Controls on chlorophyll-*a* in nutrient-rich agricultural streams in Illinois, USA

Débora Figueroa-Nieves[1], Todd V. Royer[2],* & Mark B. David[1]
[1]*Department of Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, W-503 Turner Hall, 1102 S. Goodwin Av., Urbana, IL 61801, USA*
[2]*School of Public and Environmental Affairs, Indiana University, 1315 East 10th Street, Bloomington, IN 47405, USA*
(*Author for correspondence: Tel.: +812-855-0971; Fax: +812-855-7802; E-mail: troyer@indiana.edu)*

Abstract

Nitrogen and phosphorus are the primary nutrients that affect water quality in streams in the midwestern USA and high concentrations of these nutrients tend to increase algal biomass. However, how nutrients interact with physical controls in regulating algal biomass is not well known in agricultural streams. Eighteen streams in east-central Illinois (USA) were sampled during June and September 2003 to analyze factors possibly regulating algal biomass. Additionally, two shaded and two non-shaded sites in the Embarras River in east-central Illinois were sampled intensively from June to December 2003. Both sestonic and periphytic chlorophyll-*a* (chl-*a*) were analyzed, and periphytic chl-*a* was assessed on natural substrata and unglazed ceramic tiles. Although high concentrations of nutrients were found in these streams (mean total $P = 0.09$–0.122 mg l^{-1} and mean NO_3-N $= 4.4$–8.4 mg l^{-1}), concentrations of sestonic chl-*a* were low among all sites and both sampling periods (<18 mg m^{-3}, median values of 5 and 3 in June and September, respectively). Filamentous algae were an important component of the algal communities in streams with stable substrata. Periphytic chl-*a* was generally not related to the concentration of N or P in the water column, and in non-shaded streams periphyton appeared at times to be light-limited due to turbid water. Turbidity was found to be an important factor controlling chl-*a* on ceramic tiles across the 18 sites and for the Embarras River sites; chl-*a* decreased exponentially in concentration (132–0 mg m^{-2}) as turbidity increased from 4 to 39 NTU ($r^2 = 0.80$). In general, the interaction between hydrology and light (turbidity) likely controlled algal biomass in these nutrient-rich, agricultural streams.

Introduction

Factors such as nutrients, turbidity, temperature, light, and hydrological regime can control sestonic algae and periphyton in streams (Bothwell, 1988; Bushong & Bachmann, 1989; Rosemond, 1993; Biggs, 1995; Van Nieuwenhuyse & Jones, 1996; Hill & Dimick, 2002). Midwestern streams tend to have high concentrations of N and P due to the application of fertilizers used in agricultural fields to enhance the productivity of crops (Fausey et al., 1995; David et al., 1997; David & Gentry, 2000).

Nitrogen and P reach these streams via subterranean tiles that discharge shallow ground water and associated solutes directly into the streams (David et al., 1997; McIsaac & Hu, 2004); surface runoff can also contribute nutrients, particularly P. Sewage effluent is also another source of N and P in streams, although generally more limited in effects in agricultural watersheds due to low population densities (David & Gentry, 2000). A major concern is that agricultural drainage contributes to adverse changes in stream water quality by increasing concentrations of nutrients and generally leading to

eutrophication. The United States Environmental Protection Agency currently is directing states to develop nutrient criteria as a step towards reducing stream N and P concentrations and improving biotic integrity. However, it is not known if nutrients are the critical factor controlling algal production in agricultural streams of the midwestern USA.

The concentration of sestonic chlorophyll-*a* (chl-*a*) has been related to total P across a range of streams (Van Nieuwenhuyse & Jones, 1996) and periphytic chl-*a* is often controlled by dissolved nutrients (e.g., Lowe et al., 1986). Although N and P can limit algae, these nutrients do not always relate strongly to the abundance of chl-*a*, particularly in P-enriched streams (Van Nieuwenhuyse & Jones, 1996). Bushong & Bachmann (1989) found that P enrichment never stimulated periphyton growth in agriculturally influenced streams in Iowa and concluded that background N and P concentrations were sufficient for algal demands. Similarly, Munn et al. (1989) and Wiley et al. (1990) suggested that periphyton growth was limited by temperature and turbidity in east-central Illinois, although neither study measured chl-*a* on naturally occurring substrata or included a direct evaluation of physical controls on chl-*a*. In general, algal biomass in streams depends upon the interaction of many factors, which can operate simultaneously, such as light, nutrients, and grazing (Rosemond, 1993). Discharge can also influence benthic chl-*a* by tumbling rocks on the streambed and scouring algae from substrata during floods (e.g., Lohman et al., 1992; Biggs, 1995, 2000).

Chlorophyll-*a* is often a central aspect of water quality programs and an indicator of eutrophication. Management and regulatory efforts aimed at reducing eutrophication in streams and rivers often focus on dissolved nutrients, although in agricultural regions the relationship between chl-*a* and dissolved nutrients is not always strong (e.g., Moore, 1977; Welch et al., 1988). Thus it is important to examine abiotic factors thought to regulate algal biomass (as indexed by chl-*a*) in addition to dissolved nutrients. We studied abiotic controls on chl-*a* abundance in shaded and non-shaded agricultural streams in east-central Illinois, USA. Our specific objectives were to: (1) investigate relationships between algal chl-*a* and abiotic factors (nutrients, turbidity, shading, and discharge related variables), (2) compare the relative abundance of sestonic versus benthic chl-*a*, and (3) examine periphyton growth on a common substrate across a series of nutrient-rich streams.

Methods

Site description

This study was conducted in the five watersheds that originate in east-central Illinois (Fig. 1). The landscape in these watersheds is relatively flat with organic matter rich Mollisols, creating an excellent environment for crops, especially corn and soybean production. Due to the flat topography the agricultural fields are drained with subterranean tiles to facilitate drainage and farming (David et al., 1997). Headwater streams in the region are extensively channelized and incised to facilitate drainage of water received from the subterranean tiles (Rhoads & Herricks, 1996). Additionally, riparian vegetation along most headwater streams is limited to grasses and the streams have open canopies. Larger streams often have a narrow riparian zone of deciduous trees that provide a closed canopy after leaf out (Munn et al., 1989).

Eighteen stream sites were used in the study, all of which drained watersheds dominated by row-crop agriculture (Table 1). The sites spanned a range of sizes and substrata conditions, and seven sites were shaded whereas 11 were non-shaded (Table 1). Three of the sites received sewage effluent upstream of our study locations: North Fork Salt Creek, Copper Slough, and the Salt Fork Vermilion River. All 18 sites were sampled during June and September 2003 to examine the range of benthic and sestonic chl-*a* that occurs in midwestern agricultural streams and to determine the factors controlling algal biomass. From the 18 sites, four sites on the Embarras River (Fig. 1) were monitored intensively (bi-weekly) from June to December 2003. Two of the Embarras River sites were non-shaded (EM1 and EM2) and the other two were shaded (EM3 and EM4).
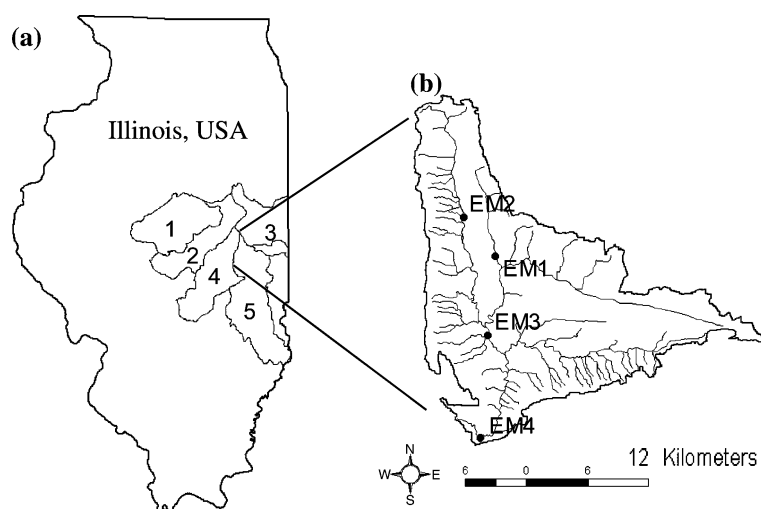
Figure 1. (a) Map showing the major river basins of east-central Illinois, USA. Study sites were located throughout the headwaters of these basins (see Table 1 for sites and basin names). (b) Detail of the locations of the intensively sampled sites in the headwaters of the Embarras River.

Physical and chemical characteristics

Streams were sampled for periphytic and sestonic chl-*a*, dissolved nutrients, temperature, turbidity, and dissolved oxygen. Three transects were established at each stream site, 15 m apart for a total study reach of 30 m. Water temperatures were recorded hourly during the study period for three sites (EM1, EM2, and EM4) using HOBO temperature loggers (Onset Computer, Bourne, MA). Turbidity was measured with a portable turbidimeter (Orbecco-Hellige Model 966). Instantaneous measures of dissolved oxygen and temperature were collected with a portable meter (Orion Model 835A or 830A). The study sites were located in proximity to existing gaging stations operated by the US Geological Survey to monitor stream discharge.

Water samples were analyzed following standard methods (APHA, 1998) for NO_3-N, using a Dionex DX-120 ion chromatograph. Ammonium and soluble reactive P (SRP) were determined colorimetrically by flow injection analysis (Lachat QuikChem© 8000). Total P was measured using a sulfuric acid and ammonium persulfate digestion followed by colorimetric detection (Lachat Quik-Chem© 8000). Internal and external standards for all solutes were analyzed routinely.

Periphytic chl-a

Two rocks were collected from each transect at each site for analysis of chl-*a* and ash free dry mass (AFDM) to estimate algal biomass in the stream reach. At some locations, the streambed was composed primarily of sand and fine sediments, but rocks were collected if present. Individual rocks were placed in plastic bags, wetted with stream water, and transported on ice to the laboratory (Steinman & Lamberti, 1996). Rocks were stored in the laboratory at 4 °C in the dark and were processed within 48 h of collection. Periphyton was removed from each rock by vigorous brushing with a plastic brush. The resulting slurry was collected onto a Whatman GF/F (0.7 μm) glass fiber filter, placed in a small petri dish, and kept in the dark at −20 °C for no longer than 4 weeks until analysis of chl-*a* (APHA, 1998).

Chlorophyll-*a* was extracted by placing filters in a test tube with 100% HPLC-grade methanol and steeping for 24 h at 4 °C in the dark (Steinman & Lamberti, 1996). After steeping, the samples were centrifuged for 20 min at 500 g. The absorbance of the chlorophyll was determined with a spectrophotometer (Thermo Spectronic, Inc.) and corrected for pheophytin (APHA, 1998). All procedures in the laboratory were conducted

Table 1. Site names, locations, and watershed and channel characteristics for the agricultural streams used in the study. All are located in east-central Illinois, USA

Site name	Coordinates	River basin*	Stream order	Watershed area (km^2)	Row crop (%)	Dominant substrata	Stream type
North Fork Salt Creek	40°15'12'' N, 88°47'43'' W	1	2	264	68	Sand/gravel	Shaded
South Branch Salt Creek	40°14'04'' N, 88°36'34'' W	1	2	60	78	Sand/gravel	Non-shaded
Salt Creek	40°15'03'' N, 88°36'59'' W	1	2	130	78	Sand/gravel	Non-shaded
Big Ditch	40°16'03'' N, 88°19'39'' W	2	2	99	90	Gravel/cobble	Non-shaded
Sangamon River site 1	40°18'39'' N, 88°19'21'' W	2	4	668	84	Gravel/cobble	Shaded
Sangamon River site 2	40°21'24'' N, 88°19'42'' W	2	4	559	84	Sand/gravel	Shaded
Jordan Creek	40°02'27'' N, 87°50'48'' W	3	1	29	74	Sand/gravel	Non-shaded
Little Vermilion River site 1	39°57'09'' N, 87°56'18'' W	3	1	67	83	Silt/sand	Non-shaded
Little Vermilion River site 2	39°54'08'' N, 87°46'12'' W	3	2	231	83	Silt/sand	Non-shaded
Stony Creek	40°09'12'' N, 87°52'23'' W	3	2	85	86	Sand/gravel	Non-shaded
North Fork Vermilion River	40°14'57'' N, 87°37'43'' W	3	3	680	66	Gravel/cobble	Shaded
Salt Fork Vermilion River	40°03'44'' N, 87°56'22'' W	3	4	839	74	Sand/gravel	Shaded
Copper Slough	40°05'03'' N, 88°20'17'' W	4	1	52	81	Gravel/cobble	Non-shaded
Lake Fork Kaskaskia River	39°53'50'' N, 88°32'16'' W	4	3	80	91	Silt/sand	Non-shaded
Black Slough (EM1)	39°57'09'' N, 88°10'08'' W	5	1	25	85	Sand/gravel	Non-shaded
Embarras River site 2 (EM2)	39°58'53'' N, 88°12'22'' W	5	2	57	85	Sand/gravel	Non-shaded
Embarras River site 3 (EM3)	39°52'47'' N, 88°10'36'' W	5	3	293	85	Sand/gravel	Shaded
Embarras River site 4 (EM4)	39°47'30'' N, 88°11'09'' W	5	4	473	85	Silt/sand	Shaded

*1, Salt Creek; 2, Sangamon River; 3, Vermilion/Little Vermilion; 4, Kaskaskia River; 5, Embarras River; see Figure 1.

under subdued light. Chl-*a* concentration was calculated from the equation of Marker et al. (1980) using coefficients appropriate for methanol extraction. Chl-*a* concentrations were expressed on an areal basis by determining the surface area on each rock from which the periphyton was scrapped. Surface areas were calculated using the aluminum foil method, which involves developing a mass–area relationship with pieces of aluminum foil (Steinman & Lamberti, 1996). After removal of the periphyton from a rock, the portion of the rock originally covered by biofilm was wrapped in a single layer of aluminum foil. The piece of foil was then weighed and the mass converted to area.

Sestonic chl-a

Before collection of periphyton, one water sample for sestonic algae was collected from the center of the stream at each of the three transects. Opaque Nalgene bottles were used and samples were placed on ice and taken to the laboratory for filtration and analysis of chl-*a*. Filtration was done immediately upon return to the laboratory. Sestonic chl-*a* was determined by filtering 475 ml through a GF/F (0.7 μm) filter. All filtration, storage and extraction procedures for the analysis of sestonic algae were conducted using the same techniques as described above for the periphyton analysis. Sestonic chl-*a* concentrations were calculated using the equation of Marker et al. (1980).

Artificial substrata

During late July and early August, 20 cm×20 cm unglazed ceramic tiles were placed in the center of the streambed in each transect at each of the 18 sites to determine periphyton growth. A small hole was drilled on the top right corner of each tile to hold the tile to the streambed with a reinforcing bar. Within each stream, an effort was made to place the tiles at similar depths, and water depth was recorded at each transect when the tiles were placed and retrieved (Table 2). Water samples were collected for nutrients and turbidity was measured at the time of tile placement. Tiles were left in the stream for 4 weeks to allow sufficient time for colonization and growth (Cattaneo & Amireault, 1992) and then retrieved in early September. Tiles were gently rinsed with stream water,

Table 2. Mean depth of the artificial substrata in the 18 sites at the time of placement (July) and retrieval (September). Missing data indicate sites at which the artificial substrata were lost or buried under sediment

Site name	Mean depth (cm)	
	July	September
Big Ditch	28	28
North Fork Salt Creek	17	9
South Branch Salt Creek	34	35
Salt Creek	44	38
Sangamon River site 1	100	76
Sangamon River site 2	88	63
Black Slough (EM1)	22	27
Embarras River site 2 (EM2)	21	35
Embarras River site 3 (EM3)	31	47
Embarras River site 4 (EM4)	41	72
Copper Slough	44	
Lake Fork Kaskaskia River	22	28
Jordan Creek	33	67
Little Vermilion River site 1	77	46
Little Vermilion River site 2	49	46
Stony Creek	59	
North Fork Vermilion River	50	
Salt Fork Vermilion River	>100	

placed in plastic bags, and transported to the laboratory for processing as described above for periphyton. Several tiles were lost or buried in sediments following a spate that occurred in September (see Table 2) and sites with lost or buried tiles were not included in the analysis.

At the Embarras River sites, 2.5 cm×2.5 cm unglazed tiles were anchored on the streambed in early July. These tiles were used to examine algal growth on a common substrate among sites. Twenty-five tiles were placed at left and right sides of each transect at each site and were sampled approximately bi-weekly from late July through December 2003. On each collection date, water depth was measured and water samples taken for measurement of dissolved nutrients (see above). Collection involved removing one individual tile from each side of the transect, rinsing it gently and placing it in an individual plastic bag for transport on ice to the laboratory. Processing was done as described above for periphyton. For the small tiles, filamentous algae were included in the analysis because it was not possible to separate the filaments

from the periphyton present on the tile. When required, samples were diluted before the absorbance was measured because of high concentration of chl-*a*. The equation for chl-*a* was then adjusted according to the dilution factors used for those samples.

Data analysis

All variables were tested for the assumption of normality by the Shapiro-Wilk statistic using the UNIVARIATE procedure (SAS, 1990). Only nitrate concentrations were normally distributed, therefore the Wilcoxon–Mann–Whitney non-parametric test was used to examine variability in sestonic and periphytic chl-a, SRP, total P, and turbidity among the Embarras River sites. The classification variable used in the test was shaded versus non-shaded. A Spearman rank order correlation (r_s; Zar, 1999) was used to examine relationships among all variables for both the 18 sites and the Embarras River sites.

Following log transformation of the data, simple and multiple linear regression were used to examine the controls on sestonic and periphytic chl-*a* from the June and September samplings of the 18 sites. Independent variables included dissolved nutrients, turbidity, temperature, streams width, watershed area, and dummy variables (Zar, 1999) for sewage effluent, substrata, and shading.

Results

Physical and chemical characteristics

During 2003, several major rainstorms led to rapid and high stream discharge, which is typical for east-central Illinois. Figure 2 shows data from EM4 and the pattern is representative of streams throughout the region. Compared to historical monthly average discharge, the timing of the high discharge was atypical; lower than average discharge generally was observed during January through June and higher than average flow occurred from July through December.

Mean turbidity during June and September 2003 for the 18 sites was 17 and 14 nephelometric turbidity units (NTU), respectively (Table 3). Most of these sites did not show a substantial
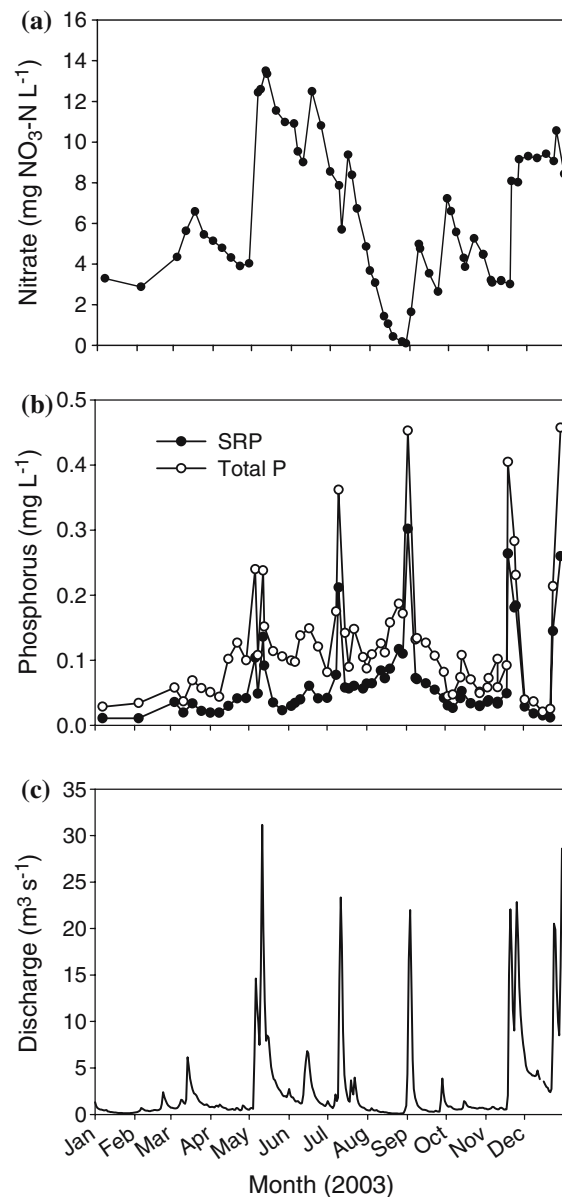

Figure 2. Stream water concentrations of (a) NO_3-N, (b) SRP and total P, and (c) daily discharge for the Embarras River at the EM4 site during 2003.

increase in turbidity during September but some sites did increase due to the high discharge that occurred in some of the watersheds. Mean NO_3-N concentrations were 8.4, 5.4, and 4.4 mg l^{-1} during June, July, and September 2003 (Table 3). Mean concentrations of total P concentrations were 0.12, 0.09, and 0.10 mg l^{-1} for these streams during June, July, and September 2003, respectively

Table 3. Mean turbidity and nutrient concentrations for all sites during the June, July, and September 2003 algal sampling. Values in parentheses are one standard deviation

Month	n	Turbidity (NTU)	NO_3-N (mg l^{-1})	NH_4-N (mg l^{-1})	SRP (mg l^{-1})	Total P (mg l^{-1})
June	18	17 (15)	8.4 (2.3)	0.054 (0.04)	0.056 (0.10)	0.12 (0.16)
July	18	13 (12)	5.4 (1.8)	0.027 (0.02)	0.060 (0.11)	0.09 (0.17)
Sept	18	14 (13)	4.4 (1.6)	0.041 (0.03)	0.067 (0.11)	0.10 (0.13)

(Table 3). The Salt Fork Vermilion River had high concentrations of P throughout the study due to a sewage treatment plant that discharges to that river approximately 40 km upstream of our study reach. Concentrations of total P were as high as 0.64, 0.72 and 0.49 mg l^{-1} during June, July and September, respectively. High concentrations of total P also were recorded in Copper Slough, which is located downstream of a sewage treatment plant that removes P to < 1 mg l^{-1}. Therefore, total P concentrations for Copper Slough were lower than Salt Fork Vermilion River but higher than the other streams studied.

The mean turbidity for EM1 and EM2 during 2003 was 4 and 8 NTU, respectively. Higher values of turbidity were observed in the shaded streams, EM3 and EM4 with means of 16 and 19 NTU, respectively. Among the Embarras River sites, turbidity was significantly higher in the shaded streams compared to the non-shaded streams ($p < 0.001$). Dissolved nutrients in the Embarras sites were temporally variable and often high (Table 4), but did not differ statistically among the sites. At EM4, the most downstream site on the Embarras River, nitrate concentrations were highest during early summer, decreased through late summer and increased substantially in late fall (Fig. 2). In contrast, SRP concentrations were more variable throughout the year with distinct peaks during periods of high discharge (Fig. 2).

Sestonic chl-a

Concentrations of sestonic chl-*a* were low among the 18 sites (<20 mg m^{-3} at all sites and sampling times) and slightly more variable in September compared to July (Fig. 3). A multiple regression failed to identify any meaningful relationships between sestonic chl-*a* and the environmental variables for either the June or September samplings. At intensively sampled sites on the Embarras River, concentrations of sestonic chl-*a* were less than 1.2 mg m^{-3} throughout the study period. Spearman rank order correlation indicated a significant but weak relationship between sestonic chl-*a* and total P ($r_s = 0.19$, $p = 0.003$) and nitrate ($r_s = 0.16$, $p = 0.007$). Sestonic chl-*a* concentrations were not statistically different between shaded and non-shaded sites on the Embarras River ($p = 0.19$).

Periphytic chl-a

Chlorophyll-*a* in periphyton varied by more than 2 orders of magnitude among the 18 sites (Fig. 3). During June, concentrations of chl-*a* were higher (reaching up to 131 mg m^{-2}) than during September when chl-*a* was below 50 mg m^{-2} for most of the sites due to a rainstorm and subsequent scouring that occurred at the beginning of September (median values of 26 and 9 mg m^{-2} for June and September, respectively). Multiple

Table 4. Mean turbidity and nutrient concentrations from sampling of the Embarras River sites during June through December 2003. Values in parentheses are one standard deviation. See Table 1 for site descriptions

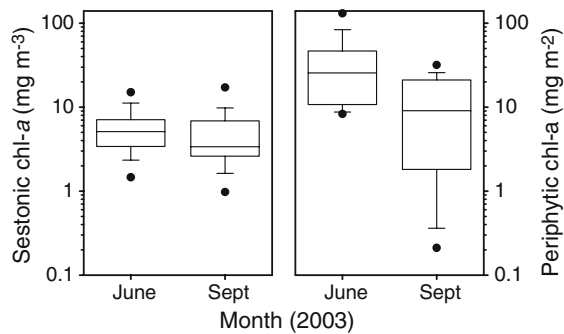| Location	n	Turbidity (NTU)	NO_3-N (mg l^{-1})	NH_4-N (mg l^{-1})	SRP (mg l^{-1})	Total P (mg l^{-1})
EM1	47	4 (3)	5.5 (2.8)	0.024 (0.03)	0.048 (0.05)	0.07 (0.07)
EM2	21	8 (5)	5.0 (2.0)	0.132 (0.26)	0.048 (0.06)	0.09 (0.10)
EM3	21	16 (11)	6.3 (3.1)	0.036 (0.03)	0.065 (0.09)	0.10 (0.12)
EM4	46	19 (11)	5.9 (3.3)	0.032 (0.02)	0.077 (0.07)	0.14 (0.10)

Figure 3. Box plots of sestonic and periphytic chl-*a* from 18 sites in east-central Illinois, USA, during June and September 2003. Box plots show 10th, 25th, 50th, 75th, and 90th percentiles as horizontal lines; values outside the 10th and 90th percentiles are shown as solid circles.

regression showed no meaningful relationships between periphytic chl-*a* and environmental variables during June. However, in September the model indicated that turbidity was a significant factor ($p < 0.001$), and explained 64% of the variability in periphytic chl-*a* among the 18 sites. In no case did a regression model with two or more factors perform better than single factor models at explaining patterns in chl-*a*.

Non-shaded sites on the Embarras River had significantly ($p = 0.009$) more periphytic chl-*a* than did the shaded sites during the June–December

study period (Fig. 4). The abundance of periphytic chl-*a* on natural substrata in the Embarras River was not related to dissolved nutrients or turbidity. Densities of periphytic chl-*a* in the Embarras River were generally between 20 and 50 mg m^{-2} except at EM4, which showed almost no occurrence of benthic chl-*a*. At EM1 there was an inverse relationship between discharge and periphytic chl-*a*, but the other Embarras sites showed no such relationship (data not shown).

Artificial substrata

Chlorophyll-*a* on tiles was not related to NO_3, NH_4, SRP, or total P among the 18 sites, but displayed a curvilinear relationship with turbidity (Fig. 5). In the Embarras River, chl-*a* was greater on tiles in the non-shaded sites compared to shaded sites, with values as high as 1460 and 1960 mg chl-*a* m^{-2} in EM1 and EM2, respectively (Fig. 6). Concentrations of chl-*a* on the 2.5 cm×2.5 cm tiles used in the Embarras were much higher than on natural substrata because filamentous algae were included in the analysis due to the difficulty of removing it without dislodging periphyton from the small tiles. During late autumn there was an increase in chl-*a* on tiles in the shaded streams (Fig. 6) possibly due to increased light reaching the streams following leaf abscission. Data were not
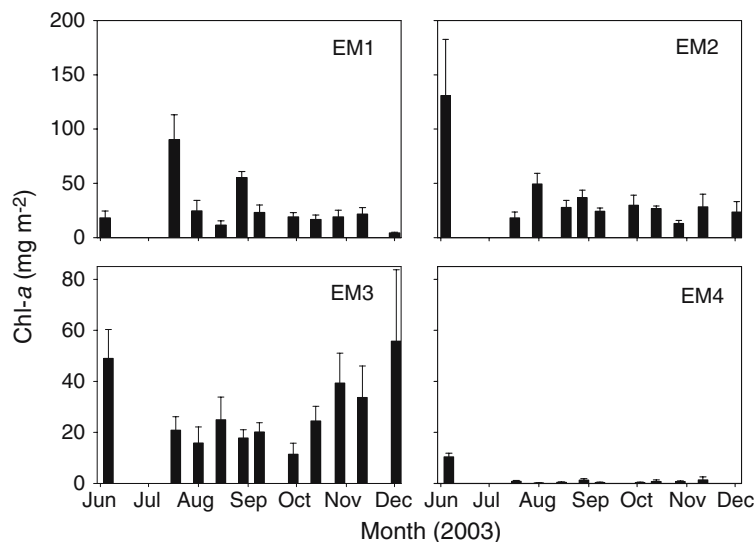

Figure 4. Mean (+1 standard error) periphytic chl-*a* concentrations on natural substrata at the Embarras River sites from June to December 2003. Note difference in scale between upper and lower graphs. EM1 and EM2 are non-shaded sites, EM3 and EM4 are shaded sites (see Table 1 for full description).
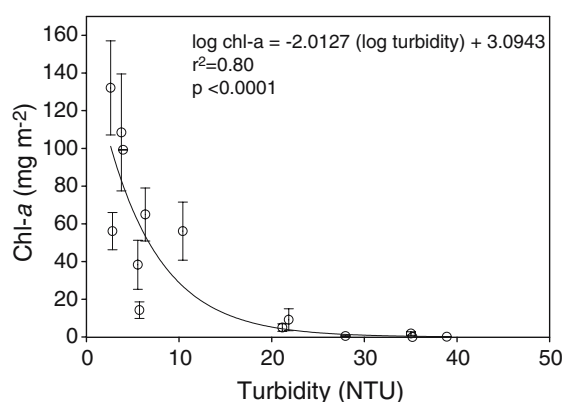
Figure 5. Relationship between mean (+1 standard error) chl-*a* on unglazed ceramic tiles and stream water turbidity for 14 sites in east-central Illinois, USA, during September 2003. The equation is based on a log–log relationship between periphytic chl-*a* and turbidity.

available for EM3 and EM4 during December because most tiles were lost during a flood.

Combining all Embarras River sites, there was a significant correlation between chl-*a* on the tiles and SRP ($p = 0.01$), total P ($p = 0.02$), and ammonium ($p < 0.0001$). However, the Spearman rank order correlations were weak ($r_s < 0.25$ for each variable) suggesting these variables were not the main factors controlling the abundance of benthic

chl-*a*. Turbidity explained over 50% of the variation in chl-*a* on tiles in the Embarras River, if both variables were log-transformed (Fig. 7).

Discussion

Sestonic chl-a

In general, concentrations of sestonic chl-*a* were not controlled by nutrients or water turbidity in the agricultural streams we examined. Low concentrations of sestonic chl-*a* were observed even though high concentrations of total P were measured. For instance, the Salt Fork Vermilion River and Copper Slough had the highest concentrations of total P, but relatively low concentrations of sestonic chl-*a*. The Salt Fork Vermilion River had 5 and 3 mg m^{-3} of sestonic chl-*a* during June and September, respectively. Moreover, Copper Slough is a non-shaded stream with abundant P and sestonic chl-*a* was low (7–9 mg m^{-3}) despite apparently good conditions for algal growth. Van Nieuwenhuyse & Jones (1996) examined streams with a wide range in total P concentrations and found that variability in sestonic chl-*a* was explained by total P. However, for streams in east-central Illinois this relationship was not observed,
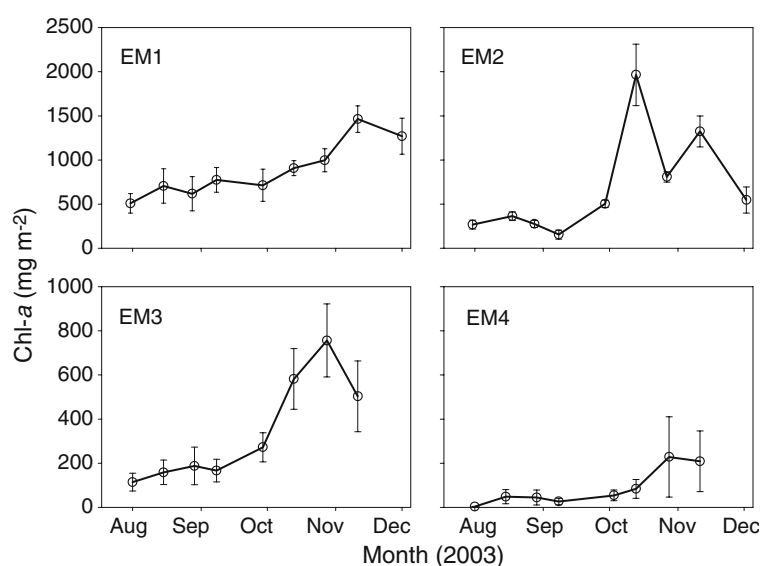

Figure 6. Mean (+1 standard error) chl-*a* concentrations on unglazed ceramic tiles at the Embarras River sites from August to December 2003. Note difference in scale between upper and lower graphs. EM1 and EM2 are non-shaded sites, EM3 and EM4 are shaded sites (see Table 1 for full description).
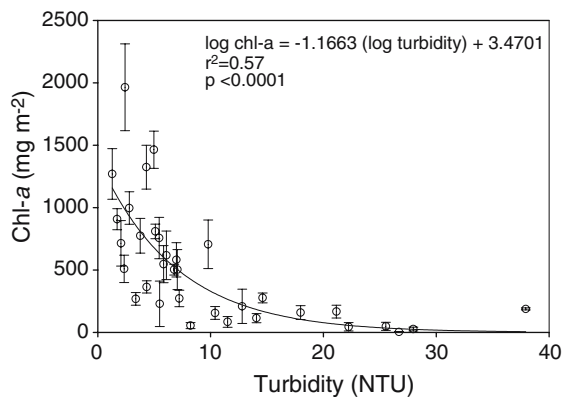
Figure 7. Relationship between mean (+1 standard error) chl-*a* on unglazed ceramic tiles and stream water turbidity for the Embarras River sites during August to December 2003. The equation is based on a log–log relationship between periphytic chl-*a* and turbidity.

probably because drainage from agricultural fields supplied the streams with excess nutrients (i.e., there were no streams with low total P concentrations). It has been suggested that in streams enriched with P the concentration of sestonic chl-*a* likely is controlled by factors other than nutrients (Van Nieuwenhuyse & Jones, 1996; Dodds et al., 1998) and our results support this conclusion.

Dodds et al. (1998) proposed that eutrophic streams could be expected to have more than 30 mg m^{-3} of sestonic chl-*a*, but also noted that this expectation might vary among stream types. We examined nutrient-rich agricultural streams and found sestonic chl-*a* concentrations in the range of only 2–20 mg m^{-3}, suggesting that the nutrient-rich condition of east-central Illinois streams is not leading to excessive amounts of sestonic chl-*a*. Alternatively, it may be that the classification scheme developed by Dodds et al. (1998) is not applicable to the heavily modified streams of Illinois or that sestonic algae is not the appropriate indicator of eutrophication for such streams. During our study, several rainstorms and subsequent spates likely limited the establishment of suspended algal populations in the streams we investigated. Additionally, the primary source of sestonic algae in agricultural streams is dislodged cells from periphyton (Swanson & Bachmann, 1976), suggesting periphytic chl-*a* may be the more appropriate measurement for examining nutrient–chlorophyll relationships. Overall, we believe sestonic chl-*a* is not a reliable indicator for identifying

eutrophication in the hydrologically variable and periodically turbid streams of east-central Illinois.

Periphytic chl-a

In the streams we examined, the abundance of periphytic chl-*a* was generally not related to the concentration of N or P in the water column. In shaded streams, light limitation may have prevented the development of dense periphyton communities despite the high concentrations of nutrients and warm water temperatures (Hill et al., 1995; Hill 1996). The generally low amounts of periphytic chl-*a* in the non-shaded streams suggests periphyton in non-shaded streams may at times be light-limited due to turbid water. This conclusion is supported by the significant relationship in September between turbidity and periphytic chl-*a*. However, no such relationship was observed in June, indicating that factors other than turbidity can be important in controlling chl-*a*. For example, some of the streams had beds composed primarily of small gravel, sand, and fine sediments, suggesting well-developed periphyton communities may have been precluded by lack of stable substrata. The placement of unglazed ceramic tiles in the streams provided a common substrate for periphyton growth that allowed us to further examine the role of turbidity. When provided with a common substrate, 80% of the among-stream variability in periphytic chl-*a* was explained by turbidity (Fig. 5). Turbidity also explained 57% of the variability in periphytic chl-*a* on ceramic tiles in the Embarras River sites (Fig. 7). Because fine sediments and clay are common in the streams of east-central Illinois, we suggest turbidity in the water column is creating light-limited conditions on the streambeds of many streams and rivers.

The majority of the streams from east-central Illinois had densities of benthic chl-*a* < 50 mg m^{-2} on natural substrata, suggesting they were not eutrophic based on the Dodds et al. (1998) classification. However, our observations suggest that filamentous algae, rather than epilithic periphyton, show the greatest response to nutrient enrichment in east-central Illinois streams. Mats of *Cladophora* can exceed 200 g m^{-2} of dry mass in these streams when discharge is low and stable (Schaller et al., 2004). The importance of filamentous algae

is particularly evident in the results from the non-shaded sites on the Embarras River (EM1 and EM2). On the artificial substrata, from which filaments were not removed, chl-*a* concentrations were an order of magnitude greater than on natural substrata that did not have attached filamentous algae or that had filaments removed prior to analysis (see Figs. 4 and 6). Coverage by filamentous algae is expected to be approximately 20% or more when total benthic chl-*a* exceeds 100 mg m^{-2} (Welch et al., 1988). We did not examine this relationship directly, but our visual observations support the notion that there is a direct relationship between coverage by filamentous algae and excessive chl-*a*.

There is good evidence that multiple factors can act to control the biomass of algae, both epilithic and filamentous, in streams (e.g., Biggs & Close, 1989; Lohman et al., 1992; Rosemond, 1993; Biggs, 2000). In the streams of east-central Illinois, hydrology and light are likely to be the main controls on algal biomass. We found that nutrient concentrations were poor predictors of chl-*a* and unlikely to be a limiting factor, a result that agrees with Munn et al. (1989) who used nutrient-diffusing substrata to examine nutrient–chlorophyll relationships in some of the same streams used in the present study. During periods of stable discharge, water clarity tends to increase with corresponding increases in periphytic chl-*a* (if hard substrata are available) and filamentous algae. Other work in these streams indicates that scouring during floods can substantially reduce the abundance of filamentous algae (Schaller et al., 2004). Time of accrual is an important factor that can determine whether or not nutrient enrichment leads to excessive algal biomass in streams (e.g., Lohman et al., 1992; Biggs, 2000). For agricultural streams with hard substrata, days of accrual might provide good explanatory power for periphytic chl-*a*, but this remains to be examined empirically in Illinois.

In general, light appeared to be the constraining factor for periphytic chl-*a* in streams of east-central Illinois. With the abundant nutrients in these streams, post-flood recovery should occur rapidly if light is available (Grimm & Fisher, 1986). There also appeared to be a seasonal effect of light, as the shaded sites on the Embarras River showed some increase in periphytic chl-*a* following leaf abscission. Although light (as indicated by turbidity) and hydrology appeared to be the main controlling factors for periphyton development in these streams, regulatory and management efforts aimed at reducing nuisance algal biomass will likely focus on dissolved nutrients. Unless nutrient loads in theses streams are reduced to the point that they become limiting for algal growth, we suggest that hydrology and light will continue to control algal biomass in streams of east-central Illinois.

Acknowledgements

References

APHA, 1998. Standard Methods for the Examination of Water and Wastewater, (20th edn). American Public Health Administration, Washington, DC.

Biggs, B. J. F., 1995. The contribution of flood disturbance, catchment geology and land use to the habitat template of periphyton in stream ecosystems. Freshwater Biology 33: 419–438.

Biggs, B. J. F., 2000. Eutrophication of streams and rivers: dissolved nutrient–chlorophyll relationships for benthic algae. Journal of the North American Benthological Society 19: 17–31.

Biggs, B. J. F. & M. E. Close, 1989. Periphyton biomass dynamics in gravel bed rivers: the relative effects of flow and nutrients. Freshwater Biology 22: 209–231.

Bothwell, M. L., 1988. Growth rate responses of lotic periphytic diatoms to experimental phosphorus enrichment: the influence of temperature and light. Canadian Journal of Fisheries and Aquatic Sciences 45: 261–270.

Bushong, S. J. & R. W. Bachmann, 1989. *In situ* nutrient enrichment experiments with periphyton in agricultural streams. Hydrobiologia 178: 1–10.

Cattaneo, A. & M. C. Amireault, 1992. How artificial are substrata for periphyton? Journal of the North American Benthological Society 11: 244–256.

David, M. B. & L. E. Gentry, 2000. Anthropogenic inputs of nitrogen and phosphorus and riverine export for Illinois, USA. Journal of Environmental Quality 29: 494–508.

David, M. B., D. A. Kovacic, L. E. Gentry & K. M. Smith, 1997. Nitrogen balance in and export from an agricultural watershed. Journal of Environmental Quality 26: 1038–1048.

Dodds, W. K., J. R. Jones & E. B. Welch, 1998. Suggested classification of stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Water Research 32: 1455–1462.

Fausey, N. R., L. C. Brown, H. W. Belcher & R. S. Kanwar, 1995. Drainage and water quality in Great Lakes cornbelt states. Journal of Irrigation and Drainage Engineering 121: 283–288.

Grimm, N. B. & S. G. Fisher, 1986. Nitrogen limitation in a Sonoran Desert stream. Journal of the North American Benthological Society 5: 2–15.

Hill, W. R., 1996. Effects of light. In Stevenson, R. J., M. L. Bothwell & R. L. Lowe (eds), Algal Ecology: Freshwater Benthic Ecosystems. Academic Press, San Diego, California, 121–148.

Hill, W. R. & S. M. Dimick, 2002. Effects of riparian leaf dynamics on periphyton photosynthesis and light utilization efficiency. Freshwater Biology 47: 1245–1256.

Hill, W. R., M. G. Ryon & E. M. Schilling, 1995. Light limitation in a stream ecosystem: responses by primary producers and consumers. Ecology 76: 1297–1309.

Lohman, K., J. R. Jones & B. D. Perkins, 1992. Effects of nutrient enrichment and flood frequency on periphyton biomass in northern Ozark streams. Canadian Journal of Fisheries and Aquatic Sciences 49: 1198–1205.

Lowe, R. L., S. W. Golladay & J. R. Webster, 1986. Periphyton response to nutrient manipulation in streams draining clearcut and forested watersheds. Journal of the North American Benthological Society 5: 221–229.

Marker, A. F. H., E. A. Nusch, H. Rai & B. Riemann, 1980. The measurement of photosynthetic pigments in freshwaters and standardization of methods: conclusions and recommendations. Archiv Fur Hydrobiologie Beiheft Ergebnisse Limnologie 14: 91–106.

McIsaac, G. F. & X. Hu, 2004. Net N input and riverine N export from Illinois agricultural watersheds with and without extensive tile drainage. Biogeochemistry 70: 251–271.

Moore, J. W., 1977. Some factors effecting algal densities in a eutrophic farmland stream. Oecologia 29: 257–267.

Munn, M. D., L. L. Osborne & M. J. Wiley, 1989. Factors influencing periphyton growth in agricultural streams of Central Illinois. Hydrobiologia 174: 89–97.

Rhoads, B. L. & E. E. Herricks, 1996. Naturalization of headwater streams in Illinois: challenges and possibilities. In Brookes, A. & F. D. Shields (eds), River Channel Restoration: Guiding Principles for Sustainable Projects. John Wiley and Sons, New York, 331–367.

Rosemond, A. D., 1993. Interactions among irradiance, nutrients, and herbivores constrain a stream algal community. Oecologia 94: 585–594.

SAS Institute, 1990. SAS/STAT User's Guide, Version 6, 4th edn. SAS Institute, Cary, North Carolina.

Schaller, J. L., T. V. Royer, M. B. David & J. L. Tank, 2004. Denitrification associated with plants and sediments in an agricultural stream. Journal of the North American Benthological Society 23: 667–676.

Steinman, A. D. & G. A. Lamberti, 1996. Biomass and pigments of benthic algae. In Hauer, F. R. & G. A. Lamberti (eds), Methods in Stream Ecology. Academic Press, San Diego, California, 295–313.

Swanson, C. D. & R. W. Bachmann, 1976. A model of algal exports in some Iowa streams. Ecology 57: 1076–1080.

Van Nieuwenhuyse, E. E. & J. R. Jones, 1996. Phosphorus–chlorophyll relationship in temperate streams and its variation with stream catchment area. Canadian Journal of Fisheries and Aquatic Sciences 53: 99–105.

Welch, E. B., J. M. Jacoby, R. R. Horner & M. R. Seeley, 1988. Nuisance biomass levels of periphytic algae in streams. Hydrobiologia 157: 162–168.

Wiley, M. J., L. L. Osborne & R. W. Larimore, 1990. Longitudinal structure of an agricultural prairie river system and its relationship to current stream ecosystem theory. Canadian Journal of Fisheries and Aquatic Sciences 47: 373–383.

Zar, J. H., 1999. Biostatistical Analysis (4th edn). Prentice Hall, Upper Saddle River, New Jersey.