Response to Comments on the 2020 Draft Wisconsin Consolidated Assessment and Listing Methodology (WisCALM)

March 2019

Water Evaluation Section, Water Quality Bureau
Environmental Management Division

A public comment period was held from January 22, 2019 to March 1, 2019. Comments were received from 5 separate entities.

Response to Dave Marshall, Underwater Habitat Investigations LLC

*"An important class of lakes is overlooked in the draft guidance. Oxbow spring lakes are the most threatened and degraded class of lakes in Wisconsin (Marshall 2012) but are overlooked due to their small size. These small lakes often provide many more ecosystem services than much larger glacial lakes and impoundments. They are vital for lateral river fish migrations, diverse floodplain wildlife communities, many endangered and threatened species and naturally functioning river systems. Oxbows often behave as spring lakes since they typically intercept massive aquifers with distinct outlets to connected rivers. They are complex since the hydrology changes drastically as soon as river stages rise. Oxbow lakes are the primary habitat for the State Endangered starhead topminnow (*Fundulus dispar*) along the Mukwonago River, Illinois Fox River, Sugar River, Black River and especially the Lower Wisconsin State Riverway. They remain endangered partly due to habitat degradation as oxbow lakes were lost due to floodplain aggradation and more recently due to groundwater pollution in the form of high NO3-N+NO2-N. Impacts of polluted groundwater on surface waters is twofold. Along the Lower Wisconsin River groundwater discharges nitrate concentrations to oxbow spring lakes at levels exceeding the Drinking Water Standard and chronic toxicity levels. Nitrogen driven eutrophication results in excessive growths of free floating plants (duckweeds and green filamentous algae) the literally smother these lakes, undermining both recreational uses and ecosystem functions (Marshall 2013, Marshall et al. 2016). John Sullivan (DNR retired) documented the water quality impacts of dense free floating plants along the Mississippi River as well."*

"In my opinion, WISCALM assessment of lakes needs to include small oxbow spring lakes, that provide outsized ecological services, and the threats that currently threaten them; nitrate pollution and excessive free floating plant cover."

> Identifying and assessing oxbow spring lakes involves establishing appropriate benchmarks. This process requires assembling and reviewing relevant studies and working with experts to develop consensus on protocols and metrics. This will be considered for future iterations of WisCALM as there is unfortunately not enough time to develop the benchmarks for implementation in the 2020 version. Thank you for providing this information.

Response to Shawn Giblin, Wisconsin DNR

"I would have an interest in seeing criteria related to duckweed and filamentous algae mats on backwaters of the Mississippi (and likely other large-river backwaters of the state). These would be based on the attached paper (Giblin et al 2014). These mats are proliferate in response to both N and P. Examination of tissue nutrient ratios suggest that the mats are P-limited early in the growing season and N-limited later in the growing season. One thing we observe on the UMR is that when duckweed coverage is > 80%

backwater hypoxia is very likely… It seems that we should have criteria for this impairment. These mats can cover up to 50% of the available backwater area under any given year."

> This concept and supporting research appear to have scientific merit and will be considered for future iterations of WisCALM. The department will need time to develop appropriate protocols and metrics. Thank you for providing this information.

"I would also like to see us develop TSS standards for large rivers in WI based on the attached paper (Giblin 2017). On the UMR, we see a threshold at 16 mg/L TSS. At TSS > 16 mg/L we see reduced biomass of native and recreational fishes."

> Total Suspended Solids (TSS) standards are recognized as an important topic for criteria development because 14% of impairment listings are based on TSS. Modeling is under way to determine a potential threshold. Your work on the Upper Mississippi River will be included in any considerations of a large river TSS criteria.

"Page 58- Will bacteria standards for the UMR switch to E. Coli at some point? We have many beaches on the Mississippi River."

> There is currently a proposed rule revision in progress to switch from a surface water bacteria standard for fecal coliform to one for E. coli, in line with Federal recommendations. The rule is expected to be promulgated at some point in 2020. If/when that happens, E. coli criteria will replace the fecal coliform criteria for surface waters, including the Mississippi River. The 2022 version of WisCALM is expected to reflect the new E. coli criteria.

General Question: What about rivers that exceed CHLa of 20 ug/L (Miss. R., La Crosse R., Red Cedar R., etc.)? Are these assumed to be covered under the TP exceedance?

> There is a proposed rule revision in progress focused on using biological metrics, including chlorophyll-*a*, in river assessments. The recreation benchmark of no greater than 20 ug/L for 30% of days during the growth period will be applied to rivers.

Response to John Sullivan, Dave Marshall, and Jim Baumann, Retired Wisconsin DNR

"Use of Free Floating Plants (filamentous algae & duckweeds) in WisCalm Guidance"
[Comments truncated, please see original for full message.]

"A critical step in the discovery of nutrient-related water quality problems is the identification of key biological response variables that are directly influence by excessive nutrient inputs. The use of algal chlorophyll concentrations, harmful algae bloom frequency, cyanotoxins, algae cell counts and other metrics are clear examples. Another useful indicator of nutrient enrichment is the development of excessive filamentous algae (metaphyton) mats or thick coverings of duckweeds that develop in shallow aquatic systems including channel borders, backwaters, floodplain lakes and deep water wetlands (Sullivan 2008, Houser and Richardson 2010, Giblin et al. 2014). Floating mats of filamentous algae or duckweeds are free floating plants (FFP) that can negatively impact dissolved oxygen levels or contribute to significant shading of submersed aquatic vegetation beds through the attenuation of surface light. Thick growths of filamentous algae on submersed vegetation may negative impact submersed vegetation due to the competition for nutrients, dissolved gasses (O^2 & CO^2) and light and may contribute to a complete collapse of these important aquatic plant communities (Phillips et al. 1998 and Hilton et al. 2006). This was likely a factor in the massive decline of submersed aquatic vegetation in the UMR in the late 1980s and has recently been observed in floodplain lakes in the Lower Wisconsin River (Marshall 2013). Further, thick mats of

filamentous algae or duckweeds seriously impact recreation use by making these areas difficult or impossible to traverse with a boat, especially paddlers."

"The Department needs to consider the impacts of nuisance growths of filamentous algae and duckweeds in the assessment of water quality use attainment. Fortunately, procedures have been developed (see attached file) and implemented to facilitate this process (Sullivan 2008, Marshall 2013 and Houser et al. 2014). However, specific impairment thresholds using FFP water quality indicators have not been adopted by the Department. We have drafted methodologies for identifying nutrient-related impairment problems using FFPs as a response factor (see table below). A tiered approach is recommended that would consider differences in surface water and use classification. We would urge the Department to consider these recommendations for Wisconsin's Consolidated Assessment and Listing Methodology for Clean Water Act Reporting."

This concept and methodology appear to have scientific merit. The Department will need some time to further consult with you and other experts to develop consensus on appropriate protocols and metrics. This topic will be considered for future iterations of WisCALM as there is unfortunately not enough time to finalize appropriate protocols and metrics for the 2020 version. Thank you for providing this information.

Response to Kristi Minahan, Wisconsin DNR

"On p. 55, it states: This approach involves the calculation of a 90% confidence limit around the median of a TP sample data… (lakes section also uses 90% CI.) In the draft code language, we state that they need to *calculate an 80% CI around the median. (for lakes, 80% CI around the mean). Let's make sure our language lines up…"*

The language within WisCALM was unclear in relation to the confidence interval method by using the terms "90% confidence limit" and "90% CI". The interval has always been calculated as an 80% confidence interval. The percentages for the CI have been updated throughout WisCALM. Rather than mentioning the percentage in the upper and lower value labels the terms "Upper Confidence Limit (UCL)" and "Lower Confidence Limit (LCL)" have been substituted. Figure 13 in WisCALM has been updated to the figure below:

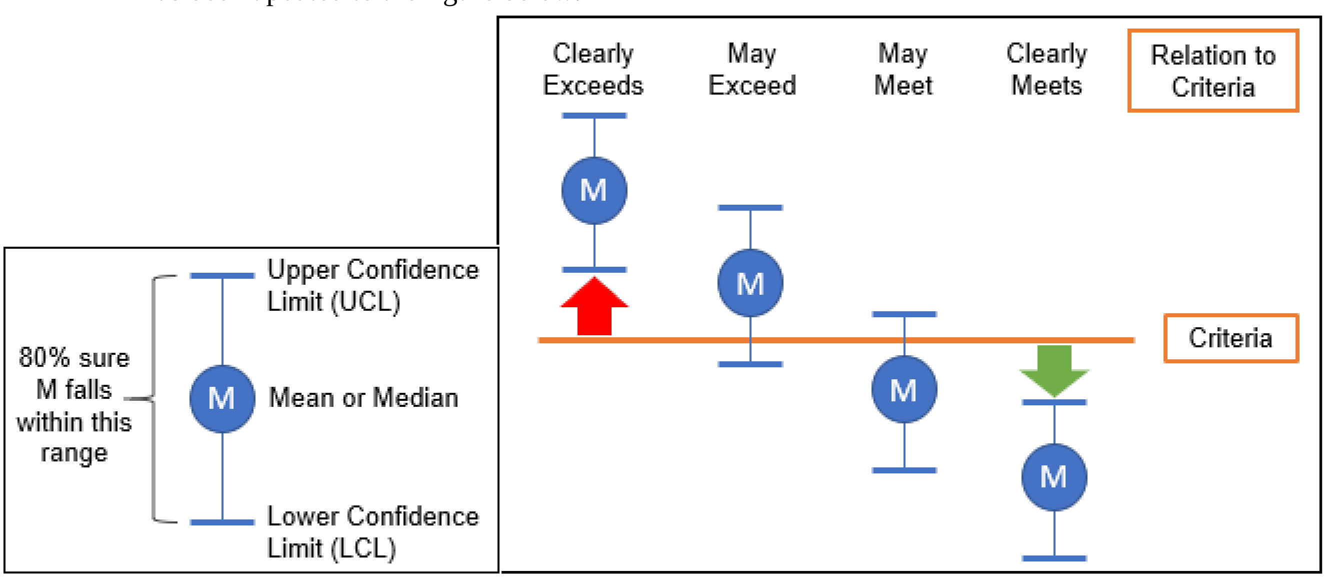

Figure 13. Comparison of the Upper and Lower Confidence Limit values and Mean/Median (M) to the criteria.

"We state that we should only use one sample per month (closest to middle of month). But for more in-depth studies, they often take a weekly or every-other-week sample, and shouldn't all those be used when they are available, as long as it was a set of dates selected ahead of time to be periodic rather than data targeting certain flows?"

While the assessment package only takes the value closest to the middle of the month, there is still the option to use all samples available by averaging all samples within a month. There is an Excel spreadsheet that can be used for individual station calculations when required or requested. There currently isn't sufficient time to update the necessary programming before assessments need to be done. For the next iteration of WisCALM (2022), the assessment package can be updated to include all samples.

Response to Mark, Citizen

"I appreciate the work being done. I wish for more funding for DNR and assistance to farmers to abate run off. Buffers need to established [sic] along water ways either by public funding, public land easement and stricter laws with or without funding to reduce run off."

Thank you for your support of Wisconsin's water quality standards and assessment methods.

"There is way too much algae growth in our lakes and streams in most of state water."

WisCALM has benchmarks for assessing algal growth in lakes and reservoirs. Benchmarks for algal growth in streams and rivers are under development and might be included in the 2022 WisCALM. With these benchmarks we can determine which waters have issues with algal growth and determine next steps.

Public Comments on Draft 2020 WisCALM

Dave Marshall, Underwater Habitat Investigations LLC

Shawn Giblin, Wisconsin DNR

John Sullivan, Dave Marshall, and Jim Baumann, Retired Wisconsin DNR

Kristi Minahan, Wisconsin DNR

Mark, Citizen

From: Dave Marshall <underh2ohab@mhtc.net>
Sent: Sunday, February 17, 2019 12:00 PM
To: DNR Impaired Waters
Subject: WISCALM comment and attached reports
Attachments: SWG11Final.pdf; Restoring Lower Wisconsin State Riverway Oxbow Lakes Final.pdf; Lower Wisconsin River Floodplain Lakes Water Pollution Investigation.pdf

Follow Up Flag: Follow up
Flag Status: Flagged

An important class of lakes is overlooked in the draft guidance. Oxbow spring lakes are the most threatened and degraded class of lakes in Wisconsin (Marshall 2012) but are overlooked due to their small size. These small lakes often provide many more ecosystem services than much larger glacial lakes and impoundments. They are vital for lateral river fish migrations, diverse floodplain wildlife communities, many endangered and threatened species and naturally functioning river systems. Oxbows often behave as spring lakes since they typically intercept massive aquifers with distinct outlets to connected rivers. They are complex since the hydrology changes drastically as soon as river stages rise. Oxbow lakes are the primary habitat for the State Endangered starhead topminnow (Fundulus dispar) along the Mukwonago River, Illinois Fox River, Sugar River, Black River and especially the Lower Wisconsin State Riverway. They remain endangered partly due to habitat degradation as oxbow lakes were lost due to floodplain aggradation and more recently due to groundwater pollution in the form of high NO3-N+NO2-N. Impacts of polluted groundwater on surface waters is twofold. Along the Lower Wisconsin River groundwater discharges nitrate concentrations to oxbow spring lakes at levels exceeding the Drinking Water Standard and chronic toxicity levels. Nitrogen driven eutrophication results in excessive growths of free floating plants (duckweeds and green filamentous algae) the literally smother these lakes, undermining both recreational uses and ecosystem functions (Marshall 2013, Marshall et al. 2016). John Sullivan (DNR retired) documented the water quality impacts of dense free floating plants along the Mississippi River as well.

In my opinion, WISCALM assessment of lakes needs to include small oxbow spring lakes, that provide outsized ecological services, and the threats that currently threaten them; nitrate pollution and excessive free floating plant cover. Thank you,

Dave Marshall, Aquatic Ecologist – P.H.
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Surveys of River Floodplain Habitats for Fish Species with Inventory Needs, SGCN and Associated Off-channel Fish Populations

State Wildlife Grant (SWG-11) Final Report

Part A: State Wildlife Grant Project SWG11-MON-004 - Resurvey a Subset of SWG09 River Floodplain Habitats for Fish Species with Inventory Needs, SGCN, and Associated off-channel Fish Populations

Part B: State Wildlife Grant Project SWG11-CAT2-007 - Surveys of New River Floodplain Habitats for Fish Species with Inventory Needs, SGCN, and Associated off-channel Fish Populations

Drought of 2012

Lower Chippewa River oxbow

David W. Marshall, Underwater Habitat Investigations LLC

November 2012

Summary

In 2009 and 2010, 143 off-channel river floodplain sites were sampled as part of a State Wildlife Grant (SWG09) to gather information on the weed shiner (Notropis texanus), pugnose minnow (*Opsopoeodus emiliae)*, pirate perch (*Aphredoderus sayanus*), mud darter (*Etheostoma asprigene*), silver chub (*Macrhybopsis storeriana)* and other off-channel species of Greatest Conservation Need including the starhead topminnow (*Fundulus dispar*) and lake chubsucker (*Erimyzon sucetta*). As part of the SWG11-MON-004 project, a subset of the SWG09 sites were resurveyed; 53 off-channel sites were sampled in 2011 and 62 sites in 2012. Sites selection was primarily based on SWG09 survey records where target fish species had been collected. A few sites were selected based solely on apparent favorable habitat and fish assemblage characteristics even though none of the target species had been found.

The frequency of occurrence for State Endangered starhead topminnows (*Fundulus dispar*) within their Wisconsin range was 56% in 2011 and was 58% in 2012. The frequency of occurrence for this species during 2011 and 2012 sampling years was lower than the SWG09 dataset (64%) when river flow rates were higher.

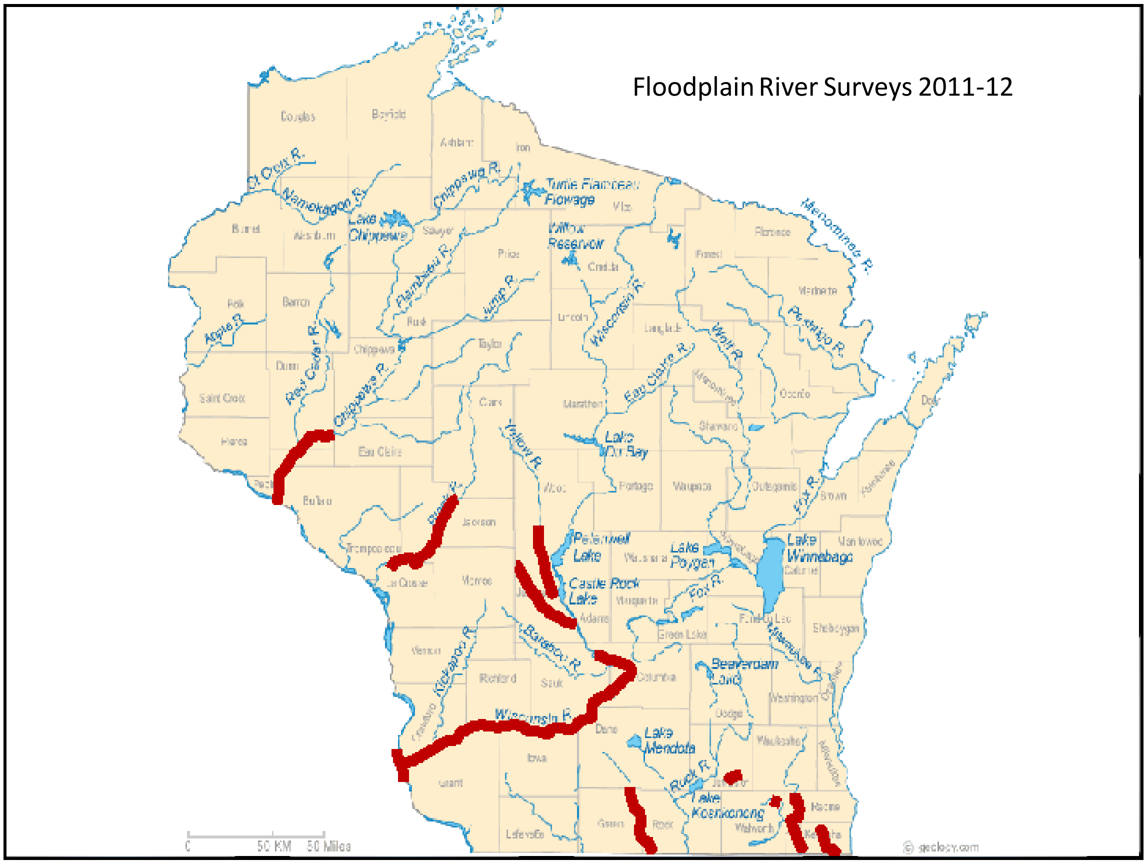
The extensive Lower Wisconsin River floodplain remains a sustainable environment for starhead topminnow populations over a range of river flow rates, including both flood and drought periods. However, the extreme drought conditions in 2012 significantly reduced starhead topminnow habitats in many areas of the state including a Black River oxbow and along the Lower Sugar River. The combination of severe drought in 2012, coupled with floodplain aggradation, resulted in complete loss of off-channel habitats along the Sugar River where starheads were found in 2010. No starhead topminnows were found along the Sugar River in 2012 even though over 300 individuals were recorded in 2010 during a period of flooding and recruitment site expansion.

Comparing 2011 with 2012 surveys, the USGS flow rates recorded for survey days declined from 97% of the long term median value in 2011 to 67% in 2012. Estimated tree roots/woody debris habitats had also declined in 2012. At numerous sites in 2012, tree roots/woody debris habitats rested above the waterline. The reduction in habitat during the drought coincided with the declines of both State Special Concern mud darter (*Etheostoma asprigene*) and pirate perch (*Aphredoderus sayanus*) collection sites. Both species display a strong affinity for tree roots. Mud darter site collections declined from 49% in 2011 to 37% in 2012 and pirate perch sites declined from 63% in 2011 to 38% in 2012 within their respective ranges. Other factors may also play a role in these changes.

Target fish species with information needs or species of greatest conservation need were found at least once at 93% of the SWG11-MON-004 sites sampled annually over three separate years. Target species were collected each of three years at 66% of the sites. Mud darters were collected from the Wisconsin River near Portage downstream to the confluence with the Mississippi River, along the Mississippi River above and below the confluence with the Wisconsin River, the Black River and Silver Birch Lake along the Chippewa River. Pirate perch were collected along the Wisconsin River, the Lemonweir River, Glenn Lake along the

Mississippi River and the Black River. Starhead topminnows were collected in weedy off-channel habitats along the Lower Wisconsin River, Lower Black River and a single specimen along the Lower Sugar River in 2011. The starhead distribution does not extend above the Prairie du Sac dam in the Wisconsin River although habitats in some areas above the dam appear favorable. Lake chubsuckers (*Erimyzon sucetta*) were occasionally found along the Lower Wisconsin River (6 in 2011 and 7 in 2012) while weed shiners (*Notropis texanus*), pugnose minnows (*Opsopoeodus emiliae)*, Mississippi grass shrimp (*Palaemonetes kadiakensis)*, and banded killifish (*Fundulus diaphanus*) were found at lower numbers of sites.

Figure 1: Map of river floodplain areas surveyed for SWG11-MON-004 and SWG11-Cat2-007



The cut-off channel oxbows and other off-channel floodplain habitats along the Lower Wisconsin State Riverway appear to be more drought resistant than some of the rivers sampled as part of this project. However, threats and impairments of Wisconsin River oxbows included contaminated groundwater and excessive growths of metaphyton (filamentous algae and duckweeds) that blanketed Jones Slough and Norton Slough in 2011. Beneath the dense metaphyton mats, native floating-leaf and submersed aquatic plants were eliminated (with the exception of coontail – *Ceratophyllum demersum*). Photosynthetic suppression and atmospheric blockage resulted in very low dissolved oxygen levels. These problems were linked to high nutrient application rates over coarse sandy soils, shallow aquifer and center pivot irrigation.

Thirty-four sites were sampled along 10 rivers as part of SWG11-Cat2-007 in 2011-12. The surveys demonstrated that pirate perch were still present in Kilbourne Road Ditch in southeast Wisconsin, despite the significant influences of urbanization and agriculture. Pirate perch were not found in the adjoining Des Plaines River. In central Wisconsin, pirate perch were collected upstream of the Lemonweir River in the Little Yellow River and in Little Roche a Cri Creek. Both streams flow into Castle Rock Flowage. Pirate perch were not found in the Yellow River where they had been collected in the past.

Low numbers of starhead topminnows were found in the Illinois Fox River less than one mile above the confluence with the Mukwonago River extending downstream of the Waterford dam. Starheads were typically collected within submersed or overhanging riparian vegetation near shore along with blackstripe topminnows, another Fundulus species. Oxbows, ponds and other off-channel habitats along the Illinois Fox River were dry or nearly dry in 2012. Surprisingly, no starhead topminnows were found in ORW Lulu Lake but a single juvenile lake chubsucker was dip netted there. Consistent with SWG09 findings, starhead topminnows were not found above the Brodhead dam along the Sugar River.

The severe drought of 2012 significantly reduced habitats along the Chippewa River in west central Wisconsin. Accessible oxbows were dry or nearly dry and therefore new sites were not found. More intensive floodplain fish surveys are needed to assess mud darters and other rare off-channel fishes along the Chippewa River and other rivers.

Recommendations

1. As Climate Change models predict greater frequency of extreme droughts and flooding, oxbows and sloughs should be restored along incised rivers channels and within aggrading floodplains. Excavation is needed to restore oxbow storage capacity lost to sedimentation. Excavation along the former thalweg banks may be the best option for restoring groundwater connectivity, an essential habitat feature required for starhead topminnow survival and survival of the entire off-channel floodplain fish assemblage.
2. The long term regulatory policy that requires "shallow wildlife scrapes only" within floodplains should be changed. Across a natural floodplain, a spectrum of habitats range from temporary floodwater pools, to shallow ponds to deeper oxbows with significant groundwater connectivity. Shallow ponds, perched above the groundwater, will not likely sustain threatened off-channel floodplain fish in aggrading floodplains, particularly during extended droughts and hard winters.
3. Conservation aquaculture should be an option for increasing numbers of rare species where populations are critically low, such as starhead topminnows in the Sugar River. Transfers should also be an option to expand the range of rare species to within a river system where dams block migrations.
4. Contaminated groundwater, causing water quality decline in some Lower Wisconsin State Riverway oxbows, is an example where main channel environmental management does not protect rare species such as starhead topminnows, pirate perch, lake chubsuckers, mud darters, weed shiners or pugnose minnows. Environmental protections are needed across the entire river floodplains. Conservation buffers are

needed to protect all off-channel habitats that are vulnerable to groundwater contamination and runoff pollution. High capacity groundwater extractions and nutrient applications should be carefully scrutinized to protect oxbow water quality and habitats.

5. Baseline surveys should become standardized for off-channel fish, macrophytes, macroinvertebrates and water quality monitoring. Surveys should be conducted in late summer following recruitment and not during high flows when sustainable habitats may not be differentiated from flooded areas. An exception could include assessing starhead topminnow populations that disperse across flooded areas.
6. Management of levee ponds should be re-evaluated. Artificially manipulating impoundments for complete drawdown and flooding can benefit fish populations that thrive on disturbances such as common carp and fathead minnows. Levee ponds should be managed to sustain the off-channel floodplain fish assemblage and macrophytes, ultimately reducing potential for common carp recruitment. Common carp reproduction is rare along the Lower Wisconsin River and these habitats should be used as models for off-channel impoundment management.
7. Monitoring to detect invasive species in off-channel habitats is needed since they pose threats to native fish populations and habitat. In 2012, an invasive mosquitofish (*Gambusia affinis*) population expanded in a small Sugar River slough while the native off-channel fish assemblage declined significantly, including State Endangered starhead topminnows.
8. Borrow pits, where permitted within floodplains, should be required to establish extensive littoral zones that support rare fish species and the entire off-channel floodplain fish assemblage.
9. Aquatic plant management activities should be carefully regulated in Waterford, Tichigan Lake and other lake/impoundments where State Endangered starhead topminnows, lake chubsuckers and other macrophyte obligate species occur.
10. Rivers that sustain favorable off-channel habitats and rare fish should be prioritized for protection.
11. Rivers or river segments that sustain only very low numbers of our target group of floodplain fish, due to floodplain aggradation or groundwater contamination, should be prioritized for restoration.
12. Degraded oxbows and other floodplain habitats should be assessed under the Clean Water Act 303d list of impaired and threatened surface waters.
13. Beaver dams should be protected along river floodplains since these natural impoundments greatly expand habitats for rare off-channel fish populations (see Tiffany Bottoms SWA).

SWG11-MON-004 Project

Survey participants: Dave Marshall - UHI LLC, Patty Avery, Dave Grey, Jean Unmuth - WDNR, Greg Searle – WDNR, Matt Miller – UW Platteville, Russ Wolf – UW Platteville, Sue Graham - WDNR, Megan Phillips -USRWA, Bill Keen - USRWA, Shane Herkert - USRWA, Amy Schmudlach – Wisconsin Canoe Company, Ryan Schmudlach – Wisconsin Canoe Company, Timm Zumm -

FLOW, Matt Krueger –River Alliance, Merel Black, Sherryl Jones, Doug Jones, Jerrod Parker, Denny Caneff – River Alliance, Allison Werner – River Alliance, Chris Clayton – River Alliance, Erin Courtenay – River Alliance, Helen Sarkinos – River Alliance, Laura McFarland – River Alliance, David Pausch – River Alliance, Corey Puzach – USFWS, Ruth Person – WDNR, Ashley – Beranek, Kristi Minahan – WDNR, Matthew Jacobson – WDNR, Samuel Betterley – WDNR, Jennifer Bergman – WDNR, Scott Provost - WDNR and Tara Bergeson WDNR for project planning and administration.

Methods

Of the 161 sites sampled in 2009 and 2010 as part of SWG09, 53 sites were re-sampled in both 2011 and 2012 and nine additional sites were re-sampled in 2012. Sites were selected if one or more of the target fish species with informational needs or SGCN (pirate perch, mud darters, weed shiners, pugnose minnows, starhead topminnows, lake chubsuckers, banded killifish) had been collected or if habitat conditions appeared highly favorable but of the target species were found. Most of the sites were located along the Wisconsin River. State Special Concern Mississippi grass shrimp were incidentally collected during the fish sampling surveys.

Fish populations were primarily sampled with a small towed single probe DC electroshocking barge. Electroshocking time was recorded for each site. Due to the high number of sites and rivers that were sampled as part of this project, sampling times were relatively brief (not exceeding 45 minutes electroshocking time). At several locations, sampling gear included small mesh dip nets and small mesh seine. Netting was the preferred sampling technique where specific conductance levels exceeded 1,000 uS/cm or were less than 100 uS/cm. Habitat information was gathered by estimating fish cover in the forms of aquatic plants, rock and tree roots/woody debris. Habitat characteristics were assigned numerical values ranging from 3 (high), 2 (medium), 1 (low) to 0 (absent). Basic water quality was assessed by measuring dissolved oxygen/temperature (YSI Model 52), temperature/specific conductance (Extech) and water transparency (120 cm transparency tube). River flow rates were recorded from the USGS National Water Information System website.

Findings

Results of the SWG11-MON-004 project demonstrated that long stretches of the Wisconsin River, Black River and Lemonweir River floodplains are often inhabited by species with information needs, SGCN and associated off-channel fish populations. Our sampling indicated that species comprising an off-channel fish assemblage can vary at a given site but at least one of our target species is typically represented in favorable habitats along these river systems. At the 62 sites that we re-sampled from the SWG09 surveys, along with several from sites sampled in 2008, rare fish were found at least once per site during a three year sampling period at 97% (57) of the sites and each year at 66% (40) of the sites.

Habitats that typically supported the target group of rare fish included oxbows and other backwater areas with abundant growths of native floating-leaf and submersed aquatic plants and large amounts of tree roots/woody debris. The nearshore fish surveys usually encompassed both of these habitats and therefore yielded fish species with different habitat preferences. Figure 2 displays mean habitat values for starhead topminnows, lake chubsuckers, mud darters and pirate perch. Regarding the former two species, aquatic vegetation was most important habitat and is consistent with published accounts. The figure also revealed that tree roots/woody debris and occasionally rock habitats were found at starhead topminnow and lake chubsucker sites, but these species were rarely found beyond vegetated areas. Both mud darters and pirate perch display a strong affinity for tree roots/woody debris but pirate perch appears to be more of a habitat generalist and was also found in aquatic vegetation at some locations. Rocky substrates were not commonly found along the floodplains. However, both mud darters and pirate perch were occasionally found these habitats, particularly where tree roots/woody debris habitat was scarce. In general, the habitat use patterns for these species were consistent with the SWG09 findings and published accounts.

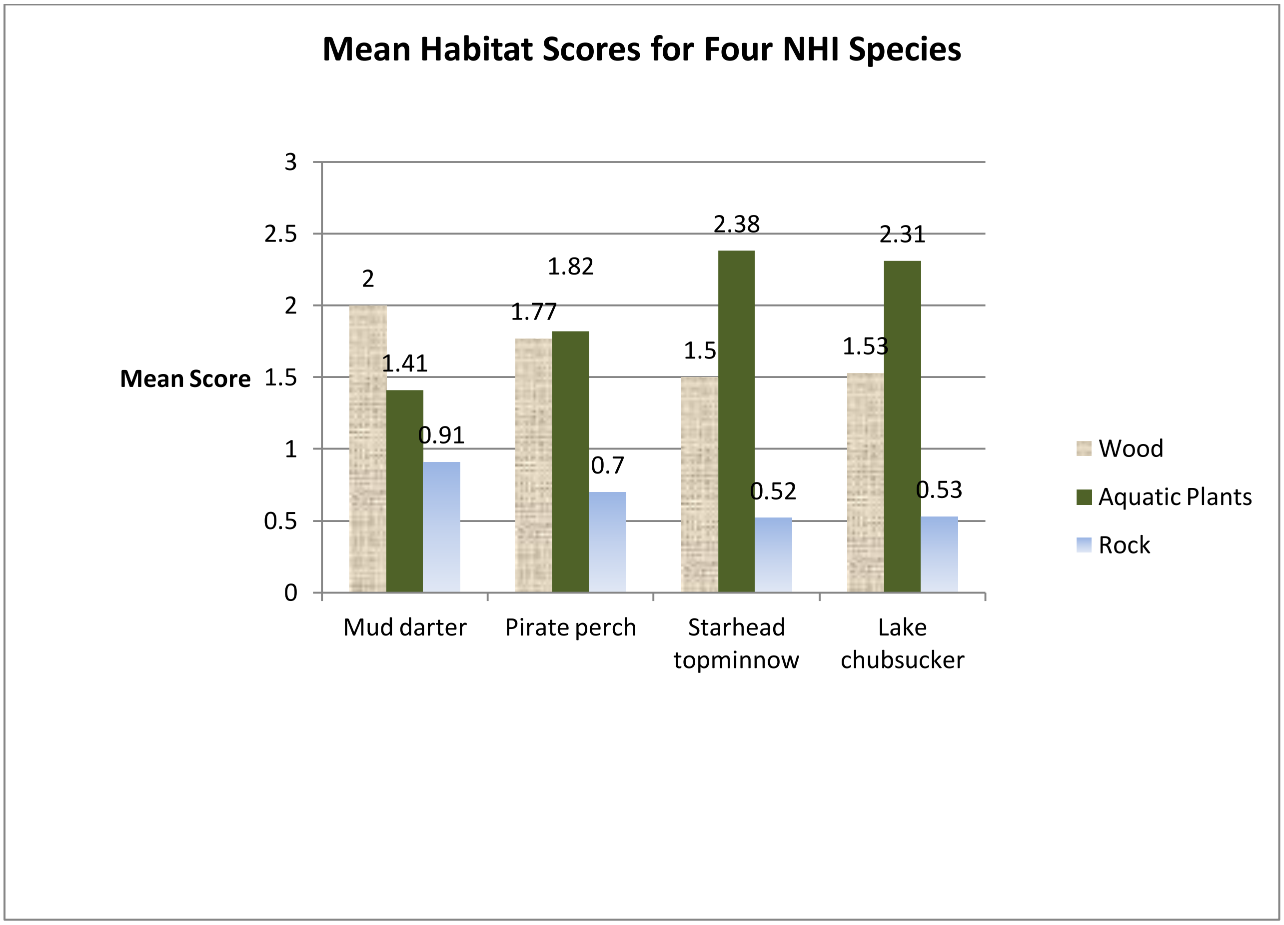
The off-channel floodplain fish assemblage varied somewhat within rivers and across the state, but common species were typically found. Figures 3-5 display common associate species of mud darter, pirate perch and starhead topminnow for both 2011 and 2012 collections. Bluegills, largemouth bass, American grass pickerel and central mudminnows were most common fish found associated with these three target species. This pattern is very similar to the SWG09 surveys conducted in 2009-10. These patterns were similar for weed shiners, pugnose minnows, and Mississippi grass shrimp since they were found in the same habitats but were far less common. Overall, the species associations were influenced by the higher number of sampling sites (40) along the Wisconsin River and can change somewhat across river basins.

Figure 7 displays the frequency for mud darter, pirate perch and starhead topminnows collections as part of the SWG11-MON-004 project (62 sites). Since the distribution for each species exceeds overlap, the histogram was adjusted to distribution for each species. The histogram compares 2011, 2012 and SWG09 data, as well as a few surveys conducted in 2007 and 2008. Starhead topminnows were found at approximately the same frequency in 2011 and 2012 but the endangered fish was collected at higher frequency in previous years. Most of the SWG09 sites were sampled in 2010 when flooding expanded starhead topminnow habitats. Pirate perch and mud darters were found at fewer sites in 2012 than in 2011 and may reflect reduced habitat (estimated tree roots/woody debris habitat at pirate perch sites: 2011 mean = 2.1, 2012 mean = 1.6, P = 0.01; estimated tree roots/woody debris habitat at mud darter sites: 2011 mean = 2.3, 2012 mean = 1.7, P = 0.01). Nearshore woody habitats often rested above the waterline during the drought of 2012. An estimated 24% decline in woody debris habitat in 2012 coincided with low river flow rates; 97% of the USGS long term median flows in 2011 compared with 67% in 2012.

The most significant effects of the 2012 drought occurred in Stevens Lake along the Black River and Sugar River in Rock County where previous starhead topminnow sites were completely dry. Few species remained in the nearly dry Stevens Lake while no starhead topminnows were found along the Sugar River in 2012, even after significant occurred during the 2010 flooding.

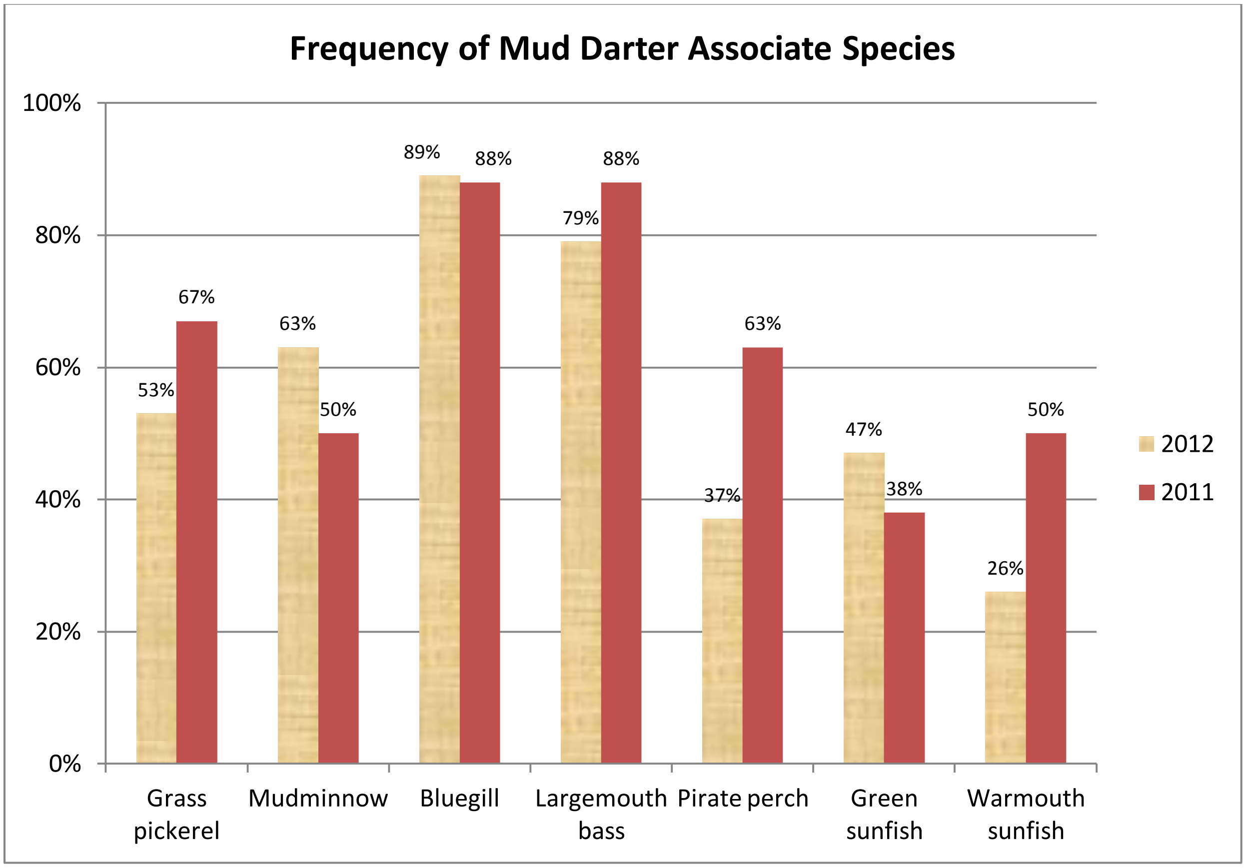
With respect to the subset of sampling sites, frequency of occurrence for pirate perch and mud darters was lower (43% each) during the SWG09 project than in 2011. Since most of the SWG09 project sampling was conducted during the floods of 2010, the lower collection site numbers may have reflected the sampling difficulty in high water. Tree roots/woody debris often hangs across steep banks that are difficult to sample in high water. As a result, accessible sampling sites were limited to flooded terrestrial areas in 2010.

Figure 2: Mean habitat scores for starhead topminnow, lake chubsucker, mud darter and pirate perch sites



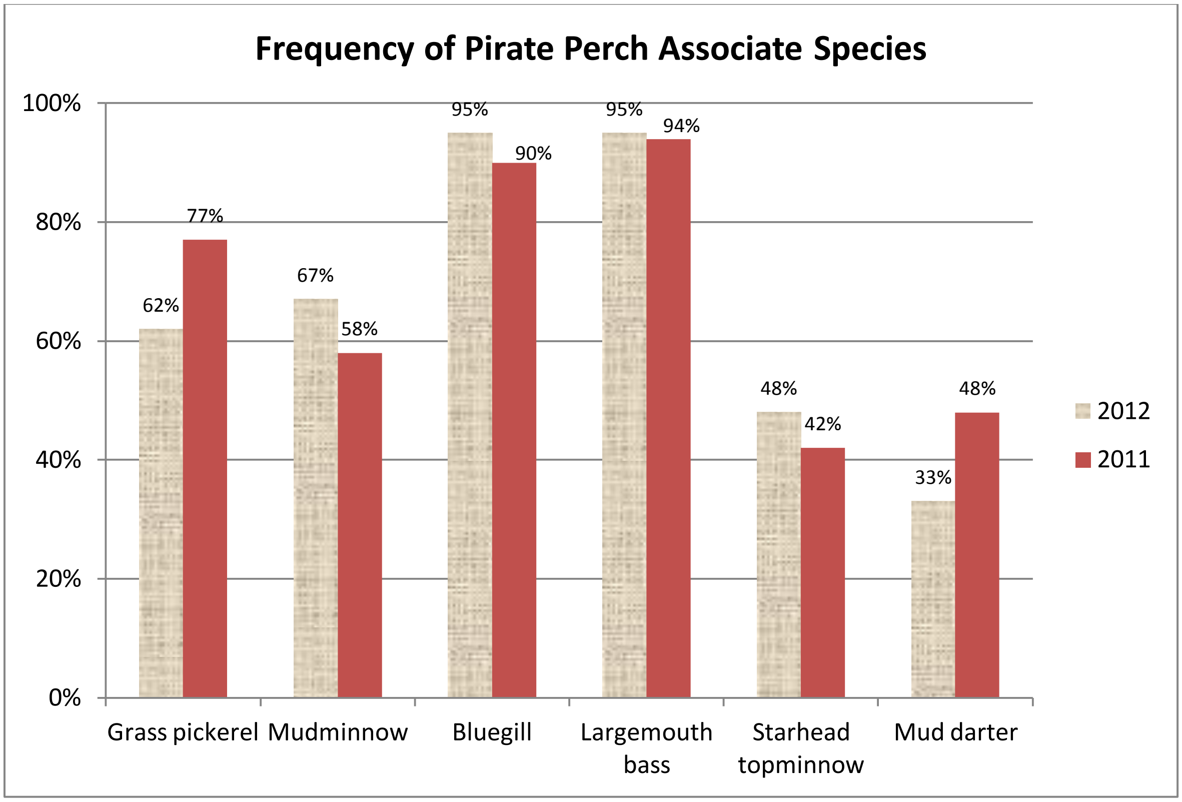
	Mean and S. D. Wood	Mean and S. D. Plants	Mean & S. D. Rock
Mud darters	2.11 - 0.78	1.41 - 0.96	0.91 - 0.90
Pirate perch	1.64 - 0.92	1.82 - 0.99	0.70 - 1.04
Starhead topminnows	1.50 - 0.93	2.38 - 0.77	0.52 - 0.85
Lake chubsuckers	1.53 - 0.88	2.31 - 0.79	0.53 - 0.62

Figure 3: Mud darter species associations



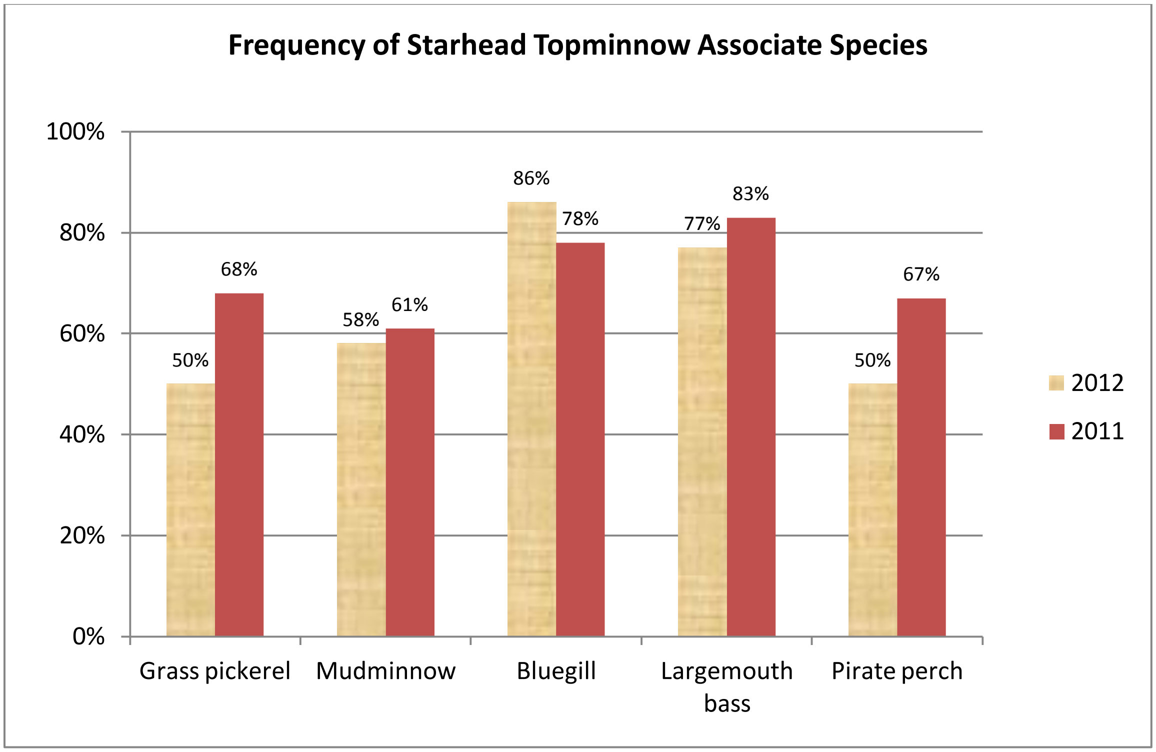
Top 14	2011	2012
1	Bluegill	Bluegill
2	Largemouth bass	Largemouth bass
3	American grass pickerel	Central mudminnow
4	**Pirate perch**	American grass pickerel
5	Central mudminnow	Green sunfish
6	Warmouth sunfish	**Pirate perch**
7	**Starhead topminnow**	**Starhead topminnow**
8	Green sunfish	Pumpkinseed sunfish
9	Johnny darter	Yellow bullhead
10	Yellow bullhead	Brook silverside
11	Tadpole madtom	Tadpole madtom
12	Pumpkinseed sunfish	Johnny darter
13	**Lake chubsucker**	Golden shiner
14	Black crappie	Black crappie

Figure 4: Common pirate perch species associations



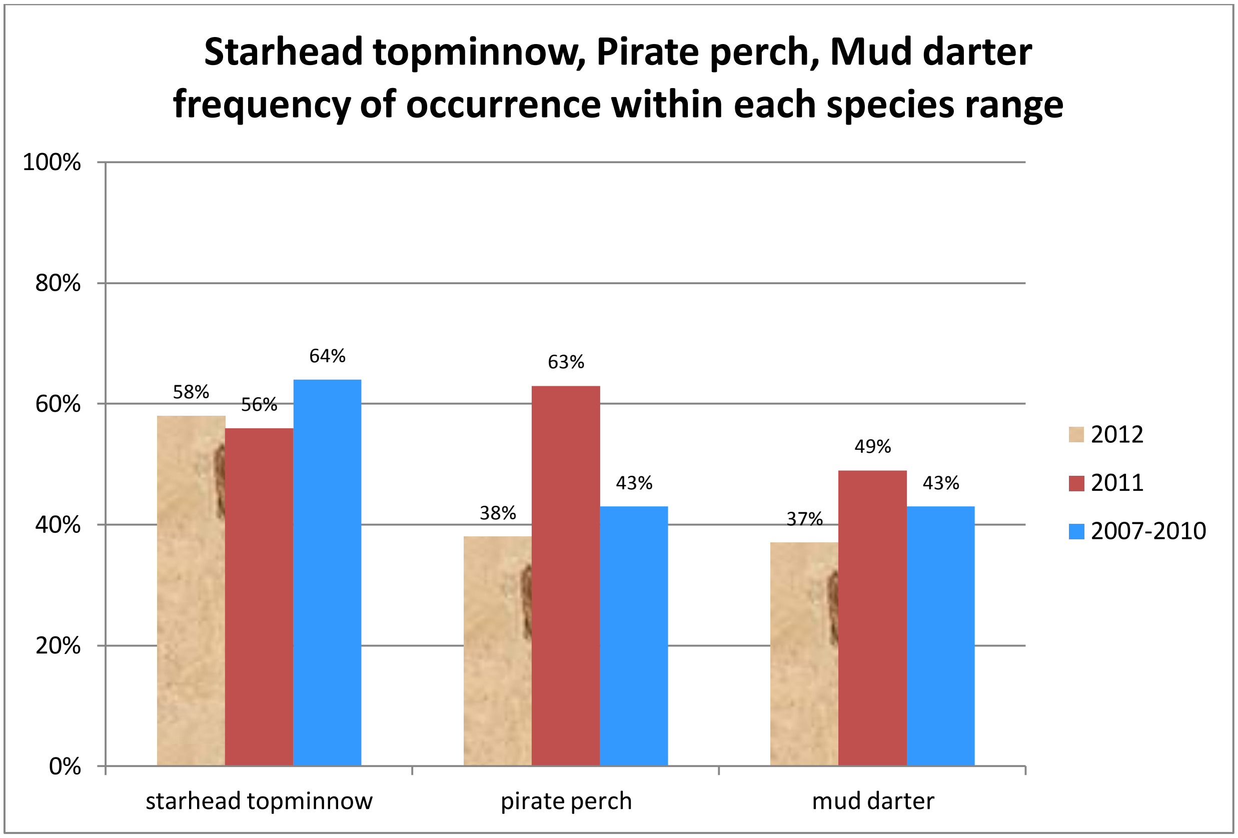
Top 16	2011	2012
1	Largemouth bass	Bluegill
2	Bluegill	Largemouth bass
3	American grass pickerel	Central mudminnow
4	Central mudminnow	American grass pickerel
5	**Mud darter**	**Starhead topminnow**
6	**Starhead topminnow**	Yellow bullhead
7	Warmouth sunfish	**Mud darter**
8	Yellow bullhead	Tadpole madtom
9	Green sunfish	Warmouth sunfish
10	Tadpole madtom	Green sunfish
11	Johnny darter	Pumpkinseed sunfish
12	Pumpkinseed sunfish	Brook silverside
13	**Lake chubsucker**	**Lake chubsucker**
14	Brook silverside	Golden shiner
15	Golden shiner	Black crappie
16	Black crappie	Johnny darter

Figure 6: Starhead topminnow associations



Top 14	2011	2012
1	Largemouth bass	Bluegill
2	Bluegill	Largemouth bass
3	American grass pickerel	Central mudminnow
4	**Pirate perch**	American grass pickerel
5	Central mudminnow	**Pirate perch**
6	**Mud darter**	**Mud darter**
7	Warmouth sunfish	Warmouth sunfish
8	Yellow bullhead	Green sunfish
9	**Lake chubsucker**	Yellow bullhead
10	Johnny darter	**Lake chubsucker**
11	Pumpkinseed sunfish	Pumpkinseed sunfish
12	**Mississippi grass shrimp**	Brook silverside
13	Green sunfish	Golden shiner
14	Tadpole madtom	Black crappie

Figure 7: Frequency of occurrence values for mud darters, pirate perch and starhead topminnows



Dissolved oxygen levels at sites sampled in 2011 and 2012 were similar with mean concentrations of 9.1 mg/l (S. D. = 2.76) and 8.4 mg/l (S. D. = 2.55) respectively. In 2011, six percent of the sites had dissolved oxygen levels less than 5 mg/l or minimum water quality criterion for diverse warmwater fish and aquatic life waterbodies. In 2012, eight percent of the sites had dissolved oxygen levels less than 5 mg/l. The lowest measurements in 2011 and 2012 were 3.5 mg/l and 1 mg/l respectively. Both of these measurements were recorded from Fishers Lake in Dane County. Rare fish were not found in this floodplain that does not intercept upland groundwater, an important feature characteristic of high quality oxbows along the Lower Wisconsin State Riverway and elsewhere. The fisheries in this lake were dominated by low dissolved oxygen tolerant central mudminnows along with low numbers of American grass pickerel, green sunfish and a single pumpkinseed sunfish. A very low measurement (2.8 mg/l) was also taken in 2012 from Clear Lake in Crawford County but this lake sustains higher diversity including pirate perch and a single mud darter specimen collected in 2009. In years prior to 2012, Clear Lake displayed characteristics of a high quality cutoff channel with that intercepted upland groundwater.

Water clarity measurements were similar in 2011 and 2012 with mean values of 95 cm (S. D. = 32.99) and 91 cm (S. D. = 33.93) respectively. Mud darters were collected from Cyanobacteria dominated Silver Birch Lake (Pepin County) three years in a row. Transparency tube measurements were 15 cm in 2010, 10 cm in 2011 and 23 cm in 2012. Starhead topminnows were typically collected from clear water habitats where rooted macrophytes suppress planktonic algae. Mean water clarity measurement for starhead topminnow habitats (113 cm, S. D. = 19). in 2011-12 was greater than the overall mean value.

Lake chubsucker collections were generally low along the Lower Wisconsin River but were consistent with six sites in 2011 and seven sites in 2012. Weed shiner and pugnose minnow collections were even more sporadic. Weed shiners were found at three sites in 2011 and only one site in 2012. Weed shiner sites included the Lower Wisconsin River and Black River. Pugnose minnows were found at four sites in 2011 and just one site in 2012. Pugnose minnow sites included the Wisconsin River and Silver Birch Lake along the Chippewa River. Mississippi grass shrimp were incidentally collected along the Lower Wisconsin River, one site in 2011 and one site in 2012.

SWG11-Cat2-007 Project

Survey participants: Dave Marshall – UHI LLC, Patty Avery, Dave Grey, Sue Graham – WDNR, Patricia Cicero – Rock River Coalition, Jerry Ziegler – TNC, Sarah Mayer – TNC, Jeanne Sherer, Suzanne Wade – Rock River Coalition, Laura Stremick-Thompson-WDNR, Gary Johncox, Ron Grasshoff, Scott Provost – WDNR, Matthew Jacobson – WDNR, Samuel Betterley – WDNR, Jennifer Bergman – WDNR, Lance Potter – WDNR administration, Tara Bergeson – WDNR administration and John Lyons – WDNR planning.

Methods

This project was designed to expand upon recent floodplain habitat surveys (SWG 09) to sample fish populations with information needs and SCGN within the floodplain habitats of the Mukwonago River, Bark River, Chippewa River portions not sampled in 2009-10, Sugar River between Albany and Brodhead, Des Plains River, Illinois Fox River and Yellow River. Fish were primarily collected with a towed DC electroshocker but also with small mes dip nets and small mesh seines. The combination of fish sampling methods allowed us to more effectively sample the different floodplain fish habitats that can escape a single sampling technique. Sites were identified using GPS and electroshocking sampling efforts were documented (single probe electroshocking minutes). Netting was the preferred sampling technique where specific conductance levels exceeded 1,000 uS/cm or were less than 100 uS/cm. Habitat information was gathered by completing by estimating fish cover in the forms of aquatic plants, rock and woody debris/tree roots. Habitat characteristics were assigned numerical values ranging from 3 (high), 2 (medium), 1 (low) to 0 (absent). Basic water quality was assessed by measuring dissolved oxygen/temperature (YSI Model 52), temperature/specific conductance (Extech) and water transparency (120 cm transparency tube). River flow rates were recorded from the USGS National Water Information System website.

Findings

Table 1 presents the complete list of species found in sites sampled in south central and south east Wisconsin (Sugar River, Bark River, Scuppernong Creek, Mukwonago River, Illinois Fox River, Des Plaines River and Kilbourn Road Ditch). The table also lists fish species collected at pirate perch, lake chubsucker and starhead topminnow sites.

The Kilbourn Road Ditch was sampled on June 1 and July 12, 2012. The stream displays relatively poor habitat due to channel modifications and nutrient loading linked to urbanization and agriculture. Specific conductance levels in 2012 surveys ranged from 920 uS/cm to 954 uS/cm. In spite of these relatively poor conditions, pirate perch were collected from two of two sites sampled. They were found in a combination of habitats including woody debris, submersed aquatic vegetation and riprap. The Des Plaines River was also sampled at three locations, including one site that was sampled on two different dates. Pirate perch were not found in the Des Plaines River where high specific conductance levels (1360 uS/cm) suggested greater environmental degradation. Water clarity was very low at 12 cm in July 2012.

Starhead topminnows were found at seven sites along the Illinois Fox River, from approximately one mile above the confluence with the Mukwonago River to about three miles below Waterford dam. The rare fish were found despite poor water quality with specific conductance level reaching 1356 uS/cm and transparency as low at 25 cm. Dissolved oxygen levels ranged from 7.7 mg/l to 19.4 mg/l within the Waterford impoundment. Starheads topminnows were found in nearshore back eddies along the Illinois Fox River, Waterford impoundment bays, an unnamed ditch and within Tichigan Lake. Starheads were often found with higher numbers of blackstripe topminnows, *Fundulus notatus*. A Waterford volunteer (Gary Johncox) also noted starhead topminnows at numerous “weedy” areas within the Waterford impoundment. Off-channel habitats including oxbows and ponds along the Illinois Fox River were dry or nearly dry in 2012.

Lulu Lake was sampled on two different dates. Starhead topminnows were not found in the lake even though they occur elsewhere in the Mukwonago River watershed. Lulu Lake displays excellent water quality and habitat that appear to be favorable conditions for starhead topminnows. The collection of two other macrophyte obligate species in Lulu Lake, environmentally intolerant blacknose shiners (*Notropis heterolepis*) and State Special Concern lake chubsucker, also suggested favorable conditions for starhead topminnows.

Table 1: Fish Species Collected at South Central and South East Wisconsin Sites in 2011-12

Common Name	Waterbody	Pirate perch (2 sites)	L. chubsucker (1 site)	Starhead t. m. (7 sites)
Bowfin	D. P. R.			
Central mudminnow	S. R., D. P. R., K. D., I. F.R.	X		X
Grass pickerel	S. R.			
Northern pike	S. R.			
Gizzard shad	D. P. R.			
Common carp	B. R., S. R.			
Golden shiner	B. R., S. R., D. P. R.			
Spotfin shiner	S. R., B. R.,			
Bluntnose minnow	S. R., Muk. R., D. P. R., B. R.		X	
Fathead minnow	S. R.			
Sand shiner	S. R.			
Emerald shiner	B. R.			
Blacknose shiner	Muk. R.		X	
White sucker	K. D.	X		
L. chubsucker	Muk. R.		----	
Yellow bullhead	S. R., B. R., I. F. R., Muk. R.		X	X
Black bullhead	K. D., S. R., B.R.	X		
Tadpole madtom	S. R., K. D., D. P. R.	X		
Pirate perch	K. D.	-----		
Starhead topminnow	I. F.R.			----
Blackstripe topminnow	B. R., I. F. R., D. P. R., K. D.	X	X	X
Brook silverside	Muk. R.		X	
Brook stickleback	B. R.			
White bass	B. R.			
Bluegill	S. R., B.R., I. F. R. Muk.R., D. P.R.		X	X
Green sunfish	S.R., B.R., l. F.R., D. P.R., K. D.	X		X
Pumpkinseed sunfish	B.R., D. P.R., K.D.	X		
Orangespotted sunfish	B.R., S.R., l. F.R., D. P.R.			X
Warmouth sunfish	Muk. R.		X	
Black crappie	B.R., S.R.,			
Rock bass	Muk. R., S.R.		X	
Largemouth bass	S.R.,B.R.,Muk. R.,D.P.R.		X	
Smallmouth bass	S.R.			
Johnny darter	S.R., B.R., Muk.R., K.D.	X	X	
Fantail darter	Muk. R.			
Banded darter	Muk.R.			
Iowa darter	Muk.R.		X	
Blackside darter	S.R.,D.P.R.,			
Yellow perch	Muk.R.		X	

B.R. = Bark River, D.P.R. = Des Plaines River, I.F.R. = Illinois Fox River, K.D. = Kilbourn Road Ditch, Muk.R. = Mukwonago River, S.R. = Sugar River. X's indicate species collected along with pirate perch, lake chubsucker and starhead topminnow.

The middle reach of the Sugar River (Lower Rock River Drainage Basin) was sampled above the Brodhead dam in 2011 and 2012. Results were consistent with the SWG09 project indicating that starhead topminnows and a few associate off-channel species were not found above the Brodhead dam. Starhead topminnows were also not found at any Sugar River location in 2012, including sites where high numbers occurred in 2010.

Table 2 lists species sampled from tributaries in the Central Wisconsin River Basin upstream of the Lemonweir River. Pirate perch were found in the Little Yellow River and Little Roche a Cri Creek where habitats and water quality appeared to be favorable. Water clarity is was very good in both streams with estimated tree roots/woody debris habitat values of 2 (medium). Pirate perch were not found at four sites that were sampled along the Yellow River and into Castle Rock Flowage. Low water levels were evident at these sites when tree roots/woody debris habitats were stranded above the waterline. Water was typically clear (120 cm) in the Central Wisconsin Basin rivers and oxbows, with the exception of Castle Rock Lake where the water was clouded (45 cm) by a Cyanobacteria bloom.

Table 2: Fish Species Collected at Central Wisconsin River Basin Sites upstream of the Lemonweir River in 2012

Common Name	Waterbody	Pirate perch sites
Unidentified brook lamprey	L. Yellow R.	X
Central mudminnow	Yellow River	
Bluntnose minnow	Yellow River	
Sand shiner	Yellow River	
Yellow bullhead	L. Roche a Cri Cr., Yellow R.	X
Tadpole madtom	All	X
Pirate perch	L. Yellow R., L. Roche a Cri Cr.	----
Bluegill	All	X
Pumpkinseed sunfish	All	X
Green sunfish	Yellow River	
Rock bass	Yellow River	
Black crappie	Yellow R., L. Roche a Cri Creek	X
Largemouth bass	All	X
Smallmouth bass	L. Yellow R.	X
Johnny darter	Yellow R.	
Fantail darter	L. Yellow R.	X
Rainbow darter	L. Roche a Cri Creek	X
Banded darter	L. Yellow R.	X
Blackside darter	Yellow R.	
Yellow perch	Yellow R.	

Species Discussion

(Combined SWG11-MON-004, SWG11-Cat2-007 and SWG09 data)

Pirate perch – *Aphredoderus sayanus*: Based on State Wildlife Grant floodplain fish surveys conducted from 2009-12, pirate perch were collected in the Black River, Mississippi River, Wisconsin River, Lemonweir River, Little Yellow River, Little Roche a Cri Creek and Kilbourne Road Ditch. Previously, pirate perch were also collected in Narrows Creek (Sauk County) and Big Roche a Cri Creek (Adams County) (Lyons et al. 2000). We did not find pirate perch in Lake Wisconsin or other Wisconsin River impoundments.

Pirate perch were collected in oxbow lakes, beaver ponds, sloughs, main channel eddies, and low gradient creeks. The secretive fish displayed an affinity for tree roots/woody debris but was also somewhat of a habitat generalist since it was also collected within aquatic plants, natural rock and riprap. Finding pirate perch within tree roots/woody debris and aquatic plants is consistent with published pirate perch accounts (Becker 1983, Fletcher et al. 2004). Fletcher et al. (2004) revealed the importance of roots since pirate perch release eggs via the urogenital pore into sheltered canals of the root masses. Aquatic plant roots are also occasionally used for nesting habitat. In southern Illinois, Poly and Wetzel (2003) found pirate perch under rocks in clear streams.

Figure 4 lists the top fish species associations of pirate perch in 2011 and 2012. These patterns were very similar both years and were also similar to SWG09 reported pirate perch associations. Bluegills, largemouth bass, central mudminnows, and American grass pickerel were the most common pirate perch associates followed closed by starhead topminnows and mud darters; the latter often occupying the same habitat. Becker (1983) listed these fish species associated with pirate perch: central mudminnow, black bullhead, tadpole madtom, American grass pickerel, northern pike, largemouth bass, bluegill, black crappie, golden shiner, emerald shiner, pugnose minnow, and fathead minnow. Where associate species distributions do not overlap with pirate perch, the assemblage composition varied somewhat. For instance, starhead topminnows and mud darters do not occur in Kilbourne Road Ditch nor were grass pickerel collected there.

Pirate perch were found at fewer numbers of sites in 2012 during an extreme drought. This finding may have reflected reduced access to tree roots/woody debris that was often stranded above the waterlines. Estimated mean tree roots/woody debris habitats declined from 2.1 in 2011 to 1.6 in 2012. In an effort to avoid exacerbating impacts of predicted climate related droughts on pirate perch habitat, surface and groundwater extractions should be avoided along rivers that sustain pirate perch populations.

Wadeable electroshocking is the preferred sampling gear for collecting pirate perch within tree roots/woody debris and occasionally rock. Late summer sampling may be desirable when seasonal populations increase after spawning and recruitment. Surveys conducted during floods can be ineffective.

Mud darter – *Etheostoma asprigene*: Mud darters were collected Glenn Lake and Garnet Lake along the Mississippi River, the Wisconsin River upstream to the Portage area, the Black River below Black River Falls, and in Silver Birch Lake along the Lower Chippewa River as part of the SWG09 and SWG11 projects. Mud darter is mentioned as a priority species in the Lower Chippewa River Properties Master Plan (WDNR 2009). It was also collected along the Lower St. Croix River in 1998 (Lyons et al. 2000). Katula (2012) described mud darters as “sporadic to common” along the Mississippi River.

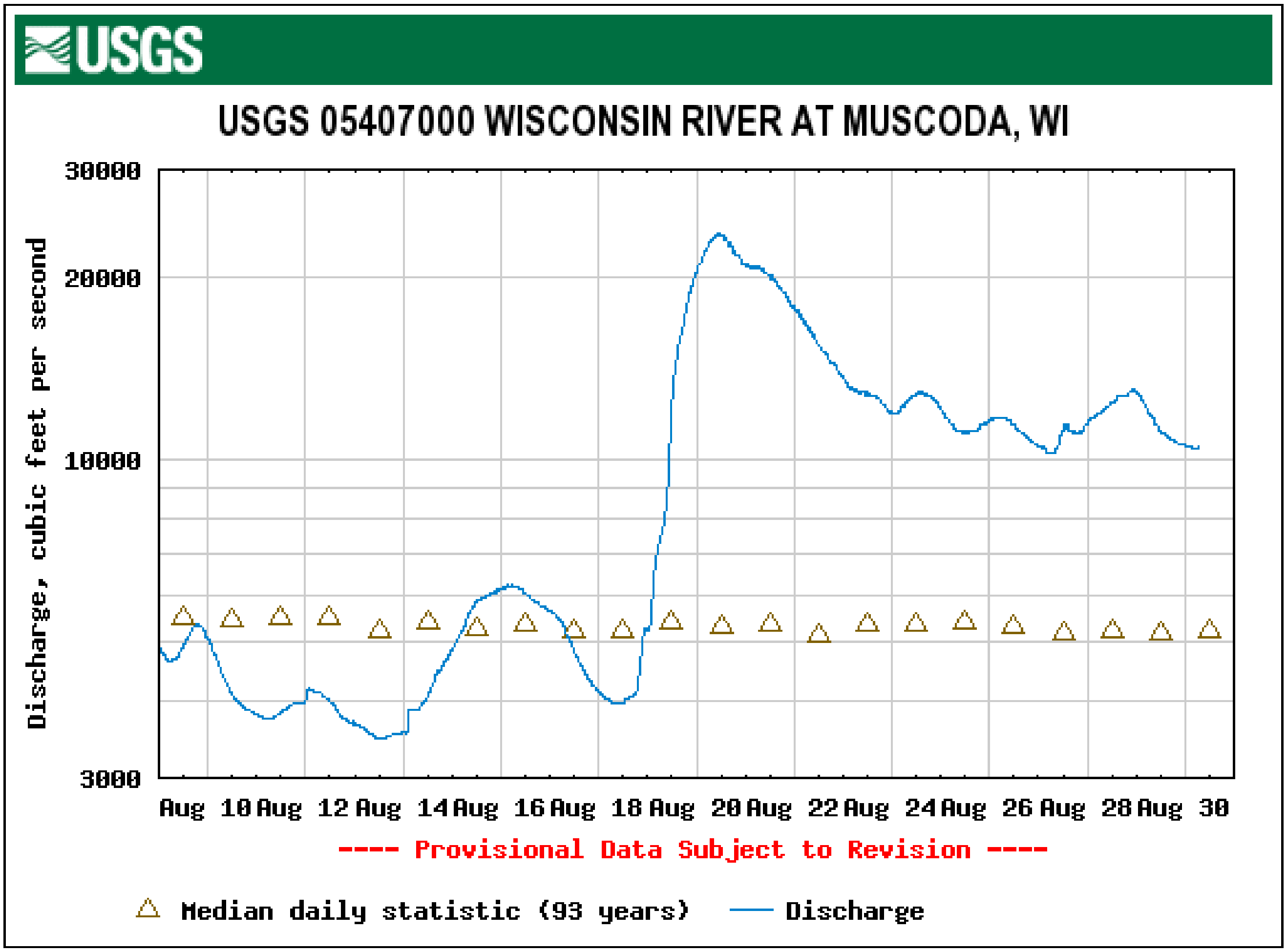
Mud darters were collected in oxbows, sloughs, main channel eddies, creek bottoms and in Lake Wisconsin. They were primarily found in tree roots/woody debris and less commonly in rock (Figure 2), and at times sharing the same habitat with pirate perch. In Lake Wisconsin, Mud darters were collected primarily along rocky wave swept shores. Mud darters were not collected within aquatic plant beds even though these habitats were often in close proximity to tree roots/woody debris. Cummings et al. (1984) reported mud darter spawning on elevated aquarium substrates while Pflieger (1997) reported mud darters spawning in “dense mats of finely divided tree roots”. Katula (2012) observed mud darters spawn on yarn filaments in an aquarium and mentioned that plant roots are likely the natural spawning habitat. Mud darter and pirate perch spawning habitats appear to be very similar and may explain why they are often found in the same site.

Figure 3 lists the common mud darter associate species collected in 2011 and 2012. Consistent assemblage patterns were found in both years and these are also similar to SWG09 findings. Not surprising given their shared habitats and overlapping ranges, the mud darters and pirate perch share a common fish assemblage.

Droughts pose similar threats to mud darters and pirate perch given their common habitats. Estimated tree roots/woody debris habitats at mud darter sites declined from 2.3 in 2011 to 1.7 during the 2012 drought. A reduction in upland groundwater poses additional threats to mud darters given their benthic niche. In 2007, very low dissolved oxygen levels were detected in Lower Wisconsin River oxbows when the river stage approached the 2-year storm event. As river flow increased in Figure 8, dissolved oxygen levels dropped from just 3 mg/l in the top few centimeters to anoxia less than a meter below the surface in several oxbow lakes (Marshall 2009). These conditions pose problems for mud darters. They will not likely survive these conditions without significant upland groundwater flow required to provide adequate dissolved oxygen levels. Drought conditions and high capacity wells could potentially reduce upland groundwater flow to oxbows and mud darter habitats.

Nearshore towed electroshocking is the most effective sampling technique to extract secretive mud darters from tree roots/woody debris and coarse rock. Consistent with pirate perch, late season sampling during near mean river flows is likely the best time to locate mud darter populations. Cummings et al. (1984) reported late season mud darter population increase in Lake Creek in Illinois due to young of year recruitment.

Figure 8: Late summer flooding (~ 2 year storm event) resulted in elevated alluvial groundwater and low dissolved oxygen levels in numerous Lower Wisconsin River oxbows (Marshall 2009)



Weed shiner – *Notropis texanus* and pugnose minnow – *Opsopoeodus emiliae*: These two minnow family species were collected far less frequently than starhead topminnows, mud darters or pirate perch. It is uncertain if the sporadic collections reflected very low population numbers, lack of habitat specificity, or perhaps that the nearshore sampling efforts that may have been ineffective. Regardless, additional surveys are needed to assess these species and should include off-shore slough sampling. Becker (1983) mentioned that adult pugnose minnows periodically inhabit deeper waters that nearshore sampling would not reach.

From 2009-12 weed shiners were sporadically collected in Mississippi River and Lower Wisconsin River sloughs. A single weed shiner was also collected within a Black River eddy. Weed shiners were collected at just four sites as part of SWG11 and at just three sites as part of SWG09. Previously, weed shiners had been collected in Garner Lake along the Lower Wisconsin River in 2004 (WDNR data) and in 2008 (Marshall 2009). Small numbers had also been collected from Chippewa River tributaries and from tributaries of the Wolf River and Fox River in the Lake Michigan Drainage Basin (Lyons et al. 2000).

Both woody debris and aquatic vegetation habitats were variable at weed shiner collection sites but they were collected along with other target off-channel species. Becker (1983) indicated that the weed shiner is not always associated with weedy habitats but is an inhabitant of sloughs, oxbow lakes and sluggish sections of medium and large rivers. Pflieger (1997) reported that weed shiners are most common in clear low-gradient streams with substrates of sand or gravel. Threats to weed shiners include droughts and floodplain aggradation that can reduce or eliminate off-channel habitats.

Pugnose minnows were collected at just four sites as part of the SWG11 project including Silver Birch Lake, the mouth of the Lemonweir River and at a few sites above the Prairie du Sac dam near Lake Wisconsin. They were found at just one site as part of SGW09 project in the upper Wisconsin Dells area. Lyons et al. (2000) reported that pugnose minnows are generally uncommon in large impoundments but can be locally common within some Mississippi River backwaters. In 2004, they were also collected along the Lower Wisconsin River in Avoca Lake and Garner Lake (WDNR data).

Aquatic plant and woody debris habitats were variable at pugnose minnow sites but coincided with mud darter and pirate perch collections at a few sites. Elsewhere, pugnose minnow habitats had been described as low gradient streams, oxbows, sloughs, borrow pits, etc., with dense aquatic vegetation (Pflieger 1997).

Starhead topminnow – *Fundulus dispar*: As part of the SWG11-MON project, starhead topminnows were collected at 26 sites along the Lower Wisconsin River between the Prairie du Sac dam and the confluence with the Mississippi River, along the Black River below Black River Falls, and the Lower Sugar River. Only one specimen was found along the Lower Sugar River in 2011 and none were found during the 2012 drought. In 2010, when Sugar River reached flood stage, hundreds of starhead topminnows were collected in flooded ponds and sloughs that became completely dry during the 2012 drought. Low numbers of starheads were collected at seven sites along the Illinois Fox River in 2012 as part of the SWG11-Cat2 project. The Illinois Fox River watershed collections extended about a mile above the confluence with the Mukwonago River to about three miles below the Waterford dam, within Tichigan Lake and in an unnamed ditch located above Waterford. Starheads were not found in the Lulu Lake State Wildlife Area even though the endangered fish occurs elsewhere in the watershed.

We typically found starhead topminnows in oxbows, beaver ponds, delta ponds, sloughs and sluggish streams in areas with abundant aquatic plants (Figure 2). The macrophyte dominated floodplain habitats where we found starheads as part of SWG11 were also consistent with the SWG09 survey results, previous Lower Wisconsin River floodplain surveys, and published accounts suggesting that they are macrophyte obligates (Becker 1983). In lakes and impoundments where starhead topminnows were found, such as Tichigan Lake and the Waterford impoundment, aquatic plant eradication efforts could reduce their habitats and threaten their survival.

The Lower Wisconsin State Riverway may hold the largest starhead topminnow population in state within a vast floodplain containing numerous cut-off channel oxbow lakes, sloughs, delta ponds, beaver ponds and creek bottoms. A massive Driftless Area groundwater flow system discharges into the Lower Wisconsin River floodplain (Pfeiffer et al. 2006). The massive aquifer can sustain starhead topminnows and the off-channel fish assemblage during both droughts and other environmentally stressful periods. The key threat to starhead topminnows in this relatively natural floodplain includes high capacity well water extraction and groundwater contamination. Contaminated groundwater, including high nitrates, along the State Riverway is a documented threat to human health and oxbow water quality. Nutrient enrichment of oxbow lakes in the Spring Green area resulted in excessive metaphyton cover that reduced habitat and dissolved oxygen to very low concentrations. Nitrate toxicity can also poses a direct threat to aquatic organisms (Camargo et al. 2005).

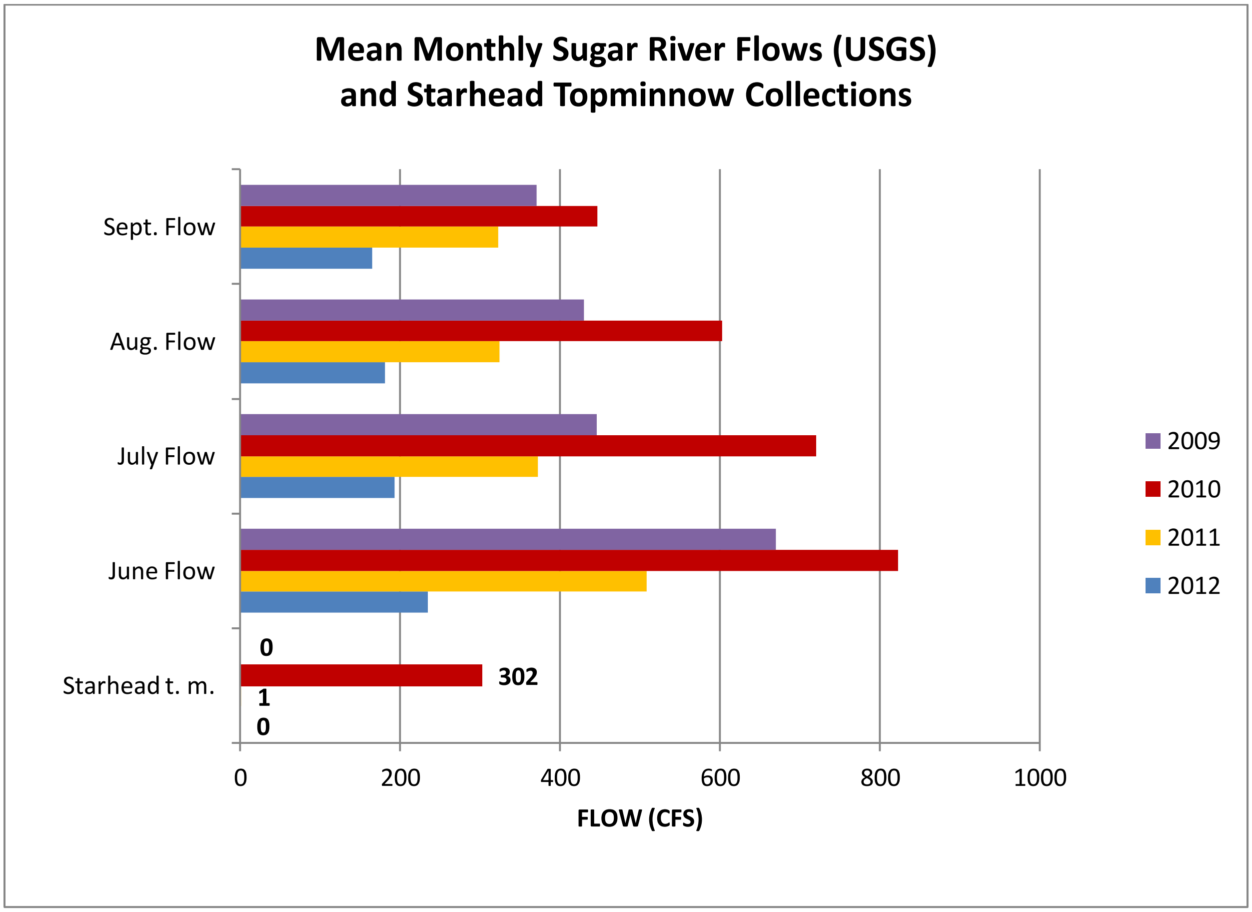
Elsewhere across their range, starhead topminnows are threatened by severe droughts. During the 2012 drought, the surface area and depth of Stevens Lake was declined significantly and habitat was greatly diminished. Starhead topminnows, pirate perch and other off-channel fish populations in the lake declined as well. Drought impacts were primarily manifested along meandered single-thread rivers where oxbows and sloughs lost storage capacity or were completely eliminated due to floodplain aggradation. Starhead topminnow habitats along the Sugar River were completely dry in 2012.

Extreme droughts and intense precipitation events can be expected across Wisconsin due to Climate Change (WICCI 2011). While droughts can significantly reduce habitats for starhead topminnows and associate off-channel species, flooding can temporarily increase starhead topminnow recruitment. Figure 9 displays the high number of starhead topminnows found along the Sugar River when the river exceeded flood stage in 2010. In 2007 we observed numerous starheads swimming in flooded reed canary grass beds near Hill Slough (Marshall 2009). However, long term impacts of flooding include increased vertical floodplain accretion and loss of oxbow and slough storage capacity (Knox 2006). As water levels declined along the Sugar River in 2011, and particularly in 2012, starhead topminnow habitats disappeared along with the starheads.

As mentioned above, the distribution of starhead topminnows is restricted by a number of dams, including the Wisconsin River Prairie du Sac dam. Habitat appears to be favorable above the dam in oxbows and sloughs between Portage and Lake Wisconsin. Expanding the starhead topminnow distribution within river systems may improve their survival and status in Wisconsin. This effort could involve transfers and conservation aquaculture. Johnson (2012) demonstrated that starhead topminnows “propagate explosively” in pond culture without competition. Taylor and Burr (1997) mentioned that starheads exhibit a multiple clutch reproductive strategy. The pond culture demonstration, along with observations of significant late summer recruitment in well vegetated oxbows, and a multiple spawning strategy indicates that conservation aquaculture should be considered for starhead topminnows. Introducing the starhead topminnows above the Prairie du Sac dam, and perhaps other dams, should not pose a problem since *Fundulus dispar* lacks the ecological niche conservatism displayed by other starheaded topminnow clade members and exhibits a far broader distribution (McNyset 2009).

Therefore, starhead topminnow conservation should include a mix of habitat protection (Lower Wisconsin State Riverway), restoration (Sugar River) and conservation aquaculture/transfers.

Figure 9: Sugar River USGS monthly mean flow rates and annual starhead topminnow collections



Lake chubsucker – *Erimyzon sucetta*: Lake chubsuckers were collected at 11 different oxbows/sloughs along the Lower Wisconsin River as part of the SWG11-MON project and at Lulu Lake as part of the SWG11-Cat2 project. As Figure 2 indicates, lake chubsuckers were primarily found in dense aquatic plant beds and are likely a macrophyte obligate species. Becker (1983) described lake chubsucker habitat: *The lake chubsucker characteristically occurs in lakes, oxbow lakes, and sloughs of large rivers with dense vegetation.* In 2009 as part of SWG09, we found lake chubsuckers in a shallow Lower Wisconsin River slough in Crawford County with a dissolved oxygen level of only 2.5 mg/l. Lake chubsuckers are considered tolerant of low dissolved oxygen levels and winterkill conditions (Becker 1983). However, habitat loss and perhaps other environmental stressors reflect their rare status and State Special Concern designation. Aquatic plant refuges are needed in developed lakes where lake chubsuckers occur. Lake chubsuckers appear to be associated with and indicators of other rare off-channel fish including mud darters, pirate perch and starhead topminnows (see Figures 3-6).

Banded killifish – *Fundulus diaphanus*: Banded killifish were only found in Lake Wisconsin during both SWG11 and SWG09 projects. It was found at two Lake Wisconsin locations in 2009 and at one site in 2012. Both both sites had moderate growths of wild celery and were moderately turbid water due to Cyanobacteria blooms. Surprisingly, we did not find banded killifish in Lulu Lake with a well documented Mukwonago River population (Lyons 2011).

Becker (1983) considered the banded killifish as common to abundant in many southeastern Wisconsin lakes and secure status in the state. However, it has since declined and is now designated as State Special Concern. In 2004, surveys indicated that banded killifish declined in southeast Wisconsin lakes where it was historically common (Marshall and Lyons 2008). Loss of banded killifish from the Madison lakes had already been documented (Lyons 1989). Banded killifish may represent another "Fundulidae" opportunity to use conservation aquaculture for reintroduction in areas where it declined or became extirpated. Becker (1983) reported that banded killifish were propagated in a small Michigan pond at the rate of 80,000 per acre. Brian Torreanno (BT Darters) has demonstrated successful banded killifish propagation in aquariums.

Mississippi grass shrimp - *Palaemonetes kadiakensis:* State Special Concern Mississippi grass shrimp were incidental catches when surveys encompassed aquatic plant beds in Mississippi River and Lower Wisconsin River sloughs. Shrimp were found in a Lower Wisconsin River Grant County slough in 2009 and were sampled in very high numbers in Spring Lake and Gremore Lake, along the Mississippi River in April 2010. However, shrimp were not found in Gremore Lake by June of 2010. In 2008, the shrimp were collected from the same Lower Wisconsin River slough in Grant County and also in a Crawford County slough. As part of the SWG11-MON project, shrimp were found at one Lower Wisconsin River slough in 2011 and another slough in 2012. Mississippi grass shrimp were not found upstream of Crawford County along the Lower Wisconsin River.

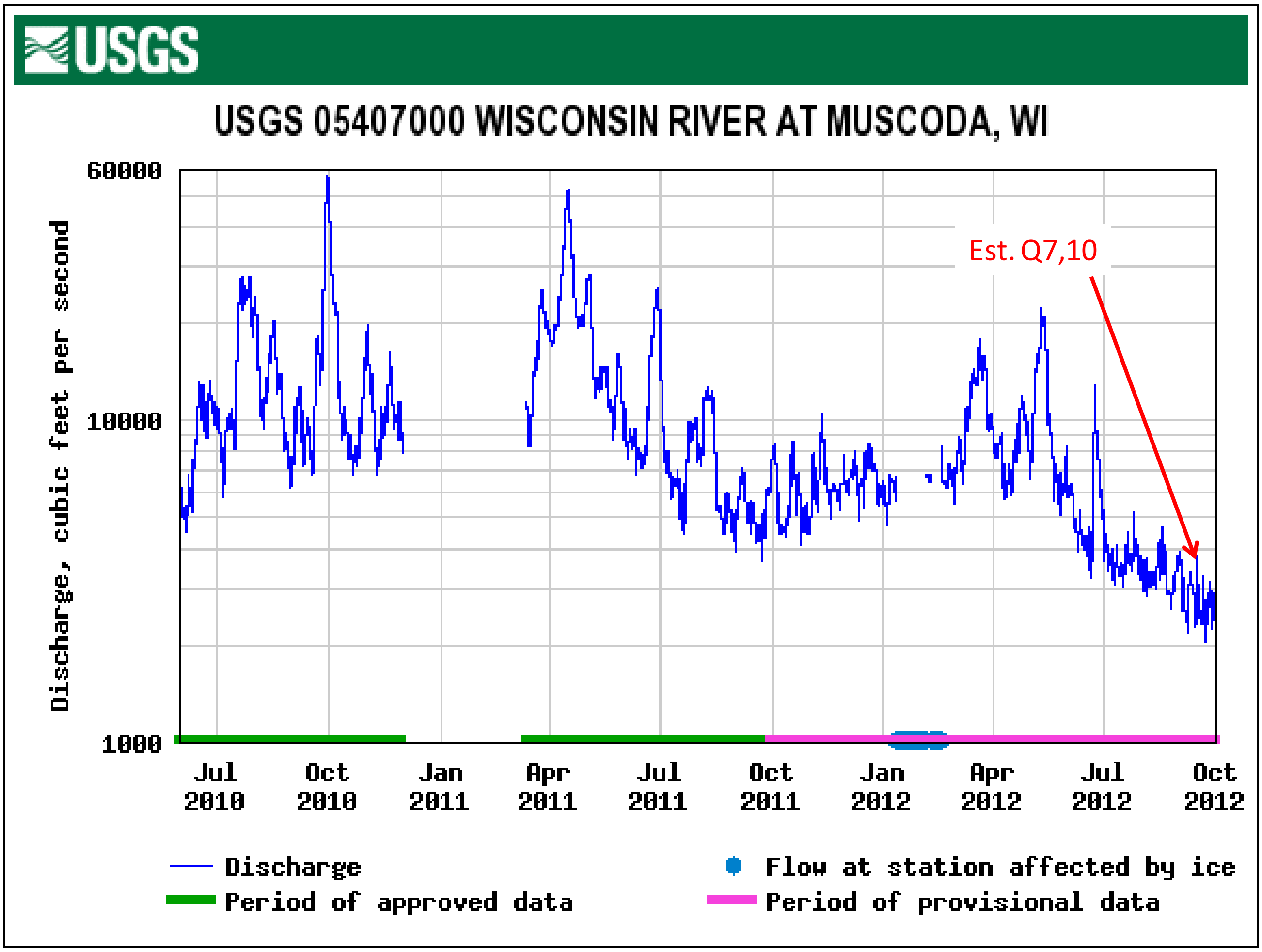
Least darter – *Etheostoma microperca*: Least darters were not found during the SWG11 or SWG09 surveys. However, they are worth mentioning since they were found in three Lower Wisconsin River oxbows in 2004 but not since. As mentioned in the mud darter discussion, darters can be more vulnerable to low dissolved oxygen stress than fish equipped with swim bladders. The decline of least darters across the state roughly coincides with the decline of banded killifish in southern Wisconsin lakes and streams. Least darters would likely become stressed in oxbows with heavy metaphyton cover due to macrophyte habitat decline and low dissolved oxygen. Least darter is another example of a species with a strong affinity to aquatic vegetation.

Rivers Discussion

Wisconsin River (WBIC 1179900: The Wisconsin River was the primary focus of the SWG11-MON-004 project. Below Prairie du Sac, the river is a braided channel floodplain that remains in a relatively natural state. The river supports 99 native fishes that are exemplified in an educational poster that was produced in 2009 in conjunction with the 20 year anniversary of the Lower Wisconsin State Riverway (Figure 10). Native fish species along the State Riverway floodplain would be significantly fewer without the vast network of oxbows, sloughs, delta ponds, beaver ponds and creek bottoms that support species adapted to lentic habitats. The Lower Wisconsin River floodplain sustains the target rare species that thrive in these habitats including starhead topminnows, mud darters, pirate perch, weed shiners, pugnose minnows, lake chubsuckers and Mississippi grass shrimp.

Figure 10: Copy of the Rare and Unusual Fishes of the Lower Wisconsin River educational poster

Figure 11: Wisconsin River at Muscoda USGS Monitored Flows



While the Lower Wisconsin River floodplain provides critical habitats for these species and the entire off-channel fish assemblage, annual and seasonal changes create both environmental stressors and opportunities. Figure 11 displays flow changes that occurred over three years, including the severe drought of 2012. In Figure 12, Amoros and Bornette (2002) illustrate how hydrological changes that occur in floodplains are closely linked to river stages. This illustration appears to be particularly applicable to the Lower Wisconsin River braided channel floodplain ecosystem with numerous cutoff channel oxbows. During high flows, the river consumes the entire floodplain and cutoff channel oxbows become active river channels. Scouring can occur due to the linear cutoff channel morphology. When river flows approach the 2-year storm event, alluvial groundwater moves toward the oxbows, causing water chemistry changes. We measured low dissolved oxygen levels in several Lower Wisconsin River oxbows during a summer storm event (Marshall 2009). Upland groundwater is the primary water source during low flow conditions. The cutoff channel oxbows along the Lower Wisconsin River intercept a massive Driftless Area groundwater flow system (Pfeiffer et al. 2006). The combination of upland groundwater flows and well connected floodplain can sustain oxbow water quality and survival of rare off-channel species during extreme droughts. Mud darters are often found in tree roots/woody debris along steep terrace side banks that intercept upland groundwater.

Figure 12: Illustration of Hydrological Connectivity Dynamics (modified from Amoros and Bornette 2002)

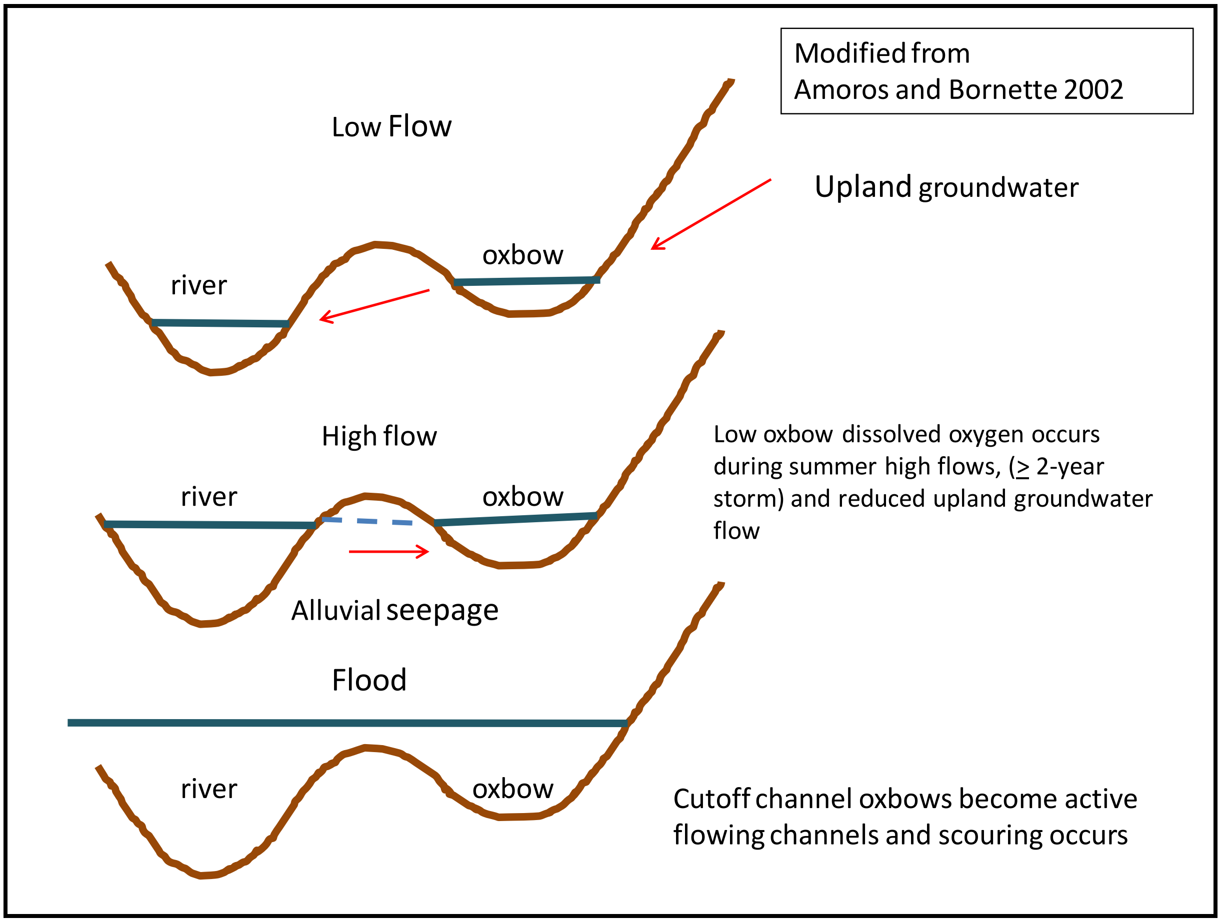


Figure 13 demonstrates the influence of upland groundwater flow on temperature and dissolved oxygen. Norton Slough lies adjacent to the river terrace and intercepts upland groundwater. Wood Slough lies closer to the river and the water quality, including low dissolved oxygen, is influenced by the alluvial groundwater. Wood Slough will not likely sustain mud darters and other benthic fish.

In addition to the cutoff channel oxbows, sloughs, beaver ponds and delta ponds, a number of borrow pits also supported rare species along the Lower Wisconsin River. Starhead topminnows, pirate perch and mud darters were occasionally collected from borrow pits in the Spring Green area. However, the sporadic occurrences of these species likely reflected flood migrations but a general lack of shallow littoral zone habitats limited their numbers and occurrences.

After hundreds of surveys conducted from 2007-12 along the Lower Wisconsin River, juvenile common carp were found at only one location. The dearth of juvenile common carp suggests egg and fry predation by the native off-channel fish assemblage including bluegills, largemouth bass, American grass pickerel, central mudminnows and many other species. Bajer and Sorensen (2010) demonstrated that common carp often reproduce successfully in winterkill habitats lacking predators. The presence of the off-channel fish assemblage in Lower Wisconsin River cutoff channel oxbows, sloughs, etc. suggest sustainable year round habitats that do not provide conditions favorable for common carp recruitment.

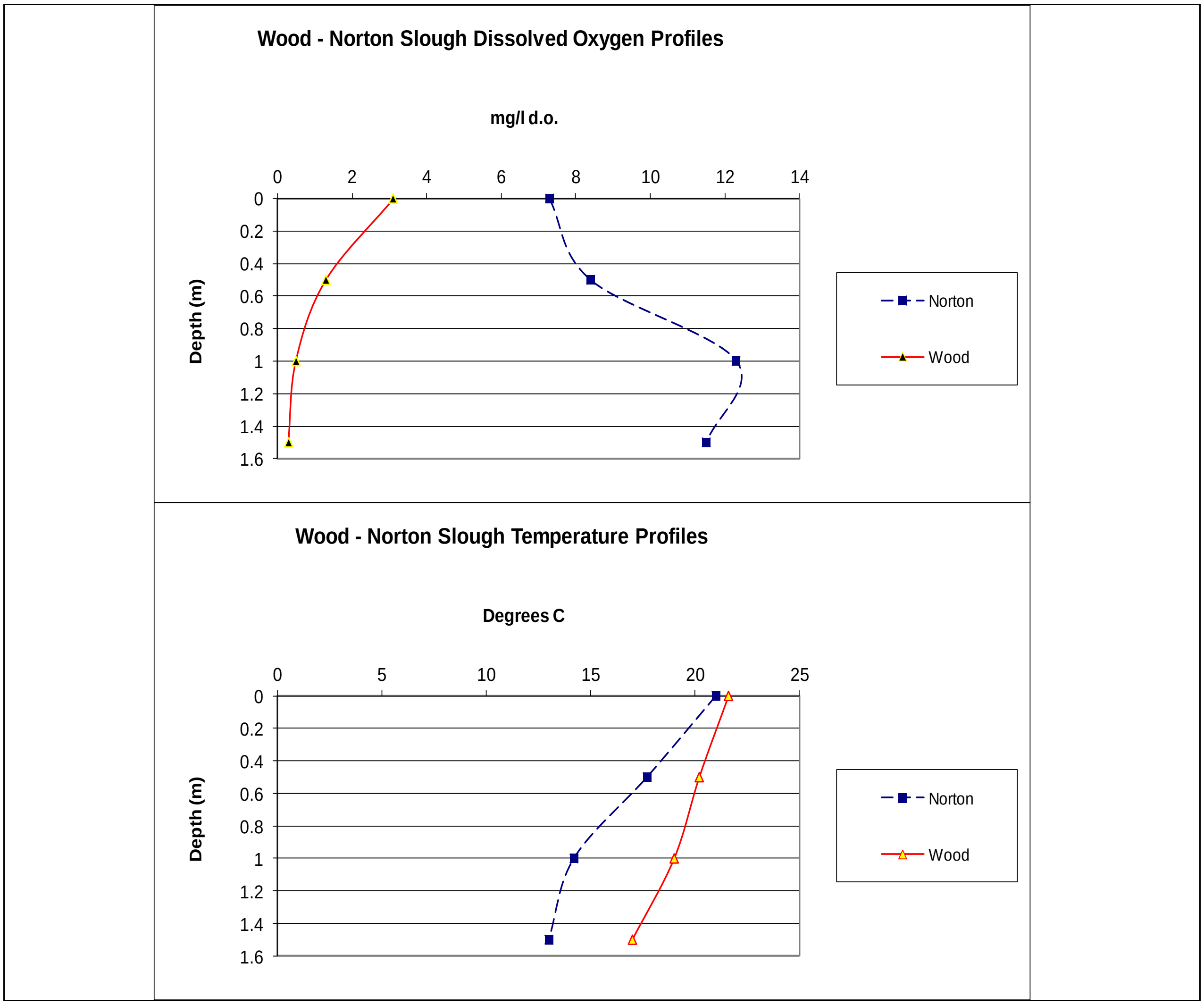
The Lower Wisconsin State Riverway was established in 1989 to protect the scenic views along the river. The prevailing assumption, previously shared by the author, was the river is well protected and the State Riverway functioned as a rare species refuge (Marshall and Lyons 2008). This assumption was largely based on existing high biodiversity in the river system, State Riverway management and extensive public lands. This view was echoed in the Land Legacy Report that mentioned only “limited protection” is needed along the Lower Wisconsin River (WDNR 2006). However, the floodplain had already been identified as highly susceptible to groundwater contamination and many wells currently exceed the health standard for drinking water. Our surveys revealed that the groundwater flow system is causing significant water quality problems in some cutoff channel oxbow lakes in the forms of eutrophication and potentially in the form of nitrate toxicity (Camargo et al. 2005). Therefore, our findings demonstrate that significant threats do exist along the State Riverway and there are moderate to substantial opportunities for additional protection efforts.

Previous surveys had demonstrated the presence of starhead topminnows and pirate perch in Norton Slough. In 2008, the top photo below demonstrates the diverse aquatic plant community including white water lily, large-leaf pondweed, bladderwort, flatstem pondweed, pickerelweed, water stargrass, slender naiad, etc. By 2011, aquatic plant community in the bottom photo largely disappeared, while a dense mat of metaphyton (filamentous algae and duckweeds) smothered the lake. The metaphyton shaded out the plants a significant amount of fish habitat. The dense metaphyton mat also eliminated photosynthesis and atmospheric exposure that resulted in very low dissolved oxygen levels. Starhead topminnows have since declined in the lake as well.

In an effort to better define the rapid water quality decline that occurred in Norton Slough, WDNR Paleolimnologist Paul Garrison, Richard Wedepohl and Dave Marshall collected sediment cores from the lake in March 2012. The sampling demonstrated that scouring had occurred during floods since the bottom was mostly sand with very little organic deposition. This finding revealed that the water quality change was not linked to internal loading but rather reflected recent nutrient loadings. Vertical temperature profiles of the lake revealed the significant upland groundwater flow that dominates the lake hydrology (Figure 13). This information coupled with the trend of increasing nitrates in adjacent wells suggests that contaminated groundwater is the primary nutrient source. Similar eutrophic conditions also occur in nearby Jones Slough and Hutter Slough. Further sampling will be conducted in 2013 to better assess the impacts of contaminated groundwater in the Spring Green area.

Norton Slough in 2008 (top) and in 2011 (bottom)

Figure 13: August temperature and dissolved oxygen profiles in two adjacent Lower Wisconsin State Riverway sloughs; Norton Slough is influenced by upland groundwater flow with colder temperatures and higher dissolved oxygen and Wood Slough that is influenced by alluvial groundwater.



The groundwater contamination issues along the Lower Wisconsin River sand terrace are primarily linked to application of nutrients and pesticides that quickly leach through sandy soils into the shallow groundwater table. The agricultural practices on the terrace are also sustained with center pivot irrigation that results in further leaching. This issue is prominent along Central Wisconsin River Basin streams (Little Roche a Cri Creek, Little Yellow River, etc.) where pirate perch, mud darters, weed shiners and pugnose minnows occur. High capacity well water extractions coupled with heavy nutrient applications on leachable sandy soils pose threats these species by reducing clean groundwater inputs and increasing nutrient loads (WICCI 2011, Kraft and Mechenich 2010, WDNR 2010). Beyond the Wisconsin River floodplain, the same hydrological impacts apply to Central Sands Region seepage lakes and some are inhabited by State Special Concern banded killifish.

Wisconsin River recommendations: Establish buffers around cutoff channel oxbows and tributaries to reduce impacts of contaminated groundwater within coarse sandy soils (Mayer et al. 2005). Regulate high capacity groundwater extractions and other activities that could alter the floodplain hydrology and water quality, particularly during droughts. Restore the annual State Stewardship funding that was specifically authorized for the State Riverway when it was formed in 1989. Stewardship funding is needed for conservation buffer easements or acquisition around environmentally sensitive cutoff channel oxbows and other floodplain habitats.

Sugar River (WBIC 875300): The Sugar River was sampled at three locations as part of the SWG11-MON-004 project in 2011 and 2012 and at three locations as part of the SWG11-Cat2-007 project. The Sugar River is a medium sized single-thread river with deeply incised banks and vertical floodplain accretion. This has resulted in significant loss of oxbows and other floodplain habitats due to sedimentation. River connectivity with the floodplain and groundwater connectivity with oxbows and ponds have been drastically reduced. Oxbows that were not completely lost to sedimentation are now very shallow and perched above the groundwater table. These conditions are not sustainable for the off-channel fish assemblage or rare fish such as starhead topminnows. The Sugar River starhead topminnow population that we discovered in 2010 likely represented a brief reproductive pulse as flooded spawning habitats became available.

Below are photographs of a shallow floodplain wildlife scrape that provided starhead topminnow habitat during the 2010 floods but became desiccated in 2012. Figure 15 displays the Sugar River USGS hydrograph with flood stages recorded in 2010 and extreme low flows in 2012. In 2011 and particularly in 2012, starhead topminnow access to the floodplain was nearly eliminated. Only one adult was collected in 2011 and no starheads were found in 2012. This suggests that the remnant population discovered along the Lower Sugar River in 2010 is again marginalized to extremely low numbers if they are present at all.

Coinciding with the loss of floodplain habitats during the 2012 drought, an invasive mosquitofish (*Gambusia affinis*) population near Brodhead significantly expanded despite the reduced off-channel habitat and reduced the native off-channel fish assemblage. The mosquitofish expansion poses a new threat to the very low numbers of starhead topminnows further downstream. Mosquitofish have been problematic for native fish species across the United States. The River Alliance of Wisconsin sponsored a WDNR Aquatic Invasive Rapid Response project in the fall of 2012 in an effort to reduce the expanding population. Preliminary results of this effort indicated that the attempted mechanical eradication effort was unsuccessful.

Shallow Wildlife Pond within the Avon Bottoms State Wildlife Area and Sugar River Floodplain

Shallow wildlife scrapes are not sustained during droughts

2010

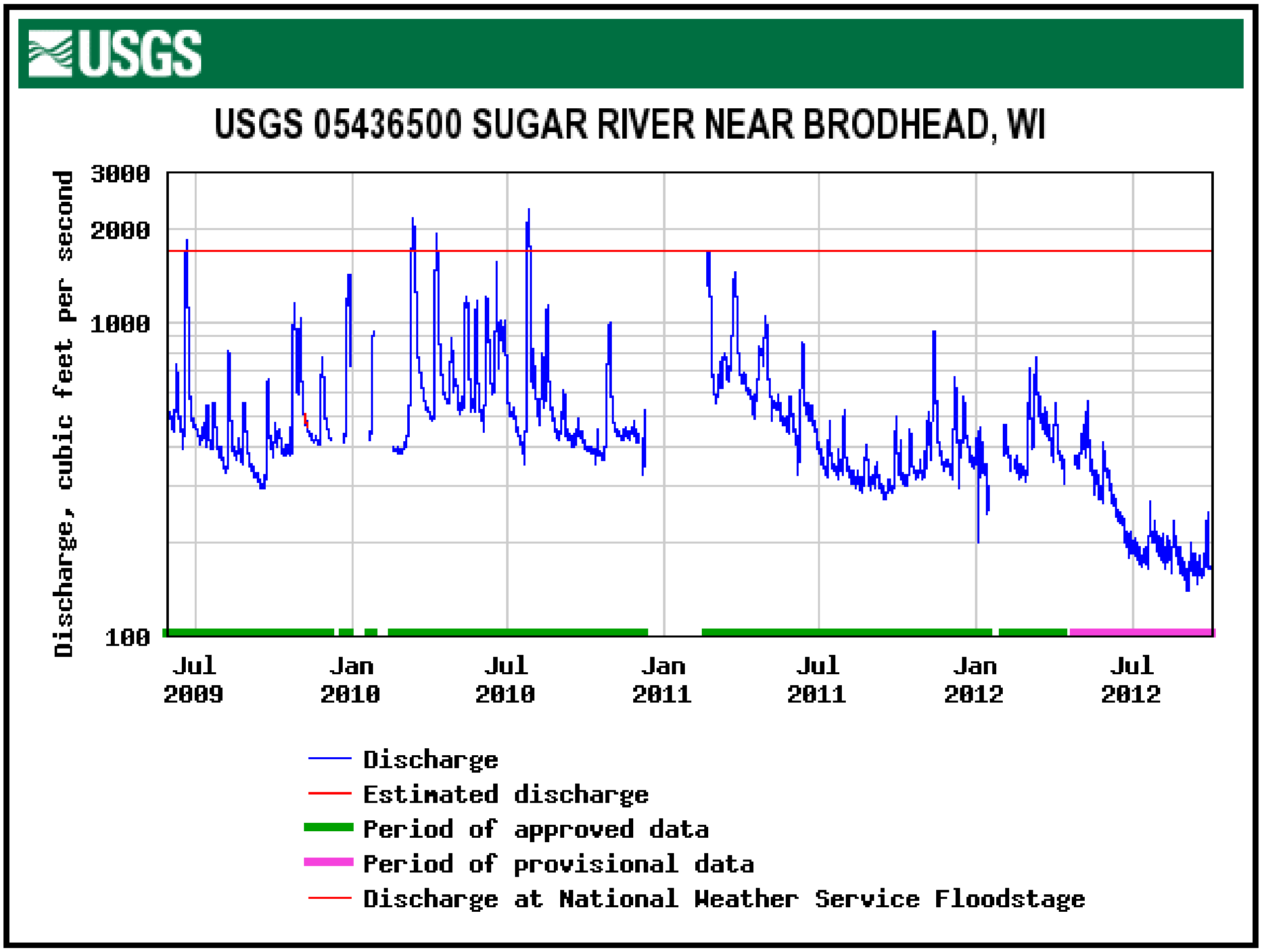
2012

Sugar River Floodplain

Floodplain habitats for the off-channel fish assemblage are very scarce along the Sugar River. Currently, three impoundments provide the most significant off-channel habitats but are common carp dominated and function as significant common carp recruitment areas for the entire river. A restoration effort was recently completed in Belleville, Wisconsin that greatly expanded off-channel habitat next to an impoundment by constructing a separation berm. The watershed diversion effort has potential to sustain native off-channel fishes by diverting sediment and pollutants but lacks the connectivity characteristic of a natural oxbow.

Sugar River recommendations: Eliminate the "shallow wildlife scrape" only model and restore selected oxbows along the Lower Sugar River to the original thalweg depths. This effort should restore groundwater connectivity and ultimately favorable for the off-channel fish assemblage. Introduce local starhead topminnows into the restored oxbows to establish a stable population. Remove the dams that form highly degraded millponds at Albany and Brodhead and restore oxbow habitats as alternatives.

Figure 15: USGS Sugar River Hydrograph at Brodhead, Wisconsin



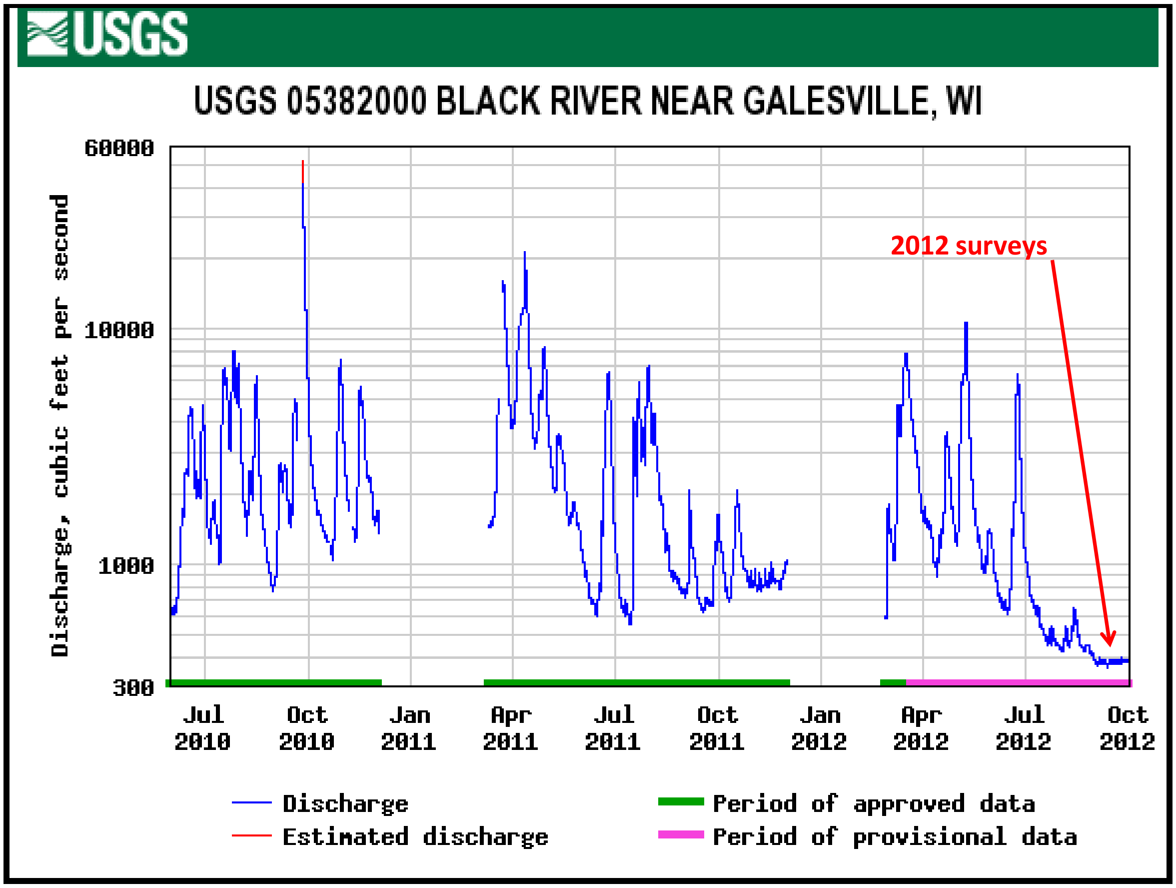
Black River (WBIC 1676700): The Black River was sampled at four sites in 2011 and 2012 as part of the SWG11-MON-004 project. Starhead topminnows, pirate perch, mud darters and a weed shiner were found in the Jackson County section of the Black River below Black River Falls. High quality oxbow habitats exist along the river in this county but the severe 2012 drought resulted in very low water levels in Stevens Lake. Low water in the lake reduced habitat and native fish, including starhead topminnows and pirate perch (see photo below). The USGS hydrograph in Figure 16 reflects the common pattern of drought impacts across the state in 2012 and flooding in 2010. The Lower Black River was surveyed as part of SWG09. Many sites were inaccessible at that time but appeared to be degraded from heavy metaphyton and perhaps floodplain aggradation.

Black River recommendations: More intensive off-channel survey work is needed to assess starhead topminnow, pirate perch, mud darter, weed shiner and other off-channel fish populations and habitats.

Lower Chippewa River (WBIC 2050000): Three sites were sampled along the Lower Chippewa River in 2011 and 2012 as part of the SWG11-MON-004 project. One of the three sites (Dead Lake) was nearly dry in 2012 and new sites sought as part of the SWG11-Cat2-007 project could not be found due to the extreme drought. The cover photo captured the prevailing low water conditions along the Chippewa River in 2012. Rare fish, mud darters and pugnose minnows, were found in only one oxbow lake that is highly eutrophic. Compared with the Wisconsin River, the Lower Chippewa River appears to lack sustainable floodplain habitats. The relative dearth of oxbows along the Chippewa River in 2012 may reflect a combination of factors linked to the complex fluvial geomorphology of the basin as well as the drought. Faulkner (2011) investigated this complex floodplain that displays post glacial aggradation and incised channel.

Lower Chippewa River recommendations: Address issues linked to the 303d designation for Silver Birch Lake that supports mud darter and pugnose minnow populations. Conduct further surveys on Thompson Lake that appears habitable for rare off-channel fish but none were found. Survey additional floodplain habitats along the Lower Chippewa River for rare off-channel fish populations, including connected near-channel sloughs and eddies that might support rare fish populaitons.

Figure 16: USGS Hydrograph of the Black River



Lemonweir River (WBIC 1301700): Scores of oxbow lakes border the Lemonweir River channel. We found pirate perch at three locations along the Lemonweir River in 2010 as part of the SWG09 project and in 2011-12 as part of the SWG11-MON-004 project. A pugnose minnow was also collected at the mouth of the river. Storage capacity loss appears to be a problem in some of the oxbows along with excessive metaphyton growths. Contaminated groundwater in the Lower Lemonweir River Watershed was ranked as a high priority for restoration. Groundwater contamination occurs along the river due to excessive manure and commercial fertilizer applications on porous soils (WDNR 2002).

Lemonweir River recommendations: Given the high number of oxbow lakes along the Lemonweir River, additional off-channel surveys are recommended. Expand riparian buffers to reduce runoff pollution, reduce polluted groundwater discharge and maintain oxbow storage capacity.

Bark River (WBIC 813500): Three sites were electroshocked within the Princess Point State Wildlife Area in 2012 as part of the SWG11-Cat2-007 project. Princess Point SWA encompasses the confluence of the Bark River and Scuppernong Creek. Both waterbodies were sampled along with a managed off-channel impoundment. Rare fish species were not found in this river system that is highly turbid within an aggrading floodplain. The managed tin-whistle impoundment was the only off-channel habitat that was not dry in the area. The impoundment was also very turbid due to common carp production.

Bark River recommendations: Adopt off-channel impoundment management techniques that will support a continuous native off-channel fish assemblage. These efforts could reduce potential for common carp recruitment and habitat destruction. Operating the impoundment for annual drawdowns may actually create disturbances that benefit common carp by eliminating native fish (common carp egg and fry predators) from the impoundment. There may be opportunities for natural oxbow restoration as well.

Des Plaines River (WBIC 734000): Pirate perch or other rare fish species were not found along the Des Plaines River. Two State Special Concern bullfrogs were observed in the river near the Illinois border. The WDNR Basin Plan (2011) identifies numerous impairments to the river including polluted runoff from urban and agricultural areas, channel modifications, streambank erosion, and loss of wetlands. The SWG11-Cat2-007 project surveys demonstrated that pirate perch are still hanging on at two sites along the Kilbourne Road Ditch (WBIC 737250) despite significant hydrologic and water quality impacts from development. Loss of sustainable off-channel habitats due to vertical floodplain accretion appears to be evident.

Des Plaines River recommendations: Implement WDNR basin recommendations. Consider restoring sustainable oxbows and sloughs that may provide refuges during environmentally stressful periods in the Des Plaines River and Kilbourne Road Ditch.

Illinois Fox River (WBIC 742500): The photo below characterizes the general lack of oxbow and slough habitats along the Illinois Fox River in 2012. The shallow small oxbows along the river were either dry or nearly dry. Small numbers of starhead topminnows were collected at various locations from about one mile above the confluence with the Mukwonago River extending downstream of the Waterford dam. They were typically found swimming with greater numbers of blackstripe topminnows near shore in eddies with aquatic vegetation. The sluggish nature of the Illinois Fox River lacks fast currents that would otherwise prohibit starhead topminnow survival in the main channels such as the Sugar River with higher velocities. Specific conductance levels in the river ranged from 960 to 1356 uS/cm with the highest measurements recorded in the 303d section of the river between Mukwonago and Big Bend. Starheads were also collected in an unnamed ditch above Waterford and in Tichigan Lake. A volunteer, Gary Johncox, also reported starheads at several locations within the Waterford impoundment.

Illinois Fox River recommendations: Conduct annual off-channel fish surveys to assess the status of starhead topminnows and other species. Restore former oxbows that lost storage capacity and sustainability due to vertical floodplain accretion. Regulate aquatic plant management activities in Tichigan Lake and Waterford impoundment to avoid loss of starhead topminnow habitat as well as other macrophyte obligate species.

Mukwonago River – Lulu Lake (WBIC 768800): Lulu Lake is listed as Outstanding Resource Waters (ORW) due to the relatively pristine watershed and water quality. Lake chubsucker was the only rare species found in the lake. The dearth of rare fish species in the lake is surprising since the Mukwonago River supports unusually high biodiversity and at least four rare fish species. Lyons (2011) reported regular collections of starhead topminnows, banded killifish, pugnose shiners and longear sunfish in the river at City of Mukwonago. Environmentally intolerant species were collected in the lake including the blacknose shiner and Iowa darter. Finding blacknose shiners and lake chubsuckers, both macrophyte obligate species, in the lake would suggest favorable habitat for starhead topminnows and other rare species.

Mukwonago River recommendations: Conduct more intensive nearshore fish surveys in Lulu Lake and surrounding tributaries.

Illinois Fox River oxbow that lost storage capacity and perhaps groundwater connectivity

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Lower Wisconsin River Floodplain Lakes Water Pollution Investigation

Diagnostic and Feasibility Study Part 1

Bakkens Pond

Prepared by David W. Marshall, Underwater Habitat Investigations LLC

Project Sponsor: River Alliance of Wisconsin

WDNR Lakes Planning Grant Project

December 2013

Thanks to study participants Doug Jones, Sherryl Jones, Timm Zumm, Merel Black, Tim Larson, Doug Risch, Matt Krueger, Susan Graham, and Sharon White. Thanks to Mike Mossman, Kevin Masarik, Cathy Bleser, Don Greenwood and Ken Wade for valuable comments.

Summary

During the growing season of 2013, severe impairments to recreation and water quality occurred in Jones Slough, Norton Slough and Bakkens Pond. Heavy metaphyton cover (Cladophora and duckweeds) rendered the sloughs unusable for recreational angling and boating. Significant environmental degradation included nitrogen hypersaturation, toxic levels of nitrates and low dissolved oxygen levels beneath the dense metaphyton cover. Nitrogen levels far exceeded the USEPA recommended criterion for the Driftless Area of 1.88 mg/l N. Very high nitrate levels were measured in most of the floodplain lakes that lie adjacent to the Pleistocene terrace but very low levels were detected in the Wisconsin River and Wagner Pond. The degraded conditions found in most of the floodplain lakes pose threats to the State Endangered starhead topminnow and other environmentally sensitive species, particularly darters that live in cutoff channel oxbow lakes. These problems are linked to excessive nutrient applications over coarse sandy soils that rapidly leach nitrates and likely phosphorus. A more detailed assessment of groundwater contaminants is needed to determine nitrogen and phosphorus loads and concentrations entering the cutoff channel oxbows.

Recommendations

1. Establish conservation buffers (~$1,300/ acre permanent easements and ~$3,300/acre fee title) around environmentally sensitive cutoff channel oxbow lakes with recommended widths of 100 meters or greater.

2. The Town of Spring Green should adopt Farmland Preservation Zoning. The town should not punish efforts to create buffers by raising property tax assessments since this activity should be considered an integral part of agricultural production.

3. Part 2 of the Lake Planning Grant Diagnostic and Feasibility Study should be conducted to better assess levels of nitrates and phosphorus entering cutoff channel lakes from the groundwater. Existing surface water data does not account for metaphyton nutrient uptake during the growing season. Additional monitoring should focus on nutrient cycling during high river flows since the floodplain may be a natural source of phosphorus and may affect de-nitrification. Groundwater monitoring should be conducted to assess the influence of existing buffers on nitrogen removal, such as the Sauk County School Forest, and this information may result in more effective buffer width recommendations. A survey of environmentally sensitive darters and other fish should be updated.

4. The State of Wisconsin should adopt USEPA recommended nitrogen criteria. Lower Wisconsin State Riverway floodplain lakes with excessive metaphyton, phosphorus and nitrogen should be placed on the USEPA 303d list of impaired waters.

Introduction

Below the Prairie du Sac dam, the Wisconsin River flows within a braided channel floodplain that remains in a relatively natural state. This high quality large river ecosystem was exemplified in an educational poster (SP River PAL River Planning Grant) that was produced in 2009 in conjunction with the 20 year anniversary of the Lower Wisconsin State Riverway. The State Riverway currently holds 99 fish species and the number of fish and overall biodiversity would be significantly lower without the vast network of oxbows, sloughs, delta ponds, beaver ponds and creek bottoms that support species adapted to off-channel habitats. The Lower Wisconsin River floodplain sustains a number rare fish species that thrive in these off-channel habitats including the State Endangered starhead topminnows (*Fundulus dispar*), State Special Concern mud darter (*Etheostoma asprigene*), State Special Concern pirate perch (*Aphredoderus sayanus*), State Special Concern weed shiner (*Notropis texanus*), State Special Concern pugnose minnow (*Opsopoeodus emiliae*), lake chubsucker (*Erimyzon sucetta*) and Mississippi grass shrimp (*Palaemonetes kadiakensis*). Off-channel habitats also provide refuges and nursery habitats for riverine fish species (Amoros 2001, Amoros and Bornette 2002, Kilgore and Miller 1995).

While the Lower Wisconsin River floodplain provides critical habitats for these species and the entire off-channel fish assemblage, the floodplain is a dynamic ecosystem that undergoes annual and seasonal changes that create both environmental opportunities and stressors. Amoros and Bornette (2002) demonstrated how hydrologic changes that occur in floodplains are closely linked to river stages. During very high river flow events, the river consumes the entire floodplain and cutoff channel oxbows become active river channels again. Scouring and removal of organic deposition can occur due to the relatively linear cutoff channel morphology. Under high river flows, alluvial groundwater moves toward the oxbows, causing significant water chemistry changes including lower dissolved oxygen, lower pH, stained water and increased organic carbon. Upland groundwater is the primary water source for cutoff channel oxbows during low flows and late summer median flows when water is very clear and alkalinity is higher. The upland groundwater is part of a massive Driftless Area groundwater flow system that begins in the bluffs, descends across the Pleistocene terrace and into the floodplain where cutoff channel oxbows and wetlands intercept groundwater before reaching the river (Pfeiffer et al. 2006). The combination of upland groundwater flows and well connected floodplain can sustain favorable oxbow water quality and habitat for rare off-channel species. Mud darters and pirate perch are often found within tree roots/woody debris along steep oxbow banks that intercept upland groundwater (Marshall 2012).

The Lower Wisconsin River was recently the focus of five lake planning grants, a river planning grant and two State Wildlife Grant projects that were conducted from 2007 to 2012 (Marshall 2012, Marshall 2009). These surveys demonstrated that water quality and habitats in some cutoff channel oxbows in the Spring Green area had become severely degraded due to contaminated groundwater. These problems were previously not assessed (Marshall and Lyons 2008, WDNR 2006) but became evident in Jones Slough in 2008 and later expanded in other oxbows in the area. By 2011, a diverse aquatic plant community in Norton Slough disappeared as it became smothered by a dense mat of metaphyton (Cladophora filamentous algae and duckweeds). The metaphyton cover shaded out the plants and significant amount of fish habitat. The dense metaphyton mat also eliminated photosynthesis and atmospheric exposure to the water that resulted in very low dissolved oxygen levels. Anecdotal reports from resident anglers also mirrored these recent findings.

In an effort to better define the rapid water quality decline that occurred in Norton Slough, WDNR Paleolimnologist Paul Garrison, Richard Wedepohl and Dave Marshall attempted to collect sediment cores from the lake in March 2012 (Marshall 2012). The sampling demonstrated that scouring had occurred during floods since the bottom was mostly sand with very little organic deposition. This finding was consistent with Amoros and Bornette (2002) but more importantly revealed that the water quality change was not linked to internal loading but rather reflected recent nutrient loadings. Vertical temperature profiles of the lake revealed the significant upland groundwater flow that dominates the lake hydrology. This information, coupled with the trend of increasing nitrates in adjacent wells, suggested that contaminated groundwater is the primary nutrient source.

This project was funded and designed to assess levels of pollutants and impairments in the Lower Wisconsin State Riverway cutoff channel oxbow lakes, identify potential pollutant sources and recommend restoration alternatives.

Methods

Floodplain lake water quality surveys were performed on May 9th, May 19th, June 1st, July 29th, August 13th, August 20th and August 30th in 2013. Nitrate sampling was not performed throughout most of June and July due to very high river flows that changed the hydrology and significantly reduced upland groundwater flows to the lakes. During each survey, a YSI Pro Plus meter was used to measure nitrates (0.5 m depth) and was calibrated each survey day using 1 mg/l NO_3 and 100 mg/l NO_3 standards. Additional quality assurance included paired SLOH nitrate samples that were collected from eight sloughs on May 19th and July 29th. The samples were also analyzed for total phosphorus and in a few instances Total Kjeldahl Nitrogen to calculate Total Nitrogen concentration. A YSI Model 52 meter was used to measure dissolved oxygen and temperature. The meter was air calibrated according to specifications. A YSI Model 63 meter was used to measure pH and specific conductance. A 120 cm secchi transparency tube was used to measure water clarity. Metaphyton cover (filamentous algae and duckweeds) was estimated using the following scale (Sullivan 2008): 0 = no filamentous algae or duckweeds present, 1 = 1 – 20 % metaphyton cover, 2 = 21 – 40 % metaphyton cover, 3 = 41 – 60 % metaphyton cover, 4 = 61 – 80 % metaphyton cover, and 5 = 81 – 100 % metaphyton cover. Most of the cutoff channel oxbow lakes had documented populations of state endangered starhead topminnows (SHTM). A long handed small mesh dip net was used to determine the presence/absence of this rare species in the lakes.

In addition to these five surveys, volunteers Doug and Sherryl Jones measured dissolved oxygen and temperature using a YSI Model 57 meter along with water levels (not linked to sea level) twice daily from their pier on Norton Slough. The water level data was also compared with the USGS flow gaging station water levels at Muscoda.

Findings

Norton Slough (WBIC 1247200 – 13 acres) and Jones Slough (WBIC 1247300 – 7 acres) drain a watershed area of approximately 1667 acres. Hutter (Rainbow, WBIC 1247000) Slough and Wood Slough (WBIC 1247100) are also part of this sand terrace oxbow drainage system. The watershed lies west of Wilson Creek where the low gradient Pleistocene terrace widens significantly, rendering surface watershed divisions somewhat nebulous.

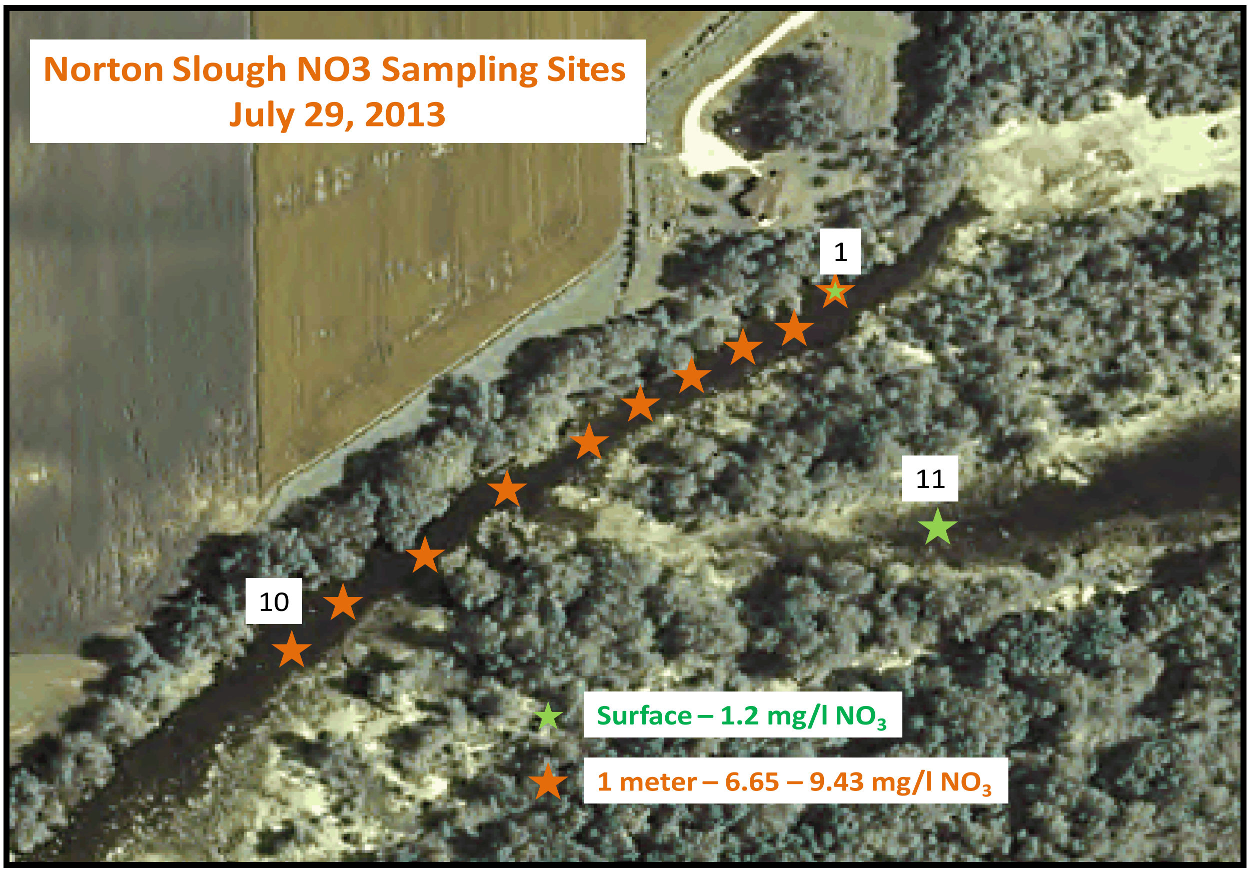
Norton Slough was sampled six times at Site 1 (Figure 1 map) in 2013 from May 9 to August 30. Nitrate levels were high (4 – 5.29 mg/l) early in the season but declined (1.2 - 2.02 mg/l) by early June (Figure 2). This trend coincided with increased metaphyton cover that was excessive and covered most of slough surface area throughout late summer. The nitrate decrease likely reflected nutrient uptake near the surface (0.5 m). On July 29th, numerous locations in Norton Slough were sampled for nitrates and significantly higher levels were found 1 – 1.5 meters below the surface where colder dense groundwater flows into the slough (Figures 3 and 4). Nitrate concentrations ranged from 6.65 to 9.43 mg/l across Norton Slough at 1 – 1.5 meters depth while the 0.5 meter samples were less than 1.5 mg/l (Figure 3).

Total phosphorus samples were also collected on May 19th and July 29th. Concentrations were below the recommended criterion of 40 ug/l at Site 1 but exceeded the concentration at Site 11 (93 ug/l) on July 29th. Site 11 was used as a surrogate for Wood Slough that was inaccessible due to low water levels and impassible metaphyton cover. Consistent with nitrates, variable phosphorus concentrations may have reflected metaphyton uptake and growth. The total kjeldahl nitrogen concentration on May 19th was 1.03 mg/l for a total nitrogen concentration of 5.15 mg/l based on State Lab of Hygiene sample results. The slough appeared to be phosphorus limited at the time given the phosphorus concentration of 0.038 mg/l (N:P > 20:1).

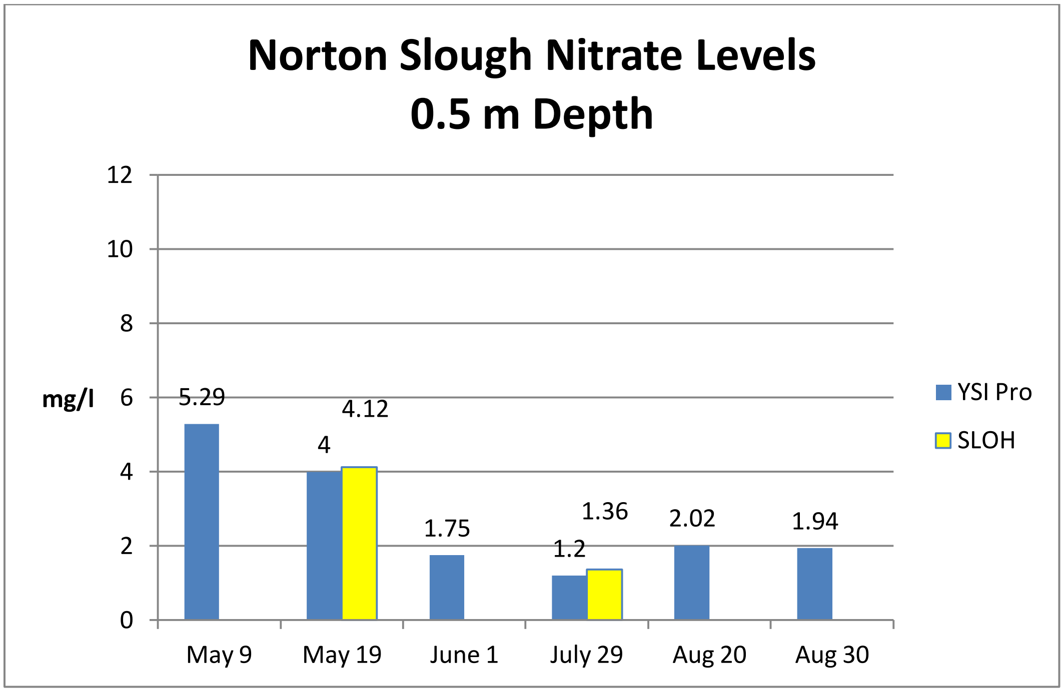
Metaphyton densities in Norton Slough are summarized in Figure 4. Over 80% of the surface was covered with metaphyton on July 29 and August 20, resulting in recreation impairments and negative effects on lake ecology and water chemistry. Doug and Sherryl Jones sampled Norton Slough twice daily for water temperature, dissolved oxygen and water level from May 20th to August 20th. Figure 5 displays dissolved oxygen versus water levels in Norton Slough and USGS stage data from the Muscoda flow station. Dissolved oxygen levels declined precipitously during high river stages when alluvial groundwater entered the slough. Extended periods of low dissolved oxygen were measured in the slough during the high river stages. During low flow periods, low dissolved oxygen levels (below water quality criterion of 5 mg/l) also occurred intermittently and likely reflected suppressed photosynthesis (and increased respiration) due to dense metaphyton cover that was somewhat variable due to shifting winds. SHTM were collected in Norton Slough on August 20th.

Jones Slough was sampled five times in 2013. It had the highest metaphyton densities, highest nitrate concentrations (13.3 mg/l on July 29th) and total phosphorus concentrations. Total phosphorus samples exceeded recommended 40 ug/l criterion (44 ug/l and 1340 ug/l). On May 19th, the total kjeldahl nitrogen concentration was 0.839 mg/l and total nitrogen concentration was 6.7 mg/l. As a result, the N:P > 20:1 ratio suggested that the slough was phosphorus limited. On July 29th, the N:P ratio was 12:1 or indeterminate. Moste likely neither nutrient was limiting given the very high concentrations at that time (TN = 16.23 mg/l and TP = 1.34 mg/l). SHTM were collected in Jones Slough on August 20th and August 30th.

Figure 1: Norton Slough sampling locations



Figures 2 and 3: Nitrate concentrations at Site 1 (0.5 m depth) and across Norton Slough on July 29th at both 0.5 and 1 meter depths.



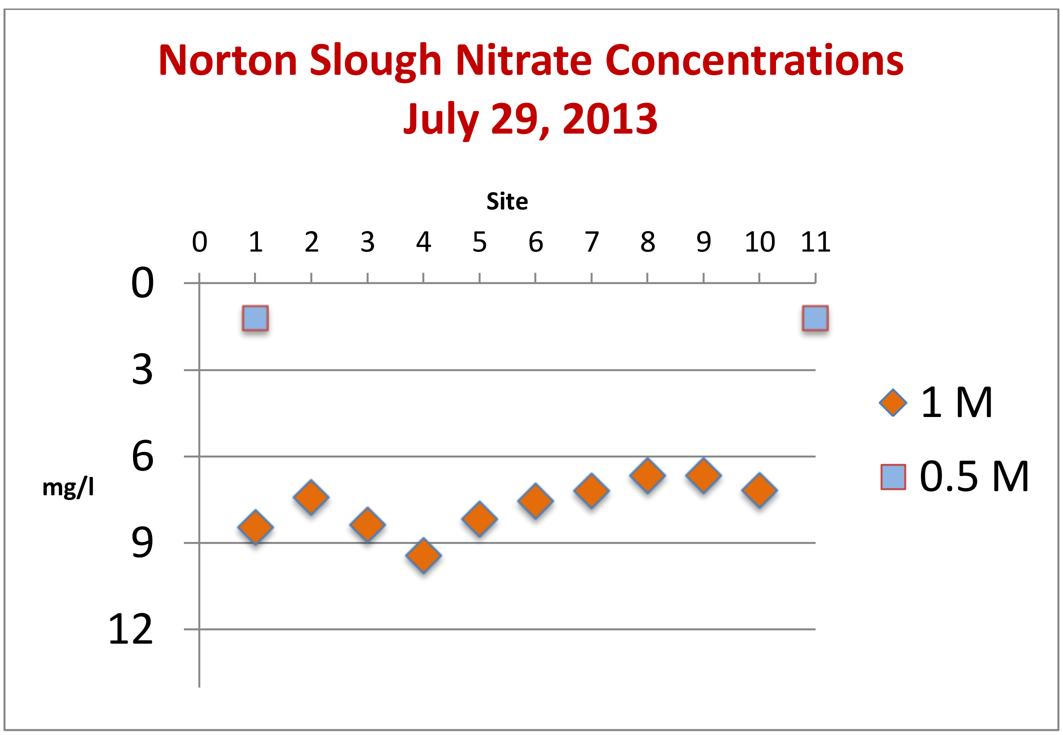


Figure 4: Norton Slough Metaphyton Cover (0 = none, 5 = 80 – 100%

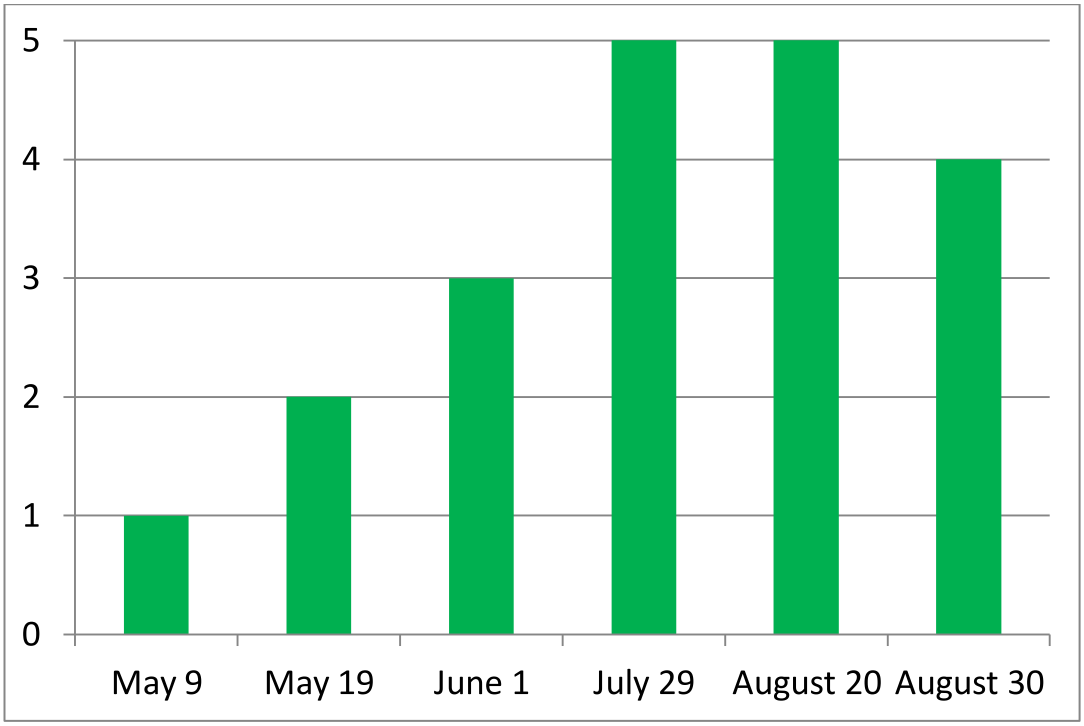


Figure 5: Water Level and Norton Slough Dissolved Oxygen Trends

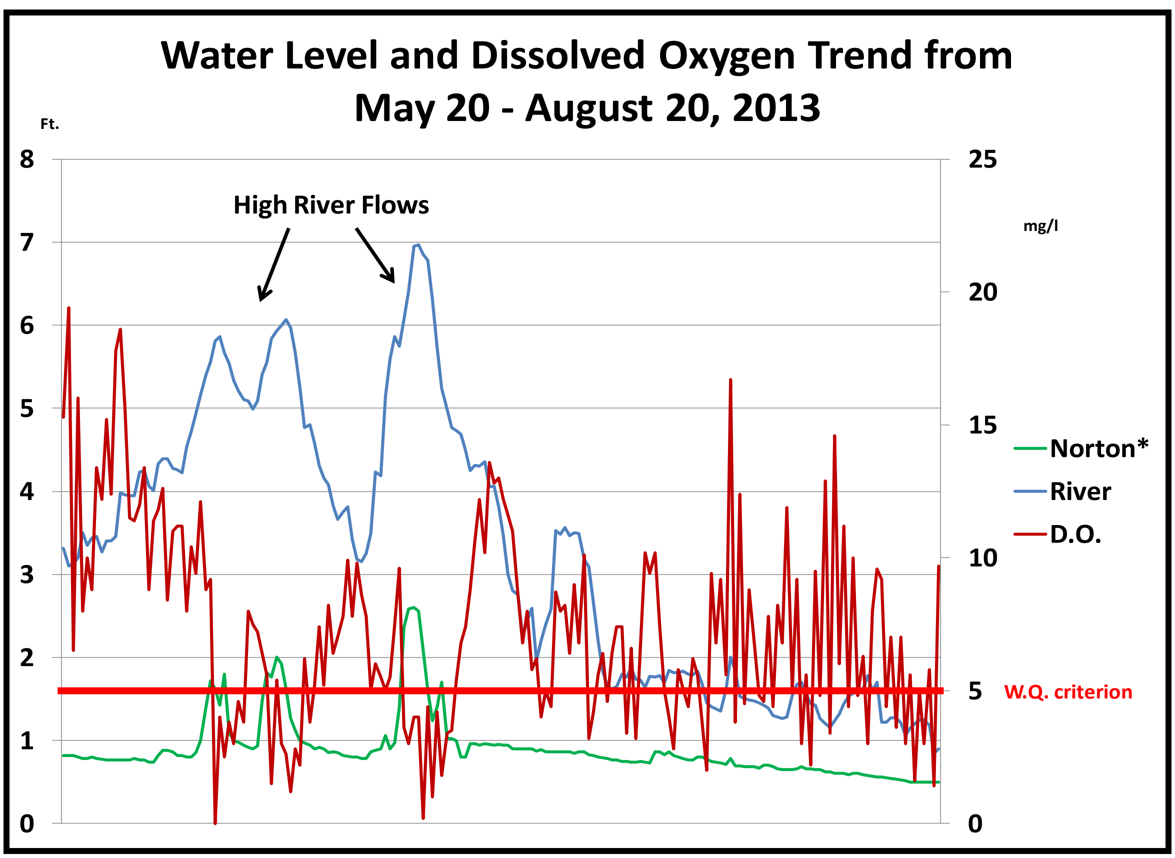


Figure 6: Jones Slough Nitrate Levels

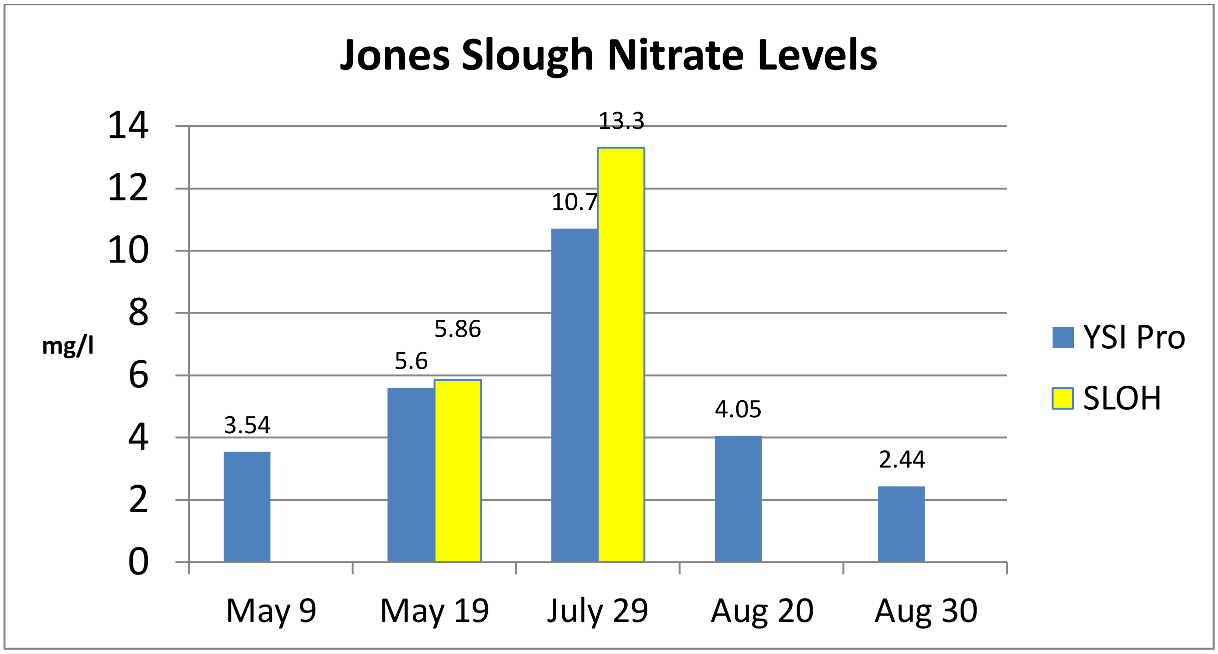
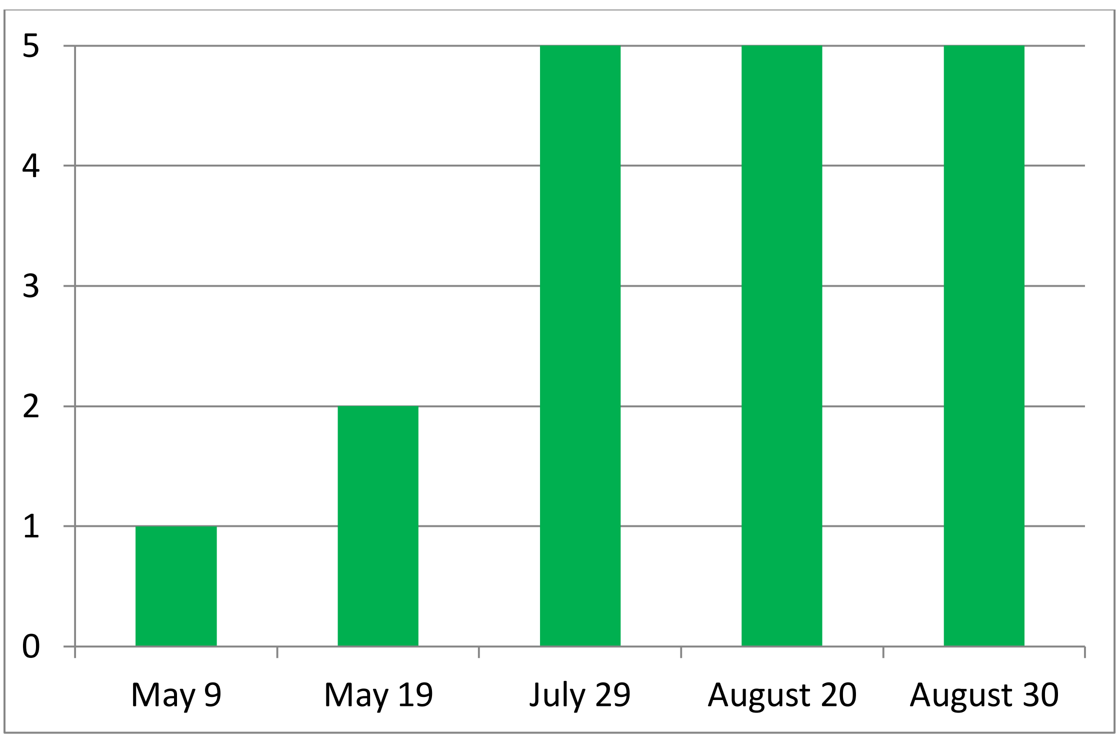
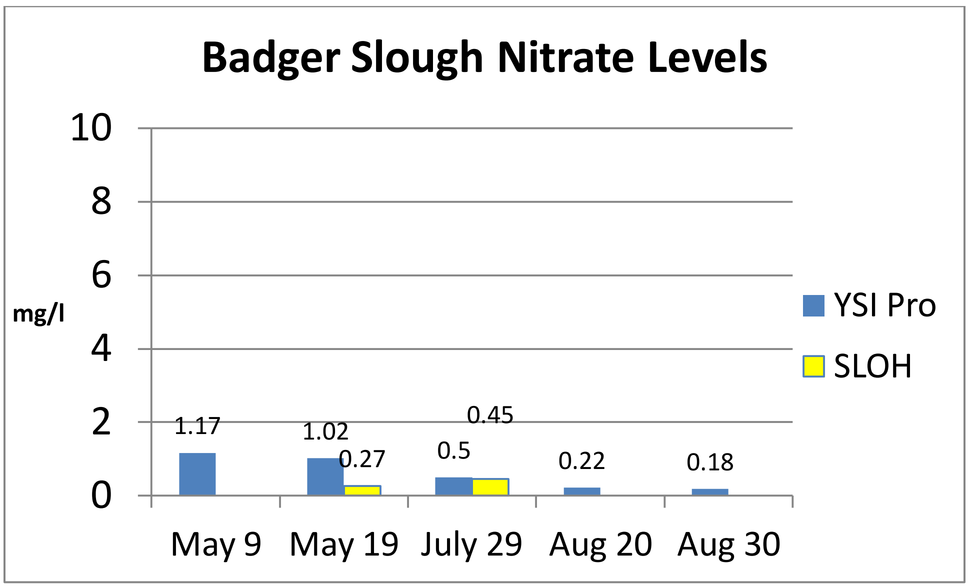


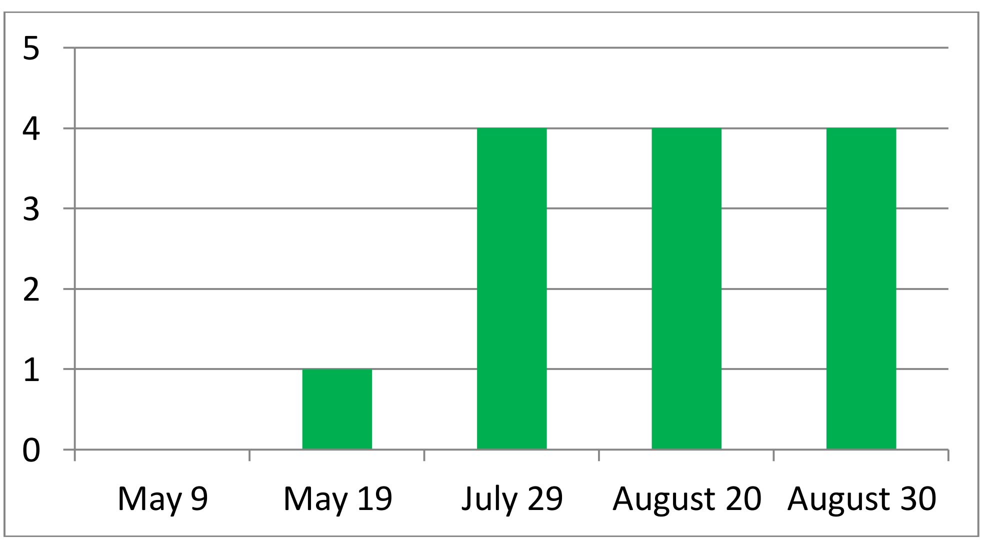
Figure 7: Jones Slough Metaphyton Cover



"Badger Valley" or unnamed (WBIC 1248000) is a Town of Troy cutoff channel oxbow lake that is sustained by both upland groundwater and a small tributary. The lake is about 5 acres in surface area with a maximum depth of about 5 feet. It lies much closer to the surrounding bluffs than the other lakes to the west and drains a very steep watershed. The 1404 acre watershed is about 22% agricultural and mostly wooded (~70%). The steep slopes have the potential to deliver significant sediment loads and nutrients. The tributary elevation change is very high at 125 ft/mile. Badger Slough contained relatively low nitrate concentrations in 2013 (Figure 8) compared with the floodplain lakes that lie adjacent to the broad Pleistocene terrace. Yet excessive metaphyton cover (Figure 9) was found late summer. While nitrogen levels were low in this slough and likely reflect wooded bluff groundwater discharge, phosphorus levels were high (81 ug/l and 77 ug/l). The high phosphorus concentrations may have reflected recent channel excavations for agricultural drainage in the steep watershed. On May 19th, the total kjeldahl nitrogen concentration was 1.1 mg/l and total nitrogen concentration was just 1.37 mg/l based on State Lab of Hygiene sample results. The N:P ratio was 17:1 and suggested a phosphorus limited ecosystem at the time. SHTM were collected the slough on August 20th and August 30th.

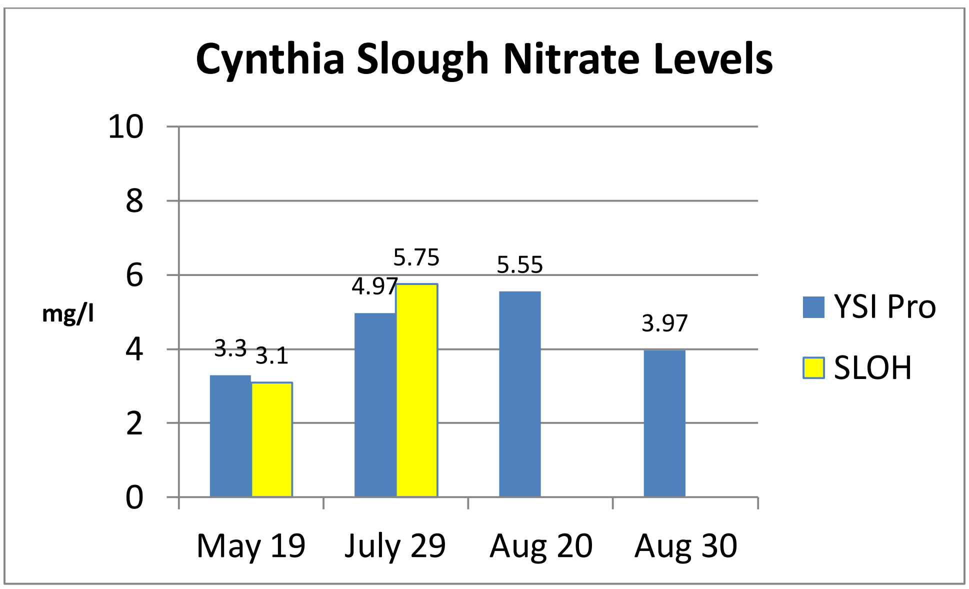
Figures 8 and 9: Badger Slough Nitrate Concentrations and Metaphyton Cover





Cynthia Slough (WBIC 5034625 – 14 acres) is located directly south of the Village of Spring Green and is surrounded by the Wisconsin Riverside Resort to the north. The floodplain lake outlet is directly connected to the river, accommodating frequent fish migrations. It shares the same elevation with the river and water levels fluctuate with river stages. The Pleistocene terrace and floodplain on the north side of the river is expansive. Surface watershed boundaries are unclear within this broad flat area. The estimated 708 acre watershed is considerably more developed than the other watersheds including about 80% moderate urban density and about 17% cropland. Cynthia Slough was sampled four times in 2013 with the nitrate data displayed in Figure 10. High levels were routinely found in Cynthia Slough but levels were somewhat lower compared to some of the other lakes in the area. A maximum concentration of 5.75 mg/l was significantly lower than a groundwater fed rivulet entering the slough with a concentration of 8.52 mg/l on August 30th. Metaphyton cover was typically low in Cynthia Slough and never exceeded a density level of 1. No STHM were collected in Cynthia Slough.

Figure 10: Cynthia Slough Nitrate Levels

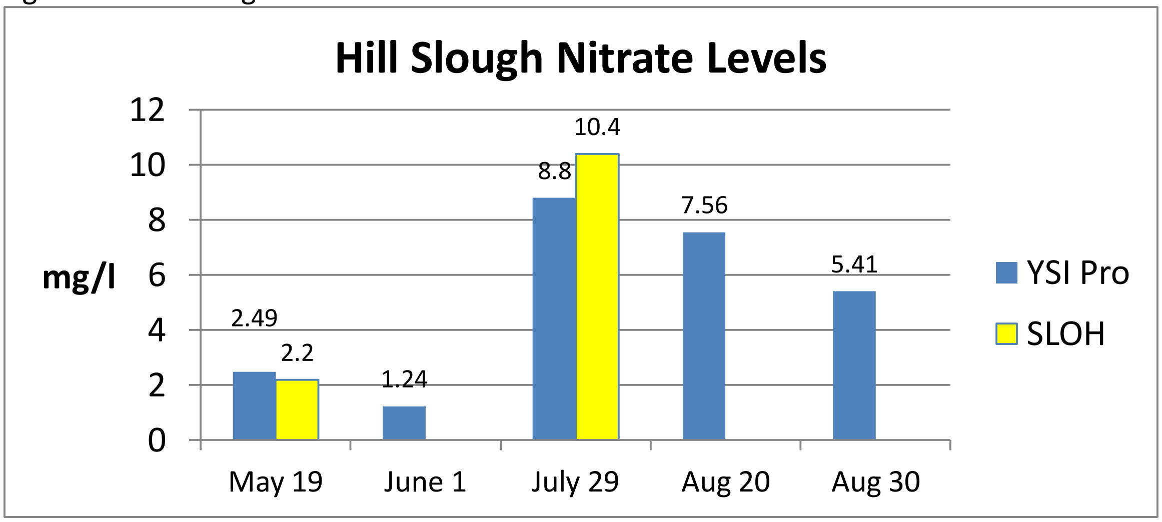


Hill Slough (WBIC 1241200 – 19 acres) is located about a mile west of Cynthia Slough and consists of a series of connected basins over 2 miles long. The far west basin drains into a river side channel. The lake drains an area of mostly sand terrace croplands of about 1220 acres. Hill Slough was sampled five times during the summer of 2013 with moderate levels of nitrates in the slough detected early in the season but increasing to very high levels by mid-summer (Figure 11). Total phosphorus concentrations exceeded the recommended criterion for drainage lakes concentration of 40 ug/l (68 ug/ and 47 ug/l$^{)}$. In spite of the very high nitrogen and phosphorus concentrations in Hill Slough, metaphyton densities never exceeded a cover value of 1. The sampling site was located on the eastern half of the slough where the hydrology and nutrient inputs are likely different from the west side. A flood control drainage ditch, that is highly polluted, discharges into the floodplain forest that lies adjacent to Hill Slough. The impacts of that ditch likely do not affect the water quality at the Hill Slough site monitored for this project since it is located (east) upstream of the ditch. SHTM were collected on June 1, July 29, August 20 and August 30.

Bakkens Pond (WBIC 1236700 – 14 acres) is a manipulated flowage system upstream of Long Lake with two earthen dams and tin-whistle discharge structures. It drains at least 660 acres of sand terrace cropland and is sustained by groundwater and a clear groundwater fed stream. Bakkens Pond is part of the Bakkens Pond State Natural and displays marshy habitat with abundant submersed, floating leaf and emergent plants. The flowage was sampled at two locations in 2013. Both sites displayed very poor water quality with excessive metaphyton cover (Figure 15). Bakkens Pond 1 (SWIMS 10017532 – cover photo) contained high nitrate concentrations in each of four sampling days (Figure 12). Phosphorus concentrations also exceeded criterion for drainage lakes (40 ug/l) at 41 ug/l and 71 ug/l. Nitrate concentrations were very high at Bakkens Pond 2 (SWIMS 10019775) ranging from 5.42 to 10.23 mg/l. Phosphorus concentrations at Bakkens Pond 2 were lower at 33 ug/l and 11 ug/l. No SHTM were found at Bakkens 1 but were found at Bakkens 2 on three sampling dates.

Jones Slough, July 29, 2013

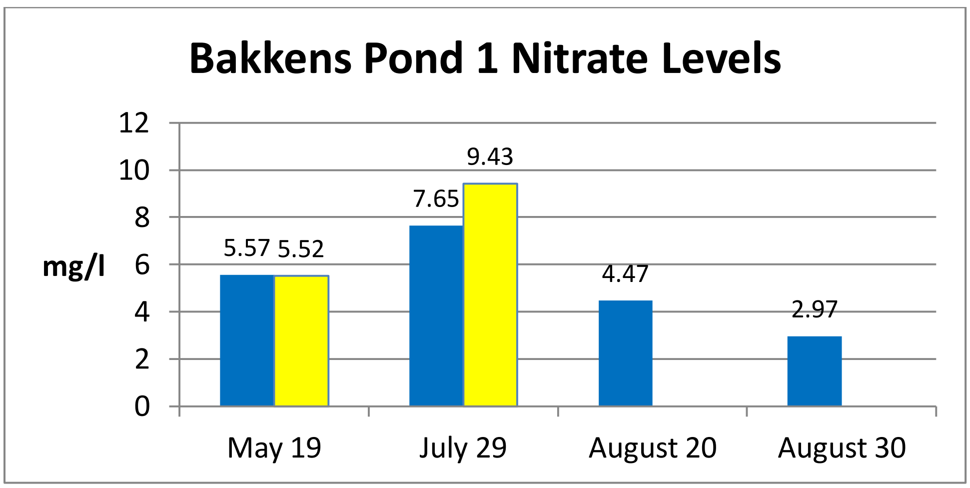
Figure 11: Hill Slough Nitrate Levels



Big Hollow Drainage Ditch (SWIMS 10037286)

Bakkens Pond drains into Long Lake (WBIC 1236600 – 37 acres). Long Lake is more typical of the long narrow cutoff channel oxbows. Long Lake is also more buffered from direct sources of contaminated groundwater than most of the lakes that lie directly adjacent to the Pleistocene terrace. The School Forest lies just to the north of the lake. The Long Lake watershed is at least 448 acres of mostly woods and low density residential. Nitrate levels in Long Lake revealed agricultural inputs with concentrations ranging from 2.54 – 4.23 mg/l but these levels were significantly lower than levels found upstream in Bakkens Pond. Phosphorus concentrations fluctuated from a high of 55 ug/l on May 19th to 33 ug/l on July 29th. Metaphyton levels, ranging from 0 to 1, were significantly lower in Long Lake compared to Bakkens Pond upstream. More sampling is needed to determine if Bakkens Pond is functioning as a nutrient sink. The primary production in Long Lake was primarily benthic algae that floated to the surface as opposed to the dense Cladophora filamentous algae and duckweed growths found in more severely degraded floodplain lakes including Bakkens Pond, Norton Slough and Jones Slough. SHTM were collected in Long Lake on July 29th and August 20th. Further monitoring is needed to determine if the elevated nitrogen and phosphorus concentrations were coming Bakkens Pond or upland groundwater.

Figure 12 and 13: Bakkens Pond Nitrate Concentrations



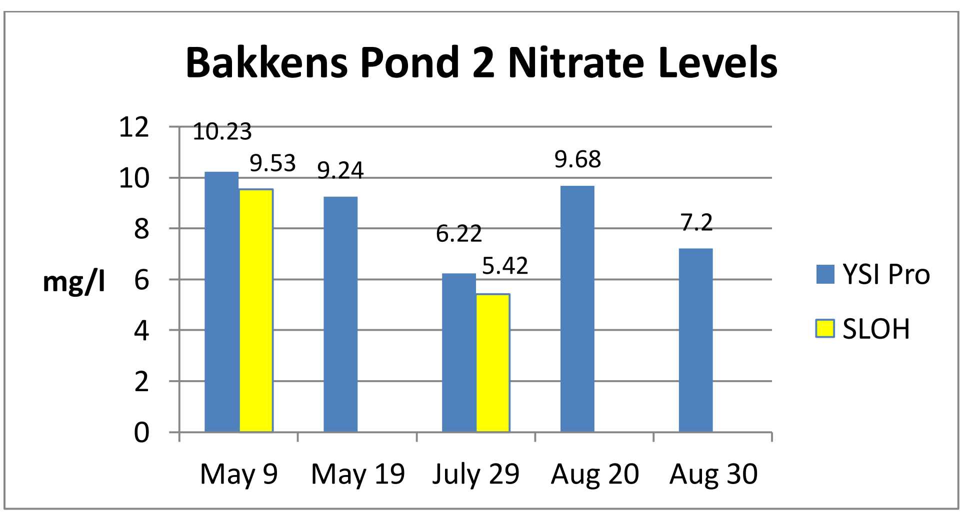


Figure 14: Bakkens Pond Metaphyton Cover

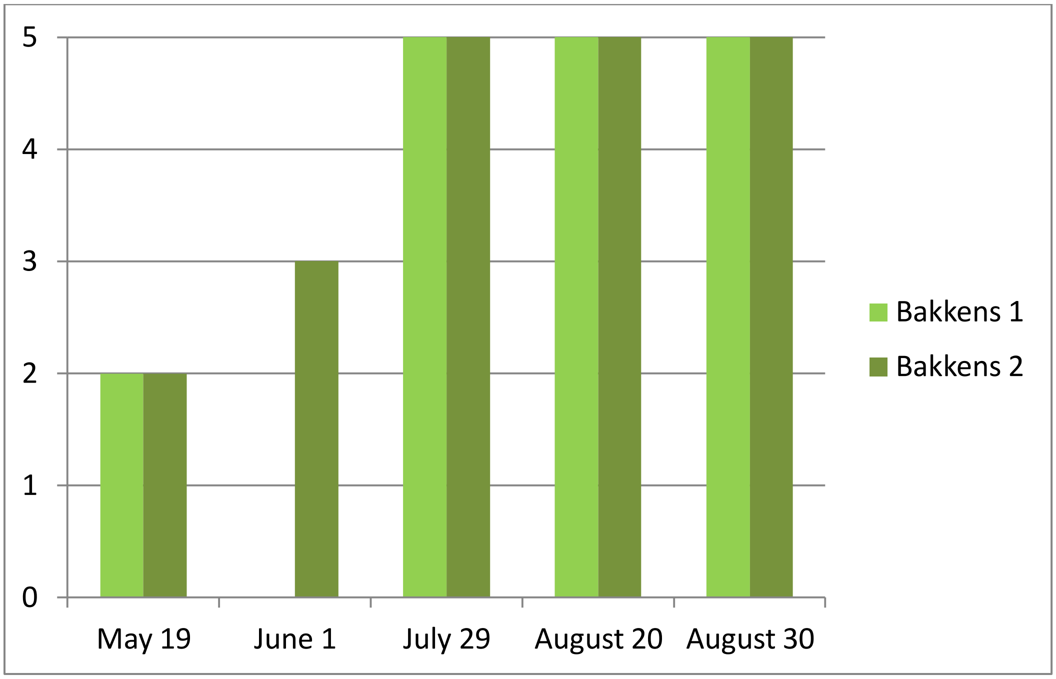
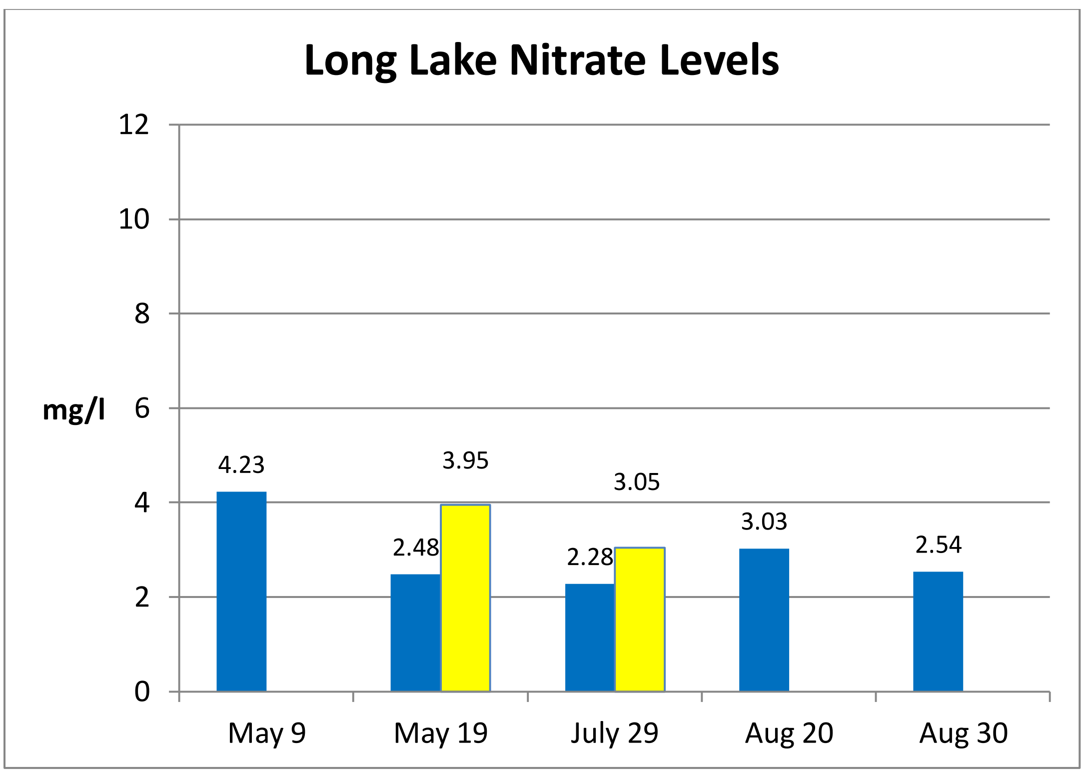


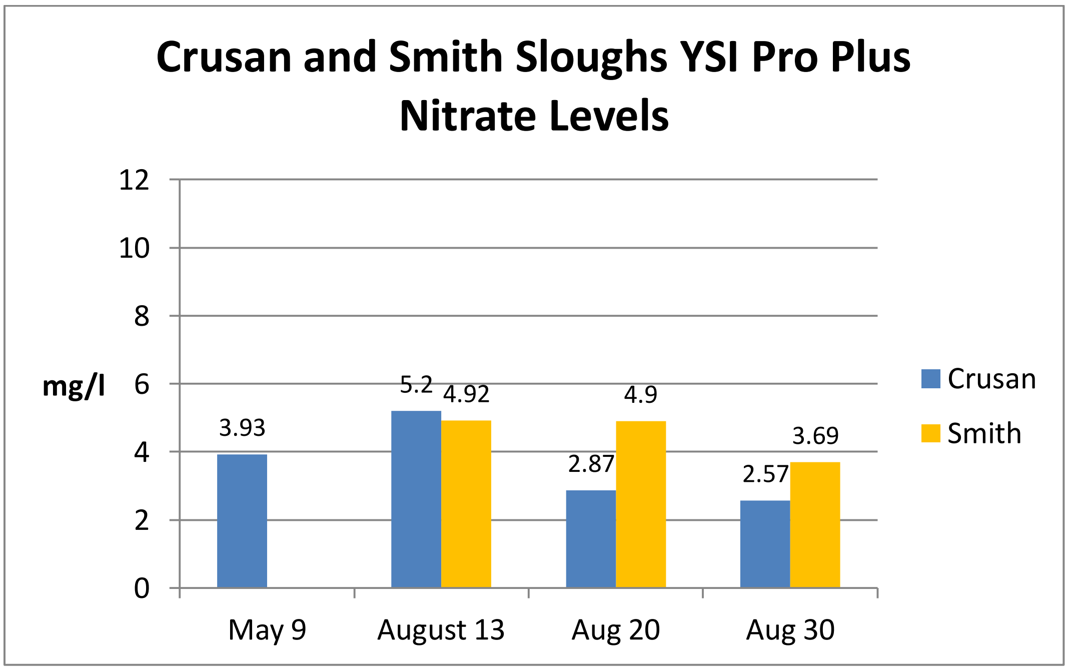
Figure 15: Long Lake Nitrate Levels



Additional Sampling Sites

West of Long Lake lie Smith (WBIC 1236400 – 17 acres) and Crusan (WBIC 5573731 – 20 acres) sloughs. These waterbodies were sampled several times using the YSI Pro nitrate meter to represent water quality conditions within the Richland County part of the Pleistocene terrace and floodplain. Both lakes contained elevated nitrate concentrations, reflecting anthropogenic pollution but the metaphyton cover was not as severe as conditions in Jones Slough, Norton Slough or Bakkens Pond. The nitrate results are presented in Figure 16. Consistent with Long Lake, the primary production in the Richland County lakes was primarily benthic algae (mostly Oscillatoria) that floated to the surface as opposed to Cladophora and duckweeds. Both sloughs had metaphyton cover of level 3 on August 20th and August 30th. SHTM were found in Crusan Slough on August 13th, August 20th and August 30th and in Smith Slough on August 20th.

Figure 16: Crusan Slough and Smith Slough Nitrate Levels



In an effort to sample reference sites, Wegner Borrow Pit (WBIC 5574197) was sampled on August 20th and August 30th. The pond is located at the Highway 23 Spring Green canoe launch and park and is well buffered by the floodplain forest. The Wisconsin River at the canoe launch was also sampled as a reference site and because comments at Town of Spring Green public meetings suggested that the river was the actual cause for water quality declines in the floodplain lakes. The results indicated that both waterbodies contained very low nitrate and total nitrogen concentrations (Figure 17). The sampling results are also consistent with the WDNR monthly monitoring data presented in Figure 18 at Muscoda. Metaphyton cover was nonexistent in Wegner Pond.

Figure 17: Reference YSI Pro Plus Nitrate Levels in Wagner Pond and Wisconsin River

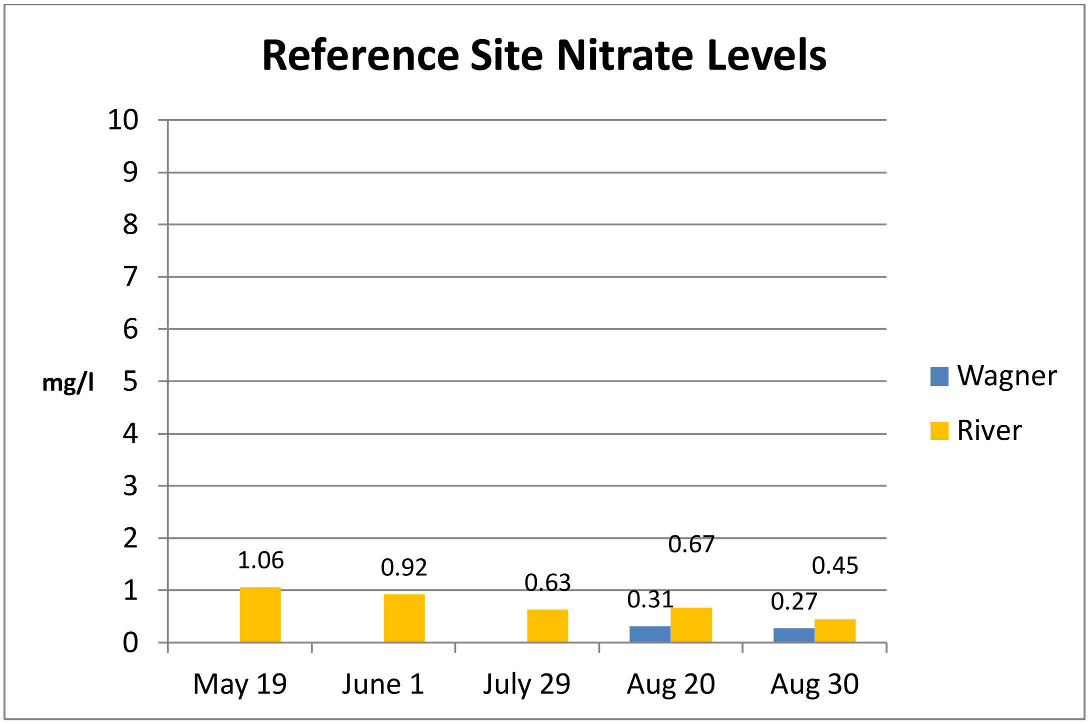
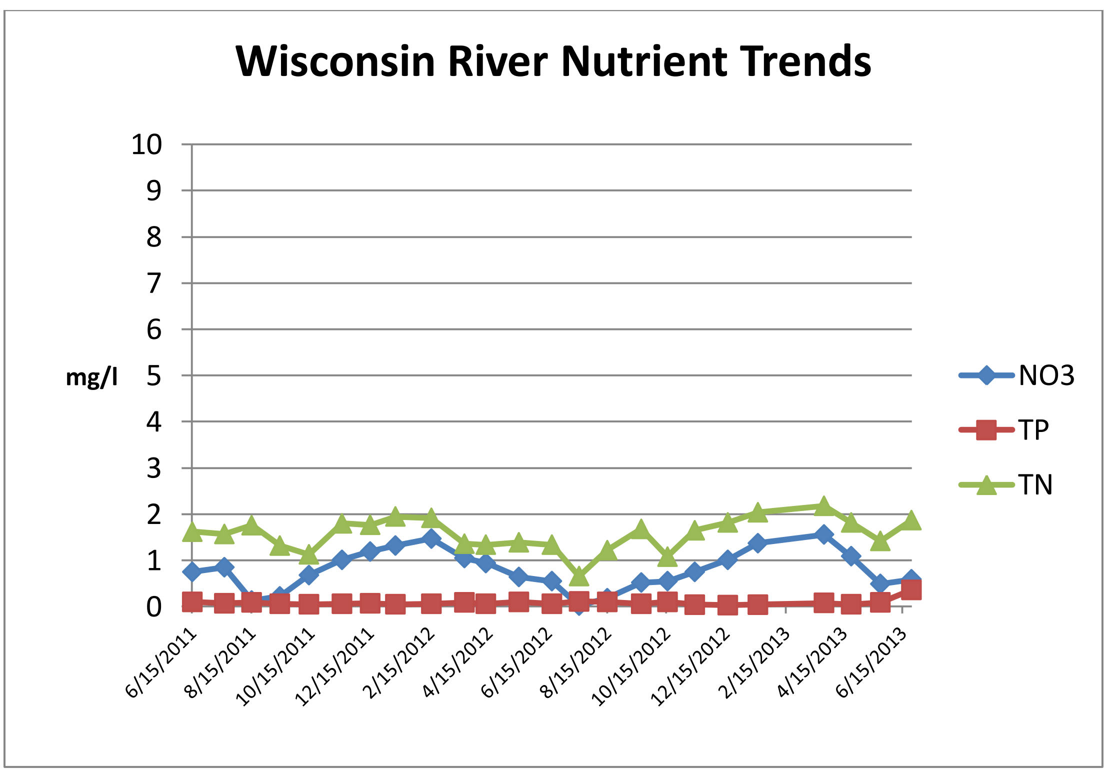
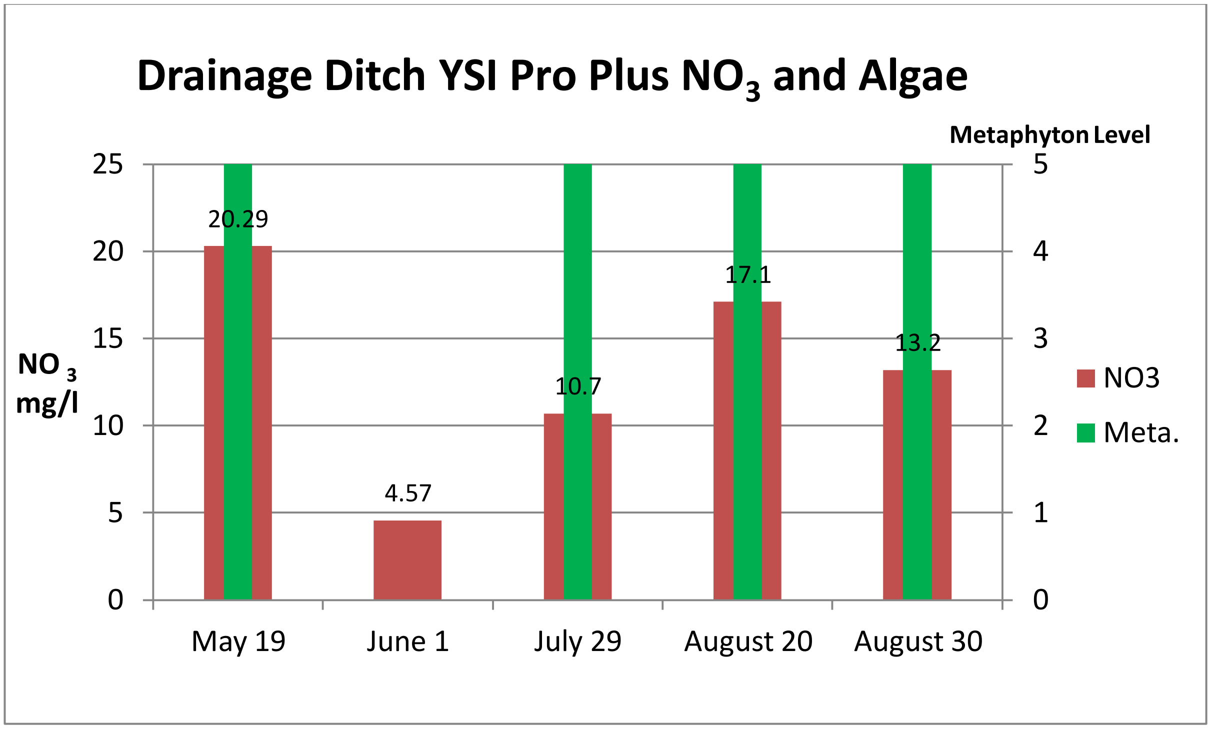


Figure 18: Wisconsin River Nutrient Concentrations at Muscoda



The flood control drainage ditch Big Hollow Drainage Ditch (SWIMS 10037286) was sampled five times in 2013 and displayed the highest nitrate concentrations found in the area. The high nitrate concentrations coincided with dry periods when groundwater dominated the ditch hydrology. The exception occurred on June 1st when the sample was collected during an extended period of precipitation. The nitrate and Cladophora levels were the lowest in the ditch after precipitation and flushing had occurred.

Figure 19: Flood Control Drainage Ditch Nitrate and Metaphyton Levels



YSI Pro Meter versus SLOH Nitrate Results

Using the recommended calibrations standards of 1 mg/l and 100 mg/l NO_3, the nitrate field measurements were in close agreement with the SLOH results ($R^2 = 0.948$). The means of 16 paired samples (meter 4.97 and SLOH 5.29) were not statistically different ($P = .05$). The greatest differences were found at the higher ranges (> 8 mg/l) where the meter measurements were lower than the SLOH in three of four samples (Figure 20).

Nutrient Management Implications

Phosphorus concentrations for Badger Valley, Jones Slough, Norton Slough, Cynthia Slough, Hill Slough, Bakkens Pond and Long Lake appear in Figure 21. 70.6 % of the samples exceeded the recommended phosphorus criterion of 40 ug/l for drainage lakes. These concentrations likely did not reflect phosphorus contained within surface and bottom metaphyton growths. All but one of the samples fell within the eutrophic TSI range, however the recommended criterion is more useful since floodplain lake eutrophication is typically expressed in different forms than in glacial lakes or impoundments where standard TSI metrics apply. Except for a few sampling dates early in the season, when river levels were high, the floodplain lakes were also very clear with minimal planktonic algae. The water clarity data were based on the 120 cm secchi transparency tube since water depths were too shallow for standard secchi measurements. As a result, water clarity data could not be transformed into TSI values. The use of WILMS to predict phosphorus loading is also of limited use given the nebulous watershed boundaries and undetermined groundwater levels of phosphorus. The WILMS predicted annual phosphorus loadings were 318 lbs./yr. for Badger Valley, 155 lbs./yr. for Bakkens Pond, 417 lbs./yr. for Cynthia Slough, 414 lbs./yr. for Hill Slough, 1061 lbs./yr. for the Jones – Norton flowage and 144 lbs./yr. for Long Lake. The use of this software is of limited value without assessing the levels of groundwater phosphorus loads and dynamics of fluctuating river stages. The effects of the fluctuating river levels were evident on dissolved oxygen levels in Figure 5 but how nutrients may change under these conditions requires further investigation. There is potential release of iron-bound phosphorus within the floodplain forest during high river stages (Loeb et al. 2008).

Figure 20: Comparing YSI Pro and SLOH Nitrate Concentrations

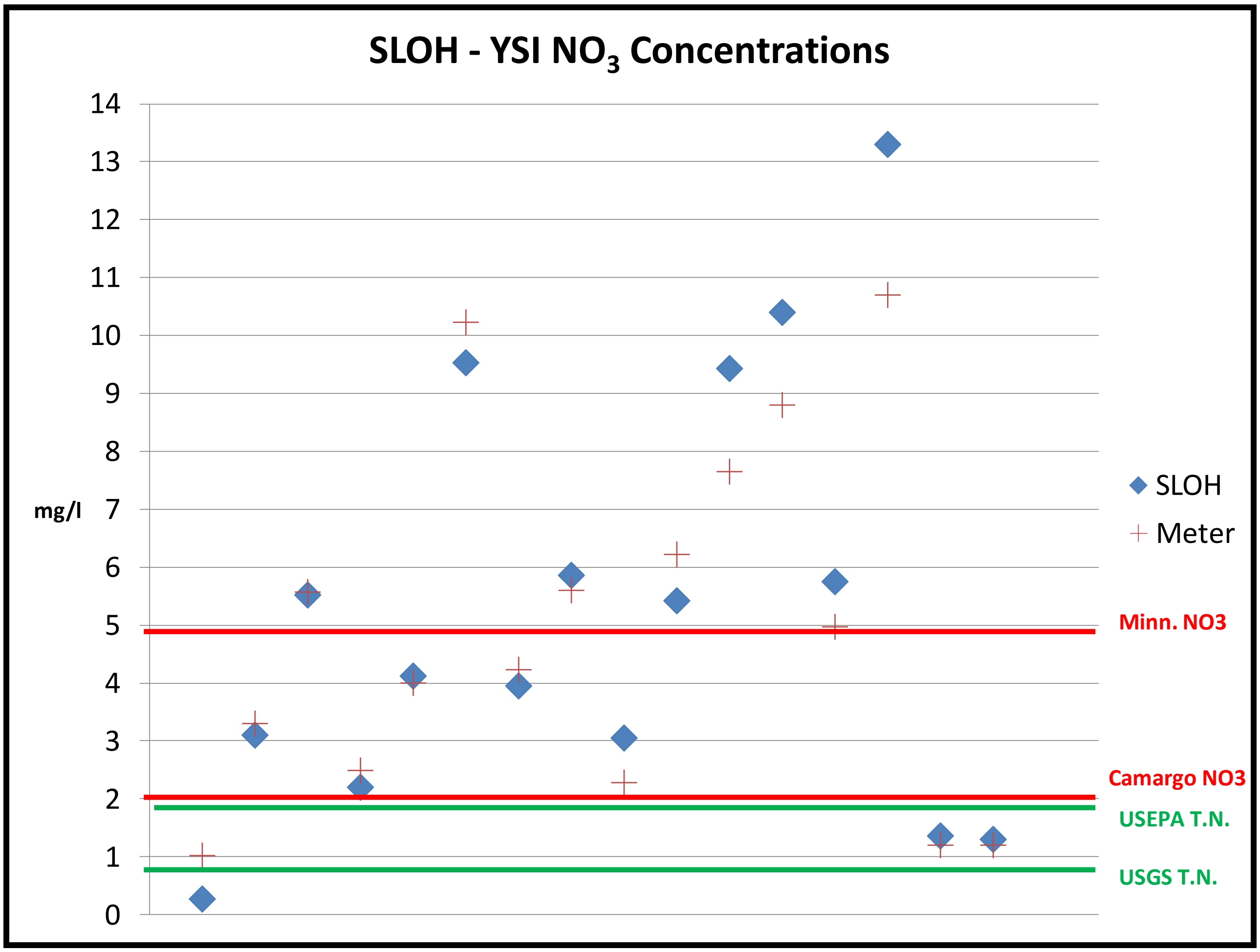
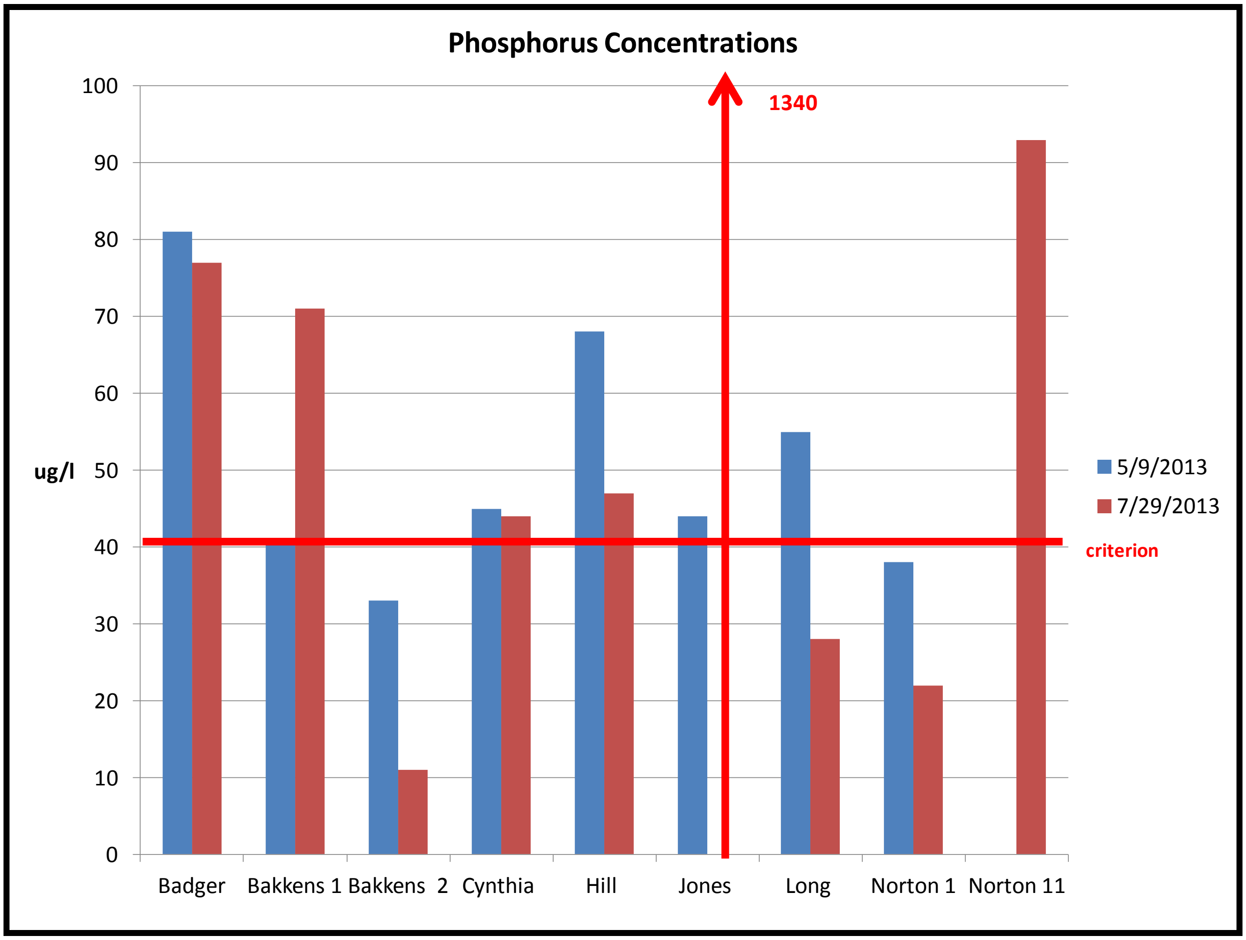
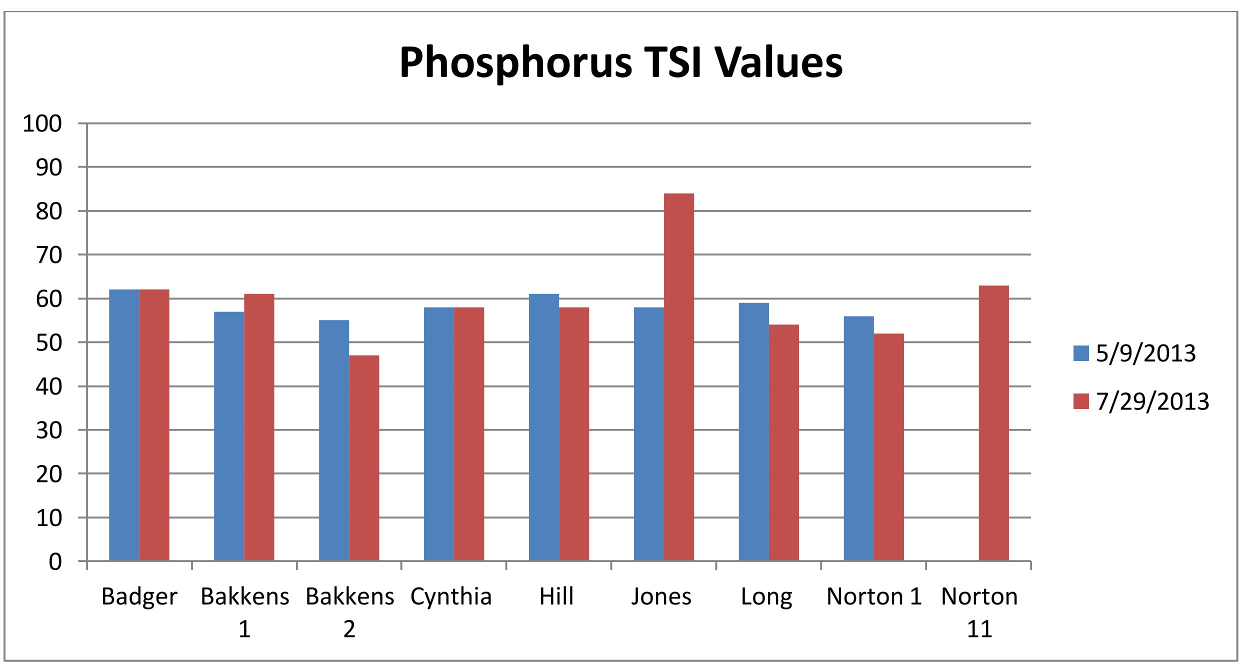


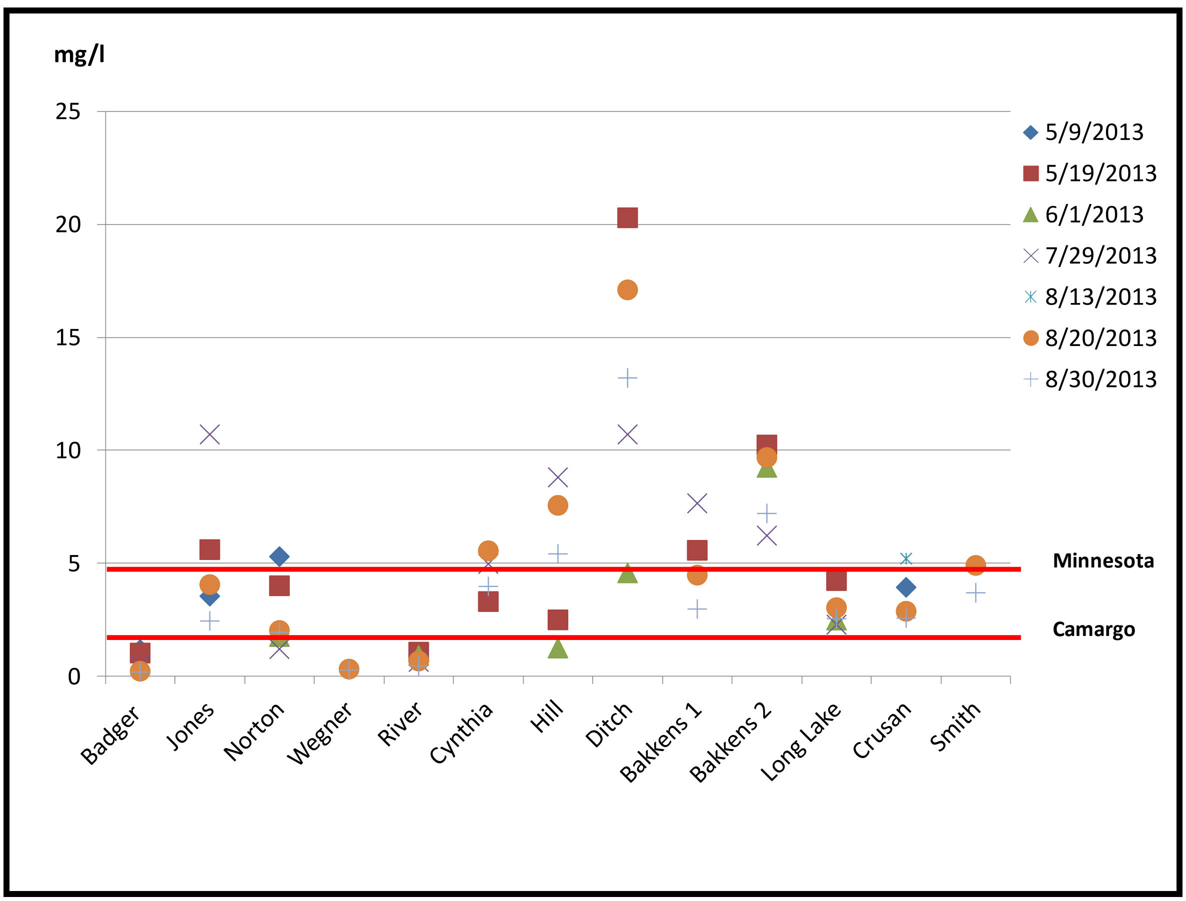
Figure 21: Phosphorus Concentrations in Lower Wisconsin River Floodplain Lakes





The focus on phosphorus in floodplain lakes must be balanced with the potential effects of nitrogen. Fried et al. (2003) and Sullivan (2008) correlated both phosphorus and nitrogen with algae and metaphyton growths. Sullivan found a slightly higher correlation between metaphyton and nitrogen. Penich et al. (2012) found a strong correlation between nitrates and Cladophora growth. Concentrations of total nitrogen exceeding 3 mg/l are considered hypersaturated and environmental pollution is clearly linked to high nitrates in groundwater and surface water (Stanley and Maxted 2008). In this study, most of the floodplain lakes had nitrate concentrations exceeding 3 mg/l at least part of the 2013 season. Robertson et al. (2006) identified 3.46 mg/l nitrates as a threshold or breakpoint concentration when fish and invertebrate communities reflected environmental degradation. Criteria and thresholds for both total nitrogen and nitrates have been proposed but none have been adopted in Wisconsin. The USEPA recommended criterion for total nitrogen is 1.88 mg/l to control eutrophication. Robertson et al. (2008) reported a total nitrogen background concentration of 0.7 mg/l as another possible criterion. Sullivan (2008) recommended a total nitrogen criterion 0.95 mg/l to prevent excessive metaphyton growths in Mississippi River sloughs. Figure 20 highlights two proposed criterion for total nitrogen (green lines).

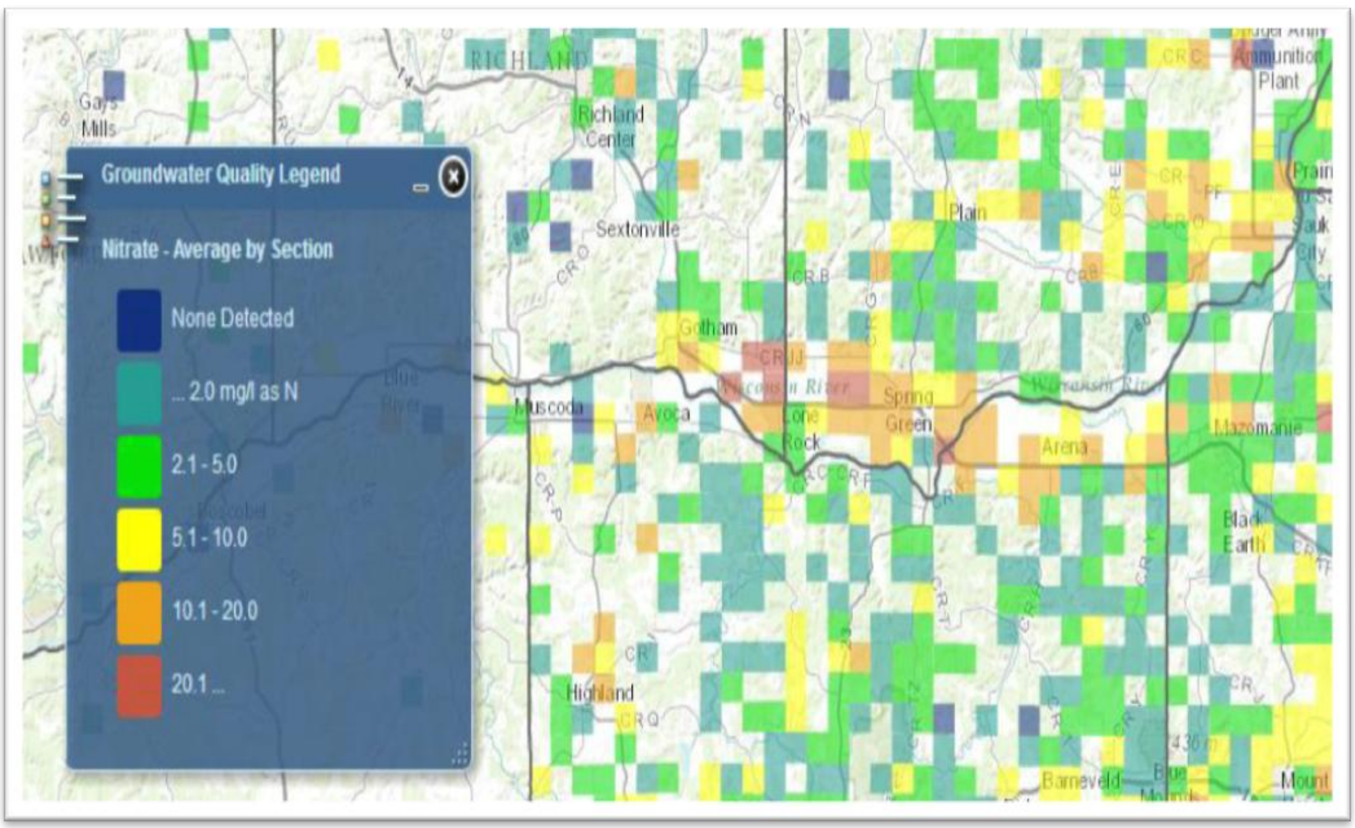
Figure 22: YSI Pro Meter Nitrate Levels Plotted with Two Proposed Thresholds for Toxicity Criterion



In addition to proposed total nitrogen criterion, nitrate criterion has been proposed due to the toxic effects on fish and other aquatic organisms. Camargo et al. (2005) recommended a criterion of 2 mg/l NO_3 to protect sensitive aquatic organisms. The Minnesota Pollution Control Agency is recommending a NO_3 limit of 3.1 mg/l for coldwater fisheries and 4.9 mg/l for other waterbodies. These proposed criteria are compared with Lower Wisconsin River floodplain lakes nitrate levels (red lines) in Figures 20 and 22. Even more conservative nitrate criterion had been proposed elsewhere. Hichen and Martin (2009) proposed 20 mg/l for acute toxic criterion, 1 mg/l chronic criterion for areas of high conservation value, 1.7 mg/l chronic criterion for waterbodies with slight to moderate environmental damage and 3.6 mg/l for degraded waterbodies. Even the proposed less conservative 4.9 mg/l NO_3-N Minnesota chronic nitrate criterion was often exceeded in Jones, Norton, Cynthia, Hill, Bakkens, Crusan and Smith lakes in 2013.

The high nitrate concentrations in Spring Green and Lone Rock area sloughs is linked to heavy nutrient applications on Pleistocene terrace coarse sandy soils. High nitrate levels in sand terrace groundwater were already identified by the UW Stevens Point Center for Watershed Science and Education in Figure 23. The higher level well concentrations are highlighted in yellow (5.1 – 10 mg/l), orange (10.1 – 20 mg/l) and red (20.1 + mg/l) and these areas lie in close proximity to the degraded floodplain lakes.

Figure 23: Nitrate Contamination in the Spring Green and Lone Rock Areas



The specific mechanism for the observed water quality declines in the Lower Wisconsin River floodplains lakes is unclear but may have mirrored modern agricultural changes including loss of CRP lands, increased corn production and increased numbers of animal units linked to larger scale farms. Large-scale farming on the Pleistocene terrace is environmentally risky due to nutrient leaching in the coarse sandy soils. Novak (2000) and Nair et al. (2004) reported substantial phosphorus buildup and ultimate leaching into a shallow aquifer beneath sandy soils. WDNR and UW Madison Extension had identified the sand terrace as highly susceptible to contaminated groundwater in 1989. Recent well testing has demonstrated that nitrate levels have been increasing well beyond the Drinking Water Standard (10 mg/l) and this trend has coincided with the floodplain lakes water quality declines.

Two options for reducing the pollution include effective nutrient management and establishing conservation buffers. The former is required if local governments adopt Farmland Preservation Zoning and tax credits are provided. However, the Town of Spring Green has not adopted the ordinance along with the required nutrient management plans. Another issue is the use of the Pleistocene terrace for liquid manure waste disposal and it is imported from outside the watershed and township. In early September 2013, at least 55 large ~ 8,000 gallon tankers hauled liquid manure from outside the township for application on 70 acres of recently harvested cropland close to Norton Slough and private wells. This situation has the potential to saturate sandy soils with phosphorus followed by leaching into the shallow groundwater table. USEPA (2003) recommends that nutrient management planning is needed if agriculture occurs in close proximity to surface waters (floodplain lakes), where well drained soils can result in leaching (Pleistocene terrace sands) and where the groundwater lies close to the surface (Pleistocene terrace). The other nutrient management option is to purchase conservation easements or fee title to establish buffers around environmentally sensitive floodplain lakes and private wells. Buffers can reduce nitrates in groundwater from 65 – 100% under the right conditions (Ranalli and Macalady 2010). Buffers often range from 7 - 100 meters in width (Mayer et al. 2006). But in the case of the Lower Wisconsin State Riverway, actual buffer widths should be based on predicted nitrate and phosphorus treatment. Wider buffers ($\geq$ 100 m) are likely needed due to the sandy soils where leaching is excessive and de-nitrification is minimal.

The goals for protecting the Lower Wisconsin State Riverway may be undermined by a clash of public policies. From 1995 to 2012, current farming practices in the Town of Spring Green had been heavily subsidized under the Farm Bill (nearly $18,000,000, Environmental Working Group 2013). Property taxes may also increase when floodplain lake riparian areas are managed as conservation buffer establishment.

The continued water quality decline in the Lower Wisconsin State Riverway cutoff channel lakes poses a significant threat to this unique large river ecosystem and the rare and popular sport fish populations that otherwise thrive in these habitats. Further declines could alter habitats more favorable for nuisance species such as common carp (Bajer and Sorenson 2010). Wisconsin’s Public Trust Doctrine requires that these unique resources are restored and protected.

Liquid manure import to the Pleistocene terrace sands, September 2013

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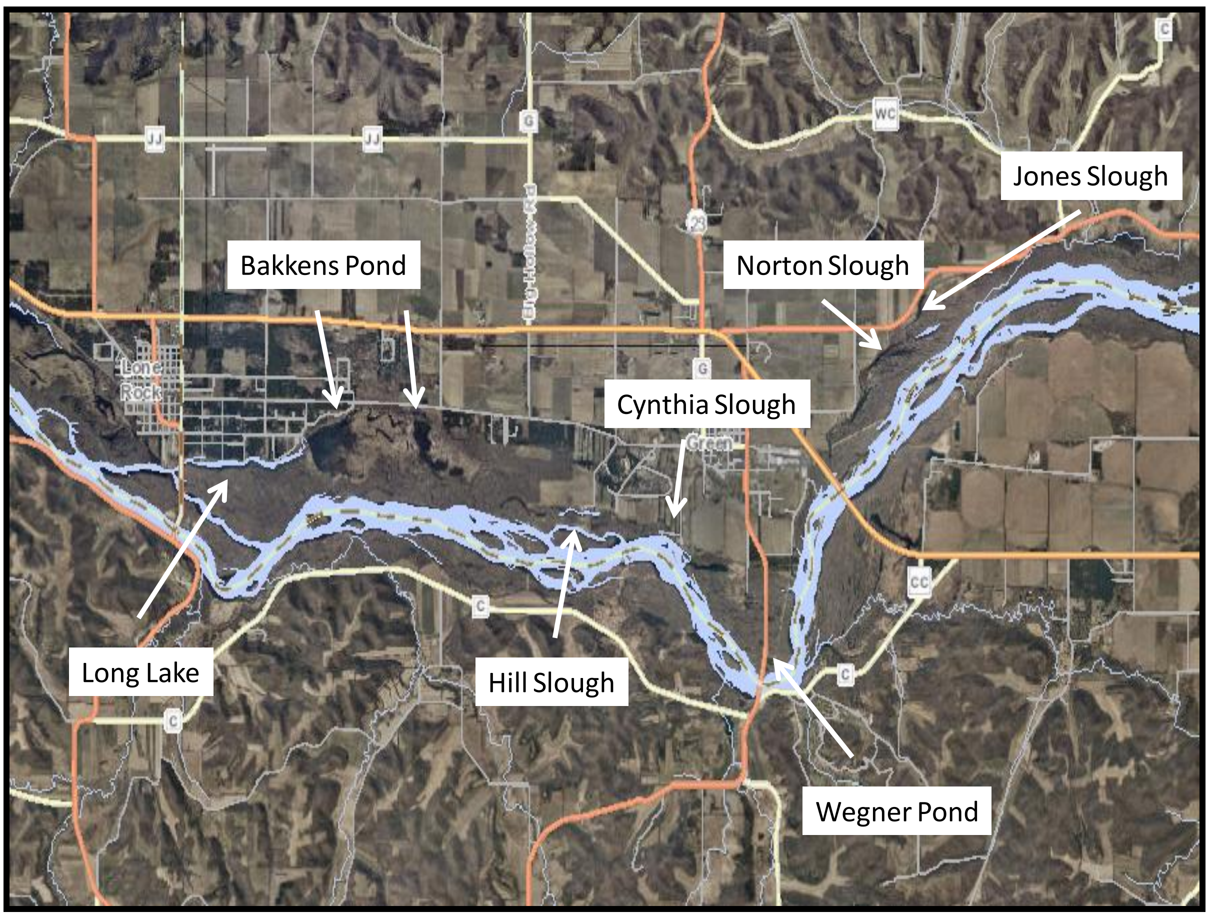

Field Temperature data (C):

Site	5/9/2013	5/19/2013	6/1/2013	7/29/2013	8/13/2013	8/20/2013	8/30/2013
Badger	17	19.5		19.8		20.9	22
Bakkens 1		21.2		19		21.6	24.9
Bakkens 2		16.2	21.1	17.3		16.3	17.7
Ditch		13.3	23.5	20.8		18.5	
Crusan					24.6	21.6	25.6
Cynthia		18.6		24.2		26.7	26.8
Hill		17.1	22	21.1		20.9	23.2
Jones	16.6	20.6		16.4		25.2	23.2
Long		19.4	24.2			21.4	24.4
Norton	18.2	21.3	24.4	18.3		22.5	22
Smith					20.4	21.3	25.3
"Wegner "						26	27.2
Wisconsin R		17.5	21.4	25.8		24.8	26.4

Field D.O. data (mg/l):

Site	5/9/2013	5/19/2013	6/1/2013	7/29/2013	8/13/2013	8/20/2013	8/30/2013
Badger	3.6	6.8		12.5		10	4.6
Bakkens 1		15.7		13.9		5.8	3.4
Bakkens 2		12.6	16.4	11.6		6.8	6.9
Ditch		4.1	10.7	12.4		5.8	
Crusan					5.2	6.4	4.7
Cynthia		7.5		11.04		18.1	10.8
Hill		9.3	10.9	13.6		9.7	7.1
Jones	5.3	15.5		4.6		12.6	6.3
Long		8.7	8.4	8.7		8.2	7.9
Norton	10.7	14.9	10.4	16.7		9.7	5.6
Smith					10.7	11.4	7.3
"Wegner "						7.8	7.5
Wisconsin R			10.3	8.6		8.8	8.2

Field pH (su):

Site	5/9/2013	5/19/2013	6/1/2013	7/29/2013	8/20/2013	8/30/2013
Badger	6.86			8.20	7.78	
Bakkens 1		10.07		9.03	7.59	6.71
Bakkens 2		8.82	8.70	8.52	7.75	8.14
Ditch		7.30		9.04	7.75	
Crusan					6.70	7.12
Cynthia		7.25		8.71	8.87	8.73
Hill		8.10	8.10	9.23	8.70	8.21
Jones	7.6	8.78		7.80	8.75	7.58
Long		8.40	8.21	8.10	7.86	8.15
Norton	7.68	9.08	8.20	9.10	8.10	7.70
Smith					8.60	8.30
"Wegner "					8.03	8.20
Wisconsin R		8.36	8.08	8.79	8.10	8.90

Field Specific Conductance (uS/cm):

Site	5/9/2013	5/19/2013	6/1/2013	7/29/2013	8/20/2013	8/30/2013
Badger	426	461		546	548	564
Bakkens 1		362		484	499	441
Bakkens 2		433	421	373	442	508
Ditch		569		408	540	
Crusan					335	324
Cynthia		342		422	501	448
Hill		220	189	372	354	344
Jones	345	441		473	410	406
Long		326	308	348	368	361
Norton	392	383	409	371	412	425
Smith					352	352
"Wegner "					249	245
Wisconsin R		190	163	214	255	241

Water clarity (cm):

Site	5/9/2013	5/19/2013	6/1/2013	7/29/2013	8/13/2013	8/20/2013	8/30/2013
Badger	120	120		120		120	120
Bakkens 1		120		120			120
Bakkens 2		120	103	120		120	120
Ditch				120		120	120
Crusan					120	120	120
Cynthia		64		88		120	120
Hill		65	90	110		120	120
Jones	120	83		92		120	120
Long		75	83	120		120	120
Norton	120	120	99	120		120	120
Smith					120	120	120
"Wegner "						120	120
Wisconsin R		57	56	56		47	45

Metaphyton Cover (0 – 5 max):

Site	5/9/2013	5/19/2013	6/1/2013	7/29/2013	8/13/2013	8/20/2013	8/30/2013
Badger	0	1		4		4	4
Bakkens 1		3		5		5	5
Bakkens 2		3	3	5		5	5
Ditch		5	0	5		5	5
Crusan					3	4	3
Cynthia		0		1		1	1
Hill		1	0	3		2	3
Jones	1	3		5		5	5
Long		0	1	1		2	2
Norton	1	4	3	4		5	4
Smith					2	3	2
"Wegner "						0	0

Starhead topminnow reports (presence/absence):

Site	5/9/2013	5/19/2013	6/1/2013	7/29/2013	8/13/2013	8/20/2013	8/30/2013
Badger						X	X
Bakkens 1							
Bakkens 2				X		X	X
Ditch							
Crusan					X	X	X
Cynthia							
Hill			X	X		X	X
Jones						X	X
Long				X		X	
Norton						X	
Smith						X	
"Wagner "						X	X

Restoring Lower Wisconsin State Riverway Oxbow Lakes

Phase 2 Diagnostic and Feasibility Study

Jones Slough 9/9/15

WDNR Lakes Planning Grants Projects

Sponsored by

River Alliance of Wisconsin

In Cooperation with WDNR Water Quality Section

Prepared by

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April 2016

Abstract

The habitat of oxbow lakes is often dependent on the quantity and quality of the groundwater discharging to them. The Phase 1 study had demonstrated that increasing nitrogen concentrations in the oxbows and subsequent habitat degradation was most evident where the river terrace groundwater discharges into the lakes. A major goal of the hydrogeological study was to measure the quality of the groundwater water discharging into the lakes. This effort was coordinated with the continuing study of the of the lakes' surface water quality and biota.

The extensive School Forest, adjacent to Long Lake, demonstrates that conservation buffers is an option for reducing groundwater and oxbow pollution. However, the data also indicate that extensive buffers are likely needed along with nutrient application reductions. The highest groundwater nitrate concentrations coincided with greatest oxbow environmental degradation in the forms of toxic nitrate concentrations, extensive free floating plant (FFP) cover, and anoxia. Toxic nitrate concentrations occurred where FFP densities were lower, along with associated plant nutrient uptake while dense FFP resulted in anoxia and internal phosphorus loading.

Introduction

Below the Prairie du Sac dam, a large river ecosystem known as the Lower Wisconsin State Riverway (LWSR) flows within a floodplain that remains in a relatively natural state. The Lower Wisconsin River is considered one of the highest-quality large rivers remaining in the Midwestern U.S. (Lyons 2005). This braided river channel ecosystem supports 98 fish species and very high biodiversity in part due to a vast network of oxbow lakes and sloughs that support numerous species adapted to slow water currents. The existence of these unique ecosystem features and high biodiversity are reasons why the State Riverway has been recommended for nomination as a Ramsar Convention wetland of international importance.

The LWSR hydrology is greatly influenced by a "dynamic aquifer" across the Driftless Area (Pfeiffer et al. 2006). This massive groundwater system is an important environmental factor that sustains this large river ecosystem but also softened the historic impacts when industrial pollution degraded upstream sections of the river prior to implementation of the Clean Water Act (Marshall 2009). By the early 1980s, over 95% of the organic pollutants discharged to the river during the 1970s and before were eliminated (WDNR 1992).

Oxbow lakes and sloughs provide important habitats for species with ecological niches that avoid fast currents. Offchannel waterbodies also function as nurseries for many riverine fishes (Amoros 2001, Killgore and Miller 1995). Oxbow lakes had been degraded or completed eliminated due to floodplain aggradation and other anthropogenic factors along many Wisconsin rivers (Marshall 2012). However, rare fish species such as the State Endangered starhead topminnow (*Fundulus dispar*), State Special Concern mud darter (*Etheostoma asprigene*), State Special Concern pirate perch (*Aphredoderus sayanus*), State Special Concern weed shiner (*Notropis texanus*), pugnose minnow (*Opsopoeodus emiliae*) and lake chubsucker (*Erimyzon sucetta*) still thrive within the many LWSR oxbows and sloughs.

They are often found where the Pleistocene river terrace aquifer discharges to oxbows lakes. Many of these oxbows are limnological definition of spring lakes and therefore should display excellent water quality given the hydrological influence of groundwater. However, beginning in the late 2000s, water quality declined rapidly in oxbows that had displayed near pristine conditions just a decade earlier (Marshall 2013 Phase 1 report). This report completes Phase 2 of the Diagnostic and Feasibility Study that expanded water quality monitoring beyond the oxbows to include the primary water source for the degraded oxbows, the groundwater discharge from the adjacent river terrace aquifer. The study was designed to better understand the groundwater dynamics as the primary nutrient delivery system to the oxbows and what measures, such as conservation buffers, can be recommended to restore and protect these environmentally sensitive off channel habitats.

Methods

Four oxbow lakes (Jones Slough, Norton Slough, Bakkens Pond and Long Lake) and the nine up-gradient clustered monitoring wells installed across the terrace and floodplain were sampled monthly during the open water seasons in 2014 and 2015. Jones Slough (WBIC 1247300), Norton Slough (WBIC 1247100) and Bakkens Pond (1236900) were selected for further monitoring since these waterbodies displayed the most severe water quality in 2013 (Phase 1 Diagnostic and Feasibility Study). Long Lake (WBIC 1236600) functioned as a reference site since it displays far less productivity and is buffered by the School Forest. A recent 12-acre conservation buffer was established (2013) adjacent to Norton Slough. Crop production and nutrient additions had ended in the twelve-acre parcel in 2010. A cluster of four monitoring wells was installed between the oxbow lake and buffer to detect potential changes in nitrate concentrations over time. Norton Slough was the only site where riparian land use changes had occurred recently.

Nitrate, ammonium and chloride were measured in the field using a YSI Pro Plus meter that was calibrated using 1 mg/l and 100 mg/l NO_3, 1mg/l and 100 mg/l NH_3, and 10 mg/l and 1000 mg/l chloride standards. Chlorides were collected for use as potential groundwater flow tracers, primarily from highway road salt applications. Water samples were collected separately WDNR and consultants but all were submitted to the State Laboratory of Hygiene (SLOH) for nitrate quality assurance, total nitrogen, chlorophyll a and total phosphorus analysis. The SLOH analysis of nitrate provided quality assurance for meter nitrate measurements. A YSI DO meter or similar meter was used to measure dissolved oxygen and temperature in the oxbows and wells. The meter was air calibrated according to specifications. A YSI Model 63 meter was used to measure pH and specific conductance. A standard secchi disc was used to measure water clarity. A Hach 2100P Turbidimeter was used for oxbow water sample analysis. Free floating cover (filamentous algae and duckweeds - FFP) was estimated using the following scale: 0 = no FFP present, 1 = 1 – 20 % FFP cover, 2 = 21 – 40 % FFP cover, 3 = 41 – 60 % FFP cover, 4 = 61 – 80 % FFP cover, and 5 = 81 – 100 % FFP cover. In late June 2015, WDNR conducted a point intercept survey on Jones Slough (84 sites) and Norton Slough (60 sites). Nearshore fish populations were sampled using towed DC electroshockers. A point intercept survey documented aquatic plant growths in Norton Slough and Jones Slough.

The hydrogeological study included water monitoring at 48 locations including: 42 wells of which 34 were installed as part of the study, two were established Town of Spring Green Flood Study monitoring wells, and six were private water supply wells monitored only for water quality. Onset pressure sensors were installed at six staff gages to monitor oxbow water levels. Soils were logged as part of the well installation and the logs provided in Appendix 1. This assessment reports on monitoring conducted from July 10, 2014 through November 4, 2015.

Study Area

The study area lakes and monitoring points are shown on Fig. 4. Nine groundwater monitoring well clusters were installed in the river terrace and floodplain to assess the aquifer that flows into Jones Slough and Norton Slough east of Spring Green and Bakkens Pond and Long Lake west of Spring Green, Sauk County. Three additional wells were installed to measure groundwater level changes in close proximity to the river which allowed determination of Wisconsin River stage elevations. Jones Slough (7.3 acres) and Norton Slough (13.6 acres) are cutoff channel oxbow lakes and both function as spring lakes (lakes with outlets but no inlets) due to the influence of the river terrace aquifer. Jones Slough has a maximum depth of 2.1 meters and Norton Slough 2.7 meters. A permanent twelve-acre conservation easement was established around the Norton Slough monitoring wells in 2014 and is a developing buffer as warm season grasses and forbs are becoming established with deep root systems. The conservation easement prohibits nutrient additions in the newly formed clean recharge buffer area. Bakkens Pond (19.1 acres) is a spring lake impoundment encompassed by a State Natural Area that begins as a series of large springs that discharge into the floodplain. The oxbow channel is impounded by two earthen dams with tin whistle structures designed to expand waterfowl habitat. Long Lake (48 acres) lies just downstream of Bakkens Pond and is classified a shallow lowland drainage lake. It is impounded by a low rock dam near the HW 130 bridge at the Wisconsin River. The north shore of Long Lake is heavily developed with seasonal and year round residences. Long Lake is also located within the Sauk County School Forest that has functioned as an extensive conservation buffer for decades. These four waterbodies and aquifer lie within the 136 square mile Bear Creek Watershed. While this watershed drains a significant area, the direct surface water drainage area around the floodplain lakes is not significant with the river terrace aquifer functioning as the primary water source except during infrequent periods of high river stages.

Findings

Groundwater Study

Historic hydrogeological information and past study in the region was reviewed and a conceptual groundwater flow model was formulated that included recharge in the bedrock highland to the north and in the adjacent Wisconsin River outwash terrace with flow to the south with discharge at the lakes and Wisconsin River. Figure 1 shows the general groundwater flow in the lake study area and Figure 2 is a cross-section showing the conceptual groundwater flow more locally. Figure 3 shows the conceptual groundwater flow into an oxbow lake. Groundwater modeling of Sauk County (Gotkowitz et al., 2005) indicates the hydraulic conductivity of the outwash and alluvial deposits is 162 ft./d with a recharge of 10.2 inches per year while the underlying sandstone bedrock and adjacent upland area has a hydraulic conductivity of 8 ft./d and a recharge rate of 5.2 inches per year. Continuous soil sampling at each well cluster location confirmed the presence of medium well sorted sand including some fine sand, coarse sand and gravel. No significant clay or organic material was observed in the saturated outwash terrace deposits nor did indications of staining, odor or mottling that would indicate the presence of reducing conditions. The near-surface floodplain deposits adjacent Norton Slough however did contain finer soils, organics and indicators of reducing conditions such as mottling and iron depletion.

A compilation of all the hydrogeological monitoring results are provided as a spreadsheet in Appendix 2. A groundwater flow map constructed for the Norton and Jones Sloughs area is presented as Fig. 5 and the flow map for the Bakkens Pond and Long Lake area is presented as Fig. 6. The Wisconsin River stage elevations used in the groundwater maps in the Jones/Norton area were interpolated and extrapolated from the water elevations measured at WR (Wisconsin River at HW 14) and WRFP (Wisconsin River adjacent Norton Slough) monitoring points. The river gradient was calculated to be 0.00033 ft./ft. (1.75 ft./mile). The Wisconsin River stage elevations used in the groundwater maps in the Bakkens Pond/Long Lake area were interpolated from the WRLR (Wisconsin River at HW 130 near Lone Rock) and WR (Wisconsin River at HW 14) monitoring points. The river gradient was calculated to be 0.00025 ft./ft. (1.33 ft./mile).

The groundwater maps demonstrate that the lakes are receiving groundwater discharge from the north. The recharge areas for the southward flowing groundwater includes significant outwash terrace agricultural development and associated nutrient additions, both as chemical fertilizer and manure. The three dams on the Bakkens Pond and Long Lake impoundments flatten the groundwater gradients in the downstream areas resulting in slower groundwater flow velocities in these floodplain areas. Table 1 displays the hydraulic gradients and calculated seepage velocities for each lake area. The location and width of the agricultural areas up-gradient from each of the lakes was determined from aerial photos. The distance of the agricultural area up-gradient from each lake was used with the groundwater seepage velocities to calculate the travel times necessary for nutrients to flow to both the lakes and the adjacent Wisconsin River location down-gradient from each lake. The nitrate component of the agricultural-derived nutrients is anticipated to act in a conservatively with negligible retardation or degradation expected due to the geochemical aquifer conditions. The results are provided in Table 2.

The time needed for nutrients to travel from the agricultural areas increases for each lake starting at Jones Slough in the eastern portion of the study area to Long Lake in the west. The longer travel time of 15.6 years calculated for Long Lake indicates the nitrate impacts from the up-gradient agricultural areas may continue to build through time.

Open water in Jones Slough during winter due to groundwater discharge. The observations of lake ice cover in December of 2014 suggest groundwater discharge and associated nutrients may not be uniform along the edge of the discharging aquifer. Below are winter photos of the other lakes demonstrating variable groundwater discharges.

Table 1: Hydraulic Gradients and Seepage Values

Oxbow lake	Location	Hydraulic Gradient (i) ft/ft	Seepage Velocity (Vs)* ft/day
Jones Slough	Terrace	0.00126	0.82
Jones Slough	Floodplain	0.001	0.65
Norton Slough	Terrace	0.0012	0.8
Norton Slough	Floodplain	0.0015	1.0
Bakkens Pond	Terrace	0.003	2.0
Bakkens Pond	Floodplain	0.0007	0.45
Long Lake	Terrace	0.002	1.3
Long Lake	Floodplain	0.0002	0.13

*Seepage Velocity $(Vs) = (K)(i/\phi)$ K = outwash & alluvial hydraulic conductivity of 162 ft/day. ϕ = porosity of 0.25.

Table 2: Agricultural Nutrient Travel Times

Lake	Distance from Ag Use Area, ft.	Lake Distance To Wis. Rr. ft.	V_S* Terrace ft./d	V_S* Floodplain ft./d	Travel Time to Lake yrs.	Travel Time to Wis. Rr. yrs. ***	N-S Width of Ag. Use Area, ft.
Jones Slough	350	2200	0.82	0.65	1.2	10.5	1600
Norton Slough	1200	2000	0.8	1.0	4.1	9.1	3700
	150**	"	"	""	0.51	6.1	"
Bakkens Pond	4400	4400	1.94	0.45	6.2	33	8500
Long Lake	7400	3700	1.3	0.13	15.6	94	12,700

* Seepage Velocities (V_S) calculated with K = 162 ft/d and porosity (ϕ) = 0.25

** 150 ft. distance used for time before 2010 when Jones field area was converted to prairie

*** Travel time to Wis. Rr. Includes both Terrace and Floodplain Travel Times

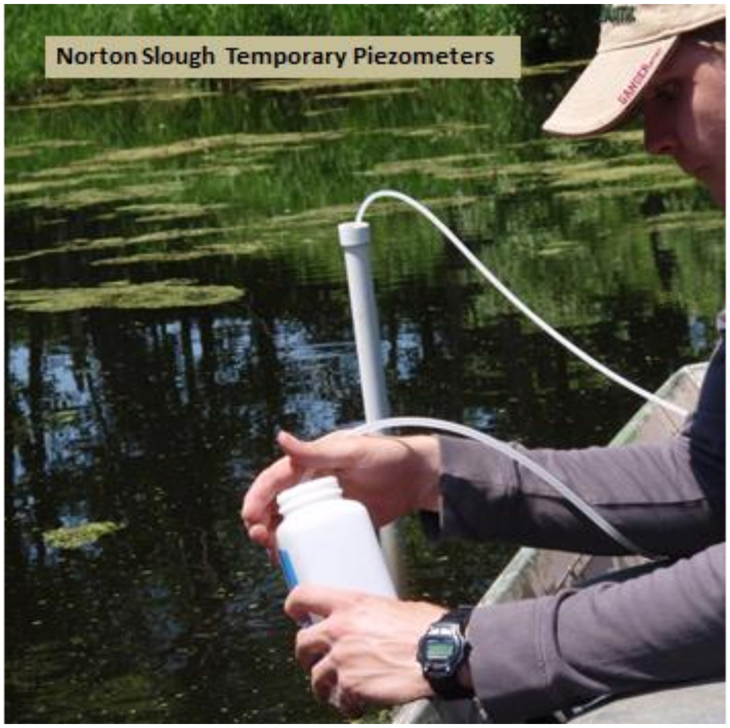
On June 10, 2015 a transect survey of Norton Slough sampled groundwater from six temporary piezometers (Table 3). The survey showed the groundwater discharging into Norton Slough with variations in nitrate of 0.9 to 11.1 mg/l NO3-N. The highest nitrate concentrations were adjacent the north shore with the greatest groundwater discharge during river low flow. A water column temperature profile at Norton Slough on the same day showed the lowest water temperatures occurred at the bottom of the water column in areas that corresponded to the highest groundwater nitrate concentrations found in the temporary piezometer transect.

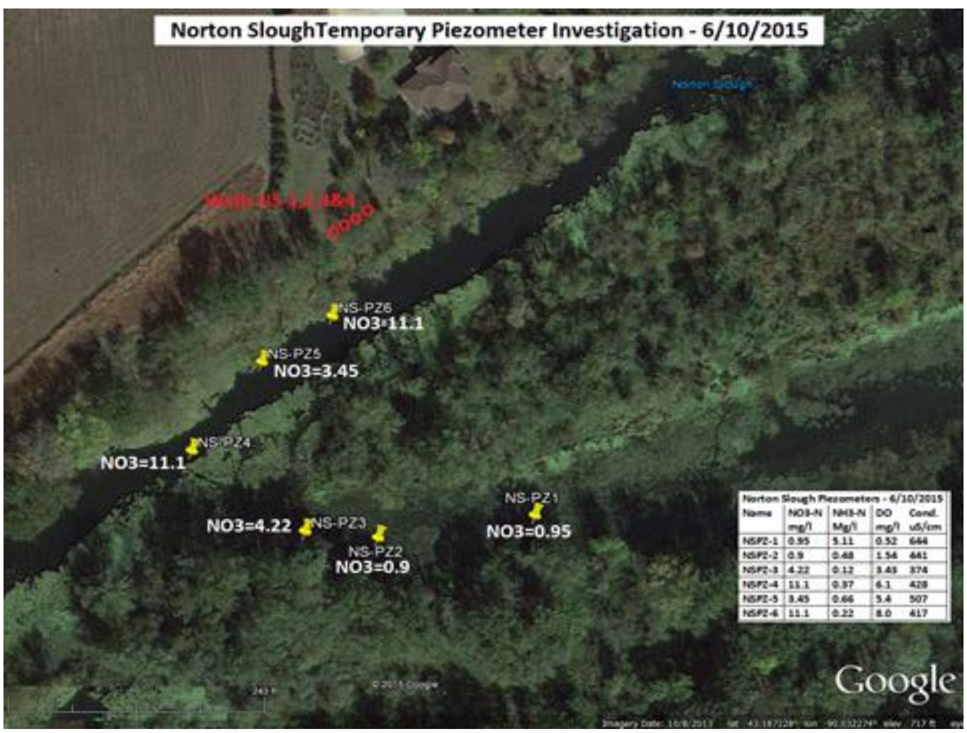

Norton Slough 12/12/2014
Moderate Open Water – Less NO3


Bakken's Pond, 12/12/2014
Most Open Water – Highest NO3

Table 3: Norton Slough Water Quality Profiles Near Temporary Piezometers

Site	Depth below water surface (m)	Nearby Temp. Peiz. NO3-N mg/l	Slough water temperature C
1	Surface 0.7	0.95	26.1 24.5
2	Surface 0.5	0.9	27 24.8
3	Surface 0.5	4.22	26.3 24.2
4	Surface 1	3.45	27.3 19.2
5	Surface 1 2	11.1	26.6 19 13.5
6	Surface 1 2	11.1	26.5 19 13.5





Norton Slough Piezometers - 6/10/2015				
Name	NO3-N mg/l	NH3-N Mg/l	DO mg/l	Cond. uS/cm
NSPZ-1	0.95	5.11	0.52	644
NSPZ-2	0.9	0.48	1.54	441
NSPZ-3	4.22	0.12	3.43	374
NSPZ-4	11.1	0.37	6.1	428
NSPZ-5	3.45	0.66	5.4	507
NSPZ-6	11.1	0.22	8.0	417

A cross-section of the study results at Jones Slough is presented as Fig. 7. A cross-section of the study results at Norton Slough is presented as Fig. 8. A cross-section of the study results at Bakkens Pond is presented as Fig. 9 and a cross-section of the study results at Long Lake is presented as Fig. 10. Jones Slough, Norton Slough and Bakkens Pond are over two feet higher than the Wisconsin River elevation. Long Lake is less than one foot higher than the river and may partly explain the classification differences in the lakes.

The results show that the hydraulic gradients adjacent each of the lakes indicate that groundwater is recharging from the north. The groundwater discharging into all four lakes contains significant concentrations of nitrate (Table 4). The nitrate concentration of the groundwater discharging into Long Lake was the lowest of the four lakes. This is most likely due to the greater width of non-agricultural buffer between the lake and the agricultural field to the north. The higher nitrate concentrations at the greatest depths monitored most likely represents water flowing from a greater distance to the north were the agricultural cropping is most intense. Some of the deeper water may be flowing under the lakes and ultimately discharge into the Wisconsin River. The shallow wells (JS1, NS1, LL1, BP1) had lower nitrate concentrations than in deeper wells since the shallow groundwater represented clean recharge within the variable buffer zones adjacent to the oxbow lakes.

Concentrations of phosphorus were measurable but were relatively modest (< 50 ug/l) when compared with nitrate concentrations. Ammonium levels were high in deeper DR and PR wells that lie within groundwater flow paths of Bakkens Pond and Long Lake and at JR2 that flows toward Jones and Norton.

The buffers can be an important factor in protecting the oxbows given both the clean recharge potential and terrestrial deep rooted plant nutrient uptake. Currently, both buffer zones and sand terrace nutrient applications areas are variable among the four study lakes. Across the sand terrace, Long Lake intercepts a polluted agricultural to buffer zone linear pathway ratio of approximately 3:1. In stark contrast, the Bakkens Pond agricultural to buffer recharge ratio is approximately 24:1 so that the clean recharge/terrestrial plant nutrient uptake zone is relatively short compared with polluted recharge. Jones Slough and Norton Slough have agricultural to buffer ratios of approximately 21:1 and 6:1 respectively. The ratios are surficial linear comparisons and do not represent actual more complex three dimensional and temporal changes in groundwater flow paths. However, Jones Slough and Bakkens Pond displayed the most degraded water quality and also have relatively longer polluted recharge zones compared with buffer zones.

Actual nitrogen and phosphorus loading rates to the oxbows will depend on a combination of factors including nutrient management, groundwater elevations intercepted by the oxbows, buffer dimensions, buffer vegetation types, and buffer locations relative to groundwater flow paths. The numerical groundwater modeling study (Sauk County River Planning Grant) now underway at the study area should be able to better determine potential groundwater flow paths including nutrient source and discharge locations along with more precise identification of the width of recharge buffers needed to insure discharging waters will not cause impacts to the sensitive lake habitats.

The floodplain wells adjacent to Norton Slough characterized the reduced conditions within alluvial aquifer. Nitrate concentrations were low while ammonium concentrations were high, suggesting denitrification within the wetland. Phosphorus concentrations were high while dissolved oxygen concentrations were very low.

Figure 1: General groundwater flow in the oxbow lake study area

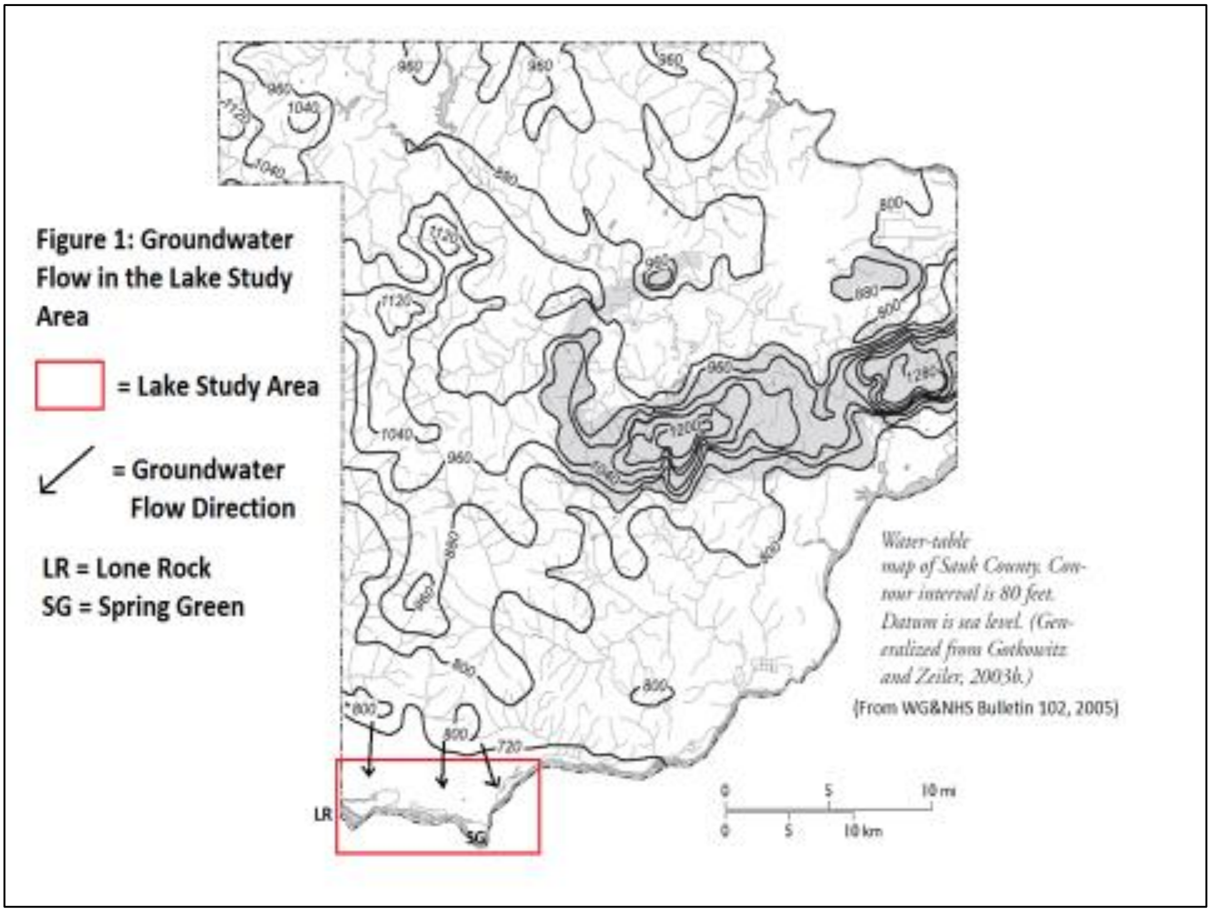


Table 4: Mean parameters concentrations from monitoring wells

Name	Sp. Cond. *u*mhos/cm	NO3-N mg/l	NH3-N mg/l	Chloride mg/l	Total Phosphorus mg/l
JS1	390	9.7	0.50	56	0.041
JS2	432	16.6	0.22	31	0.022
JS3	389	19.1	0.16	14	0.016
NS1	322	8.6	0.33	11	0.026
NS2	476	12.5	0.46	27	0.018
NS3	457	13.1	0.33	24	0.022
NS4	450	16.4	0.18	24	0.031
JW	395	12.3	0.27	21	
JP1	711	8.9	0.59	79	
JP2	441	10.7	0.50	19	
JP3	460	14.3	0.39	14	
JR1	190	3.0	0.30	7	
JR2	423	16.1	1.21	18	
JR3	395	11.8	0.161	14	
AW	387	1.0	0.16	3	
NW	329	4.6	0.87	17	
FP1	613	0.6	2.20	26	0.716
FP2	477	0.3	0.28	10	0.156
BP1	58	0.8	0.43	5	0.012
BP2	413	13.6	0.24	19	0.013
BP3	464	23.0	0.20	20	0.013
BP4	526	33.3	0.15	24	0.017
BP5	493	32.5	0.09	21	0.046
BP6	513	28.1	0.07	23	0.042
BPS	439	18.7	0.16	33	
DR1	407	6.3	0.62	53	
DR2	724	23.5	2.82	60	
DR3	533	15.6	1.86	31	
LW	570	13.2	1.50	59	
LL1	311	3.7	0.71	55	0.017
LL2	669	4.0	0.45	131	0.024
LL3	582	6.2	0.44	130	0.043
LL4	753	9.5	0.26	135	0.014
RW	371	23.2	0.20	25	
PW	281	2.3	0.22	2	
PR1	377	2.5	0.38	48	
PR2	719	13.1	0.38	107	
PR3	585	10.2	3.86	36	0.012

Figure 2: Cross-section showing the conceptual groundwater flow

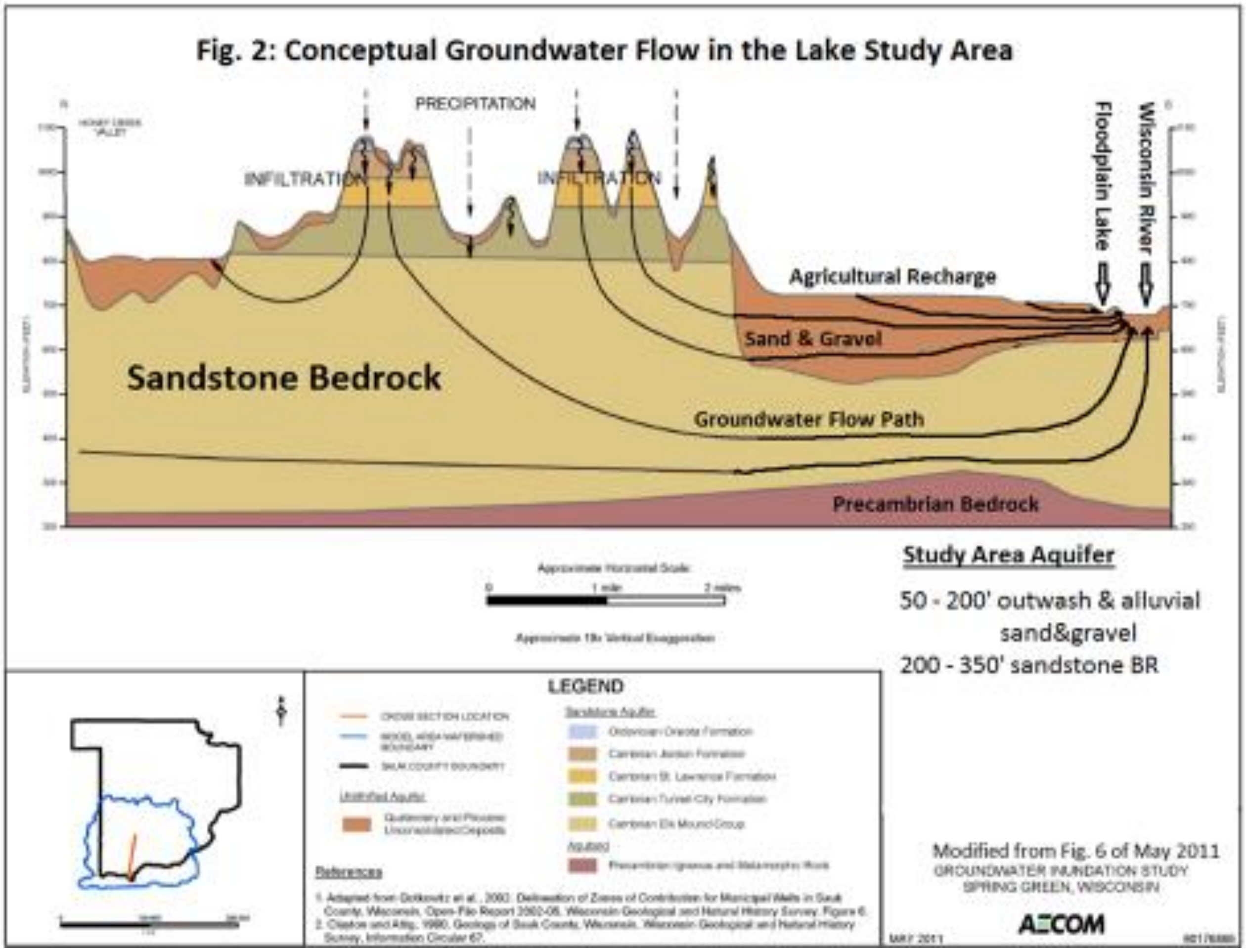


Figure 3: Conceptual groundwater flow into an oxbow lake

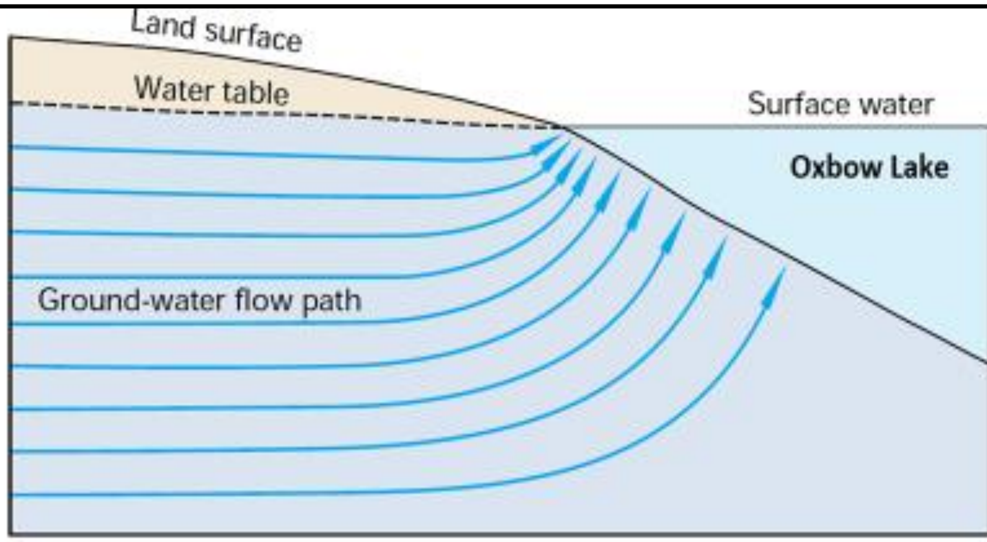


Figure 3. *Ground-water seepage into surface water usually is greatest near shore. In flow diagrams such as that shown here, the quantity of discharge is equal between any two flow lines; therefore, the closer flow lines indicate greater discharge per unit of bottom area.*

Figure 4: Study area lakes and monitoring points

Figure 5: Groundwater flow map of the Norton and Jones Sloughs area

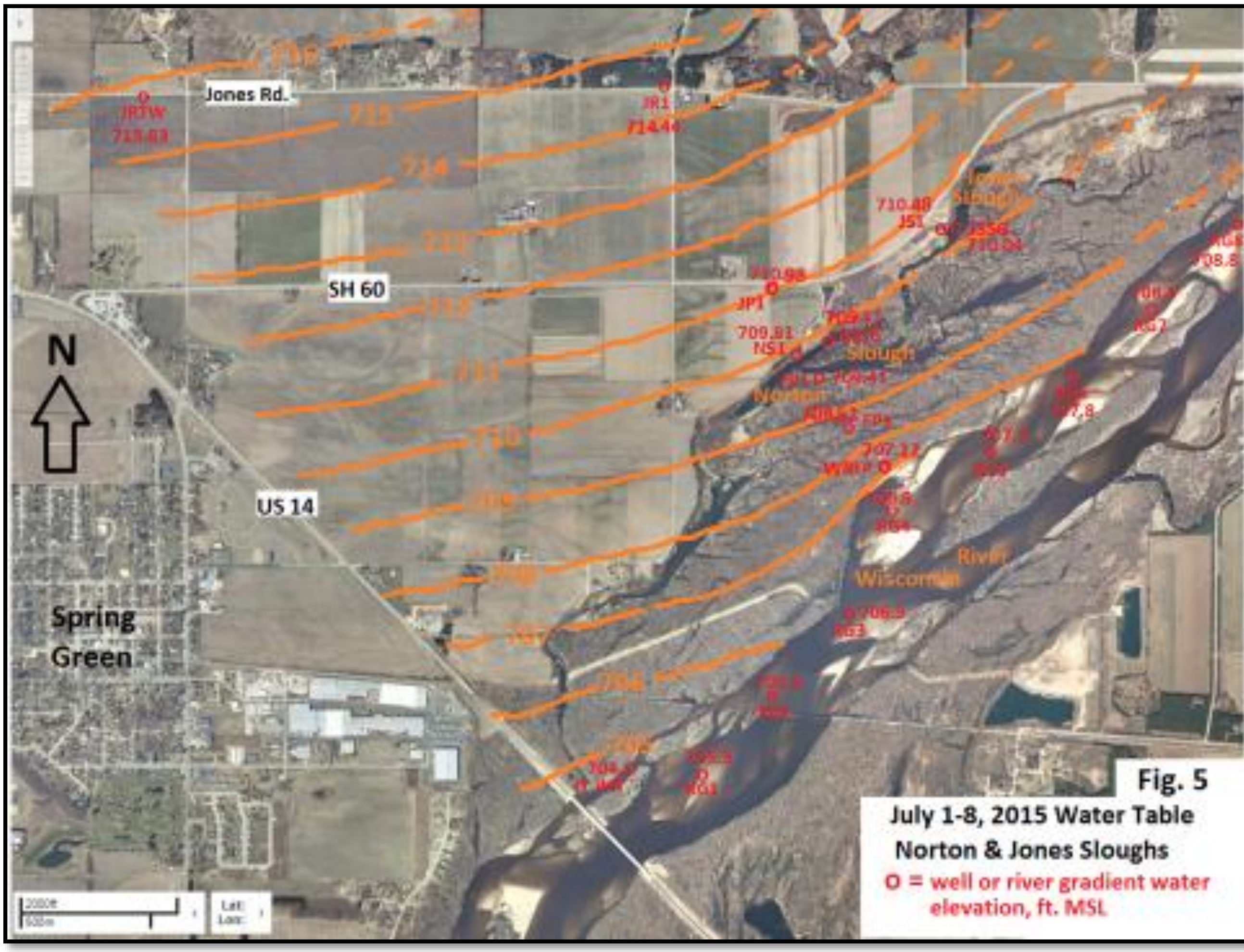


Figure 6: Groundwater flow map for the Bakkens Pond and Long Lake area

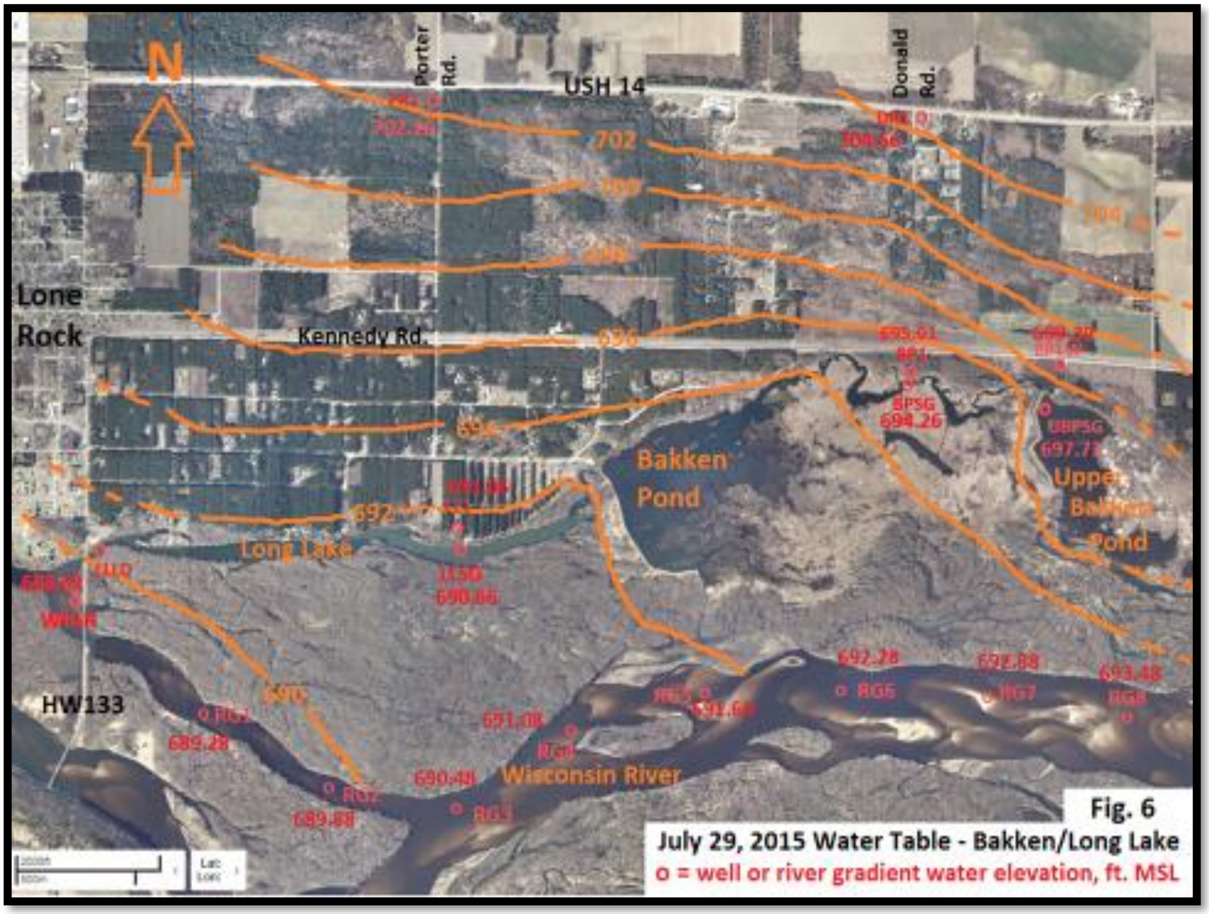


Figure 7: Elevation cross-section at Jones Slough

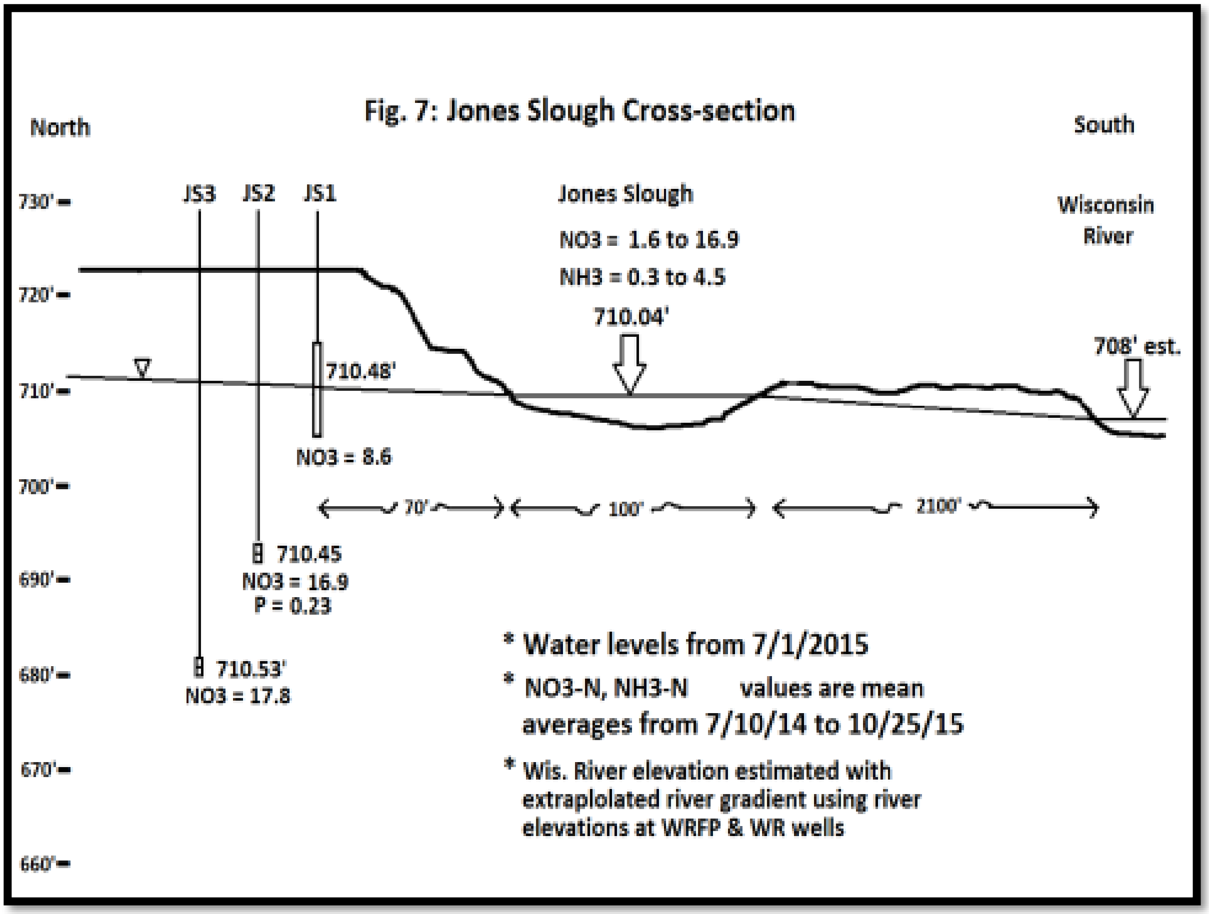


Figure 8: Elevation cross-section at Norton Slough

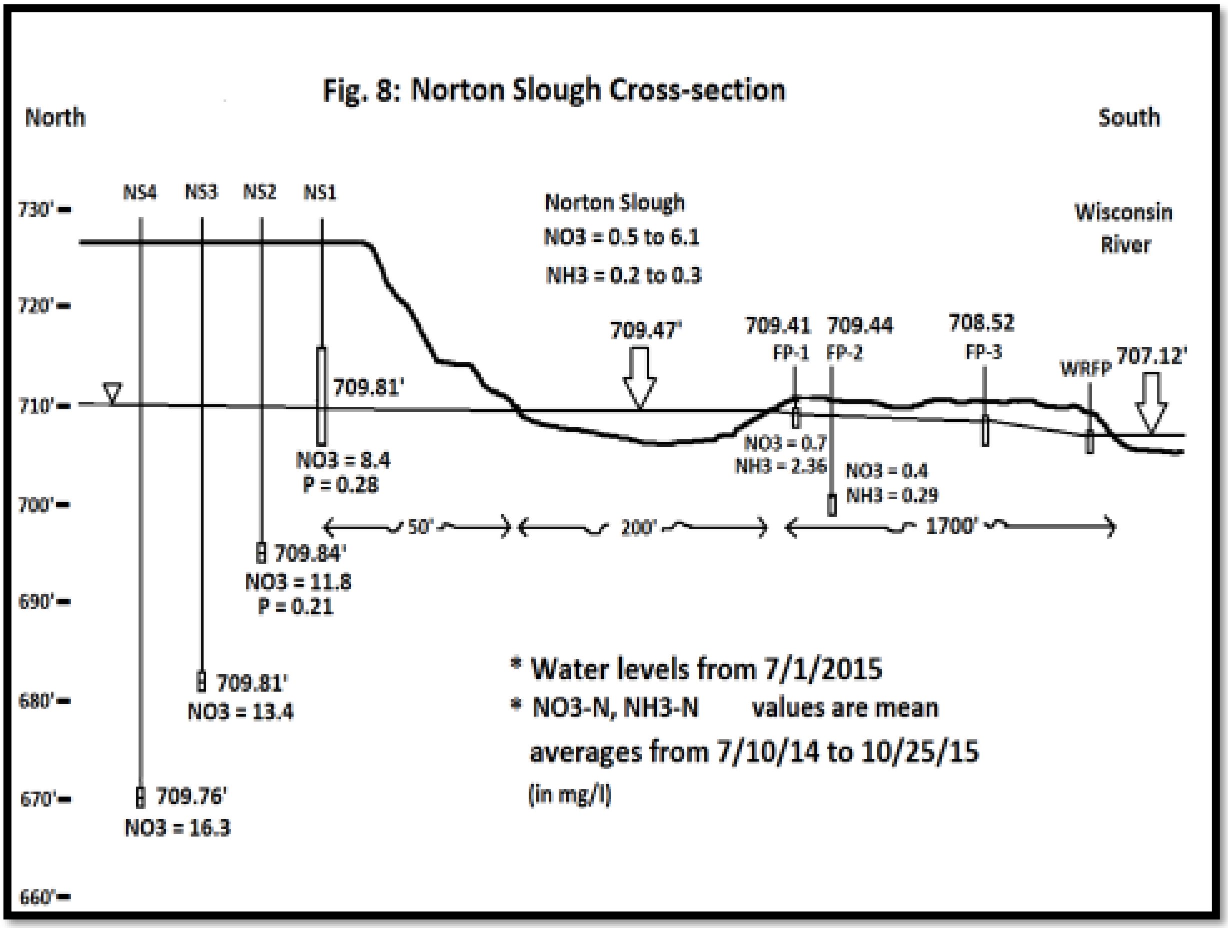


Figure 9: Elevation cross-section at Bakkens Pond

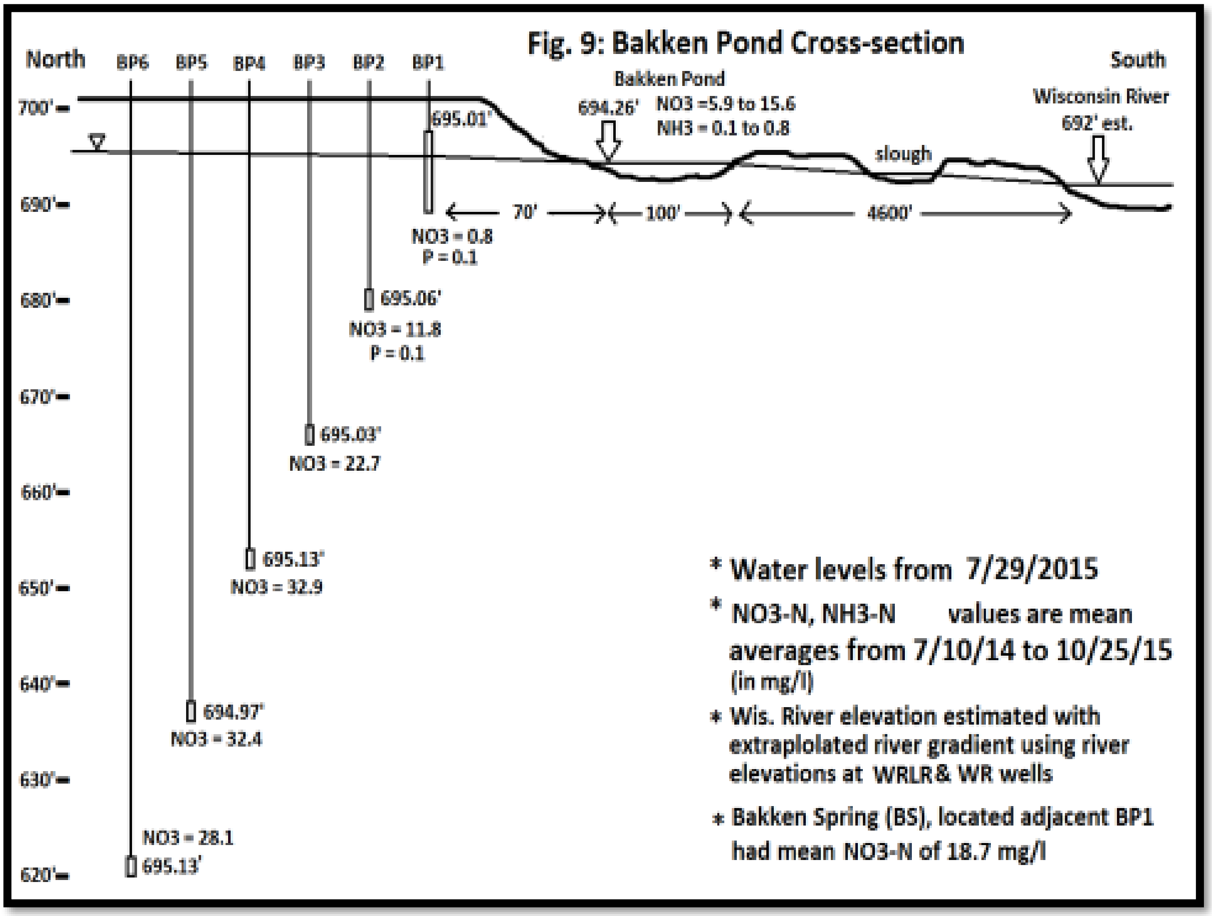
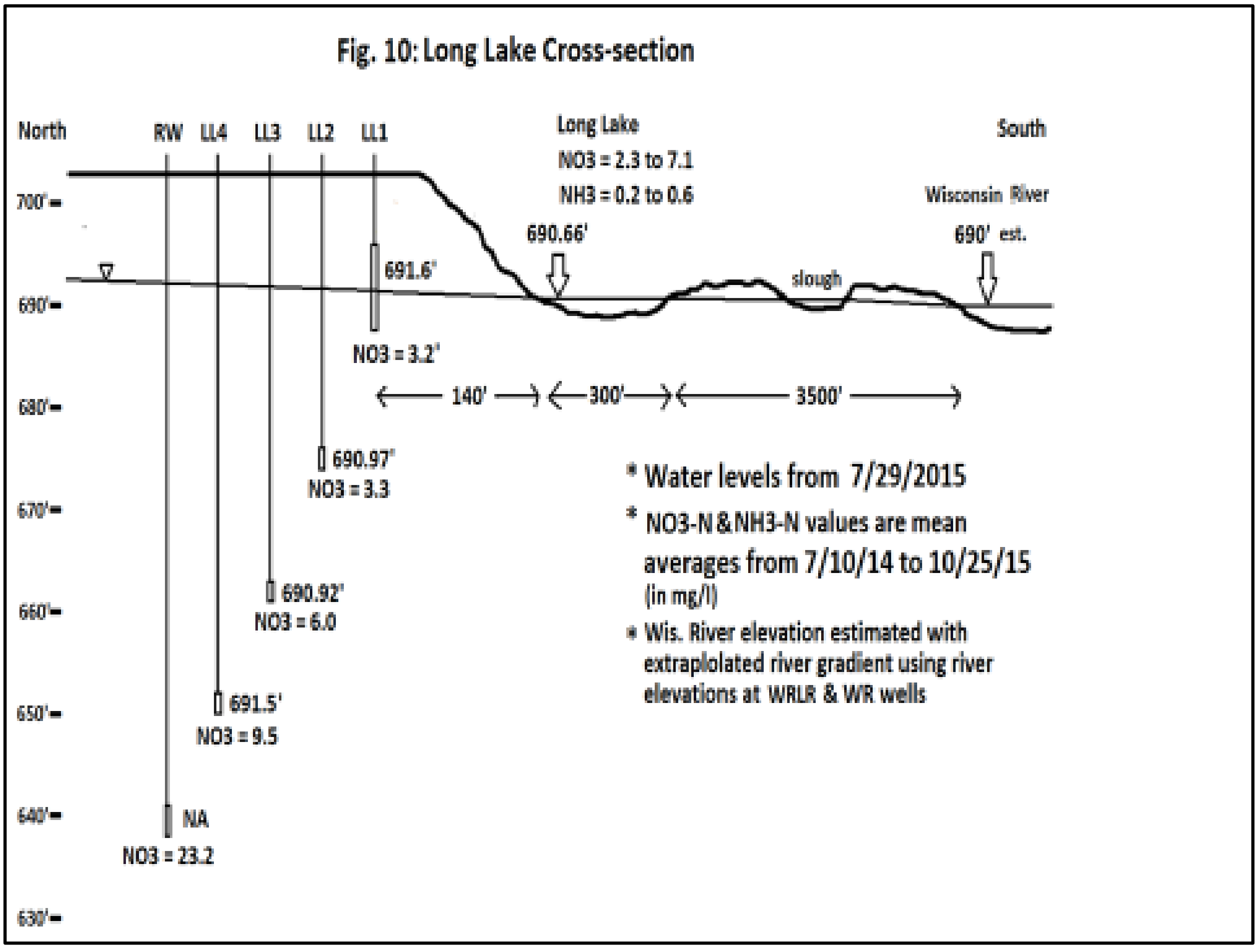


Figure 10: Elevation cross-section at Long Lake



The groundwater elevations found at the well clusters adjacent each lake are very similar in that there is not much difference in elevation from the water table and each of the adjacent piezometers. Since the error associated with each of the noted values due to surveying or field reading accuracy is likely to be approximately 0.02 to 0.03 feet, it is difficult to determine relative vertical groundwater gradients for each cluster. However, from the comparison of the groundwater and lake elevations it is clear that there is always a potential upward gradient from each groundwater monitoring location to the adjacent lake. Based on water elevation data alone, it is difficult to determine the significance of potential groundwater underflow beneath the lakes before reaching the Wisconsin River discharge point. A summary of the maximum lake nitrate concentrations and the adjacent mean groundwater nitrate concentrations is provided below in Table 5.

The results indicate that for Jones Slough, Norton Slough, Bakkens Pond and Long Lake the adjacent groundwater is the source for the nutrients entering the lake. It was noted that while the upper portion of the aquifer adjacent to Bakkens Pond had low nitrate concentrations that correspond to the adjacent natural area land use, the lower portions must be discharging into

Bakken Pond to account for the high nitrate concentration observed in the lake. The discharge of groundwater at the spring adjacent to the well cluster at Bakken Pond shows there can also be localized points of discharge of groundwater from the deeper portions of the aquifer with recharge originating at the agricultural sources located 4400 feet up-gradient. The mean nitrate in the spring was 18.7 mg/l. At Long Lake, the upper portion of the aquifer contained relatively low nitrate concentrations compared to the lake. The private water supply well adjacent the monitoring well cluster, screened approximately 10 feet below the deepest piezometer, had mean nitrate concentrations of 23.2 mg/l NO3-N. This suggests it is possible the upper aquifer water is discharging into Long Lake while the deeper water may underflow the lake with eventual discharge at the Wisconsin River. The 15.6-year travel time calculated for agricultural nutrient recharge to arrive at Long Lake also indicates the nitrate concentration in the deeper aquifer adjacent Long Lake is likely to increase over time.

The ongoing numerical modeling of the groundwater flow system in the study area being conducted by the UW-Madison Geosciences and Ken Wade under a WDNR River Planning Grant administered by Sauk County is scheduled to be completed by the end of 2016. This modeling will allow for more precise delineation of the groundwater flow paths and the recharge areas discharging to the oxbow lakes. This information will help to target the oxbow recharge areas where strategies for maintaining or improving recharge water quality can be applied.

Table 5: Mean Nitrate Concentrations for Lakes and Adjacent Groundwater

Lake	Maximum Lake NOx-N	Groundwater NOx-N*
Jones Slough	16.9	17.4**
Norton Slough	6.1	13.8
Bakkens Pond	15.6	29.3
Long Lake	7.1	6.3***

* Mean of piezometers (excluding water table wells)

** Only 2 piezometers at Jones Slough (shallower samples)

*** NO3-N of 23.2 mg/l observed in adjacent private well screened 10 ft. below piezometer LL4

Oxbow Lake Responses

Jones Slough and Norton Slough: These oxbow spring lakes lie in close proximity but exhibit different degrees of water quality degradation and nuisance levels of FFP growths (Figure 11). While both oxbows have a recent history of complete (100%) free floating plant cover (see Phase 1 report), in 2014 and 2015 the frequency occurrence was lower in Norton Slough (no FFP greater than 80% and mean FFP of 42.5%) compared with Jones Slough that exceeded 90 % cover every year (78% frequency of 90% FFP or greater and mean FFP of 80.6%). The water quality impacts of prolonged duckweeds and filamentous algae cover are evident by looking at vertical profile temperature and water chemistry. The June 2015 aquatic plant point intercept survey indicated that the FFP frequency of occurrence was 81% in Norton Slough and 88% in Jones Slough. The total relative frequency of FFP mats in Jones Slough (Table 6) was 47%, and 32.8% in Norton Slough (Table 7).

The vegetation statistics from the point intercept survey suggested that both Norton and Jones Sloughs exhibited relatively high biological integrity and a low level of disturbance impacts although there is no baseline data for comparison. Jones slough showed just slightly lower values than Norton Slough. Norton Slough supported greater species richness at 30 along with a higher Simpson diversity index of 0.92 and a higher floristic quality of 32 (Table 8). Species richness in Jones Slough was 25, along with a Simpson diversity index of 0.87 and floristic quality 29 (Table 9). Plants were found at all depths in Jones and Norton Sloughs, but the majority of plants growing in Jones Slough occurred in water depths three feet or less, while in Norton Slough the majority of plants grew in out to the five-foot contour. Greater aquatic plant growths in deeper zones within Norton Slough may reflect less FFP cover (Figure 30) as opposed to Jones Slough (Figure 31). Since the point intercept surveys were conducted after the recent water quality degradation, quantitative comparisons of aquatic plant communities that existed prior to recent water quality declines cannot be made. However, previous qualitative surveys suggested greater densities of floating leaf plant species including white water lily, spatterdock and large leaf pondweed.

The dominant submersed aquatic plants in Jones and Norton Sloughs were coontail (*Ceratophyllum demersum*), common waterweed (*Elodea Canadensis*), white water crowfoot *(Ranunculus aquaticus)*, muskgrasses (*Chara*), and Long-leaf pondweed (*Potamogeton nodosus*), while the frequency of floating leaf plants such as white water lily (*Nymphaea odorata*) was much lower (Figure 32).

In Figures 12 and 13, water temperature profiles demonstrate the influence of the massive hillslope (terrace) aquifer since growing season temperatures remain very cold in these shallow spring lakes. The significant rapid water temperature changes with depth reflect a combination of heavier cold water that moves along the bottom and direct groundwater discharges. Despite the cold temperatures and associated high oxygen saturation potential, anoxic conditions were measured in Jones Slough but not in Norton Slough (Figures 14 and 15). Very high phosphorus concentrations in Jones Slough likely reflected internal loading under anoxic conditions. Nested well phosphorus concentrations adjacent to the oxbow (Md = 22 ug/l, u = 25 ug/l) were significantly lower (P = 0.07) than lake concentrations (Md = 71 ug/l, u = 236 ug/l) that exceeded 1 mg/l on two occasions (Figure 16). Phosphorus concentrations in Norton Slough (u = 46.3 ug/l) were also higher than in adjacent wells (22.8 ug/l), but were typically lower than Jones Slough phosphorus levels (Figure 17) where anoxia was more prevalent.

While phosphorus concentrations in Jones Slough were often higher than adjacent well concentrations, oxbow nitrate concentrations displayed (u = 4.36) were generally less than concentrations (u = 15.11) found in the adjacent wells (Figure 18). The loss of nitrogen in the oxbows likely reflected plant uptake since the lowest nitrate concentrations coincided when free floating plant densities were highest. Despite this observed reduction, nitrate concentrations in both Jones Slough and Norton Slough exceeded USEPA, USGS (Robertson et al. 2006) and Camargo et al. (2006) recommended criteria of $\leq$ 2 mg/l in more than 50% of the samples (Figures 18 and 19). In both Norton Slough and Jones Slough, higher nitrate concentrations were detected where the groundwater entered the oxbows. In Norton Slough the highest NOx occurred in deeper water (2+ meters) but in Jones Slough highest NOx was found in shallow areas adjacent to the steep terrace bank. In Jones Slough, dense filamentous algal growths were often prevalent throughout the water column where plant uptake can occur. While high NOx concentration occurred in deeper water areas in Norton Slough, very high NH_3 was measured in deeper water (2 meters) on 9/9/15 when anoxia occurred in Jones Slough (Figure 20).

The nested well phosphorus concentrations were not particularly high, however nitrogen loading in the form of nitrates was very high (Figure 21). Nitrate concentrations were exceedingly high in comparison with phosphorus concentrations. Based on groundwater NOx concentrations, N:P ratios were approximately 696:1 in Jones Slough wells and 605:1 in Norton Slough wells. In both Jones Slough and Norton Slough, longitudinal nitrate sampling transects were completed on May 20, 2015 and indicated that the highest NOx concentrations in the oxbows occurred near the well clusters at both oxbows (Figures 22 and 23). This information suggested that the primary aquifer discharges were near the monitoring wells. Significant concentrations of dissolved oxygen were prevalent in the terrace monitoring wells adjacent to both Jones and Norton sloughs (JS1, JS2, JS3 – u = 7.3, 7.0, 8.6 mg/l respectively and NS1, NS2, NS3, NS4 – u = 7.0, 5.1, 4.1, 3.9 mg/l respectively). This source of cold dissolved oxygenated water is an important factor for sustaining environmentally sensitive fish and other aquatic organisms during periods of high river stages.

The water chemistry of the floodplain aquifer was significantly different from terrace aquifer. High phosphorus concentrations were found in the two floodplain wells located adjacent to Norton Slough. Floodplain well phosphorus concentrations ranged from 99 ug/l to 716 ug/l while levels were much lower in the terrace wells. Dissolved oxygen concentrations in the floodplain wells were very low, approaching anoxia, while ammonium levels were high in the shallow zone. As described in the Phase 1 report, the floodplain well data demonstrated how the river aquifer can be an infrequent but significant source of high nutrients and organic loading to the oxbows when river stages are high.

Specific conductance and pH data appear in the Appendix. Mean specific conductance levels in Jones Slough and Norton Slough were 497 and 397 uS/cm respectively. The adjacent wells were similar with levels at 403 uS/cm and 426 uS/cm respectively. Given that the Wisconsin River channel specific conductance levels are typically less than 300 uS/cm, the significantly higher oxbow levels reflect river terrace aquifer influence on the slough hydrology. Water clarity in the oxbows is also significantly different from the river. Mean turbidity levels in Jones Slough and Norton Slough were 2.1 and 2.5 NTU compared with mean river turbidity at 10.6 NTU. Secchi measurements also generally exceeded 2 meters in both oxbows that indicated minimal planktonic algal productivity.

Table 10 contains the fish electroshocking data for 2014 and 2014. State Endangered starhead topminnows still thrive in both oxbows although access to surface "topminnow" habitat is often limited, particularly in Jones Slough. State Special Concern lake chubsuckers were found in Jones Slough and State Special Concern pirate perch were found in both sloughs. State Special Concern mud darters were historically found in the slough but none were found recently. Mud darters, in part due to their benthic existence and sensitivity to degraded water quality, are particularly vulnerable to the recent environmental degradation in the oxbows.

Figure 11: Jones Slough and Norton Slough Estimated Free Floating Plant Cover

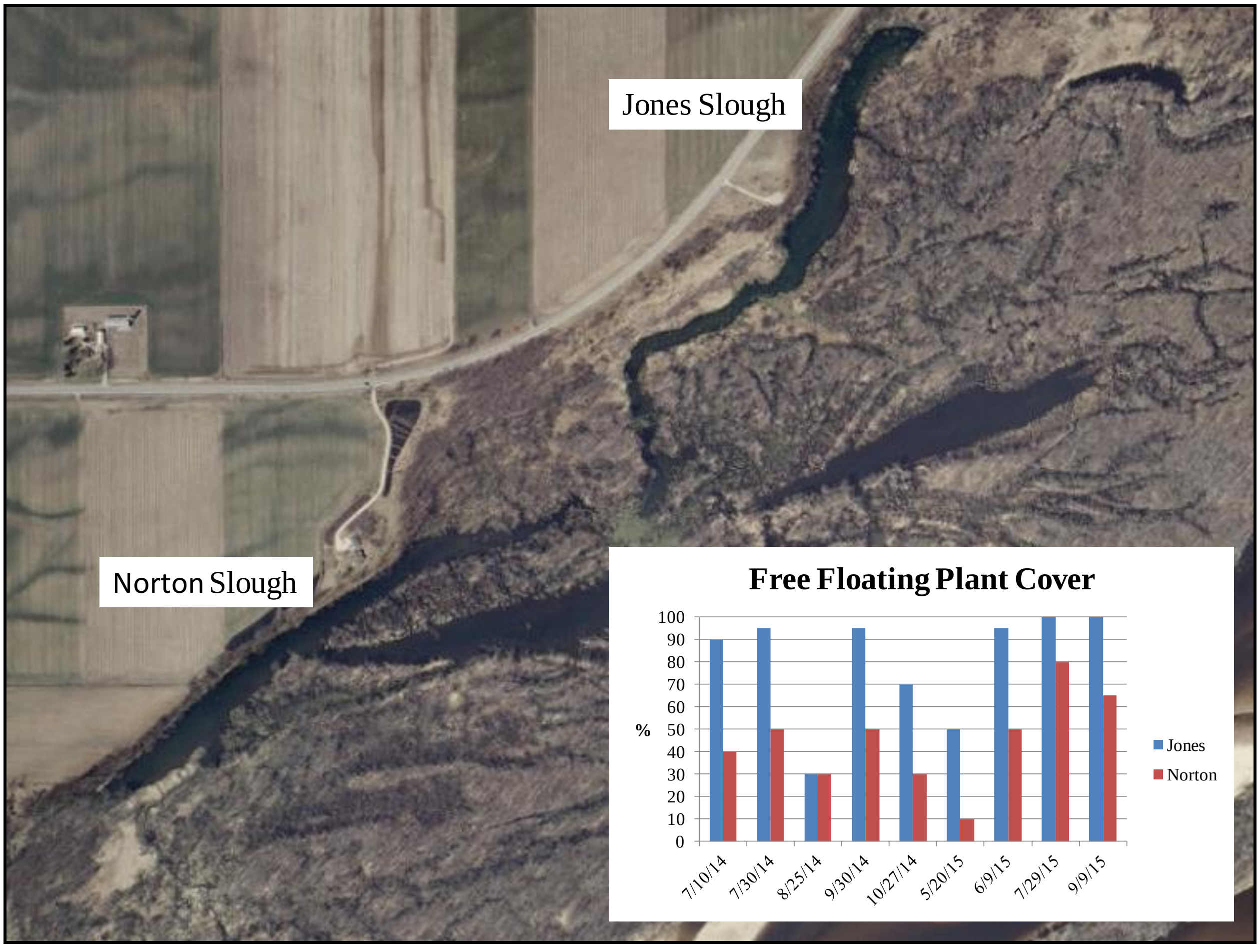


Figure 12: Jones Slough Temperature Profiles

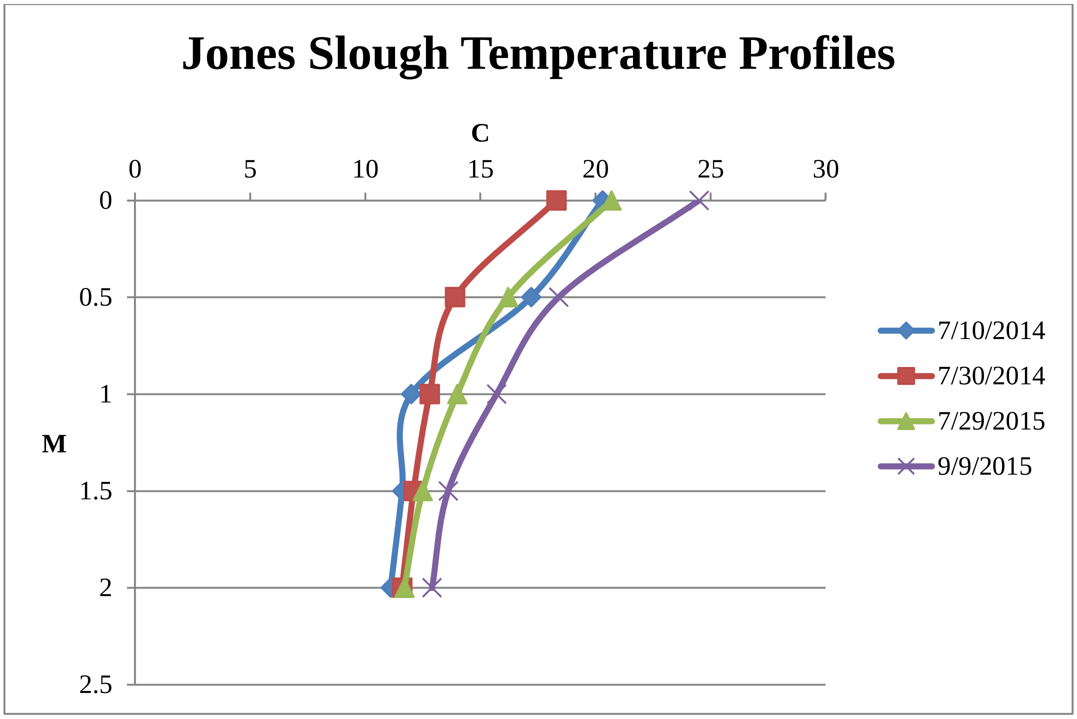
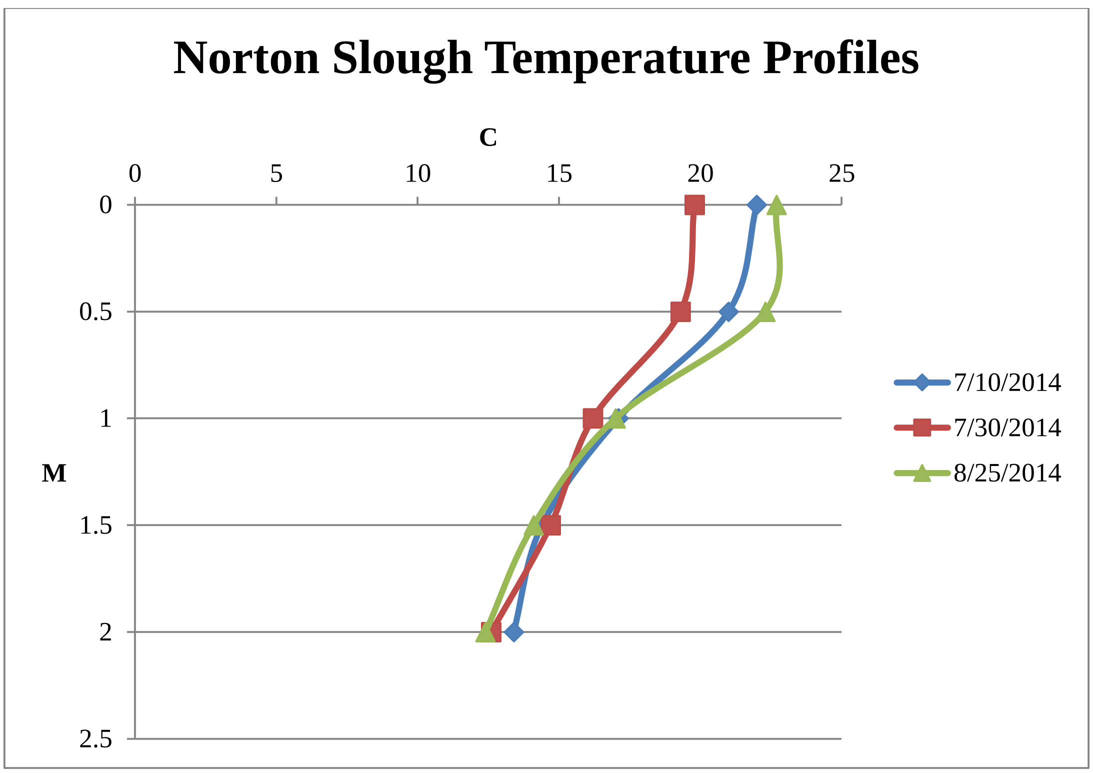


Figure 13: Norton Slough Temperature Profiles



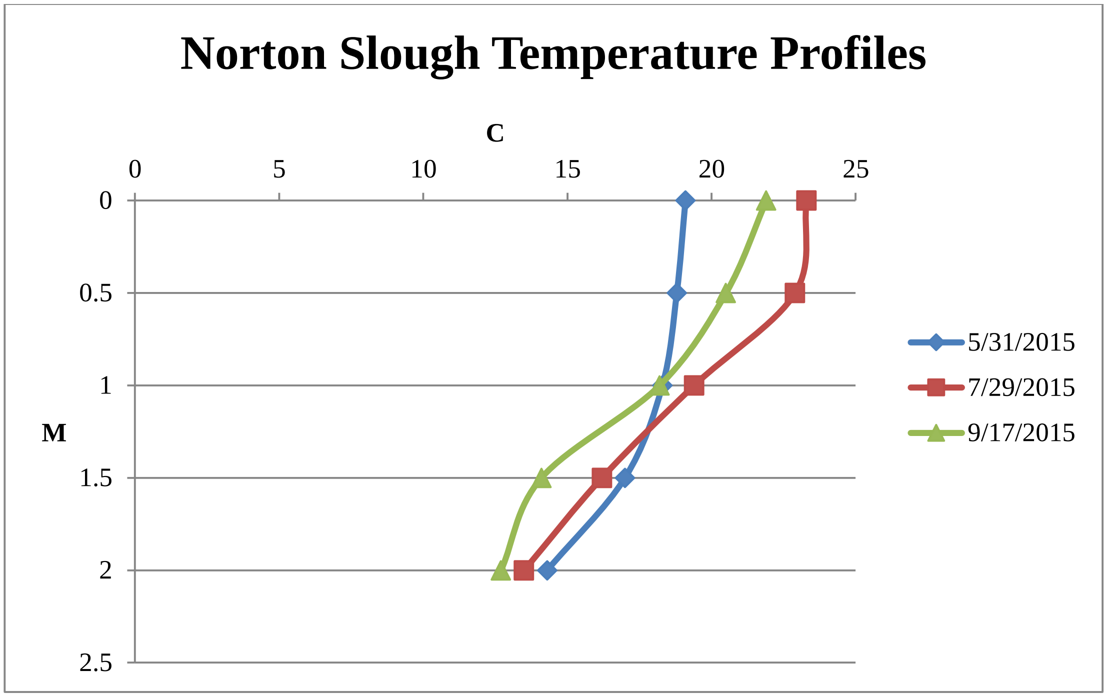


Figure 14: Jones Slough Dissolved Oxygen Profiles

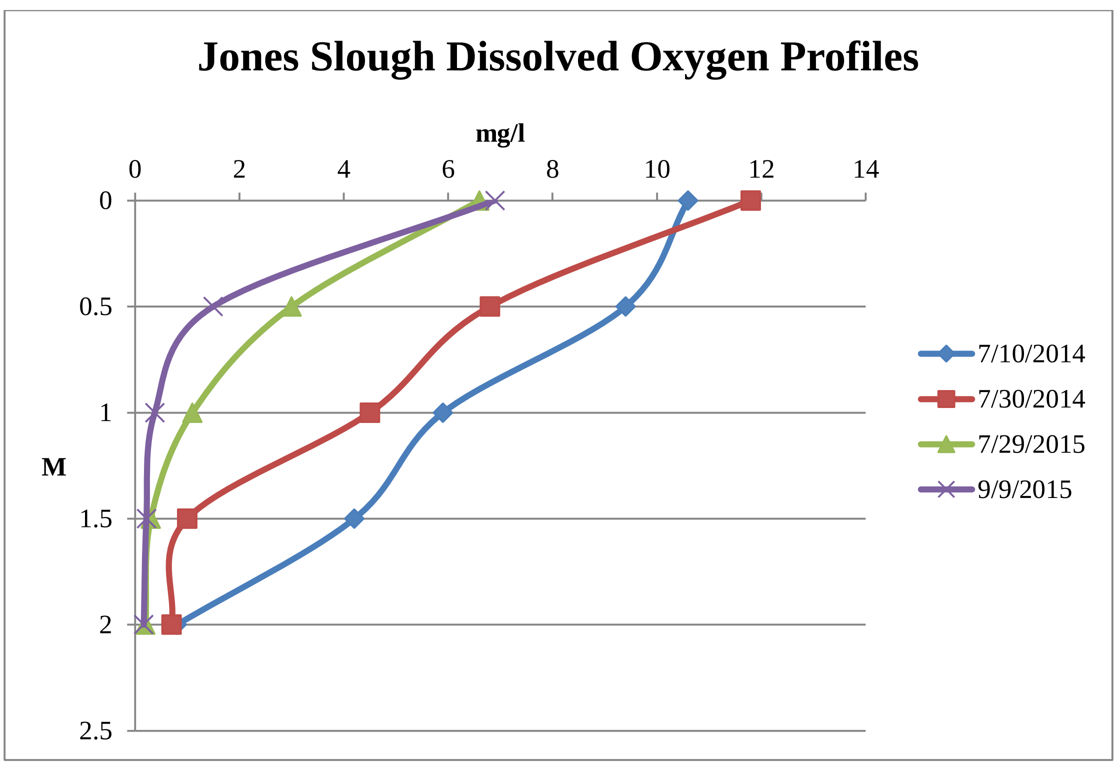
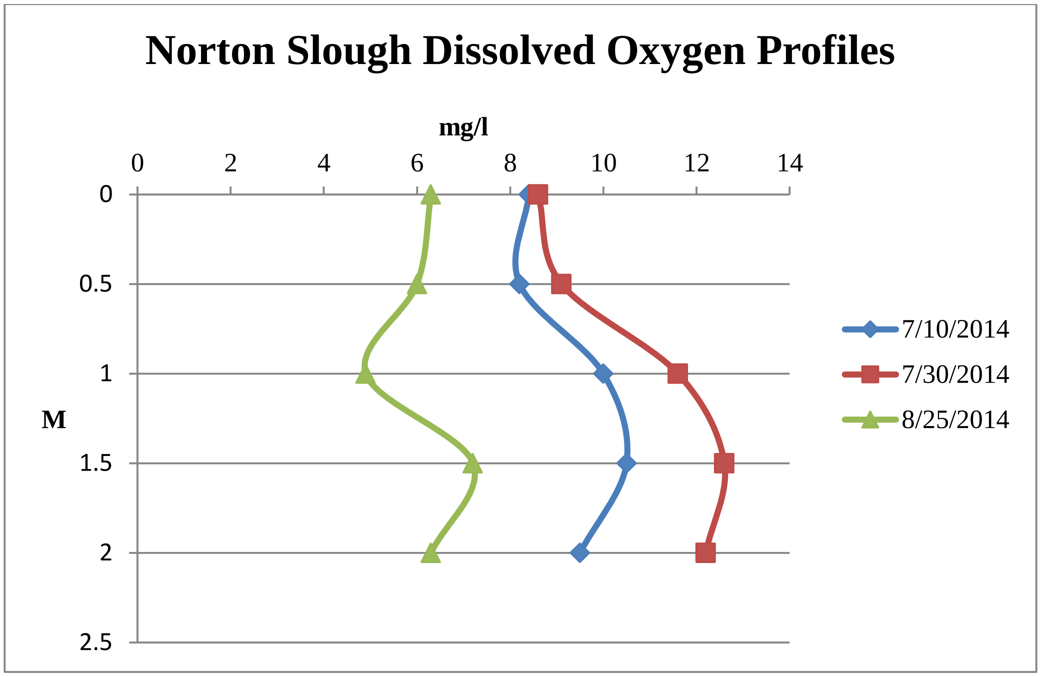


Figure 15: Norton Slough Dissolved Oxygen Profiles



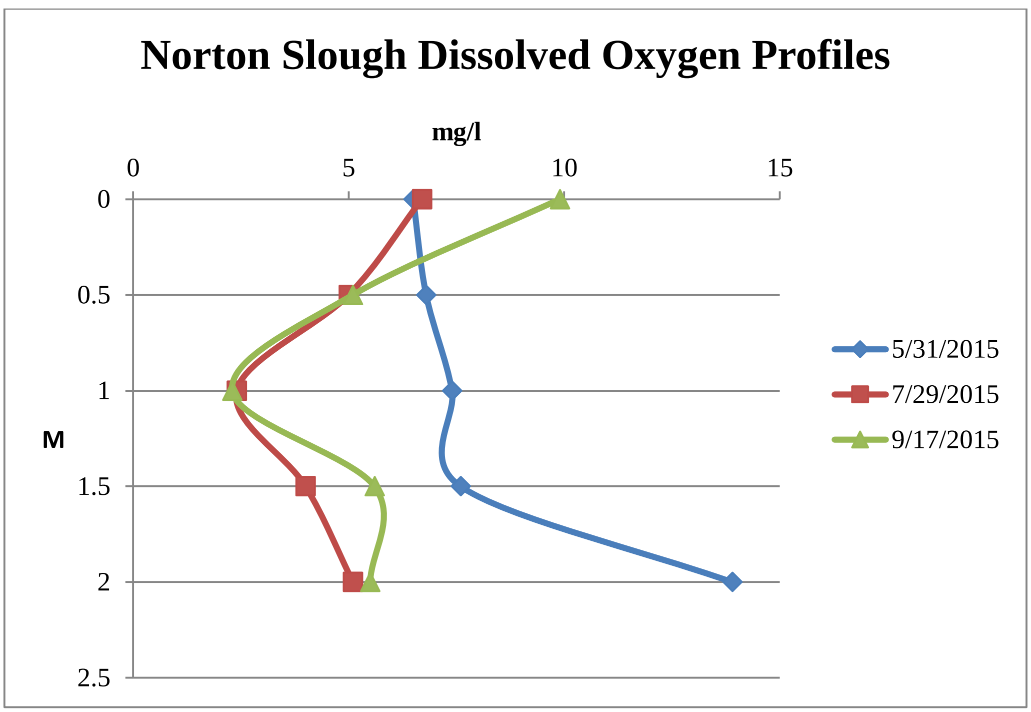


Figure 16: Jones Slough Phosphorus Concentrations

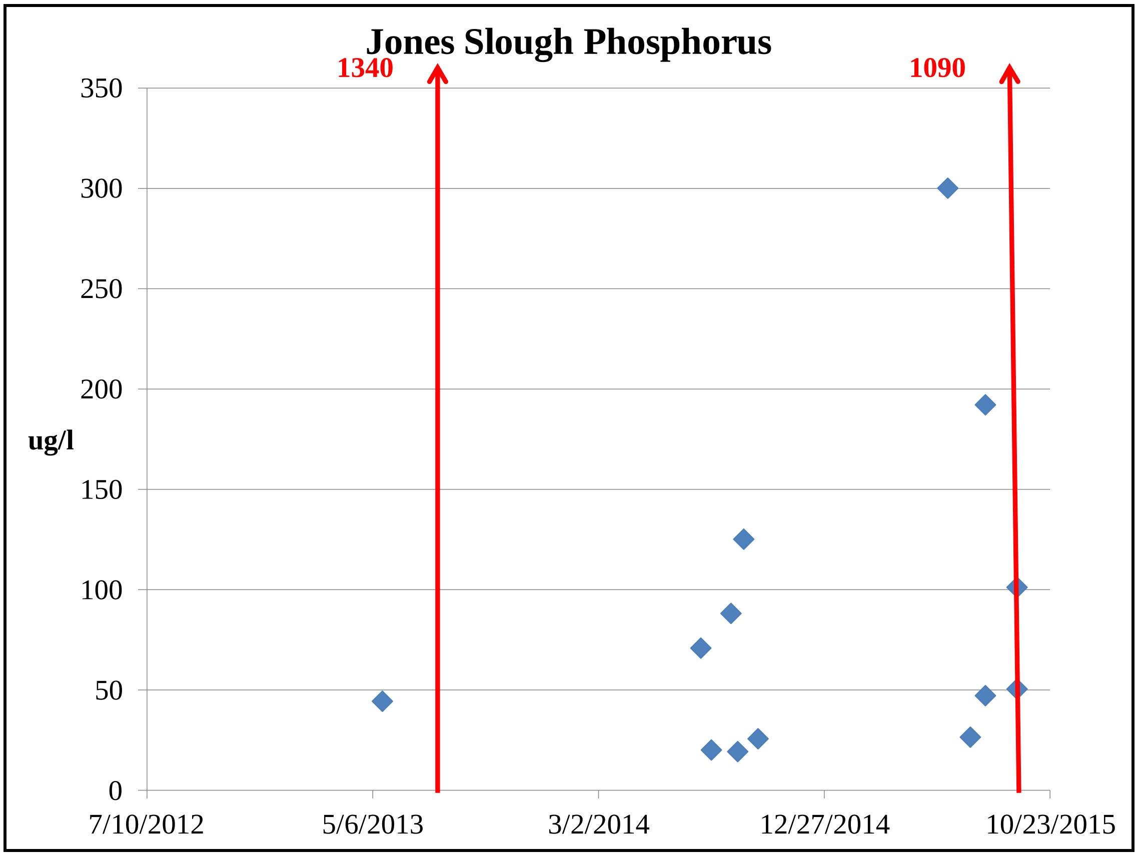


Figure 17: Norton Slough Phosphorus Concentrations

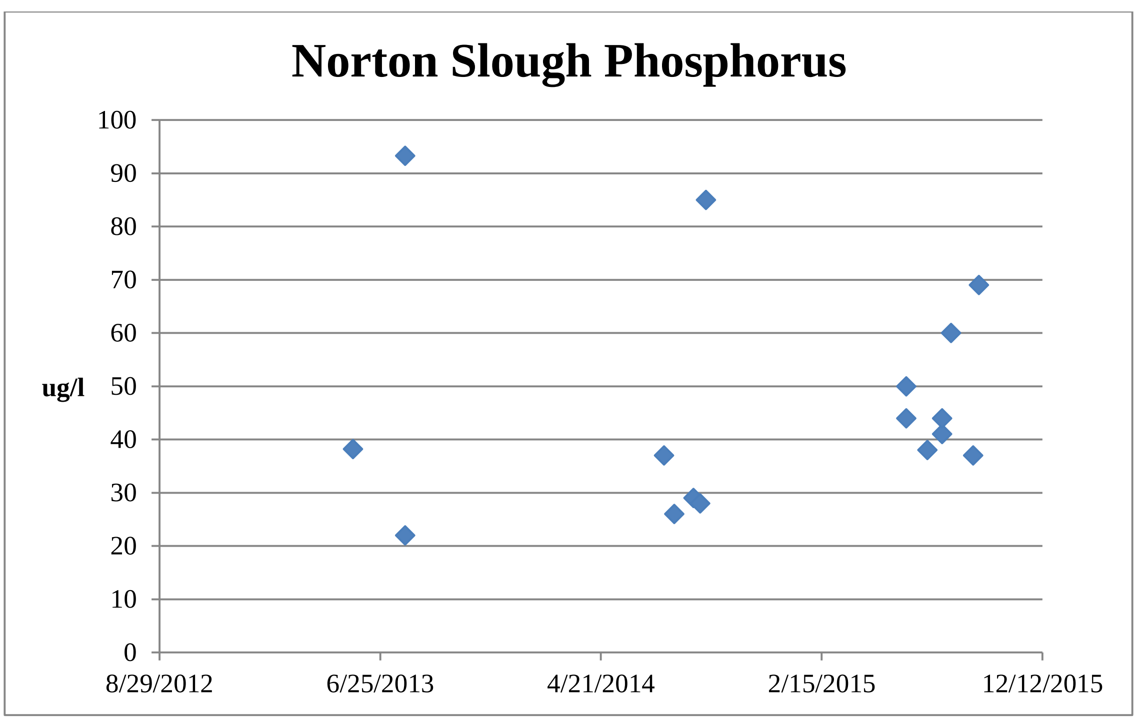


Figure 18: Jones Slough Nitrate Concentrations

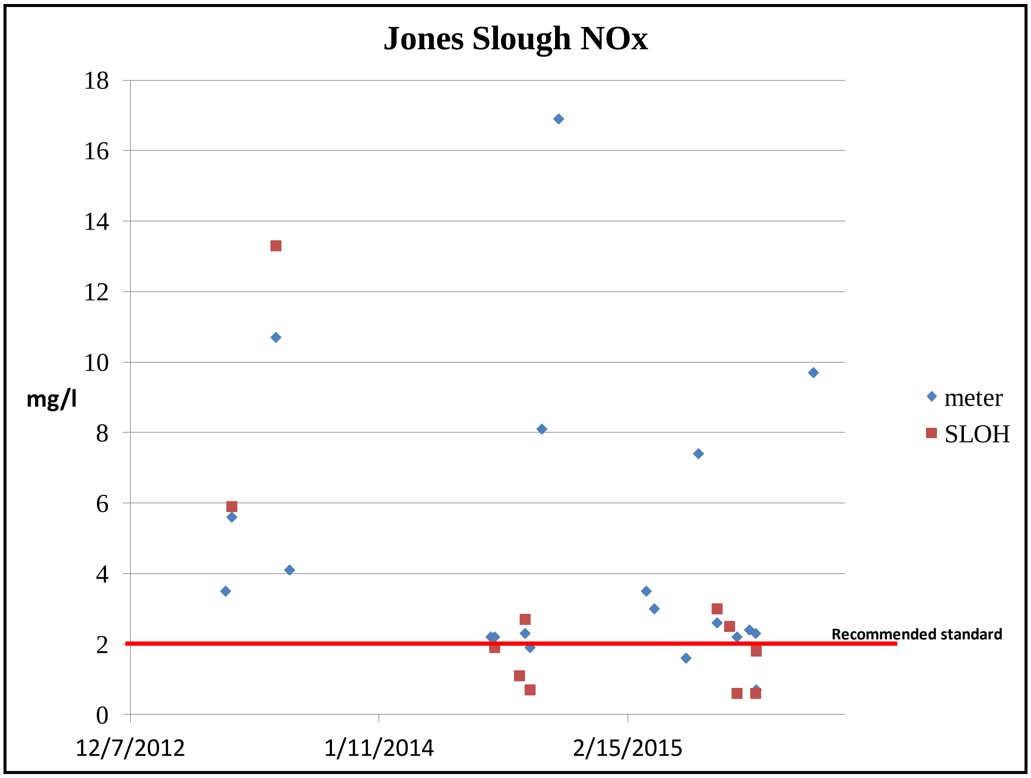


Figure 19: Norton Slough Nitrate Concentrations

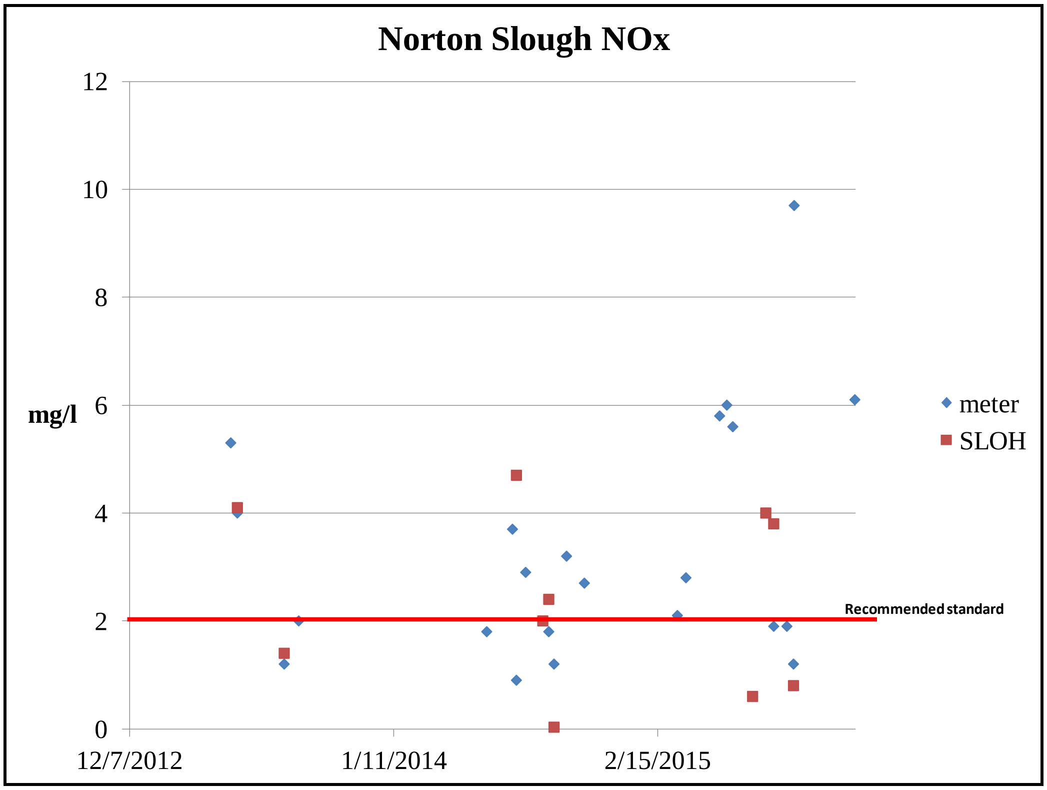


Figure 20: Jones Slough Ammonium and Nitrate Profiles

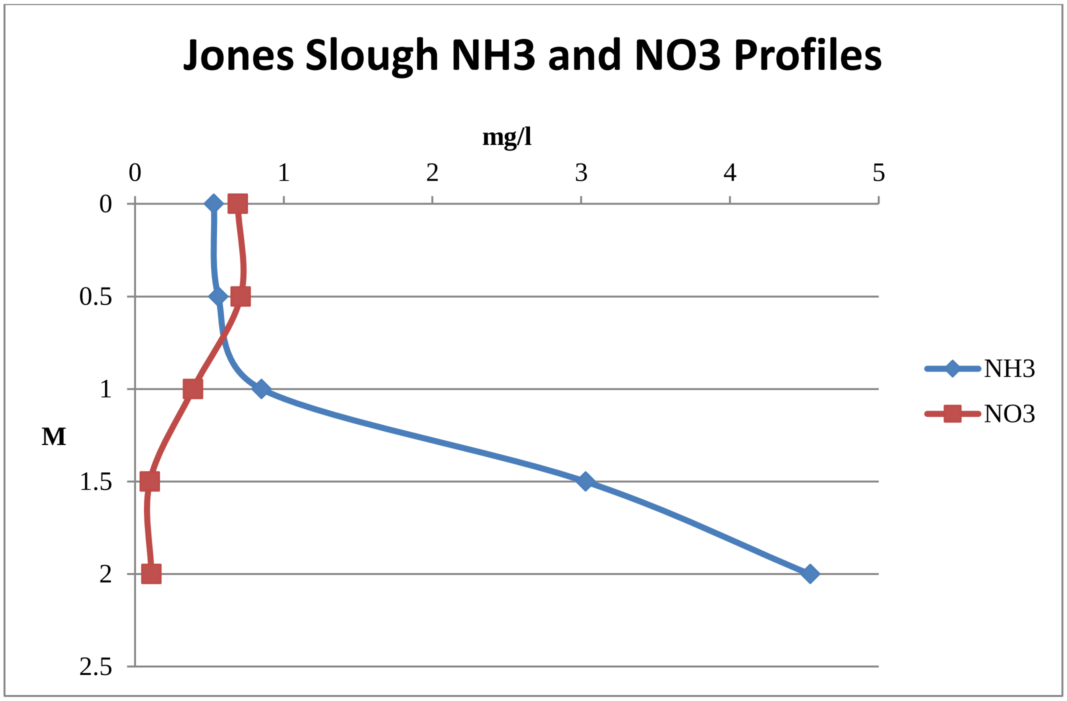


Figure 21: Mean Well Nitrate and Phosphorus Concentrations

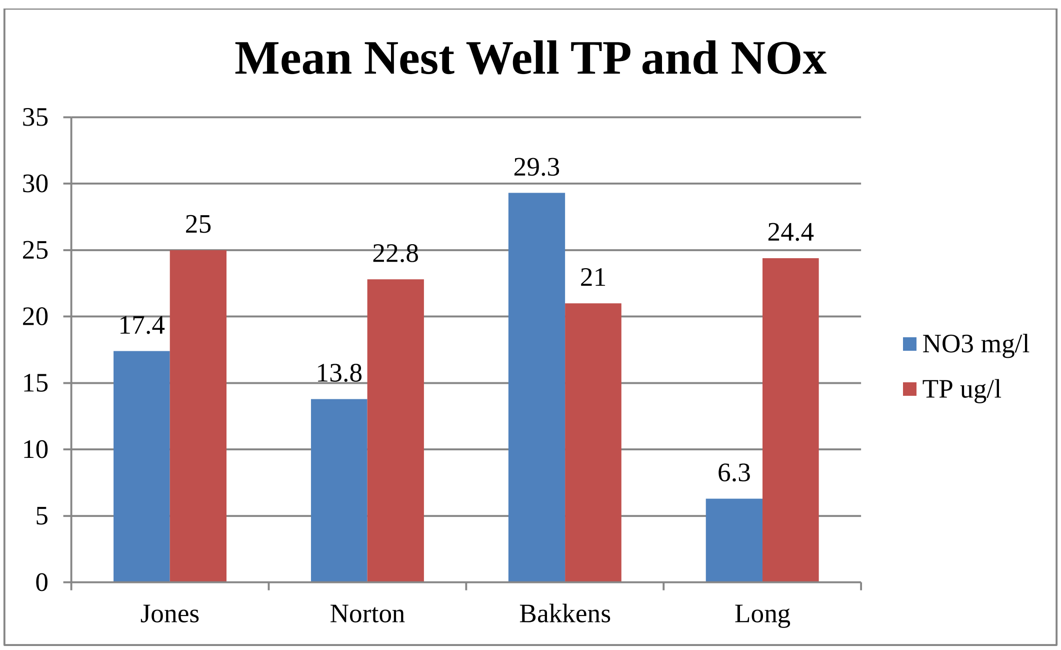


Figure 22: Jones Slough Nitrate Transect in Relation to Monitoring Wells

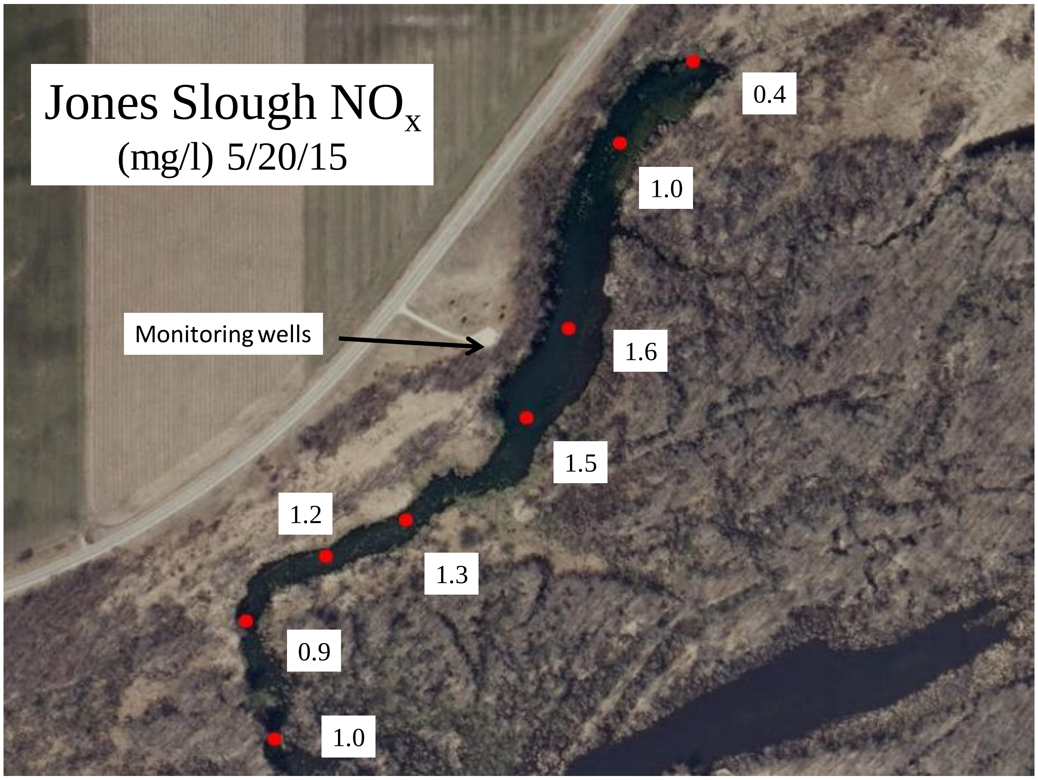
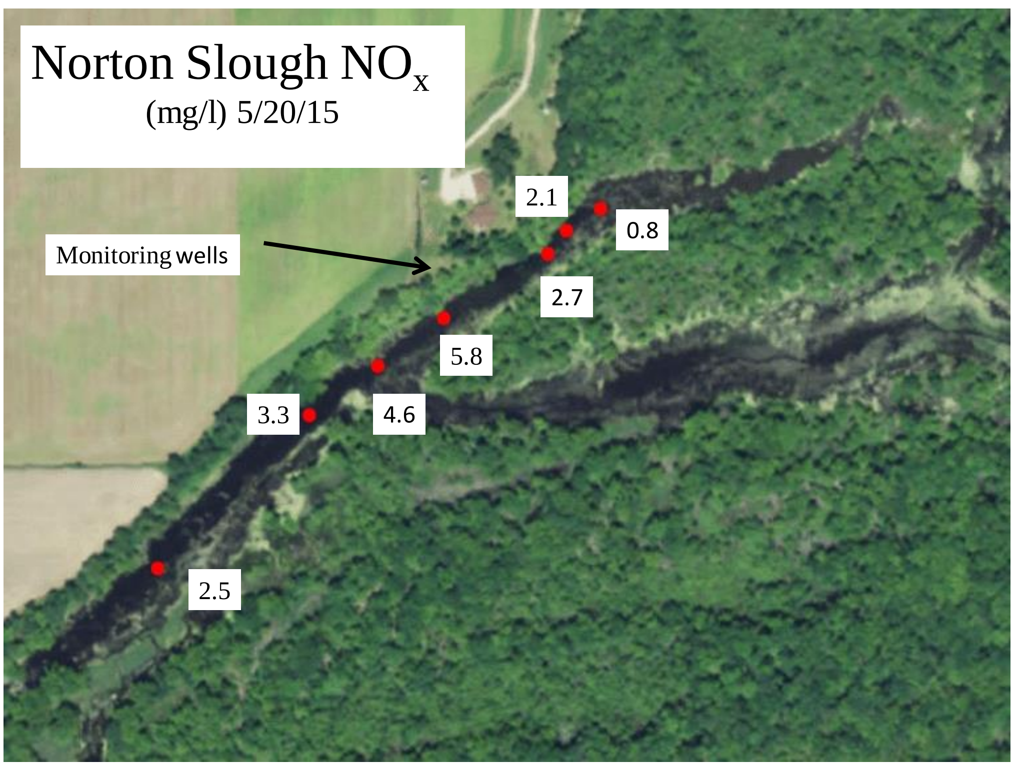


Figure 23: Norton Slough Nitrate Transect in Relation to Monitoring Wells



Bakkens Pond and Long Lake: Bakkens Pond is frequently covered with FFP within the impounded areas. The monitoring well cluster and surface water sampling site were located at a free flowing section of the pond (Figure 4) that was often choked with filamentous algae. The water column nitrate concentrations in the area averaged 11.3 mg/l (Md = 12.7). All of the NOx measurements in the channel exceeded recommended nitrogen criteria (Figure 24). The high nitrate concentrations were traced to springs that discharge in close proximity to the nested monitoring wells. These high values also coincided with very high well nitrate concentrations (u = 17.5 mg/l in Figure 21). Ignoring shallow well BP1 that represents the wooded buffer recharge zone that is located between Kennedy Road and the wells, then the average well nitrate concentration increased to 29.3 mg/l. This information suggested that the approximate 100 meter wooded buffer was not sufficient to reduce nitrate concentrations due to very high concentrations in deeper aquifer zones. Relative to the high nitrate concentrations, the phosphorus levels were relatively low in both the lake and well samples (Figure 25). Bakkens Pond mean, median and maximum phosphorus concentrations were 35, 39 and 42 ug/l respectively and the surface water N: P ratio was approximately 322:1. The monitoring wells mean, median and maximum phosphorus concentrations were 21, 14 and 46 ug/l respectively and the N:P ratio was about 1,395:1 based on NOx data.

Downstream of the second Bakkens Pond impoundment, Long Lake is a drainage oxbow that displays significantly lower levels of nuisance FFP growths. FFP occupy about 5% of the water surface throughout the summer. Low densities of FFP along with floating organic matter, that includes Oscillatoria sp., reflect a relatively moderate level of eutrophication in Long Lake compared with the other three lakes. Similar conditions were also found in Smith Lake and Crusan Slough located west of Lone Rock (Phase 1 report). Long Lake well nitrate concentrations were the lowest of the four oxbows surveyed in 2014 and 2015 even though the well cluster intercepts a septic system drain field (Figure 21). Clear evidence of the septic field influence included higher specific conductance measurements along with very high chloride concentrations that exceeded 300 mg/l on two occasions. Long Lake nitrate concentrations (u = 4.6 mg/l) consistently exceeded recommended criteria (Figure 26) but lacked the high densities of free floating plants found in the other three oxbows that can rapidly absorb nutrients. Based on two lake wide nitrate surveys conducted in 2015, the highest nitrate concentrations occurred near the east end of the lake, near Bakkens Pond outlet, and gradually decreased to the west (Figure 27). The mean and median phosphorus concentrations in Long Lake were 43.1 and 35 ug/l respectively (Figure 28) and the N: P ratio was about 258:1. Well phosphorus concentrations averaged 24.4 ug/l after removing an outlier concentration of 434 ug/l that was collected in the shallow well on September 20, 2015. The lower productivity in Long Lake likely reflects the influence of the School Forest as a buffer along with potentially greater dilution since Long Lake is significantly larger (~144 acre feet) that both Norton Slough (~54 acre feet) and Jones Slough (~30 acre feet) where groundwater could potentially have a greater influence. Another difference that Long Lake exhibited from the other three oxbows was clear evidence of significant groundwater discharge. The temperature profiles in Figure 29 demonstrate lack of cold water found in Long Lake compared with the Bakkens Pond springs or cold water layers at the bottom of both Jones Slough and Norton Slough.

Consistent with both Norton Slough and Jones Slough, specific conductance levels were higher in Bakkens Pond (u = 463 uS/cm) and Long Lake (u = 396 uS/cm) than in the river. Water clarity was also higher in both (Bakkens Pond = 1.4 NTU and Long Lake 4.2 NTU) than the river. These data again suggest greater river terrace influence on oxbows hydrology during lowflow conditions. Mean specific conductance in Bakkens Pond wells was 482 uS/cm without including the shallow well that had only 57.6 uS/cm. The low conductance in the shallow well reflected clean soft water recharge within the wooded buffer. Long Lake well specific conductance levels were very high (u = 579 uS/cm) and reflected influence of chlorides down gradient of the septic system.

Both Bakkens Pond and Long Lake support a number of rare and environmentally sensitive fish species even though nitrate toxicity is a concern. Table 11 displays electroshocking surveys conducted in 2014 and 2015. Mud darters and other darters are still found in both Bakkens Pond and Long Lake. Flowing water appears to mitigate some of the impacts of nitrogen loading and eutrophication in Bakkens Pond while more favorable water quality was likely a greater factor for darter survival in Long Lake.

Figure 24: Bakkens Pond Nitrate Concentrations

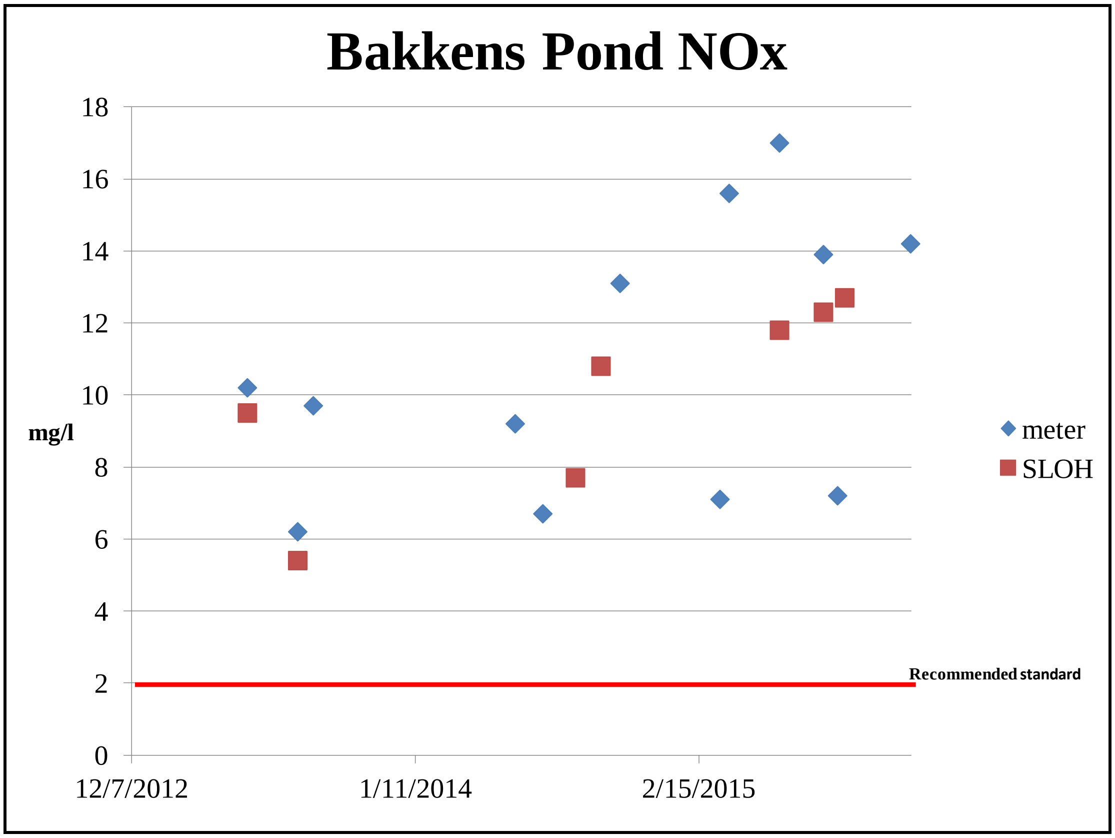


Figure 25: Bakkens Pond Phosphorus Concentrations

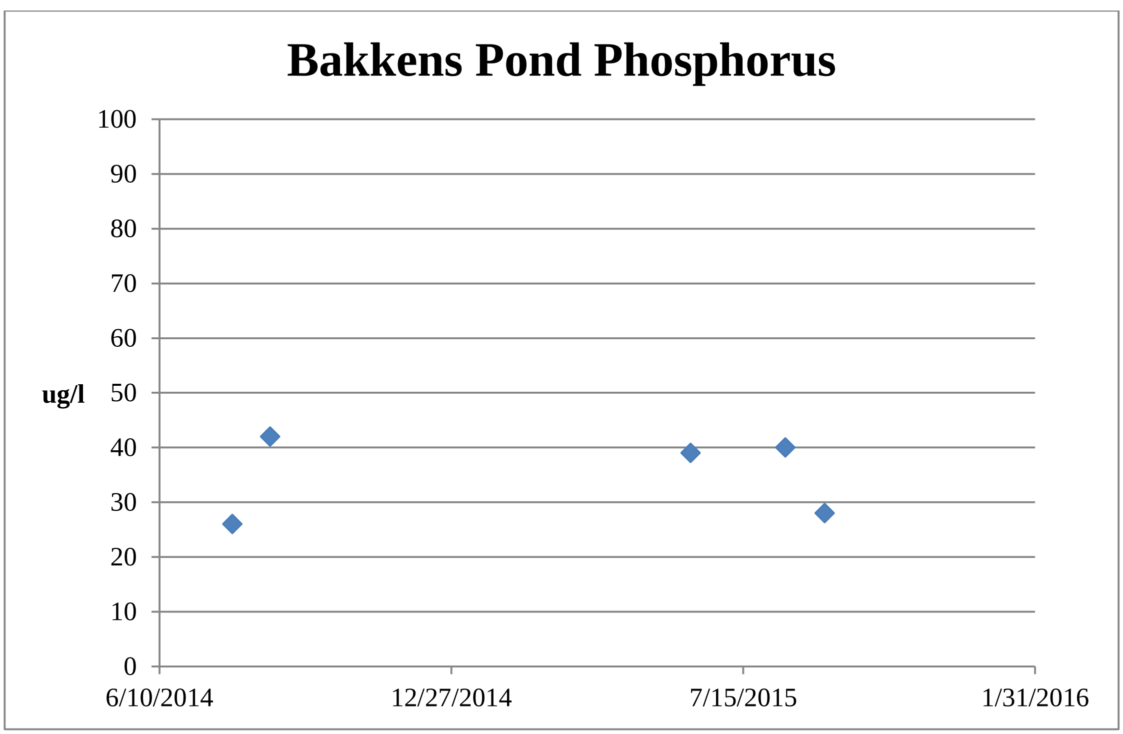


Figure 26: Long Lake Nitrate Concentrations

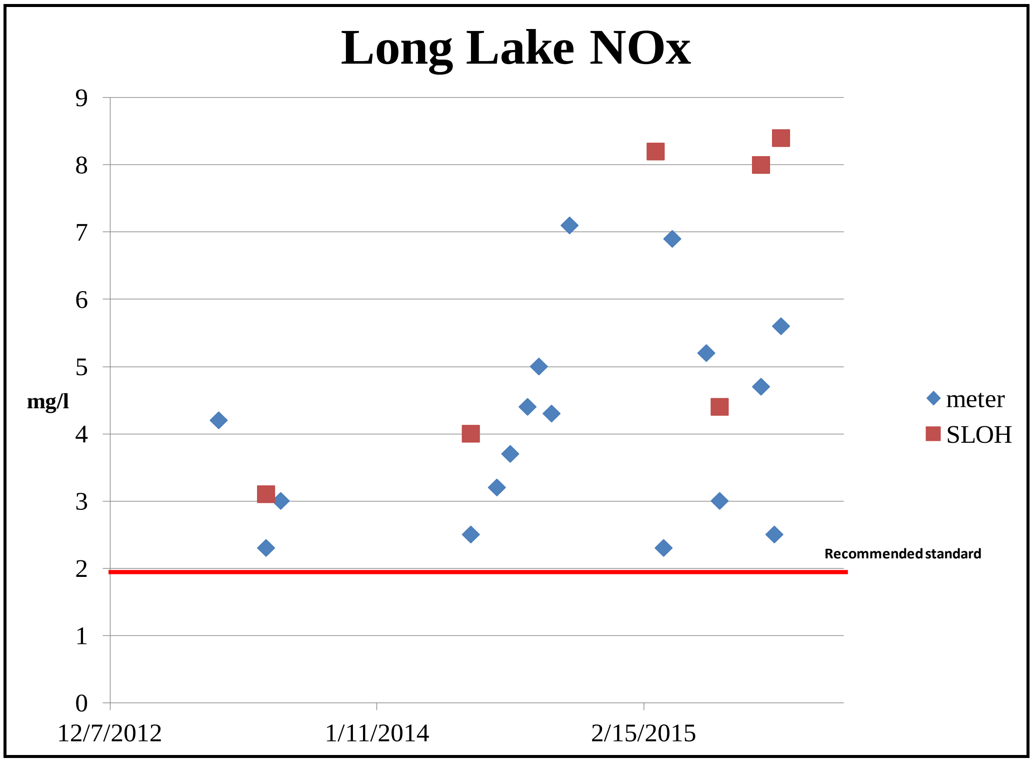


Figure 27: Nitrate Concentrations across Long Lake

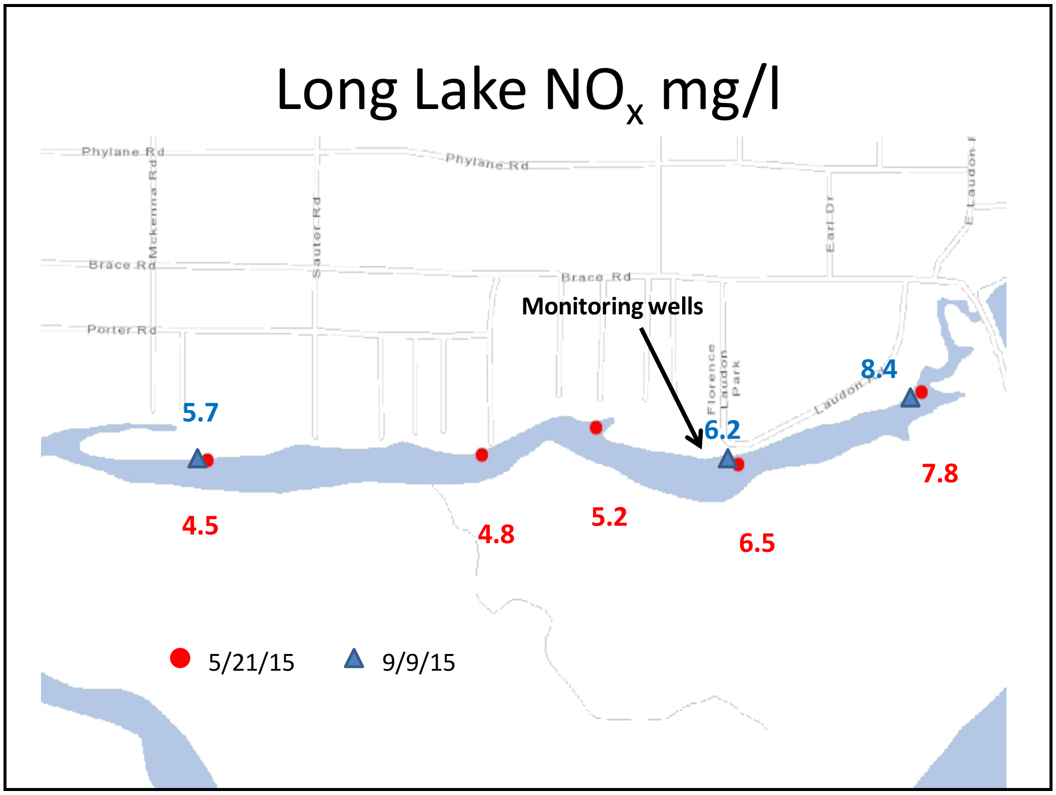


Figure 28: Long Lake Phosphorus Concentrations

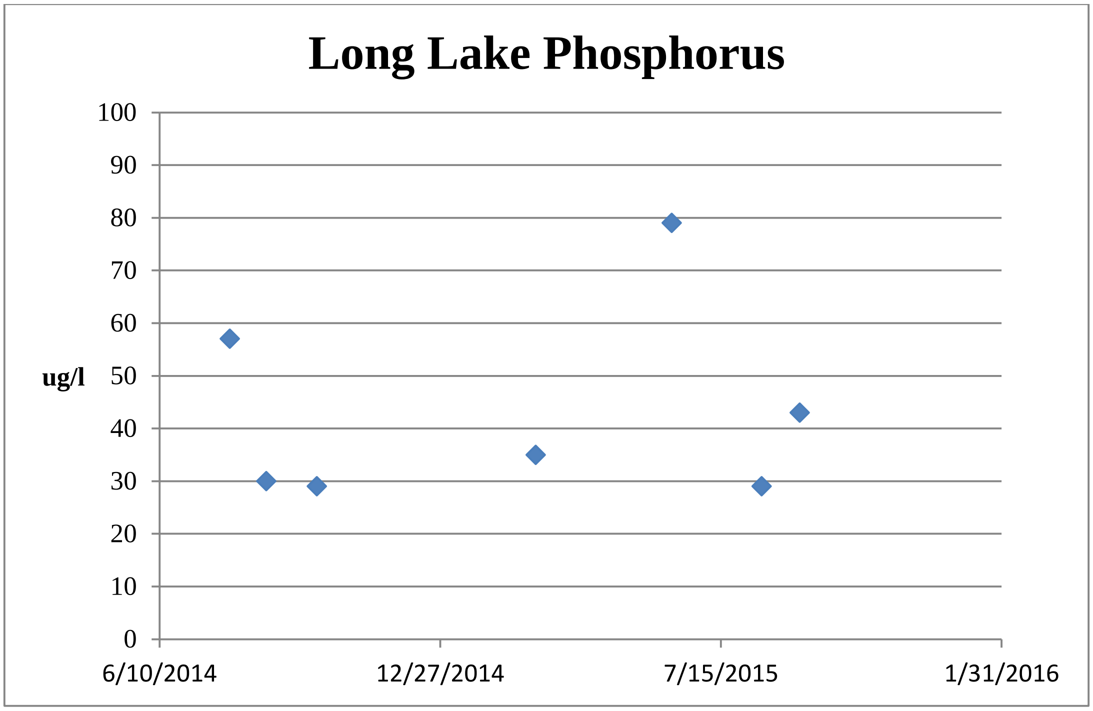


Figure 29: Long Lake Temperature Profiles

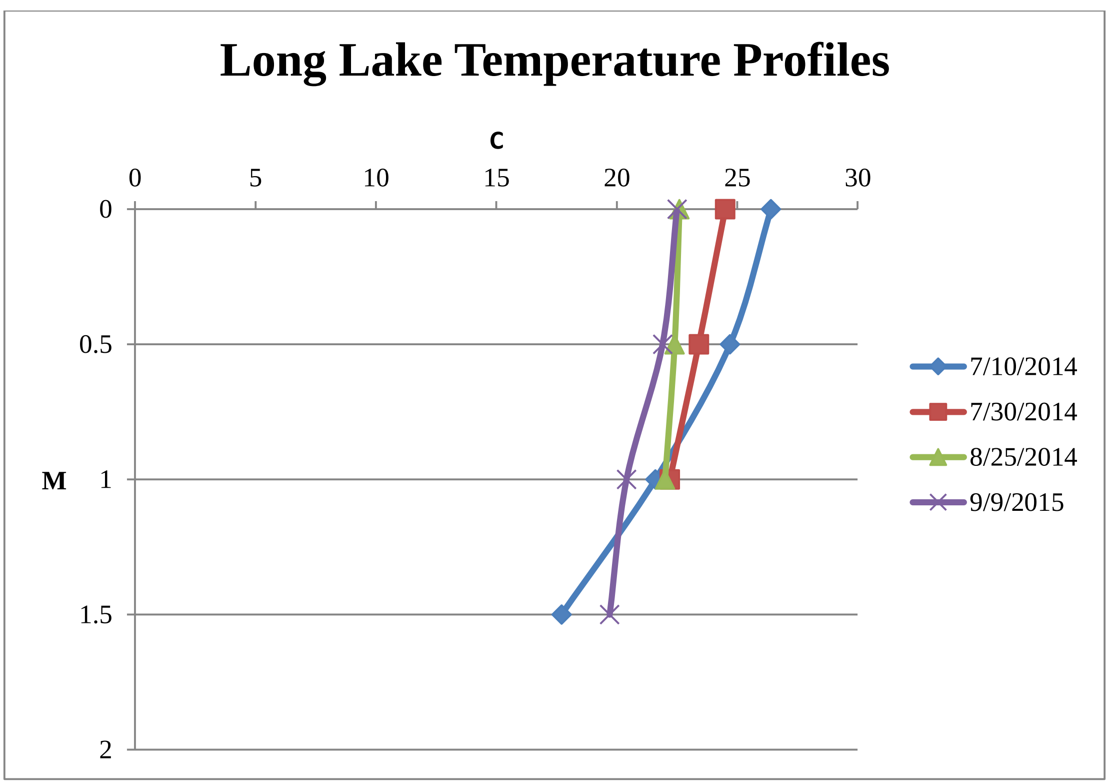


Figure 30: Norton Slough sample sites with aquatic plant colonization by water depth

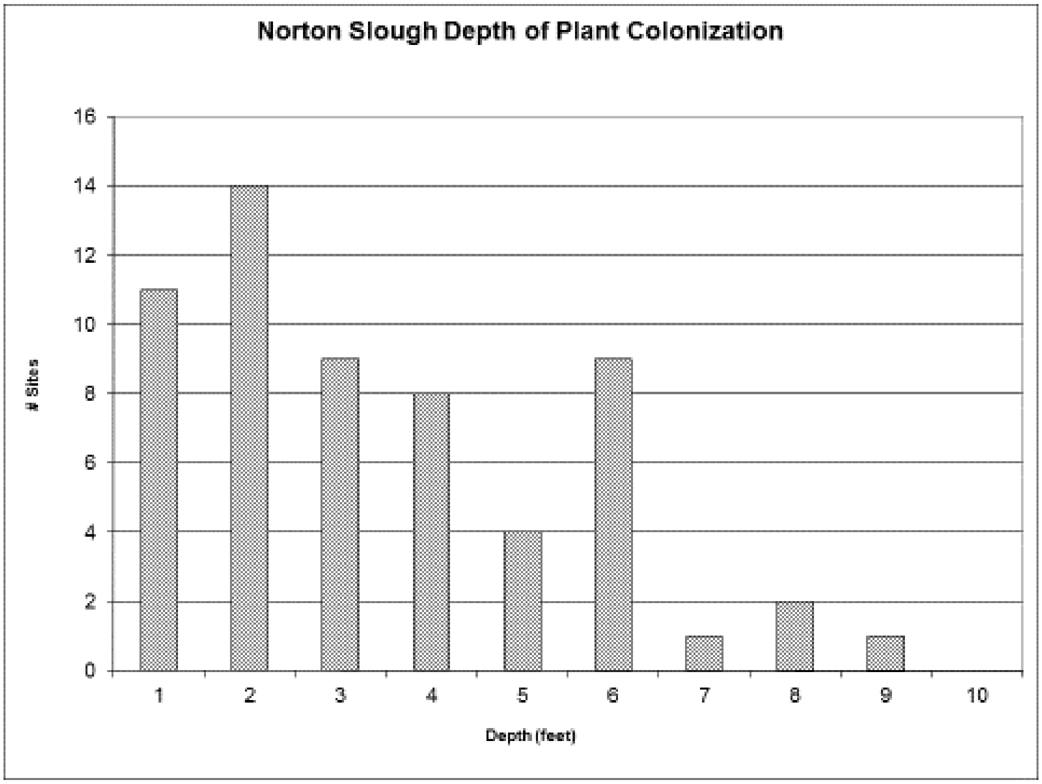


Figure 31: Jones Slough sample sites with aquatic plant colonization by water depth.

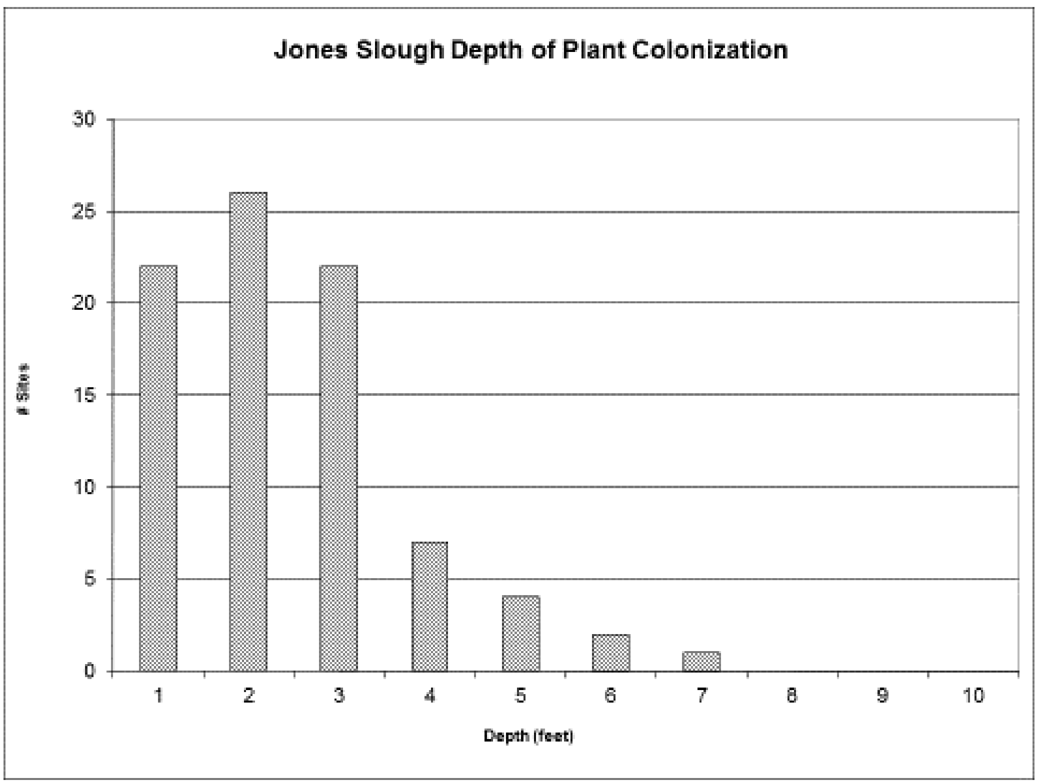


Figure 32: Relative Importance of dominant aquatic plants found in Jones and Norton Sloughs

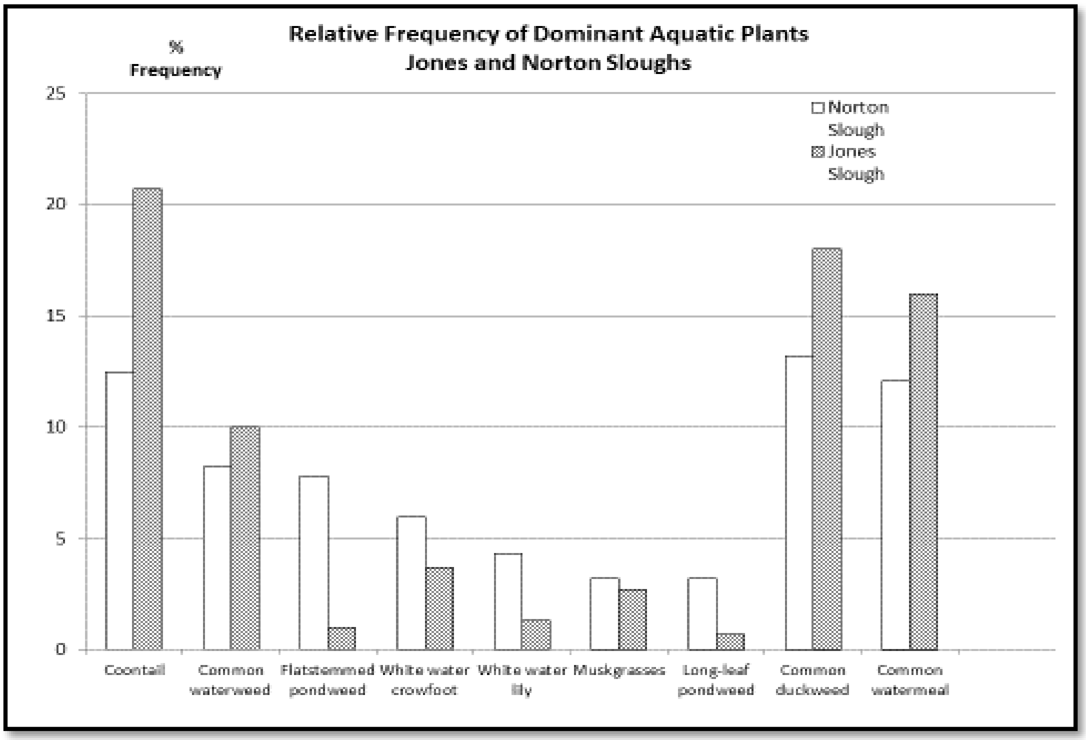


Table 6: Jones Slough relative importance of aquatic plants

Species	Common Name	% Frequency	No. of Sites
Bidens beckii	Water marigold	1.3	4
Ceratophyllum demersum	Coontail	20.7	62
Chara	Muskgrasses	1.7	8
Elodea canadensis	Common waterweed	10.0	30
Elodea nuttallii	Slender waterweed	0.7	2
Heteranthera dubia	Water star-grass	0.3	1
Lemna minor	Small duckweed	18.0	54
Myriophyllum sibiricum	Northern water-milfoil	1.3	4
Myriophyllum verticillatum	Whorled water-milfoil	1.3	4
Nitella	Nitella	2.7	8
Nymphaea odorata	White water lily	1.3	4
Potamogeton confervoides	Algal-leaved pondweed	0.3	1
Potamogeton foliosus	Leafy pondweed	0.3	1
Potamogeton illinoensis	Illinois pondweed	0.3	1
Potamogeton nodosus	Long-leaf pondweed	0.7	2
Potamogeton obtusifolius	Blunt-leaf pondweed	0.7	2
Potamogeton zosteriformis	Flat-stem pondweed	1.0	3
Ranunculus aquatilis	White water crowfoot	3.7	11
Spirodela polyrhiza	Large duckweed	13.0	39
Stuckenia pectinatus	Sago pondweed	1.0	3
Typha latifolia	Broad-leaved cattail	0.3	1
Vallisneria americana	Wild celery	0.3	1
Wolffia columbiana	Common watermeal	16.0	48
Eurasian watermilfoil	Eurasian watermilfoil	1.3	4
Potamogeton crispus	curly-leaf pondweed	0.7	2

Table 7: Norton Slough relative importance of aquatic plants

Species	Common Name	%Frequency	No. of Sites
Ceratophyllum demersum	Coontail	12.5	35
Chara	Muskgrasses	3.2	9
Elodea canadensis	Common waterweed	8.2	23
Elodea nuttallii	Slender waterweed	1.4	4
Heteranthera dubia	Water star-grass	0.4	1
Lemna minor	Small duckweed	13.2	37
Lemna trisulca	Forked duckweed	0.4	1
Myriophyllum sibiricum	Northern water-milfoil	0.4	1
Nuphar variegata	Spatterdock	1.8	5
Nymphaea odorata	White water lily	4.3	12
Potamogeton amplifolius	Large-leaf pondweed	1.1	3
Potamogeton foliosus	Leafy pondweed	1.1	3
Potamogeton fresii	Fries' pondweed	2.5	7
Potamogeton illinoensis	Illinois pondweed	0.4	1
Potamogeton nodosus	Long-leaf pondweed	3.2	9
Potamogeton oakesianus	Oakes' pondweed	1.1	3
Potamogeton obtusifolius	Blunt-leaf pondweed	1.8	5
Potamogeton praelongus	White-stem pondweed	0.7	2
Potamogeton pusillus	Small pondweed	1.8	5
Potamogeton richardsonii	Clasping-leaf pondweed	2.1	6
Potamogeton zosterformus	Flat-stem pondweed	7.8	22
Ranunculus aquaticus	White water crowfoot	6.0	17
Sparganium eurycarpum	Common bur-reed	0.4	1
Spirodela polyrhiza	Large duckweed	7.1	20
Stuckenia pectinatus	Sago pondweed	0.7	2
Utricularia vulgaris	Common bladderwort	0.4	1
Wolffia columbiana	Common watermeal	12.1	34

Table 8: Norton Slough aquatic vegetation summary statistics

Point Intercept Survey Parameters	Result
Total number of sites visited	60
Total number of sites with vegetation	59
Total number of sites shallower than maximum depth of plants	60
Frequency of occurrence at sites shallower than maximum depth of plants	98.3
Simpson Diversity Index	0.92
Maximum depth of plants (ft)**	9
Number of sites sampled using rake on Rope (R)	1
Number of sites sampled using rake on Pole (P)	59
Average number of all species per site (shallower than max depth)	4.7
Average number of all species per site (veg. sites only)	4.8
Average number of native species per site (shallower than max depth)	4.5
Average number of native species per site (veg. sites only)	4.7
Species Richness	30
Floristic Quality Index	32

Table 9: Jones Slough aquatic vegetation summary statistics

Point Intercept Survey Parameters	Result
Total number of sites visited	84
Total number of sites with vegetation	84
Total number of sites shallower than maximum depth of plants	84
Frequency of occurrence at sites shallower than maximum depth of plants	100.0
Simpson Diversity Index	0.87
Maximum depth of plants (ft)**	7
Number of sites sampled using rake on Rope (R)	0
Number of sites sampled using rake on Pole (P)	84
Average number of all species per site (shallower than max depth)	3.5
Average number of all species per site (veg. sites only)	3.6
Average number of native species per site (shallower than max depth)	3.4
Average number of native species per site (veg. sites only)	3.5
Species Richness	25
Floristic Quality Index	29

Table 10: 2014 and 2015 Nearshore Electroshocking Results for Jones Slough and Norton Slough

Species	Classification	Jones Slough	Jones Slough	Norton Slough	Norton Slough
		2014	2015	2014	2015
Central mudminnow		1	84	2	9
Grass pickerel			17		5
Lake chubsucker	Special Concern		23		4
Starhead topminnow	State Endangered	35		45	3
Pirate Perch	Special Concern	1	40	2	10
Bluegill		2		7	3
Warmouth sunfish		3	2	26	4
Green sunfish					
Largemouth bass		1			2
Mud darter	Special Concern	0	0	0	0

Table 11: Bakkens Pond and Long Lake Nearshore Fish Survey Results

Species	Classification	Bakkens Pond	Bakkens Pond	Long Lake	Long Lake
		2014	2015	2014	2015
Bowfin				1	
YOY gar				2	
Central mudminnow		15	8	4	
Grass pickerel				1	2
Bluntnose minnow	Tolerant				1
Yellow bullhead	Tolerant	1		1	1
Tadpole madtom				1	
Starhead topminnow	State Endangered	2		13	6
Pirate Perch	Special Concern			1	
Bluegill			38	28	26
Warmouth sunfish				3	5
Black crappie					1
Pumpkinseed			1		
Largemouth bass				1	
Mud darter	Special Concern	3			5
Least darter	Special Concern	3	4		
Iowa darter	Intolerant	2	1		7
Rainbow darter	Intolerant	2	2		1

Discussion

The SLOH and meter results for NOx demonstrated that the two datasets were not significantly different ($R^2 = 0.98$), as was the case during the Phase 1 part of the study ($R^2 = 0.95$). The well sampling indicated that deeper groundwater elevations contained greater nitrate concentrations. Shallow groundwater wells had lower nitrate concentrations that reflected clean recharge and plant uptake via root systems (Ranalli and Macalady 2010, Pfieffer et al. 2006, Mayer et al. 2006). Denitrification within the sand terrace is probably minimal given the relatively low organic matter content (Ranalli and Macalady 2010). High ammonium concentrations found in deep wells DR2, DR3, PR3 and JR2 may have reflected applied nitrification inhibitors applied to certain fields and not likely natural denitrification processes.

Quantitative nitrate loading to individual oxbows is unclear at this point without knowing what groundwater elevations that the oxbows intercept and what groundwater elevations pass beneath them. The results suggest that minimum buffer dimensions will have to be determined for each oxbow given multiple factors such as extent of relative polluted recharge area and local groundwater flow patterns. The proposed conservative minimum 100-meter buffer in the Phase 1 report was probably not sufficient given numerous agricultural nutrient sources that occur within the extensive sand terrace. At Long Lake, the approximate 3:1 terrace agricultural to buffer length ratio appears to significantly reduce nitrate concentrations in the groundwater but this ratio is probably unrealistic given that most of the land in the sand terrace are privately owned and are intensively farmed for row crops. The Long Lake wells contained the lowest NOx concentrations even though the wells lie within a private septic system field, as reflected by very high chloride concentrations. The volume of groundwater relative to surface water may also be a factor. Sustained winter open water areas in Jones Slough and Bakkens Pond may reflect greater groundwater discharge and therefore greater nitrate loading.

While both Jones Slough and Norton Slough function as the limnological definition of spring lakes (significant groundwater discharge and defined stream outlets) and are classified as spring lakes under the Lakes Management Program, the water quality criteria applied to these waterbodies is based on criteria developed for shallow lowland seepage lakes. Based on these criteria applied to six monthly phosphorus samples that were collected over two years, the lower 90% confidence interval was used to determine if phosphorus exceeded the impairment criteria of 40 ug/l for recreational use, and 100 ug/l for fish and aquatic life use in these lakes. Jones Slough phosphorus clearly exceeded the recreational use criteria with a mean value of 84.9 ug/l (CI, 52.5 to 137.2), but fish and aquatic life use criteria was met. By contrast, the mean phosphorus in Norton Slough was much lower at 44.5 ug/l (CI, 36.6 to 54.1) and clearly met both the recreation and fish and aquatic use impairment criteria.

The water quality criteria applied here represents shallow lowland lakes that typically intercept wetland drainage. These conditions only occur along the LWSR during infrequent brief periods of high river stages and may not be appropriate for assessing these spring lakes. Given the sand terrace aquifer driven hydrology that occurs most of the time, oligotrophic conditions would be expected if the landscape was unaltered. Near pristine conditions had occurred prior to recent increased nutrient applications over the sand terrace. Furthermore, lake criteria in general were developed using trophic state indicator (TSI) parameters and phosphorus as the limiting nutrient. These parameters do not match the type of eutrophication found in most oxbow lakes. As documented along the Mississippi River and elsewhere (Houser et al. 2013, Sullivan 2008, Giblin et al. 2010), eutrophication of oxbows and sloughs is often manifested as dense FFP mats and not phytoplankton that standard TSI parameters typically reflect. Integrated phosphorus and chlorophyll a samples will often miss surface blankets of FFP and filamentous algae that also growths on the bottom and on rooted plants. Secchi water clarity measurements can range from 0 to > 2 meters depending where the blankets of FFP mats occur within a single oxbow.

Nitrogen, in the inorganic form of NOx, instead of phosphorus is the primary cause for the oxbows eutrophication. Therefore, the standard phosphorus limited TSI criteria are of limited usefulness for assessing LWSR oxbow lakes. This finding is consistent with Conley et al. (2009), Lewis et al. (2011) and Howarth and Marino (2006) where nitrogen is at least as significant as phosphorus for controlling primary production and eutrophication.

Terrace groundwater phosphorus concentrations were relatively low (< 50 ug/l) compared with very high nitrate as the likely driver of eutrophication and environmental degradation in oxbow lakes. While high phosphorus concentrations were periodically detected in Jones Slough, internal loading linked to FFP cover and anoxia was likely a response to nitrogen loading and not the primary eutrophication driver. Bakkens Pond displays similar levels of eutrophication and FFP levels while phosphorus concentrations were much lower. Anoxia, and subsequent internal phosphorus loading, was not evident in Bakkens Pond at the monitoring sites, likely due to measurable water flow near the monitoring site.

At Norton Slough, the shallow lowland lake phosphorus criteria suggested that significant environmental degradation did not occur. However, recreational use and fish habitat impairments were evident due to thick mats of FFP. Blankets of FFP can significantly reduce water column dissolved oxygen levels, result in internal loading, reduce environmentally sensitive aquatic macrophytes and alter plankton communities (Fontanarrosa et al. 2010, Parr and Mason 2004), Janes et al. 1996, Houser et al. 2013). Mud darters in Jones Slough and Norton Slough appear to be impacted as well. High nutrient concentrations can result in a FFP alternative stable state in waterbodies previously dominated by rooted macrophytes (Scheffer et al. 2003). Surveys conducted by WDNR over a decade ago reported near pristine conditions in Jones Slough and other oxbows that lie adjacent to the sand terrace. The recent eutrophication appears to include a shift toward greater FFP cover and likely alternative stable state.

FFP cover on Long Lake remains low at around 5% or less even though mean lake nitrate concentrations are very high at 4.6 mg/l. Lower nitrate concentrations were frequently found in Jones Slough and Norton Slough except near the groundwater discharge locations. These differences may be related to plant uptake and densities are much lower in Long Lake. The dearth of FFP in Long Lake may also reflect lower nitrogen loading, as indicated by the well data, and greater fetch that limits quiet areas where FFP often proliferate. The much small surface areas of Jones Slough and Norton Slough, along with significant nutrient inputs, likely provide optimum conditions for FFP. Duckweeds are limited in the upper reaches of Bakkens Pond due to water movement but accumulate downstream in the impounded area.

Standard lake models typically reflect phosphorus loading associated with watershed land uses. The LWSR oxbows often have very small watersheds and the hydrology is dominated by groundwater. A groundwater model focusing on nitrate pollution appears to be more appropriate for this particular setting and is in the development stage.

Sullivan (2008) recommended water quality criteria of Mississippi River sloughs to include FFP cover. This approach appears to be reasonable given the impacts on ecology and recreational uses in the oxbows. Criteria should also include nitrogen and nitrates (Robertson et al. 2006, USEPA 2003) since nitrogen appears to be the primary driver of oxbow eutrophication.

The water quality decline in the Exceptional Resource Waters (ERW) classified oxbows is a recent event and in the public interest should be reversed. The following recommendations are designed to restore and protect LWSR oxbows:

Recommendations

1. Adopt the USEPA (2003) and Robertson et al. (2006) total nitrogen criterion of about 2 mg/l. That would also limit nitrate concentrations to 2 mg/l as recommended by Camargo et al. (2006),
2. Given that deeper groundwater elevations with high nitrates flow into oxbows but also beneath and ultimately discharge as springs along the Lower Wisconsin River, nutrient management planning should establish a goal of reducing groundwater nitrate concentrations below the drinking water standard of 10 mg/l, to protect the main channel, reduce nitrogen loading related to Gulf hypoxia, protect drinking water supplies and oxbow lakes.
3. Oxbows FFP cover should not exceed 40% to prevent impairments to recreational uses and ecology.
4. Extensive buffer zones ($\geq$ 300 meters) are needed to increase clean recharge and terrestrial plant uptake of nutrients. Expanding buffer areas also enhances wildlife habitat along the LWSR. Grassland grazing is one alternative to nutrient intensive row cropping beyond buffer zones may help achieve the goal of less than 10 mg/l nitrate in groundwater.
5. Norton and Jones Sloughs should be formally designated as critical habitat.

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From: John Sullivan <Irishvoyageur@aol.com>
Sent: Thursday, February 28, 2019 9:30 PM
To: Beranek, Ashley E - DNR; DNR Impaired Waters
Cc: 'Dave Marshall'; 'Jim Baumann'
Subject: [WARNING: ATTACHMENT(S) MAY CONTAIN MALWARE]2020 WisCalm Guidance Comments
Attachments: WisCalm Guidance Comments Sullivan Marshall Baumann Feb 28 2019.docx; Filamentous algae and Duckweed photos jfs.pdf; Methods for evaluating filamentous algae & duckweeds.doc

Follow Up Flag: Follow up
Flag Status: Flagged

Ashley,

Attached are comments concerning the use of free floating plants (filamentous algae and duckweeds) to evaluate the impacts of excessive nitrogen and phosphorus in surface waters of Wisconsin. I prepared this with consultation and support from Dave Marshall and Jim Baumann, former Water Resource employees for the Wisconsin Department of Natural Resources. We ask that you consider these recommendations for assessing water quality impairments in surface waters of Wisconsin by incorporating them into the 2020 WisCalm Guidance.

Thank you,

John F. Sullivan
349 24th St. S.
La Crosse, WI 54601
608-785-2194

Date: February 28, 2019

To: Ashley Beranek, Bureau of Water Quality, Madison, WI

From: John Sullivan (Irishvoyageur@aol.com)
349 24th St. S, La Crosse, WI 54601 (home phone: 608-785-2194)
Dave Marshall (underh2ohab@mhtc.net)
Jim Baumann (kayakerjb@gmail.com)

Subject: Use of Free Floating Plants (filamentous algae & duckweeds) in WisCalm Guidance

Nutrient impairments in Wisconsin's waters have been identified as a serious problem impacting aquatic life and recreational uses as identified in Wisconsin's Impaired Waters Listings (Section 303d of the Clean Water Act). Wisconsin has taken critical steps to address nutrient-related problems with the adoption of water quality standards for phosphorus in December 2010. Although phosphorus has been identified as a key limiting nutrient, there is growing information indicating that excessive nitrogen may play an equally important role in nutrient impairments in surface waters. This has been especially identified in the Gulf of Mexico where hypoxic conditions (low dissolved oxygen) have been attributed to excessive nitrogen inputs. EPA, in working with States and other Federal agencies, developed a national goal to reduce nitrogen and phosphorus loadings to the Gulf by 45% (2008 Gulf Hypoxia Plan). Wisconsin and other States have developed strategies to achieve these goals and have identified mechanisms for load reductions from point and nonpoint sources. Monitoring and research on the Upper Mississippi River (UMR) and lower Wisconsin Rivers have also identified nitrogen or nitrogen and phosphorus (co-limitation) as the primary nutrients contributing to excessive growths of metaphyton (filamentous algae) and duckweeds in backwaters, sloughs and floodplain lakes (Sullivan 2008, Sullivan & Giblin 2012, Giblin et al. 2014 and Marshall 2013).

Although nutrient reduction strategies are an appropriate initial step to address the Gulf Hypoxia problem, EPA also recognizes a critical need for states to adopt numeric nutrient criteria for phosphorus and nitrogen for surface waters. Many states have or are working towards the adoption of surface water criteria for phosphorus to address eutrophication problems. However, there has been limited progress by states in the development of numeric nitrogen criteria even though initial EPA guidance for nitrogen criteria were proposed almost twenty years ago (USEPA 2000a & 2000b).

A critical step in the discovery of nutrient-related water quality problems is the identification of key biological response variables that are directly influence by excessive nutrient inputs. The use of algal chlorophyll concentrations, harmful algae bloom frequency, cyanotoxins, algae cell counts and other metrics are clear examples. Another useful indicator of nutrient enrichment is the development of excessive filamentous algae (metaphyton) mats or thick coverings of duckweeds that develop in shallow aquatic systems including channel borders, backwaters, floodplain lakes and deep water wetlands (Sullivan 2008, Houser and Richardson 2010, Giblin et al. 2014). Floating mats of filamentous algae or duckweeds are free floating plants (FFP) that can negatively impact dissolved oxygen levels or contribute to significant shading of submersed aquatic vegetation beds through the attenuation of surface light. Thick growths of filamentous algae on submersed vegetation may negative impact submersed vegetation due to the competition for nutrients, dissolved gasses (O^2 & CO^2) and light and may contribute to a complete collapse of these important aquatic plant communities (Phillips et al. 1998 and Hilton et al. 2006). This was likely a factor in the massive decline of submersed aquatic vegetation in the UMR in the late 1980s and has recently been observed in floodplain lakes in the Lower Wisconsin River (Marshall 2013). Further, thick mats of filamentous algae or duckweeds seriously impact recreation use by making these areas difficult or impossible to traverse with a boat, especially paddlers. Example photos showing excessive growths of these floating plants in aquatic areas of the Mississippi River are included in the attached file.

The Department needs to consider the impacts of nuisance growths of filamentous algae and duckweeds in the assessment of water quality use attainment. Fortunately, procedures have been developed (see attached file) and implemented to facilitate this process (Sullivan 2008, Marshall 2013 and Houser et al. 2014). However, specific impairment thresholds using FFP water quality indicators have not been adopted by the Department. We have drafted methodologies for identifying nutrient-related impairment problems using FFPs as a response factor (see table below). A tired approach is recommended that would consider differences in surface water and use classification. We would urge the Department to consider these recommendations for Wisconsin's Consolidated Assessment and Listing Methodology for Clean Water Act Reporting.

If you have any questions concerning this recommendation or need additional information, please contact John Sullivan.

Proposed nutrient-related water quality impairment critieria for free floating plants (FFP) including metaphyton (filamentous algae) and duckweeds.

Tier	Measurement	Surface Water & Use Class *	Basis
I	FFP Cover > 20% & Biomass > 10 g dry wt m^{-2}	ORW & ERW Waters Rivers, Streams, Lakes and associated waters supporting fish & aquatic life	Protect fish & aquatic life including rare or unique aquatic plant communties
II	FFP Cover > 40% & Biomass > 25 g dry wt m^{-2}	Rivers, Streams, Lakes and associated waters supporting fish & aquatic life	Light Shading of SAV Reduced surface re-areation Recreational use impacts
III	FFP Cover > 60% & Biomass > 30 g dry wt m^{-2}	Deep Water Marshes supporting seasonal fish & aquatic life use	Light Shading of SAV Severe reduction in surface re-areation

*Water depths > 0.5 m supporting submersed aqautic vegetation (SAV).
ERW - Exceptional Resource Waters
ORW - Outstanding Resource Waters

Additional Notes:

The 10 g dry wt m^{-2} critieria for FFP biomass is point where surface light is reduced by more than 40%.

The 25 g dry wt m^{-2} critieria for FFP biomass is point where surface light is reduced by more than 80%.
Duckweed cover > 60% associated mid-day dissolved oxygen < 5 mg/L.

The 30 g dry wt m^{-2} critieria for FFP biomass is point where surface light is reduced by more than 90%.
Duckweed cover > 60% associated mid-day dissolved oxygen < 3 mg/L.

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Weaver Bottom, Pool 5 – Filamentous algae

Lake Onalaska Pool 7 – Filamentous algae

Lake Onalaska Pool 7 – Filamentous algae

Lower Pool 8 off Raft Channel – Filamentous algae

Pool 9 Above Lynxville, WI – Filamentous algae

Long Lake Pool 7 – July 2010 - Duckweed

Trempealeau Marina – Pool 6 – July 2010

Stoddard Wetland – Pool 8 – August 2005 (near Stoddard WWTP Outfall)

Blue Lake – Pool 8 – Fall 2004 (LTRMP Survey) - Duckweed

Blue Lake – Pool 6 – Oct 2004 (near La Cresent WWTP outfall) - Duckweed

Broken Arrow Slough – Pool 8 – July 2009- Duckweed

Method for Estimating the Abundance of Filamentous Algae and Duckweeds

The purpose of this monitoring is to generally quantify the percent cover and relative abundance (density) of filamentous algae and duckweeds in aquatic areas. These aquatic plants are common in aquatic systems but can reach nuisance levels especially when stimulated by a constant source of nutrients (phosphorus and nitrogen) and favorable habitat such as quiescent areas including aquatic plant beds. Excessive filamentous algae or duckweed may negatively impact submersed aquatic plants directly as a result of light limitation (shading) or indirectly by competing for nutrients or dissolved gasses (oxygen and carbon dioxide) through respiratory processes.

This monitoring activity will help us better understand factors contributing to abundance and distribution of this plant community and to help us evaluate the potential impacts on fish and aquatic life and general water quality use (boating and swimming).

Filamentous algae – A mass of hair-like strands of slimy aquatic plant material that is found attached to hard surfaces including sediments, plants, rocks, woody debris or found floating at the surface. The color may vary from yellow, yellowish-green, bluish-green to dark green. Common names include pond-scum or moss.

Duckweeds – A tiny aquatic plant having a round or oval shape that is typically found floating at the surface in quiescent areas protected from wind and waves. Some forms may be found below the surface intertwined in submersed aquatic vegetation. The size can ranged from less than a millimeter to about ten millimeters in diameter. The plant is typically pale green to green in color but certain forms may be reddish-purple.

Percent Cover Rating - A scoring system to generally quantify the percentage of the water surface area or aquatic plant bed that is "covered" by filamentous algae or duckweeds. Confine this assessment to within 25m (~80 ft) of the sampling site.

Score	Definition
0	**0% - No filamentous algae or duckweeds are present**
1	**1-20% of the area is covered**
2	**21-40% of the area is covered**
3	**41-60% of the area is covered**
4	**61-80% of the area is covered**
5	**81-100% of the area is covered**

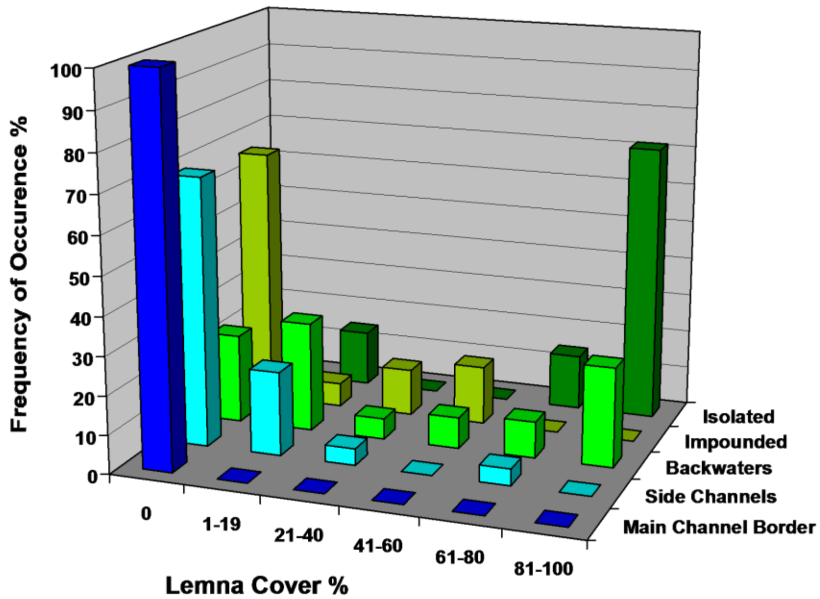
Relative Plant Density – A scoring system to generally classify the abundance or thickness of the filamentous algae or duckweed mat.

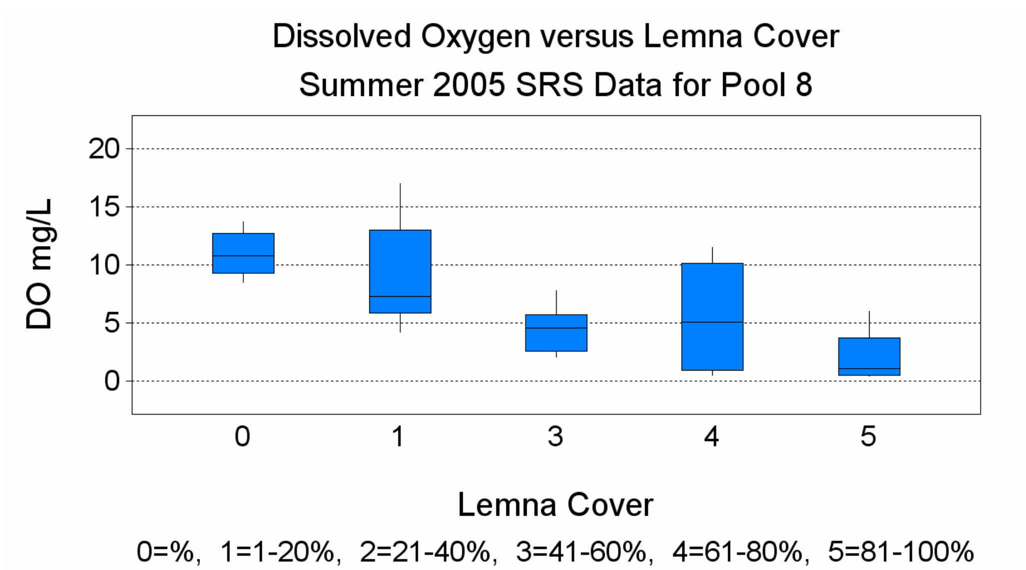
Score	Definition
0	**No filamentous algae or duckweeds are present**
1	**A thin to moderate layer of filamentous algae or duckweeds**
2	**A dense mat of filamentous algae or duckweed exceeding a thickness of ¼ inch (5 mm)**

Example Data Sheet Layout

Site	Date	Water Depth (ft)	Filamentous Algae Cover Score	Filamentous Algae Density Score	Duckweed Cover Score	Duckweed Density Score	Comments

Example of data collected by Wisconsin's LTRMP Field Station at La Crosse using the above methods during summer 2005 stratified random sampling of Pool 8 (Jim Fischer & Kraig Hoff, WDNR). Figures prepared by John Sullivan, WDNR.





Beranek, Ashley E - DNR

From: Giblin, Shawn M - DNR
Sent: Monday, February 18, 2019 3:53 PM
To: Beranek, Ashley E - DNR
Cc: Willhite, Marcia T - DNR
Subject: WisCALM Comments
Attachments: Giblin_2014_Final.pdf; Giblin_2017.pdf

Follow Up Flag: Follow up
Flag Status: Flagged

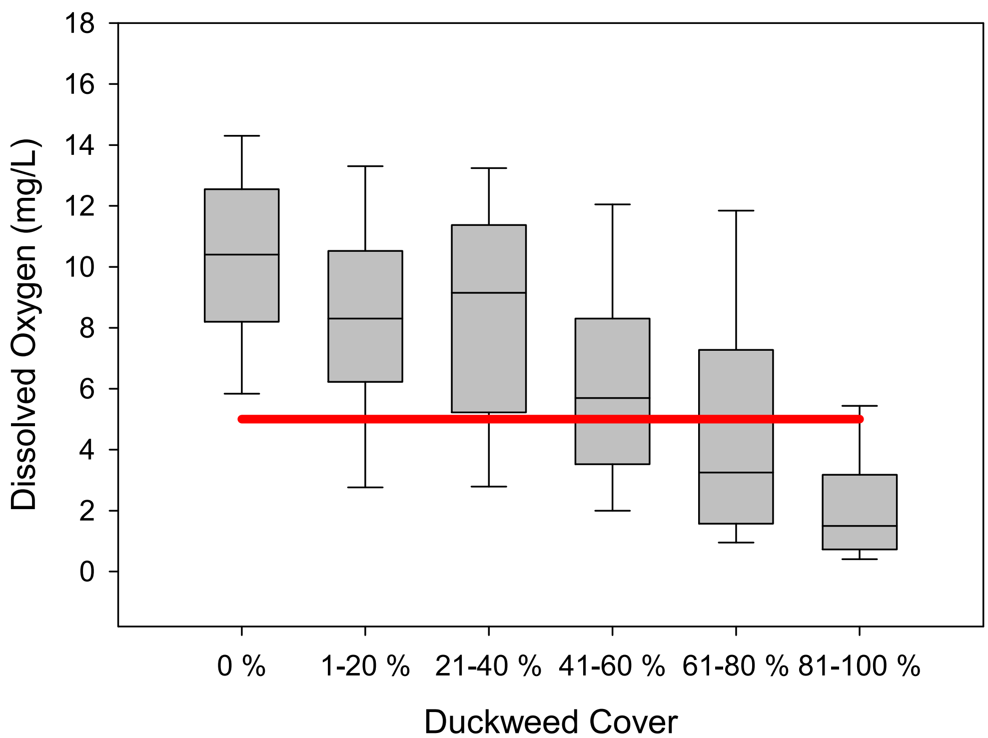
Hi Ashley and Marcia,

My WisCALM comments/questions are as follows:

- I would have an interest in seeing criteria related to duckweed and filamentous algae mats on backwaters of the Mississippi (and likely other large-river backwaters of the state). These would be based on the attached paper (Giblin et al 2014). These mats are proliferate in response to both N and P. Examination of tissue nutrient ratios suggest that the mats are P-limited early in the growing season and N-limited later in the growing season. One thing we observe on the UMR is that when duckweed coverage is > 80% backwater hypoxia is very likely- see plot below. It seems that we should have criteria for this impairment. These mats can cover up to 50% of the available backwater area under any given year.
- I would also like to see us develop TSS standards for large rivers in WI based on the attached paper (Giblin 2017). On the UMR, we see a threshold at 16 mg/L TSS. At TSS > 16 mg/L we see reduced biomass of native and recreational fishes.
- Page 58- Will bacteria standards for the UMR switch to *E. Coli* at some point? We have many beaches on the Mississippi River.
- General Question: What about rivers that exceed CHLa of 20 ug/L (Miss. R., La Crosse R., Red Cedar R., etc.)? Are these assumed to be covered under the TP exceedance?

Thanks,

Shawn



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Shawn Giblin
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From: Beranek, Ashley E - DNR
Sent: Monday, January 28, 2019 2:14 PM
To: Giblin, Shawn M - DNR <Shawn.Giblin@wisconsin.gov>
Cc: Willhite, Marcia T - DNR <Marcia.Willhite@wisconsin.gov>
Subject: RE: 303d/305b Updates

Hi Shawn,

There aren't any new assessment results since your last meeting (Sept. 2018) – we'll be starting new assessments in February. We are currently in the process of soliciting public comments on our 2020 WisCALM and requesting public data submittals (https://dnr.wi.gov/topic/surfacewater/assessments.html), open until March 1st. We welcome any comments team members have on WisCALM and if they have any external partners that would like to submit data for us to use in 2020 assessments we recommend they let them know about this process.

I will be getting in contact with you later in February to go over assessments for the Mississippi.

Thanks!
~Ashley

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From: Giblin, Shawn M - DNR
Sent: Monday, January 28, 2019 1:15 PM
To: Beranek, Ashley E - DNR <Ashley.Beranek@wisconsin.gov>
Subject: 303d/305b Updates

Hi Ashley,

Can you provide some bullet points for 303d/305b updates? This will be a topic at the WQTF meeting tomorrow at 1 PM. Items related to nutrients/chloride would be especially useful.

Thanks,

Shawn

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ARTICLE

Thresholds in the Response of Free-Floating Plant Abundance to Variation in Hydraulic Connectivity, Nutrients, and Macrophyte Abundance in a Large Floodplain River

Shawn M. Giblin · Jeffrey N. Houser · John F. Sullivan · Heidi A. Langrehr · James T. Rogala · Benjamin D. Campbell

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Abstract Duckweed and other free-floating plants (FFP) can form dense surface mats that affect ecosystem condition and processes, and can impair public use of aquatic resources. FFP obtain their nutrients from the water column, and the formation of dense FFP mats can be a consequence and indicator of river eutrophication. We conducted two complementary surveys of diverse aquatic areas of the Upper Mississippi River as an in situ approach for estimating thresholds in the response of FFP abundance to nutrient concentration and physical conditions in a large, floodplain river. Local regression analysis was used to estimate thresholds in the relations between FFP abundance and phosphorus (P) concentration (0.167 mg l^{-1}), nitrogen (N) concentration (0.808 mg l^{-1}), water velocity (0.095 m s^{-1}), and aquatic macrophyte abundance (65 % cover). FFP tissue concentrations suggested P limitation was more likely in spring, N limitation was more likely in late summer, and N limitation was most likely in backwaters with minimal hydraulic connection to the channel. The thresholds estimated here, along with observed patterns in nutrient limitation, provide river scientists and managers with criteria to consider when attempting to modify FFP abundance in off-channel areas of large river systems.

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Keywords Mississippi River · Free-floating plants · Duckweed · Nitrogen · Phosphorus · Connectivity

Introduction

Free floating plants (FFP) are common in aquatic ecosystems and, when conditions are favorable, can form thick surface mats that substantially affect ecosystem processes and condition (Parr and Mason 2004). Such surface mats are often dominated by duckweeds (*e.g.*, *Lemna* spp.) and may contain filamentous algae (*e.g.*, *Cladophora* spp.) and other species. Specific ecosystem effects of abundant FFP can include reductions in the following: dissolved oxygen concentration (Pokorny and Rejmankova 1983), phytoplankton growth rate and abundance (O'Farrell et al. 2009), and zooplankton growth rate and abundance (Fontanarrosa et al. 2010). Abundant FFP have also been associated with increased sediment oxygen demand (Parr and Mason 2004), and increased sediment nutrient release (Boedeltje et al. 2005). In addition, thick mats of abundant FFP can interfere with public recreation (Hall and Cox 1995), provide minimal benefits for invertebrates (Neill and Cornwell 1992), negatively affect fish and wildlife (Parr and Mason 2004), and can reduce submersed aquatic vegetation (SAV) abundance due to reduced light penetration (Portielje and Roijackers 1995; Morris et al. 2003).

Free floating plants are capable of growing and reproducing rapidly under favorable conditions. FFP obtain nutrients solely from the water column, and can respond rapidly to sustained nutrient availability (Landolt and Kandeler 1987). Specifically, elevated phosphorus (P) and nitrogen (N) concentrations have been associated with high FFP biomass (De Groot et al. 1987; Luond 1990; Szabo et al. 2005; Makarewicz et al. 2007). Favorable environmental conditions for FFP include warm water temperature (Landolt and Kandeler 1987), shallow water depth (Janse and Van Puijenbroek 1998), low water velocity (Duffield and Edwards 1981), and low pH (Szabo et al. 2005). Additionally, the presence of rooted aquatic macrophytes (submersed, rootedfloating-leaved, and emergent), which act as a substrate to hold FFP in place, has been associated with high FFP biomass (McDougal et al. 1997).

The concentration of N and P varies spatially and temporally in large floodplain river ecosystems (Knowlton and Jones 1997; Tockner et al. 1999). In the Upper Mississippi River, total N and P concentration tend to decline and increase through the growing season, respectively (Houser and Richardson 2010). During the growing season, N concentration tends to be highest in the main channel and, among backwaters, N concentration tends to be higher in backwaters which are more connected to the main channel (Richardson et al. 2004). Nutrient sediment release is an important source of P input to the water column during the growing season in the Upper Mississippi River (James and Barko 2004; Houser et al. 2013).

Because FFP are dependent on water column nutrient concentrations, it is possible that the spatio-temporal variability in nutrient availability leads to spatio-temporal variability in nutrient limitation of FFP. FFP tissue nutrient concentration and stoichiometry can indicate which nutrients are most likely limiting growth (Sterner and Elser 2002; Klausmeier et al. 2004; Hall et al. 2005; Demars and Edwards 2007). Minimum water column nutrient concentrations required to maintain tissue nutrients, and therefore continuous growth of FFP, have been identified in a laboratory setting (Roijackers et al. 2004 and references therein). Additionally, tissue nutrient ratios can indicate which nutrient can be limiting FFP growth or abundance (Demars and Edwards 2007).

The Upper Mississippi River has experienced severe proliferation of FFP biomass in recent years. As a result, large areas of backwater habitat have been covered by FFP mats, which reduce surface re-aeration and photosynthesis and generate hypoxic conditions (Houser and Richardson 2010). To improve ecosystem function, it is important for river scientists to understand factors driving FFP production in floodplain rivers so management strategies can be developed to prevent the proliferation of these extensive mats.

Because the Upper Mississippi River exhibits extensive variation in nutrient availability, hydraulic connectivity, current velocity, depth, and vegetation abundance, the specific factors limiting FFP abundance likely vary spatially and temporally across the aquatic areas of the floodplain. Thus, the Upper Mississippi River provides a useful system for estimating thresholds in the response of FFP to a suite of chemical and physical factors in a large floodplain river. We address the following questions: 1. Do the changes in nutrient concentrations across gradients of hydraulic connectivity and season result in temporal and spatial patterns in FFP tissue nutrient content and nutrient limitation of FFP? 2. What thresholds can be detected in the relations between FFP biomass, water velocity, water column N and P concentrations, and aquatic macrophyte cover?

Methods

Study Area

The Upper Mississippi River consists of a series of navigation pools extending from Minneapolis, Minnesota to the confluence of the Ohio River at Cairo, Illinois, USA. The navigation dams 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 discharge or water level during high flow and flood conditions (Sparks 1995). Thus navigation pools are unlike reservoirs in that they remain riverine in nature. The study was conducted in Navigation Pool 8 of the Upper Mississippi River. Pool 8 is located between Lock and Dam 7 (Dresbach, MN, USA) and Lock and Dam 8 (Genoa, WI, USA; Fig. 1). It is 39 km long and encompasses 9,000 aquatic ha. Pool 8, typical of the navigation pools of the Upper Mississippi River, is composed of a diverse array of aquatic areas (Wilcox 1993) and has been stratified for sampling purposes into main channel, side channel, contiguous backwaters, isolated backwaters, and impounded areas (Soballe and Fischer 2004). The main channel is deep and is characterized by relatively high water velocity (0.20–0.60 m s^{-1}). Side channels are lotic but exhibit depth and water velocity that is generally less than that of the main channel. Contiguous backwaters typically exhibit very low water velocity (often below detection) and are connected by surface water to main or side channel habitat at normal river stage. Isolated backwaters typically exhibit undetectable water velocity and lack a surface water connection to channel habitat at normal river stage. The impounded area is the large expanse of open water located directly upstream of the lock and dam. The Upper Mississippi River is highly modified for navigation and is unusual among large rivers in that the contiguous backwaters retain surface water connections 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

The study design consisted of two complementary components. The connectivity component consisted of monthly (May–October 2010) sampling at five locations in each of two high connectivity (to the main channel) backwaters (Round and Horseshoe Lakes) and two low connectivity backwaters (Markle and Beiers Lakes) (Figs. 1a and 2a). Data from the connectivity component were used to describe

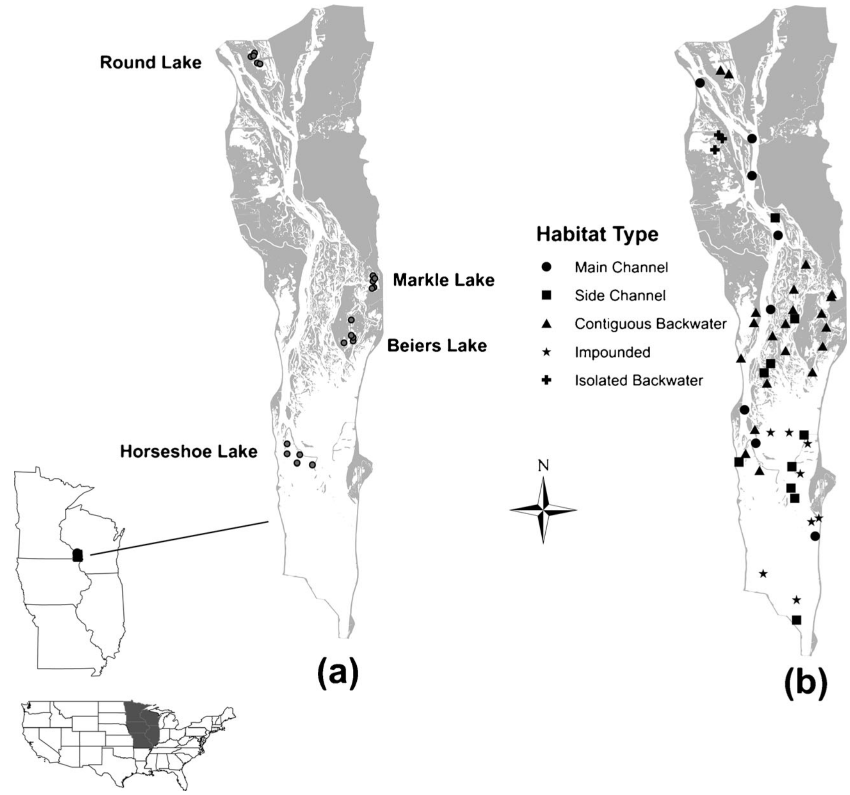

Fig. 1 Location of sampling sites within Navigation Pool 8 of the Upper Mississippi River for **a** the connectivity sampling component, and **b** stratified random sampling (SRS) component. *Grey* indicates land, white indicates water

temporal trends and estimate thresholds for FFP biomass response. All statistical analyses were conducted using the connectivity component data unless otherwise stated.

The stratified random sampling component consisted of 50 sites distributed across main channel (n=8), side channel (n=10), impounded (n=8), contiguous backwater (n=21), and isolated backwater (n=3) strata that were sampled during a 2-week period beginning the last week of July (Figs. 1b and 2a). The data from this design were collected as ancillary data in conjunction with routine data collection by the U.S. Army Corps of Engineers' Upper Mississippi River Restoration-Environmental Management Program-Long Term Resource Monitoring Program element. The stratified random sampling sites spanned a broad range of hydraulic connectivity, but were restricted in temporal extent. The differences among sample sizes for different strata resulted from a compromise between the portion of Pool 8 encompassed by that strata and variability within the strata (i.e. more limnological variability exists among the contiguous backwaters than within the main channel). Data from the stratified random sampling component were used to test the threshold estimates derived from the connectivity component data.

Water Sampling

Water samples were taken at a depth of 0.20 m at each site to assess water column total nitrogen (TN) and total phosphorus (TP). 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). Measurements of water depth and water velocity (Marsh-McBirney, model 2000, Flo-Mate, Frederick, MD, USA) were collected at each site. Water temperature measurements were taken at 0.20 m using a multiparameter sonde (Minisonde MS5, Hach Company, Loveland, CO, USA). Discharge data were collected by the U.S. Army Corps of Engineers at Lock and Dam 8, Genoa, WI, USA (www.mvp-wc.usace.army.mil). Further details regarding field methods can be found in Soballe and Fischer (2004).

Free-Floating Plant Biomass and Aquatic Macrophyte Cover Sampling

FFP samples for the determination of biomass, dominant taxa and tissue C, N and P content were collected from a 0.25 m^2 quadrat on a randomly selected side of the boat (center-starboard, center-port) at each sampling site. The dominant FFP taxon of the sample was recorded based on visual inspection. Dry weight biomass (g m^{-2}) was determined by drying the FFP samples at 80 °C for 48 h and weighing. FFP samples were then ground with a Wiley Mill (Thomas Scientific, Swedesboro, NJ, USA) to pass a 1 mm sieve and then further ground in a ring and puck grinder to pass a 100 mesh sieve (0.15 mm) and analyzed for C, N and P concentrations. A LECO CNS-2000 analyzer (St. Joseph, MI, USA) was used to

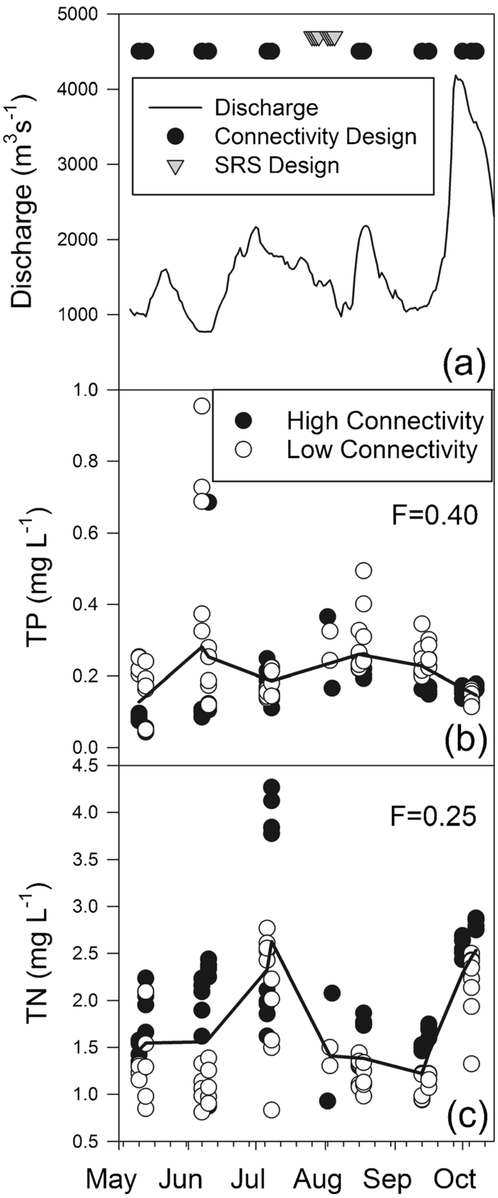

Fig. 2 **a** Discharge (m^3 s^{-1}) at Lock and Dam 8 during the sampling period. The symbols represent sampling days for the connectivity and stratified random sampling components. **b** Water column total phosphorus concentration at high and low connectivity backwater sampling sites by date from the connectivity sampling component. The *solid line* indicates the LOESS regression trend with individual smoothing parameter (F) and **c** water column total nitrogen

analyze FFP C and N by dry combustion with infrared detection of CO_2 plus thermal conductivity detection of N_2 in combustion gases. FFP tissue P concentration was analyzed as part of the total mineral analysis. Total mineral analysis included the digestion of the plant material in concentrated nitric acid with heating at 120–130 °C for 14 h followed by further digestion with two cycles of 0.5 mL of 30 % hydrogen peroxide with heating to 120–130 °C for 30 min and final dilution to 50 mL. The determination of the elements in the diluted digest was performed using a Thermo Jarrel Ash IRIS Advantage ICP-OES (Inductively Coupled Plasma Optical Emission Spectrometry; Franklin, MA, USA). FFP tissue and water column nutrient ratios (N:P, C:N, C:P; by moles) were determined as molar ratios.

Aquatic macrophyte cover at each site was estimated visually (to the nearest 5 %) for the area within 25-m of the boat. Cover was estimated separately for rooted floating-leaved, emergent, and SAV aquatic macrophyte life forms and then summed to generate an aquatic macrophyte score (Yin et al. 2000). It was possible for all three macrophyte life forms to overlap, therefore total cover score sometimes exceeded 100 %.

Backwater Connectivity

An index of backwater connectivity (connectivity score) was developed using water velocity data from the summer, fall and winter from 1993 to 2008 that was collected by the Upper Mississippi River Restoration- Environmental Management Program-Long Term Resource Monitoring Program. Mean water velocity for each backwater was used as an indicator of backwater connectivity (Horseshoe Lake, mean velocity= 0.112 m s^{-1}, n=102; Round Lake=0.048 m s^{-1}, n=48; Markle Lake=0.014 m s^{-1}, n=35; Beiers Lake=0.008 m s^{-1}, n=53).

Statistical Analyses

Spearman rank correlation (R_s) was used to assess the relations between water column and FFP tissue nutrient concentrations and ratios and to describe temporal trends (correlation with day of year) in nutrient concentrations and ratios through the growing season as described by Rodrigues and Williams (2002). Non-parametric Mann-Whitney rank sum tests were used to evaluate differences in water column and FFP tissue N and P concentration among the high and low connectivity backwaters. Thresholds in the relations between FFP abundance and various covariates were estimated using a nonparametric method for estimating local regression surfaces (PROC LOESS; SAS 9.2, 2008). The regression smoothing parameter was generated by default in SAS to strike a balance between residual sum of squares and the complexity of the fit. The PROC LOESS output was used to detect abrupt breakpoints, or thresholds, in the relations between selected variables. Once the thresholds were identified, linear or nonlinear regression was performed to estimate the x-axis intercept for select environmental variables.

Results

Seasonal Patterns in Discharge, Nutrients, and FFP Biomass

Discharge was characterized by three spring and summer peaks followed by a major flood event in late-September into early-October (Fig. 2a). P concentration was variable over the growing season with notably high concentrations occurring in June (Fig. 2b). N concentration peaked in July, decreased from July–September, and increased again in October in association with the fall flood (Fig. 2c).

There were significant temporal trends in FFP biomass and tissue nutrient concentration and ratios through the growing season. Median (across all sites) FFP biomass was very low in late-spring and early-summer, reached a maximum in August, declined in September and disappeared in October as a result of fall flooding (May, 0 g dry mass m^{-2}; June, 3.45; July, 11.39; August, 46.79; September, 10.57; October, 0). Tissue P increased (R_s= 0.57, n=74, P<0.001) and tissue N decreased (R_s=−0.37, n=74, P=0.001) during the growing season at sites where sufficient FFP biomass was collected (~60 % of sites) to measure tissue nutrient concentrations (Fig. 3). Tissue N:P (R_s=−0.68, n=74, P<0.001) (Fig. 4a) and tissue C:P (R_s= −0.47, n=74, P<0.001) (Fig. 4b) decreased during the growing season. Tissue C:N, increased, albeit weakly, during the growing season (R_s=0.41, n=74, P<0.001) (Fig. 4c). Tissue N:P ratios generally indicated an excess of P relative to N by the criteria developed for aquatic angiosperms (N:P<24:1) (Demars and Edwards 2007).

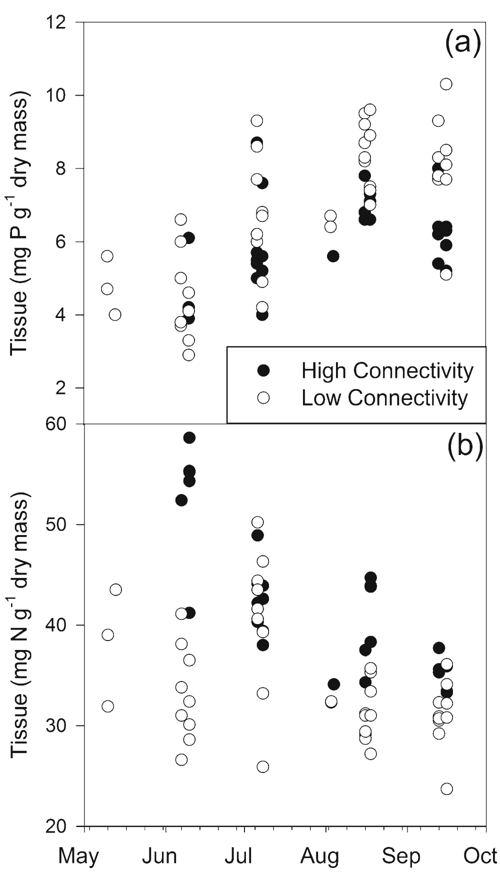

Fig. 3 **a** Free-floating plant (FFP) tissue phosphorus and **b** nitrogen concentration by date from the connectivity sampling component

N:P ratios high enough to suggest P limitation were only observed in the highest connectivity backwater during June (Fig. 4a).

Connectivity, Nutrients, and FFP Biomass

Median water column P concentrations were higher in the low connectivity backwaters (0.22 and 0.163 mg l^{-1} in the low and high connectivity backwaters, respectively) (Fig. 2b); the distributions of the two groups differed significantly (Mann-Whitney test, T=2997, n_1=n_2=62, P<0.001). Median N concentrations were lower in the low connectivity backwaters (1.293 and 1.88 mg l^{-1} in the low and high connectivity backwaters, respectively) (Fig. 2c); the distributions of the two groups differed significantly (Mann-Whitney test, T= 4851.5, n_1=n_2=62, P<0.001).

There were significant correlations between water column nutrient concentrations and FFP tissue nutrient concentrations and ratios. Tissue P was positively correlated with water column TP (R_s=0.30, n=74, P=0.01) (Fig. 5a). Tissue N was positively correlated with water column TN (R_s=0.65, n=74, P<0.001) (Fig. 5b). Tissue N:P was significantly correlated with both water column TP (R_s= −0.49, n=74, P<0.001) (Fig. 5c) and TN (R_s=0.37, n=74, P=0.001) (Fig. 5d). Tissue samples were generally dominated by duckweed species (>90 % of samples). Dominant taxa observed among the tissue samples included: *Lemna minor* (n=35), *Spirodela polyrhiza* (n=23), *Lemna trisulca* (n=7), *Wolffia columbiana* (n=3), *Cladophora* sp. (n=5), and *Spirogyra* sp. (n=1).

There were substantial trends in FFP biomass and tissue nutrient concentrations and ratios across the connectivity gradient most likely reflecting differences in water column nutrient availability across the connectivity gradient. FFP biomass was notably lower in the two higher connectivity backwaters compared to the two lower connectivity backwaters (Fig. 6). Median tissue P concentrations were lower in the high connectivity backwaters (7.0 and 5.9 mg P g^{-1} in the low and high connectivity backwaters, respectively) (Fig. 3a); the distributions of the two groups differed significantly (Mann-Whitney test, T=975, n_1=31 n_2=43, P=0.04). Median tissue N concentrations were higher in the high connectivity backwaters (32.3 and 40.3 mg N g^{-1} in the low and high connectivity backwaters, respectively) (Fig. 3b); the distributions of the two groups differed significantly (Mann-Whitney test, T=1572, n_1=31 n_2= 43, P<0.001). Low N:P ratios, suggestive of N deficiency, were observed more often in low connectivity backwaters and high N:P ratios (>24:1), suggestive of P deficiency, were observed more often in high connectivity backwaters (Fig. 7).

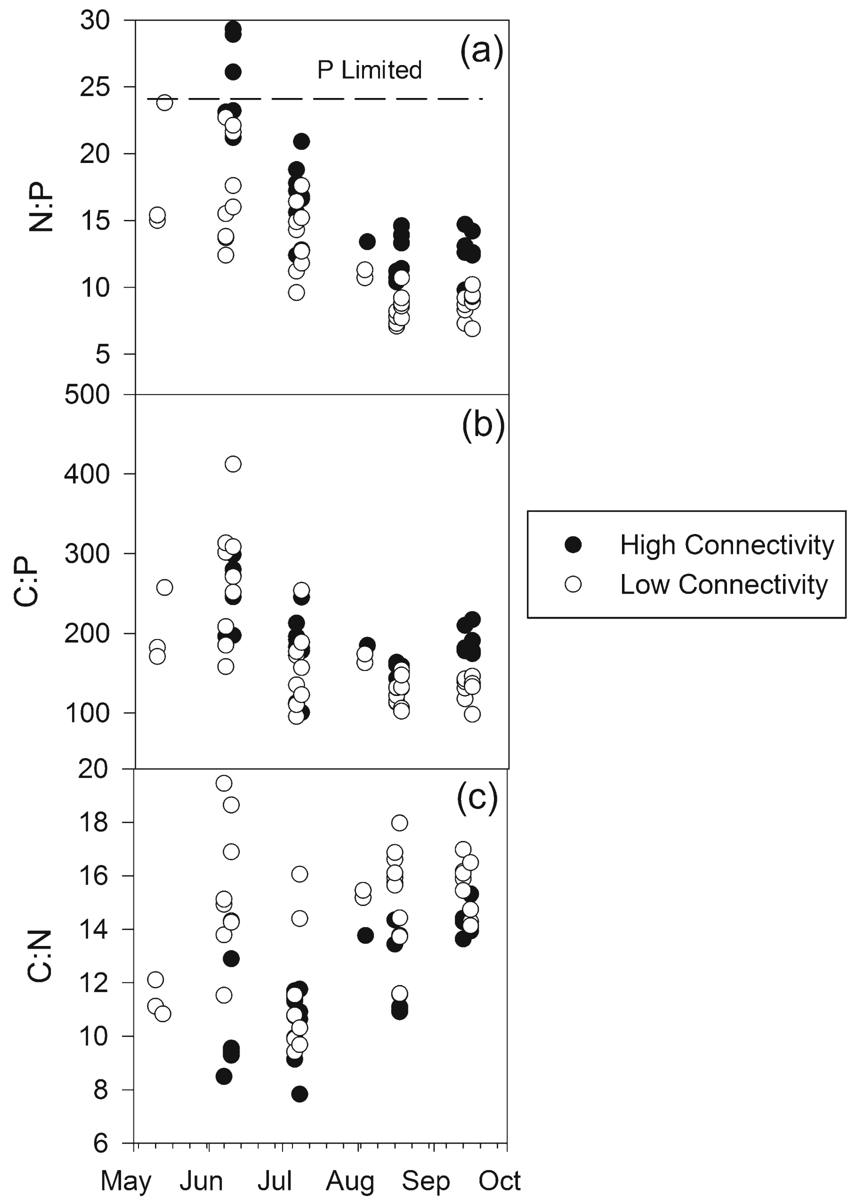

Fig. 4 **a** Free-floating plant (FFP) tissue N:P by date from the connectivity sampling component. Values above the dashed line are suggestive of P limitation for aquatic angiosperms (Demars and Edwards 2007; critical N:P value 24:1). **b** FFP tissue C:P by date from the connectivity sampling component. **c** FFP tissue C:N by date from the connectivity sampling component

Thresholds for FFP Biomass Response to Environmental Conditions

Thresholds were detected in the relations between FFP and water column TP, water column TN, water depth, water velocity, and aquatic macrophyte cover. FFP biomass increased gradually from 0.043 mg l^{-1} to a threshold at 0.167 mg l^{-1} TP above which biomass increased rapidly until TP concentration reached 0.25 mg l^{-1} (Fig. 8a; Table 1). Above 0.25 mg l^{-1} TP, FFP biomass declined with increasing TP concentration. FFP biomass increased gradually as TN concentration increased from 0.808 mg l^{-1} (minimum observed in this study) to 1.308 mg l^{-1} (Fig. 8b). Above this threshold, FFP biomass decreased with increasing TN concentration. The absence of TN values below 0.808 mg l^{-1} prevented the calculation of the x-axis intercept using the LOESS procedure. Although the x-axis intercept is below 0.808 mg l^{-1}, a precise estimate could not be calculated due to limitations in the data distribution.

No thresholds were detected in the relation between FFP biomass and water temperature, and the estimated x-axis intercept was 13.14 °C (Fig. 8c). The fitted relation between water depth and FFP biomass peaked at 1.52 m (Fig. 8d). The fitted relation between water velocity and FFP biomass exhibited a steep negative slope from 0 to 0.095 m s^{-1}, reflecting that most sites with detectable FFP exhibited water velocities of about 0 m s^{-1}; at sites where water velocity was greater than 0.095 m s^{-1}, FFP biomass was nearly always 0 (Fig. 8e).

The fitted relation between FFP biomass and aquatic macrophyte cover exhibited two thresholds (Fig. 8f). From minimal

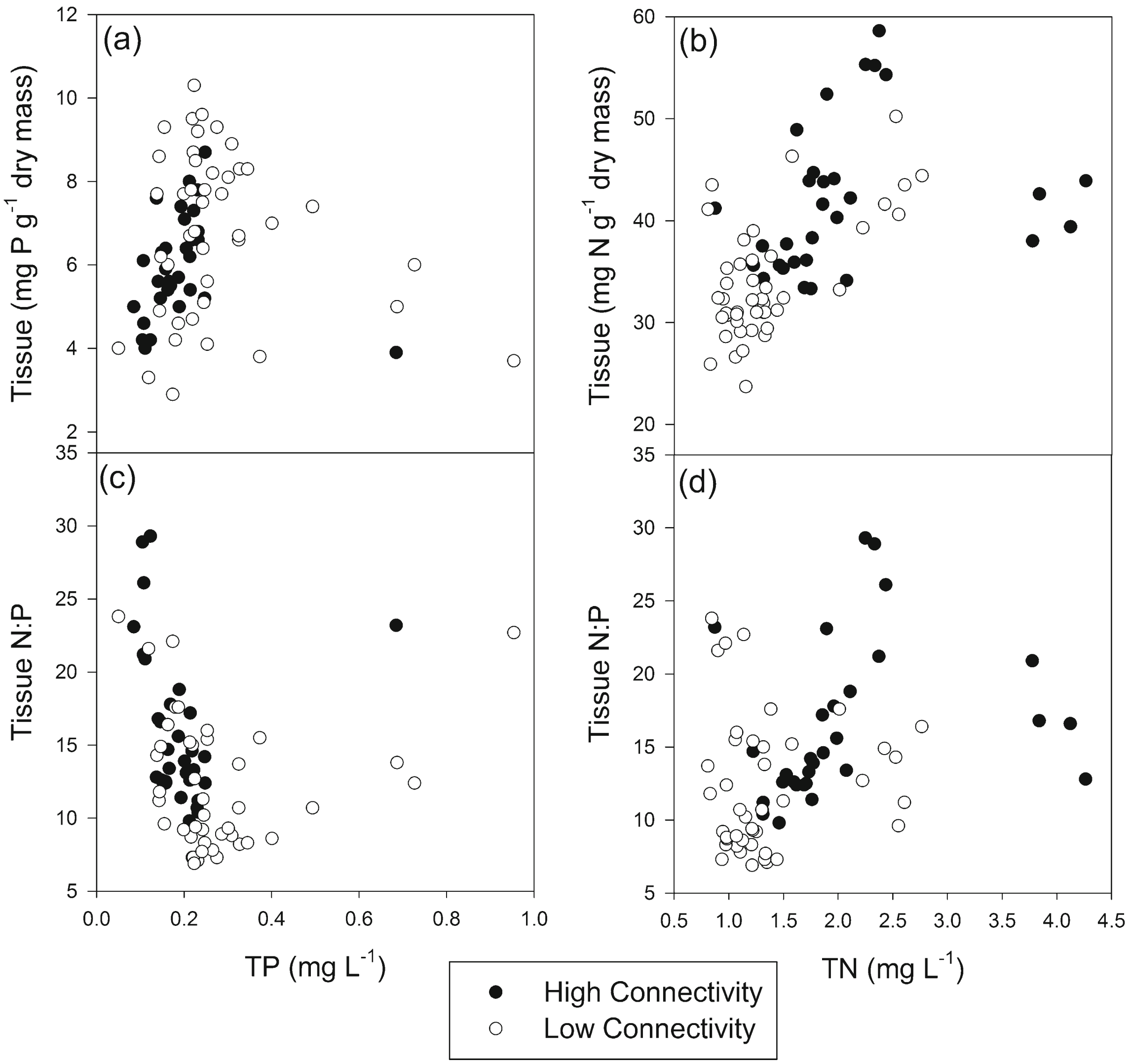

Fig. 5 **a** Relation between free-floating plant (FFP) tissue phosphorus concentration and water column total phosphorus concentration. **b** Relation between FFP tissue nitrogen concentration and water column total nitrogen concentration. Relation between FFP tissue N:P and water column **c** total phosphorus and **d** total nitrogen. All data from the connectivity sampling component

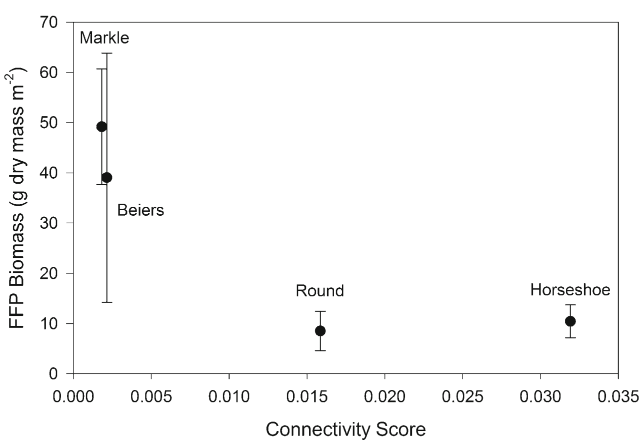

Fig. 6 Mean free-floating plant (FFP) biomass by backwater from the connectivity sampling component. A lower connectivity score indicates less connection to the main channel. Error bars represent +/− one standard error

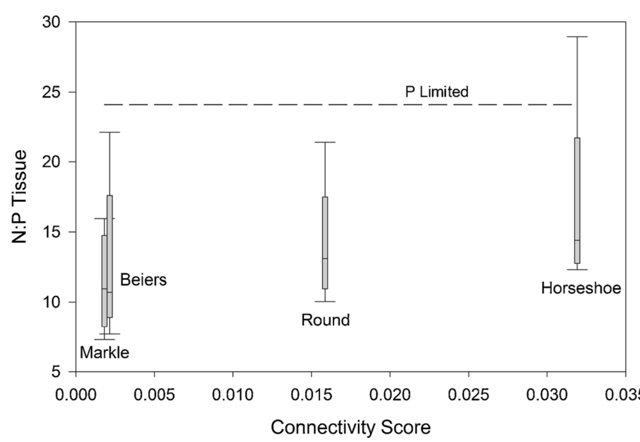

Fig. 7 Box plot of free-floating plant (FFP) tissue N:P by backwater connectivity. A lower connectivity score indicates less connection to the main channel. Values above the dashed line are suggestive of P limitation (Demars and Edwards 2007). *Box plots* represent the 10th, 25th, 50th, 75th and 90th percentiles

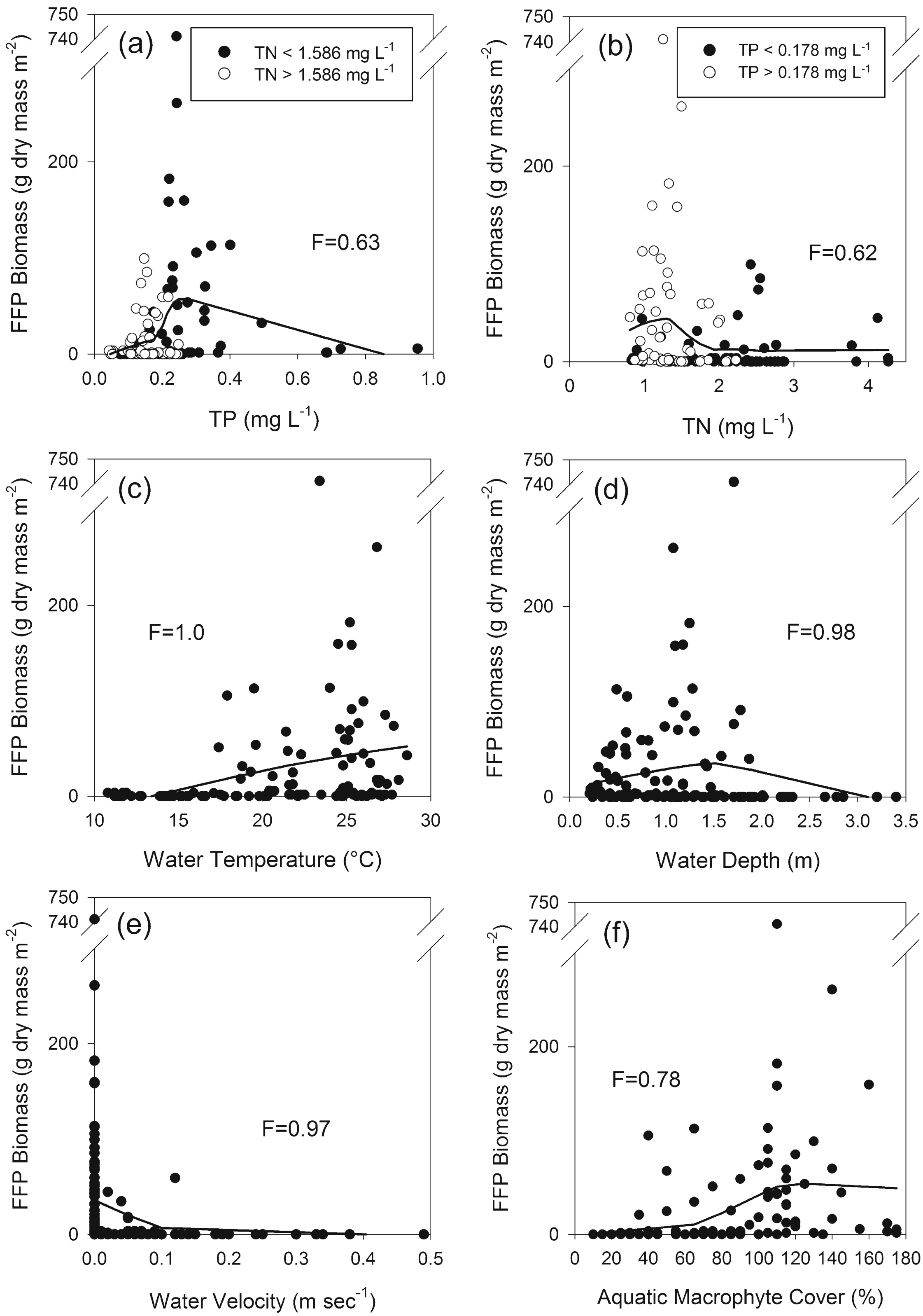

Fig. 8 **a** Relation between free-floating plant (FFP) biomass and total phosphorus concentration from the connectivity sampling component. *Closed circles* represent sites at which total nitrogen was below the median. *Open circles* represent sites at which total nitrogen was above the median. The *solid line* indicates the LOESS regression trend with individual smoothing parameter (F) indicated for each covariate. **b** Relation between FFP biomass and total nitrogen concentration. *Closed circles* represent sites at which total phosphorus was below the median. *Open circles* represent sites at which total phosphorus was above the median. Relation between FFP biomass and **c** water temperature, **d** water depth, **e** water velocity, and **f** aquatic macrophyte cover

cover to 65 % cover there was a gradual increase in FFP. From about 65 % to 110 % cover, FFP biomass increased more rapidly as macrophyte cover increased. Above 110 % macrophyte cover, FFP biomass was highly variable among sites.

The thresholds derived from the connectivity component were tested using the data from the stratified random sampling component. The main channel and side channel sampling strata exhibited lower FFP biomass than the impounded, contiguous backwater and isolated backwater sampling strata (Fig. 9). The main channel and side channel strata exhibited mean velocities above the upper threshold of 0.095 m sec^{-1}, and exhibited aquatic macrophyte cover below the lower threshold of 65 % cover indicating that velocity and macrophyte cover conditions were unfavorable for high FFP biomass. Conversely, the impounded, contiguous backwater and isolated backwater strata exhibited water velocity below the upper threshold and macrophyte cover above the lower threshold indicating favorable water velocity and macrophyte cover conditions for FFP in these strata. Mean nutrient concentrations were greater than estimated minimum threshold in all strata.

Discussion

The spatial and temporal variability typical of large floodplain river ecosystems was reflected in the temporal and spatial patterns of water column and FFP tissue nutrient concentrations observed in this study. There was evidence of P limitation during June when FFP tissue N:P ratios greater than N:P>24:1, the threshold suggested for P limitation in aquatic angiosperms by Demars and Edwards (2007), were occasionally observed. During July, nearly all of the tissue samples suggested surplus N and P relative to criteria determined for aquatic angiosperms (Demars and Edwards 2007 and references therein) and nutrient concentrations were generally high. Elevated water column N concentrations were likely due to relatively high discharge conditions. Elevated water column P concentrations likely resulted from anoxic sediment P release which has been observed to be substantial in the Upper Mississippi River during warm mid-summer conditions (James et al. 1995; Houser et al. 2013). Tissue N:P declined and tissue C:N increased through August and September suggesting that N limitation might be more common in late summer and fall compared to spring. All of the observations suggesting P limitation occurred in high connectivity backwaters, whereas the lowest N:P ratios occurred in the low connectivity backwaters largely due to the low water column N concentration that occurred there.

We detected thresholds in the relations between FFP biomass and TP, TN, depth, water velocity, and aquatic macrophyte cover. Our finding of increasing FFP biomass up to 0.25 mg l^{-1} TP is consistent with the observation of De Groot et al. (1987) who suggested that phosphate is often limiting when water column concentrations are below 0.2 mg l^{-1}. Our finding of decreasing biomass with increasing P concentration above 0.28 mg l^{-1} was an unexpected result. A plausible explanation is that the majority of the observations on the right descending limb of the plot (TP>0.28 mg l^{-1}) occurred at sites with relatively low TN concentrations suggesting N limitation might explain the lower FFP biomass at these sites. These high TP and low TN concentrations often occurred in low connectivity backwaters later in the growing season. Such conditions can result from persistent denitrification, high biotic N uptake, (Richardson et al. 2004; James et al. 2008) and high rates of P release from the sediments (James et al. 1995).

Table 1 Relation between free-floating plant (FFP) biomass and environmental variables (n=124 for all tests) during the connectivity sampling component. Range indicates the range of values which can be used to apply the corresponding regression (equation) for estimating FFP biomass. Threshold represents the value of regression breakpoints. Threshold values with lower and upper designations indicate the lower and upper boundaries of the zone of rapidly increasing FFP biomass. The x-axis intercept value indicates where the regression line intersects the x-axis

Dependent variable	Independent variable	Range	Equation	Threshold	X-axis intercept
FFP biomass	TP (mg L^{-1})	0.043–0.167	$y=-4.959+(116.459\times TP)$	0.167- lower	0.043
		0.168–0.25	$y=-90.233+(606.909\times TP)$	0.25- upper	
	TN (mg L^{-1})	0.808–1.308	$y=(-40.656\times TN^2)+(108.84\times TN)-29.1$	0.808- lower 1.308- upper	<0.808
	Temperature (°C)	10.8–28.6	$y=-47.162+(3.590\times Temperature)$	None	13.14
	Water depth (m)	0.21–1.52	$y=11.202+(17.059\times Water Depth)$	1.52	3.12
	Water velocity (m s^{-1})	<0.1	$y=35.751-(302.420\times Water Velocity)$	0.095	0.405
		>0.1	$y=9.031-(22.350\times Water Velocity)$		
	Aquatic macrophyte cover (%)[a]	10–65	$y=-0.852+(0.175\times Macrophyte Cover)$	65- lower	4.85
		70–110	$y=-0.852+(0.175\times Macrophyte Cover)$	130- upper	
		115–125	$y=27.092+(0.216\times Macrophyte Cover)$		
		130–175	$y=66.608-(0.0999\times Macrophyte Cover)$		

[a] Sum of percent submersed, rooted floating and emergent macrophyte cover at site

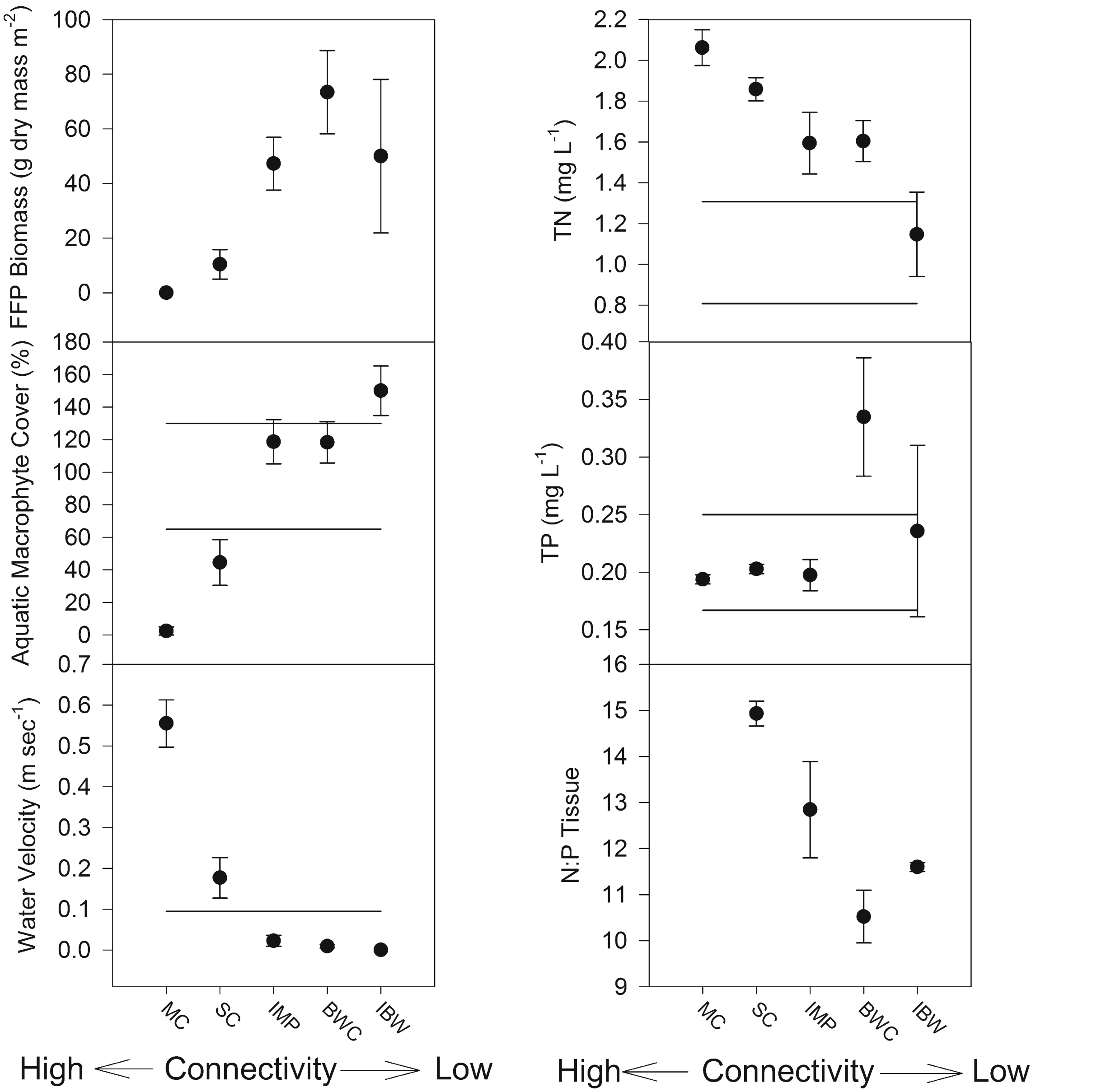

Fig. 9 Mean free-floating plant (FFP) biomass, aquatic macrophyte cover, water velocity, water TN concentration, water TP concentration, and FFP tissue N:P by habitat type from the SRS sampling component. The *solid lines* represent thresholds developed using data from the connectivity design. Error bars represent +/− one standard error

The x-intercept for the relation between water column nutrient concentration and FFP biomass can be interpreted as the minimum required concentration for FFP biomass sustainability (Landolt and Kandeler 1987). The lower x-intercept for water column TP was 0.043 mg l^{-1} similar to the 0.03 mg l^{-1} threshold for minimal FFP growth suggested by Roijackers et al. (2004 and references therein).

In other studies, waters with high N concentration tend to exhibit high FFP biomass and growth rates (Landolt and Kandeler 1987; Szabo et al. 2010). We found biomass increased with water column TN up to 1.308 mg TN l^{-1} and declined above this concentration. Examination of the data on the descending limb of the fitted line (TN>1.308 mg l^{-1}) revealed that the majority of these data points occurred at sites where the TP concentration was relatively low suggesting these sites might have been P limited. The exact x-axis intercept could not be estimated due to limitations in the data distribution. However, visual inspection of the trendline does suggest the value is substantially below 0.808 mg l^{-1}, which appears to be consistent with the concentration required for minimal FFP growth (0.22 mg l^{-1} TN) suggested by Szabo et al. (2005 and references therein).

Physical conditions such as water velocity, water depth and wind and wave action are important in determining FFP distribution (Scheffer et al. 2003), and we detected significant thresholds for several physical variables. We observed a water velocity threshold of 0.095 m sec^{-1}, above which FFP were rarely found. This is similar to the findings of Duffield and Edwards (1981) who reported that water velocity less than 0.1 m sec^{-1} is required for the accumulation of FFP biomass. Wind and wave action can also affect FFP abundance (Boedeltje et al. 2001; Scheffer et al. 2003). Heavy wind and wave action due to the large fetch in the impounded area of our study reach might partially explain why FFP biomass was less abundant in that area relative to that of protected backwaters.

Water depth and its effect on macrophyte cover can influence FFP plant abundance. Maximum FFP biomass was observed at sites with a depth of approximately 1.5 m in our study. Duckweed is known to grow more vigorously in shallow water due to elevated P concentrations near the sediment-water interface (Landolt and Kandeler 1987). The range of aquatic macrophyte cover with the greatest rate of FFP biomass increase (65–110 % cover; Fig. 8f) coincided with depths of 1.5–2.25 m, which approximates the zone of SAV growth in the Upper Mississippi River (Moore et al. 2012). The lower FFP biomass values at depths >2 m are associated with lower SAV abundance and may reflect a lack of physical structure provided by SAV, which could be important during periods of high wind. Lower FFP biomass at sites <1.5 m may have been the result of emergent macrophyte shading (Peck and Smart 1986).

These results indicate that the strong differences in the physical environment (current velocity and macrophyte abundance) between channel and off-channel areas likely explains the high contrast in FFP abundance between these two types of hydrologic regimes, whereas differences in nutrient availability may explain the smaller differences in FFP among off-channel areas. The main channel and side channel sampling strata exhibited FFP biomass far below that of the impounded, contiguous backwater and isolated backwater sampling strata. The major differences between the channels and other aquatic areas were water velocity and aquatic macrophyte cover. The main channel and side channel strata exhibited unfavorable water velocities and aquatic macrophyte cover (too high and too low, respectively). The impounded, contiguous backwater and isolated backwater strata exhibited much lower water velocities, and had higher aquatic macrophyte coverage. Among the three off-channel strata, contiguous backwaters exhibited favorable conditions for FFP by nearly all examined criteria and exhibited the highest mean FFP biomass. Specifically, the contiguous backwaters exhibited relatively high TP and TN concentrations, macrophyte cover in the ideal range, and water velocity near zero. The impounded and isolated backwater strata exhibited marginally lower FFP biomass perhaps due to lower N or P concentrations.

There were clear temporal and spatial patterns in the relative availability of N and P corresponding to spatial and temporal variability in potential nutrient limitation. Early in the growing season and in more connected backwaters, N availability was high relative to that of P whereas the opposite was true later in the growing season and in less connected backwaters. Because these differences in water column nutrient availability were reflected in FFP tissue nutrient content, the influence of connectivity on FFP nutrient content is evident. Thus, these results contribute to the growing body of work illustrating that temporal and spatial variability in connectivity to free-flowing channels is an important component of ecosystem processes within the backwaters of floodplain river systems (e.g., Amoros and Bornette 2002).

The thresholds detected here suggest that under certain conditions, small changes in drivers such as water velocity, aquatic macrophytes, and nutrient concentrations can produce relatively large changes in FFP biomass. Increasing depth >1.52 m, increasing water velocity above zero, and reducing aquatic macrophyte cover below 65 % would all likely result in a reduction in FFP biomass. In addition to improving our understanding of the basic ecology of FFP, our results can provide useful insights into how the future management of off-channel areas may reduce the occurrence of mats formed by over-abundant FFP. For example, management actions on the Upper Mississippi River are often designed to manipulate water velocity (e.g., by modifying connections between channel and off-channel areas (e.g., Johnson et al. 1998)) or to facilitate the establishment of aquatic macrophytes (e.g., constructing islands to reduce wind fetch and create shallow, sheltered areas (e.g., Gray et al. 2010)) Furthermore, the estimated TP threshold supports the numeric P criterion of <0.1 mg l^{-1} TP for Wisconsin non-wadeable rivers (Wisconsin Administrative Code NR 102.06(3)), as achieving this value would likely result in a reduction of FFP biomass. Overabundant FFP in aquatic systems can be an important symptom of eutrophication (Croft and Chow-Fraswer 2007), and a better understanding of its response to various key drivers should help to inform future management actions.

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

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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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Aquatic macrophytes; fisheries guilds; trophic shifts; Mississippi River; total suspended solids; ecological shift; alternative stable state

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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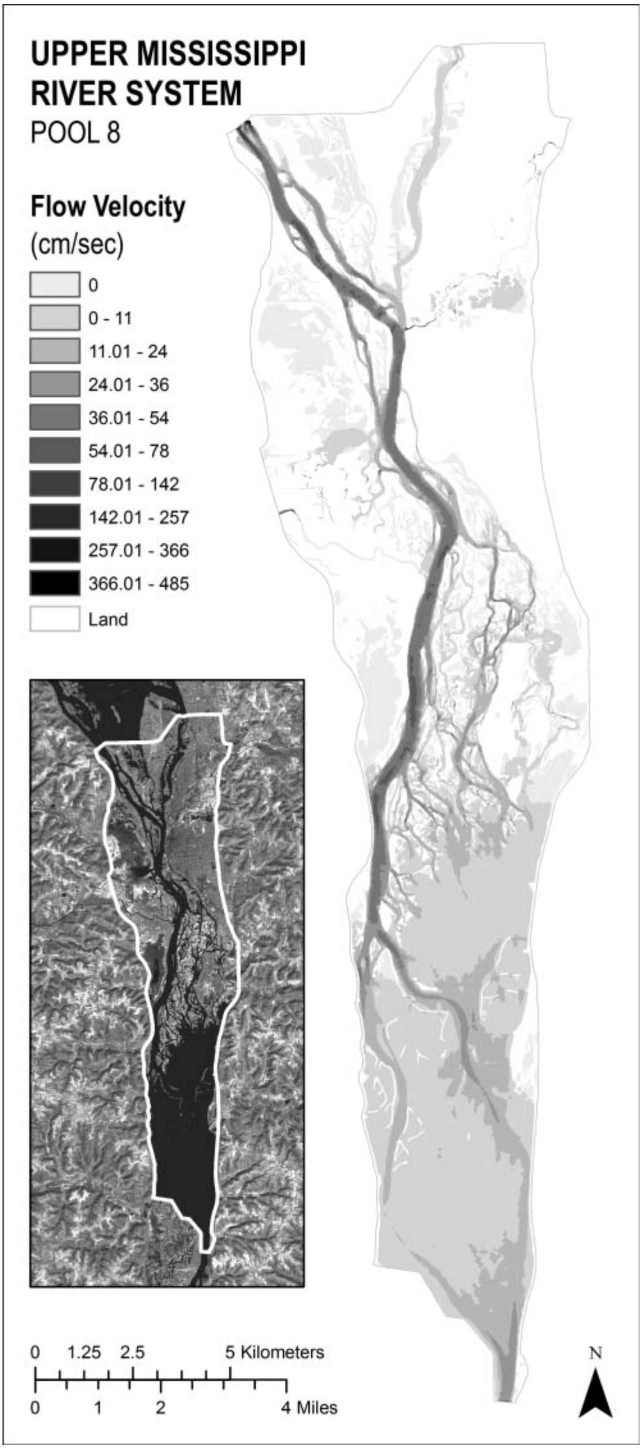

Figure 1. Navigation Pool 8 of the Upper Mississippi River extending from 1093.1 to 1130.6 km. The extent of water coverage and velocities are shown at a mean discharge of 1133 $m^3\ s^{-1}$, and the inset is a shaded representation at the same discharge. The main navigation channel is maintained at 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.

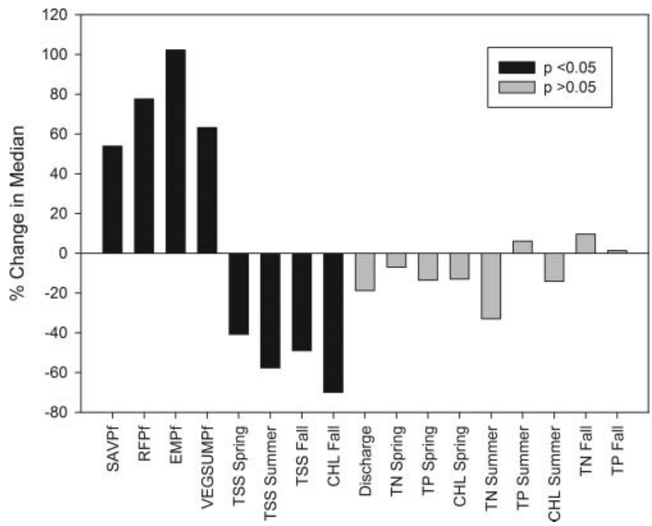

Figure 2. Change in median between the early environmental period (1993–2001) and the late environmental period (2002–2011) among the environmental variables in Pool 8 of the Upper Mississippi River. Changes significant at the $p < 0.05$ level are denoted with black bars.

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

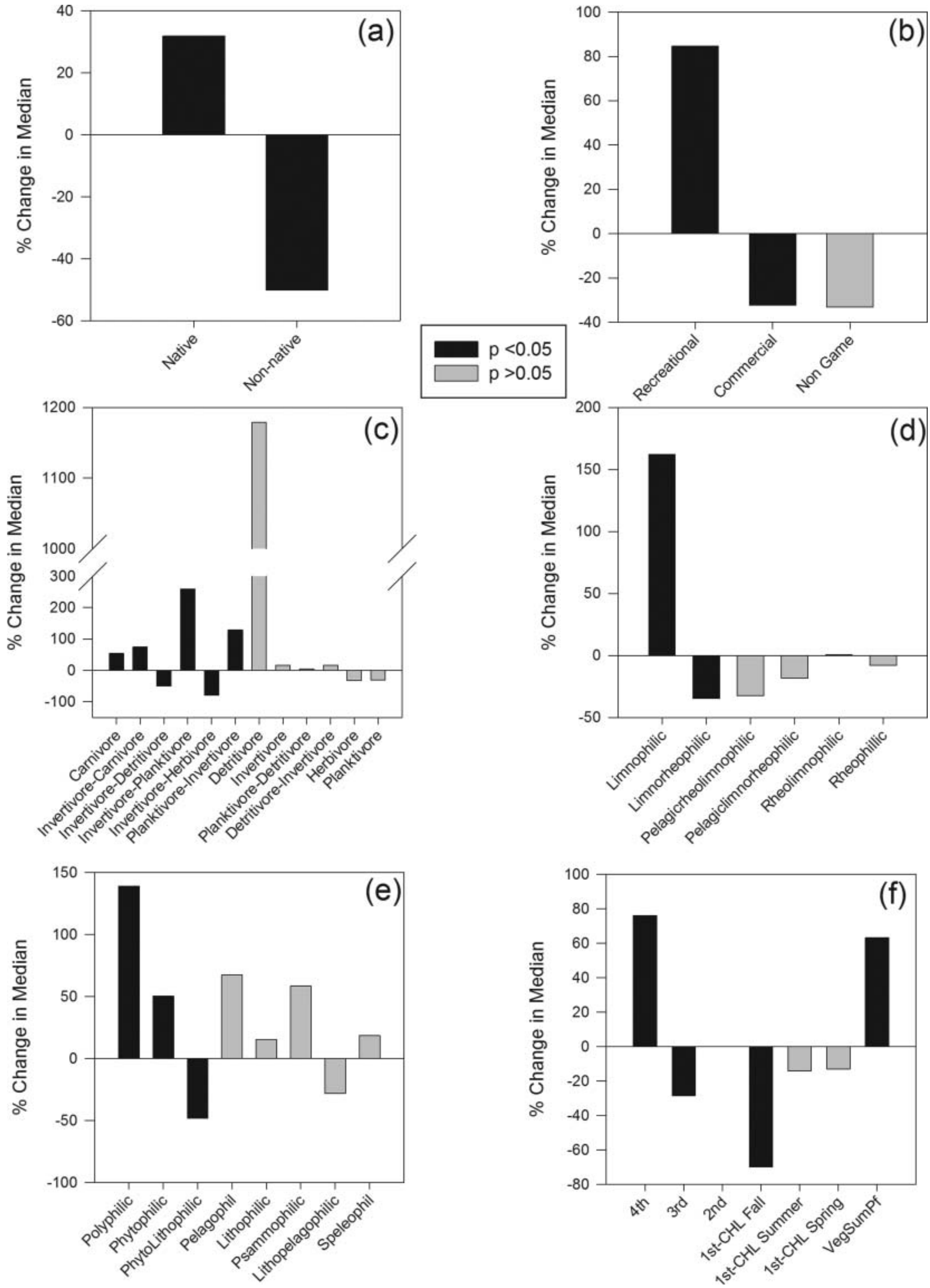

Figure 3. Change in median between the early period (1993–2001) and the late period (2002–2011) among (a) native/non-native status; (b) exploitation status; (c) feeding guild; (d) habitat guild; (e) reproductive guild; and (f) trophic position in Pool 8 of the Upper Mississippi River (1993–2011). Changes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

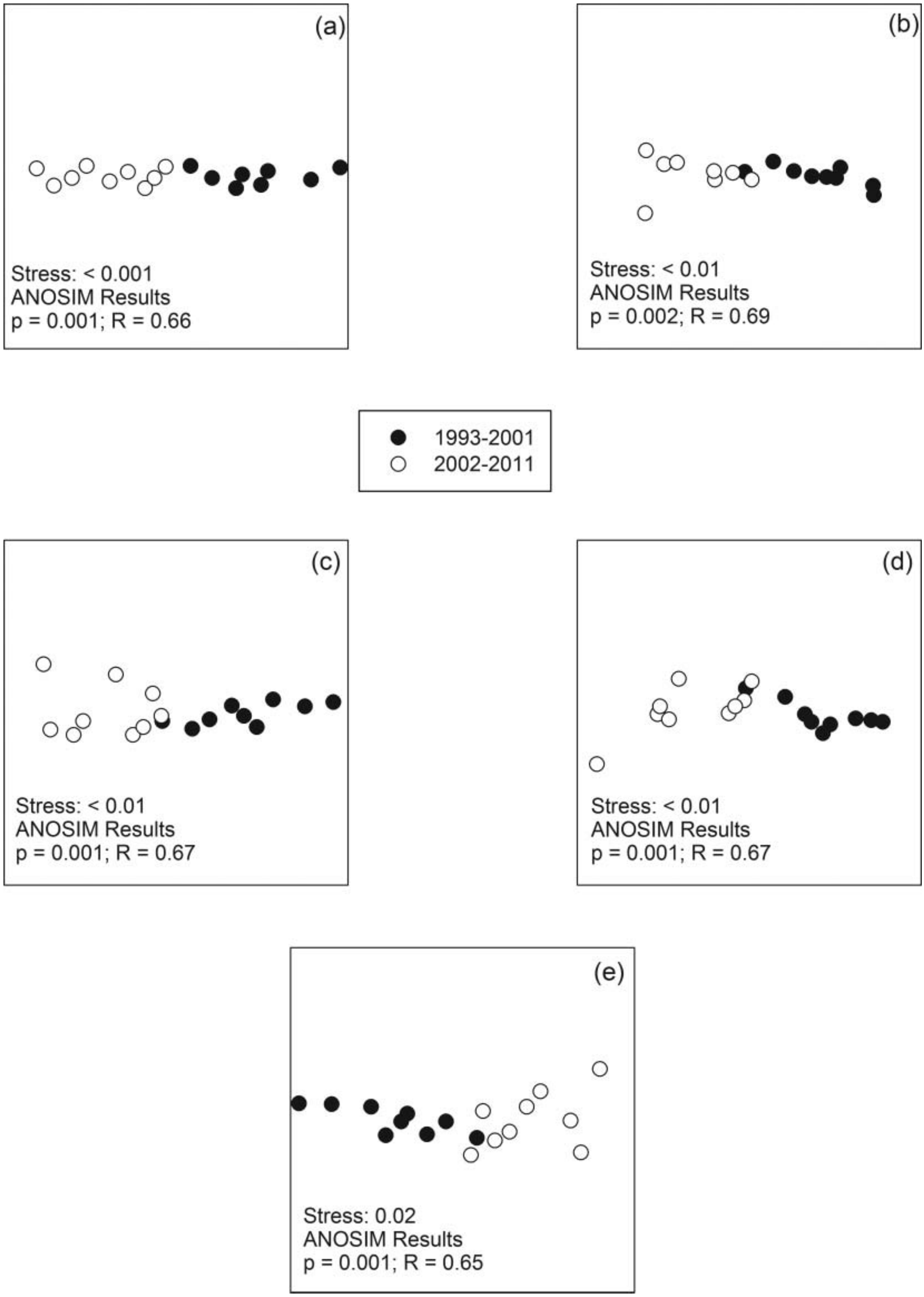

Figure 4. Two-dimensional non-metric scaling ordination (NMDS) between the early period (1993–2001) and the late period (2002-2011) among (a) native/non-native status; (b) exploitation status; (c) feeding guild; (d) habitat guild; and (e) reproductive guild in Pool 8 of the Upper Mississippi River (1993–2011).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

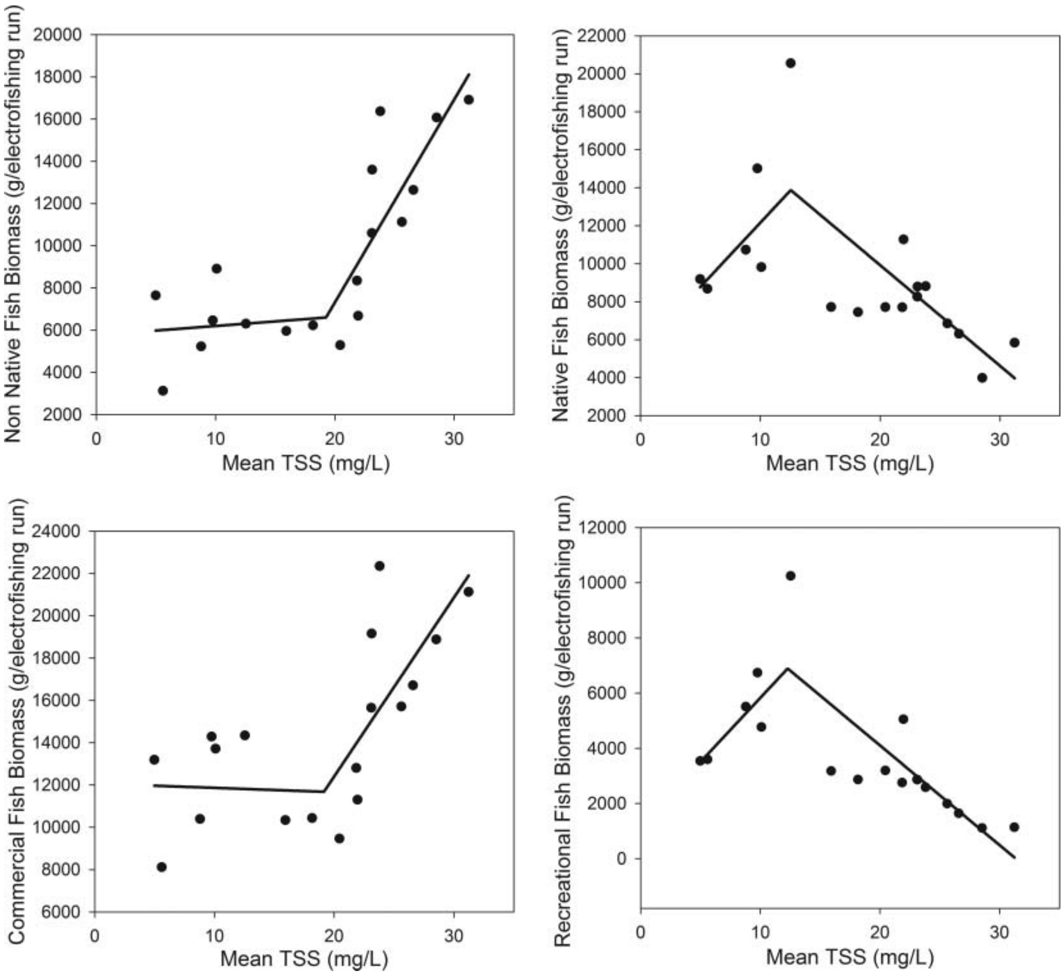

Figure 5. Relation between mean annual fish guild biomass per electrofishing run and mean summer TSS in Pool 8 of the Upper Mississippi River (1993–2011). Thresholds are indicated by the breakpoint in the piecewise regression line.

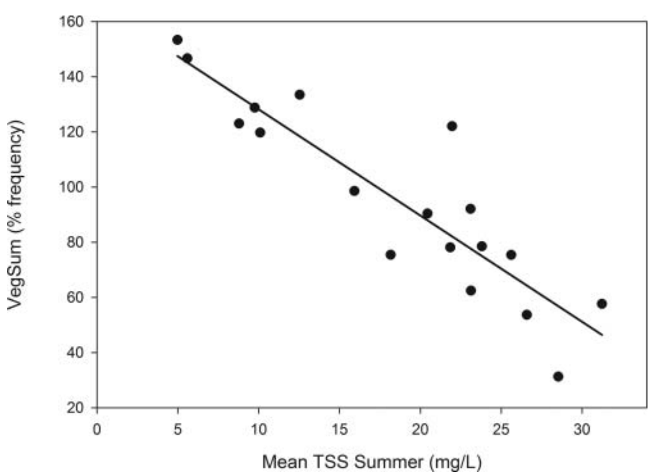

Figure 6. Relation between VegSum (percent frequency) and mean summer TSS in Pool 8 of the Upper Mississippi River (1993–2011). The line indicates the linear regression result ($y = -3.85x + 166.64$; $r^2 = 0.807$).

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.

Beranek, Ashley E - DNR

From: Minahan, Kristi L - DNR
Sent: Wednesday, January 23, 2019 2:58 PM
To: Beranek, Ashley E - DNR; Diebel, Matthew W - DNR
Subject: comments on WisCALM-TP methods

Hi Ashley (& Matt)—a couple comments on the TP calculation section of WisCALM:

- On p. 55, it states: This approach involves the calculation of a 90% confidence limit around the median of a TP sample data… (lakes section also uses 90% CI.) In the draft code language, we state that they need to calculate an 80% CI around the median. (for lakes, 80% CI around the mean). Let's make sure our language lines up unless there was a reason not to do that yet?
- We state that we should only use one sample per month (closest to middle of month). But for more in-depth studies, they often take a weekly or every-other-week sample, and shouldn't all those be used when they are available, as long as it was a set of dates selected ahead of time to be periodic rather than data targeting certain flows? We ran into a question like this from a permittee in the Fox-Illinois last week. Perhaps we should add a sentence to that effect? This comment ties in to my current dive into how the permit P calcs differ from WisCALM; they use all available data (all samples over a 28 day pd are averaged), not a single sample.

K.

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From: Mark <c185.mark@gmail.com>
Sent: Wednesday, January 23, 2019 10:07 AM
To: DNR Impaired Waters
Subject: Comments

Follow Up Flag: Follow up
Flag Status: Flagged

I appreciate the work being done. I wish for more funding for DNR and assistance to farmers to abate run off.
Buffers need to established along water ways either by public funding, public land easement and stricter laws with or without funding to reduce run off.
There is way too much algae growth in our lakes and streams in most of state water.

Sent from my iPad