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Floodplain lake assessment and fish assemblage dynamics in the Mississippi Alluvial Valley

By

Caroline Noelle Steiner Andrews

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Floodplain lake assessment and fish assemblage dynamics in the Mississippi Alluvial Valley

By

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Floodplain lakes in the Mississippi Alluvial Valley (MAV) provide valuable freshwater resources for states in which they occur. Thirty lakes in portions of Mississippi and Arkansas were surveyed for chlorophyll-*a* fluorescence and turbidity using handheld meters to determine relationships between chlorophyll-*a* concentrations and suspended solids. High applicability of handheld meters in the MAV presents economic benefits for monitoring the numerous lakes in the region. Additionally, twelve lakes within Bear Creek watershed, Mississippi were studied to determine how hydrologic connectivity shapes fish communities. Isolated and permanently connected floodplain lakes exhibited characteristically lacustrine and rheophilic fish communities, respectively, diversifying fishery management opportunities. Lastly, spring diel temperature and oxygen dynamics, as well as juvenile fish communities, were assessed within three habitats in a floodplain lake – pelagic environment, margin and contiguous wetlands. Variability in temperature and oxygen across the three habitats promotes spring habitat heterogeneity while supporting distinct but overlapping juvenile fish assemblages.

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CHAPTER I

INTRODUCTION

An increase in studies examining terrestrial-aquatic interactions on various spatial scales can be attributed to development of two major concepts: river continuum (Vannote et al. 1980) and flood pulse (Junk et al. 1989). These concepts have brought relevance of interactions between aquatic systems and surrounding landscapes to the forefront of ecological study. Advancement in understanding of aquatic systems has been a direct result of applications of these concepts and consideration of landscape perspectives to interactions among aquatic and terrestrial systems (Johnson and Host 2010; Lamberti et al. 2010). However, such interactions have not received equal attention relative to type of aquatic ecosystem. Whereas studies of lotic systems have generally benefitted from an expanded perspective, studies of lentic systems have remained primarily focused on internal processing despite the fact that considerable exchanges between systems exist that can influence lentic processes (Lamberti et al. 2010).

The Mississippi Alluvial Valley (MAV) is a particularly dynamic system in which to study interactions of lentic systems and the landscape. Historically, the MAV has been an active floodplain, with meander courses from the main channel of the Mississippi River as well as several current and ancient tributary streams and river systems (Fisk 1944). Meandering has left behind hundreds of floodplain lakes in the region, varying in size, shape, formation, and age: from small pools to entire reaches of abandoned channels (Fisk 1944; Baker et al. 1991). Introduction of anthropogenic manipulation has also introduced lakes into the floodplain, namely meander cutoffs and borrow pits associated with channelization and levee formation.

Levee formation in the MAV was a result of needs to protect and increase use of land for agricultural and urban purposes (Biedenharn et al. 2000). Agriculture dominates the region due to rich, alluvial soils, and abundant freshwater resources. As such, agriculture presently plays a key role in aquatic-terrestrial interactions in the MAV. Freshwater supplies are becoming increasingly limited, and irrigation needs are depleting groundwater resources at an alarming rate, resulting in the loss of base flow in some streams in the region (Pennington 2006). Sedimentation drastically increased after agriculture was introduced to the region, thus reducing the lifespan of floodplain water bodies (Wren et al. 2008). Widespread use of fertilizers and pesticides introduces high nutrient concentrations to floodplain waters enhancing eutrophication (Turner and Rabalais 2003), as well as bio-accumulating contaminants that persist in sediments and biota (Moore et al. 2007).

Traditionally, floodplain lakes support a diverse assemblage of biota, including fish communities comprised of up to 70 species (Baker et al. 1991). As such, recreational and artisanal fisheries in the region have provided a valuable economic resource (Brown and Toth Jr 2001). However, biotic integrity in most floodplain lakes of the region has been found to be impaired (Aycock 2008). Fish communities are limited by primary and secondary environmental factors, including variables such as depth, land use, connectivity, and water quality that are increasingly negatively influenced by anthropogenic uses (Alfermann 2011; Dembkowski 2011).

With freshwater resources in the MAV increasingly threatened for water quantity and quality, effective management options are needed to ensure continued availability for future generations. This thesis extends concepts of river continuum and flood pulse and their implications on a regional scale to provide insight to biological structuring and feasible management options for managers of freshwater resources. Chapter II addresses needs for efficient lake monitoring and assessment with handheld meters for chlorophyll-*a* and turbidity across many lake types, eutrophication status, and surrounding land uses. Chapter III addresses hydrologic connectivity and its implications on structuring fish communities within a watershed. Chapter IV represents an assessment of the potential spatial heterogeneity provided to juvenile fish communities by wetlands adjacent to floodplain lakes.

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CHAPTER II

USE OF HANDHELD METERS IN WATER QUALITY MONITORING OF FLOODPLAIN LAKES

Introduction

Thorough water quality surveys and monitoring are vital tools for water managers to use for protecting water resources (Ballance and Bartram 1996). However, with an overwhelming number of factors contributing to aquatic ecosystem functioning, simplified accurate measures are needed to optimize this process. Instantaneous estimates of water quality variables, with field meters, provide time and cost advantages relative to measurements made by analytical laboratory methods. In agricultural areas, where runoff often transports extraordinary amounts of sediments and nutrients, two metrics commonly monitored with field meters are suspended solids and chlorophyll-*a*.

Suspended solids have great influence on aquatic ecosystems, ranging from biotic functioning to lifespan of the water body, and are essential for multiple aspects of lake monitoring (Bilotta and Brazier 2008). Due to the strong association between nutrients (primarily phosphorus and nitrogen) and primary production in freshwater food webs, phytoplankton concentration is an ideal candidate for monitoring trophic state (Reynolds 1984). Chlorophyll-*a* concentration, a measure of pigments found in all phytoplankton, and a surrogate for phytoplankton biomass, is a commonly used metric when determining primary production and trophic status (Goodwin 1965). Water quality variables, including suspended solids and chlorophyll-*a*, can typically be measured *in-situ* using field meters, or processed and measured in an analytical laboratory. However, measurements provided by field meters may be subject to more error than analytical methods and may not serve as absolute substitutions.

Most portable turbidimeters measure amount of light scattered or attenuated by suspended particles (i.e., the principle of nephelometry). Differences between meters and even small errors from calibrations with formazin (especially at relative turbidity units < 1) can reduce reliability and comparability of nephelometric turbidity units (NTU) as a standard measurement (Letterman 2002). Nephelometric turbidity measurements can vary greatly due to effects of particle composition, size, and distribution on light scattering (Gippel 1989). Due to optical principles, low-range turbidity measurements can contain a large proportion of dissolved components that can distort measurement (Duchrow and Everhart 1971). Typically, turbidity has high correlation with measurements of total suspended solids (TSS; $r = 0.99$) in a laboratory setting (Holliday et al. 2003). In field studies, turbidity and TSS correlations remain high ($r = 0.91 – 0.98$); however, TSS predictions from TSS-turbidity regression models bring error contributed by other factors that affect light transmission (Grayson et al. 1996; Suk et al. 1998; Packman et al. 1999). Relative importance of three components of turbidity was assessed before developing a calibration model with TSS (Figure 2.1).

Estimation of chlorophyll-*a* by *in-vivo* use of portable fluorescence meters is often used because of a substantial cost and time benefit (Moulton et al. 2009). High sensitivity of chlorophyll-*a* fluorescence, light reemitted by a discrete sample after absorption of a specific wavelength of light, allows portable meters to function with just a few milliliters of sample directly taken from the water body of interest. For *in-vivo* chlorophyll-*a* fluorescence, meters can be calibrated with a solid standard, resulting in unit-less relative value measurements for each individual sample. To further quantify these values, regression of *in-vivo* fluorescence on volumetric measurements made with one of three standard methods is commonly derived. Often *in-vivo* fluorescence and chlorophyll-*a* concentration are well correlated (Heaney 1978; Vyhnalek et al. 1993; Lesko and Boekestein 2008; Seppälä and Olli 2008; Moulton et al. 2009) but may be associated with wide confidence and prediction intervals, limiting precise prediction capabilities. This high variability can be due to several factors. Uncontrollably, species composition and relative population health affect fluorescence intensity and chlorophyll-*a* concentration, especially when sampling multiple water bodies or integrating various depths (Heaney 1978; Reynolds 1984). Optical discrepancies such as clay particles intercepting light transmission, or additional presence of dissolved ions or phytoplankton less than 0.45µm not accounted for in laboratory analysis could also potentially alter the relationship between relative fluorescence and concentration of chlorophyll-*a*.

Given limitations in use of handheld meters, an understanding of meter mechanics and applicability, followed by validation before extensive use, is essential. When measuring chlorophyll-*a* fluorescence or nephelometric turbidity, handheld meters rely on optical principles measured on gross water samples often just collected at the surface, resulting in several restrictions. First, optical discrepancies and small sample volumes may limit measurement precision through biased sample grabs. Second, units read from field meters represent relative fluorescence or light transmission units rather than volumetric concentrations, and as such are not consistent between instruments (Gippel 1989) and not directly comparable to laboratory measurements. Finally, optical composition can often reflect multiple water column components, hindering meter measurements from being a direct surrogate for the variable of interest (Bilotta and

Brazier 2008). Specifically, there is a need for improved understanding of relationships between measurements obtained with field meters and volumetric estimations made with standard laboratory methods through evaluation of (a) sampling protocol, (b) sample composition, and (c) environmental covariates. To this end, objectives of this study were to provide the best standardized field measurements of turbidity and chlorophyll-*a* fluorescence and describe their relationship to total suspended solids and chlorophyll-*a* concentration.

Methods

Study sites

Oxbow lakes are former channels of floodplain rivers that have separated naturally or been diverted artificially. Hundreds of oxbow lakes, fostering a wide range of water quality conditions exist in the Lower Mississippi River alluvial valley (MAV) along floodplains of several contemporary and historic rivers (Baker et al. 1991). These varying conditions correspond to different lake morphometries, connectivity with adjacent water bodies, and surrounding land use and practices. While this region is primarily agricultural due to its rich, fertile soils, some protected lands do include oxbow lakes; most notably the White River National Wildlife Refuge (WRNWR) in Arkansas, has more than 300 oxbows on site. Thirty oxbow lakes were selected for this study from the Yazoo River basin, around Greenwood, Mississippi, and from the White River basin, within the WRNWR in eastern Arkansas (Figure 2.2).

Sample collection

Prior to sampling, one handheld turbidimeter (HACH 2100p; HACH, Loveland, CO) and one handheld fluorometer (Turner Aquaflor; Turner Designs, Sunnyvale, CA) were calibrated using solid standards. Relative fluorescence units (RFUs) were calibrated around a value of 400 for the solid standard.

Surface water samples were collected at 30 floodplain lakes between 1100 and 1330 hours in attempt to sample under similar light conditions. Sampling was conducted in late-June through July, representing the dry and warm period of high phytoplankton production, while avoiding potential lake dry-out. Ten 1-L surface (< 0.5 m) grab samples were collected from haphazardly-selected locations near the center of each lake and combined to form one composite sample. Care was taken to avoid sampling from water disturbed by boat passage.

At shore, alkalinity was estimated using LaMotte test kit number 9844-01 (LaMotte, Chestertown, MD). Approximately 150 mL of composite sample was filtered through a 0.45-μm glass-fiber filter paper for chlorophyll-*a* analysis at a later date. Filter paper was folded, placed in aluminum foil within a sealed plastic bag and preserved on ice for transport to laboratory facilities at Mississippi State University, where it was frozen at -20°C. The filtrate (filtered composite) was used for measurement of turbidity and chlorophyll-*a* alongside unfiltered composite samples. Additionally, 1 L of unfiltered sample was preserved on ice and transported to the laboratory, kept refrigerated (4°C), and processed within 7 days to measure suspended solids.

Three subsamples were taken from unfiltered and filtered samples and each subsample read 3 times on the fluorometer, for a total of 9 measurements for unfiltered and filtered samples. On the turbidimeter, signal-averaging mode was used, combining 10 individual readings into a single, averaged reading, and each subsample was read singly, for a total of 3 measurements of unfiltered and filtered samples. Use of features like signal-averaging mode suggests that optical measurements may lack in precision, and averaging may compensate for these inaccuracies.

At the laboratory, measurement of chlorophyll-*a* concentration, total suspended solids (TSS), and volatile suspended solids (VSS) was performed according to standard methods (APHA 1998). Total suspended solids were filtered on pre-ashed, pre-weighed, 0.45-µm glass-fiber filter paper. Filters were dried at 105°C until a constant weight. Filters were then ashed in a 550°C muffle furnace until a constant weight to determine VSS content. Non-volatile suspended solids (NVSS) were determined as the difference between TSS and VSS. Chlorophyll-*a* from macerated filters was extracted in 90% buffered acetone overnight. The solution was agitated and centrifuged, and the supernatant absorbance at 664, 647, and 630 nm read on a HACH DR5000 spectrophotometer.

Statistical analysis

Turbidity

Effect of subsample order, possibly due to settling, for either the unfiltered or filtered sample was first examined. Analysis of covariance (ANCOVA) was used with NTU as the dependent variable, lake as an independent class variable, and subsample order (1-3) as the continuous covariate. Multiple regression was then used to assess relative importance of turbidity components on NTU values. In this analysis, NTU (dependent variable) was regressed on VSS, NVSS, and the filtrate (independent variables) to estimate standardized regression coefficients. Standardized coefficients identify relative strength of variables by placing all variables in the same scale, so that larger coefficients suggest a stronger association with the dependent variable. Whereas

VSS and NVSS represent concentrations (mg/L) of suspended organic and inorganic particulate, respectively, the dissolved components, smaller than the filter used to capture TSS, are also included in turbidity measurement. Simple linear regression was then used to determine the relationship between turbidity and TSS. Multiple linear regression was also used to determine if depth could be added as a covariate to improve the relationship through addition of a settling/resuspension factor (Table 2.1). Normal probability plots of residuals were examined to insure approximate normality for all analyses, assuming ANCOVAs were robust to minor deviations in normality. Additionally, residuals were examined against dependent variables and predicted responses to ensure homogeneity of variance. Variables were transformed ($\log_e$ or square root) where appropriate to maintain assumptions of normality and homogeneous variance.

Chlorophyll-a

Consistent with the turbidity analysis, effect of subsample order for unfiltered and filtered samples through ANCOVA was determined following the procedure described above. As there were multiple readings (1-3) within a subsample, the same method was used to determine effect of reading number. To further investigate fluorometer dynamics, a nested model was used to account for amount of variation in RFUs explained by multiple readings (reading error) given multiple subsamples (sampling error) within each of the thirty lakes. To accurately reflect portions of chlorophyll-*a* retained on the filter for laboratory analysis, RFU of the filtrate sample was removed from the average RFU of the unfiltered sample. Simple linear regression was then used to determine the relationship between fluorescence and chlorophyll-*a*. Stepwise selection was used to see if incorporation of covariates including turbidity or alkalinity would improve the simple regression relationship (Table 2.1). Residual plots were again examined for extreme deviations from assumptions. Variables were transformed ($\log_e$ or square root) where appropriate to maintain assumptions of normality and homogeneous variance.

Results

Turbidity

There was no linear trend associated with subsample order ($P > 0.05$), allowing use of the average of the three subsamples to represent a lake in further analysis. Standardized regression coefficients for the filtrate, VSS, and NVSS were 0.46, 0.38, and 0.36, respectively (Table 2.2), suggesting that all three components contributed about equal proportions of optical turbidity. Moreover, all three components were positively correlated with NTU (Figure 2.3). Considering that the filtrate had an important effect on NTU, it may be advisable to filter water samples when the goal is to index VSS+NVSS turbidity. Therefore, while TSS, a summation of VSS and NVSS, has the most effect on NTU, it is important to remove the turbidity associated with the filtered sample before post calibration to laboratory samples. Optical interference from dissolved components (i.e., filtrate) composed 6-34% of total NTU measured, and seemed to be related to maximum lake depth (Figure 2.4). The corrected NTU model, removing the filtrate from total NTU, produced a model that best described the relationship between TSS and NTU with a curvilinear form (Figure 2.5):

$$\text{TSS (mg / L)} = 3.64 \times \text{Corrected NTU}^{0.647} \tag{2.1}$$

Addition of depth as a covariate did not improve the fit significantly ($P = 0.68$). The r^2 for equation 2.1, derived from the linear model derived from a log-log equation, was 0.76. The present model was compared to other published relationships, having a somewhat lesser fit than other models (Table 2.3).

Chlorophyll-a

Unlike turbidity, ANCOVA results for the filtrate did show a significant subsample order effect ($P < 0.01$) but, although statistically significant, the effect was negligible (slope = -0.03) and manifested in a 0.1-10.3% difference in the average of any two individual readings compared with the average of all three readings. Therefore, all three readings were averaged for further analysis. There was no linear trend expressed by subsamples of the unfiltered fluorescence values ($P = 0.55$) or by reading number for either unfiltered or filtered samples ($P = 0.45$, 0.12, respectively). Nested analysis revealed that lake differences composed most the majority of unfiltered sample variation (99.1%), followed by subsample (0.5%), and meter error (0.4%). Similar results were found for the unfiltered sample (98.4, 1.1, and 0.5% of variation, in the same respective order). Fluorescence of the filtrate composed 3-45% of total fluorescence of the original, unfiltered sample, providing support for the removal of this component before calibration to laboratory chlorophyll-*a* values. The original and corrected fluorescence models describing the relationship between relative fluorescence units (RFU) and µg/L chlorophyll-*a* were:

$$\text{Chlorophyll} - a\ (\mu g / L) = 0.105 \times \text{RFU}^{1.17} \qquad (2.2)$$

$$\text{Chlorophyll} - a\ (\mu g / L) = 0.157 \times \text{Corrected RFU}^{1.12} \qquad (2.3)$$

where corrected RFU increased the r^2 from 0.70 to 0.74. These models were validated by estimating trophic state for all 30 lakes (Table 2.4); however, the corrected equation over-classified one eutrophic lake.

Finally, stepwise selection of covariates including alkalinity, and unfiltered turbidity identified turbidity as a useful covariate for improving predictability of chlorophyll-*a* from corrected RFU. Therefore, the final calibration of field chlorophyll-*a* to laboratory chlorophyll-*a* was ($R^2 = 0.80$):

$$\text{Chlorophyll} - a\ (\mu g / L) = 0.25 \times \text{Corrected RFU}^{0.83} \times \text{Unfiltered NTU}^{0.35} \qquad (2.4)$$

Accounting for differences in measurement units, standardized coefficients showed that the turbidity component is about one third of the entire regression model, suggesting it is warranted as a corrective covariate. All analyses represented approximately normal residuals according to normal probability plots and exhibited no trends with the dependent variable or predicted values.

Discussion

Turbidity is a general measurement of particulates in the water. However, for its use across studies, some quantification of what comprises turbidity is needed. The objective of this study was to index a standardized measurement of total suspended solids from nephelometric turbidity. When quantifying turbidity with suspended solids, dissolved particulates that are included in turbidity measurement are often overlooked and unaccounted for. Results showed that dissolved components can have large effects on optical turbidity measurements, a portion not captured by traditional TSS measurement (Knowlton and Jones 2000). Dissolved components may include humic substances or mineral ions, even phytoplankton smaller than the filter (Bilotta and Brazier 2008), and could be influenced by watershed size and land use, as demonstrated by the transport of dissolved organic carbon (McGlynn and McDonnell 2003).

Depth could be an important covariate to include in lentic systems when describing turbidity, even though it was not significant in the model. Depth has a major influence on re-suspension and settling properties of particulates, as well as temperature and phytoplankton growth. There was only a weak trend in depth and filtered turbidity across study lakes, possibly because all lakes were shallower than 8 m maximum depth.

Using chlorophyll-*a* to index primary productivity is ubiquitous in lentic systems. While spatial techniques are being used to remotely index chlorophyll-*a*, small natural lakes with limited surface area may prove excluded from the abilities of these techniques (Gitelson et al. 1993). Calibration of handheld optical meters must consider discrepancies from between field measurements and laboratory techniques. Similar to turbidity, filtered components can be a large portion of total fluorescence. Adding turbidity as a covariate can improve relationships but must also be carefully considered as a spurious effect (Moulton et al. 2009). In this study, turbidity was found to contribute a significant portion of the calibration model, even though the model fit did not increase very much (r^2=0.74 to 0.80). Models suggest that trophic status can be instantaneously determined with a handheld meter, which is a useful tool for lake classification.

Traditional calibration curves for these meters suggest linear relationships between meter readings and laboratory counterparts. Curvilinear relationships exhibited by NTU-TSS and RFU-chlorophyll-*a* suggests that handheld meters may not be performing linearly in all systems across all ranges. Furthermore, particle shape and refractory characteristics are greatly different by region and system and can change relationships between turbidity and particulate concentration (Table 2.3) and might also be a factor in phytoplankton fluorescence. Calibration of field to laboratory measurements is crucial to determine the relationship for the application desired.

Use of field meters requires established protocol for sample and meter replicates. While the turbidimeter automatically averages multiple meter readings per sample, the fluorometer does not perform this function. However, meter readings did not differ significantly and contributed less than 1% of total variability for the handheld fluorometer, suggesting that one reading would most likely be sufficient. Subsamples did differ significantly, except for RFU of the filtered sample. While this effect was not large, it does reinforce that care needs to be taken, especially with samples in the low range of meters, in order to avoid unnecessary user error. In such case, multiple samples would be recommended to avoid bias with subsampling.

Table 2.1 Sources of meter interference

	Potential interference addressed	**Phytoplankton**	**Turbidity**
Samples			
Unfiltered composite	None	✓	✓
Filtered site water	Non-filterable substances: Picoplankton or unique dissolved ions	✓	✓
Covariates			
Turbidity	Interception of light transmission	✓	
Depth	Settling/resuspension factor - size of particles		✓
Alkalinity	Phytoplankton health	✓	

Components of nephelometric turbidity and chlorophyll-*a* fluorescence measurement and potential sources of interference addressed.

Table 2.2 Linear regression coefficients

Variable	Standardized Estimate	t value	P value
Intercept	0	2.98	< 0.01
Filtered Turbidity (NTU)$^+$	0.46	5.26	< 0.01
Volatile Suspended Solids (VSS)*	0.38	5.35	< 0.01
Non-volatile Suspended Solids (NVSS)*	0.36	4.22	< 0.01

Standardized linear regression coefficients for component analysis of unfiltered turbidity (NTU) from thirty lakes in the Mississippi Alluvial Valley, June – July 2011. The model fitted was: Unfiltered Turbidity (NTU) = Filtered Turbidity (NTU) + VSS + NVSS ($R^2 = 0.89$; $P < 0.01$).
+ indicates natural log transformation on optical variables.
* indicates square root transformation for concentration variables.

Table 2.3 Published relationships between turbidity and particulate concentration

Suspended solids (y)	Turbidity (x)	Relationship	Range	R^2	Region	Source
TSS (mg/L)	Tain FAU	$y = 32.15 + 0.73\ x$	< 800 FAU (<616 mg/L TSS)	0.93	Australia – Latrobe River	Grayson et al. (1996)
TSS (mg/L)	Tain FAU	$y = -990 + 2.70\ x$	> 800 FAU (>1170 mg/L TSS)	0.88	Australia – Latrobe River	Grayson et al. (1996)
TSS (mg/L)	NTU	$y = 0.15\ x^{1.32}$	0-240 NTU (0-220 TSS)	0.96	Puget – Lowland Streams	Packman et al. (1999)
SSC (g/L)	NTU	$y = 2.78 + 0.00065\ x$	0-60000 NTU (1.5-30 SSC)	0.79	Laboratory	Pavanelli and Bigi (2005)
SPM (g/m^3)	NTU	$y = 15.2 + 0.0103\ Q^{0.7384}\ x$	Q (flow) < 600 m^3/s	0.97	Germany – Elbe River	Pfannkuche and Schmidt (2003)
TSS (mg/L)	FTU	$y = 2.107 + 1.584\ x$	0-50 FTU (0-80 TSS)	0.83	Northeast US – Tidal Saltmarsh	Suk et al. (1998)
TSS (mg/L)	NTU(Unfiltered-Filtered)	$y = 3.64\ x^{0.647}$	0-66 NTU (0-64 TSS)	0.76	Mississippi – Oxbow Lakes	This study (2013)

Table 2.4 Trophic state criteria

Trophic State	**Chlorophyll-*a* (µg/L) criterion**	**# lakes**	**RFU criterion**	**RFU corrected criterion**	**# lakes estimated**
Oligotrophic	<4.2	0	<23.4	<18.8	0
Mesotrophic	<16.1	5	<73.8	<62.4	5
Eutrophic	<42.6	13	<169.5	<148.9	13 (12)
Hypereutrophic	≥42.6	12	≥169.5	≥148.9	12 (13)

Trophic state criteria based on maximum chlorophyll-*a* values proposed by Vollenweider and Kerekes (1980) and based on the RFU values derived in the Mississippi Alluvial Valley, June – July 2011.

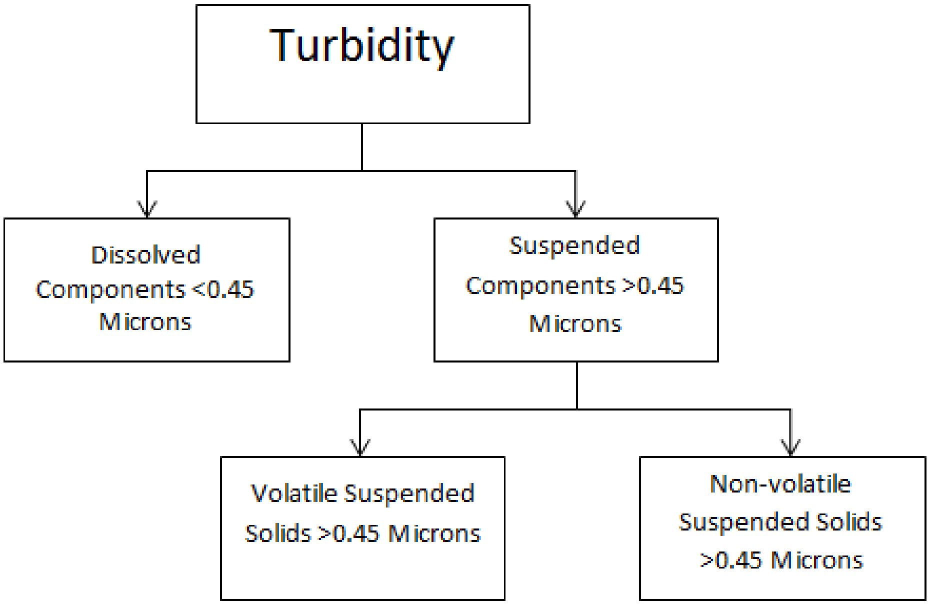

Figure 2.1 Turbidity components

Components of unfiltered, nephelometric turbidity measured in the Mississippi Alluvial Valley, June – July 2011.

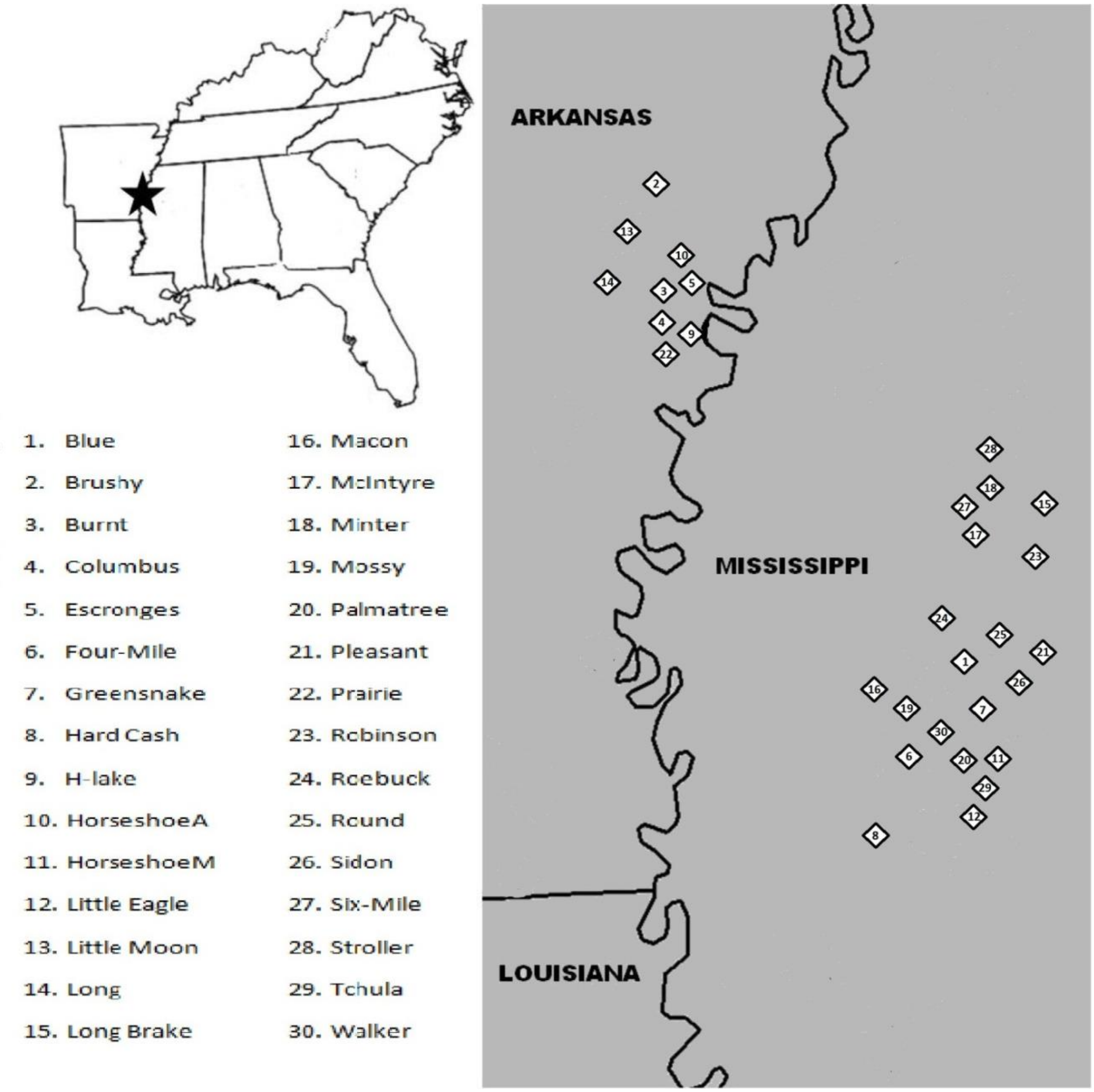

Figure 2.2 Maps of Mississippi Alluvial Valley

Approximate location of thirty oxbow lakes in the Mississippi Alluvial Valley of Mississippi And Arkansas sampled in 2011 (Adapted from Alfermann 2011).

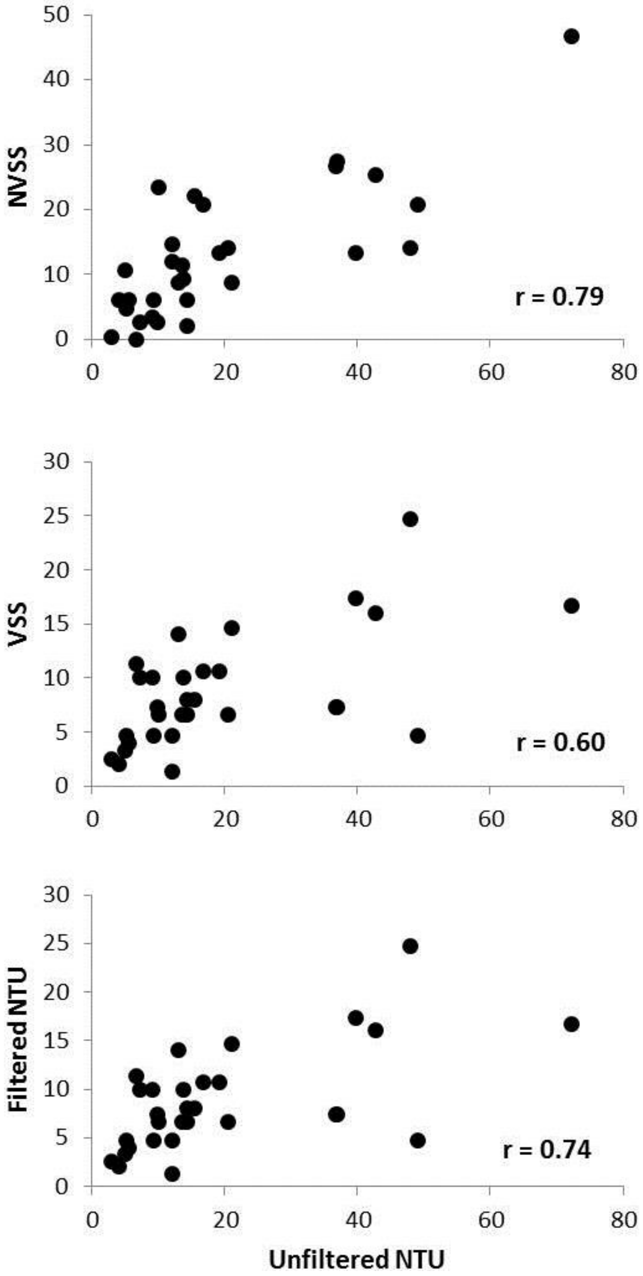

Figure 2.3 Turbidity and suspended solids

Relationships between unfiltered turbidity (NTU) and its individual components (nonvolatile suspended solids – NVSS, volatile suspended solids – VSS, filtered turbidity – filtered NTU) from thirty lakes in the Mississippi Alluvial Valley, June – July 2011.

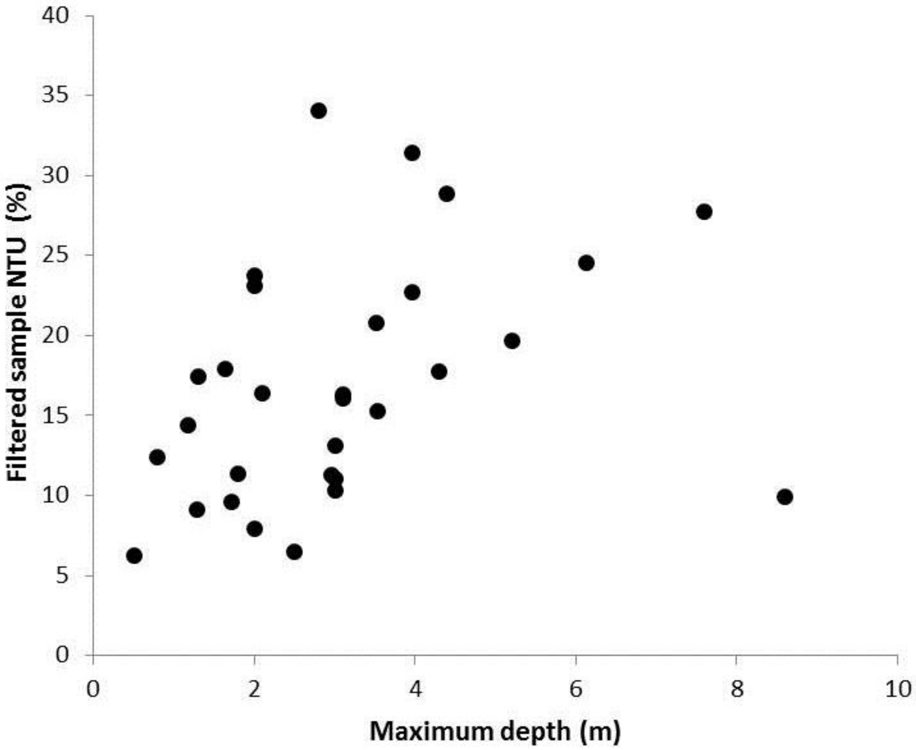

Figure 2.4 Depth and filtered turbidity

Percentage contribution of filtered sample NTU (100 * NTU of filtered sample / NTU of unfiltered sample) versus maximum lake depth from thirty lakes in the Mississippi Alluvial Valley, June – July 2011.

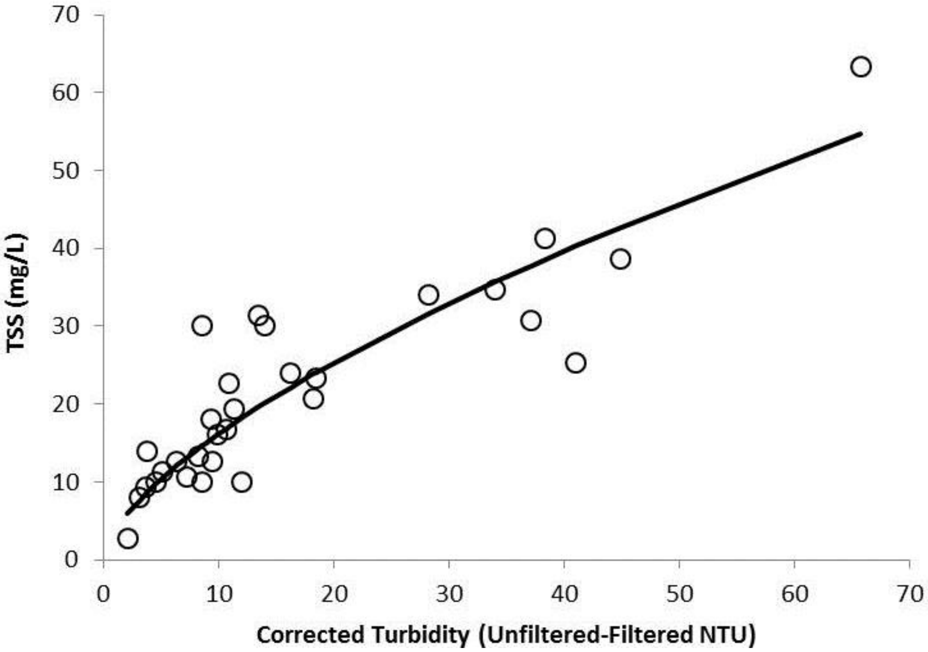

Figure 2.5 Corrected turbidity and total suspended solids

Relationship between total suspended solids (TSS- mg/L) and corrected turbidity (unfiltered-filtered NTU) from thirty lakes in the Mississippi Alluvial Valley, June – July 2011.

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CHAPTER III

SPATIAL PATTERNS OF FISH ASSEMBLAGES IN BEAR CREEK WATERSHED

Introduction

One of the most fundamental terrestrial-aquatic linking units is the watershed. Land use within a watershed has profound effects on water quality within streams and rivers (Allan et al. 1997). Traditionally, land surrounding lotic systems consists of floodplains that provide many ecological functions. For instance, floodplains consisting of bottomland hardwood forests serve to filter and transform sediments and nutrients, especially during periods of high biological activity and moderate to low flow on the floodplain (Harris and Gosselink 1990). Floodplains create important transitional habitat that promote biodiversity associated with additional habitat for aquatic species and opportunities to transform nutrient inputs (Junk et al. 1989; Hoover and Killgore 1998). However, urban development, intensive agricultural practices, and hydrologic manipulation have increased nutrient and sediment loads and drastically altered natural flood pulses, leading to reduced hydrologic connectivity and cultural eutrophication in freshwater environments (Kellison et al. 1998; Allan and Castillo 2007). As a result, anthropogenic influences have been associated with impaired water quality, decreased heterogeneity, and ultimately diminished biodiversity in freshwater systems (Jackson et al. 2001; Tockner and Stanford 2002). Differing hydrologic regimes and internal processes can influence extent of terrestrial-aquatic interactions (Kling et al. 2000), further influencing the magnitude of anthropogenic effects.

Hydrological connectivity, and subsequent transport of biological components through direct water interaction, provides a framework to observe ecosystem effects within and among individual watersheds (Pringle 2001). The hydrological pattern of serially connected lakes provides a unique opportunity to observe hierarchical patterns that may result, as each community depends on parameters related to its position. Specifically, serially connected lakes inherently accumulate watershed drainage area, and as such, surface drained chains experience increased concentrations of total nutrients and algal biomass with increased distance downstream in a chain (Soranno et al. 1999). Furthermore, it has been shown that landscape position, especially as related to stream connection, has a direct influence on water chemistry and lake clarity (Martin and Soranno 2006). Whereas spatial scale along with degree and permanence of connection may influence robustness of patterns, lake position has been used as a primary explanation for variables related to changes in water quality conditions.

Variables such as nutrient concentration, sediments, and phytoplankton, are generally transported by flow which limits influence of water bodies lower in the chain on those higher in the chain. These variables tend to be relatively stochastic and difficult to measure. Conversely, vertebrate communities may provide more stable representations of aquatic systems, and generally are not limited to unidirectional accumulation in areas of low grade and velocity. Yet effects of spatial isolation on fish communities have not been consistent between studies (Jackson et al. 2001; Olden et al. 2001; Beisner et al. 2006). As such, hydrologic connectivity may have a different role in structuring relationships depending on spatial scale. However, most distributions of motile aquatic species, such as fish, are first limited by hydrologic connectivity in the sense that individuals in isolated water bodies have limited possibility of encountering other water bodies. After the initial restrictions of flow on an individual, its resulting position on the landscape is then influenced by conditions related to abiotic conditions and food web dynamics (Jackson et al. 2001). Habitat isolation therefore may even shape fish communities more dominantly than internal processes and water quality (Lonzarich et al. 1998; Olden et al. 2001).

There is debate as to the scale at which landscape metrics associated with individual isolated habitats are appropriate at predicting biotic communities (Hawkins et al. 2000; Johnson and Host 2010). Variability in landcover and inter-basin species differences can make it difficult to investigate local patterns in biotic communities. As such, limiting spatial scale to a single watershed or region may elucidate fine scale trends in hydrologic connectivity as they relate to landscape position and features. Differing hydrologic patterns then may be responsible for influencing and regulating variability of water quality and morphology which in turn structure biotic communities. Within the Mississippi Alluvial Valley (MAV), floodplain lakes express this varying hydrologic connectivity between lakes and to their parent river. Often these lakes are isolated but may become temporarily connected during flood or irrigation events, creating uncertain patterns in connectivity, flow velocity, and biotic movement. Ultimately, frequent perennial hydrologic connectivity incorporates several interrelated dependent variables such as morphology, flow velocity, water quality, and aquatic vertebrate community assemblages. Bear Creek provides a distinctive perennial hydrologic pattern with a string of five serially connected lakes from which to observe potential patterns in fish assemblages relative to lakes formed earlier in geologic time and therefore disconnected from the creek to varying degrees.

This study aims to extend hydrologic connectivity concepts to an artificially disjoined stream to explore floodplain lake fish assemblage dynamics in Bear Creek watershed in west-central Mississippi. Bear Creek connects a chain of five lakes (remnants of an ancient river system), and its watershed includes numerous other such lakes positioned outside the chain but with varying distance and connectivity to the chain. The objective of this study was to explore patterns in fish assemblages and biodiversity within lakes relative to their position in Bear Creek watershed and their serial position along the creek continuum.

Methods

Study sites

Bear Creek watershed encompasses 33,000-44,000 ha of land in Leflore, Sunflower, and Humphreys counties in a flat, highly agricultural, alluvial region of Mississippi (Cooper and Knight 1978). Four lakes: Three-Mile, Six-Mile, Four-Mile, and Wasp, listed in downstream order, are hydrologically connected annually. Blue Lake, the uppermost lake in the string of five lakes, is intermittently connected to the lower four lakes, and is permanently connected to a large deepwater wetland, Gayden Brake, it its upper end (Figure 3.1). Due to its meander size, this combination of lakes is thought to have formed from the most recent track of the Ohio River through this region, and is henceforth referred to as “on-channel lakes” (Fisk 1944). Bear Creek spans approximately 80 km before reaching a water control structure, managed by the U.S. Army Corps of Engineers, which separates the creek form the Yazoo River just north of Belzoni, MS. Eight of twelve additional lakes (henceforth referred to as “off-channel lakes”) of at least 3 ha in size within the watershed, were sampled (Table 3.1). Some of the larger off-channel lakes are believed to be remnants of the former Mississippi-Ohio River that once flowed through the region (Fisk 1944). Other water bodies found throughout the watershed include fifty small ponds (total – 42 ha), numerous sloughs and brakes (total – 2436 ha), and multiple catfish aquaculture ponds (total – 1965 ha) as of 2006 (NHD 2006).

Sample collection

Fish assemblages were assessed with boat electrofishing on each of the five on-channel lakes and eight off-channel lakes in the watershed. Sampling occurred up to twice per lake on independent days at least seven days apart from early summer to early fall of 2012 in addition to ten prior sampling events between 2006 and 2010. Summer represents a time of severe water quality conditions, with greater probabilities of lake isolation due to low water levels, increased algal production, and highly variable dissolved oxygen. Therefore, this is an ideal time to sample fish assemblages as only perennial species exist in such conditions. A GPP 7.5 Smith-Root (Vancouver, Washington) electrofishing unit with pulsed DC 60-Hz current was used for all sampling. Usually four, but up to eight, 15-minute samples were taken per sampling event at each lake. Two netters used 2.7-m dip nets with 0.4-cm mesh to collect fish. All fish were identified to species and enumerated immediately following each sample; unidentifiable species were preserved in 10% formalin and identified with taxonomic keys at a later date (Ross 2001).

Fish collections were arranged by species composition, biodiversity metrics, and Jester et al. (1992) flow affinity guild classification. Relative species percentage composition was calculated from species counts (total count of individuals of a species divided by total count of individuals of all species) for each of the 13 lakes. Eleven biodiversity metrics were calculated using PAST software (Hammer et al. 2001) including four measures of species richness: raw species (total number of species collected), rarefied species (standardized number of species based on smallest sample - 141 individuals), and Margalef's and Menhinick's (both ratios of species to individuals) species richness were included to incorporate various sensitivities to sample size and relative abundance. Shannon-Wiener, Brillouin, and Fisher's diversity indices, dominance (Simpson, Berger-Parker), and evenness (Buzas and Gibson, Pielou's) were additional metrics of biodiversity included. Considering on-channel and off-channel was a major theme in this analysis, fish species were further classified as rheophilic or lacustrine according to Jester et al. (1992).

Statistical analysis

The three fish classification schemes listed above were independently analyzed to assess differences in patterns between on-channel and off-channel lake, and for serial continuity among on-channel lakes. Indirect gradient analysis with Principal Coordinates Analysis (PCoA) was applied to ordinate lakes relative to species composition (Bray-Curtis resemblance) and normalized biodiversity metrics (Euclidean resemblance). Species composition and biodiversity metrics of on-channel lakes were tested for differences against those of off-channel lakes through permutational multivariate ANOVA (PERMANOVA). Similarly, percentage rheophilic species composition of on-channel lakes was tested for differences against those of off-channel lakes through permutational univariate ANOVA (PERANOVA).

Spearman correlations between scores of major PCoA axes and individual lake component values for (1) species composition and (2) biodiversity metrics were used to determine which aspects of species composition and biodiversity metrics most strongly separated lakes. Spearman correlations between scores of major PCoA axes and ranked on-channel lake position (i.e., 1-5) were used to determine if fish assemblages exhibited any serial on-channel pattern for (1) species composition, and (2) biodiversity metrics. Flow affinity guilds were tested to determine if they exhibited any serial on-channel pattern by examining Spearman correlations between percentage rheophilic species and ranked on-channel lake position. Spearman correlations test the relationship between axis score or rheophilic percentage and the specified metric (individual component value or ranked position) of each lake on a linear basis. Therefore, using a two sided Spearman correlation table for individual component values of all thirteen lakes to detect significant correlations with PCoA axes, a significant ($P \leq 0.05$) correlation coefficient (r) would be $r \geq 0.56$. However, to detect serial on-channel patterns, correlation between the five ranked on-channel lakes and specified metrics of PCoA axes scores or rheophilic percentage would need to show all lakes in exactly ascending or descending order to correspond to $P \leq 0.05$, which would result in a correlation coefficient (r) of 1 or -1. All analyses were performed in Program R using the base package (R Core Team 2012), ecodist (Goslee and Urban 2007), and vegan (Jari Oksanen et al. 2012) with $\alpha = 0.05$.

Results

In all, 39 species were collected from lakes in Bear Creek watershed (Table 3.2). Due to inability to capture gizzard shad and threadfin shad because of fleeting behavior in electric current, these species were not included in any analysis or computation except for raw species richness, and provide their inclusion for descriptive narrative. Three of 15 rheophilic species were found only in on-channel lakes (spotted bass, white bass, and American eel) whereas three of 24 lacustrine and one rheophilic species were found only in off-channel lakes (taillight shiner, swamp darter, brown bullhead, river carpsucker). Buffalo species were among the most numerous species in on-channel lakes, with 31% smallmouth buffalo and 9% bigmouth buffalo compared to 8% and 2%, respectively, in off-channel lakes. Conversely, bluegill was the most common species in off-channel lakes and composed an average 37% of catch in off-channel lakes and 15% of catch in on-channel lakes. Swamp darter and brown bullhead were each only found in one lake and composed less than 0.05% of total catch in those lakes; however, removal of these observations did not noticeably change the ordination or eigenvalue scores.

Across the study lakes, raw species richness ranged from 10 to 31 species (Table 3.3). Biodiversity metrics showed a greater range in off-channel lakes compared to on-channel lakes. On-channel lakes generally tended to have greater species richness (rarefied, raw, Margalef and Menhinick), yet there was substantial overlap. On-channel lakes showed elevated median biodiversity scores for all metrics except dominance.

Species PCoA ordination separated on-channel lakes from most off-channel lakes in Axis 1 (Figure 3.2). Axis 1 accounted for 50% of species variability, with axes 2, 3 and 4 accounting for 15, 12, and 7% of species variability, respectively, for a cumulative variance of 85% with four axes. The PERMANOVA indicated significant differences between on-channel and off-channel lakes based on species assemblages (pseudo-F = 5.29, $P < 0.01$). Longitudinal separation was found in on-channel lakes along axis 2, as a trend appeared orienting upstream to downstream channel lakes from positive to negative. Spearman rank correlations confirmed strong correlation between on-channel lake position and axis 2 (r = -0.3, -1.0, -0.7, and -0.2 for axes 1-4, respectively).

Several species were correlated with PCoA axes that separate assemblages between on-channel and off-channel lakes. On-channel lakes (negative axis 1) and two off-channel lakes were strongly correlated with three rheophilic species: bigmouth buffalo (r = -0.82), freshwater drum (r = -0.79), smallmouth buffalo (r = -0.95), followed by three lacustrine species: common carp (r = -0.77), shortnose gar (r = -0.62), and white crappie (r = -0.68). Off-channel lakes (positive axis 1) were correlated (r = 0.85) with increased bluegill contribution. Additionally, lakes in the top half (positive axis 2) of the ordination were correlated with brook silverside (r = 0.90), whereas lakes in the bottom half (negative axis 2) were correlated with orangespotted sunfish (r = -0.87). Five of 37 species showed correlations with axis 3 and two species with axis 4, but these axes, like axis 2 did not separate on-channel and off-channel lakes.

The biodiversity PCoA ordination also separated on-channel lakes from most off-channel lakes along axis 1 (Figure 3.3). The PERMANOVA reflected a moderately significant difference in biodiversity between on-channel and off-channel lakes (pseudo-F = 2.50, P = 0.10). Axis 1 accounted for 85% of biodiversity variation, with axis 2 accounting for an additional 11%. Axis 1 was correlated with nine biodiversity metrics: rarefied (r = -0.87), Margalef's (r = -0.84) and Menhinick's (r = -0.85) richness indices, Brillouin (r = -0.90) and Shannon (r = -0.96) diversity metrics, Simpson's (r = -0.92) and Burger-Parker (r = 0.74) dominance and Fisher's alpha (r = -0.90), and Pielou's evenness (r = -0.82). Axis 2 only correlated with the remaining two biodiversity estimates: raw species richness (r = 0.80) and Buzas and Gibson evenness (r = -0.72). On-channel lake position was not significantly correlated with either axis (r = 0.5, 0.6 for axes 1 and 2, respectively).

Environmental preferences of fish species showed variable contributions to fish assemblages in on-channel and off-channel lakes. Rheophilic species composed 37 to > 67% of on-channel lake assemblages, while only contributing < 1 to 34% to off-channel assemblages. However, the only rheophilic species dominating (> 20% by count) assemblages was smallmouth buffalo in four of five on-channel lakes and one off-channel lake. The PERANOVA confirmed significant difference between on-channel and off-channel contributions of rheophilic species (pseudo-F = 21.86, P < 0.01; Figure 3.4). Conversely, bluegill dominated seven of eight off-channel lakes, and brook silverside, largemouth bass, orangespotted sunfish, and white crappie each separately dominated one of eight off-channel lakes. Spotted gar, a lacustrine species, rather than smallmouth buffalo, dominated Blue Lake, the first of five on-channel lakes – the most isolated and only lake connected to a large brake. There was no correlation between on-channel lake position and rheophilic contribution (r = 0.0).

Discussion

Floodplain lakes are integrated into the landscape of the MAV. Whereas lake morphometry, landscape features, and connectivity to the parent river channel have been shown to have an effect on fish assemblages in MAV floodplain lakes, watershed-level connectivity can also have substantial effects on fish assemblages (Lubinski et al. 2008; Dembkowski and Miranda 2011; Dembkowski and Miranda 2012; Alfermann and Miranda 2013). The dichotomous classification used attempts to simplify a more empirical factor such as flow within a framework of hydrologic connectivity. The homogeneous nature of on-channel assemblages most likely reflects connectivity along the chain of lakes, connectivity to the Yazoo River, as well as homogeneous channel size and fluvial dynamics (Table 3.1). On-channel lake assemblages are also more subject to riverine processes and rely on conditions of the channel above and below each lake. Conversely, off-channel lakes show more heterogeneity in fish assemblages, possibly a reflection of varying degrees of isolation and heterogeneous morphometry. Furthermore, on-channel lakes are potentially more subject to accumulation of nutrients and sediments downstream as time of highest connectivity coincides with time of excess fertilizer runoff and transport, whereas off-channel lakes are more likely to capture and retain inputs from smaller catchments. Considering these differences, biotic functioning in off-channel lakes may rely more on processes associated with flood pulse and in-lake factors.

While longitudinal gradients were found in fish assemblages along on-channel lakes, gradients were not in complete accordance with river continuum concepts. As expected, the fish assemblage within an on-channel lake was more similar to the lakes above and below than to other lakes on the channel, yet no one species or set of species was solely responsible for this trend. Species accumulation was expected, with downstream channel lakes including species from upstream lakes plus additional species, as seen in reservoirs along the length of the Tennessee River (Miranda et al. 2008). This trend was not seen in on-channel lakes, possibly due to a relatively short 80-km system and overall high connectivity among on-channel lakes. Conversely, some physical and resource boundaries contributing to species separation, rather than gradual transitions between lakes could have been unapparent in this analysis, as creek environments were not sampled (Naiman et al. 1988). Additionally, the physical barrier created by the water control structure at the last on-channel lake that accumulates water from Bear Creek watershed, as well as intermittent connectivity between the first and second on-channel lake, may limit flow along the channel, resulting in no correlation between rheophilic species contribution and channel position. However, no off-channel lakes exhibited higher contribution of rheophilic species to fish assemblages, suggesting that fluvial dynamics of Bear Creek, not hydrologic connectivity between lakes exclusively, are important in structuring on-channel assemblages.

On-channel and off-channel lakes differed relative to key fish assemblage characteristics that lend themselves to managing fishery resources to provide diversified fishing opportunities. Six species adequately separated on-channel lakes, representing mostly large-bodied species compared with species associated with off-channel lakes. The large biomass of buffalo in on-channel lakes makes them ideal candidates for commercial fishing; yet buffalo fisheries in this region have declined in recent years due to low profitability (Jackson and Jackson 1989). Recreational species in on-channel lakes, such as white crappie and bluegill, are generally in poor body condition due to high sedimentation loads (Cooper and Knight 1978). The uppermost on-channel lake and all off-channel lakes provide the most opportunity for management of water quality and lacustrine species such as centrarchids, as a result of their isolated nature and smaller proportions of watershed drainage. Off-channel lakes also have the potential to reach greater overall biodiversity as evidenced by several lakes exhibiting greater species richness and evenness than any on-channel lake. Furthermore, sedimentation rates in off-channel lakes are generally slower than in on-channel lakes, decreasing issues with turbidity and allowing manipulation options (e.g., dredging, weirs) to have a larger impact and longer lifespan (Cooper and Knight 1978).

However, these lakes do not simply provide habitat for biotic communities. Surface water in the MAV is used to meet demands of agriculture and aquaculture, and as such, lakes, streams and the surrounding landscape are subject to alteration to meet field and pond irrigation and drainage needs as well as provide flood protection (Wilber et al. 1996). As of 2005, approximately 3.5% (over 26 million liters per day) of freshwater demands were met with surface waters in the three counties containing the Bear Creek watershed (Kenny et al. 2009). However, groundwater supplies, which provide the other 96.5% of freshwater, are being depleted and may not keep up with irrigation demands, requiring escalating supplementation with surface water capture and recovery (Evett et al. 2003). Declines in the alluvial aquifer underlying the MAV have already shown failure to maintain base flow in several creeks and rivers in the region (Pennington 2006). Drawing of groundwater and manipulations to surface waters to provide freshwaters for irrigation demands have potential to alter depth, connectivity, and flow velocity between lakes within a watershed.

Agricultural practices have traditionally introduced pulsed hydrologic events, moving water off the catchment and into creek systems quickly. Maintaining some hydrologic connectivity to larger agriculturally induced habitats reportedly provides some benefit to smaller stream fishes (Smiley et al. 1998). However, edge-of-field best management practices are increasingly holding back water and shifting to on-site storage to supply freshwater – improving water quality in streams and floodplain lakes, but potentially altering amount and intensity of runoff that feeds connecting channels and streams and further exacerbating isolation (Kröger et al. 2008; Kröger et al. 2013). Floodplain lakes themselves are often subject to flow control structures to avoid complete desiccation, for maintenance of surface water supply, and/or recreational opportunities.

Sky Lake, another floodplain lake in the Bear Creek watershed but not included in this study, had a flow control structure installed in 1980, but still remains subject to periodic desiccation (Davidson et al. 2007). Variability of influence of small-scale flow control structures on hydrologic connectivity adds complexity not accounted for in this account of fish assemblages in the Bear Creek watershed.

The water-control structure installed at the confluence of Bear Creek and the Yazoo River in 1983 increased residence time of water originating from the Bear Creek watershed and limits back-flooding of agricultural fields from the Yazoo (Pennington et al. 1991). Currently, the water control structure is not managed to release water from Bear Creek watershed when water drops below 30 m above mean sea level (which regularly occurs in summer), and does not allow high flow conditions from the Yazoo to back into Bear Creek (MDEQ 2003). As such, normal hydrologic regimes are not maintained, and the creek and its respective on-channel lakes function more like a parent river as source water, and consequently biota, at the bottommost lake, Wasp Lake, is retained from the watershed rather than intermixed with the Yazoo.

Without the water control structure retaining water at 30 m, creek connections would be more likely to desiccate during dry months but improve during wet months when the Yazoo River would be backing into the Bear Creek watershed, thereby changing dynamics of connectivity and disturbance. To partially simulate this natural flooding, operation of high water gates could potentially be manipulated to increase water level in Bear Creek and thus allow hydrologic connectivity to isolated lakes on a scheduled basis. While potential effects of such enhanced connectivity were not investigated directly, these results show such manipulations to connectivity could have a substantial impact on fish assemblages, and potentially shift lake systems to more homogenized assemblages with increased incidence of rheophilic species, possibly more in line with their pre-water control structure condition. Finer scales of hydrologic connectivity and potential responses and interaction of channels (natural and anthropogenic) and smaller water control structures (e.g., weirs, culverts) need further investigation before managing an entire watershed to restore or artificially induce flood regimes and connectivity.

Table 3.1 Characteristics of Bear Creek watershed

Lake name	Lake code	Length (km)	Maximum width (km)	Maximum summer depth (m)	Area (ha)
Blue	1	3.8	0.08	4.4	20.8
Threemile	2	2.4	0.07	n/a	10.3
Sixmile	3	5.3	0.11	4	40.3
Fourmile	4	5.4	0.13	2.9	36.3
Wasp	5	17.2	0.19	2.3	211
McCoy	A	1.4	0.1	2.7	10
Jug	B	1.1	0.07	3.2	5.3
Cat	C	0.7	0.6	1.2	3.5
Mossy	D	4.5	0.27	2	79.7
Macon	E	1.8	0.12	3.1	14
Walker	F	2	0.11	3.1	16.3
Otter	G	0.5	0.92	1.7	3.1
Ole	H	1	0.22	0.7	14.7

Morphometric characteristics of length, maximum width, maximum summer depth, and area for 13 lakes in Bear Creek watershed. Alphanumeric lake code corresponds to Figure 3.1, with numbers representing on-channel lakes and letters representing off-channel lakes.

Table 3.2 Fish species collected in Bear Creek watershed

Species	**Rheophilic**	**On-channel lakes (n=5)**	**Off-channel lakes (n=8)**
American eel (*Anguilla rostrata*)	Y	2 (0.03)	0 (0)
Bigmouth buffalo (*Ictiobus cyprinellus*)	Y	5 (8.78)	6 (1.85)
Black buffalo (*Ictiobus niger*)	Y	3 (0.05)	2 (0.02)
Black crappie (*Pomoxis nigromaculatus*)	N	5 (0.34)	5 (0.17)
Blackspotted topminnow (*Fundulus olivaceus*)	N	2 (0.15)	5 (0.74)
Blue catfish (*Ictalurus furcatus*)	Y	3 (0.09)	1 (0.01)
Bluegill (*Lepomis macrochirus*)	N	5 (14.71)	8 (36.72)
Bowfin (*Amia calva*)	N	5 (0.92)	5 (0.54)
Brook Silverside (*Labidesthes sicculus*)	Y	5 (3.27)	7 (6.44)
Brown bullhead (*Ameiurus nebulosus*)	N	0 (0)	1 (0.01)
Channel catfish (*Ictalurus punctatus*)	Y	3 (0.46)	6 (0.71)
Common carp (*Cyprinus carpio*)	N	5 (1.81)	5 (0.45)
Emerald shiner (*Notropis atherinoides*)	Y	2 (0.15)	3 (0.18)
Flathead catfish (*Pylodictis olivaris*)	Y	3 (0.19)	1 (0.01)
Freshwater drum (*Aplodinotus grunniens*)	Y	5 (4.26)	5 (1.23)
Gizzard shad (*Dorosoma cepedianum*)	N	5 (NA)	8 (NA)
Golden shiner (*Notemigonus crysoleucas*)	N	3 (0.57)	2 (0.43)
Green sunfish (*Lepomis cyanellus*)	N	4 (0.55)	4 (0.53)
Golden topminnow (*Fundulus chrysotus*)	N	1 (0.28)	0 (0)
Largemouth bass (*Micropterus salmoides*)	N	5 (3.55)	8 (7.45)
Longear sunfish (*Lepomis megalotis*)	Y	5 (5.54)	4 (2.51)
Longnose gar (*Lepisosteus osseus*)	Y	3 (0.62)	2 (0.03)
Orangespotted sunfish (*Lepomis humilis*)	N	4 (3.69)	7 (13.82)
Pirate perch (*Aphredoderus sayanus*)	N	2 (0.12)	1 (0.01)
Pugnose minnow (*Opsopoeodus emiliae*)	N	3 (0.40)	5 (0.94)
Redear sunfish (*Lepomis microlophus*)	N	1 (0.02)	3 (1.20)
River carpsucker (*Carpiodes carpio*)	Y	0 (0)	1 (0.20)
Shortnose gar (*Lepisosteus platostomus*)	N	5 (0.86)	3 (0.09)
Smallmouth buffalo (*Ictiobus bubalus*)	Y	5 (31.88)	6 (7.93)
Spotted bass (*Micropterus punctulatus*)	Y	1 (0.02)	0 (0)
Spotted gar (*Lepisosteus oculatus*)	N	5 (8.85)	8 (3.97)
Swamp darter (*Etheostoma fusiform*)	N	0 (0)	1 (0.01)
Taillight shiner (*Notropis maculatus*)	N	0 (0)	6 (1.64)
Threadfin shad (*Dorosoma petenense*)	N	5 (NA)	5 (NA)
Warmouth (*Lepomis gulosus*)	N	5 (1.17)	8 (1.62)
Western mosquitofish (*Gambusia affinis*)	N	5 (1.13)	7 (2.29)
White bass (*Morone chrysops*)	Y	2 (0.18)	0 (0)
White crappie (*Pomoxis annularis*)	N	5 (6.38)	7 (6.18)
Yellow bullhead (*Ameiurus natalis*)	N	1 (0.04)	3 (0.08)

Fish species collected and percentage composition in 13 Bear Creek watershed floodplain lakes sampled 2006-2012. Rheophilic classification follows Jester et al. (1992) where Y=rheophilic and N=lacustrine. Numbers represent count of on-channel and off-channel lakes from which the species was collected. Parentheses next to counts represent the average percentage (by count) of species within all on-channel and off-channel lakes.

Table 3.3 Fish biodiversity in Bear Creek watershed

		Statistic				
Metric	Channel	min	25%	median	75%	max
Raw species richness	**Off**	10	15.5	21	22.5	31
	On	19	26	26	28	28
Rarefied species richness	**Off**	5.1	10.7	13.7	15.7	19.6
	On	15.6	16.0	16.1	17.1	17.5
Margalef's species richness	**Off**	1.1	2.1	2.7	3.1	4.1
	On	3.3	3.3	3.3	3.4	3.7
Menhinick's species richness	**Off**	0.3	0.5	0.6	0.8	1.3
	On	0.6	0.8	0.8	0.8	1.4
Shannon-Wiener diversity	**Off**	0.9	1.6	1.8	2.1	2.4
	On	2.1	2.1	2.2	2.2	2.3
Brillouin diversity	**Off**	0.9	1.6	1.8	2.1	2.2
	On	2.0	2.1	2.2	2.2	2.2
Fisher's diversity	**Off**	1.3	2.6	3.4	4.3	5.6
	On	4.3	4.4	4.5	4.9	5.6
Simpson's dominance	**Off**	0.5	0.7	0.8	0.8	0.9
	On	0.8	0.8	0.8	0.8	0.9
Berger-Parker dominance	**Off**	0.2	0.3	0.4	0.5	0.6
	On	0.2	0.3	0.4	0.4	0.4
Buzas and Gibson's evenness	**Off**	0.2	0.3	0.4	0.4	0.5
	On	0.3	0.4	0.4	0.4	0.5
Pielou's evenness	**Off**	0.4	0.6	0.7	0.7	0.8
	On	0.7	0.7	0.7	0.7	0.7

Fish biodiversity metrics of on-channel and off-channel lakes in Bear Creek watershed, Mississippi, sampled 2006-2012.

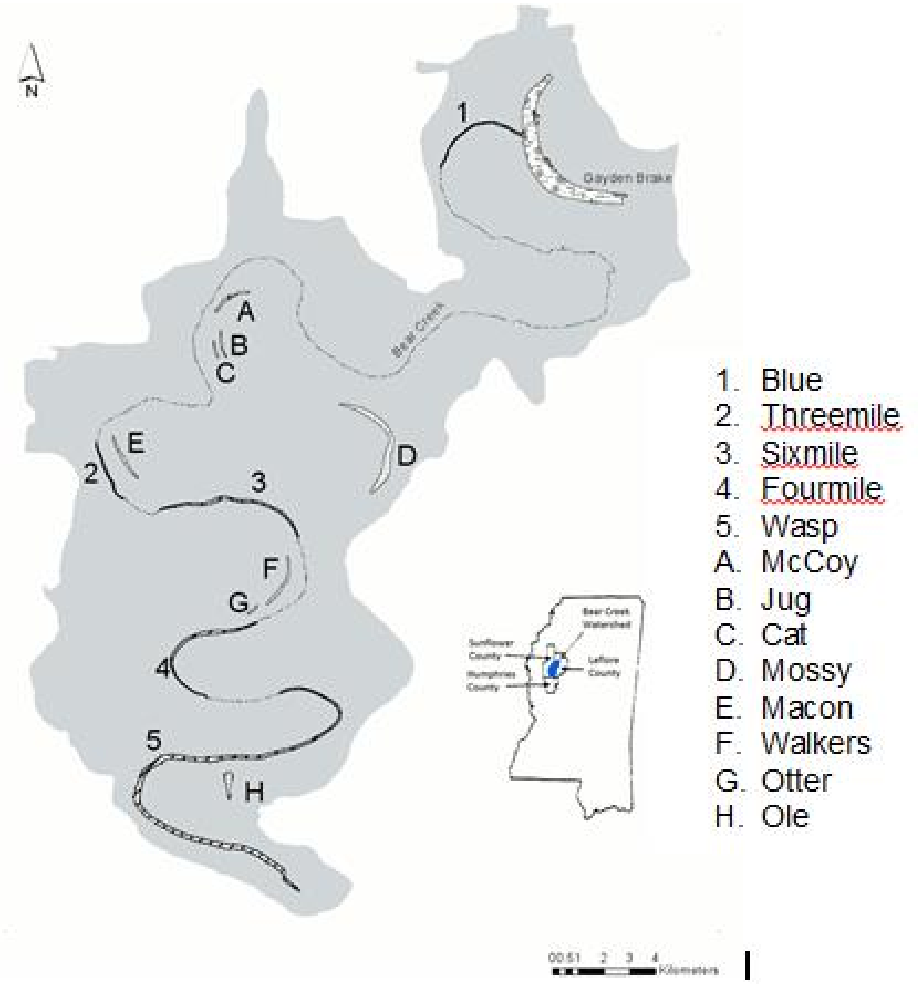

Figure 3.1 Schematic map of Bear Creek watershed

Only water bodies sampled 2006-2012 are included. Numbers represent on-channel lakes, whereas letters represent off-channel lakes, in longitudinal alpha-numeric order.

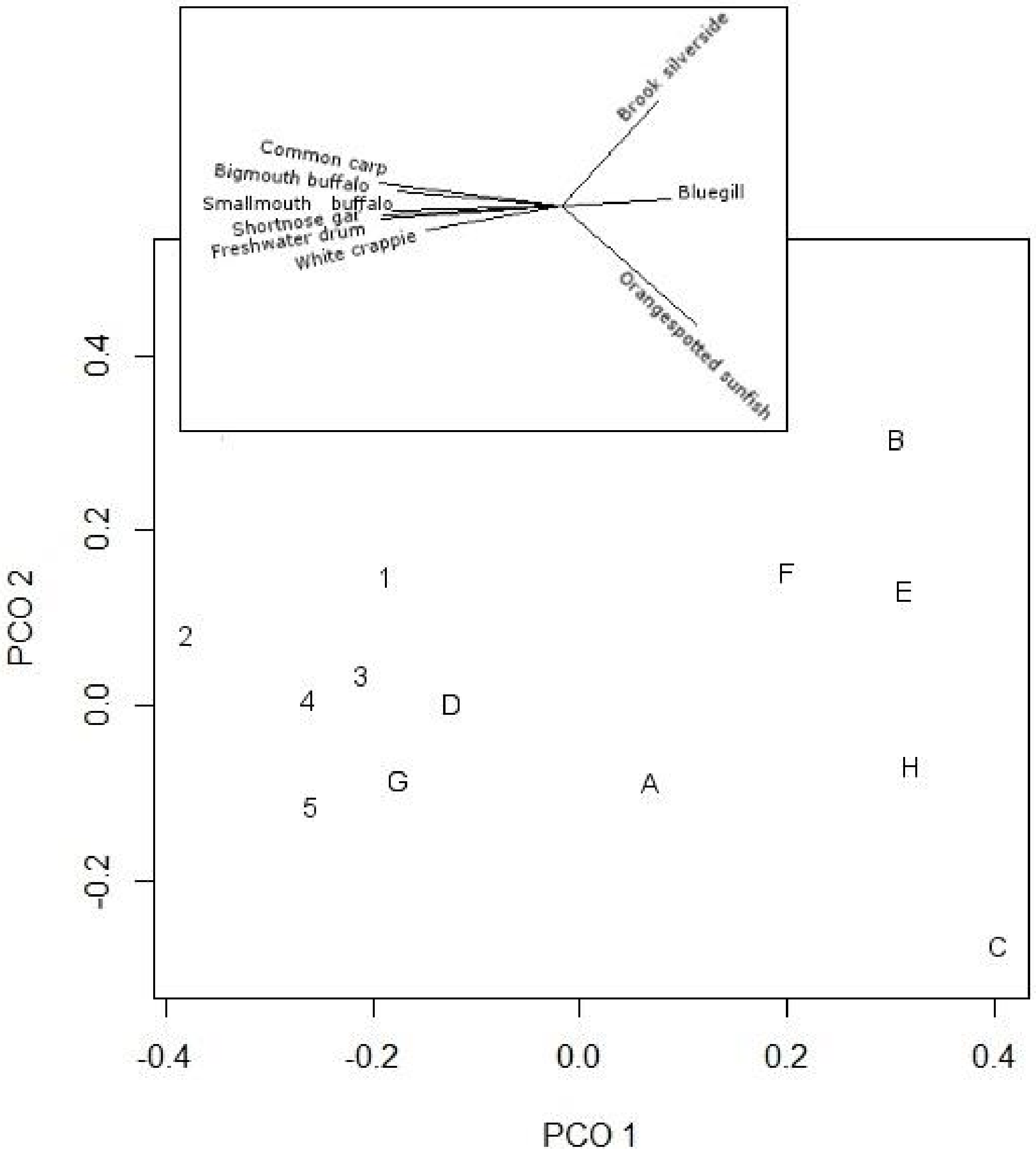

Figure 3.2 Species principal coordinates analysis

Principal coordinates analysis (PCoA) of 13 lakes sampled 2006-2013 in Bear Creek watershed relative to species assemblage composition. Alphanumeric representations of lakes correspond to Figure 3.1, with numbers representing on-channel lakes and letters representing off-channel lakes. Species showing a significant Spearman’s rank correlation ($r \geq 0.56$, $P \leq 0.05$) with Axes 1 or 2 are shown in upper inset.

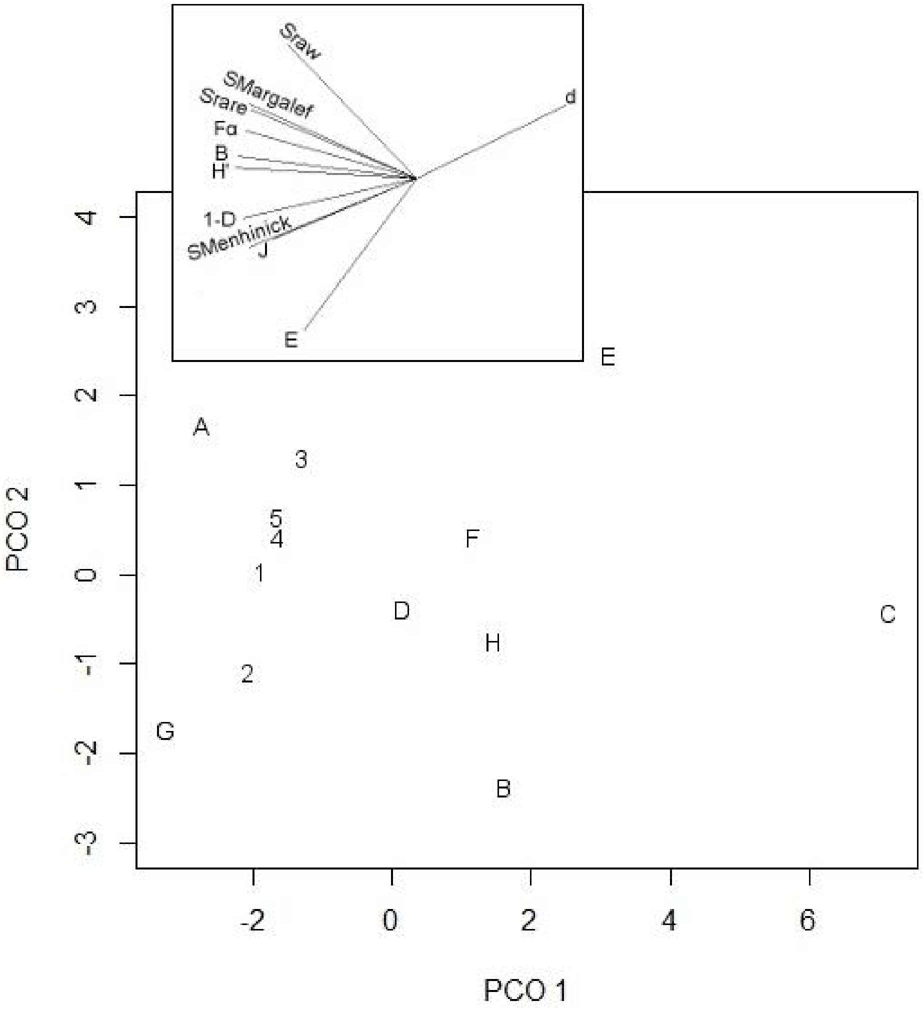

Figure 3.3 Biodiversity principal coordinates analysis

Principal coordinates analysis (PCoA) of 13 lakes sampled 2006-2012 in Bear Creek watershed relative to biodiversity metrics. Alphanumeric representations of lakes correspond to Figure 3.1, with numbers representing on-channel lakes and letters representing off-channel lakes. Fish biodiversity metrics raw species richness (Sraw), rarefied species richness (Srare), Margalef's species richness (SMargalef), Menhinick's species richness (SMenhinick), Shannon-Wiener diversity (H'), Brilloiun diversity (B), Fisher's diversity (Fα), Simpson's dominance (1-D), Berger-Parker dominance (d), Buzas and Gibson's evenness (E), and Pielou's evenness (J) showing a significant Spearman's rank correlation ($r \geq 0.56$, $P \leq 0.05$) with Axes 1 or 2 are shown in upper inset.

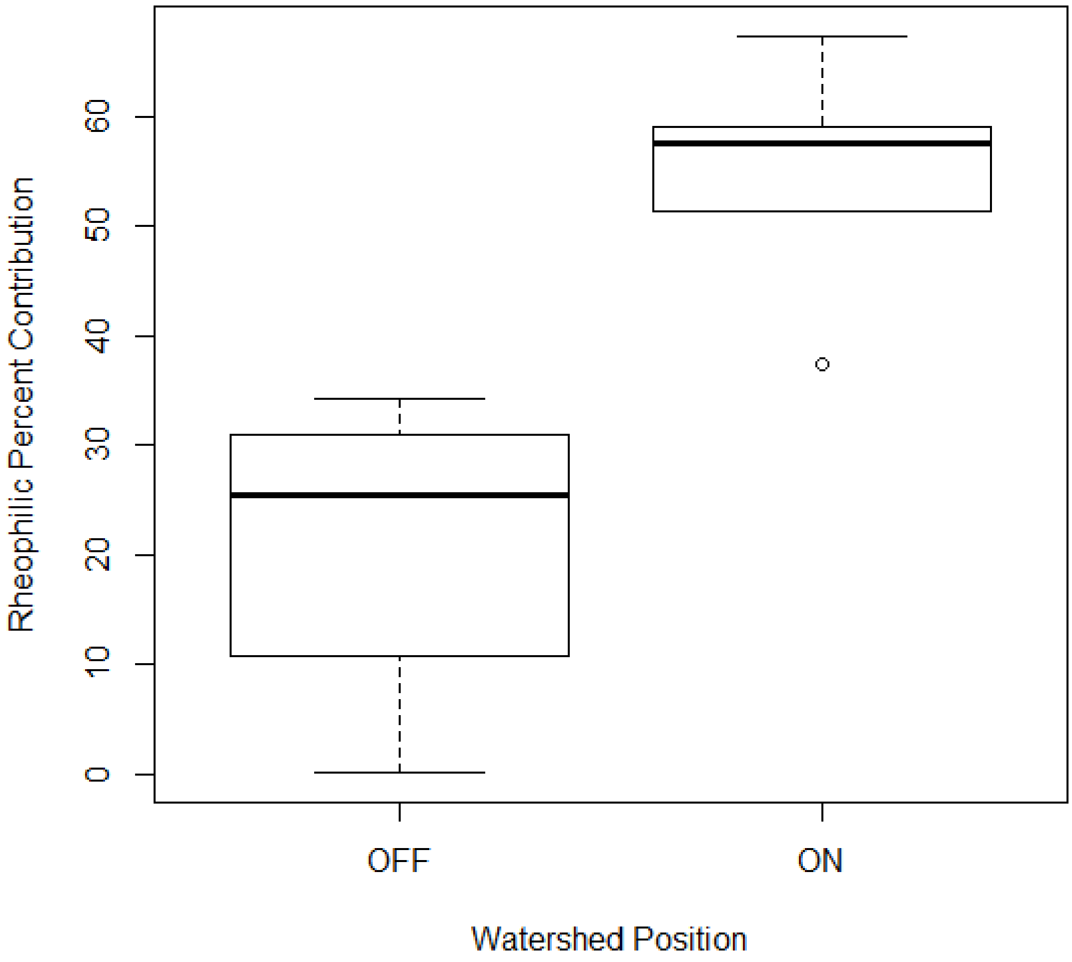

Figure 3.4 Rheophilic contribution

Differences in percentage of summer rheophilic species contribution to fish assemblages in Bear Creek watershed on-channel and off-channel lakes sampled intermittently 2006-2012. Boxes represent the 25th and 75th percentile, with the dark band representing the 50th percentile, and whiskers extending to 1.5 times interquartile range. One outlier is represented by the open circle for on-channel lakes.

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CHAPTER IV

WATER QUALITY AND FISH DYNAMICS IN WETLANDS ASSOCIATED WITH A FLOODPLAIN LAKE

Introduction

Rivers meandering in their floodplain create a variety of water bodies including isolated and contiguous sloughs, oxbow lakes, seasonally inundated floodplains, and small, shallow floodplain pools, as described in detail by Baker et al. (1991). In the southeastern United States, forested wetlands comprise more than 75% of all wetlands in floodplains and are typified by a mix of willows (*Salix* spp.), river birch (*Betula nigra*), cottonwood (*Populus deltoides*), overcup oak (*Quercus lyrata*), water hickory (*Carya aquatica*), bald cypress (*Taxodium distichum*), tupelos (*Nyssa* spp.), and other species depending on wetland age, hydrology, succession stage, and proximity to a stream channel (Hodges 1997; Shepard et al. 1998). Other than large woody vegetation, wetland plant communities in littoral zones of water bodies in the region consist of submerged, emergent, and floating species such as southern naiad (*Najas guadalupensis*), pondweeds (*Potamogeton* spp.), spikerushes (*Eleocharis* spp.), American lotus (*Nelumbo lutea*), common duckweed (*Lemna minor*), and numerous others (Godfrey and Wooten 1981; Smart et al. 1996; Dibble et al. 1997).

Three forms of wetlands are often associated with floodplain lakes. First, transitional areas between lake and upland areas form margin wetlands that may be narrow or wide depending on topography (Snodgrass and Burger 2001). Dependent on how the lake was formed, the littoral zone may have a natural levee that allows only a narrow margin wetland. “Ridge and swale” formations found in large meanders may provide a wider margin wetland on the swale side of the lake or access to additional depressional lands inside the meander bend (Hodges 1997). Moderate among-year and low to moderate within-year variation in water level promotes the development of short-lived and facultative wetland plant communities in the margins of floodplain water bodies (Hill et al. 1998). Additionally, water level fluctuation is extremely important in determining aquatic macrophyte succession in floodplain lakes (Van Geest et al. 2005). Oxbow lakes, once surrounded by forested systems, may retain remnants of tree communities mentioned above, incorporated into margin wetlands exclusively or in conjunction with vegetation (Gastaldo et al. 1989; Baker et al. 1991; Mitsch and Gosselink 2007). As lakes age, sedimentation and associated loss of depth can enlarge margin wetlands. Second, if the floodplain lake formed as a true oxbow, sediment plugs would be found at one or both ends of the horseshoe shape (Mitsch and Gosselink 2007). These ends form gradual sloping transitional zones from the deep meander bend outwards, at some point becoming shallow and fluctuating enough to support vegetated wetland ecotones (Westlake et al. 2009). Third, lakes may abut contiguous brakes and ancient oxbow lakes in forested or vegetative succession, remnants of the same river system or of prehistoric river systems that once meandered over the same floodplain (Fisk 1944). Wetlands adjacent to floodplain lakes may provide additional habitat heterogeneity and function similar to aquatic/terrestrial transition zones in rivers (Junk et al. 1989), especially in isolated systems with infrequent connection to main channels.

Surface water conditions in forested wetlands associated with floodplain lakes are often warm, stagnant, and oxygen depleted, depending on season (Conner and Buford

1998). Yet, within a region with high fish biodiversity such as the Mississippi Alluvial Valley, forested wetlands support over 75% of fish species within the region (Baker et al. 1991; Hoover and Killgore 1998). Typically, shallow wetland systems are dominated by small species, especially centrarchids, percids, cyprinids, and particular species adapted to tolerate high temperatures and low dissolved oxygen (e.g., *Amia calva*, *Lepisosteus* spp.; Baker et al. 1991; Killgore and Miller 1995; Killgore and Baker 1996; Hoover and Killgore 1998). Surface water conditions, and thereby fish communities, are more heterogeneous in floodplain lakes. Fish communities are often regulated by connectivity, depth, turbidity, and eutrophication, nearly inextricably (Scheffer 2004; Miranda 2011; Dembkowski and Miranda 2012). While conditions may vary from lake to lake, within deep water bodies, conditions are generally more stable than within forested wetlands and thus promote communities of large-bodied centrarchids, ictalurids, and catostomids capable of supporting recreational or artisanal fisheries (Baker et al. 1991). Eventually, floodplain lakes transition into shallow, forested wetlands unless recaptured by a river channel (Wren et al. 2008).

Inhabitance of permanent wetlands adjacent to floodplain lakes by juvenile fish may provide explanations for habitat-specific recruitment effort; yet little, if any, is known on the interaction of wetlands with floodplain lakes. Occurrence and survival of larval and juvenile fish depends on numerous factors including size of the spawning stock, environmental conditions, and predator/prey densities. These factors result in interacting mechanisms that drive differences between juvenile fish occurrence within a lake. The objective of this chapter was to explore differences in spring environmental conditions (temperature and oxygen) as well as differences in small, mostly juvenile, fish assemblages among three habitat types including the pelagic environment, as well as margin and contiguous forested wetlands next to the floodplain lake. This chapter is not an attempt to link environmental conditions to fish assemblage composition; rather, it is an attempt to uncover where differences may lie between permanent habitats associated with a floodplain lake.

Methods

Study site

Blue Lake is an abandoned-channel floodplain lake within Bear Creek watershed in Leflore County, Mississippi (Figure 4.1). The lake is curved but does not form a strong meander neck, and exhibits shallowing at its lower end before turning into Bear Creek. The lake has a maximum depth of approximately 4.4 m, a length of 3.5 km, total area of 21 ha, and is permanently connected to Gayden Brake (total area of 334 ha), a cypress-tupelo swamp with depths of approximately 0.5 – 1 m. Reportedly, Blue Lake has the greatest water clarity (mean Secchi depth > 60 cm) and least mean dissolved oxygen (DO; 6.5 mg/L) of seven lakes in the Bear Creek watershed (Price 1980). The northeastern end of Blue Lake that connects to Gayden Brake represents a unique transition zone that includes pelagic environment as well as margin and contiguous forested wetlands (Figure 4.1).

Sample collection

Diel temperature and oxygen trends were determined in the three habitats with a Manta Multiprobe (Eureka Environmental, Austin, TX). Each multiprobe was situated 0.5-m below the surface and set to record water quality parameters every hour in a stationary location within 100-m transects in each habitat (Figure 4.1). Multiprobes were collected at least every other week for data download and maintenance. Water quality trends were represented with four parameters including mean temperature, maximum temperature, mean DO, and minimum DO computed from complete 24-hour diel cycles.

Trends in juvenile fish composition were assessed with passive larval light traps fished in transects within each of the three habitats. Passive larval light traps, rather than active sampling gear, were used due to the structural complexity in the margin and contiguous wetlands, making push or tow nets difficult to operate. Light traps are selective for small fish attracted to light and floating debris. Light trap design was based on a modified quatrefoil design, approximately 25x25x30 cm with a 7-mm wide vertical entrance. A total of 18 units were deployed equally along the three habitats. Traps were suspended, with the bottom of the trap positioned approximately in line with the Manta multiprobes. Four of six traps fished in each habitat were illuminated with modified solar LED lights; the remaining two traps were unlighted. Lighted and unlighted traps were expected to select different but overlapping aspects of the fish assemblage, but trap selectivity was not an objective of this study (Gregory and Powles 1985; Doherty 1987). Traps were placed approximately 20-m apart so that they would be independent of the light produced by a nearby trap. Traps were deployed weekly from 3 March through 29 June, 2012 to ensure sampling of species over a wide range of spawning temperatures. All traps were deployed at dusk and retrieved at dawn approximately 40 h later, allowing for two nightly cycles. Upon collection, individual trap contents were sieved, preserved in 10% formalin, and stored frozen until identification at Mississippi State University with taxonomic keys (Wallus and Simon 2008). Catches were pooled across light traps and enumerated by habitat type. Selected species were further separated into size groups suggested based on life history characteristics as compiled by Ross (2001) and Wallus and Simon (2008), or bimodality in the length data.

Statistical analysis

Water quality and fish data matrices were independently analyzed to assess differences in patterns between the three habitats. Each data matrix was analyzed with a multivariate analysis of covariance (MANCOVA) with temperature and oxygen variables or species counts as the response variables and habitat type as the predictor variable, Julian day as a covariate, and an interaction term between habitat type and Julian day. Julian day was included to account for temporal variation across the season, whereas the interaction term was included to determine potential changes in response of water quality and fish variables among habitats as the season progressed. Variables were $\log_e$ or $\log_e$ (x+1) transformed, when appropriate, to meet assumptions of linearity or distribution prior to MANCOVA.

If MANCOVA identified statistically significant differences ($P \leq 0.10$) among habitats, a univariate analysis of covariance (ANCOVA) was applied to describe trends in individual variables. Water quality variables were examined over mean and maxima. Fish variables were examined as cumulative catch across Julian day for the most abundant taxa. Cumulative catch was used to facilitate analysis and interpretation of collections that sometimes were low or variable across time. Cumulative catches were expected to exhibit a sigmoidal response across time, with a relatively flat curve initially as few fish were caught early in the season, a rising curve as catch rates increased through mid-season, and a flattening curve once catch rates slowed down or stopped later in the season. The steepness of the rising curve and time of rise differentiated among habitats according to taxa. The sigmoidal pattern was fit with a linearized logistic model as:

$$y = b_0 + b_1 Julian + b_2 habitat + b_3 (Julian \times habitat) \tag{4.1}$$

where

$$y = \log_e\left(\frac{1}{Pcum} - 1\right)$$

Pcum = cumulative frequency of catch express as a proportion ranging from 0 to 1,

b_0= intercept of linear model,

b_1= slope of linear model for Julian day,

b_2= linear intercept modification for habitat, and,

b_3= linear slope modification for Julian day according to habitat.

In equation 4.1, $(b_0 + b_2) / -(b_1 + b_3)$ represents the inflection point (i) in the logistic curve given in Julian days. Conversely, $b_1 + b_3$ represents the steepness (s) of the logistic curve, with smaller (more negative) values corresponding to a steeper rise to total cumulative catch. The parameters i and s are provided to compare among habitats as they relate to plotted logistic curves back-transformed to cumulative counts from the cumulative proportions predicted by equation 4.1. No statistical testing was applied to the ANCOVAs as the prior MANCOVA already provided a global test for differences in taxa between habitats while accounting for Julian day. All analyses were run using the GLM procedure (SAS 2008).

Results

During the study period, water depths in the sampling transects were 0.5 - 1 m along the margin wetland, 1 m in the contiguous wetland, and 3 m in the pelagic environment. Complete diel water quality cycles were recorded over 95 days between 6 April (Julian day 96) and 22 July (Julian day 203), 2012. The water quality data overlapped but did not directly coincide with light-trap samples, as the water quality equipment could be deployed for longer periods without attention. However, due to unforeseen errors associated with multiprobe deployment, habitat types were unequally sampled, with 20 cycles in pelagic environment, 45 cycles in the margin wetland, and 30 cycles in the contiguous wetland.

The MANCOVA detected spatial ($F = 4.8$, $P < 0.001$), temporal ($F = 80.1$, $P < 0.001$), and interaction ($F = 5.1$, $P < 0.001$) effects on temperature and DO descriptors. Overall pairwise comparisons revealed the pelagic environment and contiguous wetlands were most similar ($F = 0.7$, $P = 0.603$), with margin wetlands markedly different than the pelagic environment or contiguous wetlands ($F = 6.7$, 6.1, respectively; $P < 0.001$, each) in water quality parameters. Examination of individual variables showed mean and maximum temperatures were markedly alike among the three habitats, and became more alike through the season (Figure 4.2). Similarly, as mean DO fell, minimum DO also fell. The contiguous wetland and pelagic environment followed similar seasonal temperature and oxygen trends. However, mean temperature (25.6°C) in the contiguous wetland was on average 1.6°C warmer than pelagic water temperature (24.0°C). Similarly, mean DO (3.4 mg/L) in the contiguous wetland was on average 1.8 mg/L less than pelagic mean DO (5.2 mg/L). In terms of oxygen saturation, mean saturation (41%) in the contiguous wetland was on average 20% less than pelagic mean saturation (61%). Given the observed differences in temperature, saturation was expected to be 2% less in the contiguous wetland rather than 20%. Given the moderate temperature, and wide range in oxygen conditions, the mean saturation in the margin wetland was 54% but ranged from 25% to 96%. Mean temperature in the margin wetland rose slower than in the other two habitats (0.09°C vs. 0.13°C per day) whereas mean DO fell faster (0.13 mg/L vs 0.02 mg/L per day at Julian day 100). Due to a parallel steep decline in minimum DO, the margin wetland reached minimum DO levels of 2 mg/L by Julian day 160, 30 days sooner than the contiguous wetland.

A total of 940 fish representing 14 taxa were collected over the total 270 traps fished in 15 sampling events (Table 4.1). The most numerous individuals represented three taxa – *Labidesthes* (n = 227), *Lepomis* (n = 324), and *Micropterus* (n = 313). While not all individuals collected represented larval or juvenile life histories, all individuals were included in analyses to incorporate potential resource overlap. To account for differences in life histories of the most numerous taxa, *Lepomis* and *Micropterus* were each classified into three size classes: < 20 mm, 20-39 mm, and ≥ 40 mm. *Labidesthes* individuals were classified as ≤ 39 mm, and 40 mm and larger. Each size classification was then analyzed as a separate variable.

Raw abundance was greatest in the margin wetland (n = 392) followed by the contiguous wetland (n = 335) and pelagic environment (n = 213; Table 4.1). Taxa richness followed a similar pattern (Table 4.1). Individual light traps generally collected fewer than 10 fish per set, except on 24 March when 80 *Micropterus* individuals < 20 mm long were captured in a single trap fished in the margin wetland. The MANCOVA detected spatial (F = 1.8, P = 0.046), temporal (F = 8.2, P < 0.001), and interaction (F = 2.6, P = 0.003) effects on counts of 19 taxa/size variables. Overall pairwise comparisons revealed contiguous wetlands were most similar to the pelagic environment and margin wetlands (F = 1.6, both; P = 0.156, 0.152, respectively) whereas the pelagic environment and margin wetlands were less similar (F = 2.1, P = 0.067) in fish communities. Logistic curves for *Labidesthes*, *Lepomis*, and *Micropterus* (Figure 4.3) had inflection points (*i*) ranging from Julian day 16 to 243 and slope values (*s*) ranging from 0 to -0.11 (Table 4.2). Estimates of *i* occasionally fell below or above the minimum and maximum Julian days sampled. The logistic model assumes symmetry; when cumulative catch was high in early weeks, the inflection point *i* was predicted to have occurred before the sampling began; conversely, when most of the cumulative catch occurred in later weeks, the inflection point was predicted to have occurred after the sampling stopped. Catch was most variable in the pelagic environment, with larger centrarchids (*Lepomis* ≥ 20 mm, *Micropterus* ≥ 40 mm) absent, or present in very small abundances, compared to the other two habitats. Contiguous and margin wetlands had similar trends in *i*, *s*, and abundance for most species. One notable difference occurred between medium (20-39 mm) and large (≥ 40 mm) *Lepomis*, with large individuals in higher occurrence in the contiguous wetland as the season progressed.

Discussion

These results focus on describing differences between three permanently connected habitats interacting within a single floodplain lake. Significant differences were found in water quality conditions and fish assemblages between the three habitats examined. Generally, the contiguous wetland was the warmest, least oxygenated of the three habitats, but had elevated fish catch compared to the cooler, better oxygenated pelagic environment. The margin wetland exhibited slower rise in temperature and steep declines in oxygen across the study, yet also showed relatively abundant fish assemblage and greater taxa richness. Results of the current study suggest that two wetland areas in particular, margin and contiguous wetlands, may contribute spatial heterogeneity in water quality yet have similar juvenile fish abundance and assemblage (especially *Lepomis* and *Micropterus*) as determined by highly selective light traps.

Oxygen conditions along the margin wetland may promote use of pelagic and contiguous wetland habitats. It was notable in the current study that DO conditions in the margin wetland deteriorated rapidly over the spawning season. Two factors are potentially linked to falling DO. First, the multiprobes were set at 0.5 m below the surface of the water in water ≤ 1 m deep. This put the probes closer to sediments in the margin wetland than either of the other habitats. Increased sediment respiration may cause lesser mean and minimum DO in the water column of shallow wetlands as the season progressed (Hargrave 1969). The second factor potentially affecting low margin wetland mean DO, was the close proximity to dense aquatic macrophytes. Miranda and Hodges (2000) found that vegetation densities of approximately 60% could foster late summer DO levels of 2 mg/L. The present study observed low (2 mg/L) levels of DO in margin wetlands by mid-June, suggesting impaired oxygen dynamics exist through a substantial portion of the growing season. Despite potential DO impairments, areas of aquatic vegetation are important for spawning and juvenile development, especially for species encountered in this study (see Dibble et al. 1997 for review). Furthermore, the contiguous wetland had generally low levels of DO across the study, beyond effects of warmer temperature and thus lessened solubility. Deepwater swamps, such as Gayden Brake, are typically high in organic matter creating a high oxygen demand due to microbial respiration (Sharitz and Mitsch 1993). Stagnant water and canopy cover, preventing dense understory vegetation or phytoplankton and promoting invertebrate consumers, may further drive low oxygen conditions (Mitsch and Gosselink 2007).

Presence of physical and chemical habitat differences as well as the resulting food resources, may have contributed to patterns observed in the three most abundant taxa. *Labidesthes sicculus,* as caught in the present study, is known to have divergent habitat preferences between juvenile and adult stages, with juveniles preferring pelagic water, possibly due to an aversion to structure (Hubbs 1921). Pelagic environments had the greatest abundance of juvenile *Labidesthes* with least abundance in margin wetlands, supporting these habitat preferences. However, logistic curves were very similar between habitats for adult *Labidesthes*, possibly a result of inshore-offshore diel migration, especially since traps had a soak time of 40 hours (Ross 2001). *Lepomis spp.* exhibited high abundance in each habitat across size classes. Intermediate sizes of *Lepomis* were found in relatively high abundance early in the season, suggesting individuals were from the late summer spawn of the prior year. The overall distributions of *Lepomis spp.* may be attributable to life histories in respect to structure and food resources. Lepomids such as *Lepomis macrochirus* are known to move offshore to begin feeding, shift to vegetated littoral areas, and then potentially shift back to pelagic areas at approximately 65 mm (Werner and Hall 1988). Finally, *Micropterus salmoides*, exclusive of the large single catch of individuals under 20 mm in the margin wetland, show nearly equitable use across habitats for individuals under 40 mm. *Micropterus salmoides* may be the least structure driven, most growth driven of the three taxa collected, and as such most opportunistic in spatial preferences (Davies et al. 1982).

Capture rates (3.5 fish/trap) in this study were less than those reported by Killgore and Miller (1995) in collections made in oxbow lakes (19.7 fish/trap) frequently connected to the Tallahatchie River, Mississippi. Additionally, Killgore and Miller's three most collected taxa, cyprinids, clupeids, and *Pomoxis* spp., were absent or collected in low numbers in light trap collections in Blue Lake. Conversely, catch composition in Blue Lake was similar to that of Aliceville Reservoir along the Tennessee-Tombigbee Waterway, Mississippi, with centrarchids (*Lepomis* spp., *Micropterus* spp., and *Pomoxis* spp.) as well as *Gambusia affinis* and *Labidesthes sicculus* comprising 96% of fish in March-August sampling (Ferrer-Montaño and Dibble 2002). Catch rates are incomparable due to major differences in sampling design between Blue Lake and Aliceville Reservoir studies. Similar taxonomic richness was found in all three systems, with differentiated genera between 10 and 14.

Traditionally, floodplain lake environments have been viewed as a single, overarching community contributing to the floodplain ecosystem. These results suggest that various habitats can contribute differently to the floodplain lake ecosystem, in water condition as well as providing spatial separation among aspects of the fish assemblage. With a long growing season and potential for additional reproductive efforts, the relative contribution of inundated wetlands may change for *Labidesthes*, *Lepomis*, *Micropterus*, and other taxa throughout the year. The primary capture of lentic species with prolonged, gradual spawning strategies, suggests that flood dynamics and other reproductive strategies also have the potential to introduce other taxa in greater abundance, which may give a better insight to community dynamics not detected by the sampling design. Overall, floodplain lakes show dynamic spatial and temporal characteristics, partially imparted by diverse wetlands, which makes them an integral part of the river floodplain ecosystem.

Table 4.1 Fish species collected in Blue Lake

Taxa	**Pelagic environment**	**Margin wetland**	**Contiguous wetland**	**Total**
Lepomis spp. <20 mm	42	12	14	68
Lepomis spp. 20-39 mm	24	50	97	171
Lepomis spp. ≥ 40 mm	2	69	14	85
Micropterus salmoides <20 mm	26	105	15	146
Micropterus salmoides 20-39 mm	32	46	41	119
Micropterus salmoides ≥ 40 mm	0	24	24	48
Labidesthes sicculus ≤ 39 mm	33	12	49	94
Labidesthes sicculus ≥ 40 mm	42	39	52	133
Gambusia affinis	3	8	20	31
Pomoxis spp.	7	4	4	15
Fundulus chrysotus	0	9	2	11
Percidae	1	8	0	9
Notemigonus crysoleucas	0	2	1	3
Opsopoeodus emiliae	0	0	2	2
Catostomidae	0	1	0	1
Centrarchus macropterus	0	1	0	1
Elassoma zonatum	0	1	0	1
Ictaluridae	0	1	0	1
Lepisosteidae	1	0	0	1
Total number individuals	213	392	335	940
Total number species	7	12	8	14

Individual counts for 14 taxa captured in three habitats of Blue Lake with passive larval light traps from 3 March to 29 June, 2012. *Lepomis* and *Pomoxis* species were difficult to differentiate and were combined to genus level only. Families with less than 10 individuals were often not differentiated. *Lepomis*, *Micropterus*, and *Labidesthes* were most abundant individuals across all habitat types and were further differentiated by total length.

Table 4.2 Logistic regression coefficients

	i			*s*		
Taxa	Pelagic	Margin	Contiguous	Pelagic	Margin	Contiguous
Labidesthes sicculus ≤ 39mm	212	177	243	-0.02	-0.05	-0.03
Labidesthes sicculus ≥ 40mm	145	132	140	-0.04	-0.05	-0.04
Lepomis spp. < 20 mm	160	106	186	-0.07	-0.02	-0.03
Lepomis spp. 20-39 mm	128	122	137	-0.03	-0.05	-0.03
Lepomis spp. ≥ 40 mm	53	160	151	0.00	-0.07	-0.05
Micropterus salmoides < 20 mm	16	58	100	-0.01	-0.08	-0.02
Micropterus salmoides 20-39 mm	134	139	147	-0.07	-0.09	-0.08
Micropterus salmoides ≥ 40 mm	*n/a*	165	168	*n/a*	-0.08	-0.11

Parameters *i* and *s* corresponding to those derived from linearized logistic regression coefficients, described in equation 4.1 for *Labidesthes*, *Lepomis*, and *Micropterus* collected 3 March to 29 June, 2012 across pelagic environment, margin, and contiguous wetlands, Blue Lake, Mississippi. The logistic model assumes symmetry; when cumulative catch was high in early weeks, the inflection point *i* was predicted to have occurred before sampling started; conversely, when most of cumulative catch occurred in later weeks, the inflection point was predicted to occur after sampling was terminated. No collections are denoted by *n/a*.

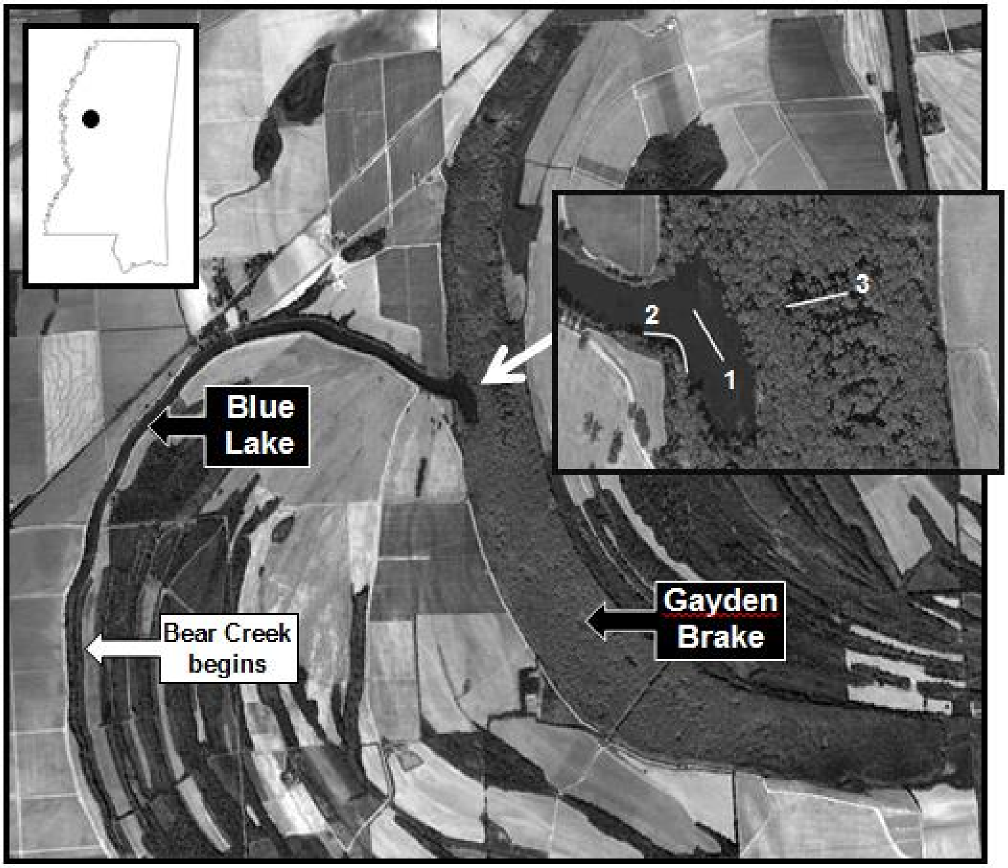

Figure 4.1 Map of Blue Lake and Gayden Brake

The northern portion of the watershed where Blue Lake is located is in Leflore County, Mississippi, shown in the top left inset. The inset on the right shows approximate location of passive larval light traps fished in transects within pelagic environment (1) as well as margin (2) and contiguous wetlands (3). Each transect, sampled 3 March to 29 June, 2012 for juvenile fish, was approximately 100 m in length.

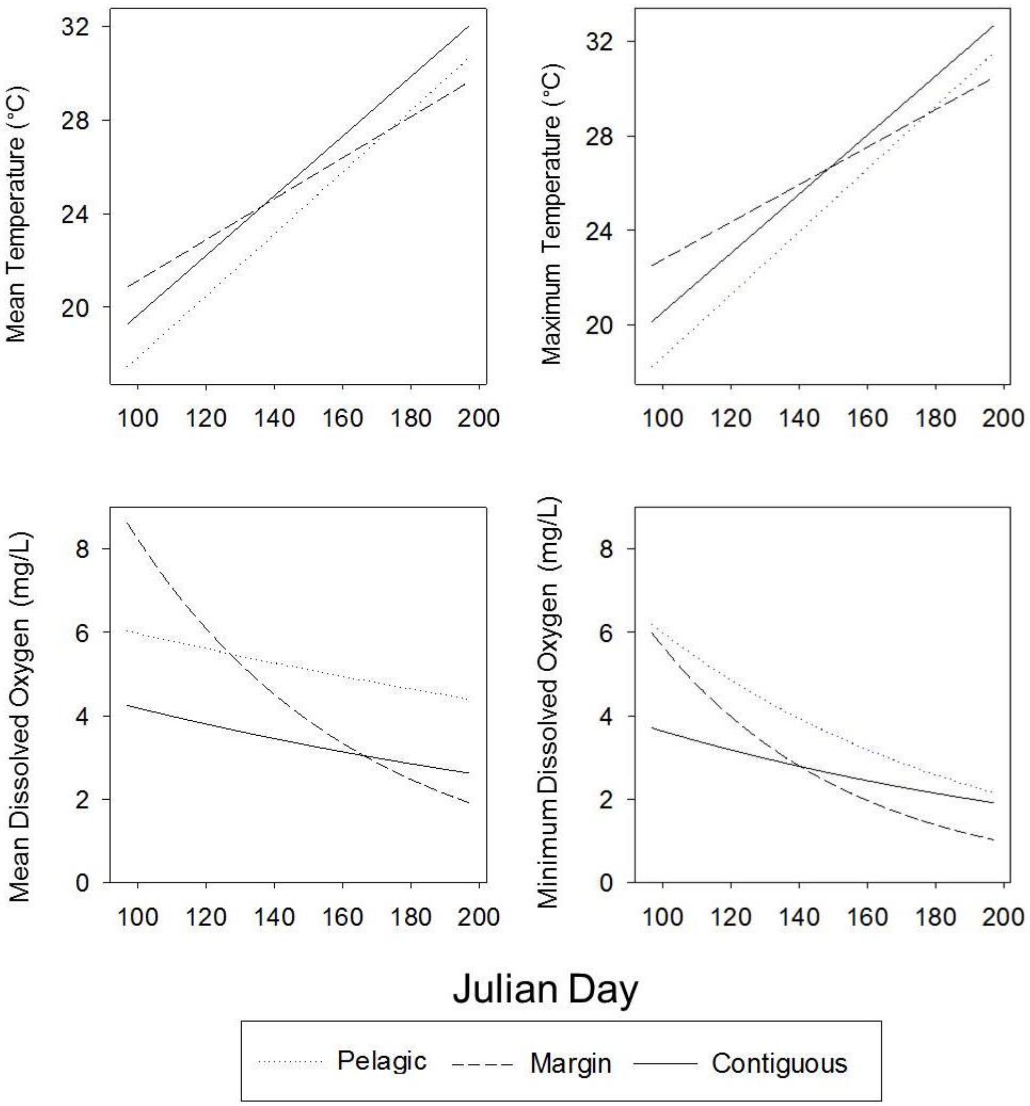

Figure 4.2 Diel water quality parameters

Diel water quality parameters in three habitats of Blue Lake, Mississippi extrapolated across 6 April to 22 July, 2012 (Julian day 96 to 203).

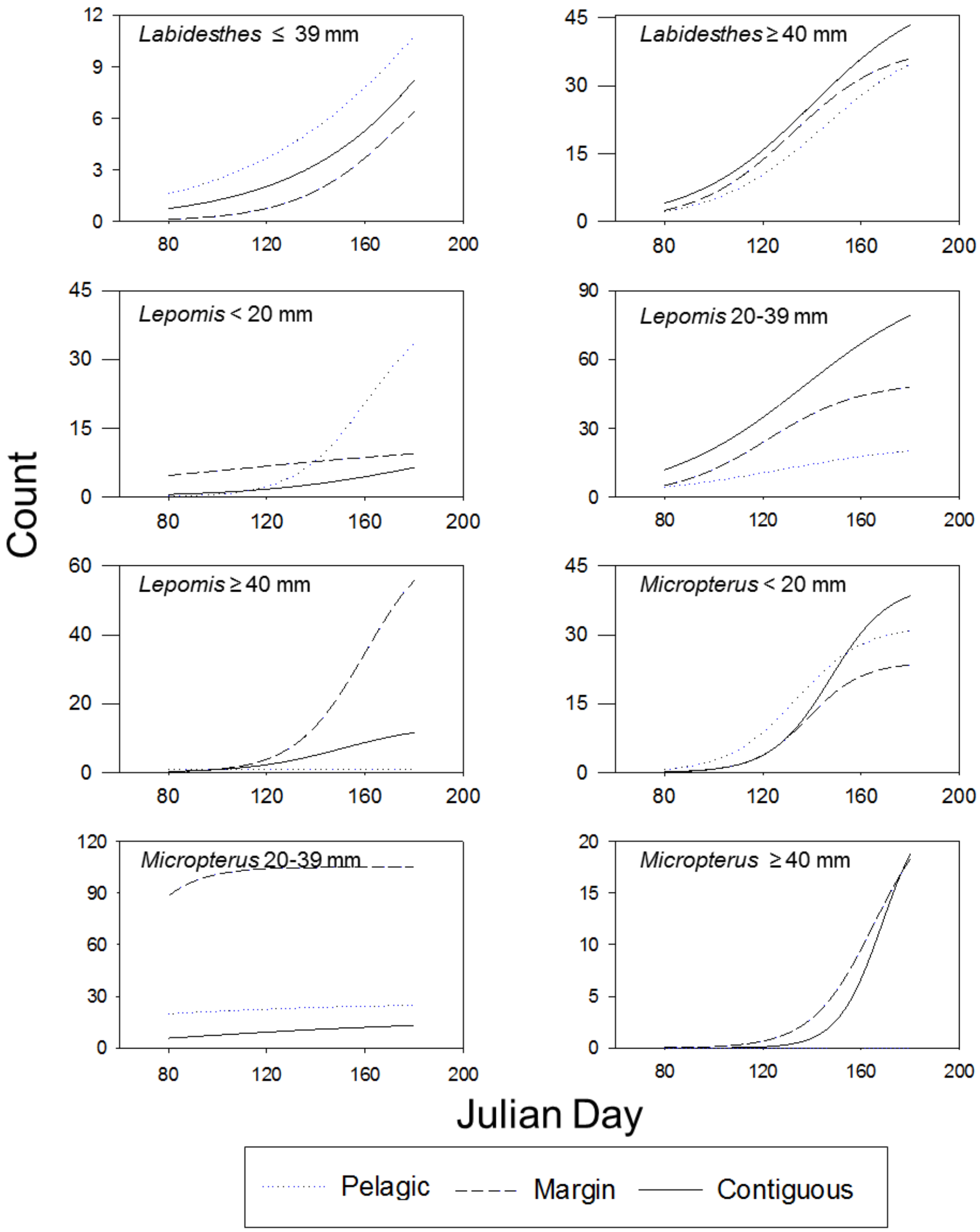

Figure 4.3 Logistic regression representations

Logistic representations of cumulative catch for *Labidesthes*, *Lepomis*, and *Micropterus*. Catches represent 3 March to 29 June, 2012 across pelagic environment, margin, and contiguous wetlands in Blue Lake, Mississippi. The y axes were derived by expressing cumulative counts from cumulative proportions predicted by equation 4.1.

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CHAPTER V

SYNTHESIS

The previous chapters identify relationships between fish communities and water quality among floodplain lakes in the Mississippi Alluvial Valley (MAV). The relationships provided are intended to facilitate efficient ways to monitor and manage floodplain lakes in the MAV. Use of *in-situ* field measurements of necessary primary limnological characteristics minimizes effort needed to monitor these systems. Identifying lake systems on a watershed level or where appropriate, lake level, that have the most potential for management and improvement will yield greater returns on investments allocated to lake restoration and management in the region. Properly managed lakes have the potential to sustain high species diversity and provide economic benefits in the MAV.

Although phytoplankton and turbidity play key roles in enhancing or hindering the base of food webs in aquatic ecosystems, excess levels of either can result in reduced water clarity, often viewed by uninformed observers as undesirable features of surface water. Obtaining reliable estimates of these variables for monitoring purposes to achieve biological and aesthetic goals is becoming required by law in some states. Mississippi is currently undergoing nutrient criteria development in accordance with these needs, and chlorophyll-*a* concentrations are often at the center of such criteria. As surface waters become increasingly imperiled, determining these criteria, implementing assessment procedures, and undertaking restoration efforts will be difficult, if not impossible, without efficient monitoring tools. I show that these needs can be partially met with handheld meters, for rapid *in-situ* estimation. While set regulations are important for cataloging potential impairments, metrics such as chlorophyll-*a* and turbidity provide only snapshots of biologic conditions. Chlorophyll-*a* and turbidity exhibit large intra-annual variability and in the modern era are exacerbated by external inputs related to anthropogenic watershed-level disturbances. Therefore, accounting for long term environmental conditions should be paramount before making management decisions.

Fish communities provide insight into environmental conditions over longer periods of time. Thus, I explored community analysis of fish species within Bear Creek watershed to determine importance of hydrologic connectivity and found a strong dichotomy between permanently connected lakes and partially connected lakes, as well as a gradient in the fish assemblages occupying the Bear Creek chain of lakes. Accurately quantifying hydrologic connectivity in the MAV is difficult, but permanently connected lakes as seen in Bear Creek watershed provides an initial framework. Importance of hydrologic connectivity should be considered before implementing management options that increase or decrease this connectivity. Yet, for watersheds already exhibiting these differences, careful management can diversify fishing opportunities within a spatially small region to include both rheophilic and lacustrine options.

Given the ease in delineating the spatially divergent communities into simple classifications, management options implemented from results of these studies should be feasible. First, systems can be connected to promote rheophilic species, or isolated to promote lacustrine communities. Lacustrine communities generally support recreational species, especially centrarchids, which will likely increase regional lake use. These systems can be manipulated to support juvenile fish communities with enhanced margin wetlands and connectivity to contiguous wetlands.

Subsetting lakes using a hierarchical decision process based on physical differences also promotes management efficiency. Once impairment criteria are established, handheld meters can facilitate ranking watersheds in most need of restoration. Within a watershed, hydrologic connectivity can be used to determine which lakes would benefit most from management action. Finally, those lakes can be managed by utilizing adjacent wetlands to promote 1) nutrient uptake, thereby mitigating eutrophication, 2) sediment capture, to slow lake fill-in and turbidity issues, and 3) fish communities, for recreational and economical gain as well as conserving regional fish diversity.