In cooperation with the Missouri Department of Natural Resources,
Division of Environmental Quality

Water Quality, Hydrology, and Invertebrate Communities of Three Remnant Wetlands in Missouri, 1995–97

Water-Resources Investigations Report 98–4190

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

Cover photograph: Spile Lake near Horton, Missouri. Photo courtesy of A.R. Walter, U.S. Geological Survey.

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By David C. Heimann and Suzanne R. Femmer

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Rolla, Missouri
1998

U.S. DEPARTMENT OF THE INTERIOR
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CONVERSION FACTORS AND VERTICAL DATUM

Multiply	By	To obtain
Length		
inch (in.)	2.54	centimeter
inch (in.)	25.4	millimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
Area		
acre	4,047	square meter
acre	0.4047	hectare
acre	0.4047	square hectometer
acre	0.004047	square kilometer
square foot (ft^2)	0.09290	square meter
square mile (mi^2)	259.0	hectare
Volume		
cubic foot (ft^3)	28.32	cubic liter
cubic foot (ft^3)	0.02832	cubic meter
acre-foot (acre-ft)	1,233	cubic meter
acre-foot (acre-ft)	0.001233	cubic hectometer
gallon (gal)	4.20	liters
Flow rate		
cubic foot per second (ft^3/s)	0.02832	cubic meter per second
Mass		
ounce, avoirdupois (oz)	28.35	gram
pound, avoirdupois (lb)	0.4536	kilogram
Density		
pound per cubic foot (lb/ft^3)	16.02	kilogram per cubic meter
pound per cubic foot (lb/ft^3)	0.01602	gram per cubic centimeter

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F = (1.8 \times °C) + 32$$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$$°C = (°F - 32) / 1.8$$

Sea level: In this report, "sea level" refers to the National Geodetic Vertical Datum of 1929 (NGVD of 1929)—a geodetic datum derived from a general adjustment of the first-order level nets of both the United States and Canada, formerly called Sea Level Datum of 1929.

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ABSTRACT

This report presents the results of a study conducted by the U.S. Geological Survey in cooperation with the Missouri Department of Natural Resources from December 1995 through May 1997 to describe the water quality, hydrologic, and invertebrate characteristics of three remnant wetlands. These data may be used to help develop selected water-quality standards for wetlands in Missouri. Wetlands monitored in this study include Spile Lake, Vernon County; Little Bean Marsh, Platte County; and Forker Oxbow, Linn County, Missouri.

Extremes in physicochemical properties in these wetlands were greatly affected by thermal stratification, hydrologic fluctuations, biological activity, and ice formation. The wetlands had dissolved-oxygen concentrations below the 5-milligrams-per-liter State water-quality standard from 40 to 60 percent of a selected 1-year period, corresponding to periods of thermal stratification. Hydrologic fluctuations were common as the water-surface elevation changes in these systems ranged up to 12 feet during the course of the study. Photosynthesis and respiration are likely causes of diurnal fluctuations in pH and dissolved oxygen throughout the study period, but particularly in the summer months. Periods of ice formation were short lived in the wetlands, but corresponded with maximum values of specific conductance and dissolved oxygen in all three systems.

Analyses of invertebrate results using the Jaccard Coefficient of Community Similarity indicated mixed results. Woody snag sample results showed little similarities between sites, while sweep net sample results indicated similarities existed. Most of the families detected at these sites are considered organic tolerant as indicated by the Hilsenhoff Biotic Index. Analysis of the dominant taxon indicates that one or two invertebrate families that are tolerant to organic enrichment generally dominate the wetlands.

The hydrologic, water quality, and invertebrate information analyzed in this study indicate that while there are similarities among wetlands, these are unique systems. The statistical comparisons between water-quality constituents in wetlands and streams indicate dissimilarities are common. Including the presence of thermal stratification in these wetlands, the exclusions and modifications in State standards that are applied to lakes and reservoirs also may be applicable.

INTRODUCTION

Wetlands are defined and delineated by their soil, hydrologic, and vegetation features and incorporate characteristics of both terrestrial and aquatic ecosystems. These overlapping areas of terrestrial and aquatic communities result in some of the most productive natural ecosystems in the world (Lieth, 1975; Richardson, 1979). Wetlands historically have been a major component of the Missouri landscape, covering some 4.8 million acres, or nearly 11 percent, of the surface area of the State in the 1780's (Dahl, 1990). About 87 percent of the wetland area in Missouri was drained and filled by the 1980's.

Water-quality standards for waters in Missouri were adopted in Chapter 644 of the Missouri Clean Water Law of 1973 (Missouri Department of Natural Resources, 1994). Revisions to this law specify that wetlands adjacent to streams are protected by the same criteria as the stream. Wetlands are unique systems and may have water-quality characteristics that differ from adjacent streams (U.S. Environmental Protection Agency, 1990). A study was undertaken by the U.S. Geological Survey in cooperation with the Missouri Department of Natural Resources (MDNR) to characterize water quality, hydrology, and invertebrate communities in three selected remnant wetland systems for use in the development of water-quality standards for wetlands in Missouri.

The objectives of the study were to establish reference or benchmark wetlands and to characterize the water quality, hydrology, and invertebrate communities in these systems. Water quality of the wetlands was monitored to characterize temporal variability and central tendencies in physicochemical properties (specific conductance, pH, temperature, dissolved oxygen, and turbidity) and major nutrients. Hydrologic characteristics were monitored to determine temporal variability and central tendencies in stage, area, and storage as well as the hydraulic residence time of these wetland systems. Invertebrate communities were sampled to characterize seasonal variation in composition and diversity.

Purpose and Scope

The purpose of this report is to summarize water quality, hydrologic, and invertebrate data collected at three selected remnant wetlands from December 1995 to May 1997. Water-quality data recorded hourly included specific conductance, pH, temperature, and dissolved oxygen. Turbidity data were collected approximately once every 2 weeks and nutrient data were collected approximately once every month during the monitoring period. Wetland stage (collected hourly) and bathymetry were used to determine temporal changes in wetland wetted surface area and storage, as well as mean residence time. Invertebrate data were collected on four occasions—March, June, August, and October 1996—to provide a biological indicator of ecosystem health and structure.

Previous Investigations

The limited previous investigations of wetlands in Missouri primarily have focused on the biological characteristics of these systems. Investigation of the invertebrate communities in Missouri's natural and managed wetlands include White (1982; 1985), Reid (1983), Batema (1987), Magee (1989), and Magee and others (1993). The water quality of managed forested wetlands in southeast Missouri are described in Wylie (1985) and Wylie and Jones (1986). A comparison of physicochemical properties and constituents in managed Missouri wetlands by physiographic regions was conducted by Hoyer and Reid (1982).

Study Site Location and Description

Three natural wetland sites were selected for inclusion in this study; Spile Lake near Horton; Little Bean Marsh near Bean Lake; and Forker Oxbow near Fountain Grove, Missouri (figs. 1–4). Spile Lake (Vernon County) is located on private property. Little Bean Marsh (Platte County) and Forker Oxbow (Linn County) are located on Missouri Department of Conservation (MDC) property. Site characteristics of the three wetlands are provided in figure 5.

The wetland sites used in this study were selected based on the following criteria: 1) the site is representative of commonly-found and widely-distributed wetland types in Missouri; 2) the sites represent wetlands in different natural divisions of the State (with emphasis on those regions that lacked previous studies); 3) the site is minimally affected by anthropogenic disturbance or control; and 4) the site contains water throughout most years. A list of potential sites was constructed from information provided by MDNR, MDC, Natural Resources Conservation Service, U. S. Fish and Wildlife Service, U.S. Environmental Protection Agency (USEPA), and the Nature Conservancy.

All three sites are riparian systems representing the most common type of wetlands in Missouri (Epperson, 1992). Based on the classification system developed by Cowardin and others (1979) most wetlands in

Figure 1. Location of three study wetlands and natural divisions of Missouri (modified from Nelson, 1987).

Missouri can be classified either as palustrine forested wetlands (swamps and forested wetlands), palustrine emergent wetlands (marshes, wet meadows, fens), palustrine scrub-shrub wetlands (shrub swamps), or lacustrine littoral wetlands (shallows of ponds and lakes) (Demas and Demcheck, 1996). Wetlands also are described in terms of their formation and management status including natural (preserved), constructed, managed, and restored (Epperson, 1992). The study wetlands were likely formed by abandoned river channels, and the hydrologic regimes of these systems are tied closely to the adjacent rivers. The hydrology of Spile Lake is tied closely with the Little Osage and Marmaton Rivers, Little Bean Marsh with the Missouri River, and Forker Oxbow with Locust Creek and the Grand River.

Each wetland was chosen to be representative of a selected natural division in the State (fig. 1). Spile Lake is located in the Osage Plains division, Little Bean Marsh in the Big Rivers division, and Forker Oxbow in the Glaciated Plains division. The purposes of selecting wetlands in different areas of the State were to encompass potential spatial variability and to establish reference wetlands that are representative of different natural divisions.

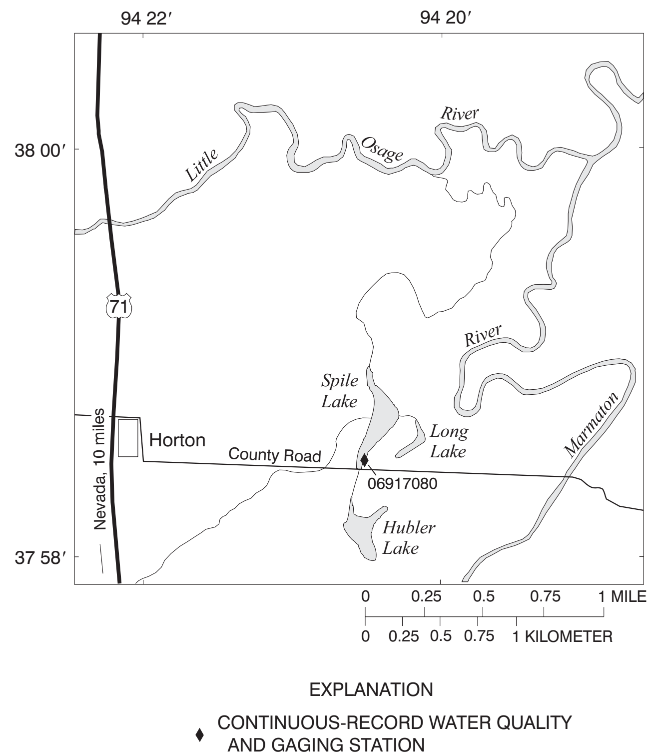


Figure 2. Location of sampling site in Spile Lake near Horton, Missouri, and vicinity.

None of the three study wetlands has outflow control structures, and water levels in these wetlands currently are not actively managed or regulated. Little Bean Marsh is located behind a levee system that has a rudimentary water control gate that can be used to prevent high Missouri River flows from entering Bean Lake (and possibly Little Bean Marsh). Occasionally, Little Bean Marsh receives floodwaters directly from the Missouri River, such as in 1952, 1973, 1983 (Castaner and LaPlante, 1992), and 1993. Although this wetland has some current (1998) and historic anthropogenic influences, Little Bean Marsh has been referred to as “the best natural marsh along the Missouri River” in the State (Thom and Iffrig, 1985).

Many wetlands in Missouri do not contain water throughout a typical year. To take full advantage of the limited monitoring period, only those wetlands which contained water in the dry periods of late summer and early fall of the 1995 selection period were considered. Maximum depths did not exceed 6 ft in the three wetlands during non-flood periods—a criterion used to distinguish wetlands from deepwater aquatic habitats.

The formation dates of the study wetlands are not known, but the location and characteristics of these features indicate that all three wetlands probably were once part of the active channel of the adjacent river(s). Little Bean Marsh was the northern section of a meander cutoff of the Missouri River first documented around 1850 (Castaner and LaPlante, 1992). Agricultural development in the area and a subsequent increase in soil erosion and sedimentation in the upper part of Bean Lake resulted in Little Bean Marsh becoming isolated from Bean Lake in the early 1900’s. Little Bean Marsh was used as a hunting club in the 1950’s and 60’s and some modifications were made to promote open water and waterfowl habitat. In 1981, the MDC purchased and began management of most of what is now Little Bean Marsh. Little is known of the origin and history of Spile Lake and Forker Oxbow. The formation of Spile Lake pre-dates the earliest land surveys from the 1840’s, as Spile Lake was indicated as a marsh on the earliest land survey notes (Missouri Department of Natural Resources, 1843).

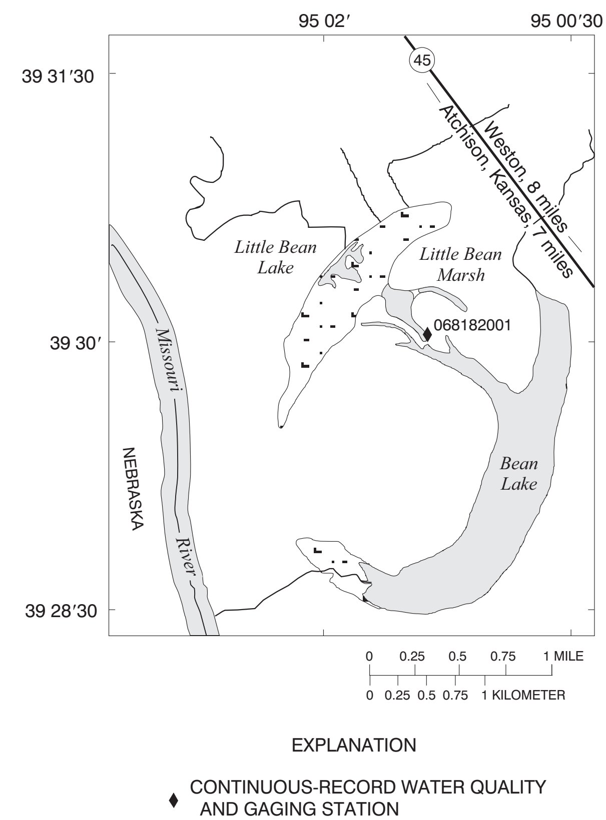


Figure 3. Location of sampling site in Little Bean Marsh near Bean Lake, Missouri, and vicinity.

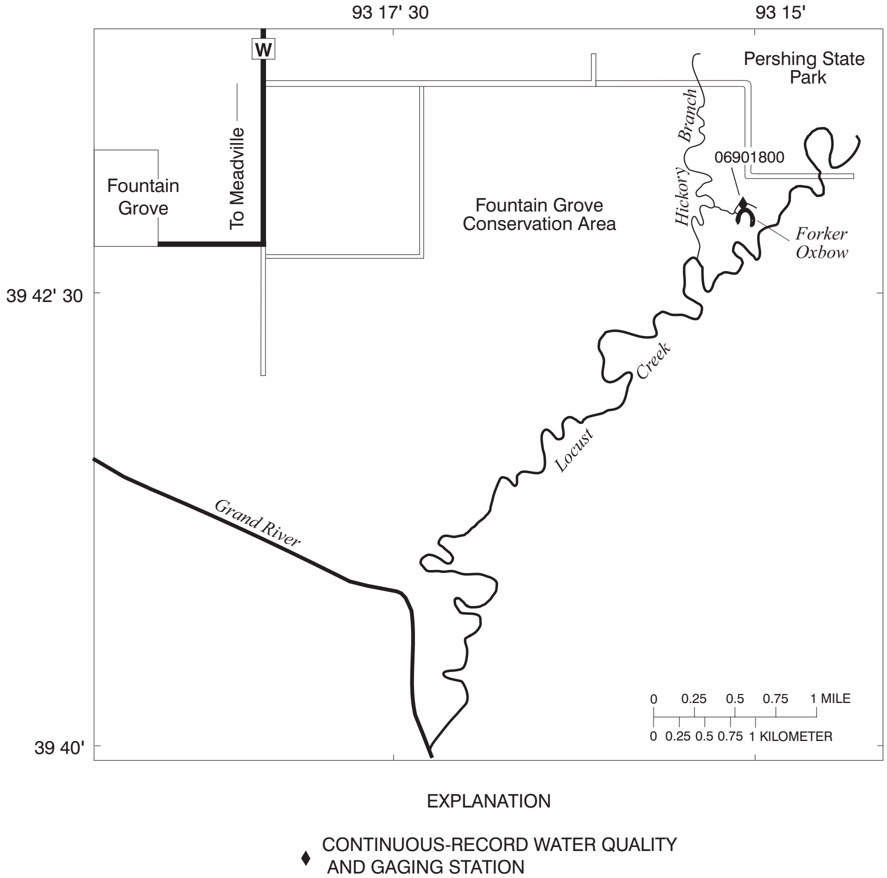


Figure 4. Location of sampling site in Forker Oxbow near Fountain Grove, Missouri, and vicinity.

The water-quality characteristics of the study wetlands are affected not only by local land use but the land-use practices in the drainage area of the adjacent streams and rivers. Land use in the Spile Lake local drainage is upland forest, pasture, and some row crop production (corn and soybeans). The land use in the Little Bean Marsh local drainage is predominantly floodplain row crop production including corn and soybeans. The local drainage area of Forker Oxbow is entirely floodplain forest. All three systems receive floodwaters from the adjacent river during overbank events, and the water quality of these wetlands is, therefore, affected by runoff and land-use practices in the entire drainage area of the adjacent river.

Acknowledgments

The authors wish to express appreciation to William Kline for his permission to access property and install monitoring equipment at Spile Lake. The authors also wish to acknowledge the Missouri Department of Conservation staff at Fountain Grove Conservation Area for their assistance in monitoring efforts at Forker Oxbow.

METHODS

The methods used in the measurement of water quality, hydrology, and invertebrate communities are discussed in the following sections. The methods were used consistently among wetlands and apply throughout the December 1995 through May 1997 monitoring period.

Water Quality

A single monitoring site was established in each wetland. This site corresponded to a relatively deep section of the wetland located within 60 to 75 ft of the bank. A relatively deep monitoring location was chosen to

Spile Lake near Horton, Missouri
Latitude: 37 58′28
Longitude: 094 20′36
Station number: 06917080
Nearby river: Little Osage and Marmaton Rivers
Wetland type: Palustrine, emergent, non-persistent vegetation
Drainage area: 3.32 square miles
Mean area: 44.4 acres
Mean storage: 54.9 acre-feet
Mean depth: 1.24 feet
Dominant vegetation: Spatterdock (*Nuphar luteum*)

Photograph courtesy of E.D. Christensen, U.S. Geological Survey

Little Bean Marsh near Bean Lake, Missouri
Latitude: 39 30′20
Longitude: 095 01′20
Station number: 068182001
Nearby river: Missouri River
Wetland type: Palustrine, emergent, persistent vegetation
Drainage area: 3.87 square miles
Mean area: 183 acres
Mean storage: 148 acre-feet
Mean depth: 0.81 feet
Dominant vegetation: Bulrushes (*Scirpus spp.*), Cattails (*Typha spp.*)

Photograph courtesy of D.W. Blevins, U.S. Geological Survey

Forker Oxbow near Fountain Grove, Missouri
Latitude: 39 42′56
Longitude: 093 15′02
Station number: 06901800
Nearby river: Locust Creek, Grand River
Wetland type: Palustrine, emergent, non-persistent vegetation
Drainage area: 0.03 square miles
Mean area: 4.34 acres
Mean storage: 4.48 acre-feet
Mean depth: 1.03 feet
Dominant vegetation: Water Pepper (*Polygonum hydropiperoides*)

Photograph courtesy of E.D. Christensen, U.S. Geological Survey

Figure 5. Site characteristics of Spile Lake, Little Bean Marsh, and Forker Oxbow in Missouri (wetland type from Cowardin and others, 1979).

ensure adequate water levels for monitoring throughout the study period. Equipment at each monitoring site included a multi-parameter sensing unit capable of measuring specific conductance, pH, temperature, and dissolved oxygen. The multi-parameterunit was protected in a case constructed of 6-in. polyvinyl chloride (PVC) pipe that had numerous holes to allow for adequate water exchange (fig. 6). Two horizontal floats were attached to the side of the vertical case to allow the monitoring unit probes to remain at a fixed depth of 1.5 ft below the water surface. A floating case also allowed the unit to be easily serviced even during high stage periods. At extreme low stages, the monitoring unit was placed in an 18-in.-wide by 36-in.-deep vertical corrugated pipe at the monitoring location. The wetland bottom in the pipe was deepened to ensure that the stage could be monitored during low-water periods. This also allowed the monitoring unit to be set vertically in the culvert without extending into the bottom sediments.The culvert also was provided with holes in the sides to allow for adequate water exchange.

Figure 6. Retrieval of multi-parameter water-quality sensor and protective case at Spile Lake, January 1996. Photograph courtesy of D.H. Wilkison, U.S. Geological Survey.

Physicochemical properties were measured each hour automatically (by the multi-parameter water-quality sensor) and manually (using individual field meters) during maintenance visits. The automatic hourly physicochemical property measurements were stored in a data logger housed onshore. A 3-minute warm-up period preceded each hourly set of measurements. A mechanical stirrer operated during the warm-up period to ensure proper water flow across the dissolved-oxygen sensor, which was equipped with a standard membrane. The hourly circulation also served to retard biofouling of the dissolved-oxygen membrane and other sensors. Temperature was measured at two depths including near-surface, by the multi-parameter water-quality sensor, and near-bottom, by a submersible pressure transducer equipped with a thermistor. The near-bottom temperatures also were recorded hourly. The multi-parameter water-quality sensor was calibrated approximately every 2 weeks at each wetland during the December 1995 to May 1997 monitoring period. The probes were checked against known standards before cleaning and calibration to determine if correction factors were necessary to compensate for fouling of the sensors during use. All probes on the multi-parameter water-quality sensor were then cleaned and calibrated in accordance with the manufacturer's specifications. Measurements of physicochemical properties also were made with individual field meters during the calibration visits, and these measurements were used for the purpose of comparison and possible adjustment of the continuous hourly record. The individual meters were calibrated in the field before use using known standards and procedures in accordance with the manufacturer's specifications.

Water samples were collected from the wetland water column during the calibration visits and were analyzed for turbidity and major nitrogen (N) and phosphorus (P) forms. Turbidity samples were collected from the top 1.5 ft of the water column at the monitoring location during each maintenance visit (approximately every 2 weeks) and analyzed at the U.S. Geological Survey's National Water Quality Laboratory in Arvada, Colorado, in accordance with methods specified in Fishman and Friedman (1989). Turbidity samples were stored in the dark to limit algal growth that could interfere with the laboratory analysis. Water samples also were collected during every other calibration visit (approximately every 4 weeks) and analyzed for dissolved ammonia (NH_3), dissolved nitrite (NO_2), dissolved nitrite plus nitrate (NO_3), total ammonia plus organic N, dissolved P, total P, and dissolved orthophosphorus (PO_4). Duplicate nutrient samples were collected on two occasions for quality assurance purposes. Nutrient samples were analyzed at the National Water Quality Laboratory in Arvada, Colorado, or at the U.S. Geological Survey's Quality Water Services Unit, Ocala, Florida, in accordance with methods described in Fishman and Friedman (1989).

Statistical Analyses

Statistical analyses were conducted comparing water-quality characteristics among wetlands, and also between the wetlands and selected streams in the same natural divisions. The null hypothesis of no significant differences in turbidity and selected nutrient concentrations among the study wetlands was tested using the Kruskal-Wallis test (Helsel and Hirsch, 1992). If a significant difference was found (null hypothesis rejected), a rank transform test (one-way ANOVA on the ranks of the data values) was conducted along with a Tukey's multiple comparison test (Helsel and Hirsch, 1992) to determine which data distributions accounted for the differences. The wetland physicochemical property and nutrient data also were compared with measurements from selected streams using a Mann-Whitney test (Helsel and Hirsch, 1992). The null hypothesis tested was that there was no statistically significant difference in the distributions of selected physicochemical properties and nutrient concentrations between the study wetlands and selected streams in the same natural division. A significance level of 0.05 was used for all statistical tests in this study. Values less than detection were quantified as one-half the detection limit in statistical analyses. The probability of error (p value) is reported with each analysis and provides a quantitative indication of the degree of differences or similarities between the analyzed data sets.

Hydrology

Stage and precipitation were monitored and logged hourly throughout the period. A submersible vented pressure transducer was used to monitor stage and was located in an 18-in. diameter vertically-mounted pipe. The datum of the pressure transducer corresponded to that of a staff gage, which was used as a reference gage for making any necessary corrections in the pressure transducer readings during the study. The staff gage datum was surveyed into the nearest vertical control benchmark allowing for the conversion of the stage readings to water-surface elevations. Hourly precipitation was measured using a tipping-bucket rain gage connected to a data logger at each of the wetlands.

A bathymetric survey was conducted at each wetland to calculate temporal changes in wetland wetted surface areas, volumes, and also to determine mean hydraulic residence times. Wetland bed elevations were surveyed along 10 to 20 transects located perpendicular to the wetland longitudinal axis. The number of survey points taken along each transect and the number of transects were determined by the wetland size and relative change in the bottom-surface elevation. Approximately 500, 1,000, and 300 points were used to define the elevation contours at Spile Lake, Little Bean Marsh, and Forker Oxbow. The wetland bottom-surface elevation contours initially were drawn using a geographic information system (GIS). The computer-generated contours and boundary lines were adjusted, if necessary, by hand to smooth jagged contour lines and to correct for erroneous assumptions made by the contouring software. The final contour information was used to develop relations between water-surface elevation and wetted surface area, as well as elevation and storage for each wetland. The wetland storage information was used to determine a mean hydraulic residence time for each wetland. Mean hydraulic residence time is an indication of the average time it takes for runoff to replace the contents of the wetland and was determined using the following equation:

$$RT = \frac{S}{(DA)(R)} \qquad (1)$$

where

RT is hydraulic residence time, in years;
S is wetland storage, in acre-ft;
DA is drainage area of wetland basin, in acres; and
R is runoff from wetland basin, in ft per year.

Invertebrates

Invertebrate populations were sampled at each wetland on four occasions (March, June, August, and October 1996) using several sampling methodologies. Four different methods were utilized to collect the invertebrate samples including woody snags, sweep net, bottom substrate cores, and Hester-Dendy artificial substrates samples. The snag, core, and Hester-Dendy samples are semi-quantitative (samples are collected over a known area) and the sweep samples are qualitative (samples are collected over a known time, not area).

Invertebrate samples were collected from the surface of woody snags, including branches, roots, and large woody fragments. Two snag samples were collected at each site per sample round and each sample consisted of 5 to 10 woody debris pieces of various sizes and stages of decay. The length and circumference of each debris piece was noted to calculate the total surface area per sample.

Two sweep net samples also were collected at each wetland over a fixed time period during each of the four sample rounds. Sweep samples were collected concurrently along the longitudinal wetland axis using a triangular shaped sweep net with a 425-micron mesh size and an opening area of 0.43 ft^2. All represented habitat types were targeted in the sweep samples. The sampling interval was one hour at Spile Lake for all four sample rounds; one-half hour for the March 1996 sample round at Little Bean Marsh and 1 hour for the three remaining rounds; and one-half hour for all four sample rounds at Forker Oxbow. The interval was increased at Little Bean Marsh after the March 1996 sample round as the surface area of the Marsh increased substantially between the first and remaining sample rounds.

Bottom substrate core samples were collected using a 1.5-in.-diameter stainless steel corer to an approximate depth of 4 in. The core samples were collected at three points along four transects oriented perpendicular to the longitudinal axis of the wetland. The longitudinal axis length was measured as the wetted perimeter length of one side of the wetland from apex to apex. The four sampling transects were located randomly within four equal subdivisions of the perimeter length. The three within-transect sample sets were collected at points roughly corresponding to one-quarter, one-half, and three-quarters the wetted length of the transect. Each of the 3 within-transect sample sets consisted of 8 cores (4 cores in the March 1996 samples) for a total of 24 cores per transect sample (12 cores in the March 1996 samples) and 4 transect samples per wetland.

Artificial substrate samplers (Hester-Dendy samplers) also were used to sample invertebrate populations in each wetland. Ten Hester-Dendy samplers with a surface area of 1.4 ft^2 were placed in each wetland at least 2 weeks before the invertebrate sampling date. This allowed adequate time for the substrate samplers to be colonized before sampling. Two Hester-Dendy samplers were suspended about one foot below each of five floats during the colonization period. The floats consisted of a 2-ft length of 2-in.-diameter PVC pipe filled with expanding foam and sealed with end caps similar in design to that described in Britton and Greeson (1987). Consideration was given in float placement within the wetland to account for factors that may affect invertebrate colonization including water depth and vegetation. The float placement locations remained consistent at each site for each of the four sample rounds.

All invertebrate samples were processed onsite or within 12 hours of collection. Benthic core samples were processed by combining the three within-transect sample sets in a 5-gal bucket, adding water, thoroughly mixing the contents, and filtering the elutriate through a 425-micron sieve. Sieve samples that exceeded 1 L (liter) were subsampled to obtain a maximum sample volume of 1 L. Snag samples were processed by brushing the exterior of the log or branch with a small stiff nylon brush into a 5-gal bucket of water. The bucket contents were then sieved through a 425-micron sieve and split, if necessary. Sweep and core samples were processed by filtering the collected sample material onto a 425-micron sieve and splitting the sample, if necessary, to obtain a maximum sample volume of 1 L. Hester-Dendy samplers were processed by disassembling the sampler in a bucket of water and brushing the colonized material off of each sampler plate and spacer using a small nylon brush. The contents of the bucket were then passed through a 425-micron sieve. All samples were preserved with 10 percent buffered formalin and sent to the Biology Department at Southwest Missouri State University (SMSU), Springfield, Missouri, for analyses. The samples were sieved using a 1.7-mm (millimeter) sieve size with the retained material sorted and sampled completely and the smaller material subsampled prior to analyses. Methods used in the analyses of the invertebrate samples at SMSU were in accordance with those listed in Downing and Rigler (1984). The taxonomic level (class, order) of some analyses of the invertebrate samples from SMSU was unsuitable for the calculation of selected desired metrics. The snag and sweep samples were later sent to BSA Environmental Services (BSA), Shaker Heights, Ohio, for more detailed taxonomic analysis. BSA conducted analyses for invertebrates using methods described in Merritt and Cummins (1984).

HYDROLOGIC CHARACTERISTICS

The hydrology of a wetland system, including the magnitude and timing of inflows and outflows, largely controls its chemical characteristics and is an important component of any wetland water-quality study. Therefore, it is presented first for better interpretation of the water-quality results. All three wetlands appear to be dominated by surface-water inflows and outflows, but the ground-water component of the hydrologic budget of these systems has not been quantified.

Water-Surface Elevation

The dominating affect of the adjacent streams and rivers on the hydrology of these wetlands is evident in water-surface elevation hydrographs from the wetlands for the study period (fig. 7). Spile Lake experienced backwater flooding from the Little Osage or Marmaton Rivers on four occasions during the study—July 1996, November 1996, February 1997, and April 1997. The water-surface elevation extremes at Spile Lake were 735.5 and 744.4 ft, a range of 8.9 ft. Little Bean Marsh experienced a water-surface elevation increase in March 1996 as a result of backwater flow from the Missouri River into Bean Lake and ultimately Little Bean Marsh. The Bean Lake control gates remained closed after this backwater event, and remained closed through September 28, 1996, for maintenance and repair of the Bean Lake outflow control channel. This resulted in artificially elevated water-surface elevation levels in Little Bean Marsh during this period (fig. 7). The extremes in water-surface elevation at Little Bean Marsh were 773.3 and 777.4 ft, a range of 4.1 ft. The levee and gate system regulated inflows and limited the range in water-surface elevation at Little Bean Marsh during much of this study. Forker Oxbow experienced several overbank events from Locust Creek, and in late May 1996, received backwater from the Grand River. Forker Oxbow is located approximately 0.25 mi from Locust Creek and represents one of several cutoff meanders in the Locust Creek floodplain. Post-flood observations indicated that these overbank events resulted in substantial amounts of clay and silt deposition on the Locust Creek floodplain. The velocities through Forker Oxbow during the late May 1996 overbank event on Locust Creek, however, were large enough to relocate the Hester-Dendy weighted floats and may have been high enough to limit sedimentation in this wetland. The overall extremes in Forker Oxbow stages were 656.6 and 668.6 ft, a range of 12.0 ft, representing the largest water-surface elevation range of the three wetlands (fig. 7).

Bathymetry, Area, and Storage

The bathymetric maps of the wetland systems were used to convert water-surface elevation data into wetland wetted surface area and storage, and changes in these hydrologic characteristics, particularly wetland storage, reflect the dynamic nature of these systems. The maximum surface area and storage of the wetlands occurred during flood events from the adjacent river(s). Only the local wetland area near the maximum maintainable pool elevation was included in the bathymetric surveys, so the maximum area and storage values for the wetlands during the floods are not known. Bathymetry data for Spile Lake, Little Bean Marsh, and Forker Oxbow are provided in figures 8 to 10. The wetland vegetation, hydrology, and soils are all used to delineate the actual wetland area (Environmental Laboratory, 1987) but for the purposes of this study, the wetted area is used as an approximation of the wetland area. The extremes in the Spile Lake wetted surface area were 14.3 acres and more than 110 acres for a range of more than 106 acres, or 770 percent (fig. 11). The extremes in the Little Bean Marsh wetted surface area were 21.2 to 303 acres, a range of about 282 acres, or 1,400 percent. The extremes in the wetted area of Forker Oxbow were 0.3 and more than 10.4 acres, a range of more than 10 acres, or about 3,500 percent. The corresponding storage extremes at Spile Lake were 3.7 and more than 282 acre-ft, a range of more than 278 acre-ft, or 7,600 percent (fig. 12). At Little Bean Marsh these extremes were 9.8 and 660 acre-ft, a range of about 650 acre-ft, or 6,700 percent. At Forker Oxbow the extremes in storage were 0.11 and 33.0 acre-ft, a range of more than 32 acre-ft, representing a 30,000 percent change. These results demonstrate how dynamic the hydrologic nature of these riparian wetlands can be and the extremes to which biological communities must adapt.

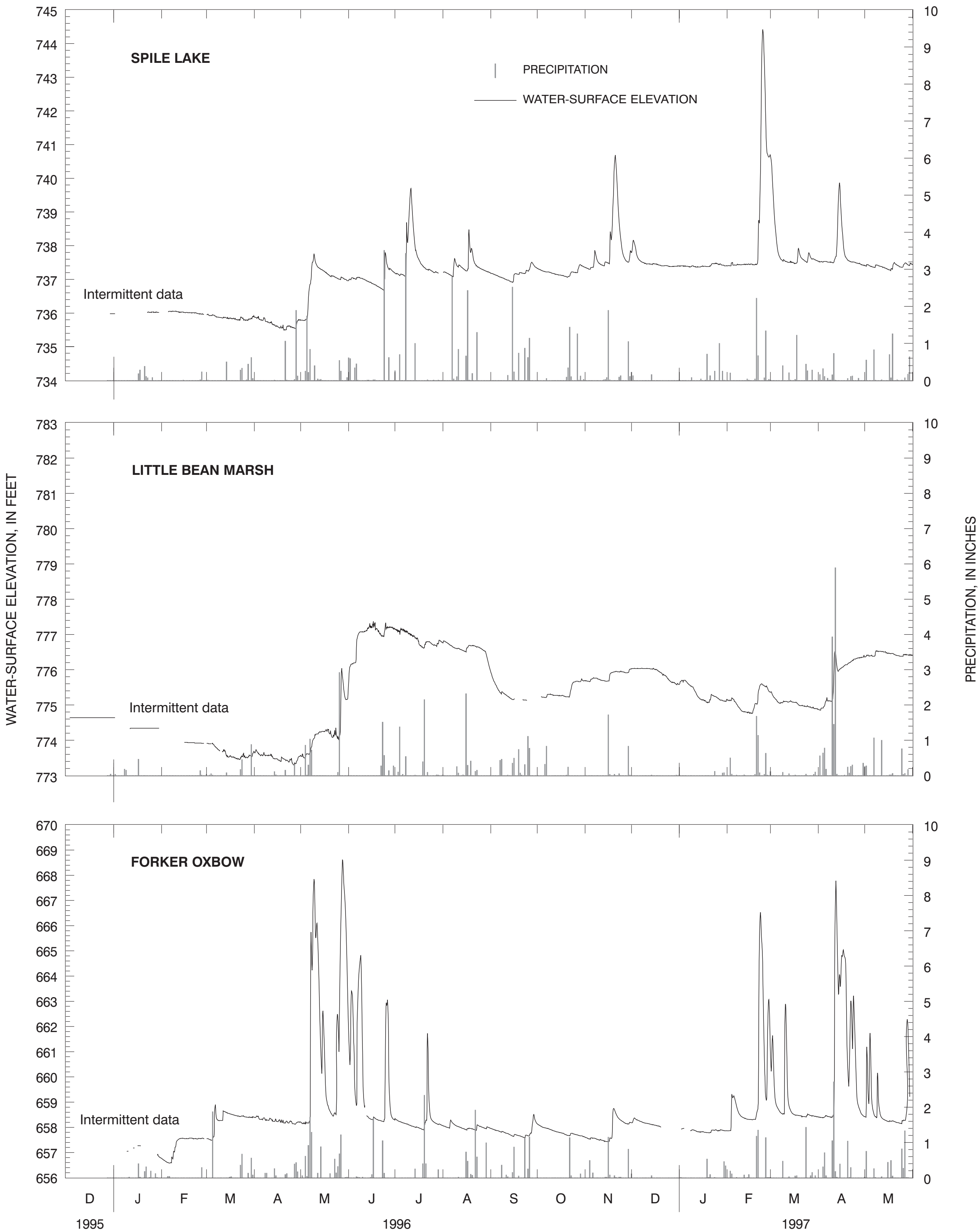


Figure 7. Temporal variability in water-surface elevations in Spile Lake, Little Bean Marsh, and Forker Oxbow.

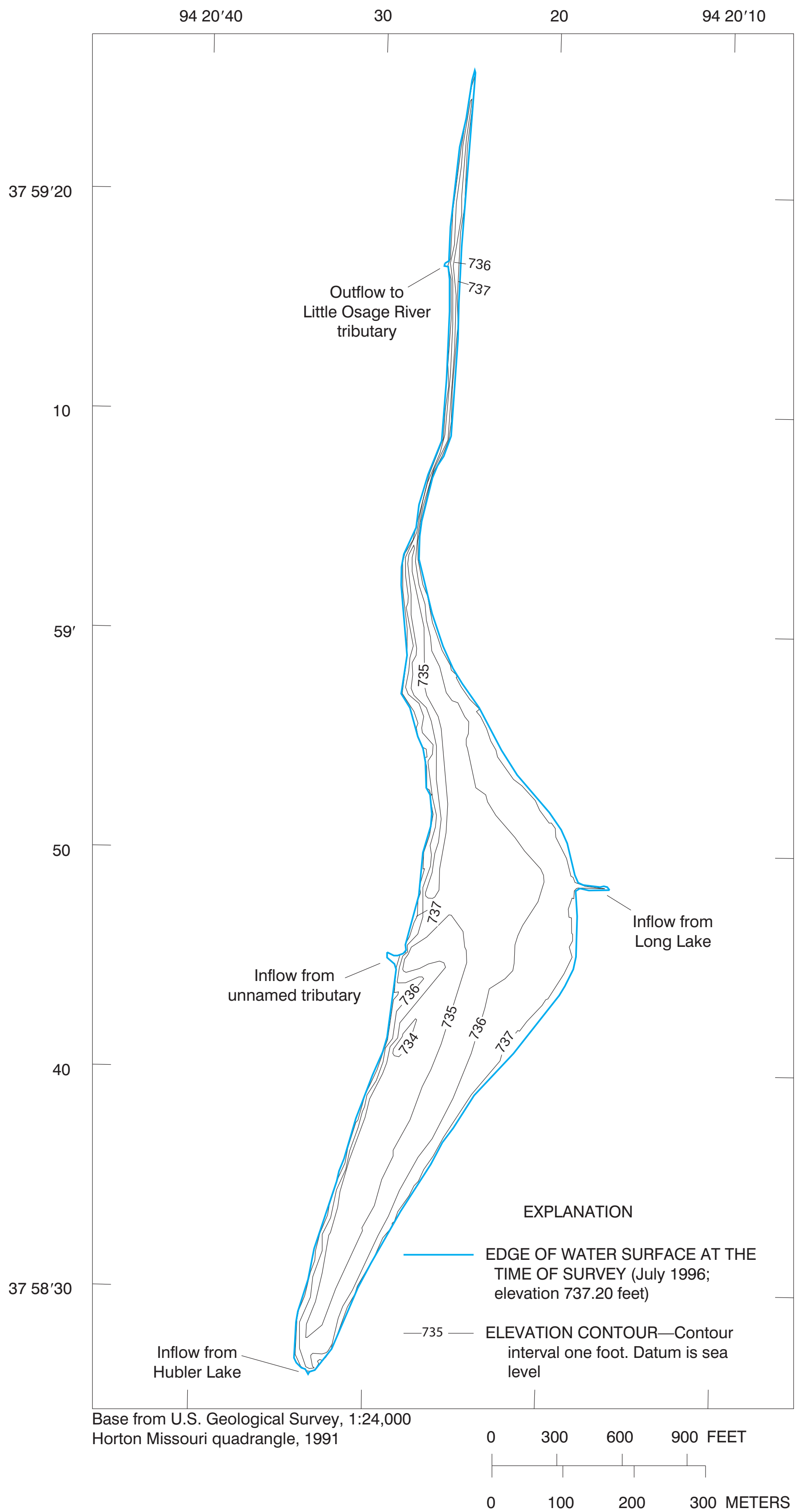


Figure 8. Bathymetry of Spile Lake near Horton, Missouri.

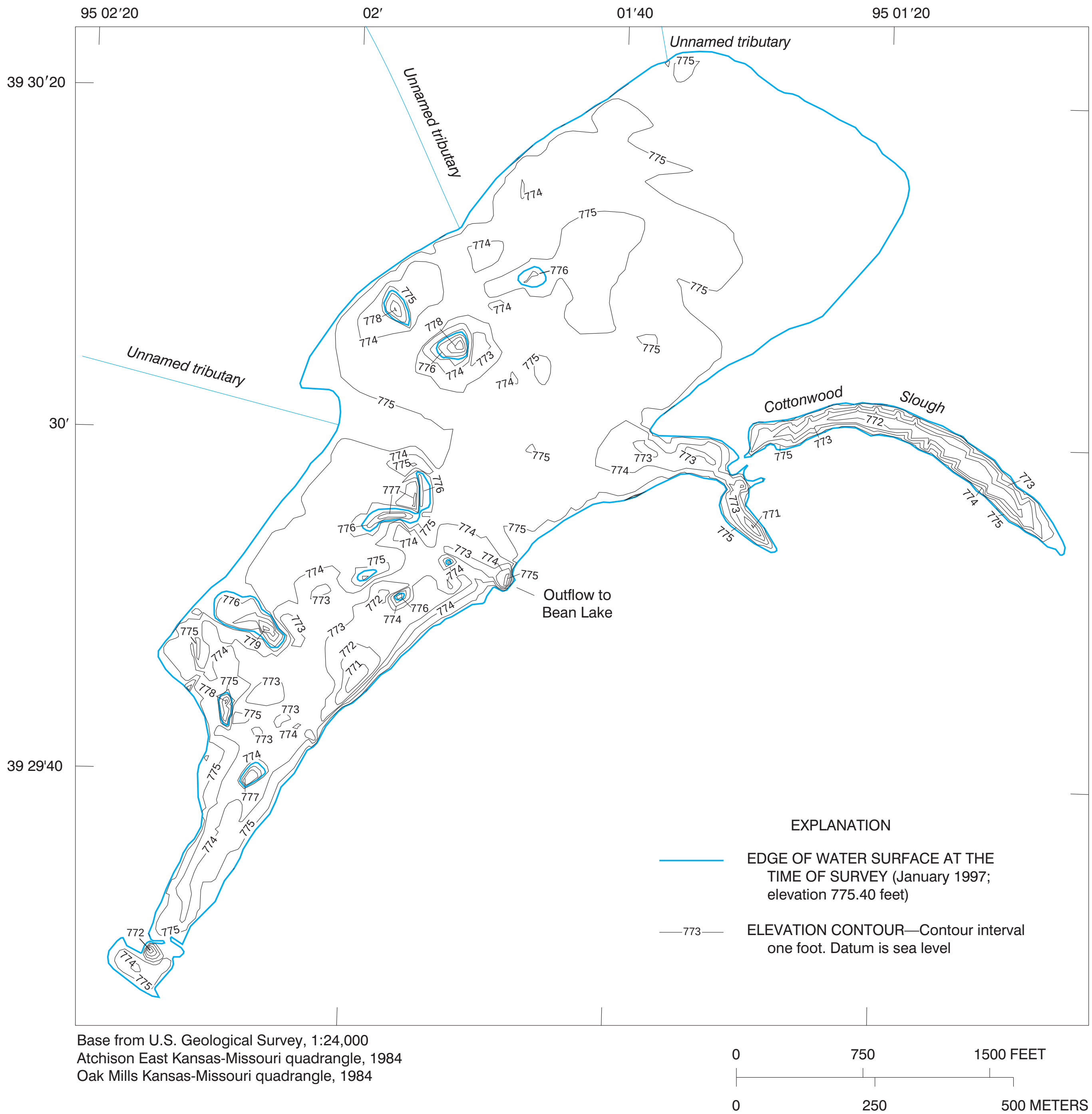
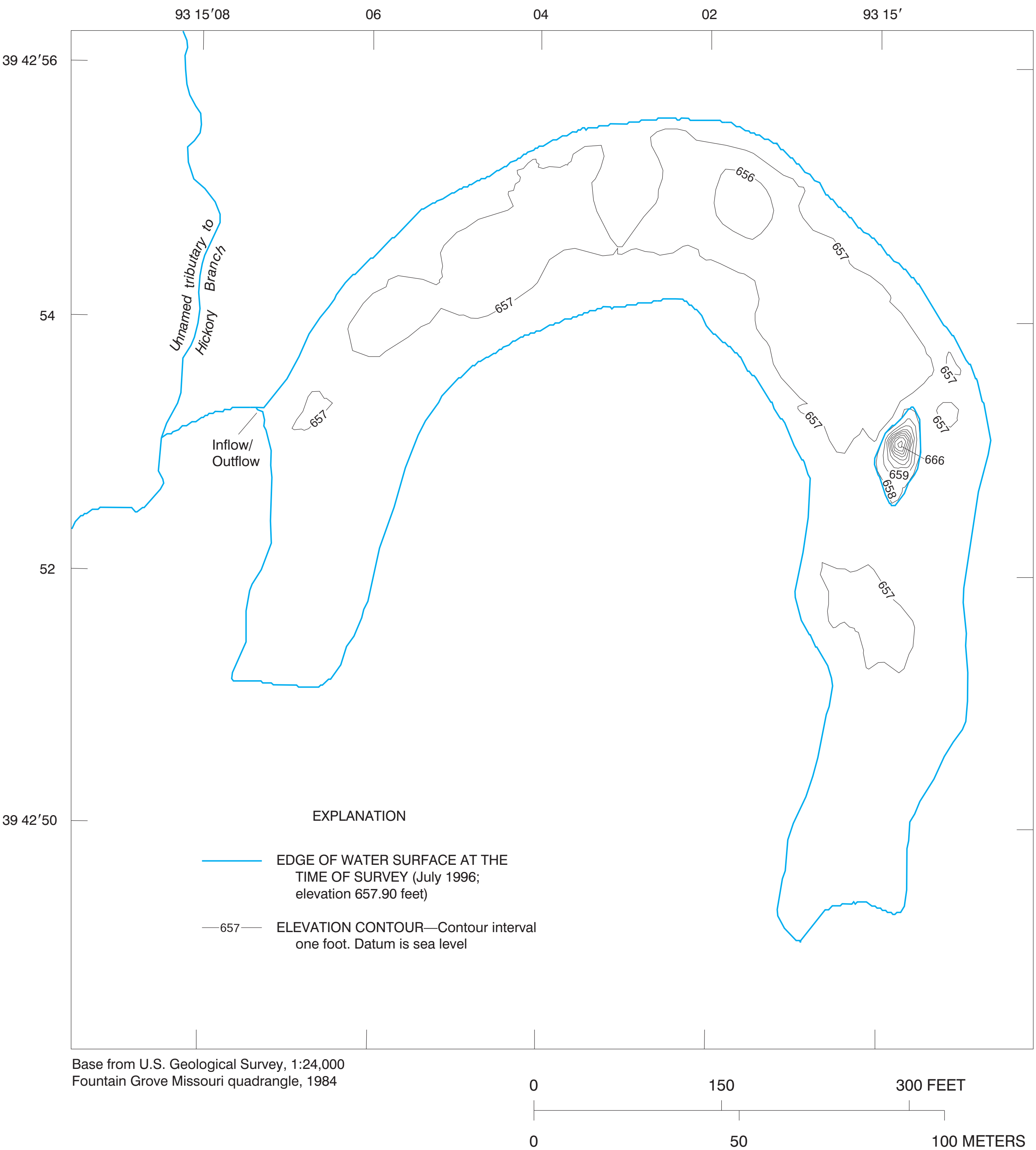


Figure 9. Bathymetry of Little Bean Marsh near Bean Lake, Missouri.



Base from U.S. Geological Survey, 1:24,000
Fountain Grove Missouri quadrangle, 1984

Figure 10. Bathymetry of Forker Oxbow near Fountain Grove, Missouri.

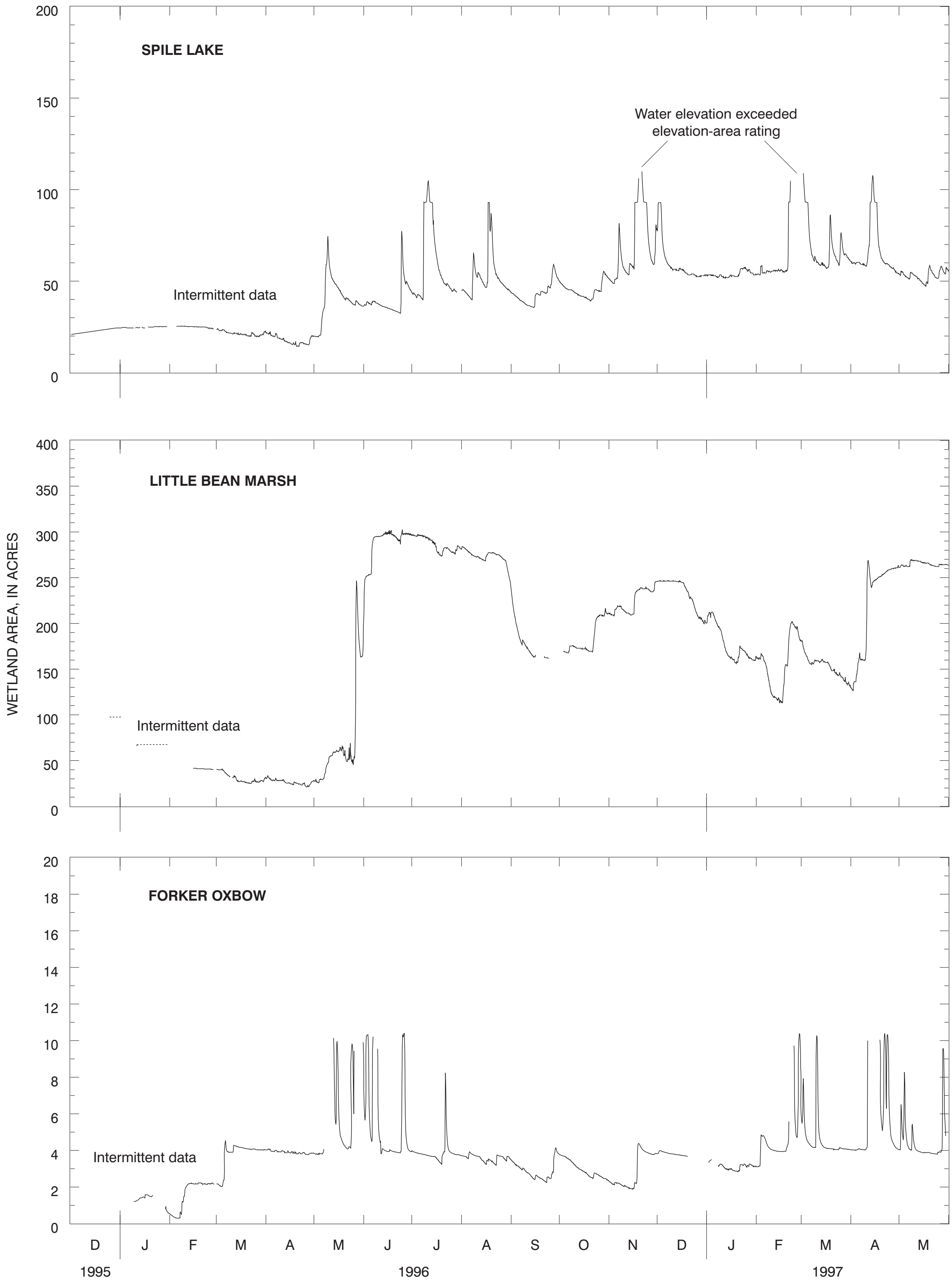


Figure 11. Temporal variability in wetland wetted-surface area at Spile Lake, Little Bean Marsh, and Forker Oxbow.

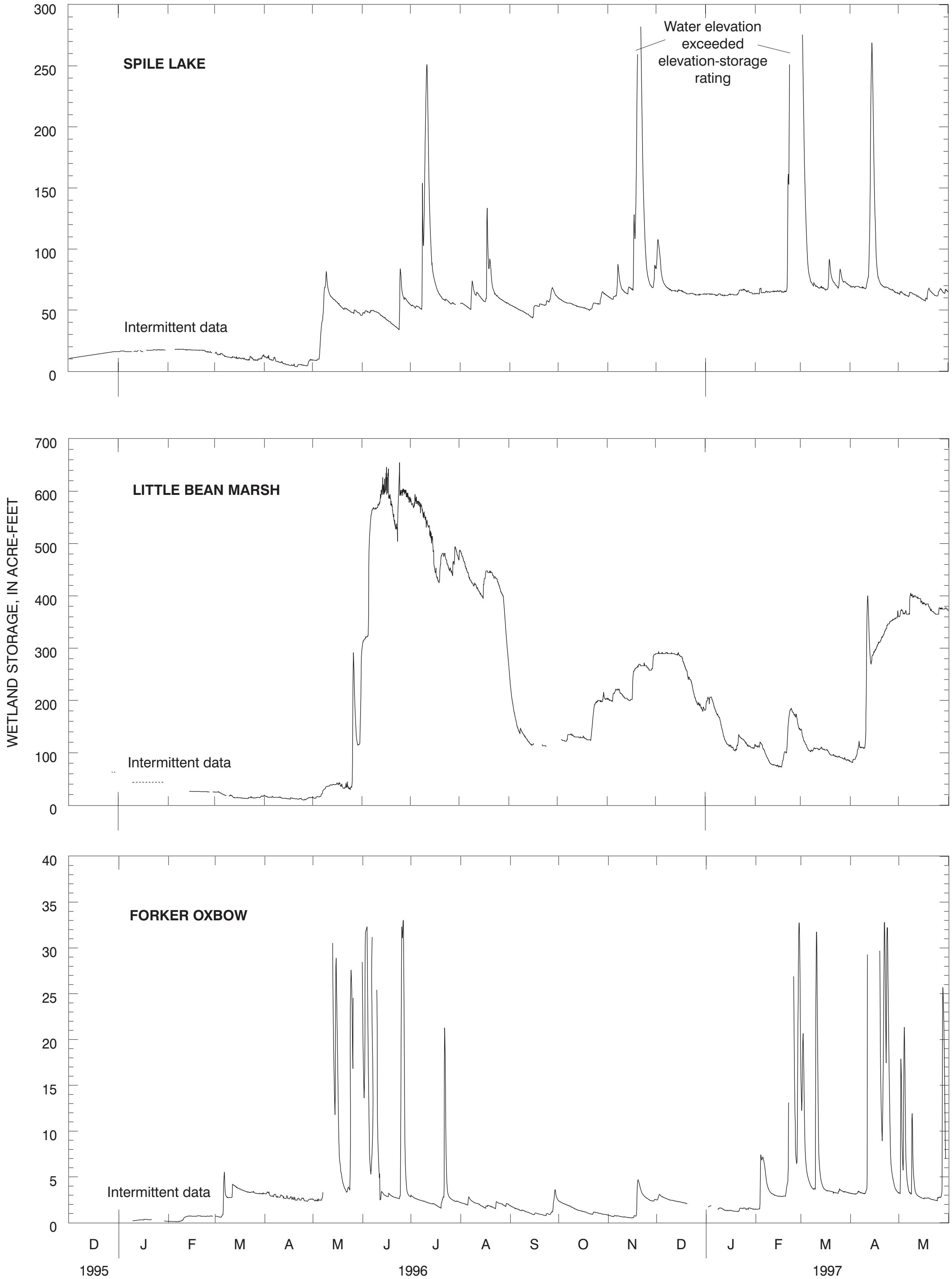


Figure 12. Temporal variability in wetland storage in Spile Lake, Little Bean Marsh, and Forker Oxbow.

Hydraulic Residence Time

Using the maximum maintainable wetland storage and mean annual local runoff, a mean hydraulic residence time was calculated for each wetland using equation 1 presented in the "Methods" section of this report. This equation is based on local runoff values only and does not take into account backwater or flood events from the adjacent rivers. The maximum maintainable wetland storage for Spile Lake occurs at a water-surface elevation of about 737.2 ft, with a corresponding storage of about 54.9 acre-ft. At Little Bean Marsh the maximum maintainable storage occurred at a water-surface elevation of about 775.2 ft, with a corresponding storage of approximately 126 acre-ft. At Forker Oxbow the maximum maintainable storage was determined to occur at an elevation of about 657.9 ft, with a corresponding storage of 1.61 acre-ft. The long-term (1951 to 1980) mean annual runoff for each basin was determined from Gebert and others (1989) and was estimated to be 9.2, 8.2, and 8.1 in. for the Spile Lake, Little Bean Marsh, and Forker Oxbow drainages. The calculated mean residence times were about 12 days for Spile Lake, 27 days for Little Bean Marsh, and 45 days for Forker Oxbow.

The calculated residence times are estimates; actual wetland residence times are quite variable because of changes in the outflow control elevations and effects from flood events. The outflow controls of these wetlands are soil berms, and the water-surface elevation at the maximum maintainable pool elevation may readily change 0.1 ft or more as a result of sizable runoff events or beaver damming. A change in the outflow control elevation of 0.1 ft would increase wetland storage and result in a change in the long-term mean hydraulic residence time of about 1 day at Spile Lake, 5 days at Little Bean Marsh, and 10 days at Forker Oxbow. Unlike Spile Lake and Little Bean Marsh, Forker Oxbow is not a typical flow through system in that the inflow and outflow channels are the same. This will likely result in a longer actual residence time than calculated. However, a complicating factor that would act to decrease the long-term mean residence times of the wetlands is the effect of overbank flood events from adjacent rivers. Floods can completely replace wetland waters in minutes, and therefore, decrease the actual mean residence times of these systems.

Direct precipitation and frequent inundation from adjacent rivers provides an obvious indication of the importance of surface-water contributions to these wetlands, but the ground-water interactions within these systems is unclear. Water-level measurements were made in nine monitoring wells in the vicinity of Little Bean Marsh, from September 1996 through April 1997, and in four of the wells from May 1997 to December 1997 (Dale Blevins, U.S. Geological Survey, oral commun., 1998). The well-level elevations indicated that the highest point of the water table was located at the northeast part of the Marsh, and the water table slopes to the southwest toward the Missouri River (fig. 3). The relative differences in the Marsh surface-water elevation and ground-water levels in surrounding monitoring wells indicate that the water table intersected the Marsh at times during the monitoring period. Ground-water inflows and outflows were not determined but would be controlled by the surface area in contact with the water table, the hydraulic conductivity of wetland substrate, and the hydraulic gradient between the wetland and substrate. The size of the contact area may be limiting ground-water surface-water interactions, as observations of soil cores collected on three occasions at five locations in Little Bean Marsh indicate the wetland substrate was unsaturated below the top few inches (Dale Blevins, oral commun., 1998). No information is available on the ground-water surface-water interactions in Spile Lake or Forker Oxbow.

WATER-QUALITY CHARACTERISTICS

Water-quality characteristics of natural wetland systems are a result of both allocthonous (outside system) and autochthonous (within system) processes. Physicochemical properties and major nutrient concentrations were monitored within the study wetlands. Heterogeneity in emergent vegetation and thermal stratification were observed, at times, to cause spatial differences in physicochemical properties in Spile Lake and especially Little Bean Marsh. The results of this study, however, represent temporal differences at a single location about 1.5 ft below the water surface in each wetland. The determination of spatial differences within individual wetlands was beyond the scope of this study.

Physicochemical Properties

More than 12,000 hourly specific conductance, pH, water temperature, and dissolved-oxygen measurements were logged at each wetland site from December 1995 to May 1997. This sampling frequency allowed for determination of the diurnal and seasonal variability of these properties as well as overall extremes.

Specific Conductance

Specific conductance is a measure of the ability of water to conduct electricity and provides an indication of the ion concentration of the water (Hem, 1985). Overall, median specific conductance values were 231 μS/cm (microsiemens per centimeter) at Spile Lake, 457 μS/cm at Little Bean Marsh, and 236 μS/cm at Forker Oxbow (table 1; fig. 13). Monitored ground-water levels near Little Bean Marsh from September 1996 to April 1997 indicated that the Marsh intersects the water table of the Missouri River alluvial aquifer, at times, and the specific conductance of this aquifer was higher than in rainfall and surface runoff. Ground-water contributions may, therefore, at least partly account for the higher median specific conductance values measured at Little Bean Marsh.

Substantial daily fluctuations in specific conductance were uncommon in the three study wetlands, and the overall extremes are closely tied with periods of ice cover and rainfall-runoff events. Maximum specific-conductance values were measured in the winter months at all three sites—February 1996 at Spile Lake, January 1997 at Little Bean Marsh, and February 1996 in Forker Oxbow (fig. 14). Possible ground-water contributions from the Missouri River alluvium would not account for the elevated specific conductance values at Little Bean Marsh in January and February of 1996 and 1997. Periods of ice formation closely correspond to these winter peaks in specific conductance. The ice formation process in these shallow systems concentrates ions in the remaining waters and substantially increases the specific conductance of these waters. At the relatively low mid-February elevation of about 657.56 ft at Forker Oxbow, a 0.5-ft ice cover would result in about a 70 percent decrease in the water volume in the wetland (0.79 acre-ft to 0.24 acre-ft), thereby concentrating dissolved ions into the remaining 30 percent of the water volume. The substantial change in specific conductance values at Forker Oxbow between February and March 1996 was the result of moving the water-quality sensor from inside the standing pipe to the open wetland. This was done because of differences in specific conductance between the deeper waters in the culvert and those of the shallower open waters during ice melt.

Periods of minimum specific conductance values in the three wetlands occurred in July 1996 at Spile Lake, and in May 1996 at Little Bean Marsh and Forker Oxbow. The minimum values correspond with rainfall-runoff events. In these wetland systems, direct rainfall of 1 to 2 in. results in a substantial dilution effect.

pH

pH is a measure of the hydrogen ion activity of a solution and in the water column of most natural waters is between 6 and 9 units (Hem, 1985). The pH of a wetland, as well as many other surface-water bodies, can be greatly affected by photosynthesis, respiration, and decomposition (Hutchinson, 1957; Wetzel, 1983; Reddy, 1981). These processes affect the carbon dioxide (CO_2) concentration in the waters, and in turn, the concentration of carbonic acid (H_2CO_3) and hydrogen ions (H_30^+) and, therefore, pH. The pH can vary among wetland types and spatially within wetland systems. Given the types of wetlands in this study and the water column sampling locations, the range provided above provides a reasonable estimate of expected pH values.

Median pH values were 7.0 at Spile Lake, 7.7 at Little Bean Marsh, and 7.2 at Forker Oxbow (table 1; fig. 13). Variations in pH in these wetlands were likely because of photosynthesis, respiration, and decomposition; all values fell within the 6 to 9 range commonly measured in natural waters. Maximum pH values were measured in the wetlands in late winter and early spring during probable algal blooms and high photosynthesis activity (fig. 15). Conversely, minimum pH values occurred in the late spring and early summer, likely because of increased respiration and decomposition in the warmer waters following early spring algal production peaks. Diurnal fluctuations in pH were greatest in Forker Oxbow during the spring of 1996 (fig. 15) and probably are associated with photosynthesis in water with low buffering capacity.

Table 1. Summary of hourly wetland physicochemical property results, December 1995 to May 1997

[µS/cm, microsiemens per centimeter at 25 degrees Celsius; °C, degrees Celsius; mg/L, milligrams per liter; NTU's, nephelometric turbidity units]

	Spile Lake			Little Bean Marsh			Forker Oxbow		
	Maximum	**Minimum**	**Median**	**Maximum**	**Minimum**	**Median**	**Maximum**	**Minimum**	**Median**
Specific conductance (µS/cm)	495	103	231	[a]1,320	187	457	928	[a]82	236
pH (standard units)	7.8	6.5	7.0	8.6	6.7	7.7	[a]8.8	[a]6.2	7.2
Temperature, near surface (°C)	31.4	[a].03	13.7	30.6	.05	12.6	[a]35.1	.04	11.6
Temperature, near bottom (°C)	[1]29.7	2.2	11.7	24.5	[a].90	11.8	29.0	1.8	8.9
Dissolved oxygen (mg/L)	14.2	[a].00	5.4	[a]26.8	.10	8.2	20.1	[a].00	4.3
Turbidity (NTU's)	[a]200	4.0	18	49	[a].10	2.2	100	1.3	26

[1] Represents overall maximum/minimum at the three wetlands.

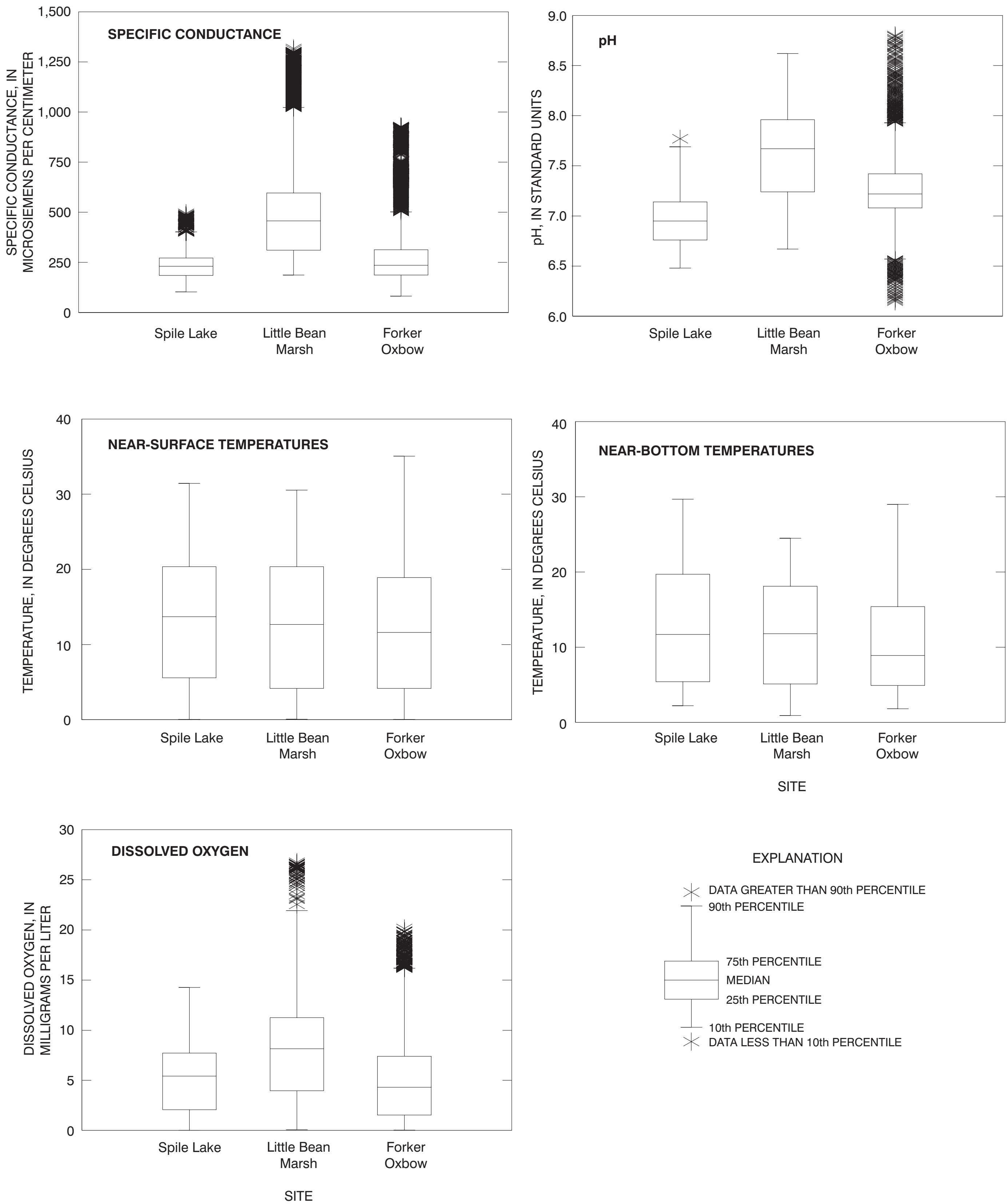


Figure 13. Distributions of specific conductance, pH, near-surface and near-bottom water temperatures, and dissolved oxygen in Spile Lake, Little Bean Marsh, and Forker Oxbow.

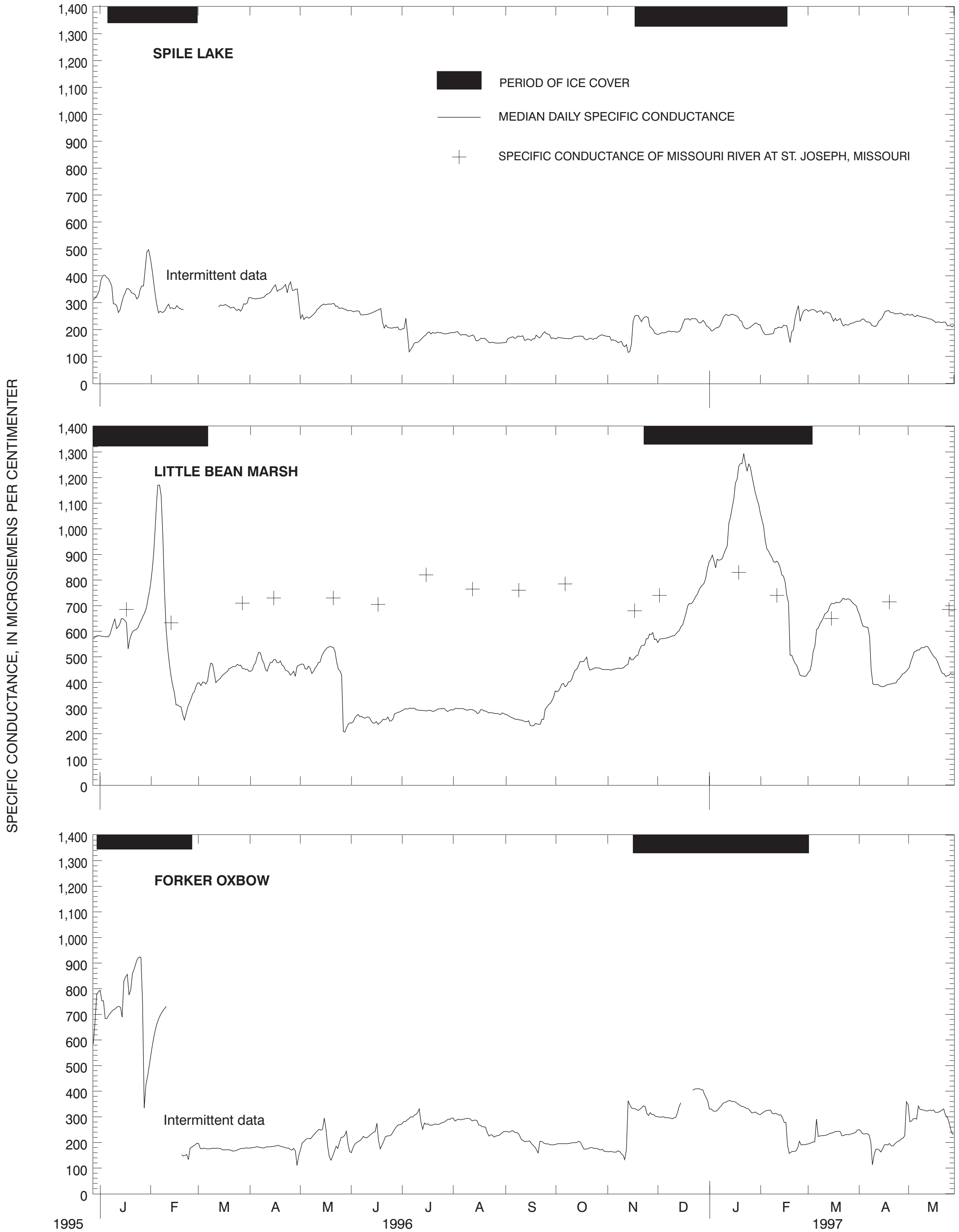


Figure 14. Median daily specific conductance at Spile Lake, Little Bean Marsh, and Forker Oxbow.

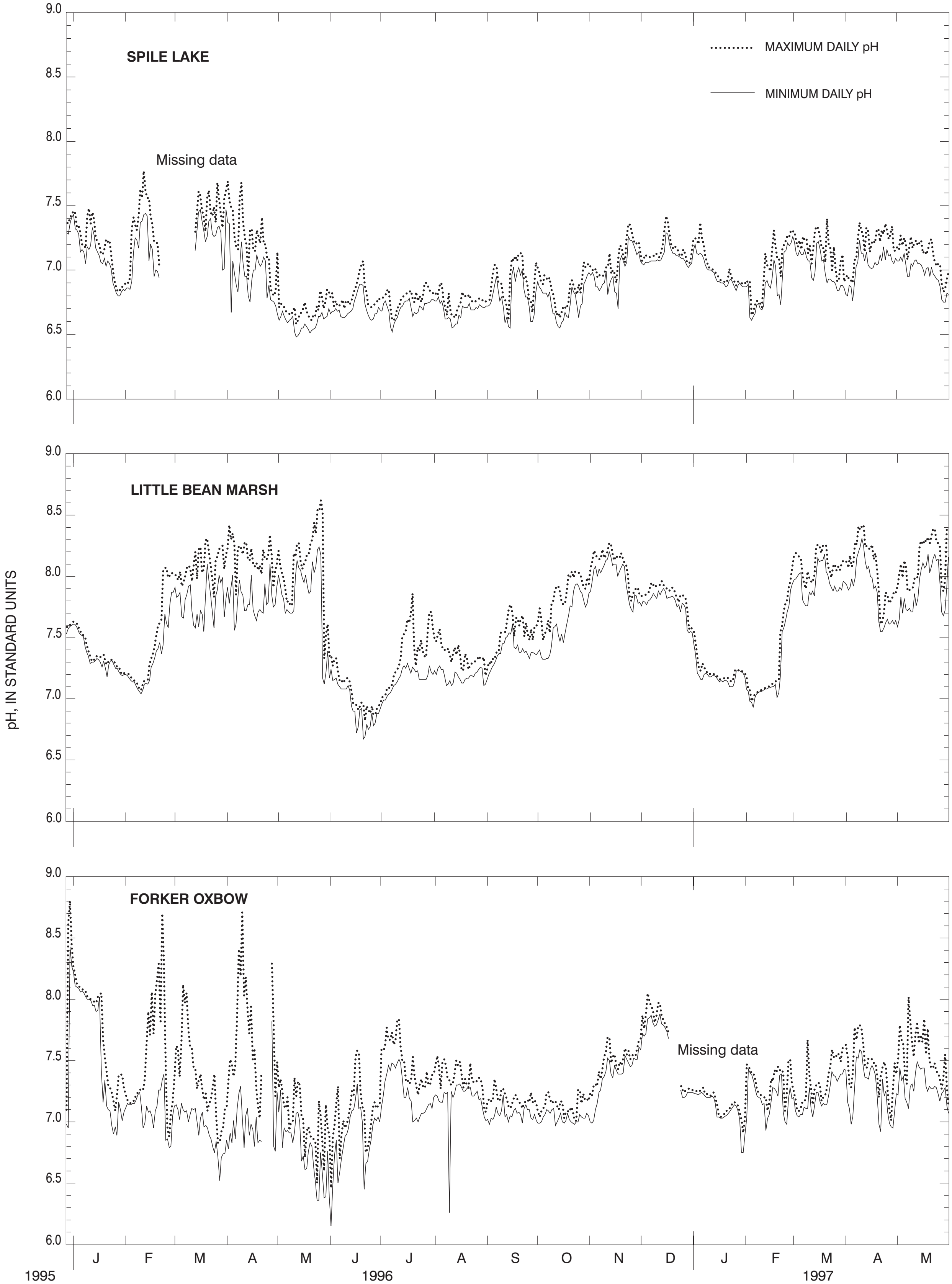


Figure 15. Maximum and minimum daily pH values at Spile Lake, Little Bean Marsh, and Forker Oxbow.

Water Temperature

The median water temperatures of the three wetlands seemed to reflect the difference in latitude of these systems. Median near-surface temperatures were greatest in Spile Lake (13.7 °C; table 1; fig. 13), the southernmost site, and least in Forker Oxbow (11.6 °C), the northernmost site. Median near-bottom temperatures were greatest in Little Bean Marsh (11.8 °C), and again least in Forker Oxbow (8.9 °C).

Despite having the lowest median near-surface and near-bottom temperature values, Forker Oxbow had a relatively high maximum near-bottom temperature and the highest maximum near-surface temperature of the three wetlands (table 1). Wetland water temperatures, particularly near-surface temperatures, closely followed changes in air temperature. Because Forker Oxbow had the lowest water volume of the three wetlands, air temperature changes would affect water temperature more rapidly and to a greater extent, which may account for the temperature extremes in this system despite its latitude.The Missouri water temperature standard of 90 °F (32.2 °C; Missouri Department of Natural Resources, 1994) was exceeded only at Forker Oxbow. The maximum temperature measured at Forker Oxbow was 35.1 °C.

Although the wetlands are shallow, there was a substantial difference between near-surface and near-bottom temperature throughout most of the monitoring period (fig. 16). This temperature gradient resulted in the formation of a density gradient and, therefore, thermal stratification. Similar to temperate lakes and reservoirs, thermal stratification in the wetlands was greatest in the summer months, less in the winter months, and absent in the fall and early spring months. Unlike many lakes and reservoirs, wetlands are shallow systems, and thermal stratification can develop and dissipate rather quickly, even diurnally. Stratification can develop during the daylight hours, as the near-surface waters become warmer than near-bottom waters, and then destratify in the evening hours as the surface waters cool (fig. 16). Thermal stratification can prevent or hinder vertical mixing of water layers and can have substantial effects on dissolved oxygen.

Dissolved Oxygen

Dissolved oxygen is an important property in aquatic systems and can have large effects on chemical and biological processes in these wetlands. Biological processes, specifically photosynthesis and respiration, can, in turn, have large effects on dissolved-oxygen levels. Thermal stratification, which was shown to be a common characteristic of the study wetlands, also will be a determining factor in the dissolved-oxygen concentrations of near-bottom waters.

Median dissolved-oxygen concentrations were 5.4 mg/L (milligrams per liter) at Spile Lake, 8.2 mg/L at Little Bean Marsh, and 4.3 mg/L at Forker Oxbow (table 1; fig. 13). Dissolved oxygen extremes in the wetlands were likely caused by biological factors and thermal stratification conditions. Maximum dissolved-oxygen levels were measured in late winter or early spring in the wetlands, and these periods probably corresponded to periods of greatest algal production and photosynthesis activity. Maximum dissolved-oxygen values at Spile Lake were about 12 mg/L less than at Little Bean Marsh and about 6 mg/L less than at Forker Oxbow (table 1). This may be the result of a relative lack of phytoplankton activity in Spile Lake, which is dominated by emergent macrophytes (spatterdock). The maximum dissolved-oxygen concentration measured in Little Bean Marsh was 26.8 mg/L in January 1997 (fig. 17), which represented about 200 percent of the saturation level. This corresponded to a probable period of photosynthesis production under ice cover. Filamentous green algae was observed to be growing on the underside and within layers of the clear ice cover during this period. The clear ice cover allowed sunlight to penetrate the surface waters but prevented the release of dissolved oxygen produced by the algae, resulting in super-saturated conditions. Observers reported “hissing sounds” emanating from cracks in the ice during this period, presumably as built-up oxygen and other gases were released from the ice cover. The super-saturated conditions may have been the cause of a fish kill observed in Little Bean Marsh on January 21, 1997. The lowest dissolved-oxygen levels were measured in late spring and summer in all three wetlands as well as in the winter months of 1996 in Forker Oxbow. While ice cover may result in super-saturated conditions as described in January 1997 at Little Bean Marsh, it also can account for the low dissolved-oxygen concentrations measured in February 1996 at all three wetlands. Snow cover on ice and “cloudy ice” cover during this period likely limited algae production while preventing reaeration. An observed February 15, 1996, fish kill affecting some 50 to 100 Gizzard shad (*Dorosoma cepedianum*) at Little Bean Marsh may be related to low (3 to 5 mg/L) dissolved-oxygen levels. Dissolved-oxygen concentrations, particularly mini-

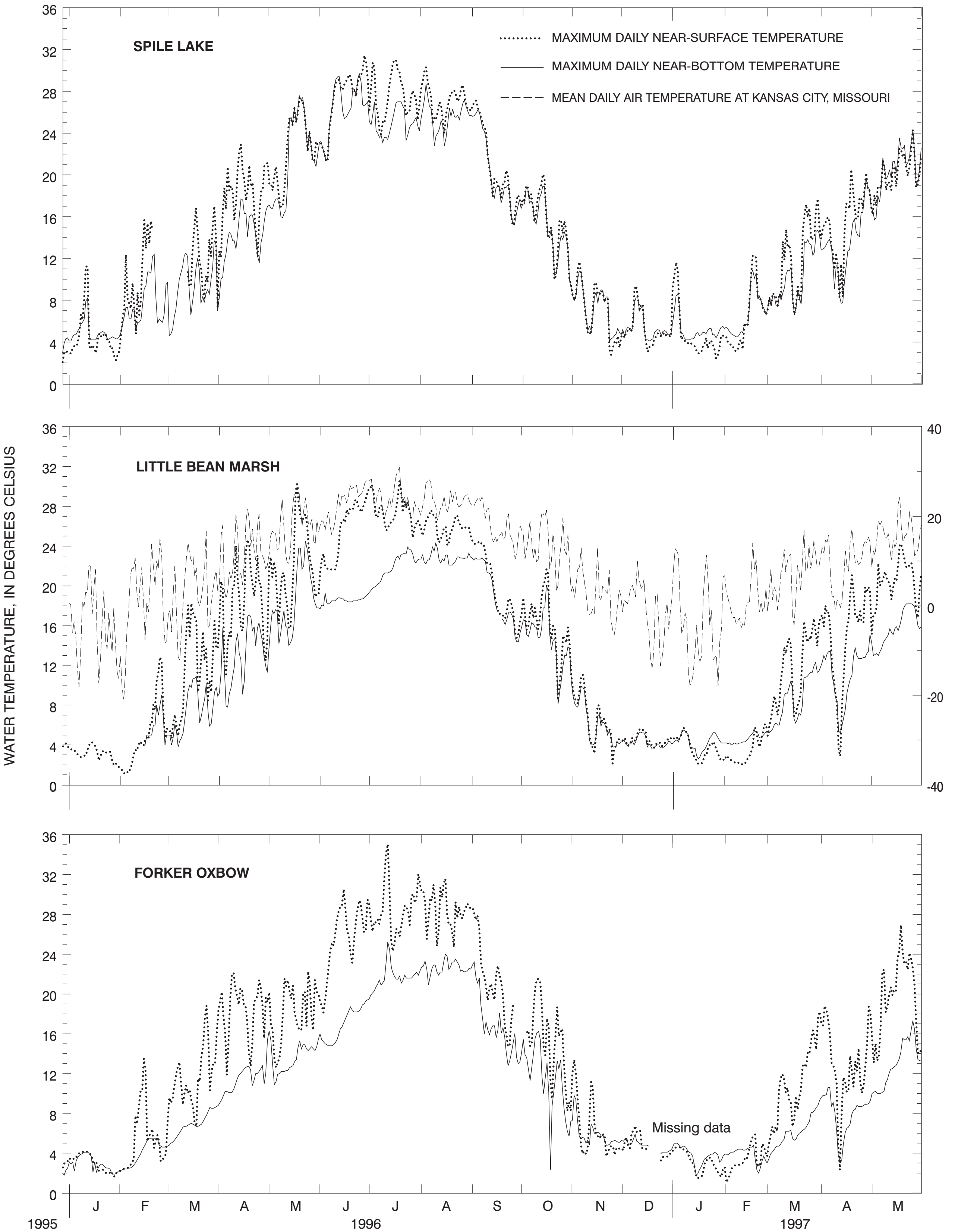


Figure 16. Maximum daily near-surface and near-bottom water temperatures for Spile Lake, Little Bean Marsh, and Forker Oxbow. Mean daily air temperature at Kansas City, Missouri, from the National Oceanic and Atmospheric Administration (1995-97).

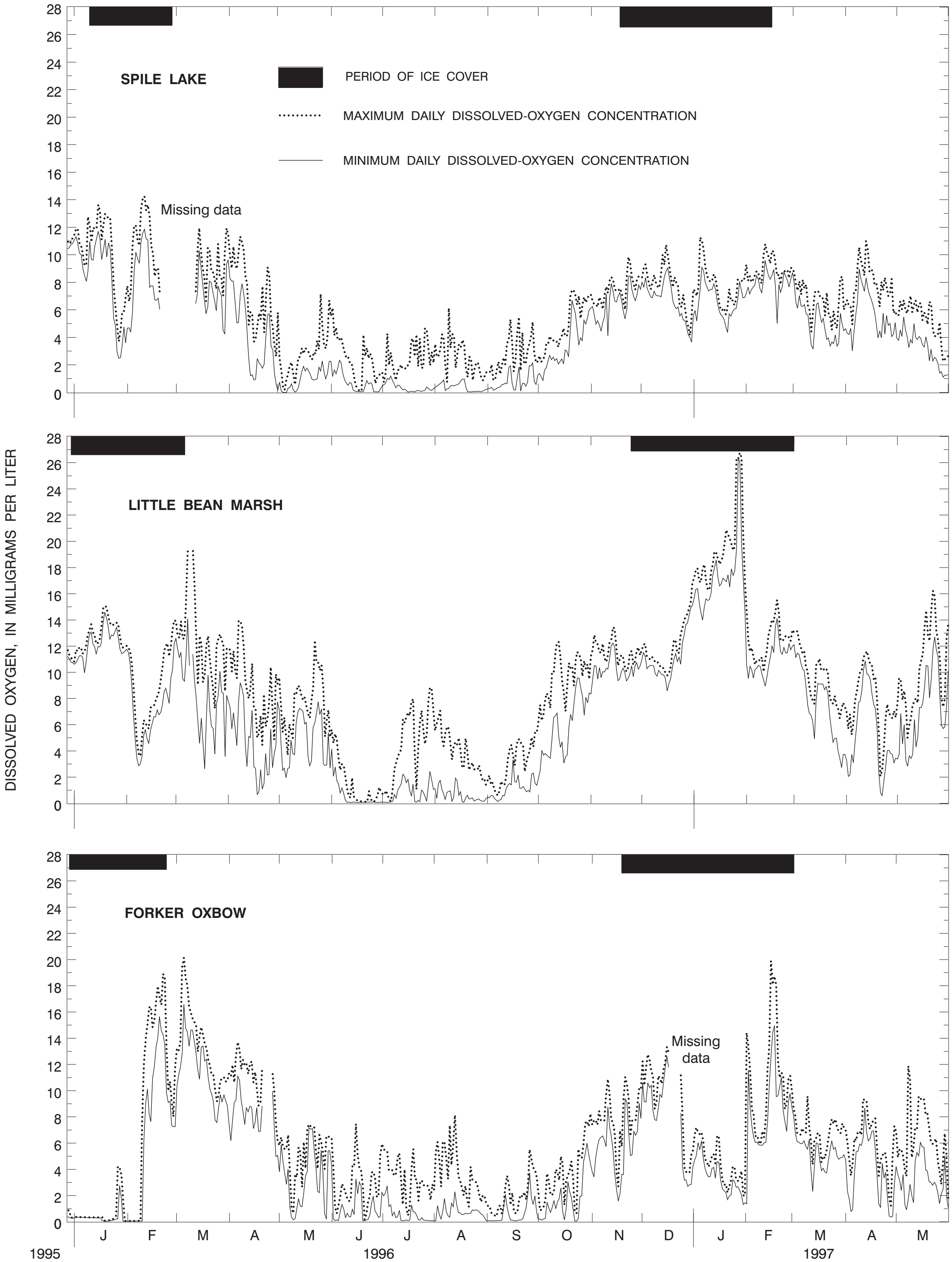


Figure 17. Maximum and minimum daily dissolved-oxygen concentrations at Spile Lake, Little Bean Marsh, and Forker Oxbow.

mum daily concentrations, are low in the summer months (fig. 17) because of increased temperature (decreased saturation levels), increased respiration, and increased frequency and magnitude of thermal stratification, which limits reaeration below the thermocline. With minimum dissolved-oxygen concentrations near zero, and maximum daily values of about 4 to 8 mg/L, the summer period marked the greatest diurnal fluctuations in dissolved oxygen at all three wetlands (fig. 17).

A cumulative frequency distribution of dissolved-oxygen levels (fig. 18) indicates that the 5 mg/L dissolved-oxygen standard set by the State of Missouri for the protection of aquatic life (Missouri Department of Natural Resources, 1994) frequently is not met in these wetlands at the sampling locations. Dissolved-oxygen concentrations at Spile Lake were below 5 mg/L about 55 percent of the time from January 9, 1996 to January 8, 1997, and 45 percent of the time for the entire monitoring period. The January 1996 to May 1997 monitoring period had a greater proportion of non-stratified high dissolved-oxygen level months (December to May); therefore, the frequency distributions for each wetland were standardized for a one-year period. At Little Bean Marsh, dissolved-oxygen levels were below 5 mg/L about 40 percent of the time from January 9, 1996 to January 8, 1997, and 30 percent of the time overall. Forker Oxbow dissolved-oxygen concentrations were below 5 mg/L about 60 percent of the time from January 9, 1996 to January 8, 1997, and 55 percent of the time overall. The relatively longer periods with dissolved-oxygen concentrations below 5 mg/L at Forker Oxbow could be explained by the longer stratification periods (fig. 16), which, in turn, may have been caused by the relatively small water volume of this system.

Turbidity

Turbidity is a measure of water clarity. Common factors affecting water clarity in natural systems include plankton, suspended inorganic matter, and suspended dead organic matter. It is not uncommon to find low water clarity in midwestern reservoirs as the result of high concentrations of suspended material, and in some cases, low water clarity may be the limiting factor in algal production in these systems (Jones and Knowlton, 1993). The degree to which water clarity limits phytoplankton production in Missouri wetlands is not known to be documented.

The median turbidity values were 18 nephelometric turbidity units (NTU's) at Spile Lake, 2.2 NTU's at Little Bean Marsh, and 26 NTU's at Forker Oxbow (table 1). A statistically significant difference existed in the median turbidity values in Little Bean Marsh and the remaining two wetlands (Kruskal-Wallis test, $p < 0.01$; rank transform test, Tukey's test). The differences in soils, ground-water contributions, and wetland orientation relative to prevailing winds may account for the lower turbidity values in Little Bean Marsh relative to the other wetlands.

The maximum turbidity value of 200 NTU's was measured on March 6, 1996, at Spile Lake, but it is not likely that water clarity limits algae or submerged macrophyte production in the study wetlands. Based on median values, turbidity was commonly less than about 26 NTU's in all three systems. Under these more typical conditions, there would be sufficient sunlight penetration in these shallow systems such that water clarity would probably not limit algae production. The dense emergent vegetation at Spile Lake and Little Bean Marsh shadows a large part of these wetlands, and this may limit algae production in some areas of these systems.

Nutrients

Much research has been conducted on the role of wetlands in the removal or transformation of nutrients in waters that pass through these systems. Wetlands have been known to act as sinks (Klopetek, 1978; van der Valk and others, 1978; Fetter and others, 1978; Dolan and others, 1981; Vega and Ewel, 1981; Klarerm and Millie, 1989), sources (Lee and others, 1975), or transformers (Klopetek, 1978; Vega and Ewel, 1981) of N or P forms entering them. Several studies stress that the nutrient retention capability of wetland systems is largely affected by the hydrology of the system (van der Valk, 1978; Kadlec and Kadlec, 1979; Peverly, 1982), which is a determining factor in the concentration and mass of nutrient imports, exports, and the interaction period of waters within the wetland with vegetation and the substrate.

Nutrient sampling efforts in this study focused on quantifying concentrations of N and P forms in the wetlands and determining temporal variability and relative differences of these forms between wetlands. A summary of nutrient results is provided in table 2 and figure 19. There are no clearly defined temporal trends in the nutrient data. The numerous sources of variabil-

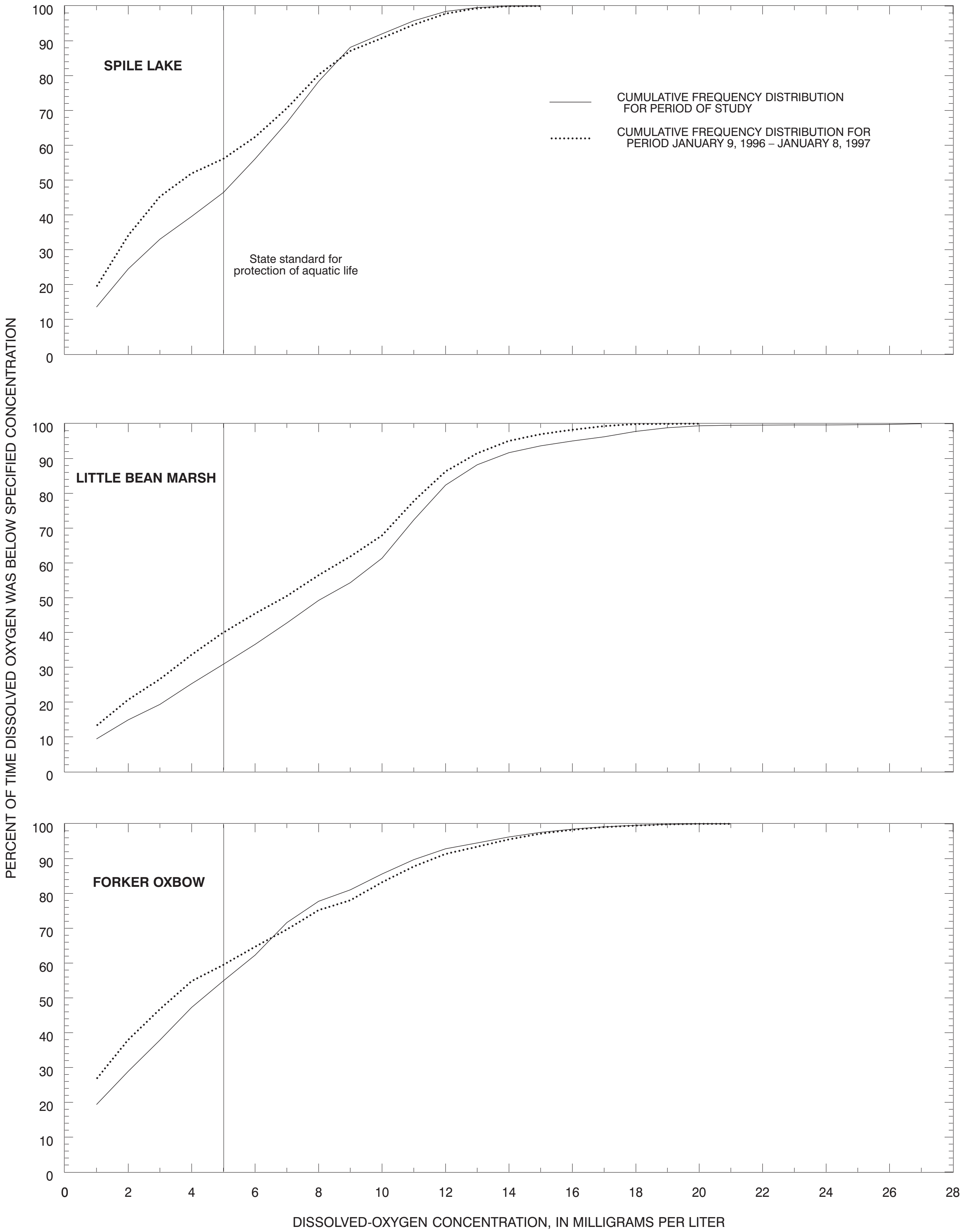


Figure 18. Cumulative time frequency distribution of dissolved-oxygen concentrations at Spile Lake, Little Bean Marsh, and Forker Oxbow for period of record and selected one-year period.

Table 2. Wetland nutrient results

[mg/L, milligrams per liter; N, nitrogen; $NO_2 + NO_3$, nitrite plus nitrate; P, phosphorus; --, no data available; <, less than]

Station number	Date	Nitrogen, ammonia dissolved (mg/L as N)	Nitrogen, nitrite dissolved (mg/L as N)	Nitrogen, $NO_2 + NO_3$ dissolved (mg/L as N)	Nitrogen, ammonia plus organic total (mg/L as N)	Phosphorus dissolved (mg/L as P)	Phosphorus total (mg/L as P)	Phosphorus ortho, dissolved (mg/L as P)
				Spile Lake				
06917080	02/23/96	--	--	--	1.8	--	0.19	--
	04/19/96	0.02	0.01	<0.05	3.2	--	.42	0.01
	05/15/96	.53	.01	.06	1.6	0.02	.21	--
	06/25/96	.01	<.01	<.02	1.7	.05	.20	.03
	07/22/96	.015	<.01	<.02	1.4	.09	.24	.07
	08/22/96	<.01	<.01	<.02	.54	.02	.12	.02
	09/06/96	<.01	<.01	<.02	.56	.03	.10	.02
	10/30/96	.02	.01	.08	1.4	.01	.19	<.01
	11/26/96	.09	.01	<.05	.70	.04	.15	.02
	12/16/96	.02	<.01	<.05	.60	<.01	.06	<.01
	01/23/97	.16	.01	.11	.60	<.01	.03	<.01
	[1]01/23/97	.16	.03	.11	.70	<.01	.05	<.01
	02/18/97	<.015	.02	<.05	.70	<.01	.04	<.01
	03/28/97	<.015	<.01	<.05	.90	.04	.16	<.01
	[1]03/28/97	<.015	<.01	<.05	1.1	.05	.15	<.01
	05/06/97	<.015	<.01	<.05	.72	.026	.063	.016
	05/26/97	<.015	<.01	<.05	1.3	.026	.15	<.01
Number of samples		**14**	**14**	**14**	**15**	**13**	**15**	**14**
Maximum		**0.53**	**0.03**	**0.11**	**3.2**	**0.09**	**0.42**	**0.07**
Minimum		**<.01**	**<.01**	**<.02**	**.54**	**<.01**	**.03**	**<.01**
Median		**<.015**	**<.01**	**<.05**	**.90**	**.02**	**.15**	**<.01**

Table 2. Wetland nutrient results—Continued

[mg/L, milligrams per liter; N, nitrogen; $NO_2 + NO_3$, nitrite plus nitrate; P, phosphorus; --, no data available; <, less than]

Station number	Date	Nitrogen, ammonia dissolved (mg/L as N)	Nitrogen, nitrite dissolved (mg/L as N)	Nitrogen, $NO_2 + NO_3$ dissolved (mg/L as N)	Nitrogen, ammonia plus organic total (mg/L as N)	Phosphorus dissolved (mg/L as P)	Phosphorus total (mg/L as P)	Phosphorus ortho, dissolved (mg/L as P)
				Little Bean Marsh				
68182001	02/15/96	--	--	--	0.26	--	0.03	--
	04/18/96	0.02	<0.01	<0.05	2.5	--	.41	0.01
	05/13/96	.70	<.01	<.02	.81	0.58	.83	.06
	06/27/96	.01	<.01	<.02	.89	.18	.24	.16
	07/23/96	.021	<.01	<.02	1	.07	.14	.03
	08/19/96	.044	<.01	<.02	.22	.02	.07	<.01
	08/30/96	<.01	<.01	<.02	.31	<.02	.04	.01
	10/28/96	.03	.02	.08	.80	<.01	.03	<.01
	11/25/96	.02	.02	<.05	.50	<.01	.02	<.01
	12/18/96	.03	<.01	<.05	.50	.01	.01	<.01
	01/23/97	.19	.02	<.05	2.2	.06	.11	.02
	02/19/97	.06	.03	.05	.80	<.01	<.01	<.01
	03/21/97	.02	<.01	<.05	.50	<.01	.07	<.01
	04/25/97	<.015	<.01	<.05	.72	<.01	.057	<.01
	05/23/97	<.015	<.01	<.05	.96	<.01	.043	<.01
Number of samples		**14**	**14**	**14**	**15**	**13**	**15**	**14**
Maximum		**0.7**	**0.03**	**0.08**	**2.5**	**0.58**	**0.83**	**0.16**
Minimum		**<.01**	**<.01**	**<.02**	**.22**	**<.01**	**<.01**	**<.01**
Median		**.02**	**<.01**	**<.05**	**.80**	**.01**	**.06**	**<.01**

Table 2. Wetland nutrient results—Continued

[mg/L, milligrams per liter; N, nitrogen; $NO_2 + NO_3$, nitrite plus nitrate; P, phosphorus; --, no data available; <, less than]

Station number	Date	Nitrogen, ammonia dissolved (mg/L as N)	Nitrogen, nitrite dissolved (mg/L as N)	Nitrogen, $NO_2 + NO_3$ dissolved (mg/L as N)	Nitrogen, ammonia plus organic total (mg/L as N)	Phosphorus dissolved (mg/L as P)	Phosphorus total (mg/L as P)	Phosphorus ortho, dissolved (mg/L as P)
				Forker Oxbow				
6901800	02/21/96	--	--	--	1.9	--	0.12	--
	04/17/96	0.02	<0.01	<0.05	.76	--	.06	0.02
	05/22/96	.03	.01	.02	.95	0.02	.17	--
	06/28/96	.01	.03	.30	1.8	.06	.26	.01
	07/26/96	.063	<.01	<.02	2.5	.04	.33	.02
	08/20/96	.051	<.01	<.02	.75	.04	.34	.02
	09/05/96	.02	<.01	<.02	.46	<.02	.34	.01
	11/01/96	<.015	.03	.10	1.1	.04	.16	<.01
	11/27/96	.07	.04	.29	.90	.01	.16	<.01
	12/20/96	<.015	<.01	<.05	.90	<.01	.07	<.01
	01/31/97	.37	.02	<.05	1.5	<.01	.10	<.01
	02/14/97	.15	.04	.69	1.6	.02	.15	<.01
	04/02/97	<.015	<.01	<.05	1	.05	.15	<.01
	04/30/97	<.015	<.01	<.05	1.6	<.01	.20	<.01
	05/30/97	<.015	.033	.428	1.2	.02	.19	<.01
Number of samples		**14**	**14**	**14**	**15**	**13**	**15**	**13**
Maximum		**0.37**	**0.04**	**0.69**	**2.5**	**0.06**	**0.34**	**0.02**
Minimum		**<.015**	**<.01**	**<.02**	**.46**	**<.01**	**.06**	**<.01**
Median		**.02**	**<.01**	**<.05**	**1.1**	**.02**	**.16**	**<.01**

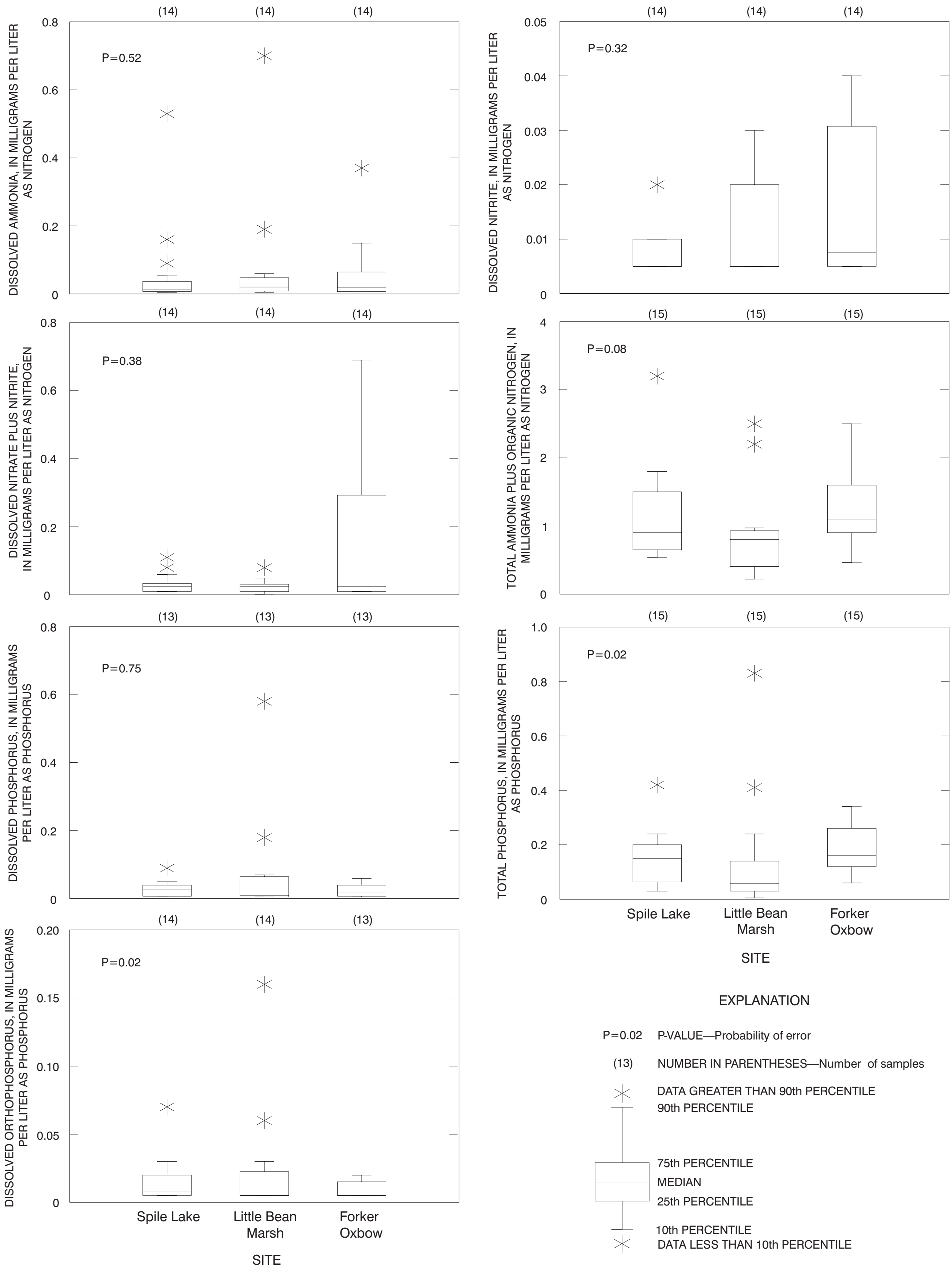


Figure 19. Distribution of selected nutrients in Spile Lake, Little Bean Marsh, and Forker Oxbow.

ity, including dramatic changes in hydrology, seasonal land-use affects (agricultural runoff), thermal stratification, and biological affects in these wetlands combined with the low concentrations and limited sample collection frequency make temporal trends indiscernible.

Whereas minimum and median concentrations of dissolved N and P constituents were at or near detection levels at all wetlands, maximum levels exceeded detection levels and reflect possible wetland substrate sources or runoff inputs. Maximum dissolved NH_3 concentrations were measured in May 1996 at Spile Lake and Little Bean Marsh, and in January 1997 at Forker Oxbow. Lowest levels were measured in summer 1996 and early spring 1997 in all three wetlands. Maximum dissolved NO_2 and NO_3 concentrations were measured in late fall 1996 and winter 1996–97 at the three wetlands. Maximum concentrations of NO_2 did not exceed 0.04 mg/L, whereas maximum concentrations of NO_3 were 0.08 to 0.69 mg/L. Highest dissolved P concentrations were measured from May to July 1996 in each wetland. This perhaps corresponds to the period of greatest P inputs from runoff and P release from bottom sediments. Lowest concentrations of dissolved P were measured from late summer 1996 through spring 1997 in each wetland. Highest dissolved PO_4 concentrations were measured in July 1996 at Spile Lake (0.07 mg/L), June 1996 at Little Bean Marsh (0.16 mg/L), and April, July, and August 1996 at Forker Oxbow (0.02 mg/L). Dissolved PO_4 concentrations generally were below the detection limit of 0.01 mg/L from October 1996 through May 1997 at Spile Lake, August 1996 through May 1997 at Little Bean Marsh, and November 1996 through May 1997 at Forker Oxbow.

Total N and P concentrations were substantially greater than dissolved values, indicating most N and P in the wetlands occurred in the particulate form. While dissolved values of N forms generally were at or near detection levels, the maximum total ammonia plus organic N concentrations were between 2.5 and 3.2 mg/L in April 1996 at Spile Lake and Little Bean Marsh and in July 1996 at Forker Oxbow. These concentrations may reflect higher particulate N amounts associated with phytoplankton populations. The Forker Oxbow total ammonia plus organic N concentrations increased in June and July 1996, following flood inundations from Locust Creek and the likely input of nutrient concentrations from agricultural runoff in this basin. The lowest total ammonia plus organic N concentrations were measured in August or September 1996 at each wetland; and minimums were from 0.2 to 0.5 mg/L. Highest total P concentrations were measured in April 1996 in Spile Lake (0.42 mg/L), May 1996 in Little Bean Marsh (0.83 mg/L) and August and September 1996 in Forker Oxbow (0.34 mg/L). Minimum total P concentrations were measured in January 1997 in Spile Lake (0.03 mg/L), February 1997 in Little Bean Marsh (less than 0.01 mg/L), and April 1996 in Forker Oxbow (0.06 mg/L). Perhaps these minimums correspond to low phytoplankton populations.

Despite differences in hydrology, location, and land-use characteristics of the wetlands, the distribution of N and P forms generally was similar in all three systems. No statistically significant difference in the dissolved N forms of NH_3 (Kruskal-Wallis test, $p = 0.52$), NO_2 ($p = 0.32$), NO_3 ($p = 0.38$), or total ammonia plus organic N concentrations ($p = 0.08$) in the three wetlands were determined. Dissolved P concentrations also were similar in all three wetlands ($p = 0.75$). The similarities in the various forms of N in the wetlands as well as dissolved P despite probable differences in loads may indicate that concentrations of these constituents are controlled by autochthonous (within wetland) processes. Total P concentrations were significantly different (Kruskal-Wallis test, $p = 0.02$), with total P concentrations significantly lower in Little Bean Marsh than in Forker Oxbow (rank transform test, Tukey's test). Hoyer and Reid (1982) did not detect a significant difference in total P concentrations when comparing December 1981 samples from wetlands in the Glaciated Plains and the Mississippi lowlands of Missouri. The distribution of dissolved PO_4 was also significantly different ($p = 0.02$) with lower concentrations in Forker Oxbow than in Spile Lake (rank transform test, Tukey's test). The low dissolved PO_4 concentrations (an indication of available phosphorus) and high total P concentrations at Forker Oxbow may be an indication of a relatively large phytoplankton population at this site.

Statistical Comparison of Physicochemical Properties and Nutrient Concentrations Between Wetlands and Streams

Currently (1998), wetlands in Missouri are subject to the same State water-quality standards as adjacent streams and rivers (Missouri Department of

Natural Resources, 1994). Because these two types of systems differ in basic hydrologic and biologic characteristics, it was hypothesized that the chemical properties of these systems also would be different. Following are the results of a statistical comparison of wetland physicochemical properties and nutrients with that of streams and rivers in the same natural divisions with similar water-quality information available.

Currently (1998), approximately 48 streams in Missouri are sampled monthly or quarterly as part of the USGS Ambient Water-Quality Monitoring Network. A data set of physicochemical properties and nutrient forms from streams and rivers in the same natural divisions as the wetlands was generated for January 1, 1996 through May 31, 1997 (U.S. Geological Survey, 1997; 1998). Four streams with suitable information were available from the Osage Plains natural division, two were available from the Big Rivers natural division, and five were available from the Glaciated Plains natural division. A data set of physicochemical properties was generated for each study wetland to match the date and time of the corresponding streams' physicochemical property measurements to the nearest hour. The two-week turbidity results and monthly nutrient results from the wetlands were compared to the monthly and quarterly nutrient data collected from the selected streams and rivers. Statistical comparisons were made between wetland and river specific conductance, pH, near-surface water temperature, dissolved oxygen, and turbidity. Statistical comparisons also were made between wetland and selected stream nutrient concentration data including dissolved NO_2, dissolved NO_3, dissolved NH_3, total ammonia plus organic N, dissolved P, total P, and dissolved PO_4. A lack of nutrient data from the four Osage Plains natural division streams allowed only comparisons of total ammonia plus organic N and total P concentrations. Statistical comparisons were made using the Mann-Whitney test at a significance level of 0.05.

The statistical comparison results indicate that there is a significant difference in the distribution of most of the physicochemical properties and nutrient forms in the three wetlands compared to these values in selected streams in the same natural divisions (table 3). Of the 31 individual comparisons, 21 resulted in a statistically significant difference between wetland and stream physicochemical properties and nutrient forms. Most of these differences represent lower values in wetlands relative to streams.

Specific conductance values were significantly less in the three wetlands than in rivers in the Big Rivers and Glaciated Plains natural divisions, but there was no significant difference between specific conductance in Spile Lake and in rivers in the Osage Plains natural division. Rivers may have a larger part of the hydrologic budget supplied by ground water than what has been observed in the wetlands. Because ground water has a greater contact time with soils and minerals, the dissolved ion concentrations would tend to be greater in rivers, resulting in higher specific conductance values. Judging by the median specific conductance values in Osage Plains streams relative to streams in the other natural divisions, it seems the lack of a difference between specific conductance values in Spile Lake and selected Osage Plains streams is because of the relatively low specific conductance values in these streams for this period, rather than higher specific conductance values in Spile Lake.

Values of pH and dissolved oxygen were significantly less in wetlands in each of the three represented natural divisions than in corresponding selected streams (table 3). The lower pH and dissolved-oxygen concentrations in wetlands could be explained by thermal stratification, which limits the transfer of dissolved oxygen and increases CO_2 concentrations below the thermocline. The limited water-quality data for the streams does not allow for examination of the diurnal differences in these systems, which could result in even greater differences during specific time periods.

Dissolved NO_3, dissolved P, and dissolved PO_4 concentrations were significantly less in Little Bean Marsh and Forker Oxbow than in selected streams in the Big Rivers and Glaciated Plains natural divisions (table 3). The lower wetland dissolved-nutrient concentrations compared with the streams may be explained by possible lower inputs, denitrification, and higher macrophyte and algae nutrient uptake in wetlands relative to streams.

The distribution of near-surface temperatures in wetlands and rivers in the same natural division are statistically similar (table 3). Temperature was the only property or constituent tested that was not statistically different between wetlands and streams for all three comparisons. This indicates that the temperatures of streams in close proximity to the studied wetlands can provide a reasonable estimate of near-surface wetland temperatures.

Table 3. Summary of statistical comparison (Mann-Whitney test) of wetland physicochemical properties and constituents to those of selected streams in the same natural division

[N, nitrogen; =, no significant difference; -, wetland values significantly less than streams; +, wetland values significantly greater than streams; na, insufficient data for comparison; p, probability of error associated with test results; <, less than; Wn, number of wetland observations; Sn, number of stream observations]

Wetland and natural division	Physicochemical properties and constituents											
	Specific conduc-tance	**pH**	**Temper-ature**	**Dissolved oxygen**	**Turbidity**	**Ammonia as N, dissolved**	**Nitrite as N, dissolved**	**Nitrate as N, dissolved**	**Organic plus ammonia N, total**	**Phos-phorus, dissolved**	**Phos-phorus, total**	**Ortho-phos-phate, dissolved**
Spile Lake[1]	=	-	=	-	+	na	na	na	=	na	+	na
Osage Plains	p=0.48	p=0.02	p=0.50	p<0.01	p<0.01				p=0.22		p=0.03	
	Wn=28	Wn=28	Wn=25	Wn=25	Wn=40				Wn=15		Wn=15	
	Sn=28	Sn=28	Sn=25	Sn=25	Sn=17				Sn=24		Sn=25	
Median values	**Wn=270**	**Wn=7.0**	**Wn=10.0**	**Wn=6.2**	**Wn=18**				**Wn=.90**		**Wn=.15**	
	Sn=264	**Sn=7.3**	**Sn=9.8**	**Sn=9.4**	**Sn=2.4**				**Sn=.84**		**Sn=.06**	
Little Bean Marsh[2]	-	-	=	-	-	=	-	-	-	-	-	-
Big Rivers	p<0.01	p<0.01	p=0.19	p<0.01	p<0.01	p=0.07	p<0.01	p<0.01	p=0.04	p<0.01	p<0.01	p<0.01
	Wn=38	Wn=38	Wn=38	Wn=38	Wn=39	Wn=14	Wn=14	Wn=14	Wn=15	Wn=13	Wn=15	Wn=14
	Sn=38	Sn=38	Sn=38	Sn=38	Sn=19	Sn=22	Sn=20	Sn=22	Sn=38	Sn=22	Sn=38	Sn=22
Median values	**Wn=452**	**Wn=7.6**	**Wn=11.8**	**Wn=6.4**	**Wn=2.2**	**Wn=.02**	**Wn=.005**	**Wn=.025**	**Wn=.80**	**Wn=.01**	**Wn=.06**	**Wn=.005**
	Sn=692	**Sn=8.0**	**Sn=10.0**	**Sn=10.1**	**Sn=51**	**Sn=.04**	**Sn=.02**	**Sn=1.3**	**Sn=.91**	**Sn=.08**	**Sn=.22**	**Sn=.069**
Forker Oxbow[3]	-	-	=	-	-	=	=	-	=	-	=	-
Glaciated Plains	p<0.01	p<0.01	p=0.26	p<0.01	p<0.01	p=0.24	p=0.2	p<0.01	p=0.36	p<0.01	p=0.26	p<0.01
	Wn=77	Wn=78	Wn=78	Wn=78	Wn=37	Wn=14	Wn=14	Wn=14	Wn=15	Wn=13	Wn=15	Wn=13
	Sn=77	Sn=78	Sn=78	Sn=78	Sn=26	Sn=25	Sn=25	Sn=25	Sn=75	Sn=25	Sn=75	Sn=25
Median values	**Wn=251**	**Wn=7.2**	**Wn=9.2**	**Wn=3.5**	**Wn=26**	**Wn=.02**	**Wn=.0075**	**Wn=.025**	**Wn=1.1**	**Wn=.02**	**Wn=.16**	**Wn=.005**
	Sn=376	**Sn=7.9**	**Sn=9.0**	**Sn=10.8**	**Sn=38**	**Sn=.04**	**Sn=.02**	**Sn=.69**	**Sn=1.0**	**Sn=.05**	**Sn=.20**	**Sn=.04**

[1] Wetland values compared to streams in Osage Plains natural division in Missouri [East Fork Drywood Creek at Prairie State Park (06917630), Fleck Creek at Prairie State Park (06917635), Osage River above Schell City, Mo. (06918070), and North Fork Panther Creek tributary near Appleton City, Mo. (06918200); see figure 1 for site locations].

[2] Wetland values compared to streams in Big Rivers natural division in Missouri [Missouri River at St. Joseph, Mo. (06818000), and Missouri River at Hermann, Mo. (06934500); see figure 1 for site locations].

[3] Wetland values compared to streams in Glaciated Plains natural division in Missouri [South Fabius River near Taylor, Mo. (05500000), Cuivre River near Troy, Mo. (05514500), Nodaway River near Graham, Mo. (06817700), Grand River near Sumner, Mo. (06902000), and Chariton River near Prairie Hill, Mo. (06905500); see figure 1 for site locations].

Turbidity was significantly less in Little Bean Marsh and Forker Oxbow than in selected streams in the Big Rivers and Glaciated Plains natural divisions (table 3). Turbidity values were significantly greater in Spile Lake than in streams in the Osage Plains natural division. Most turbidity values for the Osage Plains streams were available from the East Fork Drywood Creek site. This site has a small, relatively undisturbed, drainage area (2.28 mi^2) and the typical low-flow conditions encountered during sampling could account for the significantly lower turbidity values.

Dissolved NO_2 concentrations were less in Little Bean Marsh relative to concentrations at the Missouri River monitoring sites in the Big Rivers natural division, but there was no significant difference between NO_2 concentrations in Forker Oxbow and selected streams in the Glaciated Plains natural division (table 3). Nitrite seldom occurs at concentrations substantially above detectable levels in natural waters. Median concentrations in all three wetlands were less than 0.01 mg/L, compared to 0.02 mg/L in the selected Glaciated Plains and Big Rivers natural division streams. At these low concentrations the statistical differences may be caused by laboratory-analysis error.

Total ammonia plus organic N concentrations were significantly less in Little Bean Marsh than in the Missouri River sites (table 3), but there was no significant difference in total ammonia plus organic N concentrations when comparing between the other wetlands and the respective selected streams. The median total ammonia plus organic N concentration in Little Bean Marsh was the lowest of the three wetlands (although it was not determined to be statistically different from concentrations in the other wetlands), whereas total ammonia plus organic N concentrations at the Missouri River sites were similar to those in the other natural divisions, indicating that the relatively lower concentrations in Little Bean Marsh may account for this significant difference.

Total P concentration was the only constituent tested that produced a different result in each of the three natural division comparisons. Total P concentrations were significantly greater in Spile Lake relative to selected streams in the Osage Plains; total P concentrations in Little Bean Marsh were significantly less than concentrations in selected streams in the Big Rivers natural divisions; and there was no significant difference between total P concentrations in Forker Oxbow and selected streams in the Glaciated Plains natural division. Most total P values in the Osage Plains streams were from East Fork Drywood Creek, located in Prairie State Park. Total P concentrations from this remnant prairie are low in comparison with the local Spile Lake drainage, which receives some agricultural runoff. The median total P concentration in Little Bean Marsh was the lowest of the three wetlands, despite a large part of the local drainage being under row crop production, whereas total P concentrations in the Big Rivers streams were similar to those in streams in the Glaciated Plains and greater than those of streams in the Osage Plains. The low total P concentrations in Little Bean Marsh account for the significant difference when compared with total P values in the Big Rivers streams. There seems to be less P tied up in seston (inorganic and organic particulate material in the water column) in this wetland than in the other two wetlands.

Results from the statistical comparison between wetland and stream physicochemical properties and nutrient forms are similar among the three natural divisions. Of the 12 properties and constituents compared, 10 showed consistent results between 2 or more tested natural divisions (pH, water temperature, dissolved oxygen, NH_3, NO_2, dissolved P, and dissolved PO_4, specific conductance, turbidity, total ammonia plus organic N), and 2 showed different results between each of the tested natural divisions (NO_2, total P).

Whereas many statistically significant differences were determined in the above comparisons between wetland physicochemical properties and constituents and those of selected streams, these differences may not represent practical or chemically-significant differences and may not form a logical basis for modifications of existing water-quality standards. A statistical difference between two data sets does not necessarily constitute a practical or chemically-significant difference; determination of a practical or chemically-significant difference between the two systems may involve other considerations. Exceptions have been made in existing State standards for unaltered water bodies that may have property or constituent values outside the range of a standard as a result of natural processes. Such is the case for dissolved oxygen in lakes and reservoirs where thermal stratification affects dissolved-oxygen concentrations in the hypolimnion of these systems (Missouri Department of Natural Resources, 1994). Wetlands can be similarly affected by this process, and differences in dissolved-oxygen concentrations between the study wetlands and selected streams were substantial. Other non-represented types of wetland systems in the State (fens, swamps, wet prairie) may have differing water-quality characteristics than those presented in this report. The physical characteristics of the study wetlands and the location of the monitoring site within the water column lead to the apparent similarities with lakes and reservoirs. There were not adequate lake or reservoir data to conduct a similar statistical comparison between the study wetlands and Missouri lakes or reservoirs.

Wetland Response During Hydrologic and Seasonal Extremes

Having physicochemical properties and concurrent hydrologic information available hourly allows for analyses of these properties over the course of short-term hydrologic events (flooding) or for viewing simultaneous diurnal fluctuations of several properties during a selected temperature period. The hourly data collected during three periods were selected to illustrate the response of selected physicochemical properties to floods, high temperatures, and ice formation. Examples of flooding responses include a flood at Spile Lake (February 20 to 28, 1997), a heavy rainfall-runoff period at Little Bean Marsh (April 10 to 15, 1997), and a flood at Forker Oxbow (May 25 to 30, 1997). Examples of seasonal extremes include a low-water high-temperature period at Spile Lake (August 1 to 7, 1996), and a low-water low-temperature period at Spile Lake (February 1 to 7, 1996).

Hydrologic Extremes

To determine the most typical responses in physicochemical properties from hydrologic extremes, 13 flooding or rainfall-runoff events were analyzed from the three wetlands. The hydrology of the wetlands was sufficiently different that a single response was not representative of all three wetlands. The source and timing of runoff is substantially different in each wetland drainage system. Spile Lake receives local runoff and, at times, delayed backwater flooding from the Little Osage and Marmaton Rivers; Little Bean Marsh is protected from flooding from the Missouri River and rainfall, local runoff, and ground-water interactions are important in this system; Forker Oxbow often receives floodwaters from Locust Creek. Therefore, an event from each site was chosen to illustrate a typical response in physicochemical properties from flooding.

An event on February 20 to 28, 1997, at Spile Lake was chosen to represent a physicochemical-properties flooding response at this site (fig. 20). A rainfall on February 20 to 21 of 2.89 in. brought about a 1-ft increase in the Spile Lake stage on February 21. A second rise occurred in the Spile Lake stage as backwater entered from the Little Osage and Marmaton drainages, resulting in an additional 6-ft rise in stage. The Spile Lake stage mirrored that of adjacent Little Osage River until the river stage decreased below flood stage on February 26.

With substantial changes in stage, the multi-parameter water-quality sensor measures both temporal and spatial variations in the wetland. Some of the fluctuations in specific conductance, pH, and dissolved oxygen may be explained by a change in the vertical position of the multi-parameter sensor.

Specific conductance values decreased from 220 μS/cm to about 140 μS/cm on February 20-21 as rainfall diluted the wetland (fig. 18). Specific conductance values fluctuated from about 150 to 240 μS/cm from February 20 to 28, with local runoff inputs and later flooding from the Little Osage River. Specific conductance values seemed to follow an inverse trend with stage during the recession hydrograph as values climbed to about 320 μS/cm on February 27, but in the later stages of the event, specific conductance values were near pre-event levels. The additional 1.40 in. of rainfall that occurred on February 26 had no substantial effects on stage or specific conductance values in the wetland.

pH values were about 6.8 at the start of the rainfall event and fluctuated between 6.7 and 7.3 over the 9 days shown in figure 18. It appears the increase in pH from 6.7 to 7.2 on February 22 corresponded with backwater inflows from the Little Osage and Marmaton Rivers.

Spile Lake was weakly thermally stratified on February 20 before the rainfall event but quickly destratified as both near-surface and near-bottom water temperatures decreased from the cooler inflows (fig. 20). The wetland was well mixed for the remainder of the event.

The Spile Lake dissolved-oxygen concentration before the start of the event was about 5 mg/L. The subsequent rise in concentration to about 9 mg/L on February 20 appears to correspond with precipitation (fig. 20). The dissolved-oxygen concentrations remained fairly constant for the remainder of the event.

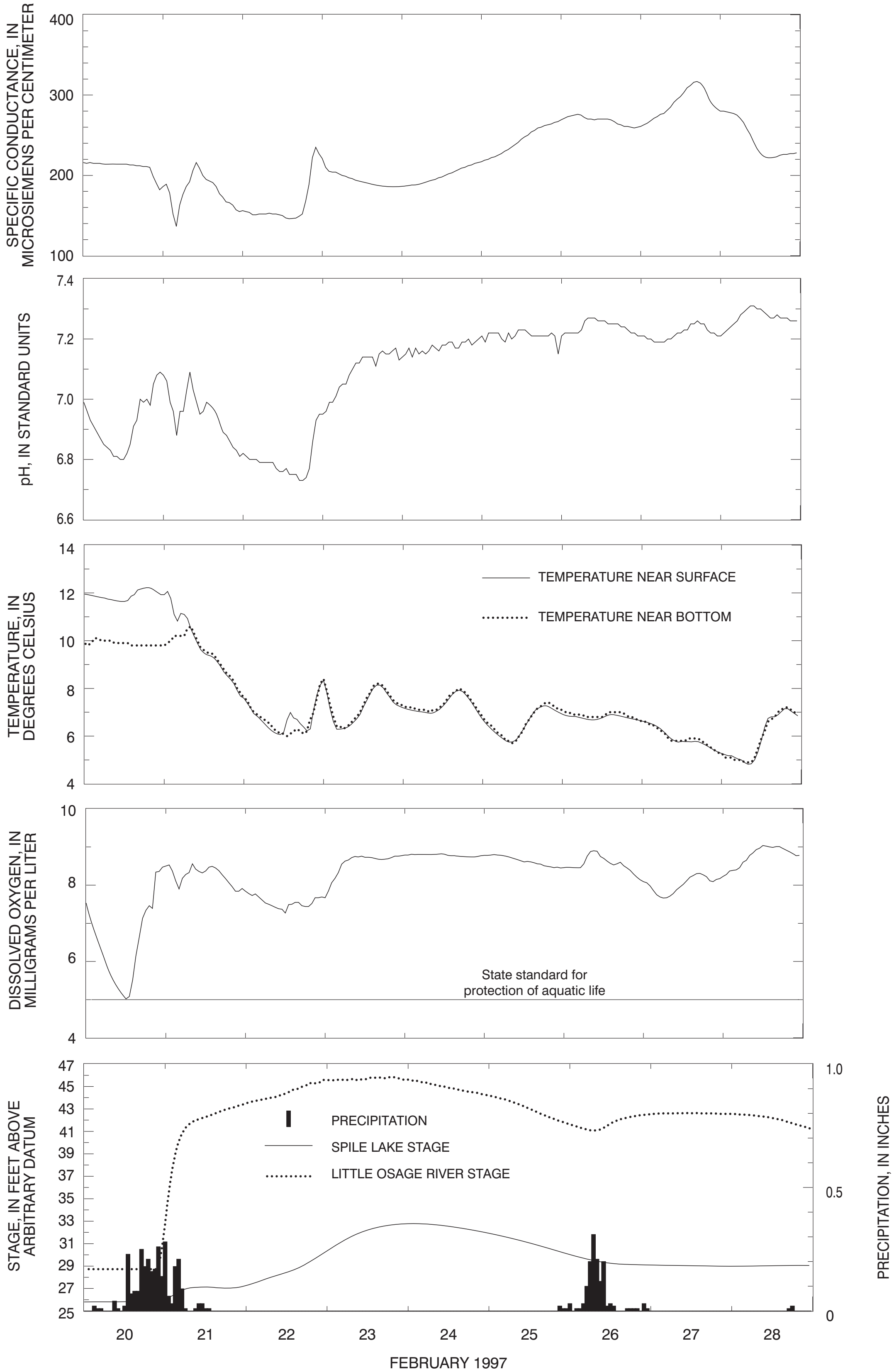


Figure 20. Temporal variability of physiochemical properties and stage at Spile Lake during a February 20-28, 1997, flood event from adjacent Little Osage and Marmaton Rivers.

Little Bean Marsh is protected by a levee system, and substantial stage changes were uncommon during the study period. Whereas there were no true floods at this site, the rainfall-runoff event during April 10 to 15, 1997, was selected to illustrate the response of this system to a hydrologic event (fig. 21). A rainfall of 5.38 in. during 2 days produced about a 1.4-ft rise in the Marsh stage on April 11, 1997. Despite an additional 5.89 in. of rainfall on April 12, there were no noticeable effects on the Little Bean Marsh stage. Apparently inflows and outflows were similar during this period.

Specific conductance values decreased from 600 μS/cm before the event to about 400 μS/cm on April 11 near the peak of the stage hydrograph (fig. 21). The specific conductance of the wetland remained near 400 μS/cm for the remainder of the event. Although characteristic of this site, the substantial drop in specific conductance followed by steady values during a runoff event was not observed at either of the two other sites.

pH values in Little Bean Marsh were about 8.4 before the event but decreased to about 8.1 during the rise in stage (fig. 21). Fluctuations in pH from April 12 to 15 mirrored those in dissolved oxygen for the remaining three days, indicating effects from photosynthesis.

The Marsh near-surface and near-bottom temperatures were similar at the start of the event, but inverse stratification intensified with rainfall and runoff inputs cooling near-surface waters (fig. 21). By April 15, temperatures were again similar throughout the water column.

Dissolved-oxygen concentrations in Little Bean Marsh were about 9 mg/L before the runoff event, and concentrations increased to near 12 mg/L following the increase in stage (fig. 21). On April 12, dissolved-oxygen concentrations and pH began showing diurnal fluctuations characteristic of photosynthesis affects.

Most flooding events that occurred during the study period took place at Forker Oxbow (7 of 13 studied events). Whereas the four events analyzed at Spile Lake seemed to produce somewhat different results, the responses from Forker Oxbow were very similar for most events. The event most representative of the response of physicochemical properties to flooding in Forker Oxbow occurred May 25 to 30, 1997 (fig. 22). Rainfall totalling 2.42 in. during 3 days, and the subsequent runoff, resulted in a 4-ft rise in stage in the wetland on May 28.

The runoff inputs on May 27 brought about an initial decrease in specific conductance from 300 μS/cm to 160 μS/cm, followed by a rapid rise to more than 300 μS/cm (fig. 22). As was the case in many of the flood and runoff events analyzed, after the initial fluctuations with runoff, the specific conductance values stabilized on May 30 and were similar to those on May 27, before the event.

pH values were about 7.3 until the stage increased on May 27 and May 28 and brought about a short-lived rise in pH to about 7.6 (fig. 22). At this point, pH appears to mirror changes in the stage hydrograph and decrease to about 7.1.

Thermal stratification was present in Forker Oxbow on May 25 and 26, but precipitation inputs resulted in a decrease in near-surface and near-bottom temperature, to a lesser extent, as the wetland became destratified (fig. 22). Thermal stratification was minimal in the wetland for the remainder of the event.

Dissolved-oxygen concentration was about 3.5 mg/L before stage increases and then increased to near 7 mg/L with the increase in stage (fig. 22). Much like pH, the dissolved-oxygen concentrations mirrored the recession hydrograph and concentrations decreased to near pre-event levels.

The examples given represent some of the more typical responses, but only one of many possible responses of each wetland to a flooding scenario. The changes in physicochemical properties of the wetlands, in response to local runoff and floodwaters from adjacent rivers, are determined by several factors including antecedent conditions in the wetlands, the timing of the event, and the characteristics of the floodwaters. To summarize all the observed responses in physicochemical properties in general terms, specific conductance in the wetlands had variable responses to the initial rainfall, but typically decreased with initial runoff inputs. The ending specific conductance values were typically near antecedent values. pH was quite variable between sites and events but typically resembled changes in the hydrograph or dissolved-oxygen concentrations. The typical effects of wetland flooding on water temperature was to temporarily destratify the wetland (Little Bean Marsh was an exception), if stratification existed before the event, with the wetland returning to pre-event conditions as floodwaters receded. Finally, dissolved-oxygen concentrations typically had variable effects from the initial rainfall, but tended to resemble changes in the hydrograph thereafter. Properties that resembled changes in the stage hydrograph may have

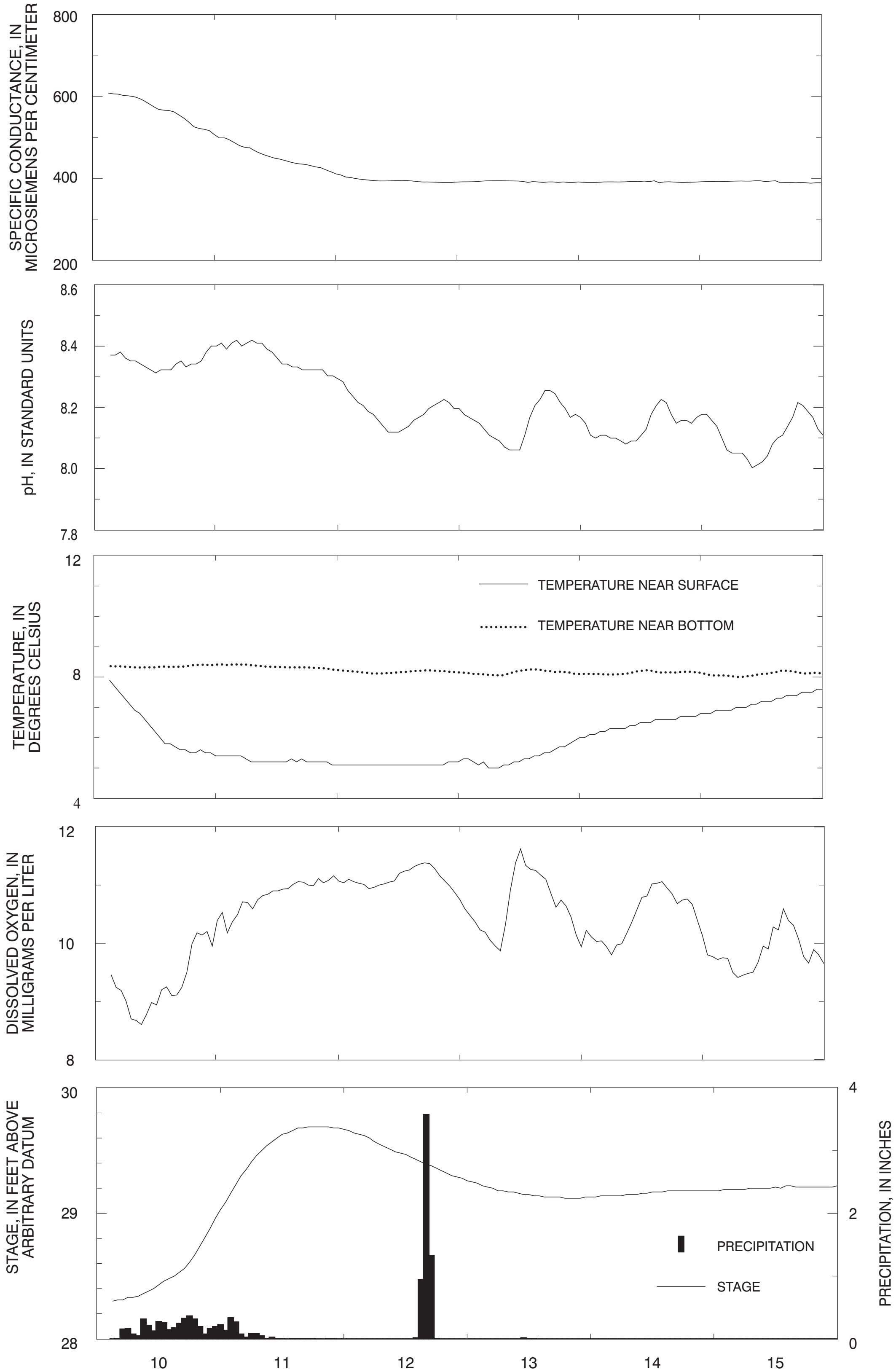


Figure 21. Temporal variability of physiochemical properties and stage at Little Bean Marsh during an April 10-15, 1997, rainfall-runoff event.

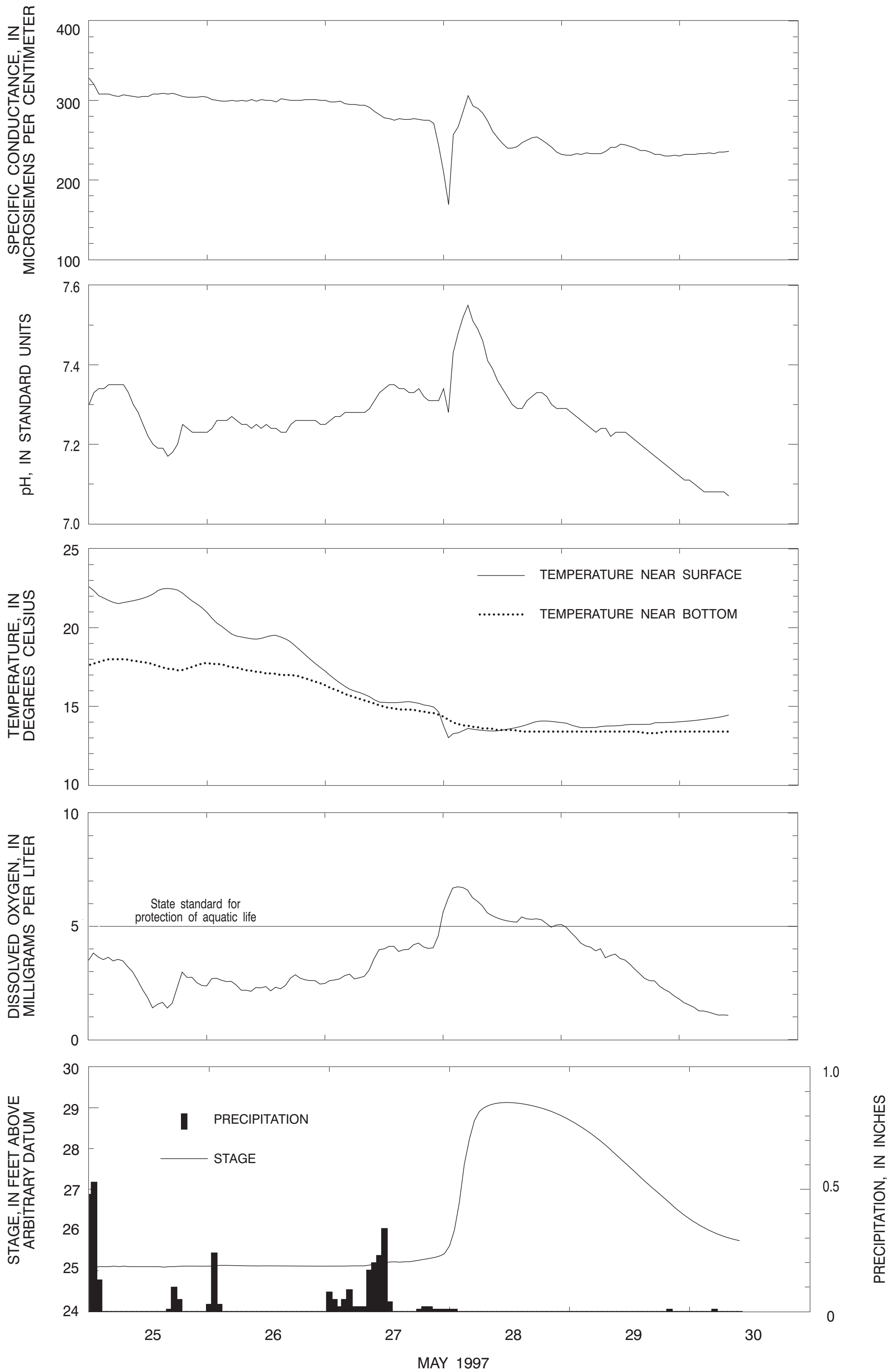


Figure 22. Temporal variability of physiochemical properties and stage at Forker Oxbow during a May 25-30, 1997, flood event from adjacent Locust Creek.

been responding to changes in the multi-parameter water-quality sensor position in the water column, rather than changes within a specific water layer, although based on homogeneous near-surface and near-bottom temperatures the wetlands usually were well mixed during the events.

Seasonal Extremes

In addition to wetland responses to hydrologic extremes such as floods, another area of interest is short-term temporal variations of physicochemical properties in wetlands as a result of seasonal variations. A summer period of relatively high water temperatures and low-stage conditions is shown for Spile Lake from August 1 to 7, 1996, followed by a winter period of low-temperature, low-stage conditions at Spile Lake from February 1 to 7, 1996.

The stage declined during the August 1 to early August 7 period (fig. 23). The small rise in stage on August 7 was the result of 2.84 in. of rain. The decline in stage was the result of evapotranspiration and possibly seepage losses.

Specific conductance values showed small (about 10 μS/cm) diurnal fluctuations from 185 to 195 μS/cm, and an overall increase with the declining stage (fig. 23). Specific conductance dropped from 190 to about 165 μS/cm in conjunction with rainfall on August 7.

pH values in Spile Lake remained near 6.80 until the August 7 rainfall, and then increased slightly to 6.9 (fig. 23). Whereas there were fluctuations in pH, there were no strong diurnal fluctuations such as those that occurred in dissolved-oxygen concentrations.

Diurnal formation and dissipation of thermal stratification was common during August 1 to 7 (fig. 23). Near-surface water temperatures increased during the daylight hours about 2 to 4 °C and cooled in the evening to near-bottom water temperatures.

Dissolved-oxygen concentrations showed strong diurnal fluctuations during the period, indicating effects of photosynthesis activity (fig. 23). Peak dissolved-oxygen concentrations remained below 4 mg/L for the period. An unusual double peak was consistently measured during the diurnal fluctuations in dissolved oxygen. The largest peak occurred between 5 and 6 p.m., and a second, smaller peak occurred between 2 and 3 a.m.

This represents just one typical summer scenario. Precipitation and runoff inputs can greatly affect the physicochemical property characteristics. This represents some common or expected responses of the study wetlands to summer conditions in that thermal stratification was present, either diurnal or strongly established; dissolved-oxygen concentrations were low but may have large diurnal fluctuations; and pH was relatively low and may or may not have exhibited diurnal fluctuations.

The last wetland response scenario presented is for a winter low-stage period at Spile Lake from February 1to 7, 1996 (fig. 24). This winter scenario corresponds to a period of ice formation on the wetland. There was no ice cover observed on Spile Lake on January 16, 1996, but by February 4, 1996, there was more than 4 in. of ice cover. Specific conductance values indicate that much of the ice cover probably formed between February 2 and 3; values increased about 25 percent during these 2 days. In shallow wetlands, even a few inches of ice formation can result in a substantial loss in water volume and increase in ion concentration in the remaining waters. Conversely, specific conductance values decreased rapidly on February 7, 1996, as temperatures increased and wetland waters were diluted by melting ice.

pH values during this period were relatively low and constant (fig. 24). The minor pH changes that occurred appeared to be associated with changes in temperature and dissolved oxygen.

Inverse thermal stratification was evident from February 1 to 7 as near-bottom water temperatures exceeded near-surface temperatures (fig. 24). Diurnal temperature fluctuations were present in near-surface waters but were less than 2 °C.

Dissolved-oxygen levels were low during this period of inverse thermal stratification (fig. 24). The minor diurnal fluctuations in dissolved oxygen and pH may be the result of limited phytoplankton activity that can occur under clear ice cover. The highest dissolved-oxygen concentration (about 7.7 mg/L) measured during this period occurred on February 7 as near-surface temperature rose, ice cover melted, and the wetland water column destratified, allowing mixing and reaeration to occur.

This represents one winter scenario in this wetland system and indicates the effects of ice formation and loss on some of the physicochemical properties of the system. Even in the relatively dry low-temperature periods, there can be substantial changes in physico-

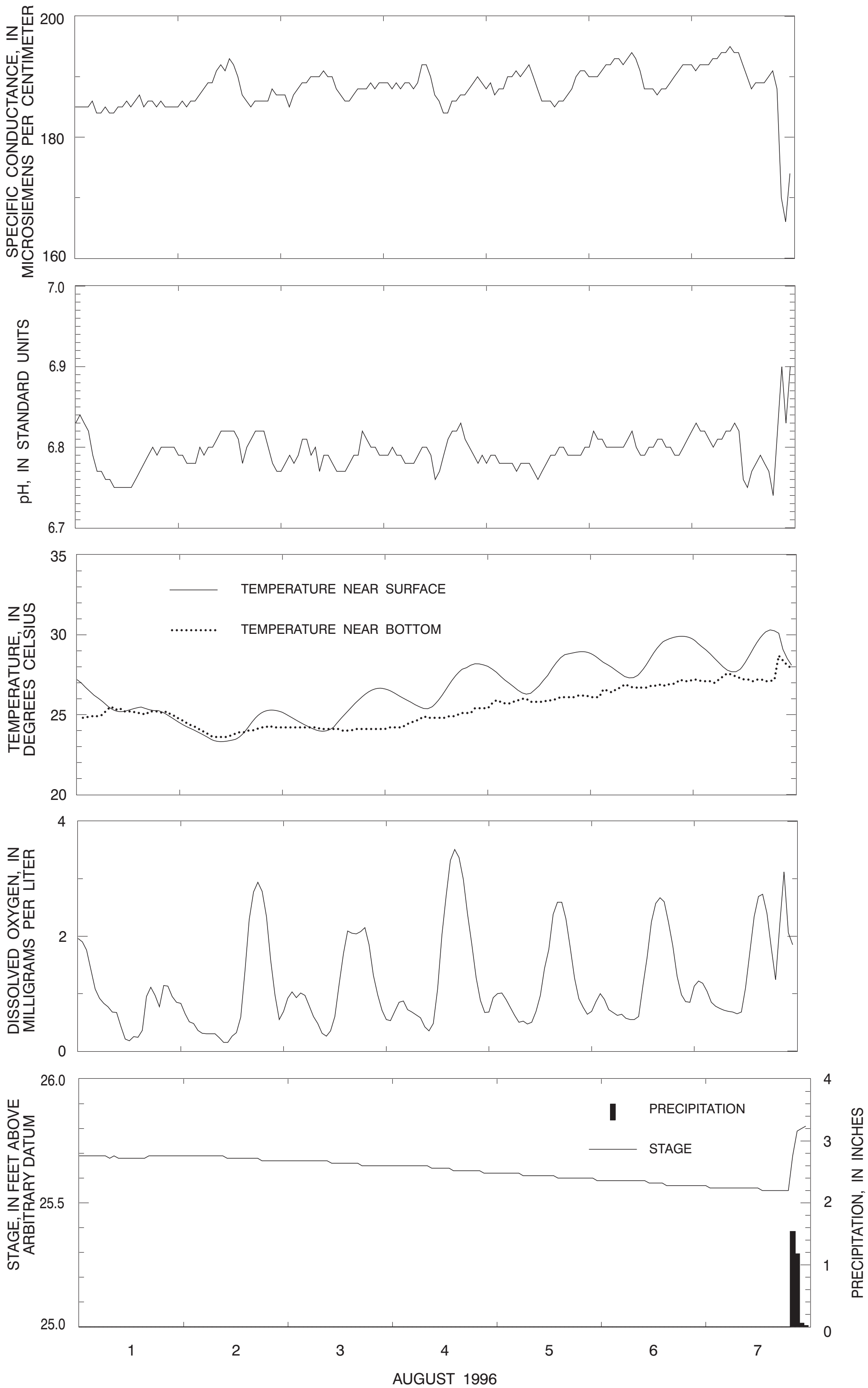


Figure 23. Temporal variability of physiochemical properties and stage at Spile Lake during a summer low-stage period.

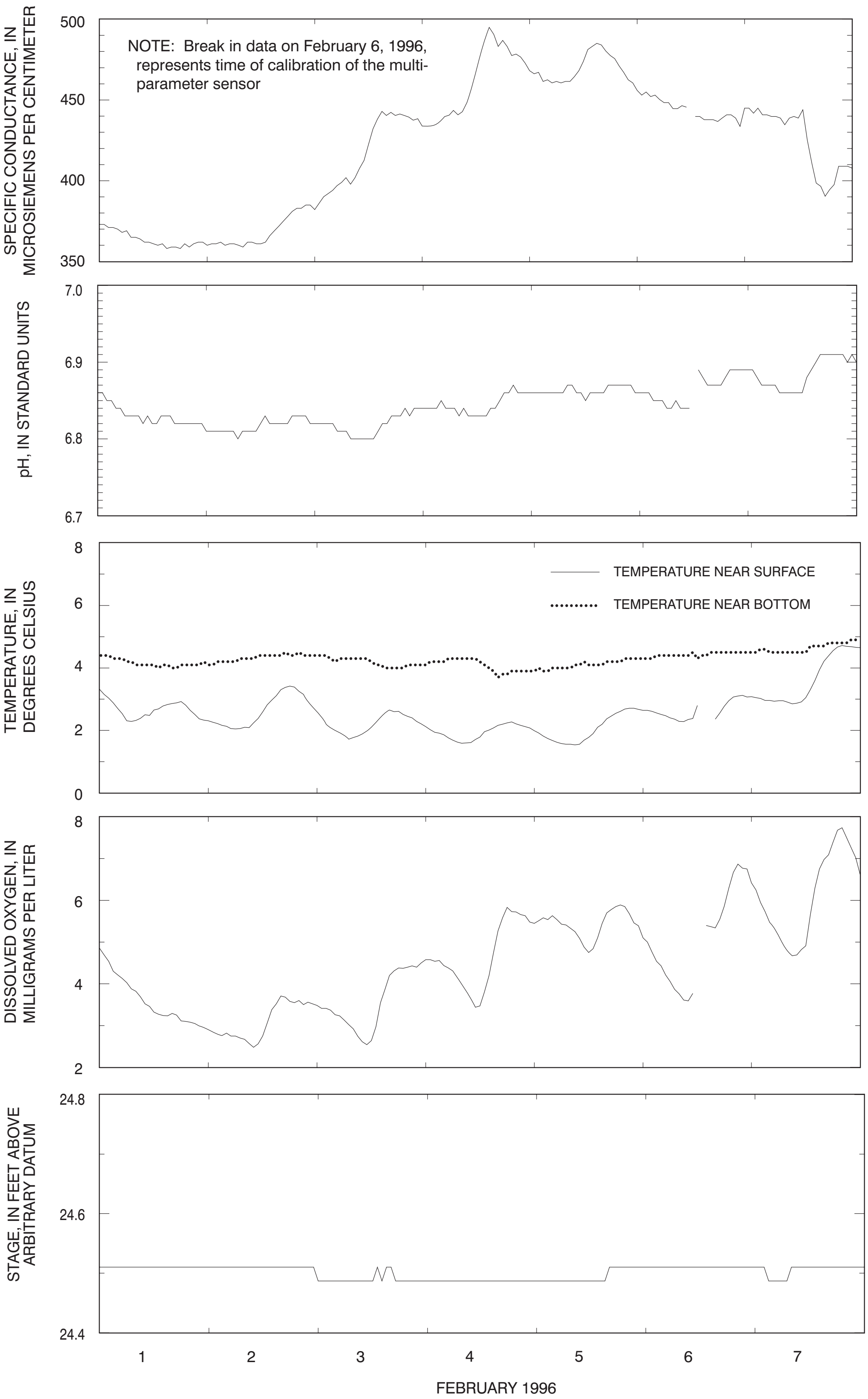


Figure 24. Temporal variability of physiochemical properties and stage at Spile Lake during a winter low-stage period.

chemical properties during 1 or 2 days as a result of ice formation and melt. This example demonstrates the important effects ice cover can have on the physicochemical characteristics of Missouri wetlands.

INVERTEBRATE COMMUNITY CHARACTERISTICS

The invertebrate communities of Spile Lake, Little Bean Marsh, and Forker Oxbow were sampled in March, June, August, and October 1996, using four different methods. Invertebrate information can be valuable in the bioassessment of wetland health and structure, with time and between wetlands. The invertebrate information from the three study sites serves as reference data sets for future comparisons.

Some of the invertebrate samples (snag and sweep samples) were analyzed by two different laboratories because the original analyses by SMSU did not have the taxonomic detail necessary for the calculation of desired metrics. The normalized invertebrate abundance and the Percent Contribution of the Dominant Taxon data for all four sample methods are presented from the data obtained from SMSU analyses. Information regarding invertebrate population diversity and organic pollution tolerance was determined from more detailed analyses of the original snag and sweep samples by BSA. The snag and sweep samples were chosen for detailed analyses by BSA because the original analyses indicated these samples may provide the best balance of abundance and diversity.

Wetland invertebrate abundances varied substantially with time and sampling method. To determine temporal variability in invertebrate communities, the abundance data were arbitrarily normalized to March samples (each site and method combination abundances are displayed as the ratio of that abundance to the March sample combination abundances) (fig. 25). For consistency, the ratios of snag, core, and Hester-Dendy samples were computed using results on abundance values per square feet, whereas sweep sample ratios were computed based on a number of individuals per 0.5 hour sample time. The Spile Lake snag sample abundances were greatest in March, sweep sample abundances were greatest in October, core sample abundances were greatest in March and October, and Hester-Dendy sample abundances were greatest in June (fig. 25). The Little Bean Marsh snag samples had the greatest abundance in March, the sweep sample abundances were greatest in March and October, and the core and Hester-Dendy samples had the greatest abundances in August. At Forker Oxbow, the snag samples had the greatest abundances in October while sweep and core samples had the greatest abundances in March. The Forker Oxbow Hester-Dendy abundances were greatest in August. The results reflect the temporal variability in the life cycles of the invertebrate populations along with the effects of the unique physical and chemical conditions at each site. There were no consistent population trends found among sites with time.

To determine variability in invertebrate communities with sampling method, the invertebrate populations were arbitrarily normalized to the snag sample abundances. Results indicate that core samples generally provided the greatest abundances (fig. 26). Spile Lake samples were dominated by abundances in core samples in all four sample rounds. The greatest abundances at Little Bean Marsh in March were detected in the core sample, although sweep samples provided the greatest abundances in remaining sample rounds. At Forker Oxbow, the greatest abundances in March and August also were from core samples, with sweep samples providing the greatest abundances in June, and snag samples in October. Little Bean Marsh samples displayed the greatest consistency in abundances between methods.

The Percent Contribution of the Dominant Taxon provides an indication of the community diversity and balance (U.S. Environmental Protection Agency, 1989), and these values vary by method and collection round. Results are presented for the dominant taxon of invertebrates in each of the three wetlands sampled for each method and sample period (fig. 27). These results indicate that while the core samples had the greatest abundances, these samples also had the lowest diversities. The major proportion of the invertebrate communities in the core samples were represented by few taxa, and, in some cases, the entire invertebrate community was represented by three or fewer taxa. The snag samples seem to have the greatest community balance, followed by the sweep and Hester-Dendy samples.

The data from snag and sweep samples analyzed by BSA (table 4, at the back of this report) were used in the determination of several metrics including Taxa Richness, Jaccard Coefficient of Community Similarity (JCCS; U.S. Environmental Protection Agency, 1989), Hilsenhoff Biotic Index (Hilsenhoff, 1977), and Percent Contribution of the Dominant Taxon. Most

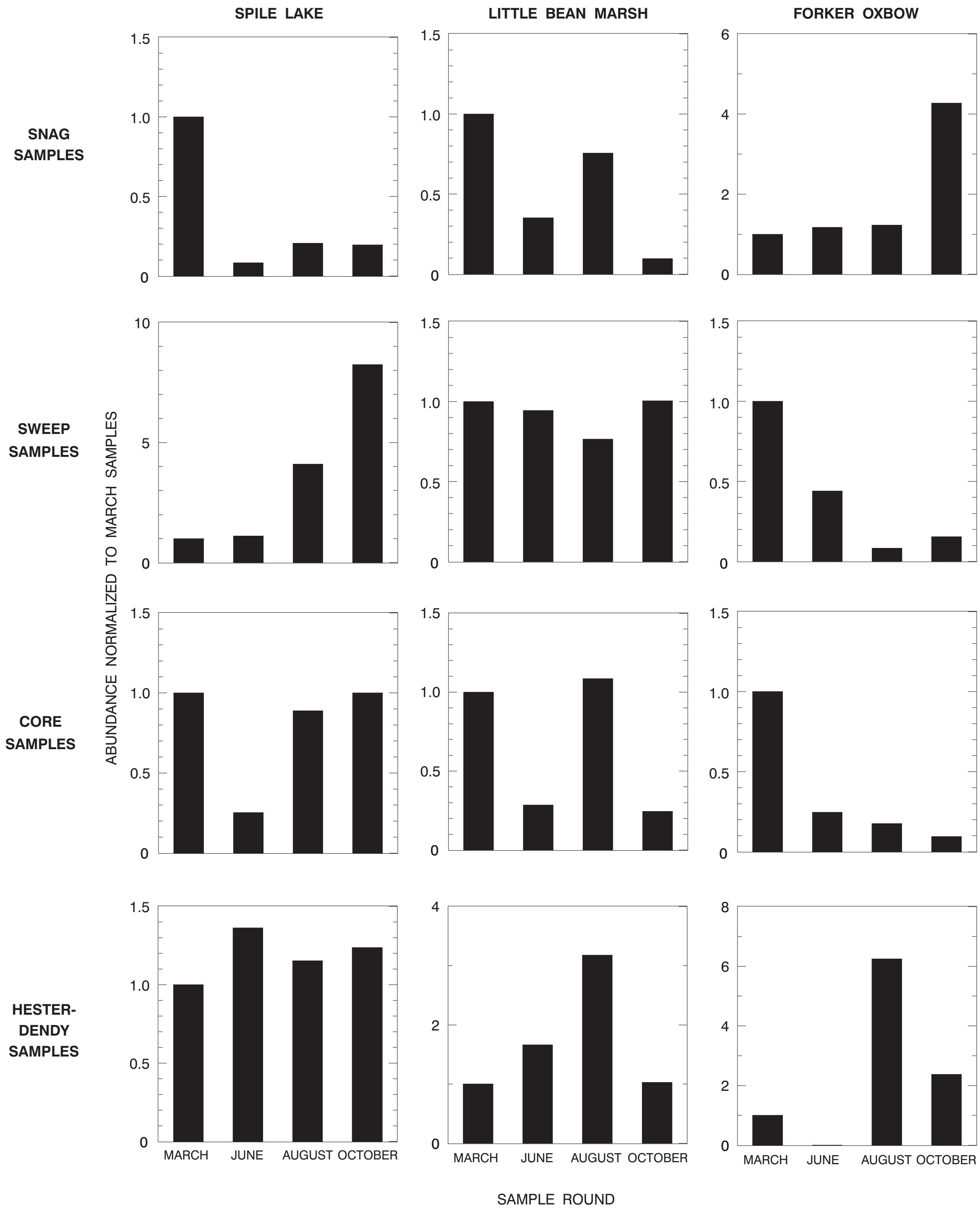


Figure 25. Temporal variation in normalized invertebrate abundance data for Spile Lake, Little Bean Marsh, and Forker Oxbow.

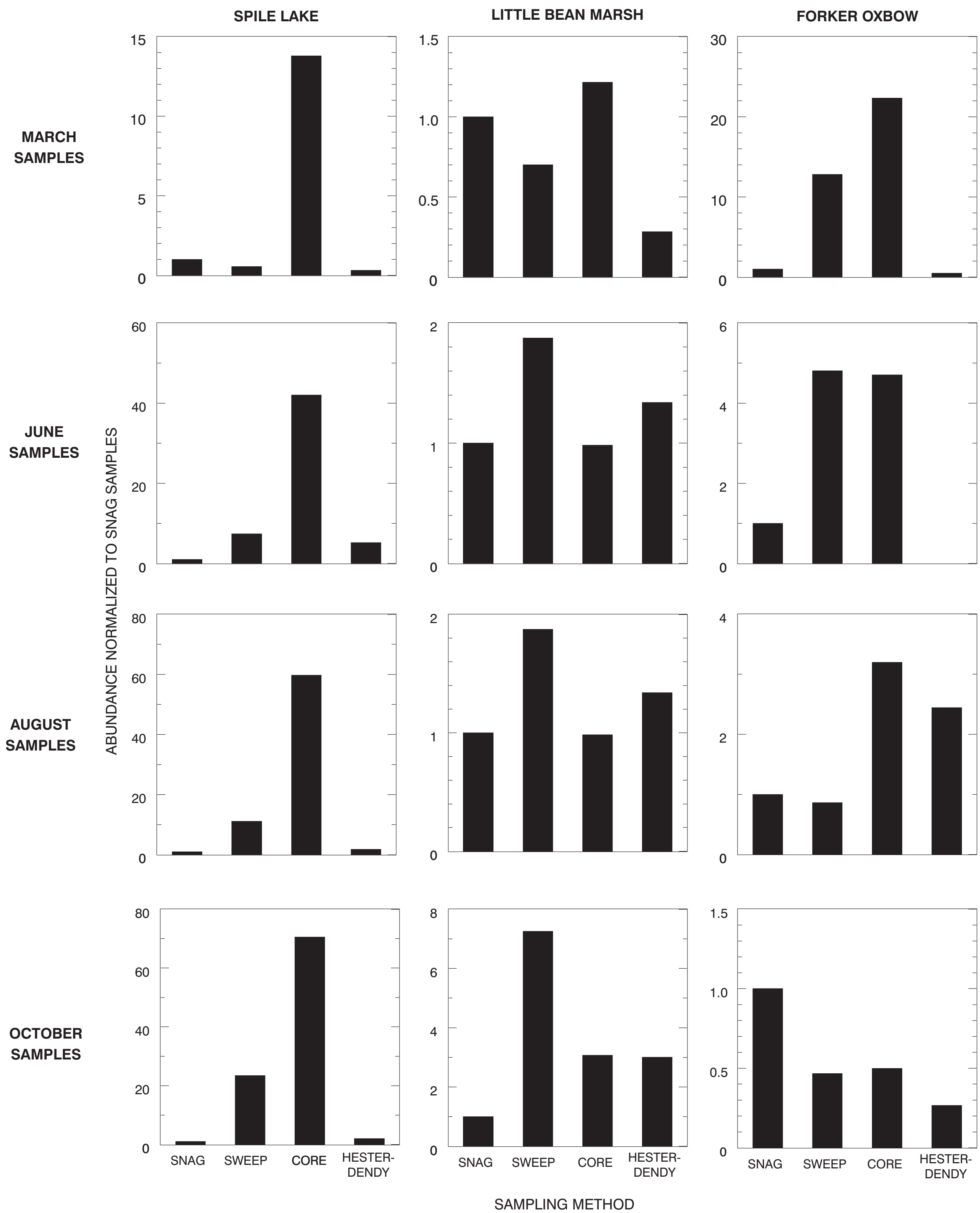


Figure 26. Variability by sampling method of normalized invertebrate abundance data for Spile Lake, Little Bean Marsh, and Forker Oxbow.

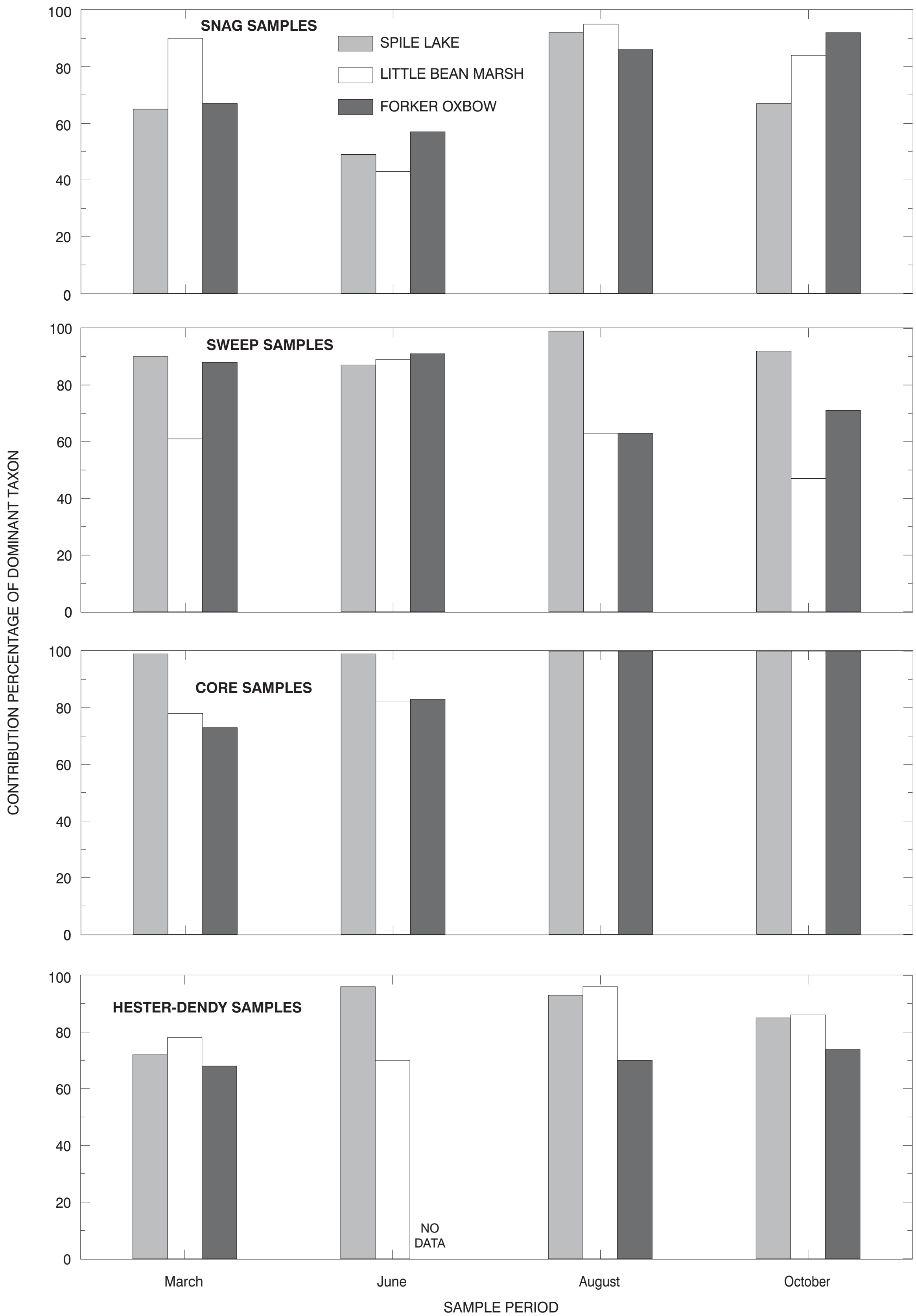


Figure 27. Contribution percentage by three dominant invertebrate taxon by sampling method and sample period.

invertebrate metrics were developed to assess populations in streams and rivers. The metrics above were selected because they are thought to be the most applicable to assess invertebrate communities in wetland systems. The analyses provided in this report compare the three study sites. In strict usage, the invertebrate population data would be compared and contrasted between a reference wetland and another wetland within the same natural division. Results would then be presented as a percent comparability to the reference site (U.S. Environmental Protection Agency, 1989).

Invertebrate diversity results were compared by site, sampling method, and season, and a similarity index was used to compare invertebrate communities at the three wetlands. Taxa Richness was determined using family-level taxa results and was greatest in the June and August snag and sweep samples. The Spile Lake invertebrate community generally had the least number of families (1 to 11) present in both the snag and sweep samples (table 4; fig. 28). Little Bean Marsh had the greatest number of families (6 to 12) present in the snag samples, whereas Forker Oxbow generally had the greatest number of families (6 to 17) present in the sweep samples. The sweep samples generally had greater diversity than the snag samples.

The JCCS (U.S. Environmental Protection Agency, 1989) was used to test the taxonomic similarity between two sites by the presence or absence of taxa using the following formula:

$$Jaccard\ Coefficient = \frac{a}{a+b+c} \qquad (2)$$

where

- a is the number of taxa common to both samples;
- b is the number of taxa present in sample B but not A; and
- c is the number of taxa present in sample A but not B.

Possible results vary from 0 (no similarities) to 1.0 (similar). The JCCS results were determined using Forker Oxbow as an arbitrary reference site, and comparisons were made using snag and sweep sample results from each of the four sample rounds (table 5).

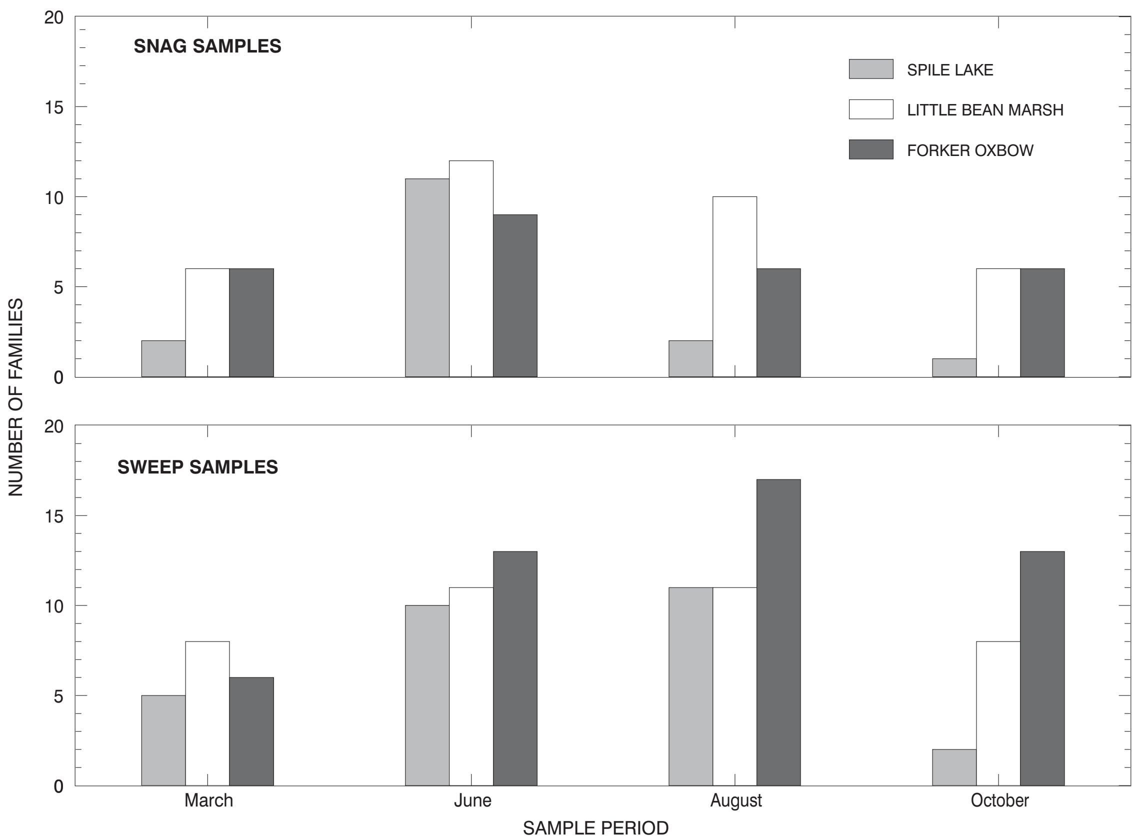


Figure 28. Number of families represented in invertebrate snag and sweep samples by site and sample period.

Table 5. Jaccard Coefficient of Community Similarity results for March, June, August, and October 1996 invertebrate snag and sweep samples

Site comparison	March	June	August	October
	Snag samples			
Spile Lake and Forker Oxbow	0.14	0.25	0.14	0.17
Little Bean Marsh and Forker Oxbow	.33	.31	.33	.20
	Sweep samples			
Spile Lake and Forker Oxbow	0.60	1.0	1.0	0.60
Little Bean Marsh and Forker Oxbow	.38	.54	1.0	.75

The JCCS results for snag samples indicate substantial differences existed between the invertebrate communities in the three wetlands.The JCCS results for invertebrate communities detected in sweep samples, however, appeared similar, except for the March comparison between Little Bean Marsh and Forker Oxbow.

A metric that commonly is used to summarize the overall organic pollution tolerance of a invertebrate community is the Hilsenhoff Biotic Index (Hilsenhoff, 1977). This index uses a classification of invertebrates according to their tolerance of organic wastes. Family-level taxonomic information was used to determine the index for the wetland samples. The sweep samples from June and August were analyzed by this method (table 6). The three wetlands studied were classified from fair to fairly poor in their organic pollution tolerance in the June samples, and from good to poor in the August samples. These classifications are reasonable as wetlands are organically-rich systems compared to most streams, and each of these wetlands are subject to potential pesticide inputs from agricultural runoff and flooding.

The Percent Contribution of the Dominant Taxon is a metric used to provide an indication of invertebrate community balance. A invertebrate community dominated by relatively few taxa would indicate environmental stress (U.S. Environmental Protection Agency, 1989). The taxa found in the August snag and sweep samples, analyzed by BSA, from each of the three wetlands are presented in figures 29 and 30. Dominance by the Chironomidae family is obvious at all three wetland sites. The Spile Lake invertebrate community in these samples is represented by only two taxa with the major family Chironomidae, an indication of substantial stresses to the wetland during this period. Little Bean Marsh also indicated a substantial dominance from the Chironomidae family, though many other families are represented to some extent. The Chironomidae family also dominated the Forker Oxbow invertebrate community, which had fewer total families than Little Bean Marsh, but a greater total than Spile Lake.

Table 6. Hilsenhoff Biotic Index Classification results for June and August 1996 invertebrate sweep samples

Site	June sample	August sample
Spile Lake	Fair	Fairly poor
Little Bean Marsh	Fairly poor	Poor
Forker Oxbow	Fairly poor	Good

The families of the Chironomidae and Palaemonidae were prominent in all three August sweep samples from the three wetlands (fig 30). The Forker Oxbow results indicate the most balanced taxon distribution. Whereas Little Bean Marsh was dominated by Palaemonidae and Libellulidae families, a relatively large part of the population was represented by additional families. Spile Lake is dominated by the Chironomidae family and, to a lesser extent, the Palaemondidae family, with several other families also represented. The results indicate the Spile Lake population to be the least balanced of the three wetlands, as a result of either natural organic enrichment, as evidenced by the dominance of the Chironomidae, or by some other physical or chemical factor.

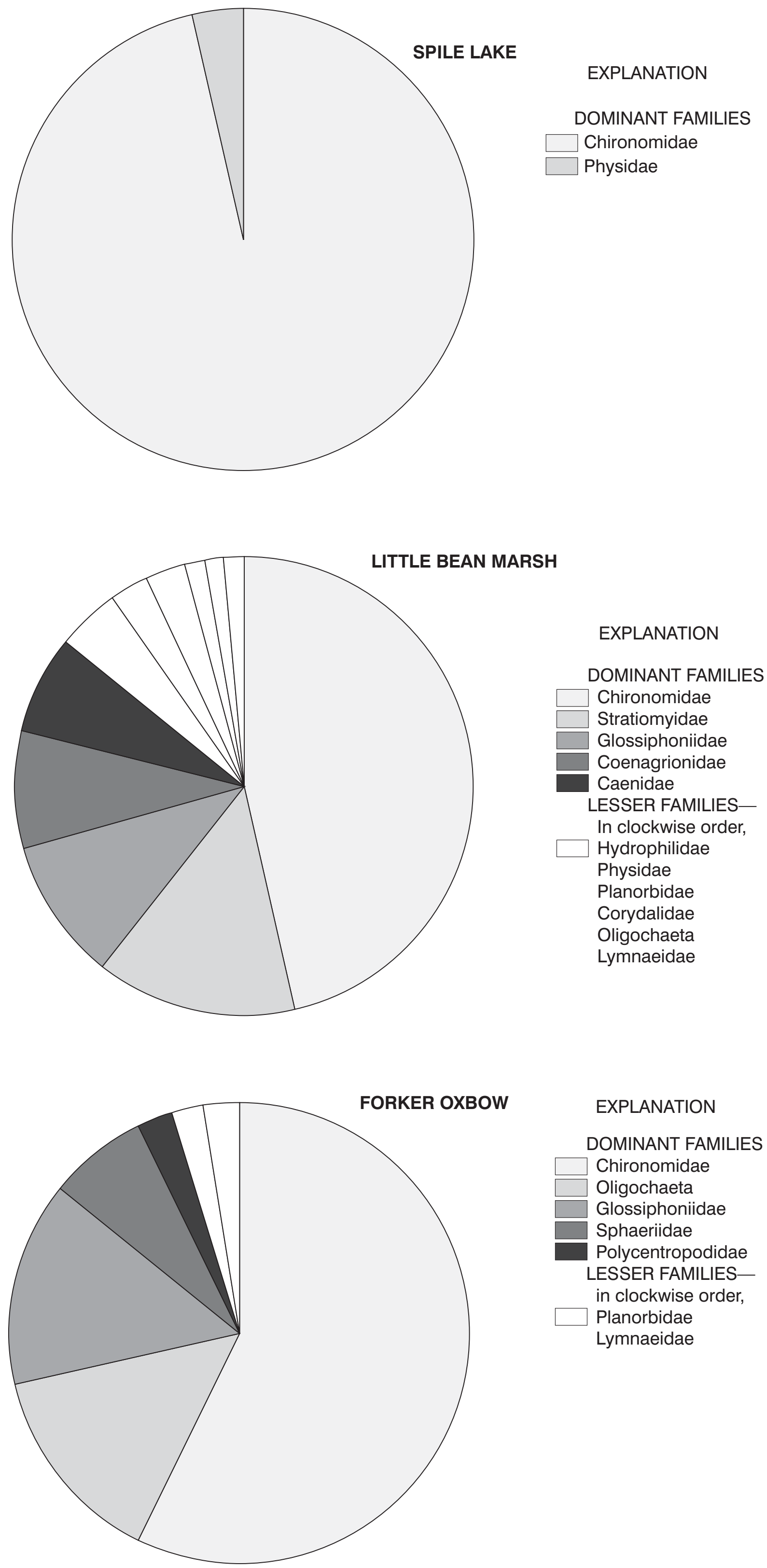


Figure 29. Taxonomic distribution of invertebrate data for August 1996 snag samples.

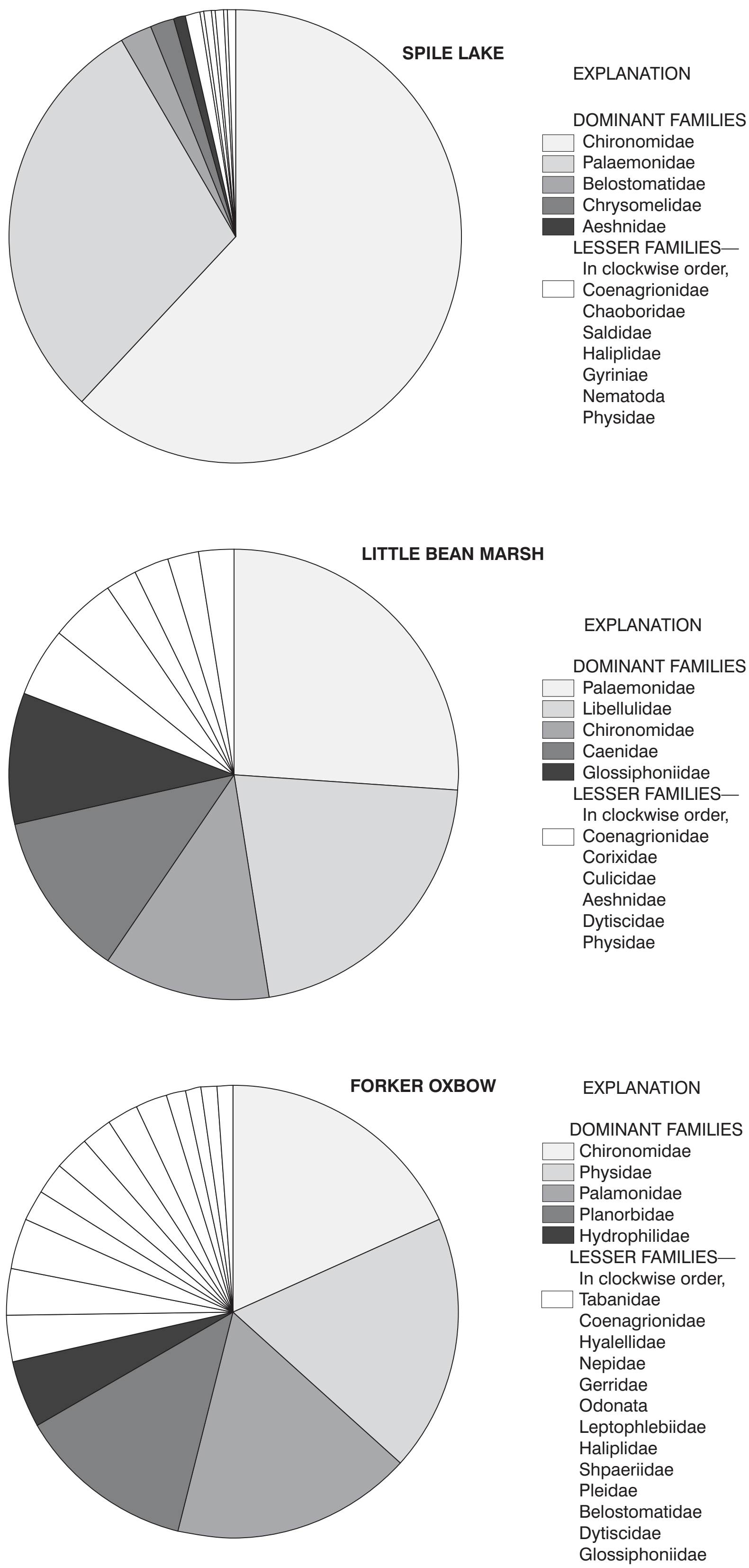


Figure 30. Taxonomic distribution of invertebrate data for August 1996 sweep samples.

SUMMARY

This report summarizes water quality, hydrologic, and invertebrate data collected at three remnant wetlands by the U.S. Geological Survey in cooperation with the Missouri Department of Natural Resources. This study was undertaken to determine basic water-quality characteristics, hydrology, and invertebrate communities of wetlands that could be used for water-quality standards and future study comparisons. Three wetland sites were selected for inclusion in this study: Spile Lake near Horton, in Vernon County; Little Bean Marsh near Bean Lake, in Platte County; and Forker Oxbow near Fountain Grove, in Linn County, Missouri. Data collection took place between December 1995 and May 1997.

The range in stages at Spile Lake, Little Bean Marsh, and Forker Oxbow were 8.9, 4.1, and 12.0 ft. Spile Lake experienced several overbank events from the Little Osage and Marmaton Rivers, and Forker Oxbow was inundated several times by overbank flow from Locust Creek. The levee and gate system regulated inflows and limited the range in stage at Little Bean Marsh during much of the study period. The bathymetry of the wetland systems was used to convert water-surface elevation data into wetted surface area and storage. There was a 770 percent range in wetted area at Spile Lake, 1,400 percent range at Little Bean Marsh, and 3,500 percent range at Forker Oxbow. Wetland volumes were more variable with a 7,600 percent change at Spile Lake, a 6,700 percent change at Little Bean Marsh, and a 30,000 percent change at Forker Oxbow during the study. These values represent conservative estimates as the total inundated area affected by floods was not surveyed at each wetland. Using the maximum maintainable wetland storage and mean annual local runoff, a mean hydraulic residence time was calculated for each wetland. The mean residence time for Spile Lake, Little Bean Marsh, and Forker Oxbow were calculated to be 12, 27, and 45 days.

Water-quality data collected at each site included specific conductance, pH, temperature, dissolved oxygen, turbidity, and major nutrient forms. Physicochemical property data were collected hourly, turbidity data were collected approximately once every two weeks, and nutrient data were collected approximately monthly during the monitoring period.

More than 12,000 hourly specific conductance, pH, temperature, and dissolved-oxygen measurements were logged at each wetland site during the monitoring period. Median specific conductance values in Spile Lake, Little Bean Marsh, and Forker Oxbow were 231, 457, and 236 μS/cm. The extremes in specific conductance values are closely tied with periods of ice cover and rainfall-runoff events.

Median pH values at Spile Lake, Little Bean Marsh, and Forker Oxbow were 7.0, 7.7, and 7.2. Extremes in pH in these wetlands were probably the result of photosynthesis, respiration, and decomposition. The overall distribution in pH values fell within the 6 to 9 unit range commonly found in natural waters.

There was a substantial difference between wetland near-surface and near-bottom water temperature throughout much of the monitoring period resulting in thermal stratification, even in these shallow waters. As in temperate lakes and reservoirs, the thermal stratification was greatest in the summer months, less of a factor in the winter months, and absent in the fall and early spring months. Median near-surface water temperatures were greatest in Spile Lake (13.7 °C), the southernmost site, and least in Forker Oxbow (11.6 °C), the northernmost site. Median near-bottom water temperatures were greatest at Little Bean Marsh (11.8 °C), and least at Forker Oxbow (8.9 °C). These differences probably reflect differences in median air temperatures because of latitudinal differences in the wetland locations. Forker Oxbow had a high maximum near-bottom water temperature and the highest maximum near-surface water temperature of the three wetlands, probably because of its relatively low volume.

Median dissolved-oxygen levels at Spile Lake, Little Bean Marsh, and Forker Oxbow were 5.4, 8.2, and 4.3 mg/L. Extremes in dissolved oxygen in the wetlands were likely tied to biological factors and thermal stratification conditions. Maximum dissolved-oxygen concentrations were measured in late winter or early spring, corresponding to periods of greatest algal production and photosynthesis activity. The lowest measured dissolved oxygen values were measured in the summer months during periods of strong thermal stratification. The cumulative frequency distributions of dissolved-oxygen concentrations indicate that the 5 mg/L dissolved-oxygen standard, set by the State of Missouri for the protection of aquatic life, was not met in these natural wetlands at the sampling locations from 40 to 60 percent of a sampled 1-year period.

The overall median turbidity values at Spile Lake, Little Bean Marsh, and Forker Oxbow were 18, 2.2, and 26 NTU's. Under most typical conditions,

there probably would be sufficient sunlight penetration so that water clarity would not limit algae or submerged macrophyte production.

Median concentration of all dissolved nutrients (NH_3, NO_2, NO_3, P, and PO_4) were at or near detection levels at all wetlands. Despite differences in location and land-use characteristics, the distribution of N and P forms generally was similar in all three systems. No statistically significant differences in dissolved NH_3, NO_2, NO_3, total ammonia plus organic N, and dissolved P concentrations were detected between the three wetlands. Consequently, it is likely that concentrations of these N and P forms in the wetlands are controlled by autochthonous rather than allochthonous processes. Total P and PO_4 concentrations were significantly different in the three wetlands.

Currently (1998), wetlands in Missouri are subject to the same State water-quality standards as adjacent streams and rivers. Comparisons between wetland and stream physicochemical property and nutrient concentration data indicate that a statistically significant difference in the distribution of most of the physicochemical properties and nutrient forms in wetlands exists when compared with streams in the same natural divisions. Thirty one individual comparisons were made, and 21 resulted in a statistically significant difference between wetland and stream physicochemical properties and nutrient forms. Most of these differences resulted from lower values in wetlands relative to streams. The wetland-stream comparisons are similar across the natural divisions. Of the 12 properties and constituents compared, 10 showed consistent results between 2 or more tested natural divisions (pH, temperature, dissolved oxygen, NH_3, NO_2, dissolved P, and dissolved PO_4, specific conductance, turbidity, total ammonia plus organic N), and 2 showed different results between each of the tested natural divisions (NO_2, total P). While a large number of statistically significant differences were detected in the comparisons between wetland physicochemical properties and constituents and those of streams, these differences may not represent practical or chemically significant differences and may not form a logical basis for modifications of existing water-quality standards.

Thirteen flooding or rainfall-runoff events were analyzed from the three wetlands to determine the most typical responses in physicochemical properties from these events. The changes in physicochemical properties of the wetlands, in response to local runoff and floodwaters from adjacent rivers, are determined by a number of factors including antecedent conditions in the wetland, the timing of the event, and the characteristics of the floodwaters. Generally, specific conductance in the wetlands had variable responses to the initial rainfall, typically decreased with runoff inputs, and the ending specific conductance values typically were near antecedent values. pH was quite variable between sites and events but typically followed changes in the hydrograph or dissolved-oxygen concentrations. The typical effects of wetland flooding on water temperature was to temporarily destratify the wetland, if stratification existed prior to the event, with the wetland returning to pre-event conditions as floodwaters receded. Finally, dissolved-oxygen concentrations typically had variable effects from the initial rainfall but tended to follow changes in the hydrograph thereafter.

In addition to wetland responses to hydrologic extremes, short-term temporal variations of physicochemical properties in wetlands as a result of seasonal variations (summer and winter) were examined. Summer conditions in the wetlands were characterized by the presence of thermal stratification, either diurnal or strongly established; dissolved-oxygen concentrations were low but may have had large diurnal fluctuations; and pH values were relatively low and may or may not have exhibited diurnal fluctuations. Winter conditions were characterized by substantial changes in physicochemical properties over the course of a day or two as a result of ice formation and melt. The winter scenario presented demonstrated the important effects ice cover can have on the physicochemical characteristics of Missouri wetlands.

Invertebrate data were collected in March, June, August, and October 1996. Invertebrate abundances varied substantially with time and sampling method in each of the wetlands. Of the three quantitative sampling methods, the core samples generally had the greatest abundances for all sample rounds. While the core samples had the greatest abundances, they also had the lowest diversities. The snag samples seem to have the greatest community balance, followed by the sweep and Hester-Dendy samples. Overall, the snag and sweep samples provided the best representation of both abundance and diversity of the four sample methods.

Using family-level taxonomic results, the Spile Lake invertebrate community had the fewest number of families (from 1 to 11) present in both snag and sweep samples. Little Bean Marsh had the greatest number of families (6 to 12) present in the snag samples, while Forker Oxbow had the greatest number of families (from 6 to 17) present in the sweep samples. The sweep samples generally had greater diversity than the snag samples. The invertebrate familial diversities were greatest during June and August for both snag and sweep samples.

Analyses of the invertebrate data using the Jaccard Coefficient of Community Similarity indicate mixed results, with analyses of snag sample results suggesting substantial differences exist between the three wetlands and sweep samples indicating substantial similarities between the three wetlands. Most of the families detected at these sites are considered organic pollution tolerant as indicated by the Hilsenhoff Biotic Index. Analysis of the dominant taxon indicate that one or two families that are tolerant to organic enrichment generally dominate the wetlands.

Wetlands constitute a wide spectrum of vegetation, soil, and hydrologic conditions, and it is difficult to define the water quality, hydrology, and invertebrate communities for all of the possible systems. The data from this study are intended to provide an indication of these characteristics for three wetlands that are representative of remnant riparian wetland systems in Missouri. Other non-represented types of wetland systems in the State (fens, swamps, wet prairie) may have differing water-quality characteristics than those presented in this report. The hydrology, water quality, and invertebrate information analyzed in this study indicate that although there are similarities between wetlands, these are unique systems. The statistical comparisons between water-quality constituents in wetlands and streams indicate dissimilarities are common. Thermal stratification in wetlands may be similar to that in lakes and reservoirs, which could be relevant in the application of water-quality standards. Additional studies of longer duration that include various wetland types and management scenarios would be useful to better define the range in physicochemical properties and constituents, the ground-water surface-water interactions, and invertebrate community characteristics in these unique ecosystems.

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TABLES

Table 4. Taxonomic summary of invertebrate snag and sweep samples from three wetlands for March, June, August, and October 1996 sampling periods

Date	Site	Phylum	Class	Order	Family	Genus
			Snag samples			
03–21–96	Spile Lake	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Copepoda			
		Nematoda				
		Arthropoda	Chelicerata	Acari	Acariformes	
		Annelida	Oligochaeta			
03–25–96	Little Bean Marsh	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Odonata	Coenagrionidae	Nehalennia
		Arthropoda	Insecta	Coleoptera	Haliplidae	Peltodytes
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Berosus
		Annelida	Oligochaeta			
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
03–27–96	Forker Oxbow	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Malacostraca	Decapoda	Palaemonidae	Macrobrachium
		Arthropoda	Malacostraca	Decapoda	Anostroca	Eubranchipus
		Arthropoda	Branchiopoda	Cladocera		
		Arthropoda	Copepoda			
		Mollusca	Bivalva		Sphaeriidae	
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
06–07–96	Spile Lake	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Diptera	Tabanidae	Tabanus
		Arthropoda	Insecta	Diptera	Ceratopogonida	Atrichopogon
		Arthropoda	Insecta	Megaloptera	Corydalidae	Chauliodes
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Berosus
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Tropisternus
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Hydrophilus
		Arthropoda	Insecta	Coleoptera	Dytiscidae	Coptotomus
		Arthropoda	Insecta	Coleoptera	Ptylodactlidae	Anchytarsus
		Arthropoda	Malacostraca	Isopoda	Asellidae	Caecidotea
		Arthropoda	Crustacea	Amphipoda	Hyalellidae	Hyalella
		Arthropoda	Malacostraca	Decapoda	Cambaridae	
		Annelida	Hirudinea		Glossiphoniidae	
		Annelida	Oligochaeta			
06–06–96	Little Bean Marsh	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Diptera	Ceratopogonida	Serromyia
		Arthropoda	Insecta	Diptera	Tabanidae	Chrysops
		Arthropoda	Insecta	Diptera	Stratiomyidae	
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Berosus
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Tropisternus
		Arthropoda	Insecta	Coleoptera	Dytiscidae	Coptotomus
		Arthropoda	Insecta	Coleoptera	Haliplidae	Peltodytes
		Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenis
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella

Table 4. Taxonomic summary of invertebrate snag and sweep samples from three wetlands for March, June, August, and October 1996 sampling periods-Continued

Date	Site	Phylum	Class	Order	Family	Genus
Snag samples—Continued						
06–06–96	Little Bean Marsh	Mollusca	Gastropoda		Lymnaeidae	Pseudosuccinea
		Annelida	Oligochaeta			
		Annelida	Hirudinea		Glossiphoniidae	
06–12–96	Forker Oxbow	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Neuroptera	Sisyridae	Sisyra
		Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenis
		Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Stenonema
		Arthropoda	Malacostraca	Isopoda	Asellidae	Caecidotea
		Arthropoda	Malacostraca	Decapoda	Cambaridae	
		Annelida	Oligochaeta			
		Annelida	Hirudinea		Glossiphoniidae	
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
08–27–96	Spile Lake	Arthropoda	Insecta	Diptera	Chironomidae	
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
08–30–96	Little Bean Marsh	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Diptera	Stratiomyidae	
		Arthropoda	Insecta	Megaloptera	Corydalidae	Chauliodes
		Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenis
		Arthropoda	Insecta	Odonata	Coenagrionidae	Nehalennia
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Berosus
		Annelida	Oligochaeta			
		Annelida	Hirudinea		Glossiphoniidae	
		Mollusca	Gastropoda		Lymnaeidae	Pseudosuccinea
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
08–28–96	Forker Oxbow	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Trichoptera	Polycentropodidae	
		Annelida	Oligochaeta			
		Annelida	Hirudinea		Glossiphoniidae	
		Mollusca	Bivalva		Sphaeriidae	
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
		Mollusca	Gastropoda		Lymnaeidae	Pseudosuccinea
10–21–96	Spile Lake	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Copepoda			
10–24–96	Little Bean Marsh	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenis
		Arthropoda	Insecta	Hemiptera	Belostomatidae	Belostoma
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Tropisternus
		Arthropoda	Insecta	Odonata	Coenagrionidae	Nehalennia
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
		Annelida	Oligochaeta			

Table 4. Taxonomic summary of invertebrate snag and sweep samples from three wetlands for March, June, August, and October 1996 sampling periods-Continued

Date	Site	Phylum	Class	Order	Family	Genus
			Snag samples—Continued			
10–23–96	Forker Oxbow	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Odonata	Coenagrionidae	Argia
		Arthropoda	Crustacea	Amphipoda	Hyalellidae	Hyalella
		Annelida	Hirudinea		Glossiphoniidae	
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
		Mollusca	Bivalva		Sphaeriidae	
			Sweep samples			
03–21–96	Spile Lake	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Hemiptera	Corixidae	Sigara
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Berosus
		Arthropoda	Copepoda			
		Arthropoda	Branchiopoda	Cladocera		
		Annelida	Oligochaeta			
		Arthropoda	Ostrocoda			
		Arthropoda	Malacostraca	Decapoda	Palaemonidae	Macrobrachium
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
03–25–96	Little Bean Marsh	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Diptera	Ceratopogonidae	Serromyia
		Arthropoda	Insecta	Hemiptera	Corixidae	Sigara
		Arthropoda	Insecta	Hemiptera	Corixidae	Hesperocorixa
		Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenis
		Arthropoda	Insecta	Coleoptera	Haliplidae	Peltodytes
		Arthropoda	Malacostraca	Decapoda	Palaemonidae	Macrobrachium
		Annelida	Hirudinea		Glossiphoniidae	
		Annelida			Oligoneuridae	
		Arthropoda	Copepoda			
03–27–96	Forker Oxbow	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Hemiptera	Corixidae	Sigara
		Arthropoda	Insecta	Hemiptera	Corixidae	Hesperocorixa
		Arthropoda	Insecta	Coleoptera	Haliplidae	Peltodytes
		Arthropoda	Malacostraca	Decapoda	Anostroca	Eubranchipus
		Arthropoda	Copepoda			
		Arthropoda	Branchiopoda	Cladocera		
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
		Mollusca (remains)				
06–07–96	Spile Lake	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Diptera (remains)		
		Arthropoda	Insecta	Coleoptera	Chrysomelidae	Donacia
		Arthropoda	Insecta	Hemiptera	Corixidae	Sigara
		Arthropoda	Insecta	Coleoptera	Dytiscidae	Coptotomus
		Arthropoda	Insecta	Hemiptera	Saldidae	
		Arthropoda	Insecta	Odonata	Coenagrionidae	

Table 4. Taxonomic summary of invertebrate snag and sweep samples from three wetlands for March, June, August, and October 1996 sampling periods-Continued

Date	Site	Phylum	Class	Order	Family	Genus
		Sweep samples—Continued				
06–07–96	Spile Lake	Arthropoda	Insecta	Coleoptera	Hydrophilidae	Tropisternus
		Arthropoda	Malacostraca	Decapoda	Palaemonidae	Macrobrachium
		Arthropoda	Malacostraca	Amphipoda	Hyalellidae	Hyalella
		Annelida	Oligochaeta			
		Arthropoda	Crustacea			
		Arthropoda	Copepoda			
		Arthropoda	Malacostraca	Decapoda	Cambaridae	
06–13–96	Little Bean Marsh	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Diptera	Stratiomyidae	Stratiomys
		Arthropoda	Insecta	Diptera	Chaoboridae	Chaoborus
		Arthropoda	Insecta	Hemiptera	Corixidae	Sigara
		Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenis
		Arthropoda	Insecta	Coleoptera	Dytiscidae	Coptotomus
		Arthropoda	Insecta	Ephemeroptera		
		Arthropoda	Insecta	Coleoptera	Haliplidae	Peltodytes
		Annelida	Oligochaeta			
		Annelida	Hirudinea		Glossiphoniidae	
		Arthropoda	Malacostraca	Decapoda	Cambaridae	
		Arthropoda	Copepoda			
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
		Mollusca	Gastropoda			
06–12–96	Forker Oxbow	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Diptera	Chaoboridae	Chaoborus
		Arthropoda	Insecta	Hemiptera	Corixidae	Sigara
		Arthropoda	Insecta	Hemiptera	Corixidae	
		Arthropoda	Insecta	Hemiptera	Pleidae	Neoplea
		Arthropoda	Insecta	Coleoptera	Haliplidae	Peltodytes
		Arthropoda	Insecta	Coleoptera	Dytiscidae	Hydroporus
		Arthropoda	Insecta	Coleoptera	Dytiscidae	Coptotomus
		Arthropoda	Insecta	Ephemeroptera	Ephemeridae	Hexagenia
		Arthropoda	Insecta	Odonata	Lestidae	Lestes
		Arthropoda	Malacostraca	Isopoda	Asellidae	Caecidotea
		Arthropoda	Copepoda			
		Arthropoda	Branchiopoda	Cladocera		
		Arthropoda	Chelicerata	Acari	Acariformes	
		Mollusca	Bivalva		Sphaeriidae	Musculium
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
08–27–96	Spile Lake	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Diptera	Chaoboridae	Chaoborus
		Arthropoda	Insecta	Odonata	Aeshnidae	Boyeria
		Arthropoda	Insecta	Odonata	Coenagrionidae	
		Arthropoda	Insecta	Hemiptera	Saldidae	
		Arthropoda	Insecta	Hemiptera	Belostomatidae	Belostoma
		Arthropoda	Insecta	Coleoptera	Haliplidae	Peltodytes

Table 4. Taxonomic summary of invertebrate snag and sweep samples from three wetlands for March, June, August, and October 1996 sampling periods-Continued

Date	Site	Phylum	Class	Order	Family	Genus
			Sweep samples—Continued			
08–27–96	Spile Lake	Arthropoda	Insecta	Coleoptera	Gyrinidae	Gyrinus
		Arthropoda	Insecta	Coleoptera	Chrysomelidae	Donacia
		Arthropoda	Malacostraca	Decapoda	Palaemonidae	Macrobrachium
		Nematoda				
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
08–29–96	Little Bean Marsh	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Diptera	Culicidae	
		Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenis
		Arthropoda	Insecta	Odonata	Aeshnidae	Aeshna
		Arthropoda	Insecta	Odonata	Libellulidae	Tramea
		Arthropoda	Insecta	Odonata	Coenagrionidae	
		Arthropoda	Insecta	Hemiptera	Corixidae	Sigara
		Arthropoda	Insecta	Coleoptera	Dytiscidae	Laccophilus
		Annelida	Hirudinea		Glossiphoniidae	
		Arthropoda	Malacostraca	Decapoda	Palaemonidae	Macrobrachium
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
08–28–96	Forker Oxbow	Arthropoda	Insecta	Diptera	Chironomidae	
		Arthropoda	Insecta	Diptera	Tabanidae	Chrysops
		Arthropoda	Insecta	Hemiptera	Nepidae	Ranatra
		Arthropoda	Insecta	Hemiptera	Gerridae	Trepobates
		Arthropoda	Insecta	Hemiptera	Pleidae	Neoplea
		Arthropoda	Insecta	Hemiptera	Belostomatidae	Belostoma
		Arthropoda	Insecta	Odonata		
		Arthropoda	Insecta	Odonata	Coenagrionidae	Nehalennia
		Arthropoda	Insecta	Ephemeroptera	Leptophlebiidae	
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Berosus
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Hydrochus
		Arthropoda	Insecta	Coleoptera	Haliplidae	Peltodytes
		Arthropoda	Insecta	Coleoptera	Dytiscidae	
		Arthropoda	Malacostraca	Decapoda	Palaemonidae	Macrobrachium
		Arthropoda	Malacostraca	Amphipoda	Hyalellidae	Hyalella
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella
		Mollusca	Bivalva		Sphaeriidae	Musculium
		Annelida	Hirudinea		Glossiphoniidae	
10–23–96	Spile Lake	Arthropoda	Insecta	Coleoptera	Chrysomelidae	Donacia
		Arthropoda	Malacostraca	Decapoda	Palaemonidae	Macrobrachium
10–24–96	Little Bean Marsh	Arthropoda	Insecta	Diptera		
		Arthropoda	Insecta	Hemiptera	Corixidae	Sigara
		Arthropoda	Insecta	Hemiptera	Notonectidae	Notonecta
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Tropisternus
		Arthropoda	Insecta	Odonata	Aeshnidae	Aeshna
		Arthropoda	Malacostraca	Decapoda	Palaemonidae	Macrobrachium
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella

Table 4. Taxonomic summary of invertebrate snag and sweep samples from three wetlands for March, June, August, and October 1996 sampling periods-Continued

Date	Site	Phylum	Class	Order	Family	Genus
Sweep samples—Continued						
10–24–96	Little Bean Marsh	Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
		Mollusca	Gastropoda		Lymnaeidae	Pseudosuccinea
10–23–96	Forker Oxbow	Arthropoda	Insecta	Diptera	Tabanidae	Chrysops
		Arthropoda	Insecta	Odonata	Libellulidae	
		Arthropoda	Insecta	Odonata	Aeshnidae	Boyeria
		Arthropoda	Insecta	Odonata	Coenagrionidae	Nehalennia
		Arthropoda	Insecta	Coleoptera	Gyrinidae	Dineutus
		Arthropoda	Insecta	Hemiptera	Notonectidae	Notonecta
		Arthropoda	Insecta	Hemiptera	Corixidae	Sigara
		Arthropoda	Insecta	Coleoptera	Hydrophilidae	Berosus
		Arthropoda	Malacostraca	Amphipoda	Hyalellidae	Hyalella
		Arthropoda	Malacostraca	Decapoda	Palaemonidae	Macrobrachium
		Mollusca	Bivalva		Sphaeriidae	
		Mollusca	Gastropoda	Basommatophora	Planorbidae	Planorbella
		Mollusca	Gastropoda	Basommatophora	Physidae	Physella

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