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WATER AND SOIL CHEMISTRY IN AGRICULTURAL WETLAND LANDSCAPES IN THE PRAIRIE POTHOLE REGION OF EASTERN SOUTH DAKOTA

BY

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1998

WATER AND SOIL CHEMISTRY IN AGRICULTURAL WETLAND LANDSCAPES IN THE PRAIRIE POTHOLE REGION OF EASTERN SOUTH DAKOTA

This thesis is approved as a creditable and independent investigation by a candidate for the Master of Science degree and is acceptable for meeting the thesis requirements for this degree. Acceptance of this thesis does not imply that the conclusions reached by the candidate are necessarily the conclusions of the major department.

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Abstract

WATER AND SOIL CHEMISTRY IN AGRICULTURAL WETLAND LANDSCAPES IN THE PRAIRIE POTHOLE REGION OF EASTERN SOUTH DAKOTA

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This study investigated the nutrient filtering capability of four seasonal and two semipermanent wetlands in the Prairie Pothole Region of eastern South Dakota. The wetlands were situated in two farming systems near Madison, SD. Wetlands were instrumented with observation wells arranged in two axes extending from the wetland border to upland sites. Nitrate-nitrogen and orthophosphate concentrations from wetland surface water and surrounding groundwater were determined on a two-week cycle throughout the 1994 and 1995 growing season using a Hach DR/2000 spectrophotometer. Results indicated that NO_3-N concentrations were higher in semipermanent wetlands (8.63 mg L^{-1}) than seasonal wetlands (4.29 mg L^{-1}) in 1994 and followed the same trend in 1995. Orthophosphate concentrations were higher in seasonal wetlands (0.57 mg L^{-1}) than in semipermanent wetlands (0.27 mg L^{-1}) in 1994 and also followed the same trend in 1995.

Soil nutrient analysis on all observation well soil cores was determined by the SDSU Soil Testing Laboratory. Soil nitrate-N, total-N, and available phosphorus concentrations were analyzed according to landscape position and depth. Results indicated that seasonal wetland landscapes may be better denitrifiers than semipermanent wetland landscapes. Significant differences in nitrate-N concentrations by landscape position and depth were found in seasonal wetland landscapes, but not in semipermanent wetland landscapes.

Determination of total phosphorus concentrations from the top 15 cm of the observation well soil cores showed an increase in total P from upland landscape positions to lowland landscape positions. Total P concentrations were also determined from sediment samples from one seasonal wetland. Results showed a concentric zonation of phosphorus with concentrations ranging from 0.673 mg P g^{-1} soil in the outer edges to 0.513 mg P g^{-1} soil near the center of the wetland. A phosphorus adsorption isotherm was determined on the same sediments and showed the maximum adsorption concentration calculated from the Langmuir equation averaged 0.36 mg P g^{-1} soil more than existing phosphorus concentration in the wetland sediment. Although the wetland still retained the capacity to sorb phosphorus, the overall capacity for agricultural wetlands to sorb P may be accelerated due to erosion and direct fertilization compared to non-agricultural wetlands.

Table of Contents

List of Abbreviations

Abbreviation	Explanation
PPR	Prairie Pothole Region
WSW	wetland surface water
WGW	wetland groundwater
UGW	upland groundwater
SAS	statistical analysis systems
GLM	general linear model
F test	parametric statistical analysis
g kg^{-1}	gram per kilogram
mg L^{-1}	milligram per liter
mg g^{-1}	milligram per gram
cm	centimeter
m	meter
ha	hectares
SEAS	seasonal wetland
SEMI	semipermanent wetland

List of Tables

Page

Page

List of Figures

CHAPTER 1

Wetland surface water and surrounding groundwater quality

INTRODUCTION

Until recently, wetlands in agricultural landscapes have been viewed as areas to drain, fill, or alter in some way to increase farm production by reclaiming rich soils. In the lower 48 states during the last century and the first two-thirds of this century, half of the original 80 million ha of wetland habitat were converted for agriculture as well as other non-wetland uses (Patrick 1994). These wetlands were drained for production agriculture because they were generally flat and lent themselves to mechanized farming, had high organic matter contents compared to surrounding uplands because of their formation under wet conditions, and were high in fertility and water holding capacity because of soil movement caused by erosion from surrounding landscapes (Patrick 1994, Reddy and Gale 1994). Little was known about the ability of wetlands to filter nutrients and pollutants, recharge groundwater, and provide flood control.

Depressional wetlands formed by glaciation during the Pleistocene era approximately 10,000 years ago exist in an area of central North America commonly known as the Prairie Pothole Region (PPR). This area consists of some of the most important waterfowl production expanses in North America with estimates that 50 to 75% of all waterfowl production comes from this region . The PPR extends from south-central Canada into the north-central United States, covers approximately 780,000 square kilometers, and includes portions of North and South Dakota, Minnesota, Iowa, and the

Canadian provinces of Manitoba, Saskatchewan, and Alberta (Mitsch and Gosselink 1993a). Among the PPR states, South Dakota has lost approximately 35% of its wetlands (Tiner 1984). This compares to a 49% decrease in North Dakota, 42% in Minnesota, and 89% in Iowa (Mitsch and Gosselink 1993a).

With the inception of the Swampbuster Provision of the Food Security Act of 1985, wetland drainage for agricultural production has essentially been curtailed. Under this provision, conversion of a wetland to cropland after 1985 would result in the farm operator becoming ineligible for price support payments, farm storage facility loans, crop insurance, and disaster payments (Carey et al. 1990). In addition to Swampbuster, programs such as the Conservation Reserve Program and the Wetland Reserve Program have taken steps to restore some of the lost wetland acreage in the United States. Even with federal protection programs, it may be beneficial to find other relationships between agriculture and wetlands to convince farm operators of the benefits wetlands provide; not only for the landowner, but the general public as well.

Relationships to be studied are wetland water quality, wetland nutrient filtering, and nutrient retention in upland landscapes. These are important relationships to be studied because of the current lack of information concerning water quality in wetlands in the PPR, especially wetlands in agricultural landscapes. In recent years, the eutrophication of surface waters as well as groundwater contamination have become major concerns in agricultural areas. A primary concern about nitrate is its potential to leach into groundwater aquifers where a large majority of rural residents obtain their drinking water. This concern stems largely from the potential of nitrate to pose serious health threats to humans. Infants under 6 months of age can develop a potentially fatal condition called methemoglobinemia, more commonly known as blue-baby syndrome, from high nitrate-N concentrations in drinking water (USDA 1996). This has led the Environmental Protection Agency (EPA) to establish a 10 mg L^{-1} standard as the maximum contaminant level (MCL) in drinking water (USDA 1996). Increased rates of gastric cancer in areas of Columbia, Japan, and Chile have been attributed to high nitrate intake (Mirvish 1983) while areas of China that experience high nitrate concentrations in their drinking water have shown an increased risk of esophageal cancer (Weisburger 1986).

Phosphorus is another nutrient that impacts water quality in agricultural areas. Over 75 percent of phosphorus loss from cropland is in runoff to surface water (USDA 1996). Phosphorus, once a limiting nutrient, causes accelerated eutrophication of these surface waters.

Beyond the water quality aspect, nutrient runoff and leaching can represent a financial loss to the landowner. Nutrients transported from the field are unavailable for crop growth and need to be replaced. Yearly N and P amendments mean dollars out of the producer's pocket when nitrogen is continually being leached below the root zone or phosphorus is being added to a wetland ecosystem due to runoff.

Knowledge about wetland water quality is important for the development of water quality standards that meet regulatory requirements and to further the understanding of wetland nutrient removal (Rickerl et al. 1995). In order to address the relationship between agricultural landscapes and water quality of wetlands, questions posed for this study were: 1) Does water quality differ among different wetland classes?; and 2) Does landscape position influence surface and groundwater quality?

To address these questions, two hypotheses were tested: 1) Water quality differs among wetland classes; and 2) Water quality in wetlands differs among landscape positions. More specifically, the objective of this study was to determine the effects of wetland classification and landscape position on the water quality of wetland areas that are hydrologically linked to groundwater in agricultural areas in the PPR of eastern South Dakota.

LITERATURE REVIEW

Before addressing how wetlands deal with nutrient inputs, it is important to understand wetland classification and the relationship different wetland classes have with groundwater. This review will then describe the potential sources, forms, and transport of nutrients into wetlands along with the abatement functions of wetlands.

WETLAND CLASSIFICATION

The Cowardin et al. (1979) classification system grew out of the need to understand and describe wetland and deepwater habitats for the purposes of inventory, evaluation, and management across the United States. This structure is hierarchical, progressing from Systems and Subsystems at its most general, to Classes, Subclasses, and Dominance Types (Cowardin et al. 1979). Of particular interest are the Water Regime Modifiers applied at the Class level to describe hydrologic characteristics. The two modifiers used in this thesis are Semipermanently Flooded and Seasonally Flooded. According to Cowardin et al. (1979), semipermanently flooded wetlands are those in which surface water persists throughout the growing season in most years. Seasonally flooded wetlands are those where surface water is present for extended periods especially early in the growing season, but is absent by the end of the season in most years.

Another wetland classification system used extensively in the glaciated northern prairie region is that by Stewart and Kantrud (1971). This system uses seven classes to define a wetland based on vegetation zones occupying the deepest portion of the wetland basin. Each zone is also defined according to its hydrologic characteristics. In this system, seasonal and semipermanent ponds (Class III and IV respectively) are similar to seasonally flooded and semipermanently flooded wetlands based on the Cowardin system.

An important aspect of northern prairie wetlands is the relationship of surface water and groundwater. Wetland groundwater flow can be classified into three categories: recharge, flowthrough, and discharge (Richardson et al. 1992). Recharge wetlands occur when the water level in the wetland is higher than the surrounding water table and groundwater flows out of the wetland (Mitsch and Gosselink 1993b). Groundwater inflows occur when the surface water level of a wetland is hydrologically lower than the surrounding groundwater (Mitsch and Gosselink 1993b). Water tables can also slope into some areas of a wetland and away from the remainder, resulting in a flow through condition (Winter 1989). Although wetlands occupying depressional areas can act as recharge sites, seasonal groundwater flow reversals can occur (Labaugh et al. 1987).

It is generally thought that semipermanent wetlands are associated with flowthrough and discharge groundwater flows while seasonal wetlands are associated with groundwater recharge flows.

NITROGEN

Nitrogen is a macronutrient essential to the growth of plants. The primary source of soil nitrogen is the atmosphere where it exists as the gaseous molecule N_2 (Stevenson 1965). The outgassing of the earth's interior during its development created this elemental form. Today, it is estimated that the bulk of nitrogen (approximately 98%) exists in the lithosphere and the remainder is found in the atmosphere (Stevenson 1982). The majority of nitrogen found today in soil is bound to organic matter, although some is mineralized to soluble forms. Soluble forms, essentially nitrate (NO_3^-) and ammonium (NH_4^+), are required for plant growth. Although these nutrients are extremely important to plant growth, excessive concentrations may negatively impact aquatic environments and groundwater supplies.

According to Stevenson (1965), the nitrogen content of soils will approach an equilibrium in a natural ecosystem. This equilibrium depends on factors such as climate, type of vegetation, nature of the terrain, physical characteristics of the soil, and activities of the microflora and microfauna. The soil-nitrogen equilibrium is dynamic and any change in the environment will lead to a new equilibrium. Disruptions of the equilibrium through activities such as cultivation will produce a marked change in the nitrogen content of the soil (Stevenson 1965). The nitrogen content of cultivated fields will generally decline over time, and, as a result, nitrogen required for plant growth must come from external sources such as fertilizers. Soluble forms of N from fertilizers and other sources can be translocated through run-off and leaching, impacting the environment.

SOURCES OF NITROGEN TO WETLANDS

A. Atmospheric deposition

Atmospheric deposition can be the major source of nitrogen in wetlands where precipitation directly on the wetland is the primary source of water. But concentrations in precipitation are generally lower than those in surface water, especially surface waters receiving point and nonpoint source pollution (Johnston 1991). According to Bowden (1987), nitrogen loading from atmospheric deposition typically average 0.5 to 1.0 g N m^{-2} $year^{-1}$ as ammonium and nitrate.

B. Nitrogen fixation

Nitrogen fixation converts elemental nitrogen (N_2) into organic forms. Although carried out by a host of different organisms, the common mechanism involved is the reduction of nitrogen to ammonia

$$N_2 + 6H + 6e^- \rightarrow 2NH_3$$

which is then combined with organic acids to form amino acids and proteins (Brady 1990).

$$NH_3 + \text{organic acids} \rightarrow \text{amino acids} \rightarrow \text{proteins}$$

This process occurs in organisms containing the enzyme nitrogenase which catalyzes the system to fix N_2 (Buresh et al. 1980). A variety of symbiotic associations as well as asymbiotic (free-living) organisms have the ability to fix nitrogen. Nitrogen fixation is known to occur within the water column, on the soil surface, in the aerobic and anaerobic layer of flooded soils, in the root zone of plants, and on the leaf and stem surfaces of plants (Buresh et al. 1980). Fixation rates of N in wetlands can range from a few hundredths g N m^{-2} y^{-1} to about 10 g N m^{-2} y^{-1}, with most values around 1g N m^{-2} y^{-1} (Bowden 1987).

C. Hydrologic inputs

According to Bowden (1987), it is difficult to summarize the importance of nitrogen inputs to wetlands by either surface or subsurface inputs. Leaching rates to a wetland will be a function of the activities on upland areas rather than the type of wetland (Bowden 1987). Reddy and Reddy (1987) cite a number of factors that play a vital role in the movement of N from source to pond including soil characteristics, N transformations in the watershed, surface runoff and subsurface flow, vegetation, rainfall, and temperature. Because land-use patterns in the PPR are dominated by intensive agriculture, prairie marshes frequently experience high nutrient loadings (Neely and Baker 1989). Davis et al. (1981) reported Eagle Lake Marsh inputs of 210 kg NO_3-N ha^{-1} and 9.1 kg NH_4-N ha^{-1} from a watershed dedicated to row-crop cultivation of corn and soybeans. According to Neely and Baker (1989), nitrogen is normally applied at rates in the range of 125 to 200 kg ha^{-1} in midwestern corn production. The amounts lost with agricultural drainage (both surface runoff and subsurface leaching) can range from 10 to 50% of the N applied (Neely and Baker 1989). Concentrations of nitrate in surface runoff are usually less than 5 mg L^{-1} and can range from 10 to 20 mg L^{-1} in subsurface drainage due to leaching losses (Neely and Baker 1989).

NITROGEN POOLS IN WETLANDS

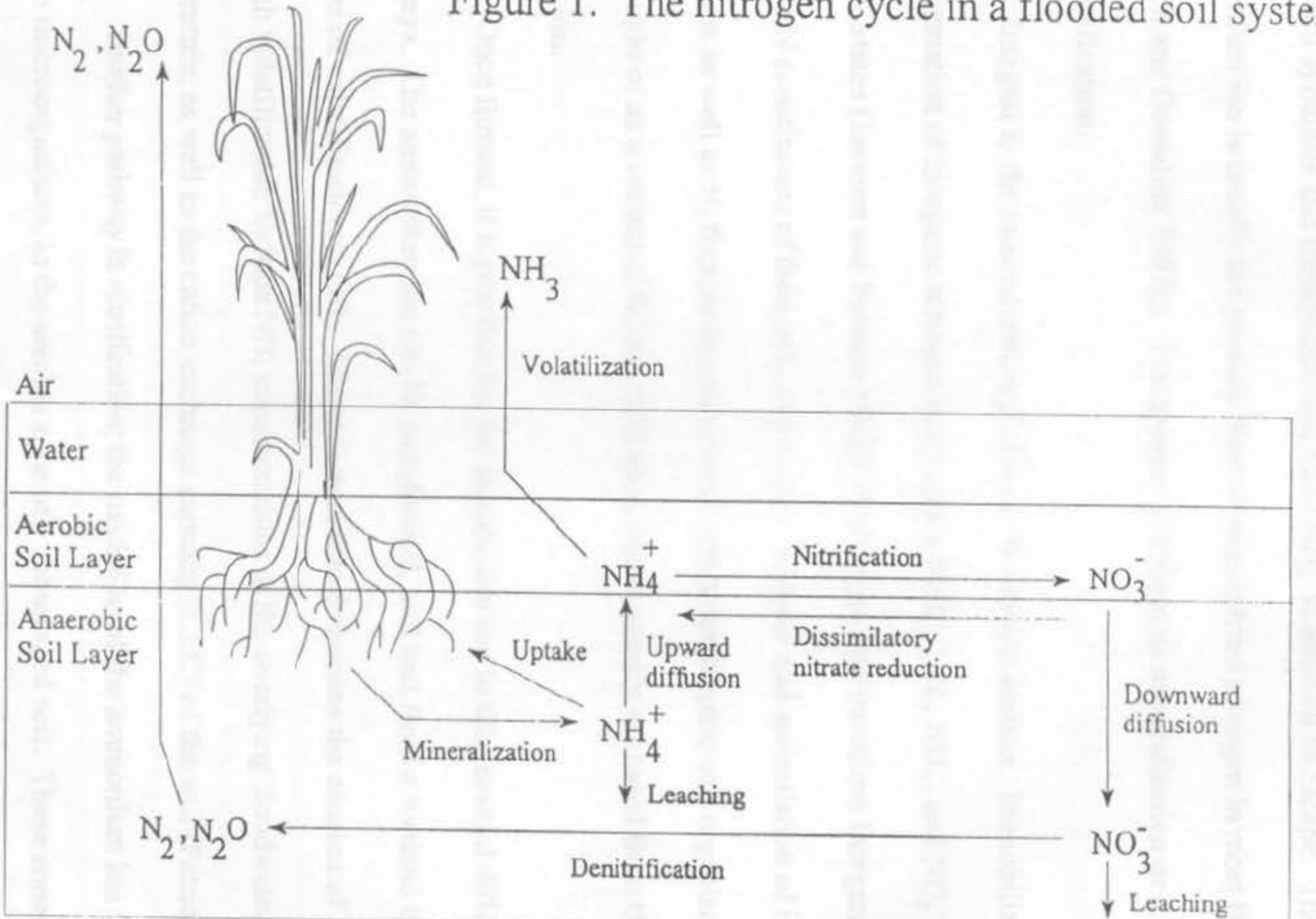
Nitrogen can exist in several different pools in a wetland. The largest pool of N exists in the sediment. This source of N is roughly one and sometimes two orders of magnitude larger than the next largest pool, plant biomass (Bowden 1987, Johnston 1991). Although there is a significant amount of N within wetland plants, it is also the most fluctuating pool due to the variability in above ground plant biomass (Johnston 1991). Inorganic N in wetland sediments and overlying water forms a third pool. This pool is usually an order of magnitude lower than plant biomass N and several orders of magnitude less than sediment total N (Bowden 1987). The balance between the supply of inorganic N by mineralization and the uptake of this form by wetland plants insures that this pool is small and turns over rapidly (Bowden 1987).

THE NITROGEN CYCLE

The soil/water interface in wetlands usually contains a thin oxidized surface layer overlying a reduced layer. The oxidized layer is created by the diffusion of dissolved O_2 from the water column. The thickness of this oxidized layer is determined by the net effect of O_2 consumption in the soil and the amount of O_2 supplied by the overlying water (Buresh et al. 1980). This interface has important implications in the nitrogen cycle (Fig. 1).

In anaerobic conditions, the decomposition of organic matter leads to a greater concentration of NH_4^+ than would normally be found in aerobic conditions (Patrick 1982). Anaerobic heterotrophic bacteria operate at a lower energy level than their aerobic

Figure 1. The nitrogen cycle in a flooded soil system.



Adapted from Patrick (1982)

counterparts and decomposition occurs at a much slower rate. Less demand is placed on N for cell synthesis and metabolism and more NH_4^+ is allowed to escape. Thus, the ammonium ion is usually the primary form of mineralized nitrogen in most flooded soils (Mitsch and Gosselink 1993c). This process is known as mineralization or ammonification.

Integral to the mineralization process is N immobilization. Immobilization is the transformation of inorganic nitrogen compounds (NH_4^+, NH_3, NO_3^-, and NO_2^-) into organic states (Jansson and Persson 1982). Soil organisms transform inorganic N into organic N constituents of their cells and tissues. Uptake and assimilation of inorganic N by plants as well as N_2 fixation by autotrophic and heterotrophic soil organisms can also be thought of as a variant of N immobilization but is usually excluded from this definition.

Once formed, it is possible for the ammonium ion to take several different pathways. The ammonium ion can be transformed, and lost from a wetland through the process of volatilization to NH_3. Major factors that determine the amount of N lost through volatilization include NH_3 concentrations in the overlying floodwater, pH, and temperature, as well as the cation exchange capacity (CEC) of the soil (Patrick 1982).

Another pathway is nitrification; the oxidation of the ammonium ion to nitrate by certain microorganisms, in the aerobic zone of a submerged soil. These zones include the water column, the surface layer of the soil, as well as the aerobic rhizosphere of aquatic plants. This process occurs in two steps. The first step includes the production of nitrite

(NO_2^-) by the *Nitrosomonas* organism while the second step involves further oxidation of nitrite to nitrate (NO_3^-) by the *Nitrobacter* organism (Brady 1990).

The downward diffusion of nitrate, created by nitrification, into the anaerobic soil zone of a wetland contributes to denitrification. Denitrification is the anaerobic process in which microbes convert NO_3^- to N gas in the absence of oxygen (Hanson et al. 1994). Broadbent and Clark (1965) define denitrification as a biological process by which facultatively anaerobic bacteria use nitrate in place of oxygen as a hydrogen (electron) acceptor. In the absence of oxygen, these bacteria are capable of nitrate respiration which can be expressed as:

$$C_6H_{12}O_6 + 4NO_3^- \rightarrow 6CO_2 + 6H_2O + 2N_2$$

The five valent nitrogen in nitrate is reduced stepwise to the zero-valent elemental form as follows (Brady 1990):

$$\underset{(+5)}{2NO_3^-} \rightarrow \underset{(+3)}{2NO_2^-} \rightarrow \underset{(+2)}{2NO} \rightarrow \underset{(+1)}{N_2O} \rightarrow \underset{(0)}{N_2}$$

As long as O_2 is present in soil, NO_3 - N will not undergo extensive biological and chemical reduction (Patrick 1982). After the oxygen has disappeared from a submerged soil, the need by facultative anaerobic and true anaerobic microorganisms for an electron acceptor will result in the reduction of NO_3 - N. According to Firestone (1982), almost all denitrifying bacteria are aerobic organisms capable of anaerobic growth only in the presence of N oxides. Respiration is similar, using either oxygen or N oxides as electron acceptors because many of the electron-carrying species used during denitrification are also used during O_2 respiration (Firestone 1982). The most important habitat for denitrifying microorganisms is the aerobic-anaerobic interface where nitrate can be produced by nitrification, an aerobic process, and then removed by denitrification in the anaerobic layer (Tiedje et al. 1982). Nitrate in the aerobic zone will reach the anaerobic layer by (i) diffusion, or by (ii) migration of the aerobic-anaerobic interface to the site of nitrate (Tiedje et al. 1982).

Under anaerobic conditions, nitrate can also be reduced through dissimilatory nitrate reduction (DNR) to ammonium (NH_4^+). Both processes occur under similar conditions, but the denitrification process results in the loss of N from the system while the other will conserve it. The denitrification reaction according to Tiedje et al. (1982) is expressed as:

$$2NO_3^- + 5H_2 + 2H^+ \rightarrow N_2 + 6H_2O$$

while the dissimilatory reaction looks like:

$$NO_3^- + 4H_2 + 2H^+ \rightarrow NH_4^+ + 3H_2O$$

Unlike the aerobic-anaerobic interface in flooded soils, the lack of electron acceptors (in this case NO_3^-) is probably the most limiting factor for growth in severely anaerobic habitats (Tiedje et al. 1982). During DNR, it is possible for the nitrogen ion in nitrate to accept eight electrons during respiration compared to the five electrons in denitrification. Because of this, DNR may be favored in strongly anaerobic environments because of the nitrate-N's capacity to receive an additional three electrons. Tiedje et al. (1982) hypothesized that the partitioning between denitrification and dissimilatory nitrate reduction may be a function of the ratio of available C to electron acceptor concentration

(C/e^- acceptor ratio). DNR would be more favored when the electron acceptor supply is limited (a higher C/e^- ratio) while the maximum energy-yielding denitrification process would be favored when carbon is more limiting but there is a sufficient supply of electron acceptors (a lower C/e^- ratio).

The oxidation status of wetland soils will have a great impact on the type of electron acceptor used during the process of microbial oxidation of organic compounds. The type of substance used is sequential with oxygen (O_2) being reduced first, followed by NO_3^-, manganese (Mn^{4+}) compounds, iron (Fe^{3+}) compounds, sulfates (SO_4^{2-}), and CO_2 (Jugsujinda et al. 1995). The redox potential, or oxidation-reduction potential, is a measure of the electron availability in a solution, and is often used to quantify the degree of electrochemical reduction of wetland soils (Mitsch and Gosselink 1993c). The redox potential is referred to as E_h when based on the hydrogen scale (Mitsch and Gosselink 1993c).

Oxygen is used first as the terminal electron acceptor at a redox potential between 400 and 600 mv.

$$O_2 + 4\,e^- + 4\,H^+ \rightarrow 2\,H_2O$$

Once the dissolved oxygen is depleted, the reduction of nitrate occurs at a redox potential of 250 mv.

$$2\,NO_3^- + 10\,e^- + 12\,H^+ \rightarrow N_2 + 6\,H_2O$$

As the redox potential continues to decrease to about 225 mv, manganese is transformed from manganic to manganous compounds.

$$MnO_2 + 2\,e^- + 4\,H^+ \rightarrow Mn^{2+} + 2\,H_2O$$

Iron is then transformed from ferric to ferrous form at about 120 mv, while sulfates are reduced to sulfides at -75 to -150 mv.

$$Fe(OH)_3 + e^- + 3\,H^+ \rightarrow Fe^{2+} + 3\,H_2O$$

$$SO_4^{2-} + 8\,e^- + 9\,H^+ \rightarrow HS^- + 4\,H_2O$$

Finally, under the most reduced conditions, the organic matter itself becomes the terminal electron acceptor at about -250 mv.

$$CO_2 + 8\,e^- + 8\,H^+ \rightarrow CH_4 + H_2O$$

Frequently one component is not completely reduced before reduction of the next component starts (Patrick 1982). Nitrate and Mn reduction will usually begin before O_2 has completely disappeared because of their relatively high redox potentials (Patrick 1982). However, SO_4^{2-} and CO_2 are reduced only under strictly anaerobic conditions with extremely low potentials and the reduction can actually be inhibited by O_2 or NO_3-N (Patrick 1982, Jugsujinda et al. 1995).

Denitrification is generally considered the major pathway of N removal from wetlands. Anaerobic conditions and large sources of carbon due to the slow breakdown of organic matter for energy favor denitrification. There is a high correlation between denitrification and the availability of organic material. Microorganisms use these organic carbon compounds as electron donors and as sources of cellular material (Firestone 1982). Firestone (1982) also maintained that the presence of abundant C substrate caused rapid O_2 consumption and depletion in soil microenvironments thus indirectly enhancing the potential for denitrification. Bremner and Shaw (1958) observed that the amount and type of organic matter had an influence on denitrification in waterlogged soils with readily decomposable compounds such as glucose and mannitol inducing rapid denitrification of nitrate compared to more difficult to decompose compounds. Burford and Bremner (1975) also demonstrated that there was a significant relationship between denitrification and the amount of total organic carbon ($r^2 = 0.77$) and that denitrification was very highly correlated ($r^2 = 0.99$) with water-soluble carbon or mineralizable carbon.

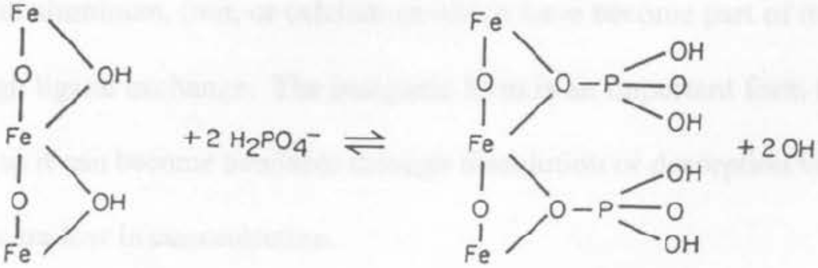
In upland landscapes, denitrification capabilities will decrease with depth in the soil profile. Cho et al. (1979) showed that the denitrification intensity of soils under irrigated conditions decreased exponentially with soil depth down to 150 cm. This decrease was attributed to microbial activity and temperature but not necessarily organic matter content. Yeomans et al. (1992) determined that the slow rate of denitrification in Iowa subsoils was not due to a deficiency of denitrifiers, but to a lack of available organic carbon utilized by these microorganisms for growth and reduction of NO_3^-. McCarty and Bremner (1992) showed that denitrifying bacteria, though reduced markedly with depth, still contained significant numbers at a depth of 150-200 cm. The ability to denitrify NO_3^- decreased greatly with depth in the absence of added organic C but did not decrease markedly with depth with the amendment of organic C. This indicated that the high level of denitrification in glucose-amended soils was due to increased growth of denitrifying bacteria promoted by the glucose. McCarty and Bremner (1992) also concluded that water-soluble organic C derived from corn and soybean residue represented a good source of available C for denitrification of nitrate in soils, but that these residues decomposed too rapidly in surface soils to be used as a leachable carbon source in subsoils, which accounted for the slow rate of denitrification in the subsoils. Although these soils were incubated in aerobic conditions and soluble C short-lived, further study by McCarty and Bremner (1993) showed that the amount of water-soluble organic C in soils incubated in anaerobic conditions was longer lived. Bradley et al. (1992) hypothesized that processes that limit denitrification rates in soils may also limit denitrification in anaerobic aquifers where these limitations may contribute to nitrate accumulations in shallow anaerobic groundwater systems. Results indicated that there was a significant relationship ($p < 0.001$, $r^2 = 0.80$) between total sediment organic content and potential denitrification rates and that reduced denitrification rates resulted in NO_3^- accumulation in anaerobic aquifers.

PHOSPHORUS

Potential sources of phosphorus to wetland environments are soil, vegetation, or phosphorus fertilizer. Transport of P in runoff from the terrestrial to the aquatic environment occurs as either soluble or particulate P (Sharpley and Menzel 1987). Particulate P includes that which is either sorbed onto soil particles or organic matter. According to Larsen (1967), phosphorus in soil is in equilibrium between solid and liquid phases. This equilibrium, $P_{solid} = P_{liquid}$, is heavily biased in favor of the solid phase which means little soluble phosphorus exists naturally in soil.

SOLUBLE FORMS OF P

The forms in which phosphorus exist in soil solution (soluble P) include phosphoric acid, H_3PO_4, and its corresponding ions, $H_2PO_4^-$, HPO_4^{-2}, and PO_4^{-3}, and the soluble complexes of these ions (Larsen 1967). The distribution of phosphorus between phosphoric acid and its ions is determined by pH. Phosphoric acid will exist at low pH but will change to its ionic forms as hydrogen [H^+] ion activity decreases. Phosphorus complexes with sodium (Na^+), potassium (K^+), magnesium (Mg^{+2}), calcium (Ca^{+2}), manganese (Mn^{+2}), aluminum (Al^{+3}), and iron (Fe^{+3}). These complexes are generally insoluble and less bioavailable than phosphoric acid and its ions. In time, phosphorus complexes will precipitate out of solution or become sorbed to a soil particle through ligand exchange, also known as anion penetration. In ligand exchange, oxygen ions on a hydrous oxide surface (such as Fe or Al oxides) can be replaced by anions such as phosphate oxyacids (Bohn et al. 1979a) and can be represented as:



Phosphorus is often associated with Al, Fe, or Ca because of their high stability constants compared to the other cations. The adsorption of phosphates onto oxides and hydroxides of iron and aluminum and subsequent precipitation as insoluble ferric phosphates and aluminum phosphates will occur in acid soils (Mitsch and Gosselink 1993c) while the precipitation of calcium phosphates dominates in soils at pH greater than 7.0 (Lindsay 1979). In addition to plant available P, other forms of soluble phosphorus may include dissolved condensed phosphates (DCP), which are compounds containing P in P-O-P bonds and dissolved organic P (DOP), which contains P-O-C bonds (Sonzogni et al. 1982). These compounds, however, usually have a short life span and are quickly converted to plant available forms (Table 35).

PARTICULATE FORMS OF P

Particulate P often comprises a high proportion of the total P in soil or water and represents the major reservoir of P to organisms in aquatic environments (Sonzogni et al. 1982). The particulate P fraction can consist of inorganic, organic, and condensed forms (Table 35).

Inorganic forms consist of those that have precipitated out of solution by complexing with aluminum, iron, or calcium or which have become part of the soil structure through ligand exchange. The inorganic form is an important form for bioavailability as it can become available through dissolution or desorption when plant available forms are low in concentration.

The condensed P compounds are generally small and comprise a small fraction of the total particulate P pool.

A significant amount of total particulate P in the soil can be in the form of organic particulate P. The organic fraction may comprise anywhere between 20 to 80 percent of the total soil phosphorus (Wells and Saunders 1960).

WETLAND SEDIMENTS

Once a substance enters a wetland, it can be stored, altered by chemical or biological action, or discharged via water or atmospheric fluxes. Howard-Williams (1985) emphasized the role of storage compartments in mediating the fluxes of nutrients through wetlands. The importance of a storage compartment from a water quality perspective is the rate of flux between storage compartments and the duration of retention in that compartment, also known as turnover time (Johnston 1991). Storage compartments have finite boundaries or carrying capacities and when the stores are full, the capacity to absorb nutrients breaks down (Howard-Williams 1985).

Soils contain by far the largest standing stock of nutrients of any wetland storage compartment (Johnston 1991). Although values can vary widely, standing stocks of soil nutrients are at least one and sometimes two orders of magnitude higher than standing stocks in vegetation. The major reason for this is the very long turnover time for soil nutrients compared to turnover time in vegetation. While storage compartments such as algae and aquatic vegetation can initially take up most of the dissolved inorganic P to store, these compartments can quickly become saturated so that long-term storage of phosphorus becomes dependent on soil absorption (Richardson 1985).

The forms and amounts of P in lake systems (in this case, wetland systems) are a function of the input of P from external sources, phosphorus output from the lake/wetland, and the interchange of P among various sediment and water components (Seyers et al. 1973, Sharpley and Menzel 1987). The amount of orthophosphate in flooded soils, swamp and marsh sediments, and shallow bodies of water depends on the capacity of the soil or sediment to release ortho-P to a solution low in P and to sorb it from a solution high in P (Patrick and Khalid 1974). Studies have shown that inorganic P added at concentrations considerably greater than those present in the interstitial water of sediments, is retained by oxides and hydrous oxides of Fe and Al (Muljadi et al. 1966) and $CaCO_3$ (Cole et al. 1953) by a sorption rather than a precipitation mechanism. Richardson (1985) showed a direct, positive relationship between the amount of extractable aluminum in the soil and that soil's ability to sorb phosphorus. Wetland types with predominately mineral soils and high aluminum content seem to be a better phosphorus sink than peatlands, meaning that peat will become saturated much more quickly than a mineral soil with a significant amount of aluminum in it.

The oxidation-reduction (redox) status of sediment also affects P in aquatic systems (Patrick and Khalid 1974). In soils and sediments exposed to oxygen, iron is in the Fe^{+3} (ferric) form, while under reduced, anaerobic conditions iron is in the Fe^{+2} (ferrous) form. This oxidation state apparently affects the phosphate equilibrium between solid and solution. Results from a study done by Patrick and Khalid (1974) determined that more phosphorus is released from soil to solution under anaerobic conditions than aerobic. Although the anaerobic soils released more P to a P-free solution, they were also capable of sorbing more P from a high-P solution. The difference between the reduced and oxidized soils in the equilibrium of P between solution and soil suggest that under reduced conditions, there is an increase of the solid material that reacts with P. The conversion of ferric oxyhydroxide to the more soluble and highly dispersed ferrous form increases the activity and the surface area of the iron compounds reactive with P. The ferric form of iron is apparently capable of binding solution P ions more firmly than the ferrous form but has less surface area exposed to the solution P, resulting in less absorption. The study done by Holford and Patrick (1979) also supports this hypothesis.

In general, phosphorus is in equilibrium between the solution phase and that which is bound by the sediment. If phosphorus in the form of solution P enters a wetland, it can be taken up by plants or sorbed by sediments. Phosphorus entering a wetland in a form that is not available for plant uptake will usually be bound to the sediment. If phosphorus inputs to a wetland system continue, there will be a point where even this storage compartment is saturated. A study done by Steward and Ornes (1975) compared phosphorus additions within wetland enclosures. Treatment rates to these enclosures simulated the quantity contained in a secondary treated domestic effluent while controls were used to simulate natural conditions. Results indicated that the marsh system seemed to have a limited capacity for assimilating continuous phosphorus additions. At a continuous rate of 2.5 kg ha^{-1} P application/week, the assimilative capacity was stressed after week 3 and was overwhelmed by week 8.

One way to abate these problems might be to include buffer strips around wetlands to act as filters for sediment and solution runoff. Cooper et al. (1987) showed that riparian areas work as filters for agricultural sediment. Results showed a significant decrease in the amount of sediment entering an aquatic system that had been buffered. With the bulk of phosphorus sorbed to soil particles, buffer strips could significantly alter the amount of this nutrient entering a wetland. Harvesting buffer strips would remove P from the system and reduce the potential to overwhelm the assimilatory capacity of a wetland.

SITE DESCRIPTION

Two farmer cooperators in Lake county, South Dakota (Fig. 2) were recruited in 1992 to participate in the study. The farms were located in the Skunk Creek-Lake Madison-Lake Herman watershed. These farms were chosen on the basis of similarities in soil type, wetland class, wetland hydrology, and proximity to each other. Principle crops for each farm location included corn, soybean, alfalfa, and some small grains and are representative of area cropland use.

The average growing season is 140 days with the last spring freeze occurring between April 30 and May 4 and the first fall frost occurring between October 5 and October 9 (S.D. Tech. Guide 1992). Average precipitation during the growing season is 40.6-45.7 cm (16-18 inches) with the majority occurring in June (S.D. Tech. Guide 1992).

Each farm contained seasonal and semipermanent wetlands (Stewart and Kantrud 1971) with comparable wetland hectares and hydric soils (Table 1). Seasonal (SEAS) wetlands had an average area of 1.38 ha with an average watershed of 14.68 ha. Semipermanent (SEMI) wetlands averaged 11.92 ha in size with an average watershed of 33.18 ha. Surface water in seasonal wetlands was usually present until mid-summer while the semipermanent wetlands stayed wet throughout the growing season for both 1994 and 1995. A physical description of each wetland in the study is listed in Table 2.

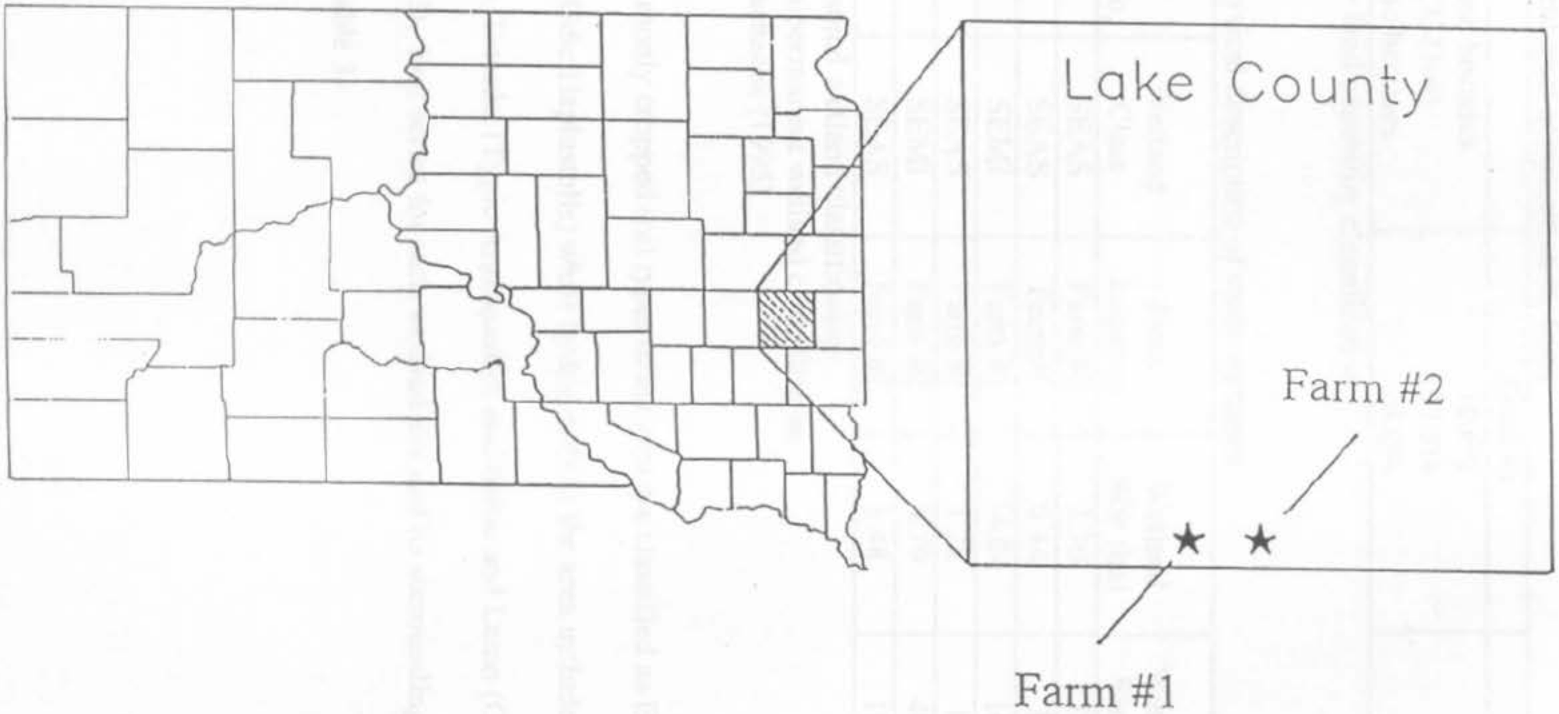


Figure 2. Location of farms in eastern South Dakota

Table 1. Field tract composition by farm.

	Farm #1	Farm #2
Wetland hectares	10.6%	6.9%
LCC (1-4)	97.5%	96.1%
Hydric hectares	24.0%	15.0%

LCC (1-4) = land capability classification

Table 2. Physical description of study wetlands.

Wetland No.	Wetland Class	Farm Location	Wetland Size (ha)	Watershed Size (ha)	Wetland Average Depth (cm)
3	SEAS	Farm #1	2.30	33.61	35.97
4	SEAS	Farm#1	0.44	3.72	17.98
5	SEMI	Farm #1	14.04	16.80	23.16
6	SEAS	Farm #2	1.28	1.41	23.77
8	SEMI	Farm #2	9.79	49.55	30.18
9	SEAS	Farm #2	1.48	19.99	19.81

SEAS = seasonal wetland classification
SEMI = semipermanent wetland classification
Source: Machacek (1995)

Commonly cropped soil types in the area are classified as Egan and Moody silty clay loams (Udic Haplustolls) while hydric soils in the area include Worthing (Typic Haplustolls), Tetonka (Typic Argiaquolls), and Baltic and Lamo (Cumulic Haplaquolls) (USDA 1973). Soil series for each wetland site and its surrounding upland soil series can be seen in Table 3.

Table 3. Soil series descriptions for wetland and upland sites.

Wetland No.	Wetland Soil Series Series	 Symbol	Upland Soil Series Series	 Symbol
3	Worthing	Wo	Egan	EbB
4	Worthing	Wo	Egan	EeC2
5	Classified Marsh Soil		Egan	EbB
6	Worthing	Wo	Egan	EbB
8	Classified Marsh Soil		Egan	EaC
9	Worthing	Wo	Egan (90%) Badus (10%)	EbB Ba

Source: Machacek (1995)

A more detailed description of each soil series is presented in Table 36 in the Appendix.

MATERIALS AND METHODS

The sampling design at each wetland site consisted of two axes extending from the wetland border to upland areas. Each axis had a sampling point located within the wetland for surface water samples and groundwater monitoring wells located at the wetland edge and 46 m from the wetland edge for a total of 6 sampling points at each wetland site. In 1993, wells were placed at each sampling point using a Gidding's hydraulic soil-probe-equipped truck (Machacek 1995). A 50.8 mm inside diameter x 122 cm length soil probe was used in conjunction with a 1.2 m and 2.4 m Kelly bar extension. Wells were 2.4 m deep at the wetland border and 3.0 m deep at the upland areas.

Upland and wetland groundwater samples were collected using a portable Masterflex sampling pump with one-half inch flexible polyurethane tubing using the third 250 ml sample for analysis. Water quality samples were collected from wetland groundwater (WGW) and upland groundwater (UGW) wells with wetland surface water (WSW) samples taken at the end of each axis. Chemical analysis was conducted on a two-week cycle throughout the growing season (approx. mid-May through September) with testing done on Farm #1 the first week and Farm #2 the next.

Nitrate (NO_3^--N) and orthophosphate (PO_4^{-3}-P) concentrations were determined using a portable Hach DR/2000 spectrophotometer. Samples were immediately measured using the cadmium reduction Method 8039 for nitrate and the Phos Ver 3 (Ascorbic Acid) Method 8048 for orthophosphate (HACH 1992).

Data analysis for water quality samples used a Proc GLM (General Linear Model) statement in SAS (Statistical Analysis Systems). The dependent variables included nitrate and orthophosphate. Means were separated using Fisher's protected least significant difference for landscape position, wetland class, and the interaction term using a LSMEANS statement.

RESULTS AND DISCUSSION

NITRATE

The probability of significant differences for landscape position, wetland class, and the interaction term for nitrate concentrations in 1994 can be seen in Table 4.

Table 4. Probability of > F(0.05) for water nitrate-N concentrations among landscape position, wetland class, and the interaction term in 1994.

Variable	Pr > F
Landscape position	0.0001
Wetland class	0.0001
Landscape position * wetland class	0.0001

By averaging across wetland class, significant differences were found for nitrate-N concentrations by landscape position. Nitrate concentrations were lowest in the wetland surface water (0.29 mg L^{-1}) and increased to the upland groundwater concentrations of 13.88 mg L^{-1} (Table 5).

Table 5. Mean water nitrate-N concentrations as effected by landscape position in 1994.

Landscape position	N	NO_3-N (mg L^{-1})
WSW (1)	68	0.29
WGW (2)	73	5.21
UGW (3)	49	13.88
Comparison*		LSD (0.05)
1 – 2		1.26
2 – 3		1.38
1 – 3		1.40

WSW = wetland surface water
WGW = wetland groundwater
UGW = upland groundwater
* Comparison based on estimate of S_D using unequally replicated means

The application of nitrogen fertilizers or production of legume N in excess of requirements of the plant-sediment ecosystem can result in the accumulation of NO_3^- in groundwater underlying cultivated land (Bradley and Chapelle 1993). The high concentrations of nitrate in the UGW underlying the cultivated fields is likely the result of excess soil N and limited denitrification potential in upland soils due to the lack of available C in the subsoils, as suggested by McCarty and Bremner (1992) and Yeomans (1992).

As groundwater nears the surface, anaerobic conditions become favorable for denitrification where available C will be present. Denitrification rates will increase and reduce NO_3-N concentrations. This process could explain the reduction in the concentrations of nitrate in the WGW and WSW positions compared to UGW.

By averaging across landscape position, significant differences were shown between seasonal and semipermanent wetland landscapes (Table 6) with seasonal wetland landscapes exhibiting lower NO_3-N concentrations (4.29 mg L^{-1}) than semipermanent wetland landscapes (8.63 mg L^{-1}).

Table 6. Mean water nitrate-N concentrations as effected by wetland class in 1994.

Wetland Class	N	NO_3-N (mg L^{-1})
SEAS	118	4.29
SEMI	72	8.63
LSD (0.05)*		1.12

SEAS = seasonal wetland classification
SEMI = semipermanent wetland classification
*Comparison based on estimate of S_D using unequally replicated means

Concentrations of nitrate-N for 1994 also showed a significant interaction (landscape position * wetland class) (Table 7, Figure 3).

Table 7. Mean water nitrate-N concentrations as effected by landscape position * wetland class in 1994.

Landscape position	Wetland class	N	NO_3-N (mg L^{-1})
WSW (1)	SEAS	32	0.10
WSW (1)	SEMI	36	0.48
WGW (2)	SEAS	50	1.77
WGW (2)	SEMI	23	8.64
UGW (3)	SEAS	36	11.00
UGW (3)	SEMI	13	16.76
Comparison*		LSD (0.05)	
1 – 1		1.82	
2 – 2		1.88	
3 – 3		2.42	

WSW = wetland surface water
WGW = wetland groundwater
UGW = upland groundwater
SEAS = seasonal wetland classification
SEMI = semipermanent wetland classification
*Comparison based on estimate of S_D using unequally replicated means

Similar trends for nitrate-N were also found in 1995 (Table 8). Significant differences in concentrations were found due to landscape position and wetland class. Although the interaction in 1995 between landscape position and wetland class was not significant, it was similar to 1994 (Fig. 4).

Table 8. Probability of > F(0.05) for water nitrate-N concentrations among landscape position, wetland class, and the interaction term for 1995.

Variable	Pr > F
Landscape position	0.0001
Wetland class	0.0048
Landscape position * wetland class	0.1002

Figure 3. Differences in nitrate-N concentrations as effected by landscape position and wetland class in 1994.

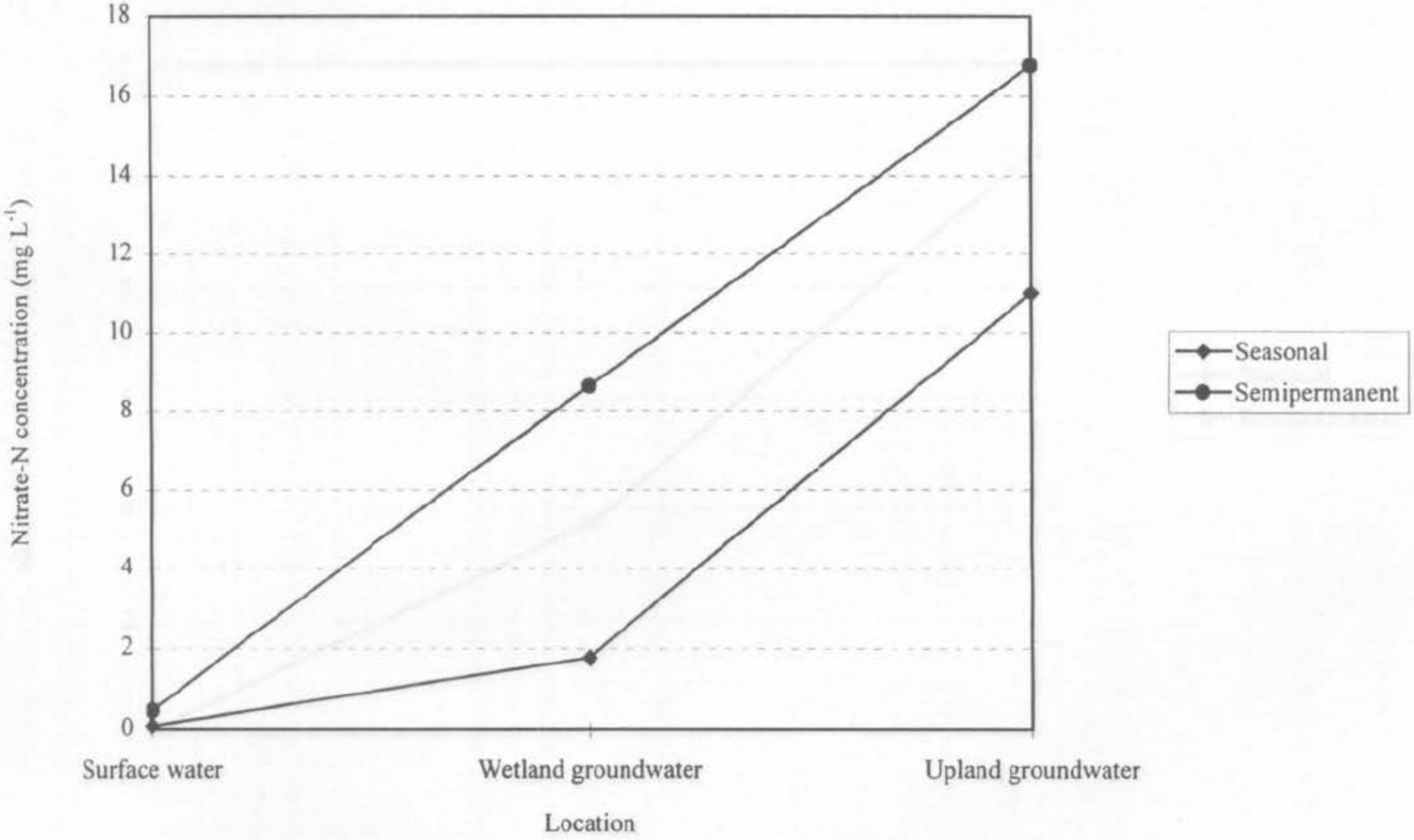
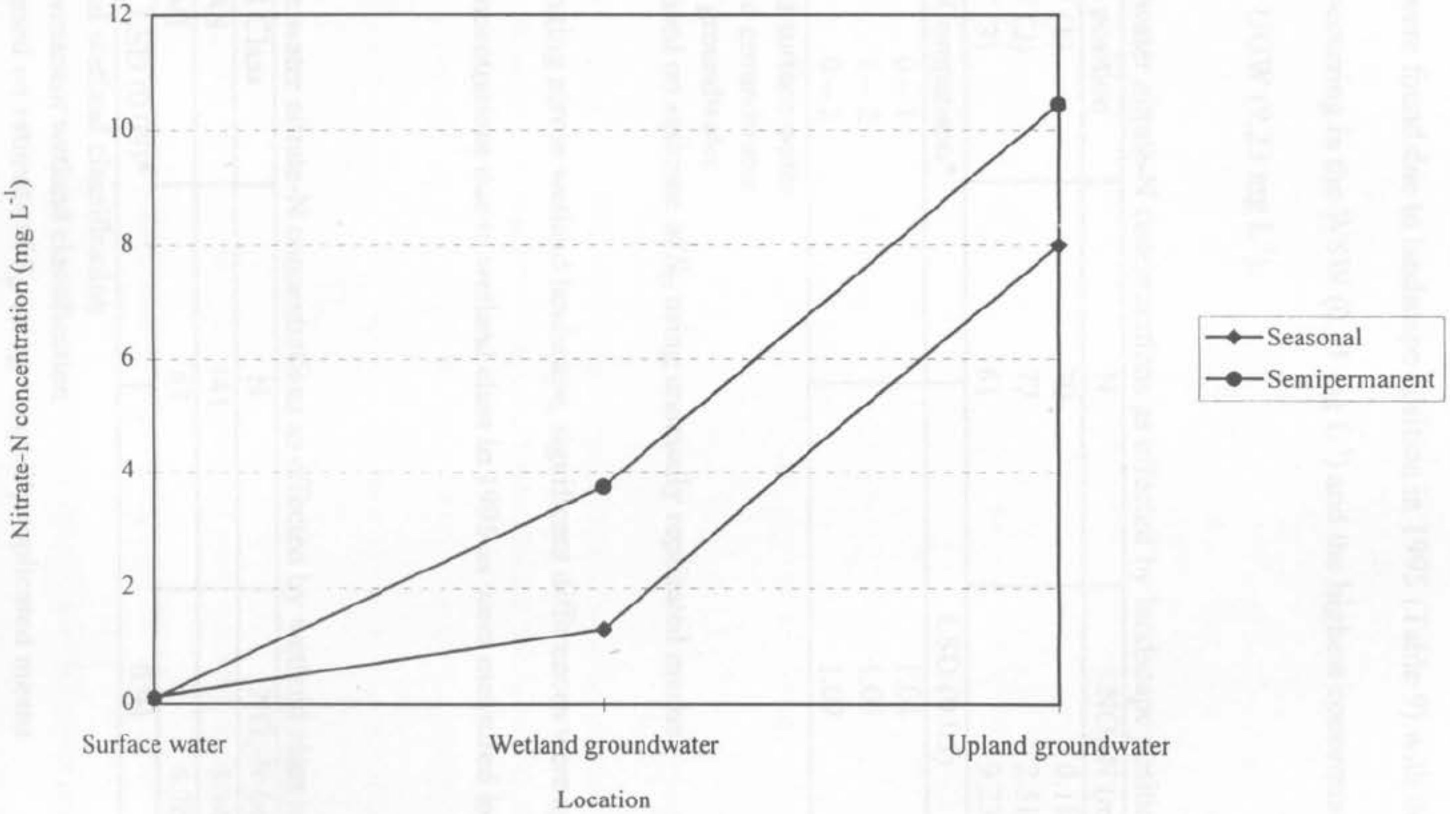


Figure 4. Differences in nitrate-N concentrations as effected by landscape position and wetland class in 1995.



By averaging across wetland class, significant differences in nitrate-N concentrations were found due to landscape position in 1995 (Table 9) with the lowest concentrations occurring in the WSW (0.11 mg L^{-1}) and the highest concentrations occurring in the UGW (9.23 mg L^{-1}).

Table 9. Mean water nitrate-N concentrations as effected by landscape position in 1995.

Landscape position	N	NO_3-N (mg L^{-1})
WSW (1)	70	0.11
WGW (2)	77	2.51
UGW (3)	61	9.23
Comparison*	LSD (0.05)	
0 – 1	1.01	
1 – 2	1.05	
0 – 2	1.07	

WSW = wetland surface water
WGW = wetland groundwater
UGW = upland groundwater
*Comparison based on estimate of S_D using unequally replicated means

By averaging across wetland landscape, significant differences were again found for nitrate-N concentrations due to wetland class in 1995 as were measured in 1994 (Table 10).

Table 10. Mean water nitrate-N concentrations as effected by wetland class in 1995.

Wetland Class	N	NO_3-N (mg L^{-1})
SEAS	141	3.14
SEMI	67	4.76
LSD (0.05)*	0.91	

SEAS = seasonal wetland classification
SEMI = semipermanent wetland classification
*Comparison based on estimate of S_D using unequally replicated means

Table 11. Mean water nitrate-N concentrations as effected by landscape position * wetland class in 1995.

Landscape position	Wetland class	N	NO_3-N (mg L^{-1})
WSW (1)	SEAS	38	0.12
WSW (1)	SEMI	32	0.10
WGW (2)	SEAS	58	1.29
WGW (2)	SEMI	19	3.73
UGW (3)	SEAS	45	8.00
UGW (3)	SEMI	16	10.45
Comparison*		LSD (0.05)	
1 – 1		NS	
2 – 2		1.62	
3 – 3		1.78	

WSW = wetland surface water
WGW = wetland groundwater
UGW = upland groundwater
SEAS = seasonal wetland classification
SEMI = semipermanent wetland classification
*Comparison based on estimate of S_D using unequally replicated means

With the exception of WSW for 1995, seasonal wetlands exhibited lower concentrations of nitrate-N in the surface water and surrounding groundwater than semipermanent wetlands. With high water table levels during the spring and early summer, seasonal wetlands may show groundwater flow reversals (Labaugh et al. 1987) by acting as discharge wetlands in the spring and becoming groundwater recharge wetlands later in the season as the water table drops. With pronounced wetting-drying cycles of the seasonal wetlands, differences in denitrification rates are enhanced and exceed those of wetlands that have water throughout the year.

Several studies have reported higher rates of denitrification in soils that have been rewet following a drying period compared to soils that were continually moist (Bremner and Shaw 1958, McKenzie and Kurtz 1976, Patten et al. 1980). These studies have indicated that denitrification was stimulated by releases of available C and N along with the reduction of soil oxygen levels by vigorous aerobic respiration following rewetting of dry soils (Groffman and Tiedje 1988, Smith and Parsons 1985). In wetland soils, the mineralization of C and N from aquatic macrophytes will occur more rapidly when oxidized than when strictly anaerobic. Brinson et al. (1981) suggested that alternating wet and dry conditions may lead to optimum litter decomposition while completely anaerobic conditions caused by constant flooding are the least favorable conditions for decomposition. Litter breakdown will release more decomposable C and N for use by organisms and cause a higher rate of denitrification during periods of rewetting.

ORTHOPHOSPHATE

The probability of > F(0.05) for landscape position, wetland class, and the interaction term for orthophosphate concentrations for 1994 can be seen in Table 12.

Table 12. Probability of > F(0.05) for orthophosphate concentrations among landscape position, wetland class, and the interaction term in 1994.

Variable	Pr > F
Landscape position	0.0001
Wetland class	0.0001
Landscape position * wetland class	0.0832

By averaging across wetland class, significant differences due to landscape position were found for orthophosphate concentrations in 1994. Ortho-P concentrations were highest in WSW (0.95 mg L^{-1}) while lowest in the upland groundwater (0.10 mg L^{-1}) (Table 13).

Table 13. Mean orthophosphate concentrations as effected by landscape position in 1994.

Landscape position	N	PO_4-P (mg L^{-1})
WSW (1)	68	0.95
WGW (2)	74	0.23
UGW (3)	49	0.07
Comparison*		**LSD (0.05)**
1 – 2		0.13
2 – 3		0.14
1 – 3		0.14

WSW = wetland surface water
WGW = wetland groundwater
UGW = upland groundwater
*Comparison based on estimate of S_D using unequally replicated means

Transport of phosphorus generally occurs in runoff as either soluble or particulate P (Sharpley and Menzel 1987). The forms in which phosphorus exists in soil solution (soluble P) include phosphoric acid, H_3PO_4, and its corresponding ions, $H_2PO_4^-$, HPO_4^{-2}, and PO_4^{-3}, and the soluble complexes of these ions (Larsen 1967). What is not taken up by plants will generally precipitate out of solution forming complexes with sodium, potassium, magnesium, calcium, manganese, aluminum, or iron or become sorbed to a soil particle through ligand exchange. Because phosphorus will usually precipitate out of solution or become complexed with soil particles, it is generally not available to be leached. In more anaerobic conditions, such as in aquatic environments, phosphorus may be released from soil to solution resulting in higher concentrations of soluble P in wetlands (Patrick and Khalid 1974).

By averaging across landscape position, significant differences in orthophosphate concentrations were found between seasonal wetland landscapes and semipermanent landscapes. Seasonal wetland landscapes exhibited a higher concentration of ortho-P

(0.57 mg L^{-1}) than semipermanent wetland landscapes (0.27 mg L^{-1}) in 1994 (Table 14).

Table 14. Mean orthophosphate concentrations as effected by wetland class in 1994.

Wetland Class	N	PO_4-P (mg L^{-1})
SEAS	120	0.57
SEMI	71	0.27
LSD (0.05)*		0.11

SEAS = seasonal wetland classification
SEMI = semipermanent wetland classification
*Comparison based on estimate of S_D using unequally replicated means

Although not significant, ortho-P concentrations for the interaction term (landscape position * wetland class) were higher in all landscape positions for the seasonal wetland landscapes (Table 15).

Table 15. Mean orthophosphate concentrations as effected by landscape position * wetland class in 1994.

Landscape position	Wetland class	N	PO_4-P (mg L^{-1})
WSW (1)	SEAS	32	1.20
WSW (1)	SEMI	36	0.69
WGW (2)	SEAS	51	0.37
WGW (2)	SEMI	23	0.09
UGW (3)	SEAS	37	0.12
UGW (3)	SEMI	12	0.03
Comparison*		LSD (0.05)	
1 – 1		0.18	
2 – 2		0.19	
3 – 3		NS	

WSW = wetland surface water
WGW = wetland groundwater
UGW = upland groundwater
SEAS = seasonal wetland classification
SEMI = semipermanent wetland classification
*Comparison based on estimate of S_D using unequally replicated means

Trends in P concentrations in 1995 were similar to 1994 (Table 16). Landscape position and wetland class showed significant differences in ortho-P concentrations although the interaction term did not .

Table 16. Probability of > F(0.05) for orthophosphate concentrations among landscape position, wetland class, and the interaction term in 1995.

Variable	Pr > F
Landscape position	0.0001
Wetland class	0.0454
Landscape position * wetland class	0.4124

By averaging across wetland class, significant differences were again found among landscape positions with the highest concentrations in the WSW (1.28 mg L^{-1}) and the lowest concentrations in the UGW (0.13 mg L^{-1}) (Table 17). This trend is similar to what was found in 1994.

Table 17. Mean orthophosphate concentrations as effected by landscape position in 1995.

Landscape position	N	PO_4-P (mg L^{-1})
WSW (1)	72	1.28
WGW (2)	77	0.32
UGW (3)	64	0.13
Comparison*		LSD (0.05)
1 – 2		0.19
2 – 3		0.19
1 – 3		0.20

WSW = wetland surface water
WGW = wetland groundwater
UGW = upland groundwater
*Comparison based on estimate of S_D using unequally replicated means

By averaging across landscape position, significant differences were found between wetland classes in 1995 with concentrations higher in the seasonal wetland landscape (0.68 mg L^{-1}) than in the semipermanent wetland landscape (0.47 mg L^{-1}) (Table 18). This trend is similar to what was measured in 1994.

Table 18. Mean orthophosphate concentrations as effected by wetland class in 1995.

Wetland Class	N	PO_4-P (mg L^{-1})
SEAS	144	0.68
SEMI	69	0.47
LSD (0.05)*		0.17

SEAS = seasonal wetland classification
SEMI = semipermanent wetland classification
*Comparison based on estimate of S_D using unequally replicated means

Although not significant, the interaction term in 1995 showed similar trends to 1994. With the exception of UGW, ortho-P concentrations were higher in the seasonal wetland landscape compared to the semipermanent wetland landscape (Table 19).

Table 19. Mean orthophosphate concentrations as effected by landscape position * wetland class in 1995.

Landscape position	Wetland class	N	PO_4-P (mg L^{-1})
WSW (1)	SEAS	40	1.43
WSW (1)	SEMI	32	1.12
WGW (2)	SEAS	58	0.47
WGW (2)	SEMI	19	0.16
UGW (3)	SEAS	46	0.13
UGW (3)	SEMI	18	0.13
Comparison*		LSD (0.05)	
1 – 1		0.27	
2 – 2		0.30	
3 – 3		NS	

WSW = wetland surface water
WGW = wetland groundwater
UGW = upland groundwater
SEAS = seasonal wetland classification
SEMI = semipermanent wetland classification
*Comparison based on estimate of S_D using unequally replicated means

The fact that seasonal wetland landscapes exhibited higher concentrations of ortho-P than semipermanent wetland landscapes may be partly due to their position in the local groundwater system and seasonal hydrologic cycle. Aquatic macrophytes are subjected to more rapid cycles of wetting-drying in seasonal wetlands, which results in more dead plant material due to flooding, drying, and freezing (LaBaugh 1987). Ulehlova (1978) and Davis and van der Valk (1978) have shown the rapid loss of phosphorus through release from standing, dead, aquatic macrophytes. Orthophosphate may be higher in seasonal wetlands due to concentration from evaporation, rapid leaching of P from aquatic macrophytes, and fertilization during farmed through dry years. The addition of phosphate fertilizers directly into the seasonal wetland system may have accelerated the wetlands capacity to assimilate phosphorus through sediment sorption. Storage compartments have finite boundaries or carrying capacities and when the stores are full, the capacity to absorb nutrients breaks down (Howard-William 1985). Once this compartment is full, solution phosphorus concentrations will be higher. This mechanism may explain why higher concentrations of P were found in the seasonal wetlands.

Another reason seasonal wetland landscapes may be higher in ortho-P than semipermanent wetland landscapes is the wetlands ability to trap phosphorus-rich runoff. Riparian area buffer strips around wetlands have been shown to work as filters for sediment and solution runoff (Cooper et al. 1987). Seasonal wetlands in agricultural landscapes may be farmed through during dry years and generally do not have large buffer strips to trap sediments before they reach the wetland. With an increase in phosphorus reaching the seasonal wetlands, the potential to overwhelm the capacity to absorb phosphorus may be accelerated and the ability to remove P diminished. As this storage compartment fills, soluble phosphorus increases in solution and may move with groundwater. In the future, wetland management should include the use of buffers to protect wetlands rather than the use of wetlands as buffers (Rickerl et al. 1995).

CHAPTER 2

Soil Analysis

INTRODUCTION

As important as it may be to monitor aquatic environments for the potential of excessive nutrient contamination, it is just as important to examine upland soil nutrient concentrations as a majority of the water can come from upland runoff. The potential threat of nitrate in our groundwater and the eutrophication of water bodies by phosphorus from surface runoff has increased the public's demand for the reduction of agricultural pollution. One way to help abate these problems is through soil testing. Regular testing for nutrient availability in soil can help alleviate the notion of the need for heavy fertilization in agricultural areas year after year. With responsible management based on an understanding of crop need and of the transport mechanisms involved with N and P, the potential for leaching or overland runoff of these nutrients into aquatic environments can be diminished.

In order to better understand nutrient distribution within an agricultural wetland landscape, observation well soil cores were analyzed by the Soil Testing Laboratory, SDSU. Nitrate-N, total-N, and available phosphorus concentrations were analyzed by depth for each soil core. Total phosphorus from the top 15 cm of each observation well soil core was also analyzed. Because orthophosphate concentrations were higher in the seasonal wetlands than the semipermanent wetlands, a closer examination of wetland sediments in a seasonal wetland was conducted. Total P concentrations in wetland sediment samples were analyzed and an adsorption isotherm test done to determine the sorption capacity of the wetland sediment.

Questions posed for this portion of the study were: 1) Does landscape position and sampling depth affect soil nutrient concentrations in agricultural landscapes?; 2) Is P spatially distributed in wetlands; and 3) Can the sorption capacity of wetland soils be exceeded?

The hypotheses tested were: 1) Landscape position and sampling depth does affect soil nutrient concentrations; 2) Nutrient loading does affect the spatial distribution of P in a wetland; and 3) The sorption capacity of wetland sediments can be exceeded.

More specifically, the objectives for this portion of the study were to: 1) show the effects of landscape position and depth on soil N and P concentrations; and 2) investigate total P and phosphorus retention of wetland sediments.

LITERATURE REVIEW

NITROGEN

Leaching of NO_3-N below the root zone in agricultural soils is a concern because it is difficult to recover and can result in groundwater contamination (Follett and Walker 1989). The likelihood of contamination will be influenced by the leaching potential and denitrification rates, both of which may vary across the landscape.

Brubaker et al. (1993) examined NO_3-N by landscape position and sampling depth in cultivated fields in eastern Nebraska. Averaging across sampling depth, no significant differences were found for NO_3-N concentrations (mg kg^{-1}) by landscape position although concentrations tended to decrease downslope. Averaging across landscape position, significant differences were found by sampling depth for nitrate, with concentrations decreasing with depth in the soil profile. Higher nitrate concentrations in the upper profile reflected fertilizer input, mineralization of organic N, and carryover of N during winter and early spring.

The process of denitrification in soil has been shown to be a large factor in the decrease of nitrate concentrations. Bremner and Shaw (1958) showed that under the right conditions, 80-86% of applied nitrogen fertilizer can be lost through denitrification. Pennock et al. (1992) examined denitrification rates in an agricultural ecosystem in southern Saskatchewan and found that denitrification activity displayed a statistically distinct landscape-scale pattern. Rates were lowest in the shoulder elements, intermediate in the footslope and level-convex elements, and highest in the level-concave elements.

Van Kessel et al. (1993) also showed that denitrification activity was highest in the footslope and lower-level landform elements with the lowest rates occurring in the shoulder and upper-level landform elements. Soil water content was the most dominant factor controlling denitrification activity.

PHOSPHORUS

Brubaker et al. (1993) examined available phosphorus in varying landscape positions and sampling depths in cultivated fields in eastern Nebraska. Averaging across sampling depths, no significant differences by landscape position were found for available phosphorus concentrations (mg kg^{-1}) although concentrations tended to decrease downslope. Averaging across landscape positions, significant differences in P concentrations were found for sampling depth, with concentrations decreasing with depth in the soil profile. Elevated levels of available P in the upper profile to a depth of 30 cm was said to be caused by fertilization and a higher organic matter content.

Other studies have shown the influence of landscape position on phosphorus concentrations due to the redistribution of soil by erosion. The erosion process from upper-level landscape positions can increase P concentrations in lower-level positions. Concentrations of P, from overlying cultivated fields, in sediments in riparian areas can actually be higher than the soil from which it is derived (Massey and Jackson 1952). This is due to the selective nature of the erosion process. Organic matter and finer mineral particles which have a higher P content are more easily eroded (Miller et al. 1982). An example of this can be seen in the study by Day et al. (1987) in northwest Florida, where the total P content in the soil was directly associated with the amount of clay in the lower landscape positions.

The spatial distribution of P within a wetland can reflect the flow patterns of erosion in agricultural settings. DeBusk et al. (1994) determined the influence of nutrient loading of total P in a portion of the northern Everglades. Mean soil total P at a depth of 0 to 10 cm was 473 mg kg^{-1} in the interior of the marsh compared to 1338 mg kg^{-1} in areas adjacent to inflows. Johnston et al. (1984) also showed that the P content of a wetland was greatly increased by sediment deposition. Alluvial soils contained much higher concentrations of total P (746 mg L^{-1}) than that of either upland soils (431 mg L^{-1}) or wetland soils not receiving alluvium (263 mg L^{-1}). Total P concentrations in soil correlated best with distance from the stream channel, indicating that the sediments and nutrients were being trapped in the wetland fringe.

Phosphorus in cropland runoff can be associated with the sediment or soil solution (Miller et al. 1982). Because P can be received by riparian areas in both forms, these areas can serve as sinks in two ways: sediment sorption (precipitation and/or adsorption reactions) of P from the overlying water sediments or deposition of P enriched soil (Cooper and Gilliam 1987). Because there is no process to remove P to the atmosphere (like the N cycle with denitrification), movement of P out of a riparian area can only occur through the removal of the enriched sediment and organic material or by desorption to the overlying water (Cooper and Gilliam 1987). Therefore the capacity of a riparian area to serve as a P sink is finite (Cooper and Gilliam 1987). A study done by Steward and Ornes (1975) compared phosphorus additions within wetland enclosures. Treatment rates to these enclosures simulated the quantity contained in a secondary treated domestic effluent while controls were used to simulate natural conditions. Results indicated that the marsh system seemed to have a limited capacity for assimilating continuous phosphorus additions. At a continuous rate of 2.5 kg ha^{-1} P application/week, the assimilative capacity was stressed after week 3 and was overwhelmed by week 8.

SITE DESCRIPTION

Two farmer cooperators in Lake county, South Dakota were recruited in 1992 to participate in the study. The farms were located in the Skunk Creek-Lake Madison-Lake Herman watershed (hereafter known as the Skunk Creek watershed). These farms were chosen on the basis of their similar soil type, wetland class, wetland hydrology, as well as location and proximity to each other. Principle crops for each management system included corn, soybean, alfalfa, and some small grains and are representative of area cropland use.

The average growing season is 140 days with the last spring freeze occurring between April 30 and May 4 and the first fall frost occurring between October 5 and October 9 (S.D. Tech. Guide 1992). Average precipitation during the growing season is 40.6-45.7 cm (16-18 inches) with the majority occurring in June (S.D. Tech. Guide 1992).

Each farm contained seasonal and semipermanent wetlands (Stewart and Kantrud 1971). Seasonal wetlands had an average area of 1.38 ha with an average watershed of 14.68 ha. Semipermanent wetlands averaged 11.92 ha in size with an average watershed size of 33.18 ha. Surface water in seasonal wetlands was usually present through mid-summer while the semipermanent wetlands stayed wet throughout the growing season.

Commonly cropped soil types in the area are classified as Egan and Moody silty clay loams (Udic Haplustolls) while hydric soils in the area include Worthing (Typic Haplustolls), Tetonka (Typic Argiaquolls), and Baltic and Lamo (Cumulic Haplaquolls)

(USDA 1973). Soil series for each wetland site and its surrounding upland soil series can be seen in Table 20.

Table 20. Soil series descriptions for wetland and upland sites.

	Wetland Soil Series		Upland Soil Series	
Wetland No.	Series	Symbol	Series	Symbol
3	Worthing	Wo	Egan	EbB
4	Worthing	Wo	Egan	EeC2
5	Classified Marsh Soil		Egan	EbB
6	Worthing	Wo	Egan	EbB
8	Classified Marsh Soil		Egan	EaC
9	Worthing	Wo	Egan (90%) Badus (10%)	EbB Ba

Source: Machacek (1995)

A more detailed description of each soil series is presented in Table 36 in the Appendix.

MATERIALS AND METHODS

The sampling design at each wetland site consisted of two axes extending from the wetland border to upland areas. In 1993, wells were placed at each sampling point using a Gidding's hydraulic soil-probe-equipped truck (Machacek 1995). Each axis had a groundwater monitoring well located at the wetland edge and 46 m from the wetland edge for a total of 4 sampling points at each wetland site. A 50.8 mm inside diameter x 122 cm length soil probe was used in conjunction with a 1.2 m and 2.4 m Kelly bar extension. Access holes were dug to depths of 2.4 m at the wetland edge and 3.0 m for the upland wells. The observation well soil cores were removed and divided into 15 cm sections and analyzed by the Soil Testing Laboratory, SDSU in March of 1995 for Nitrate-N, total-N, and available phosphorus (Bulletin #499 1988).

The statistical design used for the observation well soil cores was a split-plot arrangement similar to that of Brubaker et al. (1993). Landscape position was used as the main plot with depth used as the subplot effect. Wetland sites for both seasonal and semipermanent wetland landscapes were used as replications (Table 21, Table 22). Mean comparisons were made using the least significant difference (LSD) method with $P > 0.05$.

Table 21. Analysis of variance for landscape position, depth, and the interaction using a split-plot design for seasonal wetland landscapes.

Source	df
Site	3
Landscape position	1
Error A (position x site)	3
Depth	6
Error B (depth x site)	18
Position x Depth	6
Error AB (position x depth x site)	18
Total	55

Table 22. Analysis of variance for landscape position, depth, and the interaction using a split-plot design for semipermanent wetland landscapes.

Source	df
Site	1
Landscape position	1
Error A (position x site)	1
Depth	6
Error B (depth x site)	6
Position x depth	6
Error AB (position x depth x site)	6
Total	27

Total phosphorus concentrations were determined from the top 15 cm of the observation well soil cores using the perchloric acid digestion method with a 40-tube aluminum digestion block as described by O'Halloran (1993). The coloremetric analysis procedure, using a Milton Roy Spectronic 21 spectrophotometer to determine final phosphorus concentrations, was described by Olsen and Sommers (1982). Data for total

P for the top 15 cm of the observation well soil cores was analyzed using a Proc GLM (General Linear Model) in SAS (Statistical Analysis Systems). Significant differences among landscape position were analyzed with axis values within a wetland pooled with individual wetlands used for replication. Total phosphorus values between landscape positions were compared within seasonal and semipermanent wetlands.

Wetland #6 (a seasonal wetland) was selected for further investigation in order to determine total phosphorus concentrations and the sorption capacity of wetland soils. Sediment core samples were taken from Wetland #6 in late October/early November, 1995. This particular wetland was chosen because it had a limited buffer strip surrounding the wetland and was farmed through and fertilized during dry years. Samples were obtained using an AMS mud bucket auger (5.72 cm diameter). A radially spoked sampling pattern with approximately 76 m intervals at the circumference was used for a total of 6 spokes around the wetland. Samples were taken along each spoke in the middle of a 7.6 m interval beginning at the edge of the wetland border and leading inwards for a total of 5 samples on each spoke (labeled A through E respectively). The top 15 cm of each soil core was placed in a plastic bag and frozen for later analysis.

After being thawed, samples were forced-air dried, ground with a mortar and pestle, and sieved through a 0.03 cm mesh screen. Total phosphorus concentrations were determined using the perchloric acid digestion method using a 40-tube aluminum digestion block as described by O'Halloran (1993). The coloremetric analysis procedure using a Milton Roy Spectronic 21 spectrophotometer was used to determine phosphorus concentrations as described by Olsen and Sommers (1982). Data for total P for Wetland #6 sediment cores was analyzed using Proc GLM in SAS. Significant differences among wetland location (Samples A-E) were analyzed with axes (spokes) used as replications.

The ability of sediment to sorb inorganic P was examined using a modified version of the adsorption isotherm as described by Holford et al. (1974). For each isotherm, 2.5 g of soil was shaken in 50 ml of 0.02 M KCl solution on a Lab-line Instruments orbital shaker at 200 rpm at room temperature for 24 hours. The initial concentrations of P as KH_2PO_4 in the KCl solution were 0, 10, 25, 50, 75, 100, and 150 mg L^{-1}. The supernatant was then filtered through Whatman #15 filter paper and analyzed using the coloremetric procedure as described by Olsen and Sommers (1982). The amount of P absorbed was calculated by the difference of the original P concentration minus the P remaining in solution after shaking. The relationship between PO_4-P concentration in solution and the amount sorbed onto the sediment particles was described by the Langmuir equation. The most common form of the Langmuir equation is:

$$x/m = kbc/1 + kc$$

where x/m = mg P absorbed per gram of soil, b = the adsorption maximum, c = equilibrium P concentration in moles/liter, and k = a constant relating to binding strength. The equation can also be represented in the linear form

$$c/x/m = 1/kb + c/b$$

where c/x/m = the equilibrium P concentration in moles per liter divided by the mg P absorbed per gram of soil , 1/b = slope and 1/kb = intercept. If the adsorption isotherm conforms to the Langmuir equation, the plot of c/x/m versus c will yield a straight line and an adsorption maxima can be estimated from the inverse of the slope.

RESULTS AND DISCUSSION

SEASONAL WETLAND LANDSCAPES

Probability of significant differences in soil N and P due to landscape position, depth, and the interaction term in seasonal wetland landscapes can be seen in Table 23.

Table 23. Probability of > F(0.05) for soil N and P as effected by landscape position, depth, and the interaction term in seasonal wetland landscapes.

Variable	Nitrate-N	Total-N	Available P
Landscape position	0.0087	0.0267	0.0913
Depth	0.0001	0.0001	0.0001
Position * Depth	0.1406	0.0002	0.4213

Averaging across depth, significant differences for nitrate-N and total-N concentrations were found among landscape positions (Table 23). Nitrate-N concentrations decreased downslope while total N percentages increased (Table 24). This difference in nitrate concentrations from the upland landscape position and lowland landscape position may suggest an increase in denitrification activity (Pennock et al. 1992, van Kessel et al. 1993).

Table 24. Differences in soil N and P as effected by landscape position in seasonal wetland landscapes.

Landscape position	Nitrate-N (mg L^{-1})	Total-N (g kg^{-1})	Available P (mg L^{-1})
Upper-level	8.39	1.4	13.05
Lower-level	4.81	1.8	23.00
LSD (0.05)	2.64	0.4	NS

Higher total-N concentrations in the footslopes may be related to the increase in the organic nitrogen pool due to deposition of eroded material and soil formation.

Pierson and Mulla (1990) have shown that soils on footslope and toeslope positions have a higher organic C content and Malo et al. (1974) found that the organic C content and clay content increases from shoulder position to footslope.

Although not significant, available P concentrations were greater downslope than in the upland in the seasonal wetland landscape.

Since nitrogen and phosphorus are added as fertilizers (either through synthetic fertilizers or cover crops and manure applications), one would expect a fairly uniform distribution of these nutrients across the landscape. Differences in properties such as higher organic matter content in lower-level landscape positions, however, could influence soil nutrients. Organic matter contributes to the pool of total N and potentially mineralizable N and available P.

With more mineralizable N and organic matter in the lower-level landscape positions, a greater potential for denitrification exists in an anaerobic condition. Anaerobic conditions and large sources of carbon due to the slow breakdown of organic matter for energy favor denitrification. There is a high correlation between denitrification and the availability of organic material. Microorganisms use these organic carbon compounds as electron donors and as sources of cellular material (Firestone 1982). Firestone (1982) maintained that the presence of abundant C substrate caused rapid O_2 consumption and thus oxygen depletion in soil microenvironments; indirectly enhancing the potential for denitrification. Bremner and Shaw (1958) observed that the amount and type of organic matter will have an influence on denitrification in waterlogged soils with readily decomposable compounds such as glucose and mannitol inducing rapid denitrification of nitrate compared to more refractory compounds. Burford and Bremner (1975) also demonstrated that there was a significant relationship between denitrification and the amount of total organic carbon ($r^2 = 0.77$) and that denitrification was very highly correlated ($r^2 = 0.99$) with water-soluble carbon or mineralizable carbon.

By averaging across landscape position, significant differences by depth were found for all soil nutrients (Table 25).

Table 25. Differences in soil N and P as effected by depth in seasonal wetland landscapes.

Depth (cm)	Nitrate-N (mg L^{-1})	Total-N (g kg^{-1})	Available P (mg L^{-1})
0-15	11.96	3.5	39.69
15-30	7.09	2.8	27.63
30-45	5.94	1.8	18.25
45-60	5.21	1.1	10.56
60-75	5.09	0.8	9.06
75-90	5.53	0.9	9.13
90-105	5.20	0.5	11.00
LSD (0.05)	2.90	0.4	13.53

Values of soil N and P tended to decrease with depth. Higher levels of NO_3-N in the upper 30 cm of the soil profile may reflect fertilizer inputs and the mineralization of N from the soil organic matter as suggested by Farrell et al. (1996). Below this plow layer, nitrate concentrations decrease suggesting plant uptake within the root zone and possible denitrification and/or nitrate leaching through the soil profile. The decrease in total N concentrations with depth in the profile would also suggest higher levels of organic matter in the upper profile.

Elevated levels of available P in the upper 30 cm may also be due to fertilizer inputs along with higher levels of organic matter as suggested by Brubaker et al. (1993).

Soil N and P means having a significant interaction between landscape position and depth were evaluated using a LSD based on a weighted average of the error terms (Table 26). Total N concentration was the only soil nutrient to show a significant interaction between landscape position and depth. The interaction is shown in Figure 5.

Table 26. Soil N and P concentrations as effected by landscape position * depth in seasonal wetland landscapes.

	Nitrate-N (mg L^{-1})		Total-N (g kg^{-1})		Available P (mg L^{-1})	
Depth (cm)	Lower-level	Upper-level	Lower-level	Upper-level	Lower-level	Upper-level
0-15	7.750	16.175	4.21	2.83	44.500	34.875
15-30	6.550	7.625	3.45	2.09	33.125	22.125
30-45	5.400	6.475	2.01	1.54	21.750	14.750
45-60	4.025	6.400	1.15	1.05	12.875	8.250
60-75	3.475	6.700	0.78	0.83	13.625	4.500
75-90	3.500	7.550	0.64	1.14	15.375	2.875
90-105	2.714	7.686	0.50	0.50	19.286	2.714
LSD (0.05)	NS		0.71		NS	

Total N concentrations generally were higher downslope than upslope and decreased with sampling depth. The higher concentrations of total N in the lower-level positions may indicate an accumulation of organic matter from non-harvested vegetation.

Total phosphorus concentrations for upper-level and lower-level landscape positions can be seen in Table 27. Although not significantly different, there is a trend for total P to be higher in the lower-level positions than that of upper-level positions.

Figure 5. Total-N concentrations as effected by landscape position and depth in seasonal wetland landscapes.

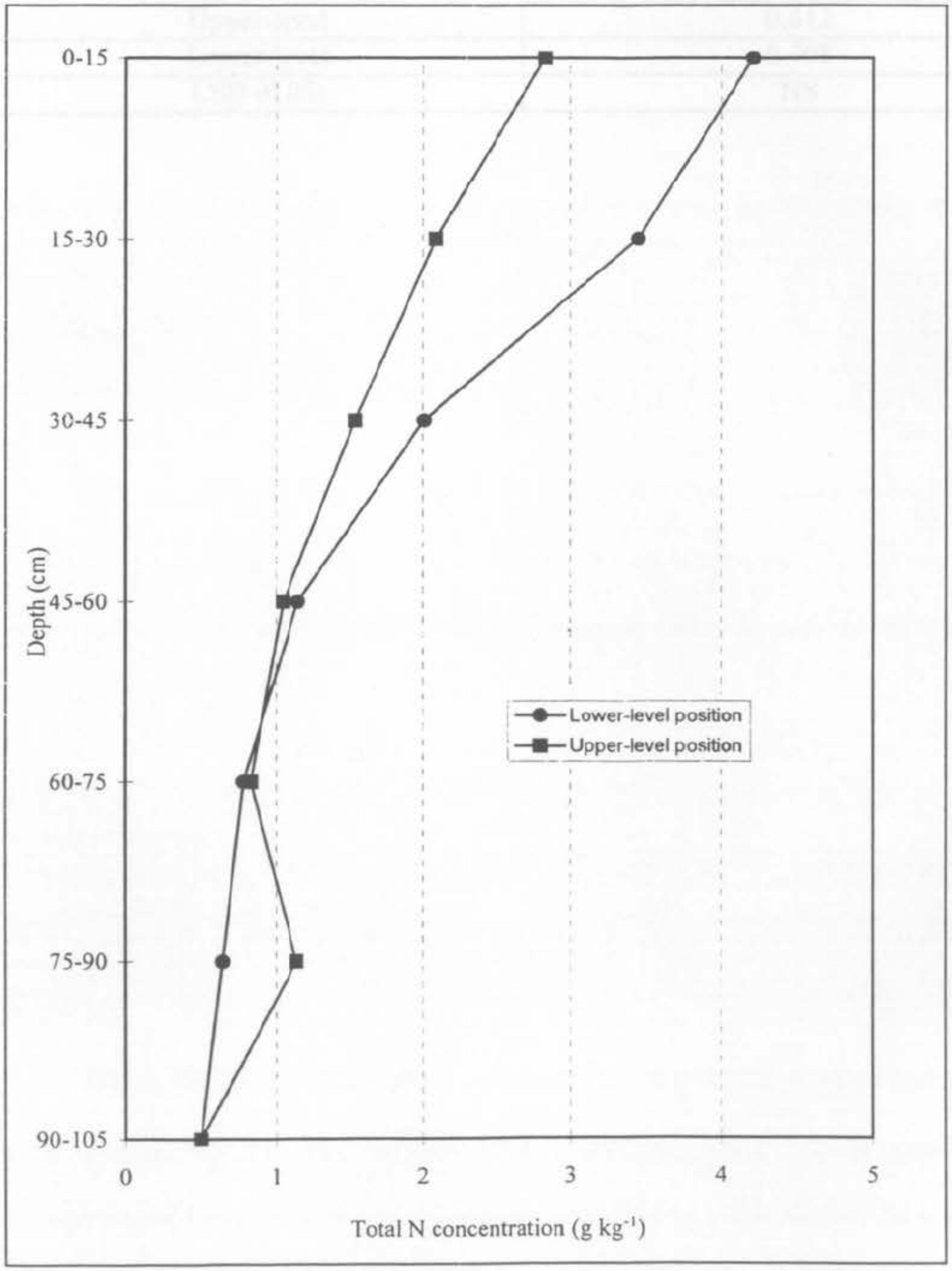


Table 27. Total P concentrations as effected by landscape position in seasonal wetland landscapes.

Landscape position	Total P (mg P g^{-1} soil)
Upper-level	0.612
Lower-level	0.708
LSD (0.05)	NS

SEMIPERMANENT WETLAND LANDSCAPES

Probability of > F(0.05) for the observation well soil cores among landscape position, depth, and the interaction term in semipermanent wetland landscapes can be seen in Table 28.

Table 28. Probability of > F(0.05) for soil N and P as effected by landscape position, depth, and interaction term in semipermanent wetland landscapes.

Variable	Nitrate-N	Total-N	Available P
Landscape position	0.3374	0.2426	0.1168
Depth	0.5005	0.0009	0.0001
Position * Depth	0.1503	0.0848	0.0170

Averaging across depth, no significant differences due to landscape position were found among soil N and P in the semipermanent wetland landscapes. Nitrate-N concentrations in the lower landscape position were not different from that of upper concentrations (Table 29).

Table 29. Differences in soil N and P as effected by landscape position in semipermanent wetland landscapes.

Landscape position	Nitrate-N (mg L^{-1})	Total-N (g kg^{-1})	Available P (mg L^{-1})
Upper-level	4.06	0.9	7.64
Lower-level	7.32	2.1	16.07
LSD (0.05)	NS	NS	NS

This is different from the nitrate concentrations in seasonal wetland landscapes where the lowland nitrate concentrations were significantly lower than the upland nitrate concentrations (Table 24). This could suggest that seasonal wetlands may be better denitrifiers than semipermanent wetlands, as supported by water data.

With high water table levels during the spring and early summer, seasonal wetlands may act more like groundwater discharge wetlands early and then become groundwater recharge wetlands later during the growing season. These fluctuating water table level wetlands, with a wetting-drying cycle typifying seasonal wetlands, will show greater differences in denitrification rates than those wetlands that have water throughout the year.

Several studies have reported higher rates of denitrification in soils that have been rewet following a drying period compared to soils that were continually moist (Bremner and Shaw 1958, McKenzie and Kurtz 1976, Patten et al. 1980). These studies have indicated that denitrification was stimulated by releases of available C and N, along with the reduction of soil oxygen levels by vigorous aerobic respiration following rewetting of dry soils (Groffman and Tiedje 1988, Smith and Parsons 1985). In wetland soils, the mineralization of C and N from aquatic macrophytes will occur more rapidly when oxidized than in a strictly anaerobic situation. Brinson et al. (1981) suggested that alternating wet and dry conditions may lead to optimum litter decomposition while anaerobic conditions caused by constant flooding are the least favorable conditions for decomposition. Litter breakdown will release more decomposable C and N for use by organisms and cause a higher rate of denitrification during periods of rewetting.

Although not significant, total N and available P concentrations showed the same trends in landscape position as the seasonal wetland landscapes, with higher concentrations occurring in the lower landscape positions.

Averaging across landscape position, significant differences in soil N and P are shown due to depth in Table 30. Total N and available P concentrations showed a significant difference due to depth but nitrate-N concentrations did not.

Table 30. Differences in soil N and P as effected by depth in semipermanent wetland landscapes.

Depth (cm)	Nitrate-N (mg L^{-1})	Total-N (g kg^{-1})	Available P (mg L^{-1})
0-15	6.73	3.1	34.88
15-30	5.20	2.2	20.63
30-45	5.25	1.7	13.13
45-60	5.93	1.5	7.25
60-75	5.95	0.9	2.13
75-90	5.60	0.6	1.63
90-105	4.89	0.3	1.57
LSD (0.05)	NS	1.0	5.04

The higher concentrations of total N and available P in the upper profile may be due to higher levels of organic matter and fertilizer input in the upper profile. The lack of a significant difference in nitrate-N concentrations by depth may suggest less favorable conditions for denitrification in semipermanent than seasonal wetlands.

The interaction between landscape position and depth for soil N and P in semipermanent wetland landscapes can be seen in Table 31. Means having a significant interaction between landscape position and depth were separated using a LSD value based on a weighted average of the error terms. Available P concentration was the only soil nutrient to show significant differences due to landscape position * depth and can be seen graphically in Figure 6.

Table 31. Soil N and P concentrations as effected by landscape position * depth in semipermanent wetland landscapes.

	Nitrate-N (mg L^{-1})		Total-N (g kg^{-1})		Available P (mg L^{-1})	
Depth (cm)	Lower-level	Upper-level	Lower-level	Upper-level	Lower-level	Upper-level
0-15	6.15	7.30	3.80	2.30	40.50	29.25
15-30	6.45	3.95	3.05	1.43	27.75	13.50
30-45	7.20	3.30	2.58	0.85	22.00	4.25
45-60	8.85	3.00	2.53	0.53	12.25	2.25
60-75	8.60	3.30	1.43	0.43	3.00	1.25
75-90	7.70	3.50	0.80	0.30	2.00	1.25
90-105	5.93	4.10	0.33	0.35	1.33	1.75
LSD (0.05)	NS		NS		20.84	

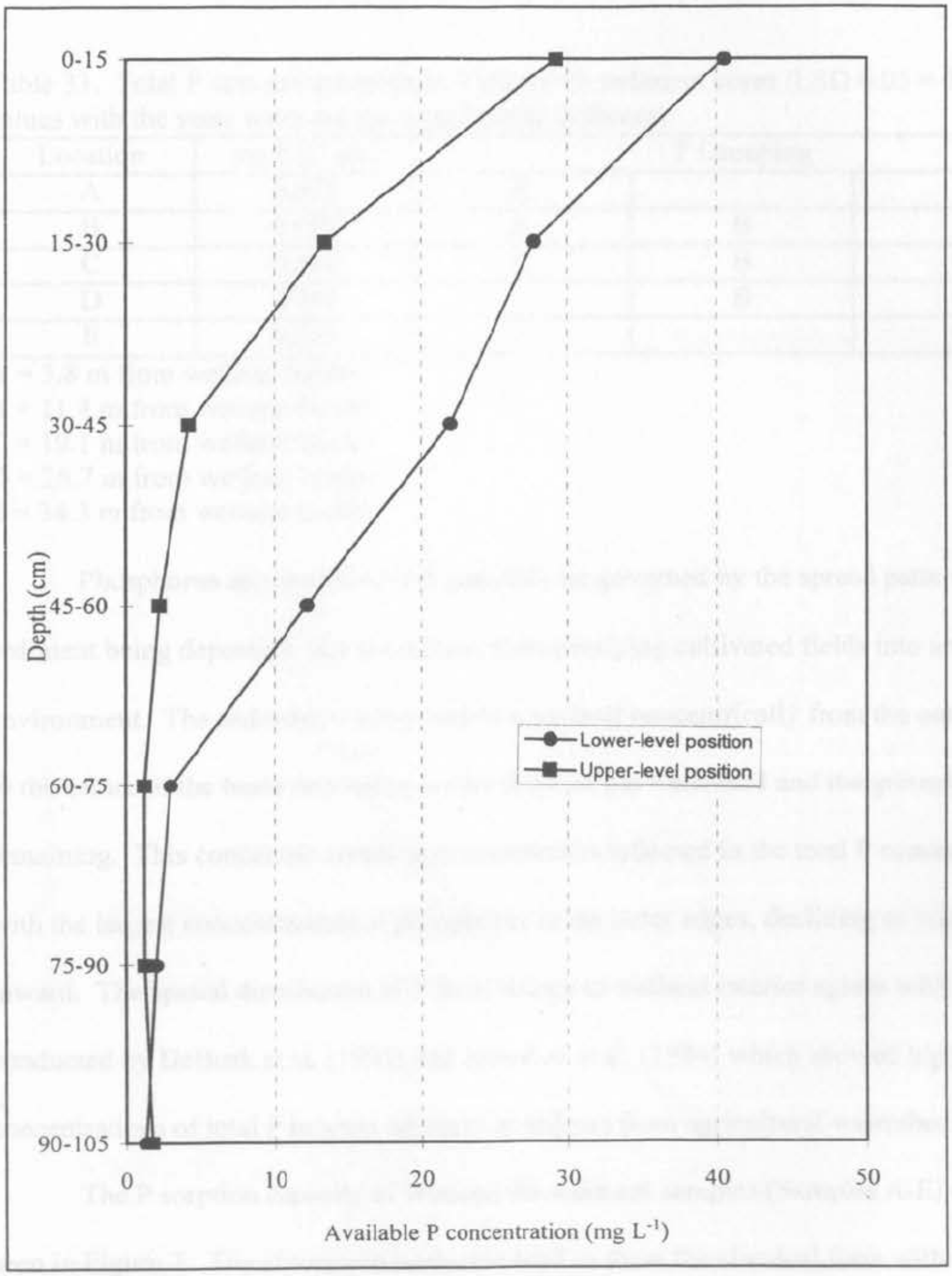
Available P concentrations generally increased downslope and decreased with sampling depth. The higher concentrations of available P in the lower-level positions may indicate a deposition of organic matter or clay-sized particles from the upper-level positions due to erosion or runoff of soluble fertilizer P.

Total phosphorus concentrations for upland and lowland landscape positions can be seen in Table 32. A significant difference between the upland and lowland positions may be due to the deposition of eroding soil from the upper-level positions.

Table 32. Total P concentrations as effected by landscape position in semipermanent wetland landscapes.

Landscape position	Total P (mg P g^{-1} soil)
Upper-level	0.572
Lower-level	0.719
LSD (0.05)	0.142

Figure 6. Available P concentrations as effected by landscape position and depth in semipermanent wetland landscapes.



WETLAND #6 SEDIMENT

Total P concentrations in Wetland #6 sediment are shown in Table 33.

Table 33. Total P concentrations from Wetland #6 sediment cores (LSD 0.05 = 0.121, values with the same letter are not significantly different).

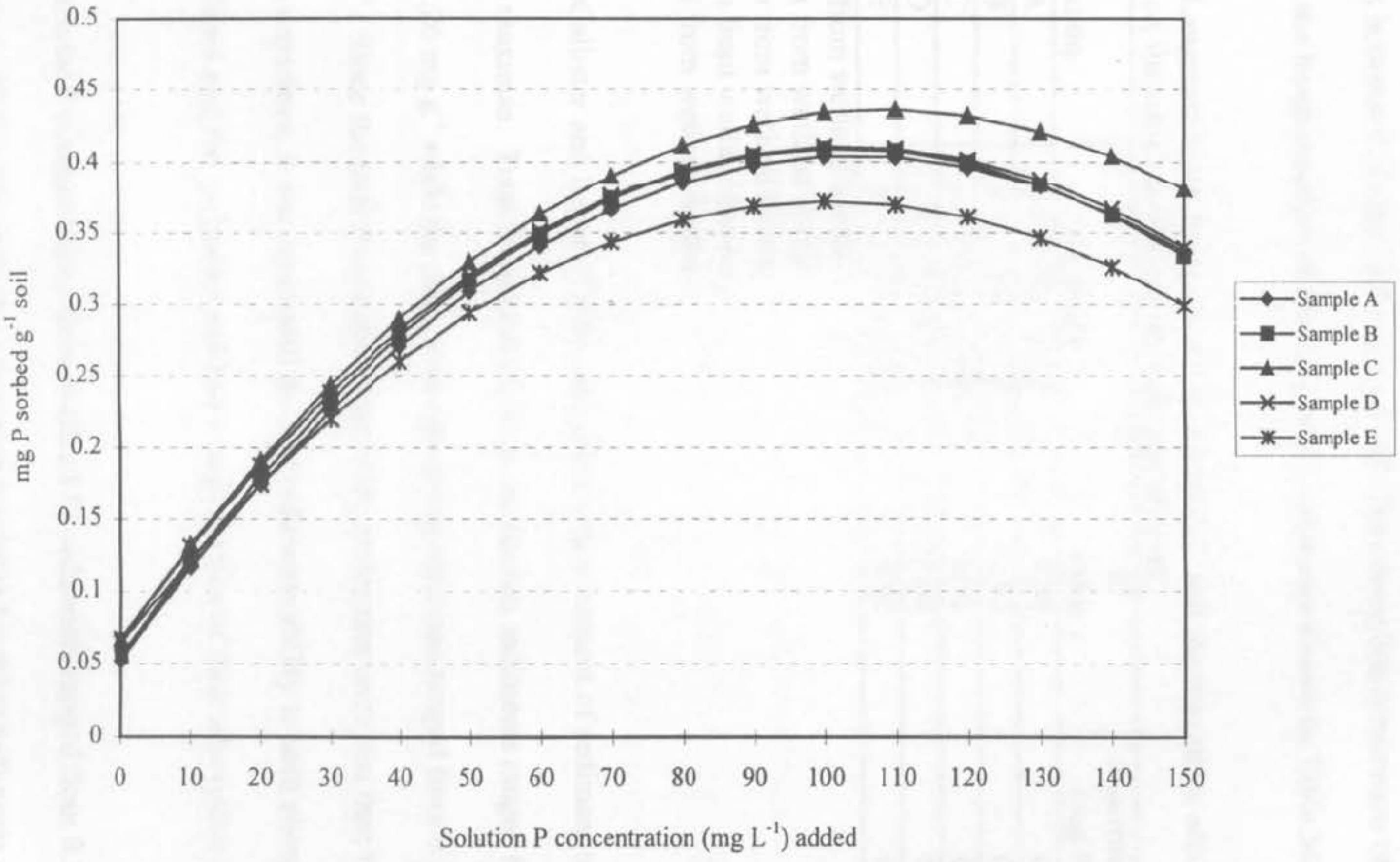
Location	mg P g^{-1} soil	T Grouping		
A	0.673	A		
B	0.657	A	B	
C	0.584	A	B	C
D	0.545		B	C
E	0.513			C

A = 3.8 m from wetland border
B = 11.4 m from wetland border
C = 19.1 m from wetland border
D = 26.7 m from wetland border
E = 34.3 m from wetland border

Phosphorus accumulation will generally be governed by the spread pattern of sediment being deposited, due to erosion, from overlying cultivated fields into an aquatic environment. The sediment is deposited in a wetland concentrically from the outer edges to the center of the basin depending on the slope of the watershed and the ground cover remaining. This concentric zonation of sediment is reflected in the total P concentrations with the largest concentrations of phosphorus in the outer edges, declining as you move inward. The spatial distribution of P from source to wetland interior agrees with studies conducted by DeBusk et al. (1994) and Johnston et al. (1984) which showed highest concentrations of total P in areas adjacent to inflows from agricultural watersheds.

The P sorption capacity of Wetland #6 sediment samples (Samples A-E) can be seen in Figure 7. The absorption isotherms tend to show the classical form with sorption capacities being reached between 0.35 and 0.45 mg P g^{-1} soil. This is slightly less than

Figure 7. Phosphate adsorption isotherms for a Worthing silty clay loam.



the capacity found by Harter (1968) which showed an absorption maxima of iron-bound phosphorus between 0.50 and 0.60 mg P g^{-1} soil. The adsorption maximums of Samples A-E using the linear equation of the Langmuir formula are shown in Table 34.

Table 34. Langmuir linear equations for locations A-E and the maximum adsorption concentration for seasonal wetland #6 sediment samples.

Location	Equation	R^2 value	adsorption maxima (mg P g^{-1} soil)
A	$y = 2.75x + 0.68$	0.89	0.36
B	$y = 2.70x + 0.58$	0.95	0.37
C	$y = 2.45x + 0.70$	0.91	0.41
D	$y = 2.70x + 0.51$	0.94	0.37
E	$y = 3.29x + 0.24$	0.82	0.30

A = 3.8 m from wetland border
B = 11.4 m from wetland border
C = 19.1 m from wetland border
D = 26.7 m from wetland border
E = 34.3 m from wetland border

McCallister and Logan (1978) related the total P content of sediment to adsorption maximas. Total P concentrations in the bottom sediments ranged from 0.48 mg g^{-1} to 1.26 mg g^{-1} while the calculated adsorption maximas ranged from 0.22 mg g^{-1} to 4.87 mg g^{-1}. Since the total P concentrations of the sediments were less than the adsorption capacities, it was concluded that the sediments ability to sorb phosphorus was not yet realized and the sediments still had a large portion of their adsorption capacity remaining.

The total P concentrations from Wetland #6 sediments ranged from 0.513 mg g^{-1} to 0.673 mg g^{-1} (Table 33) while the adsorption maximas for these sediments ranged an additional 0.30 mg g^{-1} to 0.41 mg g^{-1} (Table 34) greater than the already existing phosphorus content. In this case, it is evident that the sorption capacities of Wetland #6 sediments had not been exceeded and that the ability for Wetland #6 to sorb more inorganic phosphorus from overlying water was still available. However, the equilibrium between solution and solid phase P will shift, resulting in higher concentrations of solution P. While Wetland #6 still retains the ability to sorb P, the rate of P sorption in agricultural wetlands may be accelerated due to erosion from overlying fields and direct fertilization.

The adsorption of phosphorus in flooded soils is complex and not all the literature seems to agree on what soil properties to use in order to predict it. Khalid et al. (1977) studied soils incubated under oxidized (aerobic) and reduced (anaerobic) conditions. Soils selected for the study were characterized for clay content, total carbon, extractable P, pH, and oxalate-extractable Fe. Results indicated that of the soil properties tested, oxalate extractable Fe (oxalate extraction is known to dissolve amorphous and poorly crystalline oxides of Fe) was the most important contributor to the sorption of added P in flooded soils. Richardson (1985) concluded that the phosphorus adsorption potential in wetland ecosystems may be predicted solely from the extractable aluminum content of the soil. Harter (1969) suggested that anion exchange sites on organic matter were important in the initial bonding of phosphorus by soils which is subsequently transformed into less soluble iron and aluminum phosphates with time.

Shukla et al. (1971) investigated sediments from calcareous and noncalcareous lakes and found that noncalcareous sediments sorbed more added P than calcareous sediments. The best single criterion explaining P sorption for both calcareous and noncalcareous sediments was the amount of oxalate-extractable Fe presumed to be derived from amorphous Fe oxides. Amer et al. (1991) also showed a high correlation of P sorption and oxalate-extractable Fe in calcareous soils.

In prairie wetlands and lakes, calcium, magnesium, sodium, and potassium have been found to be the most abundant cations (LaBaugh 1989). A high Ca^{2+} status is desirable because it reflects low concentrations of other cations such as Al^{3+} in acidic soils (Bohn et al. 1979b). Because phosphorus sorption is correlated best with iron and aluminum, prairie wetlands that contain higher concentrations of calcium may reach their sorption maxima sooner than wetlands containing acidic soils. If the wetlands potential to sorb more inorganic phosphorus from overlying water has been realized, contamination of P to hydrologically linked groundwater or to surface water may become a reality because of the inability to remove this inorganic form. More study may be required to understand the behavior of prairie wetlands and its relationship to phosphorus adsorption.

SUMMARY AND CONCLUSIONS

Denitrification is a biological process by which anaerobic bacteria use nitrate in place of oxygen as a hydrogen (electron) acceptor. This process takes place in the aerobic-anaerobic interface of saturated soils with readily decomposable sources of carbon present and high concentrations of nitrate.

Today, denitrification is occurring at much higher rates than it did in the past because of the increased use of nitrogen fertilizer due to the conversion of native prairie to intensive row-crop agriculture (Lemme 1988). Surface applications of nitrogen fertilizers in excess of crop needs can lead to accumulation of NO_3^- in groundwater underlying cultivated lands (Bradley and Chapelle 1993). This supply of nitrate in agricultural discharge to riparian areas leads to favorable conditions for denitrification.

This study examined the effects of landscape position and wetland class on nitrate concentrations in agricultural wetland landscapes. Results indicated that higher concentrations of NO_3^- exist in groundwater in upland landscapes compared to groundwater in lowland landscape positions and the surface water in wetlands. The higher concentrations of nitrate in upland landscapes were possibly due to a lack of available organic carbon that is needed by denitrifying microorganisms for growth and reduction of NO_3^- and to a more aerobic environment. Results also indicated that seasonal wetland landscapes were better denitrifiers than semipermanent wetland landscapes. Wetlands that follow seasonal wetting/drying cycles, typical of seasonal wetlands, will stimulate the release of available C and N when oxidized more so than in a strictly anaerobic habitat (Bremner and Shaw 1958, McKenzie and Kurtz 1976, Brinson et al. 1981) and will promote higher rates of denitrification.

Orthophosphate concentrations were also analyzed according to landscape position and wetland class. Concentrations of ortho-P were lower in groundwater in upland landscape positions compared to ortho-P concentrations in wetland groundwater and the surface water in wetlands. Phosphorus is generally not available to be leached in upland groundwater due to the precipitation/adsorption mechanisms that occur with this anion before it reaches the groundwater. As soils are flooded and conditions become more anaerobic, as is the case in overlying surface water and surrounding wetland groundwater, phosphorus will become more available as it is released from soil to solution by the reduction of iron (Patrick and Khalid 1974) or aluminum. Results also indicated that seasonal wetland landscapes had higher concentrations of ortho-P than semipermanent wetland landscapes did. This may be due in part to the lack of buffer strips surrounding the seasonal wetlands and to the fertilization of the wetland when farmed through in dry years.

Soil N and P concentrations from observation well soil cores were analyzed by landscape position and depth. Results indicated that seasonal wetland landscapes exhibited a greater potential for denitrification than semipermanent wetland landscapes. Soil nitrate in seasonal wetland landscapes showed a significant decrease from upper-level landscape positions to lower-level landscape positions while semipermanent wetland landscapes did not. Soil nitrate concentrations also showed a significant difference by depth in seasonal wetland landscapes where semipermanent wetland landscapes did not. Higher concentrations in the upper profile may have reflected fertilizer inputs and the mineralization of N from organic matter as suggested by Farrell et al. (1996). The decrease of soil nitrate in the lower profile may suggest denitrification and/or nitrate leaching. Soil P concentrations showed significant differences due to depth for both seasonal and semipermanent wetland landscapes. Elevated levels of available P in the upper profile may have been due to fertilizer inputs along with elevated levels of organic matter as suggested by Brubaker et al. (1993).

Total P concentrations within seasonal wetland sediments showed a spatial distribution with highest concentrations (0.673 mg P g^{-1}) in the outer edges of the wetland and steadily declining towards the middle (0.513 mg P g^{-1}). Adsorption maximas found by the Langmuir equation indicated that the sorption capacity for the wetland had not yet been exceeded. While Wetland #6 still retained the ability to sorb P, the rate of P sorption in agricultural wetlands may be accelerated due to erosion from overlying fields and direct fertilization. This may attest as to why orthophosphate concentrations were higher in seasonal wetlands as indicated in Chapter 1.

In conclusion, wetlands and hydrologically linked groundwater are dynamic systems. A comprehensive study to understand wetland ecosystems would be at best difficult to accomplish because of all the factors involved such as hydrology, chemistry, vegetation, etc. as they seem to all be interrelated. Information regarding agricultural wetlands in the PPR is lacking. Suggestions for further study include a more extensive and comprehensive examination of chemistry between different wetland classes.

Comparisons of wetland chemistry between wetlands in undisturbed watersheds and those wetlands affected by agriculture is also needed. A more comprehensive investigation on the sorption capacities of wetland classes in calcareous soils in the PPR should also be better undertaken.

LITERATURE CITED

Amer, F., M. E. Saleh, and H. E. Mostafa. 1991. Phosphate behavior in submerged calcareous soils. Soil Science 151:306-311.

Bohn, H. L., B. L. McNeal, and G. A. O'Connor. 1979a. Anion and molecular retention. p.171-194. *In* H. L. Bohn, B. L. McNeal, and G. A. O'Conner (ed.) Soil Chemistry. John Wiley & Sons, Inc., New York, NY, USA.

Bohn, H. L., B. L. McNeal, and G. A. O'Connor. 1979b. Important ions. p. 272-315. *In* H. L. Bohn, B. L. McNeal, and G. A. O'Conner (ed.) Soil Chemistry. John Wiley & Sons, Inc., New York, NY, USA.

Bowden, W. B. 1987. The biogeochemistry of nitrogen in freshwater wetlands. Biogeochemistry 4:313-348.

Bradley, P. M., M. Fernandez, Jr., and F. H. Chapelle. 1992. Carbon limitation of denitrification rates in an anaerobic groundwater system. Environmental Science and Technology 26:2377-2381.

Bradley, P. M. and F. H. Chapelle. 1993. Arsenate inhibition of denitrification in nitrate contaminated sediments. Soil Biology and Biochemistry 25:1459-1462.

Brady, N. C. 1990. Nitrogen and sulfur economy of soils. p. 315-350. *In* N.C. Brady (ed.) The Nature and Properties of Soils, 10th ed.. Macmillan Publishing Company, New York, NY, USA.

Bremner, J. M. and K. Shaw. 1958. Denitrification in soil. II. Factors affecting denitrification. Journal of Agricultural Science 51:40-52.

Brinson, M. M., A. E. Lugo, and S. Brown. 1981. Primary productivity, decomposition and consumer activity in freshwater wetlands. Annual Reviews in Ecological Systematics 12:123-161.

Broadbent, F. E. and F. Clark. 1965. Denitrification. p. 344-359. *In* W. V. Bartholomew and F. E. Clark (ed.) Soil Nitrogen. American Society of Agronomy, Inc., Madison, WI, USA.

Brubaker, S. C., A. J. Jones, D. T. Lewis, and K. Frank. 1993. Soil properties associated with landscape position. Soil Science Society of America Journal 57:235-239.

Bulletin #499. 1988. Recommended chemical soil test procedures for the North Central Region. W. C. Dahnke (ed.). NDSU. 37 p.

Buresh, R. J., M. E. Casselman, and W. H. Patrick, Jr. 1980. Nitrogen fixation in flooded soil systems, a review. p. 149-192. *In* N. C. Brady (ed.) Advances in Agronomy 33. Academic Press, Inc., New York, NY, USA.

Burford, J. R. and J. M. Bremner. 1975. Relationships between the denitrification capacities of soils and total, water-soluble and readily decomposable soil organic matter. Soil Biology and Biochemistry 7:389-394.

Carey, M., R. Heimlich, and R. Brazee. 1990. A permanent wetland reserve: analysis of a new approach to wetland protection. U. S. Department of Agriculture, Economic Research Service, Agriculture Information Bulletin Number 610.

Cho, C. M., L. Sakdinan, and C. Chang. 1979. Denitrification intensity and capacity of three irrigated Alberta soils. Soil Science Society of America Journal 43:945-950.

Cole, C. V., S. R. Olsen, and C. O. Scott. 1953. The nature of phosphate sorption by calcium carbonate. Soil Science Society of America Journal 17:352-356.

Cooper, J. R., J. W. Gilliam, R. B. Daniels, and W. P. Robarge. 1987. Riparian areas as filters for agricultural sediment. Soil Science Society of America Journal 51:416-420.

Cooper, J. R. and J. W. Gilliam. 1987. Phosphorus redistribution from cultivated fields into riparian areas. Soil Science Society of America Journal 51:1600-1604.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U. S. Fish and Wildlife Service FWS/OBS-79/31. U. S. Government Printing Office, Washington, D. C., USA.

Davis, C. B. and A. G. van der Valk. 1978. Litter decomposition in prairie glacial marshes. p. 99-113. *In* R. E. Good, D. F. Whigham, and R. L. Simpson (ed.) Freshwater wetlands: ecological processes and management potential. Academic Press, New York, New York, USA.

Davis, C. B., A. G. van der Valk, and J. L. Baker. 1981. Prairie pothole marshes as traps for nitrogen and phosphorus in agricultural runoff. p. 154-163. *In* B. Richardson (ed.) Proceedings from the Midwest Conference on Wetland Values and Management.

Day, L. D., M. E. Collins, and N. E. Washer. 1987. Landscape position and particle-size effects on soil phosphorus distributions. Soil Science Society of America Journal 51:1547-1553.

DeBusk, W. F., K. R. Reddy, M. S. Koch, and Y. Wang. 1994. Spatial distribution of soil nutrients in a northern Everglades marsh: Water Conservation Area 2A. Soil Science Society of America Journal 58:543-552.

Farrell, R. E., P. J. Sandercock, D. J. Pennock, and C. van Kessel. 1996. Landscape-scale variations in leached nitrate: relationship to denitrification and natural nitrogen-15 abundance. Soil Science Society of America Journal 60:1410-1415.

Firestone, M. K. 1982. Biological denitrification. p. 289-326. *In* F. J. Stevenson (ed.) Nitrogen in Agricultural Soils. American Society of Agronomy, Inc., Crop Science Society of America, Inc., Soil Science Society of America, Inc., Madison, WI, USA.

Follett, R. F. and D. J. Walker. 1989. Groundwater quality concerns about nitrogen. p. 1-22. *In* R. F. Follett (ed.) Nitrogen management and groundwater protection. Elsevier, Amsterdam.

Groffman, P. M. and J. M. Tiedje. 1988. Denitrification hysteresis during wetting and drying cycles in soil. Soil Science Society of America Journal 52:1626-1629.

HACH. 1992. DR/2000 Spectrophotometer Handbook. Loveland, CO. 557 p.

Hanson, G. C., P. M. Groffman, and A. J. Gold. 1994. Denitrification in riparian wetlands receiving high and low groundwater nitrate inputs. Journal of Environmental Quality 23:917-922.

Harter, R. D. 1968. Adsorption of phosphorus by lake sediment. Soil Science Society of America Proceedings 32:514-518.

Harter, R. D. 1969. Phosphorus adsorption sites in soils. Soil Science Society of America Proceedings 33:630-632.

Holford, I. C. R., R. W. M. Wedderburn, and G. E. G. Mattingly. 1974. A Langmuir two-surface equation as a model for phosphate adsorption by soils. Journal of Soil Science 25:242-255.

Holford, I. C. R. and W. H. Patrick, Jr. 1979. Effects of reduction and pH changes on phosphate sorption and mobility in an acid soil. Soil Science Society of America

Journal 43:292-297.

Howard-Williams, C. 1985. Cycling and retention of nitrogen and phosphorus in wetlands: a theoretical and applied perspective. Freshwater Biology 15:391-431.

Jansson, S. L., and J. Persson. 1982. Mineralization and immobilization of soil nitrogen. p. 229-252. *In* F. J. Stevenson (ed.) Nitrogen in Agricultural Soils. American Society of Agronomy, Inc., Crop Science Society of America, Inc., Soil Science Society of America, Inc., Madison, WI, USA.

Johnston, C. A. 1991. Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. Critical Reviews in Environmental Control 21:491-565.

Johnston, C. A., G. D. Bubenzer, G. B. Lee, F. W. Madison, and J. R. McHenry. 1984. Nutrient trapping by sediment deposition in a seasonally flooded lakeside wetland. Journal of Environmental Quality 13:283-290.

Jugsujinda, A., R. D. DeLaune, and C. W. Lindau. 1995. Influence of nitrate on methane production and oxidation in flooded soil. Communications in Soil Science and Plant Analysis 26:2449-2459.

Khalid, R. A., W. H. Patrick, Jr., and R. D. DeLaune. 1977. Phosphorus sorption characteristics of flooded soils. Soil Science Society of America Journal 41:305-310.

LaBaugh, J. W., T. C. Winter, V. A. Adomaitis, and G. A. Swanson. 1987. Hydrology and chemistry of selected prairie wetlands in the Cottonwood Lake Area, Stutsman County, North Dakota, 1979-82. U.S. Geological Survey Professional Paper 1431. United States Government Printing Office, Washington D.C., USA.

LaBaugh, J. W. 1989. Chemical characteristics of water in northern prairie wetlands. p. 56-90. *In* A. van der Valk (ed.) Northern Prairie Wetlands. Iowa State University Press, Ames, IA, USA.

Larsen, S. 1967. Soil phosphorus. Advances in Agronomy 19:151-210.

Lemme, T. H. 1988. Denitrification and tillage relationships on two eastern South Dakota soils. M. S. Thesis. South Dakota State University, Brookings, SD, USA.

Lindsay, W. L. 1979. Chemical equilibria in soils. John Wiley & Sons, Inc, New York, NY, USA. 449 p.

Machacek, T. A. 1995. Water budget, hydrology, and above-ground biomass relationships between wetlands and uplands in agricultural environments. M. S. Thesis. South Dakota State University, Brookings, SD, USA.

Malo, D. D., B. K. Worcester, D. K. Cassel, and K. D. Matzdorf. 1974. Soil-landscape relationships in a closed drainage system. Soil Science Society of America Journal 38:813-818.

Massey, H. F. and M. L. Jackson. 1952. Selective erosion of soil fertility constituents. Soil Science Society of America Proceedings 16:353-356.

McCallister, D. L. and T. J. Logan. 1978. Phosphate adsorption-desorption characteristics of soils and bottom sediments in the Maumee river basin of Ohio. Journal of Environmental Quality 7:87-92.

McCarty, G. W. and J. M. Bremner. 1992. Availability of organic carbon for denitrification of nitrate in subsoils. Biology and Fertility of Soils 14:219-222.

McCarty, G. W. and J. M. Bremner. 1993. Factors affecting the availability of organic carbon for denitrification of nitrate in subsoils. Biology and Fertility of Soils 15:132-136.

McKenzie, E. and L. T. Kurtz. 1976. Effect of pretreatment on loss of nitrogen-15-labelled fertilizer from waterlogged soil during incubation. Soil Science Society of America Journal 40:534-537.

Miller, M. H., J. B. Robinson, D. R. Coote, A. C. Spires, and D. W. Draper. 1982. Agriculture and water quality in the Canadian Great Lake Basin: III. Phosphorus. Journal of Environmental Quality 11:487-493.

Mirvish, S. S. 1983. The etiology of gastric cancer: intragastric nitrosamide formation and other theories. Journal of the National Cancer Institute 71:630-647.

Mitsch, W. J. and J. G. Gosselink. 1993a. Wetland types and wetland resources of North America. p. 30-63. *In* W. J. Mitsch and J. G. Gosselink (ed.) Wetlands. Van Nostrand Reinhold, New York, NY, USA.

Mitsch, W. J. and J. G. Gosselink. 1993b. Hydrology of wetlands. p. 67-113. *In* W. J. Mitsch and J. G. Gosselink (ed.) Wetlands. Van Nostrand Reinhold, New York, NY, USA.

Mitsch, W. J. and J. G. Gosselink. 1993c. Biogeochemistry of wetlands. p. 114-163. *In* W. J. Mitsch and J. G. Gosselink (ed.) Wetlands. Van Nostrand Reinhold, New York, NY, USA.

Muljadi, D., A. M. Posner, and J. P. Quirk. 1966. The mechanism of phosphate adsorption by kaolinite, gibbsite and pseudoboehmite. Journal of Soil Science 17:212-229.

Neely, R. K. and J. L. Baker. 1989. Nitrogen and phosphorus dynamics and the fate of agricultural runoff. p. 92-131. *In* A. van der Valk (ed.) Northern Prairie Wetlands. Iowa State University Press, Ames, IA, USA.

O'Halloran, I. P. 1993. Total and organic phosphorus. p. 213-230. *In* M. R. Carter (ed.) Soil sampling and methods of analysis. Lewis Publishing, Boca Raton, FL, USA.

Olsen, S. R. and L. E. Sommers. 1982. Phosphorus. p. 403-430. *In* A. L. Page (ed.) Methods of soil analysis, Part 2, 2nd edition. American Society of Agronomy, Madison, WI, USA.

Patrick, W. H., Jr. and R. A. Khalid. 1974. Phosphate release and sorption by soils and sediments: effect of aerobic and anaerobic conditions. Science 186:53-55.

Patrick, W. H., Jr. 1982. Nitrogen transformations in submerged soils. p. 449-465. *In* F. J. Stevenson (ed.) Nitrogen in Agricultural Soils. American Society of Agronomy, Inc., Crop Science Society of America, Inc., Soil Science Society of America, Inc., Madison, WI, USA.

Patrick, W. H., Jr. 1994. From wastelands to wetlands. Journal of Environmental Quality 23:892-896.

Patten, D. K., J. M. Bremner, and A. M. Blackmer. 1980. Effects of drying and air-dry storage of soils on their capacity for denitrification of nitrate. Soil Science Society of America Journal 44:67-70.

Pennock, D. J., C. van Kessel, R. E. Farrell, and R. A. Sutherland. 1992. Landscape-scale variations in denitrification. Soil Science Society of America Journal 56:770-776.

Pierson, F. B. and D. J. Mulla. 1990. Aggregate stability in the Palouse region of Washington: effect of landscape position. Soil Science Society of America Journal 54:1407-1412.

Reddy, G. B. and K. R. Reddy. 1987. Nitrogen transformations in ponds receiving polluted water from nonpoint sources. Journal of Environmental Quality 16:1-5.

Reddy, K. R. and P. M. Gale. 1994. Wetland processes and water quality: a symposium overview. Journal of Environmental Quality 23:875-877.

Richardson, C. J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228:1424-1427.

Richardson, J. L., Wilding, L. P., and R. B. Daniels. 1992. Recharge and discharge of groundwater in aquic conditions illustrated with flownet analysis. Geoderma 53:65-78.

Rickerl, D. H., D. E. Kringen, and T. A. Machacek. 1995. Wetlands/groundwater quality in agricultural landscapes. Journal of the Minnesota Academy of Science 59:18-24.

Seyers, J. K., R. F. Harris, and D. E. Armstrong. 1973. Phosphate chemistry in lake sediments. Journal of Environmental Quality 2:1-14.

Sharpley, A. N. and R. G. Menzel. 1987. The impact of soil and fertilizer phosphorus on the environment. Advances in Agronomy 41:297-324.

Shukla, S. S., J. K. Syers, J. D. H. Williams, D. E. Armstrong, and R. F. Harris. 1971. Sorption of inorganic phosphate by lake sediments. Soil Science Society of America Proceedings 35:244-249.

Smith, M. S. and L. L. Parsons. 1985. Persistence of denitrifying enzyme activity in dried soils. Applied and Environmental Microbiology 49:316-320.

Sonzogni, W. C., S. C. Shapra, D. E. Armstrong, and T. J. Logan. 1982. Bioavailability of phosphorus inputs to lakes. Journal of Environmental Quality 11:555-563.

South Dakota Technical Guide. 1992. Section 1 – General Resources References.

Stevenson, F. J. 1965. Origin and distribution of nitrogen in soil. p.1-42. *In* W. V. Bartholomew and F. E. Clark (ed.) Soil Nitrogen. American Society of Agronomy, Inc., Madison, WI, USA.

Stevenson, F. J. 1982. Origin and distribution of nitrogen in soil. p. 1-42. *In* F. J. Stevenson (ed.) Nitrogen in Agricultural Soils. American Society of Agronomy, Inc., Crop Science Society of America, Inc., Soil Science Society of America, Inc., Madison, WI, USA.

Steward, K. K. and W. H. Ornes. 1975. Assessing a marsh environment for wastewater renovation. Journal of the Water Pollution Control Federation 47:1880-1891.

Stewart, R. E., and H. A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. U.S. Fish and Wildlife Service Resource Publication 92.

Tiedje, J. M., A. J. Sexstone, D. D. Myrold, and J. A. Robinson. 1982. Denitrification: ecological niches, competition and survival. Antonie van Leeuwenhoek 48:569-583.

Tiner, R. W., Jr. 1984. Wetlands of the United States: current status and recent trends. Washington D. C.: U.S. Fish and Wildlife Service, March, 1984.

Ulehlova, B. 1978. Decomposition processes in the fishpond littoral. pp. 341-353. *In* D. Dykyjova and J. Kvet (ed.) Pond littoral ecosystems. Springer-Verlag, Berlin.

United States Department of Agriculture, Soil Conservation Service. 1973. Soil survey of Lake County, South Dakota. United States Government Printing Office, Washington, D. C., USA.

United States Department of Agriculture, Natural Resources Conservation Service. 1996. Water quality and agriculture: status, conditions, and trends. Working Paper #16.

van Kessel, C., D. J. Pennock, and R. E. Farrell. 1993. Seasonal variations in denitrification and nitrous oxide evolution at the landscape scale. Soil Science Society of America Journal 57:988-995.

Weisburger, J. H. 1986. Role of fat, fiber, nitrate and food additives in carcinogenesis: a critical evaluation and recommendations. Nutrition and Cancer 8:47-61.

Wells, N., and W. M. H. Saunders. 1960. Soil studies using sweet vernal to assess element availability. New Zealand Journal of Agricultural Research 3:279-299.

Winter, T. C. 1989. Hydrologic studies of wetlands in the northern prairie. p. 16-54. *In* A. van der Valk (ed.) Northern Prairie Wetlands. Iowa State University Press. Ames, IA, USA.

Yeomans, J. C., J. M. Bremner, and G. W. McCarty. 1992. Denitrification capacity and denitrification potential of subsurface soils. Communications in Soil Science and Plant Analysis 23:919-927.

APPENDICES

Table 35. Phosphorus forms and their biological availabilities.

Phosphorus fraction and form	Bioavailability
Dissolved	
Inorganic P: DRP, $H_2PO_4^-$, and HPO_4^{-2}	Directly available
Condensed P: DCP, P-O-P bonds	Converted to DRP through fairly rapid hydrolysis
Organic P: DOP, P-O-C bonds	Converted to DRP through biological mineralization; slow for high-molecular-weight compounds; may be rapid for simple organic compounds
Particulate	
Inorganic P: NAIP; P absorbed on metal hydrous oxides (Fe, Al), Fe- and Al-P minerals, non-apatite Ca-P	Partially available through dissolution or desorption of phosphate when DRP concentration is low due to dilution, biological uptake, or chemical immobilization
AIP, Ca-P mineral	Essentially unavailable because of slow dissolution of apatites
Organic P: nucleic acids, phospholipids, inositol phosphates, other	Converted to DRP through biological mineralization; may be rapid for substantial fraction of fresh plant tissue or animal wastes, slow for soil and sediment organic P
Condensed P	Released from plant tissues at senescence and hydrolyzed to DRP; small fraction compared with organic P

DRP = dissolved reactive P
DCP = dissolved condensed P
DOP = dissolved organic P
AIP = apatite inorganic P

Source: Sonzogni et al. (1982)

Table 36. Description of soil series within study area.

Soil Series Description			
Soil Series	Hydric	Description	Slope (%)
Worthing (Wo)	yes	Silty clay loam	< 1
Egan (EbB)	no	Silty clay loam	2 - 6
Egan (EeC2)	no	Silty clay loam	6 - 9
Egan (EaC)	no	Silty clay loam	6 - 9
Badus (Ba)	no	Silty clay loam	< 1

Table 37. Statistical analysis for the dependent variable nitrate for 1994.

Variable	df	Sum of Squares	Mean Square	F Value	Pr > F
LOCATION	2	4513.43	2256.71	111.12	0.0001
WCLASS	1	745.23	745.23	36.70	0.0001
LOCATION * WCLASS	2	380.68	190.34	9.37	0.0001
ERROR	184	3736.67	20.31		
TOTALS	189	9156.97			

Table 38. Statistical analysis for the dependent variable nitrate for 1995.

Variable	df	Sum of Squares	Mean Square	F Value	Pr > F
LOCATION	2	2398.96	1199.48	87.68	0.0001
WCLASS	1	111.41	111.41	8.14	0.0048
LOCATION * WCLASS	2	63.67	31.84	2.33	0.1002
ERROR	202	2763.32	13.68		
TOTALS	207	5524.02			

Table 39. Statistical analysis for the dependent variable orthophosphate for 1994.

Variable	df	Sum of Squares	Mean Square	F Value	Pr > F
LOCATION	2	24.75	12.37	60.55	0.0001
WCLASS	1	3.42	3.42	16.72	0.0011
LOCATION * WCLASS	2	1.03	0.52	2.52	0.0832
ERROR	185	37.80	0.20		
TOTALS	190	67.35			

Table 40. Statistical analysis for the dependent variable orthophosphate for 1995.

Variable	df	Sum of Squares	Mean Square	F Value	Pr > F
LOCATION	2	48.54	24.27	50.21	0.0001
WCLASS	1	1.96	1.96	4.05	0.0454
LOCATION * WCLASS	2	0.86	0.43	0.89	0.4124
ERROR	207	100.06	0.48		
TOTALS	212	155.33			

Table 41. Nitrate (NO_3-N) and orthophosphate (PO_4-P) concentrations (mg L^{-1}) for wetland surface water (WSW), wetland groundwater (WGW), and upland groundwater (UGW) samples from seasonal wetland #3 for 1994 and 1995.

1994					1995				
Date	Axis	Location	NO_3-N	PO_4-P	Date	Axis	Location	NO_3-N	PO_4-P
05/23	1	WSW	00.6	0.34	05/22	1	WSW	00.4	0.18
	1	WGW	00.5	0.06		1	WGW	00.9	0.05
	1	UGW	13.7	0.14		1	UGW	2.3	0.17
	2	WSW	00.6	0.34		2	WSW	00.1	0.02
	2	WGW	-	-		2	WGW	00.2	0.01
	2	UGW	-	-		2	UGW	-	-
06/06	1	WSW	00.0	1.34	06/12	1	WSW	00.2	0.82
	1	WGW	00.0	0.19		1	WGW	1.8	0.03
	1	UGW	00.0	0.08		1	UGW	14.9	0.18
	2	WSW	00.0	1.34		2	WSW	00.2	0.70
	2	WGW	00.0	0.06		2	WGW	00.4	0.06
	2	UGW	-	-		2	UGW	7.7	0.07
06/21	1	WSW	00.0	0.54	06/27	1	WSW	00.2	2.66
	1	WGW	00.0	0.15		1	WGW	2.7	0.12
	1	UGW	-	-		1	UGW	5.4	0.15
	2	WSW	00.0	0.54		2	WSW	00.3	2.45
	2	WGW	00.0	0.02		2	WGW	00.3	0.10
	2	UGW	-	-		2	UGW	8.6	0.13
07/06	1	WSW	-	-	07/18	1	WSW	00.0	2.75
	1	WGW	00.0	0.00		1	WGW	2.7	0.33
	1	UGW	00.8	0.19		1	UGW	3.3	0.16
	2	WSW	-	-		2	WSW	00.1	2.75
	2	WGW	-	-		2	WGW	00.3	0.00
	2	UGW	-	-		2	UGW	-	-
07/18	1	WSW	-	-	08/02	1	WSW	00.1	2.75
	1	WGW	00.0	0.68		1	WGW	2.4	0.34
	1	UGW	1.1	0.14		1	UGW	2.9	0.17
	2	WSW	-	-		2	WSW	00.0	2.75
	2	WGW	-	-		2	WGW	00.2	0.03
	2	UGW	-	-		2	UGW	-	-
					08/16	1	WSW	00.1	2.75
						1	WGW	1.2	0.18
						1	UGW	1.4	0.11
						2	WSW	00.0	2.75
						2	WGW	00.2	0.07
						2	UGW	-	-
					08/30	1	WSW	00.0	2.75
						1	WGW	1.3	0.25
						1	UGW	1.0	0.10
						2	WSW	00.0	2.75
						2	WGW	00.0	0.07
						2	UGW	-	-
					09/19	1	WSW	-	-
						1	WGW	1.0	0.24
						1	UGW	00.5	0.05
						2	WSW	-	-
						2	WGW	00.1	0.04
						2	UGW	-	-

Table 42. Nitrate (NO_3-N) and orthophosphate (PO_4-P) concentrations (mg L^{-1}) for wetland surface water (WSW), wetland groundwater (WGW), and upland groundwater (UGW) samples from seasonal wetland #4 for 1994 and 1995.

1994					1995				
Date	Axis	Location	NO_3-N	PO_4-P	Date	Axis	Location	NO_3-N	PO_4-P
05/25	1	WSW	00.0	2.23	05/24	1	WSW	00.5	0.24
	1	WGW	00.0	0.85		1	WGW	00.0	0.65
	1	UGW	15.7	0.00		1	UGW	10.6	0.12
	2	WSW	00.0	2.23		2	WSW	00.2	0.30
	2	WGW	00.0	1.42		2	WGW	1.7	2.63
	2	UGW	15.5	0.00		2	UGW	4.9	0.32
06/09	1	WSW	00.0	2.11	06/14	1	WSW	00.0	0.61
	1	WGW	00.8	0.51		1	WGW	00.1	0.34
	1	UGW	-	-		1	UGW	10.9	0.14
	2	WSW	00.0	2.11		2	WSW	00.2	0.45
	2	WGW	00.7	1.50		2	WGW	00.7	2.75
	2	UGW	-	-		2	UGW	4.6	0.19
06/22	1	WSW	00.0	0.87	06/29	1	WSW	00.0	1.27
	1	WGW	00.2	0.61		1	WGW	-	-
	1	UGW	19.6	0.08		1	UGW	6.2	0.10
	2	WSW	00.0	1.17		2	WSW	00.1	0.49
	2	WGW	00.0	1.34		2	WGW	00.8	2.10
	2	UGW	13.6	0.00		2	UGW	5.0	0.21
07/06	1	WSW	-	-	07/20	1	WSW	00.0	1.33
	1	WGW	-	-		1	WGW	-	-
	1	UGW	20.3	0.21		1	UGW	4.8	0.08
	2	WSW	-	-		2	WSW	00.2	1.39
	2	WGW	00.0	1.78		2	WGW	00.7	2.00
	2	UGW	10.8	0.17		2	UGW	4.6	0.17
07/20	1	WSW	-	-	08/04	1	WSW	-	-
	1	WGW	-	-		1	WGW	-	-
	1	UGW	11.9	0.04		1	UGW	4.2	0.09
	2	WSW	-	-		2	WSW	-	-
	2	WGW	1.0	0.21		2	WGW	00.0	2.04
	2	UGW	13.4	0.06		2	UGW	3.4	0.22
08/03	1	WSW	-	-	08/18	1	WSW	-	-
	1	WGW	-	-		1	WGW	-	-
	1	UGW	12.4	0.16		1	UGW	9.7	0.08
	2	WSW	-	-		2	WSW	-	-
	2	WGW	00.0	1.41		2	WGW	1.4	2.15
	2	UGW	5.2	0.60		2	UGW	3.7	0.07
08/17	1	WSW	-	-	09/01	1	WSW	-	-
	1	WGW	-	-		1	WGW	-	-
	1	UGW	9.1	0.10		1	UGW	4.9	0.15
	2	WSW	-	-		2	WSW	-	-
	2	WGW	00.0	0.79		2	WGW	00.0	2.05
	2	UGW	-	-		2	UGW	-	-
08/31	1	WSW	-	-	09/25	1	WSW	-	-
	1	WGW	-	-		1	WGW	-	-
	1	UGW	6.6	0.16		1	UGW	3.9	0.09
	2	WSW	-	-		2	WSW	-	-
	2	WGW	-	-		2	WGW	00.0	2.07
	2	UGW	-	-		2	UGW	-	-
09/16	1	WSW	-	-					
	1	WGW	-	-					
	1	UGW	12.1	0.21					
	2	WSW	-	-					
	2	WGW	-	-					

Table 42 cont.

	2	UGW	-	-					

Table 43. Nitrate (NO_3-N) and orthophosphate (PO_4-P) concentrations (mg L^{-1}) for wetland surface water (WSW), wetland groundwater (WGW), and upland groundwater (UGW) samples from semipermanent wetland #5 for 1994 and 1995.

1994					1995				
Date	Axis	Location	NO_3-N	PO_4-P	Date	Axis	Location	NO_3-N	PO_4-P
05/25	1	WSW	00.9	0.44	05/30	1	WSW	00.3	0.33
	1	WGW	6.1	0.21		1	WGW	00.5	0.20
	1	UGW	14.4	0.05		1	UGW	10.1	0.12
	2	WSW	00.9	0.44		2	WSW	00.2	0.30
	2	WGW	10.6	0.04		2	WGW	2.2	0.28
	2	UGW	35.0	0.00		2	UGW	4.8	0.23
06/10	1	WSW	00.6	0.18	06/16	1	WSW	00.2	0.47
	1	WGW	3.6	0.07		1	WGW	-	-
	1	UGW	15.0	0.01		1	UGW	12.7	0.24
	2	WSW	00.6	0.18		2	WSW	00.2	0.55
	2	WGW	12.2	0.05		2	WGW	1.9	0.17
	2	UGW	35.2	0.00		2	UGW	5.2	0.29
06/24	1	WSW	00.0	0.46	06/30	1	WSW	00.4	0.28
	1	WGW	-	-		1	WGW	-	-
	1	UGW	-	-		1	UGW	11.8	0.16
	2	WSW	00.5	0.08		2	WSW	00.2	0.42
	2	WGW	12.9	0.07		2	WGW	00.5	0.13
	2	UGW	35.0	0.00		2	UGW	4.5	0.11
07/08	1	WSW	00.8	0.04	07/21	1	WSW	00.2	0.41
	1	WGW	2.3	0.03		1	WGW	-	-
	1	UGW	-	-		1	UGW	-	-
	2	WSW	00.1	0.00		2	WSW	00.1	0.15
	2	WGW	9.8	0.04		2	WGW	1.9	0.34
	2	UGW	28.9	0.00		2	UGW	4.1	0.05
07/22	1	WSW	00.8	0.02	08/04	1	WSW	00.0	0.32
	1	WGW	-	-		1	WGW	-	-
	1	UGW	-	-		1	UGW	-	-
	2	WSW	00.8	0.00		2	WSW	00.2	0.44
	2	WGW	10.7	0.06		2	WGW	1.6	0.19
	2	UGW	19.3	0.00		2	UGW	17.0	0.08
08/05	1	WSW	00.0	0.35	08/18	1	WSW	00.0	0.51
	1	WGW	-	-		1	WGW	00.0	0.27
	1	UGW	-	-		1	UGW	-	-
	2	WSW	00.0	0.09		2	WSW	00.0	0.52
	2	WGW	11.1	0.10		2	WGW	00.9	0.07
	2	UGW	12.3	0.15		2	UGW	35.8	0.06
08/17	1	WSW	00.7	0.15	09/01	1	WSW	00.0	0.37
	1	WGW	-	-		1	WGW	-	-
	1	UGW	-	-		1	UGW	-	-
	2	WSW	00.5	0.07		2	WSW	00.0	0.35
	2	WGW	3.1	0.23		2	WGW	00.6	0.14
	2	UGW	-	-		2	UGW	-	-
09/02	1	WSW	00.3	0.09	09/25	1	WSW	00.0	0.13
	1	WGW	-	-		1	WGW	-	-
	1	UGW	-	-		1	UGW	-	-
	2	WSW	00.7	0.02		2	WSW	00.0	0.18
	2	WGW	8.0	0.12		2	WGW	00.2	0.19
	2	UGW	-	-		2	UGW	-	-
09/16	1	WSW	00.2	0.30					

Table 43 cont.

	1	WGW	-	-					
	1	UGW	-	-					
	2	WSW	00.3	0.29					
	2	WGW	10.2	0.21					
	2	UGW	-	-					

Table 44. Nitrate (NO_3-N) and orthophosphate (PO_4-P) concentrations (mg L^{-1}) for wetland surface water (WSW), wetland groundwater (WGW), and upland groundwater (UGW) samples from seasonal wetland #6 for 1994 and 1995.

1994					1995				
Date	Axis	Location	NO_3-N	PO_4-P	Date	Axis	Location	NO_3-N	PO_4-P
05/27	1	WSW	00.0	0.45	05/31	1	WSW	00.0	0.24
	1	WGW	6.1	0.14		1	WGW	00.9	0.20
	1	UGW	7.7	0.07		1	UGW	16.8	0.20
	2	WSW	00.0	0.45		2	WSW	00.0	0.07
	2	WGW	00.0	0.12		2	WGW	10.4	0.47
	2	UGW	6.3	0.00		2	UGW	6.1	0.27
06/13	1	WSW	00.0	0.97	06/21	1	WSW	00.0	0.49
	1	WGW	6.5	0.03		1	WGW	00.5	0.11
	1	UGW	11.1	0.01		1	UGW	17.9	0.15
	2	WSW	00.0	0.97		2	WSW	00.1	2.75
	2	WGW	1.4	0.21		2	WGW	6.9	0.09
	2	UGW	8.3	0.00		2	UGW	15.0	0.25
06/29	1	WSW	00.3	0.43	07/13	1	WSW	00.1	0.90
	1	WGW	6.3	0.04		1	WGW	1.3	0.14
	1	UGW	8.5	0.02		1	UGW	15.5	0.10
	2	WSW	00.0	1.35		2	WSW	00.0	2.21
	2	WGW	00.3	0.04		2	WGW	8.0	0.15
	2	UGW	-	-		2	UGW	22.2	0.17
07/11	1	WSW	00.4	0.20	07/25	1	WSW	00.0	0.53
	1	WGW	9.6	0.03		1	WGW	1.7	0.10
	1	UGW	-	-		1	UGW	12.4	0.19
	2	WSW	00.2	0.64		2	WSW	00.0	2.75
	2	WGW	1.4	0.36		2	WGW	5.8	0.17
	2	UGW	16.8	0.10		2	UGW	12.8	0.09
07/25	1	WSW	00.0	1.32	08/10	1	WSW	00.0	0.50
	1	WGW	9.4	0.20		1	WGW	2.0	0.13
	1	UGW	-	-		1	UGW	-	-
	2	WSW	00.0	1.17		2	WSW	00.0	2.17
	2	WGW	00.0	0.16		2	WGW	5.2	0.16
	2	UGW	18.8	0.07		2	UGW	17.9	0.08
08/08	1	WSW	-	-	08/22	1	WSW	00.0	1.79
	1	WGW	5.7	0.09		1	WGW	1.0	0.18
	1	UGW	-	-		1	UGW	-	-
	2	WSW	-	-		2	WSW	00.0	2.75
	2	WGW	-	0.03		2	WGW	3.6	0.18
	2	UGW	12.2	0.03		2	UGW	14.0	0.11
08/22	1	WSW	-	-	09/08	1	WSW	-	-
	1	WGW	11.9	0.51		1	WGW	1.0	0.24
	1	UGW	-	-		1	UGW	-	-
	2	WSW	-	-		2	WSW	-	-
	2	WGW	8.7	0.22		2	WGW	2.9	0.26
	2	UGW	17.7	0.49		2	UGW	14.1	0.12
09/07	1	WSW	-	-	10/02	1	WSW	-	-
	1	WGW	9.2	0.18		1	WGW	00.0	0.12
	1	UGW	-	-		1	UGW	-	-

Table 44 cont.

	2	WSW	-	-		2	WSW	-	-
	2	WGW	2.1	0.25		2	WGW	00.6	0.65
	2	UGW	16.3	0.63		2	UGW	-	-

Table 45. Nitrate (NO_3-N) and orthophosphate (PO_4-P) concentrations (mg L^{-1}) for wetland surface water (WSW), wetland groundwater (WGW), and upland groundwater (UGW) samples from semipermanent wetland #8 for 1994 and 1995.

1994					1995				
Date	Axis	Location	NO_3-N	PO_4-P	Date	Axis	Location	NO_3-N	PO_4-P
06/01	1	WSW	00.8	1.65	06/02	1	WSW	00.1	0.94
	1	WGW	9.0	0.00		1	WGW	8.1	0.21
	1	UGW	00.0	0.00		1	UGW	-	0.15
	2	WSW	00.8	1.65		2	WSW	00.1	0.49
	2	WGW	3.9	0.00		2	WGW	8.2	0.19
	2	UGW	7.2	-		2	UGW	10.9	0.18
06/15	1	WSW	00.4	1.65	06/22	1	WSW	00.2	2.73
	1	WGW	8.9	0.01		1	WGW	10.1	0.06
	1	UGW	-	-		1	UGW	-	0.09
	2	WSW	00.4	1.65		2	WSW	00.2	2.75
	2	WGW	9.7	0.05		2	WGW	-	-
	2	UGW	1.8	0.03		2	UGW	10.1	0.12
06/29	1	WSW	00.2	0.51	07/14	1	WSW	00.2	2.54
	1	WGW	9.5	0.00		1	WGW	10.6	0.15
	1	UGW	-	-		1	UGW	00.8	0.07
	2	WSW	00.3	1.16		2	WSW	00.1	2.69
	2	WGW	-	-		2	WGW	-	-
	2	UGW	00.9	0.00		2	UGW	10.4	0.08
07/14	1	WSW	00.6	1.32	07/26	1	WSW	00.1	2.40
	1	WGW	13.4	0.11		1	WGW	6.5	0.07
	1	UGW	-	-		1	UGW	2.0	0.11
	2	WSW	00.6	1.82		2	WSW	00.0	2.46
	2	WGW	-	-		2	WGW	-	-
	2	UGW	12.9	0.06		2	UGW	11.6	0.12
07/29	1	WSW	00.6	0.37	08/14	1	WSW	00.0	2.40
	1	WGW	8.8	0.09		1	WGW	5.9	0.13
	1	UGW	-	-		1	UGW	-	-
	2	WSW	00.8	0.53		2	WSW	00.0	2.35
	2	WGW	-	-		2	WGW	-	-
	2	UGW	-	-		2	UGW	15.4	0.00
08/11	1	WSW	00.7	0.15	08/23	1	WSW	00.0	2.12
	1	WGW	7.4	0.05		1	WGW	4.9	0.10
	1	UGW	-	-		1	UGW	-	-
	2	WSW	00.6	0.07		2	WSW	00.1	2.21
	2	WGW	-	-		2	WGW	-	-
	2	UGW	-	-		2	UGW	-	-
08/24	1	WSW	00.3	1.51	09/11	1	WSW	00.0	1.19
	1	WGW	10.0	0.17		1	WGW	3.8	0.05
	1	UGW	-	-		1	UGW	-	-
	2	WSW	00.4	1.66		2	WSW	00.0	1.29
	2	WGW	-	-		2	WGW	-	-
	2	UGW	-	-		2	UGW	-	-
09/09	1	WSW	00.3	1.82	10/03	1	WSW	00.0	0.80
	1	WGW	10.6	0.15		1	WGW	2.5	0.09
	1	UGW	-	-		1	UGW	-	-
	2	WSW	00.2	1.92		2	WSW	00.0	0.83
	2	WGW	-	-		2	WGW	-	-

Table 45 cont.

	2	UGW	-	-		2	UGW	-	-
09/23	1	WSW	00.2	1.17					
	1	WGW	6.9	0.10					
	1	UGW	-	-					
	2	WSW	00.4	1.20					
	2	WGW	-	-					
	2	UGW	-	-					

Table 46. Nitrate (NO_3-N) and orthophosphate (PO_4-P) concentrations (mg L^{-1}) for wetland surface water (WSW), wetland groundwater (WGW), and upland groundwater (UGW) samples from seasonal wetland #9 for 1994 and 1995.

1994					1995				
Date	Axis	Location	NO_3-N	PO_4-P	Date	Axis	Location	NO_3-N	PO_4-P
06/03	1	WSW	00.0	1.38	06/06	1	WSW	00.4	0.85
	1	WGW	00.4	0.38		1	WGW	00.2	0.12
	1	UGW	1.2	0.24		1	UGW	-	0.17
	2	WSW	00.0	1.38		2	WSW	00.2	0.84
	2	WGW	00.0	0.00		2	WGW	00.2	0.07
	2	UGW	9.5	0.00		2	UGW	13.1	0.10
06/20	1	WSW	00.1	1.61	06/23	1	WSW	-	1.05
	1	WGW	00.2	0.37		1	WGW	00.1	0.10
	1	UGW	10.0	0.14		1	UGW	1.8	0.09
	2	WSW	00.2	1.84		2	WSW	-	0.73
	2	WGW	00.3	0.09		2	WGW	00.2	0.06
	2	UGW	10.9	0.04		2	UGW	8.8	0.01
07/01	1	WSW	00.0	1.28	07/17	1	WSW	00.0	0.76
	1	WGW	00.0	0.49		1	WGW	00.2	0.12
	1	UGW	12.8	0.08		1	UGW	3.6	0.11
	2	WSW	00.6	1.98		2	WSW	1.0	1.10
	2	WGW	00.0	0.09		2	WGW	00.4	0.08
	2	UGW	-	-		2	UGW	10.1	0.04
07/15	1	WSW	00.0	1.42	07/28	1	WSW	-	-
	1	WGW	00.6	0.14		1	WGW	00.2	0.40
	1	UGW	13.5	0.09		1	UGW	2.3	0.19
	2	WSW	00.0	1.89		2	WSW	-	-
	2	WGW	00.5	0.14		2	WGW	00.0	0.19
	2	UGW	-	0.00		2	UGW	6.8	0.12
07/29	1	WSW	-	-	08/16	1	WSW	-	-
	1	WGW	1.4	0.47		1	WGW	00.2	0.54
	1	UGW	16.5	0.08		1	UGW	-	-
	2	WSW	-	-		2	WSW	-	-
	2	WGW	00.0	0.34		2	WGW	00.1	0.18
	2	UGW	7.1	0.09		2	UGW	7.3	0.03
08/12	1	WSW	-	-	08/24	1	WSW	-	-
	1	WGW	00.6	0.09		1	WGW	00.1	0.55
	1	UGW	-	-		1	UGW	-	-
	2	WSW	-	-		2	WSW	-	-
	2	WGW	00.6	0.08		2	WGW	00.0	0.17
	2	UGW	9.0	0.07		2	UGW	6.8	0.09
08/26	1	WSW	-	-	09/11	1	WSW	-	-
	1	WGW	00.0	0.39		1	WGW	00.0	0.24
	1	UGW	-	-		1	UGW	-	-
	2	WSW	-	-		2	WSW	-	-
	2	WGW	00.0	0.33		2	WGW	00.0	0.16
	2	UGW	-	-		2	UGW	5.3	0.09
09/09	1	WSW	-	-	10/03	1	WSW	-	-

Table 46 cont.

	1	WGW	1.1	0.34		1	WGW	00.2	0.19
	1	UGW	-	-		1	UGW	-	-
	2	WSW	-	-		2	WSW	-	-
	2	WGW	00.5	0.42		2	WGW	00.0	0.14
	2	UGW	-	-		2	UGW	-	-
09/23	1	WSW	-	-					
	1	WGW	00.6	0.22					
	1	UGW	-	-					
	2	WSW	-	-					
	2	WGW	00.0	0.42					
	2	UGW	-	-					

Table 47. Chemical analysis from the Soil Testing Laboratory, SDSU for seasonal wetland #3 soil cores.

Wetland	Axis	Depth (cm)	Nitrate-N (mg L^{-1})	Total N (%)	Avail P (mg L^{-1})	Total P (mg g^{-1})
3	2-1	0-15	9.2	0.49	31	0.613
3	2-1	15-30	6.6	0.38	44	
3	2-1	30-45	4.8	0.23	54	
3	2-1	45-60	4.2	0.11	47	
3	2-1	60-75	3.8	0.08	40	
3	2-1	75-90	3.4	0.07	38	
3	2-1	90-105	2.6	0.05	44	
3	2-1	105-114	3.4	0.04	47	
3	2-2	0-15	15.4	0.37	24	0.612
3	2-2	15-30	5.8	0.28	17	
3	2-2	30-45	6.2	0.23	13	
3	2-2	45-60	6.0	0.14	5	
3	2-2	60-75	6.8	0.12	1	
3	2-2	75-90	6.6	0.50	0	
3	2-2	90-105	6.2	0.06	0	
3	2-2	105-120	5.4	0.04	0	
3	3-1	0-15	7.0	0.41	30	0.773
3	3-1	15-30	9.4	0.36	17	
3	3-1	30-45	7.4	0.27	16	
3	3-1	45-60	7.6	0.17	1	
3	3-1	60-75	5.6	0.07	4	
3	3-1	75-90	5.8	0.03	0	
3	3-2	0-15	25.8	0.27	25	0.622
3	3-2	15-30	9.0	0.19	19	
3	3-2	30-45	5.0	0.12	10	
3	3-2	45-60	4.4	0.10	8	
3	3-2	60-75	3.4	0.08	5	
3	3-2	75-90	5.0	0.05	2	
3	3-2	90-105	4.6	0.04	1	
3	3-2	105-112	4.8	0.03	1	
3	3-5	0-15	7.4	0.21	26	0.600
3	3-5	15-30	6.2	0.15	17	
3	3-5	30-45	5.2	0.08	1	
3	3-5	45-60	3.4	0.04	2	
3	3-5	60-75	3.2	0.03	1	
3	3-5	75-90	3.6	0.02	1	
3	3-5	90-97	5.4	0.02	1	

Table 48. Chemical analysis from the Soil Testing Laboratory, SDSU for seasonal wetland #4 soil cores.

Wetland	Axis	Depth (cm)	Nitrate-N (mg L^{-1})	Total N (%)	Avail P (mg L^{-1})	Total P (mg g^{-1})
4	1-1	0-15	5.6	0.38	47	0.647
4	1-1	15-30	5.4	0.33	32	
4	1-1	30-45	5.6	0.17	16	
4	1-1	45-60	2.4	0.09	7	
4	1-1	60-75	1.2	0.07	10	
4	1-1	75-90	1.6	0.05	9	
4	1-1	90-105	1.6	0.03	11	
4	1-1	105-117	1.4	0.03	10	
4	1-2	0-15	7.2	0.26	35	0.543
4	1-2	15-30	7.2	0.23	25	
4	1-2	30-45	6.0	0.17	16	
4	1-2	45-60	6.0	0.14	11	
4	1-2	60-75	8.6	0.10	7	
4	1-2	75-90	11.8	0.08	5	
4	1-2	90-105	11.0	0.04	0	
4	1-2	105-119	10.4	0.03	1	
4	2-1	0-15	10.2	0.44	32	0.663
4	2-1	15-30	6.0	0.33	32	
4	2-1	30-45	5.4	0.22	19	
4	2-1	45-60	3.2	0.14	11	
4	2-1	60-75	2.0	0.08	32	
4	2-1	75-90	1.8	0.07	48	
4	2-1	90-105	2.0	0.07	57	
4	2-1	105-109	2.2	0.07	54	
4	2-2	0-15	6.4	0.23	25	0.536
4	2-2	15-30	6.0	0.13	11	
4	2-2	30-45	4.0	0.10	9	
4	2-2	45-60	5.6	0.05	0	
4	2-2	60-75	5.4	0.04	0	
4	2-2	75-90	4.0	0.03	0	
4	2-2	90-105	3.8	0.03	0	

Table 49. Chemical analysis from the Soil Testing Laboratory, SDSU for semipermanent wetland #5 soil cores.

Wetland	Axis	Depth (cm)	Nitrate-N (mg L^{-1})	Total N (%)	Avail P (mg L^{-1})	Total P (mg g^{-1})
5	1-1	0-15	6.0	0.33	43	0.725
5	1-1	15-30	5.2	0.29	25	
5	1-1	30-45	5.2	0.24	20	
5	1-1	45-60	2.8	0.23	16	
5	1-1	60-75	4.8	0.09	1	
5	1-1	75-90	2.4	0.03	0	
5	1-1	90-105	2.4	0.00	2	
5	1-1	105-120	3.8	0.00	3	
5	1-2	0-15	6.6	0.23	32	0.576
5	1-2	15-30	3.8	0.13	9	
5	1-2	30-45	2.6	0.07	3	
5	1-2	45-60	3.4	0.05	1	
5	1-2	60-75	3.2	0.04	1	
5	1-2	75-90	3.8	0.02	1	
5	1-2	90-105	5.6	0.03	0	
5	1-2	105-116	5.4	0.02	0	
5	1-5	0-15	5.2	0.25	29	0.593
5	1-5	15-30	3.6	0.14	15	
5	1-5	30-45	2.8	0.07	1	
5	1-5	45-60	3.4	0.05	0	
5	1-5	60-75	3.4	0.05	0	
5	1-5	75-90	4.4	0.04	0	
5	1-5	90-105	5.4	0.02	1	
5	1-5	105-112	5.2	0.01	1	
5	2-1	0-15	4.4	0.46	37	0.760
5	2-1	15-30	5.6	0.26	24	
5	2-1	30-45	5.6	0.12	12	
5	2-1	45-60	6.0	0.07	6	
5	2-1	60-75	7.4	0.05	5	
5	2-1	75-90	8.0	0.03	4	
5	2-1	90-105	8.4	0.03	1	
5	2-1	105-114	9.2	0.03	0	
5	2-2	0-15	6.4	0.24	26	0.572
5	2-2	15-30	4.0	0.16	16	
5	2-2	30-45	3.8	0.10	8	
5	2-2	45-60	3.4	0.05	3	
5	2-2	60-75	3.2	0.04	2	
5	2-2	75-90	2.6	0.03	2	
5	2-2	90-105	3.0	0.03	2	
5	2-2	105-120	4.4	0.02	1	

Table 50. Chemical analysis from the Soil Testing Laboratory, SDSU for seasonal wetland #6 soil cores.

Wetland	Axis	Depth (cm)	Nitrate-N (mg L^{-1})	Total N (%)	Avail P (mg L^{-1})	Total P (mg g^{-1})
6	1-1	0-15	4.4	0.40	56	0.771
6	1-1	15-30	3.6	0.33	24	
6	1-1	30-45	3.4	0.13	11	
6	1-1	45-60	3.0	0.07	7	
6	1-1	60-75	3.0	0.06	4	
6	1-1	75-90	3.2	0.06	6	
6	1-1	90-105	2.6	0.05	5	
6	1-1	105-120	4.4	0.04	3	
6	1-2	0-15	26.8	0.32	45	0.675
6	1-2	15-30	7.4	0.18	21	
6	1-2	30-45	5.2	0.10	8	
6	1-2	45-60	4.8	0.07	3	
6	1-2	60-75	5.4	0.05	0	
6	1-2	75-90	5.2	0.04	0	
6	1-2	90-105	6.6	0.04	1	
6	1-2	105-119	6.4	0.03	2	
6	2-1	0-15	8.4	0.38	44	0.607
6	2-1	15-30	6.2	0.24	24	
6	2-1	30-45	5.6	0.12	13	
6	2-1	45-60	5.4	0.05	6	
6	2-1	60-75	3.6	0.05	5	
6	2-1	75-90	4.6	0.05	7	
6	2-1	90-105	4.8	0.04	3	
6	2-1	105-117	3.6	0.04	3	
6	2-2	0-15	18.4	0.30	41	0.669
6	2-2	15-30	10.2	0.20	28	
6	2-2	30-45	8.6	0.12	17	
6	2-2	45-60	5.2	0.07	8	
6	2-2	60-75	5.0	0.05	6	
6	2-2	75-90	6.4	0.02	1	
6	2-2	90-105	6.8	0.02	1	
6	2-2	105-114	8.2	0.03	0	

Table 51. Chemical analysis from the Soil Testing Laboratory, SDSU for semipermanent wetland #8 soil cores.

Wetland	Axis	Depth (cm)	Nitrate-N (mg L^{-1})	Total N (%)	Avail P (mg L^{-1})	Total P (mg g^{-1})
8	1-1	0-15	6.8	0.37	33	0.695
8	1-1	15-30	7.0	0.30	25	
8	1-1	30-45	9.8	0.35	27	
8	1-1	45-60	16.0	0.39	11	
8	1-1	60-75	9.0	0.23	2	
8	1-1	75-90	8.4	0.12	3	
8	1-1	90-105	7.0	0.07	1	
8	1-1	105-120	4.2	0.04	1	
8	1-2	0-15	8.8	0.25	34	0.586
8	1-2	15-30	4.8	0.18	18	
8	1-2	30-45	3.4	0.10	5	
8	1-2	45-60	2.4	0.07	4	
8	1-2	60-75	3.0	0.06	1	
8	1-2	75-90	2.6	0.04	1	
8	1-2	90-105	3.0	0.05	5	
8	1-2	105-114	4.0	0.04	1	
8	1-5	0-15	16.0	0.27	37	0.681
8	1-5	15-30	5.8	0.17	22	
8	1-5	30-45	3.0	0.11	7	
8	1-5	45-60	3.0	0.08	5	
8	1-5	60-75	3.8	0.07	3	
8	1-5	75-90	4.4	0.05	1	
8	1-5	90-105	4.8	0.04	1	
8	1-5	105-120	5.8	0.04	1	
8	2-1	0-15	7.4	0.36	49	0.697
8	2-1	15-30	8.0	0.37	37	
8	2-1	30-45	8.2	0.32	29	
8	2-1	45-60	10.6	0.32	16	
8	2-1	60-75	13.2	0.20	4	
8	2-1	75-90	12.0	0.14	1	
8	2-1	90-105	8.4	0.12	1	
8	2-2	0-15	7.4	0.20	25	0.555
8	2-2	15-30	3.2	0.10	11	
8	2-2	30-45	3.4	0.07	1	
8	2-2	45-60	2.8	0.04	1	
8	2-2	60-75	3.8	0.03	1	
8	2-2	75-90	5.0	0.03	1	
8	2-2	90-105	4.8	0.03	0	
8	2-2	105-114	5.4	0.03	1	

Table 52. Chemical analysis from the Soil Testing Laboratory, SDSU for seasonal wetland #9 soil cores.

Wetland	Axis	Depth (cm)	Nitrate-N (mg L^{-1})	Total N (%)	Avail P (mg L^{-1})	Total P (mg g^{-1})
9	1-1	0-15	8.0	0.43	59	0.784
9	1-1	15-30	7.6	0.39	33	
9	1-1	30-45	5.2	0.24	18	
9	1-1	45-60	1.4	0.14	11	
9	1-1	60-75	4.6	0.11	7	
9	1-1	75-90	5.0	0.10	7	
9	1-1	90-105	3.4	0.05	6	
9	1-1	105-117	4.4	0.06	0	
9	1-2	0-15	11.6	0.27	57	0.627
9	1-2	15-30	6.6	0.27	36	
9	1-2	30-45	10.8	0.24	31	
9	1-2	45-60	11.8	0.16	22	
9	1-2	60-75	11.6	0.13	16	
9	1-2	75-90	12.8	0.13	15	
9	1-2	90-105	11.0	0.12	16	
9	1-2	105-120	11.2	0.12	15	
9	2-1	0-15	9.2	0.44	57	0.805
9	2-1	15-30	7.6	0.40	59	
9	2-1	30-45	5.8	0.23	27	
9	2-1	45-60	5.0	0.15	13	
9	2-1	60-75	4.0	0.10	7	
9	2-1	75-90	2.6	0.08	8	
9	2-1	90-105	2.0	0.06	9	
9	2-1	105-114	3.2	0.04	1	
9	2-2	0-15	17.8	0.24	27	0.518
9	2-2	15-30	8.8	0.19	20	
9	2-2	30-45	6.0	0.15	14	
9	2-2	45-60	7.4	0.11	9	
9	2-2	60-75	7.4	0.09	1	
9	2-2	75-90	8.6	0.06	0	
9	2-2	90-105	7.6	0.03	0	
9	2-2	105-114	8.4	0.04	0	
9	2-5	0-15	10.6	0.18	34	0.518
9	2-5	15-30	8.4	0.13	27	
9	2-5	30-45	6.0	0.08	12	
9	2-5	45-60	5.4	0.04	11	
9	2-5	60-75	6.8	0.02	11	
9	2-5	75-79	7.0	0.03	3	

Table 53. Statistical analysis for total phosphorus as effected by landscape position in seasonal wetland landscapes.

Variable	df	Sum of Squares	Mean Square	F value	Pr > F	LSD 0.05
Position	1	31416.89	31416.89	4.74	0.0723	NS
Error	6	39741.88	6623.65			
Totals	7	71158.77				

Table 54. Statistical analysis for total phosphorus as effected by landscape position in semipermanent wetland landscapes.

Variable	df	Sum of Squares	Mean Square	F value	Pr > F	LSD 0.05
Position	1	43218.00	43218.00	39.75	0.0242	0.142
Error	2	2174.50	1087.25			
Totals	3	45392.50				

Table 55. Statistical analysis for total phosphorus as effected by position from seasonal wetland #6 sediment samples.

Variable	df	Sum of Squares	Mean Square	F Value	Pr > F	LSD 0.05
Position	4	115772.87	28943.22	2.78	0.0487	0.121
Error	25	260137.00	10405.48			
Total	29	375909.87				

Table 56. Total phosphorus concentrations as effected by position for sediment samples from seasonal wetland #6.

Axis	Location	mg P g^{-1} soil
1	A	0.876
	B	0.898
	C	0.696
	D	0.498
	E	0.483
2	A	0.588
	B	0.476
	C	0.586
	D	0.538
	E	0.604
3	A	0.655
	B	0.563
	C	0.468
	D	0.395
	E	0.500
4	A	0.648
	B	0.730
	C	0.567
	D	0.588
	E	0.399
5	A	0.629
	B	0.709
	C	0.607
	D	0.642
	E	0.544
6	A	0.644
	B	0.563
	C	0.578
	D	0.609
	E	0.547

A = 3.8 m from wetland border
B = 11.4 m from wetland border
C = 19.1 m from wetland border
D = 26.7 m from wetland border
E = 34.3 m from wetland border