ABSTRACT


Non-point pollution has received significant attention nationwide with seepage from lagoons one potential source. The research presented here discusses the hydrology and shallow groundwater quality associated with leakage from an old, unlined lagoon located in the Middle Coastal Plain of North Carolina. Ammonium-N concentrations from wells installed between the lagoon and a nearby stream averaged 121 mg/L, with the highest concentrations exceeding 170 mg/L. Mean annual NH$_4$-N concentrations in the stream ranged from 10 to 25 mg/L indicating that the seepage plume was reaching the stream. A water control structure was installed down stream of the lagoon to reduce the hydraulic gradient towards the stream. In addition, the lagoon was closed out in April 2001. The hydraulic gradient has decreased from 0.023 m/m when the lagoon was in production to 0.0026 m/m since closure. Over a 33-month pre- and post- closure period, NH$_4$-N concentrations in wells 15 m down gradient of the lagoon have decreased from 121 mg/L to 96 mg/L.

A series of pumping wells were installed in the seepage plume to remove and route the contaminated groundwater to a 0.35 ha constructed wetland for treatment. Inflow and outflow of the wetland were continuously monitored to determine nutrient loading and reduction rates. Fourteen monthly mass balances were computed to compare the inflow and outflow of the wetland and to assess monthly nutrient reduction for TKN, NH$_4$-N, NO$_3$-N, TP and OP. Overall, greater than 79 % of the nitrogen and 26 % of the phosphorus were assimilated on a mass basis while concentrations decreased by more than 87 % across all nutrient species.
The Use of Constructed Wetlands to Remove Nitrogen and Phosphorus from Pumped Shallow Groundwater

By

Michael J. Cook

A dissertation submitted to the Graduate Faculty of North Carolina State University in partial fulfillment of the requirements for the Degree of Doctor of Philosophy

BIOLOGICAL AND AGRICULTURAL ENGINEERING

Raleigh, North Carolina

2001

APPROVED BY:

________________________  ________________________
Robert O. Evans           Stephen Broome
(Biological and Agricultural Engineering) (Soil Science)
Chair of Advisory Committee Minor Representative

________________________  ________________________
R. Wayne Skaggs            Garry Grabow
(Biological and Agricultural Engineering) (Biological and Agricultural Engineering)
Committee Memeber Committee Memeber
BIOGRAPHY

Michael Cook grew up in Camanche, Iowa where he attended Camanche High School. He competed on the Cross Country and Track team and was Captain of both teams his Senior year. Upon graduation from High School, in 1989, Michael attended Clinton Community College (CCC) to determine a specific field of study. After taking several general education requirements, a Microbiology course sparked his interest. During Michael’s studies at Clinton Community College, Michael worked holidays and summers at Archer Daniel’s Midland Company as an analytical chemist. Following two years of study at CCC, Michael transferred to Iowa State to obtain a Bachelor’s degree in Microbiology.

While at Iowa State University (ISU), Michael obtained skills in microbiology, chemistry and genetics. He completed an internship in a clinical microbiology lab at the veterinary school his senior year. Upon graduation in 1994 with a degree in Microbiology, Michael worked full time at ISU as a research aid and was promoted to a full-time Research Associate with the responsibility of coordinating an Integrated Pest Management (IPM) project between three academic departments. Upon completion of the IPM project, Michael began graduate school at ISU in January 1996 with an interdepartmental major in Water Resources with the home department Agricultural and Biosystems Engineering. Under the direction of Dr. James Baker, Michael evaluated the effects of manure application of surface and groundwater quality. In addition to conducting research, Michael initiated and maintained a Microbiology Laboratory used for the evaluation of bacteria in agricultural drainage for the Agricultural and Biosystems Engineering
department. During this time Michael made several presentations on his research at national and international meetings.

Upon completion of the Master’s degree from ISU, Michael moved to North Carolina where began his Doctoral Degree under the direction of Dr. Robert Evans at North Carolina State University (NCSU). The focus of Michael’s research was the use of constructed wetlands for improving shallow groundwater quality contaminated with swine lagoon seepage. During his work at NCSU, Michael gave a number of presentations at national and international meetings and field days. While at NCSU, Michael married his wife, Mireia, in Barcelona, Spain. After completing his degree Michael plans to either work in the environmental consulting field or continue academic research by taking a post-doctoral position at the University of Barcelona.
ACKNOWLEDGEMENTS

I would like to thank everyone involved in this project, and those who provided support throughout this process. First, I greatly appreciate the generosity and patience of Dr. Robert Evans. His mentoring taught me a great deal with taking a project from the beginning to end, while understanding all of the details along the way. In addition many thanks to Drs. Wayne Skaggs, Stephen Broome, and Garry Grabow for their review of this manuscript and for all of the insight and guidance for this project. Also, thanks go to the research group (Ron Sheffield, Jonathon Smith, L.T. Woodlief, Mike Adcock and Michael Dukes) all of whom contributed greatly to the project, not only from a scientific, but also from a personal level.

I would also like to thank Rachel Huie and the Environmental Water Quality Laboratory in the Biological and Agricultural Engineering Department. Without their hard work I may still be waiting for samples to be analyzed. I still think that you are too efficient! Also I express my appreciation to the farm staff at the study site. From constructing the wetland to mowing the grass around the wetland, you always do a great job.

I would like to give all of my appreciation and thanks to my family. My parents, Jan and Larry, have supported me unconditionally through the whole process. Thanks for instilling in me a work ethic that allowed this work to be completed. Finally, I would like to recognize my wife Mireia, who gave me her full support and believed in me from the beginning to end. Thanks for being so patient these last months as I worked late, and taking care of all of life’s details in the middle. I don’t think that this work could have been completed without your moral support.
# TABLE OF CONTENTS

| List of Tables | viii |
| List of Figures | x |

Chapter I: The Use of Constructed Wetlands for the Amelioration of Elevated Nutrient Concentrations in Shallow Groundwater: Background and Justification.

| Introduction | 1 |
| Lagoon Seepage | 1 |
| Constructed Wetlands | 4 |
| Justification | 7 |
| References | 9 |

Chapter II: The Impact of Lagoon Seepage on Shallow Groundwater Quality: Pre- and Post-Lagoon Closure.

| Abstract | 13 |
| Introduction | 14 |
| Materials and Methods | 15 |
| Site Background | 15 |
| Well Installation | 17 |
| Water Quality Sampling | 18 |
| Hydrologic Monitoring | 19 |
| Results | 20 |
| Lagoon Seepage Rate | 20 |
| Hydrologic Analysis | 21 |
| Water Quality Data | 24 |
| Summary and Conclusions | 26 |
| References | 38 |


| Abstract | 40 |
| Introduction | 41 |
| Materials and Methods | 43 |
| Site Background | 43 |
| Wetland design and Construction | 44 |
| Removal of Contaminated Groundwater | 45 |
### TABLE OF CONTENTS (continued)

<table>
<thead>
<tr>
<th>Section</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow Quantification</td>
<td>45</td>
</tr>
<tr>
<td>Water Quality Sampling</td>
<td>47</td>
</tr>
<tr>
<td>Plant Uptake</td>
<td>48</td>
</tr>
<tr>
<td>Results and Discussion</td>
<td>49</td>
</tr>
<tr>
<td>Concentration Data</td>
<td>49</td>
</tr>
<tr>
<td>Mass Determinations</td>
<td>51</td>
</tr>
<tr>
<td>Summary and Conclusions</td>
<td>54</td>
</tr>
<tr>
<td>References</td>
<td>67</td>
</tr>
<tr>
<td>Abstract</td>
<td>69</td>
</tr>
<tr>
<td>Introduction</td>
<td>70</td>
</tr>
<tr>
<td>Materials and Methods</td>
<td>73</td>
</tr>
<tr>
<td>Site Background</td>
<td>73</td>
</tr>
<tr>
<td>Water Quality Analysis</td>
<td>74</td>
</tr>
<tr>
<td>Dissolved Oxygen</td>
<td>75</td>
</tr>
<tr>
<td>Temperature</td>
<td>76</td>
</tr>
<tr>
<td>Oxidation-Reduction Potential</td>
<td>76</td>
</tr>
<tr>
<td>Statistical Analysis</td>
<td>77</td>
</tr>
<tr>
<td>Results and Discussion</td>
<td>77</td>
</tr>
<tr>
<td>Data Analysis</td>
<td>80</td>
</tr>
<tr>
<td>Statistical Analysis</td>
<td>81</td>
</tr>
<tr>
<td>Summary and Conclusions</td>
<td>84</td>
</tr>
<tr>
<td>References</td>
<td>98</td>
</tr>
<tr>
<td>Chapter V: A Steady-State Hydrologic Evaluation of a Site Contaminated with Aerobic Lagoon Seepage using MODFLOW-GMS.</td>
<td></td>
</tr>
<tr>
<td>Abstract</td>
<td>100</td>
</tr>
<tr>
<td>Introduction</td>
<td>101</td>
</tr>
<tr>
<td>Materials and Methods</td>
<td>106</td>
</tr>
<tr>
<td>Site Description</td>
<td>106</td>
</tr>
<tr>
<td>Water Quality</td>
<td>106</td>
</tr>
<tr>
<td>Hydrology</td>
<td>107</td>
</tr>
<tr>
<td>Hydraulic Conductivity</td>
<td>109</td>
</tr>
<tr>
<td>Modeling</td>
<td>112</td>
</tr>
<tr>
<td>Pumping and Plume Capture</td>
<td>112</td>
</tr>
<tr>
<td>Simulated Lagoon</td>
<td>113</td>
</tr>
<tr>
<td>Statistical Methods</td>
<td>114</td>
</tr>
</tbody>
</table>
# TABLE OF CONTENTS (continued)

<table>
<thead>
<tr>
<th>Section</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>Results</td>
<td>114</td>
</tr>
<tr>
<td>Hydraulic Conductivity</td>
<td>114</td>
</tr>
<tr>
<td>Model Inputs and Calibration</td>
<td>115</td>
</tr>
<tr>
<td>Hydraulic Gradient Evaluation</td>
<td>115</td>
</tr>
<tr>
<td>Plume Capture Scenario</td>
<td>116</td>
</tr>
<tr>
<td>Simulated Lagoon</td>
<td>118</td>
</tr>
<tr>
<td>Summary and Conclusions</td>
<td>119</td>
</tr>
<tr>
<td>References</td>
<td>128</td>
</tr>
<tr>
<td>Appendix A: Manual Water Level Measurements</td>
<td>131</td>
</tr>
<tr>
<td>Appendix B: Continuous Measurements</td>
<td>143</td>
</tr>
<tr>
<td>Appendix C: Concentration Data</td>
<td>151</td>
</tr>
<tr>
<td>Appendix D: Pictures</td>
<td>164</td>
</tr>
</tbody>
</table>
LIST OF TABLES

CHAPTER III

Table 1. Monthly water balance values (m$^3$)……………………………………………….56
Table 2. Overall water balance values (m$^3$)………………………………………………….57
Table 3. Mass balance and nutrient reductions………………………………………………57
Table 4. Mass of nutrients from plant samples applied to the entire wetland area……….57

CHAPTER IV

Table 1. Average water depths at DO sampling locations………………………………….86
Table 2. Monthly nutrient mass attenuated (kg)* for each nutrient………………………..86
Table 3. Monthly nutrient reduction* (%) for each nutrient………………………………86
Table 4. Summary of average monthly measurements…………………………………….87
Table 5. Correlation coefficients from regression analysis for the study period comparing mass monthly nutrient attenuation (kg) to average monthly temporal measurements………………………………………………………….88
Table 6. Correlation coefficients from regression analysis for the growing season comparing mass monthly nutrient retention (%) to average monthly temporal measurements………………………………………………….88
Table 7. Correlation coefficients from regression analysis for the entire year comparing mass monthly nutrient retention (%) to average monthly temporal measurements…………………………………………………………89
Table 8. Student t-test data for temporal effects on treatment with 95% confidence intervals for the slope, for the entire year…………………………………………………………90

CHAPTER V

Table 1. Field saturated hydraulic conductivity from slug test evaluation……………….121
Table 2. Comparison of observed heads vs. simulated heads for MODFLOW calibration………………………………………………………………………………………121
LIST OF TABLES (continued)

Table 3. Calibrated model inputs used in MODFLOW……………………………………122

Table 4. Summary of gradient data with respect to outlet elevation………………………122

Table 5. Pumping rates needed from each well to reverse the gradient from the stream……………………………………………………………………………………………………122
# LIST OF FIGURES

## CHAPTER II

<table>
<thead>
<tr>
<th>Figure</th>
<th>Description</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Locations of groundwater wells and surface water sampling locations</td>
<td>28</td>
</tr>
<tr>
<td>2</td>
<td>Topographic map of pre- (brown) and post- (green) wetland construction</td>
<td>29</td>
</tr>
<tr>
<td>3</td>
<td>Schematic of pumping well installation</td>
<td>30</td>
</tr>
<tr>
<td>4</td>
<td>Relative elevation of free water surface of anaerobic lagoon</td>
<td>30</td>
</tr>
<tr>
<td>5</td>
<td>Manual water level reading for wells M1-M4</td>
<td>31</td>
</tr>
<tr>
<td>6</td>
<td>Water level topographic map pre- (orange) and post- (green)</td>
<td>31</td>
</tr>
<tr>
<td>7</td>
<td>Continuous water level measurements at wells F1 and F2</td>
<td>32</td>
</tr>
<tr>
<td>8</td>
<td>Continuous water level measurements at wells L1-L3</td>
<td>32</td>
</tr>
<tr>
<td>9</td>
<td>Continuous water level measurements at wells D1 and D2</td>
<td>33</td>
</tr>
<tr>
<td>10</td>
<td>Continuous water level measurements at wells D2, F1 and L2 Perpendicular to flow</td>
<td>33</td>
</tr>
<tr>
<td>11</td>
<td>Continuous water level measurements at water control structure</td>
<td>34</td>
</tr>
<tr>
<td>12</td>
<td>Topographic map comparing groundwater flow in high flow (green) and low flow (orange) periods</td>
<td>34</td>
</tr>
<tr>
<td>13</td>
<td>NH₄-N concentrations in wells M1-M5</td>
<td>35</td>
</tr>
<tr>
<td>14</td>
<td>NH₄-N concentrations in transducer wells L1-L3</td>
<td>35</td>
</tr>
<tr>
<td>15</td>
<td>NH₄-N concentrations in transducer wells D1-D2</td>
<td>36</td>
</tr>
<tr>
<td>16</td>
<td>NH₄-N concentrations of wetland inflow</td>
<td>36</td>
</tr>
<tr>
<td>17</td>
<td>Regression line comparing inflow concentration from March 20, 2001 to current</td>
<td>37</td>
</tr>
</tbody>
</table>
# LIST OF FIGURES (continued)

<table>
<thead>
<tr>
<th>CHAPTER III</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>Figure 1. Plan view of research site showing locations of pumping wells</td>
<td>58</td>
</tr>
<tr>
<td>Figure 2. NH$_4$-N concentrations from monitoring wells along the periphery of the closed lagoon</td>
<td>58</td>
</tr>
<tr>
<td>Figure 3. Aerial photograph of the site indicating the location of all monitoring wells</td>
<td>59</td>
</tr>
<tr>
<td>Figure 4. Photograph of bulkhead</td>
<td>59</td>
</tr>
<tr>
<td>Figure 5. Photograph of inflow box with details</td>
<td>60</td>
</tr>
<tr>
<td>Figure 6. Photograph of inflow box with overland flow distribution</td>
<td>60</td>
</tr>
<tr>
<td>Figure 7. Photograph of wetland outlet</td>
<td>61</td>
</tr>
<tr>
<td>Figure 8. Calibration data for inflow weir</td>
<td>61</td>
</tr>
<tr>
<td>Figure 9. Calibration for outflow under weir conditions</td>
<td>62</td>
</tr>
<tr>
<td>Figure 10. Calibration data for outflow: orifice conditions</td>
<td>62</td>
</tr>
<tr>
<td>Figure 11. TKN concentrations at sampling locations in constructed wetland (85% of TKN was in the NH$_4$-N form)</td>
<td>63</td>
</tr>
<tr>
<td>Figure 12. NO$_3$-N concentrations at sampling locations in the constructed wetland</td>
<td>63</td>
</tr>
<tr>
<td>Figure 13. TP concentrations at sampling locations in the constructed wetland (86% of the TP was in the OP form)</td>
<td>64</td>
</tr>
<tr>
<td>Figure 14. CL concentrations at sampling locations in the constructed wetland</td>
<td>64</td>
</tr>
<tr>
<td>Figure 15. Comparison of nutrient concentrations at each sampling location</td>
<td>65</td>
</tr>
<tr>
<td>Figure 16. Constructed wetland stage data</td>
<td>65</td>
</tr>
<tr>
<td>Figure 17. Monthly fluxes (kg) of total nitrogen and phosphorus</td>
<td>66</td>
</tr>
</tbody>
</table>
LIST OF FIGURES (continued)

CHAPTER IV

Figure 1. NH₄-N concentrations from wells proximate to the closed out lagoon………91

Figure 2. Plan view of the research site.................................................................91

Figure 3. Locations of temporal measurements in the constructed wetland..............92

Figure 4. Comparison of daily average air and water temperatures.......................92

Figure 5. Soil temperature (°C) at each sampling location with respect to time........93

Figure 6. Dissolved oxygen concentrations at four sampling locations with respect to time...........................................................................................................93

Figure 7. Oxidation-potential measurements with respect to time..........................94

Figure 8. Comparison of seasonal nutrient mass fluctuations..............................94

Figure 9. Regression analysis comparing monthly mass attenuated as a function of average monthly water temperature (°C).........................................................95

Figure 10. Regression analysis comparing monthly mass attenuated as a function of average monthly redox-potential (mV)......................................................95

Figure 11. Regression analysis comparing monthly mass attenuated as a function of average monthly dissolved oxygen (mg/L)..................................................96

Figure 12. Regression analysis comparing monthly nutrient retention (%) as a function of average monthly water temperature (°C).....................................................96

Figure 13. Regression analysis comparing monthly nutrient retention (%) as a function of average monthly redox-potential (mV)......................................................97

Figure 14. Regression analysis comparing monthly nutrient retention (%) as a function of average monthly dissolved oxygen (mg/L).............................................97

CHAPTER V

Figure 1. Plan view of the research site.................................................................123
**LIST OF FIGURES (continued)**

<table>
<thead>
<tr>
<th>Figure</th>
<th>Description</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>2</td>
<td>Schematic of monitoring well locations</td>
<td>123</td>
</tr>
<tr>
<td>3</td>
<td>NH$_4$-N concentrations of monitoring adjacent the former lagoon (M1-M4)</td>
<td>124</td>
</tr>
<tr>
<td>4</td>
<td>Aerial photograph indicating groundwater flow boundaries</td>
<td>124</td>
</tr>
<tr>
<td>5</td>
<td>Conceptual model base map used in MODFLOW-GMS</td>
<td>125</td>
</tr>
<tr>
<td>6</td>
<td>Average observed water levels from wells at the research site</td>
<td>125</td>
</tr>
<tr>
<td>7</td>
<td>Schematic displaying flux from the simulated lagoon</td>
<td>126</td>
</tr>
<tr>
<td>8</td>
<td>Close-up of the flux from the simulated lagoon</td>
<td>126</td>
</tr>
<tr>
<td>9</td>
<td>Average NH$_4$-N concentrations from wells at the research site</td>
<td>127</td>
</tr>
</tbody>
</table>
CHAPTER I
The Use of Constructed Wetlands to Remove Nitrogen and Phosphorus from Pumped Shallow Groundwater: Background and Justification

Introduction

In the 1990’s, North America, Europe and Asia have experienced a large expansion of the hog industry. In North Carolina alone, the industry quadrupled. This rapid expansion has led to large-scale production (or confinement operations) where the primary method of animal waste handling is anaerobic lagoons. Anaerobic lagoons pose potential non-point source water quality problems due to seepage. Seepage from lagoons may impact human health and threaten aquatic ecosystems. The presence of excessive nitrate-nitrogen (NO$_3$-N) in drinking water is believed to cause methglobemia in infants, more commonly known as blue-baby syndrome. Excessive nutrients in aquatic systems cause eutrophication, which can lead to decreased dissolved oxygen levels and fish kills.

Lagoon Seepage

In general, the literature indicates that properly constructed lagoons with liners pose little threat to the surrounding groundwater quality. However, improperly or poorly constructed lagoons and those lagoons that are situated in coarse-textured soils generally pose a threat to water resources. A number of studies have documented natural sealing of lagoons and concluded that there was no long-term water quality impact. Miller et al. (1985) monitored an unlined earthen storage pond that received liquid beef manure. In that study, the surrounding soil was a coarse sand, with the lagoon constructed below the normal water table. The researchers observed that two weeks following the initiation of
waste deposition, the hydraulic conductivity of the lagoon bottom layer decreased to $10^{-8}$ cm/s, or effectively sealed. Nevertheless, the authors pointed out that there was an initial rapid flush of effluent in the first two weeks of operation. Final results based on chloride (Cl) and nitrate-nitrogen (NO$_3$-N) data suggested that the lagoon was completely sealed after 12 weeks of waste deposition. Other studies documenting natural sealing of lagoons include Rowsell et al. (1985), Barrington and Broughton, (1988), Ritter (1985) and Phillips et al. (1983). Despite observations of natural sealing, Ritter (1983) recommended that unlined animal waste lagoons should not be installed at locations where coarse-textured soils and high water tables are present. This suggestion was also supported by Phillips et al. (1983), who observed that clogging of soil pores did occur, but they concluded that the creation of a no flow boundary was unlikely.

Huffman and Westerman (1995) conducted a study on older (10 to 20 years) unlined anaerobic lagoons in the lower Coastal Plain of North Carolina. Their research conclusions were that 45% of lagoons in the study had small seepage losses (<0.5 kg/day), while the remaining lagoons had moderate to severe seepage losses (1-10 kg/day). In another study, Huffman and Revels (1998) surveyed 34 older (pre-1993: NRCS standards) lagoons in North Carolina to evaluate their influence on shallow groundwater quality. That research found that about two-thirds of the sites involved in the study showed seepage contamination exceeding drinking water standards 38-m down gradient. They also found that, of those sites where surface water was proximate to the lagoon, fifty percent showed nitrogen enrichment attributable to lagoon seepage. Despite these results, the authors concluded that threats to deep ground water quality are minimal due to the geology of the North Carolina Coastal Plain. In this region, a dense clay layer
exists just below the surficial aquifer that forces water to travel laterally and discharge to surface water, as opposed to migrating downward to deeper groundwater or potential drinking water sources. This may be the case for some situations, but where the confining clay layer is non-continuous, downward movement of seepage water may occur. Although the deeper groundwater may not be exposed to the elevated nutrient concentrations of seepage water, surface water remains a concern for contamination.

Korom and Jeppson (1994) monitored groundwater quality near a newly constructed unlined dairy lagoon. In that study, seepage rates were monitored for five years, and were estimated to range from 1.3 to 9.1-cm/day. Additionally, concentrations of the leachate surrounding the lagoon were found to range from 10 to 100 mg/L NO$_3$-N. These results suggested that natural sealing of the lagoon via waste deposition did not occur. Similarly, a study by Westerman et al. (1995) found significant seepage from 3.5 to 5 year old unlined lagoons in sandy soil. Their research detected significant concentrations of NH$_4$-N (75-300 mg/L) and Cl (50-140 mg/l) in shallow subsurface waters down gradient from the lagoon.

From the information outlined above, it can be concluded that unlined lagoons situated in coarse-textured or sandy soils appear to pose a threat to surface and groundwater resources. Many states are now implementing best management practices (BMPs) in order to alleviate pressure on ecosystems derived from nonpoint source pollution (NPS). Considering the successes of constructed wetlands in the assimilation of nutrients, wetlands may provide a viable option for the clean up of sites where groundwater is contaminated with seepage from lagoons.
**Constructed Wetlands**

Constructed wetlands have been designed for the treatment of stormwater (Scheuler, 1992; Bass and Evans, 2000) and wastewater (Sikora et al. 1995; Cronk, 1996; Sievers 1997; Hunt et al. 1999). Wetlands create a unique environment where aerobic and anaerobic conditions exist in the soil matrix. The saturation of wetland soil limits the transport of oxygen, thus creating anaerobic conditions. However, growing plants create aerobic microsites in the soil matrix by translocation of oxygen to roots that allows for a wide range of microbe-dependent biochemical reactions to occur. Nutrient assimilation in constructed wetlands can be attributed to the combination of biological, chemical and physical processes that transform and immobilize contaminants. For example, nitrogen can be transformed to N\(_2\) gas by nitrification and denitrification, which are dependent on aerobic and anaerobic conditions, respectively.

Several reviews have been published relating the mechanisms of nutrient and sediment removal in wetlands. Bowden (1987) discussed the biogeochemistry of nitrogen in freshwater wetlands. That review described in detail the nitrogen cycle, and the effect of varying hydrologic conditions that regulate specific microbial processes like nitrification, denitrification, as well as plant up-take, mineralization and nitrogen fixation. Bowden reported that hydrologic conditions influence nitrogen export in wetlands, suggesting that high flow periods provide relatively little nitrogen removal. For example, Chescheir et al. (1991) evaluated the efficacy of two forested wetlands in the removal of sediments and nutrients from pumped agricultural drainage. Their study determined that higher pumping rates were less effective in reducing nutrients and sediments than lower pumping rates. Maehlum and Staelnackle (1999) observed similar results, reporting that
lower pumping rates provided an increased treatment or residence time, and improved nutrient reduction. The treatment of nitrogen in wetlands is also influenced by a number of physical properties such as temperature (Anderson, 1977; Cerco, 1989; Hill and Payton, 1998). Research by Sikora et al. (1995) showed that NH$_4$-N reductions during the growing season were higher in wetland cells with plants compared to wetland cells without plants. That research also showed that in the winter months, when plant growth was minimized, wetland cells with plants performed similarly to those cells without plants. Soil type can also be a factor in nutrient removal. Gale et al. (1993) reported that natural wetland soils reduced NO$_3$-N more effectively than constructed wetland soils. In addition, proper sizing (Scheuler, 1992; Reed et al. 1989) and position in the landscape (Johnson et al. 1990; van der Valk and Jolly 1992) contribute to the effectiveness of nutrient treatment.

Research conducted by Reddy et al. (1989) found that in wetlands, the two-step process of nitrification-denitrification occurs in the plant root-sediment interface. Their study was conducted in a growth chamber and determined that oxygen uptake by the plant produced a diffuse boundary around the roots where NH$_4$-N was converted to NO$_3$-N. The NO$_3$-N was then determined to be lost to denitrification after diffusing from the root zone to the soil matrix. Adsorption is another mechanism that occurs in wetland systems. Nutrients, particularly phosphorus, are bound to the soil matrix due to the cation exchange capacity of the soil. This process, although very effective in binding phosphorus, is finite, as soil-phosphorus binding sites are limited. Bass and Evans (2000) used an in-stream wetland to improve water quality and found that phosphorus levels increased by 55% over the two-year study period. Further nutrient removal in wetlands is
observed by plant up-take. However, this method of removal is often considered a non-
dominant process and must involve plant harvesting to completely remove the nutrients
since plant senescence would lead to nutrient export after decay of plant materials.

From the studies described above, the use of wetlands for the assimilation of
nutrients derived from wastewater has become an area of intense research. The
aforementioned problems associated with animal production systems have stimulated the
development of new technologies with minimal maintenance and cost. Constructed
wetlands may provide an answer to the problems faced by producers, as wetlands provide
treatment benefits by assimilating nutrients. Hammer et al. (1994) discussed the design of
constructed wetlands for the removal of nitrogen. They recommended that relatively low
loading rates (<3kg/ha/d for TKN) are required for wetland treatment systems to produce
low NH$_4$-N effluent levels (< 2-mg/L NH$_4$-N). In addition, their study reviewed the
results of 52 constructed wastewater wetlands and reported that total nitrogen was
reduced on average by 63%, while results for phosphorus removal ranged from −33 to
9%. Similarly, Knight et al. (2000) summarized 1300 records of wetlands receiving
livestock wastewater ranging from pilot to full scale. The study reported average
concentration reductions of 48% for NH$_4$-N, 42% for total nitrogen (TN), and 42% for
total phosphorus (TP). Other studies have observed similar nutrient reductions (Hunt et
al. 1995; Sievers 1997; and Newman and Clausen 1997).

Further investigations have evaluated wetlands for other types of waste treatment.
Kemp et al. (1997) studied subsurface flow constructed wetlands for the treatment of
municipal wastewater. The wetland treatment system received wastewater from a septic
system and found a 71% reduction in NH$_4$-N concentrations when comparing inflow to
outflow. Similarly, Coleman et al. (2001) evaluated the treatment of domestic wastewater. In addition to municipal wastewater, Vrhovsek et al. (1997) studied a constructed wetland that received industrial wastewater. Their analysis showed that the constructed wetland decreased chemical oxygen demand (COD) and biological oxygen demand (BOD) concentrations by 91% with reductions of 96% and 86% in OP and NH$_4$-N concentrations, respectively. Based on this information, constructed wetlands are an effective method of transforming and assimilating nutrients associated with nonpoint source pollution and wastewater from both human and animal sources.

**Justification**

North Carolina has recently ranked among the top in the nation for swine production, with a large portion of these farms located in the Coastal Plain. The Coastal Plain is characterized by a flat landscape, coarse textured soils and channelized streams that provide drainage for agricultural fields. Anaerobic lagoons are the primary method of waste storage in this region, with a great number of the lagoons without liners. Huffman (1999) estimated that one-third of the older lagoons in North Carolina do not meet the 10-mg/L standard for NO$_3$-N at a distance of 250-ft down gradient. Regulations have been adopted to reduce annual nitrogen loads to the Neuse and Tar-Pamlico Rivers, potentially targeting older lagoons for closure. Considering the large number of unlined lagoons and the potential nutrient loads that are lost due to seepage, clean-up procedures for older, unlined lagoons need to be explored.

The objectives of the study were to:

- Characterize the hydraulic gradients and subsurface water quality pre- and post-lagoon closure.
• Conduct a mass balance on a constructed wetland loaded with nutrients derived from anaerobic swine lagoon seepage.

• Analyze water quality improvements in the constructed wetland by comparing monthly nutrient retention as a function of average monthly redox, dissolved oxygen and water temperature measurements.

• Use MODFLOW-GMS to estimate the effect of water level in a nearby steam on seepage plume migration.

The results are presented in paper format. The first chapter provides a literature review and justification of the research. Chapter II presents a description and evaluation of the groundwater quality at the site. The site was characterized before and after lagoon closure and wetland construction, observing trends in water quality and water table data. Chapter III presents a water and nutrient mass balance for the constructed wetland with a quantification of the nutrients assimilated by the wetland. Chapter IV evaluates the effects of temporal variations (water temperature, dissolved oxygen, and oxidation-reduction potential) on monthly nutrient reduction. Finally, Chapter V evaluates the site from a hydraulic perspective using MODFLOW-GMS to estimate pumping rates required to collect the seepage plume as a function of the water level in the adjacent stream.
REFERENCES


Ammonium removal in constructed wetlands with recirculating subsurface flow:

for the use of wetlands to control rural non-point source pollution. Created and
natural wetlands for controlling nonpoint source pollution. *Ecological
Engineering*. 1:115-134.

Vrhovsek, D., V. Kukanja and T. Bulc. 1996. Constructed wetlands for industrial

Oxidation-reduction, air and water temperature, pH, and dissolved oxygen were measured weekly at several locations within the constructed wetland. Regression analyses were conducted to examine the relationship between these parameters and monthly nutrient reductions. Nutrient export from the wetland was positively correlated to water temperature (i.e., nutrient export increased as water temperature increased). In general, lower redox and DO were correlated to higher nutrient levels within the wetland and subsequently to higher export from the wetland. Caution should be taken on interpretation of these regression analyses as the conditions in the wetland changed over the course of the study. The loading rate was doubled at the beginning of the first full growing season (i.e., the loading rate was two times higher during the growing season than the previous dormant season). Another factor that likely impacted N and P assimilation is many plants in the upper portion of the wetland died in July and August of 2001.

The hydrology of the site was evaluated using MODFLOW-GMS. MODFLOW was calibrated using water level data from the site and was used to evaluate the influence of water levels in the adjacent channelized stream on the movement of contaminants in the groundwater plume. Model results indicated a decrease in the hydraulic gradient from the former lagoon from 0.0045 m/m in the free drainage case to 0.0027 m/m in the controlled drainage case. From the gradient calculations, travel time of the seepage plume to the stream increased from 380 days free drainage scenario to 640 days in the controlled drainage case. MODFLOW analysis of the pumping wells indicated that 6.3 gpm was required in the free drainage mode to reverse the gradient from the stream and capture the seepage plume. In the controlled drainage mode, 4.7 gpm was required to reverse the
direction of groundwater flow. A hydrologic analysis was also conducted to evaluate pumping requirements to mitigate an actively leaking lagoon. Simulations were performed using an interceptor drain (French drain) adjacent to the lagoon for collection of seepage discharge from the lagoon. Under controlled drainage, the interceptor drain collected 51.5 m$^3$/day (9.4 gpm) while under free drainage 67.4 m$^3$/day (12.4 gpm) would be collected. This analysis assumed a worst-case scenario where all wastewater deposited in the lagoon was lost to seepage.

The research presented here provides strategies for clean-up of leaking lagoons or those lagoons targeted for closure. Overall, the wetland assimilated 383 kg of total nitrogen and 60 kg of total phosphorus during the 14 month study period. The wetland surface area was originally based on a pumping rate of 1.5 gpm which was the estimated seepage rate of the lagoon. In March 2001, the pumping rate was increased to 3.5 gpm to match growing season ET rates. Initial results indicated that the wetland assimilated this increased flux of nutrients; but as loading continued at this rate, nutrient concentrations in outflow began to increase, suggesting that this higher rate exceeded the assimilative capacity of the wetland. The MODFLOW analysis indicated that a pumping rate of 4.7 to 6.3 gpm was required to reverse the groundwater gradient from the stream. These pumping rates would require a wetland surface area three to four times larger than the wetland area used.

The mass balance indicated that phosphorus mass exports were higher than imports for the final two months. From this, further study is needed to determine processes that are affecting removal of phosphorus. Plant uptake was responsible for only 9% (35 kg) of the N and 18% (11 kg) of the P assimilated. Oxidation-reduction
reactions were judged to be the dominant nitrogen reduction mechanism. Considering the importance of plants in providing conditions conducive for nutrient assimilation, further investigations are needed to determine the cause of the plant die-off. The influence of temperature, dissolved oxygen and redox on nutrient assimilation should be further documented under conditions of constant loading.
CHAPTER II

The Impact of Lagoon Seepage on Shallow Groundwater Quality: Pre- and Post-Lagoon Closure

ABSTRACT

The structural integrity of a swine lagoon located in the Middle Coastal Plain of North Carolina, has been questioned. Research by Huffman and Revels (1998) characterized the site and detected elevated NH$_4$-N concentrations 38-m down gradient from the lagoon. The research presented here discusses hydrologic and water quality monitoring of the site from February 1999 to October 2001, a period during which the lagoon went from being semi-active to completely closed out. Analysis of data collected prior to lagoon closure indicated that fluxes from the lagoon traveled to a nearby, channelized stream. NH$_4$-N concentrations from monitoring wells installed between the lagoon and the stream averaged 121 mg/L of NH$_4$-N, with the highest concentrations averaging 172 mg/L. NH$_4$-N concentrations in the stream 150 m down gradient from the lagoon ranged from 10 to 25 mg/L. Monitoring of water levels and subsurface water quality continued following the closure of the lagoon, as well as through the installation of a groundwater pumping and wetland system. Since the closure of the lagoon, the hydraulic gradient has decreased from 0.023-m/m, when the lagoon was semi-active, to 0.0026-m/m. Since the installation of the pumping system, 4600-m$^3$ of subsurface water has been pumped, removing a total of 430-kg of NH$_4$-N. Samples from monitoring wells showed that NH$_4$-N concentrations have decreased from 121 to 96 mg/L.
INTRODUCTION

Over the past decade, North America, Europe, and Asia have experienced a large expansion in the hog industry, in North Carolina alone the industry quadrupled. As the industry has grown, large-scale production or confinement operations have dominated the industry with anaerobic lagoons the primary method of waste handling. Although generally safe, anaerobic lagoons do pose potential non-point source water quality problems derived from seepage. Seepage from lagoons presents an environmental threat to surrounding groundwater and surface water resources, which may impact human health and threaten aquatic ecosystems. The impact of non-point source pollution in fresh water systems can cause fish kills, eutrophication (Paerl, 1988), and the contamination of drinking water sources.

The literature indicates that improperly or poorly constructed lagoons and those lagoons that are situated in a coarse-textured soil pose threats to water resources, although some studies (Miller et al., 1985; Rowsell et al. 1985; Barrington and Broughton, 1988; and Ritter, 1983) suggested that natural sealing processes eliminate most lagoon seepage. Huffman and Westerman (1995) conducted a study on older (10 to 20 years) unlined anaerobic lagoons in the lower Coastal Plain of North Carolina to determine seepage losses from established lagoons. Their research concluded that 45% of lagoons in the study had small seepage losses (<0.5 kg/day), whereas the remaining lagoons had moderate to severe seepage losses (1-10 kg/day). In another study, Huffman and Revels (1998) surveyed 34 older (pre-1993: NRCS standards) lagoons in North Carolina for their influence on shallow groundwater quality. That research found that about two-thirds of the sites had seepage contamination exceeding drinking water
standards 38-m down gradient. In addition, nitrogen enrichment in surface water was
found in 50% of the sites where surface water was proximate to the lagoon.

Korom and Jeppson (1994) monitored groundwater quality near to a newly
constructed unlined dairy lagoon for five years. In that study, seepage rates were
estimated to range from 1.3 to 9.1 cm/day and concentrations of the leachate surrounding
the lagoon were found to range from 10 to 100 mg/L NO$_3$-N. The data from that study
suggested that natural sealing of the lagoon via waste deposition did not occur. Similarly,
a study by Westerman et al. (1995) found significant seepage from 3.5 to 5 year old
unlined lagoons in sandy soil and found significant concentrations of NH$_4$-N (75-300
mg/L) and Cl (50-140 mg/l) in shallow subsurface waters down gradient.

There is clear evidence that some unlined lagoons situated in coarse-textured or
sandy soils pose a threat to surface and groundwater resources. The study described here
evaluates shallow groundwater quality at a site that has undergone significant changes,
including the closure of the lagoon and installation of a shallow-groundwater removal
system. The data were evaluated for two distinct periods; the first, February 1999 to
March 2000, represents a period when the lagoon was still active. The second period,
April 2000 to present, includes the initiation of groundwater pumping and lagoon closure.

MATERIALS AND METHODS

Site Background

The unlined lagoon, located in the Middle Coastal Plain of North Carolina was
constructed in the 1960’s in a sandy soil where the seasonal high water table was within
1-m of the ground surface. It was learned that pumping of the lagoon was rarely a
necessity; even after heavy rains, the lagoon effluent level would stabilize in a few days.
This site was initially evaluated by Huffman and Revels (1998) who found elevated NH$_4$-N concentrations in shallow groundwater 38-m down gradient of the lagoon. Due to lack of pumping, it was suspected that the lagoon was leaking and surface water quality sampling of the channelized stream adjacent to the lagoon was initiated in 1997. The data from bi-weekly surface water samples indicated that ammonium-nitrogen (NH$_4$-N) concentrations ranged from detection levels (<0.01 mg/L) upstream of the lagoon to 10 to 25-mg/L immediately downstream from the lagoon. Monitoring wells were installed in February 1999 to characterize the location of the seepage plume and to determine groundwater flow paths. An aerial photograph of the research site, including surface water sampling sites and all well locations, is displayed in Figure 1.

The construction of an artificial wetland was completed in March 2000, with the installation of a shallow groundwater pumping system ending in May 2000. The groundwater pumping-constructed wetland system was used to collect and assimilate lagoon seepage water, to reduce the offsite impacts associated with the seepage plume. Figure 2 displays both a pre- and post-construction scaled topographic map; green lines indicate post-construction contour lines, while brown lines display pre-construction contours. The general outline of the constructed wetland can be observed, as well as contours associated with the filling of the lagoon. In April 2000, hog production ceased and waste deposition ended, with a plan developed to close out the lagoon. In October 2000, a water control structure was installed on the channelized stream down gradient of the lagoon and suspected seepage plume (Figure 1). The water control structure raised the water level in the stream, which lowered the hydraulic gradient from the lagoon to the stream. Procedures to close the lagoon were completed in March of 2001. The close out
process included removal and land application of nutrient rich sludge that previously occupied the lagoon floor, as well as back-filling the lagoon with soil.

Well Installation

Groundwater monitoring wells were installed around the periphery and down gradient of the lagoon and along the channelized stream in February of 1999 (wells M1-M11 in Figure 1). The wells were used to locate the seepage plume and to determine hydraulic gradient. Using a hand auger, the 50-mm wells were initially installed to a depth of 1.5-m, and screened from 1 to 1.5-m. Wells had 5-mm holes drilled for the screened portion and were covered with a fabric well sock. Sand was backfilled around the screened portion followed by a short section of bentonite to prevent preferential flow from entering the well. The upper section of the well was backfilled with sand. These wells failed to intercept the seepage plume, as only low levels of NH$_4$-N were found. Therefore, deeper wells were installed at the same locations to a depth of 3-m, screened from 2.4 to 3-m. Due to the coarse sand at the 1.2 to 1.8-m depth and the high water tables, deep wells had to be installed using a pressurized jet of water, which was used to flush down a solid 100-mm PVC casing. Once a depth of 3-m was reached with the casing, a 50-mm PVC monitoring well was inserted and the casing was removed. Typically, native sand filled around the screened portion of the well, but if necessary, backfill sand was used. A 15-cm layer of bentonite was used as backfill above the screened portion of the well to prevent preferential flow and the remaining well length was backfilled with sand.

In May 2000, five pumping wells were installed along the periphery of the lagoon, centered along a transect of highest detected groundwater NH$_4$-N concentrations.
Pumping wells were installed to a depth of 6-m (depth of confining clay layer) using a drill rig and hollow stem auger. These wells were 25-mm in diameter and installed inside 50-mm casings, which were screened 3 to 6-m using 0.4-mm slotted PVC. The screened portion of the casing was back-filled with sand followed by a 1.5-m section of bentonite. The last 1.5-m portion of the pumping well was back-filled with sand and covered with a 0.9-m by 0.9-m concrete slab that was 0.15-m thick. Figure 3 displays a cross-sectional schematic of a pumping well. Water table monitoring wells were installed at the site in October 2000 (D1-DW, F1- F3, L1-L3, and P1 in Figure 1). Water levels were monitored continuously using transducers and sampled bi-weekly to provide additional water quality data. These wells were made of 76-mm diameter schedule 40 PVC and installed to a depth of 3-m, and each well was screened from 1.5 to 3-m using 0.4-mm well screen. The screened section of the well was fitted with a well sock fabric to prevent coarse soil particles from plugging the well screen. Wells were installed using a hollow stem auger and drill rig and wells were backfilled with 1.5-m sand, followed by 0.3-m of bentonite, with the remainder of the well backfilled with sand.

**Water Quality Sampling**

All wells were sampled monthly, although bi-weekly samples were collected during some periods. Prior to sampling, wells were purged for approximately 3-min, or the equivalent of three well volumes using a peristaltic pump. Following adequate well purging, samples were collected and stored on ice until transported to the Biological and Agricultural Engineering Environmental Analysis Laboratory located at North Carolina State University. Following arrival at the laboratory, samples were transferred and stored at 4° C until analyzed. Samples were analyzed for total Kjeldahl nitrogen (TKN), nitrate-
nitrogen (NO₃-N), ammonia nitrogen (NH₃-N), total phosphorus (TP), orthophosphate phosphorus (OP), and chloride (CL). TKN and NH₃-N was measured using automated analysis. TKN was quantified using persulfate digestion followed by ammonia-salicylate method for NH₃-N analysis. NO₃-N was determined by the automated cadmium reduction method. TP was analyzed using the persulfate digestion and ascorbic acid method, while OP was quantified using ascorbic acid method. Chloride determination was made using the ferricyanide method for automated analysis. All analyses were completed using automated protocols as approved by The Environmental Protection Agency (The Standard Methods for the examination of Water and Wastewater 18th edition (1992).

**Hydrologic Monitoring**

The lagoon in question was constructed without a liner in sandy soil, by excavating to a finished depth of 2-m. The finished surface area was 3720-m², with a 90-day storage of 1870-m³. An alternative waste treatment system was installed in 1996, and except for by-pass periods, the lagoon was inactive and no longer receiving manure. However, there was normally a free water surface in the lagoon, which remained a potential for nutrient export via seepage as the lagoon contained nutrient rich sludge. A water level recorder was installed in the lagoon in 1999 and a water balance was conducted, considering rainfall and Potential Evapotranspiration (PET). PET was estimated using the Thornwaite method based on mean monthly temperature (Thornwaite, 1957). The monthly temperature values were adjusted using a correction factor based on the Penman method for estimating PET (Amatya et. al., 1995). Since the lagoon was not being loaded, inflow was not considered. The lagoon seepage rate was
estimated as the difference between rainfall and PET. In addition, lagoon water level was also used to compute gradient to the adjacent channelized stream.

Water tables measurements were taken from all wells on a bi-weekly basis, whether or not a sample was taken. An electronic slope instrument was used to measure the depth of the water level with an accuracy of 2.5-mm (0.01-ft). Continuous water level measurements were made in wells D1-DW, L1-LW, F1-F3 and P1 using transducers (Figure 1). A weatherproof box was installed on each transducer well to protect the data logger, transducer, and battery power source from adverse weather. A Blue Earth® data logger was used to record and store raw data as well as to convert hourly raw voltage readings to water table depth. The conversion of raw voltage reading to water table depth was completed using standard equations developed from laboratory calibrations. Data loggers were downloaded using a palmtop computer on bi-weekly intervals. At the time of download, manual water level measurements were made to correct any errors associated with stray transducer readings and provided back-up water levels in the event of data logger failure. A continuous recording, tipping bucket rain gauge was installed near the lagoon. A HOBO® event loggers recorded rainfall in 0.25-mm (0.01-in) increments. The gauges were downloaded on a monthly basis or more frequently if high rainfall amounts occurred.

RESULTS

Lagoon Seepage Rate

Figure 4 displays continuous measurements of the lagoon water level. Initially, liquid levels in the lagoon were relatively constant from June until September 1999, when three hurricanes produced 42.9-cm (16.92-in) of rainfall at the site. Although the lagoon
did not overtop, liquid levels were high. A lagoon water balance was conducted from June 16, 1999 to June 30, 1999 when water level was low in the lagoon, and from October 25, 1999 to November 2, 1999 when lagoon levels were high following three hurricanes. The water balance considered potential evapotranspiration (PET) and change in lagoon storage. Since the lagoon was no longer receiving wastewater from the production facility, inflow was not a factor. In addition, rainfall was not a consideration for the water balance, as there were no events during these two periods of evaluation.

Water balance data from the low stage of the lagoon (June 16, 1999 to June 30, 1999) indicated that there was no seepage discharge. During this 14-day period, change in liquid level was due to ET, as the lagoon level decreased 3.7-cm, while PET for that time period was 5.2-cm. Data from the 8-day period (October 25, 1999 to November 2, 1999) when the lagoon levels were high showed that the stage decreased 11.6-cm, of which 1-cm was accounted for by PET. Based on the surface area of the lagoon and change in stage, it was determined that the lagoon leaked at a rate of 1.3-cm/day.

**Hydrologic Analysis**

A hydraulic analysis was conducted to evaluate the site pre and post-lagoon closure to determine the flow path of the seepage fluxes from the lagoon. Pre-closure analysis was based on manual water level measurements from the wells along the periphery of the lagoon, lagoon stage, and the channelized stream bottom. The high stages associated with the flooding caused by the hurricanes in late September and October 1999 were omitted from gradient determinations. Figure 5 displays manual water level measurements for wells M1-M4. The water levels were similar in all wells at low stage. At high stage, M1-M3 was similar, while M4 was lower than the other three.
The lower water level at M4 was due to the change in direction of the channelized stream allowing flow from M4 to the stream in two directions, in effect increasing the drainage rate at M4. The water table between M1-M3 was relatively flat.

The average hydraulic head for all wells pre- and post-construction is shown in Figure 6. The average water level from all wells (blue dots) was used to generate post-construction (orange) and pre-construction (green) contours. Under pre-construction conditions, the general direction of groundwater flow was from the lagoon to the channelized stream. Applying the average lagoon stage prior to the hurricanes (29.79-m or 97.7-ft), the ditch bottom elevation (28.66-m or 94.0-ft), and the distance to the channelized stream from the lagoon (47.3-m or 155-ft), the hydraulic gradient was determined to be 0.023-m/m. The gradient may have been larger when the lagoon was in full operation, however there were no measurements at that time.

After closure of the lagoon, the hydraulic gradient and flow direction was determined using continuous water level data from transducer wells. The water levels were similar in F1 and F2, displayed in Figure 7, indicating that the hydraulic gradient between these two wells was small. There was a slight gradient from L1 to L3 under high water table conditions, Figure 8, although under low water level conditions, the water table was flat between these wells indicating little gradient in the transect between L1 to L3. Figure 9 shows that the water table was relatively flat between wells D1 and D2, indicating that the flow was primarily perpendicular to the ditch in this region. The flow direction was similar before and after the lagoon was closed, with subsurface flow moving from the closed lagoon to the stream. The gradient was determined from average water levels from wells L1-L3 (29.54-m or 96.90-ft) down gradient of the lagoon,
average water level in the stream as a result of the water control structure (29.42-m or 96.5-ft), and the distance of wells L1-L3 from the channelized stream (155-ft, 47.3-m). The post-construction hydraulic gradient to the channelized stream was determined to be 0.0026-m/m. Figure 10 displays data from continuous water level recorders moving from the stream up gradient to F2. The data in this Figure indicates a small hydraulic gradient from the area just up gradient from the former lagoon to the stream. Water levels in wells M1-M4, installed between the former lagoon and channelized stream, indicated a drop in average water level from 29.57-m (96.98-ft) to 29.43-m (96.52-ft) when comparing pre- and post-construction measurements.

The water control structure was installed on the channelized stream in October 2000 to reduce the gradient to the stream, which should also lower seepage to the stream. The stream water level often dropped gradually for a period of days, followed by a rapid increase in the water level, Figure 11. This can be explained by normal hydrological events since evaporation from the ditch would account for the gradual decline in water level, while rainfall would explain the rapid increase in stream level. The sharp drop in stage in August 2001 was due to the removal of boards in the structure for field operations. At that time the water level in the field was too high for repairs needed on wells for an unrelated study. Additional analysis of the post-construction water levels was completed to determine flow direction under high and low flow conditions. Figure 12 displays a contour map developed from maximum and minimum observed water levels. It can be observed that post-construction flow direction did not change and that flow direction was not affected by water table depth.
Data from the deep groundwater wells was used to locate the seepage plume from the lagoon. Graphs of all shallow groundwater quality are presented in Appendix C. The initial analysis for the period March 1999 to March 2000 included wells M1-M11. Concentration data from wells M1-M5 is shown in Figure 13. Average NH$_4$-N concentration in wells M1-M4 was 121 mg/L, with M2 and M3 having an average concentration of 172 mg/L. The average concentration for well M5 was 34.5 mg/L, and was considered to be near the edge of the seepage plume, although the concentrations were much higher in wells M1-M4. Another indicator that M5 represented the edge of the plume was that NH$_4$-N concentrations were near zero during some periods. The variations in concentrations of M5 might also reflect minor fluctuations in the direction of the groundwater flow related to stage. Wells installed further down gradient along the channelized stream detected low concentrations of NH$_4$-N, averaging 3.5 mg/L. These relatively low concentrations also indicate minor seepage, with the majority of the seepage located between the lagoon and channelized stream.

The process to close the lagoon began in March 2001 and in June 2000 pumping wells began withdrawal of the contaminated groundwater. To evaluate the impact of these changes, additional wells were installed up gradient of lagoon, and around the constructed wetland. Water quality data from wells 12-m up gradient from the lagoon (F1 and F2 from Figure 1) indicated that NH$_4$-N concentrations were negligible, however NO$_3$-N concentrations ranged from 6-21 mg/L. NO$_3$-N concentrations can be explained by years of wastewater and commercial fertilizer application up gradient. The nutrient concentrations for L1-L3 located at the toe of the former lagoon are displayed in Figure
14. Well L3 was located in the seepage plume as average NH$_4$-N concentrations were 81-mg/L. NH$_4$-N concentrations remained constant through lagoon closure, but have steadily declined since lagoon was closed. Wells L1 and L2 detected low concentrations following well installation, suggesting that either they were not located in the areal extent of the plume or that the plume was deeper than the screened depth of the well. The increased NH$_4$-N concentrations for wells L1 and L2 in April and May 2001 were likely related to the lagoon closure procedures which involved mechanical mixing and removal of sludge in the lagoon. Water was pumped into the lagoon to produce a slurry so that the sludge could be land applied. The methods of sludge removal were physical in nature, involving collapse of the lagoon sidewalls, which could rupture any natural sealing resulting in additional leakage of nutrients. Any sealing of the lagoon bottom by the sludge would be broken, yielding higher nutrient concentrations in subsurface drainage. Concentrations in L1 and L2 returned to pre-closure levels soon after the procedure was completed.

Continuing in the direction of flow from the former lagoon perpendicular toward the channelized stream, average concentrations for wells M1-M5 prior to close out and pumping were 121 mg/L NH$_4$-N, while post-closure and pumping had averaged 96 mg/L NH$_4$-N. Wells M2-M3 decreased from 172 to 123 mg/L NH$_4$-N, while well M5 concentration decreased from 35 to 17 mg/L NH$_4$-N. Wells D1 and D2 located 15-m from the channelized stream and down gradient from M1-M4, had average NH$_4$-N concentrations of 93-mg/L and 76-mg/L, respectively (Figure 15). NH$_4$-N concentrations were relatively constant at D1, while concentrations at D2 dropped significantly after lagoon closure. The remainder of monitoring wells downstream between the wetland and
the stream averaged 2.7 mg/L, while transducer wells installed along the steam, adjacent to the wetland followed similar trends to the monitoring wells (M6-M11) averaging 3.8 mg/L of NH$_4$-N.

Concentration data from daily (composited weekly) water quality samples taken at the wetland inflow are shown in Figure 16. Inflow NH$_4$-N concentrations have decreased from 123 mg/L in January 2001, to 75.1 mg/L by the middle of October 2001. Regression analysis of concentration versus date indicated a linear trend ($r^2 = 0.95$) as seen in Figure 17. Assuming that concentrations continue on this trend, it was estimated that 480-days will be necessary to decrease NH$_4$-N concentrations below 1-mg/L.

In general, NH$_4$-N concentrations in subsurface water throughout the site have decreased since March 2001. NH$_4$-N concentrations in the channelized stream have decreased from 10 to 25 mg/L pre-closure to 3.5-mg/L since the lagoon was closed.

**SUMMARY AND CONCLUSIONS**

A water balance was conducted on a lagoon and indicated that under low stage conditions the seepage rates were negligible. However, under high lagoon stages (created by large precipitation events), the lagoon lost 420m$^3$ over an 8-day period, which equated to a seepage rate of 1.3-cm/day. By evaluating water level measurements and water quality samples, it was determined that seepage water traveled from the lagoon directly to the channelized stream. The hydraulic gradient between the lagoon and stream was determined to be 0.023-m/m. NH$_4$-N concentrations in monitoring wells located along the periphery of the lagoon ranged from 34.7 to 172 mg/L, with concentrations directly down gradient from the lagoon averaging 121 mg/L. The closure of the lagoon, initiated in March 2001, and the installation of a water control structure on the channelized stream
decreased the hydraulic gradient towards the stream from 0.023-m/m to 0.0026-m/m. Since the lagoon closure, nutrient concentrations in monitoring wells adjacent the lagoon ranged from 16.7 to 123 mg/L NH$_4$-N, with concentrations directly down gradient averaging 96 mg/L NH$_4$-N. Surface water samples from the channelized stream post-closure averaged 3.5 mg/L NH$_4$-N.

A comparison of the NH$_4$-N data between the two time periods (pre- and post-lagoon closure) indicated that the levels of NH$_4$-N decreased over time. The decrease in NH$_4$-N concentration in the subsurface water was the result of lagoon closure and the pumping system used to remove the contaminated groundwater. Since the initiation of this system, 4600-m$^3$ of subsurface water have been pumped which removed of 430-kg of NH$_4$-N. Although the current data strongly indicated that nutrient concentrations are decreasing over time, several years of pumping may be required to mitigate the long-term seepage losses from the lagoon. Assuming that the NH$_4$-N concentration continues to decline at the current linear rate, regression analysis ($r^2=0.95$) predicted that 480-days (1.3 years) of pumping will be required to lower the concentrations below 1-mg/L.
Figure 1. Locations of groundwater wells and surface water sampling locations.
Figure 2. Topographic map of pre- (brown) and post- (green) wetland construction.
Figure 3. Schematic of pumping well installation.

Figure 4. Relative elevation of the free water surface of the anaerobic lagoon.
Figure 5. Manual water level reading for wells M1-M4.

Figure 6. Water level topographic map pre- (orange) and post-wetland construction (green).
Figure 7. Continuous water levels measurements at wells F1 and F2.

Figure 8. Continuous water levels measurements at wells L1-L3.
Figure 9. Continuous water levels measurements at wells D1 and D2.

Figure 10. Continuous water levels measurements from wells D2, F1, and L1 perpendicular to flow.
Figure 11. Continuous water levels measurements at the water control structure.

Figure 12. Topographic map comparing groundwater in high flow (green) and low flow (orange) periods.
Figure 13. NH$_4$-N concentrations at M1-M5.

Figure 14. NH$_4$-N concentrations in transducer wells L1-L3.
Figure 15. NH₄-N concentrations in transducer wells D1-D2.

Figure 16. NH₄-N concentrations of wetland inflow.
Figure 17. Regression line comparing inflow concentration to date since March 20, 2001.

\[ y = -0.2324x + 112.42 \]

\[ R^2 = 0.9528 \]
REFERENCES


CHAPTER III

The Use of a Constructed Wetland for the Amelioration of Elevated Nutrient Concentrations in Shallow Groundwater

ABSTRACT

Seepage from improperly constructed anaerobic lagoons and manure storage basins pose a threat to water quality, via nutrient transport to the surrounding water resources. Nutrient concentrations in the shallow groundwater beneath a recently closed-out lagoon in the Middle Coastal Plain of North Carolina were observed. Considering the recent successes of constructed wetlands in assimilating nutrients from various sources, a wetland system was created in order to ameliorate nutrient contamination in the shallow groundwater. A series of pumping wells were installed to a depth of 6-m (screened 3 to 6-m) and used to deliver the nutrient rich groundwater to the constructed wetland for treatment. Inflow and outflow of the wetland were continuously monitored to determine both nutrient loading and reduction rates. Samples of flow were collected daily and composited weekly for water quality analysis. A mass balance for nutrients was completed by combining the flow and concentrations data. In addition to flow data, measurements of oxidation-reduction, air and water temperature, and dissolved oxygen were collected at bi-weekly intervals at various locations within the constructed wetland to document ongoing wetland processes. Overall, the wetland assimilated 383 kg of total nitrogen (79% of inflow) and 60 kg of total phosphorus (27% of inflow) on a mass basis while concentration dropped by an average of 87% across all nutrient species. Plant uptake was responsible for only 35 kg (9%) of the total nitrogen and 11 kg (18%) of the
total phosphorus assimilated. Oxidation-reduction reactions judged to be the dominant nutrient reduction mechanism.

**INTRODUCTION**

In the last decade North America, Europe and Asia have observed a large expansion in the hog industry. In North Carolina alone, the industry quadrupled while other regions of the United States, particularly the Midwest, have observed similar growth. Large-scale hog confinement operations dominate the industry. Anaerobic lagoons are the primary method of waste handling in these operations. Such lagoons may decrease water quality in the form of seepage to surrounding groundwater or surface water. Seepage poses a threat to human health as well as to aquatic life. Excessive nitrogen in drinking water is believed to cause methglobemia in infants, more commonly known as blue-baby syndrome. In addition, high nutrient levels in aquatic systems cause eutrophication, which may lead to fish kills due to decreased dissolved oxygen levels.

Barrington and Broughton (1988) concluded that solids entering the lagoon seal or fill soil pores, thus reducing seepage potential and the associated threats to the surrounding water resources. Similar results were observed in research completed by Ritter (1983), who monitored the groundwater impact of a newly constructed unlined swine lagoon situated in course-textured soils for four years. Although the study initially found increased NO$_3$-N, NH$_4$-N, CL and organic nitrogen concentrations, seepage rates decreased over time indicating that animal manure solids likely sealed the manure storage basin. Contrary to these results, Huffman and Revels (1998), Korom and Jeppson (1994) and Huffman and Westerman (1995), concluded that older unlined lagoons contributed a
significant amount of seepage to subsurface and surface water. Korom and Jeppson (1994), monitored unlined dairy lagoons for five years, and seepage rates were estimated to range from 1.3 to 9.1-cm/day. A study by Huffman et al. (1995) found significant seepage from 3.5 to 5-yr old unlined lagoons in sandy soil. Significant concentrations of NH$_4$-N (75-300-mg/L) and CL (50-140-mg/L) were detected in shallow subsurface waters down gradient from the lagoon, indicating that natural sealing of the lagoons had not occurred.

The use of constructed wetlands for managing livestock wastewater has grown considerably in recent years. Knight et al. (2000) described the use of constructed wetlands for the treatment of various types of animal manure including dairy manure, milk house wash water, runoff derived from cattle feeding operations, poultry manure, and swine manure. The study compiled data for removal rate of nutrients, bacteria, and sediments from 1300 locations. Average removal rates for total nitrogen (TN) and total phosphorus (TP) were 42%. Sievers (1997) conducted a study evaluating the treatment efficacy of four constructed wetlands receiving swine lagoon effluents. In that study, two constructed wetlands received wastewater from a primary anaerobic lagoon while the other two wetlands received effluent from a secondary lagoon. Comparing inflow and outflow concentrations, the wetlands reduced NH$_4$-N concentrations by an average of 30%. Sievers suggested that the highly reduced conditions in the wetland limited the nitrification process. It was also determined that phosphorus retention was poor (ranging from 15-27%) and influenced by the strongly reduced soil environment and the limited soil microsites responsible for adsorption. A review article by Cronk (1996) suggested
that properly designed, constructed and maintained treatment wetlands could effectively reduce nutrient loads downstream.

The research presented here evaluates the performance of a constructed wetland used for improvement of shallow groundwater quality. A pumping system was installed to collect groundwater, which was contaminated by lagoon seepage. Nutrient rich groundwater was routed to a constructed wetland for treatment and a mass balance calculated to characterize the nutrient assimilation capacity of the wetland.

**MATERIALS AND METHODS**

**Site Background**

The study site is located in the Middle Coastal Plain of North Carolina and was determined to be a threat to the surrounding water resources by Huffman and Revels (1998). Surface water quality sampling of the channelized stream adjacent to the lagoon found that ammonium-nitrogen (NH$_4$-N) concentrations ranged from 10-25 mg/L. Groundwater-monitoring wells were installed to a depth of 3 m (screened 2.4 to 3 m) around the periphery and down gradient of the lagoon in February 1999, Figure 1.

Data from these wells indicated that concentrations of NH$_4$-N along the periphery of the lagoon ranged from 100-200 mg/L, Figure 2. Groundwater quality samples from wells further down gradient showed no significant elevation in NH$_4$-N concentrations. It was concluded that the seepage water was traveling from the lagoon to the local discharge area, the channelized stream. Since monitoring of the surface water and groundwater quality adjacent to the lagoon began, hog production has ceased, and waste deposition into the lagoon has also stopped. Recently, the lagoon was closed, which involved the physical removal of the nutrient rich sludge.
Wetland Design and Construction

The wetland surface area was designed using criteria outlined by Reed et al. (1989). Criteria included inflow concentration, inflow rate, temperature factor, and desired outflow concentration, along with several other factors. The design inflow rate ranged from 0.3-m/yr to 0.9-m/yr based on the estimated seepage rates from the lagoon. A surface area ranging from 0.09- 0.28-ha (0.21 to 0.69-ac) was required for an outflow concentration of 0.1 mg/L at a loading rate of 8.2-m\(^3\)/day (1.5-gpm) at 100 mg/L of NH\(_4\)-N. Reed et al. (1989) recommended a 10% increase in the treatment surface area as a built-in safety factor to ensure adequate treatment by the wetland.

Wetland construction was initiated in late February 2000 and was completed in early March 2000. The finished surface area of the constructed wetland was 0.35-ha (0.85-acre). The wetland was constructed with a 15-cm clay liner and 15-cm of topsoil that served as the soil substrate for vegetation. Construction incorporated the use of a sinuous curve, as well as a variable slope and undulating bottom. The curvature was incorporated to increase flow path and travel time for the contaminants entering the wetland. The variable slope of the wetland was implemented to provide variable water levels for different plant species, with deeper pools expected to enhance denitrification. The water table was intercepted during construction at the design depth in the upper half of the wetland. Excavation depth therefore had to be raised. A bulkhead was installed to compensate for this loss in wetland depth and staged the flow on the upper end, Figure 4. The bulkhead structure was 20-cm tall (measured from the soil surface) with a series of discharge ports drilled along its 19-m length. Grab samples for measurements of water quality were collected along the bulkhead board weekly.
The finished elevation of the wetland ranged from 98.0-ft (29.9-m) at the inlet to 95.5-ft (29.12-m) in the deep pools near the outlet. The wetland was planted on a 1-m grid using various plants species that were either harvested locally or obtained from a nursery. The area above the bulkhead was planted with soft rush (*Juncus* sp.) that was transplanted from the research site. Below the bulkhead, several species were planted including lilies (*Nephar* sp.), pickeral weed (*Pontederia* sp.), cattails (*Typha* sp.), and bulrush (*Scirpus* sp.).

**Removal of Contaminated Groundwater**

Five pumping wells were installed along the periphery of the lagoon to intercept the seepage water before it reached the channelized stream. Wells were centered on the area where elevated concentrations of NH$_4$-N were present in the groundwater, Figure 3. Pumping wells were 19-mm in diameter and inserted inside a 50-mm casing. A drill rig was used to install the wells to 6-m, which is the depth of the impermeable clay layer: all well casings were screened 3 to 6-m. The screened portion of the casing was back-filled with sand followed by a 1.5-m section of bentonite. The last 0.5-m portion of the well was back-filled with sand and covered with a 0.15-m by 1-m by 1-m concrete slab. The pumping wells were plumbed in series to a jet pump that routes the nutrient rich groundwater to the constructed wetland.

**Flow Quantification**

Groundwater from the pumping system was routed to an inflow box that contained a 22.5° weir used to measure inflow volume (Figure 5). A PVC pipe was plumbed to the inflow box in order to evenly distribute the inflow across the wetland ground surface, Figure 6. The wetland outlet consisted of a flashboards riser with a 7.6-cm
tall, 30° weir/orifice plate, Figure 8. Continuous measurements of inflow and outflow were quantified using a pulley float mechanism with a pressure transducer used as a back up. The water level recorders were wired to a Blue Earth® microprocessor that measured voltage changes with regard to changes in water level (flow over the weir) at pre-programmed intervals. Stage was determined by converting measured voltage using referenced equations, derived from laboratory calibration. Water level was measured at the inflow, bulkhead, and outflow locations on an hourly basis and downloaded from the field at two-week intervals. Manually read gages were installed at each location to confirm the accuracy of microprocessor readings and were used to adjust any stray microprocessor readings.

Calibrations of the inflow and outflow weir were completed in August 2001. The calibration was done by taking measurements of head on the weir/orifice and quantifying the respective flow rate. The inflow weir was calibrated on-site by manually adjusting flow rate to the weir, measuring head on the 22.5° weir and quantifying flow volume. A duplication of the outflow weir/orifice was constructed with exact specifications and materials as the weir/orifice in the field, and calibrated using a flume in the Biological and Agricultural Engineering Department at North Carolina State University. Calibration data are displayed in Figures 8-10. Flow was quantified for the water balance by applying the hourly stage data from the Blue Earth® measurement to the appropriate weir/orifice equations generated from calibration, thus computing flow on an hourly basis. Flow was summed on a weekly basis and combined with weekly concentration data for mass balance determination.
Water Quality Sampling

Wells were generally sampled monthly but were often sampled bi-weekly to monitor water quality. Prior to sampling, wells were purged for approximately 3-min, or the equivalent of three well volumes, using a peristaltic pump. Following adequate well purging, samples were collected in clean plastic bottles and stored on ice. Surface water grab samples were taken at four locations: inflow box, downstream of nitrification field, bulkhead, and wetland outflow structure. Wetland inflow and outflow samples were taken on a daily basis and composited weekly via ISCO® automated samplers. Sample bottles stored in the ISCO® sampler were acidified prior to sample collection and gathered at weekly intervals. Grab samples were taken weekly at the bulkhead (near the wetland mid-point) and just downstream from the nitrification field. Grab samples collected at the bulkhead were taken as a composite of the flow through the ports located in the bulkhead. If no flow occurred at the bulkhead, samples were taken just upstream by submerging the sample bottle into the water column, taking care not to contaminate the sample with debris from the wetland. Grab samples collected downstream of the nitrification field were taken following the same protocol and immediately acidified following collection.

All water quality samples were stored on ice after collection and transported to the Biological and Agricultural Engineering Environmental Analysis Laboratory located at North Carolina State University. Following arrival at the laboratory, samples were transferred and stored at 4° C until analyzed. Samples were analyzed for total Kjeldahl nitrogen (TKN), nitrate nitrogen (NO₃-N), ammonia nitrogen (NH₃-N), total phosphorus (TP), orthophosphate phosphorus (OP), and chloride (CL). TKN was quantified using
persulfate digestion followed by ammonia-salicylate method for NH$_3$-N analysis. NO$_3$-N was determined by the cadmium reduction method. TP was analyzed using the persulfate digestion and ascorbic acid method, while OP was quantified using ascorbic acid method. Chloride determination was made using the ferricyanide method. All analysis were automated and described in The Standard Methods for the examination of Water and Wastewater 18$^{th}$ edition (1992).

**Plant Uptake**

Plant samples were taken at four locations to quantify nutrient uptake by plants and to determine the possibility of toxicity as an excessive number of the plants began to die-off in late July 2001. Four 1-m$^2$ plant samples were taken and dried prior to nutrient analysis. Laboratory values were applied to the dry weight of the samples to obtain kilograms of each nutrient bound in plant tissue. All dry plant samples were weighed and used for nutrient mass determination. Samples were then extracted from solid plant material for nutrients (NH$_4$-N, NO$_3$-N and CL) using potassium sulfate extraction (Page et. al. 1982), and analyzed for nutrient concentration using the protocols for water quality analysis outlined above. Available P was determined from the plant samples using a weak acid extraction and concentration was quantified using methods described by Page et. al (1982). Average values from the plant analysis ($\mu g/g/m^2$) were used to estimate mass of nutrients in plant tissue in the wetland area by applying the values to the entire area of the wetland (3500-m$^2$).
RESULTS AND DISCUSSION

Concentration Data

Figures 11-14 display nutrient concentrations with respect to time from the three sampling locations (inflow, bulkhead, and outlet) in the constructed wetland. With the exception of NO$_3$-N, a decrease in nutrient concentrations was generally observed moving down the wetland flow path. However, nutrient concentrations increased over time at each sampling location downstream from the inlet. The increase in the concentration may be explained by an increase in pumping/inflow rate, which decreased retention time of contaminants. Pumping rate was 1.5-gpm until late February 2001 when the rate was increased to 3.5 gpm to match evapotranspiration. Prior to February the pumping system was often shut down briefly to provide for adequate retention of the influent while newly planted wetland vegetation was becoming established.

Inflow TKN concentrations averaged 120 mg/L of which 85-percent was NH$_4$-N. Inflow NO$_3$-N concentration was typically less than 1-mg/L. Although lower, these concentrations reflect the source, lagoon seepage, which typically has high nutrient concentrations of NH$_4$-N and TKN and very low concentrations of NO$_3$-N. Since the pumping wells were screened from 3 to 6-m from the soil surface, no changes in nutrient speciation were expected, as anaerobic conditions prevail 1.2-m below the soil surface. Variation in the concentrations of these nutrients was expected considering that the source is groundwater, where infiltrating rainwater and drainage from up gradient will dilute the contaminant plume. NO$_3$-N concentrations were low throughout the wetland, with highest concentrations observed at the bulkhead; higher bulkhead NO$_3$-N
concentrations, relative to inflow concentrations, indicate that modest nitrification of the influent was occurring between the inflow and bulkhead.

In a constructed wetland system, the methods of removal of nitrogen and phosphorus are very different. Nitrogen species are removed by oxidation-reduction reactions that occur in the soil-water matrix by microbes, as well as by plant uptake for growth. Phosphorus binds to soil particles that have a finite number of exchange/binding sites. Plant uptake can also account for some phosphorus removal. Trends indicate that the phosphorus assimilative capacity may have been reached by late June 2001 as inflow and outflow concentrations were nearly equivalent.

Nutrient concentrations at the wetland outlet gradually increased beginning in March 2000. However, concentrations at the outlet were much lower than those at the bulkhead. The wetland was effective in retaining the two species of nitrogen, TKN and NO$_3$-N, as shown in Figures 11 and 12, respectively. Typically, TKN and NO$_3$-N concentrations were less than 2-mg/L and 1-mg/L at the outlet, respectively. Figure 13 shows increases in TP concentrations with respect to time, at each location downstream from the inlet. Concentrations at the bulkhead (midpoint of the wetland) began to increase within 2-months of pumping, but concentration at the outlet did not increase for 6-months following pumping. The most recent TP data show that the outlet concentrations were approximately equal to the inflow concentrations. Chloride data, displayed in Figure 14, followed a similar trend, with the latest inflow and outflow concentrations nearly equal (63 and 59-mg/L, respectively).

Mean nutrient concentration at each sampling location in the constructed wetland is shown in Figure 15. Little reduction in nutrient concentrations occurred between the
inflow box and the end of the nitrification bed (data not shown). When comparing the nutrient concentrations between the inflow box and bulkhead, TKN and NH$_4$-N concentrations decreased by 65%, while reductions in phosphorus concentrations were 34%. NO$_3$-N concentrations ranged from 0 to 14-mg/L at the bulkhead, again suggesting that some nitrification was occurring. All nutrient concentrations declined by an average of 85% between the bulkhead and the outlet. Similar results were observed when comparing inlet and outlet concentrations, where inlet concentrations were reduced by an average of 87%. Although the concentration data indicates that nitrification was occurring between the inlet and bulkhead, NO$_3$-N concentrations decreased between the bulkhead and outlet suggesting that denitrification was occurring.

**Mass Determinations**

Table 1 shows the monthly water balance for July 8, 2000 to August 20, 2001. Potential evapotranspiration (PET) was estimated using the Thornwaite method based on mean monthly temperature (Thornwaite, 1957). The monthly temperature values were adjusted using a correction factor based on the Penman method for estimating PET (Amatya et. al., 1995). Monthly PET volume was determined by multiplying the amount of estimated monthly PET (depth) by the water surface area of the wetland (3500-m$^2$). The volume of rainfall entering the wetland was computed by multiplying the amount of monthly rainfall by the surface area of the wetland plus the area surrounding the wetland crowned by berms (4300-m$^2$). Wetland inflow and outflow were calculated using data from water level recorders and applying the appropriate regression equations from flow calibrations. Change in storage was included in the water balance and determined by applying change in water depth in the wetland to the area of the wetland for volume
A daily water balance was conducted to observe any outlying data, which confirmed no unusual increase or decrease in wetland stage as a function of inflow, outflow or rainfall. Although calibration of the inflow and outflow quantification weirs was conducted, measurement errors associated with the water level recorders could occur. From the monthly water balance, Table 1, it appears that the largest errors were observed under high flow conditions. Considering that the points most difficult to calibrate in a weir/orifice system are the highest and lowest stages, even small measurement errors in the field water level stage may have large impact on flow calculation. Seepage into or out of the wetland could also be a factor, but the use of a liner and implementation of a water control structure in the adjacent channelized stream maintained water levels high enough in the stream to minimize the hydraulic gradient between the wetland and the stream. Overall, the water balance closed to within 6%, Table 2, and provided a sufficient estimation for a nutrient mass balance.

A mass balance was conducted to access the overall effectiveness of the constructed wetland in assimilating nutrients. By applying weekly flow volumes to weekly concentration data, the mass of nutrients entering and exiting the constructed wetland was determined. The wetland assimilated 81% of the TKN and \( \text{NH}_4\text{-N} \) from the pumped groundwater system, and retained 27% of the phosphorus, Table 3. Although CL is a conservative nutrient, not taken up by plants or microbes, 45 kg is unaccounted for in the mass balance estimate. Calculations were made to determine the mass of CL still residing in the wetland to further validate the mass balance. Mass of CL residing in the wetland was determined by multiplying the volume of water in the wetland by the average nutrient concentrations from the sampling locations. This calculation indicated
that 47 kg of CL was residing in the wetland therefore bringing the CL balance within 1%
. The mass balance of CL is also an indicator that the overall water balance was an
accurate estimation of the true water balance.

NH$_4$-N volatilization to the atmosphere is often a concern when dealing with
constructed wetlands and needs to be addressed in the mass balance. The pH of the
wetland water column and the pumped influent was typically 6.0 (ranging from 6.0 to
6.4) indicating that losses of NH$_4$-N due to volatilization were likely small, and therefore
insignificant in the mass balance of NH$_4$-N. The pKa of ammonia is 9.3 and suggests that
less than one-thousandth of the ammonical nitrogen entering the wetland was in a volatile
form. This estimate was corroborated by a model developed by Liang et al. (In Press),
where emissions of ammonia from swine lagoons were modeled. The model considered
four parameters wind speed, lagoon liquid properties, pH, temperature, and total
ammonia nitrogen concentration. Consequently, the wetland pH was determined to be the
dominant factor in determining NH$_4$-N volatilization at this low level. Plant uptake was
also a consideration for the mass balance. By adding data from the plant analysis to the
area of the wetland it was determined that wetland plants 35 kg of total nitrogen and 11
kg of total phosphorus, Table 4.

The stage-discharge of the constructed wetland is shown in Figure 16. Initially,
there were several periods of no outflow from the wetland that allowed for extended
periods for exposure to wetland assimilative processes and nutrient reduction. During the
last four months, discharge from the wetland was much higher than previous periods,
indicating greater potential for increased nutrient export from the wetland due to
decreased retention time. Monthly fluxes of total nitrogen and phosphorus are displayed
in Figure 17. Monthly loading rates were low until February 2001 to allow for plant development. Loading rates were increased from 2.3-kg/ha/day (1.5-gpm) early in the growing season to 5.4-kg/ha/day (3.5-gpm) to compensate for higher growing season evapotranspiration rates. Nitrogen mass reductions were fairly consistent throughout the study, with larger exports of total nitrogen observed in July and August of 2001. This could be associated with the die-off of plants in the upper portion (above the bulkhead) of the wetland. Although nitrogen uptake by plants was not a dominant mechanism of nitrogen removal, plants create an environment conducive for nitrogen removal in the soil matrix. Considering that the area above the bulkhead accounts for approximately 40% of the overall wetland surface area, the loss of plants in this section could have a significant impact on wetland performance. Total phosphorus export progressively increased to the point that inflow and outflow concentrations are equal, indicating that steady state assimilation of phosphorus may have been reached. Redox readings indicated the development of a strongly reduced soil environment, with values from March 2001-August 2001 averaging less than 0-mv. The pH of the water column conditions was favorable for the dissolution of the iron-phosphorus complex in the soil matrix, which is believed to be a major mechanism for phosphorus retention in constructed wetlands. These conditions limit phosphorus adsorption to soil particles, and might explain the recent large export of phosphorus from the wetland.

**SUMMARY AND CONCLUSIONS**

The constructed wetland was effective in retaining nitrogen from the shallow groundwater pumping system during two growing seasons. Based on the mass balance, the wetland assimilated 79% of the total nitrogen and 26% of the total phosphorus
pumped from the shallow groundwater. As a check of the mass and water balance, the CL balance was within 1%. Additionally, the wetland was effective at reducing nutrient concentrations at the outlet by an average of 87% for all nutrient species. Total nitrogen concentrations were reduced by 65% and total phosphorus by 34% at the bulkhead, which was approximately one-third of the way down the flow path. An increase in NO$_3$-N concentration was observed at the bulkhead, indicating that some nitrification had occurred above the bulkhead. The concentration and mass balance data indicated that denitrification was occurring below the bulkhead as overall mass and concentration of NO$_3$-N was lower at the outlet, than the bulkhead.

Total nitrogen export from the wetland was low through most of the study; however, nitrogen export increased in July and August of 2001. This could be attributed to the die-off of the wetland plants in the upper portion of the wetland, which decreased the effective treatment area. Further investigations are needed to determine why the plants died. The highly reduced conditions, continuous saturation, and high loading rates are all possible explanations. Phosphorus export increased with time until inflow and outflow concentrations became equal. Phosphorus assimilation is generally considered a function of the binding sites in soil. When binding sites become saturated, further adsorption is not possible, which leads to export of phosphorus. The wetland soil was found to be highly reduced eliminating any further possibility of phosphorus binding in the soil matrix and potentially limiting the nitrification process. As expected, the wetland assimilated large amounts of nitrogen, while phosphorus retention progressively declined. Although wetland processes were successful in assimilating nitrogen, further natural processes must be investigated for the removal of phosphorus.
Table 1. Monthly water balance values (m$^3$).

<table>
<thead>
<tr>
<th>Month</th>
<th>Inflow (m$^3$)</th>
<th>Rain (m$^3$)</th>
<th>Outflow (m$^3$)</th>
<th>PET (m$^3$)</th>
<th>Δ Storage (m$^3$)</th>
<th>Balance (m$^3$)</th>
<th>Error %</th>
</tr>
</thead>
<tbody>
<tr>
<td>July</td>
<td>0</td>
<td>440</td>
<td>160</td>
<td>280</td>
<td>-180</td>
<td>-180</td>
<td>-41</td>
</tr>
<tr>
<td>August</td>
<td>0</td>
<td>410</td>
<td>350</td>
<td>380</td>
<td>50</td>
<td>-270</td>
<td>-65</td>
</tr>
<tr>
<td>September</td>
<td>60</td>
<td>790</td>
<td>490</td>
<td>280</td>
<td>60</td>
<td>140</td>
<td>17</td>
</tr>
<tr>
<td>October</td>
<td>280</td>
<td>10</td>
<td>0</td>
<td>130</td>
<td>180</td>
<td>340</td>
<td>117</td>
</tr>
<tr>
<td>November</td>
<td>270</td>
<td>210</td>
<td>0</td>
<td>40</td>
<td>-170</td>
<td>270</td>
<td>56</td>
</tr>
<tr>
<td>December</td>
<td>280</td>
<td>70</td>
<td>0</td>
<td>0</td>
<td>20</td>
<td>370</td>
<td>106</td>
</tr>
<tr>
<td>January</td>
<td>70</td>
<td>50</td>
<td>0</td>
<td>10</td>
<td>70</td>
<td>180</td>
<td>150</td>
</tr>
<tr>
<td>February</td>
<td>20</td>
<td>180</td>
<td>0</td>
<td>45</td>
<td>-10</td>
<td>145</td>
<td>73</td>
</tr>
<tr>
<td>March</td>
<td>310</td>
<td>630</td>
<td>660</td>
<td>80</td>
<td>-110</td>
<td>90</td>
<td>10</td>
</tr>
<tr>
<td>April</td>
<td>680</td>
<td>130</td>
<td>450</td>
<td>230</td>
<td>30</td>
<td>160</td>
<td>20</td>
</tr>
<tr>
<td>May</td>
<td>720</td>
<td>410</td>
<td>470</td>
<td>340</td>
<td>-50</td>
<td>270</td>
<td>24</td>
</tr>
<tr>
<td>June</td>
<td>570</td>
<td>600</td>
<td>900</td>
<td>440</td>
<td>-20</td>
<td>-190</td>
<td>-16</td>
</tr>
<tr>
<td>July</td>
<td>660</td>
<td>700</td>
<td>1360</td>
<td>360</td>
<td>-100</td>
<td>-460</td>
<td>-34</td>
</tr>
<tr>
<td>August</td>
<td>390</td>
<td>590</td>
<td>980</td>
<td>290</td>
<td>-20</td>
<td>-310</td>
<td>-32</td>
</tr>
</tbody>
</table>

* % Error computed as ((inputs-outs)/inputs) x 100
Table 2. Overall Water balance values (m$^3$).

<table>
<thead>
<tr>
<th>Rainfall (m$^3$)</th>
<th>Inflow (m$^3$)</th>
<th>Outflow (m$^3$)</th>
<th>PET (m$^3$)</th>
<th>∆ Storage (m$^3$)</th>
<th>Balance (m$^3$)</th>
<th>Error%</th>
</tr>
</thead>
<tbody>
<tr>
<td>5200</td>
<td>4300</td>
<td>5800</td>
<td>2900</td>
<td>-250</td>
<td>+550</td>
<td>6.0</td>
</tr>
</tbody>
</table>

* % Error computed as ((inputs-outputs)/inputs) x 100

Table 3. Mass balance and nutrient reductions

<table>
<thead>
<tr>
<th></th>
<th>TKN (kg)</th>
<th>NH$_4$-N (kg)</th>
<th>NO$_3$-N (kg)</th>
<th>TP (kg)</th>
<th>OP (kg)</th>
<th>CL (kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Inflow</strong></td>
<td>485</td>
<td>415</td>
<td>2.90</td>
<td>226</td>
<td>203</td>
<td>346</td>
</tr>
<tr>
<td><strong>Outflow</strong></td>
<td>64</td>
<td>38</td>
<td>1.90</td>
<td>136</td>
<td>110</td>
<td>301</td>
</tr>
<tr>
<td><strong>Stored in Water Column</strong></td>
<td>38</td>
<td>32</td>
<td>0.9</td>
<td>30</td>
<td>28</td>
<td>47</td>
</tr>
<tr>
<td><strong>Assimilated</strong></td>
<td>383</td>
<td>345</td>
<td>0.1</td>
<td>60</td>
<td>65</td>
<td>-2</td>
</tr>
<tr>
<td>% Assimilated</td>
<td>79</td>
<td>83</td>
<td>3</td>
<td>27</td>
<td>32</td>
<td>-1</td>
</tr>
</tbody>
</table>

Table 4. Mass of nutrients in plant samples applied for the entire wetland area

<table>
<thead>
<tr>
<th>Mass (Kg)</th>
<th>TKN</th>
<th>NH$_4$-N</th>
<th>NO$_3$-N</th>
<th>OP</th>
<th>CL</th>
</tr>
</thead>
<tbody>
<tr>
<td>34</td>
<td>3.6</td>
<td>1.0</td>
<td>11</td>
<td>2.0</td>
<td></td>
</tr>
</tbody>
</table>
Figure 1. Plan view of research site showing the locations of pumping wells.

Figure 2. NH$_4$-N concentrations from monitoring wells along the periphery of the closed out lagoon.
Figure 3. Aerial Photograph of the site indicating the location of all monitoring wells.

Figure 4. Photograph of the wetland bulkhead.
Figure 5. Photograph of inflow box with details.

Figure 6. Photograph of inflow box with overland flow distribution.
Figure 7. Photograph of the wetland outlet.

![Photograph of the wetland outlet](image)

Figure 8. Calibration data for inflow weir.

![Calibration data for inflow weir](image)

\[ y = 0.0311x^{2.5891} \]

\[ R^2 = 0.9956 \]
Figure 9. Calibration data for outflow weir conditions.

Figure 10. Calibration data for outflow orifice conditions.
Figure 11. TKN concentrations at sampling locations in the constructed wetland (85% of TKN was in the NH$_4$-N form).

Figure 12. NO$_3$-N concentrations at sampling locations in the constructed wetland.
Figure 13. TP concentrations at sampling locations in the constructed wetland (86% of TP was in the OP form).

Figure 14. CL concentrations at sampling locations in the constructed wetland.
Figure 15. Comparison of nutrient concentrations at each sampling location.

Figure 16. Constructed wetland outlet stage data.
Figure 17. Monthly Fluxes (kg) of Total Nitrogen and Phosphorus.
REFERENCES


CHAPTER IV

The Effect of Monthly Variations in $E_h$, Dissolved Oxygen, and Temperature on Nutrient Retention in a Constructed Wetland

ABSTRACT

A free water surface wetland (0.35-ha) was utilized to clean up shallow groundwater that had been contaminated by anaerobic swine lagoon seepage. Fourteen monthly mass balances were calculated to compare the inflow and outflow of the wetland and were used to evaluate trends in nutrient reduction for nitrogen and phosphorus reduction. Nutrient assimilation on a percent reduction basis appeared to be higher in the dormant months. To explain nutrient retention variation, regression analysis of monthly nutrient reductions versus water temperature, dissolved oxygen (DO), and oxidation-reduction (redox) potential was completed. Nitrogen and phosphorus export from the wetland was positively correlated to water temperature (i.e., export increased as water temperature increased). In general, lower redox and DO were correlated to higher nutrient levels within the wetland, and subsequently to higher export from the wetland. Statistical analyses were performed to determine whether the observed trends were significantly different than zero by applying 95% confidence intervals to the slope of the regression line. Temporal parameters were found to be significant with respect to nutrient reduction for all nutrients except for $\text{NO}_3$-N, where redox potential was the only significant parameter. No definitive conclusions or recommendations can be made due to changes in dormant and growing season loading rates and plant death late in the second growing season.
INTRODUCTION

Constructed wetlands have received significant attention for their effectiveness in assimilating nutrients. Constructed wetlands have been used in several regions of the World as part of a waste treatment system (Cronk, 1996), and have been evaluated as a best management practice (BMP) to alleviate non-point source pollution (NPS) (Raisin and Mitchell (1995)). Wetland nutrient assimilating processes are quite variable and can include physical, chemical and biological processes that are influenced by environmental factors such as temperature and water level. Biological processes are typically regulated by temperature. However, water depth can dictate the amount of oxygen available in the soil substrate, as saturation and inundation can create anaerobic conditions, which can also be a limiting factor.

Wetlands used in waste treatment systems require a range of conditions for nutrient assimilations to take place. The most obvious biological process controlled by temperature is nutrient uptake by plants. In general, the warmer months are the most effective for nutrient uptake, since the plants are actively growing, while in the colder months plant uptake is low as plants undergo senescence. Other processes influenced by temperature include oxidation-reduction reactions. Nitrification and denitrification are processes in which nitrogen is effectively transformed in constructed wetland systems. Transformation of ammonium-nitrogen (NH$_4$-N) to N$_2$ gas occurs in wetland systems by a two-step process. In the nitrification process, NH$_4$-N is oxidized to nitrate-nitrogen (NO$_3$-N). Nitrification can be accomplished by shallow overland flow or by rapid vertical flow through coarse environments. Next, NO$_3$-N can be converted to N$_2$ gas via denitrification, which is a reductive process that requires anaerobic conditions, such as
saturated or inundated soils found in wetlands. When anaerobic or reducing conditions prevail, microbes consume organic molecules in the soil and produce electrons that reduce NO₃-N to harmless N₂ gas. Brusch and Nilsson (1993) studied denitrification rates with respect to temperature and found that denitrification rates changed rapidly between 5°C and 10°C, but also found that these rates fluctuated very little outside of these limits. Phipps and Crumpton (1994) showed that temperature and NO₃-N concentration were significant factors contributing to the loss of NO₃-N in wetland mesocosm studies. Temperature was significant in 3 of the 6 experiments, while NO₃-N concentration was statistically significant in all six cases, meaning that higher concentrations led to higher nutrient reduction. This data suggests, that temperature is not the only factor affecting nitrogen assimilations in wetlands, which is consistent with other literature (Anderson, 1977; Cerco, 1989 and Hill and Payton, 1998).

Research conducted by Reddy et al. (1989) found that the two-step process of nitrification-denitrification occurs in the plant root-sediment interface in wetlands. Their study, which was conducted in a growth chamber, determined that oxygen taken up by the plant provided a diffuse boundary around the roots where NH₄-N is converted to NO₃-N. The NO₃-N was then determined to be lost by denitrification after diffusing from the root zone to the soil matrix. Overall, the study found that 29% of the applied NH₄⁺ was lost to nitrification-denitrification in the root zone. Similarly, van Kessel (1978) found that denitrification occurred in the soil matrix and not in the water column by measuring oxidation-reduction potential of the soil and water column at various depths while monitoring NO₃-N concentrations. In his study, van Kessel determined the rate of denitrification was 160 mg NO₃-N/m²/day. Gale et al. (1993) reported that natural
wetland soils reduced NO₃-N more effectively than constructed wetland soils due to the increased build-up of carbon materials in mature natural wetlands. Results also indicated that the addition of wastewater enhanced the potential for denitrification in wetland soils. Bass and Evans (2000) studied the effect of temperature on nutrient reductions in a constructed wetland and found that greater reductions in NO₃-N and NH₄-N concentrations resulted when mean monthly temperatures were higher. However, their research indicated that increased temperatures resulted in higher concentrations of TKN and TP (total phosphorus) when comparing the outlet to the inlet.

The adsorption process that occurs in wetlands can best be defined by soluble phosphorus binding to iron in the soil matrix. Also, under reducing conditions, which typically exist in wetlands, mineral soils tend to export phosphorus as binding sites become saturated. This is a result of the reduction and solublization of the iron-phosphorus complex that occurs in the soil and results in phosphorus export. This process may be limited by the finite number of binding sites in the soil, and therefore be an ineffective mechanism for continuous storage of large phosphorus inputs. Research conducted by Kent (1994) on the performance of constructed wetlands showed that phosphorus levels were reduced by 37-90% when comparing inflow and outflow. In contrast, Kadlec (1995) observed that the adsorption potential of phosphorus in constructed wetlands was finite. The data indicated that phosphorus adsorption was only effective for a short period of time, after which large exports of phosphorus were observed. Richardson (1985) found that the initial removal of dissolved phosphorus was high in wetlands, due to microbial and plant uptake, and binding to soil particles.
Although binding to soil particles seemed to be effective initially, this mechanism was rapidly saturated, eventually leading to large exports of phosphorus.

To understand the ongoing processes of nutrient assimilation in a constructed wetland, redox, DO, and water temperature were measured at regular intervals. These data were compared to wetland monthly nutrient reduction (%) and mass attenuation (kg) using regression analysis to determine the effect of redox, DO or water temperature on wetland performance.

**MATERIALS AND METHODS**

**Site Background**

The study site, located in the Middle Coastal Plain of North Carolina was previously determined to be threatening to the surrounding water resources (Huffman and Revels, 1998). Monitoring wells were subsequently installed around the periphery and down gradient of the lagoon in February 1999. Data from these wells were collected on a monthly basis and indicated that concentrations of NH$_4$-N along the periphery of the lagoon ranged from 100 to 200 mg/L, as seen in Figure 1. To intercept the seepage water before reaching the drainage ditch, five pumping wells were installed along the periphery of the lagoon, centered on the area where elevated concentrations of NH$_4$-N occurred in the groundwater, Figure 2. The nutrient-rich groundwater from the pumping wells was routed to a constructed wetland.

Construction of the wetland was initiated in late February 2000 and completed in early March of the same year. Nutrient additions to the wetland began in June 2000. The wetland design surface area was determined using design criteria outlined by Reed et al. (1988), and discussed in Chapter 3. The initial inflow design rate was selected based on a
seepage value of 0.9-m/yr, which corresponded to a pumping rate of 1.6-gpm. The inflow rate could be varied from 0 to 13-gpm by operating the pump periodically or regulating the pumping rate manually using a ball valve. Initially, the pump was operated at low flow rates (1.5-gpm) to allow for wetland plants to mature and provide adequate nutrient retention in the colder months. With warmer spring air temperatures, pumping rates were increased to 3.5-gpm from March through October. Near the midpoint of the constructed wetland, a bulkhead was installed to divide the wetland into two cells and provide a means of staging flow from the upper cell.

**Water Quality Analysis**

Surface water grab samples were taken at four locations: inflow box, downstream of nitrification field, bulkhead, and wetland outflow structure. Wetland inflow and outflow samples were taken on a daily basis and composited weekly via ISCO® automated samplers. Sample bottles stored in the ISCO® sampler were acidified prior to sample collection and collected at weekly intervals. Grab samples were taken weekly at the bulkhead (near the wetland mid-point) and just downstream from the nitrification field. Grab samples collected at the bulkhead were taken as a composite of the flow through the ports located in the bulkhead. If no flow occurred at the bulkhead, samples were taken just upstream of the bulkhead by submerging the sample bottle into the water column, making sure not to contaminate the sample with debris from the wetland. Grab samples collected downstream of the nitrification field were taken following the same protocol and immediately acidified following collection.

All water quality samples were stored on ice and transported to the Biological and Agricultural Engineering Environmental Analysis Laboratory located at North Carolina
State University for analysis. Following arrival at the laboratory, samples were transferred and stored at 4\(^\circ\) C until analyzed. Samples were analyzed for total Kjeldahl nitrogen (TKN), nitrate nitrogen (NO\(_3\)-N), ammonia nitrogen (NH\(_3\)-N), total phosphorus (TP), orthophosphate phosphorus (OP), and chloride (CL). TKN was quantified using persulfate digestion followed by ammonia-salicylate method for NH\(_3\)-N analysis. NO\(_3\)-N was determined by the cadmium reduction method. TP was analyzed using the persulfate digestion and ascorbic acid method, while OP was quantified using ascorbic acid method. Chloride determination was made using ferricyanide method for automated analysis. All analyses were completed using automated protocols described in The Standard Methods for the examination of Water and Wastewater 18\(^{th}\) edition (1992).

A detailed water and mass balance was completed to determine wetland treatment efficacy (these results are described in detail in Chapter 3). Monthly nutrient reductions were evaluated by determining the monthly mass efflux of nutrients leaving the wetland divided by the monthly loading to the wetland. Monthly nutrient reduction was then compared to average monthly water temperature, wetland soil oxidation-reduction potential, and dissolved oxygen concentrations of the wetland water column to determine their effects on monthly nutrient reduction and mass attenuation by the wetland.

**Dissolved Oxygen**

Measurements of dissolved oxygen (DO) were made using a YSI\(^\text{®}\) multi-meter at four locations in the constructed wetland, Figure 3. The multi-meter was calibrated on a regular basis using a 100% saturated reference. Prior to taking a measurement, the probe was evaluated for accuracy using the ambient air as a reference. The measurement of dissolved oxygen concentration (mg/L) in the water column was taken by submerging the
probe to the mid-point of the water column at each sampling location and sustained there until DO measurements stabilized, which was usually about 2-min. The stabilized reading was recorded after each measurement. Table 1 shows the water depths in which the measurements were taken. The probe was rinsed between measurements to ensure accuracy at each sampling location.

**Temperature**

Water temperature was monitored continuously at three locations (Figure 3) using submersible Hobo® data loggers. Sinkers were attached to the waterproof protective case of the logger in order to maintain the logger at the bottom of the water column. Data loggers were programmed to record temperature every two hours. The Hobo® data logger was also used to measure ambient air temperature. The recorder was attached to the underside eave of the roof of the constructed wetland pump house to protect it from adverse weather. Additionally, holes were drilled in the protective case to allow for adequate ventilation, preventing any possible greenhouse effects from occurring in the plastic case, and introducing a potential overestimate of ambient air temperature. Temperature loggers were downloaded on regular intervals to ensure a continuous record and minimize loss of data due to equipment failure.

**Oxidation-Reduction Potential**

Wetland soil oxidation-reduction (redox) potential was measured at the same four locations. Platinum electrodes were fabricated in the Biological and Agricultural Engineering Department, following procedures described by Hayes and Vepraskas (1998, 2000). Prior to installation in the field, all probes were calibrated with a standard solution for redox measurements (Light, 1973). Probes that did not measure the standard solution
within 1% were not used for measurement in the field. Following calibration, five probes were installed in the wetland at a depth of 15-cm into the soil matrix and were situated 1-m apart. Measurements were made using a Cole-Parmer® multi-meter equipped with an Ag-AgCl reference electrode. All measurements were made and recorded in millivolts (mV). Measurements of redox can vary with temperature. Type T thermocouples were installed to the same depth (15-cm) as the redox probes at each location to record soil temperature. Soil redox and soil temperature measurements were collected at the same time to minimize any errors.

**Statistical Analysis**

To evaluate spatial differences in the temporal data, analysis of variance (ANOVA) was completed using SAS (1990). In addition, linear regression analyzes were performed on wetland nutrient reduction data for each nutrient species with average temporal measurements used as the independent variables. The slope of the regression line was determined and correlation coefficients ($R^2$) were used to evaluate the strength of the relationship. Student t-tests were performed on each slope of the regression line to determine whether they were significantly different than zero. Based on sample size, t-values were multiplied by the standard error estimate for the slope and used to create a confidence interval about the slope of the regression line (slope plus or minus t-value times standard error). A slope of zero indicates that variations of temporal measurement have no effect on nutrient reduction.

**RESULTS AND DISCUSSION**

The results presented here evaluate nutrient reduction in the constructed wetland as influenced by water and ambient air temperature, oxidation-reduction potential in the
wetland soil, and dissolved oxygen (DO) concentration in the water column. Conditions in the field were variable, with the growing season (March to October) receiving a higher loading rate than the dormant season. Consequently, the data was analyzed according to this seasonal variation. Data from the growing season was not collected under steady state conditions because wetlands plants above the bulkhead started to die-off in July 2001. A large majority of the plants were dead in August 2001. The death of the plants changed the dynamic conditions in the wetland. Wetland plants are typically responsible for a small amount of nutrient up-take, but provide conditions in the soil that enhance nutrient transformation and assimilation. Plant die-off likely impacted nutrient reduction, which should be taken into consideration when interpreting the results.

There was little variation in water temperature amongst sampling locations with no statistical difference in mean values between locations. Average daily air and water temperature data are displayed in Figure 4. Maximum average daily temperatures for the air and for water were 32.7°C and 27.0°C, respectively, while minimum temperatures were –2.8°C and 5.0°C for the air and water, respectively. Overall, average daily air temperature was 17.0°C while average water temperature was 16.6°C. Since the wetland was in the first full growing season, the overall canopy coverage was not complete, therefore some areas were not fully shaded. Although average air and water temperatures were the same, water temperature was less affected by spikes in daily air temperature. The recorded air temperature data was comparable to the measurements taken by North Carolina Climate Office at a nearby location in Goldsboro.

Soil temperature was measured bi-weekly at a depth 15-cm (Figure 5). These measurements were taken at the same time as redox measurements and were used to
adjust redox potential measurements. The average water temperature and soil temperature were the same at 16.6°C. Maximum temperatures for water and soil were 27.0°C and 25.5°C, and minimum temperatures were 5.0°C and 7.2°C, respectively. Soil temperature, was more buffered than air temperature, similar to that of the water temperature.

DO levels, Figure 6, were slightly variable, although similarities between location and time were observed, with no statistical difference between locations. Early in the study, DO levels were high because water holds more dissolved oxygen at lower temperatures with saturation at 4°C. As temperatures increased in the spring and summer, DO levels decreased rapidly. The ability of water to hold oxygen decreases at higher temperatures, and biological activity increases, which consumes dissolved oxygen in the water column. The local differences are believed to be due to variable water depth. Since oxygen is translocated from air to water, measurements made in the shallow water were closer to the water/air boundary and have more dissolved oxygen than do deeper measurements (all measurements were made at the midpoint in the water column, the deeper the water column, the further the measurement from ambient air).

Figure 7 displays bi-weekly measurements of redox potential (mV). Variability was observed amongst locations, but the general trend was a decrease in redox potential over time at each location. Deep and shallow locations show similar measured values and their means were determined to be statistically equivalent, while the mid-depth and bulkhead-upstream measurements were statistically different. It was initially assumed that deeper water would lower the redox potential due to the poor translocation of oxygen through water. However, many variables influence redox potential, including vegetation
Data Analysis

Average monthly values of redox potential, DO, and temperature were compared to monthly mass nutrient (kg) attenuation which was determined by subtracting the mass leaving the wetland from the mass loading (wetland mass balance evaluation is described in Chapter 3). Table 2 shows monthly mass attenuation values for each nutrient species, while Figure 8 shows a comparison of seasonal mass nutrient fluctuations. The growing season was defined as the months March through September, while the dormant season was defined as the months of October through February. It should be noted that the growing season loading rates were nearly 3 times higher than the dormant season. The mass export of nutrients was much higher for the growing season than the dormant season with nutrient attenuation in the dormant season nearly 100%.

Additional analysis compared average monthly values of redox potential, DO, and temperature to wetland nutrient reduction (%). Monthly nutrient reduction variations were calculated, by dividing the mass leaving by the mass loading, and subtracting that value from 1. Table 3 displays monthly nutrient reduction for each nutrient species. Negative values indicate that more nutrient was exported from the wetland than was pumped into the wetland. Table 4 summarizes average redox, DO, and temperature data. Monthly nutrient reduction was compared to redox, DO, and temperature monthly data to determine what effects, if any that they had on wetland nutrient attenuation. Data from Table 4 was combined with the data from Tables 2 and 3 to complete regression analysis. Monthly air temperature was excluded from regression analysis due to its similarity to the
water temperature data, and the increased daily variability. Since the analysis describes processes on-going in the wetland, water temperature data were more representative of wetland conditions. Data analysis indicated that redox potential decreased over time, as did dissolved oxygen. Further statistical analysis was conducted by season, since loading rates and nutrient retention differed greatly across season.

**Statistical Analysis**

The dormant season nutrient attenuation was consistently 100% for all nutrient species, and therefore regression analysis was not conducted for this period. The inflow rate during that time was 1.5-gpm (2.3 kg/ha/day) and equivalent to the design inflow rate. Although defined as the dormant period, some wetland plants were actively growing and did not undergo seasonal senescence during the first dormant season. Plant growth is essential for wetland processes as plants not only take up nutrients, but also create an environment conducive to redox reactions by translocating oxygen to the root zone and providing a carbon source for microbes. An additional consideration is that the dormant season was the startup period for the wetland system with short-term storage in the wetland buffering initial export values.

Regression analysis was conducted by comparing average monthly measurements to monthly mass nutrient attenuation for the entire study period. Table 5 displays the correlation coefficients for this regression. This analysis found no strong relationships in the data, DO measurements more strongly correlated to TKN and NH₄-N mass attenuation than other nutrient species. A positive slope of the regression line indicates that nutrient retention was positively correlated with a particular factor, while a negative slope indicates that nutrient retention was negatively correlated to the parameter. Figures
9-11 display graphs from the linear regression. Increasing water temperatures led to increases in TKN and NH$_4$-N mass attenuation, while decreasing DO levels and redox potential had the same effect. The results for OP and TP were different, as increases in water temperature corresponded to decreases in mass attenuation and decreases in DO and redox potential decreased OP and TP mass attenuation.

Further, regression analysis was completed comparing nutrient reduction to average monthly redox potential, DO, and water temperature for the growing season. The data for the growing season shows monthly nutrient reduction (%) progressively decreasing for each nutrient species. This result was likely due to the fact that wetland plants in the upper portion (above the bulkhead) started to die off in July with most of the plants dead by August 2001. The correlation coefficients from the regression analysis for the growing season are summarized in Table 6, which shows that redox potential was more strongly associated to nutrient reduction than any other parameter.

Final regression analysis was conducted to evaluate nutrient reduction (%) as a function of redox potential, DO, and temperature using the data from the entire study period. This regression analysis determined stronger correlation coefficients than the growing season and therefore further analyses were conducted to evaluate the slopes of the regression lines. Redox potential was more strongly correlated with nutrient reduction than DO or water temperature, Table 7. Figures 12-14 display graphs from the linear regression, indicating trends for the entire year. Decreasing redox levels corresponded to a decrease in nutrient reduction for all nutrient species while, increasing water temperatures, led to a decrease in nutrient reduction. A decrease in nutrient reduction
corresponded to a decrease in DO levels for all nutrient species with the exception of nitrate, where no effect was observed.

Student t-tests were performed on the slope of each regression line to determine whether they were significantly different than zero. A slope of zero indicates that variations in temporal measurement had no effect on nutrient reduction. Table 8 shows 95% confidence intervals for the slopes of the regression lines. Redox potential was found to have a significant effect on all nutrient reductions. In addition, DO and temperature had a significant impact on all nutrient species except NO\textsubscript{3}-N. The largest slopes (negative) were observed in the DO parameter for TP and OP reduction, suggesting that decreasing DO led to large exports of TP from the wetland. A negative slope was also observed in water temperature regression indicating that an increase of water temperature led to lower reduction.

TKN and NH\textsubscript{4}-N reduction was found to decrease with increasing water temperature and with decreasing levels of DO and redox potential. For a wetland to be effective, NH\textsubscript{4}-N must first be oxidized to NO\textsubscript{3}-N, followed by reduction of NO\textsubscript{3}-N, or denitrification, to N\textsubscript{2} gas. Since the predominant conditions were reducing, oxidation of TKN and NH\textsubscript{4}-N was limited, therefore limiting the overall reduction of these nutrient species in the constructed wetland. Surprisingly, lower redox potential led to lower NO\textsubscript{3}-N assimilation, since it was thought that with a reduced environment within the wetland, NO\textsubscript{3}-N would be denitrified. However, this observation is complicated by the conversion of NH\textsubscript{4}-N to NO\textsubscript{3}-N within the wetland. This conversion of reduced forms of nitrogen increase the NO\textsubscript{3}-N load in the wetland when compared to the inlet load. TP and OP reduction decreased as redox potential decreased (\(r^2\) of 0.89 and 0.91, respectively). Iron
oxides in the soil matrix provide binding sites for influxes of phosphorus. When the redox potential reaches the level of 100-mV, iron oxides in the soil matrix dissolve, releasing bound phosphorus and limiting further phosphorus adsorption. When that occurs, phosphorus assimilation capacity is exceeded, not only failing to assimilate additional phosphorus, but also releasing phosphorus, which was previously retained. TP and OP reduction was found to decrease with increasing water temperature.

**SUMMARY AND CONCLUSIONS**

Wetland nutrient reduction progressively decreased between March and October of 2001, with a large decrease in nutrient reduction in July and August 2001. Regression analysis was conducted to describe monthly nutrient reduction as a function of average monthly temporal variations. Average monthly redox potential, DO, and water temperature had significant effects on wetland assimilation processes for all nutrient species analyzed, with the exception of NO$_3$-N. For NO$_3$-N, only redox potential was observed to have a significant effect on nutrient reduction. Decreasing DO and redox levels led to decreased nutrient reduction. Although these conditions are conducive for denitrification, they are not suitable for oxidation of NH$_4$-N to NO$_3$-N. Therefore, insufficient nitrification of NH$_4$-N could explain limited nitrogen removal in this system. Reducing conditions in the wetland also affects the phosphorus export. When strongly reduced conditions (100-mV) develop, solubilization of the iron-phosphorus complex occurs such that phosphorus is no longer adsorbed to the soil matrix and any influx of phosphorus will be transported in outflow. These results indicate that many processes affect nutrient assimilation in wetlands and that not all processes enhance the assimilation of all nutrients species.
The analysis presented here does not take into consideration variation in loading rates by season, or the fact that plants in the upper portion of the wetland started to die off in July with a large portion of the plants dead in August. Plant die-off can have a significant impact on nutrient assimilation. Plants not only are involved in the uptake of nutrients but also provide an environment in the soil that is conducive for redox reactions, which transform and assimilate the influx of nutrients. In addition, the early part of the study period included low loading rates and wetland start-up (initial filling of the wetland with influent), which would provide for high nutrient assimilation rates. Combining the increased nutrient retention during the start up process, and the potential decreased attenuation due to plant death, regression line slopes may be exaggerated. Further studies are therefore required to clarify the mechanisms affecting nutrient assimilation and export from the wetland system.
Table 1. Average water depths at DO sampling locations.

<table>
<thead>
<tr>
<th>Average Water Depth (cm)</th>
<th>Bulkhead-Upstream</th>
<th>Bulkhead-Downstream</th>
<th>Mid-Depth</th>
<th>Deep</th>
</tr>
</thead>
</table>

Table 2. Monthly nutrient mass attenuated (kg)* for each nutrient.

<table>
<thead>
<tr>
<th>Month</th>
<th>TKN</th>
<th>NH₄-N</th>
<th>NO₃-N</th>
<th>TP</th>
<th>OP</th>
</tr>
</thead>
<tbody>
<tr>
<td>October</td>
<td>30.6</td>
<td>27.2</td>
<td>0.01</td>
<td>13.1</td>
<td>12.0</td>
</tr>
<tr>
<td>November</td>
<td>26.9</td>
<td>26.2</td>
<td>0.03</td>
<td>13.0</td>
<td>12.5</td>
</tr>
<tr>
<td>December</td>
<td>35.3</td>
<td>28.6</td>
<td>0.01</td>
<td>16.6</td>
<td>16.2</td>
</tr>
<tr>
<td>January</td>
<td>13.1</td>
<td>12.2</td>
<td>0.00</td>
<td>6.64</td>
<td>6.4</td>
</tr>
<tr>
<td>February</td>
<td>2.8</td>
<td>2.4</td>
<td>0.00</td>
<td>1.40</td>
<td>1.1</td>
</tr>
<tr>
<td>March</td>
<td>33.4</td>
<td>27.6</td>
<td>0.00</td>
<td>10.4</td>
<td>8.5</td>
</tr>
<tr>
<td>April</td>
<td>93.6</td>
<td>79.4</td>
<td>0.14</td>
<td>34.2</td>
<td>31.3</td>
</tr>
<tr>
<td>May</td>
<td>65.0</td>
<td>61.5</td>
<td>0.12</td>
<td>26.1</td>
<td>21.8</td>
</tr>
<tr>
<td>June</td>
<td>55.7</td>
<td>49.8</td>
<td>0.04</td>
<td>3.58</td>
<td>6.2</td>
</tr>
<tr>
<td>July</td>
<td>49.1</td>
<td>45.3</td>
<td>1.05</td>
<td>-23.1</td>
<td>-13.5</td>
</tr>
<tr>
<td>August</td>
<td>22.3</td>
<td>21.2</td>
<td>-0.20</td>
<td>-8.61</td>
<td>-7.1</td>
</tr>
</tbody>
</table>

* Mass Attenuated = Mass in – Mass out

Table 3. Monthly nutrient reduction* (%) for each nutrient.

<table>
<thead>
<tr>
<th>Month</th>
<th>TKN</th>
<th>NH₄-N</th>
<th>NO₃-N</th>
<th>TP</th>
<th>OP</th>
</tr>
</thead>
<tbody>
<tr>
<td>October</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>November</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>December</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>January</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>February</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>March</td>
<td>95.6</td>
<td>99.4</td>
<td>37.8</td>
<td>80</td>
<td>77.6</td>
</tr>
<tr>
<td>April</td>
<td>98.3</td>
<td>99.9</td>
<td>100</td>
<td>89.8</td>
<td>90.0</td>
</tr>
<tr>
<td>May</td>
<td>94.6</td>
<td>98.0</td>
<td>100</td>
<td>64.2</td>
<td>62.4</td>
</tr>
<tr>
<td>June</td>
<td>84.2</td>
<td>88.9</td>
<td>15.5</td>
<td>10.9</td>
<td>23.1</td>
</tr>
<tr>
<td>July</td>
<td>68.7</td>
<td>77.3</td>
<td>69.6</td>
<td>-72.1</td>
<td>-45.3</td>
</tr>
<tr>
<td>August</td>
<td>49.9</td>
<td>56.2</td>
<td>-19.8</td>
<td>-39.1</td>
<td>-36.1</td>
</tr>
</tbody>
</table>

* Reduction % = (1 –mass\_out/mass\_in) × 100
Table 4. Summary of average monthly measurements.

<table>
<thead>
<tr>
<th>MONTH</th>
<th>Redox (mV)</th>
<th>DO (mg/L)</th>
<th>Water Temp. (°C)</th>
</tr>
</thead>
<tbody>
<tr>
<td>October</td>
<td>-12</td>
<td>10.4</td>
<td>16.4</td>
</tr>
<tr>
<td>November</td>
<td>-4</td>
<td>10.5</td>
<td>12.0</td>
</tr>
<tr>
<td>December</td>
<td>3</td>
<td>9.4</td>
<td>6.7</td>
</tr>
<tr>
<td>January</td>
<td>-15</td>
<td>9.9</td>
<td>6.9</td>
</tr>
<tr>
<td>February</td>
<td>-4</td>
<td>8.5</td>
<td>9.6</td>
</tr>
<tr>
<td>March</td>
<td>-15</td>
<td>7.5</td>
<td>11.0</td>
</tr>
<tr>
<td>April</td>
<td>-50</td>
<td>2.9</td>
<td>15.8</td>
</tr>
<tr>
<td>May</td>
<td>-70</td>
<td>2.2</td>
<td>20.9</td>
</tr>
<tr>
<td>June</td>
<td>-95</td>
<td>0.8</td>
<td>25.1</td>
</tr>
<tr>
<td>July</td>
<td>-132</td>
<td>0.6</td>
<td>23.2</td>
</tr>
<tr>
<td>August</td>
<td>-151</td>
<td>1.9</td>
<td>24.5</td>
</tr>
</tbody>
</table>
Table 5. Correlation coefficients from regression analysis for the study period comparing monthly mass nutrient attenuation (kg) to average monthly temporal measurements.

<table>
<thead>
<tr>
<th>Nutrient</th>
<th>Redox (mV)</th>
<th>DO (mg/L)</th>
<th>Water Temp. (°C)</th>
</tr>
</thead>
<tbody>
<tr>
<td>TKN</td>
<td>0.09</td>
<td>0.37</td>
<td>0.19</td>
</tr>
<tr>
<td>NH₄-N</td>
<td>0.12</td>
<td>0.41</td>
<td>0.24</td>
</tr>
<tr>
<td>NO₃-N</td>
<td>0.15</td>
<td>0.18</td>
<td>0.10</td>
</tr>
<tr>
<td>TP</td>
<td>0.29</td>
<td>0.07</td>
<td>0.12</td>
</tr>
<tr>
<td>OP</td>
<td>0.26</td>
<td>0.04</td>
<td>0.09</td>
</tr>
</tbody>
</table>

Table 6. Correlation coefficients from regression analysis for the growing season comparing monthly nutrient reduction (%) to average monthly temporal measurements.

<table>
<thead>
<tr>
<th>Nutrient</th>
<th>Redox (mV)</th>
<th>DO (mg/L)</th>
<th>Water Temp. (°C)</th>
</tr>
</thead>
<tbody>
<tr>
<td>TKN</td>
<td>0.83</td>
<td>0.21</td>
<td>0.44</td>
</tr>
<tr>
<td>NH₄-N</td>
<td>0.81</td>
<td>0.20</td>
<td>0.43</td>
</tr>
<tr>
<td>NO₃-N</td>
<td>0.21</td>
<td>0.00</td>
<td>0.13</td>
</tr>
<tr>
<td>TP</td>
<td>0.84</td>
<td>0.40</td>
<td>0.56</td>
</tr>
<tr>
<td>OP</td>
<td>0.87</td>
<td>0.37</td>
<td>0.56</td>
</tr>
</tbody>
</table>
Table 7. Correlation coefficients from regression analysis for the entire year comparing monthly nutrient reduction (%) to average monthly temporal measurements.

<table>
<thead>
<tr>
<th>Nutrient</th>
<th>Redox (mV)</th>
<th>DO (mg/L)</th>
<th>Water Temp. (°C)</th>
</tr>
</thead>
<tbody>
<tr>
<td>TKN</td>
<td>0.84</td>
<td>0.46</td>
<td>0.53</td>
</tr>
<tr>
<td>NH$_4$-N</td>
<td>0.78</td>
<td>0.37</td>
<td>0.47</td>
</tr>
<tr>
<td>NO$_3$-N</td>
<td>0.48</td>
<td>0.29</td>
<td>0.37</td>
</tr>
<tr>
<td>TP</td>
<td>0.89</td>
<td>0.62</td>
<td>0.64</td>
</tr>
<tr>
<td>OP</td>
<td>0.91</td>
<td>0.63</td>
<td>0.65</td>
</tr>
</tbody>
</table>
Table 8. Student t-test data for temporal effects on treatment with 95% confidence intervals for slope, for the entire year.

<table>
<thead>
<tr>
<th></th>
<th>TKN</th>
<th>NH₄-N</th>
<th>NO₃-N</th>
<th>TP</th>
<th>OP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Slope</td>
<td>RD</td>
<td>DO</td>
<td>T</td>
<td>RD</td>
<td>DO</td>
</tr>
<tr>
<td></td>
<td>0.28</td>
<td>2.69</td>
<td>-1.96</td>
<td>0.23</td>
<td>2.08</td>
</tr>
<tr>
<td>UCL</td>
<td>0.37</td>
<td>4.95</td>
<td>-0.67</td>
<td>0.32</td>
<td>4.13</td>
</tr>
<tr>
<td>LCL</td>
<td>0.19</td>
<td>0.43</td>
<td>-3.25</td>
<td>0.13</td>
<td>0.03</td>
</tr>
<tr>
<td>S/NS</td>
<td>S</td>
<td>S</td>
<td>S</td>
<td>S</td>
<td>S</td>
</tr>
</tbody>
</table>

UCL = Upper Confidence Limit, LCL = Lower Confidence Limit, S = Significant, NS = Not Significant, RD = Redox potential, DO = Dissolved oxygen, T = Water temperature °C
Figure 1. NH$_4$-N concentrations from wells proximate to the closed out lagoon.

Figure 2. Plan view of pumping wells and constructed wetland.
Figure 3. Locations of temporal measurements in constructed wetland.

Figure 4. Comparison of daily average air and water temperatures.
Figure 5. Soil temperature (°C) at each sampling location with respect to time.

Figure 6. Dissolved oxygen concentrations at four sampling locations with respect to time.
Figure 7. Oxidation-reduction potential measurements with respect to time.

Figure 8. Comparison of seasonal nutrient mass fluctuations.
Figure 9. Regression analysis comparing monthly mass attenuation (kg) as a function of average monthly water temperature (°C).

Figure 10. Regression analysis comparing monthly mass attenuated (kg) as a function of average monthly redox potential (mV)
Figure 11. Regression analysis comparing monthly mass attenuation (kg) as a function of average monthly DO (mg/L).

Figure 12. Regression analysis comparing monthly nutrient reduction (%) as a function of average monthly water temperature (°C).
Figure 13. Regression analysis comparing monthly nutrient reduction (\%) as a function of average monthly redox potential (mV).

Figure 14. Regression analysis comparing monthly nutrient reduction (\%) as a function of average monthly DO (mg/L).
REFERENCES


CHAPTER V
A Steady-State Hydrologic Evaluation of a Site Contaminated with Anaerobic Lagoon Seepage Using MODFLOW-GMS

ABSTRACT
The shallow groundwater near a swine production facility in the Coastal Plain of North Carolina was identified as being contaminated with lagoon seepage. The hydrology of the site was evaluated using MODFLOW to assess the impact of the seepage plume on nearby surface waters. MODFLOW is a finite element groundwater model that combines boundary conditions and model inputs to generate head values for each cell in a finite element grid. MODFLOW was calibrated using water level data and measurements from the site. Once calibrated, MODFLOW was used to evaluate the influence of channelized stream levels on movement of contaminants in the groundwater plume. On-site plume mitigation actions included installation of a water control structure in the stream outlet to manage groundwater gradient and installation and pumping of five 50-mm diameter wells in the groundwater plume. The implementation of the water control structure raised the water level in the stream, reducing the gradient from the closed-out lagoon. Model simulations indicated a decrease in the hydraulic gradient from the former lagoon from 0.0045, in the free drainage case, to 0.0027 in the controlled drainage case. From the gradient calculations, travel time of the seepage plume to the stream increased from 380-days free drainage scenario, to 640-days in the controlled drainage case.

MODFLOW analysis of the pumping system indicated that 6.3-gpm was needed to reverse the gradient to the stream and capture the seepage plume in the free drainage
mode, while 4.7-gpm was required to reverse the direction of groundwater flow with controlled drainage. A hydrologic analysis was also conducted to evaluate pumping rates required to mitigate an actively leaking lagoon. Simulations were performed using an interceptor drain (French drain) adjacent to the lagoon for collection of seepage discharge. Model simulations predicted that under controlled drainage the drain line collected 51.5-m$^3$/day, while under free drainage 67.4-m$^3$/day was collected. This modeling scenario assumed the worst-case, that all waste deposited in the lagoon was lost to seepage.

INTRODUCTION

Non-point pollution derived from agriculture has received significant attention nationwide. Hypoxic zones in the Gulf of Mexico have been blamed on the excessive nutrients in agricultural drainage from cropping systems in the Midwest. The North Carolina Division of Water Quality (NCDWQ) suggested that agricultural non-point source pollution is a significant contributor to water quality problems in North Carolina (NCDWQ, 1996), stating that agriculture is the largest contributor of nitrogen in the Neuse River Basin. Impacts of non-point source pollution (NPS) in fresh water systems can include eutrophication (Paerl, 1988), fish kills, and the contamination of drinking water sources. The state of North Carolina has mandated the implementation several Best Management Practices (BMP) to alleviate the nitrogen load entering the fresh water sources derived from agriculture. The goal of the BMP mandate is to reduce nitrogen loading to the Neuse River by 30-percent by the year 2003.

Anaerobic swine lagoons have received significant attention in the literature due to their potential for nonpoint source pollution (NPS). One concern associated with
anaerobic swine lagoons is seepage. The literature indicates that improperly or poorly constructed lagoons and those lagoons that are situated in a coarse-textured environment have high potential for seepage (Huffman and Westerman, 1995; Huffman and Revels, 1998; Westerman et al. 1995). Several citations suggest that properly constructed lagoons or those lagoons that utilize liners pose little threat to the surrounding water quality. Some researchers have also observed biological sealing of unlined animal manure lagoons in coarse-textured soils (Barrington and Broughton, 1988; Ritter, 1983). However, despite these observations, Ritter recommended that unlined animal waste lagoons not be installed at locations where coarse-textured soils and high water tables are present. This recommendation was also corroborated by Phillips et al. (1983), whose observations suggested that clogging of soil pores can occur. However, the authors were careful to suggest that total sealing or the creation of an impermeable boundary was unlikely.

Considering this information, there is a significant number of older lagoons that were not required to use liners as a protective barrier against seepage. Huffman and Westerman (1995) conducted a study on older (10 to 20 years old) unlined anaerobic lagoons in the Lower Coastal Plain of North Carolina and concluded that 45% of lagoons, in the study, had small seepage losses (<0.5-kg/day), while the remaining lagoons had moderate to severe seepage losses (1-10-kg/day). In another study, Korom and Jeppson (1994) monitored groundwater quality proximate to a newly constructed unlined lagoon dairy lagoon. In that study, seepage rates were monitored for five years and estimated to range from 1.3 to 9.1cm/d. Concentrations of the leachate surrounding the lagoon ranged from 10 to 100 mg/L NO\textsubscript{3}-N.
Huffman and Revels (1998) surveyed 34 older (pre-1993 NRCS standards) lagoons in North Carolina for their impacts to shallow groundwater quality. The research found that two-thirds of the sites involved in that study showed seepage contamination exceeding drinking water standards 38-m down gradient. The researchers also found that among the sites where surface water was proximate to the lagoon, 50% showed nitrogen enrichment attributable to lagoon seepage. Similarly, a study by Westerman et al. (1995) found significant seepage from 3.5 to 5 year old unlined lagoons in sandy soil. The research found significant concentrations of NH$_4$-N (75-300 mg/L) and Cl (50-140mg/l) in shallow subsurface waters down gradient, indicating that natural sealing of the lagoon had not occurred. This evidence suggested that a properly constructed lagoon containing an impervious liner is required to prevent seepage from contaminating the surrounding water resources. In 1992, the NRCS established minimum design and construction standards for lagoon liners.

In North Carolina, threats from leaking lagoons to deep ground water quality are believed to be small due to the geology of the Coastal Plain (Huffman and Revels, 1998). In this region, a dense clay layer exists just below the surficial aquifer that forces most drainage water to travel laterally and discharge to surface water, as opposed to migrating downward to deeper groundwater contaminating drinking water sources. However, where the confining clay layer is non-continuous, downward movement of seepage water may occur. Although deeper groundwater may not be exposed to the elevated nutrient concentrations, surface water remains a concern for contamination, for the impacts on aquatic systems (Paerl, 1988) and for potential surface water intakes for municipal surface water supplies.
A common method for remediating a site with contaminated groundwater is a pump-and-treat regime (Mackay et al., 1989). In this method, polluted groundwater is pumped from the contaminated zone and treated in a manner that transforms the contaminant into a non-reactive or non-toxic form above ground. The pump-and-treat method employs two hydrologic techniques: removing and immobilizing the contaminant. Ideally, the pumping system will create a hydraulic gradient toward the pumping well such that the contaminant plume flows toward the well, reducing pollution further down gradient. Pump-and-treat methods have received criticism since this method of remediation is not capable of completely cleaning-up a contaminated site (Langwaldt et al., 2000). Although pump-and-treat systems are capable of draining the large pores containing contamination, these systems do not generate a large enough gradient to drain the small pores that contain contamination.

Contaminant adsorption to soil particles reduces the efficacy of the pump-and-treat systems. Mackay and Cherry (1989) reported that pump-and-treat systems were inefficient when dealing with non-aqueous phase liquid contaminants (NAPL) due to slow desorption, diffusion and dissolution processes. Borden and Kao (1992) observed that pumping of large volumes of water were required to meet cleanup criteria, resulting in extended pumping times and increasing costs of site remediation. Overall, the literature indicates that a well evaluated site with a properly designed pumping system can effectively remediate groundwater contamination by removing large volumes of pollutant while immobilizing contaminant migration (Langwaldt et al., 2000, Mackay et al., 2000).

Computer based models have been developed to aid in the understanding and evaluation of many environmental and agricultural based problems (AGNPS, Young et
Models allow researchers to simulate hydrology and contaminant transport associated with a variety of agricultural NPS. Modeling can save time in site evaluation as direct on-site measurements may take many years. Although the computing power has increased exponentially in recent years, computer modelers still require the collection of on-site data, calibration, and verification of simulated data. In recent years, a number of models have been developed to simulate groundwater hydrology and contaminant transport. MODFLOW is a modular three-dimensional finite-difference groundwater flow model that was developed by McDonald and Harbaugh (1988). MODFLOW is a physically based model that applies Darcy’s law for the movement of fluids in saturated porous media and allows for a wide array of inputs utilizing multiple layers on irregularly shaped regimes. In a study by Reeve et al. (2001), MODFLOW was used to evaluate the effect of regional groundwater flow on the water budget of the peatlands in Northern Minnesota. Lasserre et al. (1999) used MODFLOW to validate simulated results from a GIS integrated transport model combined AgriFlux, and found good agreement of both hydrologic and transport processes.

The study presented here utilizes MODFLOW to evaluate the pumping rates needed to prevent the seepage plume from a closed out lagoon from migrating off-site. The study also determined pumping rates required to collect seepage fluxes from an active lagoon as a function of drainage outlet level.
MATERIALS AND METHODS

Site Description

The study site involved a lagoon located in the Middle Coastal Plain of North Carolina. Huffman and Revels. (1998) found nutrient enrichment in groundwater 38-m down gradient of the swine lagoon and concluded that the lagoon was leaking. Data from bi-weekly samples obtained from a nearby channelized stream (Figure 1) indicated that ammonium-nitrogen (NH$_4$-N) concentrations ranged from 10 to 25 mg/L down gradient from the lagoon, while upstream concentrations were near detection levels (<0.1-mg/L). Groundwater monitoring wells were installed around the periphery and down gradient of the lagoon along the channelized stream in February 1999 to evaluate water level gradients and locate nutrient concentrations associated with the seepage plume, Figure 2. Data from the wells indicated that concentrations of NH$_4$-N along the periphery of the lagoon ranged from 100- 200 mg/L in wells (M1-M4), as seen in Figure 3. Water quality results were presented and summarized in Chapter II.

Since monitoring of the surface and ground water quality hog production has terminated, and therefore waste deposition into the lagoon has also ceased. In March 2001, the lagoon was closed-out, which involved the mechanical removal and land application of the nutrient rich sludge from the lagoon bottom.

Water Quality

Initial monitoring wells were installed to a depth of 1.5-m (screened 0.9-m-1.5-m) and 3-m (screened from 2.4 to 3-m). Groundwater wells were sampled monthly or bi-weekly to monitor and evaluate nutrient concentrations in the shallow groundwater. Water quality analysis protocols are described in detail in Chapter 2. All water quality
data samples described here were taken from a depth of 3-m, since the previous water quality analysis of the shallow wells found no indications of seepage. NH$_4$-N is typically the nutrient in the highest concentration in lagoons and therefore a strong indicator of seepage. Groundwater samples from deep wells further down stream from the lagoon showed no significant NH$_4$-N concentrations. Therefore, it was concluded that the seepage water was traveling along the shortest path from the lagoon to the channelized stream.

**Hydrology**

A detailed topographic survey was conducted to determine land surface elevations and discharge areas such as the nearby-channelized stream and the Neuse River, Figure 4. The free water surface elevation with respect to the channelized streams was important, as the streams provide an outlet for shallow subsurface drainage (in this case seepage water) from the agricultural fields. A water control structure was installed in the stream to manage stream water level. By raising the water level in the stream, the hydraulic gradient from the closed out lagoon to the stream can be decreased reducing the movement of the lagoon seepage plume. The water level in the stream was recorded at 4-hr intervals with a Blue Earth® data logger.

Five pumping wells were installed along the down gradient periphery of the lagoon in May 2000, centered on the area where elevated concentrations of NH$_4$-N occurred in the ground water. These wells were used to intercept the seepage water before being discharged to the stream. The pumping wells were 19-mm in diameter and placed inside a 50-mm diameter casing, both of which were schedule 40 PVC pipe. All wells were spaced 15.3-m apart. A drill rig with hollow stem auger was used to install
the wells to 6-m. All wells were screened 3 to 6-m using 0.4 mm well screen with a nylon
drain sock fitted around the screened portion of the well in order to prevent sediments
from filling in the well screen inhibiting flow to the well. The screened portion of the
well was back-filled with coarse sand followed by a 1.5-m section of bentonite. The
section of bentonite was used to prevent preferential flow from the surface from entering
the well. The last 0.5-m portion of the well was back-filled with sand and the land surface
surrounding the well was covered with a 0.15-m by 1-m by 1-m concrete slab.

Twelve monitoring wells (D1-D2, F1-F3, L1-LW and P1 in Figure 2) were
installed in May 2000 to continuously monitor water levels and determine hydraulic
gradient. Seven of these wells consisted of 76-mm diameter schedule 40 PVC pipe and
were installed to a depth of 3-m, screened from 1.5 to 3-m with 4.8-mm hand-drilled
holes. All wells were surveyed relative to a local benchmark, which provided a reference
elevation for water-level determination. Each well was installed with a pressure
transducer and Blue Earth® data logger. The transducer measured voltage changes that
are proportional to changes in water level with measurements recorded on an hourly
basis. Voltage reading was calibrated to water level in the laboratory prior to field
installation. Prior to each download, water level was measured manually using a slope
instrument. This measurement was recorded for use in correction of stray readings
recorded by the transducer. Downloading of data from the logger was performed on a bi-
weekly basis using a palmtop computer; data files were then uploaded to a personal
computer for analysis. In addition to continuous measurement, water levels in all wells
were measured manually on a bi-weekly basis.
During the installation of the wells, general observations were made with regard to soil type and depth of the impermeable layer. From this information, it was determined that the soil was typically sandy throughout the profile, with the impermeable layer 6-m below the soil surface. This information was corroborated by unpublished data from the Huffman and Revels (1998) study.

**Hydraulic Conductivity**

Five of the twelve-transducer wells (F1, LW, F3, P1, and L2 as seen in Figure 2) were installed so that saturated hydraulic conductivity measurements could be made. These wells were made of 76-cm diameter schedule 40 PVC and installed to a depth of 3-m. Each well was screened from 1.5 to 3-m using 0.4-mm well screen. The screened section of the well was fitted with a nylon drain sock to prevent coarse soil particles from plugging the well screen. In July 2001, slug tests were conducted on these wells following the criteria outlined by Bouwer and Rice (1976) using an update by Bouwer (1989). During the slug test, hydraulic head was measured with respect to time using a Global Water® pressure transducer and laptop computer.

**Modeling**

MODFLOW is a modular three-dimensional finite-difference groundwater flow model that was developed by McDonald and Harbaugh (1984). MODFLOW is a physically based model that combines Darcy’s law for the movement of fluids in saturated porous media and allows for a wide range of inputs utilizing multiple layers on irregularly shaped regimes. Simulations can be carried out in steady and unsteady state. By providing initial conditions and hydraulic properties, the model generates head values for each cell in the finite element grid. The Department of Defense Groundwater
Modeling System (GMS) is a pre- and post-processor used to perform simulations with MODFLOW. GMS provides a graphical user interface for site characterization, model conceptualization, grid generation, geostatistics and post-data processing. Data input files for MODFLOW are entered into GMS. Input files are then read and executed by MODFLOW, and the output generated from the MODFLOW simulations is then imported into GMS for post-processing. MODFLOW utilizes a site map to develop a conceptual model for analysis. The conceptual model utilizes Geographical Information System (GIS) tools to develop a representative model for the site in question. The locations of sources and sinks, layer parameters, site boundaries and all other data necessary for simulation can be defined at the conceptual model level. After completing the conceptual model, a grid model is generated with all cell-by-cell assignments performed automatically.

An aerial photograph of the site was obtained from the United States Geological Survey (USGS) website (Figure 4). This image includes the channelized streams as well as the Neuse River, both of which serve as groundwater flow boundaries. The image was imported into MODFLOW-GMS and registered. The registration process involves entering three known World coordinates from the site. The known World coordinates were determined using a Global Positioning System (GPS) unit from the Biological and Agricultural Engineering Department at North Carolina State University. The known coordinates were used by the model to properly scale the research site for further calculations and simulations. In addition, the base map registration process establishes proper real World coordinates for the entire site. The GPS unit was also used to locate all wells. MODFLOW was calibrated by comparing and fitting simulated hydraulic heads to
field observations. The calibration process was used to fine-tune the model simulations and to better-fit simulated heads to observed heads from wells in the field.

Simulations in MODFLOW-GMS were performed using field data as well as applying values from the literature. The average value from the five hydraulic conductivity measurement was used as a starting value for horizontal hydraulic conductivity ($K_h$). Since no vertical hydraulic conductivity ($K_v$) was measured, a value of one-half the $K_h$ was used. Typical values for a medium sand were taken from the literature for specific yield ($S_y$) and porosity ($\Theta$). An initial groundwater recharge value of 0.3-m/yr (0.0027 ft/day) was chosen and used as a starting point in the calibration process. This value corresponds to a typical value of excess rainfall in the North Carolina Coastal Plain. The initial simulations utilized the water control structure in the stream adjacent to the former lagoon (elevation 29.4-m) and the Neuse River (elevation 24.7-m) as constant heads. The only steady state period occurred after the lagoon was closed out with the pumping system operated at 3.5-gpm, and the water control structure at 29.4-m.

Figure 5 displays the MODFLOW conceptual model base map for this situation. In this figure, the solid red lines at opposite ends of the model indicate the constant head boundaries of the stream (on the right) and the Neuse River (left). The dark area in the middle of the figure displays the grid for the conceptual model. Grid spacing was every 6-m, with a refined grid near the pumping wells spaced at 3-m. The five blue and yellow symbols near the channelized stream represent the exact locations of the pumping wells. MODFLOW assumes that the bottom layer is impermeable, so only one layer was utilized for the simulations. Data from the site indicated a fairly consistent coarse sand with the impermeable layer, a dense marine clay, situated 6-m from the surface.
Contaminant transport was simulated in MODFLOW using MT3DMS. The hydraulic head output generated in MODFLOW is used by MT3DMS along with additional parameters specific to the contaminant (i.e., seepage concentration and dispersivity) to generate solutions for contaminant migration. The location of nutrients associated with lagoon seepage was determined by using MODFLOW-MT3DMS. A typical NH$_4$-N concentration for lagoons, 600-mg/L, was used. The seepage rates were varied from 2.3-m$^3$/day (the calibrated groundwater recharge rate) to a maximum of 13 m$^3$/day, which simulated a worst-case scenario, where all liquid waste deposited in the lagoon was lost to seepage.

The model was used to evaluate the groundwater discharge and seepage plume migration to the channelized stream. Model simulations were conducted for two steady state stream water elevations. The water level in the stream was controlled by manipulating the number of boards in the water control structure; adding boards to the water control structure raises the water level in the stream, while removing boards lowers the water level in the stream.

**Pumping and Plume Capture**

MODFLOW was used to evaluate pumping rates needed to reverse the gradient away from the stream. This scenario assumed equivalent drawdowns in each pumping well, which results in different yields from each well. This approach was used as the pumping system in the field utilized a manifold, where the pumping wells are connected in series to a jet pump. Since the pump draws an equal pressure across the manifold, steady state pumping yields equivalent drawdowns. Iterative simulations were completed until the desired drawdown was found. The desired drawdown is one that is low enough
to create a gradient from the stream to the pumping system, while not pumping water from the stream. Following each iteration, flux to the pumping well and flux from the stream were read from the MODFLOW base map. MODFLOW simulations continued until the highest pumping rate that did not create a flux from the stream was determined. The resulting pumping rates for free and controlled drainage scenarios were compared.

**Simulated Lagoon**

The unlined lagoon was constructed over 30 years ago by excavating soil to the depth of the seasonal high water table, which resulted in a lagoon depth of 2-m. The design of the lagoon at that time was for 0.6-m of storage for 90-days, and the lagoon received 4800-m$^3$ of wastewater per year. From discussions with the systems operator, it was determined that significant pumping from the lagoon was rarely a necessity, even after heavy rains. Using this information, simulations were conducted to determine the effects of the former lagoon on shallow groundwater flow and quality.

MODFLOW simulations were also conducted to determine the pumping rate needed to capture the lagoon seepage assuming that the lagoon was active. These simulations were conducted using an interceptor drain (French drain) to collect the lagoon seepage. The drain was placed just down gradient from the lagoon near the toe of the lagoon berm. The drain was placed at the same depth as the drainage outlet elevation. Placing the drain at a greater distance down gradient from the lagoon would be inefficient as the drainage system would collect additional drainage water from groundwater recharge (rainfall) not associated with seepage. To evaluate the effect of flow rates to the drain, the model was run under both free and controlled drainage conditions.
Statistical Methods

Statistical methods were used to determine the errors associated with model calibration. The model was calibrated in an iterative process by adjusting groundwater recharge and horizontal hydraulic conductivity values until simulated hydraulic heads in monitoring wells were minimized and within +/- one standard error interval of observed field data. The standard error interval is an interval about the average observed hydraulic head plus or minus one standard error. To evaluate model precision, average error was determined by summing all of the errors from the observed and simulated values, and dividing by the number of wells used in the calibration process. Another parameter that estimates model precision is average absolute error, which is the sum of the absolute values of the difference between simulated and observed heads divided by the number of wells used in the calibration process.

RESULTS

Hydraulic Conductivity

During the installation of the wells, it was determined that the soil at the site was fairly homogeneous. Table 1 presents hydraulic conductivity values calculated from the slug tests data collected in the field. Values ranged from 27 to 51 cm/hr and averaged 43 cm/hr. Throughout the installation of the wells, observations were made to evaluate variations in the soil profile. Although no soil pits were dug and evaluated, all observations of the soil cores indicated that the sandy soil was consistent throughout the site. From the analysis of the hydraulic conductivity, plots were made comparing the water level with respect to time and all plots produced smooth curves indicating that the soil drained as a single layer.
Model Inputs and Calibration

Since the focus of this analysis is on the area of the former lagoon and channelized stream, model calibration focused on the nine wells installed in this area. Data from the calibration for observed and predicted hydraulic heads are summarized in Table 2. The average error for the model was –1.5-cm and indicates that the model on average underestimated head in the well by 1.5-cm. However, this parameter can be somewhat misleading, as overestimates and underestimates can cancel each other out. The absolute average error was determined to be 2.4-cm. It can be observed from the data in Table 2 that the model underestimated heads near the stream and overestimates water levels further up gradient. A summary of calibrated model inputs are displayed in Table 3.

Hydraulic Gradient Evaluation

The hydraulic gradient of the site prior to installation of the water control structure was determined utilizing both survey information and water level observations made from monitoring wells. Average water level in monitoring wells is displayed in Figure 6. Applying data from the survey, it was determined that the stream bottom elevation was 28.70-m, and the average observed head in monitoring wells 1-4 (M1-M4 in Figure 5) was 29.54-m. This data indicates that the subsurface water traveled from the area of the closed-out lagoon to the channelized stream. In order to prevent the seepage plume from discharging to the drainage stream, the pumping system must lower water levels in the wells below the observed hydraulic head in the stream, making the series of pumping wells the area with the lowest hydraulic head, and a sink for the subsurface flow and seepage plume.
Hydraulic gradient determinations were made to evaluate the effect of the water control structure by maintaining the water level in the stream at 29.42-m. Several scenarios were evaluated by changing the water level in the stream, without the pumping system operating. The simulation results indicated that under controlled drainage (water control structure at 29.42-m) the gradient from the closed-out lagoon area was 0.0027-m/m, while under free drainage the gradient was 0.0045-m/m. Applying the calibrated hydraulic conductivity and soil porosity values to the gradient determinations, flux and travel time from the closed out lagoon (205-ft or 62.5-m) to the stream were calculated and are displayed in Table 4. Travel time was calculated by dividing the groundwater flux by the soil porosity to obtain seepage water velocity. Under controlled drainage, the flux and travel time from the closed out lagoon were determined to be 0.034 m/day and 640-days, respectively while under free drainage these values were 0.057 m/day and 380-days.

**Plume Capture Scenario**

Analyses were conducted to determine if the monitoring wells gave an accurate description of the location of the seepage plume. Modeling scenarios in MT3DMS, the contaminant transport package associated with GMS, confirmed that the seepage fluxes from the lagoon travel directly toward the channelized stream, as shown in Figure 7 (a close-up is displayed in Figure 8). The contour lines indicate lines of equal concentration; it can be observed that the concentration gradient is from the lagoon to the stream. In this case, the simulation was based on advection, which is the dominant mechanism of contaminant transport. Results indicated that the groundwater plume did not migrate any further down the stream than monitoring well 5 (M5 in Figure 2). The remaining
MODFLOW simulations evaluated the flux to the stream from the point where the stream starts up gradient to monitoring well 5 (M5). The simulated flow to the stream (as far as M5) in the controlled drainage case was 38.5-m$^3$/day, compared to the free drainage case, which was 53.8-m$^3$/day. Well concentrations indicated that the seepage plume exists in the area between the former lagoon and the drainage stream, with the highest concentrations centered where the pumping wells were installed. Average nutrient concentrations from all monitoring wells are shown in Figure 8. These data show that wells proximate to the stream (D1 and D2) had an average NH$_4$-N concentration of 95-mg/L.

Pumping rates should be high enough to capture contaminants near the stream, creating a gradient toward the pumping system, and away from the stream. To achieve this, drawdown in each of the pumping wells must be lower than the constant head in the stream, but not so low that water from the stream flows to the pumping system, which would result in excessive pumping. Following these criteria, simulations were conducted under controlled drainage and free drainage for pumping rate determination. Since the field system utilizes a manifold connection for each well, uniform drawdown should be observed at steady state. Results from these simulations show that for the controlled drainage case, 4.7-gpm was required to reverse the gradient to the ditch and 6.3-gpm was needed in the free drainage mode. In the case where the water control structure was used, pumping rates for each individual well ranged from 1.27 to 0.77-gpm with an average drawdown of 0.11-m. In the free drainage case, pumping rates for each well ranged from 1.79 to 1.04-gpm with an average drawdown of 0.18-m (Table 5).
Simulated Lagoon

Since the previous analysis considered the case with a closed-out lagoon, an alternative option was explored to collect seepage fluxes from an active lagoon. A French drain was installed down gradient of the lagoon to determine pumping rates necessary to mitigate seepage discharges to the stream. Simulations were conducted to determine the optimum drain depth for seepage collection based on varied rates of seepage. Additional evaluation of this data was conducted to determine pumping rates needed to capture the entire lagoon flux. This system needed to collect not only seepage from the lagoon, but drainage water from up gradient. Ideally, the drainage system would collect only the seepage fluxes, but considering the layout of the site, the drain must be situated down gradient from the lagoon, collecting all of the drainage water from up gradient.

For effective collection of the lagoon seepage, the drain had to be placed at the same elevation as the water level in the stream, 29.42-m for the controlled drainage case, and 28.66-m for the free drainage case. Installing the drain deeper collected additional drainage from down gradient, while installing the drain more shallow did not collect the entire seepage flux from the lagoon. The placement of the drain at the proper depth created a no flow boundary, capturing all seepage fluxes from the lagoon, and also collecting drainage water from up gradient, thus preventing any plume migration to the stream. With a no-flow boundary at the drain, 38.5-m$^3$/day (7.05-gpm) of total drainage was captured under controlled drainage conditions, and 54.4 m$^3$/day (9.96-gpm) under free drainage conditions with an active lagoon leaking at the calibrated model groundwater recharge rate. Assuming that all the waste generated (4800-m$^3$/year) was lost (since the lagoon was constructed without a liner) 51.5-m$^3$/day (9.43-gpm) was
collected for the controlled drainage case, while using free drainage 67.4-m³/day (12.3-gpm) would be collected by the drain line. This system completely captured all lagoon seepage, and mitigated lagoon discharge to the drainage stream. These data again indicated the significance of the use of the water control structure.

**SUMMARY AND CONCLUSIONS**

Seepage from a leaking swine lagoon was evaluated using MODFLOW-GMS. MODFLOW is a physically based model that applies Darcy’s law for the movement of fluids in saturated porous media and allows for a wide range of inputs utilizing multiple layers on irregularly shaped regimes. Data collected from the field was used to calibrate the model. Hydraulic conductivity and groundwater recharge values were varied until the errors were minimized when comparing simulated heads to observed heads from nine monitoring wells in the field. Nine wells were chosen as calibration wells since they were located in the portion of the field that was most affected by seepage migration.

A water control structure was installed in the channelized stream to lower the discharge gradient from the lagoon to the stream. Following calibration, MODFLOW was used to evaluate plume migration with and without the water control structure. Raising the water level in the stream decreased the hydraulic gradient from the former lagoon, from the free drainage case of 0.0045 to 0.0027 in the controlled drainage case. From the gradient calculations, travel time to the stream increased from 380-days free drainage scenario to 640-days in the controlled drainage case.

MODFLOW was used to determine the pumping rates needed to reverse the gradient to the stream, and capture the seepage plume. For the free drainage case, 6.3-gpm was needed to create a gradient towards the pumping system, while 4.7-gpm was
required for the controlled drainage case. The simulation results described here assumed that the pumping wells had equivalent drawdown. This assumption was chosen since the field pumping system utilized a manifold, which would result in equivalent drawdowns at steady state.

Several simulations were conducted in MODFLOW to evaluate collection of seepage from an active lagoon. A French drain was inserted in the model just downstream from the lagoon at the toe of the berm. The model was used to determine drainage collection rates from this system under the free and controlled drainage criteria. The French drain captured all the fluxes from the lagoon. The controlled drainage scenario decreased the amount of water collected by the French drain from 67.4-m$^3$/day in the free drainage case to 51.5-m$^3$/day for the controlled drainage case. For this application, it appears the water level in the stream could be increased further reducing the groundwater gradient to the stream. However the maximum practical height of the water level in the stream has been achieved. Low-lying farmland up steam from the water control structure adjacent to the drainage stream has experienced saturation during periods of normal rainfall, consequently restricting the height of the boards in the outlet structure. Despite the benefits of the controlled drainage system, these pumping rates are too high for adequate treatment of the contaminated groundwater with the current constructed wetland (Chapter 3). As an alternative, if the lagoon was still in production, drainage water collected by this system could be re-routed to the lagoon with a portion of the flow routed to the constructed wetland for nutrient assimilation.
Table 1. Field saturated hydraulic conductivity from slug test evaluation.

<table>
<thead>
<tr>
<th>Location</th>
<th>Water Table Depth (cm)</th>
<th>Hydraulic Conductivity (cm/hr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>F1</td>
<td>183</td>
<td>51</td>
</tr>
<tr>
<td>F3</td>
<td>66</td>
<td>45</td>
</tr>
<tr>
<td>L2</td>
<td>94</td>
<td>43</td>
</tr>
<tr>
<td>LW</td>
<td>138</td>
<td>27</td>
</tr>
<tr>
<td>P1</td>
<td>61</td>
<td>48</td>
</tr>
</tbody>
</table>

Table 2. Comparison of observed head vs. simulated heads for MODFLOW calibration.

<table>
<thead>
<tr>
<th>Well</th>
<th>Observed (m)</th>
<th>Simulated (m)</th>
<th>Upper Limit (m)</th>
<th>Lower Limit (m)</th>
<th>Error (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>L1</td>
<td>29.55</td>
<td>29.54</td>
<td>29.72</td>
<td>29.37</td>
<td>-0.01</td>
</tr>
<tr>
<td>L2</td>
<td>29.54</td>
<td>29.52</td>
<td>29.71</td>
<td>29.38</td>
<td>-0.02</td>
</tr>
<tr>
<td>L3</td>
<td>29.52</td>
<td>29.51</td>
<td>29.66</td>
<td>29.38</td>
<td>-0.01</td>
</tr>
<tr>
<td>D1</td>
<td>29.50</td>
<td>29.43</td>
<td>29.63</td>
<td>29.36</td>
<td>-0.07</td>
</tr>
<tr>
<td>D2</td>
<td>29.48</td>
<td>29.43</td>
<td>29.60</td>
<td>29.36</td>
<td>-0.05</td>
</tr>
<tr>
<td>D3</td>
<td>29.48</td>
<td>29.47</td>
<td>29.57</td>
<td>29.39</td>
<td>-0.01</td>
</tr>
<tr>
<td>F1</td>
<td>29.57</td>
<td>29.58</td>
<td>29.24</td>
<td>29.40</td>
<td>+0.01</td>
</tr>
<tr>
<td>F2</td>
<td>29.59</td>
<td>29.60</td>
<td>29.77</td>
<td>29.40</td>
<td>+0.01</td>
</tr>
<tr>
<td>LW</td>
<td>29.52</td>
<td>29.53</td>
<td>29.65</td>
<td>29.39</td>
<td>+0.01</td>
</tr>
</tbody>
</table>

*Average error = -0.015-m, Average absolute error = 0.024-m, Root MSE = 0.30
Table 3. Calibrated model inputs used in MODFLOW.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Horizontal Hydraulic Conductivity ($K_h$) (cm/hr)</td>
<td>53</td>
</tr>
<tr>
<td>Vertical Hydraulic Conductivity ($K_v$) (cm/hr)</td>
<td>27</td>
</tr>
<tr>
<td>Specific Yield ($S_y$)</td>
<td>0.15</td>
</tr>
<tr>
<td>Porosity ($\Theta$)</td>
<td>0.35</td>
</tr>
<tr>
<td>Groundwater Recharge (m/day)</td>
<td>0.0006</td>
</tr>
</tbody>
</table>

Table 4. Summary of gradient data with respect to outlet elevation.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Controlled Drainage (96.5-ft, 29.4-m)</th>
<th>Free Drainage (94.0-ft, 28.7-m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gradient (m/m)</td>
<td>0.0027</td>
<td>0.0045</td>
</tr>
<tr>
<td>Flux (m/day)</td>
<td>0.033</td>
<td>0.057</td>
</tr>
<tr>
<td>Travel Time (days)</td>
<td>640</td>
<td>380</td>
</tr>
</tbody>
</table>

Table 5. Pumping rates needed from each pumping well to reverse the gradient from the stream.

<table>
<thead>
<tr>
<th>Pumping Well</th>
<th>Controlled Drainage Scenario (gpm)</th>
<th>Controlled Drainage Drawdown (m)</th>
<th>Free Drainage Scenario (gpm)</th>
<th>Free Drainage Drawdown (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Well 1</td>
<td>1.27</td>
<td>0.12</td>
<td>1.79</td>
<td>0.20</td>
</tr>
<tr>
<td>Well 2</td>
<td>0.88</td>
<td>0.12</td>
<td>1.21</td>
<td>0.19</td>
</tr>
<tr>
<td>Well 3</td>
<td>0.77</td>
<td>0.11</td>
<td>1.04</td>
<td>0.18</td>
</tr>
<tr>
<td>Well 4</td>
<td>0.79</td>
<td>0.11</td>
<td>1.05</td>
<td>0.17</td>
</tr>
<tr>
<td>Well 5</td>
<td>0.95</td>
<td>0.10</td>
<td>1.24</td>
<td>0.16</td>
</tr>
</tbody>
</table>
Figure 1. Plan view of the research site.

Figure 2. Schematic of monitoring well locations.
Figure 3. NH$_4$-N concentrations of monitoring wells adjacent the former lagoon (wells M1-M4).

Figure 4. Aerial photograph indicating groundwater flow boundaries.
Figure 5. Conceptual model base map used in MODFLOW-GMS.

Figure 6. Average observed water levels from wells at the research site.
Figure 7. Schematic displaying flux from the lagoon.

Figure 8. Close-up of the flux from the simulated lagoon.
Figure 8. Average NH$_4$-N concentrations from wells at the research site.
REFERENCES


APPENDIX A: Manual Water Level Measurements
Figure A1. Groundwater elevation at monitoring well 1 (M1).

Figure A2. Groundwater elevation at monitoring well 2 (M2).
Figure A3. Groundwater elevation at monitoring well 3 (M3).

Figure A4. Groundwater elevation at monitoring well 4 (M4).
Figure A5. Groundwater elevation at monitoring well 5 (M5).

Figure A6. Groundwater elevation at monitoring well 6 (M6).
Figure A7. Groundwater elevation at monitoring well 7 (M7).

Figure A8. Groundwater elevation at monitoring well 9 (M9).
Figure A9. Groundwater elevation at monitoring well 10 (M10).

Figure A10. Groundwater elevation at monitoring well 11 (M11).
Figure A11. Groundwater elevation at transducer well D1.

Figure A12. Groundwater elevation at transducer well D2.
Figure A13. Groundwater elevation at transducer well D3.

Figure A14. Groundwater elevation at transducer well DW.

138
Figure A15. Groundwater elevation at transducer well L1.

Figure A16. Groundwater elevation at transducer well L2.
Figure A17. Groundwater elevation at transducer well L3.

Figure A18. Groundwater elevation at transducer well LW.
Figure A.19. Groundwater elevation at transducer well F1.

Figure A20. Groundwater elevation at transducer well F2.
Figure A21. Groundwater elevation at transducer well F3.

Figure A22. Groundwater elevation at transducer well P1.
APPENDIX B: Continuous Water Table Measurements
Figure B1. Continuous water level data from transducer D1 and D2.

Figure B2. Continuous water level data from transducer D3.
Figure B3. Continuous water level data from transducer DW.

Figure B4. Continuous water level data from transducers F1 and F2.
Figure B5. Continuous water level data from transducer F3.

Figure B6. Continuous water level data from transducer L1.
Figure B7. Continuous water level data from transducer L2.

Figure B8. Continuous water level data from transducer L3.
Figure B9. Continuous water level data from transducer LW.

Figure B10. Continuous water level data from transducer P1.
Figure B11. Continuous water level data from transducer on the ditch structure (DS).

Figure B12. Constructed wetland outlet stage data.
Figure B13. Average daily head (cm) on inflow weir.

Figure B14. Daily rainfall measurements from the study site.
APPENDIX C: Concentration Data
Figure C1. Nutrient concentrations from monitoring well M1.

Figure C2. Nutrient concentrations from monitoring well M2.
Figure C3. Nutrient concentrations from monitoring well M3.

Figure C4. Nutrient concentrations from monitoring well M4.
Figure C5. Nutrient concentrations from monitoring well M5.

Figure C6. Nutrient concentrations from monitoring well M6.
Figure C7. Nutrient concentrations from monitoring well M7.

Figure C8. Nutrient concentrations from monitoring well M9.
Figure C9. Nutrient concentrations from monitoring well M10.

Figure C10. Nutrient concentrations from monitoring well M11.
Figure C11. Nutrient concentrations from transducer well D1.

Figure C12. Nutrient concentrations from transducer well D2.
Figure C13. Nutrient concentrations from transducer well D3.

Figure C14. Nutrient concentrations from transducer well DW.
Figure C15. Nutrient concentrations from transducer well F1.

Figure C16. Nutrient concentrations from transducer well F2.
Figure C17. Nutrient concentrations from transducer well F3.

Figure C18. Nutrient concentrations from transducer well L1.
Figure C19. Nutrient concentrations from transducer well L2.

Figure C20. Nutrient concentrations from transducer well L3.
Figure C21. Nutrient concentrations from transducer well LW.

Figure C22. Nutrient concentrations from transducer well P1.
Figure C23. NH$_4$-N concentrations in the stream adjacent the ditch.
APPENDIX D: Pictures
Figure D1. Site map and close-up aerial photograph of the research site.

Figure D2. Schematic of final research site layout.
Figure D3. Picture of Wetland Construction.

Figure D4. Picture of wetland construction with sinuous curve.
Figure D5. Photo of clay liner compaction.

Figure D6. Photo of wetland construction and inactive lagoon.
Figure D7. Picture of the close-out/filling of lagoon.

Figure D8. Picture of wetland out prior to installation of water control structure.
Figure D9. Picture of wetland outlet with flow.

Figure D10. Wetland inflow quantification and overland distribution.
Figure D11. Photo of Wetland bulkhead structure.

Figure D12. Picture of bulkhead structure under flow conditions.
Figure D13. Photo of pump house and pumping wells.

Figure D14. Picture of individual pumping well.
Figure D 15. Photo of jump used to route groundwater to wetland.

Figure D16. Picture of soil up gradient form the lagoon (left) and soil from down gradient from the lagoon (right).
Figure D17. Bulkhead sampling location for Redox, Dissolved Oxygen, and Temperature.

Figure D18. Photo of planting needle rush on a 3-ft grid.
Figure D19. Picture of needle rush seeding out in the summer.

Figure D20. Photo of summer pickeral weed stand.
Figure D21. Photo of spring growth of Lizard’s Tail.

Figure D22. Picture of stand of Giant Soft-Stemmed Bulrush.
Figure D23. Photo of Cattails stand with Pickeral Weed and Arrowhead.

Figure D24. Picture of Spatterdock and Cattail stand.
Figure D25. Photo of Fall Spatterdock growth.

Figure D26. Photo of muskrat hole on island.
Figure D27. Photo of newly planted Cyprus tree.

Figure D 28. Photo of the lower portion of the wetland.
Figure D29. Picture of the upper portion of the wetland, toward the inlet.

Figure D30. Picture of bulkhead and wetland island.