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Multiyear Synthesis of Limnological Data from 1993 to 2001 for the Long Term Resource Monitoring Program

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Multiyear Synthesis of Limnological Data from 1993 to 2001 for the Long Term Resource Monitoring Program

Edited by

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Executive Summary

This report presents a broad overview of spatial and temporal variation in the limnological characteristics of the Upper Mississippi River System (UMRS). Important sources of variability within the UMRS include a longitudinal (north–south) gradient, and differences among years, seasons, and aquatic areas (e.g., main channel versus contiguous backwaters). Understanding spatial and temporal variation at these scales requires a systemic perspective on the river. The Long Term Resource Monitoring Program (LTRMP) provides a systemic perspective through the collection and analysis of monitoring data from six study reaches representing the upper, lower, and open river reaches of the UMRS: Pool 4 near Lake City, Minnesota; Pool 8 near Onalaska, Wisconsin; Pool 13 near Bellevue, Iowa; Pool 26 near Alton, Illinois; and Open River, near Cape Girardeau, Missouri; and La Grange Pool of the Illinois River near Havana, Illinois. This report presents data from 1993 to 2001 (or 2002 when available). The focus is on spring and summer conditions because spring is the time of maximum discharge and, therefore, maximum transport of suspended solids and nutrients and summer is the time of peak biological activity. Winter data are included in the analyses of dissolved oxygen because dissolved oxygen may be a critical component of winter habitat for many fish species within the northern reaches of the UMRS.

Discharge is an important driver of physical, chemical, and biotic conditions in large river systems such as the UMRS. Discharge determines longitudinal transport of dissolved and particulate constituents and is correlated with water surface elevation that determines the degree of connection among habitats within the UMRS. Discharge from 1993 to 2001 was variable among reaches and years, but general patterns were evident. In most years, maximum discharge was observed in spring and early summer with an occasional secondary peak in fall. In the southern reaches, maximum discharge between 1993 and 2001 was observed during the 1993 flood; in the northern reaches, maximum discharge was observed in spring 2001. Discharge generally was low in 2000, particularly in the northern reaches of the UMRS.

Limnological characteristics determine water quality and affect aquatic habitat availability in the UMRS. Turbidity and suspended solids and, to a lesser extent, chlorophyll *a*, are the primary determinants of light penetration and, therefore, may affect the distribution of aquatic vegetation. In addition, chlorophyll *a*, an indicator of phytoplankton abundance, serves as a measure of one component of biotic production at the base of the food web. Dissolved nutrients are necessary for vegetation and phytoplankton growth, but excess nutrients can cause noxious algal blooms which can reduce dissolved oxygen concentrations. Sufficient concentration of dissolved oxygen (DO) is a critical aspect of habitat suitability for many species of fish.

Spatial patterns in turbidity, total suspended solids, and chlorophyll *a* concentrations were apparent within the UMRS. There were distinct longitudinal patterns in turbidity and suspended solids concentrations; both were much lower in lower Pool 4 than in upper Pool 4 because of the trapping of sediments by Lake Pepin. Turbidity and suspended solids increased in each of the study reaches from lower Pool 4 to Open River. There were not clear longitudinal patterns in chlorophyll *a* concentrations, but there were spatial patterns within study reaches. For example, chlorophyll *a* concentrations were generally higher in the contiguous backwaters (hereafter referred to as backwaters) than in the main channel.

There were no clear longitudinal patterns in dissolved nutrients, but there were differences among aquatic areas within study reaches. Spring nitrate + nitrite (NO_x) concentrations varied substantially among years, but all of the UMRS study reaches exhibited markedly similar NO_x concentrations in any given year. The differences among years appeared to be driven by discharge. Low discharge years generally exhibited low NO_x concentrations. Within study reaches, summer NO_x is generally lower in backwaters than in the main channel. This was probably because of uptake by primary producers (phytoplankton and aquatic vegetation) and dinitrification combined with limited delivery of NO_x

to backwaters in low summer flows. La Grange Pool of the Illinois River usually exhibited notably higher dissolved nutrient concentrations than did any study reaches of the Upper Mississippi River.

Seasonal patterns in total suspended solids (TSS), turbidity, chlorophyll *a,* and nutrient concentrations were evident. Turbidity, TSS, and chlorophyll *a* minima usually occurred in winter. In the northern study reaches (4, 8, and 13), winter turbidity and TSS were particularly low and exhibited little variation. Seasonal chlorophyll *a* concentrations in main channels and backwaters show peaks in late summer and fall, with minima in winter and early summer. An exception to this pattern is the backwaters of La Grange Pool, where chlorophyll *a* concentrations remained elevated from April through September. Contrasting seasonal patterns were seen for dissolved nitrogen and phosphorus. Main channel soluble reactive phosphorus (SRP) concentrations peaked in September and exhibited minima in April and May; main channel and backwater NO_x concentrations exhibited minima in fall when SRP concentrations are at their maximum.

There were not strong longitudinal trends in DO concentrations, although there were differences among study reaches. Seasonal DO patterns differed slightly among the northern (Pools 4, 8, and 13) and southern (Pool 26, Open River, La Grange Pool) study reaches. In the northern study reaches, DO concentrations are generally highest in spring, lowest in summer and winter, and intermediate in fall. In the southern study reaches, DO concentrations are highest in winter, lowest in summer, and intermediate in spring and fall. The overall occurrence of low, daytime, surface DO concentration was infrequent in UMRS (about 4% of the sites monitored were <5 mg/L); however, during certain seasons and in certain areas, low surface DO did occur. Low DO on the Upper Mississippi River was most frequently observed in summer (12–21% of backwater sampling sites in Pools 8, 13, and 26) followed by winter (10–14% of backwater sites in Pools 4, 8, and 13) and was regularly observed only in backwater areas. In La Grange Pool, low summer DO occurred regularly in the main (25% of sites sampled) and side channel (35% of sites sampled) areas.

Spatial patterns in conductivity were used to examine the spatial extent of tributary influence on the UMRS. The effects of tributary inputs on spatial patterns in conductivity in the UMRS vary greatly among tributaries because of differences in discharge among tributaries and annual and seasonal hydrologic variations in the LTRMP study reaches. Of the six LTRMP study reaches, Pools 4, 8, and 26 showed clear spatial patterns in conductivity related to tributary input in some years whereas Pool 13, Open River, and La Grange Pool did not. The effect of tributaries on the spatial patterns in specific conductivity varied from year-to-year most likely because of differences in tributary and river discharge. There were no obvious effects of tributaries on spatial patterns in total phosphorus (TP), total nitrogen (TN), or TSS.

Three studies of the relation among sampling sites in the backwaters, tributaries, and main channels suggest that concentrations of TP, TN, and TSS at backwater sampling sites were correlated with concentration at both tributary and main channel sites. However, concentrations at the backwater sites were more strongly correlated with concentrations at the main channel sites than with concentrations at the tributary sites.

The patterns described here suggest possible processes that affect water quality in the UMRS. Additional analyses and specifically designed studies are needed to better understand the processes creating the patterns described here. Specific recommendations for additional analyses are presented in individual chapters.

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Preface

The Long Term Resource Monitoring Program (LTRMP) was authorized under the Water Resources Development Act of 1986 (Public Law 99-662) as an element of the U.S. Army Corps of Engineers' Environmental Management Program. The LTRMP is implemented by the Upper Midwest Environmental Sciences Center, a U.S. Geological Survey science center, in cooperation with the five Upper Mississippi River System (UMRS) States of Illinois, Iowa, Minnesota, Missouri, and Wisconsin. The U.S. Army Corps of Engineers provides guidance and has overall Program responsibility. The mode of operation and respective roles of the agencies are outlined in a 1988 Memorandum of Agreement.

The UMRS encompasses the commercially navigable reaches of the Upper Mississippi River, as well as the Illinois River and navigable portions of the Kaskaskia, Black, St. Croix, and Minnesota Rivers. Congress has declared the UMRS as both a nationally significant ecosystem and a nationally significant commercial navigation system. The mission of the LTRMP is to provide decision makers with information for maintaining the UMRS as a sustainable large river ecosystem given its multiple-use character. The long-term goals of the Program are to understand the system, determine resource trends and effects, develop management alternatives, manage information, and develop useful products.

This multiyear report supports Task 2.2.3 as specified in Goal 2, *Monitor Resource Change*, of the LTRMP Operating Plan (U.S. Fish and Wildlife Service 1993). This report was developed with funding provided by the LTRMP.

Multiyear Synthesis of Limnological Data from 1993 to 2001 for the Long Term Resource Monitoring Program

Edited by Jeffrey N. Houser[1]

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Abstract: This report presents a broad overview of spatial and temporal variation in the water quality of the Upper Mississippi River System (UMRS). The Long Term Resource Monitoring Program (LTRMP) provides a systemic perspective through the collection and analysis of monitoring data from six study reaches representing the upper, lower, and open river reaches of the UMRS (Upper Mississippi River: Pools 4, 8, 13, and 26, Open River [near Cape Girardeau, Missouri]; Illinois River: La Grange Pool). This report presents data from 1993 to 2001 (or 2002 when available) and focuses on spring and summer conditions. Water quality constituents (e.g., turbidity, suspended solids, chlorophyll, nutrients, and dissolved oxygen) varied among study reaches, aquatic area (e.g., main channel, contiguous backwaters, etc.) and seasons. For example, turbidity and suspended solids varied substantially among pools. Turbidity and suspended solids were much lower in lower Pool 4 than in upper Pool 4 because of the trapping of sediments by Lake Pepin, but increased in each of the study reaches from Pool 4 to Open River. Chlorophyll *a* and nutrient concentrations often differed between the main channel and contiguous backwater areas (hereafter referred to as backwaters). Summer chlorophyll *a* concentrations were generally higher in backwaters than in the main channel, and summer nitrate + nitrite (NO_x) concentrations were generally lower in backwaters than in the main channel. Seasonal patterns were evident in chlorophyll *a*, nutrient, and dissolved oxygen (DO) concentrations. Main channel soluble reactive phosphorus (SRP) concentrations peaked in September and exhibited minima in April and May. In contrast, main channel and backwater NO_x concentrations exhibited minima in fall when SRP concentrations are at their maximum. Seasonal chlorophyll *a* concentrations in main channels and backwaters show peaks in late summer and fall, with minima in winter and early summer. Seasonal DO patterns differed slightly among the northern (Pools 4, 8, and 13) and southern (Pool 26, Open River, La Grange Pool) study reaches. In the northern study reaches, DO concentrations are generally highest in spring, lowest in summer and winter, and intermediate in fall. In the southern study reaches, DO concentrations are highest in winter, lowest in summer, and intermediate in spring and fall. Spatial patterns within study reaches caused by tributary inputs were shown by the spatial patterns in specific conductivity. Of the six LTRMP study reaches, Pools 4, 8, and 26 showed clear spatial patterns in conductivity related to tributary input in some years whereas Pool 13, Open River, and La Grange Pool did not.

Key words: dissolved oxygen, ecosystem monitoring, hydrology, Mississippi River, nitrate, nitrogen, phosphorus, spatial patterns, suspended solids, temporal patterns

Chapter 1: Introduction

by

Jeffrey N. Houser

Rivers ecosystems are highly heterogeneous in time and space. Our understanding of what drives this heterogeneity is poor. Because of its systemic perspective, the Long Term Resource Monitoring Program (LTRMP) is well suited to improve our understanding of this temporal and spatial heterogeneity in the Upper Mississippi River System (UMRS). Important sources of variability in water quality include the longitudinal (north–south) gradient and differences among years, seasons, and aquatic areas.

A number of important drivers of river water quality change along the north–south gradient of the UMRS. Climatic conditions differ between the northern and southern reaches of the UMRS. For example, winters are longer and colder in the northern reaches. The river is affected by the tributaries that enter it as it flows south (Wasley 2000). Geomorphology and land use in the basin also change along a north–south gradient. The proportion of the floodplain used for agriculture increases moving from north to south (Yin 1998), and levees become more extensive. To what extent are these changes in geomorphology and land use reflected in river water quality?

There are also likely to be spatial patterns in water quality at a smaller scale because of the broad range of geomorphic characteristics in the river. For example, main channel areas are characterized by deeper water and faster current velocities than are backwater areas. The LTRMP uses a sampling design stratified across the geomorphic units (referred to as strata in this report) to accommodate the variation among different geomorphic areas in the UMRS (Soballe and Fischer 2004).

There is temporal variation in the water quality of the UMRS because of seasonal and interannual differences. Differences among years may be driven by year-to-year differences in precipitation and temperature. Trends across many years may be driven by long-term changes in climate or land use. Seasonal patterns may be driven by seasonal differences in discharge, solar irradiation, precipitation, evapotransporation, and temperature. What effect does seasonality in these drivers have on river water quality?

The design of the monitoring program was modified substantially in 1993, and this report covers data from 1993 through 2001 or 2002 depending on data availability at the time of analysis. Soballe and Fischer (2004) provide detailed information concerning the LTRMP field sampling and laboratory procedures; only a brief discussion is included here (an electronic copy of Soballe and Fischer [2004] and all of the LTRMP water quality data are available online at http://www.umesc.usgs.gov/data_library/water_quality/water_quality_page.html). To gain a systemic perspective on the UMRS, six study reaches representing the upper, lower, and open river reaches of the UMR and the lower Illinois River are monitored by LTRMP: Pool 4 near Lake City, Minnesota (river mile 753 to 797); Pool 8 near Onalaska, Wisconsin (river mile 679 to 702.5); Pool 13 near Bellevue, Iowa (river mile 522.5 to 557); Pool 26 near Alton, Illinois (river mile 203 to 241.5); Open River near Cape Girardeau, Missouri (river mile 29 to 80); and La Grange Pool of the Illinois River near Havana, Illinois (river mile 80 to 158; Figure 1.1). The LTRMP water quality component uses two complimentary sampling programs: fixed-site and stratified random sampling. The fixed-site sampling monitors inflows, outflows, selected tributaries, and other areas of special interest at a limited number of sites (13–15) in each study reach at biweekly or monthly intervals. The relatively high temporal resolution of the fixed-site data makes it useful to examine seasonal patterns. However, the fixed-site data are not suitable for making inferences about strata or study reaches because of the subjective criteria used for their selection.

Stratified random sampling is more limited in temporal resolution, conducted in four

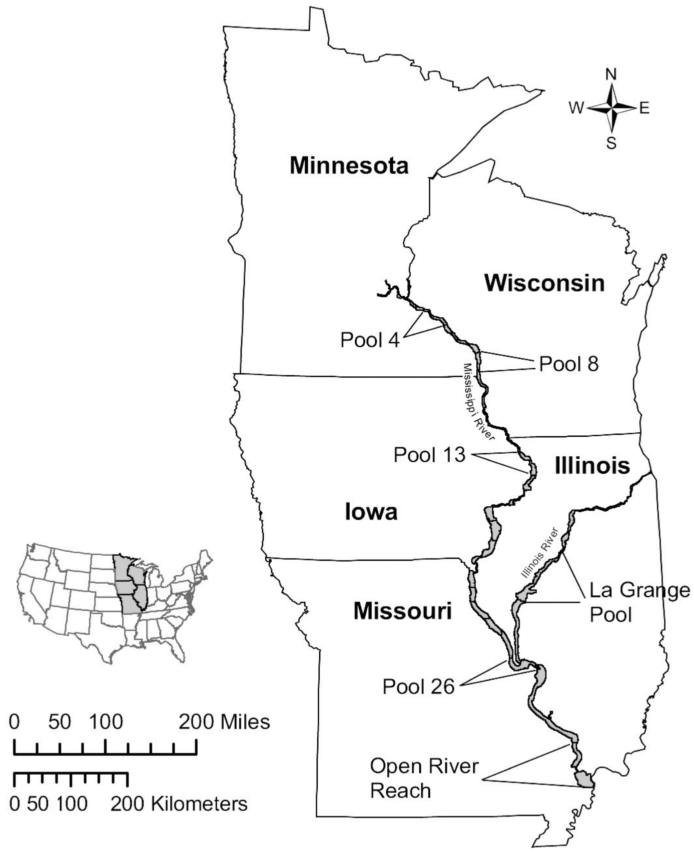

Figure 1.1. Study reaches monitored in the Upper Mississippi River System by the Long Term Resource Monitoring Program field stations.

quarterly episodes each year, but is at a much higher spatial resolution. The SRS sampling began in summer 1993 for Pools 4, 8, and 13 and La Grange Pool and began in fall 1993 for Pool 26 and Open River. Each quarterly sampling episode is generally completed in 10 to 14 days. The spring episode begins the last week of April, the summer episode begins the last week of July, the fall episode begins the second week of October, and the winter episode begins the last week of January. Between 135 and 150 sampling sites are randomly selected from six strata. The sampling strata are based on geomorphic characteristics as described in Wilcox (1993) and included the following: main channel, side channel, impounded areas, contiguous backwaters, and isolated backwaters. This report focuses on data from contiguous backwaters (hereafter referred to as backwaters) and the main channel because they represent an informative contrast in both depth and current velocity. The main channel is deep and has high current velocity; backwaters are generally shallow and have low current velocity. Other common types of aquatic areas in the river include side channels and impoundments that are generally intermediate in both depth and current velocity. Details concerning the distribution of sampling effort among these aquatic areas can be found in Soballe and Fischer (2004, Table 3).

The main objective of this multiyear report is to provide a summary of the patterns observed in the UMRS for selected water quality variables from 1993 to 2001 (or 2002 when available). The report focuses on various aspects of temporal and spatial variability in selected water quality parameters in the UMRS. The specific aspects addressed include (1) differences among study reaches; (2) differences among geomorphic strata (e.g., main channel, backwaters); and (3) temporal patterns including differences among years and seasonal patterns.

Chapter 2: Hydrology

by

James T. Rogala

Background

Discharge is an important driver of physical, chemical, and biotic conditions in large river systems such as the Upper Mississippi River System (UMRS). Discharge determines longitudinal transport of dissolved and particulate constituents, and these constituents are important determinants of habitat conditions. Discharge is correlated with water surface elevation which determines the degree of connection among habitats within the UMRS. When water elevations are high, lateral transport occurs among the main channel, off-channel and terrestrial areas. Thus, discharge provides a measure of lateral transport (i.e., channel materials delivered to off-channels and terrestrial areas), as well as a measure of habitat conditions (e.g., water depth). Physical energy is also a function of discharge; thus, discharge affects the movement of bed materials, and high discharge can affect biotic components (e.g., aquatic plant breakage).

A summary of discharge conditions in 1993 to 2001 provides context for the observed patterns in limnological characteristics. In addition, the 9-year sampling period is placed into a larger temporal context using historical discharge data.

Methods

Daily discharge data were acquired from the U.S. Geological Survey gaging stations (U.S. Geological Survey 2003) closest to the six Long Term Resource Monitoring Program (LTRMP) study reaches (Table 2.1). Only gaging stations were used that had been in operation at the time of impoundment in the 1930s and had discharge records through the entire LTRMP sampling period of 1993 to 2001. The only gap in these discharge records was for the Kingston, Illinois gage, where recording ended in September 2001. Means and standard deviations by study reach and year were calculated from daily data for the following seasons: (1) winter—previous December through February, (2) spring—March through May, (3) summer—June through August, and (4) fall—September through November.

Table 2.1. U.S. Geological Survey (USGS) gaging stations selected to represent the Long Term Resource Monitoring Program (LTRMP) study reaches (see Figure 1.1).

LTRMP sampling reach	USGS gaging station
Pool 4	Prescott, Minnesota
Pool 8	Winona, Minnesota
Pool 13	Clinton, Iowa
Pool 26	Grafton, Illinois
Open River	Thebes, Missouri
La Grange Pool	Kingston, Illinois

The results below are focused on annual patterns of discharge within each reach by season. Because the interest is mainly in interannual patterns in discharge within each study reach for each season, the data were standardized by study reach for each season in two distinct ways. First, the discharge data were standardized to the 1993–2001 mean to emphasize differences within the LTRMP sampling period. This provides for comparisons that are relevant when addressing differences in water quality values observed in the LTRMP sampling period. The discharge data were also standardized to a long-term (60-year) mean representing the period of impoundment to illustrate how the sampling period compares to past discharge conditions. In both these examples, the data were standardized by setting the mean for each reach and season during the period of comparison (9 or 60 years) equal to zero, and the standard deviation for each reach and season during the period of comparison to one.

Results

The annual hydrograph is variable among study reaches and years; however, general patterns are evident. In most years, maximum discharge was observed in spring and early summer (Figure 2.1), with an occasional secondary peak in fall. The highest discharge over the 9-year sampling period occurred during the 1993 flood, which in addition to the high discharge, was unusual in that it extended much later into summer than is typical for the Upper Mississippi River.

Seasonal patterns in discharge can also be observed by comparing seasonal means for each study reach (Table 2.2). Discharge was highest in spring for all study reaches, with summer discharge being the next highest (ranging from 70% to 90% of the spring discharge) for all study reaches. In the northern three study reaches, discharge was higher in fall than winter, whereas fall discharge was similar to winter discharge in Pool 26 and Open River. Winter discharges were higher than fall discharges in La Grange Pool. Discharge for all seasons increased with location downstream in the Upper Mississippi River.

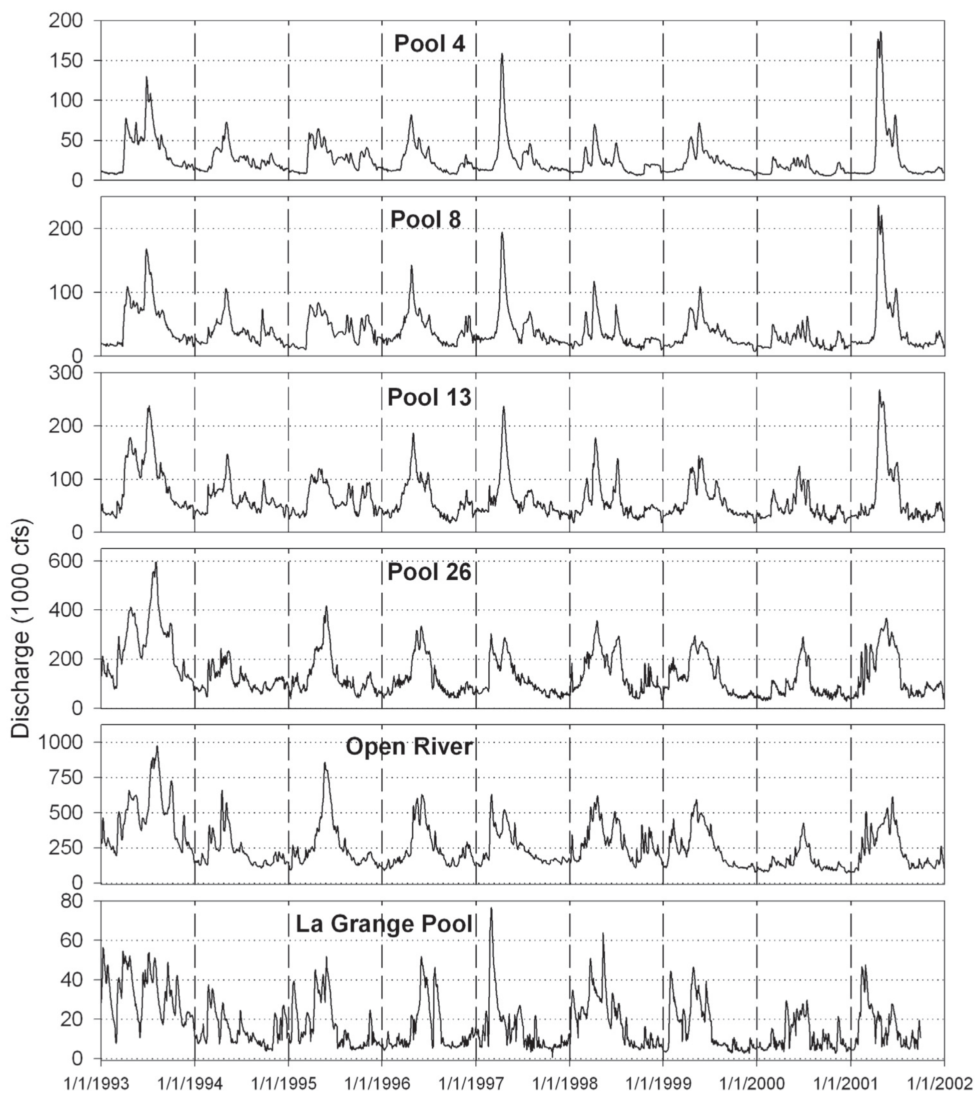

Figure 2.1. Daily discharge (cfs) for the nearest U.S. Geological Station gaging station for each of the six Long Term Resource Monitoring Program study reaches, 1993–2001. Note that the y-axis differs among study reaches.

Table 2.2. Mean and standard deviation (SD) for discharge (cfs) from the 9-year sampling period for each study reach by season.

Study reach	Winter (December–February) Mean	Winter SD	Spring (March–May) Mean	Spring SD	Summer (June–August) Mean	Summer SD	Fall (September–November) Mean	Fall SD
Pool 4	12,215	1,815	42,919	14,845	30,161	14,971	16,520	6,016
Pool 8	21,791	4,395	63,154	18,875	45,590	19,503	27,649	8,483
Pool 13	38,019	3,832	95,314	22,218	74,599	29,434	45,032	12,660
Pool 26	88,240	22,903	203,659	59,715	174,322	80,604	88,268	50,039
Open River	177,119	46,876	365,809	96,970	328,034	136,060	184,121	98,108
La Grange Pool	16,122	6,728	24,521	8,442	18,787	8,060	10,179	7,165

These overall patterns can be used to interpret the standardized discharge data and to assess the annual discharge conditions in the 9-year study period.

The standardized discharge data illustrate seasonal patterns across years for each study reach. First, the differences were summarized among study reaches, as observed as differences along the y-axis, within years by season using discharge standardized to the 1993–2001 mean. In some years, conditions within seasons were similar among all study reaches, such as the high discharge in summer 1993 and the low discharge conditions in spring 2000 (Figure 2.2). However, in other years, there were differences in discharge patterns among the study reaches. For example, discharge was high relative to other years in only the northern three study reaches (Pools 4, 8, and 13) in winter 1993. Figure 2.2 can also be used to look for trends over the study period. In general, fall discharge steadily decreased from 1993 to 2001 but no trend is apparent for any other season.

To simplify the summary of annual discharge conditions across seasons, years of extreme discharge were identified on the basis of the standard deviation from the 9-year mean, by season within study reaches (Table 2.3). Despite the differences among study reaches, several annual patterns are evident across study reaches. Discharge was high in 1993 across all seasons, and highest in 1993 for all three southern study reaches (Pool 26, Open River, and La Grange Pool). Discharge was low in 2000, particularly in winter and spring.

Discharge data standardized to the 60-year mean show that yearly seasonal discharge tended to be above the historical mean in winter, spring, and summer for most years and study reaches, whereas fall discharge during the study period was below the historical mean for the latter years of the study period (Figure 2.3). Discharge recorded since 1874 at the Clinton, Iowa, gage suggests that the recent 60-year period may be on the rising limb of a large-scale pattern for all seasons (Figure 2.4). A similar pattern was reported by Knox (2000), where large floods were found to be more common in the late 1800s and since about 1950. Knox (2000) also suggests that large-scale atmospheric circulation patterns play a large role in this pattern, as well as larger-scale patterns in Holocene floods.

In summary, the period 1993–2001 was one of relatively high discharge on both 60- and 140-year scales. The most striking hydrologic event in this 9-year period was the 1993 flood. At the other end of the spectrum, 2000 was the driest year, particularly in winter and spring. These two contrasting hydrologic events may provide useful comparisons in analyses of ecological dynamics. Further, these patterns in discharge provide a useful context for considering the annual and seasonal patterns observed in UMRS water quality.

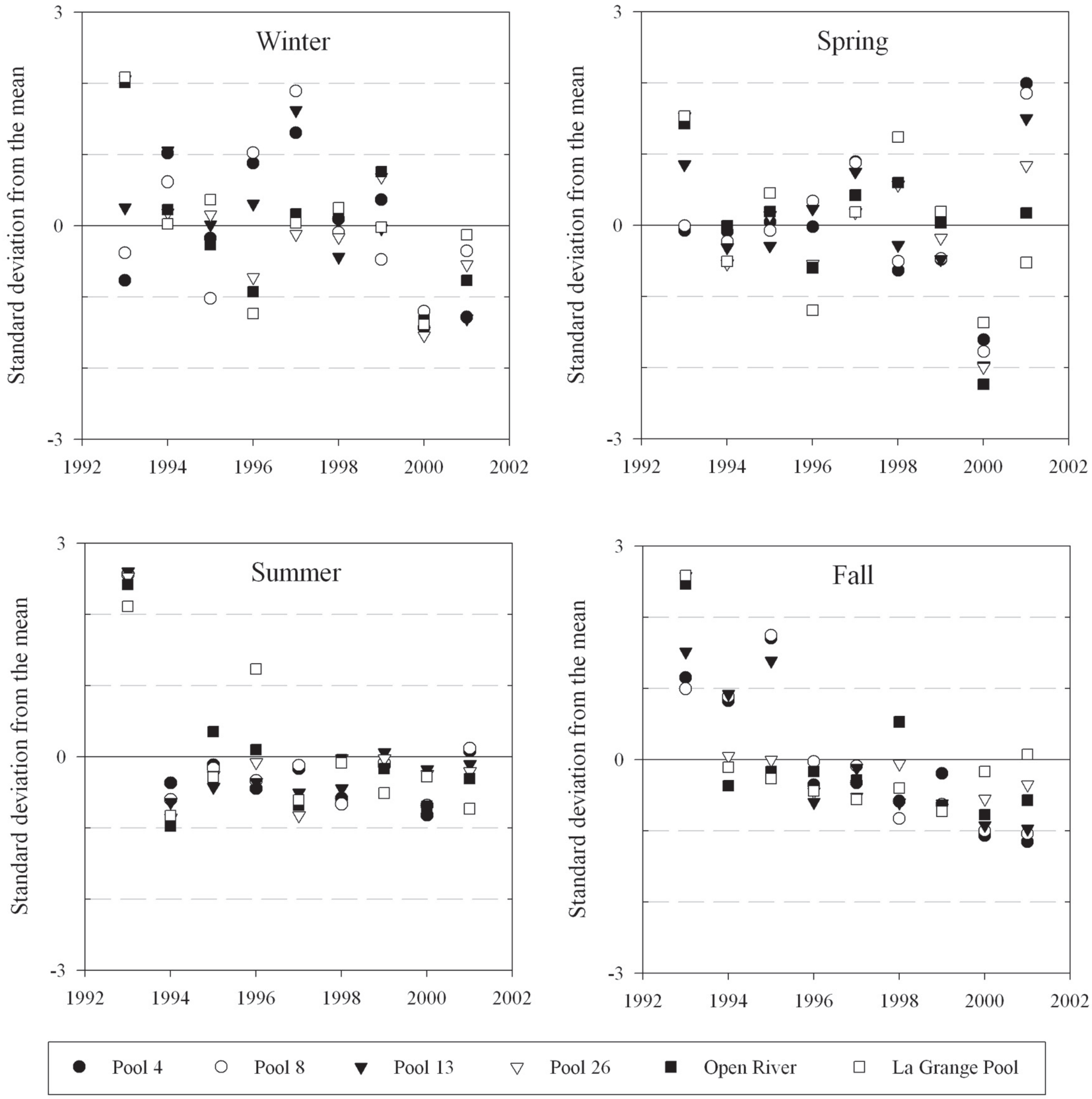

Figure 2.2. Standardized seasonal annual mean discharges for the six gages near the Long Term Resource Monitoring Program study reaches during the 9-year sampling period (1993–2001). Discharge data were standardized using the mean and standard deviations by season for each study reach for the 9-year period, where the units on the y-axis are standard deviations from the 9-year period mean.

Table 2.3. Discharge conditions relative to mean discharge (1993–2001) by season and by study reach for the 9-year study period. Conditions are ranked very high (**H**), high (H), low (L), or very low (**L**) based on standard deviation (SD) from the mean, with very high and very low being >1 SD from the mean and high and low being between 0.5 and 1 SD from the mean. Column heading is a numerical code where 1 = Pool 4; 2 = Pool 8; 3= Pool 13; 4 = Pool 26; 5 = Open River; and 6 = La Grange Pool.

	Winter						Spring						Summer						Fall					
Year	1	2	3	4	5	6	1	2	3	4	5	6	1	2	3	4	5	6	1	2	3	4	5	6
1993	L			**H**	**H**	**H**			H	**H**	**H**	**H**	**H**	**H**	**H**	**H**	**H**	**H**	**H**	H	**H**	**H**	**H**	**H**
1994	**H**	H	**H**							L		L		L	L	L	L	L	H	H	H			
1995		**L**																	**H**	**H**	**H**			
1996	H	**H**		L	L	**L**				L	L	**L**						**H**			L			
1997	**H**	**H**	**H**				H	H	H						L	L	L	L				L		L
1998							L	L		H	H	**H**	L	L					L	L	L		H	
1999				H	H													L		L	L	L	L	L
2000	**L**	**L**	**L**	**L**	**L**	**L**	**L**	**L**	**L**	**L**	**L**	L	L	L			L		**L**	**L**	L	L	L	
2001	**L**		**L**	L	L		**H**	**H**	**H**	H		L						L	**L**	**L**	L		L	

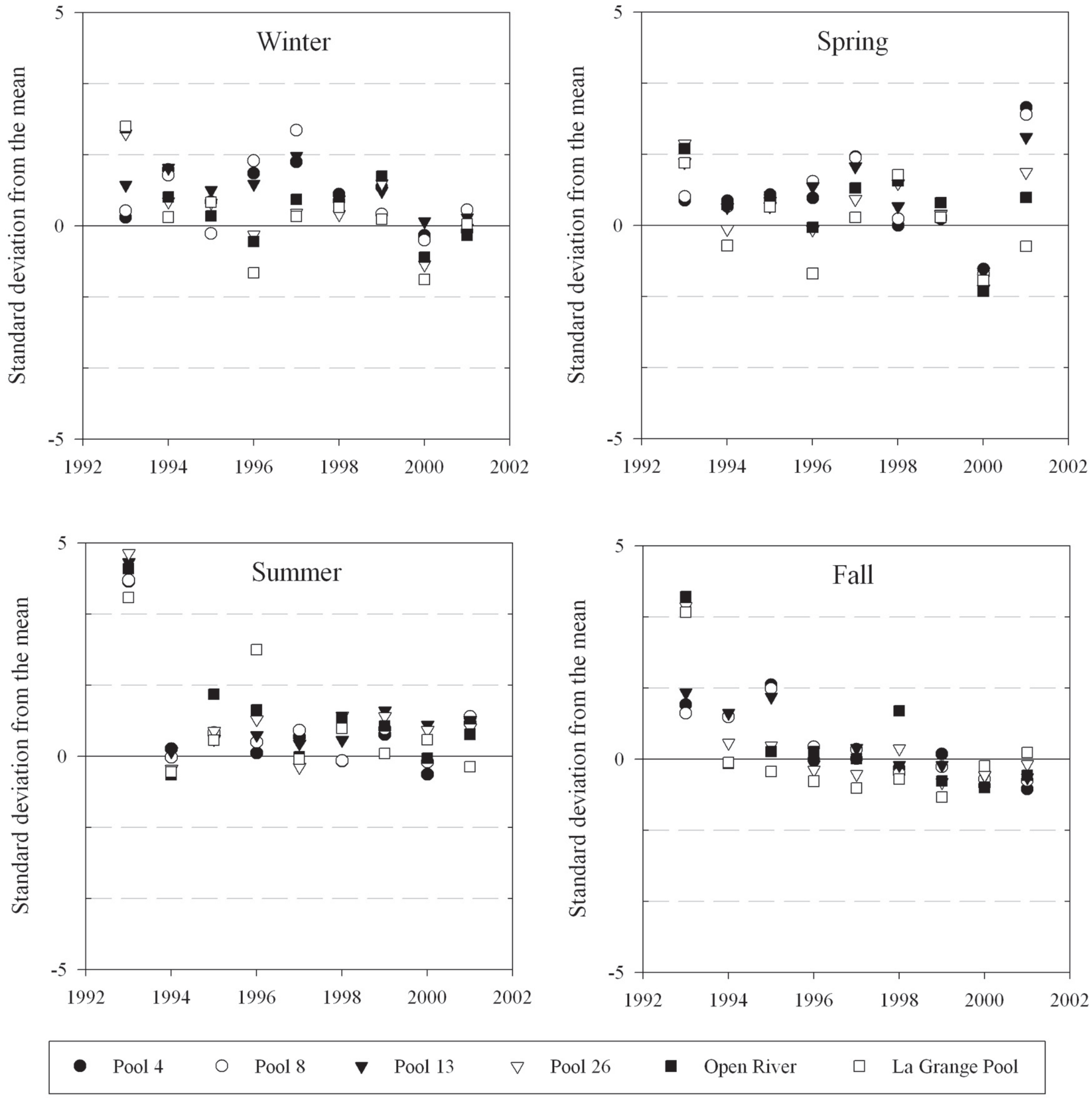

Figure 2.3. Standardized seasonal annual mean discharges for the six gages near the Long Term Resource Monitoring Program study reaches during the 9-year sampling period (1993–2001). Discharge data were standardized using the mean and standard deviations by season for each study reach for a 60-year period (1942–2001), where the units on the y-axis are standard deviations from the 60-year mean.

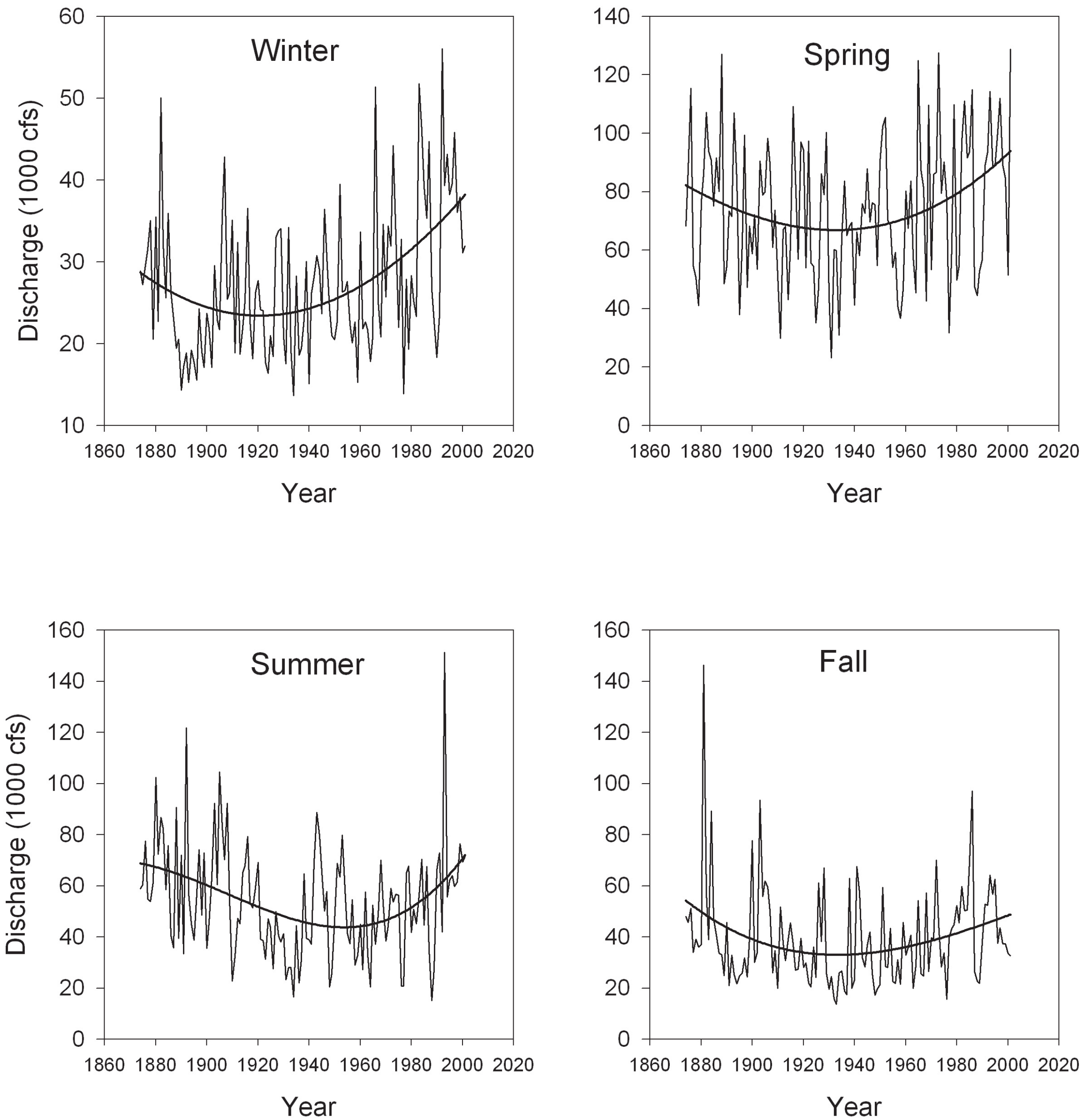

Figure 2.4. Yearly seasonal mean discharge (cfs) at Clinton, Iowa, for a 128-year period (1874–2001). The fitted line represents a third-order polynomial relation.

Chapter 3: Spatial Patterns in Suspended Solids, Turbidity, and Chlorophyll *a*

by

Robert M. Burdis

Background

Anthropogenic activity in the United States has altered land use and land cover causing increased runoff and erosion of susceptible land. This is particularly evident in streams and rivers in areas of intense agriculture where farming activities on floodplains and in riparian zones have been identified as principal sources of sediment (Waters 1995). Agriculture, which accounts for more than 60% of the land use in the Upper Mississippi River Basin (Gowda 1999), has greatly affected water quality in the Upper Mississippi River System (UMRS). In some areas of the UMRS, biological assemblages have been altered because of the increase of sediment and nutrients into the system. For example, benthic algal communities, common in Lake Pepin in presettlement times, have been greatly reduced because of shading by increased turbidity from suspended sediments and phytoplankton (Engstrom and Almendinger 2000).

Suspended solids, turbidity, and chlorophyll *a* are common measures of water quality that reflect the amount of various materials in the water column. Total suspended solids (TSS) is all the material suspended in the water, both biotic and abiotic, that can be removed by filtration. Total suspended solids is composed of organic (volatile) and inorganic fractions. Phytoplankton, bacteria, and nonliving organic material in the water column comprise the volatile suspended solids. Chlorophyll *a* is an indicator of the abundance of phytoplankton biomass in the water column. Turbidity is a measure of the light scattering properties of water. As turbidity increases, light is attenuated more rapidly with depth because of scattering and absorption.

Many contaminants in the UMRS are associated with suspended particles and, thus, the transport and biological availability of contaminants are tied to the movement of suspended material in the system (Wiener et al. 1984). The negative effects on aquatic organisms produced by the deposition of suspended material and the degradation of habitat have been well documented (Waters 1995). Total suspended solids strongly affects the depth of light penetration, which can affect the distribution and abundance of macrophytes (Korschgen et al. 1997).

Methods

The analyses in this chapter focused on suspended solids, turbidity, and chlorophyll *a* in contiguous backwaters (hereafter referred to as backwaters) and the main channel. These areas were chosen because they represent an informative contrast in both depth and current velocity. The main channel areas are deep and have high current velocity, backwaters are generally shallow and have low current velocity. Other common types of aquatic areas in the river include side channels and impoundments that are generally intermediate in both depth and current velocity. Surface water samples collected during stratified random sampling (SRS) episodes from 1993 to 2001 were analyzed for spatial and seasonal patterns. In the Upper Mississippi River (UMR), Pool 4 is unique because it contains Lake Pepin, a natural 10,300-ha lake created by the Chippewa River delta. For analysis, Pool 4 was partitioned into an upper reach (above Lake Pepin) and a lower reach (below Lake Pepin) because of the marked effects Lake Pepin has on suspended solids and turbidity.

Results

Longitudinal Patterns in Summer Concentrations of Suspended Solids and Chlorophyll *a*

There were strong longitudinal patterns in summer TSS concentrations among the study reaches. Summer TSS concentrations in upper Pool 4 are high compared to the other two northern study reaches (Pools 8 and 13; Figure 3.1A). Most of this suspended material is inorganic sediment that originates in the agriculture-dominated Minnesota River Basin (Nielsen et al. 1984; James et al. 2000). The Minnesota River has a large influence on water quality in the upper reach of the UMR because it not only carries a large amount of suspended material, but it also contributes about 40% of the discharge at the confluence with the Mississippi River.

As the Mississippi River enters Lake Pepin, it becomes more lacustrine because of the change in morphometry. This results in the settling of suspended material in Lake Pepin where about 85% of the inorganic suspended solids are retained. Mean total suspended solids concentrations on the entire UMR are lowest just below Lake Pepin and increase downstream in each of the study reaches from lower Pool 4 to Open River (Figure 3.1A). Pool 26 is the first of the downstream study reaches to exhibit TSS concentrations similar to upper Pool 4. Tributaries in the southern reaches of the system drain agricultural areas that can contribute high concentrations of suspended material (Soballe and Wiener 1999; Wasley 2000). In contrast, tributaries draining forested regions, typical in the upper portions of the UMRS (e.g., Chippewa, Black, and Wisconsin Rivers), have lower suspended solids concentrations and may cause a dilution effect. The increase in TSS in study reaches from lower Pool 4 to Open River is probably because of the cumulative impacts of tributary inputs (Chapter 6). The Illinois River Basin is also dominated by agriculture. Total suspended solids concentrations in La Grange Pool of the Illinois River are similar to those of Pool 26 and the Open River of the Mississippi River.

Summer TSS concentrations are typically higher in the backwaters than in the main channel (Figure 3.1A), with the exception of Pool 13 and La Grange Pool where main channel solids concentrations are slightly higher. The higher concentrations in backwaters may be due in part to greater phytoplankton abundance (measured as chlorophyll *a*; Figure 3.1E) and sediment resuspension. Backwaters typically have relatively shallow water depths that makes them subject to sediment resuspension because of wind and wave action. Bottom-feeding fish (e.g., common carp *Cyprinus carpio*) have been implicated as the main cause of turbidity in shallow lakes (Scheffer et al. 1993) and may be contributing to resuspension of sediments in the backwaters of the UMR. Johnson (1994) reported an increase in turbidity in the littoral zone of the main channel in upper Pool 4 associated with an increase in recreational boating activity.

Phytoplankton biomass (measured as chlorophyll *a*) is usually greater in backwaters than in the main channel (Figure 3.1E). The higher phytoplankton abundance in backwaters results in organic material (volatile suspended solids) comprising a larger fraction of TSS in the backwaters relative to the main channel (Figure 3.1C). An exception to this occurs in lower Pool 4 where the proportion of volatile suspended solids in the main channel is slightly higher. This is the result of Lake Pepin being both a source of algae and a trap for inorganic material.

In the study reaches downstream from lower Pool 4, the proportion of volatile suspended solids decreases (Figure 3.1C), even though summer chlorophyll *a* concentrations are similar in lower Pool 4, Pools 8 and 13, and increase in Pool 26 and La Grange Pool (Figure 3.1E). This pattern reflects that inorganic suspended solids increase downstream at a greater rate than volatile suspended solids (Figures 3.1B and 3.1D). The three Long Term Resource Monitoring Program (LTRMP) study reaches with the highest mean chlorophyll *a* concentrations in the backwaters (i.e., upper Pool 4, Pool 26, and La Grange Pool) also have the highest proportion of inorganic suspended solids (Figures 3.1C and 3.1E). These high chlorophyll *a* concentrations may be because of the longer residence time of water in some backwaters, which allows phytoplankton biomass to accumulate despite the high inorganic suspended solids concentrations that can inhibit phytoplankton growth by reducing light penetration.

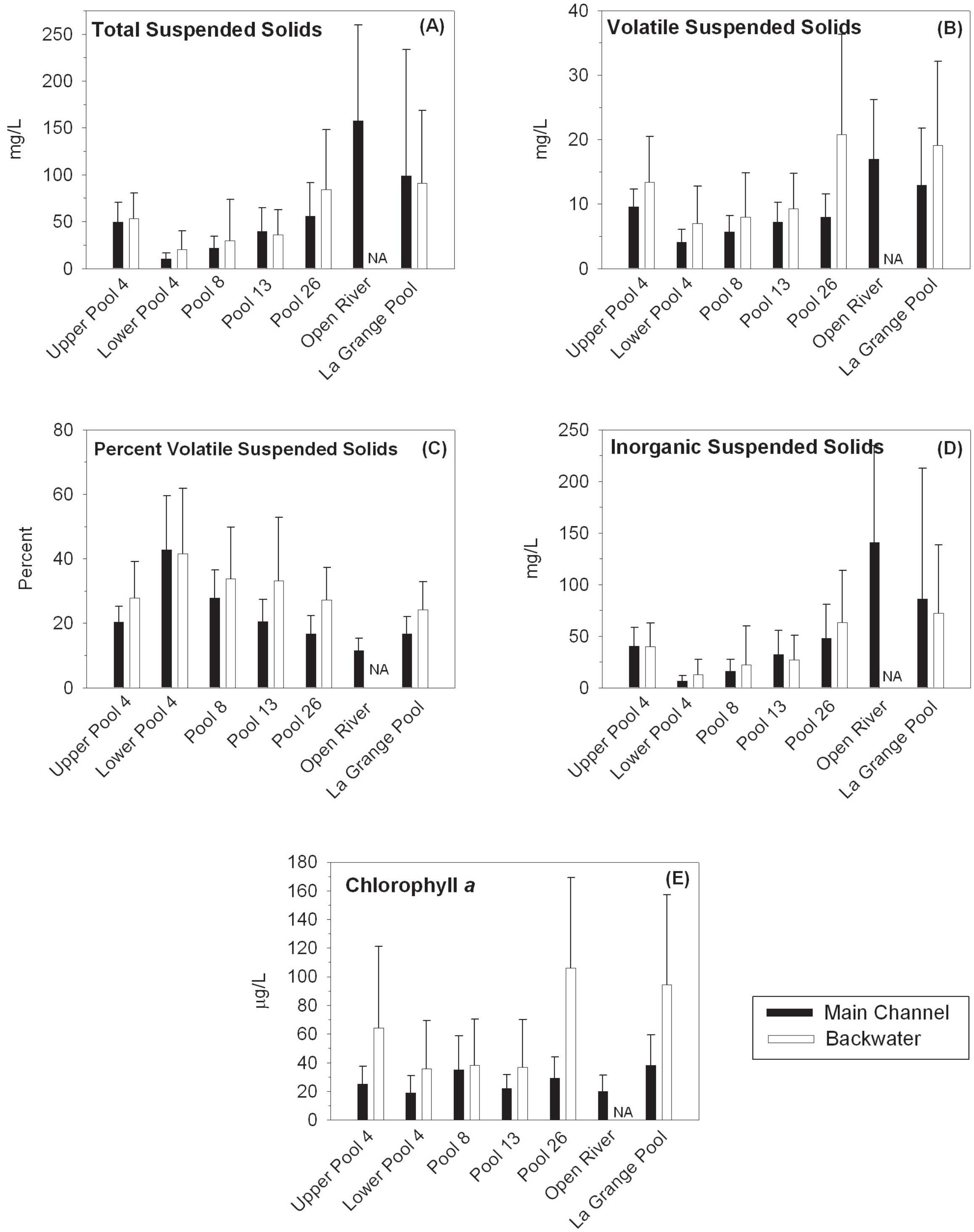

Figure 3.1. (*A*) Mean total suspended solids (mg/L), (*B*) volatile suspended solids (mg/L), (*C*) percent volatile suspended solids, (*D*) inorganic suspended solids (mg/L), and (*E*) chlorophyll *a* (µg/L) in the main channel and backwaters in the Long Term Resource Monitoring Program (LTRMP) study reaches during summer stratified random sampling. Means include all years from 1993 through 2001. Error bars represent +/- one standard deviation.

Longitudinal and Seasonal Patterns in Turbidity and Chlorophyll *a*

Seasonal and spatial differences in turbidity exist among the backwaters and main channels of the LTRMP study reaches (Figure 3.2). Because turbidity is largely determined by TSS, the variation among study reaches in turbidity was similar to that of TSS. There was a marked decrease in turbidity between upper and lower Pool 4; lower Pools 4 and 8 were similar; and turbidity increased in Pools 13 and 26. These longitudinal patterns are similar for the main channel and backwaters, although turbidity is generally higher in backwaters (Figure 3.2).

In the main channel, distinct winter minima in turbidity were observed in the northern three study reaches (Pools 4, 8, and 13; Figure 3.2A). This is most likely caused by the minimal runoff and ice and snow cover in winter. The ice and snow cover reduces light availability for algal production and eliminate sediment resuspension by wind and waves. Winter turbidity minima were not as distinct in the main channel of Pool 26 and Open River where ice cover is uncommon. Minimum median turbidity for the main channel of Pool 26 and Open River occurred in winter, but unlike the northern three study reaches, the variability in winter turbidity remained high, and the difference between winter and the other seasons was not as distinct. In the main channel of La Grange Pool, which also experiences less ice cover than the northern three study reaches, similar turbidity is observed in all seasons. In Pool 26 and Open River, maximum main channel turbidity occurred in spring.

In backwaters, distinct winter minima were observed in all six study reaches (Figure 3.2B). This contrasts with the main channel turbidity that only exhibited distinct winter minima in the northern three study reaches. Thus, for Pool 26, Open River, and La Grange Pool, the winter minima were more pronounced, and variability was lower in the backwaters than in the main channel. Highest turbidity was observed in summer in all six study reaches. In upper Pool 4 backwaters, fall turbidity was similar to summer and notably higher than winter and spring. The high turbidity in summer may be caused by high algal biomass, wind-driven sediment resuspension, or fish activity.

Chlorophyll *a* exhibits a distinct winter minimum in the main channel and backwaters of all study reaches (Figure 3.3B). Fewer hours of daylight, lower temperatures, and the reduction of light intensity in the water column because of ice and snow cover (particularly in the upper study reaches), may contribute to this pattern. Median main channel chlorophyll *a* concentration is highest for Pools 4, 8, and 13 in spring (Figure 3.3A); for the remaining study reaches (Pool 26, Open River, and La Grange Pool), median chlorophyll *a* concentration is generally lower in spring than summer and fall, perhaps because of reduced light penetration caused by high turbidity (Figure 3.2). The main channel of Pool 26 is an exception to this pattern and has lower chlorophyll *a* concentrations in fall than in spring. In Pools 4, 8, and 13, backwater chlorophyll *a* concentration spans a similar range in summer, fall, and winter, but in Pool 26 and La Grange Pool there is a distinct summer maximum in chlorophyll *a* concentrations (Figure 3.3B).

Habitat and Management Implications

Suspended material in aquatic systems is a natural and important part of the ecosystem. Algae, for example, are an important source of food in the aquatic food web. However, excessive amounts of suspended material, whether organic or inorganic, can have deleterious effects on the biota of an aquatic system. Several studies have documented the negative effects of turbidity on prey capture by fishes (Gardner 1981; Vandenbyllaardt et al. 1991; Reid et al. 1999). Miner and Stein (1996) documented turbidity affecting the selection of habitat use by fish. Turbidity can affect the growth and productivity of aquatic macrophytes (Kimber et al. 1995; Korchgen et al. 1997; Doyle and Smart 2001) by reducing light penetration and may have contributed to a decline in submersed macrophytes on parts of the UMRS in the early 1990s (Rogers 1994). In shallow lakes, turbidity levels can determine whether a system is in a

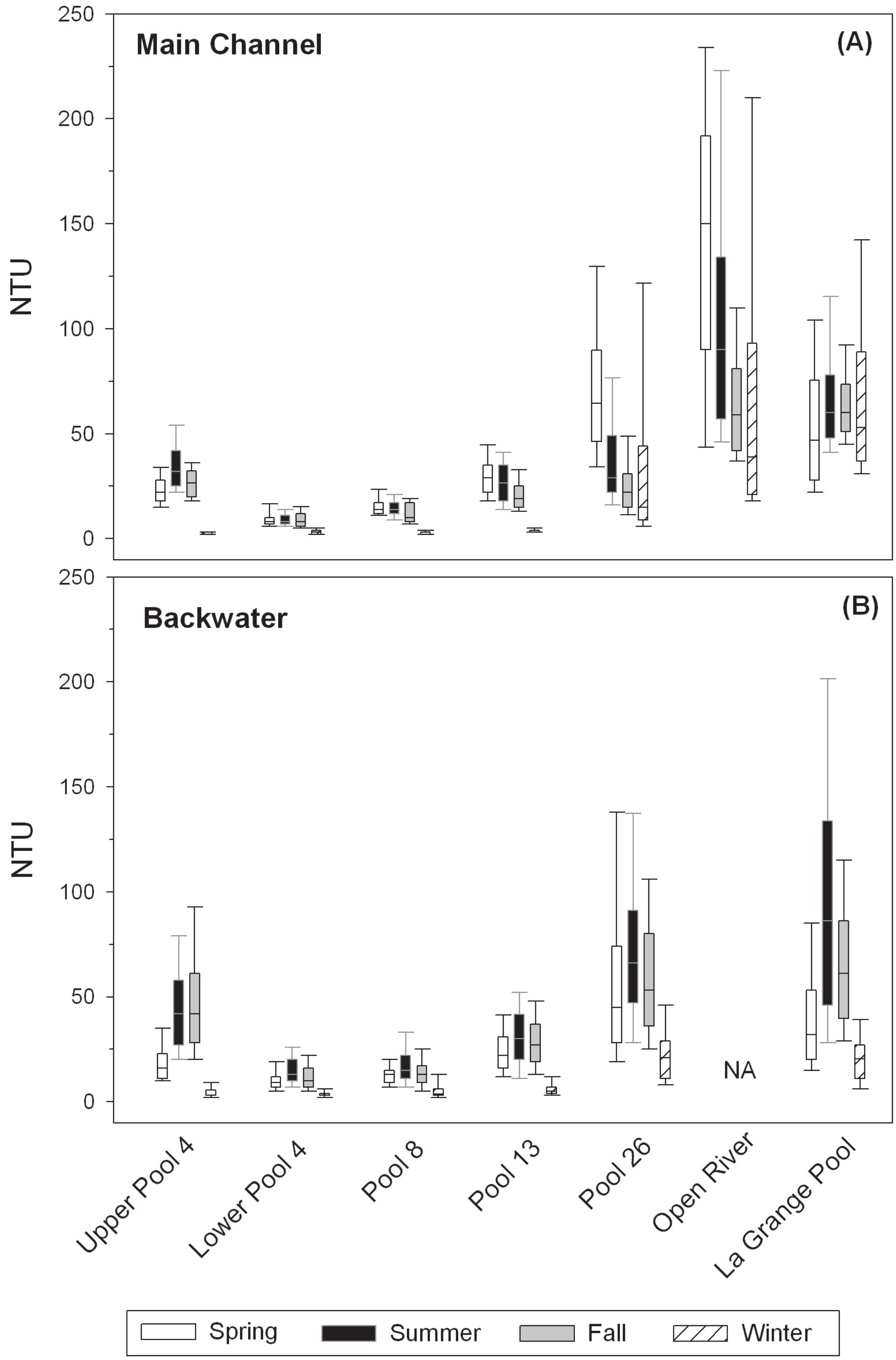

Figure 3.2. Box plots of winter, spring, summer, and fall turbidity (in nephelometric turbidity units [NTU]) in (*A*) main channel and (*B*) backwaters in the Long Term Resource Monitoring Program study reaches during stratified random sampling from 1993 through 2001. Box plots represent the 10th, 25th, 50th, 75th, and 90th percentiles.

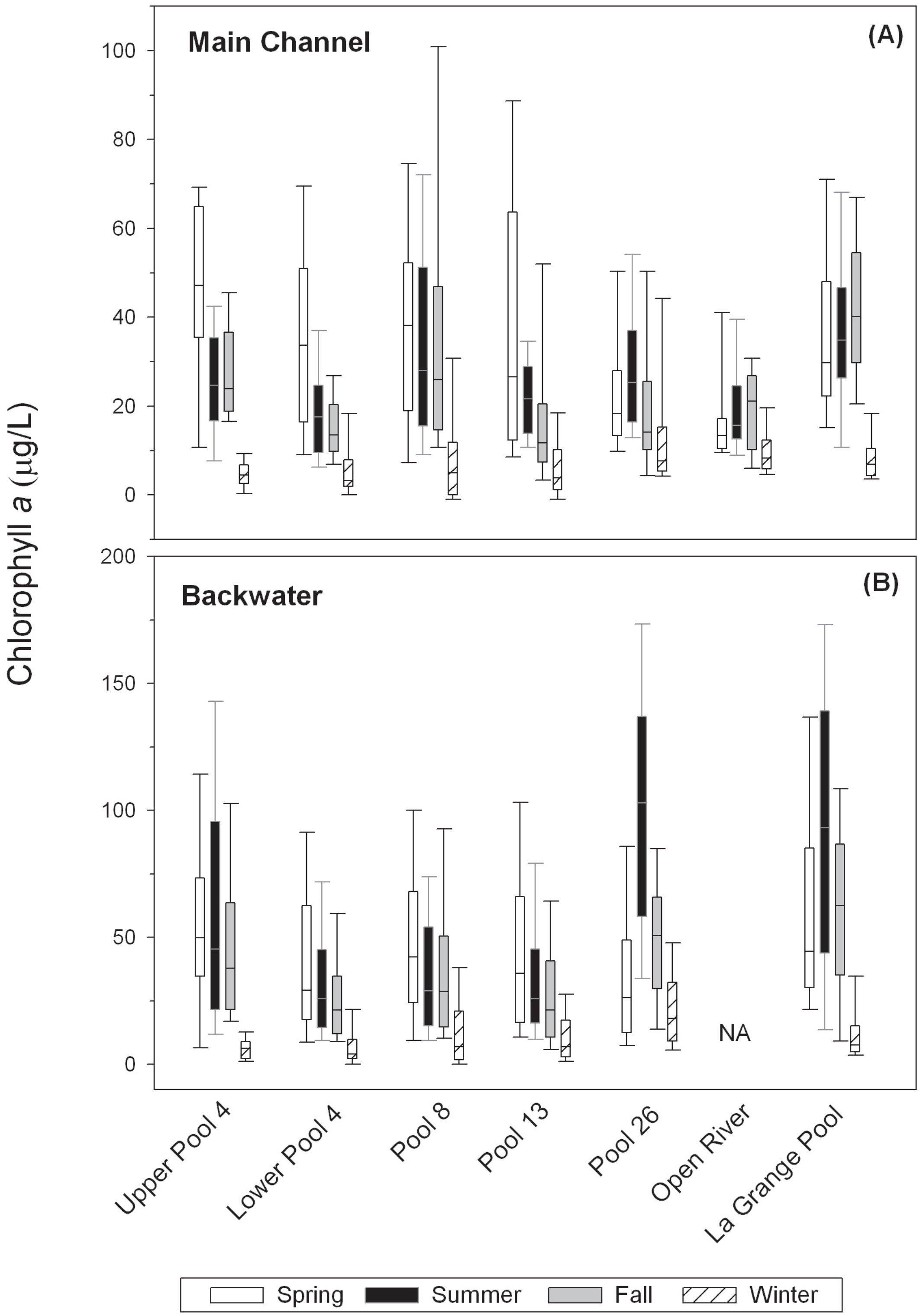

Figure 3.3. Box plots of winter, spring, summer and fall chlorophyll a concentrations (µg/L) in (A) main channel and (B) backwaters in the Long Term Resource Monitoring Program study reaches during stratified random sampling from 1993 through 2001. Box plots represent the 10th, 25th, 50th, 75th, and 90th percentiles.

clear state dominated by aquatic macrophytes or in a turbid state characterized by high algal biomass (Scheffer 1993). Suspended sediments can also negatively affect filter-feeding organisms, such as zooplankton (Arruda et al. 1983) and mussels (Aldridge et al. 1987).

The LTRMP data show that turbidity and suspended solids concentrations in the UMRS (Figures 3.1 and 3.2) are sometimes at levels that negatively affect aquatic organisms. Gardner (1981) found feeding rates for bluegills *Lepomis macrochirus* were reduced at 60 nephelometric turbidity units (NTU) and above; Reid et al. (1999) found lower rates of prey capture for largemouth bass *Micropterus salmoides* at 70 NTU; and Vandenbyllaardt et al. (1991) found walleye *Stizostedion vitreum* feeding was inhibited at 100 NTU. Doyle and Smart (2001) found that average turbidities of approximately 40 NTU decreased growth and survival of wild celery *Vallisneria americana* winterbuds and seedlings. Arruda et al. (1983) found suspended sediment concentrations of 50 to 100 mg/L reduced the amount of algal carbon ingested by *Daphnia* to potential starvation levels. The LTRMP data indicate these levels of suspended solids concentrations and turbidity are common in some parts of the UMRS. The mean TSS concentrations in upper Pool 4, Pool 26, Open River, and La Grange Pool (Figure 2.1A) indicate that these study reaches are chronically turbid. Although the turbidity and concentration of suspended solids are at levels to affect biota, the duration that these levels persist may be just as important. Newcombe and MacDonald (1991) reviewed more than 70 articles on the effects of inorganic suspended sediments on fish and invertebrates and found that suspended sediment concentration alone was a poor predictor of effects, but that concentration along with duration was a much better indicator. The timing of high turbidity events may also be significant to the growth and survival of organisms. Aquatic organisms may be exceptionally vulnerable to high turbidity during certain life-history stages. A high turbidity event in late fall, for example, may not have the same effect as an event in spring or summer when many organisms are in early life-history stages and when peak growth of most organisms occurs.

The discharge regime over the period of record was above average with no extended periods of low flow (Chapter 2). In general, inorganic material was the main source of turbidity, particularly in the more turbid reaches of the system. Under low discharge conditions, suspended solids consist of a higher proportion of volatile suspended solids because of an increase in algal biomass and a reduction of inorganic sediment load into the system. Many studies have shown the importance of discharge and water residence time in controlling algal biomass (Baker and Baker 1979; Soballe and Kimmel 1987; Heiskary and Markus 2001). As discharge decreases, water residence time increases, which allows algal biomass to accumulate. In the UMRS, significant negative correlations exist between discharge and chlorophyll *a* in the main channel area of upper Pool 4, Pools 8 and 13, and La Grange Pool and in the backwaters of upper Pool 4, Pool 8, and La Grange Pool (Figure 3.4).

Scheffer et al. (1993) suggested shallow lakes can have two alternative stable states: a clear state dominated by aquatic macrophytes and a turbid state characterized by high algal biomass. Although many mechanisms are probably involved, Scheffer et al. (1993) hypothesized that shallow lake systems are driven primarily by the interaction between submersed vegetation and turbidity. The mechanisms include macrophyte enhancement of water clarity and the inhibition of vegetation growth because of turbidity-induced light limitation. The LTRMP data suggest a turbidity threshold may have been reached for aquatic macrophytes in upper Pool 4, Pool 26, and La Grange Pool on the basis of high turbidities, high chlorophyll *a* concentrations, and a lack of submerged aquatic macrophytes in these study reaches (Rogers and Theiling 1999; Upper Mississippi River Conservation Committee 2003). Aquatic macrophytes declined rapidly in the late 1950s on the Illinois River (Sparks 1984) in large part because of an increased sediment load. Upper Pool 4 and upper Lake Pepin, which are presently turbid and nearly devoid of vegetation, had extensive macrophyte beds until the 1960s (Jack Enblom, Minnesota Department of Natural Resources, personal communication) indicating that the

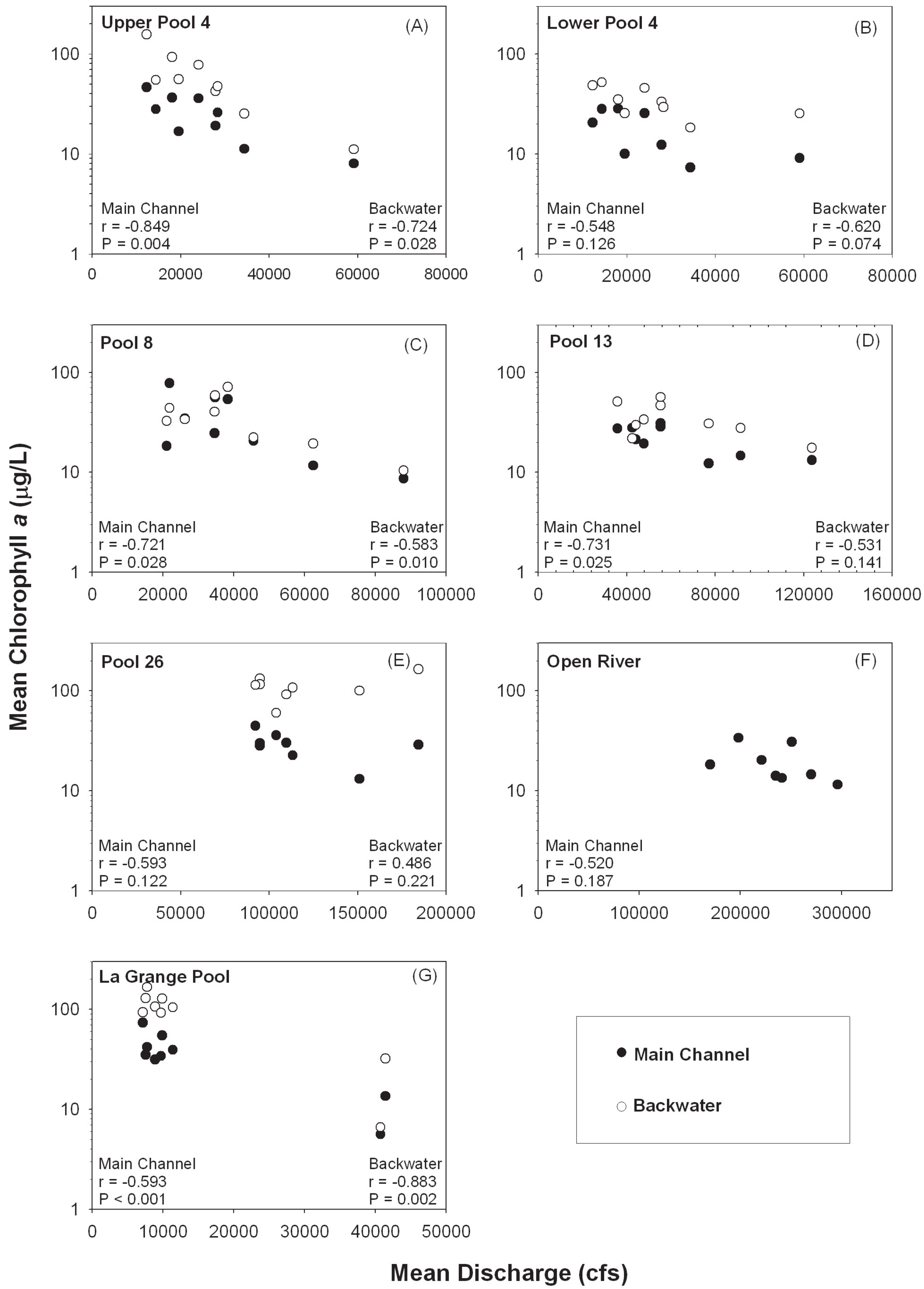

Figure 3.4. Correlation between mean discharge (cfs) and mean chlorophyll *a* (µg/L) in the main channel and backwaters in the Long Term Resource Monitoring Program study reaches ([*A*] upper Pool 4, [*B*] lower Pool 4, [*C*] Pool 8, [*D*] Pool 13, [*E*] Pool 26, [*F*] Open River, and [*G*] La Grange Pool) during summer stratified random sampling (1993–2001). No data are available from 1993 for Pool 26 and Open River. Open River has no backwater stratum.

system received perturbations creating conditions no longer conducive to the growth of submersed macrophytes. This points to the important role Lake Pepin plays in the UMRS and suggests that study reaches below Lake Pepin would be more turbid and have less vegetation without the extensive settling of sediment in the lake.

Chapter 4: Seasonal and Annual Variation in Nitrogen, Phosphorous, and Chlorophyll *a*

by

Lori A. Soeken-Gittinger and Jeffrey N. Houser

Background

Nitrogen and phosphorous are both critical plant nutrients and play an essential role in biochemical reactions and biological metabolism. The waters of the Upper Mississippi River System (UMRS) are eutrophic and nitrogen and phosphorous typically do not limit primary production (Patrick 1988), but these nutrients may be limiting at certain times and places. Chlorophyll *a* is an indicator of phytoplankton biomass and is one component of primary production in the UMRS. Phytoplankton is a food source for many organisms in the river, including mussels, invertebrates, and fish.The seasonal and annual cycles of nutrients and chlorophyll *a* can affect the biota of the river floodplain ecosystems including aquatic plants, macroinvertebrates, and fish. Understanding nutrient sources, transport, storage, cycling, and losses in the UMRS is essential to system management and maintenance.

Nutrients and chlorophyll *a* concentrations vary temporally (seasonally and annually) and spatially. Annual variations in nutrient concentrations are more pronounced in streams and rivers than in lakes (Horne and Goldman 1994). The causes of this variation are not well understood, and many factors influence year-to-year fluctuations such as climate, flood timing and duration, anthropogenic disturbance, and internal loading. A peak in nutrient concentrations is expected during the spring flood because of increased inputs from the floodplain and watershed, but might also occur in certain locations during periods of low flow if point source effluents do not decline in conjunction with declining water levels (Sparks 1984) or because of internal loading from the sediments.

The objectives of this chapter were to examine the seasonal and annual fluctuations of nutrients and chlorophyll *a* within the UMRS in main channel and contiguous backwater areas (hereafter referred to as backwaters). We focused on three nutrient and primary production constituents: soluble reactive phosphorous, nitrate + nitrite nitrogen, and chlorophyll *a* (Horne and Goldman 1994). Soluble reactive phosphorous (SRP) approximates the amount of phosphorous available for use by phytoplankton, bacteria, algae, and aquatic macrophytes (Lampert and Sommer 1997). Total oxidized nitrogen is the sum of nitrate and nitrite nitrogen (American Public Health Association 1992), referred to in this chapter as nitrate + nitrite nitrogen (NO_x). Nitrate is the most highly oxidized form of nitrogen and usually the most abundant form of inorganic nitrogen in aquatic systems, whereas nitrite is usually present only in small amounts (Horne and Goldman 1994).

Methods

The analyses in this chapter focus on NO_x, SRP, and chlorophyll *a* in the main channel and backwaters sampled during fixed-site and stratified random sampling (SRS) periods (Chapter 1). Only surface samples (about 20 cm below surface) were included in the analyses. Field and laboratory methods are described in Soballe and Fischer (2004).

Means and standard error were calculated in SAS (version 8.02). Fixed-site means were calculated by month, and SRS means were calculated by seasonal episode. The SRS data were collected from 1993 to 2002. However, SRS sampling did not begin until summer 1993, so there are no SRS data for spring 1993. Fixed-site data included in this chapter span from 1993 to 2002.

Annual Patterns

Patterns in nutrient concentrations in spring and summer are of particular interest for the UMRS. Highest flows generally occur in spring (Figure 2.1) resulting in high rates of nutrient export. Summer represents the growing season when the interactions of nutrient concentrations are most important. In the main channel and backwaters of the Upper Mississippi and Illinois Rivers, spring SRP concentrations were similar among all study reaches (Figure 4.1), suggesting that differences in local climate and land use had minimal effects on dissolved phosphorous concentrations in spring. La Grange Pool showed much higher SRP concentration in 1996 than in any other years or study reaches; the cause of high SRP concentrations in this year is unknown.

Spring NO_x concentrations in the main channel and backwaters of the Mississippi River showed a strong covariance among study reaches (Figure 4.1) probably driven by annual differences in discharge. Years of low spring NO_x concentrations were years of relatively low discharge (Chapter 2). Nitrate concentrations in streams and rivers have been related to rainfall, with nitrogen increasing in proportion to discharge (Horne and Goldman 1994). Reduced precipitation can result in less export of nitrogen from the surrounding agricultural land into the river.

La Grange Pool of the Illinois River consistently showed substantially higher NO_x concentrations when compared to Mississippi River study reaches. High inputs of nitrogen to the Illinois River were not surprising because the Illinois River flows through the portion of the UMRS watershed with the highest estimated nitrogen fertilizer use (Gowda 1999). In addition, the Illinois River had a much lower discharge than the Mississippi River, which resulted in less dilution of these terrestrial NO_x inputs.

Spring means of chlorophyll *a* did not exhibit the strong covariance among Long Term Resource Monitoring Program study reaches that was seen for NO_x (Figure 4.1). This lack of covariance suggests local factors, such as aquatic vegetation abundance, climate, watershed conditions, or local nutrient inputs affected year-to-year variation and systemic patterns in chlorophyll *a*. In spring 2000, most study reaches exhibited similar and high chlorophyll *a* concentrations. This was probably because of low discharge throughout the system that spring (Figure 2.2) that increased hydraulic retention time and reduced turbidity from inorganic suspended solids.

Summer nutrient concentrations were lower than spring concentrations, particularly for NO_x (Figure 4.2). In contrast, maximum chlorophyll *a* concentrations were higher in summer than in spring, particularly for the backwaters of La Grange Pool of the Illinois River and Pool 26.

Summer SRP concentrations in the main channel of the Upper Mississippi River were similar among all study reaches, but, as was seen for NO_x concentrations, the main channel of La Grange Pool (Illinois River) exhibited higher concentrations of SRP than any of the Mississippi River study reaches (Figure 4.2). In some years (1997, 1999, 2001), summer SRP concentrations in backwaters were significantly greater in Pool 13 and La Grange Pool than in other study reaches (Figure 4.2). The cause of this high variation is unknown, but these differences did not appear to correspond to high or low discharge (Chapter 2).

Summer NO_x in the Illinois and Mississippi Rivers showed no clear differences among study reaches in the main channel and backwaters (Figure 4.2). The NO_x concentration was generally lower in the backwaters than in the main channel. This is probably because of higher rates of denitrification (Richardson et al. 2004) and primary production in the backwaters and limited delivery of NO_x to backwaters during low summer flows.

Summer chlorophyll *a* concentrations in the main channel of Pool 8 and the Illinois River were significantly higher in certain years than in other study reaches. The Illinois River also had higher concentrations of SRP than other study reaches, and this nutrient availability may have contributed to higher chlorophyll *a* concentrations. The reasons for higher

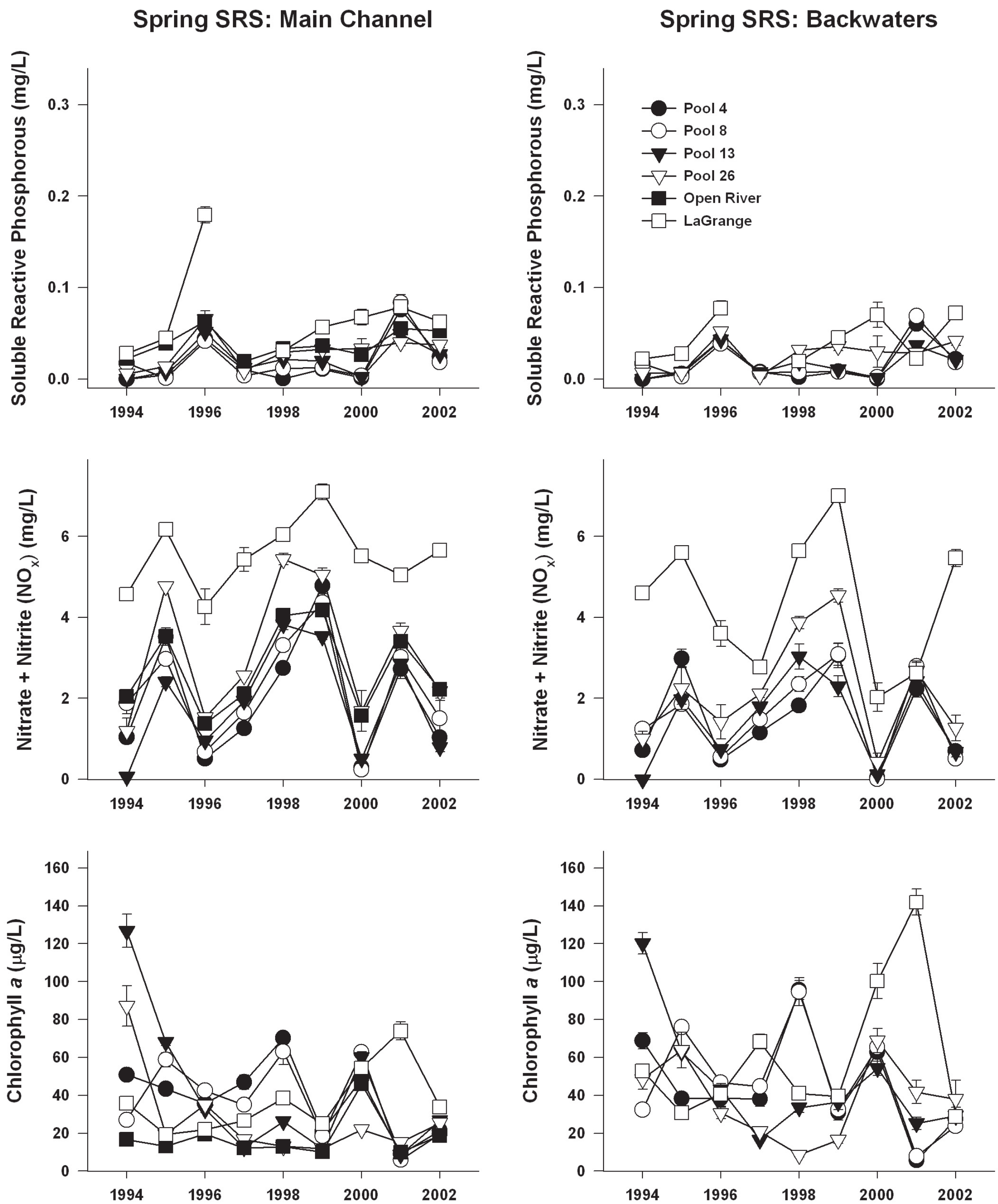

Figure 4.1. Annual means of soluble reactive phosphorous (mg/L), nitrate + nitrite oxygen (mg/L; NO_x), and chlorophyll *a* (µg/L) in the main channel and backwaters of the Upper Mississippi and Illinois Rivers based on spring stratified random sampling (SRS) from 1994 to 2002. Error bars represent +/- one standard error.

chlorophyll *a* in Pool 8 are unknown. Backwaters of the Illinois River and Pool 26 exhibited much higher summer chlorophyll *a* concentrations compared to other study reaches (Figure 4.2). This may be because the backwaters of these two study reaches had little aquatic vegetation to compete with phytoplankton for nutrients. These backwaters were also relatively shallow,

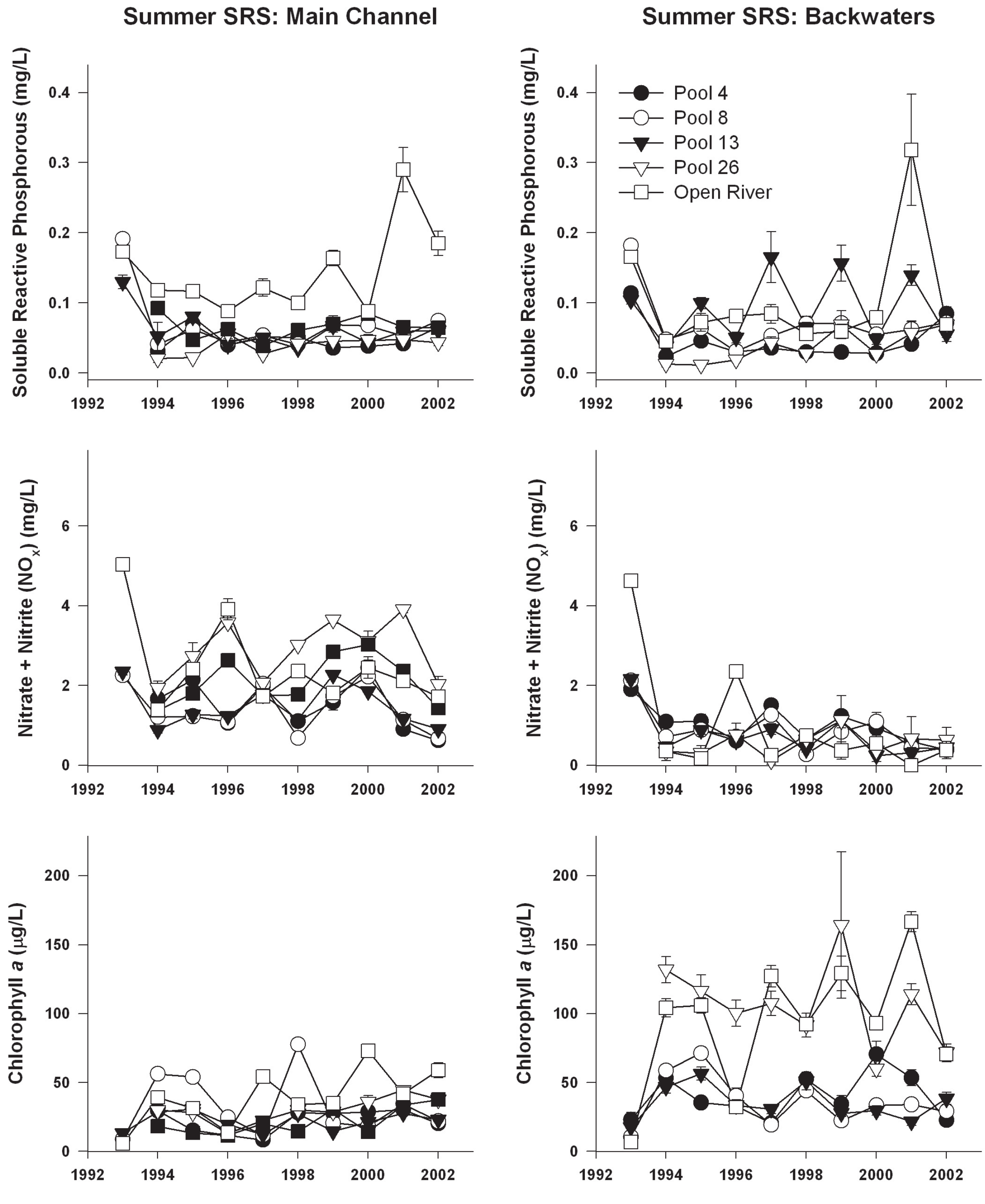

Figure 4.2. Annual means of soluble reactive phosphorous (mg/L), nitrate + nitrite nitrogen (mg/L; NO_x), and chlorophyll *a* (µg/L) in the main channel and backwaters of the Upper Mississippi and Illinois Rivers based on summer stratified random sampling (SRS) from 1993 to 2002. Error bars represent +/- one standard error.

allowing for frequent resuspension of sediment and nutrients. The suspended sediments reduce the depth of light penetration, making it more difficult for aquatic vegetation to become established, and the resuspended nutrients promote algal production.

Spring and summer SRS data clearly showed differences among the Illinois River and the

Mississippi River study reaches. In spring of most years, both backwaters and the main channel of the Illinois River had higher NO_x and SRP concentrations (Figure 4.1), and in summer the main channel of the Illinois River exhibited noticeably higher SRP concentrations than Mississippi River study reaches (Figure 4.2). In most years, the backwaters of the Illinois River had higher summer chlorophyll *a* concentrations than all of the Mississippi study reaches except for Pool 26 (Figure 4.2). The Illinois River flows into the Mississippi River at Pool 26, and in spring and summer the main channel in Pool 26 often had the highest NO_x concentrations of the Mississippi River study reaches. The relative contributions of nutrients and chlorophyll *a* from the Illinois River and from the many agriculture-dominated catchments that enter the river between Pools 13 and 26 to the high chlorophyll *a* concentrations in Pool 26 remain unclear.

The UMR Basin accounts for 31% of the total nitrogen delivered from the Mississippi River to the Gulf of Mexico (Alexander et al. 1995) and, as stated previously, the Illinois River flows through the portion of the UMR Basin with the highest estimated nitrogen fertilizer use (Gowda 1999). The Mississippi River has a greater discharge and dilution capacity than the Illinois River (Sparks 1984); thus, higher fertilizer use combined with less dilution most likely accounted for the higher NO_x concentrations in the Illinois River in spring. It is also probable that the high NO_x concentrations in the Illinois River were influenced by loadings from sewage treatment plants in Chicago, Illinois; the Upper Illinois River exhibited high ammonia concentrations that declined downstream as the ammonia was converted to nitrate (Sparks 1984). Fertilizer use, along with industrial and domestic wastes, may also have accounted for higher SRP concentrations in summer as well (Starrett 1972).

Seasonal Patterns

Soluble reactive phosphorous concentrations in the main channel of the Upper Mississippi and Illinois Rivers tended to peak in September and exhibited minimum concentrations in April and May (Figure 4.3). This is contrary to the conventional idea that nutrient concentrations in large rivers often peak in late spring or early summer in conjunction with spring flooding (Horne and Goldman 1994). This trend was also seen in backwaters with large SRP peaks in summer or early fall and minimum concentrations in April and May (Figure 4.3).

Seasonal NO_x concentrations in the main channel and backwaters indicated a pattern nearly opposite of that shown by seasonal SRP patterns, with the lowest concentrations of NO_x in fall, when SRP concentrations are at their highest (Figure 4.4). As the lows in nitrogen occurred directly following peaks in chlorophyll *a* (Figures 4.4–4.5), it was possible that the NO_x concentrations were reduced by algal assimilation and aquatic macrophytes. High rates of denitrification in summer, particularly in backwaters (Richardson et al. 2004), probably also contributed to the low NO_x concentrations observed in late summer. A similar seasonal distribution of nitrogen was found in many eutrophic temperate lakes, where the fall NO_x minima increase to maxima in late winter through summer (Wetzel 1983). This raises the possibility that nitrogen was limiting algal growth in late summer, especially during low flow periods, possibly explaining the higher SRP concentrations found during this time.

Chlorophyll *a* concentrations in main channel and backwaters showed peaks in late summer and fall, with minima in winter and early summer (Figure 4.5). The one exception was the backwaters of La Grange Pool where chlorophyll *a* concentrations remained elevated from April through September (Figure 4.5). These peak chlorophyll *a* concentrations in early spring and or late summer, along with the minima in SRP in this same period (Figure 4.3), suggest that the low SRP concentrations may have been because of algal uptake of SRP.

The fixed-site and SRS data generally show good agreement and similar annual patterns. There are several notable exceptions. The most consistent difference is seen in Pool 26 where nutrient and chlorophyll *a* concentrations measured at fixed sites are generally substantially lower than the SRS means. In Pool 13 backwaters, NO_x concentration measured at

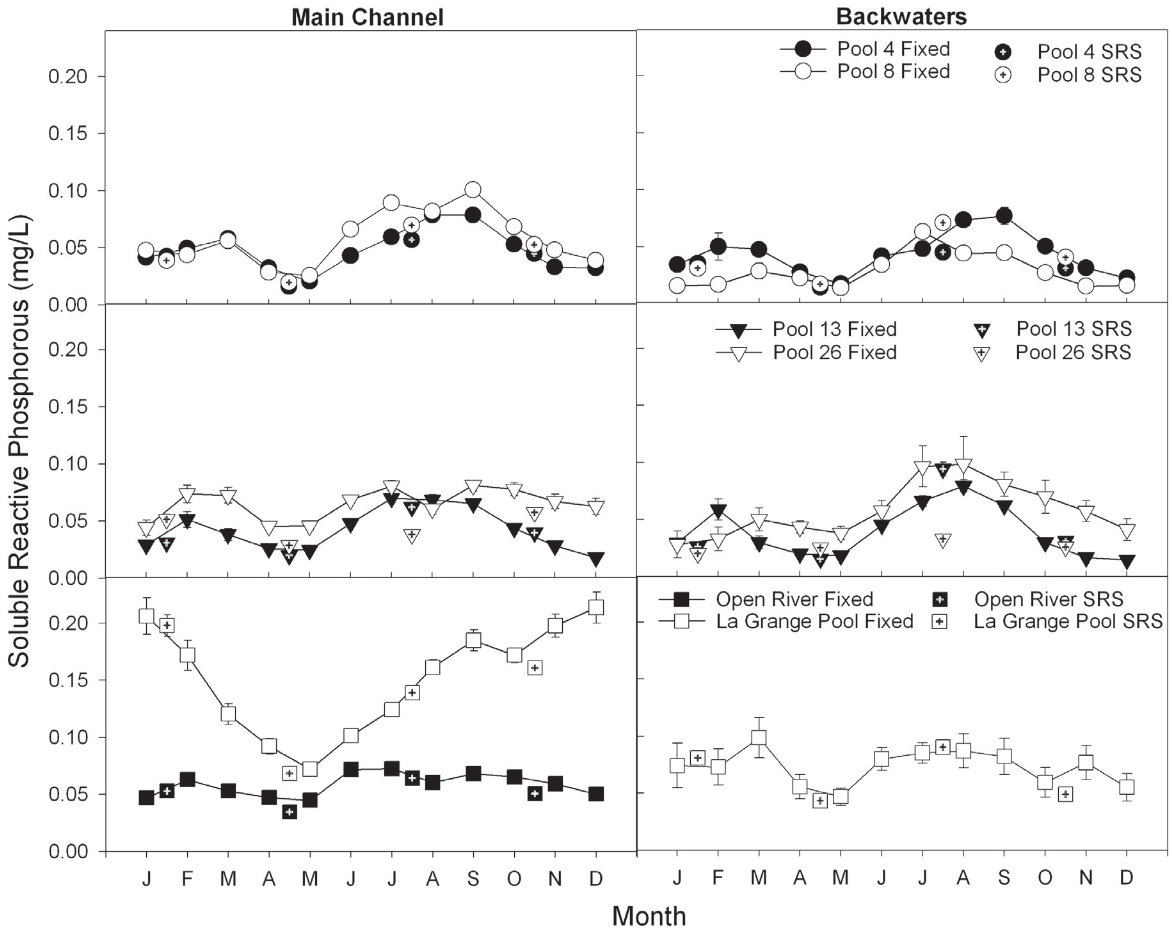

Figure 4.3. Monthly mean (1993–2001) soluble reactive phosphorous (mg/L) concentrations in the main channel and backwaters of the Upper Mississippi and Illinois Rivers. Monthly data are from fixed-site sampling. Stratified random sampling (SRS) points are plotted quarterly between the 2 months that the 2-week SRS episode spans. Error bars represent +/- one standard error.

the fixed sites is generally lower than the SRS mean, particularly in winter. In La Grange Pool backwaters, the fixed-site fall mean NO_x concentration is lower than the SRS mean. In Pool 8 backwaters, fixed-site fall chlorophyll *a* concentrations are higher than the fall SRS mean.

The fixed-site and SRS regimes were designed for different purposes. The SRS design provides unbiased estimates for each study reach and strata, but is limited in its temporal coverage. The fixed-site data provide information at a higher temporal resolution, which illustrates seasonal patterns in nutrients and chlorophyll *a*. Because of the limited spatial extent of the fixed-site sampling and subjective-site selection, these data do not provide unbiased means for study reaches or strata.

Conclusions

The seasonal and annual patterns addressed in this chapter were similar among study reaches for some parameters. There were notable differences in nutrients in Pool 26 and La Grange Pool, and in chlorophyll *a* in Pools 8 and 26 and La Grange Pool. Spring NO_x concentrations were highly variable among years in both the main channel and backwaters, but these annual differences were similar among study reaches. These variations were probably attributable to

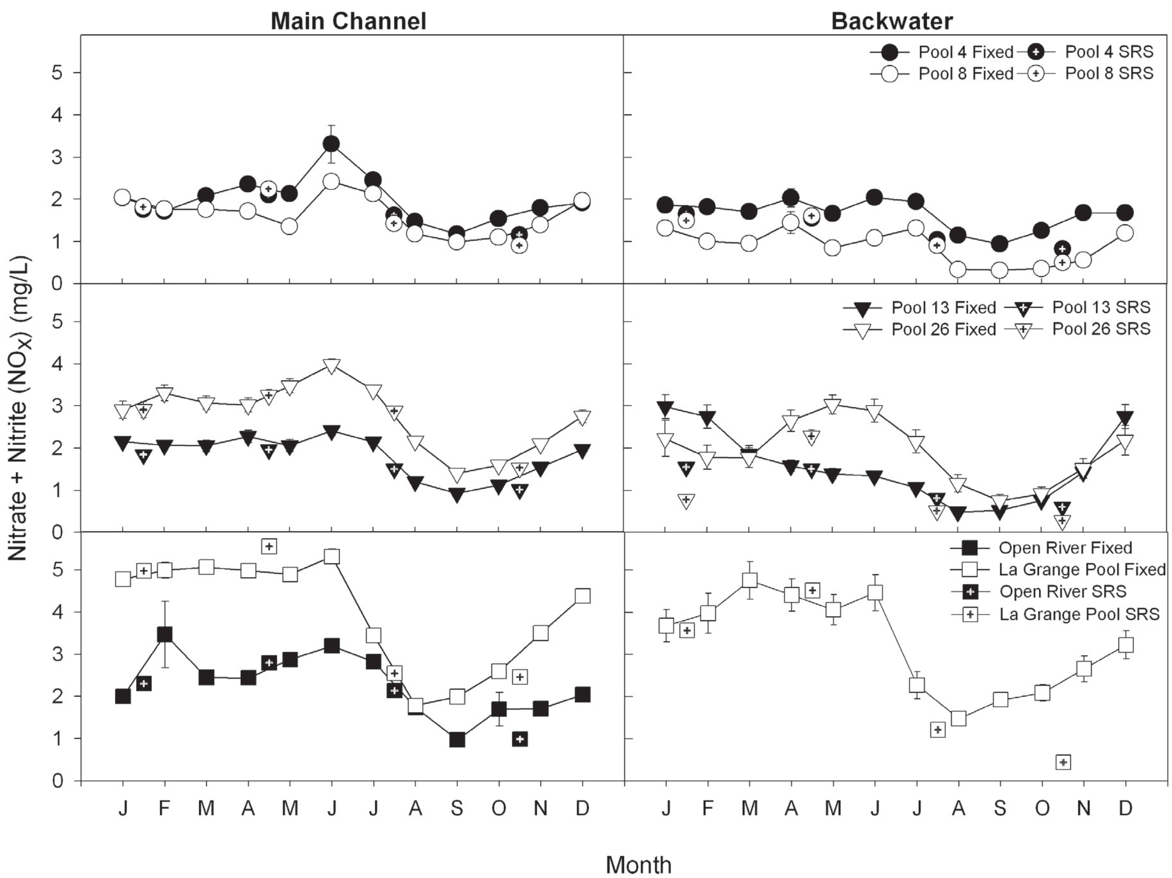

Figure 4.4. Monthly mean (1993–2001) nitrate + nitrite (mg/L; NO_x) concentrations in the main channel and backwaters of the Upper Mississippi and Illinois Rivers. Monthly data are from fixed-site sampling. Stratified random sampling (SRS) points are plotted as in Figure 4.3. Error bars represent +/- one standard error.

differences in rainfall and river discharge among years.

There was a substantial difference in nutrient and chlorophyll *a* concentrations in La Grange Pool of the Illinois River compared to study reaches of the Mississippi River in spring and summer. These differences were possibly related to the watershed and land-use differences, particularly the high use of nitrogen fertilizer in the Illinois River Basin. There were also differences in the hydrologic regimes (particularly residence time) of the Illinois and Mississippi Rivers (and among study reaches within the Mississippi River). Further analysis of the effects of hydraulic residence time on nutrient concentrations would be worthwhile.

Contrasting seasonal patterns in nitrogen and phosphorous were observed. Evidence suggested that algal production was not limited by SRP in summer, particularly in June and at the end of summer in August and September. High concentrations of SRP in association with low concentrations of NO_x and chlorophyll *a* in summer suggest algal growth may have been limited by something other than SRP (e.g., NO_x, light, etc.) during these times.

A number of additional analyses would be useful in understanding chlorophyll *a* (and therefore phytoplankton) dynamics. Questions that remain include the following: Is there a consistent relationship between nutrient and chlorophyll *a* concentrations? Is this relationship different in the main channel and backwaters or among study reaches? What is the role of resuspension of sediments in determining chlorophyll *a* concentrations? The resulting

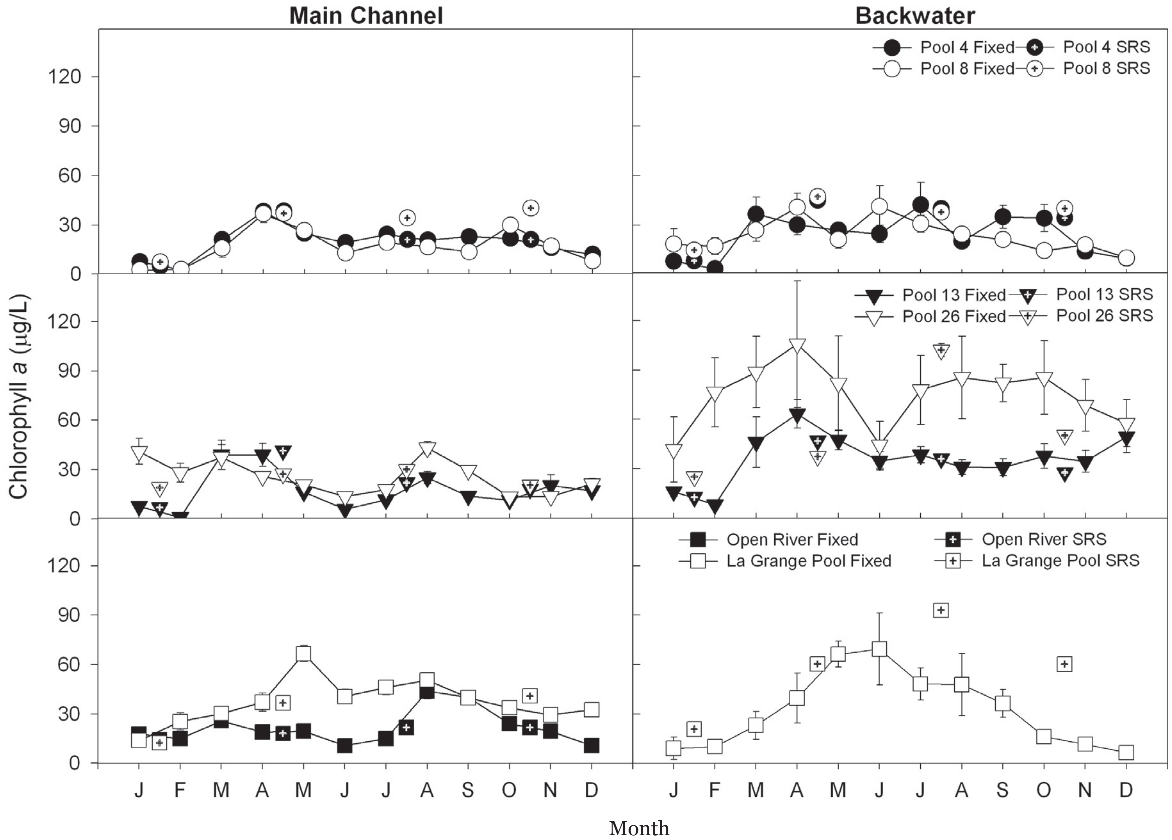

Figure 4.5. Monthly mean (1993–2001) chlorophyll *a* concentrations (µg/L) in the main channel and backwaters of the Upper

as in Figure 4.3. Error bars represent +/- one standard error.

increased turbidity will reduce light availability (potential negative effect on chlorophyll *a*), but the resuspension may increase nutrient inputs to the water column (potential positive effect on chlorophyll *a*). Studies designed to specifically address such questions combined with additional analysis of the Long Term Resource Monitoring Program data will improve our understanding of chlorophyll *a* and nutrient dynamics and increase our ability to effectively manage the UMRS.

Chapter 5: Spatial and Temporal Variation of Dissolved Oxygen

by

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Background

Dissolved oxygen (DO) is an important factor in determining habitat suitability and can affect physicochemical processes within the water column and underlying sediments. The DO concentrations in the Upper Mississippi River (UMR) are affected by temperature, current velocity, and biological processes that produce oxygen through photosynthesis and consume oxygen through respiration. Variations in these factors can produce large spatial and temporal variation in DO concentrations. Because low DO concentration is a critical factor in determining habitat suitability for fish and invertebrates, most states along the river recognize a criterion of 5 mg/L DO as the concentration necessary to protect fish and other aquatic organisms (e.g., Wisconsin Administrative Code Department of Natural Resources Chapter NR 102 [1998]).

Municipal sewage disposal has historically been an important point source of oxygen-demanding materials to the UMR and a major contributor to oxygen depletion in many parts of the river (Wiebe 1927; Fremling 1964). Before changes made in response to the Clean Water Act (1977), oxygen concentrations in some parts of the river were low enough that sensitive invertebrate species (e.g., mayflies such as *Hexagenia* spp.) disappeared. Improvement in municipal sewage treatment has reduced the effect of sewage discharge on the river and sensitive species have returned to many areas where they were historically present (Fremling 1989).

In addition to point sources, oxygen demand also results from nonpoint terrestrial and aquatic sources. The Minnesota River, for example, is a major source of organic matter to Pool 2 on the UMR. Microbial decomposition of such organic matter is an important source of oxygen demand. Zebra mussels, *Dreissena polymorpha*, first documented in the system in 1991 (Cope et al. 1997), may contribute to oxygen demand through excretion and respiration at a rate of about 15 to 20 mg O_2 m^{-2} d^{-1} (Sullivan and Endris 1998). High zebra mussel densities have depleted oxygen concentrations in the Seneca River, New York (Effler and Siegfried 1994), and in some reaches of the UMR.

An evaluation of DO dynamics in space and time is necessary to develop potential management strategies for the UMR. This chapter provides a broad overview of the oxygen dynamics within study reaches of the Long Term Resource Monitoring Program (LTRMP) on the Upper Mississippi River System (UMRS) and illustrates some of the spatial and temporal variability of DO in this complex system. This initial review establishes a baseline summary and identifies areas of interest for additional analyses. Specifically this chapter addresses (1) differences among river reaches and among years, (2) seasonal patterns, and (3) the occurrence of low DO concentrations in winter and summer.

Methods

The following variables from LTRMP stratified random sampling (SRS) data were used in our analysis: surface dissolved oxygen concentrations, percent dissolved oxygen saturation, and snow and ice thickness. Surface (0.2 m) dissolved oxygen was measured *in situ* with a multiparameter sonde (Hydrolab Datasonde 3® or Minisonde 4a®; Hach Company, Loveland, CO) equipped with a polarographic type membrance electrode (Soballe and Fischer 2004). A small number of data points (<1%) were DO concentrations denoted as “off scale high” (Quality factor code = 2; Soballe and Fischer 2004), and these data points were omitted from the figures presented in this chapter. Data from 1993 to 2002 for all study reaches and the following strata were included unless otherwise specified: contiguous backwaters, impoundment, side channel, main channel, and Lake Pepin (Pool 4) and Swan Lake (Pool 26). The SRS sampling began in summer 1993 for Pools 4, 8, and 13 and La Grange Pool and began in fall 1993 for Pool 26 and Open River.

We determined the area of low DO (<5 mg L) in contiguous backwater areas (hereafter referred to as backwaters) by assuming that surface DO measured at each SRS site characterized the grid cell it represented and summing the area of all grid cells with low DO. Grid cells were 200-m squares for all strata in 1993 to 1995. In 1995, the grid cells in backwaters and side channels were reduced to 50-m squares. For winter, sites that were frozen to the bottom were included in the total area of low DO because these frozen areas were also unsuitable habitat for most organisms of interest.

Results and Discussion

Seasonal Means by Study Reaches

In spring, median DO in Pools 4, 8, and 13 were higher (10 to 15 mg/L) than in Open River (7 to 10 mg/L; Figure 5.1). Concentrations in Pool 26 and La Grange Pool were intermediate. Virtually, no DO concentration <5 mg/L were observed in spring.

In summer, median DO concentrations in Open River were lower than in northern reaches (Pools 4, 8, and 13), but no obvious differences existed among the three northern study reaches and Pool 26 (Figure 5.2). Median concentrations were between 5 and 10 mg/L across all study reaches. All reaches occasionally had low summer DO (<5 mg/L). Pools 8 and 13 and La Grange Pool showed the highest frequency of low summer DO conditions (Figure 5.2) and Open River had the fewest occurrences. The northern study reaches (Pools 4, 8, and 13) have more backwater areas than the southern study reaches (Pool 26, La Grange Pool, and Open River [which has no backwaters]); and, in summer, backwaters were characterized by relatively long residence times that facilitate DO depletion. It is unknown why Pool 4 experienced less frequent low DO conditions than the other northern study reaches.

In fall, DO patterns were similar to summer, although the occurrence of low DO concentrations was less frequent (Figure 5.3). In winter, distinct north–south differences were apparent (Figure 5.4). Median DO concentrations were generally between 10 and 16 mg/L in all study reaches. However, DO concentrations <5 mg L^{-1} occurred regularly in the northern three study reaches, but rarely occurred in Pool 26 and La Grange Pool and never occurred in Open River. The northern study reaches showed a large decrease in median DO concentration between 1995 and 1996 and a general increase from 1997 through 2002 (Figure 5.4).

Summary of Seasonal Patterns

In the northern study reaches, median DO concentrations are higher in spring and winter, and lower in summer and fall. In the southern study reaches, DO concentrations are highest in winter, lowest in summer, and intermediate in spring and fall. Important drivers of these seasonal patterns in DO are water temperature, current velocity (a function of discharge), and ice and snow cover. The solubility of oxygen in water is negatively correlated with water temperature. Current velocity determines the extent of vertical mixing and turbulence, which affects the rate of oxygen exchange with the atmosphere, and residence time, which affects oxygen depletion rates. Ice prevents gas exchange with the atmosphere. However, if the ice is clear, high rates of photosynthesis can occur resulting in high DO concentrations. If significant snow cover reduces light penetration, respiration will be the dominant biological process resulting in low DO concentrations. The effects of snow and ice cover can be clearly seen for the UMR. Total snow and ice thickness explain about half of the variability among study reaches in mean winter backwater DO (Figure 5.5).

For the northern study reaches, spring is a time of low water temperature, high discharge, and little ice cover resulting in relatively high DO concentrations. In winter, low discharge and substantial ice and snow cover, particularly in backwaters, result in lower DO concentrations relative to spring and low DO concentrations occur regularly in backwaters. In summer, low

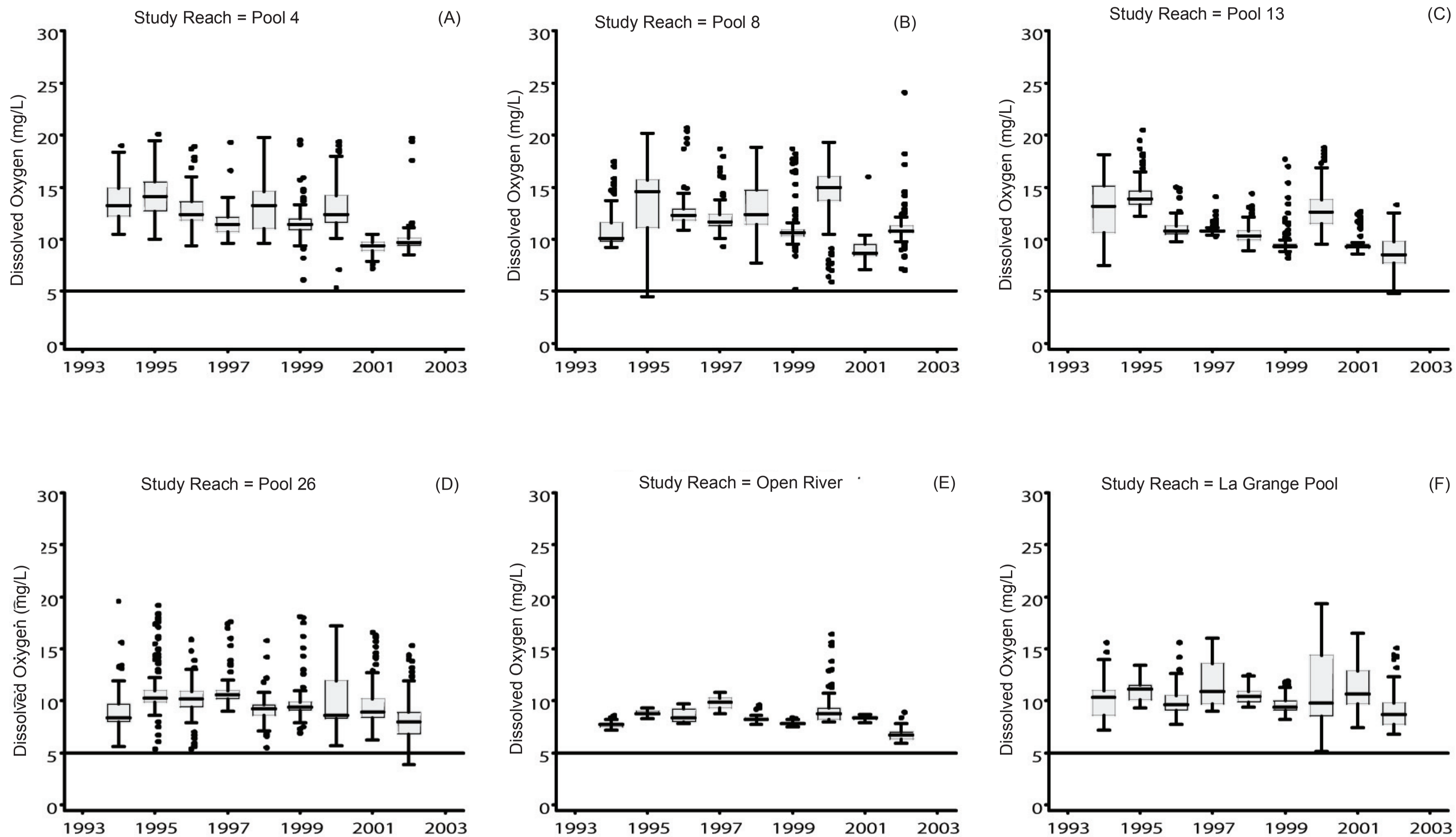

Figure 5.1. Spring dissolved oxygen (DO) in all strata of the six study reaches based on the Long Term Resource Monitoring Program stratified random sampling data, 1994–2002. Boxes indicate the 25th–75th percentile, the horizontal line within the box represents the median, whiskers extend to the largest and or smallest value within 1.5 × the interquartile range (75th–25th percentile), and points represent observations outside this range. The solid line at DO = 5 mg/L indicates DO concentration generally recognized as necessary to protect most aquatic organisms.

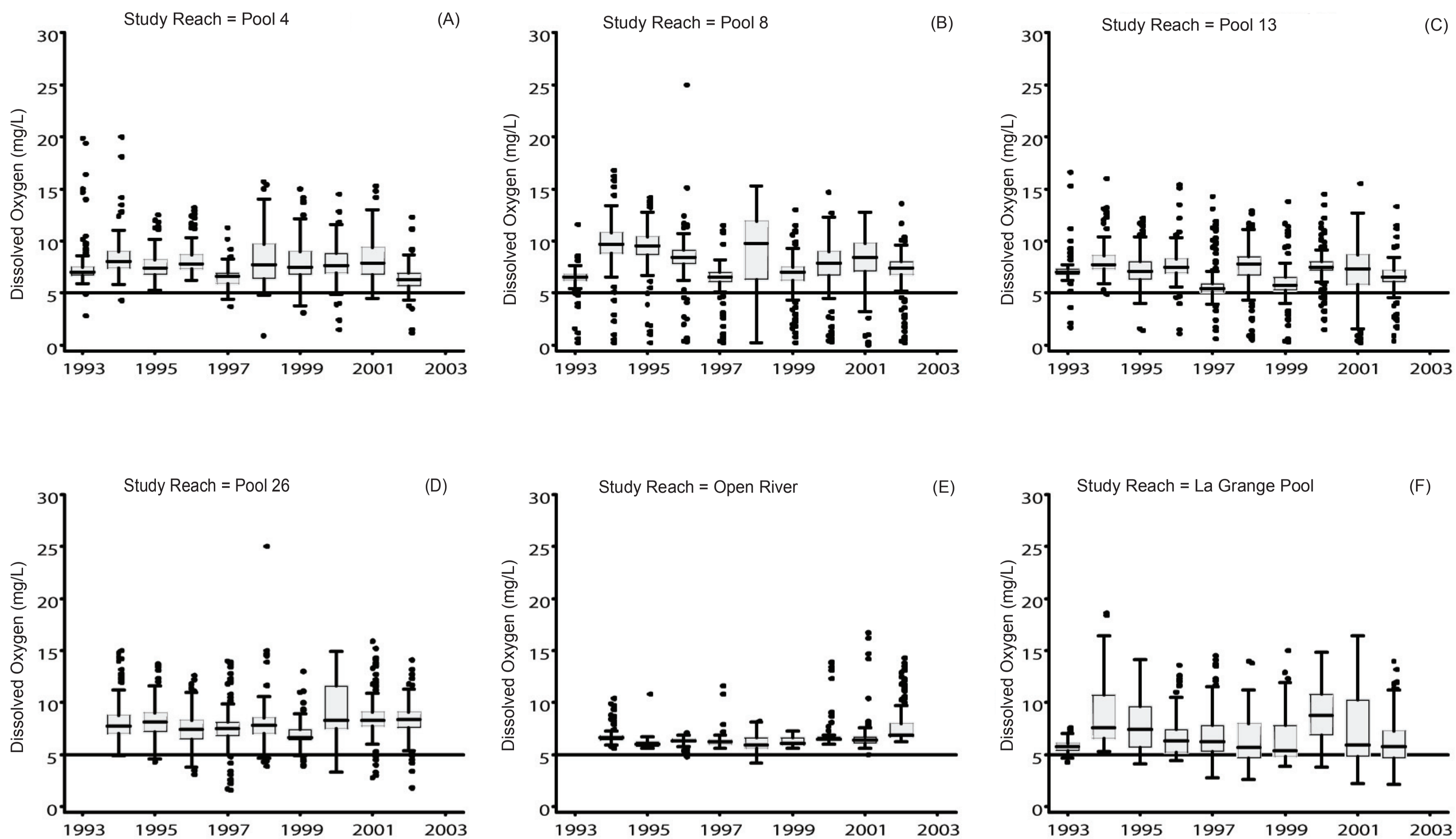

Figure 5.2. Summer dissolved oxygen (DO) in all strata of the six study reaches based on the Long Term Resource Monitoring Program stratified random sampling data, 1993–2002. Boxes indicate the 25th–75th percentile, the horizontal line within the box represents the median, whiskers extend to the largest and or smallest value within 1.5 × the interquartile range (75th–25th percentile), and points represent observations outside this range. The solid line at DO = 5 mg/L indicates DO concentration generally recognized as necessary to protect most aquatic organisms.

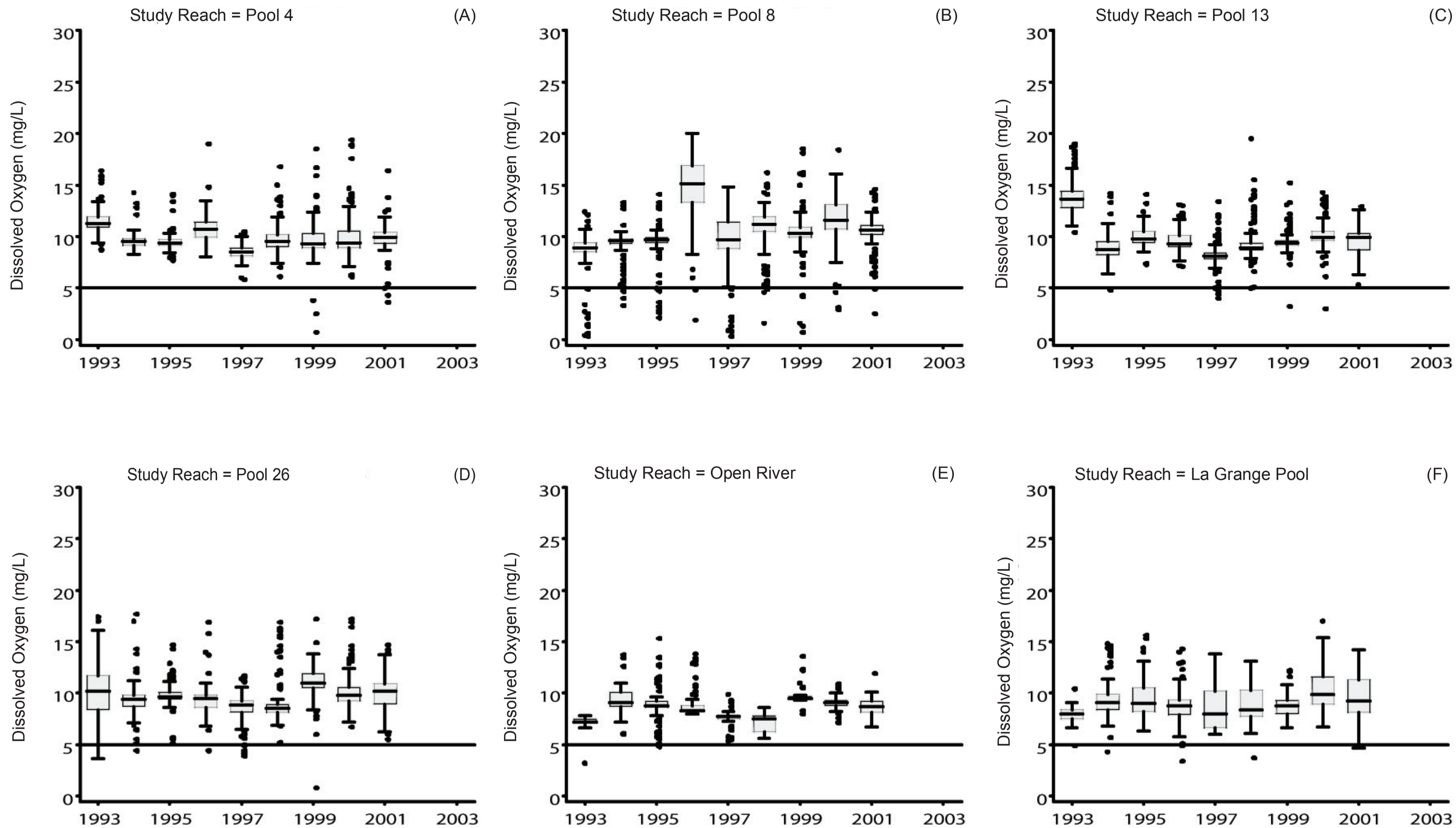

Figure 5.3. Fall dissolved oxygen (DO) in all strata of the six study reaches based on the Long Term Resource Monitoring Program stratified random sampling data, 1993–2002. Boxes indicate the 25th–75th percentile, the horizontal line within the box represents the median, whiskers extend to the largest and or smallest value within 1.5 × the interquartile range (75th–25th percentile), and points represent observations outside this range. The solid line at DO = 5 mg/L indicates DO concentration generally recognized as necessary to protect most aquatic organisms.

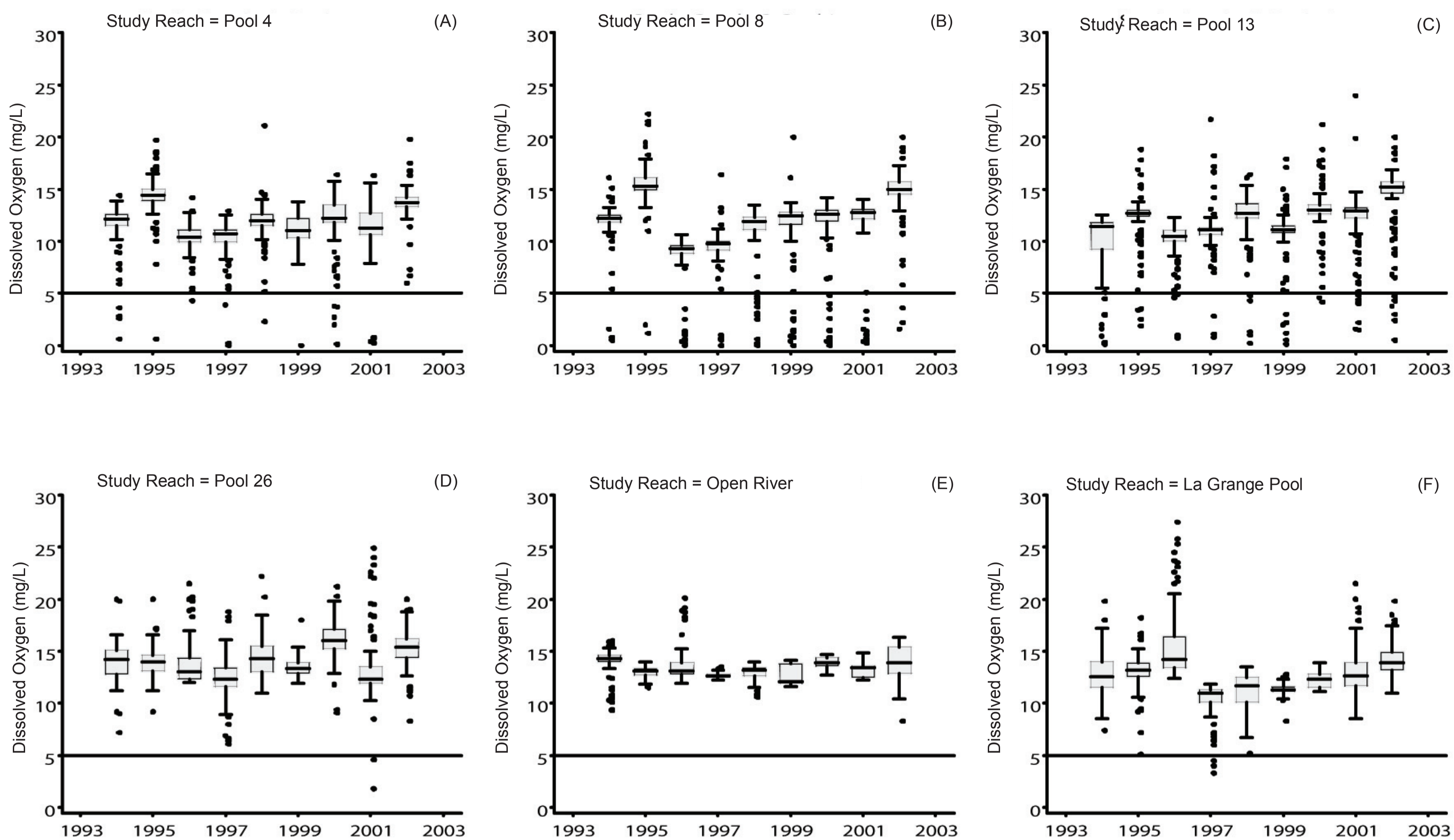

Figure 5.4. Winter dissolved oxygen (DO) in all strata of the six study reaches based on the Long Term Resource Monitoring Program stratified random sampling data, 1994–2002. Boxes indicate the 25th–75th percentile, the horizontal line within the box represents the median, whiskers extend to the largest and or smallest value within 1.5 × the interquartile range (75th–25th percentile), and points represent observations outside this range. The solid line at DO = 5 mg/L indicates DO concentration generally recognized as necessary to protect most aquatic organisms.

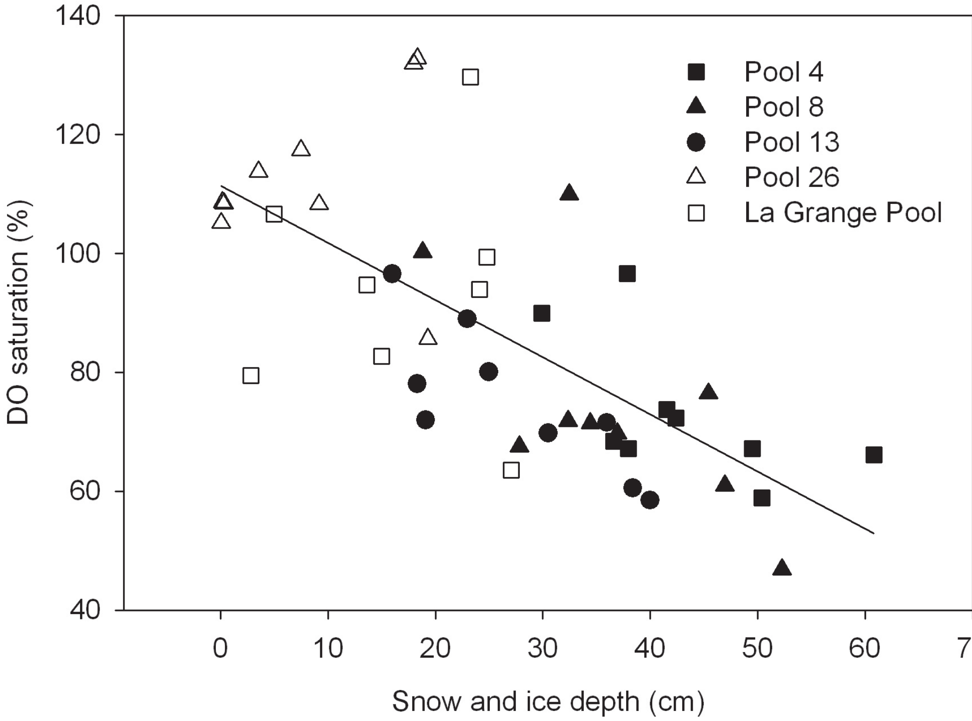

Figure 5.5. Dissolved oxygen (DO) saturation versus average snow and ice depth for the backwaters of the Long Term Resource Monitoring study reaches, 1994–2002; Open River has no backwater stratum.

discharge reduces the exchange between the main channel and backwaters and side channels. The increased residence time and high rates of biological respiration in summer often produce low DO conditions in backwaters.

For the southern study reaches (Pool 26, Open River, and La Grange Pool), seasonal patterns are somewhat different. The DO concentrations are highest in winter partly because there is little ice and snow cover in these study reaches in winter, and water temperatures are cool. In spring, DO concentrations are lower in the southern study reaches than in the northern study reaches, perhaps because of higher water temperatures. Like the northern study reaches, the DO concentrations are generally lowest in summer in the southern study reaches. Fall DO concentrations are generally intermediate relative to spring and summer concentrations.

Among the study reaches, Open River tends to exhibit lower DO concentrations (except in winter) and lower variability than the other study reaches. These differences may have been because of higher temperatures in Open River (oxygen solubility decreases with higher temperature), or because of higher suspended solids (Chapter 3), which can reduce light penetration and, therefore, photosynthesis, and increase DO demand because of increased concentrations of suspended organic material. The lower variance in DO in Open River may have been because of the reduced habitat diversity of this reach relative to the other study reaches. There were no backwaters in Open River, and backwaters were generally where the extremely high and extremely low DO values were observed in the other study reaches.

Spatial Patterns in Low DO Concentrations

The overall occurrence of low, daytime, surface DO concentration was infrequent in the UMRS (about 4% of the sites monitored were <5 mg/L), but low DO was most common in summer and winter (Figures 5.2 and 5.4). Except in La Grange Pool, backwaters were the only strata where low DO concentrations were regularly observed (Table 5.1). In winter, low

Table 5.1. Mean proportion of stratified random sampling sites with dissolved oxygen concentration <5 mg/L in winter and summer from 1993 to 2001. One standard deviation is shown parenthetically. Bold font indicates where the proportion of sites with low dissolved oxygen is >10%.

Study reach	Strata				
	Main channel	Side channel	Backwater	Lake[a]	Impounded
Winter					
Pool 4	0.004 (0.013)	0.008 (0.02)	**0.14 (0.14)**	0.007 (0.015)	—[b]
Pool 8	0	0.005 (0.014)	**0.13 (0.09)**	—	0
Pool 13	0	0	**0.1 (0.05)**	—	0.004 (0.01)
Pool 26	0	0.003 (0.008)	0.006 (0.02)	0	0
Open River	0	0	--	—	—
La Grange Pool	0	0	0.009 (0.03)	—	—
Summer					
Pool 4	0.008 (0.3)	0.013 (0.03)	0.058 (0.07)	0.003 (0.01)	
Pool 8	0.008 (0.02)	0.013 (0.02)	**0.12 (0.08)**		0
Pool 13	0.003 (0.01)	0.023 (0.05)	**0.21 (0.14)**		0.057 (0.1)
Pool 26	0.014 (0.04)	0.022 (0.05)	**0.13 (0.1)**	0.14 (0.2)	0.022 (0.05)
Open River	0.013 (0.04)	0.032 (0.06)			
La Grange Pool	**0.25 (0.24)**	0.35 (0.27)	0.067 (0.067)		

[a]Lake Pepin in Pool 4; Swan Lake in Pool 26.
[b]— indicates that stratum was not present in that reach.

DO occurred, on average, in 10% to 14% of the backwater sites sampled in Pools 4, 8, and 13, but did not commonly occur in other study reaches or strata (Table 5.1). In summer, the backwaters of Pools 8, 13, and 26 exhibit low DO concentrations in 12% to 21% of the sites sampled (Table 5.1). For La Grange Pool, low summer DO is observed regularly in the main (25% of sites sampled) and side channel (35% of sites sampled) areas.

Areal Extent of Low DO Concentrations

Summer

There was substantial interannual variability in the areal extent of low DO conditions in backwaters in summer (Figure 5.6). The smallest areal extent of low DO was observed in 1994 and 1995 for most study reaches. From 1995 through 2001, Pool 13 had the greatest area of low DO concentration and exhibited a substantial increase between 1996 and 1997. From 1997 through 2001, Pool 13 exhibited by far the largest backwater areas of low DO concentration. The estimated area of low DO concentration for Pool 13 was around 800 ha (nearly 30% of the backwater stratum) from 1997 to 2000 and approached 1,300 ha (45% of the stratum) in summer 2001 (Figure 5.6). The estimated area affected in summer for Pool 8 ranged from 3% to 23% (up to 400 ha in 1998). Between 60 and 80 ha (~8%) of La Grange Pool was affected by low DO concentration in summer from 1996 to 2000. That area increased somewhat in 2001 to about 110 ha (~10%). Pool 26 generally had the least estimated area affected by summer low DO concentration among study reaches (usually <100 ha), but the area affected represented a relatively large proportion (up to 30%) of the backwater stratum in Pool 26. This reflects the relatively small proportion of the study reach consisting of backwaters compared to the northern study reaches.

Winter

In winter, the areal extent of low DO concentrations was much greater in the northern study reaches than in Pool 26 and La Grange Pool (there are no backwaters in Open River). The ranking of the northern study reaches in terms of the areal extent of low DO conditions varied among years (Figure 5.7). In 1994 and 1995, Pool 13 exhibited the highest area of low DO conditions. In 1996, Pools 8 and 13 were similar and exhibited a greater areal extent of

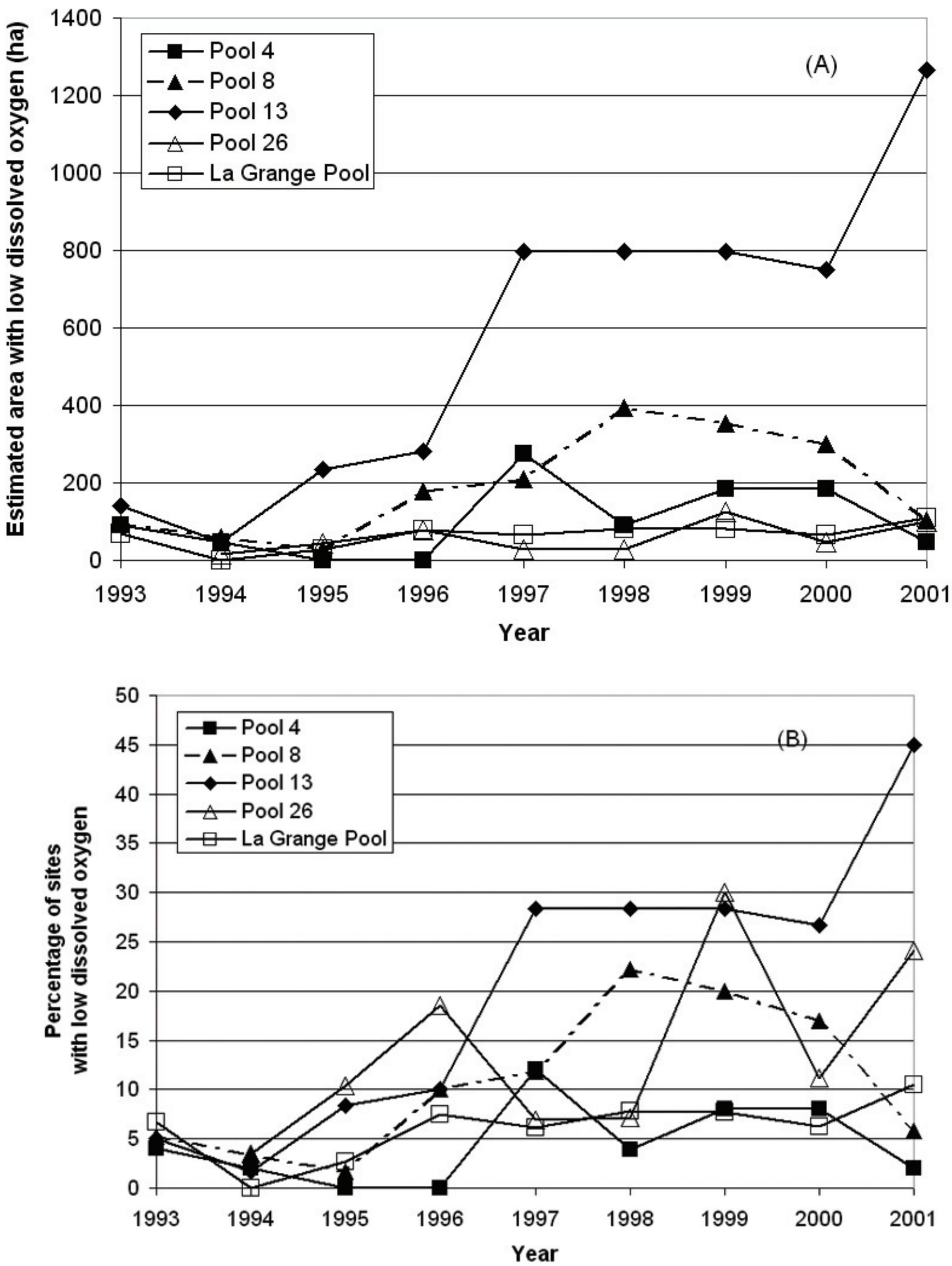

Figure 5.6. Extent of low dissolved oxygen (DO) concentration (<5 mg/L) observed in summer stratified random sampling in backwaters of the Long Term Resource Monitoring Program study reaches, 1993–2001. (*A*) Estimated area exhibiting low DO concentration, (*B*) proportion of sites that exhibited low DO concentration; Open River has no backwater stratum.

low DO conditions than Pool 4. From 1997 to 1999, low DO was much more common in Pool 4 than in Pools 8 or 13, which were similar to one another. In 2000 and 2001, the differences among the northern study reaches were moderate.

The variability among the northern study reaches in the extent of winter low DO conditions suggests that local factors varying among years caused these differences. Differences in reach morphology were not a likely explanation because these would have remained approximately constant over the time scales examined here. Regional differences in climate, particularly as it affected snow and ice cover,

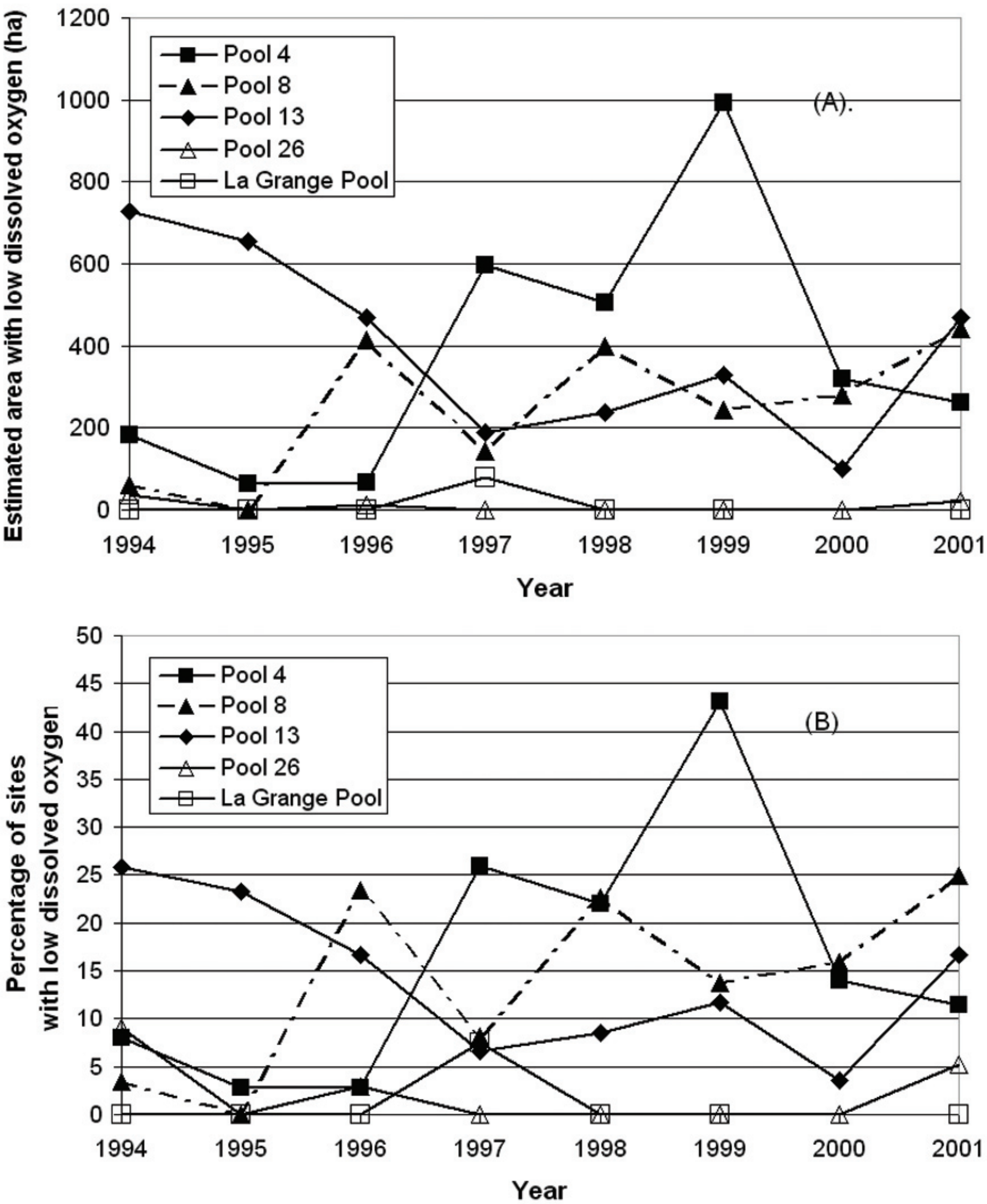

Figure 5.7. Extent of low dissolved oxygen (DO) concentration (<5 mg/L) observed in winter stratified random sampling in backwaters of the Long Term Resource Monitoring Program study reaches, 1994–2001. (*A*) Estimated area of exhibiting low DO concentration, (*B*) proportion of sites that exhibited low DO concentration; Open River has no backwater stratum.

and current velocity in the backwaters of each of these study reaches were probably responsible for much of this variation.

Low DO concentrations were rarely observed in the backwaters of Pool 26 and La Grange Pool in winter (Figure 5.7; Table 5.1). This is probably because of the minimal ice cover (which means oxygen exchange with the atmosphere is generally uninhibited) and the lower water temperatures (which increases oxygen solubility and slows oxygen consuming processes like microbial respiration). Although the total backwater area of low DO concentration is lower in winter than in summer, the biotic effects are potentially more severe in winter, given the more specific habitat requirements (i.e., low current velocity) for fish in winter (Knights et al. 1995).

Summary

These results show that there were distinct seasonal and longitudinal patterns in DO concentration in the UMRS. Low surface DO concentrations were most frequently observed in summer, followed by winter. Low surface DO concentrations were rarely observed in spring or fall. Low DO concentrations in winter occurred more frequently in the northern study reaches, primarily in backwaters. In summer, Pool 4 and Open River rarely exhibited low DO concentrations, but the other four study reaches did at frequencies that varied among years and study reaches. The data represent surface DO concentration measurements taken during the day. The DO concentration exhibits diel cycles with minima usually occurring slightly before dawn and maxima occurring in the afternoon. In addition, DO concentration can change significantly with depth. As a result, the data do not reflect the minimum DO concentrations that occur at a site and provide an incomplete picture of the extent to which anoxia may impair UMRS habitat. However, the data do indicate where low DO concentrations are likely to be a problem. The patterns in surface DO concentration suggest that detailed studies of the distribution and frequency of low DO concentration would be most productive if focused on Pools 8 and 13 in summer and winter. Likewise, management strategies, such as introducing flow to backwaters and using water-level manipulation to “pulse” water back into backwaters, might be tested or used in these areas.

The results illustrate important temporal and spatial patterns in DO concentrations. Additional studies addressing potential mechanisms for these studies would provide important additional understanding of the UMRS. Examples of areas for further inquiry include the following: What is the role of current velocity in determining backwater DO concentrations? Is there a current velocity threshold above which low DO concentrations seldom occur? How important is water temperature in determining the north–south differences in DO concentrations? Why does Pool 4 experience less frequent low DO concentrations than Pools 8 and 13? Combining studies designed to directly investigate questions such as these with the existing LTRMP water quality database would provide critical information concerning DO dynamics in the UMRS.

Chapter 6: Tributaries and Their Influence on Water Quality Spatial Patterns

by

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Background

Water quality varies laterally and longitudinally in large river systems such as the Upper Mississippi River System (UMRS). The UMRS consists of various aquatic areas (e.g., the main channel, backwaters), and the water quality in each of these aquatic areas is dependent on internal processes and the degree of hydrologic connectivity among them. Tributary inputs can strongly affect lateral and longitudinal patterns in water quality in the UMRS. The water quality of each tributary is a reflection of the size, climate, land use, geology, hydrology, and chemical characteristics of its own basin (U.S. Geological Survey 1999). The Upper Mississippi River (UMR) tributaries differ greatly in their physical and chemical characteristics and their effects on the water quality of the mainstem river (U.S. Geological Survey 1999; Wasley 2000). Tributaries that drain into the UMR range from small intermittent streams to large, commercially navigable rivers such as the Illinois and Missouri Rivers.

Since 1988, the Long Term Resource Monitoring Program (LTRMP) has monitored water quality on 43 UMRS tributaries. Tributaries to 11 of the 26 navigational pools on the UMR have been sampled, as well as several tributaries to the Open River reach of the UMR and La Grange Pool of the Illinois River. These data help identify short- and long-term trends in the tributaries themselves and to ascertain in-pool effects that may result from tributary inputs.

Contiguous backwater areas (hereafter referred to as backwaters) are important habitat for much of the UMRS biota. They serve as spawning, rearing, and overwintering areas for several species of fish (Pitlo 1992; Gent et al. 1995). Waterfowl use these areas heavily in spring and fall migrations. Abundance of vegetation and invertebrates is typically greater in backwaters than in the main channel (Sauer 2004; Yin and Langrehr 2005).

Construction of the lock and dam system in the 1930s substantially affected UMRS backwaters, increasing their areal extent, particularly in the northern navigational pools (includes Pools 4, 8 and 13; U.S. Geological Survey 1999). Construction of the lock and dam system has also greatly increased sedimentation rates in backwaters because of changes in natural hydrologic patterns (Bhowmik and Adams 1989). Tributaries may contribute to sediment accumulation in backwaters when they flow directly into, or upstream of a backwater area. Tributaries typically carry high suspended solids loads after heavy rains. This suspended material settles out as the tributaries enter the quiescent backwaters increasing the sedimentation rates in these backwaters (Bhowmik and Demissie 1989).

In this chapter, the UMRS tributaries and their influence are examined by looking at two distinct topics: (1) effects of tributary inputs on lateral and longitudinal patterns in water quality within LTRMP study reaches, and (2) differences among the tributaries and associated patterns in UMRS water quality. Specifically, we address three questions concerning the interactions of tributaries, backwaters, and the main channel of the UMRS:

1. To what extent can specific conductivity (a conservative hydrologic tracer) be used to track individual tributary water masses in the UMR?

2. Is water quality at backwater sampling sites similar to that of nearby tributary sampling sites? Is it similar to that of adjacent main channel sampling sites?

3. How does water quality vary among tributaries of the UMRS?

Methods

The period of record covered in this chapter is 1993 through 2001, although data are only available beginning in 1998 for some tributaries. All water quality measurements and samples were collected following standard LTRMP water quality component procedures (Soballe and Fischer 2004). Only data from surface measurements (collected at a depth of 0.2 m) are included in this chapter.

Patterns in Tributary Conductivity

Spatial patterns in conductivity, which serves as a conservative hydrologic tracer, were used to determine the spatial extent of tributary influence in the six LTRMP study reaches. Differences often existed between specific conductivity of tributaries and the mainstem of the UMRS (Table 6.1). This situation resulted in spatial patterns in conductivity in LTRMP study reaches below the mouths of some tributaries. Sometimes, the spatial patterns in conductivity reflected tributary inputs for a considerable distance downstream.

Specific conductivity readings from summer (late July–early August) stratified random sampling (SRS; Chapter 1) from each year in each of the six study reaches were compared to the mean summer specific conductivity (July through September 1993–2001) of major tributaries. The mean conductivity for each tributary was calculated across all years to reduce the effects of year-to-year variability inherent in tributary data. Data from summer SRS events were chosen because in summer water levels of the LTRMP study reaches and their tributaries are typically stable and at low flow.

Conductivity measurements from each summer SRS event in each study reach were categorized into quartiles and plotted by year on a map of the study reach. The maps were examined for longitudinal and lateral tributary influences by assessing the spatial pattern in conductivity near and below the tributary mouths.

To determine if similar patterns were evident for other water quality constituents, we plotted total nitrogen, total phosphorus, and total suspended solids, categorized by quartile, from spring SRS events (performed in late April–early May) and compared them to the 1993–2001 spring (April through June) tributary means for each respective constituent for each year and each study reach. Data from spring sampling were used because nitrogen and suspended solids concentrations are typically relatively high in spring.

Tributary Influences on Backwaters

Water quality data from tributary, backwater, and main channel sampling sites in relative proximity were analyzed for correlations among these sites. Appropriate data for this analysis were available for three locations in the UMRS: (1) Galena River/Sunfish Lake in Pool 12, (2) Rush Creek/Savanna Bay in Pool 13, and (3) Rock Creek/Shrickers Lake in Pool 14. Data for the Galena River/Sunfish Lake example were from 1998 to 2001 (all seasons), and data for the Rock Creek/Shrickers Slough example were from 1996 to 2001 (all seasons).

Spearman rank correlation was used to test the strength of bivariate relations in water quality constituents (i.e., total nitrogen, total phosphorus, and total suspended solids) between tributary and backwater sites and between main channel and backwater sites.

Longitudinal Tributary Trends

To investigate longitudinal differences among UMRS tributaries, 1993–2001 means for select tributary constituents (total nitrogen, total phosphorus, total suspended solids, and turbidity) were used and compared graphically to 1993–2001 means from main channel aquatic areas in the LTRMP study reaches. Means from two tributaries entering into each study reach were compared to the main channel means from the same study reach. The two tributaries selected from each study area had data strings encompassed the entire period of record and were major tributaries to the study reach. The means represent the overall mean for the period of record from each respective tributary (fixed-site data) and study reach (main channel data from SRS sampling) over all seasons.

Spatial Patterns in Specific Conductivity

The effects of tributary inputs on spatial patterns in conductivity in the UMR varied greatly among tributaries because of differences in specific conductivity and discharge among tributaries and annual and seasonal hydrologic variations in the LTRMP study reaches. Pools 4, 8, and 26 showed clear spatial patterns in conductivity related to tributary input in some years (Figures 6.1–6.3), whereas the other study reaches did not.

Pool 4

In Pool 4, the Chippewa River enters the lower third of the pool below Lake Pepin. Chippewa River water consistently exhibited conductivity readings that were distinctly lower than the adjacent Pool 4 main channel mean (Table 6.1). Chippewa River water remained close to the eastern border of the main channel below its discharge and appeared to track further downstream through the backwater complex in the eastern side of lower Pool 4. This spatial pattern was evident in 1997, 1998, and 2000, and somewhat evident in 1999, but was not seen in 1993 to 1996. Of these years, 1997 and 2000 most clearly showed the influence of the Chippewa River on downstream aquatic areas (Figure 6.1).

Pool 8

The Black River enters the eastern portion of upper Pool 8, and like the Chippewa River in Pool 4, the Black River had conductivity readings consistently lower (261 µS/cm) than the Mississippi River main channel river water (429 µS/cm) at their confluence (Table 6.1). Generally, there was a clear east–west conductivity gradient in Pool 8 and it was most readily apparent in 1998 and 1999 (Figure 6.2), but is present for all years except 1994–1995.

The La Crosse River (334 µS/cm), which enters the eastern side of the pool at the mouth of the Black River, and the Root River (509 µS/cm), which enters the western side of the pool from Minnesota contribute to the lateral conductivity gradient in Pool 8. The lower conductivity readings typically observed in the eastern half of Pool 8 were probably because of inflows from the Black and La Crosse Rivers, whereas the higher readings in the western half of the pool probably reflected the influence of the Root and Mississippi River inflows from Lock and Dam 7.

Pool 13

In Pool 13, the six primary tributaries all had mean summer conductivities higher than the main channel of the Mississippi River (424 µS/cm) and ranged from 565 µS/cm in the Maquoketa River to 730 µS/cm in the Elk River (Table 6.1). However, compared to other LTRMP study reaches—particularly Pools 4 and 8—Pool 13 showed much less spatial variability in specific conductivity. This was probably because the flows of the tributaries are low relative to the UMR flow in Pool 13, which restricts their influence to small areas near their mouths.

Pool 26

The major tributary to Pool 26 is the Illinois River. The Illinois River had a mean summer conductivity of 640 µS/cm, which was higher than the Pool 26 summer main channel mean of 469 µS/cm. As a result, the Illinois River affected the spatial patterns in conductivity in Pool 26. Although this was not evident in each year, it was apparent in 1997, 1998, and 1999, and strongest in 1998 and 1999 (Figure 6.3). In these years, higher conductivity readings in mainstem Mississippi River water below the confluence of the Illinois River were apparent. The discharges of the other five tributaries to Pool 26 were minor compared to the Illinois River, and no local effects of these five tributaries on Pool 26 conductivity were noted in this analysis.

Open River

The portion of the Open River reach monitored by the LTRMP begins about 110 river miles below the confluence of the Missouri and Mississippi Rivers. Only two tributaries in this study area were monitored, the Big Muddy River and the Headwaters Diversion Channel (formerly

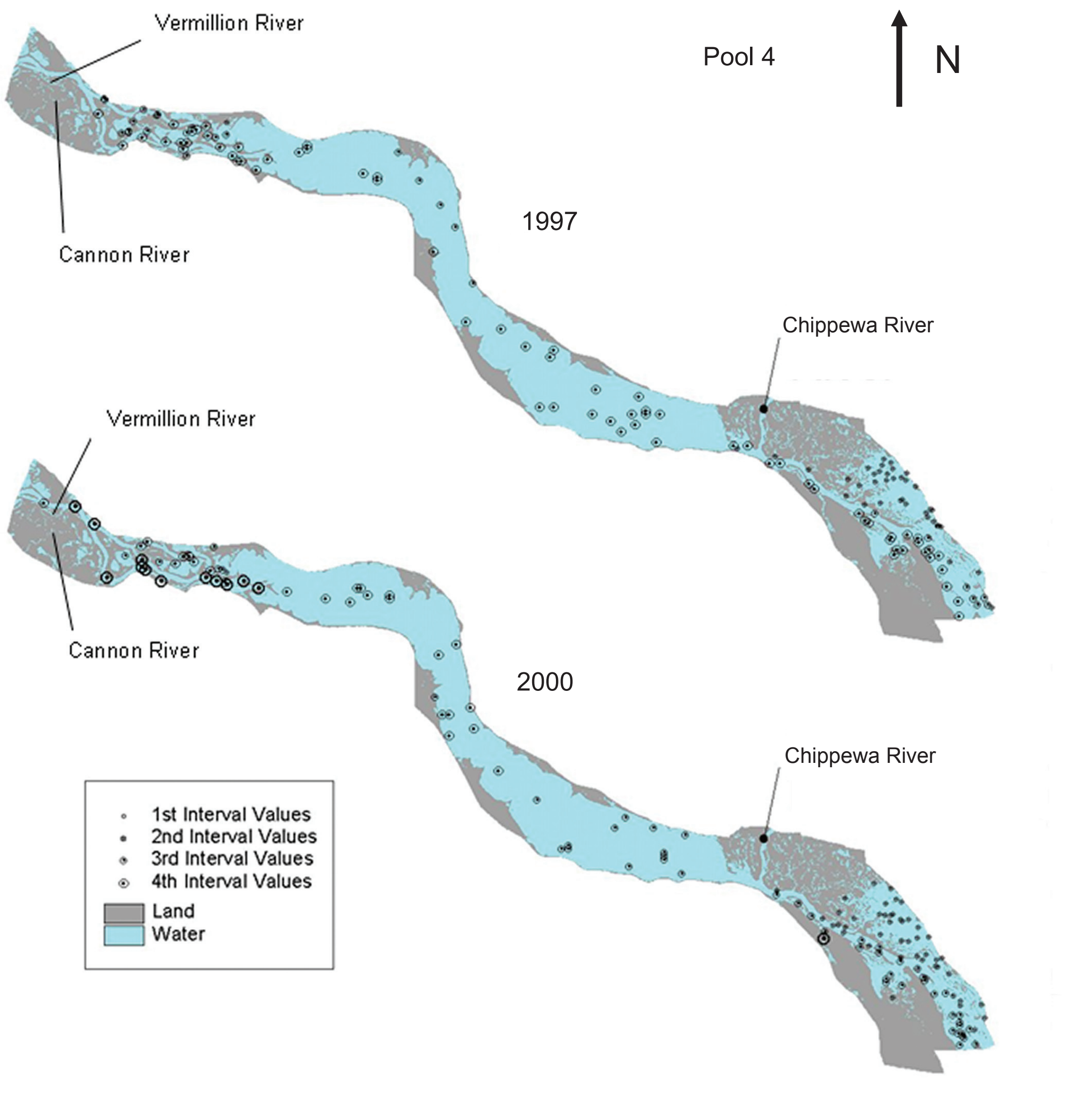

Figure 6.1. Local influence of the Chippewa River on water quality of lower Pool 4, based upon specific conductivity (µS/cm) readings from summer stratified random sampling. Influence of the Chippewa River is evident from lower specific conductivity readings in the main channel and backwaters on the eastern side of Pool 4 below the mouth of the Chippewa River in 1997 and 2000.

the Little River). No localized effects of these tributaries were evident in any year. This was most likely because of the small discharge of these tributaries relative to the Mississippi River.

La Grange Pool of the Illinois River

Five tributaries to La Grange Pool of the Illinois River are included in the LTRMP fixed-site sampling regime. The conductivities of these tributaries were similar to the Illinois River (654 µS/cm), and conductivity in La Grange Pool was spatially homogeneous especially during years of high summer discharge (e.g., 1993 and 1996). Mean summer conductivities for all five tributaries ranged from 518 µS/cm in the La Moines River to 666 µS/cm in the Sangamon

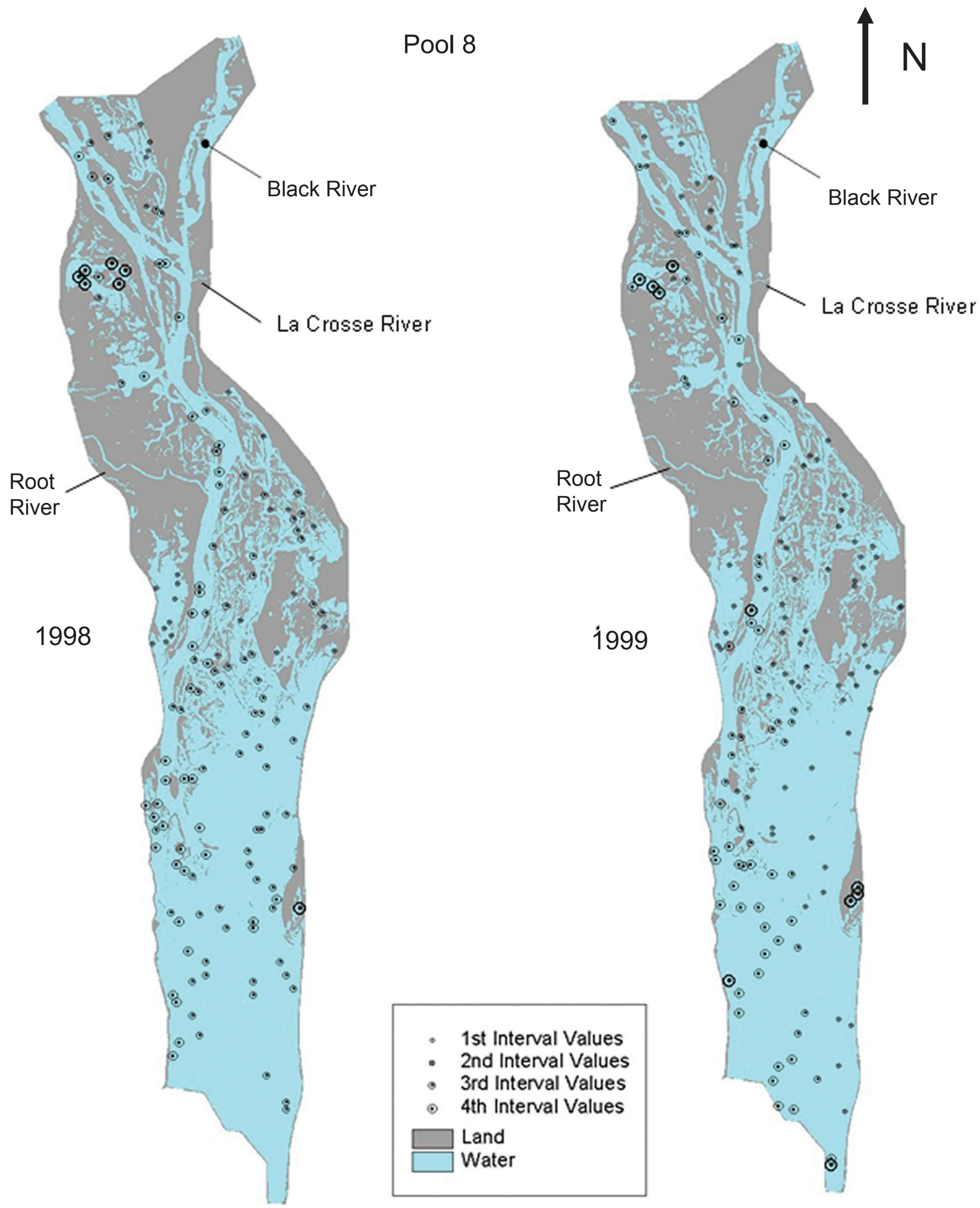

Figure 6.2. Local influence of the Black River on water quality in the eastern half of Pool 8, based upon specific conductivity (µS/cm) from summer stratified random sampling. Influence of the Black River is evident from lower specific conductivity in various aquatic areas along the eastern side of Pool 8 below the mouth of the Black River compared to values in the main channel of the Mississippi River in 1998 and 1999.

River (Table 6.1). Summer conductivity readings in La Grange Pool were typically all in the same quartile within any specific year. Thus, it was not possible to determine spatial effects of tributaries in this pool using conductivity as a tracer.

Total Nitrogen, Total Phosphorus, and Total Suspended Solids

Local influence of tributary inputs on total nitrogen (TN), total phosphorus (TP), and total

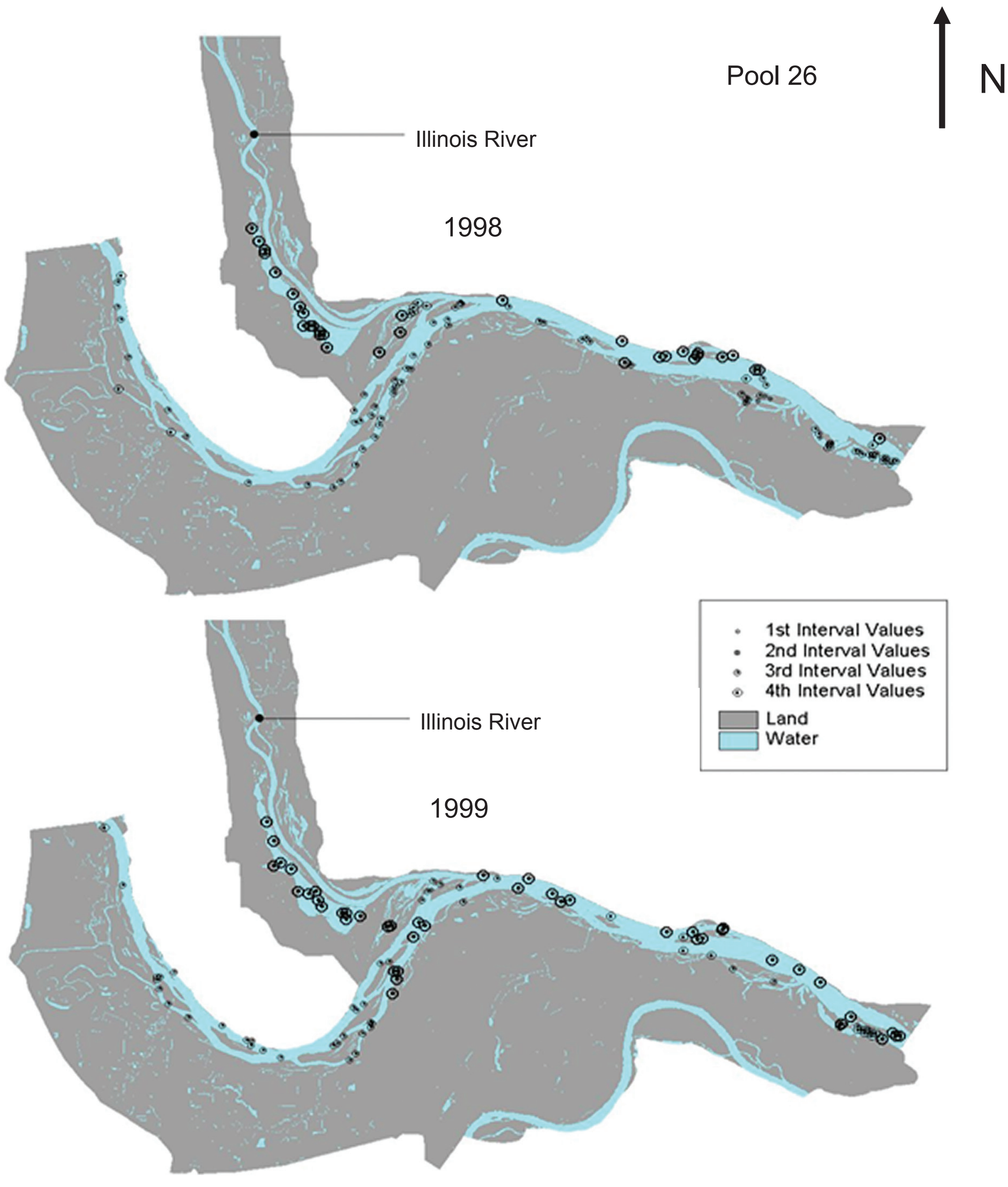

Figure 6.3. Local influence of the Illinois River on main channel water quality of Pool 26, based on specific conductivity (µS/cm) from summer stratified random sampling. Influence of the Illinois River is evident from higher specific conductivity readings in main channel areas of Pool 26 below the mouth of the Illinois River in 1998 and 1999.

suspended solids (TSS) in the Mississippi River were generally not seen in this analysis. It may be that these effects occurred at spatial and temporal scales that LTRMP sampling was not designed to detect (e.g., local effects during storm events). Occasionally, a local effect was noted for a specific parameter at a specific location, but no trends similar to those observed using conductivity intervals were obvious. However, concentrations of these constituents

Table 6.1. Mean conductivity (µS/cm) for study reaches and selected tributaries in the Upper Mississippi and Illinois Rivers, 1993–2001. Data for tributaries are from the Long Term Resource Monitoring Program fixed sites in July–September. Data for the main channel of the study reaches are from stratified random sampling in late July–early August. OR = Open River and LG = La Grange Pool.

Tributary	Study reach	Mean summer conductivity of tributary	Mean summer conductivity of study reach
Cannon River, Minnesota	4	576	453
Vermillion River, Minnesota	4	550	453
Chippewa River, Wisconsin	4	156	453
Black River, Wisconsin	8	261	429
La Crosse River, Wisconsin	8	334	429
Root River, Minnesota	8	509	429
Maquoketa River, Iowa	13	565	424
Elk River, Iowa	13	730	424
Illinois River, Illinois	26	640	469
Missouri River, Missouri	OR	665	515
Big Muddy River, Illinois	OR	667	515
Headwaters Diversion, Missouri	OR	306	515
La Moines River, Illinois	LG	518	654
Sangamon River, Illinois	LG	666	654

can be altered by biological processes creating short-term dynamics that are not likely to be detected by our seasonal sampling. Because spatial patterns were evident in conductivity, a conservative tracer of tributary influence, then those same patterns may appear in concentrations of other chemical constituents. The magnitude of tributary discharge relative to main channel discharge is an important factor in determining whether a tributary will have substantial effect on spatial patterns in water quality within a study reach.

Water Quality Comparison of Sampling Sites in Backwater, Tributary, and Main Channel Areas

Water quality relations among sets of sampling sites in backwater, tributary, and nearby main channel areas where tributaries enter into or near a backwater provide some information about the hydrologic linkages among these three areas. In these situations, one might expect that the backwater to be affected more by the tributary than by the main channel and that water quality measures in the backwater would be more closely correlated with the tributary.

Because of geomorphologic differences among study reaches in the UMRS, this configuration is not found in all LTRMP study reaches. It is most likely to occur in the Upper Impounded Reach (Pools 1 through 13) of the UMR, where tributaries enter the pools by way of braided side channels or directly into backwaters. It is not common in the Lower Impounded Reach (Pools 14 through 26) and does not occur in the Open River Reach (below the confluence of the Mississippi and Missouri Rivers). We investigated three examples of backwater-tributary-main channel complexes, one each from Pools 12, 13, and 14. In Pool 12, Sunfish Lake is the backwater, the Galena River is the local tributary, and the Lock and Dam 11 tailwaters in Dubuque, Iowa, is the main channel site (Figure 6.4). In Pool 13, Savanna Bay is the backwater, Rush Creek is the local tributary, and the Lock and Dam 12 tailwaters in Bellevue, Iowa, is the main channel site (Figure 6.5). In Pool 14, Shrickers Lake is the backwater, Rock Creek is the local tributary, and a fixed sampling site at river mile 511.4 near Camanche, Iowa, is the main channel site (Figure 6.6). For each of these examples, data from a single fixed

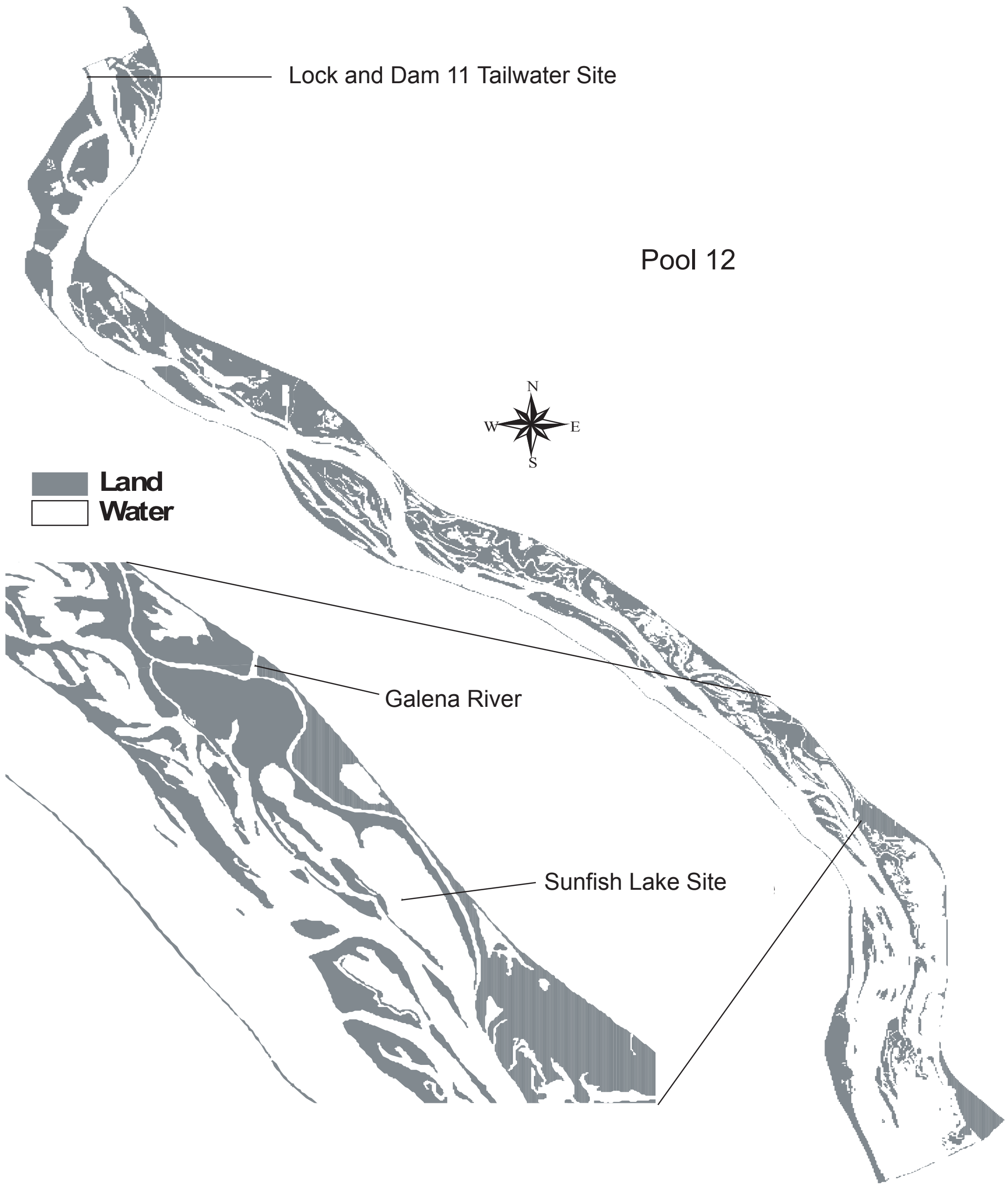

Figure 6.4. Pool 12 of the Upper Mississippi River showing the Lock and Dam 11 tailwater site, Sunfish Lake site, and the Galena River. Inset map shows spatial relation of the Galena River to the Sunfish Lake site.

site from each aquatic area (backwater, tributary, and main channel) were analyzed. These fixed sites do not provide an unbiased estimate of concentrations of water quality constituents in these different aquatic areas, but can provide some insight into the relations that may exist among them.

Galena River/Sunfish Lake and Rush Creek/Savanna Bay

The TN, TP, and TSS concentrations at the Sunfish Lake site were positively correlated with concentrations at the Galena River site (tributary) and the Lock and Dam 11 tailwaters

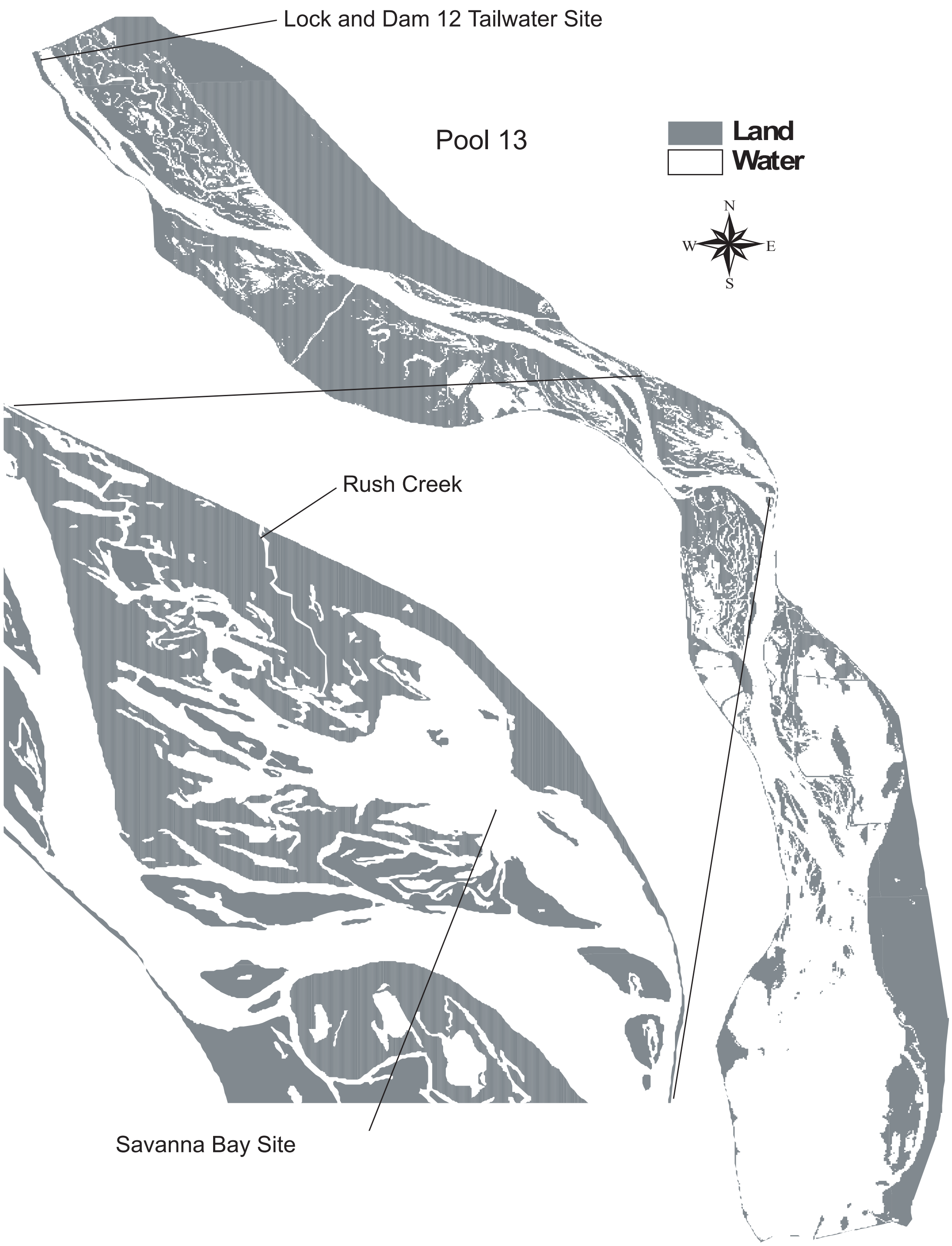

Figure 6.5. Pool 13 of the Upper Mississippi River showing the Lock and Dam 12 tailwater site, Savanna Bay site, and Rush Creek. Inset map shows spatial relation of Rush Creek to the Savanna Bay site.

site (main channel; Figures 6.7–6.9, Panels A and B). For TN, concentrations were much higher at the tributary site than at the backwater site suggesting substantial dilution of tributary inputs in Sunfish Lake. The TP was generally higher at the Sunfish Lake site than at the tributary site. There was not a consistent difference between the Sunfish Lake site and the Galena River site

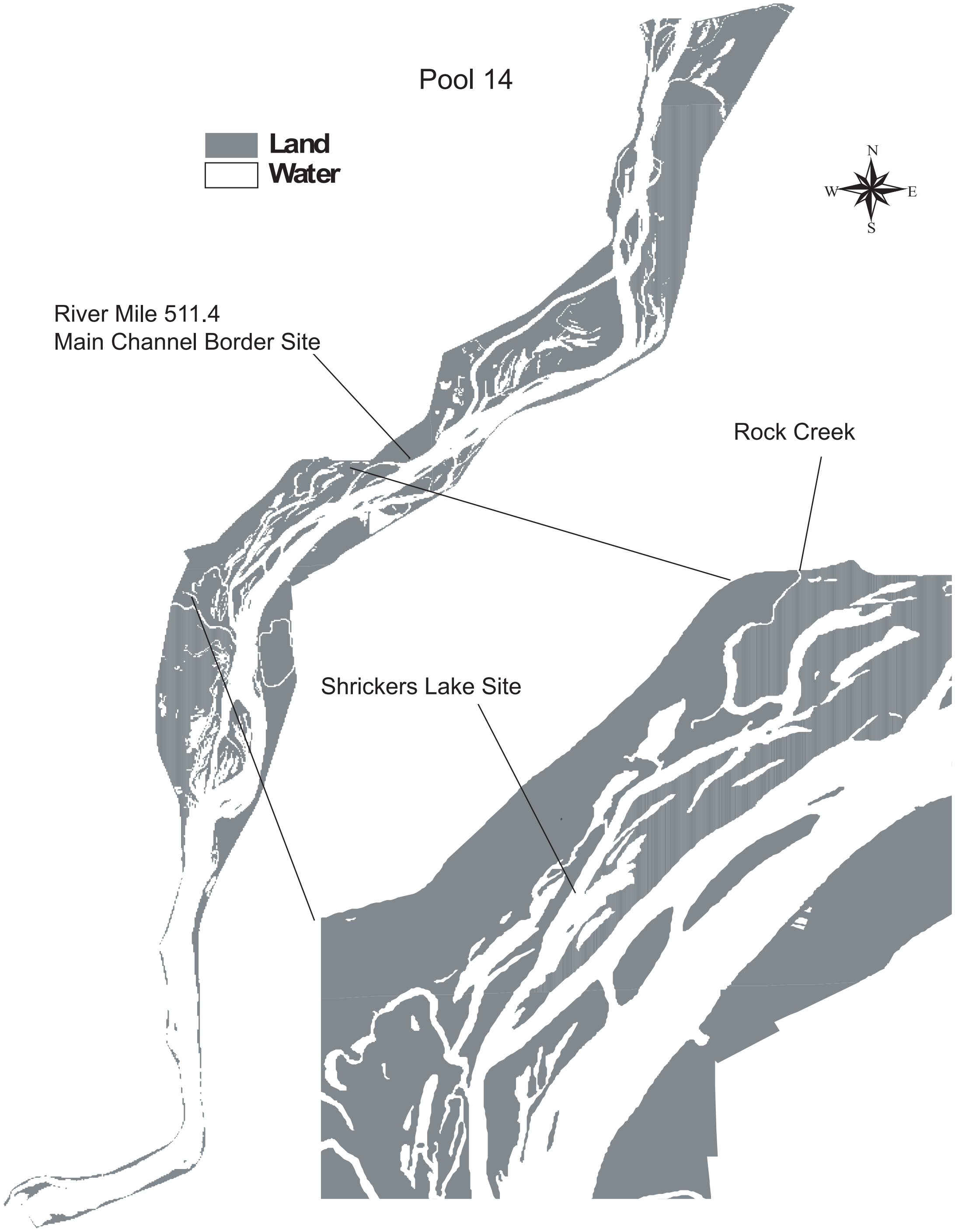

Figure 6.6. Pool 14 of the Upper Mississippi River showing Pool 14 main channel border fixed-site sampling, Shrickers Lake site, and Rock Creek. Inset map shows spatial relation of Rock Creek to the Shrickers Lake site.

for TSS. A much stronger positive relationship existed between the Sunfish Lake site and the fixed main channel site in the Lock and Dam 11 tailwaters than between the Sunfish Lake site and the Galena River site (Figures 6.7–6.9; Panels A and B), particularly for TN and TP. Large differences in concentration between the main channel and backwater sites were not observed,

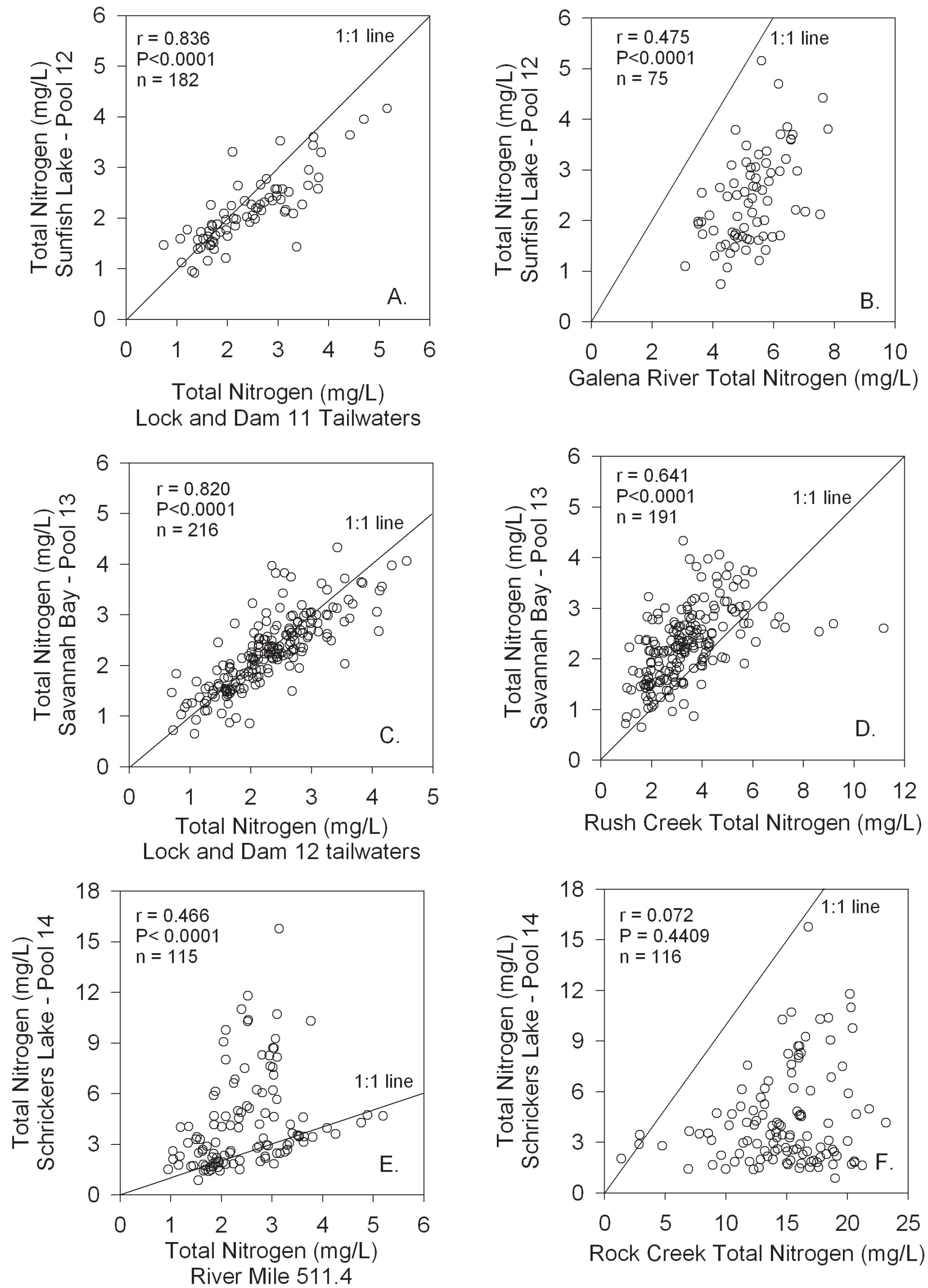

Figure 6.7. Scatter plots with Spearman correlation values for total nitrogen (mg/L) at backwater, main channel, and tributary sites in (*A*) Sunfish Lake compared to Lock and Dam 11 tailwaters (Pool 12), (*B*) Sunfish Lake compared to Galena River (Pool 12), (*C*) Savanna Bay compared to Lock and Dam 12 tailwaters (Pool 13), (*D*) Savanna Bay compared to Rush Creek (Pool 13), (*E*) Shrickers Lake compared to a Pool 14 main channel site (Pool 14), and (*F*) Shrickers Lake compared to Rock Creek (Pool 14). The solid diagonal line in each chart represents the one-to-one line.

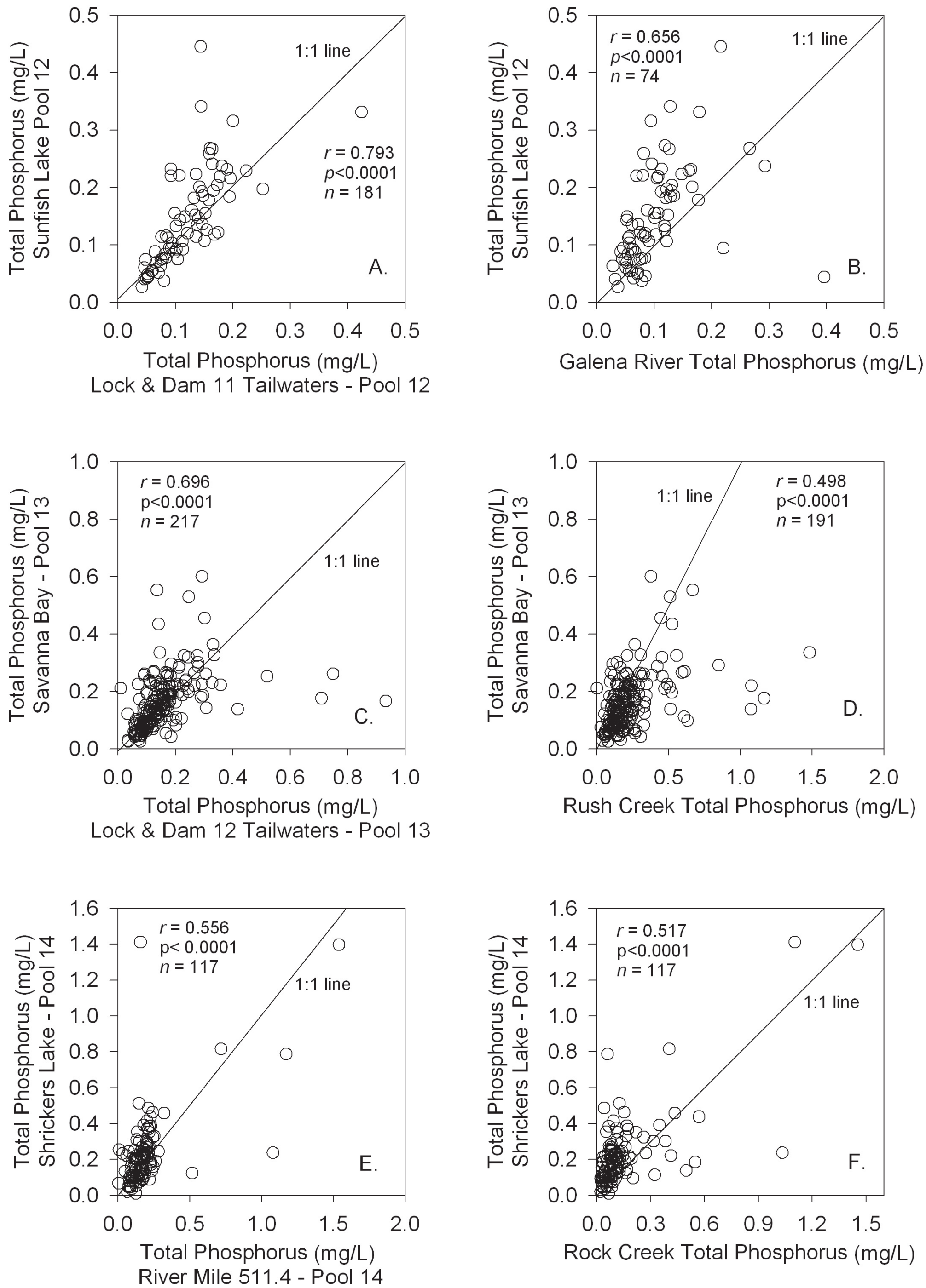

Figure 6.8. Scatter plots with Spearman correlation values for total phosphorus (mg/L) at backwater, main channel, and tributary sites in (*A*) Sunfish Lake compared to Lock and Dam 11 tailwaters (Pool 12), (*B*) Sunfish Lake compared to Galena River (Pool 12), (C) Savanna Bay compared to Lock and Dam 12 tailwaters (Pool 13), (*D*) Savanna Bay compared to Rush Creek (Pool 13), (*E*) Shrickers Lake compared to a Pool 14 main channel site (Pool 14), and (*F*) Shrickers Lake compared to Rock Creek (Pool 14). The solid diagonal line in each chart represents the one-to-one line.

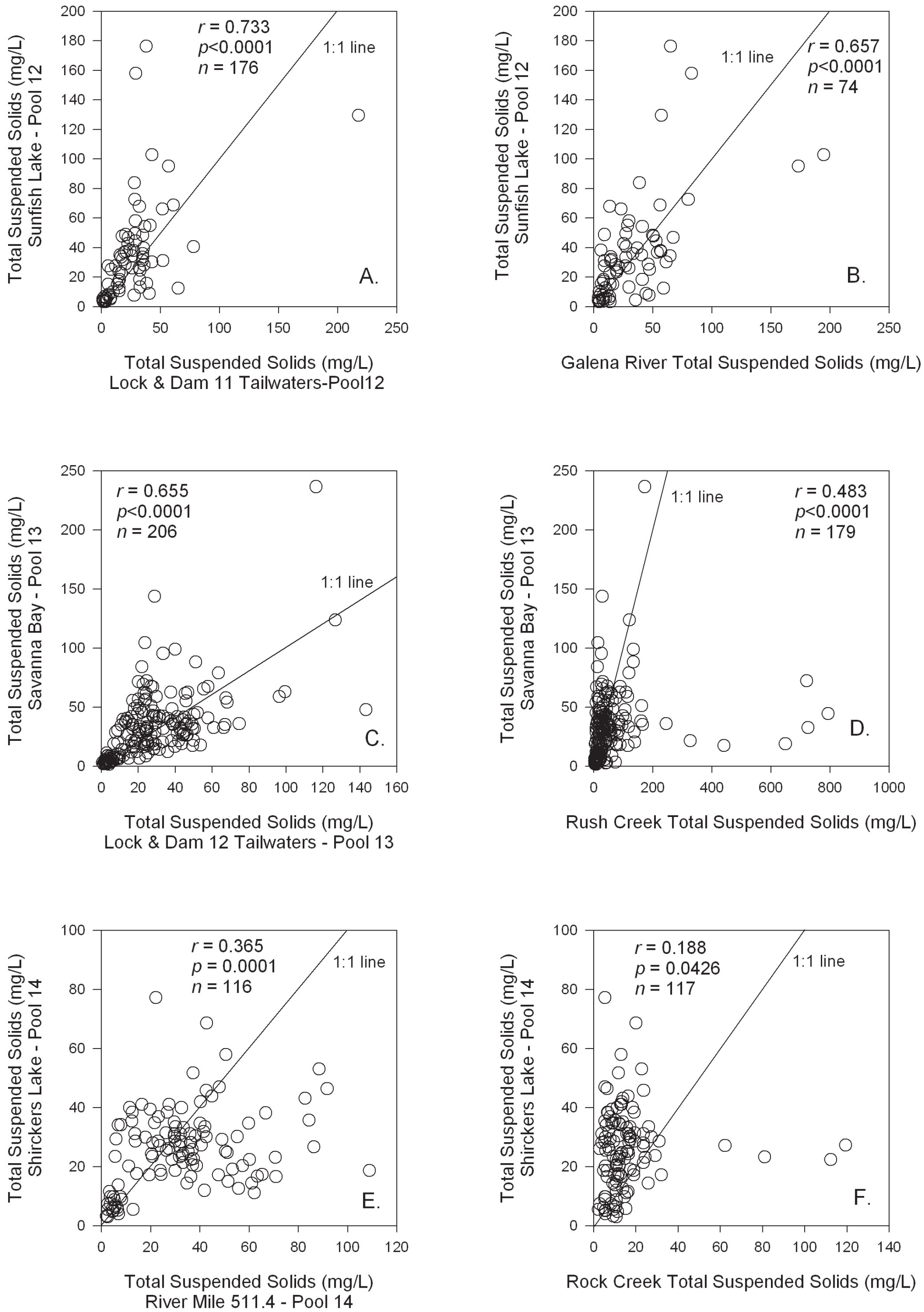

Figure 6.9. Scatter plots with Spearman correlation values for total suspended solids (mg/L) at backwater, main channel, and tributary sites in (*A*) Sunfish Lake compared to Lock and Dam 11 tailwaters (Pool 12), (*B*) Sunfish Lake compared to Galena River (Pool 12), (*C*) Savanna Bay compared to Lock and Dam 12 tailwaters (Pool 13), (*D*) Savanna Bay compared to Rush Creek (Pool 13), (*E*) Shrickers Lake compared to a Pool 14 main channel site (Pool 14), and (F) Shrickers Lake compared to Rock Creek (Pool 14). The solid diagonal line in each chart represents the one-to-one line.

although TSS is more often higher at the Sunfish Lake site then at the main channel site (Figure 6.9A).

A similar pattern occurred in Savanna Bay (Figures 6.7–6.9; Panels C and D). Water quality constituents at the Lock and Dam 12 tailwater site were more strongly correlated to the same constituents at the Savanna Bay site than those observed at the Rush Creek site. The TN was usually higher at the Rush Creek site (tributary) than at the Savannah Bay site (backwater). For TP and TSS, there were no clear differences between the tributary and backwater sites. The TN, TP, and TSS were well correlated between the main channel and backwater sites in this example, and there were no consistent differences in concentrations in any of these variables between these two sites.

For these two case studies, the correlations between backwater (Sunfish Lake and Savanna Bay) and main channel sites (tailwaters) were stronger for TN (Figure 6.7; Panels A and C) than TP (Figure 6.8, Panels A and C) and TSS (Figure 6.9, Panels A and C). Factors affecting this relationship probably included the degree of connectivity of the backwater to the main channel and the discharge of the tributary relative to the residence time of the receiving backwater.

Rock Creek/Shrickers Lake

Gritters and Gould (1996) showed that water quality in Shrickers Lake, a backwater lake in Pool 14, was greatly impaired by high nutrient levels compared to backwaters in Pool 13. Except during high water conditions, Shrickers Lake does not receive flow from the main channel. However, it does receive flow from a small tributary, Rock Creek. Rock Creek has extremely elevated levels of nitrogenous compounds (i.e., TN, nitrate + nitrite, and ammonia nitrogen), below the outflow from an ammonia nitrogen fertilizer production plant (Gritters and Gould 1996). Concentrations of these constituents were significantly elevated in Shrickers Lake as well. Shrickers Lake has been shown to have major algal blooms and die-offs and elevated levels of chlorophyll *a*, which is further evidence of nutrient enrichment (Gritters and Gould 1996).

Despite the evidence that water quality in Shrickers Lake has been greatly affected by Rock Creek (Gritters and Gould 1996), there was not a significant correlation between water quality constituents (TN, TP, and TSS) at the Rock Creek and Shrickers Lake sites (Figures 6.7–6.9, Panel F). As with the other backwater, tributary, and main channel complexes, water quality measurements at the Shrickers Lake site were more strongly correlated with the Pool 14 main channel site than the Rock Creek site. In this instance, however, the relationship between the main channel and backwater sites was weaker than those found in the Sunfish Lake and Savanna Bay studies.

This analysis shows that, at these temporal scales, water quality at the backwater sites was better correlated with the main channel sites than the tributary fixed sites. Correlational analysis measures how well short-term changes in concentration covary over time. It does not measure the cumulative effect of elevated tributary inputs on a backwater. The cumulative effect of tributary inputs has been documented for Shrickers Lake (Gritters and Gould 1996). For the other backwaters, additional analyses would aid in understanding the role of tributary inputs. Specifically, backwaters receiving direct tributary inputs should be compared to those that do not receive direct tributary inputs. Such a comparison could indicate differences in long-term trends and differences in relationship to main channel conditions for backwaters receiving tributary inputs versus those that do not and would improve our understanding of the factors that control water quality in backwaters.

Water Quality in Tributaries of the Upper Mississippi and Illinois Rivers

The water quality of tributaries to the UMRS reflect the land use of their catchment basins (Wasley 2000). As a result, there is substantial variation in the water quality among the tributaries of the UMRS. Here we illustrate this variation by comparing water quality in a group of UMRS tributaries to water quality in nearby main channel areas (Figure 6.10).

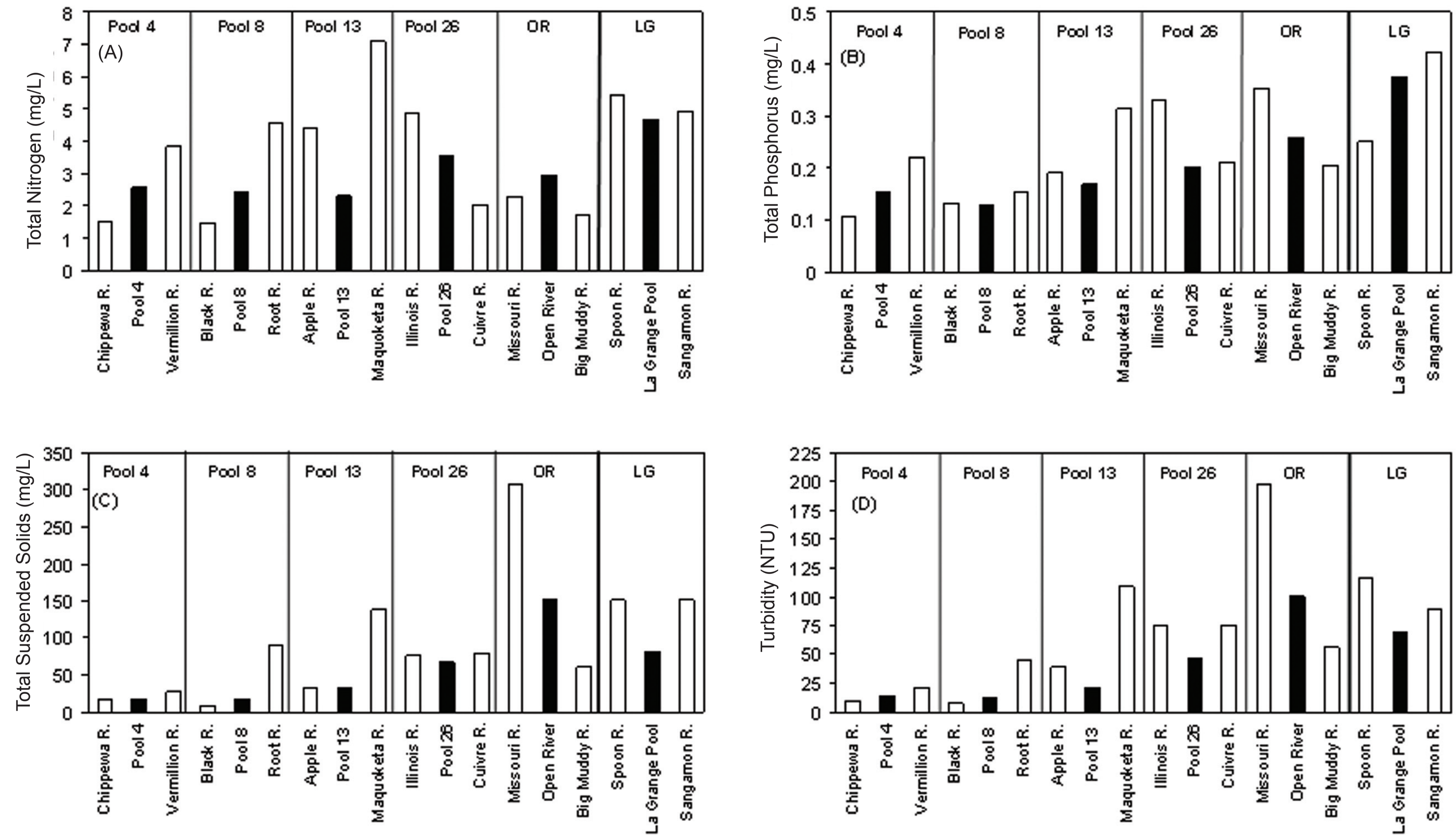

Figure 6.10. Longitudinal patterns in total nitrogen (mg/L), total phosphorus (mg/L), total suspended solids (mg/L), and turbidity (nephelometric turbidity units [NTU]) in selected tributaries to, and main channels of, the six Long Term Resource Monitoring Program study areas, based on overall means from 1993 to 2001. Note that La Grange Pool (LG) is part of the Illinois River. Open bars are the means from two major tributaries entering into each study reach and solid bars are the main channel mean in each respective study reach). OR = Open River and LG = La Grange Pool.

Upper Mississippi River Tributaries

The TN concentrations were variable among tributaries and no north–south trend was evident among the tributaries of the main channel (Figure 6.10A; Table 6.2). The Maquoketa River had the highest TN, whereas the Chippewa and Black Rivers had TN concentrations that were lower than the mainstem of the UMR. Below the confluence with the Missouri River, TN concentrations decreased, perhaps because of dilution from the lower TN concentrations in the Missouri River relative to the UMR.

The TP concentrations were less variable among tributaries than TN concentrations (Figure 6.10B; Table 6.2). The TP concentrations in the tributaries examined were equal or greater than in the UMR in 8 of 10 UMR tributaries examined here. The Maquoketa, Illinois, and Missouri Rivers had particularly high TP. The rest of the tributaries showed concentrations similar to the mainstem of the UMR. In contrast to TN, there was an increasing north–south trend in TP concentrations in the main channel of the LTRMP study reaches from Pool 8 to Open River, probably reflecting the additive affect of tributary inputs. Although the Missouri River had a comparatively low TN concentration, it had one of the highest TP concentrations of all the monitored tributaries. Phosphorus is strongly adsorbed to suspended particles and the high TP concentration observed in the Missouri River may be partly because of its high suspended sediment load (Table 6.2).

The TSS concentration is also highly variable among the tributaries (Figure 6.10; Table 6.2). The Missouri River had the highest TSS concentrations of any of the tributaries examined, followed by the Maquoketa River. Much lower TSS concentrations were seen in the tributaries to Pools 4 and 8 such as the Chippewa and Black Rivers. Mean turbidity and suspended solids levels increased downstream through the LTRMP study reaches (Figure 6.10; Chapter 3). This was probably because of the increased size of the drainage basin, changes in soil types and characteristics, and land-use practices. For example, flow in the Mississippi River increased nearly 50% below its confluence with the Missouri River (U.S. Geological Survey 1999). The mean TSS concentration of the main channel in Open River was nearly triple the mean main channel TSS concentration in Pool 26 because of the influence of the Missouri River (Figure 6.10).

Table 6.2. Baseflow discharge and mean concentrations of total nitrogen (TN), total phosphorus (TP), total suspended solids (TSS), and turbidity for selected tributaries based on fixed-site sampling by the Long Term Resource Monitoring Program in 1993–2001. Discharge data are 90% excedence data from Wasley (2000) for all but Spoon, Sangamon, and Missouri Rivers that were calculated separately from the U.S. Geological Survey discharge data available online at http://nwis.waterdata.usgs.gov/il/nwis/annual/. (Accessed January 2005.) OR = Open River and LG = La Grange Pool.

Tributary	Study reach	TN (mg/L)	TP (mg/L)	TSS (mg/L)	Turbidity (NTU[a])	Discharge ($m^3 s^{-1}$)
Vermillion River, Minnesota	4	3.86	0.22	28.4	21.2	0.6
Chippewa River, Wisconsin	4	1.52	0.106	15.8	9.7	96.3
Black River, Wisconsin	8	1.48	0.132	8.6	7.4	14.2
Root River, Minnesota	8	4.56	0.153	88.9	45.1	10.0
Apple River, Illinois	13	4.39	0.191	31.8	39.0	1.0
Maquoketa River, Iowa	13	7.08	0.314	138.9	108.9	10.9
Illinois River, Illinois	26	4.89	0.329	76.8	74.6	212.1
Cuivre River, Missouri	26	2.04	0.211	80.1	75.5	0.2
Missouri River, Missouri	OR	2.3	0.354	307.9	198.2	1,189.3
Big Muddy River, Illinois	OR	1.74	0.204	60.1	56.5	2.8
Spoon River, Illinois	LG	5.42	0.251	151.4	116.8	2.6
Sangamon River, Illinois	LG	4.93	0.424	151.8	89.6	11.4

[a]NTU = nephelometric turbidity units.

Illinois River Tributaries

The two tributaries of the Illinois River—Spoon and Sangamon Rivers—generally had high concentrations of TN, TP, TSS, and turbidity relative to the UMR (Figure 6.10). The Spoon River has higher TN concentration than all UMR tributaries except the Maquoketa River. The Sangamon River had higher concentrations of TP than any other tributaries on the UMRS. Both the Spoon and Sangamon Rivers had higher TSS concentrations than all UMR tributaries except the Missouri and Maquoketa Rivers. La Grange Pool of the Illinois River itself had higher TN and TP than any of the study reaches on the UMR reflecting the high input of its tributaries. Concentrations of TN, TP, and TSS, as well as turbidity, were similar in La Grange Pool of the Illinois River and the mouth of the Illinois River in Pool 26 of the UMR (Figure 6.10).

Iowa and Illinois are two of the most intensively farmed states in the UMR Basin, which was reflected in water quality of the tributaries that drain these two states. Of the tributaries monitored by the LTRMP, 12 of the top 15 mean TN concentrations are found in rivers and streams that drain land primarily in Iowa or Illinois. Of the 10 tributaries with the highest mean nitrogen concentration, Iowa contains six.

Summary

As determined by analysis of conductivity patterns below tributary inflows, it was apparent that water quality conditions within a study reach are not spatially homogeneous. Of the six LTRMP study reaches, Pools 4, 8, and 26 showed clear spatial patterns in conductivity related to tributary input in some years whereas Pool 13, Open River, and La Grange Pool did not. Sometimes, the direct effects of tributary inputs persisted for several miles downstream of their mouths. There were no obvious affects of tributaries on spatial patterns in TP, TN, or TSS. However, there were differences among tributaries in relation to these water quality constituents. These differences have the potential to affect aquatic vegetation abundance, species composition, and biomass. Analyses of additional water quality constituents, particularly dissolved nutrients (nitrate + nitrite, nitrogen, soluble reactive phosphorous) may reveal additional spatial patterns associated with tributary inputs.

The LTRMP SRS data were useful in assessing differences among study reaches and tributaries in various water quality constituents. However, SRS was not designed to address fine-scale patterns such as the local effects of tributaries. Because of the sampling density and the random distribution of SRS sites within each study reach, sufficient SRS sites are often not within the vicinity of tributary mouths. Spatially intensive sampling near the mouths of major tributaries may reveal local spatial patterns associated with tributary inputs.

Fixed-site monitoring data were used to assess how water quality at backwater sites is related to water quality at local tributary and main channel sites. Analysis of the relation among sampling sites in the backwaters, tributaries, and main channel showed that concentrations of TP, TN, and TSS at backwater sites were correlated with concentration at both tributary and main channel sites. However, concentrations at the backwater sites were more strongly correlated with concentrations at the main channel sites than with concentrations at the tributary sites. Fully understanding the effects of tributary inputs on the UMRS requires calculating tributary loads (discharge × concentration). Including tributary discharge in the data collected during tributary sampling would facilitate calculation of loads and increase the use of the tributary monitoring data.

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13. ABSTRACT (Maximum 200 words) This report presents a broad overview of spatial and temporal variation in the water quality of the Upper Mississippi River System (UMRS). The Long Term Resource Monitoring Program (LTRMP) provides a systemic perspective through the collection and analysis of monitoring data from six study reaches representing the upper, lower, and open river reaches of the UMRS (Upper Mississippi River: Pools 4, 8, 13, and 26, Open River [near Cape Girardeau, Missouri]; Illinois River: La Grange Pool). This report presents data from 1993 to 2001 (or 2002 when available) and focuses on spring and summer conditions. Water quality constituents (e.g., turbidity, suspended solids, chlorophyll, nutrients, and dissolved oxygen) varied among study reaches, aquatic area (e.g., main channel, contiguous backwaters, etc.) and seasons. For example, turbidity and suspended solids varied substantially among pools. Turbidity and suspended solids were much lower in lower Pool 4 than in upper Pool 4 because of the trapping of sediments by Lake Pepin, but increased in each of the study reaches from Pool 4 to Open River. Chlorophyll *a* and nutrient concentrations often differed between the main channel and contiguous backwater areas (hereafter referred to as backwaters). Summer chlorophyll *a* concentrations were generally higher in backwaters than in the main channel, and summer nitrate + nitrite (NO_x) concentrations were generally lower in backwaters than in the main channel. Seasonal patterns were evident in chlorophyll *a*, nutrient, and dissolved oxygen (DO) concentrations. Main channel soluble reactive phosphorus (SRP) concentrations peaked in September and exhibited minima in April and May. In contrast, main channel and backwater NO_x concentrations exhibited minima in fall when SRP concentrations are at their maximum. Seasonal chlorophyll *a* concentrations in main channels and backwaters show peaks in late summer and fall, with minima in winter and early summer. Seasonal DO patterns differed slightly among the northern (Pools 4, 8, and 13) and southern (Pool 26, Open River, La Grange Pool) study reaches. In the northern study reaches, DO concentrations are generally highest in spring, lowest in summer and winter, and intermediate in fall. In the southern study reaches, DO concentrations are highest in winter, lowest in summer, and intermediate in spring and fall. Spatial patterns within study reaches caused by tributary inputs were shown by the spatial patterns in specific conductivity. Of the six LTRMP study reaches, Pools 4, 8, and 26 showed clear spatial patterns in conductivity related to tributary input in some years whereas Pool 13, Open River, and La Grange Pool did not.			
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The Long Term Resource Monitoring Program (LTRMP) for the Upper Mississippi River System was authorized under the Water Resources Development Act of 1986 as an element of the Environmental Management Program. The mission of the LTRMP is to provide river managers with information for maintaining the Upper Mississippi River System as a sustainable large river ecosystem given its multiple-use character. The LTRMP is a cooperative effort by the U.S. Geological Survey, the U.S. Army Corps of Engineers, and the States of Illinois, Iowa, Minnesota, Missouri, and Wisconsin.