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EFFECTS OF ARTIFICIAL FLOODING ON WATER QUALITY OF A FLOODPLAIN BACKWATER

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ABSTRACT

Seasonal flooding of riverine backwaters is important in maintaining diverse aquatic habitats, but anthropogenic impacts have reduced the frequency and duration of such flooding. This study, conducted in a 2.5-km-long shallow floodplain severed meander backwater adjacent to the Coldwater River in Tunica County, Mississippi, USA, compared water quality during a late summer 30-day artificial flooding period with 28-day pre-flood and 26-day post-flood periods. Flooding was simulated by pumping 0.22 to 0.35 $m^3 s^{-1}$ from the river into the upstream portion of the backwater. *In situ* parameters (temperature, pH, dissolved oxygen, conductivity and fluorescent chlorophyll) were measured every 30 min at one site within the backwater. Solids (dissolved and suspended) and nutrients (phosphorus and nitrogen) were measured at three sites in the backwater and in the river every 3 to 5 days. Decreases in the amplitude of temperature, dissolved oxygen and pH diel cycles within the backwater were observed during flooding. Changes in patterns of solids and nutrients were also associated with flooding. Complex patterns in phosphorus and nitrogen emerged as a result of utilization by autotrophs (measured as chlorophyll) and seasonal changes. Artificial flooding in a shallow floodplain water body stabilized and improved water quality for aquatic biota and is a viable method for habitat rehabilitation in these systems.

KEY WORDS: flow augmentation; riverine floodplain; water chemistry; chlorophyll

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INTRODUCTION

River floodplain backwaters have dynamic habitats that can have high biological productivity (Buijse *et al*., 2002). Water flow via flooding from the mainstem river channel is important in providing ecosystem stability and high biodiversity within floodplain backwater habitats (Kingsford, 2000; Bunn and Arthington, 2002). Such stability has been observed in abiotic water quality variables such as dissolved oxygen and nutrients as well as physical habitat variables such as water depth (Pithart *et al*., 2007; Thomaz *et al*., 2007). Alterations in flow regimes due to anthropogenic channelization and water impoundments (dams) lead to decreased flooding and increased isolation of the floodplain. As a consequence, floodplain backwater habitats are altered via decreased water levels, poorer water quality and reduced biodiversity (Bunn and Arthington, 2002; Lake, 2003). In order to regain the high biological productivity of floodplain backwaters, restoration projects are needed in highly altered river systems such as those found in the lower Mississippi River alluvial plain.

The lower Mississippi River alluvial plain (i.e. 'the Delta') has numerous floodplain backwaters and oxbow lakes that have been increasingly isolated from their respective main river channels (Knight and Welch, 2004) and experience varying intermittent levels of hydrologic connectivity during high stage periods of adjacent rivers and streams. The Delta of Mississippi, a 18 130-km^2 portion of the Lower Mississippi River floodplain is one of the most intensively cultivated areas in the USA (Locke, 2004; Snipes *et al*., 2004) and produces a variety of crops including cotton (*Gossypium hirsutum*), soybeans (*Glycine max*), corn (*Zea mays*) and rice (*Oryza sativa*). Seasonal flooding of river floodplain backwaters can be important for maintaining diverse aquatic habitats (Dodds, 2002). Anthropogenic impacts such as river channelization and flood control have reduced frequency and duration of such flooding at many sites (Gore and Shields, 1995; Kingsford, 2000). This loss of hydrologic connectivity coupled with habitat degradation, destruction and non-point source pollution has major implications for riverine ecosystems and the services they provide. Floodplain backwaters in the Mississippi Delta are particularly susceptible to aggradation because of accelerated sedimentation associated with watershed cultivation (Wren *et al*., 2008), water table declines associated with pumping for irrigation (Pennington, 1993) and isolation from river overflows by flood control

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levees, sedimentation and channel incision. Shallow depths lead to extreme diel variations in water quality including night time anoxia as respiration demand exceeds water column oxygen content (Miranda *et al*., 2001). These effects are so endemic throughout the Mississippi Delta that local fish assemblage composition is directly related to backwater or lake area and depth (Miranda and Lucas, 2004; Miranda, 2005).

Various approaches have been suggested to address aquatic habitat degradation and loss within agricultural landscapes (Brookes and Shields, 1996; Buijse *et al*., 2002; Shields *et al*., 2005). One approach is the rehabilitation of backwaters through hydrologic manipulations such as flow augmentation to produce artificial flooding. Artificial flooding may result in aesthetic, recreational, ecological and water quality benefits (Cooper and Knight, 1991; Shields *et al*., 2002). The purpose of this study was to assess the impact of flooding on backwater habitat stability by comparing water quality during a late summer 30-day artificial flood produced by flow augmentation to 28-day pre-treatment and 26-day post-treatment periods.

Our study was part of an investigation of the feasibility of improving aquatic habitat in a shallow (<1.5 m) floodplain backwater along the lower Coldwater River (34°40′21″N, 90°13′44″W) in Tunica County, Mississippi, USA. This study assessed water quality before, during and after a 30-day artificial flood in the backwater by examining a suite of 14 water quality parameters assessing physical, chemical and biological processes.

MATERIALS AND METHODS

Study site

This study was conducted in a 2.5-km-long shallow floodplain waterbody of a severed meander bend adjacent to the Coldwater River in Tunica County, Mississippi, USA (Figure 1). The segment is inside the Coldwater River mainstem flood control levee and is the result of a 0.4-km cutoff constructed in 1941–1942. The study reach is about 20 km downstream of Arkabutla Dam, a flood control structure in northwestern Mississippi, USA. Land use both inside and outside the water body are in row-crop cultivation; however, there is a buffer of natural vegetation 5–100 m wide on both banks (Figure 1). The backwater water body has connections with the mainstem channel river and becomes inundated primarily during the wet season (December–May) and rarely inundated during the dry season (June–November) and true lotic conditions (inundated with inflow and outflow) occur only about 8 days year^{-1} lasting an average of 2 days (Shields *et al*., 2005). As a result, backwater water levels average approximately 0.7 m deep annually with

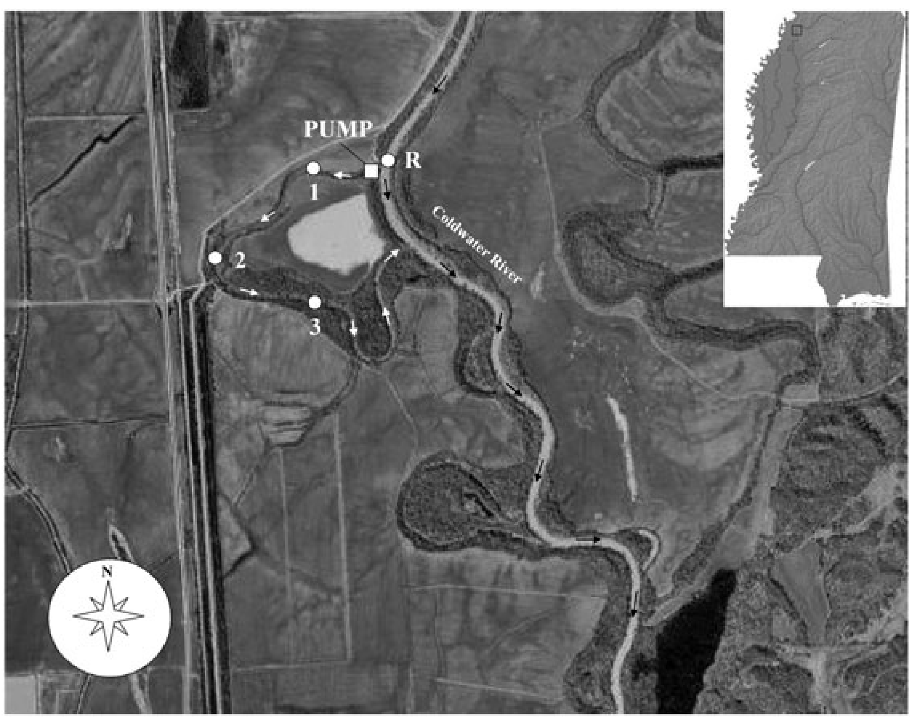

Figure 1. Aerial photograph of the backwater artificial flooding project along the lower Coldwater River, Mississippi. Arrows indicate water flow direction. Numbered symbols indicate water quality monitoring/sampling sites. Pumps were located at the white square.

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minimum water levels approximately 0.4 m deep allowing some water to remain in the backwater year-round. Three sites were monitored within the backwater during the study period, and one site was monitored along the Coldwater River at the inflow point (Figure 1).

Artificial flooding and hydrology

Artificial backwater flooding consisted of pumping water 0.22 to 0.35 $m^3 s^{-1}$ from the main channel of the river into the upstream end of the severed backwater bend for 30 days. Flooding was initiated on 17 August 2005 and ended 16 September 2005. Two Godwin DPC 300 (Bridgeport, NJ, USA) centrifugal diesel-powered pumps were used to simulate a low discharge flood event of significant duration. The pumps lifted water 2–3 m from the Coldwater River and discharged through a 0.75-m diameter culvert into a riprap-lined stilling basin at the upstream end of the backwater. When the backwater surface elevation exceeded the elevation of aggraded sediment deposits in the lower limb of the old backwater meander (approximately 53 m, Figure 2), flow returned to the river via a small ditch. When pumping ceased (post-treatment), water levels declined to the sediment deposit elevation and then continued to decline gradually as a result of evaporation.

Data collection

During pumping, stage records were logged by self-contained pressure transducers in the Coldwater River near the pump intake (site R, Figure 1) and at two sites in the segment (site 1 and downstream from site 3, Figure 1). Water depth and current velocity were monitored at the channel centerline at a transect in the downstream portion of the

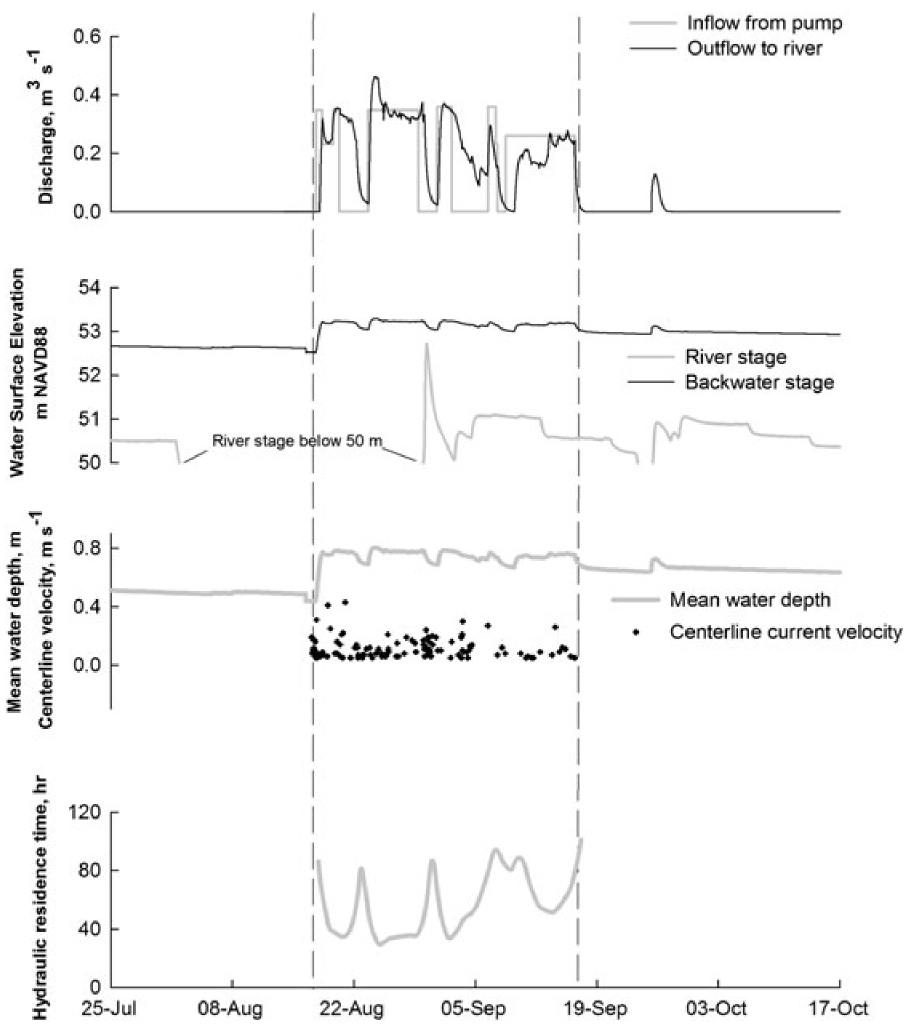

Figure 2. Time series of backwater inflow and outflow, river stage, backwater stage, mean backwater depth and centerline current velocity before, during and after the pumping experiment. Beginning and ending of pumping period is shown by vertical dashed lines.

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segment (site 3, Figure 1). A digital elevation model of the segment was constructed using LiDAR coverage of overbank areas and cross-sectional and bathymetric survey data collected from the segment using total station and echosounder coupled with differentially corrected global positioning system. Pump operation records were provided by the contractor responsible for pump operation. Periodic manual discharge measurements were obtained using a wading rod and electromagnetic flow meter in a constriction at the apex of the segment where a road culvert had formerly been placed and within the downstream connecting channel (near site 2, Figure 1). The resulting discharge values were used to produce empirical coefficients for a broad-crested weir equation for backwater discharge to the river. A time series of mean hydraulic residence time for the backwater was computed as the ratio of moving average backwater volume to moving average backwater discharge, modified by a factor to allow for hydraulic inefficiencies (Thackston *et al*., 1987). Hydrologic and geometric data were collated to produce time series of water inflow, outflow, bend water surface elevation, surface area, volume, mean depth (volume/area) and hydraulic residence time. Daily solar radiation in langleys for the experimental period and locale were obtained from the Delta Research and Extension Center of Mississippi State University (http://www.deltaweather.msstate.edu). Solar radiation was measured and logged at a permanent weather station about 70 km south of our site at Vance, MS.

Water quality was assessed during a 28-day pre-treatment period, a 30-day flow augmentation period and a 26-day post-treatment period at each of the three backwater sites and the Coldwater River inflow site (Figure 1). Pre-treatment water quality samples were collected from each site every 2–5 days during pre-treatment ($n = 8$) from 21 July to 15 August 2005. Treatment samples were collected every 1–4 days ($n = 12$) from 17 August to 15 September 2005. Post-treatment samples were collected every 2–7 days ($n = 8$) from 19 September to 13 October 2005. Samples were 1-L surface water grab collections preserved via chilling on wet ice and transported to the US Department of Agriculture Agricultural Research Service National Sedimentation Laboratory. Site specific point water depth was measured concurrently with surface water grab sampling at each of the three backwater sites 1, 2 and 3. Physical, chemical and biological water parameters consisting of turbidity (calibrated Hach electronic turbidimeter, Loveland, CO, USA), total suspended solids (TSS, dried at 180 °C), total dissolved solids (TDS, filtered through a 45-µm cellulose nitrate filter and dried at 180 °C), alkalinity (titration method), hardness (EDTA titrametric method), soluble reactive P (SRP, filtered through a 45-µm cellulose nitrate filter and analyzed using the ascorbic acid method), total P (TP, persulfate digestion with ascorbic acid method), NH_4-N (phenate method), NO_3-N (cadmium reduction method), NO_2-N (colorimetric method), total N [TN, NO_3-N + NO_2-N + total Kjeldahl N (block digestion and flow injection analysis method)] and chlorophyll *a* (pigment extraction and spectrophotometric determination) were analyzed using standard methods for the examination of water and wastewater (APHA, 1998). Molar TN:TP ratios were calculated based upon measured TN and TP as described earlier. In addition, calibrated multi-parameter water quality data loggers, Yellow Springs Instruments, Inc., (YSI) 6600 (Yellow Springs, OH, USA) were concurrently deployed to collect *in situ* measurements of temperature, pH, conductivity, dissolved oxygen and fluorescent chlorophyll every 30 min at backwater site 1. Fluorescent chlorophyll probe was calibrated against samples collected approximately weekly and analyzed using the trichromatic method for total chlorophyll (APHA, 1998) according to Çelik (2006). Logger sensors were deployed at approximately mid-depth to minimize influences from either the air–water interface or sediment–water interface; temperature loggers concurrently deployed at 0.2 m and 1.0 m depths indicated weak (0–3.5 °C) vertical thermal stratification that peaked in mid-afternoon and disappeared at night.

Data analysis

Diel mean, maximum and range amplitude were determined for *in situ* parameters. Diel range was calculated as the difference between the daily maximum and minimum values for that parameter. For all water quality parameters assessed, a one-way analysis of variance (ANOVA) with Dunnett's multiple comparison test versus controls was used to compare backwater pre-treatment period (control) with treatment and post-treatment periods. When water quality data failed parametric assumptions, a Kruskal–Wallis one-way ANOVA on ranks with Dunn's multiple comparison test versus backwater pre-treatment controls was used. In addition, a one-tailed *t*-test was used to compare backwater and Coldwater River water quality parameters within pre-treatment, treatment and post-treatment periods. When water quality data failed parametric assumptions, a Mann–Whitney Rank Sum test was used. Spearman rank order correlations were used to assess associations of nutrient parameters within the backwater and backwater *in situ* fluorescent chlorophyll. Forward stepwise regressions were run to fit equations using log-transformed data to compute standardized regression coefficients. Regression equations were of the form

$$Y = b_0 + b_1X_1 + b_2X_2 + b_3X_3 + b_4X_4$$

where Y is the predicted value of a water quality constituent measured in the backwater, X_1 is the total solar radiation (representing seasonality) on the day Y was measured, X_2

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is the water depth at the time and location where Y was measured, X_3 is either the average water discharge in the bend during the 36 h prior to Y measurement or the product of this discharge and the value of constituent Y simultaneously measured in the adjacent river (pumped inflow) when such data were available and X_4 is the distance from the inflow point to the point of measurement measured along the segment. The values of the standardized regression coefficients, b_i were dimensionless values produced by multiplying each of the stepwise regression coefficients by the ratio of the standard deviation of the observed values of Y to the standard deviation of the observed values of X_i. Because the study occurred during summer and into autumn and seasonality is well known to influence several water quality variables (Wetzel, 2001; Scheffer, 2004), the influence of seasonality was addressed using solar radiation as a representative seasonal variable. Computations proceeded by forcing the solar radiation X_1 (representing seasonality) into the equation and then allowing the algorithm to add and remove variables using F to enter and remove values of 4.0 and 3.9, respectively. The relative influence of seasonal factors associated with solar radiation and factors influenced by flow augmentation (water depth and inflow × river quality) was assessed by comparing the magnitudes of the standardized coefficients. Significance of the resulting regressions, each regression coefficient and coefficients of determination, r^2, were noted. *In situ* water quality data (temperature, dissolved oxygen, pH, turbidity, fluorescent chlorophyll and conductivity) required a slightly modified approach. Because *in situ* measurements were made at 30-min intervals, daily extremes (maximum temperature and pH and minimum dissolved oxygen) or averages (mean turbidity, fluorescent chlorophyll and conductivity) were used as independent variables, Y, in regressions. The distance from the inflow point, X_4, was not included in these analyses because all *in situ* measurements were made at the same point. Furthermore, only temperature measurements were made in the river, so except for temperature, the X_3 values were simply water discharge. Statistical significance level was set at 5% ($p \leq 0.05$) for all analyses (Steel *et al*., 1997). All data analyses were conducted using SigmaStat software, Chicago, IL, USA (SPSS, 1997).

RESULTS

Hydrology

Pump operation and resulting backwater inflow was sporadic during the experiment because of equipment failures and forced removal of the portable pumps because of high river stages associated with the passage of Hurricane Katrina on 29 August 2005 (Figure 2). However, backwater volume, surface area and outflow were less variable than inflow during the experiment because of the buffering effect of water storage in the backwater (Figure 2). At maximum stage, backwater volume and mean water depth were approximately doubled over the pre-pumping level whereas surface area increased by about 23%. Average mean water depth in the backwater was 0.49 m prior to pumping, and the average for the entire period of pump operation (17/8/05 through 16/9/05) was 0.75 m. Point water depths at the water quality sampling sites 1, 2 and 3 increased from about 0.3 m from sites 1 to 3 and ranged from ~0.3 to 0.6 m prior to pumping, 0.8 to 1.2 m during pumping and 0.6 to 1.1 m afterward. Hydraulic residence time during pumping ranged from 30 to 94 h and averaged 55 h. Flow velocities in the backwater

Table I. Daily mean, maximum and range of *in situ* water quality parameters measured at site 1 of the Coldwater River backwater

Parameter		Pre-treatment ($n = 21$)	Treatment ($n = 29$)	Post-treatment ($n = 26$)
Temperature (°C)	Mean	30.6 (1.2)	27.9 (1.9)*	23.7 (3.3)*
	Maximum	36.9 (2.4)	29.4 (1.9)*	25.1 (3.3)*
	Range	10.7 (2.3)	2.7 (0.7)*	2.6 (0.6)*
Dissolved oxygen (mg L^{-1})	Mean	5.8 (1.5)	5.9 (0.5)	6.7 (1.5)*
	Maximum	13.7 (3.0)	7.1 (0.7)*	9.5 (2.3)*
	Range	13.2 (2.9)	2.1 (1.2)*	5.0 (1.8)*
Conductivity (μS cm^{-1})	Mean	180.3 (25.6)	129.2 (30.8)*	95.3 (7.8)*
	Maximum	198.0 (26.9)	137.2 (33.5)*	100.0 (11.1)*
	Range	33.8 (17.7)	15.8 (10.5)*	7.2 (5.6)*
pH (s.u.)	Mean	7.7 (0.2)	7.2 (0.1)*	7.2 (0.3)*
	Maximum	8.8 (0.4)	7.4 (0.1)*	7.8 (0.7)*
	Range	1.7 (0.4)	0.3 (0.2)*	1.0 (0.6)*
Fluorescent chlorophyll (μg L^{-1})	Mean	95.6 (16.6)	64.4 (4.6)*	104.3 (20.5)
	Maximum	124.4 (25.7)	75.2 (15.8)*	126.5 (24.1)
	Range	64.6 (24.8)	15.7 (15.0)*	38.2 (15.8)*

Values in parentheses are standard deviations; values with an asterisk indicate backwater value significantly different from backwater pre-treatment value ($p \leq 0.05$).

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during pumping were quite low, with centerline measurements averaging 0.1 m s^{-1}. Cross-sectional average velocities were less than 0.04 m s^{-1}.

In situ water quality

In situ backwater site 1 physical and chemical water quality constituents showed immediate and dramatic response to flow augmentation and artificial flooding (Table 1 and Figures 3 and 4). Temperature, dissolved oxygen, pH and fluorescent chlorophyll showed large diel ranges during the pre-treatment observation period (Figures 3 and 4). Maximum pre-treatment temperatures were often >35 °C with diel ranges typically between 8 and 12 °C (Figure 3). After initiation of flow during the treatment period, amplitude of minimum and maximum temperatures during diel cycles decreased within 24 h of initiation of flow augmentation and concomitant increase in depth (Figures 2 and 4). During flooding, maximum temperatures were <32 °C with diel ranges of only 2–4 °C. During post-treatment, maximum temperatures were <30 °C with diel ranges of only 2–3 °C. For dissolved oxygen, maximum pre-treatment concentrations were often between 12 and 13 mg L^{-1} and minimum values were below 1 mg L^{-1} (Figure 3). During treatment, maximum concentrations were typically between 6 and 8 mg L^{-1} and minimum concentrations were between 4 and 5 mg L^{-1} with only 3 days below 4 mg L^{-1} and diel ranges damped to only 1–3 mg L^{-1}. During post-treatment, maximum dissolved oxygen concentrations increased to 8–12 mg L^{-1} and minimum concentrations ranged from 3 to 6 mg L^{-1} with 6 days below 4 mg L^{-1}. Measurements of pH oscillated from slightly acidic (6.7) to basic (8.5–9.5).

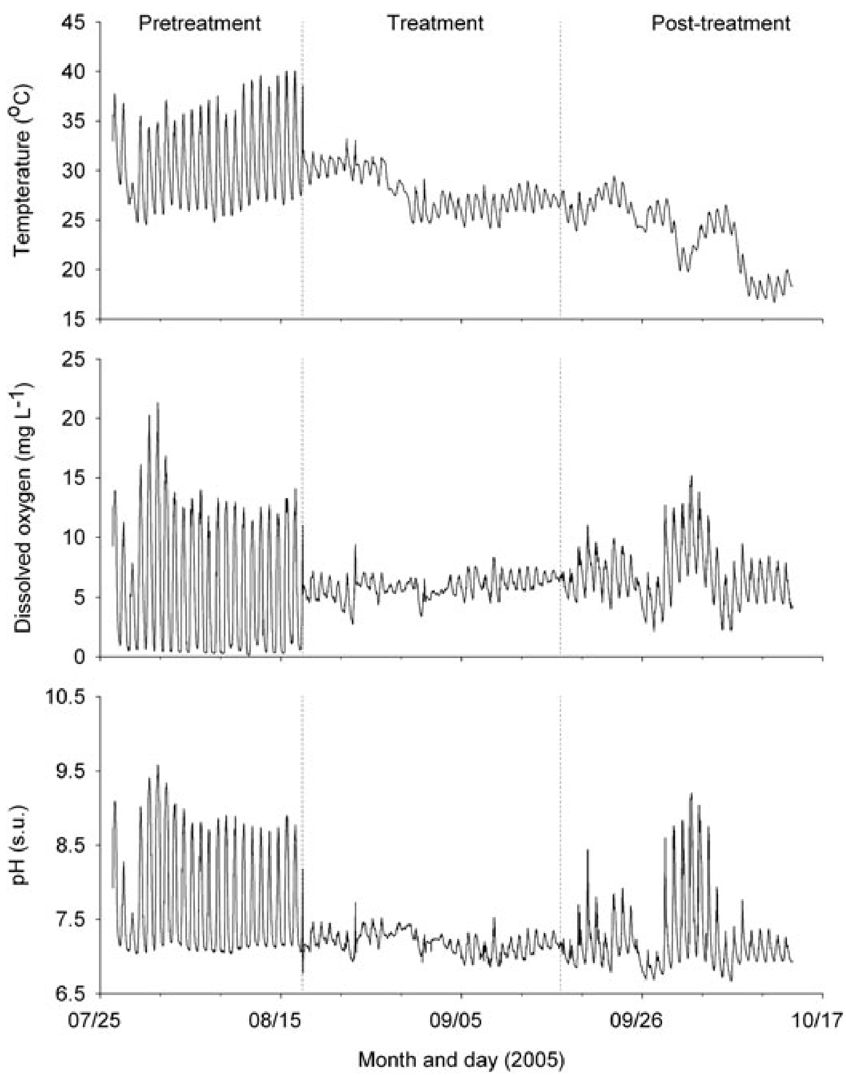

Figure 3. *In situ* water temperature, dissolved oxygen and pH measured in the Coldwater River backwater site 1 during pre-treatment, treatment and post-treatment of an artificial flood.

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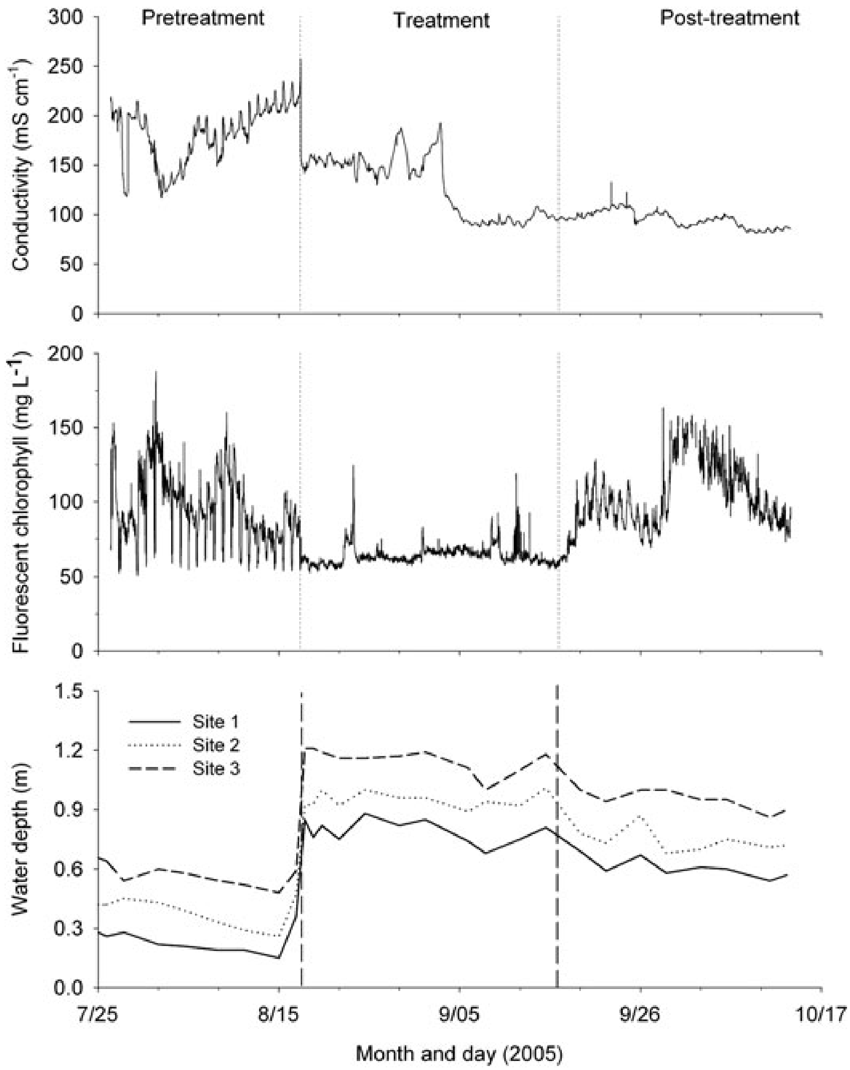

Figure 4. *In situ* conductivity (site 1), fluorescent chlorophyll (site 1) and point water depth (sites 1, 2 and 3) in the Coldwater River backwater during pre-treatment, treatment and post-treatment of an artificial flood.

Minimum and maximum pH values during diel cycles were greatest during the pre-treatment phase ranging from 6.8 to 9.5 and decreased with the initiation of flooding (Figure 3) and increases in depth (Figures 2 and 4). Overall pH remained relatively stable (6.7–7.5) during the flow period. Diel fluctuations in pH increased again after flow ceased in conjunction with fluctuations in dissolved oxygen (Figure 3). Conductivity during pre-treatment was typically between 150 and 200 $\mu S\,cm^{-1}$ with diel ranges between 30 and 50 $\mu S\,cm^{-1}$ (Figure 4). Conductivity declined within 3 h of treatment initiation and typically ranged from 90 to 150 $\mu S\,cm^{-1}$ with diel ranges damping to about 10 $\mu S\,cm^{-1}$. During post-treatment, conductivity was usually <100 $\mu S\,cm^{-1}$ with diel ranges <10 $\mu S\,cm^{-1}$. Fluorescent chlorophyll concentrations ranged from 50 to 190 $\mu g\,L^{-1}$ during the study period (Figure 4). Ranges of minimum and maximum fluorescent chlorophyll concentrations during diel cycles were greatest during the pre-treatment period and decreased with the initiation of flow augmentation with patterns very similar to dissolved oxygen and pH. Fluorescent chlorophyll concentrations remained stable during the flow period and increased again after treatment ceased (Figure 4).

Solids, nutrients and chlorophyll a

Mean Secchi visibility and laboratory-derived physicochemical water quality data from both the backwater and Coldwater River sites are presented in Table 2. Mean backwater Secchi visibility ranged from 0.18 to 0.27 m with only post-treatment significantly greater than pre-treatment measurements. Variations in mean backwater turbidity ranged from 34.1 to 83.6 NTU and were greater than those in

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Table II. Mean physicochemical water quality values in the Coldwater River (River) and in the adjacent shallow floodplain backwater (Backwater) before, during and after rehabilitation using flow augmentation

Parameter	Pre-treatment ($n = 8$)		Treatment ($n = 12$)		Post-treatment ($n = 8$)	
	River	Backwater	River	Backwater	River	Backwater
Secchi visibility (m)	–	0.18 (0.02)	–	0.22 (0.03)	–	0.27 (0.04)[a]
Temperature (°C)	28.3 (0.8)	29.8 (1.4)[b]	27.3 (1.8)	27.9 (2.2)	23.9 (2.9)	22.9 (3.4)[a]
Turbidity (NTU)	57.8 (7.4)	83.6 (37.6)[b]	77.1 (41.4)	62.1 (23.7)	157.1 (187.4)	34.1 (8.8)[a]
TSS ($mg\,L^{-1}$)	86.1 (15.8)	104.6 (55.3)	105.4 (81.9)	66.1 (22.3)[a]	220.6 (257.7)	41.8 (15.6)[a]
TDS ($mg\,L^{-1}$)	77.1 (15.6)	117.1 (10.0)[b]	87.4 (19.0)	83.1 (16.2)[a]	62.8 (13.2)	59.4 (8.0)[a]
Alkalinity ($mg\,L^{-1}$)	42.6 (11.3)	69.4 (8.4)[b]	45.5 (13.2)	45.2 (13.2)[a]	30.3 (6.0)	34.0 (1.9)[a]
Hardness ($mg\,L^{-1}$)	62.9 (7.6)	81.0 (13.5)[b]	62.7 (19.0)	58.5 (15.1)[a]	57.5 (12.8)	56.3 (6.4)[a]
SRP ($mg\,L^{-1}$)	0.025 (0.021)	0.021 (0.019)	0.060 (0.049)	0.061 (0.050)[a]	0.071 (0.080)	0.013 (0.011)[b]
TP ($mg\,L^{-1}$)	0.245 (0.052)	0.460 (0.072)[b]	0.343 (0.249)	0.247 (0.082)[a]	0.303 (0.179)	0.170 (0.029)[a]
NH_4-N ($mg\,L^{-1}$)	0.028 (0.023)	0.114 (0.067)[b]	0.008 (0.014)	0.017 (0.020)[a]	0.002 (0.006)	0.016 (0.031)[a]
NO_3-N ($mg\,L^{-1}$)	0.155 (0.107)	0.055 (0.051)[b]	0.128 (0.066)	0.120 (0.047)	0.135 (0.089)	0.081 (0.102)
NO_2-N ($\mu g\,L^{-1}$)	0.014 (0.008)	0.009 (0.005)	0.031 (0.035)	0.034 (0.032)[a]	0.029 (0.010)	0.016 (0.013)[b]
TN ($mg\,L^{-1}$)	0.9 (0.2)	3.1 (0.4)[b]	1.1 (0.3)	1.4 (0.4)[a]	1.1 (0.3)	1.5 (0.1)[a]
TN:TP ratio	8.6 (1.5)	15.3 (2.9)[b]	9.5 (4.5)	13.3 (4.8)	9.1 (2.5)	19.3 (2.3)[b]
Chlorophyll *a* ($\mu g\,L^{-1}$)	18.3 (16.2)	41.8 (33.7)	24.0 (5.5)	34.1 (12.5)[b]	23.2 (18.1)	37.1 (35.4)

–, data not collected at site. Values in parentheses are standard deviations.
[a]Backwater value significantly different from backwater pre-treatment value ($p \le 0.05$).
[b]Backwater value significantly different from Coldwater River value ($p \le 0.05$) within pre-treatment, treatment or post-treatment.

the Coldwater River site during pre-treatment and less than those in the Coldwater River site during post-treatment, and post-treatment was less than the pre-treatment measurements. Backwater TSS ranged from 41.8 to 104.6 $mg\,L^{-1}$ with treatment and post-treatment concentrations significantly less than those in pre-treatment and post-treatment concentrations significantly less in the backwater than in the Coldwater River site (Table 2). Mean backwater TDS, alkalinity and hardness decreased from pre-treatment to treatment and post-treatment and generally were greater than Coldwater River values during pre-treatment. Backwater SRP concentrations ranged from 0.013 to 0.061 $mg\,L^{-1}$ and significantly increased from pre-treatment to treatment and were significantly lower than Coldwater River concentrations only during post-treatment. Backwater TP concentrations ranged from 0.17 to 0.46 $mg\,L^{-1}$ and were significantly greater than those in the Coldwater River during pre-treatment, whereas treatment and post-treatment backwater concentrations were significantly lower than those in pre-treatment. Backwater dissolved inorganic nitrogen (NH_4-N, NO_3-N and NO_2-N) had sporadic variation in measured concentrations. Patterns of backwater NH_4-N concentrations were similar to TP. Backwater NO_3-N was lower than that in the Coldwater River during pre-treatment but was not significantly affected by flooding at levels of 0.9–1.2 m (Figure 4). Backwater NO_2-N was lower than that in the Coldwater River during post-treatment and significantly increased during flooding treatment. Backwater mean TN concentrations ranged from 0.9 to 3.1 $mg\,L^{-1}$ and were significantly greater than those in the Coldwater River during all three periods, whereas pre-treatment backwater concentrations were greater than treatment and post-treatment concentrations. Backwater mean TN:TP ratios ranged from 13:1 to 19:1 and were significantly greater than those in the Coldwater River during pre-treatment and post-treatment periods. Backwater mean chlorophyll *a* concentrations ranged from 34.1 to 41.8 $\mu g\,L^{-1}$ and were significantly greater than Coldwater River concentrations during flooding.

Water quality associations

Correlation analysis revealed several significant ($p \le 0.05$) associations between nutrients and *in situ* fluorescent

Table III. Spearman rank order correlation coefficients ($n = 23$) with diel mean, maximum and range *in situ* fluorescent chlorophyll concentrations and nutrients measured at site 1 of the Coldwater River backwater

Nutrients	*In situ* fluorescent chlorophyll		
	Mean	Maximum	Range
SRP	−0.629*	−0.614*	−0.589*
TP	0.323	0.345	0.513*
NH_4-N	0.187	0.286	0.419*
NO_3-N	−0.336	−0.353	−0.427*
NO_2-N	−0.427*	−0.436*	−0.470*
TN	0.676*	0.708*	0.681*
TN:TP ratio	0.129	0.170	0.022

Values with an asterisk are statistically significantly correlated ($p \le 0.05$).
SRP, soluble reactive P; TN, total N; TP, total P.

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Table IV. Standardized regression coefficients and coefficients of determination for forward stepwise regressions ($n = 79$) computed using $\log_{10}$-transformed *in situ* water quality constituents as dependent variables and $\log_{10}$-transformed values of point water depth, pumped flow rate and daily solar radiation as independent variables. Solar radiation was forced into each equation. Values with an asterisk indicate $p \leq 0.05$, and bold font indicates the largest standardized coefficient in each regression. Blank cells indicate that variables were dropped from stepwise regression because of a lack of significance.

In situ water quality constituent	Standardized dimensionless regression coefficients			Coefficient of determination, r^2
	Point water depth	Pumped flow rate[a]	Solar radiation	
Daily maximum temperature	**−1.154***	0.775*	0.077	0.711*
Daily minimum dissolved oxygen	**1.025***	−0.196	0.020	0.828*
Conductivity	**−1.103***	0.779*	0.005	0.684*
Daily maximum pH	**−0.762***		0.106	0.622*
Fluorescent chlorophyll		**−0.764***	−0.022	0.570*
Mean daily turbidity	−0.275	**0.617***	0.262*	0.276*

[a]Average flow rate during 36 h prior to water quality measurement. Transformed using $\log_{10}(Q+1)$. For temperature, the product of river temperature and average flow rate was used.

chlorophyll at site 1 within the backwater (Table 3). SRP was significantly negatively correlated with diel mean, maximum and oscillating *in situ* fluorescent chlorophyll concentrations. In contrast, TP was significantly positively correlated with only oscillating *in situ* fluorescent chlorophyll concentrations. Dissolved inorganic nitrogen as NH_4-N was also significantly positively correlated only with oscillating *in situ* fluorescent chlorophyll, whereas NO_3-N was negatively correlated with the same chlorophyll constituent. Backwater NO_2-N was negatively correlated with diel mean, maximum and oscillating *in situ* chlorophyll measures. In comparison, TN was positively correlated with diel mean, maximum and oscillating *in situ* chlorophyll measures. No significant correlation was observed between TN:TP ratio and chlorophyll.

Forward stepwise multiple linear regressions revealed that point water depth generally was more influential than inflow quantity and quality and the distance from the point of inflow (Tables 4 and 5). Regression coefficients for solar radiation were significantly different from zero only for two constituents (alkalinity and NH_4-N, Table 5), indicating that the large differences in water quality amongst the pre-

Table V. Standardized regression coefficients and adjusted coefficients of determination for forward stepwise regressions computed using $\log_{10}$-transformed water quality constituent as the dependent variable and $\log_{10}$-transformed values of point water depth, pumped flow rate, daily solar radiation and distance from inflow as independent variables. Solar radiation was forced into each equation. Values with an asterisk indicate $p \leq 0.05$, and bold font indicates the largest standardized coefficient in each regression. Blank cells indicate variables were dropped from stepwise regression because of a lack of significance.

Water quality constituent	(*n*)	Standardized dimensionless regression coefficients				Adjusted coefficient of determination, r^2
		Point water depth	Pumped flow rate × river value of constituent[a]	Solar radiation	Distance from inflow	
Secchi visibility	72	**0.593***	0.319*	−0.014	0.332*	0.160*
Turbidity	79	**−0.843***	0.560*	0.022	0.291*	0.280*
TSS	79	**−0.901***	0.529*	−0.036	0.277*	0.337*
TDS	79	**−1.204***	0.699*	−0.080	0.394*	0.595*
Alkalinity	73	**−0.822***	0.175*	−0.140*	0.263*	0.605*
Hardness	73	**−1.143***	0.580*	−0.198	0.460*	0.605*
SRP	79	0.282*	0.289*	−0.178	**−0.349***	0.259*
TP	79	**−0.869***	0.244*	−0.077	0.297*	0.550*
NH_4-N	79	−0.324*	**0.637***	−0.142*		0.605*
NO_3-N	79		**0.362***	−0.205		0.186*
NO_2-N	79		**0.881***	0.019		0.752*
TN	79	**−0.930***		−0.079	0.349*	0.674*
TN:TP	79		**0.400***	0.024		0.140*
Chlorophyll *a*	79		**0.499***	−0.044		0.236*

SRP, soluble reactive P; TN, total N; TP, total P.
[a]Product of the average flow rate during 36 h prior to water quality measurement, Q, and the value of the constituent measured in the river (pumped inflow), C, transformed using $\log_{10}(Q * C + 1)$. Secchi visibility data were not available for the river; average flow rate was used in this regression.

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treatment, treatment and post-treatment periods were primarily due to flooding and flow augmentation rather than seasonal shifts in solar radiation and air temperature. Increasing water depth associated with artificial flooding reduced daily maximum temperature, daily maximum pH and daily mean conductivity and increased daily minimum dissolved oxygen (Table 4). Turbidity was lowered by increasing backwater depth, but elevated by inflow from the river. Fluorescent chlorophyll levels were inversely related to river inflow, perhaps because of dilution, higher turbidity and lowered light penetration. All measures of water clarity, Secchi visibility, turbidity and suspended solids, indicated that water became clearer as depth increased but showed higher levels of solids in the water as artificial flooding increased. Nutrients responded to flooding and impoundment in a complex fashion, indicating a strong association with suspended sediments (TP and TN) or flood water quality (NH_4-N, NO_3-N and NO_2-N). Soluble reactive phosphorus and TN were influenced by the distance between the inflow and the sampled point, indicating that possible in-channel processes within the backwater were modifying nutrient concentrations.

DISCUSSION

Direct effects of artificial flooding rehabilitation during the summer season on water quality within the Coldwater River backwater were significant and, for several parameters, complex (Figure 5), with rapid (hours to minutes) and broad (1.4-fold to 7-fold) changes. All *in situ* parameters examined

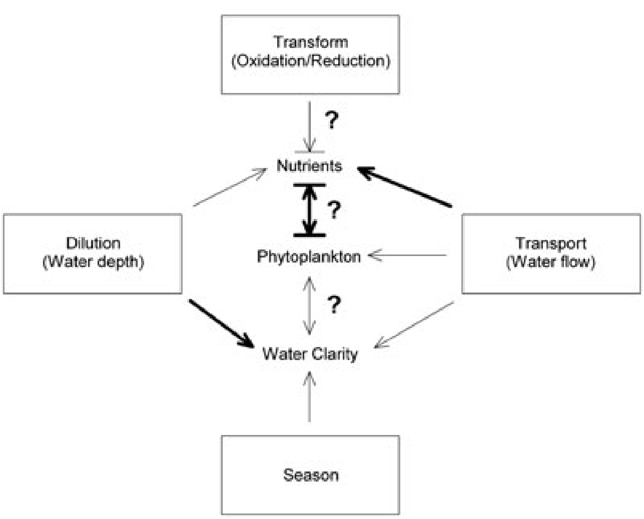

Figure 5. Conceptual model of direct (arrows) and process mediated (arrow-line) effects of artificial flooding on nutrients, phytoplankton and water clarity in a shallow backwater. Thicker arrows represent greater coefficients; question marks represent areas in need of further research.

during pre-treatment exhibited typical responses to lentic (no flow) drought conditions (Lake, 2003). During this period, rapid evaporation during summer months in addition to lower rainfall amounts typical of north Mississippi elicited direct and indirect effects of stagnant water with decreased water volume on water quality. In this study, backwater diel ranges in dissolved oxygen and pH were directly mitigated by increased water depth with flow and seasonality having little to no influence. Large diel ranges in dissolved oxygen and pH can occur in ever increasingly shallow systems as nutrients concentrate, leading to algal blooms (Matthews, 1998). More complex direct effects occurred with backwater diel ranges of temperature and conductivity mitigated by increases in both water depth and water flow due to dilution with influent river water. Wide ranges in temperature, dissolved oxygen and pH such as those observed in this study can stress biota such as fish (Tramer, 1977; Miranda *et al*., 2001; Miranda, 2005) eventually contributing to lethality. These changes occur annually in anthropogenically aggraded backwaters and bayous across this region (Justus, 2006; Hicks and Stocks, 2010) and may reduce the ecological, recreational and aesthetic benefits these systems provide. Artificial flooding during treatment significantly improved and stabilized water quality parameters measured *in situ* within minutes to hours of flow initiation with effects observed until flooding was halted. However, the current study was conducted during the time of year when the greatest stress on aquatic biota is most likely. Further research would need to address whether artificial flooding would be more effective in stabilizing these *in situ* parameters if it was conducted earlier in the spring or summer and possibly continued for a longer duration.

Changes in solids (total dissolved and suspended) and nutrients (phosphorus and nitrogen) varied from 1.4-fold to 7-fold, were initiated minutes to days after initial flow, and often lasted through post-treatment. Direct effects of artificial flooding on solids included decreased concentrations in conjunction with increases in water depth, water flow and downstream distance due to dilution with influent river water and transport downstream. As a result, backwater water clarity improved and buffering capacity (hardness and alkalinity) declined to levels found in river water. Total N and P were directly affected by water depth and downstream distance indicative of dilution with influent river water and transport downstream. Solids and nutrient concentrations during and after artificial flooding were similar to deeper, more stable aquatic systems containing sufficient water volume mitigating the effects of desiccation (Lake, 2003; Knight and Welch, 2004; Pithart *et al*., 2007). Mean total nitrogen and total phosphorus values in the Coldwater River backwater (1.2–3.5 mg TN L^{-1}; 0.147–0.571 mg TP L^{-1}) were comparable with floodplain water bodies within

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the Danube delta floodplain (3.0 mg TN L^{-1}; 0.16 mg TP L^{-1}), Austria (Buijse *et al.*, 2002) and Tualitin River flood plain (0.226–4.517 mg TN L^{-1}; 0.037–0.415 mg TP L^{-1}), Oregon, USA (Weilhoefer *et al.*, 2008). A shallow water body (5–7 m), the Shingu Reservoir in South Korea, had comparable TN concentrations (1.6–3.2 mg TN L^{-1}) but not TP concentrations (0.048–0.126 mg TP L^{-1}) (Kim *et al.*, 2007), showing the variation in these shallow aquatic systems. In contrast, Coldwater River backwater mean total nitrogen and total phosphorus values were greater than those in other floodplain water bodies in an intensively cultivated region along the Tsiza River, Hungary (0.44 mg TN L^{-1}; 0.068 mg TP L^{-1}) (Acreman *et al.*, 2007). Additionally, flooding of Tsiza River floodplain water bodies increased total nitrogen and decreased total phosphorus (Acreman *et al.*, 2007), whereas within the Coldwater River backwater, artificial flooding decreased both total nitrogen and total phosphorus.

Effects of artificial flooding on TN:TP ratios were a direct result of dilution with influent river water as observed by the decrease in ratios from pre-treatment to treatment and association with water flow. Mean floodplain backwater TN:TP ratios, although greater than Coldwater River both before and after treatment, were often less than 20:1 during the study period. This is similar to TN:TP ratios measured in the shallow (5–7 m), hypertrophic Shingu Reservoir in South Korea where TN:TP rations ranged from 13 to 60 (Kim *et al.*, 2007). This range of TN:TP ratios occurring in the Coldwater floodplain backwater suggests a possible coinciding nitrogen and phosphorus limitation in the system (Allen, 1995). Despite these observed changes, TN:TP ratios remained relatively stable in comparison with most other water quality parameters assessed in this study. One possible explanation is the remineralization of nutrients (release of inorganic nutrients by organisms) (Dodds, 2002) allowing for stabilization of these ratios. However, this is beyond the scope of the current study and further research is needed to understand why TN:TP ratios were more resilient than other variables with flooding.

Chlorophyll concentrations have been used as indirect measures of phytoplankton biomass in lentic systems dominated by these organisms (Wetzel, 2001; Scheffer, 2004; Çelik, 2006). Flooding of shallow riverine floodplain backwaters can significantly directly affect chlorophyll concentrations through a flushing effect of transporting algal biomass from the backwater into the mainstem river channel (Figure 5) (Junk *et al.*, 1989; Scheffer, 2004; Ahearn *et al.*, 2006). This has been referred to by Ahearn *et al.* (2006) as 'priming the productivity pump' during flood pulse events. The current study showed a strong negative correlation between mean, maximum and oscillating *in situ* fluorescent chlorophyll concentrations in the Coldwater backwater and water depth, corroborating a flushing of algal biomass in the system during the artificial flooding event. In addition, backwater chlorophyll *a* concentrations showed a positive regression coefficient with water flow further supporting downstream transport of algal biomass. Coldwater River floodplain shallow backwater chlorophyll *a* concentrations ranged widely from <1 to 133.7 μg L^{-1} and were comparable with several other temperate shallow floodplain water bodies (Tockner *et al.*, 1999; Roozen *et al.*, 2003; Kim *et al.*, 2007). Coldwater River backwater *in situ* fluorescent chlorophyll also ranged widely (50.9–188 μg L^{-1}) and was similar to fluorescent chlorophyll concentrations observed by Çelik (2006) in the shallow (mean depth 1.5 m) hypertrophic Lake Manyas, Turkey, ranging from ~70 to 105 μg L^{-1}.

Within shallow water bodies such as floodplain backwaters, indirect effects of artificial flooding that are process-mediated can elicit highly complex responses for several water quality parameters such as nutrients and chlorophyll concentrations (phytoplankton biomass) (Scheffer, 2004; Kim *et al.*, 2007) (Figure 5). In the current study, process-mediated effects on SRP can include the following: (i) nutrient uptake as observed by negative associations with downstream distance and phytoplankton biomass; and (ii) phosphorus release from sediment (Scheffer, 2004) as observed by positive associations with increasing water depth. Process-mediated effects on dissolved inorganic nitrogen can include chemical transformation via changes in oxidation and reduction potential of the backwater environment. Observations of negative associations with NH_4-N and water depth in conjunction with positive associations with NO_3-N and NO_2-N with flow and increased minimum dissolved oxygen levels, suggest backwater conditions became increasingly oxidized under artificial flow, providing an environment conducive to oxidation of NH_4-N to NO_2-N and NO_3-N (Figure 5) (Dodds, 2002).

Process-mediated effects on phytoplankton biomass have, most often, had nutrients as the primary focus. However, other factors such as suspended sediments and macrophyte densities can significantly influence phytoplankton biomass (Scheffer, 2004; Jeppesen *et al.*, 2007; Knight *et al.*, 2008). In the present study, chlorophyll, assessed as mean, maximum and oscillating *in situ* fluorescent chlorophyll concentrations, were associated with both phosphorus (negative correlation with SRP; positive correlation with TP) and nitrogen (as dissolved inorganic nitrogen and TN) but not TN:TP ratios. In the present study, *in situ* fluorescent chlorophyll concentrations in the Coldwater backwater showed a pattern of high concentrations prior to artificial flooding, a decrease in concentrations during the treatment period and a return to pre-treatment concentrations after treatment ceased. However, laboratory-derived chlorophyll *a* concentrations did not show any clear pattern due, in part, to insufficient statistical power from the small sample size and high variability of the data. In comparison, Çelik (2006) observed

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chlorophyll in the shallow hypertrophic Lake Manyas, Turkey, to be positively associated with PO_4, and dissolved inorganic nitrogen NO_3-N and NH_4-N. Kim *et al*. (2007) also observed associations of chlorophyll (as chlorophyll *a*) and nutrients in the shallow Shinga Reservoir, South Korea, where chlorophyll was positively correlated with TP but negatively correlated with TN and positively correlated with TN:TP ratios. Niemistö *et al*. (2008) working in the shallow eutrophic basin, Kirkkojärvi, Finland, also observed associations between chlorophyll (as chlorophyll *a*) and TN:TP ratios, but these associations were seasonally dependent. Correlations between chlorophyll and TN:TP occurred during May but not during July–August. The current study was conducted during July–October, and as a result, these associations may not have been as apparent.

Varying flow regimes within river watersheds and respective floodplains influence physical chemical and biological processes (Killgore and Hoover, 2001). Within shallow (<2 m) riverine backwaters, flow is dependent upon connectivity with the main river channel. The effects of flow and concomitant increases in water level on shallow riverine backwater wetlands include sediments, nutrients and biota influx to and/or efflux from the system that can be sporadic or seasonal. Flushing via reconnection with mainstem channel river water can benefit shallow lentic aquatic systems (Scheffer, 2004). Under these conditions, artificial flooding can transport nutrients, solids and algal biomass downstream and outside the riverine backwater or dilute constituents of eutrophic and hypertrophic conditions. In addition, water depth in lentic shallow (<5–7 m) water bodies in temperate regions exhibits poor thermal stratification for any significant amount of time (months), and as a result, stratification of *in situ* water quality measurements would be very limited or non-existent. Although riverine backwaters can improve mainstem river water quality and ecology with increased connectivity of the river with its floodplain (Acreman *et al*., 2007), the impacts of increased connectivity on riverine floodplain backwaters are not as well understood. In the present study, changes in water depth were associated with stabilization (measured as diel range) of diel temperatures, dissolved oxygen concentrations, conductivity, pH and fluorescent chlorophyll. In addition, increases in water depth were associated with decreases in nutrients such as TN and TP. Results of this study are in good agreement with other studies observing the influence of water depth, flow and volume on the physical, chemical and biological characteristics of shallow floodplain water bodies (Lake, 2003; Roozen *et al*., 2003; Scheffer, 2004; Sommer *et al*., 2004; Çelik, 2006; Novo *et al*., 2006). Also, artificial flooding of the Coldwater floodplain backwater with mainstem channel water increased the similarity of several abiotic variables, specifically dissolved constituents such as TDS, alkalinity, hardness and dissolved inorganic nitrogen. Such responses are typical in floodplain lakes and backwaters during floods, reducing habitat heterogeneity and increasing homogeneity among these systems within a river's floodplain ecosystem (Thomaz *et al*., 2007).

In addition to improving and stabilizing physical and chemical water quality and reducing eutrophication and hypertrophication in riverine floodplain backwaters, artificial flooding increased water depth and habitat. This may allow the backwater to provide refugia for fish and other aquatic organisms, mimicking conditions of highly connective river–floodplain watersheds ultimately resulting in increased ecosystem services. For many riverine species, re-establishment and maintenance of more natural patterns of longitudinal and lateral connectivity of floodplain water bodies with the parent river mainstem channel are essential for the viability of the populations of these species (Bunn and Arthington, 2002). Although natural floodplain backwater ecosystems developed in response to frequent flooding can have high biodiversity, those ecosystems developed under infrequently flooded conditions may not be as well adapted (Acreman *et al*., 2007). Renewed flooding from the Coldwater River into an isolated floodplain backwater, such as the study site, could result in an initial decrease in ecosystem services and water quality before the backwater system adapts to the new flood regime, as observed in the agriculturally impacted Tsiza river floodplain (Acreman *et al*., 2007). In order to better understand the rehabilitative influence of riverine flow on shallow, isolated riverine floodplain backwaters, there is a need to assess the hydrogeological, physical, chemical and biological changes associated with changes in flow and flooding conditions.

CONCLUSIONS

The studied Coldwater River backwater has connections with the mainstem channel river primarily during the wet season (December–May) and rarely during the dry season (June–November). True lotic conditions occur only about 8 days year^{-1}, lasting an average of 2 days (Shields *et al*., 2005). Although connectivity occurs regularly, these connections occur during periods that are least stressful to aquatic biota. Backwater rehabilitation should therefore emphasize improving water quality and habitat conditions during the most stressful time periods whenever possible. The greatest suspected stress on measured aquatic biota occurs during summer and fall when water levels are lowest, and ambient temperatures and autotrophic productivity are highest. The addition of water and flushing via flow augmentation from rapid artificial flooding (within hours to days) improved and stabilized water quality within the backwater, concomitantly improving habitat conditions for fish and aquatic invertebrates. Programmes that manage for

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ecological rehabilitation should include management strategies that target mitigating the most stressful conditions to provide improvements in ecosystem services for these types of aquatic systems. Finally, areas of additional research should include a better understanding of timing (when and how long) of artificial flooding for greatest backwater rehabilitation efficiency and a better understanding of process-mediated effects of flooding on these systems.

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REFERENCES

Acreman MC, Fisher J, Stratford CJ, Mould DJ, Mountford JO. 2007. Hydrological science and wetland restoration: some case studies from Europe. *Hydrology and Earth System Sciences* **11**: 158–169.

Ahearn AS, Viers JH, Mount JF, Dahlgren RA. 2006. Priming the productivity pump: flood pulse driven trends in suspended algal biomass distribution across a restored floodplain. *Freshwater Biology* **51**: 1417–1433.

Allen JD. 1995. Stream Ecology: Structure and Function of Running Waters. Chapman & Hall: New York, NY; 388 pp.

American Public Health Association (APHA). 1998. Standard Methods for the Examination of Water and Wastewater. 20th ed. American Public Health Association (APHA): Washington, DC.

Brookes A, Shields FD Jr. 1996. River Channel Rehabilitation: Guiding Principles for Sustainable Projects. John Wiley & Sons: New York, NY; 433 pp.

Buijse AD, Coops H, Startas M, Jans LH, Van Geest GJ, Grifts RE, Ibelings BW, Oosterberg W, Roozen FCJM. 2002. Rehabilitation strategies for river floodplain along large lowland rivers in Europe. *Freshwater Biology* **47**: 889–907.

Bunn SE, Arthington AH. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management* **30**: 492–507.

Çelik K. 2006. Spatial and seasonal variations in chlorophyll–nutrient relationships in the shallow hypertrophic Lake Manyas, Turkey. *Environmental Monitoring and Assessment* **117**:261–269.

Cooper CM, Knight SS. 1991. Nutrient and sediment trapping efficiency in small farm ponds. *Proceedings Environmentally Sound Agriculture* **2**: 673–680.

Dodds WK. 2002. Freshwater Ecology: Concepts and Environmental Applications. Academic Press: New York, NY; 569 pp.

Gore JA, Shields FD Jr. 1995. Can large rivers be restored? *Bioscience* **45** (3): 142–152.

Hicks MB, Stocks SJ. 2010. Water-quality data for selected streams in the Mississippi Alluvial Plain ecoregion, northwestern Mississippi, September–October 2007. U.S. Geological Survey Data Series Report 493, 118 p.

Jeppesen E, Meerhoff M, Jacobsen BA, Hansen RS, Søndergaard M, Jensen JP, Lauridsen TL, Mazzeo N, Branco CWC. 2007. Restoration of shallow lakes by nutrient control and biomanipulation—the successful strategy varies with lake size and climate. *Hydrobiologia* **581**: 269–285.

Junk WJ, Bayley PB, Sparks RE. 1989. The flood pulse concept in river–floodplain systems. In Proceedings of the International Large River Symposium, **106**, Canadian Special Publication of Fisheries and Aquatic Sciences, Ottawa.

Justus BG. 2006. Water Quality of Eleven Lakes in Eastern and Southern Arkansas from August 2004–July 2005. Scientific Investigations Report 2006–5047, US Geological Survey: Reston VA.

Killgore KJ, Hoover JJ. 2001. Effects of hypoxia on fish assemblages in a vegetated waterbody. *Journal of Aquatic Plant Management* **39**: 40–44.

Kim HS, Hwang SJ, Shin JK, An KG, Yoon CG. 2007. Effects of limiting nutrients and N:P ratios on the phytoplankton growth in a shallow hypertrophic reservoir. *Hydrobiologia* **581**: 255–267.

Kingsford RT. 2000. Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. *Austral Ecology* **25**: 109–127.

Knight SS, Cullum RF, Cooper CM, Lizotte RE. 2008. Effects of suspended sediments on the chlorophyll–phosphorus relationship in oxbow lakes. *International Journal of Ecology and Environmental Sciences* **34**: 1–6.

Knight SS, Welch TD. 2004. Evaluation of watershed management practices on oxbow lake ecology and water quality. In Water Quality Assessments in the Mississippi Delta: Regional Solutions, National Scope, Nett MT, Locke MA, Pennington DA (eds). American Chemical Society: Washington, DC; 284 pp.

Lake PS. 2003. Ecological effects of perturbation by drought in flowing waters. *Freshwater Biology* **48**: 1161–1172.

Locke MA. 2004. Mississippi Delta management systems evaluation area: overview of water quality issues on a watershed scale. In Water Quality Assessments in the Mississippi Delta: Regional Solutions, National Scope, Nett M, Locke M, Pennington D (eds). ACS Symposium series 877. Chicago: American Chemical Society, Oxford University Press; 1–15.

Matthews WJ. 1998. Patterns in Freshwater Fish Ecology. Chapman & Hall: New York, NY; 784 pp.

Miranda LE. 2005. Fish assemblages in oxbow lakes relative to connectivity with the Mississippi River. *Transactions of the American Fisheries Society* **134**(6): 1480–1489.

Miranda LE, Hargreaves JA, Raborn SW. 2001. Predicting and managing risk of unsuitable dissolved oxygen in a eutrophic lake. *Hydrobiologia* **457**: 177–185.

Miranda LE, Lucas GM. 2004. Determinism in fish assemblages of floodplain lakes of the vastly disturbed Mississippi Alluvial Valley. *Transactions of the American Fisheries Society* **133**(2): 358–370.

Niemistö J, Holmroos H, Pekcan-Hekim Z, Horppila J. 2008. Interactions between sediment resuspension and sediment quality decrease the TN:TP ratio in a shallow lake. *Limnology and Oceanography* **53**(6): 2407–2415.

Novo EMLM, Barbosa CCF, Freitas RM, Shimabukuro YE, Melack JM, Filho WP. 2006. Seasonal changes in chlorophyll distributions in Amazon floodplain lakes derived from MODIS images. *Limnology* **7**: 153–161.

Pennington DA. 1993. Augmentation of low flows of the Upper Sunflower River. Proceedings of the Mississippi Water Resources Research Conference. http://www.wrri.msstate.edu/pdf/Pennington93.pdf, accessed 4 Jan 2010.

Pithart D, Pichlová R, Bílý M, Hrbáček J, Novotná K, Pechar L. 2007. Spatial and temporal diversity of small shallow waters in river Lužnice floodplain. *Hydrobiologia* **584**: 265–275.

Roozen FCJM, Van Geest GJ, Ibelings BW, Roijackers R, Scheffer M, Buijse AD. 2003. Lake age and water level affect the turbidity of floodplain lakes along the lower Rhine. *Freshwater Biology* **48**: 519–531.

DOI: 10.1002/rra

Scheffer M. 2004. Ecology of Shallow Lakes. Kluwer Academic Publishers: Dordrecht; 357 pp.

Shields, FD Jr., Knight SS, Stofleth JM. 2005. Alternatives for riverine backwater restoration by manipulation of severed meander bend. In Proceedings of the 2005 World Water and Environmental Resources Congress: Impacts of global climate change, Walton R (ed). American Society of Civil Engineers Proceedings **173**(566), Reston, Virginia.

Shields FD Jr., Smiley PC Jr., Cooper CM. 2002. Design and management of edge-of-field water control structures for ecological benefits. *Journal of Soil and Water Conservation* **57**(3): 151–157.

Snipes CE, Evans LP, Poston DH, Nichols SP. 2004. Agricultural practices of the Mississippi Delta. In Water Quality Assessments in the Mississippi Delta: Regional Solutions, National Scope, Nett M, Locke M, Pennington D (eds). ACS Symposium series 877. Chicago: American Chemical Society, Oxford University Press; 43–60.

Sommer TR, Harrell WC, Solger AM, Tom B, Kimmerer W. 2004. Effects of flow variation on channel and floodplain biota and habitats of the Scramento River, California, USA. *Aquatic Conservation: Marine and Freshwater Ecosystems* **14**: 247–261.

Statistical Package for the Social Sciences (SPSS), Inc. 1997. SigmaStat for Windows version 2.03.

Steel, RGD, JH Torrie, DA Dickey. 1997. Principles and Procedures of Statistics: A Biometrical Approach. McGraw-Hill Companies, Inc.: New York, NY; 665 pp.

Thackston EL, Shields FD Jr., Schroeder PR. 1987. Residence time distributions of shallow basins. *Journal of Environmental Engineering* **113**(6): 1319–1332.

Thomaz SM, Bini LM, Bozelli RL. 2007. Floods increase similarity among aquatic habitats in river–floodplain systems. *Hydrobiologia* **579**: 1–13.

Tockner K, Pennetzdorfer D, Reiner N, Schiemer F, Ward JV. 1999. Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river–floodplain system (Danube, Austria). *Freshwater Biology* **41**: 521–535.

Tramer EJ. 1977. Catastrophic mortality of stream fishes trapped in shrinking pools. *The American Midland Naturalist* **97**: 469–478.

Weilhoefer CL, Pan Y, Eppard S. 2008. The effects of river floodwaters on floodplain wetland water quality and diatom assemblages. *Wetlands* **28**: 473–486.

Wetzel RG. 2001. Limnology: Lake and River Ecosystems. Academic Press: San Diego, CA; 1006 pp.

Wren, DG, Davidson GR, Walker WG, Galicki S. 2008. The evolution of an oxbow lake in the Mississippi alluvial floodplain. *Journal of Soil and Water Conservation* **63**(3): 129–135.

DOI: 10.1002/rra