Connectivity and Variability: Metrics for Riverine Floodplain Backwater Rehabilitation

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The importance of floodplain aquatic habitats that are seasonally or periodically connected to the main channel (backwaters) within lowland riverine ecosystems is well established. However, backwaters are becoming rare as development is transforming floodplain landscapes. Therefore, rehabilitation, protection, and management of riverine backwaters are becoming increasingly common, with annual expenditures in the millions of dollars. Even with the increasing number of projects, general criteria for selecting restoration goals and evaluating project outcomes are lacking. To address this need, Kondolf et al. (2006) proposed an approach for evaluating river restorations that is based on assigning a position to the system in a four-dimensional space that represents hydrologic temporal variability on one axis and connectivity in the three spatial dimensions on the remaining three axes. Use of the Kondolf approach for evaluating restoration of a backwater adjacent to a medium-sized river in northern Mississippi is presented as a case study, in which nearby degraded and less impacted backwaters were used as references. The restoration project resulted in a reduction in main-channel connectivity and lower levels of variability for the treated backwater. Additional responses to treatment included increased summer water depth, moderation of severe diurnal water quality fluctuations, and reductions in concentrations of solids, nutrients, and chlorophyll *a*. Fish species richness, numbers, and biomass were unchanged following rehabilitation, but trophic structure shifted away from omnivorous species and toward predators. Ecological services provided by floodplain riverine backwaters may be enhanced by modest management measures, but regaining and maintaining connectivity with adjacent ecological functional patches remains difficult.

1. INTRODUCTION

Freshwater ecosystems in the United States are exceptionally diverse, even compared with the tropics [*Master et al.*, 1998]. In particular, streams in the southeastern United States ("Southeast") are important ecological resources, but resident aquatic fauna are experiencing accelerated extinction rates [*Ricciardi and Rasmussen*, 1999; *Warren et al.*, 2000; *Karr et al.*, 2000]. Apparently, faunal declines reflect disruption of important connections between main channel and slack water habitats such as wetlands, abandoned channels, sloughs, severed meander bendways, and borrow pits [*Buijse et al.*, 2002; *Ward et al.*, 2001; *Wiens*, 2002; *Jackson*, 2003; *Kondolf et al.*, 2006], or in more current terminology, disruptions of connections between hydrogeomorphic patches [*Thorp et al.*, 2006]. The timing, frequency, and duration of hydrologic connections between rivers and their backwaters have important ecological implications. Many plant and animal species native

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to river corridors have life cycles that require access to backwater habitats during certain seasons or while certain climatic conditions exist. For example, reproduction may jointly depend on certain ranges of water temperature, photoperiod, and flooding. Furthermore, disrupted connections usually lead to drying and gradual terrestrialization of backwaters [*Gore and Shields*, 1995; *Schramm and Spencer*, 2006].

The Lower Mississippi River alluvial plain ("the Delta") is a case in point. This region contains numerous floodplain lakes that experience varying levels of hydrologic connectivity during periods of high stage in adjacent streams and rivers. Many of these water bodies receive significant inflows of water and associated pollutants from cultivated lands and have experienced precipitous declines in water quality and fisheries in recent decades. Recent studies of Delta lake fisheries indicate that lake area, lake elongation, and lake water clarity are key abiotic variables that control fish community structure, with small, shallow, elongated lakes most seriously degraded [*Miranda and Lucas*, 2004]. Backwater ecosystems often suffer from problems associated with hydrologic perturbation due to levees, dams, main channel incision, and backwater sedimentation [*Bellrose et al.*, 1983; *Bhowmik and Demissie*, 1989; *Hesse and Sheets*, 1993; *Jackson*, 2003]. Additional issues include water quality degradation, aquatic plant infestation and die-off, and extreme variation in water temperature and habitat volume [*Claflin and Fischer*, 1995; *Light et al.*, 2006; *Justus*, 2009]. One of the most pernicious problems may be described as vertical disruption of lateral connectivity. Hydrologic connections between the river and backwaters become shorter and less frequent when river stages are lowered through channel incision or when controlling elevations for floodplain water bodies are raised by sediment deposition [*Light et al.*, 2006].

Ecological restoration may be thought of as an attempt to return an ecosystem to its historic (predegradation) trajectory [*Society for Ecological Restoration International Science and Policy Working Group*, 2004] (accessed 23 November 2009). Restoration workers attempt to establish this "trajectory" through a combination of information about the system's previous state, studies on comparable intact ecosystems, information about regional environmental conditions, and analysis of other ecological, cultural, and historical reference information [*Society for Ecological Restoration International Science and Policy Working Group*, 2004]. In lightly altered natural systems, backwaters tend to follow a trajectory similar to classical lake eutrophication: due to sedimentation and perhaps migration of the river main stem, these areas become shallower, and connections to the river become briefer and less frequent. However, the formation of new backwaters due to main channel avulsion and more gradual processes continues as old backwaters become wetlands and eventually terrestrial systems. In altered floodplains, however, backwater formation processes are hindered or absent. Flood control and channel stabilization prevent formation of new backwaters as existing backwaters age, becoming shallower, more turbid and often experiencing lower dissolved oxygen (DO) concentrations [*Miranda*, 2005]. Extremely shallow backwaters tend to experience lower DO than deeper ones due to respiration occurring throughout the water column [*Miranda et al.*, 2001]. These systems also tend to have lower water transparency due to benthivorous fish and phytoplankton [*Roozen et al.*, 2003; *Miranda and Lucas*, 2004; *Lin and Caramaschi*, 2005]. Fish communities in such systems exhibit strong linkages to abiotic factors and are dominated by tolerant omnivores with few predators [*Miranda and Lucas*, 2004].

Since a hallmark of river corridor development is reduction of lateral linkages, many river restoration projects have focused on managing floodplain water bodies and their connectivity with the main channel (e.g., *Holubova et al.* [2005], but see *Pegg et al.* [2006]). Based on a study of 29 floodplain lakes in the region containing our sites, *Miranda and Lucas* [2004] recommended rehabilitation efforts focus on watershed management, dredging or water level control, and fishery management. Existing backwater rehabilitation projects feature practices such as pumping in water, breaching levees, reopening relatively small connecting channels, or by constructing water control structures to increase water depth during dry periods [*Shields and Abt*, 1989; *Theiling*, 1995; *Galat et al.*, 1998; *Amoros*, 2001; *Buijse et al.*, 2002; *Valdez and Wick*, 1981; *Grift et al.*, 2001; *Shields et al.*, 2005; *Schultz et al.*, 2007; *Julien et al.*, 2008]. Substantial sums have been spent in these efforts, but little information is available regarding the performance of existing projects to guide future design efforts [*O'Donnell and Galat*, 2007; *Palmer et al.*, 2007]. At least three approaches (or combinations of these) for generating criteria are possible. First, backwater treatments may be designed, maintained, and operated to meet habitat requirements for a selected species or group of species [*Galat et al.*, 1998]. Second, criteria may be set to produce selected characteristics of a reference site. Third, using an approach described by *Kondolf et al.* [2006], backwater physical conditions may be assessed in terms of hydrologic variation and main channel connectivity, as described below. The objective of this paper is to show how the ecological performance of a backwater rehabilitation project may be assessed using the Kondolf approach.

2. KONDOLF DIAGRAM

Kondolf et al. [2006] proposed use of hydrologic connectivity and variability (also referred to as flow dynamics) as key descriptors of riverine ecosystem status. Hydrologic connectivity was defined as water-mediated fluxes of material, energy, and organisms among the major ecosystem components: main channel, floodplain, aquifer, etc. [*Amoros and Bornette*, 2002]. Connectivity occurs in all three spatial dimensions: longitudinal (upstream and downstream), lateral (main channel and floodplain), and vertical (surface water and the hyporheic or deeper subsurface regions). Variability was primarily defined as temporal variation in discharge, but it also encompasses parameters such as temperature, sediment, and trophic levels [*Hughes et al.*, 2005]. Connectivity and variability tend to be related. For example, construction of a dam to regulate flow often reduces the frequency and duration of floods downstream, reducing lateral connectivity and flow variation. Furthermore, the dam may reduce longitudinal connectivity by presenting a barrier to movements of sediment and organisms. Connectivity tends to be reduced by human activities (e.g., construction of dams, levees, channelization, flood reduction, and blockage of side channels) or by geomorphic change produced by human activities (e.g., channel incision). The status of a given riverine ecosystem may be mapped by plotting a point representing the system within a Cartesian plane with the horizontal axis representing variability and the vertical axis representing connectivity in a selected dimension (Figure 1). Multidimensional plots may be used if connectivity is mapped in more than one dimension. If information is available, points may be plotted representing predegradation and current conditions, giving a degradation trajectory. Ideally, restoration would simply follow the reverse path of the degradation vector, returning the system to its predegradation connectivity and flow variability. If predegradation data are not available, reference conditions may be inferred from lightly degraded sites.

To illustrate this concept, *Kondolf et al.* [2006] plotted degradation trajectories for 23 rivers using at least one dimension to measure connectivity. Month-to-month flow variation, with special emphasis on the probability of intermittent flow, was used to indicate streamflow variability, with sites arrayed along a continuum ranging from spring fed to snowmelt to rain fed to intermittent or ephemeral regimes. Restoration trajectories were plotted for systems that were sites for restoration projects. In general, the bivariate plots showed that systems tended to follow paths that resulted in reduced connectivity and variability as they degraded, although some sites (e.g., base flow diversions and channelization) became more variable as they were degraded. Rehabilitation or restoration often increased connectivity but rarely increased variability. Preparing a “Kondolf diagram” for a system selected for restoration requires completion of four key tasks: assessment of historical conditions, definition of degradation in process-based terms, identification of factors triggering degradation, and setting goal trajectories for

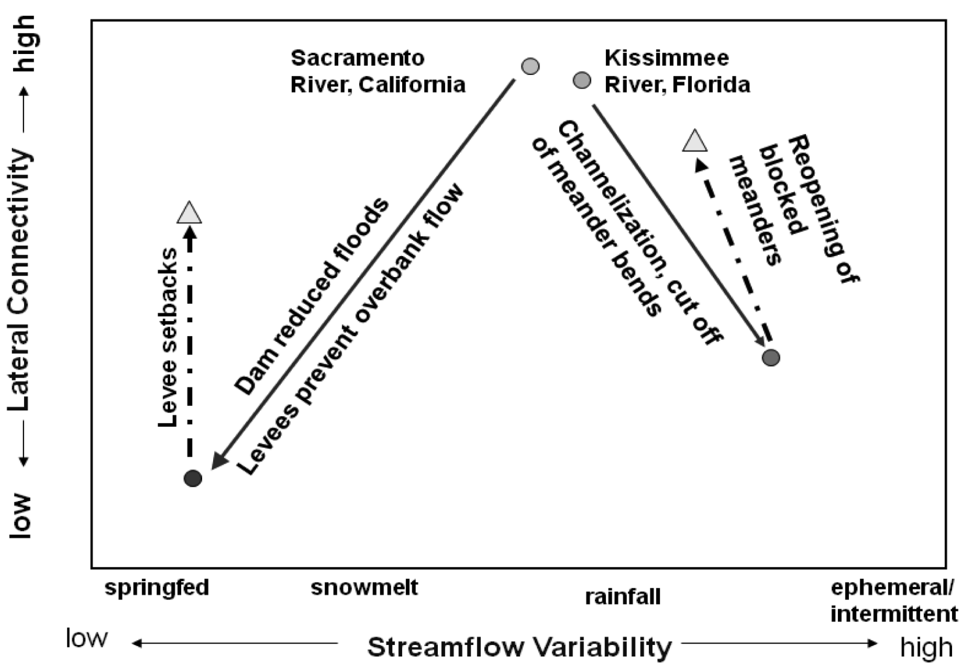

Figure 1. Diagram for assessment of aquatic ecosystem status. Solid arrows represent ecological degradation, and dashed arrows represent restoration trajectories plotted on axes of lateral connectivity and flow dynamics. After the work of *Kondolf et al.* [2006].

selected processes. Herein, we adopt this approach, not for river reaches as originally proposed, but for individual floodplain water bodies or backwaters. Clearly, the overarching goal of backwater rehabilitation is to contribute positively to the entire river ecosystem, but the open nature of the river system and the mobility of its fauna make measurement of the effects of restoring one or a few backwaters on the entire river ecosystem impossible. This project seeks to gauge impacts of rehabilitation of a single backwater body on its connectivity and variability and to relate these outcomes to the backwater's ecological functions as manifest in water quality and fish populations.

3. STUDY SITES

A reach of the Coldwater River about 20 km downstream from Arkabutla Dam in northwestern Mississippi was selected for study due to the presence of more than 20 severed meander

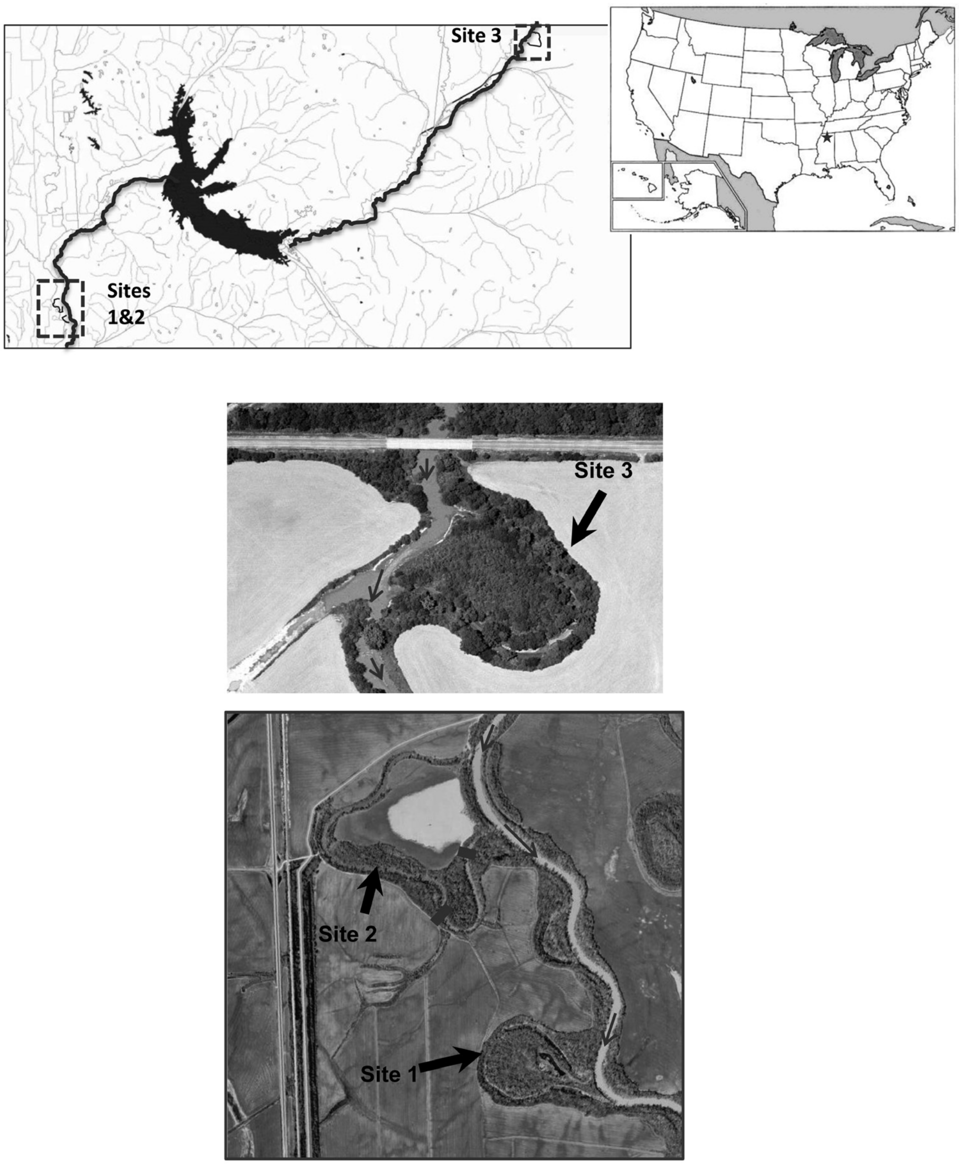

Figure 2. Study locations upstream and downstream from Arkabutla Reservoir. Air photos show site 3 in 2000 and study sites 1 and 2 in 1957. Sites 1 and 2 (34°40.024′N, 90°13.373′W) were cut off from Coldwater River in 1941–1942, and site 3 (34°51.572′N, 89°48.375′W) was cut off prior to 1991. Site 2 was treated by addition of weirs (gray rectangles) in 2006.

bends and other floodplain water bodies along the river. Elevated suspended sediment concentrations, habitat reduction associated with sedimentation and water pollution associated with agriculture are primary resource problems in this locale [*Mississippi Department of Environmental Quality*, 2003, U.S. Corps of Engineers, Coldwater River Basin below Arkabutla Lake, Mississippi, Section 905(b) reconnaissance report, undated, Vicksburg, Mississippi]. In addition, flows are highly regulated by the upstream impoundment, which is operated for flood control and recreation. Despite these problems, 13 to 22 species of fish were captured annually between 1990 and 1994 from this stretch of the Coldwater. Catch per unit effort (hoop nets) for the Coldwater River (all species and seasons) exceeded the four other Yazoo basin rivers sampled during the same time period [*Jackson et al.*, 1995].

Three Coldwater River floodplain backwaters were selected for study (Figure 2). Two were severed meander bends along the aforementioned reach below Arkabutla Dam, while the third was upstream from the reservoir. Sites were designated 1, 2, and 3 from downstream to upstream and were used as degraded reference, rehabilitation site, and least-impacted reference, respectively. Sites 1 and 2 were severed meander bends created by man-made cutoffs constructed in 1941–1942 [*Whitten and Patrick*, 1981]. Both were 1.5 to 2 km long and 40 m wide and were inside the main stem flood control levee. Lands outside the old bends were in row-crop cultivation, while lands inside the bends were in forest (site 1) or fallow (site 2). Buffers of natural vegetation 5–100 m wide were on both banks of the old channels. Both backwaters received runoff from cultivated fields. Backwater levels were tightly coupled with Coldwater River stage when the river stage exceeded the controlling elevation in the downstream connecting channel, but during the warmer months, the river was 1 to 3 m lower than the backwaters, and the backwaters became quite shallow. Site 1 was almost completely choked with aquatic plants during warmer months. Probing bed sediments at both sites with metal rods and sampling site 2 with a Vibracore apparatus revealed 2–2.5 m of fine-sediment deposition, with mean annual rates of about 3.1 cm yr^{-1} based on vertical profiles of sediment density and Cs-137 activity [*Shields et al.*, 2010]. Previously reported sediment sample chemical analyses and invertebrate bioassays indicated sediment metal concentrations were likely not high enough to create toxic impacts, but several insecticides were detected and impacted bioassays [*Knight et al.*, 2009a, 2009b].

A third severed bendway (site 3) located on the same river, but upstream from Arkabutla Lake and outside of the zone of reservoir influence, was used as a less impacted reference. There were no significant local inflows, and runoff from adjacent fields was diverted away from the bend by a low levee. The backwater channel was about 0.35 km long and 20 m wide and was subjected to more frequent connection with the river, with fully developed lotic conditions (velocities ~0.3 m s^{-1}) occurring during high river stage. This type of long-duration, pulsed connectivity is typical of the regime that persisted at the degraded site downstream of the reservoir prior to reservoir and levee construction, and fish species in this system are adapted to such conditions [*Jackson*, 2003]. However, although the stage hydrograph at site 3 was less perturbed relative to sites 1 and 2, investigations after this study began revealed that site 3 habitat was impacted by deposition of sandy sediments contaminated with the organochlorine insecticide heptachlor [*Knight et al.*, 2009b]. Probing bed sediments with metal rods revealed 1–2 m of deposition. The backwater was simply a series of small, isolated pools during periods of low river stage. Therefore, site 3 provided a hydrologic reference, but not a suitable reference for less impacted backwater water quality and ecology.

4. REHABILITATION

For rehabilitation, site 2 was modified by constructing two low weirs across the old channel. Weirs divided the backwater into two compartments: a lake cell and a wetland cell. The southern (upstream) weir was located so as to divert runoff from agricultural fields away from the lake cell. The remainder of this paper focuses on our effort to restore the lake cell as a riverine backwater and gauge progress toward that goal using the Kondolf axes representing hydrologic variability and connectivity. The wetland cell was managed using the downstream weir in order to reduce loadings of sediment, nutrients, and pesticides to the river, and results of that work have been reported elsewhere [*Lizotte et al.*, 2009; *Shields and Pearce*, 2010]. Weirs consisted of low (<2 m high) earthen embankments placed at right angles to the old river channel and covered with stone riprap. Each weir included a

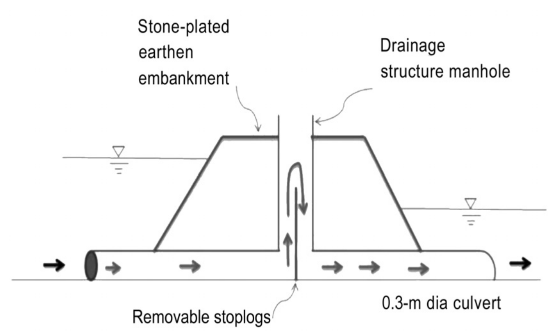

Figure 3. Schematic of weir structure.

water control structure that consisted of a 0.3 m diameter pipe that penetrated the embankment bisected by a flashboard riser "manhole" (Figure 3). Weir water control structures were operated to retain water during March–November and were opened to allow more frequent connection to the Coldwater River during December, January and February.

5. METHODS

Once daily, Coldwater River stage data for a 44 year period of record (1960–2004) were transferred from nearby gauges to the backwater sites using regression formulas between the gauge data and measurements made at the backwater mouths during this study. These data were analyzed to determine the average annual duration (or probability) of connection between the backwaters and the main channel given the site geometry found at the outset of this study, assuming stationary hydrologic conditions. In addition, backwater stage and temperature were logged at all three sites at 30 min intervals for about 18 months before and 36 months after rehabilitation of site 2 (2004–2009). Additional water quality constituents were determined for site 1 (degraded reference) during the first and final years of this period, while water quality in site 2 (rehabilitated) was sampled throughout the study period. Specifically, pH, dissolved oxygen, turbidity, and specific conductance were logged at 4 h intervals and were measured weekly using handheld meters. Grab samples were collected weekly and analyzed for solids, nutrients, and chlorophyll. Water quality loggers were placed near the apex of each of the old bends, and grab samples were collected at the same sites. Loggers were deployed so that their sensors were 0.2 to 0.6 m below the water surface; warm season vertical stratification in these waters was weak due to the shallow depths. At site 1, sensors were sometimes more deeply submerged during floods, but these occurred only during colder periods when there was no vertical stratification. At sites 1 and 2, fish were collected using a boat-mounted electroshocker at least semiannually in spring and fall. Fish were collected from each site during four, 20 min sampling periods using pulsed DC current. Because conductivities varied at the collection sites both temporally and spatially, voltages were adjusted to provide the maximum catch possible for the given conditions. All major habitats were sampled including shorelines, debris piles, and open water. At site 3, fish were sampled on two dates, 1 year apart using a backpack electroshocker due to the extremely shallow depths. Each collection consisted of one or two 20 min sampling runs depending upon the amount of surface water present such that all major habitat patches were sampled. All fish collections were processed in the same fashion. Fish were identified to species, enumerated, and measured for length, which was used to calculate weight. Weights and numbers of fish were used to calculate catch by numbers, catch by weight, catch per unit of effort, and numbers per unit of effort for each sample.

The backwater stages measured during 2004–2009 were used to compute mean depth at each measurement interval using digital elevation models based on lidar coverage of terrestrial zones and bathymetric data collected using boat-mounted echo sounders coupled with differentially corrected GPS. Mean daily values of backwater stage and mean depth were further examined using the suite of indices of hydrologic alteration proposed by *Richter et al.* [1998]. Hydrologic data from all three study sites were used to construct a two-dimensional (2-D) Kondolf diagram featuring lateral connectivity. Since water quality data were not normally distributed, nonparametric analysis of variance (ANOVA) (Kruskal-Wallis one-way ANOVA on ranks with Dunn's method for multiple comparisons) was used to compare distributions before and after rehabilitation [*Glantz*, 1992; *Systat Software, Inc.*, 2009]. Effects of rehabilitation on fish community structure similarity was examined by computing Bray-Curtis coefficients using lists of the numerical abundances of the 11 most abundant fish species from each of the backwaters [*Pegg et al.*, 2006]. For this analysis, collections from site 2 before and after rehabilitation were listed separately. Bray-Curtis values are lower for higher levels of similarity, with identical collections having values equal to 0 and collections with no species in common having coefficients equal to 1 [*Bray and Curtis*, 1957]. All abundances were fourth-root transformed prior to computation of Bray-Curtis coefficients to meet assumptions of multivariate normality and to moderate the influence of species abundance extremes. Pearson product-moment correlation coefficients were computed between key descriptors of fish collections and physical (hydrologic) variables [*Systat Software, Inc.*, 2009]. For correlation analyses, all fish samples from a given site on a given date were pooled to compute collection characteristics (number of fish, mean size of fish, percent of catch biomass composed of piscivores, etc.). Correlation coefficients were computed between these values and the mean water depth computed for that date.

6. RESULTS

Comparison of 1960–2004 once daily river stages with prerehabilitation (circa 2005) geometry indicated that sites 1 and 2 experienced backwater connection with the river channel 15% (site 1) and 12% (site 2) of the time, while the less impacted reference (site 3) was connected an average of 22% of the time (assuming static geometry circa 2005). Sites

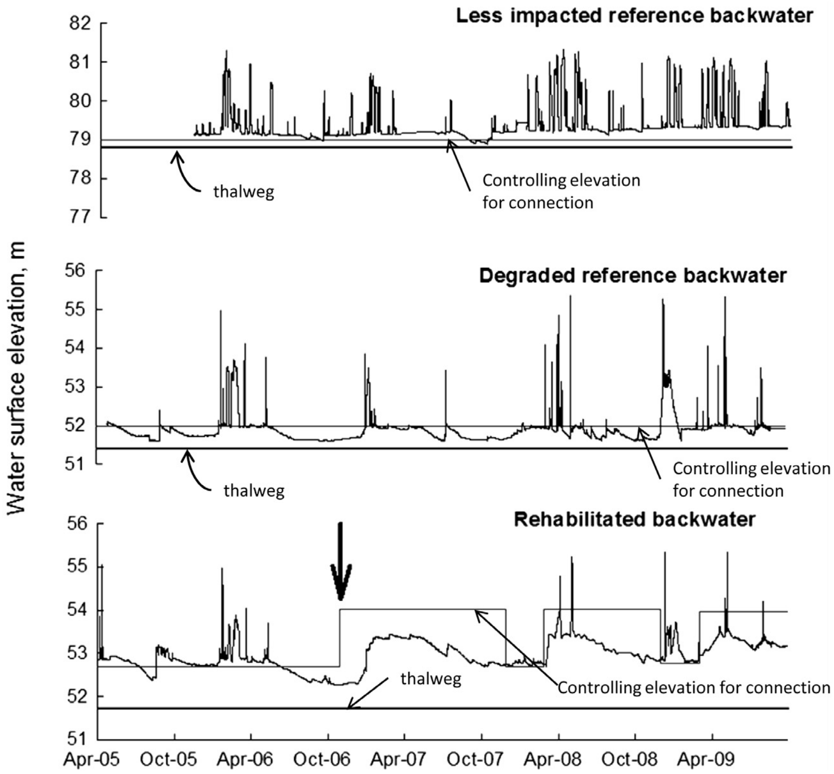

Figure 4. Stage hydrographs for less impacted reference backwaters, degraded reference backwater, and rehabilitated backwater. Vertical black arrow indicates date for completion of weir construction.

1 and 2 were connected with the river at both their upstream and downstream ends, allowing lotic conditions to develop, 2% and 3% of the time, respectively, while site 3 enjoyed such connection 10% of the time.

The actual effects of rehabilitation on backwater hydrology were measured using data we collected before and after construction of the weir in site 2 (Figure 4). Prior to rehabilitation, water depths in both degraded backwaters were extremely shallow, with monthly mean water depths generally <0.65 m. Periods with deeper water, which were driven by high river stages, were brief and limited to winter and spring. The rehabilitation weir increased dry season (summer-fall)

Table 1. Hydrologic Conditions in Study Backwaters Before and After Rehabilitation[a]

	Mean (STD) Water Depth (m)	River Connection (% of time)	Median Rise Time (m d^{-1})	Median Fall Time (m d^{-1})
Degraded reference (site 1)				
Before rehabilitation	0.53 (0.22)	5.7	0.046	0.073
After rehabilitation	0.54 (0.20)	5.5	0.193	0.087
Rehabilitated backwater (site 2)				
Before rehabilitation	0.59 (0.15)	4.4	0.028	0.073
After rehabilitation	0.69 (0.16)	2.2	0.013	0.008
Less impacted reference (site 3)				
Before rehabilitation[b]	NA	24.5	0.172	0.028
After rehabilitation	NA	55.6	0.190	0.018

[a]Median rise and fall times were computed using software package Indices of Hydrologic Alteration [*Richter et al.*, 1998]. NA indicates not applicable.

[b]These values are based on shorter period of record (only one water year, 2006) than the site 1 and site 2 “before rehabilitation” entries, which were based on two water years (2005–2006).

water depths there by 0.15 to 0.30 m, while conditions in site 1 remained unchanged (Table 1). Dry season extreme lows were greatly moderated by the presence of the weirs in site 2 (Table 2).

In general, the weir moderated stage fluctuations and made hydrologic conditions less variable (Figure 4). The high stage rise rate (median of all positive differences between consecutive mean water depths that exceeded base stage elevation) decreased 50%, while the fall rate decreased by an order of magnitude (Table 1). Rise and fall rates for the treated site 2 were similar to those for the degraded reference backwater (site 1) before rehabilitation, but an order of magnitude smaller afterward. Weir placement made stage variability at site 2 more similar to the less impacted reference site 3 during the postrehabilitation period.

Weir placement reduced connectivity between the backwater and river channel (Table 1). The degraded backwater, site 1, was hydraulically connected to the river about 6% of the time during the period of observation. The rehabilitated backwater, site 2, was connected about 4% of the time prior to weir construction, but only 2% following weir placement. These values are far lower than those observed at the less impacted site 3, which was connected to the river about one fourth of the time during the water year immediately prior to rehabilitation of site 2 and more than half the time during the three water years following rehabilitation. The ecological importance of the connection of lateral habitats to main channels is a function of timing as well as the duration of such connections. Native organisms are adapted to a hydrograph dominated by the Lower Mississippi River, which features regular high stages during the December–May timeframe, with highest stages in April [*Baker et al.*, 1991]. Prior to construction of flood control levees and dams, this regime likely produced flooding of low-lying areas across the alluvial plain containing our sites. Seasonality of connection frequency for our sites followed these trends before and after rehabilitation, although the fraction of time connection occurred tended to be low for the degraded and rehabilitated sites relative to site 3 (Figure 5).

Table 2. Medians of Annual Extreme Mean Depths (m)[a]

	30 Day Minimum	90 Day Minimum
Degraded reference backwater (site 1)	0.37	0.39
Site 2 before rehabilitation	0.39	0.45
Site 2 after rehabilitation	0.54	0.55

[a]Values computed using software package Indices of Hydrologic Alteration [*Richter et al.*, 1998].

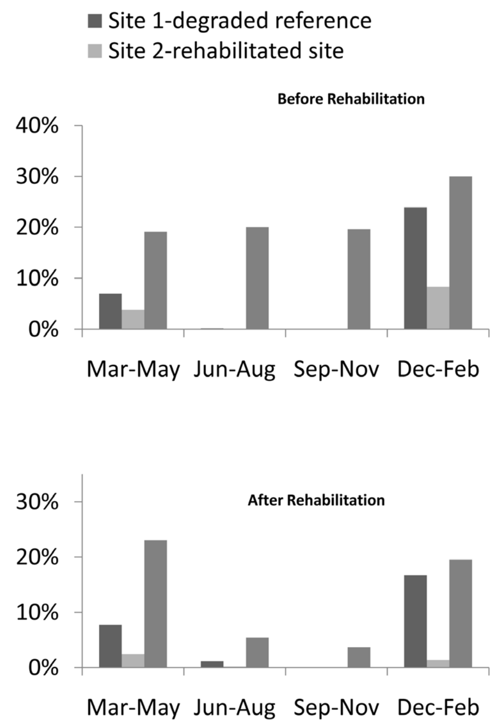

Figure 5. Percent of time backwaters were connected to main channel by quarter before (2005–2006) and after (2007–2009) rehabilitation.

In order to gauge the effects of rehabilitation, a 2-D Kondolf diagram was constructed using the less impacted reference site 3 as a predegradation condition, site 1 as an indicator of degraded status, and the postrehabilitation conditions at site 2 (Figure 6). The fraction of time that the backwaters were hydraulically connected to the river channel was used as a measure of connectivity [*Heiler et al.*, 1995], while the median rate of stage change during the falling limbs of high stage events was adopted as a measure of variability. Sites 1 and 2, both degraded backwaters, plotted very close to each other prior to rehabilitation. Since the weirs reduced connectivity and moderated the flashy stage hydrographs, rehabilitation made the treated site 2 plot closer to the less impacted reference (site 3) on the variability (x) axis, but translated it farther away from the target condition on the connectivity (y) axis.

Prior to weir placement, water quality conditions in the two degraded backwaters were similar, except dissolved oxygen and chlorophyll *a* were lower, and total N was greater, in the degraded reference, site 1 (Table 3). These differences were likely due to the heavy mat of floating duckweed (*Lemna* sp.) that covered the water surface in the degraded reference site during all but the coldest months.

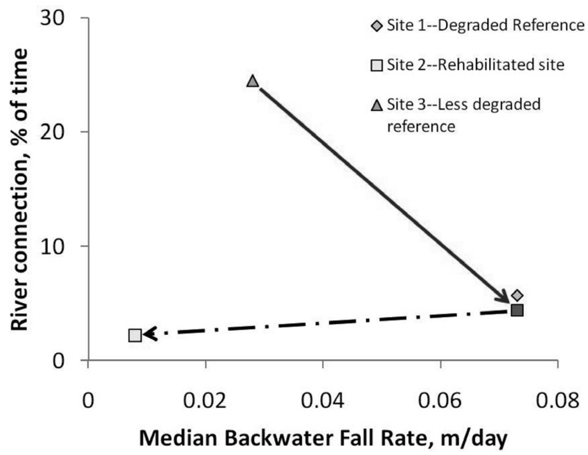

Figure 6. Kondolf diagram for study backwater. Since there were no predegradation data for the rehabilitation site 2, the upstream site 3 was used as a less impacted reference. Rehabilitation translated the status of the treated site away from the degraded reference and toward the less impacted reference on the variability axis (median fall rate for high stage events) but had the opposite effect on connectivity (y axis).

Weir placement transformed site 2 water quality, making it less similar to the degraded site 1. Diversion of agricultural runoff away from the lake cell in site 2 (Figure 2) resulted in reductions in turbidity and suspended solids of about 70%, while nutrient levels were 30% to 60% lower. Accordingly, chlorophyll *a* values were about half as great after weir placement. Summer diurnal fluctuations in temperature and dissolved oxygen were moderated by greater depths produced by the weir, but maximum temperatures (Figure 7) and minimum dissolved oxygen levels were not (Figure 8). The less impacted site continued to experience maximum summer temperatures that were about 5°C cooler than the other two sites following rehabilitation.

The rehabilitated backwater was sampled for fish on 11 occasions over the course of the study, with a total effort of 818 min of electrofishing, producing 2523 fish representing 32 species with a total mass of 259 kg. The degraded reference site yielded 402 fish representing 19 species with a total mass of 20 kg when sampled on two dates with a total effort of 65 min. The less impacted reference site was sampled twice, but yielded only eight individuals of two species total, likely due to the aforementioned insecticide contamination [*Knight et al.*, 2009b]. Two species, *Ictiobus bubalus* and *Lepisosteus oculatus* composed 53% of the biomass from the rehabilitated site 2 and 66% of the biomass from the degraded reference site 1. Fish populations in both backwaters appeared relatively insensitive to antecedent connection to the river but were influenced by mean water depth (Table 4). Greater depths in the treated backwater were associated with larger fish, more fish species, and a shift in species composition from planktivores to piscivores (Figure 9). When all fish collections from sites 1 and 2 were considered, dominance (as percent of sample biomass) of the top predator, *Micropterus salmoides*, was positively correlated with mean water depth, while the tolerant insectivore, *Lepomis humilis*, and the planktivore, *Dorosoma cepedianum*, were negatively correlated with mean water depth (Table 4). Thus, as site 2 water depth increased following rehabilitation, its fish assemblage became less similar to site 1. The Bray-Curtis dissimilarity coefficient between

Table 3. Medians for Mean Water Depth and Selected Water Quality Variables From Degraded Reference Backwater and Rehabilitated Backwater Before and After Addition of Weirs[a]

Variable	Degraded Reference Backwater (Site 1)	Rehabilitated Backwater (Site 2)	
		Before Weir	After Weir
Mean depth on days when samples were collected (m)	0.56*	0.60*	0.71**
pH	6.8*	6.7*	6.0**
Dissolved oxygen (mg L^{-1})	4.2*	5.4**	6.2**
Secchi disk depth (cm)		21*	40**
Turbidity (NTU)	38*	51*	16**
Suspended solids (mg L^{-1})	40*	60*	17**
Total P (mg L^{-1})	1.18*	0.77*	0.33**
Filterable P (mg L^{-1})	0.061*	0.052*	0.041**
NH_3– (mg L^{-1})	0.001*	0.012*	0.009*
Total N (mg L^{-1})	1.056*	0.132**	0.092***
Chlorophyll *a* (μg L^{-1})	23*	78**	39***

[a]Medians with different superscripts are significantly different ($p < 0.05$, Dunn's method for multiple comparisons, Kruskal-Wallis analysis of variance (ANOVA) on ranks). Boldface variable names indicate significant differences in the rehabilitated backwater before and after rehabilitation.

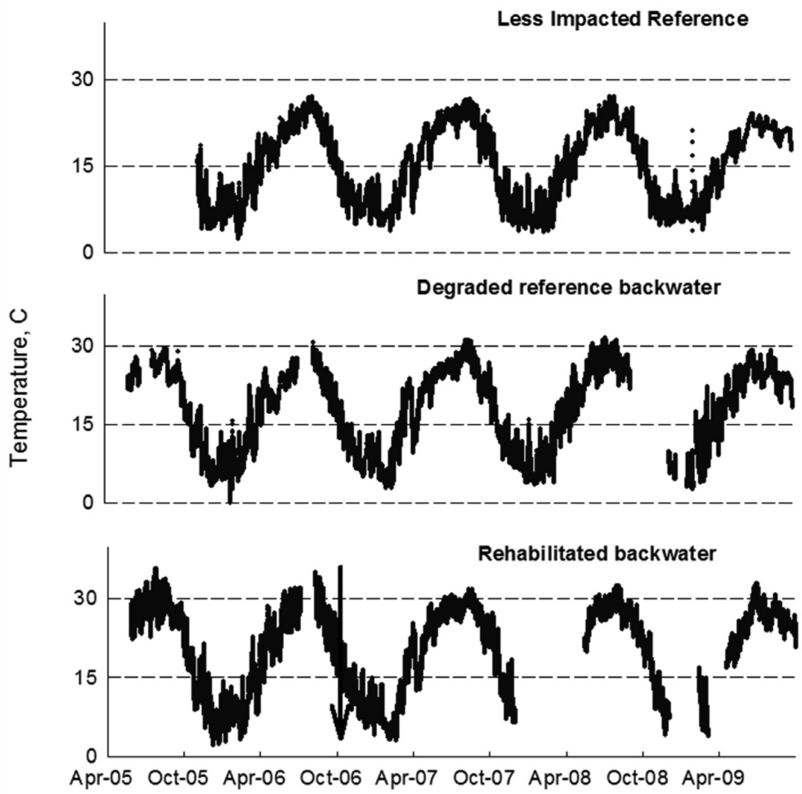

Figure 7. Water temperature for less impacted reference backwater, degraded reference backwater, and rehabilitated backwater. Vertical black arrow indicates date for completion of weir construction.

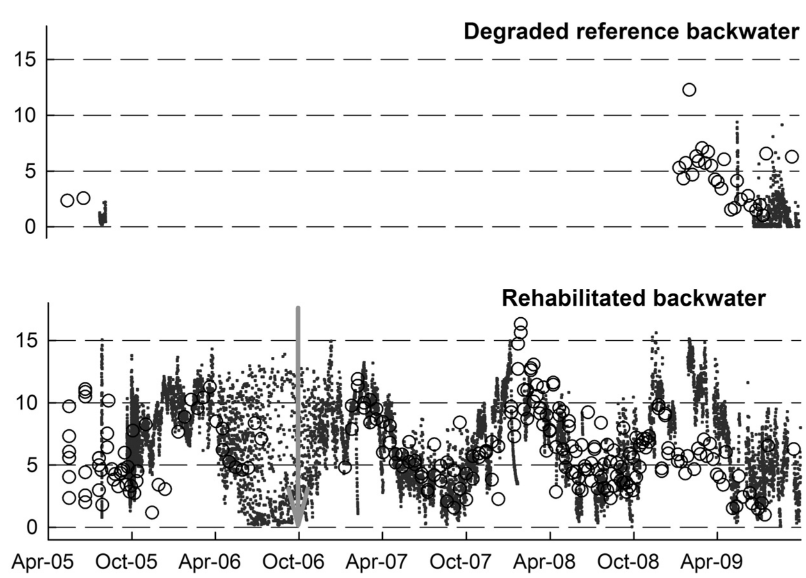

Figure 8. Dissolved oxygen concentrations for degraded reference backwater and rehabilitated backwater. Open circles are weekly measurements using handheld meter, while solid symbols are values from loggers collected at 4 h intervals. Vertical arrow indicates date for completion of weir construction.

Table 4. Pearson Correlation Coefficients r Between Descriptors of Electrofishing Samples From the Rehabilitated Backwater and Key Physical Variables[a]

	Days With Hydraulic Connection to River		
	During Previous 6 Months	During Previous 3 Months	Mean Water Depth
Number of fish species	0.139 (0.684)	0.329 (0.323)	**0.579 (0.062)**
Mean fish size (g)	−0.287 (0.391)	−0.209 (0.537)	0.501 (0.116)
Catch biomass as piscivores (%)	−0.132 (0.698)	0.117 (0.733)	0.512 (0.107)
Catch biomass as planktivores (%)	−0.161 (0.599)	−0.170 (0.616)	**−0.589 (0.057)**
Catch biomass as *Micropterus salmoides* (%)	0.233 (0.491)	0.417 (0.202)	0.515 (0.105)
Catch biomass as *Lepomis humilis* (%)	0.311 (0.351)	0.156 (0.647)	**−0.817 (0.002)**
Catch biomass as *Dorosoma cepadium* (%)	−0.081 (0.813)	−0.170 (0.618)	**−0.592 (0.055)**

[a]Numbers in parentheses are p values. Values in boldface indicate $p < 0.10$.

sites 1 and 2 was 0.28 prior to rehabilitation but 0.13 afterward (Table 5).

7. DISCUSSION AND CONCLUSIONS

Future development of stream corridors should adopt an ecological engineering paradigm [*Mitsch and Jørgensen*, 2004] that manages ecosystems for the totality of services they can provide. Since cutoff bends and other types of floodplain backwaters are common along large, lowland rivers, these areas merit special attention [*Zalewski*, 2006]. Cutoff bends may be managed using a combination of water control/flow diversion techniques [*Shields et al.*, 2005]. Key questions regarding the design of these measures have to do with the timing and duration of flow connection with the main channel [*Shields et al.*, 2009]. Alternative designs may be evaluated by comparing the level of main channel connectivity and hydrologic variability they produce relative to degraded and least impacted sites [*Kondolf et al.*, 2006].

Installation and operation of a low weir in the degraded cutoff bend described here reduced main channel connectivity and stage variation relative to the preconstruction and degraded reference site conditions. Observed chemical and biological changes were evidently related to moderating temporal hydrologic variations by increasing dry season water depths by about 0.15 m and by diverting agricultural runoff from about 350 ha of cultivated fields. In general, water quality improved as solids and nutrient concentrations declined. Others have reported floodplain lake quality improvements following diversion of polluted runoff [*Cooper*, 1993; *Filipek et al.*, 1993; *Cooper et al.*, 1995]. Water quality impairment has been directly linked to shallow depths in floodplain lakes within this region due to coincident problems associated with nutrient enrichment and biochemical oxygen

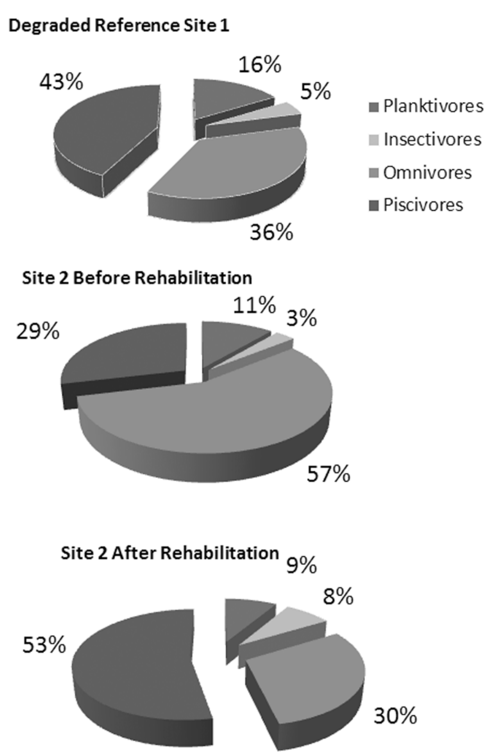

Figure 9. Trophic structure of fish assemblages in study backwater sites 1 and 2.

Table 5. Bray-Curtis Dissimilarity Coefficients for Sites 1 and 2 Based on Abundances of the 11 Most Abundant Fish Species

	Degraded Reference Backwater (Site 1)	Site 2 Before Rehabilitation	Site 2 After Rehabilitation
Degraded reference backwater (site 1)	0.00		
Site 2 before rehabilitation	0.13	0.00	
Site 2 after rehabilitation	0.28	0.16	0.00

demand from allochthonous organic matter [*Miranda and Lucas*, 2004]. In addition, shallow depths are more susceptible to increased turbidity from wind action and bottom-feeding fishes and to DO depletion by benthic respiration. Reduced depth means there is less oxygen in the water column to support such respiration [*Miranda et al.*, 2001] and less water to absorb incident solar energy. Shallow lakes in this region often experience wide diurnal swings in temperature, DO, and pH during warmer months [*Justus*, 2006].

Despite the continued problems with low DO in summer at the rehabilitated site 2, fish responded to greater depth and reduced variability in a fashion similar to that reported by *Miranda and Lucas* [2004] based on a study of 11 oxbow lakes along the Mississippi River. Degraded backwaters supported assemblages dominated by, "species that thrive in turbid, shallow systems with few predators and low oxygen content." Fish assemblages in the treated site 2 trended away from those typical of shallow, small systems studied by *Miranda and Lucas* [2004], but did not shift toward assemblages Miranda found in highly connected backwaters. Water level stabilization using a levee controlling an Illinois River backwater produced a shift in fish species community structure similar to the one reported here [*Pegg et al.*, 2006].

The backwater rehabilitation project described here had three main shortcomings. First, it failed to fully address the problem of hypoxia during warmer months. Evidently, the cyclic hypoxia reflects the high level of nutrient enrichment and attendant algal activity common to shallow backwater systems in cultivated floodplains in this region [*Miranda et al.*, 2001; *Justus*, 2006]. Others have reported anoxia in riverine backwaters and have suggested that these conditions may be ameliorated by introducing flow from the river through the backwater [e.g., *Theiling*, 1995]. In fact, we were able to produce dramatic water quality improvements in site 2 by pumping a modest amount of water from the adjacent river into the backwater for about 4 weeks early in the prerehabilitation period [*Cooper et al.*, 2006]. The second main failing of the rehabilitation project was its adverse effect on lateral connectivity between the backwater and the river main stem. The less impacted reference site was connected to the river more than half of the time in the postrehabilitation period, while the rehabilitated site enjoyed connection only about 2% of the time. Although many attest to the importance of connectivity as a determinant of backwater fish community structure [e.g., *Valdez and Wick*, 1981; *Grift et al.*, 2001; *Lusk et al.*, 2003; *Penczak et al.*, 2004; *Miranda*, 2005] and perhaps the value of the backwater as nursery habitat for river species [*Csoboth and Garvey*, 2008], the level of connectivity needed to produce a given level of ecological benefits is unknown. Others have reported fish migrating over and through water control structures to access floodplain backwaters [*Schultz et al.*, 2007; *Csoboth and Garvey*, 2008]. Third and finally, questions arise regarding the sustainability of restoration efforts like ours. The weirs we constructed create more favorable hydrologic variation based on observations of the reference sites, and diversion of polluted agricultural runoff will slow degradation of the rehabilitated site. Nevertheless, over the long term, this area will continue to experience sedimentation and eutrophication even if at a reduced rate. Full recovery of floodplain ecosystem services will require manipulation of main channel flows [*Theiling*, 1995] and floodplain vegetation and topography [*Baptist et al.*, 2004].

Rehabilitation research is challenging due to the complexity of natural ecosystems and our inability to replicate these systems or isolate the influences of key variables. This study attempted to use a before-after-control-impact approach to assess rehabilitation effects with the "control" role filled by a degraded and a less degraded site. However, our efforts were hampered by lack of resources required to study the degraded site in as great a detail as the rehabilitated site. Furthermore, the site selected as "less degraded" proved to be a poor biological reference, perhaps due to toxic residues [*Knight et al.*, 2009a, 2009b], demonstrating yet again how hard it is to find suitable reference sites for studies such as this. The Kondolf method was a useful tool in assessing physical performance of our project, but selection of the most ecologically appropriate measures of variability and connectivity is a key step. More work is needed to refine the use of the Kondolf approach for aquatic ecosystem evaluation.

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REFERENCES

Amoros, C. (2001), The concept of habitat diversity between and within ecosystems applied to river side-arm restoration, *Environ. Manage.*, *28*(6), 805–817.

Amoros, C., and G. Bornette (2002), Connectivity and biocomplexity in waterbodies of riverine floodplains, *Freshwater Biol.*, *47*, 761–776.

Baker, J. A., K. J. Kilgore, and R. L. Kasul (1991), Aquatic habitats and fish communities in the Lower Mississippi River, *Rev. Aquat. Sci.*, *3*(4), 313–356.

Baptist, M. J., W. E. Penning, H. Duel, A. J. M. Smits, G. W. Geerling, G. E. M. Van Der Lee, and J. S. L. Van Alphen (2004), Assessment of the effects of cyclic floodplain rejuvenation on flood levels and biodiversity along the Rhine River, *River Res. Appl.*, *20*, 1–13.

Bellrose, F. C., S. P. Havera, F. L. Paveglio Jr., and D. W. Steffeck (1983), The fate of lakes in the Illinois River Valley, *Ill. Nat. Hist. Surv. Biol. Notes*, *119*, 1–27.

Bhowmik, N. G., and M. Demissie (1989), Sedimentation in the Illinois River valley and backwater lakes, *J. Hydrol.*, *105*(1–2), 187–195.

Bray, J. R., and J. T. Curtis (1957), An ordination of the upland forest communities of Southern Wisconsin, *Ecol. Monogr.*, *27*, 325–349.

Buijse, A. D., H. Coops, M. Startas, L. H. Jans, G. J. Van Geest, R. E. Grifts, B. W. Ibelings, W. Oosterberg, and F. M. Roozen (2002), Restoration strategies for river floodplain along large lowland rivers in Europe, *Freshwater Biol.*, *47*, 889–907.

Claflin, T. O., and J. R. Fischer (1995), Declines in aquatic vegetation in Navigation Pool No. 8, Upper Mississippi River between 1975 and 1991, *Reg. Rivers Res. Manage.*, *11*, 157–165.

Cooper, C. M. (1993), Effects of suspended sediment reduction on chlorophyll and water quality in a large natural lake, *Verh. Int. Verein. Limnol.*, *25*, 392–395.

Cooper, C. M., S. S. Knight, F. R. Schiebe, and J. C. Ritchie (1995), Restoration of Lake Chicot, Arkansas, *Adv. Hydrosci. Eng.*, *2*, 97–108.

Cooper, C. M., R. E. Lizotte Jr., and F. D. Shields Jr. (2006), Rehabilitation of a severed meander bendway: Effects of flow augmentation on water quality, paper presented at Annual Meeting, N. Am. Benthol. Soc., Anchorage, Alaska. (Available at http://www.benthos.org/Other-Publications/NABStracts/2006/710.aspx)

Csoboth, L. A., and J. E. Garvey (2008), Lateral exchange of a larval fish between a restored backwater and a large river in the east-central USA, *Trans. Am. Fish. Soc.*, *137*, 33–44.

Filipek, S. P., J. D. Ellis, W. J. Smith, D. R. Johnson, W. M. Bivin, and L. L. Rider (1993), The effect of the Lake Chicot renovation project on the fishery of a Mississippi River oxbow lake, *Proc. Annu. Conf. Southeast. Assoc. Fish Wildlife Agencies*, *45*, 293–306.

Galat, D. L., et al. (1998), Flooding to restore connectivity of regulated, large-river wetlands, *BioScience*, *48*(9), 721–733.

Glantz, S. A. (1992), *Primer of Biostatistics*, 3rd ed., McGraw-Hill, Inc., New York.

Gore, J. A., and F. D. Shields Jr. (1995), Can large rivers be restored?, *Bioscience*, *45*(3), 142–152.

Grift, R. E., A. D. Buijse, W. T. Van Densen, and J. P. Klein Breteler (2001), Restoration of the river-floodplain interaction: Benefits for the fish community in the River Rhine, *Large Rivers*, *12*(2–4), 173–185.

Heiler, G., T. Hein, F. Schiemer, and G. Bornette (1995), Hydrological connectivity and flood pulses as the central aspects for the integrity of a river-floodplain system, *Reg. Rivers Res. Manage.*, *11*, 351–361.

Hesse, L. W., and W. Sheets (1993), The Missouri River Hydrosystem, *Fisheries*, *18*(5), 5–14.

Holubova, K., R. D. Hey, and M. J. Lisicky (2005), Middle Danube tributaries: Constraints and opportunities in lowland river restoration, *Large Rivers*, *15*(1–4), 507–519.

Hughes, F. M. R., A. Colston, and J. O. Mountford (2005), Restoring riparian ecosystems: The challenge of accommodating variability and designing restoration trajectories, *Ecol. Soc.*, *10*(1), 12.

Jackson, D. C. (2003), Fisheries dynamics in the Yazoo River Basin, in *Proceedings of the Second International Symposium on the Management of Large Rivers for Fisheries, Volume II*, edited by R. L. Welcomme and T. Petr, pp. 103–115, Reg. Off. for Asia and the Pac., Food and Agric. Organ., Bangkok. (Available at http://www.fao.org/docrep/007/ad526e0b.htm#bm11)

Jackson, D. C., Q. Ye, M. E. Stopha, T. D. Rhine, and N. J. Brown-Peterson (1995), Riverine fisheries resources assessments in the upper Yazoo River basin, *Fed. Aid Proj. F-94*, Miss. Dep. of Wildlife, Fish. and Parks, Jackson.

Julien, P. Y., S. C. Shah-Fairbank, J. Kim (2008). Restoration of abandoned channels, report, Eng. Res. Cent., Colo. State Univ., Fort Collins. (Available at http://www.engr.colostate.edu/~pierre/ce_old/Projects/linkfiles/Restoration%20of%20Abandoned%20Channels%204-21-08.pdf)

Justus, B. (2009), Water quality of least-impaired lakes in eastern and southern Arkansas, *Environ. Monit. Assess.*, *168*, 363–383, doi:10.1007/s10661-009-1120-5.

Justus, B. G. (2006), Water quality of eleven lakes in eastern and southern Arkansas from August 2004–July 2005, *U.S. Geol. Surv. Sci. Invest. Rep., 2006-5047.*

Karr, J. R., J. D. Allan, and A. C. Benke (2000), *River Conservation in the United States and Canada*, John Wiley, New York.

Knight, S. S., R. E. Lizotte Jr., M. T. Moore, S. Smith Jr., and F. D. Shields Jr. (2009a), Mississippi oxbow lake sediment quality during an artificial flood, *Bull. Environ. Contam. Toxicol.*, *82*(4), 496–500.

Knight, S. S., R. E. Lizotte Jr., and F. D. Shields Jr. (2009b), *Hyalella azteca* responses to Coldwater River backwater sediments in Mississippi, USA, *Bull. Environ. Contam. Toxicol.*, *83*(4), 493–496.

Kondolf, G. M., et al. (2006), Process-based ecological river restoration: Visualizing three-dimensional connectivity and dynamic vectors to recover lost linkages, *Ecol. Soc., 11*(2), 5. (Available at http://www.ecologyandsociety.org/vol11/iss2/art5/).

Light, H. M., K. R. Vincent, M. R. Darst, and F. D. Price (2006), Water-level decline in the Apalachicola River, Florida, from 1954 to 2004, and effects on floodplain habitats, *U.S. Geol. Surv. Sci. Invest. Rep., 2006-5173*.

Lin, D. S. C., and E. P. Caramaschi (2005), Responses of the fish community to the flood pulse and siltation in a floodplain lake of the Trombetas River, Brazil, *Hydrobiologia*, *545*, 75–91.

Lizotte, R. E., F. D. Shields Jr., S. S. Knight, and C. T. Bryant (2009), Efficiency of a modified backwater wetland in trapping a pesticide mixture, *Ecohydrology*, *2*(3), 287–293, doi:10.1002/eco.52.

Lusk, S., K. Halacka, and V. Luskova (2003), Rehabilitating the floodplain of the lower river Dyje for fish, *River Res. Appl.*, *19*, 281–288.

Master, L. L., S. R. Flack, and B. A. Stein (Eds.) (1998), *Rivers of Life: Critical Watersheds for Protecting Freshwater Biodiversity*, The Nat. Conserv., Arlington, Va.

Miranda, L. E. (2005), Fish assemblages in oxbow lakes relative to connectivity with the Mississippi River, *Trans. Am. Fish. Soc.*, *134*(6), 1480–1489.

Miranda, L. E., and G. M. Lucas (2004), Determinism in fish assemblages of floodplain lakes of the vastly disturbed Mississippi Alluvial Valley, *Trans. Am. Fish. Soc.*, *133*(2), 358–370.

Miranda, L. E., J. A. Hargreaves, and S. W. Raborn (2001), Predicting and managing risk of unsuitable dissolved oxygen in a eutrophic lake, *Hydrobiologia*, *457*, 177–185.

Mississippi Department of Environmental Quality (2003), Sediment TMDL for the Coldwater River, report, Jackson, Miss. (Available at http://www.deq.state.ms.us/MDEQ.nsf/pdf/TWB_ColdwaterRiverSedimentJun03/$File/YazooRBColdwaterRiverSedimentJun03.pdf?OpenElement)

Mitsch, W. J., and S. E. Jørgensen (2004), *Ecological Engineering and Ecosystem Restoration*, John Wiley, New York.

O'Donnell, T. K., and D. L. Galat (2007), River enhancement in the Upper Mississippi River Basin: Approaches based on river uses, alterations, and management agencies, *Restor. Ecol.*, *15*(3), 538–549.

Palmer, M., J. D. Allan, J. Meyer, and E. S. Bernhardt (2007), River restoration in the twenty-first century: Data and experiential knowledge to inform future efforts, *Restor. Ecol.*, *15*(3), 472–481.

Pegg, M. A., K. S. Irons, T. M. O'Hara, and M. A. McClelland (2006), Initial response of a floodplain community to water-level stabilization, *Ecol. Freshwater Fish*, *15*, 40–47.

Penczak, T., W. Galicka, L. Glowacki, H. Koszalinski, A. Kruk, G. Zieba, J. Kostrzewa, and L. Marszal (2004), Fish assemblage changes relative to environmental factors and time in the Warta River, Poland, and its oxbow lakes, *J. Fish Biol.*, *64*(2), 483–501.

Ricciardi, A., and J. B. Rasmussen (1999), Extinction rates of North American freshwater fauna, *Conserv. Biol.*, *13*(5), 1220–1222.

Richter, B. D., J. V. Baumgartner, D. P. Braun, and J. Powell (1998), A spatial assessment of hydrologic alteration within a river network, *Reg. Rivers Res. Manage.*, *14*, 329–340.

Roozen, F. M., G. J. van Geest, B. W. Ibelings, R. M. Roijackers, M. Scheffer, and A. D. Buijse (2003), Lake age and water level affect the turbidity of floodplain lakes along the lower Rhine, *Freshwater Biol.*, *48*, 519–531.

Schramm, H. L., Jr., and A. B. Spencer (2006), A spatial analysis of aquatic habitat change in the Tennessee-Tombigbee waterway, report, Miss. Dep. of Wildl., Fish., and Parks, Jackson.

Schultz, D. W., J. E. Garvery, and R. C. Brooks (2007), Backwater immigration by fishes though water control structure: Implications for connectivity and restoration, *North Am. J. Fish. Manage.*, *27*(1), 172–180.

Shields, F. D., Jr., and S. R. Abt (1989), Sediment deposition in cutoff meander bends and implications for effective management, *Reg. Rivers Res. Manage.*, *4*, 381–396.

Shields, F. D., Jr., and C. W. Pearce (2010), Control of agricultural nonpoint source pollution by natural wetland management, *J. Environ. Sci. Eng.*, *4*(4), 62–70.

Shields, F. D., Jr., S. S. Knight, and J. M. Stofleth (2005), Alternatives for riverine backwater restoration by manipulation of severed meander bend, in *Proceedings of 2005 World Water and Environmental Resources Congress* [CD-ROM], Am. Soc. of Civ. Eng., Reston, Va.

Shields, F. D., Jr., S. S. Knight, J. M. Stofleth, and D. G. Wren (2009), Towards a basis for designing backwater and side channel restorations, in *Proceedings of the 33rd IAHR Congress* [CD-ROM], Int. Assoc. for Hydraul. Res., Madrid.

Shields, F. D., Jr., S. S. Knight, R. E. Lizotte Jr., C. T. Bryan, and D. G. Wren (2010), Floodplain river backwater restoration: A case study, in *Proceedings of the 9th Federal Interagency Sedimentation Conference, Las Vegas, NV* [CD-ROM], Joint Fed. Interagency Conf. Organ. Comm., Washington, D. C.

Society for Ecological Restoration International Science and Policy Working Group (2004), The SER International Primer on Ecological Restoration, version 2, report, Soc. for Ecol. Restor. Int., Washington, D. C. (Available at http://www.ser.org/pdf/primer3.pdf)

Systat Software, Inc. (2009), *SigmaPlot 11.2 User's Guide*, San Jose, Calif.

Theiling, C. H. (1995), Habitat rehabilitation on the upper Mississippi River, *Reg. Rivers Res. Manage.*, *11*, 227–238.

Thorp, J. H., M. C. Thoms, and M. D. Delong (2006), The riverine ecosystem synthesis: Biocomplexity in river networks across space and time, *River Res. Appl.*, *22*, 123–147.

Valdez, R. A., and E. J. Wick (1981), Natural vs manmade backwaters as native fish habitat, in *Aquatic Resources Management of the Colorado River Ecosystem*, edited by V. D. Adams and V. A. Lamarra, pp. 519–536, Ann Arbor Sci., Ann Arbor, Mich.

Ward, J. V., K. Tockner, U. Uehlinger, and F. Malard (2001), Understanding natural patterns and processes in river corridors as the basis for effective river restoration, *Reg. Rivers Res. Manage.*, *17*, 311–323.

Warren, M. L., Jr., et al. (2000), Diversity, distribution, and conservation status of the native freshwater fishes of the southern United States, *Fisheries*, *24*(10), 7–29.

Wiens, J. A. (2002), Riverine landscapes: Taking landscapes ecology into the water, *Freshwater Biol.*, *47*, 501–515.

Whitten, C. B., and D. M. Patrick (1981), Engineering geology and geomorphology of streambank erosion (Report 2: Yazoo River Basin Uplands, Mississippi), *Tech. Rep. GL-79-7, Rep. 2, TR GL-79-7*, U.S. Army Eng. Waterw. Exp. Stn., Vicksburg, Miss.

Zalewski, M. (2006), Flood pulses and river ecosystem robustness, *IAHS Publ.*, *305*, 143–154.

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