AN INTEGRATED ASSESSMENT OF THE EFFECTS OF INTERNAL PHOSPHORUS CYCLING AND SEDIMENT RESUSPENSION ON THE EUTROPHICATION OF LAKES AND RESERVOIRS IN THE CENTRAL PLAINS

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Project Summary

Historically, lake and reservoir management has focused on controlling external nutrient loading. However, it is becoming increasingly clear that internal mechanisms including the resuspension of surficial sediments and direct flux of nutrients from the sediment under anoxic conditions can contribute significant amount of phosphorus (P) back into the water column. As a result, reductions in external nutrient loading alone may not effectively curb the negative processes associated with eutrophication. Relatively little is known about internal nutrient cycling in reservoirs of the Central Plains. To better understand mechanisms and conditions that may influence internal nutrient cycling in this region, we collected *in-situ* reservoir data and conducted laboratory sediment bioassay and core incubation experiments in and with water collected from four eastern Kansas reservoirs: Clinton Lake (CL), Gardner Lake (GL), Pomona Lake (PL), and Pony Creek Lake (PC). The goals of the project were to: 1) determine how resuspended sediments impacted dissolved and total nutrient concentrations (nitrogen and phosphorus) and algal biomass in laboratory bioassay studies; and 2) quantify P-release rates from anoxic sediments collected from three different reservoir zones using laboratory sediment core incubations. Furthermore, to better understand the extent to which there is potential for these processes to occur in reservoirs of the Central Plains, we used *in-situ* meters to quantify sediment/water interface (within 1m of the sediment surface): 1) turbidity concentrations to determine how often sediment resuspension occurred in each reservoir and at what concentrations; and 2) dissolved oxygen concentrations to determine how often reservoirs experienced anoxia and therefore have the potential for sediment nutrient release.

In-situ sediment resuspension events were observed near the sediment/water interface of each reservoir. Higher maximum turbidity concentrations were observed in the smaller reservoirs (near 500 NTU's) than in the larger reservoir (~100 NTU's). However, even relatively low turbidity concentrations (50 NTU's) had significant effects on algal biomass and nutrient concentrations in each of the subsequent sediment resuspension bioassays. Evidence suggests that resuspended sediments affected algal biomass in several ways. Specifically, resuspended sediments increased nutrient concentrations, and meroplankton, algal cells within the sediment, became entrained in the water column following sediment resuspension events.

Relatively long periods of anoxia were also observed near the sediment/water interface of each reservoir. In the two smaller reservoirs, anoxia appeared to be associated with reservoirs turnover as they were anoxic until the late summer or early fall. Surprisingly, the larger reservoir was also anoxic near the sediment/water interface during most of the summer. Laboratory sediment core incubation experiments showed that under anoxic conditions, sediment from each reservoir released high concentrations of P relative to control (oxic) cores. For example, P-release rates in anoxic cores were up to 54 times greater than in oxic cores.

The results from the *in-situ* probes combined with the laboratory experiments, suggest that internal nutrient cycling is an important process in reservoirs of the Central

Plains. Specifically, our results indicate that resuspension events occur in both small and large reservoirs, and that these events can increase turbidity concentrations to levels that have significant effects on algal biomass and nutrient concentrations. Additionally, the i*n-situ* oxygen measurements and the high P-release rates observed under anoxic conditions indicate that reservoirs have a high potential for sediment nutrient release. Overall, our results highlight the importance of considering internal nutrient cycling in management and restoration projects.

Introduction

Eutrophication, the enrichment of waterbodies with plant nutrients, affects lakes and reservoirs worldwide (see recent review by Smith, 2003). The effects of eutrophication are particularly pronounced in the Central Plains region of the United States, where many lakes and reservoirs are located in agriculturally dominated watersheds (e.g. Smith *et al*., 2002; Dzialowski *et al*., 2005; Wang *et al*., 2005). As a result, effective management strategies and realistic restoration goals are needed for these important ecosystems. Historically, lake and reservoir management has focused on controlling external nutrient inputs (e.g. Walker and Havens, 2003; Havens and Walker, 2002). However, it is becoming increasingly clear that internal mechanisms can also be important sources of phosphorus (P) into the water column.

Several in-lake processes can contribute to internal nutrient recycling. First, the episodic resuspension of benthic sediments by wind induced mixing, reservoir water release, and recreational use (e.g. Anthony and Downing, 2003; Schallenberg and Burns, 2004) may shuttle nutrients from the sediment back into the water column. Specifically, high concentrations of nutrients, which often stimulate algal production, can be released during resuspension events (e.g. Hamilton and Mitchell, 1997; Ogilvie and Mitchell, 1998). Furthermore, resuspended sediments can also contribute meroplankton, algae cells found at the interface between water and sediment, to the water column following resuspension events (Schelske *et al*., 1995). Second, sediments may directly release nutrients into the water column under anoxic (no oxygen) conditions (Nurnberg, 1984; Scheffer, 1998). During periods of high oxygen (oxic) sediments retain P. However, during periods of anoxia the redox-potential at the water-sediment surface is reduced. Under these conditions, Fe (II) is converted into Fe (III), resulting in a release of PO_4^{3-} back into the water column (Gachter and Muller, 2003).

If internal mechanisms such as those mentioned above are important sources of P to the water column, then it is likely that reductions in external nutrient loads alone will not effectively curb the negative impacts of eutrophication (Marsden, 1989). It is currently unknown how sediment resuspension and nutrient release from the sediment affect nutrient concentrations in reservoirs of the Central Plains. Therefore, we used several complimentary approaches to assess internal nutrient recycling and its affects on algal biomass in four reservoirs to: 1) determine how resuspended sediments impacted dissolved and total nutrient concentrations (nitrogen and phosphorus) and algal biomass in laboratory bioassay studies; and 2) quantify P release rates from anoxic sediments collected from three different reservoir zones using laboratory sediment core incubations. Furthermore, to better understand the extent to which there is potential for these processes to occur in reservoirs of the Central Plains, we used *in-situ* meters to quantify sediment/water interface: 1) turbidity concentrations to determine how often sediment resuspension occurred in each reservoir and at what concentrations; and 2) dissolved oxygen concentrations to determine how often reservoirs experienced anoxia and therefore have the potential for sediment nutrient release.

Methods

Study Reservoirs

Sediment resuspension and sediment nutrient release were studied in four eastern Kansas drinking water reservoirs: Clinton Lake (CL), Gardner Lake (GL), Pomona Lake (PL), and Pony Creek Lake (PC). CL and PL are relatively large federal reservoirs, while PC and GL are smaller city reservoirs (Table 1). Additional background and ecological information on CL, PC, and GL is provided in Wang *et al*. (1999) and Dzialowski *et al*. (2005).

Reservoir	Watershed Area (km^2)	Surface Area (km^2)	Watershed: Surface Area Ratio	Maximum Depth (m)
Clinton	953	28	34:1	16.8
Gardner	13	0.28	46:1	11.6
Pomona	808	16	51:1	15.4
Pony Creek	17	0.73	23:1	10.3

Table 1. General reservoir and watershed characteristics of the four study reservoirs.

In situ Surface Water Quality Data

In-situ water quality data were collected near the surface of each reservoir to quantify general limnological conditions. Samples were collected from the main basins of each reservoir on at least four sampling dates between June and November of 2005. A Horiba® Model 10 multi-probe unit was used to collect depth profiles of temperature, pH, dissolved oxygen, and turbidity during each sampling event. We also collected a 1 L surface sample (0.25 m – 1.5 m depth) that was returned to the Ecotoxicology Laboratory for analysis of chlorophyll *a*, nutrients (dissolved and total), and total suspended solids. Chlorophyll *a* concentrations were determined after filtering reservoir samples through Whatman GF/F glass fiber filters. Chlorophyll was then extracted using 90% basic methanol (10% saturated $MgCO_3$) and concentrations were determined (corrected for pheophytin *a*) with an Optical Technologies fluorometer before and after acidification (APHA, 1995).

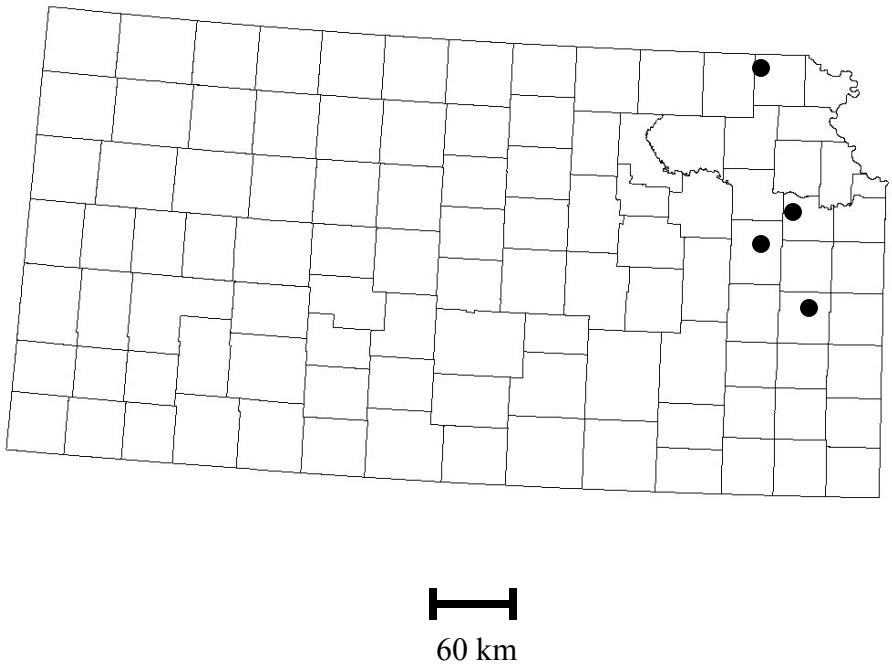

Figure 1. Locations of the four study reservoirs in eastern Kansas.

***In situ* Near-Sediment Turbidity and Dissolved Oxygen Measurements**

A Hydrolab multiprobe unit equipped with both a dissolved oxygen probe and a self-cleaning turbidity probe was positioned within roughly 1 m of the sediment surface in the main basin of each reservoir. The probes were used to quantify oxygen and turbidity concentrations near the sediment surface of each reservoir. The meters were programmed to collect dissolved oxygen and turbidity measurements at 15-30 minute intervals. The probes were maintained in each reservoir from June through September 2005. Data was downloaded from the probes at roughly bi-weekly intervals. Shortly after the start of the experiment, it was determined that the probe positioned in PL was non-functional. Therefore, *in-situ* near-sediment dissolved oxygen and turbidity measurements are presented for three reservoirs only.

Water levels fluctuated in the reservoirs throughout the summer, and as a result, several of the probes came into contact with the reservoir sediment and recorded unrealistically high turbidity values for brief periods of time. Therefore, when the probes were retrieved from each reservoir to download the data, they were checked for evidence of sediment contact. If it was obvious that a probe had come into contact with the bottom sediment, it was repositioned above the sediment surface and high turbidity values associated with sediment contact were removed from the dataset.

Sediment Bioassay Experiments

Bioassay experiments were conducted between June and November 2005 at the Environmental Bioassay Research Facility (EBRF) at the KBS. For each reservoir, a 2 x 5 factorial design consisting of 2 types of reservoir water (non-filtered and filtered to remove algae from the source reservoir water) and 5 sediment concentrations (0, 50, 150, 250, and 500 NTU's) was conducted in thirty 1-L glass bioassay bottles. Each treatment was replicated in triplicate. To initiate the experiment, a 20-L surface water sample and 6 sediment cores were collected from the main basin of each reservoir using a 5 cm diameter Wildco Sediment Corer. We were interested in the responses of algal biomass to sediment additions; therefore, the reservoir water was initially filtered (200 μm) to remove all macrozooplankton. This coarsely filtered water, with intact algal communities, was then added to half of the bottles (non-filtered – with algae). The remaining reservoir water was filtered (GF/F filter; pore size = 0.7 μm) to remove all algae and added to the remaining 15-bioassay bottles (filtered – without algae).

To create the five sediment resuspension treatments, the sediment from the upper 3 cm of the sediment cores from each reservoir were extruded and combined in individual glass beakers. Sediment was then added to 1 L of deionized water until the turbidity concentration (measured with a laboratory turbidity meter) in the beaker was equal to the five target turbidity concentrations: 0 (control – no sediment added); 50; 150; 250; and 500 NTU's.

After the treatments were established, the bioassay bottles were incubated in a growth chamber at 20°C, where they were exposed to roughly 200 μE m^{-2} s^{-1} of light on a 12-hour light/dark cycle. *In vivo* fluorescence, which is often used as a surrogate for

algal biomass in bioassay experiments (Elser *et al*. 1990), was measured daily using a Turner Model 10 Fluorometer. Each day, the sediment within the bottles were thoroughly mixed into suspension with an electronic stirrer, and fluorescence was recorded. Water samples were also collected at the beginning of each experiment (between day 1 and 3) for determinations of dissolved and total forms of nitrogen and phosphorus (Ebina, *et al*., 1983). Nutrient concentrations were measured just after the initiation of the bioassay experiments (between day 1 and 3) to minimize the amount of time that resident algae or meroplankton had for nutrient uptake. The bioassay experiments were conducted for 11-14 days.

Repeated Measure Analysis of Variance (RM-ANOVA) was used to determine if sediment resuspension had significant effects on nutrient concentrations and algal biomass. Greenhouse-Geiser corrections were used to account for potential violations of the assumptions of sphericity (von Ende, 2001). RM-ANOVA provides a number of comparisons both within and between treatments; however, for the purposes of this research we focused on several specific comparisons. First, the results from the non-filtered bioassay experiments (resident algae in tact) were used to determine how resident algal communities responded to resuspension events presumably through nutrient additions. Second, the results from the filtered bioassay experiments (resident algae were initially removed) were used to determine if meroplankton became established within the water column following resuspension events. Tukey's HSD (P=0.05) was used to determine which individual treatments were different when significant treatment differences were identified with RM-ANOVA, and all ANOVA described below.

Two-way ANOVA was used to determine if there were significant effects of the reservoir water (filtered and non-filtered) and sediment addition (0, 50, 150, 250, and 500 NTU's) treatments on concentrations of total and dissolved nutrients. When necessary, nutrient data was log transformed to help meet the assumptions of normality.

Nutrient Release Studies

Nutrient release experiments were conducted between June and November 2005 using sediment core incubations at the EBRF. Sediment cores were collected using a 5 cm diameter Wildco Sediment Corer. Eighteen cores were collected from each reservoir: 6 from the lacustrine zone, 6 from the transition zone, and 6 from the riverine zone. The cores were returned to the laboratory and processed within 24 hours.

The reservoir water was drained off of the top of each core and replaced with filtered lake water (0.7 μm), so that there are not confounding influences of plankton on nutrient recycling rates (Elser *et al*., 1988). The cores were initially bubbled with air so that all of the P in the water column was able to settle in the sediment under oxic conditions.

Internal P release was examined using a 2 x 3 experimental design consisting of two core types (anoxic and oxic) from three reservoir locations (lacustrine, transition, and riverine zones). Each core was bubbled with either N_2 gas to create anoxic conditions, or

bubbled with air to create oxic (control) conditions. Incubation experiments were run in triplicate for nine-eighteen days in a growth chamber for nine days at 20°C that was kept dark throughout the experiment. Water samples (10 ml) were collected daily from the cores and analyzed for dissolved P (PO_4^{3-}; see methods above). Filtered lake water was added to the test cores to account for sample loss.

P-release rates were calculated for each core (both anoxic and oxic) using the maximum increase in P that was observed over at least a two-day period during the course of the study. Individual two-way ANOVA was used to determine if there were significant differences in release rates between anoxic and oxic cores and the three locations within each reservoir. A two-way ANOVA was also used to determine if there were differences in release rates between reservoirs where reservoirs and location were used as the two factors (only anoxic cores were used in this ANOVA).

Results

In situ *Water Quality Conditions*

Mean total nutrient concentrations were the lowest in CL (TP=41.3$\pm$3.1 µg L^{-1}; TN=557.3$\pm$30.9 µg L^{-1}) and the highest in GL (TP=63.6$\pm$13.0; TN=855.7$\pm$138.9 µg L^{-1}) (Table 2). With respect to dissolved nutrients, CL (104.3$\pm$24.7 µg L^{-1}) and GL (82.9$\pm$35.0 µg L^{-1}) had the highest mean concentrations of NO_3-N, and GL (37.3$\pm$21.7 µg L^{-1}) had the highest concentration of NH_3-N. In contrast, mean dissolved phosphorus (PO_4-P) concentrations were relatively similar between the four reservoirs (Table 2). Chlorophyll *a* (20.8$\pm$2.4 µg L^{-1}) and total suspended solids (30.1$\pm$0.8 mg L^{-1}) concentrations were highest in PC. In general, there were few differences in the average surface concentrations of pH, dissolved oxygen, temperature, and turbidity between the four reservoirs (Table 2).

In situ Near-Sediment Measurements

The mean turbidity concentrations measured near the sediment surface of CL, GL, and PC were 20, 14, and 5 NTU's respectively. Higher maximum turbidity concentrations were observed in the two smaller reservoirs (471 in GL and 449 in PC) than were observed in CL (97 NTU's). Turbidity concentrations were more variables in CL, where they peaked several times over the course of the study. In contrast, turbidity concentrations tended to remain relatively low throughout the summer in the two smaller reservoirs, and peak concentrations were observed in late October in GL and in mid-August in PC (Figure 2).

Reservoir	pH	TSS. (mg L^{-1})	DO (mg L^{-1})	Turbidity (NTU)	Temp. (˚C)	NO_3-N (µg L^{-1})	NH_3-N (µg L^{-1})	TN (µg L^{-1})	PO_4-P (µg L^{-1})	TP (µg L^{-1})	Chl *a* (µg L^{-1})
Clinton Lake	8.2 (0.1)	24.5 (1.7)	6.3 (0.4)	16.4 (2.4)	26.0 (0.6)	104.3 (24.7)	22.2 (6.5)	557.3 (30.9)	8.4 (1.4)	41.3 (3.1)	12.7 (2.5)
Gardner Lake	7.9 (0.1)	26.2 (0.6)	3.0 (0.3)	15.3 (1.0)	23.1 (0.4)	82.9 (35.0)	37.3 (21.7)	855.7 (138.9)	9.0 (1.3)	63.6 (13.0)	17.6 (2.6)
Pomona Lake	8.3 (0.1)	28.4 (0.6)	5.2 (0.5)	23.0 (4.0)	26.4 (0.6)	67.4 (23.4)	12.5 (3.7)	615.7 (34.3)	8.4 (1.9)	45.8 (3.7)	14.9 (2.2)
Pony Creek	8.2 (0.9)	30.1 (0.8)	5.2 (0.7)	21.9 (3.2)	24.4 (0.8)	54.0 (19.4)	18.0 (6.4)	806.0 (41.8)	7.0 (1.7)	50.6 (3.4)	20.8 (2.4)

Table 2. General water quality data collected near the surface of each reservoir. For each variable, the data represent mean values based on at least four sampling events. The standard error is presented in parenthesis.

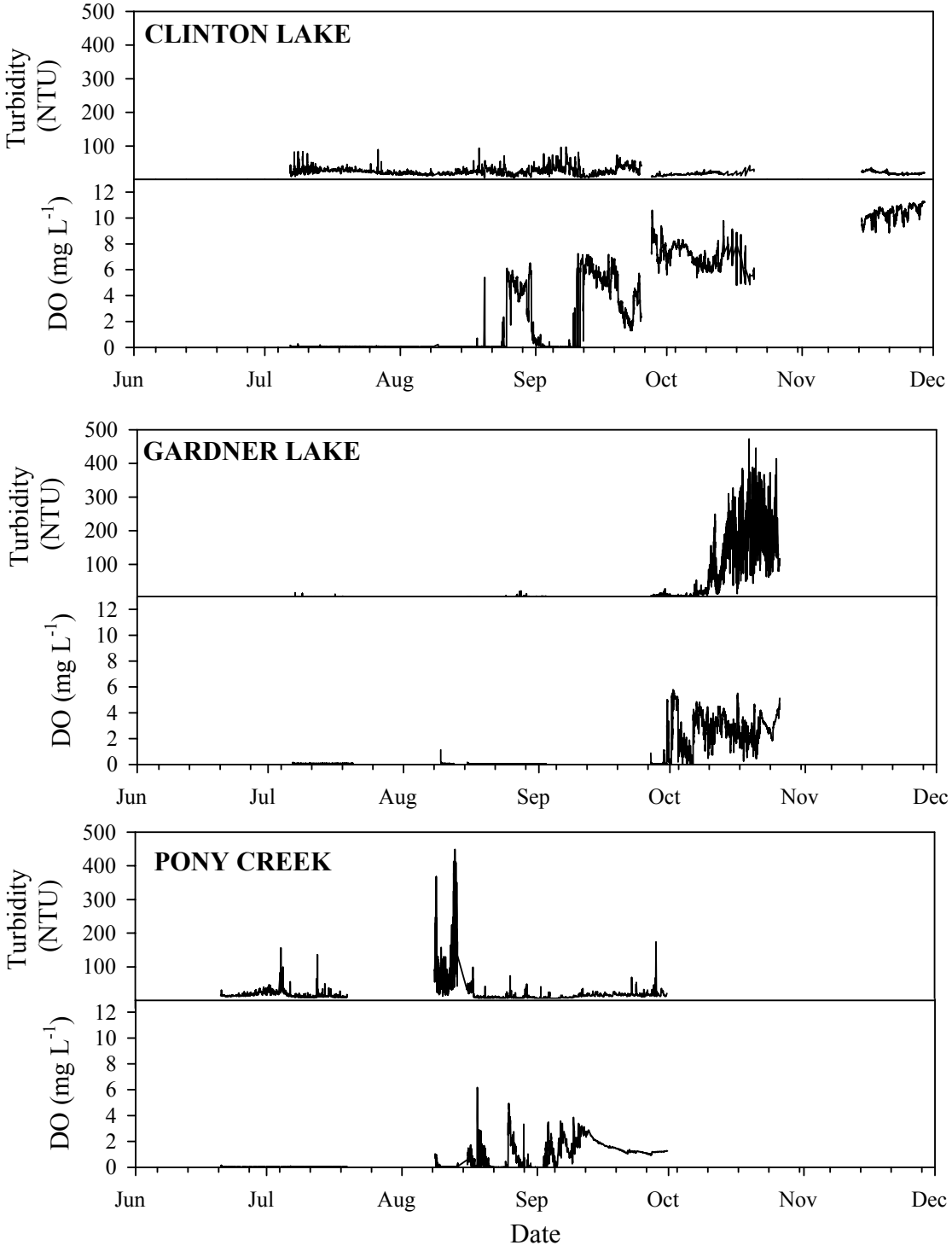

Figure 2. *In-situ* sediment/water interface measurements of turbidity and dissolved oxygen. Data was not collected from Pomona Lake due to a non-functional probe.

Anoxia was observed near the sediment surface of each reservoir, particularly during the early summer (Figure 2). In the two smaller reservoirs, increases in oxygen concentrations appeared to be associated with changes in turbidity concentrations in the late summer or early fall. In CL, there did not appear to be a relationship between dissolved oxygen and turbidity near the sediment surface (Figure 2). Instead, dissolved oxygen concentrations tended to increase into the fall/winter.

Sediment Bioassay Experiments

Algal biomass increased following sediment resuspension in the filtered and non-filtered bioassay experiments for each of the four reservoirs (significant sediment (S) effect, $P<0.001$ for each reservoir; Figure 3). PO was the only reservoir for which algal biomass was not significantly greater in all four sediment addition treatments than it was in the control treatments. Specifically, algal biomass in the 250 and 500 NTU treatments did not increase relative to the control treatment in the non-filtered bioassay, and while it did increase in these two treatments relative to the control treatment in the filtered bioassay, it did not increase to the levels observed in the 50 and 150 NTU treatments (Figure 3).

Algal biomass responded differently to sediment resuspension in the filtered and non-filtered bioassays (all W x T interactions, $P<0.001$; Figure 3). In the filtered bioassays, algal biomass did not increase in any of the control treatments; however, algal biomass tended to increase at or near day 6 in each of the sediment addition treatments. In contrast, algal biomass tended to increase in response to sediment resuspension at an earlier day in the non-filtered bioassays (Figure 3).

Total nutrient concentrations increased following sediment resuspension in the bioassay experiments for each reservoir (significant sediment (S) effect, $P<0.001$, for each reservoir; Figures 4 and 5). TP increased in each of the successively higher sediment addition treatments in all of the bioassays except for CL, where there were no differences between TP in the control and 50 NTU treatments (Figure 4). TN increased in each of the successively higher sediment addition treatments in the CL and GL bioassays (Tukey's, $P<0.05$; Figure 5). With respect to the PC and PO bioassays, TN was significantly greater in the control treatments than it was in each of the sediment addition treatments except 50 NTU's in PC and 50 and 150 NTU's in PO (Figure 5).

Increases in dissolved nutrient concentrations following sediment resuspension were more variable than those observed for total nutrient concentrations. There were increases in PO_4-P concentrations in at least some of the sediment addition treatments in all reservoirs except CL (Figure 6). There were also increases in NH_3-N concentrations in several of the sediment resuspension treatments relative to the control treatments in the GL, CL, and PC bioassays (Figure 7). Sediment resuspension did not have a significant effect on NO_3-N concentrations in either the GL or PO bioassays (Figure 8). In CL, the four sediment addition treatments had significantly higher NO_3-N concentrations than the control treatment in the filtered bioassay only. Sediment resuspension had the greatest effect on NO_3-N concentrations in the PC bioassay experiments, where there were

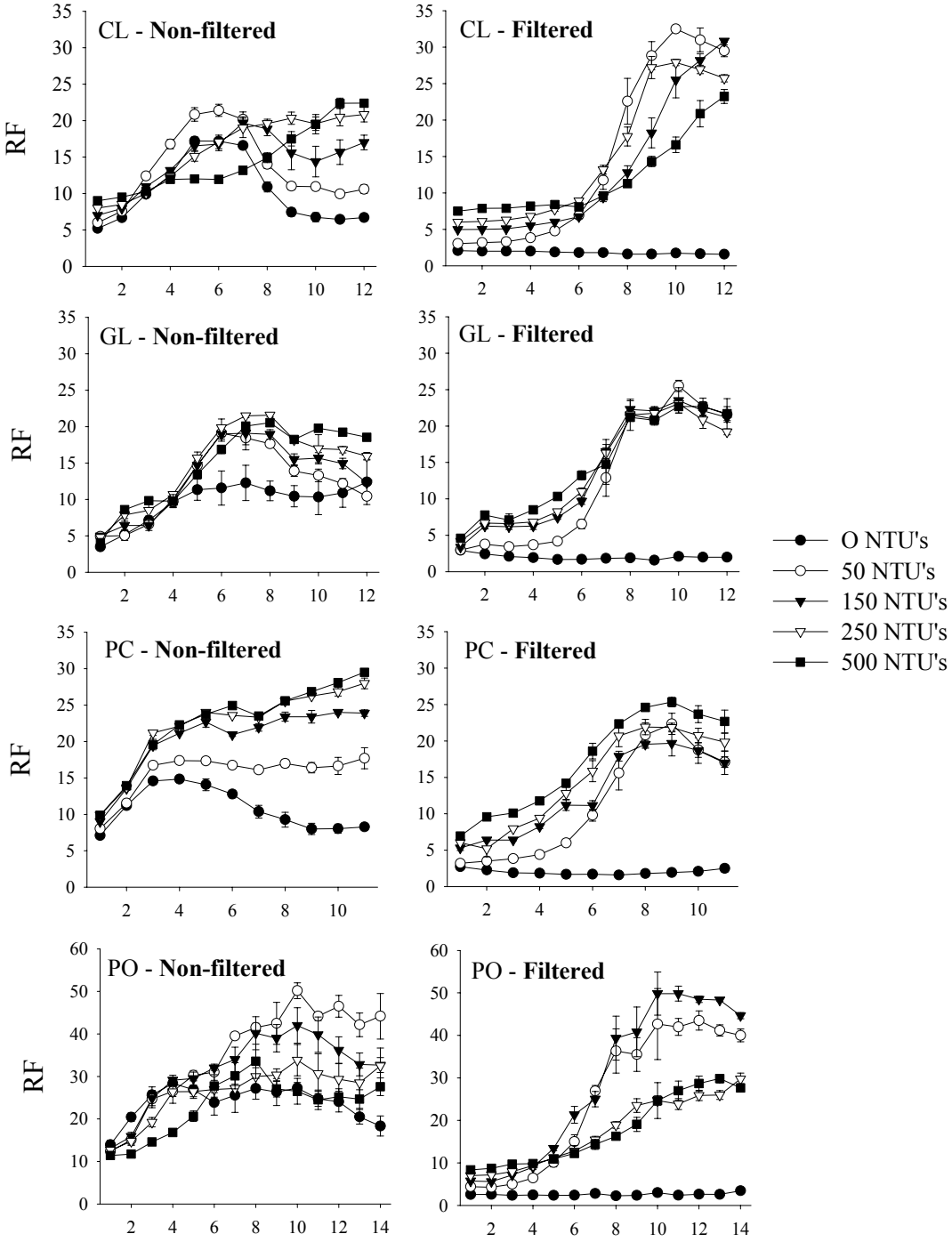

Figure 3. Relative fluorescence (RF) values from each bioassay experiment. Data are presented for both non-filtered (with algae) and filtered (without algae) reservoir water. Note differences in scale for Pomona Reservoir.

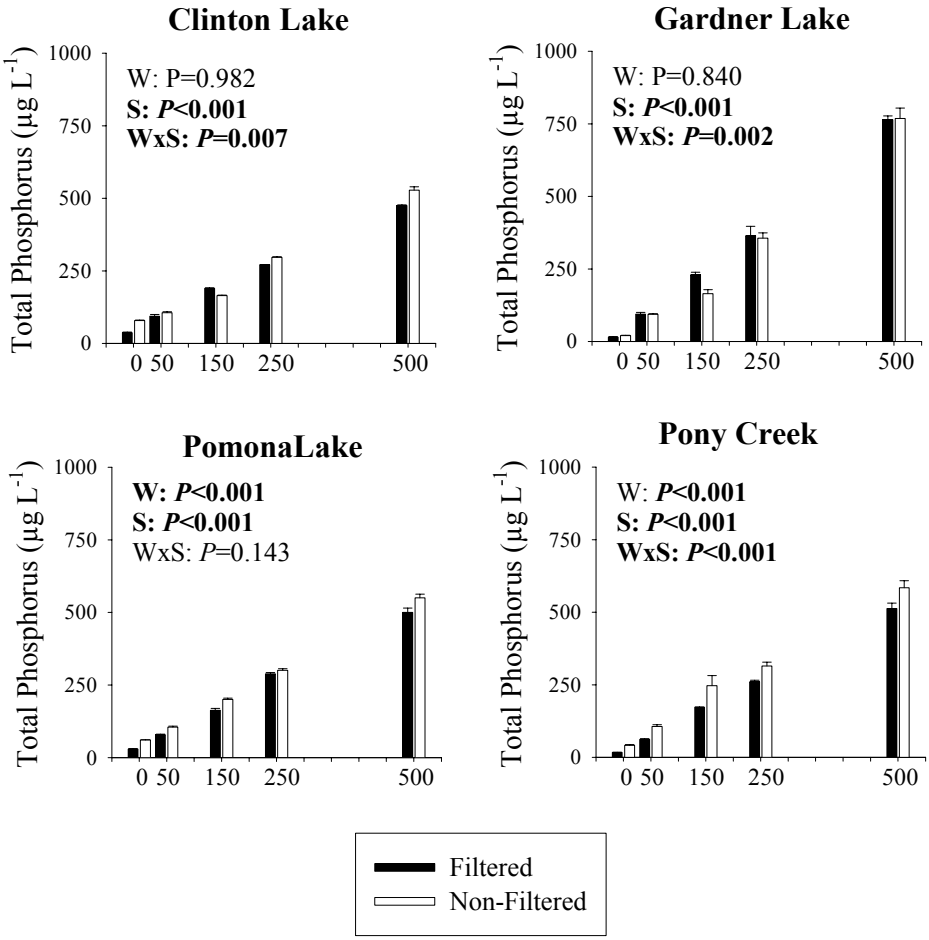

Figure 4. Total phosphorus concentrations measured at the beginning (day 1-3) of each bioassay experiment. Data is presented for both filtered (black) and non-filtered (white) treatments. Statistics for the reservoir water (W; filtered and non-filtered) and sediment (S; 0, 50, 150, 250 and 500 NTU's) treatments were determined with two-way ANOVA.

differences between the filtered and non-filtered bioassay experiments. In the filtered bioassay experiment, NO_3-N increased with sediment resuspension in the two highest sediment addition treatments (250 and 500 NTU's) relative to the control treatment. In contrast, NO_3-N concentrations were significantly lower in the four sediment addition treatments than in the control treatments in the non-filtered bioassay.

Nutrient Release Studies

P increased in all of the cores that were exposed to anoxic conditions (Figures 9-12) with the exception of two cores from the CL experiment, and one core from the PC treatment. P did not increase in these three cores suggesting that they were not anoxic, and therefore, they were removed from the analysis. Also, one core from the PO oxic treatment was lost and therefore not used.

P-release rates were significantly greater in anoxic cores than they were in oxic cores in each reservoir (significant core effect, $P<0.001$ for each reservoir; Figure 13). Specifically, release rates were up to 54, 29, 86, and 15 times greater in the anoxic treatments than in the oxic treatments in CL, GC, PC, and PO experiments respectively. P-release rates were significantly higher in CL cores than they were in cores from the other three reservoirs (reservoir effect, $P=0.018$); release rates were on average 53.1 mg P/m^2/day in the lacustrine zone cores from CL (Figure 13). There were no differences in release rates between the GL, PC, and PO cores.

Differences were observed in release rates for the three locations in each reservoir except CL (Figure 13). In GL, release rates were significantly greater in the cores from the lacustrine zone (average release rate was 39.8 mg P/m^2/day) than they were in the cores from the riverine zone (average release rate was 16.1 mg P/m^2/day) (Tukey's, $P<0.05$). In PO, release rates were significantly greater in the cores from the transition zone (average release rate was 47.1 mg P/m^2/day) than they were in the cores from the main basin zone (average release rate was 20.2 mg P/m^2/day) (Tukey's, $P<0.05$). Cores from all three zones of PC had significantly different release rates (Tukey's, $P<0.05$) (Figure 13).

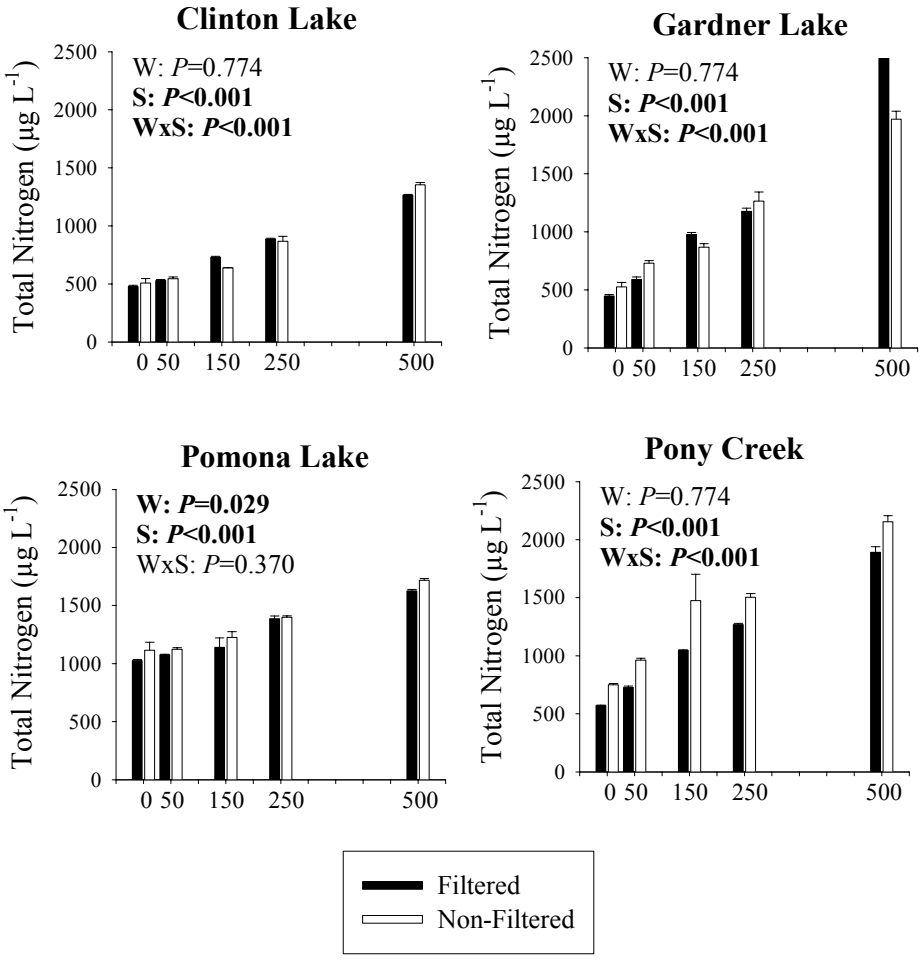

Figure 5. Total nitrogen concentrations measured at the beginning (day 1-3) of each bioassay experiment. Data is presented for both filtered (black) and non-filtered (white) treatments. Statistics for the reservoir water (W; filtered and non-filtered) and sediment (S; 0, 50, 150, 250 and 500 NTU's) treatments were determined with two-way ANOVA.

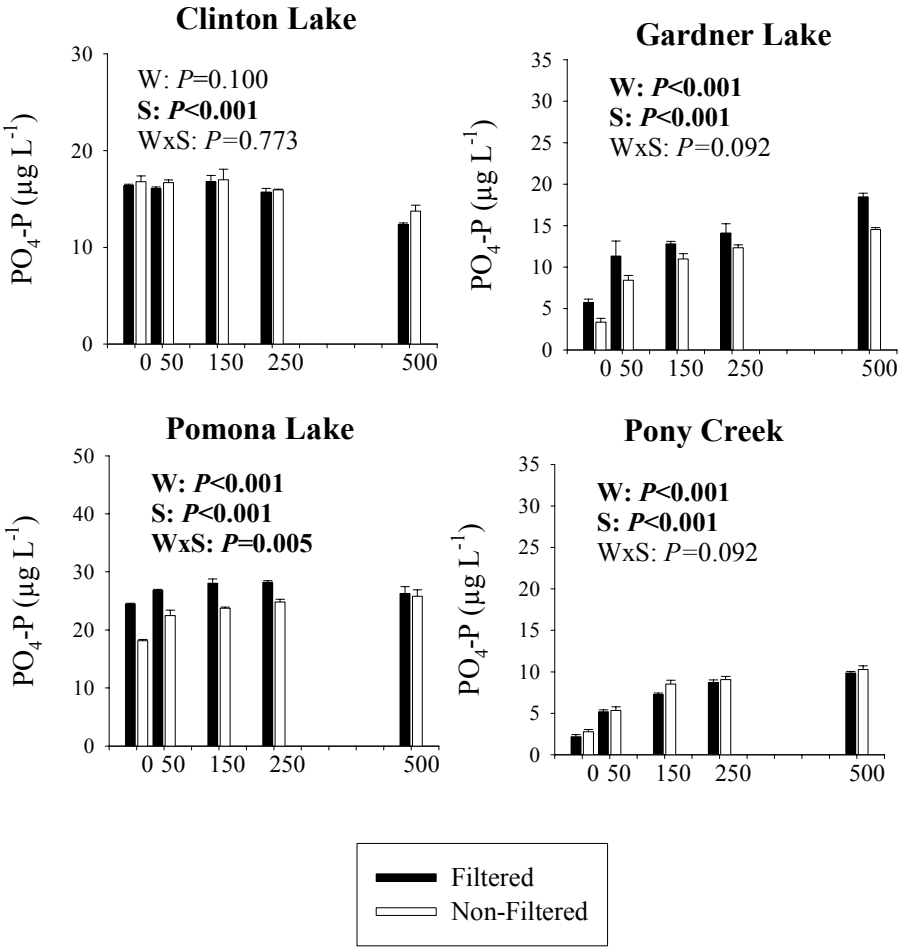

Figure 6. PO_4-P concentrations measured at the beginning (day 1-3) of each bioassay experiment. Data is presented for both filtered (black) and non-filtered (white) treatments. Statistics for the reservoir water (W; filtered and non-filtered) and sediment (S; 0, 50, 150, 250 and 500 NTU's) treatments were determined with two-way ANOVA.

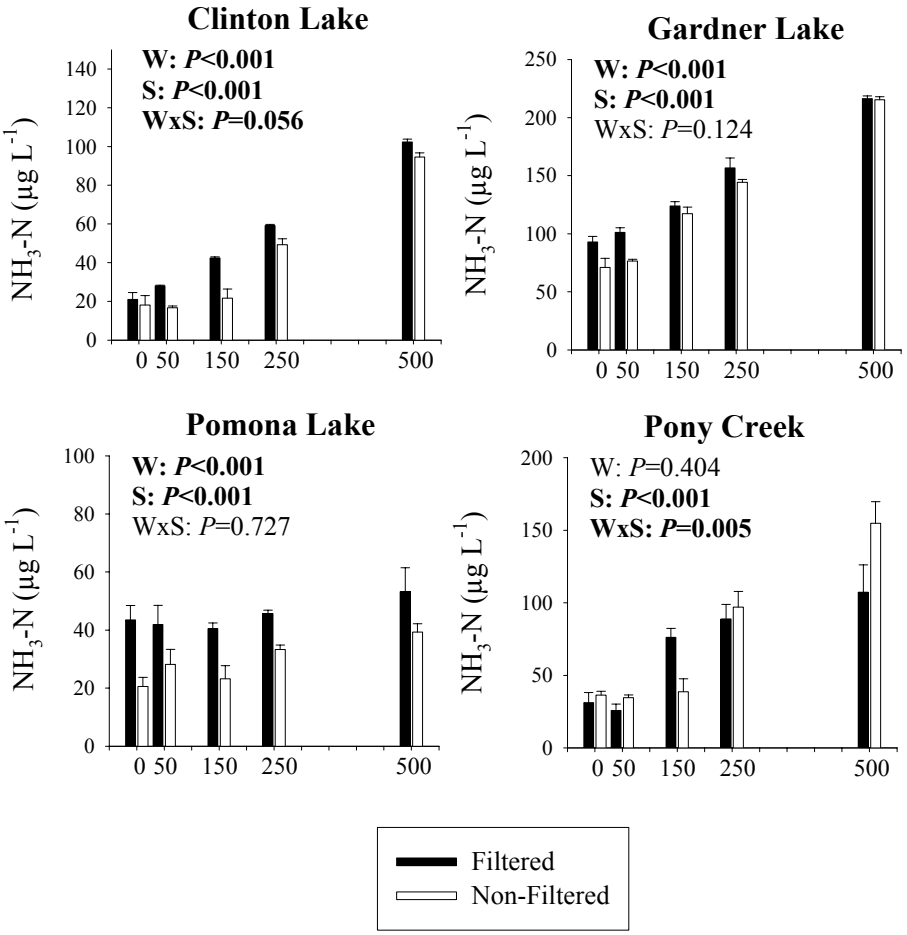

Figure 7. NH_3-N concentrations measured at the beginning (day 1-3) of each bioassay experiment. Data is presented for both filtered (black) and non-filtered (white) treatments. Statistics for the reservoir water (W; filtered and non-filtered) and sediment (S; 0, 50, 150, 250 and 500 NTU's) treatments were determined with two-way ANOVA.

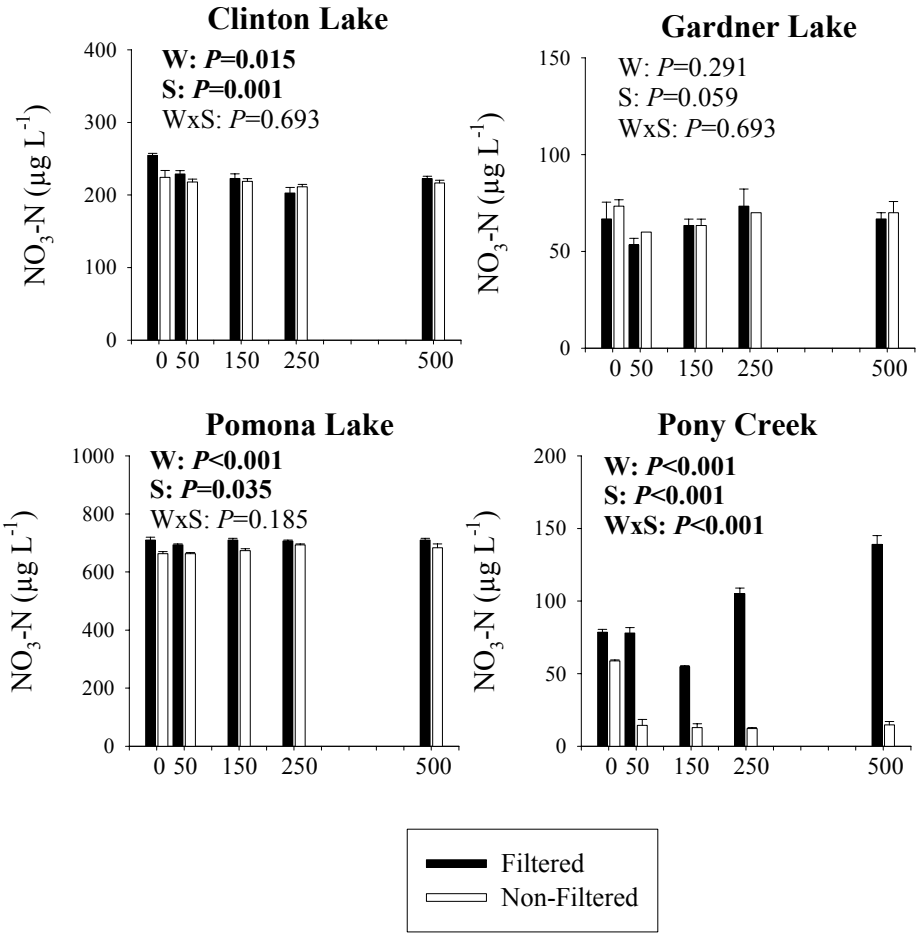

Figure 8. NO_3-N concentrations measured at the beginning (day 1-3) of each bioassay experiment. Data is presented for both filtered (black) and non-filtered (white) treatments. Statistics for the reservoir water (W; filtered and non-filtered) and sediment (S; 0, 50, 150, 250 and 500 NTU's) treatments were determined with two-way ANOVA.

Discussion

The four study reservoirs exhibited nutrient and chlorophyll concentrations that are generally characteristic of eutrophic lakes and reservoirs (Table 2; Smith, 1998). While the management of eutrophic systems has historically focused on controlling external nutrient loading, it is becoming increasingly clear that internal mechanisms can also contribute to eutrophication. Here we have shown that both sediment resuspension and internal P-release from the sediment can contribute large concentrations of nutrients and/or algae back into the water column. As such, our results strongly suggest that internal nutrient sources play important roles in the eutrophication of reservoirs in the Central Plains region.

In situ Near-Sediment Measurements

Elevated turbidity concentrations were observed near the sediment surface of each reservoir (Figure 2). Several processes can affect sediment resuspension including weather (i.e. wind and rainfall), reservoir water release, reservoir mixing patterns, and recreational use (e.g. Anthony and Downing, 2003; Schallenberg and Burns, 2004). Sediment resuspension in the two smaller reservoirs appeared to be associated with reservoir mixing. In these two reservoirs, increases in turbidity concentrations (near 500 NTU's) were closely related to increases in dissolved oxygen concentration presumably following reservoir turnover. Therefore, smaller reservoirs that experience long periods of thermal stratification throughout the summer are most vulnerable to sediment resuspension when they turnover in the late summer or early fall.

It is more difficult to determine the processes that affected sediment resuspension in the larger CL. Although CL appeared to be thermally stratified in July and part of August (data not shown), increases in sediment/water interface turbidity concentrations did not appear to be associated with thermal stratification or lake mixing. Instead, it is likely that a combination of weather, reservoir water release, and recreational use affected sediment resuspension in CL. Regardless of the mechanism, however, it is important to note that resuspended turbidity concentrations were observed in CL (>50 NTU's) that had highly significant effects on algal biomass in the subsequent laboratory bioassay experiments.

Each reservoir also experienced long periods of near-sediment anoxia during the study (Figure 2). In general, each of the smaller stratified reservoirs was anoxic in the hypolimnion until they experienced turnover in mid-August, and in the case of GL, October. Surprisingly, CL also experienced a long period of anoxia in the summer. It is generally assumed that because larger reservoirs in the region do not thermally stratify for long periods of time and as a result do not become anoxic, they do not become anoxic. However, our results clearly show that anoxic conditions existed near the sediment/water interface in CL from the beginning of the study until mid-August.

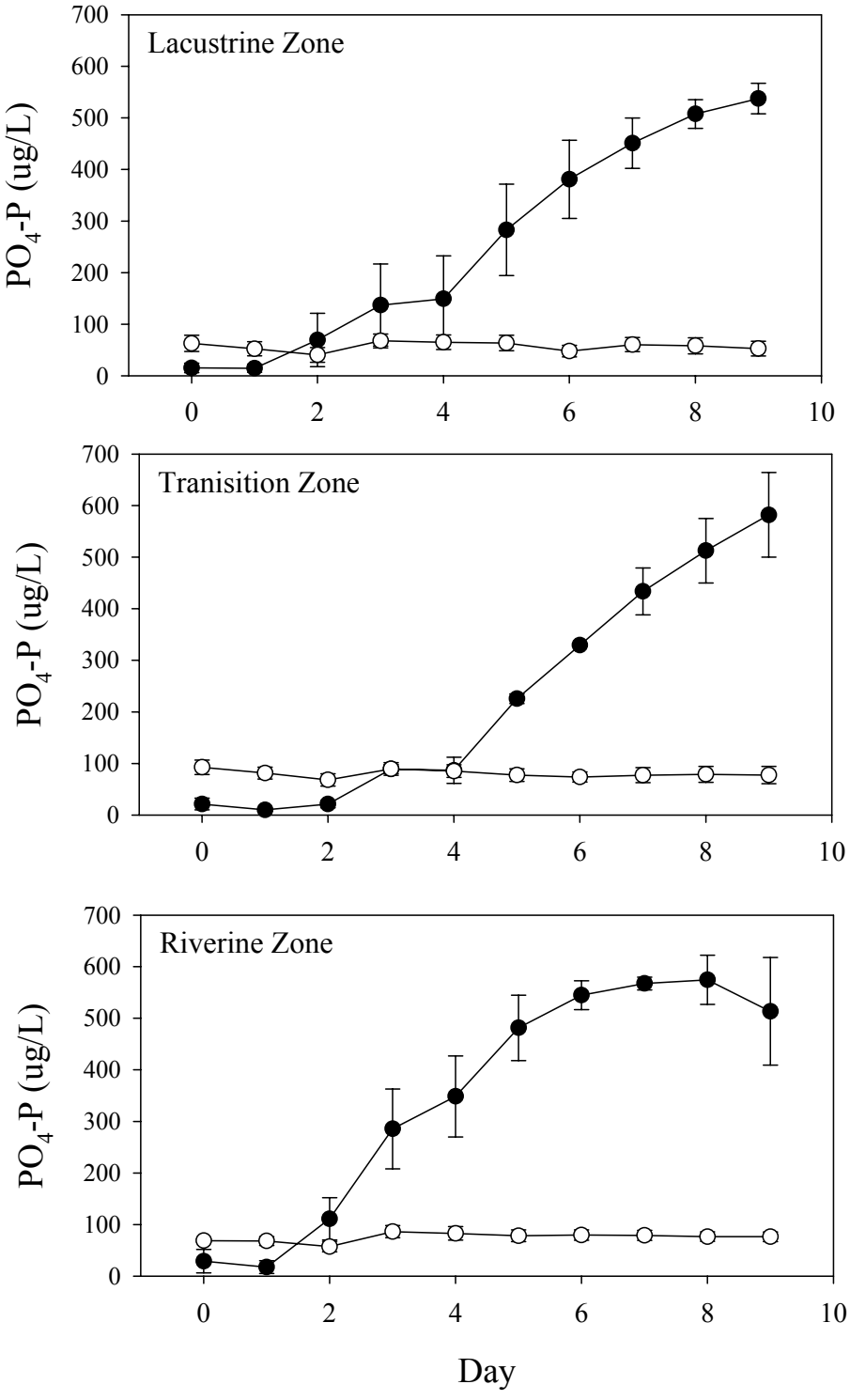

Figure 9. PO_4-P concentrations released from both anoxic (●) and oxic (○) cores from Clinton Lake.

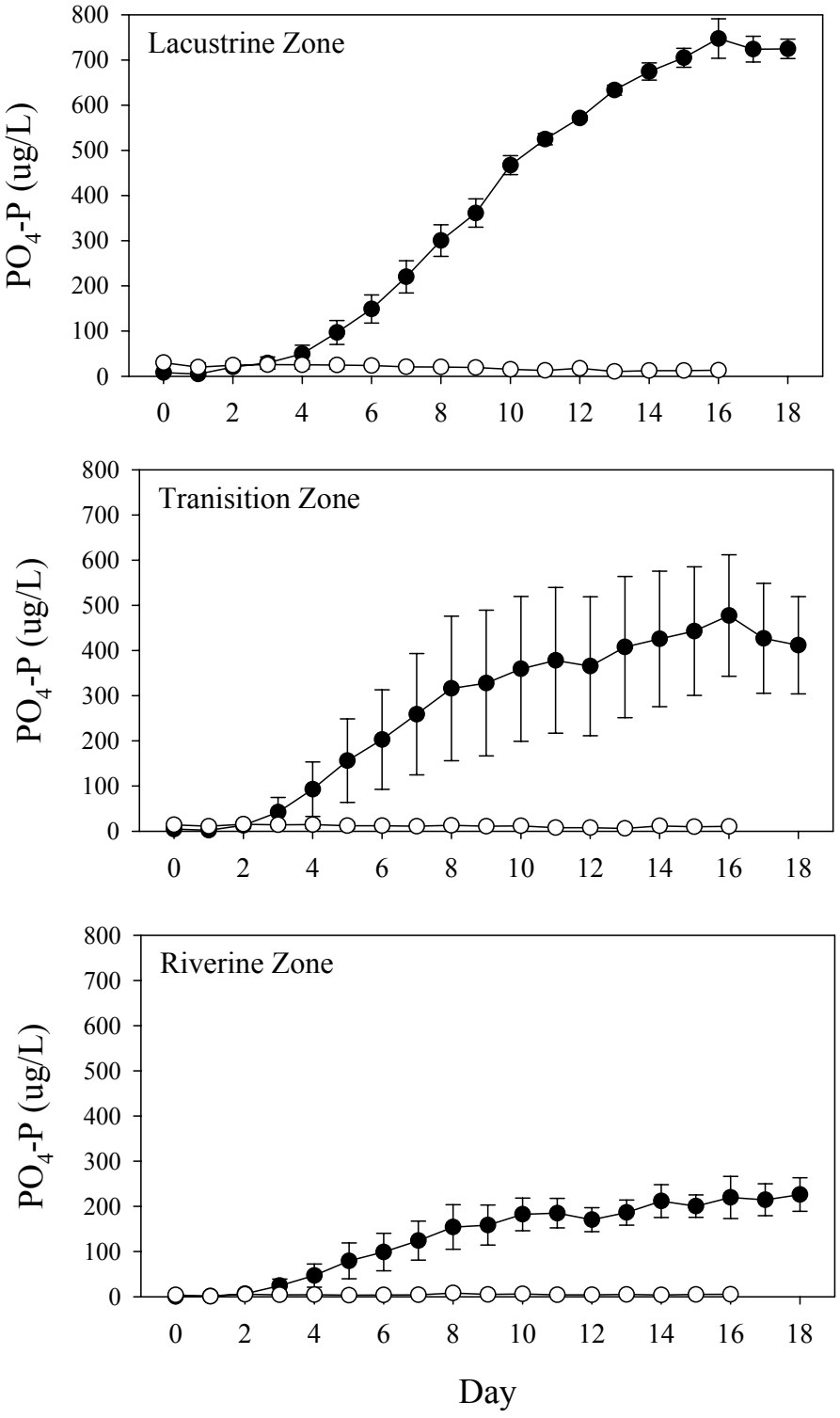

Figure 10. PO_4-P concentrations released from both anoxic (●) and oxic (○) cores from Gardner Lake.

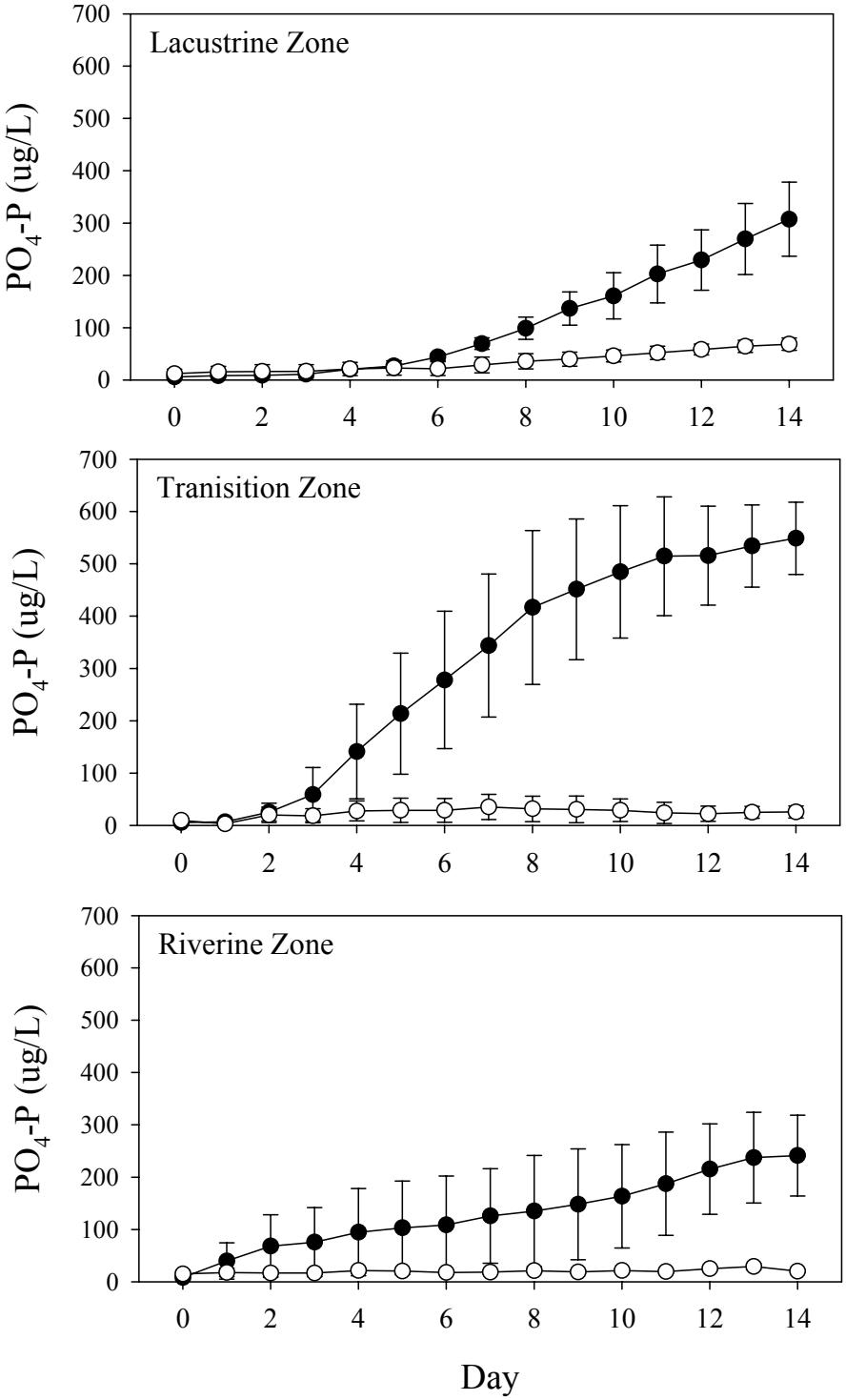

Figure 11. PO_4-P concentrations released from both anoxic (●) and oxic (○) cores from Pomona Lake.

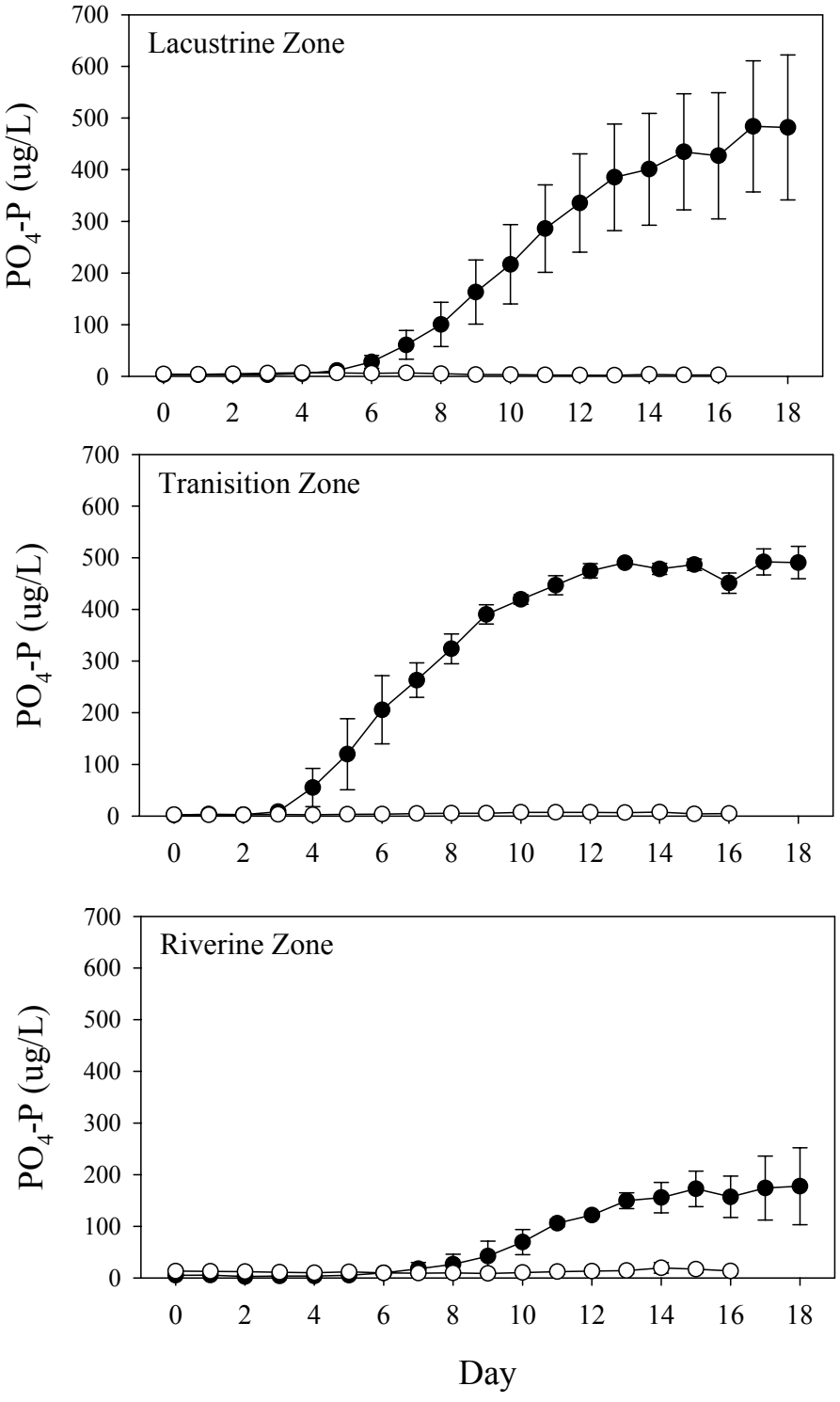

Figure 12. PO_4-P concentrations released from both anoxic (●) and oxic (○) cores from Pony Creek.

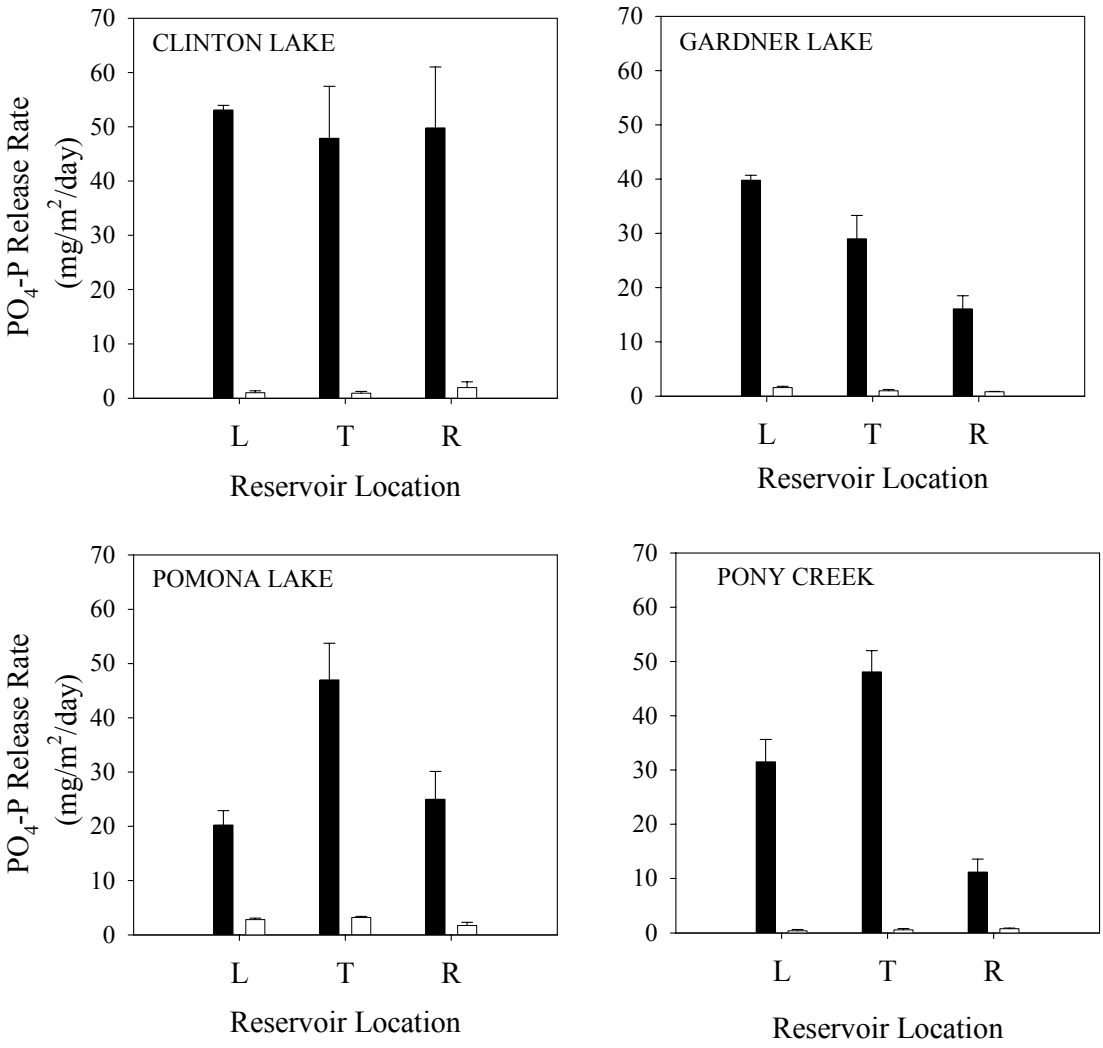

Figure 13. P-release rates calculated from the data presented in Figures 9-12. Data is presented for both anoxic (black bars) and oxic (white bars) cores. L=lacustrine; T=Transitional; and R=Riverine.

It should be noted that the *in-situ* probes only measured turbidity and dissolved oxygen concentrations in the main basins of the reservoirs. However, most reservoirs are spatially diverse having strong longitudinal gradients of physical and chemical properties. Shallower zones (e.g. transition and riverine zones) have a greater potential for wind induced sediment resuspension; however, these areas are less likely to stratify and become anoxic (Nowlin *et al*., 2005). Probes should be placed in each zone to quantify whole reservoir sediment resuspension rates and the occurrence of anoxia. However, our results show that sediment resuspension at lacustrine zone sites commonly results in turbidity concentrations that can have significant effects on reservoir function (see below), and that anoxic conditions commonly occur creating conditions that favor sediment nutrient release in Central Plains reservoirs (Bostrom *et al*., 1988).

Sediment Bioassay Experiments

Algal biomass increased following resuspension in the bioassays from each of the four reservoirs (Figure 3). A number of studies have shown that resuspended sediments often result in increases in the concentrations of N and P in the water column, and in at least some studies, a release of resident algae from nutrient limitation (e.g. Ogilvie and Mitchell, 1998; Schallenberg *et al*., 2004). Dzialowski *et al*. (2005) conducted bioassay experiments with water collected from 19 reservoirs from the Central Plains (including GL and PO) and found that algae were often nutrient limited, with the majority of reservoirs being co-limited by N and P. Resuspended sediments resulted in highly significant increases in the concentrations of both TN and TP relative to control conditions in each reservoir (Figures 4 and 5). Dissolved nutrient concentrations following resuspension were more variable than total nutrient concentrations (Figure 6-8) (see below). However, dissolved nutrients are available for direct uptake by resident algae or meroplankton so that it is difficult to assess nutrient limitation from dissolved nutrient concentrations alone.

In contrast, it is possible that algal biomass did not increase because of resuspended nutrients, but because of the establishment of meroplankton (dormant algal cells within the sediment) that became entrained in the water column following sediment resuspension events (e.g. Schelske *et al*. 1995). The results from the filtered bioassay experiments allowed us to test this hypothesis because resident algal communities were initially removed from the source water. As such, algal biomass did not increase in any of the control treatments during the bioassay experiments. In contrast, there were significant increases in algal biomass in most of the resuspension treatments (Figure 3). Unfortunately, we do not know if the algae that established within the bioassay experiments were similar species to those originally in the source water before it was filtered. However, environmental conditions in the water column can help to determine what species are able to establish from the sediment following resuspension events. Nalewajko and Murphy (1998) found that 11 algal species from the sediment of Lake Biwa were viable and able to establish in bioassays under varying nutrient and light conditions. Sediment resuspension events might be important factors leading to blooms of nuisance algal species such as cyanobacteria. For example, if cyanobacterial cells become entrained in the water column during resuspension events, they may have a

greater probability of establishing under N-limiting conditions because of their ability to fix atmospheric nitrogen (Scheffer, 1998). Verspagen *et al*., (2005) reported that recruitment of cyanobacteria from the sediment was prevalent in a eutrophic lake, accounting for up to 50% of the summer blooms. Therefore, the timing of resuspension events in combination with water quality conditions may be particularly important in determining how resuspended sediments affect algal species composition in reservoirs of the Central Plains.

Based on the available data, it is difficult to determine if nutrients or meroplankton from the resuspended sediments had a greater relative impact on algal biomass in the bioassay experiments. Several studies have shown that it can be difficult to tease apart the relative impacts of these two important processes. Schallenberg *et al*. (2004) reported that resuspended sediments rarely released algal communities from nutrient limitation in a meso-trophic lake. Instead, resuspended sediments more often contributed meroplankton to the water column leading to direct increases in algal biomass. However, Ogilvie and Mitchell (1998) found that resuspended sediments released algal communities from nitrogen limitation in at least some of their New Zealand study lakes. It is likely that both processes are important in Central Plains reservoirs and that resident algal communities, available nutrients, and environmental conditions help to determine which process has a greater relative impact during a particular resuspension event. Additional bioassay studies that include detailed algal community composition data will help to determine under what conditions resuspended nutrients or meroplankton have greater relative impacts on algal biomass following resuspension events.

PO was the only reservoir in which the results suggest that the highest turbidity concentrations had negative effects on algal biomass. For example, there was not an increase in algal biomass in the 250 and 500 NTU treatments relative to the control treatment suggesting that algal communities were light limited at elevated turbidity levels in PO. Schallenberg and Burns (2004) reported that algae experienced light limitation at similarly high turbidity concentrations (>200 NTU's). It is possible that light limitation was not observed in the bioassay experiments from the other three reservoirs because there were shifts in species composition to taxa that perform well at low light conditions at high turbidity levels (e.g. Padisak *et al*., 1990). However, the current experiments were not designed to explicitly test the hypothesis that resuspension events affected light conditions because the simulated resuspension events occurred daily, and were therefore not continuous. Additionally studies are needed to assess how the duration of individual sediment resuspension events impact algal biomass for extended periods of time.

It is important to note that resuspended sediments and resulting increases in turbidity concentrations can also have significant impacts on food web components not studied here. For example, high turbidity concentrations have been shown to impact bacteria (e.g. Pusceddu *et al*., 2005) zooplankton (e.g. Levine *et al*., 2005) and fish (e.g. Nurminen and Horppila, 2006). As such, additional research is needed to determine how sediment resuspension affects additional food web components in reservoirs in the Central Plains.

Nutrient Release Studies

Sediment cores exposed to anoxic conditions released large concentrations of P into the water column in each experiment (Figures 9-12). Furthermore, P-release rates tended to differ between at least some of the reservoirs, and within the three zones in the majority of individual reservoirs. These results are consistent with a number of previous studies (e.g. Nurnburg 1988; Weston *et al*., 2005) and highlight the importance of internal P-release in reservoirs of the Central Plains.

Sediment nutrient release rates are highly correlated with sediment P concentrations (Nurnburg, 1988). Nurnburg (1988) presented regression equations using lakes from throughout the world and found that lakes with higher sediment P concentrations had higher P-release rates under anoxic conditions. Based on the relationships presented in Nurnburg (1988), we can infer that sediment nutrient concentrations were greatest at CL because it had the highest release rates. Similarly we can infer that sediment nutrient concentrations tend to be greatest in the lacustrine zone of GL, and the transition zones of PO and PC (Figure 13). Therefore, it is important to consider spatial variation when assessing internal nutrient cycling. Furthermore, a single sediment P-release rate cannot be assumed for all reservoirs within a region (Gachter and Muller, 2003).

Although sediments released relatively high concentrations of P under anoxic conditions, it is important to note that additional abiotic and biotic processes affect actual in-lake release rates. For example, NO_3 concentrations in the water column can have either a positive or negative effect on release rates depending on bacterial activity (Bostrom *et al*., 1988). Nowlin *et a*l. (2005) reported that when they added NO_3 to sediment cores, there was a substantial decrease in P-release rates relative to control cores. Furthermore, Gachter and Muller (2003) showed that anoxia is not always a good indicator of release rates. They found that P-release rates from the sediment did not always decrease when the hypolimnion of a eutrophic reservoir was oxygenated (Gachter and Muller, 2003). While the concentrations of P-released in the current study appear to be high, it is important to consider how processes within the individual lakes affect actual release rates. Therefore, additional research is needed to monitor abiotic and biotic factors that influence P-release rates (such as NO_3 concentrations and bacterial activity) during periods of anoxia in order to quantify total lake release rates.

A number of studies have concluded that internal P cycling can hinder restoration efforts so that water column P concentrations do not respond to external load reductions (Marsden, 1989). For example, Shagawa Lake, Minnesota remained eutrophic despite reductions of external P-inputs due to sediment release that occurred during periods of anoxia (Larson *et al*., 1981). In contrast, others have shown that internal P-release from the sediment does not always hinder restoration efforts. Reductions in external nutrient loads resulted in lower water column P concentrations in a shallow, eutrophic lake in Florida despite the effects of sediment resuspension and nutrient release under anoxic conditions (Coveney *et al*, 2005).

Conclusions

In-situ sediment resuspension events were observed near the sediment/water interface of each reservoir. Higher maximum turbidity concentrations were observed in the smaller reservoirs (near 500 NTU's) than in the larger reservoir (~100 NTU's). However, even relatively low turbidity concentrations (50 NTU's) had significant effects on algal biomass and nutrient concentrations in each of the subsequent sediment resuspension bioassays. Evidence suggests that resuspended sediments affected algal biomass in several ways. For example, resuspended nutrients can release resident algal communities from nutrient limitation, and meroplankton, algal cells within the sediment, can become entrained in the water column following sediment resuspension events.

Relatively long periods of anoxia were also observed near the sediment/water interface of each reservoir. In the two smaller reservoirs, anoxia appeared to be associated with reservoirs turnover as they were anoxic until the late summer or early fall. Surprisingly, the larger reservoir was also anoxic during most of the summer, despite a lack of thermal stratification. Laboratory sediment core incubation experiments showed that under anoxic conditions, sediment from each reservoir released high concentrations of P relative to control (oxic) cores. For example, P-release rates in anoxic cores were up to 54 times greater than in oxic cores.

The results from the *in-situ* probes combined with the laboratory experiments, suggest internal nutrient cycling is an important process in reservoirs of the Central Plains. Specifically, our results indicate that resuspension events occur in both small and large reservoirs, and that these events can increase turbidity concentrations to levels that have significant effects on algal biomass and nutrient concentrations. Additionally, the i*n-situ* oxygen measurements and the high P-release rates observed under anoxic conditions indicate that reservoirs have a high potential for sediment nutrient release. Overall, our results highlight the importance of considering internal nutrient cycling in management and restoration projects.

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