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Research Letter

Sediment Phosphorus Flux at Lake Tenkiller, Oklahoma: How Important Are Internal Sources?

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Core Ideas

- Sediment P release needs to be measured in lakes and reservoirs that are eutrophic.
- Sediment P release data is essential for calibrating lake and reservoir models used in a TMDL.
- Sediment P release at Lake Tenkiller was very high relative to that predicted by the model.
- Sediment P release was also very high relative to that observed in regional reservoirs.
- Internal P sources need to be addressed in watershed management strategies.

Abstract: Watershed management for lakes and reservoirs largely focuses on external sources, but internal sources, such as sediment P release, can be important. This study quantified sediment P release in Lake Tenkiller, Oklahoma, in summer 2016. Eight intact sediment–water cores were collected from the riverine and transition zone and then incubated at 22°C for 8 d under aerobic and anaerobic conditions. Phosphorus mass in the overlying water of all cores increased with time, and the rate of increase was significantly greater under anaerobic conditions. Sediment P release was reflective of eutrophic and hypereutrophic conditions, measuring up to 15.2 mg m^{-2} d^{-1}. Sediment P fluxes were four to five times greater than the net flux predicted by the model of Lake Tenkiller, suggesting that internal P sources need to be considered in the total maximum daily load process.

Eutrophication is a natural aging process, involving gradual nutrient enrichment and increased productivity with plant and animal life, but this process can be accelerated because of human activities adding nutrients into aquatic ecosystems (Anderson et al., 2002). Nutrient concentrations and ratios are strongly correlated to nuisance algal blooms, leading to a variety of water quality issues. Phosphorus (P) and nitrogen (N) can individually limit algal growth, although co-limitation and seasonal shifts in nutrient limitation tend to be more predominant (Dzialowski, 2005; Ludwig et al., 2012; Maberly et al., 2002). Nuisance algal blooms jeopardize the quality of waters used for recreation, drinking water sources, and aquatic life habitats; therefore, watershed management has largely focused on P and N concentrations, transport, and loadings (Dzialowski, 2005; Paerl et al., 2001; Saadoun et al., 2001).

Nutrients enter lakes and reservoirs from the watershed (i.e., external sources) but are then stored within the system. In particular, bottom sediments are important internal sinks and sources in lakes and reservoirs, especially in terms of denitrification and sediment P release (e.g., see Grantz et al., 2014). This internal flux can maintain P availability and concentrations even when external sources are decreased, resulting in continued eutrophication and water quality issues (Christophoridis and Fytianos, 2006). Sediment P release is an important consideration in water quality modeling and total maximum daily load (TMDL) development for lakes and reservoirs. Sediment P release is commonly used as a calibration parameter in lake modeling, which can potentially over- or underestimate the importance of the internal P source (Haggard et al., 2012).

A watershed model and a lake model were developed using USEPA's Hydrological Simulation Program-Fortran (HSPF) and the Environmental

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Abbreviations: EFDC, Environmental Fluid Dynamics Computations; HSPF, Hydrological Simulation Program-Fortran; SRP, soluble reactive phosphorus; TMDL, total maximum daily load.

Fluid Dynamics Computations (EFDC), respectively, for Lake Tenkiller, Oklahoma, in the transboundary Illinois River Watershed (AQUA TERRA Consultants, 2015). The watershed model (HSPF) simulated nutrient loadings to the lake model (EFDC), which would be used in the TMDL for Lake Tenkiller. The EFDC model includes a sediment diagenesis routine to simulate the deposition and release of P from bottom sediments (DiToro, 2001). The purpose of this study was to quantify sediment P release in Lake Tenkiller under aerobic and anaerobic conditions and compare to the net P fluxes predicted by the sediment diagenesis routine of the EFDC model.

Methods

Study Site Description

The Illinois River flows into northeastern Oklahoma from northwestern Arkansas and then through the mountains of eastern Oklahoma, draining into Lake Tenkiller before flowing into the Arkansas River. The Illinois River watershed covers 4331 km^2 within the Ozark Highlands and Boston Mountains (Cooke et al., 2011). The geology is characterized by karst and cherty limestone, sandstone, and shale, and the common soil types are Ultisols, which have high levels of iron and aluminum and are able to store excess nutrients. Land use within the Illinois River watershed is approximately 48% pasture, 46% forested; the majority of the agriculture is for poultry production (Olsen et al., 2012). This watershed (i.e., upstream from Lake Tenkiller) has historically been one of the largest producers of broiler chicken where the poultry litter was subsequently land applied.

Lake Tenkiller has a volume of 0.835 km^3, average depth of 15.5 m, maximum depth of 50 m, 210 km of shoreline, and surface area of 52 km^2 (Cooke et al., 2011). Before 1974, Lake Tenkiller was considered oligo-mesotrophic and one of Oklahoma's highest-quality water resources. From 1975 to 1986, P in the lake water increased from external sources, causing symptoms of decreased water clarity and increased algal blooms (Nolen et al., 1986). The reservoir became eutrophic to hypereutrophic by 1986 and continued to express conditions through 2008 (Cooke et al., 2011). However, P concentrations and loads in the Illinois River have decreased significantly since 2002, after changes in effluent inputs and pasture management (Haggard, 2010; Scott et al., 2011).

Lake Tenkiller is currently listed on Oklahoma's 303(d) list for P, dissolved oxygen (O_2), and chlorophyll-*a* impairments. Oklahoma's water quality standards require a long-term average of chlorophyll-*a* concentrations 0.5 m below the surface to not exceed 10 μg L^{-1}. Portions of the Illinois River are also on Oklahoma's 303(d) list for dissolved O_2, total P, lead, turbidity, enterococcus, and *Escherichia coli* (Oklahoma Department of Environmental Quality, 2015) and Arkansas's 303(d) list for chlorides, sulfates, and pathogens (Arkansas Department of Environmental Quality, 2016). In 2010, the USEPA began the TMDL process using HSPF and EFDC to model nutrient loads and water quality in Lake Tenkiller. The EFDC model was calibrated using only data from 2005 and 2006 (AQUA TERRA Consultants, 2015).

Sediment Phosphorus Release

Two sampling sites were selected at Lake Tenkiller, one in the riverine zone (Site 1: 35.758285, −94.904385) and another in the transition zone (Site 2: 35.706831, −94.964832). Eight intact sediment–water cores were collected from each sampling site. Plexiglas tubes, with a length of 0.6 m and inner diameter of 7.9 cm, were inserted approximately 0.3 m into the sediment using a UWITEC corer, and a cap was then placed on the bottom end and a rubber stopper on top. A properly collected sample had undisturbed sediment at the surface and through the depth of the core, with relatively clear overlying water.

Once the cores were returned to the laboratory, the depth of the overlying reservoir water was adjusted so that each core contained 1 L of water. The cores were wrapped in aluminum foil to eliminate light and were incubated at room temperature, approximately 22°C. For approximately 24 h, the overlying water in all the cores was bubbled with air. Then for 8 d, four cores from each site were incubated under anaerobic conditions (bubbled with N_2), and the other cores were incubated under aerobic conditions (bubbled with air). Bubbling with N_2 gas purged the dissolved O_2 from the water column in the anaerobic cores and prevented O_2 from diffusing in by putting the core under positive pressure. Light was excluded from all cores to limit algal growth.

A 50-mL sample was removed from the overlying water of each core at daily intervals during incubation. This water sample was filtered (0.45 μm), acidified using concentrated HCl, and analyzed for soluble reactive P (SRP) using the automated ascorbic acid reduction technique (APHA, 2005). The overlying water in the cores was maintained at a volume of 1 L by adding filtered (0.45 μm) water collected from Lake Tenkiller with a measured SRP concentration.

Sediment P release rates (mg m^{-2} d^{-1}) were calculated as linear changes in P mass in the overlying water as a function of time (mg d^{-1}) divided by the inside area of the sediment–water cores (0.00495 m^2). The P mass in the overlying water was corrected for water removal and addition to the cores during incubation. The SRP concentration of the filtered replacement water was less than the detection limit (<0.005 mg L^{-1}) for every sample throughout the experiment. Simple linear regression was used to determine the slope between adjusted SRP mass and time. Analysis of covariance was used to evaluate whether release rates were different between sites or treatment ($P \leq 0.05$).

Results

Riverine Zone

After the overlying water was aerated overnight, SRP concentrations ranged from 0.024 to 0.082 mg L^{-1}, with an average of 0.061 mg L^{-1} in the water above sediments. During the incubation, SRP concentrations in all cores increased with time, although the rate of increase was greater under anaerobic conditions (Fig. 1). By the end of 8 incubation days, all cores under anaerobic conditions had SRP concentrations ≥0.400 mg L^{-1}, whereas SRP concentrations under aerobic conditions averaged 0.089 mg L^{-1}.

During the incubation, SRP mass in the overlying water in all cores significantly increased over time ($P < 0.01$) (Table 1). The slopes of the relation used to estimate sediment P flux were significantly greater under anaerobic conditions ($P < 0.01$) compared with aerobic conditions. Average sediment P flux for anaerobic cores was 15.2 mg m^{-2} d^{-1} (range: 11.6–19.0 mg m^{-2} d^{-1}), whereas average sediment P flux for aerobic cores was an order of magnitude less at 1.5 mg m^{-2} d^{-1} (range: 0.70–2.1 mg m^{-2} d^{-1}) in the riverine zone.

Transition Zone

After 24 h of aeration, SRP concentrations ranged from 0.014 to 0.029 mg L^{-1}, with an average of 0.019 mg L^{-1}, in the overlying water of the sediment cores. During incubation, SRP concentrations in all cores increased with time, although the rate of increase was greater under anaerobic conditions (Fig. 1). On average, aerobic cores increased by only 0.031 mg L^{-1}, whereas anaerobic cores increased by 0.510 mg L^{-1}. The cores under anaerobic conditions showed more variability in SRP concentrations, with two cores never reaching concentrations above 0.373 mg L^{-1} and the other two cores reaching concentrations >0.700 mg L^{-1}. Final SRP concentrations for aerobic and anaerobic cores were 0.047 and 0.533 mg L^{-1} on average in the overlying water.

During the incubation, SRP mass in the overlying water in all cores significantly increased over time ($P \le 0.02$) (Table 1). The slope of the relation used to estimate sediment P flux was significantly greater under anaerobic conditions ($P = 0.01$) compared with aerobic conditions, but sediment P release was not different between sites. Average sediment P flux for anaerobic cores was 12.3 mg m^{-2} d^{-1} (range: 4.8–17.6 mg m^{-2} d^{-1}) in the transition zone, whereas average sediment P flux for aerobic cores was 0.86 mg m^{-2} d^{-1} (range: 0.25–1.65 mg m^{-2} d^{-1}).

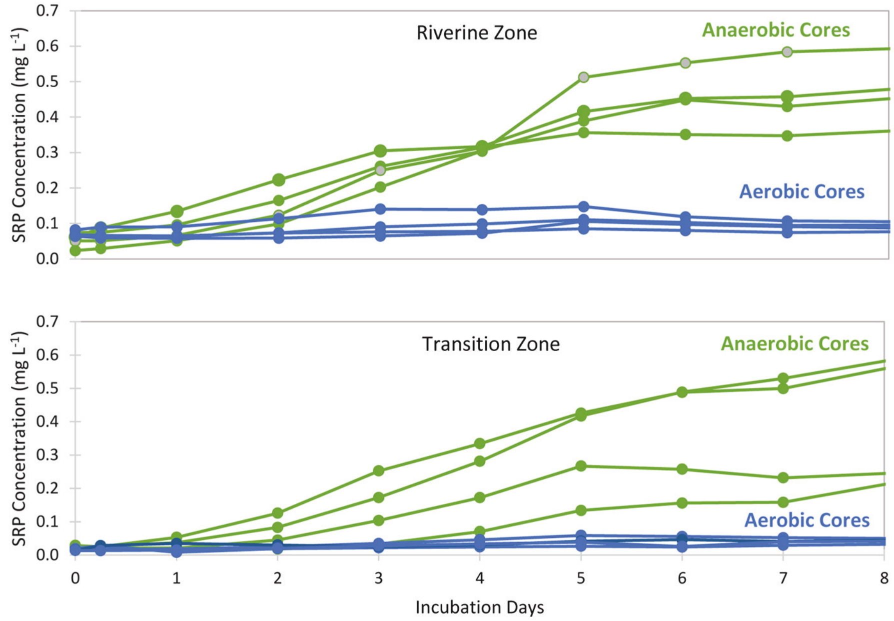

Fig. 1. Soluble reactive phosphorus (SRP) concentrations in the overlying water of cores from the transition and riverine zones at Lake Tenkiller, Oklahoma.

Discussion

Local and Regional Comparisons

Several studies have measured sediment P release rates in lakes and reservoirs across North America and found fluxes ranging between 1 and 50 mg m^{-2} d^{-1} under aerobic and anaerobic conditions (e.g., Holdren and Armstrong, 1980; Moore and Reddy, 1994; Riley and Prepas, 1984). The range in mean P release rates under aerobic and anaerobic

Table 1. Soluble reactive phosphorus fluxes measured from intact sediment cores collected in the riverine and transitional zones at Lake Tenkiller, June 2016.

Zone	Treatment	Slope	R^2	P value	Phosphorus flux
		mg d^{-1}			mg m^{-2} d^{-1}
Riverine	Aerobic	0.0003	0.75	0.005	1.55
Riverine	Aerobic	0.0004	0.93	<0.001	1.82
Riverine	Aerobic	0.0004	0.67	0.013	2.08
Riverine	Aerobic	0.0001	0.89	<0.001	0.70
Riverine	Anaerobic	0.0029	0.99	<0.001	14.1
Riverine	Anaerobic	0.0024	0.96	<0.001	11.6
Riverine	Anaerobic	0.0039	0.95	<0.001	19.0
Riverine	Anaerobic	0.0033	0.98	<0.001	16.0
Transition	Aerobic	0.0003	0.95	<0.001	1.65
Transition	Aerobic	0.0002	0.62	0.021	0.75
Transition	Aerobic	0.0002	0.74	0.006	0.79
Transition	Aerobic	0.0001	0.20	<0.001	0.25
Transition	Anaerobic	0.0020	0.93	<0.001	9.81
Transition	Anaerobic	0.0036	0.98	<0.001	17.6
Transition	Anaerobic	0.0010	0.82	0.002	4.82
Transition	Anaerobic	0.0035	0.95	<0.001	17.2

conditions in local reservoirs has been between approximately zero and 6 mg m^{-2} d^{-1} (Haggard et al., 2005, 2012; Sen et al., 2006). Our measured P release rates at Lake Tenkiller were within the range of that observed locally under aerobic conditions but double the maximum average under anaerobic conditions.

Longitudinal gradients in nutrient concentrations and algal productivity are often observed in reservoirs, where the lacustrine zone is less productive than up-reservoir (James et al., 1987). There are also longitudinal gradients in sediment deposition, and the deposition rates generally are greater in the transition and riverine zones (James et al., 1987). The changes in sediment deposition are also likely to result in spatial variability in sediment P release and the importance of internal processes. For example, sediment P release at Beaver Reservoir, Arkansas, was greater in the transitional and riverine zones than down reservoir (Sen et al., 2006), and at Lake Eucha, the greatest sediment P fluxes were in the transitional zone (Haggard et al., 2005). Based on these typical gradients, sediment P release at Lake Tenkiller down-reservoir should be less than ~12.3 mg m^{-2} d^{-1}. However, it might be worth collecting cores from the deeper water closer to the dam to quantify sediment P release.

Lake Tenkiller has been classified as a eutrophic reservoir based on eplimnetic total P concentrations (Cooke et al., 2011), where the riverine zone fluctuates from eutrophic to hypereutrophic during summer months. Sediment P release rates from this study fall within typical ranges found in hypereutrophic lakes and reservoirs under anaerobic conditions (9–36 mg m^{-2} d^{-1}; Auer et al., 1993; Holdren and Armstrong, 1980; Penn et al., 2000; Xie et al., 2003). The rates at Lake Tenkiller were similar to that observed in summer 2004 at Lake Frances (~15 mg m^{-2} d^{-1}), a small eutrophic to hypereutrophic impoundment on the Illinois River at the Arkansas and Oklahoma border (Haggard and Soerens, 2006).

EFDC Model Comparisons

The EFDC model predicts net P flux from the sediments to the water column, so P can be deposited in the bottom sediments or released at Lake Tenkiller (AQUA TERRA Consultants, 2015). Based on the EFDC model, the maximum net P flux from the sediment was approximately 3 mg m^{-2} d^{-1} occurring between July and September, when Lake Tenkiller would be stratified with an anoxic hypolimniom (T. Shaikh, personal communication, 2016). Our measured data showed average P release under anaerobic conditions to be four to five times greater than the predicted maximum values from the EFDC model. The EFDC model of Lake Tenkiller likely underestimates the importance of internal P sources, which would influence waste load allocations in the TMDL process. This study should inform the USEPA that the sediment diagenesis component of the EFDC model needs to be updated to reflect the magnitude of observed sediment P release.

When internal sources of P are low, management strategies focus on reducing external sources. Because internal sources can maintain P availability when external sources are decreased (Christophoridis and Fytianos, 2006), Lake Tenkiller might remain eutrophic even if external sources are reduced. The implications could mean millions of dollars invested in wastewater treatment plant upgrades and implementation of best management practices throughout the Illinois River watershed that do not result in improved water quality at Lake Tenkiller.

Sediment P flux at Lake Tenkiller was high relative to regional reservoirs used to guide EFDC model calibration, but how do the fluxes compare with that from the watershed? The mean flux from the Illinois River watershed draining to Lake Tenkiller was 0.2 mg m^{-2} d^{-1} (Tortorelli and Pickup, 2006) and looking only at the Arkansas portion, 0.4 mg m^{-2} d^{-1} (Haggard, 2010). The flux from the bottom sediments at Lake Tenkiller was orders of magnitude greater than the external flux. However, in terms of load, P from the sediments was less because of the difference in lake and watershed area. Internal sources could be up to 65,000 kg yr^{-1} (25% of external load), making some basic assumptions (i.e., half of lake surface area goes anoxic over the sediments about half of the time). Therefore, internal sources might be important not only in the lake model but also when developing mitigation strategies to improve water quality.

The history of total P content in Lake Tenkiller sediments shows an increase from about 300 to 500 mg kg^{-1} between 1954 and 1965 to about 1000 to 1,500 mg kg^{-1} between 2000 and 2005 (Fisher et al., 2009). Because total P concentrations and loads in the Illinois River have decreased since 2002 (Haggard, 2010; Scott et al., 2011), if sediment P content also peaked in the 2005 time period, then it is possible that release rates were even higher during 2005 and 2006 relative to the rates measured in 2016. Thus, the EFDC model may be underestimating sediment P release even more than our data show.

Conclusions

This study quantified sediment P fluxes for Lake Tenkiller, Oklahoma, under aerobic and anaerobic conditions and made the following observations:

- Sediment P fluxes were significantly greater under anaerobic conditions compared with aerobic conditions for all cores.
- Calculated sediment P fluxes from this study were four to five times greater than maximum values predicted by the EFDC model.
- The sediment diagenesis component of the EFDC model needs to be updated to reflect the magnitude of observed sediment P release.
- If internal sources are not addressed in watershed management for Lake Tenkiller, continued eutrophication and water quality issues could occur even after external sources are further reduced.

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