Algal Growth Response in Two Illinois Rivers Receiving Sewage Effluent

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ABSTRACT

Phosphorus (P) primarily enters streams in Illinois as effluent released from sewage treatment plants and runoff from agricultural fields. As a result, water quality can be affected and large amounts of algal growth are possible. We determined the growth of periphytic algae (as chl*a*) relative to differing amounts of P (factor of 10) released in sewage effluent in two rivers. The Salt Fork Vermilion River and the Copper Slough branch of the Kaskaskia River both have a sewage treatment plant near their sources. Periphytic algal growth was assayed in each river with unglazed ceramic tiles (five week period) at 10 sites, each 10 km apart downstream from where the treatment plant was located. Field measurements included canopy cover, turbidity, water depth (to the tile surface), and water temperature. The concentrations of sestonic algae (as chl*a*), total P, dissolved reactive P, nitrate-N, dissolved organic carbon, and Si were determined in water samples. Total P concentrations were different between the two rivers, ranging from 1.9 mg L^{-1} just below the Salt Fork Vermilion River plant to 0.67 mg L^{-1} 90 km downstream; corresponding values were 0.19 and 0.16 mg L^{-1} for the Kaskaskia River. Phosphorus concentrations were not related to sestonic or tile periphytic chl*a* in either river. Canopy cover, turbidity, and unstable sediments apparently regulated algal growth by limiting the penetration of light. Therefore, P was not the primary regulator of algal growth, and removing sewage effluent P from these rivers is unlikely to alter algal growth.

INTRODUCTION

The growth of algae in rivers in Illinois is controlled by various factors such as nutrient concentrations, turbidity, and light (Figueroa-Nieves et al. 2006). Although each of these factors can regulate algal production, phosphorus (P) has been thought to have a critically important effect on algal growth (Van Nieuwenhuyse and Jones 1996, Dodds et al. 1998). In Illinois, P is added to streams from both sewage effluent and agricultural runoff (David and Gentry 2000, Gentry et al. 2007). The increase in P and other nutrients to rivers has been thought to cause environmental degradation because it decreases the quality of the water and can cause eutrophication. As a result, there can be an increase in algal blooms, which can be toxic to other organisms. Eutrophication can result in large fish kills because the amount of oxygen in the water at night is reduced as a result of the large algal blooms (Morgan et al. 2006). Loss of biodiversity is also a consequence because many organisms are lost that cannot tolerate these conditions.

Nutrient enrichment, particularly for P in fresh waters, is commonly listed as a primary impairment in midwestern rivers (USEPA 2000). States are currently under federal pressure to develop nutrient criteria (Dodds and Welch 2000, USEPA 2007) that would be likely to include concentration standards for nitrogen (N) and P in streams and rivers. If standards are put into place, one of the first regulated sources is likely to be sewage effluent, primarily because it is a permitted source with technology available to reduce both nutrients. However, it is not clear for many midwestern rivers if sewage effluent N and P were greatly reduced (even to zero) that stream quality would improve

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unless there was a concomitant reduction in N and P inputs from agricultural sources. Reductions in N and P from agricultural fields will be much more difficult to achieve in the Midwest, given the intensity of agriculture and extensive artificial drainage (David et al. 1997, David and Gentry 2000).

Previous research has been performed on the topic of algal growth in many rivers in Illinois to determine some of the regulators of algal growth. Two related studies were performed, and they resulted in similar findings. In the first study by Morgan et al. (2006), samples were collected to measure the N and P concentrations as well as the turbidity, canopy cover, and dissolved oxygen concentration of the water in five streams in eastcentral Illinois. Samples of sestonic and periphytic algae were taken and the amount of chl*a* was measured. The study found that nutrients did not have a significant impact on the growth of the algae but that the amount of light had more of an impact. In the streams with more shade, there was less periphyton and sestonic chl*a* than in streams with no shade (Morgan et al. 2006). In another study (Figueroa-Nieves et al. 2006), eighteen streams throughout eastcentral Illinois were sampled and measured for nutrients and turbidity. Ceramic tiles and natural rocks were used to collect chl*a*. It was concluded that in addition to nutrients, the abundance of light helps to control the growth of algal biomass.

The objective of our study was to assess the response of algal growth to P in streams located in an agricultural area of Illinois but which also receive sewage effluent. Phosphorus enters these streams through discharge from sewage plants upstream as well as from runoff of the surrounding agricultural fields. The two specific streams were chosen because both are located downstream from a sewage treatment plant. However, each plant releases differing amounts of P through its discharged effluent, providing a wide range of P concentrations. Therefore, we were able to evaluate how these streams might respond to a decrease in P concentrations if sewage effluent discharge were regulated and reduced.

MATERIALS AND METHODS

Study sites

Water and algae samples were collected from two rivers in Illinois, the Salt Fork Vermilion River and the Kaskaskia River, with headwaters in Champaign County, Illinois. Both are located downstream of Champaign-Urbana sewage treatments plants, which release discharge containing P. The plant upstream of the Kaskaskia River on Copper Slough is required to remove P to a concentration less than 1 mg L^{-1}, while the plant upstream of the Salt Fork Vermilion River on the Saline Branch does not remove P and therefore has effluent concentrations of total P between 3 and 5 mg L^{-1}. Both rivers were divided into ten longitudinal sites for the water sampling and collection of algae. These sites were 10 km apart along a total distance of 90 km. The Salt Fork Vermilion River and Kaskaskia River are both fourth order streams with drainage areas of approximately 1,660 and 1,240 km^2, respectively, at the most downstream sampling locations. The streambeds of both rivers consisted of silt, sand, and rocks depending on the location of the site. There were agricultural fields bordering both rivers, as the watersheds of both rivers are > 90% row crop agriculture.

Water sampling and analysis

Sampling was performed at each of the 20 sites at areas of the rivers that were wadeable. Water samples for nutrient analysis were collected at all ten sites in each river as were three samples taken at the center and near both banks for sestonic algae analysis. Samples were collected during the fall of 2006 in the Kaskaskia River on September 19 and 20 and October 24 and in the Salt Fork Vermilion River on September 20 and 21 and October 25 and placed on ice in the field before returning to the laboratory. Samples were preserved and processed in accordance with standard methods (APHA 1998). Samples analyzed for NO_3-N, NH_4-N, dissolved reactive phosphorus (DRP), and silica were filtered through a 0.45 μm membrane. Nitrate concentrations were measured using ion chromatography (DX-120, Dionex, Sunnyvale, California) with a detection limit of 0.1 mg L^{-1} of NO_3-N. Ammonium, DRP, and silica concentrations were analyzed colorimetrically by flow injection analysis with a QuikChem® 8000 (Lachat, Loveland, Colorado) using the automated sodium salicylate, automated ascorbic acid, and automated heteropoly blue methods, respectively. Method detection limits were 10 μg NH_4-N L^{-1}, 5 μg P L^{-1}, and 200 μg SiO_2 L^{-1}. Dissolved organic carbon (DOC) concentrations were analyzed on a Dohrmann-Xertex DC-80 analyzer with a detection limit of 0.5 mg DOC L^{-1}. Unfiltered water samples were analyzed for total P and were digested with sulfuric acid and ammonium persulfate, converting all forms of P into DRP, and then analyzed as above.

Turbidity was measured in the field on samples from the left, center, and right portions of the river channel with a portable turbidimeter (Model 966, Orbeco-Hellige, Farmingdale, New York). The water temperature and the amount of light were continuously (15 min interval) measured at three sites in each river with a HOBO temperature and light sensor pendant that was placed on the river bed near where one of the tiles was located (Onset Computer, Bourne, Massachusetts). The percent canopy cover was also measured in the rivers at each site with a densiometer.

Periphytic and sestonic chlorophylla

Potential growth of benthic algae (periphytic chl*a*) was assessed at each site using an artificial substrate; these were unglazed beige ceramic tiles, 20-cm square. In September, tiles were anchored in the center of the streambed at each site at three transects separated by 10 m. The depth of the water at the tiles was also measured. After five weeks, the tiles were removed, rinsed with stream water, and placed in plastics bags. They were then transported to the laboratory in a dark cooler and held at -20º C until processed. The tiles were rinsed with distilled deionized water, and a sample of the periphytic chl*a* was scraped from a known area of each tile, the area being representative of the periphytic growth on the entire tile. The algae was then filtered (0.7 μm Whatman GF/F) and frozen until extracted following methods detailed in Morgan et al. (2006). For chl*a* extraction, each filter was placed in 10 mL of 90% ethanol, followed by sonnication for 30 sec and centrifugation for 20 minutes. Chl*a* was determined using a UV-Vis spectrophotometer (Aquamate, ThermoElectron, Waltham, MA). To correct for pheophytin, absorbance was

Table 1. Mean concentrations of total P, dissolved reactive P (DRP), nitrate-N, dissolved organic carbon (DOC), and Si in the Salt Fork Vermilion River and Kaskaskia River for two sampling dates in September and October 2006 (n=10).

River and sampling date	Total P	DRP	NO_3-N	DOC	Si as SiO_2
	----------	----------	mg L^{-1}	----------	----------
Salt Fork Vermilion River					
21-Sep-06	1.30	1.10	5.0	4.9	7.9
25-Oct-06	0.84	0.70	6.9	3.6	9.4
Kaskaskia River					
19-Sep-06	0.24	0.13	1.6	5.3	5.6
24-Oct-06	0.20	0.17	3.4	4.2	8.4

determined before and after acidification as described in Morgan et al. (2006). All chl*a* procedures were carried out in subdued light.

Water samples (500 mL) for sestonic algal analysis were collected at each site in each river at the beginning and end of the tile incubations. For each sestonic chl*a* sample, a known volume of water was filtered through a Whatman GF/F filter (0.7 µm), and the filters were immediately placed in individual plastic petri dishes, wrapped in aluminum foil, and frozen. The filter was placed in 10 mL of 90% acetone, and sonication was used to extract the chl*a*. The samples were then placed in a centrifuge for 20 minutes, and the absorbance was measured with a spectrophotometer in the same manner as the periphytic chl*a*.

Discharge and precipitation

Discharge was monitored by the U.S. Geological Survey at Chesterville (station no. 05590950) and Cooks Mills (station no. 05591200) along the Kaskaskia River. There was no gauge on the Salt Fork Vermilion River, but discharge was estimated by the gauge on the Middle Fork Vermilion River near Oakwood (station no. 03336645), which was located nearby. Daily precipitation in both watersheds was estimated using data from the National Climatic Data Center for sites at Rantoul, Sidell, and Mattoon and from the Illinois State Water Survey for Champaign (National Climatic Data Center 2007 and Illinois State Water Survey 2007). The average rainfall for Champaign and Mattoon was used for the Kaskaskia River watershed, and the average for Champaign, Rantoul, and Sidell was used for the Salt Fork Vermilion River watershed.

Statistical analyses

Linear regression was used to examine P concentration as a function of distance downstream from the sewage treatment plant in each stream, as well as sestonic and periphytic chl*a* and total P concentrations. Overall mean concentrations of sestonic and periphytic chl*a*, canopy cover, total P concentrations, and turbidity in each stream were compared using t-tests. All statistics were computed using SAS version 9.1.

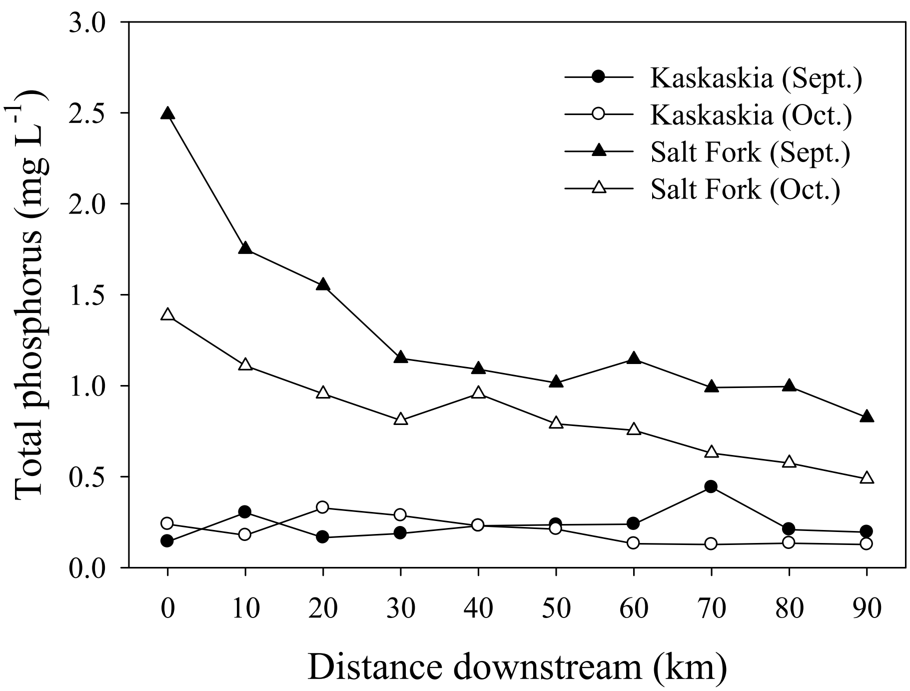

Figure 1. Mean total P concentrations for the Kaskaskia River and Salt Fork Vermilion Rivers in September and October 2006 at each site downstream from the wastewater treatment plant.

RESULTS AND DISCUSSION

During the first four weeks of the study, discharge in both rivers was consistent with the long-term median discharge for the time of year (USGS 2007). However, on October 17, 2006, widespread rainfall affected both watersheds. The Kaskaskia River watershed had rains exceeding 50 mm at its headwaters with less than 1 mm downstream; this increased discharge the next day but not enough to scour periphyton from any tiles. The Salt Fork Vermilion River had widespread rains (average 48 mm) across the watershed that caused discharge to increase two days later, but it appeared that periphyton growth already established on the artificial substrate was not affected.

Stream water total P concentrations were much greater in the Salt Fork Vermilion River than the Kaskaskia River (Table 1, Fig. 1). In the Salt Fork Vermilion River, the concentration of P was greatest at the site located closest to the wastewater treatment plant with an average total P concentration of 1.9 mg L^{-1}, with the concentration significantly decreasing to 0.66 mg L^{-1} as the water moved 90 km downstream ($p < 0.01$). In the Kaskaskia River, concentrations of P were much lower leaving the plant and did not decrease significantly downstream in the September sampling ($p > 0.05$) but did slightly in October ($p = 0.02$). Near the plant, the average P concentration was 0.19 mg L^{-1}, and the site 90 km from the plant had an average P concentration of 0.16 mg L^{-1}. This consistently low concentration of P was likely due to agricultural inputs, which continually add P to the river through surface runoff and tile drainage (Gentry et al. 2007). Even though the total P concentration was low in the Kaskaskia River as compared to the Salt Fork Vermilion River, both streams had total P concentrations that were high at every site when considering stream water quality from a eutrophication perspective (Dodds et al. 1998). These concentrations were likely above concentrations where P would be limiting to algal growth (Dodds et al. 1998, Dodds and Welch 2000). Dissolved reactive P was about 60 to 88% of the total P concentration, and the percent DRP of total P was lower and more variable in the Kaskaskia River than the Salt Fork

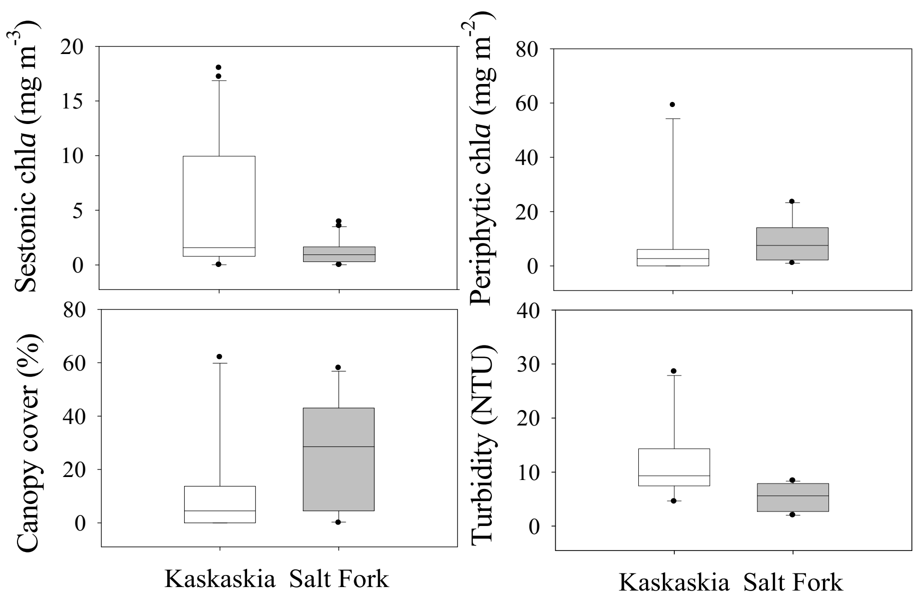

Figure 2. Box-and-whisker plots of sestonic chl*a*, periphytic chl*a*, canopy cover, and turbidity in the Salt Fork Vermilion River and the Kaskaskia River. The 25th, 50th, and 75th percentiles are shown by the horizontal lines in the box. The whiskers show the 10th and 90th percentiles, and the solid circles depict values outside the 10th and 90th percentiles. An * indicates that means were significantly difference at the 0.05 level, whereas NS indicates no significant difference.

Vermilion River. This variability in DRP in the Kaskaskia River was most likely due to the fluxes in agricultural runoff of P. Mean nitrate-N concentrations ranged from 1.6 to 6.9 mg N L^{-1}, typical of streams in this agricultural region during low flow conditions (Royer et al. 2004, Opdyke and David 2007).

There was significantly less sestonic chl*a* in the Salt Fork Vermilion River than the Kaskaskia River with an overall mean concentration of 1.2 μg L^{-1} in the Salt Fork Vermilion River compared to 4.8 μg L^{-1} in the Kaskaskia River. Contrastingly, the Salt Fork Vermilion River and the Kaskaskia River had about the same amount of periphytic chl*a* production on the artificial tiles with concentrations of 8.9 and 8.5 mg chl*a* m^{-2}, respectively, after the five-week incubation period. Also, the Salt Fork Vermilion River had considerably more and a greater range of canopy cover than the Kaskaskia River. The mean turbidity was about double in the Kaskaskia River compared to the Salt Fork

Table 2. Mean turbidity, mean canopy cover, and water depth measured at each tile, and water temperature in the Salt Fork Vermilion River and Kaskaskia River for two sampling dates in September and October 2006 (n=10). N/A = not available; data not collected.

River and sampling date	Turbidity (NTU)	Canopy Cover (%)	Depth (cm)	Temperature (ºC)
Salt Fork Vermilion River				
21-Sep-06	6.8	26	43	17.3
25-Oct-06	3.9	N/A	N/A	9.0
Kaskaskia River				
19-Sep-06	18	12	49	17.9
24-Oct-06	5.0	N/A	N/A	8.6

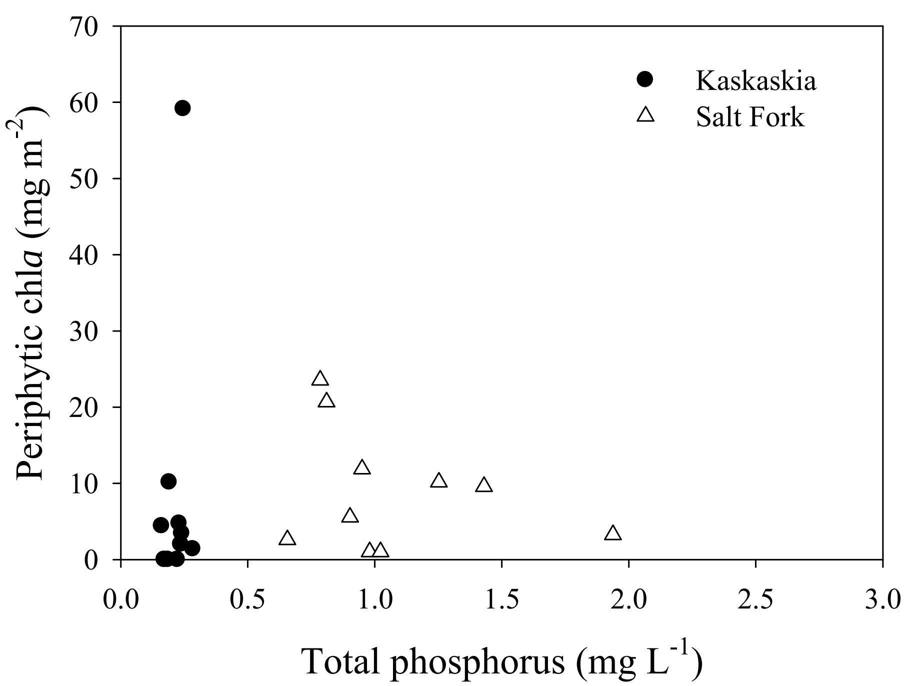

Figure 3. Relationship between total P and periphytic chl*a* growth on tiles for the Salt Fork Vermilion River and the Kaskaskia River.

Vermilion River with values of 12 NTU in the Kaskaskia River and 5 NTU in the Salt Fork Vermilion River (Fig. 2 and Table 2). Overall however, these concentrations of sestonic and periphytic chl*a* were quite low and did not reflect the high nutrient concentrations in the water.

As the P concentration increased in the river water, the concentration of sestonic or periphytic chl*a* did not increase accordingly ($p > 0.05$), probably because P was in great excess (Bushong and Bachmann 1989, Figueroa-Nieves et al. 2006). The concentration of periphytic chl*a* was generally below 15 mg chl*a* m^2 in both the Kaskaskia River and the Salt Fork Vermilion River, values more typical of low nutrient concentration streams (Dodds et al. 1998). There were high and low concentrations of periphytic chl*a,* but no pattern was evident (Fig. 3). These results strongly suggest that P was not the regulating factor for periphytic algal growth, despite the tiles providing a consistent and suitable benthic substrate. These variable periphytic chl*a* concentrations were likely due to the shifting of sediments, which covered the tiles and limited the surface area available for algal growth, as well as light limitations discussed below.

As the amounts of turbidity and canopy cover increased, the periphytic chl*a* concentration decreased. When looking at only turbidity or canopy cover and the presence of periphytic chl*a*, there was no trend in the amount of periphytic chl*a* that grew on the tiles. Therefore, there was no individual correlation between periphytic chl*a* growth and canopy cover or turbidity. However, the combination of turbidity and canopy cover, which produce low light levels, caused low concentrations of periphytic chl*a* in both rivers (Fig. 4). This is supported by the findings of Figueroa-Nieves et al. (2006), who showed that light was the primary factor limiting the amount of periphytic chl*a* in streams. They also showed that N and P and the amount of periphytic chl*a* were not related, which our findings confirm.

In these two rivers located in agricultural watersheds, sewage effluent P inputs at two different concentrations were not related to sestonic or periphytic chl*a* up to 90 km

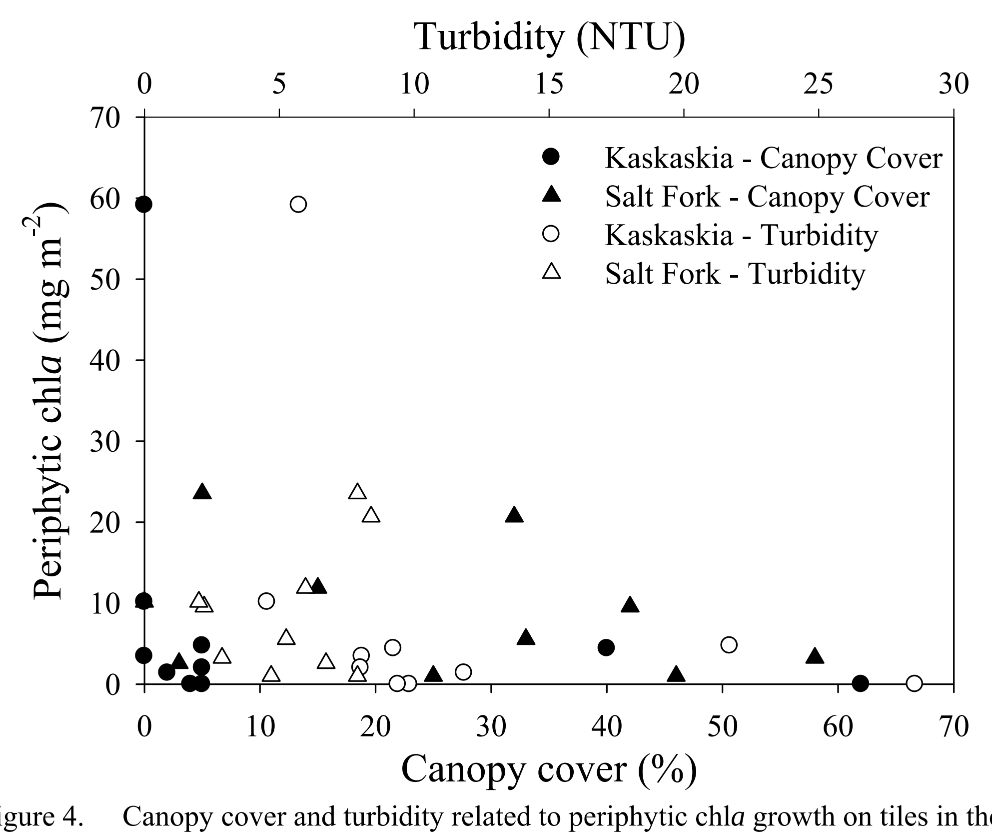

Figure 4. Canopy cover and turbidity related to periphytic chl*a* growth on tiles in the Salt Fork Vermilion River and the Kaskaskia River.

downstream from the source. Due to agricultural inputs, stream P concentrations did not decrease greatly in the low P effluent river. Even if P were removed completely from the effluent discharged into these two rivers, it seems unlikely that any change would occur in water quality with respect to algal growth (sestonic or periphytic), because agricultural runoff adds sufficient P. Limited light due to canopy cover, turbidity, and river depth is most likely the regulator of algal growth in these rivers.

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