Sediment phosphorus release at a small impoundment on the Illinois River, Arkansas and Oklahoma, USA

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ARTICLE INFO

ABSTRACT

The Illinois River in northeast Oklahoma and northwest Arkansas has been the focal point of environmental, political and legal debate over elevated P concentrations and loads transported across the Arkansas and Oklahoma border. The Oklahoma Water Resources Board has adopted numeric total P criteria ($0.037\,mg\,P\,L^{-1}$) in Oklahoma's Scenic Rivers, including the Illinois River. The US Geological Survey has reported flow-weighted total P concentrations approximately an order of magnitude greater than the newly adopted criterion. Furthermore, elevated dissolved P concentrations have been traced over 45 river kilometers upstream to municipal wastewater discharges in the headwaters of the Illinois River. The point of regulation regarding the total P criteria will be where the Illinois River flows from Arkansas into Oklahoma; however, a small impoundment exists at the Arkansas and Oklahoma border and the states respectively monitor water quality upstream and downstream from this impoundment. The purpose of this study was to evaluate P release from sediments accumulated in this small impoundment. Sediment P release measured in laboratory incubations was as much as $4\,mg\,P\,m^{-2}\,day^{-1}$ under aerobic conditions and approximately $15\,mg\,P\,m^{-2}\,day^{-1}$ under anaerobic conditions. Sediment equilibrium P concentrations (EPC_0) in laboratory equilibration studies were $0.05–0.20\,mg\,P\,L^{-1}$, which is greater than the total P criteria for this river in Oklahoma. Thus, it is conceivable that P released from bottom sediments in this small impoundment may, in fact, increase dissolved P concentrations in the Illinois River. The effect of internal P cycling at Lake Frances on P concentration in the Illinois River downstream might be greatest in the near future, because municipal discharges have recently reduced effluent P concentrations.

1. Introduction

In the USA, we are often faced with environmental, political and legal conflict between states when streams, rivers and catchments transcend the state borders. These conflicts are likely to increase in the near future from the impending development of nutrient criteria in streams and rivers and from differences between states and respective internal agencies on how these nutrient criteria are developed, implemented and embraced within each state. Differences between states also exist in their beneficial use and water quality monitoring programs, including how states interpret data collected.

As a case in point, the Illinois River Watershed in Arkansas and Oklahoma, USA, has been the focal point of environmen-

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tal controversy over elevated phosphorus (P) concentrations in the water column near the state border. The Oklahoma Water Resources Board (OWRB, 2002) has promulgated a total P (TP) criterion of 0.037 mg L^{-1} in Oklahoma's Scenic Rivers in 2002, including the Illinois River and two tributaries; this TP criterion was based the 75th percentile flow-weighted TP concentration reported in Clarke et al. (2000). Green and Haggard (2001) showed that flow-weighted TP concentrations at the Illinois River in Arkansas near the state border were over 10-fold greater than the TP criterion. Thus, Arkansas could be held liable to meet Oklahoma's TP criterion based on a US Supreme Court ruling that the federal EPA could require upstream states to adhere to downstream states water-quality standards at the state border (Arkansas versus Oklahoma, 503 US 91; http://laws.findlaw.com/us/503/91.html).

Phosphorus concentrations at the Illinois River Drainage Area in northwest Arkansas have been decreasing over time because of reduced effluent P concentrations from municipal discharges in the headwaters (Ekka et al., 2006). However, TP concentrations are still generally greater than the Oklahoma TP criterion, especially during summer base flow when effluent discharge dominates (Haggard, 2005). State agencies at the Arkansas–Oklahoma River Compact Commission reported conflicting trends in P concentrations and loads at the Illinois River during 2002, where P was decreasing in Arkansas and increasing in Oklahoma. One potential confounding factor in the water-quality monitoring programs between states may be that Arkansas monitors the Illinois River upstream of a small impoundment (Lake Frances) and Oklahoma monitors downstream from the spillway.

Impoundments increase sediment deposition, and these sediments were likely derived from upstream within the stream channel and catchment. These deposited sediments may have scavenged dissolved P from the water column in the impoundment when settling (Sonzogni et al., 1982). Benthic sediments from upstream sources within stream channels have been P enriched by effluent discharges over time (Ekka et al., 2006; Haggard et al., 2004). Thus, Lake Frances may have stored P in its benthic sediments, especially during the years when P concentrations were high at the Illinois River. This stored P could be released into the water column when dissolved P concentrations are less the equilibrium P concentration between bottom sediments and the overlying water (EPC$_0$ *sensu* Froelich, 1988) or when oxygen is depleted near the sediment–water interface and anaerobic processes occur. Under anaerobic conditions, P may be released when manganese (Mn) and iron (Fe) minerals are subsequently reduced (Mortimer, 1941, 1942). Sediment P flux to the water column was generally much greater under anaerobic conditions compared to aerobic desorption processes (Holdren and Armstrong, 1980; Moore and Reddy, 1994; Haggard et al., 2005). However, dissolved oxygen depletion in shallow reservoirs is often episodic and likely related to relatively calm weather (Welch and Cooke, 1995) and crashes in algal populations (Kann and Welch, 2005).

Several other mechanisms exist that may enhance internal P cycling in shallow impoundments, some of which are related to the reduction of Mn and Fe compounds. The molar ratio of Fe to P in bottom sediments may influence sediment P release rates, where greater release rates occur with lower

molar ratios (Jensen et al., 1992), and the amount of easily exchangeable P in sediments can influence P release and EPC$_0$ (Ekka et al., 2006). Macrophyte senescence and decomposition can be important P source to the water column, especially if molar Fe to P ratios are low and loosely bound P in sediments is high (Welch and Cooke, 1995). Wind resuspension of bottom sediments enriched with P might facilitate increased sediment P release rates in shallow reservoirs (Søndergaard et al., 1992). Wind mixing can maintain uniform pH and other physico-chemical conditions in the water column of shallow reservoirs, and these factors may increase or decrease P release rates from bottom sediments. Wind and climatic conditions often play a large role in the water quality of shallow impoundments.

The purpose of this investigation was to: (i) evaluate changes in P concentrations at the Illinois River upstream and downstream of Lake Frances using US Geological Survey data from 1997 to 2003; (ii) show differences in P concentrations from data previously collected by Parker et al. (1996) at the Illinois River upstream and downstream of Lake Frances; and (iii) quantify bottom sediment EPC$_0$ and P release rates under aerobic and anaerobic conditions at Lake Frances. The ultimate goal is to elucidate whether sediment P release at Lake Frances could result in increased P concentration at the Illinois River downstream. We hypothesize that bottom sediments at Lake Frances will significantly release P under aerobic or anaerobic conditions, potentially increasing water column P concentrations of the impoundment and the Illinois River downstream.

2. Methods

2.1. Study site description

Lake Frances is (or was) a run of the river small impoundment built in 1931 that was historically used as the municipal water supply for Siloam Springs, Arkansas (Fig. 1). This municipality still pumps some water from Lake Frances or the Illinois River into its drinking water treatment system. Lake Frances characteristics were (prior to 1990) 2.3 km^2 surface area, 1.2 m mean depth, 6.5 m maximum depth, 2.5 km maximum length and a 2–3 days mean hydraulic retention time (Søballe and Threlkeld, 1985). On 4 May 1990, the dam was breached by a high flow event and approximately 1 m was lost from the top of the spillway. Thus, Lake Frances has been reduced in surface area and previous bottom sediments in the impoundment are now part of the flood plain.

Søballe and Threlkeld (1985) have published the most comprehensive study on Lake Frances, providing the majority of background data. This shallow impoundment had TP, chlorophyll *a*, and particulate organic matter (POM) concentrations ranging from 0.29 to 0.31 mg L^{-1}, 4 to 37 μg L^{-1}, and 3.1 to 8.0 mg L^{-1}, respectively. The concentrations of chlorophyll *a* and POM were significantly greater at the outflow compared to the inflow, whereas TP concentrations were similar throughout the shallow reservoir (Søballe and Threlkeld, 1985). Total P concentrations suggest that this shallow impoundment is hypereutrophic, with regard to classic trophic state indices; however, chlorophyll *a* concentrations were within the range for eutrophic conditions and less than half that predicted
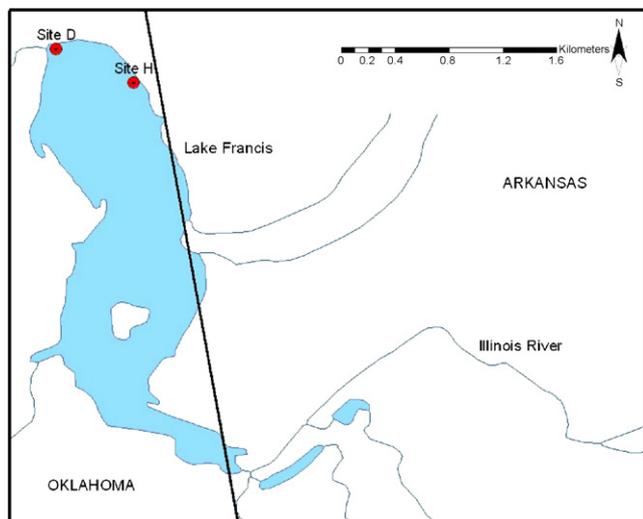
Fig. 1 – Lake Frances (prior to 1990) on the Illinois River near the Arkansas and Oklahoma border with the sediment sampling stations near the dam (Site D) and up-reservoir closer to the headwaters (Site H); Lake Frances was reduced in surface area following the breach of the dam in 1990, and bottom sediments of the shallow impoundment became part of the floodplain, especially upstream from Site H.

by empirical chlorophyll *a* and nutrient models (Søballe and Threlkeld, 1985). In-reservoir processes and hydraulic retention time play a large role in the transformation of nutrients through this eutrophic to hypereutrophic reservoir.

2.2. Stream discharge and phosphorus concentrations

The US Geological Survey (USGS) operates and maintains stream discharge and water-quality monitoring stations upstream and downstream from Lake Frances at the Illinois River; these are the Illinois River South of Siloam Springs, Arkansas (station no. 07195430) and the Illinois River near Watts, Oklahoma (station no. 07195500). The USGS has collected routine water samples every other month and six supplemental storm event water samples per water year. Water samples were analyzed for TP, dissolved P (DP) and PO_4–P at the USGS National Water Quality Laboratory, Lakewood, Colorado; all data are available at http://ar.water.usgs.gov/sun/illinois_basin/. Phosphorus concentrations and loads during base flow and storm event (i.e., surface runoff) conditions were compared using Student's *t*-test on ln-transformed data. Base flow water samples were defined as those collected when base flow discharge accounts for greater than or equal to 70% of mean daily discharge (total stream flow), and surface runoff water samples were defined as those collected when base flow accounts for less than 70% of total stream flow (e.g., see White et al., 2004).

Parker et al. (1996) installed autosamplers at sites upstream and downstream from Lake Frances at the Illinois River and collected water samples during base flow conditions and episodic storm events. The authors did not report any significant difference ($p = 0.059$) between mean TP concentrations at the Illinois River upstream ($0.28 \, mg \, L^{-1}$) and downstream

($0.33 \, mg \, L^{-1}$) from the small impoundment; however, TP concentrations in the water samples were neither ln-transformed nor compared in a pair-wise manner. Of the collected water samples (Parker et al., 1996) 76 could be paired upstream and downstream of Lake Frances and TP concentrations in these water samples were compared using Student's paired *t*-test on ln-transformed data.

2.3. Sediment phosphorus release

One sediment sampling site was selected near the breached dam at Lake Frances and another site was selected up-reservoir near the headwaters of Lake Frances (Fig. 1). Six intact sediment–water columns (hereafter, cores) were collected using a KB^{TM} Corer (Wildlife Supply Company, Buffalo, NY, USA) at each of the two sediment sampling sites. A properly collected core had relatively undisturbed sediment at the surface and with depth, and the overlying water in the core was not excessively turbid from sediment disturbance but relatively clear. Upon return to the laboratory, the depth of overlying water in the core was adjusted, so that each core contained 0.5 L of overlying water. The cores were wrapped in aluminum (Al) foil to exclude light and incubated 13 days at room temperature, approximately 22 °C. The overlying water in all cores was bubbled with air for approximately 24 h, and then three cores were incubated under aerobic conditions and three under anaerobic conditions from each site. Aerobic cores had air bubbled through the overlying water whereas anaerobic cores had nitrogen gas (N_2) with 300 ppm CO_2 bubbled through the overlying water.

A 60 mL water sample was removed from the overlying water in the cores at 1–3 days intervals during the incubation period. This water sample was used to measure pH and conductivity, and a portion of this sample was filtered through a 0.45 μm membrane, acidified using concentrated HCl, and analyzed for soluble reactive P (SRP) using the automated ascorbic acid reduction technique (American Public Health Association, 1998). The overlying water in the cores was maintained at a volume of 0.5 L using filtered (0.45 μm membrane) water that was collected at Lake Frances with a known (i.e., measured) SRP concentration. Sediment P fluxes were calculated at linear changes in the P mass in the overlying water (after corrections for the 60 mL removal of overlying water in the core and replacement with filtered lake water) as a function of time divided by the inside area of the sediment–water cores. Simple linear regression was used to determine the slope of the relation between adjusted SRP mass and time. This slope represented the linear change in SRP mass with time ($mg \, day^{-1}$) and was divided by the core area ($0.0018 \, m^2$) to determine the sediment P flux ($mg \, m^{-2} \, day^{-1}$).

2.4. Sediment phosphorus sorption

Three benthic sediment samples were collected using Ekman Grab Sampler (Wildlife Supply Company, Buffalo, NY) at each of the two sediment sampling stations, and these surficial sediments were used in P sorption isotherms to determine the sediment EPC_0. Sediment EPC_0 is defined as the aqueous phase P concentration at which net P adsorption or desorption by

benthic sediments is zero (Taylor and Kunishi, 1971; Haggard et al., 1999). We added 100 mL of filtered lake water spiked with additional PO_4–P to 20–30 g of wet sediments where the additional PO_4–P rates were 0, 0.5, 1.0, 2.0 and 5.0 mg L^{-1}. For example, if the filtered lake water had an SRP concentration of 0.15 mg L^{-1} then the series of P standards would be 0.15, 0.65, 1.15, 2.15 and 5.15 mg L^{-1}. The sediment slurries were shaken at low speed for 1 h, and then a 15 mL aliquot was filtered though a 0.45 μm membrane for SRP analysis. The remaining sediment slurry was transferred to a pre-weighed Al pan and oven-dried at 80 °C for 48 h to determine dry weight. The amount of P sorbed per dry weight sediment (mg P kg^{-1} dry sediment) was regressed against initial P concentration of the standard solutions (mg L^{-1}), and sediment EPC_0 was estimated as the x-intercept of this plot.

A Langmuir sorption isotherm was also used to evaluate P uptake by benthic sediments at Lake Frances. The previously described series of P standards and additional PO_4–P rates of 10, 25, 50, 100 and 150 mg L^{-1} were used in the lab sorption experiments. The linearized form of the Langmuir equation relates the final SRP concentration (C, mg L^{-1}) in the aqueous phase to the final concentration (C) divided by the amount of P adsorbed by sediments (X/m, mg kg^{-1} dry sediment).

$$\text{Langmuir equation}: \quad \frac{X}{m} = b\frac{KC}{1+KC} \tag{1}$$

$$\text{Langmuir equation}: \quad \frac{C}{X/m} = \frac{1}{Kb} + \frac{C}{b} \tag{2}$$

This linear relation was used to estimate the PO_4–P binding energy (K) and the P sorption maxima (b, mg kg^{-1} dry sediment) of sediments at each site.

3. Results

3.1. Phosphorus concentrations and daily loads

Phosphorus concentrations and daily loads from the 1997–2003 USGS databases were similar (ln-transformed data, t-test, $p > 0.106$) at sites on the Illinois River upstream (USGS station no. 07195430) and downstream (USGS station no. 07195500) from Lake Frances (Figs. 2 and 3). However, data from Parker et al. (1996) when ln-transformed did show that TP concentrations were significantly greater downstream from Lake Frances compared to upstream (ln-transformed, t-test, $p = 0.016$), and 'paired' data from Parker et al. (1996) further showed greater concentrations downstream (ln-transformed, paired t-test, $p = 0.003$). Unfortunately, Parker et al. (1996) did not measure any of the dissolved P fractions.

Using the 1997–2003 USGS databases, we can decipher differences between TP, DP and PO_4–P within and between sites and evaluate P transformation that may occur through Lake Frances. Phosphorus concentration differences were variable during base flow conditions at the upstream (TP > DP = PO_4–P) and downstream (TP > DP > PO_4–P) sites (ln-transformed data, paired t-test). The fraction of TP in the dissolved form was high upstream (94%) and downstream (87%) during base flow conditions at the Illinois River. Phosphorus concentrations were all different (TP > DP > PO_4–P) during surface runoff conditions
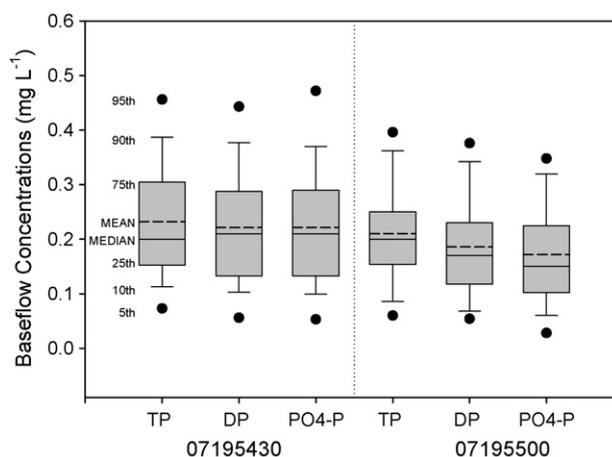

Fig. 2 – Total phosphorus (TP), dissolved phosphorus (DP) and *ortho*-phosphorus (PO_4–P) concentration during base flow conditions at the Illinois River upstream (USGS station no. 07195430) and downstream (USGS station no. 07195500) from Lake Frances, 1997–2003. [Data source was http://ar.water.usgs.gov/sun/illinois_basin/.]

at the two sampling sites (ln-transformed data, paired t-test, $p < 0.025$), and the fraction of TP as DP was 68–69% at the two sites. Only TP concentrations were significantly greater during surface runoff conditions compared to base flow conditions at each site (ln-transformed data, t-test, $p < 0.014$); DP and PO_4–P were not different (ln-transformed data, t-test, $p > 0.383$).

3.2. Sediment phosphorus release rates

Average SRP concentration of the filtered replacement water was 0.17 mg L^{-1}, whereas average SRP concentrations in the overlying water of the cores were 0.02–0.36 mg L^{-1} after air had been bubbled through the overlying water in the cores for 24 h
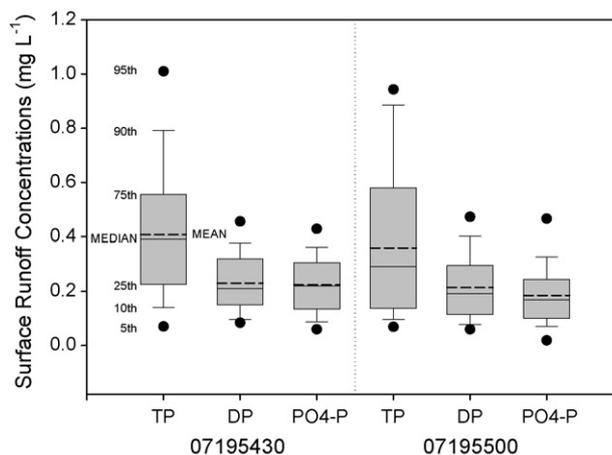

Fig. 3 – Total phosphorus (TP), dissolved phosphorus (DP) and *ortho*-phosphorus (PO_4–P) concentration during surface runoff conditions at the Illinois River upstream (USGS station no. 07195430) and downstream (USGS station no. 07195500) from Lake Frances, 1997–2003. [Data source was http://ar.water.usgs.gov/sun/illinois_basin/.]
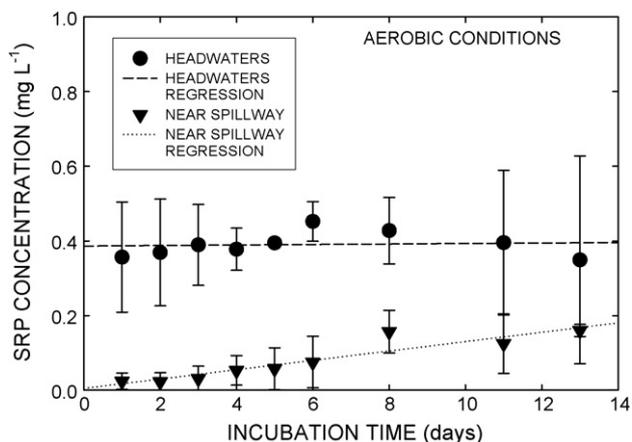
Fig. 4 – Soluble reactive phosphorus (SRP) concentrations in the overlying water of the intact sediment cores from two sites at Lake Frances with incubation time under aerobic conditions. [Symbols are the mean ± standard deviation of the data from the three sediment cores.]
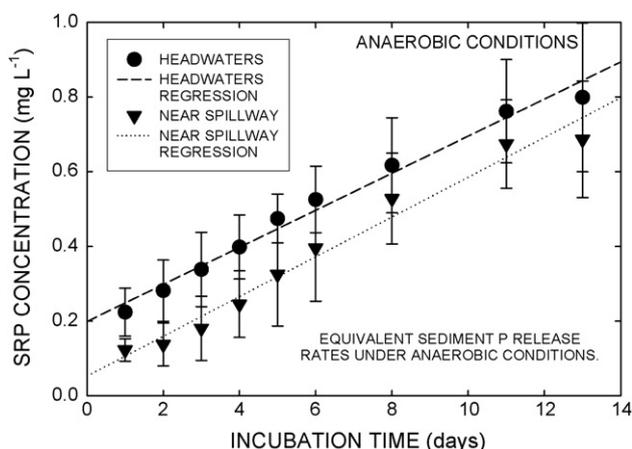

Fig. 5 – Soluble reactive phosphorus (SRP) concentrations in the overlying water of the intact sediment cores from two sites at Lake Frances with incubation time under anaerobic conditions. [Symbols are the mean ± standard deviation of the data from the three sediment cores.]

(Figs. 4 and 5). Average SRP concentrations increased with time under anaerobic conditions at both sites, but this increase was only observed under aerobic conditions in cores collected near the dam. After 13 days of incubation, average SRP concentrations in the overlying core water were 0.16–0.34 mg L^{-1} under aerobic conditions and 0.66–0.77 mg L^{-1} under anaerobic conditions. The greatest variability in SRP concentration was in the overlying water of the sediment cores under aerobic conditions from the headwaters site.

The changes in SRP concentrations in the overlying core water with time produced significantly different estimates of sediment P flux under aerobic and anaerobic conditions (Table 1). The slope of the relation used to estimate average sediment P flux was significantly greater at the two sites under anaerobic conditions compared with aerobic conditions (Dam site: $F = 157$, $p < 0.001$; Headwaters site: $F = 128$, $p < 0.001$). Average sediment P flux was similar between sites under

anaerobic conditions (slope comparison, $F = 0.77$, $p = 0.395$), but average sediment P flux was significantly greater at the site near the dam under aerobic conditions compared to the headwaters (slope comparison, $F = 9.28$, $p = 0.009$). Average sediment P flux under aerobic and anaerobic conditions was 3.6 and 15.5 mg m^{-2} day^{-1} at the dam site and 0.4 and 14.5 mg m^{-2} day^{-1} at the headwaters site. Overall, sediment P flux varied from 9.5 to 22.6 mg m^{-2} day^{-1} under anaerobic conditions in the cores collected from these two sites.

3.3. Sediment EPC$_0$ and phosphorus sorption maxima

The relation between the amount of P sorbed and initial SRP concentration was significant in all extractions ($p < 0.001$). This linear relation produced sediment EPC$_0$ estimates ranging from <0.01 to 0.18 mg L^{-1} at the headwaters and 0.07 to 0.28 mg L^{-1} near the dam (Table 2). Average sediment EPC$_0$

Table 1 – Release rates of soluble reactive phosphorus (SRP) from bottom sediments at Lake Frances using change in phosphorus mass in the overlying water of the sediment cores as a function of time and inside area of the cores

Sampling station	Treatment	Linear regression of P mass as a function of time			Sediment P flux (mg m^{-2} say^{-1})
		R^2	Slope (mg day^{-1})	Intercept (mg)	
Headwaters	Anaerobic	0.969	0.017[a]	0.123	9.5
Headwaters	Anaerobic	0.989	0.041[a]	0.024	22.6
Headwaters	Anaerobic	0.797	0.021[a]	0.149	11.5
Headwaters	Aerobic	0.130	−0.003	0.163	−1.6
Headwaters	Aerobic	0.040	−0.002	0.239	−1.0
Headwaters	Aerobic	0.418	0.007	0.188	3.7
Dam	Anaerobic	0.972	0.033[a]	0.040	18.2
Dam	Anaerobic	0.946	0.023[a]	−0.012	12.8
Dam	Anaerobic	0.969	0.028[a]	0.040	15.6
Dam	Aerobic	0.960	0.006[a]	−0.001	3.3
Dam	Aerobic	0.786	0.008[a]	0.016	4.2
Dam	Aerobic	0.576	0.006[a]	−0.019	3.2

[a] Denotes that the slope was significant at an apriori α of 0.050.

Table 2 – Sediment equilibrium phosphorus concentrations (EPC_0) of bottom sediments at Lake Frances near the dam and headwaters

Sampling station	Simple linear regression of P sorbed and initial aqueous phase SRP concentration		
	R^2	Slope	Sediment EPC_0 ($mg\,L^{-1}$)
Headwaters	0.989	8.65[a]	0.18
Headwaters	1.000	7.01[a]	<0.01
Headwaters	0.997	8.35[a]	<0.01
Dam	0.997	8.75[a]	0.22
Dam	0.996	9.57[a]	0.28
Dam	0.999	7.47[a]	0.07

[a] Denotes that the slope was significant at an apriori α of 0.050.

was 0.06 and 0.20 $mg\,L^{-1}$ at the headwaters and near the dam, respectively. Thus, SRP concentrations in the overlying water in Lake Frances would be expected to be within this range if desorption processes are controlling sediment P release, especially under aerobic conditions.

The sediments from these sites displayed an L-type Langmuir sorption isotherm for SRP. The linearized Langmuir equation was significant using sediments near the dam ($R^2 = 0.978$, $p < 0.001$) and from the headwaters ($R^2 = 0.970$, $p < 0.001$) (Table 2). The average P sorption maxima (*b*) estimated using this isotherm was approximately 460 $mg\,P\,kg^{-1}$ dry sediments near the dam and at the headwaters. The PO_4–P binding energy (*K*) was on average 0.14 and 0.16 at the headwaters and near the dam, respectively. These sorption isotherms were based on 1 h equilibration time, whereas most studies on P adsorption in soils have an equilibration time of about 24 h. Future studies on these sediments should also consider the possibility of two phase P sorption isotherms.

4. Discussion

4.1. Phosphorus concentrations and daily loads

The 'paired' analysis of the data from Parker et al. (1996) did show greater TP concentrations downstream from Lake Frances, but these samples were collected using automated sampling equipment from one point in this river. Water samples collected from a point (i.e., using autosamplers) may show variations in constituents associated with suspended sediment transport (Ging, 1999; Martin et al., 1992), related to velocity profiles across the cross-section of the stream. The USGS has collected water samples at the Illinois River representative of the integrated stream cross-section, but these water samples are not currently 'paired' upstream and downstream from Lake Frances. Future monitoring efforts at the Illinois River should consider collecting water samples representative of the integrated stream cross section that are 'paired' during base flow and surface runoff conditions upstream and downstream of Lake Frances.

Lake Frances likely plays a significant role in the transformation of P through this rapidly flushed impoundment, especially during base flow conditions and warmer climatic conditions. Søballe and Threlkeld (1985) observed that substantial algal growth occurred within Lake Frances, where average chlorophyll *a* concentrations increased from 4 $\mu g\,L^{-1}$ at the

headwaters to 37 $\mu g\,L^{-1}$ near the spillway. Chlorophyll *a* concentrations in Lake Frances also significantly increased with increasing hydraulic retention time (Søballe and Threlkeld, 1985). Thus, algal production within this shallow impoundment would likely increase dissolved P uptake and reduce dissolved fraction of TP. However, we did not have 'paired' water samples to ascertain these differences, and our analysis showed that a large fraction (87% on average) of TP was in the dissolved form at the Illinois River downstream from Lake Frances. Søballe and Threlkeld (1985) observed that more than 70% of TP was in the reactive form, although these observations were prior to the dam failure in 1990.

4.2. Sediment phosphorus release

Bottom sediments in Lake Frances have the ability to release PO_4 into the water column, and this buffering mechanism (i.e., sediment EPC_0) would likely maintain concentrations between 0.05 to 0.20 $mg\,SRP\,L^{-1}$ based on our lab experiments. However, the intact sediment cores from the headwaters incubated under aerobic conditions maintained an SRP concentration of approximately 0.40 $mg\,L^{-1}$ in the overlying water. Sediment EPC_0 in Lake Frances was within the range of values observed in sediments downstream from WWTP effluent discharges at headwater streams of the Illinois River (0.10–7.00 $mg\,L^{-1}$; Haggard et al., 2004; Ekka et al., 2006). Municipal effluent discharges generally have a more substantial effect on sediment EPC_0 and P bioavailability than agricultural land use does within regional catchments (Popova et al., 2006). These effluent discharges contribute a large portion of the annual TP load in this system, and P-rich sediments have been deposited over many years in this impoundment.

The legacy P present in the bottom sediments may explain the high P release rates observed under anaerobic and even aerobic conditions. Sediment P flux (0.4–3.6 $mg\,m^{-2}\,day^{-1}$) under aerobic conditions in cores from Lake Frances rivaled that observed under anaerobic conditions in many eutrophic reservoirs (2.0–6.5 $mg\,m^{-2}\,day^{-1}$: Cooke et al., 1977; Haggard et al., 2005; Malecki et al., 2004; Moore and Reddy, 1994). Sediment P flux (~15 $mg\,m^{-2}\,day^{-1}$) under anaerobic conditions was comparable to that reported in hypereutrophic systems worldwide (e.g., see Kim et al., 2003). Thus, bottom sediments in Lake Frances have the potential to release high amounts of P and also to maintain P concentrations downstream at the Illinois River elevated above Oklahoma's Scenic River TP criterion (0.037 $mg\,L^{-1}$). Other studies have shown that bottom

sediments in rivers and shallow impoundments are an important P source, especially during summer low-flow conditions (e.g., see Banaszuk and Wysocka-Czubaszek, 2005; Welch and Cooke, 1995).

4.3. The Illinois River and Lake Frances

The question remains as to what function Lake Frances will play in P retention and transport within the Illinois River. We know that Lake Frances will provide the opportunity for algal P uptake and growth, thereby transforming dissolved P into particulate P (Søballe and Threlkeld, 1985). This study showed the potential for bottom sediments in Lake Frances to increase P transport at the Illinois River, especially if water column dissolved P concentrations upstream from Lake Frances decrease. Municipal WWTP discharging effluent into the Illinois River and its tributaries have adopted a $1\,mg\,L^{-1}$ limit on effluent P concentrations, and these management changes have and will continue to result in reduced P concentrations and transport at the Illinois River (see Haggard, 2005). It is possible that remediation strategies should be considered for Lake Frances and the P-rich sediments stored within this former impoundment, if Oklahoma's Scenic River TP criterion will be achieved.

The spillway could be rebuilt to even a higher elevation in order to increase potential sediment deposition and its ability to adsorb dissolved P from the water column. Although higher surface water elevation will expose more P-rich sediment to potential anaerobic conditions, these sediments could be treated with aluminum sulfate to reduce sediment EPC_0 (Haggard et al., 2004) and sediment P release rates under anaerobic conditions (Kennedy and Cooke, 1982; Welch and Schrieve, 1994; Haggard et al., 2005). Chemical treatment would need to be periodic as additional sediment and associated-P from the catchment settle to the bottom of Lake Frances. This impoundment was once a well-known fishery and several recreational ecological services could be returned.

The alternative may be removing the spillway and returning the Illinois River to its pseudo-natural state. This action will ultimately transport P-rich sediments further downstream in the Illinois River and potentially to the receiving reservoir, Lake Tenkiller Ferry, Oklahoma. At this point, sediment P release via equilibrium concentration under aerobic conditions or mineral reduction under anaerobic conditions becomes a serious concern for these downstream water resources. The small impoundment could also be left as is, but chemical remediation of the sediments to reduce P release should still be considered, as well as chemical remediation of the flood plain and wetland soils (e.g., see Ann et al., 1999; Gilbert et al., 2003).

Acknowledgements

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