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Phosphorus Release from Bottom Sediments at Lake Wister, Oklahoma, Summer 2010

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Phosphorus Release from Bottom Sediments at Lake Wister, Oklahoma, Summer 2010

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A previous reservoir model of Lake Wister, Oklahoma suggested that internal P sources were dominant, and that a watershed management plan need not be developed to address external sources. The objectives of this study were to collect intact sediment cores from three sites at Lake Wister and measure sediment O_2 demand (SOD) and soluble reactive P (SRP) release to the overlying water during lab incubations under quiescent conditions. The measured SOD rates were between 9.9 and 22.6 mg m^{-2} hr^{-1} on average across the three sites, where the shallow headwaters site had the least SOD. The SRP release rates were estimated to vary from less than zero to 3.30 mg m^{-2} d^{-1} on average across the sites, and release rates were greatest under anaerobic conditions in the cores collected from deep waters near the dam and water intake structure. These measured values were an order of magnitude less than used in the calibration of the previous reservoir model, suggesting that internal P cycling was likely not the dominant P source; thus, it might be premature to neglect external P sources to Lake Wister. A watershed-based strategy that focuses on internal and external P sources is needed for this drinking water supply reservoir to improve overall water quality.

KEY WORDS: Phosphorus, Sediment, Reservoir, Oklahoma, Poteau Valley Improvement Authority

INTRODUCTION

Pathogens, metals, nutrients and sediment are generally the four main causes of beneficial use impairment across surface waters in the U.S., based upon the Environmental Protection Agency's Total Maximum Daily Load Program assessment (EPA, 2009). Nutrients might represent the causal factor with the most complex relation to various ecosystem goods and services like designated beneficial uses, including primary and secondary contact recreation, aquatic communities, and even drinking water supply (Dodds et al., 2009). The potential impacts of elevated nutrients can be expressed at different trophic levels, resulting in shifts in algae, macroinvertebrates and fish assemblage indices (e.g., see Evans-White et al., 2009; Justus et al., 2010) in streams and rivers. However, nutrients and algae (phytoplankton production measured as chlorophyll-a concentration) often show a stronger relation in lakes and reservoirs than that observed in streams. Nutrients from external and internal sources supply the demand for phytoplankton growth in lakes and reservoirs, which sequesters inorganic carbon into the water column and eventually the bottom sediments.

External sources of nutrients focus on effluent discharges from agricultural, industrial, and municipal facilities, and on diffuse sources from the landscape such as runoff from agricultural fields and urban development. The loading of nutrients, especially P, from external sources to lakes and reservoirs is not only a concern regarding phytoplankton production (e.g., Dillon and Rigler, 1974; Canfield and Bachman, 1981), but also internal nutrient cycling. Nutrients entering lakes and reservoirs from the watershed settle to the bottom, especially as sedimentation scavenges reactive nutrients like soluble P from the water column (Sognozi et al., 1982; Correll, 1998). The nutrients taken up through phytoplankton production also reach the bottom, when phytoplankton settle. These nutrients that settle to the bottom sediments can be released back into the overlying water through several processes, including reductive dissolution (Mortimer, 1941), organic matter mineralization (Lovley and Phillips, 1986; Andersen and Jensen, 1992), and equilibrium concentration gradients (Haggard and Sharpley, 2007).

Reductive dissolution of Fe and Mn minerals in sediments is generally considered to be the largest source of internal P release when the overlying waters become anoxic. The P bound to these minerals is released into the overlying waters, when Fe and Mn are used as electron acceptors and reduced. This phenomenon has been studied world-wide and over the last several decades (e.g., see Mortimer, 1941, Cooke et al., 1977, Riley and Prepas, 1984; Moore and Reddy, 1994; Haggard et al., 2005), showing that greater P release occurs with anaerobic conditions. Release rates of P from bottom sediments under anaerobic conditions varies from less than 1 mg m^{-2} d^{-1} in relatively oligo- to mesotrophic systems (Sen et al., 2007) to more than 15 mg m^{-2} d^{-1} in highly eutrophic systems (e.g., Haggard and Soerens, 2006). Sediment-P release under aerobic conditions is probably regulated by mineralization rates (Andersen and Jensen, 1992), and the equilibrium P concentration of the bottom sediments (Haggard and Soerens, 2006; Spears et al., 2007). However, organic matter mineralization under anaerobic conditions can also be a significant source of P release from bottom sediments.

The overall goal of this project was to evaluate the potential for bottom sediments at Lake Wister, Oklahoma to release P when the hypolimnion became anoxic. The specific objectives were to (1) measure the O_2 demand of bottom sediments (sediment O_2 demand, SOD) using flow through cores, and (2) quantify P release rates from bottom sediments using static cores incubated under aerobic and anaerobic con-

ditions. The typical P release rates from bottom sediments in regional reservoirs are typically less than that from external sources (Haggard et al., 2005; Sen et al., 2007), providing only up to 25% of the total P inputs on an average, annual basis. However, sediments offer a management point from which P release to overlying water can be reduced by chemical treatment or oxygenation of the overlying waters (e.g., see Kennedy and Cooke, 1982; Welch and Schrieve, 1994).

STUDY SITE DESCRIPTIONS

Lake Wister is a drinking water supply reservoir in east-central Oklahoma with two primary inflows, the Poteau River and Fourche Maline Creek (Figure 1). The U.S. Army Corp of Engineers (USACE) began reservoir and dam construction in 1946, and its flood control operations fully started in 1949. The Poteau Valley Improvement Authority (PVIA) uses raw water from Lake Wister to provide drinking water to area residents (~40,000 residents in LeFlore County, Oklahoma), and the raw water for drinking water treatment is pulled from the 60 ha inlet or cove on the north-eastern corner of this impoundment. The drinking water produced by the PVIA meets regulations and guidelines established by the Safe Drinking Water Act; however, PVIA has recently become more concerned over total organic carbon (TOC) concentrations in the raw water and its potential to impact the production of disinfection byproducts (DBPs) in the finished drinking water. The concentrations of TOC in raw water have been linked to DBPs, and when TOC concentrations exceed 2 mg L^{-1} in the raw water, Stage 1 and 2 DBP Rules must be implemented. The source of TOC in Lake Wister

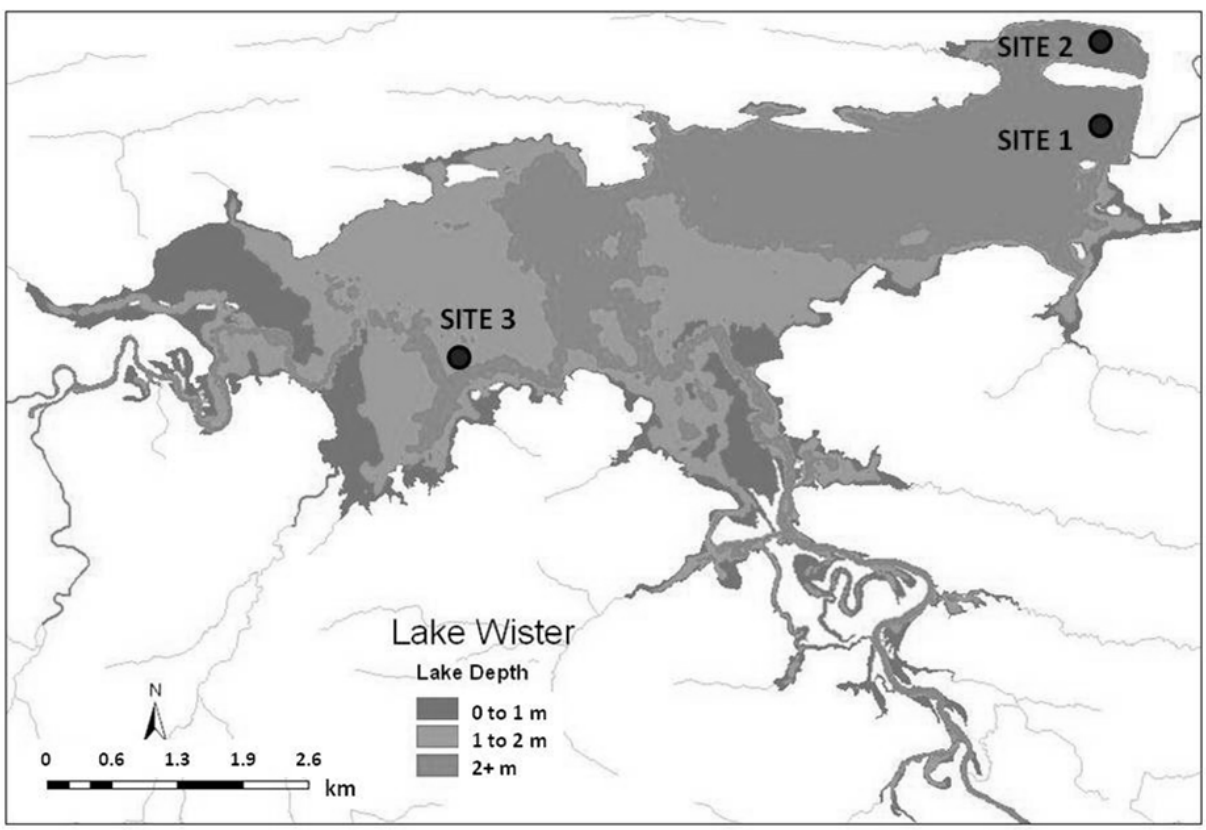

Figure 1. Lake Wister showing lake depths and sediment sampling sites where intact sediment cores were collected in June 2010 (adapted from PVIA, 2009).

is thought to be autochthonous production (i.e., phytoplankton), which is linked to nutrient supply from external and internal sources. Lake Wister is important not only for water supply, but it provides recreation, fishing and water fowl hunting opportunities for local residents and tourists. Elevated sediments and nutrients entering Lake Wister from its watershed have been identified as an important water quality issue, reducing its value for recreational opportunities and as a water supply. The reservoir is listed by the Oklahoma Department of Environmental Quality (ODEQ, 2008) for not supporting these designated beneficial uses, including public or private water supply, warm water aquatic community, and aesthetics based on the 2008 303(d) list. Lake Wister remained on the 2010 303(d) list as not supporting these designated beneficial uses, plus primary body contact recreation (ODEQ, 2010a).

Since 1991, PVIA, the Oklahoma Water Resources Board (OWRB), and USACE have been evaluating the nature of the water quality issues in the reservoir, and working to implement projects to improve water quality. In late 2009, PVIA completed its strategic planning process for Lake Wister with the purpose to accomplish four things, (1) review issues critical to reservoir restoration and management, (2) review opportunities for restoration and enhancement, (3) develop a strategy to establish priorities and next steps, and (4) document discussions and decisions (summarized from PVIA, 2009). The purpose of this study was to fill a perceived knowledge gap established by the PVIA, regarding the P release rates of bottom sediments at Lake Wister. The OWRB (2003) suggested that internal nutrient loading will increase with increases in hypolimnetic anoxia at Lake Wister, and to date there have been no laboratory measures of sediment-P release rates from this impoundment.

This project focused on internal P release at Lake Wister (Figure 1), where intact sediment cores were collected at three locations in July 2010 (Site 1: 34° 56' 33.8424", -94° 43' 24.0672"; Site 2: 34° 56' 45.8334", -94° 43' 12.4752”; and Site 3: 34° 55' 10.3038", -94° 47' 20.7918"). Site 1 was located in the deepest part of the channel near the dam, and Site 2 was in the deepest part of the cove near the PVIA water intake structure; whereas, Site 3 was located in the headwaters where water depths were much less (2 m or less). Lake Wister generally shows gradients in turbidity and other measures of water quality from the headwaters down-reservoir to the outlet (OWRB, 2003); turbidity is usually greatest in the headwaters, and decreases toward the dam and the water intake structure.

MATERIALS AND METHODS

Six intact sediment cores were collected at each site using a modified HYPOX corer (see Figure 2; Gardner et al., 2009) and 0.5 m Plexiglas tubes with a 6.35 cm inside diameter for the sediment-P release incubations. The tubes were inserted into the sediment approximately 25 cm using a weighted release mechanism on the coring device, and then the tubes were sealed at each end manually using number 13 rubber stoppers. A properly collected core had a definitive sediment-water interface with no more turbid overlying water than what was observed visually in the reservoir; the collected cores represented sediment-water interfaces with minimal disturbance. Upon return to the lab, these cores have the water volume reduced to 0.8 L above the sediment interface, and then each core was wrapped in Al foil to exclude light. The overlying water in all cores were brought up to dissolved O_2 saturation by aerating over night, and the cores were incubated at room temperature (~21°C) under aerobic and anaerobic conditions. Three cores had air bubbled through the water overlying the sediments (providing aerobic conditions), and three cores had N_2 gas with 270 ppm CO_2 bubbled through the overlying water (pro-

Figure 2. Modified HYPOX (see Gardner et al., 2009) sampler used to collect sediment cores from Lake Wister for sediment O2 demand and P release rates, as measured in the laboratory.

moting anaerobic conditions). The bubble rates were sufficient to maintain dissolved O_2 near saturation, and or to reduce O_2 in the head-space above the overlying waters in the intact sediment cores. After the first few days the cores intended to maintain at anaerobic conditions were bubbled with the N_2 gas immediately after water was collected and replaced, and then sealed until the next sampling interval.

Water samples (~0.05 L) were collected at 1 to 3 d intervals for the 10 d incubation, and the removed water was replaced with reservoir water having a known P concentration. The water samples were filtered through a 0.5 µm glass-fiber filter, acidified to pH less than 2 using concentrated HCl, and analyzed for soluble reactive P (SRP). SRP was measured using the automated ascorbic acid reduction technique following the standard operating procedures (SOPs) and quality assurance plan (QAP) of the Arkansas Water Resources Center Water Quality Lab; this includes blank, duplicate and spike analyses. Sediment release rates were estimated using the increase in P accumulated in the overlying water, which was corrected for SRP removed during sampling and added during water replacement. The amount of SRP released into the overlying water was normalized based upon initial SRP mass in the overlying water, and then the SRP mass (mg) released over time (d) was used to estimate sediment P flux (mg m^{-2} d^{-1}) from the surface area within the cores (0.0032 m^2). Simple linear regression relating SRP mass released over time was used to estimate a sediment P flux from individual cores, and an average flux based upon all data from a site under each treatment (e.g., see Haggard et al., 2005).

For SOD measurements, four intact sediment cores were collected at each site using the modified apparatus and 0.3 m Plexiglas tubes with a 10 cm inside diameter. The inflow water was saturated with dissolved O_2, and then pumped into the cores at 0.03 L hr^{-1} during the incubation at 21°C and in the dark. The inflow and outflow water collected and analyzed for dissolved O_2 to Ar ratios using a membrane inlet mass spectrometer (MIMS). The expected dissolved Ar concentration (mg L^{-1}) was calculated at 21°C and multiplied by the O_2 to Ar ratio to get the dissolved O_2 concentration in each water sample. The difference in dissolved O_2 concentrations between the inflow (~8.5 mg L^{-1}) and outflow were used to estimate the SOD (mg m^{-2} hr^{-1}), based on the flow rate (0.03 L hr^{-1}) and sediment area within the cores (0.008 m^2). The inflow water was aerated for all sites, so the SOD rates represent a potential rate of areal O_2 loss across sites.

The SOD rates were transformed using natural log to account for the typical log-distribution of water quality data, and then the means were

separated using analysis of variance and least significant difference (LSD, $p<0.05$). The sediment-P release rates were compared across sites by comparing the slopes (p, 0.05) estimated by the simple linear regression of SRP mass released over time (using all data normalized for initial P mass in the overlying water of each intact sediment core). The slopes were compared pair-wise across these three sites to determine significant differences between SRP flux under aerobic and anaerobic conditions at sites where sediment cores were collected.

RESULTS

Phosphorus Concentrations in the Overlying Water

Initial SRP concentrations measured in the overlying water of the intact sediment cores were variable after the water column was aerated over night, which was a surprising observation. The SRP concentration ranged from 0.025 to 0.162 mg L^{-1} in the overlying water on the first day samples (1 d) were collected. However, the majority (12 out of 15) of the sediment cores had SRP concentrations in the overlying water ranging from 0.025 to 0.075 mg L^{-1}, and only three cores had concentrations of 0.109 mg L^{-1} or greater in the water column. Thus, SRP mass in the overlying water of each individual core at day 1 was used to normalize SRP mass measured in the water column throughout the incubation.

In general, SRP concentrations in the water column increased from the first sampling through the ten day incubation in the cores collected from all sites and under both aerobic and anaerobic conditions. SRP concentrations in the overlying water of aerated cores increased to a maximum of 0.182 mg L^{-1} over the ten day incubation; this was only observed in one core from Site 2, whereas maximum SRP concentration was generally less than 0.100 mg L^{-1} in all other aerated cores. Maximum SRP concentration in the overlying water of the cores bubbled with N_2 gas was 0.228 mg L^{-1} after ten days, and it was generally above 0.150 mg L^{-1} in the overlying water of all cores at the end of the incubation and treatment, except those from Site 3. SRP concentrations in the overlying water of the cores incubated under anaerobic conditions from Site 3 showed a slight decrease over time (from 0.044 to 0.037 mg L^{-1} on average), which contrasted the increase observed under aerobic conditions (from 0.064 to 0.093 mg L^{-1}).

Phosphorus Release Rates from Bottom Sediments

The initial SRP mass in the overlying water of each core was used to normalize the SRP mass released over time and estimate release rates from the bottom sediments. The change in SRP mass over the first 24 hours of incubation was variable between treatments and sites, ranging from -0.003 to 0.021 mg SRP. Within 48 hours, the change in SRP mass was greater than zero in all cores, except those collected from Site 3 in the shallow headwaters. The end of the incubation showed that change in SRP mass ranged between 0.025 and 0.026 mg under aerobic conditions at all sites, whereas the change in SRP mass under anaerobic conditions was 0.049, 0.090, and -0.005 mg SRP for Sites 1, 2 and 3, respectively. SRP mass accumulated in the water column over time was used to estimate release rates.

The slopes from the simple linear regression of normalized SRP mass over time were not significantly different than zero ($P>0.05$) under aerobic conditions in the cores collected from deep water (Sites 1 and 2; Table 1, Figures 3 and 4). Time during the incubation explained 42% or less of the variability in mean SRP mass accumulated in the overlying water, and mean release rates were estimated to be 0.75 and 1.13 mg m^{-2} d^{-1} from bottom sediments at Sites 1 and 2, when the overlying water would be

Table 1. Mean soluble reactive phosphorus (SRP) release rates or flux ((mg m^{-2} d^{-1}) from intact sediment cores collected on 14 July 2010 from three sites on Lake Wister, Oklahoma.

Site	Treatment	Slope (mg d^{-1})	R^2	P Value	Mean Flux* (mg m^{-2} d^{-1})
1	Aerobic	0.0024	0.41	0.17	0.75
1	Anaerobic	0.0048	0.78	0.02	1.52
2	Aerobic	0.0036	0.42	0.16	1.13
2	Anaerobic	0.0104	0.98	<0.01	3.30
3	Aerobic	0.0030	0.75	0.03	0.94
3	Anaerobic	-0.0007	0.42	0.16	-0.23

oxygenated. The individual cores from these sites had SRP release rates ranging from less than zero to over 1 mg m^{-2} d^{-1} under aerobic conditions. The regression slope was significant (Site 3; 0.0030 mg d^{-1}, R^2=0.75, *P*=0.03) in the aerated cores collected from the headwaters of Lake Wister (Figure 5). Thus, SRP was significantly released in these cores under aerobic conditions at 0.94 mg m^{-2} d^{-1} on average.

Mass release rates calculated from the regression slopes showed that significant SRP release occurred under anaerobic conditions in the cores collected from deep water (Sites 1 and 2), ranging from 1.52 to 3.30 mg m^{-2} d^{-1} (Table 1, Figure 3 and 4). SRP release rates were estimated to be greatest in the cove with the raw water intake (Site 2, Lake Wister), ranging from 2.95 to 3.75 mg m^{-2} d^{-1} across the three cores incubated under anaerobic conditions. Whereas, SRP release rates near the dam (Site 1) was fairly consistent across the cores under anaerobic conditions (1.51–1.52 mg m^{-2} d^{-1}). In contrast, the slope of mean SRP mass accumulated over time in the overlying water of the cores from Site 3 was not significantly different than zero (-0.0007 mg d^{-1}, R^2=0.42, *P*=0.16) under anaerobic conditions (Figure 5).

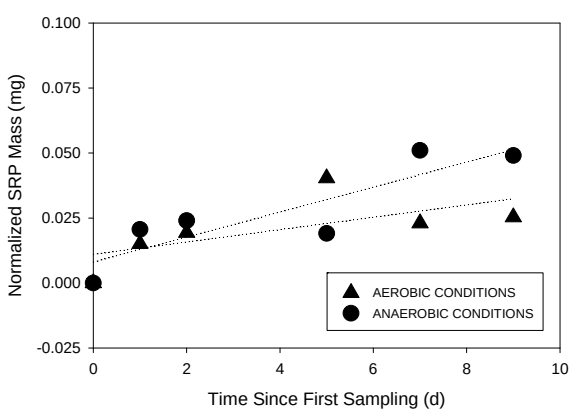

Figure 3. Mean soluble reactive phosphorus (SRP) mass as a function of time near the dam (Site 1) at Lake Wister, Oklahoma; data represents SRP mass released normalized to initial mass for three intact sediment cores, and the dotted lines are to visualize the general relation.

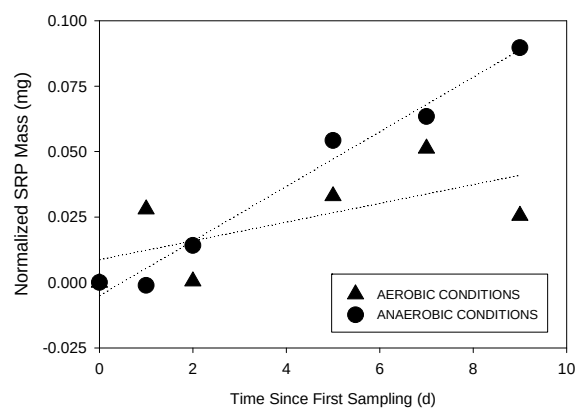

Figure 4. Mean soluble reactive phosphorus (SRP) mass as a function of time within the cove containing the raw water intake (Site 2) for the Poteau Valley Improvement Authority (PVIA) at Lake Wister, Oklahoma; data represents SRP mass released normalized to initial mass for three intact sediment cores, and the dotted lines are to visualize the general relation.

Table 2. Potential sediment oxygen demand (SOD) rates or flux (mg m^{-2} hr^{-1}) from intact sediment cores collected on 14 July 2010 from three sites at Lake Wister, Oklahoma.

Site	Observations	Mean SOD (mg m^{-2} hr^{-1})	SOD Range (mg m^{-2} hr^{-1})
1	4	22.6	18.1 – 27.6
2	4	22.5	16.2 – 26.8
3	4	9.9	9.4 – 11.2

Sediment Oxygen Demand

The water used to flow through the cores for potential had an initial DO concentration of 8.5 mg L^{-1} for the cores collected from Site 1 and 2, and 8.4 mg L^{-1} for those collected from Site 3. DO concentrations in the outflow water were variable between the cores collected at deep sites (Site 1 and 2) and those collected at the shallow site in the headwaters (Site 3). The outflow from the cores collected at Site 1 and 2 had DO concentrations ranging from 1.1 to 4.1 mg L^{-1}, whereas the outflow DO concentration from Site 3 cores was between 5.4 and 5.8 mg L^{-1}. The differences in outflow DO concentrations translated into differences in potential SOD (Table 2), where SOD varied from just over 22 mg m^{-2} hr^{-1} on average from the cores collected in deep water to less than 10 mg m^{-2} hr^{-1} on average from cores from the shallow headwaters.

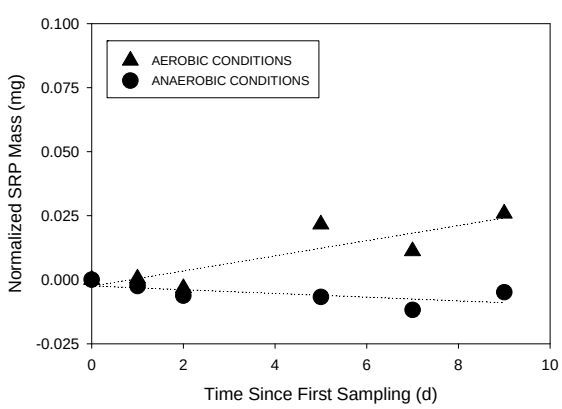

Figure 5. Mean soluble reactive phosphorus (SRP) mass as a function of time in the headwaters (Site 3) at Lake Wister, Oklahoma; data represents SRP mass released normalized to initial mass for three intact sediment cores, and the dotted lines are to visualize the general relation.

Discussion

Lake Wister has been listed on ODEQ's 303(d) list, which resulted in a landscape and reservoir modeling effort to identify P sources within its watershed (AMEC, 2008). One of the conclusions based upon these modeling efforts was that 'with the internal load as the dominate pollutant source, this modeling project could not provide a reduction goal to draft a watershed management plan' (ODEQ, 2010b). The reservoir model included sediment-P interactions, and this parameter representing P release rates from bottom sediment was increased from zero until the calibration improved. The release rates finally used in model parameterization were 30 mg m^{-2} d^{-1} in zone 1, representing the deep water (i.e., near dam and the PVIA raw water intake structure), and 50 mg m^{-2} d^{-1} in zone 2, representing the shallow waters (e.g., headwaters and shorelines) (AMEC, 2008). Furthermore, the SOD rates used in the model were approximately 150 and 200 mg m^{-2} hr^{-1} in zones 1 and 2, respectively. These parameters were not measured, but used a calibration tools when fitting the observed data.

This study measured potential P release rates and O_2 demand from bottom sediments at Lake Wister, and the measured rates differed quite a

Table 3. Estimated soluble reactive phosphorus (SRP) release rates or flux (mg m^{-2} d^{-1}) from intact sediment cores collected during summer at other regional reservoirs in or near Oklahoma.

Reservoir	Site Description	Aerobic Flux (mg m^{-2} d^{-1})	Anaerobic Flux (mg m^{-2} d^{-1})	Reference
Beaver Lake, Arkansas	Riverine Zone	0.13	0.85	Sen et al. (2007)
	Transition Zone	0.15	1.77	
	Lacustrine Zone	0.04	<0.01	
Lake Eucha, Oklahoma	Riverine Zone	1.14	4.70	Haggard et al. (2005)
	Transition Zone	1.01	2.46	
	Transition Zone	0.95	6.05	
Lake Frances, Arkansas – Oklahoma	Headwaters	0.37	14.53	Haggard and Soerens (2006)

bit from those used to parameterize the reservoir model. First, potential SOD measured from the collected cores (9.9–27.6 mg m^{-2} hr^{-1}) was an order of magnitude less than that used in model parameterization (150–200 mg m^{-2} hr^{-1}; AMEC, 2008). For comparison, potential SOD rates at Beaver Lake, Arkansas were estimated to be from 8.8 to 28.5 mg m^{-2} hr^{-1} using this same method (Hamdan et al., 2010). There are various ways to measure potential SOD rates of bottom sediments including lab and *in situ* methods, and lab methods are generally more consistent, reproducible and efficient than in situ methods (Bowman and Delfino, 1980). This study used lab methods, which produced repeatable results (n=4) based upon the potential SOD range at each site. These SOD rates were measured under quiescent conditions in the lab, and Beutel et al. (2007) observed a fivefold increase in SOD under turbulent conditions (i.e., mixing water column and disturbance at the sediment-water interface). These differences suggest that SOD parameterization needs to be able to have values reflective of the reservoir conditions, i.e. quiescent verse turbulent. Overall, SOD is an important component of reservoir sediment dynamics and modeling because it drives the reducing conditions promoting P release from bottom sediments.

The P release rates used in model parameterization for Lake Wister (30 to 50 mg $^{-2}$ d^{-1}) were much greater than that measured in this study using intact sediment cores (near zero to 3.30 mg m^{-2} d^{-1}). SRP release rates measured at Lake Wister were within the range reported for other regional reservoirs (<0.01 to 15.53 mg m^{-2} d^{-1}), which were considered to express meso- to eutrophic conditions (Table 3). Perhaps, the best comparison is to Lake Eucha, Oklahoma where SRP release rates from bottom sediments were estimated to be ~1 mg m^{-2} d^{-1} under aerobic conditions and from 2.46 to 6.05 mg m^{-2} d^{-1} under anaerobic conditions (Table 3; Haggard et al., 2005). The P release rates measured were an order of magnitude less than the value used in the reservoir model calibration, suggesting that internal P cycling might not be the dominant source to Lake Wister. Thus, it might be premature to neglect catchment P sources to Lake Wister, including both the effluent discharges and diffuse sources from the landscape.

The reservoir model was parameterized to match data available from the 2001, where total P (TP) concentrations were available from the surface and bottom of the water column at sites near the dam and PVIA raw water intake structure (AMEC, 2008; ODEQ, 2010b). In 2001, the observed TP concentrations near the reservoir bottom increased to ~0.13 mg L^{-1} near

the dam and to ~0.27 mg L^{-1} near Quarry Island (where the PVIA water intake structure is located) during summer. The overlying water in the sediment cores (collected from near the dam and the PVIA raw water intake structure) had a mean SRP concentration of 0.16 mg L^{-1} under anaerobic conditions over the last few days of the incubations. These concentrations from the incubations were within the range of that used in reservoir model calibration, especially considering that SRP only makes up a portion of TP. However, the overlying water in the collected cores was relatively clear at the end of the incubation suggesting P would be mostly in the dissolved form.

Phosphorus can be released from sediments to the overlying water in the dissolved reactive form (i.e., SRP as measured in this study) or as dissolved organic P (DOP, not measured in this study). Hamdan et al. (2010) observed that DOP releases were up to five times greater than SRP in sediment cores collected from deep water at Beaver Lake, Arkansas. However, the SRP release rates at Beaver Lake were generally less than that observed at Lake Wister (Table 3; Sen et al., 2007; Hamdan et al., 2010). Wang et al. (2009) found the DOP release was the most important factor in total dissolved P release from sediments, representing 30 to 60% of the P released from dried sediments into aqueous solution under aerobic conditions. It is possible that DOP release could be important at Lake Wister, and this should be the focus of future sediment studies. However, most studies have shown that DOP release rates were at the same order of magnitude of SRP release, suggesting that the values used in the reservoir model (see AMEC, 2008) still overestimated the total dissolved P released from the bottom sediments into overlying water at Lake Wister.

Phosphorus can also be resuspended from bottom sediments under turbulent mixing conditions, especially in shallow headwaters. For example, wind resuspension of bottom sediments might increase TP concentrations in the water column of shallow lakes or in shallow waters (Søndergaard et al., 1992). Vertical mixing in the water column generally maintains uniform physico-chemical conditions especially DO concentrations near saturation, which can influence SRP release rates. However, this P is usually bound in particulate forms that are not bioavailable (i.e., biologically reactive) under the conditions resulting in resuspension. The equilibrium P concentrations (EPC_0) of the bottom sediments would probably regulate dissolved P in the overlying waters, and Spears et al. (2007) showed that EPC_0 was less in shaken (i.e., resuspended) sediments (0.03 mg L^{-1}) than quiescent, intact sediment cores (0.18 mg L^{-1}). Sediment resuspension occurs in Lake Wister, although the bioavailability and equilibrium concentrations of this P source have not been evaluated. The release of P from resuspended sediments at Lake Wister is likely small given the aerobic release rates (<1 mg m^{-2} d^{-1}) in this study, and reported decreases in EPC_0 during resuspension by other studies.

CONCLUSIONS

This study measured SOD and P release rates under quiescent conditions from intact sediment cores collected at Lake Wister in July 2010, and then compared these measured values to those used in calibration of the previous reservoir model (see AMEC, 2008). The O_2 demand by bottom sediments was from 9.9 to 22.6 mg m^{-2} hr^{-1} on average across the sites, where the least SOD was measured in the shallow reservoir headwaters. These measured rates were an order of magnitude less than those used the reservoir model parameterization, suggesting that the selected reservoir model would need to be account for SOD under varying conditions, i.e. quiescent and turbulent. Oxygen demand drives hypolimnetic waters to anaerobic conditions, potentially releasing SRP from the bottom sediments through reductive dissolution and microbial mineralization. The

measured SRP release rates (near zero to 3.30 mg m^{-2} d^{-1}) were also an order of magnitude less than those used in the reservoir model calibration. The previous modeling report (see AMEC, 2008) suggested that internal P sources were dominant, and that this source precluded the development of a watershed-based strategy to reduce P inputs from external sources. This study illustrates the importance of measuring SOD and SRP release rates to assist in the parameterization of reservoir models, and not simply relying on those parameters as calibration tools.

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