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Ecological Dynamics of Wetlands at Lisbon Bottom, Big Muddy National Fish and Wildlife Refuge, Missouri

Final Report to
U.S. Fish and Wildlife Service
Big Muddy National Fish and Wildlife Refuge

December 2002
Revised December 2003

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

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Conversion Factors and Vertical Datum

Multiply	By	To obtain
	Length	
foot (ft)	3.048×10^{-1}	meters
mile (mi)	1.609×10^{0}	kilometers
	Area	
acre (ac)	4.047×10^{3}	square meters
	4.047×10^{-3}	square kilometers
	4.047×10^{-1}	hectares
square mile (mi^2)	2.59×10^{0}	square kilometers
	Flow	
cubic feet per second (cfs, ft^3/s)	2.832×10^{-2}	cubic meters per second

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 for the first-order level nets of both the United States and Canada, formerly called Sea Level Datum of 1929.

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Executive Summary

Intensive physical modification of the Missouri River for navigation, flood control, and power generation has dramatically changed the river corridor. Dams, revetments, channelization, and levee construction activities have constrained the river into a single fast, deep channel and disconnected the river from the flood plain. These dramatic hydrological alterations have reduced the available habitat for numerous bird, fish, invertebrate, and plant species. Several native species have declined, leading to their federal listing under the Endangered Species Act. It is widely recognized that recovery of endangered species is primarily dependent on physical habitat rehabilitation. Land acquisition for ecological rehabilitation in the Lower Missouri River has historically been impeded due to the lack of land available from willing sellers and lack of funds and authority. However, this situation changed in 1993 following a severe flood. Flooding broke many levees and scoured numerous deep holes in the flood plain; in addition, thick layers of sand were deposited on numerous crop fields. Subsequently, much flood-altered farmland was willingly offered for sale by private landowners. Much of this land was then purchased by governmental agencies for environmental rehabilitation. Lisbon Bottom is one of several parcels of flood-damaged land that was purchased from willing sellers by the U.S. Fish and Wildlife Service as part of the Big Muddy National Fish and Wildlife Refuge. Lisbon Bottom is a loop bend in the river near Glasgow, Missouri. Flooding at Lisbon Bottom in 1993 and 1995 breeched local levees and created a diverse wetland complex. Lisbon Bottom is managed in a passive manner. The levees have not been rebuilt, and floods and vegetative succession continue to alter the landscape. A side-channel chute was formed by further flooding in 1996 and 1997. The diversity of wetland types and the continued connection to the river make Lisbon Bottom an excellent natural laboratory for the study of flood-plain wetland processes and the use of flood-plain wetlands by biota.

This study had 3 objectives: 1) quantify spatial and temporal distribution of biota in aquatic habitats of Lisbon Bottom in relation to changes in hydrological variables that are associated with the spring flooding regime, 2) document biological responses as they are related to habitat dynamics, and 3) analyze and interpret these results to provide managers with information necessary to develop management strategies for Lisbon

Bottom and other tracts of the Big Muddy National Fish and Wildlife Refuge. To accomplish these objectives we conducted a study of the hydrology, limnology, and biological dynamics of zooplankton, macroinvertebrate, fish, and waterbird communities during Spring 1999.

The hydrology of Lisbon Bottom was influenced by overbank flooding from the river and chute, groundwater, rainfall, and valley-wall tributaries. Lisbon wetland types include deep scours (formed during the 1993 and 1995 floods), shallow temporary wetlands with minimal direct surface drainage area, and shallow temporary wetlands with direct surface-water connections to valley-wall tributaries.

Wetlands along the valley wall, far from the main channel, are recharged by main-channel flow only when flow is well over bank. These wetlands are recharged more frequently by local rainfall and by flow from valley-wall tributaries. Hydrologic variation in wetlands that are recharged by local rainfall is of greater magnitude and much more frequent than variation in deep scours. Deep scours, in contrast, had more stable water levels.

The deep scours at the top and bottom margins of Lisbon Bottom are similar in some respects, but differ in others. These wetlands are thermally stratified during the summer and have anoxic hypolimnia. They are devoid of aquatic macrophytes and are strongly influenced by river flooding. Differences are due to the periodicity and energy of flood events. Because the down-valley gradient of Lisbon Bottom is greater than the channel slope, backflooding from the river connects the river to downstream scours more often and for longer duration than the upstream scours. Topflooding events, while less frequent, occur with more energy, and temporarily destratify the upstream scours.

Nutrient availability was greater in river-influenced wetlands than in the valley-wall wetlands due to nutrient-rich conditions of the river. Overbank floods provide pulses of nutrient inputs to the wetlands, which are followed by increases in algal production and subsequently by increases in zooplankton production. In contrast, the valley-wall tributaries deliver lower levels of nutrients and suspended sediment to flood-plain wetlands which promotes clear, shallow conditions that favor growth of emergent and submergent aquatic macrophytes. Lower nutrient inputs and removal of nutrients by macrophytes in valley-wall wetlands result in lower phytoplankton and zooplankton density. Macroinvertebrate density, on the other hand, is strongly correlated to the presence of inundated vegetation, and thus the stream-influenced valley-wall wetlands are highest in macroinvertebrate density and biomass.

Twenty-seven species of crustacean zooplankton, mostly cladocerans and herbivorous copepods, were captured in Lisbon Bottom wetlands. Predacious copepods were not common in the wetlands. Zooplankton density and diversity were related to flood events and nutrient pulses resulting from flood events. Topflooding wetlands had higher densities and diversities of zooplankton than backflooding wetlands, due to greater phytoplankton availability and possibly due to greater predation by fishes in the backflooding wetlands. Crustacean zooplankton density was much lower in stream-influenced wetlands than in the river-influenced wetlands, probably owing to lower nutrient availability and thus lower phytoplankton production in those wetlands.

We captured 167 species of macroinvertebrates in the Lisbon wetlands; 128 of these species are unique to the flood-plain wetlands and are not found in the mainstem. Thus, most of the invertebrates found in these wetlands are not likely to have been colonized from the river during flooding. Temporary wetlands that held

water throughout the winter months due to the fall 1998 flood were likely in part responsible for the species assemblage found; the invertebrate community was dominated by overwintering species and groups of pioneer taxa that were available for dispersal to other basins after flooding occurred in mid-April, 1999. Macroinvertebrate species richness and density were highest in shallow, seasonal, vegetated wetlands and lowest in deep scour habitats. Although scours had lowest species richness throughout the early part of the study, they increased by late spring and summer periods. In all but the deep scours, the ratio of predator / herbivore-detritivores gradually declined during the study period, and the ratio of benthic / pelagic invertebrates peaked during the post-flood period. Both of these indicators appear to correspond with changes in the availability of organic matter over time due to flooding.

Forty species of fish were captured in Lisbon Bottom wetlands. Flood-plain spawning riverine fishes such as buffalo, gizzard shad, gar, and common carp used Lisbon Bottom for spawning during flood events; therefore fishes of the river clearly take advantage of this newly connected portion of the flood plain. Both topflooding and backflooding portions of the flood plain were used for spawning by riverine fishes, but topflooded areas were connected to the river less often. Temporary, topflooding, and stream-influenced wetlands of the Lisbon Bottom flood plain that held many young-of-the-year buffalo dried without being reconnected to the river; thus these fish became trapped and died. Backflooding areas were connected to the river more often and were not observed to trap large numbers of fish. Relative abundance and species composition of fishes using the flood plain were very different from the fish communities associated with the river and chute. Relative abundances of fishes also varied between wetlands, at least in part due to the wetland morphology, water source, and connectivity to the river. For example, deep scours were dominated by centrarchids, and backflooding wetlands had greater relative abundances of riverine fishes such as emerald shiner and shortnose gar. Young-of-the-year buffalo dominated the most ephemeral wetlands while red shiners and other small cyprinids and centrarchids were more common in deeper, less ephemeral temporary and seasonal wetlands. Crappie were common and large in the permanent scours. Crappie growth rates in the scours were high, despite narrow oxic epilimnia and temperatures that were higher than those known to provide good crappie survival and growth.

Thirty-one species of waterbirds were observed at Lisbon Bottom. Chronology of the presence of individual species was related to the spring migration periods of the species rather than the period of flooding, which occurred after the main migration of several species of waterbirds. Most waterbirds were observed on the river and chute prior to the flood, which occurred on April 16, 1999. After the flood, ducks, especially blue-winged teal, were mostly found on the vegetated, stream-influenced wetlands near the valley wall or on an intermittently connected exit scour. Total number of waterbirds, total numbers of ducks, and numbers of wood ducks were significantly greater in the valley-wall wetlands compared to other basin areas during the flood and post-flood period. Waterbirds that used the interior flood-plain wetlands tended to stay longer than those using the chute or river habitats. The river and chute habitats were used primarily for short migratory stopovers as opposed to longer, sustained use for feeding such as that observed for the valley-wall tributary wetlands.

Collectively, these results indicated that wetlands fed by valley-wall tributaries appeared to be different from scour and seasonal wetlands in other locations at Lisbon Bottom. Valley-wall wetlands were hydrologically fed by streams and thus had more sustained periods of inundation compared to interior wetlands

that lack direct stream connections. These factors were important in providing the basis for emergent and submergent macrophyte growth, which was in turn important for invertebrate production and associated waterbird use. Thus, valley-wall wetlands might be important habitats sought by land managers in future land acquisition and management efforts. However, further studies may be necessary to fully understand these unique habitats. Fish communities that used the flood plain during flooding differed from communities documented in the chute and river habitats. Similarly, the invertebrate community of the flood plain contains a unique community composition compared to the river and chute habitats.

This study documented the interaction between hydrology and the biological dynamics within a single spring season at Lisbon Bottom. The wetlands of Lisbon Bottom are continuing to change due to ongoing scouring and sedimentation from floods. The vegetative community is beginning to mature and be dominated by woody species. It is anticipated that as the flood-plain forest matures it will alter hydraulic roughness and the subsequent distribution of flow velocities and sedimentation. Beavers both create wetlands by damming, and drain wetlands by constructing channels. The deep scours currently present are a result of a severe flood in which very high flows overtopped or broke agricultural levees; such wetlands would have been rare or nonexistent prior to channelization of the river. Without extremely high river flows and associated rescouring, sedimentation will continue in the deep scours and they will eventually cease to exist. Continued study of the hydrology, vegetation, and animal community dynamics of Lisbon Bottom could provide important information concerning the formation and ecological trajectory of newly created flood-plain habitats on the Lower Missouri River. Wetlands in the Lower Missouri River flood plain are highly dynamic and currently do not fit well within traditional wetland classification systems. Additional knowledge concerning wetland habitat classification approaches is needed for modified riverine systems. This information is important for development of an adaptive management framework for the Big Muddy National Fish and Wildlife Refuge.

Keywords: Lower Missouri River, hydrology, flood plain, fish, invertebrates, zooplankton, birds, spring rise, management, aquatic

Introduction

Before the late 1800's, a shifting, braided channel with abundant unvegetated sandbars and extensive flood- plain connectivity characterized the Missouri River. The shifting channel provided a wide variety of hydraulic environments, including a substantial amount of shallow, slow water. Historically, this natural fluvial regime resulted in a large quantity of connected and non-connected off-channel water bodies. Within the last century and continuing to the present, the channel of the Lower Missouri River (downstream from Sioux City, Iowa) has been trained into a fast, deep, single-thread channel (fig. 1A). Intensive physical modification of the Missouri River for navigation, flood control, and power generation has resulted in dramatic changes to the river corridor. At present, artificially created habitats such as wing dikes, revetments, and levees concentrate the flow to help maintain a deep navigation channel, while disconnecting it from the flood plain. In addition, reservoir regulation of the Missouri River has substantially changed the annual hydrograph (fig. 1B), sediment loads, temperature regime, and nutrient budgets. Collectively, these physical changes have drastically altered the hydrological and ecological function of the Missouri River system. Today, the river largely consists of high velocity habitats with deep water, and low-flow backwaters that are lacking in shallow water depths and connectivity. Shallow sandbars and flooded wetland habitats that are frequently connected are now rare.

Flooding in the 1990's also caused dramatic changes in flood-plain habitats of the Lower Missouri River. These changes have inhibited continued agricultural use in many areas, making lands available for

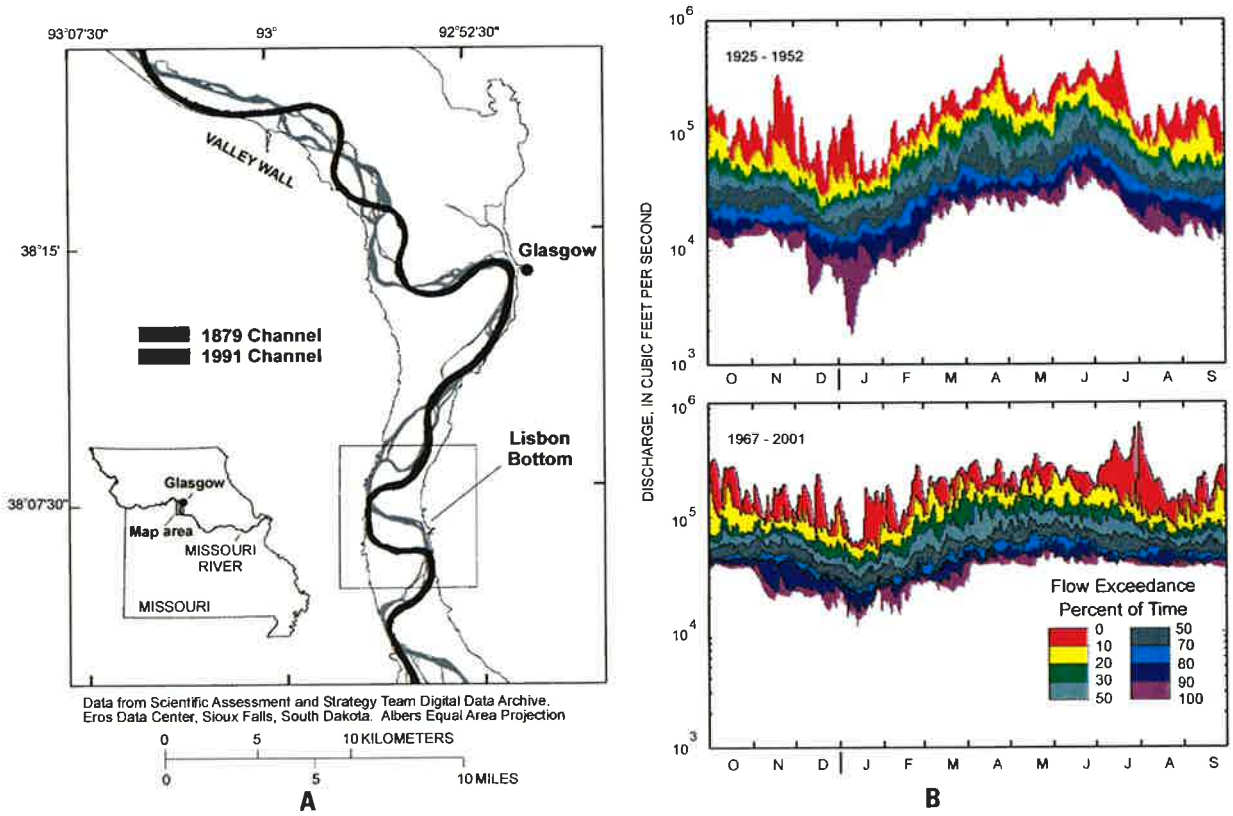

Figure 1. Typical changes in Lower Missouri River. **A.** Channel changes in Grand-Osage sediment, 1879-1991. **B.** Hydrologic changes at Boonville, Missouri pre- and post-regulation

purchase by government agencies. These lands are serving as rehabilitation areas where existing levee breaks have allowed flood-plain landscapes to be directly or indirectly shaped by natural riverine processes such as flood-pulses, scouring, sedimentation, and changes in vegetation. To date, more than 28,000 ha of flood-plain lands in the Lower Missouri corridor have been purchased by state and federal agencies, and some areas are being allowed to revert back toward a more natural or historical state. One parcel included in the land acquired by the U.S. Fish and Wildlife Service (USFWS) Big Muddy National Fish and Wildlife Refuge is Lisbon Bottom, which is approximately 875 ha in size, and is situated at a large bend in the Lower Missouri River about 5 km south of Glasgow, Missouri (river mile[1] 213-218, fig. 2). Here, additional flooding since 1993 has created a new chute and a hydrologically diverse complex of permanent and ephemeral wetlands.

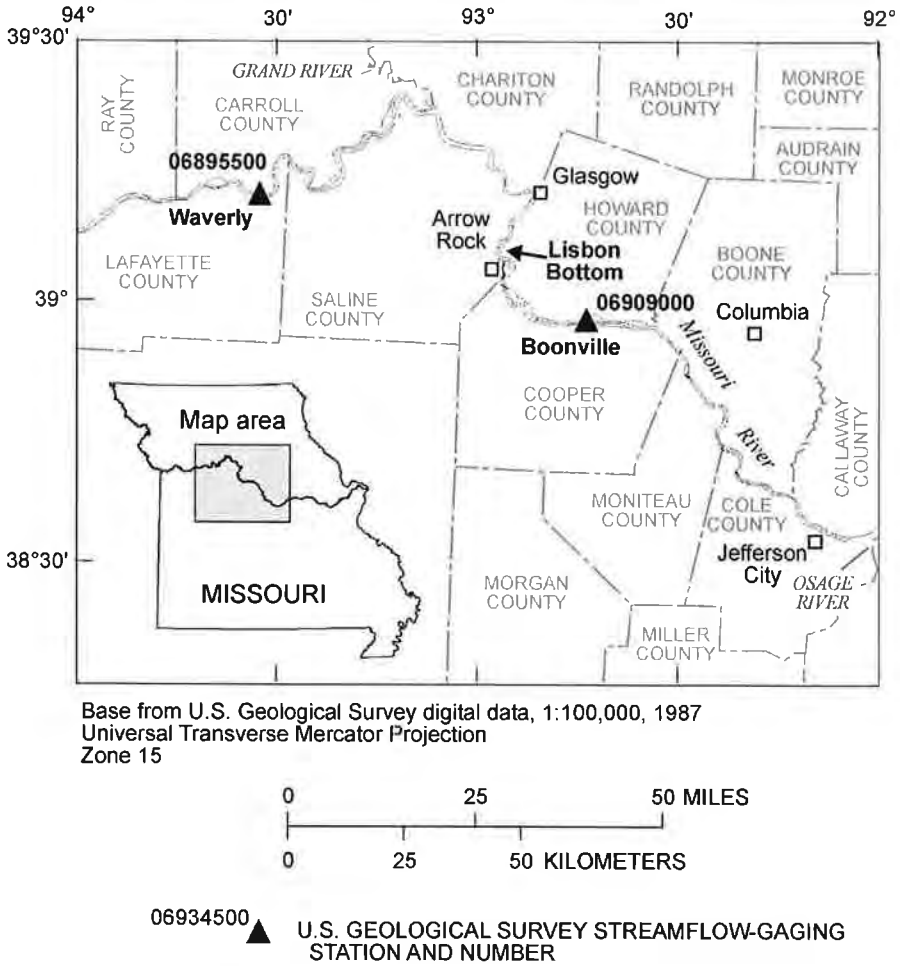

Figure 2. Location of Lisbon Bottom and nearby stream gages on the Lower Missouri River.

[1] River miles are the long-established addressing system for distances on the Missouri River. River miles increase upstream from 0 at the junction of the Missouri and Mississippi rivers at St. Louis. A mile is equal to 1.6 kilometers.

Habitat alterations over the last 50 years have led to significant changes in the quality and quantity of habitat available for fish and wildlife resources. Until the floods within the last decade, the biota within the lower river corridor had been poorly studied. Basin-wide changes have been correlated with significant declines in native species including fishes (for example, pallid sturgeon, lake sturgeon, and sicklefin chub) and birds (for example, piping plover and least tern). However, many declines in biota have not been adequately quantified, in particular those at smaller spatial and temporal scales. There is a lack of information concerning cause/effect relationships linking biological measurements to the hydrologic factors that are responsible for creating and maintaining these habitats. Moreover, there is a general lack of information available to compare the relative value of habitats in the existing navigation channel and agricultural lands with those of more natural reaches. Efforts to rehabilitate the Missouri River depend on developing quantitative understanding of management links, for example, how can changes in flow management regimes and river-corridor topography affect the availability, relative contribution, and productivity of flood-plain habitats? And if habitat is provided, how do riverine biota respond? Because hydrologic and geomorphic dynamics are the primary factors responsible for structuring the river corridor, the links must be established at a scale at which these factors can be quantified so that future management of flood-plain habitats can be based on the relationships between biological responses and changes in physical characteristics.

General Objectives and Study Approach

The general objectives of this project are to:

- Quantify spatial and temporal distribution of biota in aquatic habitats of Lisbon Bottom in relation to changes in hydrological variables that are associated with the spring flooding regime.
- Document biological responses as they relate to habitat dynamics.
- Analyze and interpret these results to provide managers with biological information necessary to develop management strategies for Lisbon Bottom and other tracts of the Big Muddy National Fish and Wildlife Refuge.

Lisbon Bottom is being managed by the USFWS in a passive manner, in other words, with a regime that allows the landscape to be formed by natural flooding and vegetative succession (fig. 3). Although still affected by flow regulation and the requirement to maintain navigation in the main channel, Lisbon Bottom is representative of many high-sinuosity portions of the Lower Missouri River, and provides an important case study to evaluate passive, least-cost approaches to river-corridor ecosystem management. The presence of existing levee breaks from the 1990s floods, and the naturally formed secondary chute (fig. 3) have re-established a dynamic connection to the Missouri River at this location. With a close proximity to the U.S. Geological Survey (USGS), Columbia Environmental Research Center and to long-term stream gages, Lisbon Bottom presents the opportunity to study ecosystem processes and dynamic geomorphology in a setting that more closely mimics the natural riverine system than any other site on the Lower Missouri River.

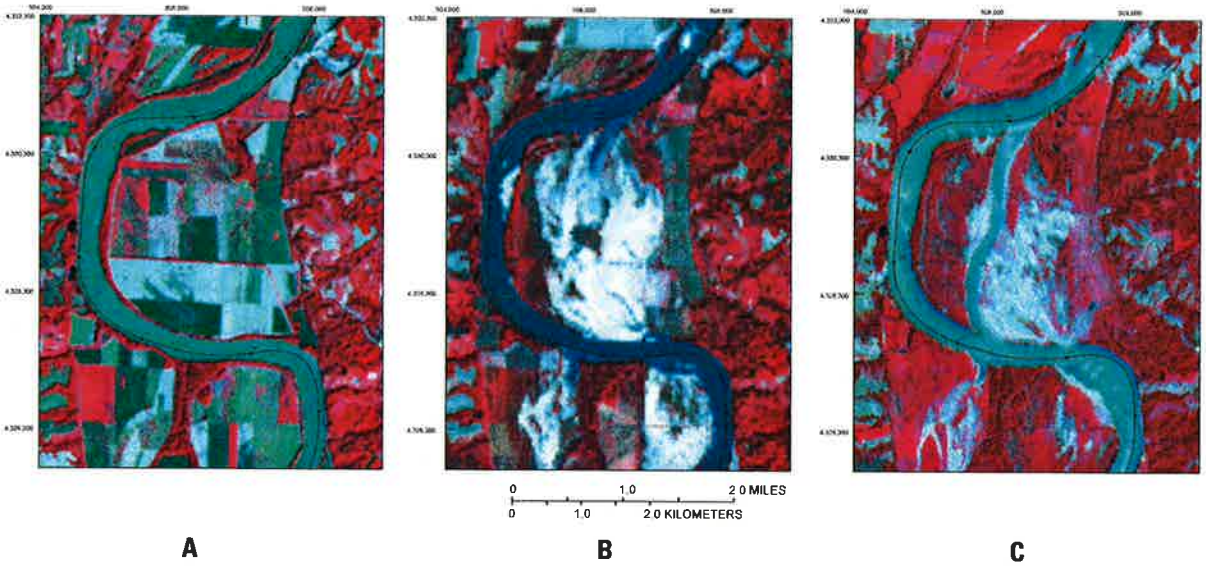

Figure 3. False color images of Lisbon Bottom showing extensive changes to land cover from 1992 to 1997. Red colors are indicative of vegetative cover; the more intense the color, the greater the biomass. The bright white in B. and C. represents sand deposits. From October 1994 to August 1997, much of Lisbon Bottom became more vegetated, despite the floods of May 1995 and June 1996. The difference in the color of the river between A., C. and B. is related to sediment load which is higher in A. and C. **A.** Pre-flood Thematic Mapper image September 24, 1992, Boonville discharge = 66,000 cfs. **B.** SPOT image October 11, 1994, Boonville discharge = 46,700 cfs. **C.** SPOT image, August 23, 1997, Boonville discharge = 91,100 cfs. Imagery courtesy of Washington University.

Wetland classification schemes represent one reference point for designing biological research studies within standing water bodies, and allow scientists to stratify sampling regimes. Cowardin and others (1979) provides the most complete, broad-scale descriptions, and under this system the wetlands at Lisbon would be classified as palustrine of either the forested, emergent or scrub-shrub types. However, the wetlands at Lisbon could not be adequately classified before the beginning of this study because the water source, permanence, and vegetation types were unknown and had not been surveyed. We expected the wetlands at Lisbon would represent a diverse continuum of these conditions, and recognized that the importance of each of these variables may not be equal for each animal group we studied. Galat and others (1998) provided a more specific categorization of different wetland types in the Lower Missouri River flood plain, which include: 1) remnants, or aged shallow wetlands surrounded by mature trees; 2) oxbows, created by channel cutoffs from tributary streams or the Missouri River mainstem, 3) scour holes, created by scouring when levees broke during the recent floods, and 4) ephemeral or temporary wetlands which consist of shallow depressions with early successional plants, including moist-soil vegetation and woody shrubs. At present, Lisbon Bottom contains wetlands that fall under all of these categories except for oxbows. Some wetlands represent either transitions between categories, and may actually change types over time, or fall under different categories from year to year. Therefore, each of the chapters in this report may use a different breakdown of wetland types for analysis and interpretation.

The studies presented as chapters below have been designed to provide an understanding of the links between biologic responses and hydrologic dynamics. This report provides biological, physical and chemical aspects of conditions at Lisbon Bottom—including surface water, limnology, and community dynamics of

benthic invertebrates, zooplankton, fish, and waterbirds. The results encompass late winter, pre-flood, flood, and post-flood conditions, as well as pre- and post-periods of waterfowl migration and flood-plain spawning or dispersal of fishes. The biological components of the study represent a chronology of changes in conditions observed throughout the study period from March through July 1999; hydrologic components cover two periods during June 1999–October 2001 (fig. 4). Biological and hydrologic components sampled a wide range of hydrologic conditions, from typical late fall low flows to floods with 2–10 year recurrence intervals. Notably, the study components were carried out at the end of a relatively wet 10-year period when the Lower Missouri River experienced as many as 18 individual floods exceeding the 2-year magnitude (fig. 4).

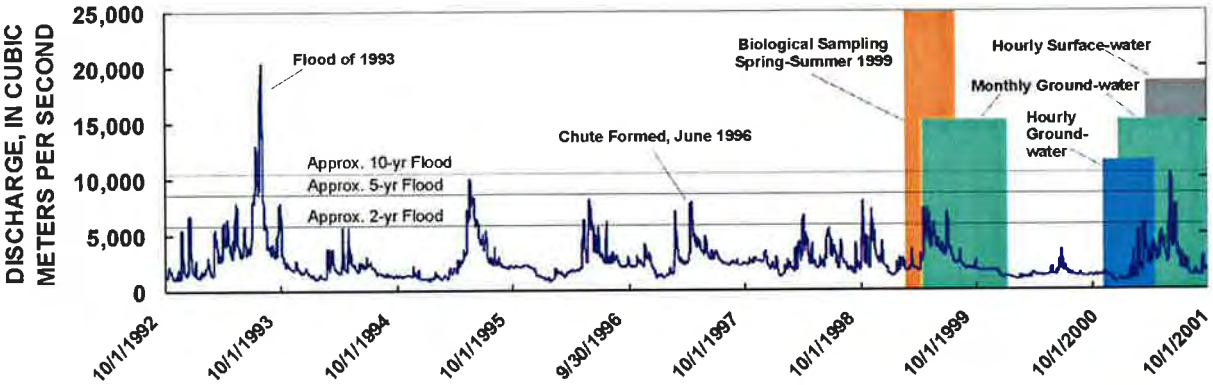

Figure 4. Daily discharges affecting Lisbon Bottom, water years 1993–2001, as measured at Boonville, Missouri, and time intervals of biological and hydrologic data collection.

Historical Background and Physical Setting

Lisbon Bottom contained mature stands of cottonwood (*Populus deltoides*) and willow (*Salix* spp.) until the late 1950's, when the forest resources were logged and the land was cleared for intensive agriculture. Row cropping covered up to 90% of the landscape (Jacobson and others, 1999) and according to aerial photographic images, the north end contained a few remnant wetlands surrounded by mature trees. These wetlands (5 and 8) (fig. 5) still exist today, and physical evidence suggests that they were used for irrigation at one time. The levee across the north end of the bottom, which protected the row crop area from flooding, had been broken and repaired eight times within the previous 20 years before the flood of 1993. The position of this levee at an outside bend of a meander in the river, and the historical propensity to flood and damage agricultural crops, made Lisbon Bottom a prime candidate for public land acquisition after the 1993 flood. Additional large floods in 1995–1997 (fig. 4) helped create a natural chute that passes through Lisbon Bottom. This channel widened and eroded laterally during high water levels in 1996–1998, and has reached a more stable configuration during the last few years. The large island resulting from this chute formation is usually inaccessible except by boat and is not included in the biological components of this study. In the spring of 2000, the U.S. Army Corps of Engineers installed a control structure at the head of the chute to regulate the percentage of flow diverted from the mainstem of the river. The chute now contains sandbars and its own secondary channels. The geomorphic evolution of the chute is documented in Jacobson and others (2001).

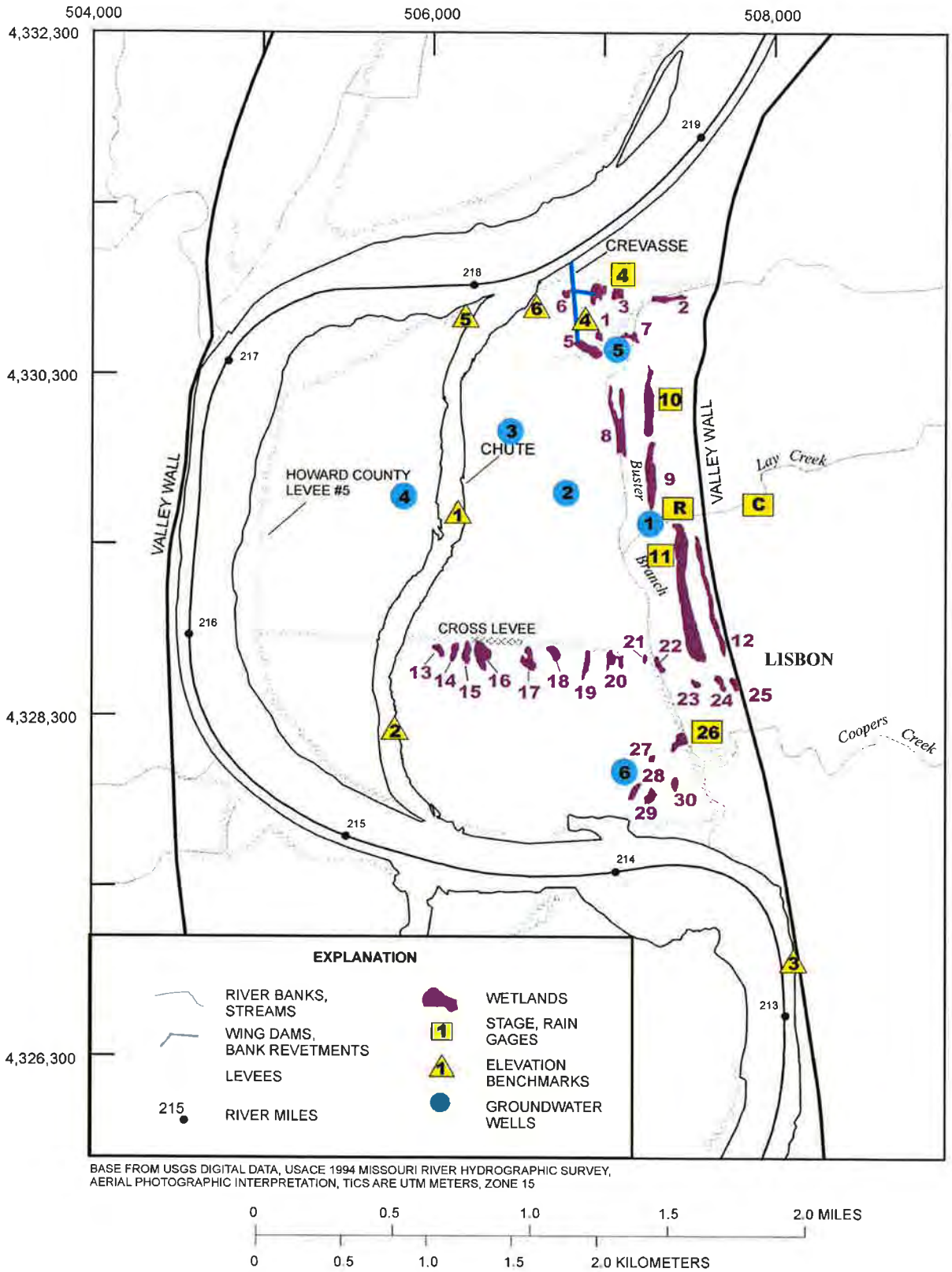

Figure 5. Map of Lisbon Bottom showing geographic features and wetlands.

In addition to large quantities of sand deposition, the 1993 flood created several connected and non-connected scours at Lisbon Bottom, including those associated with an east-west cross-levee that transects the area (fig. 5). The Missouri River Post-Flood Evaluation Project (MRPE) was initiated in late 1993 by the Missouri Department of Conservation in cooperation with the University of Missouri and included multidisciplinary studies designed to examine the effects of the 1993 flood and use of newly created habitats by different organism groups (Humburg and Burke, 1999). Three wetlands and one connected scour hole involved in the MRPE project are located at Lisbon Bottom, and included TLIS (Wetland 10 in our study), LP-1 (Wetland 7 in our study), LP-4 (Wetland 4 in our study), and S-14 (connected scour in our study). However, the area has changed dramatically since the MRPE study was completed; more flooding in 1995 and 1997 caused additional sedimentation and a surge in vegetation growth. Thick stands of cottonwood and willow in the northern portion of Lisbon Bottom slow the current and reduce scouring potential when floods cover the area. Even-aged stands of these tree species resulting from seed-bank germination after all three major floods (1993, 1995, 1997) are distinguishable from one another in both aerial photographs and on the ground, and border several of the wetlands in the northern part of the area. Sedimentation and changes in basin morphology have occurred since the 1993 flood, and can most easily be observed in Wetland 5. Sedimentation has significantly reduced the water volume in this wetland, which dried in September of 1999 for the first time since the 1993 flood.

Wetlands receiving part of their water supply from creek systems were not recognized in most definitions of flood-plain wetland types because streams entering the Missouri River flood plain have historically been diverted away from bottomland areas that are used for agriculture. This was accomplished by channelization, and by diverting stream water along secondary levees to provide more direct routes to the Missouri River mainstem while improving drainage for agricultural production. Lisbon Bottom has three separate creek systems entering it, two of which were originally diverted in this manner and one that flows into a privately owned remnant wetland along the north border (Buster Branch, fig. 5). Shortly after the Lisbon tract was purchased by the U.S. Fish and Wildlife Service in 1995, earth-moving equipment was used to construct an opening in a secondary levee to allow one of the previously diverted creeks to flow onto the flood plain. This resulted in marsh areas, an increase in moist-soil vegetation in that vicinity, and a new water connection between Wetlands 11 and 22. This surface water input changed the relative contribution of water sources, vegetation type, and permanence of some of the wetlands.

At present, the wide array of wetland types at Lisbon represent a continuum of aquatic and terrestrial plant diversity, basin morphology, permanence, and influence from various water sources. The dominant semi-aquatic emergents and moist-soil plants observed adjacent to the wetlands at Lisbon include cocklebur (*Xanthium* spp.), smartweed (*Polygonum amphibium* var. *emersum*), sedge (*Carex* spp and *Cyperus* sp.), American bulrush (*Schoenoplectus fluviatilis*), arrowhead (*Saggitaria* spp.), reeds (*Phragmites* sp.), cattail (*Typha* sp.), spikerush (*Eleocharis* spp.), and rice cutgrass (*Leersia oryzoides*). Young willow, cottonwood, and alder (*Alnus serrulata*) were the dominant woody plants along all wetland margins. Mature cottonwoods, American sycamore (*Platanus occidentalis*), and spirea (*Spiraea* spp.) are also present in some areas.

Lisbon Bottom also has two additional features that contribute significantly to the amount of surface water entering the flood plain from the river. A crevasse passes through the levee located on the north end,

allowing water to enter the area during higher river levels (fig. 5). This feature supplies six different wetlands with surface water both directly (Wetlands 4 and5), and indirectly through seepage or overflow (Wetlands 2, 3, 7, and 8). During water level increases in spring, this begins to occur at river levels a few feet below flood stage. The other feature is a connected scour hole formed by a levee break on the south end of the bottom during the flood of 1993 (S-14 from the MRPE study). Even though it has undergone sedimentation, the levee breaks allow water to enter from the river by backflooding, filling Wetlands 26, 27, 28, 29, and 30 at river levels somewhat below the stage of overall flooding of Lisbon Bottom. Both of these features are important because if they did not exist, higher river stages would be required to rapidly fill the wetlands during spring flood events, and most basins would have to rely more heavily on groundwater recharge, rainfall, and creek systems for their water source. Flooding and influx of river water is required to hydrate some of the temporary wetlands at Lisbon, such as Wetland 10 that has very little creek influence and remains dry unless river flooding or significant local rainfall events occur. For more detailed information on the physical setting and hydrological characteristics associated with Lisbon Bottom, see the summaries by Jacobson and others (1999, 2001) and Chapter 1 of this report.

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Chapter 1. Hydrology of Lisbon Bottom

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Abstract

Lisbon Bottom consists of approximately 875 ha of river bottom along the Missouri River in Howard County, Missouri, from approximately river mile (RM) 213 to RM 219 (figs. 1-1–1-3). Before regulation and structuring of the Missouri River, riverine[1] areas like Lisbon Bottom were shifting mosaics of dynamic habitat patches that were created and maintained by hydrologic and geomorphic processes. Flow regulation, navigation structures, and bank-stabilization projects isolated Lisbon Bottom from the river by decreasing the magnitude and frequency with which hydrologic and geomorphic processes could alter habitat characteristics. The flood of 1993 breached agricultural levees around Lisbon Bottom, creating numerous levee-break scours, and re-establishing a connection to the Missouri River.

Management of wetland resources requires an understanding of how water recharges wetlands, how quality of the water may vary with source, and the costs and ecological benefits associated with manipulations of water sources. Observations and monitoring of surface- and ground water at Lisbon Bottom indicate the relative contributions to wetlands of water from the main channel by overbank flooding, water from the main channel by subsurface connection, water from direct rainfall, and water discharging from valley-wall tributaries. These sources of water are distributed among the many types of wetlands that exist at Lisbon Bottom, including deep scours formed during the 1993 and 1995 floods, shallow temporary wetlands with minimal direct surface drainage area, and shallow temporary wetlands with direct surface-water connections to valley-wall tributaries.

Deep scours associated with levee breaks and crevasse splays at upstream and downstream ends of Lisbon Bottom are connected through subsurface flow to the main channel. Water levels in these scours vary with flow in the main channel, direct rainfall, and to a lesser extent, valley-wall tributaries, but because of ground-water sources, the water levels change slowly over time. Because the down-valley gradient of Lisbon Bottom is greater than the channel slope, flooding in the main channel recharges downstream scours through surface-water flow before recharging upstream scours. Wetlands along the valley wall, far from the main channel, are recharged by main-channel flow only when flow is well over bank. These wetlands are recharged more frequently by local rainfall that falls directly into surficial drainage areas and by flow from valley-wall tributaries. Hydrologic variation in wetlands that are recharged by local rainfall is of greater magnitude and much more frequent than variation in deep scours, resulting in markedly different hydrologic disturbance regimes.

[1] The term *riverine* is used to describe the area encompassing the channel and adjacent flood-plain areas; the flood plain is considered to extend to those areas that would potentially flood with an average frequency of at least once in one hundred years in the absence of bank revetments and levees.

Introduction

Lisbon Bottom is located in a valley segment[2] between the junctions of the Grand and Osage Rivers (figs. 1-1, 1-2). This segment is on the margin of the Ozark Plateaus Physiographic Province and it is cut into relatively uniform Paleozoic cherty limestone, cherty dolomite, and minor quantities of sandstone and shale. This segment is characterized by a relatively wide valley subsegment from the Grand River junction to near Glasgow, Missouri (RM 224-250) and a narrow valley subsegment from Glasgow to the Osage River junction (RM 131-224). In the wide-valley subsegment and the segment upstream of the Grand River junction, the Missouri Valley is nearly five times wider than in the narrow-valley subsegment. Downstream of the Osage River junction, the Missouri River is increasingly affected by hydrologic characteristics and the addition of coarse sediment from the Ozark Plateaus. Lisbon Bottom is within the narrow-valley subsegment where the valley is about 3.5 km wide, with few alluvial terrace remnants, and steep, bedrock bluffs. The narrow valley and bedrock walls act to confine large floods and to promote scour and secondary currents where the channel impinges on the valley walls. A more complete description of Missouri River physiography can be found in Kelmelis (1994); a description of physiography and geomorphology of Lisbon Bottom can be found in Jacobson and others (1999).

Lisbon Bottom is a typical loop bottom (fig. 1-3; Schmudde, 1963). Before stabilization, loop bottoms would migrate downstream due to lateral erosion at the upstream margins and deposition on the downstream margins. Flood flows that overtopped the upstream margin would tend to build up sandy natural levees, which might be separated by interposed crevasses where concentrated flows cut through the levees. Crevasses commonly occupied swales left from previous channel migration and so acted to guide flood flows from the channel, across the loop bottom, and toward the valley wall. Because of this, there is a tendency for loop bottoms to have wetlands along the downstream one-half of the valley wall. At Lisbon Bottom small tributary basins also provide water for these low, wet areas. Because levees and splays build up the upstream margins of loop bottoms, natural loop bottoms have higher gradients than the channel, and they tend to flood from downstream to upstream as water backs up through old overflow channels. As a result, when large floods overtopped natural levees at the upstream margin, they would typically encounter slackwater from backflooding.

The upstream margin of Lisbon Bottom has natural levees in excess of 186 m above sea level (asl); the downstream margin has elevations as low 181 m asl. Ridges and swales oriented northwest to southeast are apparent on the 1979 topographic map of the bottom but have been partly obliterated by erosion and deposition by the 1993 flood (fig. 1-4). The slope of the bottom is approximately 0.8 m/km (0.0008 m/m), compared to 0.2 m/km (0.0002 m/m) in the channel.

The soils of Howard County including the Lisbon Bottom area were mapped in the 1970s (Grogger and Landtiser, 1978) and have not been remapped since. As a result of the floods of 1993–1996, the surface materials of Lisbon Bottom have changed extensively in distribution; however, the description of soils from

[2] A valley segment is a length of a river valley between substantive tributaries and having relatively uniform physiographic and geologic characteristics. For the purposes of this report, a tributary is considered to be substantive if it adds greater than approximately 5% of the cumulative drainage area and (or) drains an area of significantly different hydrologic response, sediment yield, or water-quality contribution.

1978 is still valid. The surface soil consists of materials ranging from well-sorted sand to silty clay, and ranging from zero pedogenic[3] alteration to development of organic-rich A horizons and weak B horizons. The soils of Lisbon Bottom are classified as entisols and mollisols, indicating the predominance of weak pedogenic development and accumulation of organic matter in wetter environments. The 1978 soil maps showed a unit classified as riverwash in lateral bars along the left bank[4] and adjacent to the channel. Sarpy sand (typic udipsamment) was mapped in natural levee positions along the upstream, left bank RM 216-218, and in a long splay extending approximately one half of the long axis of Lisbon Bottom, adjacent to and east of the chute. This sand splay is indicative of historic, high-energy deposition on Lisbon Bottom similar to that which occurred in 1993, but prior to 1978. Hodge loamy fine sand (typic udipsamment) was mapped on low-relief ridges and also indicates deposition of bars or splays. Hagni silt loam (mollic udifluvent) is stratified silt loam and fine sandy loam, and was mapped on low ridges and intermediate elevations on the bottom. Leta silty clay (fluvaquentic hapludoll) is the wettest soil mapped at Lisbon Bottom, and consists of fine sediments deposited in overflow channels, swales, and low areas subjected to back flooding. Nodaway silt loam (mollic udifluvent) was mapped on alluvial fans from the tributary valleys of Buster Branch, Cooper Creek, and unnamed smaller tributary basins along the eastern valley wall. These alluvial fans were formed from re-worked loess from the uplands located to the east of Lisbon Bottom; the fans provide bench areas at somewhat higher elevations immediately adjacent to the valley wall.

Climatology, Hydrology, and Regulation History of the Grand-Osage Segment

The regional climate for Lisbon Bottom is temperate; average annual temperature is 12.2 °C (54 °F) and average annual precipitation is 990 mm (39 inches) (NOAA, 1997). Low temperatures and low precipitation tend to coincide in January, but peak precipitation tends to occur in May, two months ahead of the peak temperature.

The closest long-term, discharge-rated stream gage is located at Boonville, Missouri (fig. 1-1). The U.S. Geological Survey has operated this stream gage continuously since 1925. Between Lisbon Bottom and Boonville, the Missouri River gains very little discharge; the drainage area increases by only approximately 0.5%. Therefore, the Boonville stream gage can be used to evaluate hydrologic characteristics at Lisbon Bottom. However, because the valley and channel cross section and hydraulic roughness are different between Lisbon Bottom and Boonville, the relative stages and areas inundated are not expected to match, especially at flows above bankfull. The U.S. Geological Survey also operates a gage approximately 12 km upstream of Lisbon Bottom at Glasgow, Missouri. This gage has a short and non-continuous record of discharge, but it provides a long record of stage for comparison with Lisbon Bottom.

[3] Pedogenesis refers to the integrated chemical, physical, and biological processes that form and differentiate soil horizons.
[4] Left and right bank refer to banks as seen while facing downstream.

Background and Objectives

River-corridor wetlands can be recharged through multiple pathways, or some combination of pathways (Kelly, 2001). The most direct source of recharge is direct rainfall, or rainfall that contributes runoff from local drainage areas around wetlands. Many river bottoms along the Missouri River also receive local runoff from valley-wall tributaries that collect runoff from drainage basins in the uplands adjacent to the Missouri River valley. Historically, the tributaries flowed onto the valley bottom and recharged wetlands along their banks and in overflow basins. Although many of these tributaries flowed naturally along the valley wall until hitting the mainstem of the river, landowners and drainage districts often stabilized this alignment with levees to protect the agricultural bottomland. Bottomland wetlands are also recharged from overflow of the main channel; when the flow is from the upstream margin of the bottom the flooding is called *topflooding* and when it is from the downstream margin it is referred to as *backflooding*. Finally, bottomland wetlands can be recharged or maintained through ground-water reservoirs, which can in turn be recharged from the main channel, valley-wall tributaries, or direct rainfall. Because ground water is not directly observable, the least is known in general about the relative influence of ground water. Studies completed in areas similar to Lisbon Bottom indicate that the ground-water reservoir can fluctuate with river level, but usually with a lagged and lower-magnitude response (Kelly, 2001). Hence the ground-water reservoir acts as a buffer that evens out wetland recharge events and ground-water observations indicate a longer-term, averaged hydrologic influence on wetlands compared to surface-water events.

Recharge pathways may also be quite dynamic on a multi-year to decadal time frame. Alterations of the land surface by erosion, sedimentation, engineering, and biota have the potential to change how surface and ground water are distributed. Deepening of side-channel chutes, for example, may contribute to drawdown of the water table and dewatering of wetlands. Beavers (*Castor canadensis*) are capable of excavating new channels and damming up old channels, thereby substantially changing the distribution of surface water and sediment. Continuing alteration of the land surface by geomorphic and biologic processes diminishes the ability to extrapolate recent hydrologic conditions over multi-year time frames.

Informed and cost-effective management of river-bottom wetlands requires an understanding of how water travels to wetlands, and how recharge pathways influence hydroperiod, water quality, and disturbance regime. A general question confronting land managers is whether the management objectives can be achieved without altering natural wetland recharge processes, or whether instead it is cost effective to manage recharge sources actively. The objective of this study was to develop a general understanding of hydrologic controls on wetland recharge, using Lisbon Bottom as a representative Missouri River bottom. Stage-discharge relations for Lisbon Bottom and short-term monitoring of hydrologic responses were intended to develop a preliminary understanding of recharge pathways and possible ecological consequences.

Methods

Separate, but related, ground-water and surface-water datasets were assembled for this study. The surface-water part of the study focused on characterizing pathway, magnitude, and frequency of recharge at selected wetlands. The ground-water part of the study was developed from widely spaced monitoring wells that characterize ground water broadly over Lisbon Bottom.

Four wetlands were chosen among the many present at Lisbon Bottom, to represent distinct hydrologic environments (fig. 1-3). Each of these wetlands can receive water from multiple sources, depending on different combinations of main-channel flow, valley-wall tributary flow, local precipitation events and the disposition of the ground-water table. Nonetheless, the four wetlands were chosen to illustrate the best end members of surface-water hydrologic environments.

Wetland 4 is a deep levee-break scour approximately 170 m from the main channel. It was chosen to represent wetlands that would be flooded from upstream (topflooding). The area between Wetland 4 and the channel is characterized by sandy soils and gently hummocky topography associated with a natural levee and small crevass splays. Wetland 4 is not connected to a valley-wall tributary most of the time, but can receive overflow when local rainfall contributes to high runoff in the Buster Branch drainage basin.

Wetland 26 was chosen to represent wetlands subject to flooding mostly from downstream (backflooding). This is also a deep, levee-break scour. It is approximately 500 m from the main channel and, although separated by a levee, approximately 150 m from a small tributary which is connected directly to the main channel (Buster Branch). Wetland 26 also can receive flow upstream from Buster Branch during large local runoff events.

Wetland 11 was chosen to represent a wetland that owes much of its recharge to valley-wall tributary flow during much of the year. It is fed by the upper reaches of Buster Branch and by Lay Creek, which was routed along the valley wall and separated from the bottom by a levee prior to USFWS ownership of Lisbon Bottom; flow to Wetland 11 was restored in 1994 when Refuge managers breached the valley-wall levee. Wetland 11 is extensively vegetated with grasses, and other emergent vegetation. Wetland 11 is topographically wide and shallow and probably typical of many wetlands that existed in this part of the Missouri River valley bottom prior to agricultural land uses.

Wetland 10 was chosen to represent a temporary wetland without direct connection to a valley-wall tributary. Only during very wet conditions does Wetland 10 receive discharge through the upper reaches of Buster Branch. Wetland 10 is a broad, shallow basin that is extensively vegetated with cottonwood and willow trees that germinated after the flood of 1993, which is in marked contrast to Wetland 11 that has extensive aquatic macrophytes and few trees.

Elevation benchmarks were installed upstream and downstream of Lisbon Bottom along the main channel, in order to relate discharge in the main channel to river stage. In addition, elevation benchmarks were installed upstream and downstream in the side-channel in order to develop stage-discharge relations in the chute (fig. 1-3). Stages measured from these benchmarks were related to discharges at Boonville. In addition, high-water marks from the flood of June 2001 were surveyed and used to extend the stage-discharge relations to a relatively infrequent flood. Pressure transducers and data loggers were installed in four wetlands and operated

during spring–summer 2001. Stages at the pressure transducers were converted to water-surface elevations by surveying true elevations into the gage sites. A rain gage and one stream stage gage (Lay Creek) were installed to characterize hydrologic inputs from local rainfall and runoff from valley-wall tributaries.

Six ground-water monitoring well locations and two staff gages were established to investigate ground-water flow relations to wetland recharge (fig. 1-3). Monitoring wells were constructed of 2-inch diameter schedule 40 PVC. Each deep well is 30 to 40 feet deep with a 10 foot screened interval at the bottom of the well and a one-foot sump at the base of the well. Staff gages were constructed of steel plates with 0.01 foot graduated marks. Ground-water levels, river stage, and chute stage were measured in June, August, October, November of 1999; January and February of 2000; and January, February, March, April, May, June, July, September, and October of 2001. Hourly river stages at RM 218.5 (calculated from the Boonville USGS gage) and dates of ground-water measurements are shown on figure 1-5.

Depth to ground water ranged from -1.08 to 18.82 feet (-0.33 to 5.74 m) for all manual ground-water measurements. Measured water-table altitudes ranged from 588.45 feet to 606.54 feet (179.36 to 184.87 m) above sea level. Measured Missouri River stage altitudes at mile 218.5 ranged from 585.37 to 611.75 feet (178.42 to 186.46 m) above sea level. Measured chute stage altitudes on the south end of the chute ranged from 584.70 to 598.63 feet (178.22 to 182.46 m) above sea level. Wetland stage measured at Wetland 5 ranged from 601.76 to 604.14 feet (183.42 to 184.14 m) above sea level.

Hourly measurements of ground-water level and rainfall were made using an automatic water level recorder and rain gage at Well 2 from November 29, 2000 until April 28, 2001. Ground-water altitudes ranged from 588.20 o 598.25 feet (179.28 to 182.35 m) above sea level. Between November 29, 2000 and April 28, 2001, maximum rainfall was 2.34 inches on April 12, 2001 and total rainfall was 17.63 inches.

Wetland water-surface elevation data and ground-water monitoring results do not completely overlap with the time period of biological and limnological sampling described in other sections of this report. Nevertheless, the hydrologic data show general characteristics of wetlands at Lisbon Bottom that help put biological and limnological data in hydrologic context. The stage-discharge data developed for the main channel lead to basic understanding of frequency of flooding from the main channel, and how frequency varies among different parts of Lisbon Bottom.

Results and Discussion

Stage-discharge and Overflow of Lisbon Bottom

Stage-discharge relations at RM 218 (the upstream margin of Lisbon Bottom) and at RM 213.4 (the downstream margin of Lisbon Bottom) have convex-upward shapes indicative of flows that spread out of the channel and over un-leveed flood plains (fig. 1-6). The relations are modeled well by relating stage to the logarithm of discharge.

The stage-discharge relation can be compared with flow frequency at the Boonville gage to calculate stage frequency at Lisbon Bottom. Stage frequencies are plotted in figure 1-7 for each day of the water year (October–September), as calculated from the post-regulation Boonville record 1967–1999. Reference lines are

provided to indicate the stage at which general flooding occurs upstream and downstream. These reference elevations are based on general elevations of the land surface adjacent to the channel, ignoring human-made levees. As shown in the figures, flooding from upstream or downstream is most likely during April–May. Downstream flooding is significantly more likely than upstream flooding due to the general slope of the bottom surface, with some periods having frequencies as high as 1 in 10. On an annual basis, flooding from upstream occurs only 3 days per year on average whereas flooding from downstream occurs 11 days per year on average.

Surface-water Relations and Wetlands

Wetlands at the upstream and downstream margins of Lisbon Bottom hold water more persistently than interior wetlands. Stage gages at Wetland 4 at the upstream end of Lisbon Bottom maintained a high water-surface elevation even when flow in the main channel was substantially lower than the wetland water surface (fig.1-8A). Wetland 4 is a relatively deep scour at a levee break. Local rainfall events—shown as Lay Creek stages in fig. 1-8B for reference—also recharged Wetland 4; the April 10, 2001 storm is a good example. The period 6/4–6/12/2001 included a flood with daily mean discharge of 365,000 cfs at Boonville, a flood of approximately 10-year recurrence (U.S. Army Corps of Engineers, written communication, 1997). Although local rainfall also helped recharge Wetland 4 during this flood, at stages of about 185.5 m asl, most of the wetland area at the upstream end receives direct flow from the Missouri River.[5] Once recharged by the June 2001 flood, Wetland 4 remained at a high stage for at least another month, even when flow in the main channel was as much as 3 m lower than the wetland in mid- July. This observation indicates that although it is relatively deep and within the part of the bottom that should be dominated by sandy natural levee deposits, Wetland 4 is not strongly connected by ground water to the river.

Wetland 26 at the downstream end of Lisbon Bottom is another deep, levee-break generated scour. Unlike Wetland 4, however, Wetland 26 shows less persistence of water surfaces after recharge events, indicating that water is flowing away from Wetland 26, probably through ground-water flow (fig. 1-8C). Topography indicates that Wetland 26 should recharge from backflooding of the Missouri River when stage at RM 213.4 reaches approximately 183.6 m asl. Wetland 10 flooded from local rainfall and runoff during the 4/10/2001 event and was then relatively unaffected by main channel flows until 5/8/2001. During 5/7–5/10/2001 there was no local precipitation and a small rise in river level to about 182.8 m asl was associated with about 0.5 m of rise in Wetland 26. As the river stage was considerably less than the 183.6 m asl that should allow surface-water connection between the wetland and the scour, this is interpreted as evidence of a ground-water or other connection between the wetland and the main channel. Possibly, beaver excavations or a mis-operating flap gate on a culvert under the levee allowed water to enter Wetland 26 at stages well below the general land surface. Subsequent rises in wetland stage are difficult to separate from local rainfall effects, but concurrent rises at RM 213.3 and in Wetland 26 indicate general backflooding when stage at RM 213.6 is about 183.6 m asl (with water levels at Wetland 26 about 184.2 m asl due to slope of the water surface between Wetland 26 and the stage measurement at RM 213.3).

[5] The minimum stage at RM 218 sufficient for surface recharge to Wetland 4 may be as low as 184 m asl through small crevasses; however these crevasses apparently plug frequently with large woody debris and sediment, so they should probably not be considered a reliable source of recharge.

Wetlands 10 and 11 contrast substantially with Wetlands 4 and 26 (fig. 1-8D). Both wetlands are broad, shallow basins. Wetland 10 has only infrequent connection to valley-wall tributaries, and due to the shallow depths, little opportunity for ground-water inflows. Wetland 10 was dry during most of this study, with exceptions that resulted from intense local rainfall and total flooding of the bottom. Wetland 10 water level increased as a result of 2.4 inches of rain received in the 4/10/2001 storm and remained dry through several subsequent storms of as much as 1.0 inch of rainfall. Wetland 10 recharged again from 2.05 inches of rainfall from two storms 6/3–6/6/2001, just before and overlapping with the large flood 6/5–6/11/2001. Wetland 10 flooded again 6/22–7/1/2001 when flow in the main channel at RM 218 again surpassed 185.5 m asl.

In contrast to Wetland 10, Wetland 11 was frequently flooded by local rainfall events as well as by infrequent but large floods from the main channel (fig. 1-8D). Local rainfall of as little as 0.42 inch (for example, 4/14–4/15/2001) resulted in water-surface elevation changes of several centimeters. Rainfall of 0.69 inch during 5/11/2001 resulted in 30 cm of rise in water surface in the wetland. Due to the broad, shallow morphology of the basin, small changes in water-surface elevation can affect large areas of wetland. Direct connection of Wetland 10 to Lay Creek results in frequent, small-magnitude recharge events in addition to any recharge from bottom-covering floods.

Ground-water Relations and Wetlands

Intersection of the water-table surface with land-surface topography indicates the extent of wetlands that would exist if wetlands were connected to ground water through highly transmissive sediments and if ground water were the only source of recharge to the wetlands. On the short term—days to weeks—actual water-surface elevations and spatial extents of wetlands will be different from the ground-water prediction because of local rainfall, runoff from valley-wall tributaries, or overflow from the main channel. The potentiometric surface (distance of the water surface above or below the land surface) and the calculated locations of wetlands at Lisbon Bottom are shown for each ground-water measurement event in figure 1-9. The potentiometric surface is the surface that represents the static head of water in an aquifer; it is defined by the levels to which water will rise in tightly cased wells from a given point in an aquifer. In the Missouri River alluvial aquifer, the potentiometric surface defines the water table. The distance of the water surface above or below the land surface was calculated by subtracting the potentiometric surface from the land surface using high-resolution digital elevation data of Lisbon Bottom. The calculation includes topography of the chute and the channel. However, the topography of the navigation channel depicted in these figures is very approximate and hydraulic control exerted by a notched structure at the upstream end of the chute is not taken into account; therefore, results in the navigation channel and chute should be interpreted with caution.

Several important features of the hydrology of Lisbon Bottom are illustrated in this figure. High ground-water levels measured during times when river stage is high as shown in figure 1-9 for August 11, 1999 and April 4, 2001–July 26, 2001, indicate the general close connection between the river and ground water at Lisbon Bottom. Low ground-water levels occur when river stage is low as shown in figure 1-9, October 7, 1999–February 21, 2001 and September 6–October 3, 2001. A persistent area of higher ground-water levels occurs near the eastern valley wall. This area is somewhat topographically lower that the rest of Lisbon

Bottom. These two characteristics result in a predicted close interaction between ground-water levels and wetland stages for the deeper wetlands located along the eastern valley wall. These include Wetlands 4, 5, 7, 8, 20, 21, 22, 25, 26, and 28. Slow recession of water surfaces after overflow during the June 2001 flood (figs. 1-8A, C) indicates that the connection between ground water and wetlands can be slow, perhaps because of fine, impermeable sediment that has been deposited in the wetlands.

Other shallower wetlands in this general area also can be affected by ground water when ground-water levels rise above the land surface (fig. 1-9, June 26, 2001). Conversely, Wetlands 13 through 19, located along the old cross levee on the southern part of Lisbon Bottom are largely unaffected by ground water. Even when measured ground-water levels were highest on June 26, 2001 (fig. 1-9) these wetlands are predicted to be unaffected by ground water. Any water present in these wetlands during this time most likely came from direct rainfall or surface runoff.

Ground-water levels were continuously measured in Well 2 from November 29, 2000 until April 28, 2001. Ground-water levels in Well 2 and Missouri River stage at mile 216.5 are shown in figure 1-10. Ground-water levels at Well 2 respond to changes in river stage within a few days and follow the general river stage trend. For example, the river peaked at 182.67 m on March 24, 2001 and ground-water level in Well 2 peaked 6 days later at 181.77 m on March 30, 2001. Ground-water levels in areas located closer to the river respond more quickly to river stage changes than in areas located farther from the river. This has important implications for wetlands located along the eastern valley wall. Although ground-water levels may respond more slowly to river stage changes in the area along the eastern valley wall, once ground-water levels rise they will not decrease quickly. Therefore, a seasonal cycle of high river stage in spring and early summer will result in wetland stage increases caused by increases in ground-water levels. Lower river stages in fall and winter will result in drainage of the wetlands as ground-water levels decrease.

Conclusions

The results of this study illustrate the diversity of wetland types and recharge pathways on the Missouri River valley bottom. As indicated in the following chapters, the hydrologic variations seen in Lisbon Bottom wetlands translate to distinct limnological and biological characteristics. Recognition of the hydrologic basis for differentiation of Missouri River wetlands should be useful to achieve the most cost-effective land management.

Schmudde (1963) recognized and articulated the fundamental truth that bottom lands naturally flood from downstream to upstream, that downstream areas flood more frequently than upstream areas, and that the effect is more pronounced on loop bottoms with flood-plain slopes that are substantially greater than the slope of the main channel. This study adds to these observations by quantifying the difference in flood frequency: the downstream margin of Lisbon Bottom is subject to flooding on average 11 times per year whereas the upstream margin floods on average only 3 times per year. In addition, upstream marginal wetlands like Wetland 4 are subject to rapid sedimentation as a result of overbank flows with high sediment concentrations (see Chapter 2). High sedimentation rates would contribute to short lives or high maintenance costs for wetlands developed on the upstream margin of river bottoms like Lisbon.

Wetlands 4 and 26 on the upstream and downstream margins of Lisbon Bottom owe their existence to intense scouring associated with failures of man-made levees, and are therefore not examples of natural wetlands that would occur in non-engineered river systems. Natural wetlands at upstream margins would more likely be associated with crevasses interspersed with sandy natural levees. Although also formed by scours associated with breaching of topographic barriers, natural crevasses would likely be shallower and more elongate than Wetland 4. Natural wetlands in downstream margins are more likely to be associated with valley-wall streams, and therefore be shallower and more extensive than Wetland 26. Although altered by the levee along Buster Branch, the land adjacent to the confluence of Buster Branch with the main channel would be an example of such a natural, downstream-margin wetland (fig. 1-4). Because natural wetlands would likely be shallow and broad, they would not have the persistence of water levels observed in Wetlands 4 and 26 at Lisbon Bottom. Nevertheless, scour wetlands are now common along the Missouri River as a result of levee breaks from the 1995, 1993 and previous floods (Galat and others, 1997).

In addition to recharge from the main channel, upstream and downstream marginal wetlands showed the effects of local rainfall and of ground-water connections to the main channel. In general, main-channel floods had greater and more persistent effects on water levels in these wetlands, but local rainfall events contributed substantially to recharge. Persistence of water levels after large events may be due to the ability of big floods to recharge the entire valley-bottom water-table aquifer.

Interior wetlands like Wetlands 10 and 11 are also subject to recharge from surface water when the main channel floods overbank. However, these wetlands differ markedly in having highly variable water-surface levels that are strongly determined by connections to valley-wall tributaries. Wetland 10, in particular, displayed high frequency variation in water-surface elevations as a result of tributary flows from Lay Creek. High-frequency variations in this shallow wetland basin may result in highly variable soil moisture conditions and therefore a distinct hydrologic disturbance regime.

Ground-water level results indicate the broad, time-averaged relations between ground-water and wetland recharge. Three prominent patterns are evident in the potentiometric contours presented in figure 1-9. During dry periods, when flow in the chute is relatively low, potentiometric contours embay in the upstream direction, indicating that ground water is discharging from the alluvial aquifer to the chute. Most of the measurement dates show this pattern; only August 11, 1999, October 7, 1999, May 14, 2001, July 26, 2001, and October 3, 2001 deviate from the pattern. The deviating dates show a second prominent pattern, with potentiometric contours embaying in the downstream direction, indicating recharge of the alluvial aquifer from the chute. August 11, 1999 (fig. 1-9) is a strong example of this pattern; this measurement was made during or shortly after a small rise on the main channel. Similarly, the October 3, 2001 measurement shows the same pattern was made shortly after a small rise (fig. 1-9). These two patterns indicate one of the prominent functions of side-channel chutes on the Missouri River. They contribute to both recharge and discharge of the alluvial aquifer, and presumably increase variability of water levels in wetlands adjacent to the chute.

The third prominent pattern in the potentiometric contours is the increased ground-water levels adjacent to the valley wall on the east side of Lisbon Bottom. This may be caused in part by lowering of the ground-water table toward the chute, and in part by recharge by valley-wall tributaries. Coupled with lower ground-surface elevations along the valley wall, this relation creates persistently high water-surface elevations

and wetland recharge along the valley-wall margin. The management implication of this observation is that, although these wetlands are located at greater distances from the main channel, they are naturally wetter for longer times during the year.

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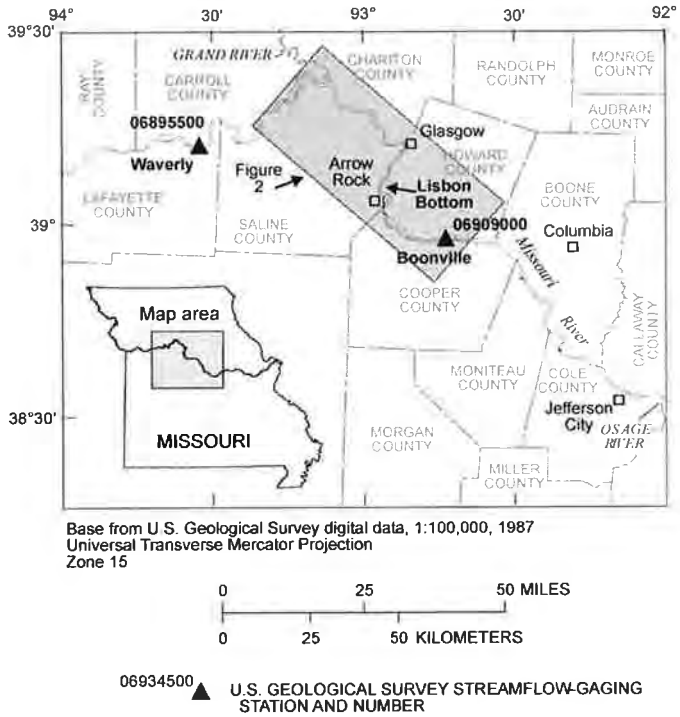
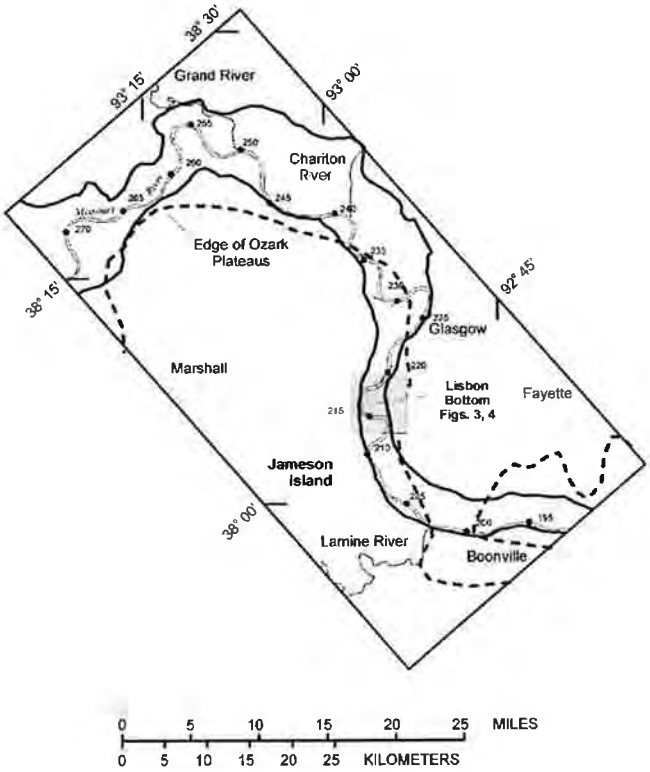


Figure 1-1. Location of Lisbon Bottom and nearby stream gages on the Lower Missouri River.



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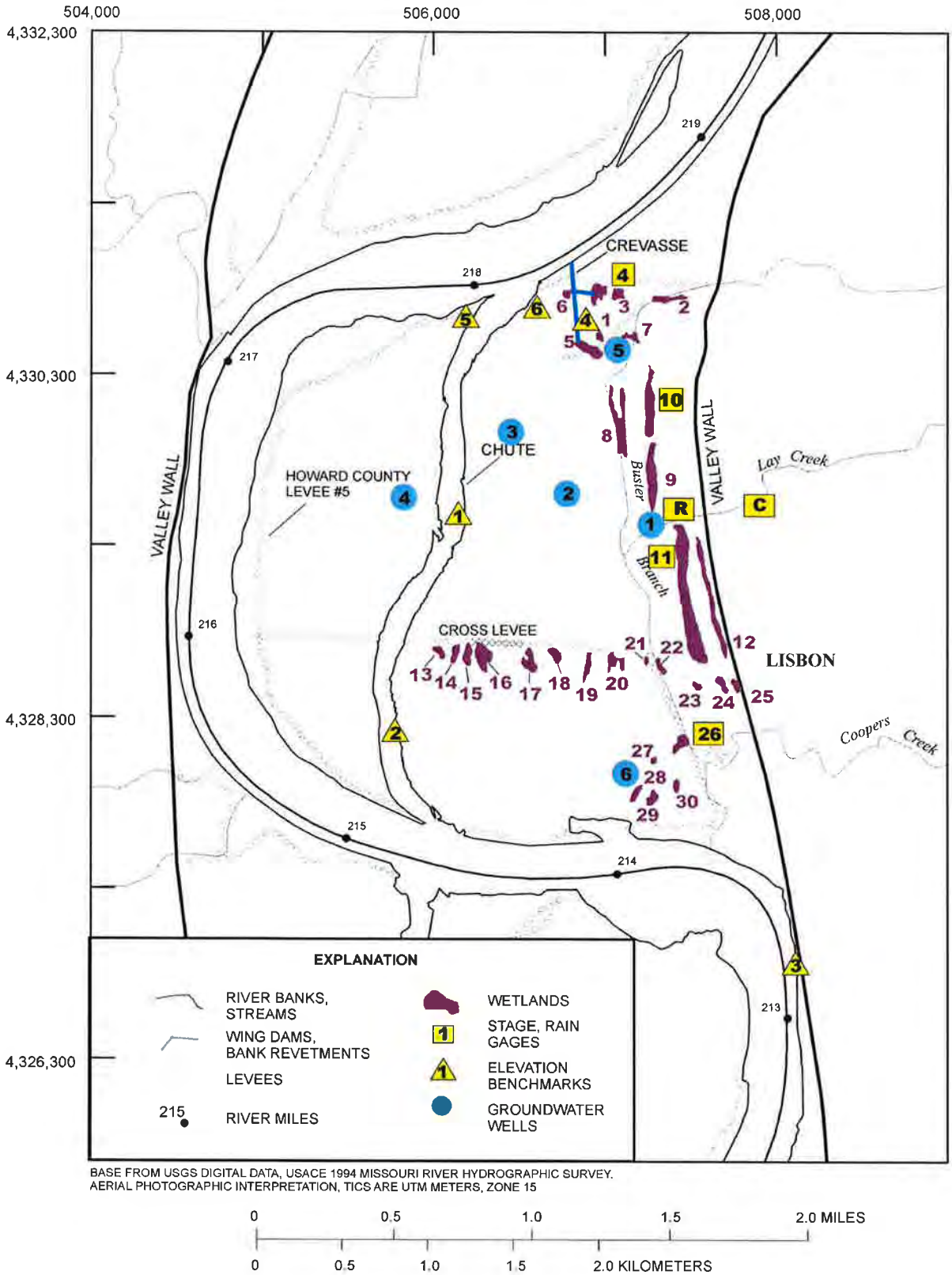

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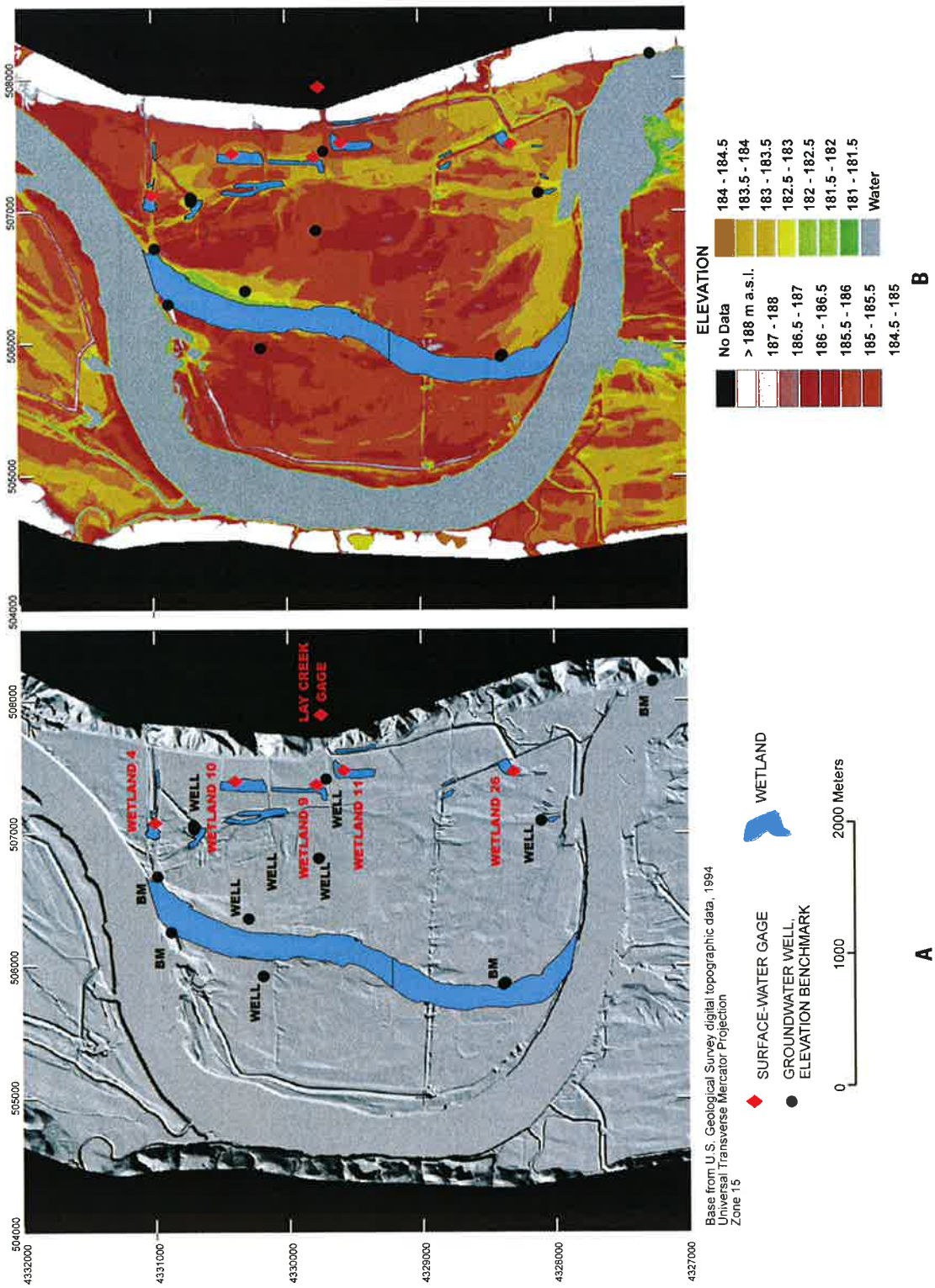

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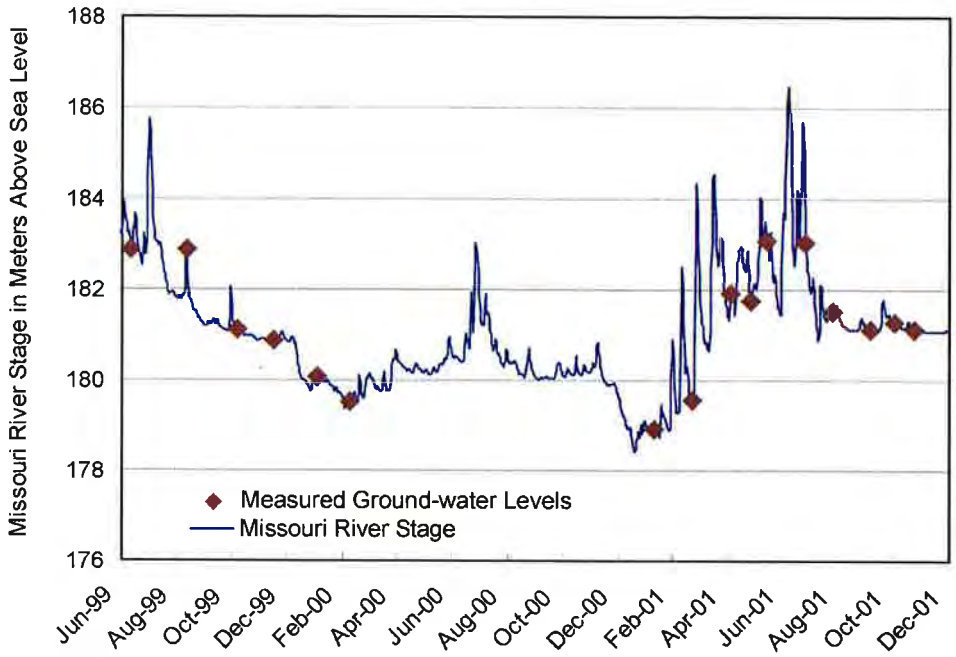

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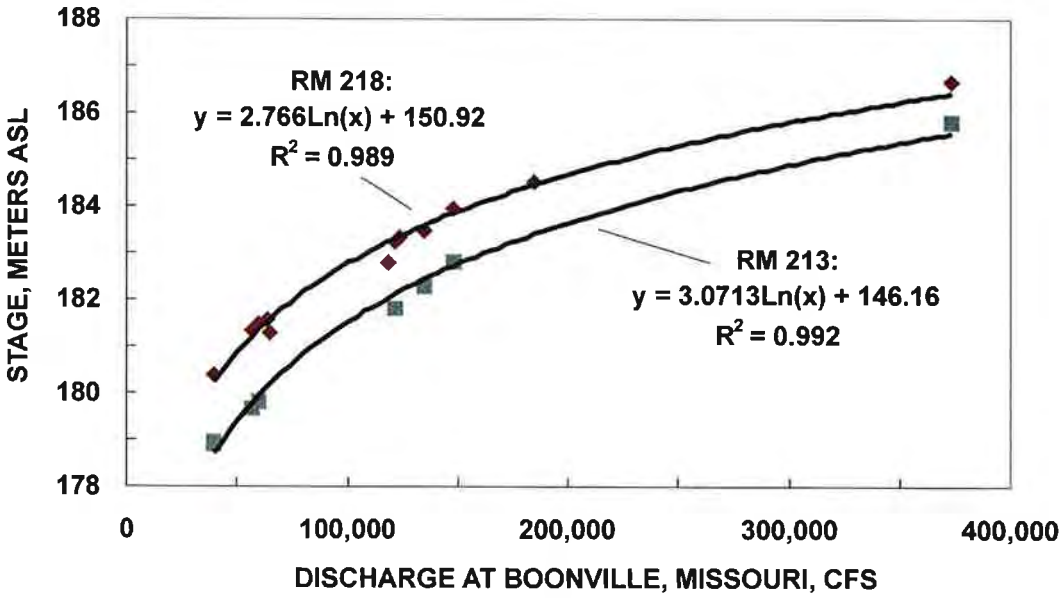

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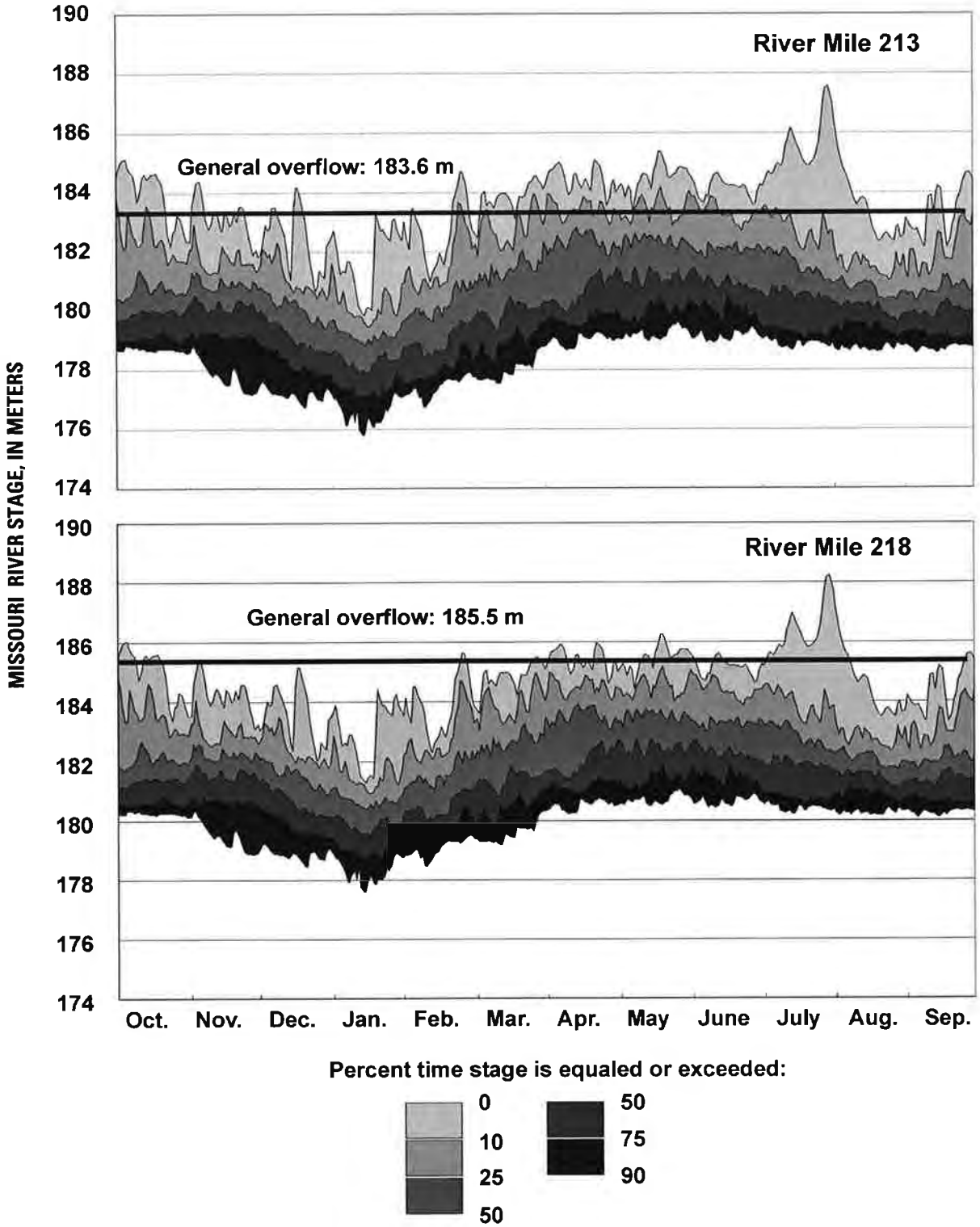

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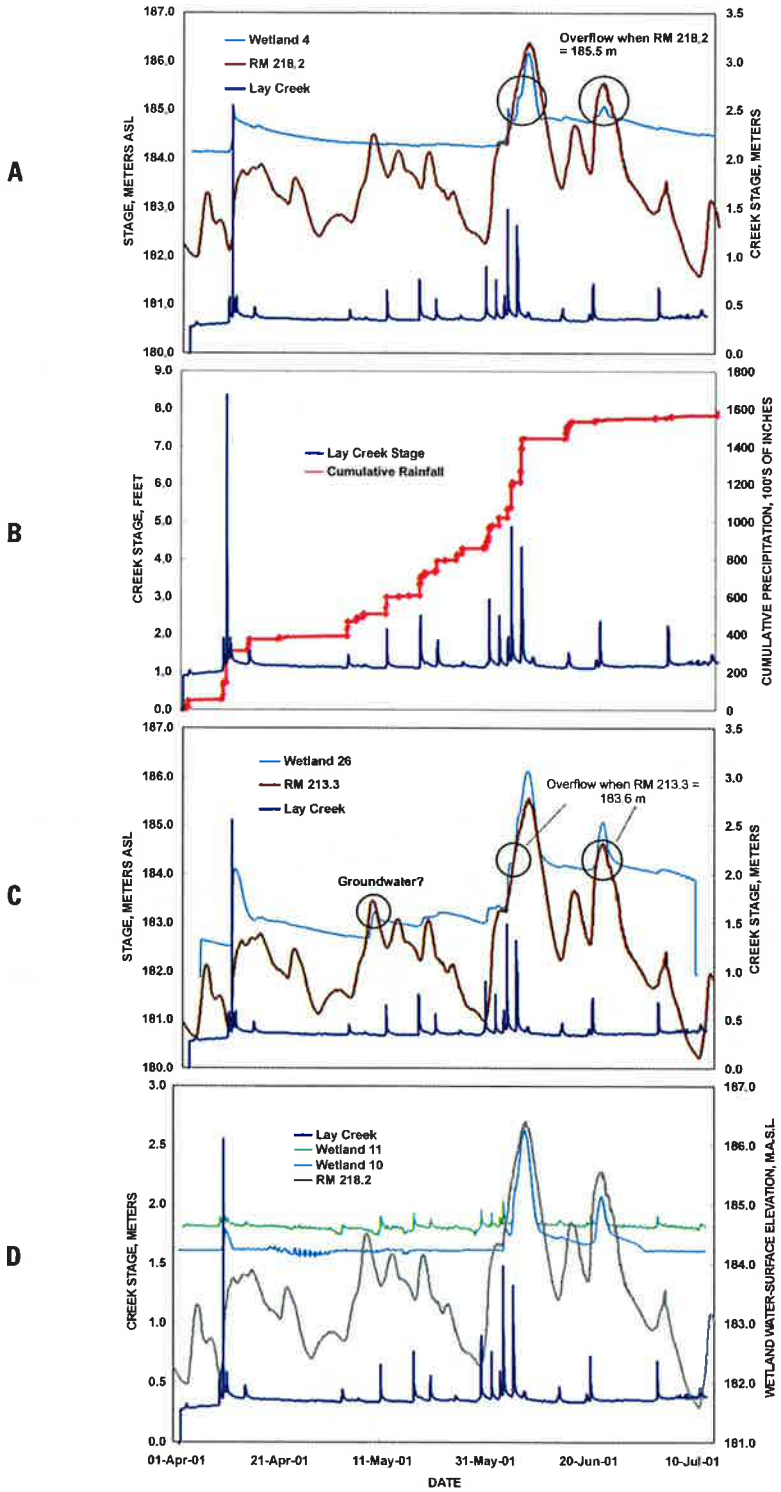

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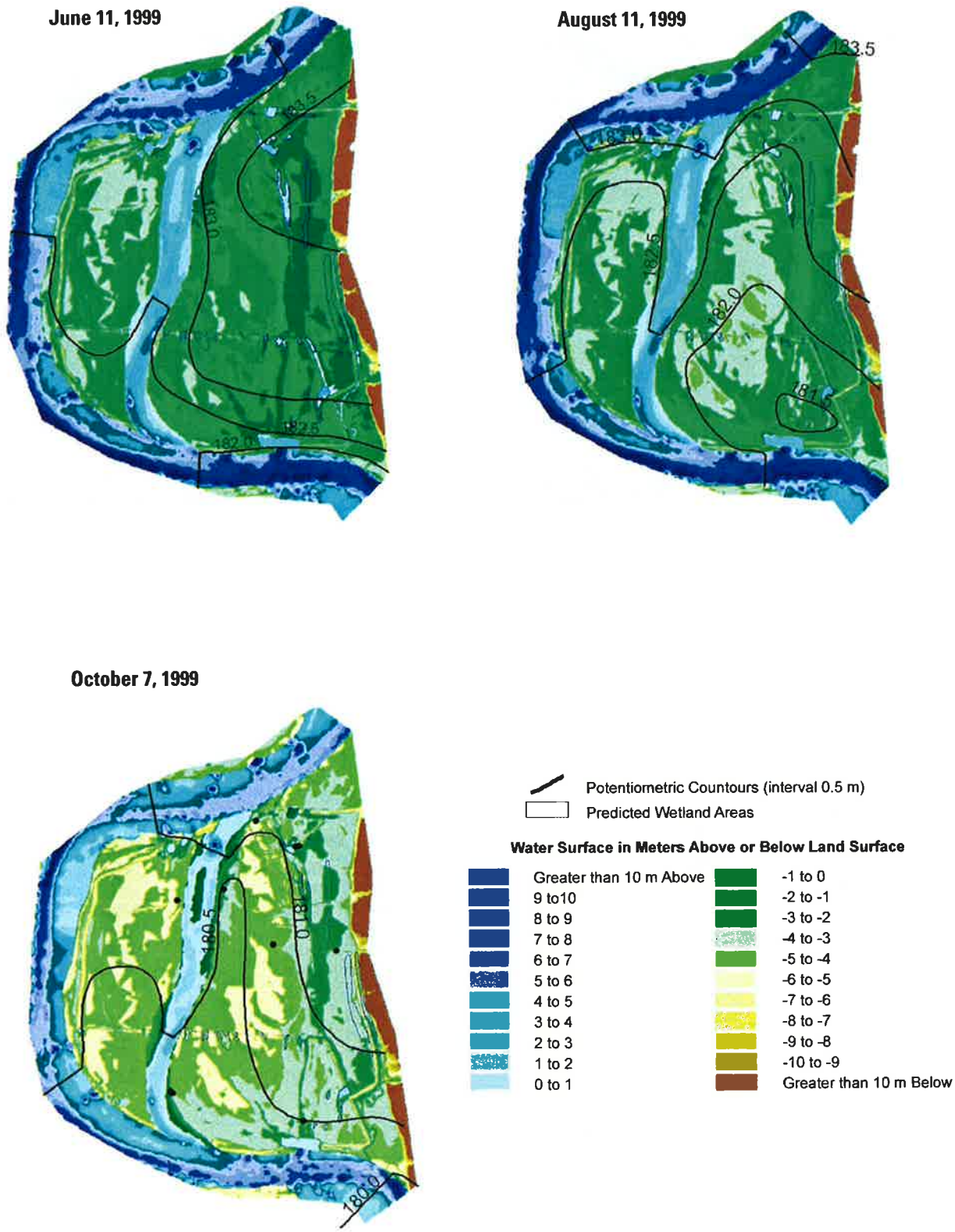


Figure 1-9. Potentiometric contours and distance of water surface above/below land surface on dates indicated.

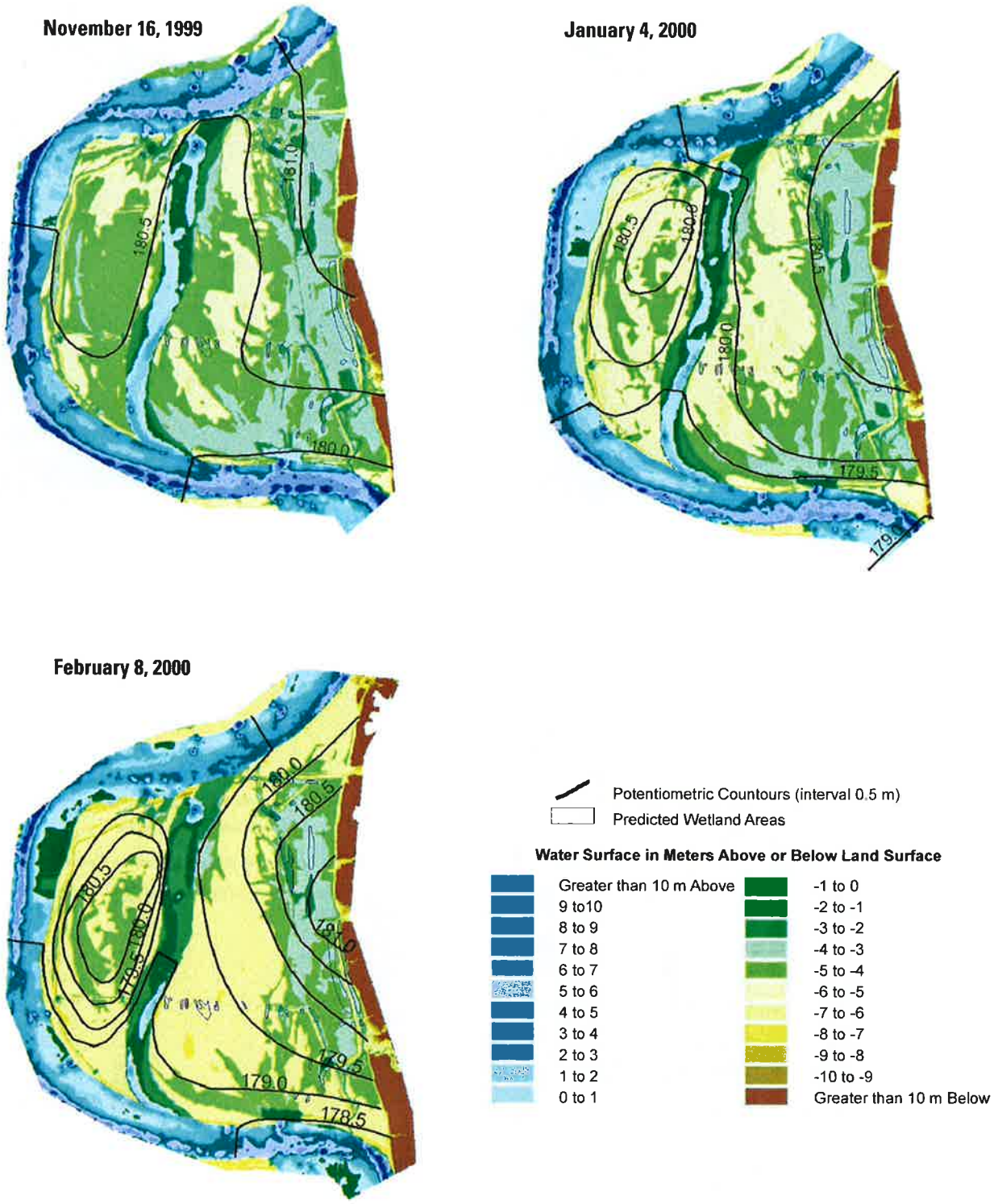


Figure 1-9. Potentiometric contours and distance of water surface above/below land surface on dates indicated–Continued.

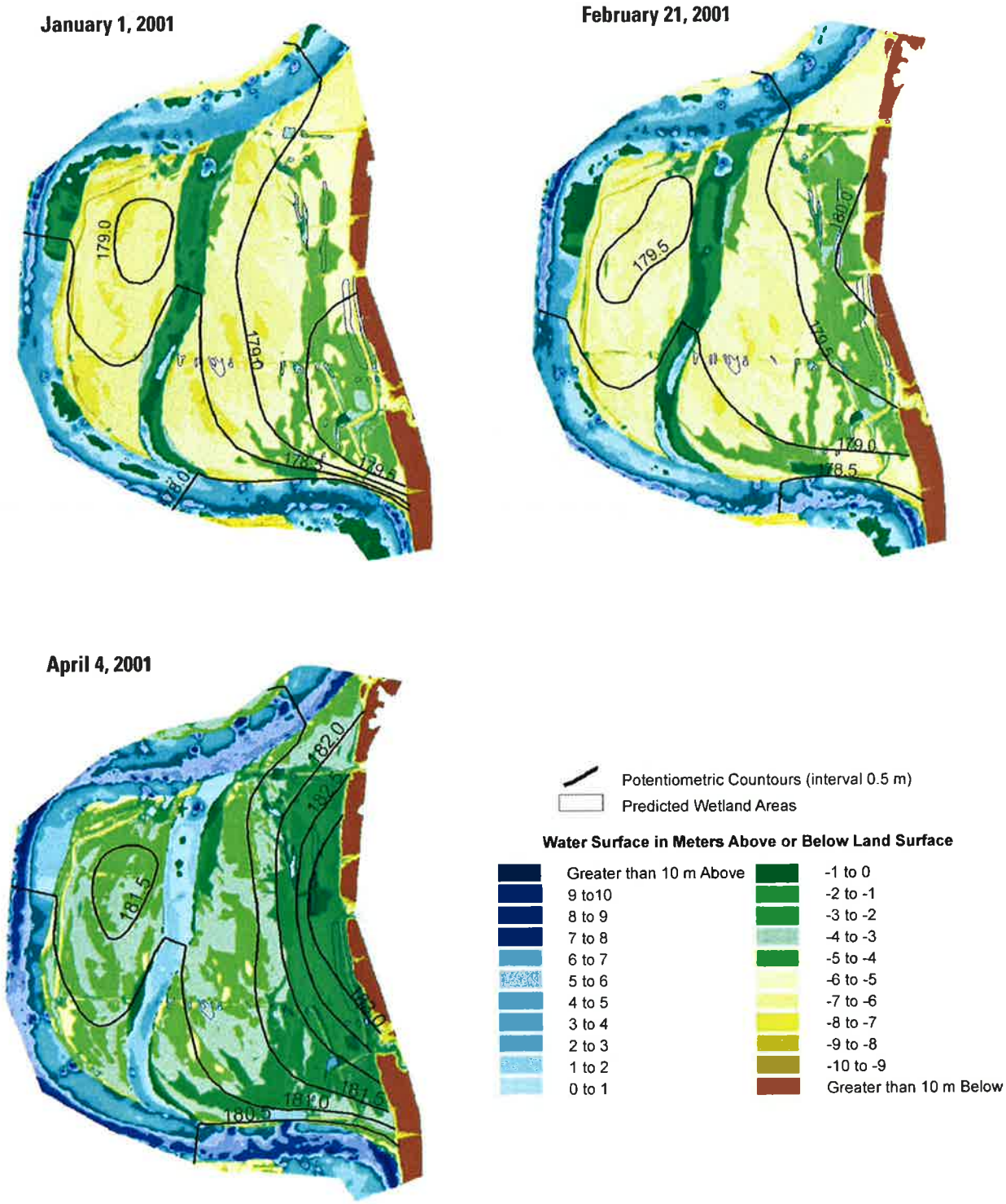


Figure 1-9 . Potentiometric contours and distance of water surface above/below land surface on dates indicated–Continued.

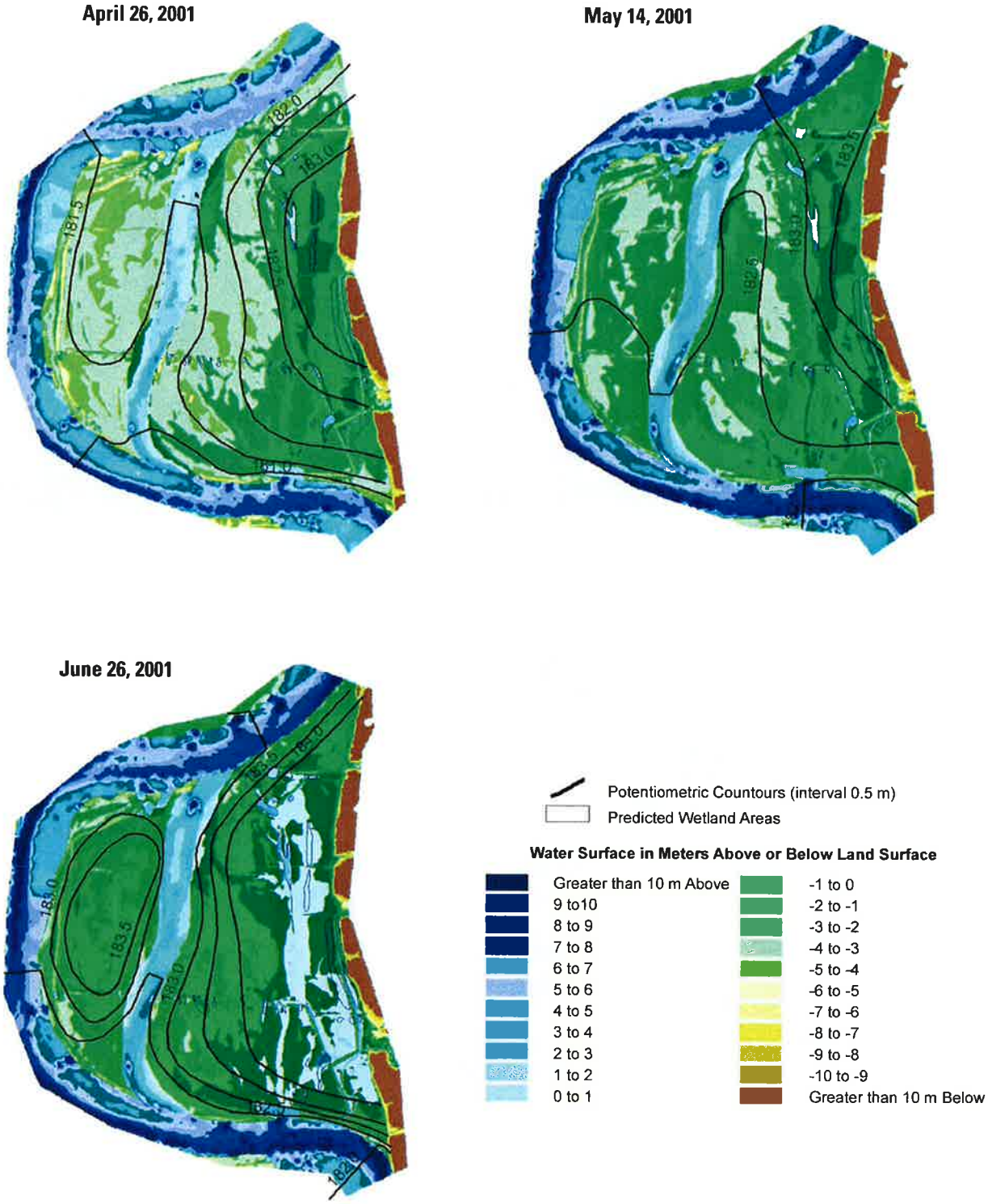


Figure 1-9. Potentiometric contours and distance of water surface above/below land surface on dates indicated–Continued.

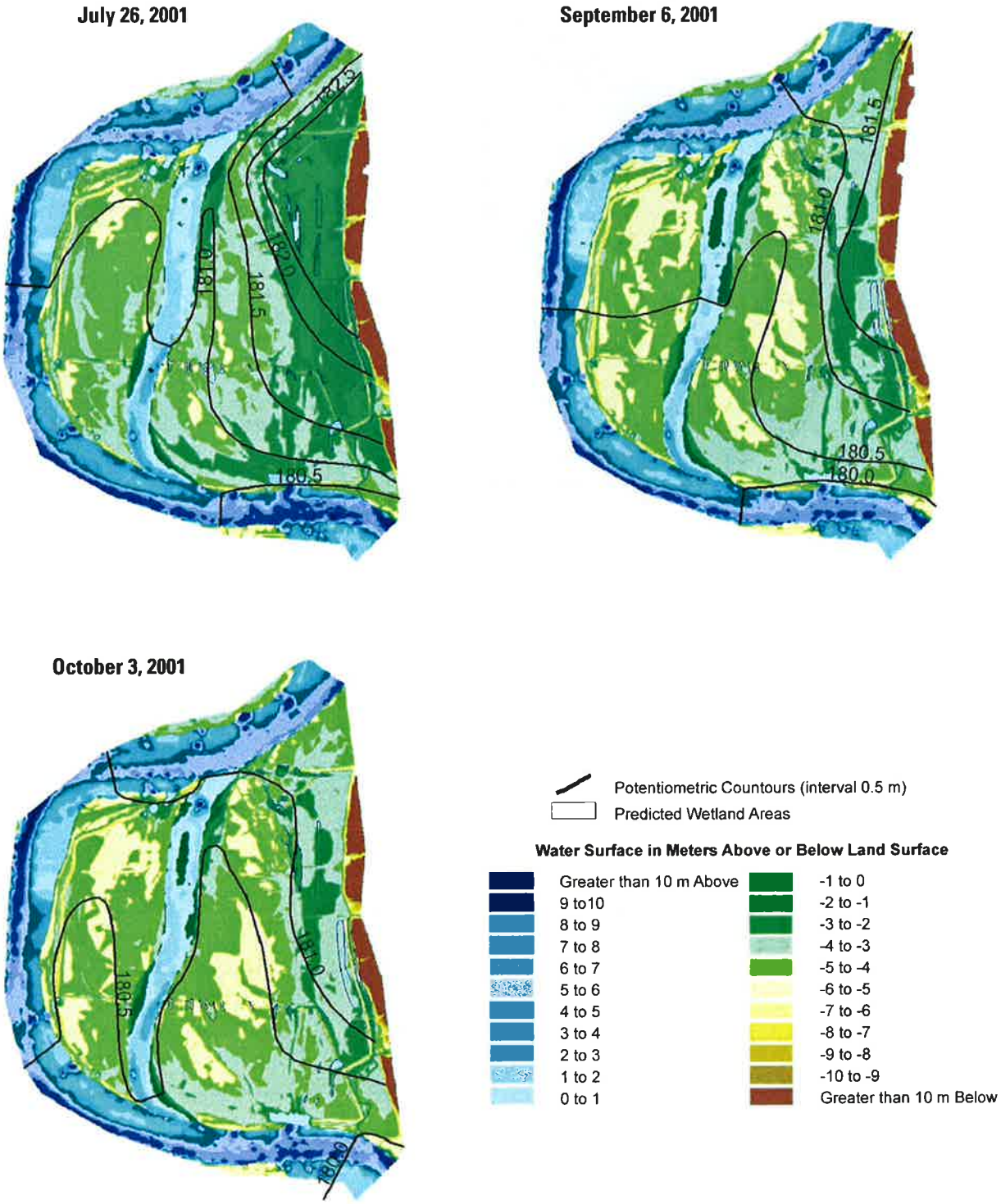
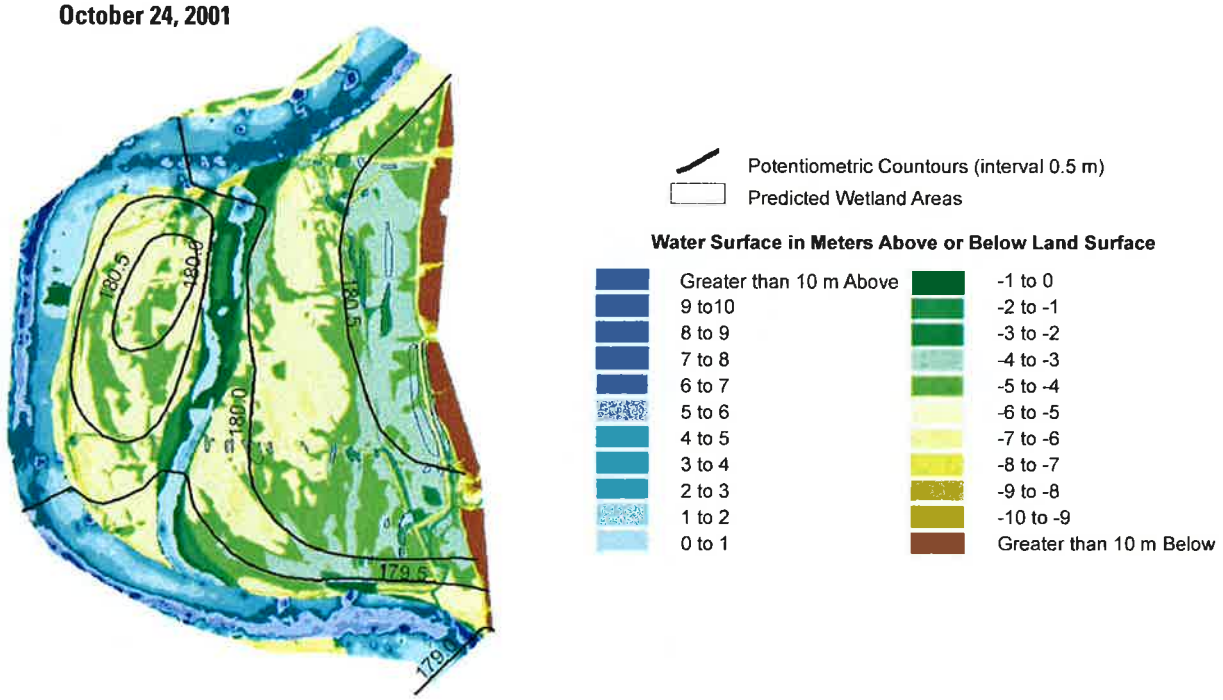


Figure 1-9. Potentiometric contours and distance of water surface above/below land surface on dates indicated–Continued.



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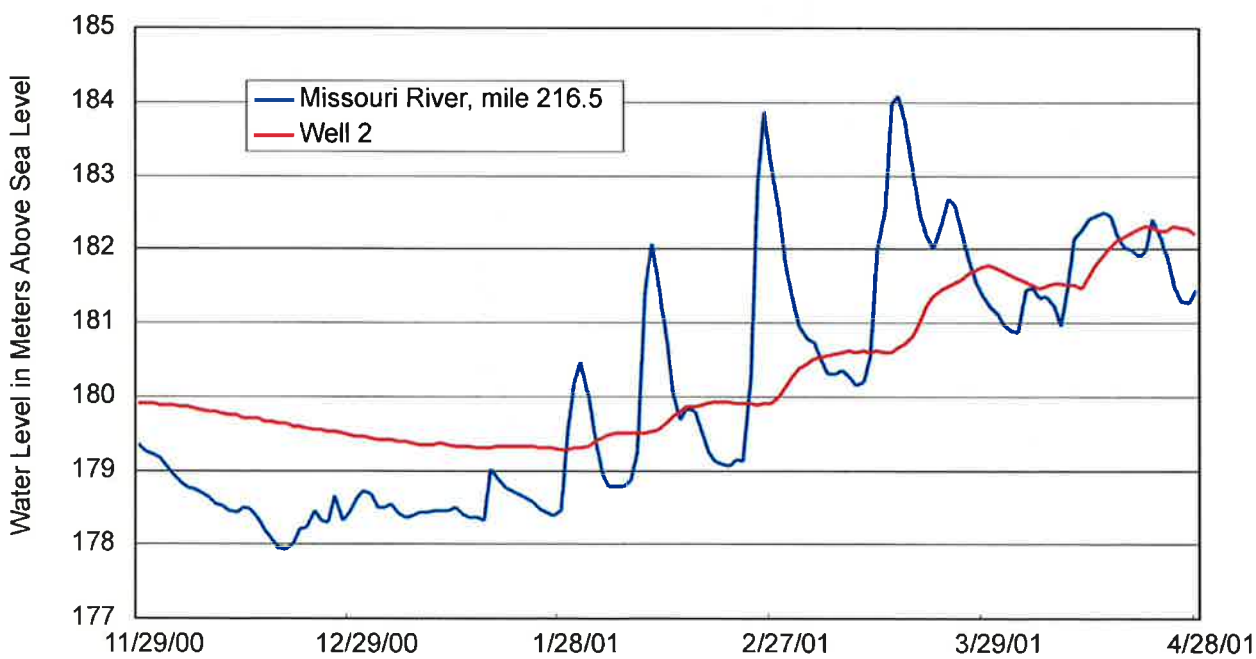

Figure 1-10. Missouri River stage at RM 216.5 and Well 2 groundwater level, November 29, 2000 to April 28, 2001.

Chapter 2. Limnology of Lisbon Bottom Wetlands

Duane C. Chapman, James F. Fairchild, and Ellen A. Ehrhardt

Abstract

In this study we examined the limnology of a continuum of wetland types at Lisbon Bottom in order to examine the physical, chemical, and biological characteristics of wetlands in relation to timing, source, and duration of flooding. We examined the relationship between inundation and water quality of thirteen Lisbon Bottom wetlands with varying degrees of permanence and water sources from late March through July 1999. Temperature and water quality of the Missouri River and of the Lisbon Chute were also measured. Growth of fathead minnows (*Pimephales promelas*) in cages was measured as an estimate of secondary productivity.

We found that nutrient availability, chlorophyll *a* concentrations, and fathead minnow growth were higher in wetlands that were influenced by the river than in wetlands that were mostly stream-influenced. Aquatic macrophytes in stream-influenced wetlands removed available nutrients from the water column after the water warmed in late April, resulting in low phytoplankton productivity and thus lower zooplankton productivity. River-influenced wetlands received large inputs of nutrients from the river during floods. Chlorophyll *a* concentrations increased in response to the nutrient inputs as soon as turbidity from the flood event diminished. In contrast to the stream-influenced wetlands, productivity in the river-influenced wetlands is likely driven by phytoplankton rather than macrophytes. Although moist-soil and inundation-tolerant vegetation such as cocklebur, cottonwoods and willows were often inundated by high water in the wetlands, obligate aquatic macrophytes were generally not present in the river-influenced wetlands because of high turbidity, sedimentation, water depth fluctuations, and sometimes short duration of inundation. Nutrient concentrations in the river-influenced wetlands peaked after floods, but nitrogen dissipated rapidly after flooding subsided.

Aquatic habitat conditions were very dynamic. Flooding events modified the topography of the wetlands during this study and between earlier studies and this one. Wetlands that were backflooded by the river were connected to the river more often than topflooded scours, but with less energy. Temperature stratification in deep topflooded scours was completely destroyed by flood events, but this did not happen in the deep backflooding scour. Dissolved oxygen concentrations in the hypolimnia of scours rapidly decreased after stratification.

Introduction

Lisbon Bottom, a unit of the Big Muddy National Fish and Wildlife Refuge, is located within a sharp bend of the Lower Missouri River south of Glasgow, Missouri. The flood of 1993 breached agricultural levees around Lisbon Bottom, thereby re-establishing a dynamic connection to the Missouri River. Repeated flooding of the bottom between 1993 and 1999 changed this previously agricultural area into a diverse ecosystem, with

unvegetated sandy areas, large areas dominated by cocklebur, dense stands of young cottonwoods and willow, and a variety of temporary and permanent wetlands.

The hydrology of Lisbon Bottom is extremely dynamic and is dependent on the sources and timing of water in the flood plain (see Chapter 1). The various sources of water entering the flood plain (river, groundwater, rainfall, and tributary inflow) vary widely in water quality characteristics; however, there have been relatively few studies to examine the limnology and water quality in the Lower Missouri River flood plain. Much of the existing data for the Lower River is contained within the databases of the U.S. Geological Survey's National Stream Quality Accounting Network (NASQAN) Program. However, this data is measured only on a monthly basis at a few, fixed sites on the river and critical water quality parameters related to aquatic structure and function (for example, particulate organic carbon and chlorophyll) are not measured (Blevins and Fairchild, 2001). Heitmeyer and others (1991) examined the water quality and hydrologic relationships among a continuum of wetland types in a managed wetland complex and found distinct differences in wetland type depending on depth and vegetative characteristics. Knowlton and Jones (1997) found that the river had a strong influence on scour holes connected to the river but that non-connected scours developed a separate limnological trajectory due to *in situ* biological processes that served to reduce concentrations of dissolved and total nutrients. Knowlton and Jones (2000) examined the suspended solids, nutrients, and chlorophyll of the Lower Missouri River over a 5-yr period and found strong, seasonal relationships in these components due to hydrologic and seasonal variation.

In this study we examined the limnology of a continuum of wetland types at Lisbon Bottom in order to examine the physical, chemical, and biological characteristics of wetlands in relation to timing, source, and duration of flooding. These data are provided as a basis for an understanding of the complexity of wetland structure and function in a dynamic ecosystem such as the Lower Missouri River. Such knowledge is essential to biologists involved in purchasing and managing habitats in the Lower Missouri River ecosystem.

Methods

Limnological studies were conducted in 13 wetlands from late March through July, 1999. A description of wetland characteristics (type, description, primary water source, and relative degree of permanence) is provided in Table 2-1. Locations of the wetlands are depicted in figure 2-1.

Wetland Water Depth and Periods of Inundation

Water elevations in individual wetlands were determined empirically using wooden staff gages installed early in the study. Staff gages were installed in the deepest portion of the shallow wetlands. In the deeper scour wetlands (also known as evorsions or blew holes) the gages were installed where they would be legible from the shore. Water elevation (or presence-absence of water) was recorded from the staff gages on each visit to a wetland, regardless of the purpose of the visit (bird, fish, invertebrate or limnology portions of the study). These data were later converted to water-surface elevations by surveying in the gages relative to established elevation benchmarks. Very high water levels impeded collection of staff gage data, because of inaccessibility to walkers or, in the case of Wetland 26, complete inundation of the staff gage by several feet of

water. Staff gage data was collected at most wetlands by canoe during the flood on April 30. These data should approximate the highest water levels experienced during the study. Beavers repeatedly destroyed the staff gages in some wetlands, resulting in some periods without water elevation data. Wooden staff gages were eventually replaced with metal gages in those wetlands, which eliminated this problem.

Water Quality

Water samples were taken twice weekly, from 3/30/99 to 6/15/99, for analysis of pH, dissolved oxygen (DO), turbidity, conductivity, hardness, alkalinity, chlorophyll *a*, particulate organic carbon (POC), total nitrogen and total phosphorus. Sampling was performed as a part of the bird observation portion of this study; the observers sampled water after the allotted observation period of each wetland. Thus, samples were taken during the morning, but the timing of sampling varied depending on when the observer completed the observation period and moved to the next wetland.

Duplicate 250 mL samples were taken by submersing the sample bottles in the wetland and filling them to the top. Air space in the container was eliminated or minimized. Though many of the samples were taken by wading, care was taken to acquire the sample without disturbing sediment and artificially increasing turbidity values. The temperature of the wetland was taken using a field thermometer and recorded on field data sheets. The bottles were iced and transported to CERC. One sample was used for DO, pH, turbidity, conductivity, hardness, and alkalinity analysis on the day of sample collection. A portion of the other duplicate water sample was filtered and extracted on the day of collection for chlorophyll analysis. The remainder of that bottle was frozen and later analyzed for nutrients.

Dissolved oxygen was determined with a YSI® model 57 dissolved oxygen meter and conductivity was determined using a YSI® S-C-T meter. Orion® EA 940 meters with glass gel electrodes were used to measure pH and alkalinity. Total nitrogen and total phosphorus were analyzed using a Technicon® Autoanalyzer. Particulate organic carbon was measured using a Coulometrics Model 5020 Carbon Analzyer. Chlorophyll *a* was determined by acetone extraction and use of a Turner Designs® 10-AU Flourometer.

In addition, *in situ* temperature loggers (Onset Corporation® tidbit loggers) were installed in selected wetlands in order to measure temporal changes in temperature and any subsequent stratification. Loggers were suspended 10 cm below a float, which was anchored in the deepest portion of the wetland. The float was attached to the anchor with sufficient cord that the float was not submerged during flooding. In the deepest scour wetlands (Wetlands 4, 5, and 26) a second logger was attached just above the anchor to record the temperature at the bottom. Additionally, three loggers were installed in the chute (one near the upstream end of the chute, one near the center and one just below the grade control structure near the lower end of the chute) and one logger was installed in the Missouri River on the outside bend of the river southeast of Lisbon Bottom.

Temperature and dissolved oxygen profiles (half-meter intervals) were taken weekly using a boat at two points in each of the deep scours using a YSI® model 57 temperature/DO meter.

A two-tailed Student's *t* test was used to compare water quality parameters between stream-influenced and river-influenced wetlands.

Fathead Minnow Growth Study

Survival and growth of fathead minnows were studied in the thirteen wetlands as an integrator of the combined influence of limnological characteristics (for example, temperature, dissolved oxygen, nutrients, and chlorophyll). Studies were conducted in small polyethylene cages that were deployed over a 20-d period (5/27 through 6/16). Fathead minnows were acquired from Aquatic Biosystems, Inc. (Fort Collins, CO) to assure low variance in starting weight. Initial fish weight was determined by taking the mean of 20 fish (29.10 mg dry weight; 144.1 mg wet weight). Both dry and wet weights were determined, but dry weights were used for the analysis because the weight of water on the surface of the fish can bias measurements of small fish. Growth in this study was defined as mean weight of surviving fish at the end of the study minus the mean starting weight.

Two 15 x 15 x 15 cm cages were installed per wetland, with 5 fish per cage. The cages were attached to the bottom side of a wooden float, so that they were completely submerged but just below the surface at all times, and at least partially shaded. Water temperature at the cages was logged every half hour using tidbit loggers (Onset® Corporation). Fish were transported using plastic bags with oxygen and installed in the cages on site. During the fathead minnow study, dissolved oxygen was measured weekly in the morning at the cages using a YSI® model 57 dissolved oxygen meter.

A two-tailed Student's *t* test was used to compare fish growth and water quality parameters between stream- and river-influenced wetlands, and between topflooding and backflooding wetlands.

Results and Discussion

Wetland Water Depth and Periods of Inundation

As is normal for spring and early summer on the Missouri River, river stage varied greatly during the period of the study. Portions of Lisbon Bottom that were completely dry for most of the period of the study were occasionally inundated with several feet of water (fig. 2-2A and 2-2B). Water surface elevation of selected wetlands by date is displayed in figure 2-3. With the exception of measurements made by canoe on 4/30/99, these data do not capture the periods of highest flooding from the river, but the figures show that most wetlands were strongly influenced by flooding from the river. Some of the wetlands were dry until they were filled by rain in early April (fig 2-2B), but almost all wetlands showed marked increase in surface elevation after the flood on April 16. The exceptions to this rule were Wetland 11 and Wetland 12. Wetland 11 had little relief and thus retained little water after the flood, although continuous flow from Lay Creek kept the wetland hydrated. Wetland 12 was separated from the flood plain by a secondary levee and thus was never flooded. Wetlands 4 and 5 were strongly connected to the river on 4/16/02 via a crevasse that passes through the upstream levee when river stage at river mile 218.2 exceeded 185.5 m above mean sea level (fig. 2-4). This river stage corresponds to approximately 25.5 ft on the Boonville gage. Stage data from Boonville is not exactly comparable to the Lisbon area because of the influence of the Lamine and other tributaries and because of differences in flood-plain morphology. Also, continued erosion of the crevasse or blockage by sediment and woody debris may have affected the water stage at which the crevasse provided water to these wetlands on later dates. However, during this study, Wetlands 4 and 5 did experience strong mixing at other times when the

Boonville river stage exceeded 25.5 ft. Other topflooding wetlands were supplied by water that passed through Wetlands 4 and 5 or entered directly from the chute (note strong water surface elevation changes on this date for all wetlands in fig 2-3). During this study there were three high-water events that caused topflooding. These events peaked on 4/18/99, 4/30/99 and 5/6/99 (fig. 2-4).

During the highest flooding periods, some water overflowed the chute and moved diagonally across the southern portion of the bottom at high velocity, returning to the river at the exit scour. Although it was impossible to view these high flow occurrences for safety reasons, the high flows were evidenced by at least 10 m head-cutting horizontally from the scour toward the chute (the head cut crossed an ATV trail that we had previously used as a path) and by cottonwood saplings of up to an inch in diameter that were pushed over and laying horizontally on the ground after the flood. In other portions of the bottom, water velocities during flooding were generally low, except possibly in the sections where Wetlands 4 and 5 were being fed by the river through the crevasse. During a comparable flood (283,000 cfs) in 1998, velocities recorded in the dense willows of the NE quadrant of Lisbon Bottom were consistently < 30 cm/sec (Robb Jacobson, personal communication). During floods, it was impossible to reach the chute or wetlands in the southern and western portions of the bottom because of safety concerns due to high and fast water. However sampling was continued as much as possible in the northern and eastern sections of the bottom, by wading and with the use of a canoe.

Wetland 26 was connected to the river via backflooding at about 20.5 ft (Boonville gage). Backflooding provided water to Wetland 22 and much of the southern bottom at river stages lower than that required for topflooding. Thus, these areas were directly connected to the river during more of this study than the topflooding wetlands.

Flooding events visibly modified the flood plain during the course of the study, and between this study and an earlier study in 1997. A single minor flood event deposited up to 10 cm of unconsolidated sediment over a large area of the southern bottom (fig. 2-5) and scouring occurred in other areas. For example, over the course of the study, Wetland 8 experienced scouring in the upper end and sedimentation in the lower end. Wetland 5 is significantly shallower than it was two years earlier as evidenced by approximately 20 cm of sedimentation around staff gages left in place from an earlier project. These fine sediments can provide a clay barrier to water movement (Chapter 1). Thus, while deposition of fine sediments tends to fill the deep scour wetlands and eventually limit their lifespan, it may in the short term provide water level stability and prevent drying of scours when the river stage and water table are low.

Staff gage data from Wetlands 11 and 12 in 1999 indicate that these wetlands were the most stable in surface elevation (fig 2-3). This contrasts with surface elevation data collected in 2000 using automatic recorders, which indicate a higher frequency and degree of surface elevation change in stream-fed valley-wall wetlands (Chapter 1). These differences may be more related to the methods than to differences in hydrology. Staff gage data were taken only when personnel were present at the site in 1999, and during very high flows, measurement was difficult. Thus the 1999 data may not have captured all of the variability of these wetlands. However, the 1999 staff gage data from these wetlands are significant in that minimum water surface elevations were extremely stable compared to the other wetlands. Stream flows during non-flooding periods maintained hydration and inundation of these wetlands much longer than other shallow wetlands elsewhere on the Lisbon

flood plain. This duration of inundation was important in providing appropriate conditions for aquatic macrophyte growth, and in providing habitat for invertebrates and waterfowl (Chapters 4 and 6).

Water Quality

Water qualities of the river and chute were almost identical. Turbidity and total nitrogen concentrations were higher (fig. 2-6 and fig. 2-7), and conductivity lower (fig 2-8), during periods of high flow. Hardness and alkalinity followed the pattern of conductivity. The temperature logger in the river was lost during the study, but data collected after the study was completed, indicate that the chute, with its higher surface area to volume ratio, is apparently more affected by changes in ambient temperature and solar radiation than the river, especially at the lower end. The river had much more thermal inertia and did not vary as much in temperature (fig. 2-9). It should be noted that the fig. 2-9 temperature data are taken from a period during which the river stage was comparable to the period prior to the spring flood, and that diurnal changes in temperature in the chute were not as pronounced during periods of higher flow (fig 2-10).

Based on the flooding regime experienced during the 1999 season, the studied wetlands were categorized loosely into topflooding, backflooding, stream- and runoff-influenced, and mixed influence groups based on the perceived major influences on each wetland. This classification is not perfect in its objectivity and distinctions are rough because all wetlands were at least somewhat mixed in influence and these influences cannot be exactly quantified by our data. However, we believe the distinctions based on our observations are adequate for making broad distinctions among these habitat types for the period examined. We considered Wetlands 4, 5, 8, 16, and 21 to be primarily topflooding wetlands, Wetlands 26 and 29 to be backflooding, and Wetlands 11 and 12 to be stream-influenced. The remainder of the wetlands were considered to have mixed or indeterminate influences. It should be noted that Wetlands 4, 5, and 8, which topflooded through a crevasse from the river probably flooded with a different frequency than Wetlands 16 and 21, which topflooded from the chute.

Of the 13 wetlands in which water quality was measured, nine (4, 5, 8, 9, 11, 12, 16, 22, and 26) held water at the beginning of the study. Wetlands 8 and 9 dried completely between the beginning of the study and the first flood.

The water quality of these wetlands was highly variable over time because of the degree of influence of flooding from the river and rain events on these small water bodies. For example, conductivities of river-influenced wetlands were lower than stream-influenced wetlands, until the river, which has a high conductivity, flooded the river-influenced wetlands on April 16 (fig. 2-11).

The overall mean pH of the wetlands was 8.2 (standard deviation 0.29). The pH varied from a low of 7.7 (Wetland 26 on the first day of the study) to 9.2 (Wetland 2 on 4/23/99). The most likely cause of pH changes and differences in these wetlands is photosynthesis. Since photosynthesis varies by time of day and degree of solar irradiance, the pH also changes similarly. Since it was impossible to sample all of the wetlands at once, or even at the same time for each wetland, pH data are useful primarily as an overall indicator.

Mean total phosphorus concentrations over the study were high, ranging from 181 to 408 μg/L in the topflooding and mixed influence wetlands, and over 300 μg/L in the backflooding and the stream-influenced wetlands and the river and chute (table 2-2). Mean N:P ratios for wetlands in this study ranged between 1.6 and

7.7. Nitrogen and phosphorus values measured in the river were similar to those measured by the U.S. Geological Survey in other years for this time period (Hauck and others, 1997).

Flooding by the river is clearly an important source of nutrients for these wetlands. The average total nitrogen concentration in the river and chute (2.7 mg/L) was nearly twice as high as that of the wetlands (1.5 mg/L). Following the 4/16/99 flood event, nitrogen concentrations in Wetland 5 changed from 2.0 mg/L to 4.3 mg/L and Wetland 4 concentrations changed from 1.2 mg/L to 1.9 mg/L. Nitrogen concentrations then rapidly decreased after the flood receded (fig. 2-12). This effect was stronger in shallow wetlands than deep wetlands. This rapid loss of nitrogen is frequently observed in shallow, lentic systems due to biological uptake by periphyton associated with sediments and detrital material (Knowlton and Jones, 1997; Flenniken, 2001). Denitrification, which occurs in carbon rich, anaerobic sediments, may have also been a factor. This study did not include an analysis of sediments and it is unclear whether sediments of shallow, temporarily flooded wetlands of Lisbon Bottom have the characteristics required for denitrification. Wetlands 11 and 12, which were the most strongly stream influenced, differed in nutrient dynamics from the rest of the wetlands. Wetland 12 did not receive a large pulse of nutrients during the flood and was the only site with submerged aquatic macrophytes; thus, nitrogen levels were much lower during the study. Wetland 11 was flooded briefly though deeply during the flood event (fig. 2-3), but since it did not retain water after the flood and it was strongly influenced by stream flows, by the time of our sampling, nitrogen concentrations were not markedly increased. Heavy rains in the watershed briefly raised nitrogen concentrations in the stream-influenced wetlands when water concentrations of nitrogen in the river-influenced wetlands were decreasing. Phosphorus concentrations were also generally higher in the river and chute than in the wetlands, although some wetlands would be considered very high in phosphorus.

POC values in this study (table 2-2) did not follow a clear pattern. In turbid systems, POC tends to correlate with turbidity, and in clear systems or systems with significant phytoplankton, POC will correlate with chlorophyll *a*. In these wetlands POC was likely influenced both by turbidity from the river and by subsequent algal production after river-born particulates precipitated. Because these two factors are negatively correlated, POC data were difficult to interpret.

Mean chlorophyll *a* concentrations, an important index of primary productivity, were high in river-influenced wetlands and low in stream-influenced wetlands and in the river and chute ($p < 0.01$; fig. 2-13). We expected turbidity to be highly correlated to nutrient concentrations in the wetlands, because suspended sediment should bear large quantities of nutrients. We found this only to be the case in the river samples ($r = 0.77$ for N and 0.73 for P correlations with river turbidity). This is probably because variables other than suspended sediment, such as phytoplankton, determine turbidity in the wetlands. Some wetlands, especially 10 and to a lesser extent 8, had generally lower chlorophyll *a* concentrations probably due to shading from trees.

Somewhat surprisingly, surface temperature did not vary according to wetland type, but all wetlands were warmer and had a higher daily variance in temperature than the river and chute (fig. 2-10).

Wetlands 4 and 5 were quick to stratify in the spring, with temperature and dissolved oxygen in the hypolimnia dropping quickly with depth (fig. 2-14). At times, the oxygenated epilimnion consisted of little more than the upper meter (fig. 2-14, 4/14/99 and 5/26/99). These wetlands were strongly influenced by flooding from the river, which destroyed the stratification (fig. 2-14, 4/30/99). However, stratification was

quickly reestablished after the flooding events. This relationship is also shown by the temperature logger data. During flooding events, the temperature of the loggers at the top and the bottom of the wetlands are equivalent, and these values separate quickly after the flood event ends (fig. 2-15). Rain events also influenced stratification. Mixing occurred in Wetland 4 on 5/17/99 at a lower river stage than that at which the crevasse would have transferred water to it. Local rains in excess of 2" that occurred on this date (from USGS rain gage, see Chapter 1) apparently provided enough inflow to mix Wetland 4. There is a small intermittent stream (Buster Branch) that enters Wetland 4 on the northeast corner. Wetland 4 can also receive runoff via channels from Wetlands 3 and 6. Water quality measurements on 5/18/99 show an increase in nitrogen concentration but a decrease in conductivity and hardness. This is additional evidence that mixing from runoff was responsible, because inflows from the river would likely have increased rather than decreased conductivity and hardness.

Wetland 26 differed from Wetlands 4 and 5 in its pattern of stratification. This scour wetland is larger and deeper and thus has more thermal inertia. It was connected to the river more often than Wetlands 4 and 5, but being primarily backflooded as opposed to topflooded, water inflows were lower in energy. Thus, Wetland 26 did not stratify as strongly early in the observation period but once stratification was established, it remained stratified during flooding events (fig. 2-14).

Most small, permanent, nutrient-rich, lentic wetlands in Missouri have a high density of submerged and/or emergent aquatic macrophytes. Wetland 12 had high densities of macrophytes, but permanent Wetlands 4, 5, 16 and 26 did not. The absence of vegetation in these river-influenced permanent wetlands is presumably a result of the turbidity, scouring, sedimentation and severe water level fluctuations caused by repeated flooding by the river. Therefore, primary productivity in the river-influenced permanent wetlands is a function of phytoplankton production, which increased during periods of low inorganic turbidity. Sequestration of nutrients by macrophytes in Wetland 12 probably limits phytoplankton productivity in these wetlands, and primary productivity in these wetlands is primarily a function of macrophyte production. These differences are important in terms of zooplankton growth and thus to birds and fish which consume the zooplankton.

Most temporary and ephemeral wetlands were unvegetated basins, surrounded on their margins by smartweed, cocklebur, willows and small cottonwoods, which are considered to be moist-soil vegetation (Fredrickson and Taylor, 1982). Wetlands 9, 10, 11 and 22 differed. When not in flood by the river, Wetland 11 was very shallow and at times had perceptible current from Lay Creek. Wetland 11 was braided and highly vegetated with emergent macrophytes, primarily rice cutgrass with some perennial smartweed and spirea as well as needlerush, duck potato, spikerush, and American bullrush. Wetland 22, which in years prior to the study had remained more moist due to a combination of runoff through Wetland 11 and repeated backflooding through Wetland 26, had cattails, sedges, and bulrushes at the margins at the beginning of the study. However, emergent aquatic vegetation in Wetland 22 died during 1999, which was dryer than previous years. Portions of Wetland 9 and most of Wetland 10 were covered with dense growths of young cottonwoods and some willows, ranging 2–5 cm in diameter at the base.

All water quality data from the wetlands are found in Korschgen and others (ArcView-based spatial decision support system for the Lisbon Bottom Unit of the Big Muddy National Fish and Wildlife Refuge, unpub. data, 2001).

Fathead minnow growth study

Survival of fathead minnows was high in all wetlands except 9 and 10, which dried during the exposure period, and in Wetland 11, where survival was 50%. There was no apparent relationship between fathead minnow growth and whether wetlands were backflooded or topflooded. Fathead minnow growth was higher in river-influenced wetlands than stream-influenced wetlands ($p = 0.015$; fig. 2-16). This is most likely a combination effect of increased productivity due to the nutrients provided by flooding, and because phytoplankton (as opposed to macrophyte) productivity dominated the river-influenced wetlands. This illustrates the difference in nutrient pathways between the stream-influenced and vegetated wetlands and the river-influenced and unvegetated wetlands as described above. Fathead minnows feed mainly on algae and on crustacean zooplankton (Pflieger, 1997). Although the stream-influenced wetlands had very high densities of macroinvertebrates (Chapter 4) the caged fish were apparently unable to take advantage of the invertebrates as a food source.

Chlorophyll *a* was higher in the river-influenced wetlands during the *in situ* growth study ($p = 0.018$; table 2-3; fig. 2-16), as well as overall during the period of the study (fig. 2-13). However, chlorophyll *a* and growth were not highly correlated across all wetlands ($r = 0.319$). This could be because fish in the river-influenced wetlands were growing at their maximum rate well below the maximum chlorophyll *a* concentration, or other factors could be the cause. POC values, which are influenced by phytoplankton concentrations as well as turbidity, were also somewhat lower in stream-influenced wetlands ($p = 0.056$; table 2-3). Zooplankton density was low (fig. 2-16, table 2-4) in the stream-influenced wetlands, which may have influenced fathead growth, but zooplankton density was also low in Wetlands 4 and 22, where the fathead minnows had high growth. However, zooplankton density was more highly correlated to growth ($r = 0.722$) than was chlorophyll *a* concentration ($r = 0.369$). It also should be noted that although zooplankton density was very low in stream-influenced Wetland 12 during the three weeks of the caged fish growth study, zooplankton density in that wetland was reasonably high on average for the study as a whole (table 3-1).

Mean temperature overall during the fathead minnow study (from loggers) in wetlands that did not dry was 25.0 °C. Maximum temperature in the wetlands ranged from 32.5 to 35.9 °C. The loggers were co-located with the cages, so these temperatures reflect the temperatures to which the fish were exposed. Fathead minnow growth was not correlated to temperature ($r = -0.186$) or maximum temperature ($r = 0.167$) during the study, and these were not significantly different between wetland types. Negative correlation of growth with maximum temperature would have been expected if heat stress was a factor in some wetlands, but this did not occur and in fact the wetland with the highest maximum temperature had the highest growth. Some positive correlation of growth with temperature was expected, but this also did not occur. However, it must be noted that an equipment malfunction resulted in the loss of most of the temperature logger data from Wetlands 2, 11, 12, and 22. Since growth in Wetlands 11 and 12 was lower than that of the other wetlands, we re-examined the data using temperature data only from the three days at the end of the study period, in which data existed for those wetlands. During this period mean temperature in the stream-influenced wetlands was similar to the river-influenced wetlands (25.9 vs 26.0 °C; $p = 0.95$). Thus, there is no evidence that temperature differences were responsible for the lower fathead minnow growth rates in the stream-influenced wetlands, but the data are

incomplete. Among only the river-influenced wetlands for which complete data was obtained, mean temperature was not correlated to growth (r = -0.123), but the mean temperature was not highly variable between wetlands.

Dissolved oxygen measured weekly in the morning at these wetlands was generally adequate for fathead minnows, with an overall average of 6.6 mg/L (fig. 2-17). However, on one date DO measured 2.4 mg/L at Wetland 5. All other measurements at Wetland 5 were over 7 mg/L. Overall survival at Wetland 5 was 80% (Fig. 2-16). Five mg/L DO is considered adequate for all fish (Piper and others, 1982) and fathead minnows are noted for their tolerance for low dissolved oxygen concentrations (Pflieger, 1997). The measured concentrations of dissolved oxygen, with the potential exception of the low value at Wetland 5, were not likely to be stressful. However, lower dissolved oxygen concentrations may have occurred earlier in the morning or on days when DO was not measured. Although an effort was made to sample these wetlands as early as possible in the morning, less accessible wetlands were sampled later in the day when photosynthesis had likely begun to raise oxygen concentrations. Because of this bias, we have not attempted to correlate measured dissolved oxygen concentration to growth.

Summary and Conclusions

The Missouri River is a primary source of nutrients for most of the wetlands. Nitrogen concentrations decreased rapidly either due to denitrification or uptake by terrestrial plants, especially in shallow wetlands. Wetlands that were river-influenced had much higher phytoplankton productivity than stream-influenced wetlands. Caged fish also grew faster in river-influenced wetlands. Zooplankton production in these wetlands was linked to phytoplankton concentrations that are in turn linked to nutrients from the river.

A crevasse through the levee at the northern end of Lisbon Bottom provided water and nutrients to wetlands that might otherwise have been inundated by the river for much less of the study or not at all. If continued river inundation of these northern wetlands is desired, the crevasse should be maintained and not plugged. There is a significant possibility that this crevasse could be plugged with river debris or it could possibly be plugged during levee maintenance operations. However, continued inundation by less than very large floods will allow continued sedimentation of Wetlands 4 and 5. The topflooding wetlands are less valuable to riverine fish (see Chapter 5) than wetlands on the downstream portion of the bottom, because they are connected to the river less often and for shorter periods, thus flooding periods are less likely to correspond with the time of year in which flood-plain spawners are actively spawning. Also, fish and their offspring spawned on the river bottom are more likely to be trapped in the topflooding wetlands because opportunities for egress to the river are more limited.

Topflooding Wetlands 4 and 5 stratified rapidly and had narrow oxic epilimnia, and hypolimnia that rapidly became anoxic. Stratification was destroyed during flood events and in Wetland 4 by runoff from a heavy rainfall, but was quickly reestablished. Wetland 26, a deeper, backflooding scour, did not completely lose stratification during flood events. Stratification patterns in these wetlands have important fishery implications (see Chapter 5). The river is continuing to modify these wetlands through sediment deposition in some areas and erosion of other areas. Further studies at this site will likely find shallow wetlands similar to

those that we studied, but the specific wetlands may disappear or change in morphology. The future of the deeper scours is unclear. Wetland 5 changed from a relatively deep "permanent" scour in 1997 to a very shallow basin by the end of this study. The exit scour S-14 at the southern edge of Lisbon Bottom (not examined in the limnology portion of this study) was a deep scour in 1996 (Tibbs and Galat, 1997) but was very shallow over most of its area by 1999, and almost entirely filled in by mid-summer of 2000. Sedimentation of the deep scours occurs during flood events. Strong flows from a very large flood would likely be required to maintain or recreate these deep wetlands.

The river and chute were very similar in water quality, except for temperature. The chute was more variable in temperature than the river because of less thermal inertia and higher surface area to volume ratio. The chute was attractive to a diverse species assemblage of fish (Louise Mauldin, fisheries biologist, USFWS, unpub. data) and the river just downstream from the chute was occupied at times by pallid sturgeon (Aaron Delonay, ecologist, personal communication). Temperature regime could be a factor in the attractiveness of the chute to these fishes.

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Table 2-1. List and description of Lisbon wetlands in which limnology and water quality was studied.

Wetland	Basin type/description	Primary water source	Permanence
2	Shallow basin, contains flooded willows and cottonwoods at high water stages	Mixed topflooding and stream influence	Temporary (4/16 > 6/15)
4	Deep scour	Topflooding, with some stream influence	Permanent
5	Scour, moderately deep, sedimenting rapidly	Topflooding	Persistent, considered permanent until it dried in the fall of 1999
8	Wide, shallow basin	Topflooding	Temporary (4/16 > 6/15)
9	Shallow basin, contains flooded willows and cottonwoods at high water stages	Mixed topflooding and stream influence	Ephemeral (first flooded 4/16, contained less than 6" water after 5/28)
10	Wide shallow basin, mostly wooded with young trees	Mixed topflooding and stream influence	Ephemeral (4-16 to 5-28)
11	Wide, very shallow marshy wetland, few trees but many wetland plants	Strongly stream influenced, at times with perceptible current, although flooded by the river during the flood.	Temporary (< 3/30 to > 6/15)
12	Moderately deep wetland formed by partial damming of an intermittent stream	Stream influenced, this wetland was never flooded by the river during the study	Persistent, considered permanent until it dried in August 1999
16	Deep scour	Topflooding from chute	Permanent
21	Moderately deep scour	Topflooding from chute	Temporary (4/16 > 6/15)
22	Shallow basin	Mixed backflooding, topflooding, and stream influences	Temporary
26	Very deep scour, steep sided	Backflooding from Cooper's Creek is primary connection to the river	Permanent
29	Narrow scour, usually less than one meter deep	Backflooding through a break in the levee is primary river connection	Temporary (4/16 > 6/15)

Table 2-2. Mean values of various water quality parameters in wetlands of Lisbon Bottom between late March and the end of July 1999. The apparent differences between the river and chute in some values (especially POC and total nitrogen) are the result of being unable to sample the river during high flows for safety reasons. River and chute water quality parameters were nearly identical on any given day when both measurements were taken.

	Wetland	Total Phosphorus (µg/L)	Total Nitrogen (mg/L)	Hardness mg/L	Alkalinity	POC[1]
Topflooding	4	212	0.665	131	286	357
	5	208	1.24	180	430	386
	8	241	1.24	179	433	667
	16	322	0.91	157	381	330
	21	408	0.87	205	496	468
Backflooding	26	316	0.97	160	387	382
	29	326	1.26	176	463	520
Mixed influence	2	181	1.30	162	347	526
	9	235	1.09	197	465	673
	10	320	1.66	175	401	394
	22	214	1.26	182	420	666
Stream influence	11	379	1.20	198	417	379
	12	309	0.44	165	351	283
River		358	1.94	232	629	612
Chute		386	2.82	239	609	854

[1]Particulate Organic Carbon

Table 2-3. Water quality in Lisbon Bottom wetlands during the 5/28 to 6/16 caged fathead minnow growth experiment. Values are means from each wetland. Wetlands 9 and 10, which dried during the experiment, are excluded.

Water source	Wetland	pH	Chlorophyll (μ/L)	Turbidity (ntu)	POC[1] (mg/L)	Temperature[2] °C
Topflooding	4	8.19	27.0	29.7	327	25.7
	5	7.95	52.3	42.8	356	25.8
	8	8.14	54.1	29.5	515	23.9
	16	8.51	36.8	22.5	392	25.1
	21	8.22	66.3	42.5	526	24.5
Backflooding	26	8.43	93.1	38.0	339	24.7
	29	8.04	64.5	29.5	450	25.7
Stream-influenced	11	8.00	31.1	42.0	302	25.6[3]
	12	7.96	13.1	23.3	242	26.3[3]
Mixed influences	2	8.22	54.3	36.0	644	20.5[3]
	22	8.11	69.1	43.3	728	26.2[3]

[1]particulate organic carbon
[2]from *in situ* temperature loggers
[3]data is mean of last three days of study, due to equipment malfunction

Table 2-4. Mean number of zooplankton organisms per liter in Lisbon Bottom wetlands during the fathead minnow growth study. Wetlands 9 and 10 are not included because they dried during the study. Rotifers may not have been adequately sampled because the mesh size was larger than that normally used for the capture of rotifers.

	Wetland	Cladocerans	Copepods	Rotifers	Total Organisms
Topflooding	4	15.6	8.7	4.7	28.9
	5	305.0	41.8	59.5	406.4
	8	553. 8	168. 9	170.3	892.9
	16	1293.1	849.6	5.3	2148.0
	21	114.2	60.7	0.0	175.0
Backflooding	26	165.8	519.6	0.7	686.0
	29	187.7	149.8	33.7	371.2
Mixed influence	2	221.6	96.1	178.9	496.7
	22	29.8	41.1	2.0	72.9
Stream influence	11	2.7	5.3	0.0	8.0
	12	26.2	21.6	1.1	48.9

Figure 2-1. Map of Lisbon Bottom with numbered wetlands where limnological measurements were made. *Background photo courtesy of U.S. Army Corps of Engineers, Kansas City, MO, March 2000.*

A

B

Figure 2-2. Two pictures taken in the same area (Wetland 22), showing normal and flooded appearances. The tree in the left of figure A is the tree to the left in figure B. Picture B was taken from a canoe. The cottonwood sapling protruding from the water in the foreground of figure B was approximately seven feet tall.

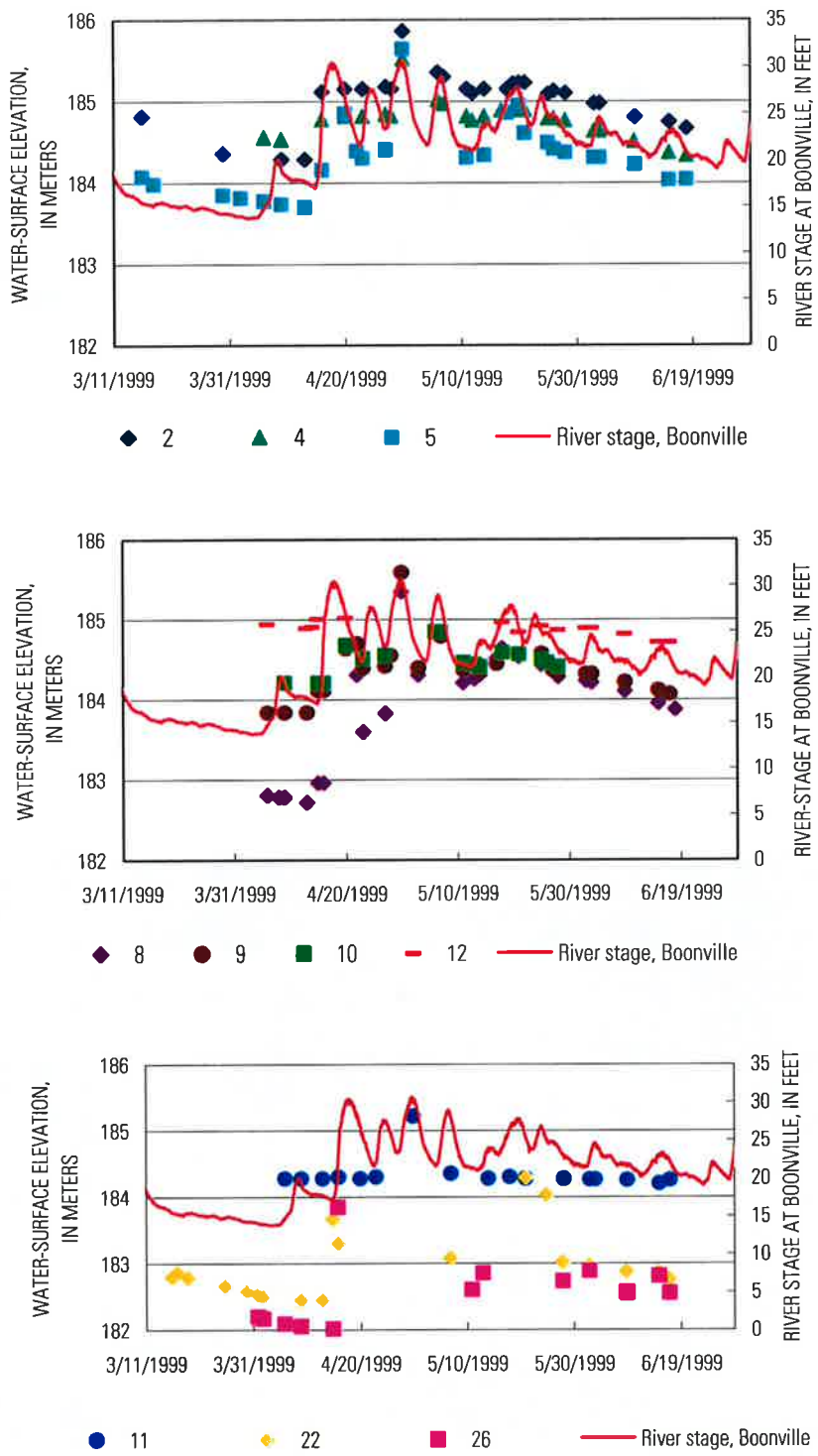

Figure 2-3. Surface elevations of wetlands at Lisbon Bottom, March–June 1999. Note that, with the exception of the measurements made by canoe on April 30, these data do not capture periods of greatest flood, when the wetlands were inaccessible to walkers.

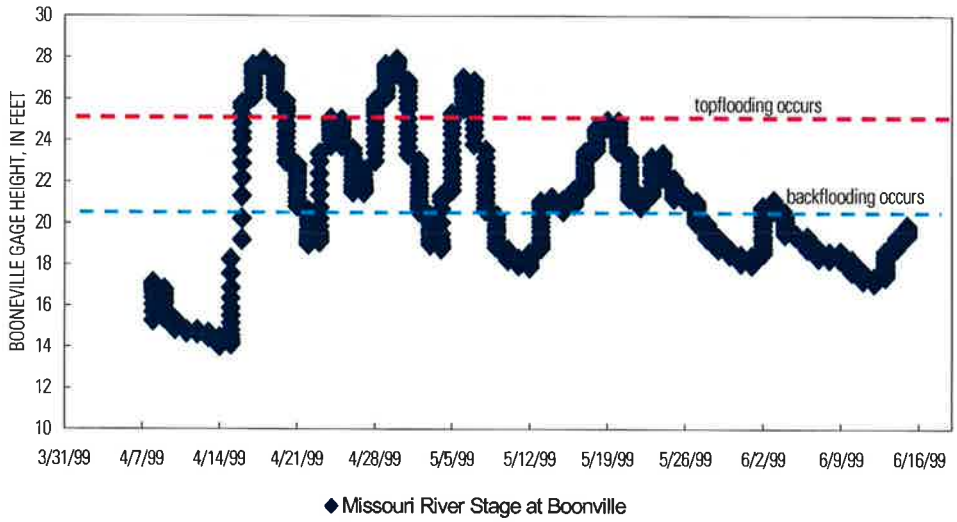

Figure 2-4. Missouri River stage at Boonville. The red horizontal line indicates the gage height (25.5 ft) at which water was seen to enter Lisbon Bottom through a crevasse that passes through the levee, topflooding Wetlands 4 and 5 and other portions of the northern bottom. Wetland 26 is believed to have connected to the river via backflooding through a narrow channel at approximately 20.5 feet on the Boonville gage (light blue horizontal line).

Figure 2-5. Unconsolidated sediment freshly deposited after a flood of Lisbon Bottom.

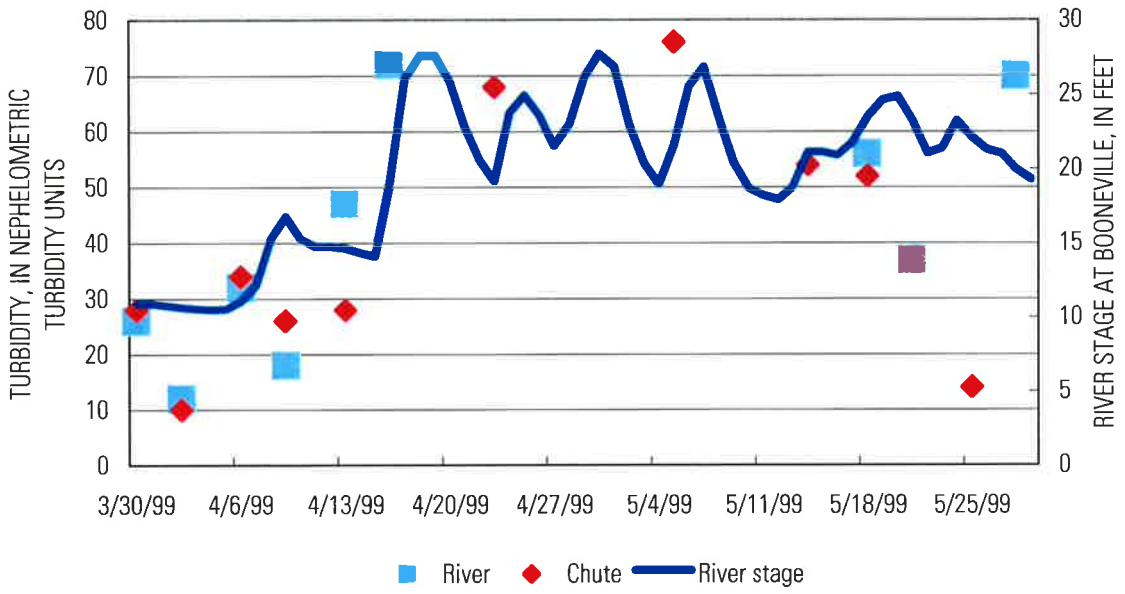

Figure 2-6. Turbidity of the Missouri River and the Lisbon Chute. Turbidity of the river and chute were very similar and both rose during the period of flooding. There are many missing values, especially from the river site, because safety concerns prohibited many collections during periods of high flow.

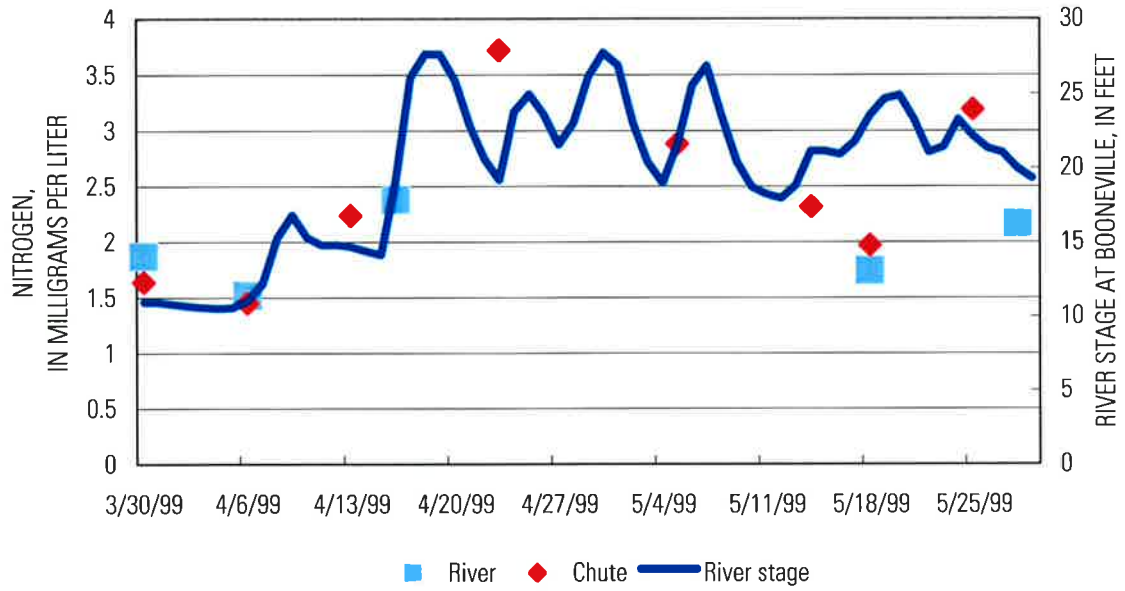

Figure 2-7. Nitrogen concentrations of the Missouri River and the Lisbon Chute. Nitrogen concentrations of the river and chute were very similar and were correlated with river stage. There are many missing values, especially from the river site, because safety concerns prohibited many collections during periods of high flow.

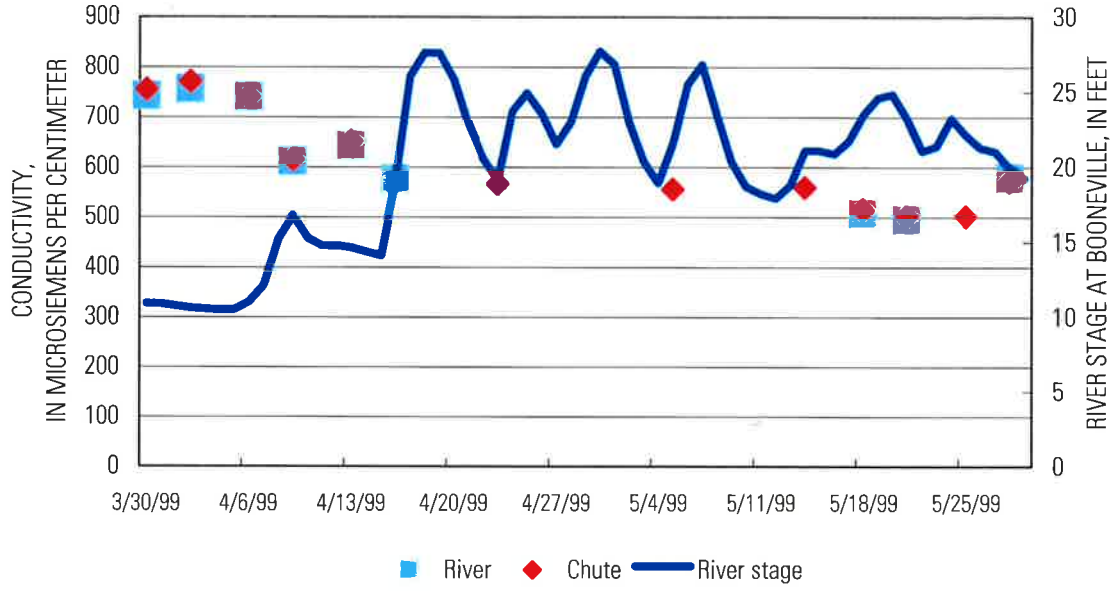

Figure 2-8. Conductivity of the Missouri River and the Lisbon Chute. Conductivity of the river and chute were nearly identical and were inversely correlated with river stage. There are many missing values, especially from the river site, because safety concerns prohibited many collections during periods of high flow.

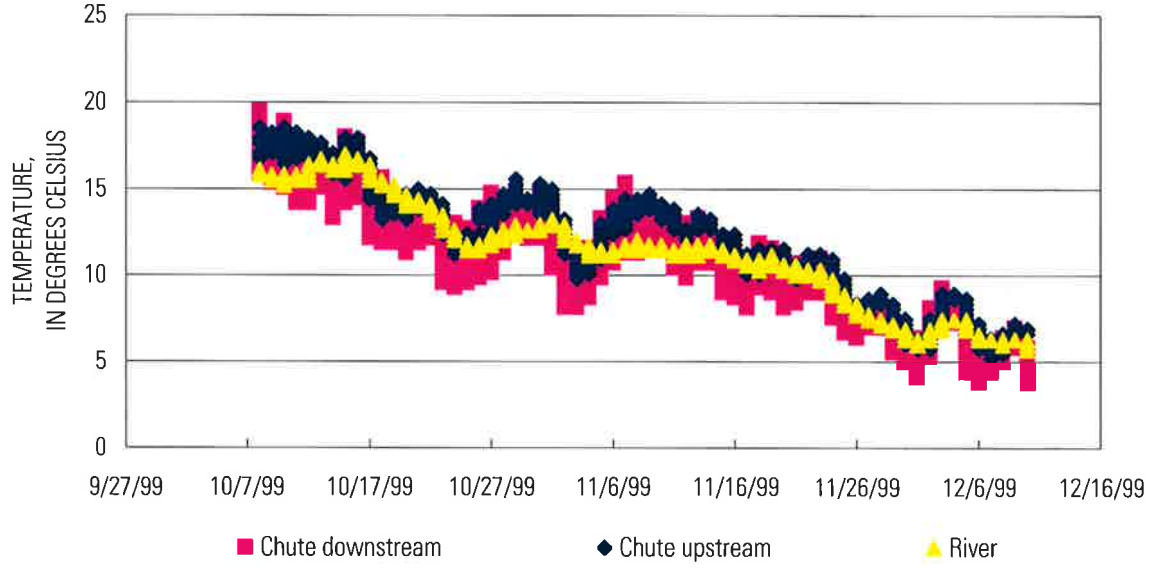

Figure 2-9. Temperature of the Missouri River and two sites within the Lisbon Chute. The temperature logger in the Missouri River was lost during the spring 1999 study; these data were collected in the fall of 1999. However, the data do indicate that the chute has less thermal inertia than the river and that diurnal fluctuations in chute temperature do occur, at least during periods of lower flow.

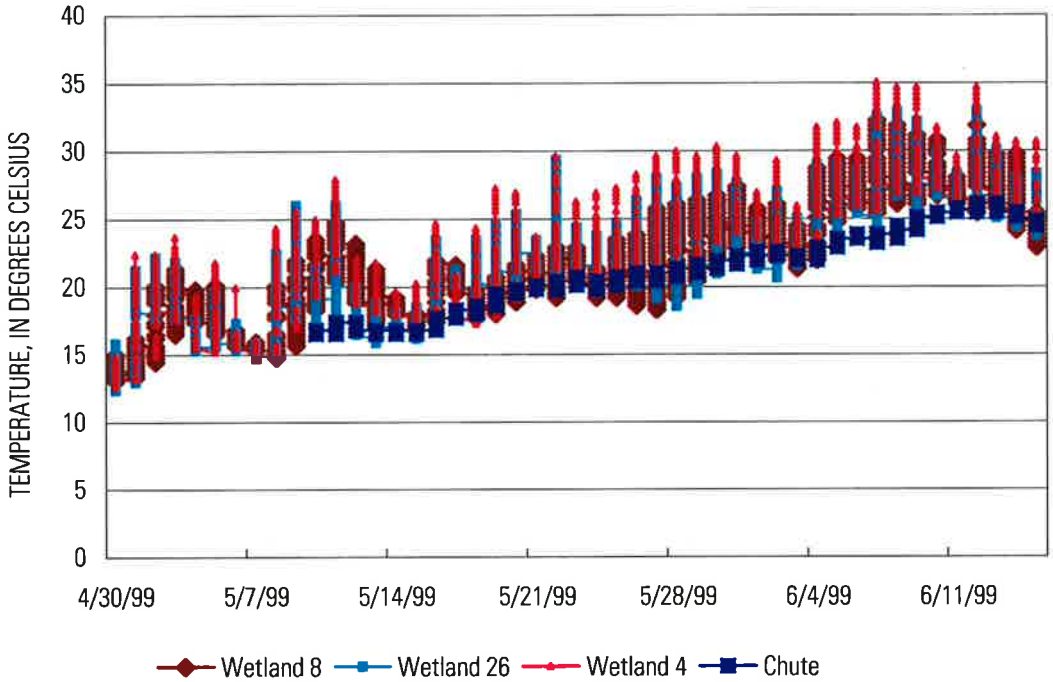

Figure 2-10. Temperature logger data from selected Lisbon Bottom wetlands and the chute. Loggers were installed 10 cm below the surface. Note that daily temperature variance was much higher in the shallow wetlands than in the chute.

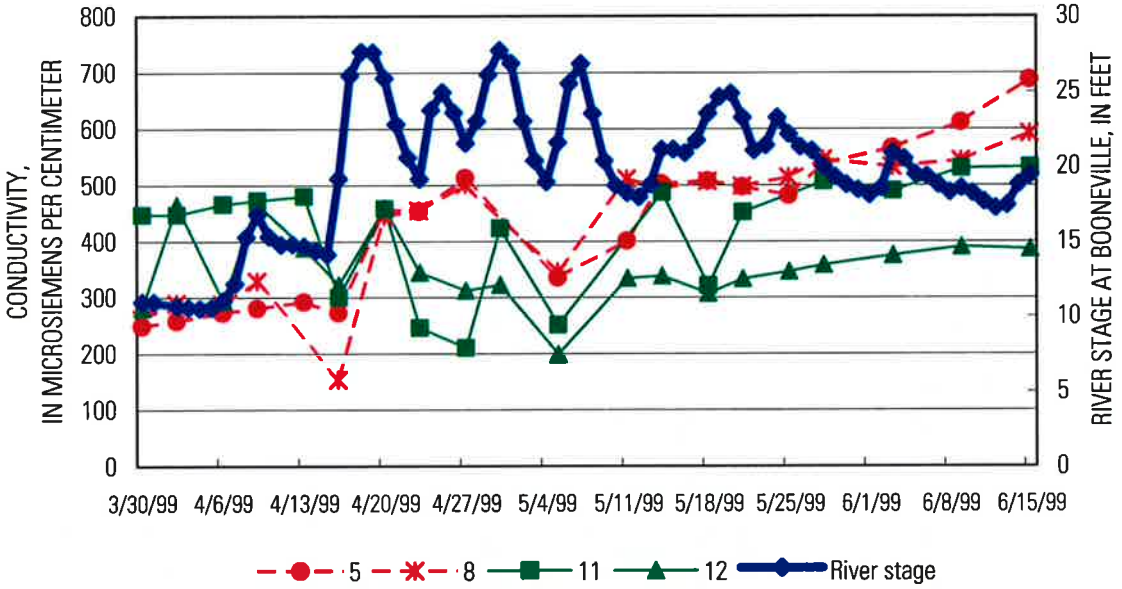

Figure 2-11. Conductivities of stream-influenced wetlands (in brown) and topflooding wetlands (in red) in relation to river stage. Note that conductivities of the top flooding wetlands increase after flooding by the river. Stream-influenced wetlands varied dramatically in response to local rain events, but were generally lower than river-influenced wetlands after flooding from the river.

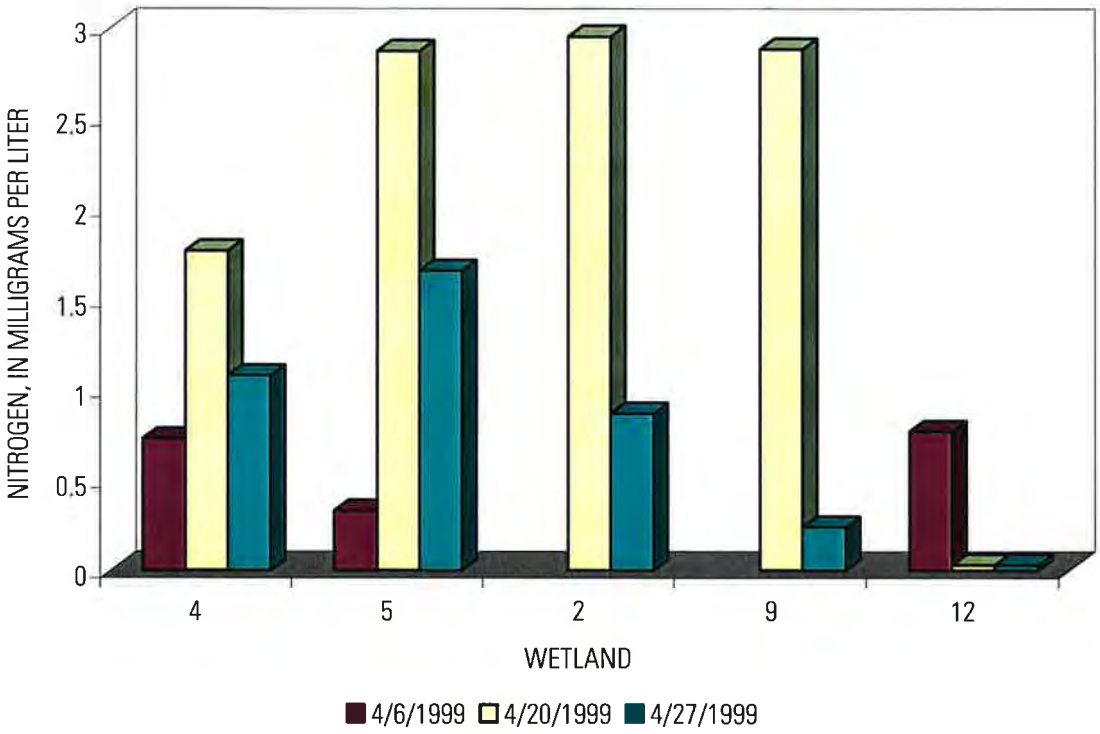

Figure 2-12. Total nitrogen concentrations in selected Lisbon Bottom wetlands before, during, and after a flood by the river. Wetlands 4 and 5 are topflooding scours. Wetlands 2 and 9 were shallow wetlands that contained no water prior to flooding. Wetland 12 was not inundated by the river. Wetlands 4, 5, 2, and 9 are in order of decreasing mean depth on 4/27/99.

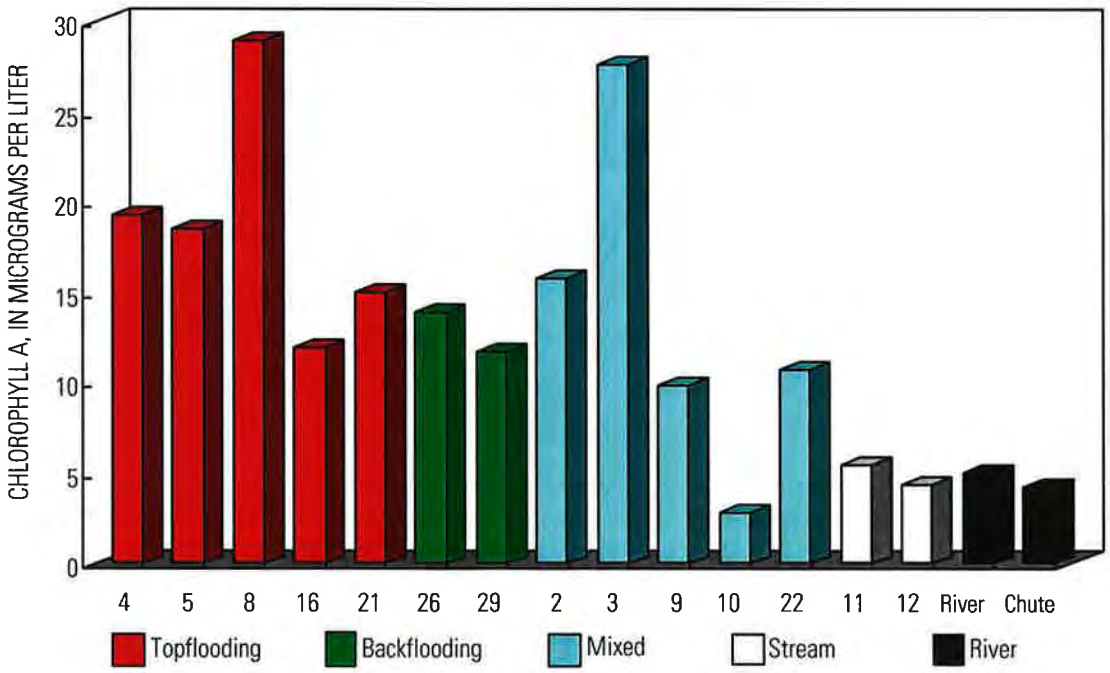

Figure 2-13. Mean chlorophyll *a* concentrations in Lisbon Bottom wetlands between March 30 and June 15, 1999.

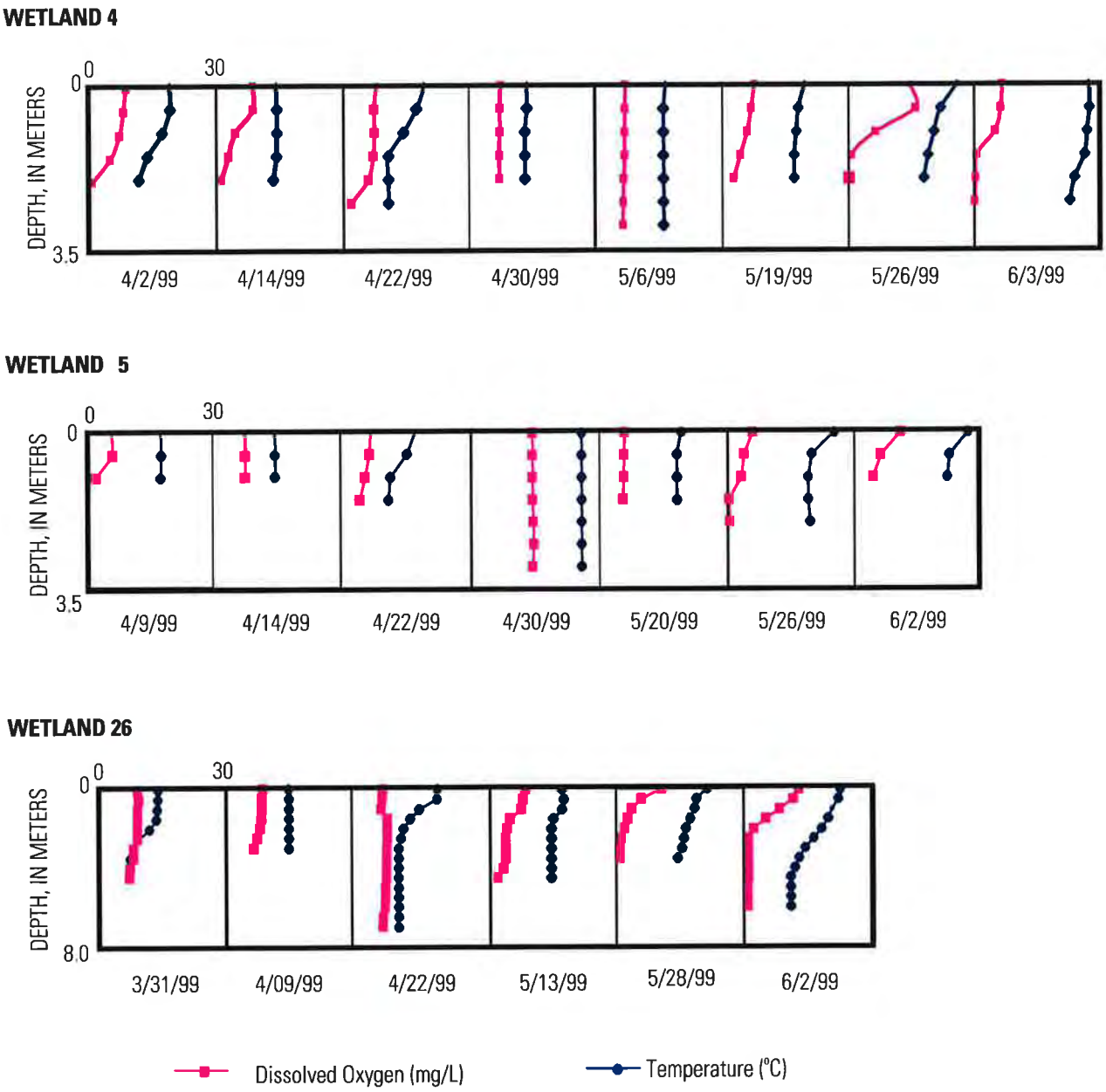

Figure 2-14. Temperature and dissolved oxygen profiles in three scour wetlands at Lisbon Bottom. Each square box is one sampling event. Surface values are at the top of the box; measurements are taken at half-meter depth intervals. These figures show data from the deepest point in each wetland. River water was passing through the crevasse April 16 to April 20, April 28 to May 1, May 5 to May 7, and May 19 to 20.

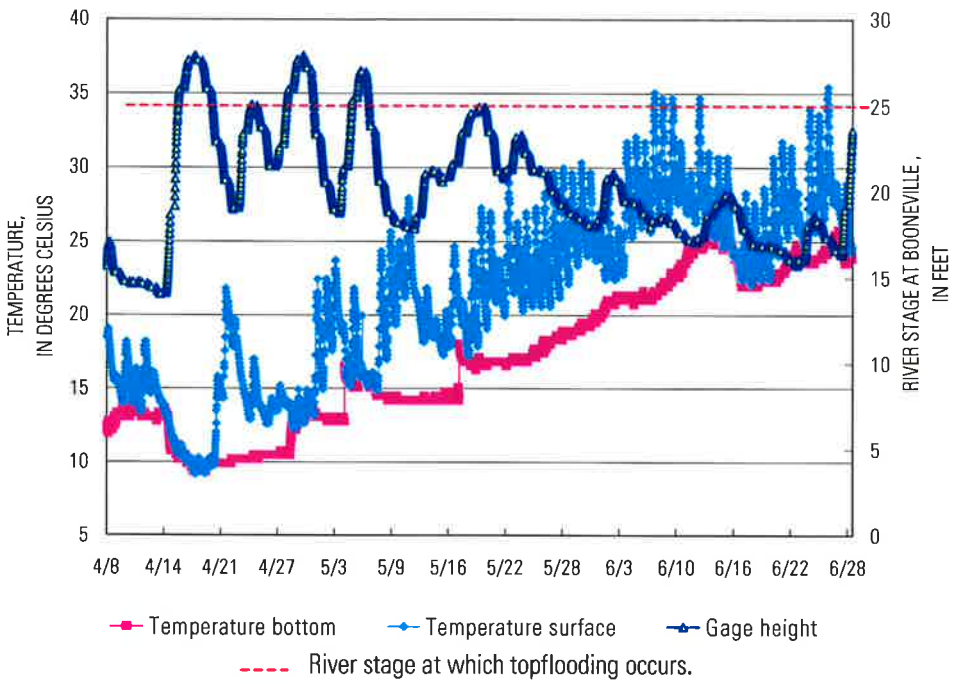

Figure 2-15. Temperature logger data from the surface and bottom of topflooding Wetland 4, and Boonville river gage height. Note that when water passes through the notch, surface and bottom temperatures are equivalent, but stratification redevelops quickly.

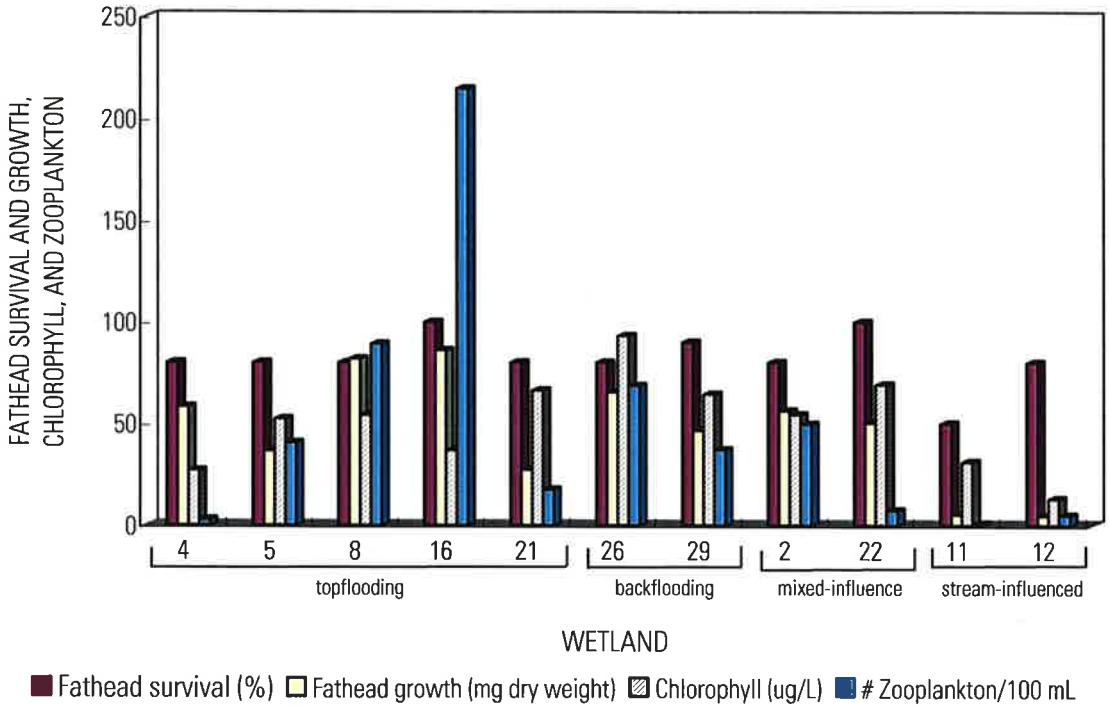

Figure 2-16. Survival and growth of caged fathead minnows in Lisbon Bottom wetlands with chlorophyll *a* concentrations and mean number of zooplankton per 100 mL water.

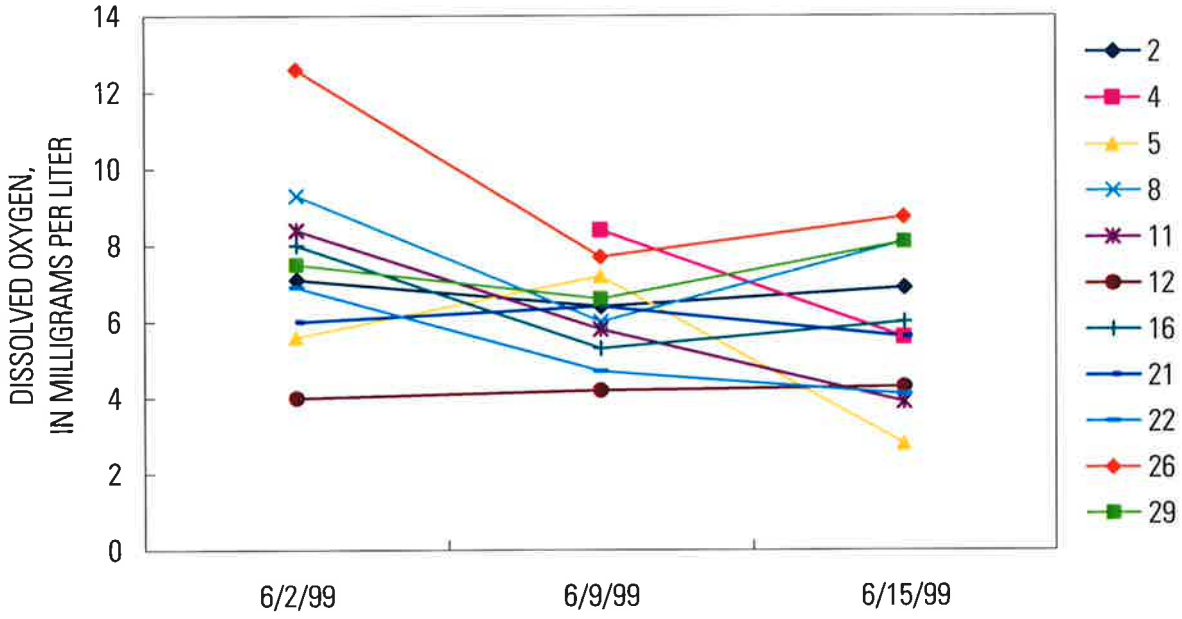

Figure 2-17. Dissolved oxygen in Lisbon Bottom wetlands during the fathead minnow experiment (5/27/99–6/16/99). Measurements were made adjacent to the floating cages.

Chapter 3. Zooplankton of Lisbon Bottom Wetlands

Duane C. Chapman, Barry C. Poulton and William R. Mabee

Abstract

In this study we examined the crustacean zooplankton assemblages of a continuum of wetland types at Lisbon Bottom and evaluated the zooplankton assemblages in relation to wetland limnology, hydrology and the fish community. Crustacean zooplankton were collected and identified in permanent and temporary wetlands of Lisbon Bottom, Missouri during the spring and early summer of 1999. Zooplankton were dominated by cladocerans and omnivorous copepods. Zooplankton density and diversity were related to flood events and nutrient pulses resulting from flood events. Topflooding wetlands had higher densities and diversities of zooplankton than backflooding wetlands, due to greater phytoplankton availability and possibly due to greater predation by fishes in the backflooding wetlands. Crustacean zooplankton density was much lower in stream-influenced wetlands than in the river-influenced wetlands, owing to lower nutrient availability and thus lower phytoplankton production in stream-influenced wetlands. Phytoplankton growth was accelerated by nutrients introduced during flood events and zooplankton populations increased thereafter, taking advantage of the increased resources.

Introduction

The Lower Missouri River system has been drastically altered over the past 50 years due to the combined effects of impoundment, channelization, bank stabilization, and levee construction. Collectively, these changes have resulted in the loss of backwater and wetland habitats. Many flood-plain-dependent fish species, including buffalos (*Ictiobus* sp.) and river carpsucker (*Carpiodes carpio*) that depend on these habitats for spawning and recruitment have also subsequently declined (Pflieger and Grace, 1987; Hesse and others, 1989).

Flooding of the vegetated flood plain is critical to these flood-plain-dependent fish species for several critical resources. Flood-plain-dependent species frequently deposit eggs on submerged vegetation as part of their specific reproductive strategy. Resulting larval fishes feed on zooplankton and other invertebrates that are produced in these nutrient and carbon-rich flooded backwater habitats. Abundant zooplankton food resources, in association with warmer temperatures of shallow water, contribute to enhanced bioenergetic conditions required for rapid growth and survival of young fishes. Numerous shorebirds and waterfowl also depend on zooplankton as food resources during late spring (Taylor, 1977; Crome, 1985). Invertebrates serve as high protein food resources necessary for egg production and feather regeneration during post-molt conditions.

Relatively little research has focused on the zooplankton assemblages of the Lower Missouri River. Early research was focused on the effects of power plant construction on fauna of the Lower Missouri River (Williams, 1973; Repsys and Rogers, 1982). This research indicated that zooplankton in the mainstem river were highly influenced by inputs from tributaries and upstream impoundments. Havel and Bethune (1999)

examined the zooplankton of various permanent connected and non-connected scour habitats of the Lower Missouri River and determined that zooplankton assemblages differed depending on frequency of river exchange and associated trophic structure of various habitats; however, temporary, shallow habitats were not studied. Beaver and others (1999) studied the midsummer zooplankton assemblages in four wetland types in northern Ohio and found higher numbers of cladoceran zooplankton in temporary wetlands compared to more permanent constructed wetlands.

In this study, we investigated the dynamics of zooplankton communities in a continuum of Lower Missouri River wetlands ranging from temporary to permanent classes on the Lisbon Bottom Tract of the Big Muddy National Fish and Wildlife Refuge. This study was conducted to determine the importance of various factors that control zooplankton dynamics including wetland morphology (that is, depth, surface area, etc.), source and timing of flooding (for example, river-connected versus non-connected), and biological factors (that is, presence and absence of fish; algal biomass).

Methods

Zooplankton were collected approximately once weekly, one sample per wetland, when water stage permitted, between 4/16/99 and 6/16/99 from Wetlands 2, 4, 8, 9, 12, and 26 (fig. 3-1). In addition, Wetlands 5, 10, 11, 16, and 21 were sampled weekly during the caged fathead minnow growth study (see Chapter 2) to provide supporting data for that study. Because some of the wetlands were very shallow and all sampling was performed by wading rather than from a boat, a special zooplankton net was designed and constructed. The net was attached to a 7.5 by 20 cm frame with a handle, rather than being pulled on a cord, and consisted of 183 µm mesh. This design allowed sampling even in very shallow wetlands with good control of the depth sampled, and reduced the possibility of bottom contact when sampling. A sample consisted of a 1 m sweep of the net pulled just below the surface. This mesh size is appropriate for the capture of adult crustacean zooplankton. In this study rotifers and copepod nauplii and copepodids were identified and enumerated, but most rotifers and many copepod nauplii probably passed through the collection device. Zooplankton were rinsed from the net into vials and preserved with ethanol. Zooplankton were sampled as close as possible to the staff gages, which were located in the deepest part of all the wetlands except the deep scours. Zooplankton were enumerated and identified to genus by BSA Environmental Services, Beachwood, Ohio. Copepod nauplii were enumerated, but not further classified, while copepodids were classified to order.

Zooplankton densities (number of organisms per liter) were calculated by dividing the number of organisms in the sample by the volume of water sampled. The Shannon-Weaver index (Shannon and Weaver, 1949) was calculated to describe the zooplankton diversity within individual wetlands and between wetlands. Analysis of variance was used to test differences in density and relative abundance of zooplankton between wetland permanence categories (SAS, 1990). Also, analysis of variance was used to test differences in zooplankton density within wetland permanence categories. Duncan's multiple range test was used to define differences. Percent relative abundance data was arc sine square root transformed before analysis (Snedecor and Cochran, 1989). A cluster analysis was performed using Ward's method to group wetlands by similarity of

zooplankton assemblages. Only wetlands that were sampled for the duration of the period (4/16/99 to 6/16/99) were included in the analysis.

Results and Discussion

All zooplankton data are reported in Korschgen and others (ArcView-based spatial decision support system for the Lisbon Bottom Unit of the Big Muddy National Fish and Wildlife Refuge, unpub. data, 2001). Mean zooplankton densities from wetlands sampled over the course of the study are shown in table 3-1. A list of zooplankton genera captured is shown in table 3-2. Overall, cladocerans were more common than copepods. Wetland 26, a deep backflooding scour that had the most connectivity with the river, had the lowest zooplankton densities; and the shallowest, most ephemeral wetlands (2 and 9) had the highest. Wetlands 2 and 9 were recharged by topflooding, and at times by intermittent streams and overland runoff. Total number of organisms varied greatly within a site between sampling periods. For example, Wetland 2 had a total of 2.7 organisms/L on April 15 and over 6000 organisms/L on May 26. No ostracods were captured during this study.

The most common crustacean zooplankton genus overall was the calanoid copepod *Skistodiaptomus*, owing to its ubiquity in the wetlands, high concentrations in shallow Wetlands 2 and 12, and very high concentrations in the shallow and terrestrially vegetated Wetland 9. *Tropocyclops* was the only other common copepod genus, being found in fairly high numbers in every wetland except 4 and 9. *Tropocyclops* is a small omnivorous cyclopoid copepod. Seven genera of adult copepods were identified in all. All of the adult copepods captured were of either the Cyclopoida or Calanoida orders, which include nektonic species (Barnes, 1987) that are susceptible to our sampling gear. Calanoid and cyclopoid copepodids were also captured. Adult harpacticoid copepods are mostly benthic and were not captured in this study, but harpacticoid copepodids were captured.

The most common cladocerans in descending order of prominence were *Moina, Scapholeberis, Daphnia, Simocephalus, Chydorus*, *Bosmina,* and *Ceriodaphnia. Bosmina and Chydorus* are considered indicative of eutrophic conditions (Beaver and others, 1999). *Bosmina* and *Chydorus* never dominated the samples, but they were common. The average combined relative abundance of these two genera was 14%. Overall diversity of Cladocera in the wetlands was higher than that of the Copepoda, with 13 different genera of Cladocera identified. The above-listed seven genera accounted for >98% of the individuals.

Number of genera of crustacean zooplankton within an individual sample ranged from 1 to 13. Number of genera within a wetland (three to six sample dates per wetland) ranged from 7 to 15. Havel and Bethune (1999), who in a 1995 study of 12 unconnected scours in the Missouri River flood plain (4 to 6 sample dates per wetland) found species richness between 4 and 14.

Topflooding wetlands had higher number of genera and Shannon-Weaver diversity than did the backflooding Wetlands 22 and 26 (table 3-3), which were also the wetlands most connected to the Missouri River. Wetland 26, a backflooding wetland and the largest and deepest wetland in the study, had the lowest number of genera, averaging 3.6 per sample over the study. Backflooding Wetland 29 had higher diversity than backflooding Wetlands 26 and 22, but it was sampled on fewer dates and only during a period in which it was not connected to the river.

Zooplankton densities were highly influenced by river flooding and the subsequent production of phytoplankton. The river repeatedly flooded wetlands in the northern portion of the bottom (Wetlands 2, 4, and 8) between April 1 and May 8, 1999 (Chapter 2, fig. 2-4), resulting in high turbidity (fig. 3-2A) and low densities of zooplankton. Crustacean zooplankton densities in the Missouri River are usually much lower than the average densities we found in wetlands (Berner, 1951; Jennings, 1979). Chlorophyll *a* concentrations increased dramatically from May 11 to May 21, 1999 (fig. 3-2B) after the turbidity from the river settled out. (Note that turbidity increased again during this period (fig. 3-2A) because of the increase in phytoplankton.) This was followed by very high numbers of zooplankton in the May 26 and June 3 samples (fig. 3-2C). Number of genera also peaked on May 26, 1999 in the topflooding wetlands (fig. 3-3). Safety concerns precluded sampling in backflooding Wetlands 22 and 26 during the highest flooding periods, but they apparently followed a pattern similar to that of the topflooding wetlands, with turbidity, chlorophyll and zooplankton peaks on approximately the same dates, although overall zooplankton density was much lower (fig. 3-4 A-C).

Wetland 12, which was not flooded by the river and was dominated by aquatic macrophytes rather than phytoplankton, showed a very different pattern in turbidity, chlorophyll, and zooplankton density (fig. 3-2, A-C). Zooplankton density in Wetland 12 was highest early in the study and quite low in late May through June. After the ambient temperature increased in late April, macrophyte growth in Wetland 12 removed all available nutrients (Chapter 2, fig. 2-7) and phytoplankton density was very low. Without the phytoplankton forage base, zooplankton density also was low. Despite the low density of zooplankton, Wetland 12 had the second highest diversity and highest number of genera of crustacean zooplankton. This differs from the report by Havel and Bethune (1999), which found that connectivity between the wetland and the river was strongly and positively correlated with species richness. In this study, Wetlands 26 and 22, which had the highest connectivity to the river, had the lowest number of genera, and Wetlands 21 and 12, which were lowest in connectivity, had the highest number of genera.

Although chlorophyll *a* concentrations were somewhat lower in Wetland 26 and 22 than in the topflooding wetlands, the lower density of zooplankton in these wetlands may not be entirely due to higher turbidity and resulting lower primary and secondary productivity. Predation on zooplankton by planktivorous fish can strongly influence zooplankton density and species composition (Devries and Stein, 1992). The fish communities in these wetlands varied by wetland type and water source (see Chapter 5), and thus likely had different influences on zooplankton density in different wetlands. Wetlands 22 and 26 were connected to the river more often than the other wetlands, but the connection was via backflooding through Coopers Creek and water exchange was not high between the river and the wetlands except during periods of extreme flood. As evidence of low water exchange, turbidity in these wetlands was not higher than in the topflooding wetlands even though they were connected more often to the highly turbid river. The connection to the river apparently provides access to the wetlands by riverine fishes such as emerald, ghost, and mimic shiners (*Notropis atherinoides, N. buchanani,* and *N. volucellus,* respectively). The numbers of zooplankton in Wetlands 22 and 26 were likely impacted by these planktivorous fishes, which were captured in large numbers in these wetlands, but not in the topflooding wetlands (see Chapter 5). Wetland 21, a topflooding wetland located very near Wetland 22, also had a high density of small fish, but had a much higher zooplankton density. Young-of-the-

year buffalo and common carp (*Cyprinus carpio*), which may not be as efficient predators of nektonic zooplankton, dominated Wetland 21.

Ephemeral wetlands in this study had the highest densities of crustacean zooplankton (table 3-4), followed by the permanent wetlands. Temporary (but not ephemeral) wetlands had the lowest densities. Ephemeral wetlands had higher numbers of calanoid than cyclopoid copepods, and permanent wetlands had higher numbers of cyclopoid than calanoid copepods. However, none of these differences were significant at the $\alpha = 0.05$ level. Beaver and others (1999) found that temporary wetlands had higher numbers of cladocerans and copepods than three categories (constructed, anthropogenically impacted, and non-impacted) of permanent wetlands, but he did not discriminate as to the degree of permanence of the temporary wetlands. Overall densities of copepods and cladocerans in permanent wetlands were higher in this study than in that of Beaver and others (1999) (table 3-4), and ephemeral wetlands in this study had higher densities than either permanent or temporary wetlands in that study. Forty-two percent of the crustacean zooplankton in temporary wetlands of the Beaver and others study were ostracods, which were not captured in any wetland in this study. Calanoid copepods were rare in the Beaver and others (1999) study. However, calanoid copepods were common in our study, especially in ephemeral basins where density was significantly higher than in temporary or permanent wetlands.

Densities of crustacean zooplankton at Lisbon Bottom were also much higher than that reported by Havel and Bethune (1999), who compared a scour wetland connected to the Missouri River to an unconnected scour on the Missouri River flood plain in March of 1997. In that study, density of crustacean zooplankton was 2.2/L in the connected scour and 7.6/L in the unconnected scour. In our study, the mean crustacean density (average of all samples in all wetlands) was 600/L. Of 64 samples analyzed in this study, only 7 had less than 10 crustacean zooplankton/L. In the study by Havel and Bethune, most of the crustacean zooplankton collected were the cyclopoid copepod *Diacyclops thomasi*. *Diacyclops* was not common in this study, occurring only in Wetland 9 on two sample dates.

Ward's minimum variance cluster analysis of the wetlands by zooplankton assemblages (fig. 3-5) identified ephemeral Wetland 2 as the most different from the others. Notably, Wetland 2 was also identified as the most different in the cluster analysis performed using large fish species assemblages. Wetland 9, the other ephemeral wetland, was most similar to Wetland 2. Wetlands 26 and 22, which were among the only backflooding wetlands in the study and the wetlands most connected to the Missouri River, were very similar. Wetlands 4 and 26, the only deep scours in the study, were grouped close together. Surprisingly, Wetlands 12 and 8 were grouped very close together, despite their very different hydrology, limnology, and fish assemblages. In general, though, wetlands that were similar in hydrology were most similar in zooplankton assemblages.

Soeken (1998) identified three genera of Cladocera (*Moina, Bosmina*, and *Diaphanasoma*) that are resistant to high turbidity and are often present in turbid rivers. Based on that study, it could be hypothesized that the relative abundances of these genera would be higher after a flood event, when turbidities are high and other zooplankton might have been flushed from the wetlands. However, we found no strong correlations between the density or relative abundances of these genera and turbidity, either when relative abundance was

expressed as a percentage of total crustacean zooplankton or when expressed as a percentage of total Cladocera. This held true regardless of whether these three genera were grouped together or considered separately (fig. 3-6). It should be noted that *Moina* and *Diaphanosoma* densities were always low at turbidities below 30 ntu, but since periods of low turbidities had low densities of many zooplankton, relative densities of these genera were sometimes high even at low turbidities.

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Table 3-1. Mean number of zooplankton organisms per liter in Lisbon Bottom wetlands. Note that the gear incorporated a mesh size designed primarily for the capture of adult crustacean zooplankton and therefore underestimates the contribution of small nauplii and rotifers.

Wetland number	Cladocerans	Copepods	Rotifers	Total organisms
2	1011.1	161.2	83.2	1255.6
4	198.9	12.2	6.7	217.7
8	252.9	117.2	76.5	446.6
9	255.1	350.6	0.8	606.5
12	447.9	78.0	0.8	526.7
22	71.8	150.9	0.0	222.8
26	21.4	24.1	12.9	58.4
Overall mean	332.8	134.1	26.1	493.0

Table 3-2. List of zooplankton genera captured in wetlands of Lisbon Bottom between late March and the end of June 1999. Thirty-two genera were captured. Rotifer genera captured are listed here, but it should be noted that the net size used (183 μm) was chosen for capture of crustacean zooplankton, and may have been too large for efficient capture of rotifers.

Cladocera	**Copepoda***	**Rotifera**
Alona		*Ascomorpha*
Bosmina	**Calanoida**	*Asplanchna*
Ceriodaphnia	*Diaptomus*	*Bdelloid*
Chydorus	*Skistodiaptomus*	*Brachionus*
Daphnia		*Conochiloides*
Diaphanosoma	**Cyclopoida**	*Filinia*
Kurzia	*Acanthocyclops*	*Keratella*
Leydigia	*Diacyclops*	*Lecane*
Macrothrix	*Eucyclops*	*Monostyla*
Moina	*Mesocyclops*	*Ploesoma*
Pleuroxus	*Tropocyclops*	*Polyarthra*
Simocephalus		*Trichocerca*
Scapholeberis		

* Harpacticoid copepodids were captured in this study, but no adults of that primarily benthic order were captured.

Table 3-3. Number of crustacean zooplankton genera and Shannon-Weaver diversity in Lisbon Bottom wetlands. All wetlands from which at least three samples were taken are shown.

Wetland number	Mean number of genera per sample	Shannon-Weaver Index	N	Water source
2	6.1	1.1	7	Topflooding and stream
4	6.9	1.3	7	Topflooding, some stream
5	7.3	1.0	3	Topflooding
8	7.9	1.2	7	Topflooding
9	6.2	1.0	8	Topflooding and stream
12	8.0	1.2	7	Stream
16	5.0	0.9	3	Topflooding
21	9.7	1.6	3	Topflooding
22	4.3	0.9	6	Backflooding
26	3.7	0.6	6	Backflooding

Table 3-4. Densities of crustacean zooplankton at Lisbon Bottom by wetland permanence, with data from Beaver and others (1999) for comparison. Wetlands 11 and 12 are excluded from this comparison because they were highly stream-influenced, and thus differed limnologically from the other wetlands. Total crustacean zooplankton may differ slightly from the horizontal sum because of the inclusion of copepod nauplii, which were not identified to order, and in the case of the Beaver and others data, from the inclusion of Ostracoda. Ostracoda were not found in this study. Different lower case superscripts indicate significant differences between wetland permanence types within zooplankton taxonomic groups ($\alpha = 0.05$, Duncan's multiple range test).

Permanence	Wetland	Mean cladocerans/L	Mean cyclopoids/L	Mean calanoids/L	Total crustacean zooplankton
Ephemeral	2	1011	78	83	1172
	9	255	27	323	606
	10	2583	70	55	2708
	Mean	**1283**	**58**	**154**[a]	**1495**
Temporary	8	253	92	25	370
	21	114	49	8	175
	22	29	19	122	170
	29	188	118	32	337
	Mean	**146**	**69**	**47**[b]	**263**
Permanent	4	203	7	6	215
	5	305	26	15	347
	16	1293	789	60	2143
	26	89	263	0	353
	Mean	**473**	**271**	**20**[b]	**764**
Temporary	Beaver and others, 1999	**152**	**85**	**0.3**	**533**
Permanent	Beaver and others, 1999	**55**	**59**	**2**	**226**

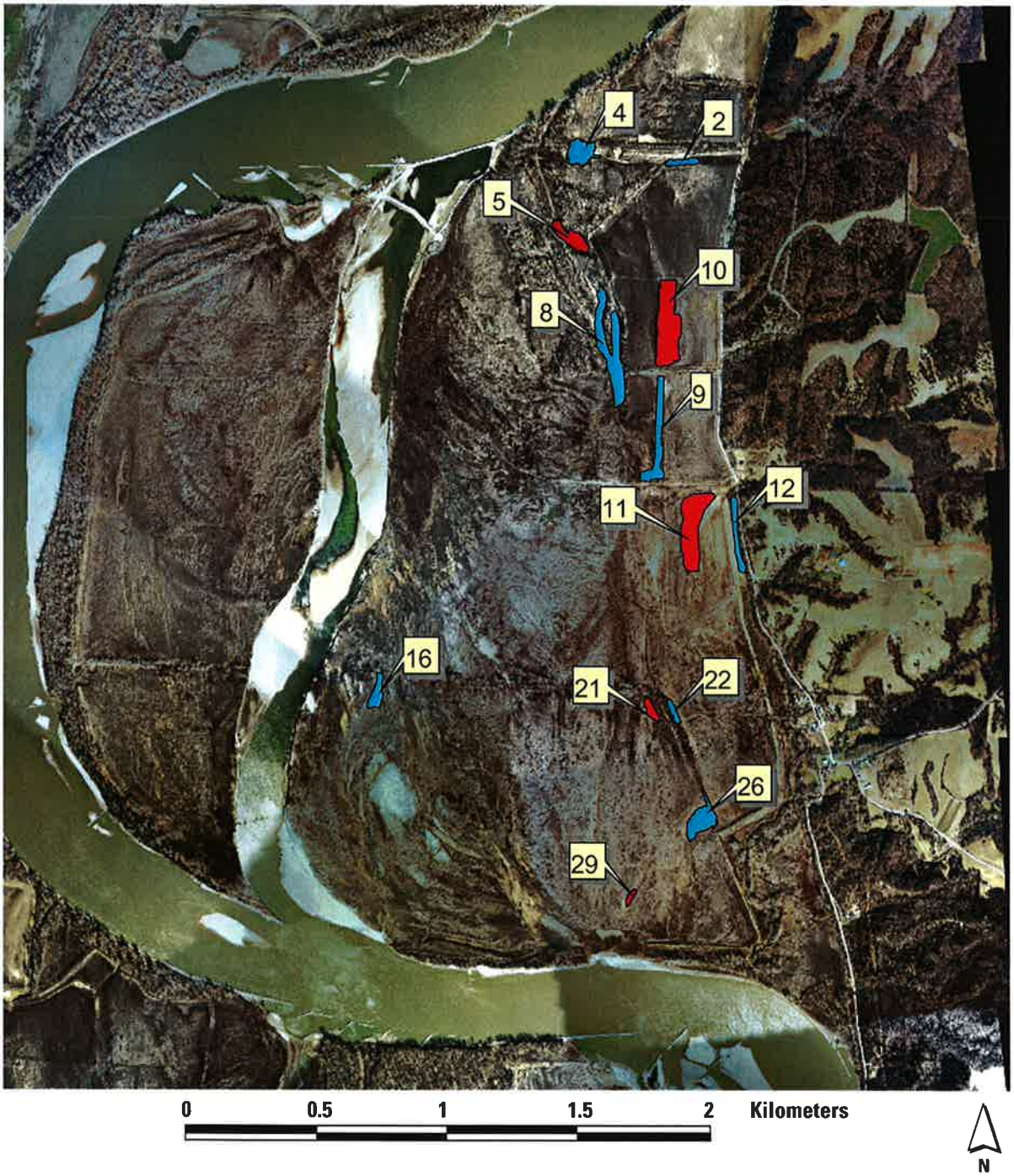

Figure 3-1. Lisbon Bottom wetlands in which zooplankton were sampled. Wetlands were sampled weekly; those illustrated in blue were sampled from April 15, 1999 through June 15, 1999; wetlands in red were sampled only during the fathead minnow growth study (May 27, 1999 through June 16, 1999). *Background photo courtesy of U.S. Army Corps of Engineers, Kansas City, MO, March 2000.*

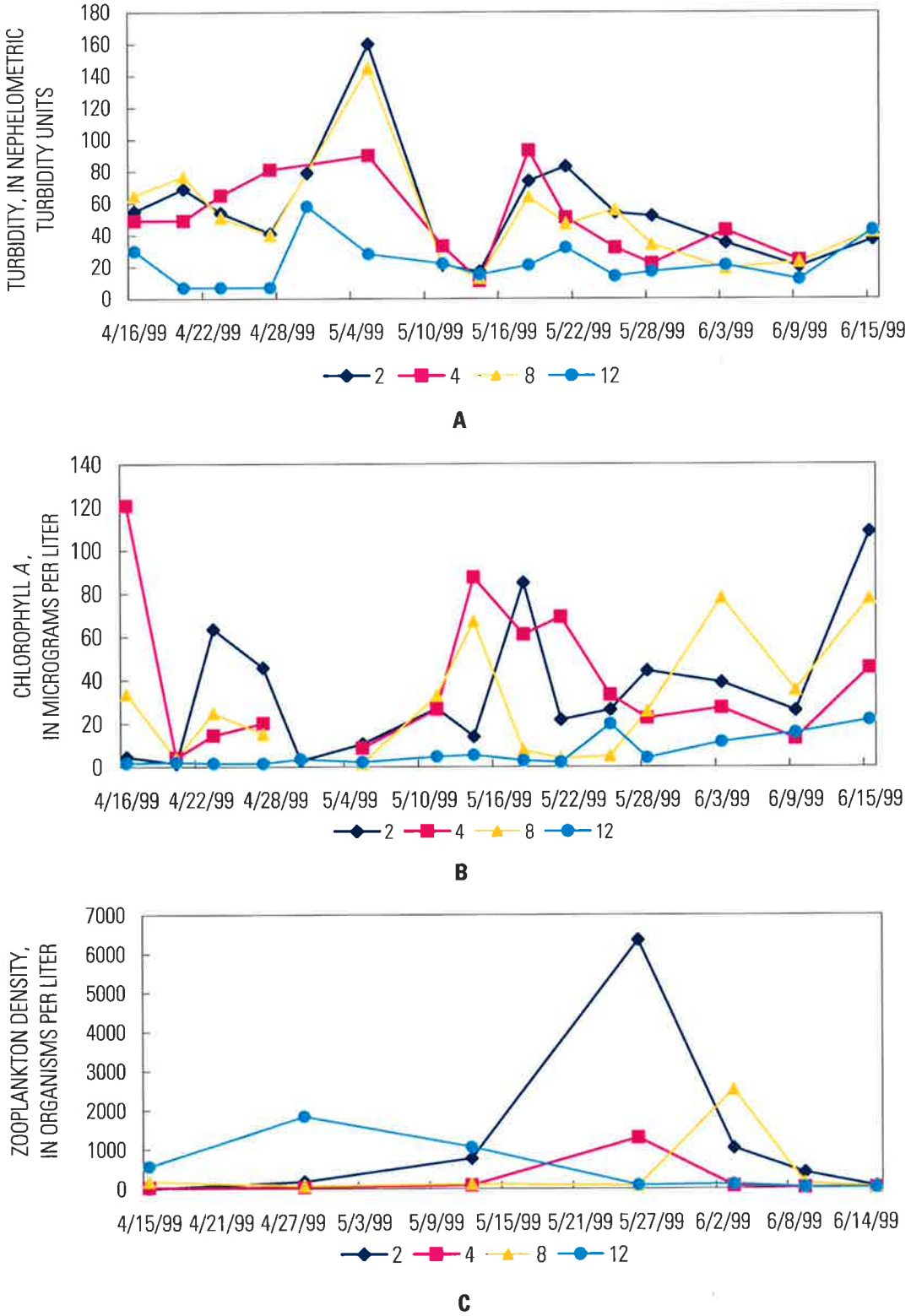

Figure 3-2. Turbidity (A), chlorophyll *a* (B), and zooplankton density (C) in Lisbon Bottom wetlands. Wetlands 2, 4, and 8 were flooded by the river between 4/16 and 5/7/99, whereas Wetland 12 was protected from flooding by a levee.

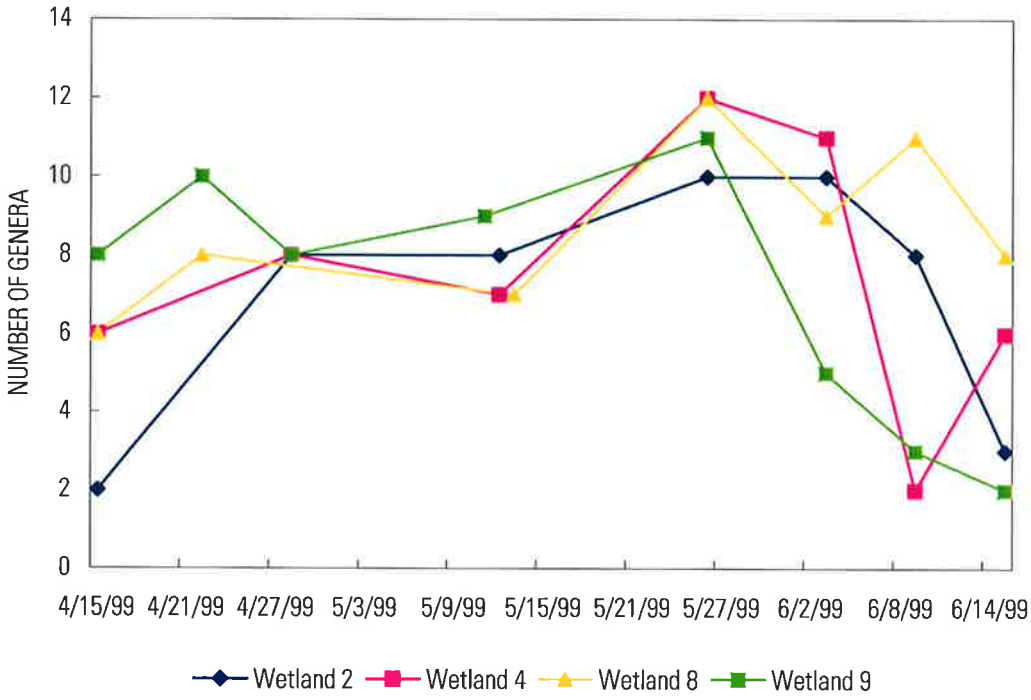

Figure 3-3. Number of crustacean zooplankton genera in topflooding wetlands of Lisbon Bottom.

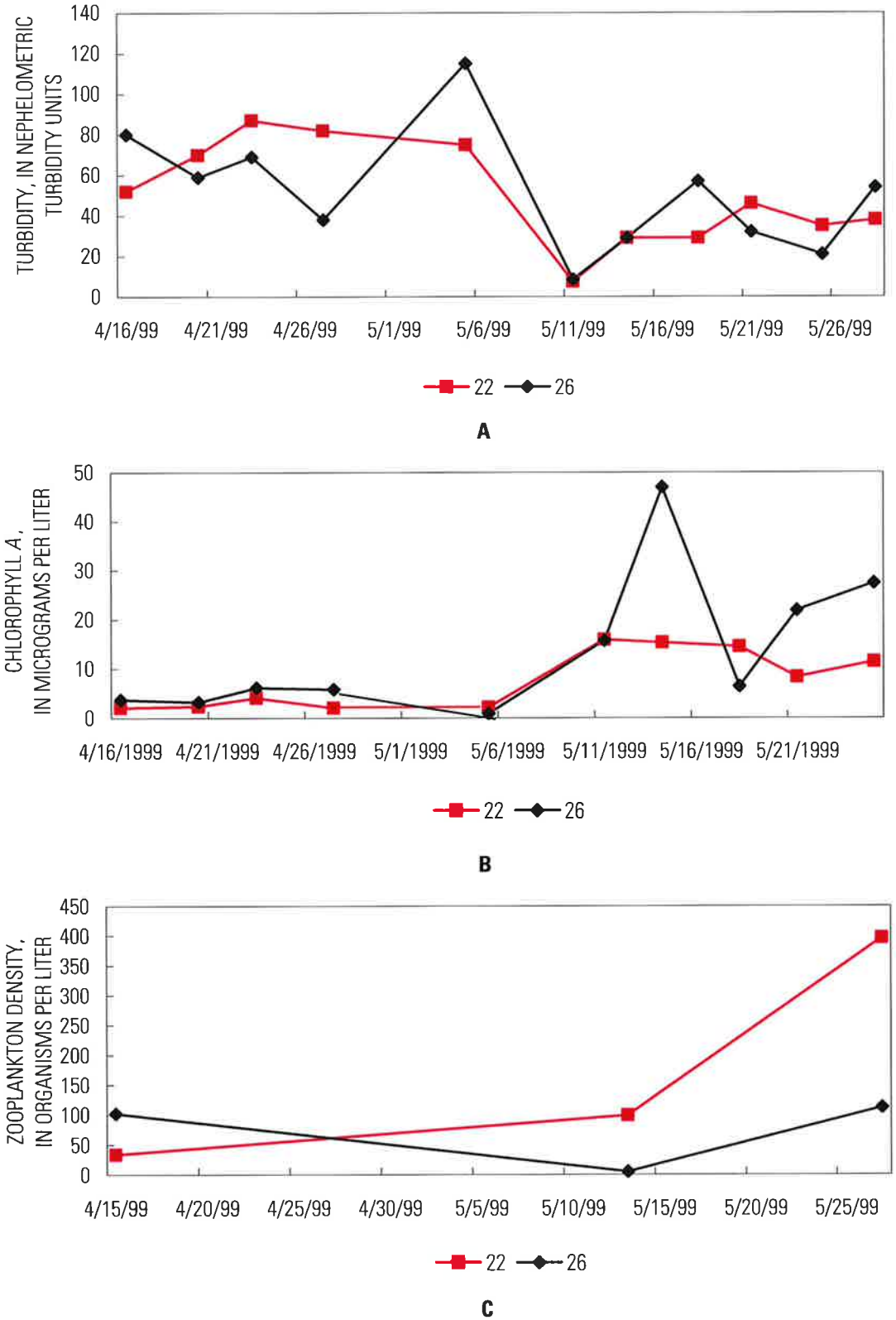

Figure 3-4. Turbidity (A), chlorophyll *a* (B), and zooplankton density (C) in two Lisbon Bottom wetlands. Wetland 26 is a large scour, often connected to the river by backflooding. Wetland 22 is connected to the river through Wetland 26 at higher water levels.

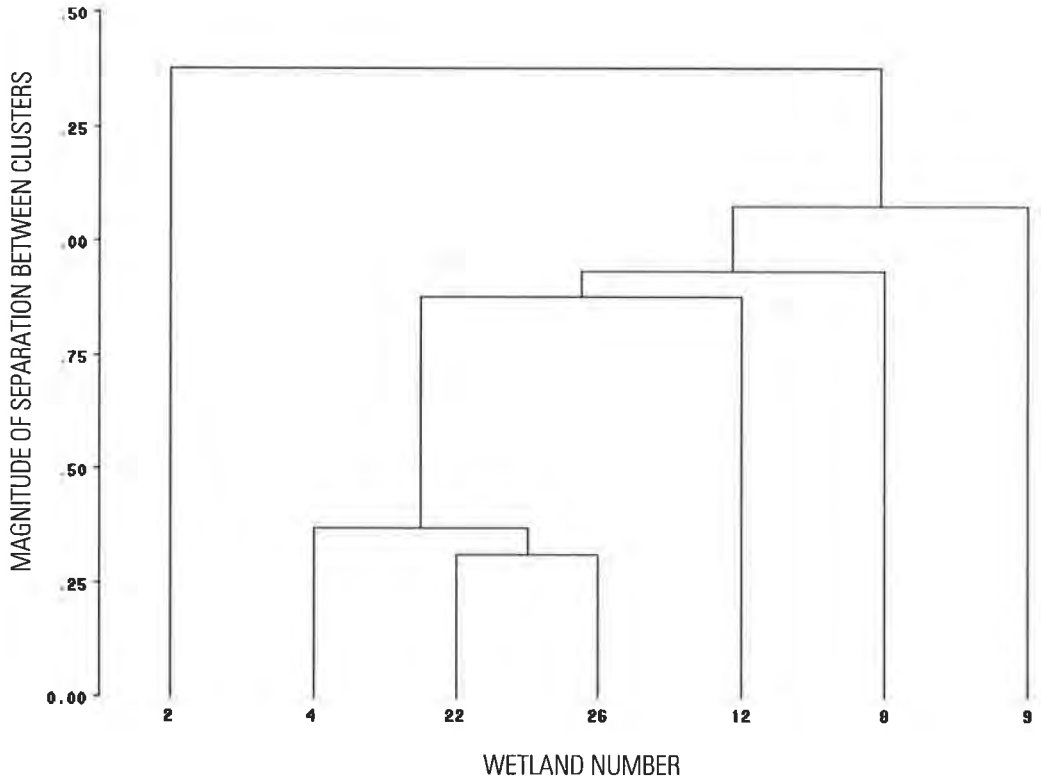

Figure 3-5. Ward's minimum variance cluster analysis of Lisbon Bottom wetlands by zooplankton genera assemblages, 4/16/99 to 6/16/99. Only wetlands that were sampled for the entire two-month period are included in this analysis.

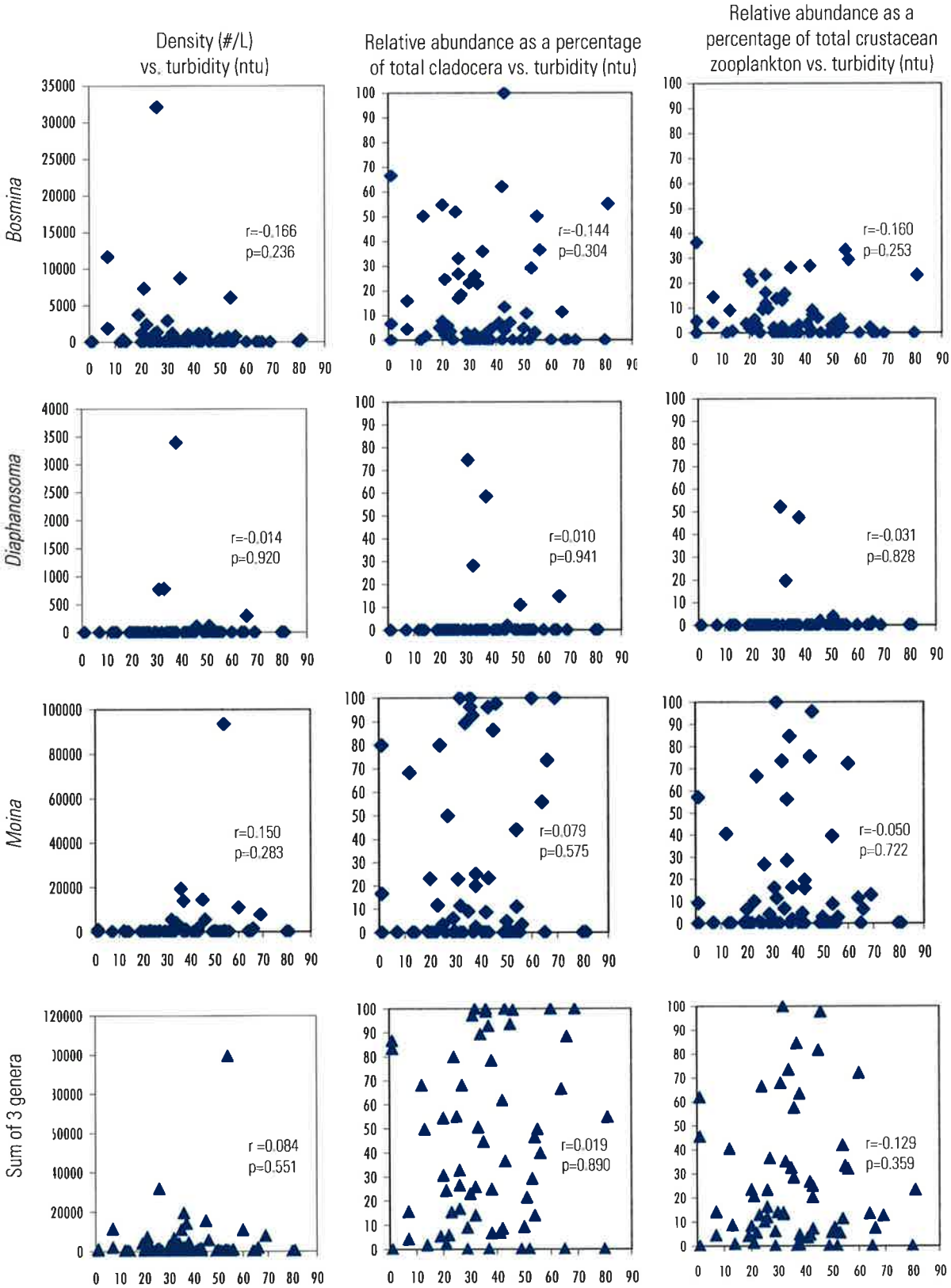

Figure 3-6. Scatterplots of density and relative abundances of three cladoceran genera that are reported to be adapted to high turbidity (Soeken, 1998) and are found in the Missouri River. Each data point represents a single sample. Samples were taken from a variety of wetland types at Lisbon Bottom, Missouri, between 4/16/99 and 6/16/99.

Chapter 4. Aquatic Invertebrates of Lisbon Bottom Wetlands

Barry C. Poulton

Abstract

Aquatic macroinvertebrates were sampled both qualitatively and quantitatively from March–July 1999 to characterize the community composition and density in different types of wetlands at Lisbon Bottom based on water source, permanence, available vegetation structure, and timing of flood pulses. Twelve wetlands were sampled 1–2 times per month to document species richness (timed sweep-net sample), and eight wetlands were sampled at least once every two weeks for measuring macroinvertebrate density (0.24-m dia. stovepipe). From this study and previous macroinvertebrate research including adjacent riverine habitats (Lisbon Chute, mainstem Missouri River, etc.), a total of 260 species are known to exist in the Lisbon area, over half of which are unique to the flood-plain wetland complex. Richness of Odonata (dragonflies, damselflies), Coleoptera (beetles), Hemiptera (true bugs), and Ephemeroptera (mayflies) was high in vegetated areas of most wetlands; however, richness of Diptera (flies and mosquitoes) was lower than that reported in other studies and the Trichoptera (caddisflies) were nearly absent. Temporary wetlands held water throughout the winter months due to the fall 1998 flood, and the invertebrate community was dominated by overwintering species and groups of pioneer taxa that were available for dispersal to other basins after flooding occurred in mid-April. Species richness was lowest in deep scours, and highest in seasonal wetlands. Both species richness and density (#/m^2) were highest when margin vegetation was inundated, which corresponds with a period of 2–3 weeks after the flood pulse. Richness and density were also highest in seasonal wetlands; scours had lowest species richness throughout the early part of the study, but increased by late spring and summer periods. In all but the deep scours, the ratio of predator / herbivore-detritivores gradually declined during the study period, and the ratio of benthic / pelagic invertebrates peaked during the post-flood period. Both of these indicators appear to correspond with changes in the availability of organic matter due to flooding. Recommendations and goals for managing flood-plain wetlands for maximization of wildlife value will also maximize the availability and productivity of macroinvertebrate food sources for other wildlife species, while increasing biodiversity.

Introduction

The aquatic macroinvertebrate communities inhabiting many flood-plain habitats had not been previously studied within the Lower Missouri River flood plain. Most studies on mainstem invertebrates in the lower river had been conducted along the Nebraska-Iowa and South Dakota borders. Several pilot projects were initiated by the U.S. Army Corps of Engineers in the 1970s and 1980s, with the primary goal of evaluating the use of artificially created habitats in the mainstem of the river such as wing deflectors and slack water areas associated with dike fields. However, many habitats in the mainstem of the river had not been surveyed, and flood-plain wetland communities were largely ignored until the Missouri River Post-Flood Evaluation Project

(MRPE) study was initiated after the 1993 flood. Bataille and others (1999) listed a total of 85 taxa in the Lower Missouri River flood-plain wetlands that were examined in the MRPE study, but most of these were not identified past the family level. Further, wetlands were not sampled during the most diverse season (early spring), and both methods used in the wetlands were passive techniques (activity and emergence traps).

Between 1992–1998, aquatic macroinvertebrates in the Lower Missouri had become well known from the bioassessment and longitudinal evaluation studies conducted by the USGS Columbia Environmental Research Center (CERC) (Poulton and others, in press). As part of these studies, which concentrated on mainstem habitats, the Lisbon-Glasgow reach was included in the sampling regime, so the distribution of species among habitats and the species richness between habitats was reasonably well known before the Lisbon wetlands were examined in the present study. The newly created Lisbon Chute was also sampled in 1997 and 1998–99, using both petite ponar methods and a benthic trawl operated as fisheries gear. The goals of most of these studies were to characterize habitat and substrate types, and to develop a comprehensive taxa list for the lower river so that information on overall species richness of invertebrates in all habitats within the flood plain could eventually be obtained. Therefore, some data included in this study report are those from mainstem habitats being studied within the same general time period as the wetlands. When this Lisbon wetland study was initiated, the sampling regime was partially designed to address gaps in our knowledge on flood-plain invertebrates. These gaps included species richness within wetland complexes, examination of species composition as an aid in the further breakdown of wetland types, and the employment of more active sampling methods (sweep and stovepipe) during time periods that included the early and mid-spring season.

Methods

A total of 12 wetland basins were sampled for aquatic invertebrates from March 15–June 17, 1999 (fig. 4-1). Samples were taken only when water was present. Water temperature, water levels, and the presence or absence of inundated vegetation along the wetland margin were recorded during each sampling event. Two sampling methods were used: 1) quantitative samples were taken with a 0.24 m diameter stovepipe sampler to acquire estimates of invertebrate density (#/m^2), and 2) qualitative samples were taken with a 500 micron mesh D-frame sweep net to determine estimates of invertebrate species richness and relative abundance. All samples were preserved in the field with 80% ethanol in labeled, wide-mouth sample containers.

Quantitative Sampling

Of the eight wetlands sampled for invertebrate density, two of these were sampled once per week (Wetlands 8, 9) and the remaining six were sampled once every two weeks (Wetlands 2, 4, 10, 12, 22, 26). At each wetland, 10–12 suitable locations (depth less than 30 cm) along the wetland margin were marked with numbered stakes, and three of these locations were randomly chosen for sampling. The stovepipe was pushed by hand into the substrate to enclose the sampling area, and organisms were removed by sweeping a 500 micron aquarium net in a circular fashion throughout the enclosure for 3 minutes (fig. 4-2A). Organisms and debris were washed from the net into a white pan, concentrated with an ASTM #30 sieve, and placed into the sample

container with preservative. Organisms were sorted from debris under a dissecting microscope with 10x magnification in the laboratory.

Qualitative Sampling

One qualitative sample was taken from each of the 12 wetland basins; four of the wetlands (5, 7, 11, 16) were sampled once per month, and the remaining 8 wetlands were sampled twice per month. For each sample, the D-frame net was swept repeatedly along the margin and emptied into a large white tray (fig. 4-2B). Organisms were hand-picked from the tray with a forceps, including those clinging to the net. An attempt was made to include as many different morphospecies as possible within a 30-minute period, or until approximately 100 organisms were picked. Invertebrates were enumerated and identified in the laboratory to the lowest possible level (usually genus or species) for each sample.

Analysis

To characterize the invertebrate species richness and density associated with the different types of flood-plain wetlands within Lisbon Bottom, and to examine the factors influencing flood-plain invertebrate biodiversity, wetlands were categorized by a combination of permanence, degree of influence from river or creek systems, basin morphology and formation, and vegetation type. Categories include: **1) deep scours** formed by levee breaks with little to no littoral zone or aquatic vegetation, **2) shallow scours** and/or remnants with a significant littoral zone, **3) semi-permanent** wetlands with significant aquatic vegetation that retain water for several months or the entire year, **4) seasonal** wetlands with prominent shoreline vegetation that hold water for at least 2–3 months and may have significant influence from flood-plain creeks, and **5) temporary** basins that are shallow depressions surrounded by moist-soil vegetation and retain water for no longer than 2–3 months after inundation. Because a few of the basins have characteristics that are associated with more than one category, some invertebrate relationships were developed with seasonal and semi-permanent basins combined into one group. This study represents an initial characterization of wetland invertebrate composition, and we have assumed that wetland classification schemes do not always account for a continuum of conditions. Therefore, the comparisons and general chronological trends reported in this chapter are based only on descriptive statistics, pending further analysis.

Samples were also qualitatively compared based on abundance and proportion of species richness for different functional feeding groups of invertebrates (Merritt and Cummins, 1996). These invertebrate categories were used to identify qualitative relationships between wetland type and function (table 4-1). Invertebrate species were subdivided into two basic functional feeding groups: **1) predators** (those feeding on other invertebrates), and **2) herbivore-detritivores** (those feeding on organic matter or living plant tissue). Species were also subdivided into: **A) benthic organisms** (those associated with bottom structure such as vegetation, organic matter, woody debris, or sediment), and **B) pelagic organisms** (those not associated with bottom structure or vegetation and are free-ranging swimmers). A third invertebrate category ("other") included pleuston (species associated with the water surface) and semi-aquatic species (those associated with wetland margins above the water line). Estimates of invertebrate density (# / m^2) were also categorized based on percentiles (high > 5500; moderate = 1430–5500; low < 1430).

In 1999, the Missouri River mainstem and Lisbon Chute were not sampled for invertebrates during the same dates as the wetlands. However, species richness information from previous studies is included here for indicating relative importance of main channel and off-channel areas and their relative contribution to flood-plain biodiversity. The list of invertebrate species found in the mainstem of the Missouri River has been generated from other previous research (Poulton and others, in press), and also includes qualitative collections made from ongoing aquatic invertebrate studies from 1991-present. Part of this research used sampling designs that included stratification by different habitats (wing dikes, revetments, scour holes, sandbars, and the chute) and substrates (rock, sand, muck, organic snags, wood).

Results and Discussion

Wetland Status in 1999

At the beginning of the study in March 1999, several temporary wetlands contained small amounts of water remaining from the October 1998 flood event, which carried water into Lisbon Bottom from both north and south ends. Within the two months before the study, we observed dried filamentous algae on emergent plant stems as evidence that water levels in some scours along the cross-levee (19, 21) had dropped rapidly after river stages declined in December 1998. These wetlands may have a more direct ground-water connection with the river and had dried by the beginning of the study. However, Wetlands 2, 7, 8, and 9 contained water throughout the winter, and Wetland 22 remained hydrated because of the downstream transport of creek water from Wetland 11. All of these wetlands had dried by mid-April just before the first flood pulse.

On April 16, Lisbon Bottom began to flood and became completely covered with water. Water levels in the river receded periodically, and rose over flood stage again in early May (Chapter 2, fig. 2-4). By May 13, river levels had receded further and all of the wetlands were accessible and full of water, including Wetland 10 that requires higher river stages to fill and maintain water presence. Two additional flood pulses occurred that carried additional river water into the wetlands, one in late May and another in late June (fig. 2-4). Most of the temporary wetlands did not dry up until early August after sampling ended.

Invertebrate Response

A total of 260 macroinvertebrate species have been identified within mainstem and off-channel habitats in this river reach, over half of which are unique to the flood-plain wetlands (table 4-2). This total includes mainstem taxa that were reported at the family level in Bataille and others (1999), and were later keyed to genus and species so that they could be included at the same taxonomic level in this report. The mainstem of the Missouri River is dominated by the EPT taxa (Ephemeroptera, Plecoptera and Trichoptera), which make up nearly half of the species present (fig. 4-3). The wetlands at Lisbon Bottom are dominated by groups associated with lentic habitats, such as the Hemiptera, Coleoptera, and Odonata. Chironomid taxa richness between the two areas was similar, but made up a larger percent of the total taxa in the mainstem (fig. 4-3). The Missouri River mainstem and chute also contain some rare invertebrate species that are restricted to very large rivers,

including some sand-dwelling mayflies and snag-dwelling stoneflies that have not been reported from any other locations within Missouri (table 4-2).

Temporary and seasonal wetlands were not expected to contribute to the local pool of overwintering invertebrate taxa because in the absence of an autumn flood, most of these basins would be dry. Fall flooding may also provide a higher degree of plant material conditioning, making organic substrates more attractive for invertebrates when they become inundated in spring (Reid, 1985). Because Wetlands 2, 7, 8, and 9 held water throughout the late fall and winter seasons, samples taken in the early part of the study period harbored pioneer species with high dispersal capabilities (adult beetles and Hemipterans), and a few species with relatively long life cycles (up to one year) that were carried in from the river or from adjacent, more permanent basins (that is, Ephemeroptera: *Hexagenia* spp.). Many of the Hemipterans and beetles that were collected in these wetlands at the beginning of the study normally overwinter in more permanent basins (Wiggins and others, 1980).

The most commonly collected pioneer taxa included the aquatic beetles *Berosus spp.* (Coleoptera: Hydrophilidae), *Peltodytes spp.* (Coleoptera: Haliplidae), and *Tropisternus spp.* (Coleoptera: Hydrophilidae), and the water boatmen *Trichocorixa spp.* (Hemiptera: Corixidae). These taxa were the dominant organisms both in the early part of the study and immediately after temporary basins were inundated after flood pulses. This finding differs from that in bottomland hardwood wetlands of the Mississippi alluvial plain, where amphipods, isopods, fingernail clams and chironomids have been reported as the dominant invertebrates (Batema and others, 1985).

Of the EPT taxa, one group that was expected to inhabit the wetlands more frequently were the Trichoptera, which normally are a diverse and dominant group in lentic habitats. In particular, the case-building families Limnephilidae, Phryganeidae, and Leptoceridae are common inhabitants of wetlands, ponds and weedy lakes in other parts of the U.S. (Wiggins, 1977). However, only one Trichoptera larvae was collected during the entire study. Because most species belonging to these families have life cycles of one year or longer, they probably require more permanent water bodies. Even though the deep scours we studied are permanent basins (4, 26), they may not be suitable for Trichoptera due to their lack of significant submerged or emergent aquatic vegetation. In contrast, there were a total of 12 species of Ephemeroptera collected from the Lisbon wetlands; several mayfly species are bivoltine and can survive periodically in seasonal or semi-permanent basins due to their shorter life cycles. The few individuals of Plecoptera collected during this study from Wetlands 11 and 22 probably drifted in from the creek system flowing through them, because some current exists there during higher flows. Our invertebrate density and species richness estimates in deep scours were low during the early part of the study. In these wetlands (4, 26), water boatmen (Hemiptera: Corixidae) and the glass shrimp (*Palaemonetes kadiakensis*) were the most dominant organisms along the margins and made up a significant portion of the taxa richness until water temperatures rose in early April. It is possible that some species may be using the deeper water as winter refuge and are inactive and not susceptible to capture during March sampling when colder water temperatures predominate.

Taxa Richness

In the wetlands, the number of both predator and herbivore-detritivore taxa peaked during the post-flood period in late May, and the relative proportion of herbivore-detritivore taxa decreased in nearly all

wetland types as the study progressed (fig. 4-4). This result is similar to other research demonstrating that prolonged flooding causes greater changes in herbivores as opposed to predators (Murkin and Kadlec, 1986b), probably due to the succession of vegetation decay. However, the overall distribution of taxa present among all functional groups for the entire study period was similar for each of the wetland types; species richness of benthic herbivore-detritivores was highest (fig. 4-5). Through time, the lowest taxa richness occurred during the flood-pulse period, indicating possible dilution effects from the flooding. However, the highest taxa richness for most of the wetland types occurred after water levels had stabilized in mid- to late May, with the possible exception of seasonal wetlands which, collectively, had the highest taxa richness during the pre-flood period before mid-April (fig. 4-6). Taxa richness in Wetland 12 was consistently high throughout the study; we attribute this to higher diversity of interspersed cover types, which has been suggested as a plausible explanation for increased invertebrate diversity in wetlands (Andrews and Hasler, 1943, Voigts, 1976).

Total taxa richness in each wetland type ranged from 14–78 per time period (mean per sample = 4–42), which is higher than that reported in sweep samples taken at Little Bean Marsh (richness = 7–15), a permanent shallow wetland in the Missouri River flood plain (Heimann and Femmer, 1998). The literature suggests that temporary wetlands are typically low in diversity (Wiggins and others, 1980); our study does not support this. Even though taxa richness may be higher in temporaries during the spring following a fall flood as eluded to earlier, we also observed relatively high taxa richness in Wetland 10 within a few weeks after spring flooding, and this wetland was dry at the beginning of the study. It is also possible that vegetation decay and conditioning that occurs before flood pulses help provide attractive conditions for invertebrates when vegetation becomes flooded.

Benthic herbivore-detritivores also made up the largest percentage of the taxa richness throughout the study period. During the flood pulse, percent of taxa richness of this group actually increased in deep scours (fig. 4-7) and declined in temporary wetlands (fig. 4-8) as opposed to the pre-flood period. In contrast, taxa richness of benthic herbivore-detritivores declined in deep scours and increased in temporary wetlands by the post-flood period in late May. The taxa richness of benthic predators also increased in temporary wetlands during this period (fig. 4-8). Percent of taxa richness for all of the functional groups stabilized after the post-flood period into the summer months, except that pelagic herbivore-detritivores increased slightly in seasonal wetlands and shallow scours by July (figs. 4-9, 4-10).

Semi-aquatic and surface-dwelling invertebrates are expected to increase in importance as air temperatures rise in spring and the wetland margins become covered with new vegetation growth. All wetland types exhibited an increase in percent of taxa richness for these groups between pre-flood and post-flood periods. By June, these invertebrates made up approximately 9–17% of the taxa richness in the wetlands (fig. 4-11).

Abundance and Density

Because invertebrates are opportunistic and are adapted to a wide range of temporal habitat changes, the ratio of abundance of predators to herbivore-detritivores can be used as an indicator of community balance and changes in habitat conditions in wetlands. Herbivore-detritivore invertebrates take advantage of inundated wetland margins and benthic habitats, where organic matter provides a more readily available source of food

and cover. This functional group is used as a food source by predators. In wetlands, when water levels drop sufficiently to the point where margin vegetation is no longer inundated, predators forage on remaining herbivorous invertebrates and become more dominant while herbivore-detritivores decline due to emergence and reduction in cover and organic matter availability. This pattern is evident at Lisbon, where gradual increases in the predator/herbivore-detritivore ratio occurred during the study. However, the ratio declined during flood pulses in deep scours (fig. 4-12).

Similarly, the ratio of benthic and pelagic invertebrate abundances can also indicate available habitat conditions for invertebrate functional groups, because true benthic invertebrates depend on bottom substrates and organic detritus that may become less available as summer progresses. Many of the pioneer invertebrate species with high dispersal capabilities are pelagic predators that can readily colonize new habitats as they become available, and are also the last taxa remaining in wetlands just before they dry up. Within each of the wetland types except the deep scours, the benthic/pelagic ratio peaked during the post-flood period, then declined through June and July (fig. 4-13).

Perennial vegetation, both new growth and that present from previous years, provides both organic matter for herbivorous invertebrates and structure for benthic predators. Availability and conditioning of this organic matter may play a critical role in determining the abundance of invertebrates during and after flood pulses. The moist-soil and woody vegetation along the margin of the Lisbon wetlands became inundated during flood pulses and remained partially flooded for several days after river levels receded. This post-flood period of mid- to late May provided a myriad of new habitats for dispersing and colonizing invertebrates, resulting in substantial increases in invertebrate density. Wetland 12 did not show this pattern, and had invertebrate densities an order of magnitude higher than most other wetlands (fig. 4-14). This wetland, which may be permanent in all but the driest of years, contained the highest diversity of submerged and emergent plants, the most stable water levels, and the least degree of influence from river flooding.

Our data suggest that inundated margin vegetation attracts higher densities and greater overall species richness of invertebrates as compared to periods when water levels in wetlands have declined or are in the process of drying up. For nearly every wetland type, a larger percentage of the number of quantitative samples taken are within the two highest density categories when margin vegetation is inundated (figs. 4-15, 4-16). About two-thirds of the samples taken from deep scours were within the lowest density class when margin vegetation was not inundated (fig. 4-16). In Wetland 12, over 80% of the invertebrate samples were taken when the vegetated margin was inundated, and it had the highest invertebrate densities observed during and throughout the entire study (figs. 4-14, 4-15). On May 3, no wetlands were accessible for sampling, but large densities of mosquito pupae (Diptera: Culicidae) were visible in flooded ditch areas near the roadway. In these areas, invertebrate densities were also among the highest observed during the study (see figs. 4-14, 4-15, 4-16). Temporary and seasonal pools and deep scours also showed an overall increase in invertebrate density after flood pulses receded, although some of this increase could be due to warmer air temperatures and a higher degree of insect activity in May and early June (fig. 4-14). Even though Wetland 11 was not sampled quantitatively, we observed high densities of snails (Gastropoda), particularly in the latter portion of the study from late May through July.

Relationship to Vertebrates

Aquatic invertebrates are an important food resource for fish and wildlife species. Fish may prey on invertebrates when they are present in wetlands, but it is unknown whether invertebrate-feeding fishes are dominant for long enough periods in the Missouri River flood plain to affect invertebrate populations. This group of fishes is sometimes poorly represented in wetlands; the Centrarchidae feed on invertebrates, but in deep scours at Lisbon the dominant species in this family are the crappies (*Pomoxis* spp.) that become piscivorous at a very early age (see Chapter 5). Temporary wetlands of Lisbon Bottom were dominated by various species that are known to be insectivorous (Pflieger, 1997), including shiners (*Notropis* spp. and *Cyprinella lutrensis*) and by orangespotted sunfish (*Lepomis humilis*) and green sunfish (*Lepomis cyanellus*) (see Chapter 5). Winged adult stages of insects are also utilized heavily by bats, many species of birds, and amphibians.

Perhaps the most well studied relationships between wetland invertebrates and higher animals that feed on them are those associated with waterfowl and shorebirds. Wetlands are important for providing invertebrate food resources because migrating waterfowl have higher protein requirements in spring just before nesting and egg-laying (Krull, 1970), and will shift food preferences from plant seeds to a higher protein diet consisting of invertebrates (Murkin and Wrubleski, 1988). Literature also suggests that wetland usage by species such as mallard (*Anas platyrhynchos*) and blue-winged teal (*Anas discors*) is strongly correlated with aquatic invertebrate density (Murkin and others, 1982), especially in the spring season (Murkin and Kadlec, 1986a). However, invertebrate groups that are listed by Eldridge (1990) as being the most often consumed by waterfowl, including Diptera, Gastropoda (snails), and zooplankton, were present at Lisbon, but were not the dominant groups collected in most of the wetlands in this study.

Several researchers have demonstrated that the combined attributes of invertebrate food resource availability, high plant stem density, and adequate interspersion of cover provides optimum conditions for waterfowl (Murkin and others, 1982; Lillie and Evrard, 1994). This presence of flooded vegetation interspersed with open water areas is also known as the hemi-marsh stage of wetland succession (Weller and Spatcher, 1965), a condition used to describe the optimum components needed for maximizing invertebrate productivity and corresponding avian use in shallow water bodies. Others have also demonstrated higher duck foraging frequency (Kaminski and Prince, 1981) in areas with highest invertebrate densities. There is also evidence that the interspersion of cover and structure in wetlands may provide cues to waterfowl that food densities are high (Mack and Flake, 1980; Nelson and Kadlec, 1984). Our research supports the conclusions found in these studies, because highest densities of invertebrates were observed in specific wetlands during time periods and conditions that corresponded with the highest observance of ducks (see Chapter 6). In our study, we observed conditions that are congruent with the hemi-marsh stage described above, both after river flooding (Wetlands 9 and10) and in seasonal wetlands that have high plant-water contact due to their connection with creek systems (Wetlands 11, 12, and 22).

Conclusions and Management Recommendations

The value of wetland invertebrates in organic matter processing, utilization and food-chain support for higher trophic levels has been well documented (Murkin and Wrubleski, 1988). Flood pulses within the Missouri River flood plain have historically occurred from early April through the middle of June in normal rainfall years. The timing of this flood-pulse is critical for supporting the needs of waterbirds because migration takes place regardless of local wetland status or condition. The invertebrate data from Lisbon suggests that the conditions for wetlands that are best for optimal usage by wildlife and fish in general, are periods within 1–2 weeks following the flood pulses when water levels have stabilized and open water areas are interspersed with inundated vegetation. These conditions can also be observed during spring seasons when surface water from creeks or localized rainfall is allowed to inundate moist-soil vegetation. This supports the results of many waterfowl studies that have demonstrated the importance of management techniques that maximize the production of invertebrate foods (Murkin and Kadlec, 1986a; Neckles and others, 1990).

Our data also suggest that temporary and seasonal wetlands may benefit significantly from a fall flood, which occurred in 1998 and also represented a historical condition. The wetlands that held water throughout the winter not only had some duck usage that may not have otherwise occurred during the early spring pre-flood period, but also harbor overwintering invertebrates that act as a local source of pioneer colonizers for recently flooded basins nearby. If the spring pulse does not occur until later in May after the peak of waterfowl migration, or if the river does not rise to a high enough level to fill up the basins, most temporary and some seasonal wetlands will not contain water or a food resource that is significant enough for optimum waterfowl usage. However, this may not be the case with wetlands that have a significant surface water input from creeks entering the flood plain. Wetlands 11 and 12 had the highest diversity of aquatic plants, among the highest species richness of invertebrates, and among the highest observed duck usage (see Chapter 6). These wetlands would likely be usable by waterfowl even if the migration peak takes place before river levels rise enough to fill other, more temporary basins. Scour wetlands (4, 26) were still used as resting areas by ducks even though these basins do not have a significant littoral zone available for invertebrate feeding (see Chapter 6).

The invertebrate data from this study suggests that both organic matter utilization by aquatic invertebrates and high invertebrate food densities can be maximized by managing wetland areas so that margin vegetation can be inundated periodically, especially when warmer water temperatures begin to dominate in spring and the flight dispersal and egg-laying habits of insects is high. This period also coincides with the migration peak for many bird species, and is also congruent with the peak in shorebird activity that occurs shortly after that of waterfowl. Invertebrate species in flood-plain wetlands are adapted to these changes, with either relatively short life cycles or dessicant-resistant stages that allow them to survive in temporary basins, or high dispersal capabilities that enable them to quickly move from more permanent wetlands to newly flooded areas (Reid, 1985). Previous research also suggests that many invertebrates adapted to temporary habitats are among the most important foods for wildlife, yet these species cannot exist in more permanent wetlands (Eldridge, 1990). Two examples of taxa that require habitats subjected to intermittent flooding and drying cycles include mosquito larvae (Diptera: Culicidae) and the clam shrimp (Crustacea: Conchostraca).

There has been very little active management at Lisbon Bottom; the wetland complex has been largely formed by natural processes. Several papers have recommended the management and preservation of entire wetland complexes because they provide a large variety of vegetation and hydration regimes, and will naturally have higher invertebrate diversity and migrating waterfowl usage because the increased range of conditions allow utilization under a wider variety of environmental conditions (Talent and others, 1982; Fredrickson and Reid, 1988). The wider range of conditions in water permanence that wetland complexes may provide also can increase overall invertebrate diversity within the area due to increased niche partitioning and the wide range of life cycle strategies that are used among wetland macroinvertebrates species. The goal for many flood-plain areas along large rivers could be to strive for a maximum diversity of wetland types based on permanence, surrounding vegetation, water source, and basin morphology. Lisbon is already such an area, even though little effort has been spent on management. Some wetlands are always available for fish and wildlife usage, regardless of season, because Lisbon contains wetland basins across a continuum from shallow temporary to deeper more permanent.

Based on the information we now have from the wetland complex at Lisbon Bottom, two primary management techniques or goals can be recommended for optimizing habitat conditions and the diversity and availability of aquatic invertebrates: 1) allow flood-plain creeks that have been historically diverted to re-enter the flood plain and provide an additional source of surface water, while also increasing beneficial moist-soil vegetation, and 2) provide additional routes (or maintain present ones) for river water to periodically enter the flood plain at stages that are lower than the flood level. This would include leaving existing levee breaks in place or adding small notches or crevasses in primary levees where appropriate. One positive result of this would be to increase the relative area and diversity of moist-soil vegetation, a management goal that is often recommended for waterfowl and other wildlife (McCrady and others, 1986). Management of some areas containing flood-plain wetlands has also included techniques that would stabilize water levels at elevations that provide inundated vegetation at the right time periods and/or increase the permanence of some wetland basins that might otherwise not hold water for long enough periods to provide maximum benefit for wildlife. At least two examples of this type of management are already occurring in other areas of the Missouri River flood plain. This type of active management may be appropriate for specific basin types or uses within an area (that is, those previously altered), as long as the dynamic processes of natural flooding and drying are maintained in other wetland basins within the same complex. If wetland succession causes the eventual elimination of some shallow temporaries, or decreases the depth and size of other wetland types (such as scours), providing water level manipulation as a form of active management in some basins would assure continued availability for both macroinvertebrates and other wildlife.

Our results suggest that habitat management and rehabilitation efforts that focus on enhancement of the natural formation of diverse wetland complexes with a wide range of permanence and vegetation types will be of great benefit to macroinvertebrate biodiversity and wildlife value within the Missouri River flood plain. Further, the life history traits of macroinvertebrates present in an individual wetland can give a past history of the water regime, and aid in the classification and management of different wetland types so that their own distinct communities can be maintained.

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Table 4-1. List of basic functional groups of aquatic macroinvertebrates collected at Lisbon Bottom, Spring 1999 (from Merritt & Cummins, 1996). Pr = Predators, Hd = Herbivore/Detritivores, Pe = Pelagic, Be = Benthic, Pl = Pleuston, Sa = Semi-aquatic

Taxonomic group	Families / Genera from Lisbon included	Pr	Hd	Pe	Be	Pl	Sa
Turbellaria (flatworms)	Planariidae	---	X	---	X	---	---
Oligochaeta (segmented worms)	Tubificidae, Naididae	---	X	---	X	---	---
Hirudinea (leeches)	Glossiphoniidae	---	X	---	X	---	---
Nematomorpha (horsehair worms)	Gordiidae	---	X	---	X	---	---
Pelecypoda (clams)	Sphaeriidae	---	X	---	X	---	---
Gastropoda (snails)	Physidae, Lymnaidae, Hydrobiidae	---	X	---	X	---	---
Decapoda (shrimps and crayfishes)	Palaemoniidae, Orconectidae	---	X	---	X	---	---
Amphipoda (scuds)	Gammaridae, Taltridae	---	X	---	X	---	---
Eubranchiopoda (clam shrimp)	Conchostraca	---	X	X	---	---	---
Collembolla (springtails)	Entomobryidae	---	X	---	---	X	---
Hemiptera (true bugs)	Gerridae, Veliidae	---	X	---	---	---	---
Hemiptera (true bugs)	Gelastocoridae, Hydrometridae, Mesoveliidae, Saldidae	X	---	---	---	---	X
Hemiptera (true bugs)	Notonectidae, Belostomatidae, Naucoridae, Corixidae (except Hesperocorixa and Sigara), Pleidae	X	---	X	---	---	---
Hemiptera (true bugs)	Corixidae (Hesperocorixa and Sigara only)	---	X	X	---	---	---
Hemiptera (true bugs)	Nepidae	X	---	---	X	---	---
Ephemeroptera (mayflies)	Tricorythidae, Leptophlebiidae, Baetidae, Ephemeridae, Caenidae, Heptageniidae	---	X	---	X	---	---
Plecoptera (stoneflies)	Perlidae	X	---	---	X	---	---
Trichoptera (caddisflies)	Phryganeidae	---	X	---	X	---	---
Odonata (dragonflies and damselflies)	Coenagrionidae, Lestidae, Calopterygidae, Gomphidae, Libellulidae, Aeshnidae	X	---	---	X	---	---
Megaloptera (alderflies)	Sialidae	X	---	---	X	---	---
Lepidoptera (moths)	Cosmopteridae, Tortricidae, Pyralidae	---	X	---	X	---	---
Coleoptera (beetles)	Dytiscidae, Hydrophilidae (larvae only)	X	---	---	X	---	---
Coleoptera (beetles)	Gyrinidae	X	---	---	---	X	---
Coleoptera (beetles)	Hydrophilidae (adults only), Hydroscaphidae	---	X	X	---	---	---
Coleoptera (beetles)	Noteridae, Scirtidae, Haliplidae	---	X	---	X	---	---
Diptera (horseflies, craneflies, biting midges)	Tabanidae, Tipulidae, Ceratopogonidae	---	X	X	---	---	---
Diptera (soldierflies)	Stratyomyidae	---	X	---	X	---	---
Diptera (phantom midges and shoreflies)	Chaoboridae, Ephydridae	X	---	X	---	---	---
Diptera (Chironomid midges)	Chironomidae (all genera except Parachironomus and Cryptochironomus)	---	X	---	X	---	---
Diptera (Chironomid midges)	Chironomidae (Parachironomus and Cryptochironomus only)	X	---	---	X	---	---
Diptera (mosquitoes)	Culicidae	---	X	---	---	X	---
Orthoptera (grasshoppers)	Tridactylidae, Tettrigidae	---	X	---	---	---	X
Acarina (water mites)	Eylaidae, Hydrachnidae, Axonopsidae	X	---	X	---	---	---

Table 4-2. Complete list of aquatic invertebrate taxa collected at Lisbon Bottom.

Main Group	Family	Genus	Species	Wetland Number[1] 2	4	5	7	8	9	10	11	12	16	22	26	Main-stem[2]	Chute[2]
Turbellaria	Planariidae	*Dugesia*				X											
Nematoda																	
Oligochaeta	Tubificidae			X	X	X	X	X			X	X	X	X		X	
Oligochaeta	Naididae			X	X	X	X	X	X	X				X	X	X	
Hirudinea	Glossiphoniidae															X	
Hirudinea	Glossiphoniidae	*Erpobdella*	*punctata*		X						X	X					
Hirudinea	Glossiphoniidae	*Placobdella*	*papillifera*	X		X					X	X			X		
Hirudinea	Glossiphoniidae	*Marvinmeveria*	*lucida*									X					
Pelecypoda	Sphaeriidae	*Sphaerium*		X	X	X						X				X	
Gastropoda	Physidae	*Physa*		X	X	X	X	X	X	X	X	X	X	X	X		
Gastropoda	Hydrobiidae	*Helisoma*	*trivalis*	X					X		X	X		X	X		
Gastropoda	Lymnaidae	*Pseudosuccinea*															
Gastropoda	Lymnaidae	*Lymnaea*	*obrussa*	X	X			X	X		X	X		X			
Gastropoda	Physidae	*Physella*														X	
Gastropoda	Hydrobiidae	*Somatogyrus*														X	
Decapoda	Orconectidae	*Orconectes*	*luteus*													X	
Decapoda	Orconectidae	*Orconectes*	*virilis*													X	
Decapoda	Orconectidae	*Orconectes*	sp.												X	X	
Decapoda	Orconectidae	*Orconectes*	*immunis*														
Decapoda	Orconectidae	*Palaemonetes*	*kadiakensis*	X	X	X	X	X	X	X		X	X	X	X		
Collembolla	Entomobryidae	*Corynothrix*		X				X									
Collembolla	Entomobryidae															X	
Hemiptera	Corixidae	*Sigara*	sp.													X	
Hemiptera	Corixidae	*Sigara*	*grossolineata*			X	X	X	X		X	X		X			
Hemiptera	Corixidae	*Sigara*	*hubbelli*						X			X	X	X			
Hemiptera	Corixidae	*Sigara*	*alternata*	X		X	X	X	X	X	X		X	X	X		
Hemiptera	Corixidae	*Corisella*					X								X		
Hemiptera	Corixidae	*Hesperocorixa*	*lucida*									X					
Hemiptera	Corixidae	*Hesperocorixa*	*obliqua*	X								X		X	X		
Hemiptera	Corixidae	*Palmacorixa*	*buenoi*		X		X	X	X			X	X		X		
Hemiptera	Corixidae	*Trichocorixa*	*calva*	X	X	X	X	X	X	X	X	X	X	X	X		
Hemiptera	Corixidae	*Trichocorixa*	*kanza*	X	X	X	X	X	X	X	X	X	X	X	X		
Hemiptera	Corixidae	*Ramphocorixa*	*acuminata*	X			X	X	X			X	X		X		
Hemiptera	Notonectidae	*Notonecta*	*irrorata*								X						
Hemiptera	Notonectidae	*Notonecta*	*undulata*			X					X						
Hemiptera	Notonectidae	*Notonecta*	*raleighii*														
Hemiptera	Notonectidae	*Notonecta*	*indica*				X			X	X						
Hemiptera	Notonectidae	*Buenoa*	*confusa*				X										
Hemiptera	Naucoridae	*Pelocoris*	*femoratus*									X	X				
Hemiptera	Saldidae	*Micracanthia*	*humulis*							X							

[1] Numbered wetlands sampled in this study
[2] Data from Poulton and others,in press; and Bataille and others, 1999

Table 4-2. Complete list of aquatic invertebrate taxa collected at Lisbon Bottom–Continued.

				Wetland Number[1]													
Main Group	**Family**	**Genus**	**Species**	**2**	**4**	**5**	**7**	**8**	**9**	**10**	**11**	**12**	**16**	**22**	**26**	**Main-stem[2]**	**Chute[2]**
Hemiptera	Gelastocoridae	*Gelastocoris*	*oculatus*			X	X					X					
Hemiptera	Nepidae	*Ranatra*	*fusca*				X	X			X						
Hemiptera	Nepidae	*Ranatra*	*australis*		X	X	X					X	X		X		
Hemiptera	Belostomatidae	*Belostoma*	*fluminea*	X			X	X	X	X	X	X	X	X	X		
Hemiptera	Mesoveliidae	*Mesovelia*	*cryptophila*			X											
Hemiptera	Mesoveliidae	*Mesovelia*	*mulsanti*		X	X	X							X			
Hemiptera	Gerridae	*Gerris*	*marginatus*	X	X	X	X		X	X	X	X	X	X	X		
Hemiptera	Gerridae	*Trepobates*	*knighti*		X	X	X	X				X		X	X		
Hemiptera	Gerridae	*Rheumatobates*	*rileyi*	X	X	X			X		X	X		X			
Hemiptera	Hydrometridae	*Hydrometra*	*martini*					X				X		X			
Hemiptera	Pleidae	*Neoplea*	*striola*		X	X	X					X			X		
Amphipoda	Gammaridae	*Gammarus*	*lacustris*		X			X	X	X	X	X		X			
Amphipoda	Taltridae	*Hyallela*	*azteca*		X	X	X	X	X		X		X	X		X	
Ephemeroptera	Isonychiidae	*Isonychia*	*sicca*													X	
Ephemeroptera	Oligoneuriidae	*Homoeoneuria*	sp.														X
Ephemeroptera	Tricorythidae	*Tricorythodes*	sp.			X							X			X	
Ephemeroptera	Leptophlebiidae	*Leptophlebia*	sp.				X	X						X		X	
Ephemeroptera	Ephemeridae	*Hexagenia*	*limbata*		X	X	X	X	X		X		X	X	X	X	
Ephemeroptera	Ephemeridae	*Hexagenia*	*bilineata*													X	
Ephemeroptera	Ephemeridae	*Hexagenia*	*munda*					X						X			
Ephemeroptera	Ephemeridae	*Hexagenia*	*atrocaudata*														
Ephemeroptera	Ephemeridae	*Hexagenia*	*rigida*			X	X	X									
Ephemeroptera	Ephemeridae	*Pentagenia*	*vittigera*													X	
Ephemeroptera	Caeniidae	*Amercaenis*	*ridens*													X	
Ephemeroptera	Caeniidae	*Caenis*	*punctata*	X	X	X		X	X		X	X	X	X	X	X	
Ephemeroptera	Caeniidae	*Caenis*	*lattipennis*		X			X			X	X		X		X	
Ephemeroptera	Caeniidae	*Caenis*	*hilaris*													X	
Ephemeroptera	Baetidae	*Baetis*	*intercalaris*													X	
Ephemeroptera	Baetidae	*Baetis*	sp.													X	
Ephemeroptera	Baetidae	*Labiobaetis*	*longipalis*													X	
Ephemeroptera	Baetidae	*Callibaetis*	*fluctuans*	X		X			X	X	X	X	X	X	X		
Ephemeroptera	Baetidae	*Paracleodes*	*minutus*								X		X		X		
Ephemeroptera	Ephemerellidae																
Ephemeroptera	Heptageniidae	*Raptoheptagenia*	*cruenata*													X	
Ephemeroptera	Heptageniidae	*Heptagenia*	*diabasia*			X			X				X				
Ephemeroptera	Heptageniidae	*Heptagenia*	*flavescens*													X	
Ephemeroptera	Heptageniidae	*Stenacron*	*interpunctatum*													X	
Ephemeroptera	Heptageniidae	*Stenonema*	*integrum*		X	X	X	X	X				X	X		X	
Ephemeroptera	Heptageniidae	*Stenonema*	*femoratum*											X		X	
Ephemeroptera	Heptageniidae	*Stenonema*	*pulchellum*													X	

[1] Numbered wetlands samples in this study
[2] Data from Poulton and others, in press; and Bataille and others, 1999

Table 4-2. Complete list of aquatic invertebrate taxas collected at Lisbon Bottom–Continued.

Main Group	Family	Genus	Species	Wetland Number[1] 2	4	5	7	8	9	10	11	12	16	22	26	Main-stem[2]	Chute[2]
Ephemeroptera	Heptageniidae	*Stenonema*	*terminatum*													X	
Ephemeroptera	Heptageniidae	*Leucrocuta*	sp.													X	
Ephemeroptera	Heptageniidae	*Nixe*	sp.													X	
Ephemeroptera	Pseudironidae	*Pseudiron*	*centralis*														X
Ephemeroptera	Baetiscidae	*Baetisca*	*obesa*														X
Ephemeroptera	Potamanthidae	*Anthopotamus*	*myops*													X	
Plecoptera	Capniidae	*Allocapnia*	*granulata*													X	
Plecoptera	Pteronarcyidae	*Pteronarcys*	sp.													X	
Plecoptera	Taeniopterygidae	*Taeniopteryx*	*burksi*													X	
Plecoptera	Taeniopterygidae	*Taeniopteryx*	*parvula*													X	
Plecoptera	Taeniopterygidae	*Strophopteryx*	*fasciata*													X	
Plecoptera	Perlodidae	*Hydroperla*	*fugitans*													X	
Plecoptera	Perlodidae	*Isoperla*	*bilineata*													X	
Plecoptera	Perlidae	*Neoperla*	sp.													X	
Plecoptera	Perlidae	*Perlesta*	*cinctipes*								X			X			
Plecoptera	Perlidae	*Perlesta*	sp.													X	
Plecoptera	Perlidae	*Paragnetina*	*kansensis*													X	
Plecoptera	Perlidae	*Attaneuria*	*ruralis*													X	
Plecoptera	Perlidae	*Acroneuria*	*abnormis*													X	
Plecoptera	Perlidae	*Acroneuria*	*evoluta*													X	
Odonata	Coenagrionidae	*Argia*	*aplicalis*		X	X							X				
Odonata	Coenagrionidae	*Argia*	sp.													X	
Odonata	Coenagrionidae	*Enallagma*	sp.													X	
Odonata	Coenagrionidae	*Enallagma*	*signatum*		X	X	X	X					X	X	X		
Odonata	Coenagrionidae	*Enallagma*	*aspersum*	X				X	X	X		X	X	X			
Odonata	Coenagrionidae	*Enallagma*	*civile*			X			X					X	X		
Odonata	Coenagrionidae	*Ischnura*	*hastata*		X	X		X			X	X					
Odonata	Coenagrionidae	*Ischnura*	*verticalis*		X	X		X	X	X	X	X	X	X			
Odonata	Coenagrionidae	*Ischnura*	*posita*	X	X			X			X	X	X	X			
Odonata	Lestidae	*Lestes*	*rectangularis*	X			X		X	X	X	X		X			
Odonata	Coenagrionidae															X	
Odonata	Calopterygidae	*Hetaerina*	sp.													X	
Odonata	Calopterygidae	*Calopteryx*	*maculatum*								X			X			
Odonata	Gomphidae	*Stylurus*	*plagiatus*			X		X					X			X	X
Odonata	Gomphidae	*Gomphurus*	*externus*													X	X
Odonata	Gomphidae	*Gomphurus*	*ozarkanus*													X	
Odonata	Gomphidae	*Gomphus*	sp.													X	
Odonata	Gomphidae	*Dromogomphus*	sp.													X	
Odonata	Gomphidae															X	
Odonata	Corduliidae	*Neurocordulia*	sp.													X	

[1] Numbered wetlands sampled in this study.
[2] Data from Poulton and others, in press; and Bataille and others, 1999.

Table 4-2. Complete list of aquatic invertebrate taxa collected at Lisbon Bottom–Continued.

Main Group	Family	Genus	Species	Wetland Number[1] 2	4	5	7	8	9	10	11	12	16	22	26	Main-stem[2]	Chute[2]
Odonata	Corduliidae	*Epicordulia*	*princeps*													X	X
Odonata	Corduliidae	*Somatochlora*	*tenebrosa*								X						
Odonata	Libellulidae	*Pachydiplax*	*longipennis*			X						X					
Odonata	Libellulidae	*Libellula*	*luctuosa*									X					
Odonata	Libellulidae	*Libellula*	*pulchella*								X	X					
Odonata	Libellulidae	*Perithemis*	*tenera*			X											
Odonata	Libellulidae	*Plathemis*	*lydia*		X		X	X				X	X				
Odonata	Libellulidae	*Sympetrum*	*vicinctum*									X					
Odonata	Libellulidae	*Erythemis*	*simplicicollis*									X					
Odonata	Libellulidae	*Tramea*	*lacerata*									X					
Odonata	Aeschnidae	*Anax*	*junius*			X					X	X	X				
Odonata	Aeschnidae	*Nasiaeschna*	*pentacantha*		X	X			X								
Odonata	Aeschnidae	*Aeschna*	*umbrosa*					X									
Megaloptera	Sialidae	*Sialis*	sp.				X									X	
Megaloptera	Corydalidae	*Corydalus*	*cornutus*													X	
Lepidoptera																X	
Lepidoptera	Cosmopterygidae	*Pyroderces*	sp.										X				
Lepidoptera	Tortricidae	*Archipes*	sp.							X							
Lepidoptera	Pyralidae	*Crambus*	sp.			X											
Trichoptera	Hydropsychidae	*Cheumatopsyche*	sp.													X	
Trichoptera	Hydropsychidae	*Hydropsyche*	*orris*													X	
Trichoptera	Hydropsychidae	*Hydropsyche*	*simulans*													X	
Trichoptera	Hydropsychidae	*Hydropsyche*	*scalaris*													X	
Trichoptera	Hydropsychidae	*Potamyia*	*flava*													X	
Trichoptera	Leptoceridae	*Nectopsyche*	sp.													X	
Trichoptera	Leptoceridae	*Oecetis*	sp.													X	
Trichoptera	Leptoceridae															X	
Trichoptera	Polycentropodidae	*Neureclipsis*	sp.													X	
Trichoptera	Polycentropodidae	*Paranyctiophylax*	sp.													X	
Trichoptera	Polycentropodidae															X	
Trichoptera	Hydroptilidae	*Hydroptila*	sp.													X	
Trichoptera	Philopotamidae															X	
Trichoptera	Phryganeidae	*Ptilostomis*	sp.				X	X									
Coleoptera	Elmidae	*Macronychus*	sp.													X	
Coleoptera	Elmidae	*Dubiraphia*	sp.													X	
Coleoptera	Elmidae	*Stenelmis*	sp.													X	
Coleoptera	Hydrophilidae	*Tropisternus*	*lateralis*	X					X	X	X	X	X		X		
Coleoptera	Hydrophilidae	*Tropisternus*	*collaris*	X	X	X			X		X	X	X	X	X		
Coleoptera	Hydrophilidae	*Tropisternus*	*blachleyi*		X	X	X	X	X	X		X	X	X			
Coleoptera	Hydrophilidae	*Tropisternus*	*natator*	X	X	X						X		X			

[1] Numbered wetlands sampled in this study.
[2] Data from Poulton and others, in press; and Bataille and others, 1999.

Table 4-2. Complete list of aquatic invertebrate taxa collected at Lisbon Bottom–Continued.

Main Group	Family	Genus	Species	Wetland Number[1] 2	4	5	7	8	9	10	11	12	16	22	26	Main-stem[2]	Chute[2]
Coleoptera	Hydrophilidae	*Tropisternus*	larvae	X	X	X	X	X	X	X	X	X	X	X	X		
Coleoptera	Hydrophilidae	*Berosus*	*pantherinus*	X	X	X	X	X				X	X		X		
Coleoptera	Hydrophilidae	*Berosus*	*striatus*	X	X		X	X	X	X		X		X	X		
Coleoptera	Hydrophilidae	*Berosus*	*infuscatus*	X	X		X					X	X		X		
Coleoptera	Hydrophilidae	*Berosus*	*peregrinus*									X					
Coleoptera	Hydrophilidae	*Berosus*	*pugnax*	X													
Coleoptera	Hydrophilidae	*Berosus*	*ordinatus*		X		X	X							X		
Coleoptera	Hydrophilidae	*Berosus*	larvae	X	X					X	X		X	X	X		
Coleoptera	Hydrophilidae	*Berosus*	sp.													X	
Coleoptera	Hydrophilidae	*Crenitis*	sp.	X						X							
Coleoptera	Hydrophilidae	*Hydrobiomorpha*	sp.			X											
Coleoptera	Hydrophilidae	*Hydrophilus*	larvae	X						X		X		X			
Coleoptera	Gyrinidae	*Dineutus*		X	X	X	X	X	X	X		X	X	X	X		
Coleoptera	Gyrinidae	*Gyrinus*		X				X	X	X			X	X		X	
Coleoptera	Noteridae					X						X		X			
Coleoptera	Scirtidae	*Prionocyphon*	sp.			X						X					
Coleoptera	Scirtidae															X	
Coleoptera	Haliplidae	*Peltodytes*	larvae														
Coleoptera	Haliplidae	*Peltodytes*	*lengi*			X	X		X			X					
Coleoptera	Haliplidae	*Peltodytes*	*edentulus*		X	X		X	X		X	X	X	X	X		
Coleoptera	Haliplidae	*Peltodytes*	*totrulosis*	X	X				X		X	X					
Coleoptera	Haliplidae	*Peltodytes*	*duodecimpunctatus*									X					
Coleoptera	Haliplidae	*Peltodytes*	*sexmaculatus*	X	X	X	X	X				X	X	X			
Coleoptera	Haliplidae	*Halplus*	sp.	X	X		X							X			
Coleoptera	Hydroscaphidae	*Hydroscapha*	sp.								X			X	X		
Coleoptera	Dytiscidae	*Acilius*	sp.											X			
Coleoptera	Dytiscidae	*Laccophilus*	sp.	X	X	X	X	X	X	X	X	X	X	X	X		
Coleoptera	Dytiscidae	*Coptotomus*	sp.	X	X		X		X	X		X	X	X	X		
Coleoptera	Dytiscidae	*Agabetes*	sp.				X	X	X			X		X			
Coleoptera	Dytiscidae	*Anodochelius*	sp.	X							X	X	X				
Coleoptera	Dytiscidae	*Hygrotus*	sp.				X						X	X	X		
Coleoptera	Dytiscidae	*Hydroporus*	sp.	X	X	X	X	X	X	X	X	X	X	X	X		
Coleoptera	Dytiscidae	*Hydrovatus*	sp.	X								X					
Coleoptera	Dytiscidae	*Cybister*	sp.				X					X		X			
Coleoptera	Dytiscidae	*Nebrioporus*	sp.		X								X				
Coleoptera	Dytiscidae	*Hydaticus*	sp.												X		
Coleoptera	Dytiscidae	*Agabus*	sp.	X					X			X		X			
Coleoptera	Dytiscidae	*Oreodytes*	sp.	X		X			X	X				X			
Coleoptera	Dytiscidae	*Copelatus*	sp.			X					X						
Diptera	Tipulidae															X	

[1] Numbered wetlands sampled in this study.
[2] Data from Poulton and others, in press; and Bataille and others, 1999.

Table 4-2. Complete list of aquatic invertebrate taxa collected at Lisbon Bottom–Continued.

Main Group	Family	Genus	Species	Wetland Number[1] 2	4	5	7	8	9	10	11	12	16	22	26	Main-stem[2]	Chute[2]
Diptera	Tipulidae	*Ormosia*	sp.								X						
Diptera	Simulidae	*Simulium*	sp.													X	
Diptera	Simulidae	*Ectemnia*	sp.													X	
Diptera	Tabanidae	*Tabanus*	sp.					X			X						
Diptera	Ceratopogonidae															X	
Diptera	Ceratopogonidae	*Bezzia*	sp.		X	X	X	X	X		X	X	X			X	
Diptera	Ceratopogonidae	*Dasyhelea*	sp.	X													
Diptera	Chironomidae	*Axaris*														X	
Diptera	Chironomidae	*Dicrotendipes*		X	X	X								X		X	
Diptera	Chironomidae	*Chironomus*		X	X	X	X	X	X	X	X		X	X	X	X	
Diptera	Chironomidae	*Tribelos*		X							X		X	X		X	
Diptera	Chironomidae	*Cryptochironomus*		X	X		X	X	X		X		X	X		X	
Diptera	Chironomidae	*Polypedilum*			X	X	X	X	X		X	X	X	X	X	X	
Diptera	Chironomidae	*Endochironomus*		X	X	X	X			X		X		X	X		
Diptera	Chironomidae	*Parachironomus*					X						X			X	
Diptera	Chironomidae	*Glyptotendipes*		X	X	X	X	X	X			X	X	X		X	
Diptera	Chironomidae	*Stenochironomus*														X	
Diptera	Chironomidae	*Paratendipes*														X	
Diptera	Chironomidae	*Paracladopelma*											X		X	X	
Diptera	Chironomidae	*Cladotanytarsus*															
Diptera	Chironomidae	*Paratanytarsus*									X					X	
Diptera	Chironomidae	*Tanytarsus*					X				X		X			X	
Diptera	Chironomidae	*Rheotanytarsus*														X	
Diptera	Chironomidae	*Robackia*														X	
Diptera	Chironomidae	*Chernovskiia*														X	
Diptera	Chironomidae	*Epoicocladius*														X	
Diptera	Chironomidae	*Eukiefferiella*									X	X				X	
Diptera	Chironomidae	*Hydrobaenus*									X			X		X	
Diptera	Chironomidae	*Orthocladius*				X					X			X		X	
Diptera	Chironomidae	*Tvetenia*														X	
Diptera	Chironomidae	*Cricotopus*		X	X	X	X			X		X	X	X		X	
Diptera	Chironomidae	*Rheocricotopus*														X	
Diptera	Chironomidae	*Lopescladius*										X					
Diptera	Chironomidae	*Procladius*		X	X		X	X	X	X	X		X	X	X	X	
Diptera	Chironomidae	*Ablabesmyia*		X	X	X	X				X	X	X			X	
Diptera	Chironomidae	*Alotanypus*														X	
Diptera	Chironomidae	*Pentaneuriella*														X	
Diptera	Chironomidae	*Tanypus*		X		X		X			X		X	X		X	
Diptera	Chironomidae	*Coelotanypus*				X											
Diptera	Chironomidae	*Thienemannimyia*			X									X			

[1] Numbered wetlands sampled in this study.
[2] Data from Poulton and others, in press; and Bataille and others, 1999.

Table 4-2. Complete list of aquatic invertebrate taxa collected at Lisbon Bottom–Continued.

				Wetland Number[1]												
Main Group	**Family**	**Genus**	**Species**	**2**	**4**	**5**	**7**	**8**	**9**	**10**	**11**	**12**	**16**	**22**	**26**	**Main-stem[2]**	**Chute[2]**
Diptera	Chironomidae	*Zavrelimyia*															
Diptera	Chaoboridae	*Chaoborus*		X			X	X	X	X			X	X		X	
Diptera	Empididae	*Hemerodromia*														X	
Diptera	Psychodidae	*Psychoda*														X	
Diptera	Stratyomyidae	*Stratyomys*				X					X						
Diptera	Ephydridae	*Ephydra*						X				X		X			
Diptera	Sciomyzidae								X		X	X					
Diptera	Culicidae	*Haemagogus*		X							X	X					
Diptera	Culicidae	*Anopheles*					X	X	X					X	X		
Diptera	Culicidae	*Culex*										X					
Diptera	Culicidae	*Culiseta*									X						
Hydracarina	Mamsersalidae	*Mamersellides*								X		X					
Hydracarina	Eylaidae	*Eylais*							X			X			X		
Hydracarina	Hydrachnidae	*Hydrachna*												X			
Hydracarina	Axonopsidae	*Albia*											X				
Hydracarina																X	
Nematomorpha	Gordiidae	*Gordius*					X										
Eubranchiopoda	Conchostraca	*Lynceus*						X	X								
Orthoptera	Tridactylidae	*Neotridactylus*	*aplicialis*	X			X	X	X								
Orthoptera	Tettrigidae	*Tettigidea*	*lateralis*	X		X	X	X	X		X	X	X	X	X		

[1] Numbered wetlands sampled in this study.
[2] Data from Poulton and others, in press; and Bataille and others, 1999.

Figure 4-1. Map of Lisbon Bottom wetlands that were sampled for aquatic invertebrates. *Background photo courtesy of U.S. Army Corps of Engineers, Kansas City, MO, March 2000.*

A

B

Figure 4-2. A. Quantitative invertebrate sampling using the stovepipe sampler.
B. Qualitative invertebrate sampling using the sweep net.

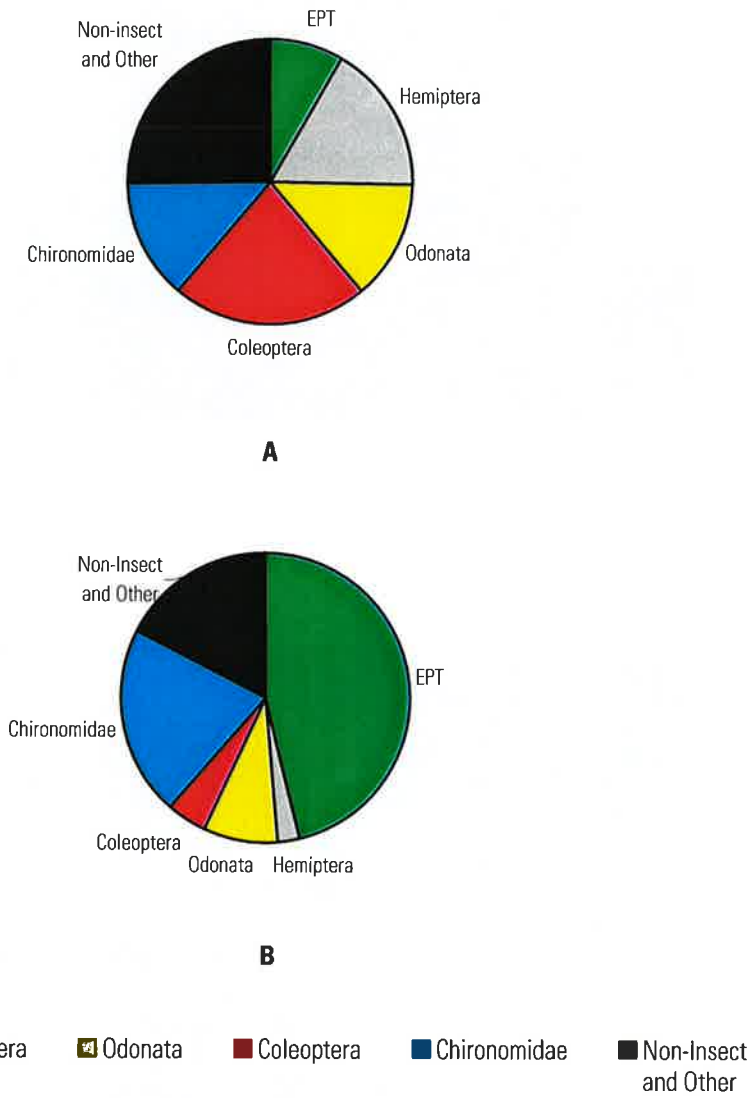

Figure 4-3. A. Number of species among different aquatic invertebrate groups known to exist in the wetlands at Lisbon Bottom. **B.** Number of species among aquatic invertebrate groups known to exist in the mainstem of the Lower Missouri River (EPT= Ephemeroptera, Plecoptera and Trichoptera).

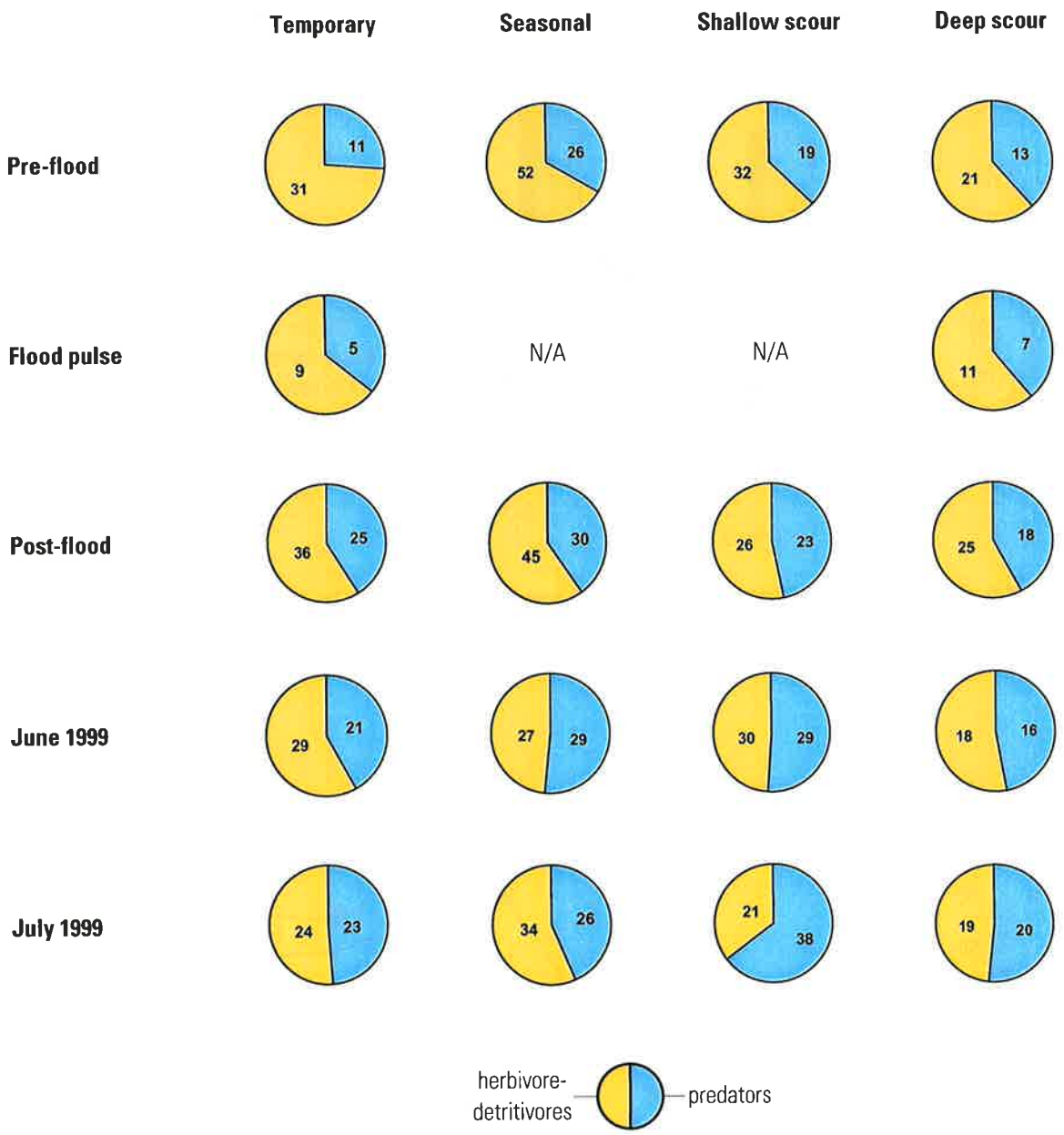

Figure 4-4. Taxa richness of invertebrate predators (in blue on right) and herbivore-detritivores (in yellow on left) over time for the different wetland types at Lisbon Bottom. Temporary wetlands (2, 9, 10) and deep scours (4, 26) were sampled during the flood-pulse period of 4/16 thru 5/13, whereas seasonal and semi-permanent wetlands (8, 11, 12, 22) and shallow scours or remnant wetlands (5, 7, 16) were not accessible for qualitative sampling during that time due to high water.

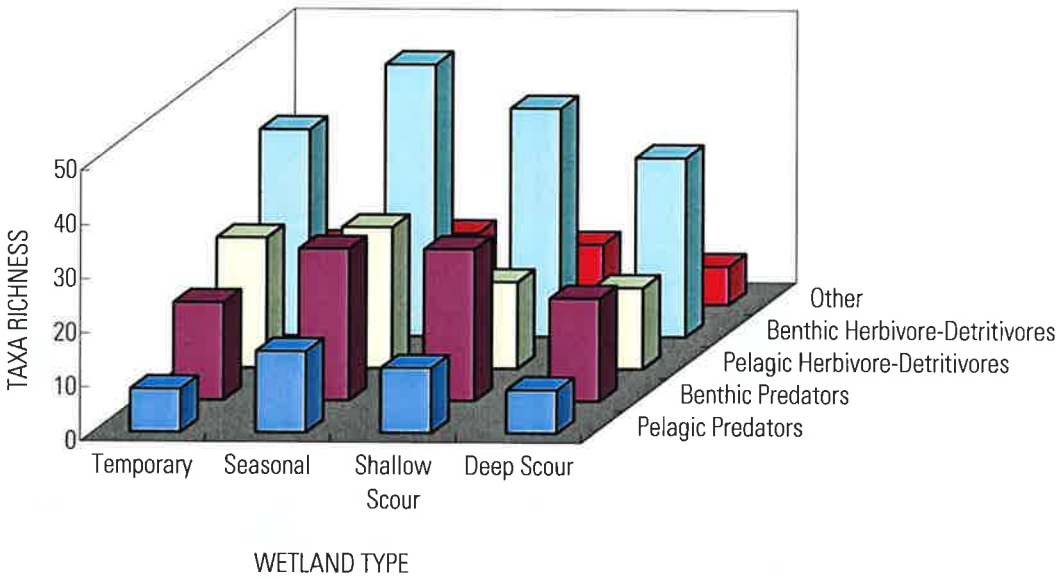

Figure 4-5. Distribution of total invertebrate taxa richness among functional groups for different wetland types at Lisbon Bottom, 1999.

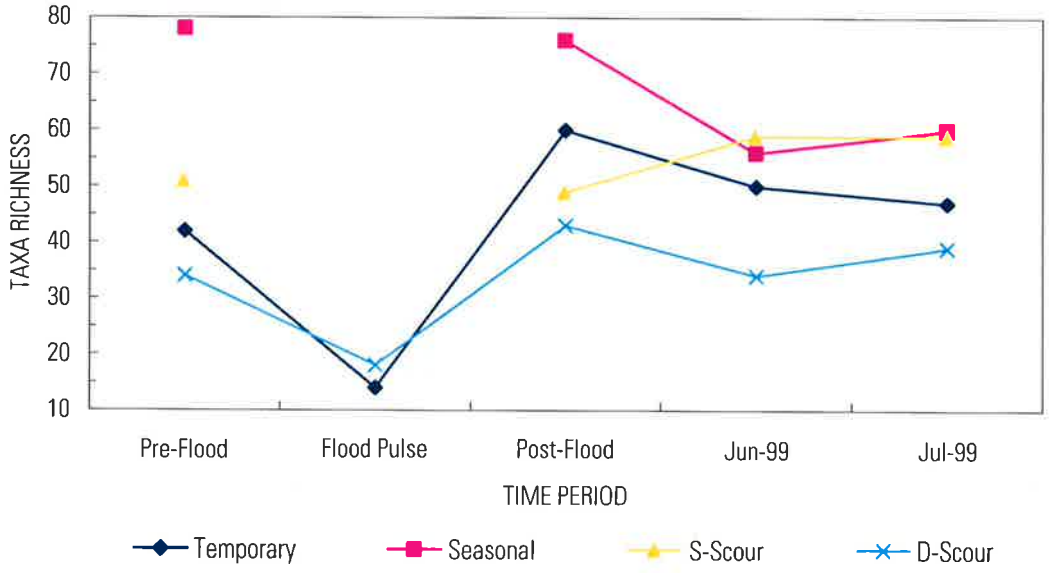

Figure 4-6. Total taxa richness of invertebrates collected over time from different wetland types at Lisbon Bottom in 1999. Temporary wetlands (2, 9, 10) and deep scours (4, 26) were sampled during the flood-pulse period of 4/16 through 5/13, whereas seasonal and semi-permanent wetlands (8, 11, 12, 22) and shallow scours or remnant wetlands (5, 7, 16) were not accessible for qualitative sampling during that time due to high water.

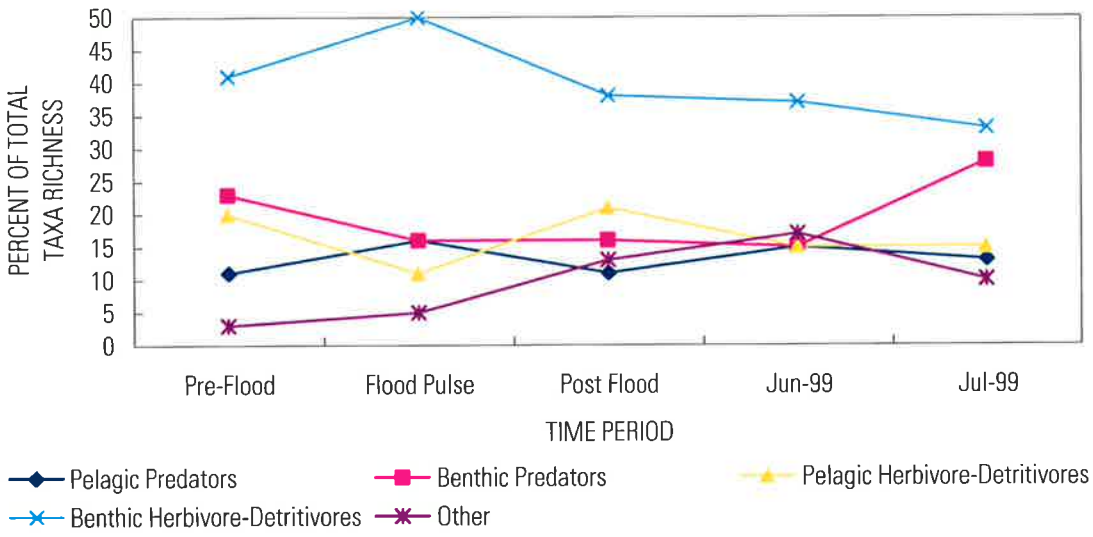

Figure 4-7. Relative percent (%) of the total taxa richness in deep scours (4, 26) of functional feeding groups of invertebrates at Lisbon Bottom. The invertebrate category "other" includes pleuston (surface-dwelling) and semi-aquatic species.

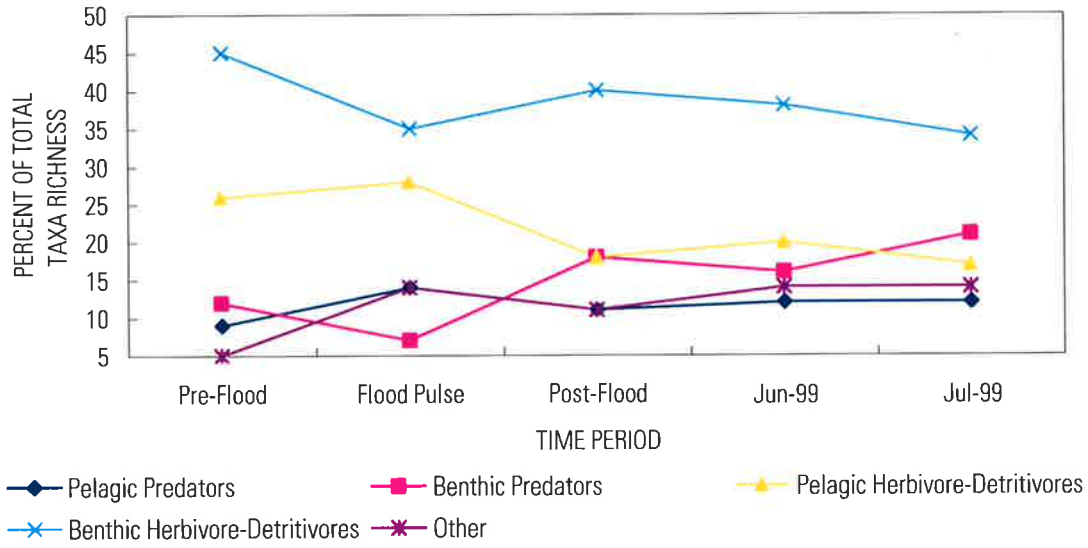

Figure 4-8. Relative percent (%) of the total taxa richness in temporary wetlands (2, 9, 10) of functional feeding groups of invertebrates at Lisbon Bottom. The invertebrate category "other" includes pleuston (surface-dwelling) and semi-aquatic species.

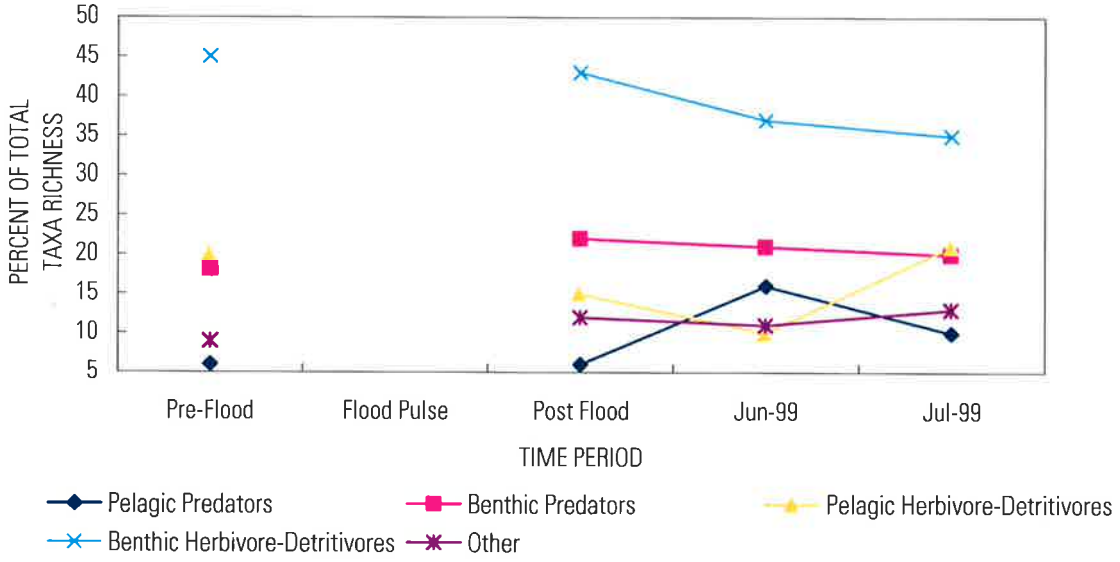

Figure 4-9. Relative percent (%) of the total taxa richness in seasonal and semi-permanent wetlands (8, 11, 12, 22) of functional feeding groups of invertebrates at Lisbon Bottom. The invertebrate category "other" includes pleuston (surface-dwelling) and semi-aquatic species.

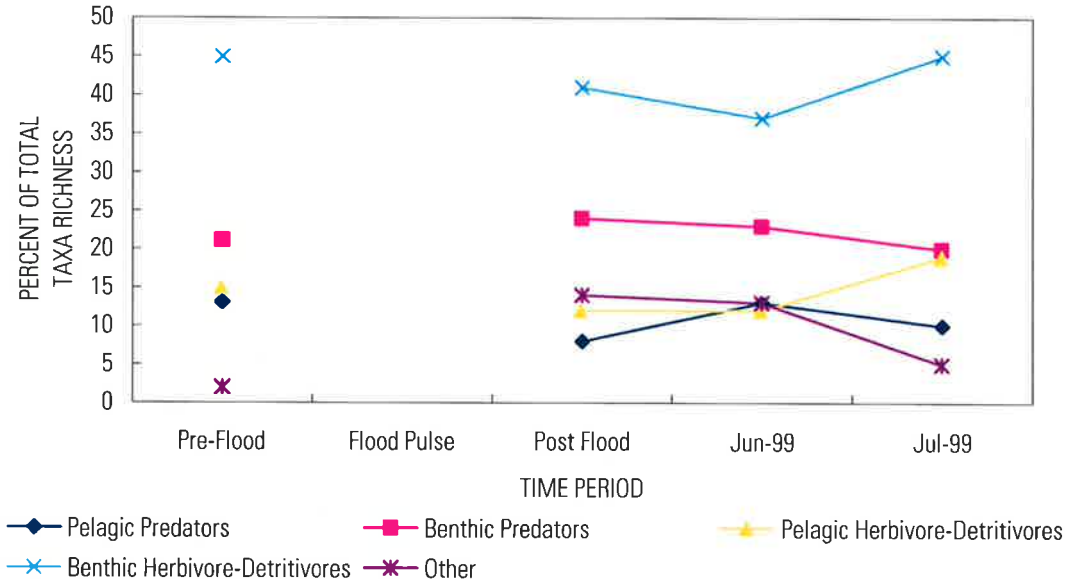

Figure 4-10. Relative percent (%) of the total taxa richness in shallow scours and remnant wetlands (5, 7, 16) for functional feeding groups of invertebrates at Lisbon Bottom. The invertebrate category "other" includes pleuston (surface-dwelling) and semi-aquatic species.

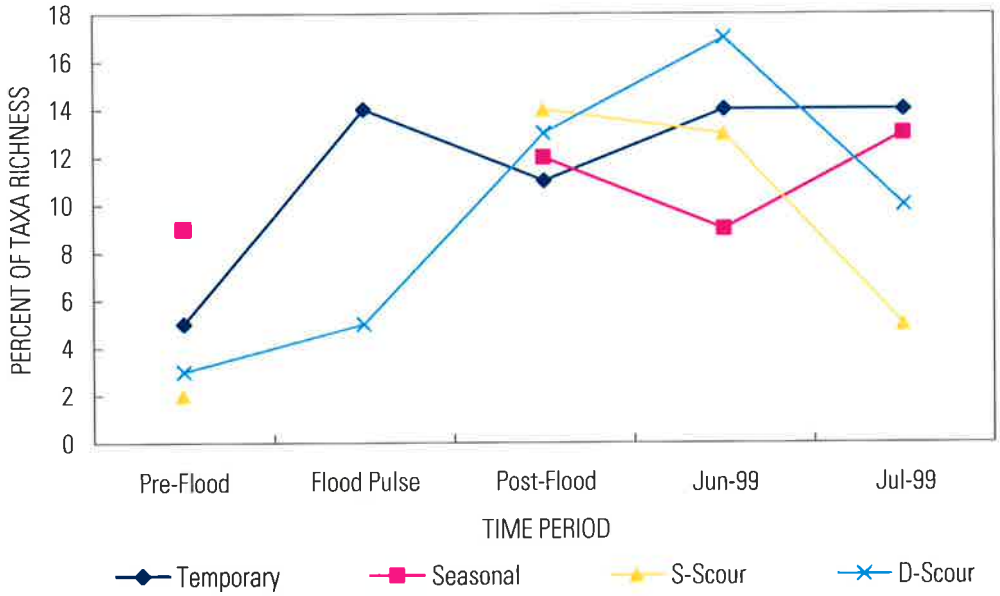

Figure 4-11. Relative percent of semi-aquatic and surface-dwelling (pleuston) invertebrates based on taxa richness observed in different wetland types at Lisbon Bottom in 1999. Temporary wetlands (2, 9, 10) and deep scours (4, 26) were sampled during the flood-pulse period of 4/16 through 5/13, whereas seasonal and semi-permanent wetlands (8, 11, 12, 22) and shallow scours or remnant wetlands (5, 7, 16) were not accessible for qualitative sampling during that time due to high water.

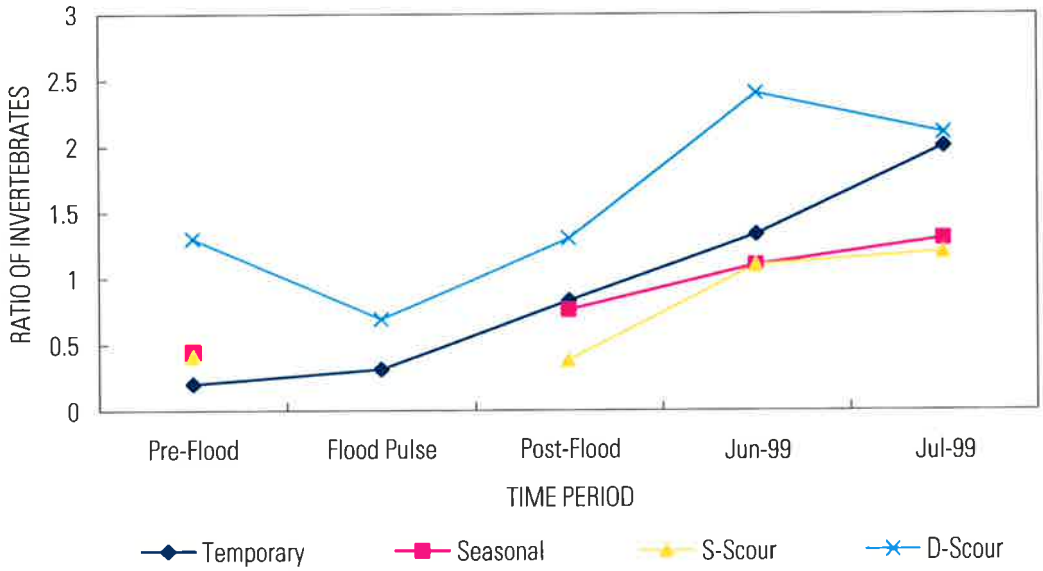

Figure 4-12. Ratio of predator / herbivore-detritivore abundances of invertebrates collected from different wetland types at Lisbon Bottom in 1999. Temporary wetlands (2, 9, 10) and deep scours (4, 26) were sampled during the flood-pulse period of 4/16 through 5/13, whereas seasonal and semi-permanent wetlands (8, 11, 12, 22) and shallow scours or remnant wetlands (5, 7, 16) were not accessible for qualitative sampling during that time due to high water.

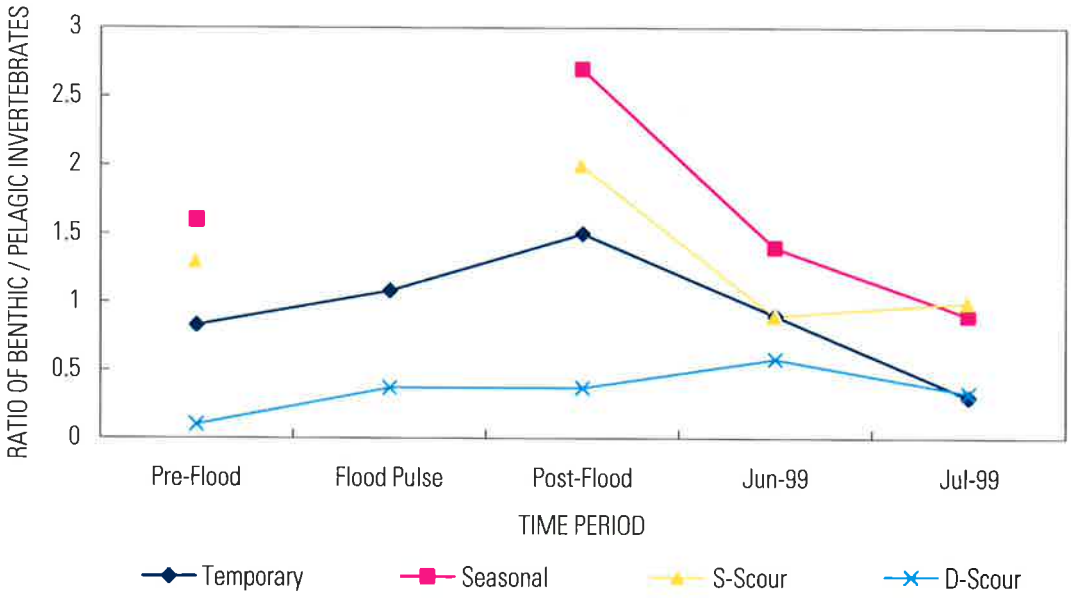

Figure 4-13. Ratio of benthic / pelagic abundances of invertebrates collected from different wetland types at Lisbon Bottom in 1999. Temporary wetlands (2, 9, 10) and deep scours (4, 26) were sampled during the flood-pulse period of 4/16 through 5/13, whereas seasonal and semi-permanent wetlands (8, 11, 12, 22) and shallow scours or remnant wetlands (5, 7, 16) were not accessible for qualitative sampling during that time due to high water.

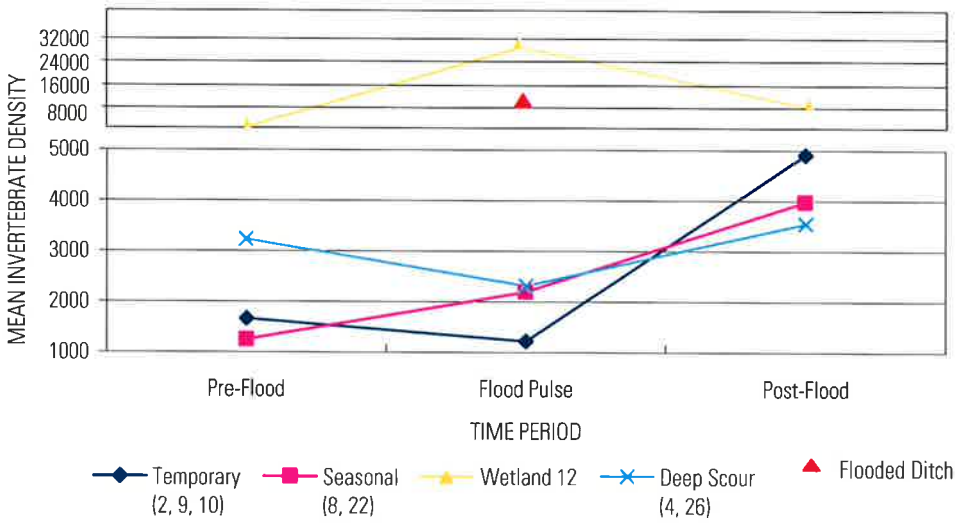

Figure 4-14. Mean invertebrate density (#/m^2) determined from stovepipe samples taken from different wetland basin types at Lisbon Bottom in 1999. During the peak of the flood-pulse period (4/16 – 5/13, with highest stage on 5/3/99) no wetland basins were accessible for quantitative sampling due to high water, and two locations along a flooded ditch were sampled to document high densities of invertebrates observed.

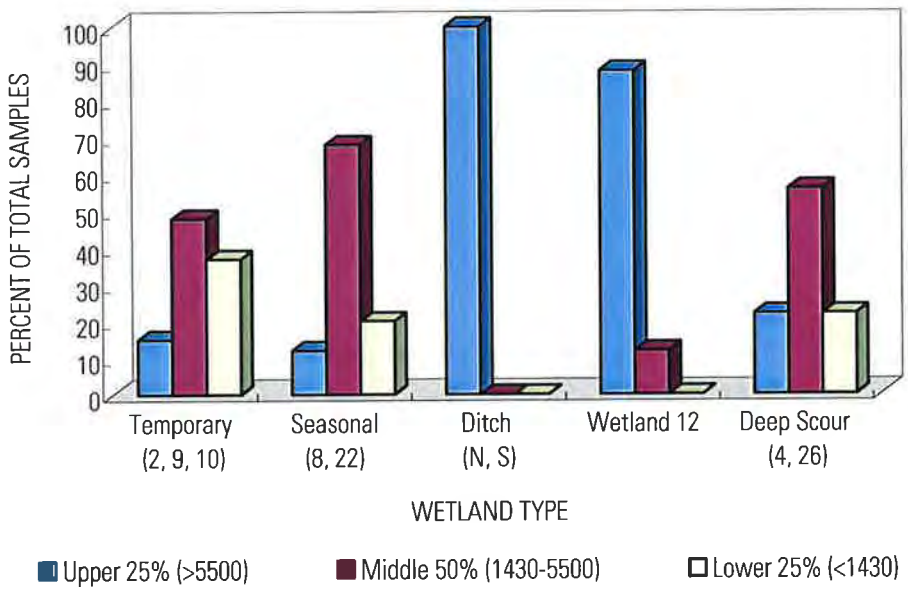

Figure 4-15. Frequency distribution of density classes (percent of total samples taken within a wetland type in #/m^2) for Lisbon aquatic invertebrate samples collected in 1999 during periods when vegetation along wetland margins was inundated.

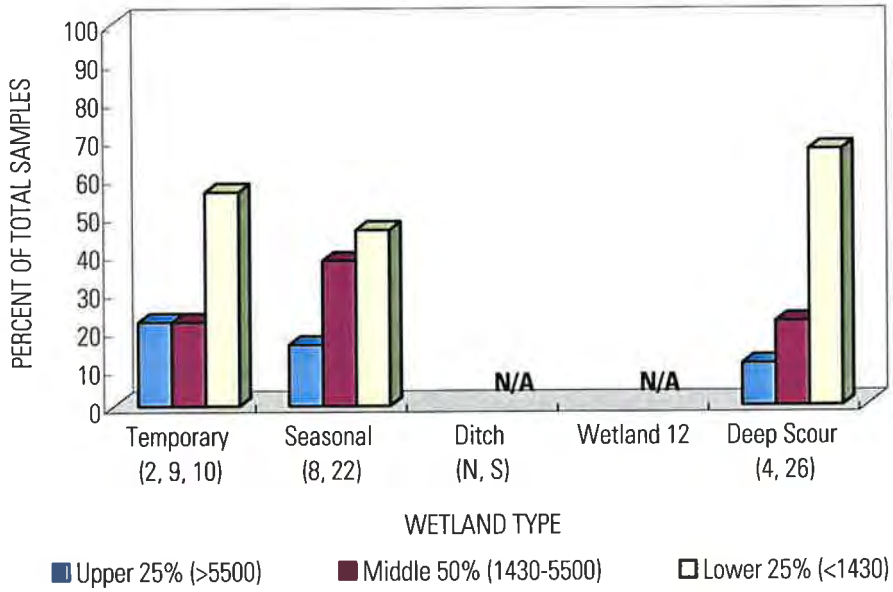

Figure 4-16. Frequency distribution of density classes (percent of total samples taken within a wetland type #/m^2) for Lisbon aquatic invertebrate samples collected in 1999 during periods when vegetation along wetland margins was above the waterline and not inundated.

Chapter 5. Fishes of Lisbon Bottom Wetlands

Duane C. Chapman

Abstract

Fish were collected and identified in permanent and temporary wetlands and ephemerally flooded areas of Lisbon Bottom, Missouri. in order to 1) Determine the seasonal use of Lisbon Bottom by flood-plain-dependent fishes, 2) compare the fish assemblage of the flood plain to that of the main river, and 3) to examine wetland fidelity of lentic, flood-plain fishes. Lisbon Bottom consists of approximately 875 ha of flood plain within a single bend of the Missouri River, with a variety of types of wetlands with different water sources and periods of flooding. Forty species of fish were captured in the wetlands. Buffalos (*Ictiobus* sp.), gizzard shad (*Dorosoma cepedianum*), common carp (*Cyprinus carpio*), and shortnose gars (*Lepisosteus platostomus*) were observed spawning and were captured exuding sex products during flood events. Therefore this flood plain, which was reconnected to the river after the 1993 floods is used by fishes from the river for spawning.

Fishes found in the flood-plain wetlands were very different from fishes captured in the Missouri River and Lisbon Chute at the same time period by another researcher (Louise Mauldin, U.S. Fish and Wildlife Service). Gizzard shad composed less than three percent of the fish captured in the wetlands, but almost seventy percent of the river and chute fish. Striped bass (*Morone saxatilis*) and white bass (*Morone chrysops*) were not captured in the wetlands but were common in the river and chute. Freshwater drum (*Aplodinotus grunniens*) was also common in the river and rare in the flood plains. In contrast, the flood-plain wetlands had higher relative abundances of cyprinids (especially *Notropis* sp. and *Cyprinellis* sp.) and centrarchids.

Although all of the wetlands except Wetland 12 were connected with a single sheet of water during the highest floods, fish species captured varied between wetlands. Basins with similar morphology and water sources generally had similar species of fish. Ephemerally flooded areas held fish during floods, but shallow very ephemeral basins left by the retreating floodwaters did not hold fish, an indication that fish have mechanisms to avoid being stranded in the very shallow areas. However, many slightly deeper wetlands apparently did concentrate fish that were unable to escape as the wetlands eventually dried. One topflooding wetland that eventually dried held many riverine fishes such as blue sucker (*Cycleptus elongatus*), sauger (*Stizostedion canadense*), and goldeye (*Hiodon alosoides*), that were most likely trapped there after a flood.

Small native cyprinids, especially red shiner (*Cyprinella lutrensis*), dominated the temporary wetlands in this study. Young-of- the-year (YOY) buffalo occupied shallower, more ephemerally flooded habitats than did small cyprinids. Orangespotted sunfish (*Lepomis humilis*) and green sunfish (*Lepomis cyanellus*) were also common in temporary wetlands.

White and black crappies (*Pomoxis annularis* and *P. nigromaculatus*) were the dominant large species in the topflooding permanent scours and were abundant in the backflooding permanent scour. These are the most likely fish in the wetlands to be targeted by recreational fishermen. Crappie were large and growth rates were rapid, despite temperatures that were significantly higher than those known to provide good crappie survival and growth. Tagged crappie were never captured in a wetland other than that in which they were tagged, and large crappie were

generally not caught in temporary wetlands, but the data were insufficient to determine the degree of wetland loyalty.

Introduction

The Lower Missouri River ecosystem has drastically changed over the past 50 years due to construction of main-stem impoundments, channelization, bank stabilization, and concomitant flow alteration. Confinement and straightening of the channel has led to the loss of approximately 72 km of river from Rulo, Nebraska to the mouth of the river at St. Louis (Funk and Robinson, 1974) and a loss of 41,000 ha of aquatic habitat (Hesse and others, 1989). Channel confinement has led to an increase in average current velocities and a loss of shallow backwater and sand island habitat. In addition, levee construction has led to a disconnection of the river with its flood plain which is known to be critical in the reproduction and recruitment of flood-plain-dependent fishes (Junk and others, 1989; Galat and others, 1998).

These combined alterations have led to significant changes in the fish communities of the Lower Missouri River (Pflieger and Grace, 1987; Hesse and others, 1989). Numerous fish species including buffalo (*Ictiobus* sp.) and carpsuckers (*Carpiodes* sp.) that are dependent on vegetated flood-plain habitats for spawning have declined (Pflieger and Grace 1987; Galat and others, 1998). However, major recent floods have served to reconnect portions of the Lower Missouri River flood plain with the river. In 1993 and 1995 floodwaters breached many levees along the Missouri River and created new habitats including shallow erosional depressions and deep, steep-sided evorsions known as "blew-holes" or "scours" that sometimes exceeded 30 m in depth. Smaller, shallower, scours occurred at secondary breaks in levees or around other obstructions on the flood plain. When the floodwaters receded these basins became a diverse assortment of wetlands ranging from shallow, vegetated temporary wetlands to both connected and non-connected scours.

Connected and non-connected scours have been the focus of several recent studies and a few of these have addressed fish assemblages. Tibbs and Galat (1997) and Galat and others (1998) investigated fish use of connected and non-connected scours and found that connected scours contained greater species richness of fishes compared to non-connected scours. In addition, the fish assemblages of these two habitat types have been shown to be quite different (Galat and others, 1998). Gelwicks (1995) studied seasonal fish use of a managed wetland complex of the Lower Missouri River and found that these habitats were extensively used by flood-plain spawners including gizzard shad (*Dorosoma cepedianum*) and *Cyprinus* sp. To date, however, there have been few studies of the entire continuum of wetland types that occurs in reconnected flood-plain habitats of the Lower Missouri River. In this study, we investigated the fish use of a series of temporary, seasonal, and permanent wetlands located at Lisbon Bottom, near Glasgow, MO. This study comprised the entire period of pre-flood, flood, and post-flood conditions during Spring 1999. Lisbon Bottom consists of approximately 875 ha of flood plain within a single bend of the Missouri River, and was recently purchased by the U.S. Fish and Wildlife Service as part of the Big Muddy National Fish and Wildlife Refuge. The Lisbon tract is passively managed as a reconnected flood-plain complex. Breaks in the levees on this bend of the river have not been repaired which makes Lisbon Bottom a natural laboratory to examine seasonal use of the flood plain by a large-river fish community. This study had the following objectives: 1)

Determine the seasonal use of Lisbon Bottom by flood-plain-dependent fishes, 2) compare the fish assemblage of the flood plain to that of the main river, and 3) to examine wetland fidelity of lentic, flood-plain fishes.

Methods

Fish were sampled with minifyke nets, trammel nets, three sizes of hoop nets, an aquarium net, and a bag seine. Metal minnow traps with leads were also used but were abandoned early in the study because they were not effective. Two sizes of the hoop nets used were the "large" and "small" hoop nets described by Gutreuter and others (1995). An intermediate size hoop net (3 ft diameter, 1 inch bar mesh, single throat) was also used. The minifyke net (fig. 5-1) and the trammel nets (fig. 5-2) are also those described by Gutreuter and others (1995). Leads (50 ft long, 1 inch bar mesh on the large and medium hoop nets and ¾ inch bar on the small hoop nets) were used on the hoop nets to guide fish into the nets.

Minifykes and hoop nets were generally placed perpendicular to the shoreline with the mouth facing the shoreline and the lead running to the bank. In some cases, especially during flooding episodes (fig. 5-3), an appropriate bank did not exist. In that case, two net/lead systems were attached together, face to face. Hoop and fyke nets were set overnight, and two consecutive nights were always fished at the same location.

Nine locations were sampled using passive gear. These consisted of three permanent scour wetlands (4, 5, and 26), five temporary wetlands (2, 3, 8, 9 and 22), and an ephemerally flooded area (E5) (fig. 5-4). Descriptions of the permanent and temporary wetlands may be found in Chapter 2, table 2-1. All or portions of those wetlands were also sampled using a 50 ft, 1/8 inch ace mesh bag seine, depending on the morphology of the wetland and the location and abundance of submerged vegetation. Trammel nets (50 ft long, 4 ft deep, 1 inch bar mesh on the small weave netting and 12 inch on the large weave netting) were set for approximately two hours, and at least two trammel sets were used in each location. Wetlands 2 and 3 were connected due to flooding at the time of sampling, and were fished as one unit, with the larger passive gear installed in Wetland 3. Wetland 2 is much shallower than Wetland 3 and more ephemerally inundated.

Four shallow (< 20 cm deep) ephemerally flooded areas (Wetlands E2, E3, E4, and E6, fig. 5-4) were seined in their entirety with the bag seine after the retreat of floodwaters to look for stranding of small fish. Wetland 21, a moderately deep temporary scour, was also sampled only by seining. Lastly, in Wetland 10 and ephemerally flooded area E1, visible schools of small fish were captured using an aquarium net. In these very shallow wooded wetlands, none of our gear other than the aquarium net would have been appropriate because of the abundance of small trees, leaf litter and woody debris.

Access to these wetlands is extremely difficult for much of the year, often being too muddy and rugged even for all-terrain vehicle access (fig. 5-5). Nets and gear often had to be carried by hand for up to a kilometer over muddy terrain. This limited the amount of sampling that was possible. Trap style nets were fished for two consecutive nights at the same location. Permanent scour Wetlands 4, 5, and 26 were each fished for two sampling occasions, with a cumulative total of at least 16 hoop-net nights. Minifyke nets were fished for a total of 8 net-nights each in Wetlands 4 and 5. Temporary and ephemeral wetlands were fished on one sampling occasion only and generally for half as many total hoop net nights. Naturally, wetlands were fished during periods in which they were inundated or contained water. Wetland morphology and vegetation precluded the use of certain gear in

different wetlands. For example, minifyke nets were not used in Wetland 26 because the steep sides of this wetland made such gear inappropriate. In Wetlands 9 and 22, only minifykes, seines, and minnow traps were used because of shallow water depths.

Seine hauls were very different in length, and also in width, depending on the physical characteristics of the wetlands being fished. It was not feasible to standardize this aspect of fishing between wetlands because seining is very dependent on an adequate bottom, absence of snags, an adequate shoreline for entrapment, depth, and other factors. We recorded the length of seine hauls, but they are not reported here because we believe that any attempt to standardize these results to catch per unit effort would not be useful. Catch efficiency was strongly affected by many factors other than length of haul. Similarly, the efficacy of the other gears depended greatly on wetland morphology and, in the case of the hoop and fyke nets, by beaver and muskrat activity. Therefore, the data herein are not discussed in terms of catch per unit effort, but rather in terms of presence-absence and relative abundance.

Large fish were identified on site, measured, and released. Centrarchids, catastomids, and shortnose gar (*Lepisosteus platostomus*) exceeding 100 mm in length were tagged before release, using individually numbered Floy®T-bar tags. Scales were taken from centrarchids and catastomids. Using methods of Carlander (1982), fish were aged and lengths at age were back calculated. Tag return data and length at age were used in an attempt to determine the fidelity of fish to the wetland in which they were captured. Since these wetlands differ in their hydrology, it is likely that growth rates of fish may differ between wetlands. Back-calculated length-at-age was compared, using Duncan's multiple range test, between wetlands as an attempt to determine fish loyalty to a given wetland. If back-calculated growth of similar-aged fishes is similar within ponds, but different between ponds, this would be evidence of pond loyalty. Adult fish were examined in the field for evidence of spawning activity, (exudation of sex products, spawning coloration or tubercles) and in some cases fish were actually observed spawning during the collections.

If circumstances allowed, small fish were identified on site and released, but generally time and accuracy of identification required that the fish be preserved on-site for later identification. Preserved fish were fixed on site with formalin, and later rinsed and transferred to ethanol. Louise Mauldin, U.S. Fish and Wildlife Service, confirmed the identification of small fish, and Matthew Winston, Missouri Department of Conservation, confirmed the identification of YOY stonerollers.

Duncan's multiple range test was used to test for differences between back-calculated length of fish between wetlands, and a Student's *t* test was used to test differences in length between black and white crappie of similar ages within a wetland. Ward's minimum variance cluster analysis was used to group wetlands by similarity of the fish species assemblages.

Results and Discussion

Forty species of fish were collected (table 5-1) including large and small fishes caught by all methods. Fish captured on Lisbon flood-plain wetlands differed greatly from fish caught in the Missouri River adjacent to Lisbon Bottom and in Lisbon Chute by another researcher during the same time period (fig. 5-6, Louise Mauldin, U.S. Fish and Wildlife Service). Gizzard shad composed almost 70% of the catch in the river and chute. Adult gizzard shad were uncommon in the wetlands and were encountered mostly during flood events when they were actively

spawning. Gizzard shad YOY were only beginning to recruit into the flood-plain wetland catch by the end of the study. For purposes of easier comparison, figure 5-6B shows the same data as figure 5-6A, with the influence of gizzard shad removed. White bass and striped bass (Moronidae; *Morone chrysops* and *M. saxatilis*) and freshwater drum (Sciaenidae; *Aplodinotus grunniens*) were abundant in the river and chute, while moronids were not captured and sciaenids were rare in the wetlands (fig 5-6B). Native cyprinids, especially shiners, had a higher relative abundance on the flood plain than in the river and chute. Relative abundance of non-native cyprinids (excluding influence of gizzard shad) was about the same between the flood plain and the riverine environments. Centrarchids also had a much higher relative abundance on the flood plain. Fishes in addition to white bass and striped bass which were caught in the riverine environments but not on the flood plain include blue catfish, (*Ictalurus furcatus*), bigmouth shiner (*Notropis dorsalus*), brassy minnow (*Hybognathus hankinsoni*), shovelnose sturgeon (*Scaphirhynchus platorynchus*), mooneye (*Hiodon tergisus*), paddlefish (*Polyodon spathula)*, river shiner (*Notropis blennius*), silverband shiner (*Notropis shumardi*), silver carp (*Hypophthalmichthys molitrix*), silver chub (*Macrohybopsis storeriana*), sturgeon chub (*Macrohybopsis gelida*), and suckermouth minnow (*Phenacobius mirabilis*). Fishes that were caught on the flood plain but not in the river or chute include channel shiner (*Notropis wickliffi*), golden shiner (*Notemigonus chrysoleucas*), mimic shiner (*Notropis volucellus*), and white sucker (*Catostomus commersoni*).

During the spring floods, we observed gar (*Lepisosteus* sp.) and buffalo actively spawning in the wetlands flooded by the river, and we captured bigmouth buffalo (*Ictiobus cyprinellus*), common carp (*Cyprinus carpio*) (fig. 5-7A), and gizzard shad (fig. 5-7B) that were releasing sex products when captured during flood events. There is little doubt that riverine fish can and do spawn on this recently reconnected flood plain.

For the purposes of analysis, we considered small and large fish separately. Small fish were defined as fish less than 80 mm total length, regardless of species. This division was chosen because it was a natural break in the data. It included almost all young-of-the-year (YOY) fish, all native cyprinids and all orangespotted sunfish (*Lepomis humilis*).

Small Fish

Over 2300 small fish were captured in Lisbon Bottom wetlands. Table 5-2 provides data on total number of fish, fish families, and fish species caught in each wetland. The entire dataset (Attributes of USGS 1999 Small fish species relative abundance) are reported in Korschgen and others (ArcView-based spatial decision support system for the Lisbon Bottom Unit of the Big Muddy National Fish and Wildlife Refuge, unpub. data, 2001).

Ephemerally flooded areas E2, E3, E4, and E6 were shallow depressions that held water briefly after the retreat of floodwaters. We seined these wetlands to determine whether they held trapped fish. The water depth of these depressions at the time of seining did not exceed 20 cm. No fish were captured in these drying areas, nor were any fish observed but not captured. These areas held some vegetation, mainly young cockleburs and dead cockleburs from the previous year. The vegetation interfered with seining and some fish may have escaped. However, we feel that if significant numbers of fish had been present at the time of sampling, that some fish would have been captured. There were very few, if any, fish remaining in these small depressions. We did catch small and large fish in ephemerally flooded areas E1, E5, and ephemeral Wetlands 2 and 10, an indication that fish do use the entire bottom when it is available to them. Therefore, fish probably have mechanisms to avoid entrapment in these

small depressions. If not, E2, E3, E4, and E6 would probably have concentrated large numbers of fish as the floods receded. However, many slightly deeper wetlands (8, 9, 21) apparently did concentrate fish that were unable to escape as the wetlands eventually dried.

Other sites where few or no small fish were captured were Wetlands 4 and 5, and ephemerally flooded area E5. No small fish were captured in Wetland 4 and only five small fish (all centrarchids; three orangespotted sunfish, one juvenile white crappie [*Pomoxis annularis*] and one juvenile bluegill [*Lepomis macrochirus*]) were captured in Wetland 5. However, schools of unidentified larval fish were seen in both of these wetlands. The lack of small fish in the catch in these wetlands stems partly from the inefficacy of the available methods used for the capture of small fish in these wetlands. Seines and minifykes were the primary gears used in this study for the capture of small fish. Neither of these scour wetlands was seined; Wetland 4 was too deep, steep-sided and full of woody snags, and Wetland 5 had very deep unconsolidated mud that made effective seining impossible. Minifykes were set in these wetlands, but the steep sides of Wetland 4 made them ineffective because the tops of the traps were inundated. However, it is likely that the number of large predatory fish in these wetlands (mostly white crappies, black crappies [*Pomoxis nigromaculatus*] and some largemouth bass [*Micropterus salmoides*]) is at least partly responsible for the low numbers of small fish and the complete absence of small cyprinids and catastomids in the catch. E5 was a normally terrestrial area of the bottom that was sampled during a major flood event. The seine was not used at E5 because the entire area was covered with temporarily submerged small trees and other vegetation that would have made seining impossible. At E5, one red shiner (*Cyprinella lutrensis*), one emerald shiner (*Notropis atherinoides*) and two western mosquitofish (*Gambusia affinis*) were caught in minifykes.

Other than E2–E6 and Wetlands 4 and 5, at least 40 small fish were captured per wetland (table 5-2). Figure 5-8 shows the relative abundance of small fishes in wetlands where 40 or more small fish were caught. Cyprinids and centrarchids replaced catastomids with increasing permanence of the wetland. All the small catastomids caught in the ephemeral wetlands in this study were YOY bigmouth buffalo, whereas the cyprinids were very diverse (fig. 5-9). Young-of-the-year bigmouth buffalo apparently selected for these very shallow habitats, which were often highly vegetated and full of leafy and woody debris. Schools of hundreds of these fish could be seen in these areas.

Wetland 21 had the highest small fish family diversity (table 5-2 and fig. 5-8) with seven families, at least one species of every family captured in the study except Poeciliidae. This is to some measure due to the topflooding nature of this wetland. This wetland contained juvenile sauger (*Stizostedion canadense*), blue suckers (*Cycleptus elongatus*), and goldeye (*Hiodon alosoides*), all of which are commonly found only in lotic habitats. These fish were probably deposited in this wetland during a flood event and were unable to find egress. Wetland 21 had many YOY common carp and very abundant crayfish at the time it was sampled. This wetland was sampled in June when wetlands were drying, and thus may have concentrated fish and crayfish; biomass of fish and crayfish in this wetland was very high. This entire small wetland was seined in two separate hauls (fish had the opportunity to move to the portion not being seined) and at least 30 kilos of fish and crayfish were captured from each haul (fig. 5-10). Subsamples were taken from each seine haul because it would have been impossible to identify that quantity of fish.

Red shiners were the most common cyprinid overall in the study (fig. 5-9). Red shiners are the most common and widely distributed fish in non-Ozark Missouri (Pflieger, 1997), and are well adapted to turbid, silty waters. Red shiners are parasitic spawners (Pflieger, 1997), usually laying their eggs in the nests of sunfishes

(especially orangespotted and green sunfish [*Lepomis cyanellus*]). This behavior allows them to reproduce efficiently in these habitats, which have very small grain size substrates. We captured both adult and YOY red shiners in most wetlands where they were found.

Common carp YOY were captured in large numbers in only two wetlands, 21 and 22. Although these two wetlands are close together, they are separated by a levee and are very different in their hydrology. Wetland 21 was filled by topflooding, while Wetland 22 has a strong stream influence and backflooded through Wetland 26 at least four separate times during the study. The abundance of YOY common carp in these two wetlands and only these two wetlands is puzzling but this may be partially explained by the dates of the sampling. Wetlands 21 and 22 were the last wetlands to be sampled. Some other wetlands may have contained common carp YOY that were too small to be captured at the time of sampling. Common carp YOY averaged 51 mm at the time that Wetlands 21 and 22 were sampled, much larger than the minimum size that would be caught by our seine. Nevertheless, common carp YOY would likely have been too small to be captured in wetlands that were sampled early in the study.

On the small fish family graph (fig. 5-8), Wetland 22 and 26 appear similar, being dominated in numbers by cyprinids and secondly by centrarchids. These two wetlands often connect through high water. However, the species of cyprinids found in these two wetlands were very different, with the more riverine shiners (mimic, channel, sand [*Notropis stamineus*], ghost, and emerald) being found in high numbers in Wetland 26, which was closer to the river (fig. 5-9). Small cyprinids in Wetland 22 were mostly red shiners, bluntnose minnows (*Pimephales notatus*), YOY common carp, and some sand shiners. While these wetlands were very different in morphology, there were two coves of Wetland 26 which were shallow and resembled Wetland 22 in morphology. These coves were seined, and were the source of all the small fish captured in Wetland 26. Therefore, the riverine shiners were captured in habitat that resembled Wetland 22 morphologically, but which had better access to deep water and to the river.

Many YOY central stonerollers (*Campostoma anomalum)* were captured in Wetland 8. No adult stonerollers were captured. The presence of these fish, which are generally considered residents of gravelly streams, (Pflieger, 1997) is surprising. Substrate in the upper end of Wetland 8 where these fish were captured was mostly consolidated clay. The substrate in the larger, shallower, downstream end of Wetland 8 was mostly unconsolidated clay when it was flooded, but this portion was dry at the time of fish sampling. One would assume that to have many YOY in this wetland (and in none other) that spawning likely occurred there. Stonerollers are usually stream riffle spawners, but Pflieger (1997) indicates that they sometimes do spawn in quiet pools

Relatively few small gizzard shad were captured in this study, because YOY gizzard shad had just begun to enter the catch by the end of this study. Gizzard shad did spawn in these wetlands. Gizzard shad spawn in tight, milling, schools at the surface of the water (Pflieger, 1997). Small spawning aggregations of gizzard shad were observed actively spawning in Wetlands 2 and 3, and some of these fish were captured while still exuding sex products (fig. 5-7). Very young gizzard shad were caught in Wetlands 21 and 22.

Wetland 12 was separated from the river by a levee and was never flooded during this study. It was also the only wetland that had more than a few submerged aquatic macrophytes. Most fish captured at this wetland were western mosquitofish, which were not dominant in any other wetland. No large fish were captured in Wetland 12.

Large Fish

More than 500 large fish were captured during this study. Distinctly different fish assemblages were captured in different wetland types (fig. 5-11). Table 5-3 lists the number of families of large fish, number of species, and total number of fish captured in each wetland. No large fish were caught in any of the ephemerally flooded areas except E5 and Wetland 2, which were deeper than the other ephemerally flooded sites at the time of sampling. Wetland 12 also had no large fish, and Wetland 9 had only one black crappie, a year-old fish. Wetland 21 had no large fish except for YOY shortnose gar, which were growing so quickly they were already exceeding 100 mm on average by the late June sampling date. Wetlands in which more than a few large fish were caught were either deep wetlands or, in the case of Wetland 2 and E5, were ephemerally flooded areas that were connected to deep water at the time of sampling and were populated largely by fishes that were actively spawning.

Permanent Wetlands 4 and 5 were similar in that they were strongly influenced by topflooding from the river during very high water stages. They differed in that Wetland 5 was shallower and more strongly flushed by the river during flooding events, and in that Wetland 4 received more runoff from rainfall events (see Chapters 1 and 2). Both of these wetlands were dominated by white and black crappies. Crappies composed 59% of the catch by number at Wetland 5 (65 total crappies) and 51% (49 crappies) at Wetland 4. More than eight crappies per large hoop net set were captured, despite beaver damage to many of the nets (fig. 5-12). White crappies outnumbered black crappies by almost 4 to 1. Crappies captured from permanent wetlands were large, averaging 242 mm total length, black and white crappies combined. In Wetland 5, 12% of the fish caught were large gizzard shad, but large gizzard shad were not caught in Wetland 4. In Wetland 4, 12% of the fish were smallmouth buffalo (*Ictiobus bubalus*), which were not caught in Wetland 5. River carpsucker (*Capriodes carpio*), largemouth bass and small numbers of bigmouth buffalo and bluegill were caught in both wetlands.

Wetland 26 was the wetland most connected to the river. This probably accounts for the prevalence of river carpsucker and shortnose gar in this wetland. Crappies, mostly white crappies, composed 16% of the large fish catch in this wetland.

Flows entering Wetlands 4 and 5 were generally strong and unidirectional; those entering Wetland 26 were usually gentle flows, and the water entered and exited by the same path. Fish had clear access to and from the river in Wetland 26 when they were connected, but this is much less clear in the case of Wetlands 4 and 5. Fish may have been swept into these wetlands through the violent currents at the crevasse, but they probably would have encountered difficulty returning to the river, except in the periods when most of the flood plain was inundated and escape downstream was possible.

Wetland 8, a temporary wetland that was deeper in a small percentage of its area than most temporary wetlands, had some crappies (12% relative abundance), but they were much smaller (mean 142 mm total length) than those caught in the permanent wetlands. Temporary Wetlands 8 and 22 were dominated by centrarchids, but these were primarily orangespotted sunfish, bluegills and green sunfish.

In E5, the ephemerally flooded (non-basin) bottom, shortnose gar composed the majority of the catch. This reflects the observed spawning of gar during that sampling period. The lack of other flood-plain spawners in this catch is probably due to the time of year (early April) that flooding of this somewhat higher area of Lisbon Bottom occurred. Most flood-plain spawning fishes have not usually begun to spawn in this portion of the Missouri River

until late April or May (Tibbs and Galat, 1997). Other fish captured on the flooded bottom included carp, immature channel catfish (*Ictalurus punctatus*), and a single very large (1016 mm) grass carp (*Ctenopharyngodon idella*).

Wetlands 2 and 3 were connected at the time of sampling and were sampled as one unit. These wetlands are shallow and ephemeral. They were sampled during a mid-May flood event when young trees at the margins were flooded. Shortnose gar, bigmouth buffalo, large gizzard shad, centrarchids (mostly bluegill), black bullhead (*Ameiurus melas*), and cyprinids (primarily juvenile bighead carp [*Hypophthalmichthys nobilis*]) were evenly represented in the catch. Wetlands 2 and 3 were sampled at a period when the fish would likely have had access to and from the river via the flooded bottomland, but they are located far from the mainstem Missouri River or the chute (fig. 1-3). However, large numbers of buffalo and some gizzard shad were observed spawning in this area at the time of sampling (fig. 5-7B).

Figure 5-13 shows a cluster analysis of the wetlands by large fish species relative abundance. Ward's minimum variance cluster analysis grouped the shallow ephemeral wetlands together, and the deep scours also were grouped together. This is an indication that wetlands with similar morphologies had similar large fishes.

Crappie Age and Growth, and Wetland Loyalty

Growth rates of the two crappie species were not significantly different except for 3-year-old fish in Wetland 4, where white crappies were significantly longer ($p = 0.02$, fig. 5-14). However, this study was hampered by unusual illegibility of the scale annuli of crappie, especially white crappie. For example, of 48 white crappie captured in Wetland 5, only 15 of them could be reliably aged. More black crappie than white crappie could be reliably aged in Wetland 5, and more white crappie than black crappie could be reliably aged in Wetland 4. Thus, comparisons of crappies between the wetlands were hampered by low sample size of same-age, same-species groupings of successfully aged crappie.

Growth of crappies was rapid, compared to other studies of crappie growth in Missouri (fig. 5-14). Two-year-old fish (white and black crappie together) averaged 200 mm, and 3-year-olds averaging 266 mm. Few 1-year-old fish were caught, probably because our gear was not effective in capturing small fish that do not enter the shallower portions of the wetlands. The oxic epilimnion was narrow, and summer temperatures within this zone exceeded 35 °C in Wetlands 4 and 5 by mid-June. This is far above optimal temperatures for crappie growth (Hayward and Arnold, 1996). Also, water levels fluctuate dramatically in the wetlands. Therefore, one would expect that these wetlands would constitute a stressful environment for crappie. However, the data indicate that crappie captured in these wetlands are growing rapidly.

YOY crappies and several yearlings were captured from temporary wetlands, indicating that there is movement of small fish between wetlands. Also, in previous work in Wetlands 5 and 8 in 1997 (Chapman and Ehrhardt, 1999) adult crappies were captured and observed on spawning beds in Wetland 5, but only YOY juvenile crappies were found in Wetland 8, which is a temporary wetland that receives floodwater through Wetland 5 during topflooding events.

Investigations into wetland fidelity by adult crappies were inconclusive. With the exception of seven adult fish captured in Wetland 8, no crappies older than 1-year-old were found in temporary or ephemeral habitats. Wetland 8 is directly downstream from Wetland 5, which was losing water depth and appropriate crappie habitat due to sedimentation. At the beginning of the study, Wetland 5 was much deeper than Wetland 8, but due to

sedimentation of Wetland 5 and scouring in Wetland 8, Wetland 8 was deeper at the time it was sampled for fishes. These fish may have been washed from Wetland 5 during a topflooding event because of inadequate deep habitat to provide refuge from the current, or it is possible that the fish selected this deeper habitat.

Likewise, tagging of fish failed to answer the question of wetland loyalty. No tagged fish were recaptured from any wetland other than that in which it was originally captured, however there may have been inadequate flooding after the fish were tagged to provide for movement. Six tagged crappie were recaptured in each of Wetland 5 and Wetland 4. One of the Wetland 5 recaptures was recaptured on two occasions. No tagged fish were recaptured from any other wetland or ephemerally flooded area. In addition, recreational fishers captured four tagged crappies from Wetland 4 in June 2000. Therefore, crappie survived the late summer and through the winter in Wetland 4. All of these fish were originally tagged in Wetland 4, but there were no occasions of topflooding in the spring of 2000, so there were no opportunities for crappies to move between wetlands and the river during this period.

Examination of back-calculated length-at-age was also inconclusive in determining questions of wetland loyalty of adult crappies. If back-calculated lengths of crappies were similar within a wetland but different between wetlands, this would be an indication that different populations of crappies exist, and thus would be an indication of loyalty to the wetlands. We did not consider the last annulus in this analysis because fish may not have had sufficient opportunities to move since the majority of the last growing season, thus only three-year-old and older fish could provide useful data. This requires an adequate sample size of same-age fish old enough to have experienced flood events at least one year earlier. In this study, the ages and species of fish that could be accurately aged varied between wetlands, therefore, sample sizes were inadequate to test this hypothesis. Figure 5-16 shows the back-calculated length-at-age of 3-year-old white crappie from three wetlands. In this age class, back-calculated lengths at the second at Wetland 4 were significantly longer than at Wetland 26 ($p = 0.02$ for both annuli). However, the data from Wetland 26 is based on only two fish of that age and species, and therefore the relationship is questionable. No other significant differences in back-calculated lengths were found.

Conclusions and Management Recommendations

Flood-plain spawning fishes did use Lisbon Bottom for spawning during flood events. Fish evidently had mechanisms to avoid entrapment in shallow wetlands, but many fish were trapped in deeper wetlands when routes of egress, and then the wetlands, eventually dried. Entrapment of fish in wetlands that dry might not always be considered a negative occurrence; wading birds and fish-eating mammals may find these concentrations of easily catchable fish useful. However, for maximum advantage to flood-plain spawning riverine fishes, the wetlands on the southern, downstream portion of the bottom were probably more useful. They were connected to the river more often, allowing more potential for access during periods when the water temperature was appropriate for spawning to occur. Also, during periods of receding floods, there was more opportunity for the fish to return to the river instead of being trapped in wetlands in the upstream portion of the bottom.

Although Wetland 26 was often connected via Coopers Creek to the Missouri River, none of the wetlands in which fish were sampled had the same degree of connectivity to the river as the connected scours described by Galat and others (1998). Scour Wetlands 4 and 5 were similar to the isolated scours described in that study, and

were similarly dominated by centrarchids. Wetland 26, with its limited connectivity, was intermediate between the connected scour and isolated scour species assemblages described by Galat and others (1998).

Scours, wetlands more often influenced by the river, and ephemerally flooded areas were more important to fishes than wetlands that were primarily influenced by streams and runoff. In contrast, the stream-influenced wetlands had the highest numbers and diversity of macroinvertebrates and of waterbirds (Chapters 4 and 6).

Topflooding wetlands sometimes contained fish that are not well adapted to life in shallow soft-bottomed wetlands. These fish were probably trapped in these wetlands after being deposited, perhaps violently, on the bottom during flood events. During flooding periods, fish used both topflooding and backflooding wetlands for spawning. Temporary topflooding wetlands, however, trapped many fish, especially YOY buffalo and gizzard shad, because there was no opportunity for egress to the river. In backflooding Wetlands 22 and 26, the fishes had much more opportunity to come and go to the river as their instincts led them (fig. 2-4). Wetland 22 and Area E5 were the only temporarily flooded areas sampled that were influenced by backflooding. Neither of these two areas were observed to trap any fish, but all of the temporary topflooding wetlands trapped fish which died when the wetlands dried. It should be noted, however, that drying wetlands which trap fish may be useful for birds and wildlife that experience a windfall food source (Chapman and Ehrhardt, 1999).

The relative abundance of fish species varied between wetlands depending on water source and wetland morphology. For maximum diversity of fish species, a diversity of wetland types should be maintained.

Crappies in the scour wetlands grew quickly, despite temperatures that are considered to be much higher than optimal. They survived through the high temperatures of late summer and through the winter. It is unclear whether these crappies were able to find thermal refugia in these wetlands, or if these fish have different requirements from crappies found in reservoirs and small impoundments.

We were unable to determine satisfactorily the degree of crappie loyalty to scour wetlands. Loyalty to wetlands is an important management question. If adult crappies are loyal to these wetlands, harvest in intensively fished wetlands should be limited. If there is no wetland loyalty, then new crappies will enter the wetland from other less intensively fished areas, and harvest limits are less important. Crappies are the species most likely to provide recreational fishing opportunities in these wetlands, and they were most abundant in the topflooding scours. Crappie growth rate comparisons between connected and unconnected scours have not been made, but crappie growth is probably tied to primary productivity in the wetland, which is strongly controlled by flooding (see Chapter 2). Flooding may also provide a source of prey to adult crappies, as small fish are washed into these wetlands from the river. However, repeated flooding from small floods may result in the eventual sedimentation and loss of the scours.

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Table 5-1. List of fish species captured in wetlands of Lisbon Bottom between late March and the end of June 1999. Forty species were captured in all.

Catostomidae	**Cyprinidae**	**Hiodontidae**	**Poeciliidae**
white sucker	red shiner	goldeye	western mosquitofish
blue sucker	bluntnose minnow		
bigmouth buffalo	central stoneroller	**Ictaluridae**	**Sciaenidae**
smallmouth buffalo	bullhead minnow	channel catfish	freshwater drum
river carpsucker	golden shiner	flathead catfish	
quillback	emerald shiner	black bullhead	
	sand shiner		
Centrarchidae	ghost shiner	**Lepisosteidae**	
bluegill	channel shiner	shortnose gar	
green sunfish	mimic shiner	longnose gar	
orangespotted sunfish	fathead minnow		
black crappie	plains minnow	**Percidae**	
white crappie	creek chub	sauger	
largemouth bass	western silvery minnow		
	common carp	**Petromyzontidae**	
Clupeidae	grass carp	chestnut lamprey	
gizzard shad	bighead carp		

quillback (*Carpiodes cyprinus*)
bullhead minnow (*Pimephales vigilax*)
plains minnow (*Hybognathus placitus*)
creek chub (*Semotilus atromaculatus*)
western silvery minnow (*Hybognathus argyritis*)
flathead catfish (*Pylodictis olivaris*)
longnose gar (*Lepisosteus osseus*)
chestnut lamprey (*Ichthyomyzon castaneus*)

Table 5-2. Small fish summary data. Sample sites are in order of increasing permanence.

Wetland	Number of families captured	Number of species captured	Number of fish captured
E5	2	3	4
E2	0	0	0
E3	0	0	0
E4	0	0	0
E6	0	0	0
E1	2	2	170
10	1	1	166
2	5	11	92
9	3	9	42
21	7	8	226
8	6	15	253
22	4	16	676
5	1	3	5
4	0	0	0
26	3	13	607
12	4	4	40

Table 5-3. Large fish summary data. Sample sites are in order of increasing permanence. Wetlands 2 and 3 were connected at the time of sampling and fished as one unit.

Wetland	Number of families captured	Number of species captured	Number of fish captured
E5	5	5	28
E2	0	0	0
E3	0	0	0
E4	0	0	0
E6	0	0	0
E1	0	0	0
10	0	0	0
2 and 3	6	12	100
9	1	1	1
21	1	1	17
8	4	9	93
22	5	6	9
5	6	9	97
4	3	12	99
26	8	14	76
12	0	0	0

Figure 5-1. Minifyke net deployment.

Figure 5-2. Deployment of trammel nets from a canoe amid flooded willows.

Figure 5-3. Setting hoop nets during flood pulse.

Figure 5-4. Fish sampling locations at Lisbon Bottom. Locations E1 through E6, in green, are very ephemerally flooded areas and are not shown on the map of numbered wetlands (fig 1-3). *Background photo courtesy of U.S. Army Corps of Engineers, Kansas City, MO, March 2000.*

Figure 5-5. Use of an ATV and trailer to transport fishing gear across Lisbon Bottom. Lisbon Bottom is often impassible even with an ATV, and transportation of nets on foot was often required.

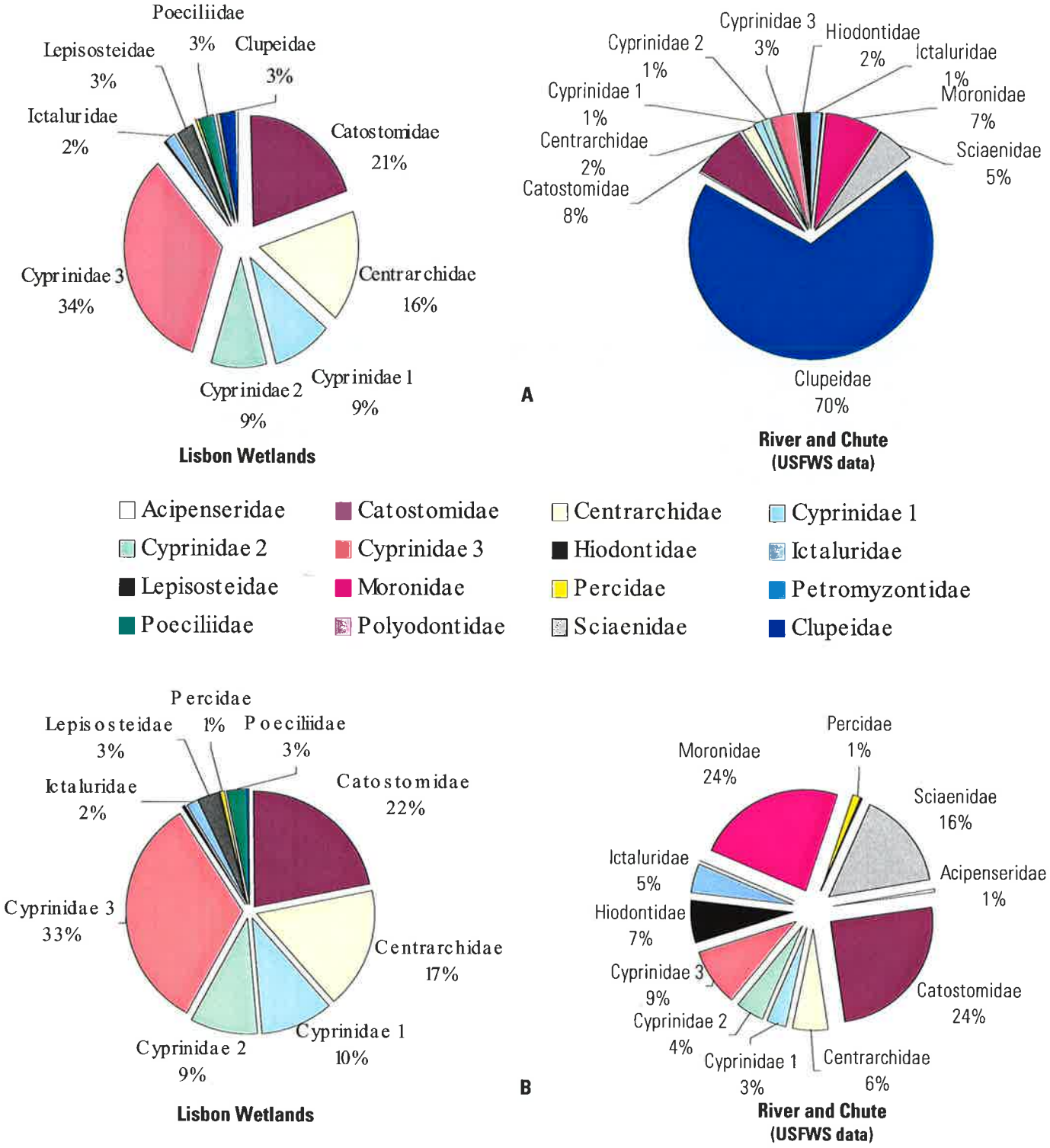
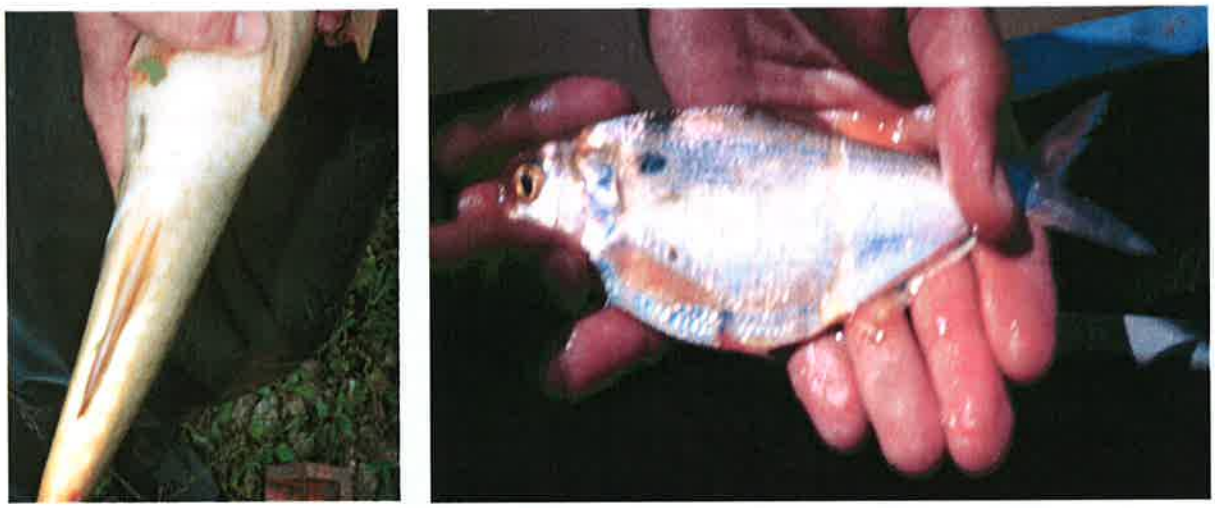


Figure 5-6. Relative abundance of fish families in Lisbon wetlands compared to the adjacent Missouri River and Lisbon Chute (A). Figure (B) with the family Clupeidae (gizzard shad) removed. Fish were sampled during the same period of 1999. Missouri River and Chute data from Louise Mauldin (US Fish and Wildlife Service, unpublished data). Cyprinidae 1 = native cyprinids exclusive of the genera *Cyprinella* and *Notropis*. Cyprinidae 2 = non-native cyprinids (common carp, bighead carp, and silver carp). Cyprinidae 3 = Cyprinids of the genera *Cyprinella* and *Notropis* (shiners).



A B

Figure 5-7. Fish captured in Lisbon Bottom wetlands that were exuding sex products when captured. **(A)** Male common carp. **(B)** Female gizzard shad.

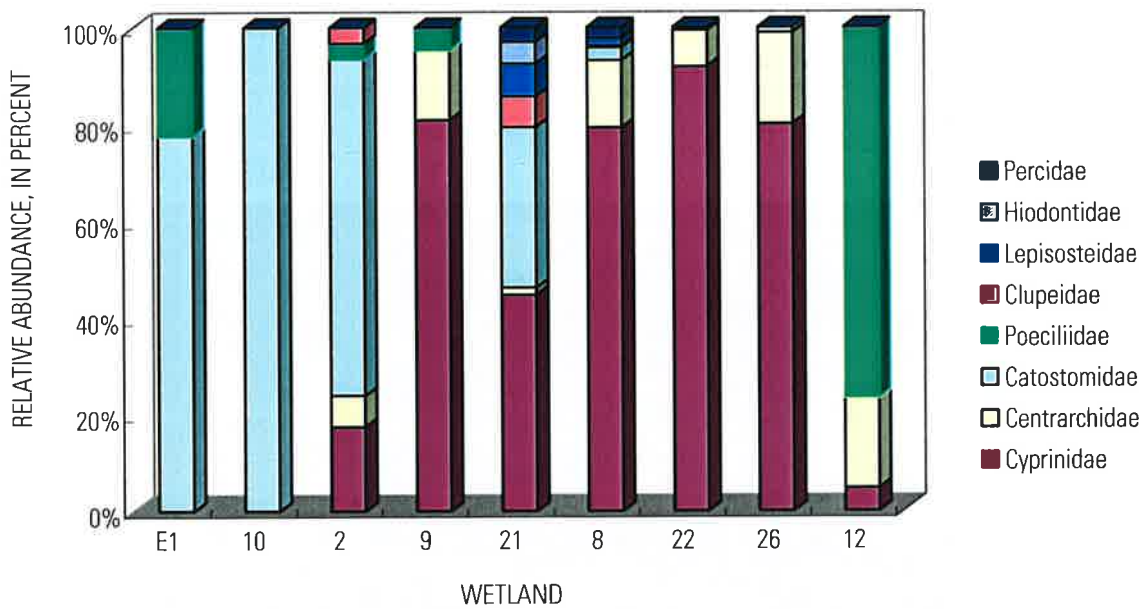


Figure 5-8. Relative abundance of small fish families in Lisbon Bottom wetlands. Wetlands E1 through 26 are ordered by increasing permanence. Catastomids were gradually replaced by cyprinids with increasing permanence. The catastomids in these wetlands were almost all young of the year bigmouth buffalo, whereas the cyprinids were very diverse. Wetland 12 is very different in hydrology from the other wetlands, being strongly stream and runoff influenced, and never inundated by the river. Wetland 12 also was the only wetland with submerged aquatic macrophytes. Wetlands with very few (< 10) small fish captured are not included in the figure. Number of captured fish in shown wetlands ranged between 42 and 678.

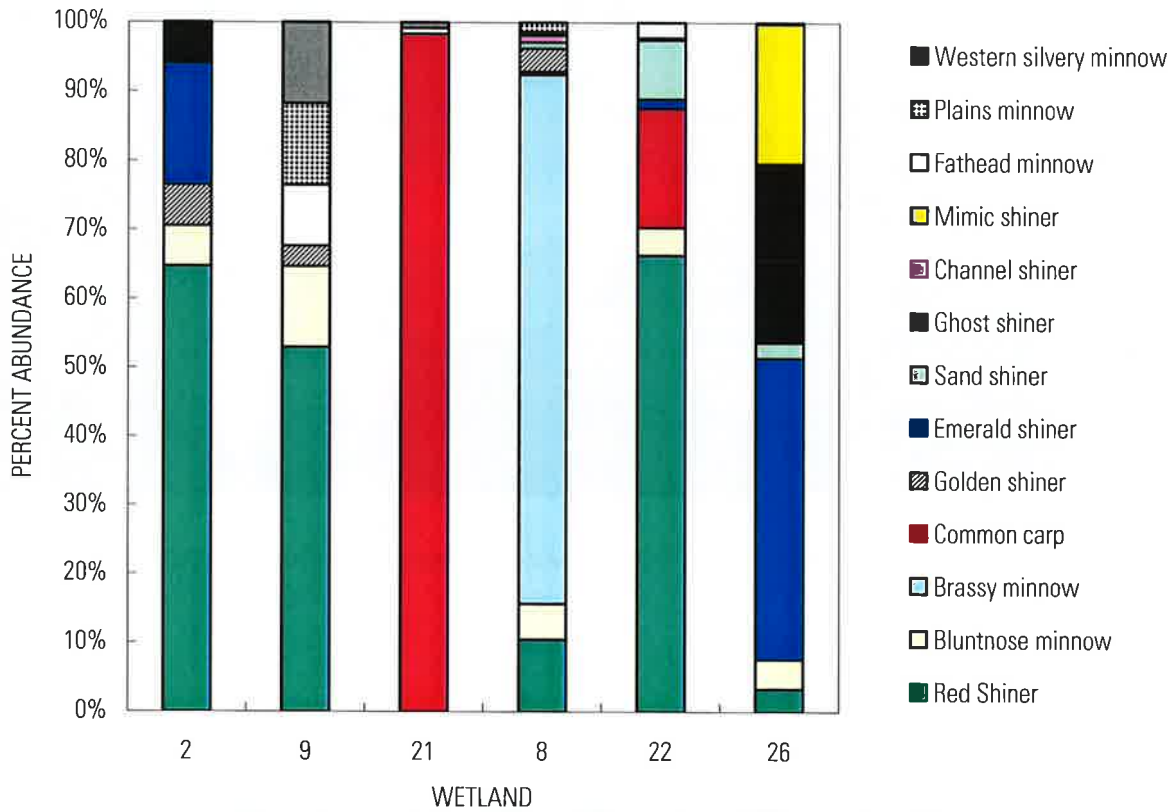


Figure 5-9. Relative abundance of small and juvenile cyprinids in Lisbon Bottom wetlands. Wetlands are ordered by increasing permanence. Only wetlands where significant numbers of small cyprinids were captured are shown.

Figure 5-10. A bag seine haul in Wetland 21, Lisbon Bottom.

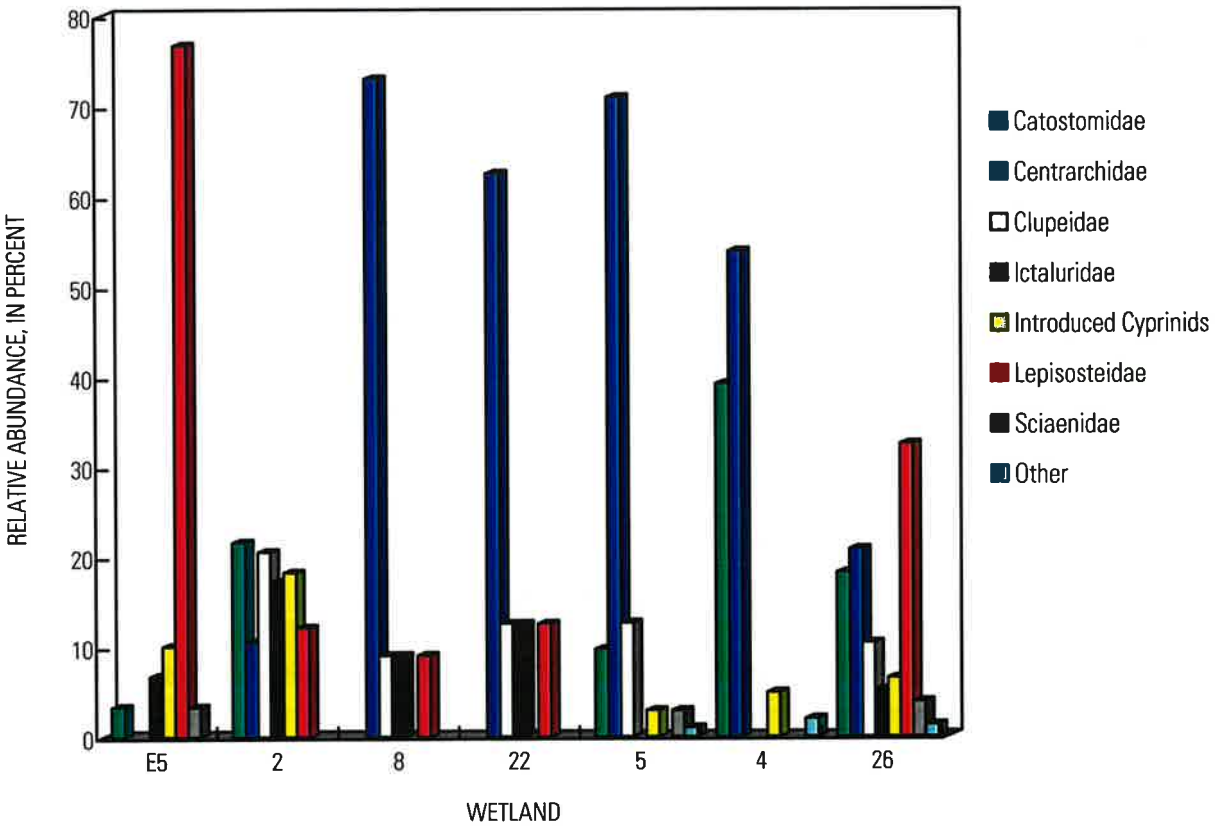


Figure 5-11. Relative abundance of large fish captured in Lisbon Bottom wetlands by family. Wetlands are ordered by increasing permanence. Only wetlands in which significant numbers of large fish were captured are shown.

Figure 5-12. Mending beaver-damaged hoop nets. Beaver damage to hoop nets was severe, often affecting more than half of the nets in place on overnight sets.

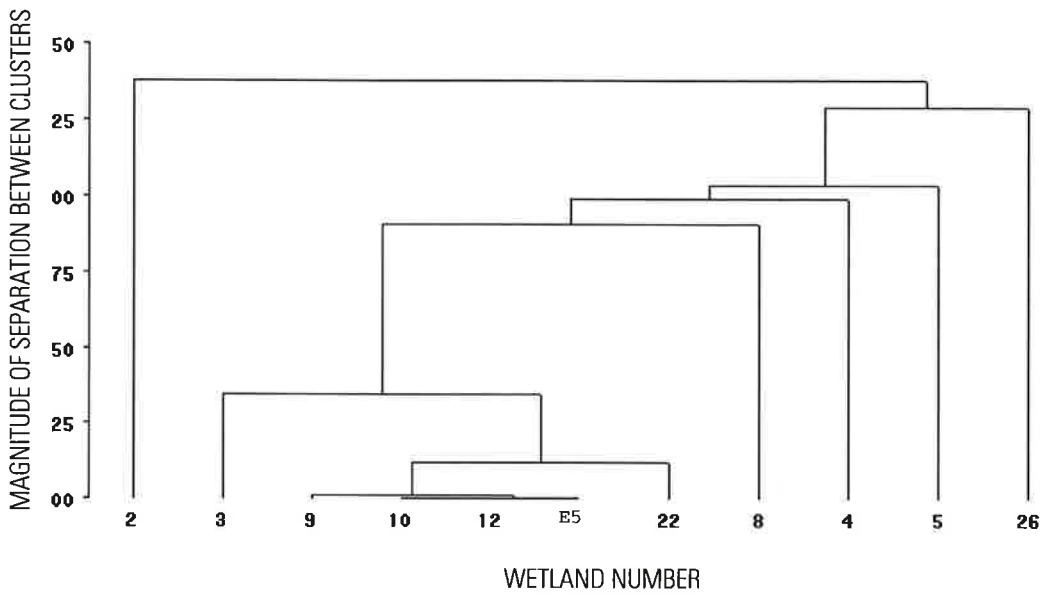


Figure 5-13. Ward's minimum variance cluster analysis of Lisbon Bottom wetlands by large fish species abundance.

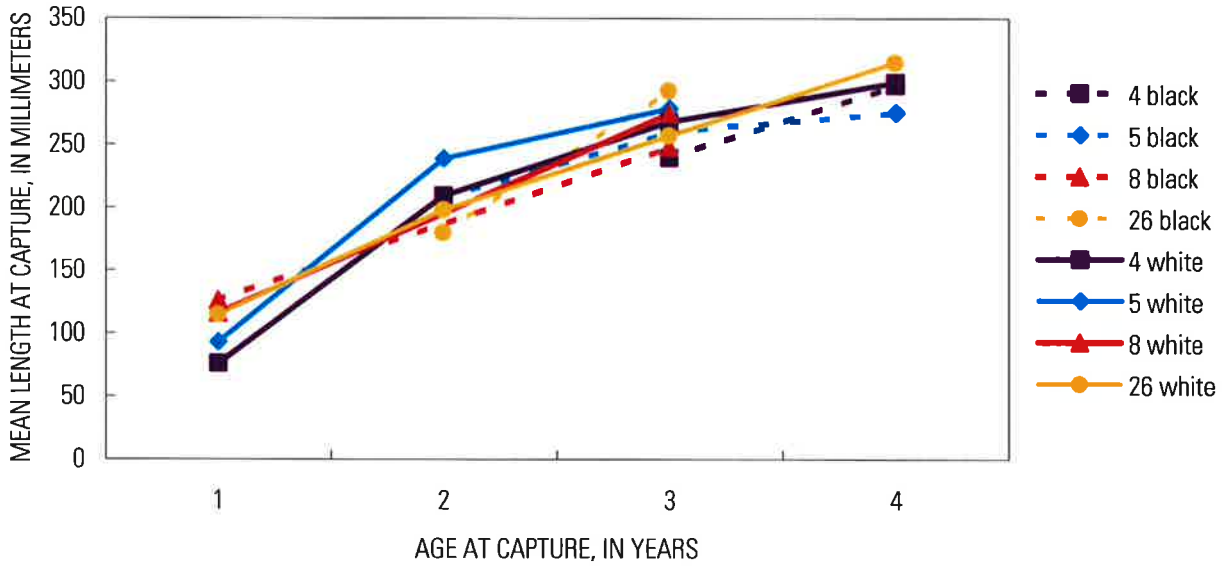


Figure 5-14. Comparison of mean lengths and ages at capture for black and white crappie captured in four wetlands of Lisbon Bottom, Missouri. Black crappies are indicated by broken lines, and white crappies by unbroken lines.

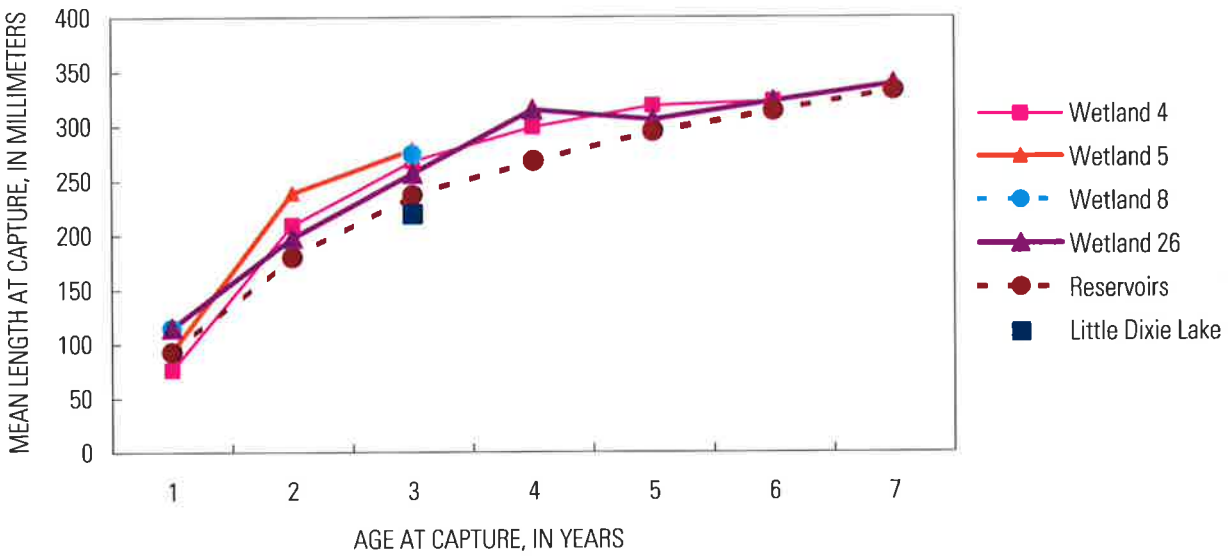


Figure 5-15. White crappie lengths and ages at capture in four wetlands at Lisbon Bottom, Missouri, compared to an eight-year study of white crappies from four Missouri reservoirs (Colvin, 1991) and to a three-year study of Little Dixie Lake, Missouri (Craig Gemming, Missouri Department of Conservation, unpublished data).

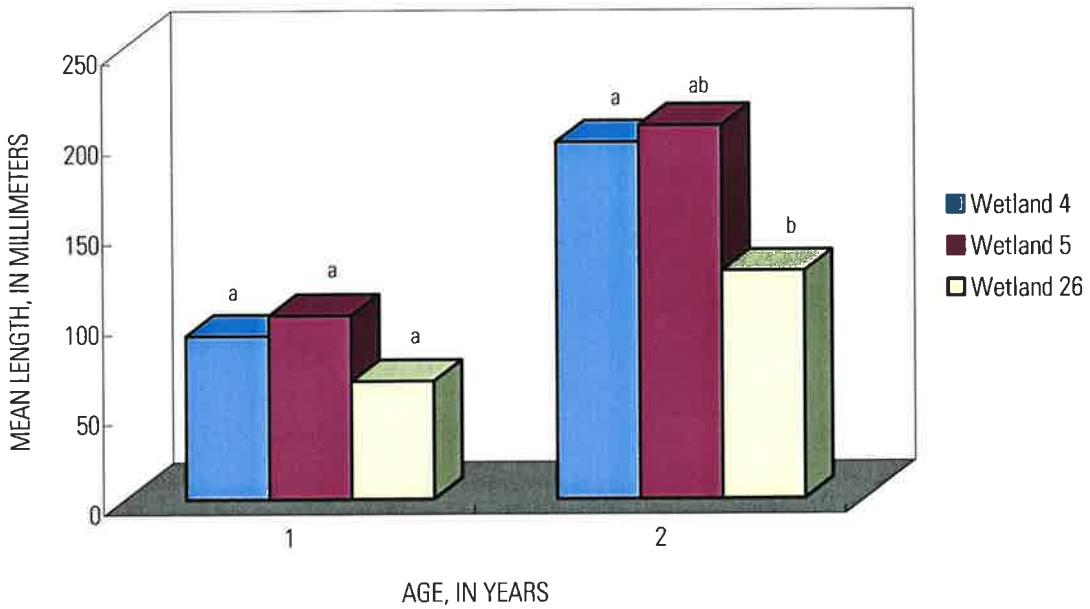


Figure 5-16. Back-calculated length at ages 1 and 2, from scales, for 3+ -year-old white crappies from Wetlands 4, 5 and 26, Lisbon Bottom, Missouri. Different letters indicate a significant difference in back-calculated length between wetlands within an age class. Different length at back-calculated ages is an indicator of distinct populations and wetland loyalty. Length at age 3 is not included in the analysis because fish may not have had an opportunity to move between wetlands since the last year's growth. These data are based on a small number of fish: 8 from Wetland 4, and 2 each from Wetlands 5 and 26.

Chapter 6. Waterbird Chronology and Habitat Use of Lisbon Bottom During Spring Migration 1999

James F. Fairchild and Linda C. Sappington

Abstract

The Lisbon Bottom Tract of the Big Muddy National Fish and Wildlife Refuge consists of an 875 ha land parcel located within a loop bend on the Lower Missouri River near Glasgow, MO. Lisbon Bottom was purchased in 1995 by the U.S. Fish and Wildlife Service to partially offset historic, cumulative losses of fish and wildlife habitat along the Lower Missouri River. Lisbon Bottom is passively managed (for example, minimal vegetative control or hydrologic management) to allow the area to evolve in a natural ecological trajectory. Lisbon Bottom is seasonally flooded by waters from the Missouri River, the Lisbon Chute, rainfall, and upland tributary inflows. Thus, Lisbon Bottom is an excellent opportunity to evaluate the biological responses of a large river flood plain in relation to localized, natural hydrologic conditions.

Missouri River flood-plain habitats are important migratory corridors for numerous waterbirds including ducks, geese, herons, and shorebirds. We surveyed the chronology of waterbird use of Lisbon Bottom over a 10-week period from mid-March to late May of 1999 to assess the distribution of waterbirds among habitats in relation to hydrologic condition. Waterbird surveys were conducted twice weekly at daybreak. The Lisbon Chute was observed from four points for 10 minutes each and the river was observed from two points for 10 minutes each. Additional 10-min surveys were conducted on 25 flood-plain wetlands distributed across 4 geomorphic basins: Upper, Valley Wall, Middle, and Lower. Thirty-one species of waterbirds, totaling 1517 individuals, were observed over the 10-wk period. The waterbird community was composed of ducks (1025), geese (203), herons (85), shorebirds (79), coots (63), cormorants (34), gulls (11), mergansers (8), terns (3), kingfishers (3) sora rails (2), and a grebe (1). Blue-winged teal were the most abundant duck followed by lesser scaup, wood ducks, and mallards. Scaup and mallards were dominant during the pre-flood period of March 15-April 15, whereas blue-winged teal use overlapped the flood period of April 16–May 18. Chronology of use of Lisbon Bottom was primarily related to spring migration of each species. A wide diversity of waterbirds was observed in the chute (20 species) and main river habitats (13 species). Highest species richness (23 species) occurred on flood-plain wetlands; in addition, flood-plain wetlands were occupied for a longer period of time than the river or chute habitats. There was a significant difference among basin areas in terms of total waterbirds and total number of ducks; however, there was no relationship between species richness and basin location. Wood ducks, mallards, and blue-winged teal had a preference for wetlands within the Valley Wall Basin compared to the Upper, Middle, or Lower Basins. The high bird use of the Valley Wall Basin (primarily Wetlands 11 and 12) may be due to the shallow, persistent moisture conditions associated with inflows from intermittent streams. These data indicate that wetlands along the valley wall at Lisbon Bottom have unique characteristics that make them attractive for waterbirds. Therefore, land managers may need to prioritize these types of geomorphic features in future land management and acquisition activities. However, subtle differences in landform, vegetation (type, density, diversity, and height), and hydrology (timing, frequency, and duration of

flooding) are highly inter-related and complex and therefore require further study to elucidate causal mechanisms.

Introduction

Missouri lies within a spring migration corridor of current and historical importance to migrating waterbirds that move through the interior of North America to reach breeding and nesting grounds in the northern U.S. and Canada (Bellrose, 1974). During springtime numerous species of waterbirds migrate through Missouri and use the Missouri River Bottom and associated wetlands extensively for feeding, resting, and courting. Spring migration is an especially important period in the life history of waterbirds enroute to northern breeding grounds since during this period they require diets high in protein and calcium in order to increase body condition prior to egg laying (Heitmeyer, 1985).

The Lisbon Bottom is a 875 ha tract that lies along an inside channel bend of the Missouri River at river miles 213-218 between Glasgow and Boonville, MO. Lisbon Bottom was purchased in 1995 as part of the newly created Big Muddy National Fish and Wildlife Refuge. A large part of Lisbon Bottom was farmed for corn, soybeans, and wheat prior to major flood events that occurred in 1993 and 1995. These floods, however, greatly altered Lisbon Bottom and resulted in formation of a new cross channel. In addition, numerous basins were scoured in the flood plain and large amounts of silt and sand were deposited. This extensive alteration rendered the property unsuitable for traditional agriculture. Thus, this tract was purchased as part of the Refuge to partly offset historic, cumulative losses of large amounts of riparian habitat that formerly occurred along the Lower Missouri River.

Currently, the Lisbon Bottom Tract is passively managed for wildlife and recreational use. Seasonal flooding of Lisbon Bottom occurs due to rainfall, upland tributary inflow, and flooding from the Missouri River. The lower end of Lisbon Bottom is nearly 5 meters lower than the upstream margin. Lowlands and swales generally back-flood before water tops the natural levees upstream (Jacobson and others, 1999). The ordering of soil textures and hydrology influence the morphology of the wetlands and the entire biological community of the flood plain. Flooding of wetland habitats is highly dynamic and is dependent on localized conditions of soil type, soil moisture, precipitation, and sources of water.

The majority of studies of bird use of the Lower Missouri River have occurred on actively managed sites such as state conservation areas and National Wildlife Refuges. Relatively few studies have been conducted on passively managed sites such as the Lisbon Bottom Tract. Recent studies have documented the species composition of Lisbon Bottom (Humburg and Burke, 1999) and other aquatic habitats of the Lower Missouri River flood plain from a landscape-scale perspective (Humburg and others, 1999). However, there have been no detailed studies of bird use at Lisbon Bottom in relation to habitat suitability and habitat type. In this study we observed the chronology of waterbird use of various habitats at Lisbon Bottom in relation to hydrologic and resource conditions. These studies were conducted to provide managers with insight into the dynamics of spring habitat conditions and how wetland dependent birds respond to these conditions.

Methods

Waterbird surveys were conducted twice weekly from mid-March through May 1999. Basin classification was made based on spatial location and hydrologic characteristics defined by existing data (for example, Chapter 1; Jacobson and others, 1999). Basin and wetland locations are presented in table 6-1 and figure 6-1. The chute was observed from four points for 10 minutes each and the river was observed from two points for 10 minutes each. Twenty-five additional flood-plain wetlands were observed for 10 minutes along walking survey routes (fig. 6-2). All waterbird surveys were conducted within the first two hours of daylight on each observation day. Wetlands were approached quietly and observed for waterbirds for 10 minutes from thick cover of willow. Care was taken not to disturb birds that might result in movement to other sites and double counting. Water levels in individual wetlands were determined using metered stakes. The chute, river, and some of the scour holes were not observed for waterbirds during the peak of the flood due to difficulty in access and concern for human safety; however, this is not expected to alter the interpretation of the data.

Statistical Analysis

Statistical analysis of biological survey data is often difficult due to the violation of normality of distributions and lack of independence (Green, 1979). However, use of exploratory statistics is valuable in discerning trends that may lead to testable hypotheses. Exploratory data analysis was conducted using the Statistical Analysis System (SAS, 1990). One-way analysis of variance of ranked data was conducted to determine differences across flood-plain basins (that is, Upper, Valley Wall, Middle, and Lower) for the following community endpoints: total # waterbirds, total # ducks, # geese, # herons, # shorebirds, # mallards, # wood ducks, and # blue-winged teal. Other individual species trends were not evaluated because of the rarity of occurrence. Additional attempts were made to test for associations of birds with various wetland types. However, these analyses proved problematic due to lack of interspersion of potential wetland types and current lack of a wetland classification system for such a dynamic system. Statistical significance levels were maintained at $p \leq 0.05$.

Results and Discussion

Hydrologic dynamics

Hydrologic regimes were defined as three separate intervals: pre-flood (March 15–April 13); flood (April 16–May 18); and post-flood (May 21–May 28). River levels prior to the spring flood averaged approximately 15 ft (184 m above mean sea level) at the Boonville gaging station (fig. 6-3). River levels rose to approximately 30 ft at Boonville on April 18th at the initiation of the spring flood. Water levels of selected wetlands are provided in figures 6-3 and 6-4 to demonstrate the effect of river stage and localized rainfall on wetland hydrologic dynamics. Prior to the flood there was water present in Wetlands 2, 3, 5, 22, and 26 that are largely deeper scour wetlands. Water levels increased immediately prior to the flood in Wetlands 10, 11, and 12 due to the combined influence of increasing river stage (via groundwater) and rainfall. Rainfall had a lesser

influence on water levels in wetlands and primarily augmented levels in valley-wall wetlands (fig. 6-4). Following the recession of the flood (May 21 and thereafter) water levels decreased within individual wetlands depending on topography, soil characteristics and evaporation. However, the relative significance of these factors for individual basins and wetlands are still not totally understood.

Waterbird community composition

Thirty-one species of waterbirds, totaling 1518 individuals, were observed over the 10-wk period (tables 6-2, 6-3 and 6-4). The waterbird community was composed of ducks (1025), geese (203), herons (85), shorebirds (79), coots (63), cormorants (34), gulls (11), mergansers (8), terns (3), kingfishers (3), sora rails (2), and a grebe (1) (fig. 6-5). Helmers and others (1999) evaluated waterbird species composition across a 4-year period (1994-1997) at Lisbon Bottom and documented 35 species of waterbirds. The species composition of our study was similar to Humburg and Burke (1999) with notable exceptions. For example, in our study we observed both a least bittern and American bittern whereas these species were not noted by Helmers and others (1999). In contrast, Helmers and others (1999) observed more shorebird species (American avocet, American golden plover, and black-bellied plover) and gull/tern species (Franklin's gull, herring gull; Caspian tern, common tern, and least tern) than in our study. However, there were distinct differences in the sampling frequency of our studies. Our study was a 10-week, intensive study of a diversity of wetland types, whereas Helmers and others (1999) conducted a long-term study of selected habitats. Therefore, direct comparisons should be made with caution. The two studies fully illustrate, however, that Lisbon Bottom is host to a large diversity of waterbirds over the course of a year.

Chronology and spatial distribution of waterbird observations

Chronology of waterbird use of Lisbon Bottom was primarily related to spring migration of each species as opposed to the onset and duration of flooding (fig. 6-6; table 6-3). For example, highest numbers of total ducks and geese were observed prior to the flood period which began on April 16, whereas herons and shorebirds exhibited differential migration patterns that varied distinctly by species (fig. 6-6, table 6-3). Major migrations of mallard and scaup occurred during the period of March 16 to April 3 (table 6-3; fig. 6-7). Similar patterns of waterbird migration in the Lower Missouri River Basin were observed by Humburg and others (1999) in their post-flood evaluation studies.

Blue-winged teal (396) (fig. 6-8) were the most commonly observed duck followed by lesser scaup (185), wood ducks (160), and mallards (126). Blue-winged teal were most commonly observed during the flood period of April 16-May 18 (fig. 6-7, table 6-3); however, this period corresponds with historic observations of migration chronology and therefore may not merely be associated with the onset of flooding (Taylor, 1977). Wood ducks were present during the entire study at lower numbers due to their use of the Lower Missouri River for courting, nesting and recruitment (Drobney and Fredrickson, 1977).

Large numbers of waterbirds were observed using the chute and river habitat over the course of the study, with peak numbers being observed in early April due to large numbers of scaup (fig. 6-7 and fig. 6-9). Waterbird use of the flood plain increased with increasing rainfall and flooding as flooded bottomland habitat increased (fig. 6-9).

Similarly, a wide diversity of waterbirds was observed in the chute (20 species) and main river habitats (13 species) (table 6-4); however, much of this use was a single day by species that used these habitats as short stopovers during migration (for example, scaup, ring-necked ducks, snow geese, terns, and gulls) (table 6-3). Highest species richness (23 species) occurred in flood-plain basin wetlands composed of a diversity of wetland types ranging from shallow, seasonal wetlands to deep scours; furthermore, birds that tended to use internal wetlands remained in these areas over a longer period of time compared to birds that used chute and river habitats (table 6-3).

Relative use of flood-plain wetlands (but not including river and chute habitats) were evaluated using analysis of variance of ranked data by basin. Results indicated that total number of waterbirds ($p = 0.037$) and total number of ducks ($p = 0.011$) were significantly higher for the Valley Wall Basin compared to other basins. Wood ducks exhibited greater use of the Upper and Valley Wall basins compared to the Middle or Lower basins. Blue-winged teal had a high use of both Valley Wall and the Lower basins but differences were not statistically different ($p = 0.056$). No significant relationships were observed for species richness, other species, or waterbird groups. Interpretation of such analyses, however, must be approached with caution. Although rank analysis is widely accepted for biological data, the classification of basins may in part reflect the type of wetlands within basins. For example, the majority of wetlands along the valley wall are shallow, seasonal wetlands with increased levels of coarse organic matter (fig. 6-10 and fig. 6-11). In contrast, wetlands in Basin 3, which exhibited less bird use, are typified by deeper scours with little emergent or submergent vegetation. Basins were selected based on location, which is influenced by hydrologic (for example, water source, timing), physical (soils type and morphology), and resulting biological (for example, vegetative biomass and species composition) factors. Classification of wetland types in such a dynamic geomorphic and hydrologic area is an emerging area of wetland ecology that deserves further study.

Blue-winged teal (396 total individuals) were the most frequently observed waterbird observed during the study (table 6-2, table 6-4, and fig. 6-7). Blue-winged teal extensively used Wetland 11, which accounted for 59% of all observations of the species. Wetland 11 consists of a long, shallow (mean 10 cm depth; maximum 1.0 m depth) seasonal wetland that is hydrologically fed from valley-wall tributaries and is extensively vegetated (see Chapter 1). High numbers of blue-winged teal were also observed in Wetland S-14, which consists of a large scour in the Lower Basin with intermittent connectance to the river.

Taylor (1977) conducted the most intensive study of spring ecology of blue-winged teal in Missouri wetlands. Blue-winged teal were observed to use average depths ranging from 13-19 cm in depth (Taylor, 1977). Crop samples of blue-winged teal collected during the period March 17- May 5, 1976 at the Mingo National Wildlife Refuge (Stoddard and Wayne counties of Southeast Missouri) contained approximately 65% animal matter and 35% plant matter. Primary animal matter in the diet consisted of snails (24%) (fig. 6-11), insects (22%; primarily chironomid larvae), and crustaceans (13%; primarily isopods); snails and chironomids were selectively fed upon at greater rates than occurred spatially in the environment. Primary plant materials consisted of seeds (elm *Ulmus* spp. and *Eleocharis* spp.), grass fragments, and algae. Future studies of waterfowl/wetland interactions at Lisbon Bottom should include Wetland 11 for intensive assessment of invertebrate, vegetation, and waterfowl relationships.

Humburg and others (1999) conducted an extensive, multi-year (1994-1998) study of waterbird use of the Lower Missouri River following the Great Flood of 1993 at approximately 140 sites distributed among 4 habitat types (scour connected, scour non-connected, remnant, and temporary) between Hartsburg, MO and Sioux City, IA. Over 70 species of waterbirds were observed. Remnant sites were used more frequently than either of the other three habitat types, however, no single habitat type accounted for all species observed. They concluded that a diversity of habitat types is necessary for the conservation of migratory waterbirds in the Lower Missouri River (Humburg and others, 1999).

Lisbon Bottom and the other tracts of the Big Muddy National Fish and Wildlife Refuge collectively represent a major contributor to the overall habitat diversity of the Lower Missouri River. Prior to the Great Flood of 1993, the majority of waterbird management emphasis was on actively managed areas associated with State Conservation Areas and National Wildlife Refuges. Actively managed wetlands are usually manipulated to produce hydrologic conditions that optimize vegetative composition, food production and availability, and other desired habitat characteristics such as mudflats (that is, for shorebirds). However, to date there have been no published studies to directly compare bird use at passively managed areas of the Big Muddy NFWR with other, more actively managed conservation areas such as Eagle Bluffs or Grand Pass Conservation Areas. Bird use at passively managed and actively managed areas may likely be different than that at Lisbon Bottom or other passively managed areas due to differences in landform, vegetation (type, density, diversity, and height), and hydrology (timing, frequency, and duration of flooding). However, these factors are highly inter-related and complex and therefore require further study to elucidate causal mechanisms. Other bird species, such as warblers and neotropical migrants may actually prefer passively managed habitats such as Lisbon Bottom; however, these groups were not the focus of this study.

This study documented the frequent waterbird use of the chute and the interior flood-plain wetlands located along the valley-wall tributaries located at Lisbon Bottom. The chute habitat was frequently used by unique groups such as terns, gulls, and other species during migration. The chute habitat, with numerous sandbars, shallow habitats, and potential fish and invertebrate food resources, may represent valuable resting areas for waterbirds during migratory flights. Valley-wall wetlands, fed by intermittent streams, were also preferred by many species; many of these species, including blue-winged teal, wood ducks, and mallards, tended to remain for longer periods during migration. These wetlands contain characteristics (that is, shallow water and vegetation) that produce optimum invertebrate food resources and feeding conditions for these dabbling ducks. Collectively, we feel that this information provides some insight into the value of passively managed areas for waterbirds during spring migration. It is hoped that this information, along with future studies of other avian fauna, can be used by refuge staff in the development of an adaptive management framework for Lisbon Bottom and other parcels within the Big Muddy National Fish and Wildlife Refuge.

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Table 6-1. List of basins, wetlands, wetland category, and hydrologic sources used for intensive studies of bird use in relation to physical, chemical, and biological variables.

Basin	Wetland	Morphology	Dominant hydrologic sources
Upper	1	Deep	River topflood
Upper	2	Shallow	Mixed stream/river topflood
Upper	3	Shallow	Mixed stream/river topflood
Upper	4	Deep	River topflood
Upper	5	Deep	River topflood
Upper	7	Shallow	River topflood
Valley Wall	8	Shallow	Mixed stream/river
Valley Wall	9	Shallow	Mixed stream/river
Valley Wall	10	Shallow	Mixed stream/river
Valley Wall	11	Shallow	Stream
Valley Wall	12	Deep	Stream
Valley Wall	22	Shallow	River backflood
Middle	13	Deep	Chute/river
Middle	15	Deep	Chute/river
Middle	16	Deep	Chute/river
Middle	19	Deep	Chute/river
Middle	20	Deep	Chute/river
Middle	21	Deep	River backflood
Lower	23	Deep	Mixed stream/river
Lower	24	Deep	Mixed stream/river
Lower	26	Deep	River backflood
Lower	28	Shallow	River backflood
Lower	29	Deep	River backflood
Lower	30	Deep	River backflood
Lower	S-14	Shallow	River backflood

Table 6-2. List of waterbird species observed at Lisbon Bottom during Initial Biotic Survey (1994-1997; Humburg and Burke, 1999) and during present study (1999). The "+" indicates that the species was observed in a given year. The "-" indicates that the species was not observed in a given year.

Family	Common Name	Species	1994	1995	1996	1997	1999
Anatidae	American wigeon	*Anas americana*	-	+	-	-	-
	Blue-winged teal	*Anas discors*	+	+	+	+	+
	Canada goose	*Branta canadensis*	+	+	+	+	+
	Common merganser	*Mergus merganser*	-	+	-	-	+
	Gadwall	*Anas strepera*	-	+	+	+	+
	Green-winged teal	*Anas crecca*	+	+	+	+	+
	Hooded merganser	*Lophodytes cucullatus*	-	-	-	-	+
	Lesser scaup	*Aythya affinis*	-	+	-	+	+
	Mallard	*Anas platyrhynchos*	+	+	+	+	+
	Northern pintail	*Anas acuta*	+	+	-	-	+
	Northern shoveler	*Anas clypeata*	+	+	+	+	+
	Ring-necked duck	*Aythya collaris*	-	-	-	-	+
	Snow goose	*Chen caerulescens*	-	-	-	-	+
	Wood duck	*Aix sponsa*	+	-	+	+	+
Ardeidae	American bittern	*Botaurus lentiginosus*	-	-	-	-	+
	Black-crowned night heron	*Nycticorax nycticorax*	-	-	-	+	+
	Great blue heron	*Ardea herodias*	+	+	+	+	+
	Great egret	*Ardea alba*	+	+	+	-	+
	Green heron	*Butorides virescens*	+	-	-	-	+
	Least bittern	*Ixobrychus exilis*	-	-	-	-	+
Cerylidae	Belted kingfisher	*Ceryle alcyon*	-	+	+	+	+
Charadriidae	American avocet	*Recurvirostra americana*	+	-	-	-	-
	American golden-plover	*Pluvialis dominica*	+	-	-	-	-
	Black-bellied plover	*Pluvialis squatorola*	-	+	+	-	-
	Killdeer	*Charadrius vociferus*	+	+	+	+	+
	Semipalmated plover	*Charadrius semipalmatus*	+	+	+	+	-
Laridae	Black tern	*Chlidonias niger*	-	+	+	-	+
	Bonaparte's gull	*Larus philadelphia*	-	-	+	-	+
	Caspian tern	*Sterna caspia*	-	-	-	+	-
	Common tern	*Sterna hirundo*	-	+	-	+	-
	Franklin's gull	*Larus pipixcan*	-	-	+	-	-
	Herring gull	*Larus argentatus*	+	-	-	-	-
	Least tern	*Sterna antillarum*	-	-	+	+	-
	Ring-billed gull	*Larus delawarensis*	-	+	+	+	+
Pelecanidae	American white pelican	*Pelecanus erythrorhynchos*	+	+	-	+	-
Phalacrocoracidae	Double-crested cormorant	*Phalacrocorax auritus*	+	+	+	+	+
Podicipedidae	Pied-billed grebe	*Podilymbus podiceps*	-	+	+	-	+
Rallidae	American coot	*Fulica americana*	-	+	+	+	+
	Sora	*Porzana carolina*	-	-	-	+	+
	Virginia rail	*Rallus limicola*	-	-	-	+	-

Table 6-3. Number of individuals of each species counted on each survey date (March 16 to May 28, 1999) at Lisbon Bottom.

		Pre-Flood								Flood										Post-Flood				
Species	**Group**	Mar 16	Mar 19	Mar 26	Mar 30	Apr 2	Apr 6	Apr 9	Apr 13	Apr 16	Apr 20	Apr 23	Apr 27	Apr 30	May 5	May 7	May 11	May 14	May 18	May 21	May 25	May 28	**Sum**	**Days Sighted**
American bittern	Heron													1									1	1
American coot	Rail	1				38		1	9		3	9	2										63	7
Belted kingfisher	Kingfisher			1		1											1						3	3
Black tern	Tern																	3					3	1
Black-crowned night heron	Heron																			1			1	1
Blue-winged teal	Duck				2	44		13	12	23	67	92	95	20	9	7	1	5	2		4		396	15
Bonaparte's gull	Gull	1																					1	1
Canada goose	Geese	2			14	2	14	6	5	2	2		13						1			2	63	11
Common merganser	Merganser	1	5			1																	7	3
Common snipe	Shorebird			6	6			1	1														14	4
Doubled-crested cormorant	Cormorant						2	30	1				1										34	4
Gadwall	Duck	1				10			10				4										25	4
Great blue heron	Heron	3		2	1	4	3	4	7	2	3	4	2	4	1	1	5	1	10	10	7	5	79	20
Great egret	Heron					1		2															3	2
Green heron	Heron																				1		1	1
Green-winged teal	Duck		1	5	8	71	2																87	5
Hooded merganser	Merganser		1																				1	1
Killdeer	Shorebird	3	2	13	2	5	4		4				1							4	2	1	41	11
Least bittern	Heron																	1					1	1
Lesser scaup	Duck			8	7	168		2															185	5
Lesser yellowlegs	Shorebird		1						1			2	2			12							18	4
Mallard	Duck	34	29	16	13	8	4	2	4		2		6				2	3	1		2		126	13
Northern pintail	Duck					1																	1	1
Northern shoveler	Duck					41		1	2														44	3
Pied-billed grebe	Grebe												1										1	1
Ring-billed gull	Gull	1		4				5															10	3
Ring-necked duck	Duck													1									1	1
Snow goose	Geese			140																			140	1
Solitary sandpiper	Shorebird												2		3		1						6	3
Sora	Rail													1				1					2	2
Wood duck	Duck	7	7	9	24	14	10	7	8	10	8	7	9	1	6	8	1	1		1	9	13	160	20
Birds/d		54	46	204	77	409	39	74	64	37	85	114	138	28	19	28	11	15	14	16	25	21	**1518**	
Species/d		9	7	10	9	15	7	12	12	4	6	5	12	6	4	4	6	7	4	4	6	4		

Table 6-4. Numbers of waterbirds by species and wetland observed at Lisbon Bottom, Spring 1999.

		Basin																											
		Upper						Valley-wall						Middle					Lower								Chute	River	Total
		Wetland																											
Common Name	**Group**	**1**	**2**	**3**	**4**	**5**	**7**	**8**	**9**	**10**	**11**	**12**	**22**	**13**	**15**	**16**	**19**	**20**	**21**	**23**	**24**	**26**	**28**	**29**	**30**	**S-14**			
American bittern	Heron								1																				**1**
American coot	Rail		1			1															1	5				4	41	10	**63**
Belted kingfisher	Kingfisher				1							1															1		**3**
Black tern	Tern																										3		**3**
Black-crowned night heron	Heron												1																**1**
Blue-winged teal	Duck	2	4	4	1			6	16		234	7	20	4		2			4		8	14		5		38	26	1	**396**
Bonaparte's gull	Gull																										1		**1**
Canada goose	Geese			6							2		13										18			2	19	3	**63**
Common merganser	Merganser																										7		**7**
Common snipe	Shorebird										13				1														**14**
Double-crested cormorant	Cormorant								1																		33		**34**
Gadwall	Duck								2		2	1															10	10	**25**
Great blue heron	Heron		1		3	3	1	3	4		3	8	8	1	4		3		2	1		8	1		3	5	10	7	**79**
Great egret	Heron																											3	**3**
Green heron	Heron																										1		**1**
Green-winged teal	Duck				5	1																				26	52	3	**87**
Hooded merganser	Merganser																										1		**1**
Killdeer	Shorebird										15		1		7											7	8	3	**41**
Least bittern	Heron										1																		**1**
Lesser scaup	Duck										2			8													8	167	**185**
Lesser yellowlegs	Shorebird										14					2										1	1		**18**
Mallard	Duck				2				3		35	42	10	2	4		1		2							16	9		**126**
Northern pintail	Duck																											1	**1**
Northern shoveler	Duck																									1	25	18	**44**
Pied-billed grebe	Grebe																					1							**1**
Ring-billed gull	Gull													3													6	1	**10**
Ring-necked duck	Duck										1																		**1**
Snow goose	Geese																										140		**140**
Solitary sandpiper	Shorebird								2		1						1						2						**6**
Sora	Rail								2																				**2**
Wood duck	Duck	1	4	8	3	26	3	26	2	11	14	27	12					2	1		2	12	4					2	**160**
Total birds / wetland		**3**	**10**	**18**	**15**	**31**	**4**	**35**	**33**	**11**	**337**	**86**	**65**	**18**	**16**	**4**	**5**	**2**	**9**	**1**	**11**	**40**	**25**	**5**	**3**	**100**	**402**	**229**	**1518**
Total species / wetland		**2**	**4**	**3**	**6**	**4**	**2**	**3**	**9**	**1**	**13**	**6**	**7**	**5**	**4**	**2**	**3**	**1**	**4**	**1**	**3**	**5**	**4**	**1**	**1**	**9**	**20**	**13**	**136**

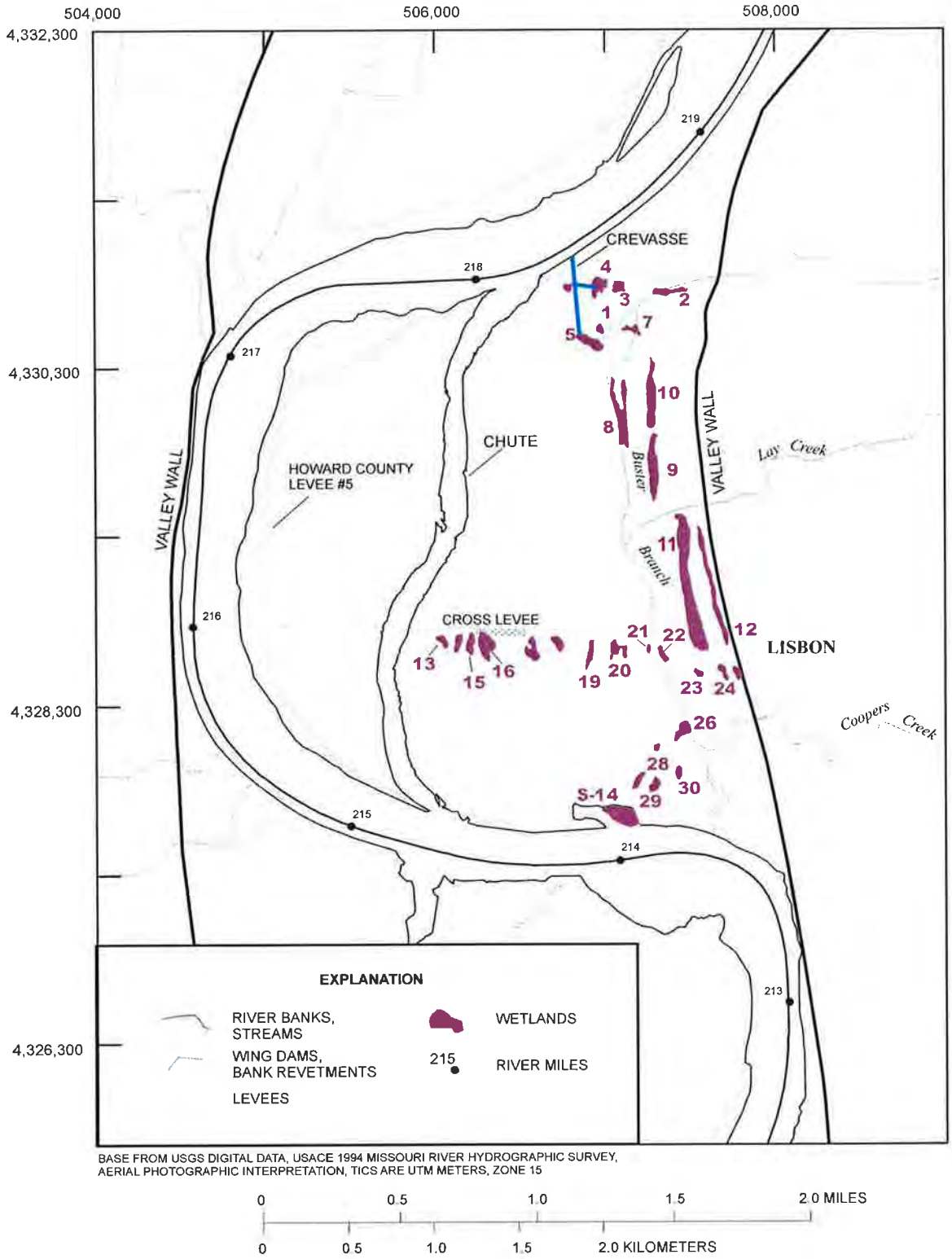

Figure 6-1. Map of Lisbon Bottom wetlands surveyed during the waterbird study, Spring1999. Refer to table 6-1 for basin assignment of wetlands.

Figure 6-2. Observation of a flood-plain wetland along the survey route at Lisbon Bottom.

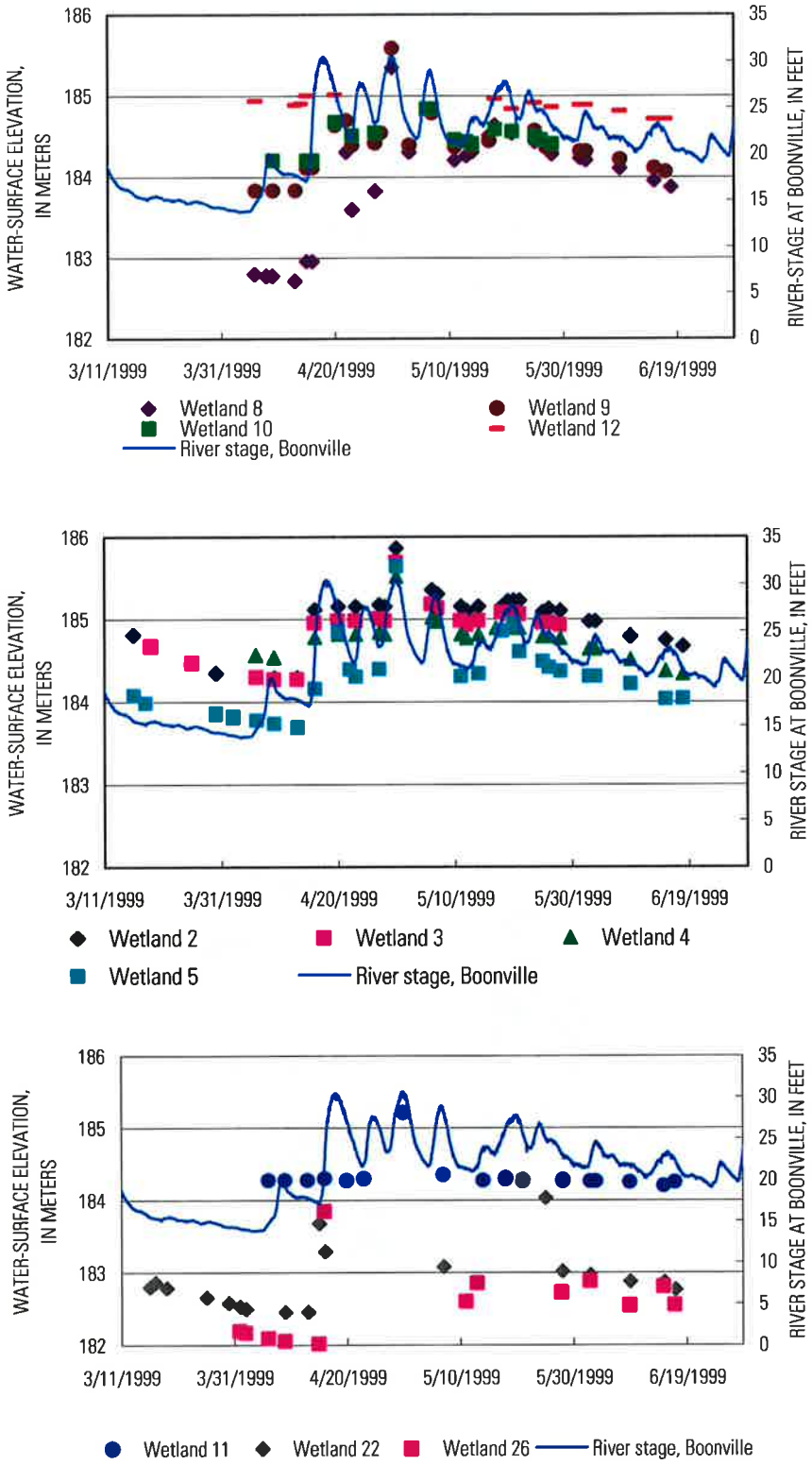

Figure 6-3. Comparison of water surface elevations in selected wetlands observed at Lisbon Bottom, Spring 1999.

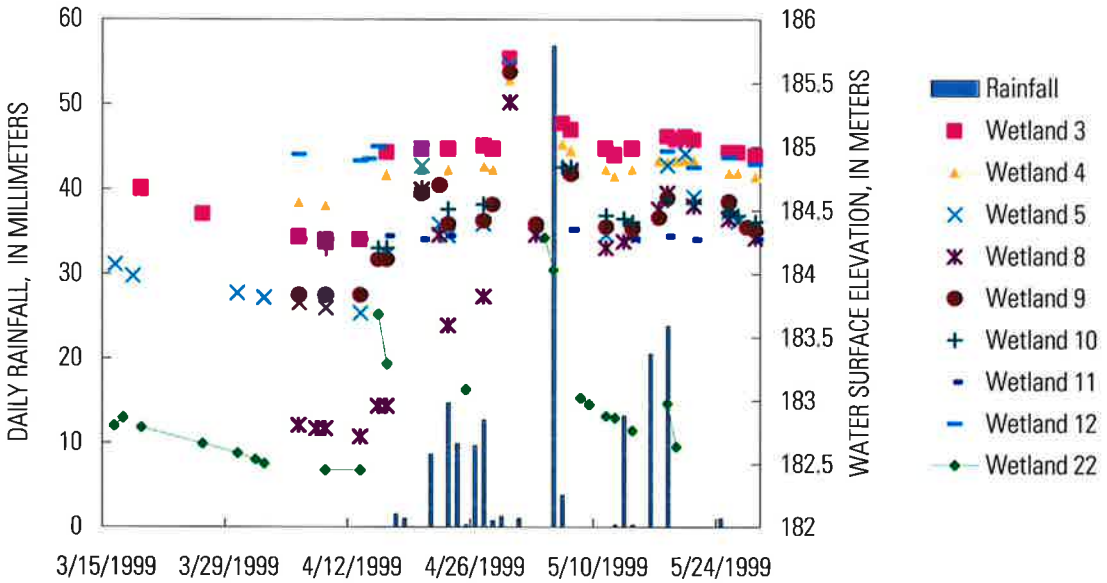

Figure 6-4. Comparison of water levels in individual wetlands compared to rainfall at Lisbon Bottom in Spring 1999.

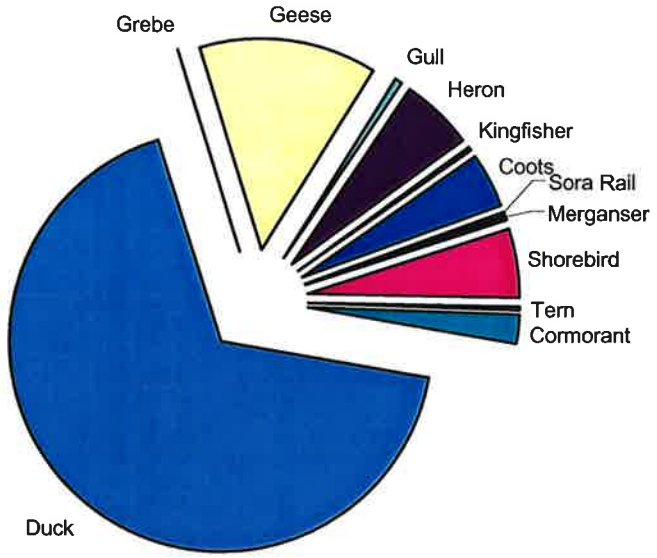

Figure 6-5. Pie chart of waterbirds by group observed at Lisbon Bottom, Spring 1999. Refer to table 6-2 for a list of species within each group.

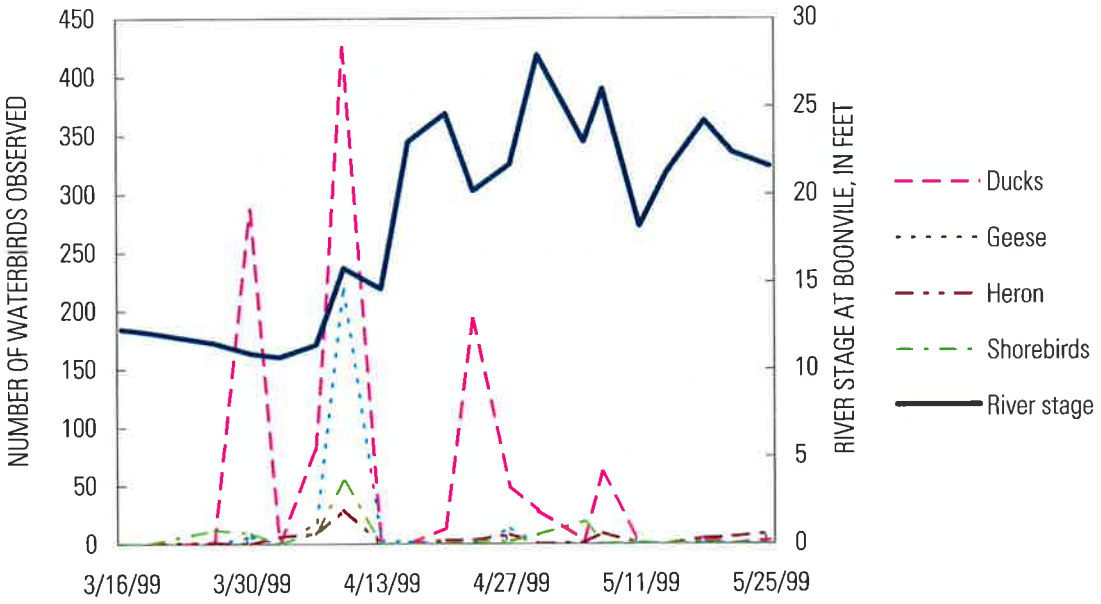

Figure 6-6. Chronology of waterbirds by group observed at Lisbon Bottom, Spring 1999.

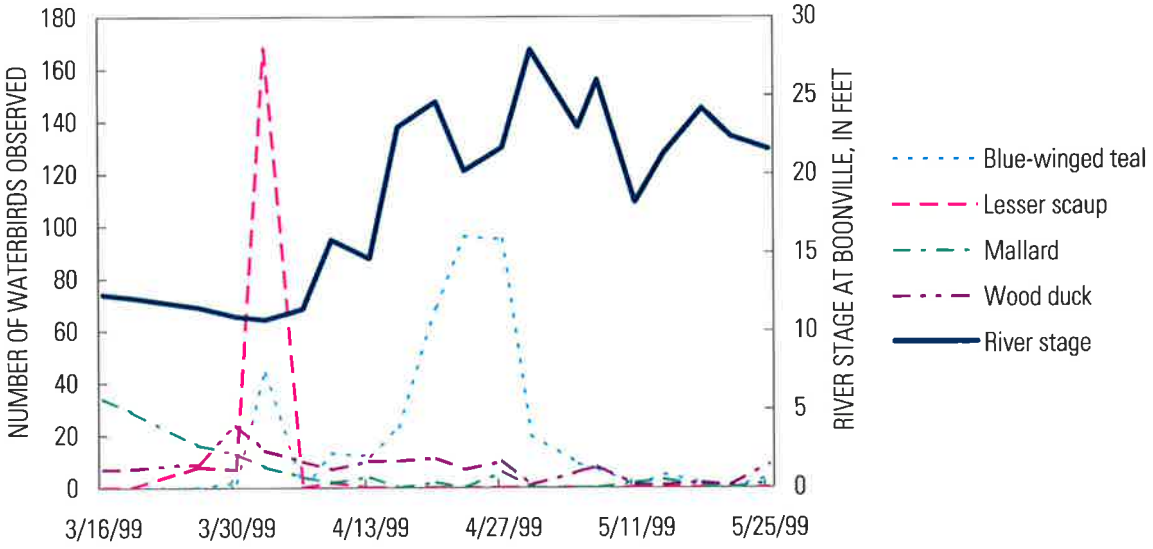

Figure 6-7. Chronology of duck numbers by species observed at Lisbon Bottom, Spring 1999.

Figure 6-8. Blue-winged teal, the most commonly observed duck during the Lisbon Bottom Spring 1999 study (*USGS photo*).

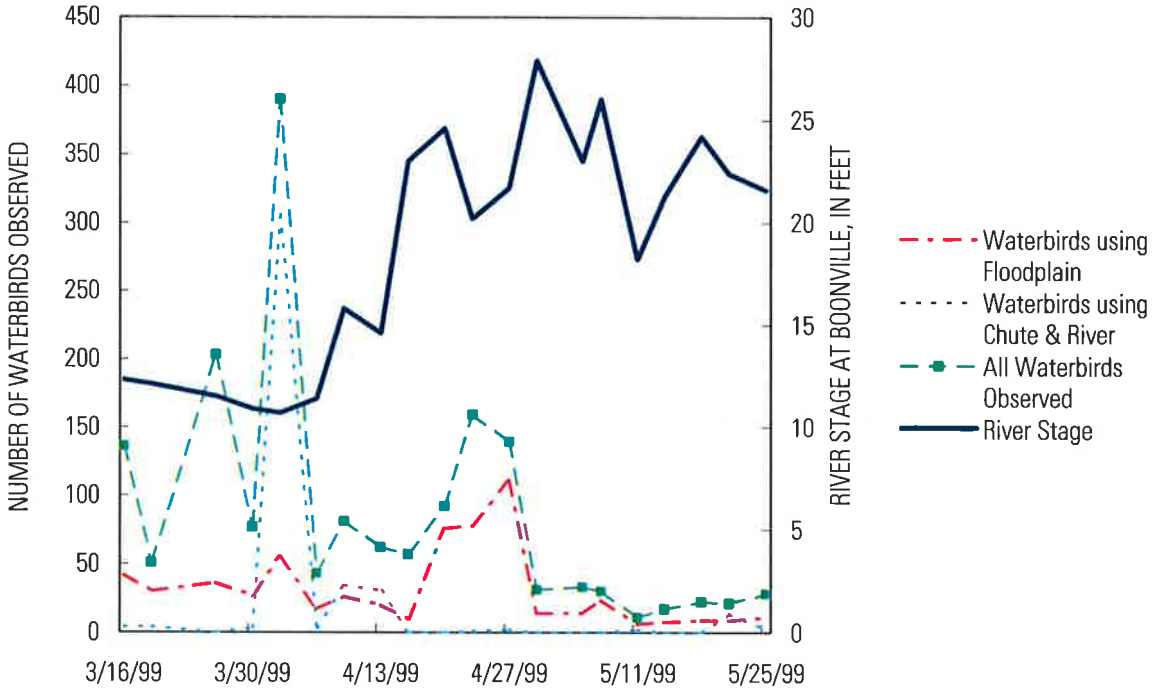

Figure 6-9. Comparison of relative habitat use by waterbirds over the duration of the study at Lisbon Bottom, Spring 1999.

Figure 6-10. Valley-wall wetland at Lisbon Bottom, Spring 1999. Valley-wall wetlands contained high numbers of invertebrates and organic matter that are important ecologically.

Figure 6-11. Snails observed in a valley-wall wetland at Lisbon Bottom. Invertebrate biomass was high in these shallow, vegetated wetlands.

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