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Relationships among Nutrients, Chlorophyll-*a*, and Dissolved Oxygen in Agricultural Streams in Illinois

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ABSTRACT

A better understanding of the controls on algae and dissolved O_2 in agricultural streams of Illinois is needed to aid in development of nutrient standards. We investigated the relationships between dissolved nutrients, algal abundance, and dissolved O_2 in five streams in east-central Illinois from March through November 2004. The streams drained watersheds from 25 to 777 km^2 that were dominated by row crop agriculture. Three sites had open canopies and two were bordered by a narrow forest of deciduous trees. Algal abundance was measured as chlorophyll-*a* (chl-*a*) concentration in the water column (sestonic) and on the streambed (periphytic). Mean NO_3–N concentrations ranged from 5.5 to 8.8 mg N L^{-1} and did not relate to algal abundance. Sestonic chl-*a* values ranged from nearly zero to >15 mg m^{-3} with no differences between open and shaded streams and only a weak correlation with dissolved reactive P (mean concentrations were 44–479 μg L^{-1}). The results suggest that sestonic chl-*a* is a poor criterion for assessing nutrient-related problems in these streams. Greatest periphytic chl-*a* occurred during low flow from August through October, but periphyton occurred consistently in only two of the five streams. The abundance of filamentous algae explained 64% of the variation in diel O_2 saturation, but was not correlated with nutrients. Currently it appears that hydrology and light, rather than nutrients, control algal abundance in these streams, and in the agricultural landscape of east-central Illinois, it may not be possible to reduce nutrient concentrations sufficiently to limit filamentous algal blooms.

Adequate dissolved O_2 is vital for the survival of aquatic organisms and is therefore an important variable in the assessment and monitoring of water quality. Short periods of anoxia can be fatal to aquatic organisms, and prolonged exposure to low O_2 concentrations can increase susceptibility to other environmental stressors (Horne and Goldman, 1994). Although O_2 concentrations in streams can vary naturally over diel and seasonal time scales, large fluctuations in O_2 concentrations often indicate excessive productivity resulting from nutrient enrichment (Walling and Webb, 1992). As algal biomass increases, respiration during nighttime can deplete O_2 concentrations to values that kill susceptible organisms and result in generally impaired biotic integrity (Portielje and Lijklema, 1995; Miltner and Rankin, 1998). In eutrophic streams and rivers, dissolved O_2 can range from supersaturated during daylight to nearly anoxic at night. Less productive, and presumably less impaired, streams are generally characterized by dissolved O_2 concentrations near saturation, with some moderate diurnal fluctuation caused by temperature and metabolism (Walling and Webb, 1992).

In freshwater systems, increased inputs of P are of particular concern because it commonly is the limiting nutrient for productivity in freshwater ecosystems. Phosphorus loading to streams can increase the biomass of periphyton, macroalgae, and sestonic algae, as measured by chl-*a* (Welch et al., 1989; Van Nieuwenhuyse and Jones, 1996; Dodds et al., 1998); however, identifying strong relationships between nutrient enrichment, chl-*a* concentrations, and biotic integrity in streams has been difficult because of confounding environmental factors such as shading, turbidity, scouring of biomass during floods, substrate characteristics, and herbivory (Miltner and Rankin, 1998; Dodds and Welch, 2000). Therefore, establishing defensible nutrient criteria for streams, as mandated by the USEPA for all states and tribes, requires an understanding of how environmental factors can influence the relationship between nutrients, chl-*a*, and dissolved O_2.

Nutrient enrichment and eutrophication are linked to a variety of human activities that can decrease water quality, such as agriculture, sewage effluent discharge, and urbanization (Biggs, 2000; Dodds and Welch, 2000). Major inputs of N and P to surface waters in the USA are from nonpoint sources, such as agricultural and urban activities (Carpenter et al., 1998). In the midwestern USA, N and P concentrations in streams tend to be high due to the dominance of fertilized agriculture and extensive artificial drainage (e.g., David and Gentry, 2000). The purpose of this study was to examine the relationships between nutrient concentrations, algal biomass, and dissolved O_2 patterns in streams draining agricultural watersheds in east-central Illinois. Algae in these streams occur as sestonic cells in the water column, as periphyton attached to hard surfaces, and as mats of the filamentous algae, *Cladophora*. Our first objective was to determine how these different groups varied in abundance through time and if that variation could be explained by P availability. Our second objective was to examine the role of physical factors, such as light and discharge, in controlling algal biomass in streams of this region. Our final objective was to determine the extent to which patterns in dissolved O_2 were related to the abundance of sestonic, periphytic, and filamentous mats of algae.

MATERIALS AND METHODS

Study Sites

This study was conducted in three watersheds in east-central Illinois: the Embarras, Vermilion, and Upper Kaskaskia rivers. These low-gradient river systems drain a flat landscape dom-

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Abbreviations: chl-*a*, chlorophyll-*a*; DRP, dissolved reactive phosphorus.

inated by row-crop agriculture of corn (*Zea mays* L.) and soybean [*Glycine max* (L.) Merr.]. Most agricultural fields in the region are artificially drained with subterranean tiles to maintain unsaturated soil for farming. Tile flow occurs primarily from late winter through early summer and during this period in-stream NO_3 concentrations are typically >8 mg N L^{-1} and occasionally exceed 15 mg N L^{-1} (David et al., 1997; Royer et al., 2004). Streams in headwater areas are extensively channelized and incised to facilitate drainage of water received from the subterranean tiles without overtopping of the stream banks (Rhoads and Herricks, 1996). The sites we examined ranged in drainage area from 25 to 777 km^2 and all are intensively farmed (Table 1). The Black Slough (BLS), upstream Embarras River (EMU), and Lake Fork Kaskaskia River (LFK) sites (see Table 1) had riparian vegetation of only grasses and thus had open canopies throughout the watersheds. The downstream Embarras River (EMD) and Salt Fork Vermilion River (SFV) sites had a narrow zone of deciduous trees that extended several river kilometers upstream of our sampling sites and provided a closed canopy during the growing season. Dissolved and total N and P concentrations tend to be high in east-central Illinois streams due to fertilization of cropland and the intensity of agricultural production (David et al., 1997; Royer et al., 2004). Streambeds consist mainly of silt, sand, and gravel (Royer et al., 2004), and large streams are often turbid much of the year whereas headwater streams tend to be clear except during floods (Wiley et al., 1990).

Physical and Chemical Variables

Sampling occurred from March through November 2004, except for Site LFK, where sampling began in May 2004. We focused our sampling on the growing season because problems associated with algal blooms and low dissolved O_2 occur mainly during this period in Illinois. At each site, five equidistant transects were established in a 40-m reach of stream. Water samples were collected weekly or biweekly, based on base flow conditions, with additional samples during high flows. During each sampling trip, water samples were collected before other collections and measurements to avoid disturbance of sediments on the streambed. Water samples were stored on ice and taken to the laboratory for analysis. Turbidity was measured at the left, center, and right portions of the wetted channel with a portable turbidimeter (Model 966, Orbeco-Hellige, Farmingdale, NY) and the values averaged. Water temperature was recorded with a portable meter (Digi-Sense Thermocouple, Cole-Parmer, Vernon Hills, IL). If the stream could be waded, mean water depth was determined from the five transects.

Water samples were analyzed for NO_3–N, NH_4–N, and dissolved reactive phosphorus (DRP) after filtration through a 0.45-μm membrane. Nitrate concentrations were determined using an ion chromatograph (DX-120, Dionex, Sunnyvale, CA), with a detection limit of 0.1 mg NO_3–N L^{-1}. Ammonium and DRP concentrations were analyzed colorimetrically by flow injection analysis with a QuikChem 8000 (Lachat, Loveland, CO) using the automated sodium salicylate and automated ascorbic acid methods, respectively. Method detection limits were 10 μg NH_4–N L^{-1} and 5 μg P L^{-1}. Water samples for total P were digested with H_2SO_4 and $(NH_4)_2S_2O_8$, which converts all forms of P into DRP, and then analyzed as above. Samples for total N were digested with H_2SO_4, $CuSO_4$, and K_2SO_4 in an aluminum block digestor (BD-46, Lachat) that converted organic N compounds to NH_3, which was then analyzed as described above. Water samples were processed, stored, and analyzed in accordance with standard methods (American Public Health Administration, 1998).

Discharge was monitored by the U.S. Geological Survey at Sites LFK (station no. 05590800) and EMD (station no. 03343400). Sites BLS and EMU are located upstream of Site EMD. Discharge at Sites BLS and EMU was calculated by scaling the discharge at Site EMD by the proportion of the watershed represented by Site BLS and Site EMU. Because of the consistent topography and land use within the EMD watershed, we believe this method gave reliable estimates of discharge patterns at Sites BLS and EMU. Discharge at Site SFV was determined in a similar manner using discharge records from USGS station no. 03339000 located downstream of Site SFV on the Vermilion River.

Algal Sampling and Chlorophyll-*a* Analysis

Sestonic Chlorophyll-*a*

Samples for sestonic chl-*a* were collected in opaque bottles from Transects 1, 3, and 5 and stored on ice for transport to the laboratory. Sestonic chl-*a* was assessed biweekly initially, then weekly from July through November. In the laboratory, water samples for sestonic chl-*a* were filtered on the same day as collection. All processing and analysis was performed in subdued light to prevent the degradation of photosynthetic pigments. A measured volume of water was passed through a Whatman GF/F (0.7-μm) glass fiber filter using a vacuum filtration apparatus. Filtered samples were stored in a petri dish wrapped in aluminum foil at −20°C for no more than 4 wk until analysis (USEPA, 1997). For analysis, each filter was placed in a 15-mL screw-cap centrifuge tube to which 10 mL of 90% acetone was added. A high-intensity ultrasonic liquid processor was used for extraction by sonication. Each sample was sonicated for two pulses of 15 s each. All samples were then shaken and allowed to steep in the dark at 4°C for 24 h.

Following steeping, samples were centrifuged for 15 to 20 min at 675*g*. Three milliliters of the supernatant were transferred to a cuvette (1-cm cell length) and absorbance was read for each sample at 664 and 750 nm on an UV-VIS spectrophotometer (Aquamate, ThermoElectron, Waltham, MA). To correct for pheophytin (a degradation pigment of chl-*a* that

Table 1. Site names, coordinates, and watershed descriptors for each of the east-central Illinois stream sites used in the study.

Site name	Site ID	Coordinates	Stream order	Drainage area	Row crop	Stream type at sampling site
				km^2	%	
Black Slough (Embarras tributary)	BLS	39°57′09″ N 88°10′08″ W	1	25	85	open
Embarras River upstream site	EMU	39°58′53″ N 88°12′22″ W	2	57	85	open
Embarras River downstream site	EMD	39°47′30″ N 88°11′09″ W	3	473	85	shaded
Lake Fork Kaskaskia River	LFK	39°48′23″ N 88°28′34″ W	3	386	91	open
Salt Fork Vermilion River	SFV	40°03′04″ N 88°01′44″ W	4	777	74	shaded

absorbs near the same wavelength), samples were acidified using 0.1 mL of 0.1 *M* HCl added to each cuvette. After 90 s, absorbance was measured at 665 and 750 nm. Samples with high concentrations of chl-*a* were diluted to be within the acceptable range for the instrument. The concentration of chl-*a* in each sample was calculated and expressed as milligrams per cubic meter (USEPA, 1997).

Periphytic Chlorophyll-*a*

At Sites BLS and EMU, a representative rock of 5 to 10 cm in diameter was collected at Transects 1, 3, and 5, placed in individual plastic freezer bags, and kept on ice for transport to the laboratory for analysis of periphytic chl-*a*. We considered periphyton to be the immediate biofilm attached to the rocks, thus long tufts of filamentous algae were not included in the periphytic chl-*a* measurements. Periphyton sampling was conducted biweekly throughout the study. Processing in the laboratory was done on the same day as collection and under subdued light. An area of each rock was scraped clean of periphyton using a stiff-bristled brush and distilled water. The resulting slurry was collected and processed as described above for sestonic chl-*a* samples with the following modifications: 90% ethanol was used as the solvent, initial absorbency readings were taken at 665 and 750 nm, then the sample was acidified with 0.06 mL of 0.3 *M* HCl for 5 min, and absorbency recorded again at the same wavelengths (Marker and Jinks, 1982). The concentration of chl-*a* was calculated using the equation presented by Steinman and Lamberti (1996) and expressed as milligrams per square meter. The surface area of rock from which periphyton was scraped was determined using the aluminum foil method (Steinman and Lamberti, 1996). The scraped area was covered completely with a single layer of aluminum foil, which was then removed and weighed. The weight of the foil was converted to surface area using a regression equation established following the procedure of Steinman and Lamberti (1996).

Filamentous Algae

When filamentous macroalgae were present, the wetted width at each transect was measured and the percentage of the streambed covered by macroalgae determined as described by Schaller et al. (2004). Cover was converted to biomass by collecting, at each transect, all filamentous algae in a 314-cm^2 area that was completely covered by algae. This material was rinsed in the laboratory, dried, weighed, and expressed as grams per square meter. The percentage cover values for each transect were scaled by this conversion to estimate mean biomass (grams per square meter) of filamentous algae for the stream reach.

Dissolved Oxygen Monitoring

Dissolved O_2 was monitored continuously during the study period at Site BLS using a datalogger (CR10, Campbell Scientific, Logan, UT) and dissolved O_2 probe (CS511-L, Campbell Scientific). Water temperature was monitored continuously with a HOBO temperature logger (Onset Computer, Bourne, MA). At the other sites, dissolved O_2 and water temperature were measured approximately once a month for a 48- or 72-h time period using a YSI (Yellow Springs, OH) 600XLM probe, a Hydrolab (Loveland, CO) MiniSonde, or a Hydrolab Datasonde (Model 4a). Regardless of the instrument, data were recorded every 15 min and averaged hourly. All probes were calibrated according to manufacturers' instructions before deployment. To account for variation between instruments and possible instrument drift, all probes were checked routinely against Winkler titrations (American Public Health Administration, 1998). The dissolved O_2 patterns we measured at the downstream transect of each site were presumed to be a reflection of conditions in the study reach, but productivity occurring further upstream could have influenced O_2 patterns at the monitoring site. Light intensity was measured continuously at Site BLS using a datalogger (CR10, Campbell Scientific) and pyranometer (LI-200SA, LI-COR, Lincoln, NE). Because Site BLS has an open canopy, the light measurements indicate incoming solar radiation and do not account for reductions by the canopy at the shaded sites.

Statistical Analyses

Variables were tested for normality using the univariate procedure (SAS Institute, 1990). No variable was normally distributed. Therefore, Spearman rank order correlation was used to examine relationships among the variables and the Spearman rank correlation coefficient (r_s) is reported (Zar, 1999). Rank order correlations do not require normally distributed data but lack the predictive power of linear regression. Nevertheless, we believe this analysis can provide insight into the relationships between nutrients, chl-*a*, and dissolved O_2.

The diel range in dissolved O_2 saturation was determined by subtracting the minimum saturation percentage value from the maximum value for that day. Twenty-one values of diel range in saturation percentage were collected across the sites, of which two values appeared to be outliers with extremely large ranges. The outliers were from open-canopy sites (EMU and BLS) and, based on our observations, we believe these outliers were not related to productivity in the study reach but instead resulted from O_2 patterns that originated upstream of the study reach. With these two values removed, the data were normally distributed with constant variance, and we regressed diel range in saturation percentage against both periphytic chl-*a* and the biomass of filamentous algae.

RESULTS

The streams ranged in mean depth from 32 to 93 cm when they could be waded, although discharge (and hence depth) varied considerably during the study period (Table 2). Turbidity ranged from <10 nephelometric turbidity units (NTU) in all sites during periods of low flow to 80 NTU or greater in the larger streams during high discharge. Mean NO_3–N concentrations were similar among the sites, ranging from 5.5 to 8.8 mg N L^{-1}. The maximum and minimum values were also comparable among sites and ranged from 0.1 to 16.1 mg N L^{-1}. Although NH_4–N concentrations were often high (>100 μg L^{-1}), NO_3–N typically accounted for 90% or more of water column total N at all sites. At each site, total P and DRP varied by at least an order of magnitude during the study, with the maximum concentration of DRP exceeding 190 μg L^{-1} at all sites (Table 2). Site SFV had particularly high P concentrations due to sewage effluent from Urbana, IL, approximately 40 km upstream of our sampling site.

Sestonic chl-*a* values ranged from nearly zero to >15 mg m^{-3} with no apparent differences between open and shaded streams (Fig. 1). Although chl-*a* in the water column could occasionally be high, the median value for both open and shaded streams was <3 mg m^{-3} and the mean values were <5 mg m^{-3}. In streams, the source of sestonic algae is generally thought to be sloughed algal

Table 2. Physical and chemical characteristics for the stream study sites during March to December 2004. Mean values are reported, with the range shown in parentheses. Depth measurements were taken only when streams were able to be waded (n = 6–14); for all other variables, n = 26.

Site†	Depth	Temperature	Discharge	Turbidity	NO_3–N	Total N	NH_4–N	DRP‡	Total P
	m	°C	$m^3\ s^{-1}$	NTU§	mg L^{-1}		µg L^{-1}		
BLS	0.32	16	0.11	5	7.3	7.5	24	44	60
	(0.17–0.48)	(7–24)	(0.002–0.44)	(1–25)	(0.1–11.3)	(0.4–11.6)	(7–56)	(BD¶–293)	(10–430)
EMU	0.33	17	0.26	8	5.5	5.8	32	46	80
	(0.20–0.49)	(6–24)	(0.005–1.01)	(1–41)	(0.6–10.1)	(1.1–10.1)	(13–127)	(3–274)	(10–400)
EMD	0.55	17	4.8	24	8.0	8.5	31	74	130
	(0.23–0.94)	(7–24)	(0.04–67)	(6–41)	(0.7–13.8)	(1.2–13.9)	(8–72)	(13–342)	(30–500)
LFK	0.43	18	4.3	46	8.8	9.8	55	50	240
	(0.31–0.53)	(6–27)	(0.01–57)	(9–111)	(0.2–16.1)	(0.6–17.0)	(5–180)	(5–194)	(10–1660)
SFV	0.93	18	9.9	21	6.8	7.4	51	479	630
	(0.90–0.94)	(7–25)	(0.6–140)	(5–86)	(1.0–11.3)	(2.2–12.2)	(19–330)	(142–1790)	(200–2750)

† BLS, Black Slough (Embarras River tributary); EMU, upstream Embarras River site; EMD, downstream Embarras River site; LFK, Lake Fork of the Kaskaskia River; SFV, Salt Fork of the Vermillion River.
‡ Dissolved reactive P.
§ Nephelometric turbidity units.
¶ Below detection limits.

cells from the periphyton (Swanson and Bachmann, 1976). We found no correlation between sestonic chl-*a* and periphytic chl-*a* values at the open sites (BLS and EMU), and the shaded sites had little or no periphyton but often had measurable amounts of sestonic chl-*a*.

Site SFV had relatively low sestonic chl-*a* values but average P concentrations three to 10 times greater than the other sites because of the upstream sewage effluent. With Site SFV included, no meaningful relationship was found between DRP or total P and sestonic chl-*a*, but with Site SFV removed, there was a moderate but significant (r_s = 0.52, $P < 0.005$) correlation between total P and sestonic chl-*a*, and a weaker correlation with DRP (r_s = 0.21, $P < 0.03$). The scatter suggests that factors other than P were important in controlling sestonic chl-*a* (Fig. 2); however, of the other variables examined, only turbidity showed a positive correlation (r_s = 0.51, $P <$ 0.005) with sestonic chl-*a*, but this probably represented an autocorrelation because algal cells in the water column contributed to the measured turbidity.

Periphyton only occurred consistently at the open sites (BLS and EMU) and the amount of periphytic chl-*a* varied during the study from 0 to 40 mg m^{-2} (Fig. 3). The greatest values of periphytic chl-*a* occurred during extended periods of low flow from August through October. Periphytic chl-*a* declined during July, possibly as a result of scouring, although similar discharge peaks during June did not affect abundance (Fig. 3). Depth was inversely correlated with periphytic chl-*a* (Fig. 4) and this relationship probably reflected light attenuation through the water column. On the dates when periphyton was sampled, the concentration of DRP ranged from about 5 to 70 µg P and was positively correlated with periphytic chl-*a* (r_s = 0.43, $P < 0.01$). Nitrate, total P, and turbidity showed no relationship with periphytic chl-*a*.

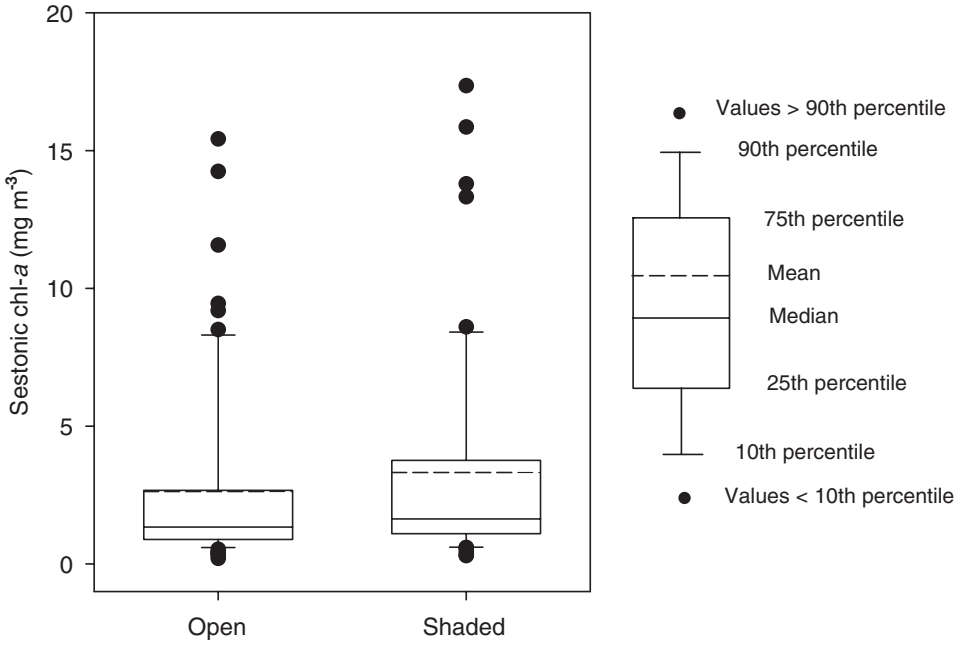

Fig. 1. Box plot of sestonic chl-*a* concentrations from March through November 2004 in three open-canopy (n = 75) and two shaded (n = 51) east-central Illinois agricultural streams.

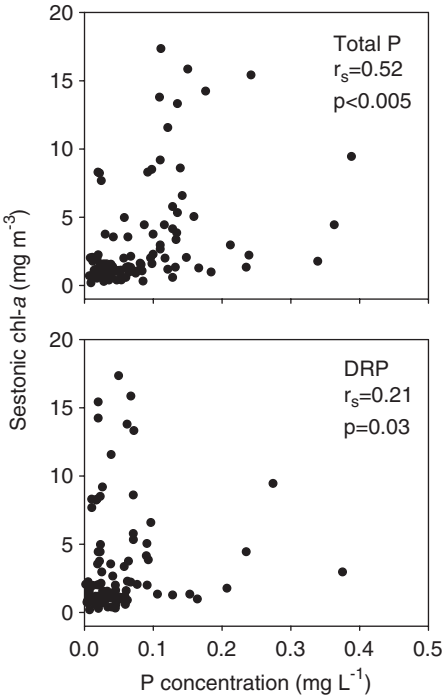

Fig. 2. Correlations between sestonic chl-*a* and (upper) total P and (lower) DRP (dissolved reactive P) from March through November 2004 in agricultural streams, excluding Site SFV.

Only the open sites had substantive amounts of filamentous algae and, at times, coverage of the streambed exceeded 40% in sites with biomass >30 g dry mass m^{-2} (Fig. 5). The abundance of filamentous algae was not correlated with discharge, turbidity, or any measure of water chemistry. There was considerable temporal variation in the abundance of filamentous algae at both sites and high discharge periods probably reduced the abundance of filamentous algae, as reported from other streams in Illinois (Schaller et al., 2004), but this was not easily discernable from the data we collected.

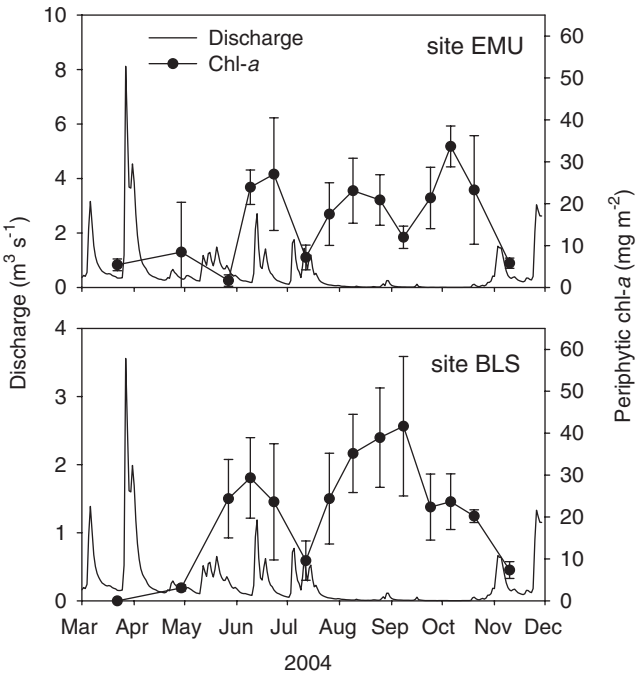

Fig. 3. Estimated discharge and mean periphytic chl-*a* (±1 SE) patterns for two open-canopy agricultural streams. Note difference in scale for discharge.

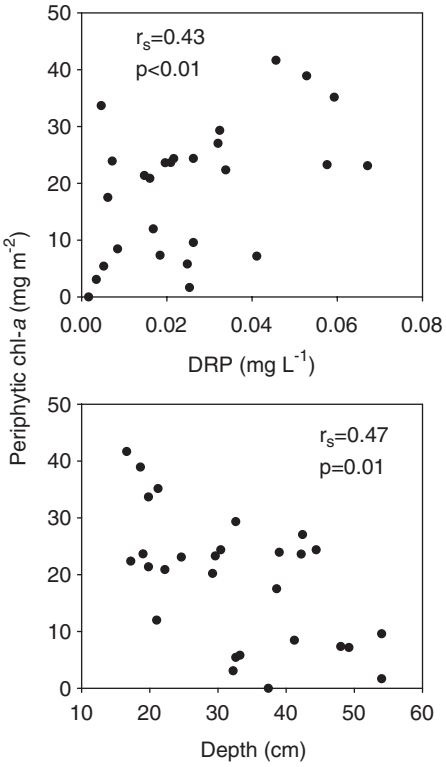

Fig. 4. Correlations between periphytic chlorophyll-*a* and (upper) DRP and (lower) stream depth for two open-canopy agricultural streams (Sites EMU and BLS) from March through November 2004.

Example dissolved O_2 curves from open (BLS and EMU) and shaded (EMD and SFV) sites from May and November are shown in Fig. 6. The diel range in dissolved O_2 saturation was several-fold greater in the open sites than the shaded sites. The open sites were routinely supersaturated (up to 160–180%) in dissolved O_2 during daylight, whereas the shaded sites were consistently undersaturated. The diel range in O_2 saturation was not related to the abundance of periphytic chl-*a*; however, the biomass of filamentous algae explained 64% of the variation in diel ranges of O_2 saturation (Fig. 7). Thus, diel patterns in dissolved O_2 appeared to be influenced by the abundance of filamentous algae but insensitive to the abundance of periphyton in these open, agricultural streams.

From April through December 2004 at Site BLS, dissolved O_2 concentrations were below the Illinois Pollution Control Board standard of 5 mg L^{-1} on a total of 57 d, including each night from 17 August through 26 September (Fig. 8). From 24 August through 29 August, dissolved O_2 declined to <3.5 mg L^{-1} and remained below this value for 8 to 14 h. The August to September period with consistently low nighttime dissolved O_2 concentrations corresponded with a period of low filamentous algae at Site BLS but the highest measured values of periphytic chl-*a*.

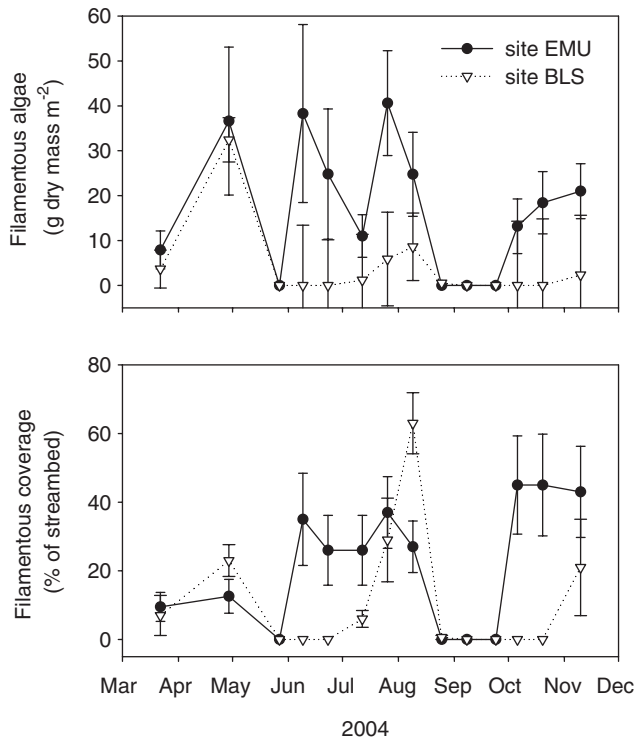

Fig. 5. Mean (±1 SE) abundance of filamentous algae as (upper) biomass and (lower) streambed coverage in two open-canopy agricultural streams.

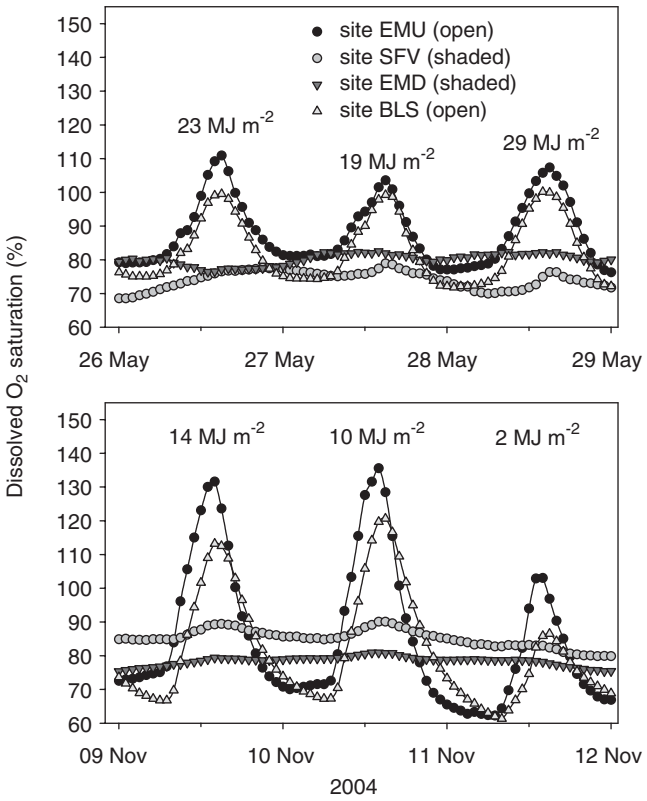

Fig. 6. Dissolved O_2 patterns during (upper) late May and (lower) early November 2004 in two open-canopy and two shaded agricultural streams. Daily solar radiation values were measured at Site BLS (open canopy) and do not reflect available light at the water surface for the shaded sites.

DISCUSSION

Agricultural streams in east-central Illinois are characterized by flashy hydrology and generally high but seasonally variable concentrations of dissolved nutrients (Royer et al., 2004; Schaller et al., 2004). No measure of N was positively correlated with the abundance of sestonic, periphytic, or filamentous algae and N was unlikely to have limited algal growth, as the minimum NO_3–N concentrations were >100 μg L^{-1} and mean concentrations exceeded 5 mg L^{-1} (Table 2). Although excessive N in streams is an environmental concern, our findings support previous work suggesting that N generally is not limiting for algal growth in many agricultural streams in the Midwest (Bushong and Bachmann, 1989; Munn et al., 1989; Wiley et al., 1990).

Total P was correlated to the concentration of sestonic chl-*a*, but the relationship was not strong and probably arose because both variables were directly related to the density of algal cells in the water column. The relationship between sestonic chl-*a* and turbidity is similarly confounded. Sestonic chl-*a* was not related to DRP, suggesting sestonic chl-*a* levels were determined by factors other than nutrients, although our analysis did not reveal which factors were important in this regard. Swanson and Bachmann (1976) presented evidence that sestonic algae in agricultural streams originates from sloughing of periphyton on the stream bed. We believe that this probably was true for the streams we examined, but the lack of correlation between periphytic and sestonic chl-*a* suggests that the mechanism was more complex than a simple displacement of cells from the streambed to the water column. Overall, sestonic chl-*a* was low despite high N and P concentrations throughout most of the study, and these streams would be classified as oligotrophic using the trophic scale presented by Dodds et al. (1998). Sestonic chl-*a* must be a reliable predictor of water quality if it is to be used as an indicator of nutrient-related impairment in streams (Reckhow et al., 2005). Because the abundance of sestonic chl-*a* was not related in any simple way with nutrients or periphyton, we suggest that chl-*a* in the water column is not a good criterion for assessing eutrophication-related problems in these low-order, agricultural streams.

The abundance of periphyton in streams is strongly affected by light, nutrients, and flow regime (e.g., Lohman et al., 1992; Biggs, 1995, 2000) and the expected direct relationship between periphyton biomass and nutrients may not be realized if light limitation or scouring occurs (Dodds et al., 2002). Periphytic chl-*a* in our study was greatest during periods of low stable flow in summer and autumn, but periphyton occurred consistently in only two of the five streams. The shaded sites (EMD and SFV) had N and P concentrations similar to or greater than the other streams but supported no periphyton, probably because insufficient light reached the streambed. The role of light was particularly evident at Site SFV, which had consistently high N and P concentrations but had no periphyton and <3.5 mg m^{-3} of sestonic chl-*a* throughout the study. Site LFK was not shaded but lacked hard substrata and contained peri-

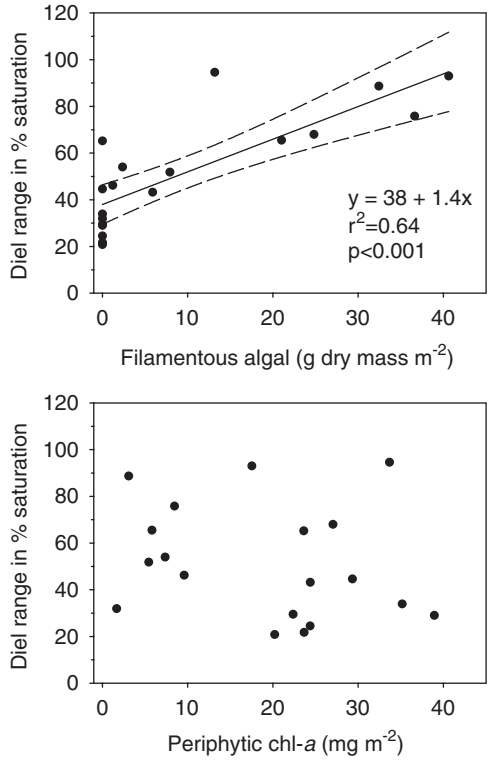

Fig. 7. Relationship between the diel range in dissolved O_2 saturation and the biomass of (upper) filamentous algae or (lower) periphytic chlorophyll-*a* in two open-canopy agricultural streams during March through November 2004.

phyton on only two dates throughout the study. In those streams where it occurred, periphytic chl-*a* was correlated positively with DRP and inversely with depth, with depth probably indicating light attenuation through the water column. Canopy cover and turbidity were important both in determining if periphyton could occur at all and in affecting temporal patterns in periphyton abundance within the streams where it occurred. The effects of nutrient enrichment will be observable as a response in periphyton biomass only in streams with the habitat conditions that allow for periphyton development (Biggs, 1995). For many nutrient-rich agricultural streams, light, temperature, flow regime, and substrata may be the controlling factors for periphyton accrual (Moore, 1977; Bushong and Bachmann, 1989; Munn et al., 1989), meaning that such streams may not show a consistent and generalized response in periphyton abundance to changes in nutrient loads.

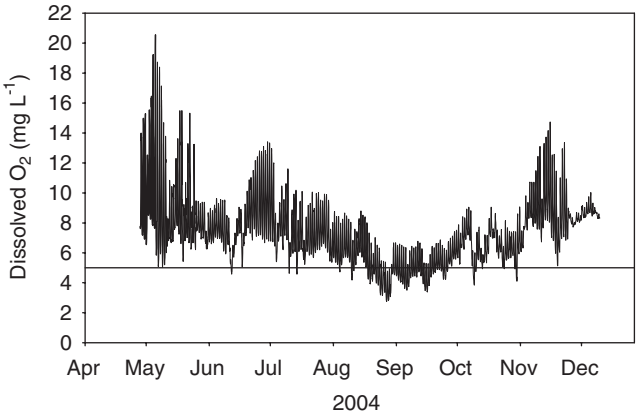

Fig. 8. Continuous dissolved O_2 concentration in an open-canopy agricultural stream (Site BLS) from late April through early December 2004. Horizontal line indicates the 5 mg L^{-1} Illinois dissolved O_2 standard at the time of the study.

The biomass of filamentous algae (*Cladophora*) during this study reached a maximum of 40 g dry mass m^{-2} (Fig. 5), which was considerably less than the maximum of 200 g dry mass m^{-2} reported by Schaller et al. (2004) during 2002 for a similar type of stream in east-central Illinois. The occurrence and abundance of filamentous algae were sporadic and unrelated to nutrient concentrations or any of the measured physical variables. Dodds et al. (1997) suggested that preventing nuisance levels of *Cladophora* in the Clark Fork River in western Montana would require an average concentration of total P below 30 μg L^{-1} and an average concentration of total N below 350 μg L^{-1}. In east-central Illinois, average total P was at least twofold greater than the value proposed by Dodds et al. (1997) and average total N was more than an order of magnitude greater (Table 2). Given the high nutrient concentrations in these agricultural streams, we suggest that flow regime, light, and temperature had greater influences on filamentous algal biomass than did nutrients. Based on our observations, the timing and density of mats of filamentous algae in these streams can vary significantly among years and between streams. Although we cannot yet explain the spatial and temporal patterns in filamentous algae, this variable did appear to have a significant influence on dissolved O_2 patterns (Fig. 7) whereas sestonic and periphytic chl-*a* did not. The spatial distribution of mats of filamentous algae within a stream network is likely to be an important factor affecting dissolved O_2 patterns, and algal blooms in upstream reaches or tributaries may create unexpected O_2 patterns at a monitoring site.

Temporal patterns in dissolved O_2, such as we present for Site BLS (Fig. 8), are influenced by multiple and often interacting factors that include temperature, physical aeration, ground water exchange, heterotrophic respiration, and algal metabolism. In general, low nighttime O_2 concentrations in eutrophic streams and rivers are thought to result from respiration by primary producers and the decay of excess biomass (e.g., Walling and Webb, 1992). The low nighttime O_2 concentrations that occurred during late August at Site BLS corresponded to a time of high periphytic chl-*a*, but the diel range in saturation percentage was not large and it is uncertain if respiration by periphyton was responsible for the low dissolved O_2 at that time. The abundance of filamentous algae influenced the diel range in saturation percentage, but not the minimum dissolved O_2 concentration. Welch et al. (1988) similarly found that the biomass of filamentous algae was not related to minimum dissolved O_2 concentrations in streams. Shading strongly influences algal photosynthesis in streams (e.g., Hill, 1996) and thus dissolved O_2 patterns. We found that open-canopy streams contained filamentous algae, whereas shaded streams did not, and this appeared to

explain the differences in dissolved O_2 patterns between the two stream types (see Fig. 6).

Streams in east-central Illinois, and throughout the Midwest, are highly modified ecosystems (e.g., Rhoads and Herricks, 1996) and strongly affected by agricultural activities and, in some cases, sewage effluent. Understanding the relationships between nutrients, algae, and dissolved O_2 in these streams is complicated by land use and hydrological modifications. Because of agricultural drainage, the highest N and P concentrations occur during periods of high flow (Royer et al., 2004), when depth and turbidity also are high but algal biomass is low. Headwater channels tend to be open, whereas canopy shading and turbidity increase as one moves from headwaters to larger streams (Wiley et al., 1990). In the shaded sites, dissolved O_2 patterns appeared to be unrelated to nutrient concentrations. For the open-canopy streams, we suggest that hydrology, through effects on depth, turbidity, and biomass scouring or accrual (Biggs, 2000), is the key factor for explaining variation in algal biomass (and O_2 patterns) in these nutrient-rich systems, but this remains to be examined mechanistically. Our results indicate that reducing the biomass of filamentous algae in the open-canopy streams may lead to improved habitat conditions. Controlling *Cladophora* is difficult (Dodds, 1991), and in an agricultural landscape such as east-central Illinois, it may not be possible to reduce nutrient concentrations sufficiently to limit the occurrence of blooms of filamentous algae.

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