SEASONAL DYNAMICS OF NUTRIENTS AND AQUATIC BIOTA IN TWO WETLANDS NEAR A SWINE CONFINED ANIMAL FEEDING OPERATION

By

WILLIAM J. LINTS

Associate of Science
Mohawk Valley Community College
Utica, New York
1995

Bachelor of Science
Oral Roberts University
Tulsa, Oklahoma
1999

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Oklahoma State University in partial fulfillment of the
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Thesis Approved:

William R. Fisher

Thesis Advisor

Dean of the Graduate College

PREFACE

This study was conducted to determine potential differences between a wetland receiving swine waste run-off and an adjacent wetland assumed to be free of swine waste. I hypothesized that there would be no differences in the biological, chemical, and physical properties between the two water bodies. This hypothesis was tested by examining nutrient limitation of periphyton and seasonal dynamics of the biological and abiotic factors of each wetland during a two year period (2000-2001). Findings from the study were designed to provide government wildlife agencies with information on the ecological condition of these systems in order to make sound assessments regarding the potential future of both wetlands. Additionally, this study may contribute new information on the seasonal dynamics in wetlands of the Southern Great Plains.

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INTRODUCTION

Public appreciation for the ecological, social, and economic values of wetlands in North America has increased greatly in recent years. Protection and preservation of these invaluable resources are quickly becoming a priority to Federal, State, and local governments. Yet, each year 23,674 hectares are lost due to the direct effects of anthropogenic activities (Dahl 2000). Of these, the potential affects of agriculture, particularly from livestock operations, on wetland ecosystems has become a concern.

It is estimated that 450,000 animal feeding operations exist in the United States, 6,500 of which are confined animal feeding operations that contain more than 300 animal units (USDA and USEPA 1999). Each of these facilities has the potential to introduce large concentrations of animal waste, particularly nitrogen and phosphorus, into the environment (Ham et al. 2000). Therefore, understanding the potential effects of these facilities on wetland environments is important for their protection.

The goal of this study was to ascertain the potential differences between a wetland directly receiving swine waste run-off and a nearby second wetland presumably free of swine waste run-off. I examined nutrient dynamics and limitation of periphyton as well as chemical and biological dynamics within each wetland. My findings will assist governmental agencies in conserving these vital aquatic resources. The research is presented as one paper formatted for submission to the journal Wetlands.

LITERATURE CITED

Dahl, T. E. 2000. Status and trends of wetlands in the conterminous United States 1986 to 1997. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. 82 pp.

Ham, J. M., T. M. DeSutter, and C. W. Rice. 2000. Animal Waste Lagoon Water Quality Study. Research Report, Kansas State University, Manhattan.

USDA (U.S. Department of Agriculture) and USEPA (U.S. Environmental Protection Agency). 1999. Unified National Strategy for Animal Feeding Operations, March 9, 1999. U.S Department of Agriculture and U.S Environmental Protection Agency, Washington D.C., U.S.A.

CHAPTER II

SEASONAL DYNAMICS OF NUTRIENTS AND AQUATIC BIOTA IN TWO WETLANDS NEAR A SWINE CONFINED ANIMAL FEEDING OPERATION (CAFO)

Abstract: The objective of this study was to assess seasonal dynamics of nutrients, aquatic biota, and nutrient limitation between two Oklahoma wetlands, one potentially receiving nutrient pollution from a swine confined animal feeding operation (Sandy Bluff wetland), and a second wetland which was not (Bull Head wetland). Sandy Bluff wetland (SB) had significantly higher phosphorus, nitrogen, and chlorophyll a concentrations than Bull Head (BH) wetland, but both wetlands were co-limited by phosphorus and nitrogen. Sandy Bluff had a higher total standing crop of phytoplankton and BH wetland had a higher standing crop of zooplankton. Each wetland contained the same two dominant orders of zooplankton, but BH contained three dominant phytoplankton orders and SB contained two. Both wetlands contained three submerged macrophytes that differed in relative abundance. *Chara vulgaris* was more abundant in BH and *Potamogeton pectinatus* and *Ceratophyllum demersum* was more abundant in SB wetland. Both wetlands were characteristic of eutrophic conditions; however, run-off from swine waste may have been responsible for the eutrophic conditions of SB wetland, whereas diffuse run-off from cattle waste, sequestering and release of nutrients (particularly phosphorus) in submerged macrophytes during

periods of growth and senescence, and concentration of nutrients associated with droughts during 2000 and 2001 may have been responsible for the conditions in BH wetland.

Key Words: wetlands, CAFO, nutrient limitation, nutrient enrichment, seasonal dynamics, phytoplankton, zooplankton, and macrophytes

INTRODUCTION

There is increasing point and non-point source pollution from agriculture, including significant contributions from confined animal feeding operations (USDA and USEPA 1999, Woltemade 2000). Confined animal feeding operations (CAFOs) have become a concern to local, state, and federal agencies because they are considered a source of contamination in lakes and streams (Cross et al. 1970). Animal feeding operations introduce by-products of animal wastes such as nitrogen and phosphorus, trace elements, and organic material into aquatic systems via run-off, groundwater infiltration, land application of manure, or directly by the animals (Giniting et al.1998, Giusquiani et al. 1998, Ham and DeSutter 1999, Moffitt and Lander 1999). As a result, excessive amounts of nitrogen and phosphorus enter these environments and begin the process of nutrient enrichment (Wiess 1969). This can lead to eutrophication and substantial ecological changes in the aquatic system (Cairns et al. 1992; Cottingham et al. 1998, USDA 1999). Such changes in wetlands are shifts in phytoplankton, zooplankton, and macrophyte assemblages (Schwartz and

Gruendling 1985, Newman et al. 1997, Moore et al. 1999, Ortega-Mayagoitia et al. 2000, Dodson and Lillie 2001, McCormick et al. 2001).

Wetlands are capable of retaining varying loads of nitrogen and phosphorus inputs (Boustany et al. 1997, Kadlec 1999, Blahnik and Day 2000, White et al. 2000, Saunders and Kalff 2001). In some cases, as much as 43% of nitrogen and 68% of phosphorus can be removed from the water through the processes of sedimentation, filtration, adsorption, and microbial or vegetation uptake by wetland systems (Woltemade 2000). However, a continuous discharge of excess nutrients can saturate a wetland creating eutrophic conditions that lead to a decrease in species richness and an increase in relative biomass of nuisance flora and fauna (Schwartz and Gruendling 1985, Daoust and Childers 1999, Miao et al. 2000, Sand-Jensen et al 2000). These changes have been documented in eutrophic areas of the Florida Everglades where emergent plants such as *Typha* have increased in biomass, frequency, and rates of colonization. (McCormick 1996, Miao et al. 2001). Stewart et al. (1997) noted that the replacement of native *Cladium* by *Typha* in the Florida Everglades coincided with changes in Everglade phosphorus levels.

Dynamic changes in phytoplankton, periphyton assemblages, and zooplankton have also been observed in wetlands with increased nutrient concentrations (Cottingham et al. 1998, McCormick et al. 1998, Zohary et al. 1998, Havens et al. 1999). Chow et al. (1998) noted that in nutrient-poor wetlands, dominant taxa included a diverse assemblage of cholorphytes and diatoms and when conditions were nutrient rich, dominant taxa shifted to more

cyanophytes and few cholorphytes. Increased abundance of bacillariophytes, cyanophytes, euglenophytes, and other smaller phytoplankton results in more food resources for smaller zooplankton such as rotifers. In contrast, the presence of large-bodied cladocerans may decline as they become food limited. However, populations of cladocerans and copepods may rebound if water quality conditions improve, promoting the growth of new food resources such as chlorophytes and cryptophytes (Beaver et al. 1999, Ortega-Mayagoitia et al. 2000).

The effects of nutrient enrichment on aquatic systems can be predicted by using a well-suited bioindicator (Matlock et al. 1999; Toetz et al. 1999, Lemly and King 2000). One such bioindicator of eutrophication is the determination of nutrient limitation as manifested through periphyton growth (Mattila and Raisanen 1998, Matlock et al. 1999). Although, there are few studies of periphyton dynamics in wetlands as compared to those in other fresh waters, evidence shows that periphyton may significantly contribute to essential wetland functions and are accurate descriptors of phytoplankton responses to nutrient limitation (McCormick et al. 1998, Smoot et al. 1998). Additionally, there have been few experimental studies on the implications of nutrient enrichment on North American wetlands (Bedford et al. 1999). Nutrients (specifically nitrogen and phosphorus) which limit productivity of phytoplankton, periphyton, and rooted aquatic macrophytes in wetlands are not well defined (Bridgham et al. 1996). Therefore, determining the limiting nutrients within wetlands could be of particular use as these habitats continue to degrade in part from various forms of rural and

urban nutrient pollution (McCormick et al. 1996, Zohary et al. 1998, Sand-Jensen et al. 2000).

The objective of this study was to assess seasonal dynamics of nutrients, nutrient limitation, and aquatic biota in a wetland receiving confined animal feeding operation (Sandy Bluff wetland), and a second wetland which was not (Bull Head wetland). Data gathered during the study provided information to assess water quality, present ecological condition, and predict potential changes in each wetland from swine CAFO pollution.

METHODS

Study Site

Sandy Bluff (SB) and Bull Head (BH) are palustrine aquatic bed wetlands located in the Canton Wildlife Management Area (CWMA) in northwestern Oklahoma (Canton NW Quadrangle T20N, R14W) at 36° 09' 47" N, 98° 44' 13" W and 36° 09' 36" N, 98° 43' 57" W, respectively (Figure 1). These are two of several wetlands within a 2.60 km^2 area. Each wetland in the CWMA contains three distinct habitat types: persistent emergents (*Typha* stands), open water aquatic macrophyte beds (sloughs), and inundated woodland (emerged and submerged tree stumps).

Sandy Bluff wetland is a 36.5ha shallow, irregularly-shaped basin surrounded entirely by dense *Typha* stands extending approximately 10-25m from shore. Submerged macrophytes are distributed throughout the basin and water clarity is excellent. Bull Head wetland is a 7.04ha shallow, oblong basin

surrounded by small shrubs, oak, willow, and cottonwood trees. *Typha* is present only in small isolated stands along the west and northwest shores of the pond. Submerged macrophytes cover about 75% of the basin and water clarity is excellent. Sandy Bluff wetland occurs along a break in slope that intercepts the water table allowing groundwater to seep into it from higher terrace deposits. Bull Head wetland is a depressional wetland formed within deposits associated with an alluvium and terrace aquifer along the North Canadian River in Dewey County, Oklahoma.

Approximately 1km north of the wetlands and up slope is a 20,000-head feeder swine CAFO. Water from rain events of 7cm or more runs off the land application field and flows south across adjacent down-gradient cropland into SB wetland. Additionally, along the wetland's northern boundary are several ground water seeps containing high nitrate and ammonia concentrations (> 10 mg/L and > 2 mg/L respectively) which flow into the wetland year-round (US Fish and Wildlife Service unpublished). Bull Head wetland was designated as a control basin because it was to assumed to not receive run-off directly from the CAFO. A current study of drainage patterns research by the EPA is attempting to verify this assumption. In addition, a herd of cattle (25-30 head) has access to SB and a separate herd (25-30 head) has access to BH 6-8 months out of the year; during the remaining 4-6 months, the cattle are absent from both areas.

Physico-chemical measurements

A 2 m PVC pipe was placed in the sediment to gauge water level. The pipe was marked at increments of 0.1 m and placed 10 m from shore in standing

water. In addition, an extendable fiberglass depth meter was used to monitor water depth fluctuations at each station throughout the study. A benchmark to delineate the water boundary for SB was determined by measuring a perpendicular line to water's edge from a large dead tree west of the boat launch area. A benchmark for the water boundary at BH was measured by extending a perpendicular line from the base of a willow tree with twin trunks on the south side of the boat launch to the water's edge.

Dissolved oxygen, temperature, and specific conductivity were determined in the field with probes using a Hydrolab Surveyor 4 multiparameter water quality meter and Minisonde (®Hydrolab Corporation). Readings were taken between 10:00 and 11:30 am at BH and between 12:30 and 2:00 pm at SB. Data were recorded every 0.5 m until the probe reached the bottom sediment. Dissolved oxygen and specific conductivity probes were calibrated at the laboratory. The pH from a composite water sample was measured at both wetland sites with a pHTesr2 field pH meter (®Oaklon). The pH meter was calibrated in the field with buffered solutions of pH 7, 10, and 4 prepared by Cole Parmer Instrument Company.

Transparency of the water column was measured with a black and white, 20 cm diameter Secchi disk (LaMotte Chemical company). The Secchi disk was lowered into the water column until visual acuity of the disk disappeared. Turbidity was measured in the field with a HACH model 2100P Nephelometer (Lind 1985). Calibration was performed in the field against three known formazin series standards of 1, 10, and 100 Nephelometric Turbidity Units (NTU).

Two 1-liter replicate water column samples were collected with from each wetland once in April bi-monthly between May and September 2000 and 2001. Water samples were stored in 1 liter acid washed amber glass bottles and transported on ice from the field. Each sample was filtered through a Whitman 45µm glass fiber filter and analyzed at the US Fish & Wildlife Service, Ecological Services laboratory in Tulsa, Oklahoma. Samples were analyzed for the following constituents: ammonia, chlorophyll a, nitrate, nitrite, total nitrogen, soluble reactive phosphate, and total phosphorus. Four 1-liter replicate water samples were collected from each wetland between April and September 2000 and 2001 were analyzed for calcium, magnesium, chloride, alkalinity, and sulfate. Methods of quantitative analysis for alkalinity, chloride, nitrite, soluble reactive phosphate, and total phosphorus followed procedures of Greenberg et al. (1992).

Methods for ammonia detection followed the procedures of Reardon et al. (1966). Nitrate and chlorophyll a were determined following the procedures of Lind (1985), and calcium, magnesium, sulfate, and total nitrogen concentrations were determined by procedures described by HACH Company (1997).

Nutrient limitation experiment

Nutrient limitation of periphyton was assessed with nutrient diffusing substrata as described by Gibeau and Miller (1989). Four plastic 30 mm x 30 mm test tube racks contained 6 replicates of 10-dram plastic vials filled with 37 ml of an 0.06 *M* agar solution augmented with either distilled water (control), 0.005 moles K_2PO_4/L (phosphorus), 0.05 moles $NaNO_3$/L (nitrogen), or 0.005

moles K_2PO_4/L + 0.05 moles $NaNO_3$/L (nitrogen + phosphorus). Each rack contained only one treatment (either control, nitrogen, phosphorus, or nitrogen + phosphorus), and was placed 1 m apart on the wetland sediment. Different treatments were not included within the same rack in order to avoid nutrient cross-contamination. Plastic vials were sealed by heating a porous silica/alumina crucible cover and molding the melted plastic of the vial around the cover. These racks were incubated *in situ* for three weeks at each wetland. At the end of the three-week incubation, the racks were retrieved and placed in an ice filled cooler and transported back to the lab. Each crucible cover was placed in a covered 25 ml wide mouth jar filled with 10 ml of a 90% acetone solution. Jars were placed in dark refrigeration and allowed to extract at least 3 days before chlorophyll a analysis.

Analysis of variance techniques were used to assess the effects of treatment and date. The PROCMIXED procedure of SAS version 8.0 (SAS Institute, Cary, N.C.) was used to perform the analysis. The experimental design was a two factor factorial arrangement of treatments in a completely randomized design. Differences in treatments at a given month were treated with a SLICE option in an LSMEANS statement. A significance level of 0.05 was set for each test.

Aquatic biota collections

Phytoplankton and zooplankton were sampled twice monthly. Phytoplankton samples were taken by lowering a 1.28 cm by 1.5 m PVC pipe

into the water column and corking the top to hold the 125ml sample in the pipe. Zooplankton samples were taken using a 12 cm diameter Wisconsin plankton net with 63 µm mesh. Samples were preserved in a 1 ml Lugol's solution. Phytoplankton cell counts and identification to lowest practical taxonomic level were preformed by Dr. Bob Lynch (University of Oklahoma, Health Science Center). Zooplankton were counted in a 1 ml Sedgwick-Rafter counting cell under a compound microscope and identified to order or family using Pennak (1978).

Submerged macrophytes were identified in both wetlands along a randomly chosen 100-m transect using a 1 m x 1 m grid made of 2.54 cm PVC pipe divided into 16 sections each section representing 6.25% of the total area of the grid. Ten points along the transect were randomly chosen for the placement of the grid and sections containing either plant or bare sediment were counted and recorded. Macrophytes were identified to species using Prescott (1980). *Typha* expansion was determined in both wetlands by measuring the edge of a randomly chosen stand from the shoreline to the furthest front edge of the stand. To ensure the stands were measured in the same spot each year, a benchmark location was determined at both wetland sites.

Fish were sampled in both wetlands by hook and line once each year and by electrofishing in 2001 only. A bait-casting rod with artificial or live bait was used to capture fish by hook and line in each wetland. Four hours of hook-and-line fishing were conducted on 9 September 2000 and 2001. A Coffelt electroshocker (model VVP-2C) was operated from an aluminum boat to capture

fish by electrofishing. Electrofishing was conducted on 26 September 2001 for 5 minutes in each wetland. Captured fish were measured, identified to species, and released.

RESULTS

Physico-chemical

Water depth in BH and SB wetland fluctuated greatly during the study as a result of a severe drought during the summer of 2000 and 2001. Water depths declined steadily from spring to fall during both years (Figure 2). The water boundary of BH was 2.3 m from the benchmark on 24 April 2000 and by 30 September 2000 this distance increased to 32.7 m. Distance from the benchmark to the water boundary at SB was 35 cm on 24 April 2000 and 13.9 m on 30 September 2000.

Turbidity in both wetlands was very low, rarely exceeding 3 NTUs, and water clarity was high. Sandy Bluff turbidity was generally higher than BH by 1-2 NTUs during each year, except for 9 and 30 September 2001 when the turbidity was 21.2 and 8.4 NTUs in SB and 5.40 and 4.10 NTUs in BH, respectively. The Secchi disk was visible to the bottom of each wetland; disk visibility was limited only by the depth of each wetland.

Both SB and BH were well-buffered, hard-water wetlands, and each wetland contained sufficient levels of oxygen to support aquatic biota. Simultaneous stratification of dissolved oxygen, specific conductivity, and temperature occurred only in BH wetland on 10 June 2001(Appendix A -E).

Nutrient Dynamics and Limitations

Global total nitrogen concentrations were higher in SB than in BH wetland. The mean and standard error (SE) for total nitrogen in SB was 1.27 mg/L ± 0.11 and 1.03 mg/L ± 0.09 in BH wetland; SB was significantly higher nine of the twenty-one sampling dates, while BH was significantly higher twice ($p < 0.05$) (Figure 3). Concentrations of ammonia and nitrate differed greatly between BH and SB wetland during the study, but nitrite concentrations were similar (Figure 3). The grand mean and SE for nitrate and ammonia in SB and BH wetland was 0.08 mg/L ± 0.01 and 0.05 mg/L ± 0.01, and 0.14 mg/L ± 0.02 and 0.05 mg/L ± 0.01, respectively. Nitrate was significantly higher in SB on fourteen of the twenty-one sampling dates and ammonia was significantly higher in SB on sixteen of the twenty-one sampling dates ($p < 0.05$). Nitrate and ammonia were significantly higher in BH on only one of the dates ($p < 0.05$). The grand mean and SE for nitrite in SB was 9.86 μg/L ± 5.35 and 2.58 μg/L ± 0.31 in BH wetland. Sandy Bluff wetland was significantly higher on eight of the twenty-one sampling dates ($p < 0.05$), while BH was not significantly higher than SB on any date ($p > 0.05$).

Total phosphorus and soluble reactive phosphorus were highly variable within and between wetlands (Figure 4). Total phosphorus was significantly higher in SB than in BH on ten of the twenty-one sampling dates, while BH was significantly higher on two sampling dates ($p < 0.05$). The grand mean and SE for total phosphorus was 125 μg/L ± 7.13 in SB and 105 μg/L ± 9.47 in BH wetland. Soluble reactive phosphorus was significantly higher in SB than in BH

wetland on six sampling dates and BH was significantly higher on two sampling dates ($p < 0.05$). The grand mean and SE for SB and BH wetland was 16.2 μg/L ± 1.99 and 15.1 μg/L ± 1.64, respectively.

Chlorophyll a concentrations differed greatly between wetlands and years in SB and BH wetland. Sandy bluff wetland was significantly higher than BH on seventeen of the twenty-one sampling dates, while BH was higher on two sampling dates ($p < 0.05$) (Figure 5). The grand mean and SE for SB was 8.06 mg/cm^3 ± 1.53 and 4.31 mg/cm^3 ± 0.99 for BH wetland.

Bull Head wetland was co-limited by nitrogen and phosphorus (N+P) in May, August, and September 2000 and May thru September 2001 (Figure 6). Insufficient data were available for June 2000 and no significant differences occurred between treatments in July 2001.

Sandy Bluff wetland was co-limited by nitrogen and phosphorus (N+P) in May and September 2000 and May, June, and September 2001 (Figure 6). Nitrogen (N) limitation occurred in SB during July and August of 2000. Insufficient data were available for June 2000 and no significant differences occurred between treatments in July or August 2001.

Aquatic biota

Phytoplankton in BH were predominately Bacillariophyta (29%), Chlorophyta (35%), and Cyanophyta (21%) (Figure 8). Bacillariophyta were exclusively pennales. Chlorophytes were primarily *Carteria sp.* (33%), *Tetraedron minimum* (20%), *Cosmarium spp.* (13%), *Oocystis spp.* (10%), and *Scenedesmus spp.* (10%). Cyanophyta were dominated by *Dactylococcopsis*

fascicularis (50%), *Anacystis sp.* (37%), and *Dactylococcopsis acicularis* (6%). Chryptophyta were primarily *Chroomonas nordstedtii* (47%), *Cryptomonas aspera* (31%), and *Cryptomonas ovata* (15%), Chrysophyta, Euglenophyta, and Pyrrophyta comprised the remaining 15% of the phytoplankton. Total standing crop for 2001 was 7.8×10^5 cells/L. There were incomplete data for 2000.

Sandy Bluff wetland was composed primarily of Chlorophyta (43%), and Chryptophyta (42%) (Figure 7). Chlorophytes were composed primarily of six species: *Crucigenia apiculata* (41%), *Gloeocystis sp.* (13%), *Scenedesmus sp.* (13%), *Oocystis spp.* (8%), *Tetraedron minimum* (8%), and *Dictyosphaerium sp.* (7%). Chryptophyta consisted primarily of *Chroomonas nordstedtii* (49%), *Cryptomonas ovata* (27%), and *Cryptomonas aspera* (18%). Cyanophyta, dominated by two genera, *Anacystis sp.* (75%) and *Oscillatoria sp.* (14%), Bacillariophyta, Chrysophyta, Euglenophyta and Pyrrophyta comprised the remaining 15% of the phytoplankton. Total standing crop for 2001 was 2.6×10^6 cells/L (incomplete data were available for 2000). A list of all species in each wetland is found in Appendix F.

Zooplankton in BP during 2000 consisted predominantly of two orders Rotatoria and Cladocera, and five primary taxa: *Daphina*, *Ceriodaphina, Brachionus*, *Keratella*, and Copepod nauplii (Table 1). In 2001, Copepoda replaced Rotatoria as the dominant zooplankton and the primary taxa were reduced from five to four (Copepod nauplii, *Keratella*, *Bosmina*, and Calanoida) (Table 2). Total standing crop was 425 cells/L in 2000 and 500 cells/L in 2001.

In SB during 2000, the dominant zooplankton orders were Copepoda and Rotatoria and the four dominant taxa were *Keratella*, Copepod nauplii , *Daphina*, and *Ceriodaphina.* In 2001, zooplankton composition paralleled 2000 with only minor differences in percent abundance (Tables 3 and 4). Total standing crop of SB was less than BH for both years with 224 cells/L in 2000 and 346 cells/L in 2001.

Three submerged macrophytes (*Chara vulgaris*, *Potamogeton pectinatus*, and *Ceratophyllum demersum*), one emergent (*Typha latifolia*) and one floating macrophyte (*Lemna* minor) were present in BH and SB during each year. The emergent macrophyte *Sagittaria brevirostra* was observed near the shore of SB only in 2001. Relative abundances (as percent) cover of submerged aquatic macrophytes in BH during 2000 were 72.5% *C. vulgaris* and 7.5% *P. pectinatus* (20% was bare sediment), and during 2001 were 99% *C. vulgaris* (1% was bare sediment). *Ceratophyllum demersum* was observed while electrofishing but was not present in the random transect sample. *Typha latifolia* was present only along the far NW edge of the wetland in small isolated stands. Lengths of the reference *Typha* stand were 7.0 m in 2000 and 10.1 m in 2001, an increase of 30.9%. Relative abundance of submerged macrophytes in SB during 2000 were 43% *P. pectinatus* and 40.5% *C. demersum* (16.5% was bare sediment), and during 2001 were 3.0 % *C. vulgaris,* 29% *P. pectinatus,* and 21.0 % *C. demersum* (41.0% was bare sediment). Lengths of the reference *Typha* stand in 2000 were 14.5 m in 2000 and 15.5 m in 2001, an increase of 6.3%.

Four species of fish were caught by hook and line in BH and SB wetland, *Micropterus salmoides*, *Lepomis macrochirus*, *Lepomis microlophus*, and *Pomoxis nigromaculatus* (Appendix G). Electrofishing yielded the same four species plus *Ameiurus melas* in both wetlands, although *A. melas* was only stunned and not captured in SB. No other species of fish were observed or captured in either wetland. Larger and fewer fish were caught by hook and line method, but smaller and more fish were caught while electrofishing (Appendix H). Forty-eight fish were captured by hook and line in each wetland in 2000, but in 2001, thirty more fish were caught in BH than in SB wetland.

DISCUSSION

Studies concerning the effects of nutrient enrichment on North American wetlands are few; the most comprehensive studies are those in the Netherlands indicating nutrient enrichment as a factor in the degradation of wetland habitats (Bedford et al. 1999, Sand-Jensen et al. 2000). Thus, demonstrating that nitrogen, phosphorus, or both are limiting may be useful for predicting changes associated with nutrient enrichment in North American wetlands. I observed that both BH and SB wetlands were co-limited by nitrogen and phosphorus, and on two occasions, SB was nitrogen limited. A reason for nitrogen limitation may be due to differences in the concentrations of available nitrogen and phosphorus in the water column during July and August of 2000. Zimba (1998) documented nitrogen limitation of epiphyton in shallow, open water regions of Lake Okeechobee, Florida when total phosphorus concentrations ranged from 35 to

120 µg/L and Philips et al. (1997) showed nitrogen was limiting for phytoplankton in Lake Okeechobee when nitrate concentrations were between 0.009 and 0.367 mg/L. During July and August, phosphorus ranged from 92 to 222 µg/L and nitrate concentrations were between 0.059 and 0.241 mg/L in SB wetland. Additionally, the ratio of nitrogen to phosphorus during July and August was 5:1 and 12:1 respectively. This is much less than Redfield's uptake ratio for phytoplankton of 16:1 (atoms). Therefore, it is likely that although the amount of available phosphorus was sufficient for periphyton growth, the amount of available nitrogen was not.

Interestingly, in months when phosphorus, nitrate, and ammonia equaled or exceeded concentrations present during July and August 2000, the additional inputs of nitrogen and phosphorus from the nutrient limitation experiment stimulated periphyton growth. The average N / P ratio in BH and SB during 2000 and 2001 was 14:1 and 18:1, and 15:1 and 13:1 respectively. Since these ratios were near Redfield's, nitrogen or phosphorus limitation was not expected. However, Istvanovics et al. (1986) noted severe nitrogen and phosphorus co-limitation in Lake Balaton at N / P ratios between 8 and 23:1. Thus, it is likely that phytoplankton and periphyton in BH and SB wetland are sensitive to additional inputs of nitrogen and phosphorus, and that limiting N / P ratios may be specific to individual water bodies.

Fluctuations in the physical and chemical properties, biota, and nutrients in BH and SB wetland were generally characteristic of most shallow water bodies in the US (Crisman et al. 1998, McCormick et al. 1998, Hambright et al. 1998,

Fisher and Willis 2000, Schell et al. 2001). However, nutrient concentrations in BH and SB were more characteristic of nutrient-enriched water bodies (Schwartz and Gruendling 1985). Possible sources of nutrients in BH include potential run-off from cattle waste, sequestering and release of nutrients (particularly phosphorus) in submerged macrophytes during periods of growth and senescence, and concentration of nutrients associated with the severe droughts during 2000 and 2001. Despite this, nutrient concentrations in BH were significantly lower than those in SB during most months. Turbidity was low in each wetland. This very clear water, combined with an ample supply of nutrients, provided good conditions for the growth autotrophic organisms. The dramatic increase in turbidity recorded in September 2001 was the result of suspended flocculent material from macrophyte senescence and not a large algal bloom.

Differences in nutrient concentrations between the wetlands were most likely related to run-off from the swine CAFO lagoon and land application field. This was apparent in the seasonal trends in ammonia and nitrate. In natural waters, ammonia is present in relatively low quantities because it is readily oxidized to nitrate in the presence of oxygen. However, elevated concentrations of ammonia can be an indication of waste contamination (Hutchinson 1957). It is likely that CAFO run off did not enter SB during the summer drought months of 2000 or 2001. Yet, contamination of SB with ammonia and nitrate might have occurred via ground water which entered the wetland through surface springs along the north shore. Ham et al. (2000) identified the potential for ground water to become contaminated from swine lagoons in a study of more than 20 CAFO

facilities across Kansas. It is possible that wastewater may have leached from the swine CAFO lagoon into the surrounding soils and eventually reached the groundwater and SB wetland. This pathway may not be available to BH because there are no surface springs along the wetland's shoreline. Further investigation of the surface springs surrounding SB will be necessary to substantiate this claim.

If conditions in SB remain eutrophic, temporal changes in the zooplankton, phytoplankton, macrophytes, and fish assemblages may occur. Noticeable differences between the plankton and macrophyte communities in each wetland are already present. Indicator species or taxa provide one form of assessment of change (McCormick et al. 1998, Zohary et al. 1998, McCormick et al. 2001). Only one genus indicative of eutrophic conditions (i.e. *Oscillatoria*) was present in small quantities in both BH and SB wetland. As a result, algal communities between BH and SB were compared at the division level, as suggested by Rojo (2000), because species assemblages from separate locations within the same wetland with similar trophic states may vary considerably.

Both wetlands had dramatically different phytoplankton assemblages even though each wetland was eutrophic. The phytoplankton assemblage of SB was composed primarily of chlorophytes and chryptomonads. These taxa are usually dominant in oligotrophic water (absent of nutrient pollution and with low nutrient concentrations), which was quite opposite to the conditions in SB during 2000 and 2001. In BH wetland, the dominant taxa were chlorophytes, bacillariophytes, and cyanophytes. These taxa are characteristic of eutrophic waters (Bronmark

and Hansson 1998, McCormick et al. 1996, McCormick et al. 1998, Zohary et al. 1998). Differences in the phytoplankton assemblages between wetlands may be related to differences in zooplankton grazing pressure (Beaver et al. 1999). The zooplankton community in SB consisted primarily of rotifers and copepods. It is possible that rotifers exhibited stronger grazing pressure on the cyanophytes than the cladocerans and copepods did on the cholorphytes and chryptomonads as these larger zooplankton made up a smaller portion of the community. In BH wetland, the opposite occured with approximately one fourth of the zooplankton community comprised of rotifers and three fourths consisting of cladocerans and copepods. Thus, heavier grazing on chlorophytes, pyrrophytes, and bacilliarophytes by cladocerans and copepodites, and less grazing on cyanophytes by rotifers may explain differences in phytoplankton assemblages between the wetlands.

Large differences in the abundance of submerged and emergent macrophytes in wetlands can be the result of hydrology, nutrient cycling, and surrounding terrestrial plant stands (Newman et al. 1998, Miao et al. 2000, Eghball et al. 2000). Expansion of the vast *Typha* stand between 2000 and 2001 surrounding SB might have been the result of nutrient enrichment from within-stand nutrient cycling, CAFO run-off and contaminated groundwater, and the lack of a terrestrial plant buffer of trees, shrubs, and grasses around the wetland. The decline in the relative abundance of submerged macrophytes from 2000 to 2001 is not completely understood. Light limitation was not a factor, and nutrients in the water column remained ample between April and September 2001. One

possibility is that the macrophytes became stressed from the extreme heat and fluctuating water levels during the summer drought (ambient air temperatures exceeded 38 °C for a period of 17 straight days) which probably led to senescence earlier in the year (before transect sampling began).

In contrast, the expansion of submerged macrophytes and *Typha* stands in BH may be the direct result of the hydrology during 2000. Areas that were underwater and uncovered by submerged macrophytes in spring of 2000 were exposed during fall 2000. This may have increased the ability for seed colonization in those areas not covered by macrophytes in the spring. When the wetland filled back up to near normal levels during the winter and spring of 2001, it may have facilitated new *Chara* growth. The 3.1 m expansion of *Typha* in BH between 2000 and 2001 was likely the result of receding water levels, vegetative expansion from extensive rhizome systems, and potentially phosphorus rich sediments from decomposed with-in stand plant litter. Miao et al. (2001) noted that lower water levels and phosphorus-enriched sediments promote *Typha* growth. As water levels recede, sediments become exposed which creates more surface area. This can concentrate seeds and increase the chance for the recruitment of new seedlings.

CONCLUSIONS

It is evident that SB and BH wetlands are dynamic systems in which nutrient concentrations, plant communities, and zooplankton assemblages vary greatly. It is probable that contamination from the swine CAFO is altering certain

ecological characteristics in SB such as *Typha* stand growth, total standing crop of phytoplankton, ammonia, nitrate, phosphorus, and chlorophyll a concentrations. However, some of the same ecological characteristics in BH wetland mimicked the progression of a wetland receiving nutrient pollution. Additionally, the natural processes that influenced the ecological characteristics of BH may also be occurring in SB, thereby influencing the ecological progression of SB as well. These natural processes may be acting synergistically with contaminants from the swine CAFO to hasten the process of eutrophication in SB wetland.

LITERATURE CITED

Bedford, B.C., M. R. Walbridge, and A. Aldous. 1999. Patterns in nutrient availability and plant diversity of temperate North American wetlands. Ecology 80:2151-2169.

Beaver, J. R., A. M. Miller-Lemke, and J. K. Acton. 1999. Midsummer zooplankton assemblages in four types of wetlands in the upper Midwest, USA. Hydrobiologia 380:209-220.

Blahnik, T., and J. Day. 2000. The effects of varied hydraulic and nutrient loading rates on water quality and hydrologic distributions in a natural forested treatment wetland. Wetlands 20:48-61.

Boustany, R. G., C. R. Crozier, J. M. Rybczyk, and R. R. Twilley. 1997. Denitrification in a south Louisiana wetland forest receiving treated swage effluent. Wetlands Ecology and Management 4:273-283.

Bridgham, S. D., J.Pastor, J. A. Janssens, C. Chapin, and T. J. Malterer. 1996. Multiple limiting gradients in peatlands: a call for a new paradigm. Wetlands 16:45-65.

Bronmark, C. and L. A. Hansson. 1998. The Biology of Lakes and Ponds. Oxford University Press, New York, USA.

Cairns, J. Jr., B. R. Niederlehner, and D. R. Orvos. 1992. Predicting Ecosystem Risk. Princeton Scientific Publishing Co. Inc. Princeton, NJ, USA.

LITERATURE CITED

Bedford, B.C., M. R. Walbridge, and A. Aldous. 1999. Patterns in nutrient availability and plant diversity of temperate North American wetlands. Ecology 80:2151-2169.

Beaver, J. R., A. M. Miller-Lemke, and J. K. Acton. 1999. Midsummer zooplankton assemblages in four types of wetlands in the upper Midwest, USA. Hydrobiologia 380:209-220.

Blahnik, T., and J. Day. 2000. The effects of varied hydraulic and nutrient loading rates on water quality and hydrologic distributions in a natural forested treatment wetland. Wetlands 20:48-61.

Boustany, R. G., C. R. Crozier, J. M. Rybczyk, and R. R. Twilley. 1997. Denitrification in a south Louisiana wetland forest receiving treated swage effluent. Wetlands Ecology and Management 4:273-283.

Bridgham, S. D., J.Pastor, J. A. Janssens, C. Chapin, and T. J. Malterer. 1996. Multiple limiting gradients in peatlands: a call for a new paradigm. Wetlands 16:45-65.

Bronmark, C. and L. A. Hansson. 1998. The Biology of Lakes and Ponds. Oxford University Press, New York, USA.

Cairns, J. Jr., B. R. Niederlehner, and D. R. Orvos. 1992. Predicting Ecosystem Risk. Princeton Scientific Publishing Co. Inc. Princeton, NJ, USA.

Crisman, T. L., L. J. Chapman, and C.A. Chapman. 1998. Predictors of seasonal oxygen levels in small Florida lakes: the importance of color. Hydrobiologia 368:149-155.

Chow-Fraser, P., V. Lougheed, V. L. Thiec, B. Crosbie, L. Simser, and J. Lord. 1998. Long-term response to the biotic community to fluctuating water levels and changes in water quality in Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario. Wetlands Ecology and Management 6:19-42.

Cottingham, K. L., S. R. Carpenter, and A. L. St. Amand. 1998. Responses of epilimnetic phytoplankton to experimental nutrient enrichment in three small seepage lakes. Journal of Plankton Research 20:1889-1914.

Cross, G. R., R. E. Shaffer, and D. D. Badger. 1970. Socio-economic impact of recent water quality legislation on confined animal feeding operators. Proceedings of the Oklahoma Academy Science 51:84-85.

Daoust, R. J. and D. L. Childers. 1999. Controls on emergent macrophyte composition, abundance and productivity in freshwater everglades wetland communities. Wetlands 19:262-275.

Dodson, S. I., and R. A. Lillie. 2001. Zooplankton communities of restored depressional wetlands in Wisconsin, USA. Wetlands 21:292-300.

Eghball, B., J. E. Gilley, L. A. Kramer, T. B. Moorman. 2000. Narrow grass hedge effects on phosphorus nitrogen in runoff following manure and fertilizer application. Journal of Soil and Water Conservation 55:172-181.

Fisher, S. J. and D. W. Willis. 2000. Seasonal dynamics of aquatic fauna and habitat parameters in a perched upper Missouri river wetland. Wetlands 20: 470-478.

Gibeau, G. G. and M. C. Miller. 1989. A micro-bioassay for epilithon using nutrient-diffusing artificial substrata. Journal of Freshwater Ecology 5:171-176.

Giniting, D., J. F. Monecrief, S.C. Gupta, and S.D Evans. 1998. Interaction between manure and tillage system on phosphorus uptake and runoff losses. Journal of Environmental Quality 27:1403-1410.

Giusquiani, P. L., L. Concezzi, M. Businelli, and A. Macchioni. 1998. Fate of pig sludge liquid fraction in calcareous soil: agricultural and environmental implications. Journal of Environmental Quality 27:364-371.

Greenberg, A. E., L. S. Clesceri, and A. D. Eaton (eds). 1992. Standard Methods for the Examination of Water and Wastewater eighteenth edition. American Public Health Association, Washington, D.C, USA.

HACH Company. 1997. HACH Water Analysis Handbook third edition. HACH Company, Loveland, Colorodo, U.S.A.

Ham, J. M., T. M. DeSutter, and C.W. Rice. 2000. Animal Waste Lagoon Water Quality Study. Research Report, Kansas State Universtiy, Mahattan, KS, USA.

Hambright, K. D., Bar-Ilan, I., and W. Eckert. 1998. General water chemistry and quality in a newly-created subtropical wetland lake. Wetlands Ecology and Management 6:121-132.

Havens, K. E., T. L. East, S. J. Hwang, A. J. Rodusky, B. Sharfstein, and A. D. Steinman. 1999. Algal responses to experimental nutrient addition in the littoral community of a subtropical lake. Freshwater Biology 42:329-343

Hutchinson, G. E. 1957. A Treatise on Limnology, volume one. John Wiley and Sons Inc., New York, USA.

Istvanovics, V., L. Voros, S. Herodek, L. G. Toth, and I. Tatrai. 1986. Changes of phosphorus and nitrogen concentration and of phytoplankton in enriched lake enclosures. American Society of Limnology and Oceanography 31: 798-811

Kadlec, R. H. 1999. The limits of phosphorus removal in wetlands. Wetlands Ecology and Management 7:165-175.

Lemly, D. A. and R. S. King. 2000. An insect-bacteria bioindicator for assessing detrimental nutrient enrichment in wetlands. Wetlands 20:91-100.

Lind, O. T. 1985. Handbook of Common Methods in Limnology. Kendall/Hunt Publishing; Dubuque, Iowa, USA.

Matlock, M. D., D. E. Storm, M. D. Smolen, M. E. Matlock, A. M. S. McFarland, and L. M. Hauck. 1999. Development and application of a lotic ecosystem trophic status index. American Society of Agricultural Engineers 42(3):651-656.

Matlock, M. D., M. E. Matlock, D. E. Storm, M. D. Smolen, and W. J. Henley. 1998. Limiting nutrient determination in lotic ecosystems using a quantitative nutrient enrichment periphytometer. Journal of the American Water Resources Association 34:1141-1147.

Mattila, J. and R. Raisanen. 1998. Periphyton growth as an indicator of eutrophication; an experimental approach. Hydrobiologia 377:15-23.

McCormick, P. V., P. S. Rawlick, K. Lurding, E. P. Smith, and F. H. Sklar. 1996. Periphyton-water quality relationships along a nutrient gradient in the Northern Florida Everglades. Journal of the North American Benthological Society 15:443-449.

McCormick, P. V., and M. B. O'Dell. 1996. Quantifying periphyton responses to phosphorus enrichment in the Florida Everglades: a synoptic-experimental approach. Journal of the North American Benthological Society. 15:450-468.

McCormick, P. V., R. B. E. Shuford III, J.G. Backus, and W.C. Kennedy. 1998. Spatial and seasonal patterns of periphyton biomass and productivity in the northern Everglades, Florida, USA Hydrobiologia 362:185-208.

McCormick, P. V., M. B. O'Dell, R. B. E. Shuford III, J. G. Backus, and W. C. Kennedy. 2001. Periphyton responses to experimental phosphorus enrichment in a subtropical wetland. Aquatic Botany 71:119-139.

Miao, S. L., S. Newman, and F. Sklar. 2000. Effects of habitat nutrients and seed sources on growth and expansion of *Typha Domingensis*. Aquatic Botany 68:297-311.

Miao, S. L., P. V. McCormick, S. Newman, and S. Rajagopalan. 2001. Interactive effects of seed availability, water depth, and phosphorus enrichment on cattail colonization in an Everglades wetland. Wetlands ecology and management 9:39-47.

Moffitt, D. C, and C. H. Lander. 1999. Nutrients available from livestock manure relative to land use. USGS Workshop on Animal Feeding Operations: Effects on Hydrologic Resources and the Environment, Aug 30 - Sept 1, 1999, Ft. Collins, CO.

Moore H. H., W. A. Niering, L. J. Marsicano, and M. Dowdell. 1999. Vegetation change in created emergent wetlands (1988-1996) in Connecticut, USA.. Wetlands Ecology and Management 7:177-191.

Newman S., J. Schuette, J. B. Grace, K. Rutchey, T. Fontaine, K. R. Reddy, and M. Pietrucha. 1997. Factors influencing cattail abundance in the northern everglades. Aquatic Botany 60:265-280.

Ortega-Mayagoitia, E., X. Armengol, and C. Rojo. 2000. Structure of zooplankton in a semi-arid wetland, the national park Las Tablas de Daimiel (Spain). Wetlands 20:629-638.

Pennak, R. W. 1978. Freshwater Invertebrates of the United States, second edition. John Wiley and Sons, USA.

Phlips, E. J., M. Cichra, K. Havens, C. Hanlon, S. Badylak, B. Rueter, M. Randall, and P. Hansen. 1997. Relationships between phytoplankton dynamics and the availability of light and nutrients in a shallow sub-tropical lake. Journal of Plankton Research 19:319-342.

Prescott, G.W. 1980. How to Know the Aquatic Plants, second edition. Wm. C. Brown Company Publishers, Dubuque, IA, USA.

Reardon, J., J. A. Foreman, and R.L. Searcy. 1966. New reactants for the colorimetric determination of ammonia. Clinica Chimica Acta 14:403-405.

Rojo, C., E. Ortega-Mayagoitia, and M. Alvarez-Cobelas. 2000. Lack of pattern among phytoplankton assemblages. Or, what does the exception to the rule mean? Hydrobiologia 424:133-139.

Sand-Jensen, K., T. Riis, O. Vestergaard, and S. E. Larsen. 2000. Macrophyte decline in Danish lakes and streams over the past 100 years. Journal of Ecology 88:1030-1040.

Saunders, D. L. and J. Kalff. 2001. Nitrogen retention in wetlands, lakes and rivers. Hydrobiologia 443:205-212

Schell, J. M., C. J. Santos-Flores, P. E. Allen, B. M. Hunker, S. Kloehn, A. Michelson, R. A. Lillie, and S. I. Dodson. 2001. Physical-chemical influences on vernal zooplankton community structure in small lakes and wetlands Wisconsin, U.S.A. Hydrobiologia 445:37-50.

Schwartz, L. N and G. K. Gruendling. 1985. The effects of sewage on a Lake Champlain wetland. Journal of Freshwater Ecology 3: 35-46.

Smoot, J. C., D. E. Langworthy, M. Levy, R. H. Findlay. 1998. Periphyton growth on submerged artificial substrate as a predictor of phytoplankton response to nutrient enrichment. Journal of Microbiological Methods 32:11-19.

Stewart, H. T., S. L. Miao, M. Colbert, and C. E. Carraher Jr. 1997. Seed germination of two cattail *(Typha)* species as a function of everglades nutrient levels. Wetlands 17:116-122

Toetz, D. W., T. Letong, D. E. Storm, T. Mihuc, and M. D. Smolen. 1999. Assessment of predictors for stream eutrophication potential. Journal of the American Water Resources Association 35: 853-865.

USDA (U.S. Department of Agriculture) and USEPA (U.S. Environmental Protection Agency). 1999. Unified National Strategy for Animal Feeding Operations, March 9, 1999. U.S Department of Agriculture and U.S Environmental Protection Agency, Washington D.C., U.S.A.

USDA (U.S. Department of Agriculture). 1999. A Procedure to Estimate the Response of Aquatic Systems to Changes in Phosphorus and Nitrogen Inputs. U.S. Department of Agriculture, National Resources Conservation Service, Washington D.C., U.S.A.

Verhoeven, J. T. A., W. Koreselman, and A. F. M. Meuleman. 1996. Nitrogen or phosphorous limited growth in herbaceous, wet vegetation; relations with atmospheric inputs and management regimes. Trends in Ecology and Evolution 11: 494-497.

Weiss, C. M 1969. Relation of phosphates to eutrophication. Journal American Water Works Association 61:387-91.

White, J. S., S. E. Bayley, and P. J. Curtis. 2000. Sediment storage of phosphorus in a northern prairie wetland receiving municipal and agro-industrial wastewater. Ecological Engineering 14:127-138.

Woltemade, C.J. 2000. Ability of restored wetlands to reduce nitrogen and phosphorus concentrations in agriculture drainage water. Journal of Soil and Water Conservation 55:303-309.

Zimba, P. V. 1998. The use of nutrient enrichment bioassays to test for limiting factors affecting epiphytic growth in Lake Okeechobee, Florida; confirmation of nitrogen and silica limitation. Archives Hydrobiologia 141:459-468.

Zohary, T., T. Fishbein, B. Kaplan, and U. Pollingehr. 1998. Phytoplankton-metaphyton seasonal dynamics in a newly created subtropical wetland lake. Wetlands Ecology and Management 6:133-142.

Table 1. Seasonal succession of zooplankton (organisms/L) in Bull Head wetland April 2000 - September 2000.

	Sampling dates									
Taxa	24-Apr	17-May	31-May	6-Jun	5-Jul	26-Jul	10-Aug	31-Aug	9-Sep	% Composition
Calanoida[a]	8	36	11	14	17	100	36	75	11	7.3
Cyclopoida[a]	8	5	6	14	14	46	4	11	0	2.5
Harpactacoida[a]	0	0	3	0	9	0	0	75	0	2.0
Copepod nauplii	18	90	42	11	119	157	85	0	29	13.0
Daphnidae[b] (*Daphnia*)	18	141	31	305	26	175	61	81	80	21.6
Daphnidae[b] (*Ceriodaphnia*)	5	0	0	0	6	46	0	86	547	16.3
Bosminidae[b] (*Bosmina*)	3	0	0	0	14	18	52	64	251	9.5
Brachionidae[c] (Keratella)	0	38	274	107	61	11	8	43	23	13.3
Brachionidae[c] (Brachionus)	0	46	321	206	12	14	0	0	0	14.1
Brachionidae[c] (*Platyias*)	0	0	0	0	3	0	4	5	6	0.4

[a] Copeopoda [b] Cladocera [c] Rotatoria

Table 2. Seasonal succession of zooplankton (organisms/L) in Bull Head wetland April 2001 - September 2001.

	Sampling dates										
Taxa	30-Apr	10-May	10-Jun	30-Jun	9-Jul	30-Jul	4-Aug	25-Aug	9-Sep	30-Sep	% Composition
Calanoida[a]	357	18	24	4	38	6	105	8	18	14	11.8
Cyclopoida[a]	14	4	10	0	5	12	11	88	18	28	3.8
Harpactacoida[a]	0	0	0	0	0	0	0	0	0	0	0.0
Copepod nauplii	82	64	137	55	115	94	72	96	1004	128	37.0
Daphnidae[b] (*Daphnia*)	32	18	0	0	5	12	61	48	129	14	6.4
Daphnidae[b] (*Ceriodaphnia*)	0	0	0	0	53	6	55	0	9	50	3.5
Bosminidae[b] (*Bosmina*)	11	14	41	25	67	18	248	0	212	163	16.0
Brachionidae[c] (Keratella)	14	667	21	165	19	12	6	1	0	0	18.1
Brachionidae[c] (Brachionus)	0	0	0	0	5	12	0	1	9	7	0.7
Brachionidae[c] (*Platyias*)	0	4	3	4	0	12	6	48	55	7	2.8

[a] Copeopoda [b] Cladocera [c] Rotatoria

Table 3. Seasonal succession of zooplankton (organisms/L) in Sandy Bluff wetland April 2000 - September 2000.

	Sampling dates									
Taxa	24-Apr	17-May	31-May	6-Jun	5-Jul	26-Jul	10-Aug	31-Aug	9-Sep	% Composition
Calanoida[a]	6	49	18	10	0	0	0	0	40	5.5
Cyclopoida[a]	28	22	9	0	3	10	0	0	44	5.2
Harpactacoida[a]	0	0	0	0	0	0	0	4	9	0.6
Copepod nauplii	28	22	46	52	45	98	33	13	128	20.8
Daphnidae[b] (*Daphnia*)	19	40	64	20	14	17	18	13	106	13.9
Daphnidae[b] (*Ceriodaphnia*)	0	0	0	0	3	0	0	0	212	9.6
Bosminidae[b] (*Bosmina*)	3	0	0	0	0	7	4	4	102	5.4
Brachionidae[c] (Keratella)	0	62	140	146	206	59	11	9	0	28.2
Brachionidae[c] (Brachionus)	0	40	70	20	14	24	0	0	0	7.5
Brachionidae[c] (*Platyias*)	0	0	0	36	0	17	4	17	0	3.3

[a] Copeopoda [b] Cladocera [c] Rotatoria

Table 4. Seasonal succession of zooplankton (organisms/L) in Sandy Bluff wetland April 2001 - September 2001.

	Sampling dates										
Taxa	30-Apr	10-May	10-Jun	30-Jun	9-Jul	30-Jul	4-Aug	25-Aug	9-Sep	30-Sep	% Composition
Calanoida[a]	21	9	7	93	4	5	11	0	61	65	8.0
Cyclopoida[a]	0	0	0	39	0	5	6	39	18	18	3.6
Harpactacoida[a]	21	0	0	0	0	0	0	0	0	0	0.6
Copepod nauplii	115	45	61	54	157	94	39	83	140	53	24.3
Daphnidae[b] (*Daphnia*)	115	9	0	0	4	0	0	66	67	47	8.9
Daphnidae[b] (*Ceriodaphnia*)	0	0	0	23	0	0	22	83	213	24	10.5
Bosminidae[b] (*Bosmina*)	7	12	11	62	25	0	39	6	55	12	6.6
Brachionidae[c] (Keratella)	0	478	360	39	83	42	11	11	6	6	29.9
Brachionidae[c] (Brachionus)	3	0	11	4	17	73	17	17	0	0	4.1
Brachionidae[c] (*Platyias*)	0	0	68	19	0	10	11	11	0	0	3.4

[a] Copeopoda [b] Cladocera [c] Rotatoria

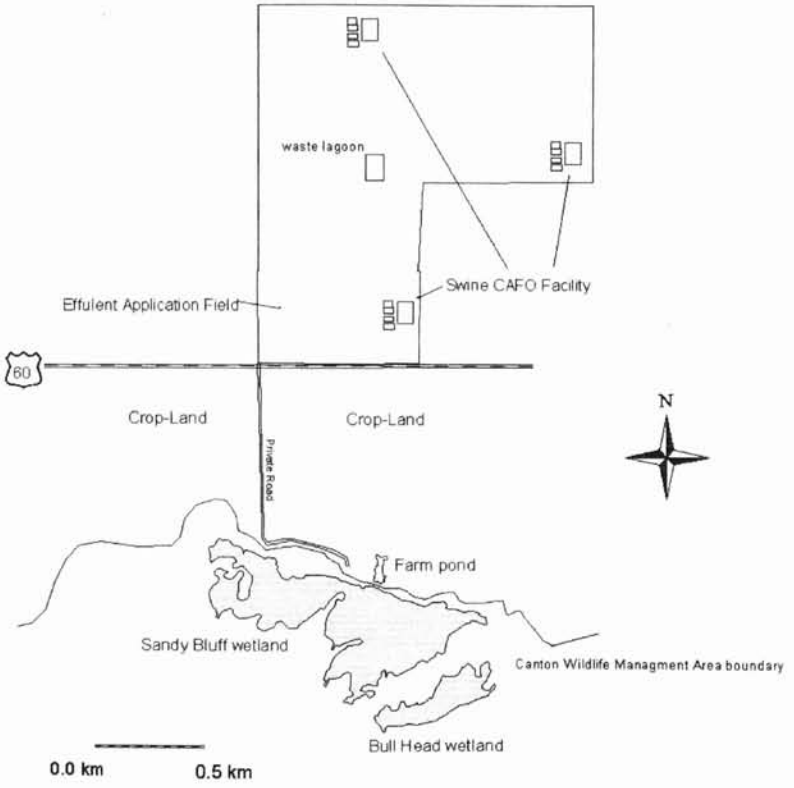

Figure 1. Map of study site.

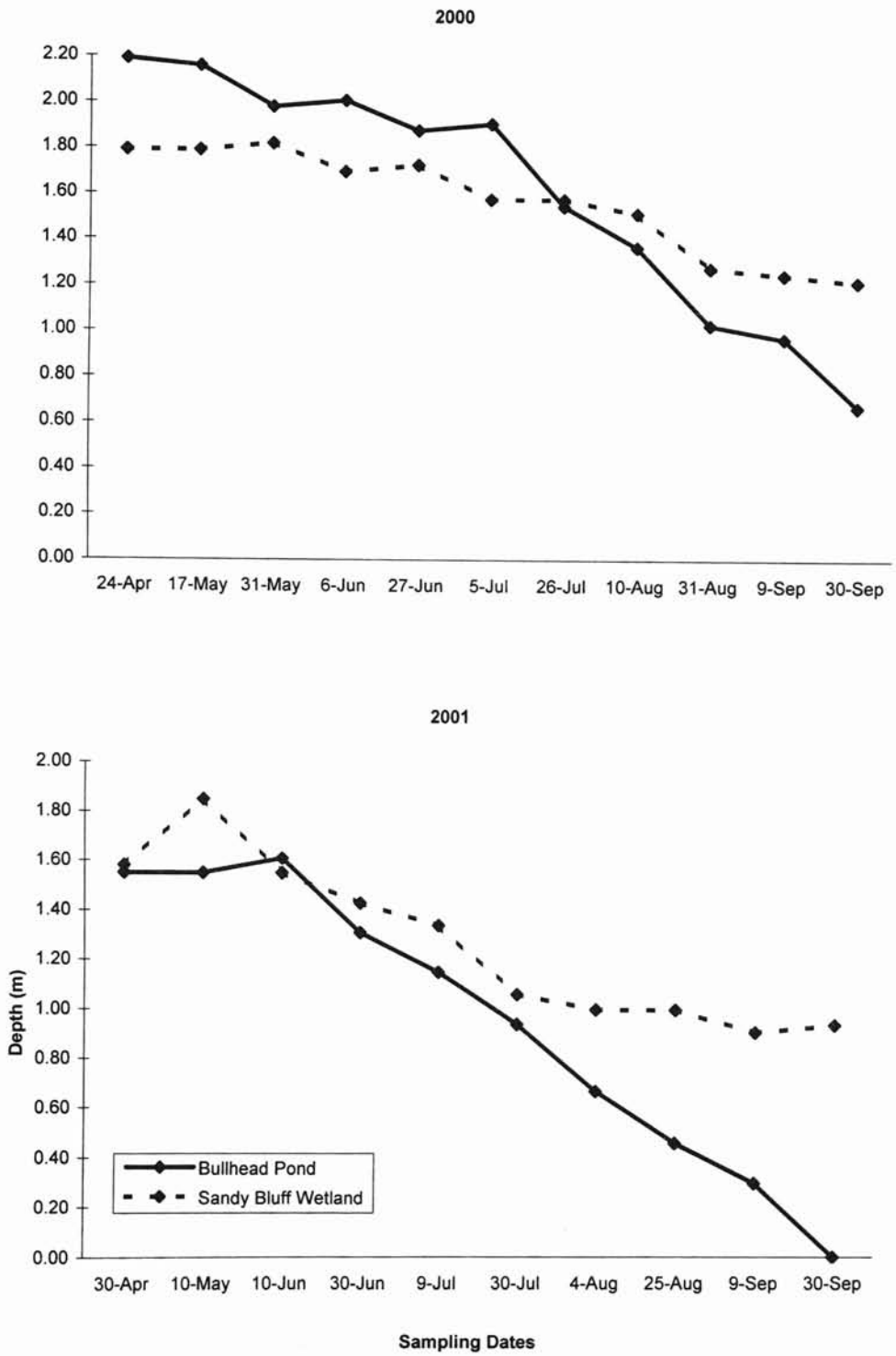

Figure 2. Depth of Bull Head and Sandy Bluff wetland 2000 - 2001.

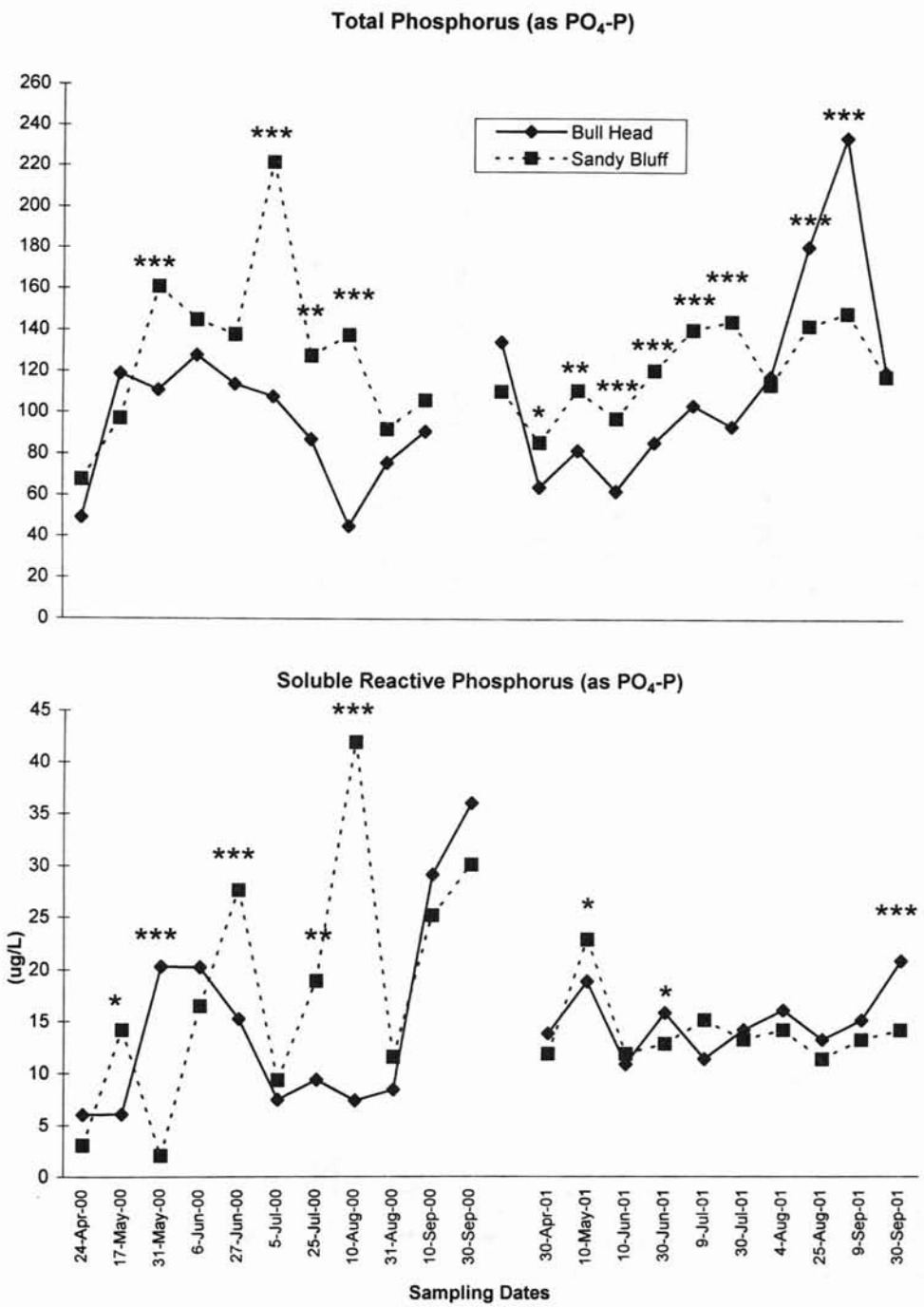

Figure 4. Mean Concentrations of total phosphorus and soluble reactive phosphorus in Bull Head and Sandy Bluff wetland 2000 - 2001 (*$p < 0.05$, **$p < 0.01$, ***$p < 0.001$).

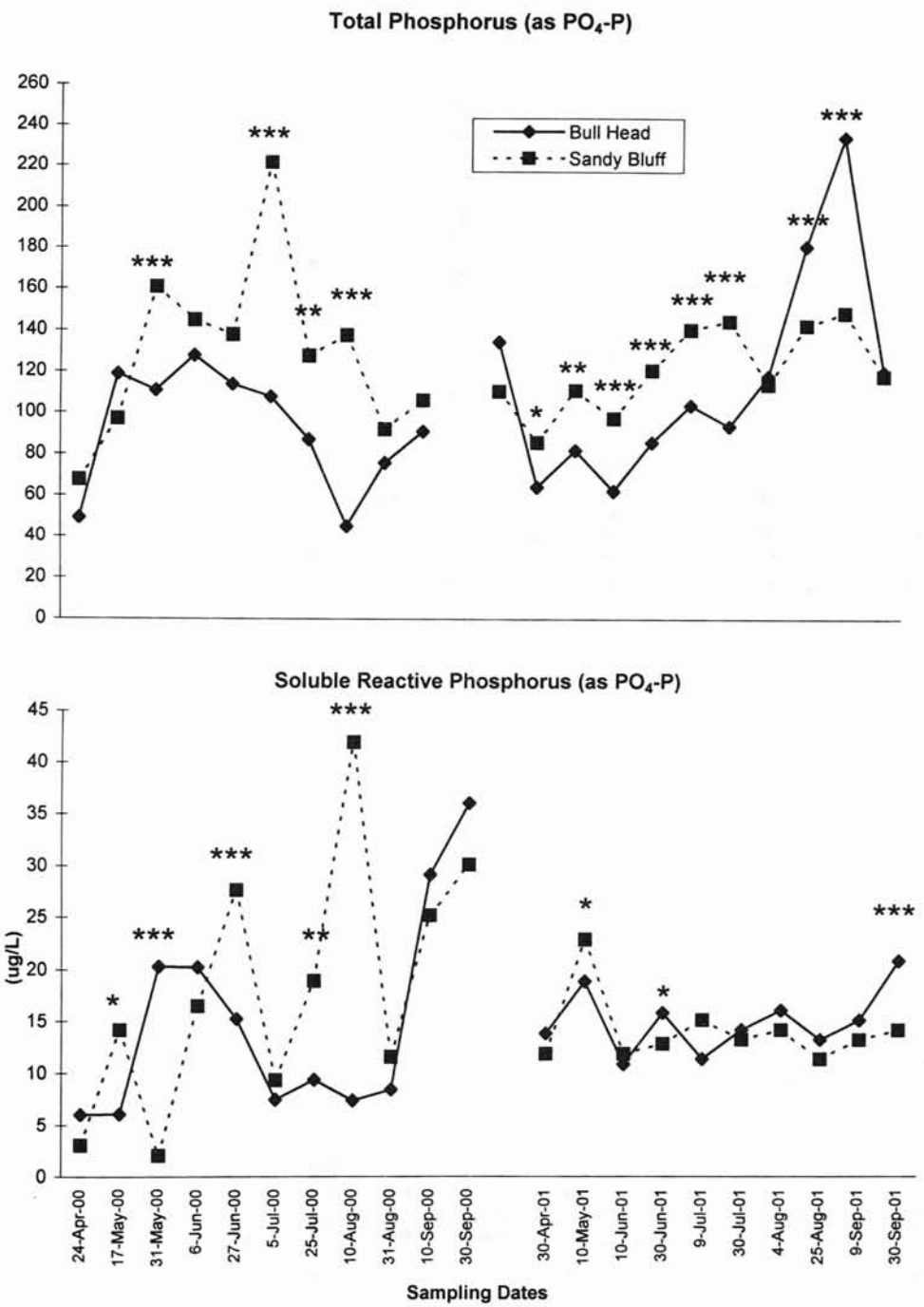

Figure 4. Mean Concentrations of total phosphorus and soluble reactive phosphorus in Bull Head and Sandy Bluff wetland 2000 - 2001 (*$p < 0.05$, **$p < 0.01$, ***$p < 0.001$).

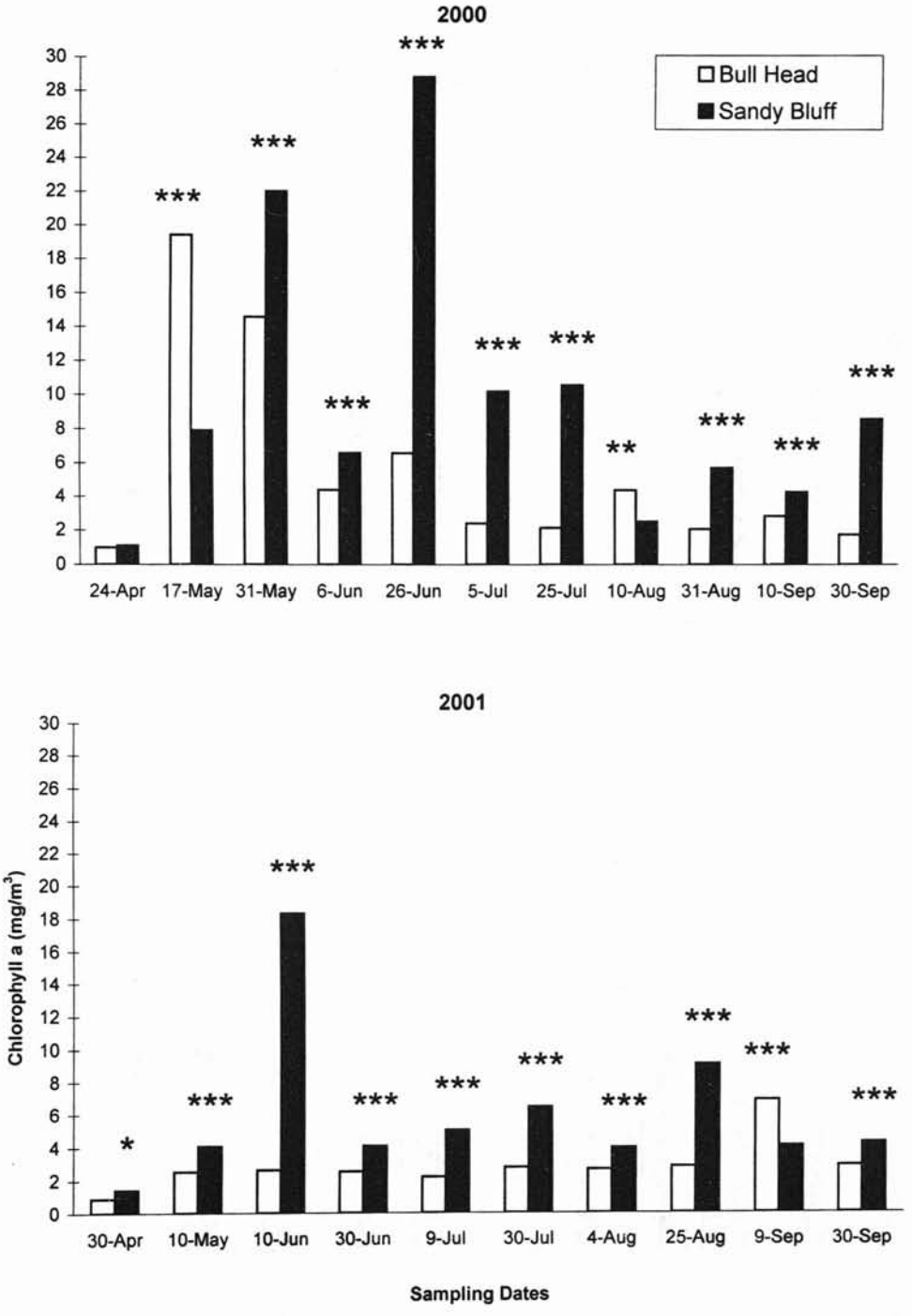

Figure 5. Mean chlorophyll a concentrations in Bull Head and Sandy Bluff wetland 2000 - 2001 (*$p < 0.05$, **$p < 0.01$, ***$p < 0.001$).

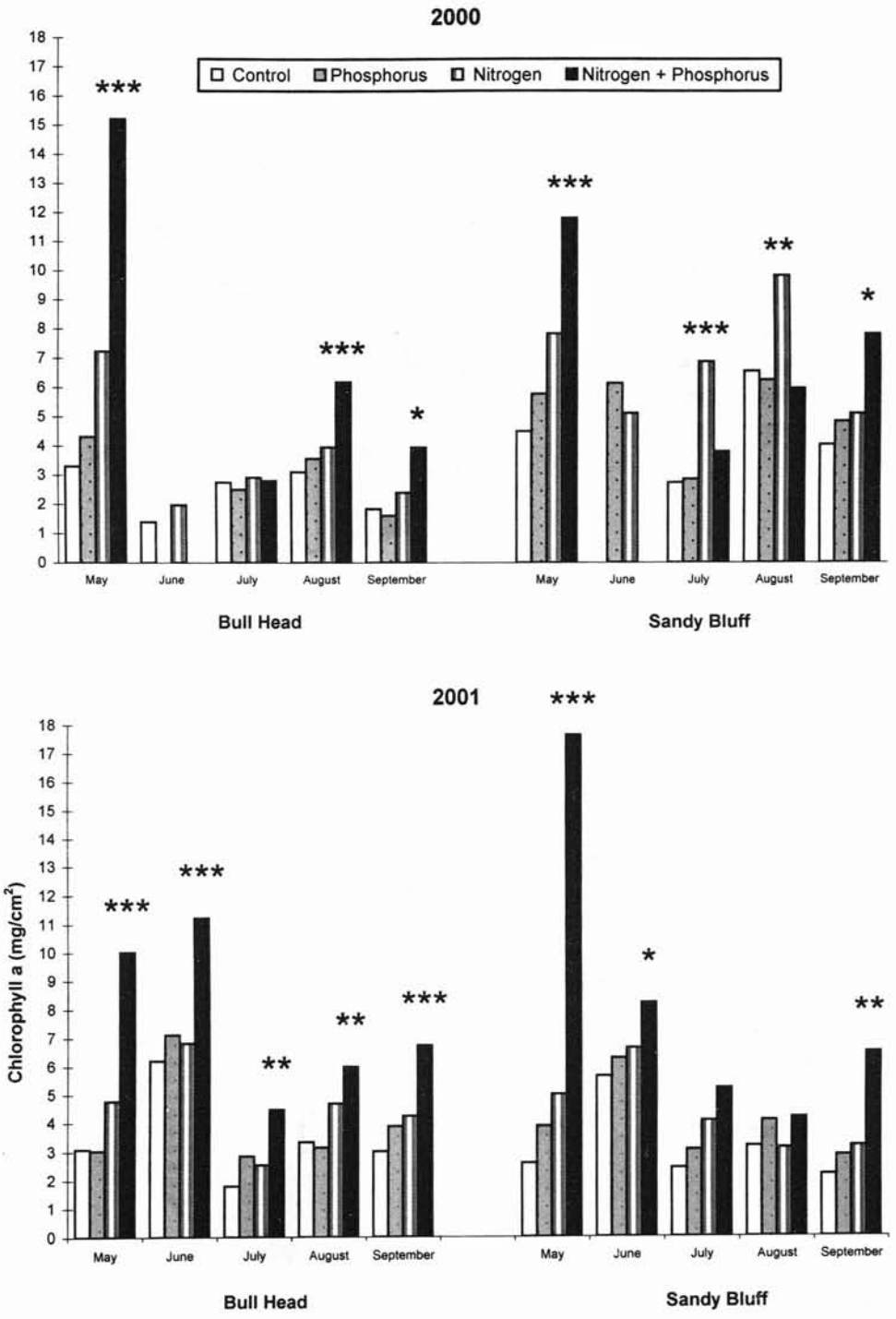

Figure 6. Mean chlorophyll a concentrations from nutrient limitation experiments in Bull Head and Sandy Bluff Wetland 2000 - 2001 (*p < 0.05,**p < 0.01, ***p < 0.001).

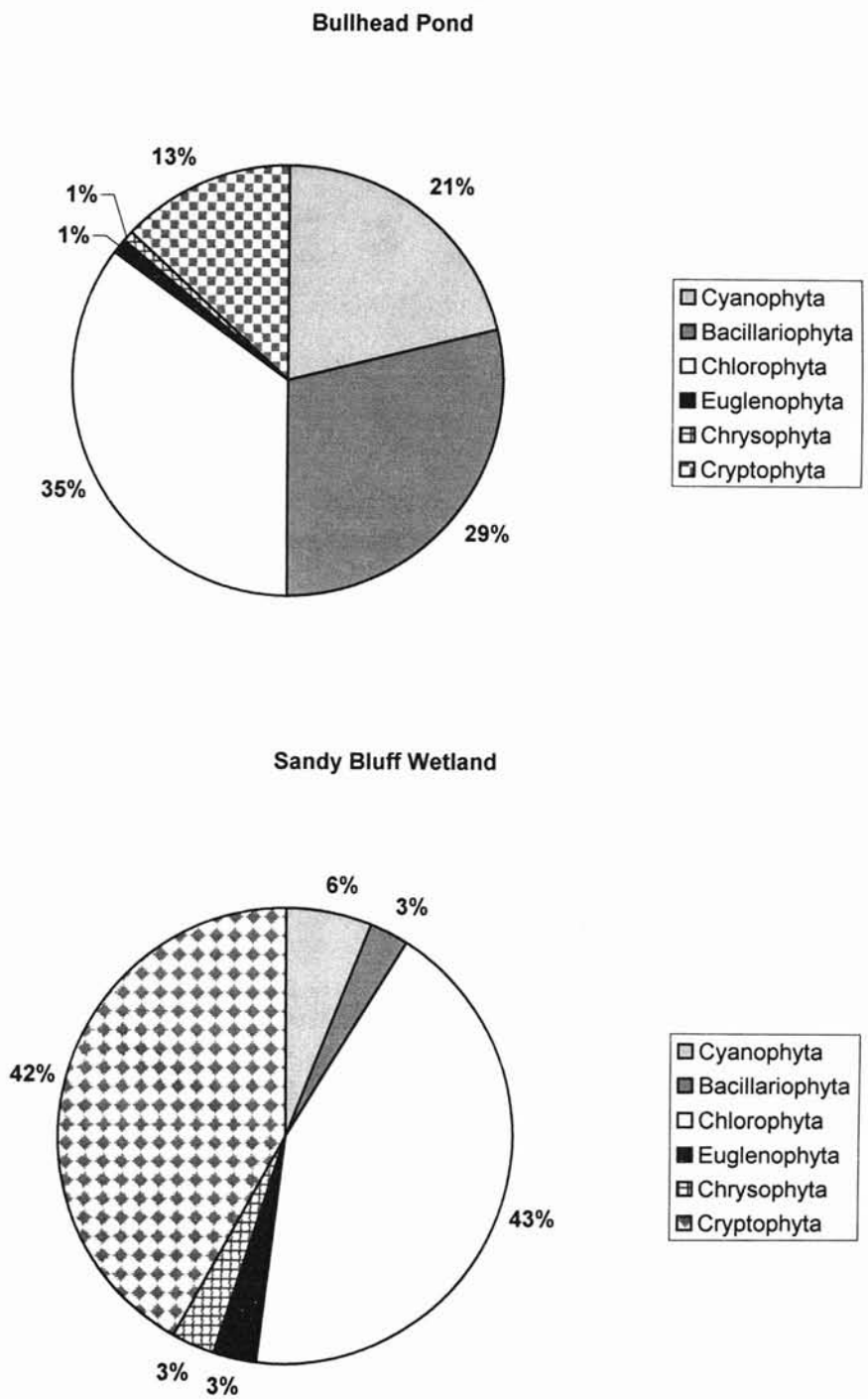

Figure 7. Combined taxonomic composition of phytoplankton in Bull Head and Sandy Bluff wetland, 2000 - 2001.

APPENDIXES

Appendix A. Mean, standard error (SE), and range of selected ions, alkalinity, dissolved oxygen, temperature, specific conductivity, and pH in Bull Head and Sandy Bluff wetland between 2000 and 2001.

	Ca^+ (mg/L)	Mg^+ (mg/L)	Alkalinity (mg/L $CaCO_3$)	Cl^- (mg/L)	SO_4^- (mg/L)	D.O. (mg/L)	Sp. Cond. (mg/L)	Temp. (C^o)	pH
Bull Head									
n[a]	8	8	8	8	8	56	56	56	18
mean	86	84	144	50	59	5.7	519	25.6	8.5
SE	4.0	3.9	5.0	3.6	3.2	0.3	9.4	0.4	0.1
range	66 -111	48 - 98	115 - 175	34 - 74	44 - 85	0.9 - 9.7	412 - 757	19.2 - 30.0	7.4 - 9.1
Sandy Bluff									
n	8	8	8	8	8	59	59	59	18
mean	109	84	147	37	73	6.6	511	26.3	8.4
SE	3.8	5.2	2.4	1.3	3	0.3	6.9	0.4	0.1
range	73 - 133	56 - 122	132 - 160	31 - 43	54 - 87	3.2 - 10.8	412 - 590	19.9 - 30.2	7.4 - 9.0

[a] sample size

Appendix B. Depth (meters) profile of dissolved oxygen,specific conductivity, temperature and pH of Bull Head wetland, 2000.

	May 31	Jun 6	Jul 27	Jul 5	Aug 10	Aug 31	Sep 9	Sep 30
Dissolved Oxygen (mg/L)								
0.0	8.10	9.00	4.70	7.40	6.60	4.90	5.90	9.20
0.5	8.20	9.40	4.30	7.10	4.80	4.50	4.90	9.00
1.0	7.90	9.70	4.20	7.10	4.00	4.30	-	-
1.5	7.90	9.00	3.90	4.30	3.40	-	-	-
2.0	7.70	7.80	3.90	-	-	-	-	-
Sp. Conductivity (*u* s/cm)								
0.0	480	474	476	476	474	525	540	594
0.5	480	473	477	477	484	523	531	573
1.0	481	474	477	478	487	521	-	-
1.5	481	480	477	-	-	-	-	-
2.0	487	495	477	-	-	-	-	-
Temperature (Celcius)								
0.0	23.6	25.5	24.7	27.9	30.0	26.7	27.0	19.2
0.5	23.4	25.3	24.7	27.8	29.4	26.7	27.0	19.3
1.0	23.0	25.4	24.7	27.9	29.2	26.5	-	-
1.5	23.0	25.5	24.7	27.2	-	-	-	-
2.0	22.9	24.3	24.7	-	-	-	-	-
pH[a]	8.70	8.50	7.80	8.30	8.80	8.60	8.70	8.80

[a]pH = composite sample

Appendix C. Depth (meters) profile of dissolved oxygen,specific conductivity, temperature and pH of Bull Head wetland, 2001.

	Apr 30	May 10	Jun 10	Jun 30	Jul 9	Jul 30	Aug 4	Aug 25	Sep 9	Sep 30
Dissolved Oxygen (mg/L)										
0.0	6.00	5.10	8.90	5.40	6.30	4.20	2.70	6.00	7.20	6.10
0.5	5.80	4.80	9.20	5.20	5.80	3.80	2.60	-	-	4.60
1.0	5.80	4.50	1.20	3.90	5.00	-	-	-	-	-
1.5	3.10	2.80	0.90	-	-	-	-	-	-	-
Sp. Conductivity (us/cm)										
0.0	481	478	459	412	556	549	600	600	630	673
0.5	485	482	456	415	553	549	599	-	-	640
1.0	485	483	649	423	553	-	-	-	-	-
1.5	623	660	757	-	-	-	-	-	-	-
Temperature (Celcius)										
0.0	23.3	24.4	28.9	29.2	29.1	27.6	28.6	26.0	26.1	20.4
0.5	23.4	23.7	28.9	28.6	29.1	27.6	28.6	-	-	20.9
1.0	23.4	23.3	25.5	27.9	28.5	-	-	-	-	-
1.5	23.1	23.1	23.4	-	-	-	-	-	-	-
pH[a]	8.60	7.50	9.00	9.10	8.80	8.90	8.30	8.80	8.50	7.70

[a]pH = composite sample

Appendix D. Depth (meters) profile of dissolved oxygen,specific conductivity, temperature and pH of Sandy Bluff wetland, 2000.

Dissolved Oxygen (mg/L)	May 31	Jun 6	Jul 27	Jul 5	Aug 10	Aug 31	Sep 9	Sep 30
0.0	8.20	9.60	6.90	8.60	6.50	5.30	7.80	10.70
0.5	8.20	9.70	6.60	8.30	5.80	5.00	7.80	10.80
1.0	8.40	9.80	6.50	8.20	5.30	5.00	7.70	10.70
1.5	8.10	9.00	6.20	8.20	5.10	-	-	-
Sp. Conductivity (*u* s/cm)								
0.0	481	490	469	416	415	477	502	559
0.5	487	490	468	417	412	477	502	558
1.0	487	492	468	420	418	477	502	558
1.5	491	502	467	435	419	-	-	-
Temperature (Celcius)								
0.0	23.9	25.7	24.6	28.2	30.2	27.0	28.4	19.9
0.5	23.9	25.7	24.6	28.2	30.2	27.0	28.4	19.9
1.0	23.6	25.7	24.6	28.2	30.1	27.0	28.4	19.9
1.5	23.6	25.7	24.6	28.2	29.8	-	-	-
pH[a]	8.60	8.60	8.10	8.60	8.70	8.60	8.50	8.70

[a]pH = composite sample

Appendix E. Depth (meters) profile of dissolved oxygen,specific conductivity, temperature and pH of Sandy Bluff wetland, 2001.

	Apr 30	May 10	Jun 10	Jun 30	Jul 9	Jul 30	Aug 4	Aug 25	Sep 9	Sep 30
Dissolved Oxygen (mg/L)										
0.0	5.10	3.80	7.30	4.00	6.90	4.80	4.70	5.90	7.40	8.60
0.5	5.00	3.40	7.70	4.00	6.80	4.70	3.60	5.90	7.90	8.40
1.0	4.90	4.00	7.40	4.50	7.20	4.10	3.20	-	-	-
1.5	4.80	3.90	6.70	-	-	-	-	-	-	-
Sp. Conductivity (us/cm)										
0.0	549	545	509	471	583	548	579	583	590	561
0.5	549	546	505	467	581	554	579	583	588	559
1.0	549	542	505	464	581	553	578	-	-	-
1.5	549	543	507	-	-	-	-	-	-	-
Temperature (Celcius)										
0.0	25.0	25.1	29.0	28.8	29.1	28.6	30.0	27.4	25.4	21.1
0.5	24.1	24.7	28.6	28.6	29.1	28.6	29.0	27.4	25.3	21.1
1.0	24.1	24.5	28.5	28.2	28.8	28.6	29.9	-	-	-
1.5	24.0	24.3	28.6	-	-	-	-	-	-	-
pH[a]	8.10	7.40	8.30	8.10	8.40	8.30	7.90	8.20	8.70	8.40

[a]pH = composite sample

Appendix F. Algal taxa present in Bull Head and Sandy Bluff wetland 2000 - 2001.

	Bull Head Wetland		Sandy Bluff Wetland	
	2000	2001	2000	2001
Cyanophyta				
Anacystis spp.	x	x	x	x
Chroococcus sp.		x	x	x
Dactylococcopsis acicularis	x	x	x	x
Dactylococcopsis fascicularis	x	x		x
Dactylococcopsis smithii	x	x		x
Dactylococcopsis sp.	x	x		x
Merismopedia s p.			x	x
Merismopedia tenuissima			x	
Anabaena sp.		x	x	x
Oscillatoria sp.	x	x	x	x
Spirulina sp.		x		
Bacillariophyta				
Centrales				x
Pennales	x	x	x	x
Chlorophyta				
Botryococcus sp.		x		
Coelastrum sp.		x	x	
Crucigenia apiculata	x	x	x	x

Appendix G. Number, mean, standard error (SE), and range of fish species captured by hook and line fish sampling in Bull Head and Sandy Bluff wetland during 2000 - 2001.

Species	Fish Length (cm)				Fish Length (cm)			
	Number[a]	Mean	SE	Range	Number[a]	Mean	SE	Range
	Bull Head Wetland							
	2000				2001			
Micropterus salmoides	7	20.8	1.75	13.5 - 27.0	16	29.8	2.05	15.2 - 45.1
Lepomis macrochirus	21	15.8	1.04	9.39 - 25.1	26	17.1	1.15	8.13 - 35.1
Lepomis microlophus	20	17.0	1.24	8.89 - 27.7	26	15.1	0.71	9.65 -24.1
Pomoxis nigromaculatus	0.0	0.0	0.0	0.0	7	28.9	0.95	24.1 - 32.0
	Sandy Bluff Wetland							
	2000				2001			
Micropterus salmoides	16	28.7	2.48	14.0 - 45.5	10	21.1	2.19	13.2 - 34.3
Lepomis macrochirus	20	15.4	0.97	8.63 - 22.6	20	16.7	0.56	9.65 - 20.6
Lepomis microlophus	12	15.6	1.67	9.14 - 21.1	11	16.4	1.45	10.2 - 24.4
Pomoxis nigromaculatus	0.0	0.0	0.0	0.0	4	23.2	1.08	20.3 - 25.4

[a] Number of fish captured during 4 hours of fishing.

Appendix H. Number, mean, standard error (SE), and range of fish species captured by electrofishing in Bull Head and Sandy Bluff wetland during 2001.

		Fish Length (cm)		
Species	Number[a]	Mean	SE	Range
		Bull Head Weltand		
Micropterus salmoides	3	6.35	NA	NA
Lepomis macrochirus	85	7.4	0.174	5.08 -12.7
Lepomis microlophus	39	9.57	0.620	5.08 - 19.1
Pomoxis nigromaculatus	3	8.47	0.847	7.62 - 10.2
Ameiurus melas	1	35.6	NA	NA
		Sandy Bluff Wetland		
Micropterus salmoides	7	9.25	0.770	7.62 -11.4
Lepomis macrochirus	33	8.69	0.350	5.08 -12.7
Lepomis microlophus	3	7.62	NA	NA
Pomoxis nigromaculatus	0	0.00	NA	NA
Ameiurus melas	0	0.0	NA	NA

[a]Equals number of fish captured during 45 minutes of electrofishing.

VITA 2

William J. Lints

Candidate for the Degree of

Master of Science

Thesis: SEASONAL DYNAMICS OF NUTRIENTS AND AQUATIC BIOTA IN TWO WETLANDS NEAR A SWINE CONFINED ANIMAL FEEDING OPERATION

Major Field: Environmental Science

Biographical:

Education: Received Associates of Science degree in Biology from Mohawk Valley Community College in May 1995; received Bachelor of Science degree in Biology with a minor in Chemistry from Oral Roberts University, May 1999. Completed the requirements for the Master of Science degree with a major in Environmental Science at Oklahoma State University in August 2002.

Experience: Born in the foothills of the Adirondack Mountains in New York State, worked with charter fishing, hunting, and camping tours year round. Employed by the United States Fish and Wildlife Service through the Student Career Experience Program; worked as an Environment Scientist from 1998 to present.

Professional Memberships: Oklahoma Academy of Science