies and improvements to existing best management practices pertaining to concentrated animal feeding operations (CAFOs).

♦ There is a lack of studies that have determined the change in income, use, non-use, and property values associated with changes in ecosystem quality, especially related to reduced nutrient loading. Thus, a major research gap is an understanding of how the monetary values generated by our nations waterbodies change as water quality improves or impairments increase. This research could be conducted immediately using publicly available information and survey research.
CHAPTER 1.0

INTRODUCTION

Concern over water quality impacts from an ever increasing array of potential contaminants has become a major challenge for water resource managers. To better understand the linkages between sources of a constituent, its effect on water quality, and the costs and benefits of controls, the Water Environment Research Foundation (WERF) has undertaken a research challenge entitled “Linking Receiving Water Impacts to Sources and to Water Quality Management Decisions”. This report describes one of the efforts undertaken as part of this research challenge.

1.1 Purpose

The purpose of the overall WERF “Linkages” research challenge is to develop a better understanding of the linkages between:

♦ Sources of a constituent.
♦ The predicted and measured adverse impacts of the constituent on receiving waters.
♦ The total cost and benefits of controls available to address these adverse impacts.

Additionally, it is a goal of this research challenge to identify the knowledge and information gaps that need to be addressed such that a comprehensive and cost-effective pollutant control strategy can be developed and implemented for a given set of constituents.

1.2 Scope

Linking water quality impacts to the many constituents of concern is a daunting task, and several research efforts are included in this WERF research challenge to adequately address the major constituent categories. The focus of this report is on developing a state-of-the-knowledge approach to establishing the linkages listed above, and to apply this approach to nutrients as a source constituent.

Nutrient pollution is one of the leading causes of water quality impairment in the nation, and the quantity of nutrients reaching the nation’s waters has dramatically escalated over the past 50 years (U.S. EPA, 2009a). Nitrogen and phosphorus loadings to waterbodies impact water quality by stimulating plant and algae growth which subsequently may result in depletion of dissolved oxygen, degradation of habitat, development of harmful algal blooms, impairment of a waterbody’s designated use, and impairment of drinking water sources.

Effectively controlling nutrient pollutant loadings to receiving waters has thus become a major challenge for resource managers. Recent requirements by the U.S. EPA for numeric nutrient criteria in Florida have made this an urgent need, especially for nitrogen since it is usually the primary limiting nutrient in Florida’s coastal waters. The linkage of Florida receiving water impacts to nitrogen sources was therefore used as a case study and basis for the approach developed in this project. Specifically, the scope of this project is to:
♦ Develop a technically defensible framework for assessment of the linkage between nitrogen sources and water quality responses.

♦ Understand how the source-response linkage is influenced by nitrogen source controls.

♦ Develop an approach to evaluate the effectiveness of nitrogen source control strategies.

♦ Gather and summarize available information and data relative to the Florida nitrogen situation, and identify any data or knowledge gaps that would need to be filled to implement the recommended approach in Florida.

1.3 Florida’s Approach to Achieving Water Quality Standards

Section 303(d) of the Clean Water Act (CWA) requires states to submit lists of surface waters that do not meet applicable water quality standards (impaired waters) after technology-based effluent limitations are implemented, and to establish Total Maximum Daily Loads (TMDLs) for these waters on a prioritized schedule. TMDLs are the maximum amount of a pollutant that a waterbody can assimilate without exceeding water quality standards.

The Florida Department of Environmental Protection (FDEP) is responsible for developing these TMDLs which it defines as “a scientific determination of the maximum amount of a given pollutant that a surface water can absorb and still meet the water quality standards that protect human health and aquatic life. Waterbodies that do not meet water quality standards are identified as “impaired” for the particular pollutants of concern – nutrients, bacteria, mercury, etc. – and TMDLs must be developed, adopted and implemented for those pollutants to reduce pollutants and clean up the waterbody” (http://www.dep.state.fl.us/water/tmdl/).

Florida’s water quality standards upon which the TMDLs are based are either narrative or numeric. Many of the water quality standards in Florida are narrative. For example, the minimum water quality standard for Florida’s surface water is as follows (from Section 62-302.500, Florida Administrative Code).

“All surface waters of the State shall at all places and at all times be free from:

(a) Domestic, industrial, agricultural, or other man-induced non-thermal components of discharges which, alone or in combination with other substances or in combination with other components of discharges (whether thermal or non-thermal):

1. Settle to form putrescent deposits or otherwise create a nuisance; or

2. Float as debris, scum, oil, or other matter in such amounts as to form nuisances; or

3. Produce color, odor, taste, turbidity, or other conditions in such degree as to create a nuisance; or

4. Are acutely toxic; or

5. Are present in concentrations which are carcinogenic, mutagenic, or teratogenic to human beings or to significant, locally occurring, wildlife or aquatic spe-
cies, unless specific standards are established for such components in Rules 62-302.5002 or 62-302.530; or

6. Pose a serious danger to the public health, safety, or welfare.

(b) Thermal components of discharges which, alone, or in combination with other discharges or components of discharges (whether thermal or non-thermal):
   1. Produce conditions so as to create a nuisance; or
   2. Do not comply with applicable provisions of Rule 62-302.520, F.A.C.

(c) Silver in concentrations above 2.3 micrograms/liter in predominately marine waters.”

Narrative water quality standards are useful in describing, in layman’s terms, the conditions under which impacts to a waterbody are unlawful and such descriptions are useful. In contrast, numeric water quality standards attempt to establish a limit on the concentration or amount of a substance that may lawfully exist in a waterbody. Both types of criteria are established in State law and are determined through scientific judgment and the political process. Benefits and costs associated with compliance are considered informally and implicitly in this process.

The basic steps used by the FDEP to establish TMDLs are as follows and were taken directly from the FDEP website cited above.

1. “Assess the quality of surface waters – are they meeting water quality standards?
2. Determine which waters are impaired – that is, which ones are not meeting water quality standards for a particular pollutant or pollutants.
3. Establish and adopt, by rule, a TMDL for each impaired water for the pollutants of concern – the ones causing the water quality problems.
4. Develop, with extensive local stakeholder input, a Basin Management Action Plan (BMAP).
5. Implement the strategies and actions in the BMAP.
6. Measure the effectiveness of the BMAP, both continuously at the local level and through a formal re-evaluation every five years.
7. Adapt – change the plan and change the actions if things aren’t working.
8. Reassess the quality of surface waters continuously.”

The development of TMDLs takes place in the context of Chapter 99-223, Laws of Florida. This law details the process for: 1) listing impaired waters, 2) determining which waters will be subjected to TMDL calculations, 3) adopting by rule those calculations and associated allocations of pollutant loadings, and 4) implementing the management strategies. These strategies are to be designed to reduce the loadings and enable the waterbody to meet water quality standards (From http://www.dep.state.fl.us/water/tmdl/faq.htm).

In the case of establishing nitrogen and phosphorus TMDLs, the process involves 1) identifying water quality indicators and targets; 2) assessing the sources of nitrogen and phosphorus loads entering the waterbodies; 3) establishing the linkages between water quality indicators/ targets and nitrogen/ phosphorus sources; 4) establishing TMDLs for nitrogen and/or phosphorus and 5) identifying and implementing methods to reduce nitrogen and/or phosphorus
loads to the waterbody. This process results in numeric goals and the methods to achieve these goals to restore water quality and remove the impairment. Nutrient reduction methods should be developed considering the benefits and costs to those who implement the methods and to the regulating agency.

While the agency attempts to consider as many benefits and costs as possible, there is no formal policy or guidelines regarding a benefit and cost evaluation. During State rulemaking to implement methods to comply with a TMDL, a Statement of Estimated Regulatory Costs (SERC) is usually required as described in Section 120.54, Florida Statutes. The requirements of a SERC include a quantification of the costs to comply with a proposed State regulation. Consideration of all other benefits and costs is not required although some costs and benefits may be established through stakeholder involvement.

Because Florida’s waterbodies have diverse characteristics they are best evaluated either individually or in small groups. Final TMDLs have been set for well over 100 waterbodies in Florida, including Lake Apopka and Lake Okeechobee, Florida’s largest lake. TMDLs for many dozens of other waterbodies have been proposed.

Florida’s TMDL process has been criticized as moving too slowly to establish TMDLs for the vast number of waterbodies in Florida. The TMDL process requires substantial time and resources to evaluate the impact of nutrient loads on receiving waterbodies. Developing nutrient criteria is more complicated than developing limits for toxic substances. While most water quality criteria in the CWA are based on a toxicity threshold where acceptable concentrations can be established below the level found to be harmful, nutrients are necessary at some level to support biological communities and their impacts are determined by many natural and man-made influences.

In establishing a TMDL, a significant amount time and effort is needed to collect the necessary water quality data, evaluate the impacts and confounding factors, and engage the key stakeholders. If these steps are implemented properly, this process results in TMDLs that are likely to be successful because they were developed by thoughtfully considering the impact of achieving the TMDL on the resulting quality of a specific receiving water.

The conceptual approach to linking receiving water impacts to sources and to appropriate water quality management decisions is shown in Figure 1-1. The linkages that must be understood to develop an approach to evaluate nutrient control effectiveness are represented by the arrows between the boxes. Nutrient sources in a watershed can be identified and quantified, but their linkages to water quality impacts and responses must be understood so that effective nutrient controls can be implemented. This in turn requires knowledge of how the source-response linkage is influenced by various nutrient controls. Finally, each source-response-control strategy must be linked to a water quality management decision process that allows assessment of ALL costs and benefits, including social, environmental, and economic benefits and costs. This is the only way that nutrient control strategies can be evaluated on like terms, and the most appropriate strategies developed and pursued. This process also includes feedbacks to response-control strategies and allows an adaptive management approach to be used to make progress towards water quality goals. However, the conceptual approach outlined in Figure 1-1 does not lend itself to application to a specific waterbody, and a more detailed and stepwise approach was desired as the outcome of this project.
1.4 Project Framework and Approach to Achieving Water Quality Criteria

Based on a state-of-the-knowledge literature review of nitrogen sources, water quality impacts, and nitrogen control strategies, an approach to linking receiving water impacts to sources and to water quality management decisions was developed for this project. The framework for this approach is illustrated in Figure 1-2. This framework was designed as a decision tree consisting of key steps and questions that lead one through the approach and includes the detailed elements required to develop a process for controlling excess nutrient inputs to surface waterbodies. This report was organized along the lines of this framework and details each key step in this approach by chapter.

Prior to initiating management activities for a waterbody, appropriate water quality criteria should be set. This is arguably the most important step in the entire process. Unless the estab-
lished criteria are meaningful with respect to meeting desired water quality conditions, subsequent activities may well prove fruitless.

Criteria development should include the use of data as required to establish technically-defensible relationships between levels of contaminants in a waterbody and the manifestations of impairment. Data used to develop criteria, especially numeric nutrient criteria, should be adequate to capture the natural, spatial and temporal variability of the specific waterbody. Characteristics of a waterbody can change in response to site-specific conditions, seasonality, inter-annual meteorological conditions, and confounding factors such as hydrologic alterations and residence time. The criteria should also provide a realistic time frame in which to gauge system response and compliance evaluation (Janicki Environmental, 2010a). Although water quality criteria can be an outcome of the approach discussed in this report, the detailed development of numeric water quality criteria based on cause and effect relationships between nutrients and specific waterbody characteristics was beyond the scope of this project.

After the appropriate criteria are established a receiving waterbody can be assessed to determine with relative certainty if it is impaired or not (Chapter 2.0). If there is no impairment then water quality criteria have been met and no corrective actions are needed. However, the Clean Water Act (CWA) mandates that total maximum daily loads (TMDLs) be set for all waterbodies, not just those exhibiting impairment. The 303(d) list of the CWA sets priorities for TMDL development for individual waterbodies. However, in some cases, for example in an area where rapid urban growth is occurring or where agricultural practices are intensifying, it would be prudent to follow the process shown in Figure ES-1 even for an unimpaired waterbody. Identifying an unimpaired waterbody’s assimilative capacity can help resource managers identify methods of precluding avoidable future water quality degradation.

If the waterbody is shown to be impaired, then the source–response relationship must be demonstrated. First, the causative agent, whether it is nutrients or another factor such as hydrologic conditions, must be identified (Chapter 3.0). If nutrients are determined to be the causative agent then the limiting nutrient must be determined. Nitrogen is most often the limiting nutrient in estuarine waters, and phosphorus is more often limiting in freshwater systems. This is by no means globally true as discussed in Chapter 3.0, and site-specific conditions must be considered when conducting the assessment.

The assimilative capacity of a waterbody must also be determined if a control strategy is to be effective. The assimilative capacity, or carrying capacity, is the pollutant loading rate that a waterbody can accommodate without showing signs of impairment. For management purposes, the loading rate corresponding to the assimilative capacity should not be considered a target, but rather a threshold for impairment that is to be avoided.
Figure 1-2. Project Framework and Approach.

Once the limiting nutrient is identified, all significant nutrient loading sources to the subject waterbody must be identified. Additionally, the sources’ relative contribution to impairment must be quantified (Chapter 4.0).

Quantifying the individual source loadings is a two-step process. First, the nitrogen, particularly the reactive nitrogen, load entering the watershed must be quantified. Second, in-watershed pathways and in-channel processes that can sequester, transform, or remove nitrogen from the watershed must be examined. In-watershed and in-channel processes represent a critical element of the source-response linkages and are central to this investigation, as discussed in Chapter 4.0. One of the most challenging aspects of nutrient control is quantifying the net affects of these processes on nutrients. The data and information gaps listed at the end of Chapter 4.0 reflect this point.

The process of evaluating and selecting one or more control measures can be initiated when nitrogen loading rates to the waterbody have been characterized (Chapter 5.0). Controls may be structural (stormwater treatment ponds, enhanced wastewater treatment practices, power plant emission scrubbers, etc.), source controls (optimizing fertilizer and animal waste handling
practices, etc.), public education, or regulatory. Controls should be evaluated with respect to existing and potential future nitrogen loading rates and the specific features of a waterbody.

All the potential benefits and costs of each measure should be assessed following the evaluation of control measures based on technological merit (Chapter 6.0). This assessment should include social, environmental, and economic costs to all parties. Issues included in the evaluation may include public health, aesthetics, supporting diverse or conflicting uses of the waterbody by stakeholders, property values, recreation, and other considerations as discussed in Chapter 6.0.

The potential benefits to water quality resulting from the use of a control should be evaluated prior to implementing a control strategy (Chapter 7.0). This step should of course be a major part of the control strategy development process. Various empirical and computer modeling approaches can be used for this step, as summarized in Chapter 7.0.

Implementing the selected control strategy (Chapter 8.0) can be a challenging process. Elements of implementation include identification and inclusion of stakeholders in the process (starting as early in this entire process as practicable), identifying responsible parties for action items, ensuring that funding sources are sustainable, maintaining the focus of the effected parties to complete their responsibilities, agreeing on how to measure success or failure of the management activities, and determining a protocol for amending the management strategy if positive results are not observed in an appropriate time frame.

Sustaining a successful management program depends on monitoring and assessment of the effectiveness of a control strategy (Chapter 9.0). Water quality changes and other relevant metrics should be reviewed on a regular basis so that the control strategy can be adjusted if needed. It is very important to share information obtained from this process with decision-makers. The demonstration of positive results can help ensure that support for the expenditure of funds and effort will continue.

If the waterbody shows signs of improvement based on the results of the monitoring and assessment, the existing plan can be continued or enhanced. If improvements are not observed, or are not adequate to restore water quality, the strategy should be re-evaluated and adjusted.

Chapter 10.0 provides the key conclusions reached based on the state-of-the-knowledge review. It also includes a list of recommended research needs based on the knowledge gaps identified in the report.
CHAPTER 2.0

WATER QUALITY IMPACTS:
IS THE WATERBODY IMPAIRED?

2.1 State-of-the-Knowledge: Linkages Between Nutrients and Water Quality Impacts

Knowledge of the links between pollutant inputs and waterbody responses is more advanced for nutrients than for some other constituents. Yet significant gaps remain in the understanding of the degree to which waterbodies can assimilate nutrient loading without reflecting the symptoms of eutrophication which may include algae blooms, excessive nuisance macrophytes, and the absence of natural biotic integrity.

The effects of eutrophication became apparent in 1960s in both fresh waterbodies in the U.S. (e.g., Lake Erie) and marine systems in Europe (e.g., Baltic Sea) (Schindler, 1974; Vallen tyne, 1974; Edmondson, 1972; Rosenberg, 1985; Nixon, 1995; Savage et al., 2010; Finni et al., 2001). The most visible symptom of eutrophication is the excessive algal growth that reduces water clarity. Eutrophication can also significantly affect phytoplankton community structure resulting in a greater abundance of less desirable taxa such as blue-green algae. These changes in the phytoplankton community can have cascading effects on higher trophic levels and the eventual transfer of organic carbon from the primary producers to less desired species – for example, the replacement of seagrasses with less desirable vegetation types.

By the 1970s, research focused on the most likely nutrient responsible for eutrophication in lakes and marine waters (Dillon and Rigler, 1974; Edmondson, 1969, 1972; Schindler, 1977; Ryther and Dunstan, 1971). While most research pointed to phosphorus limitation in lakes, some hypothesized that carbon was the limiting factor. Research methods used to identify the limiting nutrient included bioassays using test algae such as Selenastrum, limnocorrals using ambient phytoplankton taxa, whole-lake treatments for freshwater systems, and mesocosms for coastal waters (Payne, 1975; Chiaudani and Vighi, 1974; Shapiro, 1980; Lynch and Shapiro, 1981; Schindler, 1974; Oviatt et al., 1986).

Since the 1970s, many scientists and managers have been studying the deterioration of estuarine ecosystems via increases in nutrient loads and accompanying eutrophication (Paerl, 2006; Bricker et al., 2008; Fisher et al., 2006). In estuarine systems functioning without large anthropogenic disturbances, dissolved nutrients in river and stream discharge constitute the primary nutrient point source for many estuaries that receive significant freshwater input. The targeting of nutrient inputs from other point sources such as sewage outfalls and industrial effluent was met with much success, yielding improved water quality following implementation of advanced wastewater treatment (Greening and Janicki, 2006). Unfortunately, population growth and the growing need for agricultural output have led to an increase in nonpoint source pollution. It is estimated that human activity has increased the total rate of formation of reactive nitrogen globally by 33-55% through increases in agriculture via synthetic fertilizer (Howarth, 2008). In-
creases in reactive nitrogen have also resulted from increases in the encouragement of biological nitrogen fixation associated with agriculture and the inadvertent creation of reactive nitrogen through reaction with oxygen as fossil fuels are burned (Howarth, 2008; Paerl, 2006).

Knowledge of the physical characteristics of streams, rivers, lakes and estuarine systems is also central to understanding how biogeochemical cycles are influenced by hydrology. Biogeochemical processes influence the concentration of nutrients during transport through stream networks, resulting in a net decrease downstream (Peterson et al., 2001). It was estimated that 60% of the nitrogen entering a watershed’s stream network may be retained through biotic sequestration or lost through denitrification within streams of the eastern United States (Seitzinger et al., 2002).

Alteration of nutrient concentrations by in-channel processes during transport also changes the timing, quantity, and quality (labile vs. refractory) of nutrients exported to downstream ecosystems (Mulholland, 2004). In most systems, low flows typically allow a longer residence time while higher flows flush nutrients, detritus, etc. out of the system more rapidly. However, the re-suspension of organic detritus may increase nutrient levels in the water column. The rate of freshwater inflow can influence hydraulic residence time, and hence the time available for nutrients to react in the estuary (Bricker et al., 2008; Hagy et al., 2000).

Streams and rivers provide hydrologic coupling and their characteristics determine the concentrations and forms of nutrients downstream (Ensign et al., 2006). In-stream nutrient retention by algal assimilation and by periphyton represents a temporary storage of watershed-derived nitrogen as dissolved inorganic nitrogen (DIN) is converted to algal biomass. As this biomass senesces, DIN and dissolved organic nitrogen (DON) are released back to the stream ecosystem and potentially delivered downstream (Dodds et al., 2002; Ensign et al., 2006).

The next step in the understanding of how eutrophication affects aquatic ecosystems involved estimation of the linkage between nutrient loads and receiving water responses. Vollenweider (1968) developed the critical loading concept. Based on the statistical relationship between phosphorus loading and chlorophyll $a$ concentrations and adjusted for lake depth, lake trophic status could be predicted. Others expanded Vollenweider’s work enhancing routines for hydraulic flushing and other confounding factors in lakes (e.g., Dillon and Rigler, 1974). Although relatively simple, these models took into account losses due to sedimentation as well as flushing effects and represented a breakthrough in the ability to simulate eutrophication processes and predict lake trophic state (Dillon, 1974). This work indicated that the allowable phosphorus loading was greater in lakes with higher flushing rates. Still other research identified the importance of internal loading (Nürnberg, 1984; Sondergaard, et al., 2001; Nürnberg and LaZerte, 2004).

Based on the results of the research on eutrophication, management actions that controlled nutrient loadings were implemented to alleviate eutrophication symptoms. Phosphorus in detergents was identified as a significant source of nutrient enrichment to the Great Lakes and other waterbodies. Management of loadings was achieved by banning of phosphorus in detergents in watersheds draining to lakes affected by excessive loading. Advanced wastewater treatment (AWT) processes were developed to enhance the removal of phosphorus and nitrogen from domestic wastewater effluent. These and other actions all helped to decrease the nutrient load to receiving waters. The management of nutrient loadings has been shown to result in significant improvements in lakes (Edmondson, 1970; Schindler, 1974; Fitzpatrick and DiToro, 1999).
As opposed to lakes, there are fewer success stories where nitrogen load reductions have resulted in alleviation of eutrophic conditions in estuaries. Most recently, Greening and Janicki (2006) have documented the improved water quality in Tampa Bay due to reduced nitrogen loading from point and nonpoint source controls. Taylor et al., (2010) report that large decreases in nutrient loading to Boston Harbor have resulted in reduced TN, TP, and chlorophyll concentrations.

In contrast to lakes and estuaries, there are very few studies or indices to determine how eutrophication specifically impacts stream and river ecosystems (Dodds and Oaks, 2004). Direct effects of eutrophication on rivers include changes in algal biomass which has impacts on ecosystem structure. With excess nutrients, rivers can become hot-spots of biological production and nutrient cycling. Unlike estuaries which are affected by tidal action, rivers can exhibit large ranges in residence times. Increased residence time and nutrient availability are commonly cited as reasons for phytoplankton blooms in river systems. Changes in microbial assemblages which are typically found in the river can be replaced or impacted by nuisance or toxic species. Research by Dodds and others found a positive response of the alga *Cladophora* to increased nitrogen in the Clark Ford River, Montana (Dodds et al., 1997). In a North Carolina river, Mallin and others found excess phytoplankton biomass was the source of liable carbon which contributed to hypoxic conditions (Mallin et al., 2006). These changes in primary producers can lead to changes in food web structure and overall water quality although these changes have not been well documented (Murdock and Dodds, 2007).

Primary production in streams is usually achieved by periphyton (algal growth on hard surfaces), phytoplankton (free-floating algae), or macrophytes. A study by Dodds showed that nutrient availability alone can account for 40% of the variation in periphyton in streams (Dodds, et al., 2002). Periphyton biomass can also be directly impacted by light availability, flooding frequency, and biomass of invertebrate grazers.

Streams play a critical role in nutrient retention and may help control nitrogen export to downstream receiving waters (Peterson et al., 2001). High biological activity in these environments is attributed to the close connection with the landscape, creating areas where nutrients supply is focused. Anthropogenic sources of nutrients can have widespread impacts to the stream ecosystem. Downstream ecosystems, specifically estuaries, are dependent on upstream rivers and streams for the magnitude and timing of nutrient delivery.

Even today there are questions regarding how regional and site-specific factors influence the relationship between nutrient loading and the symptoms of eutrophication (Kelly and Whitton, 1998). Crain (2007) reports on how nutrient limitation shifted across estuarine salinity gradients. Salt and brackish marsh vegetation in Maine was nitrogen limited, while oligohaline marsh vegetation was co-limited by nitrogen and phosphorus.

Freshwater inflow and its influence on residence time is a key factor that influences how a waterbody responds to increases in nutrient inputs (Howarth and Marino, 2006). Geomorphic features (e.g., depth, volume), whether in lakes, streams, rivers, or estuaries, are also important regulators of hydraulic residence time. Hydrologic alterations can also significantly affect residence times. These alterations to hydrologic systems can either increase retention time (in the case of impoundments) or decrease retention time (in channelized streams). Many stream networks in the southeastern coastal plain have been altered to the point that they are now essentially engineered drainage networks. These modifications can alter the timing and duration of storm event discharges and as a result the in-stream nutrient biogeochemistry.
The importance of residence time as a major determinant of the response to nutrient loading in estuaries has been well-documented (Monson et al., 2002; Hagy et al., 2000; Borsuck et al., 2004; and Kemp, 2000). Dettman (2001) modeled nitrogen inputs, exports, and denitrification in 11 estuaries with a wide range of physical characteristics, nitrogen loading rates, and climate. In comparing model results to measured data he determined that the percent of imported nitrogen that is exported or denitrified can be predicted based on water residence time in the estuary. However, unlike estuaries which receive tidal action and regular flushing, rivers can have large ranges of residence times. During low flow periods, long residence times allow prolonged nutrient uptake and this is commonly cited as a reason for phytoplankton blooms in river systems.

Lakes and rivers, including many in Florida, are characterized by their high concentrations of humic and fulvic acids. High humic acid concentrations are often due to terrestrial inputs, including runoff from wetlands and recently deforested areas and the leaching of humic substances from soils by acid precipitation. Humic acids act as a strong chelating agent and bind trace metals that are essential for growth, thus limiting their availability to the phytoplankton. In an experiment in which a phytoplankton community dominated by diatoms and green algae (but lacking cyanobacteria) was compared to an Anabaena-dominated population, humic acids inhibited photosynthesis and N2-fixation in the diatom-dominated community but had no effect on the Anabaena-dominated community (Paerl, 1982). Cyanobacteria appear to have hydroxamate chelators which are capable of sequestering trace metals in growth-limiting concentrations, circumventing humic acid-induced trace metal deficiencies.

Despite ongoing controversies and unsolved issues regarding eutrophication, the knowledge base has supported informed management decisions that have improved water quality in some waterbodies and will facilitate future resource protection efforts. However, the effects of other constituents (PPCPs and PCBs, for example) in the aquatic system are less well-understood. The impacts of many toxic contaminants in aquatic systems have been well-studied, and much laboratory work has been done to identify dose-response effects of many constituents. Also, the ionic behavior of contaminants such as metals is fairly well understood. However, the fate and transport of organic toxins is much less so.

In summary, our knowledge of nutrient dynamics in the environment, and potential effects of excessive nutrient loadings to aquatic systems is well known in relation to some other contaminants. However, numerous questions regarding eutrophication remain to be answered.

### 2.2 Symptoms of Water Quality Impairment

Recognizing the symptoms of eutrophication is key to addressing the problem. The symptoms of water quality decline due to eutrophication are typically increases in chlorophyll \(a\), reductions in water clarity, occurrences of Harmful Algal Blooms (HABs), low DO, loss of desirable submerged aquatic vegetation (SAV), and excessive growth of less desirable SAV forms (Bricker et al., 2008).

Excess N in aquatic ecosystems including streams, lakes, rivers, and estuaries has led to increased primary production and algal biomass (Nixon, 1995). Chlorophyll \(a\), a pigment used in photosynthesis, serves as a measure of biomass (abundance) of phytoplankton and periphyton in surface waters. Using either chemical-optical or direct optical measurement methods allows easy quantification of phytoplankton biomass. Planktonic algae provide a food source for higher...
trophic levels. Chlorophyll $a$ concentrations can also be a significant determinant of water clarity in receiving waters and thereby providing a measure of ecosystem health.

Water clarity is a measure of the amount of sunlight that can penetrate through the water. Clear waters are indicative of a healthy waterbody, although many factors impact water clarity. Excessive nutrient supply can fuel the growth of photosynthesizing algae. High chlorophyll $a$ concentrations associated with high algal biomass can decrease light penetration, decreasing water clarity. Decreased water clarity can negatively impact aquatic ecosystems in many ways. Decreased water clarity can limit the growth and reproduction of (SAV). SAV provide critical habitat for many aquatic organisms. Loss of this habitat can reduce the availability of commercially important finfish and shellfish species. Loss of habitat can cause shifts in trophic structure resulting in less desirable species in both composition and overall abundance. Water clarity in lakes can be affected when the density of particulate matter suspended in the water (seston) is elevated and a color can be imparted to the water. In eutrophic lakes, cyanobacteria or diatoms can exist in large numbers and may produce blue-green or yellowish-brown colors, respectively (Wetzel, 2001).

Most microscopic algal species that inhabit the oceanic, coastal, and freshwaters of the world are considered useful and important as the base of the aquatic food web. Planktonic algae provide a food source for filter-feeding bivalves (oysters, mussels, scallops, clams) and zooplankton (including the larvae of crustaceans and finfish) (Sherr and Sherr, 1988). However, some of these algae have been identified as “harmful algal bloom” species or HABs. A “bloom” occurs when the proliferation of these species results in dense macroscopic concentrations, which can exceed millions of cells per liter (Oliver and Ganf, 2000).

Longer residence times tend to promote slower-growing taxa which include dinoflagellates, cyanobacteria, and HABs (Pinckney et al., 1999). Changes in flow can also impact the community composition with less desirable species such as HABs thriving during times of low flow when algal biomass can accumulate (Bricker et al., 2008). HAB blooms generally have negative, “harmful” effects on the aquatic community through the degradation of water quality and detrimental impacts on higher trophic levels. Blooms may alter food web dynamics due to the poor edibility and food quality of many HAB species to zooplankton and thus reduce carbon flow to upper trophic levels, potentially affecting fisheries (Pinckney, 2006). Such a large input of biomass into an aquatic system may also have impacts on biogeochemical cycling of nitrogen, phosphorus, and carbon (Howarth et al., 1988). The excessive carbon loading and microbial degradation of senescent blooms may result in hypoxia and anoxia in bottom waters. Recreational use of waters containing cyanobacterial blooms is affected due to unpleasant odors and tastes associated with bloom species (Paerl, 1996). In addition, HAB species produce toxic compounds that can have deleterious effects throughout the food web from zooplankton to humans.

The HAB species that are often found in freshwater and coastal systems include cyanobacteria (“blue-green algae”) and dinoflagellates. Cyanobacterial HAB species (cyanoHABs) are prolific in both temperate and tropical environments throughout the world. They are often found in shallow, stratified freshwater lakes (often with significant terrigenous nutrient input). CyanoHABs can be particularly prolific in the shallow, subtropical lakes and rivers in Florida. These aquatic systems experience relatively high light levels and warm water temperatures year-round and are also subject to nutrient over-enrichment from agricultural and urban sources (Canfield et al., 1989). Significant HAB blooms have also been observed in estuarine waters including the several Chesapeake Bay tidal rivers, the Neuse River, NC, and the lower St. Johns River, FL.
Nutrient concentrations play a large role in controlling the presence and bloom formation of cyanobacterial species in a waterbody. Eutrophication is thought to be responsible for much of the expansion of cyanoHABs and cyanobacteria are a very visible and well-documented indicator of this nutrient over-enrichment (Paerl and Zehr, 2000; Paerl et al., 1988). Increasing nutrient inputs from drainage water, atmospheric deposition of nitrogen, and release of nutrients from sediments during anoxic conditions have stimulated cyanobacterial blooms in many lakes, coastal rivers, and estuaries (Paerl et al., 2001; Paerl and Whitall, 1999; and Rizzo and Christian, 1996). The increasing formation and use of land for animal operations, agriculture, industrial production, and urban activities are the major contributors to excessive nutrient loads into waterbodies from drainage, effluent disposal, and storm water.

DO is a critical indicator of the health of the aquatic ecosystem. Cultural eutrophication (nutrient excess leading to overproduction of microalage and associated trophic imbalances) is common in estuaries near human population centers. DO can exhibit extreme diel cycles in eutrophic waters. Diel oxygen dynamics depend upon the relative rates of oxygen production and consumption due to photosynthesis and respiration, respectively. During the day, phytoplankton, periphyton, and higher plants all produce oxygen through photosynthesis. At night, photosynthesis ceases and plant respiration increases the rate of oxygen consumption. Other oxygen consumers include bacteria and fauna. The DO sag typically observed at night is the net result of the cessation of photosynthetic oxygen production and increased oxygen consumption through respiration.

Hypoxia occurs in many lakes, river, streams, estuaries, and coastal regions worldwide (Diaz et al., 2003). Increased anthropogenic nutrient loading has led to frequent and widespread occurrences of low DO in a variety of environments as eutrophication can lead to periodic or long term hypoxia and anoxia in estuarine ecosystems. DO levels are often quite variable due to fluctuations in temperature, basin morphology, overall productivity and respiration, and salinity (estuaries). In the aqueous environment, hypoxia is defined as oxygen tension that is too low for maintenance of homoeostasis. Technically, hypoxia is frequently identified as concentrations of oxygen from 0.1 to 2.0 mg O₂/L, or less than 30% oxygen saturation. In the literature, the typical threshold for hypoxia is 2 mg/L (= 24% saturation at 20°C and 15 psu) (Breitburg, 2002). Total loss of oxygen is termed anoxia and is defined as less than 0.1 mg O₂/L (Diaz et al., 2003). Low DO can have both lethal and sublethal effects on organisms, including shifts in vertical habitat use and trophic interactions. Fishes and invertebrates will attempt to move away from hypoxic conditions and few aquatic animals survive in prolonged exposure to it. Stress caused by reduced oxygen levels is perhaps the most biologically significant symptom because it significantly alters the zooplankton, benthic, and fish community structure in eutrophic waters.

Low DO also affects the oxidation and reduction (redox) reactions which determine the bioavailability of many inorganic compounds, as well as biologically important materials such as 2-2 nitrogen and sulfur. For example, lower redox potential may decrease the release of precipitated metals, which actually may benefit organisms by reducing bioavailability. However, it also may increase the release of precipitated phosphates, encouraging the proliferation of nitrogen-fixing cyanobacteria and potentially altering food resources for fish and invertebrate assemblages (U.S. EPA, 2010a).

Loss of SAV is often an indicator of eutrophication in rivers, lakes, and particularly in estuaries. SAV serve significant ecological functions. They help maintain water clarity by trapping fine sediments and particles with their leaves and stabilizing the estuarine sediments with
their roots. SAV can be very effective at removing dissolved nutrients from water that can enter from runoff from land. The removal of sediment and nutrients help improve water clarity, thereby improving overall ecosystem health. SAV provide habitats for fish and invertebrates, providing a nursery ground for the early life stages of many recreationally and commercially valuable species.

Excessive growth of some SAV species can be a nuisance, particularly in lakes and rivers. Nutrient enrichment tends to stimulate the growth of opportunistic plant species. Resulting increased biomass of opportunistic species and associated species composition alterations leads to impacts throughout all trophic levels in freshwater and marine ecosystems. Protection for many fauna groups from predators, food supply, and spawning areas may show impacts as well as changes in grazing pressure and metabolic pathways. In freshwater springs and rivers, the steady flow of water provides a constant supply of nutrients and is responsible for flushing planktonic organisms downstream while creating a competitive advantage for attached primary producers like SAV (commonly *Vallisneria* spp.) and filamentous species (cyanobacteria and green algae) and their epiphytes. The filamentous benthic cyanobacterium *Lyngbya wollei* is becoming more prevalent in Florida springs and rivers which have been affected by increases in nutrient loads. *Lyngbya wollei* forms thick benthic mats that can grow to significant densities (1-1.5 kg/square meter) (Joyner, 2007). The benthic mats are usually perennial and surface mats are frequently attached to SAV in Florida springs. Surface mats shade benthic mats and can allow as little as 2% of incident irradiance to pass through the mats (Joyner, 2007).

In the United States, one of the most common invasive SAV species is a hydrilla (*Hydrilla verticillata*). Native to central and south Asia, it was introduced to Florida in the 1950s and is now well established in the southeastern United States. Hydrilla forms dense submerged mats of vegetation which interfere with recreation and destroys fish and wildlife habitat. In the Chesapeake Bay, improved water quality through the reduction of nitrogen and phosphorus over an 18-year field study showed a marked decrease in hydrilla (Ruhl and Rybickib, 2010). In 1990, more than 80% of the total SAV was hydrilla; in 2007, hydrilla comprised only 20% of the total SAV (Ruhl and Rybickib, 2010).

Understanding the impacts of eutrophication on the structure and function of aquatic ecosystems continues to be a research goal for scientists and managers worldwide (Paerl et al., 2006). Eutrophication has resulted in documented cases of reduced biodiversity, habitat degradation and food web alterations (Nixon, 1995; Rabalais et al., 2009; Paerl, 2006; Bricker et al., 2008).

### 2.3 Identification of Impaired Waters: The Florida Impaired Waters Rule

The management of eutrophication should depend upon understanding how rivers, lakes, and estuaries are coupled to nutrient export from their watersheds (Howarth et al., 2002) and the factors that affect the ultimate responses within the receiving waterbody. When a receiving waterbody has been identified as impaired for nutrients it generally has shown one or more symptoms of eutrophication – low DO, elevated chlorophyll concentrations, or excessive growth of aquatic macrophytes. In Florida, water quality impairment is determined through protocols defined in the Impaired Waters Rule (IWR). Florida Administrative Code (F.A.C.) Chapter 62-303.351, .352, and .353 contain the thresholds for identifying streams, lakes, and marine waters on a list of potentially impaired waterbodies.
In the late 1990s, after court action brought by environmental groups, U.S. EPA was charged with establishing TMDLs for waters on Florida’s 1998 Clean Water Act Section 303(d) list, if the State failed to do so. Concerns over flaws in the 1998 Section 303(d) list prompted the Florida Legislature to enact the Watershed Restoration Act which authorized the Florida Department of Environmental Protection (FDEP) to implement a methodology for listing waters as impaired and setting TMDLs (Norgart, 2004).

There are two key terms used in the IWR that set out the basic framework of the impairment listing methodology. These are “planning list” and “verified list.” Planning list is defined as “the list of surface waters or segments for which assessments will be conducted to evaluate whether the water is impaired and a TMDL is needed” (Norgart, 2004). Essentially, the planning list is a screening mechanism. This type of process helps to weed out water bodies that are not in immediate need of TMDLs even though at first glance they appear to be impaired.

Waters on this planning list are assessed to determine whether the waterbody is impaired and whether the impairment is due to pollutant discharges. Waters are put on the verified list as being impaired if they meet the requirements for the planning list and the additional requirements in the IWR (Aquatic Life-Based Water Quality Criteria Assessment). A waterbody that fails to meet the minimum criteria for surface waters established in Rule 62-302.500, F.A.C.; any of its designated uses; or applicable water quality criteria as described here is classified as impaired (FDEP, 1999). The resultant verified list of impaired waters, which is the list of waters for which TMDLs will be developed by FDEP must then be adopted by Secretarial Order. Once adopted, the list is submitted to the U.S. EPA pursuant the CWA.

The thresholds used to identify impaired water bodies for each waterbody type are summarized below, and a rationale for selecting for each threshold is provided:

**Lakes**
Florida has developed an impairment threshold for nutrients in lakes based on either an exceedance of a TSI of 40 or an increasing trend in the Florida Trophic State Index (TSI). Typically, a TSI incorporates Secchi disc depth and concentrations of chlorophyll \( a \) and total phosphorus to indicate a lake’s trophic status (Carlson, 1977). In contrast, the TSI used in Florida uses total nitrogen as a third indicator, rather than Secchi disc depth. Secchi depth can provide misleading information when used in naturally dark water lakes and estuaries in the state. The Florida-specific TSI is based on a regression analysis of data from over 300 Florida lakes, and was developed so that a chlorophyll \( a \) concentration of 20 μg/L corresponds to a TSI of 60. This threshold was chosen based on the assumption that phytoplankton assemblages often evolve to communities of mainly blue-green algae when chlorophyll \( a \) concentrations exceed 20 μg/L.

Because color can affect the relationship between nutrient conditions and chlorophyll, the TSI criterion that defines an impaired lake depends upon the ambient color in that lake. In clear lakes (color less than 40 PtCo units) the critical TSI is 40. In highly colored lakes, (color greater than 40 PtCo units) the critical TSI is 60.

Florida lakes possess great diversity and range of productivity, and some lakes have natural TSI values exceeding 60. Paleolimnological data can be used to estimate a pre-development TSI and establish a site-specific criterion in those cases.

Also, lakes or lake segments with mean annual TSIs that are trending toward undesirable conditions, i.e., those that have increased over the assigned assessment period, or with an the
mean annual TSI that has increased by more than 10 units over historical values can also be defined as impaired.

Lakes can also be defined as impaired when their DO conditions do not meet State standards. The following are the DO standards (Chapter 62-302,530(30), F.A.C.). for lakes:

- Class I water (potable supply) – DO shall not be less than 5.0 mg/L
- Class III freshwater (recreation, propagation and maintenance of a healthy, well-balanced population of fish and wildlife) – DO shall not be less than 5.0 mg/L
- Class IV water (agricultural water supplies) shall not average less than 4.0 mg/L in a 24-hour period and shall never be less than 3.0 mg/L
- Class V water (navigation, utility, and industrial use) shall not be less than 0.3 mg/L 50% of the time on an annual basis for flows greater than or equal to 250 cubic feet per second and shall never be less than 0.1 mg/L

Normal daily and seasonal fluctuations above these levels shall be maintained for all classes.

**Streams**

Florida uses a narrative criterion and two chlorophyll $a$ concentration thresholds to assess nutrient impairment in streams. The narrative criterion is “algal mats are present in sufficient quantities to pose a nuisance or hinder reproduction of a threatened or endangered species”. So, if algal mats cause a nuisance or hinder threatened and endangered species, the stream will be listed as impaired. The narrative approach was used because the U.S. EPA numeric threshold was thought to be too high, and an alternative value could not be agreed upon (Joyner, 2008). Narrative criteria are sometimes referred to as “free from” criteria, as they often state that the waterbody must be “free from” something (e.g., nuisance algal growths). The narrative criterion can also be used to list a stream as impaired if other information indicates an imbalance (e.g., excessive macrophyte growth, decreases in distribution of seagrasses or other SAV, changes in algal species richness, and excessive diel DO swings) (Joyner, 2008).

The first chlorophyll $a$ threshold (impaired if the annual mean is greater than 20 μg/L) is more protective for streams with naturally high chlorophyll $a$ concentrations. To establish this limit, FDEP evaluated historical chlorophyll $a$ data from state streams and found that the 90th percentile for state streams was approximately 23 μg/L. The slightly lower threshold of 20 μg/L approximates the 95th percentile and was selected to provide additional protection.

The second chlorophyll $a$ threshold for streams states that mean chlorophyll $a$ values may not increase more than 50% over historical values for two or more consecutive years. Historical concentrations are based on the lowest consecutive five-year average for the period of record. This threshold is conservative for streams with naturally low chlorophyll $a$ levels, and is very protective of oligotrophic streams. As with lakes, a change in primary production from an historical or baseline condition is a valid means to identify a change in the balance of stream condition. This relative comparison protects lakes when application of general threshold values is not appropriate.

Like lakes, streams can also be defined as impaired when their DO conditions do not meet State standards. The DO standards for streams are identical to those defined above for lakes.
Estuaries and Open Coastal Waters

The threshold previously used for estuaries was based on the lake TSI and was determined to be inappropriate for marine waters (Joyner, 2008). The adopted impairment thresholds for estuaries and open coastal waters are similar to those for streams, although there is no narrative provision. The upper bound for estuaries is an annual mean chlorophyll $a$ concentration of 11 μg/L, rather than 20 μg/L. This level is based on an evaluation conducted under contract to FDEP that found that a median annual chlorophyll $a$ concentration of 11 μg/L reflected the breakpoint for highly eutrophic estuaries. FDEP established the threshold as a mean rather a median, to make the threshold more environmentally protective. A second criterion states that the annual mean chlorophyll $a$ may not have increased by over 50% from historical values, the same as for streams.

Like lakes and streams, marine waterbodies can also be defined as impaired when their DO conditions do not meet State standards. The DO standards for marine waterbodies are identical to those defined above for lakes with the exception of Class III waters where DO shall not average less than 5.0 mg/L in a 24-hour period and shall never be less than 4.0 mg/L.

Having identified an impaired waterbody based on the criteria and thresholds defined above, the first logical question is whether there is a linkage between nutrients and the observed impairment (This linkage was included in the Project Framework presented in Figure 1-2). The answer is often assumed to be “yes” which can lead to erroneous conclusions that would require nutrient load reductions. Rather, this question should be answered by either analysis of the available data or, when more definitive answers are required, experimental manipulation of nutrient conditions and assessment of responses to these manipulations.

FDEP develops a list of verified impaired waterbodies annually. This list includes information regarding the location of the waterbody, the reason for impairment (i.e., nutrient, DO, etc.), and identification of the presumed specific causative agent. Table 2-1 presents the numbers of lakes, streams, and marine waterbodies that were deemed nutrient impaired and whether nitrogen, phosphorus, or both were the limiting nutrient. A waterbody that is nitrogen limited presumes that the algae are using as much nitrogen as is available. So if the amount of nitrogen is reduced, then the amount of algae and chlorophyll in the water can be expected to be reduced. The information provided in Table 2-1 indicates that nitrogen was implicated as a causative agent in about half of the lake impairments and about three-quarters of the nutrient-impaired streams and marine waterbodies.
Table 2-1. Number of Nutrient Impaired Waterbodies in Florida Based on the Most Recent Verified List.

<table>
<thead>
<tr>
<th>Waterbody</th>
<th>Total Nutrient Impaired</th>
<th>N Limited</th>
<th>P Limited</th>
<th>N+P Limited</th>
<th>No Limiting Nutrient Listed</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lakes</td>
<td>173</td>
<td>19</td>
<td>78</td>
<td>66</td>
<td>17</td>
</tr>
<tr>
<td>Streams</td>
<td>129</td>
<td>45</td>
<td>9</td>
<td>46</td>
<td>28</td>
</tr>
<tr>
<td>Marine</td>
<td>99</td>
<td>55</td>
<td>1</td>
<td>20</td>
<td>18</td>
</tr>
<tr>
<td>Total</td>
<td>401</td>
<td>119</td>
<td>88</td>
<td>132</td>
<td>63</td>
</tr>
</tbody>
</table>

Table 2-2 presents the numbers of lakes, streams, and marine waterbodies that were deemed DO impaired and whether nitrogen or phosphorus or both were the limiting nutrient. The information in this table indicates that nitrogen was implicated as a causative agent in 51% of the impaired lakes and 65% of the impaired streams and marine waters. BOD and “other” causative agents were also identified in a number of the DO impaired waterbodies. The assumption of nutrient enrichment as a cause of DO impairments has been recently questioned in a number of streams identified as DO impaired in the Tampa Bay watershed (Janicki Environmental, 2010b).

Table 2-2. Number of DO Impaired Waterbodies in Florida Based on the Most Recent Verified List.

<table>
<thead>
<tr>
<th>Waterbody</th>
<th>Total DO Impaired</th>
<th>N Limited</th>
<th>P Limited</th>
<th>N+P Limited</th>
<th>No Limiting Nutrient Listed</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lakes</td>
<td>35</td>
<td>11</td>
<td>5</td>
<td>7</td>
<td>12</td>
</tr>
<tr>
<td>Streams</td>
<td>198</td>
<td>82</td>
<td>29</td>
<td>47</td>
<td>40</td>
</tr>
<tr>
<td>Marine</td>
<td>78</td>
<td>33</td>
<td>8</td>
<td>17</td>
<td>10</td>
</tr>
<tr>
<td>Total</td>
<td>311</td>
<td>126</td>
<td>42</td>
<td>71</td>
<td>62</td>
</tr>
</tbody>
</table>

There are numerous examples of waterbodies where factors other than nutrients have been shown to affect DO and chlorophyll levels. Hydrologic alterations, in particular, can significantly affect chlorophyll and DO without excessive nutrient loading. Data are presented below for two Florida rivers – the Hillsborough River in west central Florida on Tampa Bay, and the Caloosahatchee River in southwest Florida – where flows are largely controlled. The City of Tampa Municipal Reservoir was created when a dam was constructed across the Hillsborough River almost 10 miles upstream of the river mouth. The Franklin Lock was constructed in the Caloosahatchee River, which flows into the Gulf of Mexico, about 27 miles upstream of the river mouth to control flows out of Lake Okeechobee. Water quality data have been collected at several sites on the tidal portion of Hillsborough River between the dam and the bay. Figure 2-1 shows the relationship between flow and bottom DO concentration at sites in the upper portion of the river below the dam. The horizontal line at 2.5 mg/L represents a critical minimum DO threshold that was determined through the analysis of site-specific data. The graphic shows that the probability of a bottom DO below the threshold decreases significantly with increasing flow. When flow is above 80 cubic feet per second (cfs) the probability of a DO less than 2.5 mg/L is effectively zero. Chlorophyll was also shown to be significantly related to flow over the dam. Figure 2-2 shows the chlorophyll downstream of the dam very rarely exceeds the state threshold of 20 μg/L when flows exceed 70 cfs.
Figure 2-1. Bottom DO in the Lower Hillsborough River as a Function of Flow Past the Dam. SWFWMD, 2006.

Figure 2-2. Chlorophyll in the Lower Hillsborough River as a Function of Flow Past the Dam. SWFWMD, 2006.
2.4 Identification of Knowledge Gaps

There are significant knowledge gaps in the process of indentifying impaired waters. While the Florida IWR is commendable, too often both chlorophyll and DO impairments are attributed to excessive nutrients by default. Such ill-conceived conclusions lead to TMDL development that can seriously affect the stakeholders associated with the impaired water by requiring nutrient controls that will have little effect on the water quality. Incomplete knowledge also leads to the determination that both nitrogen and phosphorus are causative which in turn requires stakeholders to address the loading of both nutrients.

Development of TMDLs for nutrients without first discerning the dose-response relationships between nutrients and chlorophyll and DO is not justified. Evaluating existing data is one method by which dose-response relationships may be developed. Lacking sufficient data for such an analysis, additional data collection targeted at providing the necessary data should be considered. Data should be available over appropriate temporal and spatial scales so that a truly representative data domain exists for a system prior to attempting relationship development. Another option for defining the dose-response relationships is by developing and implementing an appropriate mechanistic model, which can be calibrated using existing data, and then used to evaluate the system responses to modified nutrient loads as a means of establishing TMDLs. The role of hydrologic effects on residence time and potential stratification should also be considered because higher loadings during high freshwater inflow periods may not result in increased chlorophyll concentrations, but may induce vertical stratification solely as a function of hydrologic load, which can lead to reduced DO concentrations in bottom waters.

In Florida, many have expressed concern regarding the DO standards in both fresh and marine (estuarine) waterbodies. FDEP has initiated research to examine the validity of the DO standards in streams. More research is clearly needed including extension into estuaries and tidal creeks. As has been achieved in the Chesapeake Bay, consideration should be given to matching the DO standards to the various habitats and the biota that are commonly found in them, and considering the time varying (seasonal, diurnal) responses of DO to both nutrient and hydrologic loadings. The result can be a more effective management of these vital ecosystem components.
CHAPTER 3.0

NUTRIENT LINKAGE:
IS NITROGEN OR PHOSPHORUS CAUSATIVE?

Biological populations are dependent upon environmental conditions suitable for growth and reproduction. For plants, these conditions include availability of light and nutrients, among others. While all other conditions may be favorable, growth can be limited by nutrient availability. Algal growth is limited by the nutrient that is in least supply. This is known as Liebig’s Law of the Minimum, first developed in the 19th century.

In aquatic ecosystems, the nutrients of importance are most often nitrogen and phosphorus. Given sufficient light, autotrophic (photosynthetic) growth is frequently limited at any given time by the availability of one of these nutrients. As discussed above, increased supplies of nitrogen and phosphorus to aquatic systems typically lead to eutrophication and a decline in desirable water quality conditions.

Three major factors control whether nitrogen or phosphorus is more likely to be limiting (NRC, 2000):
♦ The N:P ratio in external and internal nutrient inputs.
♦ The preferential loss from the photic zone of nitrogen or phosphorus due to biogeochemical processes such as denitrification, sedimentation, or absorption of phosphorus.
♦ The amount of nitrogen-fixation.

The limiting nutrient in aquatic ecosystems varies depending upon the specific system being examined, but there are some generalizations that are accepted for different types of systems. Freshwater systems, including lakes and streams, are generally considered phosphorus limited, although they may at times be nitrogen limited (Smith, 1984; Hecky and Kilham, 1988; Howarth, 1988a; and Elser et al., 1990). Marine systems, including estuaries, are generally considered nitrogen limited, although there may be times and locations when phosphorus limitation may occur (Thomas, 1970a,b; Ryther and Dunstan, 1971; Boynton et al., 1982; Smith, 1984; Howarth, 1988, 2008; Howarth et al., 1988a,b; Nixon et al., 1996; Howarth and Marino, 2006; Chapra, 1997; National Research Council, 2000; Conley, 2000; Conley et al., 2009; and Malone et al., 1996).

Evidence for phosphorus limitation in lakes has been summarized previously and includes results from whole lake experiments, long-term monitoring, and mesocosm experiments (Correll, 1999; Conley, 2000; Schindler, 1974, 1975, 1977; Edmondson, 1970; Edmondson and Lehman, 1981; Boyce et al., 1987; Powers et al., 1972; Schelske and Stoermer, 1972). Similarly, phosphorus limitation in streams and rivers is generally accepted in part based on the results of phosphorus enrichment studies in British Columbia, Alaska, and Tennessee (Correll, 1999; Stockner and Shortreed, 1978; Peterson et al., 1985; and Elwood et al., 1981).
Since nitrogen is considered the most likely limiting nutrient in estuarine systems, it has been identified as the primary nutrient of concern in estuarine ecosystems nationwide (see review in National Research Council (NRC), 1993). As noted in Correll (1999), however, since estuaries are part of the transition zone between the open ocean and the phosphorus supplied from the land, it is possible that both phosphorus and nitrogen may be limiting in estuaries depending on the time of year, location in the estuary, and nutrient supplies. Ryther and Dunstan (1971) noted the change from phosphorus limitation in freshwaters to nitrogen limitation in near-shore marine waters, although Hecky and Kilham (1988) indicated that the extent and severity of marine nitrogen limitation has not been conclusively determined. Other studies report estuarine phosphorus limitation in the spring and nitrogen limitation in the summer and fall (Fisher et al., 1992; Lee et al., 1996). Depending upon the relative rates of nitrogen and phosphorus supply, the limitation has been found to shift between nitrogen and phosphorus in coastal lagoons in the northeastern U.S. (Taylor et al., 1995). Documentation has also shown that residence times play a significant role in determining the estuarine responses to nutrient loads (Monsen et al., 2002; Hagy et al., 2000; Borsuck et al., 2004; Boynton and Kemp, 2008).

3.1 Methods to Determine Limitation

There are two general methods that have been used to define which nutrient is limiting in a waterbody. They include a method that depends upon ambient water quality data collected over a wide range of environmental conditions. The second type of method involves experimental manipulation of nutrient conditions, either in the lab or in situ. Both methods are discussed herein.

3.1.1 Evaluation of Ambient Water Quality Data

This method depends on a metric typically used to evaluate nutrient limitation: the nitrogen to phosphorus ratio (N:P ratio). The N:P ratio indicative of balanced conditions is typically taken to be 16:1 (molar), based on the work of Redfield (1934, 1958). This N:P ratio was based on the elemental composition of algae, under both laboratory and natural conditions. When N:P ratios are greater than 16:1 in a system, this is indicative of phosphorus limitation, although site-specific conditions may alter this threshold. However, there may be considerable variation in this ratio within an algal culture, dependent upon cell division status, light conditions, and precedent conditions (Correll, 1999; Correll and Tolbert, 1962; Terry et al., 1985).

Molar ratios of N:P are easily determined from water quality monitoring data. In freshwater systems the N:P ratio is usually higher than 16:1, indicating that phosphorus is usually most limiting to primary production in freshwater ecosystems (Schindler, 1977; Elser et al., 2007). This becomes evident by examining the concentrations of the forms of nitrogen and phosphorus that are available for algal uptake. Little if any dissolved inorganic phosphorus (DIP) is generally found in relatively productive fresh waters while measurable concentrations of dissolved inorganic nitrogen (DIN) remain.
Most marine systems are nitrogen limited because there are relatively low concentrations of dissolved inorganic nitrogen compared to dissolved phosphorus. Since Redfield’s observations were published, research has shown that ratios from 10:1 to 20:1 for N:P are typically found in estuaries (Parsons et al., 1984). Howarth (1988) observed that the correlation between nitrogen and the primary production was better for estuaries that received nutrient concentrations with smaller N:P ratios than the one studied by Redfield. It has been concluded that estuaries receiving nutrient concentrations with high N:P ratios were limited by phosphorus and only those with low ratios are limited by nitrogen (Boynton et al., 1982). Boyton et al. (1982) and Howarth (1988) compiled data on the ratio of inorganic nitrogen to phosphorus in a variety of estuaries. Of the 27 estuaries studied, 22 had N:P ratios below the Redfield ratio and may have been nitrogen limited (Howarth, 1988). Because phytoplankton can assimilate some organic nutrient forms and all forms are relatively labile, it is useful to examine the ratio of total nutrient concentrations (TN:TP).

Reductions of nutrient levels in a waterbody will usually result in reduction in algal growth. Reducing phosphorus, however, will have no effect unless the reduction results in an N:P ratio greater than 16:1. Phosphorus would then become the limiting nutrient. In contrast, a reduction of nitrogen concentrations will result in a reduction of primary productivity when the ratio is less than 16:1. There are exceptions to this general rule. Some coastal areas are phosphorus limited due to strict phosphorus control measures or natural conditions while some freshwaters are nitrogen limited due to natural sources of phosphorus.

3.1.2 Experimental Methods

Experimental manipulation of nutrient conditions, either in the lab or in situ typically involve nitrogen and phosphorus additions to either a test alga or a phytoplankton assemblage singularly and in combination. The responses to the additions determine the limiting nutrient. If growth is found only during nitrogen addition, nitrogen-limitation is indicated. Conversely, if growth is found only during phosphorus addition, phosphorus-limitation is indicated. Bioassays are especially beneficial because they can be performed under tightly controlled conditions, and have proven very useful (Priscu, 1995; Rudek et al., 1991; Elser and Kimmel, 1986).

In situ methods have included:

♦ Limnocorals or bags in which nutrient additions are made and resultant growth responses are measured (Shapiro, 1980; Lynch and Shapiro, 1981; Havens and DeCosta, 1986; Perez et al., 1994).

♦ Mesocosm studies in which water is collected and placed in separate containers or enclosures for application of separate treatments over multiple day time scales (Oviatt et al., 1986; Taylor et al., 1995).

♦ Whole-lake studies that are performed on entire lakes or portions of lakes separated by a curtain (Schindler, 1974, 1975).
3.2 Confounding Factors

Determination of the limiting nutrient based solely on N:P ratios estimated from water quality data or from experimental uptake rates should be performed with some idea of potentially confounding effects. Algal cell interior N:P ratios and uptake rates may vary due to:

♦ Cell division status (Correll and Tolbert, 1962)
♦ Light intensity or light quality (Wynne and Rhee, 1986)
♦ Light and temperature (Jahnke et al., 1986)
♦ P deprivation and then subsequent availability (Sicko-Goad and Jensen, 1976)

It has been found that the limiting nutrient in freshwaters, which are typically considered phosphorus limited, may vary seasonally, with summer N limitation occurring when photic zone inorganic nutrients are low (Elser et al., 1990). It has also been demonstrated that some estuaries show seasonal shifts in limitation (D’Elia et al., 1986; McComb et al., 1981; Conley, 2000). The best available information should be used to determine the limiting nutrient of a system before management decisions are made with the objective of improved water quality via nutrient load control.

3.3 Nutrient Limitation: Florida Applications

In Florida, as part of its TMDL process, the FDEP attempts to identify the limiting nutrient(s) in impaired waterbodies. The TMDL for a specific waterbody specifies the maximum amount of the limiting nutrient that may enter the waterbody, with this limitation being defined with the aim of improving water quality. If the N:P ratio does not clearly suggest the limiting nutrient, TMDLs for both nitrogen and phosphorus are typically defined. The primary method for determining the limiting nutrient employed by the FDEP is the use of existing water quality data to derive an *in situ* N:P ratio, but more complicated methods, including field tests and laboratory algal growth potential bioassays, may be employed.

Receiving waters with ratios less than 10:1 (molar) are considered nitrogen limited, ratios of greater than 30:1 (molar) indicate phosphorus limitation, and ratios of 10-30:1 (molar) indicate co-limitation (FDEP, 2002).

The City of Tampa Bay Study Group routinely performed bioassays throughout Tampa Bay during 1993-2009 as part of the evaluation of the effects of discharge from the H.F. Curren wastewater treatment facility (Johansson, 2009). Bioassays in late winter and late summer have been conducted for most of this period. This work was in support of the phosphorus waiver held by the City, as regulations require all point sources that discharge directly to Tampa Bay not exceed 1 mg/L. The results of these 152 bioassays have supported that nitrogen is the primary limiting nutrient in the bay, with no results showing phosphorus as the limiting nutrient in the bay, including within that portion of the bay which receives the wastewater effluent. Therefore, the discharge phosphorus limitation is not required for this wastewater treatment facility. This is an excellent example of determining the linkage between nutrients and water quality, and applying that knowledge to development of rational nutrient controls.
3.4 Identification of Knowledge Gaps

Determining a causative agent for eutrophication impairment is often not straightforward. Nutrient enrichment and the resulting problems may be widespread and are frequently manifested at a location remote from the sources. In addition, symptoms of impairment may not show themselves for some time after significant inputs to the system have occurred. Finally, when control measures are applied to an impaired waterbody there may be a non-linear recovery. That is, there may not be a direct one-to-one relationship between reduced loadings and improved water quality (Childers et al., 2006). It is recognized that a significant change in equilibrium may be needed to begin the recovery process. However, the specific degree of change that is necessary is not widely known.

All of the above issues were addressed during the development of a nitrogen management strategy for Florida’s Tampa Bay estuary. The City of Tampa municipal wastewater plant had historically discharged poorly treated effluent directly into Hillsborough Bay, a segment of Tampa Bay, and was the largest single source of nitrogen loading to the bay. In response to these high loading rates the bay was highly eutrophic. The plant was upgraded to advanced treatment in 1978 and nutrient loadings to the bay were greatly reduced. Water quality did significantly improve, but the chlorophyll response to the reduced loading was not immediate, demonstrating that responses to control measures may be delayed as the system adjusts to new conditions (Greening and Janicki, 2006; Taylor et al., 2010).

Knowledge gaps have led to one of the critical errors often made when faced with the identification of an impaired waterbody – the presumption that one or more of the primary nutrients is causative. Clearly, physical changes in either the receiving waterbody or its watershed can influence the waterbody’s responses to nutrient enrichment. Hydrologic alterations can also affect residence times in the receiving waterbody and in turn affect the level of primary production.

The unsupported implication of nutrient causality in the occurrence of DO impairments in surface waters is also common. The assumption of nutrient effects is typically the result of a knowledge gap that limits our understanding of the entire system.

N:P ratios can be useful in addressing nutrient controls. When very costly controls are needed to address nutrient loading, the increased certainty afforded by the application of either laboratory or in situ methods to identify the limiting nutrient should be considered. As discussed above, the City of Tampa has been successful in demonstrating that nitrogen is the sole limiting nutrient in Tampa Bay and has thus precluded the requirement for costly and unnecessary phosphorus controls.

The N:P ratios are a commonly used approach to identify nutrient limitation. However, they can vary spatially and most notably temporally within a single waterbody. Of course, the analysis of water quality data is central to determining nutrient limitation. The specific sampling design that defines the spatial and temporal resolution of the data collection can greatly affect the resultant N:P ratios and may bias the assessment as to whether nitrogen or phosphorus is limiting. Any sampling design that lends a bias to the relative N and P concentrations will lead to unrepresentative results.

Nutrient addition experiments are more complex than in situ sampling but can be more definitive than relying solely on the N:P approach. Typically, additions of nitrogen and phospho-
rus alone and in combination to surface water samples can provide conclusive results. Criticisms of this methodology typically revolve around whether the algal taxa exposed to the nutrient additions come from laboratory strains, such as *Selenastrum capricornutum*, or are taxa commonly found in the waterbody of interest.

The lack of limitation by either nitrogen or phosphorus can lead to unnecessary controls. Nutrient load reductions to a waterbody that is light-limited due to colored dissolved organic matter (CDOM) may show no change in resultant water quality. Hydrologic alterations affect residence time in a waterbody and can also confound the relationship between nutrient loading and water quality conditions.
CHAPTER 4.0

NUTRIENT SOURCES:
CAN THE MAJOR NITROGEN SOURCES AND LOADS BE QUANTIFIED?

This chapter discusses how to determine which nutrient sources are most important with respect to managing the effects of eutrophication, and presents methods that can be used to quantify the corresponding load inputs to the watershed. The specific focus of this chapter is on nitrogen, but a similar approach could apply to phosphorus. Once in a watershed, nitrogen can undergo any number of transformations and be transported via numerous linkages and pathways prior to entering the receiving water. Alternatively, nitrogen can become sequestered and stored in the watershed, or exported from the watershed and never reach the receiving water.

This chapter specifically summarizes the following:
♦ Identification of nitrogen sources to the watershed
♦ Quantification of the nitrogen inputs to the watershed from each source
♦ Quantification of nitrogen loading to the receiving waterbody
♦ Ranking of nitrogen loads to the waterbody
♦ Determination of waterbody carrying capacity
♦ Determination of need for nitrogen controls
♦ Identification of knowledge gaps

4.1 What Are the Sources of Nitrogen Inputs to the Watershed?

Nitrogen is one of the elements that are essential for both plant and animal life within the biosphere. As such, nitrogen differs from many other pollutants. There clearly is some amount of nitrogen that is needed to support the productivity of both terrestrial and aquatic ecosystems. However, as discussed above, excessive rates of supply of nitrogen (along with phosphorus) can result in significant degradation of aquatic ecosystems in particular.

♦ Understanding the various nitrogen sources in a watershed depends upon knowledge of the nitrogen cycle within the environment. Figure 4-1 presents a graphical depiction of the key linkages between different nitrogen forms within the environment and the processes that provide these linkages. Following the law of the conservation of mass, nitrogen is an element that cannot be created or destroyed and as such the mass of nitrogen on earth remains the same, but it can be transformed from one form to another.
Overall, most environmental nitrogen is in the gaseous form (N\textsubscript{2}). Nitrogen in this form is essentially biologically unavailable to most life forms. It is only after the N\textsubscript{2} is transformed via nitrogen fixation to reactive nitrogen (i.e., biologically available nitrogen) that can be utilized by plants. The products of fixation include ammonium-nitrogen (NH\textsubscript{4}\textsuperscript{+}), which can subsequently transform to nitrate-nitrogen (NO\textsubscript{3}\textsuperscript{-}). Biological nitrogen fixation (BNF) is carried out by a variety of organisms, including blue-green algae, the soil bacterium \textit{Azobacter}, legume plants (e.g., beans, peas, and clover), and the symbiotic bacterium \textit{Rhizobium}. Biological nitrogen fixation can be enhanced via cultivation in agricultural settings. Additionally, nitrogen can be fixed by some natural inorganic processes such as lightning, solar radiation, and meteorite trails (Sauer et al., 2008). Conversion of elemental N\textsubscript{2} to reactive nitrogen can also be achieved by the Haber-Bosch method. This industrial method, used to produce nitrogen fertilizers, utilizes high temperature, elevated pressure, a metal catalyst, and gas circulation to synthesize reactive ammonia from gaseous N\textsubscript{2}. The Haber-Bosch process was developed in the early 1900s and began to be used extensively for fertilizer production in the 1950s.

When plants and animals die or are eaten, cellular organic nitrogen is decomposed by bacteria and released as ammonium (NH\textsubscript{4}\textsuperscript{+}). This decay process is termed ammonification. Ammonia, can also be transformed (oxidized) by other bacteria (\textit{Nitrosomonas}) to form nitrite (NO\textsubscript{2}\textsuperscript{-}), which is then readily transformed by the bacterium (\textit{Nitrobacter}) to form nitrate (NO\textsubscript{3}\textsuperscript{-}). This two-step conversion of ammonium to nitrite and nitrate by autotrophic nitrifying bacteria is termed nitrification. Nitrate can then be taken up by plants for nutrition but it generally must be biologically reduced to ammonia before incorporation. As such, ammonia will be utilized first when it is available. The assimilative uptake of nitrate or ammonia produces organic nitrogen (Wetzel, 2001). Nitrification only occurs in the presence of oxygen in circulating water or surf-
cial soils. Plant uptake serves as a sink for nitrogen and removes it from active cycling, at least temporarily.

Most nitrogen species are very mobile in the environment and are subject to a variety of biological, chemical, and physical processes. Quantifying these effects is challenging because of the wide temporal and spatial variability at which these processes occur, and has been the subject of numerous research studies, many of which have been summarized by Sauer et al., (2008).

There are some forms of nitrogen that are most common in water. Nitrate (NO$_3^-$) is highly soluble in water and is stable over a wide range of conditions. It is easily transported in streams and groundwater. Nitrite (NO$_2^-$) in water is generally quickly converted to nitrate by bacteria or spontaneous (chemical) oxidation. The behavior of ammonia and nitrate in sediments and soils is very different. Ammonia will sorb (stick) to clayey sediment and soil particles and remain fairly immobile. Nitrate, on the other hand, is very labile and travels freely through surface water, groundwater, and sediment, often reaching surface waters. Thus, groundwater contamination from nitrogen is usually from the nitrate form as ammonia stays near its point of entry into the subsurface environment or is biologically converted to nitrate.

Completing the nitrogen cycle, nitrates can be returned to the atmosphere as gaseous nitrogen by denitrification. Denitrification is performed under anaerobic DO conditions by organisms such as fungi and the bacterium *Pseudomonas* which reduce nitrates as an alternative electron acceptor when oxygen is not present. The conversion of nitrate to nitrogen gas through the denitrification process can significantly reduce the mass of nitrogen that is exported from a watershed. Boynton (2010) estimated that tidal marshes in a reach of the Patuxent River in Maryland remove 48% of nitrogen inputs through immobilization and denitrification.

The production of reactive nitrogen has increased very significantly over the last 50 years. As a result, the natural nitrogen cycle has been upset by several culturally-induced activities. Figure 4-2 illustrates estimated global trends in the relative amounts of reactive nitrogen from various sources (Galloway et al., 2004; Nielsen, R. 2005). The most significant development affecting the earth’s nitrogen balance is the manufacture of synthetic fertilizer using the Haber-Bosch method. The invention of this process has had immense influence on our society and the environment (Nielsen, 2005; Galloway et al., 2008; Bittman and Mikkelsen, 2009). The use of synthetic fertilizer began to grow exponentially after 1960 and today it is the single largest source of biologically active nitrogen. The availability of inexpensive, easy-to-use fertilizer has been a great benefit to global food production.
Combustion of fossil fuels for electrical generation and motorized vehicles has also increased significantly since the mid-20th century, as has reactive nitrogen generated from BNF from crop cultivation. Only BNF from natural vegetation has decreased over time, resulting from land use changes and urbanization. Model extrapolations predict even greater reactive nitrogen production over the next 50 years (Galloway, et al., 2004; Nielsen, 2005). These trends have broad implications for the health of our surface waters (U.S. EPA, 2009a).

The increased production of reactive nitrogen has made it very apparent that too much of a good thing can result in the eutrophication and alteration of the structure and function of surface water ecosystems (Nixon and Buckley, 2007). Therefore, nitrogen control strategies must necessarily focus on the potential reduction of reactive nitrogen as well as the rates of supply (i.e., loading) of these biologically-available nitrogen forms.

### 4.2 Quantify Nitrogen Inputs to Watershed and Linkages to Receiving Waterbody

The first step in quantifying nitrogen loads that ultimately reach a receiving water is to identify all significant nitrogen inputs to the watershed. The ultimate nitrogen sources to a waterbody and its watershed are:

- Natural sources – e.g., soil emissions, oxidation of N₂ by lightning
- Human waste
- Industrial waste
- Fertilizer and animal waste
- Combustion products of fossil fuels
- Internal loads from sediment and biota
Nitrogen from these sources is delivered to the watershed via atmospheric deposition, nonpoint source discharges, or point source discharges (Figure 4-3). Linkages or delivery routes to the receiving water include direct atmospheric deposition, channelized flow, groundwater seepage, internals loads from sediment and biota, and direct discharge point sources. In addition, sediment interactions can result in the sediments acting as a nitrogen source or a sink. The following provides a discussion of the nitrogen sources that are delivered by these modes of delivery and how quantifying these inputs can be accomplished.

![Figure 4-3. Nitrogen Inputs to a Waterbody and Its Watershed.](image)

### 4.2.1 Atmospheric Deposition Sources and Loads

Although not as well understood as point sources and stormwater runoff, much has been learned over the past decades regarding atmospheric deposition. During research over the past 30 years, atmospheric deposition has been identified as a major source of nitrogen and phosphorus loadings to receiving waters (Environmental Science & Engineering (ES&E), 1987; U.S. EPA, 2000). Along the eastern U.S. coast and eastern Gulf of Mexico, atmospheric deposition of nitrogen currently accounts for 10% to over 40% of new nitrogen loading to estuaries (Paerl et al., 2002). Atmospheric deposition delivers nitrogen as wet deposition (precipitation) and dry deposition, which is the fallout of gases and particles not associated with precipitation. Wet deposition is a function of rainfall, so delivery can vary greatly on a seasonal, annual, and spatial scale.

The sources of nitrogen that are ultimately deposited include both natural and anthropogenic sources. One of the natural sources is soil emissions. Williams and Fehsenfeld (1991) measured soil emissions in three North American ecosystems. Results varied significantly, with a grassland site near Nunn, Colorado, having the highest NOx emissions (mean: 10 ng N/m²/s), with emission rates at both a coastal marine environment (mean: 0.034 ng N/m²/s) in South Carolina and a deciduous forest (mean: 0.28 ng N/m²/s) near Oak Ridge, Tennessee much lower. A comparison of the results from the Williams and Fehsenfeld study with previous measurements indicates that NOx emissions from grasslands and temperate forests are similar within each ecosystem independent of location.

There are many potential chemical reactions that occur within the atmosphere before the nitrogen in ultimately deposited. These reactions result in a number of nitrogen forms that are commonly deposited. Nitric acid (HNO₃), aerosol nitrate (NO₃⁻), ammonia and ammonium nitrogen (NH₃ and NH₄⁺), and organic nitrogen are the nitrogen species that are deposited from the atmosphere to the land and water surface in the watershed (U.S. EPA, 2000). Nitrogen deposition associated with urbanization includes mainly NOx and particulate ammonium nitrate (PAN).
Agricultural sources are primarily ammonia and organic nitrogen (Paerl et al., 2002). Although atmospheric deposition in the U.S. is still dominated by NOx emitted from fossil fuel combustion, annual NH$_4^+$ deposition is increasing, and in some regions is approaching total NO$_3^-$ deposition (Paerl et al, 2002).

Nitrogen can be deposited both on the watershed and on the surface of receiving water (Figure 4-4). The atmospheric deposition of nitrogen to the land surface is a component of stormwater runoff loadings, while fallout on the water surface is classified as direct atmospheric deposition loading. Atmospheric nitrogen falling on the land can be entrained in runoff and carried to a surface waterbody, or it can seep into the groundwater and undergo subsurface processes in the soil and be transported in the shallow groundwater to a surface waterbody. The amount and nature of the atmospheric nitrogen loading that eventually reaches the receiving waterbody depends upon which of these pathways is taken.

![Figure 4-4. Major Atmospheric Deposition Linkages Within the Watershed and To the Receiving Waterbody.](image)

### 4.2.1.1 Estimating Atmospheric Deposition Loads

Direct estimation of wet atmospheric deposition inputs to a watershed requires two sets of measured data: rainfall volume and nitrogen concentration in the rainfall. Dry deposition is much more difficult to measure than wet deposition and is often estimated using an empirically derived wet:dry deposition ratio factor. Precipitation data can be obtained from the National Weather Service or regional and local sources. Data should be summarized on a monthly, seasonal, or annual basis.

Precipitation nitrogen concentrations are obtained from the analysis of precipitation water samples. There are several national or regional atmospheric monitoring programs that can provide concentration data (Environmental Science and Engineering, 1987; U.S. EPA, 2000; NADP, 2001; Dennis and Arnold, 2007; NWQMN, 2007). Data obtained from these programs may be the best available information for a watershed, but unique regional or local characteristics may make the use of local data essential. Deposition monitoring sites are sparsely distributed.
and factors affecting deposition rates are highly variable, particularly in mountainous regions, thus regional deposition estimates carry some unquantifiable uncertainty (Miller, 2005).

When atmospheric deposition estimates were first made for the Tampa Bay Estuary Program in 1994 (Zarbock et al., 1994), the closest nitrogen concentration data available were from a National Atmospheric Deposition Program (NADP) site at the edge of the Tampa Bay watershed. Atmospheric phosphorus loading was also to be estimated, but rainfall phosphorus concentration data were available only from the Tampa National Urban Runoff Study (NURP) (Metcalf & Eddy, 1983), a study conducted several years earlier. Dry deposition was calculated by multiplying the wet deposition by an empirically-derived factor obtained from the Florida Acid Deposition Study (FADS), a state-wide study conducted by the Florida Electric Power Coordinating Group, an association of Florida electrical power generating companies (ES&E, 1987).

As it became evident that atmospheric sources of nutrients were a major source of excess nutrient loadings the Tampa Bay Atmospheric Deposition Study (TBADS) monitoring program was established to conduct sampling within the Tampa Bay watershed (Poor et al., 2001; Poor, 2002). Data from this program included local measurements of nitrogen and phosphorus, as well as the wet to dry deposition ratio, and allowed very site-specific loading estimates to be made. This allowed for the refinement of the original loading estimates.

The wet deposition input to the watershed can be quantified by multiplying a water volume (precipitation) by a concentration. Precipitation volume is calculated by multiplying the precipitation depth for a given time period by the surface area of the waterbody. The water volume is then multiplied by a concentration to yield a mass (nitrogen or other constituent), which can be calculated on a daily, monthly, seasonal, or annual basis. The sum of the wet and dry deposition estimates is the total atmospheric deposition load. Janicki Environmental (2005) provides a discussion of methods that can be used to quantify atmospheric deposition loading.

As an alternative to calculating loads using rainfall and concentration data, atmospheric deposition loadings of nitrogen are often evaluated utilizing numerical computer models. Models may be LaGrangian, which track emission plumes, or Eulerian, which are grid-based. Both types of models require input of emissions, from point (factories and power plants), area (small fixed emitters), mobile (transportation sources), and biogenic (vegetation and microbial activity) sources, as well as meteorological information, including winds and temperatures (see U.S. EPA, 2001, for a more complete discussion). LaGrangian models are most useful for those chemical constituents having relatively conservative properties, so that chemical reactions that result in changing chemistry do not play a major role during transport. Eulerian models are best employed when non-linear chemistry is involved in the constituents of interest, such as ozone, nitrogen, and sulfur. Depending on the area of interest, computer model run time is often a consideration when selecting the spatial resolution of an Eulerian model, with coarser grids (tens of kilometers) for larger areas, and more refined grids (kilometer scale) when examining more local areas. Inputs from sources and deposition to the surface are averaged over an entire grid cell.

Estimates of atmospheric nitrogen yields (kilograms per hectare) have been developed nationwide by NADP (2000). These values can be obtained graphically or digitally. The load can be calculated by multiplying the yield times the surface area of the subject watershed or receiving water to obtain a mass delivery of nitrogen.
4.2.1.2 Atmospheric Deposition Sources and Loads in Florida

The major emission sources of nitrogen oxides (NOx) in Florida include fossil fuel combustion, emissions from CAFOs and other agricultural sources, and natural sources. Emissions from mobile and stationary sources contribute large masses of nitrogen to the atmosphere in the form of nitrogen dioxide (NO2), which is one of a group of highly reactive gasses known as nitrogen oxides (NOx). Other nitrogen oxides include nitrous acid and nitric acid. While U.S. EPA’s National Ambient Air Quality Standard covers this entire group of NOx, NO2 is the component of greatest interest and the indicator for the larger group of nitrogen oxides. NO2 forms quickly from emissions from cars, trucks and buses, power plants, and off-road equipment. In addition to contributing to the formation of ground-level ozone, and fine particle pollution, NO2 has been linked with a number of adverse effects on the respiratory system.

U.S. EPA prepares a national database (http://www.epa.gov/ttn/chief/eiinformation.html) of air emissions information with input from numerous state and local air agencies, from tribes, and from industry. Table 4-1 below shows estimated NOx emissions in Florida for 2005 from various sources.

<table>
<thead>
<tr>
<th>Source Sector</th>
<th>NOx Emission (tons/year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Electricity Generation</td>
<td>216,737</td>
</tr>
<tr>
<td>Fires</td>
<td>114</td>
</tr>
<tr>
<td>Fossil Fuel Combustion</td>
<td>45,419</td>
</tr>
<tr>
<td>Industrial Processes</td>
<td>17,560</td>
</tr>
<tr>
<td>Miscellaneous</td>
<td>336</td>
</tr>
<tr>
<td>Non Road Equipment</td>
<td>161,124</td>
</tr>
<tr>
<td>On Road Vehicles</td>
<td>378,063</td>
</tr>
<tr>
<td>Residential Wood Combustion</td>
<td>399</td>
</tr>
<tr>
<td>Solvent Use</td>
<td>22</td>
</tr>
<tr>
<td>Waste Disposal</td>
<td>11,798</td>
</tr>
</tbody>
</table>

Atmospheric deposition for the Tampa Bay area has been estimated by Dr. Robin Dennis of the U.S. EPA (Dennis and Arnold, 2007). Dr. Dennis compiled information from output of the CMAQ-UCD Eulerian model, utilizing 2 km resolution over Tampa and nearby regions, nested in an 8 km grid domain extending over South Carolina, Georgia, Alabama, and Florida, which was in turn nested in a 32 km grid extending over the entire continental U.S. The model was implemented to assist the Tampa Bay TMDL implementation planning process by providing estimates of current and future nitrogen deposition to the bay and its watershed. The model was run considering estimates of reductions in N deposition associated with power plant upgrades to the Gannon and Big Bend power plants, and considering estimated reductions in nitrogen deposition associated with the Clean Air Interstate Rule for 2020.

The base year for the CMAC-UCD model was 2002, with mobile source and power plant emissions representing approximately 75% of the emissions in the watershed. The modeling effort indicated that about half the oxidized nitrogen deposition to the watershed was from watershed NOx emissions, with some spatial variation. Repowering of the two power plants resulted in reduction of oxidized nitrogen deposition to the watershed of about 10%. Implementation of the Clean Air Interstate Rule (CAIR) was predicted to result in a decrease in oxidized nitrogen deposition to the watershed of approximately 40%. The CAIR was vacated by the U.S.
Court of Appeals in July 2008, with rehearing in December 2008 resulting in the CAIR remaining in place until issuance by U.S. EPA of a replacement rule.

In some relatively undeveloped regions of the state, atmospheric sources appear to be by far the largest contributor of nitrogen to aquatic systems. Fu and Winchester (1994) examined the Apalachicola-Chattahoochee-Flint (ACF) river system, while Winchester et al. (1995) examined a 12-county area extending across north Florida from Pensacola on the Panhandle to Gainesville in the north central portion of the state. Both studies were based on statistical analysis of long-term data from the National Atmospheric Deposition Program (NADP) rain chemistry monitoring network and the USGS water monitoring program. It was concluded that in the ACF system, the average nitrate and ammonium atmospheric deposition flux is sufficient to account for the nitrogen that flows to Apalachicola Bay, a potentially nitrogen limited estuary (Fu and Winchester, 1994). Other nitrogen inputs were thought to be small, including urban sources such as point source discharges, and agricultural fertilizer and manure that would be retained on crops and rangeland.

The results from these studies also suggested that after atmospheric deposition, inorganic nitrogen undergoes biogeochemical transformation within the watersheds, transforming the inorganic nitrogen to organic forms. The results indicate that excess dissolved nitrogen could be temporarily recycled in the watersheds but not retained (Winchester et al., 1995). In both studies nitrate and sulfate, which are both introduced to the atmosphere through the combustion of fossil fuel, were correlated during high flows in the river but not during low flows, adding evidence that atmospheric deposition dominated nitrogen loadings to the systems and also demonstrating the importance of river transport of nutrients in the environment (Fu and Winchester, 1994; Winchester et al., 1995).

The relative contribution by atmospheric deposition depends on the size of the watershed with respect to the open waterbody. Rivers and streams, with much smaller open water area per watershed land area, will have proportionally less direct atmospheric deposition loading in comparison to other sources. Recent studies in southwest Florida show that atmospheric deposition loads range from one-quarter to one-third of the total nutrient loading to the Tampa Bay and Charlotte Harbor estuaries (Janicki Environmental, 2005, 2010b). Table 4-2 presents the relative contributions by source of nitrogen loads to a number of Florida receiving waterbodies (Figure 4-5). Clearly, these relative contributions vary appreciably among waterbodies but nonpoint sources are invariably a major contributor.

Table 4-2 illustrates the relative contributions of different sources of nutrient loadings to a selection of Florida lakes, rivers, and estuaries. Locations of the watersheds are shown in Figure 4-5. The purpose of the table is not to compare loads between watersheds, but to show how dominant nonpoint source loadings are across all these examples of different waterbody types. While comparison of these watersheds is limited since the methods used can vary appreciably, nonpoint sources are invariably a major contributor in all cases. In some examples OWTS loads were calculated separately, while being reported as part of nonpoint source or domestic waste sources in others. The contribution of nonpoint source loads ranged from 45-90%. The only waterbody with another source nearly as significant as nonpoint source was Lake Howard, which had almost equal point and nonpoint source loadings. However, since that 1998 study, a concerted effort to stop direct point source discharges to the lake and to take OWTS off-line in favor of sanitary sewer has taken place and current point source discharges are likely much lower now, although no update of the loading analysis has been done.
It should be noted that both MS4 and non-MS4 stormwater is included in the nonpoint source loading category. The studies included in Table 4-2 estimated nonpoint source loadings as a whole, and did not differentiate between regulated (MS4) and unregulated nonpoint source loads. Methods used to estimate nonpoint source loadings are usually the same for both.

One consistency in the loadings is that all watersheds separated direct atmospheric deposition and did not break out nitrogen deposition to the watershed land area. A consequence of this is that estuaries, with a larger open water to land ratio, have the largest direct atmospheric deposition contribution and rivers, with a smaller water to land ratio have a much smaller atmospheric load, while lakes are intermediate.

The large contribution of nonpoint source loads is not unique for nitrogen, or for Florida. A comparable analysis for other constituents, or in other states, would likely yield very similar results.

<table>
<thead>
<tr>
<th>Waterbody</th>
<th>Sources</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Nonpoint Sources</td>
</tr>
<tr>
<td>Lakes</td>
<td></td>
</tr>
<tr>
<td>Apopka (1) (E)</td>
<td>88.3</td>
</tr>
<tr>
<td>Lake Howard (2) (E)</td>
<td>44.7</td>
</tr>
<tr>
<td>Rivers</td>
<td></td>
</tr>
<tr>
<td>Lower St. Johns (3)</td>
<td>51.6</td>
</tr>
<tr>
<td>Caloosahatchee (4)</td>
<td>90.8</td>
</tr>
<tr>
<td>Peace (5)</td>
<td>78</td>
</tr>
<tr>
<td>Wekiva (6)</td>
<td>66</td>
</tr>
<tr>
<td>Estuaries</td>
<td></td>
</tr>
<tr>
<td>Tampa Bay (7)</td>
<td>63</td>
</tr>
<tr>
<td>Sarasota Bay (8)</td>
<td>45</td>
</tr>
<tr>
<td>Charlotte Harbor (5)</td>
<td>67.3</td>
</tr>
<tr>
<td>Indian River Lagoon (9)</td>
<td>82.6</td>
</tr>
<tr>
<td>Indian River Lagoon (10)</td>
<td>90</td>
</tr>
</tbody>
</table>

Table 4-2. Relative Contributions by Various Sources to the Nitrogen Loads to Selected Florida Receiving Waterbodies (Percent).

Sources
(1) SJRWMD, 2003
(2) SWFWMD, 1998
(3) FDEP, 2008
(4) FDEP, 2009
(5) Coastal Environmental, Inc., 1995
(6) MACTEC, 2007
(7) Janicki Environmental, Inc. 2005
(8) Sarasota Bay Estuary Program, 2006
(9) Sigua and Tweedale, 2003
(10) Woodward and Clyde, 1995

Notes
(A) Included in nonpoint source loadings
(B) Domestic and Industrial combined
(C) Not assessed
(D) Included in DPS as wastewater
(E) TP load ratio, TN not done
4.2.2 Nonpoint Sources and Loads

The sources of nitrogen that are delivered as nonpoint sources to the receiving waterbody are shown in Figure 4-6. In addition to atmospheric deposition (discussed above), fertilizers, animal waste, and biological nitrogen fixation are the major sources of nitrogen in nonpoint sources.
The common forms of nitrogen found in manufactured fertilizers are anhydrous ammonia (82% N), urea (46% N), solutions (28-32% N), ammonium sulfate (21% N) and ammonium nitrate (34% N). The forms all vary with respect to their volatility, susceptibility to leaching, and rate of conversion to bioavailable forms. Of these, anhydrous ammonia is the slowest to react, and urea and ammonium nitrate are the most mobile (Vitosh and Johnson, 1997). Cropping agricultural practices often focus on water delivery and control and can be a significant source of nitrate seepage and export (Burkart and Stoner, 2008; Randall and Ross, 2008; Tomer et al., 2009).

When applied, the ammonia in fertilizer is converted to nitrate through nitrification by soil organisms. The nitrate is soluble and leaches easily. Plants and microbes both use nitrate and ammonia to create organic compounds. Nitrate travels easily through the shallow groundwater. Estimates of pollutant loads from groundwater are among the least precise among nonpoint sources. Groundwater can reach the receiving waterbody either as baseflow to tributary streams and rivers or can be delivered directly to a receiving water.

As discussed above, fertilizer use has increased rapidly during the past decades and represents a major source of culturally-derived nitrogen. Application rates in the U.S. are estimated to have risen by 12-fold since the 1950s and represent up to 57% of nitrogen inputs to the environment above natural levels. Over 50% of the applied fertilizer is consumed as food and feed products by humans and livestock. Livestock consume most of the nitrogen in harvested crops and forage. Fertilizer inputs are greatest, and should be given high priority for management, in areas with intensive agricultural operations.

Animal waste (i.e., urine and feces) is another major source of nitrogen that contributes to the overall nonpoint source loading to a receiving waterbody. Although pets contribute to nitrogen loadings in urban areas, the vast majority of animal waste originates from agricultural operations from cattle, hogs, sheep, and poultry. Some of the ammonia in waste is transformed through nitrification to NO2 and then NO3. Nitrogen loading through animal waste is highest in areas where animals are kept and raised in confined situations. Confined Animal Feeding Operations (CAFOs) congregate animals, feed, manure and urine, dead animals, and production operations on a small land area. Feed is brought to the animals rather than the animals grazing or otherwise seeking feed in pastures, fields, or on rangeland. There are approximately 450,000 CAFOs in the United States (USDA, 2009a).

Ammonia is a major component in fertilizer and animal waste, and is very volatile. Volatilization rates are dependent on the nitrogen species in the fertilizer, meteorological factors (temperature, wind, and humidity) and physical factors (soil type and saturation) and can account for 10-40% of nitrogen applied as fertilizer or animal waste (Hatfield and Follett, 2008).

Nitrogen inputs from fertilizer and animal waste can be delivered to a receiving waterbody through surface water runoff, groundwater infiltration, or volatilization to the atmosphere. Some of the nitrogen will seep into the soil and be processed or will enter the groundwater system, as shown in Figure 4-3. When stormwater runoff occurs, the nitrogen from the land surface can be entrained in the water and carried to a tributary stream or river, or directly to a receiving water in both organic and inorganic forms.

Depending on local conditions a large fraction of fertilizer and animal waste can seep into the soil and be transformed while being transported in groundwater. Fertilizer nitrogen inputs to groundwater are estimated using infiltration rates that vary with soil type, season, depth to the water table, land surface slope, and cultivation practices. Infiltration rates can be measured at a
site using geotechnical practices such as a double ring infiltrometer, and monitoring wells used to obtain nutrient concentrations. Local hydrogeologic features must be considered when calculating groundwater fertilizer and animal waste loads. Data required to estimate loadings includes potentiometric surface (pressure head) elevations for confined aquifers, preferably for wet season and dry season conditions, aquifer transmissivity, topography data for the watershed, and N concentration data. Nitrate + nitrite (as N) concentrations are typically used to estimate nitrogen loads, as these species are most often cited as groundwater pollution threats. Flows and loads can be estimated using a simple flow net hand calculation, or using a complex computer model such as MODFLOW.

Biological nitrogen fixation (BNF) includes both natural and anthropogenic sources. In particular, BNF can be enhanced in agricultural settings. Cultivation can enhance BNF by the association of legume plants (e.g., beans, peas, and clover) and the symbiotic bacteria *Rhizobium*.

### 4.2.2.1 Estimating Nonpoint Source Loads

Nutrients that enter the watershed from nonpoint sources and are delivered to the receiving waterbody can be quantified using a variety of approaches. These include methods to estimate the actual sources that generate the nitrogen and the linkages between these sources and the receiving waterbody.

The fertilizer gross nitrogen inputs to the watershed can be estimated by multiplying the number of acres fertilized by the application rate of nitrogen in the fertilizer. Application rates are crop-specific and also vary by land management methods, region, season, soil type, and precipitation patterns. The net nitrogen input to the watershed from fertilizer is calculated by partitioning the amount of fertilizer applied to the land to the fractions that are taken up by plants, entrained in runoff, seep into the shallow groundwater, or that are volatilized back into the atmosphere. Nitrogen entering the transfer linkages is then quantified to estimate the mass of nitrogen delivered to the subject receiving waterbody.

The amount of nitrogen from fertilizer and animal waste that becomes entrained in surface water runoff is very site-specific and depends on many factors including mass of nitrogen on the ground, terrestrial plant uptake, land management practices, precipitation patterns, soil type, topography, and season. Literature values are available for representative nitrogen concentrations and nitrogen yields (lb/acre/year) from agricultural lands (Andrews, 1992; Haith and Shoemaker, 1987). Runoff loads can be roughly estimated multiplying these values by the number of acres of an agricultural facility.

Runoff loads can also be estimated using measured flow and water quality samples of runoff. Samples are usually taken for a number of storm events. Although this can provide useful data sampling is site and storm event specific, and can be difficult and expensive to get representative samples that are appropriate for use over a wide area. Also, if periodic sampling is not conducted the samples may become inappropriate for an area, as land use or management changes over time can greatly change the characteristics of runoff (U.S. EPA, 2008a).

Channelized flows include both stormwater runoff and baseflow. Separation of baseflow and direct runoff can be an important part of a loading analysis. Baseflow is the groundwater which contributes to stream flow during dry periods. Direct runoff is the surface flow that contributes to a stream during and immediately after precipitation (Gupta, 1989). By separating baseflow and direct runoff, the actual volume of runoff and nonpoint loading from the storm can
be estimated. Several techniques are available to accomplish this task. Some of them are mostly qualitative, such as drawing a line across the base of the hydrograph or construction of an exponential depletion curve. There are more rigorous methods of baseflow separation, some incorporating watershed characteristics, some using sophisticated statistical methods, (Nejadhashemi et al., 2005). Baseflow separation can also be accomplished by assessing the concentrations of water quality constituents that are unique to either groundwater or surface water. It should be noted that in many urban areas or in areas with a deep surficial aquifer the separation of baseflow and direct runoff can be neglected, because almost all flow comes from direct runoff.

Models are frequently used to estimate nitrogen loading via stormwater runoff. There are numerous types of models, with some specific to non-urban applications that simulate “off the land” loadings and transformations. The simplest include spreadsheet based models with a minimum of input values needed. The Simple Model (Schueler, 1987) is a prototype of a simple model that estimates annual loads using only a few variables (precipitation, nutrient concentration, runoff coefficient).

Another simple approach to estimate nonpoint source loads is the unit area load approach. Literature values exist for land use-specific unit area loads (e.g., pounds per acre per year) that can be used (Baginska et al, 2003; NALMS, 1988). The unit area load is multiplied by the number of acres of a particular land use in the basin to obtain a load from that land use, and then the individual loads are summed to obtain a total basin load.

SPARROW (Spatially Referenced Regressions on Watershed attributes) is a modeling technique used to relate observed water quality data to watershed attributes. SPARROW uses a nonlinear regression equation describing the non-conservative transport of pollutants from a watershed through its hydrologic network. SPARROW predicts contaminant flux, concentrations, and yields (Schwarz et al., 2006).

At the other end of the spectrum, there are complex mechanistic models that require data for a large number of model variables, boundary conditions, and rate processes. SWMM is a relatively straightforward model that is commonly used to simulate both channel flow and overland flow (Huber and Dickinson, 1988). BASINS (U.S. EPA, 2010c), MIKESHE, and HSPF (U.S. EPA, 2010d) are highly data intensive models that have also been applied to estimate loads for different waterbody types. Models suited for urban areas are better able to simulate storm sewer network flow and include STORM, SWMM, and ICPR (NRC, 2000).

WAMView (Watershed Assessment Model) (SWET, 2002) simulates the primary physical processes involved in watershed hydrologic and pollutant transport. The model makes use of GIS-based coverages of land use, soils, topography, hydrography, basin boundaries, point sources, and meteorological data. The model is a grid-based system that can be used to assess spatial impact of land use changes on water quality and quantity, and provides load estimates as a function of modifications to source loadings.

The grid-based representation of the model allows for identification of flows and nutrient concentrations for each cell, with routing of the surface water and groundwater to assess the flow and water quality throughout the watershed and at the discharge to the receiving waterbody. The model provides for simulation of phosphorus and nitrogen species, total suspended solids, and biological oxygen demand, and has been linked to WASP (U.S. EPA, 2010e) to provide simulation of dissolved oxygen and chlorophyll.
WAMView has been used to simulate flow and water quality constituents for several Florida watersheds in support of Florida’s TMDL program, including Lake Okeechobee tributaries (U.S. EPA, 2008b), Lakes Lena and Bonny (Shelley et al., 2005a,b) and Lake Parker (U.S. EPA, 2006a) in the Peace River watershed, and multiple creeks in the Hillsborough River basin (U.S. EPA, 2004), among others.

WAMView and other field-scale models estimate nonpoint source loadings by inputting data describing actual nutrient source loads (fertilizer, animal waste, etc.) to the land surface and then tracking the material as it follows the various pathways described in this report (surface runoff, leaching, etc.). This approach was first used in agricultural applications, for example to compare how much nitrogen would reach a waterbody if fertilizer application rates were changed and can result in a very detailed examination of nutrient inputs within a watershed.

This approach can also be used in urban settings, if sufficient information is available for different types of land uses. Loadings from residential fertilizer may vary greatly between lots, blocks, and neighborhoods. However, professionally-maintained areas of turf are much more likely to have fertilizer applied within fairly tight ranges of fertilizer type, application rates, and periodicity of application. Areas where information may be most representative include golf courses, cemeteries, parks, sporting fields, and large public grounds.

There are several watershed loading models that are designed specifically for agricultural applications. These models simulate loadings based on specific agricultural activities (row crops, livestock, citrus, etc.) Required data may include tillage, fertilization, planting, harvest, rotation, chemical operations, and irrigation schedules. Information describing feedlot operations includes daily manure production rates, times of manure removal, and residual amounts from previous operations. Several of these agricultural models are discussed below.

Agricultural Nonpoint Source Pollution Model (AGNPS) (Bingner and Theurer, 2009): AGNPS is a tool used to evaluate the effects of agricultural management decisions on the resultant water quality conditions in the watershed. The term “AGNPS” now refers to the system of modeling components instead of the original single event AGNPS, which was discontinued in the mid-1990s.

The emphasis of the model is on nutrient transport, soil erosion, and sediment yield to compare the benefits of best management practices (BMPs) on agricultural pollutant loadings. AGNPS can simulate nutrient and sediment loadings from agricultural basins on an event-based or continuous simulation. AGNPS can also simulate point source inputs such as feedlots or wastewater discharges. In addition, sediment transport such as stream bed and gulley erosion can be simulated. The universal soil erosion equation is used to predict erosion for different soil particle sizes. The pollutant transport routine simulates movement of both soluble constituents and sorbed materials. Input data are mainly derived from topographic maps, soil surveys, meteorological data, and other literature values.

Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS) (Knisel, 1980) and GLEAMS, Groundwater Loading Effects on Agricultural Management Systems (Knisel, 1993) are also typically used in agricultural applications. CREAMS is classified as a field-scale model used to assess agricultural BMPs for surface water and groundwater. Erosion, sediment yield, and subsequent nitrogen and phosphorus loads are estimated in an “edge of the field” assessment. GLEAMS may be used to provide a groundwater component to off-site load-
ings. CREAMS and GLEAMS are maintained by the USDA Agricultural Research Service (NRC, 2000).

Simulator for Water Resources in Rural Basins (SWRRB): (Arnold and Williams, 1994) was intended to simulate daily hydrology, nutrient transport, and other loads from agricultural basins. SWAT (Soil and Water Assessment Tool) (Arnold et al., 1995) is intended to scale down SWRRB from a watershed to basin level. Surface water and groundwater relationships are simulated using regression relationships, chemical properties, and sediment yield and runoff volume. Nitrate and organic nitrogen are simulated using different algorithms in different soil layers. The models are supported by USDA Agricultural Research Service.

Data that are typically required to conduct a runoff loading assessment for nitrogen or any pollutant using a mechanistic model includes, from simple to complex, with likely information sources:

♦ Land Use – Local government planning departments or state agencies, often in GIS format. Future land use may also be available for running future conditions loading scenarios.

♦ Rainfall – National Weather Service, water management districts (Florida), local monitoring programs. Data can be used on a daily, monthly, or annual basis. Event-based rainfall is usually not used for pollutant loading estimates.

♦ Soils – USDA Natural Resources Conservation Service. Hydrologic Soil Groups (HSGs) give an indication of a soil’s runoff generating features. Likely available in GIS format.

♦ Runoff Coefficients – Land use-specific values are obtained from numerous literature sources, and give an indication of a land use’s runoff generating features.

♦ Event Mean Concentrations (EMCs) – Land use-specific values are obtained from numerous literature sources and provide estimates of stormwater quality concentrations for many parameters.

♦ Watershed and Subbasin Boundaries – May be available from water management agencies, U.S. EPA, or can be generated using topographic maps either manually or using GIS routines.

♦ Surface Water Hydrography – Stream and river networks are important if in-stream and watershed processes are to be modeled. Data can be obtained from the National Hydrography Dataset (USGS, 2010a) in GIS format or state or local sources. USGS quadrangle maps also show this information in manual form.

♦ Conveyance Information – Conveyance information is important if in-stream and watershed processes or downstream loadings are to be modeled. Characteristics such as channel dimensions and slope, Manning’s friction coefficient, and other parameters allow improved hydrology/hydrologic model results, which will improve the loading estimates.

♦ Pollutant Process Rate Constants – If in-stream processes are to be modeled rate constants for pollutant transformations (denitrification, etc.) are needed. Literature values are available if site-specific data are not available (U.S. EPA, 1985).

♦ Sediment Characteristics – If internal cycling and nutrient spiraling are to be considered, sediment grain size, chemical composition, and other features should be obtained through field data collection.
There are other modeling approaches that can be used to estimate loads including statistical methods. Also, a mass balance approach can be used (Borgotti, 2008). Each method has benefits and shortcomings, so it is often useful to use more than one method if possible to corroborate results and develop a weight-of-evidence for the findings.

In general, the least complex model that will meet the needs of a particular analysis should be used. Complex models are more expensive and take longer to construct and run. Also, the enhanced capabilities of a complex model may be offset by uncertainties in the numerous assumptions that have to be made (Freni et al., 2009). There are many stormwater modeling studies with several good comparisons of models of varying complexity (e.g., Donigian and Huber, 1991; DeVries and Hromadka, 1993; Novotny and Olem, 1994; Donigian et al., 1995; Singh, 1995; U.S. EPA, 1997; Center for Watershed Protection, 2010; U.S. EPA, 2010b; USGS, 2010).

4.2.2.2 Nonpoint Sources and Loads in Florida

Nutrient enrichment of surface water, groundwater, and the air from animal waste near CAFOs is a growing issue across Florida and the nation. U.S. EPA has criteria for developing Comprehensive Nutrient Management Plans for CAFOs but implementing the plans is voluntary for the operator (USDA, 2009a). CAFOs are regulated under the NPDES program governed by the 2008 U.S. EPA CAFO rule. U.S. EPA has delegated the responsibility for regulating CAFOs to FDEP’s industrial wastewater program. These include dairies, poultry, horse, and swine operations.

The number of CAFOs operating in Florida was estimated using the most current information available. The U.S. Department of Agriculture 2007 Census of Agriculture provides a variety of agricultural statistics for all states (USDA, 2009b). Agricultural facilities in Florida meeting U.S. EPA animal feeding operations criteria are shown in Table 4-3.

<table>
<thead>
<tr>
<th>Animal</th>
<th>U.S. EPA Criteria (Minimum Number of Head)</th>
<th>Number of Farms</th>
</tr>
</thead>
<tbody>
<tr>
<td>Swine</td>
<td>750</td>
<td>3</td>
</tr>
<tr>
<td>Cows – dairy</td>
<td>200</td>
<td>109</td>
</tr>
<tr>
<td>Poultry</td>
<td>9000</td>
<td>22</td>
</tr>
<tr>
<td>Total</td>
<td>–</td>
<td>134</td>
</tr>
</tbody>
</table>

There are also a number of additional agricultural operations that do not meet the minimum size criteria for CAFOs but are still sizable, including 10 swine farms with more than 200 head, and 42 poultry farms with more than 400 head. It should be noted that the number of CAFOs in Florida has been decreasing over the past years, likely partially in response to the spread of urban development and increased regulatory scrutiny.

Animal waste (manure) is also used as fertilizer. It is composed of organic nitrogen and ammonia and is another major source of agricultural nitrogen inputs to watersheds. The nitrogen content of manure varies according to the type of animal and age of herd/flock, feed and feed supplements, bedding, collection and storage system, application method and timing, and soil texture and weather. Manure can be stored in a variety of ways, in barns, outside, or in lagoons. It can be treated, composted, or applied directly to a field. Nitrogen transformation and migration to the air, surface water, and groundwater can occur almost immediately after manure is generated.
Estimates were made of state-wide manure generation by animals, several of which were not included because of their smaller numbers across Florida. The state-wide number of animals in 2007, the species-specific amount of manure generated on a daily basis, and manure nitrogen content are shown in Table 4-4. Total nitrogen generated is summed on an annual basis.

Table 4-4. Livestock and Poultry Manure Generated in Florida, 2007.

<table>
<thead>
<tr>
<th>Animal</th>
<th>Number of Animals (1)</th>
<th>Manure (lb/animal/day)</th>
<th>Nitrogen Content (Percent)</th>
<th>Nitrogen Load (lb/animal/day)</th>
<th>Nitrogen Load (lb/day)</th>
<th>Nitrogen Load (tons/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Swine</td>
<td>19,932</td>
<td>75</td>
<td>5 (3)</td>
<td>–</td>
<td>74,745</td>
<td>13,641</td>
</tr>
<tr>
<td>Cows – dairy</td>
<td>1,711,011</td>
<td>100 (2)</td>
<td>–</td>
<td>0.4 (4)</td>
<td>684,404</td>
<td>124,904</td>
</tr>
<tr>
<td>Poultry</td>
<td>28,494,105</td>
<td>0.3 (5)</td>
<td>1.7 (5)</td>
<td>–</td>
<td>145,320</td>
<td>26,521</td>
</tr>
<tr>
<td>Total</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>165,066</td>
</tr>
</tbody>
</table>

Sources:
(1) USDA, 2009a
(3) Cords, W., Bigpumpkins.com web site.
(4) Van Horn et al., 1990.
(5) Naber and Bermudez, 1990.

Commercial fertilizer is the other major nitrogen source associated with agricultural operations, although significant loads of nutrients are introduced in the environment from urban landscape fertilization as well. Commercial fertilizer is synthesized from gaseous nitrogen, as discussed previously. It comes with a variety of nitrogen and phosphorus concentrations but behaves similarly to manure with ammonia volatilizing to the atmosphere or entering surface water or groundwater with nitrates.

The Florida Department of Agriculture and Consumer Affairs (FDACS) publishes annual records of fertilizer usage in the state, (FDACS, 2010). For the period July 2008 through June 2009, the most recent data available, 1,368,495 tons of commercial fertilizer were used in Florida. Synthetic fertilizer commonly contains phosphorus and potassium and sometimes other assorted micro-nutrients. Nitrogen content can range from 0% up to 50%, but a common, mid-range value is 15%. Multiplying the fertilizer applied by the nitrogen content yields an input to Florida watersheds of 205,274 tons/year in 2008-9. How the fertilizer affects the atmosphere, surface waters and groundwaters depends on the fertilizer type, application method and timing, season, and soil characteristics.

The relative contribution of nonpoint source nitrogen loads to a number of Florida waterbodies was discussed above, and the relative magnitude of nitrogen loading to selected lakes, rivers, and estuaries is presented in Table 4-2. One of the clear conclusions that can be drawn from these estimates is the importance of nonpoint source loads to all waterbody types. In the waterbodies shown in Table 4-2 above, nonpoint source loading accounted for no less than 45% of the total nitrogen load in any watershed, and was at or near 90% in three watersheds. The average nonpoint source nitrogen loading for the 11 watersheds was approximately 70% of the total load. No other loading source was so uniformly dominant.
4.2.3 Wastewater Sources and Loads

Wastewater discharges, in particular point sources, have long been subject to regulatory scrutiny in efforts to control nutrient over-enrichment, as discussed in Chapter 1.0. As the relationship between excess nutrient loading to a waterbody and the effects of eutrophication became understood, point sources, especially domestic point sources, were identified as major contributors to the problem (Edmonson, 1969; Ryther and Dunstan, 1971; Cairns, 1977; Hansen, 1994; Nixon, 1995). Management of point sources has proven successful in many locations, with significant over-enrichment problems caused by point source discharges greatly reduced through upgrades and advances in treatment technology, modification of facility operation, and innovative effluent disposal methods (reuse water for irrigation and industrial use, and deep well injection). In Florida, many problematic point source discharges were brought under control as early as the 1970s (Fulton, 1995). Sources and pathways of wastewater in the watershed that are ultimately delivered to the receiving waterbody are shown in Figure 4-7.

The effluent from domestic and industrial facilities delivers nutrients either directly to the receiving waterbody or via pathways through the watershed prior to entering the receiving waterbody. The nutrient sources treated by domestic facilities include human waste and other household and commercial waste products. Industrial discharges vary significantly and depend on facility-specific processes. Other pathways by which nutrients are delivered to the receiving waterbody discussed in this chapter include Combined Sewer Overflow (CSO)/Sanitary Sewer Overflow (SSO), reuse water, and septic systems.

![Figure 4-7. Major Wastewater Pathways in the Watershed.](image)

CSOs and SSOs are emergency discharge features of sewage collection systems that allow either combined sanitary sewage and stormwater (CSO), or straight sewage (SSO) to discharge from the piped collection system if flow exceeds the ability of the system to convey and treat. Although these high flow events release untreated sewage into the environment, it is intended to prevent large scale damage to the entire wastewater system which would have much greater and long-term consequences.

Reuse water is simply treated domestic effluent that is put to beneficial use, usually for landscape irrigation. Septic systems are actually a diffuse source of nitrogen loadings but are discussed in this chapter as a source of human waste.
### 4.2.3.1 State-of-the-Knowledge – Wastewater

**Domestic Discharges – Nitrogen**

Nitrogen in untreated domestic wastewater is mainly in the form of ammonium nitrogen (NH$_4^+$), and organic nitrogen. Sometimes wastewaters contain traces of oxidized forms of nitrogen, mainly nitrite (NO$_2$) and nitrate (NO$_3$). The organic N and NH$_4^+$ concentrations combined is referred to as total Kjeldahl nitrogen (TKN). In the treatment facility, the processes of nitrification, denitrification, and ammonification/assimilation result in the transformation of some of the aqueous nitrogen into gaseous molecular nitrogen gas (N$_2$) and solid sludge. The effluent contains mainly nitrate and ammonia (NH$_3$). Depending on the treatment level, nitrogen removal can exceed 90% (van Haandel and van der Lubbe, 2007).

Organic nitrogen is generally controlled by treatment upgrades, and advanced treatment processes have resulted in decreased ammonia discharge. However, these processes nitrify ammonia that results in higher nitrate concentrations. Thus, concerns about toxicity to aquatic organisms have decreased, but the potential for eutrophication can still be an issue (Mueller and Helsel, 1999). The TN concentration in raw domestic sewage typically ranges from 20-40 mg/L, and that of effluent with secondary treatment is in the range of 20-25 mg/L. Effluent from enhanced nutrient removal treatment facilities can have a TN concentration of 10 mg/L or less but ultimate performance usually reflects permit limits. Additional nutrient removal can be achieved through advanced wastewater treatment, resulting in effluent with TN concentrations of 3 mg/L or less. Advanced wastewater treatment with nutrient removal is mandatory in the Tampa Bay and Sarasota Bay watersheds for all domestic facilities with direct surface discharges. Requiring this level of nitrogen removal efficiency in wastewater treatment greatly reduces point source contributions to eutrophication in the area.

The nitrogen content of industrial waste effluent is a function of the processes that occur within an individual facility. Nitrite has many industrial uses, such as to reduce corrosive action in industrial process water. Ammonia is also used in the production of nitric acid, urea, and other nitrogen compounds, and in the production of ice and in refrigerating plants. Ammonia is also used in cleaning supplies and in water softeners. Food processing facilities generate high levels of nutrient waste including ammonia or TKN, requiring treatment prior to final discharge. Besides nutrients, food processing effluent can also contain elevated levels of biochemical oxygen demand (BOD), total suspended solids (TSS), pathogenic organisms, and residual chlorine and pesticide (UNIDO, undated).

Food processing waste may be discharged to a municipal sanitary sewer, treated using traditional processes (mechanical/chemical, aerobic or anaerobic treatment), or via alternate treatment/disposal methods. These methods include discharge into ponds or treatment wetlands, or land application on crops to take advantage of the high nutrient content (Sherrard et al., 1978; DeBusk et al., 2004; Huang et al., 2005). It should be noted that conventional anaerobic ponds used to purify these effluents give cost-effective removal of BOD but little nutrient removal (Raper and Green, 2001).

There are several methods by which wastewater effluents can be discharged to the environment, including:

- Direct discharge from a pipe to surface receiving waters.
- Land application (discharge to percolation ponds, sprayfields or reuse).
- Percolation through soil to groundwater from OWTS.
♦ Combined sewer and sanitary sewer overflows.

♦ Deep well injection.

The transformations of nitrogen that occur once the effluent exits the facility depend on the discharge method. Effluent discharged directly to a receiving water will deliver nitrogen in much the same form that it left the treatment plant due to the effluent’s isolation from environmental processes for the short time it is in the discharge pipe.

**Direct Discharge**

The load leaving the facility should be relatively unchanged at the discharge point of the pipe unless the pipe is very long. Undesirable chemical reactions in wastewater can occur in very long discharge pipes but these can be rectified. Rodríguez-Gómez et al. (2009) successfully used oxygen injection in a 62 km long wastewater discharge pipe to inhibit anaerobic conditions.

Nitrogen in conventionally treated domestic effluent consists mainly of nitrate and ammonia. Subsequent to entering the receiving water, the ammonia is taken up by phytoplankton or aquatic vascular plants or undergoes fixation to nitrate. The nitrate in the effluent is either taken up by phytoplankton or aquatic vascular plants, undergoes denitrification and returns to the atmosphere, or is sorbed to receiving waterbody sediments. Estimating the fraction of nitrogen that undergoes each transformation and the rate at which each transformation occurs is very difficult to quantify. Bench testing and field measurements have shown that the rates of these processes can vary greatly under normally variable conditions (Bernhardt et al., 2003; Corbet, 2010; Battaglin et al., 2001; Bohlke et al., 2004; DeLuca et al., 2002; Earl, 2004; Boynton et al., 1995; Hall et al., 2005; Johnson, undated; Marcarelli et al., 2008; NRC, 2000; Smith, 1990). Field sampling and bench testing can provide valuable information describing individual processes. Those data can also be used in a material budget to develop a holistic interpretation of the entire system rather than individual components (Crump et al., 1987; Boynton et al., 2008).

Several steps are involved with estimating nutrient loading from point source discharges. The pollutant load to a receiving waterbody is a function of the water flow rate (e.g., MGD) and pollutant concentration (e.g., mg/L) (Rice and Izuno, 2009). The product of these two components yields a calculated load (e.g., tons per year). Depending on the data available, loads can be quantified at a variety of time scales such as month, season, or year.

Domestic point source facilities usually report effluent flow and quality, including nitrogen concentration on a monthly basis. The frequency of sampling during the month is usually facility-specific. The domestic waste flow rate and quality often do not vary significantly over time. Records are available through NPDES Discharge Monitoring Reports (DMR). These records are typically reported monthly and are available in a number of databases maintained by U.S. EPA and state agencies. If the effluent is discharged into a ditch that in turn flows to a receiving waterbody, standard methods for discharge measurements and water quality sampling can be used to quantify the load.

If effluent discharge and concentration monitoring data are not available then other methods must be used to provide gross load estimates. For example, if the number of people in a domestic facility service area is known then estimates of flows and loads can be made based on literature values of per capita waste generation and water used.

Because discharges from CSO/SSOs may contain both untreated domestic and industrial waste, as well as stormwater, the quality of the discharges combines characteristics of all sources.
and can vary significantly. Each storm is different in the quantity and type of pollutants it contributes. For example, storms that occur in the dry season have the most pollutants from runoff. Pollutants such as nutrients, oil, grease, fecal coliform from pet and wildlife waste, and pesticides get flushed into the sewer system. In cold weather areas, pollutants from cars, people, and animals also accumulate on land surfaces during the winter and then are flushed into the sewer systems during heavy spring rains (Hogland et al., 1984; U.S. EPA, 2010f).

CSO/SSOs usually discharge directly into a ditch, river, stream, lake, or estuary. The frequency and duration of overflows is storm-event specific. Some CSO outfalls discharge infrequently, while others are used during every large rainfall. CSO/SSOs discharge directly to a surface waterbody, thus only minimal transformations take place during the discharge process. Discharges to ditches or streams will be subject to the same processes as described for nonpoint sources during conveyance to the receiving water.

There are several methods that can be use to estimate loadings from CSOs/SSOs. Automated samplers can be installed at the discharge points and be programmed to estimate flow and obtain water quality samples during the duration of a discharge (Hogland et al., 1984). Alternatively, flows can be estimated by other means, and concentration data can be obtained from representative data collected from the system or from other systems. If concentration estimates from other locations or storms are used, it may be desirable to adjust the concentration to account for variability in the discharge quality based on the storm event season, magnitude, and duration (U.S. EPA, 2010f).

Industrial discharge loads are calculated the same way as domestic discharges. However, permit requirements for discharge quality and flow rate record keeping for industrial facilities are often not as consistent as for domestic discharges and adds uncertainty to estimates of discharge volume and quality. If a monitoring plan for an industrial source is to be initiated, the magnitude and timing of changes in flow rates and quality should be identified to help select the timing and frequency of sampling.

**Land Application and Reuse**

Treated effluent that is discharged to the land surface (sprayfield disposal or reuse landscape irrigation) or a percolation pond can become entrained in stormwater runoff, return to the atmosphere, be taken up by plants or seep into the groundwater. Nitrogen in wastewater that is discharged via some sort of land application can be sequestered by vegetative uptake, burial in soil, or storage in lakes, groundwater, and wetland areas. This sequestration can be relatively temporary, with the release of this nitrogen occurring in a matter of days, to as long as years. The net result is that many watershed-scale studies comparing nitrogen inputs to outputs have found that large fractions of nitrogen inputs to watersheds are sequestered for years at a time (Sauer et al., 2008).

The fraction of effluent that is discharged as land application can seep into the groundwater and be acted on by subsurface processes. Subsurface retention depends upon the form of nitrogen. Nitrate-nitrogen does not attach to soil particles and, as a result, is easily transported by water. Therefore, the degree to which soils can hold subsurface water strongly influences how much transport will take place. Sandy soils do not hold as much water as clays, so leaching of nitrates will take place much more readily in sandy soil than in a clayey soil. Other factors that can affect nitrate leaching include the amount of rainfall, amount of water use by plants, level of microbial action, and how much nitrogen is already present in the soils (Freeze and Cherry, 1979).
Nitrogen inputs to a receiving waterbody from groundwater linkages can be quantified by collecting samples from a network of monitor wells (Shreffler et al., 2005; Misztal et al., 1992). Wells should be installed on both the upgradient and downgradient side of the application site. Groundwater levels and water quality samples should be collected to characterize the groundwater/effluent plume leaving the site and to establish background conditions. The up- and downgradient samples can be compared to determine nutrient inputs for the effluent. Using the water level information and Darcy’s equation, a well-known tool for estimating groundwater flow rates, estimates of groundwater flux can be made. Multiplying the flow rate by the nutrient concentration provides an estimated load. To estimate groundwater nutrient loads entering a receiving water, a similar monitoring well network can be set up adjacent to the receiving water shoreline. The wells should be far enough away from the water’s edge that surface water will not mix with the groundwater near the shoreline. It should be noted that because subsurface flow and chemical conditions can vary spatially, there can be high levels of uncertainty in estimates of groundwater loading.

Groundwater loads can also be quantified by the use of computer models. Flows and loads can be estimated using a simple flow net hand or spreadsheet calculation, or by using a complex computer model such as MODFLOW. Integrated surface and groundwater models are also available, but they are uniformly data intensive and complex. Local hydrogeologic features must be considered when calculating groundwater loads using any mechanistic groundwater model. Data required to estimate loadings include shallow aquifer water table elevations, soil types, potentiometric surface (pressure head) elevations for confined aquifers, preferably for wet season and dry season conditions, aquifer transmissivity, topography data for the watershed, and nitrogen concentration data. Nitrate and nitrite concentrations are typically used to estimate nitrogen loads, as these species are most often cited as the common forms of nitrogen in groundwater.

Reuse water is domestic effluent and thus has the same characteristics as domestic point source discharges. The delivery pathway of reuse water is the same as other sources applied to the land surface. Reuse water is used as irrigation water on turf and landscape vegetation on public or private land, mainly in urban areas. The nitrogen transformations of reuse water after discharge are the same as other sources applied to the land surface. As discussed above, from the land surface nitrogen can be volatilized, be entrained by stormwater runoff, taken up by vegetation, or seep into the soil.

**Onsite Wastewater Treatment Systems**

Onsite wastewater treatment systems (OWTS), many times referred to as “septic tank systems”, consist of an underground tank for primary treatment followed by a subsurface wastewater infiltration system, sometimes called a “drainfield”. Waste enters the tank for solids separation and primary treatment, and then flows into the infiltration system for further treatment and percolation into the soil. Influent to OWTS consists of raw human waste and other domestic wastewater. The total nitrogen content of water leaving the infiltration system can be high, a recent review reported an average of 60 mg/L in a study of 11 OWTS sites (Hazen and Sawyer, 2009a). Forms of nitrogen present in the septic tank effluent include ammonia, and organic nitrogen, and nitrification in the wastewater infiltration system converts most of the nitrogen to nitrate in a properly functioning system. (Brown, 1992).

Most if not all states have regulations mandating design standards for OWTS. In Florida, the bottom of the infiltration system must be a minimum of two feet above the seasonal high wa-
ter table, and must be installed in soils with good infiltration capacities. This allows the effluent
to percolate vertically from the infiltration system to the water table with remaining ammonia
converting to nitrate through the process of nitrification.

OWTS discharges can be either into the soil or via surface water, depending on whether
the unit is functioning correctly or has failed. Effluent from failed OWTS may leave the site
mainly through surface flow, as most failures result in system back-up, ponding, and subsequent
surface runoff (Janicki Environmental, Inc., 2010c). The percent of in-use OWTS considered to
be failed has been reported by states to range from 1% in Wyoming to over 50% in Minnesota
with most states in the 10-30% range (U.S. EPA, 2002).

Discharges from operating OWTS are usually assumed to be mainly via soil and shallow
groundwater. Assuming that the drainfield is above the water table, effluent must first migrate
downward through the soil, and then travels horizontally along the water table gradient until it
reaches a surface waterbody.

Only a small part, perhaps 10%, of the total nitrogen in raw wastewater discharged to the
septic tanks is removed via sludge that accumulates in the bottom of the tank. Nitrification, i.e.,
conversion of ammonium-nitrogen into the nitrate form, occurs in the first foot or so of soil be-
low the drainfield provided that unsaturated conditions exist in that zone. Nitrate is very soluble
and does not interact with soil components under aerobic conditions, travelling through the soil
practically unimpeded. Denitrification (conversion to nitrogen gas in an anoxic environment)
will occur in the soil and shallow groundwater, but the rate of nitrate reduction by this process is
very site dependent. Absent denitrification, nitrate will not undergo further transformation in
groundwater unless it is taken up by plants (Brown, 1992). Unlike nitrate, most phosphorus in
the effluent is bound to soil particles or taken up by plants near the drainfield.

Estimating loads from OWTS is two-part process. Loads from functioning and failed
septic tanks must be estimated separately. Attempts to estimate nitrogen loadings from operating
septic tanks encounter many of the same issues as similar exercises for atmospheric deposition
and land application of domestic wastewater effluent and reuse water (Schreffler et al., 2005).
Although not subject to nitrogen transformations on the land surface, except for failed septic
tanks, the movement of nitrogen through the soil and shallow groundwater is the subject of con-
siderable past and ongoing research (Hazen and Sawyer, 2009a, 2009b; Aley et al., 2007; Bauman
and Shafer, 1985; Kuphal, 2005; MACTEC, 2007; Maizel et al., 1997; Waller et al., 1987;
CHEC, 2003).

Nitrogen loads from failed OWTS on a basin scale are difficult to estimate with certain-
ty. It is often assumed that the effluent leaving the failed system is near-raw sewage quality, but
estimating the fraction of that transported to a receiving waterbody includes making many as-
sumptions including estimating the relative fractions of flow that remain as surface flow, become
volatilized, or seep into the soil. Then the nitrogen transformations must be estimated for each
fraction using methods discussed above. The variability of physical settings and individual sys-
tem features makes estimating loads from OWTS subject to considerable uncertainty (Hazen and
Sawyer, 2009b).

OWTS nitrogen inputs are usually completed by assembling the best available informa-
tion on the following items:

♦ Number of active OWTS in the study area
Average rate of failure of OWTS in the study area
Average number of people in a household using an OWTS
Average influent nitrogen load entering an OWTS
Average effluent nitrogen load leaving a working OWTS
Average effluent nitrogen load leaving a failing OWTS
Vertical and horizontal soil attenuation rates of TN

A series of simple multiplications and unit conversions done by hand or spreadsheet can be used to calculate estimated inputs.

Septage (the sludge removed from septic tanks during maintenance) is a by-product of OWTS operation and maintenance and is also source of nitrogen loading to the watershed. For an OWTS to function properly, every few (3-5) years the septic tank should be cleaned to remove accumulated semi-solid organic material that accumulates. The sludge reduces the tank’s volume and treatment capacity and can lead to system failure. If a septic tank ceases to operate properly it may release untreated domestic waste to the surroundings. Septage is routinely collected and is delivered for treatment at a municipal wastewater plant, or is spread on the land surface, often pasture or rangeland, for assimilation. Septage contains approximately 2% nitrogen (U.S. EPA, 1995) which is taken up in the watershed through the linkages described above.

4.2.3.2 Wastewater Sources and Loads in Florida

Point source discharges in Florida include effluent from domestic plants (including reuse irrigation), industrial facilities, and OWTS. Although OWTS are actually a diffuse source of nutrient discharges they are included here as a source of human waste. There are no known combined sewers in Florida so CSOs are not a problem, but SSOs occur in some domestic wastewater systems after heavy rainfall events. As in the rest of the nation, early nutrient control actions in Florida targeted domestic wastewater facilities as a source of nutrient enrichment. There has been much success in reducing nitrogen loads from domestic waste over the past decades. An order-of-magnitude estimate of current nitrogen loading to the Florida environment from domestic wastewater are described below.

The 2009 Florida population was estimated to be 18,750,500 (Florida Office of Economic and Demographic Research, 2010). A value of 9.2 lb nitrogen/capita/year was used for waste generation (Janicki Environmental, Inc. 2010c), so an estimate of total nitrogen generated in Florida in human waste is 172,504,600 lb/year (86,252 tons/year). It was assumed that the entire population had waste treated by either sanitary treatment facility (secondary or advanced treatment levels) or OWTS.

Domestic wastewater treatment plants vary in treatment efficiency according to processes employed, operational methods, regulatory permit requirements and influent quality. However nitrogen content of domestic wastewater is somewhat consistent. Moderate strength domestic sewage has a TN concentration of about 40 mg/L. Secondary treatment yields effluent with a TN concentration of about 25 mg/L, and effluent from nutrient removal facilities will generally have a TN content of 10 mg/L or less. Advanced wastewater treatment (AWT) which is required in some parts of Florida for surface discharge, gives an effluent with a TN concentration of 3 mg/L or less, which is considered the practical limits of technology for domestic wastewater treatment. Given the influent and effluent qualities, efficiencies for nitrogen removal for domes-
tic plants using secondary, nutrient removal, and advanced wastewater treatment technology, re-
movals are in the range of 62%, 75%, and 92%, respectively.

Advanced wastewater treatment with nutrient removal has been required for many do-
mestic wastewater plants with surface discharge to Tampa Bay and its tributaries since the pas-
sage of the Grizzle-Fig Act in 1972. This mandated significant treatment upgrades to meet
discharge standards of 3 mg/L TN, along with 5 mg/L BOD₅, 5 mg/L TSS, and 1 mg/L phos-
phorus. Other state bills have caused surface discharges to be severely curtailed or eliminated, such
as the Indian River Lagoon Act of 1990.

The other common form of human waste treatment is with OWTS. The Florida Depart-
ment of Health reports that 26% of Floridians use OWTSs (Janicki Environmental, Inc., 2010c).
OWTS efficiencies vary greatly depending on soil type, construction, use level, maintenance,
etc. Reported nitrogen removal efficiencies are correspondingly variable but have been estimated
at 35% for conventional OWTS in sandy Florida soils (Hazen and Sawyer, 2009a).

An estimate of nitrogen released into the Florida environment from human waste can be
made by calculating treatment plant and OWTS loads separately. Multiplying the total nitrogen
mass generated by humans in Florida (172,504,600 lb/year) by the fraction of the population that
uses treatment plants (74%) and by an estimate of plant efficiency (70%, assuming 75% of efflu-
ent state-wide has secondary treatment and 25% has advanced) gives an annual load released in
wastewater treatment plant effluent of 26,307 tons/year.

Using a similar method for OWTS the total nitrogen mass generated by humans in Flori-
da (172,504,600 lb/year) is multiplied by the faction of the population that uses septic tanks
(26%) and by an estimate of treatment efficiency (35%, the average of available reported effi-
ciencies) gives an annual load released in septic tank effluent of 14,577 tons/year. Adding the
two values gives a total of 40,884 tons/year TN released into the environment from human
waste, substantially less than other major sources.

Industrial discharges in Florida are also regulated under the NPDES program. State-wide
nitrogen loading estimates are not codified, and the broad range of effluent types, volumes and
quality makes developing such a number quite complex and uncertain. The largest industrial
sources of nitrogen discharges in Florida include the electrical power industry, surface mining
and processing of phosphate, paper manufacturing, and agricultural activities (sugar cane, citrus,
livestock/CAFOs). In Florida, historically significant nutrient discharges from food processing
facilities included citrus and seafood processing plants, livestock operations, and sugar manufac-
turing.

Table 4-2 above provided examples of the relative contributions of point sources to ni-
trogen loading in Florida watersheds. Domestic and industrial point sources combined comprise
less than 10% of the total nitrogen loads to many of the watersheds shown. Additionally, in sev-
eral of the watersheds where point sources are significant sources of nitrogen, current manage-
ment initiatives are in place to lower their contributions through process upgrades or the
elimination of surface discharges.

### 4.3 Summarize and Rank Sources by Loading to Waterbody

The section above described how to quantify the nitrogen loading contributions from in-
dividual sources to a receiving waterbody, either by any of a variety of modeling approaches, or
by the use of measured data. To focus on the most significant loading sources, the loadings should be ranked according to the source load that reaches the waterbody. It is critical to use the load reaching the waterbody because a large fraction of the nitrogen entering a watershed can be stored or transported out of the watershed and never affect the receiving water. This means that loads from each of the multiple pathways for each source must be assessed.

The spatial and temporal variability in the nitrogen loads from each source should also be examined. This analysis can provide useful information to be used in identifying individual source controls, but can also indicate what combination of controls may be most effective in affecting eventual loads to the receiving waterbody.

4.4 Determine Assimilative Capacity of a Receiving Waterbody

Assimilative capacity has been defined as the ability of a natural system to absorb various materials, including anthropogenic wastes, at certain concentrations without becoming degraded (Cairns, 1977). The term was originally used to describe the use of streams to process simple organic wastes such as sewage (Cairns, 1999). Another definition is the ability of a waterbody to cleanse itself; its capacity to receive wastewaters or toxic materials without deleterious effects and without damage to aquatic life or humans who use the water. The assimilative capacity is sometimes referred to as the critical load.

The primary task in determining a waterbody’s assimilative capacity is to set a threshold value for an indicator of degradation. Typically, this threshold is a water quality standard but can be any stated water quality goal. The commonly used indicators to determine nutrient assimilative capacities are chlorophyll $a$ and DO since they are the most noticeable symptoms of eutrophication. Thus, the assimilative capacity of a nitrogen-limited waterbody is the maximum nitrogen load that a receiving water can process without showing symptoms of eutrophication or exceeding some water quality standard.

Various methods have been developed to derive assimilative capacity of streams, lakes, and estuaries with respect to nutrients. These methods typically require significant data collection activities and analysis to complete. Data collection must be adequate to capture the major spatial and temporal heterogeneity within a system. The methods should be capable of resolving relatively small changes in loads or concentrations that can result in significant ecological changes (Howes et al., 2003).

One method of determining assimilative capacity includes evaluation of the effects of increased loadings on selected water quality criteria through mechanistic modeling. These methods involve developing hydrodynamic and water quality models of the waterbody of interest, then comparing results when various levels of loadings are introduced to the system. These models require physiographic data for the system, hydrologic and nutrient loadings, and any additional boundary condition data, all over some representative time period. Candidate water quality models for such applications include QUAL2E, WASP, EFDC, and CE-QUAL.

Another method involves development of empirical relationships between loadings and/or concentrations and the water quality constituent of concern. As for mechanistic modeling, empirical modeling is best applied to systems for which sufficient data exist to allow for evaluation of relationships over sufficient temporal extents and system conditions to discern ecological changes in response to relatively small changes in loads or concentrations. The system should be sufficiently defined with respect to data collection locations so that the assimilative capacity is
representative of the system, not extrapolated from relationships developed from non-
representative data.

A Bayesian empirical approach was developed and applied by Faulkner (2008) to model
the assimilative capacity component of a nutrient TMDL for a stream. In this study, the author
developed prior distributions of nutrient retention based on Monte Carlo sampling of 495 stream
metrics that were obtained via literature review. These prior distributions were then modified
based on data from sites for a particular stream for which distributions for various metrics were
needed, and where measured data were available and thus posterior distributions could be calcu-
lated. The metrics used for the development of the prior distributions included stream discharge,
cross-sectional area, fraction of storage volume to free stream volume, denitrification rate con-
stant, storage zone mass transfer rate, dispersion coefficient, among others. The Bayesian ap-
proach was also used to estimate the confidence of compliance that a given level of retention had
been achieved.

In Florida, the definition of a total maximum daily load (TMDL) established by the
FDEP is “…the maximum amount of a pollutant that a waterbody or water segment can assimili-
ate from all sources without exceeding water quality standards…” (Chapter 403.031, Florida
Administrative Code). Therefore, the TMDL by definition is the assimilative capacity of a wa-
terbody for a particular pollutant.

Florida’s nutrient criteria are currently narrative only and state that nutrient concentra-
tions of a waterbody shall not be altered enough to cause an imbalance in natural populations of
aquatic flora or fauna. Thus, a numeric nutrient-related threshold was needed to represent levels
at which an imbalance in flora or fauna is expected to occur. The impaired waters rule (IWR),
which provides a protocol to determine impairment for TMDLs, provides a numeric nutrient im-
pairment threshold for chlorophyll in streams (20 μg/L) and estuaries (11 μg/L). The thresholds
are not water quality standards and should not necessarily be used as thresholds for TMDL de-
development. The IWR thresholds were developed using statewide average conditions, but the
IWR (Section 62-303.450, F.A.C.) allows the use of alternative, site-specific thresholds that
more accurately reflect conditions beyond which an ecological imbalance could occur in the wa-
terbody (FDEP, 2008).

There are several examples of how TMDLs and by definition assimilative capacities
have been determined in Florida waters. The Lower St. Johns River (LSJR) is located in north-
east Florida, and flows north to discharge near the City of Jacksonville and thereby extending
from euryhaline downstream regions to upstream freshwater reaches. In 2003, FDEP used the
IWR protocol to assess water quality impairments in the main stem of the LSJR and verified that
11 of the 15 freshwater and estuarine segments of the lower river were impaired. The impairment
determination was based on annual mean chlorophyll $a$ concentrations or annual mean TSI val-
ues (indicators), and was corroborated by the evidence of numerous symptoms of eutrophication,
including fish kills, submersed aquatic shoreline vegetation covered in algal mats, excessive epi-
phyte growth, accounts of shoreline vegetation losses, reported reduced recreational fishing qual-
ity, excessive organic sediments, prolonged anoxia, and the presence of potentially toxic
dinoflagellates (FDEP, 2008). FDEP deemed nutrients as the causative agents in the eutrophica-
tion of the lower river. Once the indicator (chlorophyll $a$) and causative agent (nutrients) were
identified, the relationship between the two could be determined.

Previously, the St. Johns River Water Management District (SJRWMD) established a
site-specific threshold for nutrient impairment for the LSJR freshwater zone based on chloro-
phyll \(a\) values (Hendrickson et al., 2003). The threshold established was the maximum algal biomass levels that would:

- Maintain the diversity of the plankton community
- Facilitate the upward transfer of primary production to higher trophic levels (and maintain zooplankton diversity)
- Minimize the potential dominance of detrimental algal species and the production of algal toxins was evaluated

A chlorophyll \(a\) target of 40 μg/L, not to be exceeded more than 10% of the time, was found to protect the river’s aquatic biota. It was also demonstrated that when chlorophyll \(a\) levels rise above 40 μg/L, a shift in algal types occurs and blue-green algae begin to dominate the system, toxic algal species begin to increase, and zooplankton communities begin to decline.

This alternative threshold for the freshwater portion of the river was discussed by the LSJR TMDL Stakeholders Committee and TMDL Executive Committee, and both groups recommended it be used for this TMDL. These groups also recommended that the threshold be applied over a long-term period rather than a worst-case, dry year. FDEP agreed with these recommendations and established the TMDL using the alternative chlorophyll \(a\) threshold and long-term average model output, rather than model predictions for a worst-case year.

To determine the LSJR assimilative capacity, an interconnected suite of basin-wide hydrologic, hydrodynamic, and water quality models was constructed. The models included a hydrologic model that calculates seasonal runoff and nutrient loads from the watershed to the river (PLSM), a hydrodynamic model of the river that simulates the mixing and transport of nutrients in the river (EFDC), and a water quality model that simulates the transformation of nutrients and processes affecting eutrophication in the river (CE-QUAL).

The model was constructed and calibrated, and predictive runs were made to establish the relationship between chlorophyll and nutrients in the river. Incremental nutrient load reductions (25%, 50%, 75%, and 100%) to the river were made to assess the relative influence of anthropogenic nitrogen and phosphorus loads from point and nonpoint sources (Sucsy and Hendrickson, 2004).

The exceedance frequency and duration of the alternative chlorophyll threshold was calculated for each year, along with the estimated reduction in the anthropogenic load necessary to meet the threshold. Based on the long-term (four years) results, the SJRWMD recommended a 30%-reduction in anthropogenic point, nonpoint, and upstream nitrogen and phosphorus loads. The 30% load reduction was accepted as the TMDL and as such is the assimilative capacity for the LSJR (FDEP, 2008).

Empirical modeling methods were employed to develop loading targets for nitrogen loads to Tampa Bay (Janicki and Wade, 1996; Greening and Janicki, 2006). The state’s IWR specifies that if reasonable assurance (RA) can be given to FDEP that existing management programs are addressing impairments, and that there is a reasonable expectation that the programs will prove successful, then the findings provided in the RA application may become a substitute for a TMDL. The Tampa Bay Estuary Program (TBEP) developed a paradigm that addressed the desired seagrass restoration to 1950s levels (Figure 4-8). Chlorophyll \(a\) targets for each bay segment which allow for adequate water clarity to restore seagrasses (Janicki and Wade, 1996). Using regression techniques, the quantitative relationship between chlorophyll \(a\) concentrations and
nitrogen loading was defined (Figure 4-9). Using this relationship, the nitrogen loads consistent with the chlorophyll \(a\) targets were estimated.

![Figure 4-8. Conceptual Basis for Tampa Bay Nitrogen Targets. Reprinted with permission from Janicki, 2010d.]

![Figure 4-9. Relationship Between Chlorophyll \(a\) Concentrations and the Three-Month Cumulative TN Load For Hillsborough Bay, FL. Reprinted with permission from Janicki and Wade, 1996.]

### 4.5 Determine Need for Nutrient Controls

When the carrying capacity has been determined using best available information, the critical loads are compared to existing loads. If the existing loads exceed the critical loads on either a seasonal or annual basis, then the excess load must be reduced using nitrogen control measures.

Even if the existing loads do not exceed the critical loads, it is prudent to predict potential future loads to the waterbody. Future loads may increase due to increased development and land use change, increases to point source discharges or atmospheric deposition, or by changes to climatic conditions. Identifying potential future excess loads will allow resource managers to plan ahead and allocate resources for use in coming years.
This proactive approach may also include adjusting existing water quality or quantity monitoring programs to develop a database of existing conditions that can be compared to data collected in the future. For example, if a monitoring program currently samples in the downstream reach of a river system but it is anticipated that development will spread to the river headwaters, sampling sites can be established in the now undeveloped areas prior to alteration.

Whatever the standard that is used to determine the need for control measures, following this approach will help ensure that available resources are utilized in the most efficient manner practicable. The selection and evaluation of control sources is discussed in the following chapter.

4.6 Identification of Knowledge Gaps

There are significant knowledge gaps both in terms of the identification and estimation of the nitrogen sources and the means by which the linkages between the sources and the receiving waterbody are estimated. The following presents some of these knowledge gaps and uncertainties.

Nitrogen Sources:

♦ Although the timing of application and the rate of application for agricultural and urban fertilizer is provided in literature values, the actual amount of material applied to the land in a given watershed is difficult to quantify. A mechanism for tracking fertilizer use (application rates, weather/season) and BMP use would be beneficial in determining related loads. Methods for tracking or estimating application rates need to be established.

♦ Although the animal waste generation rate for livestock and poultry is provided in literature values, the actual rate and method of application to the land in a given watershed is difficult to quantify.

♦ Information on the number of people who reside within specific sanitary sewer service areas and are hooked up to the collection system and the number of people that continue to use OWTS could be greatly improved. The Florida Department of Health has recently developed a GIS-based data base of OWTS permits with information provided such as the permit status (constructed, inactive, etc.). Because the records are spatially-referenced, site conditions (elevation, soil type, etc.) can be determined and the relative likelihood of failures can be estimated. However, older, unpermitted OWTS are not yet included in this data base. Because it is the older, less rigorously constructed systems that are most prone to failure having this information would be very beneficial. A database of sewer customers and OWTS permittees would be valuable in estimating wastewater N loading.

♦ Accurate quantification of septage sludge production and disposal is typically lacking, and better data collection and development of a database would be beneficial.

♦ Accurate emissions inventories are critical inputs to the atmospheric deposition models used to estimate the spatial distribution and temporal variation in nitrogen deposition, and these are sometimes lacking.

Nitrogen Loads, Linkages and Confounding Factors:

♦ The adequacy of data provided by precipitation and precipitation chemistry monitoring programs is critical to the estimation of nitrogen loads due to atmospheric deposition either empirically or by the application of deposition modeling techniques. Using regional data for this
important nutrient source can lead to inaccurate estimation, and more deposition monitoring sites are needed to obtain accurate loading estimates from nitrogen deposition.

♦ Defensible wet:dry deposition ratios are needed to estimate atmospheric deposition accurately.

♦ Empirical estimation of nutrient loads delivered as channelized flows depends upon both accurate flow estimates and representative water quality characterization. Many important stream channels are not gauged and major programs like the USGS and others are scaling back networks due to budgetary constraints and a diminishing number of cooperators. Simulating surface flow with any certainty is very time consuming and expensive and requires complete and accurate data sets of land use, soils, drainage patterns and conveyances, topography, basin boundaries, precipitation volume, modifications to historical drainage features, among other data. USGS and other agencies responsible for streamflow monitoring should be given adequate funding to maintain and even add monitoring locations for streamflow.

♦ Since significant land use changes and hydrologic alterations can be common, watershed model constructs, if not updated regularly, will provide inaccurate estimates of current flows.

♦ Given the trend in increased reliance of modeling over data collection, it is imperative that the most accurate data practicable be available. Features including land use, drainage networks, and basin boundaries must all reflect current physical conditions within the watershed to support accurate flow estimation.

♦ Accurate basin boundaries are very challenging to establish in some areas such as peninsular Florida. A very flat terrain combined with an extensive canal network and mechanical pumping of canal water make basin boundaries very difficult to define and some boundaries will change between seasons based on flow patterns.

♦ Given that the number of failed OWTS is a critical factor in the estimation of nitrogen loads from these systems, accurate estimates of the number of these systems is essential, and a routine inspection and operating permit system should be established for OWTS. Also, Florida Senate Bill 550, which was passed in June 2010, will require all septic tanks in the state to be evaluated by the Florida Department of Health every five years at the property owner’s expense. The bill is scheduled to go into effect on January 1, 2011, and will require an evaluation of all OWTS, along with an assessment of the system’s overall condition. If needed, a pump out, repairs or replacement may be required.

♦ Attribution of the nitrogen load delivered to a receiving waterbody via channelized flows to the various nitrogen sources is currently not practical and potentially hinders the identification of appropriate controls to address the critical sources.

♦ Application of watershed models that do not incorporate surface-groundwater interactions in areas where such interactions are common will lead to incomplete understanding of the watershed hydrology and as a result contribute to the inaccurate estimation of nutrient loads.

♦ Our knowledge of in-stream, in-lake, and in-estuary nutrient kinetics is insufficient to accurately represent the rates of nutrient transformations, immobilization, import and export in water quality models. Further study should be conducted in these areas.

♦ Model selection should be determined by the specific objectives and questions to be answered as well as the watershed characteristics that significantly influence the estimation of flows and resultant nutrient loads. Many watershed loading models are constructed to simu-
late surface water flows in specific settings such as SWMM for urban areas and GLEAMS for rural and agricultural settings.

♦ Understanding the behavior of nutrients within shallow subsurface waters is limited at best to areas that have been extensively studied. Although the processes are known, the ability to quantify the potential loads from these waters is lacking.

♦ The ineffective management of the data generated by point source facilities by the regulatory agencies has resulted in critical gaps and discrepancies and as a result inaccurate estimation of point source loads.

♦ The estimation of nutrient loads via CSOs and SSOs depends upon difficult and relatively expensive methods and therefore accurate estimates have been difficult to obtain.
CHAPTER 5.0

NUTRIENT CONTROLS:
ARE THERE AVAILABLE CONTROLS
TO ADDRESS MAJOR SOURCE TYPES?

5.1 Potential Controls for the Primary Sources

The issue which we are confronted with is simply – how do we control the major nutrient sources that impact our nation’s water? By focusing on the nutrient sources, we will ultimately have more success with available controls in terms of improved water quality, regardless of whether nutrients are delivered via point or nonpoint sources. Therefore our focus should be centered on controlling sources of nutrients that impact our streams, rivers, estuaries, and lakes. The major cultural sources of nitrogen include:

♦ Human wastes
♦ Industrial wastes
♦ Fertilizer and animal wastes
♦ Natural and atmospheric emissions
♦ Combustion of fossil fuels

These sources can have a variety of delivery mechanisms. As discussed in the previous chapter, nitrogen from human wastes can enter our surface waters primarily as a consequence of either 1) wastewater treatment facilities, 2) combined sewer overflows, 3) sanitary sewer overflows, and 4) onsite wastewater treatment systems (OWTSs). Similarly, nitrogen sources which are derived from industrial wastes enter primarily from wastewater treatment facilities.

Nitrogen sources that are derived from fertilizer and animal wastes enter our nation’s waters primarily as point sources via stormwater discharges and/or as nonpoint sources via stormwater runoff. Nutrient controls for stormwater runoff can occur in a number of different ways, primarily through the implementation of structural and non-structural best management practices (BMPs).

Therefore, for the purposes of this case study, the discussion of nitrogen controls will focus on the following:

♦ Wastewater treatment plants
♦ Onsite wastewater treatment systems
♦ Controls for stormwater runoff
♦ Controls for stormwater discharges
♦ Controls for atmospheric deposition
5.1.1 Wastewater Treatment Plants (WWTPs)

This section discusses the technologies currently available and utilized for nutrient control in centralized wastewater treatment facilities. Table 5-1 presents the potential capabilities of wastewater treatment technologies for treating wastewater effluent using various levels of treatment. Influent wastewater concentrations are compared with different effluent total nitrogen (TN) levels achieved using various treatment technologies as well as the U.S. Environmental Protection Agency (U.S. EPA) proposed in-stream nutrient criteria for Florida’s rivers and streams. If point sources from WWTPs in a watershed are required to meet the proposed in-stream standards, this table illustrates that it will be difficult for utilities to meet these standards with the current accepted limits of wastewater treatment technology.

Table 5-1. Comparison of Nitrogen Levels Found in Wastewater Effluent from Advanced Treatment and Typical In-Stream Nutrient Criteria. WERF, 2010.

<table>
<thead>
<tr>
<th>Water Quality Parameter</th>
<th>Municipal Raw Wastewater, mg/L</th>
<th>Typical Secondary Effluent, mg/L</th>
<th>Typical Treatment With Nutrient Removal (BNR), mg/L</th>
<th>Enhanced Nutrient Removal (ENR), mg/L</th>
<th>Practical Limits of Treatment Technology, mg/L</th>
<th>Florida In-stream Nutrient Criteria, mg/L</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total Nitrogen</td>
<td>25-35</td>
<td>20-30</td>
<td>~ 10</td>
<td>4 to 6</td>
<td>3</td>
<td>Varies$^{1,2}$ 0.824 – 1.798</td>
</tr>
</tbody>
</table>

1 Currently Florida has narrative criteria for total nitrogen.
2 U.S. EPA has proposed in-stream nitrogen criteria for Florida based on regions that include: Panhandle (0.824 mg/L), Bone Valley (1.798 mg/L), Peninsula (1.205 mg/L), and North Central (1.479 mg/L). These criteria were scheduled to be finalized by November 14, 2010.

A large amount of recent research has been done to document available controls that can be used for nitrogen removal with wastewater treatment facilities. In September 2008, U.S. EPA published *Municipal Nutrient Removal Technologies Reference Document – Volumes 1 and 2 (U.S. EPA, 2008c)*. This document presents detailed information on commonly used biological and physiochemical nutrient removal technologies in addition to detailed technical and cost information. Twelve treatment technologies were identified as viable nutrient removal technologies that produce average effluent TN concentrations in the range of 1-17 mg/L.

Out of twelve (12) treatment technologies that were identified as viable nutrient removal technologies, only four technologies (identified in Table 5-2) can provide treatment of wastewater where effluent can meet very low N removal concentration limits of 3 mg/L or less. Typically, these technologies would also require a supplemental carbon source for denitrification.

Table 5-2. Viable Wastewater Treatment Technologies for Effluent Limits of 3 mg/L TN.

<table>
<thead>
<tr>
<th>Treatment Technology</th>
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</thead>
<tbody>
<tr>
<td>4-Stage Bardenpho Process with Filter</td>
</tr>
<tr>
<td>Step Feed Activated Sludge Process with Filter</td>
</tr>
<tr>
<td>Concentric Oxidation Ditches</td>
</tr>
<tr>
<td>Denitrification Filters with Carbon Sources</td>
</tr>
</tbody>
</table>

The average annual total nitrogen effluent concentration for these processes was less than 3 mg/L with a coefficient of variation (CV) of less than 50%. Providing wastewater treatment beyond these limits requires more complex and expensive treatment technologies such as membranes (reverse osmosis and nanofiltration), oxidation (ozone, UV, peroxides) and/or ion exchange.
Florida currently has more than 2,100 domestic wastewater treatment facilities with a total permitted capacity of 2.4 billion gallons per day (Reardon, 2009). About 80 of these wastewater treatment facilities have NPDES permits that authorize discharges to surface waters, including permits for discharges to streams and rivers (Childs, 2009). While many surface water discharge WWTPs in Florida have permitted TN and Total Phosphorus (TP) limits of 3 mg/L and 1 mg/L, respectively, relatively few wastewater treatment plants in Florida have permitted limits lower than those levels (Levesque et al., 2010). However, a survey of Florida plants found 15 plants that were discharging effluent TN levels below 2.5 mg/L. Nine of those facilities were discharging less than 2.0 mg/L TN and 5 were discharging less than 1.5 mg/L TN (Jimenez et al., 2007). Two examples of the best performing facilities are Lee County’s Fiesta Village Advanced Wastewater Treatment Plant (AWT) and Hillsborough County’s River Oaks Advanced Wastewater Treatment Plant.

The Fiesta Village AWT facility is a 5.0 mgd facility which employs an extended air oxidation ditch process followed by denitrification filters with methanol feed. This facility is permitted for a surface water discharge to the Caloosahatchee River with TN limits of 3.0 mg/L (annual average and monthly average) and 6.0 mg/L (daily maximum). Nitrogen is removed in two stages at this facility: 1) an oxidation ditch provides treatment with an effluent down to an average TN of 3 mg/L and 2) the process treatment is followed with denitrification filters which produce an effluent TN of 1.45 mg/L (annual average) with a CV of 28% (Levesque et al., 2010).

The Hillsborough County River Oaks facility uses two activated sludge processes in series to achieve nitrification, followed by denitrification with methanol feed. Deep bed filters are used for final polishing of the effluent stream. The Hillsborough County Plant produces a final effluent TN of approximately 1.5 mg/L (Parker et al., 2009). Both plants produce an effluent containing approximately 0.7 mg/L soluble organic nitrogen, which is assumed to be refractory (Levesque et al., 2010). To achieve lower effluent total nitrogen concentrations would require adding additional treatment processes, at a significant cost, to remove the remaining refractory material.

5.1.2 Onsite Wastewater Treatment and Disposal Systems (OWTSs)

Onsite wastewater treatment systems are used for household wastewater treatment where centralized sewers are unavailable. They are actually point sources, but because of their diffuse location and lack of operational permitting they are generally considered one of the nonpoint sources of nitrogen which can impact water quality. Onsite wastewater treatment systems (OWTSs) serve approximately 25% of the U.S. population and approximately 37% of new residential development (U.S. EPA, 1997).

Approximately one third of Florida’s population is served by OWTS representing approximately 2.5 million systems (Briggs et al., 2007). This number is expected to increase with the rising population in the state. Few OWTSs are designed to remove nitrogen beyond secondary treatment levels. Consequently, nitrogen can reach drinking water wells or surface waters raising concerns over risks to human health and the environment.

Nitrogen and phosphorus removals by conventional OWTSs are highly variable. Nitrogen in raw wastewater and septic tank effluent exists in the organic and ammonium N forms and is oxidized to nitrate-nitrogen by microorganisms in the soil as wastewater percolates through the
unsaturated zone. Some denitrification can take place during this percolation process, depending on the local characteristics of the soil medium.

The Florida Department of Health (FDOH) has listed several OWTSs for nitrogen removal in Florida, under the National Science Foundation/Environmental Technology Verification (NSF/ETV) protocol. The systems tested as part of this program and approved for use by the FDOH demonstrated total nitrogen removals ranging from 51% to 77%, with effluent TN concentrations ranging from 5.2-14.9 mg/L for those systems (FDOH, 2008).

Field evaluations of several onsite wastewater treatment nutrient reduction systems (OWNRS) have been conducted in the Florida Keys for the purpose of evaluating the effectiveness of nutrient removal (Anderson et al., 1998). Results of this work demonstrated that while excellent secondary treatment of wastewater was demonstrated (CBOD₅ and TSS < 5 mg/L), none of the individual systems tested were capable of meeting Florida advanced waste treatment (AWT) effluent standards of 5 mg/L CBOD₅, 5 mg/L TSS, 3 mg/L TN, and 1 mg/L TP. Available technologies for nitrogen reduction with OWTSs include biological nitrification/denitrification treatment processes and natural systems. Nearly all of these treatment technologies can achieve over 50% total nitrogen reduction, however, as nutrient removal requirements increase, fewer technologies become available. Few available treatment technologies have been identified that are capable of consistently reducing influent total nitrogen more than 70%.

The Florida DOH is currently conducting research to develop passive nitrogen reduction treatment technologies for onsite wastewater and treatment systems. These systems utilize a reactive media for denitrification, and preliminary testing has indicated the potential for these technologies to be able to meet the 3 mg/L total N standard (Smith et al., 2008).

Florida Senate Bill 550 was passed in 2010 that requires DOH to create a statewide septic tank evaluation program to ensure that all OWTSs are working properly and to identify failures. This evaluation program will be phased in beginning in early January 2011. Evaluation procedures currently under development will include an evaluation of the septic tank and the drainfield in addition to an overall evaluation of system condition. These evaluations will be performed by registered septic tank contractors, professional engineers, or certified environmental health professionals. This program is oriented toward the prevention of groundwater and surface water contamination resulting from the inadequate maintenance and subsequent failing OWTSs that can contribute to public health concerns as well as nutrient loadings.

5.1.3 Controls for Stormwater Runoff

Stormwater runoff adversely impacts surface water quality through the introduction of nonpoint source pollutants. Stormwater runoff includes both rural and urban nonpoint sources. Rural nonpoint sources are land uses usually classified as agricultural use or woodlands. Nonpoint source inputs of N and P from fertilizer are more highly correlated with waterbody nutrient concentrations in agricultural areas (ex. row crops, CAFOs) than in non-agricultural areas. The amounts of pollutants contained in rural nonpoint sources vary considerably and typically include animal wastes, fertilizers, and pesticides. This section discusses controls for animal wastes and fertilizers since these two sources are considered the largest contributors of nitrogen in stormwater runoff.

Riparian restoration has been considered a cost-effective method for restoring and controlling stream water quality that has been impacted by nonpoint source pollution. Methods of
active riparian restoration include bank stabilization, grazing exclusion fencing, riparian easements, silviculture, and bio-engineering. In developed urban areas, both wetland and riparian areas have been disturbed by construction, filling, channelization and other alteration means. In agriculture areas, riparian areas can be impacted by the overuse of areas for grazing or other agricultural activities. Water quality improvements can occur with the removal, storage, and transforming of sediments, nutrients, and some heavy metals (U.S. EPA, 2005). However, the effectiveness of wetlands and riparian areas for removing nonpoint source pollutants depends on many factors, e.g. soil and vegetation types. Dodds and Oakes (2006) demonstrated that both total nitrogen and nitrate concentrations were strongly related to riparian cover in the subcatchment areas above their respective sampling sites. Additional stream water quality data indicated that riparian conditions were important determinants of water quality even after catchment influences were taken into account.

5.1.3.1 Concentrated Animal Feed Operations (CAFOs)

The U.S. Department of Agriculture (USDA) estimates that confined livestock and poultry animals generate an estimated 500 million tons of manure annually. Based on U.S. EPA estimates, this compares with about 160 million tons (wet weight) of human sanitary wastes produced annually assuming a U.S. population of 310 million (2010) and an average waste generation of about 0.518 tons per person per year. By this estimate, confined animal wastes account for more than three (3) times the raw waste generated by humans in the U.S.

In November 2008, U.S. EPA issued revised NPDES regulations and effluent limitation guidelines for concentrated animal feeding operations (CAFOs) in response to the court order issued by the Second Circuit Court of Appeals (Federal Register, 2008). These revised regulations were issued with the goal of ensuring that CAFOs are properly managing manure by including the requirement for the submittal of Nutrient Management Plans (NMPs).

Recently, U.S. EPA initiated new CAFO rulemaking under the Clean Water Act in February 2010 to reduce water pollution in Chesapeake Bay watershed and more effectively achieve the objectives of the Chesapeake Bay Total Maximum Daily Load (TMDL). In developing the proposed rule, U.S. EPA may consider requiring more information about off-site manure transfers with supporting recordkeeping and more stringent permitting requirements for land application of manure, litter and process wastewater. U.S. EPA plans to propose revisions to its regulations by June 30, 2012, and take final action on the CAFO regulation by June 30, 2014.

Many existing CAFOs utilize surface storage lagoons for the anaerobic digestion of wastes. Contaminants from animal wastes can enter into the environment through pathways that include leakage that occurs with improperly design lagoons, major rainfall events resulting in overflows and runoff, and air emissions from the wastes resulting in atmospheric deposition via dry or wet fallout.

According to U.S. EPA, Florida and all states in Region IV have programs in place to address nutrient management associated with poultry operations. Currently, Florida has a total of 134 Large CAFOs in operation that meet U.S. EPA’s requirement for obtaining an NPDES permits for operation. These include swine, dairy, and poultry operations (USDA, 2009a).

The agriculture industry has focused on educating animal feeding operation owners and operators whereby voluntary actions can be used to minimize potential water pollutants from confinement facilities and land application of manure and organic by-products. To do so, the USDA works with the owners and operators to develop and implement Comprehensive Nutrient
Management Plans (CNMP). The CNMP identifies management and conservation actions that are utilized as soil and water conservation goals, including nutrient management, where agriculture operations are in place.

5.1.3.2 Florida BMPs for Animal Wastes

The University of Florida’s Soil and Water Science Department (2009) has recently conducted funded research entitled “Evaluating Effectiveness of Best Management Practices (BMPs) for Animal Waste Fertilizer Management to Reduce Nutrient Inputs into Ground Water in Suwannee Basin, Phase I & Phase II”. The project is directed toward the implementation of BMPs at the farm level to reduce nutrient loadings to groundwater from agricultural activities and evaluate their effectiveness. This project complements ongoing BMP implementation and evaluation programs that are being funded through the USDA, the Suwannee River Water Management District’s (SRWMD) Surface Water Improvement and Management (SWIM) program, the Florida Department of Agriculture and Consumer Services (FDACS) Agriculture BMP Cost Share Program, and the State of Florida’s Nitrate BMP Program.

This project is focused on finding nutrient management solutions to improve water quality in the Suwannee River Basin where increasing concentration of nutrients are being found in groundwater, spring water, and private drinking water wells. In response to the State of Florida’s TMDL program, the agricultural community and public agencies are implementing a watershed-based process for development, demonstration and implementation of BMPs that will reduce nutrient loadings to the water resources found in the Suwannee River Basin.

5.1.3.3 Agricultural Structural BMPs for Drainage Waters

Natural and constructed wetlands have also been recognized as effective means for nitrogen removal from agricultural drainage waters. Denitrification primarily occurs in the wetland sediments and the periphyton films of the water column where dissolved oxygen is limited and carbon availability is high. Following wetland construction, nutrient uptake is typically high due to the growth rate of plants. Varying degrees of total nitrogen removal have been reported, with rates ranging from 37-65% (Appelboom and Fouss, 2006; Day et al., 2004).

Vegetated treatment systems are another structural BMP which has been effective with agricultural runoff. Properly designed vegetated treatment areas (VTAs) can achieve an average of 70% total nitrogen removal, with some VTA performance results suggesting 100% reduction in situations where soil infiltration of runoff prevented effluent from leaving the vegetated area (Koelsch et al., 2006).

5.1.3.4 Fertilizer Restrictions and Ordinances

In August 2007, the Florida Department of Agriculture and Consumer Services (FDACS) adopted the final statewide Urban Turf Fertilizer Rule which limits the phosphorus and nitrogen content in fertilizers for urban turf and lawns. The FDACS expects a 20-25% reduction in nitrogen and a 15% reduction in phosphorus in every bag of fertilizer sold to the public. This ruling was developed by the FDACS with input from the University of Florida’s Institute of Food and Agricultural Services, the Florida Department of Environmental Protection (FDEP), the State’s five water management districts, the Florida Association of Counties, the Florida League of Cities, fertilizer manufacturers, and concerned citizens. This rule making was a component of the $200 million Lake Okeechobee and Estuary Recovery (LOER) Plan and was considered an essential component to improve nutrient control from nonpoint source loadings using a statewide approach.
The Florida Green Industries Manual (FGIM), published in 2002, provided Best Management Practices (BMPs) for turfgrass and landscape maintenance to be used by professional landscaping firms throughout the state. The FGIM authors acknowledged using the Professional Lawn Care Association of America’s BMPs for this guide. Prior to the publication of the FGIM, St. Johns County and the Village of Wellington were the only two local governments which had promulgated fertilizer restrictions for the purpose of improving environmental quality. St. Johns County’s ordinance was directed toward the improvement of the Guana River basin while the Wellington ordinance was targeted at decreasing phosphorus runoff from ranching operations into the Everglades Agricultural Areas which were undergoing restoration.

FDEP published the first statutory guide in September 2003 and a recent model ordinance entitled “Model Ordinance for Florida-Friendly Fertilizer Use of Urban Landscapes (FDEP, 2010a)”. This model ordinance is a tool for Florida municipal governments to use for reducing nutrient sources from urban landscapes that may impact Florida’s surface and groundwaters.

More than 20 city and county ordinances have since been adopted in Florida for the purpose of restricting the use of fertilizers on lawns. These ordinances rely heavily on education of the public on the maintenance schedules for proper fertilizer applications as well as seasonal restrictions on fertilizer sales. The majority of these ordinances are more restrictive with respect to the summer application of fertilizers than the State of Florida’s model ordinance.

As part of the effort to decrease nitrogen loadings and improve water quality, the Tampa Bay Estuary Program developed a Model Fertilizer Use Ordinance that is being used by several communities in the Tampa Bay region. The estimated baywide reductions in total nitrogen, based on different assumptions for urban runoff and compliance as illustrated in Figure 5-1, presents an excellent example of the impact that a nonstructural BMP can have on a watershed.

![Figure 5-1. Estimated Total Nitrogen Load Baywide Based on Runoff and Compliance Assumptions. TBEP, 2008.](image)
5.1.4 Controls for Stormwater Discharges

Stormwater discharges are one of the primary causes of pollution in lakes, streams, rivers and estuaries throughout the United States. Stormwater problems, unlike wastewater effluent discharges, are not steady state occurrences that are linked with a community’s population. Because of the irregularities associated with weather patterns, particularly in the subtropical climate of Florida, stormwater flows are not predictable on a daily or even a monthly basis. Florida continues to experience periods of explosive development which create new stormwater problems and in many cases, exacerbate existing problems within existing communities. New developments also present challenges resulting in increasing runoff volume and runoff rates, reducing stream baseflow, and increasing pollutant loadings.

Stormwater discharges from urban areas typically include a variety of pollutants resulting from vehicles, animals, lawns, and human activities. The major nonpoint source pollutants found in urban watersheds include sediment, nutrients, trace metals, oxygen-demanding substances, bacteria, hydrocarbons, and chlorides. The major source of nutrients includes organic matter such as lawn clippings, leaves, improper and excessive use of fertilizers, and pet wastes. Phosphorus contributions from auto emissions have been found in urban areas which have heavy traffic. Controlling these sources of nutrients is a key to reducing the impacts that stormwater discharges have on receiving waters.

The important steps that occur with the delivery of nutrients via nonpoint source discharges include: availability, detachment and transport. Breaking up this delivery procedure can help control nutrients from reaching a receiving waterbody. For example, the availability of fertilizer is function of the quality and the method in which it’s applied. Application of excess amounts of fertilizer will result in excess nutrients leaving the ground surface with runoff and entering the groundwater or surface water. Reducing the fertilizer application amount, and even adjusting the timing of the fertilizer application, will provide a means for controlling nutrients (found in nonpoint source discharges) from entering a surface waterbody.

Studies have shown that certain pollutants found in stormwater runoff are contributed primarily by rainwater, particularly nitrogen (Wanielista and Yousef, 1993). Several investigators, including Yousef (1985) demonstrated that total nitrogen in rainfall represented a large fraction of TN found in runoff (approximately 84%). Wanielista (1976) demonstrated that the concentration of total nitrogen found in urban stormwater varied widely (0.01 – 4.5 mg/L) when compared among various metropolitan areas, including several Florida cities.

5.1.4.1 Existing Florida Rules for Stormwater Discharges

In Florida, stormwater can be regulated by the Florida Department of Environmental Protection (FDEP), one of the five water management districts (WMDs) within the State, a local government, or any combination of the above depending on where the discharge is located within the State, the date the discharge was permitted, and the activity responsible for the stormwater discharge. In 1972, the Water Resources Act (Chapter 373, Florida Statues) was enacted for the purpose of establishing permit programs for the regulation of surface water management systems within the boundaries of each water management district.

Late in 1993, the Water Resources Act was amended by consolidating wetland resource, mangrove alteration, and surface water management permits into a single regulatory approval process referred to the “environmental resource permit” (ERP). Rules implementing the Water Resources Act became effective in October 1995 and are required for any activity that can affect
wetlands, alter surface water flows or contribute to water pollution. The regulation of stormwater permitting in most areas of the state is currently handled by with a single permit issued by a single agency, either FDEP or a WMD. The water management districts regulate residential and commercial developments, while the FDEP oversees power plants, wastewater treatment plants and single-family home projects.

An additional complexity in the permitting requirements for stormwater discharges involves the requirement for some discharges to obtain NPDES permits. Starting in 1990, Phase I of U.S. EPA’s stormwater program required counties and cities to secure NPDES permits for stormwater discharges from municipal separate storm sewer systems (MS4s) serving populations over 100,000, construction activity disturbing more than five (5) acres, and ten categories of industrial activity. The Stormwater Phase II Final Rule, published December 8, 1999, expanded the Phase I program by requiring operators of MS4s in urbanized area and operators of smaller construction sites to implement programs and practices to control polluted stormwater runoff.

Palm Beach County, Florida and its co-permittees (41 total) are currently in the renewal process of its Phase I MS4 permit. The re-issuance of this permit includes an increased number of requirements which add to the accountability of each co-permittee. The Draft Permit was submitted to PBC in September 2010 with a Final Permit scheduled to be issued in February 2011. Each permittee will be required to implement a comprehensive Stormwater Management Program (SWMP) that includes pollution prevention measures, treatment or removal techniques, stormwater monitoring, use of legal authority, and other appropriate means to control the quality of stormwater discharged from an MS4. The SWMP must also include controls necessary to prohibit the discharge of non-stormwater into the MS4 and reduce the discharge of pollutants from the MS4 to the Maximum Extent Practicable (MEP). New permit requirements include conduction programs for:

- Litter control
- Street sweeping
- Fleet maintenance yard(s)
- Waster transfer station
- Illicit discharge for commercial areas
- High risk facilities/industrial areas
- Construction sites

Monitoring requirements for structural and non-structural controls area also identified in the Draft Permit. Additionally, permittees must implement a public education and outreach plan, address land development activities through the requirement of state permits and the adoption of ordinances for landscaping and fertilizer usage, and implement a plan to achieve established TMDL reduction goals.

5.1.4.2 New Florida Rulemaking for Stormwater Discharges

To address growing concerns about over-nutrient enrichment of Florida’s surface waters, groundwaters, and springs, the FDEP and the Water Management Districts (WMDs) are currently in the process of developing a new statewide stormwater treatment rule (Chapter 62-347, Florida Administrative Code). This rule is being developed to achieve more uniformity in the
stormwater regulatory arena and to control nutrient loadings from stormwater discharges. The draft stormwater rule is directed toward providing statewide regulatory criteria for stormwater treatment systems which are designed and constructed to control stormwater pollutant loads. Additionally, FDEP and the WMDs are developing a draft Applicant’s Handbook that provides design criteria for identified best management practices, as well as focusing on the development of inspection programs and operation and maintenance programs for stormwater treatment facilities.

As part of the rule development, the FDEP is presently seeking input on the frequency and requirements for inspections, the frequency of recertifying whether a stormwater treatment system operates as designed and permitted, and how the processing of recertification should be administered. The FDEP is also seeking input on the nutrient load reduction effectiveness of various structural BMPs that include:

- Dry detention
- Underdrain filtration
- Wetland treatment trains
- Vegetated natural buffers
- Managed aquatic plant systems

Florida’s current BMP design criteria are outdated and need to be revised to assure that nutrients in stormwater discharges do not cause or contribute to violations of nutrient criteria, either individually or cumulatively.

5.1.4.3 Structural Stormwater BMPs

The 1999 Florida Watershed Restoration Act (FWRA) provides FDEP with funding to conduct research on the design, implementation, and effectiveness of BMPs to reduce pollutant loadings from stormwater discharges and urban nonpoint sources of pollution. Several research projects have been conducted recently, many which focus on water quality improvements and nutrient removal effectiveness. Those projects which have shown promising nutrient level reductions are summarized below.

- **Baffle Box Effectiveness Monitoring Report (GPI Southeast for FDEP, 2010)** This report summarizes the results of stormwater monitoring at four baffle boxes to determine the effectiveness of TN and TP removals. First generation boxes removed only 0.50% TN while second generation boxes were shown to remove nearly 20% TN.

- **Performance Efficiency of the New York Avenue Exfiltration Facility (Environmental Research and Design for FDEP, 2008)** This stormwater retrofitting project was located in Winter Garden, FL where an exfiltration system was utilized for the purpose of demonstrating nutrient removal efficiencies. The project was funded in part by a TMDL Water Quality Restoration Grant. Results of the monitoring effort demonstrated that the exfiltration system reduced TN loads, TP loads, and TSS loads by 83%. TN loadings were reduced from 105.5 kg/yr to 17.9 kg/yr.

- **Hillsborough Filter Pilot Project (D. P. Smith & Associates and Berryman & Henigar, Inc., 2006)** This research project was conducted at the Talifero Stormwater Research site in Hillsborough County for the purpose of evaluating the use of coagulating agents for improving the nitrogen removal efficiency with stormwater treatment BMPs. A zeolite filter pilot
plant was operated over a 216-day period to evaluate its ability to remove nitrogen from stormwater discharges. When operated at a steady filtration rate, the pilot plant produced an effluent ammonia nitrogen concentration of 0.07 mg/L, and removed 93% of ammonia and 32% of total inorganic nitrogen.

A review of TN removal efficiencies for typical stormwater management systems used extensively throughout Florida is presented in Table 5-3. Retention facilities were considered as any method used for infiltrating water into the ground and included retention ponds, stormwater reuse, swales, or the use of exfiltration trenches.

<table>
<thead>
<tr>
<th>Stormwater Management System</th>
<th>Estimated Removal Efficiencies (Percent) for Total Nitrogen</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Wet Retention</td>
<td></td>
</tr>
<tr>
<td>a. 7-day Detention Time</td>
<td>20</td>
</tr>
<tr>
<td>b. 14-day Detention Time</td>
<td>30</td>
</tr>
<tr>
<td>2. Dry Detention</td>
<td>0 - 30</td>
</tr>
</tbody>
</table>

Wet detention stormwater treatment systems are also used throughout Florida for stormwater management. Design criteria for these systems are provided by the St. Johns River and South Florida Water Management Districts (SJRWMD and SFWMD). Performance efficiency pertaining to wet detention ponds according to SJRWMD criteria indicates annual removal rates of 25% for total nitrogen. These removal rates fail to meet the 80% removal criteria stipulated in the Florida Administrative Code (62-40.432). However, achieving 80% removal for nitrogen in stormwater discharges is extremely difficult when using a single BMP. Several BMPs used in series would be needed to achieve this required level of treatment.

Based on a study of removal efficiencies in wet detention ponds, approximately 40% of total nitrogen can be removed in 50 days of retention time. With longer detention times, only about 45% removal of total nitrogen can be achieved (Figure 5-2). The lack of additional removal efficiency is possibly due to phosphorus limitation or that irreducible concentrations of total nitrogen have been achieved during that time period (FDEP, 2007). Regardless, this demonstrates that stormwater treatment facilities currently permitted for use in Florida are most likely unable to meet proposed total nitrogen concentrations for discharges from some land use categories in Florida.
5.1.4.4 Non-Structural Stormwater BMPs

Non-structural stormwater BMPs used for improving water quality in stormwater discharges include institutional controls, pollution prevention methods, educational programs, and regulatory controls. The benefits from using non-structural BMPs include a number of factors such as cost, coverage, economic incentives/disincentives, potential effectiveness, community participation, flexibility, and secondary benefits (Taylor and Wong, 2002).

FDEP has a wide range of publications, handouts, videos, and training aids that are used for assisting with the implementation of various non-structural BMPs that can used in urban area, golf course, agricultural, silviculture, lake management, and onsite wastewater treatment and disposal (FDEP, 2010b). Limited data is available on the effectiveness of individual BMPs once implemented.

5.1.4.5 Stormwater Maintenance Programs

Stormwater management facilities, conveyance or treatment systems, must be maintained so that they can operate as originally intended and designed. However, typically in Florida, most communities cannot keep up with the maintenance demands which ultimately reduce the capacity of their stormwater conveyance facilities. Inadequate routine maintenance has been identified as one of the factors repeatedly contributing to major and nuisance flooding events, which ultimately contribute to water quality degradation.

The majority of these problems are related to the lack of capital and operational funding dedicated to stormwater management. To effectively operate stormwater management systems, which will ultimately lead to improve water quality, communities must be able to fund their programs that will allow for renewal and replacement of stormwater treatment facilities, maintaining existing facilities, and acquisition of the land and rights-of-way required to facilitate identified capital improvements.
5.1.4.6 Water Quality Trading

Water quality trading (WQT) is an approach supported by U.S. EPA for controlling pollutants from multiple sources that collectively impact water quality conditions in a designated waterbody. U.S. EPA has supported the concept and implementation of water quality trading for several years, including the watershed-based trading efforts on the Tar-Pamlico River (North Carolina), the Long Island Sound, the Chesapeake Bay, and the Lower Boise and Snake Rivers in Idaho. Several water quality trading markets that are currently under operation focus on either phosphorus or nitrogen-based trading.

Revisions to the Florida Watershed Restoration Act authorized water quality trading, but limited its use to a pilot project in the Lower St. John’s River Basin as part of the initial rulemaking process. General requirements for Florida’s water quality trading stipulate that credits generated by a point source must be measured while credits generated by a nonpoint source can be estimated. Activities that can be utilized to generate credits include installation of water pollution control equipment, operational changes that reduce the load of nutrients discharged, implementation of structural nonpoint source management controls, and the installation, operation and maintenance of drainage projects to control stormwater. FDEP is scheduled to report on the pilot program to the Governor and legislature in October 2010.

5.1.5 Controls for Atmospheric Deposition

Under the 1990 amendments to the Clean Air Act (CAA), legislation was put into place that addressed three major threats to the nation’s environment and the public health of the American people – acid rain, urban air pollution, and toxic air emissions. Ozone, the most widespread and persistent urban pollution problem, and nitrogen oxides (NOx) are both well documented examples of air pollution problems currently existing in the United States. Transportation sources account for an estimated 55% of NOx emissions while stationary sources (power plants and industries) account for remaining 45% of NOx emissions. The 1990 amendments included specific requirements for reducing emissions of nitrogen oxides. U.S. EPA’s 2006-2010 strategic plan calls for reducing NOx emissions from mobile sources by 3.7 million tons from the 2000 level of 11.8 million tons. An example of Florida’s reduction of NOx emissions from a stationary source is presented in the remainder of this section.

Tampa Electric (TECO) entered into binding legal agreements with the U.S. EPA and FDEP in 1999 and 2000 to reduce NOx emissions from its Gannon and Big Bend facilities which are located near and on Tampa Bay. The nitrogen emissions from the facilities result in nitrogen loadings directly to the surface of Tampa Bay, and to the bay via transport of deposition from the watershed. Reduction of the atmospheric emissions was expected to lead to reduced nitrogen loadings to Tampa Bay.

In December 1999, TECO and FDEP reached an agreement requiring that TECO add NOx controls, or shut down or re-power the coal-fired units at the Gannon Station and Big Bend plants by May 2010. This agreement also called for the shutdown of three of the Gannon Station units, while the other three units were to be switched to natural gas.

In February 2000, TECO and the U.S. EPA settled the November 1999 federal litigation, providing significant additional details for the implementation requirements of the agreement. The agreement featured the conversion of the TECO Gannon Station facility from coal to natural gas, with Gannon Station Units 1, 2, and 6 shutting down in 2004 and Units 3, 4, and 5 converting to natural gas. The repowered Gannon Station was renamed the Bayside Power Station. The
agreement also called for major reductions in NOx emissions by 2010 from the TECO Big Bend Station, to be achieved by installing NOx controls.

During 1997, the TECO Big Bend and Gannon facilities accounted for approximately 80% of the stationary point source atmospheric nitrogen emissions from Hillsborough and Pinellas counties (FDEP, 2000; TECO, 2000). One of the results of the activities called for in the federal agreement for the TECO facilities involves the reduction in NOx emissions from the two facilities by approximately 90% between 1997 and 2010 (TECO, 2000). These reductions alone constitute a 46% reduction in atmospheric NOx emissions from Hillsborough and Pinellas counties compared to 1997 levels.

The total reduction in TN loading to Tampa Bay due to the repowering and selective catalytic reduction (SCR) system installation was estimated by the U.S. EPA for the Tampa Bay Estuary Program (TBEP) (Dennis and Arnold, 2007). The CMAQ-UCD atmospheric deposition model was implemented to assist the Tampa Bay TMDL implementation planning process by providing estimates of current and future nitrogen deposition to the bay and its watershed. The model was run considering estimates of reductions in N deposition associated with power plant upgrades to the Gannon and Big Bend power plants. Repowering of the two power plants resulted in estimated reduction of TN loading to Tampa Bay by about 15 tons/year during each year of the eight year period from 2002 to 2010, assuming linear reductions over time, or about 120 tons/year by 2010.

The Bayside Power Station (formerly Gannon facility) was repowered in 2003-2004. As of June 2010, the SCR control systems to Units 1 through 4 at the Big Bend facility were completed. According to a water quality assessment, the SCR system would reduce the total nitrogen load to Tampa Bay from the Big Bend Station. After installation of the SCR on Unit 4 (the first unit with SCR installed, in June 2007), net nitrogen load to the bay for 2007 was reduced by more than 19 tons/year from those projected without SCR installation (ECT, 2008).

Installation of the SCR units at the Big Bend facility results in removal of NOx from the air emissions, but some of this N is then discharged to the bay via surface water discharge. Based on discussions with TECO, the Tampa Bay Nitrogen Management Consortium (TBNMC) provided an interim allocation for surface water discharge of 56.5 tons TN annually for the Big Bend facility. It is expected that the discharge from the facility will be at least 35 tons/year, with the additional 21.5 tons allocated only through 2012 while the company determines the discharge loads actually occurring as a result of the SCR project completion and operation (TBEP and Janicki Environmental, 2010). Still, the net result of the repowering and SCR installation will be a net removal of nitrogen load from Tampa Bay of at least 60 tons/year.

5.2 Identification of Knowledge Gaps

This section identifies data and knowledge gaps pertaining to the specific nutrient controls that were identified and discussed in the previous sections.

5.2.1 Wastewater Treatment Facilities

U.S. EPA’s February 2008 report on “Emerging Technologies for Wastewater Treatment and In-Plant Wet Weather Management” identified the need for research into technologies that are capable of improving nutrient removal and finding processes that can achieve low total nitrogen levels. Effluent dissolved organic nitrogen cannot be removed using current technolo-
gies for biological wastewater treatment (excluding reverse osmosis) since they are not subject to degradation. New technologies are needed to address the treatment and removal of effluent dissolved organic nitrogen from wastewater effluent streams if lower nitrogen levels will be required for wastewater treatment facilities.

Since many wastewater treatment plants would need to construct new process facilities to meet effluent goals for nitrogen removal, supplemental carbon sources would need to be considered. As a result, more research is needed on the appropriate use of supplemental carbon. Side-stream treatment was suggested as a research area that should be further explored in order to reduce the use of supplemental carbon and aeration energy.

Additional research is needed on technology and design procedures for nutrient removal processes based on different treatment plant sizes and discharge limits. Additionally, municipalities and consultants need to better understand the impact of limits of technology for nutrient removal on sludge treatment and reuse and the implications of the potential for additional solids volume produced as a byproduct of new and improved technologies for nutrient removal.

5.2.2 Controls for Stormwater Runoff

Riparian areas play an important role in the reduction of nonpoint source pollution by intercepting surface water runoff, subsurface flows, and some groundwater flows. In effect, wetlands and riparian areas buffer receiving waters from the impacts resulting from pollutants typically found in nonpoint source discharges. When wetlands and riparian systems are damaged or removed, nonpoint source pollution has a greater potential to reach receiving waters. In some cases, degraded wetlands and riparian areas can contribute to nonpoint source pollution (U.S. EPA, 2005).

Best management practices (BMPs) utilized and recommended for agricultural control of nonpoint sources are not closely monitored for nutrient control effectiveness. Basinwide systems utilized for implementing nutrient control strategies need to be monitored, results compared across geographic regions, and information shared to affect change. Site-specific techniques need to be identified to improve the efficiency of using nutrients in fertilizers and manure applications.

There is clearly no best treatment solution for manure wastes resulting from CAFOs. The development of a range of options is needed for this point source that can be integrated, adapted, and implemented based on the specific agricultural operation and location. Furthermore, there is a need for the development of environmentally superior technologies that recycle nutrients from manure wastes that will allow for both an economic and environmental benefit.

Recent findings from a national workshop indicate there is a “surprising lack of information about the environmental impacts of CAFOs to adjacent lands and receiving waters” (Burkholder et al., 2007). An understanding of the impacts of many CAFO pollutants on receiving waters and their associated ecosystems continues to be a critical data gap. Examples include “the poor understanding of the impacts of fecal bacteria and other microbial pathogens from CAFO waste effluent contamination on aquatic communities; impacts of antibiotic-resistant bacteria created from CAFO wastes on aquatic life; impacts of organic nutrient forms preferred by certain noxious plankton; impacts from the contributed pesticides and heavy metals; and impacts from these pollutants acting in concert, additively or synergistically.”
The USDA’s Agricultural Research Service (ARS) is performing ongoing research into several technologies that include improved solid-liquid separation, enhanced biological nitrogen treatment, phosphorus extraction, anaerobic ammonia oxidation, material science and green oxidant application development. Investigations are also being conducted by ARS to improve our limited knowledge on the biology of anaerobic lagoons and the development of technologies that can be used to retrofit existing manure treatment systems. Additionally, research is also being conducted on natural treatment technologies that include constructed wetlands, floating wetlands, and riparian zones to effectively manage nutrients.

5.2.3 Controls for Stormwater Discharges

The National Research Council (NRC) states that the “lack of rigorous end-of-pipe monitoring, coupled with U.S. EPA’s failure to use flow or alternative measures for regulating stormwater, make it difficult to develop enforceable requirements for stormwater discharges” (NRC, 2009).

Florida’s current BMP design criteria need revision to assure that nutrients in stormwater discharges do not cause or contribute to violations of nutrient criteria, either individually or cumulatively. FDEP recognizes this deficiency and is working toward the development of new stormwater rules that focus on BMP design criteria.

Pollutant load reductions from structural controls have been well studied, but estimating load reductions from non-structural controls is much more difficult. Much of this difficulty stems from the dependence of a non-structural control’s effectiveness on voluntary actions (e.g. compliance with a fertilizer ordinance). Additionally, many non-structural BMPs have disadvantages that include the uncertainty associated with their performance and ability to improve stormwater quality and the corresponding health of the receiving waters (Taylor and Wong, 2002). Efforts are needed to quantify the effectiveness of non-structural controls.

Stormwater permits lend themselves to a great deal of discretion to let communities set their own standards and self monitor. The NRC states that radical changes to the current regulatory program are needed before meaningful regulation of stormwater discharges will have any impact on improving water quality.

5.2.4 Controls for Atmospheric Deposition

There are several knowledge gaps, or sources of uncertainty, involved in estimating the loadings to a waterbody from atmospheric emissions. Estimates of deposition are dependent upon some method to translate atmospheric emissions to deposition on the waterbody and to its watershed.

Emission sources are typically classified as point sources (factories and power plants), area sources (small fixed emitters), mobile sources (automobiles, planes, ships, trucks, rail), and biogenic sources (vegetation and microbial activity). For the point sources, accurate emissions data are often available. However, estimates of emissions for the other three types of sources are necessary as well, and require a set of assumptions to derive. For example, emissions from mobile sources are often based on population figures, biogenic source emissions are based on estimates of biological activity, and area source emissions are based on surveys of small emitters. Uncertainty exists in all these estimates, but most particularly in estimates of emissions from mobile and biogenic sources.
An acceptable estimate of emissions must then be translated into estimates of deposition to the watershed and the waterbody. This is often accomplished by application of a deposition or air quality computer model which incorporates meteorological data with emissions data and provides estimates of deposition via a series of accepted algorithms. Depending upon the objectives of the application, spatial resolution may be very large (for a model domain including the entire continental U.S.) or more resolved (for a particular watershed). Results are reported as average deposition within a particular grid section of the model. The more resolved the model domain, the more locally specific the results, but it should always be kept in mind that the results are averaged over some area and time.

The total contribution of atmospheric deposition to a waterbody is the sum of the deposition directly to a waterbody and that which has passed through the watershed. Various studies have been performed to assess the percentage of deposition load reaching the receiving water, but these are often specific to a selected waterbody. Generalizations based on land use type are available (U.S. EPA, 2001), but this pass-through is subject to a large uncertainty for any particular system. These uncertainties can be addressed via utilization of watershed-specific water quality models, but the inputs to these models also have associated uncertainties with regard to loading sources, chemical and biological rate processes, and travel times.

Throughout the process of estimating the relationships between atmospheric emissions and eventual loading to a waterbody, there are many sources of uncertainty because of lack of specific knowledge. The ability exists to improve on the knowledge needed for this process, but the costs of filling knowledge gaps must be weighed against the objectives of the overall effort prior to mapping out a plan to develop the necessary information.
CHAPTER 6.0

WATER QUALITY MANAGEMENT DECISIONS: ARE ALL BENEFITS AND COSTS OF NUTRIENT CONTROLS CONSIDERED?

6.1 Evaluating the Benefits and Costs of Nutrient Management to Receiving Waters

A significant number of waterbodies in the United States continue to suffer from water quality impairment due to excess nutrient loading. Nutrient over-enrichment affects many of Florida’s major waterways including Tampa Bay, the St. Johns River, Lake Okeechobee, the Florida Keys and the Gulf of Mexico, among others. These waterbodies provide valuable ecosystem services to local and regional households and businesses, including those who benefit from commercial fishing, recreational fishing, water-based recreation, and tourism. The aesthetics provided by these waterbodies increase property values and human well-being. Decaying plant and animal life caused by nutrient enriched water reduces water quality, creates odors and is visually unappealing. In addition, this water quality impairment can affect surface and groundwaters used for public water supply causing health problems and the need for additional water treatment.

Water quality management decisions required to restore nutrient impaired waterbodies are complex and extend far beyond the technical and scientific decisions. The development and implementation of nutrient control strategies must also consider economic, social and environmental considerations. In the past, water quality improvement project implementation has been based on a simple cost comparison of the top technically ranked project alternatives. In reality, there are many costs and benefits associated with water quality improvements, and the best water quality management decisions are based on consideration of all benefits and costs.

The economic value of avoiding nutrient enrichment of waterbodies is likely to be substantial and includes the following income, use, non-use and property values.

1. **Value to the economy** measured by household income (salaries, wages, rents and profits) generated from commercial fishing; water-based tourism (swimming, diving/snorkeling, power and non-power boating, recreational fishing, bird watching, picnicking, beautiful/unique environment, etc.); and retirees who reside in close proximity to the waterbody.

2. **Resident wealth and well-being** measured by the impact of the waterbody on property values as it provides visual aesthetics and recreation opportunities.

3. **Use value of the recreation activities** provided by the waterbody which is defined as the recreators’ maximum willingness-to-pay to enjoy these recreation activities currently and in the future. This willingness to pay includes the expenditures made to participate in the recreation activity, including travel costs, if any, plus the additional
value that the recreator receives from the activity that he/she does not have to pay. Economists call this additional value “consumer surplus” and its meaning is similar to how a business views its profits. Economists call profits “producer surplus”.

4. **Non-use value of the ecosystem** supported by the waterbody is defined as the maximum willingness of all households to pay to maintain or improve the ecosystem services provided by the waterbody. This non-use value includes benefits to future generations and the benefit of knowing that the ecosystem exists and is protected. These benefits may include the value of the waterbody’s potential to provide research opportunities to find new cures for disease and the educational value of the waterbody. This value is separate from the value of actual or future use by the individual which is included in the use value described above.

5. **Value as a potable water supply source** which is measured as the maximum willingness of households and businesses to pay for the water from that source. This value is statistically estimated using water demand functions and would be constrained by the cost of the alternative water supply source that could be used instead.

6. **Avoided cost of diseases** that may be transmitted by drinking water from the waterbody, or swimming in the waterbody, or consuming the seafood products harvested from the waterbody.

7. **Replacement cost of the ecosystem services** provided by the waterbody such as improved water quality (for potable water supply use) and nutrient cycling as might be provided by marshes, mangroves, seagrasses and other aquatic plant life.

8. **Value of the commercial fish harvested** from the waterbody which is the price paid for the species by consumers and the “consumer surplus” enjoyed by the consumer as he/she consumes the species.

In Florida, many of the economic values listed above have been estimated for the following waterbodies and marine resources: the Indian River Lagoon; Biscayne Bay; and the coral and artificial reef systems off the coasts of Martin, Palm Beach, Broward, Miami-Dade and Monroe counties (Hazen and Sawyer, 2001, 2003, 2005, 2008). Other economic valuation studies have been conducted for waterbodies including the Charlotte Harbor National Estuary (Hazen and Sawyer, 1998) and other smaller waterbodies in Florida (for example, Hazen and Sawyer, 1997).

Management measures that reduce the amount of nitrogen (and phosphorus) in waterbodies can preserve and increase these economic values. Economists recommend that this chosen level of nutrient reduction should be based on maximizing the net benefits of nutrient management, where net benefits are the benefits to all persons and entities minus the costs to all persons or entities now and in the future. The benefit of the measure is the value of the nutrient reduction as it affects the ecosystem services and includes the values listed in (1) through (8) above, as applicable, in addition to any other benefits of the measure that are identified. The cost of nutrient reduction includes the change in the financial condition of those who pay for the alternative, such as the costs paid to implement the alternative, and any negative external effects to other persons and entities that are caused by implementing the alternative.

For some types of benefits and costs, dollar values of these benefits and costs can be relatively easy to estimate. For other types of benefits and costs, the amount of research necessary to estimate dollar values is relatively high. In this case other non-monetary measures can be used that are relatively inexpensive to estimate and that reflect the dollar value.
In practice, the chosen level of nutrient reduction in a waterbody is based on achieving the most desirable level of ecosystem services as determined by the regulatory agency. The extent to which benefits, costs, and public input are considered by the agency in choosing the desired level of nutrient reduction and in choosing management alternatives varies among waterbodies. When benefits, costs and public input are not sufficiently considered, the actual design and implementation of alternatives to reduce nutrient levels in waterbodies can result in choices where the costs are greater than the benefits and/or where the most cost-effective alternatives are not implemented.

Ecosystem restoration and protection projects are often a complex web of technical, legal and economic issues involving numerous stakeholders. Ecosystem restoration challenges that appear to lie in a scientific and engineering realm have erupted into heated political battles. Affected persons recognize that their livelihood may be affected by ecosystem projects and are taking a much more active role in the decision-making process. Affected entities such as utilities and businesses fear that increasing costs will result from implementing nutrient reduction strategies where the benefits are perceived to be limited or non-existent. Stakeholders such as affected communities, environmentalists, developers and industry groups are taking full advantage of the open public forum where the fate of many ecosystem projects is ultimately decided. Accommodating the diverse interests of stakeholders may be one of the biggest challenges in implementing ecosystem restoration and protection projects.

With all of these considerations, a benefit-cost evaluation model to evaluate and rank nitrogen control alternatives is described in this section. An application of this model is provided in a study of phosphorus control alternatives for the Lake Okeechobee watershed in 2003 for the South Florida Water Management District (Hazen and Sawyer, 2003). The alternative included on-farm and regional methods to reduce phosphorus load. The evaluation model considers the benefits and costs of each alternative to all persons and entities that would be affected by the alternative, either directly or indirectly. The best available information is used to assess current and future benefits and costs associated with the alternative. This benefit-cost evaluation model provides a structured approach to: 1) achieve an apples-to-apples comparison of nitrogen control alternatives; 2) identify data and information gaps; 3) evaluate the impact of uncertainty on the ranking of each alternative; 4) provide useful input into the design of nutrient reduction programs; and 5) update the conclusions as new data and information become available. As a result of using this evaluation approach, the benefits and costs of nutrient reduction alternatives are better understood; politically sensitive issues can be put into perspective; and stakeholders’ interests are placed on a level playing field.

6.2 Previous Uses of Evaluation Models

A traditional life-cycle cost analysis (LCCA) considers the cost impacts to those who will implement the project. Usually costs and benefits to others (also called external costs and benefits) and the uncertainty of project outcomes is not considered in the evaluation. The result of this type of analysis may not always be the best fit for a community’s sustainability values and goals. A complete analysis of alternatives should include the costs and benefits of the project to all affected persons and entities regardless of location and should include social, cultural and environmental benefits and costs.
Several such approaches have been used to assist water management decision-making including Benefit Cost Analysis (BCA), Triple Bottom Line (TBL) assessment, and other Multi-Criteria Assessments. These types of analyses along with risk and uncertainty analyses are similar, in that they are used for decision-support and help balance risks and rewards while keeping in mind sustainability goals and stakeholder expectations (Parker and Williams, 2008).

A BCA attempts to quantify all costs and benefits including externalities and includes economic, environmental and social (employee and community) costs and benefits. The analysis considers operation and life cycle maintenance costs. Benefits and costs are discounted over the lifetime of a system into present values (Pearce et al., 2006). Sensitivity analyses can be used to vary individual assumptions to determine their impact on the overall analysis (Parker and Williams, 2008).

The TBL assessment originated as a reporting tool but can be used as a decision-making framework which incorporates economic, social and environmental considerations (Lai et al., 2008). TBL or other multi-criteria assessments do not attempt to quantify all externalities when time and/or money is insufficient to estimate a dollar value of each type of external benefit and cost. These benefits and costs are still included and evaluated in a quantitative or qualitative manner using the best available information. Criticisms of the various multivariate approaches include issues regarding double counting of benefits and costs when there are interdependencies among criteria or when criteria are redundant (Lai, et al., 2008). In addition, some complex analyses may be seen by non-experts as lacking transparency (i.e. “black-box”). These issues can be adequately addressed through proper selection and measurement of each criterion.

An example of a Benefit Cost Analysis (BCA) for a nutrient impaired waterbody is the study prepared for the South Florida Water Management District on phosphorus management for Lake Okeechobee (Hazen and Sawyer, 2003). This study developed an evaluation model of 10 criteria and used it to score and rank 18 combinations of phosphorus controls to reduce the amount of phosphorus entering Lake Okeechobee. The controls included specific agricultural best management practices and regional phosphorus treatment systems. The benefits and costs to the District, landowners and the regional economy were quantified for each phosphorus control combination and uncertainty analysis was conducted. This study was used in the development of the 2004 Lake Okeechobee Protection Plan, a state mandated program that was implemented by multiple agencies (Johns and O’Dell, 2004).

Another example is the Sonoma Valley County Sanitation District (SVCSD) Recycled Water Project in California which evaluated the use of reclaimed water for grape irrigation. The relevant water agencies (SVCSD, Sonoma County Water Agency, Valley of the Moon Water District and the City of Sonoma) conducted an analysis that included both a qualitative Triple Bottom Line analysis and a BCA to evaluate the full cost of the project (Atapattu et al., 2010). The BCA monetized the financial, social and environmental benefits of the project in comparison to the “without-project” alternative. The analysis demonstrated that the project had numerous qualitative and quantitative benefits and helped to identify the beneficiaries beyond the identified agencies.

There are also several examples that used multi-criteria assessment in municipal water/wastewater planning and evaluation of alternatives. In 1999 the City of Los Angeles, California developed an Integrated Resources Plan to manage wastewater, runoff, water recycling and conservation (CH2M HILL and CDM, 2004). The City sought to meet sustainability goals with respect to environmental quality (water, air quality), resource use (water conservation, water reuse,
biosolids reuse, energy, materials, land area) and social impacts (environmental justice, job creation, land enhancement, education) (Kennedy and Holmes, 2006). Four water management alternatives that were improvements to the City’s wastewater treatment facilities were evaluated with respect to their sustainability and their costs. Care was taken to develop evaluation criteria that: measured sustainability; were comprehensive to the alternates being evaluated; were relevant to stakeholders; were relevant over time; and were convenient to use (Vos et al., 2005). The criteria used to evaluate the alternatives were social impacts, economic development, natural resource consumption, environmental pollution, urban ecology, system flexibility and adaptability, and institutional capacity (political will, staff training and commitment, public education, pricing signals, data development, and interagency partnerships). The analysis also used a Life Cycle Assessment to determine the “indirect impacts” of the wastewater treatment plant alternatives (CH2M HILL and CDM, 2004).

Other examples of multi-criteria assessments include King County, Washington’s planning for the Brightwater Wastewater Treatment Facility, and the City of San Francisco, California’s Local Water Supply Alternatives Analysis and Sewer System Master Plan. In both of these studies, sustainability was evaluated simultaneously with the technical and economic analyses (Kennedy and Holmes, 2006).

Numerous European and Australian examples of assessment methodologies exist that use a TBL or similar approach in water management decision-making. Examples include the Sustainable Water Industry Asset Resource Decisions (SWARD) project (Ashley et al., 2004, 2005, 2008), the DayWater decision-support system (Ellis et al., 2005), the IBOS Regenwater decision-support tool (Kluck et al., 2005) and the Water Cycle Management for New Developments (WaND) decision-support system (Makropoulos et al., 2005). The SWARD project included the typical three TBL categories with a fourth technical category (Ashley et al., 2008). Another approach called the Quadruple Bottom Line is used in New Zealand and includes the typical three TBL categories plus a cultural and/or governance issues category (Feeney et al., 2009). Taylor and Fletcher (2005) provide several more examples using similar analyses in Australian stormwater projects.

6.3 Evaluation Criteria for Initial Screening of Nitrogen Management Technologies

The following preliminary evaluation criteria could be used to screen a large list of potential nitrogen management methods and technologies in order to develop nitrogen management alternatives that would be further evaluated. These criteria are as follows.

1. Sufficient information – There is sufficient information to allow for an adequate evaluation of benefits and costs associated with the proposed technology or nutrient management method.

2. Nitrogen reduction potential – The technology or management method has the potential to successfully reduce nitrogen loads to the waterbody.

3. Confidence of Sustained Performance – The technology has been demonstrated to be effective in reducing nitrogen loads and its operational theory and application indicates that the technology is likely to reliably reduce nitrogen loads on a long-term basis.
4. Timeliness – The technology is timely in that it is available and is, or could be, considered in planning programs.

5. No Significant Negative Side Effects – The technology will not have a significant negative side effect (e.g. increased greenhouse gas emissions) that is thought to be detrimental to the region from the perspectives of public health and safety and the mission of the regulatory agencies.

For the management method or technology to be considered for inclusion in the benefit-cost evaluation of nitrogen control alternatives, the method or technology must pass all of the identified screening criteria.

### 6.4 Criteria that Consider All Benefits and Costs of Each Nitrogen Management Alternative

An example nitrogen management alternative for a watershed could include one or more of the following actions:

- Wastewater treatment technologies implemented by specific utilities
- Connection of onsite wastewater treatment systems (OWTS) to a centralized wastewater treatment plant, or development of a management program for OWTS that included nitrogen reduction technologies or measures
- Stormwater management systems installed and maintained by a government or by developers, businesses and farms
- Nonpoint source runoff controls and management methods including, but not limited to fertilizer restrictions or ordinances limiting the amount of nitrogen fertilizer that can be applied
- Best management practices, including the use of nitrogen reducing technologies, implemented by households, businesses, and farms
- Zoning laws that dictate the types of land uses that may exist on and near waterbodies
- Controls for atmospheric deposition
- Other identified methods

Evaluation criteria are developed that reflect the benefits and costs of the specific alternatives. The criteria presented in this section are for illustration.

Each alternative is evaluated with respect to evaluation criteria that consider the benefits and costs of the alternative’s action items. The description of each alternative must include the types of persons and entities that: 1) will pay for and 2) will implement each action item included in the alternative. This information is necessary to evaluate some of the benefits and costs of the alternative.

All benefits and costs would be relative to a specific set of baseline conditions that generally reflect a “no action” alternative. For example, the current and future land uses within the study area under the “no action” alternative and all other laws, policies and regulations that could affect the benefits and costs should be specified.
The study period over which the benefits and costs will be evaluated should be specified and should be long enough to consider the full nitrogen reduction benefits of alternatives that are phased in, or where time is needed to realize the full benefits. Alternatives with useful lives that are reached before the end of the study period should be replaced. Salvage values should be estimated and placed in the last year of the study period if the useful life of the alternative has not been reached. Even though it is difficult to forecast anything after five years, a long time period is needed to adequately evaluate the benefits and costs of nutrient reduction technologies. Applying the best available information over a long time period will provide more information to decision makers than focusing only on a five year period. For the illustrative purposes of this report, the evaluation period is 50 years. Other studies have used evaluation periods of 20-30 years.

Each nitrogen management alternative would be evaluated with respect to each of the criteria described in the following sub-sections. These criteria were developed assuming that estimates of the dollar values of the items listed in (1) through (8) at the beginning of this chapter are generally not available. The criteria described herein demonstrate how to consider benefits and costs without using dollar values. Where estimates of the dollar values are available, they should be incorporated into the criteria. The evaluation model can be applied to various combinations of nutrient reduction actions to help decision makers evaluate which combination may best meet the nitrogen removal and sustainability goals.

A. Nitrogen Reduction
1. Average annual change in the amount of nitrogen entering the waterbody in pounds per year during the study period (or other weight measure such as kilograms) due to the alternative chosen for implementation.
2. Expected nitrogen concentration leaving the source property after the chosen alternative is implemented as measured in parts per million (ppm) or mg/L.

B. Cost-Effectiveness
1. Present value cost per pound of nitrogen removed from the waterbody during the study period. The present value should include all capital, operation and maintenance costs to implement the alternative over the study period.

C. Other Benefits and Costs
1. Present value change in State or regional income to the entire economy
2. Affordability of wastewater and storm water utility bills
3. Success in achieving other water management objectives
4. Water supply benefits
5. Acres of increased/improved wildlife habitat, including water-based ecosystems
6. Change in recreation opportunities
7. Change in property values
8. Effect on greenhouse gas emissions

D. Risk and Uncertainty Measures
1. Engineering / Technological track record
2. Permitting uncertainty

All of the 13 criteria can be applied to point sources and nonpoint sources of nitrogen.

The answer associated with each alternative for some or all of the criteria could be assigned a “moderate” or “high” confidence level that refers to the level of uncertainty associated with the data and information used to derive the answer. Moderate means that the studies used to obtain the answer provided reservations about the accuracy of the results or that insufficient data and information exists to provide a high level of confidence. The high level implies that the data and information used to develop the answer are reasonable for a planning-level analysis. It is expected that there would be no answers associated with a “low” confidence level because any such action items associated with an alternative would have been screened out during the preliminary evaluation.

6.4.1 Nitrogen Reduction – Average Annual Change

This criterion is the average annual value of the estimated amount of nitrogen reduction at the waterbody that can reasonably be expected from the alternative each year during the study period. Variations in this criterion definition may be made to suit the purposes and goals of the alternatives evaluation.

The method by which the nitrogen reduction is estimated will depend on the alternative being considered and the available data. In most cases the land uses that are expected to exist during the study period will need to be considered to estimate the change in nitrogen loads to the waterbody with and without the alternative. For example, if the study period is 50 years, then the nitrogen generation from land uses each year from year 1 to year 50 would need to be considered.

In general, the change in nitrogen loads entering the waterbody each year would be estimated in three steps, as follows.

Step 1:  Estimate the amounts of nitrogen leaving each property (including both point and nonpoint sources) where the alternative would be implemented under 1) the baseline (no action) condition and 2) the alternative condition.

Step 2:  Estimate the amounts of nitrogen entering the waterbody from each property (point or nonpoint source) under the baseline condition and under the alternative condition by taking the estimated nitrogen load at the edge of the source property estimated in Step 1 and multiplying it by 1.0 minus the estimated nitrogen reduction rate (if applicable). The nitrogen reduction rate is the proportion of nitrogen load leaving the property as nitrate nitrogen that is denitrified as the nitrogen travels to the waterbody. The load reduction increases with increasing distance between the source property and the waterbody, all other factors equal. The term “1 minus the reduction rate” is between 0.0 and 1.0 due to biochemical interactions with the soil and water as the nitrogen travels to the waterbody. Controls for point sources that discharge directly to the impaired waterbody would not include a load reduction. Controls for point sources that indirectly discharge to the waterbody because they impact groundwater or discharge to a flowing waterbody that affects the impaired waterbody would potentially include a reduction rate.
**Step 3:** The nitrogen load reduction at the waterbody is calculated as the difference between the total nitrogen load entering the waterbody under the baseline condition and the total nitrogen load entering the waterbody after the alternative is implemented. The result is the amount of nitrogen reduction to the waterbody in pounds that would be expected during the year.

Each of the 50 years would be assigned a nitrogen load reduction value and the relative values from year to year would depend on the extent to which the alternative is implemented over time. A similar approach would be used for point source controls assigning a nitrogen load reduction value from year to year if the alternative is implemented in phases or if increased removal is required as flows increase over time. For example, this may include additional or enhanced nitrogen removal at a wastewater treatment facility as flows to the facilities increase (due to population growth) in order to comply with a loading limit to the waterbody.

### 6.4.2 Nitrogen Reduction – Nitrogen Concentration

The expected total nitrogen concentration leaving the source property (point or nonpoint source) after the alternative is implemented as measured in parts per million (ppm) or mg/L may be a useful evaluation criterion if there are concentration goals associated with managing nitrogen loads to the waterbody. It is possible that an alternative may significantly reduce total nitrogen loads to a waterbody because it affects a significant amount of water but the total nitrogen concentration of the water could be higher than desired. In the event that the nonpoint source control alternative is implemented under different types of land uses that result in different nitrogen concentrations, then the weighted average concentration based on the proportion of land in each use should be calculated.

### 6.4.3 Cost-Effectiveness

This criterion considers the cost-effectiveness of each alternative as it reduces nitrogen loads to the waterbody. The measurement for this criterion is the present value of the alternative’s annual costs in current dollars divided by the present value of the reduction in nitrogen entering the waterbody due to the alternative. The annual data for the change in nitrogen is taken from data used to assess the criterion called “average annual reduction in the amount of nitrogen entering the waterbody during the study period”.

The cost-effectiveness formula is as follows:

$$PV \text{ Cost Per Pound of Nitrogen Removed} = \frac{\sum_{t=1}^{50} \text{Total Annual Cost} \times (1 + d)^t}{\sum_{t=1}^{50} \text{(Annual Pounds of Nitrogen Removed)} \times (1 + d)^t}$$

(6.4.3-1)

Where: “d” = discount rate which represents the opportunity cost of money,

$t = \text{year 1, 2, 3, }... \text{ 50}$

All costs include the costs to construct, implement, operate and maintain the alternative, the costs to government agencies to implement the alternative and any estimated changes in costs or revenues to landowners. The timing of these costs would be the expected year that each type
of cost would be incurred. The costs and changes in revenues should not include inflation – they should be in current year dollars. In this example, the study period is 50 years in order to consider the full life cycle of each alternative.

The discount rate represents the opportunity cost of money to the people or entities that are financing the alternative. If the costs and changes in revenue are estimated in current dollars then the discount rate should be net of annual inflation. For example, the 2010 discount rate net of inflation used by the U.S. Office of Management and Budget (OMB, 2009) is 2.7% per year for a 30-year or longer study period. If the costs and change in revenue are measured in nominal dollars, then the discount rate should include inflation. The 2010 OMB discount rate, including inflation, is 4.5% per year.

This present value cost per pound of nitrogen removed is directly proportional to the cost-benefit ratio for the alternative (not including other costs and benefits of the alternative that are considered by other criteria). To calculate the cost-benefit ratio, the value per pound of nitrogen removed is needed. This value usually is not known, yet we do know that it has a dollar value. Even if the value of the nitrogen removed is not known, the amount of nitrogen removed each year must still be discounted because it is the benefit of the alternative and should be discounted in the same manner as the costs. As with costs, a pound of nitrogen removed today is worth more than a pound of nitrogen removed twenty years from now.

Reardon et al., 2010, recently published a WERF report estimating the incremental cost of increasing nutrient removal at municipal WWTPs. Figure 6-1 illustrates the incremental operational and maintenance (O&M) cost increase per additional pound of nitrogen removed as the treatment level increases from secondary to a TN effluent limit of 1 mg/L, as determined in that study. As shown in the graphic, the incremental cost to achieve an effluent TN of 1 mg/L is more than 25 times the cost of providing secondary treatment.

![Figure 6-1. Incremental O&M Cost for Increasing Wastewater Treatment Levels. Reardon et al., 2010.](image-url)
6.4.4 Present Value Change in Regional or Statewide Income

This criterion addresses how the cost of the alternative will change regional or statewide resident income relative to the “no action” alternative. Under the “no action” alternative, the money that would have been used to pay for the alternative under consideration moves through the economy as savings, investment, and production that generates income to residents. Money spent to pay for the alternative changes this flow of money through the economy and will result in resident income that is higher, lower or unchanged.

The income change is the income generated from constructing and implementing the alternative minus the income generated under the “no action” alternative. The potential ways in which an alternative would change resident income are listed as follows:

♦ Change in land use
♦ Change in demand for goods and services
♦ Change in business profit

Such changes will change the value of the goods and services produced in the region or State which will change resident income. Income is defined as employee compensation, proprietor’s income, interest, rents, profits, sales taxes, and excise taxes, including business and household property taxes.

The changes can be direct, indirect or induced. The direct change in income is the change in income to the businesses and employees that are directly affected by the alternative either because they are building the alternative or selling goods to implement and maintain the alternative or because they have lost their jobs because of the alternative or had to pay for the alternative. The indirect change in income is the change in income earned by businesses and employees who provide goods and services to the businesses building the alternative and/or those that closed due to a land use change (the direct businesses). The induced change in income is the change in income earned by businesses and employees that provide goods and services to the indirect businesses and the direct, indirect and induced employees.

Part of the cost of the alternative might be financed from sources that are outside of the region or State whereby this money would not have been otherwise spent in the region or State. This part of the alternative’s cost will generate direct, indirect and induced income to residents as they are hired to construct and implement the alternative, supply the goods and labor needed to build and maintain the alternative, or supply the goods and labor to the employees as they spend this generated income.

In addition, the alternative may take land out of production. If this production was exported outside of the region or state, such as tomato production, then the lost income would be the lost value of production plus the indirect and induced income that would have been earned. If part or all of the cost of the alternative will be financed from sources within the region or state, then only the direct change in income is the relevant measure of income change.

These impacts would be considered and estimated using an economic input-output model such as IMPLAN or RIMS to obtain the direct, indirect and induced income generated under the nitrogen reduction alternative and under the “no action” alternative.

The equation to calculate the present value change in regional or statewide income is as follows.
Present Value of the Change in Income due to Alternative =

$$\sum_{t=1}^{50} (Income_{Alternative, t} - Income_{No Action, t}) \times (1 + d)^{-t}$$

(6.4.4-1)

Where: “d” = discount rate which represents the opportunity cost of money,

\( t \) = year 1, 2, 3, ... 50.

The discount rate represents the opportunity cost of money to the people or entities that are financing the alternative. If the costs and changes in revenue are estimated in current dollars then the discount rate should be net of annual inflation. For example, the 2010 discount rate net of inflation used by the U.S. Office of Management and Budget (OMB, 2009) is 2.7% per year for a 30-year or longer study period. If the costs and change in revenue are measured in nominal dollars, then the discount rate should include inflation. The 2010 OMB discount rate, including inflation, is 4.5% per year.

6.4.5 Affordability of Wastewater and Stormwater Bills

In the event that the alternative would be financed through a wastewater or stormwater utility, then the impact of the alternative on the cost of living in the area should be considered. This criterion could be measured as the percent change in the utility bill relative to the “no action” alternative or the utility bill under the alternative as a percent of median household income in the area. Under this criterion, the alternatives would be compared based on the relative increase in the utility bill.

6.4.6 Success in Achieving Other Water Management Objectives

The alternative may provide benefits in addition to a reduction in nitrogen loads. These benefits may include the following.

1. The alternative will increase water supply availability. This can occur directly due to increased water supply availability from a particular waterbody, or indirectly due to a diversion of wastewater effluent from a surface water discharge to non-potable reuse. A new or expanded non-potable water reuse system from a wastewater treatment facility can help to offset potable withdrawals thereby increasing water supply availability.

2. The alternative will improve raw water quality.

3. The alternative will significantly reduce soil erosion and sediment transport.

4. The alternative will attenuate flood runoff to receiving waters.

5. The alternative will improve flood control.

6. The alternative will also remove or reduce other contaminants. For example, enhanced levels of municipal wastewater treatment may remove endocrine disrupting compounds, pharmaceutically active compounds and contaminants associated with personal care products, such as anti-bacterial compounds.
7. The alternative will increase public health and safety due to reduced pollutant load to a waterbody. This benefit could also be reworded to address a specific health and safety issue such as achieving a “swimmable” waterbody.

8. Other benefits relevant to the alternative.

The criterion can be as simple as counting the number of benefits from the list and equating this number to the number of points for this criterion. Alternatively, some of these benefits could be separated out as individual criterion and be measured quantitatively, such as the amount of water that would be made available for potable or irrigation use, as discussed in this next criterion.

6.4.7 Water Supply Benefits

This criterion measures the value of the water supply benefits expected under the alternative relative to “no action”. If the alternative will improve water quality such that additional treatment would be avoided, then the cost savings could be measured under this criterion. If the alternative would increase water supply, then the value of this additional water supply would be used. This criterion could be measured as follows.

\[
\text{Present Value of the Change in the Value of Water Supply due to Alternative} = \\
\sum_{t=1}^{50} (\text{Value of Water Supply from Alternative}, t - \text{Value of Water Supply from No Action}, t) \times (1 + d)^{-t}
\]  

(6.4.7-1)

Where: “d” = discount rate which represents the opportunity cost of money, 
\[ t = \text{year 1, 2, 3, ... 50} \]

The difference in the value of water supply could be the cost savings from avoided additional treatment and/or the value of the additional water provided by the alternative. This value of additional water would be calculated using estimated water demand equations that have been estimated for the region.

6.4.8 Acres of Increased or Improved Wildlife Habitat

This criterion measures the contribution of the alternative by directly increasing or by improving wildlife habitat either because of the reduced nitrogen loads or because the alternative will improve the quality of the acreage for wildlife habitat that otherwise would be converted to other uses. This criterion is measured in number of acres and may be adjusted to reflect differences in the quality of the habitat among the alternatives. For example, this criterion could be measured as the total use and non-use value of the habitat protected by the alternative. Different values for the different habitats that would be protected or restored could be used to identify those alternatives that protected higher-valued habitats versus lower-valued habitats.
6.4.9 Change in Recreation Opportunities

This criterion can be measured in terms of the types and amount of recreation opportunities created or it could be a simple 0, 1 point criterion as follows.

0 Points = The alternative will not provide an increase in recreational opportunities in the study area through additions to capacity and/or increases in recreation quality over and above that which would be achieved by nitrogen removal.

1 Point = The alternative will provide an increase in recreational opportunities in the study area through additions to capacity and/or increases in recreation quality over and above that which would be achieved by nitrogen removal.

The point assignment descriptions under this criterion should take care not to double count recreation benefits that may already be considered in other criteria such as the nitrogen reduction criteria and in the other water supply benefits criterion.

6.4.10 Change in Property Values

The criterion measures the estimated change in property values associated with improvements to aesthetics provided by a waterbody due to the nutrient load reductions and the positive or negative impacts associated with other features of the alternative. The criterion would be useful if the alternatives differ in terms of the waterbody that will benefit and the extent to which nitrogen loads are reduced. Other features of the alternative may increase or reduce property values and this impact would be considered in developing this criterion.

6.4.11 Effect on Greenhouse Gas and NOx Emissions

This criterion measures the effect of the alternative on greenhouse gas (GHG) emissions to the atmosphere as compared to the no action alternative. GHG emissions have received significant attention as a major contributor to climate change. Fossil fuel combustion is a major contributor to GHG and NOx emissions. NOx emissions are a source of nitrogen to a watershed through atmospheric deposition and these effects should also be considered.

The GHG emissions that may be affected by a nutrient management alternative include: carbon dioxide (CO₂); nitrous oxide (N₂O); and methane (CH₄). The amount of these three emissions can be measured as mass of CO₂ equivalent (kilograms, metric tons, pounds). Thus, this criterion could be measured as the average annual change in the pounds of CO₂ equivalent, relative to the “no action” alternative, emitted into the atmosphere that results directly from implementing the alternative and indirectly as the alternative consumes energy generated from the production of electricity or fuel.

Other methods for measuring this criterion can be developed based on the GHG emissions generated by each alternative, the sustainability goals of the utility and/or community, and the data and information available to score each alternative with respect to the criterion.

Although the United States did not ratify the Kyoto Protocol, there are several regional mandatory and voluntary climate change initiatives including the Regional Greenhouse Gas Initiative for 10 northeast and mid-Atlantic states, and the Western Climate Initiative which includes 7 U.S. states and 4 Canadian provinces. Currently the U.S. EPA requires GHG reporting for the largest emitters (>25,000 metric tons CO₂e/yr). Although it is unlikely that wastewater treatment facilities will be required to report under the U.S. EPA rule, it is possible that regula-
tions may exist in the future that will encourage facilities to minimize their carbon footprint. GHG emissions are also often used as a way to evaluate the sustainability of a process.

Typical greenhouse gases that would be impacted by nitrogen reduction alternatives are carbon dioxide (CO$_2$), and nitrous oxide (N$_2$O). Alternatives that impact animal production operations could change the amount of methane (CH$_4$) emitted into the atmosphere. For a typical wastewater treatment facility the majority of CO$_2$ emissions is from energy demand (aeration, pumping, mixing, building uses), while the majority of N$_2$O emissions are from the biological treatment basins. N$_2$O emissions are typically higher from facilities with incomplete nitrification and denitrification processes (Ahn et al., 2009). Another indirect impact on greenhouse gas is the effect on N$_2$O emissions in the environment due to nitrogen released in the wastewater treatment facility effluent. N$_2$O emissions are of particular importance because their greenhouse impact is approximately 300 times the impact of CO$_2$.

Although it has been demonstrated that wastewater treatment plants can approach carbon neutral operation (Wett, 2007) for a total nitrogen limit of 14 mg/L, energy neutrality becomes increasingly unlikely as stricter nitrogen limits are imposed (Reardon et al., 2010). The numeric nutrient criteria currently being proposed in several states will likely require very high levels of treatment including processes that will remove recalcitrant dissolved organic nitrogen (rDON). Practical methods for removing these compounds are limited and may potentially require the addition of microfiltration, reverse osmosis (RO) and brine disposal to an enhanced nutrient removal facility. A typical method of brine disposal is deep well injection. This level of treatment is likely to more than double the GHG emissions from a typical enhanced nutrient removal wastewater facility (TN < 3 mg/L, TP < 0.1) and produce more than six times the GHG as conventional secondary treatment (Reardon et al., 2010). While adding microfiltration, RO and brine disposal to a typical nutrient removal facility may increase the amount of nitrogen removed, the incremental increase in GHG emissions (CO$_{2e}$ mass/yr) per lb of nitrogen removed will be 30 times higher than a typical enhanced nutrient removal facility due to the high energy demand. Figure 6-2 shows the incremental GHG emissions increase associated with the increases in treatment level from secondary to TN effluent limits of 1 mg/L.

![Figure 6-2. Incremental Increase in GHG Emissions with Increasing Wastewater Treatment Level.](Image)

Reardon, et al., 2010.
Reardon et al., 2010, concluded that the point of diminishing returns with regards to treatment level and GHG emissions occurs at the enhanced nutrient removal level of treatment (TN < 3 mg/L, TP < 0.1). In addition, extremely low levels of effluent nitrogen will likely require the addition of supplemental carbon for the biological nutrient removal process. The chemical production of methanol, the most commonly used compound for supplemental carbon addition, produces 0.67 lb of CO₂ per pound of methanol (Willis et al., 2009) not including the fossil fuels required to deliver the chemical to the wastewater treatment facility. Due to the large incremental increase in GHG emission with progressively lower levels of effluent nitrogen (and phosphorus) the study concluded that a more moderate level of treatment (TN from 4-8 mg/L, TP from 0.1-0.3 mg/L) combined with best management practices for nonpoint source loads might be the most sustainable approach to reaching water quality goals for a particular waterbody (Reardon et al., 2010). Nonpoint source controls typically have low energy requirements and do not use chemicals. Nonpoint source controls may also lead to reduced carbon emissions by carbon sequestration if natural control systems are used. Natural control systems offer other benefits including enhanced watershed habitat and aesthetics (WERF, 2010).

The evaluation of this criterion for each alternative would be based on the available literature and expert judgment. Typical references for reporting of GHG emissions include the Intergovernmental Panel on Climate Change (IPCC 2006), the U.S. EPA’s Emission and Generation Resource Integrated Database – eGRID (2007), and the Climate Registry’s General Reporting Protocol Version 1.1 (2008).

6.4.12 Engineering/Technological Track Record

This criterion measures the degree to which the alternative can be built and operated successfully in the study area. This criterion includes evaluation of the technologic track record of the alternative that considers whether it has been successfully implemented and operated in the study area or elsewhere. Knowing that an alternative, or its components, has been successfully implemented in a similar application demonstrates its viability and provides information regarding the extent to which the technology can be customized to site-specific conditions. This criterion also considers the extent to which studies have demonstrated the likelihood of the alternative’s success.

There are four aspects to consider when scoring this criterion: 1) whether the alternative, or similar program, has been implemented in other similar situations; 2) whether the alternative, or similar program, was successful; 3) whether studies have shown that the alternative is likely to be successful under this application, and 4) whether or not there are specific unresolved issues that may hinder the success of the alternative. The scoring for this criterion is as follows:

0 Points = The alternative has not been successfully implemented elsewhere in similar applications and the alternative is not likely (< 80% probability) to be successful under this application.

1 Point = The alternative has not been successfully implemented in the study area or elsewhere in similar applications but studies have shown that it is likely (> 80% probability) to be successful under this application.

2 Points = The alternative has been successfully implemented elsewhere in similar applications but further experience is needed to assess one or more specific unresolved issues that may hinder the success of the alternative.
3 Points = The alternative has been successfully implemented in a relatively small number of similar applications and is likely (> 80% probability) to be successful under this application.

4 Points = The alternative has been implemented in many similar applications and has demonstrated success and is likely (> 80% probability) to be successful under this application.

The evaluation of this criterion for each alternative would be based on the available literature and expert judgment. The statistical reliability of nitrogen removal can vary with different control alternatives. The probability of success of a nitrogen control alternative will depend on the permitted monitoring frequency and averaging period. Weekly, daily or single sample limits require a much greater level of reliability from a particular control than annual average or multi-year average limits. For example, a typical advanced biological nutrient removal wastewater facility may be able to reliably meet a Total Nitrogen limit of 3 mg/L on an annual average basis but would require additional treatment process(es) to meet a weekly or single sample Total Nitrogen limit of 3 mg/L.

6.4.13 Permitting Uncertainty

This criterion is measured as the relative amount of effort and time needed to obtain all applicable permit approvals and the likelihood of obtaining all necessary permits from all applicable permitting agencies. This criterion considers potential negative environmental impacts of the project.

Likelihood is defined by probabilities as follows: 1) “not likely” is less than 50% probability of obtaining all necessary permits; 2) “somewhat likely” is between 50% and 70% probability; and 3) “moderately likely” is greater than 70%. Alternatives where all components do not need permits would receive the highest score (4 points) for this category.

This criterion does not attempt to measure the cost of permitting. Such a cost is included under the cost criterion. The scoring is as follows.

0 Points = The project is not likely to receive all necessary permits (less than 50% probability) or the State has denied one or more permits for a project of this type and circumstance in the past and would likely do so again.

1 Point = Insufficient information exists to adequately determine the likelihood that the project would obtain all necessary permits.

2 Points = This type of project is somewhat likely (50% to 70%) to receive all necessary permits or specific identified issue(s) will need to be resolved by the permitting agencies that will likely increase the time and effort above “normal” levels needed to obtain all necessary permits.

3 Points = This project is moderately likely (greater than 70%) to receive all necessary permits given “normal” time and effort and the State has never had the opportunity to permit this type of project in the past.

4 Points = This project does not require any permits or this project is moderately likely (greater than 70%) to receive all necessary permits given “normal” time and effort and the State has approved permits for this type of project in the past.
The evaluation of this criterion for each alternative would be based on the available information and expert judgment.

6.5 Scoring and Weighting the Alternatives and Sensitivity/Uncertainty Analysis

Each alternative is evaluated with respect to the evaluation criteria. Computerized evaluation models such as the Criterium Decision Plus (CDP, Infoharvest, Inc.) model can be used to score, weight and rank the alternatives and perform sensitivity and uncertainty analyses. The evaluation model scales the quantitative and qualitative scores assigned to each alternative and criterion to a number between zero and one. For each alternative, the scaled value of each criterion is then weighted based on the importance of each criterion and then summed among all criteria to obtain the total score for the alternative. The lowest possible total score for an alternative is zero and the highest possible total score is one (0 ≤ total score ≤ 1).

Each criterion is assigned a weight that reflects the relative importance of each criterion as determined by the preferences of the decision makers. If one criterion receives 100 points and another criterion receives 25 points, this means that former criterion is four times more important in the evaluation of the alternatives than the latter criterion. The weights are then converted into a number between zero and one by dividing the number assigned to the criterion by the total of all the numbers of assigned to all of the criteria. For example, given two criteria, one with a weighting number of 100 points and the other with a weighting number of 25 points, the weight of the former criterion would be 0.80 (= 100 / 125) and the weight of the latter criterion would be 0.20 (= 25 / 125). Sensitivity analysis can be performed to determine if alternative weighting would result in a different ranking of the alternatives.

There will likely be some degree of uncertainty associated with the answers (or values) for a criterion as it applies to an alternative. To address this uncertainty, probability distributions can be developed for each criteria value using the best available information. These distributions are used by the evaluation model to recalculate the alternative’s scores to generate a probability distribution of scores for each alternative. These probability distributions of scores can then be compared visually among the alternative. In addition, the percent of the time that each alternative would be ranked number 1 can be calculated. Uncertainty analysis helps to identify those aspects of an alternative that would improve confidence in the ranking which can then be used to develop future research objectives.

6.6 Summary of Benefit-Cost Evaluation Model

The benefit-cost evaluation model includes developing nitrogen reduction alternatives; designing criteria to evaluate the benefits and costs of the alternatives; employing a scoring and weighting system to rank the alternatives; and assessing data gaps and uncertainties.

The criteria presented in Section 6.4 were developed to reflect the potential benefits and costs of a wide range of nutrient reduction alternatives. The actual criteria should be tailored to the alternatives proposed for evaluation and the available data and information. The criteria may be simplified but a purely qualitative review process is not recommended. Qualitative approaches can introduce biases and reduce process transparency. To address the criteria, certain assumptions may need to be made and the answers may be uncertain. The assumptions and uncertainties
identified in the process will help to recognize data gaps and areas where more research is needed.

The evaluation method can be tailored and used as a planning tool and as a decision-making tool. In addition, this process can be reassessed over time to evaluate how successful the selected alternatives were in meeting their targets and if assumptions made in the original analysis were valid. In addition, new information may be available to address data gaps in the original analysis. An evaluation can be preformed to address whether continued use and maintenance of the selected alternatives are warranted or if new technologies or new approaches should be considered or implemented.

6.7 Water Quality Trading

Water quality trading is a market-based approach to achieving nutrient removal goals at the lowest possible cost by allowing the voluntary exchange of nutrient credits for money. When certain conditions exist, those who can reduce nutrient loads at the lowest cost will sell nutrient credits to others and a market price (or value) of the credit will evolve. Those who can reduce nutrient loads at a cost lower than the market price will sell credits and those with nutrient removal costs that are higher than the market price will purchase the credits.

The Florida Department of Environmental Protection has established a Water Credit Trading and Pollutant Trading Policy Advisory Committee (http://www.dep.state.fl.us/water/watersheds/ptpac.htm). Currently there is a water quality credit trading program pilot project in the lower St. Johns River Basin in Florida that facilitates the trading of nitrogen and phosphorus credits (Lower St. Johns River TMDL Executive Committee 2008). Statutory authority for a water quality credit trading program is provided only for the Lower St. Johns River Basin pilot program (Chapter 403.067, Florida Statutes). This program was established in October 2008 and a report on this pilot project is due to the Governor and Legislature by October 2010. Other than this pilot program, water quality trading approaches have been discussed in Florida but have not been implemented.

Descriptions of the nitrogen reduction program of the Tampa Bay National Estuary Program and the developing trading initiative by the Florida Ranchlands Environmental Services Project have been provided in the literature (Selman, et al., 2009 and Breetz, et al., 2004). However, these programs are not water quality trading programs as defined in this sub-section.

According to the U.S. EPA, (2008d, page 1-2), “Despite the theoretical promise of water quality trading and U.S. EPA’s efforts, however, water quality trading to date has met with limited practical success. Only 100 facilities have participated in trading, and 80% of trades have occurred within a single trading program (Long Island Sound). Moreover, relatively few trading programs have been scaled up from pilot projects to permanent programs, and even fewer can claim to have had a significant impact in improving water quality or reducing pollutant control costs.”

Under the right circumstances, a well-designed and properly implemented water quality trading system can be an effective tool to achieve water quality goals. The Water Environment Research Foundation (WERF) has supported the research and development of water quality trading programs over the past 14 years. This Foundation has developed guidelines that take into account the successes and lessons learned from implementing five water quality trading studies. A recent compendium of documents regarding quality trading and market based approaches pre-
pared by WERF, the U.S. EPA and other authors is provided on page 5-16 and 5-17 of the 2010 WERF document titled, “Nutrient Management: Regulatory Approaches to Protect Water Quality, Volume 1 – Review of Existing Practices”. This bibliography is reprinted at the end of this sub-section.

Water quality trading approaches are based on the concept of creating a market for the nutrient reduction service. Thus, the economics profession is best trained to assist in evaluating the potential for such markets and in providing design and implementation advice. Economists will first evaluate whether favorable conditions exist that will support market-based trading. These conditions, applied to nitrogen reductions in waterbodies, include the following, and are worded as questions where a “yes” answer identifies a positive characteristic that would facilitate a successful water quality trading program.

1. **Will there be enough buyers and sellers such that a trading program will be more cost-effective than a traditional regulatory program in reducing nitrogen loads?** The answer is “yes” if there is an adequate mix of dischargers that have different cost structures such that nitrogen reduction costs are lower for some and significantly higher for others. The answer is “no” if it is clear that the number of dischargers is so low and so similar that traditional regulatory mechanisms will cost less or about the same as a trading program. In this case there is no need to further evaluate the usefulness of water quality trading and this approach should not be further pursued. If the answer to this question is “yes”, then evaluate the next conditions as follows.

2. **Two part question:** (a) Is there a clear, legally mandated water quality goal, such as a nitrogen total maximum daily load or a nitrogen discharge cap, in water leaving the discharging property from which credits can be deducted or added? (b) Will this goal be an effective standard by which to judge compliance with the nitrogen water quality goals of the receiving waterbodies?

3. **Two part question:** (a) Can the nitrogen reduction credit system be designed so that the contribution of one unit of credit to the water quality goal is the same regardless of the location of the entity or type of entity that is discharging the nitrogen-laden water? (b) Will the resulting credit system not inhibit trading to the extent that the answer to question (1) is “no”? For example, can effective credit ratios be developed and applied to individual trades so that the nitrogen reducing service being bought and sold is standardized while still providing enough of a cost-difference among dischargers so that will there still be enough buyers and sellers to establish and sustain a market?

4. **Will it be technically, economically and environmentally feasible to install and maintain the water quality monitoring devices and data collection systems needed to monitor the nitrogen loads of water leaving the discharging properties?**

5. **Does the agency that regulates the trades and enforces compliance with the nitrogen reduction goals have sufficient legislative authority to legally ensure that individual trades are completed and that individuals are in compliance with their nitrogen allocation?** Is there an agency that has the legal authority to handle trading disputes?

6. **Are there no potential trades that could be made that would produce negative impacts, such as nitrogen “hot spots”, or would be in violation of Federal or State law?** If such trades could occur, can mechanisms be put in place to prevent this from hap-
pening while still resulting in sufficient trades such that the answer to question (1) above is still “yes”?

7. Are there no other policies and/or regulations in place that would lower the incentive for dischargers to trade with one another? If the answer is “no”, can these policies and/or regulations be changed to remove any barriers to trade?

If the answers to questions (1) through (7) are “yes”, then a nitrogen trading program has the potential to successfully reduce nitrogen loads at the lowest possible cost. However, all costs and benefits of this program must be estimated and compared to other alternatives. A nitrogen trading program would then be designed as a nitrogen reduction alternative and evaluated using the benefit-cost evaluation model.

Design of a nitrogen trading system should consider the issues addressed in 1) through 7) above and other considerations as follows.

(a) Develop the permit trading system so that it minimizes the cost of completing the trade, including the time and cost needed to locate potential buyers and sellers, to learn the current and historic market prices of the credit, to install the monitoring and reporting system, and to negotiate the transaction.

(b) Establish and disseminate clear procedures for enforcement when the seller of the credit does not comply with its nutrient reduction requirements. Consider ways to reduce uncertainty to those who are purchasing the credits.

(c) Provide constant dissemination of the information regarding the requirements, enforcement and status of the trading program to all interested parties in a manner that is easy to understand and follow.

(d) Establish and implement effective methods to monitor the success of the trading program in achieving the nitrogen reduction goals of the receiving waterbodies and the satisfaction levels of the participants and the communities. To this end, implement data collection, reporting and evaluation methods that are reviewed on an annual basis to determine the success of the trading program. Issues to address include the extent to which the water quality goals are achieved, ancillary benefits, problems being experienced by the regulating agency, traders, dischargers and the communities, and methods to address the shortfalls and problems.

(e) Design the trading program so that it can be legally re-structured or changed without negatively affecting the benefits of past trades to those who purchased and sold the credits.

(f) Develop stable funding sources to finance the monitoring and enforcement of the trading system.

(g) Learn from the experiences of other agencies that have implemented water quality trading programs.

These listed issues and considerations are necessary to design and implement a water quality trading program but are not, by themselves, sufficient to develop a successful program. Consultations with economists, those who have implemented past programs, and the experts at the U.S. EPA and WERF are also needed.
A list of studies that address the experiences of past and current water quality trading systems and that discuss design and implementation issues are provided as follows.


http://www.epa.gov/ow/waternews/2007/071113.html#1

http://www.epa.gov/owow/watershed/trading.htm


Reckhow, K.H., et al., 2006. Water Quality Indicators: Nutrient Impacts on Chlorophyll or Algae Species Composition. WERF / IWAP. (02ECO1).


2. Other relevant documents are as follows.


6.8 Develop the Nutrient Control Strategy

As noted in previous chapters, no two waterbodies are identical and therefore there are no “one size fits all” approaches to nutrient management. A site-specific evaluation of the waterbody is required in order to properly understand the linkages between nutrients and response variables including chlorophyll \(a\), dissolved oxygen and transparency. The unique watershed characteristics and combination of point and nonpoint sources must be understood in order to properly select effective control alternatives to reduce nutrient loading to a waterbody.

Once potential nutrient removal actions or combinations of actions are identified, a benefit cost analysis of these alternatives, as described above, will help decision makers develop the best nutrient control strategy to meet the nitrogen reduction and sustainability goals for a particular waterbody. This evaluation method attempts to identify and quantify the social and environmental benefits and costs resulting from a variety of nutrient management alternatives. The true measure of the benefits and costs associated with an investment is the change in total economic value caused by the alternative.
The change in total economic value is the sum of the changes in producer surplus and consumer surplus associated with the alternative relative to “no action”. Benefits and costs should be assigned monetary values to the extent that the available time and budget will allow. Where insufficient information exists to assign a monetary value, benefits and costs should be described and quantified to the extent practical. In addition, stakeholder involvement in the decision-making process and in the implementation of the nutrient control strategy is critical to success. The implementation of nutrient control strategies is discussed briefly in Chapter 8.0.

6.9 Data Gaps and Research Needs

Each benefit-cost analysis will require significant site-specific data collection and quantification. Missing, incomplete or uncertain information will almost always exist for one or more of the alternatives and criteria. The criteria presented above were developed to assess benefits and costs when the values of the ecosystem services provided by the alternative have not been estimated. Thus, most of the uncertainty in evaluating these specific criteria will be associated with expected nitrogen reductions from a particular alternative, all costs associated with the alternative, and the impact on GHG emissions.

Time and money are needed to estimate the benefits and costs of each alternative, including any desired estimates of changes in income, use, non-use and property values associated with changes in nitrogen or nutrient levels. This is why it is important to assess nutrient reduction technologies and alternatives well before they need to be implemented. Economic studies can be tailored to the needs of the evaluation and can provide the needed benefit and cost information given sufficient time, say one to two years, and a budget that is likely to be very nominal compared to an alternative’s cost to implement. The actual budget and time needed will depend on the size of the geographic area and number of benefit and cost values to be estimated.

To date, there are no publicly available studies in Florida that have determined the change in income, use, non-use, and property values associated with changes in ecosystem quality, including changes in nitrogen concentrations of waterbodies. The existing studies have estimated the income, use, non-use and property values generated by waterbodies in their existing condition and are typically one-time studies in that they are not updated over time. The exception is the Indian River Lagoon where use and non-use values were estimated in 1995 and 2007. Thus a major research gap is an understanding of how the values generated by Florida’s waterbodies change as water quality impairments increase. This research could be conducted immediately using publicly available information and survey research.

Water demand equations needed to estimate the value of additional water supply have been estimated for single-family homes throughout Florida (Whitcomb, 2005). However, estimated water demand equations are not common in Florida for all other uses including multifamily residential, commercial and industrial. Additional research to estimate these water demand equations for all users and locations in Florida would benefit the evaluation of nutrient reduction alternatives and contribute significantly to the evaluation of water supply and conservation management in Florida.

Advanced technologies that have not traditionally been used to achieve extremely low levels of nitrogen are being proposed as potential treatment methods to reduce total nitrogen in wastewater treatment plant effluent. These technologies include but are not limited to reverse osmosis, ion exchange, and oxidative processes. Recently, a water recycling facility in Sydney,
Australia was brought online (October 2010) utilizing reverse osmosis to remove nutrients and other contaminants before discharge to the Hawkesbury-Nepean River (Sydney Water, 2010). Additional research and monitoring of completed facilities is needed on the nitrogen removal potential of these current and emerging treatment technologies in order to properly evaluate which processes will perform reliably to meet the permitted limits for a waterbody. Understanding the ancillary benefits of these advanced technologies, including the removal of endocrine disrupting compounds and pharmaceutically active compounds, is a topic currently undergoing significant research.

As discussed in Chapter 5.0 there is a lack of “end of pipe” monitoring for storm water discharges which often makes it difficult to understand their contribution to a waterbody’s impairment. Increased storm water collection and testing for nutrients and toxins are needed at key watershed locations throughout the State where discharges are flowing into waterbodies. In addition, in some areas, concentrated animal feeding operations (CAFOs) contribute significant nutrient loading to the watershed. Research on CAFOs is ongoing and includes the economic and environmental benefits of nutrient recovery and recycling. Additional research is also being performed on natural treatment technologies and improvements to existing best management practices.

Additional research is also needed to assess the effectiveness of stormwater runoff controls and management methods including fertilizer ordinances and public education. In addition, the effect of point and nonpoint source atmospheric deposition to many waterbodies is poorly understood. In certain watersheds, understanding the impact of atmospheric deposition is important to achieving water quality goals as described in Chapter 5.0 for the Tampa Bay watershed.
CHAPTER 7.0

WATER QUALITY IMPROVEMENTS: CAN RECEIVING WATER RESPONSES BE ESTIMATED?

Once a waterbody has been identified as impaired and the causative pollutant inputs (nitrogen, for example) and their sources have been identified, the loads of these pollutants quantified, and the potential control strategies have been defined, an evaluation of the expected waterbody responses is the next critical step. This evaluation depends upon the ability to predict likely receiving water responses to nutrient load reductions based on estimates of control measure treatment efficiencies. Conducting this evaluation will help determine if a particular combination of control measures could be expected to provide acceptable water quality benefits if implemented. The potential methods to predict the waterbody responses to the proposed control strategy are discussed in this chapter.

A variety of modeling tools are available to assess the waterbody responses to nutrient controls. Generally, these include empirical and mechanistic modeling techniques. Each method is presented below. The linkages between loading and response should be sufficiently understood and quantified to enable selection of the appropriate predictive tool and its use. Clearly, these considerations depend upon the specific predictions required (e.g., steady-state vs. dynamic), the spatial and temporal resolution desired, and the degree of certainty in the model’s ability to simulate all required processes and make reasonable predictions. Other common questions that are considered in model selection are:

♦ How will the control strategy affect the magnitude, timing, and distribution of nutrient loadings to the receiving water?

♦ Can the model predict phased responses and ultimately a new equilibrium condition?

The appropriate model selection and application depends on numerous factors including nutrient loading magnitude, timing, and distribution; availability of data for model calibration; available time and budget for model application; and the types of control that are proposed (Chapra, 1997; Martin and McCutcheon, 1999; NRC, 2000).

7.1 Empirical Models

There are several empirical modeling methods that can be used to assess waterbody response to changes in nutrient loads. They range in complexity from simple regression models to artificial neural networks (ANNs).

7.1.1 Regression Models

Statistical regression is a procedure for predicting the value of a changing (dependent) variable based on the value of a different (independent) variable when the relationship between the variables can be quantified, based on observed information. Regression models can be very
useful in ecological analysis because the outcomes do not depend on specific knowledge (i.e., data) of the dynamics of the relationships between the variables. The lack of such knowledge requires significant assumptions if a more complex mechanistic model is applied. For example, in using a mechanistic model to predict the behavior of a receiving waterbody in response to changing nitrogen inputs, one must make assumptions about site-specific biological, chemical, and physical transport and transformation processes that affect nitrogen in the watershed and in the waterbody. Each assumption introduces additional uncertainty into the predictions.

Care should also be taken when interpreting empirical model results as valid conclusions regarding the inferred cause-effect relationships. The validity of the conclusions drawn will depend upon the specific knowledge of the nature of the relationship between dependent and independent variables.

Linear regression is a common statistical technique used to explore the relationship between two or more variables. The relationship between the dependent variable (y-axis) and independent variable (x-axis) is developed. This is done by fitting a straight line through the set of points that minimizes the sum of squared residuals (i.e. the difference between observed and model predicted values). Assumptions of linear regression include independence, homoscedasticity (equality of variance) and normal errors.

An example of a linear regression applied to the relationship between total nitrogen (TN) and chlorophyll $a$ for Upper Lemon Bay in southwest Florida is provided in Figure 7-1. There is a statistically significant relationship between these two variables.

![Figure 7-1. Linear Regression of Monthly Average TN and Chlorophyll $a$ Concentrations in Upper Lemon Bay. Janicki Environmental, 2010d.](image)

An example of one measure of the goodness-of-fit of a regression model that predicts chlorophyll $a$ concentrations given the TN loadings to Tampa Bay is shown in Figure 7-2. Other critical diagnostics that should be investigated in the assessment of the validity of the regression
model developed include residual analysis and tests of the assumptions defined above (i.e., independence, homoscedasticity (equality of variance), and normal errors).

Figure 7-2. Goodness of Fit for the Regression of Chlorophyll $a$ Concentrations on TN Loads. Janicki and Wade, 1996.

Regression models that predict the responses in waterbodies to changes in nutrient loads have been developed for a variety of waterbody types. Phosphorus-chlorophyll relationships in lakes were developed by Vollenweider (1968, 1972) and have been refined and expanded by others, as discussed in Chapter 2.0. Similar approaches to predict chlorophyll responses to varying nutrient loads have also been applied in estuaries (Boynton et al., 1982; Nixon, 1992), streams (Lin et al., 2007), and wetlands (Mitsch et al., 2005).

As a result of not using rate constants and other mechanistic tools, regression models do not indicate what drives the relationship between dependent and independent variables, but only describes the magnitude of the change. Therefore, a regression model cannot tell us why a receiving waterbody responds to a change in nutrient load.

Also, many ecological interactions are progressive and dynamic. A change in conditions may trigger a response immediately, or the response may be delayed. Most regression modeling assumes a steady state condition and may not be appropriate for tracking trends in conditions. Despite these limitations, regression modeling is an extremely useful and cost-effective means of predicting the benefits of control measures.

EUTROMOD is a spreadsheet-based model that is used for predicting nutrient runoff and eutrophication for lakes in the U.S. (Reckow, 1990). Lake eutrophication response is predicted based on a set of regional statistical models. The model input data include meteorology, lake morphology, and land use to predict waterbody nitrogen, phosphorus, and chlorophyll $a$ concentrations. Phosphorus and nitrogen loads via stormwater runoff may be predicted using ei-
ther nutrient loading functions or nutrient export coefficients. The nutrient loading functions are based on the rational formula for dissolved nutrients, and the universal soil loss equation for sediment-attached nutrients. The sediment delivery ratio is addressed with user-defined trapping zones. Response variables include total phosphorus (TP) concentration, TN concentration, chlorophyll \( a \) level, Secchi disk depth, and in some cases, probability of hypolimnetic anoxia, and probability of blue-green algae dominance.

### 7.1.2 Artificial Neural Networks

Another type of empirical modeling approach that has become popular for assessing water quality is called an artificial neural network (ANN), which is a mathematical model or computational model that is inspired by the structure and/or functional aspects of biological neural networks. This mathematical model consists of an interconnected group of decision points that process information using a connectionist approach to computation. That is, an ANN is an adaptive system that changes its structure based on external or internal information (input) that flows through the network during the learning phase (hidden) and determines a logical outcome (output). Artificial neural networks are non-linear statistical data modeling tools that are typically used to model complex relationships between inputs and outputs or to find patterns in data. This ability of a neural network provides a useful tool for modeling water quality changes in response to environmental change. Figure 7-3 presents a neural network used to predict chlorophyll \( a \) concentrations from a series of nutrient supply and physical factors.

![Figure 7-3. Artificial Neural Network Construct Used to Predict DO.](image)

*Figure 7-3. Artificial Neural Network Construct Used to Predict DO.*

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The ANN approach has been used to examine eutrophication (Karul et al., 2000), predict stream quality (DO, TSS) based on physical processes (Lek. et al., 1999; Bowers, and Shedrow,
2000), as well as predicting lake chlorophyll concentrations (Panda et al., 2004), reservoir phosphorus concentrations (Kuo et al., 2006), phytoplankton succession (Olden, 2000), and primary production in estuaries (Scardi, 1996; Scardi and Harding, 1999). Rochelle-Newall et al. (2007) used an ANN approach to develop a mathematical model of the gross primary production to community respiration ratio (GPP:CR) based on input variables derived from widely contrasting coastal ecosystems.

### 7.2 Mechanistic Models

Mechanistic modeling is based on the underlying physics and biogeochemistry governing the behavior of individual processes in receiving waterbodies (Chapra, 1997; Martin and McCutcheon, 1998; Edinger, 2002). Knowledge of the interactions among process variables is used to define the model structure. Mechanistic modeling requires determination of the critical model state variables and parameters, that is, what features of the waterbody are important in the process of assessing the responses to changes in nutrient loads. The numerous processes that can be addressed in a mechanistic eutrophication model such as WASP are shown in Figure 7-4.

![TMDL Modeling Toolbox -- WASP](Figure 7-4. Schematic Illustration of Eutrophication Processes Modeled in WASP. U.S. EPA, 2010e.)

Typical inputs include pollutant and hydrologic loads, boundary conditions, and initial conditions within the receiving waterbody. Also, dissolved and particulate fractions of each nutrient are typically specified (Wool et al., 2003). There is also a large number of rate constants that must be specified. Given that many constants vary from waterbody to waterbody and that these constants are typically lacking for a given waterbody, literature values are often used. This approach can lead to increased uncertainty in the model predictions (NRC, 2000).
A mechanistic waterbody response model should predict water quality responses while accounting for a variety of important factors including waterbody geometry, advective and dispersive flows, settling and resuspension rates, boundary concentrations, pollutant loadings, and initial conditions. For many mechanistic models, the waterbody is divided into several finite elements or segments. Individual segment volumes, connectivity, and type (surface or subsurface flow) can usually be specified. The model must also incorporate a variety of transport mechanisms besides advection and dispersion in the water column (Figure 7-5). These mechanisms include advection and dispersion in pore water, settling, resuspension, and sedimentation, and evaporation or precipitation (Wool et al., 2003).

![Figure 7-5. Sediment Transport Mechanisms. U.S. EPA, undated.](image)

Typically, the predicted pollutant load generated by a watershed model (examples of these models are discussed in Chapter 4.0) is estimated for existing conditions. Subsequent model runs estimate the responses due to some load reduction or more specifically the load reduction expected from a particular control strategy. The existing and proposed control strategy output from the watershed model, often in a time series format, is used as input to the waterbody response model that simulates processes occurring within the waterbody.

Accurate model predictions depend upon effective model calibration and verification. Model calibration consists of modifying input parameters until the output from the model achieves a predefined level of agreement with observed data. After a model is sufficiently calibrated, the model is validated by making predictions for a time period and conditions other than those used for calibration, with the robustness of the model determined by comparison of validation output and observed data.

Calibration and validation require the collection and use of substantial measured data describing the system. Although mechanistic models are widely used and are very useful in predicting a wide variety of ecological processes, they tend to be iterative, and are more complex than other approaches.
As with empirical models, several types of mechanistic models of varying complexity have been developed to assess waterbody responses to changing nutrient conditions. Simply, these include box models and dynamic simulation models.

BATHTUB is a simple box model developed by Walker (1985) that consists of three interrelated programs (FLUX, PROFILE and BATHTUB) used to make simplifying assumptions regarding eutrophication-related processes and effects (Figure 7-6). FLUX allows estimation of tributary mass discharges and loading data and daily flow records. PROFILE facilitates analysis and reduction of in-lake water quality data. BATHTUB applies a series of empirical eutrophication models to lakes and reservoirs. The program performs steady-state water and nutrient balance calculations in a spatially segmented hydraulic network which accounts for advective and diffusive transport, and nutrient sedimentation. Eutrophication-related water quality conditions (total phosphorus, total nitrogen, chlorophyll $a$, transparency, and hypolimnetic oxygen depletion) are predicted using empirical relationships derived from assessments of reservoir data. Applications of BATHTUB are limited to steady-state evaluations of relations between nutrient loading, transparency and hydrology, and eutrophication responses.

![Figure 7-6. BATHTUB Model Construct. FDEP, 2006.](image)

Other more complex mechanistic waterbody response models include WASP (Wool et al., 2003), QUAL2E (USGS, 2010b), MIKE 11 (DHI, 2010), EFDC (U.S. EPA, 2010g,h), CEQUAL-ICM (Cerco, 1995), and EPD-RIV1 (U.S. EPA, 2010c). These models are commonly used to simulate and predict hydrodynamics and water quality fate and transport in the surface receiving waters in one, two, or three dimensions. In addition to nutrients, mechanistic models can be used to simulate biochemical oxygen demand (BOD), dissolved oxygen (DO) dynamics, among other response variables. There are many model reviews that examine the nature and applicability of mechanistic models that predict changes in waterbodies as a result of changes in nutrient loads (e.g., Donigian and Huber, 1991; DeVries and Hromadka, 1993; Novotny and Olem, 1994; Donigian et al., 1995; Singh, 1995; U.S. EPA, 1997b; Deliman et al., 1999; NRC, 2000; U.S. EPA, 2010f; USGS, 2010c,d).

Cerco (2000) developed and used an eutrophication model package to examine the response of the Chesapeake Bay to nitrogen and phosphorus load reductions. The bay was examined to predict conditions under two extreme conditions. Best available technology nutrient controls were simulated in one scenario and for an all-forested watershed for the other scenario.
Ten-year simulations were run to examine the mature response of the bay to nutrient load reductions. Analyses indicated that the primary effect of phosphorus controls was the limitation of algal biomass during the spring bloom and in the upper bay during the summer. Nitrogen controls limited algal growth in the lower bay. Nitrogen controls were more effective than phosphorus controls in limiting algal production and in reducing the extent of anoxic conditions.

QUAL2E (USGS, 2008, 2010b) is a steady-state, one-dimensional model used to simulate flow and water quality in streams and rivers that can be assumed to be well-mixed. Branched networks can be modeled. Importantly, QUAL2E is not suitable for unsteady flows, or for domains receiving variable inputs of water quality constituents. However, the effects of daily variations in meteorological forcing functions on water temperature and algal photosynthesis can be simulated. The model is intended as a water quality planning tool and can predict concentrations of nutrients, BOD, DO, chlorophyll, and other constituents.

WASP (Ambrose et al., 1993) is a mechanistic, deterministic, dynamic model for aquatic systems, including both the water column and the benthic sediments. The time-varying processes of advection, dispersion, point and diffuse loading, and boundary exchange are represented in the model. WASP consists of two stand-alone computer programs, DYNHYD and WASP, which can be run in conjunction or separately (Figure 7-7). DYNHYD is a hydrodynamics program, while WASP, the water quality program, simulates DO, carbonaceous BOD (CBOD), phytoplankton, carbon, chlorophyll $a$, ammonia, nitrate, organic nitrogen, and orthophosphate in benthic sediments and overlying waters. WASP can also be linked to several other hydrodynamic models, such as EFDC.

CE-QUAL-ICM (Cerco, 1995) is a finite volume eutrophication model that can be used to simulate water quality responses in either one-, two-, or three-dimensions. The model can be applied to most surface waterbodies, including in mixed dimensions (e.g., a river discharging to an estuary). CE-QUAL-ICM incorporates 22 state variables that include multiple forms of algae, carbon, nitrogen, phosphorus, and silica; and DO. The model incorporates benthic processes including sediment oxygen demand and sediment-water nutrient flux. The model does not compute hydrodynamics. Flows, diffusion coefficients, and volume must be specified externally and read into the model. Hydrodynamics are usually obtained from a model such as CH3D-WES (http://smig.usgs.gov/cgi-bin/SMIC/model_home_pages/model_home?selection=ch3dwes) or EFDC.
MIKE 11 is a 3D modeling system for free surface flows, including advection-dispersion, water quality, heat exchange with the atmosphere, heavy metals, eutrophication, flooding and drying of intertidal areas, and sediment processes (DHI, 2010). Water quality modules simulate the fate and transport of conservative or linearly decaying constituents, eutrophication processes including nutrient cycling, phytoplankton, zooplankton, and benthic vegetation growth, processes affecting dissolved oxygen, exchange of metals between the bed sediments and the water column, and sediment transport/deposition/erosion.

EFDC (Environmental Fluid Dynamics Code) is surface water model developed to simulate hydrodynamic and water quality in two- or three-dimensions in rivers, lakes, reservoirs, wetland systems, estuaries, and the coastal ocean (Tetra Tech, 2002). EFDC (U.S. EPA, 2010h) has become one of the more commonly applied hydrodynamic and water quality model and includes eutrophication processes and sediment biogeochemical processes. The eutrophication kinetics and sediment processes are based on those in CE-QUAL-ICM (http://smig.usgs.gov/cgi-bin/SMIC/model_home_pages/model_home?selection=cequalicm)

In summary, there are numerous approaches to modeling waterbody responses to changes in nutrient loads, and the approaches range from simple to complex. The selection of an appropriate model depends on available resources, data availability, scale of analysis, and the desired level of certainty. Effective use of a model assists in evaluating control measure effectiveness, which is a critical element of managing surface water resources.

7.3 Water Quality Response Modeling in Florida

There are several instances of predictive models being successfully applied to Florida waterbodies to assess the potential effects of control measures to water quality problems. Some examples of these are summarized below.
7.3.1 Caloosahatchee River

FDEP published a final TMDL for nutrients for the tidal Caloosahatchee River in 2009. Using the IWR protocol, the Tidal Caloosahatchee River had been verified as being impaired (FDEP, 2009). Specifically, analysis of water quality data revealed that the waterbody was impaired for both DO and nutrients (based on chlorophyll *a* concentrations that exceeded the adopted threshold for impairment). Nutrient data indicated that the system was nitrogen limited based on the TN:TP ratio.

The technical approach to determining the potential load reductions needed from control measures to meet water quality targets required the use of a mechanistic model. The waterbody (Figure 7-8) consists of estuarine and freshwater segments. Although the Caloosahatchee River extends from the Gulf of Mexico to Lake Okeechobee, the analysis included only the river reach from its mouth upstream to a control structure, the Franklin Lock, about one half-way between the river mouth and the lake.

An attempt was made to determine regression relationships between DO and potential causative pollutants as well as chlorophyll *a* and nutrients. Since the system was large and hydrologically complex, only weak relationships between nutrients and chlorophyll were discerned. Because of these complications, computer modeling was used to simulate the Tidal Caloosahatchee River.

![Figure 7-8. EFDC Model Segmentation of the Tidal Caloosahatchee River. FDEP, 2009.](image-url)
A water quality target was selected based on protecting seagrasses by ensuring adequate light penetration in the water column. A demonstrated relationship between TN and chlorophyll concentrations in the river allowed for the connection between TN concentration and percent photosynthetically available radiation (PAR). Thus, the objective became to determine the nitrogen loading rate to the river that would correspond to an acceptable level of light penetration for seagrass growth.

Nitrogen sources in the watershed were identified and loadings were simulated using HSPF (U.S. EPA, 2010d). Loading estimates from three load reduction scenarios were obtained, and EFDC was used to simulate the river’s response to changes in loading rates. The three scenarios were: current conditions, background conditions (pre-development land uses with Lake Okeechobee at its TMDL nutrient target levels), and proposed nutrient reductions that would be adequate to meet the light attenuation target. The required load reductions were determined by first running the response model using existing conditions loading rates as input. The loads were then reduced incrementally until the loading rate corresponding to chlorophyll \(a\) levels that permit the target light attenuation (consistent with healthy seagrass meadows) was identified. This meant ensuring that the maximum amount of area at a depth suitable for seagrass growth received a minimum 25% PAR. Results indicated that a 20% reduction in TN loading from the watershed would meet this target. A margin of safety was added based on pre-development TN loading rates and the final load reduction for the TMDL was set at 23%.

### 7.3.2 Tampa Bay

The Tampa Bay Estuary Program (TBEP) has established one of its major goals was the restoration of seagrass acreage in the bay to that found in the 1950s, a total of 38,000 acres bay-wide. To address water quality in Tampa Bay, the TBEP Nutrient Management Consortium utilized a paradigm relating seagrasses, light, chlorophyll, and nitrogen loadings, as shown in Figure 4-8. Empirical relationships were developed to quantify the linkages between nitrogen loading, water quality responses, and seagrasses recovery (Janicki and Wade, 1996). These linkages were:

- ♦ Between total external loads and chlorophyll \(a\) concentrations
- ♦ Chlorophyll \(a\) concentrations and subsurface light conditions
- ♦ Between subsurface light conditions and seagrasses recovery

These linkages were developed for each segment of Tampa Bay. A variety of methods were used to estimate external loadings to the bay. Measured streamflow and water quality data and a regression model were used to estimate nitrogen loads in stormwater runoff. Other loads were estimated using available measured atmospheric deposition data, point source permitting data as described in Chapter 4.0, and a groundwater flow net analysis.

Based on the results of these empirical modeling efforts, it was determined that seagrass restoration goals could be met by constraining external nitrogen loads and bay segment chlorophyll \(a\) concentrations to remain at the estimated average levels of 1992-1994 (Janicki and Wade, 1996; Greening and Janicki, 2006). As described in the 2007 Reasonable Assurance Update approved by the Nitrogen Management Consortium in 2008 (TBEP, 2009; 2010), the chlorophyll \(a\) thresholds derived from this effort were approved by FDEP as indicators of impairment in each bay segment. The nitrogen load-chlorophyll \(a\) regression model was the key to determining the magnitude of future load reductions that will be required to maintain chlorophyll \(a\) levels.
at or below the target concentrations. Figure 7-2 above shows the goodness of fit of that regression model.

A mechanistic waterbody response model (WASP) was also developed for Tampa Bay to provide a process-oriented examination of relationships between nutrient loadings, chlorophyll concentration, and light attenuation (Martin et al., 1996). Both the empirical and mechanistic models produced similar results, providing a weight of evidence that acceptable nutrient management targets could be developed (Brown, 1999). The critical relationships that were established were external nitrogen (limiting nutrient) loads and resulting chlorophyll concentrations; chlorophyll concentrations and density of phytoplankton in the water column; and chlorophyll concentrations and light levels at the deep edges of historic seagrass beds.

7.3.3 Lower St. Johns River

The Lower St. Johns River (LSJR) is located in northeast Florida, and extends from its mouth near Jacksonville south to near Daytona Beach. The Nutrient TMDLs for the main stem of the LSJR basin were adopted by the FDEP in 2008. The LSJR segments had been verified as impaired by nutrients based on annual mean chlorophyll \( a \) concentrations and annual mean TSI values. In addition to the elevated chlorophyll \( a \) and low DO levels, a number of widespread water quality problems were identified that indicate eutrophic conditions, including fish kills, submerged aquatic shoreline vegetation covered in algal mats, and excessive epiphyte growth, among others, as described in Section 4.4.

The TMDL requires that TP and TN in the freshwater reach of the river be reduced to achieve reductions in chlorophyll \( a \) levels. In addition, a TN TMDL was also established in the marine reach of the river to meet DO levels that would be protective of aquatic life uses. A mechanistic modeling approach was used to identify the load reductions required to restore the river, as described in Section 4.4. Based on the long-term average results of the modeling, a 30% reduction in anthropogenic point, nonpoint, and upstream boundary nitrogen and phosphorus loads was deemed adequate to meet the target chlorophyll \( a \) target (FDEP, 2008).

A similar analysis was completed for the combined downstream reach of the river. In this portion of the river, nitrogen was the key nutrient that needed to be reduced to meet DO targets. The modeling indicated that a 28.5% reduction in anthropogenic point and nonpoint nitrogen loads was needed from within this reach to attain the DO target, contingent on the 30% reduction occurring in the upstream, freshwater reach. These nutrient load reductions were adopted as part of the TMDL.

7.4 Knowledge Gaps and Uncertainties

Water quality models may be classified as either empirical or mechanistic. Empirical models depend on relationships between constituents derived solely from observed data, and can be an efficient way to develop usable relationships under certain data domains. Mechanistic models utilize mathematical relationships to represent chemical, biological, and physical transformations. When linked with hydrodynamic models, mechanistic water quality models can supply information regarding movement and transformations of constituents of interest within a system. The advantage of mechanistic models based on first principles is that they can be used to make predictions outside the domain of the data used for initial development.
There are uncertainties and knowledge gaps in application of both these types of models, as follows:

**Empirical Models.** The assurance of cause and effect linkages can be misleading. Many times the linkages are treated as a black box. This can lead to inappropriate assumptions and conclusions regarding causal relationships.

It is important to have an adequate data (i.e., adequate model domain) to capture the range of variability in loadings and water quality response data needed to make predictions. Empirical models should only be used to predict conditions that exist within the model domain. As an example, estimating the effects of a hurricane using a relationship developed from data which do not capture the extreme conditions present during such an event is an inappropriate extrapolation of any derived relationships.

**Mechanistic Models (Water Quality and Hydrodynamic).** The model chosen for a particular application must be:

- appropriate for the waterbody being simulated, with consideration of waterbody type (e.g., river, estuary) and its physical nature (e.g., bathymetry) and
- appropriate for the required spatial resolution (one-, two-, or three-dimensional) and temporal resolution (e.g., hourly, monthly, annually).

Other knowledge gaps and uncertainties include:

- Input data (including boundary conditions, initial conditions, and calibration and validation data) are rarely measured without error. Lack of consideration of this error can lead to very spurious results.
- The literature value rate processes used in these models typically vary over a wide range and are often not waterbody-specific. Even less complex mechanistic models utilize many coefficients and rates to represent transformations, and more complex models (those including dissolved oxygen and phytoplankton kinetics, for example) require many more. Therefore, when data for such rate processes are lacking, critical assumptions involving the values used in the model are necessary. These assumptions are often not tested for a specific model application.
- Rate processes may vary spatially and temporally (e.g., sediment oxygen demand variation seasonally and with sediment type), and many mechanistic models have provision for time- and space-variable rates. However, without adequate data, using simplified assumptions for the values for these variable rates will result in erroneous conclusions, especially with respect to the importance of certain parameterized processes.
- Uncertainties also include the availability of adequate boundary conditions, as well as, adequate data to drive the model.
- Adequate data for model calibration and validation affects the confidence that the model can be applied to evaluate system responses, when driving conditions are other than those represented by the calibration and validation domains. This is in contrast to predictions using regression models, which are constrained to only those conditions represented in the model domain.
♦ Inadequate model performance testing can lead to misleading model predictions and can contribute to model uncertainty.

♦ Sensitivity analyses are essential in assessing model uncertainty with respect to critical rate processes and state variables.

♦ The objectives of a modeling study are aided by selecting an appropriate horizontal and vertical grid structure. The most common horizontal grid systems are fixed or stretched rectangular or curvilinear systems, with curvilinear systems providing better resolution of curving coastlines. The most common vertical grid structures are rectangular and stretched, with stretched grids allowing the same number of vertical cells at any depth.

♦ The numerical solution scheme employed within a model depends upon the objectives of the study. Schemes are classified as implicit or explicit, with the optimal scheme a function of the problem being addressed. Explicit schemes have stricter stability limits on allowable time step lengths, which impact model run time. Implicit schemes allow longer time steps, but may introduce phase errors in results.
Implementation of nutrient control strategies is not the focus of this report; however, the topic deserves mention due to its level of importance in successfully achieving water quality standards. Any pollution control plan, no matter how well considered and developed, will be ineffectual unless implemented in a manner that allows it to succeed.

In the process of finalizing a nutrient control strategy, the stakeholder group must ensure that each element of the plan will be implemented in a coordinated, timely manner. This will be better accomplished if all group members are involved in the development of the plan. To make sure that this happens, the control strategy should reflect the nature of the goals, the leadership structure, the complexity and size of the proposed actions, and the expertise of the individuals and organizations involved.

Stakeholder consensus must be reached on the path from the original goals and expectations, to the required task inputs and how they will contribute to the achievement of those expectations, the expected outcomes, the assumptions made about those links; and how progress will be evaluated.

Once the goals of the strategy are agreed upon, responsibilities should be assigned to all participating parties. To facilitate progress, specific functional elements of the plan should be examined and a stepwise approach developed for each, considering required costs and level of effort involved, expected time frame for completion, and any anticipated obstacles. Implementation timelines are especially critical when activities are interdependent. Additionally, any potential changes in the economic or legislative environment that may affect plan implementation should be identified. Clear measures of success for each step of implementation should be identified so that corrective measures can be taken as appropriate if needed. The entire strategy must be thought through so that any information or resource gaps can be addressed proactively.

The successful implementation of a pollution control strategy is illustrated in the work of the Nitrogen Management Consortium (NMC) of the Tampa Bay Estuary Program (TBEP). In 1996, the TBEP’s governmental partners joined with key industries in the Tampa Bay region to create a unique public/private partnership for the express purpose of developing a Consortium Action Plan to meet the “hold the line” nitrogen loading target for Tampa Bay. The goal of this approach remains to reduce current nitrogen loads to Tampa Bay at a rate that offsets predicted increased loads caused by continuing development.

The original Action Plan consisted of more than 100 projects, voluntarily implemented by NMC members, which collectively reduced or precluded nitrogen discharges to the bay by an estimated 134 tons/year between 1995 and 1999. The NMC Action Plan was the core of a nitrogen management strategy that included: meeting chlorophyll and nitrogen reduction targets for all parts of the bay, apportionment of responsibility for meeting the nitrogen reduction targets, and sustaining a process to track whether the targets were being met.
In November 2002, the Florida Department of Environmental Protection (FDEP) concluded that the TBEP’s nitrogen management strategy provided reasonable assurance that the state water quality criteria for nutrients in Tampa Bay would be met. The U.S. Environmental Protection Agency (U.S. EPA) in the meantime continues to recognize a 1998 action by FDEP that proposed a total maximum load of nitrogen that could be discharged to the bay annually and still meet state standard water quality criteria. Both FDEP’s reasonable assurance determination and the total maximum nitrogen loading recognized by the U.S. EPA are based on statistical modeling and data analyses done by TBEP and its partners.

To help ensure continued recovery of vital seagrass habitat and the successful nitrogen management strategy that makes it possible, local governments, agencies, and industry members of the NMC continue to work together to provide FDEP with an updated reasonable assurance document, ensuring that state water quality criteria for nutrients will continue to be met in the bay. The partners are committed to develop an equitable allocation of responsibility for achieving nitrogen reduction targets as well as the total maximum loading of nitrogen to the bay. The amount of nitrogen discharge (i.e. load) allocated to each municipal and industrial facility will take into consideration expected growth in population and/or demand for manufactured products. The NMC partners accept that the agreed-upon allocations of nitrogen load will be the basis for future permitting of nitrogen discharges from municipal and industrial facilities.
A key element in the overall framework linking nutrient sources to water quality impacts and the mitigation of these impacts is the development of an effective monitoring and assessment approach. The following discusses some of the considerations to be made in developing a monitoring and assessment program whose goal is to determine whether the implementation of a control strategy has resulted in improved water quality conditions in the receiving waterbody.

9.1 Monitoring Potential Water Quality Improvements Due to Nutrient Control Strategies

There are multiple scales of inference when establishing a process for monitoring and assessing the effectiveness of nutrient controls. A local scale assessment can be used to evaluate the effectiveness of a particular nutrient control strategy such as a stormwater treatment pond. A watershed scale monitoring and assessment program is more likely targeted to evaluate the effectiveness of the sum of source control strategies within the watershed. Both assessments rely on chemical or biological indicators or “endpoints” characterizing the effects of nutrients in the system. Irrespective of the scale of inference, measuring the effects of source controls is heavily dependent on the design of the monitoring program. Water quality monitoring designs should be based on ensuring that the effects of nutrients and the nutrient control strategy can be quantified. Important considerations in the design include:

♦ Existing knowledge
♦ Identifying the spatial scale of inference
♦ Identifying the temporal scale over which the control should be evaluated
♦ Identifying methods and limitations of the methods (e.g. detection limits)
♦ Identifying the effect level that would constitute a significant outcome
♦ Identifying the sample size required to determine with statistical confidence that the effect level has been achieved
♦ Identifying any potential confounding factors that might mask the true relationship between the control and its effect
♦ Identifying limitations of the design in scope and inference

Cost is often a major limiting factor in the design and implementation of water quality monitoring programs and one that often results in the failure of a program to meet its objectives over time. Cost projections should be explicitly outlined in the design with the understanding that many associated costs are not fixed. Major program costs include laboratory, equipment and data handling costs. However, often overlooked are the logistical costs associated with the data collection itself. This cost often represents a major component of the overall cost and it is imper-
ative that inefficiencies in data collection efforts be minimized in order to maintain an effective program.

One example of a broad scale water quality monitoring network in Florida is the South Florida Water Management District’s Water Quality Monitoring Network with over 2,000 sampling stations, 35,000 sampling events each year, and an annual cost of 18 million dollars (SFWMD, 2008). This program includes data collection efforts that span from tribal lands to residential canals to nonpoint source treatment areas associated with the Comprehensive Everglades Restoration Plan (CERP). This program has undergone nearly 40 optimizations over time to refine the program to meet the demands and mandates of Florida’s regulatory arena (SFWMD, 2008). Despite these optimizations, refinements continue in an attempt to maximize the information derived while minimizing the substantial associated cost. Much of this cost has been identified as field and logistic costs. This serves as but one example of how cumbersome a task it is to establish a monitoring program to meet multiple, sometimes conflicting, monitoring objectives over a regional scale by combining different previously established sampling programs with different objectives.

There are three general categories of controls for nutrients in the environment: regulatory controls; technological controls, and educational controls. Each of these categories may require a different implementation strategy and different analytical techniques to evaluate the effectiveness of the control. While there are many types of water quality monitoring designs, there are generally three types of assessments used to assess water quality:

- Statistical differences in populations
- Violations of water quality standards
- Trends over time

The selection of which of these three types of analyses to use is dependent upon the goals, objectives, and design of the program, emphasizing again that the establishment of quantifiable objectives and sound design are imperative to establishing an effective program. Judging the effectiveness of the control often relies on statistical analysis which employs hypothesis testing. Therefore, it is not unreasonable that the goals and objectives could be developed to include a specific set of hypotheses to be tested after implementation. Program implementation without careful consideration of the ways the data will be analyzed leads to skepticism regarding inferences drawn based on data mining techniques rather than well constructed hypotheses tests. These considerations are especially important in regulatory and legal frameworks where results and conclusion are heavily scrutinized because the resulting inference can literally be millions of dollars in regulatory impositions.

Within each of the categories described above, there are a host of statistical techniques available to the analyst. Analysis of differences in populations can be constructed to test the effectiveness of structural BMPs using, for example, a sampling program designed with upstream and downstream sampling stations or inflow and outflow stations (U.S. EPA, 2010i). In this design, samples are paired as they are taken from the same stream at the same time. This type of analysis would benefit from baseline water quality data collected prior to the implementation of the BMP. Other examples may include case control studies or paired watershed designs in which a “treatment” is assigned to one of two systems which are similar in all other aspects related to water quality. Again, in these designs it is important to consider the potential confounding factors such as potential differences in the biogeochemistry of the watersheds under study (Helsel
and Hirsh, 1992). Many times, if differences are known a priori they can often be accounted for and factored out of the analysis through the use of appropriate statistical analysis techniques such as analysis of covariance (Zar, 1984). Time series analysis is an effective means of assessing the effectiveness of regulatory and educational controls on water quality provided that other nutrient sources can be accounted for. Advantages of time series analysis is that seasonal variability can be accounted for as well as serial autocorrelation that often exists from sampling the same sites sequentially through time (Reckhow, 1993). Implementation of such controls may result in discrete changes in the time series signal such as a step function or in linear or non-linear reductions in concentrations which can be pinpointed through the appropriate use of trend detection methods. Consideration for if and how natural variability in environmental condition over multiple time scales is an important attribute to consider as well as how events such as floods, drought or hurricanes may affect water quality assessments.

Recently in Florida there have been several successful efforts to legislate local ordinances as nutrient pollution control measures. For example, Florida’s Sarasota and Pinellas counties have recently passed ordinances banning the sale and application of fertilizers during Florida’s summer rainy season. These types of controls are both regulatory and educational in that they inform the public of the sources, transport, and fate of an important anthropogenic nutrient source.

9.2 Evaluate Effectiveness of Plan Components

Irrespective of the type of control or the type of analytical technique used to measure the effectiveness of the control, there are principal questions that the assessment framework needs to address. These questions include:

♦ Was the control implemented properly?
♦ Were actions effective in meeting objectives and obtaining desired outcome?
♦ Can the results be generalized to other systems?
♦ What are the confounding factors and limitations?

These questions can be addressed as follows. Data collected from the monitoring program is assessed based on a previously established model construct developed during the planning phase. Analytical outcomes are then reviewed and reported. The relevance to controls is assessed; the effects attributable to source controls are documented, and any potential confounding factors that lend additional insights are identified that may be used to refine the model construct.

Feedback from this assessment process then guides refinement of the models, the source control strategy and future monitoring efforts.

Pilot projects in smaller watersheds are also an excellent way of evaluating potential control strategies and are recommended as an effective means of providing a concise evaluation of the efficacy of a potential control. Pilot studies provide information on knowledge gaps and potential confounding factors and provide feedback of the potential success of the controls as well as the analytical methods chosen for inference. Pilot studies provide an important feedback mechanism that can guide subsequent steps in the evaluation of controls. Findings from initial pilot studies should be documented and presented for review. A list of potential enhancements to
the control design, study design, and analytical approach should be documented and considered. Factors that limit the potential generalization of the techniques should also be documented. If the control is to be implemented on a broader scale based on findings from initial studies, coordination and standardization of methods are of utmost importance. The integration of information from multiple agencies is a big challenge in reporting on the effectiveness of the control. Generalization of the controls also provides additional information on how the optimal efficiencies of the controls will respond over a more general set of conditions and therefore, the generalization should be considered as a larger study that includes all the components of the initial plan. Therefore, the management paradigm begins again at a larger scale with the establishment of specific objectives, careful planning and a holistic understanding of the costs and expected benefits of implementing the control.

An example of a successful nutrient control assessment program is the Tampa Bay Estuary Program (TBEP) Nitrogen Management Consortium (NMC), which is a public/private partnership made of local governments and industries and regulatory agencies that have collectively agreed to contribute to the management of nitrogen sources in the Tampa Bay watershed (Greening and Janicki, 2006). The TBEP developed a decision framework to guide management actions. The decision framework incorporates the results of annual assessments of two key biological indicators for the ecological health and success of seagrasses in Tampa Bay. The result of the decision framework leads to several potential outcomes dependent upon magnitude and duration of events in excess of the established targets (Janicki et al., 2000). When outcomes for the indicators suggest conditions are sufficient for seagrass success, no management response is required. When outcomes suggest conditions are marginal, further examination into the potential factors causing those marginal conditions is investigated. When conditions are problematic, such that there are relatively large, longer term differences from either or both targets, then stronger management responses are warranted. For example, a strong El Nino pattern in 1998 resulted in abnormally high rainfalls and poor scores for the decision matrix. Repair of sewer transport and pumping systems and industrial treatment water holding systems that had failed during high rainfall amounts were identified as problems and actions were taken by municipalities and industrial facilities to address these failed systems. This illustrates the benefits of consortia and the involvement of local stakeholders in the overall nutrient control strategy. This assessment mechanism provides an expedient and effective local response that expedites corrective measures in lieu of regulatory impositions which can be obscure, lengthy and inefficient in identifying sources and specific corrective actions.
CHAPTER 10.0

CONCLUSIONS AND RECOMMENDATIONS

10.1 Summary

The Water Environment Research Foundation (WERF) has undertaken a research challenge entitled “Linking Receiving Water Impacts to Sources and to Water Quality Management Decisions”. The research described in this report provides an initial case study towards this larger challenge and focuses on nutrients as a source constituent, primarily nitrogen. The purpose of this effort was to provide a state-of-the-knowledge approach to establishing the linkages between sources of nitrogen, the predicted and measured adverse impacts of nitrogen on receiving waters, and the costs and benefits of controls available to address these adverse impacts. Additionally, a goal of the research was to identify the knowledge and information gaps that need to be addressed such that the approach can be developed and implemented for a specific waterbody.

Based on an extensive review of the literature, a framework for an approach to establishing these linkages was developed. The framework included establishing water quality impacts, linking these impacts to nutrients, quantifying major nitrogen sources, evaluating the costs and benefits of available nitrogen controls, estimating receiving water responses to controls, and finally, implementing a nitrogen control strategy and assessing water quality for potential improvements.

10.2 Conclusions

Nutrient pollution is one of the leading causes of water quality impairment in the nation, and the quantity of nutrients reaching the nation’s waters has dramatically escalated over the past 50 years (U.S. EPA, 2009a). Nitrogen and phosphorus loadings to waterbodies impact water quality by stimulating plant and algae growth which subsequently may result in depletion of dissolved oxygen, degradation of habitat, harmful algal blooms, impairment of the waterbody’s designated use, and impairment of drinking water sources.

The following sections present the major conclusions, by major subject area, drawn from the state-of-the-knowledge review and approach developed for this project.

10.2.1 Water Quality Impacts

There were numerous studies and sources of data providing evidence that waterbodies respond negatively to an excessive supply (loading rate) of the nutrients nitrogen and phosphorus, resulting in the characteristic symptoms of eutrophication. While knowledge of the links between pollutant inputs and waterbody responses may be more advanced for nutrients than for other emerging constituents, significant gaps remain in the understanding of the degree to which waterbodies can assimilate nutrient loading without reflecting these symptoms. The following conclusions were drawn from the literature reviewed in this study.
Since nitrogen and phosphorus are necessary ingredients for life in aquatic ecosystems, developing nutrient criteria is more complicated than it is for toxic substances, or constituents whose desired quantity is extremely low or zero. While most water quality criteria in the CWA are based on a toxicity threshold where acceptable concentrations can be established below the level found to be harmful, nutrients at some level are necessary to support biological communities, and these natural levels change with environmental conditions. This can lead to considerable difficulty in establishing nutrient concentration limits to meet water quality goals.

Each waterbody is different, and there are numerous factors which impact how a particular waterbody will respond to excess nutrient loading, including hydraulic residence time, freshwater inflow, clarity and light attenuation, geologic substrate, depth, temperature, and degree of physical alterations, such as impoundment or channelization. Additionally, it is often difficult to identify the causative nutrient in eutrophic waterbodies. Some systems vary between nitrogen limitation, phosphorus limitation, or nitrogen-phosphorus co-limitation in response to the relative contribution of each nutrient by season and/or source.

Determining a causative agent for eutrophication impairment is often not straightforward, therefore, establishing a dose-response relationship between nutrients and water quality is challenging. Nutrient enrichment and the resulting problems are frequently widespread and are manifested at a location remote from the sources, and may not show themselves for some time after significant inputs to the system have occurred. Without an understanding of the dose-response relationship, setting numeric concentration criteria is difficult. Clearly, such a relationship is very difficult to develop for a group of waterbodies, and therefore typically requires a site-specific approach based on the characteristics of the individual waterbody.

Development of TMDLs for nutrients without first discerning the dose-response relationships between nutrients and chlorophyll and DO is not justified. Too often both chlorophyll and DO impairments are attributed to excessive nutrients by default. Such ill-conceived conclusions lead to TMDL development that can seriously affect the stakeholders associated with the impaired water by requiring nutrient controls that will have little effect on the water quality.

The cause of impairment of a waterbody should be determined with relative certainty prior to making any management decisions. The cause may be excess nitrogen, excess phosphorus, or other agent (hydrologic alteration, etc.).

The response of an impaired waterbody to control measures may be non-linear. That is, there may not be a direct one-to-one relationship between reduced loadings and improved water quality. At times the aquatic system may need a considerable change in equilibrium to initiate the recovery process. This may result in a significant lag between the time of applying controls and observing a response from the waterbody.

Information obtained from examining N:P ratios can be useful for nutrient management. However, the ratios can vary spatially and most notably temporally. When very costly controls are needed to address nutrient loading, the increased certainty afforded by the application of either laboratory or in situ testing to identify the limiting nutrients should be considered.
The lack of limitation by either nitrogen or phosphorus can lead to ineffective and unnecessary controls. Nutrient load reductions to a waterbody that is light-limited due to colored dissolved organic matter (CDOM) may result in no change in resultant water quality, for example.

The role of alterations to the hydrologic system on residence time should be considered when investigating the causes of eutrophication. Lower freshwater inflows to a waterbody, or retention of water by a control structure such as a dam, may result in less flushing and longer residence time, while channelization or flood control alterations may increase flow rates and decrease residence times. Either types of alteration could impact water quality and be a cause of impairment.

10.2.2 Nitrogen Sources

Nitrogen is one of the earth’s most abundant elements, and is essential for both plant and animal life within the biosphere, that portion of the earth’s crust and atmosphere where living things reside. However, as a result of man’s activities there has been a tremendous increase in the amount of reactive nitrogen available in the global environment over the past 50 years. Figure 4-2 illustrated the increase in reactive nitrogen in the biosphere over the last century. Globally, the increase in biologically available nitrogen in our biosphere has clearly been influenced by the combustion of fossil fuels and the production of synthetic fertilizers, and these sources are projected to increase further in the future. Conclusions regarding nitrogen sources are summarized below:

- Biological nitrogen fixation (BNF) has been enhanced via increased cultivation of nitrogen fixing crops for agriculture.
- The combustion of fossil fuels, previously locked away in the earth’s crust, releases new reactive nitrogen compounds into the atmosphere that then return to earth via atmospheric deposition.
- The Haber-Bosch process was developed to synthesize reactive ammonia from gaseous N\textsubscript{2}. This industrial process began to be used extensively for synthetic fertilizer production in the 1950s, and its use has grown exponentially since.
- Nitrogen can enter a receiving waterbody through atmospheric deposition, nonpoint source discharge, or point source discharge pathways. However, these linkages are not the ultimate sources of nutrients. The key culturally-derived nitrogen sources are combustion of fossil fuels, fertilizer and animal waste, and human and industrial waste.
- In Florida, nitrogen loading estimates have been developed for numerous watersheds. Table 4-2 provided a summary of the relative contributions (%) by source of nitrogen loads to a number of Florida receiving waters. While these relative contributions varied appreciably among waterbodies, it is clear that nonpoint sources were a major contributor to nitrogen load in every case.
- There are many tools available with which to quantify nitrogen loadings to receiving waters, including empirical and mechanistic models. The proper use of an appropriate model can greatly aid in the management of nutrient impacts to surface waters.
10.2.3 Nitrogen Controls

There are two ways to control nutrient loadings to receiving waterbodies. The first method of nutrient control is to manage the quantity of nutrients introduced into the environment from the ultimate sources. For example, agricultural and urban fertilization practices can be optimized to reduce the amount of excess nutrient inputs to the environment.

The second method is to remove the nutrients after they have entered the network of pathways and reservoirs between release and delivery to receiving waterbodies. Improvement in wastewater treatment is a good example of this approach that has already been successful. Nutrients in nonpoint sources can also be removed by the use of a variety of stormwater treatment alternatives. Atmospheric sources can be reduced by treatment of power plant and vehicle emissions as they are discharged. Key conclusions developed regarding nitrogen controls follow:

♦ If WWTP effluent in a watershed was required to meet the recently proposed Florida in-stream numeric nutrient criteria, it would be difficult to do so consistently with currently accepted limits of wastewater treatment technology. The currently accepted practical limit of treatment technology for nitrogen is approximately 3 mg/L TN, although emerging technologies have shown that further reductions to 1-2 mg/L may be possible with additional research.

♦ The cost of increasing wastewater treatment to remove nutrients to very low levels is substantial. Figure 6-1 illustrated that the incremental O&M cost increase per additional pound of nitrogen removed as treatment level increases from secondary to a TN effluent limit of 1 mg/L (Reardon et al., 2010) is very high.

♦ Conventional onsite wastewater treatment systems (OWTS) are not designed for nitrogen removal, and there are an estimated 2.5 million OWTS in Florida (Briggs, Roeder, et al., 2007). While some level of nitrogen removal occurs naturally in the soil below these systems, they can be a substantial contributor to nitrogen loading in some watersheds. New developments in OWTS technology have resulted in systems that achieve nitrogen removals of up to 75%, but these systems are expensive, difficult for homeowner operation, and sometimes difficult to retrofit into existing properties.

♦ The management of OWTS by routine evaluations and maintenance requirements can improve OWTS performance and identify failures that may be impacting water quality. Requirements for such a program have recently been passed into law in Florida, with implementation scheduled for 2011.

♦ Concentrated animal feeding operations (CAFOs) are estimated to generate an estimated 500 million tons of manure annually, over 3 times the estimated waste generated by the human population of the U.S. This agricultural waste can impact water quality via point source discharges or as a component of nonpoint source inputs to receiving waters and are not required to meet the same effluent quality requirements as domestic waste.

♦ Fertilizer restriction ordinances and education are controls that can be used to reduce the nitrogen loads in stormwater runoff to receiving waters from urban landscapes. A recent statewide Urban Turf Fertilizer Rule adopted in Florida is expected to reduce the nitrogen fertilizer applied to urban turf and lawns by 20-25%.

♦ Florida requires treatment of all stormwater discharges for new development. Current technologies used for stormwater treatment systems in Florida effectively remove 20-30% of TN
from stormwater discharges. However, the lack of end-of-pipe monitoring for stormwater discharges makes it extremely difficult to estimate stormwater discharges for nutrients.

- Atmospheric deposition of nitrogen is a major source of nitrogen loading in urban watersheds. The Tampa Bay Estuary Program lists atmospheric deposition as the second leading source of nitrogen loads to Tampa Bay. Nitrogen controls recently completed at the TECO Big Bend power plant are expected to reduce the nitrogen load to Tampa Bay by approximately 60 tons per year.

- Water quality trading is a market-based approach to achieving nutrient removal goals at the lowest possible cost by allowing the voluntary exchange of nutrient credits for money. Under the right circumstances, a well-designed and properly implemented water quality trading system can be an effective tool to achieve water quality goals. Consideration of many specific issues is necessary to evaluate, design and implement a successful water quality trading program. Consultations with economists, those who have implemented past programs, and experts at the U.S. EPA and WERF are recommended.

### 10.2.4 Water Quality Management Decisions

Water quality management decisions required to restore nutrient impaired waterbodies are complex and extend far beyond the technical decisions. The development and implementation of nutrient control strategies must consider the economic, social, and environmental considerations as well as the technical. In the past, water quality improvement projects have been based on a simple cost comparison of the top technically ranked alternatives far too often. In reality, there are many costs and benefits associated with water quality improvements, and the best water quality management decisions are based on consideration of all these costs and benefits. Key conclusions from this review are listed below:

- The economic value of avoiding nutrient enrichment of waterbodies is likely to be substantial and includes income, use, non-use, and property values.

- To properly evaluate nutrient control strategies on like terms, all benefits and costs must be considered. The chosen level of nutrient reduction should be based on maximizing the net benefits of nutrient management, where net benefits are the benefits to all persons and entities minus the costs to all persons or entities now and in the future.

- When benefits, costs and public input are not sufficiently considered, the actual design and implementation of nutrient management alternatives can result in choices where the costs are greater than the benefits and/or where the most cost-effective alternatives are not implemented.

- An example illustrating this is wastewater treatment improvements. Although it has been demonstrated that wastewater treatment plants can approach carbon neutral operation (Wett, 2007) for a total nitrogen limit of 14 mg/L, energy neutrality becomes increasingly unlikely as stricter nitrogen limits are imposed. Figure 6-2 showed the incremental greenhouse gas (GHG) emissions increase associated with the increases in treatment level from secondary to TN effluent limits of 1 mg/L (Reardon et al., 2010). The social and environmental costs of these emissions need to be considered in determining the net environmental benefit of improved treatment.
A benefit-cost evaluation model can be developed to evaluate and rank nitrogen control alternatives. Such models provide a structured approach to:

- Achieving an apples-to-apples comparison of nitrogen control alternatives
- Identifying data and information gaps
- Evaluating the impact of uncertainty on the ranking of each alternative
- Providing useful input into the design of nutrient reduction programs
- Updating the conclusions as new data and information become available

The benefit-cost evaluation model should be designed so that the benefits and costs of nutrient reduction alternatives are better understood; politically sensitive issues can be put into perspective; and stakeholders’ interests are placed on a level playing field.

For water quality improvement alternatives, benefits and costs typically include: value to the economy measured by household income; resident wealth and well-being; use value of recreation activities; non-use value of the ecosystem supported by the waterbody; value as a potable water supply source; avoided costs of disease; replacement cost of ecosystem services; and value of commercial fish harvested.

For some types of benefits and costs, dollar values can be relatively easy to estimate. For other types of benefits and costs, the amount of research necessary to estimate dollar values is relatively high so other measures that are relatively inexpensive to estimate and that reflect the dollar value are often used instead.

Benefits and costs should be assigned monetary values to the extent that the available time and budget will allow. Where insufficient information exists to assign a monetary value, benefits and costs should be described and quantified to the extent practical. In addition, stakeholder involvement in the decision-making process and in the implementation of the nutrient control strategy is critical to success.

It is important to assess nutrient reduction technologies and alternatives well before they need to be implemented because time is needed to estimate the benefits and costs of each alternative, including estimates of changes in income, use, non-use and property values associated with changes in nitrogen or nutrient levels.

10.2.5 Water Quality Improvements – Estimating Receiving Water Response to Controls

Once impairment of a waterbody has been substantiated and sources and controls identified, an evaluation of the expected waterbody response must be conducted. This can be accomplished using either empirical or mechanistic modeling techniques. Key conclusions drawn from the review of this topic are summarized below:

- Both empirical and mechanistic models can be useful in predicting the potential water quality benefits from control measures. Information obtained from the use of these tools can be helpful in establishing water quality targets and determining the assimilative capacity of waterbodies. They may also be useful in determining potential dose-response relationships.

- Empirical models depend on relationships between constituents derived solely from observed data, and can be an efficient way to develop usable relationships under certain data domains.
However, the assurance of cause and effect linkages can be misleading when using empirical models. Many times the linkages are treated as a black box that can lead to inappropriate assumptions and conclusions regarding causal relationships.

♦ It is important to have adequate data to capture the range of variability in loadings and water quality response. Empirical models should only be used to predict conditions that exist within the model domain.

♦ Mechanistic models utilize mathematical relationships to represent chemical, biological, and physical transformations. When linked with hydrodynamic models, mechanistic water quality models can supply information regarding movement and transformations of constituents of interest within a system. Mechanistic models are often very data intensive and are based on rate constant values that may not be well-understood. Significant assumptions may also be necessary when site-specific data for model inputs are lacking.

### 10.3 Recommendations for Research Needs

Numerous knowledge gaps and the need for additional data were discovered during the literature review and in the development of the framework for this project. These identified needs range in scope from simple data collection needs to major research efforts related to cause and effect relationships between water quality and the major constituents of concern. Knowledge gaps were identified at the end of each chapter, and the key research needs formulated during the conduct of this project are summarized below.

♦ Lack of compliance with an existing water quality standards or criterion often provides the impetus for the investigation of the linkages between nutrient sources and resulting water quality. Improper criteria, i.e., those that are not defensible or overly-protective, can result in improper water quality management decisions. Therefore, it is absolutely essential that the development and implementation of nutrient criteria be built upon robust, defensible methodologies and data sources. Many states are facing the imposition of numeric nutrient criteria by the U.S. EPA and the lack of appropriate methodologies and data sources remains as a significant gap. Clearly, efforts to fill this gap are warranted if unnecessary nutrient controls are to be avoided. Therefore, it is recommended that a toolbox of potential methodological approaches to the establishment of numeric nutrient criteria be developed. These approaches must build on the specific knowledge of how waterbodies respond to variation in nutrient conditions and most importantly on the choice of appropriate response variables and defensible thresholds. The nature and volume of the available data for both stressor and response variables, as well as for any significant confounding factors, must also be considered.

♦ To better understand the linkage between nutrient sources and receiving water impacts, research should be undertaken to better identify and quantify the transformations of nutrients in the environment, including the confounding factors that affect the behavior of nutrients in aquatic systems.

♦ While it is obvious from the literature that fertilizer use and combustion of fossil fuels are key sources of nitrogen to the environment, few studies have quantified the nutrient load to a given watershed resulting specifically from these sources. Detailed watershed studies documenting the specific contributions from these sources should be conducted to develop a bet-
ter understanding of the potential role that nutrient controls can play in managing their impacts.

♦ Additional research is needed on the nitrogen removal potential of current and emerging wastewater treatment technologies in order to properly evaluate which processes will perform reliably to meet extremely low proposed nitrogen limits for a waterbody. The impact of increased energy use, GHG emissions, and other social and environmental costs associated with increased treatment also needs to be considered in the evaluation of net benefits.

♦ Onsite wastewater treatment systems serve approximately one-third of the population, yet research expenditures to improve the treatment performance of these systems for nutrient removal has been only a small fraction of that allocated for municipal WWTPs. Further research on nutrient removal technologies for new as well as existing OWTS, is warranted.

♦ Additional monitoring of “end of pipe” stormwater discharges is needed to understand their contribution to a waterbody’s impairment and to properly evaluate and implement controls methods.

♦ Research is needed on the economic and environmental benefits of nutrient recovery and recycling, as well natural treatment technologies and improvements to existing best management practices pertaining to CAFOs.

♦ There is a lack of studies that have determined the change in income, use, non-use, and property values associated with changes in ecosystem quality, especially related to reduced nutrient loading. Thus, a major research gap is an understanding of how the monetary values generated by our nations waterbodies change as water quality improves or impairments increase. This research could be conducted immediately using publicly available information and survey research.
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Linking Receiving Water Impacts to Sources and to Water Quality Management Decisions: R-41 Using Nutrients As An Initial Case Study


WASTEWATER UTILITY

Alabama
Montgomery Water Works & Sanitary Sewer Board

Alaska
Anchorage Water & Wastewater Utility

Arizona
Avondale, City of
Glendale, City of,
Utilities Department
Mesa, City of
Peoria, City of
Phoenix Water Services Dept.
Pima County Wastewater Management
Tempe, City of

Arkansas
Little Rock Wastewater Utility

California
Central Contra Costa Sanitary District
Corona, City of
Crestline Sanitation District
Delta Diablo Sanitation District
Dublin San Ramon Services District
East Bay Dischargers Authority
East Bay Municipal Utility District
El Dorado Irrigation District
Fairfield-Suisun Sewer District
Fresno Department of Public Utilities
Inland Empire Utilities Agency
Irvine Ranch Water District
Las Gallinas Valley Sanitary District
Las Virgenes Municipal Water District
Livermore, City of
Los Angeles, City of
Los Angeles County, Sanitation Districts of
Napa Sanitation District
Novato Sanitary District
Orange County Sanitation District
Palo Alto, City of
Riverside, City of
Sacramento Regional County Sanitation District
San Diego Metropolitan Wastewater Department, City of
San Francisco, City & County of
San Jose, City of
Santa Barbara, City of
Santa Cruz, City of
Santa Rosa, City of
South Bayside System Authority
South Coast Water District
South Orange County Wastewater Authority
South Tahoe Public Utility District
Stege Sanitary District
Sunnyvale, City of
Union Sanitary District
West Valley Sanitation District

Colorado
Aurora, City of
Boulder, City of
Greeley, City of
Littleton/Englewood Water Pollution Control Plant
Metro Wastewater Reclamation District, Denver
Platte Canyon Water & Sanitation District

Connecticut
Greater New Haven WPCA
Stamford, City of

District of Columbia
District of Columbia Water & Sewer Authority

Florida
Broward, County of
Fort Lauderdale, City of
Jacksonville Electric Authority (JEFA)
Loxahatchee River District
Miami-Dade Water & Sewer Authority
Orange County Utilities Department
Pinellas, County of
Reedy Creek Improvement District
St. Petersburg, City of
Tallahassee, City of
Tahoe Water Authority
West Palm Beach, City of

Georgia
Atlanta Department of Watershed Management
Augusta, City of
Clayton County Water Authority
Cobb County Water System
Columbus Water Works
Fulton County
Gwinnett County Department of Public Utilities
Savannah, City of

Hawaii
Honolulu, City & County of

Idaho
Boise, City of

Illinois
Decatur, Sanitary District of
Greater Peoria Sanitary District
Kankakee River Metropolitan Agency
Metropolitan Water Reclamation District of Greater Chicago
Wheaton Sanitary District

Indiana
Jeffersonville, City of

Iowa
Ames, City of
Cedar Rapids Wastewater Facility
Des Moines, City of
Iowa City

Kansas
Johnson County Wastewater Unified Government of Wyandotte County/Kansas City, City of

Kentucky
Louisville & Jefferson County Metropolitan Sewer District
Sanitation District No. 1

Louisiana
Sewerage & Water Board of New Orleans

Maine
Bangor, City of
Portland Water District

Maryland
Anne Arundel County Bureau of Utility Operations
Howard County Bureau of Utilities
Washington Suburban Sanitary Commission

Massachusetts
Boston Water & Sewer Commission
Massachusetts Water Resources Authority (MWRA)
Upper Blackstone Water Pollution Abatement District

Michigan
Ann Arbor, City of
Detroit, City of
Holland Board of Public Works
Saginaw, City of
Wayne County Department of Environment
Wyoming, City of

Minnesota
Rochester, City of
Western Lake Superior Sanitary District

Missouri
Independence, City of
Kansas City Missouri Water Services Department
Little Blue Valley Sewer District
Metropolitan St. Louis Sewer District

Nebraska
Lincoln Wastewater & Solid Waste System

Nevada
Henderson, City of

New Jersey
Bergen County Utilities Authority
Ocean County Utilities Authority

New York
New York City Department of Environmental Protection

North Carolina
Charlotte/Mecklenburg Utilities
Durham, City of
Metropolitan Sewerage District of Buncombe County
Orange Water & Sewer Authority

Ohio
Akron, City of
Butler County Department of Environmental Services
Columbus, City of
Metropolitan Sewer District of Greater Cincinnati
Montgomery, County of
Northeast Ohio Regional Sewer District
Summit, County of

Oklahoma
Oklahoma City Water & Wastewater Utility Department
Tulsa, City of

Oregon
Albany, City of
Clean Water Services
Eugene, City of
Gresham, City of
Portland, City of
Bureau of Environmental Services
Lake Oswego, City of
Oak Lodge Sanitary District
Water Environment Services

Pennsylvania
Philadelphia, City of
University Area Joint Authority

South Carolina
Beaufort-Jasper Water & Sewer Authority
Charleston Water System
Mount Pleasant Waterworks & Sewer Commission
Spaatzburg Water

Tennessee
Cleveland Utilities
Murfreesboro Water & Sewer Department
Nashville Metro Water Services

Texas
Austin, City of
Dallas Water Utilities
Denton, City of
El Paso Water Utilities
Fort Worth, City of
Houston, City of
San Antonio Water System
Trinity River Authority

Utah
Salt Lake City Corporation

Virginia
Alexandria Sanitation Authority
Fairfax, County of
Hampton Roads Sanitation District
Hanover, County of
Henrico, County of
Hopewell Regional Wastewater Treatment Facility

Louisiana Water

Lynchburg Regional Wastewater Treatment Plant
Prince William County Service Authority
Richmond, City of
Rivanna Water & Sewer Authority

Washington
Everett, City of
King County Department of Natural Resources
Seattle Public Utilities
Sunnyside, Port of
Yakima, City of

Wisconsin
Green Bay Metro Sewerage District
Kenosha Water Utility
Madison Metropolitan Sewerage District
Milwaukee Metropolitan Sewerage District
Racine, City of
Sheboygan Regional Wastewater Treatment
Wausau Water Works

Water Services Association of Australia
ACTEW Corporation
Barwon Water
Central Highlands Water
City West Water
Coliban Water Corporation
Cradle Mountain Water
Gippsland Water
Gladstone Area Water Board
Gold Coast Water
Gosford City Council
Hunter Water Corporation
Lagon Water
Melbourne Water
Onstream
Power & Water Corporation
Queensland Urban Utilities
South Australia Water Corporation
Sydney Catchment Authority
Sydney Water
Unity Water
Wannon Regional Water Corporation

Water Corporation
Water Distribution Brisbane City Council
Western Water
Yarra Valley Water

Canada
Edmonton, City of/Edmonton Waste Management Centre of Excellence
Lethbridge, City of
Regina, City of,
Saskatchewan
Toronto, City of,
Ontario
Winnipeg, City of,
Manitoba

New Zealand
Watercare Services Limited

STORMWATER UTILITY

California
Fresno Metropolitan Flood Control District
Los Angeles, City of,
Department of Public Works
Monterey, City of
San Francisco, City of
San Francisco, City of & County of
Santa Rosa, City of
Sunnyvale, City of

Colorado
Aurora, City of
Boulder, City of
Florida
Orlando, City of
Georgia
Griffin, City of
Iowa
Cedar Rapids Wastewater Facility
Des Moines, City of
Kansas
Lenexa, City of
Overland Park, City of
Kentucky
Louisville & Jefferson County Metropolitan Sewer District
Maine
Portland Water District
North Carolina
Charlotte, City of,
Stormwater Services
Pennsylvania
Philadelphia, City of
Tennessee
Chattanooga Stormwater Management
Texas
Harris County Flood Control District, Texas
Washington
Bellevue Utilities Department
Seattle Public Utilities

Connecticut Department of Environmental Protection

Kansas Department of Health & Environment
New England Interstate Water Pollution Control Commission (NEWPCC)
Ohio River Valley Sanitation Commission
Urban Drainage & Flood Control District, CO

CORPORATE

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AECOM
Alan Plummer & Associates
Alpine Technology Inc.
American Cleaning Institute
Aqua-Aerobic Systems Inc.
Aquateam–Norwegian Water Technology Centre A/S
ARCADIS
Associated Engineering
Bernhard Loehmuller & Associates
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Blue Water Technologies, Inc.
Brown & Caldwell
Burgess & Niple, Ltd.
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CABE Associates Inc.
The Cadmus Group
Camp Dresser & McKee Inc.
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Carpenter Environmental Associates Inc.
CET Engineering Services
CH2M HILL
CONTECH Stormwater Solutions
CRA Infrastructure & Engineering
CSIRO (Commonwealth Scientific and Industrial Research Organisation)
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EMA Inc.
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Post, Buckley, Schuh & Jernigan
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RMC Water & Environment
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Siemens Water Technologies
Southeast Environmental Engineering, LLC
Stone Environmental Inc.
Stratus Consulting Inc.
Synagro Technologies Inc.
Tata & Howard, Inc.
Tetra Tech Inc.
Tranier Technologies Inc.
Trussell Technologies, Inc.
URS Corporation
Westin Engineering Inc.
Wright Water Engineers
Zoeller Pump Company

INDUSTRY

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The Coca-Cola Company
Dow Chemical Company
DuPont Company
Eastman Chemical Company
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InsinkErator
Johnson & Johnson
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Suez Environment
United Utilities North West (UUNW)
United Water Services LLC
Veolia Water North America

List as of 12/8/10
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<th>Role</th>
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<th>Organization</th>
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<td>Chair</td>
<td>Alan H. Vicory, Jr., P.E., BCEE</td>
<td>Ohio River Valley Water Sanitation Co</td>
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<td>Vice-Chair</td>
<td>William P. Dee, P.E., BCEE</td>
<td>Malcolm Pirnie, Inc.</td>
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<td>Secretary</td>
<td>William J. Bertera</td>
<td>Water Environment Federation</td>
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<td>Jeff Taylor</td>
<td>Freese and Nichols, Inc.</td>
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<tr>
<td>Patricia J. Anderson, P.E.</td>
<td>Florida Department of Health</td>
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<td>Jeanette A. Brown, P.E., BCEE, D.WRE</td>
<td>Stamford Water Pollution Control Authority</td>
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<td>Catherine R. Gerali</td>
<td>Metro Wastewater Reclamation District</td>
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<tr>
<td>Charles N. Haas, Ph.D., BCEEEM</td>
<td>Drexel University</td>
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<tr>
<td>Stephen R. Maguin</td>
<td>Sanitation Districts of Los Angeles County</td>
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<td>Karen L. Pallansch, P.E., BCEE</td>
<td>Alexandria Sanitation Authority</td>
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<td>Robert A. Reich, P.E.</td>
<td>DuPont Company</td>
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<td>R. Rhodes Trussell, Ph.D., P.E.</td>
<td>Trussell Technologies Inc.</td>
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<td>Rebecca F. West</td>
<td>Spartanburg Water</td>
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<td>Brian L. Wheeler</td>
<td>Toho Water Authority</td>
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<td>Joseph E. Zuback</td>
<td>Global Water Advisors, Inc.</td>
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<tr>
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<td>University of California-Irvine</td>
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<td>Robbin W. Finch</td>
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<td>Thomas Granato, Ph.D.</td>
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<td>U.S. Environmental Protection Agency</td>
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<td>James A. Hodges, CPEng.</td>
<td>Watercare Services Limited</td>
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<td>David Jenkins, Ph.D.</td>
<td>University of California at Berkeley</td>
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<td>Terry L. Johnson, Ph.D., P.E., BCEE</td>
<td>Black &amp; Veatch Corporation</td>
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<td>Beverley M. Stinson, Ph.D.</td>
<td>AECOM</td>
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<tr>
<td>Susan J. Sullivan</td>
<td>New England Interstate Water Pollution Control Commission (NEWPCC)</td>
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<td>$20.00</td>
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<td>More than $200.00</td>
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