Prepared in cooperation with the Missouri Department of Conservation

Hydrology and Cycling of Nitrogen and Phosphorus in Little Bean Marsh: A Remnant Riparian Wetland along the Missouri River in Platte County, Missouri, 1996–97

Scientific Investigations Report 2004–5171

U.S. Department of the Interior
U.S. Geological Survey

Hydrology and Cycling of Nitrogen and Phosphorus in Little Bean Marsh: A Remnant Riparian Wetland along the Missouri River in Platte County, Missouri, 1996–97

By Dale W. Blevins

Prepared in cooperation with the
Missouri Department of Conservation

Scientific Investigations Report 2004-5171

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
Gale A. Norton, Secretary

U.S. Geological Survey
Charles G. Groat, Director

U.S. Geological Survey, Reston, Virginia: 2004
For sale by U.S. Geological Survey, Information Services
Box 25286, Denver Federal Center
Denver, CO 80225

For more information about the USGS and its products:
Telephone: 1-888-ASK-USGS
World Wide Web: http://www.usgs.gov/

Suggested citation:
Blevins, D.W., 2004, Hydrology and cycling of nitrogen and phosphorus in Little Bean Marsh: A remnant riparian wetland along the Missouri River in Platte County, Missouri, 1996–97: U.S. Geological Survey Scientific Investigations Report 2004-5171, 80 p.

CONTENTS

Figures

Tables

Conversion Factors and Datum

Multiply	By	To obtain
Length		
centimeter (cm)	0.3937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
Area		
hectare (ha)	2.471	acre
hectare (ha)	0.003861	square mile (mi^2)
Volume		
cubic meter (m^3)	264.2	gallon (gal)
Flow rate		
cubic meter per day (m^3/d)	35.31	cubic foot per day (ft^3/d)
cubic meter per day (m^3/d)	264.2	gallon per day (gal/d)
kilometer per hour (km/h)	0.6214	mile per hour (mi/hr)
Mass		
gram (g)	0.03527	ounce, avoirdupois (oz)
kilogram (kg)	2.205	pound avoirdupois (lb)
Density		
gram per cubic centimeter (g/cm^3)	62.4220	pound per cubic foot (lb/ft^3)
Hydraulic conductivity		
meter per day (m/d)	3.281	foot per day (ft/d)
Hydraulic gradient		
meter per kilometer (m/km)	5.27983	foot per mile (ft/mi)
Application rate		
kilograms per hectare per year [(kg/ha)/yr]	0.8921	pounds per acre per year [(lb/acre)/yr]

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:
°F = (1.8 x °C) + 32

Vertical coordinate information is referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29).

Horizontal coordinate information is referenced to the North American Datum of 1927 (NAD 27).

Altitude, as used in this report, refers to distance above the National Geodetic Vertical Datum of 1929 (NGVD 29).

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25°C).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L).

Hydrology and Cycling of Nitrogen and Phosphorus in Little Bean Marsh: A Remnant Riparian Wetland along the Missouri River in Platte County, Missouri, 1996–97

By Dale W. Blevins

Abstract

The lack of concurrent water-quality and hydrologic data on riparian wetlands in the Midwestern United States has resulted in a lack of knowledge about the water-quality functions that these wetlands provide. Therefore, Little Bean Marsh, a remnant riparian wetland along the Missouri River, was investigated in 1996 and 1997 primarily to determine the magnitude and character of selected water-quality benefits that can be produced in such a wetland and to identify critical processes that can be managed in remnant or restored riparian wetlands for amelioration of water quality.

Little Bean Marsh averages 69 hectares in size, has a maximum depth of about 1 meter, and the majority of the marsh is covered by macrophytes. In 1997, 41 percent of the water received by Little Bean Marsh was from direct precipitation, 14 percent was from ground-water seepage, 30 percent from watershed runoff, and 15 percent was backflow from Bean Lake. Although, Little Bean Marsh was both a ground-water recharge and discharge area, discharge to the marsh was three times the recharge to ground water. Ground-water levels closely tracked marsh water levels indicating a strong hydraulic connection between ground water and the marsh. Reduced surface runoff and ground-water availability are stabilizing influences on marsh hydrology and probably contribute to the persistence of emergent vegetation. The rapid hydraulic connection between Little Bean Marsh and ground water indicates that the hydrologic regime of most wetlands along the lower Missouri River is largely a function of the altitude of the marsh bottom relative to the altitude of the water table.

More water was lost from the marsh through evapotranspiration (59 percent) than all other pathways combined. This is partially because the transpiration process of abundant macrophytes can greatly contribute to the evapotranspiration above that lost from open water surfaces. Surface outflow accounted for 36 percent and ground-water seepage accounted for only 5 percent of the losses. Large residence times allows the marsh to greatly affect water quality before water escapes as ground-water recharge or surface outflow.

The shallowness of Little Bean Marsh and ion exclusion during ice formation caused the highest specific conductances of 1,100 to 1,300 microsiemens per centimeter at 25 degrees Celsius to occur during the winter. This concentration of dissolved solutes under ice can make wetlands more vulnerable to toxic contaminants than deeper surface-water bodies.

Dissolved oxygen was less than 5 mg/L (milligrams per liter) for 3 to 4 months and near 0 mg/L for about 1 month in summer. Despite depths of less than 1 meter, temperature stratification persisted more than 3 months during the summers of 1996 and 1997, preventing mixing and contributing to periods of anoxia. Shallow depths and extended periods of anoxia in the marsh limit the ability of some organisms to escape high-temperature stress.

Turbidity in Little Bean Marsh usually was low for several reasons: sediment loadings from the largely flood-plain drainage were low, emergent vegetation shade out algae and shield the water from wind, and high concentrations of bivalent cations increase flocculation rates of inorganic suspended material. The high concentrations of bivalent cations was largely because of a substantial amount of ground-water seepage into the marsh.

Dissolved organic nitrogen was the dominant nitrogen species in Little Bean Marsh. Denitrification and biotic uptake kept more than 62 percent of nitrate (NO_3) and 43 percent of ammonium (NH_4) concentrations in marsh samples less than a detection limit of 0.005 mg/L. This contrasts with the Missouri River where inorganic NO_3 dominates. Consequently, artificial flood-plain drainage that bypasses riparian wetlands likely deliver substantially more biotically available inorganic nitrogen to receiving waters than surface water that has been routed through a remnant wetland. Average total nitrogen concentrations in Little Bean Marsh were substantially less than those at other Missouri River wetlands, roughly one-half the mean concentrations in the Missouri River, but roughly twice the average nitrogen values in reservoirs of the glaciated plains of Missouri.

The largest concentrations of nearly all species of nitrogen and phosphorus and the most intense period of hypereutrophy coincided with a phytoplankton bloom and senescence of River Bulrush (*Scirpus fluviatilis*) and common cattail (*Typha latifolia*) in September 1997. The rapid leaching of nitrogen that

occurs soon after macrophyte senescence combined with a recent destratification of the marsh probably provided nitrogen to the nitrogen-limited open-water areas and triggered a phytoplankton bloom. Despite the rarity of runoff events, surface runoff from the watershed, combined with atmospheric deposition, contributed more than seven times the 530 kg (kilograms) of nitrogen that escaped Little Bean Marsh in surface outflow during 1997. Atmospheric deposition alone was more than 530 kg. Seepage to ground water contained less than 1.5 percent of the nitrogen leaving the marsh in surface outflow. The slow decay rate of *Scirpus fluviatilis* and reducing conditions in bottom sediments make burial of organic nitrogen a substantial sink of nitrogen.

Denitrification experiments indicate that denitrification rates were limited by NO_3 in the water column. Consequently, decomposition and nitrification of NH_4 and organic nitrogen are the rate limiting steps of nitrogen removal in Little Bean Marsh. The NO_3-limited rates of denitrification also indicate that Little Bean Marsh has a large unused capacity for nitrogen removal. These data indicate that the vast extent of riparian marshes along the Missouri and Mississippi Rivers may have had a substantial role in limiting NO_3 loads to the Gulf of Mexico before agricultural development of flood plains. Drainage and removal of riparian marshes may be a major cause of the increased NO_3 loads to the Gulf of Mexico.

Periods of anoxia had much larger effects on phosphorus release than the other variables. The largest concentrations of phosphorus occurred in late summer and corresponded with senescing macrophytes, periods of anoxia, and a large algal bloom in Little Bean Marsh. Low water levels prevented the escape of phosphorus in surface outflow during these periods of highest phosphorus concentrations. Dry weather in late summer is typical and probably makes the correspondence of low water levels, anoxia, and consequent low phosphorus release a common occurrence in marshes along the Missouri River. Little Bean Marsh retained more than 95 percent of the phosphorus it received. The amount of phosphorus in surface inflows to the marsh were more than one order of magnitude greater than that escaping in surface outflows. The long hydraulic residence time of the marsh and large contributions of iron from ground water (that provide many sorption sites for phosphorus) make the marsh an effective sediment and phosphorus trap.

Background and Problem

Little Bean Marsh is a riparian wetland located along the Missouri River in northwestern Missouri (fig. 1). Surface runoff flows through this remnant native wetland. More than one-half of Little Bean Marsh and some surrounding land [totaling 100 ha (hectares)] was purchased by the Missouri Department of Conservation in 1981, and 61 ha of this tract was designated as the Little Bean Marsh Natural Area. The marsh is characterized by shallow and fluctuating water levels and an abundance of aquatic and marsh plants.

Water-quality benefits commonly are ascribed to remnant native wetlands to justify their preservation or restoration. These benefits typically include removal or trapping of nitrogen (N), phosphorus (P), and sediment from surface waters; however, few limnological studies of wetlands in the Midwestern United States have been undertaken to document or quantify these benefits. Water-quality benefits, which have been demonstrated in a small number of coastal and northern wetlands (Mitsch and others, 1979; Kemp and Day, 1984), have been assumed to occur in Missouri and other parts of the Midwest. Also, wetland scientists and managers frequently have overstated the ability of wetlands to retain sediment and nutrients (Johnston, 1991). Wetland nutrient studies have a wide range of nutrient removal rates and processing characteristics depending on wetland characteristics—especially hydrology (Gosselink and Turner, 1978; Mitsch and Gosselink, 1993). Numerous studies have shown that some wetlands actually are net nutrient sources or nutrient transformers (Peverly, 1982; Elder, 1985; Baillie, 1995). Only three water-quality studies have been undertaken in wetlands along large rivers in Missouri, and none of these studies collected hydrologic data needed to explain many nutrient cycling phenomena. The first study was done in the Mississippi Embayment (Wylie and Jones, 1986) where the soils, vegetation, and hydrology are substantially different than those present elsewhere in Missouri. These factors (especially the altered hydrology) may substantially affect the water quality of wetlands. The second study (Knowlton and Jones, 1997) focused on nutrients in flood-plain water bodies created along the Missouri River by the 1993 flood. Although most of these water bodies were immature and too deep to be considered wetlands, four of these water bodies were remnant oxbows with monthly samples that are very comparable to Little Bean Marsh. However, few hydrologic data and no continuous data are available for these Missouri River oxbows. Hoyer and Reid (1982) analyzed samples from 10 seasonally flooded waterfowl refuges in December 1981, but the sites had neither natural hydrology nor vegetation, and the short sampling period limited comparisons with data collected in this study at Little Bean Marsh.

High N concentrations in many agricultural streams of the Midwestern United States (Goolsby and others, 1993) have highlighted the need for opportunities to reduce N loads in surface runoff before they are received by larger rivers and streams. The primary mechanisms for removal of N by wetlands are plant uptake, denitrification, and volatilization of ammonia (NH_3) from soil, plant, and water surfaces. Generally, physical and microbial processes are more important than vegetative uptake in controlling nutrient retention in wetlands (Johnston, 1991). Because most N uptake by plants is recycled annually in a mature wetland and NH_3 volatilization has not been demonstrated to be substantial, denitrification is considered the most important mechanism for N removal in remnant riparian wetlands (Brinson and others, 1984; Gale and others, 1993; Johnston, 1993; Hanson and others, 1994; Lowrance and others, 1995). Denitrification has been shown to remove substantial nitrate (NO_3) from overlying surface water (Lindau and others, 1994) as well as from sediments. Consequently, denitrification

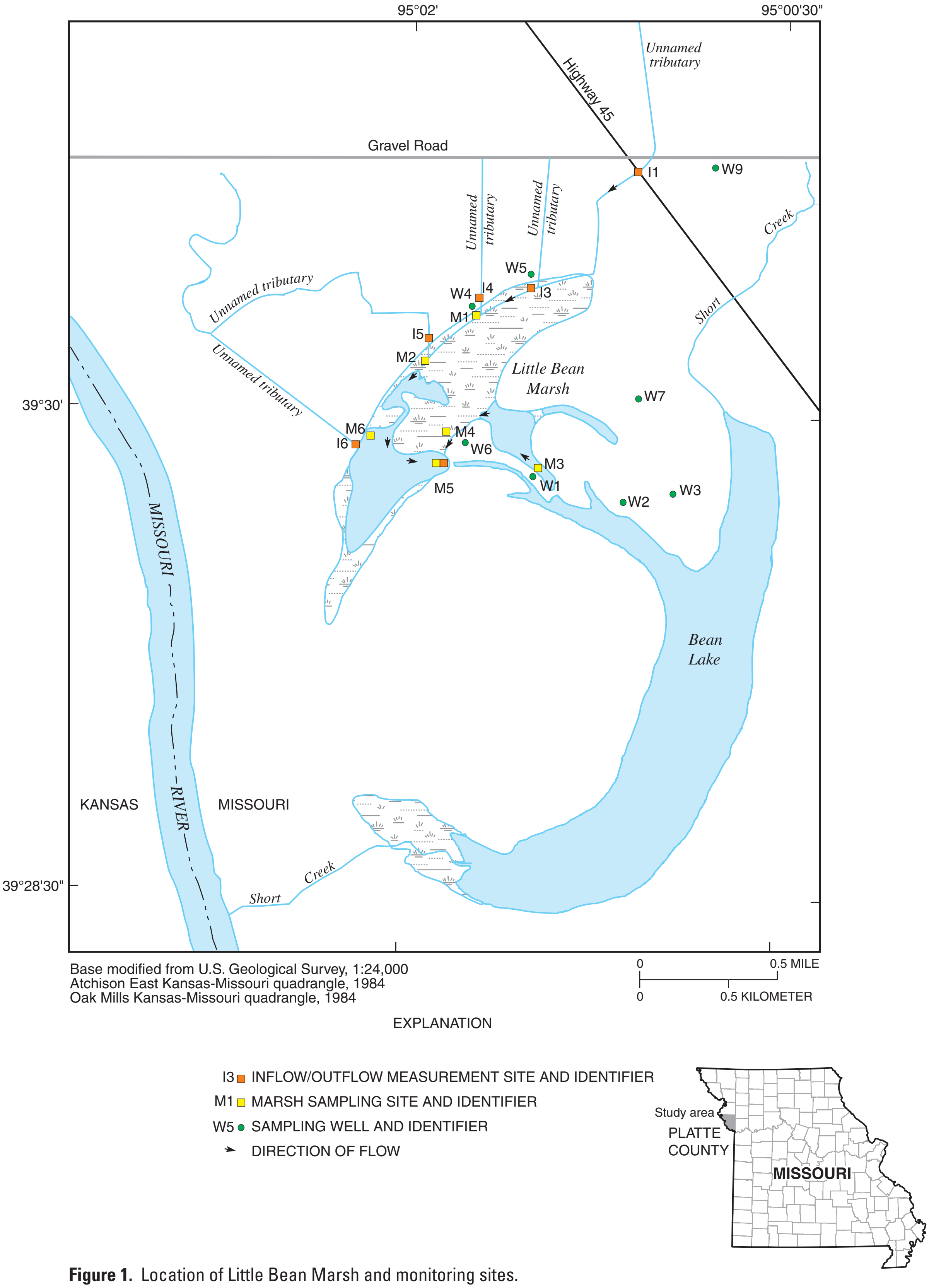


Figure 1. Location of Little Bean Marsh and monitoring sites.

rates usually are the primary mechanism of removal of N from surface water by wetlands. Denitrification rates can be limited by temperature, pH, redox potential, NO_3 availability, and carbon availability. Dissolved organic carbon (DOC) concentrations may be greatest soon after senescence of rushes and cattails (Klopatek, 1978) and winter temperatures may severely limit denitrification rates, causing seasonal patterns in concentrations of N species. Nitrate availability commonly is identified as limiting denitrification in native wetlands (Broderick and others, 1988; Cooper, 1990) because oxidizing conditions required for nitrification are sometimes limited (Zak and Grigal, 1991). Therefore, alternations of oxidizing and reducing conditions or the presence of an aerobic surface zone overlying anaerobic sediments, which are common in wetlands, are conducive to denitrification (Brinson and others, 1984; Patrick, 1991). Some wetlands just transform inorganic N to organic N (Elder, 1985) without substantially decreasing the total N export. This transformation also can be beneficial, however, because refractory organic N generally is not available for uptake by algae.

Phosphorus availability often limits primary productivity and trophic state in surface-water bodies. Therefore, the ability for wetlands to sequester particulate P by sedimentation and burial (Mitsch, 1978; Patrick, 1991) can be important to receiving streams. Particulate P is both sorbed on particulates and precipitated as iron and aluminum compounds (Cooke, 1992). However, large amounts of particulate P commonly are released to overlying water under various biogeochemical conditions common to wetlands and shallow lakes. Although release of sorbed P is most commonly associated with anoxic conditions that reduce and solubilize ferric iron compounds (Mortimer, 1941a, 1941b; Cooke and others, 1992), numerous biogeochemical conditions can cause a release of P from bottom sediments. In fact, aerobic release of P sometimes exceeds anaerobic release over an annual cycle (Bostrom and others, 1982). High temperatures, high pH, bioturbation, macrophytes, wind-induced resuspension of sediment, “P saturation” of the sediments, and carbonaceous sediments are all factors in aerobic processes or characteristics that can release P to overlying aerobic water. Sondergaard and others (1992) determined that wind induced resuspension of sediment in a shallow eutrophic lake released 20 to 30 times more P than seeped from undisturbed sediments. It also is possible for denitrification to at least indirectly affect P in two ways: removal of NO_3 allows ferric iron (Fe^{+3}) to serve as an electron acceptor with subsequent P release, and high rates of denitrification can increase rates of mineralization of organic matter that can release P to the water column (Bostrom and others, 1988). P-mobilization processes in surface sediments commonly result only in a change in mineralogical species and not a release to the water column. For example, P mobilized by mineralization of sedimented organic matter may be rapidly adsorbed by iron oxyhydroxides under aerobic conditions (Bostrom and others, 1988). Nearly every process that mobilizes P in the sediment (both aerobic and anoxic) and every mechanism that transports P to overlying water is maximized in wetlands compared to deeper lakes (Bostrom and others, 1982; Baccini, 1985). Therefore, wetlands may not be efficient removers of P (as reported by Peverly, 1982) and relations between P concentrations and temperature, wind, invertebrate density, macrophyte populations, oxidation/reduction potential (Eh), pH, and other characteristics have implications for wetland restoration and management of water quality.

To address the need for water-quality and hydrologic data to gain knowledge about nutrient cycling in remnant riparian wetlands, the U.S. Geological Survey (USGS), in cooperation with the Missouri Department of Conservation, conducted a study of Little Bean Marsh in 1996–97.

Purpose and Scope

The purpose of this report is to describe results of a study of hydrology and cycling of nitrogen and phosphorus in Little Bean Marsh. The general goals of this study were to determine the magnitude and character of selected water-quality benefits that may occur in a remnant riparian wetland along the Missouri River and to identify critical processes such as denitrification and P release that can be managed in remnant or restored riparian wetlands for water-quality improvement. Specific objectives to achieve these goals were:

1. Quantify the sources and losses of water to a remnant riparian wetland and determine the affects of hydrologic variation on water quality.
2. Characterize concentrations and speciation of N and P in a remnant riparian wetland to identify the role of the wetland as a sink, source, or transformer of N and P for 1 year.
3. Identify the factors limiting denitrification and determine whether or not denitrification rates are large enough to remove most of the N received by Little Bean Marsh.
4. Determine whether sedimentation within the wetland results in a net removal of P from runoff and atmospheric deposition by the wetland.
5. Relate P concentrations in water to various biogeochemical conditions such as temperature, pH, Eh, dissolved oxygen, macrophyte life cycles, wind, and external loadings to identify the primary mechanisms of P release and the consequent ability of the wetland to retain P.
6. Compare N and P concentrations and speciation in a remnant riparian wetland with similar data from other wetland studies.

Hydrologic and water-quality data used in this report were collected from November 1996 through December 1997. Sampling and measurement locations included four stream sites, six marsh sites, eight wells, and six denitrification sites. Continuous water-level, water-temperature, dissolved-oxygen, specific conductance, and pH data were collected from January 1996 through December 1997 at one site.

Study Area Description

The drainage area of Little Bean Marsh is 952 ha. About 80 percent of this drainage area is bottomland in row-crop agriculture, 5 percent is the bordering loess bluffs, about 7 percent (69 ha) is the marsh, and the remaining 8 percent is uncultivated flood plain. The surrounding terrain is flat with low ridges and a large terrace forms the north and west boundary of the marsh. Discharge from Little Bean Marsh flows directly into Bean Lake to the south. The area is protected from small Missouri River floods by levees, but large floods in 1952, 1973, 1983, and 1993 breached levees so the marsh is not entirely isolated from flooding disturbances, which can prevent an ecosystem from forming a stable community. Bean Lake and Little Bean Marsh also are occasionally flooded by local runoff. The outflow from Bean Lake is controlled by a flood gate in the Missouri River levee. When the gate is closed during high river stage, water can back all the way into Little Bean Marsh if large amounts of local runoff enter Bean Lake. Stage and discharge from Little Bean Marsh usually are controlled by a mud riffle between Bean Lake and Little Bean Marsh.

The average annual air temperature at Little Bean Marsh is 12.0 °C (degrees Celsius) with average monthly temperatures ranging from -3.5 °C in January to 24.5 °C in July (National Oceanic and Atmospheric Administration, 1997) (fig. 2). The largest departure from the 30-year mean monthly temperature was -3.2 °C in April, and the mean annual departure was -0.2 °C. Average annual rainfall at Kansas City International Airport (KCI) is 95.5 cm/yr (centimeters per year), with the greatest amount occurring in the summer [30.5 cm (centimeters) from June through August], and the least in the winter (10.1 cm from December through February) (Missouri Department of Natural Resources, 1986). Annual evaporation averages 152 cm/yr. During 1997, the total precipitation was 98.7 cm with 25.6 cm from June through August and 17.2 cm during January, February, and December. The 30-year mean monthly precipitation values at KCI and monthly precipitation at Little Bean Marsh are shown in figure 3. Although the summer was a little drier than average, the winter was wetter than average, and the spring was a little cooler than average, 1997 was not an unusually wet, dry, cool, or warm year.

In 1997 about 42 ha (60 percent) of Little Bean Marsh was dominated by River Bulrush (*Scirpus fluviatilis*), common cattail (*Typha latifolia*), smartweed (*Polygonum coccineum*), and arrowhead (*Sagittaria brevirostra*). There also were about 28 ha of bottomland hardwood forest on areas of higher ground. About 40 percent of the marsh is open water with a maximum depth of about 1 m (meter) at typical water levels. A detailed description of the vegetation and natural history of Little Bean Marsh since 1850 is in Castaner and LaPlante (1992) (fig. 4).

Methods and Materials

Concurrent hydrologic and hydrochemical data were collected from November 1996 through December 1997 at six marsh sites (M1 to M6, fig. 1). Continuous water-level, water-temperature, dissolved-oxygen, specific-conductance, and pH data were collected for 2 years from January 1996 through December 1997 at site M3. Miscellaneous data were collected before and after these dates in support of several experiments. Sampling, measuring, and experiment sites were located along two flow paths to investigate changes in nutrient concentrations and speciation as water passed through the marsh. The first flow path begins with inflow sites I1 and I3 (fig. 1) at the northeast end of the marsh and continues through marsh sites M1, M2, and M5. The second flow path passes through marsh sites M3, M4, and M5. Sites M2, M3, and M5 were sampled 26 times from November 1996 through December 1997. Vegetated sites

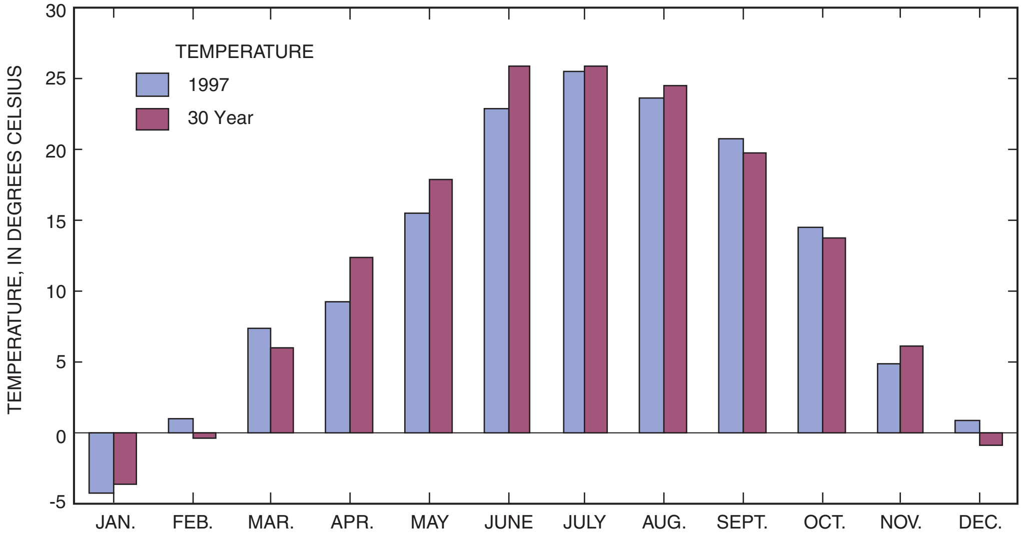


Figure 2. Mean monthly air temperatures in 1997 compared to 30-year (1968–97) means recorded at the Kansas City International Airport.

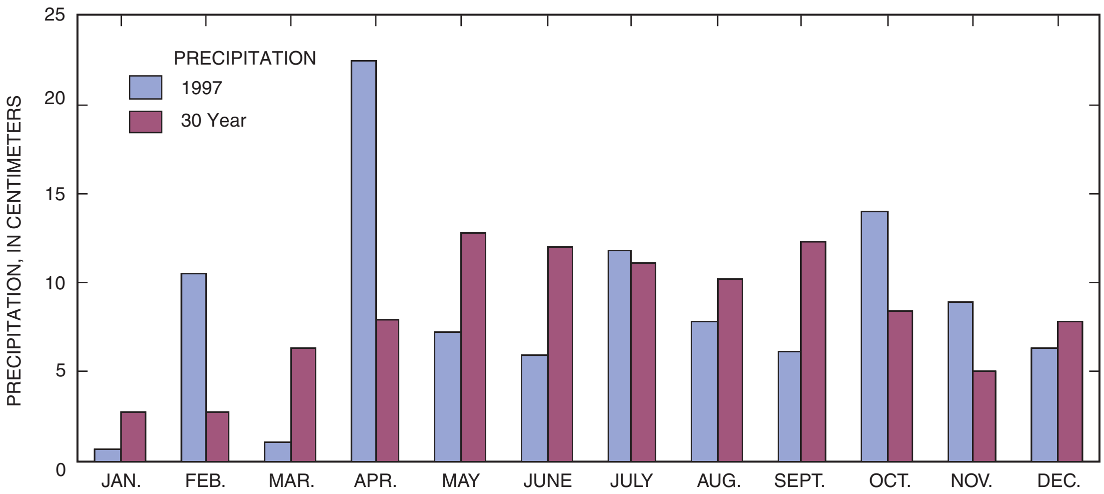


Figure 3. Monthly precipitation measured at Little Bean Marsh in 1997 compared to 30-year (1968–97) means recorded at the Kansas City International Airport.

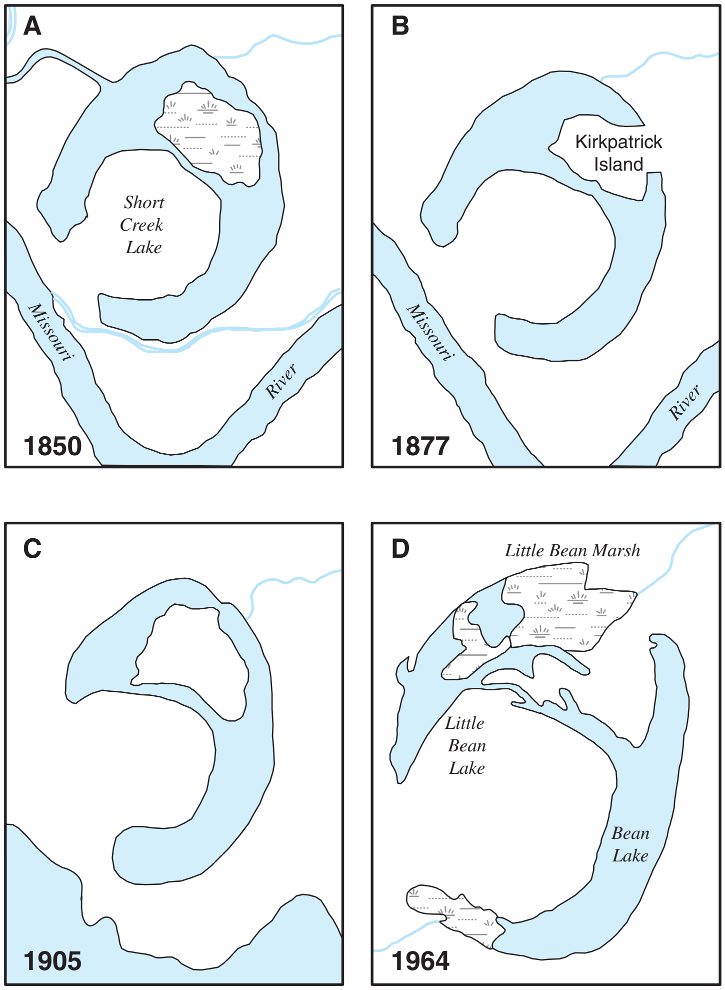


Figure 4. Historical evolution of Little Bean Marsh (Castaner and LaPlante, 1992).

M1 and M4 were sampled at 2-week intervals from April through December 1997 when standing water was present. Open-water site M6 was sampled six times from September through December 1997 when site M1 was dry and was located along the northern flow path between sites M2 and M5. The southern end of Little Bean Marsh receives little surface runoff and mostly is an open-water area that includes outflow site M5.

Marsh water samples were synoptically collected twice per month at three to six locations (sites M1 to M6 depending on the area inundated) in high-density polyethylene bottles. A few additional samples were collected within a few days after floods inundated the marsh. When water depths were sufficient, the bottles were raised and lowered through the water column at an equal transit rate during filling to make sure the sample was representative of the entire depth. Samples were refrigerated near 4 °C immediately after sample collection. The day after sample collection, samples were poured into a churn splitter and subsamples of suspended solids, turbidity, and total N, P, and DOC were split into separate bottles for laboratory analysis. The remaining water was then filtered through 0.45-micron polycarbonate filters into separate bottles for laboratory analysis of dissolved N and P species. All laboratory analysis were done at the University of Missouri Limnology Laboratory in Columbia, Missouri. Mean concentrations of N and P in the marsh samples were multiplied times the volume of water in the marsh to compute the change in mass stored in the marsh between sample events.

Laboratory analyses of water samples for total N were done using a persulfate digestion as described by Crumpton and others (1992), for dissolved orthophosphate (PO_4) using a colorimetric method and NO_3 and nitrite (NO_2) using a cadmium-reduction method as described in U.S. Environmental Protection Agency (1979), for ammonia nitrogen using a phenate method as described in Fisheries and Environment Canada (1977), for total and dissolved P using a persulfate reduction as described in Prepas and Rigler (1982), for suspended chlorophyll using a flourometric determination described in Knowlton (1984) and Sartory and Grobbelaar (1984), for total and volatile suspended solids using methods described in American Public Health Association (1980), and for DOC using a wet persulfate oxidation described in Crowther and Evans (1978). Suspended ammonium N was calculated by subtracting dissolved from total ammonium N. A single analysis for calcium and magnesium was performed with atomic adsorption spectrometry, for sodium and potassium with flame emission spectroscopy, and for alkalinity by titration according to methods described in American Public Health Association (1980). Laboratory measurements of turbidity and color were made with a colorimeter using methods described by Hach Company (1987).

Field measurements of water temperature, specific conductance, and dissolved oxygen were made following methods given in Wildke and others (1998). Measurements of water temperature and dissolved oxygen were made *in situ* and measurements of specific conductance and pH of water samples were made in the field at the time of sampling. Dissolved oxygen was measured using an electronic meter that had been calibrated in an air chamber saturated with water vapor near the ambient temperature of the marsh. Temperature was measured using a thermistor attached to the dissolved-oxygen meter. The thermistor occasionally was checked against a mercury thermometer. Specific conductance and pH were measured with electronic meters that were calibrated with standard solutions that bracketed measured values.

The significance of correlations between site characteristics and various water samples was determined statistically. Tukey's multiple comparison tests (Tukey, 1977), Wilcoxon rank-sum tests (Wilcoxon, 1945), and Kruskal-Wallis tests are nonparametric statistical tests that were used to analyze the spatial variability of water chemistry (Ott, 1993). Correlation and significance testing also was performed between various water-sample constituents. In this report, "significant" denotes that the p-value of a comparative statistical test is less than 0.05 or that there is a 95 percent chance that statistical relations were not false.

A bathymetric survey (fig. 5) was conducted (Heimann and Femmer, 1998) to calculate temporal changes in surface area and volume for hydrologic and nutrient budgets and to determine hydraulic residence time, which is defined as the average time that water remains in an impoundment, and is determined using the following equation (modified from Mitsch and Gosselink, 1993):

$$Rt = S/(DaR) \tag{1}$$

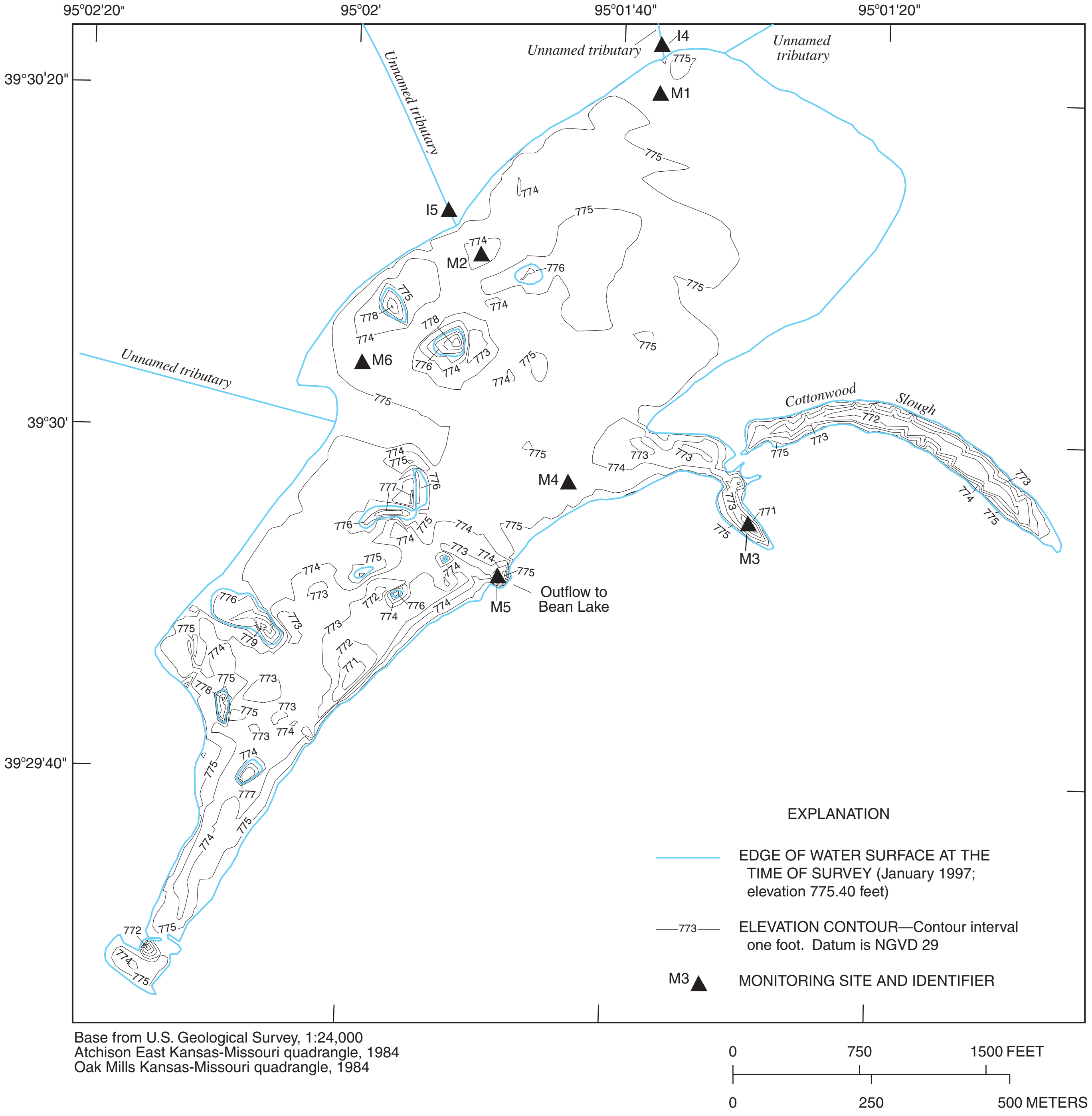
where

Rt = hydraulic residence time, in years;
S = storage volume, in cubic meters;
Da = the drainage area of the watershed, in square meters; and
R = the runoff from the local watershed, in depth (meters) per year.

Approximately 1,000 points were surveyed along 20 transects through the marsh. The bathymetric survey was tied to the water-level gage so that surface area and volume could be determined for any gage height.

Marsh water levels were measured at site M3 with a pressure transducer and recorded to the nearest 0.3 cm on a data logger equipped with a real-time satellite radio transmitter. The pressure transducer was set and calibrated to an outside staff gage anchored into the marsh bottom with concrete. Altitudes were surveyed into the gage using standard surveying techniques from a U.S. Army Corps of Engineers' reference mark located on the Missouri River levee.

Inflow site I1 (fig. 1) was located on the upstream side of a concrete box culvert with a concrete bottom. Stage was measured with a pressure transducer to the nearest 0.6 cm and set to an outside staff gage mounted on the culvert wing wall. Discharge measurements were made over a range of gage heights to obtain a rating curve (Rantz and others, 1982). The rating curve was programmed into an automatic sampler, which was connected to a pressure transducer. The autosampler was pro-



Base from U.S. Geological Survey, 1:24,000
Atchison East Kansas-Missouri quadrangle, 1984
Oak Mills Kansas-Missouri quadrangle, 1984

Figure 5. Bathymetry of Little Bean Marsh (modified from Heimann and Femmer, 1998).

grammed in such a way as to sample after every 1,130 m^3 (cubic meters) of runoff passed the gage.

Discharge at the outflow site M5 (fig. 1) was computed using discharge measurements and marsh water levels to prepare a rating curve. However, the frequent presence of beaver dams on the outflow control point made rating curves unusable for some periods. During these periods, outflow was estimated with water-budget calculations made from marsh water-level data, inflow data, and estimates of evapotranspiration and ground-water seepage. These estimates were checked and constrained with frequent discharge measurements of the outflow. Fortunately, daily discharge fluctuations usually were a small percentage of average daily streamflow, which minimized the error induced by these estimates. Long periods of no outflow were documented.

Annual mass balance data were used to determine the masses of N or P removed, added, or transformed by the marsh and for comparison of volumetric and areal removal rates with those of other wetlands. Wet atmospheric deposition was estimated with on-site precipitation measurements, and concentrations of NO_3 and NH_4 were collected and published by the National Atmospheric Deposition Program (1997) at two sites approximately 120 km (kilometers) southwest and northeast of Little Bean Marsh. Dry atmospheric deposition was estimated from data collected as a part of the U.S. Environmental Protection Agency's Castnet network at Alhambra, Illinois (http://www.epa.gov/castnet/data.html, accessed 2002). Wet deposition of P was estimated from data collected at Centralia, Missouri, by the Agricultural Research Service (written commun., 1997).

Inflow from the easternmost tributary to Little Bean Marsh were measured and sampled at site I3 (fig. 1) when wetland water levels were low (usually base-flow conditions) and at site I1 when marsh water levels were high (usually during runoff events). About 70 percent of the drainage area of the tributary at site I3 is bottomland in row-crop agriculture and 30 percent loess uplands in mixed agriculture and forest. The land use of the other tributaries is almost 100 percent bottomland agriculture. Scant runoff was observed from these tributaries after rains as large as 3.8 and 7.6 cm. Substantial runoff was observed in these bottomland tributaries only after the 14.8-cm rain on April 10 and 11, 1997. Therefore, automatic samples and continuous water-level and discharge data were collected only at inflow site I1. Data collected at I1, changes in marsh concentrations of N and P (sites M1 to M6), and marsh volume measured immediately before and after rainfall events were used to compute the mass of N and P received by the marsh in runoff. Inflow sites received some ground-water seepage (base flow) when ground-water levels were high. Therefore, all inflow sites were measured and sampled twice per month (when flow was present). Sites I3, I4, I5, and I6 also were sampled during the largest runoff events. Water-budget data also were computed using changes in marsh volume and specific conductance values, which were diluted by surface runoff [between 50 to 200 µS/cm (microsiemens per centimeter at 25 degrees Celsius)] and increased by ground-water seepage (800 to 1,000 µS/cm) (equation 2):

$$Mc = Gc \times \frac{Gv}{Mv} + Sc \times \frac{Sv}{Mv} \qquad (2)$$

where

Mc = specific conductance of marsh, in microsiemens per centimeter at 25 degrees Celsius;
Gc = specific conductance of ground water, in microsiemens per centimeter at 25 degrees Celsius;
Gv = ground water volume, in cubic meters;
Mv = marsh volume, in cubic meters;
Sc = specific conductance of surface water, in microsiemens per centimeter at 25 degrees Celsius;
Sv = surface water volume, in cubic meters; and
$Gv + Sv$ = Mv.

The primary process removing N from most wetlands is denitrification (Johnston, 1993), which seldom occurs below 8 cm deep in wetland sediments (Broderick and others, 1988). Measurements of denitrification rates were made using *in situ* acetylene-block methods every other month from April through November when temperatures were high enough to support denitrification. *In situ* acetylene-block experiments are considered the most accurate way to measure field denitrification rates. Acetylene causes denitrified N to accumulate as nitrous oxide (N_2O) [instead of nitrogen gas (N_2)] that can be detected and quantified with a gas chromatograph (Yoshinari and others, 1977; Ryden and others, 1979a, 1979b). This method consisted of setting a capped 15-cm diameter Polyvinylchloride (PVC) pipe 10 cm into the bottom sediments leaving 20 cm of water in the enclosure (fig. 6). The overlying water inside the enclosure was then saturated with acetylene (modified from Ryden and others, 1979b) by bubbling acetylene through the overlying water through needle tubes that extended down through the top of the enclosure. Water samples were collected in syringes through the needle tubes at 24-hour intervals for N_2O analysis. These experiments lasted 48 to 72 hours. Measurements were made at 3 locations in the open water and three locations in the macrophytes at sites corresponding to sampling sites M1 to M6 (fig. 1).

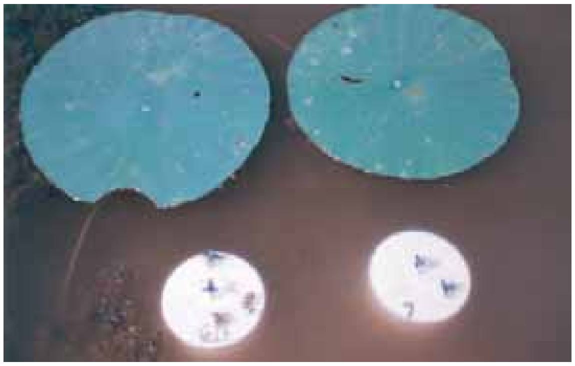


Figure 6. Denitrification chambers in Little Bean Marsh.

In situ acetylene-block experiments were used to determine whether NO_3 concentrations in the water column limit denitrification rates by artificially adding NO_3 to the overlying water of a second chamber. If more N_2O were to be produced in the spiked chamber than the unspiked chamber, then denitrification was NO_3-limited. Wetlands with NO_3-limited denitrification rates have unused N-removal capacity. Because NO_2 is typically very small compared to NO_3 in surface waters, NO_2+NO_3-N will be referred to as NO_3 in this report.

Eight wells (W1 to W7 and W9) were hand augered to 3 m deep; screened with 1.6 m of slotted, 5-cm diameter, PVC pipe, and cased with solid PVC pipe for a separate study by Walter (1998). Altitudes of measuring points on wells were determined by running levels from the staff gage on Little Bean Marsh using a transit, rod, and conventional surveying methods. Ground-water levels were measured at 2 to 4 week intervals from September 1996 through December 1997 with an electric tape to the nearest 0.3 cm. Daily ground-water levels were estimated for this report by straight-line interpolation between measurements. Darcy's Law for vertical flow (equation 3) was used to determine ground-water seepage into and out of the marsh (Darcy, 1856; Freeze and Cherry, 1979; Carter, 1986):

$$Q = KA(dh/dl) \tag{3}$$

where

Q = flow through the bottom sediments, in volume per unit time (can be positive or negative);
K = hydraulic conductivity of the bottom sediments, in length per unit time;
A = inundated area of bottom sediments, in length squared;
dh = the head difference between the marsh and the underlying water table, in length (can be positive or negative); and
dl = the distance between the marsh bottom and the underlying water table.

Area of inundation was determined from the bathymetric survey for every daily marsh water level. The term dh was determined by subtracting the average ground-water altitude in wells W1, W4, W5, and W6 from the altitude of water in the marsh. The term dl was the distance between the average ground-water altitude and the average altitude of the marsh bottom determined for every stage. A value of K was selected that minimized error in the annual marsh water budget.

In the Walter (1998) study, ground-water samples were collected from 8 monitoring wells (W1 to W7 and W9, fig. 1) every other month from September 1996 to April 1997, and analyzed at the University of Missouri at Kansas City for dissolved NO_3+NO_2-N, P, magnesium, calcium, sodium, potassium, sulfate, chloride, bicarbonate, aluminum, iron, manganese, nickel, lead, and zinc. For this report monthly samples collected by USGS personnel after April 1997 through December 1997 at four of the wells (W1, W4, W5, and W6) were analyzed for dissolved total N, dissolved ammonium (NH_3+NH_4-N), NO_2+NO_3-N, PO_4, and total P (table 1) at the University of Missouri Limnology Laboratory in Columbia, Missouri. The mass of N and P in ground-water seepage was then calculated by multiplying the seepage volume times the average N or P concentration of ground water if flow is into the marsh, or the negative seepage volume times N and P concentrations of the marsh if seepage is out of the marsh as shown in equation 4:

$$Mnp = Cnp \times S \tag{4}$$

where

Mnp = mass of nitrogen or phosphorus entering or leaving the marsh through ground water, in milligrams;
Cnp = concentration of nitrogen or phosphorus in the marsh, in milligrams per liter; and
S = the volume of seepage entering or leaving the marsh into or from ground water, in liters.

The Eh and pH of bottom sediments were measured at each of the marsh-sampling sites (sites M1 to M6) by collecting a core in a 2.54-cm diameter soil-sampling tube. Eh and pH probes were inserted into the bottom end of the core while it was still in the tube and the meter readings were allowed to reach temporal stability before being recorded. Sampling depth intervals were shorter near the sediment-water interface and increase away from the sediment-water interface, and ranged from 0.5 to 15 cm. Visual observations of the color, wetness, and organic material in the core were recorded as each section of the core was extruded from the tube. The Eh meter was calibrated with standard Zobell solution before each series of measurements and checked against Zobell solution again at the end of the measurements to check for drift. The pH meter was calibrated with pH 4, 7, and 10 buffer solutions according to methods described in the USGS National Field Manual for the collection of water-quality data (Wildke and others, 1998).

Hourly measurements of temperature, dissolved oxygen, specific conductance, and pH were made with a multiparameter water-quality monitor at a single point in the marsh. This point was at site M3 in open water in one of the deepest parts of the marsh so that measurements were possible at low water levels. The hourly data were stored in a data logger. A mechanical stirrer operated during a 3-minute warm-up period to ensure proper water flow across the dissolved-oxygen membrane. The hourly circulation also minimized biofouling of the sensors. The monitor was protected in a 15-cm diameter PVC pipe with numerous holes to provide for water exchange. Two horizontal floats were attached to the side of the vertical PVC pipe to allow the monitor sensors to float at a fixed depth of 0.5 m below the surface. The monitor was checked against field meters, cleaned, and calibrated with standard solutions at 2-week intervals to determine if corrections were necessary and to recalibrate the monitor.

Precipitation was collected at site M3 with a tipping bucket rain gage at hourly intervals and recorded on the data logger. Precipitation data collected by the National Weather Service (NWS) at the KCI (National Atmospheric and Oceanic Administration, 1996 and 1997) were used to verify the data collected on site and to estimate short periods of missing daily rainfall. KCI is 34 km southeast of Little Bean Marsh. Daily average wind-speed data, also collected at KCI, were obtained from the NWS for correlation with P concentrations.

Evapotranspiration from November 1996 through April 1997 was estimated using climatological data collected at DeSoto, Kansas; St. Joseph, Missouri; and Powhatten, Kansas (National Oceanic and Atmospheric Administration, 1996 and 1997) and the Penman-Montieth method (Penman, 1948; Montieth, 1965) of estimating potential evapotranspiration. The Penman-Montieth method has been successfully used in wetlands where water tables are high and abundant macrophytes keep evapotranspiration near potential values (Souch and others, 1998). Potential evaporation estimates from all three sites were averaged to produce estimates at Little Bean Marsh. Penman-Montieth evapotranspiration values are calculated from solar radiation, wind, humidity, and air temperature measurements using the Penman equation (equation 5). Pan evaporation data collected at Smithville Reservoir (39 km east of Little Bean Marsh) by the National Oceanic and Atmospheric Administration (1996 and 1997) were used from May through October along with a “pan coefficient” to estimate evapotranspiration

Table 1. Specific conductance and concentrations of selected nutrients in ground-water samples collected from monitoring wells near Little Bean Marsh.

[N, nitrogen; mg/L, milligrams per liter; P, phosphorus; µS/cm, microsiemens per centimeter at 25 degrees Celsius; --, no data; <, less than]

Well number (fig. 1)	Date	Total dissolved N, in mg/L	Nitrite plus nitrate as N, in mg/L	Dissolved ammonia N as N, in mg/L	Dissolved P, in mg/L	Soluable reactive P, in mg/L	Specific conductance, in µS/cm
W1	01/20/1997	--	<0.02	--	<0.02	--	807
W1	02/17/1997	--	<.005	--	.006	--	768
W1	02/27/1997	--	--	--	--	--	584
W1	03/09/1997	--	--	--	--	--	671
W1	04/13/1997	--	--	--	--	--	482
W1	04/25/1997	0.25	<.005	<0.005	.013	0.007	--
W1	08/12/1997	.68	<.005	.065	.148	.140	--
W1	12/02/1997	.33	<.005	.042	.019	.010	796
W4	01/20/1997	--	<.02	--	<.02	--	645
W4	02/17/1997	--	<.005	--	.002	--	779
W4	02/27/1997	--	--	--	--	--	639
W4	03/09/1997	--	--	--	--	--	625
W4	04/13/1997	--	--	--	--	--	639
W4	04/25/1997	--	<.005	.049	.285	.286	--
W4	08/12/1997	--	<.005	.127	.760	.816	--
W4	12/02/1997	--	.010	.142	.874	.866	--
W5	01/20/1997	--	.04	--	.04	--	763
W5	02/17/1997	--	.010	--	.012	--	728
W5	02/17/1997	--	--	--	--	--	739
W5	03/09/1997	--	--	--	--	--	709
W5	04/13/1997	--	--	--	--	--	720
W5	04/25/1997	.19	<.005	<.005	.024	.025	--
W5	08/12/1997	.90	<.005	.098	.124	.125	--
W5	12/02/1997	.28	.024	.052	.025	.024	--
W6	01/20/1997	--	<.02	--	.03	--	775
W6	02/17/1997	--	<.005	--	.012	--	731
W6	02/27/1997	--	--	--	--	--	755
W6	03/09/1997	--	--	--	--	--	749
W6	04/13/1997	--	--	--	--	--	794
W6	04/25/1997	.10	.019	<.005	.020	.019	--
W6	08/12/1997	.07	<.005	.013	.014	.022	--
W6	12/02/1997	.14	<.005	.025	.060	.055	761
W9	01/20/1997	--	<.02	--	.17	--	1,112
W9	02/17/1997	--	--	--	--	--	936
W9	02/27/1997	--	--	--	--	--	1,068
W9	03/09/1997	--	--	--	--	--	1,034
W9	04/13/1997	--	--	--	--	--	1,017

from May through October. A coefficient of 1.0 was determined by selecting a value that gave the best match with monthly Penman-Montieth estimates:

$$Q_e = \frac{s(Q^{*}\text{-}\hat{}Q_{\underline{s}}) + @_{\underline{a}}C_{\underline{p}}D/R_a}{s+g(1 + R_s/R_a)} \tag{5}$$

where

Q_e = the latent heat flux of evaporation, in megajoules per square meter per day;
s = the slope of the saturation vapor pressure and temperature curve, in millimeters of mercury per degree Celsius;
Q^{*} = the net all wave radiation, in megajoules per square meter per day;
$Q_{\underline{s}}$ = the net storage heat flux, in megajoules per square meter per day;
$@_{\underline{a}}$ = the density of air, in kilograms per cubic meter;
$C_{\underline{p}}$ = the specific heat of moist air, in joules per kilogram;
D = the vapor pressure deficit, in millimeters of mercury;
R_a = aerodynamic resistance, in seconds per meter;
R_s = the surface resistance, in seconds per meter; and
g = the density of water, in kilograms per cubic meter.

Stages on the Missouri River near Little Bean Marsh were estimated using continuous stage data collected by the USGS at St. Joseph, Missouri, and 59 daily river-stage readings at Atchison, Kansas, as measured by a U.S. Army Corps of Engineers observer and recorded by the National Weather Service River Forecast Center in Pleasant Hill, Missouri (written commun., 2000). The Atchison data were collected between October 6, 1996, and July 4, 1997. Concurrent stage readings at the two sites were used to develop average altitude drops and river slopes between St. Joseph and Atchison. The altitude drop averaged 7.11 m with a range of 6.77 to 7.43 and the slope averaged 0.173 m/km (meter per kilometer) and ranged from 0.165 to 0.180. The average altitude drop was subtracted from the continuous stage data at St. Joseph to estimate stage at Atchison for days where Atchison stage data were unavailable. The average river slope was then used to extrapolate stage data 9.7 km downstream from Atchison to Little Bean Marsh. The ranges of altitude drop and river slope indicate stage estimates at Little Bean Marsh usually were within 1 foot of actual stages. Given the large fluctuations in river stage, this error was considered acceptable for use in comparing stage with ground-water and marsh levels in the vicinity of the marsh.

Hydrology

Although hydrologic characteristics such as source of water, hydraulic residence time, and duration and timing of inundation are the primary determinants of many wetland functions (Moustafa, 1997) and are necessary for improved understanding of wetland limnology and ecology, hydrologic measurements are rarely made in wetland studies (Kibby, 1978; Carter, 1986; Mitsch and Gosselink, 1993). Therefore, a hydrologic analysis of Little Bean Marsh was made to better measure nutrient cycling processes and to quantify inputs and outputs of nutrients to the marsh.

Hydrologic Inputs

In the absence of active management at Little Bean Marsh, marsh vegetation has been fairly stable (Castaner and LaPlante, 1992) since acquisition by the Missouri Department of Conservation in 1981. Because emergent vegetation responds fairly quickly to changes in inundation frequency, duration, and water level (Carter, 1986), hydrologic conditions probably have been fairly stable as well. In 1997, 41 percent of the water received by Little Bean Marsh was direct precipitation and 14 percent was ground-water seepage (fig. 7). Surface-water inflow constituted 45 percent of the marsh inputs, although about one-third of surface inflow (15 percent of marsh inputs) were backflows from Bean Lake. Nearly all the backflow occurred after two large rainstorms in April and May when the outflow from Bean Lake was blocked by closed flood gates in the Missouri River levee. The relatively high percentage of direct precipitation is because of the low ratio of drainage area to water-surface area (12:1). Like many flood-plain water bodies, more than 90 percent of the marsh's drainage area is flood plain. The nearly flat slopes, sandy soils, and large shallow depressions in the flood plain keep runoff coefficients low. No runoff was observed from several storms that had more than 3.8 cm of rain; consequently, surface-water contributions from the watershed were small compared to upland watersheds. The tributary that most frequently produced runoff was one that contained a small percentage of upland soils (site I1, fig. 1).

Minimal surface runoff and a substantial hydraulic connection with ground water tend to stabilize water levels in the marsh. Ditches created for agricultural drainage cause runoff from the loess hills to the east to bypass Little Bean Marsh, and thus bypass its potential for flood attenuation and nutrient removal. Flood-plain ditches and river levees have converted Little Bean Marsh from an open to a closed system. This conversion is not an unusual occurrence in the Missouri River flood plain as landowners have drained valuable agricultural bottomlands by routing runoff from the uplands straight across the flood plain into the river. Consequently, the hydrology, water quality, and ecological functions of remnant wetlands located along today's Missouri River usually are vastly different from the predevelopment condition. Open wetlands often are large sinks for sediment and nutrients, whereas closed wetlands primarily rely on precipitation for nutrients (Craft and Casey, 2000).

The large percentage of precipitation as an inflow source has diluting effects on water quality, as indicated by low specific conductance values and concentrations of nutrients immediately after large rainstorms and during periods of low groundwater levels (fig. 8). If precipitation did not make up a large proportion of the inflow, concentrations of most constituents would

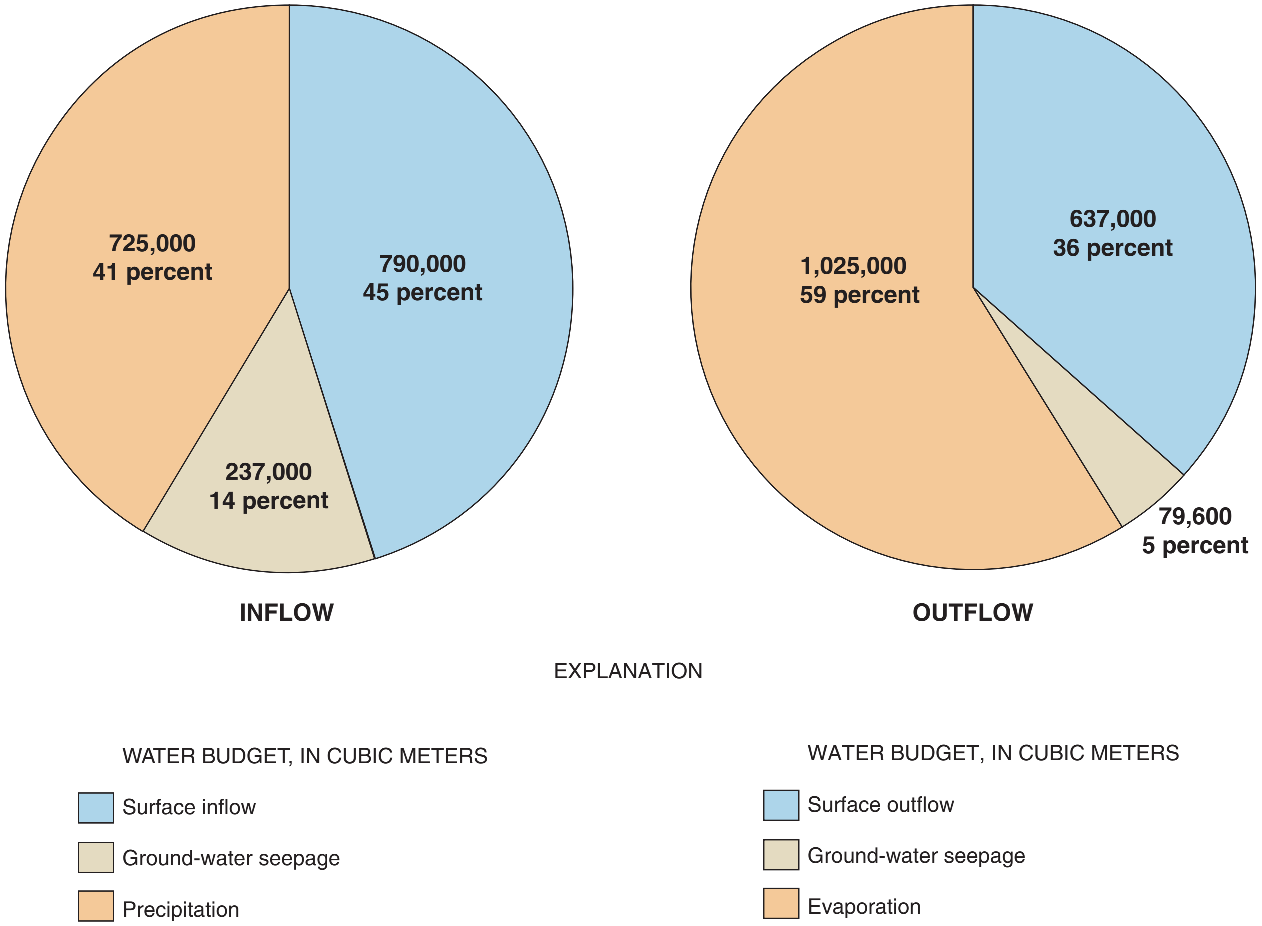


Figure 7. Inflow and outflow water budgets for Little Bean Marsh during 1997.

have been greater, especially during periods of intense evapotranspiration and ice formation.

Ground-Water Interactions

Although the bottom of Little Bean Marsh is composed of fine organic material mixed with sand, underlying sands and gravels have large hydraulic conductivities that typically range from 126 to 326 m/d (meters per day) (Emmett and Jeffery, 1969; Kelly and Blevins, 1995). Also, Little Bean Marsh is incised 3 to 4 m below the surrounding flood plain, which allows rapid hydraulic communication between the marsh and ground water. The bottom of Little Bean Marsh is only 1 to 2 m above the Missouri River stage at median discharge (Missouri River Bottoms Work Group, written commun., 1979). The marsh bottom was between 1.0 m below and 0.7 m above the median stage of the river in 1997 and the water surface in the marsh averaged only 0.5 m above the river stage. Higher-than-normal river stage in 1997 (Hauck and others, 1998) probably increased ground-water levels above normal, which may have caused greater-than-normal ground-water contributions to Little Bean Marsh.

Attempts often have been made to justify wetland protection on the basis that wetlands are recharge areas. However, many wetlands have been shown to be ground-water discharge areas (Carter, 1986). Although Little Bean Marsh was both a recharge and discharge area, discharge to the marsh was three times greater than recharge to ground water during the 12 months of monitoring. Ground-water levels were measured in two shallow (3 m) wells along the northern edge of Little Bean Marsh (W4 and W5, fig. 1), and two shallow wells along the southern edge (W1 and W6, fig. 1). The water-table gradient slopes to the southwest, downstream, and toward the river (fig. 9) as typically occurs in the Missouri River alluvial aquifer (Emmett and Jeffery, 1969). Ground-water levels on the south side of the marsh typically were 6 to 18 cm below the marsh water level, whereas ground-water levels on the north side (wells 4 and 5) typically were 6 to 18 cm above the marsh water level. Accumulations of iron deposits on the north side of the marsh bottom were frequently observed, indicating ground-water seepage from the alluvial aquifer, which typically has large concentrations of iron (Ward, 1973; Ziegler and others, 1993). Also, three marsh tributaries, incised into the terrace scarp that comprises the northern and western boundaries of the marsh, were observed to receive small amounts of ground-water

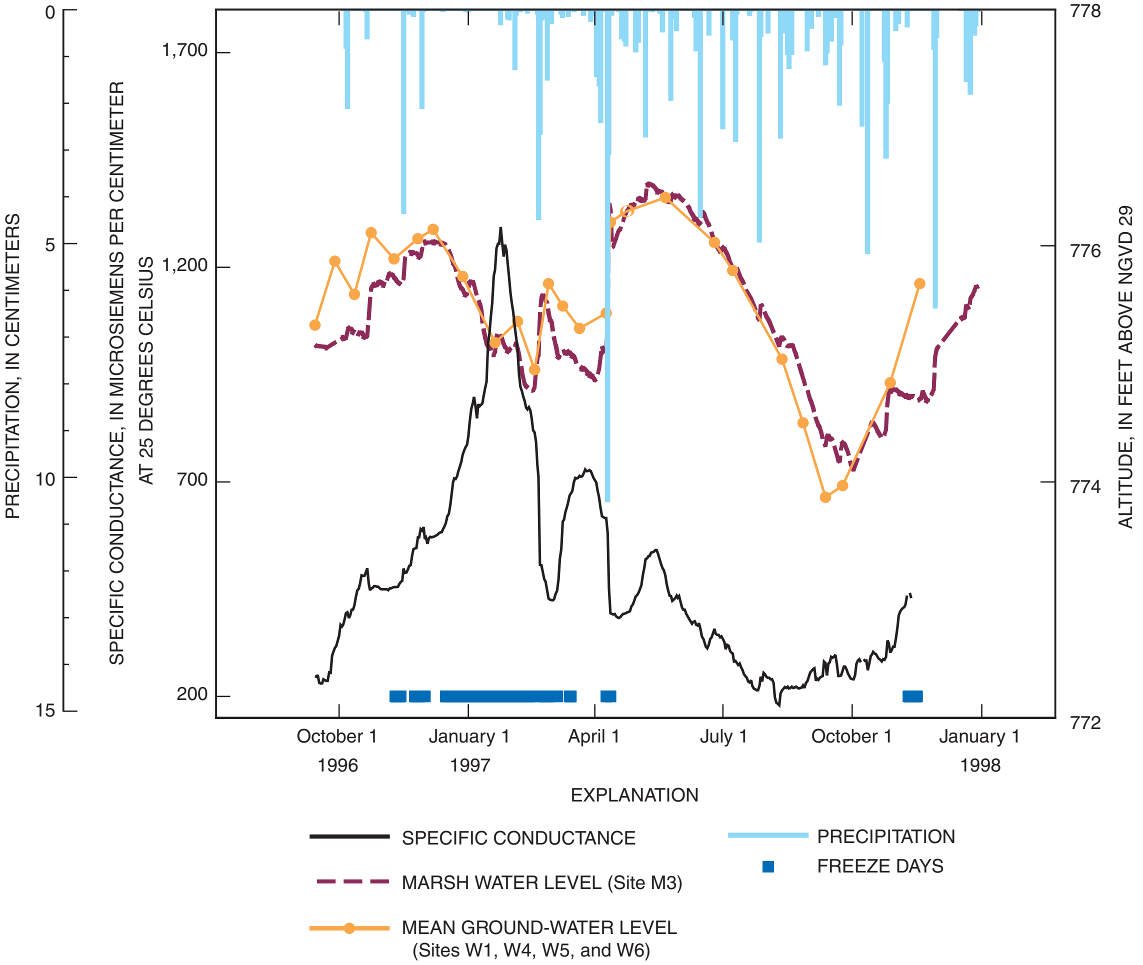


Figure 8. Specific conductance, marsh water level, mean ground-water level, precipitation, and days with mean air temperatures below 0 degrees Celsius at Little Bean Marsh.

seepage during periods of high ground-water levels and wet weather. These factors indicate that ground water typically recharges the marsh on the north side and seeps out of the marsh bottom on the south side.

Ground-water levels were more closely related to water levels in Little Bean Marsh than to stages in the Missouri River (fig. 10), indicating the marsh has good hydraulic connection with ground water. Although the interaction between the marsh and ground water is close, it also is complex. Even in the southern, downgradient wells, the lowest ground-water levels were always above the lowest parts of the marsh bottom. The ground-water connection of Little Bean Marsh indicates that the hydrologic regime of most wetlands along the lower Missouri River largely is a function of bottom altitude relative to water-table altitude. In the past, when Little Bean Marsh was much deeper, the connection between the marsh and ground water would have been even greater. Ground water would likely have caused most of the marsh to remain permanently inundated. In the future, as Little Bean Marsh continues to fill with sediment and detritus, the ground-water connection will decrease, slowly changing the vegetation to terrestrial plants. Although the marsh was inundated during this study, local farmers reported some years when the marsh has been nearly dry. After large runoff events, all ground-water levels are less than the marsh level, and the entire marsh recharges the aquifer. At these times, the marsh can remove selected nutrients and contaminants before they enter the aquifer where they are difficult to remove. Conversely, the marsh also can be an area of aquifer vulnerability where soluble surface-water contaminants, such as agricultural pesticides, can be directly recharged into the aquifer.

During long periods of high evapotranspiration rates, marsh water levels often are sustained by ground water, providing conditions favorable for the cattails (*Typha latifolia*) and bulrushes (*Scirpus fluviatilis*) that are so abundant in the marsh. This interaction with shallow ground water indicates that lowering the water table by less than a meter by pumping would likely make Little Bean Marsh drier and could substantially affect the marsh ecosystem. Specific conductance values of ground water in the Missouri River alluvium typically are about 1,180 µS/cm (Ziegler and others, 1993); somewhat greater than values in shallow ground water near Little Bean Marsh, which average about 750 µS/cm (table 1). Well W9 is located about

1.6 km upgradient from Little Bean Marsh near the bluff line, and has a more typical (Ziegler and others, 1993) ground-water conductance of about 1,000 µS/cm. The lesser conductance values of ground water near the marsh may reflect surface recharge from the marsh itself. Conversely, the alternating effects of precipitation/runoff and ground-water seepage on specific conductance in the marsh are evident (fig. 8). Except during extended ice periods, specific conductance is a good indicator of water source in the marsh. When mean ground-water levels are higher than marsh water levels, specific conductance increases in the marsh; the opposite is true when ground-water levels decline.

Hydrologic Outputs

More water was lost from the marsh through evapotranspiration than by all other pathways combined (59 percent, fig. 7). This partially is because of abundant macrophytes that transpire water, in addition to that lost from evaporation of open-water surfaces. Jensen and Rahman (1987) determined that evaporation from the free water surface of a Bangkok wetland rice paddy was only 13 percent of the total evaporative loss. In Little Bean Marsh surface outflow accounted for 36 percent and ground-water seepage accounted for only 5 percent of the water losses (fig. 7). Consequently, the marsh seems to serve the hydrologic function of surface-runoff reduction, and may even lower the water table when ground-water levels are high in the summer. The percentage of outputs from evapotranspiration would have been less, and surface outflow would have been greater, had Short Creek still run through Little Bean Marsh. This evaporative loss obviously is not present where marshes have been drained and surface water is routed directly to the river. Evaporation leaves behind dissolved constituents and causes higher specific conductances during the summer than would be expected from a precipitation-dominated system. The shallowness of the marsh, ground-water inflows, and large hydrologic-budget components of precipitation and evapotranspiration lead to large fluctuations in specific conductance and dissolved-constituent concentrations (fig. 8). The shallowness of wetlands makes open-water evaporation in wetlands extremely responsive to solar radiation (Parkhurst and others, 1998; Souch and others, 1998). Consequently, the amount of surface outflow also is sensitive to solar radiation. Although macrophytes in Little Bean Marsh transpire much water, they also substantially decrease evaporation by shading. This factor is most important during the early spring and fall when transpiration from *Scirpus fluviatilis* is low, but senescent plants still shade most of the water surface.

The importance of precipitation and evapotranspiration in a Midwestern riparian cypress swamp (*Taxodium distichum*) in southern Illinois also was noted in a study by Mitsch (1978) where precipitation accounted for 70 percent of inflow and evapotranspiration accounted for almost one-half of the outflow. The ratio of drainage area to surface area for this Illinois swamp was 3.5 to 1, and total annual precipitation was only 74.3 cm. Low drainage area to surface area ratios, annual precipitation averaging 75 to 130 cm/yr, average potential evapotranspiration values of 100 to 175 cm, and high percentages of flood plain in the drainage area likely make precipitation and evapotranspiration the largest components of the hydrologic budgets of most riparian wetlands in the Midwest. Ground water interactions are more likely to have increased importance along larger rivers associated with highly productive alluvial aquifers, such as the Missouri River alluvium.

Dividing the average storage of Little Bean Marsh in calendar year 1997 (200,000 m^3) by the average rate of inputs [480 m^3/d (cubic meters per day)] gives a hydraulic residence time of 42 years. Although ground-water inflows and evapotranspiration are not accounted for in the formula for hydraulic residence time and may decrease the real residence time somewhat, the average amount of time that water remains in the marsh is still very long (probably years or decades) and allows the marsh to greatly attenuate peak flows. In fact, the marsh has the capacity to substantially attenuate much larger peak flows, but drainage ditches, designed to remove water as quickly as possible, have routed flows around Little Bean Marsh and deprive it of this function. The large residence time also allows the marsh to greatly affect the water quality of outflow (Moustafa, 1997).

Water Chemistry

The lack of water-quality data on remnant native wetlands requires that certain parameters be characterized before specific processes can be investigated. For example, denitrification and P release occur during low redox conditions, but the temporal and spatial distribution of reducing conditions in remnant wetlands along the lower Missouri River are not known. Therefore, the central tendencies and seasonal variations of N, P, pH, Eh, specific conductance, temperature, and dissolved oxygen were computed on data measured in Little Bean Marsh for 12- or 24-month periods. The intermixing of emergent macrophytes and open water is expected to create substantial spatial variation in these parameters (Rose and Crumpton, 1996). Therefore, samples collected at six sites (M1 to M6, fig. 1) were used to quantify the spatial variability and to investigate statistical differences in water quality between vegetated and open-water sites.

Specific Conductance

Daily fluctuations of specific conductance were less than 50 µS/cm and seemingly nonsystematic except during large rain storms. However, longer-term variability in specific conductance ranged from 180 to 1,300 µS/cm and was related to the source of water in the marsh, the amount of evaporation, and the amount of ice (fig. 8). The primary sources of water to the marsh were precipitation (specific-conductance values usually less than 25 µS/cm) (National Atmospheric Deposition Program, 1997), runoff (50 to 240 µS/cm, table 2), and ground-water seepage (about 750 µS/cm, table 1). The large difference between specific conductance of ground water and precipita-

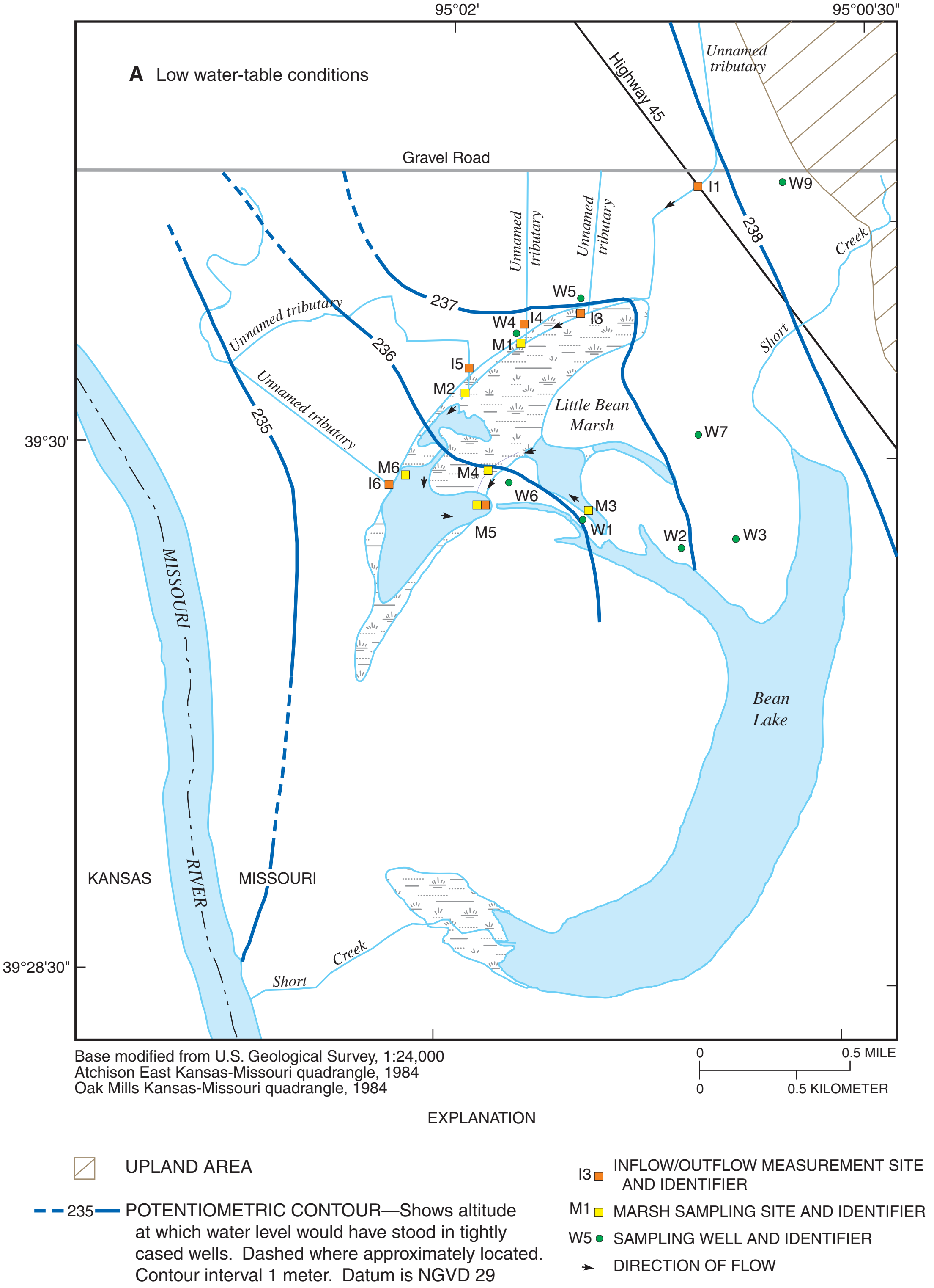


Figure 9. Altitude of potentiometric surface near Little Bean Marsh during (A) low water-table conditions on February 17, 1997, and (B) high water-table conditions on October 24, 1996.

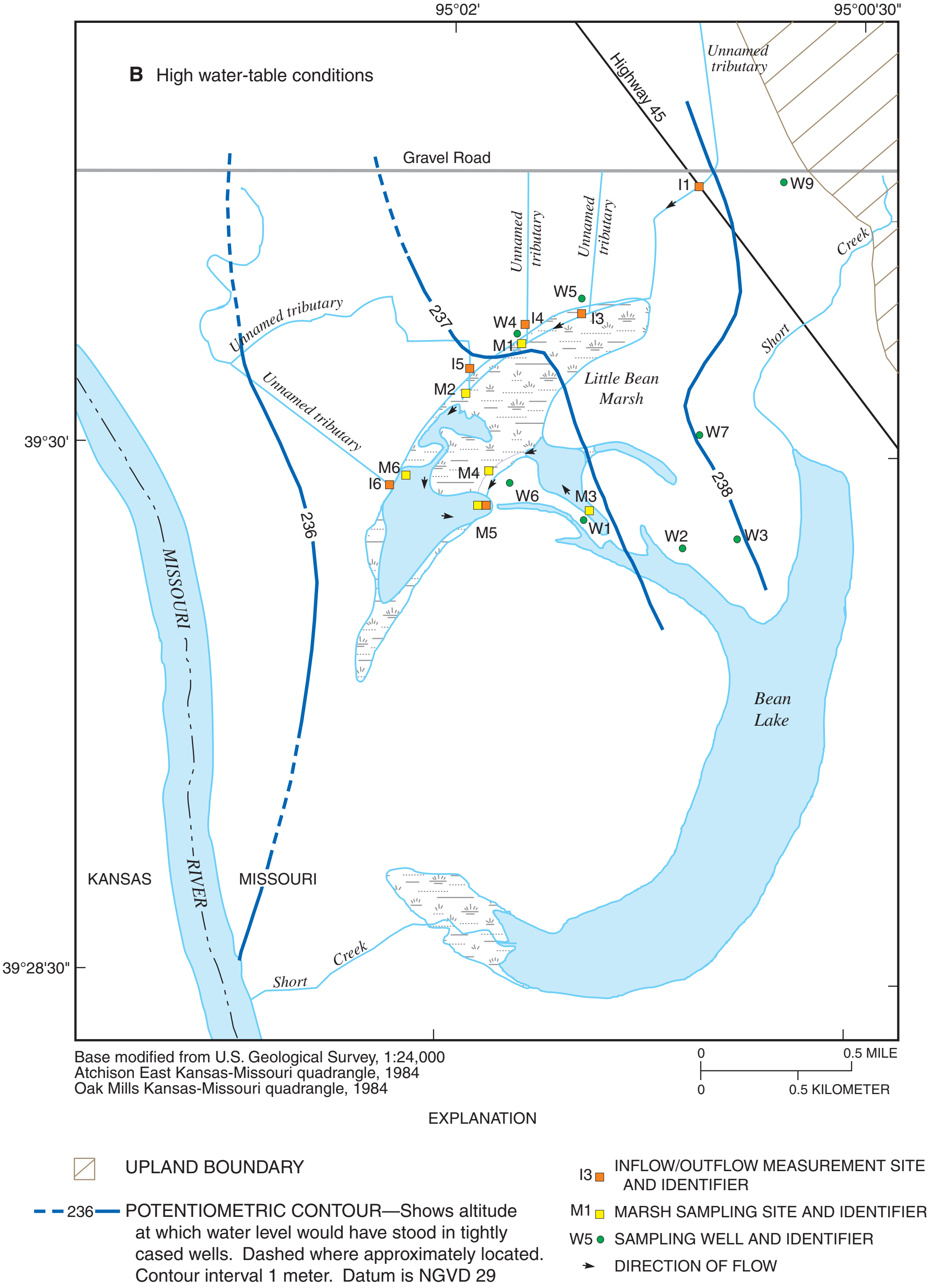


Figure 9. Altitude of potentiometric surface near Little Bean Marsh during (A) low water-table conditions on February 17, 1997, and (B) high water-table conditions on October 24, 1996—Continued.

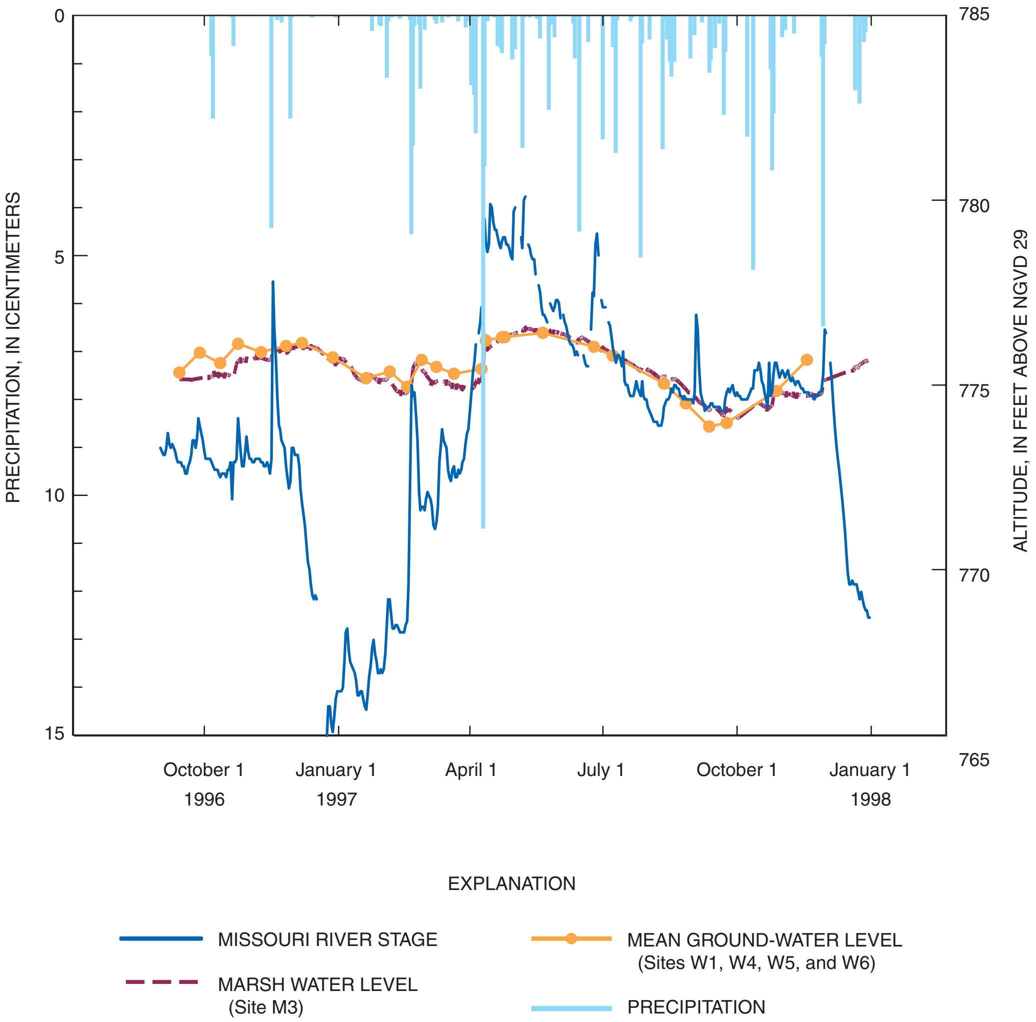


Figure 10. Missouri River stage, marsh water level, mean ground-water level, and precipitation at Little Bean Marsh.

tion/runoff, and the short periods of input from precipitation and runoff, allow the use of specific conductance as an indicator of the primary source water in the marsh. For example, two storm events in late February and early April 1997 substantially decreased specific conductance to about 400 µS/cm. These events were followed by a rapid increase in specific conductance that were likely caused by ground-water seepage as these periods correspond with periods of low evapotranspiration and high ground-water levels. Conversely, a late May 1996 storm decreased specific conductance to 200 µS/cm with only a small rebound (Heimann and Femmer, 1998), when ground-water levels were too low to be a substantial source of water to the marsh.

Another important factor affecting specific conductance was ice formation. Both the 1996 and 1997 annual peaks in specific conductance corresponded with ice formation (Heimann and Femmer, 1998; fig. 8). These peaks were 1,140 and 1,300 µS/cm, which were substantially higher than those associated with ground water (about 800 µS/cm) and about 500 µS/cm larger than any peaks not associated with ice formation. These extremely large increases probably are a result of ion exclusion from the ice and the concentration of excluded solutes in shallow water under the ice. This is likely not an uncommon phenomenon in high-latitude wetlands, where ice can make up a large percentage of the water in shallow water bodies. As the volume of ice in the marsh increases, solute concentrations would be expected to increase geometrically. The concentration of solutes could have significant effects on biota that survive under the ice, especially for solutes with high toxicity. Therefore, concentration of solutes by ice formation could make wetlands more vulnerable to toxic contaminants than other water bodies.

During 1997, the mean specific conductance at site M2 was more than 50 percent larger than the mean conductance at site M5 near the outflow. Tukey's multiple comparison test indicates specific conductance at sites M1 and M2 on the north

Table 2. Sample analyses and storm loads of surface inflow to Little Bean Marsh.

[mg/L, milligrams per liter; kg, kilograms; µS/cm, microsiemens per centimeter at 25 degrees Celsius; m^3, cubic meters; --, no data available; <, less than]

Site number (fig. 1)	Date	Total nitrogen		Total ammonium nitrogen		Total phosphorus		Total dissolved nitrogen		Nitrite plus nitrate nitrogen	
		Concentration, in mg/L	Load, in kg	Concentration, in mg/L	Load, in kg	Concentration, in mg/L	Load, in kg	Concentration, in mg/L	Load, in kg	Concentration, in mg/L	Load, in kg
I3	April 5, 1997	1.14	1.90	--	--	0.26	0.43	1.14	1.90	0.16	0.27
I1	April 10–11, 1997	3.62	104	--	--	.82	24	3.05	8.73	2.51	71.8
I4	April 10–11, 1997	2.03	47.7	--	--	.71	17	1.97	46.2	1.30	30.4
I1	May 8, 1997	22.26	228	4.78	48.8	9.32	95.3	11.39	116	2.18	22.3
Total			381				136		252		125

Site number (fig. 1)	Date	Dissolved ammonium nitrogen		Total dissolved phosphorus		Soluable reactive phosphorus		Total suspended solids		Nonvolatile suspended solids	
		Concentration, in mg/L	Load, in kg	Concentration, in mg/L	Load, in kg	Concentration, in mg/L	Load, in kg	Concentration, in mg/L	Load, in kg	Concentration, in mg/L	Load, in kg
I3	April 5, 1997	<0.005	<0.008	0.17	0.29	0.15	0.25	--	--	--	--
I1	April 10–11, 1997	<.005	<.14	.26	2.4	.26	7.3	429	12,300	393	11,200
I4	April 10–11, 1997	.18	4.3	.37	8.6	.35	8.3	120	2,810	113	2,650
I1	May 8, 1997	2.34	23.9	.53	5.4	.050	.51	9,640	98,500	8,920	91,200
Total			28		22		16		113,600		105,000

Table 2. Sample analyses and storm loads of surface inflow to Little Bean Marsh.—Continued

[mg/L, milligrams per liter; kg, kilograms; µS/cm, microsiemens per centimeter at 25 degrees Celsius; m^3, cubic meters; --, no data available; <, less than]

Site number (fig. 1)	Date	Volatile suspended solids		Specific conductance range, in µS/cm	Storm runoff volume, in m^3
		Concentration, in mg/L	Load, in kg		
I3	April 5, 1997	--	--	--	1,670
I1	April 10–11, 1997	36	1,000	147–241	28,600
I4	April 10–11, 1997	17	400	52–151	23,500
I1	May 8, 1997	720	7,360	--	10,200
Total			8,790		64,000

side of the marsh to be significantly higher than at sites M3 and M5 on the south side of the marsh ($p<0.05$). Whereas M4, on the south side of the marsh, was not significantly different from the north or south sites ($p>0.05$), specific conductance was less at M4 than at either one of the northern sites (fig. 11). These statistics indicate that areal differences were both large and nonrandom. The higher conductance values on the north side of the marsh are likely related to the higher ground-water levels on the north side that provided a constant source of high-conductance ground-water seepage compared to the south side, where ground-water levels usually were below marsh water levels. No consistent statistical significance was observed between specific conductance in open-water and vegetated sites. The areal distribution of specific conductance seems to be a good locator of areas where ground-water seepage is prevalent.

Dissolved Oxygen

Dissolved-oxygen concentrations showed both seasonal and daily patterns. Low dissolved-oxygen concentrations are expected in the summer because of decreased water levels, lower solubility of oxygen, and increased rates of respiration and decomposition at higher temperatures. However, summertime dissolved-oxygen values in Little Bean Marsh were less than most streams and the epilimnions of most lakes in the region, which typically range from 7 to 10 mg/L (milligrams per liter) (Davis, 1991). The water-quality standard for dissolved oxygen for protection of aquatic life in Missouri streams and lakes is 5 mg/L (Missouri Department of Natural Resources, 1999). Dissolved oxygen was near zero for about a month in the summer of 1996 and 1997 (fig. 12). Dissolved-oxygen concentrations in both years remained less than 5 mg/L for 3 to 4 months each summer. Additionally, dissolved oxygen decreased to less than 5 mg/L under the ice in February 1996. These extended periods of anoxia indicate reducing conditions probably were present in the water column as well as in the bottom sediments. Reducing conditions in the water column would inhibit nitrification, enhance denitrification, and possibly cause the release of P from sediments through the reduction of iron. The periods of least dissolved oxygen corresponded with the period of highest water levels in the marsh in 1996 and with the lowest water levels in 1997. Apparently, anoxia is not limited to periods of low water.

The decreased availability of oxygen in areas of emergent macrophytes likely has profound effects on biochemical processes in a marsh. For example, removal of oxygen from the water column decreases rates of nitrification, which is the rate limiting step for N removal in the marsh. Oxidizing energy flow paths are replaced by reducing flow paths such as denitrification, sulfate reduction, and methanogensis, and anoxic water favors invertebrates adapted for low oxygen conditions and, therefore, affects rates of secondary production and consumers higher up the food chain.

Although a short period of anoxia occurred under ice, ice also contributed to extremely high concentrations of dissolved oxygen in the underlying water. Concentrations as high as 26.5 mg/L (nearly 200 percent of saturation) occurred when heavy coats of algae were both in and on the undersides of the ice. Evidently, while algae was producing oxygen, the ice was preventing escape to the atmosphere and low temperatures increased solubility and minimized respiration and decomposition. Also, ice formation may have concentrated some oxygen into the underlying water through ion exclusion as happened with other dissolved constituents. Once again, the shallowness of the wetland contributes to large fluctuations in water quality as all changes in constituent mass are confined to a shallow and, thus, small volume of water.

Dissolved-oxygen concentrations followed typical diel patterns caused by algal photosynthesis (fig. 13). The largest diurnal changes were about 5 mg/L that typically occurred during periods of low dissolved-oxygen concentrations.

Median dissolved-oxygen concentrations ranged from 10.1 to 11.2 mg/L at open-water sites M5 and M3 (fig. 11) and were significantly higher than the three sites with emergent vegetation (M1, M2, and M4) (median concentrations ranged from 4.1 to 7.5 mg/L) ($p<0.05$) in four of the six possible pairwise comparisons. Open-water sites (M5 and M3) also had high dissolved-oxygen concentrations in the other two pairwise comparisons as well, but those differences were not statistically significant ($p>0.05$). Although all measurements were made during the day and within 6 hours of each other, some of the scatter in these data might be from the lack of simultaneous measurement, which would decrease the sensitivity of statistical tests. Nevertheless, these data confirm what has been noted by others (Reddy, 1981), that emergent and floating macrophytes decrease dissolved-oxygen concentrations by shading out algal photosynthesizers and decreasing contact with the wind. Rose and Crumpton (1996) reported that in a prairie pothole wetland, emergent macrophytic vegetation (primarily *Typha spp* and *Scirpus fluviatilis*) causes much lower concentrations of dissolved oxygen than those in open water areas. Macrophytes also contribute to the oxygen demand through production of large amounts of detritus. Macrophyte limitation of dissolved oxygen can be an important wetland characteristic, as it can decrease the wetland's ability to generate dissolved oxygen for fish and other aquatic fauna during summer periods of anoxia.

Temperature

Continuous temperature measurements were made at one of the deeper parts of the marsh in an area of open water (site M3). Water depth at the measuring site was about 1 m at a typical stage. Daily mean water temperatures were computed from continuous data measured at the marsh bottom and 0.5 m below the water surface. Despite shallow depths and the closeness of the two temperature sensors, vertical temperature differences of 3 to 9 °C persisted during June, July, and August of 1996 and 1997, indicating persistent stratification in summer (fig. 14). Persistent stratification also occurred in April and May of 1997.

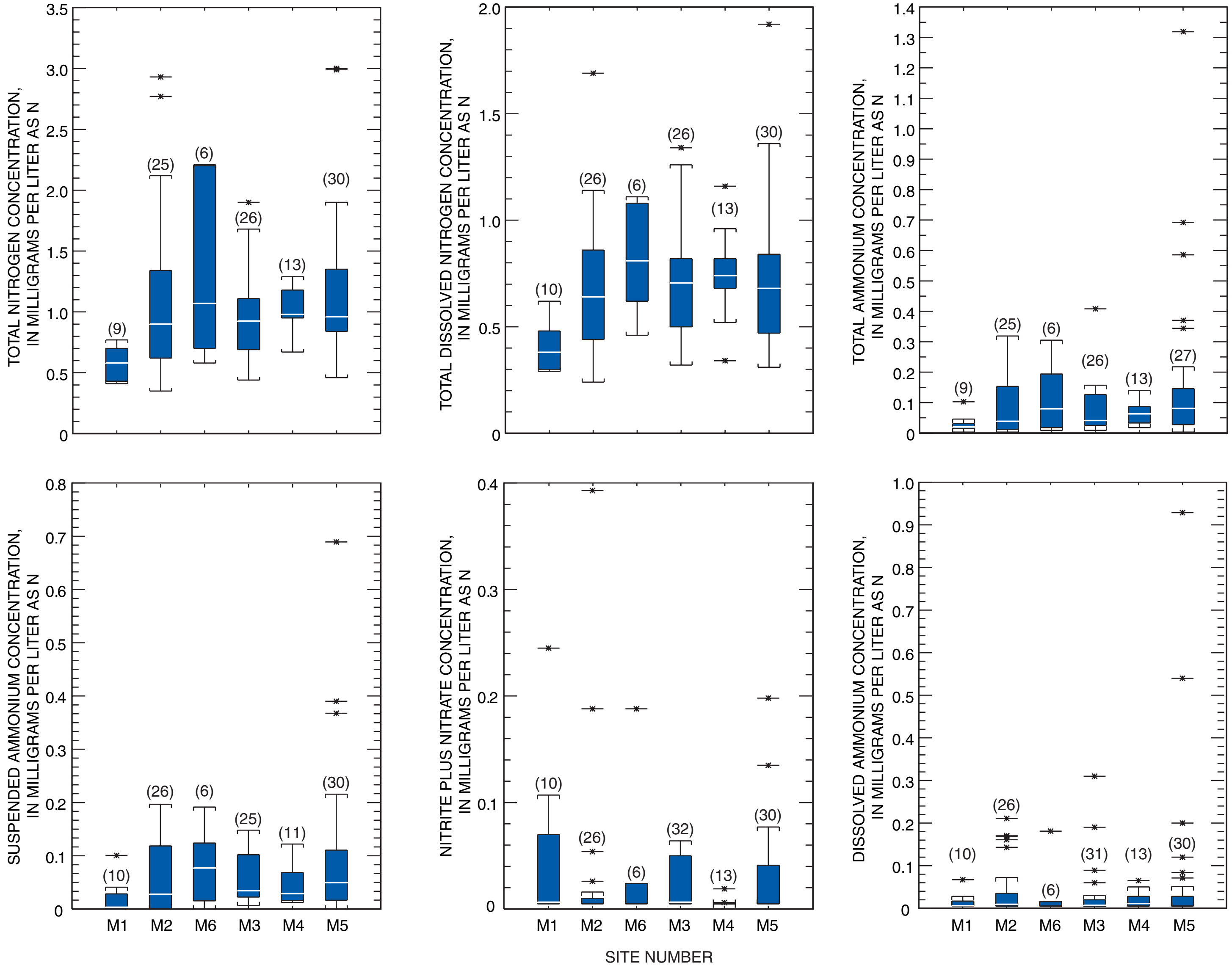


Figure 11. Boxplots of constituent concentrations in samples collected from Little Bean Marsh in 1997.

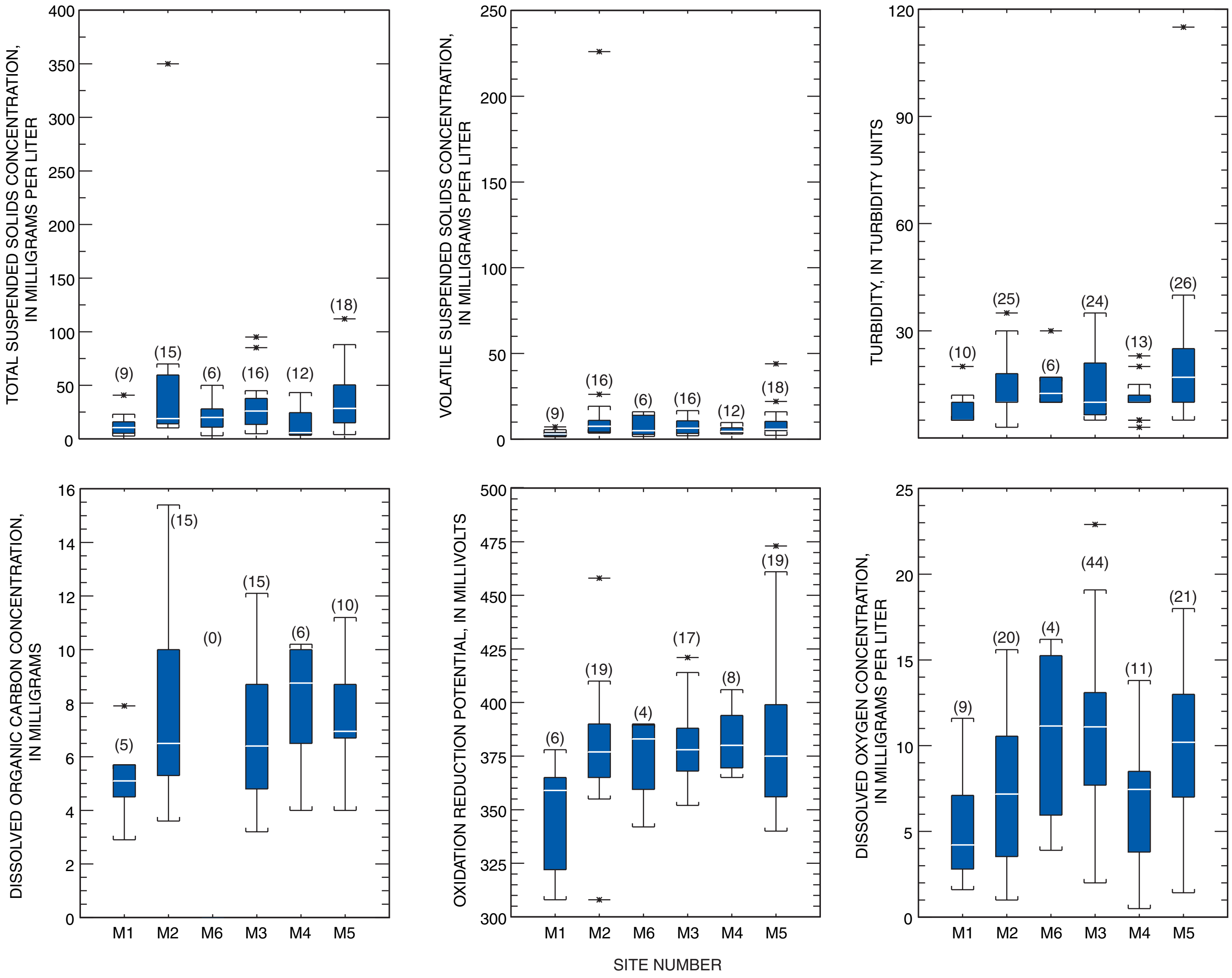


Figure 11. Boxplots of constituent concentrations in samples collected from Little Bean Marsh in 1997   Continued.

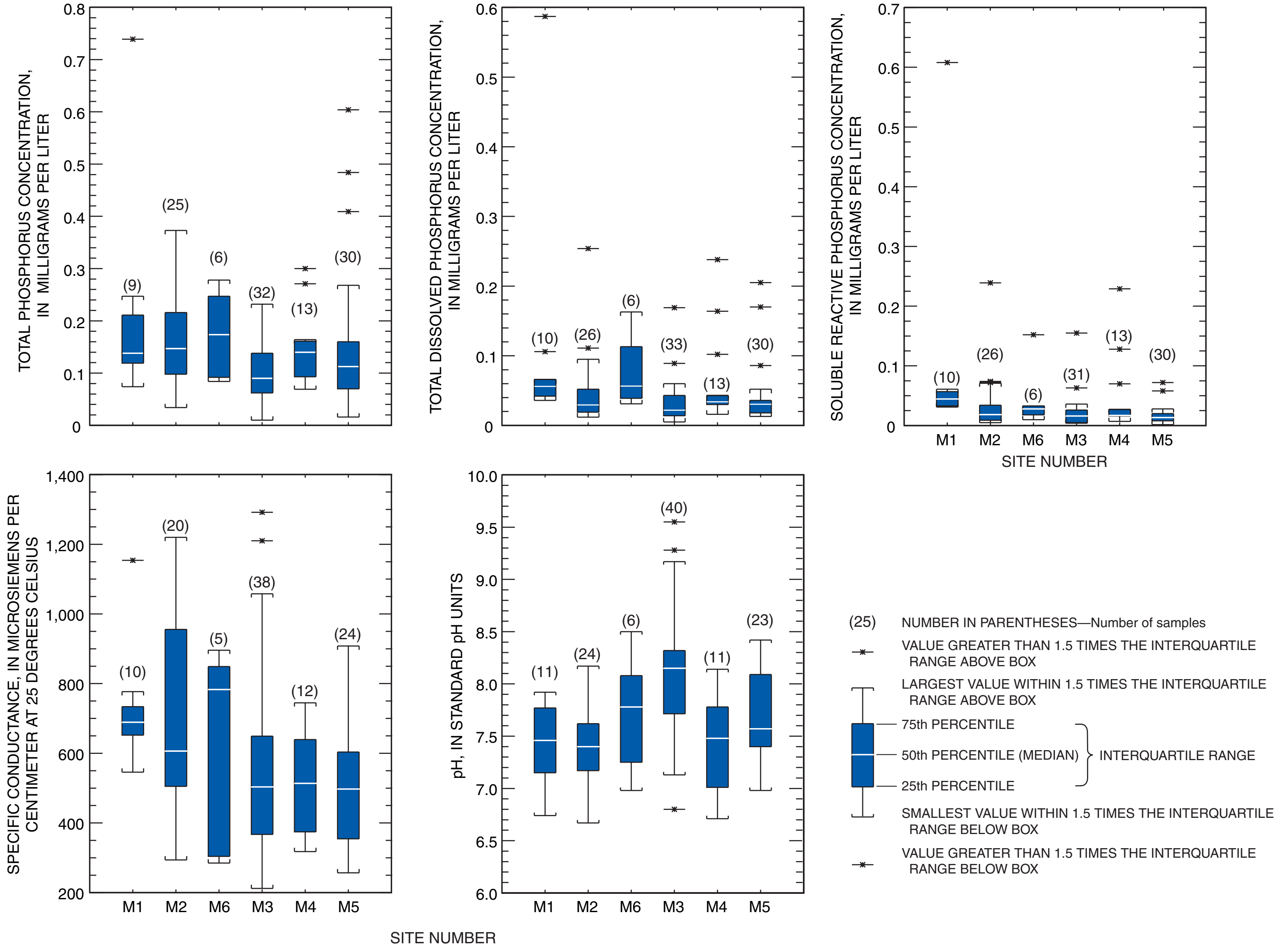


Figure 11. Boxplots of constituent concentrations in samples collected from Little Bean Marsh in 1997—Continued.

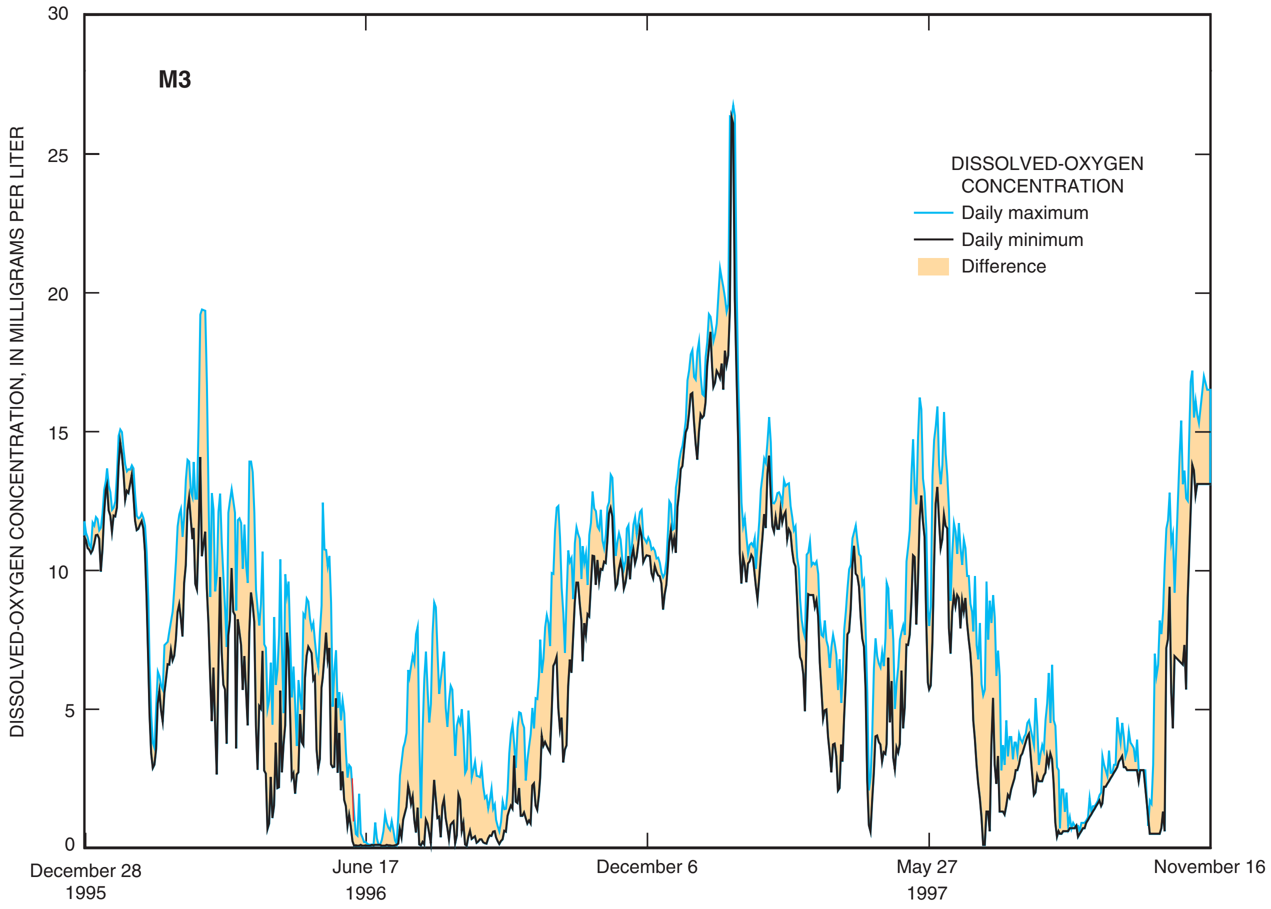


Figure 12. Daily maximum and minimum dissolved-oxygen concentrations at site M3 in Little Bean Marsh.

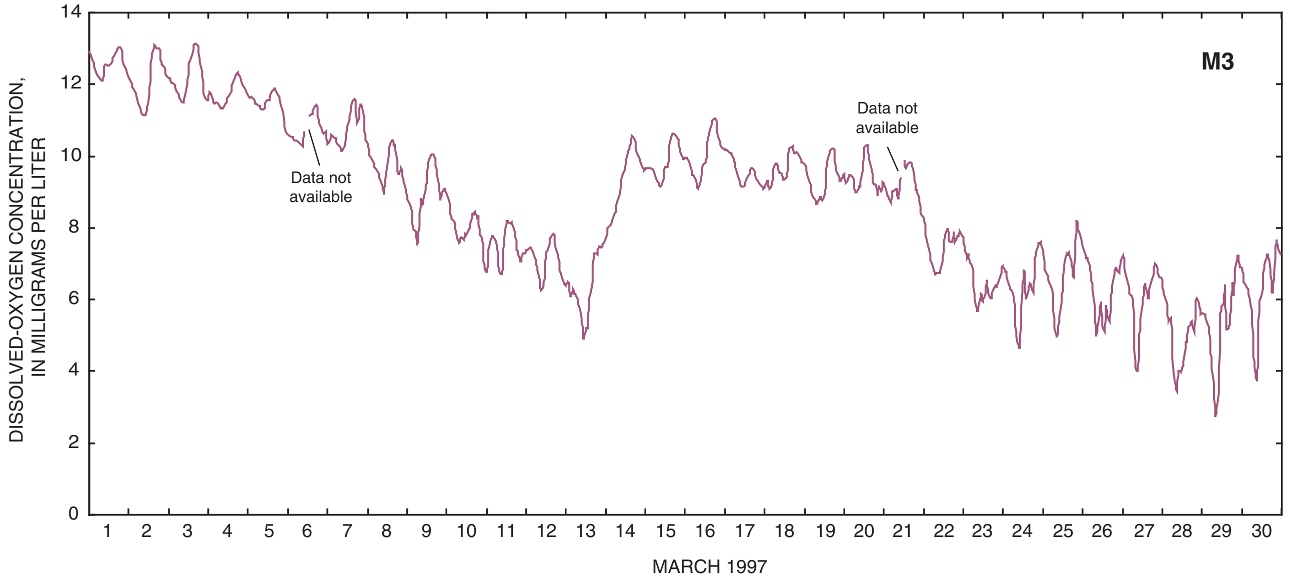


Figure 13. Dissolved-oxygen concentrations at site M3 in Little Bean Marsh during March 1997.

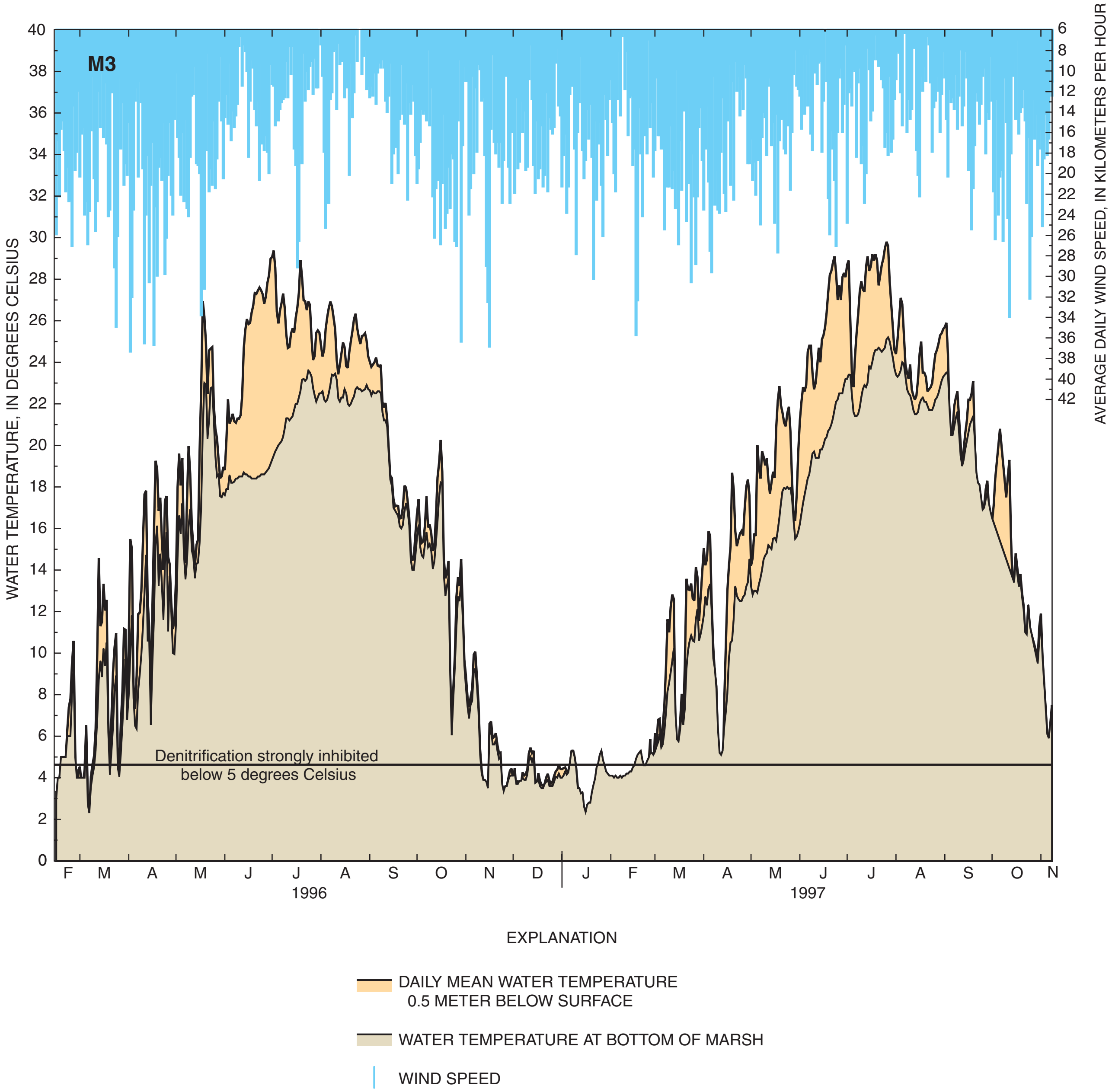


Figure 14. Water temperatures near the top and bottom of Little Bean Marsh at site M3 and average daily wind speed at the Kansas City International Airport.

Temporary stratifications were common in the spring and fall months (fig. 14). Had the upper temperature measurement been made at the water surface, the extent of stratification would have been even more evident. Persistent temperature differences were not expected in such shallow open water, where wind and night-time temperatures were expected to make stratification a temporary phenomenon. At low latitudes, shallow lakes rarely have persistent stratification (Dye and others, 1980). However, persistent temperature differences also were observed by Heimann and Femmer (1998) at two other wetlands in Missouri. The greatest differences occurred above 20 °C where the density of water changes more rapidly with temperature, thus creating stable stratification able to withstand wind mixing. Seepage of 13 ºC ground water into and through the bottom of the marsh may have contributed to stratification in some places. The persistence of stratification also limits the movement of dissolved oxygen to the bottom of the marsh, making the bottom sediments and deeper water susceptible to anoxia. The period of lowest dissolved-oxygen concentration corresponded with the period of maximum stratification (fig. 14, June and July 1996). In July 1997, a strong period of stratification also corresponded with a period of reduced dissolved-oxygen concentrations. Earlier and cooler periods of stratification in spring 1997, however, did not correspond with periods of low dissolved oxygen. The ability of open-water marshes to stratify may increase areas and periods of reducing conditions, which inhibit nitrification and potentially allow denitrification and P release from iron reduction. Chlorophyll data indicate that much of Little Bean Marsh underwent a major algal bloom in September 1997 (table 3, at the back of this report) shortly after summer stratification abruptly ended. The bloom may have been triggered by the mixing of the water column that made ammonia nitrogen from the hypolimnion available to a N-limited algae population.

Daily mean water temperatures 0.5 m below the surface closely corresponded to the daily peaks of air temperatures in the summer (fig. 15). Rarely did the water temperature fall as far as air temperature on cool days. The higher mean water temperatures were likely caused by solar heating of the open water areas that add heat in addition to conductive heat from the air. Summer stratification held much of this heat in the upper 0.5 m. The annual pattern of water temperature was similar for both years with temperatures peaking at 25 to 30 °C for almost 2 months during the summer each year. The mean July temperatures at KCI (34 km southeast of Little Bean Marsh) were 24 °C in 1996 and 25.4 °C in 1997, which are 1.8 and 0.4 degrees below normal (National Oceanic and Atmospheric Administration, 1996 and 1997). However, the close relation between air temperature and water temperature indicates that abnormally warm summers would increase water temperatures (and thermal stresses on biota) in direct proportion to higher air temperatures. Shallow depths and extended periods of anoxia in the marsh would limit the ability of some organisms to escape high-temperature stress.

Hourly near-surface water temperatures (0.5 m below the surface) at site M3 showed seasonal patterns of diel variability. The diel variability was substantially larger in the spring and fall than the summer and winter (fig. 16). The low variability of summer temperatures closely corresponded to periods of persistent stratification, whereas periods of maximum temperature variation occurred during mixed conditions or short periods of temporary stratification (figs. 14 and 16). The lower summer variations likely result from a lack of mixing.

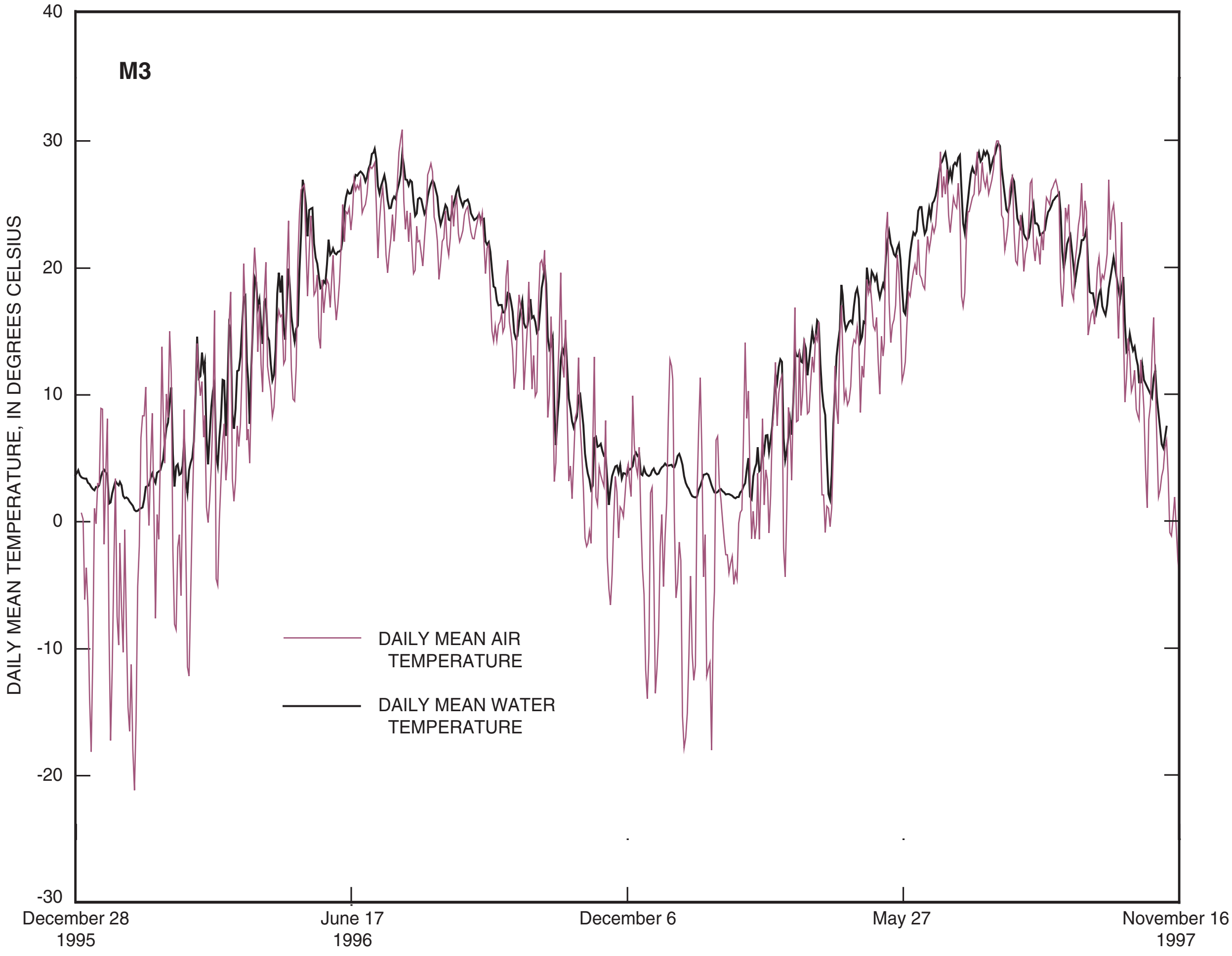


Figure 15. Daily mean air and water temperatures at site M3 in Little Bean Marsh.

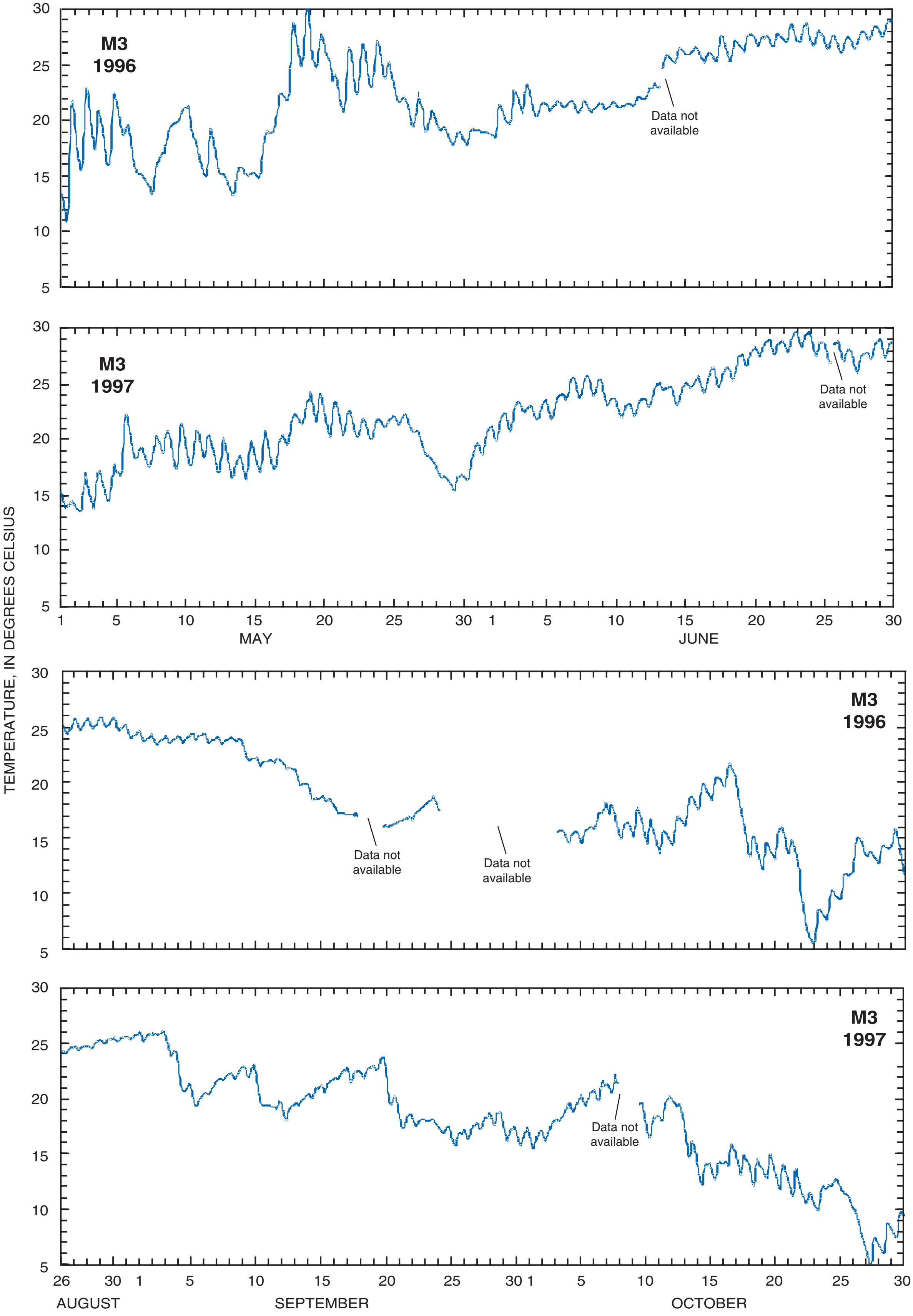


Figure 16. Hourl water temperatures 0.5 meter below water surface at site M3 in Little Bean Marsh during 1996 and 1997.

Surface temperatures near 30 °C were recorded at the open-water sites (table 3, fig. 14). Qualitative observation of other open water areas indicate even higher surface temperatures where dark algal masses, just below the water surface of Little Bean Marsh, added solar heat to the top 1 cm. High surface-water temperatures contribute to high evaporative losses and help account for the large part of the hydrologic outflows attributed to evapotranspiration in Little Bean Marsh. For all of 1997, sample sites with dense emergent vegetation had cooler temperatures than open water sites in 7 out of 10 pairwise comparisons of 5 sites where at least 10 concurrent samples were available. However, a Tukey's multiple comparison test did not show these differences to be statistically significant at the 95 percent confidence level ($p>0.05$) because the maximum difference of mean temperatures between sites was only 2.3 °C. This difference is much less than seasonal variability, which greatly limits the ability of statistical tests to identify small daily differences.

pH

Daily median values of pH ranged from 6.8 to 10.2 during a 2-year period (fig. 17). The lowest values occurred in winter during extreme ice and during large backflow from Bean Lake in June 1996. A rapid, one-unit rebound in February 1997 occurred during and immediately after a large, ice-melting storm. Evidently, the dominant source of water to the marsh plays an important role in determining the pH. During periods when high-bicarbonate ground water constituted the majority of water in the marsh, pH fluctuations were buffered between 7.6 and 8.4. Higher pH values with much larger fluctuations occurred during late summer and fall of 1997, when Little Bean Marsh was filled with low specific-conductance (less than 350 µS/cm) surface water with little buffering capacity. Diurnal fluctuations of pH usually ranged from 0.2 to 0.5 units, with the minimum values typically occurring between 0800 and 1000 hours, and the maximum values around noon (fig. 18), although this pattern occasionally was concealed by other factors such as storm events. When most of the water in the marsh was from lightly buffered rainfall and runoff, the larger fluctuations of pH caused by algal carbon dioxide were much larger.

The median of synoptic pH measurements in Little Bean Marsh ranged from 7.4 at site M1 (vegetated) to 8.2 at site M3 (open water). The median pH of vegetated sites was significantly less than those of open-water sites. All six pairwise comparisons of vegetated and open-water sites had significantly dif-

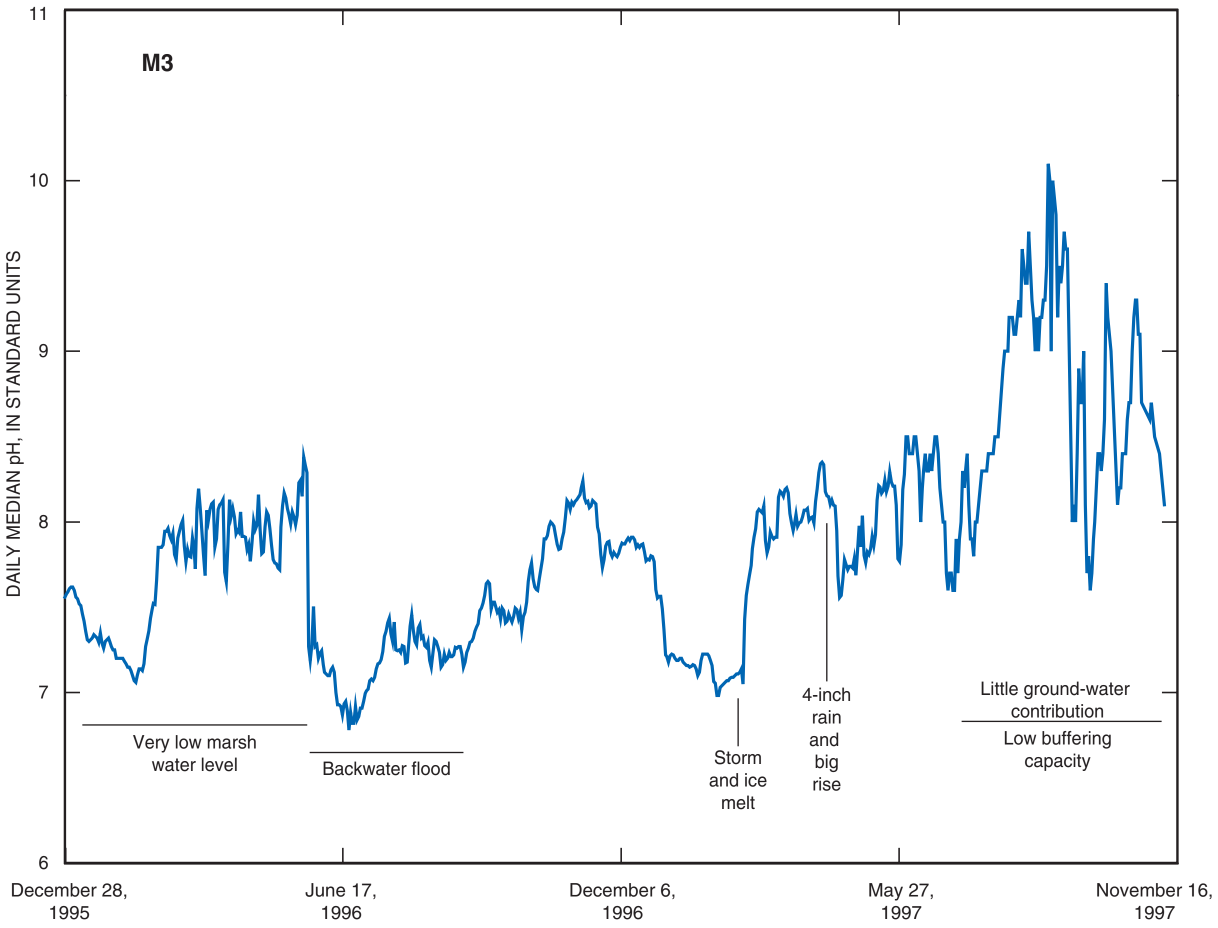


Figure 17. Median daily fluctuations of pH at site M3 in the Little Bean Marsh, 1996–1997.

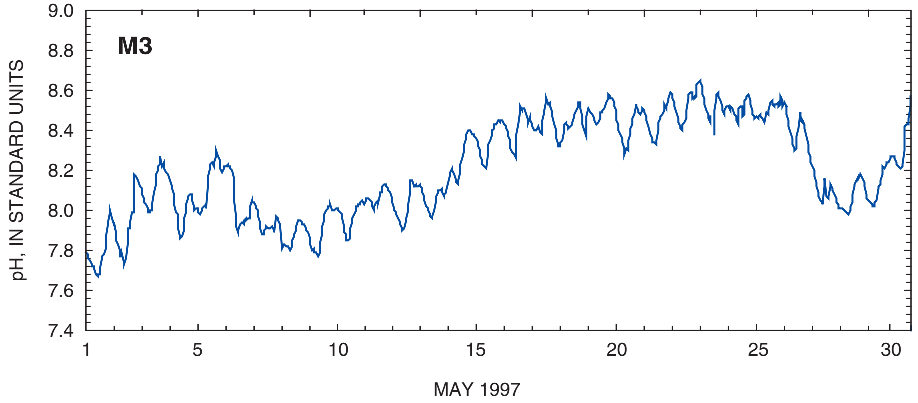


Figure 18. Hourly fluctuations of pH at site M3 in the Little Bean Marsh during May 1997.

ferent median pH values ($p<0.05$). This difference can be explained by daytime sampling, when algal photosynthesis increases the pH (Horne and Goldman, 1994) at the open-water sites more than vegetated sites. Only 1 of the 4 pairwise comparisons of same-cover sites had significant differences at the 95 percent confidence level ($p<0.05$).

Nitrogen

Although most emergent macrophytes receive 95 to 99 percent of their nutrients from the soil (Wetzel, 1983), enrichment of N and other nutrients in the water column can affect the plant composition of a marsh. For example, nutrient enrichment can cause algal blooms, decrease water transparency and impede growth or kill macrophytes (Stewart and Ornes, 1973; Carter and Haramis, 1980; Carter and others, 1985; Carter, 1986). Average total N concentrations in Little Bean Marsh ranged from 0.6 mg/L at site M1 to 1.7 mg/L at site M6 (fig. 11). Concentrations of N would have been larger except for the near absence of inorganic N (NH_4+NO_3+NO_2-N) in Little Bean Marsh (figs. 19 and 20). Apparently, denitrification and biotic uptake keep dissolved inorganic N concentrations low. The average percentage of total N in the dissolved organic form at the five sites ranged from 61 percent at site M5 to 78 percent at site M4. Most of the remaining N was particulate (attached to particles) organic N (table 3). Therefore, most N lost from the marsh was organic N, and most of that was dissolved. This situation contrasts with those of the Missouri River and other streams in the region where oxidizing conditions dominate and the dominant N species is NO_3 (table 4). Nitrogen in the organic form generally is less usable by plants than inorganic N. Therefore, artificial flood-plain drainage that routes water directly to the river and bypasses wetlands are more likely to contribute to excess inorganic N in the river where deep water and high turbidity can limit uptake of the readily available forms of N.

Generally, only the largest storms (greater than 5 cm) produced enough runoff to measurably affect the concentrations of nutrients in Little Bean Marsh. Concentrations of both total N and total dissolved N in the marsh generally were unchanged by storms (figs. 19 and 20, table 3) at outflow site M5. At site M2, concentrations of total N and NO_3 also increased after the second largest storm (fig. 20), which occurred February 20 to 22, 1997. This site was located near the largest tributary of Little Bean Marsh (fig. 1). N concentrations at site M3, which is somewhat isolated from tributaries, showed little response to any runoff events (figs. 19 and 20). The reduced quantities of surface runoff limits the Marsh's allochthonous sources of nutrients, and concentrations of N and P in precipitation usually are much less than those present in surface runoff. Consequently, the response of N concentrations in the marsh to rainfall events is erratic and none systematic.

No statistically significant differences were seen among any sites for dissolved NO_3 ($p = 0.15$), dissolved NH_4 ($p = 0.22$), and total NH_4 ($p = 0.30$) using the Kruskal-Wallis test on ranks of the data. This lack of relation primarily was because of the large number of NO_3 concentrations that were less than the laboratory reporting limit, which were estimated to be one-half of the reporting limit. Consequently, the full variability of NO_3 concentrations between sites was not available for the Tukey test. Nevertheless, the large demand on inorganic N from nitrification, denitrification, and biotic uptake keeps concentrations and spatial variability of NO_3 minimal within the marsh.

No significant differences were detected among sites M2, M3, M4, and M5 for total N and total dissolved N ($p>0.05$) using Tukey's multiple comparison test. However, concentrations of total N and total dissolved N were significantly less at site M1 ($p<0.05$) than all other sites despite being a heavily vegetated site. This difference may have been because site M1 is near the largest surface inflow and is in the area of greatest ground-water inflow. Consequently, the water at site M1 may not have had a chance to dissolve or suspend large quantities of organic N from the marsh vegetation. Also, any NO_3 seeping into the marsh from ground water may be removed in the anoxic bottom sediments (Chen and others, 1972) before entry into the water column.

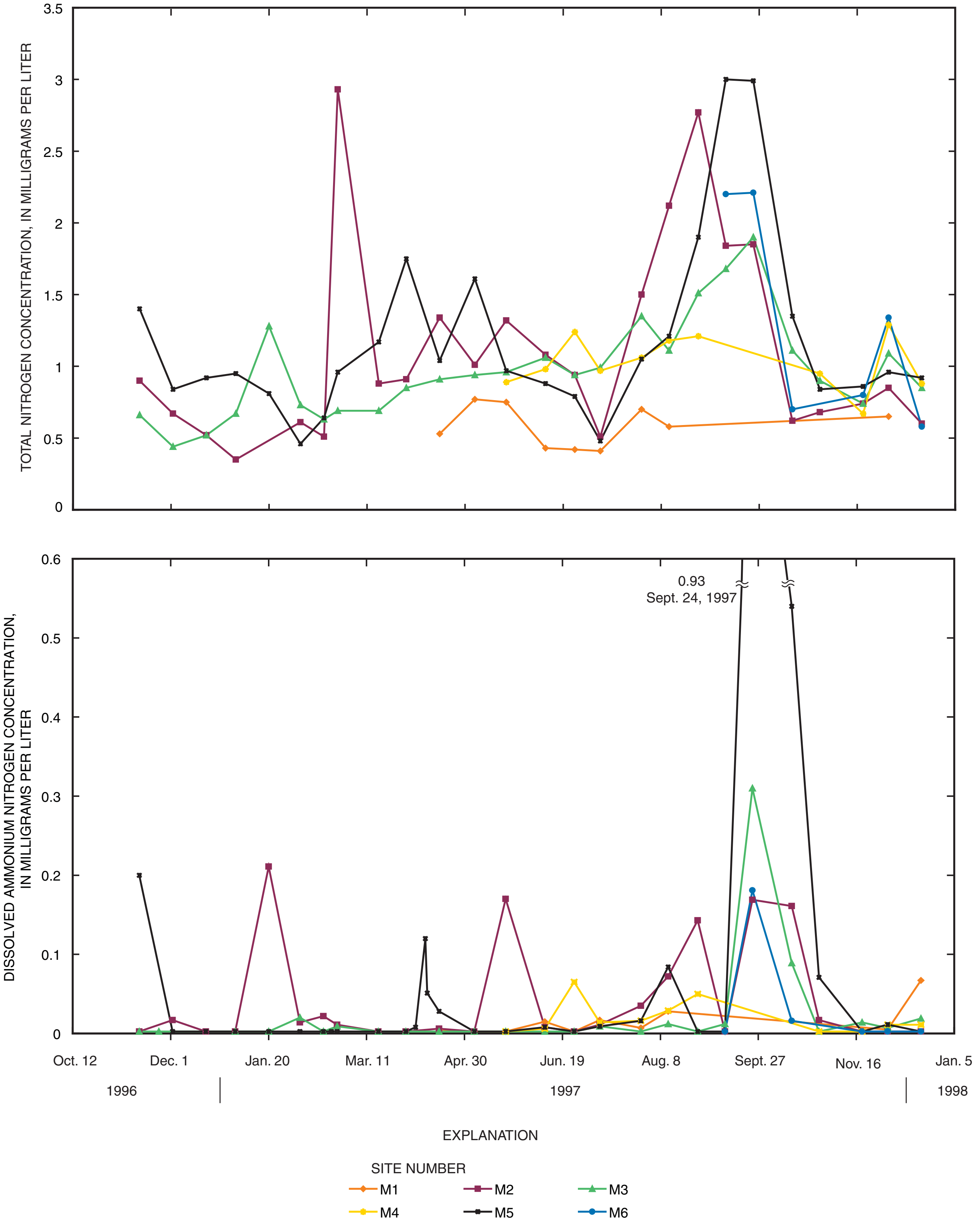


Figure 19. Total nitrogen and dissolved ammonium nitrogen concentrations with time at Little Bean Marsh.

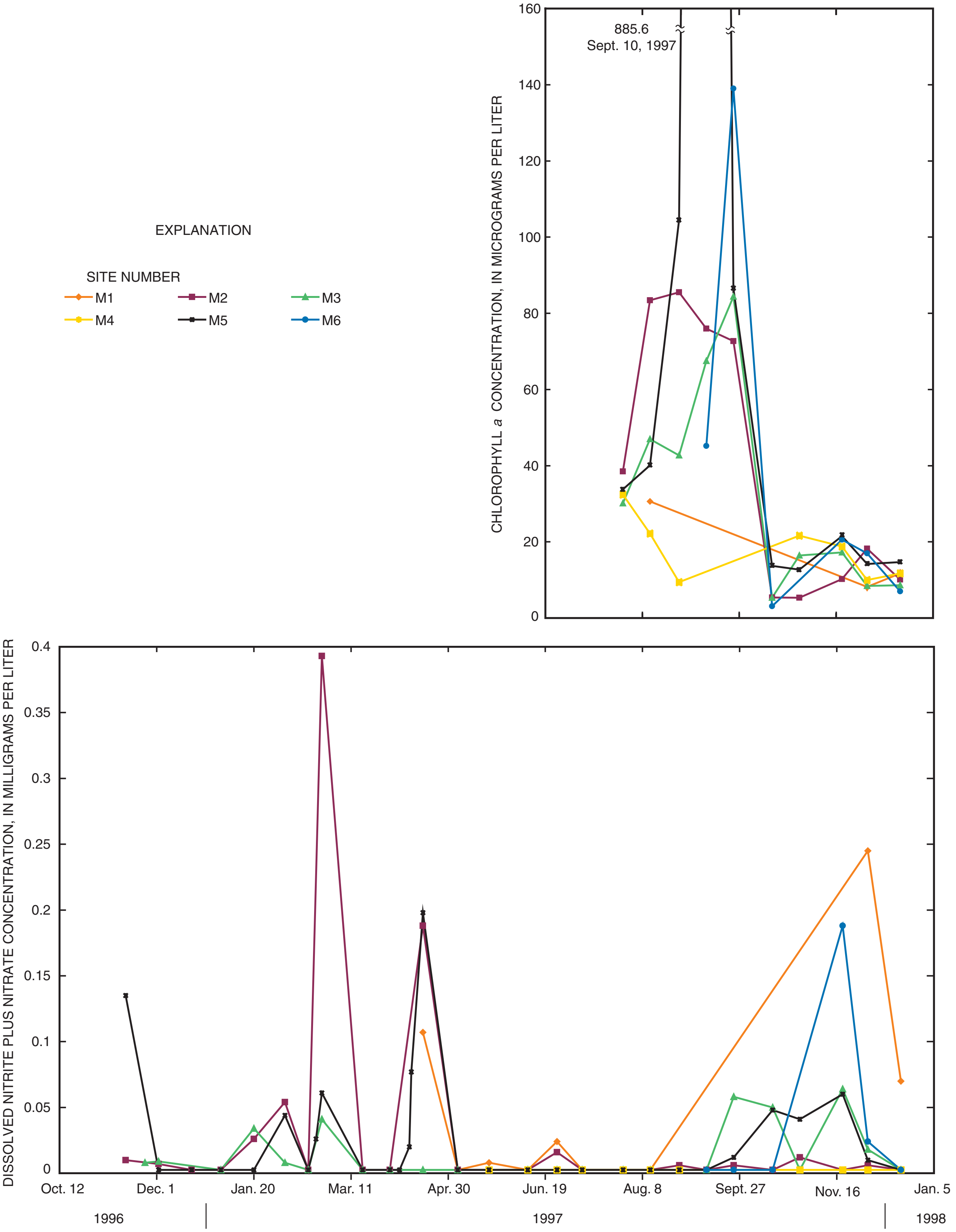


Figure 20. Chlorophyll *a* and nitrite plus nitrate concentrations with time in Little Bean Marsh.

Table 4. Means and ranges of selected constituent concentrations and properties of water samples collected from the Missouri River, Little Bean Marsh, and other wetlands along large rivers of the Midwestern United States.

[mg/L, milligrams per liter; µS/cm, microsiemens per centimeter at 25 degrees Celsius; --, no data; <, less than]

Site	Statistic	Number of sample dates	Period of data collection	Total nitrogen, in mg/L	Total organic plus ammonia nitrogen, in mg/L	Total ammonium nitrogen, in mg/L	Nitrate nitrogen, in mg/L	Total phosphorus, in mg/L	Ortho-phosphorus, in mg/L	Specific conductance, in µS/cm	Dissolved organic carbon, in mg/L	pH, in standard units
Missouri River	Mean	14	Dec. 1996–	--	1.1	0.1	1.1	0.23	0.06	718	--	8.2
at St. Joseph, MO[a]	Range		Dec. 1997	--	0.36–2.50	0.03–0.21	0.32–1.7	0.10–0.57	0.030–0.11	650–830	--	8.0–8.4
Heron Pond,	Mean	6–9	Aug. 1976–	--	1.64	1.00	<0.01	0.47	0.16	--	--	
Johnson Co., IL[b]	Range		July 1977	--	--	--	--	--	--	--	--	
Eagle Bluffs wetland	Mean	5	Dec. 1996–	--	2.6	[c]0.88	2.2	0.97	0.72	899	6.4	8.4
Boone Co., MO[d]	Range		Dec. 1997	--	1–3.9	[c]0.19–1.43	0.41–5.1	0.8–1.21	0.53–0.91	774–1,070	5.0–7.8	8.0–9.0
Eagle Bluffs scour	Mean	3	Dec. 1996–	--	0.98	[c]0.54	0.05	0.082	<0.01	801	3.5	7.5
Boone Co., MO[d]	Range		Dec. 1997	--	0.88–1.1	[c]<0.015–0.92	<0.05–0.068	0.05–0.13	<0.01–0.01	763–801	2.5–5.2	7.2–7.7
Little Bean Marsh	Mean	12	March 1996–	0.58	0.54	0.03	0.047	0.21	0.1	730	--	7.4
Site M1	Range		Dec. 1997	0.41–0.77	0.40–0.77	<0.05–0.10	<0.005–0.24	0.074–0.74	0.031–0.61	530–1,150	--	6.7–7.9
Little Bean Marsh	Mean	27	Dec. 1996–	1.12	1.09	0.086	0.029	0.16	0.033	694	[e]6.3	7.4
Site M2	Range		Dec. 1997	0.35–2.77	0.35–2.76	0.003–0.32	<0.005–0.39	0.034–0.37	0.006–0.074	294–1,220	[e]3.6–10.0	6.7–8.2
Little Bean Marsh	Mean	26	Dec. 1996–	0.97	0.96	0.077	0.013	0.12	0.025	560	[e]5.7	7.9
Site M3	Range		Dec. 1997	0.44–1.90	0.43–1.84	0.009–0.41	<0.005–0.064	0.030–0.23	0.004–0.16	250–1,210	[e]3.2–12.1	7.50–8.44
Little Bean Marsh	Mean	11	March 1996–	1.03	1.03	0.062	<0.005	0.14	0.052	510	--	7.3
Site M4	Range		Dec. 1997	0.67–1.29	0.67–1.29	0.018–0.13	<0.005–<0.005	0.076–0.30	0.007–0.23	320–700	--	6.7–7.9
Little Bean Marsh	Mean	30	Dec. 1996–	1.16	1.13	0.18	0.026	0.16	0.017	500	--	7.7
Site M5	Range		Dec. 1997	0.46–3.00	0.42–3.00	0.003–1.32	<0.005–0.20	0.037–0.60	0.002–0.072	260–910	--	7.0–8.4

[a] Hauck and others, 1999.
[b] Mitsch, 1978.
[c] Dissolved phase only.
[d] Richards, 1999.
[e] Eleven to 13 samples collected between November 15, 1996, and June 25, 1997, only.

Phosphorus

The rerouting of flow from Short Creek around Little Bean Marsh in the 1960's undoubtedly decreased the amount of P entering the marsh. The fine sediments present in a loessal stream such as Short Creek likely added substantial quantities of attached P (Craft and Casey, 2000) in the past; however, that stream now bypasses Little Bean Marsh. Concentrations of total P increased in the marsh in response to the large storm of April 10 and 11 (fig. 21, table 3) at outflow site M5. At site M2, concentrations of total P also increased after the second largest storm, which occurred on February 20 to 22, 1997. This site was located near one of the larger tributaries. The minimal amounts of runoff from all other storms generally precluded increases in P concentrations from storm runoff.

Dissolved to total P ratios for the five marsh sites ranged from 0.30 to 0.43, indicating that the majority of the P usually was in the particulate phase (table 3), although these ratios varied widely for individual samples (range = 0.04 to 1.00) with time. Average annual ratios of soluble reactive P (free PO_4-P plus metal-bound PO_4-P) to dissolved P for five sites (M1, M2, M3, M4, and M5) ranged from 0.50 to 0.85, indicating that most of the dissolved P usually was PO_4. These ratios also ranged widely (from 0.12 to 1.0) for individual samples with time. These variabilities were likely because of varying amounts of algae, sediment, and senescing plants in wetland water in response to season, algal blooms, hydrologic events, and other sources of particulates.

Although mean concentrations of total P and total dissolved P were both larger at vegetated sites (M1, M4, and M2) than at open-water sites (M3 and M5), no statistically significant difference was detected in either constituent between vegetated and open-water sites for 13 concurrent samples collected between April and December 1997 ($p>0.05$). Soluble reactive P was significantly greater at site M1 ($p<0.05$) than at all other sites, despite the fact that it was a heavily vegetated site. Those greater concentrations may have been because site M1 is located in the area of largest surface- and ground-water inflow to the marsh and the lack of time for emergent vegetation to remove substantial amounts of PO_4 from the water. Emergent vegetation is slower to take up PO_4 from the water column than algae and nonrooted vegetation that are shaded out by emergent plants at site M1.

Turbidity and Total Suspended Solids

The annual median turbidities for sites M2 and M3 were 10 formazine turbidity units (FTU) and 17 FTU for site M5. The largest turbidities were associated with the two largest storm events and a September algal bloom, which increased turbidity to 115 FTU (table 3). Excluding two September 1997 samples collected during an algal bloom, the total range of turbidities from all three sites was only 5 to 35 FTU. Turbidities usually were low in Little Bean Marsh (table 3) for several reasons: sediment loadings from the largely flood-plain drainage area were minimal, emergent vegetation shade out algae and shield the water from wind, and high concentrations of bivalent cations increase flocculation rates of inorganic suspended material. The clarity of water in Little Bean Marsh contrasts with northern Missouri reservoirs where inorganic material often limits light penetration and subsequent algal growth (Jones and Knowlton, 1993). Akhurst and Breen (1988) determined that floculation rates of inorganic material in shallow African lakes increased from 0.6 to almost 1.7 percent per hour as specific conductance increased from 370 to 622 µS/cm. The settling rate of clays at 1,054 µS/cm was nearly 2 percent per hour. Settling rates of clays are preferentially increased by bivalent ions such as calcium and magnesium that dominated the ionic content of the Little Bean Marsh (table 5). The high divalent cation content is enhanced by ground water received by Little Bean Marsh, which indirectly increases the trapping efficiency of sediment-affinitive constituents such as P. Also, ground water generally is much less turbid than surface water when they first enter the wetland. Although emergent vegetation increases water clarity in shallower areas, high ionic-content water can decrease the duration of high-turbidity episodes (caused by wind, runoff events, and backwater through the flood gates) in open-water areas by increasing the flocculation rate. For example, approximately one week after the 12.5 cm rain on April 10–11, 1997, turbidities had decreased to 5, 35, 22, and 30 FTU at sites M1, M2, M3, and M5 (table 3). Consequently, the amount of ground water received by other wetlands in the lower Missouri River flood plain may have a substantial effect on water clarity. A more incised, ground-water-fed wetland may be expected to have better water clarity than a shallow surface-water-fed wetland. In addition to enhancing algal growth, increased water clarity also can enhance the growth and establishment of submerged and emergent vegetation.

The average concentration of volatile suspended solids expressed as a percentage of total suspended solids for each marsh site ranged from 28 to 56 percent with an average of 35 percent. Assuming chlorophyll represented 2 percent of the living algal biomass, mean concentrations of algae at the marsh sites ranged from 10 to 22 percent of the volatile suspended solids, indicating most of the organic seston was detritus.

The turbidity and suspended-solids differences between open-water and vegetated sites was likely caused by light limitation of algae in areas shaded by emergent macrophytes. The spatial variability of turbidity was similar to total suspended solids in that vegetated sites M1, M2, and M4 had lesser turbidities and suspended solids concentrations than open-water sites M3 and M5. Tukey's multiple comparison test indicated differences in concentrations of total suspended solids at vegetated sites M1 and M4 and open-water sites M2, M3, and M5 were statistically significant ($p<0.05$). Turbidity at vegetated sites M1, M2, and M4 was significantly less than at the two open-water sites M3 and M5 ($p<0.05$). Site M2 was less densely vegetated than sites M1 and M4, and both turbidities and suspended solids concentrations at site M2 fell in between values for the open-water sites (M3 and M5) and the more densely-vegetated sites.

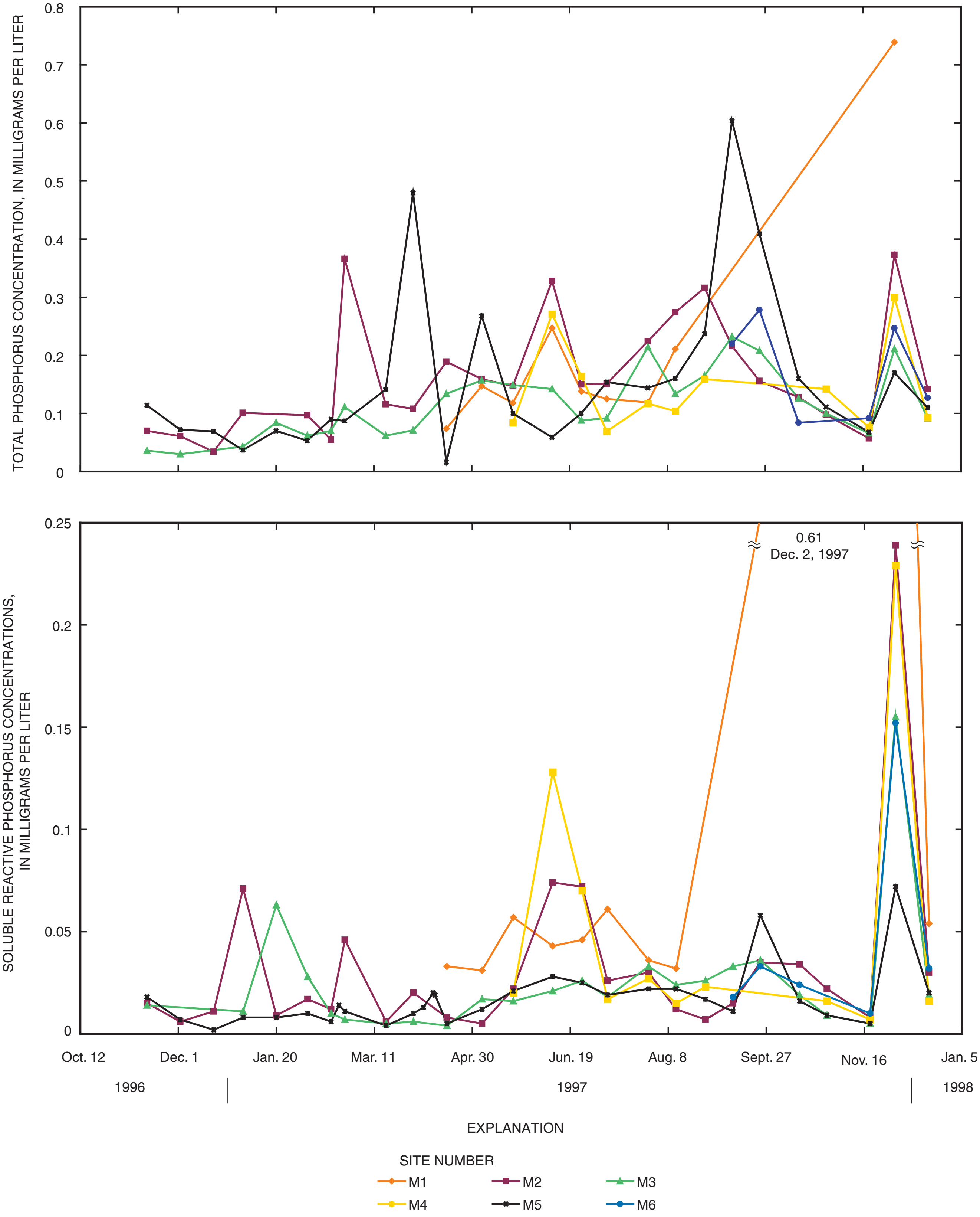


Figure 21. Total phosphorus and soluble reactive phosphorus concentrations with time at Little Bean Marsh.

Table 5. Concentrations of selected ions in water samples collected from Little Bean Marsh.

[mg/L, milligrams per liter; --, no data]

Site number (fig. 1)	Date	Alkalinity, in mg/L	Dissolved bicarbonate, in mg/L	Dissolved chloride, in mg/L	Dissolved calcium, in mg/L	Dissolved magnesium, in mg/L	Dissolved sodium, in mg/L	Dissolved potassium, in mg/L
M2	02/05/1997	554	688	--	--	--	--	--
M2	02/17/1997	688	840	--	--	--	--	--
M2	11/19/1997	532	--	4.6	168.0	47.4	24.5	16.1
M3	01/20/1997	681	849	--	--	--	--	--
M3	02/05/1997	598	747	--	--	--	--	--
M3	11/19/1997	280	--	5.3	80.5	30.9	16.5	13.1
M4	11/19/1997	295	--	5.7	86.4	30.7	17.0	14.0
M5	01/20/1997	456	556	--	--	--	--	--
M5	02/05/1997	255	317	--	--	--	--	--
M5	02/17/1997	480	586	--	--	--	--	--
M5	10/28/1997	273	--	--	--	--	--	--
M5	11/19/1997	--	--	3.8	74.7	29.0	16.8	11.7

Total suspended solids were not significantly correlated with windspeed at open-water sites M2, M3, and M6 (Spearman's rho ranged from 0.30 to 0.54 and p ranged from 0.10 to 0.13) or at the vegetated sites (maximum Spearman's rho = 0.02, minimum p = 0.49) M1 and M4. This result contrasts with the open-water oxbows sampled by Knowlton and Jones (1997) where a good relation existed. The minimal effect of wind on total suspended solids was likely because of the wind-breaking affects of emergent macrophytes. Wind affects were limited even in open water areas by emergent macrophytes that break these areas up into small patches with short fetches. Relations between windspeed and total suspended solids also may have been masked, at least in September 1997, by algal blooms that were not related to windspeed.

Dissolved Organic Carbon

Dissolved organic carbon (DOC) samples were collected from Little Bean Marsh in the winter and spring of 1997 only. Concentrations averaged 6.2 mg/L, and ranged from 2.9 to 12.1 mg/L (tables 3 and 4). These values are large enough that they probably do not limit denitrification rates in Little Bean Marsh. These values also are somewhat larger than the narrow range of DOC measured in samples collected from the Missouri River at Herman, Missouri, in 1997 (Hauck and others, 1998), and are evidence that wetland outflows from vegetated marshes, such as Little Bean, increase concentrations of DOC in the Missouri River.

Trophic Indicators

Particulate N:P ratios were nearly always less than 22 (fig. 22), which is indicative of N deficiency in phytoplankton according to Hecky and Hendzel (1980). This relative deficit in N is likely because of the removal of N from the marsh through denitrification rather than an overabundance of P. N:P ratios in the inorganic fractions were less than 10, which also is indicative of N-limited growth of phytoplankton (Horne and Goldman, 1994).

Chlorophyll concentrations were significantly correlated with total N at open-water sites (Spearman's rho = 0.69 to 0.94, p = 0.003 to 0.03) in Little Bean Marsh (table 6). Correlations with total P usually were not as strong. These relations also indicate that N typically is more limiting than P. N limitation is not uncommon in other types of wetlands (Kibby, 1978). The lack of correlation between total N and chlorophyll at vegetated site M4 is evidence of light limitation.

Jones and Knowlton (1993) classified the trophic state of Missouri reservoirs using concentrations of chlorophyll *a*, total N, or total P (table 7). Concentrations of these constituents were determined in samples from the six sampling sites in Little Bean Marsh collected in 1997 (M1 to M6). Comparison of these criteria with concentrations in Little Bean Marsh indicate the marsh typically is eutrophic or hypereutrophic, regardless which of the three criteria are used. If the high primary productivity of the water column indicated by the trophic state was added to that of the macrophytes, Little Bean Marsh is indeed a highly productive system. Four Missouri River oxbow lakes sampled by Knowlton and Jones (1997) were even more eutrophic than Little Bean Marsh (table 7). The mean concentrations of total N and total P were less than those of the four

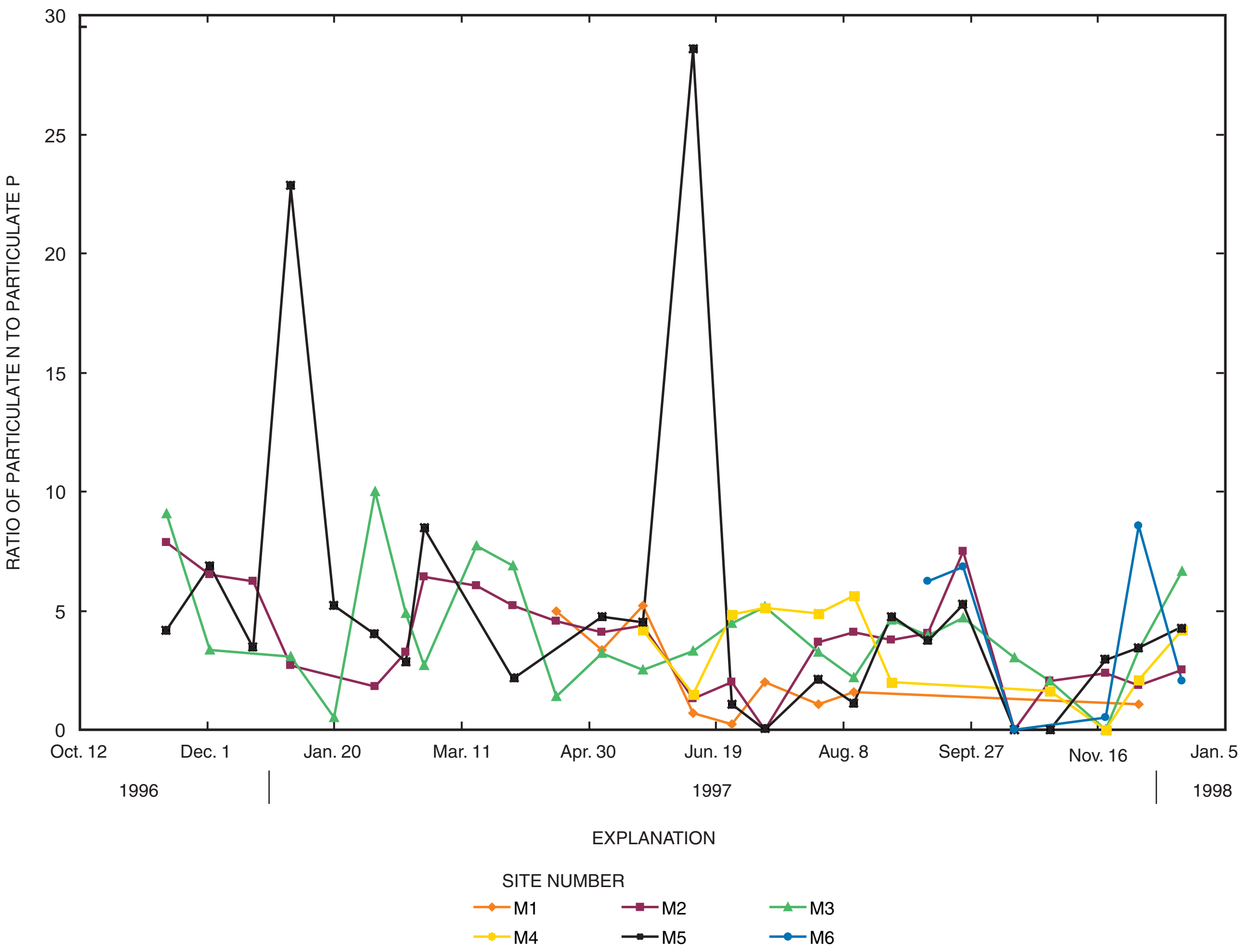


Figure 22. Ratios of particulate nitrogen to particulate phosphorus with time at Little Bean Marsh.

Table 6. Correlation coefficients (r), coefficients of determination (r^2), and levels of significance (p) for the relation between chlorophyll concentrations and concentrations of total nitrogen, total phosphorus, and total suspended solids in Little Bean Marsh.

[N, number]

	Observations	**Total nitrogen**		**Total phosphorus**		**Total suspended solids**	
Site	**N**	**Spearman's rho**	**p**	**Spearman's rho**	**p**	**Spearman's rho**	**p**
M2 (semi-open water)	10	0.94	0.003	0.71	0.018	0.26	0.16
M3 (open water)	10	.69	.02	.44	.095	.59	.015
M4 (vegetated)	7	-.32	.80	-.39	.842	.09	.52
M5 (open water)	10	.76	.01	.76	.002	.64	.005
M6 (open water)	6	.89	.03	.71	.063	.83	.011

Table 7. Classification of the trophic state of Missouri reservoirs used by Jones and Knowlton (1993); May–September means of total phosphorus, total nitrogen, chlorophyll *a*, and total suspended solids for Missouri reservoirs by physiographic region (Jones and Knowlton, 1993); means of monthly samples of four oxbows along the Missouri River (Knowlton and Jones, 1997) from 1994–1996; and 1997 annual means for Little Bean Marsh.

[µg/L, micrograms per liter; mg/L, milligrams per liter; <, less than; --, no data; >, greater than]

Trophic state or water body	Chlorophyll *a*, in µg/L	Total nitrogen, in mg/L	Total phosphorus, in mg/L	Total suspended solids, in mg/L
Trophic State				
Oligotrophic	<3	<0.3	<0.01	--
Mesotrophic	3–7	0.3–0.5	0.01–0.025	--
Eutrophic	7–40	0.5–1.2	0.025–0.100	--
Hypereutrophic	>40	>1.2	>0.10	--
[a]Missouri Reservoirs	15	0.6	0.045	11
[a]Glaciated Plains Region	15	0.7	0.047	10
[a]Osage Plains Region	23	0.8	0.067	20
[b]Four oxbows along Missouri River	113	2.3	0.371	103
Little Bean Marsh	[c]56	[d]0.9	[d]0.114	[e]20

[a] Samples collected in the summers only from 1978 through 1984.

[b] Samples collected at monthly intervals between June 13, 1994, and June 18, 1996.

[c] Samples collected at biweekly intervals from July 29 through December 19, 1997.

[d] Samples collected at biweekly intervals for all of 1997.

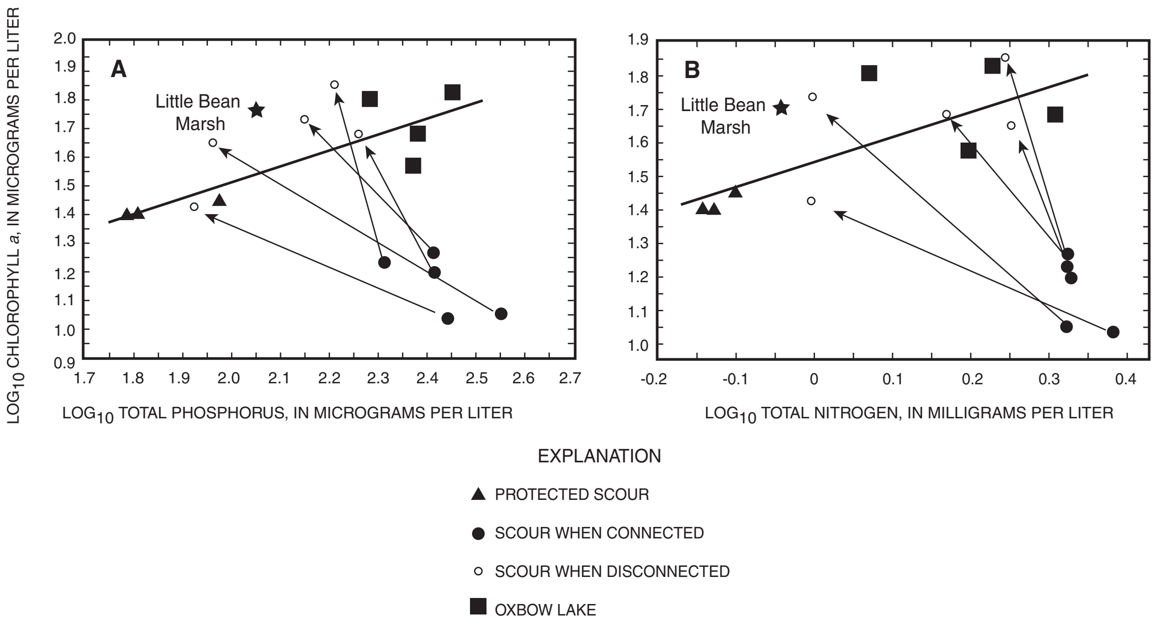
[e] Samples collected at biweekly intervals from April 17 through December 19, 1997.

oxbows (fig. 23), and the ratios of chlorophyll a to both N and P were greater than these ratios in the oxbows, perhaps indicating more detritus in the oxbows. The mean concentrations of all three constituents were larger in these open-water oxbows than in Little Bean Marsh. Periods of hypereutrophy in Little Bean Marsh, as indicated by concentrations of chlorophyll and total P, occurred for much of the summer at open-water sites M3, M5, and at partially open site M2 (figs. 20 and 21). Vegetated sites had less persistent periods of hypereutrophy, especially as indicated by chlorophyll a concentrations. The most intense periods of hypereutrophy occurred in September, coinciding with both an algal bloom and senescence of *Scirpus fluviatilis* and *Typha latifolia.* Even total N concentrations indicated hypereutrophic conditions during this period. These two events may not have been coincidental. The rapid leaching of N that occurs soon after macrophyte senescence may have provided a load of N to the N-limited open-water areas, which triggered a phytoplankton bloom and an intense period of hypereutrophy. However, the time interval between sample collection was too long to document this sequence of events. The triggering of fall periods of hypereutrophy by senescing macrophytes could be a common phenomenon in remnant riparian wetlands along the Missouri River where emergent vegetation is abundant. The period of intense hypereutrophy was ended abruptly in early October by a 2-inch rain and cold temperatures (figs. 10 and 15).

Eutrophication and hypereutrophication occurred in Little Bean Marsh in spite of the fact that most of the marsh is shaded by macrophytes and is N limited. However, the somewhat lesser concentrations of trophic indicators in Little Bean Marsh, compared to the four open-water oxbows of Knowlton and Jones (1997) (table 7, fig. 23), are most pronounced in the large areas of emergent macrophytes and may be caused, in part, by shading by that vegetation. This interpretation indicates some light limitation in addition to the N limitation identified earlier. Chlorophyll a concentrations were significantly correlated with total suspended solids at open-water sites M3, M5, and M6 (table 6), but were not significantly correlated at vegetated sites M2 and M4. Therefore, at open-water sites, a substantial part of the particulate matter probably is phytoplankton, but at light-limited vegetated sites is either detritus or mineral sediment.

Nitrogen Cycling and Denitrification

One of the primary beneficial functions commonly ascribed to wetlands is the removal of N from surface waters.



NOTE: Data from scours connected to the Missouri River were averaged separately for periods of low and high connectivity. Regression lines shown are based on data from the disconnected lakes only (modified from Knowlton and Jones, 1997),

Figure 23. Geometric means of chlorophyll *a* concentrations with (A) total phosphorus concentrations and with (B) total nitrogen concentrations for scour and oxbow lakes along the Missouri River and Little Bean Marsh.

Therefore, the following section addresses N cycling and denitrification in Little Bean Marsh.

Sinks and Sources of Nitrogen

Nitrogen limitation of algal growth and delivery of excess N into the Missouri River, and ultimately the Gulf of Mexico, make quantification of N cycling and denitrification in riparian wetlands an important element in the management of trophic problems such as Gulf Coast hypoxia (Goolsby and others, 1999). However, the compartments and processes of N in wetlands are many and complex (fig. 24). Therefore, detailed analysis and special denitrification experiments were done on Little Bean Marsh to assess its effectiveness in removing N from surface water before it enters the Missouri River.

The amount of N leaving the marsh in surface outflow is critical to evaluating the nutrient removal function of a marsh. Outflow loads can be used to evaluate the impact of the marsh on receiving water bodies, such as the Missouri River, as well as evaluating the relative impacts of various N sources to the marsh. Approximately 530 kg (kilograms) of total N left Little Bean Marsh in surface outflows in 1997 (table 8). Dividing this value by the 952 ha drainage area gives an areal yield of 0.56 kg/ha (kilogram per hectare), which is less than one-half of the 1.4 kg/ha areal yield of the Missouri River at St. Joseph, Missouri, in 1997 (Hauck and others, 1998) and slightly more than one-half of 0.9 kg/ha average total N yield of the entire Missouri River Basin in 1980–1996 (Goolsby and others, 1999). The majority of the N in Little Bean Marsh was dissolved organic N; very little was inorganic N (table 3). The largest N loads leaving the marsh in surface outflows occurred in response to the two largest storms on February 20–22 and April 10–11, 1997.

Atmospheric Contributions

Direct atmospheric inputs of N to wetlands generally are small (Peterjohn and Correl, 1984; Bowden, 1987; Morris, 1991). Concentrations of N in rainfall generally are less than those present in the Missouri River (Hauck and others, 1998; http://nadp.sws.uiuc.edu/nadpdata), or in runoff from its tributaries in Missouri and Kansas. However, because precipitation constituted 41 percent of the inflow to Little Bean Marsh and other sources of N were relatively small, precipitation was the second largest source of N to Little Bean Marsh. Data from the National Atmospheric Data Program (http://nadp.sws.uiuc.edu/nadpdata, accessed 2000) show deposition of NO_3 and NH_4-N at two sites within 322 km and on either side of Little Bean Marsh in 1997 were 4.7 and 6.0 kg/ha respectively. Using an annual precipitation loading of 5.4 kg/ha and an average surface area of 81 ha, the estimated precipitation load of NO_3-N and

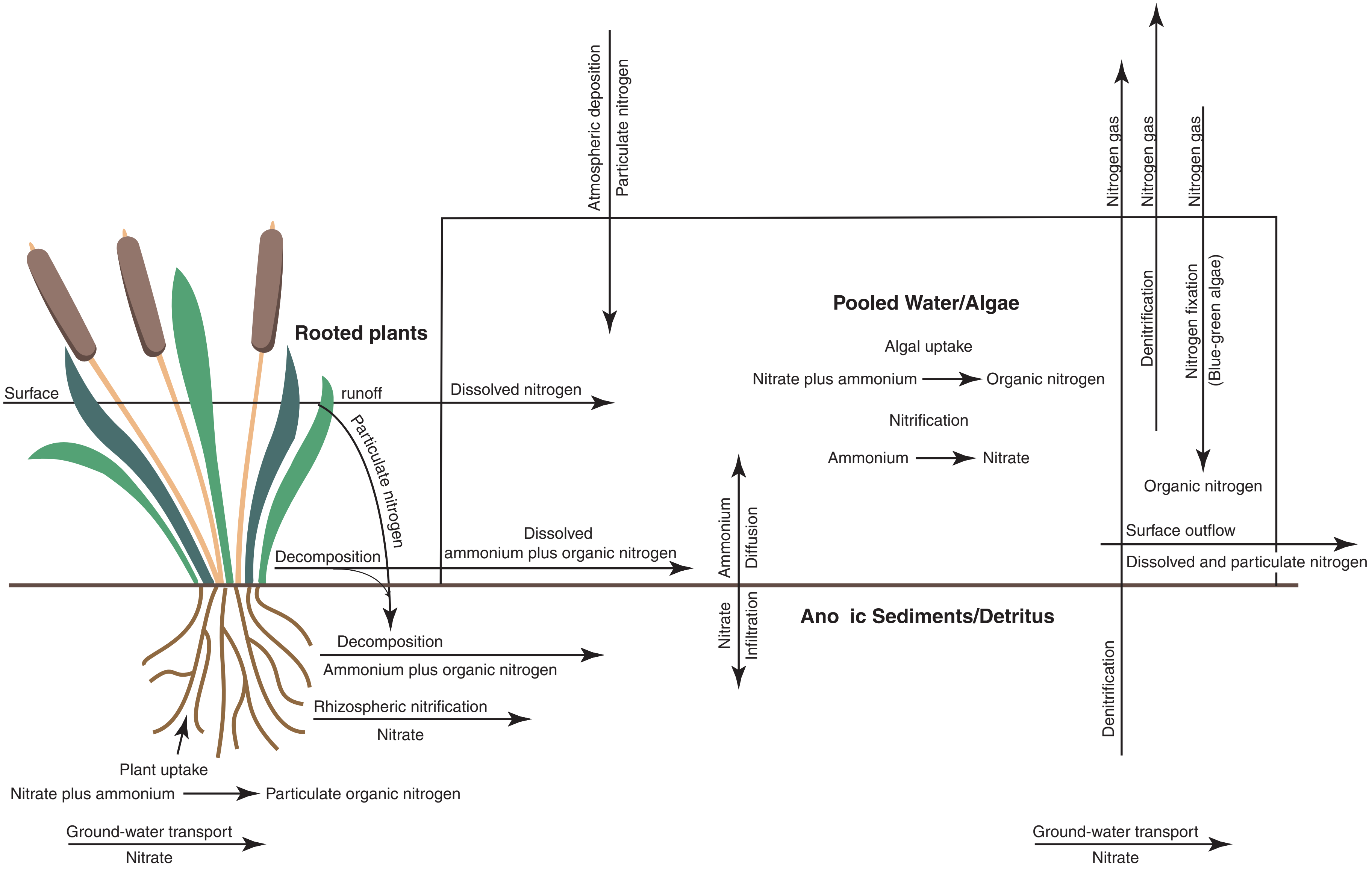


Figure 24. Conceptual compartments and dominant processes involving nitrogen in Little Bean Marsh.

Table 8. Loads of selected constituents in surface outflows from Little Bean Marsh.

[Bold figures are total ammonium (NH_4) estimated from ratio of total to dissolved phase of NH_4; m^3/s, cubic meters per second; g/d, grams per day; g/period, grams per period; kg, kilograms]

Site number (fig. 1)	Date	Mean discharge, in m^3/s	Period (days)	Total nitrogen, in g/d	Total dissolved nitrogen, in g/d	Total suspended nitrogen, in g/d	Total ammonium nitrogen, in g/d	Dissolved ammonium nitrogen, in g/d	Suspended ammonium nitrogen, in g/d	Nitrite plus nitrate, in g/d	Total ammonium plus organic nitrogen, in g/d	Dissolved ammonium plus organic nitrogen, in g/d
M5	11/15/1996	0.0054	17	651	465	186	160	93	67	63	588	402
M5	12/02/1996	.0003	17	21	12	9	5	0	5	0	21	12
M5	12/19/1996	.074	17	5,853	4,708	1,145	732	16	719	16	5,840	4,695
M5	01/03/1997	.013	15	1,093	518	575	426	3	423	3	1,090	515
M5	01/20/1997	.0048	17	337	229	108	1	1	0	1	336	228
M5	02/05/1997	.002	14	79	55	24	7	0	7	8	71	47
M5	02/17/1997	.000	8	0	0	0	0	0	0	0	0	0
M5	02/21/1997	.0037	3	188	99	89	19	1	18	8	179	90
M5	02/24/1997	.146	12	12,097	4,536	7,561	1,096	32	1,071	769	11,329	3,768
M5	03/17/1997	.057	18	573	206	367	9	1	8	1	572	205
M5	03/31/1997	.002	10	257	115	142	11	0	11	0	257	114
M5	04/05/1997	.002	6	231	106	125	0	1	0	0	231	106
M5	04/10/1997	.042	1	4,478	2,973	1,505	0	440	0	73	4,404	2,900
M5	04/11/1997	.747	3	51,031	43,280	7,752	0	3,294	0	4,974	46,058	38,306
M5	04/17/1997	.000	13	0	0	0	0	0	0	0	0	0
M5	05/05/1997	.000	17	0	0	0	0	0	0	0	0	0
M5	05/21/1997	.000	18	0	0	0	0	0	0	0	0	0
M5	06/10/1997	.0085	17	646	499	147	12	6	7	2	644	498
M5	06/25/1997	.018	15	1,256	1,161	95	159	4	156	4	1,253	1,158
M5	07/08/1997	.0300	16	1,245	2,179	0	44	23	21	6	1,240	2,173
M5	07/29/1997	.013	18	1,208	943	265	46	18	28	3	1,205	941
M5	08/12/1997	.001	15	148	131	17	14	10	4	0	148	131
M5	08/27/1997	.000	14	0	0	0	0	0	0	0	0	0
M5	09/10/1997	.000	15	0	0	0	0	0	0	0	0	0
M5	09/24/1997	.000	16	0	0	0	0	0	0	0	0	0
M5	10/14/1997	.0051	17	595	599	0	258	238	20	21	573	578
M5	10/28/1997	.002	18	123	126	0	19	10	9	6	117	120
M5	11/19/1997	.000	18	0	0	0	0	0	0	0	0	0
M5	12/02/1997	.001	14	94	66	28	2	1	1	1	93	65
M5	12/19/1997	.015	21	1,216	872	344	37	3	34	3	1,213	869

Table 8. Loads of selected constituents in surface outflows from Little Bean Marsh.—Continued

[Bold figures are total ammonium (NH_4) estimated from ratio of total to dissolved phase of NH_4; m^3/s, cubic meters per second; g/d, grams per day; g/period, grams per period; kg, kilograms]

Site number (fig. 1)	Date	Total organic nitrogen, in g/d	Dissolved organic nitrogen, in g/d	Total phosphorus, in g/d	Dissolved phosphorus, in g/d	Soluable reactive phosphorus, in g/d	Suspended phosphorus, in g/d	Chlorophyll mean, in g/d	Total suspended solids, in g/d	Volatile suspended solids, in g/d	Dissolved organic carbon, in g/d	Total nitrogen, in g/period
M5	11/15/1996	428	309	53	8	8	45	0	0	0	3,301	11,065
M5	12/02/1996	15	12	2	1	0	1	0	0	0	166	349
M5	12/19/1996	5,109	4,676	439	108	13	331	0	0	0	43,260	99,499
M5	01/03/1997	665	512	43	17	9	25	0	0	0	7,705	16,388
M5	01/20/1997	335	227	29	8	3	21	0	0	0	3,411	5,728
M5	02/05/1997	64	47	9	3	2	6	0	0	0	685	1,103
M5	02/17/1997	0	0	0	0	0	0	0	0	0	0	0
M5	02/21/1997	161	90	41	5	4	36	0	0	0	0	563
M5	02/24/1997	10,232	3,743	1,096	202	139	895	0	0	0	0	145,167
M5	03/17/1997	562	203	69	83	2	0	0	0	0	0	10,306
M5	03/31/1997	245	114	70	4	1	67	0	0	0	0	2,569
M5	04/05/1997	231	104	40	5	2	35	0	0	0	0	1,387
M5	04/10/1997	4,404	2,459	551	139	73	411	0	141,672	20,921	0	4,478
M5	04/11/1997	46,058	35,011	4,457	2,196	1,227	2,261	0	258,387	148,573	0	153,094
M5	04/17/1997	0	0	0	0	0	0	0	0	0	0	0
M5	05/05/1997	0	0	0	0	0	0	0	0	0	0	0
M5	05/21/1997	0	0	0	0	0	0	0	0	0	0	0
M5	06/10/1997	632	492	43	38	21	5	0	18,645	3,964	0	10,981
M5	06/25/1997	1,094	1,153	159	70	40	89	0	35,467	7,952	13,837	18,847
M5	07/08/1997	1,196	2,150	399	80	49	319	0	141,354	25,937	0	19,919
M5	07/29/1997	1,159	922	166	41	25	124	38,871	33,351	7,475	0	21,735
M5	08/12/1997	133	120	20	4	3	16	4,906	3,426	648	0	2,221
M5	08/27/1997	0	0	0	0	0	0	0	0	0	0	0
M5	09/10/1997	0	0	0	0	0	0	0	0	0	0	0
M5	09/24/1997	0	0	0	0	0	0	0	0	0	0	0
M5	10/14/1997	315	340	70	14	7	57	6,034	15,503	3,171	0	10,108
M5	10/28/1997	98	110	16	3	1	13	1,864	2,349	587	0	2,220
M5	11/19/1997	0	0	0	0	0	0	0	0	0	0	0
M5	12/02/1997	91	64	17	8	7	8	1,390	1,370	587	0	1,315
M5	12/19/1997	1,176	865	145	66	26	79	19,423	14,402	4,228	0	25,527
13-month total in kg												565
12-month total in kg												528

Table 8. Loads of selected constituents in surface outflows from Little Bean Marsh.—Continued

[Bold figures are total ammonium (NH_4) estimated from ratio of total to dissolved phase of NH_4; m^3/s, cubic meters per second; g/d, grams per day; g/period, grams per period; kg, kilograms]

Site number (fig. 1)	Date	Total dissolved nitrogen, in g/period	Total suspended nitrogen, in g/period	Total ammonium nitrogen, in g/period	Dissolved ammonium nitrogen, in g/period	Suspended ammonium nitrogen, in g/period	Nitrite plus nitrate, in g/period	Total ammonium plus organic nitrogen, in g/period	Dissolved ammonium plus organic nitrogen, in g/period	Total organic nitrogen, in g/period	Dissolved organic nitrogen, in g/period
M5	11/15/1996	7,903	3,161	2,719	1,581	1,138	1,067	9,998	6,836	7,279	5,256
M5	12/02/1996	204	146	91	1	90	1	349	203	258	202
M5	12/19/1996	80,032	19,467	12,437	270	12,221	270	99,282	79,815	86,845	79,491
M5	01/03/1997	7,763	8,625	6,383	43	6,348	43	16,353	7,728	9,971	7,676
M5	01/20/1997	3,889	1,839	18	18	0	18	5,714	3,875	5,692	3,854
M5	02/05/1997	767	336	103	6	98	106	998	662	894	657
M5	02/17/1997	0	0	0	0	0	0	0	0	0	0
M5	02/21/1997	296	267	56	2	54	25	538	271	482	269
M5	02/24/1997	54,437	90,729	13,156	378	12,853	9,224	135,942	45,213	122,787	44,911
M5	03/17/1997	3,700	6,606	167	22	150	22	10,289	3,682	10,121	3,656
M5	03/31/1997	1,145	1,424	115	4	112	4	2,566	1,142	2,452	1,138
M5	04/05/1997	637	750	**16**	8	**8**	3	1,385	635	1,385	627
M5	04/10/1997	2,973	1,505	**880**	440	**440**	73	4,404	2,900	4,404	2,459
M5	04/11/1997	129,840	23,255	**19,766**	9,883	**9,883**	14,922	138,173	114,918	138,173	105,034
M5	04/17/1997	0	0	0	0	0	0	0	0	0	0
M5	05/05/1997	0	0	0	0	0	0	0	0	0	0
M5	05/21/1997	0	0	0	0	0	0	0	0	0	0
M5	06/10/1997	8,486	2,496	212	100	112	31	10,956	8,461	10,744	8,361
M5	06/25/1997	17,415	1,431	2,386	60	2,338	60	18,799	17,368	16,413	17,296
M5	07/08/1997	34,859	0	705	373	332	104	19,836	34,776	19,131	34,402
M5	07/29/1997	16,974	4,761	828	331	497	52	21,694	16,933	20,866	16,602
M5	08/12/1997	1,964	257	217	154	62	5	2,217	1,960	2,000	1,806
M5	08/27/1997	0	0	0	0	0	0	0	0	0	0
M5	09/10/1997	0	0	0	0	0	0	0	0	0	0
M5	09/24/1997	0	0	0	0	0	0	0	0	0	0
M5	10/14/1997	10,183	0	4,388	4,043	344	359	9,749	9,823	5,361	5,780
M5	10/28/1997	2,273	0	341	188	153	108	2,111	2,164	1,771	1,977
M5	11/19/1997	0	0	0	0	0	0	0	0	0	0
M5	12/02/1997	918	397	27	15	12	14	1,302	904	1,274	889
M5	12/19/1997	18,313	7,214	777	69	721	69	25,472	18,258	24,695	18,174
13-month total in kg		405	175	**66**	18	**48**	27	538	379	493	361
12-month total in kg		379	164	**62**	16	**46**	25	502	353	461	337

Table 8. Loads of selected constituents in surface outflows from Little Bean Marsh.—Continued

[Bold figures are total ammonium (NH_4) estimated from ratio of total to dissolved phase of NH_4; m^3/s, cubic meters per second; g/d, grams per day; g/period, grams per period; kg, kilograms]

Site number (fig. 1)	Date	Total phosphorus, in g/period	Dissolved phosphorus, in g/period	Soluable reactive phosphorus, in g/period	Suspended phosphorus, in g/period	Chlorophyll mean, in g/period	Total suspended solids, in g/period	Volatile suspended solids, in g/period	Dissolved organic carbon, in g/period
M5	11/15/1996	901	142	142	759	0	0	0	56,114
M5	12/02/1996	30	9	0	21	0	0	0	2,829
M5	12/19/1996	7,462	1,839	216	5,624	0	0	0	735,425
M5	01/03/1997	638	259	138	380	0	0	0	115,577
M5	01/20/1997	495	141	57	354	0	0	0	57,985
M5	02/05/1997	127	43	24	84	0	0	0	9,592
M5	02/17/1997	0	0	0	0	0	0	0	0
M5	02/21/1997	124	16	13	108	0	0	0	0
M5	02/24/1997	13,156	2,419	1,663	10,736	0	0	0	0
M5	03/17/1997	1,242	1,497	35	0	0	0	0	0
M5	03/31/1997	705	44	15	667	0	0	0	0
M5	04/05/1997	242	32	13	210	0	0	0	0
M5	04/10/1997	551	139	73	411	0	141,672	20,921	0
M5	04/11/1997	13,372	6,589	3,682	6,783	0	775,161	445,718	0
M5	04/17/1997	0	0	0	0	0	0	0	0
M5	05/05/1997	0	0	0	0	0	0	0	0
M5	05/21/1997	0	0	0	0	0	0	0	0
M5	06/10/1997	736	649	349	87	0	316,965	67,386	0
M5	06/25/1997	2,386	1,050	596	1,336	0	532,006	119,284	207,554
M5	07/08/1997	6,391	1,286	788	5,104	0	2,261,671	414,985	0
M5	07/29/1997	2,981	745	455	2,236	699,671	600,310	134,552	0
M5	08/12/1997	294	61	40	233	73,589	51,384	9,726	0
M5	08/27/1997	0	0	0	0	0	0	0	0
M5	09/10/1997	0	0	0	0	0	0	0	0
M5	09/24/1997	0	0	0	0	0	0	0	0
M5	10/14/1997	1,198	232	120	966	102,577	263,555	53,909	0
M5	10/28/1997	293	61	24	233	33,561	42,282	10,570	0
M5	11/19/1997	0	0	0	0	0	0	0	0
M5	12/02/1997	233	118	99	115	19,457	19,183	8,221	0
M5	12/19/1997	3,052	1,387	555	1,665	407,885	302,445	88,791	0
13-month total in kg		57	19	9	38	1,337	5,307	1,374	1,185
12-month total in kg		53	17	8	36	929	5,004	1,285	1,126

NH_4-N (inorganic N) to Little Bean Marsh in 1997 was approximately 437 kg; which was greater than 70 percent of the N leaving Little Bean Marsh in surface outflows. The 437 kg computation does not include organic N or N in dry atmospheric deposition.

Lawrence and others (2000) reported that the dry deposition rate of NO_3-N and NH_4-N usually is about one-half of the wet deposition rate, and that the dominant form of dry N deposition in southern Minnesota, Iowa, and northern Missouri is NH_x, which originates primarily from senescing crops within 50 km of deposition sites. The closest dry deposition monitoring station to Little Bean Marsh was a U.S. Environmental Protection Agency Clean Air Status and Trends site at Alhambra in western Illinois (http://www.epa.gov/castnet, accessed 2000). The average annual NO_3 and NH_4 deposition rate at this site was 2.0 kg/ha during 1989 to 1999 and 1.7 kg/ha in 1997. When organic N and loadings from dry deposition are added to wet deposition of inorganic N, it is likely that atmospheric deposition alone could account for more than the entire 530 kg of N leaving the marsh in surface outflow. Consequently, it is likely that Little Bean Marsh removes much inorganic N, probably through denitrification.

Contributions from Storm Runoff

Storm loads from four storm events were measured at three sites in the spring of 1997. Loads were measured from two storms at site I1. These storms contributed 64,000 m^3 of surface runoff to Little Bean Marsh (table 2), which was approximately 12 percent of the total surface runoff into the marsh in 1997. These four storm events contributed approximately 381 kg of N to Little Bean Marsh. If the loading of total N from these storms is prorated to the total annual amount of surface runoff (516,000 m^3), then surface runoff contributed several times the amount of N that was released in surface outflow from the marsh. When the storm runoff loads of N are added to the amount of N contributed by direct precipitation, the amount of N contributed to Little Bean Marsh from storms is almost seven times more than the amount of N leaving the marsh in surface outflow. Unlike N speciation in the marsh and in the surface outflow, a substantial amount, roughly one-third, of the N in storm loads was in the form of NO_3. Another one-third was dissolved organic nitrogen, and the remaining one-third was suspended nitrogen that probably also is in the organic form. Little of the N in storm runoff entering the marsh was in the NH_4 form. The small concentrations of NO_3 in Little Bean Marsh and the small loads of NO_3 leaving the marsh are strong evidence of substantial N removal.

Ground-Water Contributions

Ground-water seepage usually is a small nutrient source to freshwater marshes (Mitsch and Gosselink, 1993). Even though the marsh receives substantial quantities of ground-water seepage, Little Bean Marsh receives little N from ground water because ground water has small concentrations of N (table 1). Nineteen of 21 ground-water samples collected from 5 monitoring wells around Little Bean Marsh (fig. 1) had concentrations of NO_3 less than 0.02 mg/L in 1996–1997. Ziegler and others (1993) determined that more than one-half of the shallow wells in the Missouri River alluvium had concentrations of NO_3 less than the detection limit of 0.05 mg/L. Ammonium (NH_4-N) concentrations in Little Bean Marsh wells averaged only 0.05 mg/L, but total dissolved N averaged 0.3 mg/L. Most of the dissolved N likely was organic N that entered ground water from the marsh during ground-water recharge periods. Ground water away from the marsh would likely have lesser concentrations of organic N. Using a net ground-water seepage of 157,400 m^3 (fig. 7) in 1997 and assuming total dissolved N concentrations in ground water averaged as high as 0.3 mg/L, ground-water seepage would have contributed less than 1 percent of the total N leaving Little Bean Marsh in surface outflow. However, NO_3 concentrations have been shown to decrease as ground water moves through anoxic lake-bottom sediments (Chen and others, 1972). Consequently, the anoxic sediments of the marsh likely remove NO_3 that may be present in ground water as it moves into the marsh, thus providing another pathway for N removal in the riparian corridor. Therefore, most wetlands along the Missouri River probably receive only small amounts of N from ground water.

Reduction of NO_3 to NH_4 can occur under intensely reducing conditions, as bacteria can rapidly immobilize and then ammonify N (Chen and others, 1972), but this process generally is insignificant compared to denitrification (Reddy and others, 1978) which actually removes N from the marsh. Reddy and others (1980) determined that 92 to 96 percent of isotopically labeled NO_3-N escaped as gaseous N in laboratory incubations lasting 21 days, leaving 4 to 8 percent as organic N and NH_4. Several authors have noted decreases in NO_3-N in water passing through wetlands (Kibby, 1978); Little Bean Marsh shares this characteristic, as NO_3 concentrations in the surface outflow commonly were less than 0.005 mg/L.

Nitrogen fixation can supply about one-half of the total N input to a wetland wholly dependent on precipitation as a source of water. Asymbotic N fixation in a *Scirpus atrovirens* marsh in Massachusetts ranged from 0.4 to 2.0 $g/m^2/yr$ (grams per square meter per year) (Kana and Tjepkema, 1978), and a rate of 1.8 $g/m^2/yr$ was measured *in situ* in a *Typha latifolia* stand in Minnesota (Biesboer, 1984). However, N inputs from fixation generally are small compared to those from surface-water inflows, and are not a substantial source of N to downstream waters (Johnston, 1991). Also, the macrophytes that dominate Little Bean Marsh do not symbiotically fix N, but do shade out floating N fixers such as bluegreen algae. Even if the greater fixation rates detected in the Massachusetts and Minnesota marshes are assumed for Little Bean Marsh, the total amount of N supplied to the marsh by fixation would be less than 0.5 percent of the amount of total N escaping the marsh in surface outflows. Therefore, N fixation rates, which commonly vary widely over short distances, were not measured in this study.

Whereas the input-output analysis in this report treats the whole marsh as a storage compartment, it is useful to review the wetland interdistributions and fluxes of N in other *Scirpus/Typha* wetlands and compare their inputs and outputs with Little Bean Marsh. Standing stocks of *Scirpus fluviatilis* were 0.066 kg/ha in an Iowa prairie pothole (Davis and Van der Valk, 1978) and 0.154 kg/ha in a Wisconsin wetland (Klopatek, 1978). Both of these densities are less than the mean (2.07 kg/ha) of all nonwoody emergent vegetation in 30 wetlands around the United States (Johnston, 1991). The above-ground part of N in the Wisconsin wetland was about three times that present in the roots and rhizomes, and two times that present in the litter, for a total of 28 g/m^2 (grams per square meter) (Klopatek, 1978). At this rate, the amount of plant material at Little Bean Marsh was about 20,000 kg of N, which is much larger than the annual loss or gain of N to the marsh from inflows or outflows. Consequently, much more of the N in Little Bean Marsh is internally recycled every year than is lost to surface outflows. Perhaps *Scirpus fluviatilis* has adapted to the low inorganic N content of Little Bean Marsh by recycling N. Annual leaching fluxes of N in *Scirpus fluviatilis* were measured as 1.49 g/m^2/yr, and annual litterfall fluxes were 8.09 g/m^2/yr in a study by Davis and Van der Valk (1978). Klopatek (1978) reported an annual leaching rate of 7.34 g/m^2 for *Scirpus fluviatilis,* which is 42 percent of the total uptake in the Wisconsin wetland. Applying these leaching rates to Little Bean Marsh would indicate that several times the amount of N were leached to the marsh that was lost in surface outflow in 1997. Because most leached inorganic N is rapidly oxidized to NO_3 and subsequently lost to denitrification, macrophytes in Little Bean Marsh likely facilitate the removal of soil N from the marsh. The largest concentrations of total N occurred during the September to early October period of senescence (figs. 19 and 20). However, *Scirpus fluviatilis* is slow to decay (Whitley and others, 1990) and in the Iowa pothole, *Scirpus fluviatilis* retained more than 50 percent of its dry weight after 525 days of decomposition. Klopatek (1978) determined that 26 percent of the N uptake by *Scirpus fluviatilis* was retained during winter months, with the rest lost to leaching and litterfall. Consequently, much of the N bound in *Scirpus fluviatilis* is prone to burial as evidenced by the large quantity of N in bottom sediments of Little Bean Marsh (table 9). Even though most (95 to 99 percent) of the nutrients used by emergent macrophytes are from the soil rather than the overlying water (Wetzel, 1983), the amount of N brought back into the water column by emergent vegetation would likely be a tiny fraction of the amount in the soil. The ratio of NH_4-N to total N in the bottom sediments would indicate that decomposition in bottom sediments is slow, and burial of N in bottom sediments also may be a substantial sink of N, despite the ability of Little Bean Marsh to remove inorganic N through denitrification. Once vegetation is buried in anoxic sediments below the rhizospheric root zone, decomposition would be slow and organic N would be more stable.

Soils contain, by far, the largest stocks of nutrients of any wetland storage compartment (Johnston, 1991). In a *Scirpus fluviatilis* marsh in Wisconsin, soils between 3 and 15 cm deep contained two orders of magnitude more N than that present in the *Scirpus fluviatilis* alone (Klopatek, 1978). The mean total N concentration in the top 15 cm of bottom sediments collected from four locations in Little Bean Marsh was 1,230 parts per million (table 9), which is near the low end of the range of total N concentrations in the mineral soils of six wetlands in Wiscon-

Table 9. Concentrations of selected nutrients with depth in bottom sediment cores collected from Little Bean Marsh in 1997.

[cm, centimeters; ppm, parts per million; >, greater than]

Site (fig. 1)	Depth, in cm	Ammonium nitrogen, in ppm	Nitrate nitrogen, in ppm	Total nitrogen, in ppm	Available phosphorus, in ppm
M1	0–4	9.1	1.7	1,053	20
M1	4–10	8.4	.5	644	24
M1	10–21	26.3	3	2,084	24
M1	21–35	12.9	2.4	1,718	27
M2	0–8.5	16	.3	580	35
M2	8.5–16	25	.3	1,257	29
M2	16–18	36.9	1	1,775	26
M3	0–3	59.4	4.1	3,900	22
M3	3–8	37.4	1	1,517	27
M3	8–16.5	51	.5	1,470	29
M3	>16.5	20.6	1.7	1,223	15
M5	0–6	40.6	.4	835	26
M5	6–12.5	35.7	.4	494	31
M5	12.5–19	67	.4	1,736	25
M5	19–29.5	55.8	2.5	1,897	27

sin, North Dakota, and North Carolina [range = 900 to 11,000 mg/kg (milligrams per kilogram); Johnston, 1991]. However, assuming a bottom-sediment density of 1.5 and a mean concentration of total N in bottom sediments of 1,230 parts per million, the top 15 cm of Little Bean Marsh contains 470 kg/ha of total N. This amount of N is still more than one order of magnitude greater than the amount of N entering the marsh in surface runoff, and more than two orders of magnitude greater than the amount of N in surface outflow during 1997. Consequently, burial of organic N is likely a substantial sink of N; however, most (95 to 99 percent) of the nutrients used by emergent macrophytes are from the soil, rather than the overlying water (Wetzel, 1983) so much of the N feeding emergent vegetation is recycled. Most of the bottom-sediment N exists in the organic form (table 9). NH_4-N concentrations in bottom sediments were minimal probably because of plant uptake and nitrification in rhizospheres. However, NH_4-N concentrations were significantly correlated with total N concentrations (Spearman's rho = 0.54, p = 0.022), indicating some new NH_4 was being supplied through slow decomposition of organic N. The extremely low concentrations of NO_3 could be expected, considering the reducing conditions present in all but the rhizospheric parts of the bottom sediments. No detectable trend in N with depth can be discerned from the data in table 9 and, surprisingly, no clear differentiation in total N concentration can be made between open-water sites (M3 and M5) and vegetated sites (M1 and M2).

Because of the large percentage of N in emergent vegetation, macrophytes can increase nutrient concentrations at senescence when nutrients are leached into the water column. In fact, time trend plots of all nondry marsh sites (M5, M4, and M3) had peak concentrations of nearly every N species during the senescing period of September and early October (figs. 19 and 20). Consequently, this is a period in which nutrients are vulnerable to escape. However, this period usually follows a period of low water levels, which also can prevent escape of nutrients in surface outflows, as happened at Little Bean Marsh in 1997.

Denitrification

In the absence of oxygen, many soil and water microbes can use NO_3 as an electron acceptor for respiration. This process converts NO_3-N to gaseous N_2O and N_2, and is thus described as denitrification (equation 6). Because anoxic conditions are prevalent in the inundated bottom sediments of Little Bean Marsh, widespread denitrification is possible. The chemical equations in the nitrification-denitrification system are:

$$\begin{array}{l} \text{Nitrification } 6NH_4^{+} + 12O_2 \rightarrow 6NO_3^{-} + 6H_2O + 12H^{+} \\ \underline{\text{Denitrification } 6NO_3^{-} + 1.25C_6H_{12}O_6 + 6H^{+} \rightarrow 3N_2 + 7.5CO_2 + 10.5H_2O} \\ \text{Combined } 6NH_4^{+} + 1.25C_6H_{12}O_6 + 12O_2 \rightarrow 3N_2 + 7.5CO_2 + 16.5H_2O + 6H^{+} \end{array} \qquad (6)$$

Measurements of Eh (oxidation/reduction potential) in bottom-sediment cores from Little Bean Marsh consistently were less than the 0.25-volt maximum Eh for denitrification (Mitsch and Gosselink, 1993) (fig. 25). Eh typically increased with depth, probably because of the increased organic matter and moisture content that was observed (not measured) near the top of the cores. Usually, the Eh ranged between 0.0 and 0.1 volts in the top 10 or 15 cm, and increased more rapidly at depths below 15 cm. Because the water column usually had concentrations of dissolved oxygen greater than 0.3 mg/L, a thin oxidized layer likely exists on top of the sediment column (Patrick and Reddy, 1976; Mitsch and Gosselink, 1993). Eh measurements were not usually made at depths less than 0.5 cm because of the size of the Eh and temperature probes used. Evidently, the oxidized layer usually was less than 0.5 cm thick at Little Bean Marsh as oxidizing conditions were detected only occasionally near the top of the sediment column (fig. 25).

Redox conditions in the open-water areas of Little Bean Marsh seem to follow a conceptual model of N cycling that has been observed in the laboratory (Patrick and Reddy, 1976) and seems to provide a large capacity for N removal. The thin oxidizing layer at the sediment/water surface and in rhizospheres oxidizes NH_4 to NO_3 and creates an NH_4 gradient within the reduced sediments, which allows NH_4 to diffuse upward toward the zone of nitrification. NO_3 then diffuses back down into the reduced sediments from the oxidized sediments and water where NO_3 is converted to N gases in the reduced sediments by denitrification, and is lost to the atmosphere. Diffusion of NO_3 into the sediments was shown to be the rate-limiting step in laboratory experiments and mathematical theory of denitrification in flooded soils by Reddy and others (1978) and Phillips and others (1978). Consequently, denitrification occurred at a first order rate (logarithmic with time) relative to soil-NO_3 concentration, typical of diffusion processes. NO_3 removal was faster in the absence of overlying water, and denitrification reverted to a zero-order rate (linear with time); however, increasing the water depth from 3 to 6 cm made little difference in NO_3 removal rates. In laboratory experiments conducted by Reddy and others (1980), NO_3 diffused less than 3 cm into bottom sediments after 96 hours and diffusion rates were only minimally affected by temperature. Soil denitrification increased the diffusion rate compared to diffusion rates into sterilized soil by increasing the concentration gradient across the soil water interface. These experiments were conducted in anoxic conditions and did not examine the rate of nitrification that is likely the rate-limiting step at Little Bean Marsh as evidenced by the very low concentrations of NO_3.

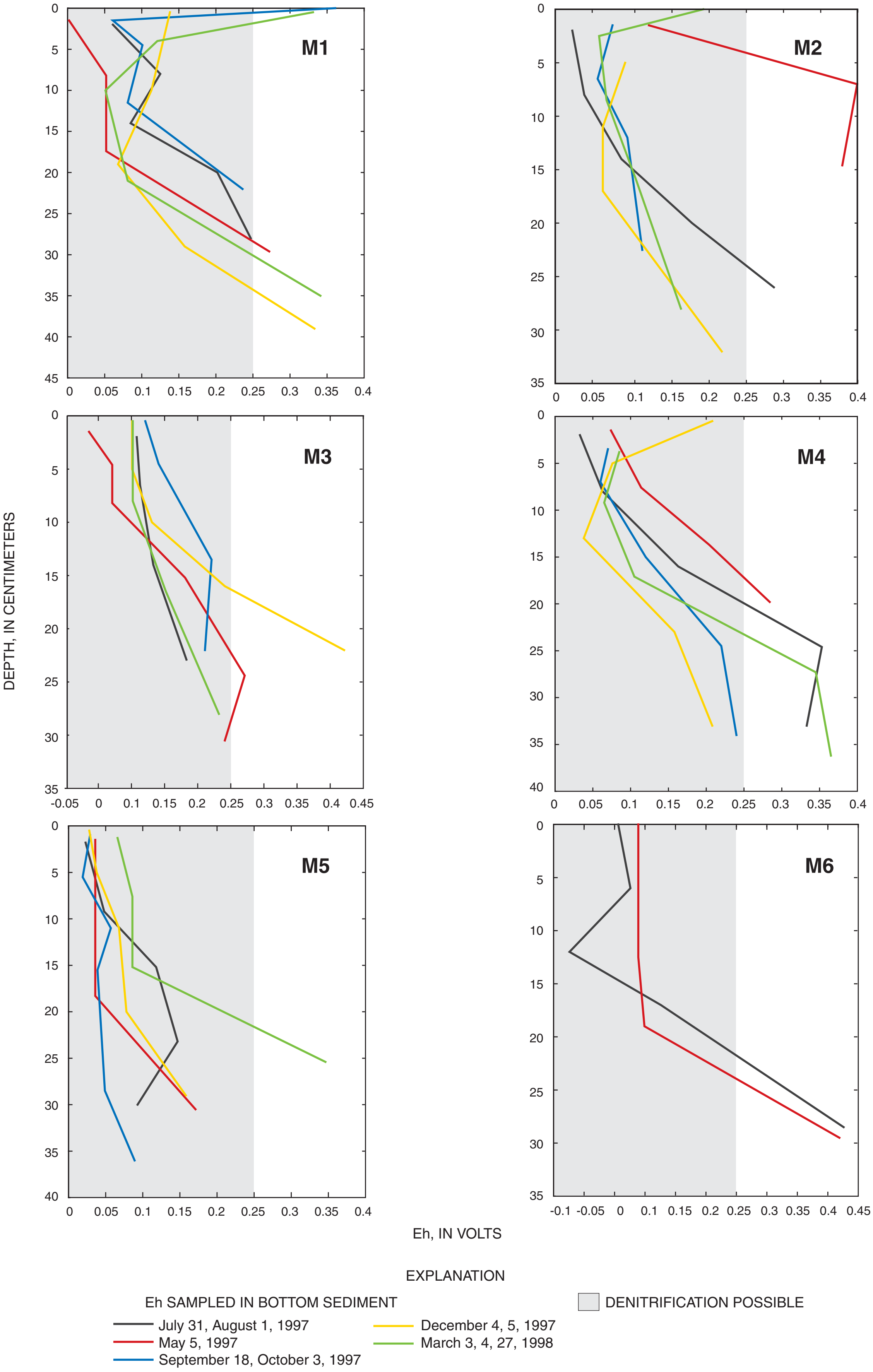


Figure 25. Eh in bottom sediment in relation to depth at six sites in Little Bean Marsh, 1997 and 1998.

Whereas the laboratory experiments described in the previous paragraph were designed to simulate open water systems, about 60 percent of Little Bean Marsh is covered with *Scirpus fluviatalis* and *Typha latifolia*. Bottom-sediment cores from macrophyte areas contained large quantities of rootlets that were easily identified by red ferric oxyhydroxide deposits that precipitated in the oxidizing rhizospheres around the rootlets. The oxidizing rhizospheres provide a massive intertwining of oxidizing and reducing zones, and many more chemical gradients over shorter distances. These rhizospheres are likely essential to the removal of soil N at depth. Phillips and others (1978) have shown that without rhizospheres, NO_3 is rapidly [about 60 μg/g (micrograms per gram) of soil per day where NO_3 was not limiting] removed from saturated soil, and is present only in the top few centimeters after a few days. Without rhizospheres, the only source of NO_3 is diffusion down from the water column. NH_4 also has to diffuse up into the surface oxidizing zone. Whereas emergent macrophytes bury large amounts of organic N in the form of root stocks and detritus, rhizospheric plants substantially increase the rate of N removal from bottom sediments by increasing the rate of organic N decomposition, increasing the rate-limiting step of nitrification in rhizospheres, decreasing the length of chemical diffusion gradients, and substantially increasing the number of gradients. Thus, removal of soil N from vegetated areas is likely much more rapid and efficient than from open water areas, because the amount of soil N generally is much larger than in the overlying water.

Denitrification is inhibited greatly or terminated when dissolved-oxygen concentrations exceed 0.3 mg/L, temperatures are below about 5 ^{o}C, or pH is less than 5 (Bremmer and Shaw, 1958; Stanford and others, 1975; Reddy and others, 1980; Broderick and others, 1988). Denitrification rates increase about 10-fold between 5 and 10 °C, double every 10 °C between 11 and 35 °C, and maximum denitrification rates occur at 60 to 65 °C (Bremmer and Shaw, 1958) and, therefore, occur at the fastest rate during the highest temperatures of the summer. At pH values less than 6, reduction of N_2O to N_2 is strongly inhibited (Wijler and Delwiche, 1954). The entire range of pH values measured (concurrently with Eh) in the cores from Little Bean Marsh was 6.1 to 7.3, which is greater than these denitrification-limiting values. Water pH was never less than 5 units (fig. 18), water temperatures were greater than 5 °C for about 9 months each year (fig. 14), and dissolved oxygen was less than 0.3 mg/L only for a month or less each summer (fig. 12) in Little Bean Marsh. Therefore, denitrification is strongly inhibited by dissolved oxygen in the water column for about 11 months each year, and most denitrification must occur in the bottom sediments.

The results of *in situ* denitrification experiments are presented in table 10 and summarized as means in figure 26. Apparently, the small ambient concentrations of NO_3 were the primary factor limiting spring, summer, and fall denitrification rates as all ambient incubations yielded little or no detectable N_2O. However, incubations with added NO_3 had large accumulations of N_2O, indicating that Little Bean Marsh has a large unused capacity to remove NO_3 from the water column. The rerouting of Short Creek around the marsh deprives the creek of this nutrient removal function. Thus, the bypassing of wetlands with drainage ditches in the flood plain may increase NO_3 loads to the Missouri River. The marsh's large capacity for denitrification also explains the low concentrations of NO_3 in the marsh. In other types of wetlands, nitrification has been determined to be the rate-limiting step in N removal. However, concentrations of dissolved NH_4 in Little Bean Marsh also were usually at or near the detection limit of 0.005 mg/L (fig. 19), indicating that NH_4 in the water column is rapidly nitrified or taken up by plants. Because about two-thirds of the marsh is covered by emergent macrophytes that derive most of their N from the soils and shade out algae that derive N from the water column, decomposition of organic N may be a more likely rate-limiting step than nitrification in the removal of N. Anoxic and acidic conditions in the bottom sediments probably limit both nitrification and decomposition rates (Chen and others, 1972).

The lack of denitrification in December when marsh temperatures averaged 6 °C verifies the findings of Broderick and others (1988) that winter denitrification rates approach 0 at approximately 5 °C. One might suspect that winter concentrations of NO_3 might increase as denitrification ceases. As previously described, ice formation also may serve to increase NO_3 concentrations in the underlying unfrozen water. However, only small increases in NO_3 occurred during the winters of 1997 and 1998 (fig. 20) with the exception of two increases in February and April 1997 that are associated with the two largest rainstorms. The seasonal pattern of inorganic N concentrations in Little Bean Marsh does not closely conform with the pattern of low concentrations in the growing season and higher concentrations in the fall and early spring suggested by Kibby (1978). The minimal increases in winter NO_3 concentrations may be explained by the fact that processes that generate NO_3, such as nitrification, decomposition, and surface-water loadings also slow down in winter. Although Reddy and others (1980) determined a laboratory Q_{10} value (the ratio of two denitrification rate constants 10 degrees apart) of 1.5 between 18 and 28 °C, a relation between temperature and potential denitrification rates in Little Bean Marsh at temperatures between 19 and 27 °C was not evident (fig. 27) indicating other factors control the process after a threshold temperature is reached. The decrease in N removal rates in winter can cause wetland management for water quality to be in conflict with management for waterfowl. Water levels commonly are raised in the late fall to attract migrating waterfowl and allowed to stay up through the spring migration, coinciding with the period of minimal N removal. Wetlands commonly are drained in the summer to allow vegetation growth for waterfowl, which is the period of maximum N removal. Without the overlying water, reducing conditions in the sediments are unlikely, and denitrification will cease. However, short periods of drying may increase rates of decomposition, and thus increase the NO_3 available for denitrification upon reinundation.

Dissolved-oxygen measurements made at the beginning of incubation ranged from 3.0 to 8.5 mg/L, and also had no apparent relation to potential denitrification rates (fig. 28). A dis-

Table 10. Amounts of denitrified nitrogen generated in *in-situ* experiments.

[mg/L/d, milligrams per liter per day; --, no data]

Date	Time	Ambient, in mg/L/d	Nitrate added, in mg/L/d	Time	Ambient, in mg/L/d	Nitrate added, in mg/L/d	Time	Ambient, in mg/L/d	Nitrate added, in mg/L/d
	Site M1			Site M2			Site M3		
06/24/1997	--	--	--	--	--	--	1445	Experiment initiated	
06/25/1997	1645	Experiment initiated		1600	Experiment initiated		1100	0.00	0.14
06/26/1997	1615	0.00	0.25	1500	0.00	0.00	1030	.00	.42
06/27/1997	1110	.00	1.36	1050	.00	.75	--	--	--
07/28/1997	--	--	--	--	--	--	1145	Experiment initiated	
07/29/1997	1400	Experiment initiated		1530	Experiment initiated		--	--	--
07/30/1997	1610	0.00	0.36	1550	0.23	1.19	1410	0.00	0.52
07/31/1997	1615	.68	.21	1550	.04	1.48	--	--	--
08/01/1997	1200	.97	1.01	--	.00	.00	--	--	--
09/08/1997	Dry	--	--	1335	Experiment initiated		--	--	--
09/09/1997	Dry	--	--	1315	0.00	1.31	--	--	--
09/10/1997	Dry	--	--	1400	.00	4.61	--	--	--
12/01/1997	--	--	--	--	--	--	1315	Experiment initiated	
12/02/1997	1030	Experiment initiated		1110	Experiment initiated		1315	0.00	0.00
12/03/1997	1130	0.00	0.00	1145	0.00	0.00	1030	.00	.00
12/04/1997	1300	.00	.00	1130	.00	.00	1330	.00	.12
04/21/1998	1450	Experiment initiated		--	--	--	--	--	--
04/22/1998	1230	0.00	0.17	1000	Experiment initiated		1200	Experiment initiated	
04/23/1998	1445	.00	.38	1400	0.00	0.41	1500	0.00	0.18
04/24/1998	1320	.00	.66	1340	.00	-.50	1130	.00	.36

Table 10. Amounts of denitrified nitrogen generated in *in-situ* experiments.—Continued

[mg/L/d, milligrams per liter per day; --, no data]

	Time	Ambient, in mg/L/d	Nitrate added, in mg/L/d	Time	Ambient, in mg/L/d	Nitrate added, in mg/L/d	Time	Ambient, in mg/L/d	Nitrate added, in mg/L/d
Date	**Site M4**			**Site M5**			**Site M6**		
06/24/1997	1420	Experiment initiated		1500	Experiment initiated		--	--	--
06/25/1997	1045	-0.04	-0.07	1115	0.00	0.00	1615	Experiment initiated	
06/26/1997	1210	.59	.85	1315	.00	.80	1515	0.00	0.00
06/27/1997	--	--	--	--	--	--	1040	.00	1.38
07/28/1997	1345	Experiment initiated		1400	Experiment initiated		1615	Experiment initiated	
07/29/1997	--	--	--	--	--	--	--	--	--
07/30/1997	1430	0.00	0.53	1445	0.00	0.53	1540	0.02	0.72
07/31/1997	1215	.00	2.75	1340	.00	2.16	1530	.00	.95
08/01/1997	--	--	--	--	--	--	--	.00	.00
09/09/1997	1120	Experiment initiated		1210	Experiment initiated		1345	Experiment initiated	
09/09/1997	1100	0.00	1.40	1200	0.00	0.43	1330	0.00	0.74
09/10/1997	1250	.00	2.30	1310	.00	5.03	1420	.00	1.65
12/01/1997	1235	Experiment initiated		1235	Experiment initiated		--	--	--
12/02/1997	1345	0.00	-0.03	1345	0.00	0.00	1145	Experiment initiated	
12/03/1997	1045	.00	-.07	1045	.00	.05	1200	0.00	0.00
12/04/1997	1400	.00	.24	1400	.08	.09	1000	.00	.00
04/21/1997	1200	Experiment initiated		1345	Experiment initiated		--	--	--
04/22/1998	1300	0.00	0.99	1330	0.00	0.12	1100	Experiment initiated	
04/23/1998	1515	.00	.19	1545	.00	.23	1400	0.00	0.00
04/24/1998	1200	.00	1.62	1220	.00	.28	1400	.00	--

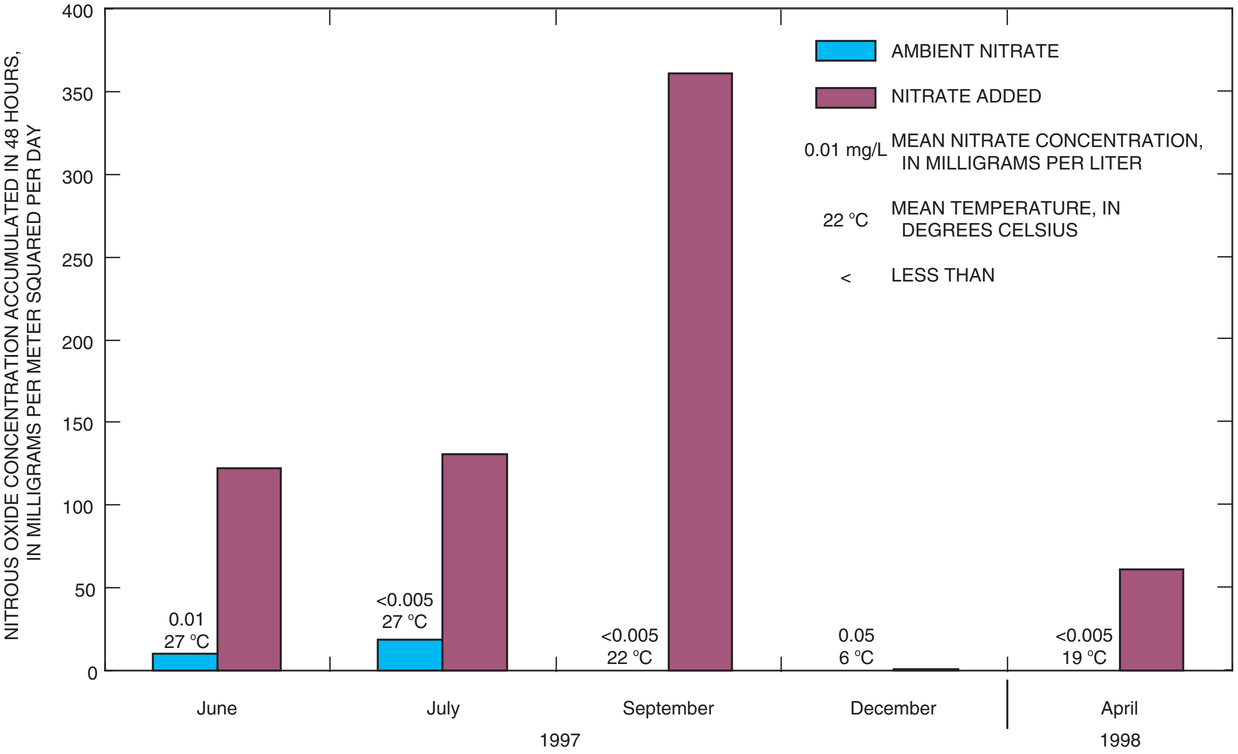


Figure 26. Concentrations of nitrous oxide in denitrification chambers after 48 hours showing differences between chambers with ambient concentrations of nitrate and those spiked with nitrate.

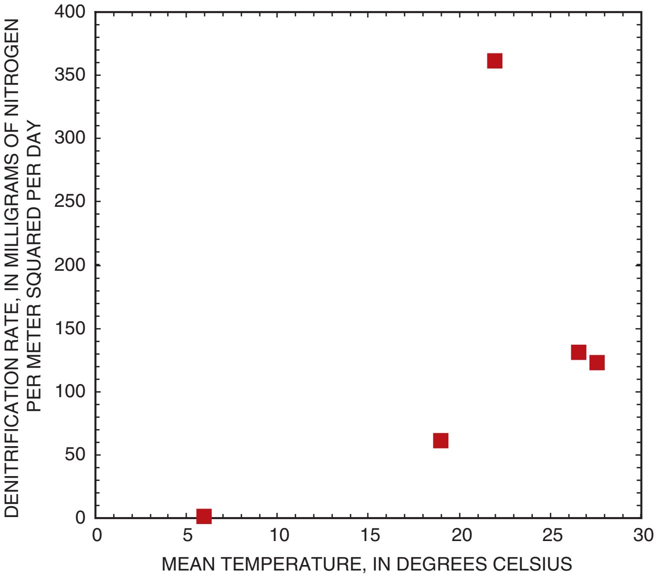


Figure 27. Denitrification rates in relation to temperature in denitrification chambers spiked with nitrate in Little Bean Marsh.

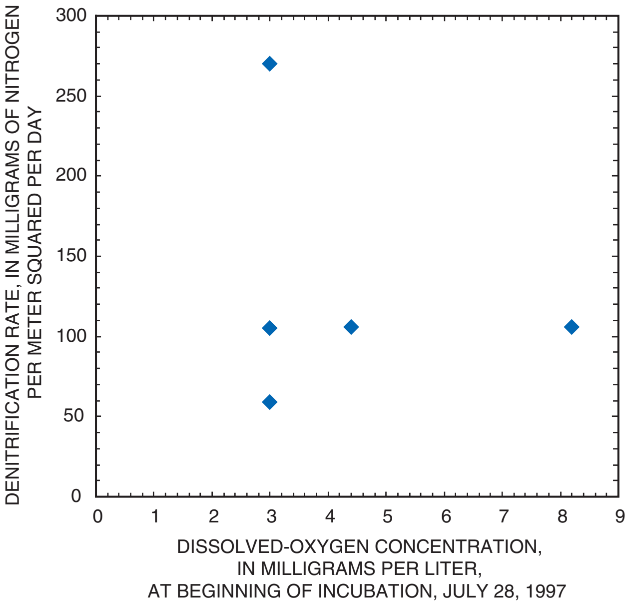


Figure 28. Denitrification rate in relation to dissolved-oxygen concentration in denitrification chambers spiked with nitrate in Little Bean Marsh on July 28, 1997.

solved-oxygen concentration of 0.3 mg/L is sufficiently oxidizing to limit denitrification in the water column (Broderick and others, 1988). Therefore, a relation with potential denitrification rates was not expected.

Although evidence for rapid denitrification in other types of wetlands is common (Brinson and others, 1984), it is not universal. Therefore, identification and characterization of denitrification in Little Bean Marsh is important for the management of water quality, wetland restoration, and flood plains in the Missouri River riparian zone.

Phosphorus Cycling

Compartments and processes involving P in wetlands are numerous and complex (fig. 29). However, sediment deposition in wetlands with mineral soils can be a primary mechanism for removing nutrients from surface water. Measurements of nutrient sedimentation fluxes in 11 wetland studies surveyed by Johnston (1991) averaged 15 grams N per square meter per year and 1.5 grams P per square meter per year. Mitsch (1978) determined that 15.7 grams P per square meter were removed in 1 year in a cypress swamp in southern Illinois, and that 3.6 grams P per square meter were from a single flood event. Less than 10 percent of the P input from all sources was released to the adjacent Cache River. However, in subsequent years, Dortch (1996) reported much lower removal rates for the Cache River wetland. Sorption and desorption of P has been reported to be substantial in wetlands with mineral soils (Dierberg and Brezonik, 1985). Phosphorus adsorption potential is best predicted by the presence of amorphous aluminum and iron minerals in the soil (Nichols, 1983; Richardson, 1985). The presence of these compounds results in more PO_4 being sorbed where dissolved PO_4 concentrations are high, and more PO_4 being solubilized where dissolved PO_4 concentrations are low under anaerobic conditions (Patrick and Khalid, 1974; Holford and Patrick, 1981). The abundant supply of dissolved iron available to Little Bean Marsh from ground water (Ziegler and others, 1993) should provide a large capacity for sorption and desorption of P. Indeed, coatings of iron on the marsh bottom commonly were observed on the north side of the marsh where ground-water seepage was most prevalent.

Several mechanisms, both aerobic and anoxic, can resuspend or redissolve P into the water column. Wind is one of these mechanisms. However, Carper and Bachmann (1984) determined that in a 104-ha Iowa lake (Little Wall Lake) with an average depth of 1.3 m, wind was rarely strong enough to resuspend mineral sediments from the bottom. Winds of 30 km/h (kilometers per hour) were required to produce waves that “felt” the lake bottom. Although the open-water depths in Little Bean Marsh (approximately 0.75 m) are less than Little Wall Lake, the fetch, and thus the wave heights, also were much less. The lack of fetch primarily is caused by macrophytes that break up open water and protect large areas from wind. Therefore, a wetland would trap sediment more efficiently when open-water areas are small, and macrophytes dominate the area near the surface outflow point of the wetland. The outflow area in Little Bean Marsh was an area of open water.

Macrophyte leaching is another source of aerobic P release. Klopatek (1978) reported that 58 percent of P taken up by *Scirpus fluviatilis* was leached back to the water.

Correlations

The P-trapping efficiency of a wetland usually is determined by P release from the sediments to the water column where P can then escape the wetland in surface outflow. As previously described, several aerobic and anaerobic processes can release P. Therefore, concentrations of P were correlated with wind speed, dissolved oxygen, pH, and turbidity to identify the primary mechanisms for P release from sediments and subsequent escape from the marsh.

Whereas total P and PO_4 concentrations were not significantly correlated with wind speed or pH at any site, P concentrations were significantly correlated with mean daily dissolved oxygen at open-water sites M3 and M5 ($p<0.05$) (table 11). Total P at site M2 also was significantly correlated with mean daily dissolved oxygen at site M3 ($p<0.05$). The significance of these correlations was somewhat unexpected because mean daily dissolved oxygen was measured only at site M3. Apparently, low dissolved oxygen values at site M3 are indicative of widespread anoxia or near anoxia over large parts of the marsh. These correlations indicate that anoxic conditions have a larger effect on P concentrations than other variables such as wind speed and pH. However, anoxic conditions usually were limited to low-water periods in 1997 (figs. 8 and 12) that prevented large quantities of P from escaping the marsh in surface outflows. Annual climatic patterns in the Midwest likely make this a common scenario in Missouri wetlands. Although surface outflows during periods of P release could cause eutrophication problems downstream, high P concentrations in wetlands are most likely during low-water periods, when wetlands are not discharging surface water and, therefore, P can be retained by the wetlands.

Total P was significantly correlated ($p<0.1$) with chlorophyll concentrations at partially open-water site M2 and open-water site M5 only (table 12), indicating that in open-water areas, most P is tied up in algae similar to most reservoirs in the region (Jones and Knowlton, 1993). Conversely, most P is not tied up in algae where emergent plants shade the water. The highest concentrations of total P occurred during a September algal bloom at open-water sites (table 3). This bloom occurred during macrophyte senescence and soon after an extended period of low dissolved oxygen and low water levels in the marsh (figs. 8 and 12). High summer temperatures likely accelerated decomposition that consumed oxygen and caused large areas of anoxic conditions in the marsh, which released P into the water that contributed to the algal bloom. The bloom and the period of low dissolved oxygen ended in early October with a sudden decrease in temperature (figs. 14 and 15) and more than

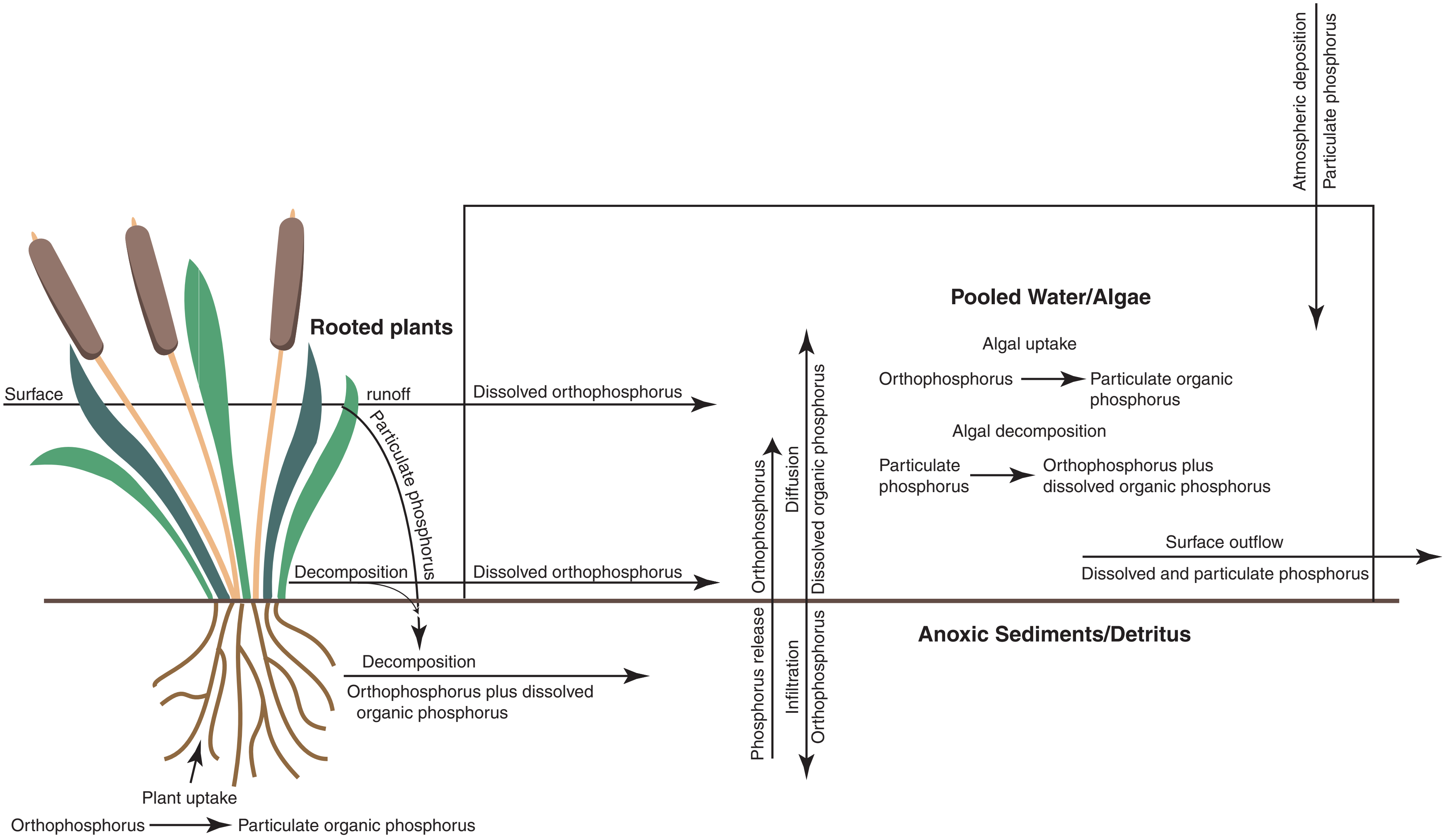


Figure 29. Conceptual compartments and dominant processes involving phosphorus in Little Bean Marsh.

Table 11. Spearman's rho values and levels of significance (p) for correlation of phosphorus with mean daily dissolved oxygen at five sites in Little Bean Marsh. Mean-daily dissolved oxygen was measured at site M3.

Site (fig. 1)	Total phosphorus		Soluble reactive phosphorus (PO_4)	
	rho	p	rho	p
M1	-0.38	0.15	+0.19	0.68
M4	-.06	.42	-.38	.11
M2	-.72	.0005	-.14	.26
M3	-.54	.0027	-.35	.034
M5	-.68	.0002	-.44	.012

Table 12. Spearman's rho values and levels of significance (p) for correlation of phosphorus with chlorophyll concentrations at five sites in Little Bean Marsh.

Site (fig. 1)	Total phosphorus		Soluble reactive phosphorus (PO_4)	
	rho	p	rho	p
M4	-0.39	0.84	-0.25	0.75
M6	.71	.063	-.03	.55
M2	.71	.018	-.40	.89
M3	.44	.095	.35	.15
M5	.76	.002	.13	.35

7.5 cm of rain. Soluble reactive P had no significant correlations ($p>0.1$) with chlorophyll at any site (table 12).

Sinks and Sources of Phosphorus

Approximately 53 kg of total P left Little Bean Marsh in surface outflows in 1997 (table 8). Dividing this value by the 952 ha drainage area gives an areal yield of 0.056 kg/ha, which is less than one-half of the 0.12 kg/ha areal yield of the Missouri River at St. Joseph, Missouri, in 1997 (Hauck and others, 1998) and less than one-third of the 1980 to 1996 average P yield of 0.19 for the entire Missouri River Watershed (Goolsby and others, 1999). Most P in Little Bean Marsh was in the particulate phase (table 3); the majority of the dissolved P usually was PO_4, although nonreactive (organic) P was a substantial part. The largest P loads leaving the marsh occurred in response to the two largest storms in February and April 1997 (table 8). Even though Little Bean Marsh removed a large percentage of the P it received, concentrations of available soluble reactive P in the outflows were large enough to stimulate phytoplankton growth in a hypothetical P-limited water body downstream (Horne and Goldman, 1994).

Concentrations of P in rainfall generally are less than those present in the Missouri River (Hauck and others, 1999) or in runoff from its tributaries in Missouri and Kansas. However, because precipitation constituted 41 percent of the inflow to Little Bean Marsh and other sources of P were relatively low, precipitation was an important source of P to Little Bean Marsh. P data are not collected by the National Atmospheric Data Program and, consequently, wet deposition data for P are difficult to obtain. At 28 nonurban locations in the contiguous 48 states, atmospheric deposition of P ranged from 0.05 to 1.1 kg/ha/yr (kilograms per hectare per year), and averaged 0.4 kg/ha/yr (Johnston, 1991). Orthophosphorus (PO_4) concentrations in precipitation measured at Centralia, Missouri, by the Agricultural Research Service (Alberts and others, written commun., 1993) averaged 0.007 mg/L, or about 0.07 kg/ha, in 1997. Because particulate and organic P concentrations are expected to be small in precipitation, PO_4 concentrations are likely good estimates of total P in precipitation. A rate 0.07 kg/ha from direct precipitation into Little Bean Marsh would result in about 3 kg of P input into Little Bean Marsh in 1997, which is less than 6 percent of the P leaving the marsh in surface outflow. The amount of dry deposition is not known, but when added to wet deposition, would be expected to substantially increase the amount of total atmospheric deposition.

Storm loads were measured from four storms at three sites during the spring of 1997. Storm loads were measured from two storms at site I1 (fig. 1). These storms contributed 64,000 m^3 of surface runoff to Little Bean Marsh (table 2), which is approximately 12 percent of the total surface runoff into the marsh during 1997. These four storm events also contributed approximately 136 kg of P to Little Bean Marsh. Eighty-four percent of the 136 kg of P was attached to suspended sediment. If one prorates the loading of total P from these four storms to the total amount of surface runoff (516,000 m^3) during 1997, it is apparent that surface runoff contributes more than one order of magnitude more P than was released in surface outflow from the marsh. Little Bean Marsh evidently traps P-laden sediments very efficiently as the long hydraulic residence time would indicate (Moustafa, 1997).

Ground-water seepage usually is a small source of nutrients to freshwater marshes (Mitsch and Gosselink, 1993). The median concentration of total dissolved P in 21 ground-water samples collected from five monitoring wells near Little Bean Marsh (fig. 1) was less than 0.02 mg/L (table 1). Nearly all of the P in ground water was in PO_4 form and is, therefore, readily available to phytoplankton. Samples from well W4 periodically had abnormally high concentrations of dissolved P. Using a net ground-water seepage of 157,000 m^3 in 1997 (fig. 7) and assuming total P concentrations of 0.02 mg/L, ground-water seepage would have contributed less than 8 percent of the total P leaving Little Bean Marsh in surface outflow and less than 1 percent of the amount of P received in storm flows. Therefore, ground water probably is a minor source of P to wetlands along the Missouri River that receive at least minimal amounts of storm runoff.

The mean concentration of iron in samples from the Missouri River alluvial aquifer in northwest Missouri was 8 mg/L (Ziegler and others, 1993). Iron in ground-water samples collected from nine wells at Little Bean Marsh at approximately monthly intervals between September 27, 1996, and April 13, 1997, ranged from 1.0 to 18 mg/L (Walter, 1998). At 8 mg/L, the net 157,000 m^3 of ground water received by Little Bean Marsh in 1997 provided approximately 1,200 kg of iron that usually would be oxidized to form a large and relatively constant source of sorption sites for P, metals, and other hydrophobic compounds. The P immobilization capacity of Little Bean Marsh would be expected to be substantial as long as the marsh was oxic. Therefore, the source of wetland water can have a substantial effect on P retention. However, the frequent lack of detectable NO_3 in Little Bean Marsh means that Fe^{+3} is the next likely electron acceptor during anoxic conditions. Decomposition and other oxygen-consuming processes can reduce ferric to ferrous iron and allow release of bound P, regardless of the amount of iron available.

Soils contain, by far, the largest stocks of P of any wetland storage component (Johnston, 1991). The mean concentration of total P in the top 15 cm of bottom sediments collected from four locations (M1, M2, M3, and M5) in Little Bean Marsh is 590 parts per million (table 9), which is near the low end of the range of total P concentrations in the mineral soils of six wetlands in Wisconsin, North Dakota, and North Carolina (ranging from 220 to 1,500 mg/kg, Johnston, 1991). However, assuming a bulk bottom-sediment density of 1.5 g/cm^3 (grams per cubic centimeter) and a mean concentration of total N in bottom sediments of 590 parts per million (table 9), the top 15 cm of Little Bean Marsh contained 1,500 kg of P. This amount of P is more than one order of magnitude greater than either the amount of P entering the marsh in surface runoff or the amount of P in surface outflow during 1997.

Annual leaching fluxes of P from *Scirpus fluviatilis* in a Wisconsin wetland were 0.37 g/m^2/yr and litterfall rates from *Typha latifolia* were 2.5 g/m^2/yr (Klopatek, 1978). Annual P litterfall fluxes in an Iowa pothole wetland were 0.63 g/m^2/yr (Davis and Van der Valk, 1978). Whereas these leaching rates would provide less than 2 percent of the total amount of P lost from the marsh in surface outflow, the highest concentrations of P in the marsh occurred during the September/early-October period of senescence (fig. 21). *Scirpus fluviatilis* is slow to decay (Whitley and others, 1990) and in the Iowa pothole, *Scirpus fluviatilis* retained more than 50 percent of its dry weight after 525 days of decomposition. Klopatek (1978) determined that 38 percent of the P uptake by *Scirpus fluviatilis* was retained over the winter months, with the rest lost to leaching and litterfall. Prentki and others (1978) reported 30 percent overwinter retention of P in *Typha latifolia*. Consequently, much of the P tied up in *Scirpus fluviatilis* is prone to burial. Even though most (95 to 99 percent) of the nutrients used by emergent macrophytes are taken in from the soil rather than the overlying water (Wetzel, 1983), the amount of P returned into the water column by emergent vegetation would likely be a tiny fraction of the amount in the soil, and, therefore, burial of P probably is a substantial sink of P in Little Bean Marsh.

Comparisons with the Missouri River, nearby Wetlands, and Missouri Reservoirs

Whereas few studies of wetland water quality have been undertaken in Missouri or along the Missouri River, reports by Knowlton and Jones (1997) and Richards (1995) provide some data for comparison. Average total N concentrations (ranging from 0.6 mg/L at site M1 to 1.7 mg/L at site M6) (fig. 11) in Little Bean Marsh, are substantially less than four Missouri River oxbows sampled by Knowlton and Jones (1997) (table 7), roughly one-half of mean concentrations of total organic, NH_3, and NO_3-N in the Missouri River reported by Hauck and others (1999) (table 4) and Ford (1982), but somewhat greater than the average total N concentrations reported for reservoirs of the glaciated plains of Missouri (0.65 mg/L) by Jones and Knowlton (1993). Low N concentrations in Little Bean Marsh primarily result from the near absence of inorganic N (figs. 19 and 20). Others also have reported low concentrations of inorganic N in riparian wetlands (Kibby, 1978; Mitsch, 1978). Apparently, denitrification and biotic uptake keep dissolved inorganic N concentrations low. This finding is similar to N speciation found in Missouri River oxbows by Knowlton and Jones (1997), who reported that dissolved organic N composed more than 95 percent of the total dissolved N, and more than 50 percent of the total N. Most N lost from Little Bean Marsh in surface outflow was organic N, and most of that was dissolved. This N speciation contrasts with the receiving Missouri River, where the dominant N species is NO_3 (Knowlton and Jones, 1997) (table 4).

Richards (1995, 1999) collected samples approximately every 3 months on an unnamed flood-plain water body near the Eagle Bluffs of the Missouri River in central Missouri beginning in 1992. This water body was created by flood scour, and is several feet deeper than Little Bean Marsh. A similar setting in the Missouri River flood plain makes comparisons with Little Bean Marsh useful. Specific conductance in the scour was similar to that in alluvial ground water. Although the scour had somewhat greater concentrations of all N and P species except organic nitrogen than Little Bean Marsh, both water bodies had low concentrations of NO_3, indicating N limitation caused by denitrification in the scour. The greater percentage of organic nitrogen in Little Bean Marsh may have been related to the large quantity of emergent vegetation compared to the deeper, open-water scour. The greater concentrations of P and inorganic N in the scour may have been because of lesser rates of vegetative uptake in the deeper scour, because much of the water in the scour was likely below the photic zone. The pH values in the scour were well within the range measured at Little Bean Marsh.

An artificially-created and intensively-managed wetland near the Eagle Bluffs scour provides another useful comparison. This water source for this artificial wetland primarily is outflow from municipal sewage-treatment wetlands, supplemented with Missouri River water. Consequently, the Eagle Bluffs wetland receives a much greater loading of N and P than Little Bean Marsh. The Eagle Bluffs wetland was filled in 1996 and was sampled at the outflow from 1996 to 1999 (Richards, 1999). As might be expected, N and P concentrations were much larger that those present in Little Bean Marsh, indicating that the created wetland may not have the nutrient removal capacity of Little Bean Marsh, a remnant native wetland. However, the cause of this difference in removal capacity is not apparent, because in addition to greater loading rates, the Eagle Bluff wetland may not yet have a thick layer of organic matter on the marsh bottom, and may have more algal growth in the abundant open water. All N and P species concentrations also were greater in the Eagle Bluffs Wetland than in the Missouri River. Therefore, outflow from the Eagle Bluffs wetland actually increases concentrations of nutrients in the Missouri River compared with Little Bean Marsh outflow that dilutes nutrient concentrations in the river.

Comparison of Little Bean Marsh water quality with Missouri River water quality is useful as, historically, much local runoff from small streams passed through riparian wetlands, such as Little Bean Marsh, before entering the river. Because there are very few remnant riparian wetlands left in the Missouri River flood plain, a comparison of the water quality in Little Bean Marsh and the Missouri River can help determine the effects that restored wetlands might have on water quality in the Missouri River, and document the effect that riparian wetlands may have had on Missouri River water quality before flood-plain development.

Annual means and ranges of selected constituents and properties of sites where at least 12 samples were collected over a 12-month period are listed in table 4. A Kruskal-Wallis rank-sum test was done on data collected monthly from the Missouri River at St. Joseph in 1997 and data collected biweekly at site M5 near the outflow of Little Bean Marsh during the same year to determine the statistical significance of the differences between the means of several constituents and properties. Test results and observation of data in table 4 indicate that concentrations of NO_3 were significantly less at Little Bean Marsh than concentrations in the Missouri River ($p<0.0001$). Nitrate concentrations in Little Bean Marsh usually were less than 0.005 mg/L, whereas concentrations in the Missouri River consistently were between 0.3 and 1.7, and averaged 1.1 mg/L. This difference in NO_3 concentration was likely the result of the rapid removal of NO_3 in the marsh by denitrification, whereas most water in the river is derived from ditches that convey water across the flood plain with little opportunity for denitrification. Large rivers, such as the Missouri, typically have low amounts of primary production because of high turbidity, large depths, and constant mixing. Consequently, inorganic nitrogen is not rapidly removed, can build up to substantial concentrations in large rivers such as the Missouri and Mississippi, and cause large zones of hypoxia in the Gulf of Mexico, which usually is N-limited (Goolsby and others, 1999). However, there was no significant difference between Little Bean Marsh and the Missouri River in the annual means of total NH_4 and NH_4 plus organic N ($p = 0.65$ and $p = 0.99$). Consequently, wetlands such as Little Bean Marsh may not reduce the amount of organic nitrogen in receiving rivers. However, suspended organic material in rivers often is refractory and not readily available to biota. Nevertheless, these data indicate that the vast extent of riparian marshes along the Missouri and Mississippi River may have had a substantial role in limiting NO_3 loads to the Gulf of Mexico before agricultural development of flood plains, and that drainage and removal of riparian marshes may be a major cause of the increased NO_3 loads to the Gulf of Mexico.

Annual mean concentrations of total and soluble reactive P were less in Little Bean Marsh than the Missouri River. However, whereas the differences between the means for soluble reactive P were statistically significant ($p<0.0001$), the difference between the means for total P were not significant at the 95 percent confidence level ($p = 0.063$). Nevertheless, it is likely that the water contributed from wetlands like Little Bean Marsh would decrease the concentration of P in the Missouri River, despite the potential for occasional P release during periods of wetland anoxia.

The annual mean of specific conductance at site M5 in Little Bean Marsh (500 µS/cm) was significantly less ($p = 0.0014$) than the annual mean conductance of the Missouri River (718 µS/cm). This probably reflects the large contribution of direct precipitation relative to runoff in the marsh compared to the river.

Summary and Conclusions

Water-quality benefits have been assumed to accrue from wetlands in Missouri and other parts of the Midwest despite the lack of concurrent water-quality and hydrologic data. Consequently, this study was undertaken to determine the magnitude and character of selected water-quality benefits that can occur in a remnant riparian wetland along the Missouri River, and to identify critical processes that can be managed in remnant or restored riparian wetlands for water-quality improvement. Little Bean Marsh was selected for this study because it is one of only a few remaining riparian wetlands along the Missouri River where nutrient cycling processes were expected to be relatively undisturbed. Nutrient cycling processes discovered in Little Bean Marsh may predict the effects that large-scale wetland restorations along the Missouri River will have on the water quality of the Missouri River.

In 1997, 41 percent of the water received by Little Bean Marsh was from direct precipitation, 14 percent from ground-water seepage, 30 percent from watershed runoff, and 15 percent from backflows from Bean Lake. Like many flood-plain water bodies, more than 90 percent of the marsh's drainage area is flood plain. The nearly flat slopes, sandy soils, and large shallow depressions keep runoff coefficients in the flood plain low. No runoff was observed from several storms that had more than 4 cm (centimeters) of rain; consequently, surface-water contributions from the watershed were small. River levees and flood-plain drainage have converted Little Bean Marsh and many other Missouri River wetlands from open to closed systems. Open wetlands typically are large sinks for alloctanous sediment and nutrients, whereas closed wetlands rely primarily on precipitation for nutrients. Consequently, the hydrology, water quality, and ecological functions of remnant wetlands located along today's Missouri River usually are substantially different from the predevelopment condition.

Ground-water levels near Little Bean Marsh did not closely track Missouri River stages primarily because of the 1.6 km (kilometers) separating them. However, ground-water levels closely tracked marsh water levels indicating a substantial hydraulic connection between ground water and the marsh. Although Little Bean Marsh was both a recharge and discharge area, ground water entering the marsh was three times the recharge to ground water. Ground water typically recharged the marsh on the north side, and marsh water seeped down through the marsh bottom on the south side. The close interaction with shallow ground water indicates lowering of the water table by as little as 1 m (meter) by pumping would likely make Little Bean Marsh much drier, and could substantially affect the marsh ecosystem. Reduced surface runoff and ground-water availability is a stabilizing influence on marsh hydrology, and probably leads to the persistence of emergent vegetation. The ground-water connection of Little Bean Marsh indicates that the hydrologic regime of most wetlands along the lower Missouri River is largely a function of marsh depth relative to the water table.

More water was lost from the marsh through evapotranspiration (59 percent) than all other pathways combined. This characteristic partially is because of abundant macrophytes that can increase evapotranspiration above that lost from open-water surfaces. Surface outflow accounted for 36 percent and ground-water seepage accounted for only 5 percent of the losses. Long residence times and large evaporative losses allow Little Bean Marsh to perform the hydrologic functions of peak-flow attenuation and surface-runoff reduction. Routing of streams around marshes obviously precludes these benefits. Large residence times also allows the marsh to greatly affect water quality before water escapes as ground-water recharge or surface outflow.

The greater percentage of precipitation as an inflow source has diluting effects on all water-quality constituents, and the substantial difference between specific conductance of ground water and precipitation/surface runoff make specific conductance a useful indicator of the primary source of water in the marsh at a given time. However, because of the shallowness of the marsh, ion exclusion during ice formation in the winter caused the greatest specific conductances during the entire monitoring period. This substantial increase in the concentration of all dissolved solutes could make wetlands more vulnerable to toxic contaminants than other surface-water bodies.

Dissolved-oxygen concentrations were less than 5 mg/L (milligrams per liter) for 3 to 4 months in 1996 and 1997, and were near zero for about 1 month during the summers of both years. These periods of anoxia produce widespread reducing conditions that inhibit nitrification, enhance denitrification, and cause iron reduction and phosphorus (P) release. The largest diel changes were about 5 mg/L that often occurred during periods of low dissolved-oxygen concentrations. Dissolved-oxygen concentrations were significantly greater at open-water sites than at sites with emergent vegetation because of shading and decomposition of a large amount of detritus. These extended periods of anoxia during the summer undoubtedly limit the habitat for certain fish and other fauna that have not adapted to low oxygen conditions and have no refuge in a shallow marsh.

Water temperatures peaked between 25 and 30 °C (degrees Celsius) for almost 2 months each year. Despite depths of less than 1 m, vertical temperature differences of 3 to 9 °C persisted for more than 3 months during the summers of 1996 and 1997. The period of least dissolved-oxygen concentration corresponded with the period of maximum stratification. The ability of open-water marshes to stratify increases the area and periods of time with reducing conditions that inhibit nitrification and potentially cause denitrification and P release from iron reduction. Chlorophyll data indicate that much of Little Bean Marsh underwent a major algal bloom in September 1997, shortly after summer stratification abruptly ended. The bloom may have been triggered by mixing of the water column that made ammonia nitrogen (N) from the hypolimnion available to a N-limited population of algae. Shallow depths and extended periods of anoxia in the marsh would limit the ability of some organisms to escape high-temperature stress.

Most of the N in Little Bean Marsh was in organic forms. Denitrification and biotic uptake kept concentrations of inorganic N extremely small. This situation contrasts with N in the Missouri River and many of its tributaries, where nitrate (NO_3) is the dominant species of N. Consequently, artificial flood-plain drainage that bypasses wetlands can deliver substantially more biotically available inorganic N to receiving streams than surface water that has been routed through a remnant wetland.

Water clarity in Little Bean Marsh usually was high for several reasons. Sediment loadings from the largely flood-plain drainage were low, emergent vegetation shade out algae and shield the water from wind, and high concentrations of bivalent cations increase flocculation rates of inorganic suspended material. The high divalent cation concentration in the marsh probably was a result of ground water received by the marsh. Open-water sites had significantly greater turbidities and suspended sediment concentrations than sites covered with emergent vegetation. This difference was likely caused by shading that limited algal growth in vegetated sites. Consequently, marshes along the lower Missouri River are likely to have better water clarity if they are incised deep enough to receive substantial amounts of ground water and grow abundant emergent macrophytes.

Little Bean Marsh usually was either eutrophic or hypereutrophic. Vegetated sites had less persistent periods of eutrophy. The most intense period of hypereutrophy occurred in September, and coincided with both an algal bloom and senescence of *Scirpus fluviatilis* and *Typha latifolia*. The rapid leaching of N that occurs soon after macrophyte senescence, combined with a recent destratification of the marsh, probably provided N to the N-limited open-water areas and triggered a phytoplankton bloom. This type of bloom could be a common phenomenon in Missouri River wetlands with thick stands of macrophytes. N was better correlated with chlorophyll than was P at open-water sites, indicating N is more limiting than P on phytoplankton growth. The lack of any correlation of chlorophyll with N or P at some vegetated sites is consistent with light limitation.

Despite the rarity of runoff events, surface runoff, combined with atmospheric deposition, contributed more than seven times the 530 kg (kilograms) of N that escaped Little Bean Marsh in surface outflow in 1997. Atmospheric deposition alone was more than the 530 kg. Ground water contributed less than 1.5 percent of the N leaving the marsh in surface outflow. Undoubtedly, much more N is internally recycled through plants in Little Bean Marsh than is externally gained or lost annually. The slow decay rate of *Scirpus fluviatilis* and reducing conditions in bottom sediments potentially make burial of organic N a substantial sink of N. Annual peak concentrations of nearly all N and P species occurred during the fall period of senescence.

Results of denitrification experiments indicate that denitrification rates were limited by NO_3 in the water column. Consequently, decomposition and nitrification of ammonium (NH_4) and organic N are the rate-limiting steps of N removal in Little Bean Marsh. Conditions for high rates of denitrification were almost ideal in the bottom sediments of the marsh. The dominant macrophytes in the marsh were *Scirpus fluviatilis* and *Typha latifolia*; both have oxidizing rhizospheres around their roots. These rhizospheres greatly increase rates of the rate-limiting decomposition and nitrification processes over rates occurring in open-water areas where all NO_3 must diffuse from the water column through the sediment/water interface, as opposed to being created in the sediments. The NO_3 limitation on rates of denitrification also indicate that Little Bean Marsh has a large unused capacity for N removal.

Only the largest storm had a measurable effect on P concentrations in Little Bean Marsh. Most dissolved P usually was in the orthophosphate (PO_4) form, but most total P was on particulates. Correlations of wind speed, pH, turbidity, and dissolved oxygen with concentrations of total P indicate that periods of anoxia have much larger effects on P release than the other variables. Periods of most extreme anoxia occurred in late summer and corresponded with high rates of decomposition and senescing macrophytes to produce the highest concentrations of both N and P and a large algal bloom in Little Bean Marsh. Low water levels prevented the escape of P in surface outflow during the periods of highest P concentrations. Annual climatic patterns in the Midwestern United States probably make the correspondence of these conditions common in marshes along the Missouri River.

Little Bean Marsh removed the majority of the P it received. The amount of P in surface inflows to the marsh were more than one order of magnitude greater than that escaping in surface outflows. P contributions from atmospheric deposition to the marsh are small compared to the 53 kg of P in surface outflows in 1997. Low concentrations of P in ground water caused ground water to be a small source of P to Little Bean Marsh as well. The long hydraulic residence time of the marsh and large contributions of iron from ground water (that provide many sorption sites for P) make the marsh an effective sediment and P trap. These characteristics, plus the large long-term retention of P by the macrophytes after senescence, make burial of P a substantial sink of P in Little Bean Marsh.

N and P concentrations were much larger in an artificially-created sewage-treatment wetland along the Missouri River in central Missouri than those present in Little Bean Marsh. Much greater loadings of N and P, a lack of organic material in wetland bottom sediments, and more abundant phytoplankton in the sewage-treatment wetland may have all contributed to this phenomenon.

Concentrations of total P in Little Bean Marsh were less than those in other Missouri River wetlands. This difference is likely related to the amount of open water that is more conducive to algal growth. Average total N concentrations in Little Bean Marsh were substantially less than other Missouri River wetlands; roughly one-half of mean concentrations in the Missouri River, but roughly twice the mean N concentrations found in reservoirs of the glaciated plains of Missouri. Low concentrations of N primarily result from the near absence of inorganic N in Little Bean Marsh. Apparently, denitrification and biotic uptake keep dissolved inorganic N concentrations low. Dissolved organic N was the dominant N species in Little Bean

Marsh and other Missouri River wetlands. This situation contrasts with the Missouri River, in which NO_3 dominates.

Concentrations of NO_3 in Little Bean Marsh usually were less than 0.005 mg/L and were significantly less than concentrations in the Missouri River ($p<0.0001$), which averaged 1.1 mg/L. Large rivers, such as the Missouri, typically have low amounts of primary production because of turbidity, large depths, and constant mixing. Consequently, inorganic nitrogen is not rapidly removed and can build up to substantial concentrations in large rivers, such as the Missouri and the Mississippi, and can cause large zones of oceanic hypoxia, such as in the Gulf of Mexico, which typically are N limited. Concentrations of organic N in Little Bean Marsh and the Missouri River, however, were not significantly different. Consequently, wetlands such as Little Bean may not reduce the amount of organic nitrogen in receiving rivers. Suspended organic material in rivers often is refractory, however, and not readily available to biota. Nevertheless, these data indicate that the vast historical extent of riparian marshes along the Missouri and Mississippi Rivers may have had a substantial role in limiting NO_3 loads to the Gulf of Mexico before agricultural development of flood plains, and that drainage and removal of riparian marshes may be a primary cause of the increased NO_3 loads to the Gulf of Mexico today. Annual mean concentrations of total and soluble reactive P were both less in Little Bean Marsh than the Missouri River; therefore, the water contributed from Little Bean Marsh and marshes like it would likely decrease the concentration of P in the Missouri River, despite the potential for occasional P release during periods of wetland anoxia.

Whereas the data indicate that Little Bean Marsh has the capacity to remove much more N and P than is presently removed, large quantities of additional nutrients could alter species composition of wetland vegetation and cause other hydrochemical changes. For example, rerouting of Short Creek through Little Bean Marsh might allow the removal of additional N from runoff through denitrification, but also might cause the marsh to fill with sediment. Because remnant riparian wetlands along the lower Missouri River are so rare, preservation of these wetlands may be more important than using them to improve the quality of downstream and downgradient waters such as the Missouri River.

References

Akhurst, E.G., and Breen, C.M., 1988, Ionic content as a factor influencing turbidity in two floodplain lakes after a flood: Hydrobiologia, v. 160, p. 19–31.

American Public Health Association, 1980, Standard methods for the examination of water and wastewater (15th ed.): New York, p. 1,134.

Baccini, P., 1985, Phosphate interactions at the sediment-water interface, *in* Wilcoxon, W. (ed.), Chemical processes in lakes: New York, Wiley Sciences, p. 189–205.

Baillie, P.W., 1995, Renovation of food processing wastewater by a riparian wetland: Environmental Management, v. 19, p. 115–126.

Biesboer, D.D., 1984, Nitrogen fixation associated with natural and cultivated stands of *Typha latifolia* (*Typhaceae*): American Journal of Botany, v. 71, p. 505.

Bostrom, B., Andersen, J.M., Fleischer, S., and Jansson, M., 1988, Exchange of phosphorus across the sediment-water interface: Hydrobiologia, v. 170, p. 229–244.

Bostrom, B., Jansson, M., and Forsberg, C., 1982, Phosphorus release from lake sediments: Arch. Hydrobiol. Beih., v. 18, p. 5–59.

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

Bremmer, J.M., and Shaw, K., 1958, Denitrification in soil. II. Factors affecting denitrification: Journal of Agricultural Science, v. 51, p. 40.

Brinson, M.M., Bradshaw, H.D., and Kane, E.S., 1984, Nutrient assimilative capacity of an alluvial floodplain swamp: Journal of Applied Ecology, v. 21, p. 1,041–1,057.

Broderick, S.J., Cullen, P., and Maher, W., 1988, Denitrification in a natural wetland receiving secondary treated effluent: Water Resources, v. 22, p. 431–439.

Carper, G.L., and Bachmann, R.W., 1984, Wind resuspension of sediments in a Prairie Lake: Canadian Journal of Fisheries and Aquatic Science, v. 41, p. 1,763–1,767.

Carter, V., 1986, An overview of the hydrologic concerns related to wetlands in the United States: Canadian Journal of Botany, v. 64, p. 364–374.

Carter, V., and Haramis, G.M., 1980, Distribution and abundance of submersed aquatic vegetation in the tidal Potomac River—implications for waterfowl: Atlantic Naturalist, v. 33, p. 14–19.

Carter, V., Paschal, J.E., and Bartow, N., 1985, Distribution and abundance of submersed aquatic vegetation in the tidal Potomac River and estuary, Maryland and Virginia, May 1978 to November 1981: U.S. Geological Survey Water-Supply Paper 2234A, p. 1–46.

Castaner, D., and LaPlante, D., 1992, The vegetation of Little Bean Marsh: Transactions, Missouri Academy of Science, v. 26, p. 27–33.

Chen, R.L., Keeney, D.R., Graetz, D.A., and Holding, A.J., 1972, Denitirification and nitrate reduction in Wisconsin lake sediments: Journal of Environmental Quality, v. 1, no. 2, p. 158–162.

Cooke, J.G., 1992, Phosphorus removal processes in a wetland after a decade of receiving a sewage effluent: Journal of Environmental Quality, v. 21, p. 733–739.

Cooke, J.G., Stub, L., and Mora, N., 1992, Fractionation of phosphorus in the sediment of a wetland after a decade of receiving sewage effluent: Journal of Environmental Quality, v. 21, p. 726–732.

Cooper, A.B., 1990, Nitrate depletion in the riparian zone and stream channel of a small headwater catchment: Hydrobiologia, v. 202, p. 13–26.

Craft, C.B., and Casey, W.P., 2000, Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA: Wetlands, v. 20, no. 2, p. 323–332.

Crowther, J., and Evans, J., 1978, Dual channel for determination of dissolved organic and inorganic carbon: Ontario Ministry of the Environment, Document JC 7501, 51 p.

Crumpton, W.G., Isenhart, T.M., and Mitchell, P.D., 1992, Nitrate and organic N analyses with second-derivative spectroscopy: Limnology and Oceanography, v. 37, p. 907–913.

Darcy, H., 1856, Les fontaines publiques de la ville de Dijon: Paris, V. Dalmont, 674 p.

Davis, C.B., and Van der Valk, A.G., 1978, Litter decomposition in prairie glacial marshes, *in* Good, R.E., Whigham, D.F., Simpson, R.L., and Jackson, C.G. (eds.), Freshwater Wetlands: Ecological Processes and Management Potential: New York, USA, Academic Press, p. 20–28.

Davis, J.V., 1991, Missouri Stream Water Quality, *in* National Water Summary 1990–91: U.S. Geological Survey Water-Supply Paper 2400, p. 351–360.

Dierberg, F.E., and Brezonik, P.L., 1985, Nitrogen and phosphorus removal cypress swamp sediments: Water Air Soil Pollution, v. 24, p. 207.

Dortch, M.S., 1996, Removal of solids, nitrogen, and phosphorus in the Cache River Wetland: Wetlands, v. 16, p. 358–365.

Dye, C.W., Jones, D.A., Ross, L.T., and Gernert, J.L., 1980, Diel variations of selected physico-chemical parameters in Lake Kissimmee, Florida: Hydrobiologia, v. 71, p. 51–60.

Elder, J.F., 1985, Nitrogen and phosphorus speciation and flux in a large Florida river wetland system: Water Resources Research, v. 21, p. 724–732.

Emmett, L.F., and Jeffery, H.G., 1969, Reconnaissance of the ground-water resources of the Missouri River alluvium between Kansas City, Missouri and the Iowa border: U.S. Geological Survey Hydrologic Investigations Atlas HA-336, 1 sheet.

Fisheries and Environment Canada, 1977, The chemical analysis of fresh water, Misc. Pub. No. 25. Second Edition: Manitoba, Freshwater Institute, 180 p.

Ford, J.C., 1982, Water quality of the lower Missouri River, Gavins Point to the mouth: Missouri Department of Natural Resources, Division of Environmental Quality, 35 p.

Freeze, A.R., and Cherry, J.A., 1979, Groundwater: Englewood Cliffs, N.J., Prentice-Hall, Inc., 604 p.

Gale, P.M., Reddy, K.R., and Graetz, D.A., 1993, Nitrogen removal from reclaimed water applied to constructed and natural wetland microcosms: Water Environment Research, v. 65, p. 162–168.

Goolsby, D.A., Battaglin, W.A., Lowrence, G.B., Artz, R.S., Aulenback, B.T., and Hooper, R.P., 1999, Flux and sources of nutrients in the Mississippi-Atchafalaya River Basin: NOAA Coastal Ocean Program, Decision Analysis Series No. 17, 50 p.

Goolsby, D.A., Boyer, L.L., and Mallard, G.E. (eds.), 1993, Selected agricultural chemicals in water resources of the midcontinental United States: U.S. Geological Survey Open-File Report 93–418, 89 p.

Gosselink, J.G., and Turner, R.E., 1978, The role of hydrology in freshwater wetland ecosystems, *in* Freshwater Wetlands, Good, R.E., Whigham, D.F., and Simpson, R.L. (eds): New York, Academic Press, 63 p.

Hach Company, 1987, Water Analysis Handbook: Loveland, Colo., 401 p.

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

Hauck, H.S., Huber, L.G., and Nagel, C.D., 1998, Water resources data for Missouri water year 1997: U.S. Geological Survey Water-Data Report MO–97–1, 318 p.

Hauck, H.S., Huber, L.G., and Nagel, C.D., 1999, Water resources data for Missouri water year 1998: U.S. Geological Survey Water-Data Report MO–98–1, 434 p.

Hecky, R.E., and Hendzel, L.L., 1980, Physiological indicators of nutrient deficiency in lake phytoplankton: Canadian Journal of Fisheries and Aquatic Science, v. 37, p. 442–494.

Heimann, D.C., and Femmer, S.R., 1998, Water quality, hydrology, and invertebrate communities of three remnant wetlands in Missouri, 1995–97: U.S. Geological Survey Water-Resources Investigation Report 98–4190, 64 p.

Holford, I.C., and Patrick, W.H., 1981, Effects of duration of anaerbiosis and reoxidation on phosphate sorption characteristics of an acid soil: Australian Journal of Soil Research, v. 19, p. 69.

Horne, A.J., and Goldman, C.R., 1994, Limnology: USA, McGraw-Hill, 576 p.

Hoyer, M.V., and Reid, F.A., 1982, Seasonally flooded Missouri wetlands: Early winter limnological characteristics in relation to physiographic regions: Transactions of the Missouri Academy of Science, v. 16, p. 67–75.

Jensen, J.R., and Rahman, M.M., 1987, Semi empirical model for calculating evaporation and transpiration from wetland rice: Agricultural and Forest Meteorology, v. 41, nos. 3–4, p. 289–306.

Johnston, C.A., 1991, Sediment and nutrient retention by freshwater wetlands: Effects on surface-water quality: Critical Reviews in Environmental Control, v. 21, nos. 5–6, p. 491–565.

Johnston, C.A., 1993, Mechanisms of wetland water quality interaction, *in* Constructed Wetlands for Water Quality Improvement, Moshiri, G.A. (ed.): Boca Raton, Fla., CRC press, p. 293–299.

Jones, J.R., and Knowlton, M., 1993, Limnology of Missouri reservoirs: An analysis of regional patterns: Lake Reservoir Management, v. 8, no. 1, p. 17–30.

Kana, T.M., and Tjepkema, J.D., 1978, Nitrogen fixation associated with *Scirpus atrovirens* and other nonnodulated plants in Massachusetts, USA: Canadian Journal of Botany, v. 56, p. 26–36.

Kelly, B.P., and Blevins, D.W., 1995, Vertical hydraulic conductivity of soil and potentiometric surface of the Missouri River Alluvial Aquifer at Kansas City, Missouri and Kansas–

August 1992 and January 1993: U.S. Geological Survey Open-File Report 95–322, 19 p.

Kemp, G.P., and Day, J.W., 1984, Nutrient dynamics in a Louisiana swamp receiving agricultural runoff, *in* Cypress Swamps, Ewel, K.C., and Odums, H.T. (eds.): Gainesville, University of Florida Press, p. 286–293.

Kibby, H.V., 1978, Effects of wetlands on water quality, *in* Proceedings of Strategies for protection and management of floodplain wetlands and other riparian ecosystems: U.S. Department of Agriculture GTR-WO-12, p. 289–297.

Klopatek, J.M., 1978, Nutrient dynamics of freshwater riverine marshes and the role of emergent macrophytes, *in* Freshwater wetlands: Ecological processes and management potentials, Good, R.E., Whigham, D.F., and Simpson, R.L. (eds.): New York, Academic Press, 195 p.

Knowlton, M.F., 1984, Flow-through microcuvette for flourometric determination of chlorophyll: Water Resources Bulletin, v. 20, p. 795–799.

Knowlton, M.F., and Jones, J.R., 1997, Trophic status of Missouri River floodplain lakes in relation to basin type and connectivity: Wetlands, v. 17, no. 4, p. 468–475.

Lawrence, G.B., Goolsby, D.A., Bataglin, W.A., and Stensland, G.J., 2000, Atmospheric nitrogen in the Mississippi River Basin–emissions, deposition, and transport: The Science of the Total Environment, v. 248, nos. 2–3, p. 87–99.

Lindau, C.W., DeLaune, R.D., and Pardue, J.H., 1994, Inorganic nitrogen processing and assimilation in a forested wetland: Hydrobiologia, v. 277, p. 171–178.

Lowrance, R., Vellidis, G., and Hubbard, R.K., 1995, Denitrification in a restored riparian forest wetland: Journal of Environmental Quality, v. 24, p. 808–815.

Missouri Department of Natural Resources, 1986, Missouri Water Atlas: Rolla, Missouri Department of Natural Resources, Geological Survey and Resource Assessment, 100 p.

Missouri Department of Natural Resources, 1999, Rules of Department of Natural Resources, Division 20–Clean Water Commission, Chapter 7-Water Quality, 136 p.

Mitsch, W.J., 1978, Interactions between a riparian swamp and a river in Southern Illinois, *in* Proceedings of symposium Strategies for protection and management of floodplain wetlands and other riparian ecosystems: U.S. Department of Agriculture GTR-WO-12, p. 63–72.

Mitsch, W.J., Dorge, C.L., and Wiemhoff, J.R., 1979, Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in southern Illinois: Ecology, v. 60, p. 1,116–1,124.

Mitsch, W.J., and Gosselink, J.G., 1993, Wetlands (2nd ed.): New York, Van Nostrand Reinhold, 722 p.

Montieth, J.L., 1965, Evaporation and the environment: Symposium Society Experimental Biology, v. 19, p. 205–234.

Morris, J.T., 1991, Effects of nitrogen loading on wetland ecosystems with particular reference to atmospheric deposition: Annual Review of Ecology and Systematics, v. 22, p. 257–279.

Mortimer, C.H., 1941a, The exchange of dissolved substances between mud and water in lakes: Journal of Ecology, v. 29, p. 280–329.

Mortimer, C.H., 1941b, The exchange of dissolved substances between mud and water in lakes: Journal of Ecology, v. 30, p. 149–201.

Moustafa, M.Z., 1997, Graphical representation of nutrient removal in constructed wetlands: Wetlands, v. 17, no. 4, p. 493–501.

National Atmospheric Deposition Program (IR-7)/National Trends Network, Annual report, 1997, NADP/NTN Coordination Office: Fort Collins, Colo., Natural Resource Ecology Laboratory, Colorado State University, p. 103.

National Oceanic and Atmospheric Administration/National Climatic Data Center, 1996 and 1997, Climatological Data Annual Summary Missouri, v. 99, no. 13, p. 3.

Nichols, D.S., 1983, Capacity of natural wetlands to remove nutrients from wastewater: Journal of Water Pollution Control Federation, v. 55, p. 495–505.

Ott, R.L., 1993, An introduction to statistical methods and data analysis: Belmont, Calif., Duxbury Press, 1,051 p.

Parkhurst, R.S., Winter, T.C., Rosenberry, D.O., and Sturrock, A.M., 1998, Evaporation from a small prairie wetland in the Cottonwood Lake area, North Dakota—and energy-budget study: Wetlands, v. 18, p. 272–287.

Patrick, W.H., 1991, Microbial reactions of nitrogen and phosphorus in wetlands: The Utrecht Plant Ecology News Report No. 11, p. 52–63.

Patrick, W.H., and Khalid, R.S., 1974, Phosphate release and sorption by soils and sediments: effect of aerobic and anaerobic conditions: Science, v. 186, p. 53.

Patrick, W.H., and Reddy, K.R., 1976, Nitrification-denitrification reactions in flooded soils and water bottoms: dependence on oxygen supply and ammonium diffusion: Journal of Environmental Quality, v. 5, no. 4, p. 469–472.

Penman, H.L., 1948, Natural evaporation from open water, bare soil and grass: Proceedings Royal Society London A193, p. 120–145.

Peterjohn, W.T., and Correl, D.L., 1984, Nutrient dynamics in an agricultural watershed observations on the role of a riparian forest: Ecology, v. 65, no. 5, p. 1,466–1,475.

Peverly, J.H., 1982, Stream transport of nutrients through a wetland: Journal of Environmental Quality, v. 11, p. 38–43.

Phillips, R.E., Reddy, K.R., and Patrick, W.H., Jr., 1978, The role of nitrate diffusion in determining the order and rate of denitrification in flooded soil II: Theoretical analysis and interpretation: Soil Science Society of America Journal, v. 42, p. 272–278.

Prentki, R.T., Gustafson, T.D., and Adams, M.S., 1978, Nutrient movements in lakeshore marshes, *in* Freshwater wetlands ecological processes and management potential, Good, R.E., Whigham, D.F., and Simpson, R.L. (eds): New York, Academic Press, p. 169–194.

Prepas, E.E., and Rigler, F.H., 1982, Improvements in quantifying the phosphorus concentration in lake water: Canadian Journal of Fisheries and Aquatic Science, v. 39, p. 822–829.

Rantz, S.E., and others, 1982, Measurement and computation of streamflow: Volume 1, Measurement of stage and discharge and Volume 2, Computation of discharge: U.S. Geological Survey Water-Supply Paper 2175, 631 p.

Reddy, K.R., 1981, Diel variations of certain physico-chemical parameters of water in selected aquatic systems: Hydrobiologia, v. 85, p. 201–207.

Reddy, K.R., Patrick, W.H., and Phillips, R.E., 1978, The role of nitrate diffusion in determining the order and rate of denitrification in flooded soil: Experimental results: Soil Science Society of America Journal, v. 42, p. 268–272.

Reddy, K.R., Saco, P.D., and Graetz, D.A., 1980, Nitrate reduction in an organic soil-water system: Journal of Environmental Quality, v. 9, p. 283–288.

Richards, J.M., 1995, Hydrologic data for the Columbia/Eagle Bluffs wetland complex, Columbia, Missouri—1992–93: U.S. Geological Survey Open-File Report 95–109, 50 p.

Richards, J.M., 1999, Hydrologic data for the Columbia/Eagle Bluffs wetland complex, Columbia, Missouri—1993–96: U.S. Geological Survey Open-File Report 99–607, 91 p.

Richardson, C.J., 1985, Mechanisms controlling phosphorus retention capacity in freshwater wetlands: Science, v. 228, p. 1,424–1,427.

Rose, C., and Crumpton, W.G., 1996, Effects of emergent macrophytes on dissolved oxygen dynamics in a prairie pothole wetland: Wetlands, v. 16, no. 4, p. 495–502.

Ryden, J.C., Lund, L.J., and Focht, D.D., 1979a, Direct measurement of denitrification from Soils: I Laboratory evaluation of acetylene inhibition of nitrous oxide reduction: Soil Science Society of America Journal, v. 43, p. 104–110.

Ryden, J.C., Lund, L.J., Letey, J., and Focht, D.D., 1979b, Direct measurement of denitrification measurement in soils: II Development and application of field methods: Soil Science Society of America Journal, v. 43, p. 110–118.

Sartory, D.P., and Grobbelaar, J.U., 1984, Extraction of chlorophyll-a from freshwater phytoplankton for spectrophotometric analysis: Hydrobiologia, v. 114, p. 177–181.

Sondergaard, M., Kristensen, P., and Jeppesen, E., 1992, Phosphorus release from resuspended sediment in the shallow and wind exposed Lake Arreso, Denmark: Hydrobiologia, v. 228, p. 91–99.

Souch, C., Grimmond, C.S.B., and Wolfe, C.P., 1998, Evapotranspiration rates from wetlands with different disturbance histories: Indiana Dunes National Lakeshore: Wetlands, v. 18, no. 2, p. 216–229.

Stanford, G., and others, 1975, Effect of temperature on denitrification rates in soils: Soil Science Society of America Proceedings, v. 39, p. 867.

Stewart, R.R., and Ornes, W.H., 1973, Assessing the capability of the Everglades marsh environment for renovating wastewater: Journal of Water Pollution Control Federation, v. 47, p. 1,880–1,891.

Tukey, J.W., 1977, Exploratory Data Analysis: Reading, Mass., Addison-Wesley Publishing Company, 506 p.

U.S. Environmental Protection Agency, 1979, Methods for chemical analysis of water and wastes: Washington D.C., 257 p.

Walter, A.R.B., 1998, The hydrogeology and geochemistry of the Missouri River alluvial aquifer near Little Bean Marsh: University of Missouri at Kansas City, unpublished Master's Thesis, 66 p.

Ward, J.R., 1973, Geohydrology of Atchison County, Northeastern Kansas: U.S. Geological Survey Hydrologic Investigations Atlas HA–467, 2 sheets.

Wetzel, R.G., 1983, Limnology (2nd ed.): New York, USA, Sounders College Press, 767 p.

Whitley, J.R., Bassett, B., Dillard, J.G., and Hefner, R.A., 1990, Water plants for Missouri Ponds: Missouri Department of Conservation, 151 p.

Wijler, J., and Delwiche, C.C., 1954, Investigations on the denitrifying process in soil: Plant Soil, v. 5, p. 155.

Wilcoxon, F., 1945, Individual comparisons by ranking methods: Biometrics, v. 1, p. 80–83.

Wildke, F.D., Radtke, D.B., Gibs, J., and Iwatsubo, R.T., 1998, National field manual for the collection of water-quality data: U.S. Geological Survey Techniques for Water-Resources Investigations, Book 9, variously paged.

Wylie, G.D., and Jones, J.R., 1986, Limnology of a wetland complex in the Mississippi alluvial valley of southeast Missouri: Archiv Hydrobiologic (Supplement), v. 74, no. 3, p. 288–314.

Yoshinari, T., Hynes, R., and Knowles, R., 1977, Acetylene inhibition of nitrous oxide reduction and measurement of denitrification and nitrogen fixation in soil: Soil Biology and Biochemistry, v. 9, p. 177–183.

Zak, D.R., and Grigal, D.G., 1991, Nitrogen mineralization, nitrification and denitrification in upland and wetlands systems: Oecologia, v. 88, p. 189–196.

Ziegler, A.C., Wallace, W.C., Blevins, D.W., and Maley, R.D., 1993, Occurrence of pesticides, nitrite plus nitrate, arsenic, and iron in water from two reaches of the Missouri River alluvium, northwestern Missouri--July 1988 and June-July 1989: U.S. Geological Survey Open-File Report 93–101, 30 p.

TABLE

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Time	Gage height, in ft	Water depth, in ft	Dissolved oxygen (top), in mg/L	Dissolved oxygen (bottom), in mg/L	Water temperature (top), in °C	Water temperature (bottom), in °C	pH, in standard units	Specific conductance, in µS/cm	Turbidity, FTU
M1	03/17/1997	--	28.25	--	8.3	--	5.3	--	7.5	1,150	--
M1	03/31/1997	1550	28.05	--	--	--	--	--	7.86	731	8
M1	04/17/1997	1200	29.27	--	11.6	--	13.3	--	7.92	637	5
M1	05/05/1997	1430	--	--	6.4	6.4	23.1	22.3	7.66	652	12
M1	05/21/1997	1330	29.59	--	4.2	4.3	23.3	--	7.77	695	5
M1	06/10/1997	--	29.56	1	2.8	2.2	20.8	20.6	7.46	734	5
M1	06/25/1997	--	--	--	--	--	--	--	--	--	5
M1	07/08/1997	1500	28.98	.8	1.6	1.2	24.9	24.6	7.28	675	5
M1	07/29/1997	1345	28.65	.6	3	--	25.2	24.9	6.74	546	5
M1	08/12/1997	1420	28.37	.5	1.6	1.4	22.8	22.8	7.06	683	20
M1	12/02/1997	--	--	--	7.1	--	5.8	--	7.15	777	10
M1	12/19/1997	1200	--	[1].4	--	--	.5	--	7.3	--	--
M2	11/15/1996	1600	--	2	12.9	12.2	6.6	6.6	7.34	585	25
M2	12/02/1996	1345	29.16	2	13.6	--	4.1	--	7.69	605	20
M2	12/19/1996	1230	29.12	--	13.7	--	2.3	--	7.66	756	10
M2	01/03/1997	1500	--	1.5	3.7	3.0	3	--	6.67	615	3
M2	01/20/1997	1545	28.27	.04	15.6	--	.2	--	7.13	--	--
M2	02/05/1997	1430	28.27	1	4.6	--	1	--	7.02	998	5
M2	02/17/1997	1500	--	.4	--	--	1.5	--	7.09	1,210	15
M2	02/24/1997	--	--	--	--	--	--	--	--	--	10
M2	03/17/1997	1450	28.25	--	10.3	--	8.3	--	7.64	917	10
M2	03/31/1997	--	28.05	--	13	--	13.5	--	7.31	1,220	10
M2	04/17/1997	1230	29.27	2	7.8	7.2	13.3	11.4	7.55	331	35
M2	05/05/1997	1530	--	--	6.6	6.5	21.5	20	8.1	522	30
M2	05/21/1997	1400	--	--	9.3	3.3	25.5	18.9	7.6	541	10
M2	06/10/1997	1100	--	--	1	.4	21	19.8	7.4	504	5
M2	06/25/1997	--	--	--	--	--	--	--	--	--	5
M2	07/08/1997	1430	28.98	1.2	3.4	2.4	27.1	24.9	7.21	409	5
M2	07/29/1997	--	28.65	1	5	1	31	27	6.76	385	15
M2	08/12/1997	1425	28.37	.7	1.4	.7	22.3	22.3	7.06	--	15

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Time	Gage height, in ft	Water depth, in ft	Dissolved oxygen (top), in mg/L	Dissolved oxygen (bottom), in mg/L	Water temperature (top), in °C	Water temperature (bottom), in °C	pH, in standard units	Specific conductance, in µS/cm	Turbidity, FTU
M2	08/27/1997	1330	28	0.3	3	--	27.7	--	7.47	--	20
M2	09/10/1997	1315	27.54	--	--	--	--	--	8.17	294	20
M2	09/24/1997	1300	27.49	.4	2.1	--	16.8	--	7.46	506	18
M2	10/14/1997	1330	27.68	.5	10.8	8.5	15.9	14.9	7.78	994	10
M2	10/28/1997	1145	27.94	.5	9.4	--	8.5	--	7.42	889	15
M2	11/19/1997	1430	27.86	.6	9.6	--	9.2	--	7.4	1,100	8
M2	11/19/1997	--	--	--	--	--	--	--	--	--	--
M2	12/02/1997		28.3	--	4.8	--	6.1	--	7.29	608	10
M2	12/19/1997	1230	--	1.5	--	--	5.2	--	7.23	--	13
M3	11/15/1996	1415	--	--	11	--	7.1	--	7.13	509	5
M3	11/25/1996	1400	29.09	2.3	12.0	--	1.6	--	8.36	555	--
M3	12/18/1996	1145	--	--	13.3	--	3.5	--	8.25	641	10
M3	01/03/1997	1117	28.85	2.1	16.1	--	5.1	--	7.32	879	5
M3	01/20/1997	1145	28.27	1.5	22.9	24.1	2.4	2.7	7.32	1,210	--
M3	02/05/1997	1100	28.27	1.5	12.8	--	1.3	2.8	6.8	1,060	5
M3	02/17/1997	1030	27.94	1.1	13	11.1	1.3	3.8	--	880	10
M3	02/24/1997	--	--	--	--	--	--	--	--	--	8
M3	03/17/1997	0945	28.25	1.5	11.7	--	7.1	--	8.19	702	5
M3	03/31/1997	1130	28.05	1.3	11.8	--	14.3	--	8.19	719	10
M3	04/17/1997	1310	29.27	2.5	11.1	11.1	15.7	15.7	8.28	380	22
M3	05/05/1997	1110	--	--	8.2	8.2	18.3	18.3	8.15	530	30
M3	05/21/1997	1120	29.59	2	14.2	14.5	21.7	214	8.44	499	25
M3	06/09/1997	--	29.42	2.6	5.8	5.5	24	24	7.68	518	10
M3	06/25/1997	1050	29.2	2.4	8.6	8.2	27.5	--	7.92	366	5
M3	07/08/1997	1315	28.98	2	9.8	7.5	28.5	28.3	7.6	461	5
M3	07/29/1997	1215	28.65	1.8	10	5	29	27.5	7.62	300	30
M3	08/12/1997	1325	28.37	1.6	5.6	5.6	23	23	7.85	272	10
M3	08/27/1997	1200	28	1.5	7	2.5	29.1	28.1	8.27	260	10
M3	09/10/1997	1115	27.54	1	6.4	6.3	22.7	22.8	8.13	248	35
M3	09/24/1997	1125	27.49	.8	2.9	2.8	16.8	16.8	7.88	275	35
M3	10/14/1997	1130	27.68	1	11.1	11	12.3	12.4	8.31	468	20

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Time	Gage height, in ft	Water depth, in ft	Dissolved oxygen (top), in mg/L	Dissolved oxygen (bottom), in mg/L	Water temperature (top), in °C	Water temperature (bottom), in °C	pH, in standard units	Specific conductance, in µS/cm	Turbidity, FTU
M3	10/28/1997	1045	27.94	1.1	13.9	13.9	7.5	--	7.67	508	15
M3	11/19/1997	1315	27.86	1.3	19	19.5	6.2	6.3	7.5	649	10
M3	12/02/1997	--	28.3	1.3	11.4	11.6	6.5	6.5	7.75	534	10
M3	12/19/1997	1115	--	2	--	--	4.3	--	8.37	--	10
M4	05/21/1997	1055	--	1.5	6.7	5.3	19.4	18.8	7.45	745	10
M4	06/10/1997	--	--	1	.5	.5	20.3	--	7.6	514	5
M4	06/25/1997	--	--	--	--	--	--	--	--	--	10
M4	07/08/1997	1245	28.98	1.25	4.6	.2	26.8	24.2	7.75	445	3
M4	07/29/1997	1145	28.65	.9	8.5	4.5	26.9	24.8	6.83	394	10
M4	08/12/1997	1145	28.37	.92	1.4	1.6	21.5	21.5	7.01	318	5
M4	08/27/1997	1120	28	.33	3.8	.1	23	--	7.3	330	10
M4	10/28/1997	0945	27.94	.25	12.3	--	5.9	--	6.71	513	20
M4	11/19/1997	1130	27.86	.17	13.8	--	.6	--	--	699	15
M4	12/02/1997	1345	28.3	--	8.4	--	5.9	--	7.48	596	10
M4	12/19/1997	1100	--	.83	--	--	1.5	--	7.78	--	10
M5	11/15/1996	1415	--	2.5	11	--	7.1	--	7.13	510	10
M5	11/25/1996	--	29.09	--	--	--	--	--	--	--	--
M5	12/02/1996	1210	--	--	12.7	--	3.9	--	7.52	530	5
M5	12/19/1996	1100	29.12	--	15.7	--	2.1	--	8.25	616	20
M5	01/03/1997	1315	28.85	--	13.1	--	2.7	--	7.48	651	15
M5	01/20/1997	1400	28.27	--	13	--	2.6	--	7.49	908	20
M5	02/05/1997	--	28.27	--	16.3	--	3.4	--	6.98	501	5
M5	02/17/1997	1210	27.94	--	9.2	--	3.7	--	7.29	900	--
M5	02/21/1997	--	--	--	--	--	--	--	--	--	17
M5	02/24/1997	--	--	--	--	--	--	--	--	--	17
M5	03/17/1997	--	28.25	--	12.4	--	6.3	--	8.37	591	35
M5	03/31/1997	--	28.05	--	--	--	--	--	--	--	35
M5	04/05/1997	--	--	--	--	--	--	--	--	424	--
M5	04/10/1997	--	--	--	--	--	--	--	--	360	--
M5	04/11/1997	--	--	--	--	--	--	--	--	--	28

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; μS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; μg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Time	Gage height, in ft	Water depth, in ft	Dissolved oxygen (top), in mg/L	Dissolved oxygen (bottom), in mg/L	Water temperature (top), in °C	Water temperature (bottom), in °C	pH, in standard units	Specific conductance, in μS/cm	Turbidity, FTU
M5	04/17/1997	1400	29.27	--	12.5	--	16.8	--	8.42	318	30
M5	05/05/1997	--	--	--	8.4	8.1	19.9	20.3	8.22	478	--
M5	05/21/1997	1015	29.59	--	1.4	1.7	19.4	19.1	7.59	494	15
M5	06/10/1997	--	--	--	5.3	4.2	21.3	21.2	7.85	556	5
M5	06/25/1997	--	29.2	--	--	--	--	--	--	--	15
M5	07/08/1997	1215	28.98	1.6	6.3	5.8	27.4	27.4	8.36	349	15
M5	07/29/1997	1120	28.65	2.2	3	3	27.4	26.8	7.18	270	25
M5	08/12/1997	1240	28.37	2.3	3.6	3.4	22.8	22.8	7.57	265	5
M5	08/27/1997	1140	28	2	7	.1	28.3	27.3	8.09	257	20
M5	09/10/1997	1105	27.54	2	7.6	9.8	22.4	22.2	7.7	276	115
M5	09/24/1997	1100	27.49	2	1.9	1.6	17.5	17.5	7.54	372	40
M5	10/14/1997	1100	27.68	--	7.2	--	12	--	7.93	464	25
M5	10/28/1997	1015	27.94	1.5	15.4	15.4	6	--	7.25	644	20
M5	11/19/1997	1215	27.86	--	18	17.5	5.5	5.7	7.62	639	10
M5	12/02/1997	--	28.3	--	10.2	--	6.3	--	7.45	585	15
M5	12/19/1997	1045	--	2	--	--	4	--	7.4	--	10
M6	09/10/1997	1330	--	--	--	--	--	--	8.5	285	17
M6	09/24/1997	--	--	1.5	3.9	3.5	17	17	7.6	304	30
M6	10/14/1997	1355	27.68	.7	16.2	--	12.7	--	8.08	896	10
M6	11/19/1997	1445	27.86	1.2	14.3	--	9	--	7.96	849	10
M6	12/02/1997	1115	--	--	8	--	5.9	--	6.98	783	15
M6	12/19/1997	1300	--	2	--	--	4.4	--	7.25	--	10

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Color	Eh, in millivolts	Total nitrogen, in mg/L	Total dissolved nitrogen, in mg/L	Total suspended nitrogen, in mg/L	Total ammonium nitrogen, in mg/L	Dissolved ammonium nitrogen, in mg/L	Suspended ammonium nitrogen, in mg/L	Nitrite plus nitrate, in mg/L	Total ammonium plus organic nitrogen, in mg/L
M1	03/17/1997	--	--	--	--	--	--	--	--	--	--
M1	03/31/1997	10	364	--	--	--	--	--	--	--	--
M1	04/17/1997	2	354	0.53	0.35	0.18	0.015	0.005	0.01	0.11	0.42
M1	05/05/1997	7	--	.77	.42	.35	.031	<.005	.029	<.005	.77
M1	05/21/1997	7	322	.75	.48	.27	.10	<.005	.10	.008	.74
M1	06/10/1997	0	365	.43	.3	.13	.015	.015	0	<.005	.43
M1	06/25/1997	0	--	.42	.4	.02	.0025	<.005	0	.024	.40
M1	07/08/1997	0	--	.41	.29	.12	.02	.017	.003	<.005	.41
M1	07/29/1997	2	378	.7	.62	.08	.011	.007	.004	<.005	.70
M1	08/12/1997	0	308	.58	.3	.28	.03	.028	.002	<.005	.58
M1	12/02/1997	17	--	.65	.49	.16	.046	.005	.041	.24	.40
M1	12/19/1997	0	--	--	--	--	--	.067	0	.07	--
M2	11/15/1996	22	390	.9	.5	.4	.18	<.005	.18	.01	.89
M2	12/02/1996	20	377	.67	.39	.28	.11	.017	.096	.007	.66
M2	12/19/1996	33	458	.52	.42	.1	.039	<.005	.037	<.005	.52
M2	01/03/1997	14	371	.35	.24	.11	.048	<.005	.046	<.005	.35
M2	01/20/1997	--	375	--	.92	0	--	.211	0	.026	--
M2	02/05/1997	6	390	.61	.48	.13	.052	.014	.038	.054	.56
M2	02/17/1997	2	410	.51	.38	.13	.006	.022	0	<.005	.51
M2	02/24/1997	15	--	2.93	.92	2.01	.16	.011	.15	.39	2.54
M2	03/17/1997	11	--	.88	.25	.63	<.005	<.005	0	<.005	.88
M2	03/31/1997	13	385	.91	.5	.41	<.005	<.005	0	<.005	.91
M2	04/17/1997	85	357	1.34	.54	.8	.15	.006	.15	.19	1.15
M2	05/05/1997	5	--	1.01	.44	.57	.12	<.005	.12	<.005	1.01
M2	05/21/1997	9	355	1.32	.83	.49	.32	.17	.15	<.005	1.32
M2	06/10/1997	15	368	1.08	.8	.28	.012	.008	.004	<.005	1.08
M2	06/25/1997	13	--	.94	.83	.11	<.005	<.005	0	.016	.92
M2	07/08/1997	16	--	.51	.84	0	.031	.011	.02	<.005	.51
M2	07/29/1997	16	368	1.5	.86	.64	.047	.035	.012	<.005	1.50
M2	08/12/1997	12	308	2.12	1.1	1.02	.038	.072	0	<.005	2.12

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Color	Eh, in millivolts	Total nitrogen, in mg/L	Total dissolved nitrogen, in mg/L	Total suspended nitrogen, in mg/L	Total ammonium nitrogen, in mg/L	Dissolved ammonium nitrogen, in mg/L	Suspended ammonium nitrogen, in mg/L	Nitrite plus nitrate, in mg/L	Total ammonium plus organic nitrogen, in mg/L
M2	08/27/1997	20	401	2.77	1.69	1.08	0.28	0.14	0.13	0.006	2.76
M2	09/10/1997	0	357	1.84	1.1	.74	.012	<.005	.01	<.005	1.84
M2	09/24/1997	26	387	1.85	1.14	.71	.24	.17	.071	.006	1.84
M2	10/14/1997	24	390	.62	.76	0	.01	.16	0	<.005	.62
M2	10/28/1997	19	365	.68	.54	.14	.023	.017	.006	.012	.67
M2	11/19/1997	8	395	.74	.65	.09	.038	<.005	.036	<.005	.74
M2	11/19/1997	--	--	--	--	--	--	--	--	--	--
M2	12/02/1997	35	--	.85	.63	.22	.20	<.005	.20	.006	.84
M2	12/19/1997	13	--	.6	.41	.19	.017	<.005	.015	<.005	.60
M3	11/15/1996	22	360	.66	.46	.2	.045	<.005	.043	.008	.65
M3	11/25/1996	--	380	.44	.39	.05	.037	<.005	.035	.009	.43
M3	12/18/1996	26	410	.52	.48	.04	.021	--	--	--	.52
M3	01/03/1997	20	377	.67	.59	.08	.022	<.005	.02	<.005	.67
M3	01/20/1997	42	378	1.28	1.26	.02	.13	<.005	.13	.034	1.25
M3	02/05/1997	9	421	.73	.5	.23	.066	.02	.046	.008	.72
M3	02/17/1997	18	414	.63	.39	.24	.037	<.005	.035	<.005	.63
M3	02/24/1997	19	--	.69	.43	.26	.16	.009	.15	.041	.65
M3	03/17/1997	2	--	.69	.32	.37	.028	<.005	.026	<.005	.69
M3	03/31/1997	7	--	.85	.5	.35	.022	<.005	.02	<.005	.85
M3	04/17/1997	75	370	.91	.74	.17	.13	<.005	.12	<.005	.91
M3	05/05/1997	5	--	.94	.53	.41	<.005	<.005	.023	<.005	.94
M3	05/21/1997	10	--	.96	.66	.3	.028	<.005	.026	<.005	.96
M3	06/09/1997	8	370	1.06	.76	.3	.009	<.005	.007	<.005	1.06
M3	06/25/1997	9	--	.94	.74	.2	.01	<.005	.008	<.005	.94
M3	07/08/1997	11	--	.99	.67	.32	.022	.009	.013	<.005	.99
M3	07/29/1997	6	352	1.35	.81	.54	.027	<.005	.025	<.005	1.35
M3	08/12/1997	5	382	1.11	.89	.22	.045	.012	.033	<.005	1.11
M3	08/27/1997	0	--	1.51	.95	.56	.14	<.005	.14	<.005	1.51
M3	09/10/1997	20	383	1.68	.99	.69	.13	.012	.11	<.005	1.68
M3	09/24/1997	24	368	1.9	1.34	.56	.41	.31	.098	.058	1.84
M3	10/14/1997	18	360	1.11	.82	.29	.12	.089	.028	.05	1.06

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Color	Eh, in millivolts	Total nitrogen, in mg/L	Total dissolved nitrogen, in mg/L	Total suspended nitrogen, in mg/L	Total ammonium nitrogen, in mg/L	Dissolved ammonium nitrogen, in mg/L	Suspended ammonium nitrogen, in mg/L	Nitrite plus nitrate, in mg/L	Total ammonium plus organic nitrogen, in mg/L
M3	10/28/1997	17	352	0.9	0.74	0.16	0.095	<0.005	0.093	<0.005	0.90
M3	11/19/1997	8	389	.74	.77	0	.13	.014	.11	.064	.68
M3	12/02/1997	33	--	1.09	.95	.14	.11	.007	.10	.018	1.07
M3	12/19/1997	8	--	.85	.55	.3	.026	.019	.007	<.005	.85
M4	05/21/1997	11	401	.89	.68	.21	.024	<.005	.022	<.005	.89
M4	06/10/1997	15	365	.98	.82	.16	.018	.006	.012	<.005	.98
M4	06/25/1997	23	--	1.24	.94	.3	.11	.065	.041	<.005	1.24
M4	07/08/1997	10	379	.97	.77	.2	.03	.014	.016	<.005	.97
M4	07/29/1997	16	374	1.06	.7	.36	.033	.016	.017	<.005	1.06
M4	08/12/1997	15	387	1.18	.74	.44	.043	.029	.014	<.005	1.18
M4	08/27/1997	15	381	1.21	.96	.25	.079	.05	.029	<.005	1.21
M4	10/28/1997	20	365	.95	.77	.18	.071	<.005	.069	<.005	.95
M4	11/19/1997	8	406	.67	.72	0	.063	<.005	.061	<.005	.67
M4	12/02/1997	10	--	1.29	1.16	.13	.13	.011	.12	<.005	1.29
M4	12/19/1997	6	--	.88	.67	.21	.087	.011	.076	<.005	.88
M5	11/15/1996	25	360	1.4	1	.4	.34	.2	.14	.13	1.26
M5	11/25/1996	--	--	--	--	--	--	--	--	--	--
M5	12/02/1996	22	399	.84	.49	.35	.22	<.005	.22	<.005	.84
M5	12/19/1996	43	473	.92	.74	.18	.11	<.005	.11	<.005	.92
M5	01/03/1997	14	367	.95	.45	.5	.37	<.005	.37	<.005	.95
M5	01/20/1997	--	377	.81	.55	.26	.0025	<.005	0	<.005	.81
M5	02/05/1997	6	410	.46	.32	.14	.043	<.005	.041	.044	.42
M5	02/17/1997	10	411	.64	.42	.22	.06	<.005	.058	<.005	.64
M5	02/21/1997	7	--	.59	.31	.28	.059	<.005	.057	.026	.56
M5	02/24/1997	9	--	.96	.36	.6	.087	<.005	.085	.061	.90
M5	03/17/1997	7	--	1.17	.42	.75	.019	<.005	.017	<.005	1.17
M5	03/31/1997	--	--	1.75	.78	.97	.078	<.005	.076	<.005	1.75
M5	04/05/1997	--	--	1.35	.62	.73	--	.008	0	<.005	1.35
M5	04/10/1997	--	--	1.22	.81	.41	--	.12	0	.02	1.2
M5	04/11/1997	--	--	.79	.67	.12	--	.051	0	.077	.71

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Color	Eh, in millivolts	Total nitrogen, in mg/L	Total dissolved nitrogen, in mg/L	Total suspended nitrogen, in mg/L	Total ammonium nitrogen, in mg/L	Dissolved ammonium nitrogen, in mg/L	Suspended ammonium nitrogen, in mg/L	Nitrite plus nitrate, in mg/L	Total ammonium plus organic nitrogen, in mg/L
M5	04/17/1997	11	348	1.04	0.34	0.7	0.081	0.028	0.053	0.20	0.84
M5	05/05/1997	--	--	1.61	.47	1.14	.031	<.005	.029	<.005	1.61
M5	05/21/1997	1	461	.97	.68	.29	.026	<.005	.024	<.005	.97
M5	06/10/1997	0	365	.88	.68	.2	.017	.008	.009	<.005	.88
M5	06/25/1997	3	--	.79	.73	.06	.1	<.005	.098	<.005	.79
M5	07/08/1997	3	376	.48	.84	0	.017	.009	.008	<.005	.48
M5	07/29/1997	0	352	1.05	.82	.23	.04	.016	.024	<.005	1.05
M5	08/12/1997	5	379	1.21	1.07	.14	.12	.084	.034	<.005	1.21
M5	08/27/1997	0	346	1.9	.92	.98	.11	<.005	.11	<.005	1.90
M5	09/10/1997	2	375	3	.85	2.15	.69	<.005	.69	<.005	3.00
M5	09/24/1997	22	340	2.99	1.92	1.07	1.32	.93	.39	.012	2.98
M5	10/14/1997	25	370	1.35	1.36	0	.59	.54	.046	.048	1.30
M5	10/28/1997	13	356	.84	.86	0	.13	.071	.058	.041	.80
M5	11/19/1997	10	389	.86	.72	.14	.15	<.005	.14	.06	.8
M5	12/02/1997	27	--	.96	.67	.29	.02	.011	.009	.01	.95
M5	12/19/1997	3	--	.92	.66	.26	.028	<.005	.026	<.005	.92
M6	09/10/1997	2	342	2.2	1.11	1.09	.19	<.005	.192	<.005	2.20
M6	09/24/1997	24	389	2.21	1.08	1.13	.30	.18	.124	<.005	2.21
M6	10/14/1997	23	377	.7	.85	0	.009	.016	0	<.005	.70
M6	11/19/1997	8	390	.8	.77	.03	.11	<.005	.108	.19	.61
M6	12/02/1997	35	--	1.34	.62	.72	.049	<.005	.047	.024	1.32
M6	12/19/1997	18	--	.58	.46	.12	.018	<.005	.016	<.005	.58

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Total organic nitrogen, in mg/L	Dissolved organic nitrogen, in mg/L	Total phosphorus, in mg/L	Dissolved phosphorus, in mg/L	Soluable reactive phosphorus, in mg/L	Suspended phosphorus, in mg/L	Chlorophyll mean, in µg/L	Total suspended solids, in mg/L	Volatile suspended solids, in mg/L	Dissolved organic carbon, in mg/L
M1	03/17/1997	--	--	--	--	--	0	--	--	--	--
M1	03/31/1997	--	--	--	--	--	0	--	--	--	--
M1	04/17/1997	0.408	0.238	0.074	0.038	0.033	.036	--	5.2	1.4	2.9
M1	05/05/1997	.737	.415	.147	.042	.031	.105	--	16	5.6	5.7
M1	05/21/1997	.639	.47	.118	.066	.057	.052	--	12.9	4	5.1
M1	06/10/1997	.413	.283	.247	.057	.043	.19	--	40.8	7.2	--
M1	06/25/1997	.394	.374	.138	.055	.046	.083	--	10	2	4.5
M1	07/08/1997	.388	.271	.125	.065	.061	.06	--	3.2	1.7	--
M1	07/29/1997	.687	.611	.119	.045	.036	.074	--	2.7	2	--
M1	08/12/1997	.548	.27	.211	.036	.032	.175	30.6	10.5	3.5	--
M1	12/02/1997	.359	.24	.739	.587	.608	.152	8.2	23	3	--
M1	12/19/1997	--	--	--	.106	.054	0	11.7	--	--	--
M2	11/15/1996	.706	.488	.070	.019	.015	.051	--	--	--	7.1
M2	12/02/1996	.550	.366	.061	.018	.006	.043	--	--	--	6.3
M2	12/19/1996	.479	.415	.034	.018	.011	.016	--	--	--	8.2
M2	01/03/1997	.300	.235	.101	.060	.071	.041	--	--	--	--
M2	01/20/1997	--	--	--	.020	.009	0	--	--	--	--
M2	02/05/1997	.504	.412	.097	.025	.017	.072	--	--	--	5.6
M2	02/17/1997	.502	.356	.055	.015	.012	.04	--	--	--	5.3
M2	02/24/1997	2.374	.516	.366	.052	.046	.314	--	--	--	--
M2	03/17/1997	.875	.245	.116	.012	.006	.104	--	--	--	3.6
M2	03/31/1997	.905	.495	.108	.029	.020	.079	--	--	--	6.5
M2	40/17/1997	.999	.346	.189	.014	.008	.175	--	56.6	26.2	4.2
M2	05/05/1997	.887	.435	.159	.019	.005	.14	--	70	12	5.8
M2	05/21/1997	.999	.658	.147	.035	.022	.112	--	59.6	10	6.6
M2	06/10/1997	1.07	.79	.328	.111	.074	.217	--	350	226	--
M2	06/25/1997	.922	.812	.150	.095	.072	.055	--	14	4.7	10
M2	07/08/1997	.477	.827	.151	.04	.026	.111	--	16.4	5.2	--
M2	07/29/1997	1.45	.823	.224	.049	.030	.175	38.5	21	9	--
M2	08/12/1997	2.08	1.03	.274	.024	.012	.25	83.4	19	8	--

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Total organic nitrogen, in mg/L	Dissolved organic nitrogen, in mg/L	Total phosphorus, in mg/L	Dissolved phosphorus, in mg/L	Soluable reactive phosphorus, in mg/L	Suspended phosphorus, in mg/L	Chlorophyll mean, in µg/L	Total suspended solids, in mg/L	Volatile suspended solids, in mg/L	Dissolved organic carbon, in mg/L
M2	08/27/1997	2.49	1.54	0.316	0.028	0.007	0.288	85.5	66.8	19.2	--
M2	09/10/1997	1.83	1.10	.216	.034	.015	.182	76	14.5	7.5	--
M2	09/24/1997	1.60	.965	.156	.061	.035	.095	72.7	19.5	7.5	--
M2	10/14/1997	.608	.597	.128	.044	.034	.084	5.4	10.2	3.5	--
M2	10/28/1997	.645	.511	.098	.03	.022	.068	5.3	11	4	--
M2	11/19/1997	.700	.645	.057	.019	.008	.038	10.2	10.4	3.6	--
M2	11/19/1997	--	--	--	--	--	0	--	--	--	--
M2	12/02/1997	.645	.622	.373	.254	.239	.119	18.2	16	4	--
M2	12/19/1997	.581	.405	.142	.066	.030	.076	10.1	20	4.4	--
M3	11/15/1996	.607	.45	.036	.014	.014	.022	--	--	--	6.9
M3	11/25/1996	.394	.379	.030	.015	--	.015	--	--	--	4.5
M3	12/18/1996	.499	.480	--	--	--	0	--	--	--	3.2
M3	01/03/1997	.646	.585	.043	.017	.011	.026	--	--	--	8.3
M3	01/20/1997	1.11	1.22	.084	.046	.063	.038	--	--	--	12.1
M3	02/05/1997	.656	.472	.062	.039	.028	.023	--	--	--	6.4
M3	02/17/1997	.591	.385	.070	.021	.010	.049	--	--	--	5.8
M3	02/24/1997	.492	.38	.111	.015	.007	.096	--	--	--	--
M3	03/17/1997	.66	.315	.062	.014	.005	.048	--	--	--	4.8
M3	03/31/1997	.826	.495	.071	.02	.006	.051	--	--	--	5.8
M3	04/17/1997	.782	.735	.134	.012	.004	.122	--	28.8	6.8	4.3
M3	05/05/1997	.913	.525	.157	.030	.017	.127	--	45	11	6.4
M3	05/21/1997	.93	.655	.149	.030	.016	.119	--	85.2	14	7.2
M3	06/09/1997	1.05	.755	.142	.051	.021	.091	--	26	7	--
M3	06/25/1997	.928	.735	.088	.043	.026	.045	--	6.3	2.7	8.7
M3	07/08/1997	.966	.659	.092	.030	.018	.062	--	22.5	16.7	--
M3	07/29/1997	1.32	.805	.214	.047	.033	.167	30.1	95	15.5	--
M3	08/12/1997	1.06	.876	.134	.034	.024	.1	47	26	6	--
M3	08/27/1997	1.37	.945	.165	.043	.026	.122	42.7	24.4	7.6	--
M3	09/10/1997	1.55	.976	.232	.058	.033	.174	67.5	36	2	--
M3	09/24/1997	1.43	.972	.208	.089	.036	.119	84.4	39.5	10.5	--
M3	10/14/1997	.943	.681	.126	.030	.019	.096	5.3	31.3	5.5	--

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Total organic nitrogen, in mg/L	Dissolved organic nitrogen, in mg/L	Total phosphorus, in mg/L	Dissolved phosphorus, in mg/L	Soluable reactive phosphorus, in mg/L	Suspended phosphorus, in mg/L	Chlorophyll mean, in µg/L	Total suspended solids, in mg/L	Volatile suspended solids, in mg/L	Dissolved organic carbon, in mg/L
M3	10/28/1997	0.803	0.735	0.100	0.022	0.009	0.078	16.4	17.5	4	--
M3	11/19/1997	.549	.692	.066	.020	.005	.046	17.2	8.4	3.6	--
M3	12/02/1997	.963	.925	.211	.169	.155	.042	8.4	4.8	2	--
M3	12/19/1997	.822	.529	.092	.047	.019	.045	8.6	9.6	3.2	--
M4	05/21/1997	.864	.675	.084	.034	.02	.05	--	12	3	7.8
M4	06/10/1997	.96	.812	.271	.164	.128	.107	--	13.7	6	--
M4	06/25/1997	1.13	.873	.164	.102	.070	.062	--	43.3	9.7	10
M4	07/08/1997	.938	.754	.069	.030	.017	.039	--	--	--	--
M4	07/29/1997	1.02	.682	.117	.043	.027	.074	32.4	5.7	4.7	--
M4	08/12/1997	1.13	.709	.104	.026	.015	.078	22.2	14.7	11.3	--
M4	08/27/1997	1.13	.908	.159	.034	.023	.125	9.4	3.6	29.2	--
M4	10/28/1997	.877	.765	.142	.031	.016	.111	21.6	27	20	--
M4	11/19/1997	.605	.715	.076	.020	.007	.056	18.8	17.6	13.2	--
M4	12/02/1997	1.15	1.15	.300	.238	.229	.062	10	11	7	--
M4	12/19/1997	.791	.657	.093	.043	.016	.05	11.7	13.5	10	--
M5	11/15/1996	.921	.665	.114	.018	.018	.096	--	--	--	7.1
M5	11/25/1996	--	--	--	--	--	--	--	--	--	--
M5	12/02/1996	.62	.485	.072	.021	.007	.051	--	--	--	6.8
M5	12/19/1996	.803	.735	.069	.017	.002	.052	--	--	--	6.8
M5	01/03/1997	.578	.445	.037	.015	.008	.022	--	--	--	6.7
M5	01/20/1997	.805	.545	.070	.020	.008	.05	--	--	--	8.2
M5	02/05/1997	.373	.274	.053	.018	.010	.035	--	--	--	4
M5	02/17/1997	.578	.415	.090	.013	.006	.077	--	--	--	5.6
M5	02/21/1997	.505	.282	.130	.017	.014	.113	--	--	--	--
M5	02/24/1997	.812	.297	.087	.016	.011	.071	--	--	--	--
M5	03/17/1997	1.15	.415	.141	.170	.004	0	--	--	--	--
M5	03/31/1997	1.67	.775	.480	.030	.010	.454	--	--	--	--
M5	04/05/1997	1.35	.61	.235	.031	.013	.204	--	--	--	--
M5	04/10/1997	1.2	.67	.150	.038	.020	.112	--	38.6	5.7	--
M5	04/11/1997	.713	.542	.069	.034	.019	.035	--	4	2.3	--

Table 3. Analyses of surface-water samples collected from Little Bean Marsh.—Continued

[ft, feet; mg/L, milligrams per liter; °C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; FTU, formazine turbidity units; Eh, oxidation-reduction potential; µg/L, microgram per liter; --, no data; <, less than]

Site number (fig. 1)	Date	Total organic nitrogen, in mg/L	Dissolved organic nitrogen, in mg/L	Total phosphorus, in mg/L	Dissolved phosphorus, in mg/L	Soluable reactive phosphorus, in mg/L	Suspended phosphorus, in mg/L	Chlorophyll mean, in µg/L	Total suspended solids, in mg/L	Volatile suspended solids, in mg/L	Dissolved organic carbon, in mg/L
M5	04/17/1997	0.761	0.114	0.016	0.016	0.005	0	--	50.4	10.4	--
M5	05/05/1997	1.58	.465	.268	.028	.012	.24	--	62	22	--
M5	05/21/1997	.942	.675	.100	.036	.021	.064	--	15	5.3	--
M5	06/10/1997	.861	.67	.059	.052	.028	.007	--	25.4	5.4	--
M5	06/25/1997	.688	.725	.100	.044	.025	.056	--	22.3	5	8.7
M5	07/08/1997	.461	.829	.154	.031	.019	.123	--	54.5	10	--
M5	07/29/1997	1.01	.802	.144	.036	.022	.108	33.8	29	6.5	--
M5	08/12/1997	1.09	.984	.16	.033	.022	.127	40.1	28	5.3	--
M5	08/27/1997	1.78	.915	.237	.033	.017	.204	104.5	44	12.4	--
M5	09/10/1997	2.31	.845	.604	.023	.011	.581	885.6	112	44	--
M5	09/24/1997	1.66	.979	.409	.205	.058	.204	86.5	88	16	--
M5	10/14/1997	.716	.772	.160	.031	.016	.129	13.7	35.2	7.2	--
M5	10/28/1997	.670	.748	.111	.023	.009	.088	12.7	16	4	--
M5	11/19/1997	.654	.658	.068	.02	.005	.048	21.5	9.2	2.8	--
M5	12/02/1997	.930	.649	.170	.086	.072	.084	14.2	14	6	--
M5	12/19/1997	.890	.655	.110	.05	.02	.06	14.7	10.9	3.2	--
M6	09/10/1997	2.00	1.10	.220	.045	.018	.175	45.2	26	14	--
M6	09/24/1997	1.90	.897	.278	.113	.033	.165	139	50	16	--
M6	10/14/1997	.689	.832	.084	.039	.024	.045	3.1	3	1.6	--
M6	11/19/1997	.502	.58	.092	.031	.01	.061	20.6	28	6	--
M6	12/02/1997	1.27	.594	.247	.163	.152	.084	17	11	4	--
M6	12/19/1997	.56	.455	.127	.068	.032	.059	7	14	2.8	--

---

[1] Depth below ice.

For additional information contact:
U.S. Geological Survey
Water Resources Discipline
1400 Independence Road
Mail Stop 100
Rolla, MO 65401
(573) 308-3664 or
http://mo.water.usgs.gov

USGS

Dale W. Blevins—**Hydrology and Cycling of Nitrogen and Phosphorus in Little Bean Marsh: A Remnant Riparian Wetland along the Missouri River in Platte County, Missouri, 1996–97**—Scientific Investigations Report 2004–5171

125 years of science for America 1879–2004