POOL 11 ISLANDS (MUD AND SUNFISH LAKES) HABITAT REHABILITATION AND ENHANCEMENT PROJECT INSPECTION OF COMPLETED WORKS 2017

I. PROJECT:
Pool 11 Habitat Rehabilitation and Enhancement Project (HREP)

II. AUTHORITY:
Upper Mississippi River Restoration (UMRR) Program

III. LOCATION:
Pool 11, Upper Mississippi River Miles 583.0 - 592.0, Grant County, Wisconsin and Dubuque County, Iowa

IV. PREVIOUS REPORTS:
Reports listed below are posted at this website:
http://www.mvr.usace.army.mil/Missions/Environmental-Protection-and-Restoration/Upper-Mississippi-River-Restoration/Habitat-Restoration/Rock-Island-District/Pool-11-Islands/

U.S. Army Corps of Engineers, Rock Island District, Upper Mississippi River System, Environmental Management Program, Definite Project Report (R-13F) with Integrated Environmental Assessment, Pool 11 Islands Rehabilitation and Enhancement, September 2001.

U.S. Army Corps of Engineers, Rock Island District, Operation and Maintenance Manual, Pool 11 Islands, Sunfish Lake and Mud Lake Habitat Rehabilitation and Enhancement Program, August 2012.

U.S. Army Corps of Engineers, Rock Island District, Adaptive Management Study, Winter Water Circulation Patterns in Mud Lake, 2014 Dye Study, Pool 11 Habitat Rehabilitation and Enhancement Program, January 2016.

U.S. Army Corps of Engineers, Rock Island District, Upper Mississippi River System, Environmental Management Program, Operation and Maintenance Manual Addendum, Pool 11 Habitat Rehabilitation and Enhancement, April 2016.

U.S. Army Corps of Engineers, Rock Island District, Upper Mississippi River System, Environmental Management Program, Post-Construction 10-Year Performance Evaluation Report, Pool 11 Habitat Rehabilitation and Enhancement, 2016.

U.S. Army Corps of Engineers, Rock Island District, Upper Mississippi River System, Environmental Management Program, Winter Water Circulation Patterns in Mud Lake, 2016 Dye Study, Pool 11 Habitat Rehabilitation and Enhancement, December 2016.

V. PROJECT GOAL & OBJECTIVES:

The project goals and objectives were outlined in the original Definite Project Report and are summarized in the following table.

Table 1: Project Goals and Objectives

Project Goals and Objectives		
Goals	**Objectives**	**Project Features**
Restore and Protect Aquatic/Backwater Habitat	Create off-channel deep-water areas Create areas with flow and depth diversity Reduce sedimentation in backwaters Reduce island erosion Reduce resuspension of sediments Enhance nesting/brooding habitat for migratory birds Increase abundance/diversity of aquatic plants Provide reliable food resources for migratory birds and resident wildlife	Excavate channels in backwater areas Construct deflection embankments Construct flow control structure Construct sediment trap

VI. MONITORING PLAN EVALUATION CRITERIA:

U.S. Army Corps of Engineers, Rock Island District, Upper Mississippi River System, Environmental Management Program, Post-Construction 10-Year Performance Evaluation Report, Pool 11 Habitat Rehabilitation and Enhancement, 2016.

No changes or discussion of these tables was made during this site assessment.

Table 2 Performance Evaluation and Monitoring Schedule

Goal	Objective	Enhancement	Units	Year 0 W/out Project	Year 1 W/ Project: Actual Conditions	Year 1 DPR Target	Year 10 W/ Project: Actual Conditions	Year 10 DPR Target	Year 50 Target W/ Project-DPR Target	Monitoring Schedule
Restore and Protect Aquatic and Backwater Habitat	Create off-channel deep-water areas to provide year-round habitat for centrarchids and associated species	Excavate channels in backwater areas	Winter water temperature. [°C(°F)]-Avg of all stations	0.5 (32.9)	0.7 (33.2)	1.0 (33.8)	1.2 (34.2)	1.0 (33.8)	1.0 (33.8)	Perform water quality tests
			Channel water depth [ha>1.2m (acre>3.9 ft)]	0	23.1 (5)	23.1 (57.0)[1]	14.1 (34.8)	23.1 (57.0)[1]	23.1 (57.0)[1]	Transect Surveys
			Channel velocity [cm/sec (ft/sec)] Summer & Winter Channel stations only: avg	>3.0 (>0.1)	6.7 (0.2)	0	2.7 (0.09)	0	0.3 (0.001)	Perform water quality tests
	Reduce sedimentation in backwater	Construct Deflection Embankments	Current velocity in backwater areas [cm/sec (ft/sec)] Summer & Winter Stations 583.4P & 588.0B	>3.0 (>0.1)	1.1 (0.04)	0	1.8 (0.06)	0	0	Perform water quality tests
		Construct flow control structure	Dissolved Oxygen (mg/L) Summer & Winter Avg all stations	3.0-5.0 (M*) 13.1 (S*)	12.2 (M) 12.1 (S)	≥5.0	10.3 (M) 10.3 (S)	≥5.0	≥5.0	Perform water quality tests
			Total Suspended Solids (mg/L)[2] range of avgs all stations	11.3-28.5 (M) 16.5-60.1 (S)	Not measured		6.2-34.6 (M) 8.2-22.75 (S)		Not established	Perform water quality tests
		Sediment Trap	Bottom Elevation (M)	183.0	179.8	Not established	Avg. 180.5	Not established	183.0	Transect Surveys

*M=Mud Lake
*S=Sunfish Lake

[1] DPR intent for target was the as built condition. DPR projected 24.3 hectares for as built condition. Actual as built condition was 23.1 hectares

[2] Not a monitoring plan parameter in DPR

VII. SIGNIFICANT EVENTS SINCE LAST INSPECTION

Significant events at the project site are listed in Table 3 below. These high water events are listed in descending elevations. Figure 1 below shows water stages at the Lock and Dam 11 gage for the last three years. Water levels have only reached flood stage once since 2014. This occurred in June of 2017.

Table 3: Significant Events at the Site

High Water Elevations Since Project Completion http://water.weather.gov/ahps2/hydrograph.php?wfo=dvn&gage=dldi4 WS slope from 10-year flood profile (2004 Mississippi River FFS)		
Date	Elevation at Mud Lake overtopping location, ft, MSL1912	Elevation at Sunfish overtopping location, ft, MSL1912
4/19/2011	610.94	610.04
6/13/2008	608.26	607.36
7/3/2014	608.15	607.25
4/26/2008	607.99	607.09
5/10/2014	606.83	605.93
6/19/2004	606.36	605.46
5/21/2014	606.13	605.23

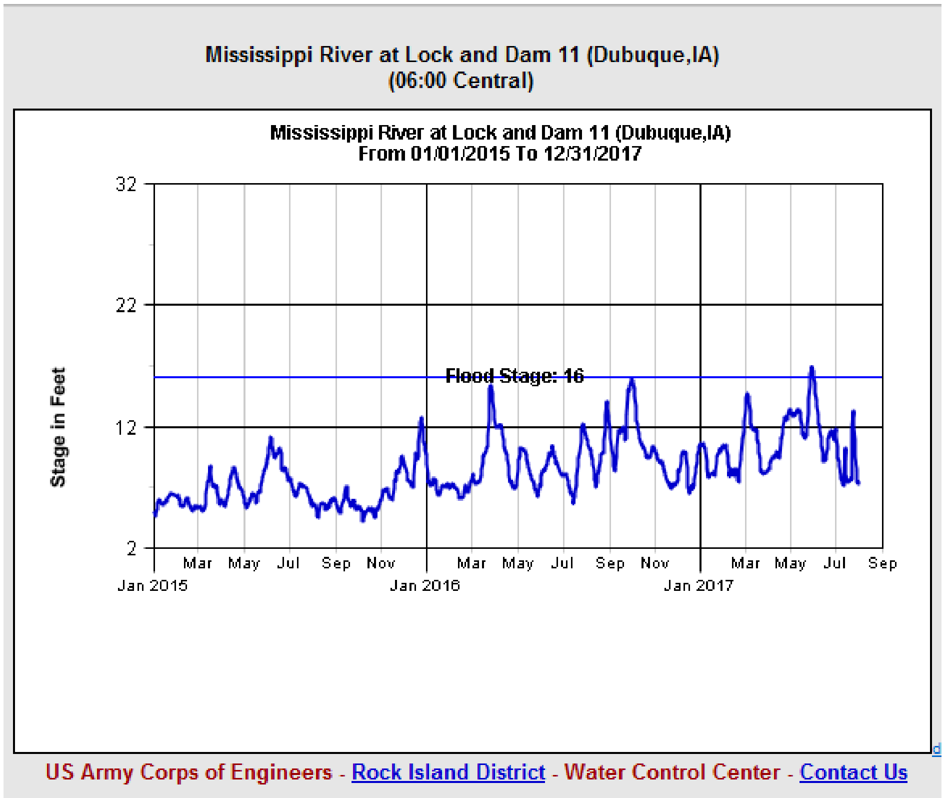


Figure 1 Pool 11 Stage from 2015 to 2017

VIII. PROJECT SPONSOR UPDATES

The US Fish and Wildlife Service (USFWS) Operation and Maintenance costs are tabulated and described below for the last several years in Table 4.

Table 4 Sponsor O&M Recent Costs (Source: Sharonne Baylor)

Fiscal Year	Cost	Description
FY13	$1700	Inspect, Mud Lake slow-no-wake buoys
FY14	$1400	Inspect, Mud Lake slow-no-wake buoys
FY15	$1700	Inspect, Mud Lake upper inlet*
FY16	$420	Inspect

*2015-2016 sponsor went from placing slow-no-wake buoys to placing “Slow No Wake” signs.

IX. DATE OF FIELD VISIT: July 12, 2017. Storms and heavy rain, clearing up in the afternoon, minimum temperature of 65°F, maximum temperature of 85°F.
Water levels were 596.24 EL on the day of the site visit.

X. ATTENDEES:

Table 5 outlines the list of personnel who conducted the site inspection.

Table 5: 2016 Site Visit Attendees

Name	Office	Title	Number
Kara Mitvalsky	USACE – Rock Island	Environmental Engineer	309-794-5623
Steve Gustafson	USACE – Rock Island	Environmental Specialist	309-794-5202
Marvin Hubbell	USACE – Rock Island	Program Manager	309-794-5428
Ben Vandermyde	USACE – Rock Island	Lead Forester	309-794-4522
Davi Michl	USACE – Rock Island	Biologist	309-794-5174
Darron Niles	USACE – Rock Island	Community Planner	309-794-5400
Kaileigh Scott	USACE – Rock Island	Civil Engineer	309-794-5318
Tara Gambon	USACE – Rock Island	Pathways Intern	309-794-5874
Elizabeth Bruns	USACE – Rock Island	Hydraulic Engineer	309-794-5762
Sarah Schmuecker	USFWS	Fish and Wildlife Biologist	309-757-5800
Jeff Janvrin	Wisconsin DNR	Habitat Specialist	608-386-0341
Sharonne Baylor	USFWS	Environmental Engineer	507-494-6207
Scott Gritters	Iowa DNR	Fisheries Biologist	563-880-8781
Mike Griffin	Iowa DNR	Wildlife Biologist	563-872-5700
William Tague	USACE – Rock Island	Engineering Technician	309-794-5164
Karen Osterkamp	Iowa DNR	Natural Resources Biologist	563-252-1156
Brandon Jones	USFWS	Refuge Manager	608-326-0515
Wendy Woyczik	USFWS	Deputy Manager	608-326-0435

XI. SITE VISIT AND RECENT SPONSOR OBSERVATIONS:

Due to severe weather there were delays accessing the project during the site visit. While waiting for storms to pass the project was discussed at length by representatives from all the organizations present. In the afternoon a portion of the attendees were able to take boats out to view Sunfish Lake. All information pertaining to Mud Lake was provided by the sponsor.

Water Control Structures: Due to heavy rains the water in the main channel was much more turbid than in the backwater areas during the site visit. Sediment laden flow was observed entering Sunfish Lake through the inlet structure. Any changes to the inlet structure will need to be initiated by the USFWS. Currently, according to the USACE Hydrology and Hydraulics branch, velocities measured in Sunfish Lake are higher than project objectives. High flows after rainfall bring in heavy sediment loads and sedimentation of Sunfish Lake is occurring at a higher rate than designed. The sediment trap continues to fill in.

Rock Revetment: Rock revetment was added at Sunfish Lake as a contract modification to alleviate erosion of the berm. The rock has held up well and allowed for the creation of an ephemeral wetland. The rock has become vegetated with shrubby plants; however, these do not appear to impact the integrity of the structure. Any fill in of sediment or debris between the rock revetment and the berm has been deemed acceptable because it will increase the structure's stability.

Erosion: The project sponsor reported that erosion is occurring on the embankment channel side at both the upstream and downstream sides of the STA 11+85 riprap slope protection in the Mud Lake project area. The eroded area is coming off in large chunks, leaving a vertical face. Images provided by the USFWS can be viewed in Attachment A.

2015 Rock Modification: In the fall of 2015 project sponsors worked with the Dubuque County Conservation Board and Iowa Department of Natural Resources (IADNR) to modify the upper inlet in Mud Lake. The modification consisted of placing rock from an adjacent stockpile in order to allow a small amount of flow at all times. The IADNR reported the following January that the riprap was performing as intended. A 2016 Dye Study performed by USACE, which is discussed in the next section, revealed that the rock closure was successful in reducing velocities in the upstream inlet. Electrofishing within Mud Lake has also shown an increase in age 1+ bluegills. However, this is only indicative of a single year of monitoring since the modification. Further monitoring will be conducted over the next several years.

Overwintering Habitat: The project goals in providing year round fish habitat has been largely successful in Mud Lake and Sunfish Lake. According to IADNR the reduced flow into both lakes has increased the number of fish utilizing the backwaters within the project site immensely. See Attachment A for images of the IADNR's workup tank filled with fish after shocking a beaver dam in Mud Lake and refer to the Summary of Pool 11 Islands HREP Fisheries Response Monitoring within the next section for more details on electrofishing within both Mud Lake and Sunfish Lake.

Vegetation: The vegetation and forest along the Sunfish Lake berm is developing well. The oldest tree on the project site was accessed during the site visit. This bur oak dates back to 1874 and is pictured in Attachment A.

Additional Comments: A side scanner was operated on one of the boats that went out in the afternoon on Sunfish Lake. The scanner was operated run along the main dredge cut within Sunfish Lake. The cut appeared fairly uniform, and submerged vegetation could be seen along the perimeter of the cut.

XII. MONITORING AND REPORTS

Hydrology and Hydraulics: Overtopping locations at both Sunfish and Mud Lake are outlined in this report. Both project locations will overtop first at the upstream end of the project. Overtop would occur between a 10 year and a 25 year flood. The project has overtopped once since project completion in April 2011. For more details refer to Attachment C.

Uses of Ecosystem Goods and Services in Adaptive Management: Mud Lake Habitat Restoration Project as a Case Study: The Case Study objectives consisted of assessing nutrient limitation in UMRR-EMP back waters and testing the relationship of timing and flow volume into the Mud Lake project area relative to nutrient inputs. Specifically this report took into consideration the low levels of nitrogen resulting from the reduced flow in the backwaters and as a result of the project features designed to produce overwintering habitat for fish. This document can be viewed in Attachment D.

2014 Dye Study: Winter Water Circulation Patterns in Mud Lake – An Adaptive Management Study of a Backwater Habitat Restoration Project in Pool 11 of the Mississippi River: In 2014 a dye study of Mud Lake was conducted in response to IADNR fish telemetry data. The fish telemetry data indicated that fish were not utilizing the backwater channels as desired or expected. The dye study showed that the increased velocities at the upstream inlet were undermining project goals, and that other measures would be necessary to result in velocities that would better support overwintering fish habitat. For further information on the 2014 Dye Study refer to Attachment E.

2016 Dye Study: Winter Water Circulation Patterns in Mud Lake Following Inlet Modification – An Adaptive Management Study of a Backwater Habitat Restoration Project in Pool 11 of the Mississippi River: As a response to project modifications, USACE performed an additional dye study in February 2016. The modified rock inlet impacted velocities in the upper dredged channels near the inlet. Velocities were significantly reduced from Mud Lake through Zollicoffer Slough. The Mud Lake inlet was successful in reducing velocities in the upstream areas of the project to support fish overwintering requirements. For additional details on the 2016 dye study refer to Attachment F.

Water Quality Data: Summer and winter water quality monitoring performed by USACE at the HREP is ongoing since December 2013. Water quality monitoring results on data collected prior to 2013 are discussed in previous Performance Evaluation Reports.

Forestry Data: A tree species survey and USDA forest codes write-up are provided for this project. Data sheets charting tree species, azimuth, radial distance, height, health, age, etc. are included in Attachment G. These sheets show the diversity of the vegetation that came in following project completion. The tree plantings, as mentioned in the observations above, have developed well and the oldest tree on site is nearing one hundred and fifty years old.

Summary of Pool 11 Islands HREP Fisheries Response Monitoring: Electrofishing during the late fall has been used to monitor the fisheries response to the project. According to these efforts Sunfish Lake began operating as an overwintering site in the fall of 2004. Mud Lake followed two years later in 2006 as effectively providing overwintering habitat. Both Mud Lake and Sunfish Lake showed an increase in age 1+ bluegills and species diversity over time post project. Catch Per Unit Effort (CPUE) is increasing slowly as the population is building. This can take 7 to 10 years to become established. Once population is established, the CPUE is expected to fluctuate based on factors influencing catchability by the electrofishing crews. The full summary and data can be viewed in Attachment H.

XIII. SUMMARY

Overall the Pool 11 HREP appears to be generally meeting its goals and objectives through continued operation and maintenance by the USFWS.

XIV. RECOMMENDATIONS

- Monitor flow into the project site and continue water quality monitoring to determine impacts on backwater water quality and depths
- Repair erosion along Mud Lake embankment

XV. LESSONS LEARNED

Provide project features that support multiple project goals and year round habitat. Additional rock available at closure structures with notches at future HREPs will help to support adaptive management. Unarmored dredged material berms may erode in flow regimes similar to this HREP, and may affect adjacent constructed channels.

Attachment A
2017 Site Visit Photos

2017 site visit attendees from USACE, Iowa DNR, Wisconsin DNR, and USFWS

Weather

Inclement weather on the day of the site visit resulted in delays in viewing the site due to safety concerns. Included are images of the storm rolling in.

View of Sunfish Lake from Eagle Point Park in Dubuque, IA.

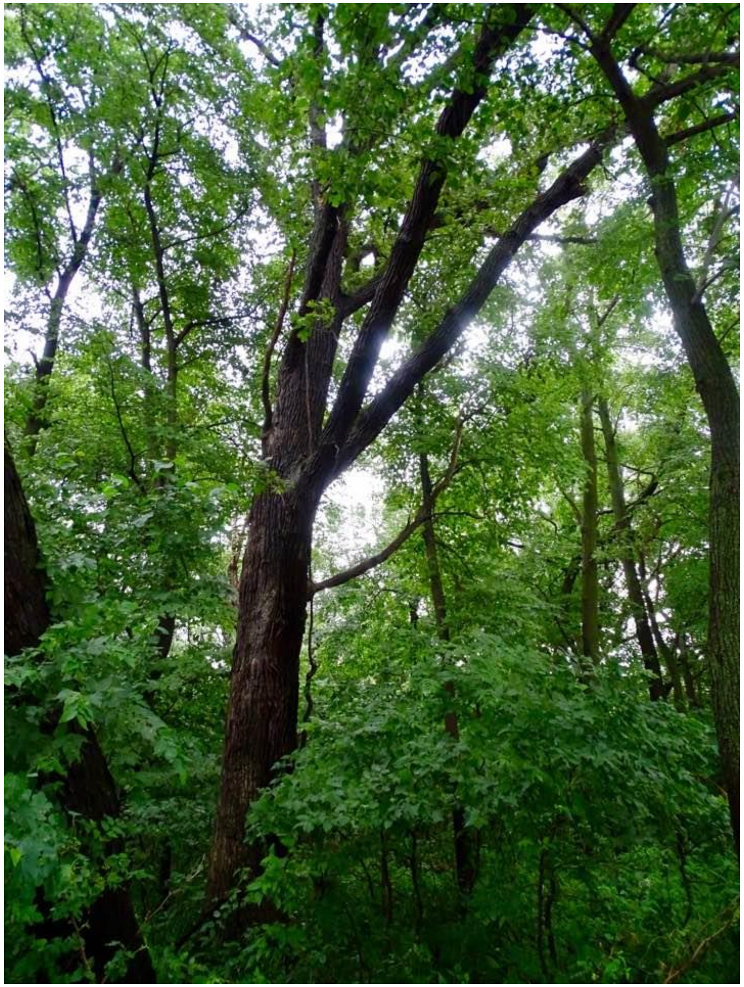
Sunfish Lake Rock Revetment

The rock revetment added as a construction modification to the project is pictured. The ephemeral wetland this feature has added to the project can be viewed as well just past the revetment.

Sunfish Lake Wildlife

Several bald eagles were observed during the site visit by attendees. No nest was noted. However, the eagles are believed to have nested nearby either within the project site or outside of the project boundaries.

Sunfish Lake Vegetation



The oldest tree on site is pictured to the left. This tree is located along the earthen embankment of the Sunfish project area.

Sunfish Lake Vegetation

View of the ephemeral wetland from within sunfish lake

Sunfish Lake looking downstream

Sunfish Lake Vegetation

Vegetation along earthen embankment of Sunfish project area

Sunfish Lake Vegetation

Wild rice at Sunfish Lake (Photo taken 9/6/17)

Sunfish Lake Inlet Structure

The inlet channel was letting in a lot of sediment and dirty water during the site visit. The structure was designed too large for the project and the notch is allowing flow with a higher velocity than desired into the project area. The clear water mixing with the water from upstream can be viewed in both images above.

Sunfish Lake Inlet Structure

Adjacent to the inlet structure on land is extra rock. This is believed to be left over from construction of the structure.

Wildlife

Pictured are fish within the IDNAR's workup tank following the shocking of a beaver dam. These were gathered within the Pool 11 project area.

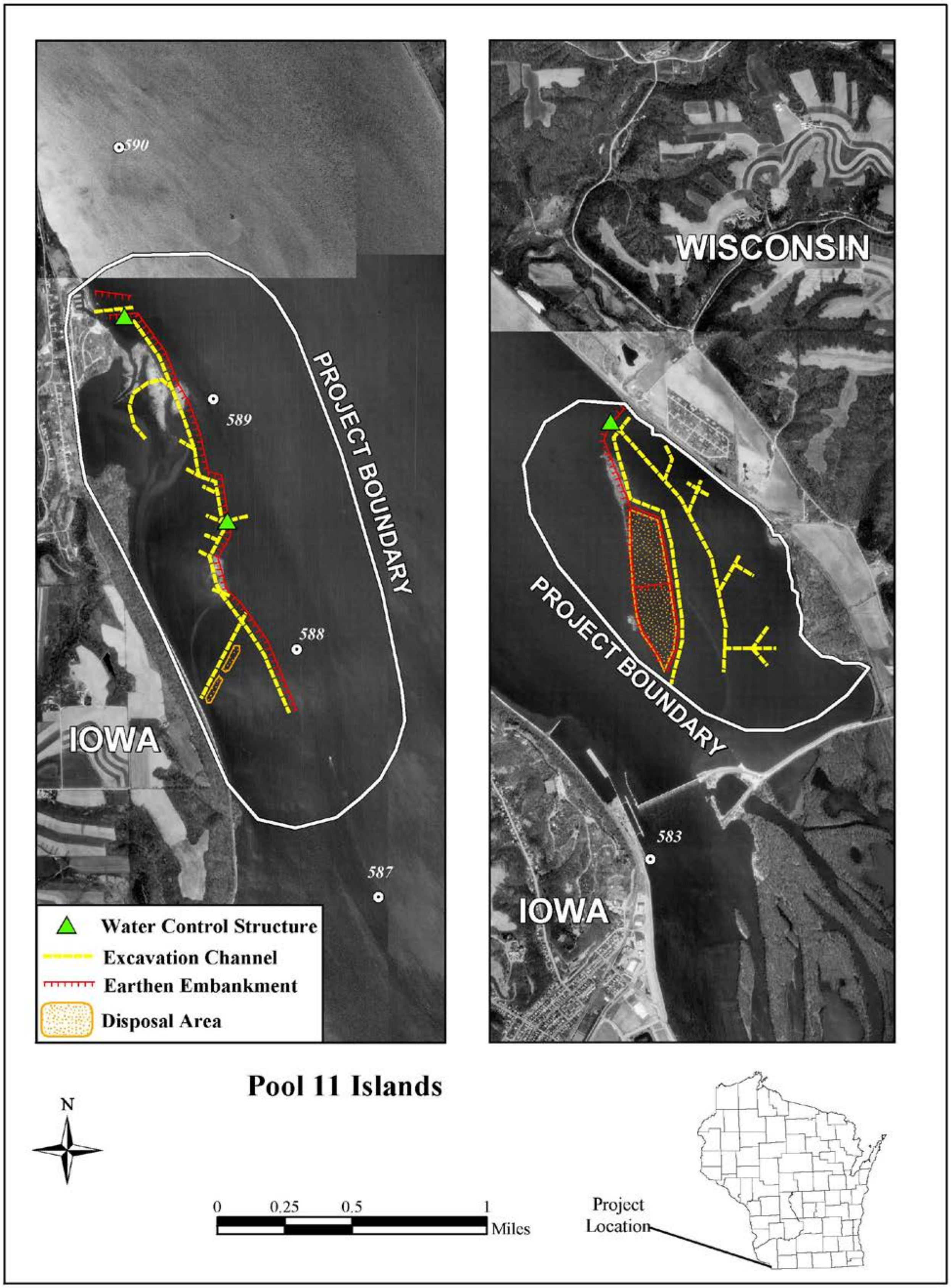
Mud Lake Erosion

The images below were provided by the sponsor. These show erosion occurring along the exterior of the Mud Lake embankment. The embankment is separating in large chunks and creating a vertical face.

Mud Lake looking downriver

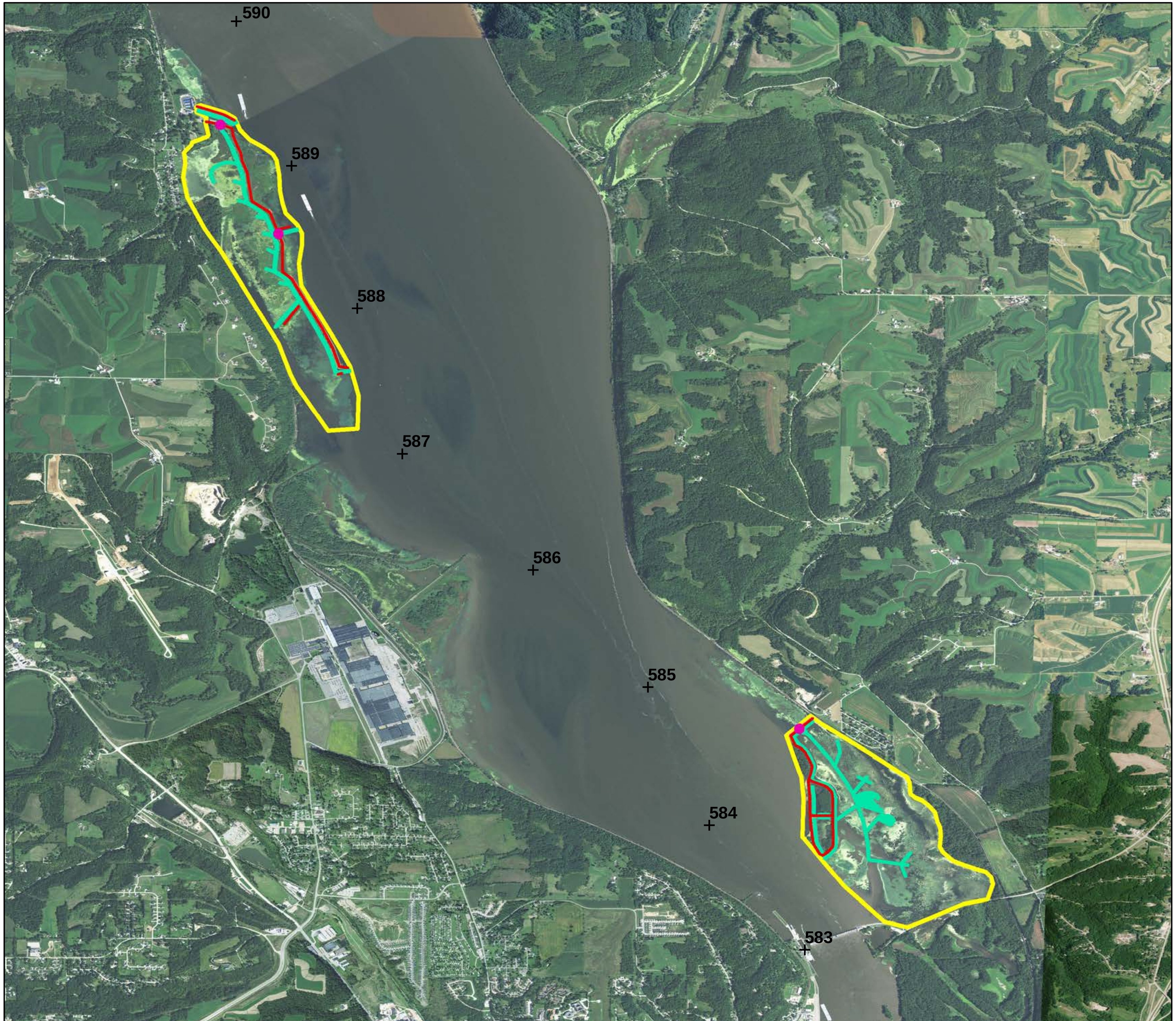
Mud Lake looking upriver

Attachment B
Site Plan



Pool 11 Islands

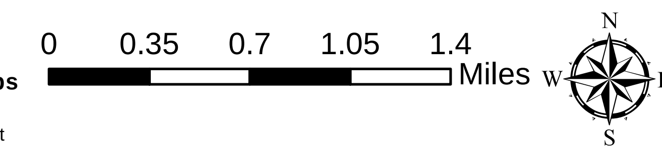
Pool 11 Islands



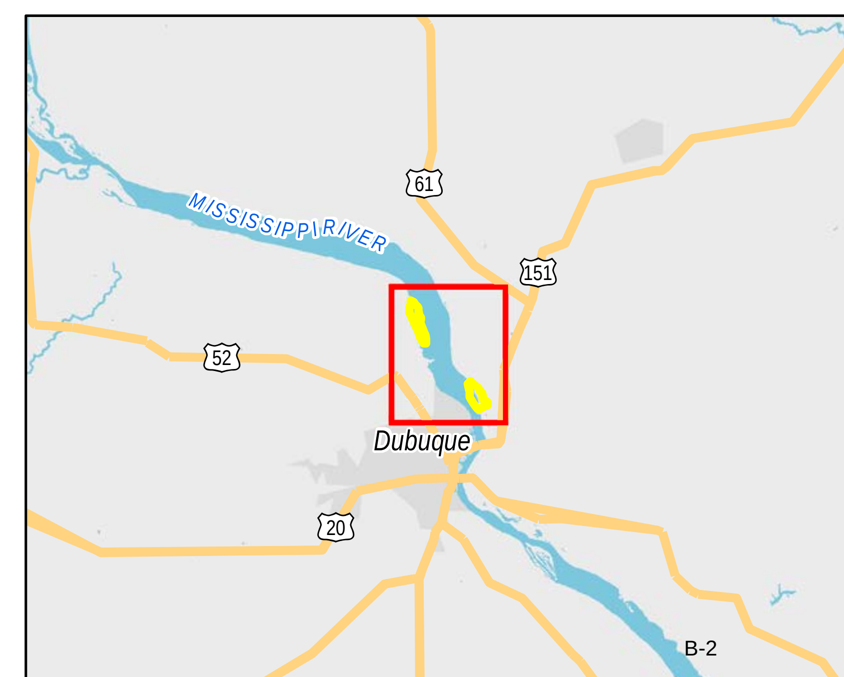
Legend

- Water Control Structure
- Embankment Centerline
- Dredging Event
- Revetment
- Project Boundary
- River Miles

US Army Corps of Engineers
Rock Island District



-- Location Map --



Attachment C

Hydrology and Hydraulics

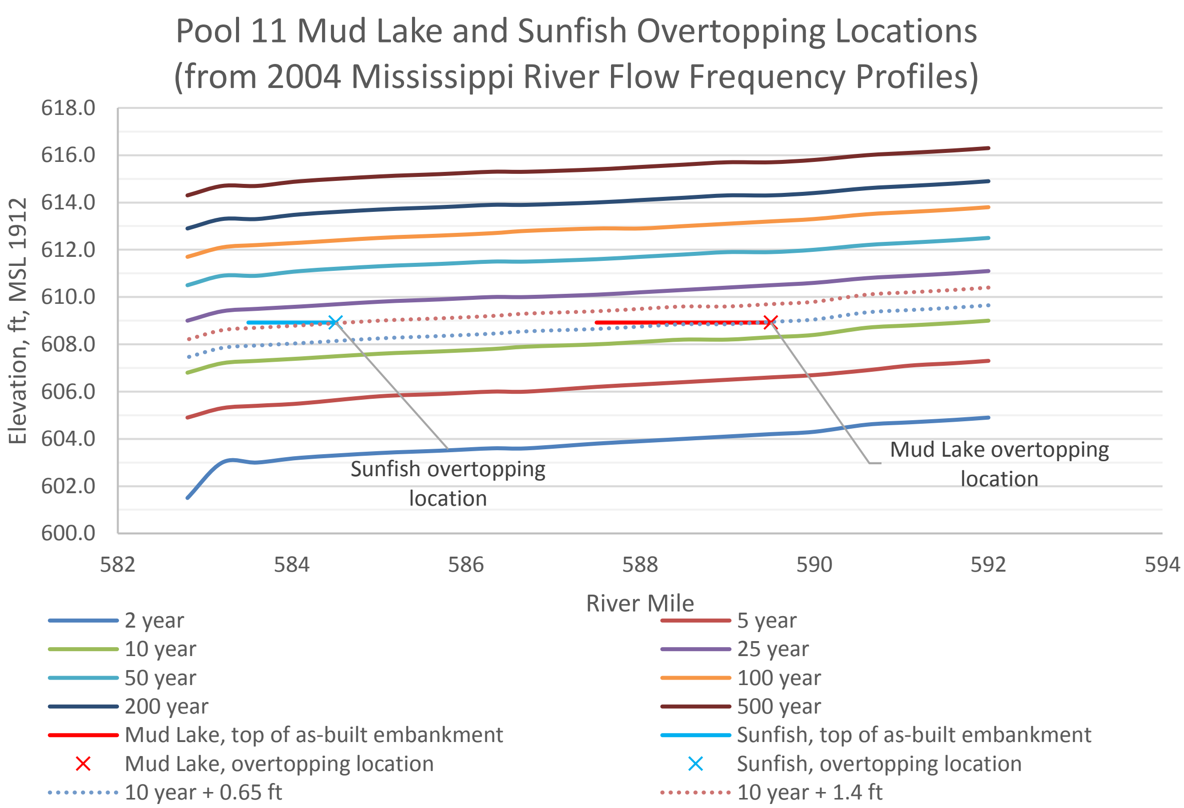
Pool 11 Island EMP
Site Visit 7/12/2017
HH observations and data

1. Overtopping location

All berms for Sunfish and Mud Lake were built to a consistent top elevation of 608.92 ft MSL 1912. This top elevation was plotted against flood profiles to determine if overtopping of each site at the upstream end first could cause damage to the berms and increase velocity head in the dredged channels.

According to the as-built top of berm elevations and the 2004 Mississippi River FFS profiles, Mud Lake will overtop first as the upstream end. When overtopping occurs, the downstream end of the project will have approximately 0.27 ft of freeboard. Sunfish will overtop after Mud Lake. Overtopping will occur at the upstream end of Sunfish first. When overtopping occurs, the downstream end of the project will have approximately 0.22 ft of freeboard.

Note that changes in the berms since original construction, errors expected in surveying in heavy vegetation, and LiDAR error are likely to exceed the head difference estimated between the upstream and downstream ends of each site.



2. High water events

High water elevations at Sunfish and Mud Lake since project completion are shown. Sponsor confirms overtopping of berms occurred once since project completion, on 4/19/2011. Top of berm elevation is 608.92 MSL 1912.

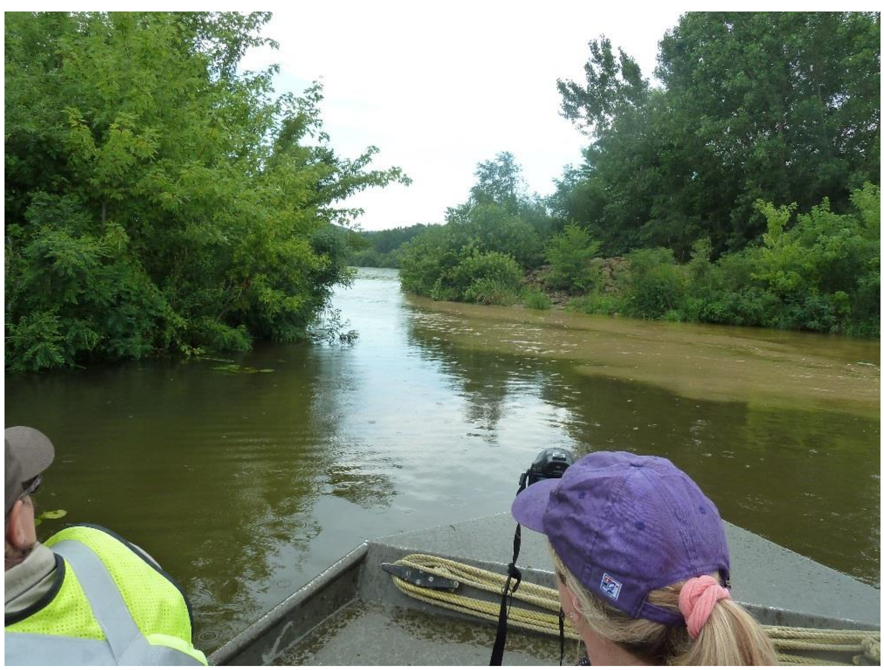
High Water Elevations Since Project Completion http://water.weather.gov/ahps2/hydrograph.php?wfo=dvn&gage=dldi4 WS slope from 10-year flood profile (2004 Mississippi River FFS)		
Date	Elevation at Mud Lake overtopping location, ft, MSL1912	Elevation at Sunfish overtopping location, ft, MSL1912
4/19/2011	610.94	610.04
6/13/2008	608.26	607.36
7/3/2014	608.15	607.25
4/26/2008	607.99	607.09
5/10/2014	606.83	605.93
6/19/2004	606.36	605.46
5/21/2014	606.13	605.23

3. Rock revetment at Sunfish

Rock revetment was added at Sunfish (as a contract modification) to alleviate erosion of the berm. The rock has held up well and allowed for the creation of an ephemeral wetland at Sunfish. The rock has become vegetated with shrubby plants. Filling in of debris and sediment between the rock revetment is acceptable because it will increase the structure's stability.

4. Closure structure with inlet notch and sediment trap at Sunfish

Closure structure rock is holding up well, but the notch is too big. Velocities measured in Sunfish backwaters are higher than project objectives. High flows after rainfall bring in heavy sediment loads and sedimentation of Sunfish is occurring at a higher rate than designed. Sediment trap is expected to fill in before 50 year design life.



Attachment D

Uses of Ecosystem Goods and Services in Adaptive Management: Mud Lake Habitat Restoration Project as a Case Study

THIS ADVANCED BIOLOGICAL PROJECT HAS BEEN EXAMINED AND APPROVED:

Name Susan P Romano

Susan Romano

Chair, Examining Committee

Name Charles Theiling

Charles Theiling

Member, Examining Committee

Name Charles Lydeard

Charles Lydeard

Member, Examining Committee

Date May 27, 2016

USES OF ECOSYSTEM GOODS AND SERVICES IN ADAPTIVE MANAGEMENT:

MUD LAKE HABITAT RESTORATION PROJECT AS A CASE STUDY

A THESIS

PRESENTED TO THE FACULTY OF

THE SCHOOL OF GRADUATE STUDIES OF

WESTERN ILLINOIS UNIVERSITY

IN PARTIAL FULFILLMENT

OF THE REQUIREMENTS FOR THE DEGREE

MASTER OF SCIENCE

BY

DAVI MICHL

DR. SUSAN P. ROMANO, ADVISOR

2016

ACKNOWLEDGEMENTS

I thank Dr. Susan Romano for her effort and support in conducting this study and review of this manuscript. I thank Dr. Charles Theiling for his significant time investment in developing this study and in sourcing partial funding from the Upper Mississippi River Restoration-Environmental Management Program. My sincerest gratitude is extended to Michael Dougherty for his assistance with both model design and manipulating R Studio for statistical analyses. I thank Dr. Charles Lydeard for his contribution in reviewing this document. I also acknowledge Jeff Houser and Bill Richardson from the Upper Mississippi Environmental Sciences Center for elucidating years of nutrient dynamics research conducted in the Upper Mississippi River System. Finally, I acknowledge Dave Bierman from the Bellevue Field Station for providing Brown's Lake nutrient datasets.

ABSTRACT

Algal blooms resulting from nitrogen (N) limitation are frequently observed in Upper Mississippi River (UMR) backwaters, and research has demonstrated that the degree of connectivity to the main channel can help introduce nitrogen to these habitats for nutrient processing. Mud Lake is a UMR-Habitat Rehabilitation and Enhancement Project (HREP) used here as a case study for a hydrological-nutrient simulation model assessing potential nutrient uptake benefits of three alternative flow regimes. The objectives of this research were to verify nutrient trends using data from a nearby backwater and to assess the effectiveness of each modeled flow regime in reducing downstream N delivery.

Verification of N limitation was determined by analyzing the difference in mean nitrite-nitrate (NO_3) concentrations between two sites: Brown's Lake HREP backwater and an adjacent main channel site using Long Term Resource Monitoring (LTRM) and Bellevue Field Station datasets (1989-2013). A two-way ANOVA with independent variables (site and season) indicated water column NO_3 in Brown's Lake was significantly lower than in the main channel ($p < 0.01$). Seasonal differences in mean NO_3 between the two sites were also observed—Brown's Lake was N limited relative to the main channel during all seasons except winter ($p < 0.01$). There was no significant difference in seasonal NO_3 observed in the main channel. These results supported the observed trend of nitrogen (N) limitation in isolated backwaters and its prevalence in the main channel.

This research also examined how the interplay between hydraulic residence time and flow volume into the Mud Lake project area can affect nutrient processing and

habitat benefits using a spreadsheet model. Three hydrologic flow management structures were evaluated, including a rock closure (0 m^3 s^{-1}), notched weir (2.0 m^3 s^{-1}), and gated culvert (0-7.9 m^3 s^{-1}). The Mud Lake model estimated total potential N denitrified per growing season for the gated culvert flow regime was 7.36 x 10^{10} mg m^{-2}, considerably higher than either the rock closure (2.9 x 10^{10} mg m^{-2}) and notched weir (4.7 x 10^{10} mg m^{-2}). The Mud Lake model demonstrated its capacity as a simple, yet relevant model with wide management and policy implications for planning restoration projects within an adaptive management framework.

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INTRODUCTION

The Upper Mississippi River System (UMRS) is recognized by Congress as an ecologically diverse and historically significant region. Humans derive both aesthetic and economic benefits including commercial and recreational fisheries, drinking water supply, a multi-billion-dollar tourism industry, and a commercial navigation system (Millennium Ecosystem Assessment, 2005). In response to environmental degradation, Congress authorized that funding be allocated to ecosystem restoration and scientific monitoring in Section 1103 of the Water Resources and Development Act (WRDA) of 1986 (P.L. 99-662), as amended (USACE, 2012). The Upper Mississippi River Restoration Environmental Management Program (UMRR-EMP) is the primary outgrowth of this mandate, charged with restoring diverse aquatic habitats and associated floodplains along the entire UMRS.

There are two primary components of the UMRR-EMP: a long-term resource monitoring program (LTRM) and habitat rehabilitation and enhancement projects (HREPs). Together, these two components create a unique and valuable learning environment for resource managers, termed adaptive management. Adaptive management is iterative learning approach to applied natural resource conservation (Williams, 2011; Fischenich and Vogt, 2012). This approach implies management strategies change both as natural systems respond to the management intervention or in response to stochastic biogeochemical variations in the environment (Williams, 2011). This management strategy has often been cited as “learning while doing”, allowing resource managers to plan, monitor, and adjust restoration projects in the face of uncertain outcomes resulting from the management intervention (Fischenich and Vogt, 2012).

It is important to note that natural resources management actions also affect ecosystem services. The Institute for Water Resources (IWR) defined ecosystem services within a Corps planning framework as “socially valued aspects or outputs of ecosystems that depend on self-regulating or managed ecosystem structures and processes” (Reed et al., 2013). These services may include: commercial navigation, nutrient delivery to floodplains and coastal regions, fisheries, recreation, dilution and transport of wastes (Millennium Ecosystem Assessment, 2005; Reed et al., 2013). Because the effects of adaptive management on ecosystem services also deal with uncertain outcomes, the integration of these concepts may inform decisions that improve management of both natural resources and ecosystem services.

Traditionally, HREPs have been designed to address project-specific natural resources goals identified early in the planning process (USACE, 2012). Due to the uncertainty inherent in ecosystem restoration planning, these objectives are often narrow in scope and fail to consider multiple restoration benefits that can be derived from a project. Incorporating ecosystem goods and services (EGS) is of increasing interest to resource managers implementing adaptive management plans to achieve multiple restoration project objectives and ecosystem service benefits.

Case Study

Mud Lake is part of the UMRR Pool 11 HREP and is a prime example of adaptive management in action. Following impoundment, Mud Lake’s former deepwater habitat and channels were completely submerged with high current velocities that precluded its use for overwintering (Wilcox, 1993; USACE, 2014). The original design and objectives of the HREP aimed to restore and protect backwater habitat, including fish overwintering habitat and diverse backwater habitat for waterfowl (USACE, 2014). Initial project monitoring revealed fish were not using much of the project area during the winter as projected. High flows were therefore reduced to an estimated 2.0 m^3 s^{-1} by filling in the notch with rock, but fish still did not use the upper dredged channels where current velocity remained high. During construction of this first closure, project planners anticipated the need for change and had the contractors leave an additional stockpile of rock in anticipation of adaptive management. This step demonstrated foresight for achieving the primary objective of creating overwintering habitat—if fish tracking and monitoring data collected during project performance evaluation periods revealed fish were not using the project or that flows were still too high, there was a potential to add more rock on site to cut off flow. While this was good example of planning with the future in mind, the decision failed to consider other EGS benefits that could potentially be achieved outside the winter period.

Like many isolated backwater habitats in the UMR, Mud Lake frequently experiences noxious algal blooms during the mid- to late-summer months when low-flow conditions do not permit adequate nitrogen delivery to backwaters (Giblin et al 2013). Instead, excess nutrient runoff is transported in the main channel, which has become a persistent problem in coastal ecosystems, contributing to hypoxic dead zones (Alexander et al., 2000; Rabalais et al., 2001). These seasonal eutrophic conditions represent an opportunity to capture EGS benefits in backwaters. Historically, the UMRR-EMP program has supported a tremendous amount of nutrient dynamics research in the UMRS that provides a foundation for the multi-objective adaptive management of Mud Lake. In Pools 4 through 8, algal blooms were significantly associated with nitrogen (N) limitation occurring in isolated backwaters (James et al., 2008a; Kreiling et al., 2010; Giblin et al., 2014). Houser et al. (2013) further demonstrated this N limitation is largely associated with the degree of connection to the main channel. Backwaters exhibit greater denitrification rates compared to other riverine habitat strata, owing to their lower current velocities and highly organic sediments (Richardson et al., 2004; Strauss et al., 2004). It has been suggested that reestablishing connectivity to backwater strata could be used to harness this nutrient cycling potential, manage algal succession, and reduce N transport downstream (James et al., 2008b; Kreiling et al., 2010; Strauss et al., 2011; Stevenson, 2014).

Research Objectives

The objectives of this research were two-fold: 1) assess nutrient limitation in UMRR-EMP backwaters, and 2) test the relationship of timing and flow volume into the Mud Lake project area relative to nutrient inputs.

These objectives will be tested with the following hypotheses:

1) Rock closure (0 $m^3 s^{-1}$): the project will remain N limited and blue-green algae will dominate the project area in the late summer, but winter habitat will be improved for fish (200 mg $m^{-2} d^{-1}$)

2) Notched weir (2.0 $m^3 s^{-1}$): there will be low to moderate nutrient processing in the summer and moderate fish usage in the winter (200 mg $m^{-2} d^{-1}$)

3) Gated culvert (0-7.9 $m^3\ s^{-1}$): will allow greatest control over derived seasonal benefits, optimizing both nutrient processing in the summer and fish habitat usage in the winter (500 mg N $m^{-2}\ d^{-1}$).

METHODS

Project Area

The Upper Mississippi River (UMR) comprises the area upstream of Cairo, Illinois. It consists of a series of low-head dams that maintain sufficient depth (2.7 m) for navigation and is divided into 27 reaches, or navigation pools (Chen and Simons, 1986; Theiling and Nestler, 2010). This structural design has altered the hydrological pattern of the UMR and created distinct aquatic habitat strata; namely the main channel, side channels, impoundments, and contiguous or isolated backwaters (Richardson et al., 2004; Strauss et al., 2004). Mud Lake is located in the impounded area created by Lock and Dam 11 on the right descending bank approximately 2.3 miles upstream of Dubuque, Iowa. The average annual precipitation for the Dubuque area was 92.3 cm, most of this precipitation occurs seasonally May-July (www.usclimatedata.com/ accessed 5 May 2016). In 2015, average discharge at Lock and Dam 11 was 1,504 $m^3 s^{-1}$ (www.rivergages.mvr.usace.army.mil/ accessed 5 May 2016). On average, higher rates of discharge are observed during the spring months (March-May) and typically correspond to the spring flood pulse.

Study Design

Brown's Lake vs. Main Channel

The assumptions for this dual hydrological-nutrient simulation model are considered to provide reasonable estimates of denitrification rates within a particular habitat type (backwater) based on earlier research conducted upriver from Mud Lake (James et al., 2008a; 2008b). However, it will be prudent to test the idea that main channel (MC) significantly differs in nutrient concentration as a function of site and season compared to a backwater site (BW), for which nutrient data is available (Houser and Richardson, 2010).

While there are no baseline water quality data currently available for the Mud Lake project area, other HREP projects with sufficient water quality datasets of post-management interventions can provide insights regarding hypothesized Mud Lake outcomes. The Brown's Lake HREP is located in Pool 13 on the right descending bank, river mile 545, approximately 40 miles downstream from Mud Lake (USACE, 1987). Brown's Lake is an ideal candidate for analysis because it is located in one of the LTRM trend pools where a broad range of water quality parameters are available—in short, Brown's Lake has a long history of nitrate samples and Mud Lake does not. Furthermore, Brown's Lake is situated closer to the Mud Lake backwater (BW), allowing for more direct comparison (similar precipitation, flooding, discharge, etc.) than Pools 4-8 where earlier denitrification research was conducted (Richardson, 2004; James et al., 2008b; Giblin et al.. 2014). It should be noted that while Brown's Lake was built with the capacity to manage connectivity (gated culverts), opening the gates is only used when monitoring demonstrates dissolved oxygen (DO) concentrations dip too low for overwintering fish survival. When waters become anoxic in either the winter or summer, managers will open the gates until DO levels recover to target levels (Gent et al., 1995). Thus, Browns Lake is currently functioning as a semi-isolated backwater with highest connectivity to the main river channel occurring during spring flooding, similar to Mud Lake. Testing for N limitation in Brown's Lake can help further validate observed UMR nutrient dynamic trends and the assumption that backwater habitat strata function similarly across UMR pools (Kreiling et al., 2010; Strauss et al., 2011).

All statistical analyses were conducted using R package “Rcmdr” (Fox, 2005). First, samples from a 1989-2013 dataset (Bellevue Field Station; LTRMP, Soballe and Fischer, 2004) were coded by creating three new categorical variables. The first variable was coded by site, either main channel (MC; n = 387) or Brown’s Lake backwater (BW; n = 265); these correspond to LTRM fixed sampling sites M556.4A and M545.5B, respectively (Figure 1). The second categorical variable was designed to separate the combined nitrite/nitrate (NO_3 in mg/L) samples of both sites by biweekly sample period, used to observe general trends in both intra- and inter-annual nutrient concentrations (Table 1). The interaction plot depicted in Figure 2 helped to visualize mean separation of the raw NO_3 by both site and biweekly sample period with standard error bars. Then, a LOESS smoothing technique was applied to demonstrate average trends in NO_3 between BW and MC sites for all years sampled (Figure 3). While this scatterplot depicted NO_3 trends more clearly, there were insufficient samples in the BW during the winter months (biweekly sample periods 1, 2, 4, 24, 25, 26) that precluded direct comparison to the MC. The lack of samples in backwater during these periods is commonly attributed to restricted access and sustained ice cover. To achieve more uniform sample replication, a third categorical variable coded biweekly sample period by its corresponding season: Spring, Sumer, Fall, and Winter (Table 1).

The frequency distribution of NO_3 samples for both sites approximately followed a normal distribution; likely a result of multiple-year sampling events (Figure 4). Boxplot separation (little overlap) between BW and MC also suggested substantial differences in water column NO_3 (Figure 5). Thus, a two-way analysis of variance (ANOVA II) was employed, owing to its robustness against minor normality deviations and the large sample size (n = 652) (Altman and Bland, 1995; Elliott and Woodward, 2007). This parametric test was used to minimize the chance of committing a Type I error and to demonstrate whether the independent variables (site, season) have an effect on the response variable (NO_3). To simplify the number of multiple comparisons for the post-hoc Tukey analysis, a third categorical variable was created by grouping each biweekly sample period by its corresponding season: Spring, Summer, Fall, or Winter (Table 1). In short, if N limitation were indeed a problem in Brown's Lake, expected mean concentrations of NO_3 would be higher in the earlier sample periods (early growing season) and plummet during the later sample periods (late growing season). In contrast, NO_3 would be expected to occur throughout each sample period in main channel (M556.4A), offering an abundant supply that could be made available for injection to backwaters as they become N limited. These analyses were employed to determine whether nutrient trends observed in Brown's Lake support the expected outcomes of adaptive management in Mud Lake.

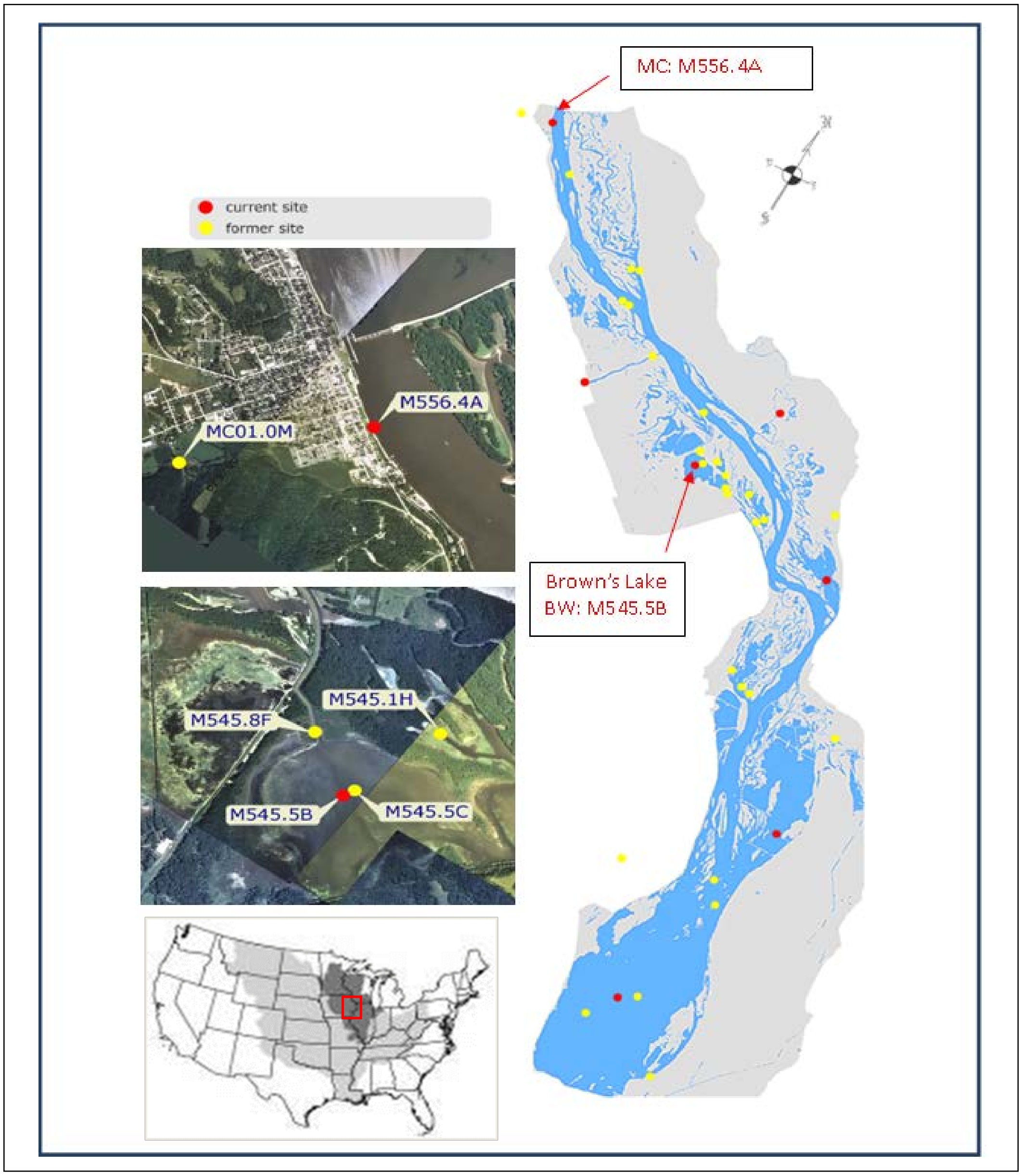

Figure 1. USGS Water Quality Graphical Browser Pool 13 fixed sample points M556.4A and M545.5B (www.umesc.usgs.gov, accessed 16 Apr 2016).

Table 1. Variable codes for ANOVA II testing the effects of two factors (site and season) on water column NO_3.

Sample Event	Biweekly Sample Period	Season
Jan 1-14	1	Winter
Jan 15-28	2	Winter
Jan 29-Feb 11	3	Winter
Feb 12- 25	4	Winter
Feb 26-Mar 11	5	Winter
Mar 12-25	6	Spring
Mar 26- Apr 8	7	Spring
Apr 9-22	8	Spring
Apr 23-May 6	9	Spring
May 7-20	10	Spring
May 21-Jun 3	11	Spring
Jun 4-17	12	Spring
Jun 18-Jul 1	13	Summer
Jul 2-15	14	Summer
Jul 16-29	15	Summer
Jul 30-Aug 12	16	Summer
Aug 13-26	17	Summer
Aug 27-Sept 9	18	Summer
Sept 10-23	19	Summer
Sept 24-Oct 7	20	Fall
Oct 8-21	21	Fall
Oct 22-Nov 4	22	Fall
Nov 5-18	23	Fall
Nov 19-Dec 2	24	Fall
Dec 3-16	25	Fall
Dec 17-31	26	Winter

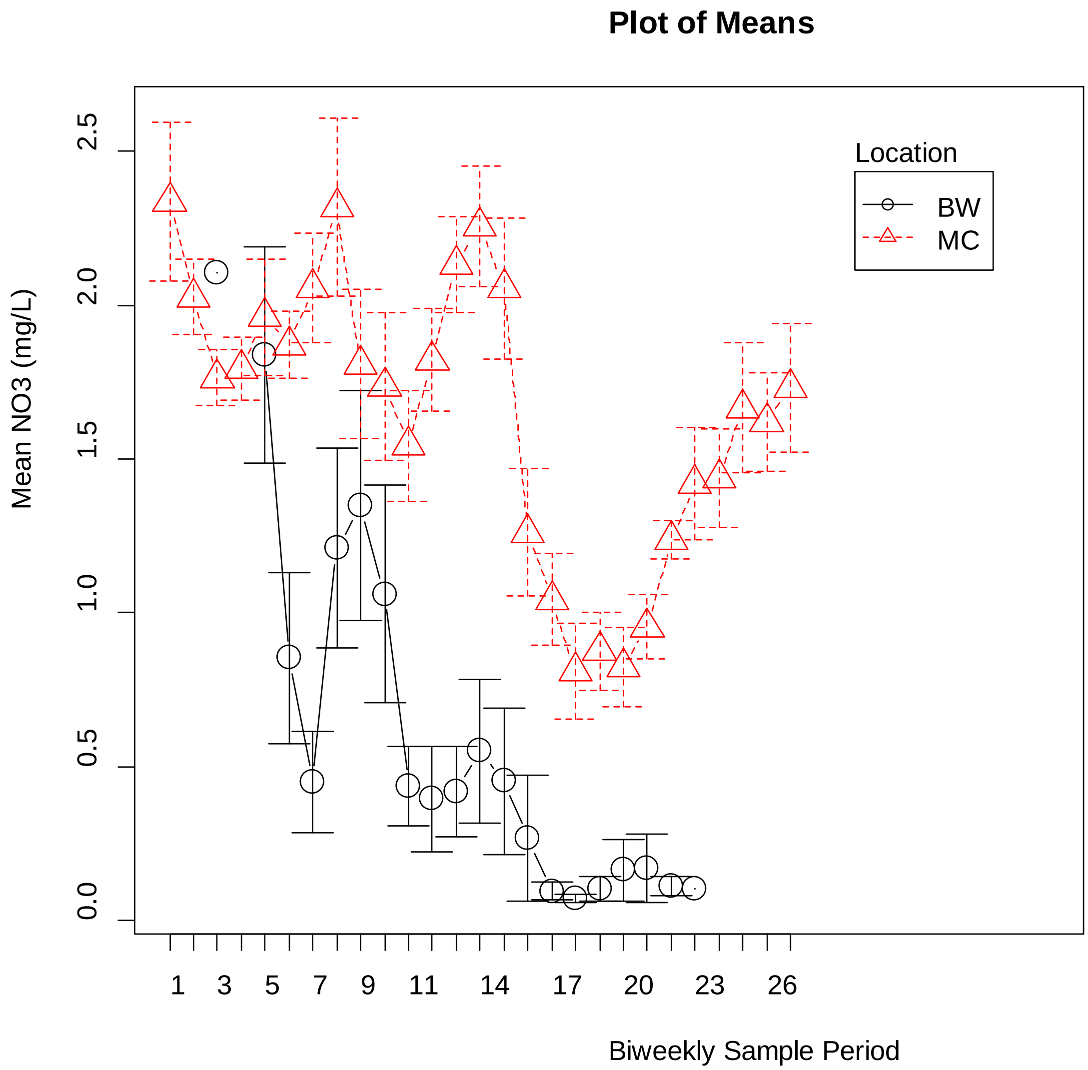

Figure 2. Interaction plot of mean water column NO_3 between MC (red triangles) and BW (black circles) for all biweekly periods sampled 1989-2013 (refer to Table 1 for x-axis codes)

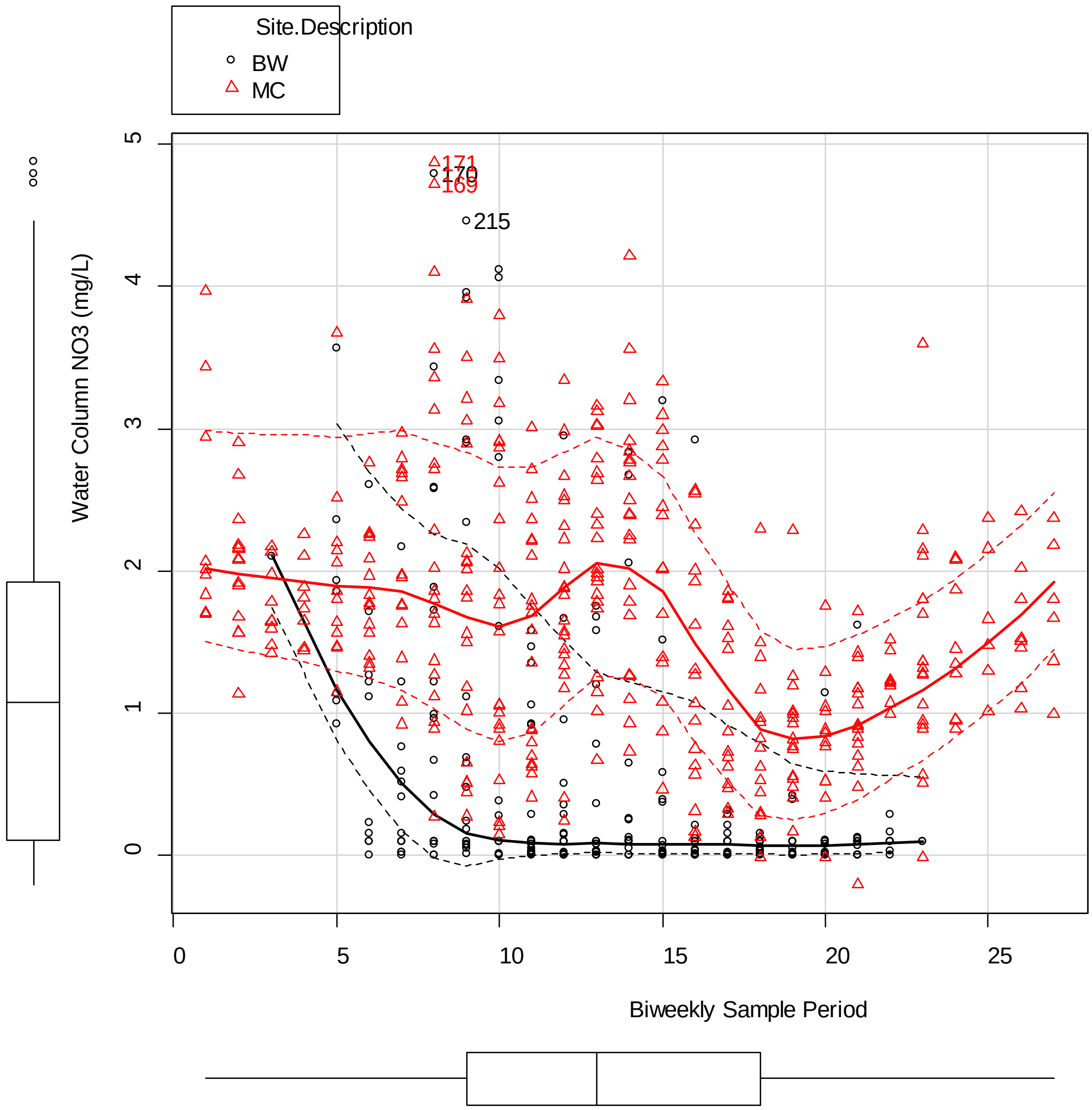

Figure 3. LOESS regression trend with smoothing parameter (solid lines) of water column NO_3 (mg/L) for Brown's Lake (black circles) and MC (red triangles). Dashed lines represent inter-annual variability (1989-2013). Refer to Table 1 for x-axis codes.

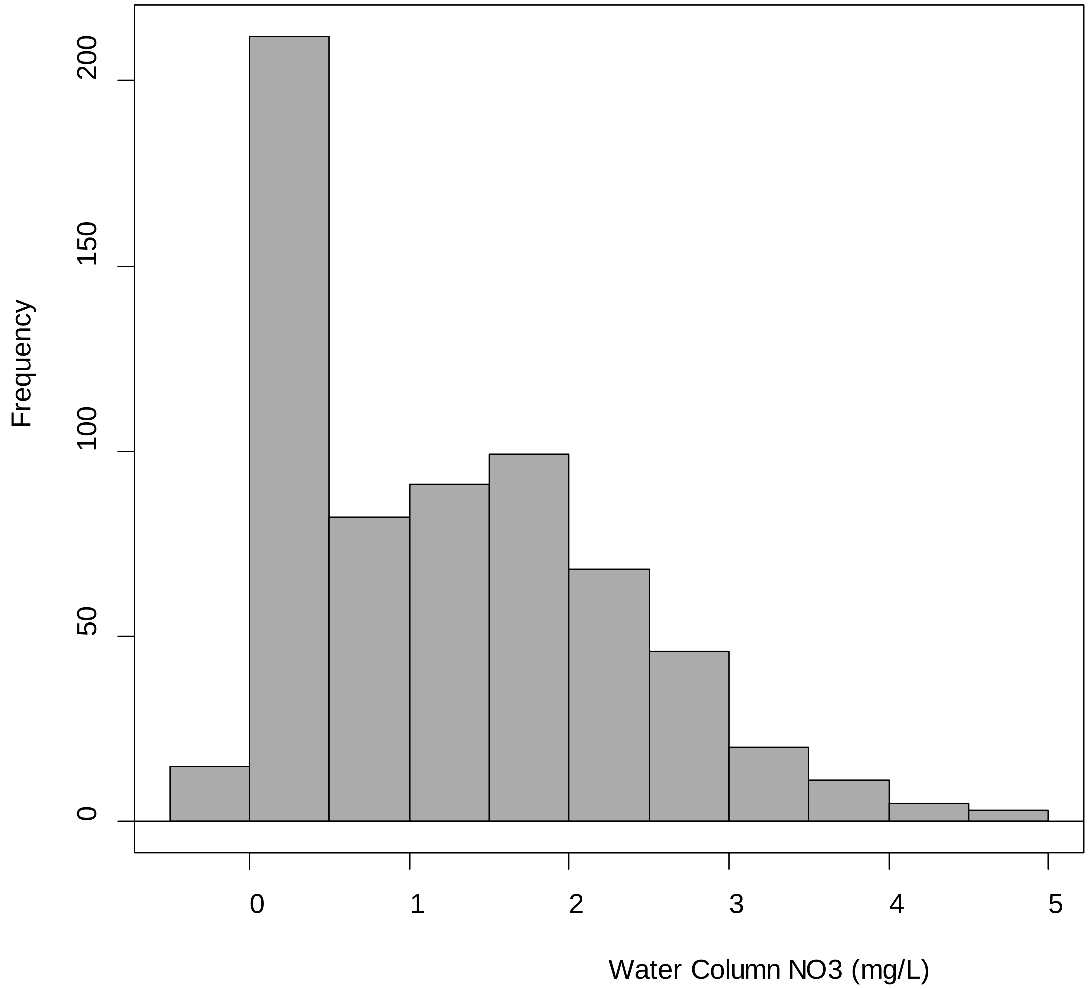

Figure 4. Frequency distribution of water column NO_3 (mg/L) for MC (M556.4A) and BW (M545.5B) sites combined.

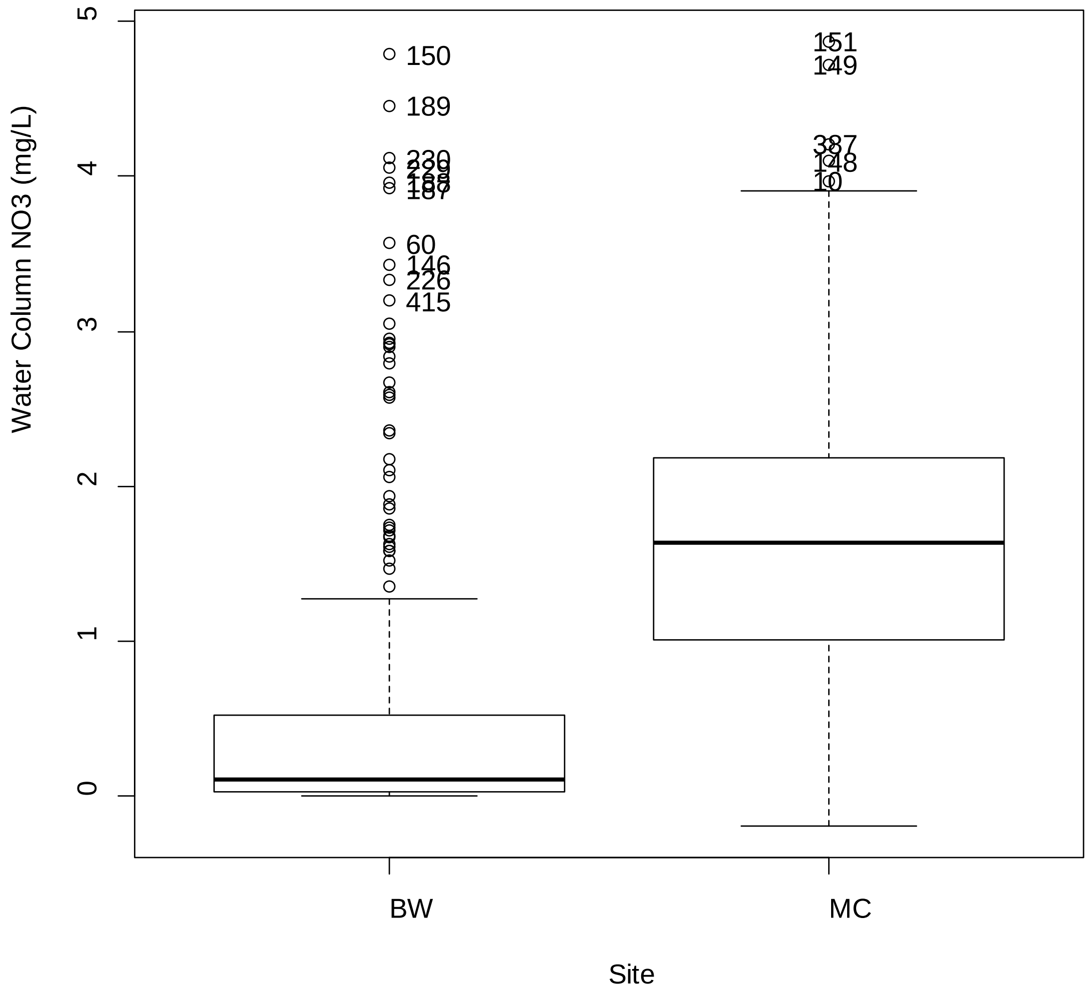

Figure 5. Box plots showing water column NO_3 (mg/L) samples from two sites: MC and BW (1989-2013). Open circles indicate extreme values for each site.

Mud Lake Hydrological/Nutrient Simulation Model

The primary objective of this research was to model how the timing and volume of flow into the Mud Lake project area can affect EGS benefits in the form of nitrogen processing for potential use as a planning tool in ecosystem restoration projects. An integrated hydrological and nutrient simulation modeling approach was used to assess potential nutrient uptake benefits of three management configurations to adjust the amount of flow entering the backwater including a notched weir, a rock closure, and a gated culvert at the head of Mud Lake. Assuming the Brown's Lake analysis confirmed nutrient processes are reasonably comparable to upstream backwaters, the denitrification rates observed by James et al. (2008b) were used as estimates to feed into the Mud Lake model.

Mud Lake was first separated into a series of compartments using a two-dimensional RMA2 hydraulic model as a guide for how water moves through the project area (Figure 6). This model was recently re-calibrated following a 2014 dye study tracking water movement conducted by the USACE and considered more reliable than previous versions (USACE, 2014). The four compartments delineated for Mud Lake were: Channel, Mud Lake, Zollicoffer Slough, and Lower Eddy (Figure 7). Aquatic surface area of each compartment (Table 4) was estimated in ArcGIS 10.2 using elevation, LIDAR, aerial photograph interpretations, and restoration project design features (Esri, 2015). Empirically-observed denitrification rates through the backwater vary based on the interplay between flow and residence time ames et al. 8b . High flow flushes N through the system too quickly for optimal processing (< 300 mg N m^{-2} d^{-1} 1 day) and too little flow restricts N processing capacity to the upper reaches of the project (200 mg N m^{-2} d^{-1} days ames et al. 8b . ptimal residence time 1-1.5 days) occurs at intermediate flow rates and maximizes nutrient processing throughout the entire project area.

These average values were used in a spreadsheet modeling nitrate diffusion in Mud Lake under each of three management alternatives (rock closure; notched weir; and gated culvert), assuming that nutrient uptake rates in backwaters are similar in all reaches of the river (Kreiling et al., 2010; Strauss, 2011). The extracted surface areas of each compartment were multiplied by the average denitrification rates reported for each flow regime (high, medium, and low) and by the number of days in the growing season (gs = 150 days).

Figure 6. Recalibrated RMA2 hydraulic model following a 2014 dye study at Mud Lake HREP, Dubuque County, Iowa. Line thickness indicates relative velocity.

Figure 7. Mud Lake model compartments used to extract aquatic surface areas to estimate potential denitrification benefits. Each homogenous unit was based on a recalibrated RMA2 hydraulic model and represent different flow paths and current velocities.

RESULTS

Brown's Lake vs. Main Channel

Nitrogen concentrations were significantly different between Brown's Lake and main channel sites ($p < 0.01$; $p = 2.2 \times 10^{-16}$; Table 2). Seasonally, water column NO_3 also changed significantly ($p < 0.01$; $p = 2.2 \times 10^{-16}$). Two-way analysis considering the dual effect of site and season on NO_3 concentration resulted in a p value < 0.05 (Table 2). The Tukey Honestly Significant Difference (HSD) test clarified which interactions between the two factors were statistically significant (Table 3), indicating relationships between habitat type and season on water column availability of nitrogen.

During the spring, summer, and fall periods, water column NO_3 was significantly higher in the MC than the BW site ($p < 0.01$). However, there was no significant difference in NO_3 between MC and BW during the winter period ($p > 0.05$). In Brown's Lake, NO_3 was significantly greater in the spring than in the summer period. No significant differences in NO_3 were observed between spring and summer in the main channel, as expected.

Table 2. ANOVA II results of two factor effects (site x season) on water column NO_3, indicating significant effects of both site (Brown's Lake backwater, Pool 13 main channel) and season (Spring, Summer, Fall, Winter) on nutrient concentrations.

Variables Response: NO_3	**Sum Sq**	**Df**	**F value**	**Pr (>F)**
Season	51.7	3	22.9	4.3×10^{-14}****
Site	172.7	1	229.2	2.2×10^{-16}****
Site x Season	9.56	3	4.2	0.01**
Residuals	407.1	605		
*Note: Signif. codes: 0 '****' 0.001 '***' 0.01 '**' 0.05 '+' 0.1 ' '*				

Table 3. Post-hoc Tukey HSD multiple comparisons (site x season) effects on water column NO_3, indicating N limitation in the Brown's Lake backwater (BW) relative to the Pool 13 main channel (MC) during spring, summer, and fall periods.

Biweekly Sample Period x Site	**Dates**	**HSD**	**Lower CI (95%)**	**Upper CI (95%)**	**Adjusted p value**
Winter:MC-Winter:BW	Dec 17-Mar 11	0.05	-0.94	1.04	1.0
Spring:MC-Spring:BW	Mar 12-Jun17	1.04	0.71	1.38	1.0×10^{-7}****
Summer:MC-Summer:BW	Jun 18-Sept-23	1.26	0.91	1.62	1.0×10^{-7}****
Fall:MC-Fall:BW	Sept 24-Dec 16	1.06	0.51	1.60	2.0×10^{-7}****
Summer:BW-Spring:BW	N/A	-0.53	-0.89	-0.18	1.7×10^{-4}***
Summer:MC-Spring:MC	N/A	-0.31	-0.64	0.03	0.1
*Note: Signif. codes: 0 '****' 0.001 '***' 0.01 '**' 0.05 '+' 0.1 ' '*					

Mud Lake Hydrological/Nutrient Simulation Model

The notch weir model provided the closest approximation to the existing condition. In this modeled management alternative (Table 4), both the Channel and Lower Eddy experienced moderate to high current velocity rates (~2.0 $m^3 s^{-1}$) and low hydraulic residence times (< 1 day). According to James et al. (2008b), this condition corresponded to an average denitrification rate of 300 mg N $m^{-2} d^{-1}$. The Zollicoffer Slough and Mud Lake compartments experienced low flow rates and high hydraulic residence times (> 5 days), resulting in an average denitrification rate of 200 mg N $m^{-2} d^{-1}$. Total average denitrification potential for the notch weir management alternative was 3.52×10^{10} mg N $m^{-2} gs^{-1}$ (gs = growing season, or 150 days).

The rock closure alternative assumed that the removal of the main channel connection would result in low current velocity rates and high hydraulic residence times for each of the compartments in the project area. The total average denitrification potential for the rock closure management alternative was 2.94×10^{10} mg N $m^{-2} gs^{-1}$.

Finally, the gated culvert alternative was assumed to offer the most precision in achieving both moderate current velocity rates and optimal hydraulic residence times for nutrient processing. The total average denitrification potential for the gated culvert management alternative was 7.36×10^{10} mg N $m^{-2} gs^{-1}$.

Table 4. Mud Lake Hydrological-Nutrient Simulation Model results indicating the gated culvert structure has the highest denitrification potential than either the notch weir or rock closure.

Alternative	Compartment	Name	Area	Flow	τ	Denitrification rate	Time	Total N Denitrified	Alternative N Denitrified
			m^2	H/M/L	H/M/L	$mg/m^2/d$	days	$mg/m^2/gs$	$mg/m^2/g$
	1	Channel	154,228	H	L	300	150	6.94E+09	
	2	Mud Lake	198,454	L	H	200	150	5.95E+09	
	3	Zollicoffer Upper	393,808	L	H	200	150	1.18E+10	
	4	Lower Eddy	234,243	H	L	300	150	1.05E+10	
Notch Weir									**3.52E+10**
	1	Channel	154,228	L	H	200	150	4.63E+09	
	2	Mud Lake	198,454	L	H	200	150	5.95E+09	
	3	Zollicoffer Upper	393,808	L	H	200	150	1.18E+10	
	4	Lower Eddy	234,243	L	H	200	150	7.03E+09	
Rock Closure									**2.94E+10**
	1	Channel	154,228	M	M	500	150	1.16E+10	
	2	Mud Lake	198,454	M	M	500	150	1.49E+10	
	3	Zollicoffer Upper	393,808	M	M	500	150	2.95E+10	
	4	Lower Eddy	234,243	M	M	500	150	1.76E+10	
Gated Culvert									**7.36E+10**

DISCUSSION

Brown's Lake vs. Main Channel

The seasonal mean differences in NO_3 between the BW and MC sites were consistent with nutrient trends reported elsewhere in the UMRS. Namely, these results supported the expected outcomes of a seasonal decrease in NO_3 availability in Brown's Lake. In contrast, there was no such observed seasonal decrease within the main channel. Numerous studies in the UMR have associated NO_3 variability with the degree of connectivity to the MC (Richardson et al., 2004; Houser and Richardson, 2010; Houser et al., 2013). Generally, the BW site appeared to exhibit small spikes in NO_3 during the spring, typically coinciding with spring flooding or high rainfall events. The MC also exhibited a similar spike in NO_3 during these sampling periods, suggestive that high NO_3 is related to runoff (James et al., 2008a). This type of seasonal flood discharge rarely permits enough contact time in the backwater to be cycled efficiently (Royer et al., 2004; James et al., 2008b). Without continuous connection to the MC, Brown's Lake BW typically experienced steep declines in NO_3 during later sample periods (late summer to early fall). A proportion of this influx is assimilated rapidly by aquatic vegetation and other metaphyton including algae, but it is the highly organic sediments in backwaters that appear to account for such high denitrification rates compared to other parts of the river (Richardson et al., 2004; James, 2010; Kreiling et al., 2010).

The results also supported the expected outcome that mean BW NO_3 was significantly higher in the spring than in the summer, which is when the highest degree of connection to the main channel was hypothesized to occur. It is also important to note that rivers are unpredictable systems affected by a variety of environmental conditions (i.e., rainfall, discharge, drought, etc.). During some years sampled, the BW experienced variable connection to the MC and did not become NO_3 limited (Figure 3, dashed lines). Similarly, there were some sample periods when the MC had lower NO_3 than were observed in the BW. Long-term monitoring of water quality parameters can help capture the ecosystem response to unpredictable conditions and provide restoration managers with a tool to test hypotheses of different management strategies. An impending post-construction performance evaluation of the Brown's Lake HREP may be an opportunity to calibrate the Mud Lake model and achieve more precise denitrification estimates for this reach of the UMRS.

Mud Lake Hydrological/Nutrient Simulation Model

The notch weir alternative approximated the existing condition at Mud Lake where current velocities are high enough to preclude overwintering in the upper project area. Suitable winter habitat exists, but fish typically restricted use to the downstream project area (USACE, 2014). Summer algal blooms were also consistently reported, a factor likely associated with nitrogen limitation in the backwater (Giblin et al., 2014., 2013; Houser et al., 2013). The rock closure alternative represented the least amount of connection to the main channel to further improve overwintering habitat across the entire project area. However, this alternative represented a trade-off between winter and summer habitat suitability—conditions for algal blooms likely to increase in the summer with sustained low current rates. The closure was designed as leaky riprap to support flow and DO conditions required by overwintering fish.

Overall, the gated culvert alternative would allow for the greatest amount of controlled flow ( ) through the project area. To extend habitat benefits throughout the year, the gated culvert would be operated to achieve optimum nutrient processing in the summer, then closed in the late fall/early winter for fish refuge. However, the installation of the culvert can often be cost-prohibitive for resource managers. Since Mud Lake had an adaptive management component, the aforementioned rock stockpile on site was used to close the notch in the fall of 2015 to mimic the rock closure alternative. Preliminary surveys conducted in the winter by the Iowa Department of Natural Resources reported fish response to this management alternative was indicated by angler success (Scott Gritters, unpublished). It will be interesting to observe how the backwater metaphyton responds during the growing season if flow into the site remains limited. While the spreadsheet model used here assumed total potential denitrification rates can be applied across the UMR, models can and should be calibrated when locally-relevant datasets become available

Future Model Integration of Phosphorus and Carbon

Based on earlier studies (Richardson et al., 2004; James et al., 1995), there appear to be two main mechanisms for nutrient input to backwater lakes: the main source of nitrogen is positively associated with hydraulic loading and the main sources and sinks for phosphorus are associated with diffusive sediment flux (James and Barko, 2008). While nutrient runoff from agricultural fields and wastewater are large contributors of the phosphorus budget in rivers and streams, the main source of P outside the main channel appears to be more related to P release from sediments (James and Barko, 2004). While accounting for P within a backwater complex should prove fairly simple by measuring P at the inflow and outflow sites, internal processes are more difficult to measure. One thing is fairly certain, phosphorus does not appear to be limiting in many of these backwater habitats (Houser and Richardson, 2010; James et al., 2004). Low N:P (< 24:1) tend to create conditions that lead to harmful algal blooms witnessed during the late summer in the UMR. (Demars and Edwards, 2007). The availability of P has been demonstrated to efficiently increase N removal in many freshwater ecosystems (Finlay et al., 2013). With regards to the management of Mud Lake and other backwaters, increasing connectivity may have the potential to inject a large source of anthropogenic nutrient inputs to reduce local algal blooms while also reducing downstream nutrient transport. Incorporating P into the Mud Lake model will require adequate data collection to examine the balance of constituents in the project area to ensure the most efficient management strategy can be selected.

The carbon cycle in backwater lakes is arguably more complex and there are several approaches to quantifying ecosystem metabolism by calculating atmospheric and sediment gas exchanges (Cole et al., 2000; Demars et al., 2007; Houser et al., 2015). While such approaches are outside the scope of this study, the diel oxygen (O_2) method may have the greatest relevance to future estimation of carbon benefits using the Mud Lake model (Staehr et al., 2010). In short, aquatic dissolved oxygen (DO) represents the balance of metabolic production and respiration rates, which can be estimated using high frequency sonde data. These data can be quickly imported into open software environments for statistical computation of ecosystem metabolism using continuous DO data (e.g. R package "LakeMetabolizer" [v.1.3.3; Winslow et al., 2015]). The USACE already deploys DO sondes at many HREP locations for performance monitoring, so there is future potential for incorporating carbon cycling benefits to the EGS model

Conclusions

The potential for diverting N through backwater habitats for increased nutrient processing is substantial in the UMR, but is alone unlikely to result in significantly offsetting N delivery to the Gulf of Mexico (Rabalais, 2002; Richardson et al., 2004; Strauss et al., 2011). Future reductions in nutrient pollution to freshwater and coastal ecosystems will necessarily require a concerted effort from a number of disciplines including agriculture, urban municipalities, natural resource management, levee and drainage districts, etc. Furthermore, there are limitations to consider when increasing hydrologic connectivity in backwaters for the purpose of biogeochemical cycling, including increased sediment loads and changes to macrophyte community structure (Houser and Richardson, 2010). The immensity of increased N flux in the UMRS need not deter resource managers from incorporating EGS in habitat restoration planning to benefit fish and wildlife. The nutrient simulation model used here is recommended as a planning tool for projects designed within an adaptive management framework as a means to explore habitat benefits beyond the overwintering season. A well-designed EGS model should be adjustable to meet the objectives of each individual restoration project and flexible enough to help resource managers to balance trade-offs and select the most cost-effective alternative in a climate of scarce funding sources (Merenlender and Matella, 2013).

As the field of ecosystem restoration evolves to engineer solutions at a system-wide scale, future habitat restoration plans can be designed to achieve multiple ecosystem benefits across seasons without compromising the immediate goals of individual projects. The integration of EGS in adaptive management plans can enable managers to apply what is learned about the system, to support smarter planning and greater efficiency, and to adjust management to enhance the functionality of both current and future restoration projects.

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Attachment E

2014 Dye Study: Winter Water Circulation Patterns in Mud Lake – An Adaptive Management Study of a Backwater Habitat Restoration Project in Pool 11 of the Mississippi River

WINTER WATER CIRCULATION PATTERNS IN MUD LAKE - AN ADAPTIVE MANAGEMENT STUDY OF A BACKWATER HABITAT RESTORATION PROJECT IN POOL 11 OF THE MISSISSIPPI RIVER

By

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Photo looking upstream into the Mud Lake HREP inlet (Courtesy of Elizabeth Bruns).

Winter Water Circulation Patterns in Mud Lake - An Adaptive Management Study of a Backwater Habitat Project In Pool 11 of the Mississippi River

Abstract

U.S. Army Corps of Engineers, Rock Island District (USACE) personnel performed a dye study during March 2014 in a backwater of the Mississippi River, Pool 11, near Dubuque, Iowa. The study was conducted in response to Iowa Department of Natural Resources (IDNR) fish telemetry data which indicated that newly created dredge channels were underutilized by overwintering fish; and velocity data that indicated Mississippi River main channel flow was entering the backwater area from the dredge channel outlet. A habitat restoration project for the backwater was completed in 2005 as part of the Upper Mississippi River Restoration program. The project included creation of deep water dredge channels in the backwater adjacent to the navigation channel to provide overwintering habitat for centrarchids and associated species.

The primary purposes of the study were to determine how inflowing water disperses, both temporally and spatially, throughout the backwater complex during winter, under ice cover; and to measure velocity, a critical factor in the selection of overwintering areas utilized by centrarchids. A single slug injection of Rhodamine WT dye was dispensed immediately downstream from the inlet structure to the backwater and was tracked for more than 24 hours as it dispersed throughout the area. When initial results indicated the dye was not traversing the full length of the main dredge channel, a second injection was dispensed in the dredge channel outlet. The results of the study suggest that implementation of adaptive management measures is necessary in order to reduce dredge channel velocities to acceptable levels for overwintering fish. The results also substantiated velocity data collected by IDNR and Wisconsin Department of Natural Resources (WDNR) personnel which indicated Mississippi River main channel flow enters the backwater area from the dredge channel outlet.

Introduction

The Pool 11 Islands Habitat Rehabilitation and Enhancement Project (HREP) under the Upper Mississippi River Restoration program includes two distinct backwater enhancement areas: Mud Lake and Sunfish Lake. A location map is included in Figure 1. All work related to the present study was performed in the Mud Lake HREP, which is located on the Mississippi River (river miles 587.6 to 589.4), approximately five miles upstream from L/D 11 and the City of Dubuque, Iowa. Construction of the Mud Lake project commenced in August 2004 and was completed in July 2005 (USACE, 2014). The HREP consists of Mud Lake at the upstream portion of the backwater area and Zollicoffer Slough at the downstream portion, with the mouth of Leisure Creek forming a depositional area between the two water bodies (see Figure 2). The recommended plan for the HREP included construction of a 3,038 m sediment deflection embankment to protect the backwater complex from sediment accretion /resuspension and mechanically dredging 8.8 ha of deep channels for fish overwintering habitat (USACE, 2001). Dredge material was used to construct the deflection embankment and an island near the lower portion of the project which was adjacent to a channel connecting Zollicoffer Slough with the main dredge channel. As part of the original design process for the Mud Lake HREP, a two dimensional hydrodynamic model (RMA-2) was utilized to evaluate various alternatives for the project. The recommended alternative included two notched rock weirs in the deflection embankment: one at the upper end and one near the middle. The primary purpose of the weirs was to allow oxygenated main channel flow into the backwater area during the winter months to help assure sufficient DO concentrations to support overwintering fish. A DO mass balance performed during project design indicated an inflow of 1.09 cm/sec would be necessary to

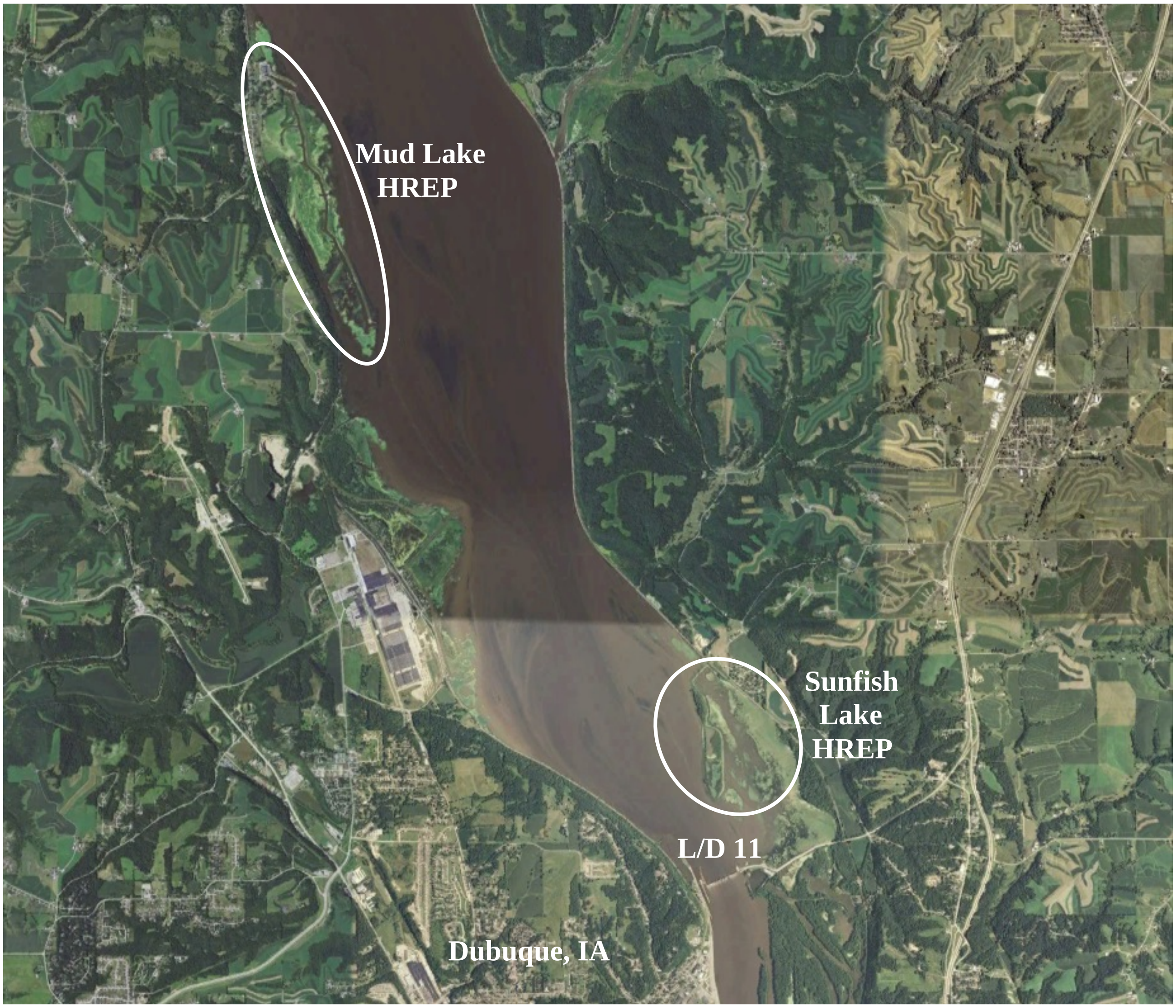

Figure 1. Location map for the Mud Lake and Sunfish Lake HREPs.

maintain a DO of 5 mg/L in the backwater. The RMA-2 model was used to size the inlets for the required inflow. Following project construction, both USACE and IDNR personnel measured velocities in the dredge channels that were excessive for overwintering centrarchids. In 2006, adaptive management measures were incorporated to reduce the inflow. The opening in the middle of the deflection embankment was completely filled with rock, while the opening at the upper end was partially filled. This resulted in a significant reduction in velocity in the dredge channels during ensuing winters; however, IDNR fish telemetry studies have indicated the HREP is still underutilized by overwintering centrarchids and velocities continue to be excessive. According to Scott Gritters (IDNR, personal communication, April 2, 2014), at the start of winter, centrarchids in the HREP prefer to stage in areas with zero flow.

In addition to issues involving velocity magnitude, velocity direction has also been a concern. A study performed jointly by IDNR and WDNR staff on February 22, 2008 indicated Mississippi River main channel flow enters the backwater area from the dredge channel outlet. The present study was performed

Figure 2. Mud Lake HREP project features.

in order to better define velocities and circulation patterns in the backwater complex in an effort to explain the underutilization of Mud Lake by overwintering fish.

Methods

As part of the district's HREP performance evaluation monitoring program, USACE personnel performed water quality sampling at Mud Lake on March 6, 2014. This trip provided an opportunity to gather reconnaissance data for the upcoming dye study. The inlet structure was investigated in order to determine ice coverage and an appropriate method for dispensing the dye, and velocity measurements were taken in order to estimate dye travel times. Ice condition and thickness were also determined at several sites in order to assess the level of effort that would be required for completing the dye study.

Sample site locations were determined prior to performing the dye study by utilizing Google Earth Pro software. Historical imagery was viewed in order to select a recent image (September 22, 2011) that provided the best view of the dredge channels and other deep areas in the backwater complex, which were readily recognized as areas devoid of emergent vegetation. The software pointer was placed on the location of each proposed sampling site and the geographical coordinates were recorded. Most of the sampling sites were located in dredge channels, while some were located in Zollicoffer Slough and other areas throughout the backwater. In this initial exercise, 21 sampling locations were identified (see Figure 3).

The fluorescent dye used for the study was a 20 percent solution of Rhodamine WT manufactured by Crompton and Knowles. To determine the amount of dye required for the study, the area of the backwater was estimated using the ruler function in Google Earth Pro. Average water depths were estimated for the dredge channels (1.5 m), Zollicoffer Slough (2.7 m) and the remainder of the backwater complex (0.3 m). The depth for each stratum was multiplied by the area to calculate water volume. The three volumes were added to determine the total water volume of the backwater complex (534,128 m^3). This value was compared to the volume calculated for Spring Lake (11,280,000 m^3), where a previous dye study was conducted. In the Spring Lake study, it was estimated that 3.5 liters of dye would be required to dye the lake to a concentration of 100 ppt (Bierl, 2002), the approximate level of detection. The volume of Spring Lake is considerably greater than Mud Lake; however, to account for dye fluorescence decay which may have occurred during storage, it was conservatively estimated that 3.75 liters of dye would be sufficient for the Mud Lake study.

Waypoints stored on a GPS (Trimble TSC1 datalogger/Pro XR receiver) were used to locate the 21 sampling sites on the first day of the study (March 10, 2014). The sites were marked with orange spray paint, holes were drilled through the ice and measurements were taken. Sites 2, 8, 9, 10 and 19 were found to have insufficient water depth to allow for collection of a representative water sample; thus, these sites were eliminated from further study. Site 12, located on the river side of the rock-filled notch in the deflection embankment was also dropped from further study when water was observed flowing from the river side of the notch to the backwater side (this site was initially considered due to the possibility of dye exiting the backwater area here). At the remaining sites, water depth, ice thickness, snow depth, dissolved oxygen (DO), water temperature and velocity were recorded. DO and water temperature values were measured at the surface (10 cm below the bottom of the ice), mid-depth (1/2 the water depth) and bottom (10 cm above the bottom) with a YSI Pro ODO Meter. A Sontek FlowTracker ADV was used for taking velocity measurements at the surface. At selected sites, pH was measured at the surface with an Extech Instruments pH100 meter and a depth integrated water sample was collected and analyzed for background fluorescence with a Turner Designs Model 10-AU fluorometer.

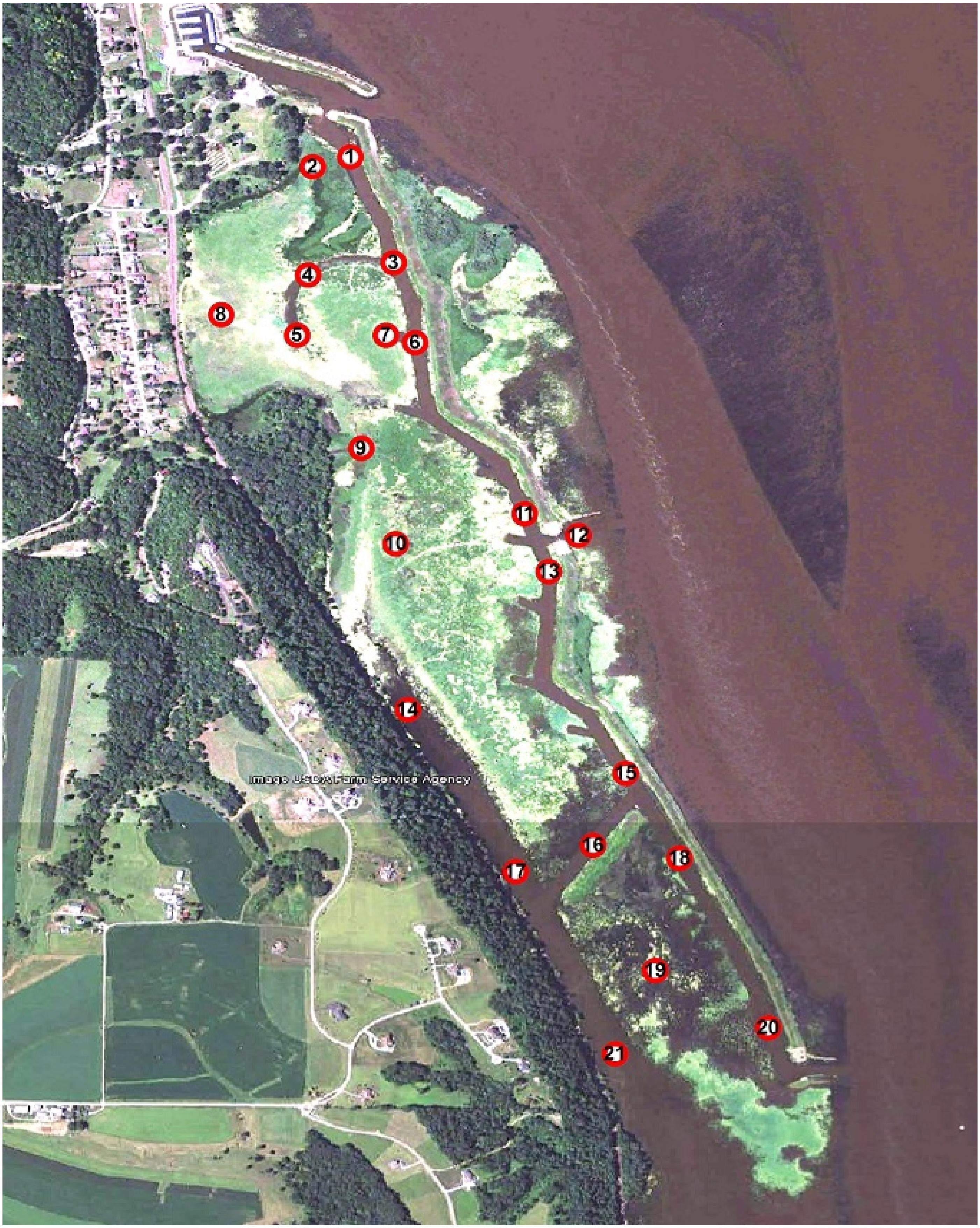

Figure 3. Mud Lake HREP initial 21 sampling locations.

On the morning of March 11, 2014, water collected from the inlet channel of the backwater area was mixed with Rhodamine WT dye in a 151 liter plastic drum fitted with a spigot and a one meter discharge tube (see Figure 4). In order to facilitate assimilation of the dye with the inflow, 3.75 liters of dye were mixed with 121 liters of river water. This helped reduce the viscosity of the dye and equilibrate the temperature of the dye with that of the inflowing river water in order to allow for more complete mixing. A single slug injection of the dye commenced at 0830 hours and was completed by 0900 hours. The dye was then tracked.

Figure 4. Dye delivery apparatus.

A water sample was collected at each site with a 2.8 m length of ½-inch diameter EMT conduit with back-to-back #0 conduit hangers fastened near one end (see Figure 5). A 40 ml, amber glass vial with silicon septum screw cap was snapped into place in the conduit hanger. The narrow opening of the cap (following removal of the silicon septum) allowed the bottle to fill relatively slowly; thus, allowing for sample collection throughout the depth profile. The sampling apparatus was lowered into the hole until it approached the bottom and was then raised at the same rate. This allowed for a depth integrated sample. Following collection, a portion of the sample was poured into a 13 mm cuvette and immediately analyzed for the presence of dye with the fluorometer. This process helped assure the temperature of all samples was similar; thus, minimizing the impact of temperature variation on dye concentration. According to Johnson (1984), Rhodamine WT fluorescence decreases approximately five percent for every 2°C increase in temperature. In order to prevent cross-contamination, the sampling apparatus and ice auger/chisel were rinsed with non-dye tainted river water after each sample containing dye was collected. DO, water temperature and velocity measurements were taken at selected sites to determine if these parameters changed significantly from day one of the study.

Figure 5. Dye sampling apparatus.

Once dye tracking commenced, additional sampling sites were identified (see Figure 6) in order to locate the leading edge of the dye at various times. When the sampling results indicated dye had not reached site 18 at the predicted time, a second dye injection was made in the dredge channel outlet to validate velocity data collected by IDNR and WDNR personnel in 2008 which indicated Mississippi River main channel flow enters the backwater area from the dredge channel outlet. The second slug injection of dye commenced at 1214 hours and was completed by 1224 hours on March 12, 2014. This injection was administered in a similar fashion as the first injection; however, approximately one-half the amount of dye and river water was mixed in the drum. Water samples were collected at the three sampling sites located near the outlet (sites 26-28) in order to verify the dye's direction of travel.

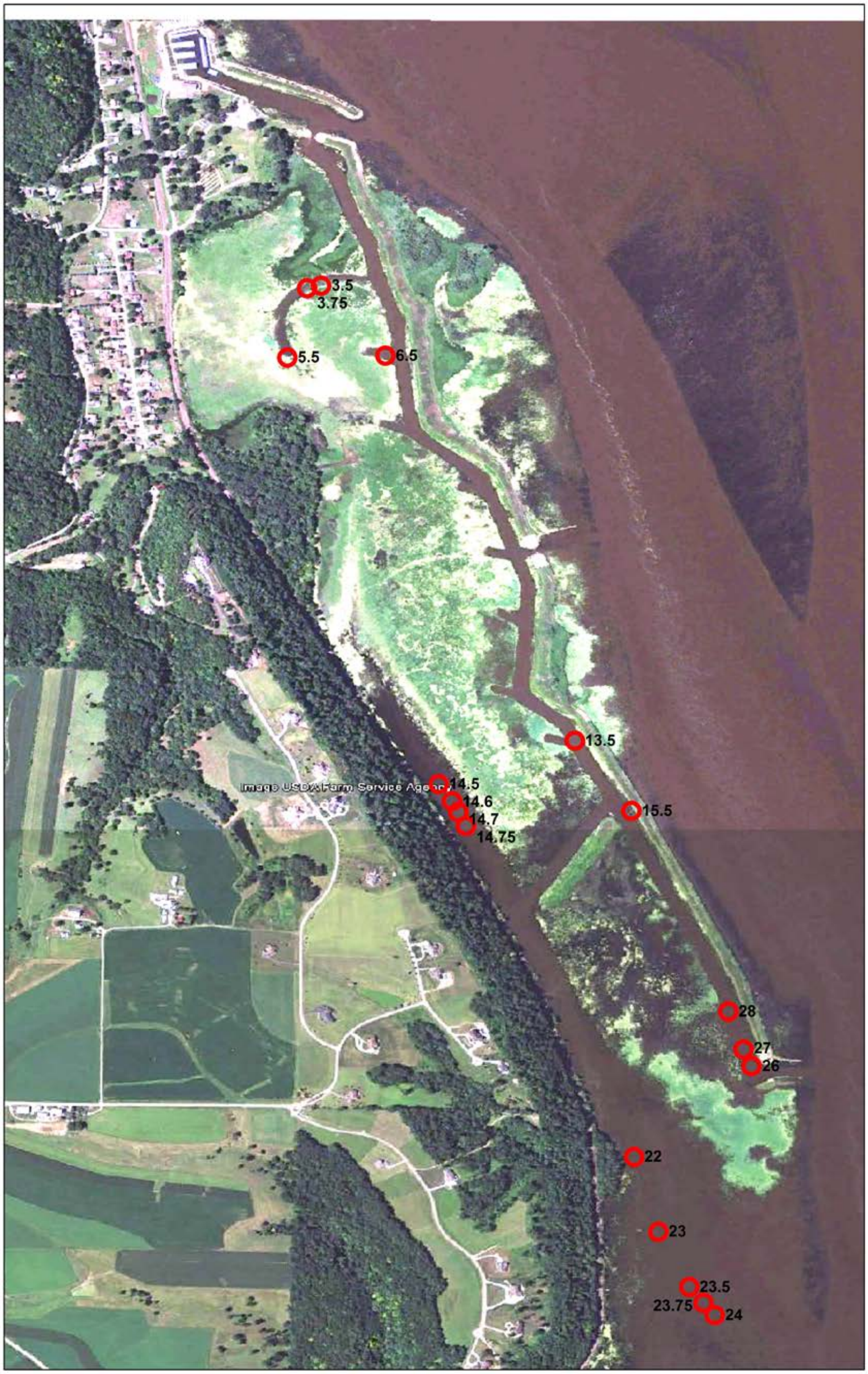

Figure 6. Mud Lake HREP additional sampling sites.

Results and Discussion

The Mississippi River elevation during the study was close to the long-term historic average as measured at the Lock and Dam 11 gage (see Figure 7). Over the course of the study, the river rose approximately 0.5 feet. The 6:00 a.m. river elevations on March 10, 2014 and March12, 2014 were 605.61' and 606.07' upstream at the Guttenberg, Iowa gage and 593.62' and 594.15' downstream at the Lock and Dam 11

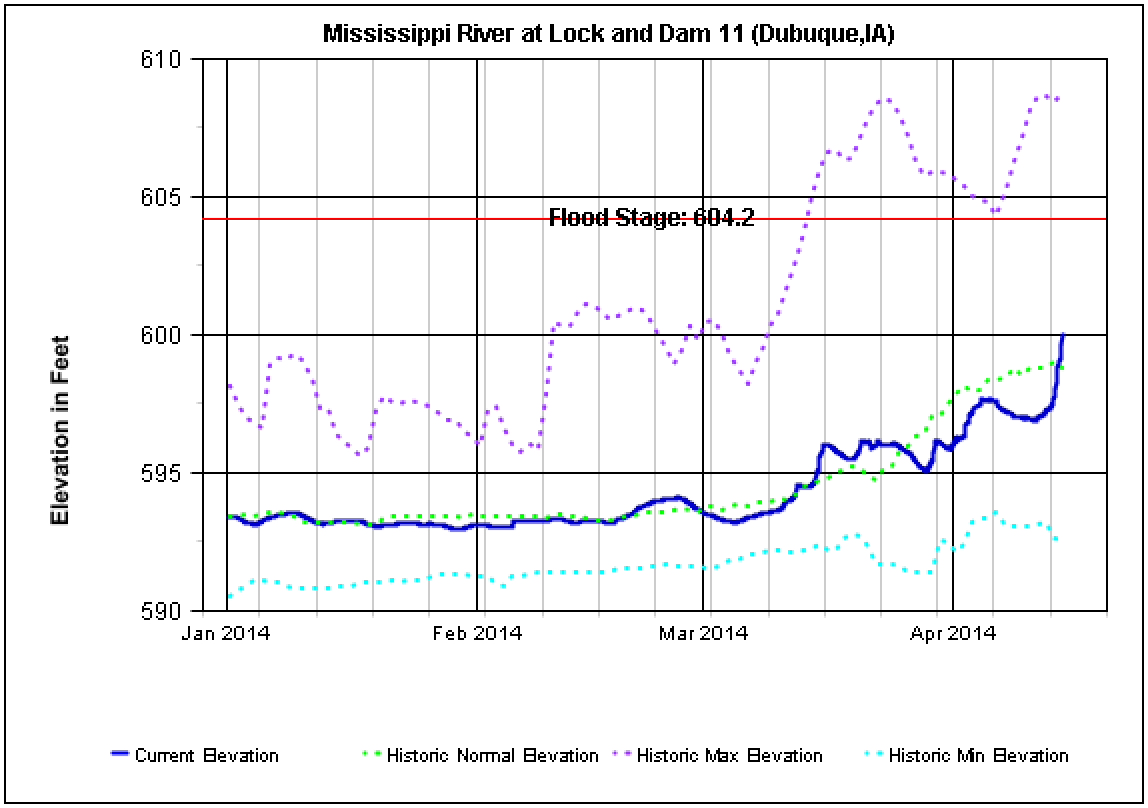

Figure 7. Mississippi River elevation at L/D 11 (Dubuque, IA).

gage, respectively. Field data collected on March 10, 2014 are given in Table 1. The winter of 2013/2014 was one of the coldest on record; thus, ice thickness was much greater than during a typical winter, with values ranging from 27.9 cm at site 16 to 66.0 cm at site 17. The combination of thick ice and shallow water depth at sites 2, 8, 9, 10 and 19 precluded collection of a representative water sample; therefore, these sites were eliminated from further study. Site 12, which was initially considered due to the possibility of dye exiting the backwater area here, was also dropped from further study when water was observed flowing from the river side of the rock-filled deflection embankment notch into the backwater side. The remaining sites, all located within dredge channels or in Zollicoffer Slough, had water depths ranging from 1.49 m at the upper end of Zollicoffer Slough (site 14), to 2.74 m at the lower end of Zollicoffer Slough (site 21). The average depth of sites located in dredge channels was 1.89 m, with the deepest area (> 2.0 m) located in the middle of the main dredge channel (sites 11, 13 and 15) and the shallowest area (1.50 m) at site 20, near the dredge channel outlet. Snow was present at all sites with depths ranging from 2.5 cm at several locations to 10.2 cm at site 17. All surface DO values in the backwater area were below saturation, but were more than sufficient to support aquatic life. Concentrations varied little, ranging from 11.34 to 12.34 mg/L. Mid-depth and bottom DO concentrations were similar to surface values, except for sites 4, 5, 14 and 21, where bottom concentrations were lower. The most prominent stratification occurred in the curved dredge channel in Mud Lake (sites 4 and 5). Here, in addition to low bottom DO concentrations (3.66 and 1.92 mg/L, respectively), velocity was also low (0.32 and 0.35 cm/s, respectively). Stratification was less prominent

Table 1. Field data collected on March 10, 2014, prior to dye dispersal.

Site*	Time	Water Depth (m)	Ice (cm)	Snow (cm)	D.O. (mg/L)	Water Temp. (°C)	Velocity (cm/s)	pH	Dye Blank (µg/L)
1S	1708	1.89	35.6	2.5	12.34	0.5	6.17	7.62	0.770
M					12.37	0.4			
B					12.38	0.3			
3S	1640	1.78	45.7	2.5	12.21	0.5	5.37		
M					12.24	0.4			
B					12.21	0.4			
4S	1653	1.88	53.3	7.6	11.97	0.7	0.32		
M					12.00	0.7			
B					3.66	1.4			
5S	1550	1.98	58.4	5.1	11.58	0.7	0.35	7.48	0.953
M					11.88	0.8			
B					1.92	1.6			
6S	1627	1.85	48.3	7.6	11.98	0.6	5.20		
M					12.03	0.4			
B					12.04	0.4			
7S	1608	1.74	61.0	5.1	11.70	0.6	0.41		
M					11.68	0.5			
B					11.62	0.7			
11S	1519	2.23	50.8	5.1	11.71	0.5	3.94		
M					11.64	0.4			
B					11.55	0.4			
12	1506	0.46	43.2	2.5	15.43	0.6			
13S	1449	2.19	50.8	5.1	11.42	0.6	3.56	7.50	0.848
M					11.45	0.5			
B					11.43	0.5			
14S	1238	1.49	55.9	5.1	11.34	0.5	0.24	7.49	0.731
M					11.25	0.4			
B					9.08	0.7			
15S	1225	2.20	43.2	7.6	11.55	0.5	4.03		
M					11.60	0.3			
B					11.50	0.2			
16S	1213	1.70	27.9	7.6	11.93	0.3	6.55	7.48	0.737
M					11.83	0.3			
B					11.72	0.2			
17S	1150	2.46	66.0	10.2	11.63	0.4	0.64		
M					11.55	0.4			
B					11.35	0.4			
18S	1127	1.68	55.9	5.1	11.68	0.5	3.02	7.52	
M					11.72	0.4			
B					11.77	0.3			
20S	1049	1.50	53.3	7.6	11.62	0.6	3.60	7.46	0.754
M					11.76	0.2			
B					11.77	0.2			
21S	1024	2.74	53.3	7.6	11.84	0.4	2.10	7.49	0.853
M					11.88	0.3			
B					8.57	1.0			

* "S" readings taken at 10 cm under the ice, "M" at 1/2 water depth, "B" at 10 cm off of bottom.
Sites 2, 8, 9, 10 and 19 were too shallow to sample.

at sites 14 and 21 located in Zollicoffer Slough (bottom DO concentrations of 9.08 and 8.57 mg/L, respectively). A similar stratification pattern was seen with water temperature, where most values changed little throughout the depth profile and surface values ranged from 0.3 to 0.7°C. Again, sites 4, 5, 14 and 21 showed some stratification, with bottom temperatures ranging from 0.7 to 1.6°C. Another parameter which showed a narrow range of variance throughout the backwater area was pH: surface values ranged from 7.46 to 7.62. Sample fluorescence blanks were collected at sites 1, 5, 13, 14, 16, 20 and 21 in order to determine background concentrations, which ranged from 0.731 to 0.953 µg/L.

All velocity readings in the main dredge channel exceeded 3.00 cm/s, ranging from 3.02 cm/s at site 18 to 6.17 cm/s near the inlet (site 1). Surprisingly, the highest velocity measured was 6.55 cm/s at site 16, in the angled channel that connects the main dredge channel with Zollicoffer Slough. Velocity measurements at sites 18 and 20 validated the findings of a 2008 IDNR/WDNR study which indicated Mississippi River main channel flow enters the backwater area from the dredge channel outlet. During the present study, it is surmised that flow from the outlet continued up the main dredge channel past site 18 until the vicinity of the dredge material island, where it either joined the flow coming from above and was routed through the angled dredge channel to Zollicoffer Slough, or was deflected toward Zollicoffer Slough just below the island, or perhaps some combination of the two scenarios. Upon entering Zollicoffer Slough, the flow split with the majority coursing downstream. Figure 8 displays the general direction and velocity of flow in the backwater complex on March 10, 2014. Lower velocities were measured in the dredge channel in Mud Lake (0.32 and 0.35 cm/s at sites 4 and 5, respectively), in a short dredge cut off of the main dredge channel (0.41 cm/s at site 7) and at sites 14 and 17 in Zollicoffer Slough (0.24 and 0.64 cm/s, respectively).

Figure 8. Mud Lake HREP velocities on March 10, 2014.

Following injection of dye at the inlet on the morning of March 11, 2014, tracking commenced. Sampling sites were added as needed in an effort to locate the leading edge of the dye before it arrived at the next established sampling point. Based on the background fluorescence concentrations measured and several initial fluorometer readings, it was determined a positive "hit" for dye would be a concentration

1. g L. Fluorescence concentration results for each site including the date and time of measurement are included in Table 2. The dye had reached site 1 before there was an opportunity to collect a sample. At most sites, at least one measurement was taken before the dye was detected; thus, giving a good indication as to when the leading edge of the dye plume had arrived. At others, dye was detected on the first measurement; therefore, it was difficult to estimate how much time had lapsed since the leading edge of the dye plume had passed. No samples were collected between 2134 hours on March 11, 2014 and 0822 hours on March 12, 2014; thus, at sites 4 and 5 the dye had likely already passed before it could be detected. Dye was detected at site 5.5 at 0846 hours on March 12, 2014; however, this may have been the trailing edge of the dye plume. The dye transited the main dredge channel until it reached the dredge material island, where flow traveling up from the outlet essentially deflected the dye to the southwest. This was substantiated because the dye arrived at site 16 at 2003 hours but did not arrive at site 15.5, at the northeast tip of the dredge material island, until approximately 2107 hours. The delay in arrival of dye at site 15.5, coupled with the non-arrival of dye at site 18, indicates an upstream movement of flow in the main dredge channel below the island. It is surmised that site 15.5 is located in an eddy where the two flow paths meet; however, lacking additional data it is difficult to determine if the majority of the flow from the outlet was deflected along the upstream or downstream side of the island.

Once the dye passed site 16, it entered Zollicoffer Slough. Here, a majority of the dye flowed downstream, while a small portion traveled upstream. The approximate elapsed time (hours) the dye took to reach selected sampling sites is given in Figure 9. The leading edge of the dye reached the farthest downstream Zollicoffer Slough site (23.75) in 25.4 hours, while it took approximately 30.3 hours to arrive at site 14.6, which was located in Zollicoffer Slough, just upstream of the dredge material island.

A second dye injection took place midday on March 12, 2014 in order to verify the upstream direction of flow from the outlet. Dye was injected in the outlet at 1214 hours and after 0.5 hour had arrived at site 26 and following 2.2 hours was detected at site 28; thus, confirming the upstream direction flow.

Conclusions and Recommendation

A Rhodamine WT dye study was performed during March 2014 in Mud Lake, a backwater of the Mississippi River, Pool 11, near Dubuque, Iowa. The study was conducted in response to the underutilization by overwintering fish of newly created dredge channels, and velocity data that indicated Mississippi River main channel flow was entering the backwater area from the dredge channel outlet. The results from the study indicate velocities in Mud Lake still exceed the level preferred by centrarchids in early winter. The results also confirmed the upstream travel of flow from the dredge channel outlet.

It is imperative that additional adaptive management measures be investigated in order to reduce velocities in Mud Lake so the area provides a viable overwintering site. It is recommended the initial RMA-2 model be revised to reflect as-built conditions and utilize data collected in the present study for model calibration. The updated model could be utilized to evaluate new adaptive management strategies for reducing/redirecting flow.

Table 2. Rhodamine WT concentrations from samples collected on March 11 and 12, 2014.

Site	Date	Time	Dye (µg/L)*	Site	Date	Time	Dye (µg/L)*
3	3/11/2014	9:33	0.557	11	3/11/2014	13:51	0.571
3	3/11/2014	9:40	0.540	11	3/11/2014	13:58	0.841
3	3/11/2014	9:45	0.732	11	3/11/2014	14:03	0.836
3	3/11/2014	9:48	0.748	11	3/11/2014	14:08	1.03
3	3/11/2014	9:50	0.763	11	3/11/2014	14:13	2.18
3	3/11/2014	9:53	0.800	13	3/11/2014	14:46	0.787
3	3/11/2014	9:57	0.775	13	3/11/2014	14:52	0.726
3	3/11/2014	10:02	0.757	13	3/11/2014	14:58	0.767
3	3/11/2014	10:07	1.09	13	3/11/2014	15:04	0.865
3	3/11/2014	10:13	19.2	13	3/11/2014	15:09	0.803
3.5	3/11/2014	13:30	0.504	13	3/11/2014	15:14	0.970
3.5	3/11/2014	15:51	1.54	13	3/11/2014	15:19	1.39
3.75	3/11/2014	17:13	0.833	14.5	3/12/2014	10:31	0.584
3.75	3/11/2014	17:20	1.61	14.5	3/12/2014	14:53	0.684
3.75	3/11/2014	21:34	0.688	14.6	3/12/2014	10:37	0.782
4	3/11/2014	13:25	0.774	14.6	3/12/2014	14:49	2.33
4	3/11/2014	15:56	0.681	14.7	3/12/2014	10:12	3.60
4	3/11/2014	16:38	0.502	14.75	3/12/2014	10:25	4.43
4	3/11/2014	16:57	0.678	15	3/11/2014	19:08	3.08
4	3/11/2014	18:34	0.573	15.5	3/11/2014	21:07	6.12
4	3/11/2014	18:46	0.539	16	3/11/2014	19:20	0.503
4	3/11/2014	18:56	0.563	16	3/11/2014	19:30	0.485
4	3/11/2014	21:33	0.671	16	3/11/2014	19:41	0.479
5	3/12/2014	8:22	0.965	16	3/11/2014	19:51	0.637
5	3/12/2014	8:32	0.781	16	3/11/2014	20:03	1.12
5.5	3/12/2014	8:46	1.58	17	3/11/2014	21:18	0.471
6	3/11/2014	10:47	0.629	17	3/12/2014	9:04	3.61
6	3/11/2014	10:54	0.685	18	3/11/2014	20:11	0.495
6	3/11/2014	10:59	0.717	18	3/11/2014	20:24	0.462
6	3/11/2014	11:04	0.714	18	3/11/2014	20:36	0.461
6	3/11/2014	11:09	0.757	18	3/12/2014	10:08	0.626
6	3/11/2014	11:14	0.766	21	3/12/2014	9:13	3.13
6	3/11/2014	11:19	0.708	22	3/12/2014	9:21	14.3
6	3/11/2014	11:24	3.67	23	3/12/2014	9:31	12.3
6.5	3/11/2014	12:24	1.04	23.5	3/12/2014	9:47	3.44
6.5	3/11/2014	13:13	6.92	23.75	3/12/2014	9:52	1.82
6.5	3/11/2014	15:31	15.5	24	3/12/2014	9:38	0.885
7	3/11/2014	11:42	0.518	20**	3/12/2014	13:23	0.760
7	3/11/2014	11:49	0.455	20**	3/12/2014	13:36	0.648
7	3/11/2014	11:59	0.634	20**	3/12/2014	13:46	77.5
7	3/11/2014	12:09	0.678	26**	3/12/2014	12:36	0.844
7	3/11/2014	12:17	0.649	26**	3/12/2014	12:42	>100
7	3/11/2014	13:11	0.469	27**	3/12/2014	12:49	0.915
7	3/11/2014	13:36	0.696	27**	3/12/2014	12:51	0.974
7	3/11/2014	14:17	0.635	27**	3/12/2014	12:54	1.11
7	3/11/2014	14:23	0.693	27**	3/12/2014	12:58	0.977
7	3/11/2014	14:28	0.728	27**	3/12/2014	13:05	0.959
7	3/11/2014	14:33	0.569	27**	3/12/2014	13:16	18.1
7	3/11/2014	14:38	0.594	28**	3/12/2014	13:56	0.705
7	3/11/2014	15:25	0.713	28**	3/12/2014	14:03	0.697
7	3/11/2014	15:33	0.643	28**	3/12/2014	14:13	0.754
7	3/11/2014	15:43	0.728	28**	3/12/2014	14:27	>100
7	3/11/2014	16:44	5.37				

* Shaded concentration indicates dye detected.
** Site tracked as part of second dye injection.

Figure 9. Mud Lake HREP Rhodamine WT dye travel times (hours).

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Attachment F

2016 Dye Study: Winter Water Circulation Patterns in Mud Lake – An Adaptive Management Study of a Backwater Habitat Restoration Project in Pool 11 of the Mississippi River

Comments on the 2016 Dye Study provided by the USFWS to be incorporated into the following report at a later date:

1. This is good water quality information. Figures 8 and 9 clearly depict how changing the upper inlet affected the interior flows. This could be useful for other HREPs such as Conway Lake.

2. The term "adaptive management" is used throughout the document, but there is no adaptive management plan for this project.

3. The report seems to be focus on centrarchids, though based on this report it could be concluded that the overwintering velocity is not suitable for any fish. Also, the word "underutilized" is used frequently, but is not defined or associated with a measurement.

4. The conclusions state recommendations for fisheries actions, but there is no fisheries data included or referenced to support those recommendations.

5. The report seems to indicate that flow entering from the lower end is a concern. As previously discussed before, there will always be flows entering from the bottom, and those velocities will continue to increase as the flows from the upper and middle inlets are reduced.

6. The report does not address if these velocities meet the design criteria. It is stated that there is not a clear answer to this since different velocities and criteria were used in design.

7. It would be helpful and appreciated if the report acknowledged that this study was performed on the Upper Mississippi River National Wildlife and Fish Refuge.

WINTER WATER CIRCULATION PATTERNS IN MUD LAKE FOLLOWING INLET MODIFICATION - AN ADAPTIVE MANAGEMENT STUDY OF A BACKWATER HABITAT RESTORATION PROJECT IN POOL 11 OF THE MISSISSIPPI RIVER

By

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December, 2016

Rock-filled upper inlet of the Mud Lake portion of the Pool 11 Islands HREP (Courtesy of Scott Gritters, Iowa DNR).

Winter Water Circulation Patterns in Mud Lake following Inlet Modification - An Adaptive Management Study of a Backwater Habitat Restoration Project in Pool 11 of the Mississippi River

Abstract

A habitat restoration project for Mud Lake, a backwater in Pool 11 of the Mississippi River, was completed in 2005 as part of the Upper Mississippi River Restoration program. The project included creation of deep-water dredged channels in the backwater adjacent to the Mississippi River navigation channel to provide overwintering habitat for centrarchids and associated species. U.S. Army Corps of Engineers, Rock Island District (USACE) personnel performed a dye study during March 2014 in the backwater in response to Iowa Department of Natural Resources (IDNR) fish telemetry data, which indicated that newly created dredged channels were underutilized by overwintering fish, and velocity data that indicated Mississippi River main-channel flow was entering the backwater area from the dredged channel outlet at the downstream end of the project. The results from the 2014 study verified that flow was entering the dredged channel "outlet" (also referred to as the lower inlet) and also indicated velocities exceeded those preferred by overwintering fish; thus, prompting a modification in the fall of 2015 to reduce flow entering the upper inlet of the backwater. The modification consisted of filling the upper inlet with rock. A subsequent dye study was performed during February 2016 to determine the effectiveness of the modification in reducing flow through the upper inlet and to ascertain its impact on flow entering the lower inlet.

The primary purposes of the two dye studies were to determine how inflowing water disperses both temporally and spatially throughout the backwater complex during winter under ice cover and to measure velocity, a critical factor in the selection of overwintering areas utilized by centrarchids. For the 2016 study, a slug injection of Rhodamine WT dye was dispensed downstream from the upper inlet of the backwater, followed immediately by an additional dye injection in the lower inlet. The dye was tracked for more than 48 hours as it dispersed throughout the backwater complex. The results of the study showed that a significant reduction in velocities occurred in the upper portion of the project following closure of the upper inlet. Velocities entering the project through the lower inlet, however, still remain high, suggesting that implementation of additional adaptive management measures may be necessary.

Introduction

The Pool 11 Islands Habitat Rehabilitation and Enhancement Project (HREP) under the Upper Mississippi River Restoration program includes two distinct backwater enhancement areas: Mud Lake and Sunfish Lake (see Figure 1). All work related to the present study was performed in Mud Lake, which is located on the Mississippi River (river miles 587.6 to 589.4), approximately five miles upstream from L/D 11 and the City of Dubuque, Iowa. Construction of the Mud Lake project commenced in August 2004 and was completed in July 2005 (USACE, 2014). The project area consists of Mud Lake at the upstream portion of the backwater area and Zollicoffer Slough at the downstream portion, with the mouth of Leisure Creek forming a depositional area between the two water bodies (see Figure 2). The recommended plan for the project included construction of a 3,038 m sediment deflection embankment to protect the backwater complex from sediment accretion /resuspension and mechanically dredging 8.8 ha of deep channels for fish overwintering habitat (USACE, 2001). Dredged material was used to construct the deflection embankment and an island near the lower portion of the project which was adjacent to a channel connecting Zollicoffer Slough with the main dredged channel.

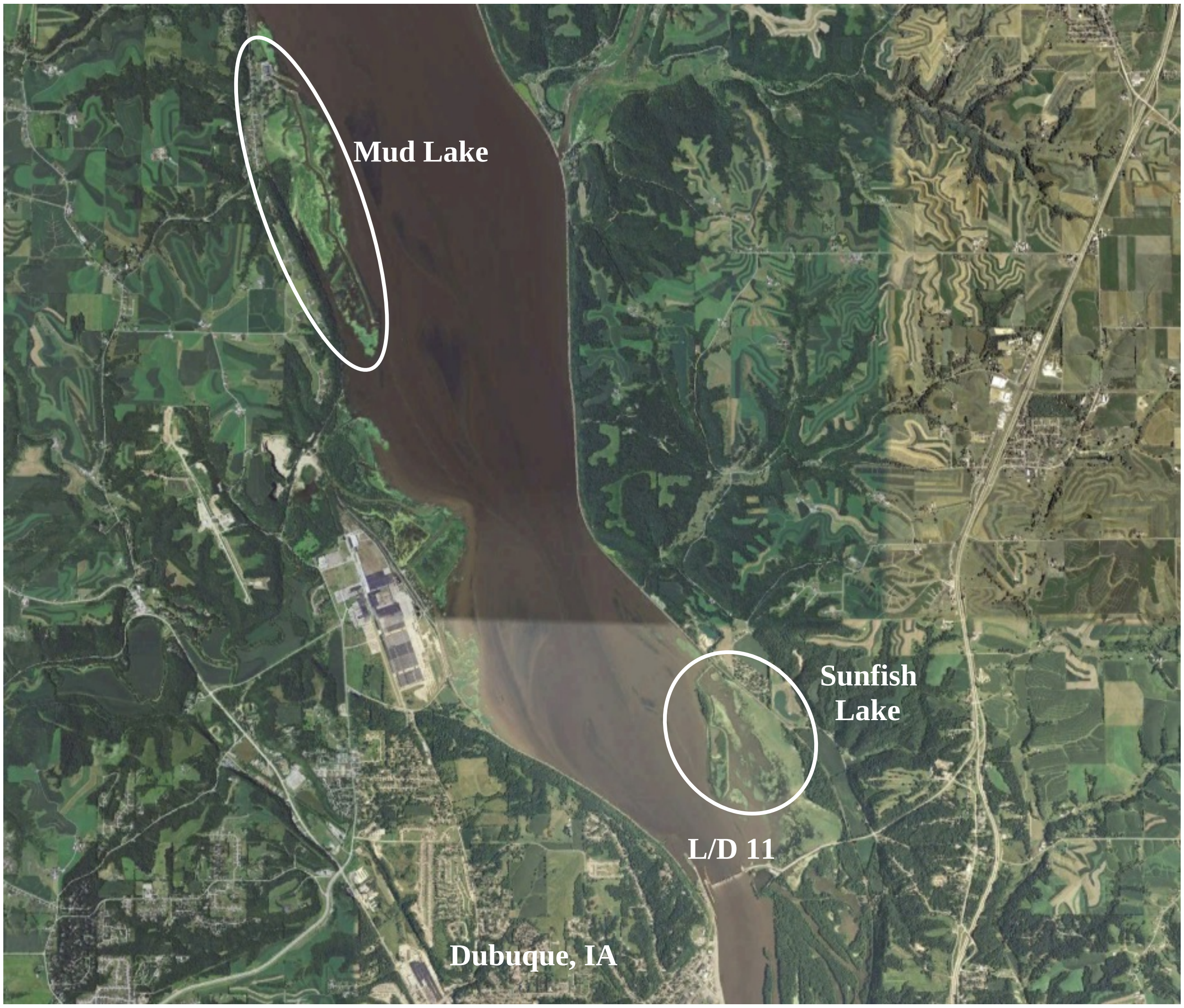

Figure 1. Location map for Mud Lake and Sunfish Lake in the Pool 11 Islands HREP.

As part of the original design process for the Mud Lake project, a two-dimensional hydrodynamic model (RMA-2) was utilized to evaluate various alternatives for the project. The recommended alternative included two notched rock weirs in the deflection embankment: one at the upper end and one near the middle. The primary purpose of the weirs was to allow oxygenated main channel flow into the backwater area during the winter months to help assure sufficient dissolved oxygen (DO) concentrations to support overwintering fish. A DO mass balance performed during project design indicated an inflow of 1.09 cm/sec would be necessary to maintain a DO of 5 mg/L in the backwater. The RMA-2 model was used to size the inlets for the required inflow.

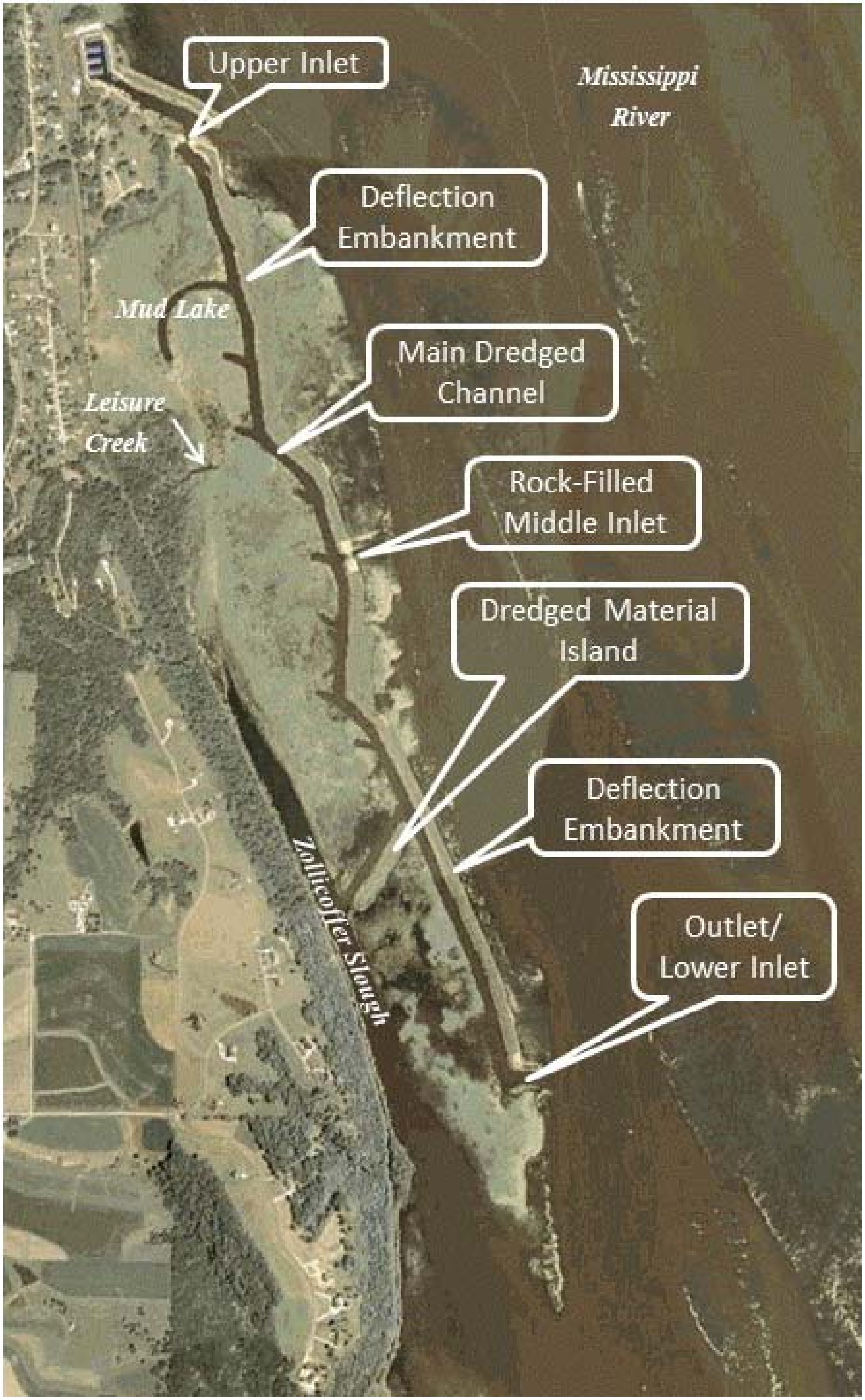

Figure 2. Mud Lake HREP project features.

Following project construction, both USACE and Iowa Department of Natural Resources (IDNR) personnel measured velocities in the dredged channels that were excessive for overwintering centrarchids. In 2006, adaptive management measures were incorporated to reduce the inflow. The opening in the middle of the deflection embankment was completely filled with rock, while the opening at the upper end was partially filled. This change resulted in a significant reduction in velocity in the dredged channels during ensuing winters; however, IDNR fish telemetry studies have indicated the HREP is still underutilized by overwintering centrarchids and velocities continue to be excessive. According to Scott Gritters (IDNR, personal communication, April 2, 2014), at the start of winter, centrarchids in the HREP prefer to stage in areas with zero flow.

In addition to issues involving velocity magnitude, velocity direction has also been a concern. A study performed jointly by IDNR and Wisconsin Department of Natural Resources (WDNR) staff on February 22, 2008 indicated Mississippi River main-channel flow enters the backwater area from the lower inlet. This was also verified by USACE in a 2014 dye study (Bierl, 2016).

In response to the persistent high velocities and underutilization of the project area by overwintering fish, another adaptive management modification was completed in the fall of 2015, which entailed filling the upper inlet with rock. The present study was performed in February 2016 in order to better define velocities and circulation patterns in the backwater complex following the project modification.

Methods

Information gathered and lessons learned by USACE personnel during the 2014 dye study were incorporated when possible during the 2016 study. Most of the 2016 initial sampling sites were the same as those utilized in 2014. Sample site locations were determined by utilizing Google Earth Pro software. Historical imagery was viewed in order to select a recent image (September 22, 2011) that provided the best view of the dredged channels and other deep areas in the backwater complex, which were readily recognized as areas devoid of emergent vegetation. The software pointer was placed on the location of each proposed sampling site and the geographical coordinates were recorded and converted with Corpscon software to NAD83 IL West State Plane, US Survey Feet for entry as waypoints into the GPS unit (see Table 1). Most of the sampling sites were located in dredged channels, while some were located in Zollicoffer Slough. For the 2016 study, most of the 2014 sites were used; sites outside the flow path or with insufficient depth were eliminated. In this initial exercise, 18 sampling locations were identified (see Figure 3). Once dye tracking commenced, additional sampling sites were identified in order to locate the leading edge of the dye at various times.

The fluorescent dye used for the study was a 20 percent solution of Rhodamine WT manufactured by Crompton and Knowles. Determination of the amount of dye required was according to the methods described in Bierl (2016) however, to account for additional dye fluorescence decay which may have occurred during storage, it was conservatively estimated that four liters of dye would be sufficient for the upper inlet injection and two liters for the lower inlet injection.

Figure 3. Mud Lake HREP dye sampling locations.

Table 1. Sampling Site Coordinates.

Site	Easting	Northing	Site	Easting	Northing
1	2152897.203	2167844.339	14.12	2153331.844	2162791.305
3	2153169.360	2166899.202	14.2	2153355.334	2162711.972
4	2152619.255	2166795.296	14.23	2153371.198	2162672.730
4.12	2152570.925	2166735.029	14.25	2153368.914	2162633.614
4.25	2152530.382	2166686.375	14.5	2153547.219	2162318.718
4.5	2152491.568	2166574.450	14.7	2153678.673	2162061.122
4.65	2152489.179	2166488.204	15	2154633.913	2162339.758
4.75	2152507.271	2166405.982	15.5	2154788.225	2162026.958
5	2152547.010	2166255.457	16	2154417.932	2161699.895
5.75	2153196.911	2163017.568	17	2153921.584	2161471.158
6	2153312.427	2166215.378	17.5	2154112.597	2160999.381
7	2153118.734	2166252.224	18	2154974.044	2161582.403
10.75	2153954.312	2164783.071	19	2154812.588	2160580.960
11	2153998.700	2164656.402	20	2155535.254	2160067.644
11.25	2154042.780	2164464.787	21	2154550.354	2159840.956
11.75	2154099.078	2164324.410	22	2154841.205	2158913.044
13	2154151.498	2164140.046	24	2155375.382	2157469.093
13.25	2154117.442	2163450.135	29	2155404.990	2160475.182
13.5	2154466.259	2162707.304	30	2155259.558	2160882.430
13.55	2154523.741	2162580.388	31	2155131.934	2161230.705
13.65	2154561.392	2162492.135	36	2153253.339	2166545.730
13.75	2154601.156	2162413.391	316	2153190.926	2166839.800
14	2153234.078	2162917.643	326	2153211.606	2166760.061

* Coordinates are NAD83 IL West State Plane, US Survey Feet.

Waypoints stored on a GPS (Trimble TSC1 datalogger/Pro XR receiver) were used to locate the 18 sampling sites on the first day of the study (February 15, 2016). The sites were marked with orange spray paint, holes were drilled through the ice and measurements were taken. Site 19 was found to have insufficient water depth to allow for collection of a representative water sample; thus, this site was eliminated from further study. At the remaining sites, water depth, ice thickness, snow depth, dissolved oxygen (DO), water temperature, pH and velocity (magnitude and direction) were recorded. DO and water temperature values were measured at the surface (10 cm below the bottom of the ice), mid-depth (1/2 the water depth) and bottom (10 cm above the bottom) with a YSI Pro ODO Meter. A Sontek FlowTracker ADV was used for taking velocity measurements at the surface. An Extech Instruments pH100 meter was used to measure pH. At selected sites, a depth integrated water sample was collected and analyzed for background fluorescence with a Turner Designs Model 10-AU fluorometer.

Figure 4. Dye delivery apparatus.

Figure 5. Dye sampling apparatus.

Water samples were collected with a 2.8 m length of ½-inch diameter EMT conduit with back-to-back #0 conduit hangers fastened near one end (see Figure 5). A 40 ml, amber glass vial with silicon septum screw cap was snapped into place in the conduit hanger. The narrow opening of the cap (following removal of the silicon septum) allowed the bottle to fill relatively slowly; thus, allowing for sample collection throughout the depth profile. The sampling apparatus was lowered into the hole until it approached the bottom and was then raised at the same rate to allow for a depth-integrated sample. Following collection, a portion of the sample was poured into a 13 mm cuvette and immediately analyzed for the presence of dye with the fluorometer. This process helped assure the temperature of all samples was similar; thus, minimizing the impact of temperature variation on dye concentration. According to Johnson (1984), Rhodamine WT fluorescence decreases approximately five percent for every 2°C increase in temperature. In order to prevent cross-contamination, the sampling apparatus, ice auger, and chisel were rinsed with non-dye tainted river water after each sample containing dye was collected.

On the morning of February 16, 2016, water collected from just below the upper inlet was mixed with Rhodamine WT dye in a 151 liter plastic drum fitted with a spigot and a one meter discharge tube (see Figure 4). In order to facilitate assimilation of the dye with the inflow, four liters of dye were mixed with 106 liters of river water. This dilution reduced the viscosity of the dye and equilibrated the temperature of the dye with that of the inflowing river water in order to allow for more complete mixing. A slug injection of the dye immediately below the upper inlet commenced at 0828 hours and was completed by 0853 hours.

A second dye injection, in the lower inlet, commenced at 0931 hours and was completed by 0946 hours. Here, two liters of dye were mixed with 61 liters of river water. During this injection, the dye froze in the valves of the delivery apparatus, thus necessitating their removal along with the attached tubing. The dye was then poured slowly from the opening in the drum into the hole in the ice.

Following injection of dye at the upper and lower inlets on the morning of February 16, 2016, tracking commenced. Based on velocities measured the preceding day, dye injected in the lower inlet was tracked first. At most sites, at least one measurement was taken before the dye was detected; thus, giving a good indication as to when the leading edge of the dye plume had arrived. At others, dye was detected on the first measurement; therefore, it was difficult to estimate how much time had lapsed since the leading edge of the dye plume had passed. Tracking was done by a single team during daylight hours for approximately 48 hours. Because the team was only on site during the day and they were tracking dye on multiple fronts, sampling sites were added as needed in an effort to locate the leading edge of the dye before it arrived at the next established sampling point.

Determination of when the leading edge of the dye reached a particular sampling site was usually readily apparent but occasionally was less evident. During these instances, three factors were considered in order to make a determination: the background fluorescence on February 15, 2016; the initial post-injection fluorescence value at a particular site; and whether consecutive readings at a particular site were stable, rising or falling. At site 13, for example, the background fluorescence on February 15, 2016 was 1.40 µg/L; however, six readings taken on February 17, 2016 were all below 1.00 µg/L. It wasn't until February 18, 2016 that a reading of 18.9 µg/L indicated dye had reached the site, although the leading edge had likely passed.

Results and Discussion

The Mississippi River elevation rose slightly, less than 0.2 feet, during the course of the study and stayed within the normal navigation pool limits as measured at the Lock and Dam 11 Pool gage (see Figure 6). The 8:00 a.m. river elevations on February 15 and 18, 2016 were 606.57'[1] and 606.75' upstream at the Lock and Dam 10 Tailwater gage. The respective measurements downstream at the Lock and Dam 11 Pool gage were 602.87' and 602.90'. Water levels during the 2016 dye study were slightly less than those measured during the March 10-12, 2014 study (see Figure 7). The interpolated water surface elevations at River Mile 589.3, at the upper inlet, may not exactly reflect water levels in the backwater due to the berm and inlet structures.

Field data collected on February 15, 2016 are given in Table 2. The winter of 2013/2014 was one of the coldest on record; thus, ice thickness during the 2014 dye study was greater than during the 2016 dye study. Ice thickness ranged from 30 to 66 cm during the 2014 study and from 18 to 44 cm during the 2016 study. In light of the thinner ice conditions during 2016, an attempt was made to sample a site located outside of a dredge cut (site 19), downstream from the dredged material island; however, conditions still precluded collection of a representative water sample.

[1] All river elevations reference the MSL 1912 datum.

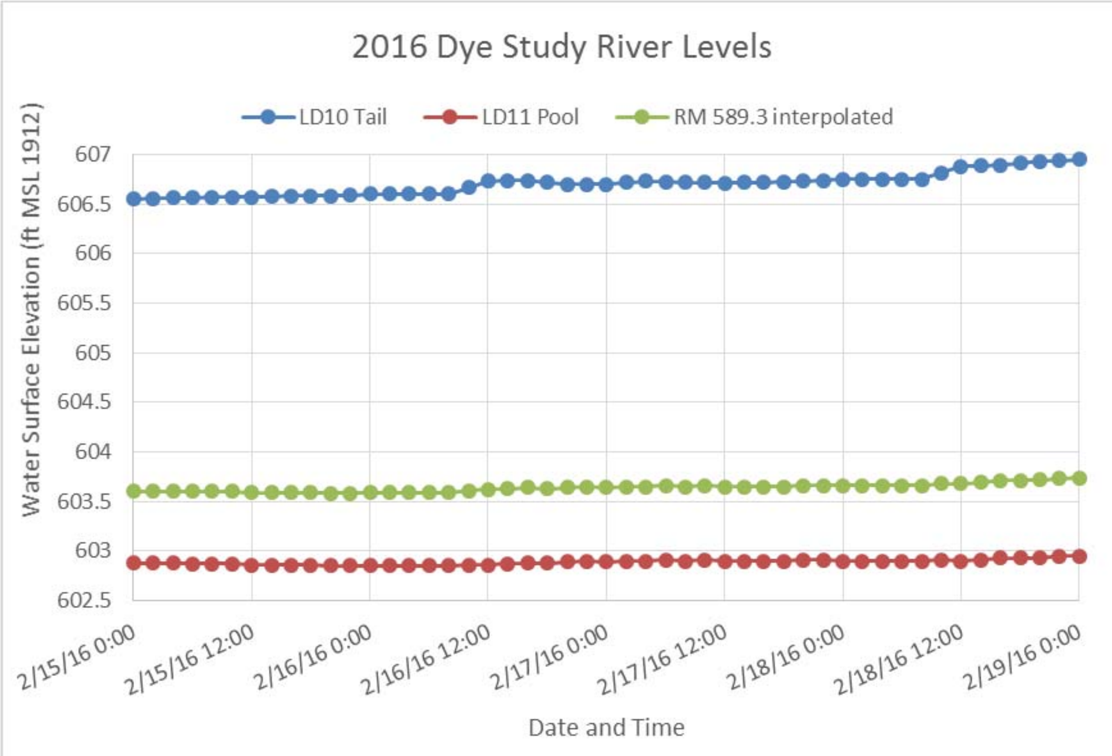

Figure 6. Mississippi River elevations during the 2016 dye study.

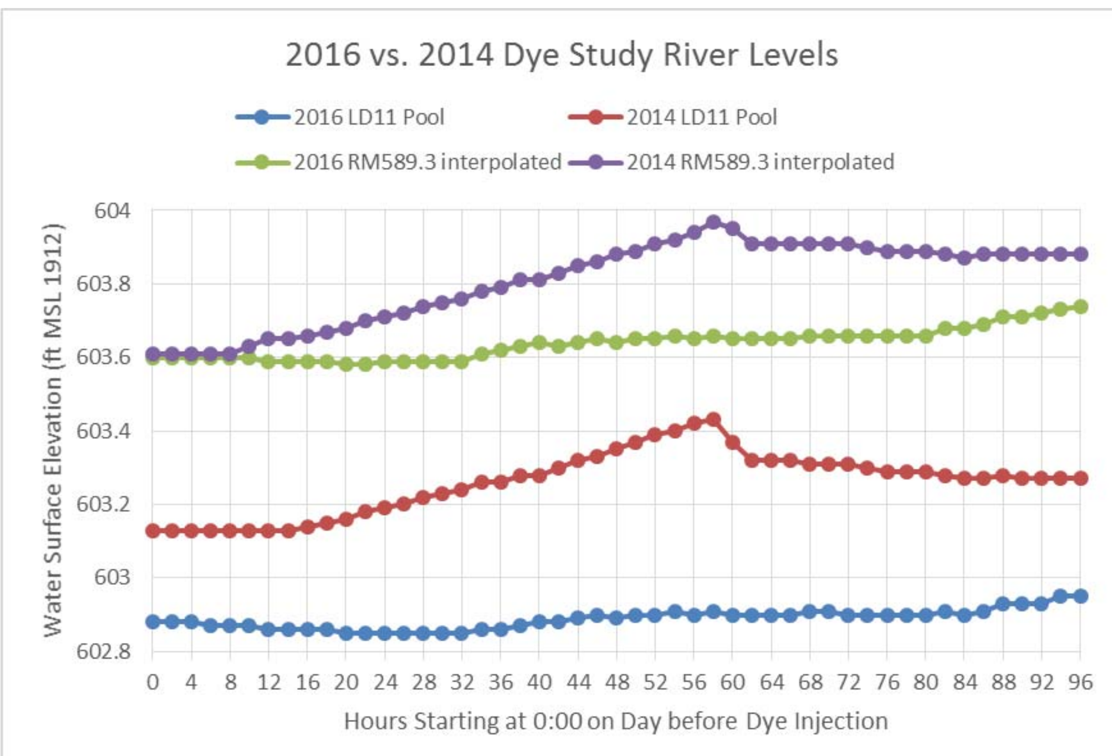

Figure 7. Mississippi River elevations during the 2014 and 2016 dye studies.

The average water depth of sites located in dredged channels was 1.69 m, with the deepest area (2.04 m) located in the middle of the main dredged channel at site 11 and the shallowest area (1.34 m) at site 20, near the dredged channel outlet. Snow was present at all sites with depths ranging from 3 cm at several locations to 8 cm at site 22. All DO values in the backwater area exceeded 13 mg/L, with concentrations ranging from 13.56 to 22.36 mg/L. DO stratification was minimal where water velocity was the highest (sites 18 and 20) and it varied at the remaining sites. Surprisingly, the most prominent stratification occurred in the main dredged channel at site 3, where the surface DO was 15.45 mg/L and the bottom DO was 22.36 mg/L. The velocity here was 1.01 cm/s, while several sites with lower velocity did not exhibit as prominent stratification.

A similar stratification pattern was seen with water temperature, where the sites with the highest velocity (18 and 20) had the least stratification and it varied at the remaining sites. Water temperature ranged from 0.0 to 2.0°C, with the greatest stratification (1.6°C) at site 22. Sites 3, 13, and 15 had the next highest difference between the surface and bottom temperatures (1.4°C). The pH of surface measurements taken throughout the backwater complex ranged from 7.89 at site 22 to 8.69 at site 13. Sample fluorescence (dye) blanks were collected at sites 1, 5, 13, 14, 15, 16, 17, 20 and 21 in order to determine background concentrations, which ranged from 0.775 to 1.40 µg/L.

The most significant change in velocity following closure of the upper inlet occurred at sites 1, 3 and 6 located in the upper third of the main dredged channel. The velocity at these sites during the 2014 dye study ranged from 5.20 to 6.17 cm/s, while the 2016 values at these sites were significantly lower, ranging from 1.00 to 1.10 cm/s. Another noticeable difference between the pre- and post-2015 modification velocity measurements was in the lower inlet. Velocities at sites 18 and 20 were 3.02 and 3.60 cm/s in 2014 and 6.31 and 6.04 cm/s respectively, in 2016. In addition, the flow entering the lower inlet extended farther up the main dredged channel in 2016. During the 2014 study, the flow at site 15 was moving in the downstream direction; while during 2016, flow here was in the upstream direction, continuing to approximately site 13.5. Difficulty obtaining a repeatable velocity measurement at site 13 indicated an eddy may have been present. One velocity measurement here indicated a slight upstream flow while another indicated a slight downstream flow. Thus, this site was likely where the flow paths from the upper and lower inlets met.

Figures 8 and 9 display the general magnitude and direction of flow during the 2014 and 2016 studies. Flow moving along the dredged-material island splits upon entering Zollicoffer Slough, with some coursing downstream and some upstream. Unlike in 2014, the velocity at site 21 was less than the velocity at site 17 during 2016. Similar to 2014, lower velocities were measured in the dredged channel in Mud Lake (0.14 cm/s at both sites 4 and 5), in a short dredge cut off of the main dredged channel (0.22 cm/s at site 7) and at site 14 in Zollicoffer Slough (0.10 cm/s).

Table 2. Field data collected on February 15, 2016, prior to dye dispersal.

Site*	Time	Water Depth (m)	Ice (cm)	Snow (cm)	D.O. (mg/L)	Water Temp. (°C)	Velocity (cm/s)	pH	Dye Blank (µg/L)
1S	0956	1.74	41	4	14.95	0.2	1.10	8.06	1.15
M					14.67	0.4			
B					16.78	1.3			
3S	1030	1.71	41	5	15.45	0.4	1.01	8.02	
M					15.94	0.6			
B					22.36	1.8			
4S	1047	1.76	41	5	15.17	0.6	0.14	7.96	
M					19.58	1.0			
B					20.24	1.3			
5S	1100	1.80	41	5	16.28	0.7	0.14	8.14	1.11
M					17.47	0.7			
B					18.12	0.9			
6S	1134	1.67	41	3	15.70	0.5	1.00	8.09	
M					15.95	0.8			
B					20.70	1.8			
7S	1118	1.65	43	3	16.56	0.7	0.22	8.16	
M					16.63	0.7			
B					16.36	1.9			
11S	1151	2.04	39	5	16.73	0.7	0.81	8.12	
M					17.53	1.4			
B					18.64	1.8			
13S	1205	1.99	38	5	18.97	0.6	0.26**	8.69	1.40
M					17.04	1.3			
B					17.89	2.0			
14S	1250	1.38	44	4	16.40	0.5	0.10	8.17	0.911
M					16.63	0.7			
B					22.19	1.5			
14.7S	1308	1.81	44	4	14.92	0.2	0.80	8.06	
M					15.25	0.3			
B					15.68	0.6			
15S	1457	1.57	43	3	14.13	0.1	0.71	7.95	0.855
M					14.09	0.1			
B					16.97	1.5			
16S	1514	1.54	18	5	13.82	0.1	5.42	7.93	0.798
M					13.80	0.2			
B					14.98	0.8			
17S	1321	1.31	44	3	14.40	0.2	1.23	7.95	0.855
M					14.40	0.2			
B					15.63	0.6			
18S	1445	1.47	33	3	13.64	0.0	6.31	7.92	
M					13.60	0.0			
B					13.62	0.0			
20S	1417	1.34	28	3	14.11	0.0	6.04	8.00	0.775
M					14.11	0.1			
B					14.16	0.1			
21S	1338	2.56	44	5	15.39	0.2	0.82	8.56	0.801
M					17.37	0.3			
B					16.35	1.1			
22S	1402	3.20	44	8	13.56	0.1	0.58	7.89	
M					14.92	0.5			
B					14.75	1.7			

* "S" readings taken at 10 cm under the ice, "M" at 1/2 water depth, "B" at 10 cm off of bottom.
** Average of two measurements.

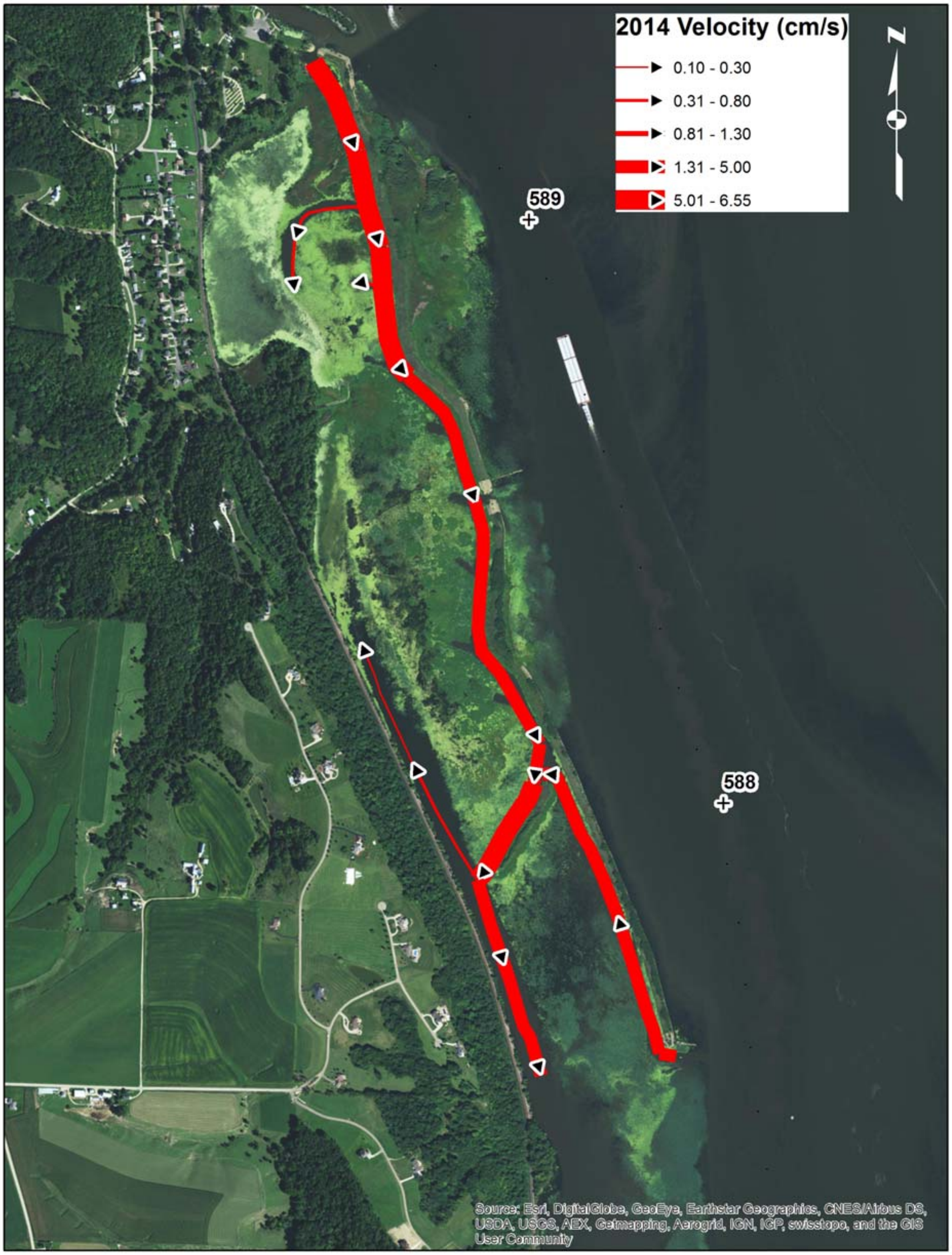

Figure 8. Mud Lake HREP velocities on March 10, 2014.

Figure 9. Mud Lake HREP velocities on February 15, 2016.

The lower inlet dye injection tracking results are shown in Table 3. The dye injection in the lower inlet commenced at 0931 hours on February 16, 2016. The leading edge of the dye had already passed site 20 at 1010 hours when a reading of 36.7 µg/L was taken. The dye quickly moved up the main dredged channel, passing sites 29, 30, 31 and 18 before arriving at site 15.5, next to the dredged material island, at 1306 hours. From here, the majority of the flow moved along the island towards Zollicoffer Slough, but unlike during the 2014 dye study, a smaller portion continued up the main dredged channel to the vicinity of site 13.5.

Once the dye passed site 16, along the dredged material island, it entered Zollicoffer Slough. Here, a majority of the dye flowed downstream, while a smaller portion traveled upstream. The leading edge of the dye had already passed the farthest downstream Zollicoffer Slough site (24) within 24 hours, while it took more than 48 hours to arrive at site 5.75, which was the farthest upstream site located in Zollicoffer Slough. The approximate elapsed time (hours) the dye took to reach selected sampling sites is given in Figure 10.

The dye injected in the upper inlet flowed at a much slower pace than the lower inlet injection. The upper inlet dye injection tracking results are shown in Table 4. The upper inlet injection commenced at 0828 hours on February 16, 2016 and the dye took more than ten hours to reach site 316, at the junction of the curved dredged channel. By comparison, during the 2014 study the dye traveled this distance in approximately 1.5 hours, indicating closure of the upper inlet was successful in reducing velocities in this portion of the project. When tracking resumed in the main dredged channel the following day, the leading edge of the dye had already passed site 6 by 0816 hours and site 11 by 1626 hours but was detected at site 11.75 at 1653 hours. As the dye flowed from the main dredged channel into the curved dredged channel it slowed again. The leading edge of the dye had passed site 4 by 0835 hours and reached site 4.25 at approximately 1724 hours, and site 4.65 the following day at 0842 hours.

Table 3. Lower inlet injection Rhodamine WT dye tracking results.

Site	Date	Time	Dye (µg/L)*
20	2/16/2016	10:10	36.7
29	2/16/2016	10:22	0.820
29	2/16/2016	10:27	0.819
29	2/16/2016	10:32	0.944
29	2/16/2016	10:37	1.02
29	2/16/2016	10:42	>100
30	2/16/2016	10:55	1.16
30	2/16/2016	10:59	0.943
30	2/16/2016	11:03	1.09
30	2/16/2016	11:07	1.01
30	2/16/2016	11:11	1.26
30	2/16/2016	11:16	1.30
31	2/16/2016	11:42	0.828
31	2/16/2016	11:47	1.14
31	2/16/2016	11:52	0.936
31	2/16/2016	11:57	0.701
31	2/16/2016	12:02	7.40
18	2/16/2016	12:17	0.769
18	2/16/2016	12:22	0.803
18	2/16/2016	12:27	0.793
18	2/16/2016	12:33	1.38
15.5	2/16/2016	12:51	1.05
15.5	2/16/2016	12:56	1.18
15.5	2/16/2016	13:01	1.03
15.5	2/16/2016	13:06	1.23
15.5	2/16/2016	13:11	2.88
16	2/16/2016	13:27	1.13
16	2/16/2016	13:32	1.18
16	2/16/2016	13:37	1.11
16	2/16/2016	13:42	1.20
16	2/16/2016	13:47	1.04
16	2/16/2016	13:52	1.03
16	2/16/2016	13:57	1.13
16	2/16/2016	14:02	1.10
16	2/16/2016	14:07	1.88
17.5	2/16/2016	15:35	0.870
17.5	2/16/2016	15:55	1.05
17.5	2/16/2016	16:00	1.21

Site	Date	Time	Dye (µg/L)*
17.5	2/16/2016	16:05	1.40
21	2/17/2016	8:56	1.84
22	2/17/2016	9:03	3.54
24	2/17/2016	9:14	1.73
17	2/16/2016	14:47	0.870
17	2/16/2016	14:57	1.19
17	2/16/2016	15:02	1.07
17	2/16/2016	15:07	1.19
17	2/16/2016	15:12	1.14
17	2/16/2016	15:17	1.21
17	2/16/2016	15:22	1.17
17	2/16/2016	16:14	0.957
17	2/16/2016	16:26	1.04
17	2/16/2016	17:21	0.944
17	2/17/2016	9:26	1.45
14.7	2/17/2016	9:36	4.79
14	2/17/2016	9:44	0.918
14	2/17/2016	17:57	0.832
14	2/18/2016	10:14	3.30
14.5	2/17/2016	9:56	4.05
14.25	2/17/2016	10:10	2.30
14:12	2/17/2016	10:24	0.864
14:12	2/17/2016	18:06	2.39
14.2	2/17/2016	10:34	1.10
14.2	2/17/2016	10:43	1.05
14.2	2/17/2016	10:51	0.933
14.23	2/17/2016	10:40	1.44
14.23	2/17/2016	10:55	1.60
5.75	2/18/2016	10:24	1.21
15	2/16/2016	16:33	3.52
13.75	2/16/2016	16:42	1.83
13.75	2/17/2016	18:27	1.39
13.65	2/16/2016	16:53	1.08
13.65	2/16/2016	16:58	1.14
13.65	2/16/2016	17:03	1.05
13.65	2/16/2016	17:08	0.982
13.65	2/16/2016	17:13	0.901
13.65	2/16/2016	17:43	0.827

* Shaded concentrations indicate dye was detected.

Figure 10. Mud Lake HREP Rhodamine WT dye travel times (hours).

Table 4. Upper inlet injection Rhodamine WT dye tracking results.

Site	Date	Time	Dye (µg/L)*	Site	Date	Time	Dye (µg/L)*
3	2/16/2016	17:51	8.09	11	2/17/2016	12:29	0.879
6	2/16/2016	18:05	0.427	11	2/17/2016	13:34	0.815
6	2/17/2016	8:16	43.7	11	2/17/2016	13:39	0.798
36	2/16/2016	18:15	0.386	11	2/17/2016	16:26	20.1
326	2/16/2016	18:25	0.449	10.75	2/17/2016	13:29	12.0
316	2/16/2016	18:35	1.54	11.25	2/17/2016	16:42	14.4
7	2/17/2016	8:26	0.790	11.75	2/17/2016	16:53	1.10
7	2/17/2016	12:17	3.22	11.75	2/17/2016	16:58	1.35
4	2/17/2016	8:35	2.34	13	2/17/2016	16:33	0.699
5	2/17/2016	8:42	0.835	13	2/17/2016	17:33	0.977
5	2/17/2016	11:29	0.721	13	2/17/2016	17:38	0.977
5	2/17/2016	17:12	0.864	13	2/17/2016	17:43	0.929
5	2/18/2016	8:15	0.677	13	2/17/2016	17:48	0.950
4.5	2/17/2016	11:41	0.774	13	2/17/2016	18:35	0.990
4.5	2/17/2016	17:19	0.820	13	2/18/2016	8:53	18.9
4.5	2/18/2016	8:23	3.25	13.25	2/18/2016	9:03	12.6
4.25	2/17/2016	11:52	0.961	13.5	2/18/2016	9:14	8.04
4.25	2/17/2016	12:10	0.880	13.75	2/18/2016	9:24	1.11
4.25	2/17/2016	17:24	1.58	13.65	2/18/2016	9:29	1.26
4.12	2/17/2016	12:01	1.74	13.55	2/18/2016	9:41	1.92
4.75	2/18/2016	8:35	0.847	15	2/18/2016	9:51	1.25
4.65	2/18/2016	8:42	1.18	18	2/18/2016	10:04	0.978

* Shaded concentrations indicate dye was detected.

Conclusions and Recommendations

A Rhodamine WT dye study was performed during February 2016 in Mud Lake, the site of a backwater rehabilitation project on the Mississippi River in Pool 11 near Dubuque, Iowa. The study was performed to determine the effectiveness of a modification to reduce flow through the upper inlet into the backwater.

Discrete water quality samples collected just prior to the dye study did not reveal adverse effects to dissolved oxygen levels in the project area due to the flow reduction. Concentrations were generally supersaturated, similar to those observed in the 2014 dye study. However, continuous DO monitoring during the critical winter months will better assess the impacts of the flow modification on DO levels in the project area. If over-summering habitat is also a concern, continuous DO monitoring during the summer months is also recommended to ensure adequate DO levels for fish year-round.

The results from the study indicate post-2015 modification velocities in the upper portion of Mud Lake under ice are considerably lower than pre-2015 modification values. A considerable portion of the project, however, is still subject to relatively high velocities from flow entering the lower inlet. It is imperative that additional adaptive management measures be investigated in order to determine if the area currently provides a viable overwintering site for fish.

Radiotelemetry and/or creel studies are two options for making this determination. If overwintering fish continue to underutilize the area, it may be necessary to evaluate new adaptive management strategies for reducing or redirecting flow.

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Johnson, M. C. 1984. *Fluorometric Techniques for Tracing Reservoir Inflows.* U. S. Army Corps of Engineers, Environmental and Water Quality Operational Studies, Instruction Report E-84-1, 46 pp.
USACE. 2014.

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U.S. Army Corps of Engineers. 2001. *Upper Mississippi River System Environmental Management Program, Definite Project Report with Integrated Environmental Assessment (R-13F), Pool 11 Islands Habitat Rehabilitation and Enhancement.* U.S. Army Corps of Engineers, Rock Island District

Attachment G

Forestry Data

Permanent Plot Data Sheet *v 03 February 2011*

Subplot > or = 1" up to less than 5" in DBH,
11.8' radius -15' from plot center. Plot 1- 0˘, Plot 2-120˘, Plot 3- 240˘

Plot #	Tree Species	DBH	Height	Health
1	no trees			
2	CACO15	1.0	7	VIG
3	CEOC	1.2	8	VIG
3	CEOC	1.1	8	VIG
3	CEOC	1.4	9	VIG
3	TIAM	1.2	10	VIG
3	CEOC	2.1	14	VIG
3	ULAM	1.1	9	VIG
3	CEOC	1.8	12	VIG
3	CEOC	1.4	8	VIG

Notable Species
POLYG4
LEVI2
PAQU2
EUAT5

Microplot < 1" DBH, 6.8' plot radius
Plot 1- 0˘, Plot 2-120˘, Plot 3- 240˘ (if < 1' tall use % cover)

Tree Species	Height	# plot 1	# plot 2	# plot 3
ACNE2	2'-4'	2		
ACSA2	<1'			1%
CEOC	<1'		3%	1%
CEOC	1'-2'	1	5	1
CEOC	2'-4'		4	1
FRPE	>6'		2	
GLTR	<1'			1%
MORU2	1'-2'			1
MORU2	4'-6'	1		
QUMA2	1'-2'		1	
QUMA2	2'-4'		1	

Herbaceous Species	% plot 1	% plot 2	% plot 3
TORA2	80	40	40
ARDR3			1
ASTER	2	3	5
LACA3		2	3
MECA3	5	40	10
POLYG2			25
SMILA2	5		1
VITIS	3		1

Permanent Plot Data Sheet v 03 February 2011

Plot #: P11LD006		Date: 14 July 2011		Crew: BJV/JRS/MWL		Pool 11	Compartment: p11c02s183			RM 583.9
Target UTM N	Target UTM E	Record UTM N	Record UTM E	Horizontal Datum	Vertical Datum	LD Gauge	Reading/Elevation		Est WSE	WS - Rod
4714092	693281	4714092	693281	NAD27 UTM15N	MSL 1912	10 tail		606.74	603.05	8.68
Comments:	Placed 1/2" diameter rebar that's 2' in length @ center & sub plots					11 pool	14.74	602.94	originally P11L87	
120° subplot/microplot offset to 110° due to CEOC (tree#4)					water surface elevation captured @ 1520 on 29 Aug 11					

Post	Rod Reading	Post Height	Elevation- top	Elevation- ground	Densiometer	Photo #	BA	Geomorphology
Center	5.53	3.78	609.98	606.20	92		130	low, wide natural levee
0°	5.95			605.78	98	1870	x	
120°	5.82			605.91	99	1871	x	
240°	5.49			606.24	97	1873	x	

Overstory >or= 5 in DBH, 37.2 ft plot radius

Tree #	Azimuth(0-360°)	Radial Dist(ft.in)	Species	DBH	Tree Hgt.	Mer. Hgt.	Crown Class	Wit.-Elev.	Health	Age	Comment
1	45	30.06	QUMA2	17.0	45	0	INT		Dead		
2	61	5.08	TIAM	12.1	55	0	INT		VIG		
3	84	33.00	ACSA2	13.8	60	0	INT		Stress		top broken
4	120	13.05	CEOC	15.0	75	1.5	DOM	606.11	VIG	1966	120 witness
5	142	30.06	TIAM	5.1	14	0	SUP		VIG		
6	143	28.07	TIAM	9.8	48	0	INT		VIG		mult. Stem
7	144	23.02	TIAM	12.8	70	2	DOM		VIG		
8	158	30.08	TIAM	13.8	70	0	CD		VIG		mult. Sweeps
9	170	31.08	TIAM	5.4	30	0	SUP		VIG		
10	187	34.01	QUMA2	30.8	78	1.5	DOM		VIG	1874	
11	210	33.03	GLTR	20.8	63	0.5	CD	606.14	VIG		240 witness(lean)
12	231	37.05	BENI	17.8	50	0.5	CD	606.15	VIG		240 witness
13	261	36.05	ULAM	8.0	30	0	SUP		VIG		
14	280	33.03	TIAM	11.4	47	0	INT		VIG		under QUMA2 crown
15	287	26.06	TIAM	9.6	47	0	INT		VIG		under QUMA2 crown
16	312	15.04	QUMA2	30.5	80	1.5	DOM		VIG		
17	312	27.02	QUMA2	18.3	60	0.5	INT		VIG		flattened crown
18	331	15.06	TIAM	10.6	65	0.5	INT	605.88	VIG	1961	0 witness
19	347	23.04	ULAM	12.7	60	0	SUP		Dead		
20											
21											
22											
23											
24											
25											
26											
27											
28											
29											
30	**Red text: trees outside of plot radius (37.2')**										
31											
32											
33											
34											
35											

Attachment H

Summary of Pool 11 Islands HREP Fisheries Response Monitoring

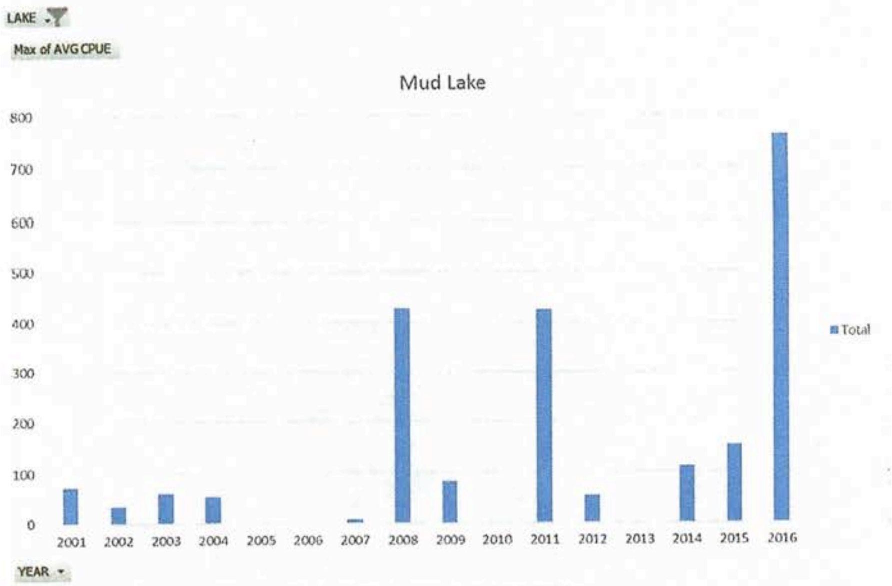
Brief summary of Pool 11 Islands HREP
Sunfish Lake, Mud Lake and Zollicoffer Lake Fisheries Response Monitoring

Late fall electrofishing has been used to document the fisheries response to the UMRR Habitat Rehabilitation and Enhancement Projects in Lower Pool 11. Sunfish Lake began functioning as an overwintering site in the fall of 2004 and Mud Lake/Zollicoffer began functioning as an overwintering site in the fall of 2006. Radio telemetry data indicates bluegills and largemouth bass home back to overwintering locations, therefore, the target for assessment was to document the use of the area by age 1 + target species which would have spent at least one winter in the restored habitat. Therefore, the first year of post project data is 2005 for Sunfish and 2007 for Mud Lake/Zollicoffer.

Both projects showed an increase in age 1 plus bluegills and species diversity over time post project. CPUE increases slowly post project as a population begins to build., which may take 7 to 10 years. Once the population becomes established, the CPUE will fluctuate year to year based on factors that can influence catchability by the electrofishing crews (i.e. vegetation and turbid water conditions make it more difficult to see and net the fish).

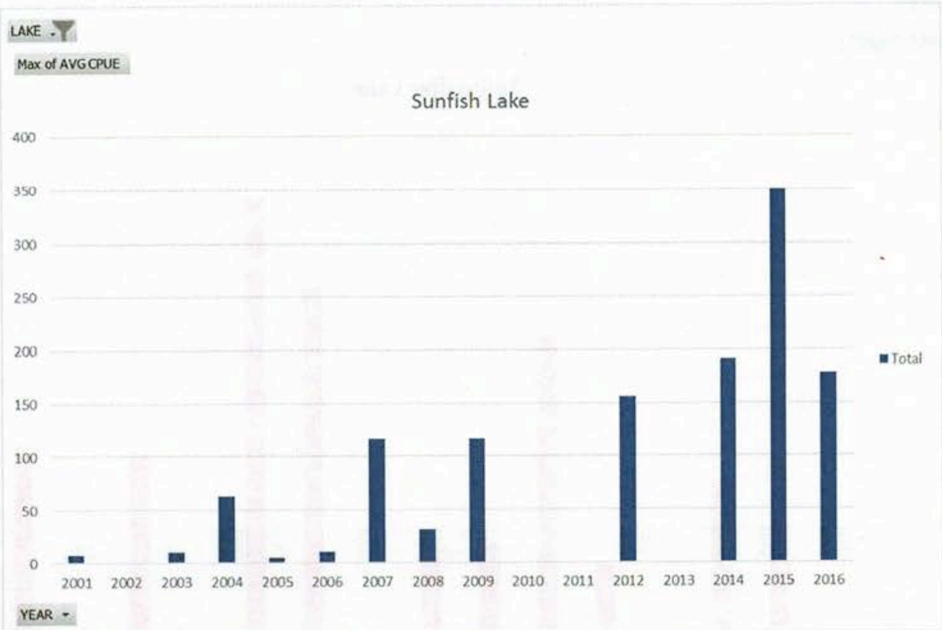
The inlet to Mud Lake was modified in 2015 to reduce flow. An increase in age 1+ bluegills has been observed, however this represents only a single year of monitoring since the modification. Additional monitoring will be conducted over the next several years.

Mud Lake Average CPUE (#/hr) of age 1 plus bluegill (≥ 3.0 inches) and Species List.



Lake	Fish Common Name	2001	2002	2003	2004	2005	2006	2007	2008	2009	2011	2012	2014	2015	2016
MUD LAKE		8	7	5	4	1	3	10	23	17	14	11	16	12	21
	bigmouth buffalo									X					X
	black crappie				X				X	X	X		X		X
	bluegill	X	X	X	X			X	X	X	X	X	X	X	X
	bluntnose minnow	X													
	bowfin														X
	brook silverside	X	X	X				X	X	X	X	X	X	X	X
	channel catfish								X						
	channel shiner												X		
	common carp								X				X	X	
	darters (Percina spp.)														X
	emerald shiner	X	X					X	X	X	X				X
	fathead minnow									X					
	gizzard shad		X				X	X	X	X		X	X	X	X
	golden redhorse						X								
	golden shiner			X				X	X	X	X	X		X	X
	grass pickerel												X		
	green sunfish							X	X	X					X
	largemouth bass	X	X	X	X			X	X	X	X	X	X	X	X
	longnose gar														X
	no fish captured at station					X		X		X					
	northern pike								X	X	X		X	X	X
	orangespotted sunfish							X	X	X	X	X	X	X	X
	pumpkinseed								X	X	X	X	X	X	X
	river carpsucker								X						
	rock bass		X						X						X
	sauger								X				X		
	smallmouth bass								X			X	X		X
	spotfin shiner														X
	spottail shiner										X				X
	spotted sucker	X	X	X			X		X	X	X	X	X	X	X
	tadpole madtom								X						
	white bass								X						
	white crappie										X		X	X	
	white sucker	X						X	X	X		X	X		
	yellow bullhead								X		X				
	yellow perch	X			X				X	X	X	X		X	X

Sunfish Lake Average CPUE (#/hr) of age 1 plus bluegill (≥ 3.0 inches) and Specie List



Lake	Fish Common Name	2001	2002	2003	2004	2005	2006	2007	2008	2009	2011	2012	2014	2015	2016
SUNFISH LAKE		12	4	4	7	6	8	13	21	12		9	13	19	13
	bigmouth buffalo													X	
	black buffalo													X	
	black bullhead								X						
	black crappie							X				X		X	
	bluegill	X		X	X	X	X	X	X	X		X	X	X	X
	bluntnose minnow	X													
	bowfin												X		
	brook silverside	X	X				X		X			X	X		X
	buffalos			X											
	channel catfish								X	X				X	X
	common carp	X			X		X		X	X			X	X	X
	emerald shiner	X						X							X
	freshwater drum													X	X
	gizzard shad	X				X	X	X	X	X			X	X	
	golden shiner							X						X	
	green sunfish		X						X						X
	highfin carpsucker								X						
	largemouth bass	X			X		X	X	X	X		X	X	X	X
	no fish captured at station			X					X						
	northern pike					X		X	X				X	X	
	orangespotted sunfish	X							X	X		X		X	
	pumpkinseed		X		X	X	X	X	X	X		X	X	X	X
	quillback	X													X
	river carpsucker				X									X	X
	river shiner								X						
	rock bass								X				X	X	
	shiners m20-29 m31-33 m35-40								X						
	shorthead redhorse								X	X			X	X	
	smallmouth bass								X						
	spottail shiner	X						X		X				X	
	spotted sucker	X	X		X			X	X	X		X	X	X	X
	usually net set						X								
	white crappie					X		X	X	X		X			
	yellow bullhead							X					X		
	yellow perch	X		X	X	X	X	X	X	X		X	X	X	X

Zollicoffer Lake Average CPUE (#/hr) of age 1 plus bluegill (≥ 3.0 inches) and Specie List

Lake	Fish Common Name	2001	2002	2003	2004	2005	2006	2007	2008	2009	2011	2012	2014	2015	2016
ZOLLICOFER LAKE		8	1	13	4	1	11	11	11	10	8	8	2		6
	black crappie			X			X	X		X	X	X			
	bluegill	X		X	X		X	X	X	X	X	X	X		X
	bowfin														X
	brook silverside	X		X			X		X	X	X	X			
	buffalos			X											
	channel catfish								X						
	common carp								X						
	emerald shiner							X							X
	gizzard shad	X						X	X	X					
	golden shiner						X			X					
	largemouth bass	X		X	X		X	X	X	X	X	X	X		X
	longnose gar						X								
	no fish captured at station		X			X									
	northern pike						X		X	X					
	orangespotted sunfish	X		X				X				X			
	pumpkinseed			X			X			X	X	X			
	river shiner										X				
	rock bass				X		X	X							
	shorthead redhorse			X											
	spotfin shiner	X													
	spotted sucker	X		X	X		X		X	2		X			X
	warmouth							X							
	weed shiner								X						
	white crappie			X			X	2	X		X	X			
	white sucker	X		X				1							
	yellow bullhead			X					X						
	yellow perch			X				2		X	X				X

Attachment I

Identifying and quantifying environmental thresholds for ecological shifts in a large semi-regulated river:

Pool 8 Study Area

Taylor & Francis
Taylor & Francis Group

Journal of Freshwater Ecology

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Identifying and quantifying environmental thresholds for ecological shifts in a large semi-regulated river

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ABSTRACT

Ecological shifts, between a clear macrophyte-dominated state and a turbid state dominated by phytoplankton and high inorganic suspended solids, have been well described in shallow lake ecosystems. While few documented examples exist in rivers, models predict regime shifts, especially in regulated rivers with high water retention time. Here I quantified ecological shifts in a large, semi-regulated floodplain river during a transition from a turbid- to a clear-water state using water quality, aquatic vegetation and fisheries data from a rigorous, standardized long-term data set. My findings indicate that significant changes occurred in total suspended solids concentration, aquatic macrophyte abundance, native and non-native fish biomass, fish functional feeding guild patterns, fish habitat guild assemblages and fish spawning guild assemblage patterns over a nearly 20-year period in Navigation Pool 8 of the Upper Mississippi River. Transitions in physical and biological indicators were examined to identify mechanisms underlying the ecological shifts. Environmental variables driving fish assemblage changes were identified (total suspended solids and aquatic vegetation) and management-relevant thresholds are presented. Awareness of management thresholds is critical for resource managers to implement measures to prevent the river from moving to a degraded state characterized by high non-native fish abundance and low predatory fish species abundance.

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Introduction

The Upper Mississippi River (UMR) near La Crosse, Wisconsin, USA (Figure 1), experienced increased turbidity and a collapse of submersed aquatic vegetation (SAV) in the late 1980s, resulting in a shift from mostly SAV-based primary production to phytoplankton-based primary production (Rogers 1994; Owens & Crumpton 1995). The collapse of SAV resulted in a dramatic decline in the recreational fishery (Rogers et al. 1995). In the early 2000s, SAV coverage expanded, and the recreational fishery recovered. Ecological shifts, between a clear water macrophyte-dominated state and a turbid, phytoplankton-dominated state, have been well described in shallow lake ecosystems (e.g. Scheffer 2004). The potential for shifts between macrophyte dominance and algal dominance in river environments with relatively long water residence time (WRT) is supported by both conceptual and spatially explicit mathematical models (Hilton et al. 2006; Hilt et al. 2011). There are, however, few published examples of this type of shift in free-flowing rivers (see Dent et al. 2002).

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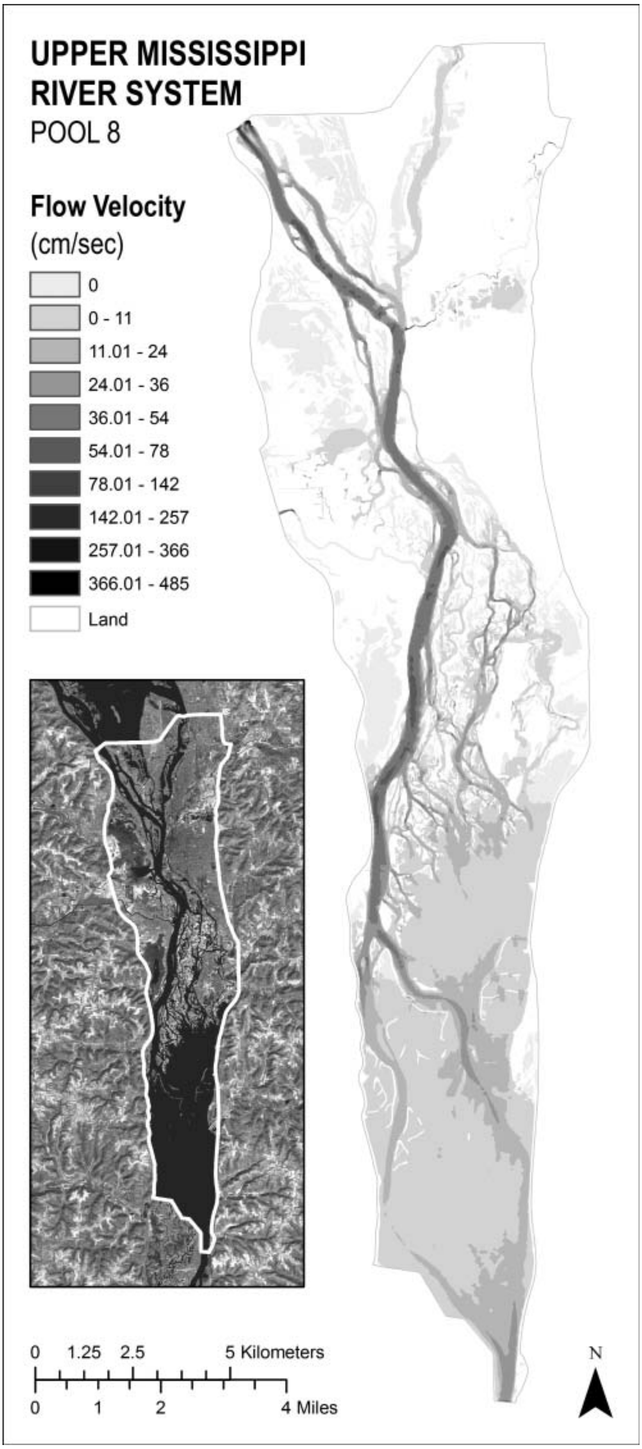

Figure 1. Navigation Pool 8 of the Upper Mississippi River extending from 1093.1 to 1130.6 km. The extent of water coverage and velocities are shown at a mean discharge of 1133 $m^3\ s^{-1}$, and the inset is a shaded representation at the same discharge. The main navigation channel is maintained at a depth of at least 2.75 m through dredging and natural erosion. Backwaters, semi-connected lakes and the impounded areas are shallower, with average depths of <1.5 m.

The positive relationship between aquatic macrophytes and water clarity is well understood (Scheffer 1990) and the prevalence of aquatic macrophytes drives a variety of ecological processes in many aquatic ecosystems (Meerhoff et al. 2003). Proliferation of aquatic macrophytes influences a variety of feedback mechanisms in large rivers including reduced sediment resuspension (James et al. 2004), reduced phytoplankton biomass via competition for nutrients and sinking (James & Barko 1994), increases in invertebrate biomass (Engel 1988), increased refuge for zooplankton (Schriver et al. 1995), increased denitrification (Weisner et al. 1994), production of allelopathic substances (Jasser 1995) and increases in waterfowl abundance (Hargeby et al. 1994; Rybicki & Landwehr 2007).

The abundance of SAV is also one of the major factors driving the fish community characteristics across the UMR (Barko et al. 2005; Chick et al. 2005; Ickes et al. 2005). Widespread landscape disturbance, resulting in increased sediment loads, has been identified as driving declines in SAV abundance resulting in declines in backwater specialists and predators with phytophilic spawning strategies (Parks et al. 2014). Relatively clear, vegetated systems tend to be dominated by visual predators such as yellow perch (*Perca flavescens*), northern pike (*Esox lucious*) and largemouth bass (*Micropterus salmoides)* (Kipling 1983; Killgore et al. 1989). Piscivorous fish such as northern pike, bowfin (*Amia calva*), largemouth bass and longnose gar (*Lepisosteus osseus*) are often able to substantially reduce recruitment among planktivorous fish (Scarnecchia 1992; Sondergaard et al. 1997). A reduction in planktivorous fish can alter food webs and results in further increases in aquatic vegetation and water clarity (Persson et al. 1988). Alternatively, benthivorous fish such as common carp (*Cyprinus carpio*) tend to be abundant in turbid systems and can maintain a turbid state due to resuspension during their feeding activities (Miller & Crowl 2006). Once substantial populations of common carp and other benthivores are high, establishing SAV can become difficult due to poor water transparency (Havens 1991).

The UMR navigational pool examined here includes multiple habitat-type characteristics of this ecologically complex river: the main channel, extensive, natural floodplain backwaters extending kilometers laterally from the main channel, semi-connected shallow lakes and a shallow impoundment in the lower third of the pool (Figure 1). Thus, it is a relatively natural, connected floodplain ecosystem influenced by a combination of riverine and shallow lake processes, and may provide an unusual example of ecosystem shifts in a large semi-regulated river.

A shift from a turbid phytoplankton-dominated system to a clear macrophyte-dominated system was captured by long-term physical and biological monitoring by Long Term Resource Monitoring (LTRM) on the UMR. Comprehensive, quality-controlled, replicated data on water quality, fish and aquatic plant communities have been collected annually since 1993 (Moore et al. 2010). This long-term data set provides an opportunity to closely examine the mechanisms underlying large ecological shifts (Holling 1973; Scheffer & Carpenter 2003), including trophic interactions at large spatial and temporal scales.

My objective was to quantify and describe changes in water quality, vegetation and fish assemblage over an 18-year period spanning a transition from turbid to clear water in a 39-km reach of the UMR (Navigation Pool 8). Specifically, I (1) examined the environmental factors associated with the observed ecological changes; and (2) identified management-relevant environmental thresholds for shifts in biological and limnological responses.

Methods

Study area

The UMR consists of a series of navigation pools extending from Minneapolis, Minnesota, USA, to the confluence of the Ohio River at Cairo, Illinois, USA. The 27 navigation dams within this area are low-head dams built to maintain sufficient depth in the river for navigation during the low-flow season and were designed to have little impact on the discharge or water level during high-flow and flood conditions (Sparks 1995; Anfinson 2005). Navigation pools are unlike reservoirs in that they remain mostly riverine in nature.

The study was conducted in Navigation Pool 8 of the UMR (Figure 1). Pool 8 is located between Lock and Dam 7 (Dresbach, Minnesota, USA) and Lock and Dam 8 (Genoa, Wisconsin, USA). It is 39 km long and encompasses ~9000 aquatic ha. Pool 8, typical of many of the navigation pools of the UMR, is composed of a diverse array of aquatic areas (Wilcox 1993), and has been spatially stratified for sampling purposes into the main channel, side channel, contiguous backwaters, isolated backwaters and impounded areas (Soballe & Fischer 2004; Ickes et al. 2014). The main channel is >3 m deep and is characterized by relatively high water velocity (0.20–0.60 ms^{-1}). Side channels are lotic but exhibit depth and water velocity that are generally less than the main channel. Contiguous backwaters typically exhibit very low water velocity (often below detection) and are connected to the main or side channel habitat at normal river stage. Isolated backwaters typically exhibit undetectable water velocity and lack connection to the channel habitat at average river stage. The impounded area is a large expanse of open water located directly upstream of the lock and dam. The average WRT in Pool 8 is 1.7 days (Wasley 2000), but this number is heavily influenced by the very large volume of water moving quickly through the main channel – WRTs in contiguous backwaters, isolated backwaters and impounded areas may range from days to months.

The UMR is modified for navigation and is somewhat unique among rivers worldwide in that the contiguous backwaters remain connected to flowing channels even during low-flow conditions. More detailed descriptions of these contrasting aquatic areas can be found in Strauss et al. (2004).

Study design

Annual pool-wide weighted mean data from a spatially stratified random sampling design were used to generate water quality (Soballe & Fischer 2004) and fisheries trends (Ickes et al. 2014; Ratcliff et al. 2014) by season and/or year for analysis. Aquatic vegetation was also measured in representative strata, and was quantified using a percent frequency index (essentially a detection rate), measured and calculated over the entire navigation pool (Yin et al. 2000). Collection of the fish and water quality data presented here began in 1993 and continued through 2011, except for 2003, when no data were collected due to budgetary constraints. I used water quality data from three seasonal sampling episodes from each year: spring, summer and autumn. In each episode, water quality data were collected at 150 randomly selected sites, weighted for stratum. Spring episodes began the last week of April, summer episodes began the last week of July and fall episodes began the second week of October. Each seasonal sampling episode was generally completed in 10–14 days. Annual fish community data were indexed using standardized day electrofishing methods from 15 June to 31 October (Ickes et al. 2014; Ratcliff et al. 2014). Aquatic vegetation data were collected annually (between 15 June and 15 August). All sampling sites were selected randomly prior to each sampling episode according to published procedures under a stratified random sampling design (Yin et al. 2000; Soballe & Fischer 2004; Ratcliff et al. 2014).

Periods (1993–2001 and 2002–2011) were delimited to provide equal sample size between the earlier period, characterized by higher total suspended solids (TSS) and less aquatic vegetation and the later period, characterized by lower TSS and increased aquatic vegetation. Water quality, vegetation and fish community metrics were compared between the two periods. Associations and potential explanatory mechanisms linking fish community responses to environmental drivers were identified using the BIOENV procedure (Primer v. 6.0).

Sampling and data collection

My data have been derived from a long-term monitoring program on the UMR, which has been observing water quality, aquatic plant and fish communities since 1993. As part of the federally mandated Upper Mississippi River Restoration (UMRR) program, the LTRM element conducts annual assessments using a spatially stratified randomized sampling design and highly standardized sampling protocols to control sampling and non-sampling error sources (Gutreuter et al. 1995; Soballe & Fischer 2004, Ickes et al. 2014; Ratcliff et al. 2014). The statistical design of the monitoring effort, and the standardized nature of the observations it collects, produce annual design-based index estimators of the measured attributes with well-understood statistical properties (Ickes et al. 2014). Relevant sampling details and descriptions of attributes used in my paper are provided below, for each data source.

Water quality and discharge

Water quality data were gained from online data repositories housed at the United States Geological Survey (USGS) Upper Midwest Environmental Sciences Center (http://www.umesc.usgs.gov/data_library/water_quality/water_quality_page.html, accessed 11 November 2016). Water samples were taken at a depth of 0.20 m at each site to assess the water column TSS, total nitrogen (TN), total phosphorus (TP) and chlorophyll *a* (CHL) concentrations. TSS was determined gravimetrically following standard methods (APHA 1992). Samples for TN and TP analyses were collected from randomly selected subsets consisting of 33% of the sampling sites. TN and TP samples were preserved in the field with concentrated H_2SO_4, transported on ice and refrigerated until analysis. TN and TP concentrations were determined colorimetrically using standard methods (APHA 1992). CHL concentrations were determined fluorometrically. Further details regarding LTRM field methods can be found in Soballe and Fischer (2004). Discharge data were collected by the U.S. Geological Survey at Winona, Minnesota, USA.

Seasonal pool-wide means (spring, summer and fall) were generated annually for TSS, TP, TN and CHL for the period of record (1993–2011) for analysis. Pool-wide means are adjusted for non-proportional sampling and standard errors for both non-proportional sampling and stratification. These statistics are calculated according to established procedures, and are published on the LTRM online database. Mean annual discharge at Winona, Minnesota, USA, was used in the analysis.

Aquatic vegetation

Aquatic vegetation community data were gained from online data repositories housed at the USGS Upper Midwest Environmental Sciences Center (http://www.umesc.usgs.gov/data_library/vegetation/vegetation_page.html, accessed 11 November 2016). Standardized sampling procedures are described in Rogers and Owens (1995) and Yin et al. (2000). Aquatic vegetation community and relative abundance data are collected annually between 15 June and 15 August, the period of maximum standing stocks. Each year, 450 randomly selected sampling sites (weighted by stratum) are visited and vegetation is identified and quantified in six subsampling units, each $\sim$1.5 m$\times$ 0.36 m. Recorded field data include species detect/non-detect and a relative abundance score that reflects either the biomass (SAV) or the percent cover over the water surface (rooted floating leaf and emergent). I used percent frequency occurrence (Yin et al. 2000) for analysis. Percent frequency occurrence is a measure of how often a species or life form is encountered. It is calculated by dividing the number of sites where a species or life form occurs by the total number of sites sampled and multiplying by 100. I used annual pool-wide design-based percent frequency estimators (Yin et al. 2000) for the submersed (SAVPf; *N* species = 18), rooted-floating leaf (RFPf; *N* species = 3) and emergent (EMPf; *N* species = 27) vegetation. This provided annual time series (1993–2001) of abundance indices for plant assemblages. Submersed, rooted floating-leaf and emergent vegetation class estimates derived from percent frequency estimators were then summed to generate a total aquatic plant index, referred to hereafter as VegSum (Yin et al. 2000). It was possible for all three life forms to overlap; therefore, VegSum can exceed 100%.

Fish

Fish community data were gained from online data repositories housed at the USGS Upper Midwest Environmental Sciences Center (http://www.umesc.usgs.gov/data_library/fisheries/fish_page.html, accessed 11 November 2016). I selected fishery-independent day electrofishing collections from a larger database, 1993–2011 (15 June–15 October each year; the average number of samples per year = 76). I retained data for all the observed species (N = 87, 1993–2011). Species catch and length data were relationally linked to a second database housing species-specific life-history traits and empirically derived allometric growth models (O'Hara et al. 2007). Using these two linked databases, I then generated estimates of mass per sample per species by applying species-specific growth models to length and catch data per sample. Species were then combined, per sample, into the following guilds as expressed in O'Hara et al. (2007): (1) native/non-native status; (2) exploitation status; (3) feeding guild; (4) habitat preference; (5) reproductive guild; and (6) trophic position (Table A1). Mass was summed by sample and guild for each year and an estimate of mean mass-per-unit-effort (g/15-minute electrofishing run) was calculated as per the statistical estimators expressed in Ickes et al. (2014) and Ratcliff et al. (2014). This resulted in annual time series of design-based functional mass expressions for each fish guild class that represent the aquatic environment of Navigation Pool 8, 1993–2011.

Analytics

Testing for changes in observed attributes

Water quality, aquatic plant and fish guild time series used in this study were parsed into two equal periods (1993–2001 and 2002–2011; both N = 9 due to no data collected in 2003) for analyses. Mann–Whitney Rank Sum Tests (SAS Institute 2008, SAS v. 9.2) were used to infer differences in water quality, fish guild and aquatic plant indices (Table 1) between the two periods. Differences in the observed medians between periods were calculated for each environmental variable and guild class and plotted (Figures 2 and 3) to both qualify and quantify the nature of significant shifts among all study variables (expressed as percent change in median).

Testing for fish guild shifts in relation to changes in environmental conditions

For each fish guild (N = 5; trophic position excluded), guild classes were treated as multivariate observations and the Bray–Curtis similarity metric was used to ascribe similarity scores among years in the guild structure. Non-metric multidimensional scaling (NMDS; Primer v. 6.0; Clarke 1993) was applied to the similarity matrices and patterns in guild structure were visualized in both two-dimensional and three-dimensional solutions and plots. I tested for shifts in guild structure between periods using an Analysis of Similarity (ANOSIM; Primer v. 6.0), with period (as described above) as the grouping factor in the analysis (Figure 4).

To identify and test which environmental attributes (discharge, water quality and aquatic plant variables) were most strongly associated with shifts in fish guild responses between the two periods, I used the BIOENV procedure (Table 2; Primer v. 6.0). To complement the similarity matrices described for the fish guild data, I generated similarity scores (Euclidean distance) among years based upon the environmental attributes data. For each fish guild, Primer's BIOENV routine was used to generate a canonical solution (maximum rank correlation) between the biological response similarity matrix and the environmental variable similarity matrix. Correlations were calculated using Spearman's rank correlation coefficient. To impose parsimony upon the maximal correlation determination, I constrained the number of environmental variables to a maximum of three variables for each fish guild analysis. Primer's BIOENV procedure is an unconstrained method and generates rank correlation solutions for all permutations of environmental variables (order and number

Table 1. Mann–Whitney rank sum test results indicating the *U*-statistic, *t*-value and *p*-value for all study parameters between environmental periods observed in Pool 8 of the Upper Mississippi River (1993–2011). The 25th percentile, median and 75th percentile for each parameter by environmental period are also presented.

	1993–2001			2002–2011					
Variable	25th	Median	75th	25th	Median	75th	U	t	p
Vegetation									
SAV[a] (% Freq.)	36.30	46.4	48.51	64.76	71.39	79.03	1	46	<0.001
RF[b] (% Freq.)	12.16	17.50	18.50	24.98	31.08	37.68	0	45	<0.001
EM[c] (% Freq.)	7.20	9.87	11.47	17.55	19.96	25.40	0	45	<0.001
VEGSUM[d] (% Freq.)	55.65	75.38	78.32	109.13	123	140.04	1	46	<0.001
Discharge									
Mean annual at Winona (m^3 s)	33,855	38,600	46,690	24,145	31,360	37,575	21	105	0.093
Water quality									
TSS[e] Spring (mg L)	20.40	25.12	27.46	12.93	14.86	21.83	12	114	0.013
TSS[e] Summer (mg L)	22.48	23.81	27.56	7.19	10.09	18.18	3	123	0.001
TSS[e] Fall (mg L)	16.83	19.80	24.10	7.44	10.10	18.47	16	110	0.034
CHL[f] Spring (μg L)	24.15	37.10	53.16	16.41	32.27	45.77	33	93	0.536
CHL[f] Summer (μg L)	14.99	25.04	55.15	12.89	21.51	34.21	36	90	0.724
CHL[f] Fall (μg L)	15.46	22.73	42.14	4.67	6.82	15.99	12	114	0.013
TN[g] Spring (mg L)	1.75	2.85	3.66	1.74	2.65	3.56	36	90	0.724
TN[g] Summer (mg L)	1.77	2.49	2.60	1.41	1.67	2.21	22	104	0.112
TN[g] Fall (mg L)	1.30	1.46	1.95	1.37	1.60	2.77	33	78	0.536
TP[h] Spring (mg L)	0.10	0.11	0.12	0.09	0.10	0.12	29	97	0.331
TP[h] Summer (mg L)	0.15	0.17	0.19	0.16	0.18	0.23	26	71	0.216
TP[h] Fall (mg L)	0.13	0.15	0.17	0.12	0.15	0.17	40	85	1.000
Fish MPUE									
Native	6070.96	7445.65	8524.23	8195.85	9814.95	13,144.96	8	53	0.005
Non-native	9472.25	12,642.28	16,216.79	5260.18	6304.35	7160.53	6	120	0.003
Exploitation status									
Recreational	1389.55	2581.95	2861.10	3368.10	4767.30	6125.07	0	45	< 0.001
Commercial	14,224.93	16,710.59	20,142.43	9899.41	11,299.64	13,997.18	9	117	0.006
Non-game	257.91	368.67	487.66	97.45	246.24	1025.66	31	95	0.427
Adult feeding guild									
Carnivore	637	731.22	801.24	973.14	1122.42	1885.73	3	48	0.001
Invertivore–carnivore	1829.55	2303.81	2559.72	3104.49	4012.74	5002.12	3	48	0.001
Invertivore–detritivore	9528.29	12,705.65	16,320.54	5339.21	6416.56	7250.48	6	120	0.003
Invertivore–planktivore	0.66	1.06	1.34	1.93	3.80	8.78	7	52	0.004
Invertivore–herbivore	35.96	107.76	129.42	17.37	22.31	45.39	14	112	0.022
Planktivore–invertivore	0.24	0.42	0.74	0.69	0.97	2.00	14	59	0.022
Detritivore	0.26	0.42	11.58	3.29	5.31	35.79	21	66	0.093
Invertivore	2677.72	3664.80	4375.28	3793.94	4266.26	5441.15	21	66	0.093
Planktivore–detritivore	25.44	35.09	137.22	0	36.77	62.10	30.5	95.5	0.399
Detritivore–invertivore	0	0.01	0.05	0	0.01	0.03	34	92	0.579
Herbivore	77.26	284.71	399.06	25.09	191.51	956.02	36	90	0.724
Planktivore	6.94	12.86	18.89	5.41	8.83	21.33	36	90	0.724
Habitat guild									
Limnophillic	1089.69	1536.16	2264.98	2805.82	4,025.70	6047.51	0	45	<0.001
Limnorheophillic	12,934.84	14,653.77	18,457.17	8140.20	9564.78	11,269.62	7	119	0.004
Pelagicrheolimnophillic	38.92	68.14	79.39	14.83	46.05	86.14	31	95	0.427
Pelagiclimnorheophillic	30.48	78.19	202.46	9.81	63.84	183.46	32	94	0.480
Rheolimnophillic	1184.14	1321.94	1521.70	1140.46	1329.54	1713.92	38	83	0.860
Rheophillic	90.83	200.92	245.51	113.03	184.78	218.10	39	87	0.930
Reproductive guild									
Polyphillic	935.85	1695	2210.53	2561.63	4047.78	5035.37	0	45	<0.001
Phytophillic	683.10	791.15	851.63	1071.34	1189.34	1923.26	4	49	0.001
Phytolithophillic	9620.32	12,771.18	16,370.70	5685.71	6593.01	7510.34	6	120	0.003
Pelagophillic	143.72	158.75	245.37	165.48	265.69	327.35	26	71	0.216
Lithophillic	2221.99	2697	3527.93	2725.97	3105.90	3860.88	29	74	0.331
Psammophillic	0	0.01	0.02	0	0.02	0.03	32	77	0.477
Lithopelagophillic	992.73	1095.14	1306.17	700.17	786.85	1681.23	33	93	0.536
Spleleophillic	329.31	364.61	539.95	293.62	431.79	507.59	39	84	0.930
Trophic status									
Fourth	2619.94	2917.59	3263.05	4179.92	5135.17	6887.85	2	47	< 0.001
Third	14,140.75	16,331.20	19,247.65	9670.18	11,691.53	13,837.96	8	118	0.005
First-CHL[f] fall	15.46	22.73	42.14	4.67	6.82	15.99	12	114	0.013
First-CHL[f] summer	14.99	25.04	55.15	12.89	21.51	34.21	36	90	0.724

(*continued*)

Table 1. (*Continued*)

Variable	1993–2001 25th	1993–2001 Median	1993–2001 75th	2002–2011 25th	2002–2011 Median	2002–2011 75th	*U*	*t*	*p*
First-CHL[f] spring	24.15	37.10	53.16	16.41	32.27	45.77	33	93	0.536
VegSum[d] (% Freq.)	55.65	75.38	78.32	109.13	123	140.04	1	46	< 0.001
Species of management interest									
Micropterus salmoides	390.43	569.19	1081.25	1793.21	2211.97	3353.93	1	46	< 0.001
Esox luscious	87.65	138.43	287.33	364.85	424.36	446.46	2	47	< 0.001
Lepomis macrochirus	136.95	393.42	457.57	430.29	782.72	1278.63	11	56	0.01
Cyprinus carpio	9472.25	12,642.28	16,216.79	5260.18	6304.35	7160.53	6	120	0.003

[a]Submersed aquatic vegetation.
[b]Rooted-floating vegetation.
[c]Emergent vegetation.
[d]Sum of submersed, rooted floating and emergent vegetation percent frequency.
[e]Total suspended solids.
[f]Chlorophyll α.
[g]Total nitrogen.
[h]Total phosphorus.

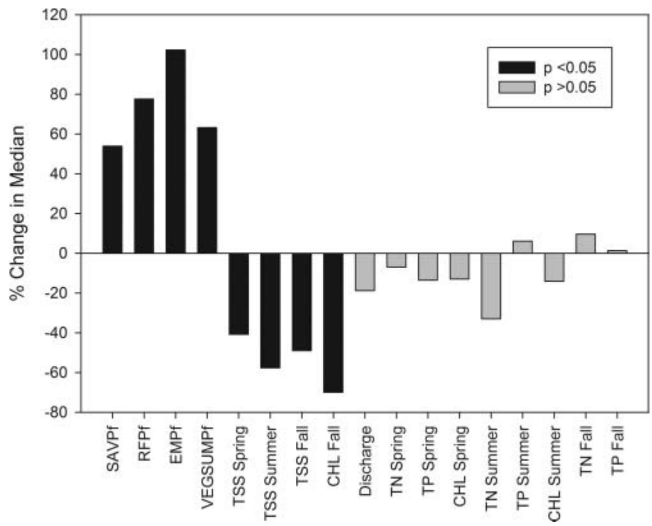

Figure 2. Change in median between the early environmental period (1993–2001) and the late environmental period (2002–2011) among the environmental variables in Pool 8 of the Upper Mississippi River. Changes significant at the $p < 0.05$ level are denoted with black bars.

of variables). Solutions were sorted by rank correlation order to identify the environmental variables most strongly associated with fish guild responses.

Identification of thresholds for environmental covariates driving fish guild responses

Once the environmental covariates associated with fish guild responses were identified, linear and piecewise regression techniques were used to determine the presence of TSS thresholds for fish guild metrics. Native/non-native and exploitation status guilds were selected for TSS threshold analysis due to their resource management importance. I selected TSS for threshold determination due to it being a more easily measured, and more management-relevant target than aquatic vegetation percent frequency (Table 3; Figure 5). Furthermore, TSS and aquatic vegetation (VegSum) tend to be tightly coupled (Figure 6; r^2 = 0.807). Linear regression was used to determine if TSS could predict fish guild metrics and generate statistics comparable to the piecewise regression method. Piecewise

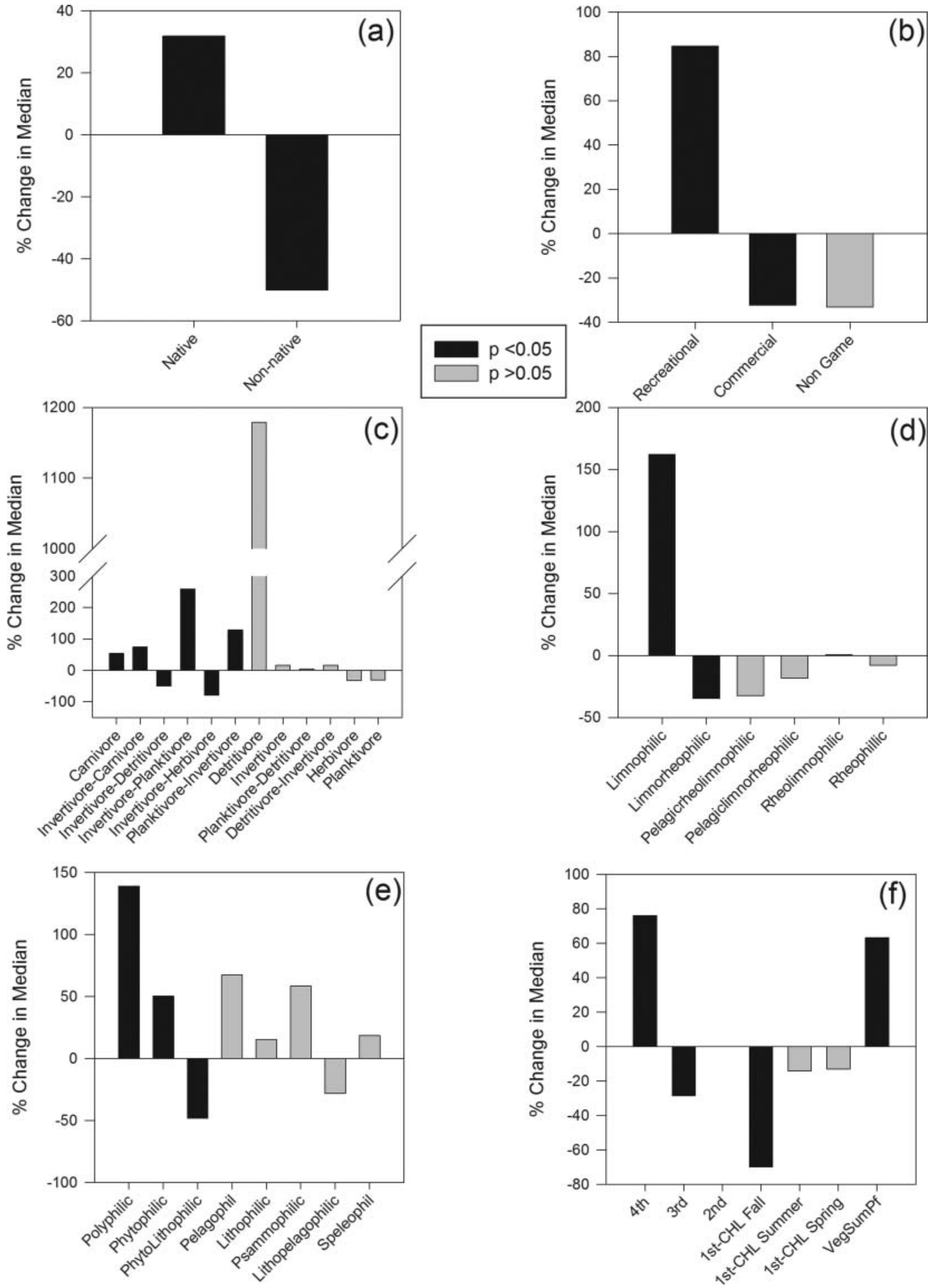

Figure 3. Change in median between the early period (1993–2001) and the late period (2002–2011) among (a) native/non-native status; (b) exploitation status; (c) feeding guild; (d) habitat guild; (e) reproductive guild; and (f) trophic position in Pool 8 of the Upper Mississippi River (1993–2011). Changes significant at the $p < 0.05$ level are denoted with black bars.

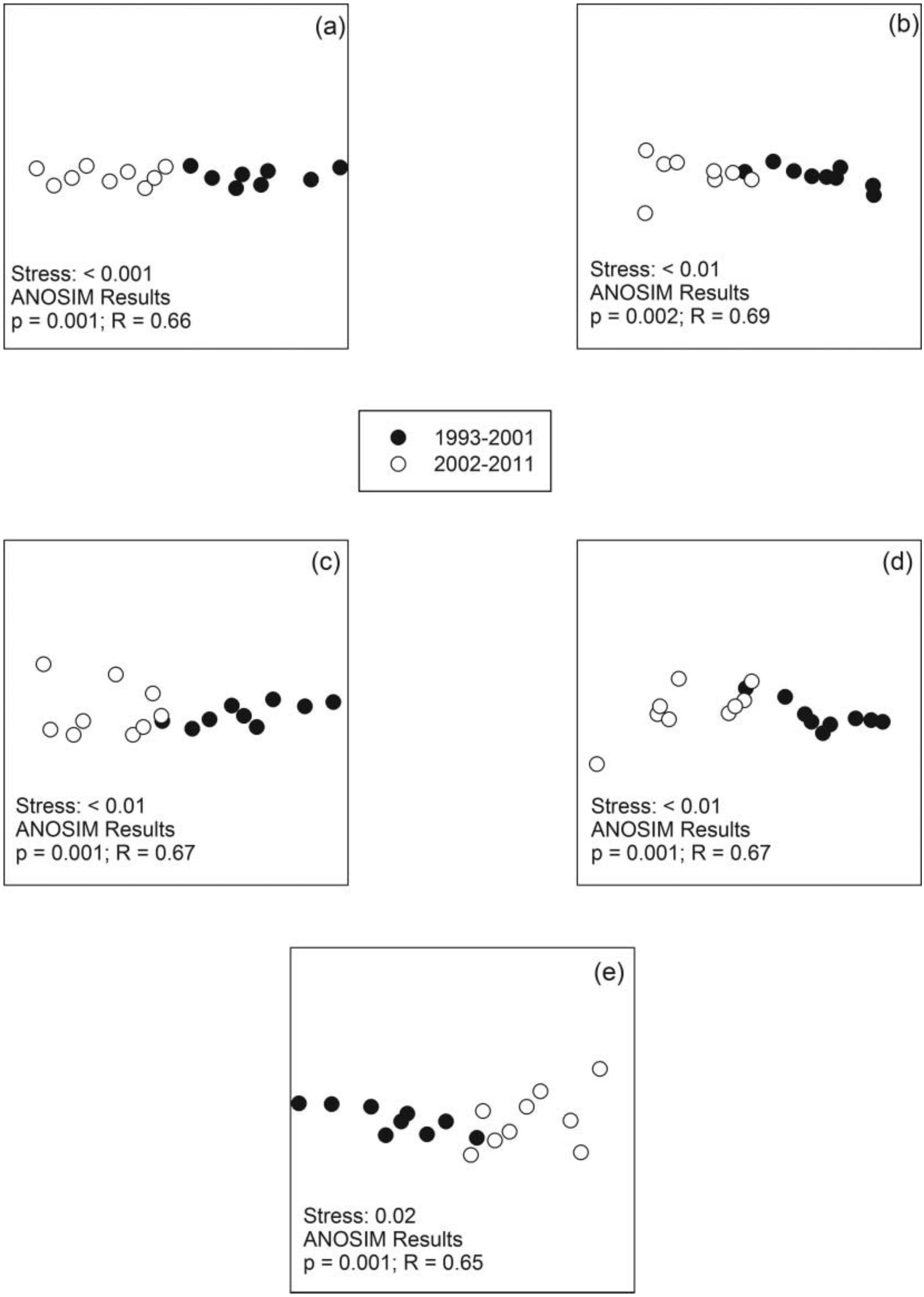

Figure 4. Two-dimensional non-metric scaling ordination (NMDS) between the early period (1993–2001) and the late period (2002-2011) among (a) native/non-native status; (b) exploitation status; (c) feeding guild; (d) habitat guild; and (e) reproductive guild in Pool 8 of the Upper Mississippi River (1993–2011). The ANOSIM results comparing the two time periods are also given.

Table 2. Primer BIOENV results indicating the top three environmental variables associated with fish guild shifts between periods in Pool 8 of the Upper Mississippi River (1993–2011). *R* indicates the maximal rank correlation for each three-variable solution.

Biological variable	First environmental variable	Second environmental variable	Third environmental variable	R
Native/non-native	VegSum[a] (% Freq.)	TSS[b] summer (mg L)	TSS[b] fall (mg L)	0.466
Exploitation status	VegSum [a] (% Freq.)	TSS[b] summer (mg L)	CHL[c] summer (μg L)	0.415
Adult feeding guild	VegSum [a] (% Freq.)	TSS[b] summer (mg L)	TSS[b] fall (mg L)	0.499
Habitat guild	VegSum [a] (% Freq.)	TSS[b] summer (mg L)	TSS[b] fall (mg L)	0.421
Reproductive guild	VegSum [a] (% Freq.)	TSS[b] summer (mg L)	TSS[b] fall (mg L)	0.358

[a]Sum of submersed, rooted floating and emergent vegetation percent frequency.
[b]Total suspended solids.
[c]Chlorophyll α.

Table 3. Thresholds for fish guild responses to mean summer TSS in Pool 8 of the Upper Mississippi River (1993–2011), and adjusted r^2 values as determined from two regression techniques. All parameter estimates are significant at the 0.05 level.

	Piecewise regression			Linear regression
Fish guild	Threshold	95% confidence interval	Adj r^2	Adj r^2
Non-native	19.26	14.235–24.275	0.6928	0.555
Native	12.55	6.424–18.666	0.4324	0.24
Commercial	19.15	12.401–25.889	0.4692	0.367
Recreational	12.29	8.155–16.414	0.5833	0.341

or 'broken-stick' regression models were used to identify thresholds or breakpoints (Toms & Lesperance 2003). Successful piecewise regression models have r^2 values >0.2 and greater than calculated r^2 values from corresponding linear regressions (Toms & Lesperance 2003; Black et al. 2011). For each identified threshold value, 95% confidence intervals were also calculated. Linear and piecewise regressions were performed in SigmaPlot 11.0 (Systat 2008).

Results

Shifts in water quality, aquatic plant and fish guild indices

Substantial shifts were observed among the environmental variables in this study. Eight of the 17 water quality, aquatic macrophyte and discharge variables demonstrated significant shifts ($p < 0.05$; Figure 2). Percent frequency of submersed, rooted-floating leaved, emergent and VegSum (all three life forms combined) increased significantly from the early-to-late environmental period (Table 1; Figure 2). Conversely, spring TSS, summer TSS, fall TSS and fall CHL decreased significantly from the early-to-late environmental period (Table 1; Figure 2). The remainder of the discharge and water quality variables exhibited no statistically significant change between the periods.

Many statistically significant differences were observed among the fish guild metrics between the two time periods. Notably, native fish biomass indicated a significant increase, while non-native fish biomass indicated a significant decrease (Table 1; Figure 3(a)). For exploitation status, recreational fish biomass increased significantly, while commercial fish biomass decreased significantly (Table 1; Figure 3(b)). Within the adult feeding guild, carnivore, invertivore–carnivore, intertivore–planktivore and planktivore–invertivore guild classes all increased significantly, while the invertivore–detritivore and invertivore–herbivore guild classes decreased significantly (Table 1; Figure 3 (c)). For the habitat preference guild, limnophils increased significantly, while limnorheophils decreased significantly (Table 1; Figure 3(d)). For the reproductive guild, polyphils and phytophils increased significantly, while phytolithophils decreased significantly (Table 1; Figure 3(e)). For the trophic position guild, the fourth trophic level increased significantly, while the third trophic level decreased significantly (Table 1; Figure 3(f)). Furthermore, ANOSIM analysis demonstrated significant differences in fish community between the two time periods for all fisheries guilds examined (Figure 4).

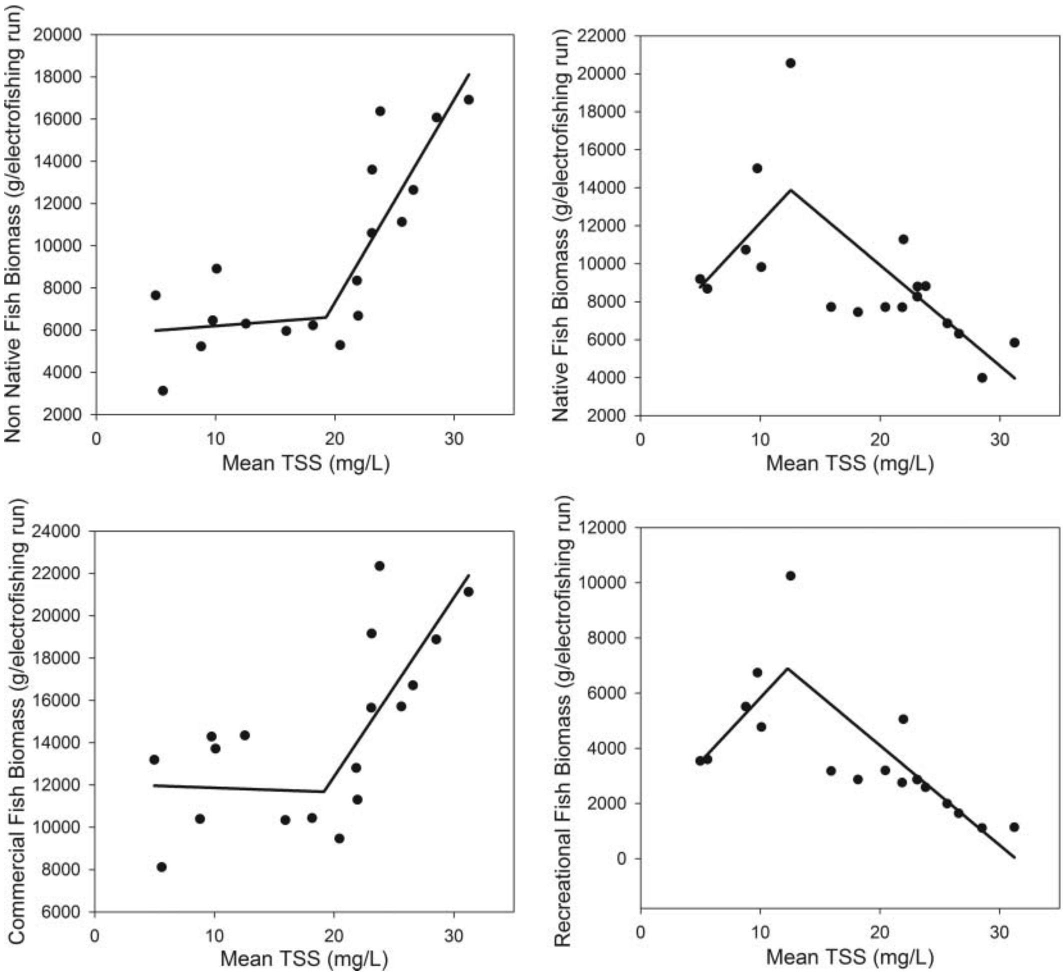

Figure 5. Relation between mean annual fish guild biomass per electrofishing run and mean summer TSS in Pool 8 of the Upper Mississippi River (1993–2011). Thresholds are indicated by the breakpoint in the piecewise regression line.

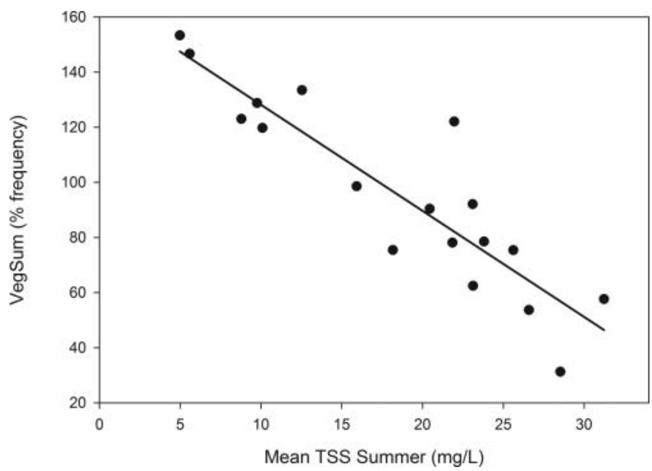

Figure 6. Relation between VegSum (percent frequency) and mean summer TSS in Pool 8 of the Upper Mississippi River (1993–2011). The line indicates the linear regression result ($y = -3.85x + 166.64$; $r^2 = 0.807$).

Environmental drivers of fish guild responses

Canonical rank correlation results from the BIOENV procedure, performed for five fish guilds considered, identified the primary environmental variables associated with fish guild responses (Table 2). For each fish guild, a three-variable solution produced the maximal rank correlation (range 0.358–0.499 among guilds; Table 2). The aquatic plant abundance index (VegSum) contributed to the canonical solution for every fish guild. Mean summer TSS also contributed to all five solutions. Mean fall TSS contributed to four of five solutions (exploitation status was the only exception). Mean summer CHL only contributed to the exploitation status guild solution. No additional environmental variables made contributions to the canonical solutions.

Thresholds for fish guild responses to environmental drivers with emphasis on native and exploitation status

Thresholds were detected in the relations between fish guild metrics and summer TSS. Fish guild response thresholds ranged from 12.29 to 19.26 mg/L summer TSS (Table 3; Figure 5). Non-native fish biomass increased and native fish biomass decreased as summer TSS increased (Table 3; Figure 5). Similarly, recreational fish biomass decreased and commercial fish biomass increased as summer TSS increased (Table 3; Figure 5).

Discussion

It is evident that portions of the UMR have undergone a shift from a turbid system with sparse vegetation during the early 1990s, to a clear water system with abundant aquatic vegetation in the recent years. There are likely multiple factors driving TSS levels within Pool 8 which makes it difficult to identify the ultimate driver of these changes, but TSS is clearly associated with changes in vegetation and fish communities in the UMR. As this shift from a turbid to vegetated condition has occurred, a number of positive and negative feedbacks have reshaped the ecosystem. The increase in vegetation has likely resulted in a decrease in wind-induced sediment resuspension due to buffering of wave action (Dent et al. 2002) and sediment stabilization. Phytoplankton production decreased, although only statistically significant in the fall, and was likely the combined result of many drivers, including allelopathic exudates from rooted vegetation inhibiting phytoplankton growth (Sondergaard & Moss 1998), higher algal sinking rates within the low-velocity environment of the plant beds that remove phytoplankton from the photic zone (Sand-Jensen 1998; Kohler et al. 2010), increased algal predation by zooplankton that use refuge within plant beds and reduce phytoplankton standing stocks (Hillbricht-Ilkowska 1999), trophic shifts resulting in suppression of planktivores by abundant top predators (Wootton & Power 1993) and nitrate becoming locally less available due to denitrification within the plant beds (Veraart et al. 2011).

The indexed mass of benthivorous, non-native, common carp decreased by approximately 50% over the transition, perhaps due to the less favorable vegetated environment that developed (Breukelaar et al. 1994). Common carp were the most abundant fish species in Pool 8, in terms of indexed mass, throughout the entire study period. Therefore, a 50% reduction in common carp likely reduced bioturbation in the system, leading to a strong positive feedback between this non-native fish and turbidity/TSS.

Indexed native fish mass showed a significant increase, while indexed non-native fish mass showed a significant decrease as TSS declined (Figure 5). Aquatic vegetation and TSS were the most explanatory variables driving native/non-native fish assemblage (Table 2). This is consistent with the results of many studies demonstrating a significant positive relationship between common carp mass (non-native to North America) and TSS concentration (Meijer et al. 1989; Meijer et al. 1990; Havens 1991; Breukelaar et al. 1994). Conversely, many studies have shown an increase in native fish biomass as TSS is reduced and vegetation coverage increases (Grift 2001; Zambrano et al. 2001;

Parks et al. 2014). Because TSS had such a pronounced effect on the dominance between native and non-native indexed fish mass, I expect that TSS reductions will be critical to native fish conservation in the upper impounded Mississippi River.

Recreational fish indexed mass increased significantly by nearly 80%, while commercial fish indexed mass decreased significantly as TSS declined and aquatic vegetation increased (Figure 5). The increase in recreational fish indexed mass was overwhelmingly tied to increases in largemouth bass, northern pike (both visual predators) and bluegill (*Lepomis macrochirus*; a visual invertivore; Table 1). Many studies have documented the link between an increase in these three species and increased vegetation (Killgore et al. 1989; Grimm & Backx 1990; Bettoli et al. 1993; Grift 2001). The reduction in commercial fish indexed mass closely mirrored the reduction in non-native fish indexed mass, and was likely driven by the observed decline in common carp, a non-native but commercially important species.

The carnivorous fish guild increased significantly, while the invertivore–detritivore fish guild decreased significantly as TSS declined and aquatic vegetation increased. The positive relationship between aquatic vegetation and visual predator species like largemouth bass and northern pike is well known, but an understanding of the ecological importance of formerly reviled fishes such as gars and bowfin has only recently come to light (Scarnecchia 1992). Having the full complement of carnivorous fishes is critical to ecosystem function, especially for controlling recruitment of ecosystem generalists of the invertivore–detritivore guild, with the most prominent of this group being the common carp (Parks et al. 2014).

Limnophilic fish showed a significant increase, while the more channel-dwelling limnorheophils decreased significantly as TSS declined and aquatic vegetation increased. This result supports recent research documenting ecological shifts in the opposite direction (from clear to turbid states) in which a decline in backwater specialists was observed in agriculturally impacted Midwestern rivers (Parks et al. 2014). TSS concentration was lower, and vegetation coverage within Pool 8 was greater, than the highly impacted rivers in Iowa, USA, studied by Parks et al. (2014). It seems likely that the expansion of vegetation beds in Pool 8 has increased areas of low water velocity within the pool, and is a possible reason for the decline in limnorheophils (Sand-Jensen 1998).

Polyphilic and phytophilic fish guilds increased significantly, while the phytolithophilic fish guild decreased significantly as TSS declined and aquatic vegetation increased. The increase in fish with phytophilic spawning strategies is encouraging and suggests that the reduction of TSS can contribute greatly to the restoration of ecological structure of North American rivers affected by agriculture. My results again corroborate those of Parks et al. (2014) who noted substantial declines in fish with phytophilic spawning strategies in Iowa, USA. Rivers as flow regimes were altered, water quality degraded and river corridors were fragmented following the onset of intensive row crop agriculture.

Significant trophic shifts in fish were observed as TSS declined and aquatic vegetation increased. Indexed mass of the fourth trophic level increased significantly; likely due to the increase in visual predators (especially northern pike and largemouth bass) experiencing increased feeding efficiency with greater water transparency (Killgore et al. 1989; Grimm & Backx 1990; Bettoli et al. 1993). Additionally, many of the top trophic-level species (northern pike, longnose gar and bowfin, specifically) are also phytophilic spawners, so they may have benefited both from increased clarity and increased vegetation abundance (Parks et al. 2014). The increase in the fourth trophic level likely resulted in the reduction of the third trophic level due to increased predation.

This study demonstrates TSS as a useful indicator for changes in ecosystem structure and function. I found it was associated with increases in aquatic vegetation (Figure 6) and important functional changes in fish community. Identification of ecological thresholds is critical to sound management of aquatic resources. Once particular thresholds are crossed, aquatic systems can move away from desired ecological conditions and it can become very difficult to shift the system back to the desired state (Groffman et al. 2006). Managers need to know where these thresholds exist due to the very high stakes associated with crossing the ecological tipping points (Sparks et al. 1990; Scheffer & Carpenter 2003). I identified thresholds ranging between 12.29 and 19.26 mg/L mean summer TSS for the UMR. The mean of the summer TSS thresholds was 16 mg/L and I suggest this value as an important management target for native fish conservation in the UMR. This value appears to be consistent with thresholds identified by other researchers in a variety of environments. Jackson et al. (2010) identified TSS in the 11–14 mg/L range as being associated with high bluegill/largemouth bass catch rates and low common carp catch rates in 129 Iowa lakes. Conversely, TSS in the 25–30 mg/L range was associated with low bluegill/largemouth bass and high common carp catch rates. Growing season TSS of 15 mg/L has been identified as a tipping point for SAV establishment, waterfowl, fish and invertebrate populations on Chesapeake Bay (Kemp et al. 2004). Lougheed et al. (1998) observed dramatic shifts in Great Lake wetlands among fish and SAV communities as turbidity values shifted from 6 NTU (equivalent to 8 mg/L TSS using relationships in Giblin et al. 2010) to 20 NTU (equivalent to 30 mg/L). When considering public perception and the value of aquatic resources, Michigan (USA) residents identified 20 mg/L TSS as the point where water was perceived to be 'clear' (http://www.michigan.gov/documents/deq/wb-npdes-TotalSuspendedSolids_247238_7.pdf, accessed 12 May 2016).

Freshwater ecosystems are constantly undergoing changes of both natural and human-induced origins, and many changes over the past century have led to ecosystems locked in degraded ecological states (Scheffer 2004). The mechanisms leading to such shifts arise from varying processes, including compromised water quality (Hilton et al. 2006), establishment of invasive and competitively superior species (Zambrano et al. 2006) and land uses and ecosystem extractions that exceed the assimilative capacity of ecosystems (Parks et al. 2014). Such ecological shifts often come with notable social and economic costs, progressing from a diverse natural system with diverse ecosystem service benefits, toward simplified ecosystems with fewer and harder-to-manage ecosystem service benefits. Such transitions are not limited to freshwater ecosystems. Examples in terrestrial ecosystems include an irreversible shift from grasslands to desert where native grazers were (even temporarily) replaced with livestock in the Sahel (Van De Koppel et al. 1997). In marine ecology, coral bleaching (Hoegh-Guldberg 1999; Fitt et al. 2001) – the loss of dinoflagellate algal symbionts from coral hosts – is a threshold response to anthropogenic disturbances, leading to fundamental change in primary production, ecosystem simplification and a loss of ecosystem services. Understanding the thresholds where ecosystems begin to shift ecological states is critical for the applied management of ecosystems. While sometimes abrupt (e.g. Hilt et al. 2011), ecosystem state shifts are most commonly slow-moving, cumulative responses to a variety of ecosystem impairments. For this reason, long-term standardized observation is a key tool for documenting these shifts, and for identifying their proximate causes, so that management can be applied before important thresholds are crossed and undesirable ecological shifts occur. Here, I have used long-term and standardized observations to identify shifts in the functional attributes of a large river fish community, and to identify the environmental factors associated with this ecological shift. I have also proposed an ecological threshold in TSS and associated changes in aquatic plant and fish community attributes where an ecosystem shift occurred for the UMR. Science-informed management is frequently required to address ecosystem shifts, and because of the size and inter-jurisdictional nature of the UMR, management will require a plurality of stakeholders to actively engage in seeking and meeting threshold targets.

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Appendix

Table A1. Native/non-native status, exploitation status, feeding guild, habitat guild, reproductive guild and trophic position by species among fishes in Navigation Pool 8 of the Upper Mississippi River (1993–2011).

Fish code	Common name	Scientific name	Native or non-native[a]	Exploitation status	Feeding guild	Habitat guild	Reproductive guild	Trophic status
ABLP	American brook lamprey	*Lampetra appendix*	N	Non-game	No feed		Lithophil	
AMEL	American eel	*Anguilla rostrata*	N	Commercial	Invertivore/ carnivore	Rheo-limnophilic		4
BDDR	Banded darter	*Etheostoma zonale*	N	Non-game	Invertivore		Phytophil	3
BHMW	Bullhead minnow	*Pimephales vigilax*	N	Non-game	Invertivore/ herbivore	Rheo-limnophilic	Speleophil	3
BKBF	Black buffalo	*Ictiobus niger*	N	Commercial	Invertivore/ herbivore	Rheo-limnophilic	Lithopelagophil	3
BKBH	Black bullhead	*Ameiurus melas*	N	Commercial	Invertivore/ carnivore	Limnophilic	Speleophil	4
BKCP	Black crappie	*Pomoxis nigromaculatus*	N	Recreational	Invertivore/ carnivore	Limnophilic	Phytophil	4
BKSB	Brook stickleback	*Culaea inconstans*	N	Non-game	Planktivore/ invertivore		Ariadnophil	3
BKSS	Brook silverside	*Labidesthes sicculus*	N	Non-game	Planktivore/ invertivore	Rheo-limnophilic	Phytolithophil	3
BLGL	Bluegill	*Lepomis macrochirus*	N	Recreational	Invertivore	Limnophilic	Polyphil	3
BMBF	Bigmouth buffalo	*Ictiobus cyprinellus*	N	Commercial	Invertivore	Pelagic Limno-rheophilic	Lithopelagophil	3
BNMW	Bluntnose minnow	*Pimephales notatus*	N	Non-game	Detritivore		Speleophil	3
BNBH	Brown bullhead	*Ameiurus nebulosus*	N	Commercial	Invertivore/ carnivore		Speleophil	4
BNTT	Brown trout	*Salmo trutta*	NN	Recreational	Invertivore/ carnivore		Lithophil	4
BRBT	Burbot	*Lota lota*	N	Recreational	Invertivore/ carnivore		Lithopelagophil	4
BSDR	Blackside darter	*Percina maculata*	N	Non-game	Invertivore		Lithophil	3
BSMW	Brassy minnow	*Hybognathus hankinsoni*	N	Non-game	Planktivore/ detritivore		Phytophil	3
BUSK	Blue sucker	*Cycleptus elongatus*	N	Non-game	Invertivore/ herbivore		Lithopelagophil	3
BWFN	Bowfin	*Amia calva*	N	Commercial	Carnivore		Phytophil	4
CARP	Common carp	*Cyprinus carpio*	NN	Commercial	Invertivore/ detritivore	Limno-rheophilic	Phytolithophil	3
CKCB	Creek chub	*Semotilus atromaculatus*	N	Non-game	Invertivore/ carnivore		Lithophil	4
CLDR	Crystal darter	*Ammocrypta asprella*	N	Non-game	Invertivore	Rheophilic	Psammophil	3
CLSR	Central stoneroller	*Campostoma anomalum*	N	Non-game	Herbivore		Lithophil	3
CMMW	Central mudminnow	*Umbra limi*	N	Non-game	Invertivore	Limnophilic	Phytophil	3
CNCF	Channel catfish	*Ictalurus punctatus*	N	Commercial	Invertivore/ carnivore	Rheophilic	Speleophil	4
CNLP	Chestnut lamprey	*Ichthyomyzon castaneus*	N	Non-game	Carnivore	Rheo-limnophilic	Lithophil	4
ERSN	Emerald shiner	*Notropis atherinoides*	N	Non-game	Planktivore	Rheo-limnophilic	Pelagophil	3
FHCF	Flathead catfish	*Pylodictis olivaris*	N	Commercial	Invertivore/ carnivore	Rheo-limnophilic	Speleophil	4
FHMW	Fathead minnow	*Pimephales promelas*	N	Non-game	Detritivore/ invertivore		Speleophil	3

(*continued*)

Table A1. (*Continued*)

Fish code	Common name	Scientific name	Native or non-native[a]	Exploitation status	Feeding guild	Habitat guild	Reproductive guild	Trophic status
FTDR	Fantail darter	*Etheostoma flabellare*	N	Non-game	Invertivore	Rheophilic	Speleophil	3
FWDM	Freshwater drum	*Aplodinotus grunniens*	N	Commercial	Invertivore/ carnivore		Pelagophil	4
GDEY	Goldeye	*Hiodon alosoides*	N	Commercial	Invertivore	Rheo-limnophilic	Lithopelagophil	3
GDRH	Golden redhorse	*Moxostoma erythrurum*	N	Commercial	Invertivore	Limno-rheophilic	Lithophil	3
GDSN	Golden shiner	*Notemigonus crysoleucas*	N	Non-game	Invertivore/ herbivore		Phytophil	3
GNSF	Green sunfish	*Lepomis cyanellus*	N	Recreational	Invertivore/ carnivore	Limnophilic	Polyphil	4
GZSD	Gizzard shad	*Dorosoma cepedianum*	N	Non-game	Herbivore	Limnophilic	Lithopelagophil	3
HFCS	Highfin carpsucker	*Carpiodes velifer*	N	Commercial	Detritivore	Limno-rheophilic	Lithopelagophil	3
IODR	Iowa darter	*Etheostoma exile*	N	Non-game	Invertivore		Phytophil	3
JYDR	Johnny darter	*Etheostoma nigrum*	N	Non-game	Invertivore	Limno-rheophilic	Speleophil	3
LGPH	Logperch	*Percina caprodes*	N	Non-game	Invertivore		Lithophil	3
LKSG	Lake sturgeon	*Acipenser fulvescens*	N	Recreational	Invertivore/ herbivore	Rheophilic	Lithopelagophil	3
LMBS	Largemouth bass	*Micropterus salmoides*	N	Recreational	Invertivore/ carnivore	Limnophilic	Polyphil	4
LNGR	Longnose gar	*Lepisosteus osseus*	N	Commercial	Carnivore	Rheo-limnophilic	Phytolithophil	4
MDDR	Mud darter	*Etheostoma asprigene*	N	Non-game	Invertivore	Limno-rheophilic	Phytophil	3
MMSN	Mimic shiner	*Notropis volucellus*	N	Non-game	Invertivore/ herbivore		Phytophil	3
MNEY	Mooneye	*Hiodon tergisus*	N	Commercial	Invertivore	Rheo-limnophilic	Lithopelagophil	3
NHSK	Northern hog sucker	*Hypentelium nigricans*	N	Commercial	Invertivore/ herbivore		Lithophil	3
NTPK	Northern pike	*Esox lucius*	N	Recreational	Carnivore	Limnophilic	Phytophil	4
OSSF	Orangespotted sunfish	*Lepomis humilis*	N	Recreational	Invertivore	Limnophilic	Lithophil	3
PDSN	Pallid shiner	*Notropis amnis*	N	Non-game				
PGMW	Pugnose minnow	*Opsopoeodus emiliae*	N	Non-game	Detritivore		Speleophil	3
PNSD	Pumpkinseed	*Lepomis gibbosus*	N	Recreational	Invertivore/ carnivore	Limnophilic	Polyphil	4
PRPH	Pirate perch	*Aphredoderus sayanus*	N	Non-game	Invertivore/ carnivore		Gill chamber brooder	4
QLBK	Quillback	*Carpiodes cyprinus*	N	Commercial	Invertivore/ detritivore	Limno-rheophilic	Lithopelagophil	3
RBST	Rainbow smelt	*Osmerus mordax*	NN	Non-game	Invertivore/ carnivore		Lithopelagophil	4
RKBS	Rock bass	*Ambloplites rupestris*	N	Recreational	Invertivore/ carnivore		Polyphil	4
RRDR	River darter	*Percina shumardi*	N	Non-game	Invertivore	Rheo-limnophilic	Lithophil	3
RVCS	River carpsucker	*Carpiodes carpio*	N	Commercial	Planktivore/ detritivore	Limno-rheophilic	Lithopelagophil	3
RVRH	River redhorse	*Moxostoma carinatum*	N	Commercial	Invertivore	Rheo-limnophilic	Lithophil	3
RVSN	River shiner	*Notropis blennius*	N	Non-game	Invertivore	Rheo-limnophilic		3
SFSN	Spotfin shiner	*Cyprinella spiloptera*	N	Non-game	Invertivore/ detritivore		Speleophil	3

(*continued*)

Table A1. (*Continued*)

Fish code	Common name	Scientific name	Native or non-native[a]	Exploitation status	Feeding guild	Habitat guild	Reproductive guild	Trophic status
SGER	Sauger	*Sander canadense*	N	Recreational	Invertivore/ carnivore	Rheo-limnophilic	Lithopelagophil	4
SHDR	Slenderhead darter	*Percina phoxocephala*	N	Non-game	Invertivore		Lithophil	3
SHRH	Shorthead redhorse	*Moxostoma macrolepidotum*	N	Commercial	Invertivore	Rheo-limnophilic	Lithophil	3
SJHR	Skipjack herring	*Alosa chrysochloris*	N	Recreational	Planktivore	Rheo-limnophilic	Phytolithophil	3
SKCB	Speckled chub	*Macrhybopsis aestivalis*	N	Non-game	Invertivore	Rheophilic	Lithopelagophil	3
SMBF	Smallmouth buffalo	*Ictiobus bubalus*	N	Commercial	Invertivore/ herbivore	Pelagic Limno-rheophilic	Lithopelagophil	3
SMBS	Smallmouth bass	*Micropterus dolomieu*	N	Recreational	Invertivore/ carnivore	Limno-rheophilic	Polyphil	4
SNGR	Shortnose gar	*Lepisosteus platostomus*	N	Commercial	Carnivore	Rheo-limnophilic	Phytophil	4
SNSG	Shovelnose sturgeon	*Scaphirhynchus platorynchus*	N	Commercial	Invertivore	Rheophilic	Lithopelagophil	3
SNSN	Sand shiner	*Notropis stramineus*	N	Non-game	Invertivore/ detritivore	Rheo-limnophilic		3
SPSK	Spotted sucker	*Minytrema melanops*	N	Commercial	Invertivore	Limno-rheophilic	Lithopelagophil	3
STCT	Stonecat	*Noturus flavus*	N	Non-game	Invertivore/ carnivore	Rheophilic	Speleophil	4
STSN	Spottail shiner	*Notropis hudsonius*	N	Non-game	Invertivore/ planktivore	Limno-rheophilic	Lithopelagophil	3
SVCB	Silver chub	*Macrhybopsis storeriana*	N	Non-game	Planktivore/ invertivore	Rheophilic	Lithopelagophil	3
SVLP	Silver lamprey	*Ichthyomyzon unicuspis*	N	Non-game	Carnivore		Lithophil	4
SVMW	Mississippi silvery minnow	*Hybognathus nuchalis*	N	Non-game	Detritivore	Rheo-limnophilic	Lithopelagophil	3
SVRH	Silver redhorse	*Moxostoma anisurum*	N	Commercial	Invertivore	Limno-rheophilic	Lithophil	3
TPMT	Tadpole madtom	*Noturus gyrinus*	N	Non-game	Invertivore/ planktivore	Limnophilic	Speleophil	3
TTPH	Trout perch	*Percopsis omiscomaycus*	N	Non-game	Invertivore/ carnivore		Lithophil	4
WDSN	Weed shiner	*Notropis texanus*	N	Non-game	Detritivore	Limno-rheophilic		3
WLYE	Walleye	*Sander vitreum*	N	Recreational	Invertivore/ carnivore	Limno-rheophilic	Lithopelagophil	4
WRMH	Warmouth	*Lepomis gulosus*	N	Recreational	Invertivore/ carnivore	Limnophilic	Lithophil	4
WSDR	Western sand darter	*Ammocrypta clara*	N	Non-game	Invertivore	Rheophilic	Psammophil	3
WTBS	White bass	*Morone chrysops*	N	Recreational	Invertivore/ carnivore	Pelagic rheo-limnophilic	Phytolithophil	4
WTCP	White crappie	*Pomoxis annularis*	N	Recreational	Invertivore/ carnivore	Limnophilic	Phytophil	4
WTSK	White sucker	*Catostomus commersoni*	N	Commercial	Invertivore/ detritivore		Lithopelagophil	3
YLBH	Yellow bullhead	*Ameiurus natalis*	N	Commercial	Invertivore/ carnivore	Limnophilic	Speleophil	4
YWBS	Yellow bass	*Morone mississippiensis*	N	Recreational	Invertivore/ carnivore	Pelagic rheo-limnophilic	Phytolithophil	4
YWPH	Yellow perch	*Perca flavescens*	N	Recreational	Invertivore/ carnivore	Limno-rheophilic	Phytolithophil	4

[a]Native, N; non-native, NN.

Attachment J

Memorandum for Record:

Pool 11 HREP Mud Lake

Modification Summary

CEMVR-EC-DN **March 6, 2017**

Memorandum for Record

Subject: Pool 11 Islands HREP Mud Lake Modification Summary

March 2014

A dye study of Mud Lake was conducted in response to Iowa Department of Natural Resources (IA DNR) fish telemetry data, indicating that overwintering fish were not utilizing newly dredged backwater channels. Restoring year-round aquatic habitat for fish was one of the primary objectives of the Pool 11 Islands Habitat Restoration and Enhancement Project (HREP), but increased velocities at the upstream inlet precluded meeting this objective in Mud Lake. The dye study suggested that the implementation of adaptive management measures would be necessary to reduce dredged channel velocities levels that support overwintering fish.

October 2014—August 2015

Not only is aquatic habitat compromised by high velocities in the winter, the opposite problem occurs during the summer. Summer habitat in Mud Lake is also inhospitable for fish due to seasonally reduced flows leading to noxious algal blooms that deplete oxygen and push fish into channel habitat. These blooms also preclude an abundance and diversity of aquatic plants, yet another objective of the P11 Islands HREP. Using Long Term Resource Monitoring (LTRM) data, Michl (2016) examined backwater lake nutrient dynamics relative to the main channel. Using experimentally-derived nutrient processing rates in the Upper Mississippi River, Michl (2016) modeled the impacts of various hydraulic flow regimes (management alternatives) on different water quality parameters. The model study tested the potential benefits for the Mud Lake HREP to manage hydrologic retention time to optimize summer habitat and denitrification rates. Many associated environmental factors, like dissolved oxygen, would also likely improve summer habitat for fishes. The model demonstrated the potential to increase denitrification with increased flow from the main channel. Michl (2016) hypothesized that the adaptive management alternative with the greatest amount of control (e.g., gated culvert) would likely improve both summer and winter aquatic habitat in Mud Lake. In short, closing the gates prior to winter would reduce flows and increase fish usage, while opening the gates following spring flooding would increase flow and nutrients, optimizing summer aquatic habitat for both vegetation and fish.

September 2015

On September 10, 2015, representatives from the USACE, USFWS, and Iowa and Wisconsin DNRs met to discuss potential adaptive management modifications to the Mud Lake inlet structure as a response to both high velocities in the dredged channels and underutilization of the project by overwintering fish. As a result, managers decided to utilize the rock stockpile at Mud Lake to fill in the upstream inlet prior to the 2015-2016 winter season. This decision provided an opportunity to study backwater response and further support or refute hypotheses raised by Michl (2016).

October 2015

On October 14, 2015, the project sponsor worked with the Dubuque County Conservation Board and Iowa DNR to modify the upper inlet in Mud Lake by placing rock from an adjacent stockpile. Rock placement was designed to allow some injection of flow at all times; i.e., leaky riprap. In January 2016, IA DNR reported that this leaky riprap was performing as intended. Local fisherman reported many small bluegill, crappie, and yellow perch using the site (Gritters, email; Note* still awaiting response from IA DNR to see if there is actual monitoring data post-mod).

February 2016

As part of this adaptive management strategy, the USACE performed another dye study in February 2016, to better characterize velocity and circulation patterns as a response to the project modification (Bierl and Bruns, 2016). Following the rock closure, velocities in the upper dredged channels near the inlet were significantly reduced from Mud Lake through Zollicoffer Slough. There did not appear to be any adverse impacts to dissolved oxygen levels due to flow reduction. Overall, the modification to the Mud Lake inlet was sucessful in reducing velocities in the upstream areas of the project to support fish overwintering requirements.

Also noticeably different in 2016, are increases in velocity from the lower inlet, causing flows to extend further upstream in the main dredged channel than in 2014. It appears that the project is still subject to higher velocities adjacent to the lower inlet; additional adaptive management measures may need to be investigated if monitoring continues to indicate underuse by fish. Figures 1 and 2 help characterize the effect of the rock closure on backwater velocities.

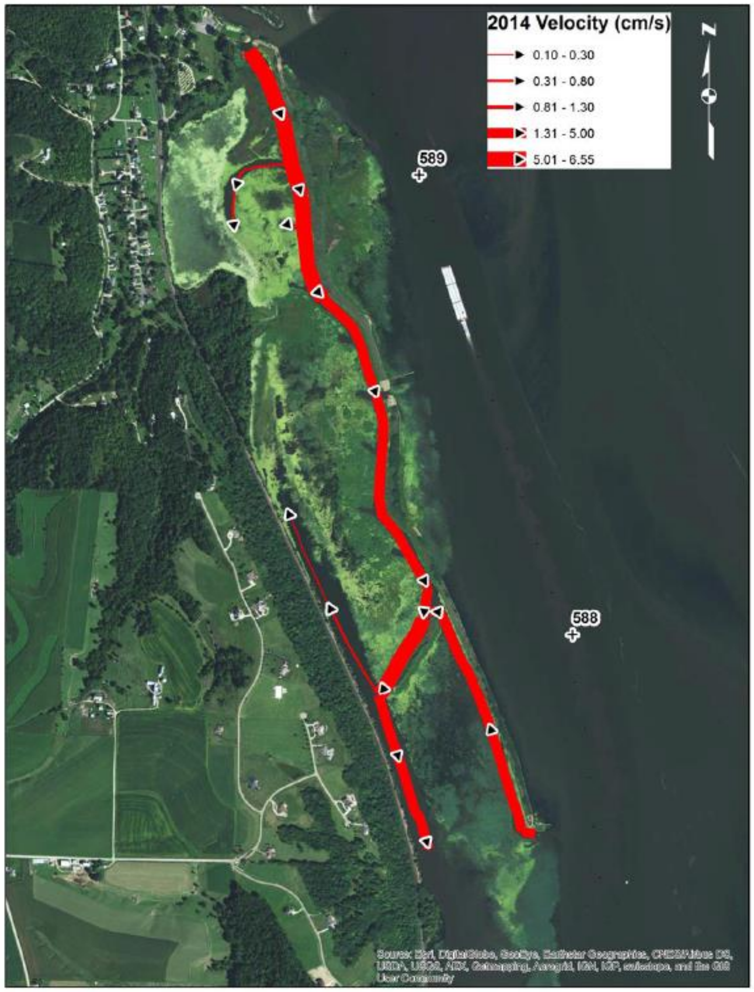

Figure 1. Mud Lake HREP velocities on March 10, 2014

Figure 2. Mud Lake HREP velocities on February 15, 2016

June-September, 2016

Preliminary water quality observations suggest that reduced flows from the rock closure at the upper inlet has an effect on summer habitat quality. Dissolved oxygen levels were reduced compared to pre-modification years (Figure 3); Chlorophyll *a* spiked significantly (Figure 4); and water clarity was reduced (Figure 5). These observations add further support for the hypothesis that decreasing backwater connectivity may impact habitat quality in the summer months by reducing potential for nutrient sequestration (Michl, 2016).

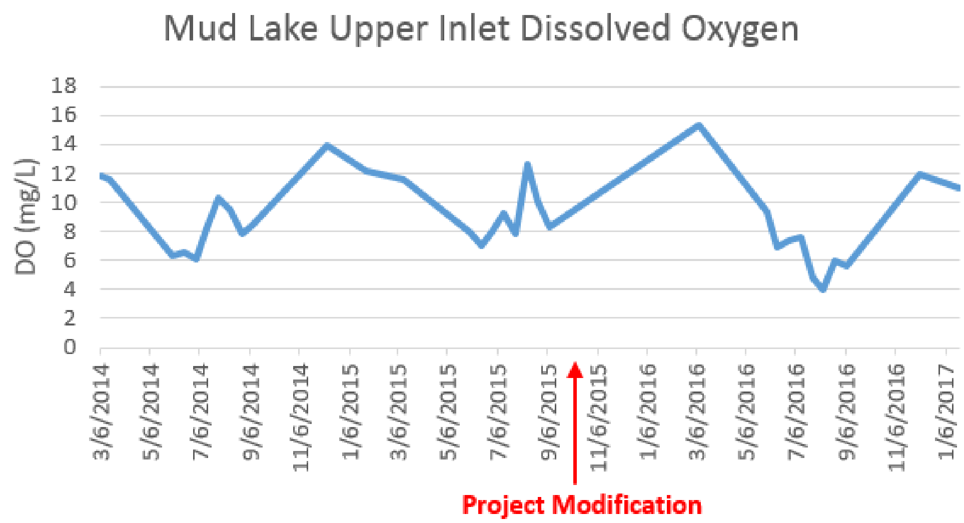

Figure 3. Mud Lake Upper Inlet Dissolved Oxygen: Pre- and Post- Modification

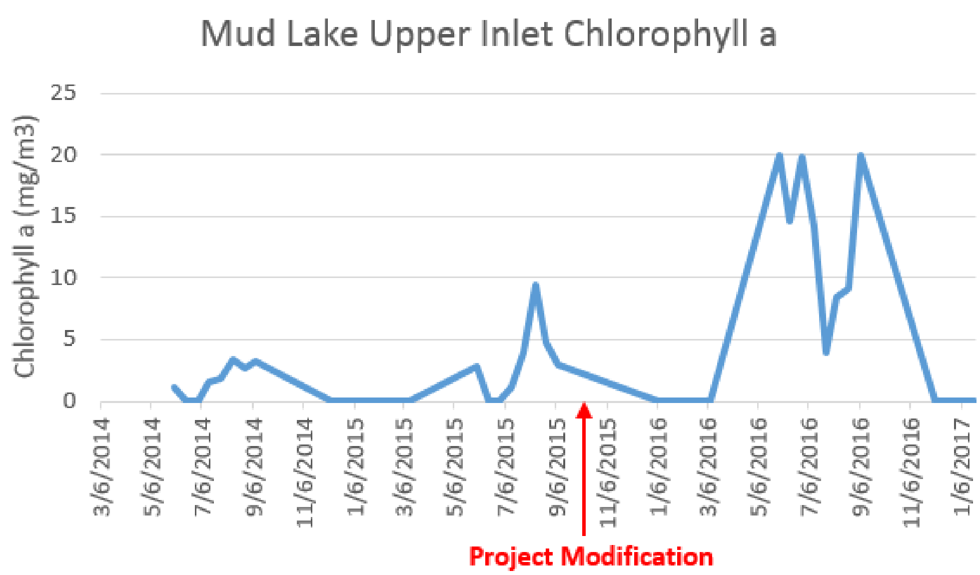

Figure 4. Mud Lake Upper Inlet Chlorophyll a: Pre- and Post- Modification

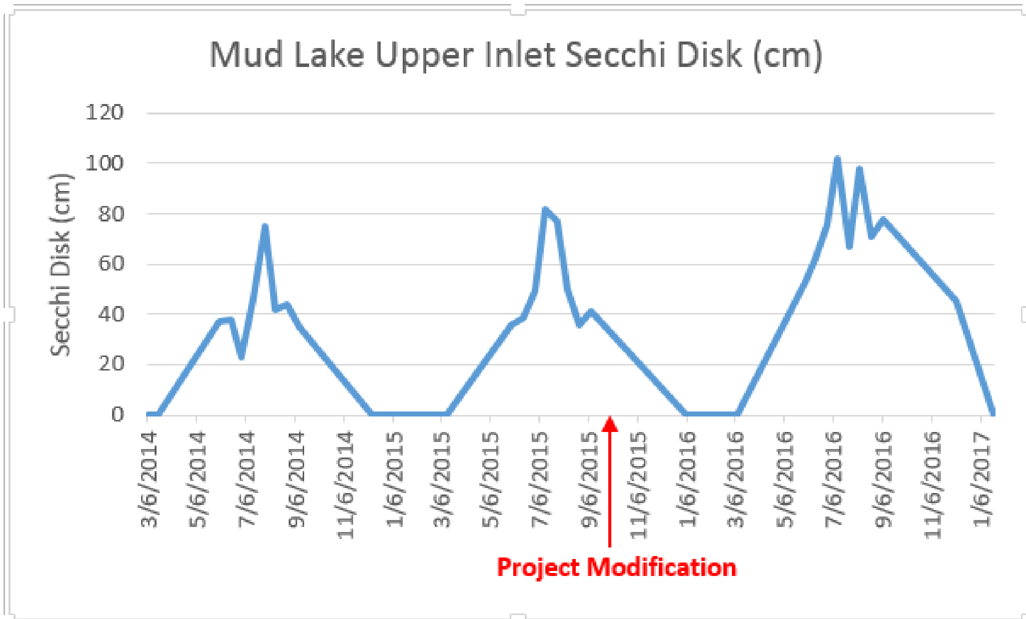

Figure 5. Mud Lake Upper Inlet Secchi Disk: Pre- and Post- Modification