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Ecological Status and Trends of the Upper Mississippi and Illinois Rivers

Open-File Report 2022–1039
Version 1.1, July 2022

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

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Front and back covers: Photographs showing U.S. Geological Survey and Illinois, Iowa, Minnesota, Missouri, and Wisconsin staff that work under the Upper Mississippi River Restoration program's Long Term Resource Monitoring element collecting samples and data in the field and analyzing samples in the laboratory.

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In addition to the Upper Mississippi River Restoration program LTRM data, we include data in this report from four USGS streamgages. We acknowledge the effort and funding that support those streamgages as follows: Mississippi River at Winona, Minnesota (05378500), is operated as a USGS Federal Priority Streamgage (formerly the National Streamflow Information Program) in cooperation with the U.S. Army Corps of Engineers, St. Paul District; Mississippi River at Keokuk, Iowa (05474500), is funded by Ameren UE, the USGS Ecosystems Mission Area, the U.S. Environmental Protection Agency Great Lakes Restoration Initiative, and the U.S. Army Corps of Engineers, Engineer Research and Development Center; Mississippi River at St. Louis, Missouri (07010000), is operated in cooperation with the U.S. Army Corps of Engineers, St. Louis District; and Illinois River at Valley City, Illinois (05586100), is operated in cooperation with the U.S. Army Corps of Engineers, St. Louis District.

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Conversion Factors

U.S. customary units to International System of Units

Multiply	By	To obtain
	Length	
foot (ft)	0.3048	meter (m)
	Area	
acre	4,047	square meter (m^2)
	Flow rate	
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)

International System of Units to U.S. customary units

Multiply	By	To obtain
	Length	
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
kilometer (km)	0.5400	mile, nautical (nmi)
	Area	
square kilometer (km^2)	247.1	acre
hectare (ha)	0.003861	square mile (mi^2)
	Flow rate	
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)
meter per second (m/s)	3.281	foot per second (ft/s)
	Mass	
gram (g)	0.03527	ounce, avoirdupois (oz)

Datum

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88).

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Supplemental Information

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

Abbreviations

AEP	annual exceedance probability
bra	Brandon Road Pool of the Illinois River
ca.	circa
CHL	chlorophyll *a* concentration
CPUE	catch per unit effort
DO	dissolved oxygen
dre	Dresden Island Pool of the Illinois River
EPA	Environmental Protection Agency
FWS	U.S. Fish and Wildlife Service
HAB	harmful algal bloom
HREP	Habitat Rehabilitation and Enhancement Project
HNA	Habitat Needs Assessment
lag	La Grange Pool of the Illinois River
lidar	light detection and ranging
loc	Lockport Pool of the Illinois River
LOESS	locally weighted regression
LTEF	Long-Term Survey and Assessment of Large River Fishes in Illinois
LTRM	Long Term Resource Monitoring
mar	Marseilles Pool of the Illinois River
N	nitrogen
or1	northern section of the Open River Reach of the Upper Mississippi River
or2	southern section of the Open River Reach of the Upper Mississippi River
p	pool
P	phosphorus
peo	Peoria Pool of the Illinois River
SAV	submersed aquatic vegetation
SRS	stratified random sampling
sta	Starved Rock Pool of the Illinois River
TN	total nitrogen
TP	total phosphorus
TSS	total suspended solids
UMRR	Upper Mississippi River Restoration
USGS	U.S. Geological Survey
WRTDS	Weighted Regressions on Time, Discharge, and Season

Executive Summary

By Jeffrey N. Houser, Kristen L. Bouska, Nathan R. De Jager, Brian S. Ickes, Kathi Jo Jankowski, Danelle M. Larson, Molly Van Appledorn

Executive Summary of

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The report summarizes the status and trends of selected ecological health indicators of the Upper Mississippi River System based on the data collected and analyzed by the Long Term Resource Monitoring (LTRM) element of the Upper Mississippi River Restoration program. The report addresses four objectives:

- provides a brief introduction of the Upper Mississippi River System including its significance, history, modern-day stressors, and recent research;
- uses ecological indicators to describe the status of the river system and where and how it has changed from circa 1993 to 2019;
- discusses management and restoration implications of these changes; and
- highlights the fundamental role of long-term monitoring in the understanding, management, and restoration of large-floodplain rivers.

Overview of the Upper Mississippi River System

The Upper Mississippi River Basin covers 500,000 square kilometers of North America. The Upper Mississippi River System is congressionally defined as the commercially navigable parts of the Mississippi River main stem north of Cairo, Illinois, and its commercially navigable tributaries, including the entire Illinois River (Water Resources Development Act of 1986, 33 U.S.C § 652). Laterally, the Upper Mississippi River System includes these rivers and their floodplains and encompasses a range of riverine and floodplain habitats.

The Upper Mississippi River System is a globally significant ecosystem that is recognized by the U.S. Congress as a nationally significant transportation system and a nationally significant river ecosystem. The ecosystem provides habitat for a diverse group of riverine biota including more than 140 species of fish, at least 80 species of aquatic vegetation, 57 species of mammals, 45 species of amphibians, and 40 species of freshwater mussels. The Upper Mississippi River System is a critical migratory corridor for as much as 40-percent of North American waterfowl and provides habitat for as much as 50-percent of the world's population of *Aythya valisineria* (canvasback ducks). The floodplain forests of the river system provide nesting and migratory habitat for as many as 180 bird species, and bird abundances along the river system can be double that of adjacent upland areas. The Upper Mississippi River System also supports a variety of human activities, and river-related economic activities generate nearly $350 billion annually (U.S. Fish and Wildlife Service, 2015).

The Upper Mississippi River System Basin has been transformed for agriculture, and the river system has been modified to support commercial navigation. Most of the challenges facing the river system reflect the combined effects of past and ongoing changes to the rivers and their watersheds. These challenges include increased frequency and magnitude of floods, altered geomorphic processes and rates, floodplain forest loss, altered river-floodplain connectivity, high rates of nutrient and sediment input, scarce aquatic vegetation in some reaches, and increased abundance of invasive species.

The Upper Mississippi River Restoration Program

The 1986 Water Resources Development Act created the Environmental Management Program to monitor and rehabilitate the Upper Mississippi River System because of the system's recognized values and ongoing stressors. The program was subsequently renamed the Upper Mississippi River Restoration program and consists of two elements: Habitat Rehabilitation and Enhancement Projects and the Long Term Resource Monitoring. The Upper Mississippi River Restoration program is administered by the U.S. Army Corps of Engineers and implemented through a broad partnership composed of multiple Federal and State agencies. The U.S. Geological Survey provides the scientific leadership for the LTRM element of the program and State agency personnel conduct the field data collection and contribute their expertise to the assessment of those data.

LTRM staff collect annual long-term data for water quality, fish, and aquatic vegetation in six study reaches of the Upper Mississippi River System. The study reaches include Pool 4, Pool 8, Pool 13, Pool 26, and an Open River Reach on the Upper Mississippi River and La Grange Pool of the Illinois River. In addition, every 10 years LTRM staff collect aerial imagery of the entire Upper Mississippi and Illinois Rivers and their floodplains. Those images are interpreted to create land cover data. The specific indicators monitored through time by LTRM reflect its objective of assessing the general habitat conditions in the river through time and across the river system. The indicators in this report represent a subset of the larger suite of metrics monitored by LTRM that measure a broad range of ecological components and collectively indicate how and where the system has changed. The quantitative assessments within this report provide a long-term and spatially extensive description of how the conditions within a large river system differ across hydrogeomorphic and climatic gradients and change through time.

Status and Trends of the Upper Mississippi River System Indicate a Diverse and Changing River System

Many of the indicators of river ecosystem health changed significantly over the nearly 30 years of our evaluation. However, there was substantial spatial variability in the magnitude and timing of those changes among study reaches. Few indicators changed everywhere or nowhere; most indicators changed in some reaches but not others.

Strong spatial gradients in the underlying physical structure of the river system contributed to the diversity of observed long-term changes among study reaches. Gradients in the Upper Mississippi River System's physical template resulted from the interactions of the underlying geology and geomorphology, navigation infrastructure (for example, locks and dams, wing dams, and closing dams), and prevalence of levees (chs. A and D; De Jager and others, 2018). The differences in geology, geomorphology, navigation infrastructure, and levees resulted in distinct temporal dynamics and feedbacks in different parts of the system, and many of the changes observed over the last two and a half decades reflected those differences.

Widespread Changes

The most widespread change was the increase in discharge that was observed throughout the system. Nearly all hydrology indicators (annual maximum, mean, and minimum discharge; monthly mean discharge; and duration of high flows) showed increasing discharge at all four U.S. Geological Survey streamgages assessed in this report (05378500 [Winona, Minnesota], 05474500 [Keokuk, Iowa], 07010000 [St. Louis, Missouri], and 05586100 [Valley City, Illinois]), and none of the included streamgages showed any evidence of downward trends (figs. ES1 and A1; ch. B). For all four streamgages, the greatest increases in discharge occurred during the summer months (May, June, and July). The number of high-flow days increased significantly through time at three of the four streamgages.

Floodplain forest area decreased except in the Unimpounded Reach where forest area increased (ch. D). Floodplain forests may be responding to several interacting disturbances and environmental changes. First, floodplain tree mortality rates have been shown to increase with increasing flood inundation duration (Yin and others, 2009; De Jager and others, 2019). This indicates that the recent increases in the duration of high flows described in this report could be contributing to the observed monotonic decline in total forest cover from 1989 to 2010 in three of the four floodplain reaches (ch. D; fig. ES1). Another factor contributing to forest loss is the recent arrival of the *Agrilus planipennis* (emerald ash borer), which kills *Fraxinus pennsylvanica* Marsh (green ash trees), one of the more abundant floodplain tree species. Tree mortality resulting from insect outbreaks or flooding may interact with invasive herbaceous species (for example, *Phalaris arundinacea* [reed canarygrass], *Phragmites australis* [common reed], and *Humulus japonicus* [Japanese hops]) to reduce rates of forest regeneration and create lasting forest loss across the system.

Regional Changes

New landform surface area in the form of crevasse deltas, tributary deltas, impounded deltas, and bar-tail limbs has accrued steadily since 1989 in the Upper and Lower Impounded Reaches (Pools 3–26; fig. ES1; ch. C). Most of the observed changes occurred in the Upper Impounded Reach (Pools 3–13). However, net change in floodplain terrestrial area remains unknown because there were not comparable indicators of landform surface area loss from ongoing erosional processes. The overall pattern of increasing bed elevations in backwater lakes in Pool 4 and Pool 8 since 1997 (fig. ES1, ch. C) indicates that many backwater lakes have become shallower, and can be expected to continue to do so, resulting in a decline in the abundance of deep water in off-channel areas. However, some backwater lakes are likely to retain their current approximate depths in areas where diversity of changes within and among backwater lakes results in little net change in bed elevation. The geomorphic changes reported in chapter C affect the amount, distribution, and condition of habitats for aquatic and terrestrial biota.

Ongoing and regional shifts in Upper Mississippi River System fish communities were apparent in several of the fisheries indicators (fig. ES2). Invasive bigheaded carps (*Hypophthalmichthys molitrix* [silver carp] and *Hypophthalmichthys nobilis* [bighead carp]) are greatly affecting Upper Mississippi River System fish communities in part of the Upper Mississippi River System (ch. I). As of 2019, bigheaded carps had not established self-sustaining populations in the upper LTRM study reaches (Pools 4, 8, and 13). In contrast, bigheaded carps were fully established and actively increasing their mass dominance in the lower LTRM study reaches and comprised about 50-percent of the fish community mass in Pool 26 and La Grange Pool and about 20-percent in the Open River study reach. Notably, increases

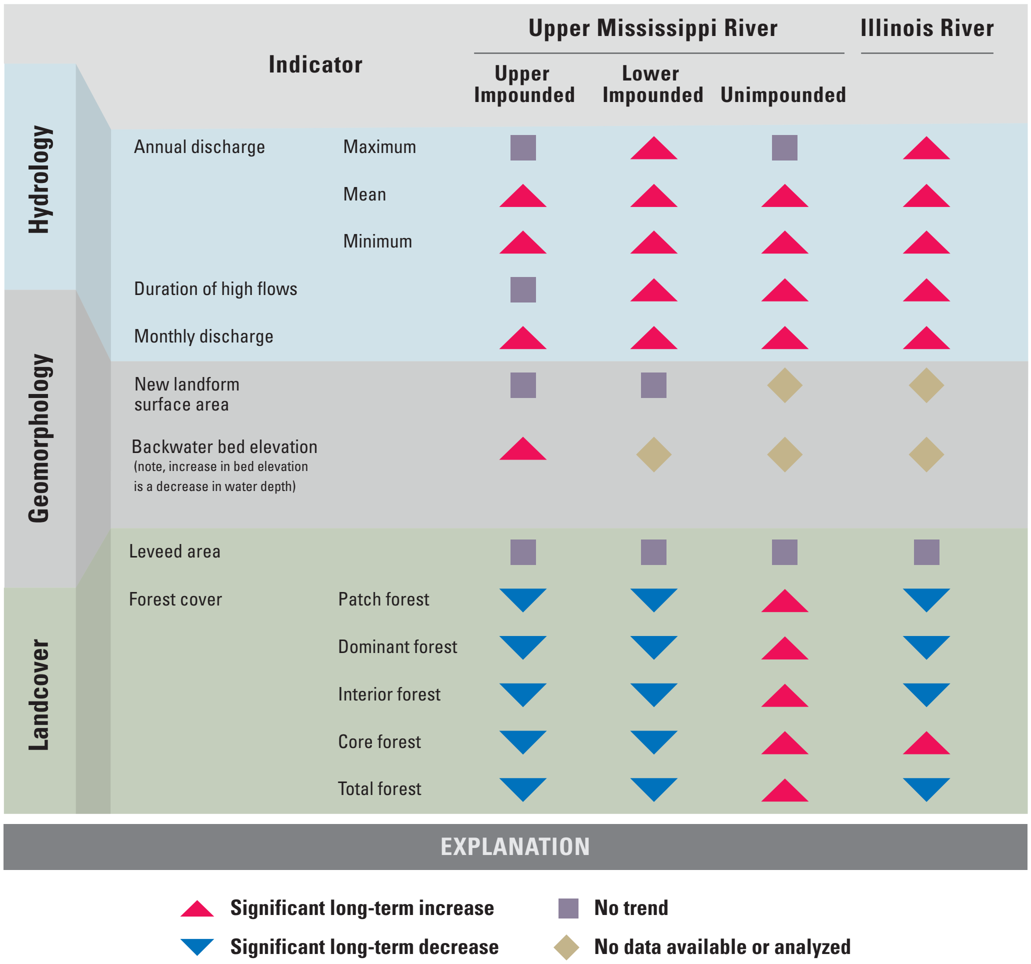

Figure ES1 Summary of long-term changes observed within the Upper Mississippi River System for hydrology (based on data from four U.S. Geological Survey streamgages [U.S. Geological Survey, 2021]—Upper Impounded Reach [Mississippi River at Winona, Minnesota, 05378500; 1940–2019], Lower Impounded Reach [Mississippi River at Keokuk, Iowa, 05474500; 1940–2019], Unimpounded Reach [Mississippi River at St. Louis, Missouri, 07010000; 1960–2019], and Illinois River Reach [Illinois River at Valley City, Illinois, 05586100; 1940–2019], ch. B); geomorphology (based on results from specific research projects; ch. C); and landcover (change in leveed area between circa 1994 and 2014 and change in floodplain forest based on Long Term Resource Monitoring element decadal land cover data from 1989, 2000, and 2010; ch. D).

in bigheaded carp mass were associated with concomitant decreases in native fish mass in Pool 26 and La Grange Pool (chs. G and I).

Regional differences were also apparent for recreationally valued native fishes—a socially and economically important resource (fig. ES2; ch. G). Recreational fishes demonstrated stability (Pool 4) or increases (Pools 8 and 13) in the upper LTRM study reaches, and in the lower study reaches, they demonstrated stability (Open River) or decreases (Pool 26 [approximately a 20-percent decrease over period of record] and La Grange Pool [approximately a 50-percent decline over the period of record]).

Forage fishes, important intermediaries in Upper Mississippi River System food webs, also exhibited notable regional differences (fig. ES2; ch. G). Forage fish declined in 4 of the 6 LTRM study reaches; in the lower three study reaches, pronounced decreases of approximately 50-percent by mass were observed over the period of record.

Submersed aquatic vegetation prevalence increased substantially in Pools 4, Pool 8, and Pool 13 between 1998 and 2010 (fig. ES2; ch. F). However, only in Pool 13, submersed aquatic vegetation prevalence generally decreased between 2011 and 2019. Invasive plants, duckweeds, and filamentous algae dominance decreased in Pools 4, 8, and 13. Emergent aquatic vegetation, especially wild rice, increased substantially in the backwaters and impoundments of Pools 4 and 8 since 2010. Field sampling of aquatic vegetation was limited to Pools 4, 8, and 13, but analyses of land cover data indicated that all aquatic vegetation life forms remain scarce in other reaches of the system (De Jager and Rohweder, 2017).

Regional changes in water quality were significant (fig. ES2; ch. E). Main channel water clarity has increased as indicated by decreases in total suspended solid concentrations (chs. E and H) in all LTRM study reaches except Pool 13. The reductions in suspended solids were most notable in Pools 4 and 8, which exhibited consistent decreases since 1994 in observed and flow-normalized total suspended solids concentrations.

In the main channel of all study reaches, total phosphorus concentration (TP) exceeded US EPA water quality criteria for the entire period of record; however, significant decreases in TP in four of the six study reaches were observed (there was no significant change in the Open River study reach and a significant increase in the La Grange Pool). TP decreases were greatest in Pools 4 and 8, and significant but progressively smaller decreases were observed as far south as Pool 26. Notably, in Pools 4 and 8, TP was substantially above water quality criteria before 2010 but decreased to concentrations near the

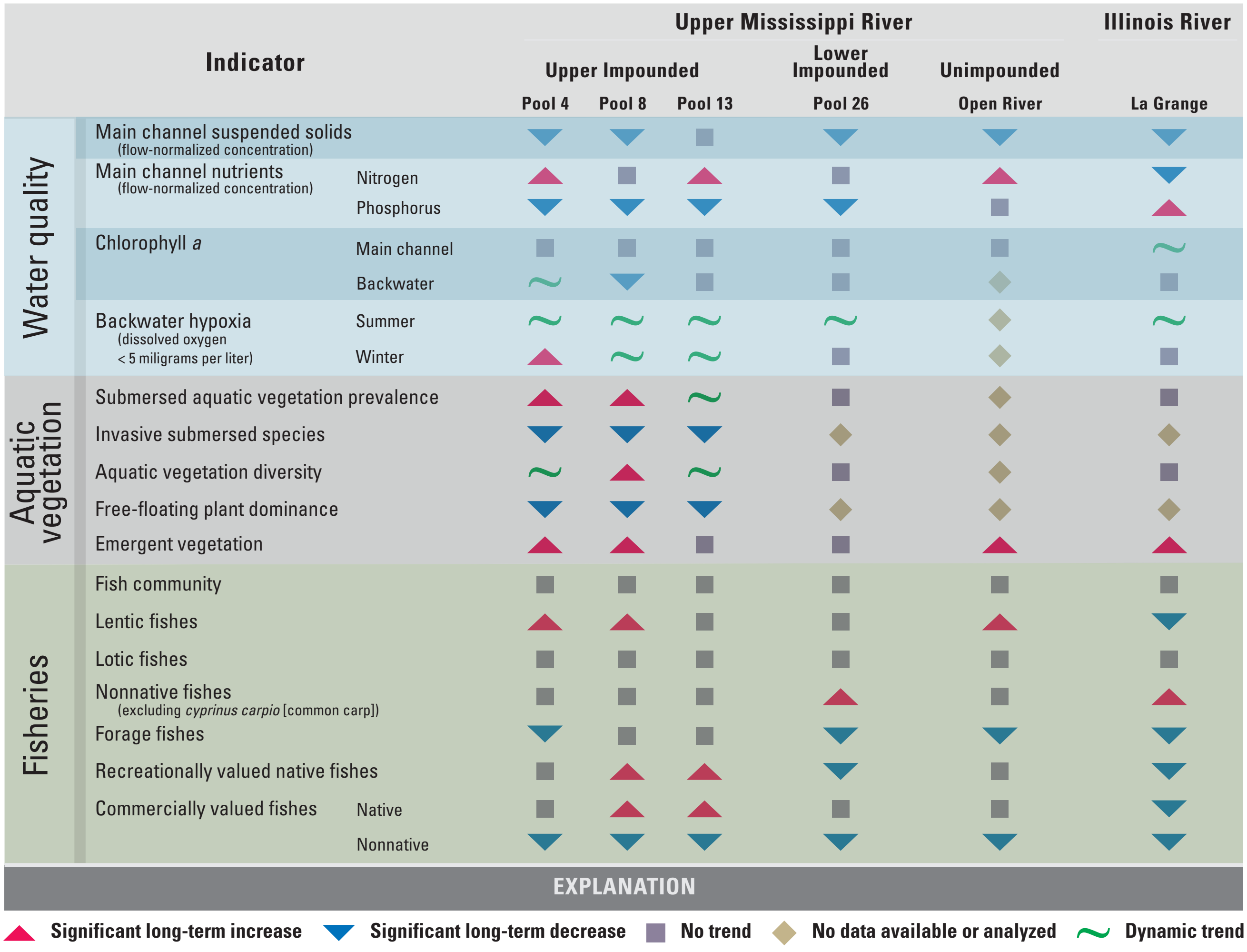

Figure ES2 Summary of long-term changes observed within the Upper Mississippi River System for water quality (ch. E), aquatic vegetation (ch. F), and fisheries (ch. G) based on the Long Term Resource Monitoring element data collected from 1993 to 2019 (Upper Mississippi River Restoration program, 2021). Data summaries include information from the Long Term Resource Monitoring element stratified random sampling and land-cover datasets. Dynamic trend indicates there was no significant, long-term, linear trend, but there was either a clear mid-period minimum or maximum or a significant nonlinear pattern.

water quality criteria during 2011–19. There were at least three possible reasons for these strong declines in TP. First, between 2000 and 2005, the Metropolitan Wastewater Treatment Plant in St. Paul, Minnesota, upstream from the Upper Mississippi River LTRM study reaches, implemented improved water treatment capacity that removes substantially more phosphorus (Metropolitan Council Environmental Services, 2018). Second, TP has decreased in several tributaries (the Chippewa and Cannon Rivers) of Pool 4, likely contributing to the reductions observed in that pool. Last, the reductions in suspended sediment discussed previously likely contributed to some reduction in TP because phosphorus easily attaches to soil particles that are transported to the river during runoff events (for example, Wood, Bormann, and Voigt, 1984).

Total nitrogen was high throughout the period of record in all study reaches. However, there were north-south differences in concentrations and trends. Concentrations were lower in the upper reaches than the lower study reaches. Trends in concentrations and fluxes in the main channel and associated tributaries were upward or neutral in upper study reaches but showed some evidence of total nitrogen concentration decrease in lower study reaches, particularly in the Illinois River Basin (figs. E2 and E3). There were significant trends in total nitrogen in four study reaches, increases in three study reaches (Pools 4, 13, and Open River), and decreases in the La Grange Pool. These spatial differences also match recently observed basin-wide trends in nitrate, which showed increases in upper reaches and northern tributaries and decreases in the Illinois River and southern portions of the Upper Mississippi River System (Crawford and others, 2019).

Stable Indicators

Some of the assessed indicators of ecosystem health showed little change throughout the system. For example, from about 1994 to about 2014, changes in the floodplain area behind levees were small, indicating that the distribution of levees continues to maintain a strong spatial gradient in the physical template of the river system. The Upper Impounded Reach and upper portion of the Illinois River have few levees and maintain a well-connected system of aquatic and floodplain habitats. In contrast, the lower Illinois River and the Lower Impounded Reach and Unimpounded Reaches of the Upper Mississippi River have approximately 50-percent or more of their floodplain behind levees. The enduring presence of levees in the river system places fundamental constraints on the abundance and distribution of aquatic and floodplain habitats and related biota.

Annual fish species richness varied little over time within each LTRM study reach, and even among LTRM study reaches, reflecting no acute or catastrophic faunistic losses over the period of record (fig. ES2; ch. G). High species richness, diversity, and lack of trends in species richness and total annual catches indicate the Upper Mississippi River System fish community remains relatively healthy. Lotic fishes exhibited no significant changes over the period of record, although their abundance is likely strongly suppressed given a century and a half of river modifications that have substantially altered flowing water habitats across the entire Upper Mississippi River System (Carlander, 1954; Shields, 1995; Tripp and others, 2014).

Submersed and emergent aquatic plant prevalence and diversity have remained scarce in the Lower Impounded, Unimpounded, and Illinois River Reaches since at least 1989 when LTRM began plant sampling (fig. ES2; ch. F; De Jager and Rohweder, 2017). Submersed plants are limited by poor water clarity, lack of shallow area, and water level fluctuations in these reaches (Carhart and others, 2022). The restoration of aquatic plant prevalence and diversity remains a high partnership priority in most of these reaches (McCain and others, 2018).

Two aspects of water quality, specifically phytoplankton abundance and the prevalence of backwater hypoxia (dissolved oxygen concentration less than 5 mg/L), were generally stable over the period of record (fig. ES2; ch. E). Over the period of record, phytoplankton abundance (measured as chlorophyll [CHL]) was stable in all study reaches except for the backwaters of Pool 8 (fig. ES2; ch. E). However, decreases in CHL were notable for at least some portion of the period of record in some main channel and backwater areas. Specifically, CHL has decreased in the main channel and backwaters of Pool 4 since circa 2005/2006 and in the main channel of the La Grange study reach since 2002. The prevalence of backwater hypoxia did not show a net change across the entire period of record in five of the six study reaches. The exception was Pool 4 where there was a long-term, significant increase in the prevalence of backwater hypoxia (dissolved oxygen less than 5 mg/L) in winter.

Long-Term Data and River Rehabilitation and Management

Based on nearly three decades of standardized and statistically robust monitoring, analyses of LTRM data have advanced the understanding of the ecological dynamics of the Upper Mississippi River System. Analysis of long-term data is the most effective way to gain insight into long-term ecosystem dynamics, which aids in identifying and designing effective rehabilitation and management actions. The results in this report quantify and describe changes observed mainly from 1993 to 2019 and the insights gained from those observed changes. As more is learned about how the Upper Mississippi River System functions, the approaches used to manage and rehabilitate continue to expand and improve. For example, the recent decreases in water clarity and vegetation abundance in Pool 13 have informed planning of a rehabilitation project intended to increase the distribution and abundance of aquatic vegetation in the impounded area of the pool. Similarly, the long-term submersed vegetation and water clarity data have been combined with other data sources to indicate where physical constraints permit or prevent the establishment of submersed aquatic vegetation (Carhart and others, 2021), providing important context for the location and design of future rehabilitation projects intended to restore aquatic vegetation. The fish community in southern portions of the Upper Mississippi River System is now dominated by invasive bigheaded carps (ch. G); in those areas, analysis of LTRM data has identified a diverse array of effects on the ecosystem that can inform decisions regarding when and where to devote resources to limiting further spread (ch. I). As a final example, although TP decreased in some reaches of the Upper Mississippi River System, nitrogen concentrations were relatively stable and nitrogen and phosphorus concentrations remain high. Thus, the Upper Mississippi River System remains a eutrophic river, and rehabilitation projects would need to work within that constraint in selecting and designing rehabilitation projects.

The results summarized in this report can be used to inform a description of the desired future conditions for Upper Mississippi River System. The quantitative descriptions of the river in this report and in De Jager and others (2018) combined with the qualitative assessment of the condition of the river based on the best professional judgment of natural resource and restoration professionals (McCain and others, 2018) provide a strong foundation for a broad, inclusive discussion to identify those desired future conditions.

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Chapter A: Introduction

By Jeffrey N. Houser, Kristen L. Bouska, Nathan R. De Jager, Brian S. Ickes, Kathi Jo Jankowski, Danelle M. Larson, Molly Van Appledorn, and Jason J. Rohweder

Chapter A of

Ecological Status and Trends of the Upper Mississippi and Illinois Rivers

Edited by Jeffrey N. Houser

In cooperation with the U.S. Army Corps of Engineers

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U.S. Geological Survey

Chapter A: Introduction

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Purpose

This report summarizes the status and trends of selected ecological health indicators of the Upper Mississippi River System derived from the data collected and analyzed by the Long Term Resource Monitoring (LTRM) element of the Upper Mississippi River Restoration (UMRR) program. This assessment is intended for a broad audience, including those interested in habitat rehabilitation, public policy, ecology, and a general understanding of the river system. The report addresses four objectives:

- provides a brief introduction to the Upper Mississippi River System including its significance, history, modern-day stressors, and recent research;
- uses ecological indicators to describe the status of the river system and where and how it has changed from circa 1993 to 2019;
- discusses management and restoration implications of these changes; and
- highlights the fundamental role of long-term monitoring in the understanding, management, and restoration of large-floodplain rivers.

Introduction to the Upper Mississippi River System

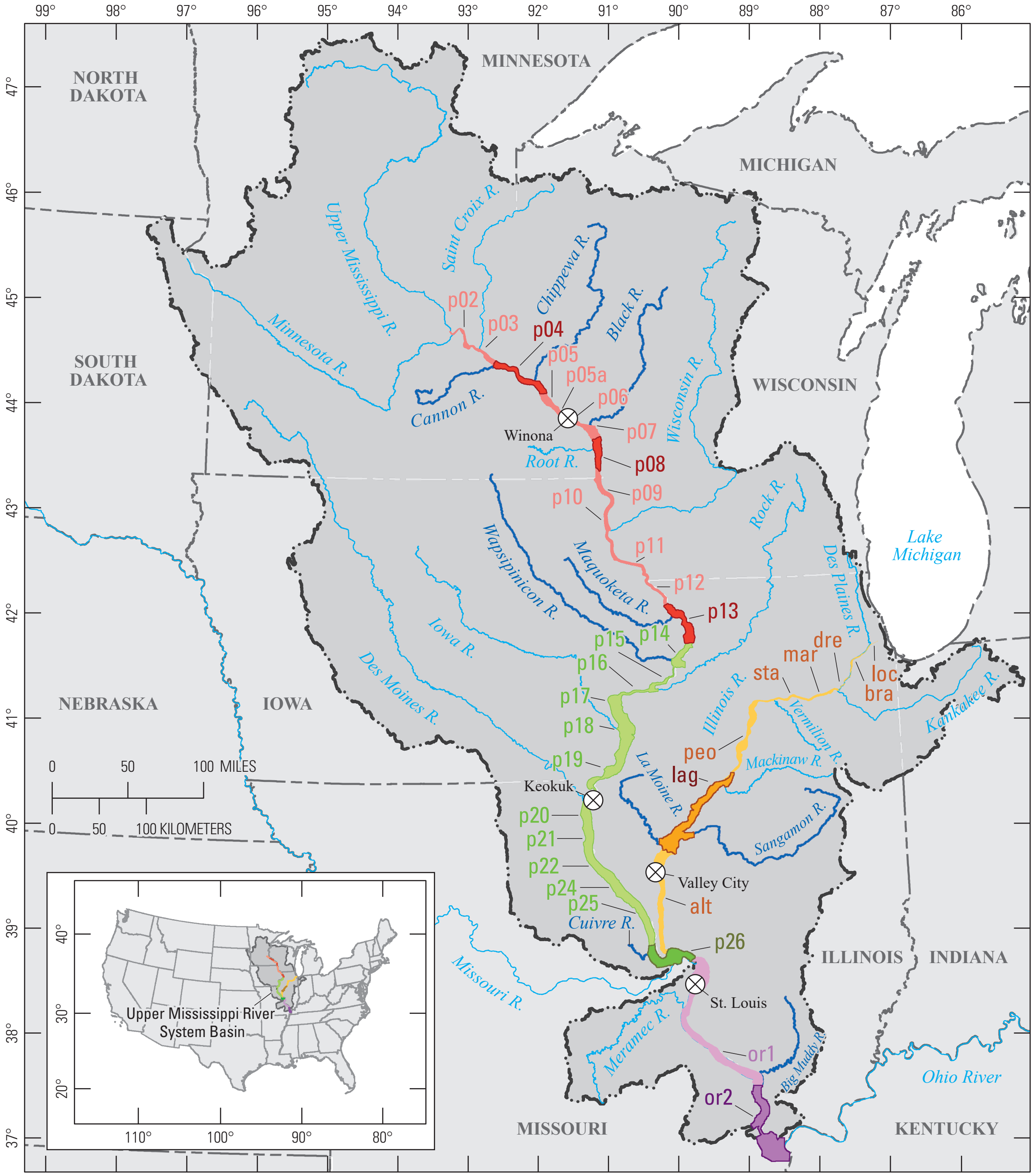
The Upper Mississippi River System is congressionally defined as the commercially navigable portions of the Mississippi River main stem north of Cairo, Illinois, and its commercially navigable tributaries, including the entire Illinois River (Water Resources Development Act of 1986, 33 U.S. Code, section 652); its basin covers 500,000 square kilometers of North America (fig. A1). Laterally, the Upper Mississippi River System includes the rivers and their floodplains and encompasses a range of riverine and floodplain habitats.

The Ecosystem

The Upper Mississippi River System reflects the spatial and temporal complexity inherent in large floodplain-river ecosystems. A useful conceptual simplification of this river system, and floodplain rivers generally, is to consider them as three interacting subsystems: lotic channels, lentic off-channel areas (backwaters and impounded areas), and floodplains (Bouska and others, 2018; fig. A2). The lotic subsystem includes areas dominated by water flow and transport and provides habitat for organisms adapted to flowing water. The lentic subsystem includes the lake-like aquatic areas with little or no current such as backwaters, floodplain lakes, wetlands, and impounded areas upstream from locks and dams. These lentic areas provide habitat for organisms adapted to still-water conditions; in some river reaches these areas are characterized by abundant aquatic vegetation and associated organisms. Floodplains include areas that experience intermittent inundation and are characterized by a diversity of trees, shrubs, forbs, and grasses.

Major resources are ecosystem components that contribute to popular uses of, and ecosystem services provided by, the river ecosystem (Bouska and others, 2018; fig. A3). For this report, major resources include “Water Quality” (ch. E), “Aquatic Vegetation” (ch. F), and “Fisheries” (ch. G). For each major resource, select ecological indicators are used to assess the status and trends of the resource and are the focus of this report. The large-scale drivers that affect major resources that are included in this report are “Hydrology” (ch. B), “Geomorphology” (ch. C), and “Floodplain Land Cover” (ch. D). External drivers are factors that affect major resources but have been previously assessed by others and are not assessed in this report, for example watershed land use (Turner and Rabalais, 2003; Zhang and Schilling, 2006; Alexander and others, 2008), climate (Naz and others, 2016; Rajib and Merwade, 2017), and navigation infrastructure (Alexander and others, 2012; WEST Consultants Inc., 2000).

The Upper Mississippi River System consists of four major floodplain reaches: Upper Impounded Reach, the Lower Impounded Reach, the Unimpounded Reach, and the Illinois River Reach (fig. A1). These reaches are geomorphically distinct in terms of extent and frequency of floodplain inundation, sedimentation rates, hydrologic regimes, and the distribution of agricultural levees (Theiling and Nestler,



Base from U.S. Geological Survey digital data, 1:1,000,000, 2014, Universal Transverse Mercator projection, zone 15, North American Datum of 1983

EXPLANATION

[Darker colors indicate long-term study areas within each floodplain reach]

Upper Mississippi River System

- Upper Mississippi River, Upper Impounded Reach
- Upper Mississippi River, Lower Impounded Reach
- Upper Mississippi River, Unimpounded Reach
- Illinois River Reach
- **Boundary of the Upper Mississippi River Basin**
- **State boundaries**
- **Study tributaries**
- **Other tributaries**
- **Streamgages**

Figure A1. The Upper Mississippi River System. The four floodplain reaches are identified by color. Navigation pools on the Upper Mississippi River are numbered, except for the two Open River reaches (or1 and or2) and those on the Illinois River, which are named (Lockport [loc], Brandon Road [bra], Dresden Island [dre], Marseilles [mar], Starved Rock [sta], Peoria [peo], La Grange [lag], Alton Pool, [alt]). The Long Term Resource Monitoring element study reaches are identified in each floodplain reach by the darker colors (Upper Mississippi River study reaches—Pool 4 [p04], Pool 8 [p08], Pool 13 [p13], Pool 26 [p26]; Open River Reach [or2]; and Illinois River study reach—La Grange Pool [lag]). Tributaries from which data are included in this report are indicated by the darker blue color; light blue tributaries are included for context. The four U.S. Geological Survey streamgages shown are those from which data are included in this report: 05378500 (Winona, Minnesota), 05474500 (Keokuk, Iowa), 07010000 (St. Louis, Missouri), and 05586100 (Valley City, Illinois).

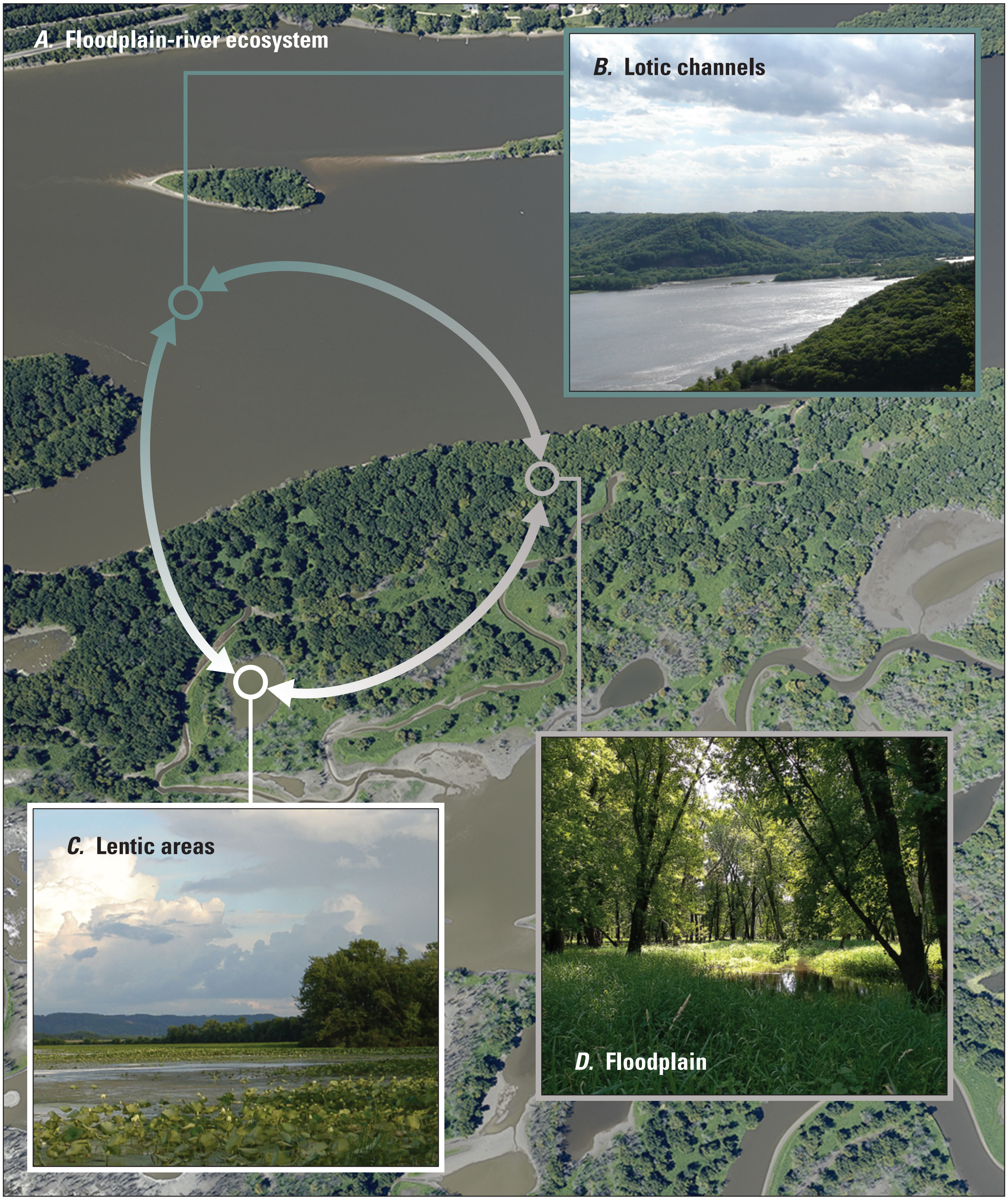


Figure A2. Photographs and a conceptual diagram of the three interacting subsystems that compose the Upper Mississippi River System. This river system can be described as consisting of the following three interacting subsystems: lotic channels (characterized by flowing water), lentic backwater lakes and impounded areas (characterized by little or no water current), and floodplains (characterized by intermittent inundation). Connectivity and exchange between these subsystems are critical to the structure and function of large floodplain rivers. Photographs by the U.S. Geological Survey.

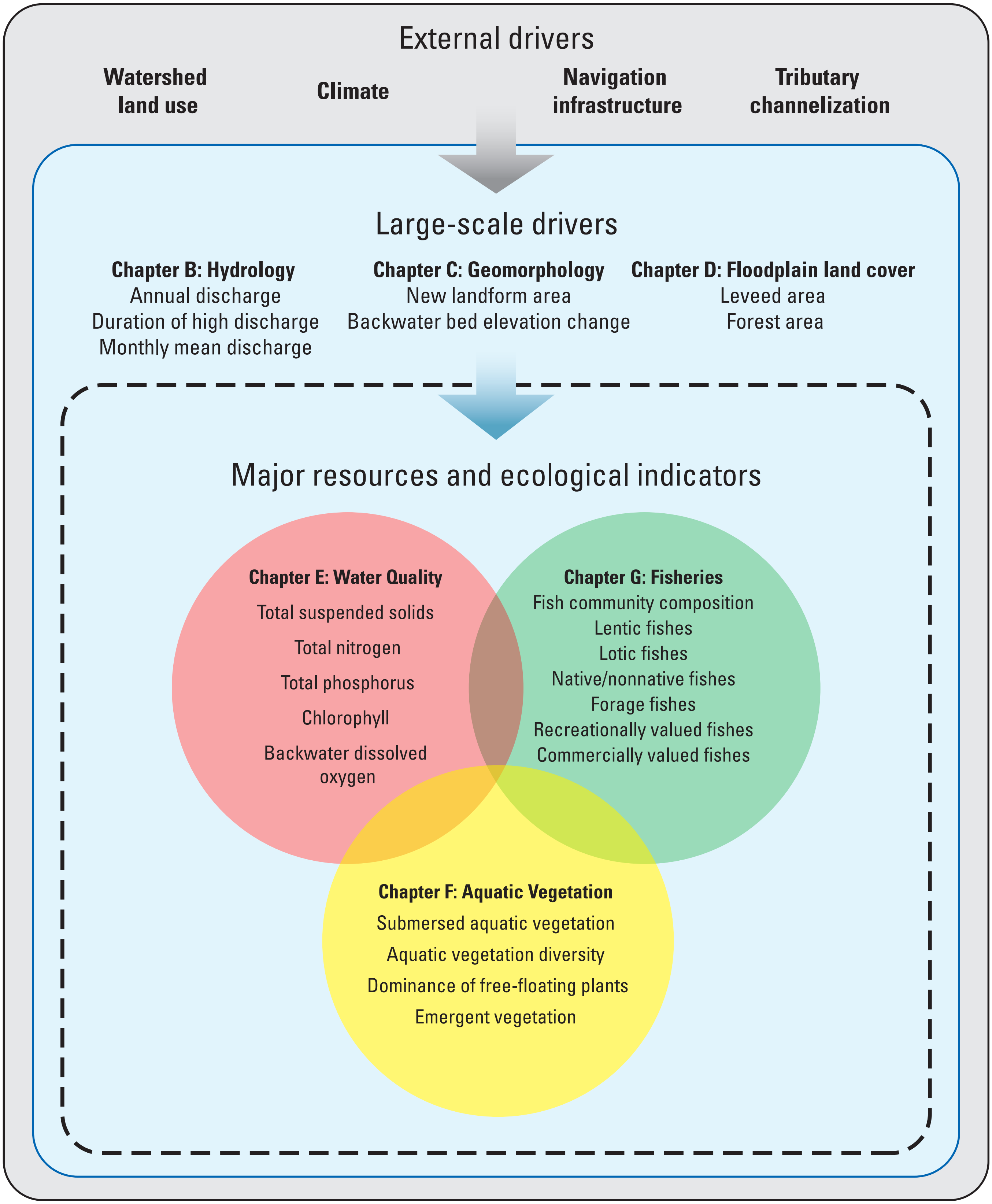


Figure A3. Conceptual diagram of the external drivers, large-scale drivers (chs. B, C, and D), and major resources and ecological indicators of the Upper Mississippi River System. Major resources are ecosystem components that contribute to popular uses of, and ecosystem services provided by, the river ecosystem and include “Water Quality” (ch. E), “Aquatic Vegetation” (ch. F), and “Fisheries” (ch. G). For each major resource, select ecological indicators (listed below each major resource) collectively are used to assess the status and trends of the resource and are the focus of this report. The large-scale drivers that affect major resources that are included in this report are hydrology, geomorphology, and floodplain land cover. External drivers are factors that affect major resources but have been previously assessed by others and are not assessed in this report (see text under “The Ecosystem” for additional detail).

2010; De Jager and others, 2018). The Upper Impounded Reach extends from Minneapolis, Minnesota, through Navigation Pool 13. This reach is characterized by abundant side channels and islands, extensive impounded areas, few levees (and therefore a floodplain that remains well-connected to the river), and public ownership of a substantial part of the floodplain, which is managed for fish and wildlife conservation. The Lower Impounded Reach extends from Navigation Pool 14 through Navigation Pool 26. In contrast to the Upper Impounded Reach, this reach generally lacks large, impounded areas, has fewer backwaters and floodplain lakes, and is generally dominated by channel environments. The Unimpounded Reach runs from Lock and Dam 26 to its confluence with the Ohio River near Cairo, Illinois. This reach contains the largest amount of floodplain behind levees, has no impounded areas, and few backwater lakes. The Illinois River reach encompasses a steeper section upstream from the Peoria Pool that is highly affected by urban development and a lower gradient downstream section that is primarily affected by agricultural land uses. The downstream section, including the Peoria, La Grange, and Alton Pools, receives high inputs of sediment, and levees are widespread.

Significance of the Ecosystem

The Upper Mississippi River System is a globally important ecosystem that is recognized by the U.S. Congress as a nationally significant transportation system and a nationally significant river ecosystem (Water Resources Development Act of 1986, 33 U.S.C., section 652). The ecological value of this river system is reflected in the diversity of biota the river-floodplain system supports, the extensive network of wildlife and fish refuges, and the public conservation lands along the river system and its floodplain. This river system is home to more than 140 species of fish, at least 25 species of submersed and floating-leaf aquatic vegetation, at least 57 species of mammals, and at least 45 species of amphibians (U.S. Fish and Wildlife Service [FWS], 2006; Garvey and others, 2010; Moore and others, 2010). The Upper Mississippi River System is a critical migratory corridor for as much as 40-percent of North American waterfowl and provides habitat for as much as 50-percent of the world's population of *Aythya valisineria* (canvasback duck) (FWS, 2006; Beatty and others, 2015). This river system contains a diverse and abundant assemblage of native freshwater mussels; of the 50 mussel species in the historical record, about 40 species have been found live in recent years (Upper Mississippi River Conservation Committee, 2015). The floodplain forests of the Upper Mississippi River provide nesting and migratory habitat for as many as 180 bird species, and bird abundances along the river can be double that of adjacent upland areas (Knutson and others, 1996). About 300,000 acres of the Upper Mississippi River are included on the Ramsar List of Wetlands of International Importance (Hawkins, 2008). The Upper Mississippi River contains six national wildlife refuges that span 300,000 acres of water, wetland, and floodplain. The Upper Mississippi River National Wildlife and Fish Refuge was established in 1924 and spans approximately 420 linear kilometers of the Upper Mississippi River (FWS, 2006). Other Upper Mississippi River national wildlife refuges include the Trempealeau, Port Louisa, Great River, Clarence Cannon, and the Middle Mississippi (not shown). The Illinois River contains the Chautauqua, Emiquon, Meredoisa, and Two Rivers National Wildlife Refuges. The States of Illinois, Iowa, Minnesota, Missouri, and Wisconsin manage more than 140,000 acres as wildlife management areas, conservation areas, and State parks at more than 80 sites along the Upper Mississippi River System (U.S. Army Corps of Engineers [USACE], 2004).

The Upper Mississippi River System supports a variety of human activities. River-related economic activities along the Upper Mississippi River generate nearly $350 billion annually (FWS, 2015). The river system is a critical commercial navigation corridor and supplies public drinking water for several major cities, and is a popular destination for fishing, hunting, wildlife viewing, boating, and photography. For example, during 2012, the Upper Mississippi River National Wildlife and Fish Refuge received approximately 1.5 million visitors annually who spent a mean of $61 USD (nonlocal visitors) and $22 USD (local visitors) in the local communities during their visit (Dietsch and others, 2012).

Historical Context

Humans have valued and used the Upper Mississippi River System for millennia, but it has been fundamentally modified since the arrival of Europeans in the basin (Anfinson, 2005; fig. A4). Much of the basin has been converted to row-crop agriculture beginning in the mid-1800s (Turner and Rabalais, 2003). Changes associated with the cultivation of agricultural crops include extensive wetland drainage, widespread installation of tile drainage, levee construction, and tributary channelization (McCorvie and Lant, 1993; Thompson, 2002; Turner and Rabalais, 2003; Schilling, 2005; Zhang and Shilling, 2006). Collectively, these changes have increased water, nutrient, and sediment input to the river system and altered the hydrology of the river system and its catchments (Knox, 2001; Turner and Rabalais, 2003; Houser and Richardson, 2010; Rajib and Merwade, 2017).

The potential economic value of the river system as a navigation corridor was recognized early in European settlement of the basin. The cumulative effects of river modification to support navigation have fundamentally changed the Illinois and Upper Mississippi Rivers. Modifications for navigation began in the mid-1800s (fig. A4). The initial changes included clearing snags (downed trees) and blasting to remove shallow rapids within the navigation channel (Thompson, 2002; Anfinson, 2005). As a result of a series of Federal authorizations to create a progressively deeper navigation channel (4 foot [ft], 4.5 ft, and then 6 ft), extensive dike construction and dredging were used to maintain a navigable channel. The 1930 authorization of the 9-ft navigation channel transformed the free-flowing rivers into a series of connected riverine

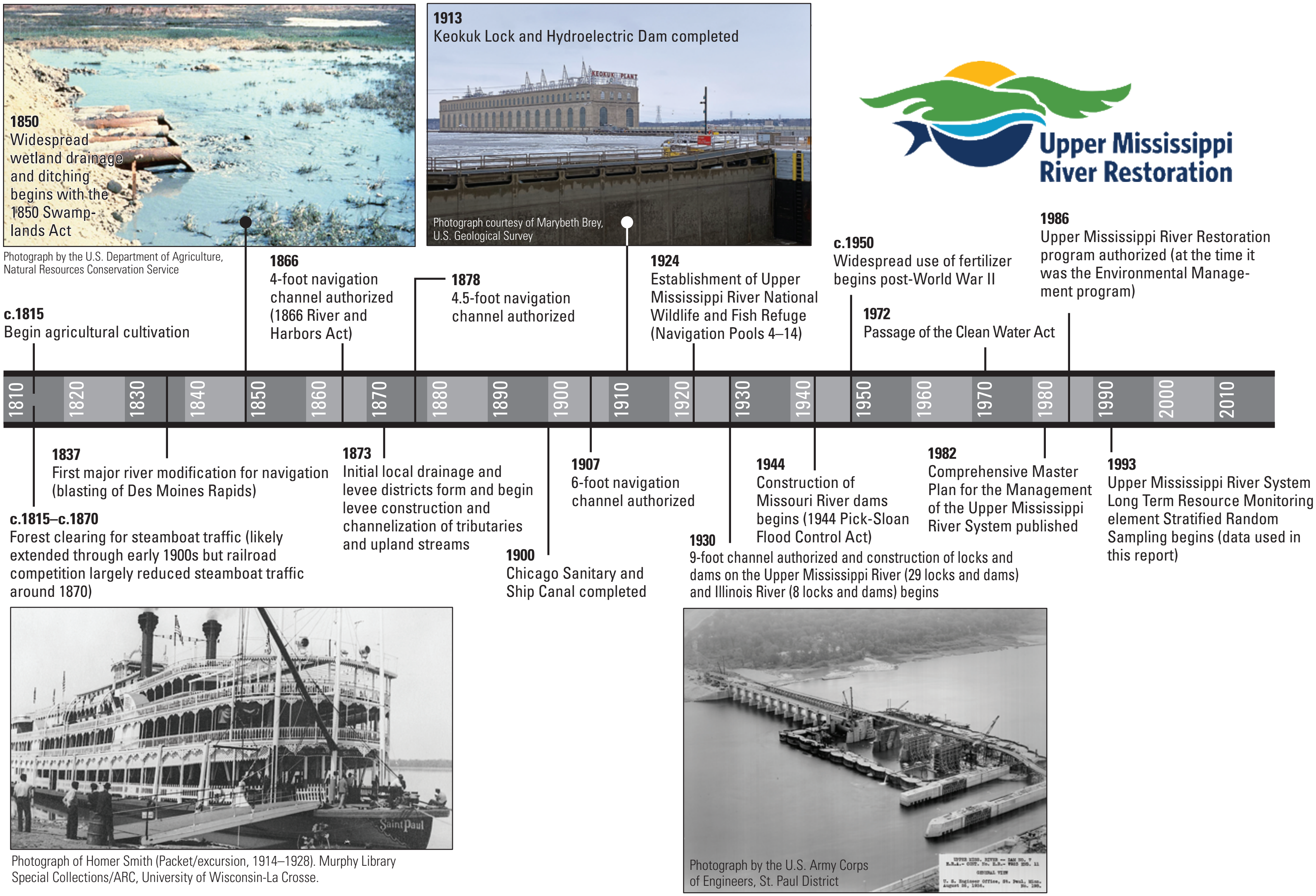


Figure A4. A timeline of the major events that have shaped the Upper Mississippi River System and the Upper Mississippi River Restoration program.

impoundments through the completion of 29 locks and dams on the Upper Mississippi River and 8 locks and dams on the Illinois River. On the Upper Mississippi River, these dams inundated 630 square kilometers of floodplain between Minneapolis, Minn., and Alton, Ill. (for example, fig. A5*A* and *B*; Fremling, 2005; Theiling and Nestler, 2010) and created large differences in the distribution of aquatic areas among the different floodplain reaches. Downstream from the last Upper Mississippi River lock and dam near Alton, Ill., the navigation channel has been channelized by hundreds of river training structures and revetments (Dobney, 1977), mostly constructed in support of the 9-ft channel authorization beginning in 1930. In contrast to the expansive lake-like environments created by impoundment in the Upper and Lower Impounded reaches, the Unimpounded Reach has experienced channel incision (because of the focusing of flows within the main channel), which in combination with agricultural levees, has reduced river-floodplain connectivity (for example, fig. A5*C* and *D*).

On the Illinois River, the combined effects of the navigation dams, diversion of water from Lake Michigan, and watershed drainage substantially increased permanent water-surface area, and many shallow marshes became large, permanent lakes (Theiling and others, 1996). Completion of the Chicago Sanitary and Ship Canal in 1900 reversed the flow of the Chicago River to carry Chicago's sewage away from Lake Michigan and into the Des Plaines River and then the Illinois River (Karr and others, 1985). The Chicago Sanitary and Ship Canal also made commercial navigation possible from the Mississippi River to Lake Michigan by way of the Illinois River. The canal continues to divert water from the Lake Michigan watershed to the Illinois River and facilitates spread of nonnative species between the Great Lakes and the Illinois and Mississippi Rivers.

Challenges Facing the River

Most of the challenges facing the Upper Mississippi River System reflect the combined impacts of historical and ongoing changes to the rivers and their watersheds (fig. A4; Upper Mississippi River Basin Commission, 1982; U.S. Geological Survey [USGS], 1999; Johnson and Hagerty, 2008; Bouska and others, 2018). These challenges include increased frequency and magnitude of floods, altered geomorphic processes and rates, floodplain forest loss, altered river-floodplain connectivity, high rates of nutrient and sediment input, scarce aquatic vegetation in some reaches, and invasive species.

The hydrologic and hydraulic regimes of the Upper Mississippi and Illinois Rivers have been affected by watershed land use, navigation infrastructure, levees, and changing climate (Sparks and others, 1998; Collins and Knox, 2003; Remo and others, 2008). Annual patterns in river flow were once characterized by a single spring flood pulse that, once the river overflowed its banks, was attenuated by expansive floodplains, low flows in the summer, and a minor increase in flows in the fall (Sparks and others, 1998). The addition of low-head navigation dams in the impounded reaches of the Upper Mississippi and Illinois Rivers purposefully increased water levels at low discharges to maintain adequate depth for navigation and permanently inundated substantial portions of the floodplains. Downstream from the Missouri River confluence, river training structures have simplified the river into a narrow, deep, fast-flowing main channel with highly variable water levels. Changes in watershed land use (for example, agricultural cultivation, installation of drain tile, increases in impervious surface area) and precipitation regimes over the past century increased discharge and changed seasonal discharge patterns in the rivers (Pielke and others, 2002; Collins and Knox, 2003; Zhang and Schilling, 2006; Schilling and others, 2010; Caldwell and others, 2012; Frans and others, 2013). Extensive levees have reduced the capacity of the floodplain to attenuate flows in some areas. The net result has been the concentration of the spring flood pulse into a shorter period with quicker water level rises, greater peak water levels, and more rapid flood recession compared to historical conditions (Sparks and others, 1998).

The physical structure of the Upper Mississippi River has been substantially changed by the construction of locks and dams, channel training structures, and agricultural levees. Navigation dams in the Upper Impounded Reach (fig. A4) created extensive permanent aquatic areas over parts of the river that previously experienced seasonal inundation (De Jager and others, 2018). As a result, this reach supports the greatest amount of lake-like aquatic habitats: approximately 20-percent of the aquatic area of this reach is in impounded areas, 28-percent is in contiguous (connected to the main channel) floodplain lakes, and 13-percent is in contiguous shallow aquatic areas (De Jager and others, 2018). The Lower Impounded Reach has been impacted by navigation dams and an extensive levee system. Approximately 50-percent of the total aquatic area of this reach is occupied by the main channel and its border, and another 15-percent is occupied by side channels (De Jager and others, 2018). Similarly, the structure of the Illinois River reflects a history of dam construction and levees. Contiguous floodplain lakes make up approximately 47-percent of the aquatic area of the lower three pools in this reach.

In all impounded areas of the Upper Mississippi and Illinois Rivers, elimination of low summer water levels, island loss through erosion, increased unconsolidated sediments, sediment resuspension, and sediment accumulation since installation of the locks and dams have affected river biota, such as aquatic vegetation (Peck and Smart, 1986; Fischer and Claflin, 1995). High sedimentation rates in the impounded areas of the river system have been a concern since the lock and dams were completed (Upper Mississippi River Basin Commission, 1982; Anfinson, 2005; Fremling, 2005). Sedimentation reduces backwater depth, and deeper backwaters are less prone to hypoxia (for example, Houser and others, 2013); deep backwaters provide overwintering habitat for some popular game fishes because they are more likely to stratify and contain warmer water (for example, Gutreuter, 2004;

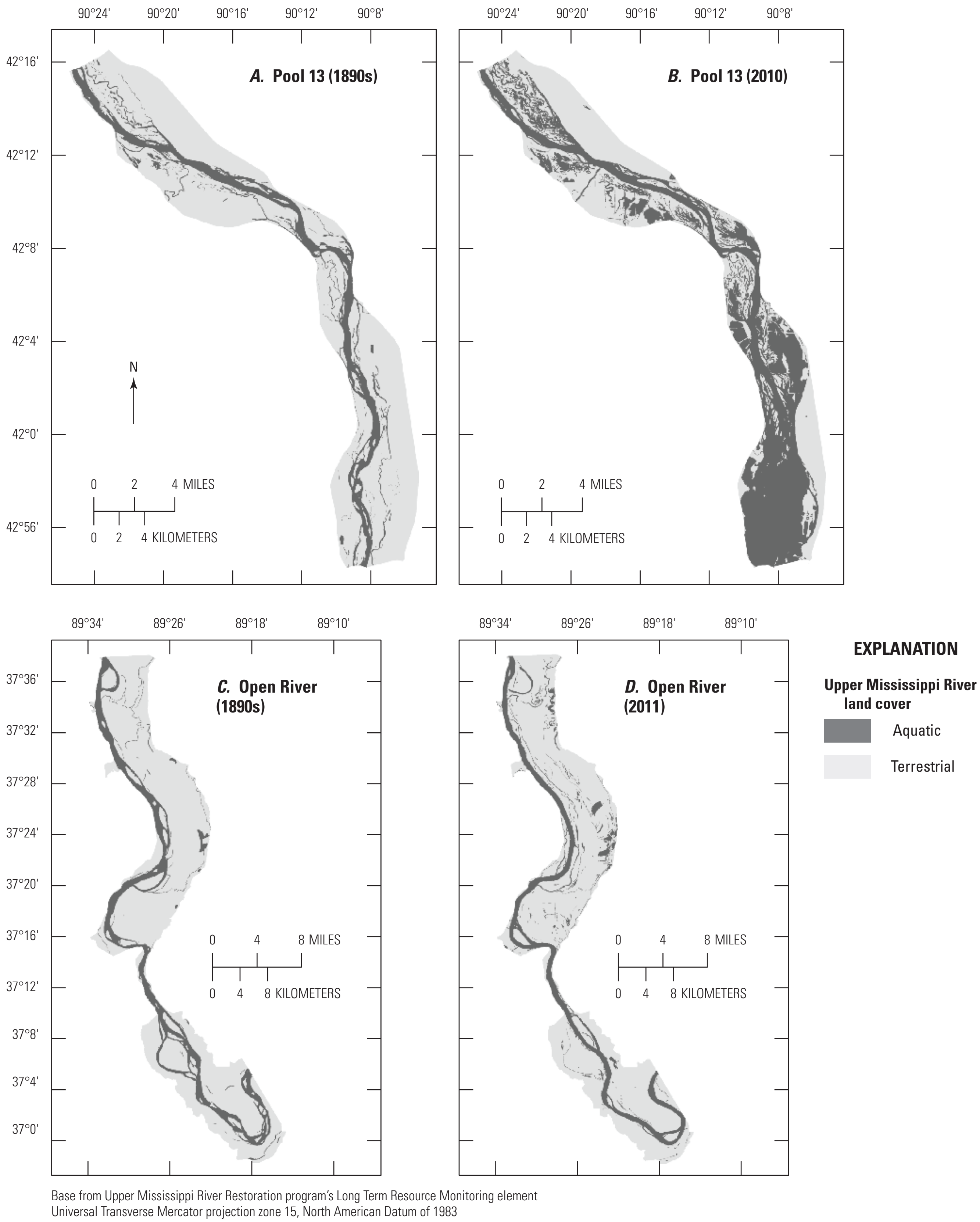


Figure A5. Maps showing two contrasting study reaches illustrating the effects of navigation modification on the Upper Mississippi River. Study reach Pool 13, *A* and *B*, is an example of how the locks and dams substantially altered aquatic area abundance and distribution on the floodplain in the Upper Impounded Reach. The Open River study reach, *C* and *D*, shows the effects of channelization on side-channel abundance in the Unimpounded Reach.

Bodensteiner and Lewis, 1992). As a result, deep backwaters are generally sought for fishing and other recreational uses. Sedimentation has been particularly problematic in the lower Illinois River, where it has led to the loss of productive aquatic habitats and continues to adversely affect biotic communities (USACE, 2006). Channelization of the Upper Mississippi River downstream from the Missouri River confluence in the Unimpounded Reach requires continuous construction of river training structures (for example, dikes and revetments) and regular sediment dredging to ensure navigable conditions (USACE, 2017).

Floodplain forests have been affected by the changes in channel morphology and hydraulics in different ways in different parts of the system. In the Upper Impounded Reach, vast areas of floodplain forests were replaced by shallow lake environments (De Jager and others, 2013), creating a modern-day pattern of highly fragmented floodplain forests (De Jager and Rohweder, 2011). In the Lower Impounded and Unimpounded Reaches, the expansive levee systems and associated agricultural areas have likewise resulted in a reduction in total floodplain forest area (De Jager and others, 2013). The Illinois River floodplain was similarly affected by lock and dam construction, levees, and agricultural land conversion. Systemwide, changes in the frequency and duration of floodplain inundation within forested areas continue to affect tree longevity; areas that experience longer periods of inundation are more prone to tree mortality and forest loss (De Jager and others, 2019). Ongoing changes in forest species composition have been attributed to the large changes in water levels caused by the locks and dams (Knutson and others, 1996), large magnitude floods (Yin, 1998), varying patterns of inundation across the floodplain (De Jager and others, 2012), and invasive species (Thomsen and others, 2012).

Water quality has improved substantially since the early to mid-20th century, particularly after the passage of Federal legislation to address degrading water quality conditions (1972 Clean Water Act; Thompson, 2002; Fremling, 2005; Wiener and Sandheinrich, 2010). High rates of suspended sediment inputs remain a longstanding concern for parts of the river system because associated sediment accumulation in off-channel areas and reduced water clarity (McCain and others, 2018; chs. C and H). Nutrient (in other words, nitrogen and phosphorus) concentrations are generally high in the Upper Mississippi and Illinois Rivers (ch. E; Houser and Richardson, 2010; Crawford and others, 2019; U.S. Environmental Protection Agency [EPA], 2017). High nutrient concentrations combined with suitable environmental conditions can lead to harmful algal blooms (HABs) in aquatic systems. Although spatially extensive HABs have not yet been reported on the Upper Mississippi River, algal species capable of producing toxins are common (Manier, 2014; Decker and others, 2015; Giblin and Gerrish, 2020), and localized HABs have been reported as problems in off-channel areas. During summer 2020, the Illinois River experienced a substantial HAB event. Other consequences of high nutrient concentrations are extensive mats of free-floating plants (for example, filamentous algae and duckweeds; ch. F; Houser and others, 2013; Giblin and others, 2014). Excessive filamentous algae may impair submersed plant growth and inhibit its re-establishment (Irfanullah and Moss, 2004), which can reduce habitat availability for fishes and macroinvertebrates (Camp and others, 2014). In addition, high abundance of free-floating plants creates aesthetic and recreational impairments and has been associated with reduced dissolved oxygen concentration in parts of the Upper Mississippi River (Houser and others, 2013; Giblin and others, 2014).

Over time, the distribution and abundance of aquatic vegetation have been affected by long-term changes in hydrogeomorphic and water quality conditions directly and through ecological and geomorphic feedbacks (Peck and Smart, 1986; Bhowmik and Adams, 1989; Fischer and Claflin, 1995; Sparks and others, 1998; Drake and others, 2018; Carhart and De Jager, 2019). For example, the expansive shallow areas and stable water levels in the Upper Impounded Reach support abundant submersed aquatic vegetation. Aquatic vegetation remains scarce or absent in the Lower Impounded, Unimpounded, and Illinois River reaches, likely because of several causes (for example, water level fluctuations, high turbidity, and few suitable off-channel areas; chs. F and H; Moore and others, 2010; De Jager and Rohweder, 2017; Bouska and others, 2020).

Nonnative species, such as *Phalaris arundinacea* L. (reed canary grass), *Humulus japonicus* (Japanese hops), *Cyprinus carpio* (common carp), and *Hypophthalmichthys* spp.(bigheaded carps), continue to affect the Upper Mississippi and Illinois Rivers (chs. G and I; Irons and others, 2009; Solomon and others, 2016; Weeks and others, 2017; DeBoer and others, 2018; FWS, 2019). Invasive species often flourish where native ecological communities decline or human-modified conditions facilitate niche opportunities (Shea and Chesson, 2002). In turn, invasive species can alter ecological interactions and facilitate long-lasting changes to abiotic conditions and biotic communities (Havel and others, 2015; Bouska and others, 2020). As a result, management agencies are challenged with balancing invasive species control with efforts to build invasion resistance in native communities.

The Upper Mississippi River Restoration Program

The 1986 Water Resources Development Act created the Environmental Management Program to monitor and rehabilitate the Upper Mississippi River System because of the system's recognized value and ongoing stressors. The program was subsequently renamed the Upper Mississippi River Restoration (UMRR) program and consists of two elements: Habitat Rehabilitation and Enhancement Projects and Long Term Resource Monitoring (LTRM). When authorized by Congress, the UMRR became the first Federal program to combine ecosystem restoration with scientific monitoring and research on a large river system. The UMRR is administered by the USACE and implemented through a broad partnership composed of multiple Federal and State agencies including USACE, FWS, USGS, Natural Resources Conservation Service, EPA, U.S. Maritime Administration, the Upper Mississippi Basin Association and the five Upper Mississippi River System States: Illinois, Iowa, Minnesota, Missouri, and Wisconsin. Other Federal and State agencies and nonprofit organizations associated with environmental protection, agriculture, and transportation are actively involved and instrumental in the success of the UMRR.

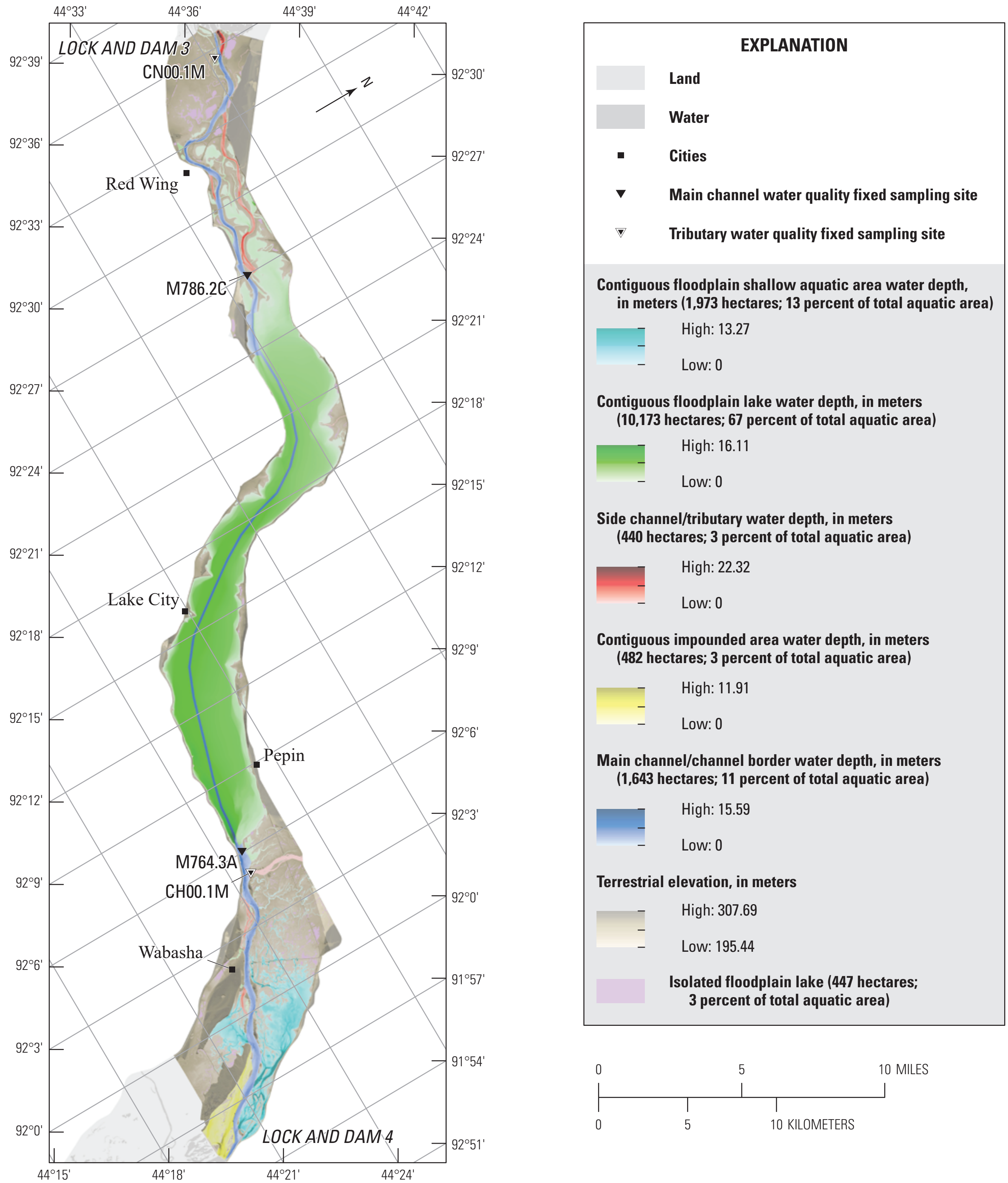
Habitat Rehabilitation and Enhancement Projects

Habitat Rehabilitation and Enhancement Projects generally modify geomorphic characteristics of a river to rehabilitate degraded ecosystem function, structure, and dynamic processes (USACE, 2012). USACE leads the planning, design, and management of Habitat Rehabilitation and Enhancement Projects in close coordination with State and Federal agency partners. The program and its partners have restored more than 112,000 acres of aquatic and floodplain habitat. Often, the proximate goal of these restoration projects is to alter or restore hydrogeomorphic processes, and the ultimate goal is to rehabilitate physical, chemical, and biological conditions to restore and maintain a healthier and more resilient ecosystem (for example, fig. ES1 in USACE, 2011; UMRR, 2015). Many of these restoration projects address slow, ongoing changes in the geomorphology of the river system: island loss and secondary channel formation and expansion, which cause increased hydraulic connectivity, sediment deposition, and floodplain forest degradation. These projects have included island rehabilitation and protection, modifications of hydraulic connections between the main channel and floodplains, dredging backwaters to remove accumulated sediments, side channel modifications, and floodplain forest restoration (USACE, 2012; Theiling and others, 2015).

Long Term Resource Monitoring

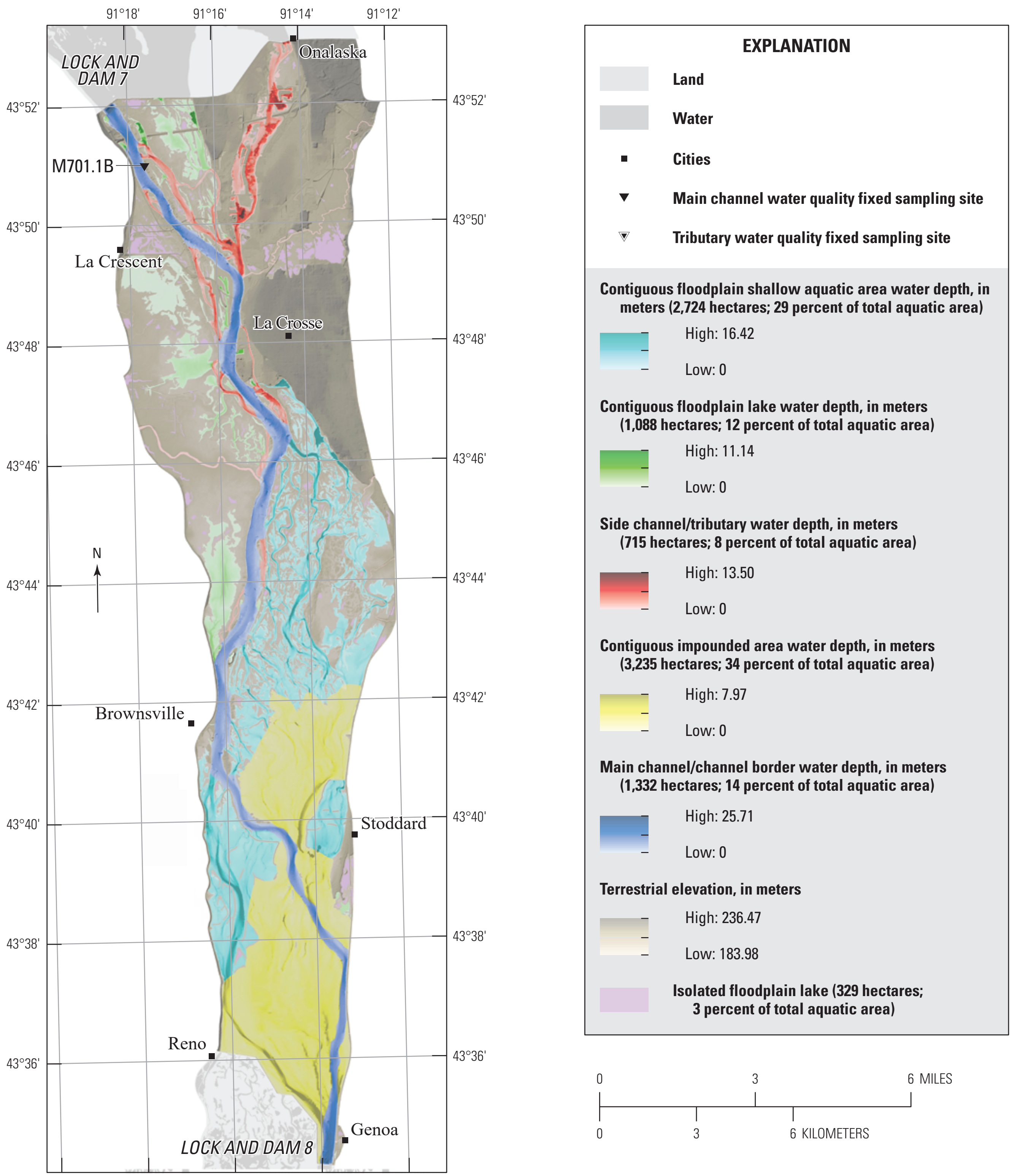
The need for coordinated, systemic monitoring was initially described in the 1982 Comprehensive Master Plan for the Upper Mississippi River System, which noted that an acute lack of information about the river system and inadequate communication among agencies was hindering effective river restoration and management (Upper Mississippi River Basin Commission, 1982). Therefore, the report recommended a large-scale and ongoing ecosystem-monitoring effort to inform successful restoration of the river system. The LTRM addresses the critical need for the standardized collection, integration, analysis, and reporting of scientific information to resource managers and decision makers. The LTRM serves as the foundation for river restoration by detecting patterns and trends, determining associations and cause-effect relations among the drivers and major resources (fig. A3), establishing benchmarks of current conditions for future reference, providing early warnings of change, and informing evaluations of the effectiveness of restoration and management actions. The USGS Upper Midwest Environmental Sciences Center provides scientific leadership for the implementation of the LTRM and analysis and interpretation of the data. Natural resource agency personnel from the five States bordering the Upper Mississippi and Illinois Rivers (Minnesota, Wisconsin, Iowa, Illinois, and Missouri) perform field data collection and contribute their expertise to analysis, interpretation, and publication of the long-term data.

The ecological conditions, biotic communities, and capacity of the Upper Mississippi River System to cope with environmental disturbance vary among floodplain reaches, reflecting the interplay among geomorphic variability and ecological interactions and dynamics (De Jager and others, 2018; Bouska and others, 2019). The LTRM sampling design accounts for this geomorphic and biological variation within study reaches using a stratified random sampling design to allocate sampling effort among different aquatic area types (in other words, strata). Furthermore, the six LTRM study reaches span the entire Upper Mississippi River System and the various gradients therein. The LTRM study reaches within the Upper Mississippi River include Navigation Pools 4, 8, 13, and 26 and the portion of the Unimpounded Reach between Grand Tower, Ill., and Cairo, Ill. (hereafter the Open River study reach). The LTRM study reach on the Illinois River is the La Grange Pool (fig. A1). Navigation Pools 4 (fig. A6), 8 (fig. A7), and 13 (fig. A8) are within the Upper Impounded Reach (fig. A1). Pool 4 is approximately 70 kilometers (km) long and encompasses Lake Pepin, a natural riverine lake (fig. A6). Pool 8, near La Crosse, Wisconsin, is approximately 37 km long (fig. A7). Pool 13, near Bellevue, Iowa, is 45 km long (fig. A8). Pools 8 and 13 have similar physical structure; the upper third of the pools is characterized by braided channels, the lower third is a large, open impounded area, and the middle third represents a transition between the upper and lower thirds with extensive braided channels and contiguous backwater areas. Pool 26 (fig. A9) is at the southern end of the lower impounded reach, is approximately 60 km long, and contains the confluence of the Illinois River. Pool 26 is dominated by main and side channel aquatic areas, scarce contiguous floodplain backwaters, and a much smaller impounded area compared to the Upper Impounded study reaches. The Open River study reach spans approximately 80 km of the Unimpounded Reach and is dominated by the main river channel (fig. A10); contiguous floodplain backwaters are mostly absent from this study reach. The La Grange Pool of the Illinois River spans approximately 120 km and contains a mix of main channel, side channel, and contiguous and isolated backwater aquatic areas (fig. A11). Data are collected for the LTRM each year from each of these six study reaches for water quality (ch. E), aquatic vegetation (ch. F), and fisheries (ch. G) using a stratified random sampling design (augmented in the water quality and fisheries components by



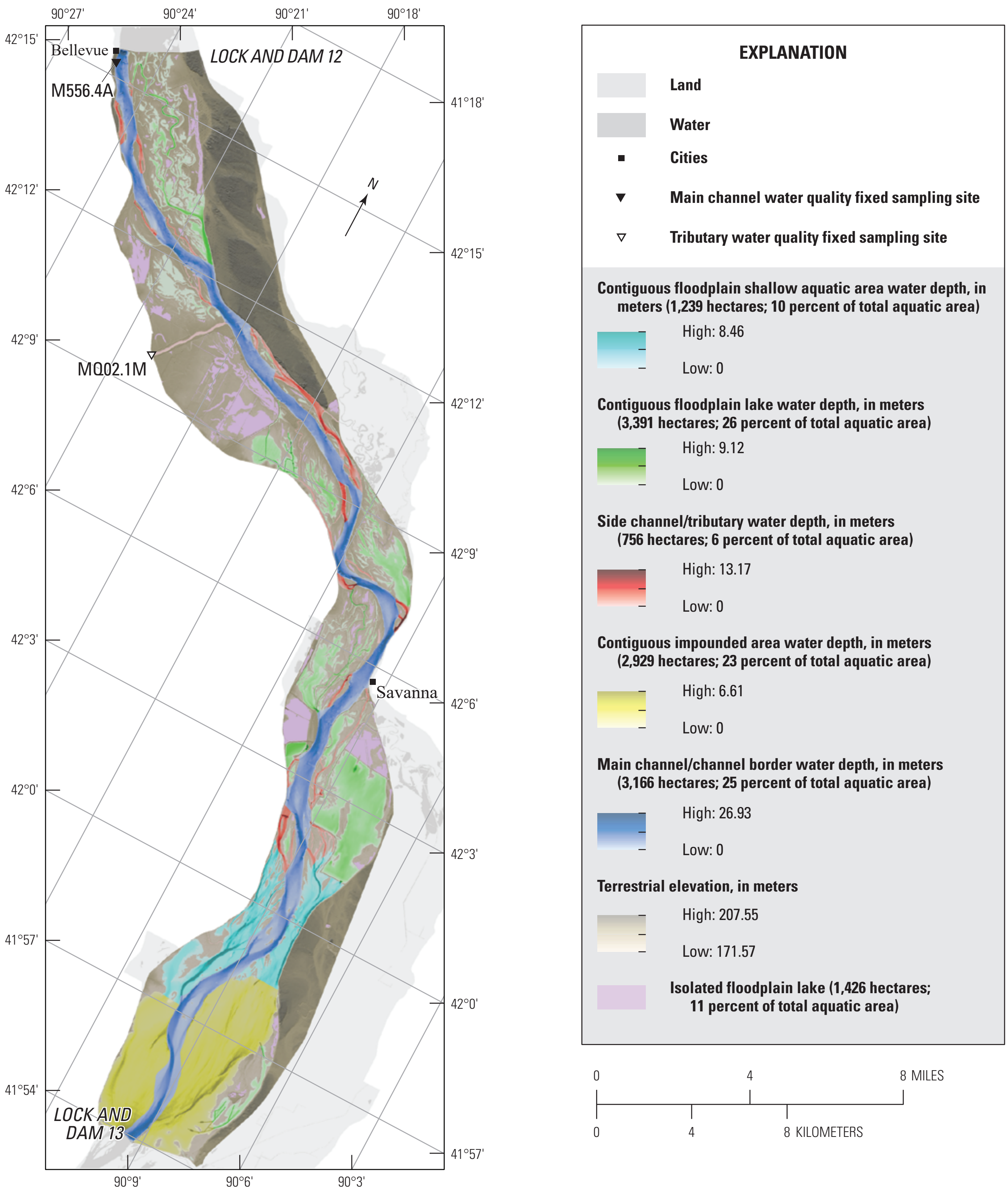
Cities from U.S. Geological Survey digital data, 1:1,000,000, 2014
Elevation, water depth, and aquatic area types from Upper Mississippi River Restoration program's Long Term Resource Monitoring element
Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88)
Universal Transverse Mercator projection, zone 15, North American Datum of 1983

Figure A6. Map of Pool 4 study reach (Upper Impounded Reach of the Upper Mississippi River) showing aquatic areas (De Jager and others, 2018), bathymetry, floodplain elevation, and locations of water quality fixed sites. The water quality fixed sites in Pool 4 include the main channel (M786.2C; M764.3A), the Cannon River (CN00.1M), and the Chippewa River (CH00.1M). Note that no depth data are available for isolated floodplain lakes, so no color ramp is used for that aquatic area. Occasional missing depth data in the other aquatic areas are classified as the lower extreme of the color ramps. Lake Pepin is shown in green (contiguous floodplain lake). Note the orientation of the compass arrow indicating "North."



Cities from U.S. Geological Survey digital data, 1:1,000,000, 2014
Elevation, water depth, and aquatic area types from Upper Mississippi River Restoration program's Long Term Resource Monitoring element
Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88)
Universal Transverse Mercator projection, zone 15, North American Datum of 1983

Figure A7. Map of Pool 8 study reach (Upper Impounded Reach of the Upper Mississippi River) showing aquatic areas (De Jager and others, 2018), bathymetry, floodplain elevation, and locations of water quality fixed sites. The water quality fixed site in Pool 8 is in the main channel (M701.1B). Note that no depth data are available for isolated floodplain lakes, so no color ramp is used for that aquatic area. Occasional missing depth data in the other aquatic areas are classified as the lower extreme of the color ramps. Note the orientation of the compass arrow indicating "North."



Cities from U.S. Geological Survey digital data, 1:1,000,000, 2014
Elevation, water depth, and aquatic area types from Upper Mississippi River Restoration program's Long Term Resource Monitoring element
Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88)
Universal Transverse Mercator projection, zone 15, North American Datum of 1983

Figure A8. Map of Pool 13 study reach (Upper Impounded Reach of the Upper Mississippi River) showing aquatic areas (De Jager and others, 2018), bathymetry, floodplain elevation, and locations of water quality fixed sites. Water quality fixed sites in Pool 13 include the main channel (M556.4A) and the Maquoketa River (MQ02.1M). Note that no depth data are available for isolated floodplain lakes, so no color ramp is used for that aquatic area. Occasional missing depth data in the other aquatic areas are classified as the lower extreme of the color ramps. Note the orientation of the compass arrow indicating "North."

a complementary set of fixed sites). A detailed description of the study design and methods are on the LTRM website and in additional program reports and methods manuals (Gutreuter and others, 1995; Yin and others, 2000; Soballe and Fischer, 2004; Ickes and others, 2014b; Ratcliff and others, 2014). All the LTRM monitoring data, graphical summary tools, and spatial maps are on the LTRM website (https://umesc.usgs.gov/ltrm-home.html).

Land cover data also are collected for the entire Upper Mississippi River System floodplain at decadal intervals. In this report, those LTRM data are used to assess long-term changes in levees, floodplain forest area (ch. D), and emergent aquatic vegetation (ch. F). Details regarding the methods used to collect and process the land cover data are described by Dieck and others (2015); land cover data layers can be downloaded in a variety of formats (https://umesc.usgs.gov/data_library/land_cover_use/land_cover_use_data.html) and many layers viewed using the Systemic Spatial Data Viewer (https://umesc.usgs.gov/management/dss/umrs_land_cover_viewer.html).

Learning About the Upper Mississippi River System from Long-Term Data

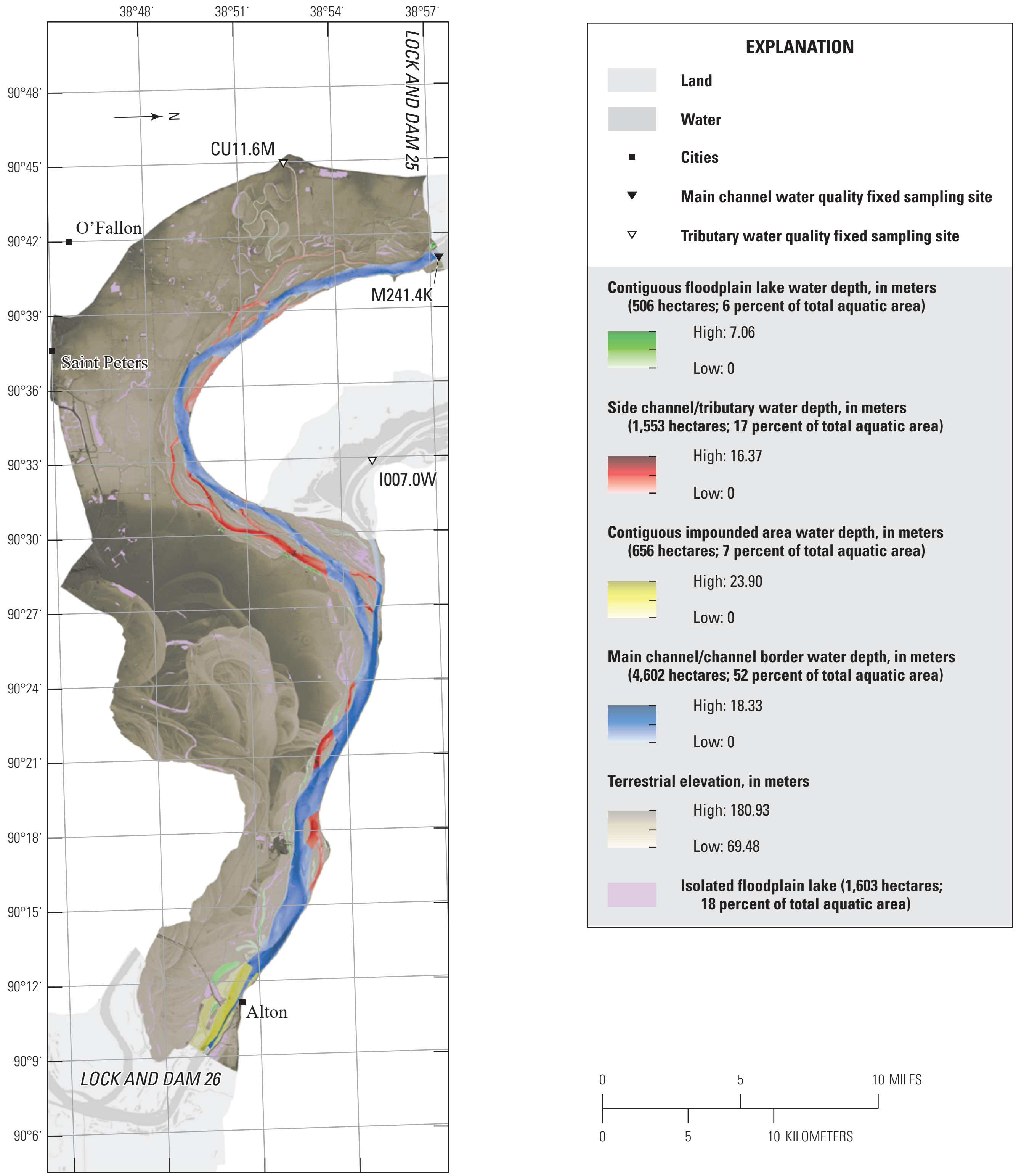
The LTRM data have provided critical information about long-term changes in the habitat conditions of the river system and its floodplain through their use in diverse research projects and previous status and trends assessments. Research using LTRM data has contributed substantially to understanding the structure and function of the Upper Mississippi River System. This research has emphasized questions regarding the causes and consequences of spatial and temporal patterns in the water quality, aquatic and floodplain vegetation, and fisheries. Here we provide a brief overview of findings produced by these studies, which often combined LTRM data with complementary data collected for study-specific purposes.

Longitudinal, lateral, and temporal patterns in water quality reflect differences in climate, land use, catchment inputs, and river hydrogeomorphology that exist across the system. There are longitudinal gradients in nutrients (nitrogen and phosphorus), silica, phytoplankton abundance, and total suspended solids (and therefore water clarity; Houser and others, 2010; Crawford and others, 2016; Carey and others, 2019; ch. E). Wetland and floodplain areas retain and remove sediments and nutrients (Richardson and others, 2004; Houser and Richardson, 2010; Kreiling and others, 2013), but the retained nitrogen has generally been a small proportion of the total nitrogen in transport, highlighting the important role of watershed-based management actions in nutrient reduction (for example, Strauss and others, 2011; Loken and others, 2018). Phytoplankton (algae suspended in the water column) biomass is generally high throughout the river system, reflecting its eutrophic condition, but there are longitudinal patterns in the abundance and community composition of phytoplankton in the river system (Houser and others, 2010; Manier, 2014; Decker and others, 2015; Crawford and others, 2016; Giblin and Gerrish, 2020).

Lateral patterns (across a gradient of conditions from the main channel to impounded areas and floodplain lakes) in nutrients, total suspended solids, and algal abundance and production are affected by the interaction of discharge, the physical structure of the rivers, and the resulting temporal and spatial variation in hydraulic connectivity of off-channel areas (De Jager and Houser, 2012; Houser and others, 2013; Giblin and others, 2014; Manier, 2014; Decker and others, 2015; Houser, 2016; Sobotka and Phelps, 2017; De Jager and others, 2018; Carey and others, 2019; Giblin and Gerrish, 2020). Phytoplankton production and community composition in the system are generally more affected by physical factors such as discharge, turbidity, and residence time than by nutrients (Kimber and others, 1995; Manier, 2014; Decker and others, 2015; Houser and others, 2015), although nutrient limitation may sometimes occur in isolated backwaters (Sullivan, 2008). Higher discharge (and therefore hydraulic connectivity) often leads to more similar phytoplankton communities among main channel and backwater environments (Manier, 2014).

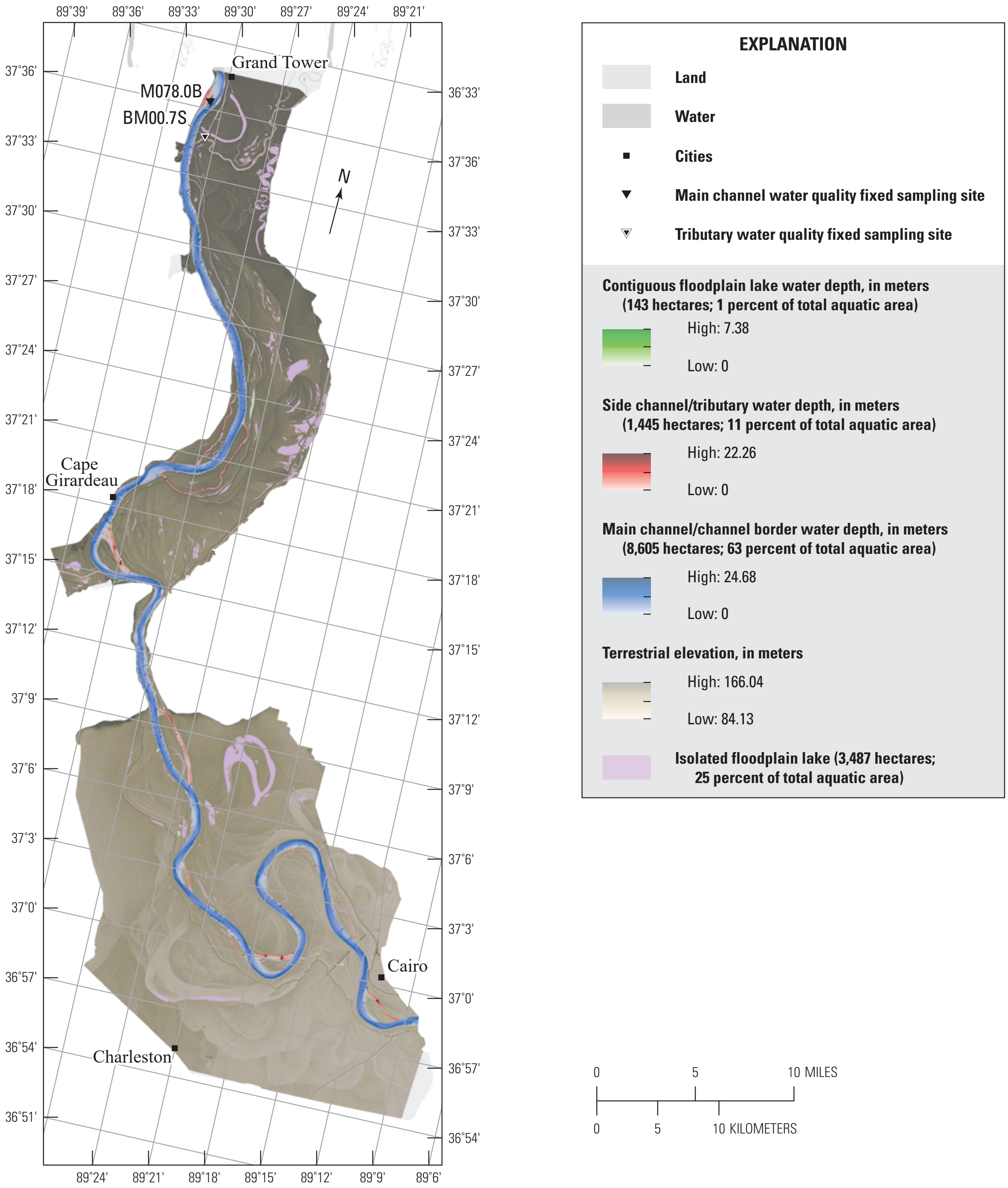
The LTRM vegetation data have been used to address research questions at multiple scales and inform restoration and management. Aquatic vegetation is abundant in the Upper Impounded Reach but scarce or absent in the other three reaches (De Jager and Rohweder, 2017). Models that incorporated hydrology, turbidity, and aquatic vegetation datasets showed three primary limitations for submersed plants depending on pool and reach: lack of shallow water for plant rooting, high turbidity that reduces light availability for photosynthesis, and extreme water level fluctuations (De Jager and others, 2018; Carhart and others, 2021). Study results have improved the LTRM sampling methods and estimates of plant biomass. Results of these studies have indicated how vegetation abundance in the Upper Mississippi River System compares to that of other large rivers, and can inform waterfowl management (for example, Yin and Kreiling, 2011; Schmidt, 2021). Studies in Pool 8 indicate that some of the changes in aquatic vegetation abundance and diversity are associated with the islands constructed as part of Habitat Restoration and Enhancement Projects (Langrehr and others, 2007; Drake and others, 2018; Carhart and De Jager, 2019). Ecological community analyses have characterized biodiversity and many distinct vegetation community types throughout the Upper Mississippi River System's three subsystems (fig. A2; De Jager and Rohweder, 2017; Bouska and others, 2022), and have noted community composition changes during a period of increasing aquatic vegetation within the Upper Impounded Reach that began in the early 2000s (Bouska and others, 2022).

Since the last status and trends publication (Johnson and Hagerty, 2008), the LTRM fisheries component has engaged in a diversity of efforts to improve our understanding of the Upper Mississippi River System fisheries. This includes international collaboration and the transfer of information and methods through The Nature Conservancy's Great River



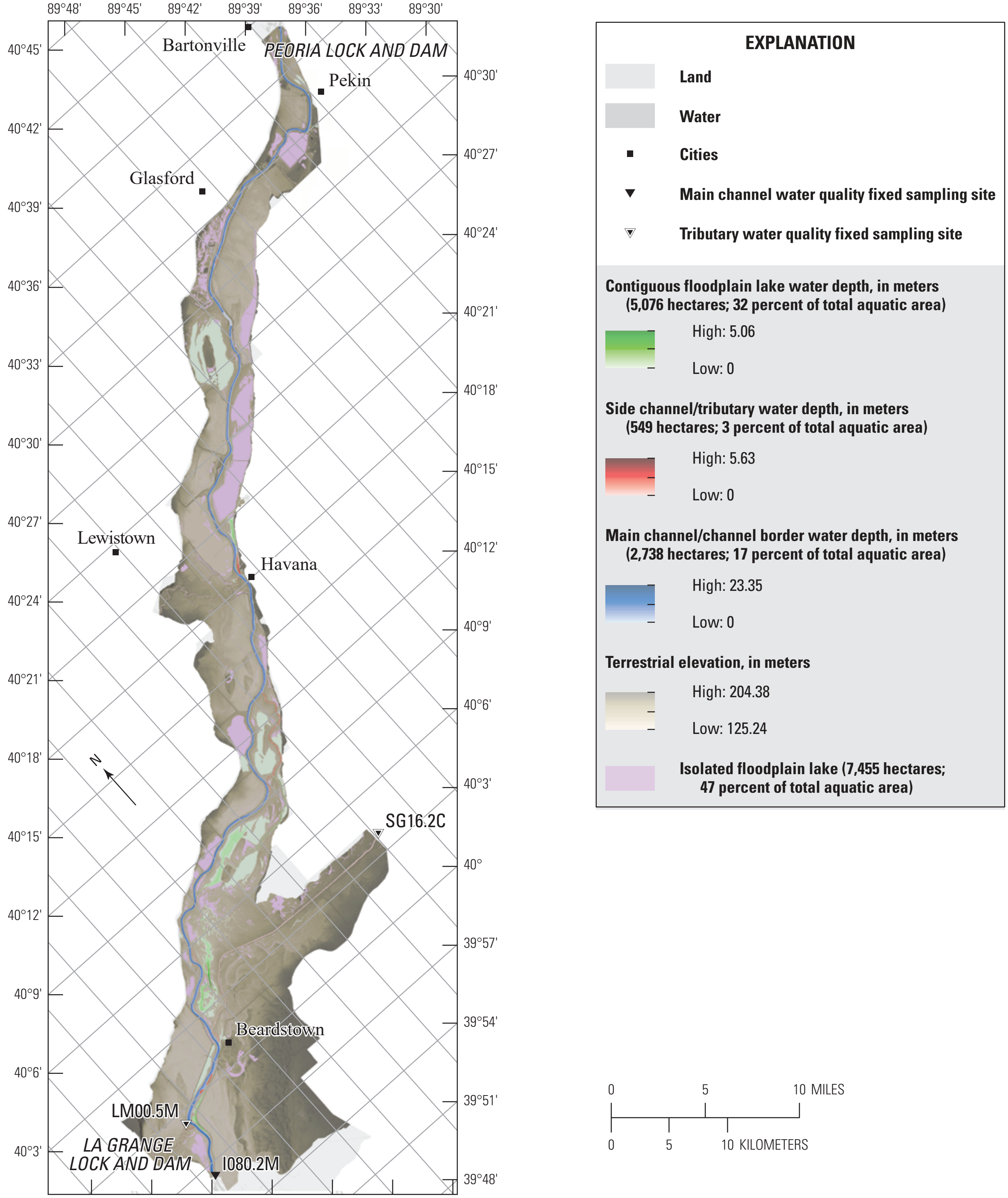
Cities from U.S. Geological Survey digital data, 1:1,000,000, 2014
Elevation, water depth, and aquatic area types from Upper Mississippi River Restoration program's Long Term Resource Monitoring element
Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88)
Universal Transverse Mercator projection, zone 15, North American Datum of 1983

Figure A9. Map of Pool 26 study reach (Lower Impounded Reach of the Upper Mississippi River) showing aquatic areas (De Jager and others, 2018), bathymetry, floodplain elevation, and locations of water quality fixed sites. Water quality fixed sites in Pool 26 include the main channel (M241.4K) and the Illinois River (I007.0W; the Illinois River and floodplain above the confluence with the Upper Mississippi River are shown in light gray). Note that no depth data are available for isolated floodplain lakes, so no color ramp is used for that aquatic area. Occasional missing depth data in the other aquatic areas are classified as the lower extreme of the color ramps. Note the orientation of the arrow indicating "North."



Cities from U.S. Geological Survey digital data, 1:1,000,000, 2014
Elevation, water depth, and aquatic area types from Upper Mississippi River Restoration program's Long Term Resource Monitoring element
Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88)
Universal Transverse Mercator projection, zone 15, North American Datum of 1983

Figure A10. Map of the Open River study reach (Unimpounded Reach of the Upper Mississippi River) showing aquatic areas (De Jager and others, 2018), bathymetry, floodplain elevation, and locations of water quality fixed sites. Water quality fixed sites in the Open River study reach include the main channel (M078.0B) and the Big Muddy River (BM00.7S). Note that no depth data are available for isolated floodplain lakes, so no color ramp is used for that aquatic area. Occasional missing depth data in the other aquatic areas are classified as the lower extreme of the color ramps. Note the orientation of the arrow indicating "North."



Cities from U.S. Geological Survey digital data, 1:1,000,000, 2014
Elevation, water depth, and aquatic area types from Upper Mississippi River Restoration program's Long Term Resource Monitoring element
Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88)
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Figure A11. Map of La Grange study reach (Illinois River) showing aquatic areas (De Jager and others, 2018), bathymetry, floodplain elevation, and locations of water quality fixed sites. Water quality fixed sites in La Grange Pool include the main channel (I080.2M), the Sangamon River (SG16.2C), and the La Moine River (LM00.5M). Note that no depth data are available for isolated floodplain lakes, so no color ramp is used for that aquatic area. Occasional missing depth data in the other aquatic areas are classified as the lower extreme of the color ramps. Note the orientation of the arrow indicating "North."

EcoPartnership with the Chinese Academy of Fisheries Science and the Yangtze River Fisheries. Nationally, LTRM data and expertise have been used in comparative studies spanning multiple large river basins (Ward and others, 2017; Counihan and others, 2018). Schramm and Ickes (2016) expanded the description of fisheries resources, habitat, and issues across the entirety of the full Mississippi River from its headwaters to the Gulf of Mexico. Within the river system and more regionally, Garvey and others (2010) identified challenges in merging fisheries research and management in these unique, dynamic, and large ecosystems. LTRM has advanced notably in scaling its analytics, research, and data serving capabilities beyond local accounts and species specific investigations to assessments of the full Upper Mississippi River System, entire fisheries assemblages and communities, and functional interactions and dynamics (for example, Gibson-Reinemer and others, 2017; Bouska, 2020; Chick and others, 2020). Several studies based on the LTRM fisheries data focused on management information needs and informing restoration project planning and potential outcome; these studies emphasized the development and evaluation of habitat association and assessment models (for example, Crites and others, 2012; Sechler and others, 2012; Ickes and others, 2014a; Gibson-Reinemer and others, 2017). Our understanding of native and nonnative fisheries ecology has advanced with an emphasis on the effects of invasive bigheaded carps on native fishes, contributions to regional planning and coordinating efforts, and development of potential control techniques (for example, Sampson and others, 2009; Solomon and others, 2016; Pendleton and others, 2017; Love and others, 2018; Chick and others, 2020).

During the past decade, broad-scale, pattern-based studies and finer-scale, short-term studies have improved our basic understanding of floodplain ecology along the Upper Mississippi River System. Spatially explicit flood inundation models have revealed basic distributional patterns of flooding and associated patterns of floodplain forest species composition, stand structure, community type distributions, soil texture, and fertility (De Jager 2012; De Jager and others, 2012, 2015, 2018; Van Appledorn and others, 2021). Finer-scale studies have provided a more mechanistic understanding of broad-scale patterns by examining seasonal nutrient cycling patterns (De Jager and others, 2015; Swanson and others, 2017) and tree survival rates (De Jager and others, 2013) across gradients of flood inundation. Perhaps most importantly, patterns of invasion by herbaceous species (for example, *Phalaris arundinacea* L. [reed canarygrass]) appear to depend on the degree to which local flooding conditions affect recruitment and tree survival (De Jager and others, 2013). Invasion by herbaceous species, such as reed canarygrass, in turn, may increase rates of nitrogen cycling to a degree that promotes its own long-term persistence, given its high demand for soil nitrogen (Swanson and others, 2017). Finally, information gained from these studies has been incorporated into a process-based model linking flood inundation and forest succession that can be used to evaluate the effects of alternative environmental conditions (for example, hydrology and climate changes) and management actions (for example, harvesting, invasive species control) on floodplain forest dynamics (De Jager and others, 2019).

Previous Assessments

This report builds on previous assessments of the Upper Mississippi River System by scientists and resource managers. The UMRR has completed two Habitat Needs Assessments (HNAs; McCain and others, 2018; Theiling and others, 2000). Information about the structural and functional attributes of the river system that create habitats suitable for various biota and information regarding the values and (or) desired conditions of the river system provided by State and Federal management agencies are assembled for HNAs. These previous assessments covered the entire river system because their primary purposes were to inform strategic decisions about the types, location, and timing of restoration projects. Both assessments used coarse-scale datasets that covered the entire river system and complement the finer-scaled temporal datasets detailed in this report. The most recent HNA took advantage of similar land and water cover datasets from 1989, 2000, and 2010, along with topobathy data (merged light detection and ranging (lidar) and bathymetry data), streamgage data, and simulation models to map and report on 13 indicators of ecosystem structure and function (De Jager and others, 2018). Used together, the HNA indicators and the indicators in this report provide a holistic account of the ecological status across the spatial extent of the Upper Mississippi River System and the temporal span of the LTRM data.

The ongoing ecological resilience assessment of the Upper Mississippi River System also informed the development of this report. The objectives of the resilience assessment are to develop and apply concepts of ecological resilience to large floodplain-river ecosystems, use existing data to infer the ecological resilience of the river system, and highlight implications for ecosystem restoration and management. The initial phase of the resilience assessment produced a system description that simplified the complex river system to its most fundamental elements by identifying valued major uses, major ecological resources, and controlling variables (Bouska and others, 2018). Adaptive capacity of each of the floodplain reaches was assessed through the quantification of a set of indicators of general resilience on the basis of principles of diversity and redundancy, connectivity and slow variables and feedbacks (Bouska, 2018; Bouska and others, 2019). Plausible alternate regimes of the ecosystem were developed to synthesize and communicate what we currently know or hypothesize regarding how specific major ecological resources can shift from desirable to undesirable conditions (Bouska, 2020; Bouska and others, 2020).

Two preceding Upper Mississippi River System status and trends reports have been produced by the UMRR. The first report provided an extensive description of the history of the basin, the geology and land use of the watershed, the basic geomorphological structure of the floodplain-river ecosystem, and the initial information about select biota in the system (USGS, 1999). The second report built on the information provided in USGS (1999) and provided a quantitative assessment of the river system using the first decade of LTRM data (Johnson and Hagerty, 2008). In this third status and trends report, we present results summarizing $2^{1}/_{2}$ decades of long-term monitoring data from the Upper Mississippi River System. Spanning 27 years, we now have a sufficiently long dataset for long-term trends to be defined, important variation through time to be captured, and nonlinear patterns to be identified.

Assessing Status and Trends in the Upper Mississippi River System

The specific metrics monitored through time by LTRM reflect its objective of assessing the general habitat conditions in the rivers. The indicators herein represent a subset of the larger suite of metrics monitored by the LTRM and have been selected to represent the broad range of ecological components of the system and to collectively indicate how and where the system has changed. Selection of the indicators was also based on a UMRR partnership report assessing the indicators included in the second Status and Trends report (UMRR, 2013), subsequent research led by LTRM scientists (mentioned previously), research completed as part of a resilience assessment of the river system (Bouska and others, 2018, 2019, 2020), and additional partnership discussions.

As of 2021, there were no specific and quantitative targets for most of the indicators herein. The exceptions were limited to a subset of water quality indicators where external standards existed (for example, nutrient concentrations developed by the EPA or States) and aquatic vegetation indicators where biologists have determined important thresholds for suitable duck habitat (Devendorf, 2013a, b). Therefore, we assessed the status of the six study reaches using externally derived criteria where possible and used internal spatial and temporal data comparisons for the remaining indicators. For example, we have identified which study reaches have the most and least aquatic vegetation or have the most and least diverse fish communities. Where applicable, we also related the indicator status to existing qualitative statements of desired conditions in McCain and others (2018). The quantitative assessments within this report describe how the conditions of the rivers differ across hydrogeomorphic and climatic gradients and through time and can provide a foundation for the development of quantitative indicator targets for river restoration.

The indicators in chapters B, C, and D describe important spatial hydrogeomorphic gradients within the Upper Mississippi River System and represent large-scale drivers within the system (fig. A3). These metrics are based on data from long-term USGS streamgages, LTRM research, and land cover data collected by the LTRM at decadal intervals. Detailed water quality (ch. E), aquatic vegetation (ch. F), and fish (ch. G) indicators that represent major resources are the emphases of this report. These indicators describe the ecological condition of the river system and were based on the LTRM data from the six LTRM study reaches (figs. A6–A11 and described previously). In recognition of broad changes that have occurred across the system and have been documented through LTRM monitoring, this report includes two chapters that present case studies describing select changes in more detail. Chapter H describes the myriad causes and consequences of long-term changes in submersed aquatic vegetation and water clarity in the Upper Impounded Reach. Chapter I describes the various ecosystem changes that have occurred since the establishment and proliferation of bigheaded carps in the Illinois River and the Unimpounded and Lower Impounded Reaches of the Upper Mississippi River and the role LTRM data and expertise have played in detecting and understanding those changes. Lastly, chapter J provides an overview and synthesis of the changes observed over the last 2.5 decades and their implications for river restoration and management.

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Chapter B: Hydrologic Indicators

By Molly Van Appledorn

Chapter B of
Ecological Status and Trends of the Upper Mississippi and Illinois Rivers
Edited by Jeffrey N. Houser

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Chapter B: Hydrologic Indicators

By Molly Van Appledorn

Introduction

A river's hydrologic regime comprises inter and intra-annual patterns of discharge. The contemporary annual hydrograph of the Upper Mississippi River System is characterized by high discharges during the spring and early summer driven largely by snowmelt or a combination of snowmelt and rainfall (WEST Consultants, Inc., 2000), declining summer discharges, a slight rise in discharge during the fall, and low winter discharges (fig. B1). The hydrology in any given year may vary substantially from mean conditions because of the distribution and nature of precipitation events, temperature patterns, and antecedent conditions in the drainage basin (Pitlick, 1997; WEST Consultants, Inc., 2000). Because the Upper Mississippi River System's hydrologic regime is an important driver of many riverine processes (such as sedimentation and biogeochemical cycling) and affects the distribution and quality of habitat conditions in the river and on the floodplain, it is important to understand whether and how discharge patterns have changed over time.

Ecological relations with river hydrology can be complex. Different organisms and processes interact and respond to a range of hydrologic cues and conditions at different time scales. Because of this context, three indicators were developed to describe the Upper Mississippi River System's hydrologic regime: (1) annual discharge; (2) duration of high discharges; and (3) monthly mean discharge. Annual discharge captures the full range of annual discharge dynamics through characterizations of peak (maximum), typical (mean), and low-discharge (minimum) conditions. Duration of high discharges measures how long the river was experiencing conditions expected to inundate the floodplain annually. Monthly mean discharge describes seasonal patterns of discharge. The intent of these indicators is to capture different yet complementary aspects of the Upper Mississippi River System's hydrologic regime and provide a holistic understanding of the system with multiple ecological endpoints in mind.

The hydrologic indicator data were derived from four U.S. Geological Survey streamgages on the Upper Mississippi and Illinois Rivers (fig. A1 and table B1). These streamgages were selected based on the availability of their long-term data in the U.S. Geological Survey's National Water Information System (U.S. Geological Survey, 2021) and their proximity to Long Term Resource Monitoring study reaches. The indicators characterize the contemporary hydrologic regime since the operationalization of the lock and dams, which were completed on the Upper Mississippi and Illinois Rivers by 1940 (see ch. A, fig. A4). To reflect the contemporary hydrology, I used a period of record spanning from 1940 to 2019 for streamgages at Winona, Minnesota (05378500); Keokuk, Iowa (05474500); and Valley City, Illinois (05586100) (table B1). For the St. Louis, Missouri, streamgage (07010000), where the hydrology is also affected by the Missouri River, I used a period of record spanning from 1960 to 2019 to reflect hydrologic conditions after the Missouri River dam system became operational. For comparisons of hydrology before and after locks and dams were operationalized, see Johnson and Hagery (2008) and Theiling and Nestler (2010). Indicators were computed for water years, which are defined beginning October 1 of a given calendar year through September 30 of the following year and named for the calendar years in which they ended. Water years, commonly used in hydrologic analyses, were used for the indicators because they account for snow, which does not affect discharges until the following spring. Daily mean discharges were used as the basis for all calculations. All records of daily discharge were complete for the periods of analysis, except for the Keokuk, Iowa, streamgage, which was missing a single day (November 22, 2016). I filled in the missing daily value by averaging discharges from the day immediately prior and after the missing date (November 21 and 23, 2016). I used locally weighted regression (LOESS) models and their confidence intervals to visualize indicator behavior through time. I statistically assessed whether indicators were significantly increasing or decreasing through time with a Mann-Kendall trend test, which is a standard test of monotonic trends used in hydrologic analyses (Helsel and others, 2020). Positive values of the Mann-Kendall test statistic (z) indicated a positive trend and negative z values indicated a downward trend. Trends were interpreted as statistically significant when p-values were less than or equal to 0.05.

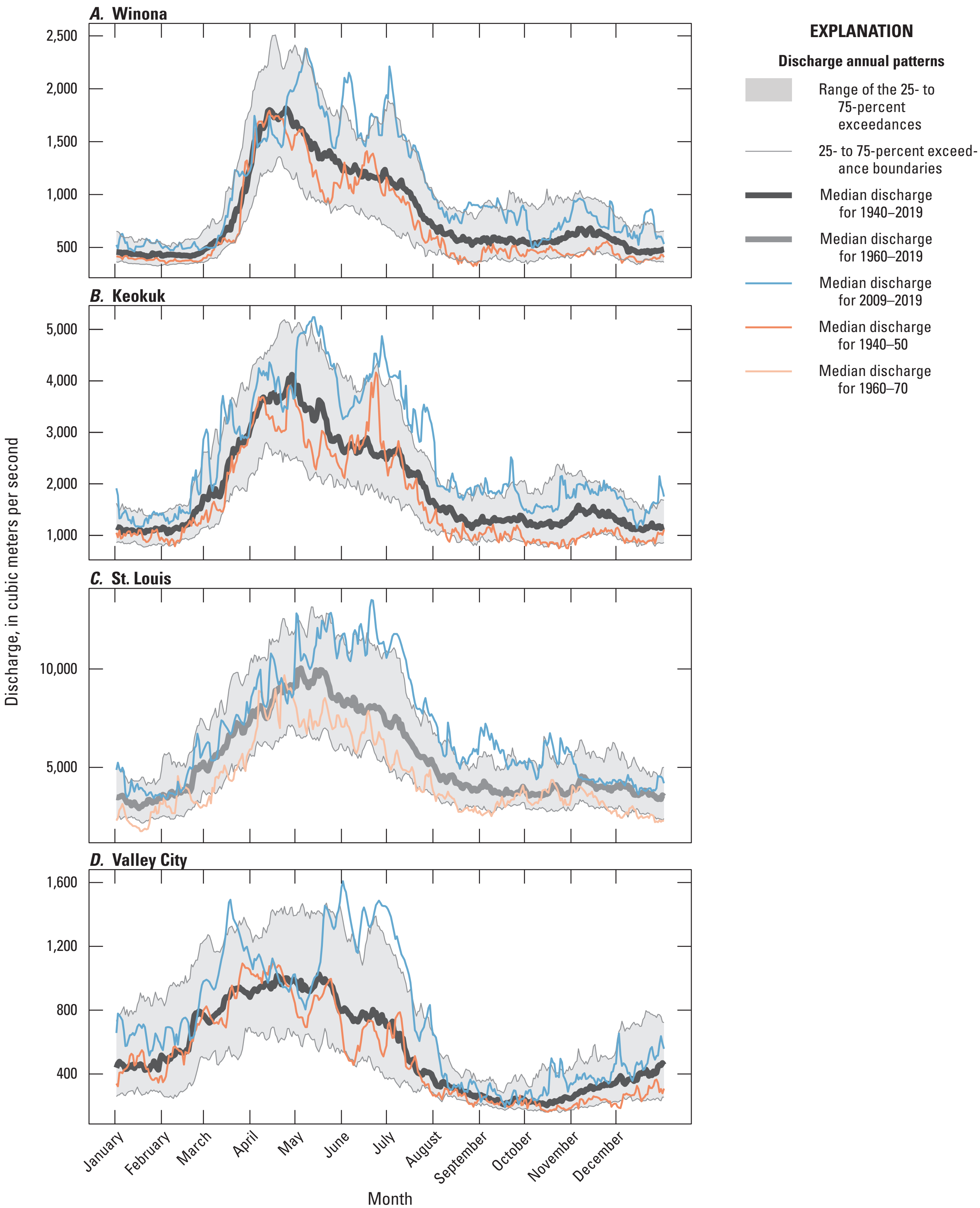


Figure B1. Graphs showing median discharge patterns for *A*, Winona, Minnesota (streamgage 05378500); *B*, Keokuk, Iowa (streamgage 05474500); and *C*, St. Louis, Missouri (streamgage 07010000), on the Upper Mississippi River and at *D*, Valley City, Illinois, on the Illinois River (streamgage 05586100). See figure A1 for map showing streamgage locations. Discharge data are from U.S. Geological Survey (2021).

Table B1. U.S. Geological Survey streamgages used in study.

[NWIS, National Water Information System]

NWIS streamgage number and link to data	NWIS streamgage name	Drainage area (square kilometer)	Navigation pool or river reach
05378500	Mississippi River at Winona, Minnesota	153,327	Pool 6
05474500	Mississippi River at Keokuk, Iowa	308,209	Pool 20
05586100	Illinois River at Valley City, Illinois	69,264	Alton Pool
07010000	Mississippi River at St. Louis, Missouri	1,805,222	Unimpounded Reach

Indicator: Annual Discharge

Annual discharge patterns are a key driver of river-floodplain ecosystem patterns and processes. I used three measures to summarize annual discharges: maximum, mean, and minimum discharges. Annual maximum discharge describes extreme events that can affect sediment scouring and deposition patterns, determine the extent and depth of floodplain inundation, and cause high-velocity conditions in various aquatic habitats. Annual mean discharge describes mean discharge conditions on the river during a given water year, and higher values indicate “wetter” years. This measure can provide insight on sediment loading and transport, floodplain inundation, and connectivity among habitats. Annual minimum discharge describes discharge conditions that typically occur during late summer and winter months when discharge variation can substantially affect the distribution and abundance of suitable habitat (for example, for centrarchid fishes). These three annual measures collectively describe the Upper Mississippi River System’s basic hydrologic regime.

Methods

Annual maximum discharge was calculated as the maximum daily discharge observed during the 12-month water year, defined as beginning October 1 through September 30 of the following year. Annual mean discharge was calculated for each water year as the mean of daily discharge over the 12-month period. Because of the large drainage area for all streamgages, annual mean discharge is similar to median annual discharge. The minimum value of a 7-day running average discharge was computed for each water year as a measure of annual minimum discharge and is commonly used in statistical analyses of low-discharge conditions (for example, Riggs, 1972). Metrics are given in cubic meters per second.

Assessment

All four streamgage locations exhibited similar, increasing patterns of annual discharge (figs. B2, B3, B4). Increases in annual maximum discharge were statistically significant for

Summary of Annual Discharge

Indicator intent: To characterize peak (maximum), typical (mean), and low-discharge (minimum) hydrologic patterns on an annual basis.

Measurement: Annual discharge patterns were summarized using three measures: annual maximum, annual mean, and annual minimum discharge. Annual maximum discharge was defined as the greatest daily discharge value observed during each 12-month water year. Annual mean discharge was computed as a mean of daily discharges during each water year. To calculate annual minimum discharge, a running average with a 7-day window was first computed for daily discharge data. The minimum value of the 7-day running average was then extracted for each water year. All units are in cubic meters per second.

Special consideration: None

Assessment

Status: All four streamgages exhibited similar, increasing patterns of annual discharges. Annual maximum discharges increased significantly only at Keokuk, Iowa, (Upper Mississippi River) and Valley City, Illinois (Illinois River), but annual mean and minimum discharges increased significantly at all four streamgages.

Trends: Annual maximum, mean, and minimum discharges have increased over the period of record, although only annual mean and minimum discharges increased significantly at all locations.

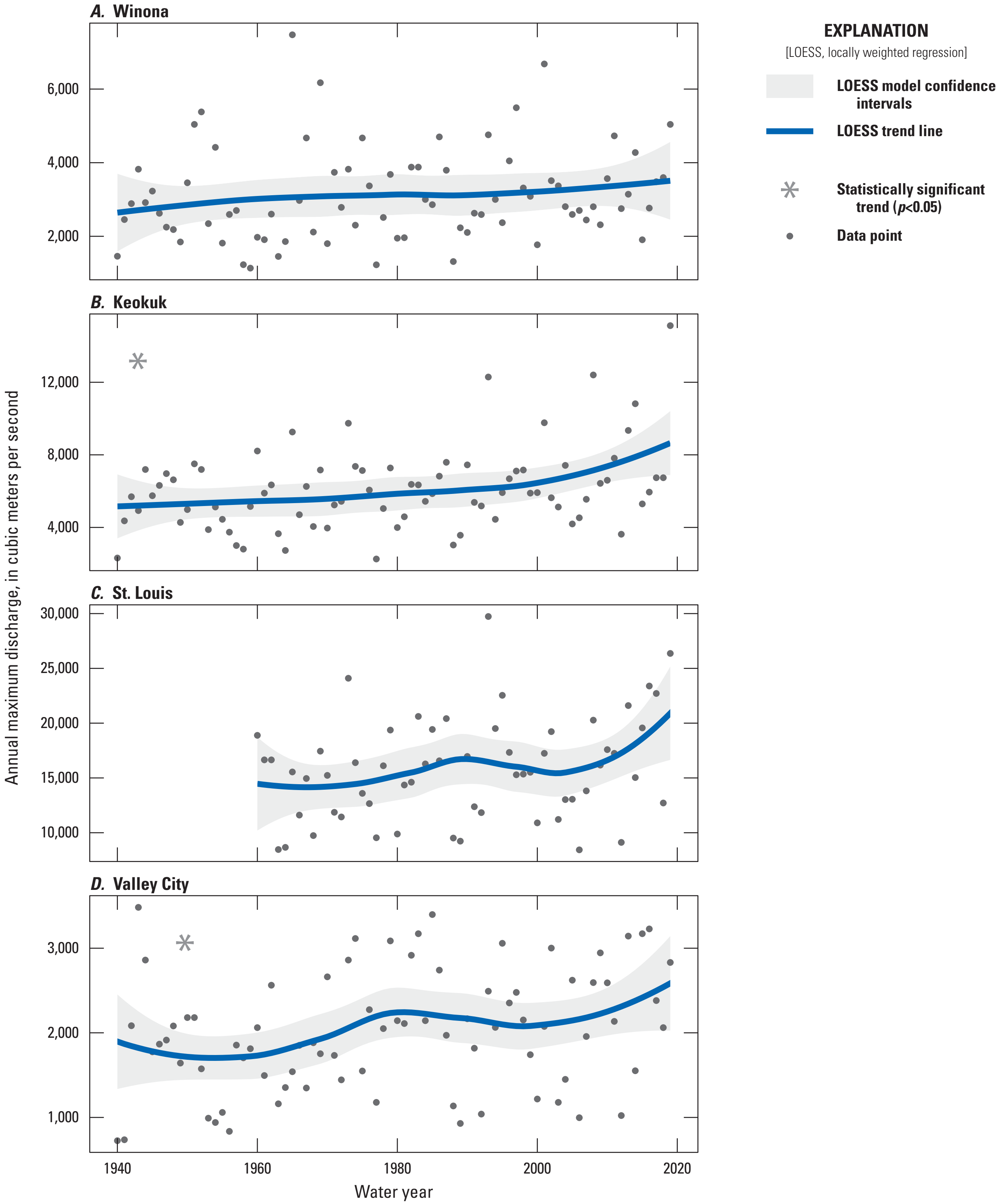


Figure B2. Graphs showing annual maximum discharge patterns for U.S. Geological Survey streamgages at *A*, Winona, Minnesota (streamgage 05378500); *B*, Keokuk, Iowa (streamgage 05474500); and *C*, St. Louis, Missouri (streamgage 07010000); on the Upper Mississippi River and at *D*, Valley City, Illinois; on the Illinois River (streamgage 05586100). Locally weighted regression (LOESS model) trendlines (solid blue lines) and confidence intervals (gray-shaded areas) depict temporal patterns over the period of 1940–2019 (Winona, Keokuk, and Valley City) or 1960–2019 (St. Louis). Asterisks denote statistically significant trends (p<0.05). See figure A1 for map of streamgage locations. Discharge data are from U.S. Geological Survey (2021).

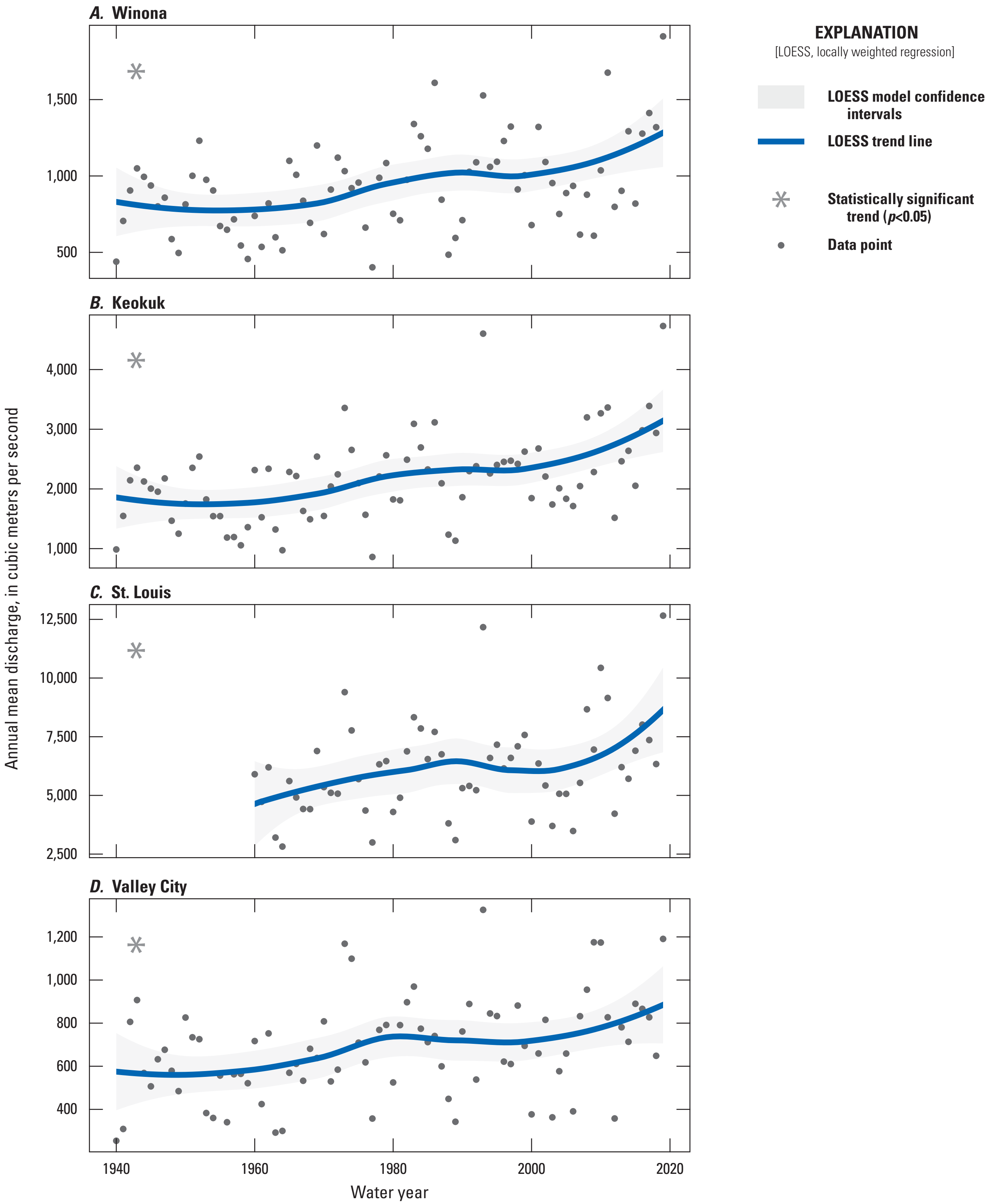


Figure B3. Graphs showing annual mean discharge patterns for U.S. Geological Survey streamgages at *A*, Winona, Minnesota (streamgage 05378500); *B*, Keokuk, Iowa (streamgage 05474500); and *C*, St. Louis, Missouri (streamgage 07010000); on the Upper Mississippi River and at *D*, Valley City, Illinois (streamgage 05586100); on the Illinois River. Locally weighted regression (LOESS model) trendlines (solid blue lines) and confidence intervals (gray-shaded areas) depict temporal patterns over the period of 1940–2019 (Winona, Keokuk, and Valley City) or 1960–2019 (St. Louis). Asterisks denote statistically significant trends (p<0.05). See figure A1 for map of streamgage locations. Discharge data are from U.S. Geological Survey (2021).

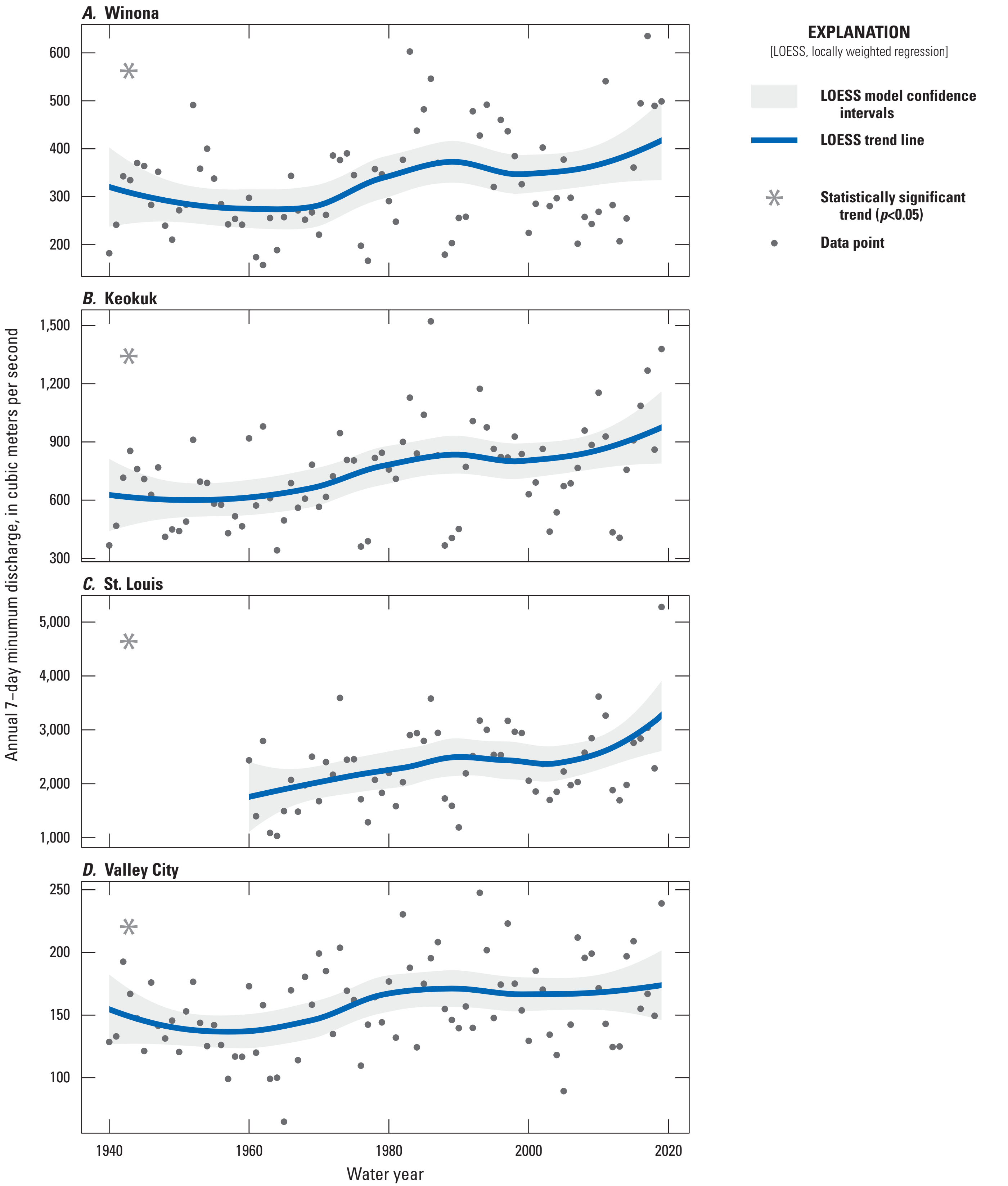


Figure B4. Graphs showing annual 7-day minimum discharge patterns for U.S. Geological Survey streamgages at *A*, Winona, Minnesota (streamgage 05378500); *B*, Keokuk, Iowa (streamgage 05474500); and *C*, St. Louis, Missouri (streamgage 07010000); on the Upper Mississippi River and at *D*, Valley City, Illinois (streamgage 05586100); on the Illinois River. Locally weighted regression (LOESS model) trendlines (solid blue lines) and confidence intervals (gray-shadedareas) depict temporal patterns over the period of 1940–2019 (Winona, Keokuk, and Valley City) or 1960–2019 (St. Louis). Asterisks denote statistically significant trends (p<0.05). See figure A1 for map of streamgage locations. Discharge data are from U.S. Geological Survey (2021).

only two locations: Keokuk, Iowa (p=0.007, z=2.684), and Valley City, Ill. (p=0.009, z=1.907) (fig. B2). Increases represent a 42-percent change for Keokuk and a 27-percent change for Valley City on average between the first and last decades of analysis. Annual mean (fig. B3) and minimum discharges (fig. B4) significantly increased at all locations (p<0.05) over the period of analysis. Annual mean discharge increased the most at Valley City, which recorded a 117-percent increase in mean values between the first and last decades of analysis (395.4 cubic meters per second [m^3/s] versus 895.1 m^3/s). Changes at other streamgages represented between 52 and 60-percent increases between the two periods. Increases in annual minimum discharge were also statistically significant at all streamgages ($p<0.05$; fig. B4), but in some cases the change was more modest. For example, between the first and last decades of analysis, minimum discharges increased by 17-percent on average at Valley City. These quantitative assessments support qualitative observations that the annual hydrograph has shifted in the past decade towards wetter conditions compared with the earliest decade of analysis (fig. B1).

Indicator: Duration of High Discharges

The duration of high-discharges indicator measures the length of time when the river is experiencing discharges at or above the 20-percent AEP discharge (that is, a discharge that has a 20-percent chance of being exceeded in any given year). The 20-percent AEP was selected because above this discharge most locks and dams are open and no longer impede discharge, the floodplain is largely submerged, and tributary discharge and sediment transport are high. High-discharge conditions can affect a variety of geomorphic and ecological patterns and processes in river-floodplain ecosystems (Junk and others, 1989; Sparks, 1995). How long such conditions last can have consequences for aquatic and terrestrial vegetation communities (Tyser and others, 2001; De Jager, 2012; De Jager and others, 2012); biogeochemical cycling (Kreiling and others, 2015); and fish growth, recruitment, and movement (Finger and Stewart, 1987; King and others, 2003; Vallazza and others, 2021).

Methods

To calculate the duration of high discharges, daily discharge records were compared to the streamgage-specific 20-percent AEP discharge as specified in U.S. Army Corps of Engineers (2004). The number of days when daily discharge met or exceeded the 20-percent AEP discharge were then summed for each water year.

Assessment

All streamgages except Winona, Minn., exhibited a statistically significant increase in number of days above a

Summary of Duration of High Discharges

Indicator intent: To identify the length of time during each water year the river experiences high-discharge conditions. High discharges are defined as discharges at or above the 20-percent annual exceedance probability (AEP) discharge (a discharge that has a 20-percent chance of being exceeded each year).

Measurement: High-discharge duration was calculated as the annual total number of days where discharge was equal to or greater than the discharge associated with a 20-percent AEP. Twenty-percent AEP discharges for each streamgage were obtained from the U.S. Army Corps of Engineers (2004).

Special consideration: None

Assessment

Status: The 2019 water year was a record year in terms of high-discharge duration for all three Upper Mississippi River streamgage locations. Water year 2019 was not record-setting at Valley City, Illinois, instead being the fourth highest year on record in terms of high-discharge duration.

Trends: The number of days for which high-discharge conditions were detected tended to increase through time. The number of years with discharges greater than the 20-percent annual exceedance discharge also increased, especially at Keokuk, Iowa, and St. Louis, Missouri, on the Mississippi River and Valley City, Illinois, on the Illinois River.

20-percent AEP discharge over the period of analysis. Comparisons of streamgages between the most recent decade (2010–19) to the earliest decade analyzed (1940–49 or 1960–69) indicate more years with high-discharges and more days above the 20 percent AEP discharge. In the most recent decade on the Upper Mississippi River, discharges greater than the 20-percent AEP discharge were observed in 6 years at St. Louis, Mo. (mean 16.3 days per water year) and in 7 years at Keokuk, Iowa (mean 23.8 days per water year). In contrast, only 2 or 3 years during the first decade of analysis had high-discharge conditions; during these years, there were fewer days of high discharges (mean 7.5 days per water year at St. Louis, Mo., and 3.6 days per water year at Keokuk, Iowa). On the Illinois River, years with high discharges were also more frequent in the last decade of record compared with the first decade of record (6 versus 2 years), and more days experienced high discharges on average in the later decade than in the earlier decade (22.0 versus 19.5 days per water year).

The 2019 water year was a record year in terms of days above the 20-percent AEP discharge for all three streamgage locations on the Upper Mississippi River (fig. B5). Winona, Minn., experienced 50 days of discharges exceeding the 20-percent AEP discharge, and high discharges were experienced for more than a month longer at St. Louis, Mo.: a total of 86 days. The greatest duration of high discharges was observed at Keokuk, Iowa, which had 97 days of discharges exceeding the 20-percent AEP discharge. Before 2019, the next highest number of high-discharge days was in 2001 (30 days) for Winona, Minn., and in 1993 for St. Louis, Mo. (80 days), and Keokuk, Iowa (90 days). Water year 2019 was not a record-setting year at Valley City, Ill., but was the fourth highest year on record for days above the 20-percent AEP discharge (35 days).

Indicator: Monthly Mean Discharge

Monthly mean discharge is an indicator of seasonal discharge conditions. Seasonal discharge conditions affect the quality and distribution of habitat for vegetation, fish and wildlife, the movement of materials and organisms, and the provisioning of ecosystem goods and services through time (Junk and others, 1989; Poff and others, 1997). Many organisms have life history strategies that are adapted to seasonal discharge patterns (Lytle and Poff, 2004). For example, summer low-discharge conditions can affect germination and mortality rates of aquatic vegetation (Harris and Marshall, 1963, Frederickson and Taylor, 1982, Kenow and others, 2018), and high-discharge conditions late in the growing season may be detrimental to germinating seeds and young seedlings in the floodplain (Van Splunder and others, 1995). Rates of biogeochemical cycling can also exhibit seasonal trends related to the interaction between temperature and discharge conditions (Richardson and others, 2004; Jicha and others, 2014). Water quality constituent levels may also be related to seasonal patterns of discharge. For example, winter discharges can affect dissolved oxygen levels (ch. E, this volume) and water temperature in off-channel areas: critical aspects of overwintering habitat in the Upper Mississippi River System.

Methods

A monthly mean discharge for each water year was calculated from daily discharges and expressed in cubic meters per second.

Summary of Monthly Mean Discharge

Indicator intent: To characterize seasonal patterns of discharge and how they may vary across the periods of record.

Measurement: Daily discharges were summarized for each month in every water year as a mean value. Units are given in cubic meter per second.

Special consideration: None

Assessment:

Status: Streamgage locations differed in which months exhibited significant positive trends of monthly mean discharge through time. There is some evidence of a seasonal shift in peak monthly mean discharge from April to May or June.

Trends: Positive trends were observed for at least 4 of the 12 months for all streamgages. No significant negative trends were observed in any month for any streamgage. No trends were observed in April for any streamgage.

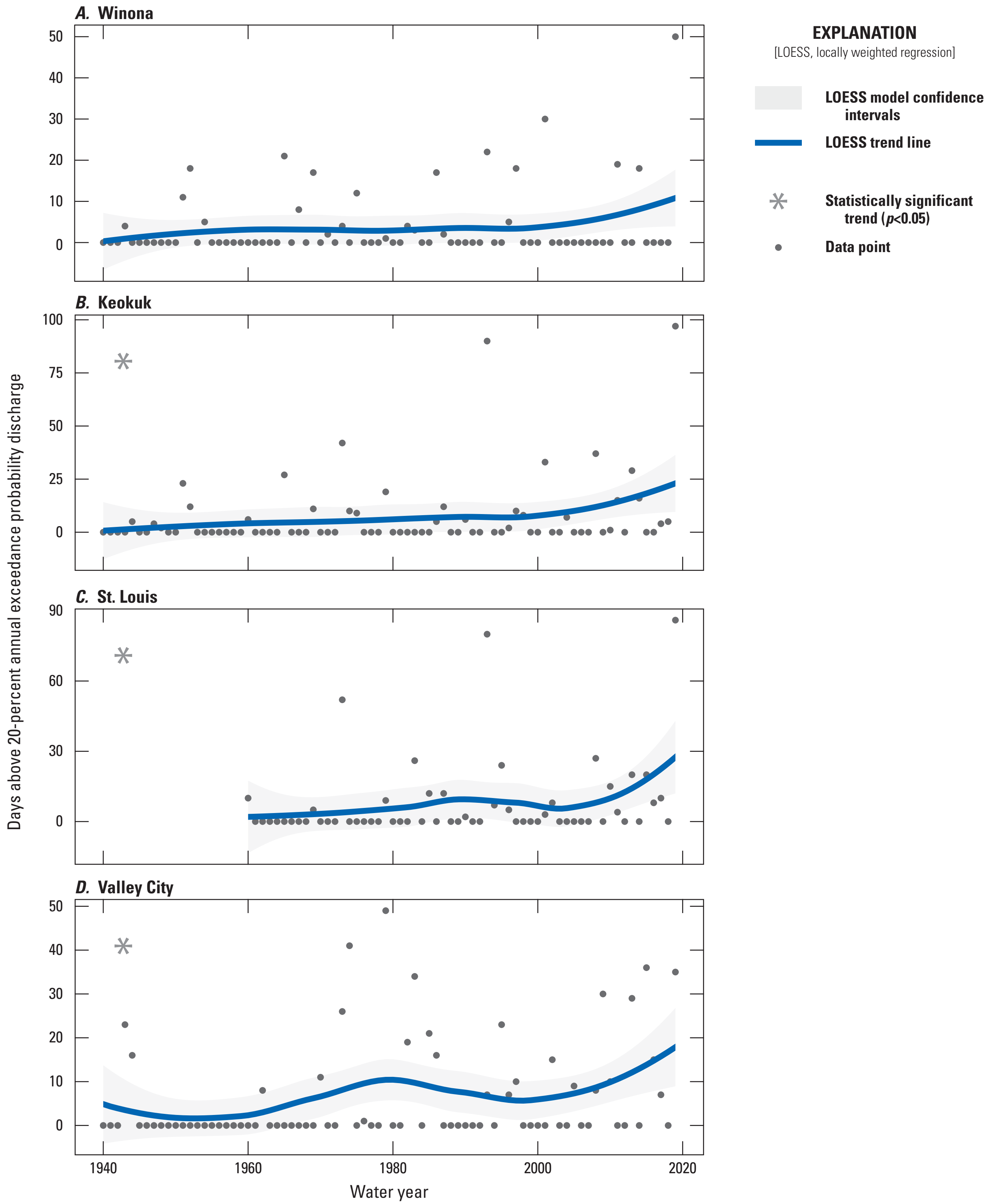


Figure B5. Plots showing the total number of days during a water year for which a discharge exceeding the 20-percent annual exceedance probability discharge was observed for U.S. Geological Survey streamgages at *A*, Winona, Minnesota (streamgage 05378500); *B*, Keokuk, Iowa (streamgage 05474500); and *C*, St. Louis, Missouri (streamgage 07010000), on the Upper Mississippi River; and at *D*, Valley City, Illinois (streamgage 05586100), on the Illinois River. Locally weighted regression (LOESS model) trendlines (solid blue lines) and confidence intervals (gray-shaded areas) depict temporal patterns over the period of 1940–2019 (Winona, Keokuk, and Valley City) or 1960–2019 (St. Louis). Asterisks denote statistically significant trends ($p<0.05$). See figure A1 for map of streamgage locations. The 20-percent annual exceedance probability discharge is from U.S. Army Corps of Engineers (2004) and discharge data are from U.S. Geological Survey (2021).

Assessment

Streamgage locations differed in which months exhibited significant positive trends of monthly mean discharge through time (fig. B6). Positive trends of monthly mean discharge were observed for at least 4 of the 12 months for all streamgages. No significant negative trends were observed for any month at any streamgage. No trends for the month of April were detected at any location. There is some evidence of a seasonal shift in peak monthly mean discharge across streamgages. Early in the period of record, monthly mean discharges were typically greatest in April, but in the last decade, monthly mean discharges were observed to be greater in May or June.

On the Mississippi River, the two northern streamgages at Winona, Minn., and Keokuk, Iowa, exhibited more similar trends than those of St. Louis, Mo. Significant positive trends in monthly mean discharge occurred at both northern streamgages for 10 of the 12 months (April trends were not significant for either streamgage, and September trends were not signifcant for Winona.), whereas St. Louis only shared positive trends in May, June, July, and August. Increases were particularly great during the months of May and June for all three streamgages. For example, typical May mean discharges in 2010–19 were more than twice as great as those from 1940 to 1949 at Winona (2,319 m^3/s versus 1,113 m^3/s) and Keokuk (5,083 m^3/s versus 2,363 m^3/s), and more than 160-percent greater at St. Louis (11,459 m^3/s versus 6,962 m^3/s; 1960–69).

Positive trends were observed in 7 of the 12 months on the Illinois River at Valley City: January, March, June, August, September, November, and December. No significant trends for the Illinois River were detected in May (p=0.121); however, May exhibited large increases for Mississippi River streamgages. This discrepancy may be due to the high variability in monthly May discharges across years. Instead, particularly large increases of monthly mean discharges were observed on the Illinois River for June, where the median indicator discharges in 2010–19 were 117-percent greater than those from 1940 to 1949 (1,406.8 m^3/s versus 649.0 m^3/s). Large increases were also observed for November and December of 2010–19: median indicator discharges in 2010–19 were 73-percent greater than those from 1940 to 1949.

State of Ecosystem

There is evidence of hydrologic change over time for the Upper Mississippi and Illinois Rivers in the form of significantly positive trends in all discharge indicators. No significant negative trends were exhibited by any indicator. Annual maximum discharge increased at two of the four streamgages, but annual mean and minimum discharges increased significantly at all streamgages. Together, these patterns indicate that greater discharges through much of the year are driving these patterns rather than increases in a single, annual event. Seasonal indicators also support this interpretation: positive trends in monthly mean discharge were found for at least 4 months per streamgage. Indeed, qualitative comparisons between annual hydrographs from the first and last decades of analysis show shifts toward overall wetter conditions (fig. B1).

The hydrologic changes observed in this study are consistent with observations from other studies that have documented instances of increasing discharge across many upper Midwest drainage basins since the 1940s, and most of the increase in discharge is associated with moderate and low discharges rather than peak discharges (Lettenmaier and others, 1994; McCabe and Wolock, 2002; Lins and Slack, 2005; and Kochendorfer and Hubbart, 2010). Johnson and Hagerty (2008) also documented increasingly wetter conditions over time in the Upper Mississippi River System. Increased discharge has typically been attributed to climate and drainage basin land-use changes that can affect the amount and delivery of water to a river system. For example, Mallakpour and Villarini (2015) detected increased frequency of high-discharge events in upper Midwest drainage basins from 1962 to 2011 and attributed the change to shifts in seasonal precipitation patterns over the same period. In contrast, Kochendorfer and Hubbart (2010) found that changes in annual peak discharges through time in subbasins of the Upper Mississippi River System were not well explained by precipitation trends, instead suggesting that rural land-use practices may have also contributed to the observed hydrologic trends. Discerning the roles of climate and land-use patterns on the magnitude and nature of hydrologic changes, and the subsequent ecological consequences, remains a challenge.

Future Pressures

Climate, drainage basin land-use changes, and engineered structures (for example, wing dikes, levees, dams, and so on) have all been shown to affect discharge dynamics in river systems (Randall and Mulla, 2001; Twine and others, 2005; Pinter and others, 2008; Kochendorfer and Hubbart, 2010; Schilling and others, 2010; Alexander and others, 2012; Pryor and others, 2014; Foufoula-Georgiou and others, 2015; Mallakpour and Villarini, 2015; Byun and others, 2019). These factors all have the potential to affect the future discharge regime of the Upper Mississippi River System, although how and to what degree are largely unknown.

Climate change can affect the frequency, intensity, and seasonality of precipitation events: factors that directly affect the delivery of water to a river system. Climate models for the upper Midwest region indicate wetter spring and winter seasons (up to 30-percent more precipitation) because of increased rain and less snow and drier summers (up to 15-percent less precipitation) under a range of scenarios (Byun and Hamlet, 2018). Using a set of 20 drainage basins in the region (maximum area of 74,164 square kilometers), 8 of which are within the Upper Mississippi River System's drainage basin, Byun and others (2019) found that such potential climate

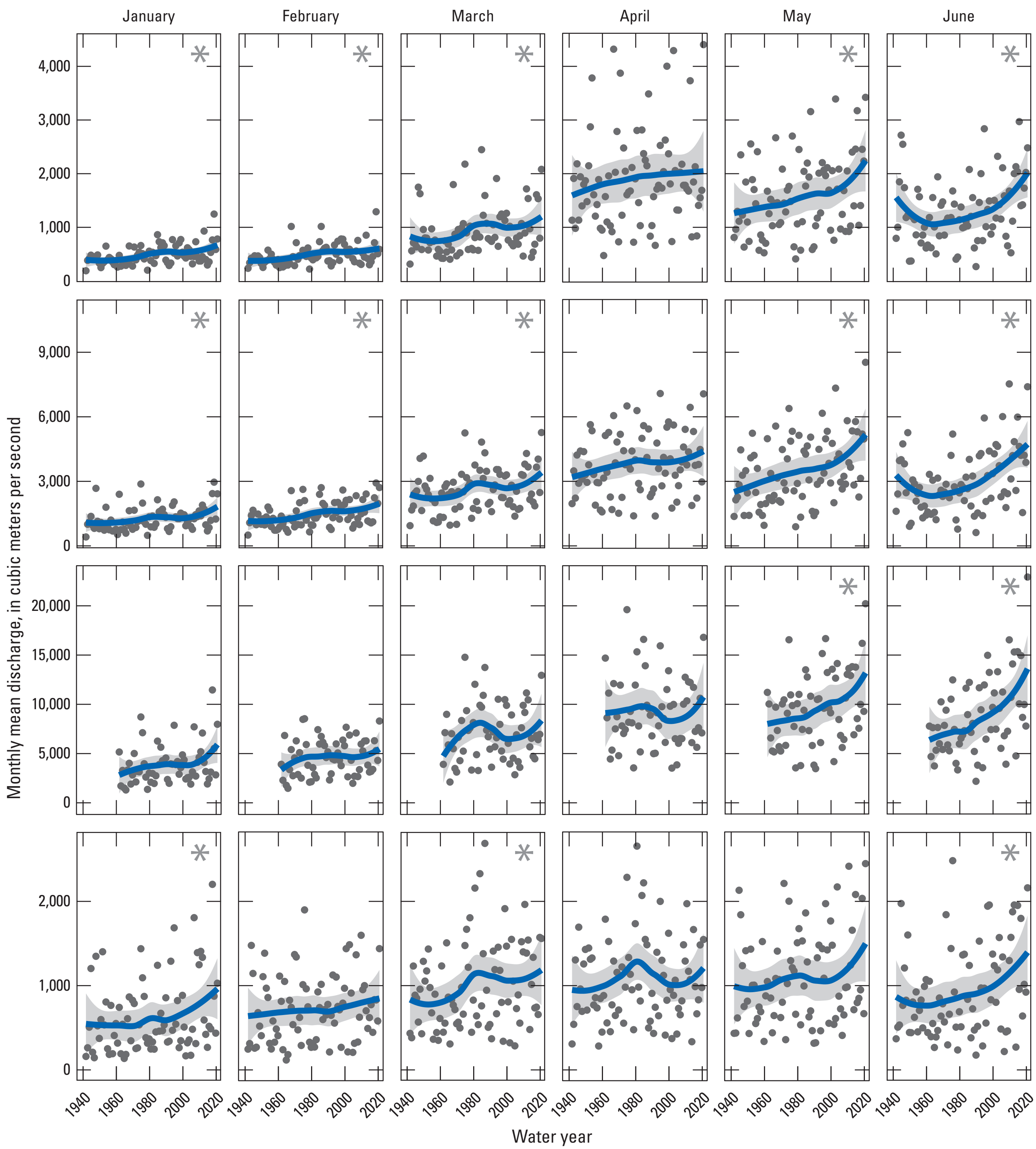
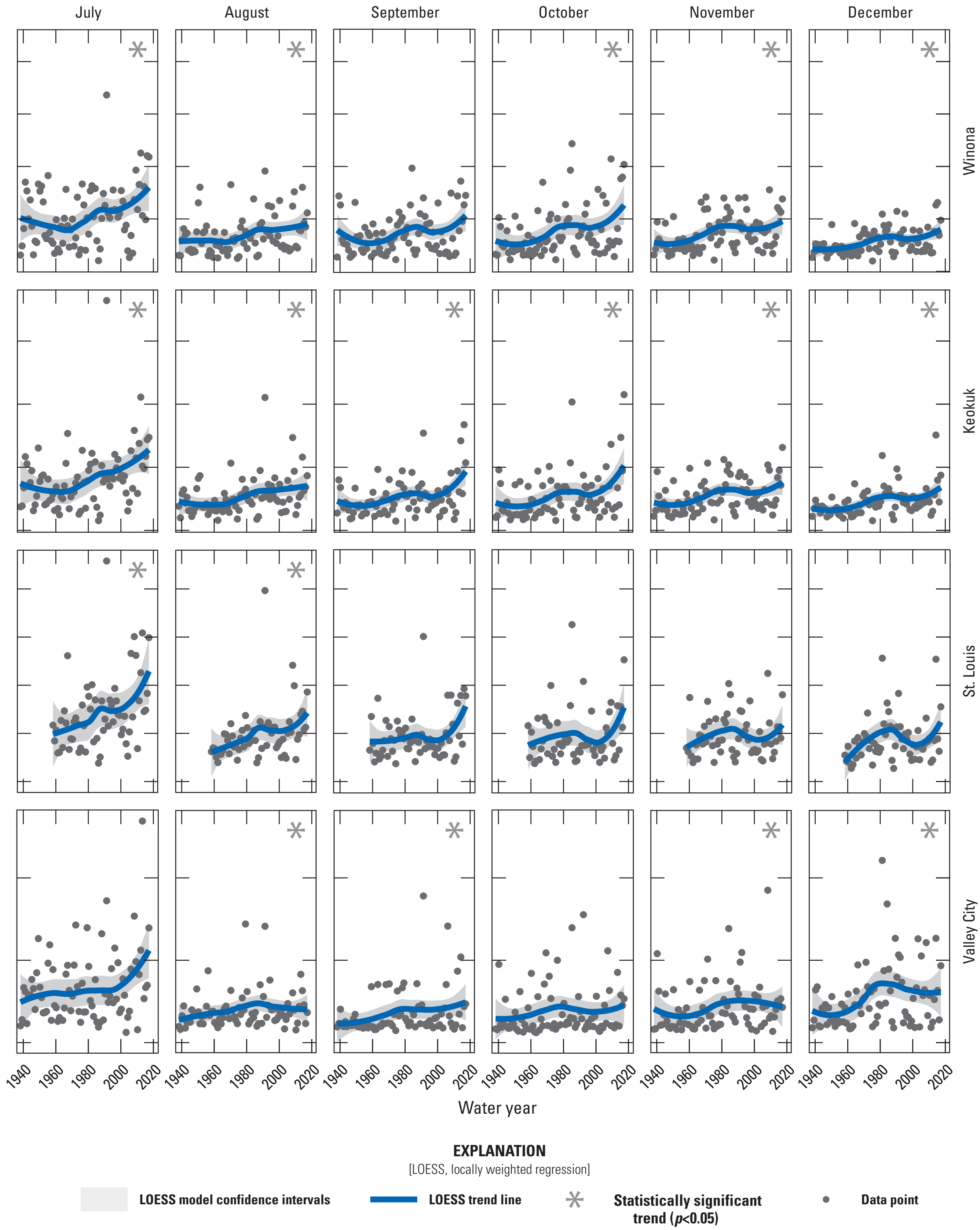


Figure B6. Graphs showing patterns of monthly mean discharges for U.S. Geological Survey streamgages at Winona, Minnesota (streamgage 05378500); Keokuk, Iowa (streamgage 05474500); and Valley City, Illinois (streamgage 05586100); for water years 1940–2019 and at St. Louis, Missouri (streamgage 07010000); for water years 1960–2019. In each panel, years are on the x-axis and monthly mean discharge (in cubic meters per second) is on the y-axis. Locally weighted regression (LOESS model) trendlines (solid blue lines) and confidence intervals (gray-shaded areas) depict temporal patterns over the period of analysis. Asterisks denote statistically significant trends ($p<0.05$). See figure A1 for map of streamgage locations. Discharge data are from U.S. Geological Survey (2021).


July
August
September
October
November
December
Winona
Keokuk
St. Louis
Valley City
1940
1960
1980
2000
2020
Water year
EXPLANATION
[LOESS, locally weighted regression]
LOESS model confidence intervals
LOESS trend line
Statistically significant trend (p<0.05)
Data point

shifts could contribute to increased annual peak discharges for the 1 percent annual exceedance probability (AEP) discharge, peak discharge events occurring up to a month earlier in the year, and shifts in extreme low discharges. Jha and others (2004) found a 50-percent net increase in total annual water yield in the Upper Mississippi River System's drainage basin using a regional climate model. Although these results were produced for subbasins within the larger Upper Mississippi River System's drainage basin, these findings indicate the potential for continued increases in discharge for the main stem of the Upper Mississippi River System. To date there is no study predicting how the main stem of the Upper Mississippi River System, a much larger river system than any of the previously studied drainage basins, may respond to future climate change scenarios at scales relevant for management decisions. Assessing potential future hydrologic changes anticipated in the main stem of the Upper Mississippi River System would be important for future studies.

Future land-use shifts within the greater Upper Mississippi River System may also have the potential to affect its hydrology. Changes in the way a drainage basin captures, stores, and routes water due to shifts in land-use practices can affect water storage discharge and rainfall-runoff relations (Foufoula-Georgiou and others, 2015). Soil conservation practices (Gerbert and Krug, 1996) and conservation tillage (Tomer and Schilling, 2009) can reduce peak discharges and increase base flow locally. Total discharge can also be reduced through conservation drainage management practices (Skaggs and others, 2012; Williams and others, 2015). However, discharge can increase where pasture is converted to row crops (Zhang and Schilling, 2006), where there is an increase in impervious surfaces (Shuster and others, 2005), or where there are changes in other land-use practices, such as an increase in tile drainages (Randall and Mulla, 2001). Intensive land-use changes are expected in the Upper Mississippi River System's drainage basin in the future (Foley and others, 2004; Sohl and others, 2012; Rajib and Merwade, 2017). Projection models of how different land-use change scenarios may affect hydrology in the Upper Mississippi River System's drainage basin document highly variable responses. For example, a 30-percent decrease of forest and grassland cover under a "worse-case" scenario is projected to generate a 70-percent increase in surface runoff (Rajib and Merwade, 2017).

Engineered structures such as wing dikes, levees, revetments, and dams have been used to modify the Upper Mississippi River System to meet transportation, agricultural, and industrial needs (Fremling, 2005). Engineered structures can affect discharge regimes of river systems by altering channel geometry, discharge dynamics within the channel, and river-floodplain connectivity. For example, reduced floodplain access can speed the propagation of flood waves, concentrating downriver discharge and increasing peak discharges (Woltemade and Potter, 1994; Jacobson and others, 2015). Changes in the degree of channel confinement can amplify hydrologic responses to other hydrologic drivers, such as climate change (Muñoz and others, 2018). Increasing discharges during our period of analysis, in addition to the uncertain hydrologic responses to potential future climatic and land-use scenarios, may have implications for restoration and river management. Restoration strategies to "make room for the river" have goals of increasing storage and conveyance of discharges through a variety of strategies (Interagency Floodplain Management Review Committee, 1994; Larson and Plasencia, 2001; Rijke and others, 2012). One of these strategies, reconnecting floodplains to their rivers, has gained prominence in the United States (Sparks and others, 2003; Opperman and others, 2009; Opperman and others, 2010; Jacobson and others, 2015; Guida and others, 2016). Floodplain reconnection studies on the Illinois River show tradeoffs among improved river conveyance, ecosystem services, and socioeconomic factors, which would have to be navigated with care to meet restoration goals and societal needs in the future (Sparks and others, 2003; Guida and others, 2016).

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Chapter C: Geomorphic Indicators

By Molly Van Appledorn and James T. Rogala

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Overview

Geomorphic changes in the Upper Mississippi River System have long been a concern of agencies charged with maintaining and restoring river habitat (GREAT I, 1980; Jackson and others, 1981; U.S. Fish and Wildlife Service, 1992). Large, meandering alluvial rivers like the Upper Mississippi River System are expected to constantly change and adjust their fluvial landforms within their corridors over time as a result of the natural interaction of hydrologic processes, sediment movement, and vegetation. Historically, erosion and sedimentation patterns in the Upper Mississippi River System also may have changed over the middle to late 20th century because of increased annual precipitation, increased variability in flood magnitudes from year to year, the possibility of more fall and winter flooding, and decreases in upland sources of sediment from many tributaries during that time (for example, Belby and others, 2019). Geomorphic changes in the Upper Mississippi River System also reflect human modifications to the river system itself such as constructed agricultural levees, navigation infrastructure and dredging, and altered flows and water levels. These changes have altered hydrologic, hydraulic, and sediment dynamics (WEST Consultants Inc., 2000; Alexander and others, 2012). For example, rates of island loss and land surface erosion since the completion of the locks and dams have been largely attributed to changes in sedimentation rates, pool elevations, and wave action in many parts of the Upper Mississippi River System (WEST Consultants Inc., 2000; Collins and Knox, 2003; Fremling, 2005). However, there have also been gains in land surface area in other areas of the Upper Mississippi River System in the form of islands, sediment filling between wing dams, and delta formations (Fremling and others, 1973; WEST Consultants Inc., 2000; Freyer and Jefferson, 2013). Changes in sediment dynamics have also affected off-channel aquatic areas: sediment deposition in backwater lakes has reduced water depths and contributed to loss of contiguous backwater lake surface area in the Upper Mississippi and Illinois Rivers (Lee and Stall, 1976; Bhowmik and Adams, 1989; WEST Consultants Inc., 2000).

In this chapter, we use two complementary indicators to describe some of the geomorphic changes in the Impounded Reaches of the Upper Mississippi River System that have been detected during the past three decades. These two indicators, new landform surface area and backwater bed elevation change (an indicator of net sedimentation in backwaters), serve as examples of the types and rates of geomorphic change in the river system occurring over a period similar to that of the data collection efforts of the Long Term Resource Monitoring element. The indicators were based on research on landform changes and sediment dynamics in the Upper Impounded Reach as reported in Rogala, Fitzpatrick, and Hendrickson (2020) and Rogala, Kalas, and Burdis (2020). For a comprehensive review of geomorphic changes in the Upper Mississippi River System, see WEST Consultants Inc. (2000).

The first indicator, new landform surface area, is defined as the accumulation of new, natural surfaces within the river and its floodplain. Mapping new landform surface area over time can provide location information, help identify potential contributing factors driving the geomorphic change, and inform expectations about how river and floodplain biota may respond to the changes. New landform surfaces can be important habitats for ecological restoration in regulated river systems (for example, Volke and others, 2015). The exposed, sandy substrate of new landforms can be preferred habitat for some biota like shorebirds and some waterbirds and can offer ideal conditions for the establishment of important early successional tree species such as willows and cottonwood. For aquatic organisms, however, expansion of certain landforms in larger backwater lakes can reduce suitable habitat for some aquatic and fish species.

The second indicator, backwater bed elevation change, is used to quantify the net effects of sedimentation and erosion within two Long Term Research and Monitoring (LTRM) element study reaches from the Upper Impounded Reach, and by extension, changes to water depth (for any given water surface elevation) over time. Increased bed elevations reduce available deepwater habitat and increase areas of shallow water, indicating areas of net sedimentation and water depth decrease. Sediment deposition or erosion can also affect water exchange rates between main channel and off-channel areas in the river, which is an indicator of habitat suitability for some organisms in the Upper Mississippi River System (De Jager and others, 2018). Bed elevation changes in deep backwater areas are of concern to resource managers because deep backwater areas can provide overwintering habitat for recreationally valued fish species and resource managers consider deep backwater areas to be uncommon (McCain and others, 2018). Additional sediment accumulation in these areas can deplete this valued lentic habitat.

Indicator: New Landform Surface Area

New landform surface area is formed by sediment deposition through time within the river-floodplain ecosystem. Four types of new landform surface area are within the river and its floodplain: bar-tail limbs, crevasse deltas, tributary deltas, and impounded deltas (fig. C1). Bar-tail limbs and delta features are described by Rogala, Fitzpatrick, and Hendrickson (2020). Bar-tail limbs are relatively linear features appearing along the margin of a main channel or side channel downstream from narrow remnant floodplains and levees. Delta landforms are characterized by their fan-like appearance, lenticular deposits, and associations with minor distributary channels. Delta features are named according to their position in the river-floodplain ecosystem: crevasse deltas form in backwaters, tributary deltas occur at tributary confluences, and impounded deltas are located in impounded areas. Bar-tail limbs and delta features represent an uncommon terrestrial habitat condition because they comprise bare mineral soils that support early successional vegetation species.

The geomorphic processes that produce new landform surface area can affect the amount and distribution of aquatic and floodplain habitats. For example, sediments forming a crevasse delta at a channel-backwater connection can accumulate in subsurface and surface areas of the backwater through time. As a result, some deeper aquatic off-channel areas can become shallower while other aquatic areas can be converted to terrestrial habitat. The implications of these processes can include a loss of deepwater off-channel habitat, new shallow-water habitat that can support submersed aquatic vegetation, and new terrestrial habitat that can be colonized by early successional tree species like willows: an underrepresented vegetation type that is important for forest successional processes. Therefore, quantifying surface area gains is important for understanding trajectories of geomorphic change and the implications for aquatic and floodplain habitats.

Methods

New landform surface area was calculated for the Upper and Lower Impounded Reaches (Pools 3 through 26) of the Upper Mississippi River (fig. A1). New landform surface area was detected by interpreting lateral differences in land-water boundaries across 1989, 2000, and 2010 LTRM land-cover datasets through a process described by Rogala, Fitzpatrick, and Hendrickson (2020) and summarized in Rohweder (2019). First, vegetation classes from the 1989, 2000, and 2010 LTRM land-cover datasets were reclassified to one of four classes: aquatic vegetation, willow community, wet meadow, and terrestrial other than willow and wet meadow. The reclassified land-cover datasets were then sequentially overlayed to identify the location and type of transitions. Transition areas greater than 0.1 hectare (ha), adjacent to older landmasses, and representing a sequential change from aquatic vegetation

Summary of New Landform Surface Area

Indicator intent: Assess geomorphic change by quantifying the amount of new, terrestrial landform surface area formed within two periods (1989–2000 and 2000–10) in Pools 3–26 of the Upper Mississippi River.

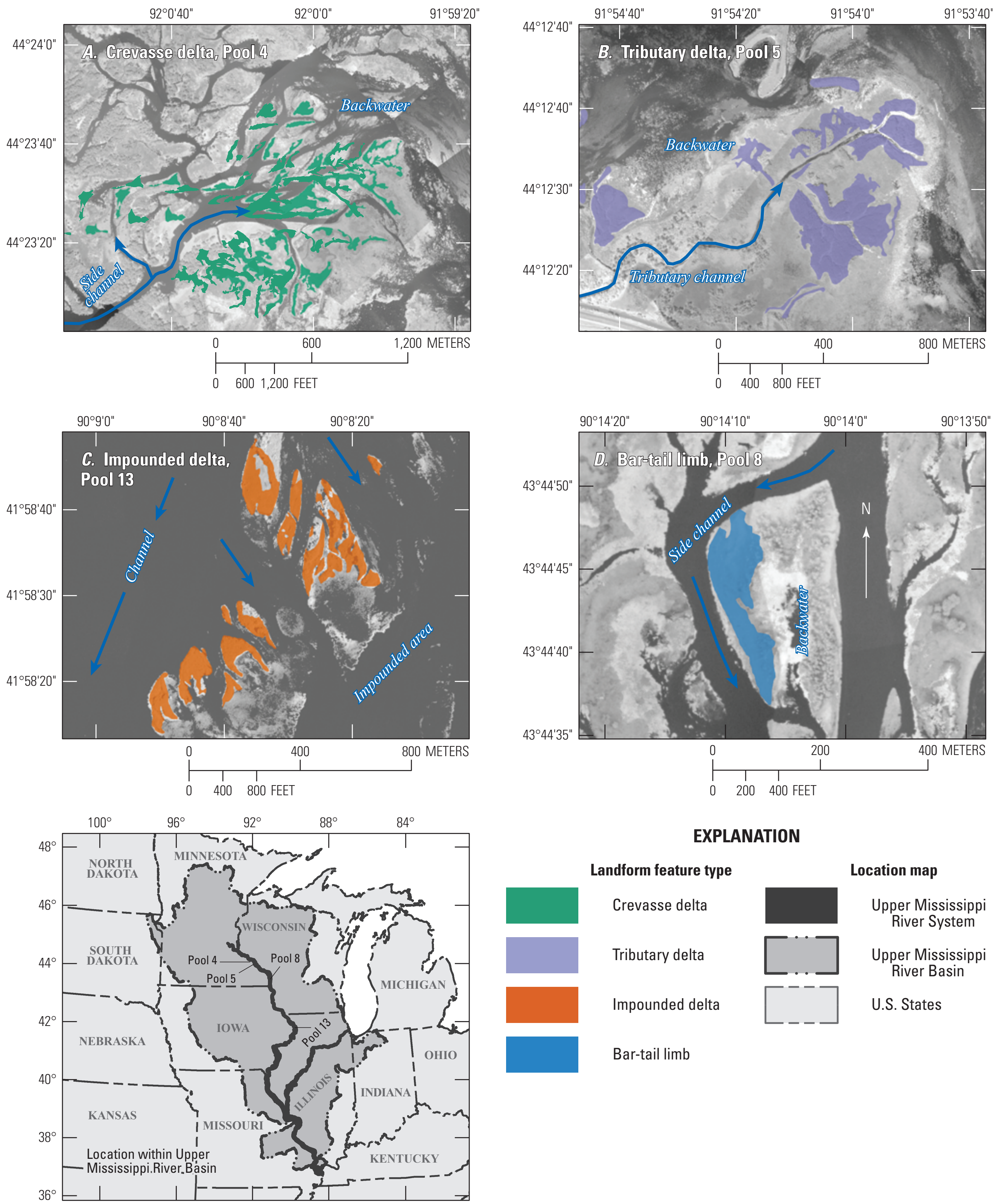
Measurement: New landform surface area was identified using geospatial overlay analysis of 1989, 2000, and 2010 Long Term Resource Monitoring element land-cover data as summarized in Rohweder (2019) and detailed by Rogala, Fitzpatrick, and Hendrickson (2020).

Special consideration: Water levels at the time of aerial imagery collection from which the Long Term Resource Monitoring element land-cover data were derived may affect surface area calculations. However, that effect is unlikely to change the conclusions of this report because new landform surface area was inferred from sequential changes from aquatic habitat to either willow, wetland, or other vegetated cover type; these changes are relatively insensitive to short-term (for example, daily) fluctuations in water level compared to detection of land-water boundaries. The data and methods used were not appropriate for detection of landform loss such as island and bank erosion, because erosional losses can be obscured by minor georeferencing errors and tree canopy overlap (Rogala, Fitzpatrick, and Hendrickson, 2020).

Assessment

Status: A total of 718.5 hectares of new landform surface area was detected across all landform feature types for the period of study (1989–2010), 80 percent of which occurred in the Upper Impounded Reach (Pools 3–13). The amount of net surface area gained varied by landform feature type and navigation pool.

Trends: The amount of new surface area detected was similar within the two periods (1989–2000 and 2000–10), and there was little difference between the two periods in the distribution among new landform feature type.



Base from U.S. Geological Survey digital data, 1:1,000,000, 2014
Aerial photography and landform features from Upper Mississippi River Restoration program's Long Term Resource Monitoring element
Universal Transverse Mercator projection, zone 15, North American Datum of 1983

Figure C1. Maps showing new landmasses formed as *A*, a crevasse delta in Pool 4; *B*, tributary delta in Pool 5; *C*, impounded delta in Pool 13; and *D*, bar-tail limb in Pool 8. Arrows depict flow direction and colors depict new landform surface areas.

to either willow, wet meadow, or other terrestrial land covers were considered to be new landform surfaces. Man-made features were excluded. Landform loss (for example, island erosion) was not measured because the data and methods used precluded its detection. For example, erosional loss tends to occur along shoreline margins, margins that can be masked by the tree canopy in aerial photos used to delineate LTRM land-cover datasets. It can, therefore, be difficult to distinguish erosional losses at shoreline margins from minor georeferencing errors of the land-cover datasets (Rogala, Fitzpatrick, and Hendrickson, 2020). Although such limitations can also affect the detection of new landform surface areas, bar-tail limbs and delta landforms are morphologically distinct from georeferencing errors in shape and size.

Assessment

Status

Most of the new surface area was detected in the Upper Impounded Reach (Pools 3–13; 575.4 ha), and only 20 percent of the new surface area was in the Lower Impounded Reach (Pools 14–26; 143.1 ha) (fig. C2). The most common types of land expansion were crevasse deltas (259.0 ha) and bar-tail limbs (253.8 ha); deltas associated with tributaries and impounded areas comprised smaller areas (90.2 and 115.4 ha, respectively). The only feature to be detected in the Upper and Lower Impounded Reaches was the bar-tail limb, but it was not observed in every pool (fig. C3). Similar total bar-tail limb areas were in the Upper and Lower Impounded Reaches (114.3 and 139.5 ha, respectively). Other features (crevasse, tributary, and impounded deltas) were only detected in pools from the Upper Impounded Reach and were not evenly distributed across pools (fig. C4). Crevasse deltas were detected in every pool, but their area was greater in upper pools (Pools 3–7; 207.8 ha) of the Upper Impounded Reach than pools lower in the reach (Pools 8–13; 51.2 ha).

Trends

Nearly identical amounts of total new area were observed for each period (333.8 ha from 1989 to 2000 and 384.7 ha from 2000 to 2010). The amount of new bar-tail limb area was similar between periods for the Upper Impounded Reach (53.3 and 61.0 ha, respectively) and the Lower Impounded Reach (71.2 and 68.3 ha, respectively) (fig. C3). New crevasse delta area was larger during the second period, increasing from 114.6 ha in 1989–2000 to 144.5 ha in 2000–2010. Amounts of new impounded delta area and tributary delta area were similar in 1989–2000 and 2000–2010 (impounded deltas, 53.4 and 62.0 ha; tributary deltas, 41.4 and 48.8 ha).

Indicator: Backwater Bed Elevation Change

Backwater bed elevation change is an indicator that characterizes the net effect of sedimentation and erosion in connected, off-channel backwater lakes. This indicator describes changes in backwater depth through time. Positive change in backwater bed elevation indicates net sedimentation and, therefore, reduced water depths at any given water surface elevation. Net sedimentation can result in loss of deepwater, off-channel habitat, conversion of shallow aquatic areas to terrestrial habitat, and burial of aquatic vegetation. Decreased backwater bed elevation indicates net erosion and greater water depths at any given water surface elevation. Backwater bed elevation changes can also affect water exchange rates between the main channel and off-channel areas.

Summary of Backwater Bed Elevation Change

Indicator intent: Assess the net effects of sedimentation and erosion on off-channel backwater lakes as measured by changes in bed elevations. Increased bed elevation indicates shallower depth for any given water surface elevation.

Measurement: Bed elevation was measured along 38 transects of backwater lakes within Pools 4 and 8 in 1997 and again in 2017 or 2018 (Rogala, Kalas, and Burdis, 2020).

Special consideration: Transects were resurveyed in either 2017 or 2018. Bed elevation changes are reported as rates in centimeter per year to allow for comparisons across 20- and 21-year periods.

Assessment

Status: Rates of bed elevation change indicate net sedimentation in backwaters (pool-wide means from 0.27 to 0.51 centimeter per year). Rates of bed elevation change were lower than many previously reported rates. Rates varied spatially within and among transects. Rates of bed elevation change tended to be greatest in deeper areas and least in shallower areas.

Trends: Backwater bed elevations have risen through time due to net sedimentation when summarized at the pool scale. However, evidence of sedimentation and erosion was observed within and among transects.

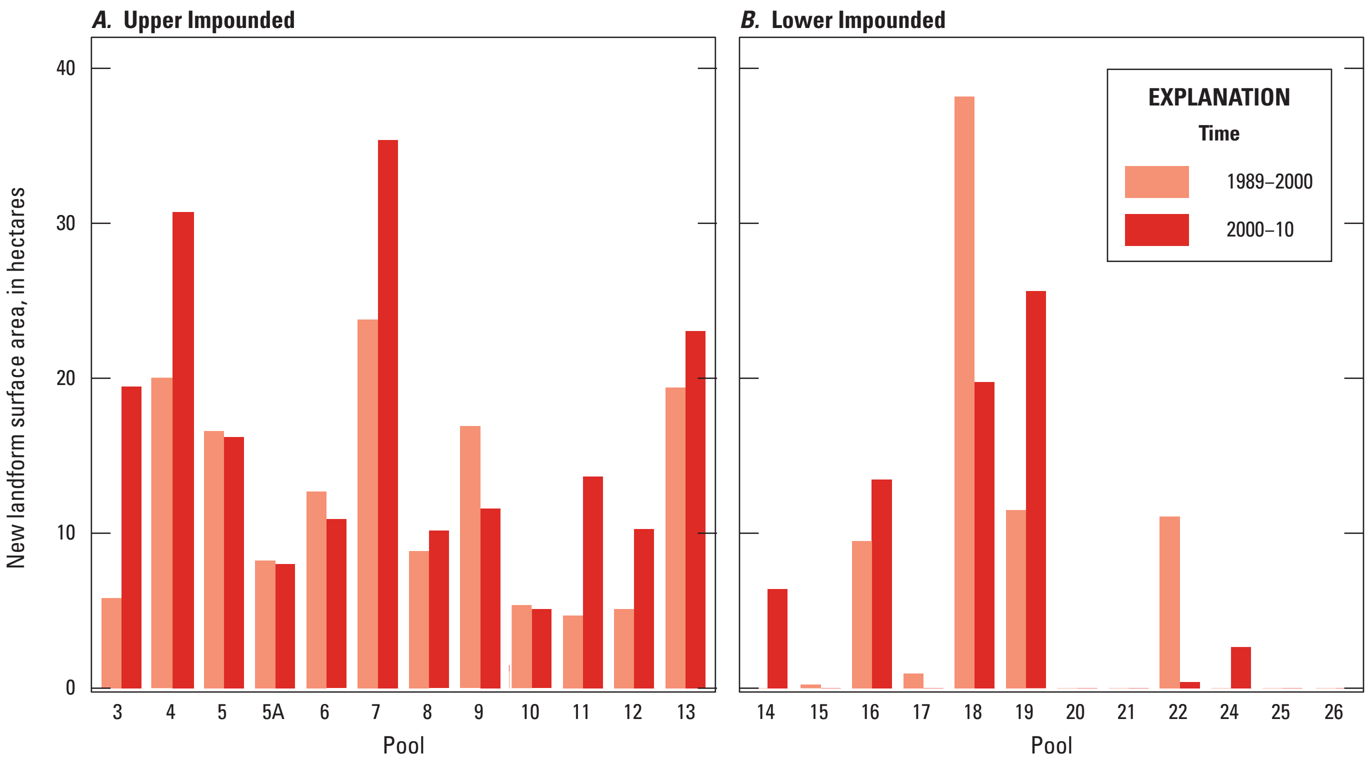


Figure C2. Graphs showing total surface area gained in hectares for all four landform features combined (bar-tail limbs, crevasse deltas, tributary deltas, and impounded deltas) across two periods (1989–2000 and 2000–10) for *A*, Pools 3–13 and *B*, Pools 4–26. X-axis indicates navigation pools of the Upper Mississippi River.

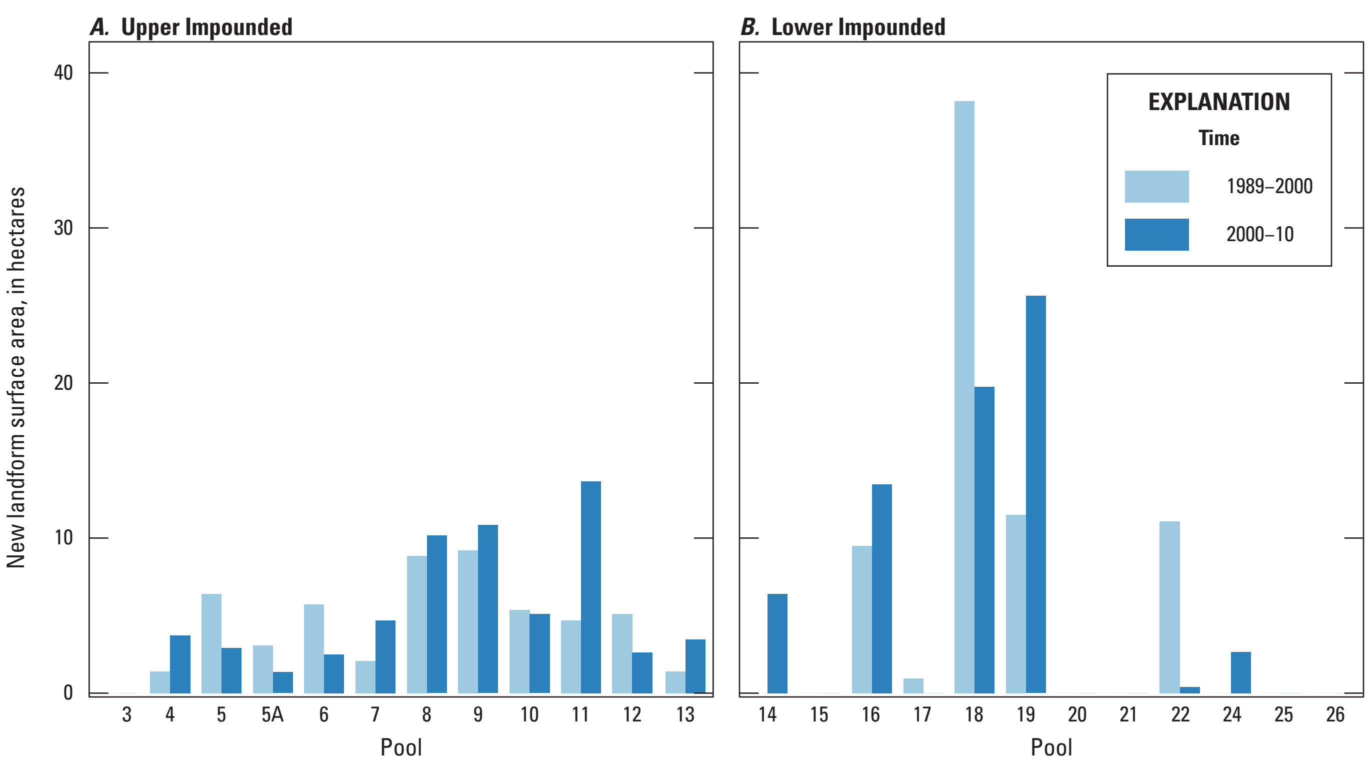


Figure C3. Graphs showing surface area gains of bar-tail limbs in *A*, Pools 3–13 and *B*, Pools 14–26 across two periods (1989–2000 and 2000–10). X-axis indicates navigation pools of the Upper Mississippi River.

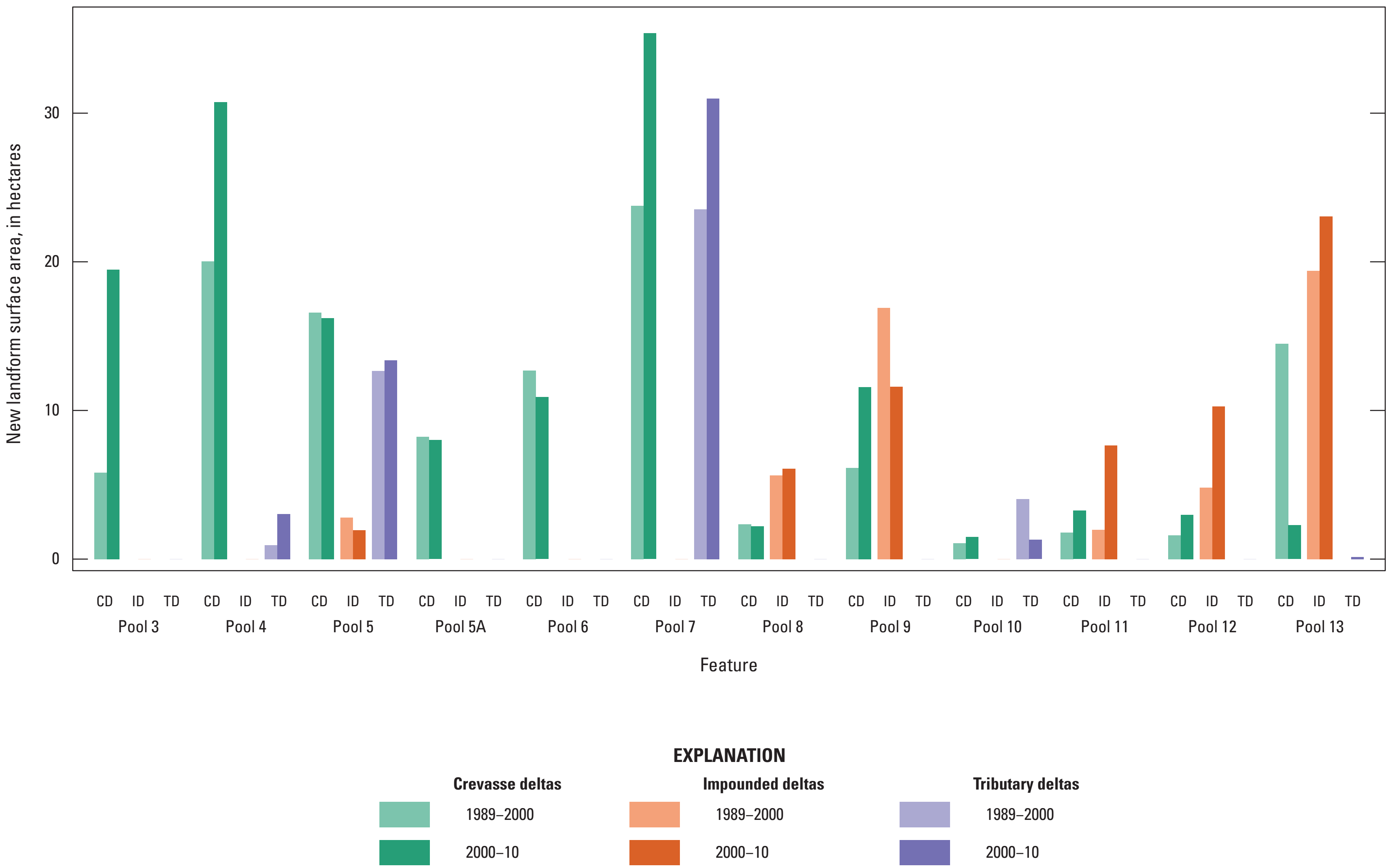


Figure C4. Graph showing surface area gains of crevasse deltas, impounded deltas, and tributary deltas in Pools 3–13 across two periods (1989–2000 and 2000–10). CD, crevasse deltas; ID, impounded deltas; and TD, tributary deltas.

Methods

Bed elevations were measured along transects spanning randomly selected backwater lakes from Pools 4 and 8 as described by Rogala, Kalas, and Burdis (2020). Thirty-eight random backwater lake transects initially sampled in 1997 from Pools 4 and 8 were revisited in 2017 or 2018 (fig. C5). Bed elevation changes were reported as mean annual rates, in centimeters per year, to standardize measurement across 20 or 21 years of record. Rates were summarized as pool-wide backwater means, individual transect means, and means within terrestrial and water depth classes (terrestrial, 0–0.25 meter [m], greater than 0.25–0.5 m, and greater than 0.5 m). Positive change reflects increases in bed elevations and, thus, implies net sedimentation and loss of backwater lake depth; negative change reflects decreases in bed elevation and indicates net erosion and gain of backwater lake depth. See Rogala, Kalas, and Burdis (2020) for detailed methods.

Assessment

Status

Pool-wide mean backwater bed elevation increased from 1997 to 2017/2018 in both pools: 0.27 (standard error =0.06) centimeter per year (cm/yr) in Pool 4 and 0.51 (standard error=0.08) cm/yr in Pool 8 (table C1). Patterns of bed elevation change exhibited high spatial variance: rates differed within and among pools (fig. C6; table C1). On average, near-shore terrestrial areas experienced a mean elevation decrease of 0.11 cm/yr in Pool 4 and a net increase of 0.15 cm/yr in Pool 8. The greatest gains in bed elevation were in backwater areas deeper than 0.50 m, where sediment accumulated at rates of 0.34 and 0.62 cm/yr in Pools 4 and 8, respectively (fig. C6). Rates of bed elevation change also varied among transects (table C1). Mean rates detected in Pool 4 transects ranged from –3.25 to 1.30 cm/yr and from –0.52 to 1.40 cm/yr in Pool 8. Most transects experienced overall increases in elevation, but some experienced decreases in elevation (table C1). Rates of bed elevation change tended to be greatest in deeper areas indicating that these areas were experiencing more net sediment deposition through time than other areas, although there were exceptions. For example, transects 11R, 1R, and 5R in Pool 4 and 1B, 1R, and 5R in Pool 8 (fig. C5) exhibited decreasing rates at greater depths (table C1), meaning that more sediment deposition was in shallower locations than in deeper locations. Greater sedimentation in topographic lows is consistent with observations in other lentic systems (Håkanson 1977; Hilton and others, 1986).

Trends

Annual rates of bed elevation change in 1997–2017/2018 were generally lower than many previously reported rates for Upper Mississippi River backwater lakes, some of which were as high as 4 cm/yr (for example, McHenry and others, 1984). These discrepancies may be due to differences in sampling design among studies, changes to the sedimentation rates over time, or hydrologic variability across different study periods (Rogala, Kalas, and Burdis, 2020). However, the rates reported here are greater than those from a previous 5-year study along the same transects by Rogala and others (2003). Rogala and others (2003) noted that the magnitude and variability of flow conditions, particularly in 2001 that had the second largest flood crest of record in Pool 8 since 1959, likely contributed to their observed rates of bed elevation change. High-discharge years can be associated with lower sediment accumulation due to scouring, reduced sediment trapping efficiency, and particle size distributions in backwater lakes (Rogala, Kalas, and Burdis, 2020). The authors conclude that the potential effects of very high discharges in 2001 may have been very influential on the 5-year study mean because it was the last year of analysis; the effects of this single year high flow event were likely less influential on the 20-year mean reported here, resulting in greater rates of sediment accumulation over the 20-year period compared to the 5-year period (Rogala, Kalas, and Burdis, 2020).

Changes in bed elevation indicate changes in the water depth of backwater lakes. Increased bed elevation indicates a loss of water depth in backwaters from 1997 to 2017/2018. Using the same bed elevation change measurements reported here, Rogala, Kalas, and Burdis (2020) calculated that five transects in Pool 8 lost more than 40 percent of their depth over the 20-year period, and two of them lost greater than 75 percent of their depth. An additional five transects in Pool 4 and eight transects in Pool 8 exhibited a loss of more than 20 percent (Rogala, Kalas, and Burdis, 2020). These losses indicate many backwater lakes are likely to become shallower over the next 50 years, although some lakes where low rates were observed may be expected to exhibit only small losses of depth.

State of Ecosystem

The two indicators reported here document recent geomorphic changes in nonaquatic and aquatic habitats. New landform surface area has been relatively consistent since 1989. New surface areas were largely in the Upper Impounded Reach (Pools 3–13), mostly in the form of crevasse deltas and bar-tail limbs. The amount of new area and the type of landform varied by pool, however, and some pools gained no new surface area over the period of analysis. Losses due to erosion were not assessed in this report but would be needed to understand the net effects of sedimentation and erosion processes on the distribution of land and water in the floodplain of the Upper Mississippi River System. Aquatic areas have also experienced recent geomorphic changes as indicated by overall increases in backwater bed elevations in Pool 4 and Pool 8 since 1997. This increase varied spatially: the pool-wide mean rate of bed elevation change was nearly twice as large for Pool 8 compared to Pool 4 (0.51 cm/yr versus 0.27 cm/yr), and rates of bed elevation change varied within

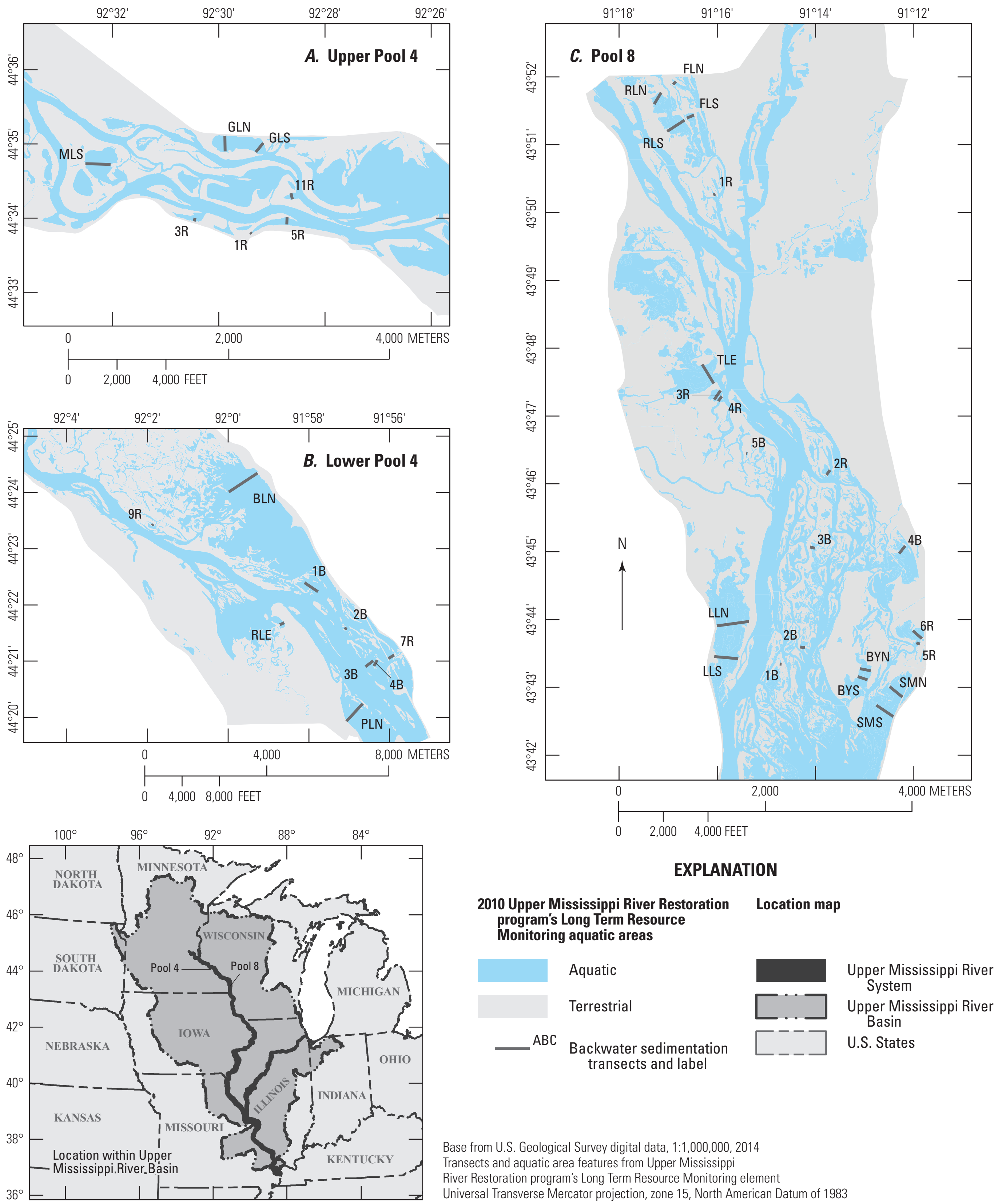


Figure C5. Maps showing backwater transects from *A*, Upper Pool 4, *B*, Lower Pool 4, and *C*, Pool 8 used in this analysis. Transect names are referenced in table C1.

Table C1. Mean bed elevation change rates in centimeters per year for backwater lake transects and four depth classes along the transects.

[Standard error is reported for pool-wide mean rates. The upper part of the table summarizes the number of transects showing different sedimentation patterns by depth class. Light brown, positive values indicate elevation increases (sediment deposition, reduced site depth) dark brown, negative values indicate elevation decreases (sediment erosion, increased site depth); cm/yr, centimeter per year; m, meter; >, greater than; SE, standard error; n.d., no data]

Rates of bed elevation change (cm/yr) from 1997 to 2017	Depth class: Terrestrial	Depth class: 0–0.25 m	Depth class: >0.25–0.5 m	Depth class: >0.5 m
Total number of transects: erosion	10	4	4	6
Total number of transects: deposition	23	29	33	22

Pool	Pool mean	Transect	Transect mean	Depth class mean: Terrestrial	Depth class mean: 0–0.25 m	Depth class mean: >0.25 m–0.5 m	Depth class mean: >0.5 m
4	0.27 (SE=0.06)	11R	0.70	0.98	0.29	0.27	0.20
		1R	0.28	0.45	−0.20	−0.05	−0.35
		2B	−0.81	−1.86	−1.17	0.09	0.95
		3B	−1.52	−3.25	0.05	−2.42	−1.21
		3R	0.05	0.06	−0.02	n.d.	n.d.
		4B	0.17	0.08	−0.16	0.29	1.13
		5R	1.08	1.30	0.73	0.61	0.56
		7R	0.56	0.61	0.23	0.27	0.92
		9R	0.29	0.18	0.31	0.47	0.56
		GLN	0.14	−0.16	0.03	0.27	0.60
		GLS	0.08	−0.22	0.07	0.18	0.40
		MLS	−0.05	−0.23	0.14	0.20	n.d.
		RLE	0.30	0.31	0.20	0.38	1.10
		BLN	0.04	n.d.	0.00	0.01	0.05
		PLN	1.23	n.d.	n.d.	0.38	1.27
		1B	0.57	n.d.	n.d.	0.64	0.55
8	0.51 (SE=0.08)	1B	0.25	0.37	0.21	0.07	−0.09
		1R	0.11	0.09	0.26	0.02	n.d.
		2B	0.18	0.04	0.17	0.87	1.01
		2R	0.23	−0.09	0.48	0.95	1.08
		3B	0.58	0.38	0.85	1.30	n.d.
		3R	0.26	0.13	0.18	0.44	0.67
		4B	0.23	0.18	0.26	0.61	n.d.
		4R	0.00	−0.24	0.20	0.59	0.83
		5B	0.57	0.46	0.89	0.96	n.d.
		5R	0.12	0.07	0.29	−0.23	−0.52
		6R	0.25	0.16	0.25	0.41	n.d.
		BYN	0.32	0.03	0.26	0.63	0.98
		BYS	0.30	−0.14	0.07	0.36	1.08
		FLN	0.13	0.05	0.25	0.11	n.d.
		FLS	0.38	0.38	0.17	0.64	n.d.
		LLN	−0.09	−0.38	0.14	−0.04	−0.05
		LLS	0.46	0.51	0.63	0.17	0.20
		RLN	0.27	0.21	0.12	0.49	n.d.
		RLS	0.11	0.04	0.01	0.24	0.36
		SMN	1.12	n.d.	n.d.	0.31	1.31
		SMS	1.34	n.d.	n.d.	0.13	1.40
		TLE	0.00	−0.36	0.23	0.51	0.24

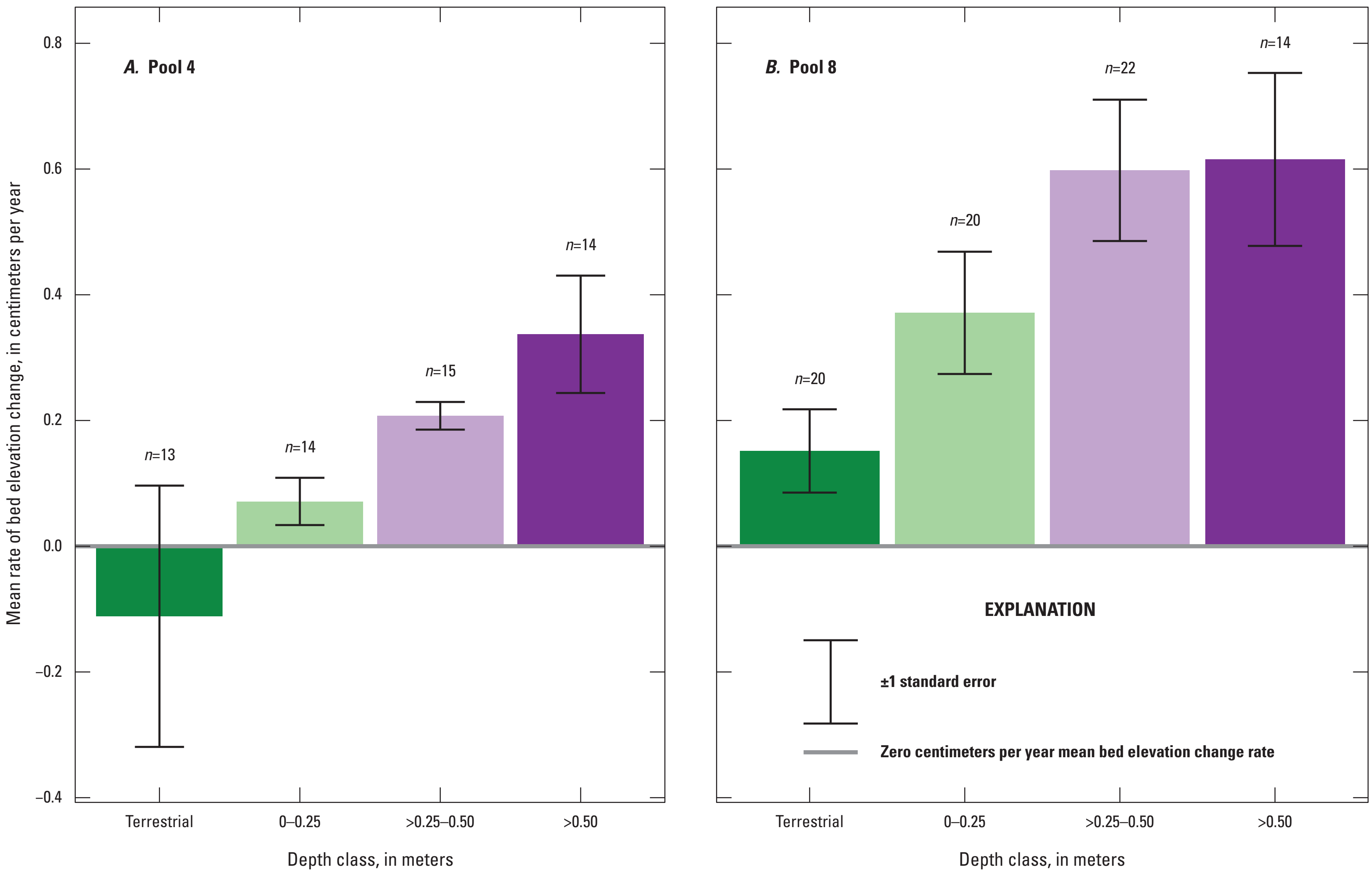


Figure C6. Bar charts showing mean rates of bed elevation change for backwater transects in *A*, Pool 4 and *B*, Pool 8 for four water depth classes. Data reflect geomorphic change from 1997 to 2017/2018. Error bars represent the standard error of the mean, and numbers above each bar are the sample sizes. Positive values indicate increases in bed elevation (net sediment deposition); negative values indicate losses in bed elevation (net sediment erosion). ±, plus or minus; >, greater than.

sampling transects such that deeper areas tended to increase more than shallower areas. Taken together, these two types of geomorphic changes—the incursion of new landform surface area, particularly tributary and crevasse deltas, and increases in backwater bed elevations—imply a gradual and consistent loss of backwater aquatic areas through time.

The geomorphic changes observed using these two indicators have implications for physical aspects of the river system. For example, hydrologic connectivity can be affected by new landform surface areas. Bar-tail limbs extending from islands decrease hydrologic connectivity between channel and off-channel waters downstream from the islands. Similarly, the impounded deltas sometimes form these limbs, but even without limbs, any landmass can decrease wind fetch and water exchange rates. In contrast, crevasse deltas can be an initial sign of higher connectivity between channel and off-channel areas because higher-velocity flows through breaches are needed to build these sandy depositional features into backwaters. Through time, however, these deltas can become self-limiting by altering within-lake flow patterns and eventually reducing connectivity with source channels as they build. Like crevasse deltas, tributary deltas can also alter within-lake flows if the new landmasses divert flow into different areas within the lake. Changes in backwater bed elevations can also change the spatial distribution of flow velocities within backwaters and can potentially change water residence times. In addition, these changes can be used to estimate where deep-water habitats may be affected by fine sediment deposition at faster rates than analyses of land expansion alone.

New landform surface area, although representing less than 1 percent of the existing land area in the floodplain, has the potential to be ecologically significant because of where it occurs and the type of habitat it creates. For example, the new tributary and crevasse deltas forming in backwaters cause a loss of deep backwater habitat while at the same time creating exposed, sandy habitats preferred by some biota such as shorebirds and waterbirds. Such sandy habitats are not common in navigation pools because of the minimum water levels that are maintained for navigation. They are also often rare in off-channel areas that typically accumulate fine sediments. A high percentage of the new terrestrial area was occupied by young willow stands (Rogala, Fitzpatrick, and Hendrickson 2020), an underrepresented vegetation type that is important for forest successional processes. This early successional state is a unique habitat that is reliant on bare mineral soil in new feature formations. In the future, these new terrestrial areas may help sustain the succession and expected evolution of diverse terrestrial and aquatic habitats in the river.

The rates of backwater bed elevation change also have the potential to be ecologically significant. Increased backwater bed elevation in the aquatic parts of the sampling transects during the 20-year study period indicates rates of sedimentation exceeding that of erosion and, on average, backwater depth has decreased over time. In addition, deeper depth classes tended to have higher rates of sedimentation. Sedimentation rates in deep areas are of most concern to resource managers (McCain and others, 2018) because additional filling can reduce the abundance of this valued habitat. In addition to concerns about the filling of the deepest areas of backwaters, loss of water surface area (in other words, terrestrial encroachment) may also be a management concern in some areas. Although the methods used to survey backwater bed elevations were not conducive to accurately measuring aquatic/terrestrial transitions, Rogala, Fitzpatrick, and Hendrickson (2020) concluded overall loss of aquatic area due to sedimentation was less than 1 percent (about 1.5 meters per transect over the 20-year sampling period).

The deposition of sand that results in new land area associated with tributary, crevasse, and impounded delta formations (new landform surface area indicator) is likely longer lasting than deposition of fine sediments in backwaters (backwater bed elevation change indicator). It is less likely that these sandy, terrestrial depositional areas will experience substantial erosion, especially when they are colonized rapidly by vegetation. In contrast, accumulated fine-grained sediment (silts and clays) in backwaters may be eroded by river currents during high flow events and wind-wave action more often and be transported farther downstream. In fact, backwater filling with fine sediments can occur at low long-term rates in part due to scouring during high-discharge years that removes some accumulated sediment (Rogala and others, 2003). The years in which fine-grained sediment are scoured are likely the same years in which sands would be deposited as deltas. Delta-building processes may eventually be somewhat self-limiting because sediment deposition and vegetation growth in the deltas may impede inflows from side channels and levee breaches.

New landform surface area and backwater bed elevation change indicators were not available for the Unimpounded Reach and the Illinois River in this study, but other studies have documented geomorphic changes in nonaquatic and aquatic habitats in these reaches. For example, qualitative comparisons between historical imagery in these reaches show small differences in land surface area over time, mostly associated with side-channel filling in the Unimpounded Reach and some conversion of backwater areas to terrestrial surface area in parts of the Illinois River (WEST Consultants Inc., 2000). Bed elevation changes in Illinois River backwaters have reduced lake volumes (Lee and Stall, 1976), a trend that is expected to continue in many parts of the Illinois River (Bellrose and others, 1983; Demissie and Bhowmik, 1986; Demissie and others, 1992). These patterns of backwater loss are qualitatively similar to those reported here.

Future Pressures

The geomorphology of the Upper Mississippi River System and its changes over time reflect ongoing responses to hydrology (ch. B), shifts in catchment land-cover patterns (ch. D; for example, Belby and others, 2015), and river modifications that support commercial navigation (for example, WEST Consultants Inc., 2000). Sediment and water interact within a river system in complex ways to affect the amount, quality, and configuration of aquatic and floodplain habitats (Gregory and others, 1991; Malanson, 1993; Ward, 1998; Montgomery, 1999; Poole, 2002; Stanford and others, 2005; Beechie and others, 2008). These interactions take place across a range of spatial and temporal scales, and often in nonlinear ways (Wohl and others, 2015). In addition, human alteration of biophysical processes has further transformed sediment dynamics in river systems, including the Upper Mississippi River System (WEST Consultants Inc., 2000; Fremling, 2005). As a result, predicting future geomorphic change and its implications for biota is challenging (Poole, 2002; Beechie and others, 2008), and this complicates river management decisions (Wohl and others, 2015).

The two indicators presented here, new landform surface area and backwater bed elevation change, describe patterns of recent geomorphic change but can also provide insights regarding potential future trajectories. For example, places where large surface area gains have occurred, such as delta formations supplied by the Chippewa River in Pool 4, might be expected to continue building landform features in the near-term because tributaries and channel connections are expected to continue to deliver sediments (WEST Consultants Inc., 2000). Landform areas experiencing little or no geomorphic change may remain in a stable state, but they may also experience natural levee breaches and the formation of new crevasse deltas. Evidence of such breaches over the period of this study was rarely found but observed increases in discharges (ch. B) and water levels may contribute to increased probability of future breaches. The changes in backwater bed elevation reported here and in Rogala, Kalas, and Burdis (2020) can be used to detect the location and prevalence of recent landmass formations and roughly determine where underwater accumulation of fine sediment may be altering deepwater habitats at faster rates than land expansion would indicate. Rogala, Kalas, and Burdis (2020) reported that the percentage of water depth lost on account of net sedimentation was high for many backwater lakes (for example, over 75 percent of depth lost). The magnitude of these losses indicates many lakes will become extremely shallow in the next 50 years, although some lakes that have low rates of net sedimentation were observed and for the most part are expected to retain their depths. However, predicting the long-term patterns of sediment deposition and erosion is difficult due to the potential for threshold changes and feedbacks that could result in abandoned depositional sites or new areas of active erosion. Continued monitoring of geomorphic changes, as well as additional research on geomorphic processes in the Upper Mississippi River System, would be important for developing management strategies aimed at maintaining desired conditions in this complex river system.

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Chapter D: Land Cover Indicators

By Nathan R. De Jager and Jason J. Rohweder

Chapter D of

Ecological Status and Trends of the Upper Mississippi and Illinois Rivers

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Chapter D: Land Cover Indicators

By Nathan R. De Jager and Jason J. Rohweder

Overview

At broad scales, such as the navigation pool, reach, or system, the abundance and distribution of physical features, such as leveed areas and floodplain forests, provides information about how the Upper Mississippi River System ecosystem is structured and how it functions (De Jager and others, 2018). The spatial patterns of such features and their temporal dynamics typically reflect even broader scale processes operating at regional to global scales (for example, hydrology and land use). Further, spatial patterns of such physical features often affect how local-scale processes play out in time and space (for example, nutrient concentrations and cycling rates, vegetation community composition, and population dynamics). In other words, landscape patterns connect changes in regional to global-scale driving processes with local-scale ecological properties and processes, which are often of importance to management agencies. As a result, river habitat rehabilitation projects often seek to modify such physical features to affect the biological productivity and diversity of river systems. In this chapter, we investigate the status and trends in leveed areas and floodplain forests of the Upper Mississippi River System. Area calculations were made at the scale of floodplain reaches (see ch. A) and navigation pools to summarize system-level distributions, changes, and variations within floodplain reaches.

Indicator: Leveed Area

For large rivers with expansive floodplains, the amount of area behind levees provides important information about how they function. Areas separated from the river by levees are generally not connected to the waters flowing in the main channel, whereas areas that are not protected by levees may become inundated during high-flow events. Such differences in connectivity can affect how effectively the floodplains of large rivers store water and capture sediments and nutrients, therefore reducing eutrophic conditions in downriver and coastal areas. In addition, levees reduce potential access to critical spawning, nursery, forage, and refuge habitats for some fish and other aquatic species.

Methods

The measurements in this section use Long Term Resource Monitoring element land cover/use maps and maps of areas behind Federal Levees. Land cover/use maps are described in Dieck and others (2015) while the Federal levee data set is described in De Jager and others (2018). For the analysis, maps of the area behind Federal levees in 1994 and 2010 were overlain by the land cover/use maps developed for the years 1989 and 2010. The data for both time periods were standardized by the common spatial extent of the Upper Mississippi River System. The slight mismatch in time periods was due to the fact that the levee datasets are continually updated and therefore reflect constant changes (if or where they occurred) in Federal levee coverage up to the date of data download. The 1994 levee data were downloaded around 1994 and saved on a server at the Upper Midwest Environmental Sciences Center, while the 2014 data were downloaded in 2015. Neither dataset accounts for private or State levees and is therefore a conservative estimate of leveed area at the time of data download. Still, the Federal levees that are mapped likely make up most leveed areas within the Upper Mississippi River System and represent areas that are hydrologically disconnected from the river. More information on each dataset can be found in De Jager and others (2018).

Assessment

Status

The Upper Impounded Reach (Pools 1–13) contains few levees (about 5-percent of total reach area) and therefore maintains a high degree of river-floodplain connectivity and an abundance of temporarily flooded habitat (fig. D1 and D2). Notable deviations from floodplain reach-wide estimates in the Upper Impounded Reach were found for Pools 5, 6, and 13, which had more than 10-percent of total area behind levees. In the Lower Impounded Reach, approximately 45-percent of the total area was behind levees, although Pools 14–16 had much less area behind levees than the rest of the reach (less than 25-percent). Farther south, in the Unimpounded Reach, the area behind levees exceeded 55-percent of total reach area for the northern and southern reaches. Finally, the Illinois River showed similar longitudinal increases in levees as found in the Upper Mississippi River, with no detected levees

Summary of Leveed Area

Indicator intent: To provide an approximation of areas generally not connected to the waters of the Upper Mississippi River System (in other words, an index of lateral river-floodplain disconnectivity). Conversely, this measure also provides an approximation of areas available as temporarily flooded habitat (in other words, an index of floodplain habitat).

Measurement: Percentage of total navigation pool or reach area (mapped in 1989 and 2010) as leveed area (mapped circa [ca.] 1994 and 2014).

Special consideration: Levee data for ca. 1994 was used in the Upper Mississippi River Restoration Status and Trends Report II (Johnson and Hagerty, 2008) and developed by a scientific assessment and advisory team for the Upper Mississippi River System Habitat Needs Assessment (Theiling and others, 2000). The ca. 2014 levee data were compiled from the National Inventory of Dams for use in the Upper Mississippi River Restoration Habitat Needs Assessment II (McCain and others, 2018). Neither dataset accounts for private nor State levees. For more information on each dataset, see table 3 in De Jager and others (2018).

Assessment

Status: The percentage of total river and floodplain area (total reach area) behind levees generally increased from upstream to downstream in the Upper Mississippi River System (table D1). The Upper Impounded Reach had approximately 5-percent of total reach area behind levees, compared with approximately 46-percent in the Lower Impounded Reach, and approximately 59-percent in the Unimpounded Reach. The Illinois River had approximately 38-percent of total reach area behind levees.

Trends: From ca. 1994 to ca. 2014, the area behind levees decreased by 2,036 hectares (ha) (1.1-percent of total reach area) in the Upper Impounded Reach and 4,390 ha (1.5 percent of total reach area) in the Lower Impounded Reach and increased by 3,403 ha (2.2-percent of total reach area) in the Unimpounded Reach. In the Illinois River, the area behind levees decreased from 1989 to 2010 by 2,941 ha or 1.5-percent of total reach area (table D1).

upstream from the Starved Rock Pool (Sta) and with levees exceeding 40-percent of total navigation pool area below Peoria (Peo) (fig. D2).

Trends

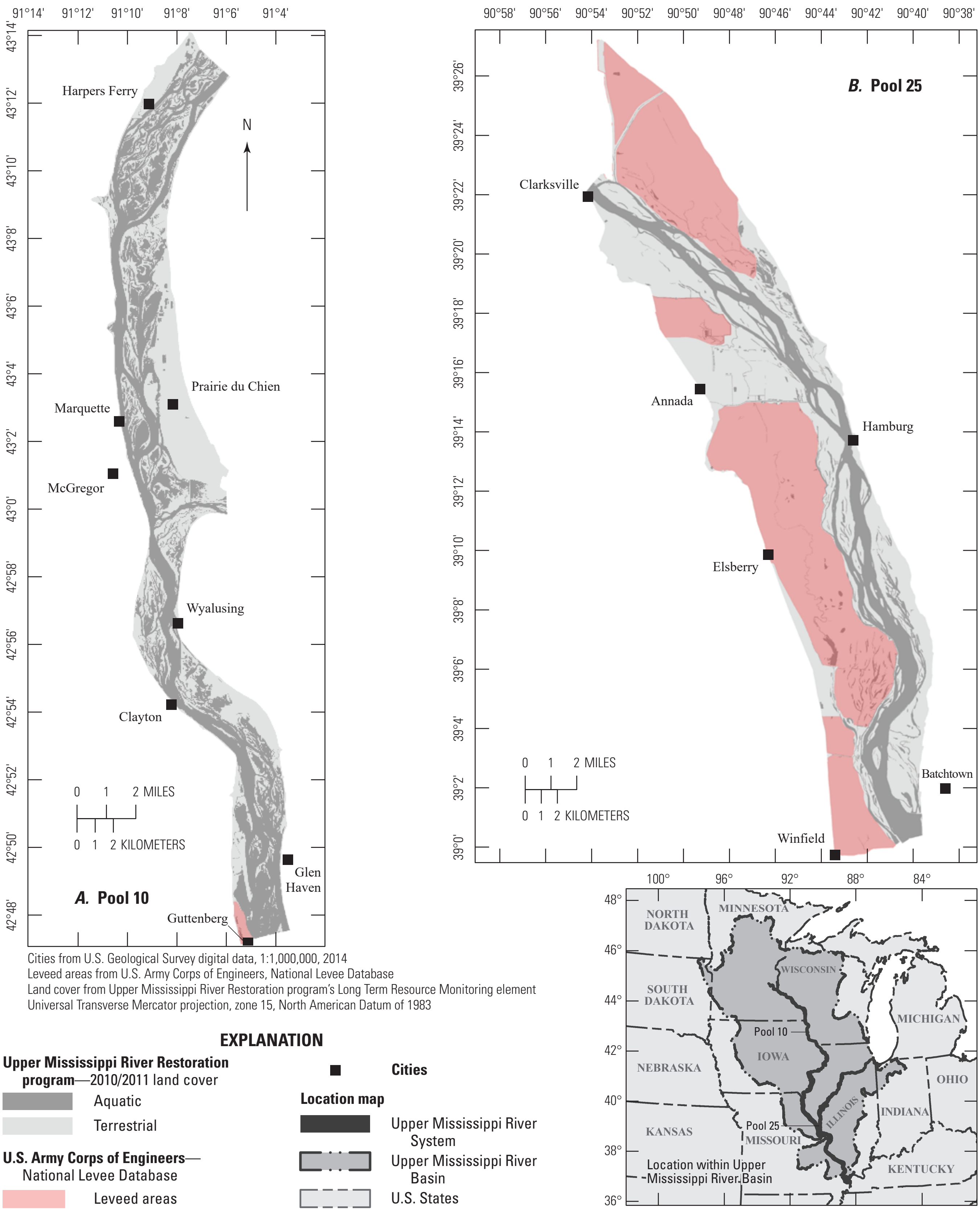
Leveed area decreased by 2,036 hectares (ha) in the Upper Impounded Reach between 1989 and 2010. These decreases primarily occurred in Pools 8 and 13. All other pools in the reach remained relatively unchanged (fig. D2). Across the entire river system, leveed area remained relatively unchanged between 1989 and 2010 for 22 of the 36 navigation pools or reaches (1, 2, 3, 4, 5a, 6, 7, 9, 10, 11, 12, 16, 17, 19, 20, 21, Lockport [Loc], Branden [Bra], Dresden [Dre], Marselles [Mar], Sta, and the northern section of Open River [or1] see fig. A1 for locations of pools). Increases in leveed area between 1994 and 2014 were found in just 3 of the 36 pools (5, 26, and the southern section of Open River [or2]). Decreases in leveed area between circa 1994 and circa 2010 were found for 11 of the 36 pools (8, 13, 14, 15, 18, 22, 24, 25, Peo, La Grange [Lag], and Alton). Such decreases ranged from 1.2 percent of total navigation pool or reach area in Peo to 10.9 percent in Pool 25.

Table D1. Percentage of total reach area behind levees in 1989 and 2010, along with the percentage difference between the two dates, for four floodplain reaches of the Upper Mississippi River System.

Floodplain reach	Percentage of total reach area		
	1989	2010	Difference
Upper Impounded	6.0	4.8	−1.1
Lower Impounded	47.0	45.4	−1.5
Unimpounded	57.1	59.3	+2.2
Illinois River	38.7	37.1	−1.5

Discussion

Our analysis of levee distributions in the Upper Mississippi River System from 1989 to 2014 revealed that the Upper Impounded Reach and the upper part of the Illinois River have few levees and maintain a well-connected system of aquatic and floodplain habitats. In contrast, the lower Illinois River, Lower Impounded Reach, and Unimpounded Reaches have ~50-percent or more of their floodplain behind levees. However, as shown by De Jager and others (2018), the overall size of the floodplain increases in the lower river so that there remain large areas of floodplain habitat connected to the river even if the proportion of connected area is smaller than in the upper river. From 1989 to 2010, changes in leveed areas were relatively minor, indicating that the distribution of levees in the system continues to maintain a strong spatial gradient across the system. The differences in levee abundances



Cities from U.S. Geological Survey digital data, 1:1,000,000, 2014
Leveed areas from U.S. Army Corps of Engineers, National Levee Database
Land cover from Upper Mississippi River Restoration program's Long Term Resource Monitoring element
Universal Transverse Mercator projection, zone 15, North American Datum of 1983

EXPLANATION

Upper Mississippi River Restoration program—2010/2011 land cover
- Aquatic
- Terrestrial

U.S. Army Corps of Engineers—National Levee Database
- Leveed areas

Cities

Location map
- Upper Mississippi River System
- Upper Mississippi River Basin
- U.S. States

Figure D1. Mapped areas behind levees circa 2014 for *A*, Pool 10, which has very few levees and *B*, pool 25, which has many levees.

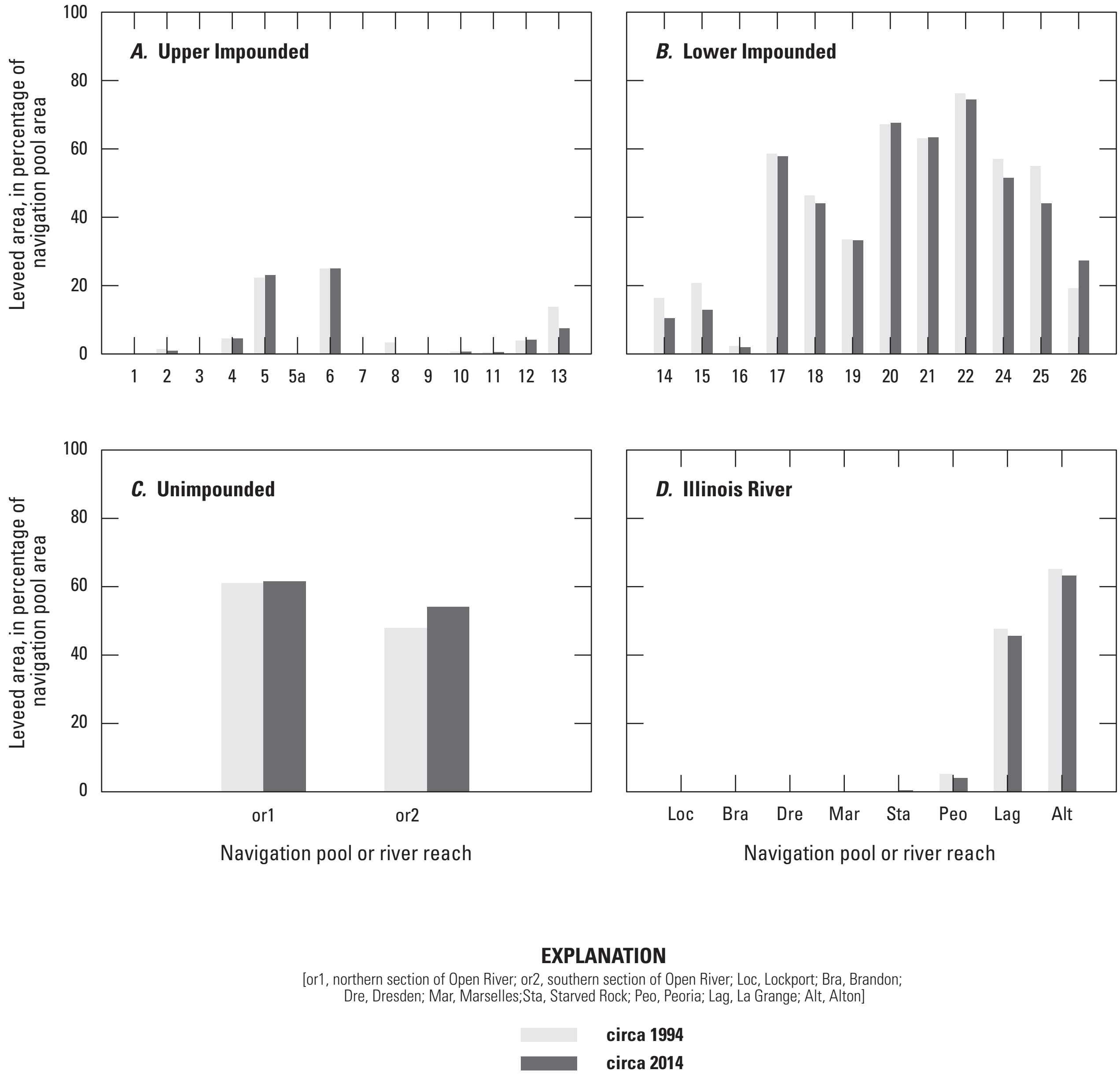


Figure D2. Status and trends circa 1994 to 2014 for all Federal levees along the Upper Mississippi River System, represented as the percentage of pool area behind levees. *A*, Pools 1–13; *B*, Pools 14–26; *C*, Unimpounded Reach; and *D*, Illinois River.

have ecological implications for various aspects of the Upper Mississippi River System ecosystem. For example, the abundance of levees coincides with different navigation infrastructures and land cover attributes that further affect aquatic habitats within the Upper Mississippi River System (De Jager and others, 2018). The Upper Impounded Reach and the Illinois Rivers are defined by extensive lentic aquatic areas, whereas the Lower Impounded and Unimpounded Reaches are defined by lotic conditions. The abundant lentic areas in portions of the Upper Impounded Reach and the Illinois River often contain suitable conditions for aquatic vegetation, and indeed aquatic vegetation is much more abundant in these areas than the rest of the system (see ch. F). In addition, the mass per unit effort of lentic fishes in Pools 4, 8, 13 is at least an order of magnitude larger than found in Pools 26 and the Unimpounded Reach (see ch. G). The lower abundance of lentic fish in the lower, leveed, and channelized portions of the river system indicates that fish species composition is dominated by lotic fishes there.

Floodplain plant communities, including forest distributions, are highly dependent on levee distributions. The same areas protected by levees and currently in agricultural land uses in the lower river were once forested areas, indicating that there was a much higher abundance of forest cover in the

Upper Mississippi River System before levee construction. Without levees, such areas would be prone to flooding and would therefore support natural floodplain vegetation as opposed to anthropogenic land cover types.

Indicator: Forest Area

Forests make up most of the floodplain vegetation in the Upper Mississippi River System (De Jager and others, 2018). This is particularly significant, considering the Upper Mississippi River System has experienced large decreases in forest cover dating back to the 1800s owing to the expansion of agriculture and urban communities and additional decreases of forest cover in the 1930s and 1940s due to lock and dam construction (De Jager and others, 2013). As forest area is lost from ecosystems, the remaining forested areas often become increasingly fragmented, isolated from each other, and with more edge per unit area (Kupfer, 2006). These structural changes can favor species that prefer edges as opposed to the interior of large blocks of forest, increase the susceptibility of forests to invasion by exotic species, alter dispersal patterns of animal and plant populations, and subdivide populations, in some cases leading to loss of genetic and species diversity (Zuidema and others, 1996; Laurance and others, 2001; Weathers and others, 2001; Lindenmayer and Franklin, 2002; Harper and others, 2005; Ramaharitra, 2006). The intent of the forest cover indicator is to provide an estimate of total forest cover and to examine spatial and temporal changes among forest types of varying degrees of fragmentation (see below) to identify where and how forest cover in the Upper Mississippi River System might be changing in terms of area and fragmentation.

Methods

Forest cover for the entire Upper Mississippi River System was mapped using aerial imagery collected by the Long Term Resource Monitoring element in 1989, 2000, and 2010. We aggregated the following forest classes to quantify total forest cover: Salix community, Populus community, floodplain forest, and lowland forest. From these maps, the percentage of area surrounding each 1-meter forested pixel was calculated within a 10-ha circular window. Thresholds were then developed to classify forested pixels based on the percentage of forest cover in the surrounding neighborhood (10-ha window). Core forest pixels were those that were

Summary of Forest Area

Indicator intent: To provide an approximation of total forest area in hectares (ha) and the degree of forest fragmentation (conversely forest habitat connectivity).

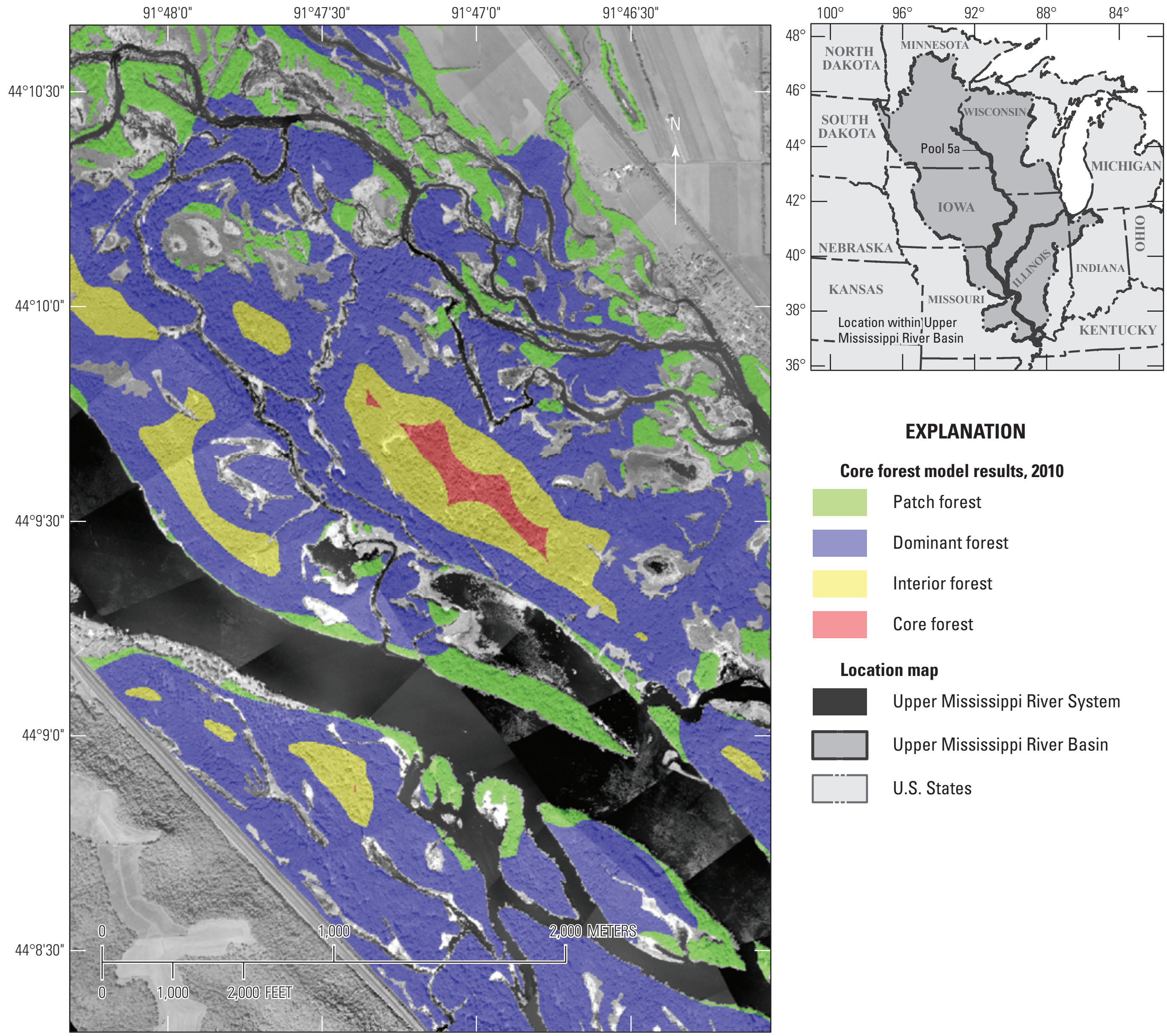
Measurement: Areas that were mapped as forest by the Long Term Resource Monitoring element in 1989, 2000, and 2010 that meet the following criteria: core forest (100-percent surrounded by forest), interior forest (between 90-percent and 100-percent surrounded by forest), dominant forest (between 50-percent and 90-percent surrounded by forest), and patch forest (less than 50-percent surrounded by forest) based on calculations made in a 10-ha circular window (fig. D3). Total forest area is the sum of all four forest types. For more details on calculation methods see De Jager and Rohweder (2011).

Special considerations: Differences in mapping units and interpretation methods among years may affect trend analysis.

Assessment

Status: Total forest area, as a percentage of total navigation reach area, ranged from a high of approximately 20.6-percent in the Upper Impounded Reach to a low of approximately 16.8-percent in the Unimpounded Reach in 2010. However, forests in the Upper Impounded Reach were the most fragmented with approximately 90.1-percent of forest cover classified as either patch or dominant (less than 90-percent surrounded by forest). In contrast, just 77.6-percent of forest cover in the Unimpounded Reach was classified as patch or dominant, indicating that forest cover was less fragmented in this reach. The Lower Impounded Reach and the Illinois River were similar to the Upper Impounded Reach and the Unimpounded Reach in that approximately 18-percent of total navigation reach area was in forest cover and approximately 84-percent of forest cover was classified as patch or dominant.

Trends: Total forest cover decreased from 1989 to 2010 in the Upper and Lower Impounded Reaches of the Upper Mississippi River and the Illinois River (table D2). Forest loss estimates ranged from 2,508 ha in the Upper Impounded Reach (6.4-percent decrease) to 1,311 ha (3.7-percent decrease) in the Illinois River. These reaches did not show increases in any forest type, indicating little overall change in forest fragmentation. In contrast to the other reaches, forest cover in the Unimpounded Reach increased by 3,823 ha (17.3-percent increase). The amount of all forest cover types also increased in the Unimpounded Reach, indicating little overall change in forest fragmentation.



Base from U.S. Geological Survey digital data, 1:1,000,000, 2014
Aerial photography from Upper Mississippi River Restoration program's Long Term Resource Monitoring element
Universal Transverse Mercator projection, zone 15, North American Datum of 1983

Figure D3. Distributions of patch, dominant, interior, and core forest for a section of the Upper Mississippi River System.

Table D2. Percentage change in the area of different forest types between 1989 and 2010 for four floodplain reaches of the Upper Mississippi River System.

[%, percent]

Floodplain reach	Patch	Dominant	Interior	Core	Total
Upper Impounded	−3.9%	−7.1%	−7.9%	−31.0%	−6.4%
Lower Impounded	−9.9%	−1.2%	−2.1%	−6.9%	−4.3%
Unimpounded	+4.5%	+25.1%	+21.2%	+20.3%	+17.3%
Illinois River	−7.2%	−1.1%	−8.9%	+7.4%	−3.7%

nested in a 100-percent forested neighborhood, which is relevant for species or processes that require extremely dense forest cover. Interior forest pixels were surrounded by greater than 90-percent but less than 100-percent forest cover and represent slightly fragmented forest. Dominant forest pixels were those surrounded by greater than 50-percent forest cover but less than 90-percent and were therefore more fragmented than interior forest. Finally, patch forest pixels were those surrounded by less than 50-percent forest cover and were therefore most fragmented. See figure D3 for example. More information regarding these methods can be found in De Jager and Rohweder (2011).

Assessment

Status

At the floodplain reach scale, the Upper Impounded Reach contained the most floodplain forest cover as a percentage of reach area (20.6-percent), followed by the Lower Impounded Reach and the Illinois River, which similarly contained 18-percent forest cover, and the Unimpounded Reach (16.8-percent) (fig. D4). Forests in the Upper Impounded Reach were the most fragmented with approximately 90.1-percent of their forest cover classified as either patch or dominant (less than 90-percent surrounded by forest) and the remaining 9.9-percent classified as core or interior forest cover (greater than 90 percent surrounded by forest). This pattern is the result of a highly braided river channel network with extensive backwater lakes interspersed with floodplain forests. The Lower Impounded and Illinois Rivers had 84-percent of their forest cover classified as patch or dominant with the remaining 16 percent classified as interior or core. Finally, forests in the Unimpounded Reach were the least fragmented with 22.4-percent of forest cover classified as either core or interior and 77.6-percent of forest cover classified as the more fragmented patch or dominant classes.

At the navigation pool scale, total forest area, as a percentage of navigation pool area, ranged from a low of 3-percent in Pool 1 to a high of 35-percent in Pool 5A. Across all pools and periods, total forest area was correlated with the amount of patch (r=0.62), dominant (r=0.96), and interior (r=0.66) forest cover, but not with core forest cover (r=0.28). Thus, differences in total forest area among pools and river reaches were generally a good predictor of differences in forest types. Patch forest was approximately 37-percent of total forest area for individual navigation pools, dominant forest was approximately 52-percent of total forest area, and interior forest was approximately 8.7-percent of total forest area. Core forest, on the other hand, was not predictable based on total forest area. It ranged from a low of 0-percent of pool area in the Bra (all years) and Dre [in 2010]) Pools of the Illinois River and Pools 1 (all years) and 15 (all years) of the Mississippi River, to a high of 2.3-percent of total pool area in or2. Or1 and 2; Pools 14, 20, 21, and 26 on the Mississippi River; and Peo and Lag Pools on the Illinois River generally contained the largest blocks of core forest on the Upper Mississippi River System.

Trends

At the floodplain reach scale, total forest cover decreased from 1989 to 2010 in all reaches except for the Unimpounded Reach (fig. D4). In the Upper Impounded Reach, 2,508 ha of forest were lost between 1989 and 2010 (6.4-percent decrease). In the Lower Impounded Reach, 2,229 ha of forest were lost (4.3 percent). In the Illinois River, 1,311 ha of forest cover were lost (3.7-percent). In contrast, the Unimpounded Reach gained 3,823 ha of forest cover between 1989 and 2010 (17.3-percent increase). At the floodplain reach scale, no large changes were among specific forest types indicative of changes in forest fragmentation. Rather, the loss or gain of forest cover appeared to occur in a way that maintained the present day distribution of forest cover types across navigation reaches.

Consistent with findings at the floodplain reach scale, floodplain forest cover generally either remained stable or decreased in the navigation pools of the Upper Mississippi River System, although increases were observed in a few places. Total forest area decreased between 1989 and 2010 for 14 of the 36 pools or reaches (4, 5, 5A, 6, 8, 9, 10, 11, 12, 14, 24, Loc, Bra, and Dre) and increased for just 2 (or1 and Mar). No change or initial increase or decrease, followed by a mixed outcome between 1989, 2000, and 2010, occurred for the remaining twenty pools or reaches (fig. D4).

For the 14 navigation pools that showed loss of forest cover from 1989 to 2010, 2 of them (Pools 6 and 9) showed indications of becoming less fragmented (loss of patch or dominant cover and a corresponding increase in core or interior cover), 4 pools (Loc, 5a, 8, and 17) did not change, and 8 pools (4, 5, 10, 11, 12, 24, Bra, and Dre) became more fragmented (increased in patch or dominant cover and a corresponding decrease in core or interior cover). Of the 20 pools that showed no sustained change in forest area, 6 pools showed a decrease in forest fragmentation (1, 20, 21, 22, Sta, and Peo), 9 pools showed no change (2, 3, 13, 15, 18, 19, 25, Lag, and Alton), and 5 pools showed an increase in fragmentation (7, 14, 16, 26, and or1). Finally, two pools showed increases in forest area (or1 and Mar), and both also became less fragmented.

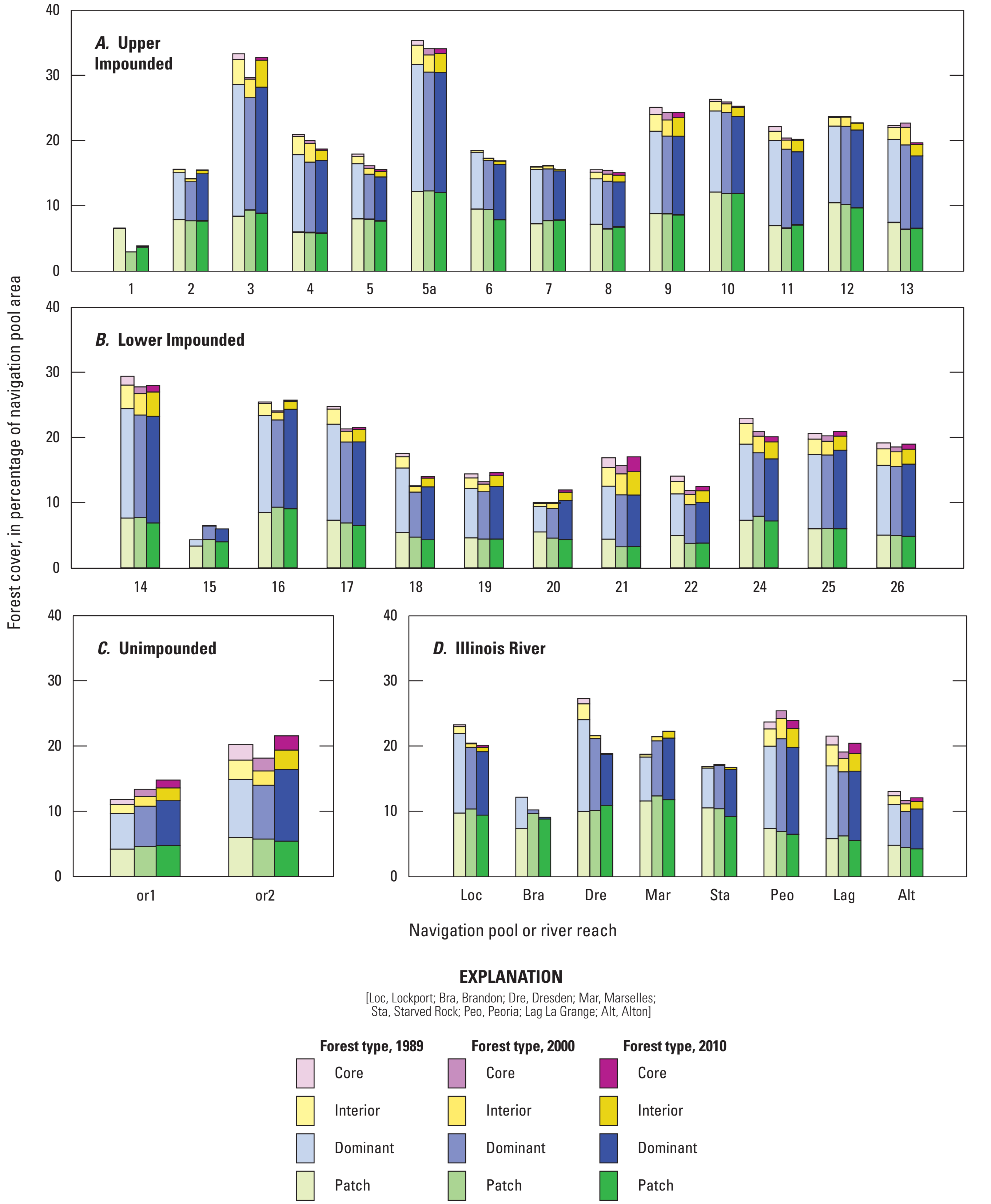


Figure D4. Total forest area and forest area meeting criteria for patch, dominant, interior, and core forest in the *A*, Upper Impounded; *B*, Lower Impounded; *C*, Unimpounded; and *D*, Illinois River reaches (see "Methods" for details).

Discussion

Forests remain the dominant floodplain land cover type in the Upper Mississippi River System, despite large decreases since pre-settlement and pre-lock and dam conditions (De Jager and others, 2013). Our data, derived from aerial imagery in 1989, 2000, and 2010, indicate that there were few places in the Upper Mississippi River System where forest area has increased over that period. Most of the system exhibited either stable or decreasing forest cover. Floodplain forests are inherently prone to fragmentation by other aquatic and terrestrial features (for example, channels, backwater lakes, and emergent marshes). However, loss relative to pre-settlement conditions indicates that the forest habitat patches were much less connected in 2010 than they were historically. Fragmentation is often correlated with forest loss, and in our analysis 12 of the 14 pools that showed forest loss also showed indications of increased fragmentation. This fragmentation was typically a shift from dominant forest to patch forest or a shift from core forest to interior forest. Forest fragmentation increases the amount of edge per unit area of forest, which can affect a range of ecological processes, such as increased pressure from invasive species that establish in high light environments along forest edges or in canopy gaps (for example, *Phalaris arundinacea* L. [reed canary grass]). Although it is possible that the effects of fragmentation might not be as severe in an ecosystem that is somewhat naturally fragmented (for example, island-braided channel system) as compared to a more upland forest system with large blocks of contiguous forests, no systematic analyses of these effects have been conducted. Regardless, our analyses signal a continued decrease in forest area and an increase in fragmentation over much of the system.

Previous land cover change analyses reported large decreases in forest cover in the Upper Mississippi River System due to the establishment of the lock and dam system (De Jager and others, 2013). The data reported here indicate a continuation of such trends for many of the pools on the Upper Mississippi River System through 2010. In addition, results from a recent simulation modelling study indicate that changes in water levels in the Upper Mississippi River System, along with the effects of the *Agrilus planipennis* (emerald ash borer), have the potential to cause additional forest losses over the next 50 to 100 years (De Jager and others, 2019). On the other hand, recent planform change analyses indicate that some new land masses could support forest cover in the Upper Impounded Reach, where approximately 575 ha of new land mass developed between 1989 and 2010 (ch. C), much of which was colonized by young willow stands. Such increases in willow cover are expected to have been accounted for in our analysis, thus, indicating that gross forest loss estimates likely exceeded 3,000 ha given that net losses in the Upper Impounded Reach were approximately 2,508 ha. Visual inspection of maps indicates that formation of new terrestrial areas may underlie some localized increases in forest cover in the unimpounded reaches, but those new land masses were not detected in our geomorphic analysis (ch. C).

State of the Ecosystem

The distribution and abundance of leveed areas and floodplain forests reflect the interplay between human-caused changes to the Upper Mississippi River System and natural geomorphic and biotic processes. Given that our results show relatively minor changes in leveed areas or floodplain forests since 1989 (the first dataset in this study), changes largely occurred before 1989, and most likely before or around the time of lock and dam construction (Fremling, 2005). Indeed, our results reflect the use of different navigation structures along the Upper Mississippi River System. In the Upper Impounded Reach, where navigation is supported with dams, we found more abundant floodplain forests (per unit area) compared with other portions of the river system. However, the higher water levels caused by the locks and dams may also contribute to the higher degree of fragmentation in this reach as forests tend to be interspersed with various floodplain water bodies and vegetation types. A somewhat similar pattern of forest cover was found in the portions of the Illinois River that contain few levees.

Future Pressures

The distribution of levees and floodplain forests is linked to variation in water levels across the Upper Mississippi River System. The presence of levees in the floodplain reflects a desire to protect certain areas from the effects of flooding. The increase in water discharge noted in chapter B, indicates that there may be increased motivation for new or higher levees in the Upper Mississippi River System for flood protection benefits. On the other hand, some organizations may seek to reconnect previously isolated areas given the prospects for higher water levels. In addition, more frequent or longer durations of higher water levels in the future may influence the distribution of floodplain forests and contribute to additional forest losses.

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Chapter E: Water Quality Indicators

By Kathi Jo Jankowski

Chapter E of
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Chapter E: Water Quality Indicators

By Kathi Jo Jankowski

Overview

Water quality plays a central role in supporting a healthy community of aquatic organisms in the riverine environment of the Upper Mississippi River System. It also is important for recreation and is key to providing healthy drinking water for cities and towns along the river corridor. Water quality in many areas of the river is sufficient to support diverse assemblages of aquatic species and migratory birds. However, sediment and nutrient-rich runoff from urban and agricultural land use and nutrient-rich groundwater continue to affect water quality conditions in many reaches of the Upper Mississippi River System, and ultimately, the Gulf of Mexico (Alexander and others, 2000; Turner and Rabalais, 2003; Sprague and others, 2011; Crawford and others, 2019).

Excess sediment and nutrient loads have affected aquatic organisms and their habitats in rivers around the globe (Val and others, 2016), including the Upper Mississippi River System. Large floodplain rivers naturally have higher loads of total suspended solids (TSS) and lower water clarity than their smaller counterparts, but several anthropogenic factors can lead to excess TSS loads. Anthropogenic sources of TSS include sediment runoff of soil from agricultural fields, erosion of riverbanks, or channelization, which reduces the potential for sediment deposition on floodplains (Hassan and others, 2017). Excess TSS concentration in the Upper Mississippi River System has contributed to filling in and reducing depth in backwater lakes (Great River Environmental Action Team, 1980; Rogala and others, 2020) and has reduced water clarity in some areas, which has affected the distribution and abundance of aquatic vegetation and animal species that rely on high visibility for foraging (ch. H,Vinyard and O'Brien, 1976; Johnson and Hagerty, 2008; Miranda and Lucas, 2011; McCain and others, 2018; Burdis and others, 2020). The organic components of TSS can also temporarily bind and transport nutrients or other contaminants that may be released under certain conditions (Wetzel, 2001).

High nutrient inputs also continue to contribute to water-quality problems in rivers globally. Nutrients, particularly nitrogen (N) and phosphorus (P), are important for riverine primary production but in excess can lead to nuisance blooms of algae and free-floating plants and eutrophication. These blooms can affect the use of the rivers for recreation, and the death and decomposition of algal biomass can cause low oxygen conditions that threaten the survival of aquatic organisms (Wetzel, 2001). Under certain conditions, often related to nutrient availability and flow dynamics, some riverine algal species can produce toxins, which can affect human and aquatic organism health. Researchers have increasingly observed toxin-producing species in large rivers throughout the United States (Graham and others, 2020). Although the Upper Mississippi River System has not had widespread issues with toxic blooms, species of cyanobacteria capable of toxin production remain dominant members of its phytoplankton community (Decker, 2015; Giblin and Gerrish, 2020; Manier and others, 2021), causing frequent, local problems in some areas. In addition, other large rivers in the region have had large, toxic cyanobacteria blooms in the last few years (Illinois Environmental Protection Agency and Department of Public Health, 2020; Ohio River Valley Water Sanitation Commission, 2021). Nutrients exported from the Upper Mississippi River System can also have effects outside the basin: high nutrient export from the Upper Mississippi River System is responsible for a large portion of the total N load to the Gulf of Mexico, which has led to long-term issues with hypoxia in the marine environment (Scavia and others, 2017).

To assess the status and trends of water quality in the Upper Mississippi River System, several indicators that describe important components of water quality that affect habitat conditions for aquatic organisms were selected (table E1). These indicators include TSS (indicative of water clarity, light penetration, and sedimentation); total nitrogen (TN) and total phosphorus (TP) (indicative of nutrients available for algal and plant growth, which in excess can cause eutrophication); chlorophyll *a* ([CHL]; measure of algal biomass which, in excess, is an indicator of eutrophication); and dissolved oxygen (DO) (necessary for survival of many aquatic organisms). For all indicators, I analyzed Long Term Resource Monitoring (LTRM) element stratified random sampling (SRS) data for trends over time. In the case of nutrients and TSS, I also used data from fixed sites in the main channel of each LTRM study reach and the mouths of 10 tributaries to model trends in concentration and flux. To do so, I used a modeling approach that statistically accounts for the effect of variation in discharge ("flow-normalization") and, by doing so, more clearly showed the effects of changes in watershed land use on river water quality.

Table E1. Summary of indicators used to assess the water quality of the Upper Mississippi River System.

[LTRM, Long Term Resource Monitoring element; <, less than; mg, milligram; DO, dissolved oxygen]

Indicator	Value	Aquatic area	LTRM dataset	Sampling frequency	Years	What it describes
Total suspended solids, total nitrogen, and total phosphorus	Annual mean concentration	Main channel	Stratified random sampling	Seasonal (winter, spring, summer, fall)	1994–2019	Conditions experienced by organisms in the river
Total suspended solids, total nitrogen, and total phosphorus	Annual mean concentration	Tributaries	Fixed sites	Bimonthly/monthly	1994–2019	Conditions experienced by organisms in the river
Total suspended solids, total nitrogen, and total phosphorus	Annual mean flow-normalized concentration	Main channel, tributaries	Fixed sites	Bimonthly/monthly	1994–2019	Variation/trends in concentration independent of fluctuations in discharge
Total suspended solids, total nitrogen, and total phosphorus	Annual mean flow-normalized flux	Main channel, tributaries	Fixed sites	Bimonthly/monthly	1994–2019	Variation/trends in flux independent of fluctuations in discharge
Chlorophyll *a*	Mean summer concentration	Main channel, backwaters	Stratified random sampling	Summer	1993–2019	Algal biomass, metric of trophic status
Dissolved oxygen	Proportion of sites <5 mg/L DO	Backwaters	Stratified random sampling	Summer, winter	1993–2019	Prevalence of hypoxic conditions that affect habitat suitability

Methods and Sampling Design

LTRM tracks water quality status and trends in six study reaches of the Upper Mississippi River System: Pool 4, Pool 8, Pool 13, Pool 26, the Open River Reach, and the La Grange Pool on the Illinois River (fig. A1). LTRM water quality monitoring includes a set of limnological variables related to habitat quality and ecosystem function that describe the physical, chemical, and biological condition of the river, including variables such as temperature, DO, conductivity, TSS, TN, TP, and CHL.

LTRM has followed a mixed sampling design since 1993 that includes both fixed monitoring locations and SRS across aquatic area types. Fixed site sampling began in 1989, but recognition of a need for unbiased pool-scale estimates of condition and a better assessment of spatial variation among aquatic areas in the river led to the development of a complementary SRS design. Water-quality fixed sites are monitored every 2 weeks from April through July and monthly from August to March, except in December and February when no sampling occurs. Fixed sites were selected to provide data on specific sites of interest, including tributary mouths and main channel sites. SRS involves seasonal sampling episodes that occur over the course of 2-week periods starting in January, April, July, and October each year (Soballe and Fischer, 2004). Between 120 and 150 sites are sampled in each LTRM study reach during every SRS episode. These sample sizes have remained constant across time with modest differences among study reaches (see table 3 in Soballe and Fischer, 2004) The sites are randomly selected within sampling strata. Strata have not changed over time, are based on the aquatic area types originally defined by Wilcox (1993), and are similar to the aquatic areas defined in De Jager and others (2018; figs. A2–A7). The water quality random

Table E2. Water quality criteria used for the assessment of the status of Long Term Resource Monitoring element reaches and tributaries.

[U.S. Environmental Protection Agency (EPA) nutrient criteria for rivers and streams (EPA, 2000a; EPA 2000b; EPA 2000c) (https://www.epa.gov/nutrient-policy-data/ecoregional-nutrient-criteria-rivers-and-streams) were used in all cases except for total suspended solids criteria. The total suspended solids criterion developed by the Upper Mississippi River Conservation Committee was used as a metric of light conditions for submersed aquatic vegetation growth (Upper Mississippi River Conservation Committee, 2003; Giblin and others, 2010). TN, total nitrogen; mg/L, milligram per liter; TP, total phosphorus; CHL, chlorophyll a; mg/L, microgram per liter]

Reach (fig. A1)	Ecoregion	Sub-region	TN (mg/L)	TP (mg/L)	CHL (µg/L)	Total suspended solids (mg/L)
Pool 4	7	52	1.51	0.070	2.32	30
Pool 8	7	52	1.51	0.070	2.32	30
Pool 13	6	47	3.26	0.118	7.85	30
Pool 26	9	72	1.669	0.083	5.74	30
Open River	9	72	1.669	0.083	5.74	30
La Grange	6	54	2.95	0.073	7.01	30

sampling stratum include the main channel, side channels, contiguous backwaters, isolated backwaters, impounded areas, and two riverine lakes (Lake Pepin in Pool 4 and Swan Lake in Pool 26). In this report, however, we only report data from the main channel and contiguous backwater strata. Additional details on the mechanics of site allocation among strata and sampling considerations are included in Soballe and Fischer (2004).

Indicator: Total Suspended Solids

TSS are a combination of inorganic and organic substances suspended in the water column, such as sediment and living and dead plankton. TSS is a diverse mixture of materials and sources that can include sediment delivered from upstream river reaches, tributaries, and overland flow; resuspended sediment in areas of high wind fetch or turbulence within the river such as shallow backwaters; or high phytoplankton biomass. TSS have several physical and ecological effects on river ecosystems. For example, TSS concentration affects the light environment for aquatic plants and visual predators, contributes to sediment accumulation in river habitats, and affects nutrient availability (Wetzel, 2001).

Methods

I assessed status and trends in TSS between 1994 and 2019 for all LTRM study reaches and 10 tributaries, which include the Cannon and Chippewa Rivers in Pool 4; the Black River in Pool 8; the Maquoketa and Wapsipinicon Rivers in Pool 13 and 14, respectively; the Cuivre and Illinois Rivers (site is located in the Alton Pool, at the mouth of the Illinois River) in Pool 26; the Big Muddy River in the Open River Reach; and the Sangamon and La Moine Rivers, which are tributaries of the Illinois River that drain into the La Grange Pool (fig. A1). Water-quality conditions have a marked contrast upstream and downstream from Lake Pepin, which divides Pool 4. Therefore, I analyzed and reported data from Upper and Lower Pool 4 separately. I focus on TSS concentration only in the main channel in this report because concentrations in the main channel integrate processes occurring upstream, in connected off-channel areas, loads from tributaries, and represent potential loads to downstream pools and off channel areas of the Upper Mississippi River System.

I evaluated three indicators of long-term change in TSS. First, in all LTRM study reaches, I used annual main channel mean SRS concentration to assess the long-term trends in observed concentration of TSS ("annual mean observed concentration;" table E1). This indicator incorporates spatial variation within the main channel through random sampling and describes the actual conditions experienced by organisms in the river over time. I used linear regression to evaluate long-term trends in this indicator, with $p<0.05$ indicating significant trends.

Second, in LTRM study reaches and tributaries, I evaluated trends in two additional indicators that account for the effects of discharge on concentration and assess potential shifts in watershed delivery to the river: flow-normalized concentration and flow-normalized flux. To do so, I analyzed fixed site data from the main channel and tributaries using the "Weighted Regressions on Time, Discharge, and Season" (WRTDS) model (Hirsch and others, 2010, 2015). I used concentration data from 7 main channel and 10 tributary fixed sites and daily discharge data from the U.S. Geological Survey streamgage (U.S. Geological Survey, 2020) nearest each site (table E3) in this analysis.

The WRTDS model uses daily discharge paired with periodically measured solute concentrations to estimate daily concentration and flux over the period of record by applying a relation of concentration with time, season, and discharge. WRTDS generates two types of outputs: estimated daily concentration and flux and flow-normalized concentration and

Table E3. Long Term Resource Monitoring element fixed sites used in Weighted Regression on Time, Discharge, and Season Models to assess total nitrogen, total phosphorus, and total suspended solids trends. Tributary fixed sites are located near the confluence of each tributary in the Upper Mississippi or Illinois Rivers.

[Streamgages not shown. USGS, U.S. Geological Survey; km^2, square kilometer; Wis., Wisconsin; Minn., Minnesota; Mo., Missouri; Ill., Illinois; LTRM, Long Term Resource Monitoring element].

Fixed site name	River reach	USGS streamgage location	State	USGS streamgage number	Drainage area (km^2)
			Tributaries		
CH00.1M	Pool 4	Chippewa	Wis.	05369500	23,336
CN00.1M	Pool 4	Cannon	Minn.	05355200	3,471
BK14.2M	Pool 8	Black	Wis.	05382000	5,385
MQ02.1M	Pool 13	Maquoketa	Iowa	05418500	4,021
WP02.6M	Pool 14	Wapsipinicon	Iowa	05422000	6,048
CU11.6M	Pool 26	Cuivre	Mo.	05514500	2,338
I007.0W	Pool 26	Illinois	Ill.	05586100	69,264
BM00.7S	Open River	Big Muddy	Ill.	05599490	5592
LM00.5M	La Grange	La Moines	Ill.	05585000	3,347
SG16.2C	La Grange	Sangamon	Ill.	05583000	13,185
			LTRM study reaches		
M786.2C	Pool 4	Prescott	Minn.	05344500	148,831
M764.3A	Pool 4	Prescott	Minn.	05344500	148,831
M701.1B	Pool 8	Winona	Wis.	05378500	165,322
M556.4A	Pool 13	Clinton	Iowa	05420500	221,444
M241.4K	Pool 26	St. Louis	Mo.	07010000	444,183
M078.0B	Open River	Thebes	Mo.	07022000	1,844,071
I080.2M	La Grange	Kingston Mines	Ill.	05568500	40,968

flux. Using this model has several advantages. First, WRTDS has been shown to be more robust for trend analysis than other load models in that it allows the relation between concentration and discharge to vary among seasons and over time (Hirsch, 2014). Second, WRTDS estimates daily values, and therefore captures periods of high concentration or export that often occur outside of standard sampling windows. For example, the most frequent sampling interval for fixed sites is twice a month and SRS only occurs seasonally, thus potentially missing periods or events that can transport a large portion of annual solute or sediment loads. Third, WRTDS provides a statistical means of evaluating the effect of flow on trends in concentration by generating flow-normalized concentration and flow-normalized flux. This is important because concentration is fundamentally affected by changes in discharge, and as a result, variation of discharge through time can obscure actual reductions in solute concentration or loads. For example, in cases where concentration increases with discharge, a high-flow period of a year or two near the end of the period of record can indicate deteriorating water quality (Hirsch and others, 2010). WRTDS applies the flow-normalization step by estimating a concentration for each day of the time series by using the mean long-term discharge on that day over the period of record. This process then generates estimates of how concentration and flux at a given discharge have changed through time and isolates the effects of changes in watershed solute sources from variation in discharge. Thus, by removing the variation associated with discharge, flow-normalized concentrations and fluxes allow for an assessment of how the river is responding to changes on the landscape (removal of point source inputs, land use change, or implementation of best-management practices) and should provide a clearer indication of progress (or lack thereof) toward water-quality goals (Hirsch and others, 2010).

I performed WRTDS analyses in R version 4.1.1 (R Core Team, 2021) using the generalized flow-normalization method (Choquette and others, 2019) in the “Exploration and Graphics for RivEr Trends” (EGRET) package in R (Hirsch and De Cicco, 2015) and evaluated the significance of trends in flow-normalized concentration and flow-normalized flux from 1994 to 2019 using the EGRETci package in R (Hirsch and others, 2015). I assessed significance of trends using the likelihood approach recommended by Hirsch and others (2015), reporting trends as significant if they were “Highly Likely”

Summary of Total Suspended Solids

Indicator intent: To characterize patterns and long-term trends in main channel total suspended solids (TSS) and, therefore, water clarity and depth of light penetration across 6 Long Term Resource Monitoring element (LTRM) study reaches and 10 Upper Mississippi River System tributaries.

Measurement: Status of TSS concentration was evaluated by comparison to criteria listed in table E2. Status of TSS in LTRM study reaches was assessed using annual mean observed concentrations. Status of tributaries was assessed using annual mean estimated concentrations (as described in table E1). In LTRM study reaches, long-term trends were evaluated for annual mean observed concentration, flow-normalized concentration, and flow-normalized flux. In tributaries, long-term trends were evaluated for flow-normalized concentration and flow-normalized flux (see details in table E1; see table E3 for list of sites). Period of analysis was 26 years (1994–2019). Only fixed sites that had at least 20 years of concentration data and nearby measurements of continuous discharge data were included in the modeling analysis because of model requirements. All analyses were done by calendar year.

Special considerations: Water-quality conditions have a marked contrast upstream and downstream from Lake Pepin, which divides Pool 4. Therefore, I report data from Upper and Lower Pool 4 separately.

Assessment:

Status: TSS concentration increased substantially downsteam, with a 10-fold increase from Pool 4 to Open River. Pools 4 and 8 rarely exceeded the TSS concentration established as adequate for sustaining submersed vegetation. Pool 13 exceeded this concentration in 54-percent of the years, and Pool 26, Open River, and La Grange had concentrations greater than this TSS criterion in nearly all years. Concentrations had a nearly 25-fold range among tributaries, and all except the Black and Chippewa Rivers exceeded the criterion.

Trends: Annual mean observed concentrations decreased in Pools 4 and 8 between 1994 and 2019. Flow-normalized concentration decreased in all LTRM study reaches except Pool 13 and decreased significantly in 7 of 10 tributaries. Flow-normalized flux trends were more variable among LTRM study reaches and tributaries, but there were declining trends in four LTRM study reaches and six tributaries.

(95–100-percent likely), “Very Likely” (90–95-percent likely), or “Likely” (70–90 percent likely).

To assess the status of TSS in LTRM study reaches, I used annual mean observed concentrations generated from SRS data. To assess TSS status for tributaries, I used annual mean estimated concentrations from the WRTDS model generated from fixed site data (see table E1).

I compared annual mean concentration to the criterion developed as adequate for sustaining submersed vegetation based on data from Pools 4, 8, and 13 (30 milligrams per liter [mg/L]; Upper Mississippi River Conservation Committee 2003; Giblin and others, 2010; table E3). Comparing TSS to this criterion provides an indication of whether the light environment is suitable for submersed vegetation growth. This criterion was not intended for the more naturally turbid reaches of the Upper Mississippi River System downstream from Pool 13. No analogous criteria exist for those river reaches; therefore, I use a single criterion across all study reaches for consistency and to provide a comparable index of water column light conditions across reaches. In addition, Giblin and others (2010) found that the total suspended solids value of 30 mg/L corresponds to a similar light extinction coefficient as the 15 nephelometric turbidity units U.S. Environmental Protection Agency (EPA) turbidity criterion for ecoregions in which the lower reaches are located.

Assessment

Status

Annual mean main channel TSS concentrations ranged from 7.43 to 317 mg/L across all LTRM study reaches, and long-term mean values increased downstream (table E4). Annual TSS concentrations in Pools 4 and 8 never exceeded the TSS criterion (30 mg/L; table E3); Pool 13 did 54-percent of the time; and Pool 26, Open River Reach, and La Grange Pool did in all years. Annual mean TSS concentrations in tributaries ranged from 5.41 to 322 mg/L, with a minimum long-term mean concentration of 13.3 mg/L in the Black River and a maximum of 144.7 mg/L in the Maquoketa River (table E5). The Black and Chippewa Rivers were the only tributaries to remain less than 30 mg/L consistently, but TSS in the Cannon River has decreased over time. The TSS concentration in the Cannon River was always greater than 30 mg/L before 2005 but has been less than that criterion in 9 of the 14 years since then. All other tributaries exceeded the TSS criterion in all years.

Trends

Among LTRM study reaches, significant decreases in observed TSS concentration were observed only in Upper Pool 4, Lower Pool 4, and Pool 8 (table E4; fig. E1). However, decreases in flow-normalized concentration were widespread, decreasing in all LTRM study reaches except Pool 13 and in 7 of 10 tributaries (fig. E2). The Maquoketa River was the only

Table E4. Long-term mean concentrations (grand mean of annual means) and trends in observed total suspended solids, total nitrogen, and total phosphorus.

[All concentrations are shown with standard error of the mean in parentheses. Estimated slope and coefficient of determination (R^2) values are from linear regressions used to evaluate long-term trends in annual mean concentrations from 1994 to 2019. Bold indicates a significant trend (linear regression, p-value less than 0.05). Analyses of suspended solids and total phosphorus were done in log space (log transformed) to meet assumptions of normality for a linear model. TSS, total suspended solids; mg/L, milligram per liter; TN, total nitrogen; TP, total phosphorus; ns, nonsignificant trend]

Reach	Total suspended solids (mg/L)	Slope	R^2	Total nitrogen (mg/L)	Slope	R^2	Total phosphorus (mg/L)	Slope	R^2
Upper Pool 4	**27.4 (3.0)**	**−0.018**	**0.24**	2.98 (0.24)	ns	ns	**0.13 (0.01)**	**−0.02**	**0.48**
Lower Pool 4	**7.5 (0.6)**	**−0.019**	**0.44**	2.47 (0.17)	ns	ns	**0.11 (0.01)**	**−0.01**	**0.34**
Pool 8	**14.8 (1.3)**	**−0.024**	**0.30**	2.53 (0.13)	ns	ns	0.12 (0.01)	ns	ns
Pool 13	33.1 (2.5)	ns	ns	2.46 (0.11)	ns	ns	0.14 (0.01)	ns	ns
Pool 26	73.2 (7.5)	ns	ns	3.62 (0.20)	ns	ns	0.21 (0.01)	ns	ns
Open River	147 (6.4)	ns	ns	3.01 (0.11)	ns	ns	0.26 (0.01)	ns	ns
La Grange	64.1 (4.2)	ns	ns	**4.45 (0.21)**	**−0.03**	**0.23**	0.38 (0.02)	ns	ns

tributary to have an increasing trend, although the magnitude was highly uncertain. Flow-normalized TSS flux decreases were slightly more limited, but decreases were significant in Lower Pool 4, Pool 8, the Open River Reach, and the La Grange Pool. In tributaries, flow-normalized flux declined in 6 of 10 rivers (Chippewa, Black, Cuivre, Illinois, Big Muddy, and Sangamon Rivers), remained stable in the Cannon and Wapsipinicon Rivers, and increased in the Maquoketa and La Moine Rivers (fig. E3).

Indicators: Total Nitrogen and Total Phosphorus

TN and TP concentrations, which include dissolved and particulate forms of nutrients, fuel primary production (algae and vegetation) in the river; however, excessive TN and TP can contribute to local and downstream eutrophication. In addition, in high enough concentrations, nitrate, a component of the TN pool, can harm human health (EPA, 2017b). In addition, nitrate export from the Upper Mississippi River System is an important contributor to Gulf of Mexico hypoxia (Turner and Rabalais, 2003; Crawford and others, 2019). TN and TP are delivered to the Upper Mississippi River System mainly from nonpoint sources such as runoff, tributary loads, and groundwater rather than point sources like wastewater treatment effluent as a result of large efforts to reduce phosphorus and nitrogen in wastewater (Robertson and Saad, 2013 and 2021). Some transformation and uptake of these nutrients can occur in the river itself, especially in off-channel areas, which reduces the effect of excess nutrients to organisms locally and downstream. For example, microbial denitrification in backwater environments can transform inorganic nitrogen to nitrogen gas, thus permanently removing it from the system (Richardson and others, 2004). Also, phosphorus can sorb to sediments and soils, and thus can be stored in sediments under certain conditions.

Methods

Data and methods for assessment of status and trends of TN and TP were the same as described above for TSS, except in details of comparison to water-quality standards. Uniform nutrient criteria are not established for the Upper Mississippi River System, and the only States that have applicable criteria only include TP (Minnesota: https://www.epa.gov/sites/production/files/2014-12/documents/mnwqs-chapter-7050.pdf#page=115 and Wisconsin: https://www.epa.gov/sites/production/files/2014-12/documents/wiwqs-nr102.pdf#page=4). Therefore, I used the applicable EPA regional nutrient criteria for each study reach of the river (EPA, 2000a, 2000b; table E2). I chose to use EPA standards for TN and TP because they were generated for specific regions based on geology and soil information and, therefore, more closely reflect the unique geology contributing to each area of the river rather than a state-wide value that may not accurately reflect watershed characteristics near the river.

Assessment

Status

Annual mean TN concentration in the main channel ranged from 1.59 to 5.34 mg/L in LTRM study reaches and generally increased downstream (fig. E1). Long-term mean TN was lowest in the upper three study reaches (2.46–2.98 mg/L) and highest in the La Grange Pool (4.45 mg/L; table E4). For all study reaches, annual mean TN values were greater than TN criteria (fig. E1; table E2), except in Pool 13, which exceeded its TN criteria in only 2 years. All reaches were also much greater than the TN concentration indicative of eutrophic conditions (0.714 mg/L; Dodds, 2006) most of the years. Annual mean TN concentration in tributaries ranged from 1.40 to 7.96 mg/L (table E5). The Chippewa River had the

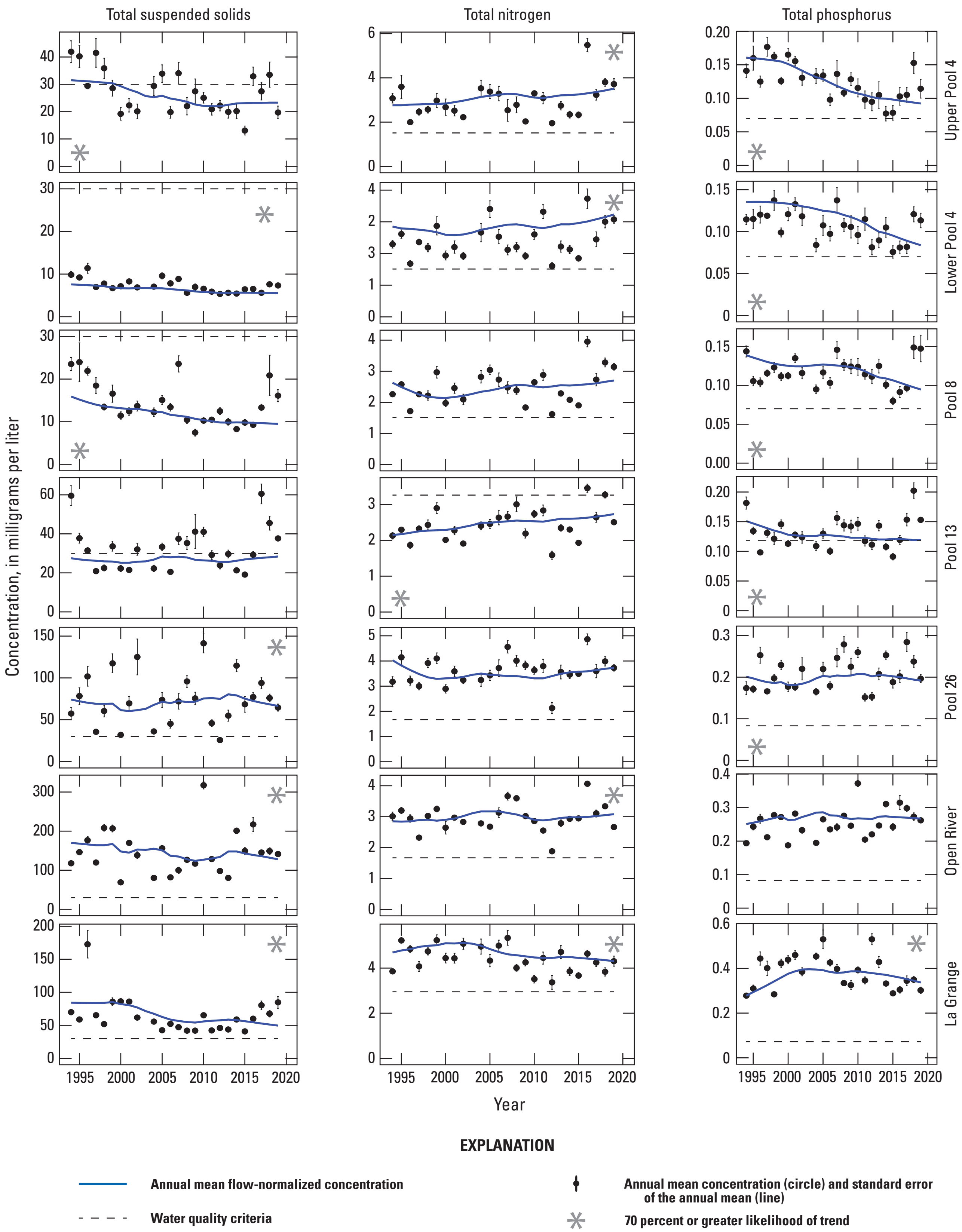


Figure E1. Annual mean total suspended solids, total nitrogen, and total phosphorus concentrations based on stratified random sampling (black points) and Weighted Regressions on Time, Discharge, and Season flow-normalized concentrations (solid blue line) based on data from main channel fixed sites in each Long Term Resource Monitoring element study reach.

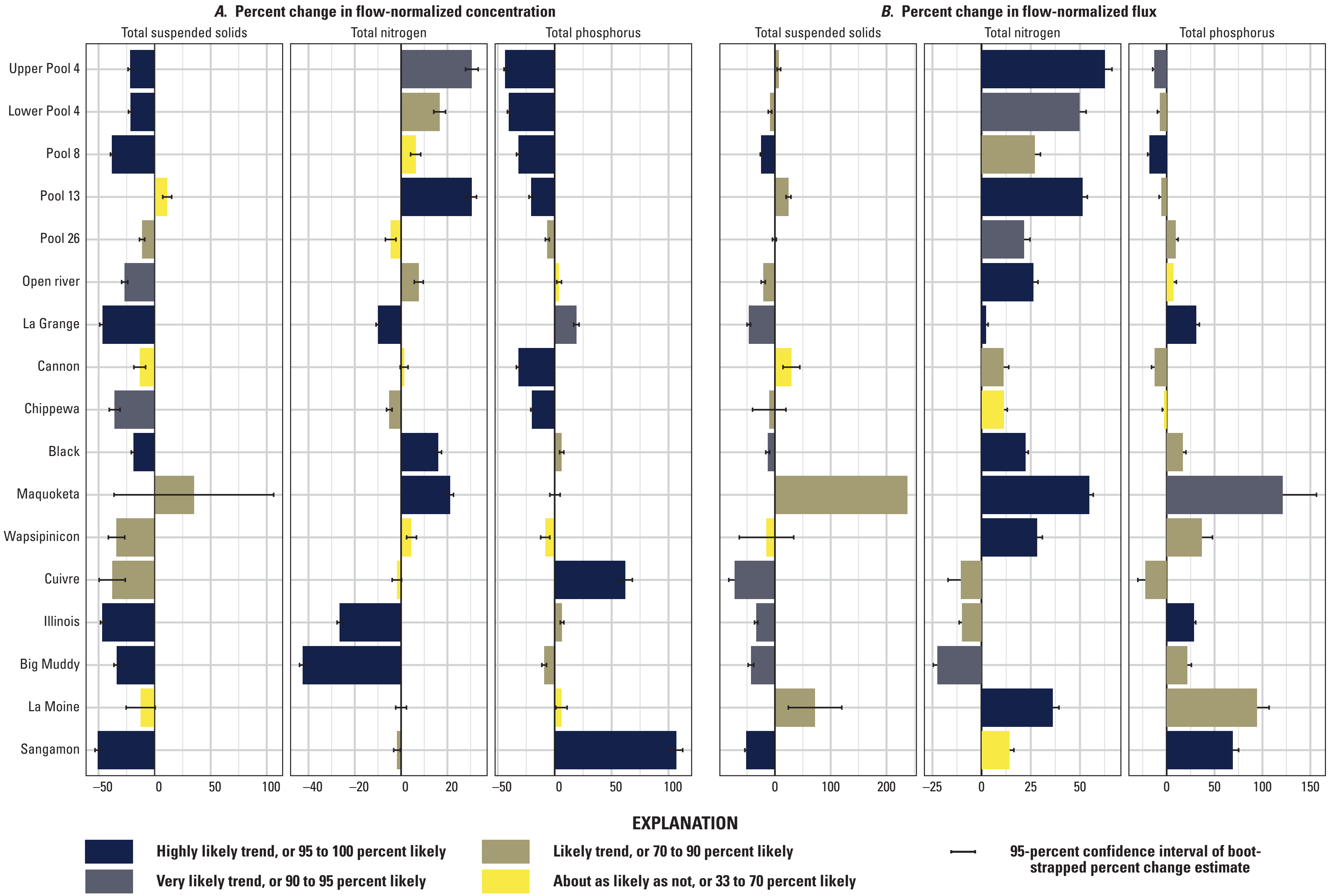


Figure E2. Percentage change in flow-normalized concentration and flow-normalized flux from Long Term Resource Monitoring element study reaches and tributaries between 1994 and 2019.

Table E5. Mean and standard deviation of estimated annual mean concentrations of total suspended solids, total nitrogen, and total phosphorus in Upper Mississippi River System tributaries.

[Estimates were generated from the Weighted Regressions on Time, Discharge, and Season model and used for the assessment of water quality status of tributaries. LTRM, Long Term Resource Monitoring element; mg/L, milligram per liter]

Tributary name	LTRM study reach	Total suspended solids (mg/L)	Total nitrogen (mg/L)	Total phosphorus (mg/L)
Cannon	Pool 4	32.9 (7.5)	4.80 (0.24)	0.16 (0.03)
Chippewa	Pool 4	10.7 (2.3)	1.53 (0.10)	0.09 (0.01)
Black	Pool 8	13.3 (1.7)	1.53 (0.08)	0.14 (0.01)
Maquoketa	Pool 13	144.7 (61.1)	7.05 (0.63)	0.29 (0.07)
Wapsipinicon	Pool 14	78.0 (11.4)	5.63 (0.67)	0.21 (0.02)
Cuivre	Pool 26	60.7 (20.4)	2.00 (0.13)	0.19 (0.03)
Illinois	Pool 26	64.6 (14.4)	4.47 (0.46)	0.33 (0.04)
Big Muddy	Open River	51.4 (12.2)	1.49 (0.42)	0.19 (0.02)
La Moine	La Grange	114.7 (63.8)	3.43 (0.64)	0.22 (0.06)
Sangamon	La Grange	91.0 (38.9)	4.72 (0.68)	0.53 (0.14)

Summary of Total Nitrogen

Indicator intent: To characterize patterns and long-term trends in main channel total nitrogen (TN) across 6 Long Term Resource Monitoring (LTRM) element study reaches and 10 tributaries of the Upper Mississippi River System.

Measurement: Overall status of TN was evaluated by comparison to criteria listed in table E2, which vary among reaches (1.51–3.26 milligrams per liter [mg/L]). Status of TN in LTRM study reaches was assessed using annual mean observed concentrations. Status of TN in tributaries was assessed using annual mean estimated concentrations (as described in table E1). In LTRM study reaches, long-term trends in TN were evaluated for annual mean observed concentration, flow-normalized concentration, and flow-normalized flux. In tributaries, long-term trends in TN were evaluated for flow-normalized concentration and flow-normalized flux (see details in table E1; see table E3 for list of sites). Period of analysis was 26 years (1994–2019). Only fixed sites that had at least 20 years of TN concentration and continuous discharge data were included in the modeling analysis because of model requirements. All analyses were done by calendar year.

Special considerations: Water quality conditions have a marked contrast upstream and downstream from Lake Pepin, which divides Pool 4. Therefore, I report data from Upper and Lower Pool 4 separately.

Assessment

Status: Annual mean observed concentration of TN in LTRM study reaches ranged from 2.46 mg/L in Pool 13 to a maximum of 4.45 mg/L in La Grange. All LTRM study reaches exceeded TN criteria in all years, except Pool 13, which only exceeded its criterion in 2 years. This lack of exceedance reflected a higher TN criterion for that region than others, not a lower concentration (table E2). Annual mean estimated TN concentration in tributaries ranged from 1.53 mg/L in the Black and Chippewa Rivers to a high of 7.05 mg/L in the Maquoketa River. TN concentration in tributaries exceeded TN criteria most years, but exceedances occurred least frequently in the Chippewa and Black Rivers (15 and 18 years, respectively).

Trends: Five of six LTRM study reaches did not exhibit significant long-term trends in annual mean observed TN concentration, except the La Grange Pool where TN decreased. Trends in flow-normalized TN concentration were more common but direction and magnitude varied among LTRM study reaches. Concentrations increased in Upper and Lower Pool 4, Pool 13, and the Open River Reach; remained stable in Pools 8 and 26; but decreased in the La Grange Pool. For tributaries, flow-normalized concentration increased in two, remained stable in three, and decreased in five rivers. Flow-normalized flux increased in all LTRM study reaches and in 5 of 10 tributaries, but decreased in the Cuivre, Illinois, and Big Muddy Rivers.

Summary of Total Phosphorus

Indicator intent: To characterize patterns and long-term trends in main channel total phosphorus (TP) across 6 Long Term Resource Monitoring (LTRM) element study reaches and 10 tributaries of the Upper Mississippi River System.

Measurement: Status of overall TP was evaluated by comparison to the criteria listed in table E2, which vary among reaches (0.070–0.118 mg/L). Status of TP in LTRM study reaches was assessed using annual mean observed concentration. Status of TP in tributaries was assessed using annual mean estimated concentration (as described in table E1). In LTRM study reaches, long-term trends in TP were evaluated for annual mean observed concentration, flow-normalized concentration, and flow-normalized flux. In tributaries, long-term trends in TP were evaluated for flow-normalized concentration and flow-normalized flux (see details in table E1; see table E3 for list of sites). Period of analysis was 26 years (1994–2019). Only fixed sites that had at least 20 years of concentration and continuous discharge data were included in the modeling analysis because of model requirements. All analyses were done by calendar year.

Special considerations: Water-quality conditions have a marked difference upstream and downstream from Lake Pepin, which divides Pool 4. Therefore, I report data from Upper and Lower Pool 4 separately.

Assessment

Status: TP concentration varied threefold among LTRM study reaches. TP increased downstream from a long-term mean of 0.11 mg/L in Lower Pool 4 to 0.38 mg/L in the La Grange Pool. All LTRM study reaches exceeded TP criteria in all 26 years, except Pool 13, which exceeded criteria in only 18 years, reflecting its higher criterion not a lower concentration. Long-term mean TP in tributaries ranged from a minimum of 0.09 mg/L in the Chippewa River to a maximum of 0.53 mg/L in the Sangamon River. All tributaries exceeded TP criteria in nearly all years.

Trends: Annual mean observed TP concentration decreased only in Upper and Lower Pool 4. Flow-normalized TP concentration decreased in all LTRM study reaches except the La Grange Pool and Open River Reach, where concentrations increased and remained stable, respectively. Tributary flow-normalized concentrations decreased in three rivers, remained stable in three, but increased in four rivers. Flow-normalized flux decreased in Pools 4, 8, and 13; remained stable in Open River Reach, and increased in Pool 26 and the La Grange Pool. Flow-normalized flux increased in 7 of 10 tributaries.

lowest long-term mean TN concentration (1.53 mg/L), and the Maquoketa River had the highest (7.05 mg/L; table E5). Only the Black and Chippewa Rivers had some years with concentrations less than TN criteria (8 and 11 years, respectively; data not shown).

Annual mean TP concentrations ranged from 0.08–0.53 mg/L in LTRM study reaches. Long-term mean TP increased downstream, and similarly to TN, was lowest in the Upper Impounded Reach (0.12–0.14 mg/L) and highest in the La Grange Pool (0.38 mg/L; table E4). All LTRM study reaches were greater than TP criteria in all years, except Pool 13, which exceeded its higher criterion in 18 of the 26 years (fig. E1; table E2). However, this reflected a higher criterion, not a lower concentration than other reaches. Annual mean concentration in tributaries ranged from 0.08–0.86 mg/L and was greater than TP criteria in nearly all years. Long-term mean TP concentration in tributaries ranged from a minimum of 0.09 mg/L in the Chippewa River to a maximum of 0.53mg/L in the Sangamon River (table E5). Annual mean TP concentration in LTRM study reaches and tributaries also consistently indicated eutrophic conditions (Dodds, 2006; greater than 0.071 mg/L).

Trends

Annual mean observed concentration of TN in LTRM study reaches remained mostly stable over time, except in the La Grange Pool where they decreased (table E4). Flow-normalized TN concentration showed more significant change, however. Three of 6 LTRM study reaches had increasing flow-normalized concentrations (fig. E1 and E3), two remained stable, and only the La Grange Pool decreased. Downward flow-normalized concentration trends were common among tributaries (5 of 10 tributaries), but these were mainly focused in the Illinois River Basin (Illinois, La Moine, Sangamon Rivers) or the state of Illinois (Big Muddy River). In contrast, increases in flow-normalized concentration were observed in the Black and Maquoketa Rivers, both in the northern portion of the basin. Thus, there was a contrasting pattern in flow-normalized concentration trends between the northern and southern portions of the basin, with northern reaches and tributaries more likely to have upward trends in TN and southern reaches and tributaries more likely to have downward trends in TN. Despite these somewhat variable flow-normalized concentration trends, flow-normalized TN flux increased in all LTRM study reaches. Trends in tributary flow-normalized flux were variable among rivers, but similarly to flow-normalized concentration, there were more upward flow-normalized TN flux trends in the northern basin (Cannon, Black, Maquoketa, and Wapsipinicon Rivers) than the southern basin (Cuivre, Illinois, and Big Muddy Rivers; fig. E2).

Annual mean observed concentration of TP decreased significantly only in Upper and Lower Pool 4 (table E4; fig. E1). Decreases in flow-normalized concentration were much more widespread, showing decreasing concentrations

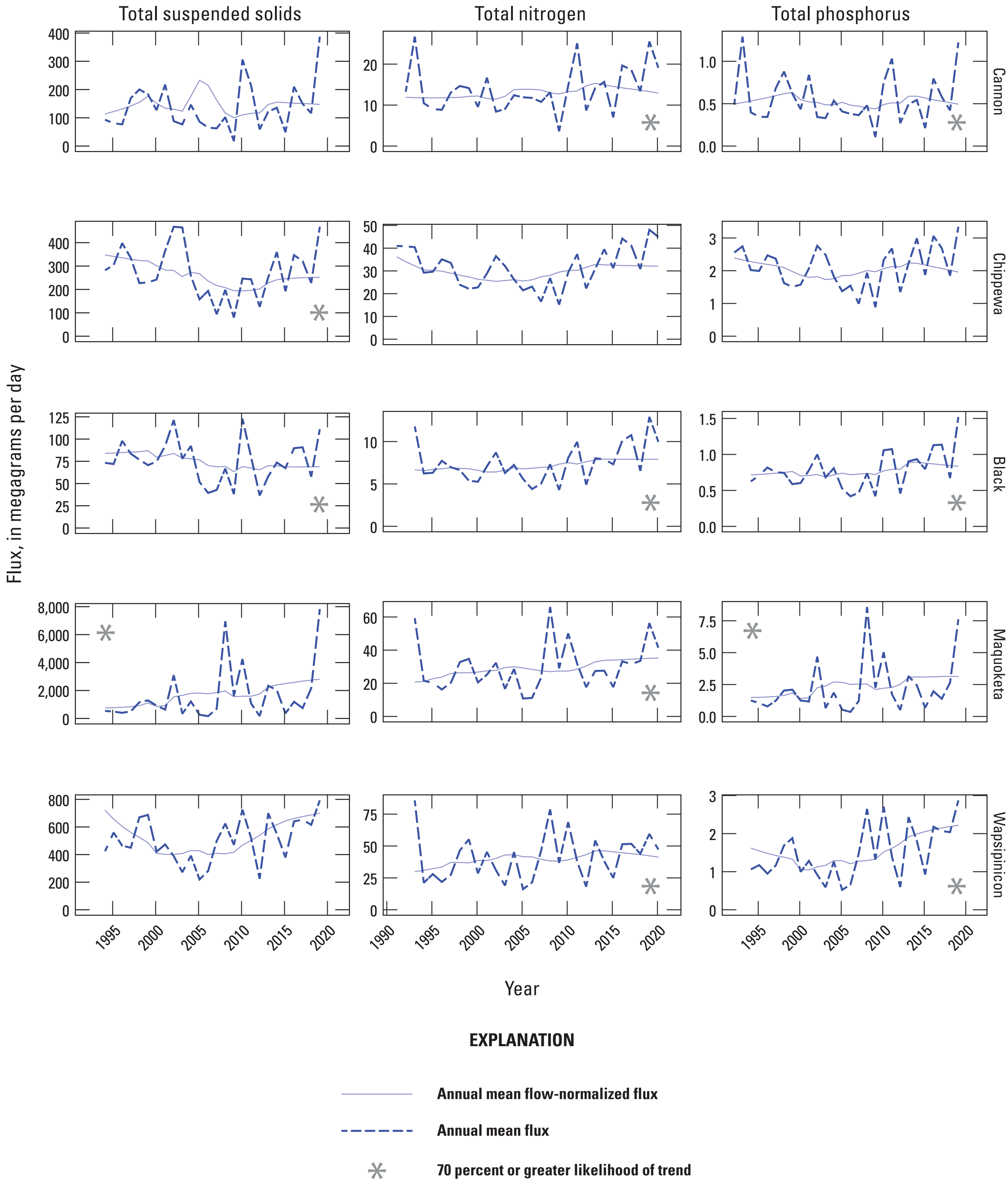


Figure E3. Trends in annual mean flux and annual mean flow-normalized flux of total suspended solids, total nitrogen, and total phosphorus from Upper Mississippi River tributaries based on a fixed sampling site near the confluence of the tributary and the Upper Mississippi River or Illinois River. Tributaries plotted from north to south from the Cannon through the Big Muddy Rivers. The La Moine and Sangamon Rivers are on the Illinois River.

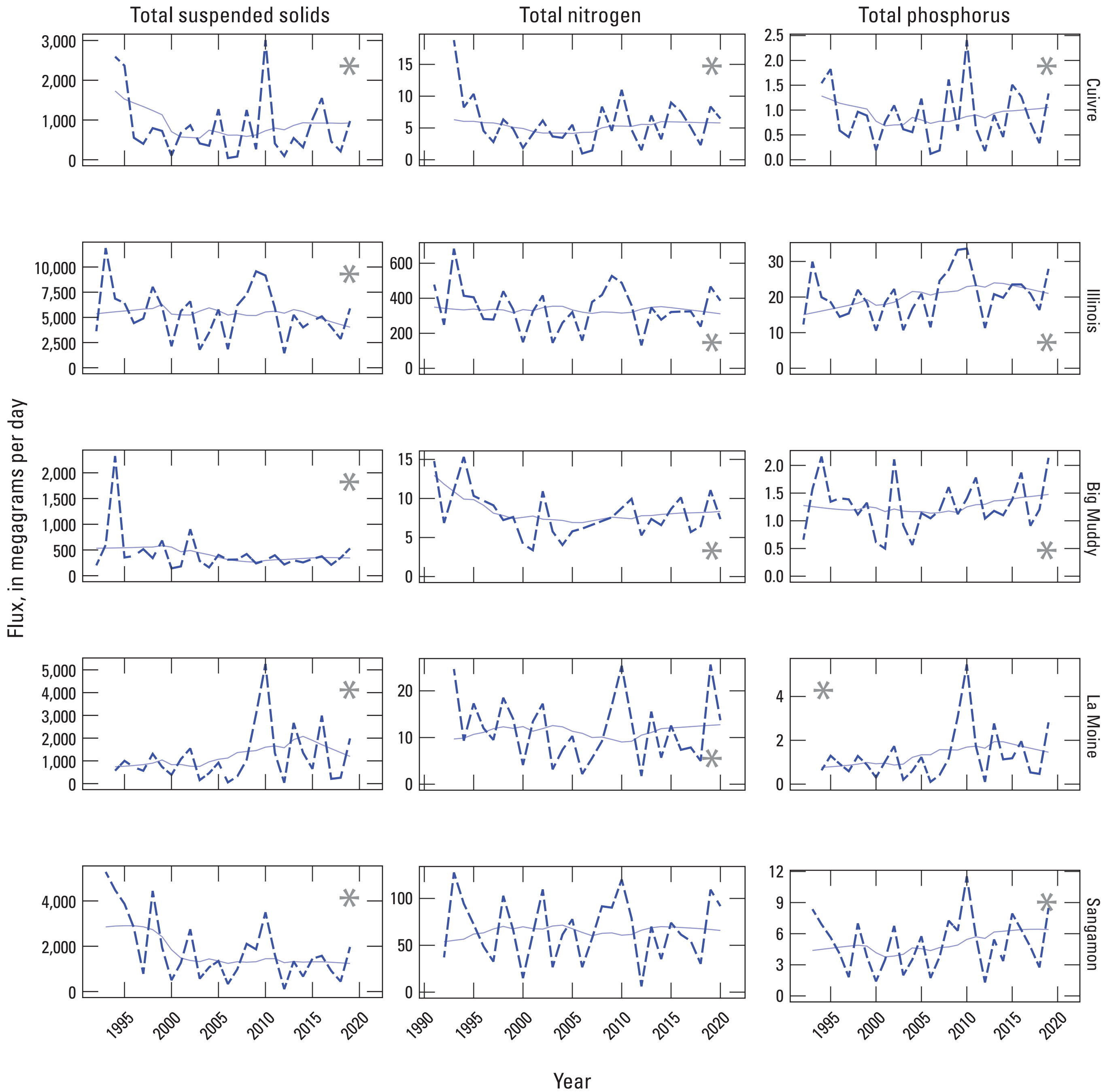


Figure E3. Continued

in most LTRM study reaches, except the Open River Reach where TP remained stable and in the La Grange Pool where TP increased (figs. E1 and E2). Tributaries had more variable patterns, with flow-normalized concentration decreasing in the Cannon, Chippewa, and Big Muddy Rivers but increasing in the Black, Cuivre, Illinois, and Sangamon Rivers. Patterns in flow-normalized flux among LTRM study reaches were mostly similar to flow-normalized concentration, but decreases were limited to Pools 4, 8, and 13. In addition, although Pool 26 had a downward flow-normalized concentration trend, its flow-normalized flux increased. Increases in flow-normalized TP flux were more common in tributaries than in LTRM study reaches, with increases in 7 of 10 tributaries (figs. E2 and E3). There were small decreases in flow-normalized flux in the Cannon and Cuivre Rivers and no change in the Chippewa River.

Indicator: Chlorophyll *a*

CHL is a measure of water column algal biomass, which is predominantly phytoplankton. Algae are a food source for upper trophic level organisms like zooplankton and fish, but in excess can negatively affect habitat and aquatic organisms. For example, algal blooms reduce light penetration into the water column, which affects growth of submersed aquatic vegetation (SAV). In addition, some phytoplankton species produce toxins, which can harm aquatic organisms and human health. Finally, the death and decay of algal blooms can lead to low DO concentration, which reduces habitat suitability for many species and alters rates of nutrient cycling and uptake.

Methods

I assessed status and trends using LTRM summer SRS CHL data from the main channel and contiguous backwaters. I evaluated differences in CHL among LTRM study reaches using analysis of variance. I compared CHL concentrations to the EPA Ecoregional Nutrient Criteria for Rivers and Streams (https://www.epa.gov/system/files/documents/2021-07/ecoregion-table-rivers-streams.pdf) and used criteria specific to the subregions surrounding LTRM study reaches (table E3). I estimated trends in CHL concentration based on annual summer averages. I assessed linear trends using generalized linear models and nonlinear trends using generalized additive models; Zuur and others, 2009; table E6). I considered linear and nonlinear trends because they provide complementary information. Linear trends identify long-term, directional changes. Nonlinear trends provide information as to whether there were significant fluctuations over time that may or may not result in a straightforward increase or decrease over the period of record. Generalized linear models were run using the stats package and generalized additive models were run using the mgcv package (Wood 2017) in R version 4.1.1 (R Core Team, 2021).

Assessment

Status

Mean summer main channel CHL concentration was similar among study reaches and ranged from 6.9 to 118.5 micrograms per liter (µg/L) (fig. E4). Pool 8 and the La Grange Pool had the highest long-term mean summer CHL concentrations (33.8 and 33.9 µg/L, respectively; table E6). CHL was similar in Pool 4, 13, and the Open River (table E6).

Summary of Chlorophyll *a*

Indicator intent: To characterize patterns and trends in summer main channel and contiguous backwater chlorophyll *a* (CHL) concentration: a metric of algal biomass and an indicator of eutrophication.

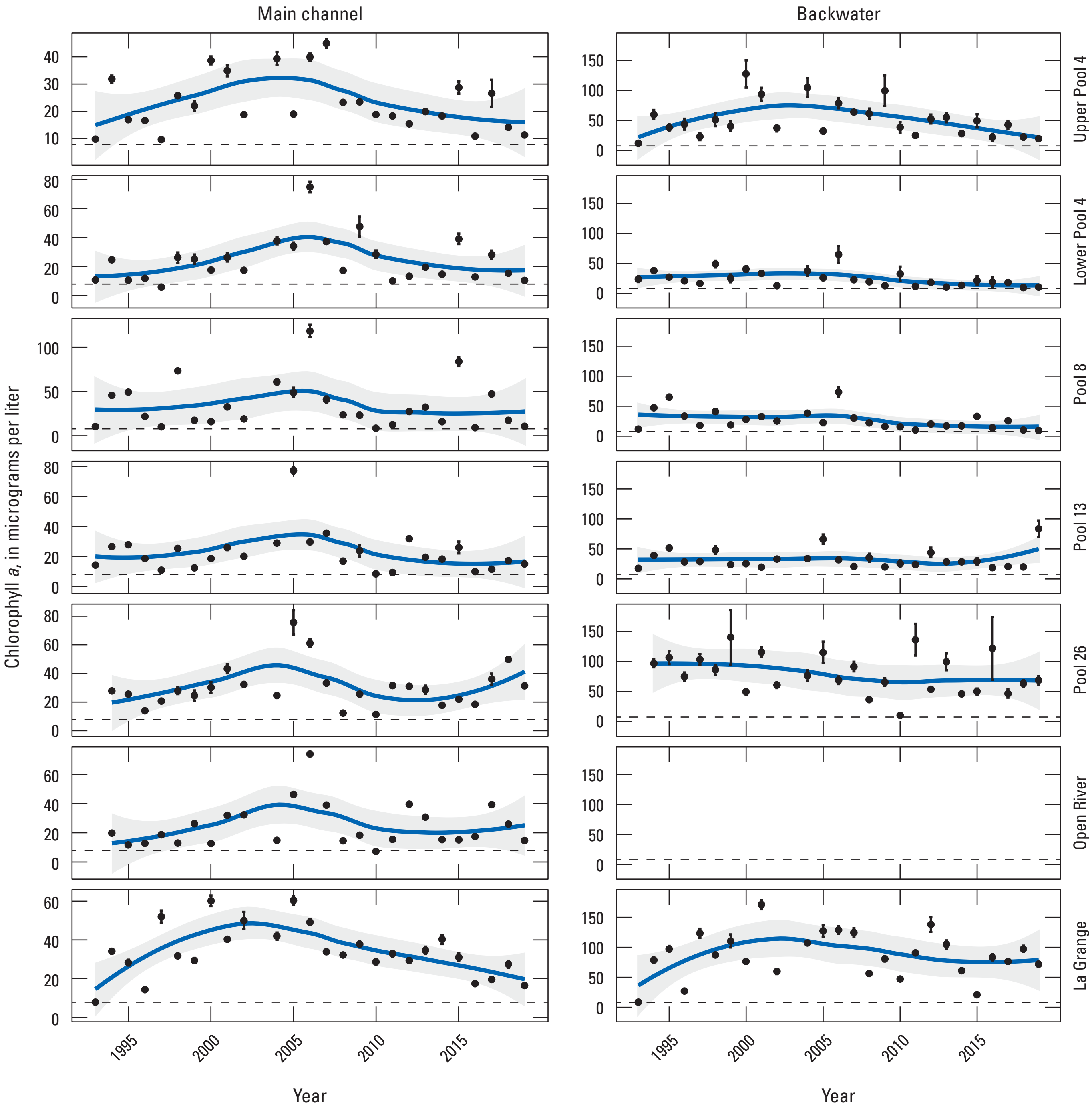
Measurement: Annual mean summer main channel and backwater CHL concentration for Long Term Resource Monitoring (LTRM) element study reaches were calculated from LTRM stratified random sampling data from 1993 to 2019.

Special considerations: Only summer concentration was included because summer is generally the season with the greatest CHL concentration and, therefore, the period during which algal biomass is most likely to affect habitat conditions for aquatic organisms. No uniform CHL criteria exist for the Upper Mississippi River System; therefore, I used U.S. Environmental Protection Agency (EPA) river and stream CHL criteria developed for subregions specific to areas surrounding LTRM study reaches (EPA, 2000a, 2000b; table E2). EPA criteria range from 2.32 to 7.85 micrograms per liter among reaches.

Assessment

Status: Backwater CHL was generally higher than main channel CHL, except in Pool 8. Main channel and backwater CHL were greater than CHL criteria in nearly all years in all LTRM study reaches (at least 25 of 26 years in all reaches).

Trends: No significant linear trends were observed for main channel CHL concentration in any reach, but a downward trend was observed for the backwaters of Pool 8. Pools 4 and the La Grange Pool also showed decreases in CHL since 2006 and 2002, respectively.



EXPLANATION

[LOESS, locally weighted regression]

LOESS model confidence intervals

LOESS trend line

Water quality criteria for chlorophyll *a*

Annual mean concentrations (circle) and standard error of the annual mean (line)

Figure E4. Annual mean summer main channel and backwater chlorophyll *a* concentration in Long Term Resource Monitoring element study reaches. Note: Axis scales differ between main channel and backwater panels.

Table E6. Mean summer main channel and backwater chlorophyll a concentrations over the Long Term Resource Monitoring element period of record (1993–2019).

[Standard deviations shown in parentheses. Values with different superscripted letters (a, b, or c) indicate significant differences as assessed using an analysis of variance; for example, a indicates that the value is significantly different than values with b or c. "Best model" refers to model with lowest (Akaike Information Criterion value (Burnham and Anderson, 2002). Bold text indicates significant model. Statistical significance assessed at a p value less than 0.05. SD; standard deviation; NA, not applicable; ±, plus or minus]

Pool	Main channel mean (SD)	Best model	Backwater mean (SD)	Best model
Upper Pool 4	23.0 (10.0)[a]	Nonlinear	51.1 (29.2)[a]	**Nonlinear**
Lower Pool 4	23.7 (15.1)[a]	Nonlinear	24.4 (13.2)[b]	**Nonlinear**
Pool 8	33.8 (26.8)[a]	No change	26.7 (16.1)[b]	**Linear**
Pool 13	22.2 (13.5)[a]	Nonlinear	32.5 (15.6)[ab]	No change
Pool 26	30.3 (14.7)[a]	Nonlinear	79.7 (32.8)[c]	Linear
Open River	24.3 (14.9)[a]	No change	NA	NA
La Grange	33.9 (13.2)[a]	**Nonlinear**	86.7 (37.9)[c]	Nonlinear
Overall	**27.9 (±17.1)**	NA	**51.2 (±36.2)**	NA

Mean annual CHL concentration exceeded criteria in nearly all years (fig. E4; table E2).

Summer backwater CHL concentration ranged from 8.7 to 171 µg/L (fig. E4) and was higher on average than main channel concentration in all pools except Pool 8 (table E6). Backwaters in Pool 26 and the La Grange Pool had a substantially greater CHL concentration than the main channel. The long-term mean backwater CHL concentration was greatest in the La Grange Pool (86.7 µg/L) and lowest in Pool 8 (26.7 µg/L; table E6). CHL concentration in backwaters exceeded criteria in nearly all years in all LTRM study reaches.

Trends

No long-term linear trends in main channel CHL concentration were observed in any of the LTRM study reaches (table E6; fig. E4). In Upper and Lower Pools 4, 13, 26, and the La Grange Pool, a nonlinear model that exhibited a peak in the middle of the time series better fit the data than a linear model but was only significant in the La Grange Pool. A CHL peak occurred between 2005 and 2006 for all Upper Mississippi River reaches, coincident with a period of low discharge in the Upper Mississippi River (ch. B), but CHL in the La Grange Pool on the Illinois River peaked earlier (~2002). In the backwaters, a downward linear trend was significant in Pool 8, and nonlinear models with a peak in 2006 and subsequent decrease were significant in Upper and Lower Pool 4.

Indicator: Dissolved Oxygen

DO is critical for the survival of many aquatic organisms and can affect nutrient cycling. Backwater environments are particularly susceptible to low DO conditions because they are shallow, have longer residence times than channel habitats, contain highly organic sediments, have low water volume to sediment ratios, and form thicker ice and snow cover in the winter which impedes light penetration for photosynthesis (Jankowski and others, 2021). Backwaters also often have a high biomass of algae, vegetation, or free-floating plants that consume DO through their own growth and respiration and when they die and decompose. Low DO can reduce the availability of suitable habitat in winter and summer but is especially problematic during winter when other factors such as temperature and velocity reduce habitat availability. Low availability of DO in the water column can also affect nutrient cycling in positive and negative ways. For example, reduced oxygen diffusion into sediments promotes denitrification, an efficient N removal mechanism from the river (Richardson and others, 2004) but can also increase sediment P release, which fosters local eutrophication through algal or floating plant growth (Houser and others, 2013).

Methods

The frequency of low DO was calculated as the proportion of backwater sites with DO less than 5 mg/L. This was calculated for summer and winter each year using backwater SRS data. I used surface measurements of DO (0.2 meter depth), except where bottom readings were measured at sites likely to exhibit vertical stratification (defined as velocity less than 0.1 meter per second and depth greater than or equal to 0.60 meter in Soballe and Fischer, 2004). Where surface and bottom DO readings were available, I used their mean to better represent conditions in the entire water column. Where bottom samples were not available, I assumed the surface sample was representative of the full water column. LTRM sampling is centered around noon, generally completed between 9:00 a.m. and 3:00 p.m. and, therefore, excludes the time of day where minimum DO concentration generally occurs (around dawn). As a result, our measurements likely underestimated the occurrence of low DO conditions.

Summary of Dissolved Oxygen

Indicator intent: To characterize patterns and long-term trends in low dissolved oxygen (DO) concentration in contiguous backwater lakes during winter and summer.

Measurement: The frequency of low DO in backwaters was assessed as the proportion of all sampled backwater sites where DO concentration was less than 5 milligrams per liter. This was assessed for winter and summer each year using Long Term Resource Monitoring element stratified random sampling data from 1993 to 2019.

Special considerations: Five milligrams per liter was used as the threshold DO concentration because it complies with Clean Water Act criteria for aquatic life protection for all States in the Upper Mississippi River System region (U.S. Environmental Protection Agency: https://www.epa.gov/wqs-tech/state-specific-water-quality-standards-effective-under-clean-water-act-cwa#tb2). Contiguous backwaters are uncommon in the Open River Reach and the Long Term Resource Monitoring element does not sample a backwater stratum there; therefore, no data are reported for the Open River Reach in this section.

Assessment

Status: Low DO in backwaters occurred more frequently during summer (mean proportion of sampled sites=0.20) than winter (mean proportion of sampled sites=0.08). Among all Long Term Resource Monitoring element study reaches, low backwater DO was most common in Pool 8 in both summer and winter.

Trends: Upper Pool 4 had an increase in low backwater DO conditions during the summer and Lower Pool 4 had an increase during the winter.

To evaluate trends, I compared linear and nonlinear models (generalized linear model and generalized additive model, respectively). I considered linear and nonlinear trends because they provide complementary information. Linear trends identify long-term, directional changes. Nonlinear trends provide information as to whether significant fluctuations occurred over time that may or may not have resulted in a straightforward increase or decrease over the period of record. Generalized linear models were run using the stats package and generalized additive models were run using the mgcv package (Wood, 2017) in R.version 4.1.1 (R Core Team, 2021). Data are plotted and shown with a local polynomial regression smoother to show nonlinear patterns in the data (fig. E5).

Assessment

Status

The proportion of backwater sites with low DO was greater in summer than winter. Low DO conditions occurred at 0–80-percent of sampled sites in summer and 0–38-percent of sampled sites in winter. Across the period of record, the overall mean proportion of sampled sites with low DO during summer ranged from a minimum of 0.08 in Upper Pool 4 to a high of 0.28 in Pool 13, but proportions were similarly high in Pool 8 and Lower Pool 4 (table E7). During the winter, the mean proportion of sites with low DO was smaller than summer, ranging from 0.003 in La Grange to 0.17 in Pool 8.

Trends

In most cases, the pattern in low DO was either nonlinear or there was no change through time (fig. E5, table E7). The only exception was Lower Pool 4, where the proportion of sites with low DO increased linearly in the winter (table E7). There have been notable decreases in low DO in the summer in recent years in Lower Pool 4 and Pool 8 after a midperiod peak, but they remained higher as of 2019 compared to 1993 (fig. E5). The year 2010 had a high frequency of low DO across all reaches during the summer. The timing of the periods with higher frequency of low DO was more variable among reaches in winter than it was during the summer. For example, low winter DO was relatively common throughout the period of record for Pool 8, but more frequent in the past 10 years than the first 10 years for Lower Pool 4.

State of the Ecosystem

The Upper Mississippi River System remains eutrophic in many reaches, but there is evidence of increased water clarity (decreased TSS) and decreased TP and CHL concentrations in some study reaches. Trends varied within the river basin, however, with some indicators having consistent trends across the system (TSS), some varying from north to south (TN), and others varying across the lotic-lentic gradient of the river (CHL). These patterns are described in more detail in the following sections.

Total Suspended Solids

Observed TSS concentration decreased significantly only in Pool 4 and 8, whereas flow-normalized TSS concentration decreased nearly everywhere (5 of 6 LTRM study reaches and 7 of 10 tributaries). This indicates that TSS concentration has decreased at the mean long-term discharge in many reaches and tributaries of the Upper Mississippi River System.

In Pools 4 and 8, several factors may contribute to the significant decrease in observed and flow-normalized concentrations of main channel TSS. First, the coincident decrease in flow-normalized concentration and flux of TSS

Table E7 Mean proportion of low dissolved oxygen sites during summer and winter from 1993 to 2019.

[Standard deviations shown in parentheses. "Best model" refers to model with lowest Akaike Information Criterion value (Burnham and Anderson, 2002). Bold indicates significance of model (*p*-value less than 0.05); ±, plus or minus]

Reach	Summer	Best model	Winter	Best model
Upper Pool 4	0.08 (±0.12)	Nonlinear	0.12 (±0.16)	No change
Lower Pool 4	0.26 (±0.22)	**Nonlinear**	0.09 (±0.07)	**Linear**
Pool 8	0.27 (±0.20)	**Nonlinear**	0.17 (±0.12)	**Nonlinear**
Pool 13	0.28 (±0.18)	**Nonlinear**	0.10 (±0.11)	**Nonlinear**
Pool 26	0.12 (±0.12)	**Nonlinear**	0.02 (±0.06)	No change
La Grange	0.11 (±0.10)	**Nonlinear**	0.003 (±0.01)	No change

from major tributaries to Pools 4 and 8 can explain some of the decrease (Chippewa and Black Rivers; figs. E2 and E3). Second, the strong correspondence between observed and flow-normalized concentrations of TSS in Pools 4 and 8 indicates that the decrease is not a function of dilution by higher flows later in the period (ch. B), but rather a true reduction in watershed inputs. If dilution were occurring, flow-normalized concentration would remain greater than observed concentrations (Hirsch and others, 2010). Finally, decreased TSS (and associated increased water clarity), especially in off-channel areas, has been linked to the increased prevalence of aquatic vegetation that has decreased sediment resuspension in backwaters and impounded areas (Giblin, 2017; Bouska and others, 2020; Burdis and others, 2020; ch. H).

Pool 26 and the Open River Reach and the La Grange Pool showed significant decreases in flow-normalized but not observed TSS concentration. In Pool 26 and Open River, trends in observed concentration were not significant and remained high during the last 10 years, indicating that high and variable discharge is likely sustaining high TSS concentrations in those reaches. In other words, although the concentration of TSS at a given discharge has decreased over time (as indicated by the flow-normalized concentration), river discharge has been high in recent years and this has resulted in a lack of trend in observed TSS concentration. A decrease in watershed input to these reaches is also indicated by decreased flow-normalized concentration and flux from tributaries that feed into Pool 26 (the Cuivre and Illinois Rivers) and and the Open River Reach (Big Muddy River; figs. E2 and E3). In the La Grange Pool, the decrease in TSS likely reflects two processes. First, tributaries to the La Grange Pool—the Sangamon River—had a downward trend in flow-normalized TSS concentration and flux, indicating lower delivery from portions of the watershed. Second, the downward trend in TSS coincides closely with onset of a decrease in CHL (fig. E4), indicating that some of the reduction in La Grange may reflect a decrease in algal biomass, which is a component of TSS (Houser and others, 2010; fig. E1).

TSS in Pool 13 was higher than Pools 4 and 8 and did not show the same significant downward trends in observed or flow-normalized concentration as other LTRM study reaches (figs. E1 and E2). Observed concentration was particularly high at the beginning and end of the period of record, whereas flow-normalized concentration and flux remained stable. Although Pool 13 also experienced a modest increase in SAV from 1998–2010 (ch. F and ch. H), SAV never reached the same frequency as it did in Pools 4 and 8, decreased from its peak in 2010, and has not been associated with the pool-wide decreases in turbidity seen in Pools 4 and 8 (ch. H). In addition, the Maquoketa River, which flows into Pool 13, had the highest TSS concentration of all tributaries analyzed here (table E5). In fact, TSS concentration in areas of Pool 13 downstream from the mouth of the Maquoketa River was slightly higher on average than the TSS concentration upstream from the Maquoketa River, indicating the Maquoketa River has a measurable effect on water quality in a large portion of the reach (data not shown). Thus, the Maquoketa River TSS inputs combined with TSS loads from upstream reaches and tributaries, result in higher TSS concentrations and light conditions that are not as suitable for SAV in Pool 13 as in Pools 4 and 8 (Carhart and others, 2021).

Nutrients

The Upper Mississippi River System remains replete with nutrients, and most reaches and tributaries have concentrations much greater than water quality criteria for both TN and TP (tables E2, E4–E5). Observed TN concentrations in Pool 26, the Open River Reach, and the La Grange Pool were consistently higher than in the upper three study reaches, but TN concentrations have remained high throughout the system. Flow-normalized TN concentrations were less variable than measured concentrations from year to year (fig. E1, table E4), indicating that high and variable discharge contributes to high and variable observed TN concentration in many reaches of the Upper Mississippi River System. In addition, there was only evidence for a downward trend in observed and flow-normalized concentration in one LTRM study reach—the La Grange Pool. This decrease matched downward TN trends at the mouth of the Illinois River (figs. E2 and E3) as well as broader decreasing patterns in nitrate observed throughout the Illinois River (Crawford and others, 2019). The decreases in flow-normalized concentration indicate that TN has decreased independently from changes in discharge, indicating that

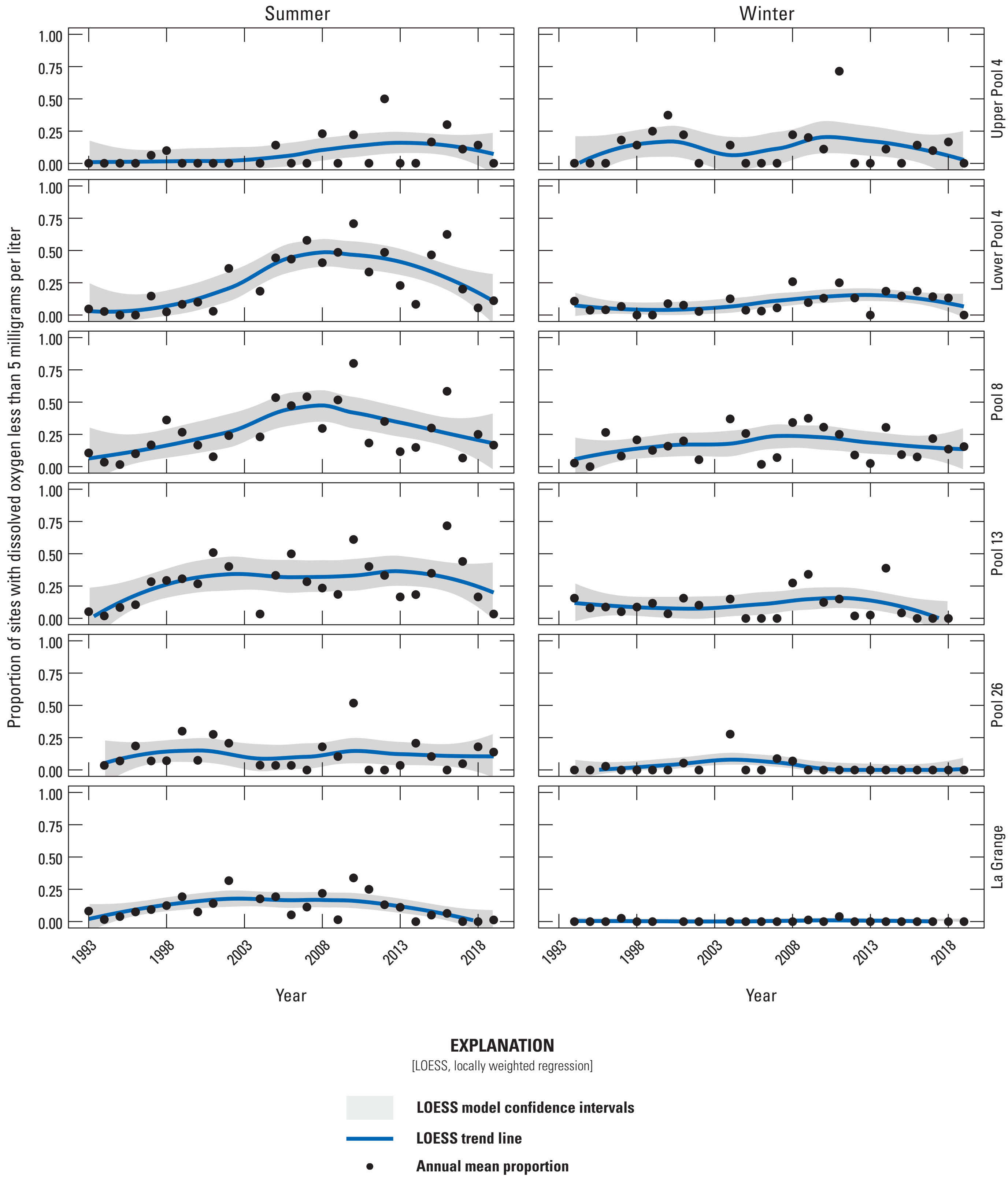


Figure E5. Proportion of backwater sites in summer and winter each year that had a dissolved oxygen concentration less than 5 milligrams per liter. Only Long Term Resource Monitoring element reaches that have a backwater strata are shown.

reductions in watershed contributions have affected TN concentrations in the Illinois river (Murphy and Sprague, 2019). In contrast, all other LTRM study reaches displayed upward or no trends in flow-normalized concentration and all LTRM study reaches had increasing flow-normalized TN flux, indicating increasing export of TN from all portions of the Upper Mississippi River System. This indicates that despite some improvements in concentration, export of TN from the basin remains high, making reductions in concentration particularly important for reducing TN export if discharge continues to increase (ch. B). (Crawford and others, 2019).

TN concentrations in the Mississippi Basin will likely remain high without large-scale change. The Mississippi Basin has a large quantity of "legacy" N stored in groundwater and soils that would take decades to drain even if N applications on the landscape were to stop altogether (Van Meter and others, 2018). In addition, high discharge is linked to greater loads of N from major tributaries of the Upper Mississippi River (Crawford and others, 2019; Smits and others, 2019). Thus, predicted increases in high-discharge events (Naz and others, 2016; Swanston and others, 2018) are likely to increase loads of N to the Upper Mississippi River System unless substantial efforts are dedicated to restoration projects that increase N uptake in the watershed (Schilling, Jacobson, and Wolter, 2018), such as wetland conservation or restoration (Cheng and others, 2020) or in enhancing the connectivity of floodplain habitats that can remove N through denitrification (Richardson and others, 2004; Hansen and others, 2018). Nationwide (Stets and others, 2020) and in some areas of the Mississippi River Basin (Crawford and others, 2019), N transport in rivers has decreased recently, indicating that nutrient management efforts can improve nutrient flux (Stets and others, 2020; Robertson and Saad, 2021).

In contrast to TN, observed TP significantly decreased in Upper and Lower Pool 4, and flow-normalized TP concentration and flux decreased in many LTRM study reaches and tributaries. At least three possible reasons can explain these significant decreases. First, between 2000 and 2005, the Metropolitan Wastewater Treatment Plant in St. Paul, Minnesota, upstream from LTRM study reaches, implemented improved water treatment capacity that removes substantially more phosphorus (Metropolitan Council Environmental Services, 2018). Decreases in TP are more apparent during winter than summer (LTRM database; https://umesc.usgs.gov/data_library/water_quality/graphical/wq_browser.html), indicating that lower TP is likely related to changes in point source inputs to the river from sources such as the Metropolitan Wastewater Treatment Plant and observable only when the watershed is frozen and nonpoint sources are reduced. In addition, progressively smaller decreases in TP were seen downstream. Given that Pool 4 is closer to Minneapolis than other reaches, it follows that the magnitude of the decrease would be most substantial there and decrease moving southward (fig. E2). Similar efforts to reduce TP in wastewater have occurred throughout the Mississippi River watershed, and thus could also be contributing to lower TP loads in other reaches (Robertson and Saad, 2021). Second, flow-normalized TP concentrations decreased in the majority of tributaries we evaluated here and flow-normalized fluxes decreased in the Cannon and Cuivre Rivers indicating some potential for reduced watershed inputs to the Upper Mississippi River System. Nationwide, decreases in phosphorus balances from watersheds have been observed, indicating that this result may also mirror broader scale trends in reduced phosphorus loss from watersheds (Stackpoole and others, 2019). Finally, the reductions in TSS likely contribute to the reduction in TP, because phosphorus easily attaches to soil particles that get transported to the river during runoff events (Wood and others, 1984; Houser and others, 2010). Despite these widespread decreases in flow-normalized TP concentration and flux, the Sangamon River, which drains into the La Grange Pool of the Illinois River, increased substantially in flow-normalized TP concentration and flux, likely contributing to the observed increase in flow-normalized TP concentration in the La Grange Pool (fig. E2). These differential north-south patterns in TN and TP across the basin are important to consider as they could affect nutrient ratios, which has implications for ecological communities in the Upper Mississippi River System and the Gulf of Mexico (Dodds and Smith, 2016; Royer, 2020).

Chlorophyll *a*

LTRM study reaches had high summer CHL concentrations, indicating widespread eutrophic conditions. Some longitudinal and lateral differences were observed in CHL concentrations, however. First, although CHL did not have the strong increasing north-south pattern apparent for TSS and nutrient concentrations, backwater CHL concentrations were consistently higher in lower reaches than upper reaches. In addition, in all reaches except Pool 8, CHL concentrations in the backwaters exceeded the main channel concentrations. The lack of longitudinal pattern in the main channel is similar to work by Houser and others (2010), who found only a weak longitudinal decrease in CHL concentrations among Upper Mississippi River reaches. They attributed the similar concentrations among reaches to several factors, including high interannual variability and light limitation in the main channel of all LTRM study reaches. For example, although Pool 26 and the La Grange Pool have high nutrient concentrations that feed phytoplankton growth, they also have high suspended sediment concentrations, which leads to light limitation in many areas of the Mississippi River main stem (Ochs and others, 2013).

Backwater CHL concentrations were highest in Pool 26 and La Grange Pool but were followed closely by CHL concentrations in Upper Pool 4 backwaters. This could be for several reasons. First, backwaters in the lower reaches of the Upper Mississippi River System are generally less hydrologically connected to the main channel (De Jager and others, 2018), which increases water residence time and favors algal growth (Soballe and Kimmel, 1987). Decreased connectivity between the main channel and backwaters, especially during low discharge years, has been shown to result in a greater CHL concentration in backwaters relative to the main channel (Houser, 2016; Jankowski and others, 2021). Second, the backwaters of Pools 4, 8, and 13 have much higher abundances of submersed aquatic vegetation than lower reaches (ch. F), which can outcompete phytoplankton for light and other resources (Scheffer and others, 2003; see further discussion below). Further, vegetation abundance was lower in the backwaters of Upper Pool 4 than Lower Pool 4, which may explain why backwater CHL concentrations remained higher there (Burdis and others, 2020; ch. F). Thus, the combination of higher connectivity and greater frequencies of submersed aquatic vegetation in Pools 4, 8, and 13 contribute to the greater similarity between main channel and backwater CHL concentrations than was observed in the lower reaches.

Although CHL concentration remained high across all reaches, there have been notable decreases in CHL concentrations for at least some portion of the period of record in main channel and backwater areas (table E6; fig. E4). Decreases in backwater CHL concentrations and turbidity in Pools 4 and 8 have been linked to increased prevalence of SAV (Giblin, 2017; Bouska and others, 2020; Burdis and others, 2020; ch. F). For example, Burdis and others (2020) demonstrated CHL concentrations decreased in backwaters after an increase of SAV in Lower Pool 4, similar to what I report here (fig. E4). Shallow lakes can alternate between a clear, vegetated state and a turbid, phytoplankton-dominated state (Scheffer and others, 2003). Vegetation competes with phytoplankton for light and nutrients, thus the increased prevalence of vegetation in the Upper Impounded Reach has likely reduced nutrients and light available to phytoplankton for growth in backwaters, thereby reducing CHL concentrations.

Main channel decreases in CHL concentrations have also coincided with decreasing TP concentrations (fig. E1) and higher summer discharge (Manier and others, 2021; ch. B). Observed TP concentration remained high enough in all LTRM study reaches that it is unlikely to have limited main channel phytoplankton; however, discharge had a significant and negative association with CHL in main channel and backwaters for all reaches over time.The period of high CHL across LTRM study reaches corresponded to a period of low discharge in the Upper Mississippi River (ch. B), indicating that the observed patterns in CHL partially reflected interannual patterns in discharge. This also aligns with previous work showing that CHL is negatively related to discharge in channel environments because of associated increases in current velocity and turbidity (Ochs and others, 2013; Houser, 2016).

In the La Grange Pool, previous work has linked decreases in CHL concentration to an increase in the biomass of *Hypophthalmichthys molitrix* (silver carp) (De Boer and others, 2018), which consume high quantities of plankton. De Boer and others (2018) also observed that this downward trend in CHL concentrations was significant in the main channel but not in the backwaters and attributed that difference to silver carp use of channel borders and side channels more often than connected backwaters. Silver carp can also reduce zooplankton biomass, releasing phytoplankton from predation pressure, which leads to increases in CHL concentrations (Shen and others, 2021).

Dissolved Oxygen

Sufficient DO is critical for the survival of a diverse aquatic community, and improving winter DO conditions is often a goal of habitat rehabilitation projects on the Upper Mississippi River System (Johnson and others, 1998). Analyses showed that low DO was more common in backwaters during the summer than during the winter and was more widespread in the upper three pools than in Pools 26 and the La Grange Pool during summer and winter.

During the summer months, sediment oxygen demand can be high in connected backwaters and, therefore, drive down DO concentration under low velocity, warm conditions. In addition, typical low summer discharge diminishes hydrologic connectivity with backwaters, thereby reducing backwater replenishment with higher DO water from the main channel and altering backwater conditions (Houser and others, 2013). Periods with a high prevalence of low summer DO in Lower Pool 4 and Pool 8 occurred primarily between 2005 and 2010, which was a period of low discharge (ch. B). This period also coincided with the greatest rates of SAV expansion in Pools 4 and 8 (ch. F). High densities of SAV can sometimes result in high oxygen demand through senescence of dead plant material, the reduction of water circulation, or by facilitating of the colonization of free-floating plants and filamentous algae, which block light from penetrating below the surface and reduce oxygen production via photosynthesis (Giblin and others, 2014; Houser and others, 2013). The proportion of sites with low DO has decreased from this midperiod peak, most notably in Lower Pool 4 and Pool 8, and is likely related to a series of particularly high-discharge years (figs. B1–B5). However, the relationship between the proportion of low DO sites and discharge was variable among years and not statistically significant in any reach, indicating that multiple factors, not just discharge, affect the occurrence of low summer DO in backwaters (Houser and others, 2013). A pronounced midperiod peak in the prevalence of low summer DO did not occur in Pool 13, Pool 26, or in the La Grange Pool as it did in Pools 4 and 8. In Pool 13, the frequency of low DO increased from 1993 to 2002 but has varied around a mean of 25-percent of sampled sites since 2002, whereas the occurrence of low DO in Pool 26 and the La Grange Pool has remained relatively stable over time.

Low winter DO was less frequent than low summer DO, but likely has greater effects on habitat availability. This is especially true in the Upper Impounded Reach where other factors like low temperature or excess current velocity reduce winter habitat suitability in addition to DO (Knights and others, 1995). Low DO in the winter typically occurs in areas of low connectivity to the main channel, areas of high sediment oxygen demand, areas of shallow depths, and after periods of ice and snow cover that reduce light for oxygen production via photosynthesis (Knights, Johnson, and Sandheinrich, 1995; Sullivan, 2004, Jankowski and others, 2021). The upper study reaches have more frequent, longer lasting, and thicker ice cover (including frozen to the bottom conditions), which often result in more frequent occurrences of low DO (Sullivan, 2004). Accordingly, low winter DO conditions were most prevalent in the upper reaches, specifically in Pool 8. Lower Pool 4 was the only reach to potentially have increased prevalence of low DO conditions in the winter. Pool 13 had reduced frequency of low winter DO in the last ~10 years.

Discharge during winter has increased in several of the study reaches (ch. B), which likely contributed to higher winter DO in connected backwaters (Johnson and others, 1998). Although our results do not show that low winter DO was a major issue in backwaters of lower reaches, other factors like high water velocity or cold temperatures could affect winter habitat availability in warmer, often ice-free southern reaches.

Future Pressures

Although there have been notable improvements, several challenges remain to improving the water quality of the Upper Mississippi River System over the long-term. Reductions in agricultural fertilizer use in the basin have not been substantial, and large amounts of legacy nutrients remain on the landscape (Van Meter, Van Cappellen, and Basu, 2018). Groundwater in the region is rich in nitrate as a result of years of fertilized agriculture (Green and others, 2014), and analyses indicate it will take decades before nitrate is sufficiently reduced even if fertilization stopped now (Van Meter and others, 2018). Phosphorus is also stored in the watershed in soils, in sediments on agricultural fields, and in many lakes and rivers in the region and can be mobilized downstream during high-flow events (Hassan and others, 2017; Stackpoole, Stets, and Sprague, 2019). Anthropogenic and climatic factors may exacerbate these effects. For example, increases in the use of tile drainage have reduced the uptake capacity of soil and plants by routing water directly into streams from farm fields (Robertson and Saad, 2021). Further, hydrologic projections for our region indicate that more frequent extreme precipitation events are likely (Naz and others, 2016), which can increase nutrient and sediment delivery to the river depending on when and where the precipitation falls (Scavia and others, 2017; Smits and others, 2019). For example, toxic algae blooms in the Ohio River have been linked to the nature of the seasonal hydrograph. High spring-discharge events, which deliver nutrients, followed by a period of low flow have resulted in harmful bloom events in the Ohio River. (Nietch and others, 2022). Finally, it will be important to consider how other emerging contaminates such as road salt (Dugan and others, 2017; Stets and others, 2018), pharmaceuticals (Hughes and others, 2013), per-and polyfluoroalkyl substances (Boone and others, 2019; Dykstra and others, 2021), microplastics (Reid and others, 2019), and a multitude of legacy (Wiener and Sandheinrich 2010; Corsi and others, 2019) contaminants are affecting water-quality and habitat conditions in the Upper Mississippi River System.

Despite these ongoing challenges, this chapter documents several notable improvements in water quality in the form of decreases in TSS and TP in several reaches and tributaries. Although measured concentrations in the river have remained high (fig. E1), our model results indicate that TP and sediment contributions from the watershed are decreasing independent of changes in discharge (figs. E2 and E3). These results agree with analyses of national-scale datasets that showed widespread decreases in TSS and TP from rivers draining urban and mixed land uses (Stets and others, 2020), which has implications for sediment and nutrient transport, water clarity, and river productivity. Reducing nitrogen input to aquatic systems remains a priority goal for many government and nongovernmental agencies throughout the Mississippi River Basin (EPA, 2017a), and the relatively stable and high TN concentrations in the river indicate much more progress is needed. Nationally, however, TN and nitrate from agricultural sites (Stets and others, 2020) and in certain areas within the Upper Mississippi River (Crawford and others, 2019) have decreased, indicating that although the Upper Mississippi River continues to contribute a significant proportion of the nitrogen load to the Gulf of Mexico, watershed interventions are showing some progress (Crawford and others, 2019; Robertson and Saad, 2021). Opportunities to build on these improvements in the Upper Mississippi River System and its basin include restoration efforts that maintain and enhance floodplain connectivity that increase the river's capacity for processing of water, sediments, and nutrients; maintaining gains in SAV that have improved water clarity in many areas of the Upper Mississippi River System (ch. H); and continued efforts toward reducing loads from the watershed (Robertson and Saad, 2021).

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Chapter F: Aquatic Vegetation Indicators

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Chapter F of

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Chapter F: Aquatic Vegetation Indicators

By Danelle M. Larson,[1] Eric M. Lund,[2] Alicia M. Carhart,[3] Deanne C. Drake,[3] Jeffrey N. Houser,[1] Nathan R. De Jager,[1] Kristen L. Bouska,[1] Kyle R. Bales,[4] and Shawn M. Giblin[3]

Importance of Aquatic Vegetation in Riverine Floodplains

Aquatic vegetation (collectively referring to multiple life forms including submersed, emergent, rooted-floating and free-floating plants) is vital to large floodplain rivers, such as the Upper Mississippi River System. Aquatic vegetation provides many ecosystem services, including regulating services like water purification (Kreiling and others, 2011), bank erosion reduction, sediment redeposition, and flood impact reduction (Gurnell, 2014), which can strengthen the functionality and resilience of the system. Submersed aquatic vegetation (SAV) provides shelter and food for the Upper Mississippi River's invertebrates, fish, amphibians, reptiles, aquatic mammals, and water birds (Holland and Huston, 1984; Korschgen and Green, 1988; Flinn and others, 2005; Stafford and others, 2007; DeLain and Popp, 2014; Weeks and others, 2017). Emergent vegetation responds to water-level variations on daily, seasonal, and annual time scales, and emergent presence indicates a river with hydrologic pulses (Haslam, 2006). *Zizania aquatica* L. (annual wildrice), *Sagittaria latifolia* Willd. (broadleaf arrowhead), and *Sagittaria ridgid*a Pursh. (sessilefruit arrowhead) are common emergent species on the Upper Mississippi River System, and they provide many ecosystem services, including food for human consumption and cultural connections to the river. Rooted-floating vegetation such as *Nymphaea odorata* Aiton (white waterlily) and *Nelumbo lutea* Willd. (American lotus) are found in shallow, warm backwaters that are important for some Upper Mississippi River fish nurseries (Shaeffer and Nickum, 1986). Free-floating plants such as duckweeds and filamentous algae are important contributors to ecosystem health and native diversity; however, their excessive growth and dominance can cause many ecological and aesthetic problems and diminish some recreational uses of the river (Scheffer and others, 2003).

Historical Context for Aquatic Vegetation in the Upper Mississippi River System

All life forms of aquatic vegetation were present in the Upper Mississippi River System before lock-and-dam construction in the 1930s (Fremling, 2005; U.S. Fish and Wildlife Service [FWS], 2011a). However, the lock-and-dam navigation systems caused major changes in the distribution of aquatic areas across the floodplain (fig. A5). In the Upper Impounded Reach, the lock-and-dams created large, impounded areas that formed expansive, off-channel aquatic areas and stabilized water levels. Therefore, all aquatic vegetation life forms flourished in these new habitats for several decades after construction (Fremling, 2005; Moore and others, 2010).

Severe decreases in SAV abundance within the Upper Mississippi River System have occurred periodically, with differences among river reaches in timing, apparent causes, and recovery. In the Illinois River, most SAV loss occurred between the 1920s and 1950s and has been attributed to urban waste discharge, increased water-level fluctuations, and high concentrations of suspended sediments (Mills and others, 1966; Bellrose and others 1979; Sparks and others, 1990). In the Upper Impounded and Lower Impounded Reaches of the Upper Mississippi River (see figure A1 for locations of reaches and pools), a major decrease in SAV occurred in the late 1980s (Fischer and Claflin, 1995; FWS, 2011a), but the causes remain poorly understood. Possible causes include a drought in the late 1980s, high nutrient concentrations leading to competition for light with phytoplankton and filamentous algae, erosion of islands, or other water quality issues (Rogers, 1994; Fischer and Claflin, 1995; FWS, 2011a). The SAV abundance remained low for a period of years in the impounded areas, possibly reinforced by the loss of islands and associated increased wind fetch, wave energy, and turbidity.

However, SAV recovered substantially beginning in the mid-2000s in much of the Upper Impounded Reach but not in the other reaches. By 2010, the Upper Impounded Reach had shown substantial recovery of SAV, but the Lower Impounded Reach and Illinois Rivers had not (De Jager and Rohweder,

[1]U.S. Geological Survey.

[2]Minnesota Department of Natural Resources.

[3]Wisconsin Department of Natural Resources.

[4]Iowa Department of Natural Resources.

2017). The aquatic vegetation in the Lower Impounded Reach is limited by lack of shallow areas, poor water clarity, and fluctuating water levels (Carhart and others, 2021). In much of the Illinois River, SAV remains scarce today likely due to severe water-level fluctuations, poor water clarity, and herbivory (Sass and others, 2017; Carhart and others, 2021).

The Upper Mississippi River System's SAV community is dynamic, and temporal fluctuations in the abundance of a few species were well-documented. Since the early 1960s, *Vallisneria americana* Michx. (watercelery) was the most common species in the impounded areas of Pools 4–14 (Rogers, 1994). At Pool 19, watercelery was the dominant species from the 1950s to 1980s (Steffeck and others, 1985; FWS, 2011a) but lost about 50-percent of its abundance in the 1980s coinciding with the increase of American lotus (Tazik and others, 1993). Populations of watercelery decreased in backwaters and impounded areas from navigation Pools 5–19 of the Upper Mississippi River during a severe drought from 1987 to 1989 (Fischer and Claflin, 1995). After this drought, the watercelery and *Ceratophyllum demersum* L. (coontail) seedbanks remained plentiful in Pool 8 (Kimber, van Der Valk, and Korschgen, 1995a). This seedbank and overwintering buds of watercelery allow the plants to be resilient to short-term disturbances like droughts and floods. The flood of 1993 reduced the abundance of most SAV species, including nonnative *Myriophyllum spicatum* L. (Eurasian watermilfoil), but was followed by an increase in watercelery the following year (Spink and Rodgers, 1996; Sparks, 2010). Significant watercelery beds occurred in Pool 8 within 2 years of the drought of 1987–89 and the flood of 1993 (Kimber and others, 1995a).

Historical information on emergent vegetation in the Upper Mississippi River System is incomplete despite its ecological importance (Moore and others, 2010). Before the Long Term Resource Monitoring (LTRM) element sampling of aquatic vegetation, other data sources reported significant fluctuations in the river system's emergent vegetation through time. Since the lock-and-dam installations, Pools 3–13 supported the greatest amount of emergent vegetation due to having open, shallow aquatic areas as suitable habitat (Peck and Smart, 1986). Below Pool 13, more open channels, levees, and agricultural lands that typically did not provide the geomorphic template necessary for emergent species emerged after the lock and dams (De Jager and others, 2018). Emergent vegetation was abundant in many sections of the Illinois River until water diversions and lock-and-dam construction in the early 1900s and has been nearly absent since the 1950s (Richardson, 1921; Mills and others, 1966). Historical records of annual wildrice indicate this emergent plant was common throughout the Upper Mississippi River System, including the Illinois River, but was rarely detected in the Lower Impounded Reach or Illinois River by 2020 (Bellrose and others, 1979; Dalrymple, 2008). In Pool 8, emergent biomass decreased threefold from 1975 to 1991 (Fischer and Claflin, 1995), a decrease that was in part responsible for the implementation of the LTRM Aquatic Vegetation Monitoring Component.

Current Upper Mississippi River System Goals of Restoring Aquatic Vegetation

McCain and others (2018) compiled multiple agency perspectives on aquatic vegetation along each major reach to assess the partnership's values and perceptions of ecological status in comparison to each management agencies' desired conditions. Aquatic vegetation abundance and diversity are considered a medium to high priority by management agencies across the entire river system (Landwehr and others, 2004; U.S. Army Corps of Engineers, 2011; FWS, 2011a; McCain and others, 2018). The FWS's Habitat Management Plan for the Upper Mississippi River Refuge System identified watercelery, *Stuckenia pectinata* (L.) Börner (sago pondweed), and *Sagittaria* spp. (arrowheads) as key habitat features because they disproportionately supported the FWS' priority bird species (FWS, 2019). That Habitat Management Plan quantitatively specified that native emergent species such as arrowheads should be greater than 60-percent of the total vegetation cover in managed areas. Similarly, the Mark Twain National Wildlife Refuge Complex along the Upper Mississippi River from northeast Iowa to Missouri and the Lower Illinois River has a Comprehensive Conservation Plan and associated Habitat Management Plan that specify goals for restoring aquatic vegetation for a variety of wetland-dependent species (FWS, 2011a).

Sampling Design and Methods

The primary objective of the LTRM aquatic vegetation component is to estimate pool- and stratum-scale prevalence and diversity of aquatic vegetation over time (Yin and others, 2000). Primary "indicators" used with the LTRM data and highlighted in this chapter are summarized in table F1. Data are collected on the prevalence (the fraction of occupied sampling units, which is also known in other literature as "percent frequency of occurrence"), relative abundance, and species composition of SAV (about 20 species). Data also are collected for the LTRM on the prevalence, percent cover, and species composition of rooted-floating, free-floating, and emergent species (about 10 species).

Since 1998, the LTRM has led stratified random sampling (SRS) annually within Pools 4, 8, and 13 in the Upper Impounded Reach with 450 or more sites per pool per year. Detailed data collection methods are in Yin and others (2000). From 1998 to 2004, SRS vegetation monitoring was conducted in five of the six LTRM study reaches (all but the Open River Reach; fig. A1); however, routine LTRM SRS of the Lower Impounded and Illinois Reaches ceased after 2004 because aquatic vegetation was rarely detected, and subsequent field studies and remote sensing analyses indicated aquatic vegetation has remained scarce in these reaches (Sass and others, 2017; De Jager and Rohweder, 2017). The Open River Reach has never been routinely sampled for aquatic vegetation because its hydrology and geomorphology are such

Table F1. Names, attributes, and intent of each aquatic vegetation indicator developed to evaluate status and trends in the ecosystem of the Upper Mississippi River System aquatic vegetation community.

[SAV, submersed aquatic vegetation]

Indicator name	Indicator attribute(s)	Indicator intent
SAV prevalence	The prevalence of submersed plants (fraction of sampled sites occupied).	Assess prevalence of total SAV and two common invasive SAV species as biological indicators of ecosystem health.
Aquatic vegetation diversity	Estimates of Shannon's diversity index, species richness, and evenness.	Investigate differences across space and changes over time as representative measures of biodiversity.
Dominance of free-floating plants	Number of sites where free-floating plants are in high abundance and SAV abundance is low or undetected.	Investigate how often free-floating plants are the dominant vegetation type, which is an indicator of river eutrophication and aesthetic impairment.
Emergent vegetation	Proportion of pool area and number of hectares with coverage scores for emergent plants, including *Zizania aquatica* L. (annual wildrice).	Note changes in emergent vegetation as indicators of dynamic hydrology and note increases of annual wildrice, which is an important resource.

that little area is suitable for aquatic vegetation (De Jager and Rohweder, 2017; Carhart and others 2021).

The vegetation sampling is divided proportionally according to strata area in each pool to produce unbiased pool-scale estimates of prevalence. The primary sampling strata for aquatic vegetation include main channel borders, side channels, impounded areas, contiguous backwaters, and isolated backwaters. The LTRM also collects land cover data across the entire Upper Mississippi River System floodplain every 10 years from which aquatic vegetation cover is extracted. Select results from analysis of the land cover data are reported in the "Emergent Vegetation" section of this chapter.

Indicator: Submersed Aquatic Vegetation Prevalence

High SAV prevalence (generally >50-percent) indicates quality habitat for waterfowl in the Upper Mississippi River System (Devendorf, 2013a, b; Larson and others, 2020) and has been the primary indicator of aquatic vegetation in the Upper Mississippi River (Yin and others, 2000; Johnson and Hagerty, 2008).

Methods

The SAV data were collected using LTRM standard procedures for SRS since 1998 (Yin and others, 2000). For assessing SAV prevalence, data from Pool 4 were further subdivided (postsampling stratification) into Upper Pool 4 and Lower

Summary of Submersed Aquatic Vegetation Prevalence

Indicator intent: Assess total submersed aquatic vegetation (SAV) prevalence and two common invasive SAV species, *Potamogeton crispus* L. (curly-leaved pondweed) and *Myriophyllum spicatum* L. (Eurasian watermilfoil), in Long Term Resource Monitoring element study reaches as biological indicators of ecosystem health.

Measurement: SAV was assessed visually and by using a plant rake according to Long Term Resource Monitoring element standard procedures (Yin and others, 2000). SAV prevalence was calculated as the proportion of sampled sites within a pool or stratum where SAV was detected.

Special considerations: Data were analyzed separately for Upper and Lower Pool 4 because of the important ecological contrasts for aquatic vegetation. Pools 26 and La Grange Pool were only sampled from 1998 to 2004. The Open River Reach was not included in vegetation sampling.

Assessment

Status: In Lower Pool 4 and 8, SAV prevalence is greater than 75-percent. In Upper Pool 4 and Pool 13, SAV prevalence is near or less than 50-percent. SAV prevalence was negligible in Pool 26 and La Grange Pool. Prevalence of all invasive SAV species was less than 25-percent system wide.

Trends: Pool-wide SAV prevalence increased ~30-percent in Upper Pool 4, Lower Pool 4, and Pool 8 during 2002 to 2010 and plateaued thereafter. Pool 13's SAV prevalence increased from 1998 to 2008 and then decreased from 2009 to 2019.

Pool 4 because of the large differences in SAV prevalence and other conditions in these two areas (Burdis and others, 2020). Prevalence was calculated based on the site-scale detection rate of SAV (for all species combined and for each individual species) as: prevalence (0–100 percent) = number of sites SAV was detected/total number of sites sampled*100.

We present the time series using annual means, standard errors, locally estimated scatterplot smoother (LOESS) trendlines, and 95-percent confidence bands to display intra- and interannual variability and long-term trends.

Assessment

The SAV prevalence varied by study reach, pool (figs. F1 and F2) and strata. The SAV prevalence was consistently greater in Lower Pool 4 and Pool 8 than in Upper Pool 4 and Pool 13. In 2019, prevalence of SAV in Lower Pool 4 and Pool 8 was greater than 75-percent, and SAV prevalence in Upper Pool 4 and Pool 13 was about 50-percent. In 2019, the maximum prevalence in Upper Pool 4 was less than the minimum prevalence in Lower Pool 4, Pool 8, and Pool 13. In all pools, prevalence was typically less than 50-percent within side channels and main channel borders, which are characterized by high water velocities and deep water. Contiguous backwaters and impounded areas of Lower Pool 4 and Pool 8 were greater than 75-percent SAV but about 50-percent SAV in Upper Pool 4 and Pool 13. From 1998 to 2004 in Pool 26, SAV was only detected at a few isolated backwater sites and was absent from all other strata; in the La Grange Pool, SAV was not detected.

The greatest rate of increase of SAV prevalence in pools and all major strata occurred during 2002 to 2010. Upper and Lower Pool 4 and Pool 8 increased in SAV prevalence from 50-percent to 80-percent during 2002–10 and have since stabilized. In Pool 8, the impounded area experienced the greatest increase in SAV, particularly in years 1998–2010. In contrast, Pool 13 increased from 40-percent to 60-percent SAV prevalence during the 1998–2010 period but subsequently decreased in SAV prevalence beginning around 2009.

Invasive SAV species prevalence generally remained low and invasive species were rarely dominant. *Potamogeton crispus* L. (curly-leaved pondweed), an invasive SAV species, fluctuated between 5 to 25 percent prevalence within the Upper Impounded Reach over the LTRM record. We caution that curly-leaved pondweed prevalence may be underestimated because it attains maximum biomass in about mid-May (spring), but LTRM sampling occurs in summer during senescence of this species. *Myriophyllum spicatum* L. (Eurasian watermilfoil) neared 40 percent prevalence in backwaters of Pools 4 and 8 but was usually a community member of a diverse SAV assemblage. System-wide, neither curly-leaved pondweed or Eurasian watermilfoil prevalence increased over the LTRM record.

Indicator: Aquatic Vegetation Diversity

Aquatic vegetation diversity is a biological indicator of systemic and local water quality, geomorphology, ecosystem integrity, and biological diversity of other biota (Larson and others, 2020a; Law and others, 2019). Maintaining diverse aquatic vegetation communities are primary ecosystem restoration objectives throughout the Upper Mississippi River System (U.S. Army Corps of Engineers, 2011; McCain and others, 2018).

Methods

Diversity was assessed using LTRM SRS data of vegetation prevalence and cover classes for all the life forms and species recorded in the field (see Yin and others, 2000). Sampling in year 2003 was constrained by budget and staffing constraints, and the data were omitted from diversity calculations and figures. All species detected in the LTRM sampling frame were included in diversity calculations, but sites where vegetation was not detected were excluded. We used the

Summary of Aquatic Vegetation Diversity

Indicator intent: Assess aquatic vegetation diversity for Pools 4, 8, and 13 from 1998 to 2019.

Measurement: Diversity indices were calculated using aquatic vegetation data for all the life forms and species recorded in the field (see Yin and others, 2000). We calculated Shannon's Diversity Index, species richness, and Shannon's evenness to assess plant community diversity. We also considered structural diversity, which is the presence of multiple life forms at a sampling site that compose habitat structure important for some wildlife species.

Special consideration: The Long Term Resource Monitoring element sampling frame effectively samples submersed aquatic vegetation but underrepresents emergent and moist soil communities. Pool 26 and La Grange Pool were only sampled from 1998 to 2004 and Open River was not sampled.

Assessment

Status: Pools 4, 8, and 13 exhibited moderately high diversity of aquatic vegetation species. Lower Pool 4 and Pool 8 had higher diversity indices and total species richness compared with Upper Pool 4 and Pool 13. Species evenness was moderately high and indicated that species were similar in abundance and frequency. Vegetation diversity was low in Pool 26 and the La Grange Pool of the Illinois River.

Trends: Upper Pool 4 and Pool 8 had notable increases in vegetation diversity since the 2000s. Diversity in Pool 13 decreased in 2015–19.

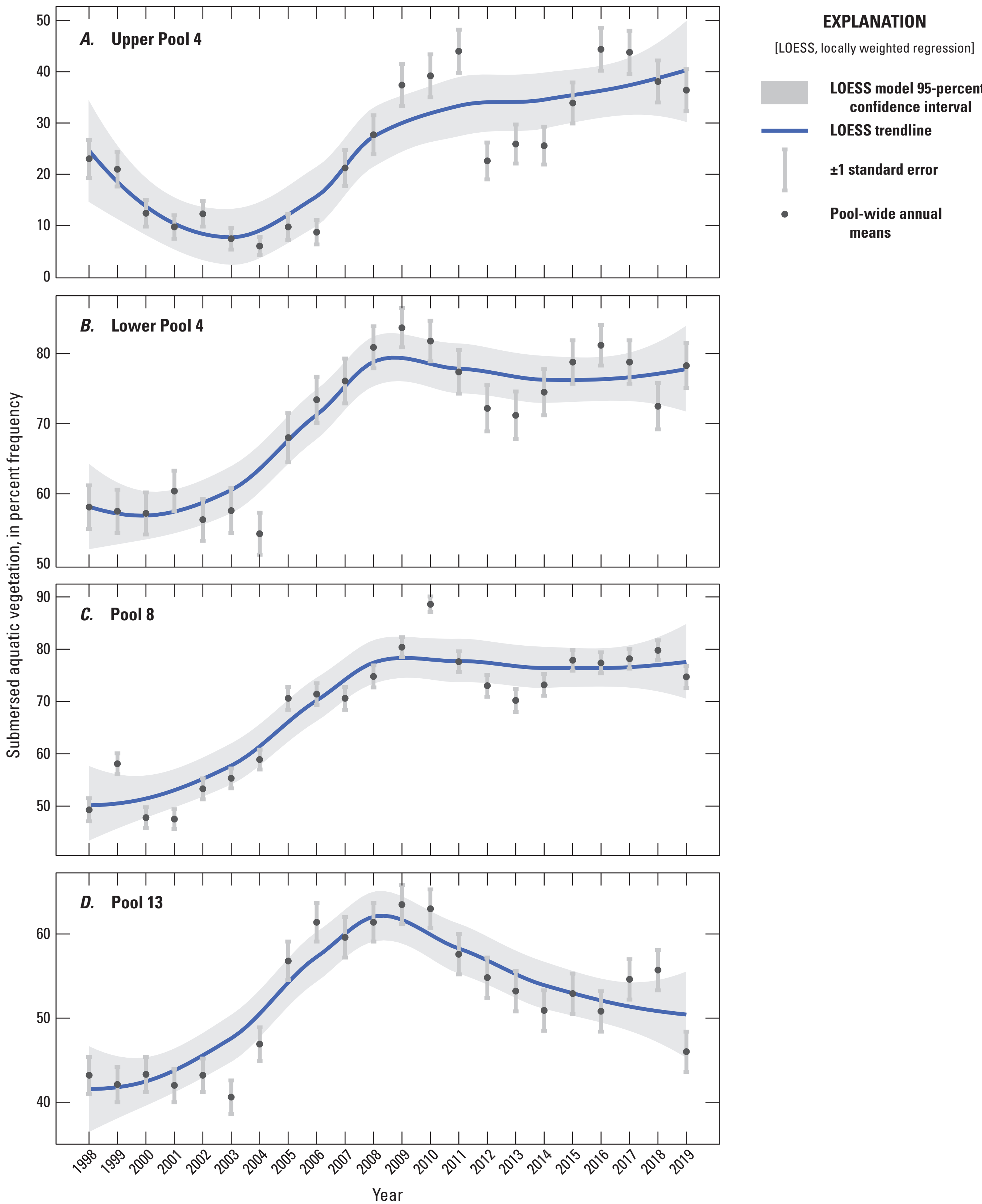

Figure F1. Time series of four pools in the Upper Mississippi River System from 1998 to 2019 for the pool-wide estimates of submersed aquatic vegetation prevalence. Note, the scales on the y-axes differ to highlight changes within pools through time.

Figure F2. Example of submersed aquatic vegetation in the Upper Mississippi River System; photograph by Deanne Drake, Wisconsin Department of Natural Resources, used with permission.

Shannon's Diversity Index and association tables to evaluate patterns in vegetation diversity and co-occurrences of species and life forms. Shannon's Diversity Index is a composite of the number of species observed (species richness) and their relative abundances (an evenness index). Shannon's Diversity Index scores increase as richness and evenness of the community increases. Evenness refers to diversity divided by the maximum diversity and is standardized to range from 0 to 1. Low evenness (values close to 0) indicate species are not distributed evenly, whereby some species are highly dominant and some species are rare. High evenness (values close to 1) indicates the plant species present have similar abundance.

We present the time series using annual means, standard errors, locally estimated scatterplot smoother (LOESS) trendlines, and 95-percent confidence bands to display intra- and interannual variability and long-term trends.

Assessment

More than 60 species of submersed, emergent, and rooted-floating and free-floating plants were detected with LTRM field sampling. The most common Upper Mississippi River aquatic vegetation species and vegetation life forms are listed in table F2. These included six submersed species, four emergent species (collected accurately within our data frame), two rooted-floating species, and two free-floating species complexes.

The LTRM study reaches in the Upper Impounded Reach exhibited diverse aquatic vegetation. Evenness was relatively high (*E*=0.75–0.85) at all LTRM study reaches, indicating the lack of a single or few dominant species. In Pool 13, Shannon's Diversity Index averaged 10 fewer aquatic species (about 25 species in Pool 13) than other pools in the Upper Impounded Reach (about 35 species in Pools 4 and 8) (fig. F3).

Trends in species richness, Shannon's Evenness, and Shannon's Diversity Index differed among Pools 4, 8, and 13 (figs. F3 and F4). Pool 4 diversity was stable throughout the study period, Pool 8 experienced a notable increase in diversity since ~2008, but Pool 13 had increasing diversity until year 2015, after which diversity declined. The mean richness at the beginning (1998) and end (2019) of the time series had decreased in Pools 4 and 13 but increased in Pool 8. There is no evidence of extirpations of plant species in the Upper Impounded Reach. Evenness remained between *E*=0.75–0.85 through time at all LTRM study reaches, which indicated that relative abundances of species have not substantially changed. Further, high evenness indicates that no specific plants are dominant and that most sites tend to have a mix of several species in similar abundance. Diversity at the stratum-scale generally mirrored that of the pool-scale, but diversity increases were most pronounced in contiguous backwater strata.

At the sampling site-scale, aquatic vegetation was often diverse. A single site often contained 6–10 species of aquatic plants. Structural diversity within the water column and above the water's surface was provided by the co-occurrence of emergent and submersed vegetation at about 80-percent of sites. The photograph in figure F5 shows an example of structural diversity. For example, annual wildrice (which rapidly expanded in Pools 4 and 8) usually co-existed with diverse assemblages of SAV species.

Factors driving the spatiotemporal patterns in aquatic vegetation diversity in the Upper Mississippi River System remain only partially understood. Aquatic vegetation communities were generally more species-rich and diverse in areas with moderate velocities and finer substrates. In strata with low connectivity and velocity, coontail is more likely to dominate (Carhart and De Jager, 2019) but can also be a member of a diverse assemblage. Strata with high velocities and deeper waters typically had low plant diversity, where only watercelery and *Heteranthera dubia* (Jacq.) MacMill

Table F2. Prevalence of the most common aquatic vegetation species in Pools 4, 8, and 13 of the Upper Mississippi River.

[%, percent]

Common name	Species name	Life form	Mean prevalence (0–100%; mean of years 2015–19)		
			Pool 4	Pool 8	Pool 13
Coontail	*Ceratophyllum demersum* L.	Submersed	34	51	32
Canadian waterweed	*Elodea canadensis* Michx.	Submersed	33	57	20
Grassleaf mudplantain	*Heteranthera dubia* (Jacq.) MacMill	Submersed	23	36	7
Watercelery	*Vallisneria americana* Michx.	Submersed	21	41	21
Sago pondweed	*Stuckenia pectinata* (L.) Börner	Submersed	16	13	18
Curly-leaved pondweed	*Potamogeton crispus* L.	Submersed	8	21	11
White waterlily	*Nymphaea odorata* Aiton	Rooted-floating	10	20	16
American lotus	*Nelumbo lutea* Willd.	Rooted-floating	2	6	17
Annual wildrice	*Zizania aquatica* L.	Emergent	11	21	0
Sessilefruited arrowhead	*Sagittaria rigida* Pursh	Emergent	7	11	3
Broadleaf arrowhead	*Sagittaria latifolia* Willd.	Emergent	4	7	6
River bulrush	*Bolboschoenus fluviatilis* (Torr.) Soják	Emergent	3	2	1
Filamentous algae	Conglomerate of algal species, not speciated	Free-floating	22	26	16
Duckweeds	*Spirodela polyrrhiza* (L.) Schleid., *Lemna minor* L., and *Lemna trisulca* L.	Free-floating	5	8	4

(grassleaf mudplantain) can establish (Bouska and others, 2022). In Upper Pool 4, where diversity increased, Burdis and others (2020) linked this diversity change to a several year period of low discharge and turbidity.

Indicator: Dominance of Free-Floating Plants

Free-floating plants in high biomass can indicate eutrophication because they have been associated with high nutrient concentrations and low oxygen conditions in the Upper Mississippi River System (Sullivan and Giblin, 2012; Houser and others, 2013; Giblin and others, 2014). Free-floating plants can become dominant, rapidly proliferate, prevent re-establishment of SAV (Irfanullah and Moss, 2004), shift habitat for riverine fishes and macroinvertebrates (Camp and others, 2014), and become a stable ecosystem state that is difficult to reverse (Scheffer and others, 2003). High abundance of free-floating plants in the Upper Mississippi River was a common source of public complaints because of its aesthetic and ecological degradation. However, free-floating plants may not necessarily inhibit SAV abundance or diversity. Free-floating plants can be a member of diverse assemblages in sites across the Upper Mississippi River System. Some free-floating plants like duckweeds can efficiently remove contaminants from river water, improve water quality (Amy-Sagers and others, 2017), and serve as a food resource for several waterfowl species.

Methods

Species that were considered in the "free-floating plants" life form categories included *Spirodela polyrrhiza* (L.) Schleid., *Lemna minor* L., *Lemna trisulca* L., *Wolffia columbiana* H. Karst., *Azolla* Lam., *Riccia fluitans* L., and *Ricciocarpos natans* L. Corda, and a conglomerate of filamentous algal groups (for example, Cladophora, Spirogyra, Microspora, and Oedogonium). Free-floating plant dominance is defined as free-floating plants at sites in relatively high abundance and SAV in low abundance or absent. To estimate relative abundances, we used rake scores and cover scores (Yin and others, 2000); specifically, a rake score or cover score of less than or equal to 1 was considered "low abundance," whereas rake or cover scores of 2 through 5 were considered "high abundance." Thus, free-floating plant dominance is indicated at all sites where the free-floating plant cover score or rake score is greater than or equal to 2 and the SAV rake score was less than 2. We reported the relations between free-floating plants and SAV because the dominance of one might lead to the scarcity of the other, and dominance of free-floating plants can hinder SAV abundance (Scheffer and others, 2003).

To assess the association between free-floating plants and SAV, we used mosaic plots and statistics of independence. The mosaic plot was based on conditional probabilities and visualized counts in a contingency table by tiles whose area is proportional to the cell count. Under the null hypothesis of independence (in other words, no association between free-floating plants and submersed plants), the heights of the

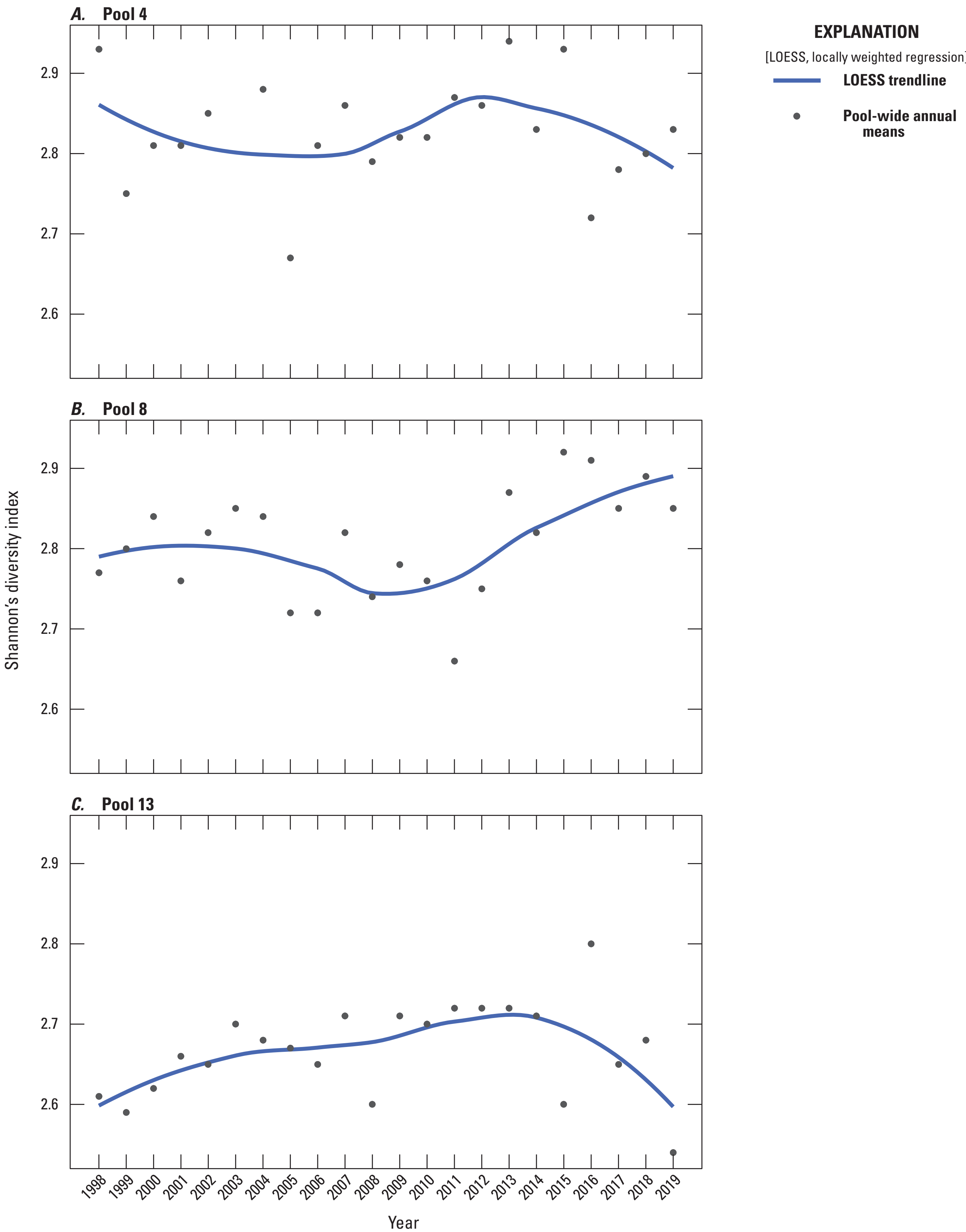

Figure F3. Time series of three Long Term Resource Monitoring element study reaches in the Upper Mississippi River System from 1998 to 2019 for aquatic vegetation diversity represented with Shannon's Diversity Index.

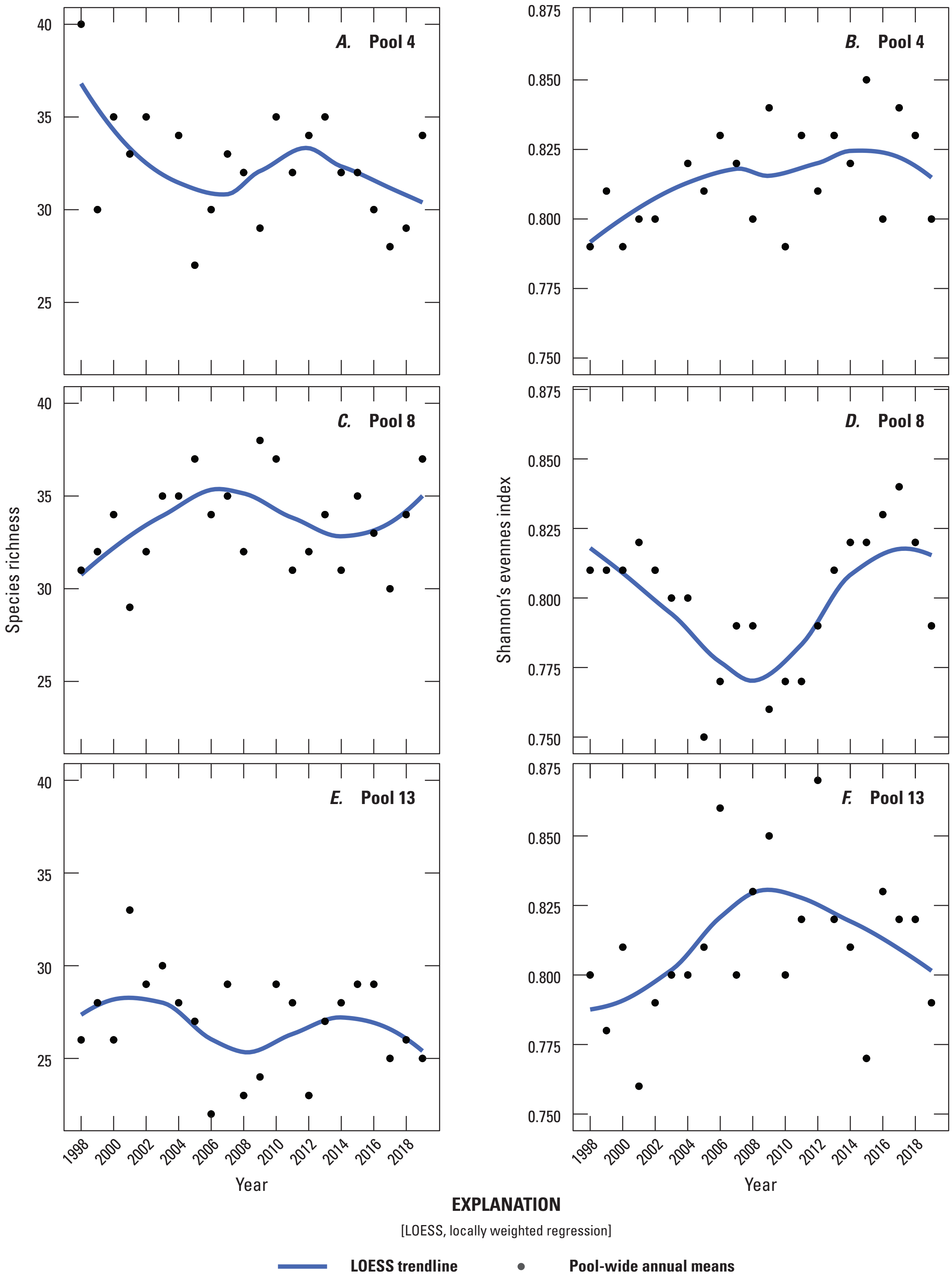

Figure F4. Time series of three Long Term Resource Monitoring element study reaches in the Upper Mississippi River System from 1998 to 2019 for species richness and Shannon's Evenness Index.

Figure F5. Structural diversity in the Upper Mississippi River was provided by multiple aquatic vegetation life forms, including rooted-floating vegetation, emergent vegetation, and free-floating plants. Photograph by Deanne Drake, Wisconsin Department of Natural Resources, used with permission.

tiles in each row are expected to be the same. To assess the strength and direction of the association, we used Pearson standardized residuals that measure the departure of each cell from independence. Residuals were measured in units of standard deviations, so residuals of >2 or <−2 are statistically significant departures from independence and roughly correspond to a p-value of 0.05; residuals of >4 or <−4 are also significant with a p-value of <0.0001.

Assessment

Across the system, sites with free-floating plant dominance were either populated by duckweeds or filamentous algae; the filamentous algal and duckweed mats rarely co-occurred in high abundance (Pearson residual =−4; p-value <0.0001). Sites were two times more likely to be dominated by duckweed than by filamentous algae.

At all LTRM study reaches, through the 21-year record, dominance of free-floating plants was measured at between 2 and 5-percent of all the LTRM's randomly selected study sites. This equates to roughly 60–300 hectares per LTRM study reaches that were dominated by free-floating plants. There were no substantial differences of dominance among LTRM study reaches (Pools 4, 8, 13, 26, La Grange) from 1989 to 2004.

The mosaic plot showed the relative number of sites and relations between free-floating plants and SAV at all LTRM study sites over the period of record (fig. F6). On average, 4-percent of the LTRM sites were dominated by free-floating plants and had little or no SAV (fig. F6, dark red boxes; Pearson residual =−4; p-value <0.0001). Collectively, >60-percent of LTRM sample sites did not have any SAV or free-floating plants detected (fig. F6, dark gray box). When SAV was

Summary of Dominance of Free-Floating Plants

Indicator: Dominance of free-floating plants and filamentous algae (hereafter "free-floating plants" and sometimes referred to as "metaphyton" in the literature).

Indicator intent: Assess free-floating plant dominance as an indicator of river eutrophication and aesthetic degradation.

Measurement: The free-floating plants in the Upper Mississippi River System are typified by duckweeds and filamentous algae. Free-floating plant dominance was defined as sites with free-floating plants in relatively high abundance and submersed vegetation in low abundance or absent.

Special consideration: Sites with dominance of free-floating plants may indicate eutrophication, poor fish habitat, and poor waterfowl habitat. However, free-floating plants in low abundance and coupled with submersed aquatic vegetation are considered a positive contributor to aquatic vegetation diversity. We include data for Pools 4, 8, 13, and 26 and La Grange Pool, but caution that Pool 26 and La Grange Pool have substantially fewer samples and no samples past 2004, which can make comparisons difficult. Temporal trends are not shown because low detection rates of free-floating plants (~10 sites per year in the Upper Impounded Reach, and no data after 2004 in the Lower Impounded Reach) precluded reasonable trend estimates.

Assessment

Status: Free-floating plants are the dominant vegetation type at ~3–20-percent of Long Term Resource Monitoring sites, with a higher probability of dominance in Pool 26 and La Grange.

Trends: Not shown; see special considerations.

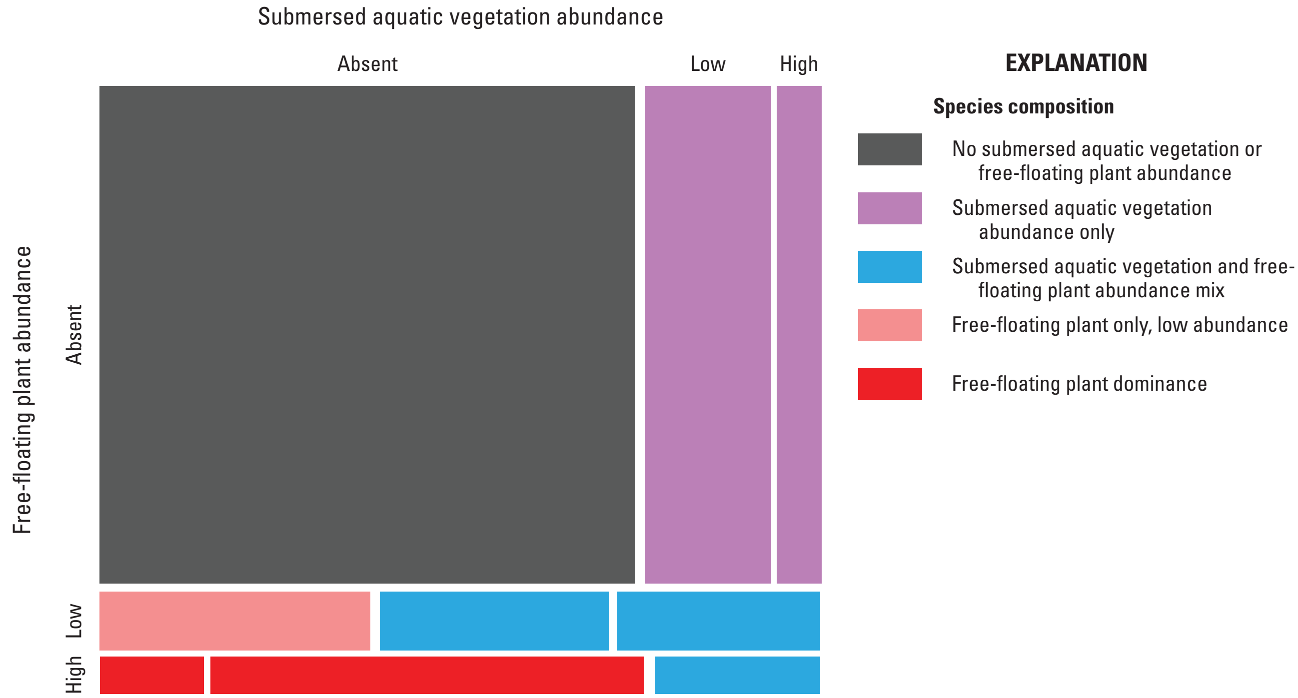

Figure F6. Mosaic plot emphasizing the co-associations between submersed aquatic vegetation and free-floating plant relative abundances at Long Term Resource Monitoring sites in Pools 4, 8, 13, and 26 and La Grange Pool (n=40,613 vegetation sampling sites over the entire record). The size of each box is proportional to the number of observations in each category.

detected, 80-percent of those sites had no free-floating plants (fig. F6, violet boxes; Pearson residual =−4; p-value <0.0001). About 8-percent of sites had free-floating plants in low abundance without SAV (fig. F6, light red box), which can either indicate potential for a shift to free-floating plant dominance, or sites simply had incidental drift by waves and wind and is not likely competitive dominance.

Free-floating plant dominance occurred mostly in contiguous and isolated backwaters (fig. F7) and occasionally in strata with greater water velocities (main channel border and impounded areas). In previous work, areas of lower connectivity to the main channel moderately increased the probabilities of free-floating plant presence (Houser and others, 2013; Giblin and others, 2014). Factors favoring free-floating plant abundance in the Upper Mississippi River System included slow water velocity, warm waters, shallow depths (<1 meter [m]), and high total nitrogen and total phosphorus concentrations (Giblin and others, 2014).

Indicator: Emergent Vegetation

Emergent vegetation provides energy and habitat for aquatic invertebrates, fish, and water birds in the Upper Mississippi River (Flinn and others, 2005; Stafford and others,

Figure F7. Examples of *A*, filamentous algae and *B*, duckweeds (both free-floating plants) dominance in the Upper Mississippi River; photographs by Shawn Giblin, Wisconsin Department of Natural Resources, used with permission.

2007; Coulter and others, 2019). Emergent vegetation species are herbaceous, vegetation that root in inundated or seasonally saturated soil, whereas most vegetative biomass and reproductive structures grow above the water.

Methods

Two complementary datasets were used to evaluate the distribution of select emergent vegetation species (land cover and SRS). The land cover data were used to estimate the surface area of every navigation pool in the Upper Mississippi River System in 1989, 2000, and 2010. The land cover data included several classified emergent marsh communities: annual and perennial deep and shallow marshes, deep and shallow marsh shrubs, rooted floating marshes, and sedge meadow marshes (Dieck and others, 2015). These communities represent the wide range of semi-aquatic species in the Upper Mississippi River System, which are fully listed in Dieck and others (2015).

We interpolated emergent coverage from SRS data and quantified areal extent of cover categories in Pools 4, 8, and 13 using ArcMap version 10.6. Input data were the LTRM categorical cover rating (0–5) recorded in the field from the nearest three sites sampled during two periods, 1998–2008 and 2009–19. We chose to pool data into these two periods because we wanted to show spatial data in maps of each pool (so annual coverage estimates would yield too many maps), and the greatest change in emergent coverage began around year 2009. Interpolations used the inverse distance weighted method (Spatial Analyst toolbox) to estimate cover from the nearest three sites. Output rasters with a 25-m pixel resolution were classified as four discrete ranges of emergent cover values (0-percent, <20-percent, 21–40-percent, and >40-percent) for each of the two periods. The sampling frame for SRS data was restricted to depths less than 2.5 m where *Sagittaria latifolia* Willd. (broadleaf arrowhead), *Sagittaria rigida* Pursh (sessilefruited arrowhead), annual wildrice, and *Bolboschoenus fluviatilis* Torr. Soják (river bulrush) comprised 95 percent emergent vegetation detections. We described prevalence of annual wildrice at the navigation pool scale using annual means, standard errors, locally estimated scatterplot smoother (LOESS) trendlines, and 95-percent confidence bands to display intra- and interannual variability and long-term trends from the SRS data.

Assessment

The land cover data indicated that emergent coverage ranged from 5 to 25-percent of total pool area of all the pools within the Upper Impounded Reach (Pools 2–13, fig. F8). For the rest of the system, emergent cover was typically less than 5-percent of each pool area. Total marsh cover decreased in

Summary of Emergent Vegetation

Indicator intent: Assess the area of emergent vegetation in all pools of the Upper Mississippi River System from 1989 to 2019.

Measurement: Two datasets were used. The land cover data are based on systemic aerial photography collected in 1989, 2000, and 2010–11; span the entire Upper Mississippi River System; and were used to describe the area (in hectares) and proportion of total area with emergent cover for each navigation pool on the Upper Mississippi River and Illinois River and two reaches within the Unimpounded Reach. The stratified random sampling (SRS) data were collected by annual field sampling in Pools 4, 8, and 13 of the Upper Mississippi River since 1998 (Yin and others, 2000) and include the metrics of emergent vegetation, prevalence, and percent cover. The SRS measures were spatially interpolated to estimate aerial coverage.

Special consideration: The land cover dataset covers the river and floodplain, including temporarily flooded areas. In contrast, the SRS dataset only includes emergent vegetation within the aquatic sampling frame, which accurately reflects deep marsh perennial and deep marsh annual species only.

Assessment

Status: Land cover data showed emergent species covered between about 5 and 25-percent of pools within the Upper Impounded Reach since 1998, but usually less than 5-percent of the pools within Lower Impounded Reach, Open River Reach, and Illinois River. The SRS field data showed that as of 2019, emergents covered about 1,500 hectares in Pool 4, about 2,000 hectares in Pool 8, and about 1,000 hectares in Pool 13. In addition, *Zizania aquatica* L. (annual wildrice) was at 30 percent prevalence in Pools 4 and 8 but not Pool 13 (where annual wildrice has not been detected).

Trends*:* Land cover data showed emergent coverage increased since 1998 in 8 out of 13 pools within the Upper Impounded Reach. The emergent coverage more than doubled in several pools of the Lower Impounded Reach, Open River Reach, and Illinois River between 2000 and 2010, although coverage remained less than 5-percent of the total pool area. The SRS data indicated that in Pools 4 and 8, arrowheads were the dominant emergent taxa before 2009, but annual wildrice dominated and continued to increase in prevalence after 2010. In Pool 8, annual wildrice expanded to 30 percent prevalence, and its coverage increased by 40-percent since 2010.

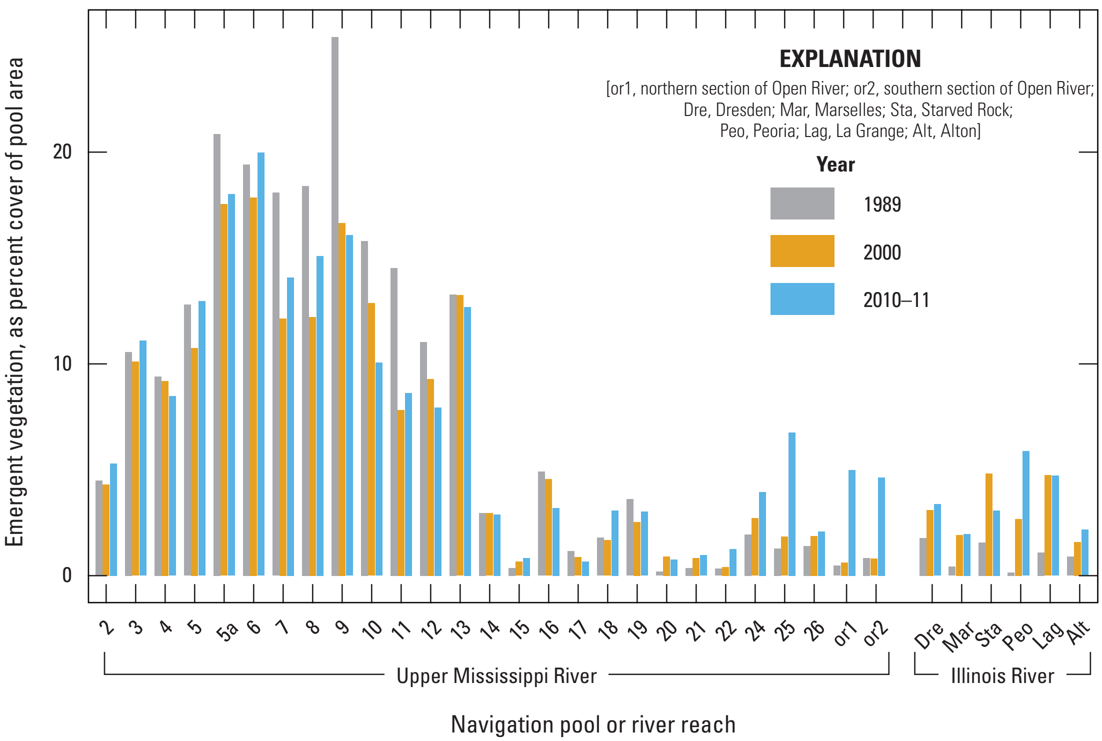

Figure F8. Percentage cover of emergent vegetation across all pools within the Upper Mississippi River System for years 1989, 2000, and 2010–11. Emergent vegetation is classified by 8 marsh classes composed of approximately 45 species as listed in Dieck and others (2015).

Pools 4, 9, 10, 12, 16, and 17 from 1989 to 2010, although the magnitude of decrease varied by pool. In contrast, total marsh cover in Pools 22–25, Open River (or1 and or2), and most pools within the Illinois River increased over this period. Emergents in Pools 25, Open River, and Peoria (within the Illinois River) increased at least fivefold from 1989 to 2010.

The SRS data indicated that percent cover of emergent vegetation expanded greatly (fig. F9, fig. F10). The annual SRS data indicated that as of 2019, emergents in aquatic habitats (excluding floodplains) covered ~1,500 hectares in Pool 4, ~2,000 hectares in Pool 8, and ~1,000 hectares in Pool 13 (fig. F9). The 10-year mean emergent vegetation coverage was greater during 2009–2019 than during 1998–2009 in Pools 4, 8, and Pool 13. After 2009, open water coverage (defined by the absence of emergent cover, but not necessarily SAV) decreased by 30–40-percent (or 556 hectares and 1,427 hectares) in Pools 4 and 8. After 2009, open water coverage decreased by only 5-percent (350 ha) in Pool 13, indicating less expansion of emergents in Pool 13 compared with Pools 4 and 8. The largest increase in emergent vegetation since year 2009 occurred in the contiguous backwaters, impounded areas (399 to 1,516 hectares), and side channels (11 to 211 hectares) of Pool 8. Arrowheads were the dominant emergent taxa in all strata before 2009 and decreased in Pools 4, 8, and 13 from 2009 to 2019.

Little to no annual wildrice was detected in the land cover data before 2010. The SRS data indicated that annual wildrice became the dominant emergent species in Pools 4 and 8 after 2010. Annual wildrice has not been detected in Pool 13. The SRS data documented a substantial increase in annual wildrice prevalence from ~2009–19 in Pools 4 and 8 (fig. F11), which covered more than 1,000 hectares in each of those pools. From year to year, coverage was more variable in the backwaters and more stable in the impounded area.

State of the Ecosystem

Aquatic vegetation in the Upper Impounded Reach of the Upper Mississippi River increased substantially from 1998 to 2009. From 2009 to 2019, SAV prevalence plateaued at ~75-percent in Lower Pool 4 and Pool 8, and this plateau that occurred for at least a decade indicated the stability of SAV prevalence. The SAV in Pools 4, 8, and 13 maintained high prevalence and demonstrated resilience to extended flood conditions over several consecutive years (2017–19). Some LTRM study pools in the Upper Impounded Reach also had notable increases of emergent vegetation after 2010. As of 2019, emergent and submersed vegetation co-occurred at 80-percent of sites, which can create substantial vegetation structure, biomass, and diversity (fig. F5). Prevalence of annual wildrice has increased by an order of magnitude in the past 10 years and covered thousands of hectares in Pools 4 and 8 (figs. F8, F9, and F11). Primary causes of increased aquatic vegetation and diversity are not yet well understood, but likely include multiple interactive factors such as increased water clarity, decreased external and internal nutrient loading, reduced bioturbation from decreasing abundance of *Cyprinus carpio* (common carp), and moderately low discharge for several consecutive years in the 2000s (see ch. H). About 2–5-percent of sites in the Upper Impounded Reach were dominated by free-floating plants, and those sites

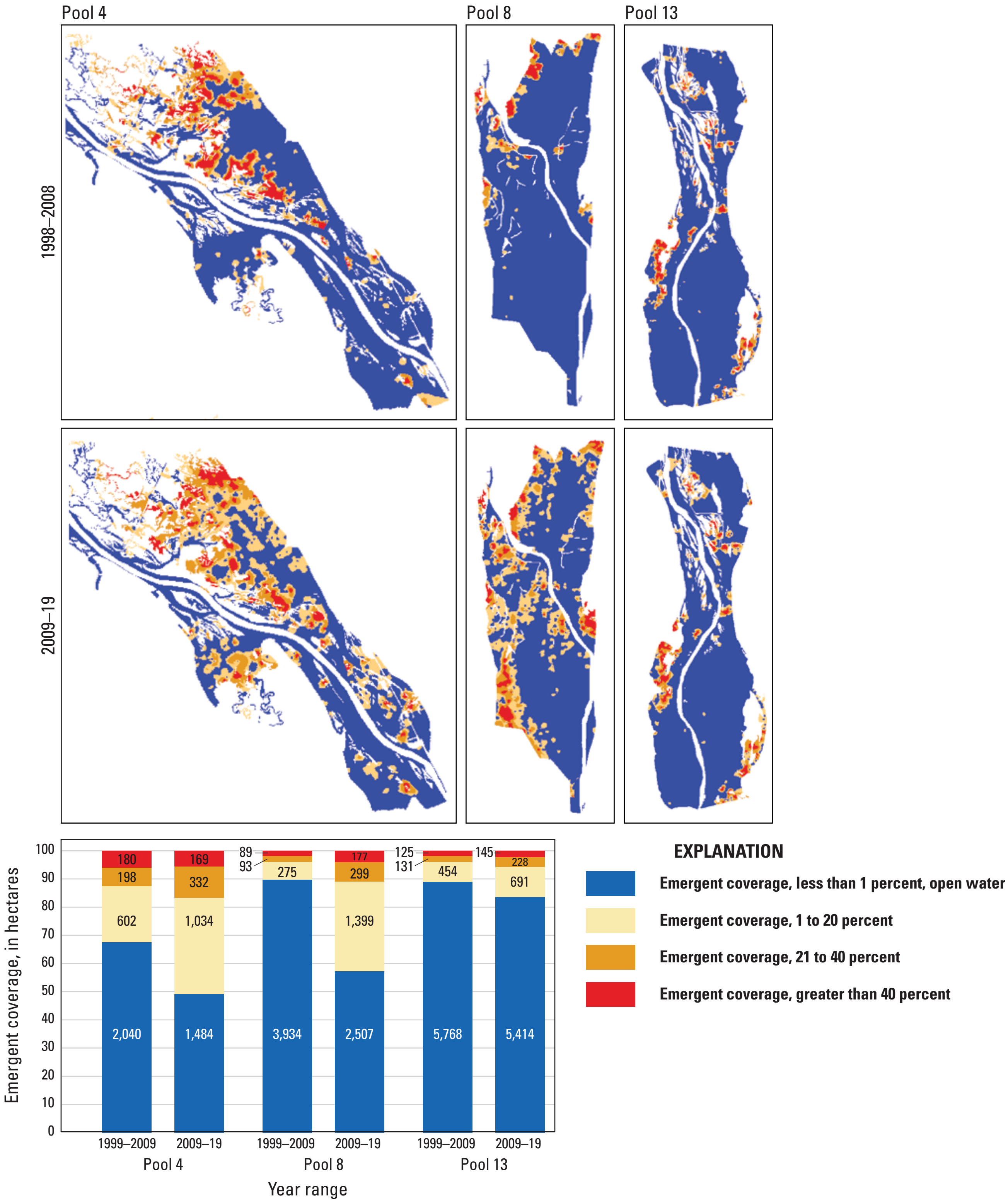

Figure F9. Interpolations of stratified random sampling data highlight the changes in emergent coverage over time in Lower Pools 4, 8, and 13. There were four cover categories: less than 1-percent emergent cover (open water where no emergents were present but submersed aquatic vegetation may or may not be present); 1–20-percent emergent coverage; 21–40-percent emergent coverage; and greater than 40-percent emergent coverage. We split the data into two periods (1998–2008 and 2009–19) because there was a notable change in emergent cover around 2009. Labels within the stacked bar charts are number of hectares in each coverage category.

Figure F10. Emergent vegetation expanse on Pool 8 of the Upper Mississippi River in 2020; photograph by Deanne Drake, Wisconsin Department of Natural Resources, used with permission.

may have been adversely affected ecologically and aesthetically by an excess of those plants.

Pool 13 differs from Pools 4 and 8 in the Upper Impounded Reach in terms of the status and trends of SAV prevalence and diversity and emergent coverage and composition. Like Pools 4 and 8, the SAV prevalence in Pool 13 increased from 1998 to 2012; but unlike the other pools, Pool 13 SAV prevalence decreased from 2012 to 2019. The SAV prevalence in Pool 13 was consistently lower than other Upper Impounded Reach LTRM study reaches. From 2005 to 2019, Pool 13 SAV prevalence hovered around 50-percent, which has been proposed as an approximate threshold for a "clear-water state" in shallow lakes regionally and worldwide (Scheffer, 2004; Larson and others, 2020a). The Upper Mississippi River diving duck model scored pools with >50-percent SAV prevalence as the maximum score for habitat suitability (Devendorf, 2013b), and Pool 13 bordered that threshold. The SAV diversity in Pool 13 also decreased since 2012, but diversity had generally increased in the other LTRM study reaches. Annual wildrice had not been detected in Pool 13 over the LTRM record, but annual wildrice flourished in Pools 4 and 8. The reasons for differences in status and trends of Pool 13 in relation to Pools 4 and 8 are not yet well understood.

Several agencies viewed the current state of aquatic vegetation diversity and prevalence in the Upper Impounded Reach as within "desirable conditions," but others suggested more actions were needed to expand vegetation distribution and diversity (McCain and others, 2018). The LTRM data indicated that all LTRM study reaches in the Upper Impounded Reach since 2005 had attained the highest scoring possible in the Upper Mississippi River Diving and Dabbling Duck Habitat Suitability Models, which aimed for pool-wide SAV prevalence greater than 50-percent and the common emergent species at 20–30-percent coverage (Devendorf, 2013a, b).

From 1998 to 2004, SRS and land cover data revealed sparse submersed and emergent vegetation in the Lower Impounded Reach, Unimpounded Reach, and most pools of the Illinois River. Restoration practitioners and natural resource professionals assessed the aquatic and floodplain vegetation in the Lower Impounded, Unimpounded, and Illinois River reaches as "deviating from desired conditions" as of 2010 (McCain and others, 2018). However, aquatic vegetation was rated as "moderate [restoration] importance" in these reaches, except for the upstream pools of the Illinois River where aquatic vegetation was the most important indicator of ecosystem structure and function.

The Lower Impounded Reach (Pools 14–26) typically had low prevalence of aquatic plants (De Jager and Rohweder, 2017) and dominance of free-floating plants on occasion. This reach was characterized by less lentic shallow area, more structured channels and levees, and lower water clarity compared with the Upper Impounded Reach (De Jager and others, 2018), which are likely reasons aquatic vegetation was scarce. In addition, Pools 21–26 experienced water-level fluctuations as much as 4 m within a growing season that likely limited plant establishment (Carhart and others, 2021). Pool 19 has the greatest amount of suitable SAV habitat at ~2,000 hectares, and this habitat is comparable to the habitat amount in Lower Pool 4 in the Upper Impounded Reach (Carhart and De Jager, 2019). Water-level management in Pool 26 after two drawdowns showed positive response of emergent vegetation and fish use (Flinn and others, 2005; Coulter and others, 2019). This reach was rated as "substantial deviations from

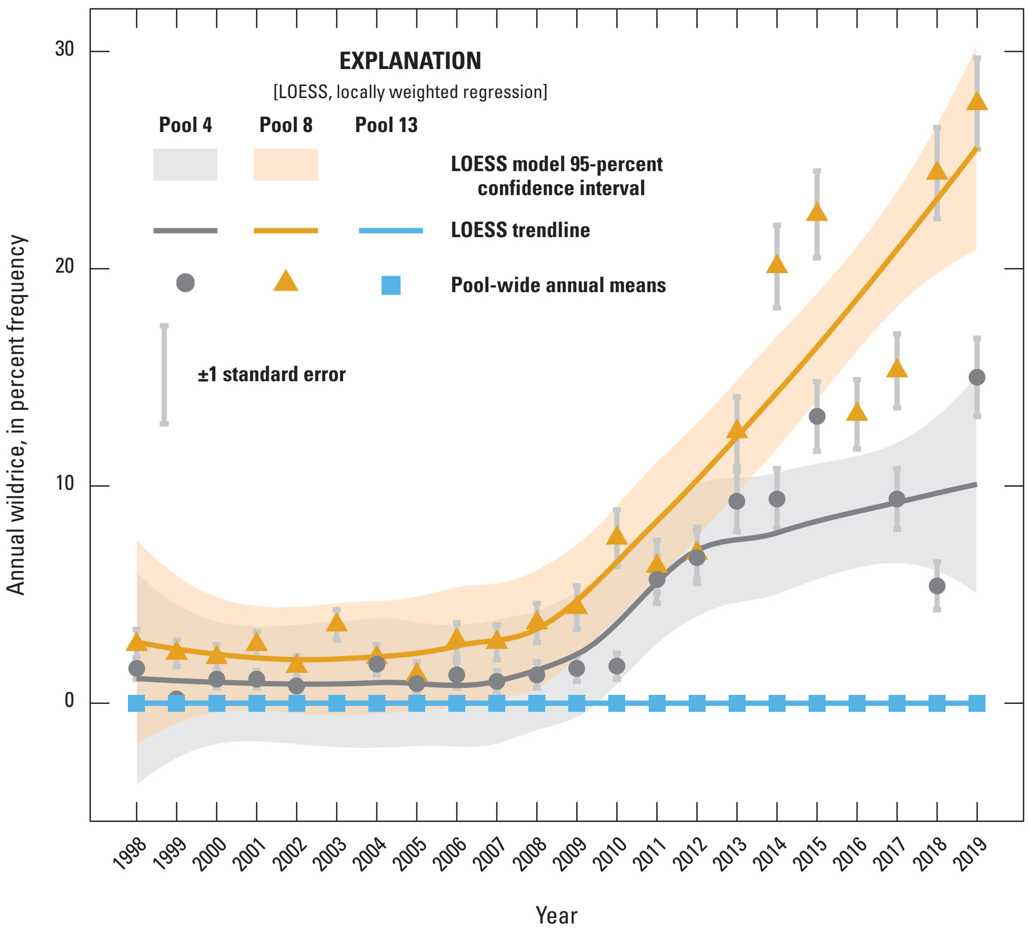

Figure F11. Time series of three Long Term Resource Monitoring element study reaches in the Upper Mississippi River System from 1998 to 2019 for the prevalence of *Zizania aquatica* L. (annual wildrice).

defined, desired conditions" with moderate restoration priority (McCain and others, 2018).

The Unimpounded Reach (Open River pools called or1 and or2) likely cannot support much aquatic vegetation because this reach contains few shallow areas and slow water velocities (Peck and Smart, 1986). The land cover data showed that emergents increased from <1-percent to 5-percent of reach area in Open River since 1989 (fig. F8). Other land cover analyses also indicated that Open River supports small areas of submersed and rooted-floating plants as well (De Jager and Rohweder, 2017). SAV is likely to remain scarce in the Open River due to scarce habitat availability (Carhart and others, 2021).

The Illinois River historically supported lush aquatic vegetation (Bellrose and others, 1979) and continues to be a vitally important area for waterfowl habitat (Stafford and others, 2007; FWS, 2011a). The land cover data showed most pools throughout the Illinois River had sparse vegetation (fig. F8; De Jager and Rohweder, 2017). All pools throughout the Upper and Lower Illinois River had abundant, shallow lentic areas as suitable geomorphic habitat for aquatic vegetation (De Jager and others, 2018).

Primary limitations to aquatic vegetation specific to the Illinois River were a lack of water clarity, localized herbivory, and large water-level fluctuations (Sass and others, 2017; Carhart and others, 2021). The Peoria Pool is estimated to have greater than 4,000 hectares suitable for vegetation restoration (Carhart and others, 2021), particularly if the other limiting factors were fully identified and addressed. A private survey conducted in 2005 showed that several pools in the Illinois River had high abundance of filamentous algae and low abundances of emergent and submersed vegetation (Basler and Elzinga, 2006), although the LTRM data indicated dominance of free-floating plants was rare before 2004. A notable exception was the Dresden Pool, which had 50-percent SAV prevalence that included watercelery. The Dresden Pool also had diversity indices comparable to the Upper Impounded Reach during the same year (Cook and McClelland, 2007). Most pools in the Illinois River "substantially deviated from defined, desired conditions" for aquatic vegetation and floodplain vegetation, and vegetation establishment is the top restoration priority in the Dresden, Marseilles, and Starved Rock Pools (McCain and others, 2018).

Future Pressures

Invasive Species

Two nonnative SAV species (Eurasian watermilfoil and curly-leaved pondweed) were present throughout the Upper Mississippi River System during the LTRM period of record. Both species' prevalence decreased in the past decade and may continue a downward trajectory. Both species were usually members of a relatively diverse SAV assemblage and did not tend to dominate in the Upper Mississippi River System as has been observed in regional wetlands and lakes. In some places, curly-leaved pondweed exhibits very high standing biomass. In these places, curly-leaved pondweed contributes to local nutrient remineralization, which may contribute to early summer proliferation of free-floating plants (Sullivan and Giblin, 2012) and to supersaturation of oxygen under winter ice that may induce fish kills from gas-bubble disease. Factors affecting the distribution and abundance of these nonnative species in the Upper Mississippi River System are not well understood.

Butomus umbellatus L. (flowering rush) is a nonnative emergent species that was first observed in the Upper Mississippi River System in 2009 as a few plants at limited locations in Pool 13. From 2009 to 2020, flowering rush detections increased in Pool 13. The first detections of flowering rush in Pools 4 and 8 were during 2020. The size of individual patches of flowering rush varied from only a few flowers to greater than several hectares. Multiple agencies within the partnership began coordinating control efforts throughout the Upper Mississippi River System in response. This nonnative species can quickly spread and outcompete native species and can flourish in rivers with similar conditions to the Upper Mississippi River System (Hudon, 2004; Madsen and others, 2016a, 2016b). Future LTRM monitoring may reveal the extent to which flowering rush establishes, spreads, and affects native vegetation diversity.

Discharge Variability and Responses

The Upper Mississippi River System experienced several years of nearly continuous high discharge from 2017 to 2019 (ch. B). High discharge has not reduced aquatic vegetation as some expected, which may complicate future predictions. For example, annual wildrice continued to expand into deeper waters, which demonstrated this species' adaptability to variable Upper Mississippi River System conditions. The predicted suitable amount of habitat for submersed plants may be sensitive to changes in bed elevation and falling water levels, particularly in the Upper Impounded Reach, Pool 19, and Peoria Pool (Carhart and others, 2021). Because emergent plants, free-floating plants, SAV, and potentially competing phytoplankton respond strongly to water-level fluctuations and water depth, future trends of primary production by phytoplankton and aquatic vegetation would likely be tightly coupled to and predicted by riverine discharge (Dodds and others, 2013; Burdis and others, 2020; Giblin and Gerrish, 2020).

Restoration and Management of Aquatic Vegetation

Many past, current, and planned Habitat Rehabilitation and Enhancement Projects used various techniques to increase the distribution and prevalence of aquatic vegetation. Restoration techniques are implemented at many spatial scales, including reach, pool, and localized scales. At reach scales, techniques include modifying hydraulic connectivity among aquatic areas to reduce high wind fetch, velocities, and wave action (Sefton, 1976; Kimber and others, 1995a). At the pool scale, techniques include constructing islands (Carhart and De Jager, 2019; Drake and others, 2018; Langrehr and others, 2007) or aiming to improve water clarity (Kreiling and others, 2007). Pool-scale water level management has been implemented throughout the Upper Mississippi River because of the demonstrated ecological benefits to emergent species (Woltemade, 1997; Sparks and others, 1998; Theiling and Nestler, 2010; Kenow and others, 2016), as well as the associated macroinvertebrate and fish communities using the emergent vegetation (Flinn and others, 2005; Coulter and others, 2019). At local scales, water level management using control structures can induce seeding (Kimber and others, 1995a, 1995b; Woltemade, 1997) or Habitat Rehabilitation and Enhancement Projects can attempt hand seeding and transplanting tubers. Management agencies may benefit from establishing specific aquatic vegetation goals during the Habitat Rehabilitation and Enhancement Projects formulation to enable better communication and evaluate progress towards achieving those goals; for example, constructed islands in Pool 8 shifted the aquatic plant assemblage to more lentic and diverse vegetation stands (Carhart and De Jager, 2019), which was either a benefit or unintended consequence depending on the agency's restoration objectives.

Measures to reduce the extent of free-floating plant dominated sites in the Upper Mississippi River System have been proposed to reduce the ecological effects and public complaints regarding aesthetic degradation (fig. F7). Restoration techniques that have been tried in other systems include harvest of free-floating plants (Scheffer and others, 2003), backwater drawdown via control structures (Larson and others, 2020a), removal of nonnative curly-leaved pondweed to also alleviate filamentous algal domination (Zhang and others, 2020), and the excavation of nutrient-laden sediments (Larson and others, 2020b). In addition, systemic management of external and internal nutrient loading to specific thresholds may greatly reduce free-floating plants in multiple Upper Mississippi River backwaters simultaneously (James and Barko, 2004; Giblin and others, 2014).

Potential Future Monitoring and Research

Re-establishing aquatic vegetation in some unvegetated reaches of the river is a longstanding restoration priority (McCain and others, 2018). In the less vegetated reaches of river (for example, Pool 13, the Lower Impounded Reach, and most of the Illinois River), continued monitoring via decadal land cover surveys would help detect large changes to vegetation. However, periodic field surveys would provide more detailed information on plant distribution and community shifts at finer scales that contribute to understanding if, when, and why vegetation rebounds in those areas. Seedbank assessments would help identify which species are present, whether seeds are viable, and the role of dispersal in plant re-establishment. Continued experimental plantings of vegetation may further identify factors limiting vegetation and how restoration may overcome those factors. Lastly, research studies where the study design targets sites dominated by free-floating plants and high chlorophyll *a* concentration (ch. E) would further improve our understanding of the factors allowing their dominance and their biotic interactions with other species.

The current LTRM datasets are valuable because they help us understand aquatic plant community assembly and associated ecological processes. Understanding community assembly (that is to say, which species are likely to co-occur or not at small spatial scales) and related ecological processes (that is to say, environmental drivers and biotic interactions) can inform river restoration and management. Understanding community assembly would improve our ability to predict the effects of restoration projects on community diversity, specific aquatic assemblages, and individual species. Important goals for continued long-term monitoring and complementary research projects include identifying the controlling variables and feedbacks of priority species and communities to improve the resiliency of the system (Bouska and others, 2019) and the incorporation of those results into predictive models to facilitate restoration and management.

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Chapter G: Fisheries Indicators

By Brian S. Ickes, Mel C. Bowler, Andrew D. Bartels, and Kristopher A. Maxson

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Chapter G: Fisheries Indicators

By Brian S. Ickes,[1] Mel C. Bowler,[2] Andrew D. Bartels,[3] and Kristopher A. Maxson[4]

Overview

Nearly one in every four freshwater fish species in North America is native to the main-stem reaches of the Upper Mississippi River System (Patrick, 1998). As such, the Upper Mississippi River System represents a nexus of freshwater fish diversity in North America and a key natural heritage resource for the United States (Ickes, 2008). As of 2019, the Long Term Resource Monitoring (LTRM) element Fisheries Component has observed 143 fish species and greater than 4.2 million individual fish in the 6 LTRM study reaches using scientific methods and protocols (Ickes and others, 2014; Ratcliff and others, 2014). The fish community consists of species of ancient evolutionary origins (Ickes, 2008), economically and socially important recreational and commercial species (Sparks, 2010; Schramm and Ickes, 2016), species of special conservation concern (Ickes and others, 2005; Ickes, 2018), and invasive and naturalized nonnative species (Ickes, 2008; Irons and others, 2009). The Upper Mississippi River System is probably the most biologically productive and economically important large floodplain river system in the United States (Patrick, 1998; U.S. Geological Survey, 1999), and fish are one of the most important goods and services provided to humans (Carlander, 1954). Scientists and fishery managers also recognize fish communities as an integrative index for a complex set of physical and biological conditions (U.S. Geological Survey, 1999; Ickes and others, 2005; Johnson and Hagerty, 2008).

The objective of the Upper Mississippi River Restoration (UMRR) program LTRM Fisheries Component is to collect quantitative information on the distribution and abundance of fish species and communities and to complete research on fish to understand resource status and trends, ecological dynamics, fish community response to disturbances, and support UMRR restoration activities. Data are collected within six study reaches (Pools 4, 8, 13, 26, and Open River Reach on the Upper Mississippi River and La Grange Pool on the Illinois River; see fig. A1). Sample allocations are based on a spatially stratified random design (Gutreuter and others, 1995; Ickes and others, 2014; Ratcliff and others, 2014), where strata include contiguous backwaters, main channel borders, impounded areas, and secondary channel borders (figs. A6–A11). Sampling effort is allocated independently and equally across three sampling periods annually (June 15–July 31; August 1–September 15; September 16–October 31) to minimize risks of annual data loss during flood periods and to characterize seasonal patterns in abundance and habitat use. Annual indices expressing the mean and variance of catch per unit effort (CPUE) and mass per unit effort (MPUE) are gained by applying standard stratified random sampling design estimators (Ickes and others, 2014). Data entry, quality assurance, data summaries, standard analyses, data serving, and report preparation occur under highly standardized protocols (Ickes and others, 2014; Ratcliff and others, 2014).

In this third UMRR Status and Trends report and after much partnership discussion after Johnson and Hagerty (2008), the fisheries indicators were reconceived to express and represent functional attributes and dynamics of the fish community. These new indicators represent functionally or socially relevant assemblages within the fish community, rather than simple faunistic ones, and were crafted to provide insights into changes in key functional interactions and attributes among assemblages, across LTRM study reaches, and over time. By recasting the indictors in this way, we sought to connect more directly observed fish community responses and drivers such as habitat, exploitation, and invasive species effects on the fish community. Key in crafting these functional indicators is the ability to group species into socially and functionally relevant assemblages to express these composite indicators in units of functional response. This was achieved using a life history database developed for the Central Basin of the United States by O'Hara and others (2007), which permitted the design-based estimation of composite functional guild classes in functional units of mass rather than counts as expressed in earlier reports. Table G1 identifies each fisheries indicator represented in this chapter and describes its attributes and indicative intention.

[1]U.S. Geological Survey.

[2]Iowa Department of Natural Resources.

[3]Wisconsin Department of Natural Resources.

[4]Illinois Natural History Survey.

Table G1. Names, attribute(s), and intent of each fisheries indicator developed to evaluate status and trends in the ecosystem functioning of the Upper Mississippi River System fish community.

Indicator name	Indicator attribute(s)	Indictor intent
Fish community	Systemic and study reach specific fish community catch, richness (species counts), and diversity (Shannon Index).	Investigate differences across space and changes over time in the richness and diversity of the fish community.
Lentic fishes	Mass per unit effort of an assemblage of low-flow (off-channel habitat) dependent fish species.	Investigate differences across space and changes over time in lentic habitats for fishes.
Lotic fishes	Mass per unit effort of an assemblage of high-flow (channel habitat), dependent fish species.	Investigate differences across space and changes over time in lotic habitats for fishes.
Native/nonnative fishes	Comparisons of indexed mass prominence of nonnative assemblages relative to native assemblages and changes in these relations over time.	Investigate the effect of nonnative fish species as extrinsic threats to the native portion of the entire fish community.
Forage fishes	Mass per unit effort of an assemblage of fishes that play a key intermediary trophic role in food webs.	Investigate differences across space and changes over time in forage fishes.
Recreationally valued native fishes	Mass per unit effort of an assemblage of fishes that are socially and economically relevant.	Investigate differences across space and changes over time in recreationally valued fishes.
Commercially valued fishes	Mass per unit effort of an assemblage of fishes that support commercial exploitation.	Investigate differences across space and changes over time in commercially valued fishes.

Indicator: Fish Community

Summary of Fish Community

Indicator intent: Investigate differences across space and changes over time in the richness and diversity of the fish community.

Measurement: Annual species counts, total annual catch, and diversity within each of six long-term study reaches (see fig. A1; Ickes and others, 2014 and Ratcliff and others, 2014).

Special considerations: Diversity measured using the Shannon index.

Assessment

Status: Species richness was high in all locales, but highest in the lower study reaches. Diversity was high in all locales, but highest in Pools 8 and 13.

Trends: Species richness is high and stable (no trends) in all locales. No trends were observed in total annual catch, although catch varies interannually.

The primary mission of the LTRM fish component is to observe fish community responses over time and differences across space, and to identify drivers of differences and changes. To this end, the LTRM Fisheries Component uses multiple sampling gears to assess the full fish community under a scientifically sound and standardized sampling design. Details concerning sampling strategies, design, methods, strengths, and limitations have been well documented (Ickes and others, 2014; Ratcliff and others, 2014). The primary metrics used in this assessment are species richness (total and mean annual species observed), community diversity (Shannon, 1948), and total catch statistics.

Within each study reach, a mean of 62–74 species have been observed annually (fig. G1), and cumulative species richness since program inception ranges from 83–89 species across the upper three study reaches, while richness increases in the lower three study reaches ranging between 86 and 98 species per study reach (fig. G1). The Shannon diversity index is a composite metric of species richness and evenness in abundance among species per study area. While richness is lower in the upper reaches, Shannon diversity is greatest in Pools 8 and 13; intermediate in Pools 4, 26, and the Open River Reach of the Upper Mississippi River; and lowest in the La Grange Pool of the Illinois River (fig. G2). Thus, while the lower study reaches possess innately richer faunas, the fish communities are dominated by fewer species than the northern reaches.

Total catches have varied over time (fig. G2); however, some of this variance is due to sampling constraints (early-mid 2000s). No notable or strong trends in total catch were observed. Annual species richness has shown long-term stability (no observable trends) in all six study reaches (fig. G2).

The fish community has remained faunistically intact over the period of observation by LTRM yet demonstrates important regional differences in species richness and community diversity, attributable to individual species' innate biogeography as well as important differences in regional aquatic habitat conditions (for example, Koel, 2004). Although

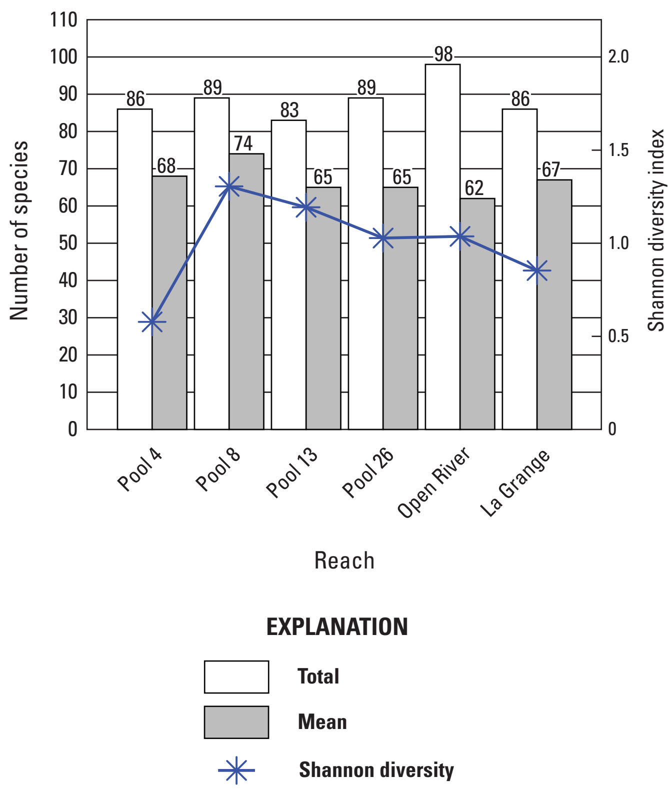

Figure G1. Total and mean annual species richness and Shannon diversity index observed over the period of record (1993–2019) for the Upper Mississippi River Restoration Long Term Resource Monitoring element in the Upper Mississippi River System. See figure A1 for the location of each pool and study reach.

such regional differences in the fish communities and assemblages are now well-understood (Barko and others, 2004; Barko and others, 2005; Chick and others, 2005; Ickes and others, 2005; Chick and others, 2006; Schramm and Ickes, 2016), emerging evidence indicates that functional attributes of the fish community are undergoing notable and active shifts (Bouska, 2020). Ongoing research is examining the nature of these functional shifts and what may be driving the shifts to better inform applied management practitioners across the basin.

The fish community continues to face extrinsic and intrinsic threats. Invasive bigheaded carps (extrinsic threat) have become dominant species in the local fish communities of the lower Upper Mississippi River System and have been demonstrated to affect the growth and health of native planktivorous fish species and affect sportfish species in these locales (ch. I). Additionally, certain elements of the fish community also remain under notable intrinsic threats associated with wild caviar harvest (ch. G; Garvey and others 2003), habitat degradation associated with commercial navigation (Johnson and Hagerty, 2008), flood control, and basin runoff from agricultural and industrial sources (Johnson and Hagerty, 2008).

Indicator: Lentic Fishes

Lentic, or lacustrine, fishes are adapted to low-flow environments and include most of the nest-spawning fish species (O'Hara and others, 2007). Although lentic fish may spend portions of their lives in flowing-water habitats to take advantage of feeding opportunities, this assemblage of species requires backwaters for spawning, nursery, or overwintering habitat, and their abundance is thus linked with the amount and quality of such habitat. Several lentic fishes, such as *Micropterus salmoides* (largemouth bass), *Lepomis macrochirus* (bluegill), *Esox lucius* (northern pike), and *Perca flavescens* (yellow perch), are considered excellent sport fish by anglers.

Four criteria were used to select species for the lentic fishes indicator: (1) native species, (2) not defined as a large river species (O'Hara and others, 2007), (3) prefer little or no current flow (O'Hara and others, 2007), and (4) listed in Schramm and Ickes (2016) as "backwater-dependent." Of the 34 candidate lentic species that met all 4 criteria, 14 species composed 98-percent of the lentic fish indexed mass in the LTRM fish database and were selected for this indicator (listed in table G2). Status and trends for lentic fishes are reported and interpreted using the annual mean MPUE (grams per 15 minutes [g/15 min]) for each LTRM study reach using daytime electrofishing (Ratcliff and others, 2014).

The upper three LTRM reaches contain more backwater habitat, and are of better quality, than the two lower Upper Mississippi River reaches (DeJager and others, 2018). This is reflected in the lentic fish MPUE, which is generally an order

Summary of Lentic Fishes

Indicator intent: Investigate differences across space and changes over time in lentic habitats for fishes.

Measurement: Annual design-based index of mass per unit effort (grams per 15 minutes of daytime electrofishing) for each of six long-term study areas (see fig. A1), (Ickes and others, 2014 and Ratcliff and others, 2014). Annual trends in mass per unit effort were assessed using simple linear regression analysis for each study reach.

Special considerations: Replaces the bluegill indicator from previous reports (U.S. Geological Survey, 1999; Johnson and Hagerty, 2008).

Assessment

Status: Lentic fishes are abundant in the three upper study reaches, moderately abundant on the La Grange Pool of the Illinois River, and have low abundance in Pool 26 and the Open River Reach.

Trends: Lentic fishes are increasing in Pools 4, 8, and the Open River Reach; stable in Pools 13 and 26; and decreasing in the La Grange Pool.

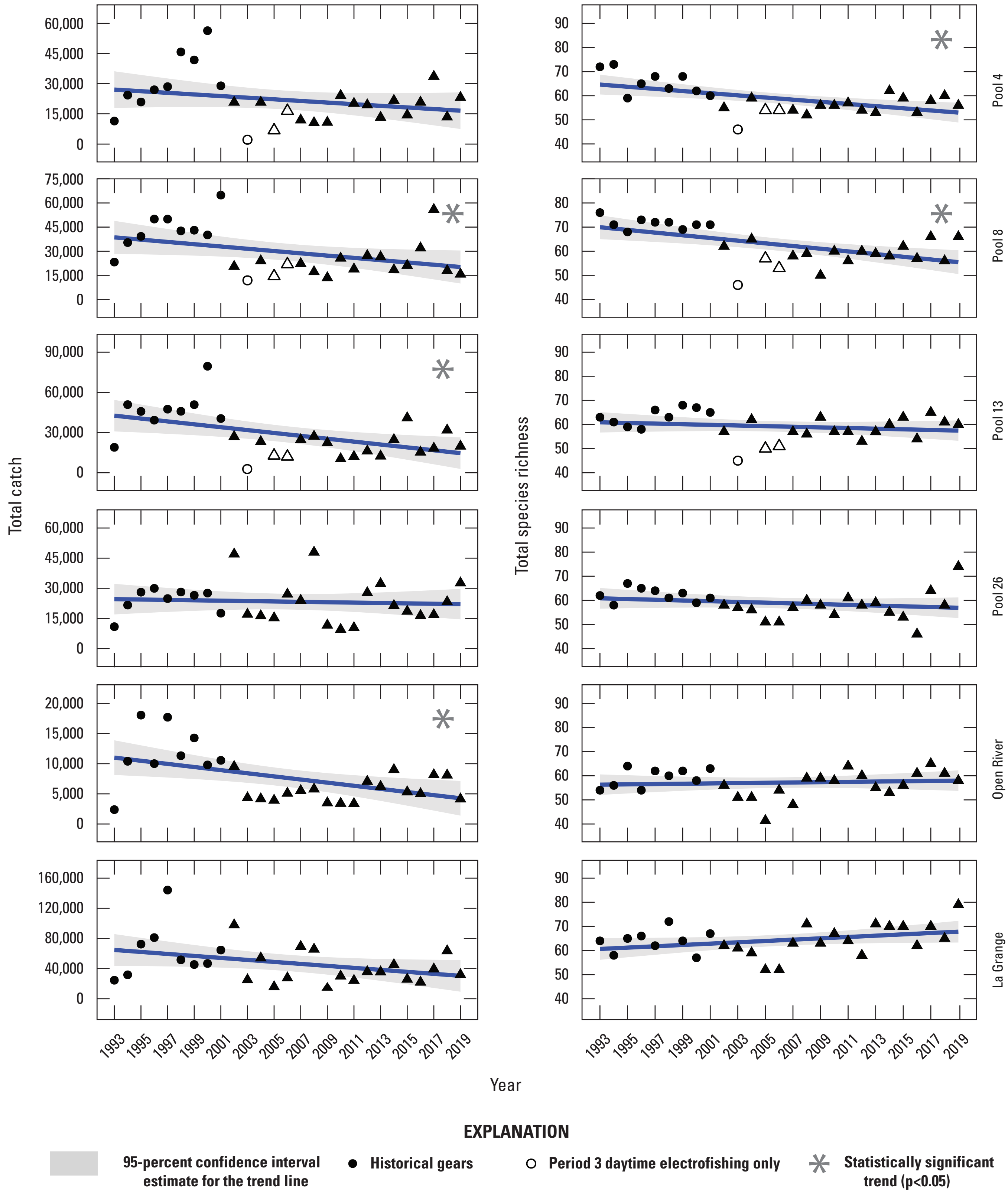

Figure G2 Annual fish total catch (left column) and annual species richness (right column) observed over the period of record (1993–2019) for the Upper Mississippi River Restoration Long Term Resource Monitoring element in each of six long-term study reaches of the Upper Mississippi River System. See figure A1 for the location of each study reach. <, less than.

Table G2. List of fish species selected for the lentic fishes indicator and their percentage contribution to total mass of lentic fishes sampled from all study reaches of the Upper Mississippi River Restoration program's Long Term Resource Monitoring element by daytime electrofishing from 1993 to 2019.

Common name	Scientific name	Total mass (percent)
Largemouth bass	*Micropterus salmoides*	40.17
Bluegill	*Lepomis macrochirus*	14.44
Bowfin	*Amia calva*	14.22
Weed shiner	*Notropis texanus*	7.04
Northern pike	*Esox lucius*	5.82
Black crappie	*Pomoxis nigromaculatus*	5.14
Longnose gar	*Lepisosteus osseus*	4.29
White crappie	*Pomoxis annularis*	3.07
Yellow perch	*Perca flavescens*	2.63
Golden shiner	*Notemigonus crysoleucas*	0.73
Spotted gar	*Lepisosteus oculatus*	0.61
Green sunfish	*Lepomis cyanellus*	0.43
Pumpkinseed	*Lepomis gibbosus*	0.38
Orangespotted sunfish	*Lepomis humilis*	0.19

of magnitude higher in Pools 4, 8, and 13 than in Pool 26 and the Open River Reach (fig. G3). When the LTRM began stratified random fish sampling in 1993, the upper reaches had just experienced a nearly complete loss of aquatic vegetation (Rogers and others, 1994). The large, gradual increase in lentic fish MPUE over time for Pools 4 and 8 (fig. G3) reflects the recovery of aquatic vegetation in these study reaches over the past 25 years (ch. F). Conversely, the La Grange Pool of the Illinois River had similar lentic fish MPUE to the northern reaches in the 1990s but has shown a downward trend since about 2000, likely caused by multiple stressors, including the establishment of bigheaded carps and water level fluctuations. The meager number of backwaters and lack of aquatic vegetation in the La Grange Pool probably decreases the resilience of this study reach to these stressors.

The close relation between lentic fishes and aquatic vegetation is also an important consideration (Burdis and others, 2020). Eutrophic conditions and turbid waters can favor an algal-dominated system, whereas moderate nutrient inputs and clearer waters favor the growth of rooted aquatic plants (ch. H). Increased frequency, duration, and magnitude of flooding events (ch. B) may cause reductions in aquatic plants, which would be detrimental to lentic fishes. However, water level management practices that mimic natural low-water conditions during the growing season, when appropriate, may promote seasonal aquatic vegetation. This seasonal vegetation could be useful in providing cover and food for lentic fishes, provided it persists long enough and reduces flow enough for lentic fishes to adapt to its presence.

Impounded systems tend to trap sediment and fill in over time, resulting in more wetlands and fewer open water habitats. However, increased intensity of precipitation and frequency of high-flow events may cause scouring and reshaping of off-channel areas (ch. C). These opposing forces may affect the amount and quality of backwater areas available to lentic fishes of the upper study reaches in the future. Water level fluctuations, channelization, and levees in the lower study reaches are likely to continue to constrain backwater habitats, and thus, lentic fish populations, from reaching their potential in those reaches. Habitat restoration projects that promote aquatic vegetation growth may also promote lentic fishes' habitat (Johnson and Hagerty, 2008).

Indicator: Lotic Fishes

The Upper Mississippi River System possesses lotic (flowing water) and lentic (still water) habitats. Fishes possess different life history traits that enable them to exploit either or both major aquatic habitat types (O'Hara and others, 2007).

Lotic habitats have been vastly altered historically (Remo, 2016) and remain subject to a variety of disturbances today. A few important examples include engineered flow diversion and control structures to maintain navigation channels, snag removal, river impoundment to promote commercial shipping, main channel dredging, commercial navigation, and bank stabilization. Species that depend upon lotic environments to fulfill at least one key life history trait (for example, reproduction, feeding, migration; "lotic dependent fishes") and species that specialize in fully exploiting lotic habitats ("lotic specialist fishes") were selected to gauge the status of lotic habitats across the six LTRM study reaches (table G3), from 1993 to 2019.

Ten species were identified as lotic dependent fishes, eight of which are migratory, while an additional 17 species were identified as lotic specialist fishes (see O'Hara and others, 2007). Annual mean MPUE was calculated for all lotic

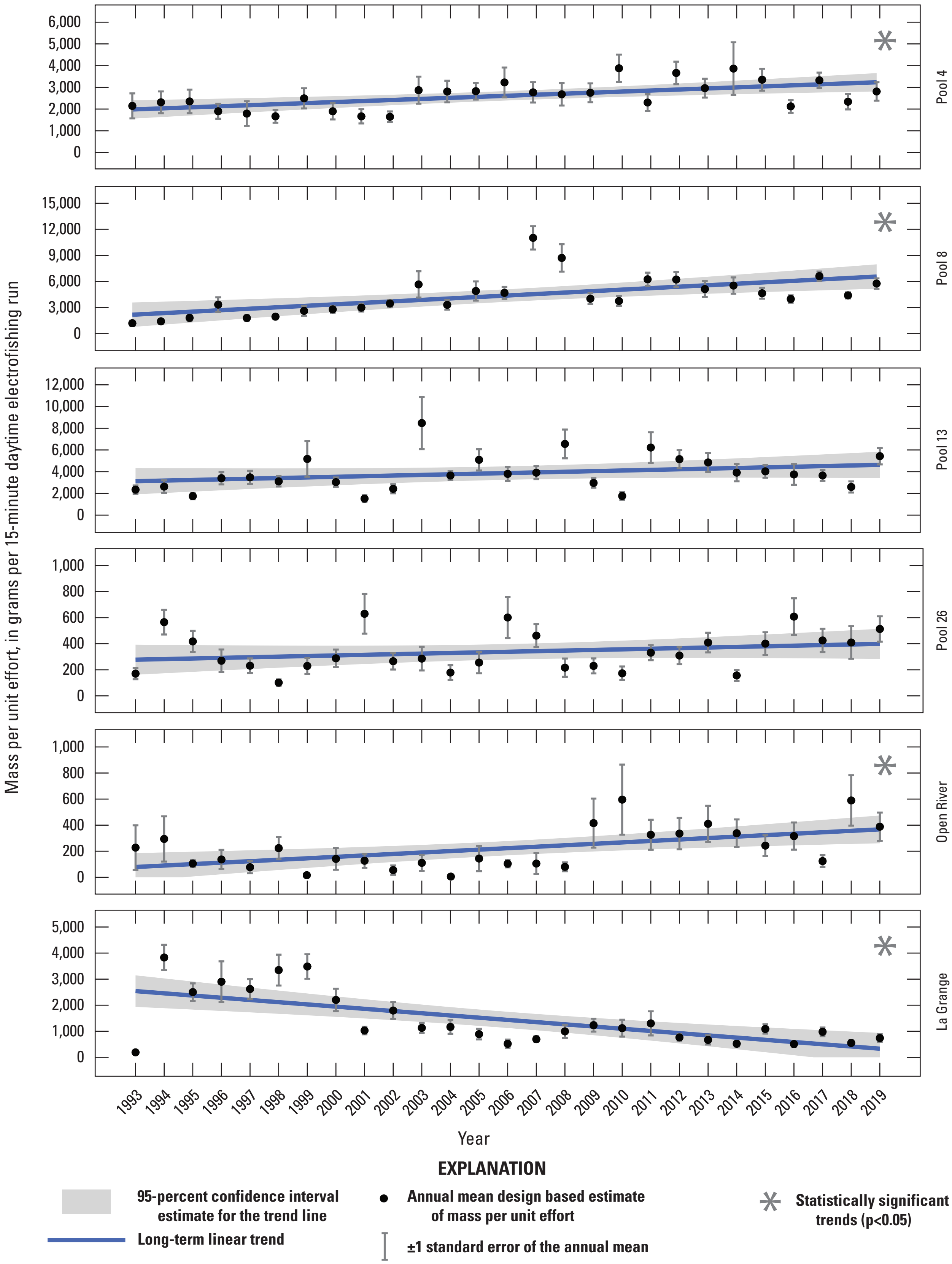

Figure G3. Mass per unit effort of lentic fish, captured by daytime electrofishing, in the Upper Mississippi River Restoration Long Term Resource Monitoring element study reaches from 1993 to 2019. Please note differences in scaling of the y-axis among study reaches. See figure A1 for the location of each study reach. ±, plus or minus; <, less than.

Summary of Lotic Fishes

Indicator intent: Investigate differences across space and changes over time in lotic habitats for fishes.

Measurement: Annual design-based index of mass per unit effort (grams per 15 minutes of daytime electrofishing) for each of six long-term study reaches (see fig. A1), (Ickes and others, 2014 and Ratcliff and others, 2014). Annual trends in mass per unit effort were assessed using simple linear regression analysis for each study reach.

Special considerations: Lotic class designations are as described in O'Hara and others (2007) and table G3.

Assessment

Status: Lotic dependents were abundant and dominant relative to lotic specialists, which displayed rarity, in all five impounded Long Term Resource Monitoring element study reaches. The converse applied to the unimpounded Long Term Resource Monitoring element study reach.

Trends: No statistically significant trends were observed; however, substantial interannual variability was observed for lotic dependents in all five impounded reaches and for lotic specialists in the Unimpounded Reach.

Table G3. List of species contributing to lotic dependent or lotic specialist classes. Only species in these classes observed by the Long Term Resource Monitoring element from 1993 to 2019 are included. Additional species are known to be present but have not been observed by the Long Term Resource Monitoring element, including the federally endangered *Scaphirhynchus albus* (pallid sturgeon).

Class	Common name	Scientific name	Life history requirements
Lotic dependents	Silver lamprey	*Ichthyomyzon unicuspis*	Species that depend upon lotic environments to fulfill one or more key life history requirements (for example, feeding, reproduction, and migration)
	Lake sturgeon	*Acipenser fulvescens*	
	Goldeye	*Hiodon alosoides*	
	Mooneye	*Hiodon tergisus*	
	Skipjack herring	*Alosa chrysochloris*	
	Flathead catfish	*Pylodictis olivaris*	
	White bass	*Morone chrysops*	
	Yellow bass	*Morone mississippiensis*	
	Chestnut lamprey	*Ichthyomyzon castaneus*	
	Plains minnow	*Hybognathus placitus*	
Lotic specialists	Shovelnose sturgeon	*Scaphirhynchus platorynchus*	Species that depend upon lotic environments to fulfill all key life history strategies (for example, feeding, reproduction, and migration)
	Blue sucker	*Cycleptus elongatus*	
	Blue catfish	*Ictalurus furcatus*	
	Western silvery minnow	*Hybognathus argyritis*	
	Mississippi silvery minnow	*Hybognathus nuchalis*	
	Speckled chub	*Macrhybopsis aestivalis*	
	Sturgeon chub	*Macrhybopsis gelida*	
	Sicklefin chub	*Macrhybopsis meeki*	
	Pallid shiner	*Notropis amnis*	
	River shiner	*Notropis blennius*	
	Silverband shiner	*Notropis shumardi*	
	Mimic shiner	*Notropis volucellus*	
	Channel shiner	*Notropis wickliffi*	
	Western sand darter	*Ammocrypta clara*	
	Crystal darter	*Ammocrypta asprella*	
	Mud darter	*Etheostoma asprigene*	
	River darter	*Percina shumardi*	

dependent fishes combined, as well as all lotic specialist fishes combined, for LTRM's main channel border sampling stratum (fig. G4) as observed by day electrofishing to evaluate status and trends in lotic habitats (1993–2019).

Between lotic dependent fishes and lotic specialist fishes, lotic dependent fishes were observed to be more dominant in lotic habitats in all five of the impounded reaches (Upper Mississippi River Pools 4, 8, 13, 26, and the La Grange Pool of the Illinois River) and lotic specialist fishes were observed to be rare within each of these five reaches (fig. G4). In the unimpounded Open River Reach, lotic specialist fishes were observed to be more dominant than lotic dependent fishes in lotic habitats, although lotic specialist fishes exhibited low MPUEs (fig. G4).

These patterns in status are indicative of the idea that river impoundment for navigation, which largely began in the 1930s and was completed by the early 1940s (fig. A4), has strongly and historically selected against lotic specialist fishes in impounded river reaches. However, lotic habitats in these impounded reaches appear to meet most intermittent life history requirements for lotic dependent fishes to maintain a faunistic foothold.

In the Open River Reach, lotic dependent fishes are the rarer class indicating critical life history requirements are not being met in the unimpounded reach (see table G3; O'Hara and others, 2007). The high degree of interannual variability in observed lotic specialist mass in the Unimpounded Reach relative to impounded reaches (fig. G4) indicates river modifications that have (a) substantially constrained the main channel (Remo, 2016), (b) promoted side channel cut-offs (Barko and Herzog, 2003), (c) increased flow velocities (ch. B), and (d) intensified high water flooding events (ch. B) that may be affecting the potential production of lotic specialists. Moreover, the only federally listed endangered fish species in the Upper Mississippi River System, *Scaphirhynchus albus* (pallid sturgeon), is a lotic specialist, further supporting substantial lotic habitat impairments in the Unimpounded Reach.

Although differences in status are observable, no significant trends were observed for any study reach for either lotic dependent or lotic specialist fishes. This indicates long-term stability in annual MPUE, yet with considerable interannual variability over the period of record.

Commercial navigation is expected to continue for the foreseeable future. Continued communication, engagement, and the crafting of novel management practices among river management agencies, programs, and industry would be beneficial for improving lotic habitats.

Indicator: Native/Nonnative Fishes

During the past 100 years, nonnative species have invaded ecosystems globally at increasing rates, frequently with profound effects upon native assemblages and species. Before the early 2000s, the Upper Mississippi River System proved itself largely resistant to, and resilient in the face of, nonnative fish species invasions. While several nonnative species established themselves before the 2000s (Irons and others, 2009), nearly all have had minimal to no effects upon native fish communities, and successful establishments have been mostly limited to the Illinois River rather than the Upper Mississippi River. The main exception to this was the successful establishment of *Cyprinus carpio* (common carp) in the mid-1800s. Common carp became widespread throughout the Upper Mississippi River System, and indeed, North America, by the beginning of the 20th century. While profound shifts in the fish community followed the establishment of common carp (Carlander, 1954), common carp have never been attributed with faunistic losses. Since 1993 when LTRM began, common carp has been the mass dominant fish species over the period of record across all study reaches, while also experiencing steady decreases over the period of record (Gibson-Reinemer and others, 2017).

In the early-mid 2000s, two new species of bigheaded carps established populations in the lower reaches of the Upper Mississippi River System. *Hypophthalmichthys molitrix* (silver carp) and *Hypophthalmichthys nobilis* (bighead carp) were first observed by LTRM in 1998 and 1993, respectively. Chapter I provides details on the sources of the invasion and some of the effects that have been observed in some native species. Nonnative species invasions represent an extrinsic threat to the native fish fauna and present unknown future effects. Moreover, the arrival of nonnative species is usually beyond the control of local and regional management jurisdictions, being driven primarily by global trade and many vectors of introduction and spread.

Silver and bighead carp are filter feeders, consuming a diet primarily of algae and zooplankton that form the base of the food web. Consequently, they compete directly with native filter feeding fishes (for example, paddlefish, bigmouth buffalo), and recent studies have observed decreases in abundance, growth, and body condition of filter feeders (ch. I). In addition, all fish species depend upon zooplankton during early life stages, placing all fish species in direct competition with bigheaded carps during critical life stages. Bigheaded carps are highly fecund, able to produce 100,000s of eggs per female, and fast growing, meaning they can escape mortality from predator species rather quickly in their first year of life.

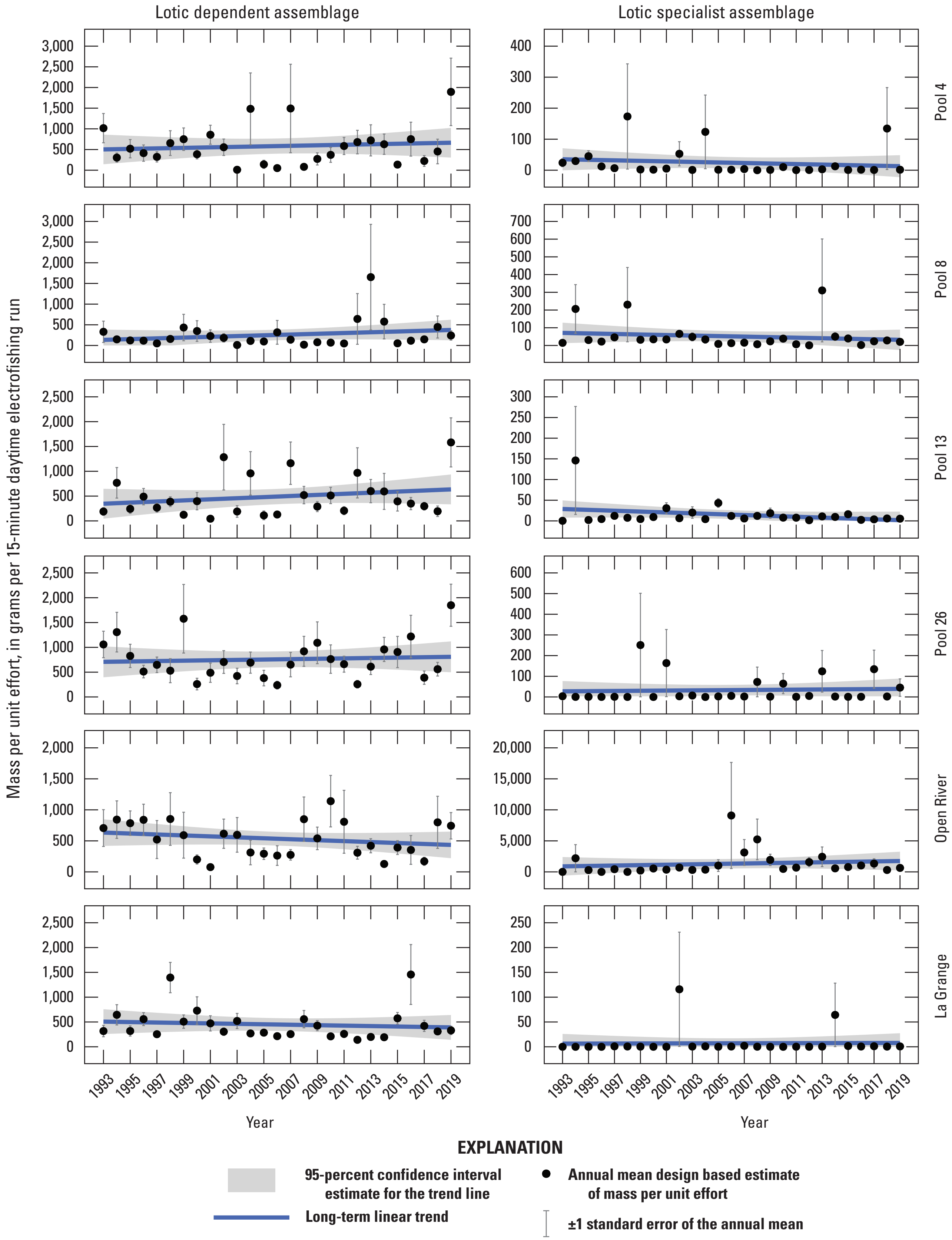

Figure G4. Mass per unit effort of lotic dependent (left panels) and lotic specialist (right panels) fishes, observed by daytime electrofishing, in the Upper Mississippi River Restoration Long Term Resource Monitoring element study reaches from 1993 to 2019. Please note differences in scaling of the y-axis among study reaches. See figure A1 for the location of each study reach. ±, plus or minus.

Summary of Native/Nonnative Fishes

Indicator intent: Investigate the effect of nonnative fish species as extrinsic threats to the native portion of the entire fish community.

Measurement: Annual design-based index of mass per unit effort (grams per 15 minutes of daytime electrofishing) of all nonnative species (see special consideration below) for each of six long-term study reaches (fig. A1; Ickes and others 2014 and Ratcliff and others 2014). Annual trends were assessed using simple linear regression analysis for each study reach.

Special considerations: *Cyprinus carpio* (common carp), a naturalized, nonnative, and mass dominant fish species in all study reaches, was excluded from all calculations and presentations because they have been established since the late 1800s, and their effect on fish communities exist in the unobservable past. Our index was tailored to reflect the effect of nonnative species over the period of observation (1993–present).

Assessment

Status: Nonnative fish species (aside from common carp) are not established in the upper three study reaches but are established and, in some cases, abundant in the lower three study reaches, especially *Hypophthalmichthys* spp. (bigheaded carps). Total native fish community mass suppression is evident in La Grange Pool of the Illinois River.

Trends: Nonnative mass within the fish communities of the three lower study areas increased, especially in Pool 26 and La Grange Pool. As of 2010, *Hypophthalmichthys molitrix* (silver carp) became the mass dominant species in the La Grange Pool of the Illinois River. Nonnative species were not observed in the upper three study reaches.

As of 2019, bigheaded carps have established self-sustaining populations in the three lower reaches of the Upper Mississippi River System but have yet to establish self-sustaining populations in the three upper Upper Mississippi River System study reaches (fig. G5). In the lower study reaches, in 2019, bigheaded carps account for half, excluding common carp, of the observed fish community mass in Pool 26 of the Mississippi River and the La Grange Pool of the Illinois River and about 15-percent of the fish community mass in the Open River Reach of the Mississippi River (fig. G5). In the upper study reaches (Pools 4–13), no observations of adult or juvenile bigheaded carps have been recorded by LTRM, and only occasionally have adult bigheaded carps been observed by other means (for example, commercial fisheries), which indicates bigheaded carps have not established populations in the upper study reaches.

The three lower study reaches offer three rather distinct responses to the establishment and increasing mass dominance of bigheaded carps. Within the Open River Reach of the Mississippi River, populations are established, but maintain a lower level of mass representation within the full fish community. In contrast, mass representation of bigheaded carps is increasing notably in Pool 26 of the Mississippi River and the La Grange Pool of the Illinois River, each approximating one-half of the indexed fish mass observed in each study reach in 2019, exclusive of common carp. In Pool 26, total fish community indexed mass has significantly increased with a small yet significant decrease in native fish mass (fig. G5). This indicates bigheaded carps are presently increasing the productive capacity of the fish community in Pool 26, but that this increase in total productivity is attributable to significant increases in the production of bigheaded carps with small yet significant decreases in the total mass of the native fish community. In comparison, recent increases in mass dominance of bigheaded carps in La Grange Pool have been associated with a strong decrease in native fish mass, indicating native fish displacement and suppression by bigheaded carps. It is also important to note that a significant increase in overall productivity was not observed in the La Grange Pool, as was observed in Pool 26 of the Mississippi River.

LTRM data demonstrate bigheaded carps represent an extrinsic threat to native fish communities in the lower reaches. The large decreases in native fish mass observed in the La Grange Pool indicate a detrimental effect that bigheaded carps can have on the ecosystem. That effect can potentially manifest in other areas if bigheaded carps continue to expand their distribution and increase where already established. These factors indicate the importance of continued vigilance regarding observed trends and additional investigation to elucidate the mechanisms leading to native fish community decreases. To date, no native extirpations have been associated with bigheaded carps; however, trends in the dominance of bigheaded carps are presently significantly upward and represent a potential native faunistic suppression and loss threat in some reaches.

Future expansion into more northern reaches remains a concern. However, even after several recent years of prolonged high discharge and river stages (ch. B), as of 2019, LTRM has not observed either adult or juvenile bigheaded carps in these study reaches, although a few adult specimens have been observed by commercial fisheries. Advancing studies that seek to determine why these upper study reaches seem to be resistant to the expansion and establishment of

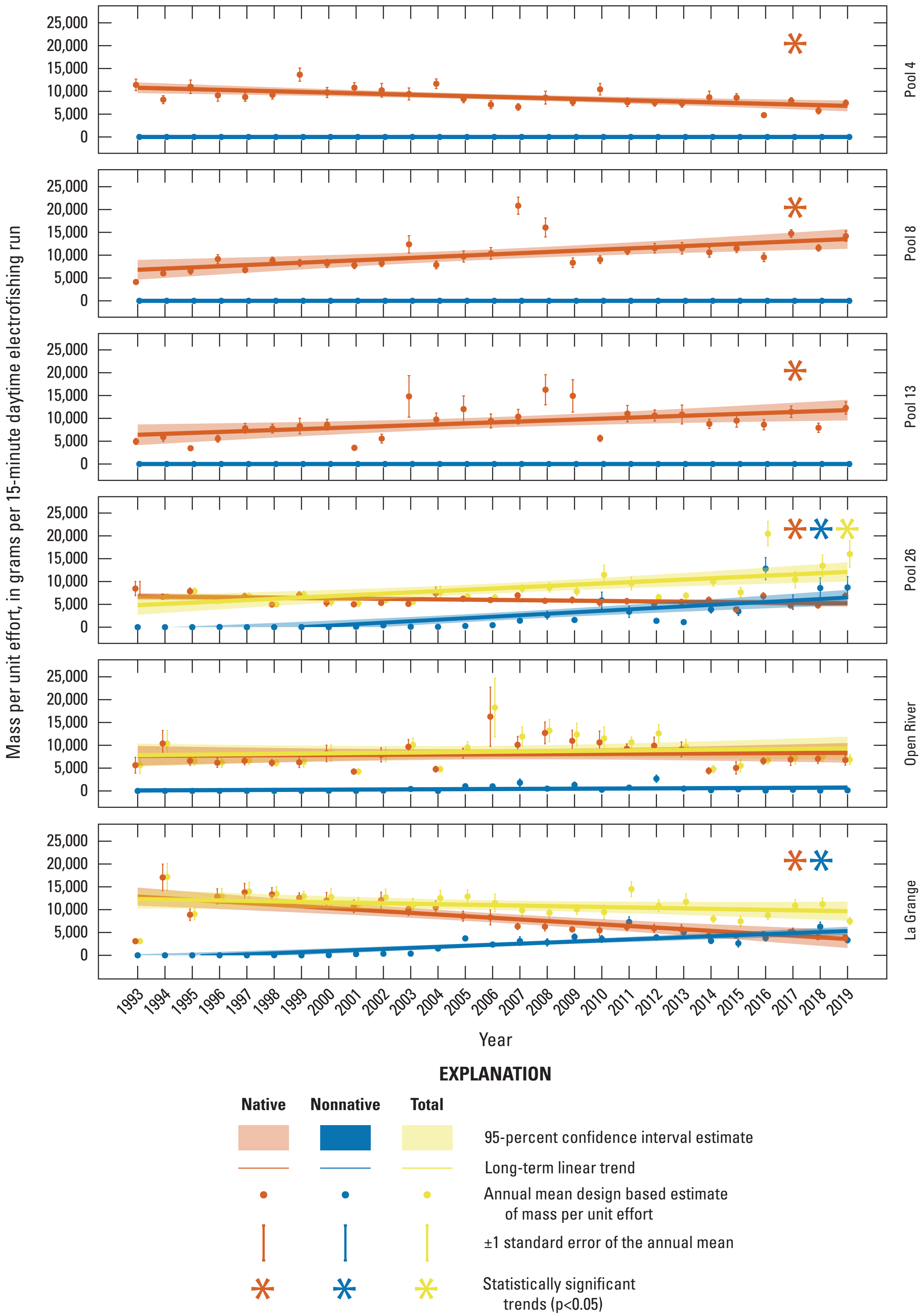

Figure G5. Mass per unit effort for the Upper Mississippi River System native fish assemblage, nonnative bigheaded carps, and total fish community in the Upper Mississippi River Restoration Long Term Resource Monitoring element study reaches from 1993 to 2019. Annual estimators exclude *Cyprinus carpio* (common carp), a naturalized and mass dominant nonnative fish species. Please note the log10 scale on the y-axis. Because bigheaded carps have not been observed in the upper three study reaches, total fish community mass per unit effort equals native fish assemblage mass per unit effort. See figure A1 for the location of each study reach. ±, plus or minus; <, less than.

bigheaded carps would prove to be beneficial for several reasons. Such research would help predict range expansions in other rivers, impounded areas, and lake systems and may also provide insights into river management options in reaches where bigheaded carps are established and increasing in fish community dominance.

Indicator: Forage Fishes

Forage fish are an important component of the fish community, occupying an intermediate role in the food web—assimilating plankton, invertebrates, and plants—while serving as food for larger predaceous fish and other animals. Given their intermediate role in the ecosystem, forage fish can greatly affect trophic levels given their large role in processing nutrients and transferring energy through the ecosystem. Nearly all species of fishes are forage fishes for at least a brief period of their lives. Forage fishes also include the minnow and darter families, which harbor a great amount of diversity that adds to the resilience of the community.

Dorosoma cepedianum (gizzard shad) and *Notropis atherinoides* (emerald shiner) were selected for the forage fish indicator because they are universally abundant on the Upper Mississippi River and are prey throughout their life cycles.

Summary of Forage Fishes

Indicator intent: Investigate differences across space and changes over time in forage fishes.

Measurement: Includes all sizes of *Dorosoma cepedianum* (gizzard shad) and *Dorosoma petenense* (threadfin shad), *Notropis atherinoides* (emerald shiners), and all fish less than 80 millimeters in total length. Annual design-based index of mass per unit effort (MPUE; grams per 15 minutes of daytime electrofishing) for six long-term study reaches (fig. A1; Ickes and others, 2014 and Ratcliff and others, 2014). Annual trends in MPUE were assessed using simple linear regression analysis for each study reach.

Special considerations: None

Assessment

Status: Forage fishes reach their greatest MPUE in the lower Upper Mississippi River study reaches and the La Grange Pool of the Illinois River. Forage fishes are moderately abundant in Pools 4, 8, and 13.

Trends: Significant downward trends in forage fish MPUE were evident in four of the six Long Term Resource Monitoring element study reaches: Pool 4, Pool 26, and the Open River Reach on the Upper Mississippi River and the La Grange Pool on the Illinois River. Forage fish MPUE in Pools 8 and 13 is stable.

Dorosoma petenense (threadfin shad) are more common in the lower Upper Mississippi River reaches and function as an ecological surrogate for gizzard shad. Fishes less than 80 millimeters in total length represent most of the remaining minnow and darter species, as well as newly hatched fish of most species, which are often vulnerable to predators.

The greatest mean annual daytime electrofishing MPUE for forage fish in LTRM samples was in Pool 26 (1,060 g/15 min) and the La Grange Pool of the Illinois River (873 g/15 min), followed by the Upper Mississippi River Open River Reach (674 g/15 min), Pool 13 (454 g/15 min), Pool 8 (433 g/15 min), and Pool 4 (285 g/15 min). These results, spatially, represent a longitudinal gradient in forage production (fig. G6). The difference between the largest and smallest mean annual MPUE was roughly fourfold between Pool 4 and Pool 26 (fig. G6).

Dorosoma spp. (shad) dominated MPUE, constituting at least 40-percent of the forage MPUE in all years for five of the six study areas. In the lower study reaches, shad was always at least 70–99-percent of the MPUE; whereas, in the upper study reaches, emerald shiners and other small fishes generally composed more of the MPUE. Among the other fish groups in the forage fish indicator, emerald shiner MPUE decreased in Pool 4 and increased in the La Grange Pool of the Illinois River, while other small fishes increased in Pool 26 and decreased in the Open River Reach.

The forage fish indicator trended significantly downward in four of the six LTRM reaches, reflecting a decrease in shad MPUE in those study reaches (fig. G6). Potential causes for this decrease in MPUE include decreasing catch rates (number of fish as CPUE), reduced fitness (mass per unit length), or a shift toward smaller sizes (length). If CPUE was stable, but MPUE decreased, the result was due to decreases in mean size. The LTRM daytime electrofishing CPUE of shad was stable in all LTRM reaches except the Open River Reach but decreased in the La Grange Pool for other gear types (in other words, fyke and mini fyke nets) (https://umesc.usgs.gov/data_library/fisheries/graphical/fish_front.html). Frequency of occurrence (percentage of species collections) for shad also decreased for large hoop nets and fyke nets in Pool 26 and the La Grange Pool and for fyke nets and daytime electrofishing in the Open River Reach. Thus, decreasing mean size, catch rate, and frequency of occurrence of shad all contributed to the observed decrease in the forage fish indicator in the lower three reaches. Irons and others (2007) demonstrated significant reductions in gizzard shad fitness commensurate with increases in the production of invasive bigheaded carps, likely associated with competition for food resources. The resulting decrease in mass per unit length of shad also may have contributed to MPUE decreases.

The decrease in forage fishes in Pool 4 was driven by three years of high shad abundance from 1999 to 2001 (fig. G6). Before and after those three years, Pool 4 forage fish MPUE has been stable, indicating the absence of a trend in recent years. Shad in the upper study reaches of the Upper Mississippi River are susceptible to winter mortality due to poor tolerance of prolonged cold water conditions. Thus, populations in these reaches tended to be low, with occasional years of high production.

Habitat limitations are possible, as the upper three LTRM study reaches are considered to have better and more varied habitats than the three lower LTRM study reaches (De Jager and others, 2018) and are the only LTRM reaches with stable forage fish MPUEs. However, severe winter conditions limit shad abundance in these reaches. The quality and complexity of habitats in the upper study reaches allows a wider variety of forage fish to flourish, and this is reflected in a diverse predator population, as well. If habitat conditions were improved in the lower reaches, it is possible that a more diverse forage base could develop that might compensate for decreases in shad. The proliferation of bigheaded carps throughout the lower reaches of the Upper Mississippi River has likely affected the forage fish indicator through reduced fitness of shad and subsequent decreases in abundance, and this stressor to shad will likely persist and possibly spread if invasive carps remain abundant and become more widely established in the Upper Mississippi River. It remains to be seen whether the higher diversity of habitat of the upper study reaches will prevent bigheaded carps from establishing or reduce the effects of such establishment on the fitness of other members of the fish community that have been observed in the La Grange Pool.

Long-term changes in hydrology (ch. B), in any season of the year, may have substantial effects upon forage fish that generally depend on ideal conditions to yield large year classes and that often occupy specialized ecological niches. Changes to seasonal flows could disrupt spawning, displace fish from refugia during the cold season, limit food availability, or increase susceptibility to predation, among a host of other possible effects.

Indicator: Recreationally Valued Native Fishes

The production of recreationally valued fishes is an important ecosystem service provided by the Upper Mississippi River System. Recreational fishers gain and express perceptions on the health of the river based upon their observations of the diversity and abundance of this socially relevant class of fishes. While little recent information is available on the economic value of recreational fishing activities, in 1979 the recreational fishery was valued at $1–1.2 billion and supported over 18,000 jobs (Schramm and Ickes, 2016). The recreationally valued native species indicator is the combined pool-wide MPUE for 19 native fish species pursued by anglers and considered sportfish by at least 1 State. Only fish of stock size or greater (O’Hara and others, 2007) were used in this analysis (table G4).

Pool 13 had the greatest long-term mean MPUE (4,697 g/15 min) of recreational fishes over the study period, followed by Pool 8 (4,447 g/15 min), Pool 4 (3,231 g/15 min), the La Grange Pool of the Illinois River (2,568 g/15 min), Pool 26 (2,296 g/15 min), and the Open River Reach (2,264 g/15 min). MPUE trends were significantly upward over the study period in Pool 8 and Pool 13, while MPUE significantly decreased in Pool 26 and the La Grange Pool of the Illinois River (fig. G7). Trends were stable in Pool 4 and the Open River Reach (fig. G7).

The contribution of each species to total MPUE remained stable in the upper three study reaches, which were dominated by centrarchid sunfish species (period of record mean centrarchid MPUE as percentage of total recreational fishes MPUE: Pool 4=46 percent, Pool 8=63 percent, Pool 13=63 percent). Although the species remained dominant, the contribution of centrarchid sunfishes to total MPUE decreased in Pool 13 (slope = −0.01, p<0.001, R^2=0.38, F=15.57). Catfishes (*Ictalurus furcatus* [blue catfish], *Ictalurus punctatus* [channel catfish], and *Pylodictis olivaris* [flathead catfish]) dominated the indexed mass of recreationally valued native fishes in the lower three study reaches (period of record mean: Pool 26=52-percent, Open River Reach =65-percent, La Grange Pool =38-percent), with significant decreases over time in the contribution of sunfishes to total MPUE on the La Grange Pool (slope = −0.01, p<0.001, R^2=0.43, F=18.53).

Regional aquatic habitat differences seem to be associated with the observed status and trends in recreationally valued native fishes. For example, aquatic vegetation remained scarce or nonexistent in the lower three study reaches since its decrease in the early 1990s (ch. F). Indeed, a strong longitudinal contrast existed between the relatively clear water and abundant vegetation of the upper three study reaches and the turbid water and scarce vegetation of the lower three study

Summary of Recreationally Valued Native Fishes

Indicator intent: Investigate differences across space and changes over time in recreationally valued fishes.

Measurement: This indicator is the combined pool-wide mass per unit effort from 19 fish species and includes fish common in backwater and channel habitats. Only fish of stock size or greater were used in this analysis (table G4). Annual design-based indices of mass per unit effort (grams per 15 minutes of daytime electrofishing) were generated for six long-term study areas from 1993 to 2019 (fig. A1; Ickes and others, 2014 and Ratcliff and others, 2014). Long-term trends in mass per unit effort were assessed using simple linear regression analysis for each study reach.

Special considerations: None.

Assessment

Status: All reaches support recreationally valued fishes and recreational fisheries.

Trends: Stable or upward in the upper three study reaches; stable or downward in the lower three study reaches.

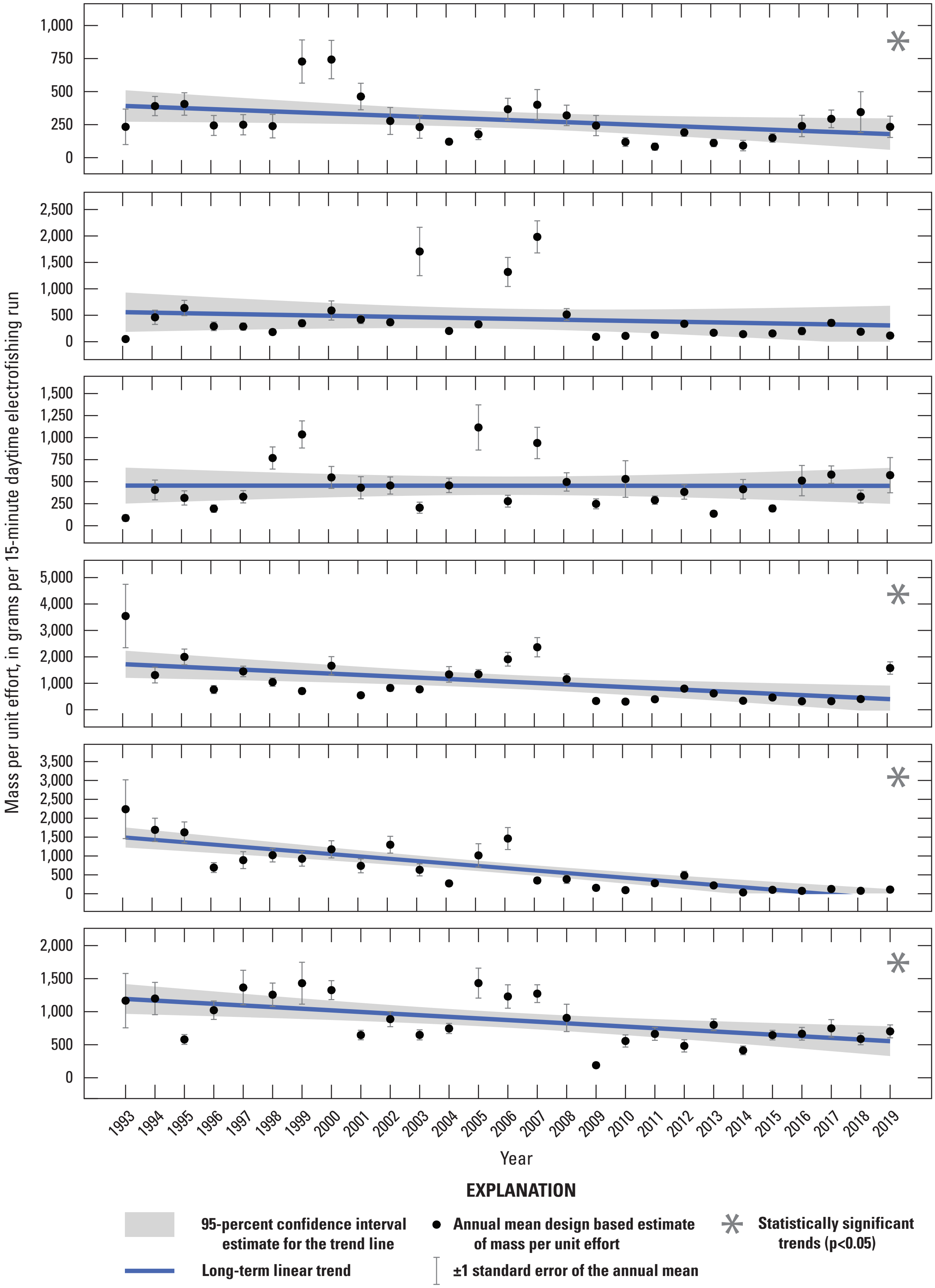

Figure G6. Mass per unit effort of forage fish, observed using daytime electrofishing, in the Upper Mississippi River Restoration Long Term Resource Monitoring element study reaches from 1993 to 2019. Please note differences in scaling of the y-axis among study reaches. See figure A1 for the location of each study reach. ±, plus or minus; <, less than.

reaches (ch. H; Bouska and others, 2020). This likely contributed to the contrast in fish species dominance noted above; phytophilic species (for example, sunfishes) dominated the upper reaches and open-water species (for example, catfishes and *Aplodinotus grunniens* [freshwater drum]) dominated the lower reaches (Bouska and others, 2020; Burdis and others, 2020), and sunfishes significantly decreased in the La Grange Pool of the Illinois River (Solomon and others, 2019).

Invasive bigheaded carps may continue to increase where they are established and spread further within the system (ch. I). On the lower three study reaches, bigheaded carps represented an increasing portion of total fish MPUE within local fish communities (ch. G). Shifts in fish community composition have been documented post invasion in the La Grange Pool of the Illinois River (Solomon and others, 2016), and increasing evidence points to effects on sportfish recruitment through direct food resource competition between larval sport fishes and bigheaded carps (Chick and others, 2020; ch. I) on the lower study reaches. The effects of bigheaded carps on native fauna, if they become established elsewhere in the system, are unknown but merit ongoing observation.

Restoration of habitats may contribute to increases in the populations of recreationally valued native fishes. Among the LTRM study reaches, only the Open River Reach lacks a completed habitat rehabilitation project (HREP). While benefits to fish communities have been observed locally because of the introduction of HREPs, reach-wide benefits have been harder to detect thus far. For example, Pool 8 has experienced a significant upward trend in MPUE for recreationally valued native fishes, and habitat rehabilitation efforts may have contributed to these observed increases. In contrast, HREPs on the La Grange Pool are semidisconnected from the main river, and their contributions to the local fish community are harder to assess and may be masked by other factors mentioned herein. Complementary studies before and after HREP construction, in addition to routine monitoring, can improve our ability to assess population level-effects of HREPs.

Habitat degradation and invasive species are likely the largest threats facing recreationally valued fishes (chs. A, G, and I). Although hydrology and geomorphology vary regionally in the Upper Mississippi River System (ch. B), increasing frequency and magnitude of flooding events, linked with shifting seasonal precipitation patterns, contribute to sedimentation, decreasing backwater depth, and associated loss of fish habitat system wide (chs. B and C). Future climate predictions for the Mississippi River Valley indicate a trend of heavy and frequent rainfall, particularly in winter and spring, to continue (ch. B).

Table G4. Common and scientific names and stock sizes of the 19 species included in the recreationally valued fishes indicator. Note that some recreationally valued species are also harvested by commercial fishers and are also included in the commercially valued fish indicator.

[mm, millimeter]

Common name	Scientific name	Stock size (mm)
Black crappie	*Pomoxis nigromaculatus*	130
Blue catfish	*Ictalurus furcatus*	300
Bluegill	*Lepomis macrochirus*	80
Channel catfish	*Ictalurus punctatus*	280
Flathead catfish	*Pylodictis olivaris*	350
Freshwater drum	*Aplodinotus grunniens*	200
Green sunfish	*Lepomis cyanellus*	80
Largemouth bass	*Micropterus salmoides*	200
Northern pike	*Esox lucius*	350
Pumpkinseed	*Lepomis gibbosus*	80
Redear sunfish	*Lepomis microlophus*	100
Rock bass	*Ambloplites rupestris*	100
Sauger	*Sander canadensis*	200
Smallmouth bass	*Micropterus dolomieu*	180
Walleye	*Sander vitreus*	250
White bass	*Morone chrysops*	150
White crappie	*Pomoxis annularis*	130
Yellow bass	*Morone mississippiensis*	100
Yellow perch	*Perca flavescens*	130

Summary of Commercially Valued Fishes

Indicator intent: Investigate differences across space and changes over time in commercially valued fishes.

Measurement: This indicator is the pool-wide mass per unit effort from eight native and four nonnative fish species, grouped by native status. Only fish of stock size or greater (O'Hara and others, 2007) were used in this analysis (table G5). The annual design-based index of mass per unit effort (grams per 15 minutes of daytime electrofishing) was estimated for six long-term study areas from 1993 to 2019 (fig. A1; Ickes and others, 2014 and Ratcliff and others, 2014). Annual trends in mass per unit effort were assessed using simple linear regression analysis for each study reach.

Special considerations: None.

Assessment

Status: All reaches support commercially valued fishes and commercial fisheries.

Trends:

Native species: Stable or upward in the upper three study reaches; stable or downward in the lower three study reaches.

Nonnative species: Stable or downward in all reaches.

Indicator: Commercially Valued Fishes

The production of commercially valued species is an important ecosystem service provided by the Upper Mississippi River System. It has been estimated that nearly 10 million pounds of fish, valued at over $3.8 million, are harvested from the basin each year (U.S. Army Corps of Engineers, 2012). This indicator is the combined pool-wide MPUE for eight native and four nonnative fish species, grouped by native status. Only fish of stock size or greater were used in this analysis (table G5).

The La Grange Pool of the Illinois River had the greatest mean annual MPUE for native commercially valued fish during the study period (4,820 g/15 min). However, annual mean MPUE in the La Grange Pool decreased and was below the long-term mean from 2007 to 2019 (slope = −197.28, $p<0.001$, $R^2=0.41$, $F=17.71$; fig. G8). Mean annual MPUE was next greatest for the Open River Reach (4,563 g/15min), followed by Pool 13 (3,572 g/15min), Pool 26 (2945 g/15min), Pool 4 (1980 g/15min), and Pool 8 (1831 g/15min).

Native commercially valued fish MPUE was stable over the period of record (1993–2019) in Pool 4, Pool 26, and the Open River Reach but increased in Pool 8 and Pool 13. Nonnative commercially valued fish MPUE was stable or decreased in all reaches (fig. G8), reflecting decreases in common carp throughout the system. The decrease in common carp MPUE was partially ameliorated within this indicator by increased mass of bigheaded carps in the lower three study reaches (ch. G). Nonnative commercial fish MPUE exceeded native commercial fish MPUE in all but the Open River Reach, making up 59–85-percent of MPUE of all commercially valued fishes in the impounded reaches and only 36-percent of MPUE in the Open River Reach.

While native commercially valued fish MPUE was stable or increasing in most reaches, the contribution of each species to total commercially valued fish MPUE changed over the period of record. For example, *Amia calva* (bowfin) made up an increasing proportion of commercial fish biomass in Pool 4, Pool 8, Pool 13, and the La Grange Pool. Meanwhile the contribution of buffalo fishes to total MPUE decreased in Pool 4 and the La Grange Pool. Although common carp was still a principal component of total commercially valued fish MPUE, common carp populations decreased in all LTRM reaches: a reflection of the broadscale decrease of common carp (Gibson-Reinemer and others, 2017). Bigheaded carps increased in prevalence in Pool 26, the Open River Reach, and the La Grange Pool of the Illinois River. This has resulted in decreases in native fish communities (chs. G and I), including the commercially important *Ictiobus cyprinellus* (bigmouth buffalo).

Habitat degradation and invasive species are likely the greatest threats to native commercially valued fish species (for example, chs. B, C, and I). However, stable or upward trends in native commercially valued fish MPUE in five of the study reaches indicate that habitat loss and harvest have not yet overtaxed the resource (Klein and others, 2018). Commercially valued fish MPUE on the La Grange Pool, historically the most productive of the study reaches, has decreased since the early 2000s, likely in relation to invasion by bigheaded carps (ch. I). This trend is likely to continue as bigheaded carps replace native fish biomass in the La Grange Pool (ch. G).

The increasing contribution of bowfin to total commercially valued fish MPUE in the upper three study reaches is noteworthy. Bowfins are an increasingly important commercial fish species, particularly as part of the roe industry (Koch and others, 2009). As world-wide sturgeon populations continue to collapse, greater pressure may be put on the bowfin, the sole member of its family of fishes (*Amiidae*) (Koch and others, 2009).

The number of full-time commercial fishing licenses issued has continued to decrease across the system (Schramm and Ickes, 2016). In addition, shifting focus by some agencies to contracted commercial harvest of bigheaded carps is changing the dynamic of commercial fishing, and it is unclear how this may affect native bycatch.

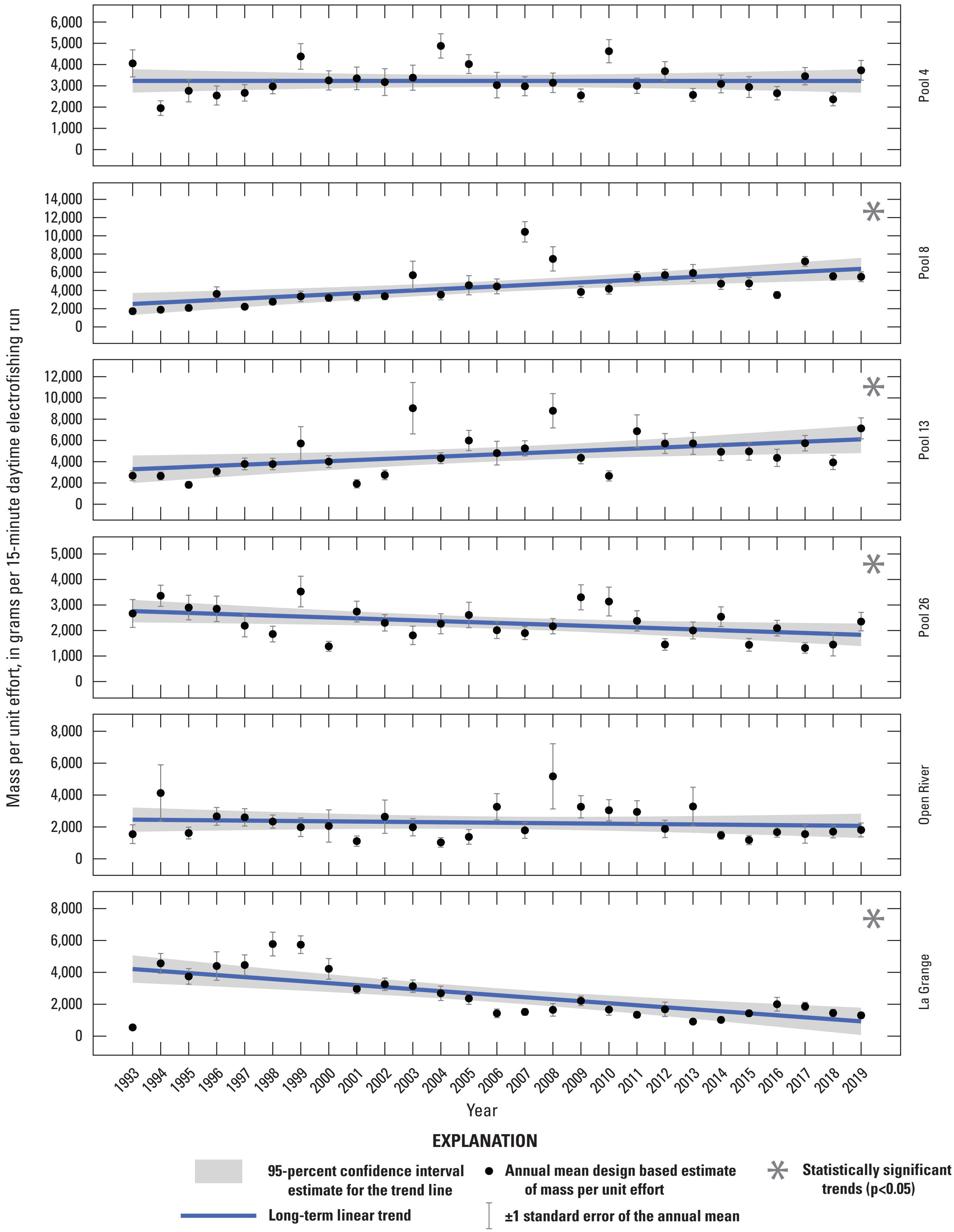

Figure G7. Mass per unit effort of recreationally valued fishes caught with daytime electrofishing in the Upper Mississippi River Restoration Long Term Resource Monitoring element study reaches from 1993 to 2019. Please note differences in scaling of the y-axis among study reaches. See figure A1 for the location of each study reach. ±, plus or minus; <, less than.

State of the Ecosystem

The Upper Mississippi River System fish community presently exists as a largely intact native fish community of ancient evolutionary origins and represents perhaps the most intact and functionally sound fish community in a large, developed, and modified temperate river anywhere. High species richness and diversity and lack of trends in species richness and total annual catches indicate the fish community remains relatively healthy. Compared to other heavily modified, large freshwater ecosystems, the fish community has generally proven resilient. However, continued vigilance is warranted. Recent research indicates the fish community is actively undergoing functional changes and potential regime shifts (Bouska, 2020). Additionally, 30-percent of the fish species observed by LTRM possess either Federal or State level conservation status designations (Ickes and others, 2005), which is a sizeable portion of fish diversity.

It is important to note that LTRM only began scientific observations in 1993, more than a century after river training and several decades after river impoundment. Two indicators were crafted to assess the contemporary ecosystem health of flowing (lotic) and slack water (lentic) aquatic habitats for fishes. Lotic habitats were stable, yet rather impaired. Lotic specialists, while stable over the LTRM period of record, were rare in the impounded reaches indicating impoundment, river training, and channel maintenance have sharply impaired this group of fishes that specialize in exploiting flowing water habitats. Lotic dependent species were more common in the impounded reaches and demonstrated stability, yet great annual variability, over the period of record. This indicates that lotic habitats in the impounded reaches can maintain self-sustaining populations of lotic dependent fishes, but also that there is a sizeable amount of interannual variability in lotic habitat quality, perhaps related to annual hydrology (ch. B). Conversely, lotic specialists were more common than lotic dependents in the Open River Reach.

Lentic habitats were abundant and generally healthy in the upper study reaches and less common or healthy in the lower study reaches. Aquatic plant recovery appears to be a major factor contributing to the health of lentic fishes (ch. F) in the upper reaches. Most lentic fishes possess dependencies upon aquatic plants for feeding or reproduction (O’Hara and others, 2007), and as most whole body particle feeders require visual acuity, they benefit from increases in water clarity associated with aquatic vegetation recovery (chs. E, F, and H) and refugia from predators the plants provide.

The establishment of bigheaded carps in the lower study reaches over the past decade has had seemingly large and negative effects on fish communities in the lower reaches (ch. I). In Pool 26 and in the La Grange Pool of the Illinois River, bigheaded carps composed roughly 50-percent of the fish community mass, exclusive of common carp. Research into the competitive interactions of these invasive species with native planktivorous species has demonstrated decreases in sympatric native species abundance and body condition (Irons and others, 2007). Additionally, recent research has also demonstrated decreases in nonsympatric sport fishes in association with increasing community dominance of bigheaded carps (Chick and others, 2020). Finally, a resilience-based assessment identified shifting species dominance by these nonnative species acts as a driver of regime change in the fish community (Bouska, 2020). Once a threshold of relatively low

Table G5. Common and scientific names and stock sizes of the eight native and four nonnative species that were included in the commercially valued fish species indicator. Note that some commercially valued fish species are also harvested by recreational fishers and are also included in the recreationally valued native fishes indicator.

[mm, millimeter]

Common name	Scientific name	Stock size (mm)
	Native	
Bigmouth buffalo	*Ictiobus cyprinellus*	280
Black buffalo	*Ictiobus niger*	280
Blue catfish	*Ictalurus furcatus*	300
Bowfin	*Amia calva*	200
Channel catfish	*Ictalurus punctatus*	280
Flathead catfish	*Pylodictis olivaris*	350
Freshwater drum	*Aplodinotus grunniens*	200
Smallmouth buffalo	*Ictiobus bubalus*	280
	Nonnative	
Bighead carp	*Hypophthalmichthys nobilis*	300
Common carp	*Cyprinus carpio*	280
Grass carp	*Ctenopharyngodon idella*	300
Silver carp	*Hypophthalmichthys molitrix*	250

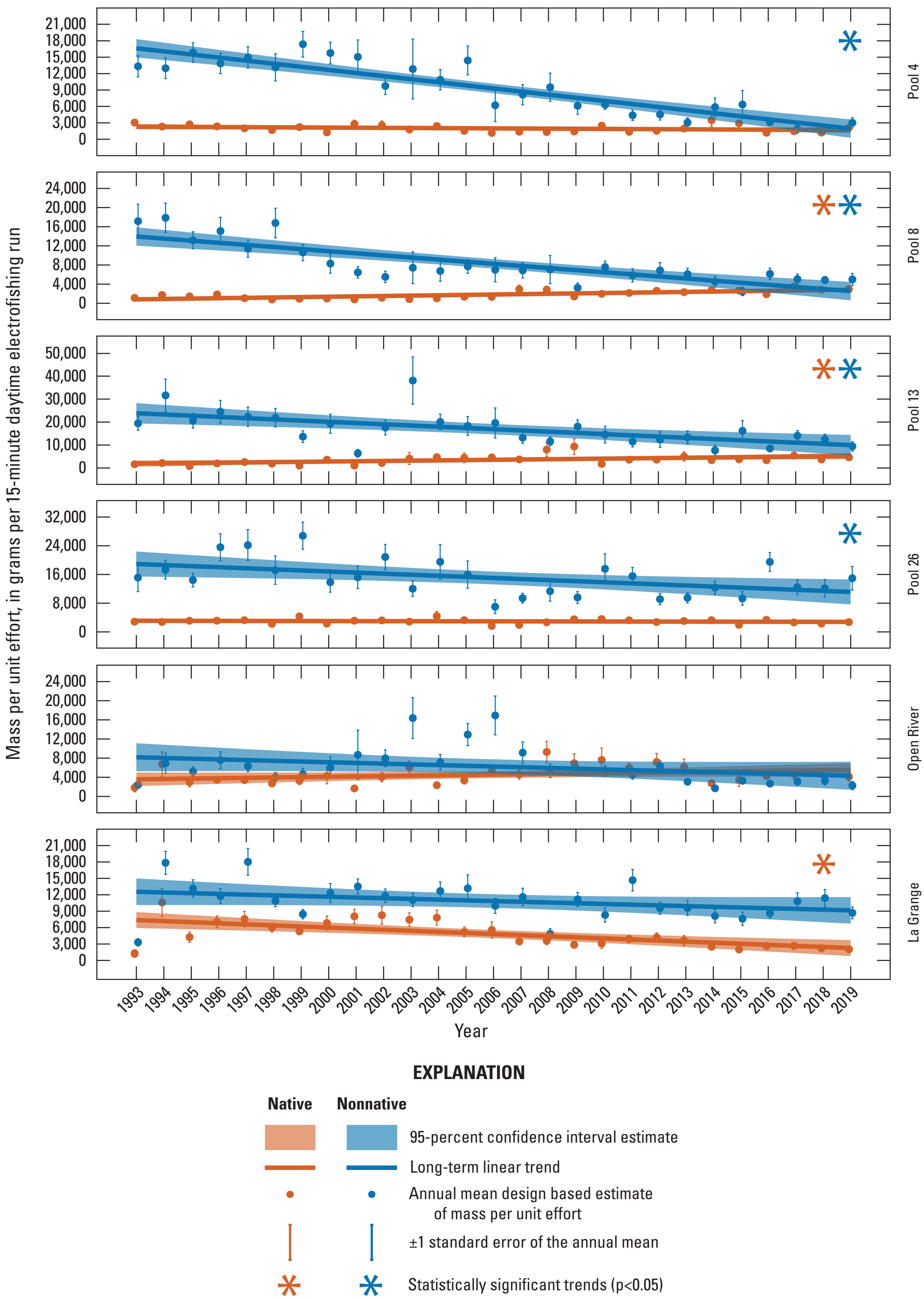

Figure G8. Mass per unit effort of commercially valued fishes observed by daytime electrofishing in the Upper Mississippi River Restoration Long Term Resource Monitoring element study reaches from 1993 to 2019. Please note differences in scaling of the y-axis among study reaches. See figure A1 for the location of each study reach. ±, plus or minus; <, less than.

silver and bighead carp biomass is passed, silver and bighead carp dominance persists as a result of reinforcing feedbacks. Notable decreases in small-bodied fishes of different trophic/functional feeding guilds in reaches dominated by bigheaded carps indicate competition for plankton resources as the principal mechanism of the decreases.

Forage fishes play a key role in aquatic trophic ecology and their abundance and mass are reflections of lower (planktonic) and upper (predaceous) trophic levels. Over the period of record (1993–2019), significant long-term decreases in forage fish production were observed in four of the six LTRM study reaches, including all three of the lower study reaches. Recent shifts in the mass dominance of bigheaded carps in the lower study reaches have been attributed to observed decreases in sport fishes (Chick and others, 2020) and wider community regime shifts (Bouska, 2020), reportedly through competition for planktonic food resources. In Pool 4, the downward trend was weak and most likely associated with intermittent and early boom years of shad, which are near their biogeographic northern distribution in Pool 4. If climate warms in the future, shad may become a more regular and steadier component of the forage fish community in the upper LTRM study reaches, with potential effects on lower trophic levels and benefits to higher tropic levels.

Recreational and commercial fishes provide social and economic benefits to States and localities. Both are important because the public perceives the health of the river, in large part, by the health of each of these indicators. Importantly, the LTRM observes recreational and commercial fishes entirely independent of the fisheries themselves, so status and trends reported herein are not dependent in any way on the behaviors of the recreational and commercial fisheries (Ickes and others, 2014). For the most part, recreational fishes represent a remarkably stable portion of the overall fish community. This stability has remained despite changes in recreational species preferences; fishing methods; new technologies in boats, tackle, and electronics; and even the very nature of recreational fishing (for example, shifts towards competitive fishing tournaments). The only observed significant changes in recreational fishes occurred in Pool 8 (increase), which may be a signal of many habitat restoration actions in this study reach, and in La Grange Pool on the Illinois River (decrease), which has been attributed to a rapid increase of bigheaded carps (Solomon and others, 2016; Chick and others, 2020; ch. I). Similarly, commercial fishes have remained reasonably stable over the period of record. The native portion of the commercial catch, overall, is identical to recreational fisheries, being stable except in Pool 8 (increasing) and La Grange Pool (decreasing), likely for the same reasons as expressed for recreational fishes. The nonnative portion of commercial fishes has decreased significantly over time in the upper study reaches, most likely attributable to nonfishery dependent decreases in common carp, which have decreased persistently over the LTRM period of record, speculated to be driven by disease (Gibson-Reinemer and others, 2017).

Future Pressures and Opportunities

Future pressures on the fish community have foundations in the historical past, contemporary river conditions, and potential future changes. Additional opportunities to better understand how contemporary and future conditions affect the fish community are readily available, especially given the increasing duration of monitoring observations and lengths of many ecological and hydrologic time series.

Historical Pressures

Three historical factors play a large role in the present-day status of the fish community. First, through the mid-late 1800s, few fishing regulations were in place and several species were overharvested. Most affected were those species that are longer lived, slower to reach sexual maturity, and had migratory habits for spawning that made them particularly vulnerable to intense seasonal exploitation (Carlander, 1954). Several of these species are prototypical large river species, such as the *Acipenseridae* (sturgeons) and the *Polyodon spathula* (American paddlefish). While initially suppressed through excessive exploitation, paddlefish and additional species remain rarer members of the fish community than they likely were historically because of decades of water quality impairments that affected most fish species and a series of successive river modifications for commercial navigation and floodplain development that have benefitted lentic species to the detriment lotic species.

Beginning in the mid-1800s, wholesale changes in the watershed ensued to bring fertile soils of the Upper Midwest into agricultural production. Land use patterns were substantially modified over a rather short period of time (ch. D). These changes affected runoff from the landscape and made basin soils more erodible, contributing high loads of inorganic sediments to the river (Turner and Rabalais, 2003). Concurrently, the United States underwent its economic industrialization phase. Point source contaminants hit the river hard and included several toxic compounds and untreated organic wastes emanating from burgeoning cities and agricultural livestock, often producing low oxygen levels in the river itself (Fremling, 2004). However, since environmental legislation in the late 1960s and early 1970s, water quality has vastly improved (Wiener and Sandheinrich, 2010). Somewhat concurrent, but in punctuated phases, the Upper Mississippi River System underwent a series of successive river engineering efforts to promote commercial shipping (Anfinson, 2005). This was accomplished through a program of successive river training projects and culminated with the establishment of a system of 27 locks and dams to ensure river navigation during summer low flows during the 1930s. Each successive project vastly altered flow and river conveyance patterns within the main-stem corridor and consequently vastly altered the distribution, types, and abundance of different aquatic habitats (ch. D).

Contemporary and Future Pressures

Contemporary and future pressures faced by the fish community include the persistent long-term effects of impoundment aging associated with the commercial navigation system, direct navigation effects associated with channel maintenance and tow operation, invasive species, and continuing changes in commercial and recreational fisheries.

Impoundment aging primarily manifests as a loss in lentic habitat quantity and quality as backwater and impounded areas receive sediments and become increasingly shallow (ch. C). The UMRR attempts to remedy some of these losses under a program of habitat rehabilitation, a series of engineering and habitat focused projects that seek to increase geomorphic and bathymetric diversity within the management areas, generally by reconstructing islands, dredging for depth, and protecting project land/water boundaries from high-flow events (Theiling, 1995).

Direct navigation effects manifest in the main channel and affect lotic environments and species. Ongoing and future pressures include direct mortality due to fish entrainment during tow operations (Gutreuter and others, 2006), habitat disruptions associated with annual channel dredging operations to ensure a 9-foot navigation channel, and emplacement of new and additional channel training structures designed to focus flow into the navigation channel to ensure sufficient commercial navigation depths (Remo, 2016).

Invasive species are an additional present day and future pressure to the fish community. Invasive species have fundamentally altered ecological communities across the world and in all types of ecosystems. Until recently, the fish community has proven resilient in the face of invasive fish species, and even largely to invertebrate and herbaceous invasive species. However, bigheaded carps in the lower study reaches continue to increase their mass dominance, and in some areas, there is evidence of total fish community production decreasing with these increases (ch. G) and decreasing abundance and body condition of species that share a life history niche with these invaders (Irons and others, 2007; Solomon and others, 2016; Chick and others, 2020).

Lastly, ongoing changes in commercial and recreational fisheries contribute to present day pressures on fishes that are likely to continue (Garvey and others, 2010). Within recreational fisheries, fishers are shifting how they pursue their sport, including shifts in species preferences; new technologies that make it easier to find, target, and catch fishes; cultural and ethical shifts between harvest and catch-and-release; and even shifts towards competitive modes of recreational fishing (for example, fishing tournaments; Hunt and others, 2013). The future combined effects of these shifts remain unknown and uncertain, and while demonstrating 25-years of stability in recreational fish production overall, certain species may come under additional pressures that may exceed their self-sustaining population capabilities, particularly if the U.S. population turns towards more harvest in economically uncertain or challenging times. Commercial fisheries are smaller, family-oriented, and more artisanal than the highly commercialized fisheries of the Gulf of Mexico, Atlantic and Pacific coasts, and the Laurentian Great Lakes. In the northern reaches, there has been a general trend towards fewer commercial licenses being granted and less overall activity. However, in the southern reaches, commercial fisheries are still reasonably strong and have even recently been incentivized and enlisted into efforts to control bigheaded carps (Schramm and Ickes, 2016). These are early experimental efforts; however, issues with unintended bycatch and mortality of nontarget native fishes may become an issue in the future.

Potential Future Opportunities

The scope, scale, duration, and scientific rigor of the LTRM are unique among large rivers in the United States and perhaps even the world. At nearly 30 years of highly standardized scientific observation, the LTRM now has a well-founded understanding of the mean condition of many ecosystem attributes, as well as their variation and dynamics. Under scenarios of future impairments or future rehabilitation, the LTRM is well-suited to capitalize on several novel opportunities.

First, the LTRM now possesses increased capabilities and opportunities to evaluate the magnitude and scale of future impairments and future restoration efficacy. Within the LTRM fish component, efforts are underway to use our long-term datasets to design impairment and restoration studies in a way that uses existing data to maximize the probability of detecting the effect of either an impairment or a rehabilitation response. The former can help to contribute towards negotiations with the source(s) of impairments while the latter can help contribute towards documenting the full effects of restoration efforts.

Second, the LTRM fish component has been proactive in developing new data sources that allow us to pursue a whole new set of ecological questions and issues. In the early years of the program, the LTRM Fisheries Component was largely limited to somewhat simplistic faunistic assessments and evaluations. These capabilities remain. However, through the development of new resources and capabilities, for example the development of a life history database (O'Hara and others, 2007), new population vital rates information, and the development of over 100 species specific growth models, it is now possible to recast the entirety of the LTRM fisheries data into functional and monetized units to pursue questions and issues pertaining to the functional dynamics of important fish assemblages or the whole community. Increasingly, these new capabilities are being used by researchers to uncover critical ecosystem dynamics that could be gained by no other methods (Bouska 2020; Chick and others, 2020; Gibson-Reinemer and others, 2017). These new capabilities permit a greater understanding of the functional dynamics of entire ecological communities in a complex, unique, and impaired ecosystem undergoing active rehabilitation efforts.

Finally, a primary goal of LTRM is to provide data and information to natural resource managers, policy makers, scientists, and the general public. The LTRM Fisheries Component has previously advanced several new methods to provide LTRM data in easily accessible and intuitive ways to better convey key ecosystem dynamics to all stakeholders. In the future, there are many additional opportunities to further develop and advance new data and information sharing tools under an open access paradigm.

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Chapter H: Using Long-Term Data to Understand the Causes and Consequences of Changes in Water Clarity and Aquatic Vegetation in the Upper Impounded Reach of the Upper Mississippi River

By Jeffrey N. Houser, Kyle R. Bales, Kristen L. Bouska, Rob M. Burdis, Alicia M. Carhart, Deanne C. Drake, Kathi Jo Jankowski, John E. Kalas, Danelle M. Larson, Eric M. Lund, and Jason J. Rohweder

Chapter H of

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In cooperation with the U.S. Army Corps of Engineers

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Chapter H: Using Long-Term Data to Understand the Causes and Consequences of Changes in Water Clarity and Aquatic Vegetation in the Upper Impounded Reach of the Upper Mississippi River

By Jeffrey N. Houser,[1] Kyle R. Bales,[2] Kristen L. Bouska,[1] Rob M. Burdis,[3] Alicia M. Carhart,[4] Deanne C. Drake,[4] Kathi Jo Jankowski,[1] John E. Kalas,[4] Danelle M. Larson,[1] Eric M. Lund,[3] and Jason J. Rohweder[1]

Overview

The distribution and abundance of aquatic vegetation in floodplain river ecosystems are affected by geomorphology, hydrology, and biogeochemistry (Franklin and others, 2008). In turn, aquatic vegetation affects geomorphology (rates of sediment erosion, transport, and sedimentation; Gurnell, 2014), biota (for example, fish habitat [Johnson and Jennings, 1998], and food/energy for waterfowl [Korschgen and others, 1988]), and rates of biogeochemical processing (Kreiling and others, 2011). In the Upper Impounded Reach of the Upper Mississippi River System, submersed aquatic vegetation (SAV) prevalence has varied dramatically since the completion of the locks and dams in the 1930s (ch. F). In the first few decades after the completion of the locks and dams, SAV was abundant and diverse in the newly created off-channel aquatic areas throughout the Upper Impounded Reach (Green, 1960; Peck and Smart, 1986; Fremling, 2005). During the late 1980s, there was a large and spatially extensive decrease in SAV prevalence (Rogers, 1994; Fischer and Claflin, 1995). The reasons for the decrease in SAV are poorly understood, and vegetation remained scarce in many areas through the 1990s (Rogers, 1994; Fischer and Claflin, 1995). Since the mid-2000s, SAV abundance has increased substantially throughout the Upper Impounded Reach, although not elsewhere in the Upper Mississippi River System (ch. F).

The magnitude and timing of the increase in SAV prevalence exhibited similarities and contrasts among and within navigation pools of the Upper Impounded Reach that have provided insights to the likely causes and consequences of those changes (Langrehr and others, 2007; De Jager and Yin, 2011; Gray and others, 2011; DeLain and Popp, 2014; Giblin, 2017; Drake and others, 2018; Bouska and others, 2020; Burdis and others, 2020). Here we summarize results from previous work and describe changes in SAV and associated ecosystem components from 1993 to 2019 for three Long Term Resource Monitoring element (LTRM) study reaches: Pool 4, Pool 8, and Pool 13. We use those study reaches as an example of a large change in the Upper Impounded Reach of the Upper Mississippi River System.

Long-Term Changes and Variability in Potential Drivers of Vegetation Change

Six main components inform our understanding of the shift from high turbidity and scarce vegetation to clear-water and abundant vegetation in the Upper Impounded Reach (fig. H1): (1) a period of low discharge from 2006 to 2009, (2) a systemic, long-term decrease in *Cyprinus carpio* (common carp), (3) a long-term decrease in turbidity (increase in water clarity), (4) local management and restoration actions intended to improve conditions for aquatic vegetation, (5) a long-term increase in aquatic vegetation since the late 1980s, and (6) interactions and feedbacks among these components that have subsequently maintained abundant aquatic vegetation in much of the Upper Impounded Reach.

Discharge

A period of low discharge predominated across most of the Upper Impounded Reach from 2006 to 2009 (fig. H2). During this period of low discharge, input of total suspended solids was reduced, water velocity was less, water-level fluctuations were smaller, and some backwaters and parts of

[1]U.S. Geological Survey.

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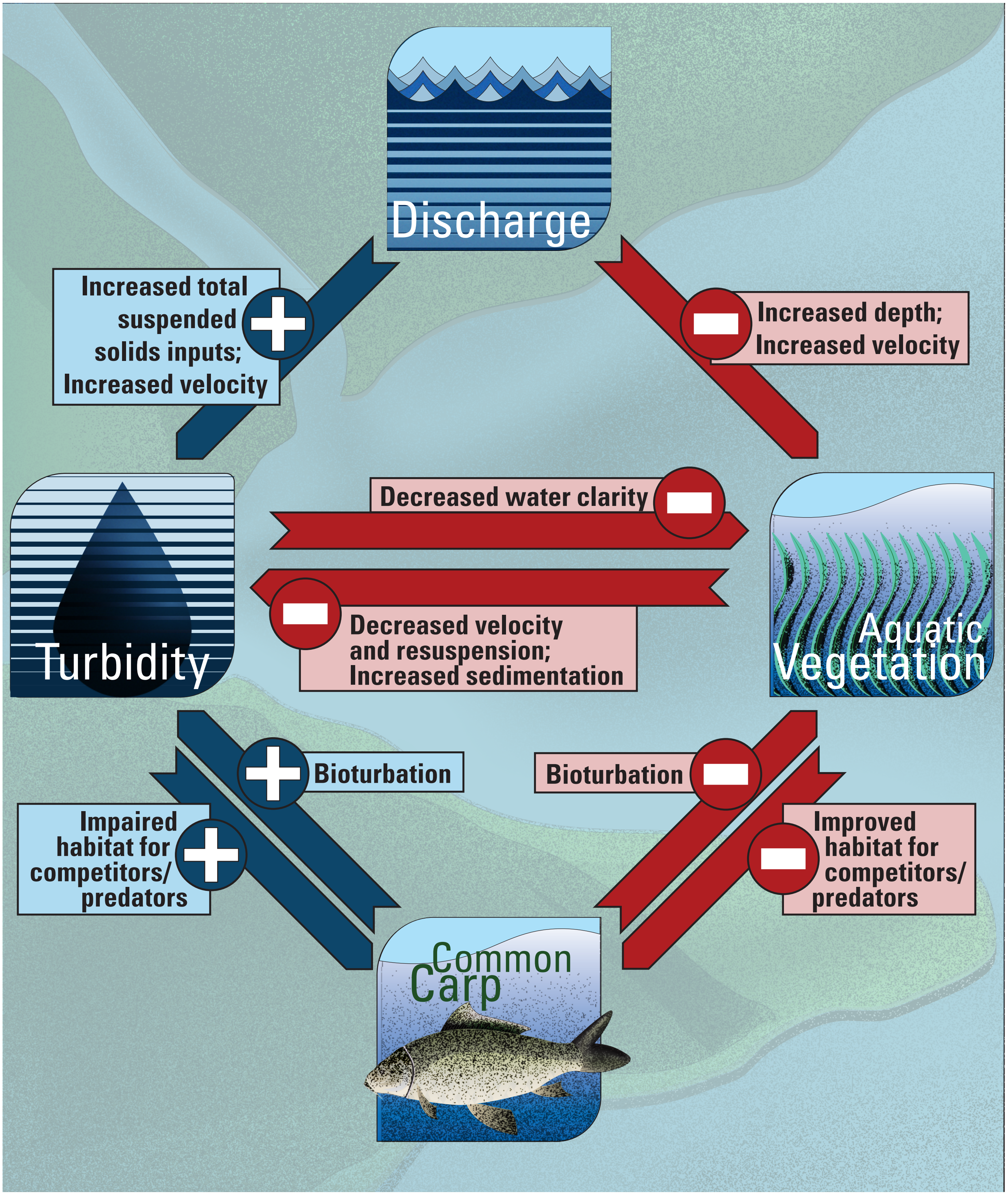

Figure H1. Conceptual model of the interactions among submersed aquatic vegetation, discharge, turbidity, and common carp. Dark red arrows indicate a negative effect, blue arrows a positive effect, and the text in boxes identifies the hypothesized mechanism.

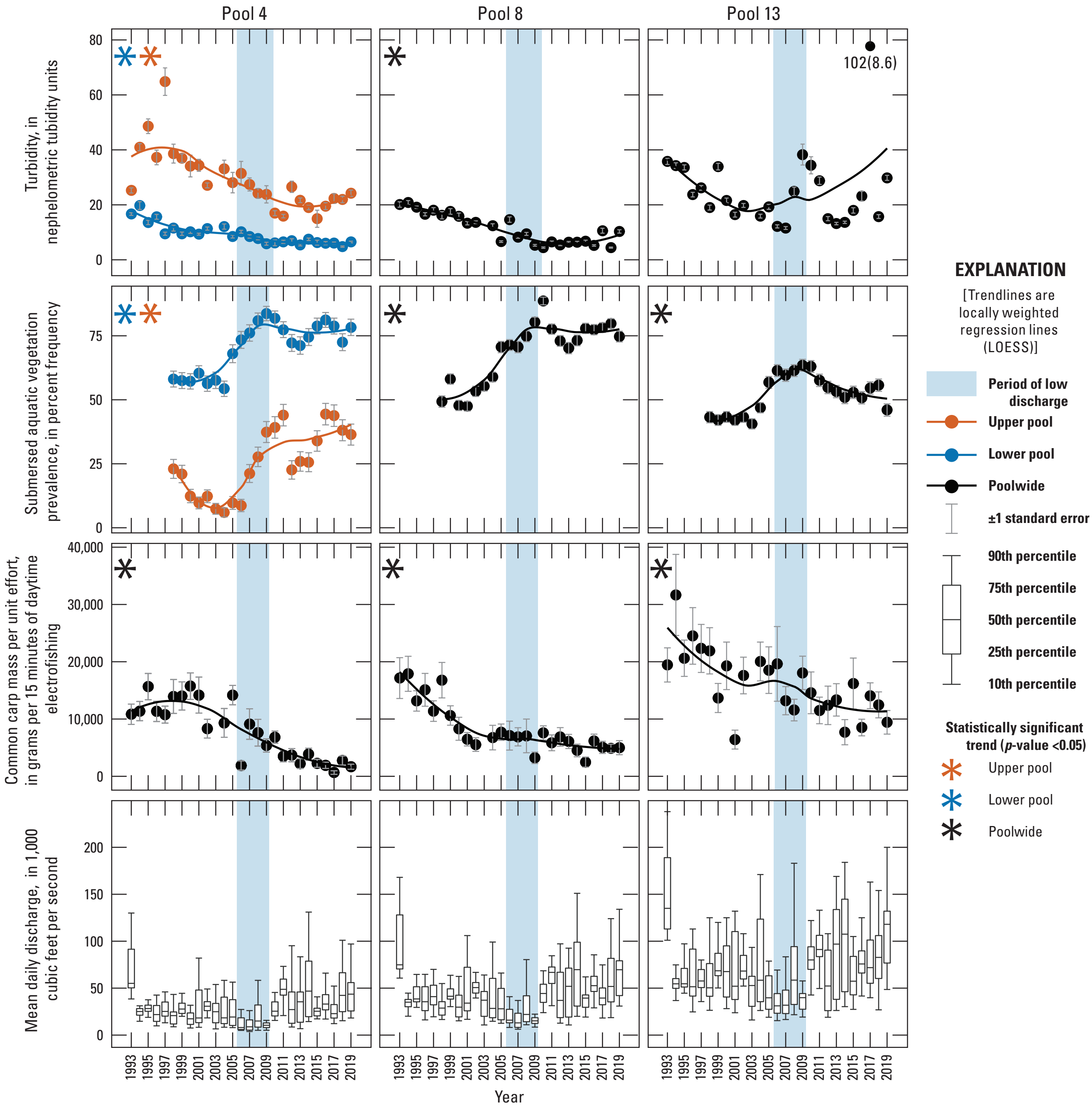

Figure H2. Times series from three Long Term Resource Monitoring element study reaches in the Upper Mississippi River System from years 1993 to 2019. Turbidity and submersed aquatic vegetation are shown separately for Upper Pool 4 and Lower Pool 4 because Lake Pepin (a natural, riverine lake formed by the Chippewa River Delta) is an effective sediment trap that reduces downstream turbidity. Shown are annual summer study reach mean turbidity, annual study reach mean submersed aquatic vegetation prevalence, annual study reach mean adult *Cyprinus carpio* (common carp) biomass per unit effort of daytime electrofishing, and box plot summaries of daily mean discharge for the summer months of June–August. Means and standard errors for turbidity, submersed aquatic vegetation prevalence, and common carp mass per unit effort were calculated using the Long Term Resource Monitoring element stratified random sampling data from the water quality, vegetation, and fisheries components, respectively (Soballe and Fisher, 2004; Yin and others, 2000b; and Ratcliff and others, 2014). Note that the Pool 13 summer mean turbidity value for 2017 is greater than the maximum y-axis value, and the 2017 summer mean (standard error) is shown in text adjacent to the point at the maximum value for the y-axis. LOESS, locally weighted regression model; ±, plus or minus; <, less than.

Table H1. Estimated slope and coefficient of determination values from linear regressions used to evaluate long-term trends in turbidity (1993–2019), submersed aquatic vegetation (1998–2019), and common carp (1993–2019). These results are the basis of the asterisks indicating significance in fig. H2 . [Except where indicated as nonsignificant, slopes and coefficient of determination values are significant (p-value less than 0.05). Turbidity and submersed aquatic vegetation data for Upper Pool 4 and Lower Pool 4 were analyzed separately because of the contrasts of these characteristics between the river upstream and downstream from Lake Pepin (see chs. E and F for details regarding the effect of Lake Pepin on turbidity). SAV, submersed aquatic vegetation; s.e., standard error; R^2, coefficient of determination; NA, not applicable; ns, nonsignificant]

Study reach	Turbidity		SAV		Common carp	
	Slope (s.e.)	R^2	Slope (s.e.)	R^2	Slope (s.e.)	R^2
Pool 4	NA	NA	NA	NA	−533.70 (45.70)	0.70
Upper Pool 4	−1.0 (0.20)	0.53	1.50 (0.30)	0.55	NA	NA
Lower Pool 4	−0.4 (0.05)	0.74	1.18 (0.22)	0.59	NA	NA
Pool 8	−0.6 (0.07)	0.74	1.49 (0.24)	0.65	−443 (62.10)	0.70
Pool 13	ns	ns	0.52 (0.23)	0.20	−496.2 (104.80)	0.50

the impounded areas were at their minimum depth for longer periods of the growing season.

Common Carp

Common carp have shown a long-term, widespread decrease in the Upper Mississippi River System (fig. H2; table H1; Gibson-Reinemer and others, 2017). The decrease in common carp has spanned at least the LTRM period of record (1993–2019). The cause of this decrease is not known. Current hypotheses include disease (for example, cyprinid herpes virus), cascading effects of improved water quality, and increased abundance of native fish populations, which can negatively affect common carp through competition for resources and predation (Giblin, 2017; Gibson-Reinemer and others, 2017).

Turbidity

Study reach mean turbidity in Pool 4 and Pool 8 has decreased significantly from 1993 through 2019 resulting in clearer water and greater light availability for SAV (fig. H2; table H1). Suspended sediment is the dominant contributor to turbidity in the Upper Mississippi River System, but modest decreases in the flux of suspended sediment from the tributaries (based on those monitored by LTRM, ch. E) can only partially explain the observed turbidity decrease in Pools 4 and 8. Flow-normalized flux of suspended sediment from the Black River (a major tributary that flows into Pool 7 just upstream from Pool 8; figs. E2 and E3) and main channel turbidity in Pool 8 decreased before and during the period of SAV increase in the impounded area of Pool 8. However, similar changes were not observed in the tributaries of Pool 4 where neither of the monitored tributaries exhibited significant long-term changes in flow-normalized flux of suspended sediment.

Turbidity in Pool 13 was more varied over time than Pools 4 and 8, exhibiting a period of decrease from 1993 through about 2007, followed by a period of higher and more varied turbidity (fig. H2). Long-term trends for Pool 13 summer mean turbidity were not detected (table H1), and in recent years summer mean turbidity was similar to that of the early 1990s. The highest turbidity during the period of record for Pool 13 occurred during 2017 immediately after a significant rainfall event (fig. H2). Pool 13 received high suspended sediment input from the Maquoketa River (which had the highest mean total suspended sediment concentration of all the sampled tributaries; ch. E) and other upstream tributaries that drain catchments dominated by agricultural land use. Total suspended solids input from the Maquoketa River exhibited high interannual variability and did not change significantly over the period of record (figs. E2 and E3).

Management and Restoration Actions

Island rehabilitation projects and summer water-level drawdowns have been used in parts of the Upper and Lower Impounded Reaches of the Upper Mississippi River System to address some of the consequences of the higher, stabilized water levels caused by the operation of the navigation locks and dams (Theiling and others, 2015).

Island Rehabilitation

In much of the Upper Impounded Reach, islands that were in the impounded portions of the navigation pools in the years immediately after the completion of locks and dams have gradually shrunk or disappeared. This island loss has generally been attributed to the erosive effects of wind and boat driven waves across the large open water expanses created by the operation of the locks and dams (WEST

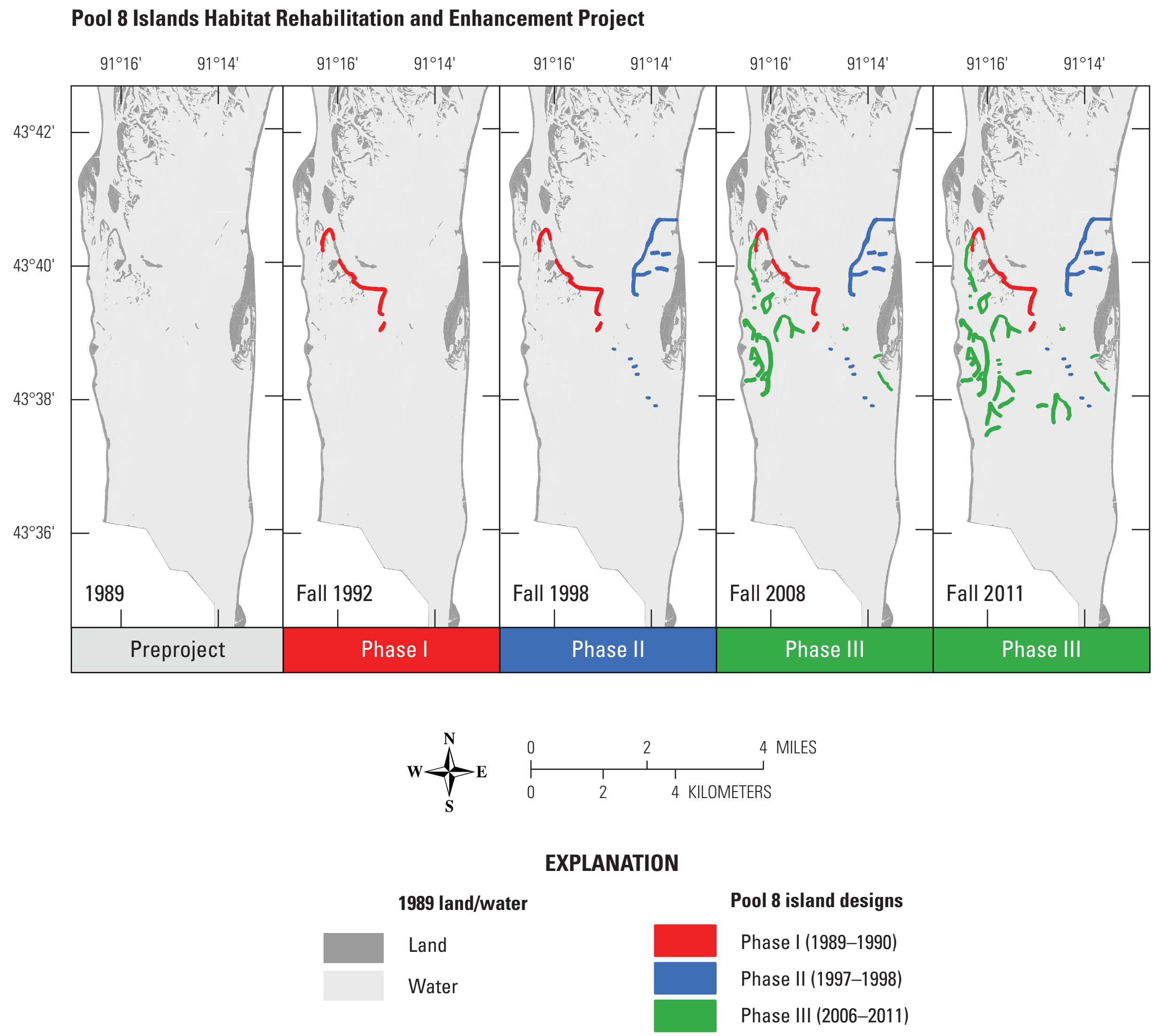

Figure H3. Maps showing the distribution of islands in Pool 8 before the Pool 8 Upper Mississippi River Restoration Habitat Rehabilitation and Enhancement Projects (1989), after completion of Phase I (1992) and Phase II of the restoration projects (1998), during Phase III (2008), and at the completion of Phase III (2011). Image by Jeff Janvrin, Wisconsin Department of Natural Resources, used with permission

Consultants Inc., 2000). As the islands eroded, wind fetch (in other words, the unobstructed distance over water that wave-generating wind can travel) increased, further increasing wave-associated erosive energy in the impounded area (Rohweder and others, 2008). Greater wave energy can exert direct physical disturbance on aquatic vegetation; it can also indirectly affect aquatic vegetation through increased resuspension of bottom sediments leading to reduced water clarity and availability of light. Some of the Upper Mississippi River Restoration program's Habitat Rehabilitation and Enhancement Projects (HREPs) reconstruct islands that have been lost to erosion in order to reduce wind fetch and water velocity in affected areas (U.S. Army Corps of Engineers, 2012). From 1989 to 2011, a subset of the islands in the impounded area of lower Pool 8 that had been lost to erosion were reconstructed as a part of three HREPs (fig. H3) that reduced wind fetch (and associated wave energy) in much of the impounded area (fig. H4) and created areas of reduced flow in the vicinity of the islands (Theiling and others, 2015).

Summer Water-Level Drawdowns

Summer water-level drawdowns have been implemented in select navigation pools of the Upper Mississippi River to partially simulate seasonal low-water conditions that no longer occur because of the operation of the navigation locks and dams (Lubinski and others, 1992; Theiling, 1995; Kenow and others, 2016).

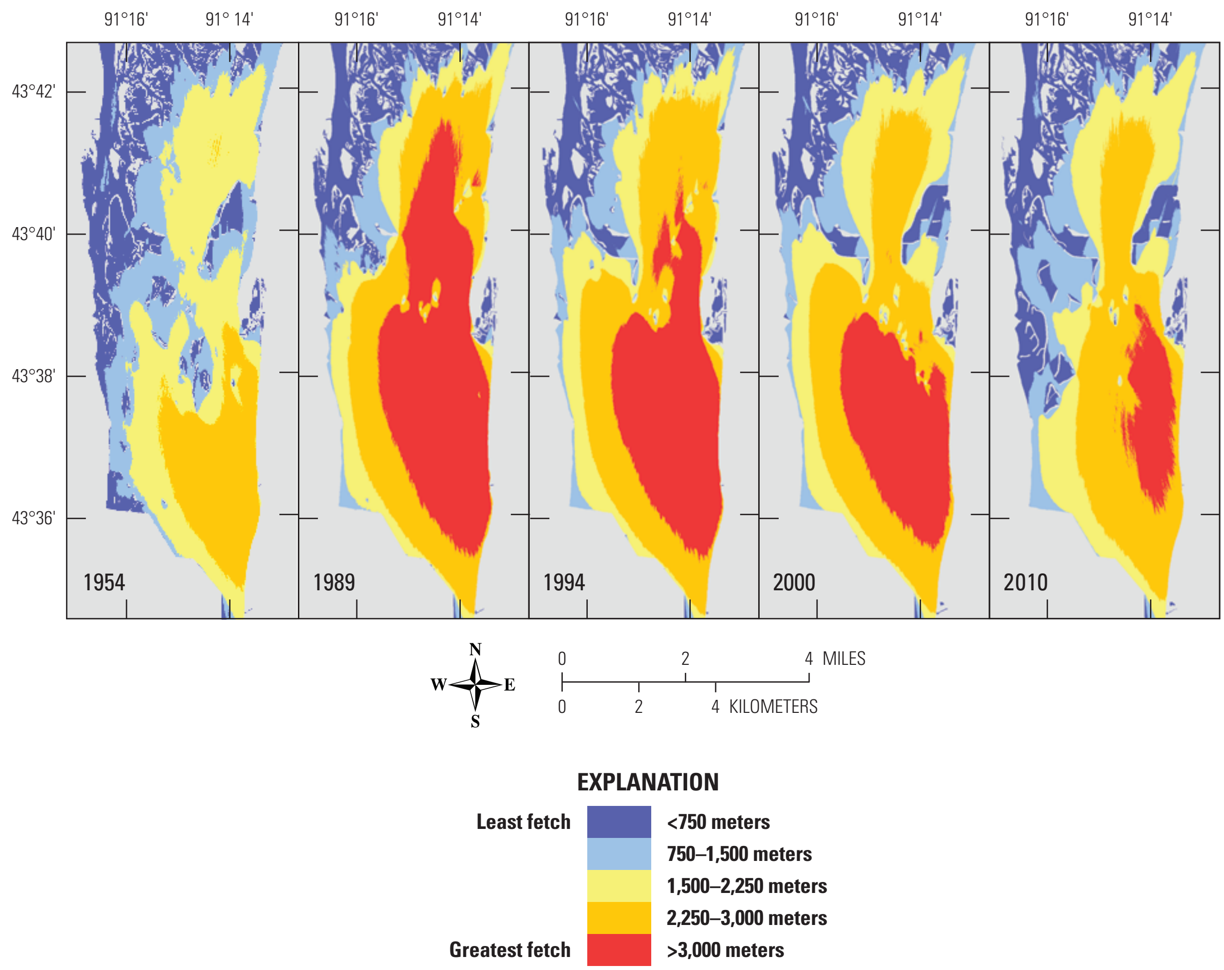

Figure H4. Long-term changes in weighted wind fetch modeled using the configuration of islands in Pool 8 from 1954 (about 20 years after lock and dam construction), 1989 (immediately prior to Phase I of the Pool 8 Habitat Restoration and Enhancement Projects), 1994 (after completion of Phase I island restoration), 2000 (after completion of Phase II island restoration), and 2010 (near the completion of Phase III island restoration). Wind fetch is the unobstructed distance over water that wave-generating wind can travel and wave energy can accumulate. The wave energy in areas with greater fetch can physically disturb plants and resuspend bottom sediments reducing water clarity and light available for plant growth. For details regarding the wind fetch model see Rohweder and others (2008).<, less than; >, greater than.

In the Upper Mississippi River, the emphasis of water-level drawdowns has been on facilitating emergent vegetation (for example, Flinn and others, 2005; Kenow and others, 2016; Coulter and others, 2019), and the effects of water-level drawdowns on submersed vegetation in the Upper Mississippi River System has received less attention. Experts expect that lower water levels expose some areas of existing SAV (negative effect), but temporarily create additional shallow areas that could support SAV (positive effect). Lower water levels may consolidate the sediment exposed during the drawdown providing longer term benefits including reduced subsequent turbidity (increased water clarity) and improved sediment rooting conditions for SAV. Two assessments of the effects on SAV of intentional, summer water-level drawdowns found either no response or a small decrease in SAV; however, these assessments were limited in scope, consisted of sampling in two backwaters in Pool 5 immediately after the conclusion of the drawdown, and did not assess longer-term responses (such as during subsequent growing seasons; Kenow and others, 2016). Positive effects of intentional reductions in water level on emergent vegetation have been previously documented for Pools 24, 25, and 26 (Wlosinski and others, 2000; Flinn and others, 2005; and Coulter and others, 2019). Previous work indicates that periods of low water can facilitate increases in SAV in the Upper Mississippi River System (Burdis and others, 2020), and water-level drawdowns in impaired shallow lake ecosystems have had positive effects on vegetation and wildlife (Larson and others, 2020).

Aquatic Vegetation

Long-term changes in aquatic vegetation occurred in Pools 4, 8, and 13. There were informative similarities and differences among these three study reaches in the timing and magnitude of those changes.

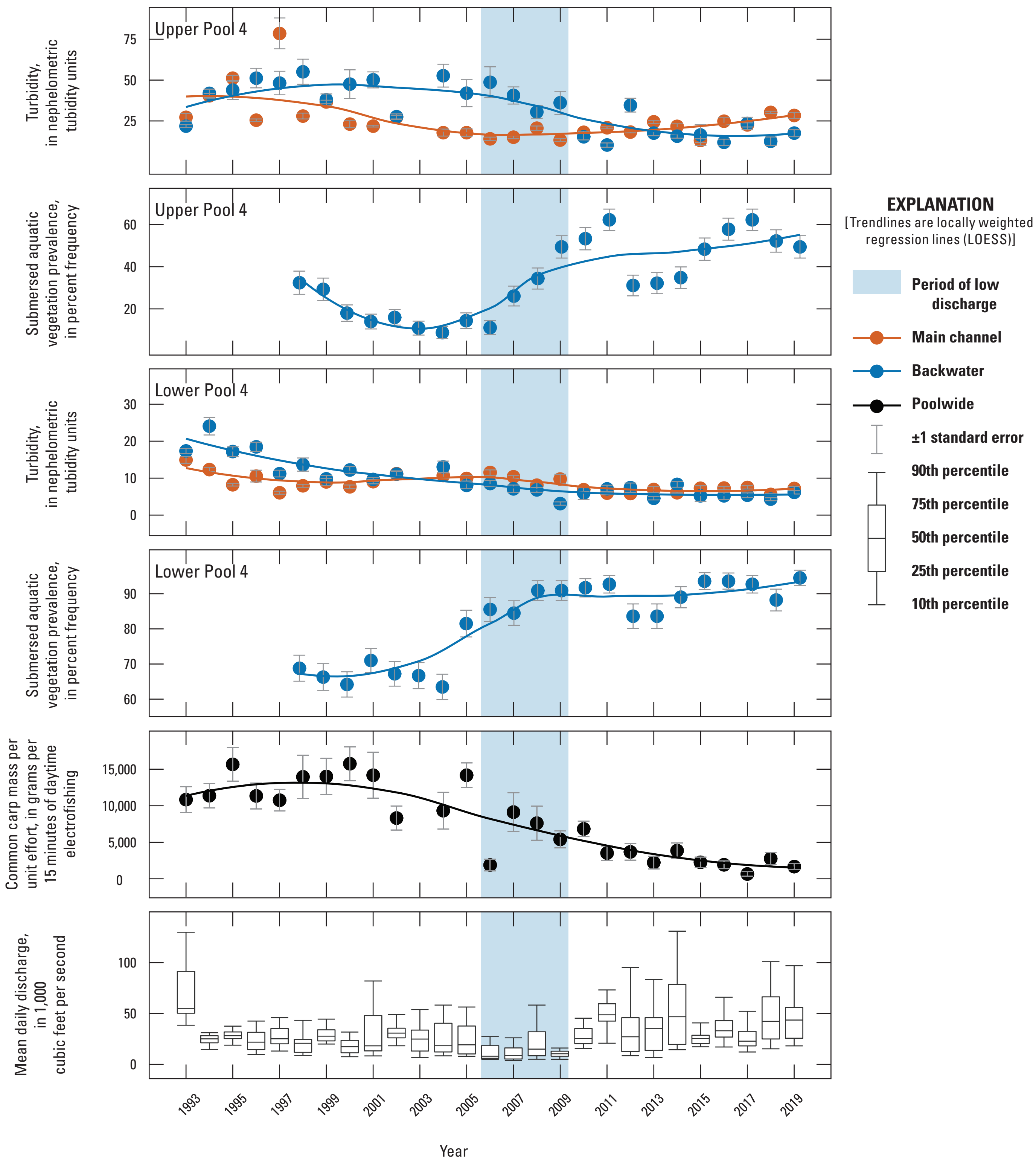

Figure H5. Times series for Pool 4 of the Upper Mississippi River System from 1993 to 2019. Turbidity and submersed aquatic vegetation are shown separately for Upper Pool 4 and Lower Pool 4 because Lake Pepin (a natural, riverine lake formed by the Chippewa River Delta) is an effective sediment trap that reduces downstream turbidity. Shown are the annual summer study reach mean turbidity; annual study reach mean submersed aquatic vegetation prevalence; annual study reach mean adult *Cyprinus carpio* (common carp) biomass per unit effort of daytime electrofishing; and box plot summaries of daily mean discharge for June–August. Means and standard errors for turbidity, submersed aquatic vegetation prevalence, and common carp mass per unit effort were calculated using the Long Term Resource Monitoring element stratified random sampling data from the water quality, vegetation, and fisheries components, respectively (Yin and others, 2000b; Soballe and Fisher, 2004; and Ratcliff and others, 2014). Discharge data are from U.S. Geological Survey streamgage 05344500 at Prescott, Minnesota (U.S. Geological Survey, 2021). LOESS, locally weighted regression model; ±, plus or minus

Long-Term Changes in Pool 4 Submersed Aquatic Vegetation

During the period of record, vegetation prevalence in the backwaters of Lower Pool 4 was consistently greater than that of the backwaters of Upper Pool 4 (fig. H5). Because of the sedimentation that occurs in Lake Pepin that separates the upper and lower pool (fig. A6), Lower Pool 4 exhibited clearer water than Upper Pool 4. Water level is regulated by the locks and dams such that water-level fluctuations were smaller in the lower pool than in the upper pool. This clearer water and smaller water-level fluctuation most likely supported greater SAV prevalence in Lower Pool 4 relative to Upper Pool 4 (Burdis and others, 2020).

Prevalence of SAV in Upper and Lower Pool 4 was low from 2000 to 2006, but increased substantially from 2006 to 2011 coincident with a period of unusually low discharge (fig. H5; ch. B). SAV prevalence increased from ~20 to ~50-percent and from ~70 to ~90-percent in Upper and Lower Pool 4, respectively, during this period of low discharge. This period of low discharge was an important contributor to the shift to more vegetated conditions (Burdis and others, 2020). During this period of low discharge, water was shallower, water-level fluctuations were smaller, and turbidity was lower, which likely contributed to the observed increase in SAV in the backwaters of Pool 4. The reduced water levels and decreased turbidity likely resulted in a larger area of Pool 4 exhibiting the shallow water, low current velocity, and light conditions required for establishment and growth of SAV (Madsen and others, 2001; Haslam, 2006). The observed increase in SAV prevalence and the decrease in turbidity also coincided with a substantial decrease in common carp abundance.

Long-Term Changes in Pool 8 Submersed Aquatic Vegetation

The SAV dynamics in Pool 8 differed substantially between its backwater and impounded areas (fig. H6). Even at the beginning of the period of record (1998), SAV prevalence was high in backwaters of Pool 8 (~80-percent prevalence), indicating that a substantial recovery from the low point in the late 1980s and early 1990s had already occurred. Results of LTRM SAV transect sampling in select backwaters from 1991 to 1998 indicate that in some backwaters SAV prevalence was high throughout that period, whereas in others it was lower in 1991 and had increased by 1998 (Yin and others, 2000a).

The greatest change in SAV prevalence since 1998 occurred in the impounded area where it changed from ~35 to ~90-percent prevalence by 2009 (fig. H6). This increase in SAV prevalence in the impounded area may have begun before the period of low discharge (which was from 2006 to 2009) that appeared to be a key part of the increase in SAV in Pool 4.

Pool 8 is unique among the study reaches because it experienced two types of large-scale management and restoration actions during the period of record: island rehabilitation and summer water-level drawdowns. Ecological changes associated with the completion of Phase I and Phase II of the HREP islands include increased SAV prevalence and decreased phytoplankton abundance in areas adjacent to the islands (Langrehr and others, 2007; Gray and others, 2011). An assessment of the effects of Phase III of the HREP islands detected no increase in the diversity or abundance of aquatic vegetation within 400 meters of the Phase III islands, likely because SAV had become well-established in those areas before completion of the project; however, in more distant parts of the impounded area, an increase in SAV associated with completion of the Phase III islands was detected—potentially as a result of decreased wind fetch (Drake and others, 2018). SAV community composition has shifted from lotic species dominance to lentic species dominance in areas sheltered by the rehabilitated islands (Carhart and De Jager, 2019).

In Pool 8, water-level drawdowns were conducted during 2001 (July 6–September 15) and 2002 (July 2–September 15). Thus, the drawdowns immediately preceded the period of greatest increase in SAV in the impounded area of Pool 8. However, this increase in SAV occurred concurrently with an ongoing period of substantial decrease in common carp, followed the completion of island rehabilitation projects, and in the context of increases in SAV throughout the Upper Impounded Reach. Thus, the role of each of these potential contributors to the increase in SAV in the impounded area of Pool 8 remains uncertain and provides an example of the challenges of assessing cause and effect regarding long-term changes in a dynamic, multiuse system. More thorough assessments of the magnitude, duration, and spatial extent of the response of SAV to future water-level drawdowns in the Upper Mississippi River System would inform management decisions regarding if, when, and where to implement additional water-level drawdowns.

The period of greatest increase in SAV in the impounded area of Pool 8 followed the completion of the second phase of island construction and two consecutive years of summer water-level drawdowns in Pool 8, and occurred in the midst of a long-term decrease in common carp (fig. H6). Constructed islands potentially increased SAV in portions of the impounded area, but not in areas where vegetation was already established (Langrehr and others, 2007; Drake and others, 2018; see also “Management and Restoration Actions” above). The increases in SAV observed in Pool 8 occurred during a period of regional increase in aquatic vegetation that was observed in much of the Upper Impounded Reach. As a result of the simultaneous changes in possible drivers of water clarity and vegetation in Pool 8, it is not yet possible to determine the relative contribution of each of the previously mentioned possible drivers of SAV change. It is likely that long-term changes in suspended sediment inputs, low discharge in the mid-2000s, island reconstruction, water-level drawdowns, and the preceding decrease in common carp all contributed to creating conditions more suitable to the establishment and growth of SAV in the impounded area of Pool 8.

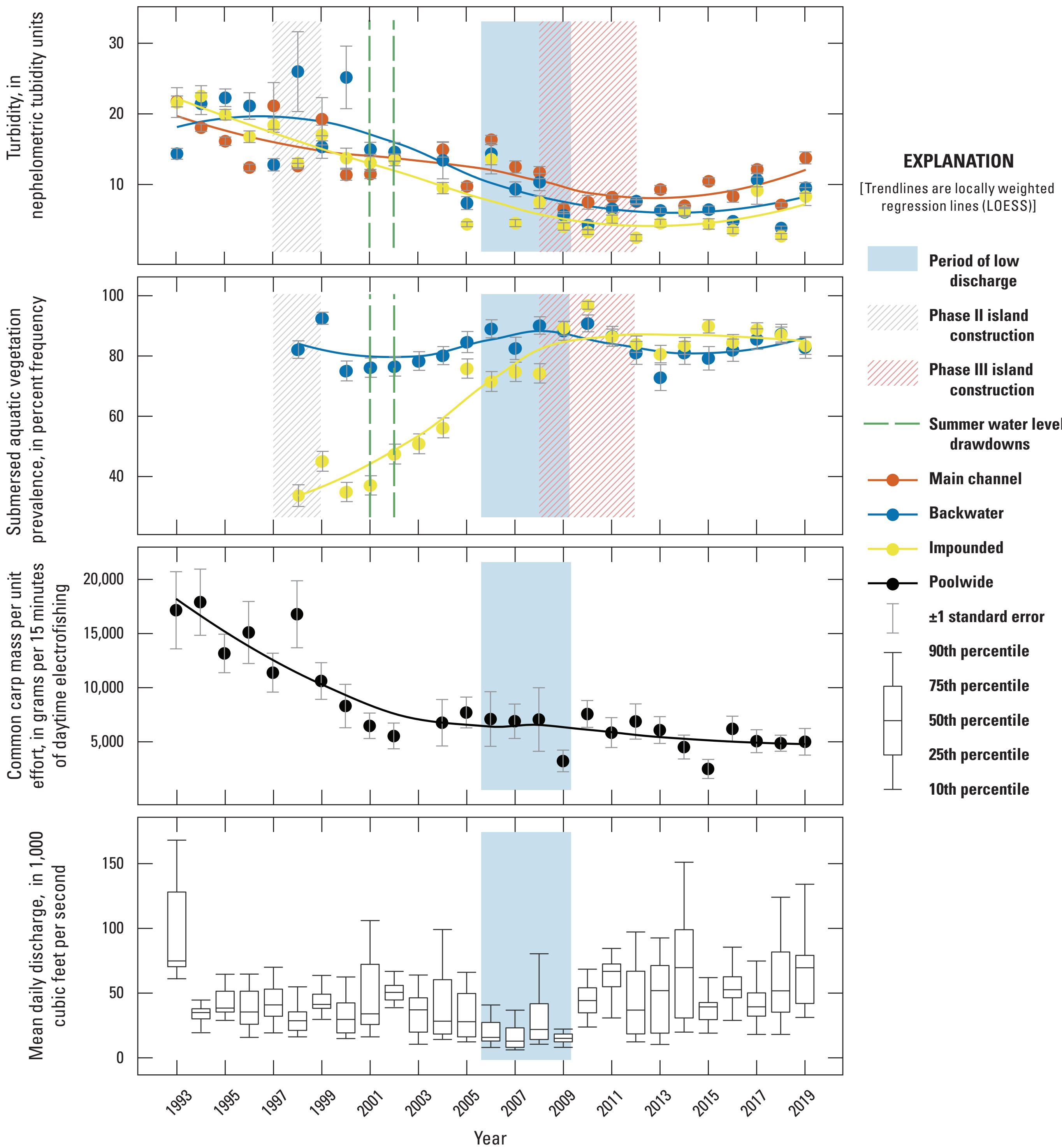

Figure H6. Times series for Pool 8 of the Upper Mississippi River System from 1993 to 2019. Shown are the annual summer study reach mean turbidity, annual study reach mean submersed aquatic vegetation prevalence, annual study reach mean adult *Cyprinus carpio* (common carp) biomass per unit effort of daytime electrofishing, and box plot summaries of daily mean discharge for June through August. Means and standard errors for turbidity, submersed aquatic vegetation prevalence, and common carp mass per unit effort were calculated using the stratified random sampling data from the Upper Mississippi River Restoration program's Long Term Resource Monitoring element water quality, vegetation, and fisheries components, respectively (Yin and others, 2000b; Soballe and Fisher, 2004; and Ratcliff and others, 2014). Discharge data are from U.S. Geological Survey streamgage 05378500 at Winona, Minnesota (U.S. Geological Survey, 2021). LOESS, locally weighted regression model; ±, plus or minus.

The turbidity and SAV trajectories over the last decade in Pool 8 indicate that the current conditions of high SAV prevalence and low turbidity have been stable and resilient to recent increases in discharge.

Long-Term Changes in Pool 13 Submersed Aquatic Vegetation

The changes in SAV in Pool 13 differed from those observed in Pool 4 and Pool 8. As in Pools 4 and 8, pool-wide prevalence of SAV increased from 2004 to 2010. However, from 2011 to 2019 SAV prevalence decreased in Pool 13, especially in the impounded area (fig. H7). Unlike Pool 8, patterns in SAV prevalence in Pool 13 were similar in backwater and impounded areas, and SAV prevalence was consistently lower in the impounded area relative to backwaters.

Pool 13 exhibited smaller long-term changes in turbidity and SAV prevalence over the period of record than Pools 4 and 8, and SAV prevalence in the impounded area of Pool 13 decreased since a midperiod peak in ~2010. Pool 13 continued to receive high suspended sediment input from tributaries (ch. E). Common carp biomass decreased in Pool 13 but remained substantially higher than in Pool 4 and Pool 8 (fig. H2). The high turbidity values and decreases in SAV prevalence observed in the later years of the period of record indicate considerable uncertainty regarding the future trajectory of vegetation in Pool 13.

Interactions and Feedbacks

There is evidence for feedbacks among some of the above described components of the ecosystem. The contrasts between main channel turbidity and that of backwaters and impounded areas of Lower Pool 4 and Pool 8 are examples. In Lower Pool 4, backwater turbidity was consistently greater than that of the main channel from 1993 until 2001, after which it was similar to, and often less than, that of the main channel (fig. H5). In Pool 8, backwater turbidity was often greater than that of the main channel until ~2001, after which it was consistently lower than that of the main channel (fig. H6). In Upper Pool 4, the inversion in relation between main channel and backwater turbidity did not occur until ~2014. Two mechanisms potentially explain this change: (1) the effects of increased SAV on water clarity and (2) decreased sediment resuspension as fewer common carp caused less bioturbation. The sources of turbidity in the backwaters included suspended sediment inputs from channels, resuspension of bottom sediments, and phytoplankton. Increased prevalence of SAV could have affected all these sources by increasing the rate at which sediment settles out of the water column, by reducing the resuspension of benthic sediments, and by competing with phytoplankton for light and nutrients. SAV can reduce current velocity and wave energy (from wind exposure and boat traffic) thereby increasing sedimentation rate, reducing sediment resuspension rate, and increasing water clarity (Barko and James, 1998; Schulz and others, 2003). The resulting clear water may have facilitated additional increases in vegetation prevalence. Evidence for the role of this feedback was observed in the backwaters of Lower Pool 4 where a notable increase in water clarity occurred for 2 years after an increase in SAV abundance (Burdis and others, 2020).

A second possible mechanism for the change in backwater turbidity relative to the main channel was a reduction in sediment resuspension in backwaters via bioturbation by common carp as their abundance decreased. Common carp bioturbation can directly affect SAV through physical disturbance of plants and indirectly by resuspending sediments, increasing turbidity, and reducing light availability (Weber and Brown, 2009). Increases in aquatic vegetation and associated increases in water clarity may also indirectly affect common carp. As vegetation abundance and water clarity increase, conditions improve for sight-feeding predators that consume common carp eggs and juveniles, potentially limiting recruitment (Bajer and others, 2015). However, in their assessment of long-term fluctuations in *Lepomis macrochirus* (bluegill) and common carp abundance in the Illinois River, Gibson-Reinemer and others (2017) did not find evidence supporting the limitation of common carp recruitment by predation, rather they suggest that the most likely explanation for the broad-scale, long-term decrease in common carp in the Upper Mississippi River System is a pathogen affecting young fish as evidenced by the scarcity of juvenile carp.

Additional changes in the fish community have been associated with the long-term changes in water clarity and SAV. In Pool 4, vegetation changes have been associated with a shift in the fish community from species more associated with open water to those associated with vegetation (DeLain and Popp, 2014; Burdis and others, 2020). The abundance of *Dorosoma cepedianum* (gizzard shad) and *Notropis atherinoides* (emerald shiner) decreased as SAV increased. Species that increased included *Notropis texanus* (weed shiners), *Micropterus salmoides* (largemouth bass), *Amia calva* (bowfin), and *Perca flavescens* (yellow perch). More broadly, the long-term increases in recreationally valued fishes in the Upper Impounded Reach (ch. G; Giblin, 2017; Burdis and others, 2020) and shifts to more functionally diverse fish communities in Pools 4 and 8 (Bouska, 2020) may be due in part to the increased SAV prevalence and associated clearer water.

Regime Shift

The dramatic changes in turbidity and SAV prevalence and evidence of internal feedbacks indicate these changes may represent a regime shift for the Upper Impounded Reach of the Upper Mississippi River System (Giblin, 2017; Burdis and others, 2020; Bouska and others, 2020). The concept of regime shifts is that ecosystems have multiple stable states in which they can exist rather than a single, global equilibrium to which

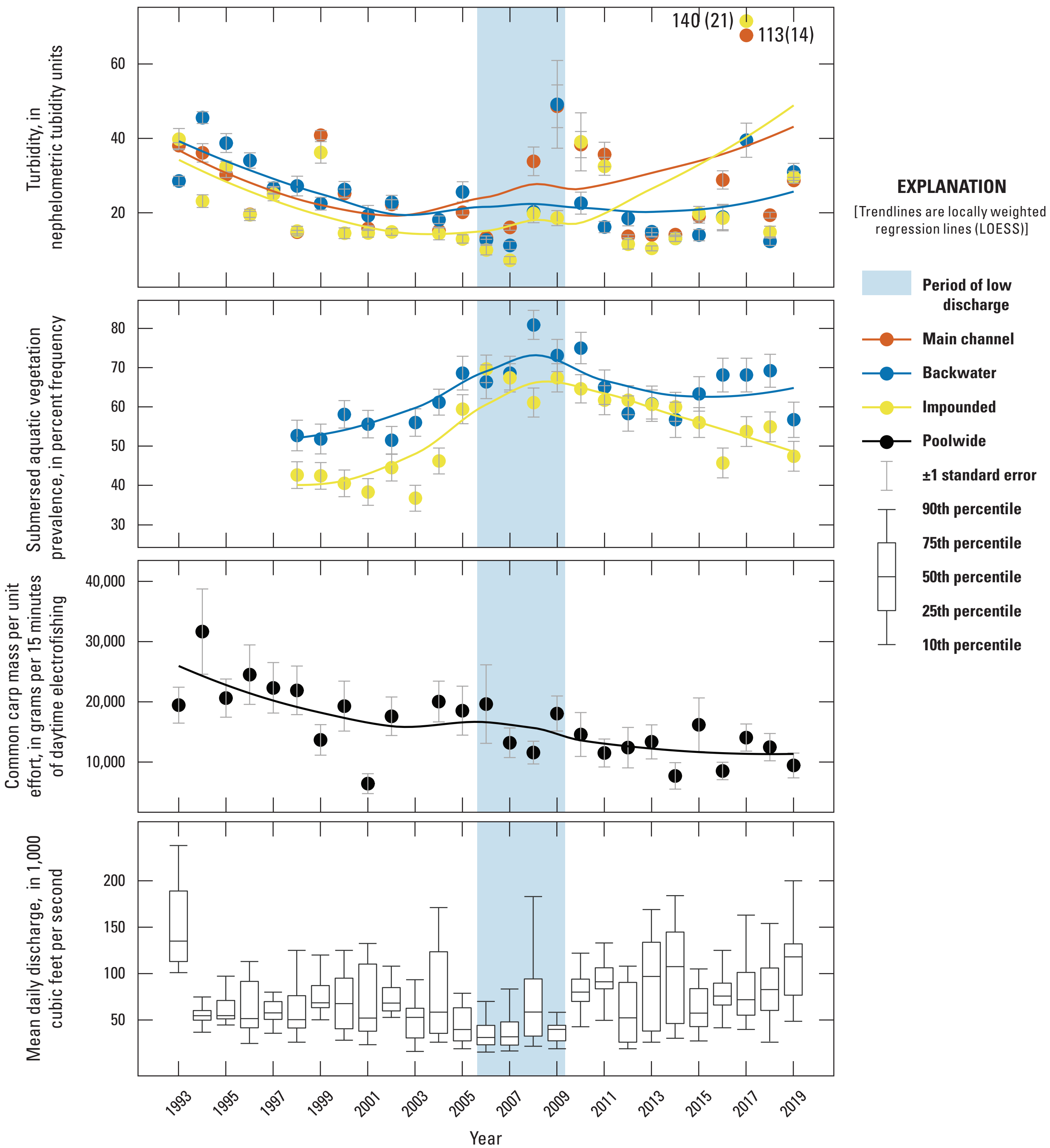

Figure H7. Times series for Pool 13 of the Upper Mississippi River System from 1993 to 2019. Shown are the annual summer study reach mean turbidity, annual study reach mean submersed aquatic vegetation prevalence, annual study reach mean adult *Cyprinus carpio* (common carp) biomass per unit effort of daytime electrofishing, and box plot summaries of daily mean discharge for June through August. Means and standard errors for turbidity, submersed aquatic vegetation prevalence, and common carp mass per unit effort were calculated using the stratified random sampling data from the Upper Mississippi River Restoration program's Long Term Resource Monitoring element water quality, vegetation, and fisheries components, respectively (Yin and others, 2000b; Soballe and Fisher, 2004; and Ratcliff and others, 2014). Discharge data are from U.S. Geological Survey streamgage 05420500 at Clinton, Iowa (U.S. Geological Survey, 2021). Note that the mean summer turbidity values for the main channel and impounded area in 2017 are greater than the maximum y-axis value, and their values (standard error) are shown in text adjacent to the points at the maximum value for the y-axis. LOESS, locally weighted regression model; ±, plus or minus.

the ecosystem can always return if the causal disturbances are mitigated or removed (for example, Folke and others, 2004). An ecosystem remains in a given regime as long as the stabilizing feedbacks are maintained. When ecological drivers cross critical thresholds, these feedbacks are destabilized, and the ecosystem can shift to a new regime. Once established in a new regime, it can be difficult, take a long time, or be impossible for the system to return to the previous regime. Such shifts have been well documented in shallow lakes (Scheffer, 2004), coral reefs (Mumby, 2009), and grasslands (Ratajczak and others, 2014).

Regime shifts have been conceptualized and demonstrated in large rivers (Hilt and others, 2011; Ibáñez and others, 2012; Bouska and others, 2020; Diamond and others, 2022), but most rivers worldwide do not have sufficient time series data to evaluate regime shifts (Biggs, and others, 2018). In rivers, the strong role of physical and external factors (mainly discharge) in maintaining or shifting regimes makes identifying and understanding regimes and regime shifts difficult. That regime shifts can occur over a period of years further complicates their assessment. However, others have found that nonlinearities and internal feedbacks can shift and maintain regimes in rivers (for example, Dent and others, 2002).

In Pools 4 and 8, the increases in water clarity and vegetation that were associated with low-flow conditions from 2006 to 2009 have persisted during an extended period of higher discharge, indicating that internal feedbacks (such as the effects of abundant vegetation on sedimentation and resuspension) were likely contributing to the resilience of the high vegetation and low turbidity regime in these pools. For example, only during one summer sampling episode (summer 2015) since the low water period has turbidity in Upper Pool 4 been similar to typically higher values observed before 2009. In contrast, the recent higher turbidity and variability and apparent decrease in SAV in Pool 13 indicate that these feedbacks may be insufficient to maintain an abundant vegetation regime in that pool, especially in the impounded area.

In contrast to the changes described in this chapter for the Upper Impounded Reach, estimates of SAV distribution from LTRM land cover data indicate that there has been little increase in vegetation in the Lower Impounded Reach and only in two navigation pools of the Illinois River since 2010 (De Jager and Rohweder, 2017; De Jager and others, 2018). In some of those areas, the lack of vegetation likely is a consequence of the combined effects of geomorphology, water clarity, and the magnitude of annual water-level fluctuations (Carhart and others, 2021). In other areas, the lack of change across 20 years of diverse discharge conditions may result from feedbacks maintaining a regime of higher turbidity and scarce vegetation.

Implications

In Pools 4 and 8, substantial increases in SAV and decreases in turbidity have occurred since 1998. These changes occurred in the context of a period of important regional scale changes: low discharge from 2006 to 2009, decreases in turbidity, and a long-term decrease in common carp. In Pool 8, local management actions likely further contributed to and reinforced these long-term changes. Although these changes initially coincided with a period of lower discharge, they have been sustained during higher discharge conditions since 2010 indicating the current state of abundant vegetation and clearer water is resilient (so far) to these higher discharge conditions. The situation in Pool 13 may be different. Although Pool 13 also experienced the regional period of low discharge (2006–09) and long-term decrease in common carp, local turbidity and SAV dynamics were different from those of Pool 4 and Pool 8. Although SAV prevalence increased and turbidity decreased during the first half of the period of record, since ca. 2010, SAV decreased and high turbidity conditions were more frequent. Thus, the abundant SAV and clear water state in Pool 13 may be less resilient to higher discharge conditions, and the future trajectory of this part of the river is more uncertain. Because of concerns about the future trajectory of Pool 13, a habitat rehabilitation project for the impounded area of Pool 13 is being planned to reduce wind and wave energy, diversify flow, and temporarily reduce water levels to improve conditions for SAV.

The increase in SAV and water clarity in much of the Upper Impounded Reach represents a significant improvement in the ecological condition of the Upper Mississippi River System. Gaining an understanding of the multiple drivers of these long-term changes can help inform ongoing efforts to better protect and restore the river. Assessing the causes and consequences of the long-term changes in water clarity and SAV reveal the value and limitations of long-term monitoring data for understanding the structure and function of floodplain river ecosystems. The long-term data provide a clear picture of where, when, and how much water clarity, vegetation, and associated other components of the ecosystem changed and show that these changes exhibited some informative similarities and some contrasts among and within the three study reaches of the Upper Impounded Reach. These spatial and temporal similarities and differences within and among study reaches provide important clues as to the causes and consequences of these changes. With each additional year of long-term data collected, different combinations of these drivers and responses are added to the record such that future analyses may further clarify some of the interactions within this complex system to guide future restoration and management.

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By Levi E. Solomon[1] and Brian S. Ickes[2]

Background on Invasive Carp

Hypophthalmichthys molitrix (silver carp; fig. I1) and *H. nobilis* (bighead carp), collectively known as invasive carps, are large, nonnative, planktivorous fishes that now extend throughout most of the Upper Mississippi River System. Native to eastern Asia, invasive carp were first brought to Arkansas by the aquaculture industry in the 1970s and the fish subsequently escaped into adjacent rivers connected to the Mississippi River. Both species have steadily expanded their range since. The Upper Mississippi River Restoration (UMRR) program's Long Term Resource Monitoring (LTRM) element fish component staff observed the first bighead carp in Pool 26 in 1993 and the first silver carp in Pool 26 and La Grange Pool in 1998—the earliest scientific detection of both species in the system (see fig. A1 for locations of study reaches, pools, and tributaries). Dispersal throughout the Upper Mississippi River System includes expansion into nearly all major and minor tributaries; the only thing limiting the expansion is man-made dams built to create reservoirs. One of many examples of this is the Sangamon River, a tributary of the Illinois River in central Illinois where invasive carp successfully invaded the entire river until they reached the dam that forms Lake Decatur (not shown on any figures). Another example is the Salt River (not shown on any figures), a tributary of the Mississippi River in northeastern Missouri that was successfully invaded up to the Clarence Cannon Dam that forms Mark Twain Lake (not shown on any figures). At this time, invasive carp have been caught as far north as Pool 2 and the St. Croix River but remain uncommon or rare north of Pool 16.

[1]Illinois Natural History Survey.

[2]U.S. Geological Survey.

Invasive carp exhibit certain life history traits that make them very successful in many habitats (fig. I2). Many of these traits (also known as r-selected traits) are shared with other highly invasive species. Some of these life history traits include high fecundity (a single female can hold in excess of 1 million eggs [Cooke and Hill, 2010]), early sexual maturation (females and males can reach maturity as early as 2 or 3 years of age), and fast growth (silver carp can pass 300 millimeters [approximately 12 inches] in their first full year of life [Stuck and others, 2015]). However, one characteristic of invasive carp that may make them especially adverse to native ecosystems is that they are planktivorous, and they are more efficient feeders than most of our native fishes. Plankton—zooplankton and phytoplankton (also known as algae)—make up the diet of all native fishes early in their lives before most species undergo a diet shift as they grow, with most native species expanding their diets to include macroinvertabrates and fish (O'Hara and others, 2007). Some native species, such as *Dorosoma cepedianum* (gizzard shad), *Ictiobus cyprinellus* (bigmouth buffalo), and *Polyodon spathula* (paddlefish), consume primarily plankton their entire lives (O'Hara and others, 2007). As a result, invasive carp compete for planktonic resources with all native fishes at some point in their lives and some native fishes for all their lives.

What LTRM Data Have to Say about Status and Trends of Invasive Carp Themselves

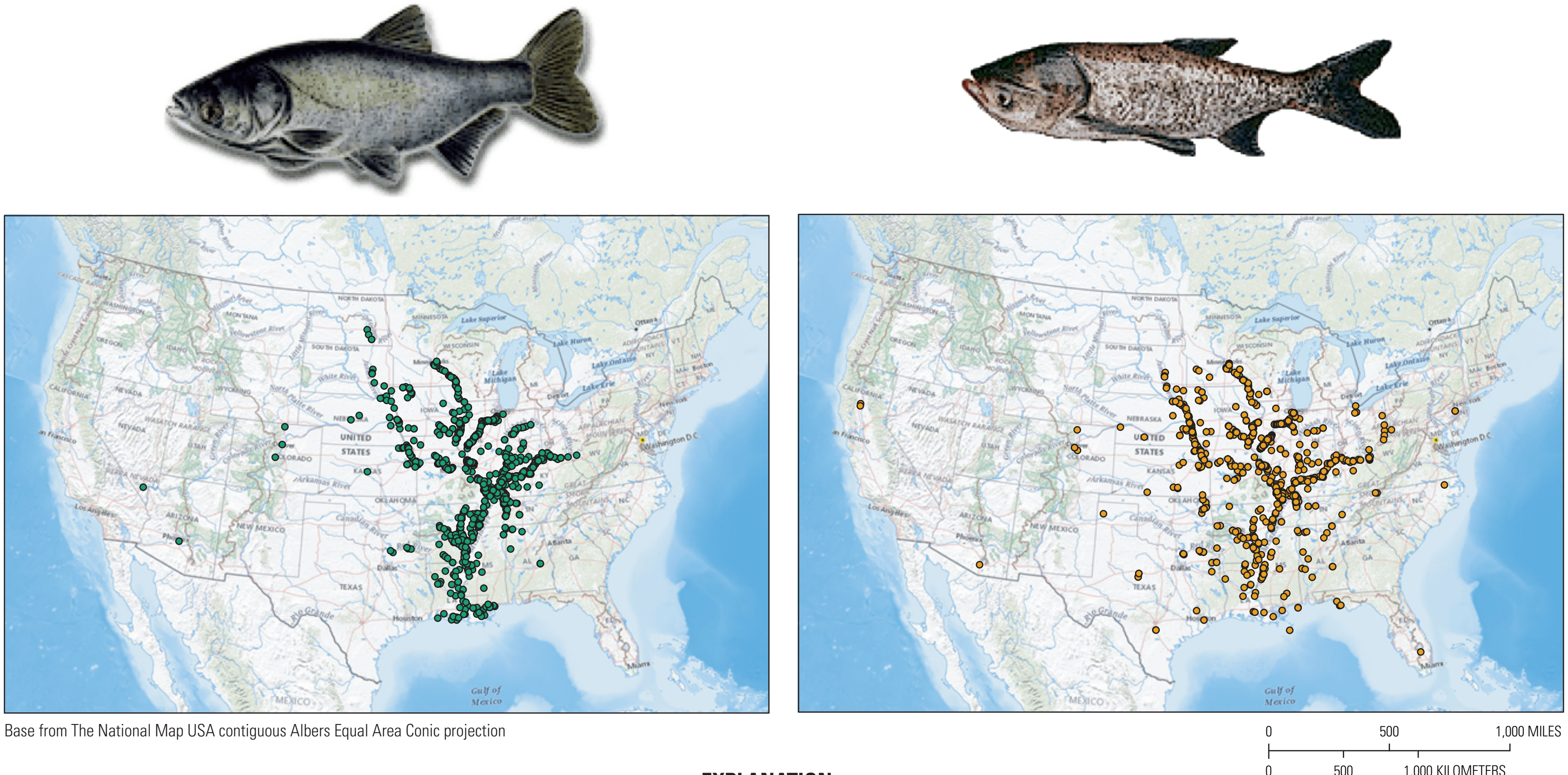
The LTRM has been observing fish community and fish species responses across 1,960 kilometers of river and for 27 years using a scientific sampling design (Ickes and others, 2014) and standardized sampling protocols (Ratcliff and others, 2014). When initially conceived, the LTRM fisheries sampling design was implemented to gain comparable data over space and time, assess the status and trends in fisheries

Figure I1. Photograph showing *Hypophthalmichthys molitrix* (silver carp). Silver Carp have achieved incredible densities and mass dominance in the La Grange Pool of the Upper Mississippi River System in less than 15 years (ch. G). Long-term monitoring efforts conducted by the Upper Mississippi River Restoration program are consequently revealing profound shifts in the diverse native fish community in this region. Photograph by Thad Cooke, Illinois River Biological Station of the Illinois Natural History Survey, U.S. Army Corps of Engineers Upper Mississippi River Restoration program, Havana, Illinois, used with permission.

resources, and to inform applied management and policy needs across multiple Federal and State resource management jurisdictional boundaries. By exercising a scientific sampling design and standardized protocols for data collection and reporting, the LTRM fish component can control common sources of ecosystem survey error (Ickes and others, 2014) and provides the best estimates possible on the status and trends of fisheries resources. In this section, we report the status and trends of invasive carp as observed by the LTRM fish component from 1993 to 2019. It is important to note, as the primary data source for the full fish community, LTRM fisheries data have played a central role in diagnosing and investigating invasive carp invasion and establishment and informing myriad State and Federal agencies and interjurisdictional coordinating committees seeking to control the spread of these invasive species (Ickes, 2008).

In the previous UMRR status and trends report (Johnson and Hagerty, 2008), Ickes (2008) presented the observed status and trends of all nonnative fishes. At that time, invasive carps had only been observed in the lower three reaches (Pool 26 and the Open River Reach of the Mississippi River and La Grange Pool on the Illinois River). Since this last reporting, there has been no change in the gross distribution of invasive carps across the LTRM fish community monitoring network; however, localized studies beyond annual status and trends monitoring by LTRM have demonstrated there are established populations as far north as Pool 16 in Iowa, indicating both species have overcome the prominent migration barrier at Lock and Dam 19 (Ickes, 2008; Larson and others, 2017; Jackson and Runstrom, 2018). Additionally, adult-sized individuals of invasive carp have now been observed as far north as Pool 2 in Minneapolis, Minnesota, and the St Croix River near Prescott, Minn., by commercial fishermen and local management authorities; however, no reproduction has been observed to date which indicates populations have yet to be established in these areas. Thus, since the last UMRR status and trends report (Johnson and Hagerty, 2008), the status of invasive carp has not changed.

There have, however, been notable changes in the trends of invasive carps in the lower three reaches. When last reported (Johnson and Hagerty, 2008) in the context of UMRR status and trends, invasive carps represented a much



Silver carp

Planktivore

Most cultured fish in the world

22degN distribution

500k–>2m eggs

Eggs semi-buoyant and drift

Can attain 60+ pounds

Leap up to 3 meters high

Bighead carp

Planktivore

Fifth most cultured fish in the world

24degN distribution

250k–>1.2m eggs

Eggs semi-buoyant and drift

Can attain 80+ pounds

Figure I2. Native range, United States distribution as of 22 July 2020, and life history traits of *Hypophthalmichthys molitrix* (silver carp) and *H. nobilis* (bighead carp) (U.S. Geological Survey, 2020) smaller proportion of the full fish community's indexed mass. Before 2008, these two invasive species represented 0–17, 0–15, and 0–33 percent of the full fish community annual indexed mass in Pool 26, the Open River Reach, and La Grange Pool, respectively. Since 2008, these two nonnative species now represent 17–65, 1–21, and 31–61 percent of the full fish community indexed annual mass in Pool 26, the Open River Reach, and La Grange Pool, respectively. In 2010, silver carp became the mass-dominant species in the La Grange Pool (ch. G.), surpassing *Cyprinus carpio* (common carp), a naturalized nonnative fish species introduced in the Mississippi River as early as the mid-1850s and the mass-dominant species across all UMRR study reaches since 1993 when standardized fish community monitoring began. Increasing mass dominance indicates invasive carp are superior competitors relative to the native fish community, and LTRM data are beginning to reveal decreases in native fish species in areas where invasive carp have gained in their mass dominance (Solomon and others, 2016; Chick and others, 2020; ch. G).

What LTRM Data Have to Say About Community-level Shifts in the Native Assemblage After Invasion

Invasive carp abundance and mass dominance in midwestern rivers has led to extensive research and analysis on effects to the existing fisheries community. For invasions such as this, it is essential that researchers understand the existing fish community before invasion to assess effects after invasion. It also is important to have accurate information about population densities and demographics (such as catch per unit effort [CPUE], length, and weight) of the invasive species in the invaded system. This fact makes the LTRM dataset (https://umesc.usgs.gov/data_library/fisheries/fish1_query.shtml, accessed 30 September 2020) vital and unusual; there are few multi-decadal datasets on major rivers that have collected standardized, comparable, pre- and postinvasion data for a nonnative species (Counihan and others, 2018). To date, LTRM staff have used this dataset to publish several studies concerning the effects of invasive carp. Irons and others (2007) published the first peer reviewed journal article to describe effects to native fishes in North America. Irons and others (2007) found statistically significant declines in body condition of native planktivores gizzard shad and bigmouth buffalo after establishment of invasive carp, specifically within the La Grange Pool of the lower Illinois River. Irons and others (2007) also described a significant decrease in bigmouth buffalo abundance (fig. I3).

Pendleton and others (2017) updated the work of Irons and others (2007) with another 7 years of data and reported the decreases in abundance of bigmouth buffalo observed by Irons and others (2007) had continued and gizzard shad had also become significantly less abundant. Both publications made extensive use of LTRM data and were led by LTRM fish component specialists. Another long-term river monitoring program focused on the Illinois River, the Long-Term Survey and Assessment of Large River Fishes in Illinois (LTEF), provided complimentary data for these analyses. Phelps and others (2017) expanded analysis of effects on gizzard shad and bigmouth buffalo body condition and CPUE to include every LTRM study reach. The three upper study reaches where invasive carp are at low numbers saw no significant change in mean CPUE for gizzard shad or bigmouth buffalo whereas the three lower invaded reaches saw CPUE of both species decrease and remain suppressed. Phelps and others (2017) also found a decline in body condition of gizzard shad and bigmouth buffalo in the Open River Reach while silver carp body condition remained consistent. Importantly, Love and others (2018) determined that gizzard shad body condition can recover when invasive carp are removed from the system.

In addition to analyzing effects on native planktivores, LTRM staff have also assessed the effects of invasion on the entirety of the fish community. Using LTRM data from the La Grange Pool and led by LTRM fish component specialists, Solomon and others (2016) investigated changes in the existing fish community pre- and postestablishment on the La Grange Pool and found significant differences in the fish communities after establishment of invasive carps (fig. I4). Specifically, most recreationally valuable sportfishes and commercially valuable Catostomids (buffalo spp., suckers) were less abundant postestablishment while less valuable *Lepisosteus* spp. (gar spp.), *Amia calva* (bowfin), *Ctenopharyngodon idella* (grass carp), *Notropis atherinoides* (emerald shiner), and *Lepomis cyanellus* (green sunfish) were more abundant postestablishment.

Chick and others (2020), using LTRM data, took a more system-wide approach and incorporated all six study reaches to analyze fish CPUE and water quality data; three northern reaches where invasive carp are at low numbers acted as a control to study the effects of invasive carp on the three lower invaded reaches. Results indicate negative effects of invasive carp on the native fish community and adult sportfish whereas trends in sportfish in noninvaded reaches were generally positive. The authors hypothesized that the mechanism may be competition for plankton resources between juvenile sportfishes and invasive carp. They also speculated that while the 20-year warming trend observed in all six study reaches is expected to increase recruitment success of sportfishes, the opposite is true in invaded reaches, indicating invasion is restricting sportfish recruitment.

More specific to invasive carp effects on water quality, DeBoer and others (2018) summarized 22 years of LTRM water quality data along with LTEF/LTRM fisheries data. Using the year 2002 as a change point in invasive carp biomass (as indicated by LTEF CPUE results) and using LTRM chlorophyll *a* as a proxy for phytoplankton biomass, DeBoer and others (2018) found a significant decrease post-2002 of chlorophyll *a* in main-channel and side-channel habitats, but not backwaters (fig. I5). Phytoplankton biomass was also negatively, although not strongly, associated with invasive carp biomass.

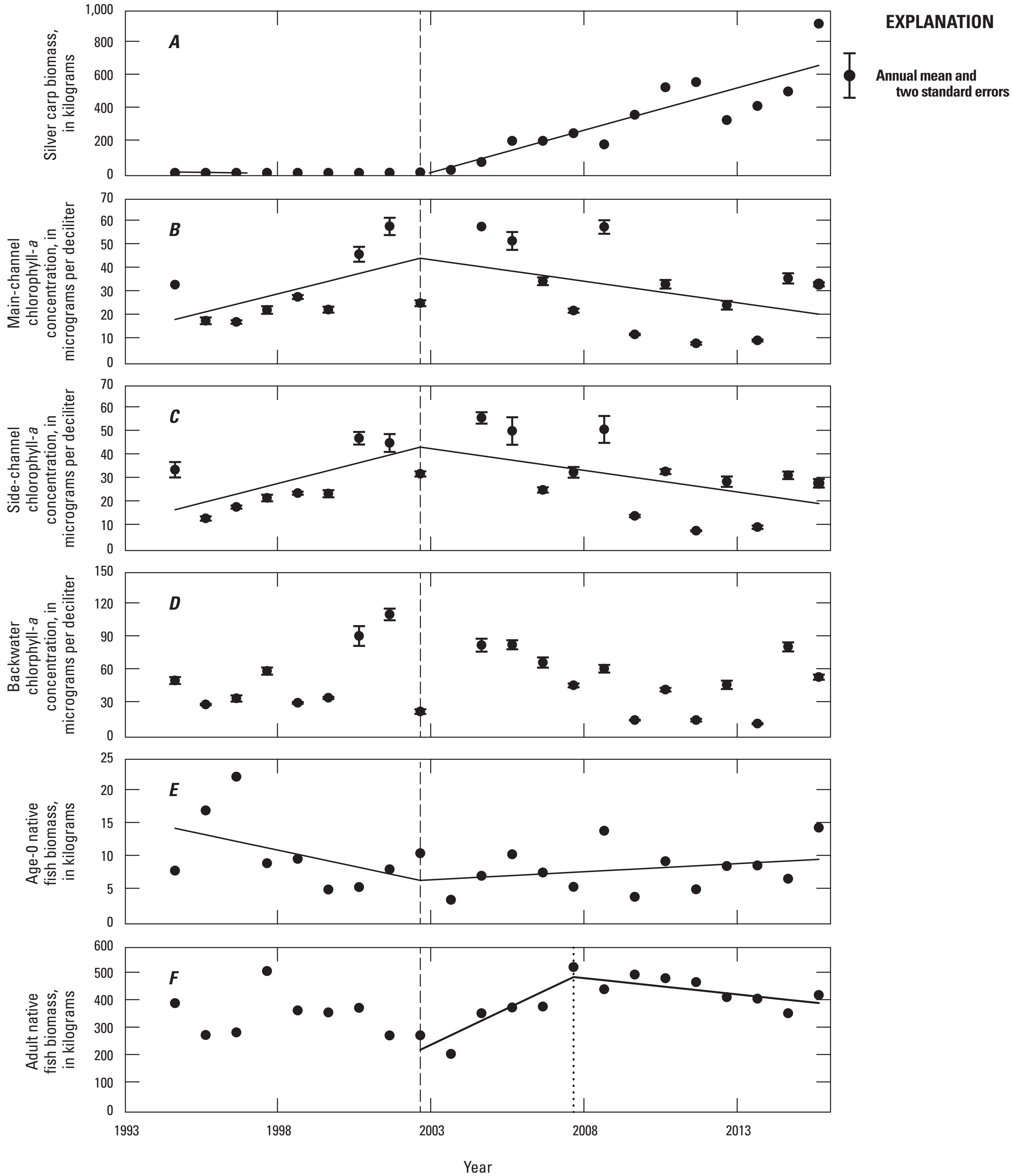


Figure I3. La Grange Pool *A*, biomass; *B*, chlorophyll *a* concentration (as an index of phytoplankton biomass) from main-channel border habitat strata sampled during summer; *C*, chlorophyll *a* concentration from side-channel border habitat strata sampled during summer; *D*, chlorophyll *a* concentration from connected-backwater habitat strata sampled during summer; *E*, age-0 native fish biomass; and *F*, adult native fish biomass. Solid dots represent annual means, vertical line intervals represent ± 2 standard errors for the annual means, solid black lines represent trends pre- and post-invasion by bigheaded carp, and the vertical dashed line represents the pre- versus post-invasion period. Graph from DeBoer and others (2018) and slightly modified by the authors.

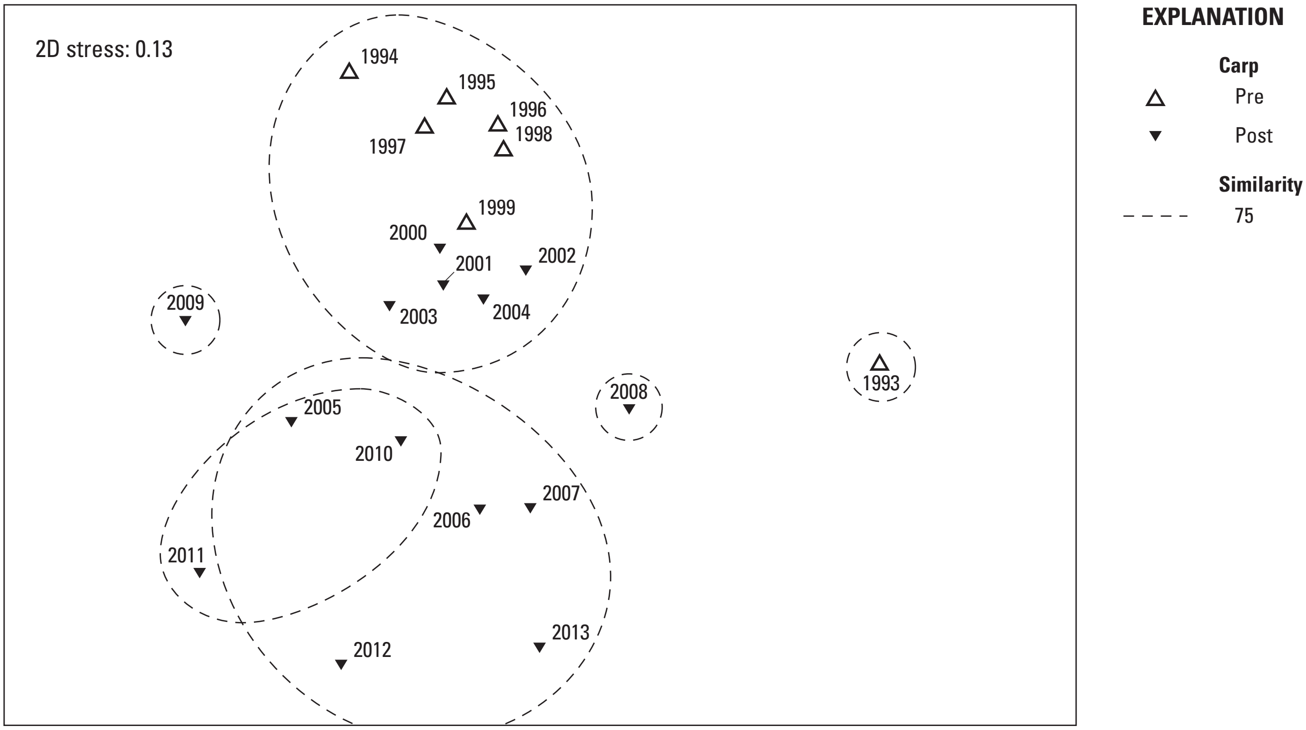


Figure I4. Nonmetric multidimensional scaling plot from poolwide day electrofishing of La Grange Pool pre- and post-*Hypopthalmichthys* spp. (invasive carp) invasion (Solomon and others, 2016). In this graphical portrayal, points in the plot closer to one another are more similar in their community structure; points farther away are more dissimilar. Points with greater than 75 percent similarity are presented within each dashed boundary. Each boundary is greater than 25 percent dissimilar in its community structure than other boundaries represented.

LTRM fish component data have value and relevance beyond the UMRR, and as a public trust data source on one of the world's great rivers, these data resources have been used repeatedly by scientists and managers across the United States and even worldwide. Outside of the LTRM, substantial effort has been put into the study, management, and control of the invasive carp invasion, and a full accounting of this effort is beyond the scope of this document. However, LTRM staff have assisted with other efforts. Highlights of those efforts (note: this is not a complete listing) include several publications using LTRM data and (or) personnel to assess the following:

- fishing-induced population collapse of silver and bighead carp on the Illinois River (Tsehays and others, 2013);
- recruitment overfishing (taking more than are naturally produced) of silver carp (Seibert and others, 2015);
- population estimates of silver carp indicating perhaps the greatest density of wild silver carp in the world (Sass and others, 2010);
- diet overlap between invasive carp and native fishes (Sampson and others, 2009);
- establishing that existing zooplankton models on the Great Lakes potentially significantly underestimated microzooplankton available and, accordingly, greatly underestimated prey resources available to invading invasive carp (Thomas and others, 2017);
- relating changes to the plankton community of the Illinois River to decreases in larger-bodied zooplankton species and increases in smaller-bodied zooplankton species postestablishment of invasive carp (Sass and others, 2014); and
- invasive carp recruitment is driven by hydrology with 4 major spawning events (2000, 2003/2004, 2007/2008, and 2014) in the 15 years following establishment on the La Grange Pool and accounting for 99.5 percent of all age-0 invasive carps collected (Gibson-Reinemer and others, 2017; see fig. I6).

Multiagency efforts throughout the invaded range of invasive carp, specifically within and bordering the state of Illinois, have been at the forefront of the efforts to keep invasive carp out of the Great Lakes to protect a multi-billion dollar fishery and prevent invasive carp from further expanding their range. LTRM personnel have been active in supporting these efforts as well; for example, by assisting with sound, bubble, and light barrier research (see Taylor and

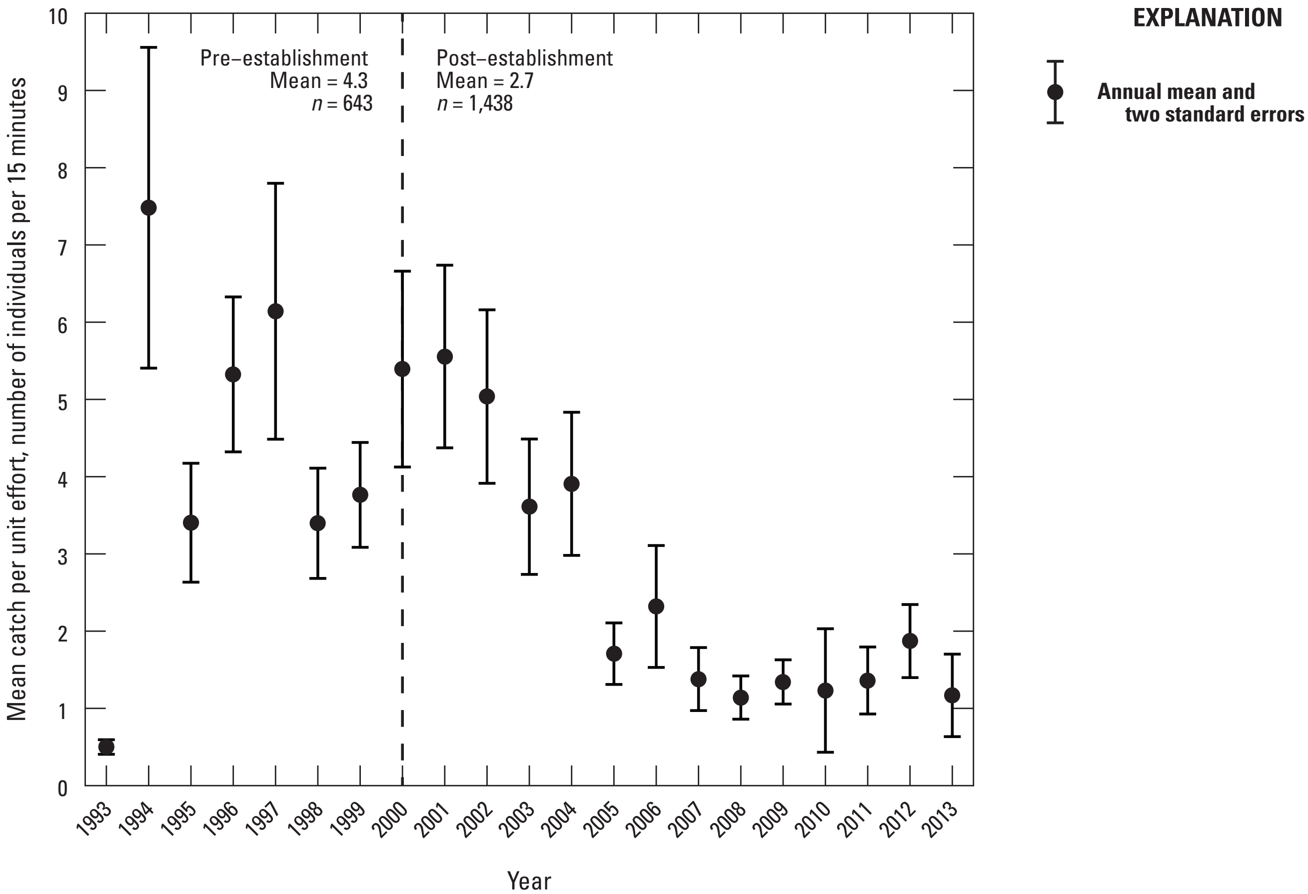


Figure I5. *Ictiobus cyprinellus* (bigmouth buffalo) abundance pre- and post-invasive carp invasion of the La Grange Pool from Pendleton and others (2017). Solid dots represent annual means, vertical line intervals represent ±2 standard errors for the annual means, and the vertical dashed line represents the pre- versus post-invasion period. Note: 1993 is likely artificially low due to highly compromised annual sampling allocations associated with a historic flood event.

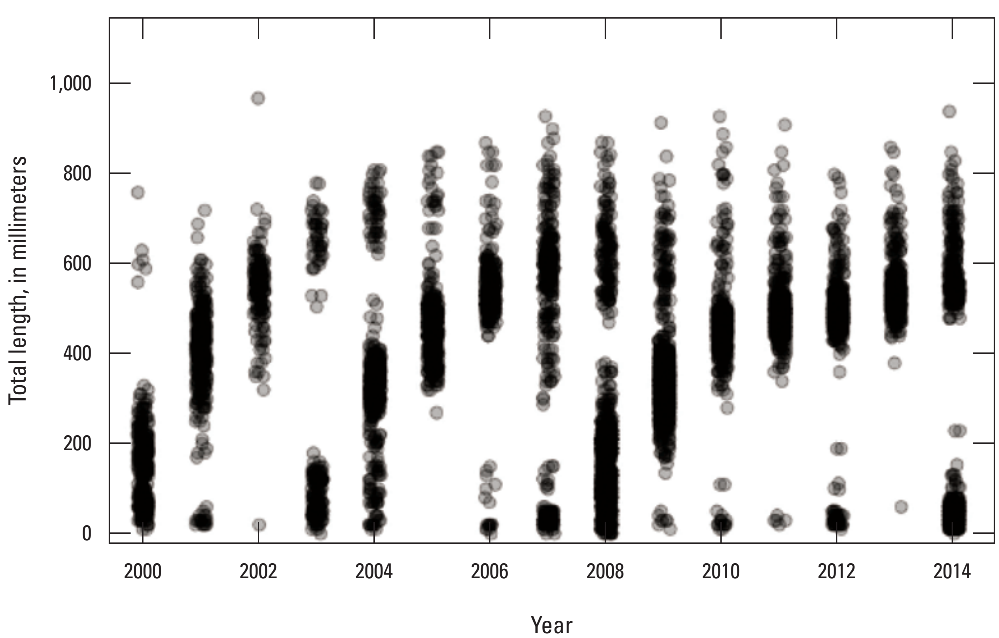


Figure I6. Total lengths of individual *Hypopthalmichthys* spp. (invasive carp) from Upper Mississippi River Restoration Long Term Resource Monitoring element sampling on La Grange Pool. Each symbol indicates an individual fish, difference in shading due to density of observations (higher density darker). Large year classes of invasive carp generally corresponded to flood pulses of near-flood stage in magnitude occurring between mid-June and mid-July. Year 2007 was the exception as late season flood pulses producing invasive carp were either less than flood stage (August) or much greater than flood stage (September) in magnitude. Figure from Daniel K. Gibson-Reinemer, used with permission.

others, 2005; Ruebush and others, 2012; Vetter and Mensinger, 2016; and Vetter and others, 2017).

Beyond LTRM, abundant sources of information are available through Invasive Carp Regional Coordinating Committee (ICRCC; http://www.invasivecarp.us/) and its action arm, the Monitoring and Response Working Group (MRWG; http://www.invasivecarp.us/early-detection-monitoring-assessment.html). In addition, the Upper Mississippi River Invasive Carp Team (UMRICT; http://www.micrarivers.org/invasive-carp-plans-and-reports/) is working in Pools 16–19 to control the spread of invasive carp northward using incentivized commercial fishing efforts, like MRWG, as well as the study of early life history. The scientific community dedicated to study and management of invasive carp is growing across the midwestern United States, and the LTRM remains a highly valued and foundational resource for several reasons. Specifically, LTRM houses a unique dataset at a spatial and temporal scale that cannot be gained by other modes of study, LTRM data have been used in a growing number of scientific products published in peer reviewed journals, and LTRM has a network of field stations with the ability to provide logistic/intellectual support and personnel that can regularly be called upon for their knowledge, expertise, and field capabilities.

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Chapter J: Summary and Synthesis

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Chapter J of

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Overview

We have provided a broad assessment of when, where, and how the Upper Mississippi River System has changed over the last 26 years. The selected indicators of ecosystem health reflect the Upper Mississippi River Restoration (UMRR) program's emphasis on understanding the ecology of the river in the context of river management and habitat restoration; the indicators were selected based on ecological understanding of the Upper Mississippi River System and partnership input. The data reported here span an unusual spatial and temporal extent for large river studies: 26 years and approximately 1,500 kilometers of the 2,092 kilometer long Upper Mississippi River System. The results provide diverse insights into the spatial and temporal variability in the Upper Mississippi River System, changes observed from 1993 through 2019, and the implications for understanding and restoring this river system.

Changes within the Upper Mississippi River System were observed at multiple scales. The most widespread long-term trend was the increase in discharge observed throughout the Upper Mississippi River System. Discharge is a fundamental characteristic of river systems, and this change has broad implications for habitat conditions and riverine biota. There is evidence of changes in the river geomorphology including loss of depth in backwater areas because of ongoing net sedimentation and accumulation of new floodplain land areas. Floodplain forest cover decreased in all floodplain reaches, except the Unimpounded Reach where forest cover increased. The Upper Mississippi River System remains a highly eutrophic system based on water quality indicators. Total phosphorus and suspended solids concentrations decreased in most of the study reaches but remain high relative to water quality criteria, and fewer significant trends in nitrogen concentrations were detected. The prevalence of aquatic vegetation (the term broadly used for many life forms like submersed, emergent, and rooted-floating leaved species), and especially submersed aquatic vegetation (SAV), increased in the Upper Impounded Reach while little change was observed in the Lower Impounded, Unimpounded, or Illinois River reaches where aquatic vegetation remains scarce. The fish indicators documented that some aspects of the fish community have not changed significantly (for example, species diversity and total catch) whereas others have exhibited important changes (for example, forage fishes and invasive bigheaded carps). In this chapter, we discuss likely associations among some of these changes, the importance of long-term data for understanding these changes, and the implications for river restoration and management.

Connections Among Observed Changes in the Upper Mississippi River System

Spatial gradients are strong in the underlying physical structure of the Upper Mississippi River System (see fig. A1 for locations of study reaches, pools, and tributaries) that contribute to the diversity of observed long-term changes among study reaches. Gradients in the rivers' physical template result from interactions of the underlying geology and geomorphology, navigation infrastructure (such as locks/dams, wing dams, and closing dams), and prevalence of levees (ch. A; ch. D; De Jager and others, 2018). Levee distribution across the Upper Mississippi River System displays an especially strong gradient; little floodplain area was behind levees in the northern part of the system and at least 50 percent of floodplain area was behind levees in the southern part.

Differences in geology, geomorphology, navigation infrastructure, and levees resulted in a strong north-south difference in connectivity between the main channel and the floodplain. In particular, the abundance of off-channel, connected lentic habitats decreased from upstream to downstream along the length of the Upper Mississippi River (De Jager and others, 2018). These off-channel lentic areas can support abundant aquatic vegetation (ch. F and ch. H) and specific lentic fish communities (ch. G, "Lentic Fishes") including several popular recreational species (ch. G, "Recreationally Valued Native Fishes"). The study reaches with an abundance of well-connected, off-channel areas had significant increases in the abundance of aquatic vegetation and water clarity in those areas (ch. E and ch. H). In contrast, river reaches where most of the aquatic area was flowing-channel environments had much less area suitable for SAV (Carhart and others, 2021); however, these flowing-channel environments provide critical habitat for important native fish species (ch. G, "Lotic Fishes;" table G3).

The navigation pools of the Illinois River have aquatic area distributions that are different from the navigation pools of the Upper Mississippi River (De Jager and others, 2018). In general, the Illinois River contained a large amount of shallow lentic area that was geomorphically suitable for aquatic vegetation. Yet aquatic vegetation was scarce in the Illinois River, likely because of a combination of high total suspended solids concentrations (low light availability), water level fluctuations, seedbank viability, and herbivory (Sass and others, 2017).

Thus, the changes observed over the last two and a half decades and the spatial variability in those changes reflected these geomorphic and structural differences across the Upper Mississippi River System. Assessing where and how these aquatic habitats continue to change into the future will be important for understanding associated changes in aquatic vegetation, water quality, and fish communities.

Two indicators, new landform surface area and backwater bed elevation change, directly addressed geomorphic changes that can alter local physical conditions such as water depth, hydrologic connectivity, water velocities, and water residence time. New landform surface area in the form of crevasse deltas, tributary deltas, impounded deltas, and bar-tail limbs increased steadily from 1989 to 2010 in the Upper and Lower Impounded Reaches (Pools 3–26 ; fig. C2). Most new landform surface area was detected in the Upper Impounded Reach (Pools 3–13). Results from a study in Pools 4 and 8 indicate an overall increase in bed elevations in backwater lakes since 1997 and indicate that many backwater lakes have become shallower (table C1) and can be expected to continue to do so. Loss of backwater depth has been a long-term concern for the Upper Mississippi River System and will likely remain so for the foreseeable future. Loss of deep, lentic, off-channel areas may adversely affect overwintering habitat for some lentic fish species (Bodensteiner and Lewis, 1992; Raibley and others, 1997; but see Gutreuter, 2004). The resulting increase in shallow areas also may affect the distribution and abundance of aquatic vegetation, with increases in some habitat types and losses in others. A third component of geomorphic change that was not assessed in this report (but is an ongoing management concern for the Upper Mississippi River System) is the erosion of islands and river shoreline. This erosion can increase connection of backwater resulting in greater sediment input and flow velocities, both of which can affect habitat suitability.

Discharge is a fundamental driver of the structure and function of rivers (Poff and others, 1997). Our results indicated that multiple aspects of discharge increased across the entire Upper Mississippi River System and the timing of seasonal flooding changed (ch. B). An increase in discharge in a given river reach can increase the magnitude and duration of floodplain inundation, hydraulic connection among floodplain water bodies, water velocities, flushing rate within floodplain water bodies, and depth throughout the system (for example, Tockner and others, 2000; Pongruktham and Ochs, 2015; Burdis and Hirsch, 2017; Van Appledorn and others, 2021). Several factors such as climate, drainage basin land use, and river engineering likely affect the magnitude of, and seasonal patterns in, Upper Mississippi River System discharge, but the effects of these drivers vary widely among watersheds (Kochendorfer and Hubbart, 2010; Alexander and others, 2012; Frans and others, 2013; Mallakapour and Villarini, 2015). Thus, the relative importance of, and interactions among, the multiple factors affecting annual variation and long-term trends in discharge remain only partially understood.

The duration of flooding during the growing season plays a key role in structuring floodplain plant communities within the Upper Mississippi River System (De Jager and others, 2018). The increased duration of high discharge may have contributed to observed forest loss in the impounded parts of the Upper Mississippi River and Illinois Rivers (fig. D4). Statistical models of tree mortality predict increased mortality with increasing flood duration (De Jager and others, 2018), and a recent simulation modeling study indicated that changes in water levels in the Upper Mississippi River System, along with effects of an invasive insect (*Agrilus planipennis* [emerald ash borer]), have the potential to contribute to forest loss in the future (De Jager and others, 2019). Furthermore, forest loss could be permanent in some locations even if and after discharge conditions become more suitable for woody plant communities because of invasion of canopy gaps by invasive plants such as *Phalaris arundinacea* (reed canary grass) and *Humulus japonicus* (Japanese hops; Thomsen and others, 2012).

Discharge is one of the main drivers of variation in water quality in rivers (Hirsch and others, 2010; Moatar and others, 2017). For example, total suspended solids (TSS), total phosphorus (TP), and total nitrogen (TN) tend to increase with discharge in the Upper Mississippi River System and high discharge is generally associated with high sediment and nutrient input (Smits and others, 2019; Marinos and others, 2020). However, despite the long-term increase in discharge, TP and TSS concentrations decreased in several of the study reaches (fig. E1). Decreases in TSS and TP concentrations were even more widespread (five out of six reaches for TSS; four out of six for TP) when we considered flow-normalized trends. Flow-normalized values account for how concentration changes at a given discharge over time, thus decreases in flow-normalized values indicated that less TSS and TP are delivered per unit discharge now than in the past. This indicates that the changes in TSS and TP were likely because of improvements in landscape land use practices (Kreiling and Houser, 2016) or processes in the river such as an interaction with vegetation (Bouska and others, 2020; ch. H) and were not simply a result of dilution by higher flows. Nitrogen concentrations and fluxes remained high through 2019, and there were upward trends in three LTRM study reaches. However, significant downward trends were observed in the La Grange Pool of the Illinois River. This indicates that the TN concentration at a given discharge remained stable and that the main drivers of TN concentrations in the Upper Mississippi River System likely differed from those of TP and TSS (Robertson and Saad, 2021).

Recent decreases in chlorophyll *a* concentration (CHL) may have been because of higher discharge (ch. B) because the relation was significantly negative between CHL concentration in the main channel and backwaters with discharge (ch. E). This is a common pattern in large rivers generally (Houser, 2016 and references therein) and is consistent with other observations in the Upper Mississippi River (Manier and others, 2021; Jankowski and others, 2021). On short time scales such as during a single storm event, high discharge can reduce CHL through dilution, downstream transport, or scouring (Uehlinger, 2006). Over longer time-scales sustained high discharge in the main channel is typically associated with increased sediment loads that reduce light availability and can limit photosynthesis (Ochs and others, 2013). In addition to discharge, several other factors may have affected CHL such as decreases in TP concentrations (fig. E1), the increase in aquatic vegetation (chs. F and H), and the proliferation of bigheaded carps, which are efficient filter feeders (ch. I; De Boer and others, 2018).

During low discharge, water exchange among channels and backwaters is generally reduced; this reduced inflow of water from the main channel where oxygen concentrations are sometimes greater, especially during winter, can result in lower backwater dissolved oxygen concentrations (Johnson and others, 1998). Across the period of record, no relation between dissolved oxygen concentration and discharge was significant. However, the greater prevalence of low dissolved oxygen conditions in backwaters (hypoxia) did coincide with relatively low discharge between 2005 and 2010, especially in Pools 4 and 8 (ch. E). This low discharge also coincided with the highest rates of increase in SAV abundance in Pools 4, 8, and 13 (ch. F). The SAV can increase oxygen demand either through growth and senescence or by facilitating free-floating plant and filamentous algae accumulation that can reduce light penetration and contribute to a high oxygen demand at night (Houser and others, 2013; Giblin and others, 2014). The greater winter discharge near the end of the period of record may have contributed to recent declines in the prevalence of low dissolved oxygen in backwaters of Pools 4, 8, and 13. Although higher winter discharge can increase oxygen concentrations in backwaters, it can also increase water current speed and reduce winter water temperatures in backwaters (Johnson and others, 1998) and reduce or eliminate their suitability as overwintering habitat for some fish species (Sheehan and others, 1990). The net effect of increased discharge on overwintering habitat has not yet been explicitly quantified, but the observed increase in lentic fish species that rely on backwaters for overwintering habitat (fig. G3) indicates that, as of 2019, increased winter discharge has not reduced the amount of suitable backwater habitat available to cause a detectable reduction in the populations of these fishes.

The response of aquatic vegetation to discharge included some unexpected ecological responses. Submersed and emergent aquatic vegetation were resilient to higher discharge conditions during the last 10 years of the period of record (and the associated greater current velocity and deeper water; ch. F; Burdis and others, 2020). In addition, wildrice expanded into deeper waters, which demonstrated this species' adaptability to variable conditions. However, SAV is sensitive to substantial water level fluctuations (Carhart and others, 2021), so increased discharge could lead to greater water level variations and affect SAV and likely other aquatic vegetation types as well.

Increased water clarity in the Upper Impounded Reach likely contributed to the increased prevalence of SAV and the subsequent resilience of aquatic vegetation to increases in discharge (ch. H; Bouska and others, 2020; Burdis and others, 2020; Carhart and others, 2021). Increased water clarity in much of the Upper Impounded reach from 1993 to 2019 was likely because of a combination of a decrease in TSS input from tributaries (ch. E), reduced *Cyprinus carpio* (common carp) biomass, and management and restoration actions (ch. H). Increased vegetation prevalence likely further contributed to increased water clarity by reducing water velocity, enhancing sedimentation, and reducing resuspension. One study indicated that improvements in water clarity to less than 20 nephelometric turbidity units (roughly equivalent to 25 milligrams per liter TSS in the Upper Mississippi River System (Upper Mississippi River Conservation Committee [UMRCC], 2003) support abundant *Vallisneria americana* Michx. (watercelery; Kreiling and others, 2007). During the greatest increase in SAV prevalence in the Upper Impounded Reach, turbidity was less than this 20 nephelometric turbidity units threshold; in the backwaters and impounded areas of Pools 4 and 8, but not Pool 13, turbidity remained less than that threshold as of 2019. In Pool 13, the increased turbidity in the second half of the period of record was coincident with a decrease in the prevalence of SAV. In the Pool 26, Open River Reach, and La Grange Pool, all aquatic vegetation life forms were still scarce as of 2019 (ch. F; De Jager and Rohweder, 2017) largely because of the combined effects of geomorphology, water clarity, and the magnitude of water level fluctuations (Carhart and others, 2021), although in the La Grange Pool of the Illinois River, additional factors such as herbivory may have contributed (Sass and others, 2017).

Some changes in the native fishes of the Upper Mississippi River System coincided with changes in aquatic vegetation (ch. G; DeLain and Popp, 2014; Giblin, 2017) and invasive fishes (Bouska, 2020). Observed increases in lentic fishes and recreationally valued fishes in the upper three Long Term Resource Monitoring (LTRM) element study reaches may be because of increased water clarity and prevalence of SAV over the period of record; for example, improved water clarity can affect foraging efficiency and predation rates while aquatic vegetation supports foraging resources and refuge from predators. Alternately, decreases in mass per unit sampling effort for native fish community, recreationally valued species, and forage fishes observed in the Pool 26, Open River Reach, and La Grange Pool likely reflect competition among invasive bigheaded carps and native fishes in these regions. For example, competition with bigheaded carps for plankton resources can reduce growth rates of larval fishes (Fletcher

	Upper Mississippi River			Illinois River
Rank	Upper Impounded	Lower Impounded	Unimpounded	Lower Illinois
More important	Lentic area	Lentic area	Lentic area	Total suspended solids concentration
	Lotic structure	Lotic structure	Lotic structure	Lentic area
Less important	Aquatic vegetation diversity	Aquatic vegetation diversity	Leveed area	Lotic structure
	Total suspended solids concentration	Leveed area	Aquatic vegetation diversity	Leveed area
	Lateral connectivity—leveed area	Total suspended solids concentration	Total suspended solids concentration	Aquatic vegetation diversity

EXPLANATION

- Existing condition substantially deviates from desired condition; may merit action to improve
- Existing conditons deviate from desired conditions; may merit actions to improve
- Existing condition is near desired condition; may merit actions to improve or maintain
- Existing condition is near desired condition; may merit actions to maintain
- Existing condition meets desired condition; continued monitoring and management may be needed

Figure J1. Summary of ratings of five indicators of ecosystem structure and function for river reaches of the Upper Mississippi River System (modified from figure ES-1 in McCain and others [2018]). The indicators are listed in decreasing order of importance as determined using a paired comparison analysis conducted by participating agencies for each river reach. Different colors refer to the proximity of the measured score of each indicator (De Jager and others, 2018) to the desired conditions of management agencies within the Upper Mississippi River System. See De Jager and others (2018) for details on measured scores for each indicator and McCain and others (2018) for methods used to define the condition descriptions and rank indicators. Lentic area refers to the total area of water with little or no current in river reaches. Lotic structure refers to the total area of flowing-water channels that contain structure, such as wing dams and closing dams.

and others, 2019) and may contribute to population declines in native species (Chick and others, 2020).

Other changes in the fish indicators may be at least in part because of long-term changes in discharge; for example, increased discharge may be contributing to the decrease in abundance of forage fishes. Forage fishes are small bodied and often depend on ideal conditions to yield large year classes. Ongoing changes in hydrology may disrupt spawning, displace fish from refugia during the cold season, limit food availability, or increase susceptibility to predation, among a host of other possible effects.

Management Implications

In addition to contributing to our basic understanding of how the Upper Mississippi River System has changed through time, the ecological indicators in this report can be used in setting management priorities and targets. McCain and others (2018) evaluated a series of indicators of landscape-scale ecosystem structure and function for the Upper Mississippi River System (from De Jager and others, 2018) in the context of the priorities of the management agencies along the river system (fig. J1). In doing so, they assessed whether each indicator in each of the four floodplain reaches of the Upper Mississippi

River System met the desired conditions of the management agencies. The results presented in the current report provide additional granularity and a long-term context for management decisions that stem from the ecosystem management and restoration framework presented in McCain and others (2018); for example, the indicators pertaining to levees, floodplain forests, suspended solids, and aquatic vegetation are directly related to stated priorities of the UMRR management agencies (fig. J1). Further, because fish communities are associated with certain habitat conditions (Ickes and others, 2014), the long-term changes in fish communities and differences among study reaches presented in this report provides important information to inform decisions based on aquatic habitat distributions presented in De Jager and others (2018) and McCain and others (2018). The priority placed on the conditions represented by each of these indicators varies across the Upper Mississippi River System as does the proximity of each indicator to the desired conditions of management agencies (fig. J1; McCain and others, 2018). Thus, many of the management implications of our results are dependent on the location along the Upper Mississippi River System and likely vary among natural resource agencies. However, there remain some general management implications that are described here.

The geomorphic changes reported in ch. C have implications for physical and ecological aspects of the Upper Mississippi River System; for example, new landform areas and increasing backwater bed elevation changes alter patterns of exposed sediment, hydrologic connectivity, flow velocities, and residence times within various aquatic habitats. Such physical changes affect the amount and distribution of habitats suitable for any specific organism. This information on the rate the river is changing and where those changes are occurring can support decisions regarding the location and design of habitat restoration projects.

Although TP concentrations have decreased in much of the system, the Upper Mississippi River System remained highly eutrophic, and river and floodplain restoration actions would need to work within the constraint of high TP, TN, and TSS concentrations. TP and TN concentrations exceeded the water quality criteria in nearly all the study reaches, and TSS concentrations were greater than or similar to the established maximum concentration for sustaining vegetation growth in three of the six study reaches (fig. E1). Major changes in these indicators are not likely to be achieved with management actions within the river and its floodplain, but rather would need to be achieved by the broad scale changes in the watershed land use in the Upper Mississippi River System basin (Santhi and others, 2014a; Santhi and others, 2014b).

The long-term increase in water clarity may lead to an increase in areas that are suitable for SAV. Carhart and others (2021) found that the area suitable for aquatic vegetation (based on water clarity, geomorphology, and growing season water level fluctuations) in most pools of the Upper Impounded Reach, Pool 19 of the Lower Impounded Reach, and Peoria Pool of the Illinois River appear to be relatively sensitive to reductions in TSS. This indicates these pools have areas likely to respond to management actions, especially if any other inhibiting factors can be addressed (such as herbivory in the Illinois River). Recent decreases in the water clarity and aquatic vegetation of Pool 13 indicate that the resilience of vegetation to recent years of higher flows may be less in Pool 13 than in Pools 4 and 8. Ongoing monitoring in Pool 13 will provide important information regarding the future trajectory of vegetation and water quality and their responses to a habitat rehabilitation and enhancement project being planned for lower Pool 13. More generally, the fact that SAV has been consistently scarce or absent in much of the Lower Impounded Reach, Unimpounded Reach, and Illinois River (De Jager and others, 2018) highlights an ongoing river restoration challenge.

The fish community in southern parts of the Upper Mississippi River System is now dominated by invasive bigheaded carps (ch. G) and in those areas a diverse array of effects on the ecosystem have been observed (ch. I). Bigheaded carps appear to have increased the productive capacity of the fish community in Pool 26, but this increase in total productivity is associated with significant increases in bigheaded carp production along with a small yet significant decrease in the total mass of the native fish community. In contrast, recent increases in mass dominance of bigheaded carps in the La Grange Pool have been associated with a strong decrease in native fish mass, indicating native fish displacement and suppression by bigheaded carps. Associated changes in the fish community include decreases in native species abundance and body condition (Irons and others, 2007) and decreases in the abundance of sport fishes (Chick and others, 2020). Furthermore, dominance by these nonnative species may act as a driver of regime change in the fish community (Bouska, 2020). Once a threshold of relatively low bigheaded carp biomass is passed, bigheaded carp dominance may persist as a result of reinforcing feedbacks and be difficult to reverse. Notable decreases in abundance of small-bodied fishes of different trophic/functional feeding guilds in reaches with bigheaded carp dominance indicate competition for plankton resources as the principal mechanism contributing to decreases in small-bodied fishes (Bouska, 2020). Understanding these effects can inform natural resource agencies when considering efforts to prevent establishment of these invasive fishes in new areas and to reduce their biomass where they have established.

Forage fish are an important component of the Upper Mississippi River System fish community, occupying an intermediate role in the food web and therefore affecting the productive capacity of lower and higher trophic levels. Abundance of forage fishes has decreased notably in several study reaches (ch. G). Habitat limitation in the lower study reaches is a possible explanation because the upper three LTRM study reaches are considered to have higher quality and more varied habitats than the three lower LTRM study reaches (De Jager and others, 2018) and are the only LTRM reaches with stable forage fish biomass. If habitat conditions were improved in the lower reaches, it is possible that a more diverse and abundant forage base could develop. The proliferation and persistence of bigheaded carps throughout the lower reaches of the Upper Mississippi River System have also likely affected the forage fish indicator through reduced fitness and abundance of *Dorosoma cepedianum* (gizzard shad). It remains to be seen whether the higher diversity of habitat and greater functional diversity within the fish communities (Bouska, 2018) of the upper study reaches can prevent establishment of bigheaded carps or reduce the effects of such establishment on the fitness of native fishes that has been observed in the La Grange Pool.

The Value of Long-term Data

Long-term data provide important information for the restoration and management of a healthier and more resilient ecosystem. Long-term observations make it possible to detect important associations among changing aspects of the river. Two examples that illustrate this in detail in this report are the long-term changes in, and interactions among, discharge, water clarity, vegetation, and fish (ch. H), and the ecological changes associated with the proliferation of invasive bigheaded carp (ch. I). Long-term data have also made it possible to detect important changes in the river that would otherwise be unnoticed or unquantified, such as the long-term decreases in abundance of forage fishes, which are an important intermediary in the aquatic food web.

Understanding long-term ecosystem dynamics can aid in identifying effective restoration/management approaches. Trends in major resources and large-scale drivers can be indicative of impending regime shifts, particularly when nearing a threshold, and inform management actions taken to reduce the probability of a regime shift. Long-term changes in hydrology, geomorphology, and floodplain land cover (chs. B, C, and D) may lead to abrupt changes in the distribution and abundance of major resources if they cross critical thresholds (Bertoldi and others, 2009). Long-term changes in water clarity and vegetation in Pools 4, 8, and 13 (ch. H) may be an example of a regime shift with alternate regimes described as a clear water and abundant vegetation regime and a turbid water and sparse vegetation regime (Giblin, 2017; Bouska and others, 2020; Burdis and others, 2020). Similarly, the rapid increase in bigheaded carps and responses of the ecosystem in La Grange Pool (Illinois River) and Pool 26 (Upper Mississippi River) may be an example of a regime shift from a native to nonnative dominated food web (ch. I; Bouska and others, 2020; Bouska, 2020). For these ecosystem shifts, long-term data have provided important insights into the causes or consequences of these changes for diverse components of the ecosystem (chs. H and I). In addition to long-term monitoring, complementary research to better understand the mechanisms causing the observed changes can aid in identifying management options that improve adaptive capacity, bolster resilience of desired conditions, or erode resilience of undesired conditions. Forecasting future changes in controlling variables, such as discharge, can inform whether desired conditions are likely to persist and inform management decision making.

Long-term monitoring has provided insights into the effects of river restoration that are otherwise unobtainable. For the Upper Mississippi River System, the LTRM data provide extensive information regarding the broad context within which management decisions are made and restoration actions are taken. The resulting understanding of spatial differences and long-term trends in water quality, vegetation, and fisheries can inform restoration priorities, selection of restoration sites, and evaluation of the effectiveness of the restoration projects. When restoration projects are constructed within areas with long-term, ongoing monitoring, information regarding the effects of those efforts not only provide information about those specific restoration projects but inform the selection and design of future projects (Langrehr and others, 2007; Gray and others, 2011; Carhart and De Jager, 2018; Drake and others, 2018). Long-term observation can also provide important information regarding the habitat associations and preferences of riverine biota that can be incorporated into the design of projects intended to rehabilitate and enhance riverine habitat (Ickes and others, 2014; Carhart and De Jager, 2018). Lastly, although the UMRR is focused on understanding, monitoring, and restoring the main stem of the Upper Mississippi River System, the program also collects long-term water quality data at the mouths of more than 25 tributaries (ch. E). These data not only provide useful information on how changes in the watershed affect conditions in the Upper Mississippi River System but also can aid partners in assessing whether and how watershed management efforts have affected nutrient loads and water quality in those rivers (Kreiling and Houser, 2016).

Looking Forward

Evaluating ecosystem status is most effective when societal values concerning the system have been specified and indicators that reflect those values have been identified and measured. At the time this report was completed, there were no specific, quantified targets for most components of the Upper Mississippi River System ecosystem. The exceptions are limited to the water quality indicators where external standards exist for some indicators (specifically, nutrient concentrations developed by the U.S. Environmental Protection Agency or states). Qualitative statements regarding management goals exist (U.S. Army Corp of Engineers, 2011), and recent progress has been made regarding some aspects of desired future conditions. McCain and others (2018) combined quantitative assessments of current conditions by De Jager and others (2018) with the expert opinion of natural resource managers and restoration professionals of the Upper Mississippi River System to produce a qualitative assessment of how closely select quantitative systemic indicators of ecosystem condition were to a “desired” future condition (fig. J1).

The Upper Mississippi River System is a spatially complex and temporally dynamic system, and continuous change is expected as the river continues to adapt to the altered hydrology caused by ongoing changes in climate, watershed land use, levee distribution, and navigation modifications to the river. The results presented in this report describe the quantitative changes observed from 1993 to 2019 and the insights gained therefrom. Ongoing long-term monitoring provides the valuable data needed to accurately assess the state of the system, determine drivers of changes, and inform river management and restoration. A useful next step for the restoration and management of the Upper Mississippi River System would be to describe the desired future conditions for the Upper Mississippi River System so that future status and trends assessments can be based on specific benchmarks for the system. The quantitative descriptions of the river presented in this report and De Jager and others (2018) and the qualitative assessment of the river condition based on the best professional judgement of natural resource and restoration professionals (McCain and others, 2018) provide a foundation for successfully identifying those desired conditions.

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