BEFORE THE ILLINOIS POLLUTION CONTROL BOARD

IN THE MATTER OF:	)	
	)	
PROPOSED AMENDMENTS TO	)	**R 04-25**
DISSOLVED OXYGEN STANDARD	)	
35 Ill. Adm. Code 302.206	)	

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TO: See Attached Service List

PLEASE TAKE NOTICE that I have today filed with the Office of the Clerk of the Pollution Control Board the following documents:

WRITTEN TESTIMONY OF DR. JAMES E. GARVEY

a copy of which is served upon you.

ILLINOIS ASSOCIATION OF WASTEWATER AGENCIES,

By: Roy Harsch
One of Its Attorneys

Dated: October 4, 2006

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The undersigned certifies that a copy of the foregoing:

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Written Testimony of James E. Garvey

I thank the Illinois Pollution Control Board (Board) for allowing me to present my testimony. My name is Dr. James E. Garvey, Associate Professor of Zoology and Associate Director of the Fisheries and Illinois Aquaculture Center at Southern Illinois University Carbondale (SIUC). I also hold several other appointments such as Chair of the American Fisheries Society (AFS) Farm Bill Advisory Task Force, Executive Officer of the Illinois Chapter of the AFS, Member of the US Army Corps of Engineers Environmental Management Program - Project Sequencing Team, and North Central Representative of the Early Life History Section of the AFS. As you know, I am an aquatic ecologist with an active research program that revolves around environmental and human-induced factors influencing the abundance and distribution of fishes in lakes and rivers. I have published well over forty publications that are widely cited in the discipline of fisheries, aquatic ecology, and general ecology. I also have an active graduate training program. My graduate students often join natural resource agencies such as the Illinois EPA, the US Fish and Wildlife Service, and the Missouri Department of Conservation.

My participation in this process began over two years ago when the Illinois Association of Wastewater Agencies (IAWA) asked Dr. Matt Whiles and me to evaluate the current dissolved oxygen standard in Illinois. After an extensive literature review, we generated a report that stated that the current standard is too simplistic for the diverse waters of Illinois. We supported many of the recommendations that were developed in the US EPA National Criteria Document (NCD) for dissolved oxygen.

Review

Over the course of two years, much data collection, literature review, and discourse among stakeholders have occurred. I have attended all the stakeholder meetings and hearings before the Board; I have had the opportunity to review all the technical information and data presented in this rulemaking process thanks to the cooperation of the stakeholders. The end result of this process is that the recommendations that Dr. Whiles and I set forth largely have been supported. I have appeared before the Board on several occasions to present my findings. Recall, we recommended that a two-season standard be adopted throughout the state. During March through June, when the majority of early life stages of many fishes and other aquatic organisms are produced, we recommended a standard dissolved oxygen concentration be met that provides sufficient oxygen to support the metabolic needs of eggs and larvae. During this time of year, streams are typically flowing, primary productivity is accelerating but not peaking, and temperatures are cool to moderate. Thus, high dissolved oxygen concentrations are expected to be available to young aquatic organisms; this expectation has been well supported by my findings described in previous testimony. The literature and growing state-wide oxygen data set demonstrate that, for warm-water, low gradient systems common in Illinois, concentrations should not decline below 5 mg/L and weekly averages should not decline below 6 mg/L. We also suggested a 30-day running average of 5.5mg/L, which has little biological support in my view, but is recommended in the NCD.

As temperatures increase during summer, increased biological activity and water's reduced oxygen capacity should reduce dissolved oxygen concentrations, particularly during night. Evidence is mounting that the majority of reproduction of aquatic organisms in Illinois either occurs before July 1 (see Csoboth 2006 thesis, SIUC; Exhibit 1) or late-spawning organisms have early life stages that are tolerant to low dissolved oxygen concentrations (e.g., freshwater mussels). Thus, we recommended that during July through February, Illinois adopt a daily acute minimum of 3.5 mg/L and a seven-day average of daily minima of 4 mg/L. In previous testimony before the Board, I have demonstrated that streams that meet these dissolved oxygen conditions appear to contain diverse, robust biological assemblages. Those that do not are typically impaired.

During the past year, the Illinois Department of Natural Resources (IDNR) and the Illinois Environmental Protection Agency (IEPA) have proposed an alternative, two-tier oxygen standard for the state and have expended much energy to develop it. The "general use" tier is very similar to the IAWA state-wide recommendation with slightly higher concentrations. Also, the criteria for early life stages are extended through July. In addition, the agencies recommended an "enhanced oxygen" tier for streams that contain fishes and invertebrates that were found by Ohio Environmental Protection Agency to occur in Ohio waters with high average oxygen concentrations. My concern about this approach is that the selection of streams based solely on associations between aquatic organisms and average oxygen concentrations ignores other potential causal factors such as habitat quality, gradient, and temperature. Thus, coining these organisms as "oxygen sensitive" and then using them to select enhanced tier waters may by completely spurious. Only through experiments that establish causality between oxygen tolerance and fish life processes can tolerance be assessed. Again, these issues have been addressed in previous testimony when I described the research by Smale and Rabeni published in the *Transactions of the American Fisheries Society.* Recall, these investigators used a combination of lab assays and surveys to develop an index of oxygen sensitivity in Missouri streams.

Overview of Testimony

I present results that continue to support the recommendations in the Garvey and Whiles report. First, I review the results of recent peer-reviewed papers that show that dissolved oxygen concentrations in Illinois streams are difficult to predict and largely influenced by characteristics of stream habitat and morphology. I then explore the implications of the two-tier oxygen standard for Illinois using data that were collected both by the IDNR and IEPA as well as data that were collected by IAWA members. In my view, the most compelling results derive from stream segments slated for enhanced dissolved oxygen protection by the proposed IDNR/IEPA two-tier approach. As I analyzed these data, it became apparent that many of these segments likely violate both the IDNR/IEPA and perhaps the IAWA proposed standards, even though "enhanced oxygen" taxa are present in the streams. Further, daily discharge (i.e., volume of water moving per second through the stream) explained as much as 50% of the variation in daily median and minimum dissolved oxygen concentrations in several of these systems. Thus, the physical characteristics of streams interacting with flow largely drove much of the oxygen dynamics. In my view, this further complicates any attempts to fit a single standard to any stream in the State and renews the urgent need to develop tiered, habitat-based criteria that incorporate how discharge affects aquatic communities and water quality.

Literature Review

Several papers that were generated by Dr. Mark David and colleagues at the University of Illinois Urbana-Champaign through support by the C-FAR program recently have been published (Exhibit 2). Although the general expectation was for dissolved oxygen dynamics in their research streams in Illinois to be affected by nutrient loading, they found that stream physical characteristics, primarily basin shape and its propensity to hold organic matter and intercept light, were more important in influencing oxygen concentrations. As I have argued throughout this process and in the original IAWA-sponsored report, these results indicate that stream physical characteristics trump water quality and need to be the primary focus of standard development.

Analysis of Historical Grab Data and 2004-2005 Continuous Data

Illinois DNR/EPA provided me with "grab" dissolved oxygen data collected during 1994 through 2003 in streams that have fully met their aquatic use designation. In addition, they provided data from 2004 and 2005 collected with semi-continuous data logging probes in streams that have been tapped for inclusion in the "enhanced oxygen" tier. I sent the results I present below to Mr. Matt Short and Mr. Joel Cross for their review. As of the date I am drafting this testimony, they have not responded. The grab data demonstrate that median dissolved oxygen concentration declines during June through August relative to other months (Exhibit 3). Concentrations did decline below a benchmark of 5 mg/L during the summer months, although rarely. Given that these grabs were typically taken during the day, it is not surprising that relatively low dissolved oxygen concentrations were not frequently encountered.

The continuous data demonstrated that dissolved oxygen in "enhanced" segments more frequently declined below 5 mg/L and even occasionally below 3.5 mg/L (Exhibit 3). These low concentrations which often exceeded both the IAWA and DNR/EPA proposed standards typically occurred during the night through dawn. Interestingly, these enhanced-tier segments more frequently (up to 20% of observations) exceeded the DNR/EPA minimum of 5 mg/L during July than the IAWA proposed standard of 3.5 mg/L during that month (Exhibit 3). The streams that contained "oxygen sensitive" species failed to meet the standard set for them by the IDNR/EPA proposal.

On 24 April 2006, Mr. Toby Frevert sent a letter to Mr. Dennis Streicher including several disclaimers about the above data set. He indicated that the grab data were a worst case scenario, including only data collected in the early morning hours. On the contrary, the data set I received from the agencies and recently sent back to them for confirmation included grab data that were collected during morning through afternoon (median collection time was 1100 hours, with times as late as 1700; Exhibit 3). Thus, it appears to me that the data represent the range of daily conditions that affect oxygen concentrations. Time of day was positively related to DO concentration in this data set, but explained less than 1% of the variation. Although the continuous data show that the enhanced streams cannot meet the IDNR/IEPA expected standard, Mr. Frevert noted that these data included results from 2005 when a drought gripped much of the state. Because these results were collected under extreme conditions, he argued they should be discounted. I respectfully disagree.

Few laws exist in the tangled and complex discipline of ecology. However, one of the most commonly agreed tenets in our discipline is Liebig's Law of the Minimum, taught in every general ecology course, including my own at SIUC. Liebig aptly noted that the distribution of all living organisms will not be dictated by the average conditions, but rather the availability of the most limiting condition. This condition does not always have to be limiting, but only when organisms are experiencing some critical period such as reproduction or growth. The condition could be an occasionally limited nutrient, or in our case, oxygen. In other words, the occasional "worst case" scenario which limits the oxygen available to the local fauna will determine the species composition and abundance present at all times. Only by identifying the limiting conditions, in other words the acute minimum oxygen concentration, can we determine what should be present through time. The extreme drought conditions in the enhanced streams likely provided the worst case scenario and thereby insight into what the acute minimum should be to support a diverse aquatic assemblage. The proposed minimum standard of 3.5 mg/L was rarely exceeded in these streams (Exhibit 3) and likely is near that extreme lower limit.

Illinois Water Survey Data

Illinois DNR via Ms. Ann Holtrop provided me with grab dissolved oxygen data from various studies compiled through the Illinois State Water Survey. These data extend from the early 1970s through the 1990s. After reviewing the reports from which these data were collected (see Exhibit 4), it was clear that the 20,101 individual observations that I analyzed were collected in many ways. Even given this caveat, I thought it might be interesting to determine whether average dissolved oxygen concentrations "improved" in Illinois surface waters through time as nutrient loading abated during the past thirty years as a function of the Clean Water Act. I was rather surprised to find that no real pattern occurred through the decades, with concentrations varying widely among sites and years for which data were available. As per the results emerging from Dr. David's laboratory as well as the results I will present below, it appears that oxygen concentrations in streams are likely influenced by habitat and its interactions with many other factors, of which nutrient loading is but one component.

IAWA 2005 and 2006 Semi-Continuous Monitoring

Several IAWA members have installed semi-continuous dissolved oxygen loggers (15-60 minute intervals depending on the source) in streams that are in segments slated for enhanced tier standards by the agencies. Segments for which I have received data are on the Fox, DuPage, Kickapoo, Rock, and Vermilion Rivers (Exhibit 5; 24,575 individual observations). With the exception of the Fox River where the data derive from 2005, the remainder of the data derives from summer 2006. I also procured USGS daily monitoring data for discharge from gauging stations near the river segments to test the hypothesis that discharge drives much of the variation in dissolved oxygen concentrations in low-gradient Illinois streams. The IAWA members who collected the data have reviewed these summary results.

Dynamics of dissolved oxygen vary widely among the enhanced tier stream segments (Exhibit 5), from daily concentrations varying widely in the Fox River to less so in the Vermilion River. Both median and minimum daily dissolved oxygen concentrations typically declined as the summer progressed in the Fox, DuPage, and Kickapoo Rivers, but not the others (Exhibit 5). Probably the most compelling result is the linear or log-linear relationship between daily discharge and median and minimum daily dissolved oxygen concentrations in the streams (Exhibit 5). In 2005 for the Fox

River, dissolved oxygen concentrations declined sharply with declining daily discharge (Exhibit 5). Conversely, in the other streams during 2006, dissolved oxygen concentrations were either unrelated to discharge or negatively related (Exhibit 5). I could speculate broadly about the underlying mechanisms including flow-related biochemical oxygen demand, hypoxic groundwater intrusion, and changes in water quality due to run-off. Regardless of the underlying causes, given that discharge can explain up to 50% of the variation in dissolved oxygen concentrations during both severe drought (2005) and non-drought years, this issue needs to be incorporated into standard development and interpretation.

I applied both the enhanced tier standard and the proposed IAWA standard to the semi-continuous data. Typically, both standards demonstrate that several of the stream segments including those in the DuPage, Fox, and Kickapoo Rivers fail to meet the season-dependent acute minima, even given the proposed enhanced status of these systems (Exhibit 6). This is not surprising given that some portions of the DuPage and Fox Rivers are currently listed with low dissolved oxygen as a probable cause for impairment (see map in Exhibit 5). However, the Rock River which is listed as impaired due to low oxygen did not fail to meet any of the minimum criteria (Exhibit 6).

Seven-day means ending in July for IAWA and August for the IDNR/IEPA proposals were generally insensitive (Exhibit 6). Interestingly, the IAWA proposed 7-day minimum standard of 4 mg/L which applies during July through February generated more violations than the IDNR/IEPA 7-day mean minimum of 4.5 mg/L which starts in August (Exhibit 6). Although I did not expect this to occur, apparently applying the mean minimum criterion during July as per the IAWA proposal is more sensitive.

Because the daily variation in dissolved oxygen concentrations differs more than the daily average (i.e., it is the variation not the mean that is sensitive), it appears that the mean-minimum criterion is more sensitive to frequent declines in oxygen during the summer. In my view, it appears that many of these streams, particularly the Fox River, fail to provide adequate oxygen for aquatic life during part of the summer. This causes me to question the linkage between the aquatic assemblages used to select the sites for enhanced status and oxygen needs of the resident organisms.

Summary

One of the major conclusions of the Garvey and Whiles report was that we have much to learn about associations between aquatic organisms and spatial and temporal heterogeneity in dissolved concentrations of warm surface waters in the US. Since that report was completed, I have had the privilege of exploring this issue in depth and receiving some unprecedented (and fun) data sets. As Liebig stated generally for all ecology, it is clear that oxygen can become a limiting dissolved gas for aquatic organisms and, below some threshold concentration, we should expect to see deleterious effects and reductions in species composition and abundance. To this date, all the data I have reviewed suggest that a threshold does exist and that it occurs during the summer when concentrations are less than or equal to 3 mg/L as stated in the NCD and the Garvey and Whiles report. If a stream remains consistently above this level (i.e., never violates a 3.5 mg/L minimum), oxygen is no longer limiting for life and some other factor then limits organisms….probably habitat. All of the stream data and the literature (see Dr. David's research) support this view.

I favor scrapping dissolved oxygen as a standard altogether. Although under extreme conditions it can become limiting (e.g., in the Gulf of Mexico hypoxic zone), variable or low concentrations are largely a symptom of habitat problems and their interactions with other factors such as chemical and biological pollutants … and, as this testimony suggests, discharge. However, given that this is not currently a possibility, it appears that the set of standards proposed in the Garvey and Whiles report stand the test of the data and should be adopted in the interim. I do urge the stakeholders to move rapidly toward a habitat-based tier designation where oxygen is but one of a suite of physical and chemical parameters used to diagnose root causes and develop sound solutions.

EARLY LIFE HISTORY OF FISHES IN RESTORED AND UNRESTORED BACKWATERS

by

Laura A. Csoboth

B.S., University of Delaware, 2002

A Thesis
Submitted in Partial Fulfillment of the Requirements for the
Master of Science Degree

Department of Zoology
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December 2006

[redacted]

THESIS APPROVAL

EARLY LIFE HISTORY OF FISHES IN RESTORED AND UNRESTORED BACKWATERS

By

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A Thesis Submitted in Partial

Fulfillment of the Requirements

for the Degree of

Master of Science Degree

in the field of Zoology

Approved by:

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AN ABSTRACT OF THE THESIS OF

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TITLE: EARLY LIFE HISTORY OF FISHES IN RESTORED AND UNRESTORED BACKWATERS

MAJOR PROFESSOR: Dr. James E. Garvey

River modifications have degraded backwaters, reducing critical habitat for larval fishes. Restoration projects, such as on the Illinois River's Swan Lake backwater, may be improving fish spawning and nursery areas. During 2004 and 2005, I (1) assessed restored and unrestored backwaters as fish nurseries by quantifying the density of larval fishes and their zooplankton prey and (2) investigated movement of larval fish between the river and Swan Lake by setting drift nets on a diel and seasonal basis.

A flood pulse occurred during June 2004, but no spring or summer flood event during 2005. When restored and unrestored backwater sites were compared to the Illinois River, families of fish changed between years, with fluvial-dependent taxa present during the flood pulse of 2004 and relatively absent during the drought of 2005. Higher fish densities and larger fish larvae were collected in backwaters as compared to the Illinois River during both years, probably due to abundant zooplankton and warmer temperatures. Larval drift was positively related to water velocity during 2004, and an estimated 32.3 million larvae drifted into Swan Lake. No other relationships between drift direction and abiotic variables occurred. In 2005, drift was related to larval density, probably due to the lack of a flood pulse.

Backwater-river connectivity in the Swan Lake HREP was preserved through the construction of water control stop-log structures, which allowed continual access by all fishes. The backwater management plans at Swan Lake enhanced habitat diversity and realized beneficial effects by creating a diverse and abundant age-0 fish community.

ACKNOWLEDGMENTS

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CHAPTER 1

OVERVIEW

River floodplain habitats are an endangered landscape (Tockner and Stanford 2002), which is a concern because they function as a critical component in the early life history of fishes by providing rich food resources and structural refuge (Junk et al. 1989). In their natural state, rivers are dynamic systems where flood pulses cue biological processes, increase productivity, and maintain diversity (Junk et al. 1989). These flood pulses also alter the river's course, where meanders become backwaters, and backwaters become marshland or low-land forests (Lusk et al. 2003). Continued meandering of the river continually creates and alters new backwaters habitats. The degree of connectivity between a river and backwater affects the backwater's ability to function as a spawning and nursery habitat (Grift et al. 2003). The higher the connectivity, the more available the backwater is for fish use.

In backwater habitats, congregations of larval fishes vary in species composition and abundance through time (Holland 1986). To explain this, I adopted a generalized life history classification based on fishes' relative dependence to flowing water (adapted from Galat and Zweimuller 2001). The first early life history class is classified as fluvial specialists which spend most of their life in flowing water and rarely use backwater habitats. Examples of these fishes are sturgeon *Acipenseridae* spp. and perches *Stizostedion* spp. The second category, fluvial dependents, have only a portion of its life history requiring flowing water (Galat and Zweimuller 2001). These fishes, like temperate basses *Morone* spp., paddlefish *Polyodon spathula*, and freshwater drum *Aplodinotus grunniens*, either spawn in the river and their larvae move laterally into backwaters or spawn in the backwaters and their larvae move into rivers at some point during their first year (Dettmers et al. 2001, Galat and Zweimuller 2001). The extent of this larval fish movement between rivers and adjacent, connected backwaters is currently undocumented. Finally, macrohabitat generalists, such as sunfishes *Lepomis* spp., shads *Dorosoma* spp., and gars *Lepisosteus* spp., conduct most of their life in backwater areas. When macrohabitat generalists are found in the river, they typically use it as a corridor to access other backwater areas (Junk et al. 1989). Based on these life history classes, many riverine fishes rely on semi- or permanently-connected backwaters.

Backwater habitat and river connectivity have been reduced because of high sedimentation rates (Havera and Bellrose 1985). In a natural floodplain system, sedimentation of some backwaters would be counteracted by the creation of other backwater areas (Miranda 2005). Unfortunately, because rivers have become immobilized due to levee construction and floodplain settlement, existing backwaters are becoming terrestrialized without new backwaters being created (, Lusk et al. 2003, Miranda 2005).

Spurred by backwater research in the 1980s, the Habitat Restoration and Enhancement Program (HREP) was initiated as a partnership between state and federal agencies to fund large-scale restoration projects in the Upper Mississippi River (USACE 1993). The continuing goal of the HREPs is to improve habitat for waterfowl and fish by reducing sedimentation and enhancing backwater-river connectivity. Few studies have documented the effects of these HREPs and their subsequent management regimes on river fishes. Fewer still have documented the effect of these improvements on spawning and nursery habitats.

This study was conducted on the lower Illinois River (ILR) and contained two backwater complexes, the HREP-managed Swan Lake (river kilometer [RKM] 8.0) and natural Calhoun Point (CP, RKM 1.6), and two adjacent river segments, RKM 1.6 and RKM 8.0 (measuring from its confluence with the Mississippi River; Figure 1). Historically, Swan Lake was only connected to the river at its downstream end where a 0.5 km-wide opening to the river existed. To reduce sediment loading, this opening was restricted to the width of a stop-log water control structure during restoration, about 5 m wide. Additionally, the backwater was compartmentalized by constructing cross-levees, creating a lower compartment (Lower Swan [LS]), which was managed to be continuously connected to the river, and a middle compartment (Middle Swan [MS]), which was drawn down each year during this study to consolidate sediments and promote moist-soil vegetation. The CP backwater complex (466 ha), at the confluence of the Illinois and Mississippi Rivers, had a low-lying levee, isolating the backwater and maintaining water levels during non-flood periods (Figure 1). Thus, LS was a restored, unmanipulated backwater, MS was a restored and manipulated counterpart, and CP was an unrestored off-channel backwater.

Therefore the objectives of this study were two-fold. First, I assessed the benefit of a restored HREP-supported backwater system relative to an unrestored, manipulated backwater by quantifying the response of larval fish communities. Secondly, I investigated the interplay of life history strategies with lateral drift dynamics on a diel and seasonal basis within a restored backwater system. I tested the hypotheses that:

i) Backwater sites, whether restored or unrestored, would be used as spawning and nursery habitat in higher densities than river segments sampled (Holland 1986, Junk et al. 1989).

ii) Seasonal abundances of larval fishes would be different among sites based on their river connectivity, with the site being the most connected (i.e., LS) to the river yielding the highest larval abundance.

iii) Family composition, described by the life history classification, would also vary among sites, where sites with the lowest river connectivity would contain relatively higher densities of macrohabitat generalists (Miranda 2005).

iv) Larval exchange would occur between the restored backwater, LS, and the lower ILR, such that an influx of larvae into the backwater would be related to rising water temperatures and river stage (Junk et al. 1989).

v) Because Swan Lake is a major backwater of the lower ILR, ingress or egress of larvae between LS and the ILR would result in density differences between river segments upstream and downstream of the backwater-river confluence (Sheaffer and Nickum 1986). However, restriction of the LS-ILR confluence may affect the magnitude of larval movement between the river and backwater.

vi) Diel periodicity in larval drift patterns may occur where drift rates and sizes of larvae differ among sampling times (Gale and Mohr 1978, Gadomski and Barfoot 1998).

These are ecological and management-oriented questions regarding backwater function and the influence backwater restoration may have in the early life history of fishes. Understanding larval dynamics and production within backwater systems, especially within restored habitats, will enable researchers to determine the quality of backwaters as larval fish nurseries and to address the influence of habitat alterations on larval fish assemblages among backwater lakes and river reaches.

CHAPTER 2

THE ROLE OF BACKWATER RESTORATION IN LARVAL FISH ECOLOGY

ABSTRACT

Large river modifications have widely degraded backwaters, reducing critical habitat for larval fishes. During 2004 and 2005, I assessed how river backwaters function as fish nurseries by quantifying the response of larval fish communities to restoration of Swan Lake, a major (>1,000 ha) backwater complex on the lower Illinois River. The response was compared to a nearby unrestored backwater complex and to adjacent river segments. The densities of zooplankton prey in each site also were quantified. Families of fish changed between years, with fluvial-dependent taxa present during the flood pulse of 2004 and relatively absent during the drought of 2005. During both years, tenfold greater larval densities were produced in backwaters than the river. Larvae were larger in backwaters and during the non-flood year, probably due to abundant zooplankton. All backwaters produced similar larval densities regardless of restoration. Growth was highest in the most isolated and regulated backwater portion of Swan Lake. Predictable flood pulses coupled with habitat heterogeneity in the backwaters may be important for larval abundance, assemblage composition, and recruitment. Current site-specific restoration efforts constructed with the built-in flexibility to adapt to other management regimes will likely be most beneficial to all early life history strategies of fishes, while also providing widespread benefits and supporting the move towards system-wide management programs on large rivers.

INTRODUCTION

Natural river ecosystems support abundant and diverse species assemblages due to high habitat diversity and physical complexity (Junk et al. 1989, Dettmers et al. 2001). Off-channel aquatic habitats, such as backwater lakes and sloughs, provide productive habitat and a lentic-lotic gradient whereby fishes find structural refuge, food resources, and spawning and nursery grounds (Junk et al. 1989). For many riverine fish species, floodplains are optimal spawning and nursery habitats and are actively sought out via lateral spring spawning migrations of adults (Molls 1999). However, river regulation and degradation have reduced backwater habitat quality and river connectivity, jeopardizing larval fish abundance and success (Havera and Bellrose 1985, Tockner and Standford 2002). Larval fish diversity and recruitment are likely related to river-backwater connectivity, such that reduced access could alter riverine fish communities (Pezold 1988, Turner et al. 1994). Although this association with the river is critical to backwater health (Gore and Shields 1995), few studies have related connectivity to early life history requirements and larval fish communities (Miranda 2005).

Despite the important links between backwaters and larval fish abundance, floodplain habitat has become severely degraded and is among the most endangered landscape in the world (Tockner and Stanford 2002). Dam and levee construction coupled with high sediment loads from agricultural lands extensively modified large river hydrology, altered the flood-pulse, reduced backwater habitat quality and river connectivity, and decreased system productivity (Havera and Bellrose 1985). In the 1980s, the loss of backwater habitat along the Illinois River spurred research and the development of projects aimed to restore essential fish and waterfowl habitat (Sheehan et al. 1990). A Habitat Restoration and Enhancement Project (HREP), through the federally supported Environmental Management Program (EMP), was initiated on Swan Lake, a major backwater of the Illinois and Mississippi Rivers. One goal of the Swan Lake HREP, to improve spawning and nursery habitat for fishes, was addressed by controlling river connectivity of the backwater and increasing habitat heterogeneity (USACE 1991).

HREP techniques must be evaluated because current paradigms that influence river management and serve as the basis for these restorations will continue to be implemented in future programs. However, a paucity of information about life histories and habitat needs of larval and juvenile fishes in large river ecosystems, which may hinder restoration efforts and effectual progress (Galat and Zweimuller 2001). All riverine fishes exhibit certain life history strategies based on their relative dependence on flowing water; some are specialized for riverine environments, while others require flowing water for only a portion of their life history. Lastly, generalist fishes reside mostly in lentic backwater areas, especially during their first year of life. Therefore, understanding larval dynamics and abundance within restored habitats will enable researchers and managers to determine the influence of habitat alterations on larval fish assemblages, where alterations that increase connectivity may produce more diverse and abundant assemblages. I quantified the response of larval fish communities within a restored HREP-supported backwater system relative to an unmanipulated off-channel complex to assess the benefit of this HREP restoration program. I tested the hypotheses that (1) fish and zooplankton densities differed among sites and between years and that (2) life history classes (i.e., family composition) varied among sites or between years. I also evaluated how flood regimes and backwater habitat heterogeneity affected larval abundance and composition within managed and natural backwater habitats.

METHODS

STUDY AREA

The study area was located on the lower Illinois River (ILR) and contained two backwater complexes, HREP-managed Swan Lake (river kilometer [RKM] 8.0) and natural Calhoun Point (CP, RKM 1.6), and two adjacent river segments, RKM 1.6 and RKM 8.0 (measuring from its confluence with the Mississippi River; Figure 1).

Historically, Swan Lake was only connected to the river at its downstream end where a 0.5 km-wide opening to the river existed. During restoration, a cross-levee was constructed to create a lower compartment (Lower Swan [LS], 567 ha) and middle compartment (Middle Swan [MS], 485 ha; Figure 1), and the historic LS connection was restricted to the width of a stop-log water control structure, about 5 m wide. An additional river connection was created in MS via a stop-log structure. During normal pool stage, the stop-log water control structures at LS and MS were the only avenue through which backwater-river movement of larval fish could occur. The cross-levee and stop-log connections allowed the compartments to be managed independently (Figure 1). Lower Swan was managed to be continuously connected to the Illinois River, while the MS water control structure was opened in early winter, allowing spawning fish to access the lake before it was disconnected from the river in early spring and pumped a meter or two below pool level each June. Moderate flooding could top the MS stop-logs in the water control structure and the cross-levee, which would serve to connect MS to the river and LS during additional times of the year. Thus, LS was a restored, unmanipulated backwater, while MS was a restored and manipulated counterpart.

The CP backwater complex (466 ha), at the confluence of the Illinois and Mississippi Rivers, was unrestored and regularly connected to the rivers during floods via low-lying levees (Figure 1). These levees provided limited flood control, and functioned more to isolate the backwater from the river and maintain water levels during non-flood periods. This backwater system was dendritic and contained many vegetated islands. However CP was also shallow and had an unconsolidated lakebed. This system represented an unrestored, unmanipulated backwater.

LARVAL ABUNDANCE

To understand drift patterns and compare the outcome of management practices on study sites, I quantified seasonal larval abundance using paired, bow-mounted ichthyoplankton nets (0.5 m diameter x 2 m long, 500-µm mesh). River and backwater habitats were sampled during late March through September 2004 and 2005. Each week, four stratified transects were randomly chosen within each Swan Lake compartment and two transects were randomly chosen in the ILR (RKM 8.0). Every two weeks, I sampled CP and its adjacent segment of the ILR (RKM 1.6). I stratified backwater transects into inshore and offshore tows, with two inshore plus two offshore transects per Swan Lake compartment, and one inshore plus one offshore tow within CP (N = 10 backwater transects). River tows were conducted within one km upstream and one km downstream of each backwater-river confluence (N = four river transects). In MS, sampling ended on 22 July 2004, and on 27 June 2005, because the summer drawdown made the lake too shallow to sample (i.e., water level management program). Calhoun Point was not sampled during 2005 due to low water levels.

At each transect, tows were conducted at the surface for five minutes at a constant speed, with a calibrated mechanical flow meter (Model 2030R, General Oceanics, Inc., Miami, Florida, USA) mounted in the mouth of one net to estimate volume sampled and standardize samples. Inshore backwater tows followed the shoreline, offshore backwater tows were straight transects, and river tows were straight transects conducted perpendicular to flow direction. River tows started at the main channel border and continued across to the opposite main channel border. If five minutes had not passed by the time the opposite side was reached, the direction was reversed with nets still in the water, and sampling continued until five minutes had elapsed.

Upon completion of each transect, net contents were flushed into the cod end and preserved in 95% ethanol. Samples were split to approximately 200 fish using a Folsom plankton splitter (Aquatic Research Instruments, Hope, Idaho, USA). All age-0 fish in the subsample were counted, identified to the lowest possible taxon, typically genus, and classified to a developmental stage (i.e., yolk-sac, larval, juvenile) using descriptions and keys in Soin and Sukhanova (1972), Auer (1982), Murty et al. (1986), Tweb et al. (1990) and voucher specimens from Southern Illinois University's Fluid Vertebrate Collection (Brooks Burr, Carbondale, Illinois, USA) and Colorado State University's Larval Fish Laboratory (Darrel Snyder, Fort Collins, Colorado, USA). A subsample of fish from each taxon and stage identified was measured (total length [TL]; mm) using Scion Image® software or metric calipers (N = 10 per net). During 2005, MS samples were counted, but not identified.

At the start of each transect, I sampled water chemistry parameters that could affect the density of larval fish, including temperature (°C), dissolved oxygen (mg/L [YSI Model 52 Dissolved Oxygen Meter; Yellow Springs Instruments, Yellow Springs, Ohio, USA]), water depth, secchi depth, and average wind speed (km/h [Kestrel 1000, NFS – Radiation Protection Systems, Inc., Groton, Connecticut, USA]). Surface water velocity (cm/s) was measured in the main channel at river transects with an electronic (Flo-Mate Model 2000, Marsh McBirney, Inc., Frederick, Maryland, USA) or mechanical flow meter (Model 2030R Flowmeter, General Oceanics, Inc., Miami, Florida, USA). Continuous monitoring temperature loggers (8-bit Minilog-TR, Vemco Ltd., Nova Scotia, Canada) were located in each backwater and the river to supplement the water temperature data taken at each transect. River stage data were recorded at Hardin, Illinois by the St. Louis District U.S. Army Corps of Engineers (river kilometer 34.4). Swan Lake channel depth data were recorded using a Doppler unit (Model 6526-51 Starflow Ultrasonic Doppler Flow Meter; Unidata America, Lake Oswega, Oregon, USA). This unit was anchored to the bottom of the water control structure and continuously recorded temperature (°C) and depth (mm). Depth data were not available for MS and CP.

LIFE HISTORY CLASSIFICATION

Larval and juvenile fish collected were grouped by family into one of three generalized classes to better explain trends between years and treatment groups (Galat and Zweimuller 2001; Table 1). The classes were: fluvial specialists, fluvial dependent, and macrohabitat generalists. Fluvial specialists inhabit streams and rivers throughout their entire life and rarely enter floodplain habitats (Galat and Zweimuller 2001). Fluvial dependent species regularly use lentic backwater or reservoir habitats, but certain life stages depend on lotic environments (Galat and Zweimuller 2001). These species are typically broadcast spawners, where developing eggs and larvae are semi-buoyant and passively drift in wind-induced or downstream currents (Holland 1986). Adult fluvial dependent fishes also may make lateral migrations into slow-flowing lentic areas to spawn—activities likely corresponding with increasing temperatures and rising water levels (Junk et al. 1989). Macrohabitat generalists include species commonly found in reservoirs and off-channel habitats that do not depend on lotic systems (Galat and Zweimuller 2001). When these fishes use the river, it is as a corridor to move among backwaters (Junk et al. 1989, Dettmers et al. 2001). Spawning usually occurs in off-channel habitats and offspring generally do not leave this habitat until the juvenile stage (Holland 1986). I grouped families based on Galat and Zweimuller (2001; taxonomy from Nelson [1994]), and the only deviation from their groupings was Sciaenidae, which I classified as fluvial dependent based on life history descriptions from Dettmers et al. (2001) and Koel and Sparks (2002).

ZOOPLANKTON DENSITY

Because zooplankton abundance and composition affect growth and survival of exogenous feeding larvae (Miller et al. 1988, Miller et al. 1990), zooplankton were sampled in each backwater and river segment. Similar to larval tows, I randomly stratified samples between inshore/offshore and upstream/downstream habitats. Backwaters were sampled before a corresponding inshore and offshore larval tow ($N =$

two samples/backwater), and river sites were sampled in the thalweg at each upstream and downstream transect (*N* = two samples/river segment). Samples consisted of four, 1-m vertical hauls from the boat using a conical net (0.5 m x 2 m, 64 μm mesh). If depth was less than 1 m, the entire water column was sampled four times, with the depth noted to adjust volume sampled. During 2005, neither MS nor CP were sampled.

After each haul, net contents were flushed into a removable collecting bucket and rinsed into a 64 μm sieve. All four hauls from one site were preserved with 10% buffered sugar-formalin in a single container (Haney and Hall 1973), and returned to the laboratory for processing. Copepods including cyclopoids, calanoids, and nauplii, cladocerans including *Bosmina* spp., *Chydorus* spp., *Daphnia* spp., *Diaphanosoma* spp., and *Moina* spp., and other zooplankters including ostracods were identified and counted. Rotifers were dyed with Rose Bengal and counted, but not identified. Using a Henson-Stempel Pipette, each sample was processed until two taxa reached counts of 200 or until 10% of the sample volume had been processed. Due to the high density of rotifers, they were counted until approximately 100 individuals had been processed. Densities were calculated by dividing number of taxa in a subsample by the fraction of subsample counted, then dividing that amount by the total volume of water filtered in the field. Zooplankton were grouped as rotifers and macro-crustaceans (i.e., copepods, cladocerans, nauplii, etc.) for data analysis due to large differences in density.

DATA ANALYSIS

To examine environmental relationships among sites and between years, weekly means of abiotic variables were analyzed using principle component analysis (PCA). The abiotic matrix, containing temperature (°C), dissolved oxygen (mg/L), depth (m), secchi (cm), and wind (km/h) data, was analyzed using PC-ORD with the correlation option to center and standardize parameters (McCune and Medford 1999).

To standardize samples by volume, densities for larval tow and zooplankton data were calculated as fish/m^3 and zooplankton/L. Total larval abundance at each site was conservatively calculated as the sum of weekly densities. For the abundance estimate, when a site was not sampled or processed during a week, the larval density from the previous weeks was substituted for the missing value. All data were log-transformed to meet assumptions of normality. Two-way repeated measures ANOVA (proc MIXED, SAS Institute 1999) was used to test for differences among sites and between years that were sampled over time (Hurlbert 1984). For tow and zooplankton data, mean densities among sites were compared over time.

Fish length data were analyzed to determine whether sizes differed among tow and drift samples. Proportions of fish per 1-mm length group were calculated and used to make pairwise comparisons. Kolmogorov-Smirnov tests were used to compare size structure of fish collected from tows and between year differences for LS and ILR tow data. Size structure among tow sites was compared using Bonferroni adjusted α-values. Because MS was drawn down before the end of the 2004 sampling season, pairwise comparisons within this site only included samples collected on or before 23 July.

I determined overall trends within and between years for larval and zooplankton densities. A one-way ANOVA design tested density differences between years for larvae and zooplankton. I investigated larval and zooplankton density differences within each year using a two-way ANOVA randomized block design. The randomized block design test assessed hypotheses about overall lake effects: (1) stratified habitats differed in density (i.e., inshore v. offshore), (2) sites differed in density (i.e., LS v. CP), and (3) fish families and zooplankton groups varied among sites. To control for experimentwise error rates, Tukey-Kramer post-hoc (Sokal and Rohlf 1995). Pearson correlation coefficients were calculated for untransformed larval and zooplankton densities to reveal any association between larvae and their food source.

RESULTS

ENVIRONMENTAL FACTORS

A moderate flood pulse occurred in 2004, during which water levels in the lower Illinois River were above flood stage for approximately five weeks during June (Figure 2). In contrast, water levels remained at or below normal pool level of 128 m during the 2005 sampling season.

Water temperatures in the river gradually rose and fell during 2004, peaking in late July at 28 °C, and varied in 2005, exceeding 30 °C twice (Figure 2). Lower Swan conditions varied in a manner similar to the ILR, although water temperatures were higher and more variable and depths fluctuated less (Figure 2). The 2004 flood pulse topped the water control structure at MS and the low-lying levee at CP, causing water levels in these backwaters to rise rapidly and connecting all the backwaters to the river. Middle Swan and CP remained isolated during 2005.

From the PCA output, two axes, which had broken-stick eigenvalues less than the actual eigenvalues, were used to graphically represent the data (Jackson 1993). River and backwater sites were spatially separated by water and secchi depths, while water temperatures and other abiotic parameters were similar among all sites (Figure 2 & 3, Table 2 & 3).

LARVAL ABUNDANCE

During both years, fish larvae in the backwaters first appeared in low densities during late March and early April, whereas larvae were not collected in river sites until late April (Figure 4). Backwaters consistently produced higher densities of larvae than the ILR ($P < 0.01$, Table 4), with 2004 and 2005 ILR densities peaking at means of 11 and 14 fish/m^3 (Figure 4), and seasonal abundance was estimated at 41 and 31 fish/m^3, respectively. A peak of larvae occurred in all backwaters and the river during June 2004, the period of floodwater inundation (Figure 4). During that time, Lower Swan and CP experienced a larval pulse of similar size (Figure 4, Table 4), and had similar total seasonal abundance estimates of 435 and 409 fish/m^3, respectively. Middle Swan larvae peaked at the highest density of 400 larvae/m^3, though they were not statistically higher than LS (Table 4), and total seasonal abundance was estimated at 1,276 fish/m^3. These among site differences (two-way repeated measures ANOVA; 2004: $F_{3,10} = 55.37$, $P < 0.01$) differed across time (2004: $F_{21,133} = 55.06$, $P < 0.01$) and also interacted (2004: $F_{43,133} = 8.07$, $P < 0.01$). During 2005, sites also differed (two-way repeated measures ANOVA: $F_{2,9} = 53.37$, $P < 0.01$) across time (2005: $F_{22,115} = 15.05$, $P < 0.01$) and interacted (2005: $F_{31,115} = 5.00$, $P < 0.01$). Larval pulses did not occur in synchrony among sites, but occurred in MS during May, LS during late May, and the ILR during June 2005 (Figure 4). My estimation of MS larval abundance during 2005, at 516 fish/m^3, was less than LS' estimated total larval abundance. However, during the same time interval, of late-March through 23 July 2005, MS had a higher estimated larval abundance than LS. Seasonal abundance in LS increased between years to 531 fish/m^3 during 2005.

Each site had homogeneous larval distributions, with no differences between inshore and offshore or upstream and downstream stratified transects ($P > 0.05$, Table 5). During 2004, offshore transects in Swan Lake had slightly higher densities than inshore transects, although they were not significantly higher (Table 5).

The larval fish assemblage in the backwaters was comprised predominantly of macrohabitat generalists during both years, but changed between years in the river (Figure 5). In the ILR, fluvial dependent taxa, consisting mostly of sciaenidae, catostomidae, and cyprinidae, occurred at higher densities during 2004, but were relatively absent during the low water year of 2005 (Figure 5). Macrohabitat generalists dominated the ILR during 2005 (Figure 5). MS contained higher densities of fluvial dependents (i.e., catostomids, cyprinids) compared to other backwaters during 2004 (Table 6, Figure 5). Clupeids, the most abundant family, drove macrohabitat generalist patterns in all sites and heavily influenced system-wide trends in total density (Table 6). Despite being continuously connected to the ILR, LS had substantially higher densities of every family sampled except sciaenidae and moronidae during 2004. During 2005, higher densities of these families were collected in LS compared to the ILR, though these larval densities in LS were lower than during 2004.

Larval sizes were larger in backwater systems than the ILR, regardless of year (Figure 6). During 2004, Calhoun Point had larger fish than MS and LS, possibly indicating this backwater provided better fish nursery habitat ($P < 0.001$, Table 7, Figure 6). Progressively smaller sized fish occurred in MS, LS, and the smallest were collected in the ILR (Table 7). The size structure of fishes in LS and ILR were significantly larger during 2005, though larger fish again occurred in the backwater (all $P < 0.001$, Table 7).

ZOOPLANKTON DENSITY

Patterns in zooplankton and rotifer density differed by site and exhibited temporal patterns. During 2004, macro-crustaceans, consisting mostly of cladocerans, copepods, and nauplii, peaked during late May and June in the ILR, LS, and CP, while MS densities steadily declined during April through June (Figure 7). The opposite occurred for rotifer densities, with the ILR, LS, and CP showing declining densities during late May and a pulse of rotifers occurring in MS during 2004 (Figure 8).

Although rotifers dominated the zooplankton assemblage during both years, 2005 rotifer densities appeared greater while macro-crustaceans occurred in lower densities than 2004. In the ILR, while there was no between-year difference in rotifer density ($F_{1,34} = 1.28$, $P = 0.27$; Figure 8), higher densities of macro-crustaceans occurred during 2004 ($F_{1,34} = 12.43$, $P < 0.01$; Figure 7). In LS, the opposite occurred, with higher rotifer densities during 2005 ($F_{1,34} = 8.46$, $P < 0.01$; Figure 8) and no difference of macro-crustacean densities between years ($F_{1,34} = 1.43$, $P = 0.24$; Figure 7). The repeated measures analyses for 2004 revealed temporal variation in density, but no difference among sites for rotifers (site: $F_{3,63} = 0.80$, $P = 0.50$; week: $F_{15,63} = 2.61$, $P < 0.01$; lake*week: $F_{32,63} = 1.20$, $P = 0.27$) or macro-crustaceans (site: $F_{3,64} = 1.63$, $P = 0.19$; week: $F_{15,64} = 11.92$, $P < 0.01$; lake*week: $F_{3,64} = 5.37$, $P = 0.01$). During 2005, all effects were significant for both rotifer densities (site: $F_{1,2} = 91.56$, $P = 0.01$; week: $F_{21,33} = 2.74$, $P < 0.01$; lake*week: $F_{16,33} = 3.70$, $P < 0.01$) and macro-crustaceans (site: $F_{1,2} = 93.56$, $P = 0.01$; week: $F_{21,33} = 3.69$, $P < 0.01$; lake*week: $F_{16,33} = 2.89$, $P < 0.01$).

Macro-crustaceans in LS and the ILR were positively correlated to larval fish density during 2004 (LS: $r = 0.71$, $P < 0.01$; ILR: $r = 0.63$, $P = 0.01$). Only rotifer densities in MS correlated with larval densities ($r = 0.80$, $P < 0.01$). No other patterns occurred during 2004, and no correlations occurred during 2005 for either rotifers or macro-crustacean densities and fish larvae.

DISCUSSION

Connectivity is a critical feature of floodplain habitats, and likely influenced larval abundance patterns in the lower ILR and its backwaters. Larval abundance during the flood year was synchronized among all sites; well-timed to the period of inundation where all sites were connected. More families in greater evenness also occurred during the flood year. Although typically isolated from the river, a large variety of families occurred in MS during 2004, most notably a large proportion of catostomids and cyprinids that likely were entrained in the backwater when floodwaters topped its levee. This restored and manipulated site also produced more larvae during both years than any other site in the same time frame, which hints towards the benefits garnered from regular periods of connectivity and increased aquatic vegetation.

During the non-flood year I saw a lack of fluvial dependent taxa in all sites, which was more pronounced in the riverine habitat than the backwaters. Moreover, macrohabitat generalists tended to have higher densities during the non-flood year, apparently capitalizing on the stable water levels and low flows. Similarly, Brown and Coon (1994) showed a decreased number of taxa during a non-flood year, with lower densities of fluvial dependent species (i.e., goldeye *Hiodon alosoides*, buffalo *Ictiobus* spp., and carpsuckers *Carpiodes* spp.) and an increased abundance of macrohabitat generalists (i.e., centrarchids) in most tributaries. Changes in family composition between years emphasize the importance of a predictable flood pulse in larval fish ecology where rising waters cue spawning activity and permit access to floodplain habitat (Junk et al. 1989).

In accordance with other studies that have found lower or no reproduction in rivers during non-flood years (Brown and Coon 1994, Agostinho et al. 2004), I expected significantly lower densities and larval abundance estimates during 2005. However, seasonal abundance of macrohabitat generalists in LS was higher than the year before, and only MS abundance and peak densities were markedly lower. Moreover, the ILR, which functioned nearly exclusively as spawning habitat during 2004, as indicated by the larval lengths, had a larger size structure during 2005, suggesting these segments of the river provided relatively better nursery habitat for larvae during the non-flood year. Lower velocities during summer 2005, often below 0.1 m/s, transformed the ILR into habitat suitable for larvae. This compensatory pattern in larval abundance occurred due to certain species, mainly gizzard shad *Dorosoma cepedianum*, mosquitofish *Gambusia affinis*, brook silversides *Labidesthes sicculus*, and emerald shiners *Notropis atherinoides*, exploiting the low flow conditions, a concept dubbed the 'low flow recruitment hypothesis' (Humphries et al. 1999, King 2004). The hypothesis postulates that during low flow periods in the river, appropriately sized prey is concentrated, and under these conditions, some species spawn and can easily make the transition from endogenous to exogenous feeding, thereby having high recruitment (Humphries et al. 1999). I do not know whether recruitment differences occurred between years, but the significantly higher rotifer densities in LS during the low-flow period likely supported the increased rates of larval abundance (Aoyagui and Bonecker 2004).

Limited information exists on the interaction between larval fishes and zooplankton densities in large rivers. The positive correlations between plankton and larvae may be due to mutually favorable abiotic conditions, such as warm temperatures (Wetzel 2001). However, it has been speculated that biotic factors, such as top-down effects of larvae, may play an important role in riverine foodwebs (Jack and Thorp 2002). During the flood year, plankton and fish densities were positively related; ILR and LS zooplankton to fish densities in those sites, and MS larvae to rotifer densities. The positive relationship may be due to autochthonous inputs that increased seasonal abundance in backwaters during inundation, allowing for large pulses of larvae and plankton (Junk et al. 1989). The drastic summer decline in zooplankton densities may be due to increased foraging from larger larvae or simply summer declines often observed in large rivers (Gosselain et al. 1998). Whether temperature related or not, these patterns of summer declines remain unexplained (Gosselain et al. 1998). Although significant patterns were not found in either CP during 2004, or in LS and the ILR during 2005, total plankton densities increased with larval abundance and decreased after mid-summer, indicating the use of plankton as a prey source by larval fishes.

Promoting vegetation growth in backwaters through annual drawdowns may have positive effects for these restored systems. Vegetation could have provided food and cover for larvae, possibly promoting invertebrate populations in densely vegetated areas and affording protection from predation (Dewey et al. 1997, Flinn et al. 2005). However, fish survival in MS was likely low due to the drawdowns. High mortality often occurs through bird and fish predation (Crowder et al. 1997) and anoxic conditions in shallower waters. The true potential of MS for larval abundance may only be realized if the compartment is managed in a rotating fashion, whereby drawdowns would be conducted every few years to maintain vegetation growth and during other years, it would be continuously connected to the river.

Floodwater stability (gradual rise and fall) coupled with habitat heterogeneity in the backwaters may be important. The unrestored backwater, CP, had the largest larval sizes during 2004, which may have been due to more stable water levels where larvae were not stranded on lake banks and fish nests were not desiccated due to rapidly receding waters (Adams et al. 1999, Brown and Coon 1994). In many species, larval survival and eventual recruitment of the adult population has been show to directly relate to larval (Miller et al. 1988). Progressively smaller larvae were collected in MS and LS, a pattern which corresponds to the degree of emergent structure available in each site, with CP having the most emergent vegetation and LS having none.

To enhance the complex biotic and abiotic interactions so valued in riverine-floodplain habitats, restoration projects must have a built-in flexibility to either create or sustain different habitat types. This approach will likely be most beneficial to all life history strategies by offering a wide variety of habitat characteristics to meet specific early life history environmental and habitat requirements (Grift et al. 2003). Restoration projects similar to the Swan Lake HREP offer the added benefit of flexibility. The compartmentalization allows managers the option of rotating management regimes between the two lower compartments to promote habitat diversity while maintaining river connectivity. One or both compartments may be left open to the river at any one time, ensuring backwater access to riverine fishes. Given the widespread benefits of these management practices and the move towards system-wide management programs on large rivers (Theiling 1995, Flinn et al. 2005), current site-specific restoration projects should be constructed with the flexibility to adapt to other management regimes.

CHAPTER 3

LATERAL EXCHANGE OF LARVAL FISH BETWEEN A RESTORED BACKWATER AND A LARGE RIVER IN THE EAST-CENTRAL U.S.

ABSTRACT

The lateral exchange of larval fish between a river's main channel and its floodplain waters may be compromised by the widespread degradation of backwaters. During 2004 and 2005, drift nets were set bi-directionally within a constructed channel between the Illinois River and an adjacent, restored backwater, Swan Lake, to investigate movement between these sites on a diel and seasonal basis. Larval density and composition data within the river and backwater were also collected. Drift was positively related with water velocity during 2004, and an estimated 32.3 million larvae drifted into Swan Lake that season. No other relationships with drift direction and abiotic variables occurred. Lateral drift patterns were strongly related to the flood pulse during 2004, but in the absence of a flood, as during 2005, the ambient biotic assemblage influenced drift timing, magnitude, and composition. Swan Lake's restoration appears to have successfully altered the backwater for multiple-use management while maintaining river connectivity and allowing exchange between the backwater and river to occur.

INTRODUCTION

Propensity to drift in streams and rivers is an adaptive response in aquatic organisms (Eckblad et al. 1984, Kennedy and Vinyard 1997). In fishes, drift is usually restricted to early life stages, such as seasonal larval fish drift in lotic systems and largely influences spatiotemporal patterns in larval density (Holland 1986). Drift in riverine larval fishes is typically downstream, but can also be lateral, where larval fishes may be exchanged between river and slackwater habitats (Humphries et al. 1999). This exchange has been hypothesized due to ichthyoplankton (hereafter termed larvae) density differences among slackwater and river habitats (Sheaffer and Nickum 1986, Brown and Coon 1994, King 2004) and due to downstream larval drift from tributaries to main channel areas (Eckblad et al. 1984, Muth and Schmulbach 1984). However, the extent of lateral larval drift into and out of backwaters relative to downstream drift in rivers is currently undocumented and would be useful to determine origins and destinations of larval fishes.

I defined lateral drift as the movement of organisms between a river's main channel and its adjacent floodplain waters. When connected to the river, these backwaters provide a lentic-lotic gradient along which fish and other aquatic organisms find spawning grounds, structural refuge, food resources, and overwinter habitat (Junk et al. 1989). Slackwater areas have been recognized as a critical component in the early life history of fishes (Holland 1986). These areas may be a productive source of age-0 fishes given the higher larval densities downstream of backwater outflow (Sheaffer and Nickum 1986) and large migrations of juveniles to the main channel from backwaters (Molls 1999). However, the contribution of river larvae to backwaters may also be significant and a vital process bringing riverine spawned larvae to productive nurseries (de Graaf et al. 1999). Due to the nature of these backwater-river confluences, with periods of inflow to the backwater, outflow to the river, and stagnant waters following high water events, larval exchange is likely complex (Brown and Coon 1994). These flow patterns at the confluence are apt to drive the ability of backwaters to function as nursery habitat for larval and juvenile fishes, potentially affecting the diversity and abundance of fishes (Brown and Coon 1994).

Most investigators have focused on patterns of downstream drift in rivers (Gale and Mohr 1978, Muth and Schmulbach 1984, Johnston et al. 1995), but not in the context of how timing and behavior may shape trends in lateral exchange. Fish spawning behavior dictates temporal patterns in larval drift, where drift duration increases with the duration of spawning (Reichard et al. 2001), and rising water temperatures and spring flood events that cue spawning often result in peak larval drift densities (Carter et al. 1986, de Graaf et al.1999, Auer and Baker 2002). Because feeding may influence the presence of absence of larvae in the drift, larval metamorphosis into exogenously feeding fish can either initiate or cease drift behavior (Carter et al. 1986, Dudley and Platania 2000), or have no influence on the species' presence or absence in the drift (Auer and Baker 2002), depending on the species. Downstream drift probably allows access to nursery habitats with amenable growing conditions, and scatters the cohort, which may reduce competition for food and space as well as reduce conspicuousness to predators (Bardonnet 2001). Thus, the interplay of abiotic and biotic factors determines the timing, duration, and taxa of fish drifting, but their influence on lateral exchange is unknown.

Larvae have diurnal, nocturnal, or crepuscular drift patterns that depend on species developmental stage (Gale and Mohr 1978, Muth and Schmulbach 1984, Gadomski and Barfoot 1998). These photokinetic responses may change in some species as larvae age (Bulkowski and Meade 1983), possibly a result of ontogenetic changes in diet or susceptibility to or avoidance of predation. However, diel patterns in drift are contradictory, where fish species of the same life stage exhibit significant nocturnal drift in some water bodies and diurnal drift in others (Muth and Schmulbach 1984, Smith and Brown 2002).

Therefore, management decisions to improve nursery and spawning habitats and survival of early life stages of fish may be ineffective due to gaps in our understanding of the timing and behavior of lateral larval drift. All riverine fishes exhibit certain life history strategies based on their relative dependence on flowing water. Some are specialized for riverine environments and adapted for downstream drift, others require flowing water for a portion of their life history where habitat changes likely entail lateral movement; and finally generalists reside mostly in lentic areas and should be less prone to enter the drift. I investigated the interplay of life history strategies with lateral drift dynamics on a diel and seasonal basis within a restored system, which will assess the success of the restoration and the backwater's use as a productive fish nursery. Quantifying lateral movement will enable researchers to determine the role of backwaters as larval fish nurseries in large rivers and to address the influence of habitat alterations on larval fish assemblages among backwater lakes and river reaches. I hypothesized lateral movement at the restored connection between Swan Lake and the lower Illinois River to be influenced by abiotic factors that cue spawning, such as temperature and river stage, and that drift composition would be linked to life history strategy, where taxa requiring flowing water would comprise a majority of the drift.

METHODS

STUDY AREA

Swan Lake, a 1,100-ha Illinois River (ILR) backwater located between river kilometer 8 and 21, is vitally important for fishes of the Illinois and Mississippi Rivers (USACE 1991). A Habitat Restoration and Enhancement Project (HREP) through the federal Environmental Management Program (EMP) was initiated in the 1980s. One goal of the Swan Lake HREP was to improve spawning and nursery habitat for fishes by improving river connectivity of the backwater and increasing habitat heterogeneity (USACE 1991). The downstream portion of the backwater (Lower Swan [LS], 567 ha) was managed to be continuously connected to the ILR through a water control structure. Historically, Swan Lake was connected to the ILR through a 0.5-km wide opening. Restoration of the backwater complex restricted the river connection to the width of a stop-log water control structure, about 5-m wide, which was nested within a rip-rap lined channel (c. 50 m long x 12 m wide). During normal pool stage, the stop-log structure at LS is the only avenue through which larval fish movement between the backwater and river occurs, making the stop-log structure the focal point of this study site.

FISH SAMPLING

To quantify bidirectional larval drift between LS and the ILR, three conical drift nets (0.5 m x 2 m, 500-µm mesh) were attached to a floating, anchored PVC frame and fished during late March through July 2004, and March through August 2005 (Figure 9). I sampled larvae for 15 minutes at the surface (approximately one-third channel depth) every two weeks on the lake-side of the LS stop-log structure (Figure 9). Two directional net sets, one sampling larvae potentially moving into LS and one set sampling larvae leaving LS, were conducted at dawn, mid-day, dusk, and mid-night within 36 hours. During flood events, sampling frequency was increased to weekly, but fewer night sets occurred.

At each sampling time (e.g., dawn, mid-day, dusk, mid-night), surface water temperature (°C) and dissolved oxygen (mg/L [YSI Model 52 Dissolved Oxygen Meter; Yellow Springs Instruments, Yellow Springs, Ohio, USA]), secchi depth (cm), surface water velocity (cm/s [Flo-Mate Model 2000, Marsh McBirney, Inc., Frederick, Maryland, USA]), and average wind speed and direction (km/h [Kestrel 1000, NFS – Radiation Protection Systems, Inc., Groton, Connecticut, USA]) were quantified. A Doppler bi-directional velocity meter (Model 6526-51 Starflow; Unidata America, Lake Oswega, Oregon, USA), anchored to the bottom of the water control structure, recorded temperature (°C), depth (mm), and mean velocity (mm/s) twice an hour. River stage data from Hardin, Illinois (river kilometer 34.4) were obtained from the U.S. Army Corps of Engineers. No Swan Lake depth data were collected during September through November 2004, and no velocity data were collected during October 2004 through April 2005.

Upon completion of each drift net set, contents were flushed into the cod end and preserved in 95% ethanol. Each sample was split to approximately 200 fish using a Folsom plankton splitter (Aquatic Research Instruments, Hope, Idaho, USA). All age-0 fish in the subsample were counted, identified to the lowest possible taxon, typically genus, and classified to a developmental stage (i.e., yolk-sac, larval, juvenile) using descriptions and keys in Soin and Sukhanova (1972), Auer (1982), Murty et al. (1986), Tweb et al. (1990) and voucher specimens from Southern Illinois University's Fluid Vertebrate Collection (B. Burr, Carbondale, Illinois, USA) and Colorado State University's Larval Fish Laboratory (D. Snyder, Fort Collins, Colorado, USA). A subsample of fish from each taxon and stage identified was measured (total length [TL]; mm) using Scion Image® software, which was calibrated to 0.5 mm, or metric calipers (N = 10 per net).

Larval samples also were collected in LS and the ILR to characterize the taxa and abundance within each site and compare larval densities to drift composition at the water control structure. I used paired, bow-mounted ichthyoplankton nets (0.5 m diameter x 2 m long, 500-μm mesh) to sample these habitats on the same dates as drift nets were set. Four stratified transects randomly chosen within LS and two in the ILR (river kilometer 7.0 to 9.0) were sampled weekly. I stratified backwater transects into two inshore and two offshore tows, while tows in the ILR were conducted within 1 km up- and downstream of the LS-ILR confluence.

At each transect, nets were towed at the surface for 5 minutes at a constant speed, with a calibrated mechanical flow meter (Model 2030R, General Oceanics, Inc., Miami, Florida, USA) mounted in the mouth of one net to standardize sampling effort (i.e., volume sampled). Inshore backwater tows followed the shoreline; offshore backwater tows were straight transects; and river tows were straight transects taken perpendicular to flow direction. River tows started at the main channel border and continued to the main channel border on the opposite side of the river. If 5 minutes had not passed during river transects, the direction was reversed with nets still in the water, and sampling continued until 5 minutes was reached. All tow samples were preserved, processed, and identified as with drift net samples.

LIFE HISTORY CLASSIFICATION

The collected fish were grouped by family into one of three generalized classes to better explain trends observed between years and treatment groups (e.g., Galat and Zweimuller 2001). The classes were as follows: fluvial specialists, fluvial dependent, and macrohabitat generalists (Table 1). Fluvial specialists are fish that inhabit streams and rivers throughout their entire life and rarely enter floodplain habitats (Galat and Zweimuller 2001). Fluvial dependent species regularly use lentic backwater or reservoir habitats, but certain life history traits depend on lotic environments (Galat and Zweimuller 2001). These species are typically broadcast spawners, where developing eggs and larvae are semi-boyant and passively drift in wind-induced or downstream currents (Holland 1986). Adult fluvial dependent fishes also may make lateral migrations into slow-flowing lentic areas to spawn—activities likely corresponding with increasing temperatures and rising water levels (Junk et al. 1989). Macrohabitat generalists include species commonly found in reservoirs and off-channel habitats that do not depend on lotic systems (Galat and Zweimuller 2001). When these fishes use the river, it is either as a corridor to move among backwaters (Junk et al. 1989, Dettmers et al. 2001). Spawning usually occurs in off-channel habitats and offspring generally do not leave this habitat until the juvenile stage (Holland 1986). I grouped families based on Galat and

Zweimuller (2001; taxonomy from Nelson [1994]), and the only deviation from their groupings was Sciaenidae, which I classified as fluvial dependent (Dettmers et al. 2001, Koel and Sparks 2002).

DATA ANALYSIS

Larval tow data at each site were standardized as fish per m^3. Larval drift represented larval exchange between the backwater and river and was calculated as number of fish per minute. Some fish were caught in nets set opposite to the direction of flowing water when velocities were 0.1 m/s or greater (e.g., frame positioned to catch fish drifting out of LS while water flowed into LS at 0.11 m/s; Scheidegger and Bain 1995). This was not considered drift and thus was removed from data sets before analyzing. Total larval abundance from tow data was calculated as the sum of weekly densities during each sampling season. All data were $\log_{10}(x+1)$ transformed to meet assumptions of normality.

Two-way repeated measures ANOVA (proc MIXED, SAS Institute 1999) was used to test for differences among treatments sampled over time (Hurlbert 1984): (1) bi-weekly drift catch rates into and out of the backwater, with catch rates as the response variable and week and direction as predictors, (2) bi-weekly drift catch rates at dawn, day, dusk, and night, where week and time of day are predictor variables, and (3) weekly mean densities of tow data between sites were compared, with density as a response variable and week and site as predictors.

I estimated total number of larvae drifting for each season and determined overall trends within and between years for larval densities. Non-zero catch rates of drift data for each direction and year were regressed (Type I regression, proc REG, SAS Institute 1999, Sokal and Rohlf 1995) as a function of environmental parameters collected from the Doppler device corresponding to the time and date of drift (e.g., depth, temperature, velocity). For significant relationships, these environmental data were used to estimate the total amount of larval drift for each season. A one-way ANOVA design tested how larval density and catch rates differed between years. To control for experimentwise error rates, I used Tukey-Kramer post-hoc (Sokal and Rohlf 1995). Relationships between larval densities and lateral drift were examined using linear regression on untransformed data.

Proportions of fish per 1-mm total length group were used to compare sizes of fish between tow sites, drift direction, year, and among diel time points. Kolmogorov-Smirnov tests were used to compare (1) size distribution of fish drifting into versus out of LS in each year, (2) total size structure (into and out combined) of fish drifting during 2004 and 2005, (3) size structure of fish collected from tows, and (4) between-year differences for LS and ILR tow data. Multiple pairwise comparisons were used to examine size structure differences among the times of day sampled (e.g., dawn vs. night, dawn vs. day, dawn vs. dusk, etc.), with Bonferroni adjusted α-values ($\alpha'' = 0.05/6$).

RESULTS

ABIOTIC PATTERNS

A moderate flood pulse occurred in 2004, during which water levels in the lower Illinois River were above flood stage for approximately five weeks during June (Figure

10). In contrast, water levels remained at or below normal pool level (i.e., 128 m) during the entire 2005 sampling season. Water temperatures in the river gradually rose and fell during 2004, peaking in late July at 28 °C (Figure 10). However, during 2005, water temperatures were more variable and rose over 30 °C twice. In general, water temperatures in LS varied in a manner similar to those in the ILR, where the amplitude and daily variation was larger because of the shallower water depths. Lower Swan depth fluctuations were slightly dampened compared to the river (Figure 10).

Movement of water through the LS water control structure was bi-directional, often changing direction multiple times a day likely due to river boat traffic and wind, and velocities varied greatly. Aside from the 2004 flood pulse which increased water depth in the channel to 3.98 m, channel depth remained above 2 m for most of 2004 and averaged 2.04 m after May 2005 (Figure 11). The bi-directional movement of water into and out of Swan Lake typically occurred daily, but on average water was moving into LS, with 86% of differential velocity values being positive (Figure 11). Mean channel velocities were typically less than 1 m/s during the two sampling seasons, and only three percent of data points equaled or exceeded this amount. In 2004, velocities flowing into LS peaked at 2.2 m/s during the rise in flood waters, dropped to near zero at flood crest, and flowed out at peak velocities of 1.5 m/s during the rapid recession of flood waters. Near-zero velocities occurred during low water periods, particularly those occurring during summer 2005, where differential velocity values were clustered around zero (Figure 11).

SEASONAL PATTERNS OF LARVAL FISHES

Larval drift patterns varied between years, while larval fish densities and timing in LS and the ILR were similar between years. Fish recruited to drift nets during 1 May through mid-July 2004, with a major pulse of fish drifting into LS during late May 2004, timed with the rising flood waters (Figure 12). During 2005, a smaller, discrete peak of larvae was exchanged between LS and the ILR; three weeks later and at less than half the magnitude of catch rates (Figure 12). Larval fish appeared (garnered from tow data) in the ILR during mid-April and during late-March 2004 in LS, and there was a synchronous pulse of larvae in tows during June 2004, the period of floodwater inundation (Figure 13). During 2005, larval fish appeared in both LS and the ILR during mid-April, and the larval pulses were asynchronous, peaking two weeks later in the ILR than in LS (Figure 13).

Higher rates (#/minute) of fish drifted into and not out of LS during 2004, although differences over time and an interaction between drift direction and time occurred (Figure 12; all $P < 0.01$; direction: $F_{1,6} = 18.53$; week: $F_{11,60} = 7.67$; direction*week: $F_{11,60} = 8.2$). Ingress and egress of ichthyoplankton were similar in 2005 (Figure 12; $F_{1,6} = 3.8$, $P = 0.10$). However, larval densities in LS were 10 times higher than the ILR during both years (Figure 13; two-way repeated measures ANOVA: $P < 0.01$; 2004: $F_{1,6} = 66.97$; 2005: $F_{1,6} = 109.78$), although they differed over time ($P < 0.01$; 2004: $F_{21,91} = 23.27$; 2005: $F_{22,76} = 15.11$) and also interacted ($P < 0.01$; 2004: $F_{20,91} = 4.43$; 2005: $F_{18,76} = 3.74$). Total annual larval abundance estimates were similar between years in the ILR, 42 and 31 larvae/m3, and LS, 435 and 531 larvae/m3, during 2004 and 2005, respectively. Mean daily drift rates of larvae by direction did not differ between years (two-way ANOVA: $F_{3,132} = 1.6$, $P = 0.21$) despite an almost eight-fold difference in rates of larvae drifting into Swan Lake during 2004 and 2005, with means of 4.2 fish/minute and 0.5 fish/minute, respectively. The non-significant between-year result was due to high variance within year and similar rates of larvae drifting out of the backwater between years, at 0.4 fish/minute each year (Figure 12).

Both LS and the ILR had homogeneous larval distributions, with no differences in LS between inshore and offshore (2004: $t_{1,19} = -2.08$, $P = 0.05$; 2005: $t_{1,18} = -0.66$, $P = 0.52$) or in the ILR between upstream and downstream stratified transects (2004: $t_{1,21} = 0.94$, $P = 0.36$; 2005: $t_{1,22} = -0.27$, $P = 0.79$), though there was a propensity for higher densities to occur downstream of LS during 2004.

No diel patterns in catch rates occurred during 2004 (Table 9; $F_{3,4} = 0.1$, $P > 0.05$), though a diel drift pattern occurred during 2005, where more fish larvae drifted at night than during other times of the day ($F_{3,4} = 15.9$, $P = 0.01$).

Grouping families by life history class revealed that drift composition varied between years (Table 10, Figure 14). Fluvial dependents were most abundant drifting into Lower Swan during 2004, where sciaenids and cyprinids represented 60% of the fish exchange (Table 10, Figure 14). Macrohabitat generalists were collected in the drift during 2004 and 2005 without much directional difference, making these taxa some of the only that drifted out of LS (Figure 14).

Temporal drift patterns varied by life history class. Drift of fluvial dependent taxa was initiated and peaked in a discrete pulse during rising flood waters. Macrohabitat generalists, comprised mostly of clupeids, also drifted in peak rates during the rising flood waters, beginning in May before the pulse and continuing for two months. During

2005, neither timing nor peak drift between fluvial dependents and macrohabitat generalists varied.

Macrohabitat generalists dominated the ambient larval composition during both years in tows conducted in LS, but not the ILR. During 2004, fluvial dependent taxa dominated the composition in the ILR, and were five times higher than densities of those taxa in LS. Regardless of year, fluvial dependent taxa were relatively absent in LS compared to the densities of macrohabitat generalists (Table 10, Figure 14). As with drift rates during the non-flood year, densities of fluvial dependents were lower in the ILR (Table 10, Figure 14). No fluvial specialists were sampled during either year. There were no differences in family composition in tows conducted upstream or downstream of LS.

Drift rates at LS were related to abiotic and biotic factors. Non-zero catch rates of fish drifting into the backwater during 2004 was positively related to velocity, but not temperature or channel depth (multiple regression model: adj. $r^2 = 0.92$, $F_{3,13} = 61.83$, $P < 0.01$; Temperature: $t = 0.64$, $P = 0.53$; Velocity: $t = 13.19$, $P < 0.01$; Depth: $t = 1.23$, $P = 0.24$). Using the regression model and continuous velocity data, I extrapolated catch rates to the seasonal duration of drift (i.e., 1 May through 15 July) and channel volume. An estimated 32.3 million fish drifted into LS during the 2004 sampling season (mean catch = [33.86 * velocity] – 2.91). Drift catch rates into LS were weakly but positively correlated with ILR tow-derived densities during 2004 ($P = 0.02$, $r^2 = 0.36$). During 2005, although drift was unrelated to abiotic variables, it was related to ambient larval densities. Drift into LS was positively related to ILR larval densities ($P < 0.01$, $r^2 = 0.78$), while LS larval tow-derived densities likely influenced catch rates of drift to the river ($P < 0.01$, $r^2 = 0.94$). Therefore, abiotic factors influenced 2004 larval drift and density affected 2005 drift.

SIZE STRUCTURE OF LARVAL FISHES

Larval fish sizes differed between years and among sites. Larger fish drifted out of LS during 2004, while during 2005, larger fish drifted into LS (Table 11, Figure 15). However, there were no differences in the size distribution of larvae caught in tows upstream and downstream of LS in either year. Higher river and channel velocities during the flood pulse (2004) did not entrain larger fish into LS, and, regardless of net direction (i.e., in versus out), larger fish occurred in drift nets during 2005 (KSa = 8.01, $P < 0.01$). Despite this change in size structure, larval sizes remained larger in the backwater than the ILR regardless of year (Table 11, Figure 15). Furthermore, both backwater and river yielded larger larval sizes during 2005 (LS: KSa = 2.53, $P < 0.01$; ILR: KSa = 10.55, $P < 0.01$) than 2004.

Lengths of fish drifting at the LS water control structure also varied among times of day sampled (i.e., dawn, day, dusk, night). Multiple pair-wise Kolmogorov-Smirnov comparisons revealed a propensity for larger fish to drift at night than at other times (Table 11). In 2004, fish were similarly sized during dawn, day, and dusk (Table 11, Figure 16). Stronger diel patterns occurred during 2005, with night catch collecting the largest fish and dusk, dawn, and day catches each sampling progressively smaller fish (Table 11). Regardless of year, ichthyoplankton drifting at dawn, day, and dusk were predominantly less than 8 mm TL, with cumulative percent frequencies between 74 - 80% for each time period in 2004, and between 56 – 82% in 2005. Almost 70% of larvae caught at night were 8 mm or greater (Figure 16). Therefore, during 2004 and 2005, larger fish drifted at night between LS and the ILR.

DISCUSSION

Seasonal lateral drift occurred at the restored connection between LS and the lower ILR, with patterns shaped by the annual flood pulse and the fishes' life history strategies. The flood pulse concept specifies the importance of coupling increasing water levels and temperatures to cue spawning and yield high recruitment of riverine fishes (Junk et al. 1989, Harvey 1987). In this study, fish abundance in LS was similar throughout two physically contrasting seasons, a flood and a non-flood year. Fish abundance estimates for the ILR were also similar between years. Yet these abiotic conditions induced a change in ambient family densities (i.e., life history classes) between 2004 and 2005, which may have influenced between-year variation in magnitude, direction, and composition of lateral larval exchange. During 2004, the large pulse of larvae drifting into LS occurred with the rising flood waters, and exchange between the river and backwater was temporally isolated to the flood event. Similarly, peak richness and lateral drift was synchronized, but not correlated, with peak densities in the river and backwater during rising flood waters. In contrast, drift had no net directionality during 2005, and peak drift rates were less than half that of 2004. Only during 2005 were ambient larval densities positively correlated to drift rates, demonstrating a strong biotic influence in the absence of the spring flood.

The change in life history classes also conveys the significance of abiotic cues to initiate and influence larval drift. The propensity for fluvial dependents to drift into backwaters during the flood year was likely related to the coupling of temperature and flooding, while the relative absence of these species in the drift and the larval assemblage was related to their decoupling during the non-flood year. A lack of a spring flood pulse, as seen during 2005, may have resulted in poor reproductive or recruitment of fluvial dependents, where many of those taxa either did not reproduce or their eggs did not hatch or develop successfully (Humphries and Lake 2000). Some between-year differences in life history classes may be related to the strong reliance of fluvial dependents on annual flood pulses.

Aside from the influence of abiotic cues on floodplain habitat use by fluvial dependents, the ecological role of the restored LS, as it contributes to fish reproduction in the river-floodplain, needs to be identified and evaluated. In other large river systems, slackwater areas contribute larvae and juveniles to the river such that densities downstream of the backwater-river confluence become higher (Muth and Schmulbach 1984, Sheaffer and Nickum 1986). However, I did not find any difference in larval fish densities in the river upstream or downstream of the backwater. Substantial movement between the backwater and river occurred, even though Swan Lake seemed to neither function as a major sink nor source of larvae. Despite this, I believe LS to be a vital component of the lower ILR, with potential benefits to the Mississippi River due to its close proximity (USACE 1991).

The low drift rates of macrohabitat generalists denote a behavioral component to lateral drift, thereby discounting suspicions of LS acting solely as a sink. The dominant taxa in the larval assemblage were found in the drift (Jurajda 1995, Reichard et al. 2001); yet clupeids were less abundant in the 2004 drift than their ambient densities would have warranted. During 2005, macrohabitat generalists drifted in lower rates despite their higher densities within the river and LS. This under-representation of macrohabitat generalists in the drift may be indicative of drift avoidance (Brown and Armstrong 1985, Reichard et al. 2001). Ultimately, these findings imply a purposeful lateral drift pattern, where some families avoid or are less prone to drift.

Additional behavioral components of larval drift were reflected by diel and size structure patterns. Nocturnal lateral movement of larger individuals occurred during both years, while higher catch rates at night only occurred during 2005. Larvae, particularly larger fish, may innately drift at the bottom during the day and move to the surface at night to feed or evade predation (Gale and Mohr 1978, Muth and Schmulbach 1984, Johnston et al. 1995). Carter et al. (1986) captured larvae drifting in densities in the Colorado River almost four times higher at night than during the day. Other abiotic factors, such as water clarity, likely drove interannual diel variations, where catch rates showed no diel patterns during the flood year and were more apparent during low flow when water transparency was likely higher (Reichard et al. 2001, Araujo-Lima et al. 2001). Therefore, the diel patterns quantified in LS probably resulted from phototaxic responses and changes in water clarity between years.

Larger fish drifted out of LS during 2004 than 2005, which supports conventional larval drift hypotheses, where backwaters function as nursery habitat, provide ideal conditions for growth, and later become a source for age-0 fish (Sheaffer and Nickum 1986). Lateral drift during 2005 was not necessary as a life history strategy given that the ILR offered amenable habitat similar to backwaters, with slow flows, warm habitat, and high plankton densities (see Chapter 3). Furthermore, additional research on the

Swan Lake backwater complex showed that a large abundance of juvenile clupeids emigrated from LS into the ILR during summer 2004, and juvenile sciaenid and moronid fish during fall 2004 (Schultz 2006). Thus, it is my thought that larvae entering the backwater or spawned in the backwater were able to feed and grow throughout the season, eventually making an ontogenetic habitat shift by exiting the backwater sometime that fall and recruiting to the river fishery (King 2004).

In restored systems, continuous connectivity should be maintained where possible, as species use the floodplain throughout the spring and summer. Limiting backwater access or reducing river access could impair the recruitment potential of certain species, such as fluvial dependents, eventually leading to a less diverse riverine fish community (Turner et al. 1994). Historically, the LS connection to the ILR was about 99% wider than its post-restoration state, and likely permitted gradual changes in water level, direction, and velocity. Currently, water levels lag behind the river, and typically flow directions and velocities change multiple times per day. Velocities are most likely higher through the restricted channel, especially during flood pulses. It is possible that the narrowed connection has negatively altered lateral movement of fishes through the stop-log structure, and that changes in water patterns since the stop-log structure construction have altered exchange patterns.

Although I have no larval exchange information from natural systems, the restricted channel connection between LS and the ILR did not seem to hinder fish movement (Schultz 2006). Instead, one effect of the narrowed channel connection between LS and the ILR on larval fishes may be to concentrate larvae as they are transported; however it is unknown whether this concentration of fishes would have a compensatory or depensatory effect on survival, if any. Few studies have evaluated restoration techniques and their influence on fish early life stages. A marine study demonstrated the efficacy of artificial channels in permitting larval exchange between estuarine and ocean habitats, showing a similar number of species and concentration between an artificial and natural estuary-ocean channel, despite a large difference in the topography and volume of water passing through the channels (Young and Potter 2003). Further research efforts should focus on the effects of interannual variability on drift patterns (Johnston et al. 1995), as well as investigate exchange between natural backwaters and riverine habitat. Although potentially tedious, studies investigating lateral movement of ichthyoplankton to backwaters or tributaries should be conducted to improve energy exchange estimates of river-floodplain systems and enable us to estimate contribution and productivity of backwater systems to the river.

This study was unique, as no other study has quantified lateral larval drift in a river-floodplain system. Lateral drift is an important component of fishes' life history in lotic systems, but these strategies and ontogenetic habitat changes may not withstand the anthropogenic disturbances in our streams and rivers, such as levees and impoundments. Altered hydrology may affect larval drift ecology and influence recruitment of fishes. This study has shown that lateral drift patterns may be related to the flood pulse, but in the absence of a flood, the ambient biotic assemblage could influence drift timing, magnitude, and composition. Generally, Swan Lake's restoration appears to have successfully altered the backwater for multiple-use management while maintaining river connectivity and allowing exchange between the backwater and river to occur. Ultimately, future conservation efforts aimed at restoring hydrology should not focus on one particular habitat, but should equally consider main channel, floodplain, and tributary habitats (Galat and Zweimuller 2001).

CHAPTER 4

MANAGEMENT IMPLICATIONS

I completed two objectives in this study: (1) to assess the benefit of a restored backwater system relative to an unrestored, manipulated backwater by quantifying the response of larval fish communities, and (2) to investigate the interplay of life history strategies with lateral drift dynamics on a diel and seasonal basis within a restored backwater system. The following evaluations were made on tested hypotheses:

i) Backwater sites, whether restored or unrestored, were used as spawning and nursery habitats in higher densities during both years than river segments sampled. During 2004, the Illinois River (ILR) was used almost exclusively as spawning habitat, based on the larval size structures. However, during 2005, the non-flood year, size structures in the ILR indicated those river segments were being used as both spawning and nursery habitat.

ii) Seasonal abundances of larval fishes varied among sites, but this variation was not solely due to backwater-river connectivity. Middle Swan (MS) experienced the highest larval fish densities during both years, but had limited river connectivity. I believe high larval fish densities can be attributed to the annual drawdowns that promoted aquatic vegetation as well as the brief periods of connectivity.

iii) Family composition, described by the life history classification, varied among sites and between years. During 2004, the ILR and

MS had high densities of fluvial dependent larval fishes compared to Lower Swan (LS) and Calhoun Point (CP), which were dominated by macrohabitat generalists. Despite LS having the highest river connectivity, macrohabitat generalists dominated the assemblage during both years. Fluvial dependent taxa were relatively absent in all sites during 2005, the non-flood year.

iv) Larval exchange occurred between the restored backwater, LS, and the lower ILR. However, the influx of larvae into the backwater occurred only during 2004 and was more related to velocity than to water temperatures or river stage. Although drift rates were not different between years, timing and composition seemed more related to ambient larval densities in the ILR and LS than abiotic conditions (i.e., water temperature, water depth, velocity).

v) Although Swan Lake is a major backwater of the lower ILR, density differences between river segments upstream and downstream of the backwater-river confluence did not occur. It is unclear whether restriction of the LS-ILR confluence affected the magnitude of larval movement between the river and backwater. It may have reduced larval movement or may have concentrated larvae moving through the connection.

vi) Diel periodicity in larval drift patterns occurred at the LS-ILR confluence. During both years, larval fish drifting at night were larger than those drifting at other times of day. During the non-flood year, larval fishes also drifted in higher rates at night.

The results from this study were intended to provide researchers and managers with information on the role restored backwaters play in the life history strategies of larval fishes and to help guide the development of future backwater management and restoration programs.

In restored systems, continuous connectivity should be maintained where possible, as species use the floodplain throughout the spring and summer. Limiting backwater access could impair the survival of some families, such as fluvial dependents that require multiple habitat types to complete their life history, eventually leading to a less diverse river fish community. Connectivity in the Swan Lake HREP was preserved through the construction of water control stop-log structures. These fixed structures either created new river connections or severely restricted original river confluences. The effect of the narrowed channels on larval movement between the river and backwaters cannot be assessed due to the lack of information on natural systems. Despite this, the Swan Lake HREP appears to have successfully restored the backwater while still maintaining its river connectivity.

In addition to reconnecting the floodplain to the river, natural flood pulses serve an important biological role in river ecosystems. The flood pulse concept specifies the importance of coupling increasing water levels with rising temperatures during spring to cue spawning and yield high recruitment of fishes. In this study, fish production in LS and the ILR was similar throughout two physically contrasting seasons, a flood and a non-flood year. These abiotic conditions induced a change in ambient family densities between 2004 and 2005, which may have influenced between-year variation in magnitude, direction and composition of lateral larval exchange. A lack of a spring flood pulse, as seen during 2004, may have reduced reproduction or recruitment of fluvial dependents.

Although river connectivity is an important attribute of backwaters and should not be overlooked, maintaining this connectivity might not be enough to ensure successful habitat restoration. Increasing the habitat heterogeneity through system-wide management regimes can positively influence larval fishes and other target organisms by directly impacting organic matter and macroinvertebrate and zooplankton densities (Flinn et al. 2005). Often, natural backwaters go through an annual succession from lotic to lentic conditions, thereby accommodating several life history strategies and promoting species richness (Aarts et al. 2004). Restored systems that are drawn down to consolidate sediments and promote vegetation growth also have greater water clarity and will likely be productive areas for aquatic vegetation and zooplankton (Flinn et al. 2005). By increasing food resources and offering protective refuge, river restoration schemes aimed at enhancing habitat diversity in off-channel areas can have similar beneficial effects by creating a stable, diverse, and abundant age-0 fish community (Pezold 1998, Langler and Smith 2001).

However, research conducted to establish the role of off-channel habitat in fish recruitment (Langler and Smith 2001) and to determine the function of connectivity in recruitment models is still necessary. Schultz (2006) showed that because LS was continuously connected to the ILR, mass emigrations of juvenile fishes could occur during the late summer and fall. Additionally, restoration projects should not be viewed as sustainable once completed. Natural floodplain succession is stifled by an immobilized main channel, so restored and unrestored backwater systems must be actively managed to prevent sedimentation from degrading the habitat (Lusk et al. 2003, Aarts et al. 2004).

Therefore, human restoration and management of floodplain habitat can potentially create valuable spawning and nursery areas for many fish species, and offer the necessary habitat diversity lost since the degradation of natural floodplain regions (Grift et al. 2003). These alterations can restore fragmented areas, providing macrohabitats in proximity, thereby meeting specific requirements of riverine species early life stages (e.g., fluvial specialists, fluvial dependents; Aarts et al. 2004). Also, the important interplay among hydrologic patterns, habitat quality and availability, and the ontogeny of larval fish in river-floodplain systems should be noted, where flood regimes influence family composition and production improves with habitat diversity. Our continued attention to river connectivity is paramount, but the importance of habitat heterogeneity in larval production and assemblage structure should not be neglected.

Table 1. Fish families commonly found in the Illinois River basin grouped into one of three life history classes (e.g., Nelson 1994, Dettmers et al. 2001, and Galat and Zweimuller 2001), fluvial specialists (FS), fluvial dependents (FD), and macrohabitat generalists (MG).

Family	Life History Class	Family	Life History Class
Acipenseridae	FS	Hiodontidae	FD
Amiidae	MG	Ictaluridae	MG
Anguillidae	MG	Lepisosteidae	MG
Aphredoderidae	MG	Moronidae	FD
Atherinidae	MG	Percidae	MG
Catostomidae	FD	Percopsidae	MG
Centrarchidae	MG	Petromyzontidae	FD
Clupeidae	MG	Poeciliidae	MG
Cyprinidae	FD	Polyodontidae	FD
Esocidae	MG	Sciaenidae	FD
Fundulidae	MG	Umbridae	MG
Gadidae	MG		

Table 2. Monthly means (± SE) for abiotic variables (temperature [°C], dissolved oxygen [DO, mg/L], water depth [m], secchi depth [cm], wind speed [km/h], channel velocity [m/s]) collected at sites during 2004 in the Illinois River system.

Site	Temp		DO		Depth		Secchi		Wind		Velocity	
Illinois River												
March-04	7.81	(0.56)	7.58	(0.98)	.	.	22.6	(0.6)	7.0	(2.2)	.	.
April-04	11.22	(1.10)	7.86	(0.24)	6.25	(1.75)	27.5	(5.5)	9.2	(1.3)	0.46	(0.07)
May-04	18.62	(1.07)	5.03	(0.70)	6.07	(0.54)	25.2	(4.2)	6.4	(1.3)	0.54	(0.15)
June-04	23.28	(0.44)	4.49	(0.80)	7.89	(0.14)	23.6	(3.7)	8.3	(2.2)	0.63	(0.11)
July-04	25.91	(0.54)	3.61	(0.15)	6.62	(0.34)	30.3	(3.1)	7.0	(0.8)	0.55	(0.13)
August-04	26.87	(0.87)	3.85	(0.13)	6.44	(0.69)	29.1	(0.7)	8.4	(3.3)	0.19	(0.07)
September-04	23.77	.	8.18	(0.35)	6.81	(0.57)	25.6	(2.4)	7.7	(1.0)	0.08	.
Lower Swan												
March-04	13.90	(1.75)	7.26	(1.00)	0.87	(0.25)	15.3	(1.9)	13.6	(2.7)	.	.
April-04	16.42	(2.30)	8.02	(0.55)	0.77	(0.03)	23.2	(0.5)	9.0	(1.6)	.	.
May-04	22.03	(1.31)	6.31	(0.55)	1.01	(0.07)	19.5	(1.2)	8.2	(1.8)	.	.
June-04	25.05	(1.08)	8.61	(1.53)	2.09	(0.10)	32.7	(1.5)	6.8	(1.3)	.	.
July-04	27.58	(0.77)	6.30	(0.65)	0.96	(0.20)	22.7	(3.5)	4.7	(0.6)	.	.
August-04	24.02	(0.64)	3.78	(0.27)	0.70	(0.14)	14.3	(0.8)	6.3	(3.0)	.	.
September-04	20.24	(0.64)	5.08	(1.03)	0.77	(0.03)	14.1	(1.4)	6.4	(0.6)	.	.
Middle Swan												
March-04	.	.	7.64	(0.95)	1.33	(0.05)	25.6	(7.6)	4.0	(0.4)	.	.
April-04	14.40	(0.76)	9.53	(0.65)	1.26	(0.06)	27.8	(1.0)	10.0	(2.1)	.	.
May-04	22.29	(2.01)	6.50	(0.54)	1.13	(0.10)	24.1	(3.5)	7.8	(2.4)	.	.
June-04	25.18	(0.96)	8.08	(1.73)	2.03	(0.17)	34.5	(2.2)	9.6	(1.4)	.	.
July-04	28.41	(1.35)	7.46	(0.86)	1.25	(0.26)	33.6	(6.2)	7.6	(1.1)	.	.
Calhoun Point												
March-04	.	.	7.12	(0.42)	0.68	(0.33)	28.0	(1.0)	9.3	(5.4)	.	.
April-04	.	.	6.13	(0.05)	0.80	(0.00)	19.0	(1.0)	12.9	(5.7)	.	.
May-04	25.27	.	4.97	(1.27)	0.82	(0.08)	19.6	(0.6)	12.6	(0.3)	.	.
June-04	24.85	(1.80)	6.00	(3.31)	1.45	(0.20)	44.1	(18.4)	4.7	(3.5)	.	.
July-04	29.34	(2.44)	7.10	(0.96)	0.79	(0.11)	29.1	(3.6)	6.0	(1.9)	.	.
August-04	26.63	(0.82)	5.15	(0.70)	0.77	(0.03)	18.3	(0.8)	8.9	(2.3)	.	.
September-04	21.34	.	4.45	(0.05)	0.83	(0.02)	19.0	(1.0)	14.5	(0.6)	.	.

Table 3. Monthly means (± SE) for abiotic variables (temperature [°C], dissolved oxygen [DO, mg/L], water depth [m], secchi depth [cm], average wind speed [km/h], channel velocity [m/s]) collected at tow sites during 2005 in the Illinois River system.

Site	Temp		DO		Depth		Secchi		Wind		Velocity	
Illinois River												
March-05	6.57	.	.	.	5.25	(0.75)	27.0	(3.0)	14.5	(0.4)	0.44	.
April-05	15.16	(1.37)	6.66	(1.77)	6.91	(0.41)	28.9	(6.6)	9.9	(1.9)	0.50	(0.06)
May-05	18.68	(1.72)	9.66	(2.19)	6.51	(0.22)	18.1	(4.3)	10.1	(1.1)	0.37	(0.03)
June-05	26.64	(1.10)	9.69	(1.15)	6.36	(0.22)	25.6	(1.3)	5.8	(1.3)	0.16	(0.03)
July-05	29.60	(0.36)	7.04	(0.51)	6.00	(0.14)	24.9	(1.9)	4.6	(0.7)	0.15	(0.04)
August-05	28.44	(0.53)	6.78	(1.24)	6.50	(0.23)	25.9	(3.2)	6.1	(2.1)	0.12	(0.04)
September-05	26.65	.	8.94	(0.31)	6.13	(1.38)	25.1	(0.6)	3.4	(1.1)	0.15	.
Lower Swan												
March-05	8.86	(0.33)	.	.	0.78	(0.03)	12.5	(0.2)	7.7	(2.7)	.	.
April-05	16.15	(2.18)	8.31	(1.42)	2.11	(1.37)	15.3	(1.8)	13.2	(0.7)	.	.
May-05	20.94	(2.35)	9.25	(1.32)	0.75	(0.03)	14.5	(1.8)	8.9	(1.7)	.	.
June-05	27.09	(0.62)	10.02	(1.61)	0.73	(0.04)	14.4	(0.6)	5.0	(3.2)	.	.
July-05	29.47	(1.34)	9.54	(1.27)	0.68	(0.06)	14.5	(1.2)	7.1	(1.2)	.	.
August-05	27.53	(1.31)	5.76	(1.18)	0.73	(0.04)	13.5	(1.0)	5.6	(1.3)	.	.
September-05	26.62	.	12.72	(1.65)	0.70	(0.09)	13.3	(0.5)	4.5	(1.4)	.	.
Middle Swan												
March-05	7.74	.	.	.	1.08	(0.21)	23.3	(1.2)	9.1	(2.5)	.	.
April-05	16.81	(1.89)	8.31	(1.40)	0.95	(0.07)	20.9	(1.6)	12.3	(2.4)	.	.
May-05	21.45	(2.34)	9.25	(1.13)	0.89	(0.07)	21.5	(0.5)	8.6	(2.0)	.	.
June-05	27.54	(0.91)	7.45	(1.26)	0.97	(0.03)	19.1	(2.4)	7.6	(2.0)	.	.

Table 4. Site comparisons tested for overall differences in larval densities during 2004 and 2005 among Calhoun Point (CP), the Illinois River (ILR), Lower Swan (LS), and Middle Swan (MS), with adjusted *P*-values.

Site Comparison	*t*	df	*P*
2004			
CP vs. ILR	1.61	48.1	0.38
CP vs. LS	-1.50	48.3	0.45
CP vs. MS	-2.51	48.0	0.07
ILR vs. LS	-4.04	46.6	< 0.01
ILR vs. MS	-4.85	47.7	< 0.01
LS vs. MS	-1.32	47.7	0.55
2005			
ILR vs. LS	-4.43	35.3	< 0.01
ILR vs. MS	-7.11	36.6	< 0.01
LS vs. MS	-3.07	37.5	0.01

Table 5. Comparisons of stratified transects within each site for 2004 and 2005 in the lower Illinois River system, where backwater systems had inshore and offshore transects (Calhoun Point [CP], Lower Swan [LS], and Middle Swan [MS]), and the Illinois River (ILR) had upstream and downstream transects at CP and LS.

Within-Site Comparison	*t*	df	*P*
Inshore vs. offshore			
CP 2004	0.72	11.0	0.49
LS 2004	-2.08	18.9	0.05
LS 2005	-0.66	18.0	0.52
MS 2004	-1.96	13.8	0.07
MS 2005	-1.49	12.2	0.16
Upstream vs. downstream			
ILR at CP 2004	-0.04	10.0	0.97
ILR at LS 2004	0.94	22.0	0.36
ILR at LS 2005	-0.27	22.0	0.79

Table 6. Fish families grouped into one of three life history classes (e.g., Galat and Zweimuller 2001) with percent of total catch during 2004 and 2005 for the Illinois River (ILR), Lower Swan (LS), Middle Swan (MS), and Calhoun Point (CP).

	2004 % Catch				2005 % Catch	
Family	ILR	LS	MS	CP	ILR	LS
Fluvial Specialist	0.0	0.0	0.0	0.0	0.0	0.0
Fluvial Dependent						
Catostomidae	1.1	0.2	24.7	0.0	0.3	0.2
Cyprinidae	5.7	2.2	5.2	4.2	7.8	3.1
Hiodontidae	0.0	0.0	< 0.1	0.0	0.0	0.0
Moronidae	1.1	< 0.1	0.0	< 0.1	0.1	< 0.1
Sciaenidae	34.0	0.2	< 0.1	0.0	0.6	0.7
Macrohabitat Generalist						
Atherinidae	< 0.1	< 0.1	0.0	0.5	0.2	0.2
Centrarchidae	1.5	1.9	2.3	8.5	1.3	0.6
Clupeidae	56.4	95.4	67.5	86.8	89.6	94.9
Gasterosteidae	< 0.1	0.0	0.0	0.0	0.0	0.0
Ictaluridae	0.0	0.0	0.0	0.0	0.0	< 0.1
Lepisosteidae	< 0.1	< 0.1	0.2	< 0.1	0.0	0.0
Percidae	0.0	< 0.1	0.0	0.0	0.0	0.0
Poeciliidae	< 0.1	< 0.1	< 0.1	0.1	< 0.1	0.4
Total *N*	**8,086**	**48,390**	**105,906**	**11,500**	**4,476**	**56,033**

Table 7. Length analysis of larval fish caught in Lower Swan (LS), Middle Swan (MS), Calhoun Point (CP), and the Illinois River (ILR) using Kolmogorov-Smirnov pairwise comparisons, with results listed for sites or years that collected larger-sized fish. MS was drawn down before the end of the 2004 sampling season. Therefore, pairwise comparisons with this site were conducted on truncated data sets that included only samples taken on before 23 July 2004.

Site Comparison			KSa	P	Results
			α″-value = 0.008, Bonferroni adjusted		
2004					
LS	vs.	MS	4.67	< 0.001*	*MS*
LS	vs.	CP	3.49	< 0.001*	*CP*
LS	vs.	ILR	8.83	< 0.001*	*LS*
MS	vs.	CP	1.74	0.005*	*CP*
MS	vs.	ILR	12.79	< 0.001*	*MS*
CP	vs.	ILR	11.54	< 0.001*	*CP*
			α-value = 0.05, not adjusted		
2005					
LS	vs.	ILR	5.37	< 0.01*	*LS*
2004	***vs.***	***2005***			
LS			2.53	< 0.01*	*2005*
ILR			10.55	< 0.01*	*2005*

* Indicates significant differences between treatments, where $P \leq \alpha$ *or* α''.

Table 8. Mean zooplankton density (#/L) ±1 SE by taxa described as a percent of the total annual density for each site in 2004 and 2005.

	Cladoceran		Copepod		Nauplii		Rotifer	
Site	Density (SE)	%	Density (SE)	%	Density (SE)	%	Density (SE)	%
2004								
Illinois River	0.20 (0.08)	< 0.1	1.53 (0.45)	< 0.1	28.79 (7.38)	0.6	4868 (2753)	99.4
Lower Swan	0.15 (0.04)	< 0.1	1.14 (0.35)	0.1	35.00 (7.87)	1.8	1897 (365)	98.1
Middle Swan	1.10 (0.54)	0.1	8.47 (3.90)	0.4	105.54 (36.22)	4.8	2101 (542)	94.8
Calhoun Point	0.32 (0.09)	< 0.1	0.98 (0.38)	< 0.1	40.50 (13.00)	1.6	2443 (745)	98.3
2005								
Illinois River	0.87 (0.31)	0.1	0.69 (0.23)	< 0.1	11.04 (3.29)	0.6	1888 (241)	99.3
Lower Swan	0.31 (0.08)	< 0.1	1.74 (0.56)	0.1	61.97 (23.65)	1.6	3774 (618)	98.3

Table 9. Drift of fish during 2004 and 2005 in Swan Lake, Illinois River, with net sets averaged by time period (standard error represents ±1 of mean catch rate).

		N		Catch (#/minute)
Year	Time of day	Net sets	Fish	Mean ± SE
2004	Dawn (0450 – 0711)	24	1,956	1.80 ± 1.76
	Day (1215 – 1500)	24	2,551	2.38 ± 2.00
	Dusk (1815 – 2050)	22	1,137	0.96 ± 0.94
	Night (0000 – 0110)	20	112	0.12 ± 0.08
2005	Dawn (0450 – 0640)	24	490	0.45 ± 0.42
	Day (1145 – 1435)	23	77	0.07 ± 0.05
	Dusk (1815 – 2110)	26	70	0.06 ± 0.04
	Night (2340 – 0110)	20	869	0.93 ± 0.79

Note: Night net sets were not conducted during rising flood waters during 2004, which reduced net set count and likely mean night catch rate.

Table 10. Fish families grouped into one of three life history classes (Galat and Zweimuller 2001) with percent of total catch during 2004 and 2005 by gear. Drift net total catch for 2004: $N = 5{,}756$; for 2005: $N = 1{,}506$. Tow net total catch for 2004: $N = 56{,}476$; for 2005: $N = 60{,}509$. No fluvial specialists were caught in either drift or tow nets.

	2004		2005	
Family	Drift Nets	Tow Nets	Drift Nets	Tow Nets
Fluvial Specialist	0.0%	0.0%	0.0%	0.0%
Fluvial Dependent				
Catostomidae	8.4%	0.3%	1.6%	0.2%
Cyprinidae	28.8%	2.7%	1.0%	4.9%
Moronidae	0.2%	0.2%	0.0%	< 0.1%
Sciaenidae	31.0%	5.0%	2.2%	0.4%
Macrohabitat Generalist				
Atherinidae	0.0%	< 0.1%	0.0%	0.2%
Centrarchidae	0.3%	1.8%	0.7%	1.1%
Clupeidae	30.6%	89.8%	92.6%	92.5%
Gasterosteidae	0.0%	< 0.1%	0.1%	0.0%
Ictaluridae	0.0%	0.0%	0.5%	< 0.1%
Lepisosteidae	0.1%	< 0.1%	0.0%	0.0%
Percidae	< 0.1%	< 0.1%	0.1%	0.0%
Poeciliidae	< 0.1%	< 0.1%	1.2%	0.6%

Table 11. Kolmogorov-Smirnov tests for 2004 and 2005 pooled diel drift net data and Lower Swan Lake (LS) and the Illinois River (ILR) larval tow data. Test results are listed for treatments which collected significantly larger-sized fish.

	2004			2005		
Treatment	KSa	*P*	Result	KSa	*P*	Result
IN vs. OUT	10.91	< 0.01*	*Out*	4.41	0.01*	*In*
LS vs. ILR	8.83	< 0.01*	*LS*	5.37	< 0.01*	*LS*
NIGHT vs. DAWN	9.48	< 0.001*	*Night*	7.78	< 0.001*	*Night*
NIGHT vs. DAY	9.57	< 0.001*	*Night*	11.99	< 0.001*	*Night*
NIGHT vs. DUSK	10.93	< 0.001*	*Night*	2.55	< 0.001*	*Night*
DAWN vs. DAY	0.56	0.914	--	9.70	< 0.001*	*Dawn*
DAWN vs. DUSK	2.03	< 0.001*	*Dusk*	6.06	< 0.001*	*Dusk*
DAY vs. DUSK	1.48	0.026	--	4.74	< 0.001*	*Dusk*

* Indicates significant differences between treatments, where $P \leq \alpha$ *or* α''.

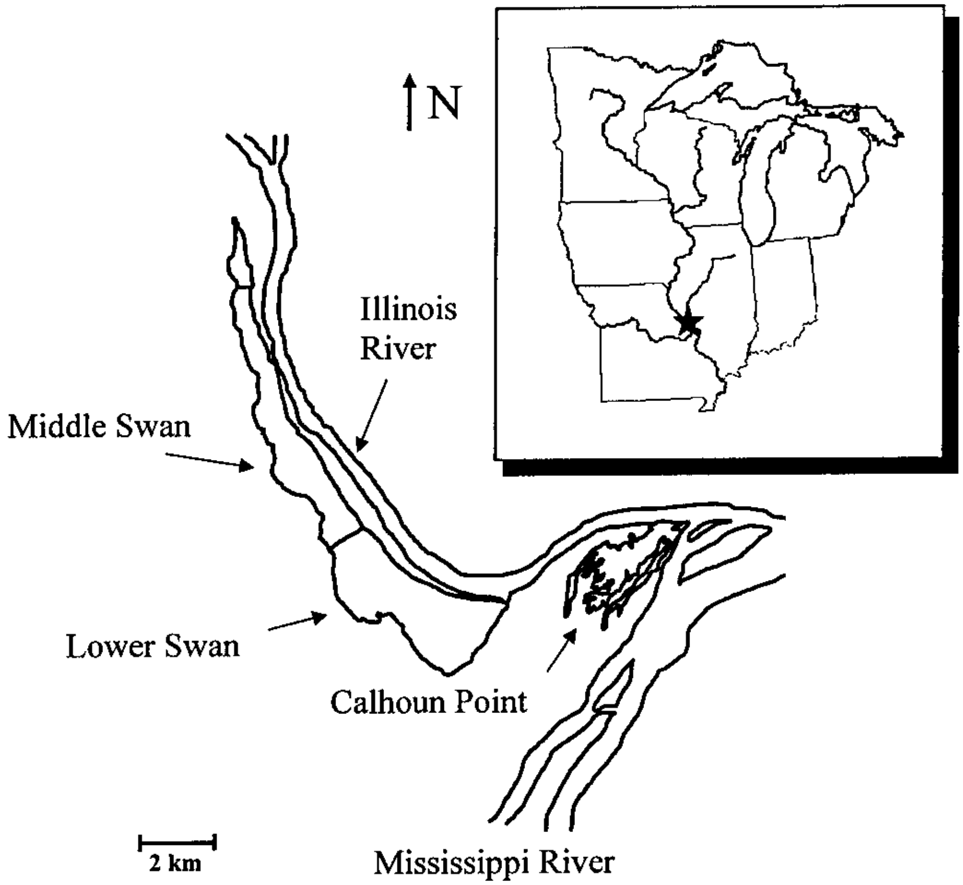

Figure 1. Study site of the lower Illinois River and two major backwaters, Swan Lake and Calhoun Point.

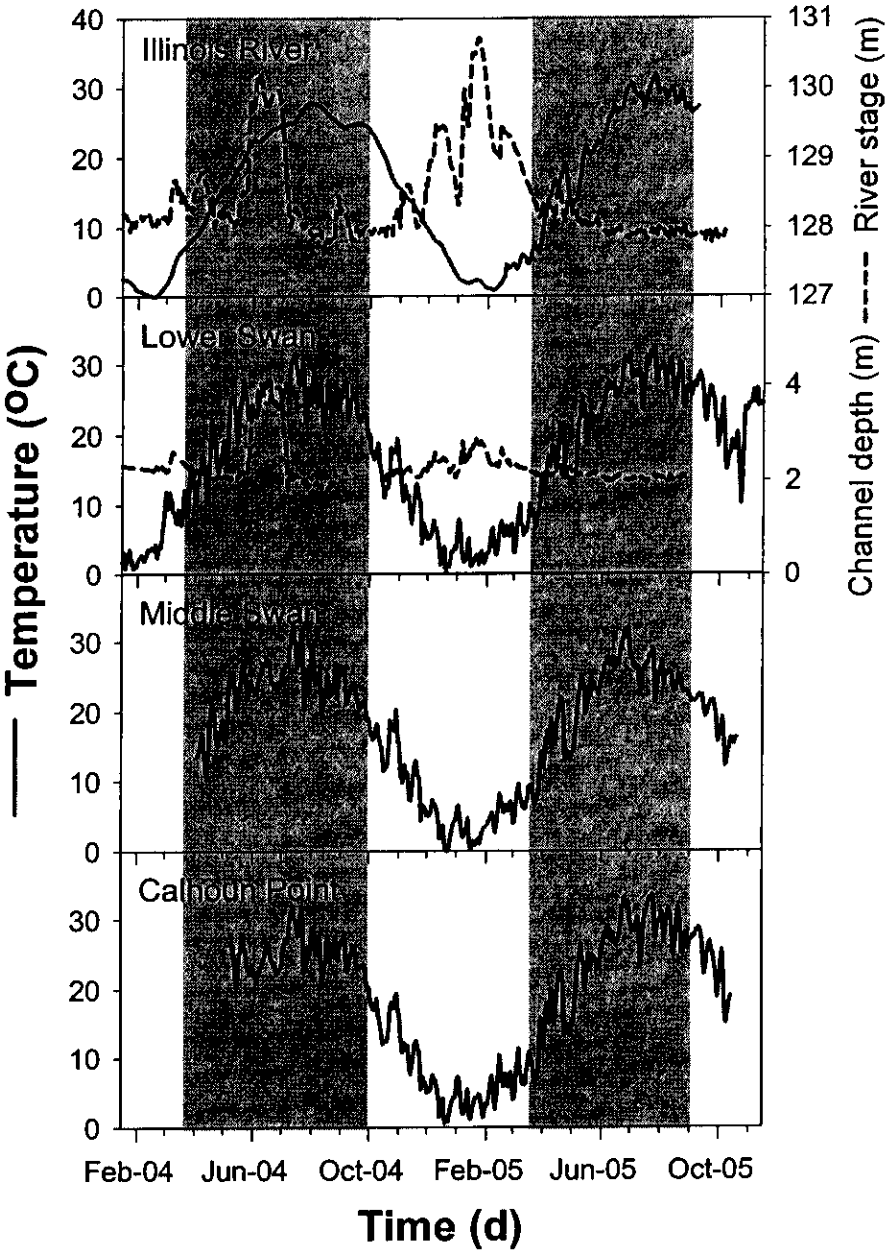

Figure 2. Mean daily temperature of Illinois River and backwater sites during 2004 through 2005, depicted as solid gray lines. Mean daily depth data are depicted as broken black lines for the Illinois River and Lower Swan. Channel depth of Lower Swan was recorded using a submerged device at the stop-log structure. River data were collected at Hardin, Illinois, USA (river kilometer 34.4). Depth was not recorded for Middle Swan and Calhoun Point. Shaded regions represent sampling periods.

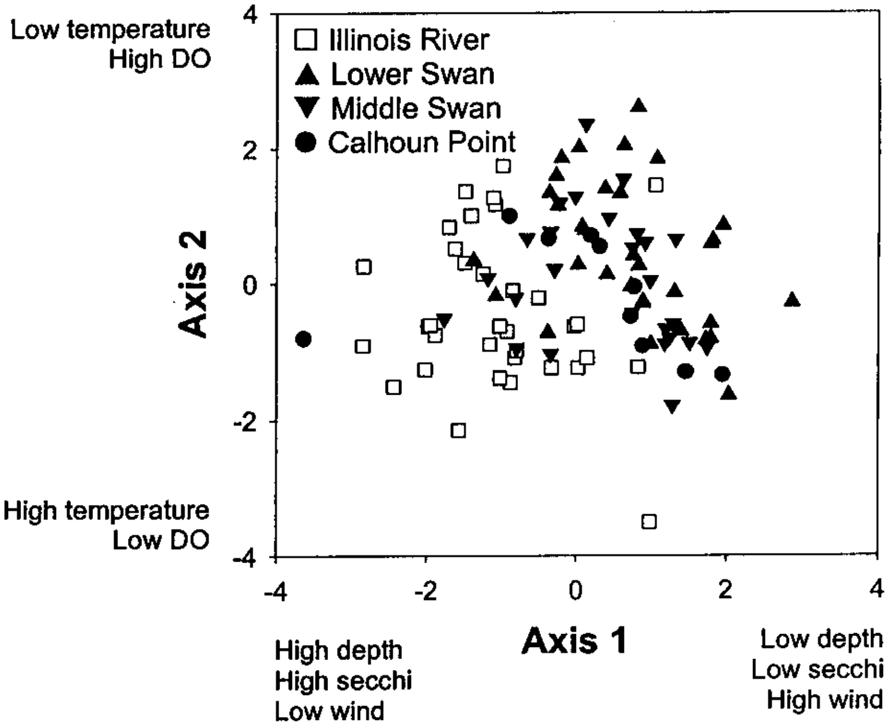

Figure 3. Principle component analysis (PCA) of abiotic variables for the lower Illinois River (□) and backwater sites, Lower Swan (▲), Middle Swan (▼), and Calhoun Point (●). Weekly means of abiotic variables were included in the analysis: temperature (°C), dissolved oxygen (DO, mg/L), depth (m), secchi depth (cm), and wind (km/h).

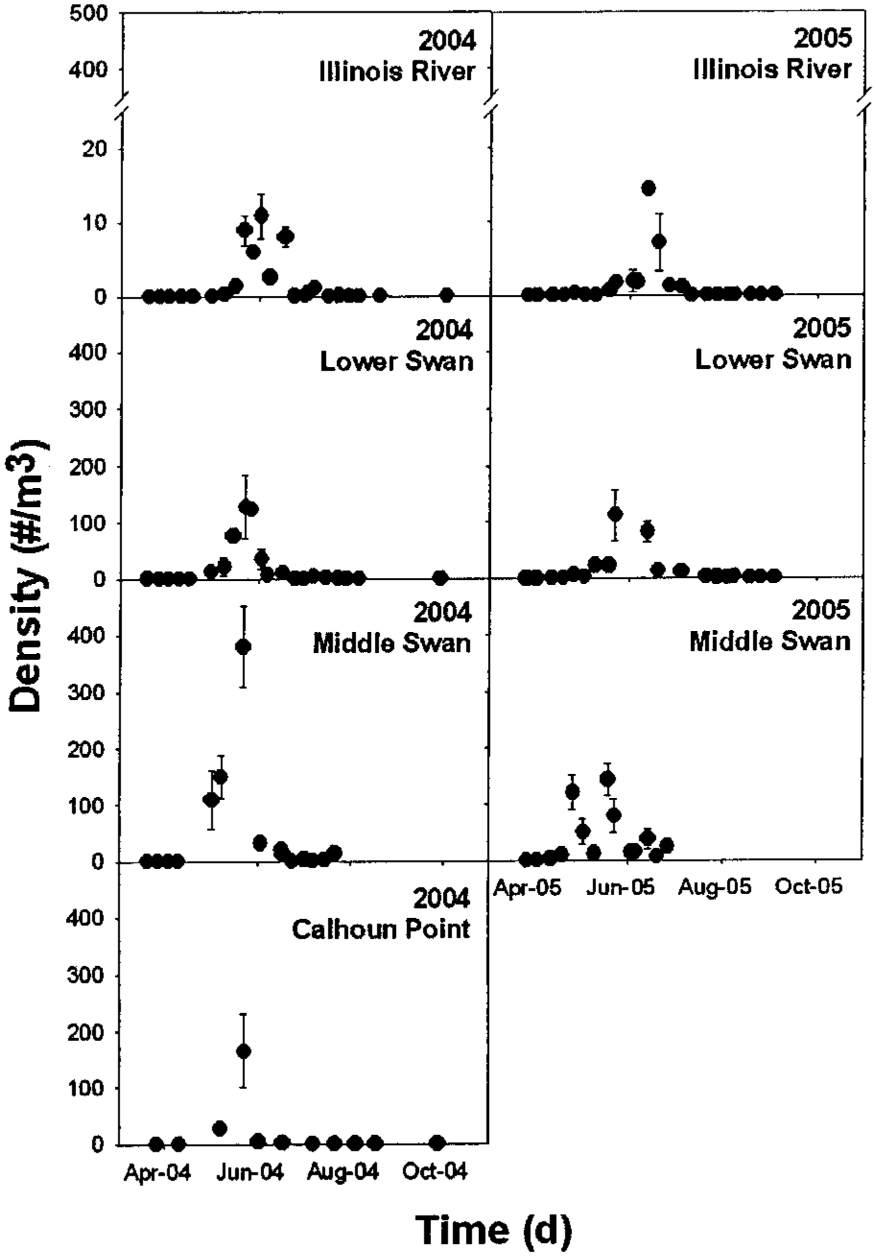

Figure 4. Mean density (#/m^3) per day of fish caught in the lower Illinois River during 2004 and 2005. Error bars represent ±1 standard error of the mean transect density. Calhoun Point was not sampled during 2005 because of low water levels.

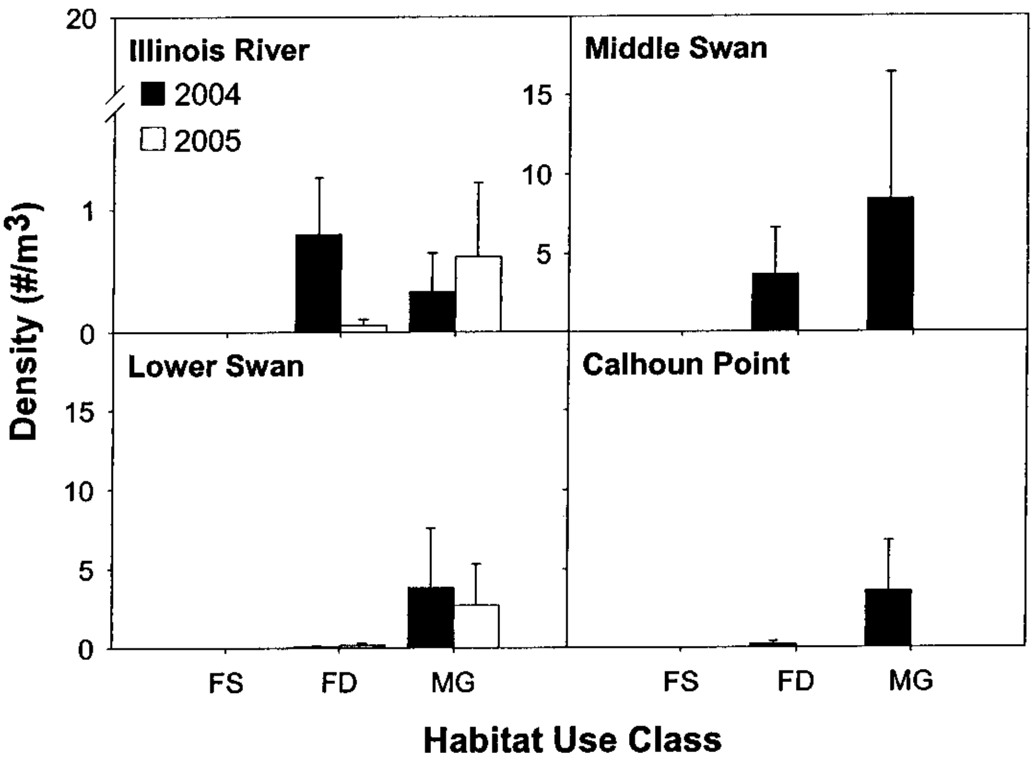

Figure 5. Fluvial specialist (FS), fluvial dependent (FD), and macrohabitat generalist (MG) mean densities by site and year in the lower Illinois River system. Error bars represent ±1 standard error of the mean of dates sampled during that year.

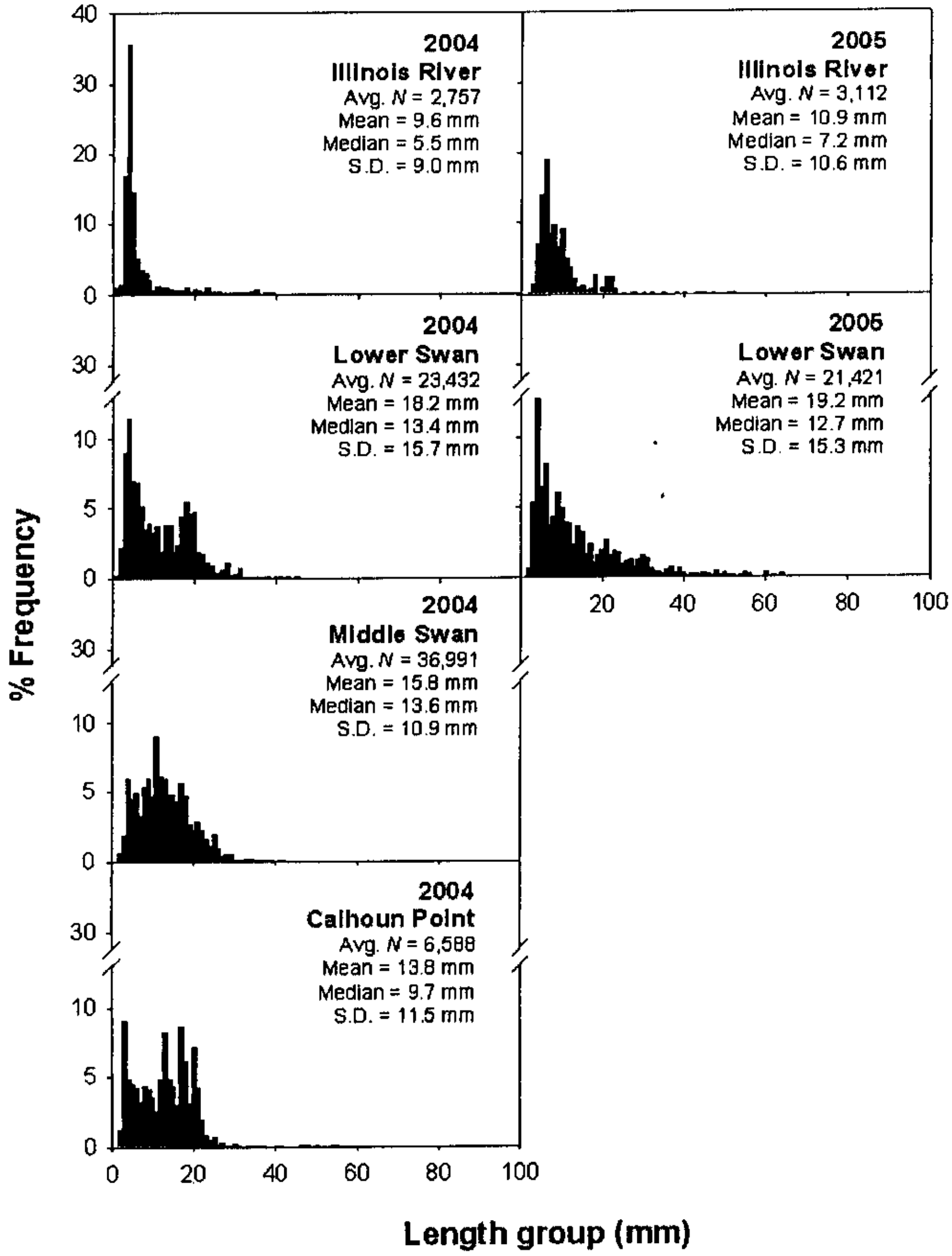

Figure 6. Length frequency distributions of all larval and juvenile fish caught in each site in of the lower Illinois River system during 2004 and 2005. Sample size is an average of transects within that site summed over the sampling season. Length frequencies are expressed as a percentage of average catch. Middle Swan samples were not collected after 23 July 2004. During 2005, lengths were collected in Middle Swan samples, and Calhoun Point was too shallow to be sampled. Descriptive statistics were calculated on raw length data.

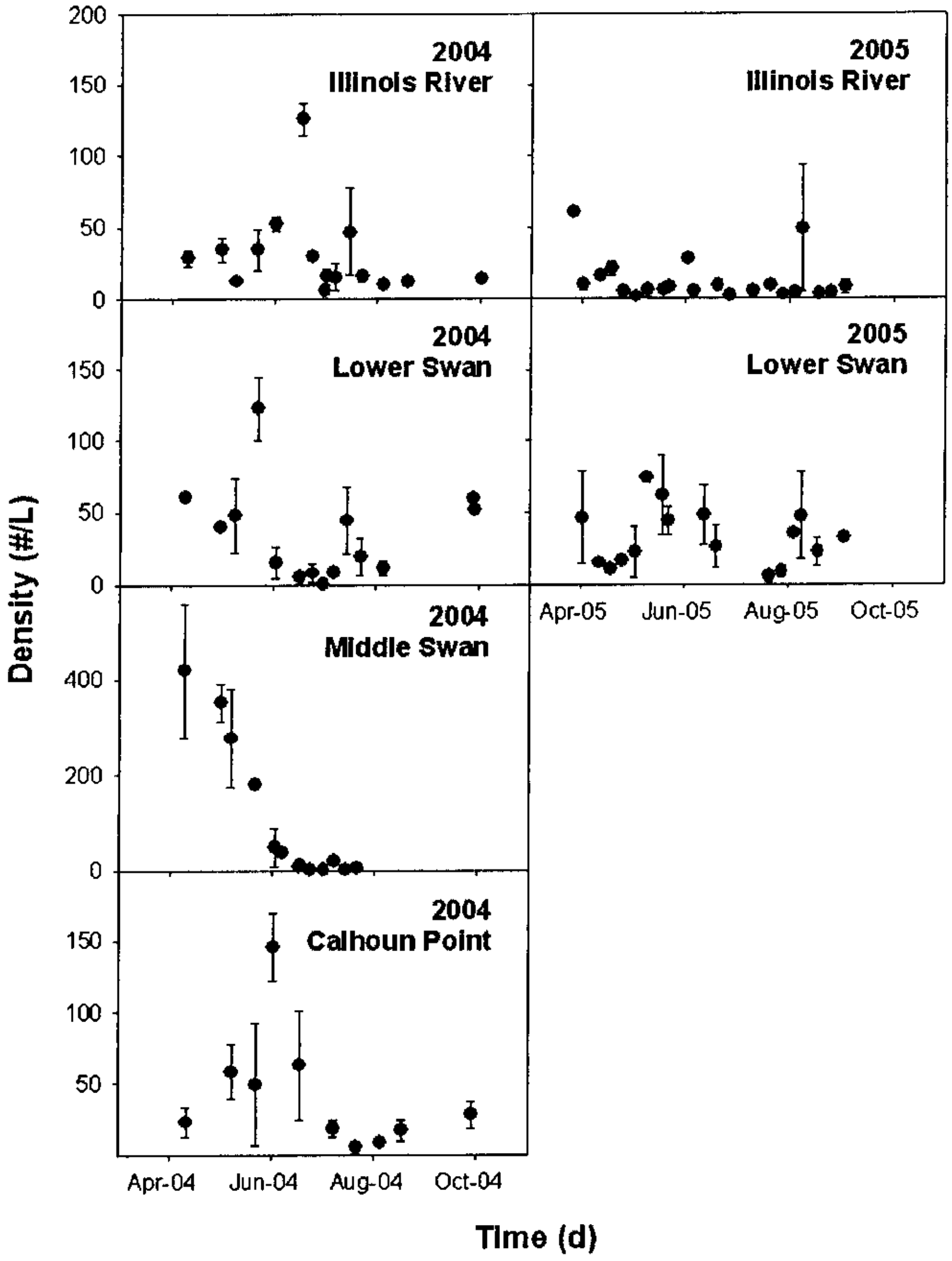

Figure 7. Mean density (#/L) of cladocerans, copepods, nauplii, and other zooplankters per day in each site of the lower Illinois River system through 2004 and 2005. Error bars represent ±1 standard error of the mean of transects sampled that day. Middle Swan samples were not collected after 23 July 2004. During 2005, zooplankton data were not identified for Middle Swan and no zooplankton samples were taken in Calhoun Point.

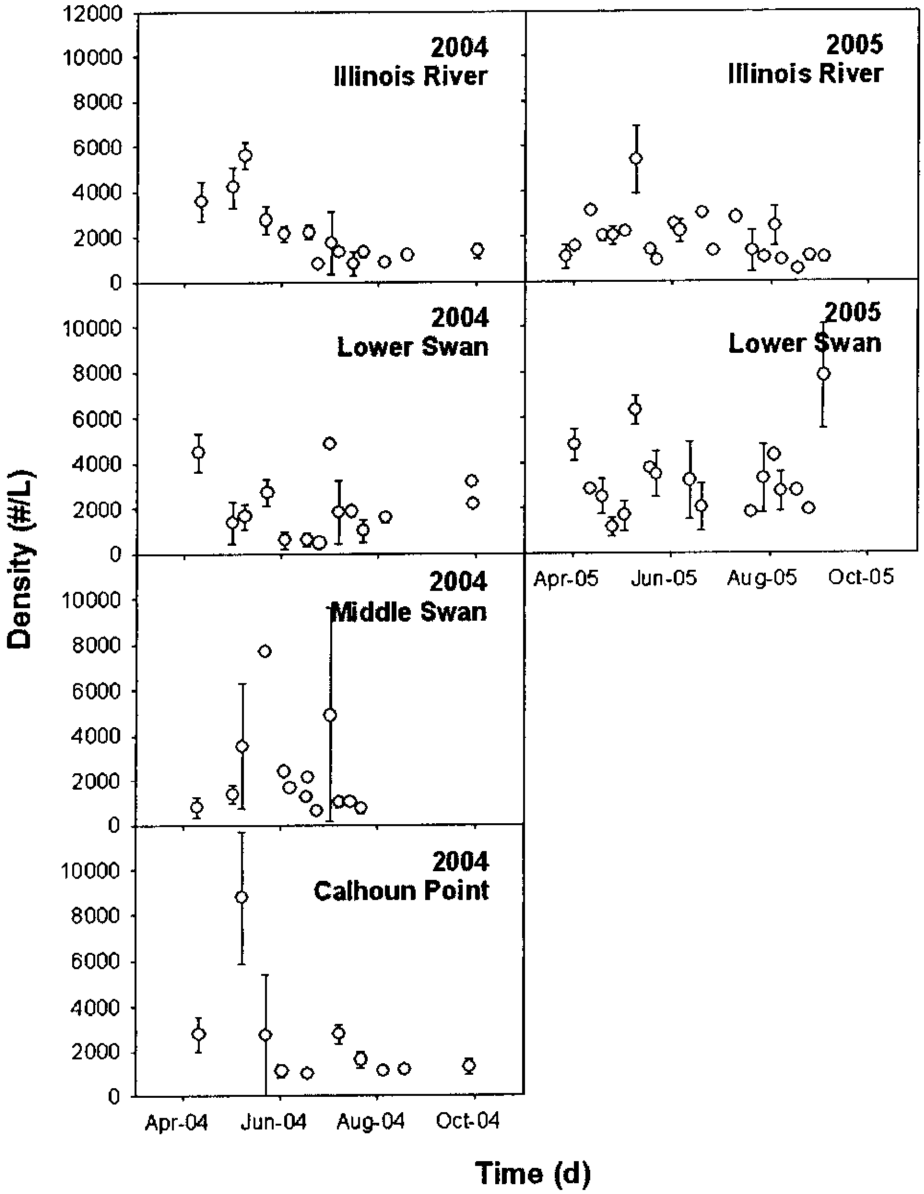

Figure 8. Mean density (#/L) of rotifers per day in each site of the lower Illinois River through 2004 and 2005. Error bars represent ±1 standard error of the mean of transects sampled that day. Middle Swan samples were not collected after 23 July 2004. In 2005, zooplankton data were not identified for Middle Swan and no zooplankton samples were taken in Calhoun Point.

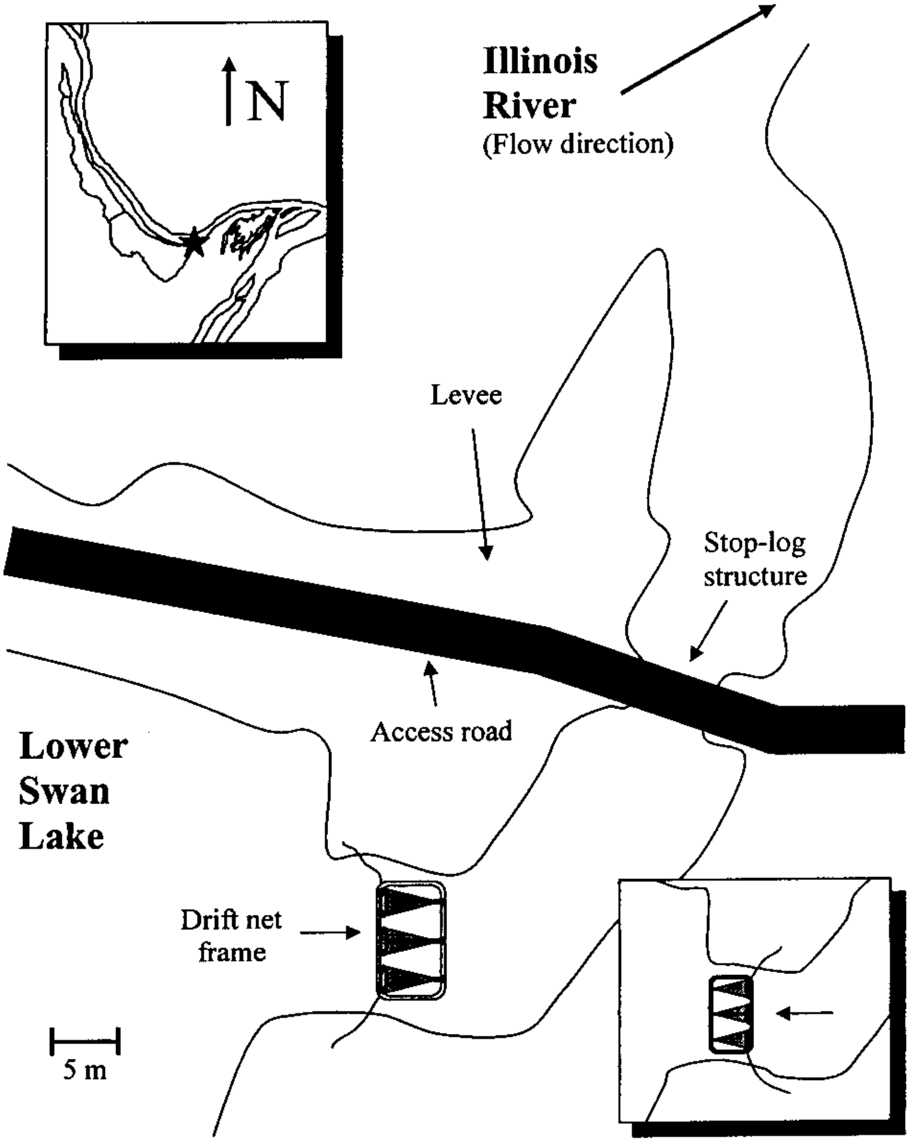

Figure 9. Fixed drift net site during 2004 and 2005 on the lower Illinois River (top inset). Tandem nets were floated on the Lower Swan (LS) side of the stop-log water control structure. Schematic depicts nets sampling larval fish drifting out of LS. Bottom inset portrays the drift net frame positioned to sample larval fish drifting into LS.

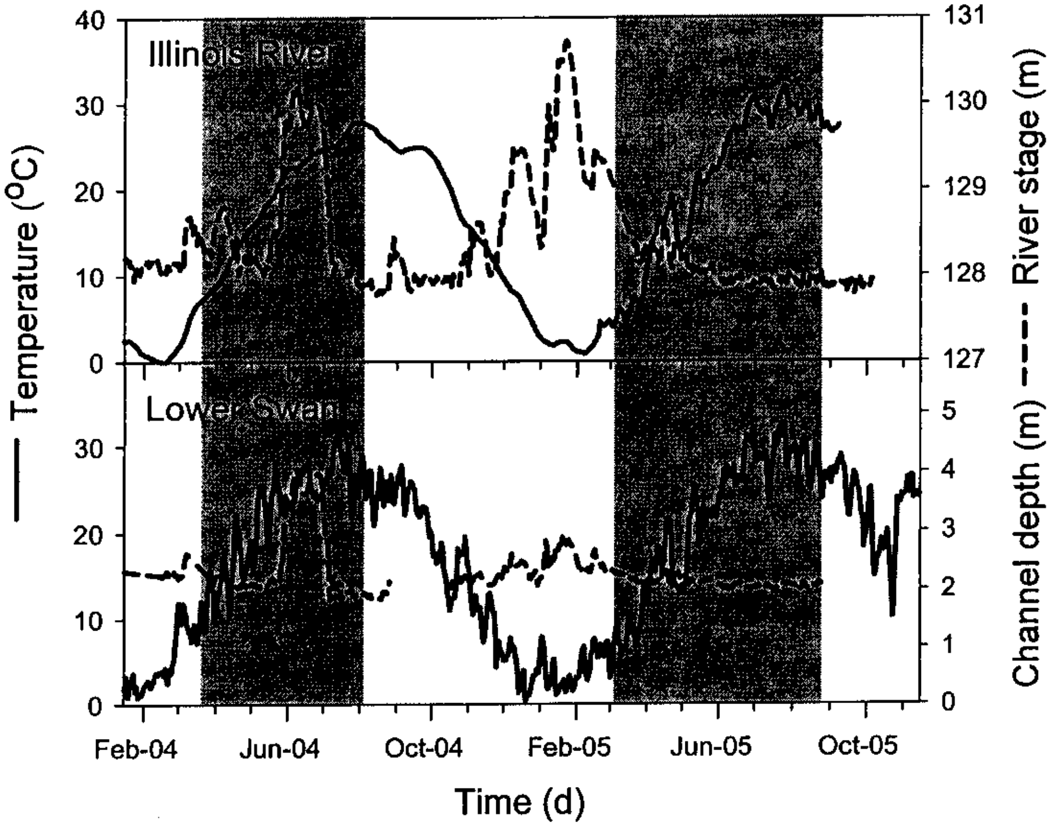

Figure 10. Mean daily temperature of the Illinois River and Lower Swan for 2004 and 2005, depicted as solid gray lines. Mean daily depth data are depicted as broken black lines. Channel depth of Lower Swan was recorded at the stop-log structure between the backwater and the river. River stage and temperature data were collected at Hardin, Illinois, USA (river kilometer 34.4). Shaded regions represent sampling periods.

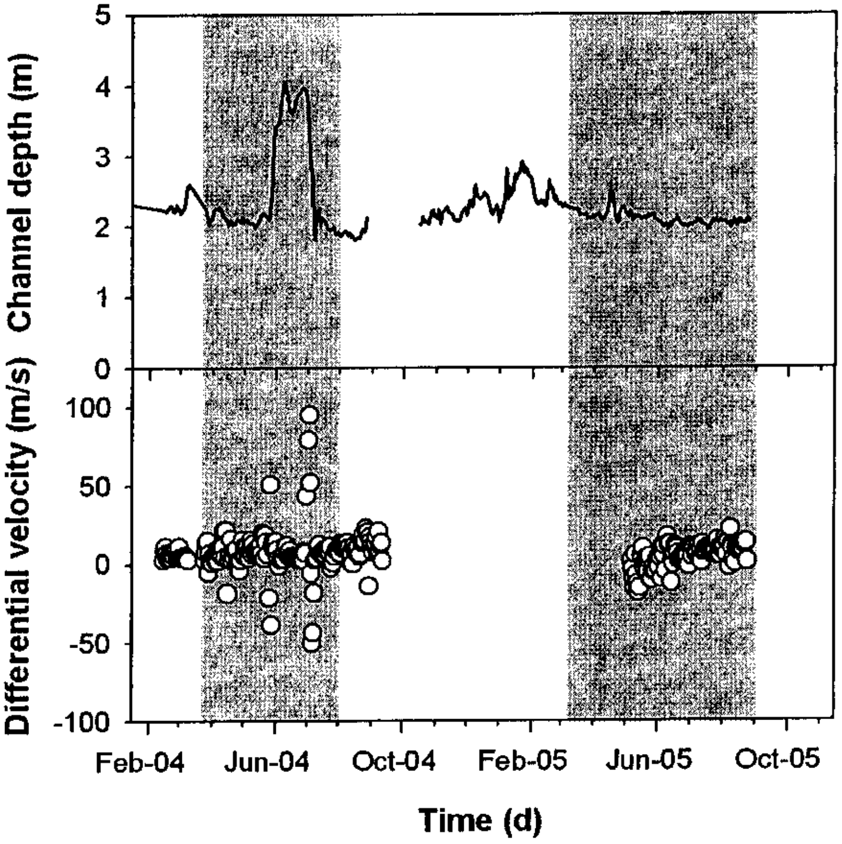

Figure 11. Mean daily depth (top) and differential velocity (bottom) measurements in the channel between Swan Lake and the Illinois River. Differential velocity was calculated as the sum of mean channel velocities per day (i.e., the sum of 48 values). Positive velocity values represent net inflow of water into Swan Lake and negative values correspond to net outflow to the river. Continuous data were recorded using a submerged device anchored at the stop-log structure. No depth data were collected from September through November 2004, and no velocity data were collected from October 2004 through April 2005. Shaded regions represent the 2004 and 2005 sampling periods.

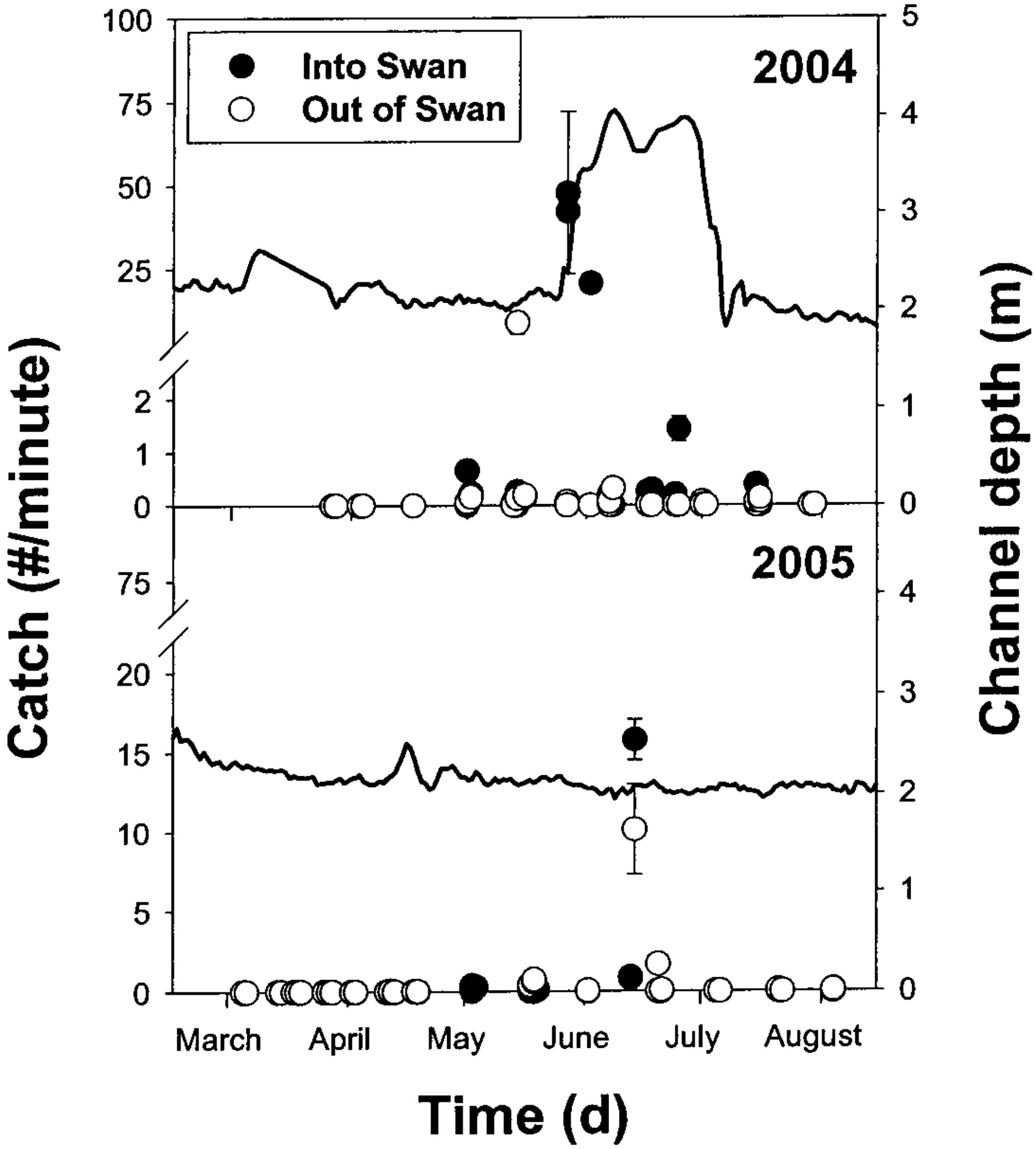

Figure 12. Drift of ichthyoplankton into (●) and out (○) of Swan Lake in 2004 and 2005. Symbols represent mean catch (#/minute) per net set and line data represent continuous depth data recorded at the stop-log structure. Error bars represent ±1 standard error for the mean of each net set.

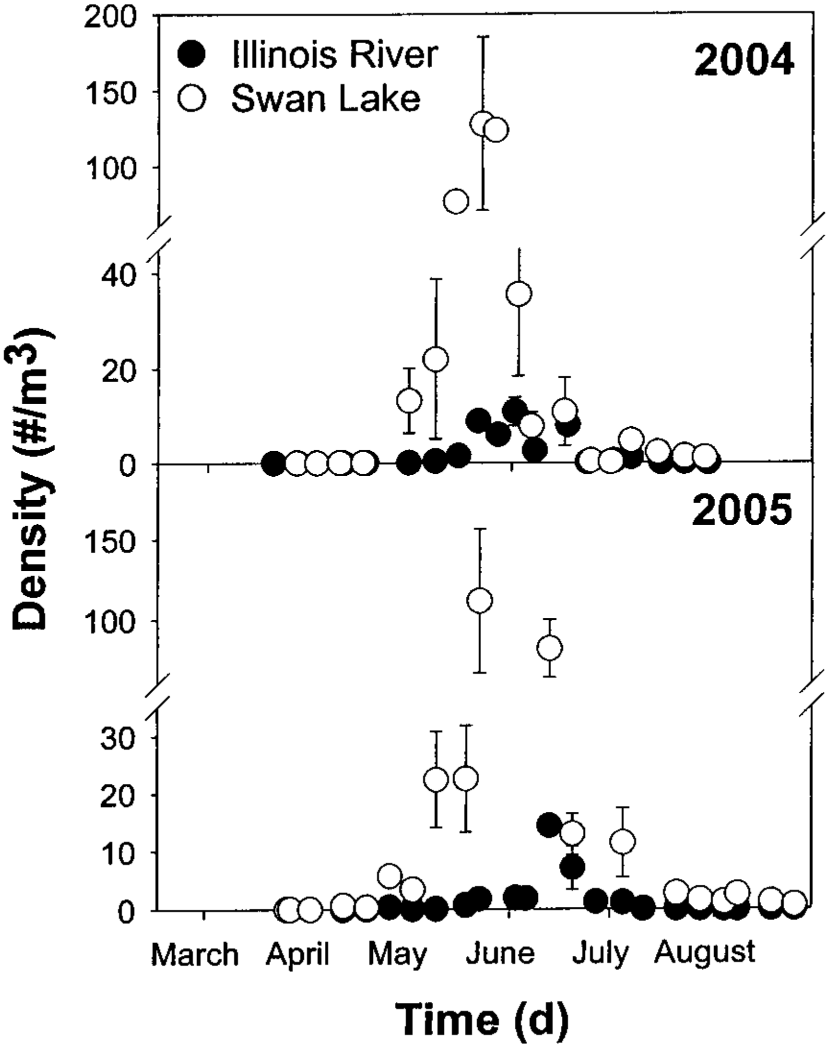

Figure 13. Mean density (#/m^3) per day of fish caught in the Illinois River and Swan Lake during 2004 and 2005. Error bars represent ±1 standard error of the mean transect density.

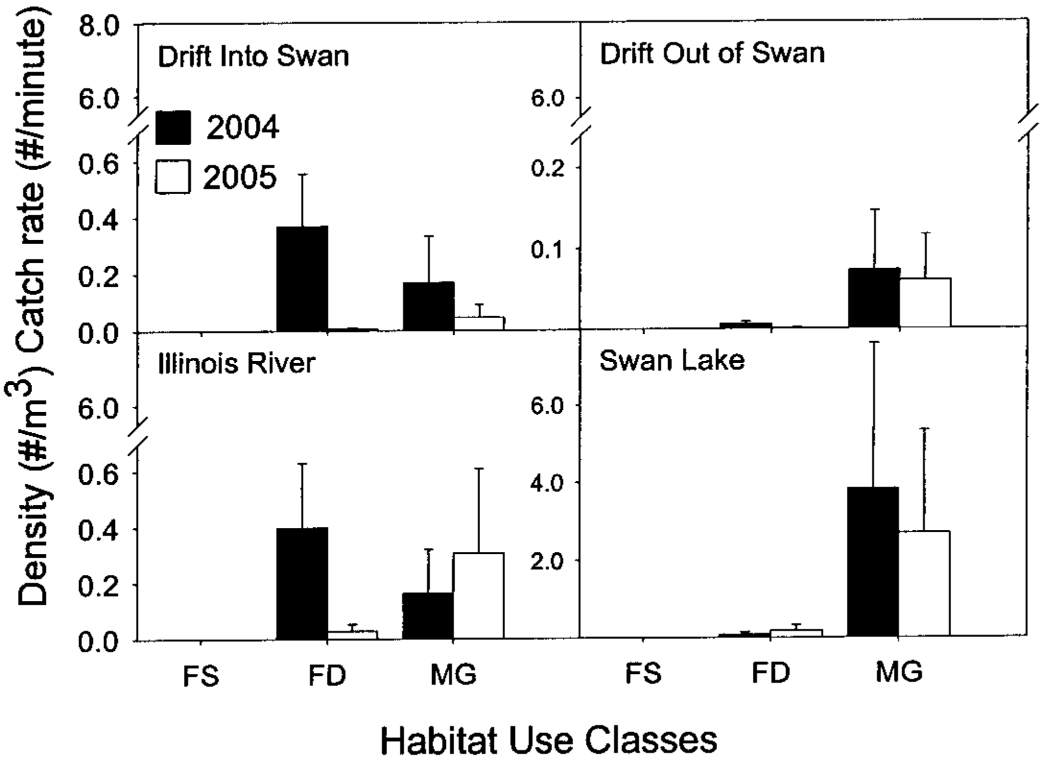

Figure 14. Mean catch rate (#/minute) of fish in three early life history classes drifting into and out of Swan Lake and mean density (#/m^3) of classes in the Illinois River and Swan Lake during 2004 and 2005. All families were included and grouped into one of three early life history classes as defined by Galat and Zweimuller (2001): fluvial specialists (FS), fluvial dependents (FD), and macrohabitat generalists (MG). Error bars represent ±1 standard error of the mean.

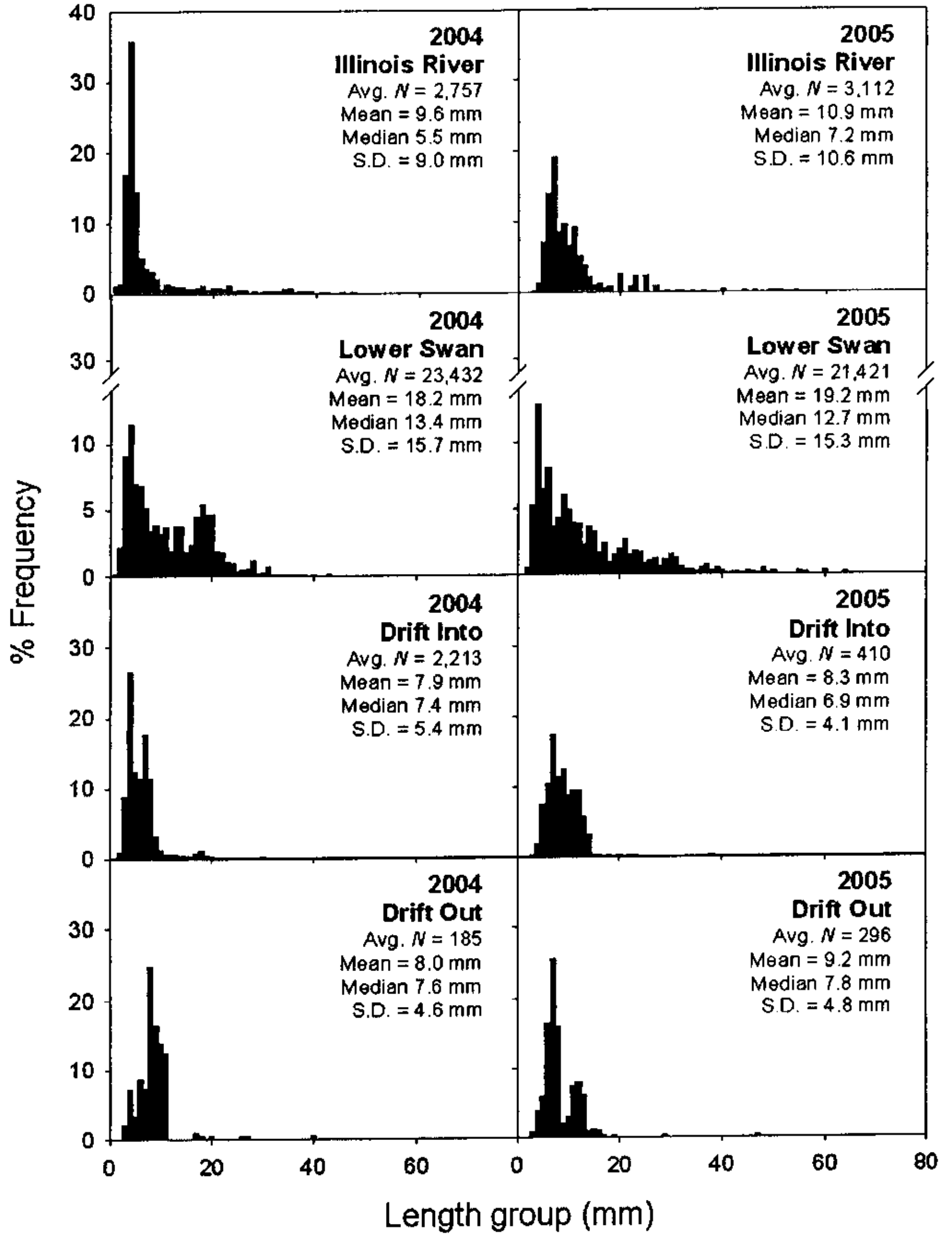

Figure 15. Length frequency distributions of all larval and juvenile fish caught in the Illinois River, Lower Swan, and in the drift during 2004 and 2005. Sample size is an average of transects or nets within that site summed over the sampling season for larval tow data or drift data, respectively. Length frequencies are expressed as a percentage of the average caught. Descriptive statistics were calculated on raw length data.

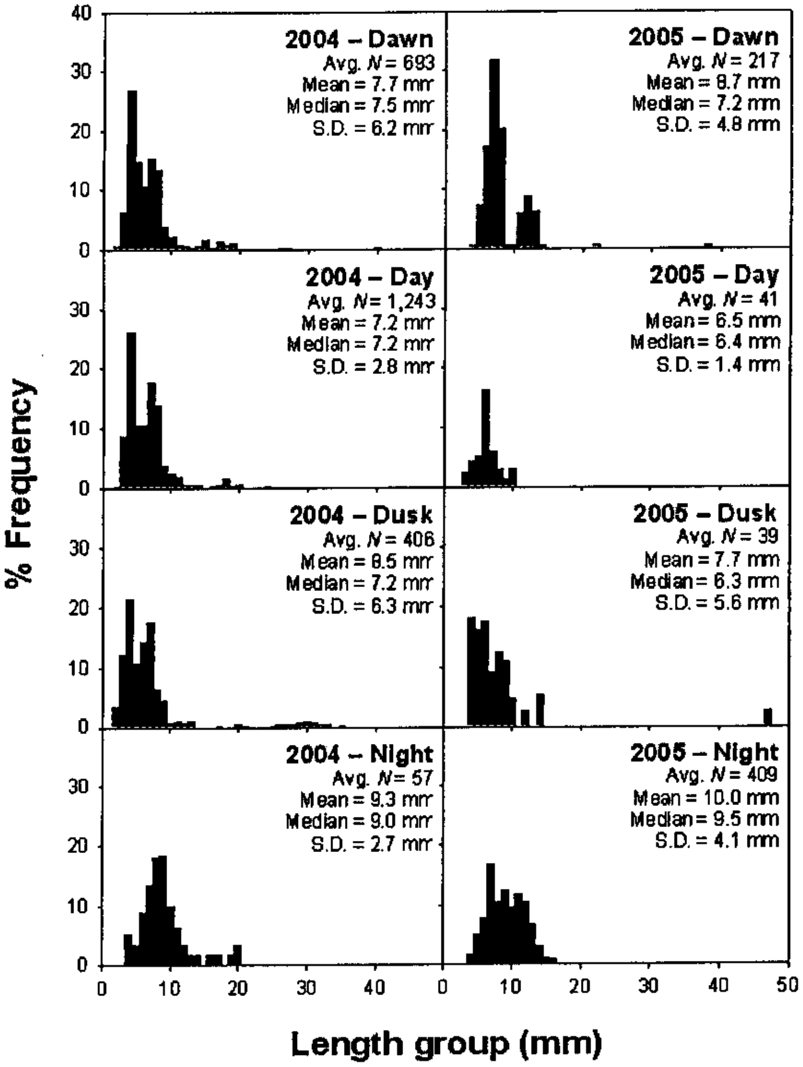

Figure 16. Length frequency distributions of all fish caught drifting during each time of day for 2004 and 2005. Data of fish drifting into and out of Lower Swan were pooled by time of day (i.e., dawn, day, dusk, night). Length frequencies are expressed as a percentage of the mean number per net set (i.e., mean of three nets) summed over the sampling season. Descriptive statistics were calculated on raw length data.

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Timing of Riverine Export of Nitrate and Phosphorus from Agricultural Watersheds in Illinois: Implications for Reducing Nutrient Loading to the Mississippi River

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Agricultural watersheds in the upper Midwest are the major source of nutrients to the Mississippi River and Gulf of Mexico, but temporal patterns in nutrient export and the role of hydrology in controlling export remain unclear. Here we report on NO_3^-−N, dissolved reactive phosphorus (DRP), and total P export from three watersheds in Illinois during the past 8−12 years. Our program of intensive, long-term monitoring allowed us to assess how nutrient export was distributed across the range of discharge that occurred at each site and to examine mechanistic differences between NO_3^-−N and DRP export from the watersheds. Last, we used simple simulations to evaluate how nutrient load reductions might affect NO_3^-−N and P export to the Mississippi River from the Illinois watersheds. Artificial drainage through under-field tiles was the primary mechanism for NO_3^-−N export from the watersheds. Tile drainage and overland flow contributed to DRP export, whereas export of particulate P was almost exclusively from overland flow. The analyses revealed that nearly all nutrient export occurred when discharge was ≥ median discharge, and extreme discharges (≥ 90th percentile) were responsible for >50% of the NO_3^-−N export and >80% of the P export. Additionally, the export occurred annually during a period beginning in mid-January and continuing through June. These patterns characterized all sites, which spanned a 4-fold range in watershed area. The simulations showed that reducing in-stream nutrient loads by as much as 50% during periods of low discharge would not affect annual nutrient export from the watersheds.

Introduction

Nitrogen and P enrichment from nonpoint sources and resulting eutrophication is a main cause of poor water quality and biotic impairment in many streams and rivers in the United States (*1*). To address nutrient inputs from nonpoint sources, states currently are developing nutrient criteria, numeric standards, and total maximum daily loads (TMDLs) for nutrient-impaired streams and rivers. A TMDL represents the maximum load (kg d^{-1}) of a nutrient that a stream can receive and still maintain water quality sufficient to meet its designated uses. Determination of the TMDL is based on the assimilative capacity of the stream for that nutrient, natural or background sources, point and nonpoint inputs, and a specified margin of safety (*2*). Because the assimilative capacity, discharge, and magnitude of inputs can vary throughout a year, the approach allows for seasonal variation in the TMDL. In streams of the midwestern United States, the effects of eutrophication are most pronounced during periods of low discharge and warm water temperatures in summer and autumn. Nutrient TMDLs tend to focus on critical periods of summer low discharge, while allowing increased nutrient loads during times of high discharge. For example, the Stillwater River drains an agricultural watershed in western Ohio and the approved TMDL for NO_2^- + NO_3^- increases from 3122 kg N d^{-1} in October and November to >6700 kg N d^{-1} for December through June (*3*). The higher load is needed to accommodate the increased discharge and nonpoint source runoff that occur from late winter through spring.

In addition to degrading local water quality, nutrient enrichment of midwestern streams has increased N and P loading to the Mississippi River and Gulf of Mexico (*4−6*). Rivers draining agricultural regions of the upper Midwest (i.e., the cornbelt) export large quantities of P and N (predominantly as NO_3^-−N) as a result of extensive fertilization and artificial drainage (*5, 7, 8*). Approximately 20 million hectares of cropland in the Mississippi River basin (MRB) are artificially drained by under-field (tile) systems, particularly in intensively farmed and fertilized areas such as Iowa, Illinois, Indiana, and Ohio (*5, 9*). Tile drainage provides a mechanism by which water and dissolved nutrients can bypass groundwater flow paths and move rapidly from fertilized cropland to streams and rivers (*10, 11*). Because of channel and hydrological modifications, streams in agricultural watersheds are not efficient at nutrient removal by processes such as denitrification, and a large fraction of the nutrient load in such streams is transported to downstream water bodies (*12, 13*).

Nutrient export to the Mississippi River and eutrophication of midwestern streams both result from nutrient inputs to surface waters but are associated with different hydrological conditions. Eutrophication of streams is a primarily biological process driven by nutrient uptake when local conditions favor rapid growth of nuisance algae, such as during extended periods of low discharge. Conversely, nutrient export from the Midwest to the Mississippi River is a primarily hydrological process driven by precipitation and drainage of the agricultural landscape (*14*). To address water quality in the Midwest, it is critical to gain a mechanistic understanding of how N and P enter, and are exported from, midwestern streams. There is a particular need for long-term data to address (i) the temporal patterns in nutrient loads and export, (ii) the role of hydrology in controlling export, and (iii) the implications of these patterns for efforts, such as TMDLs, to reduce eutrophication and nutrient export from the agricultural Midwest.

A robust analysis of these issues requires long-term data on NO_3^-−N and P concentrations and river discharge, and a sampling scheme that targets periods of high discharge and rapidly changing nutrient loads. In this paper we use long-term, intensive monitoring data from three agricultural watersheds in Illinois to examine the above issues in relation

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TABLE 1. Location, Watershed Characteristics, Period of Record, and Discharge Statistics for the River Sites Used in the Analysis[a]

river system	site coordinates	watershed area (km^2)	row-crop agriculture (% land cover)	period of record (water years)	maximum discharge (m^3 s^{-1})	median discharge (m^3 s^{-1})	mean discharge (m^3 s^{-1})
Embarras	39°47′29″N, 88°11′08″W	481	91	1994–2005	198	1.6	5.0
Kaskaskia	39°50′09″N, 88°29′18″W	386	91	1998–2005	108	1.0	3.6
Sangamon	40°16′06″N, 88°19′35″W	101	86	1994–2003	71	0.3	0.8

[a] Discharge statistics based on average daily discharge values from the period of record.

to nutrient management scenarios. Using simple simulations of nutrient load reductions, we evaluate how such reductions might affect NO_3^-–N and DRP export to the Mississippi River from areas such as Illinois. Last, we discuss the implications of our results for designing programs to reduce riverine export of nutrients from agricultural regions of the upper Midwest.

Site Descriptions. We used long-term data on discharge and nutrient concentrations from sites in the Embarras, Kaskaskia, and Sangamon river systems in east-central Illinois. The sites range in drainage area and peak discharge but have similar land use dominated by row-crop agriculture, mainly corn and soybean (Table 1). East-central Illinois soils are poorly drained Mollisols and the landscape was mostly wetland and mesic prairie prior to settlement. Much of the landscape is now tile-drained, with tile densities of 3–5 km km^{-2} (*15*), and headwater streams have been extensively channelized and dredged to accommodate high discharges (*16*). Nitrogen losses from the watersheds typically range from 20 to 50 kg N ha^{-1} yr^{-1}, depending on precipitation, and are among the highest in the MRB (*5, 7, 17*). In the streams, inorganic N loads are 90% or more NO_3^-–N (*7*). The land use and drainage modifications at the sites are representative of tile-drained and intensively farmed areas of the upper Midwest, and patterns in these streams likely characterize much of the cornbelt region.

Methods

Stream discharge was monitored at the Embarras River and Kaskaskia River sites by the U.S. Geological Survey (stations 03343400 and 05590800, respectively). Discharge at the Sangamon River site was monitored by the Illinois State Water Survey (station 106). For each site, mean daily discharge was determined from hourly or 15-min readings. The Illinois State Water Survey provided precipitation data (Champaign station) and NO_3^-–N concentrations for the Sangamon River site for 1994–1999. All other nutrient concentrations were determined from samples we collected approximately weekly, either manually or with automated samplers. Streams and small rivers in the Midwest have flashy hydrology, and discharge during floods represents a significant fraction of annual discharge (*18*). To account for this, we collected additional samples when discharge was changing rapidly. During most floods samples were collected daily, but in some cases 2–4 samples were collected in a 24-hr period. In total, our analysis is based on >4000 nutrient concentrations determined from >2000 individual water samples collected from October 1993 through September 2005.

Nitrate was determined, after filtration through a 0.45 μm membrane, on an ion chromatograph (Dionex, Inc. model DX-120 or model 2000i). Dissolved reactive phosphorus (DRP) was determined colorimetrically on filtered samples using a spectrophotometer or (after 2001) a Lachat QuikChem8000 flow injection analyzer. Total P was determined as described for DRP except that samples were unfiltered and digested with sulfuric acid and ammonium persulfate prior to analysis. Particulate P is defined as the difference between total P and DRP. Internal and external standards for each nutrient were analyzed routinely throughout the study as part of a quality assurance plan. Daily in-stream nutrient loads were determined by multiplying mean daily discharge (m^3 d^{-1}) by nutrient concentration (kg m^{-3}). Linear interpolation in SAS (*19*) was used to estimate nutrient concentrations between sampling dates. Annual and total nutrient export were determined by summing the daily nutrient loads for each water year or the period of record, respectively. For a 1406 km^2 agricultural watershed in Illinois, weekly NO_3^-–N sampling resulted in a load estimate with a root-mean-square error of <4% (*20*). We examined smaller watersheds (Table 1), but with a greater sampling frequency, and believe our load estimates have similarly small error.

For the period of record at each site, we ranked the daily discharge values and calculated the fraction of the total nutrient export attributable to each day, and the fraction that occurred between various discharge percentiles. This allowed us to assess how NO_3^-–N, DRP, and total P export were distributed across the range of discharge that occurred at each site. The mass of nutrients exported each year from the MRB is controlled to a large extent by precipitation (*14*). To examine patterns across years and account for inter-annual differences in precipitation and export, we constructed graphs of cumulative NO_3^-–N and DRP export based on percentage of the total for each water year.

To examine how reduced NO_3^-–N and DRP loads during different hydrological conditions might affect export of these nutrients to downstream water bodies, we performed three simple simulations for each site. The first simulation reduced in-stream NO_3^-–N and DRP loads by 50% on all days with discharge <50th percentile; this simulation represented the effect of focusing nutrient reductions on periods of low discharge only. The second simulation reduced in-stream loads by 25% on all days with discharge <75th percentile; this represented a smaller reduction in nutrient loads but applied to a larger range of discharge (i.e., more days of the year). The final simulation focused on high flow periods and reduced in-stream loads of NO_3^-–N and DRP by 25% on all days with discharge ≥75th percentile. For all simulations, NO_3^-–N and DRP export during the period of record was recalculated based on the adjusted loads and expressed as a percent of the original export. The reductions used in the simulations were selected to evaluate the interaction between hydrology and nutrient export and do not necessarily reflect management goals, although N load reductions of 20–30% in the Mississippi River will be required to reduce hypoxia in the Gulf of Mexico (*21*).

Results

The volume of water and mass of NO_3^-–N, DRP, and total P exported from the watersheds varied considerably during the study (Table 2), mainly as a consequence of variable precipitation among years. In wet years, such as 1998 and

TABLE 2. Annual Water and Nutrient Export from the Three Watersheds during the Study

water year	discharge (10^6 m^3)	Mg			kg ha^{-1}		
		NO_3^-–N	DRP	total P	NO_3^-–N	DRP	total P
			Embarras				
1994	235	1599	43		33.3	0.9	
1995	125	1081	11		22.5	0.2	
1996	186	1999	24		41.5	0.5	
1997	137	1140	21		23.7	0.4	
1998	228	2219	31		46.1	0.6	
1999	132	1372	21		28.5	0.4	
2000	52	490	4	7	10.2	0.1	0.2
2001	92	962	12	16	20.0	0.2	0.3
2002	290	2728	35	102	56.7	0.7	2.1
2003	53	426	5	9	8.9	0.1	0.2
2004	170	1506	24	51	31.3	0.5	1.1
2005	180	1396	29	51	29.0	0.6	1.1
			Kaskaskia				
1998	179	2223	21		57.6	0.5	
1999	96	1389	8		36.0	0.2	
2000	46	514	3	6	13.3	0.1	0.2
2001	100	1164	10	14	30.2	0.3	0.4
2002	187	2129	13	38	55.2	0.3	1.0
2003	31	292	2	4	7.6	< 0.1	0.1
2004	130	1382	11	45	35.8	0.3	1.2
2005	150	1288	22	40	33.4	0.6	1.1
			Sangamon				
1994	38	262			26.0		
1995	29	328			32.5		
1996	31	318			31.5		
1997	35	337			33.3		
1998	36	369			36.6		
1999	19	216			21.4		
2000	8	91			9.0		
2001	19	193	4	7	19.1	0.4	0.7
2002	34	473	3	8	46.8	0.3	0.8
2003	16	146	1	3	14.5	0.1	0.3

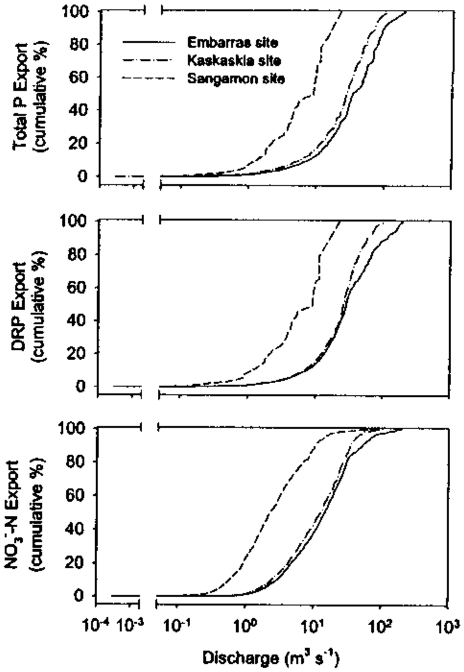

FIGURE 1. Cumulative nutrient export as a function of discharge during the period of record for each site.

TABLE 3. Percentage of the Total Nutrient Export that Occurred at or above Various Discharge (*Q*) Percentiles during the Periods of Record

river system	% of export		
	Q ≥ 90th percentile	*Q* ≥ 75th percentile	*Q* ≥ 50th percentile
	NO_3^-–N		
Embarras	54	81	97
Kaskaskia	58	84	98
Sangamon	57	82	97
mean	56	82	97
	DRP		
Embarras	80	94	98
Kaskaskia	85	96	99
Sangamon	86	94	98
mean	84	95	98

2002 (117 and 108 cm of precipitation, respectively), NO_3^-–N yields ranged from 45 to 55 kg ha^{-1}. In dry years, such as 2003 (82 cm of precipitation) yields were <15 kg ha^{-1}. Across the sites there was a clear pattern of nutrient export occurring predominantly at the high end of the discharge range (Figure 1). During the period of record, days with discharge ≥ median discharge accounted for 97–98% of the NO_3^-–N export and 98–99% of the DRP export (Table 3). Extremes in discharge (>90th percentile) accounted for an average of 56% of the total NO_3^-–N export and 84% of the DRP export.

The temporal distribution of NO_3^-–N export indicated that the majority of the annual export occurred during a 5.5 month period from mid-January through June across all sites and years (Figure 2). Within a water year, the first 3.5 months (October–mid-January) and the last 3 months (July–September) together typically accounted for <30% of the annual NO_3^-–N export. Within the January–June period, NO_3^-–N export often occurred in discrete events, as evidenced by the abrupt increases in the slopes of the lines in Figure 2. The watersheds spanned a 4-fold range in size, but the pattern in temporal distribution of NO_3^-–N export was consistent across sites. For P, we focus on DRP rather than total P because we have longer records for DRP and it represents the immediately available P. As with NO_3^-–N, DRP export occurred mainly from January to June in most years, although substantial export occasionally occurred later in the summer (Figure 3). Export of DRP was often associated with individual floods and in several years 40–80% of the annual DRP export occurred during a period of <1 month.

Mechanistically, NO_3^-–N, DRP, and total P responded differently to the occurrence of overland flow and we illustrate these differences with the 2002 and 2003 water years (wet and dry years, respectively) at the Embarras River site (Figure 4). In 2002 a series of precipitation events from February through mid-April initiated flow through agricultural tile drains, increased discharge, and resulted in a steady export of NO_3^-–N, with that period accounting for approximately 55% of the total 2002 NO_3^-–N export. During that same period, DRP and total P exports were 33 and 17%, respectively, of 2002 annual export. From April 8 through May 17, 2002, total precipitation was 21 cm and much of that water entered the Embarras River as overland flow and caused sustained flooding. There also were several periods of high discharge through tile drains throughout the Embarras River watershed. During these floods and high tile discharge periods, DRP and total P export increased substantially from the previous months, whereas NO_3^-–N export continued at approximately the same rate (Figure 4). Overland flow was not important for NO_3^-–N export from the watershed, but during 2002 overland flow was important for export of DRP and total P (with particulate P accounting for a large fraction of total P).

We cannot assign precise values to the fraction of the DRP and total P export originating from tile drainage versus

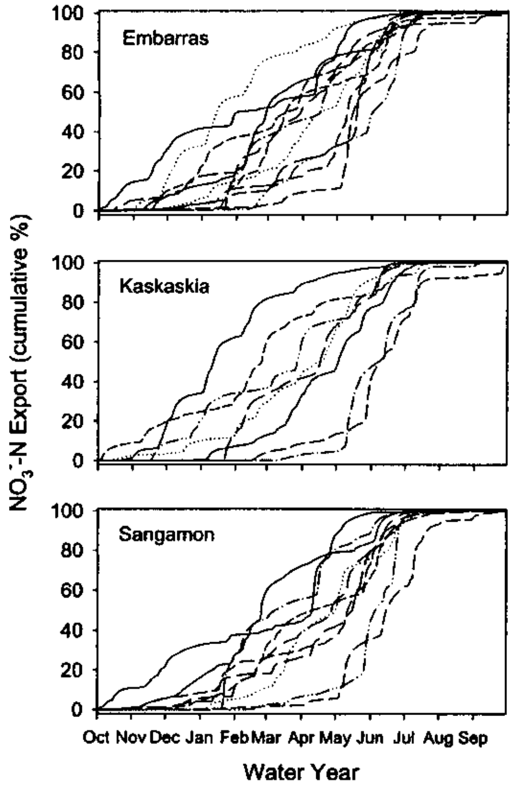

FIGURE 2. Cumulative NO_3^-−N export during each water year during the period of record for each site.

overland flow, because DRP export through tiles is greatest during peak tile flows but then decreases rapidly and disproportionately as tile discharge declines (*11*). This contrasts with NO_3^-−N export which continues via tile drainage approximately in proportion to tile discharge (*10*). This pattern is evident in the dry 2003 water year, when DRP and total P both increased along with NO_3^-−N in the first flow event in May, but then diverged from NO_3^-−N as discharge and tile flow declined (Figure 4). Dissolved reactive P and total P export increased discretely during the other two flow events, but these were much smaller than the 2002 floods and not associated with surface runoff. Therefore, in a dry year with small N and P export (Table 2), nearly all DRP and total P inputs to the river appeared to be from tile drainage.

The simulations showed that a 50% reduction in NO_3^-−N and DRP loads during periods of low discharge (i.e., <median discharge) would reduce total export of these nutrients by <2% in the case of NO_3^-−N and <1% in case of DRP (Table 4). Reducing loads 25% across a wider range in discharge (<75th percentile) resulted in a larger, but nonetheless disproportional, reduction in total export, with an average decline of 13.0% for NO_3^-−N and 4.2% for DRP. Conversely, reducing loads 25% during periods of high discharge (≥75th percentile) gave a nearly proportional reduction in total export of NO_3^-−N (20.7%) and DRP (23.6%) (Table 4).

Discussion

In midwestern agricultural watersheds, in-stream nutrient concentrations are greatest during winter through spring because of the increase in runoff that occurs during that time (*22, 23*). This pattern has been previously documented in the streams we examined (*7, 12, 24*) and we believe that our study sites, although limited geographically to east-central Illinois, are representative of intensively drained and farmed watersheds of the upper Midwest. Our analyses revealed two important characteristics regarding the export of nutrients from the agricultural Midwest during the past 8−12 years. First, nearly all nutrient export occurred when discharge was ≥ median discharge, and extreme discharges (Q ≥ 90th percentile) were responsible for >50% of the NO_3^-−N export and >80% of the P export. Second, the annual export of NO_3^-−N and P from the watersheds occurred consistently from mid-January through June.

As discharge increases, there is less opportunity for the exchange of nutrients between the water column and the

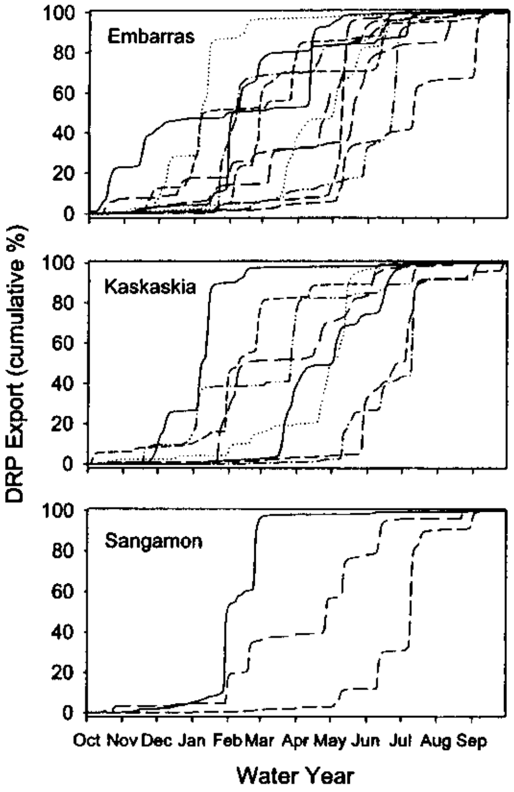

FIGURE 3. Cumulative DRP export during each water year during the period of record for each site.

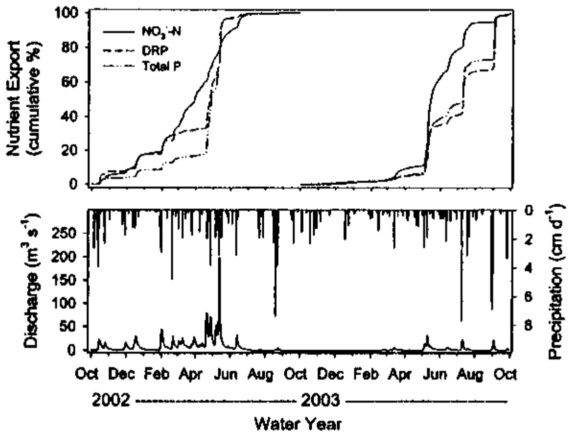

FIGURE 4. Cumulative nutrient export (upper) and precipitation and discharge (lower) from the Embarras River site during the 2002 and 2003 water years.

TABLE 4. Decrease in Total Nutrient Export (%) as a Result of Simulated Load Reductions during Various Hydrological Conditions

	simulated reduction in nutrient loads		
river system	loads reduced 50% on days with $Q < 50$th percentile	loads reduced 25% on days with $Q < 75$th percentile	loads reduced 25% on days with $Q \geq 75$th percentile
	NO_3^--N		
Embarras	1.7	14.2	20.3
Kaskaskia	0.9	11.8	21.1
Sangamon	1.4	13.2	20.6
mean (1 SD)	1.3 (0.4)	13.0 (1.2)	20.7 (0.4)
	DRP		
Embarras	1.0	4.6	23.5
Kaskaskia	0.5	3.3	23.9
Sangamon	0.9	4.6	23.5
mean (1 SD)	0.8 (0.2)	4.2 (0.8)	23.6 (0.3)

benthic sediments where biological uptake and denitrification occur (*12*). Streams can switch from a nutrient retention and processing mode at low discharge, to a through-put mode at high discharge in which nutrient inputs from the landscape are transported downstream without biological processing (*25*). For the streams we examined, median discharge appeared to be the approximate transition point at which the streams switched from a state of nutrient retention to a state of nutrient export. For example, 97% of the NO_3^-−N export occurred above median discharge when denitrification in these streams has little effect on NO_3^-−N retention (*12*, *26*). The hydrology of the agricultural landscape in the upper Midwest has been greatly altered during the last 125 years by wetland drainage, stream channelization, and installation of under-field (tile) drainage systems (*16*). Thus the patterns we observed, although characteristic of the upper Midwest, may not represent landscapes with less altered hydrological regimes.

In addition to the association with high discharge, nutrient export from our sites occurred mainly during late winter through spring of each year. Elevated nutrient concentrations during spring are common in agricultural streams, and the present analysis clearly documents the extent to which nutrient export is generally confined to this time period. These patterns in the timing of nutrient export likely result from the seasonality of fertilizer application and the prevalence of tile drainage within the watersheds, both of which influence the timing and magnitude of nutrient loads (*23*). From 1994 to 2003, an average of 55% of the annual N fertilizer used in east-central Illinois was applied during autumn (*27*). This fertilizer is susceptible to nitrification and loss during late winter and spring as NO_3^-−N in drainage water. In intensively farmed areas, fertilizer use and disturbance from tillage appear to interact to produce large losses of NO_3^-−N through tiles (*7*), and evidence suggests that spring peaks in NO_3^-−N concentrations in the lower Mississippi River are a result of increased fertilizer use in the MRB during the past 50 years (*6*). Our analysis of watersheds in Illinois supports these conclusions.

Water entering the Embarras River through tile drains can at times have NO_3^-−N concentrations >30 mg L^{-1} and N input to the river from tiles can exceed 45 kg ha^{-1} yr^{-1} (*7*). Artificial drainage through under-field tiles is clearly a mechanism by which NO_3^-−N entered the streams we examined. Phosphorus also can enter streams through tile drainage in the Midwest (*11*), although this mechanism appears relatively more important in dry years with limited or no overland flow (Figure 4). Phosphorus transport processes are more difficult to separate in tile drained watersheds, because DRP inputs occur by either mechanism. The ratio of total P to DRP can be used to examine these flow paths, with larger ratios indicating more surface transport of particulate P (Table 2). This ratio was 2.9 in 2002, a wet year with significant overland flow and particulate P export, but only 1.8 in 2003, a dry year with no overland flow and much less particulate P export. However, regardless of the mechanism or the form of the P, extreme discharges were the driving factor for P export from the watersheds. For example, extreme discharges (≥ 90th percentile) accounted for >80% of the DRP and total P export, compared to 56% of the NO_3^-−N export.

Implications for Nutrient Loading to the Mississippi River. Reducing riverine export of NO_3^-−N from the Midwest is a key component to addressing hypoxia in the Gulf of Mexico (*6*). Likewise, reducing in-stream nutrient concentrations is an important step toward protecting water quality and aquatic life in the streams and rivers of the Midwest. In east-central Illinois, peak algal productivity and associated dissolved O_2 depletion occur most commonly during summer and early autumn when discharge is low (*26*, *28*). These same time periods, however, account for little of the annual NO_3^-−N or DRP export (Figures 2 and 3) due to low discharge and high nutrient uptake in the streams. There is therefore a temporal separation between periods of poor water quality and periods of high nutrient loads and export. Nutrient export from other areas of the Midwest also occurs mainly during high discharge (*22*, *29*) and we conclude that concerns about nutrient loading to the Mississippi River and Gulf of Mexico (*6*) must be addressed by reducing nutrient export from the Midwest during times of high discharge. The simulations support this conclusion, showing that nutrient export from the watersheds could be reduced only by reducing in-stream loads during high discharge (Table 4).

Efforts are underway to restore and protect local water quality using the TMDL approach and the development of nutrient criteria (*30*). Local water quality in the Midwest and nutrient loading to the Gulf of Mexico are both nutrient-related problems, but the TMDL approach, with its current focus on periods of low discharge, is not conducive to reducing nutrient loads at times relevant to affecting water quality in the Mississippi River and hypoxia in the Gulf of Mexico. However, our analysis indicates that if TMDLs in the Midwest are directed at reducing nutrient loads during periods of high discharge in late winter and spring, such efforts could reduce nutrient export from agricultural watersheds to the Mississippi River.

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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 $\mu g\ L^{-1}$). 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*.

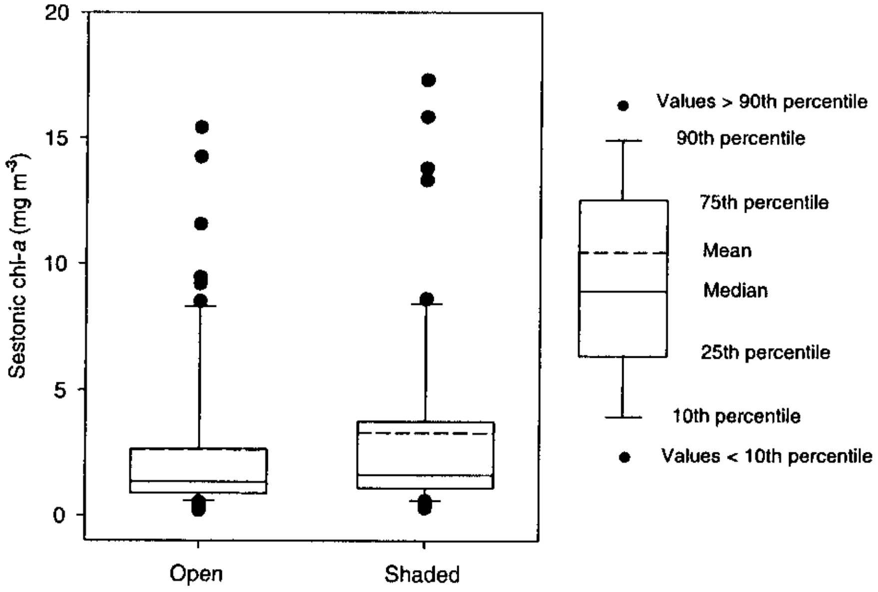

Fig. 1. Box plot of sestonic chl-*a* concentrations from March through November 2004 in three open-canopy (n = 75) and two shaded (n = 51) east-central Illinois agricultural streams.

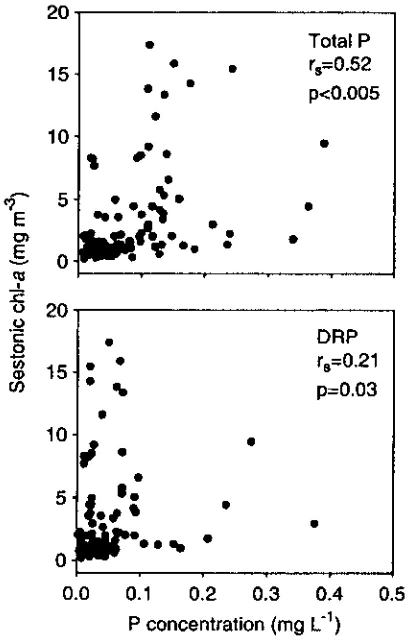

Fig. 2. Correlations between sestonic chl-*a* and (upper) total P and (lower) DRP (dissolved reactive P) from March through November 2004 in agricultural streams, excluding Site SFV.

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.

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*.

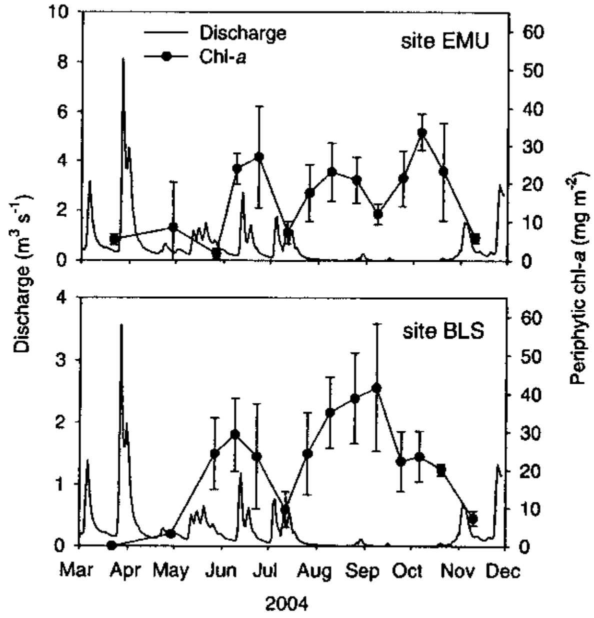

Fig. 3. Estimated discharge and mean periphytic chl-*a* (±1 SE) patterns for two open-canopy agricultural streams. Note difference in scale for discharge.

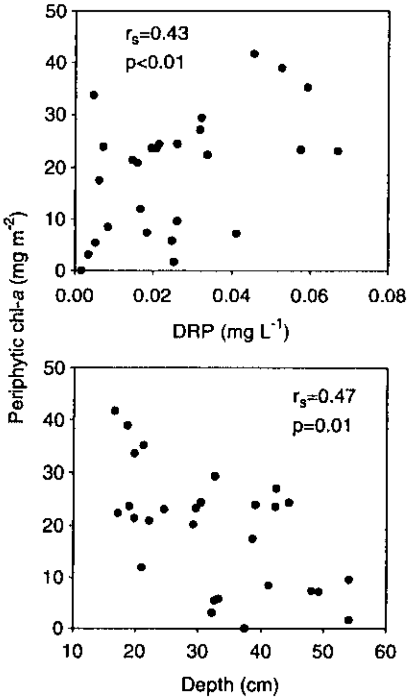

Fig. 4. Correlations between periphytic chlorophyll-*a* and (upper) DRP and (lower) stream depth for two open-canopy agricultural streams (sites EMU and BLS) from March through November 2004.

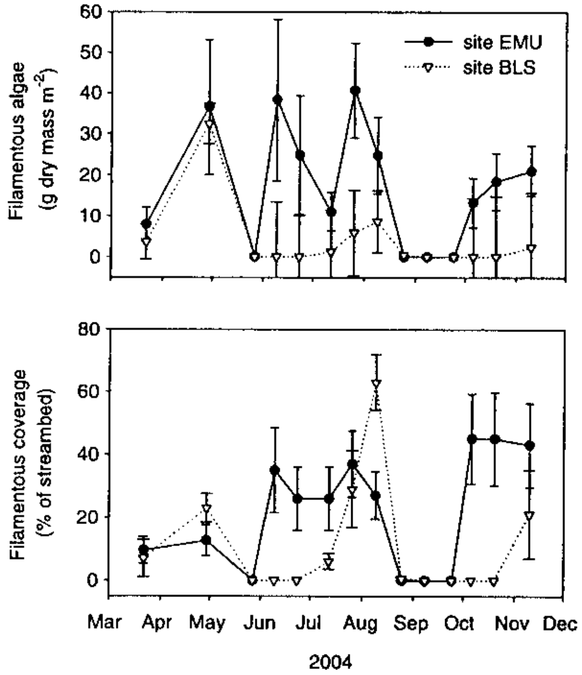

Fig. 5. Mean (±1 SE) abundance of filamentous algae as (upper) biomass and (lower) streambed coverage in two open-canopy agricultural streams.

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.

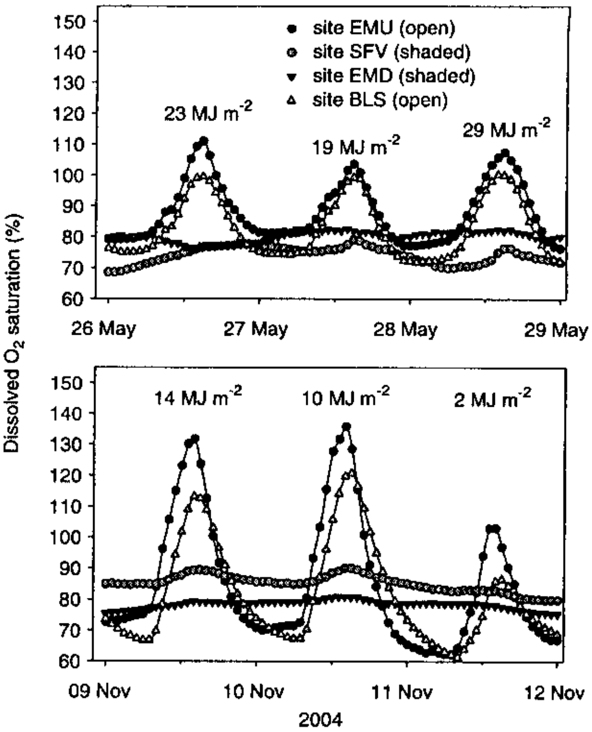

Fig. 6. Dissolved O_2 patterns during (upper) late May and (lower) early November 2004 in two open-canopy and two shaded agricultural streams. Daily solar radiation values were measured at Site BLS (open canopy) and do not reflect available light at the water surface for the shaded sites.

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-

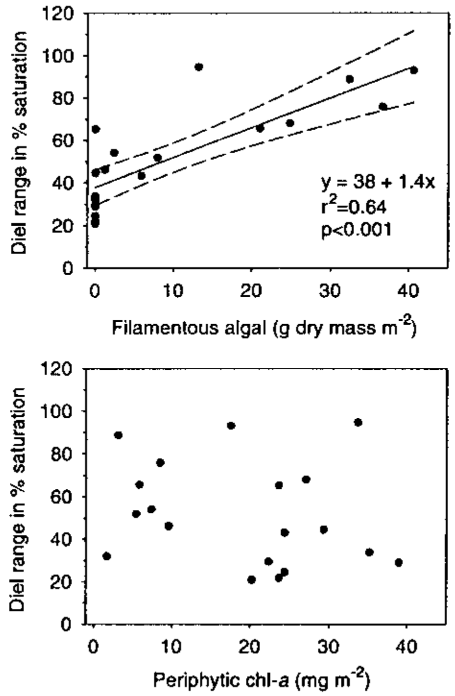

Fig. 7. Relationship between the diel range in dissolved O_2 saturation and the biomass of (upper) filamentous algae or (lower) periphytic chlorophyll-*a* in two open-canopy agricultural streams during March through November 2004.

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.

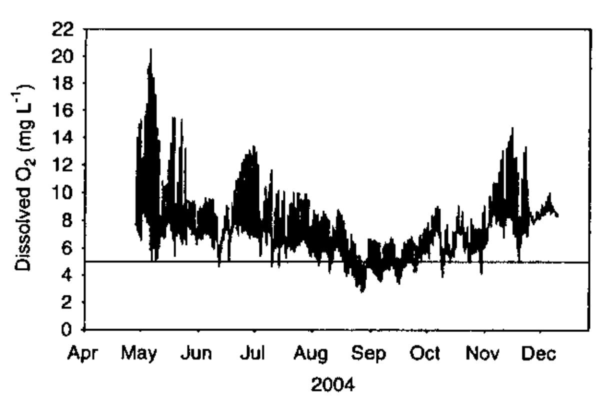

Fig. 8. Continuous dissolved O_2 concentration in an open-canopy agricultural stream (Site BLS) from late April through early December 2004. Horizontal line indicates the 5 mg L^{-1} Illinois dissolved O_2 standard at the time of the study.

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 $\mu g\ L^{-1}$ and an average concentration of total N below 350 $\mu g\ L^{-1}$. 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 algae 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), however, 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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Primary Research Paper

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

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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).

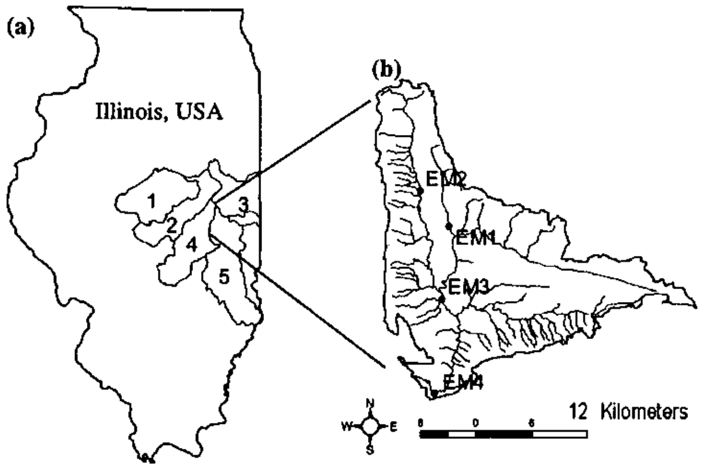

Figure 1. (a) Map showing the major river basins of east-central Illinois, USA. Study sites were located throughout the headwaters of these basins (see Table 1 for sites and basin names). (b) Detail of the locations of the intensively sampled sites in the headwaters of the Embarras River.

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 QuikChem© 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, 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.

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	

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 nonparametric 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.

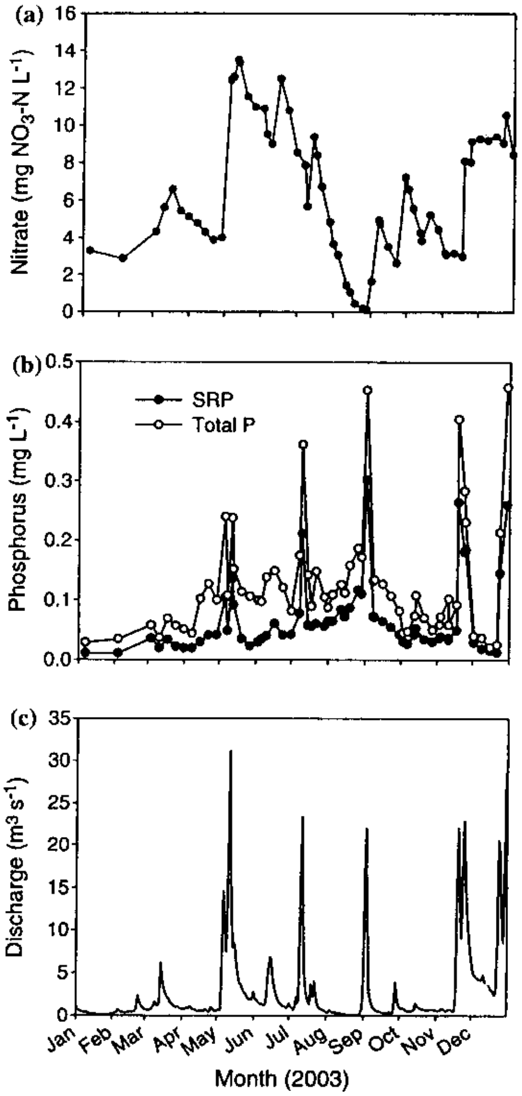

Figure 2. Stream water concentrations of (a) NO_3-N, (b) SRP and total P, and (c) daily discharge for the Embarras River at the EM4 site during 2003.

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 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)

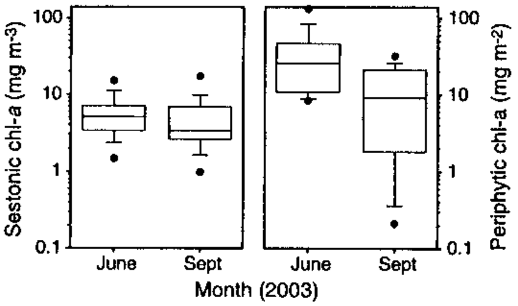

Figure 3. Box plots of sestonic and periphytic chl-*a* from 18 sites in east-central Illinois, USA, during June and September 2003. Box plots show 10th, 25th, 50th, 75th, and 90th percentiles as horizontal lines; values outside the 10th and 90th percentiles are shown as solid circles.

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

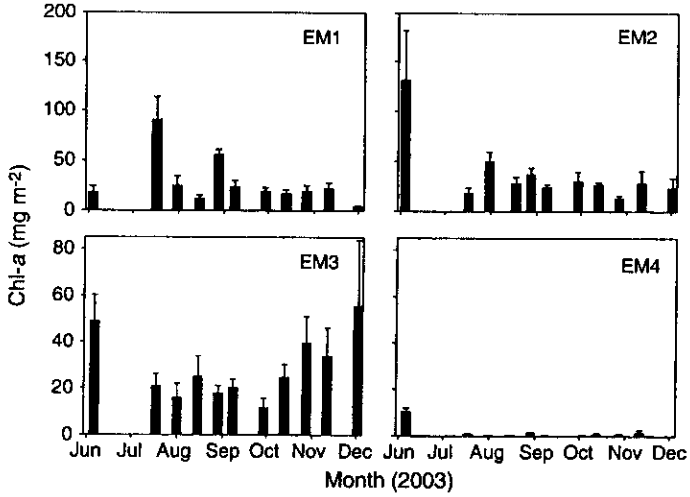

Figure 4. Mean (+1 standard error) periphytic chl-*a* concentrations on natural substrata at the Embarras River sites from June to December 2003. Note difference in scale between upper and lower graphs. EM1 and EM2 are non-shaded sites, EM3 and EM4 are shaded sites (see Table 1 for full description).

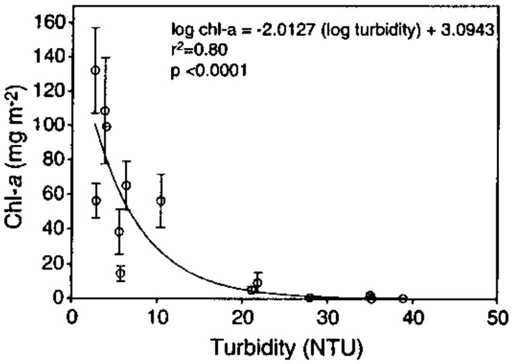

Figure 5. Relationship between mean (+1 standard error) chl-*a* on unglazed ceramic tiles and stream water turbidity for 14 sites in east-central Illinois, USA, during September 2003. The equation is based on a log–log relationship between periphytic chl-*a* and turbidity.

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,

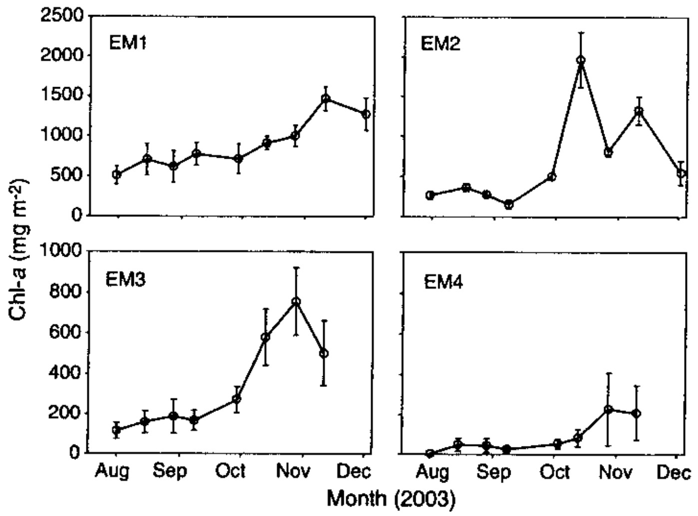

Figure 6. Mean (+1 standard error) chl-*a* concentrations on unglazed ceramic tiles at the Embarras River sites from August to December 2003. Note difference in scale between upper and lower graphs. EM1 and EM2 are non-shaded sites, EM3 and EM4 are shaded sites (see Table 1 for full description).

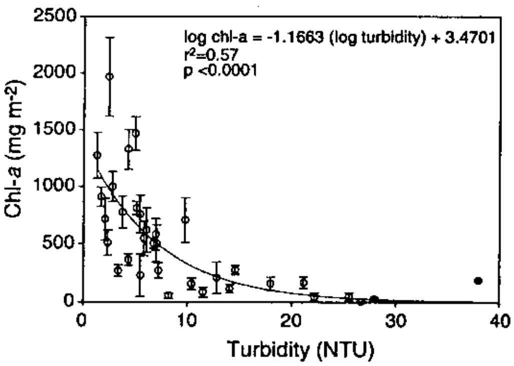

Figure 7. Relationship between mean (+1 standard error) chl-*a* on unglazed ceramic tiles and stream water turbidity for the Embarras River sites during August to December 2003. The equation is based on a log–log relationship between periphytic chl-*a* and turbidity.

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

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Garvey – October 2006 – Exhibit 3

Analysis of Grab and Continuous Data.

Analyzed: September 2006 by Jim Garvey

Data were procured from IDNR and IEPA during spring 2006.

1994-2003 grab data derive from:

APPLE RIVER
AUXSABLE CREEK
BAKER CREEK
BEAR CREEK
BEAVER CREEK
BECK CREEK
BIG BUREAU CREEK
BIG CREEK
BIG GRAND PIERRE CREEK
BIG ROCK CREEK
BLACKBERRY CREEK
BLUEGRASS CREEK
BOONE CREEK
BROUILLETTS CREEK
BRUSHY FORK
BUCK CREEK
BUFFALO CREEK
BURTON CREEK
CAMP CREEK
CAMP RUN
CARROLL CREEK
CEDAR CREEK
CLEAR CREEK
COAL CREEK
COON CREEK
COOPERS DEFEAT CR
COPPERAS CREEK
COURT CREEK
COX CREEK
CRABAPPLE CREEK
CRANE CREEK
CRANE GROVE CREEK
CROOKED CREEK
DEER CREEK
DES PLAINES RIVER
HAW CREEK
HAYES CREEK
HENDERSON CREEK
HENLINE CREEK
HERMON CREEK
HICKORY CREEK
HICKORY GROVE DITCH
HORSE CREEK
HURRICANE CREEK
HUTCHINS CREEK
ILLINOIS RIVER
INDIAN CREEK
IROQUOIS RIVER
JACK CREEK
JACKSON CREEK
JOES CREEK
JOHNNY RUN
JORDAN CREEK
KANKAKEE RIVER
KICKAPOO CREEK
KILLBUCK CREEK
KISHWAUKEE RIVER
KYTE RIVER
LAHARPE CREEK
LAKE FORK
LAMOINE RIVER
LANGAN CREEK
LEAF RIVER
LITTLE EMBARRAS R
LITTLE INDIAN CREEK
LITTLE KICKAPOO CREEK NORTH
LITTLE MACKINAW R
LITTLE ROCK CREEK
LITTLE SALINE RIVER
LITTLE VERMILION RIVER
NINEMILE CREEK
NIPPERSINK CREEK
NORTH BRANCH
NORTH HENDERSON CREEK
OPOSSUM CREEK
OTTER CREEK
PANTHER CREEK
PINE CREEK
POLECAT CREEK
POPE CREEK
POPLAR CREEK
PRAIRIE CREEK
PRAIRIE DULONG CREEK
RACCOON CREEK
RICHLAND CREEK
RILEY CREEK
ROBINSON CREEK
ROCK CREEK
ROCK RIVER
ROCK RUN
ROOKS CREEK
RUSH CREEK
S BR KISHWAUKEE R
S FK APPLE RIVER
S FK BEAR CREEK
S FK SALINE RIVER
S FK VERMILION RIVER
S KINNIKINNIK CREEK
SALT CREEK
SALT FORK VERMILION R
SANGAMON RIVER
SHOAL CREEK
SILVER CREEK
SIX MILE CREEK
SMITH CREEK
WALNUT CREEK
WEST BUREAU CREEK
WEST OKAW RIVER
WILLOW CREEK
YELLOW CREEK

Garvey – October 2006 – Exhibit 3

DICKERSON SLOUGH
DRUMMER CREEK
DUPAGE RIVER
DUTCH CREEK
E BR S BR KISHWAUKEE RIVER
E FK GALENA RIVER
E FK LA MOINE RIVER
E FK MAZON RIVER
E FK SHOAL CREEK
E FK SPOON RIVER
EAGLE CREEK
EDWARDS RIVER
ELKHORN CREEK
EMBARRAS RIVER
EXLINE SLOUGH
FERSON CREEK
FORKED CREEK
FOX RIVER
FRANKLIN CREEK
FRIENDS CREEK
FURNACE CREEK
GALENA RIVER
GREEN RIVER
GRINDSTONE CREEK
HADLEY CREEK
LONG POINT CREEK
LUSK CREEK
MACKINAW RIVER
MACOUPIN CREEK
MADDEN CREEK
MANHATTAN CREEK
MAZON RIVER
MCCRANEY CREEK
MIDDLE BRANCH
MIDDLE FORK SHOAL CR
MIDDLE FORK SUGAR CR
MIDDLE FORK VERMILION R
MILL CREEK
MISSISSIPPI RIVER N
MISSISSIPPI RIVER SC
MITCHELL CREEK
MOSQUITO CREEK
MUD CREEK EAST
MUDDY CREEK-NORTH
N FK EMBARRAS RIVER
N FK KENT CREEK
N FK LAKE FORK
N FK SALT CREEK
N FK VERMILION RIVER
NETTLE CREEK
SOUTH BRANCH-EAST
SOUTH HENDERSON CREEK
SPOON RIVER
SPRING BROOK
STEVENS CREEK
STILLMAN CREEK
SUGAR CREEK
SUGAR CREEK-NORTH
SUGAR RIVER
SUMNER CREEK
SWAB RUN
TEN MILE CREEK
TIMBER CREEK
TRIM CREEK
TURKEY CREEK
TYLER CREEK
U-TRIB MACKINAW R BRAY CREEK
U-TRIB MACKINAW R FROG ALLEY
VERMILION RIVER
W BR DUPAGE RIVER
W BR LAMARSH CREEK
W BR PANTHER CREEK
W FK MAZON RIVER
W FK SHOAL CREEK
WABASH RIVER

2004-2005 continuous data derive from:

Big Creek, Big Grand Pierre Creek, Edwards River, Exline Slough, Hickory Grove Ditch, Indian Creek, Lake Fork, Rock Creek, Sangamon River, South Henderson Creek, Spoon River, Sugar Creek

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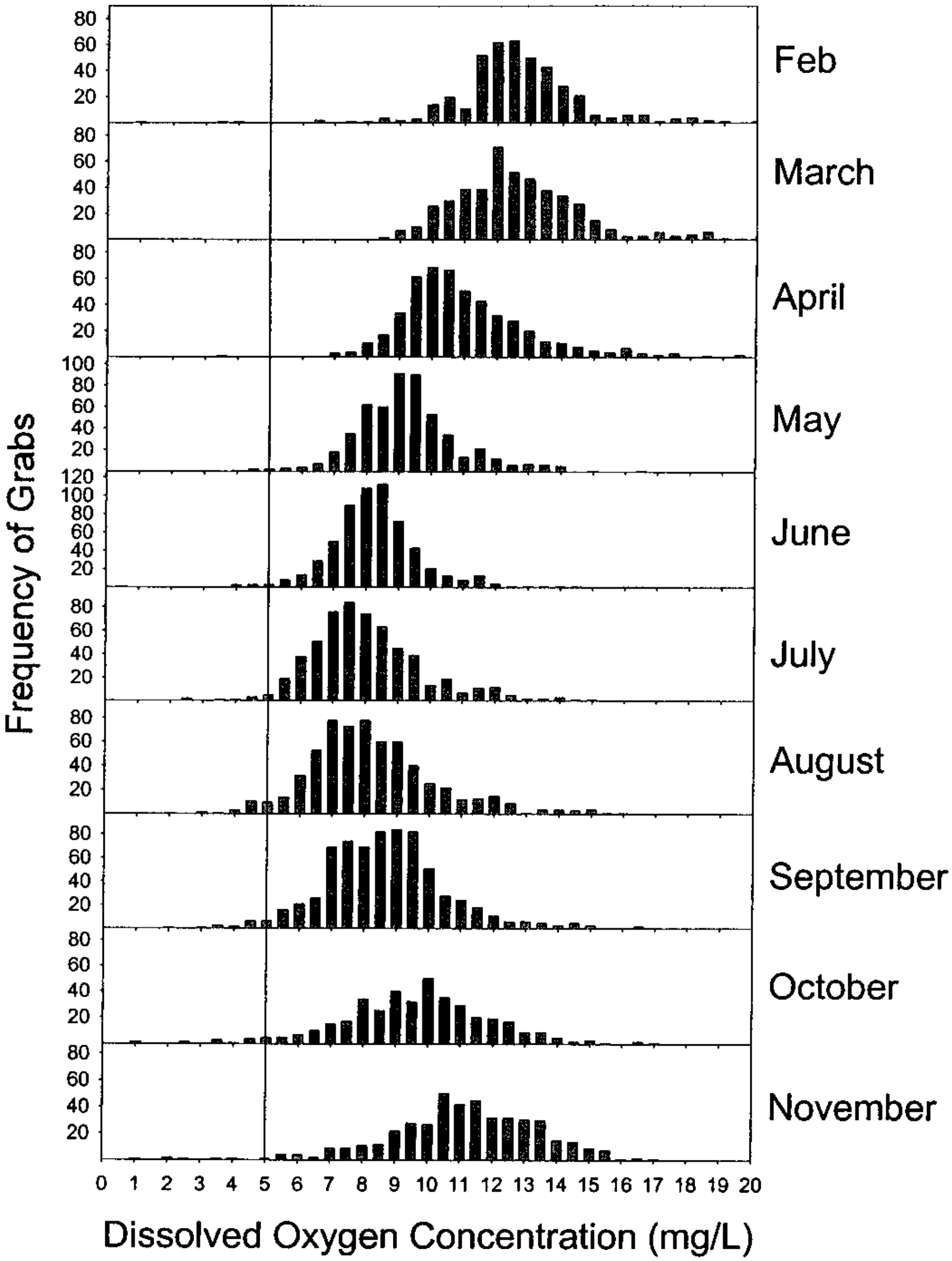

Figure 1. Frequency of grab data from several Illinois streams that are being considered for enhanced dissolved oxygen tier standards.

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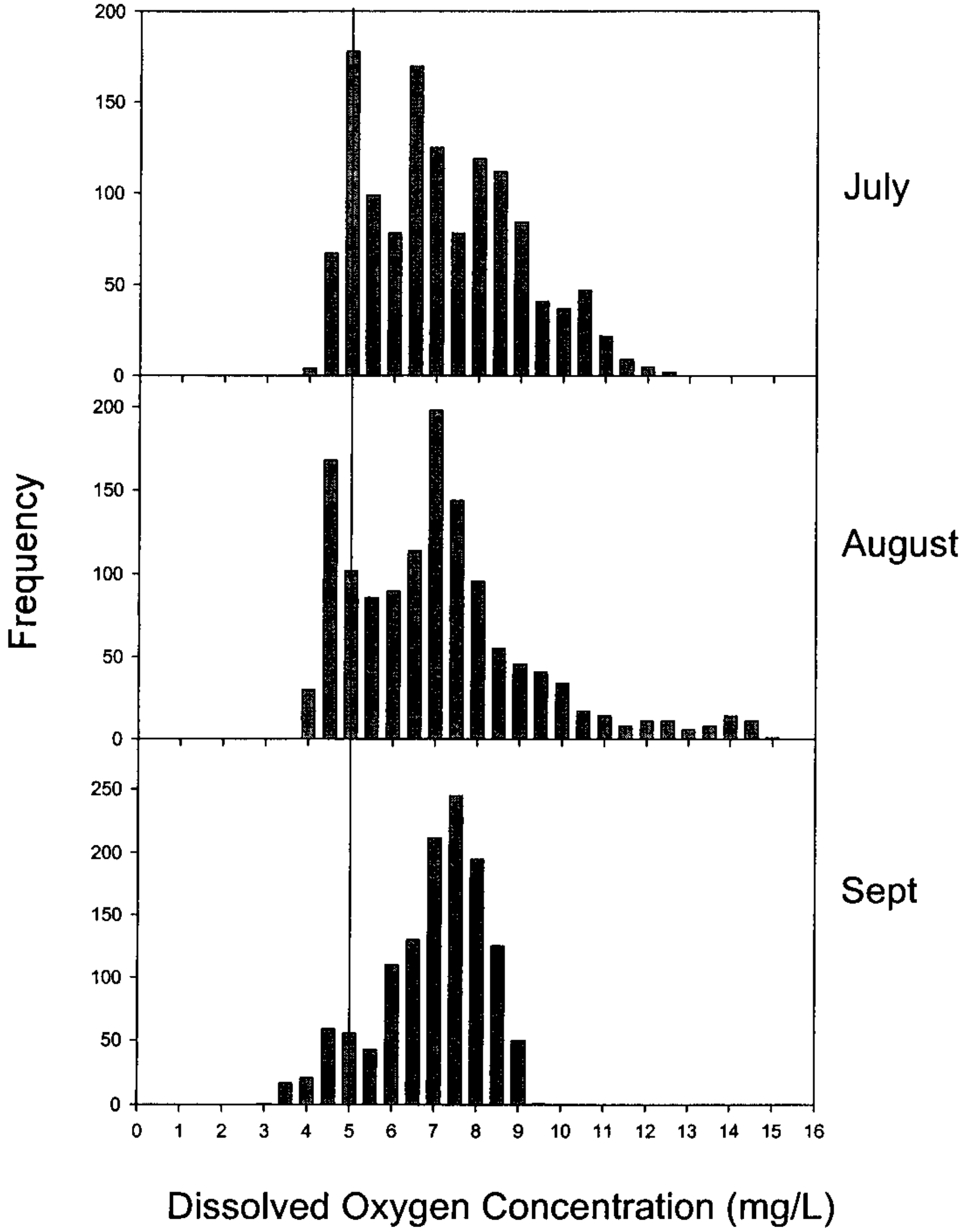
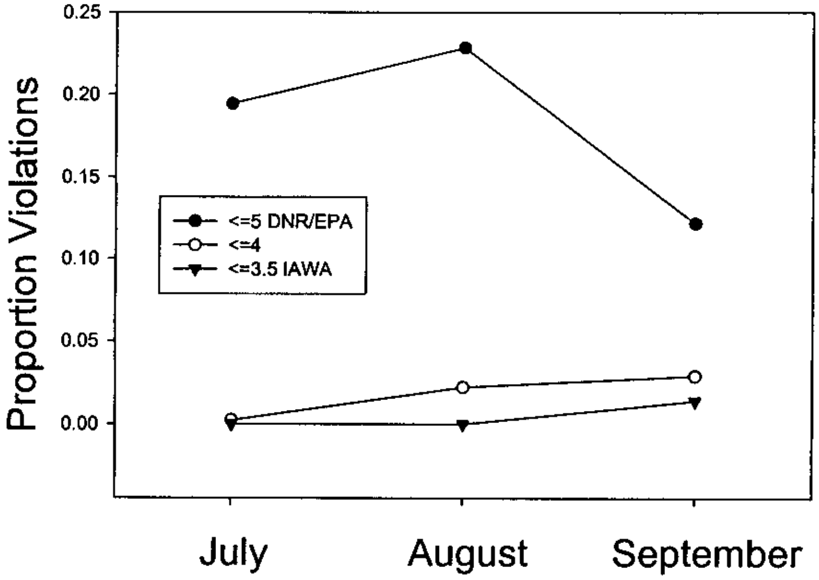


Figure 2. Frequency of dissolved oxygen observations collected with continuous loggers in selected streams that are being considered for the enhanced oxygen tier.

Garvey – October 2006 – Exhibit 3



Proportion of potential violations in continuous data with three proposed daily minima from proposed enhanced DO sites in Illinois. Data from Roy Smogor, IL EPA.

Figure 3. Proportion of observations below 5 mg/L, 4 mg/L or 3.5 mg/L during the months that continuous dissolved oxygen data were collected from streams considered for enhanced oxygen tier.

Garvey – October 2006 – Exhibit 3

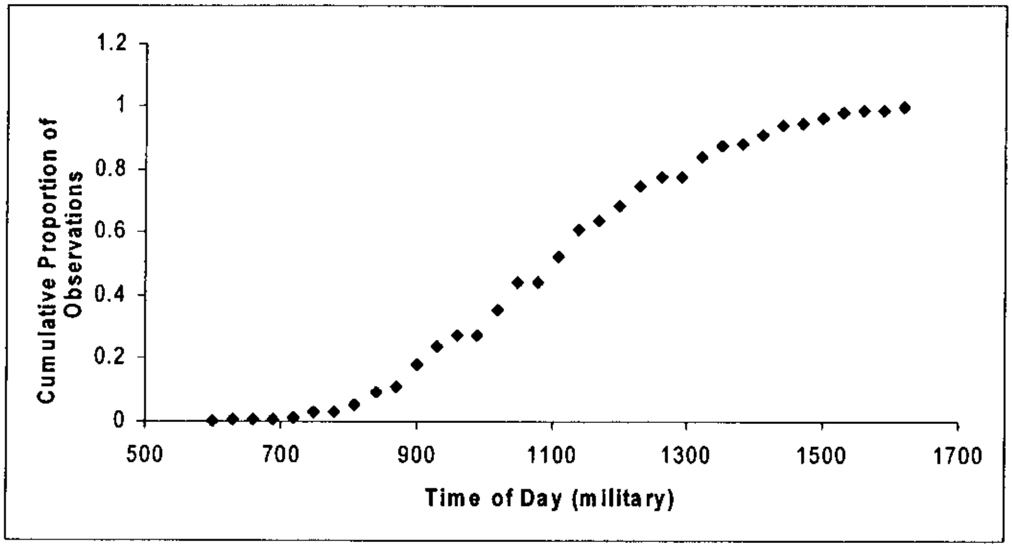


Figure 4. The cumulative proportion of data collection times for grab samples displayed in Figure 1. About 50% of the data were collected before 11 AM. All data were collected by 1700 hours.

Garvey - October 2006 - Exhibit 4

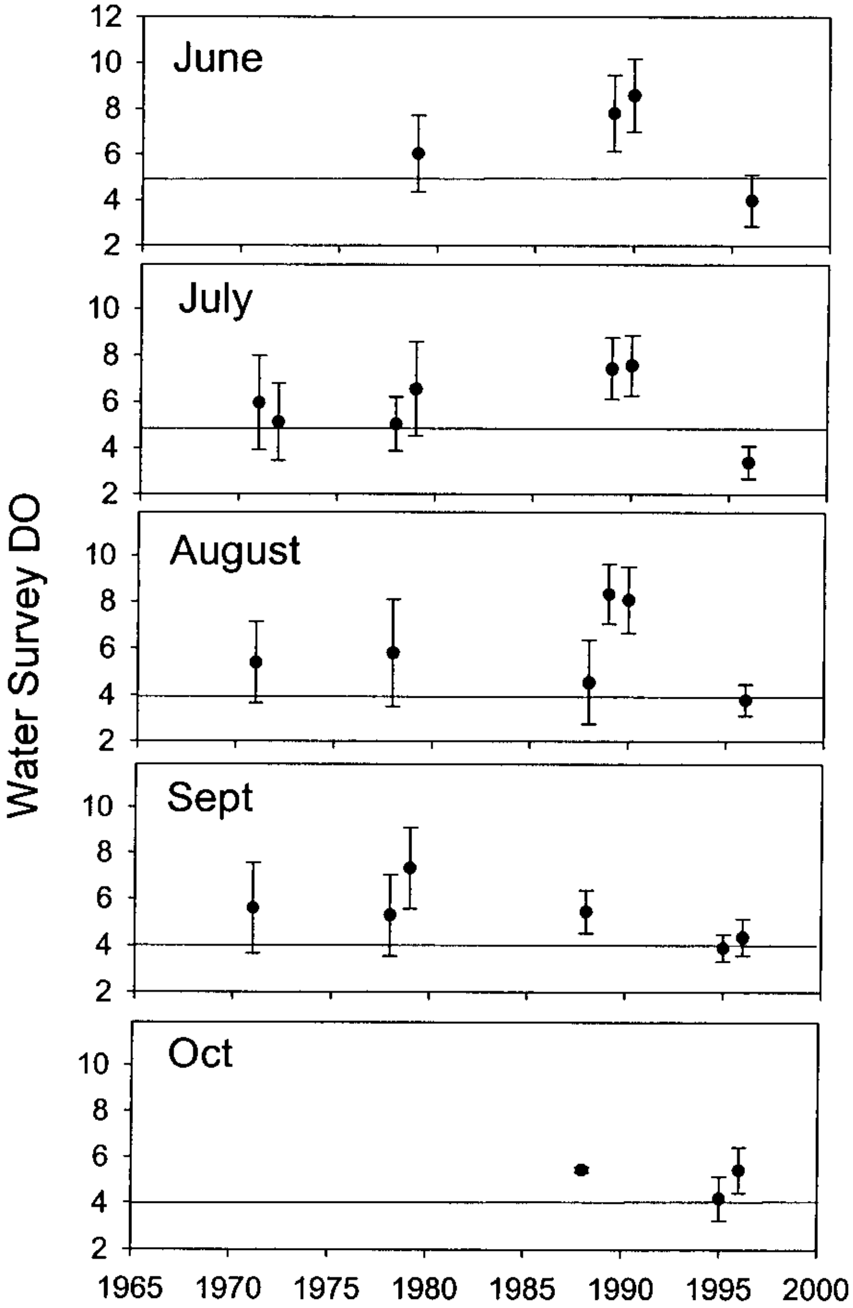


Figure 1. Mean ± standard deviation dissolved oxygen concentrations collected across multiple stream sites in Illinois during 1971 through 1996. Average sample size per data point is 400 observations. All observations were restricted to grabs taken either at the surface or mid water. No bottom values are included.

[redacted]

ISWS Illinois Waterway Studies Referencing Dissolved Oxygen Data

Subject of Study	*Study Period*	*Tabulated*	*ISWS Rpts*	*Report Titles and Authors*
LaGrange Pool Water Quality	1965-67		RI-64	**Dissolved Oxygen Resources and Waste Assimilative Capacity of the LaGrange Pool, Illinois River** Butts, Thomas A., Donald H. Schnepper, and Ralph L. Evans, 1970
Upper Waterway Water Quality	1971-72	Yes	RI-79	**Water Quality Features of the Upper Illinois Waterway** Butts, Thomas A., Ralph L. Evans, and Shun Dar Lin, 1975
LaGrange Pool Water Quality	1977-79		CR-260	**Water Quality Assessment and Waste Assimilative Analysis of the LaGrange Pool, Illinois River** Butts, Thomas A., Donald P. Roseboom, Thomas E. Hill, Shundar Lin, Davis B. Beuscher, Richard M. Twait, and Ralph L. Evans, 1981
Lake Michigan Diversion/ Water Quality	1982	Yes	CR-324	**The Effects of Lake Michigan Discretionary Diversion Strategies on Illinois Waterway Dissolved Oxygen Resources** Butts, Thomas A., Donald H. Schnepper, and Krishan P. Singh, 1983
Peoria Pool Water Quality	1986	Yes	no rpt	
Side Stream Elevated Pool Aeration (SEPA) Stations	1987	Yes	CR-452	**Development of Design Criteria for Sidestream Elevated Pool Aeration Stations** Butts, Thomas A., 1988
Commercial Navigation/ Water Quality	1987-88	Yes	RR-122	**Impacts of Commercial Navigation on Water Quality in the Illinois River Channel** Butts, Thomas A., and Dana B. Shackleford, 1992
TARP Effects on Water Quality	1988-89	Yes	CR-256, CR-256a	**Reduction in Peak Flows and Improvements in Water Quality in the Illinois Waterway Downstream of Lockport Dam Due to Implementation of Phases I and II of TARP** Durgunoglu, Ali, Krishan P. Singh, Thomas A. Butts, and Dana B. Shackleford, 1992 **Appendices for Reduction in Peak Flows and Improvements in Water Quality in the Illinois Waterway Downstream of Lockport Due to Implementation of Phases I and II of TARP** Butts, Thomas A., and Dana B. Shackleford, 1992
QUAL-2E Water Quality Modeling (above Lockport)	1989	Yes	no rpt	
QUAL-2E Water Quality Modeling (LM to Chillicothe)	1990-91	Yes	no rpt	
Illinois River @ Peoria Monitoring	1966-94	Yes	no rpt	
LaGrange Pool	1973	Yes	no rpt	
Effects of SEPA on Water Quality	1995-97	Yes	CR-200-2, CR-653	**Sidestream Elevated Pool Aeration (SEPA) Stations: Effects on In-stream Dissolved Oxygen** Butts, Thomas A., Dana B. Shackleford, and Thomas R. Bergerhouse, 2000 **Evaluation of Reaeration Efficiencies of Sidestream Elevated Pool Aeration (SEPA) Stations** Butts, Thomas A., Dana B. Shackleford, and Thomas R. Bergerhouse, 1999
Water Chemistry of Illinois Waterway	1978-79	Yes	C-147	**Water Chemistry of the Illinois Waterway** Kothandaraman, Veerasamy, Robert A. Sinclair, and Ralph L. Evans, 1981
Other ISWS Authored Reports Referencing Dissolved Oxygen Data				
Upper Sangamon	1990		CR-513	**Incorporation of Dissolved Oxygen in Aquatic Habitat Assessment for the Upper Sangamon River** Broeren, S McConkey, Thomas Butts, and Krishan Singh, 1991
Lower Sangamon	1991		CR-571	**Water Quality and Habitat Suitability Assesment: Sangamon River between Decatur and Petersburg** Larson, R.B., Thomas Butts, and Krishan P. Singh, 1994
Fox River St Charles Pool	1993-94		CR-586	**Considerations in Water Use Planning for the Fox River** Singh, Krishan, Thomas Butts, Vernon Knapp, Dana Shackleford, and Robert Larson, 1995
Effects of Channel Dams on DO Northeastern IL	1976-77		C-132	**Effects of Channel Dams on Dissolved Oxygen Concentrations in Northeastern Illinois Streams** Butts, Thomas and R. Evans, 1978

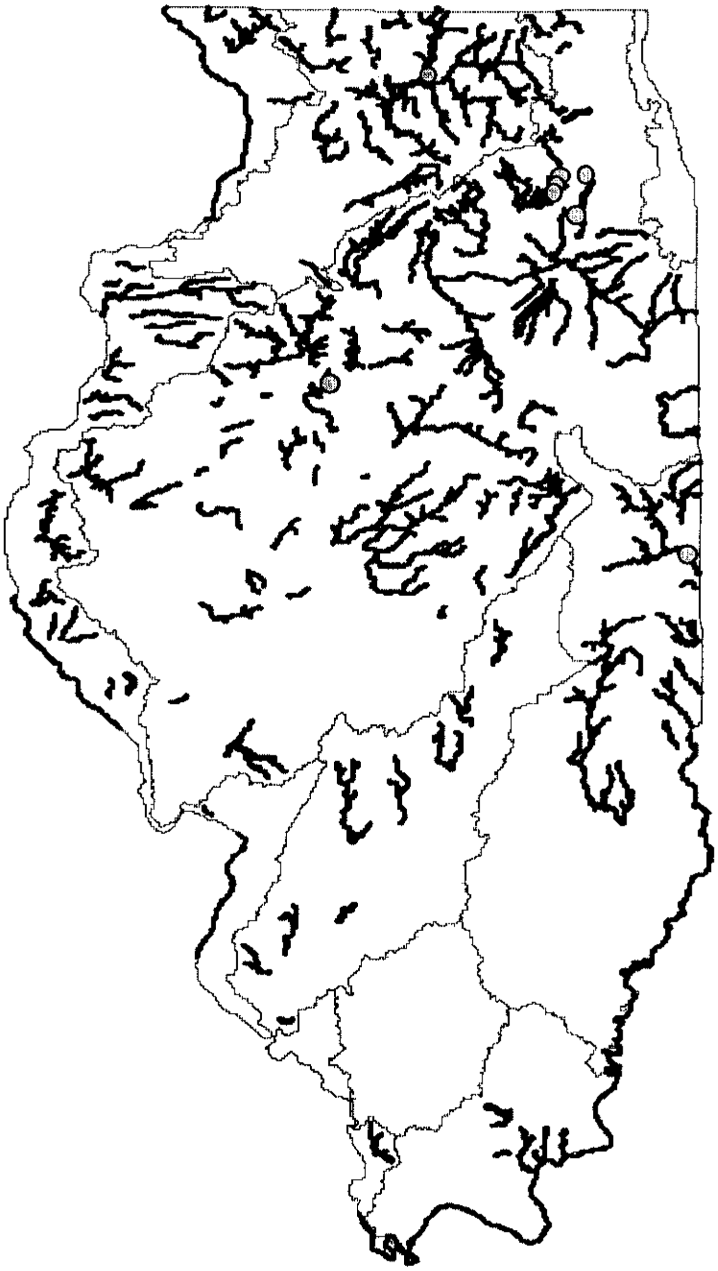


Figure 1. Map of streams to receive proposed enhanced dissolved oxygen tier (blue) and IAWA sampling sites (green circles). Reaches highlighted in red are enhanced waters that are 303d listed with low oxygen as a probable cause of impairment.

[redacted]

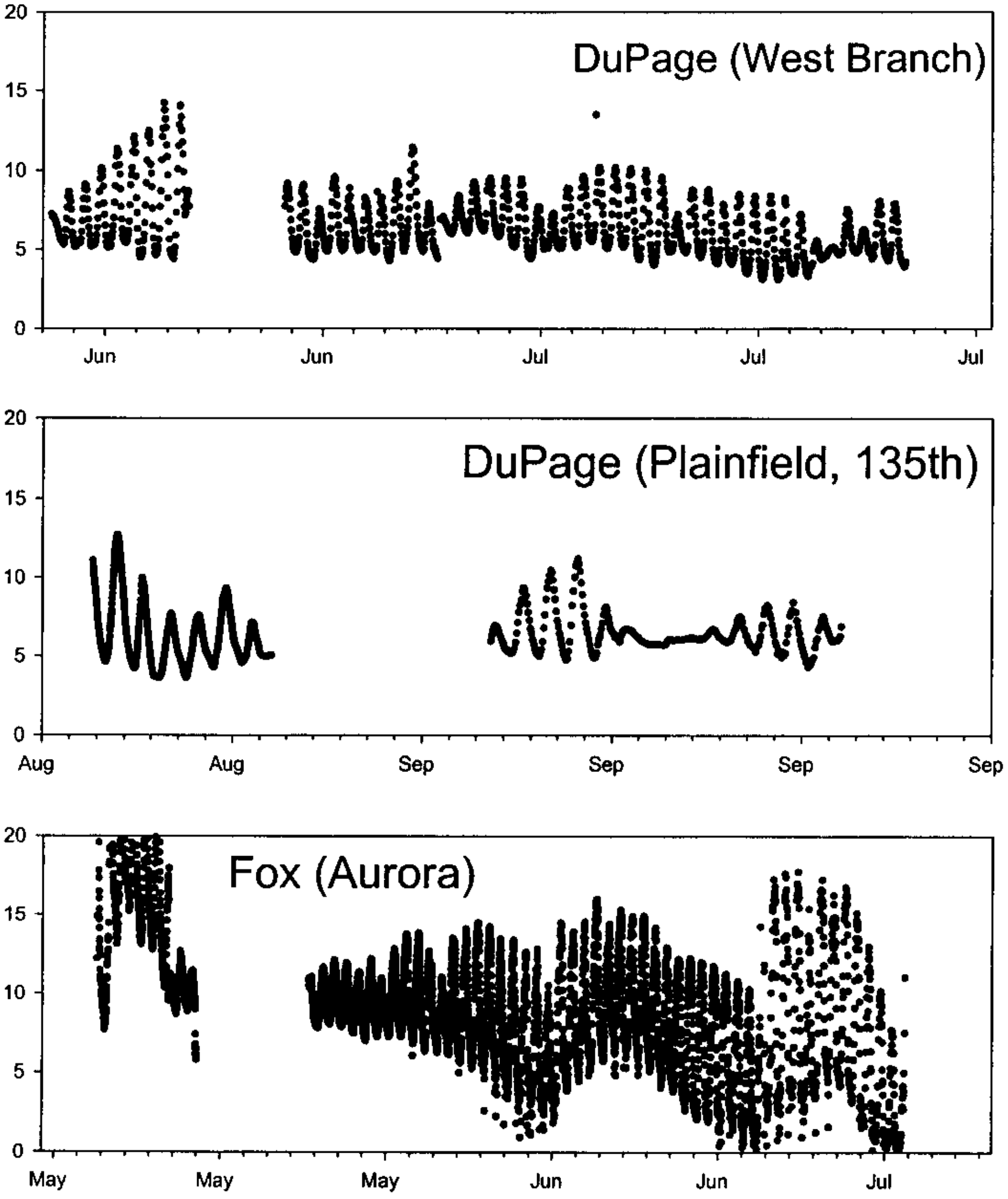


Figure 2. Daily changes in oxygen concentrations collected by IAWA during 2005 (Fox River) and 2006 (Du Page River).

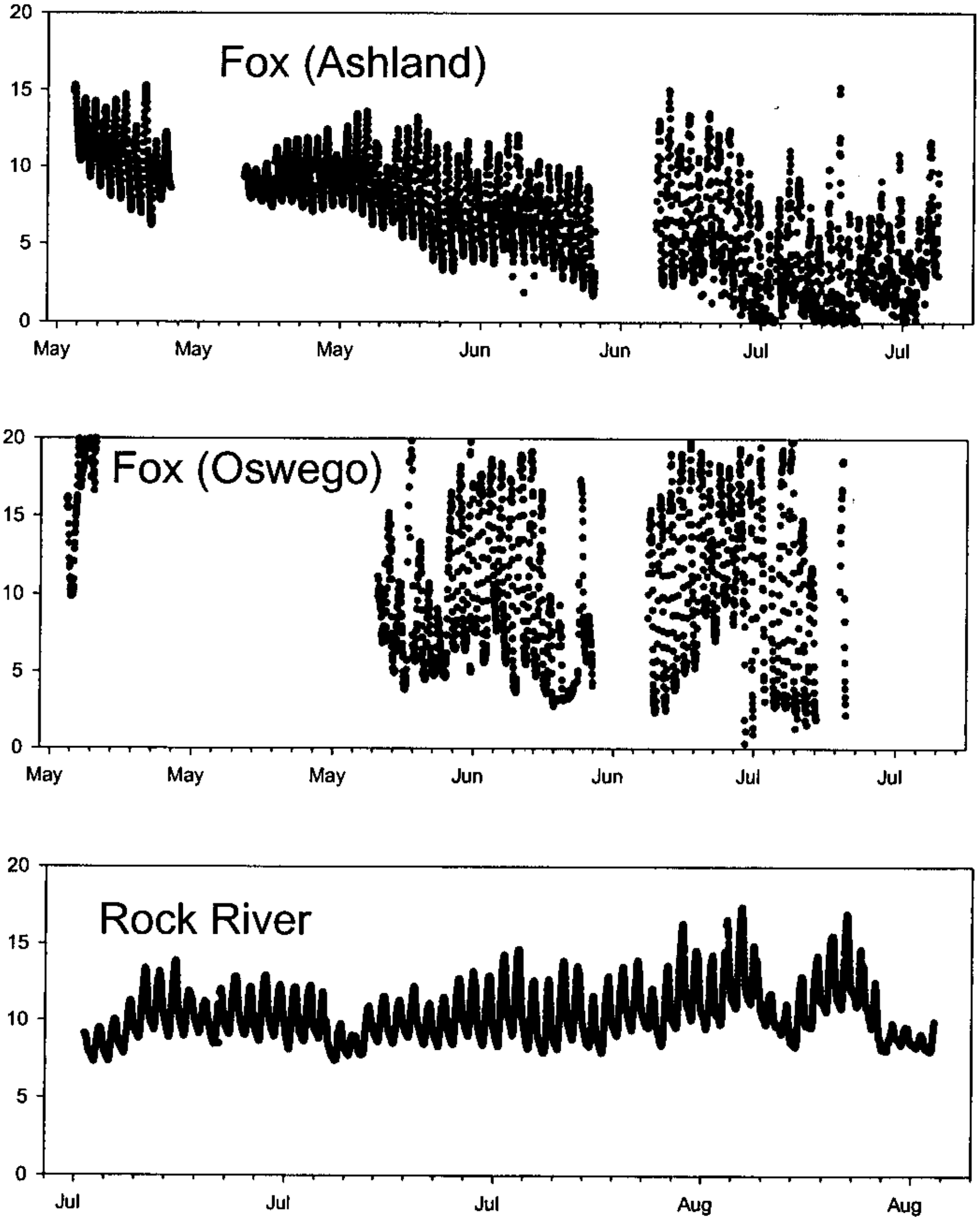


Figure 3. Daily changes in oxygen concentrations collected by IAWA during 2005 (Fox River) and 2006 (Rock River).

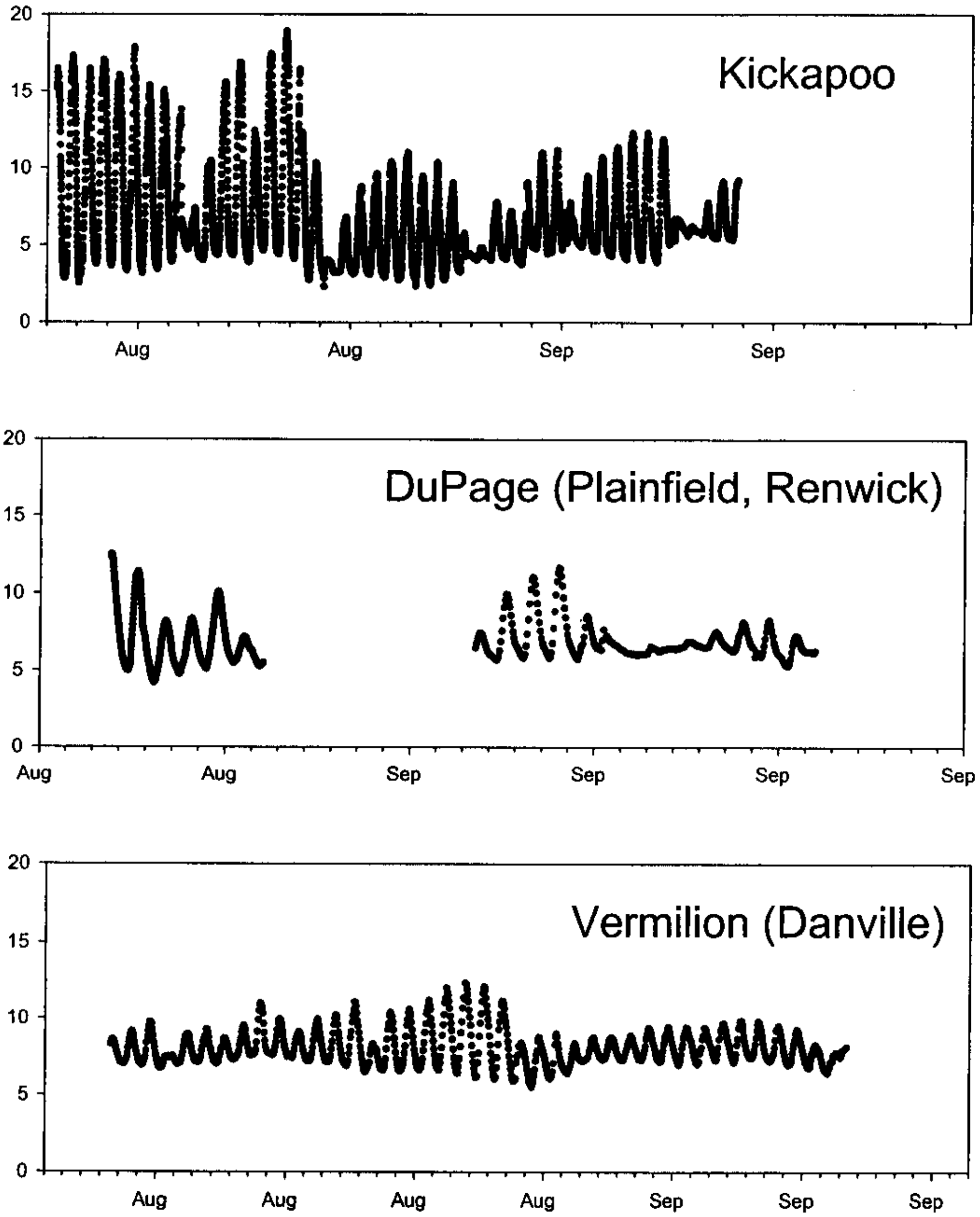


Figure 4. Daily changes in oxygen concentrations collected by IAWA during 2006.

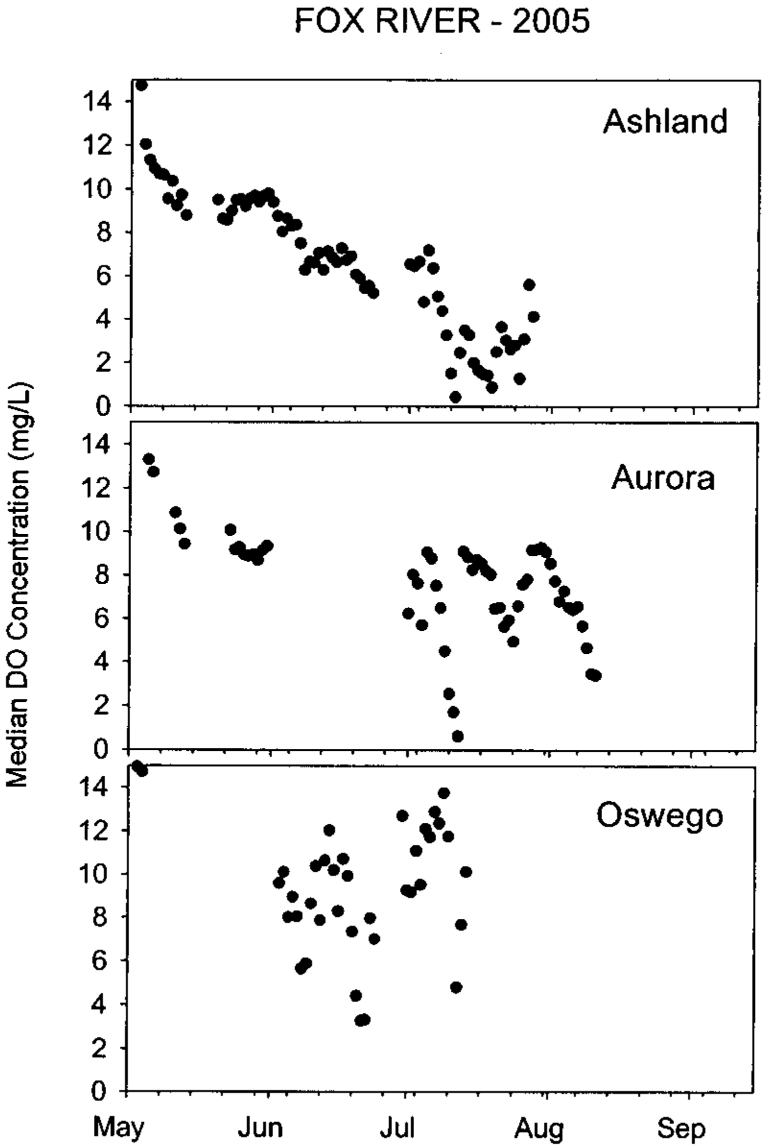


Figure 5. Median daily dissolved oxygen concentrations collected by IAWA during 2005.

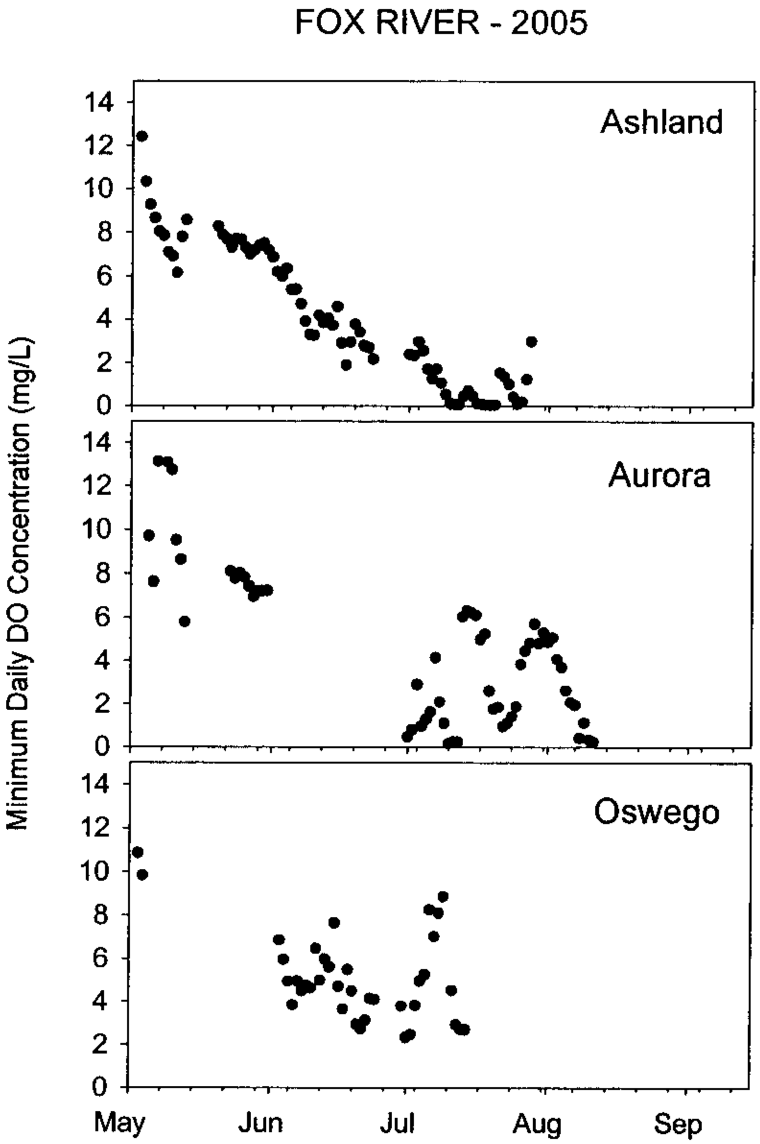


Figure 6. Minimum daily dissolved oxygen concentrations collected by IAWA during 2005.

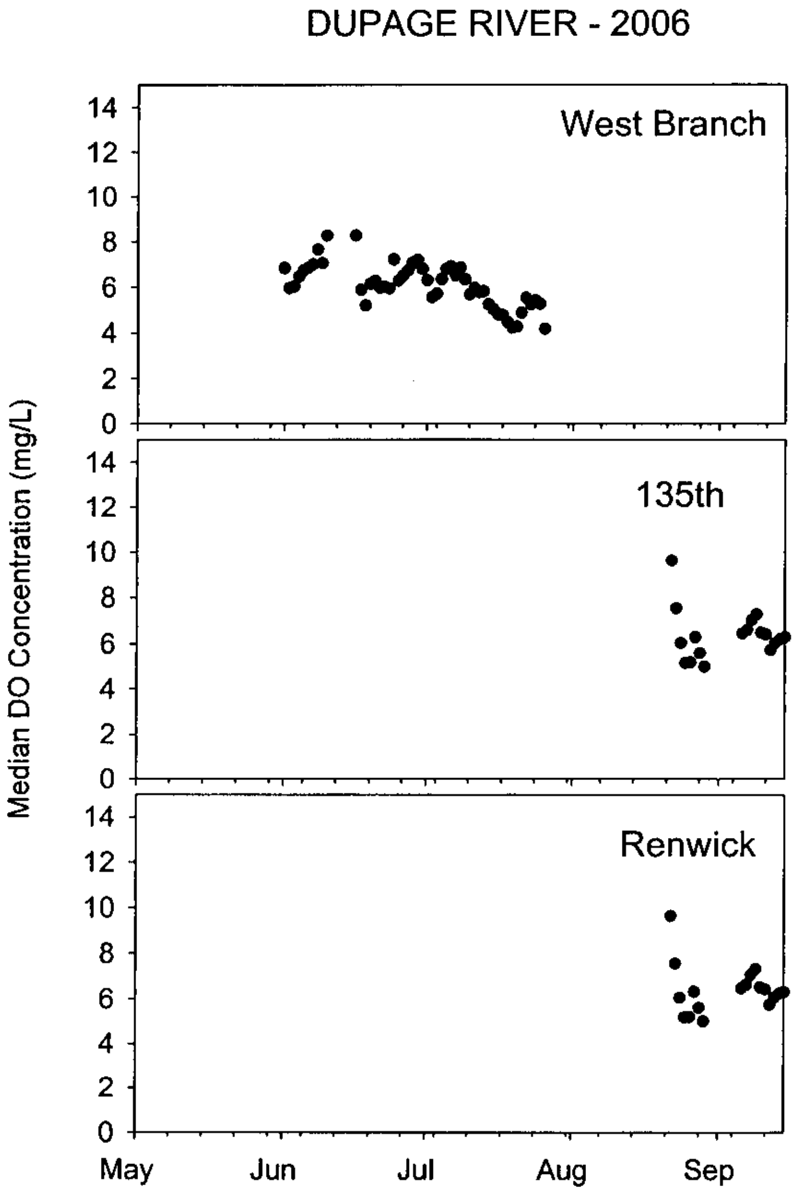


Figure 7. Median daily dissolved oxygen concentrations collected by IAWA during 2006.

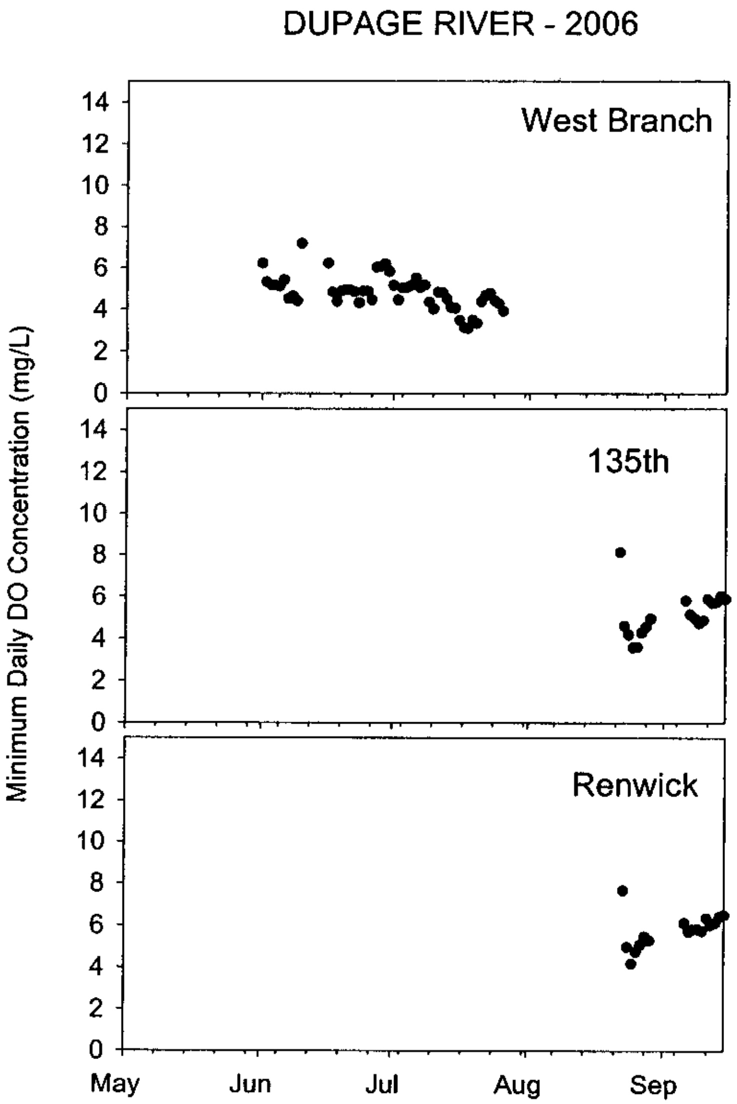


Figure 8. Minimum daily dissolved oxygen concentrations collected by IAWA during 2006.

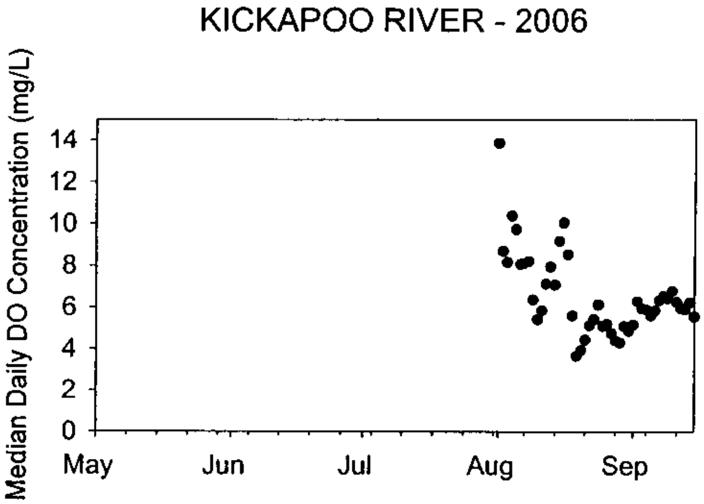


Figure 9. Median daily dissolved oxygen concentrations collected by IAWA during 2006.

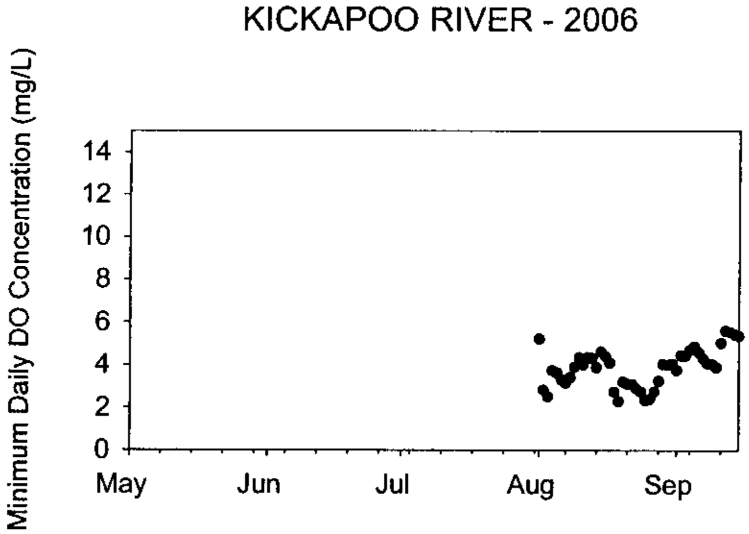


Figure 10. Minimum daily dissolved oxygen concentrations collected by IAWA during 2006.

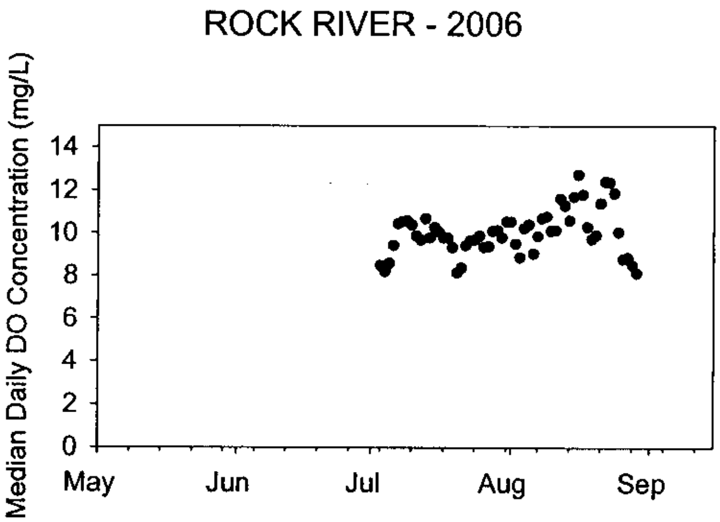


Figure 11. Median daily dissolved oxygen concentrations collected by IAWA during 2006.

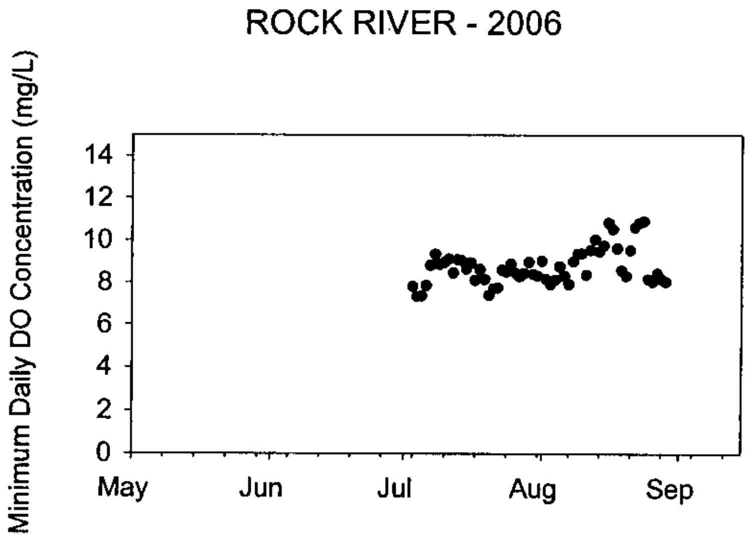


Figure 12. Minimum daily dissolved oxygen concentrations collected by IAWA during 2006.

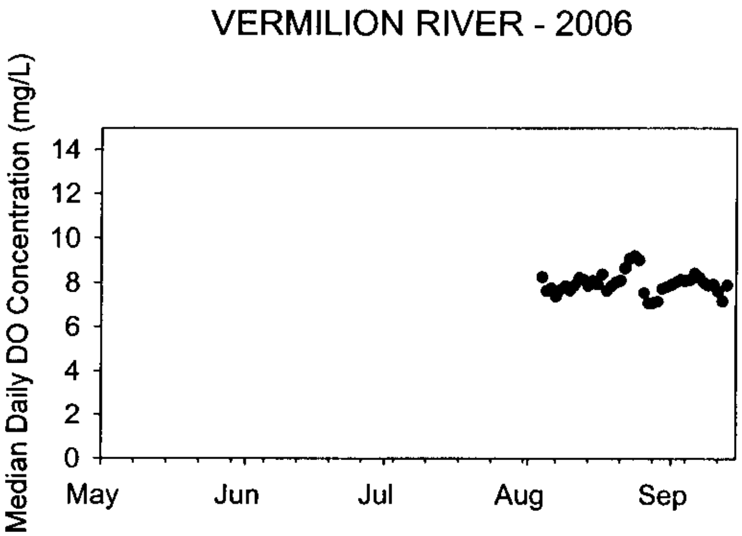


Figure 13. Median daily dissolved oxygen concentrations collected by IAWA during 2006.

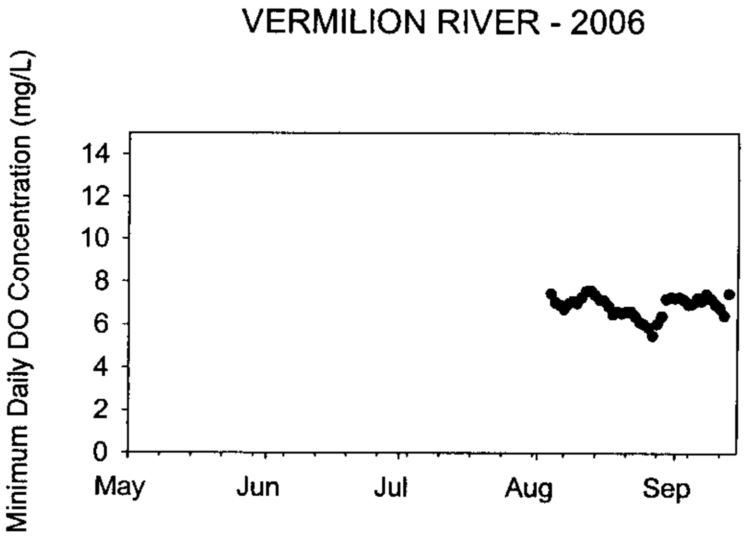


Figure 14. Minimum daily dissolved oxygen concentrations collected by IAWA during 2006.

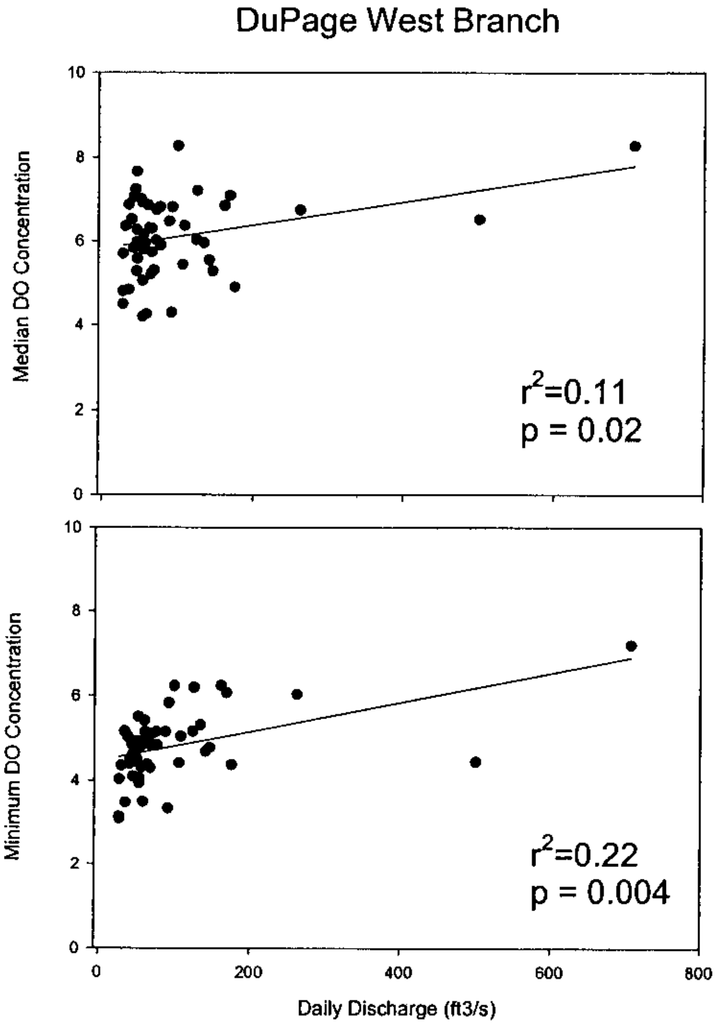


Figure 15. Daily discharge versus daily median and minimum dissolved oxygen concentration. All regressions were linear.

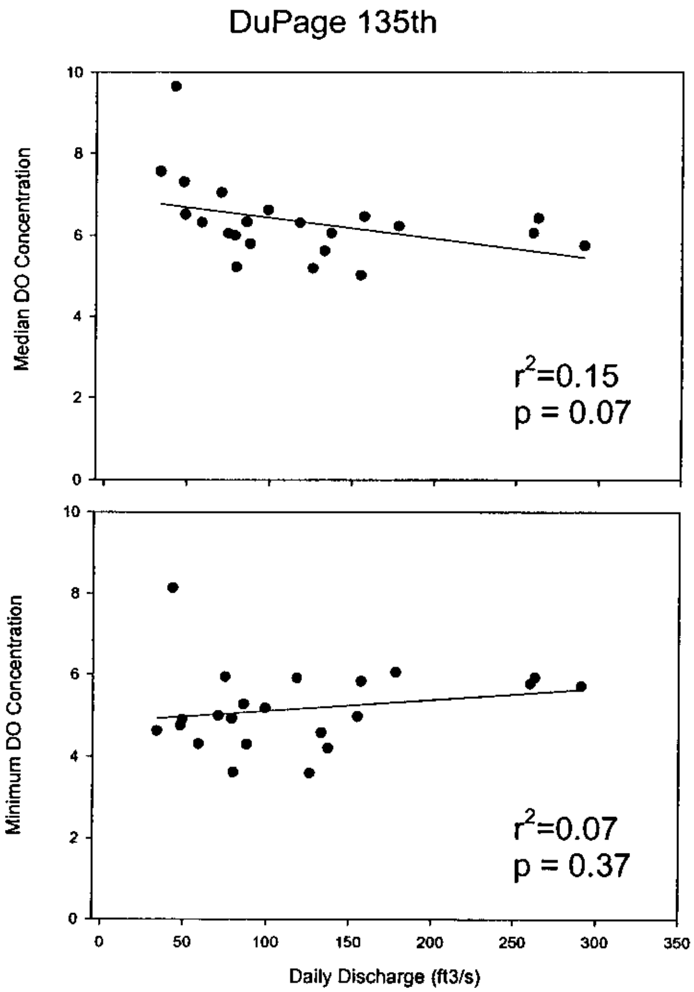


Figure 16. Daily discharge versus daily median and minimum dissolved oxygen concentration. All regressions were linear.

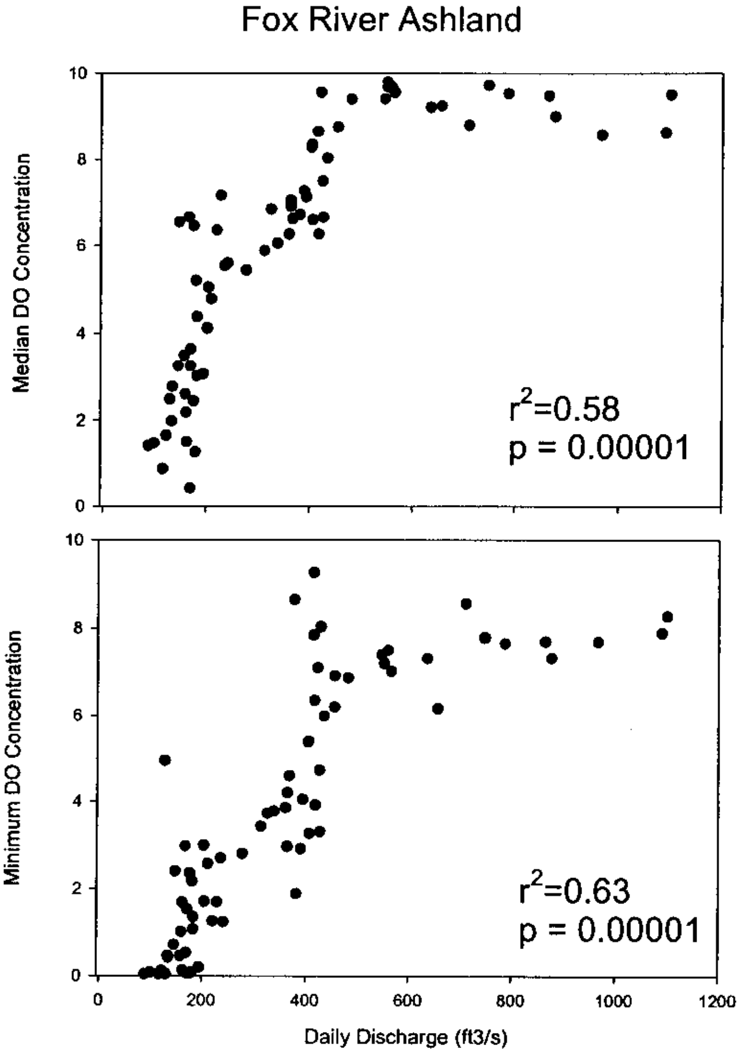


Figure 17. Daily discharge versus daily median and minimum dissolved oxygen concentration. All regressions were log-linear.

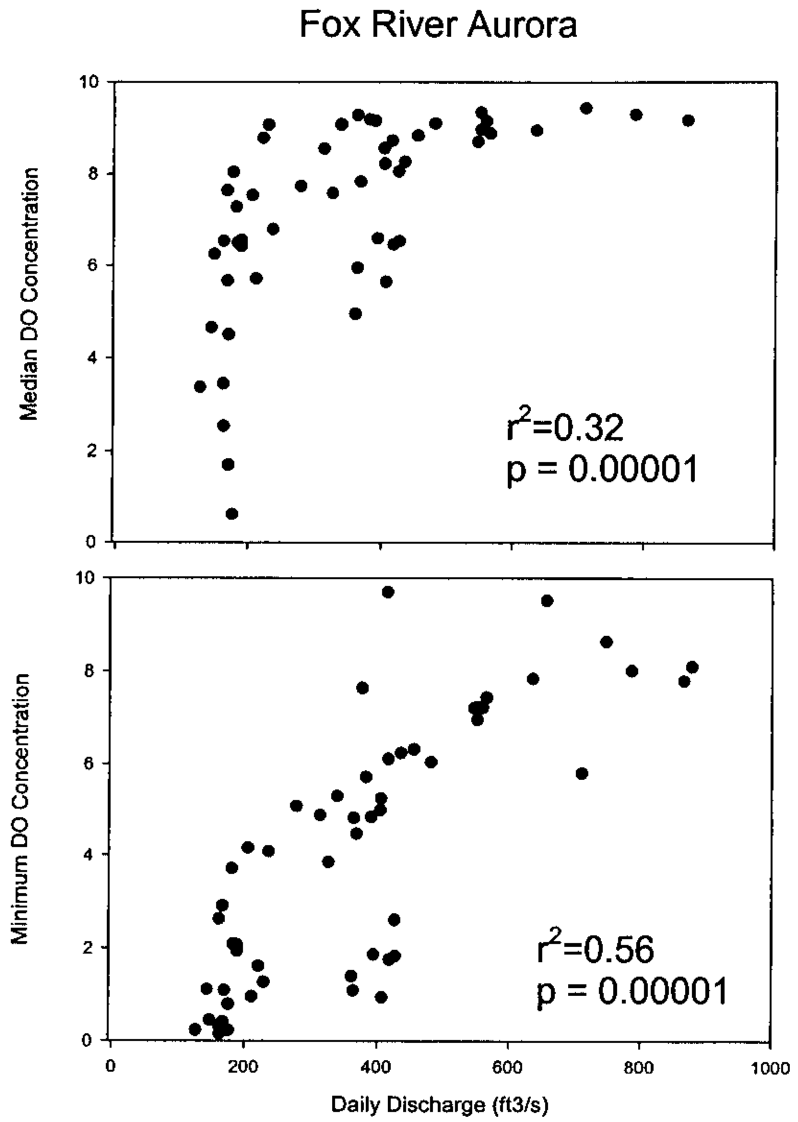


Figure 18. Daily discharge versus daily median and minimum dissolved oxygen concentration. All regressions were log-linear.

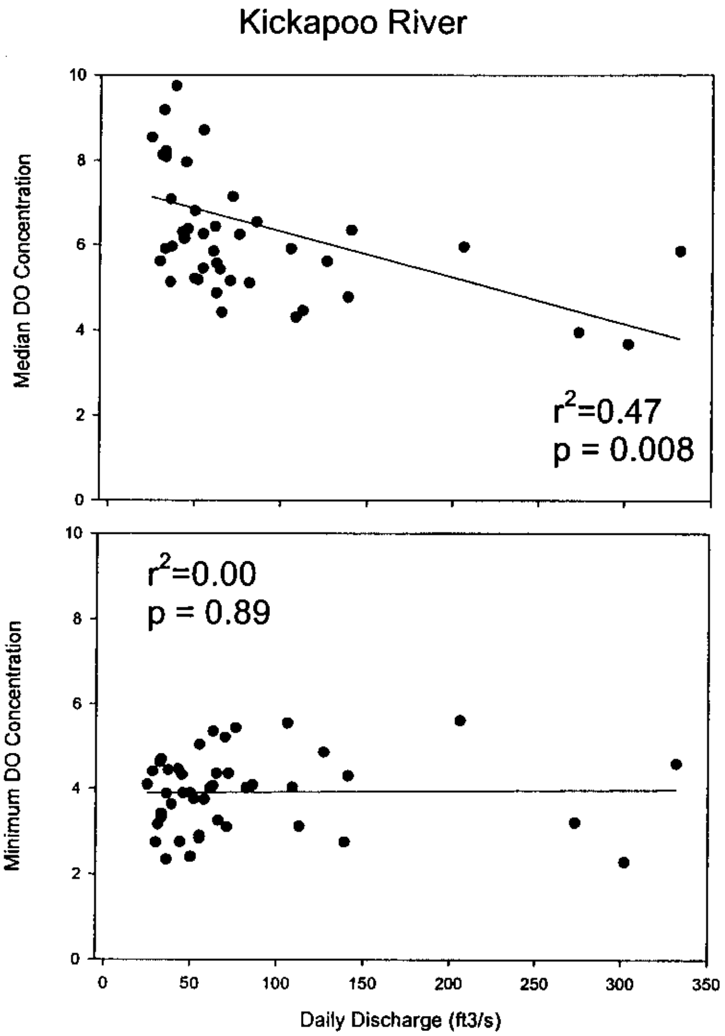


Figure 19. Daily discharge versus daily median and minimum dissolved oxygen concentration. All regressions were linear.

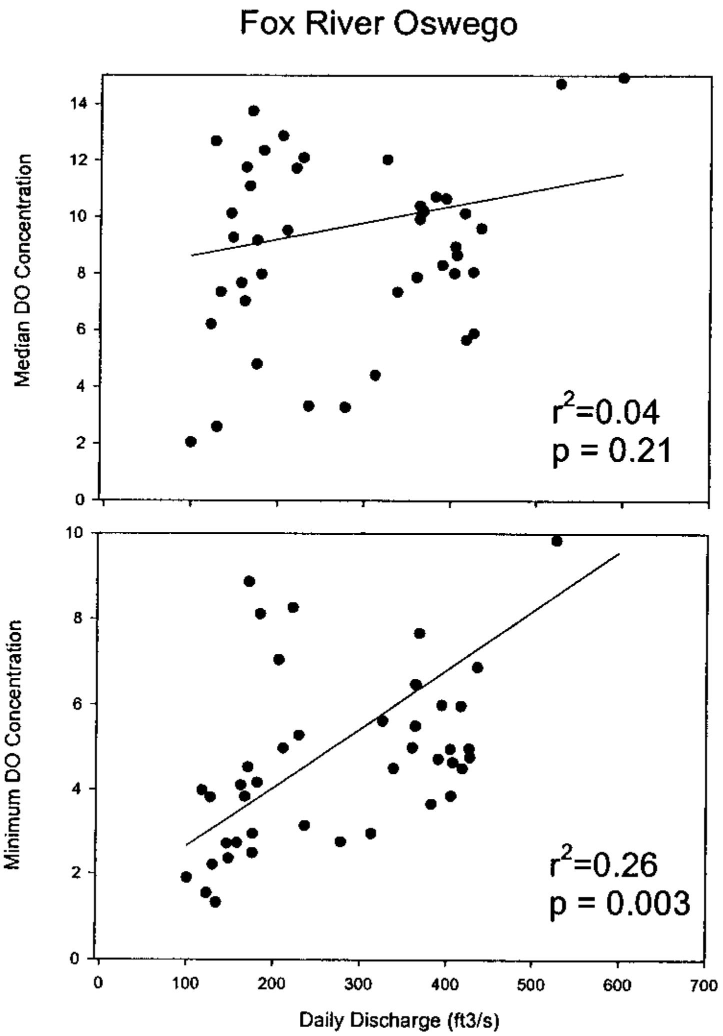


Figure 20. Daily discharge versus daily median and minimum dissolved oxygen concentration. All regressions were linear.

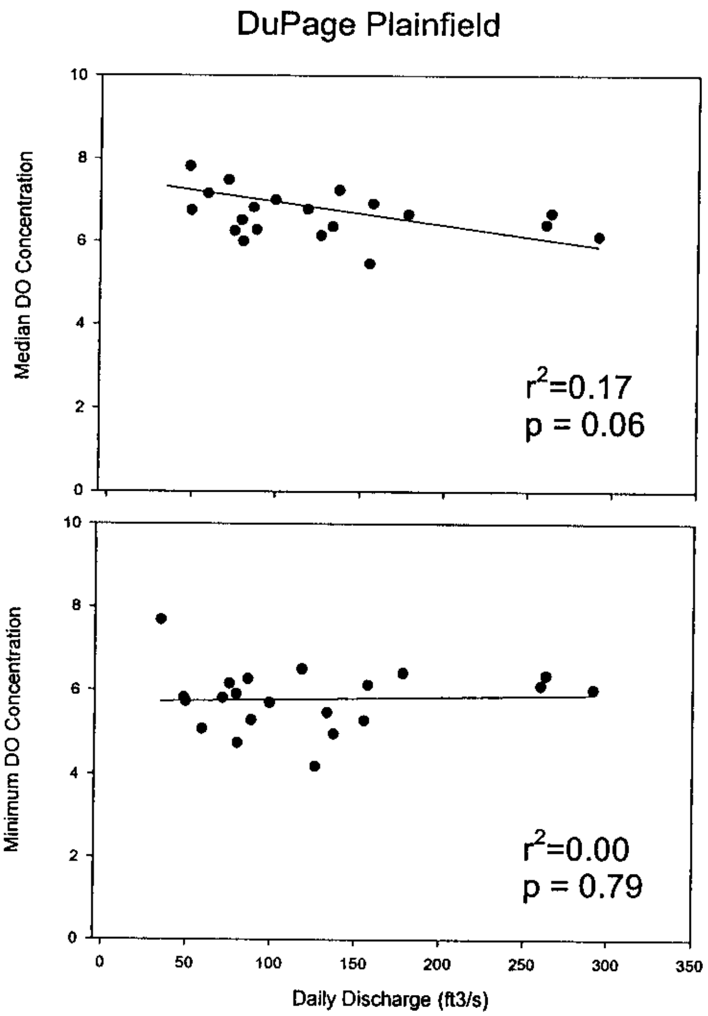


Figure 21. Daily discharge versus daily median and minimum dissolved oxygen concentration. All regressions were linear.

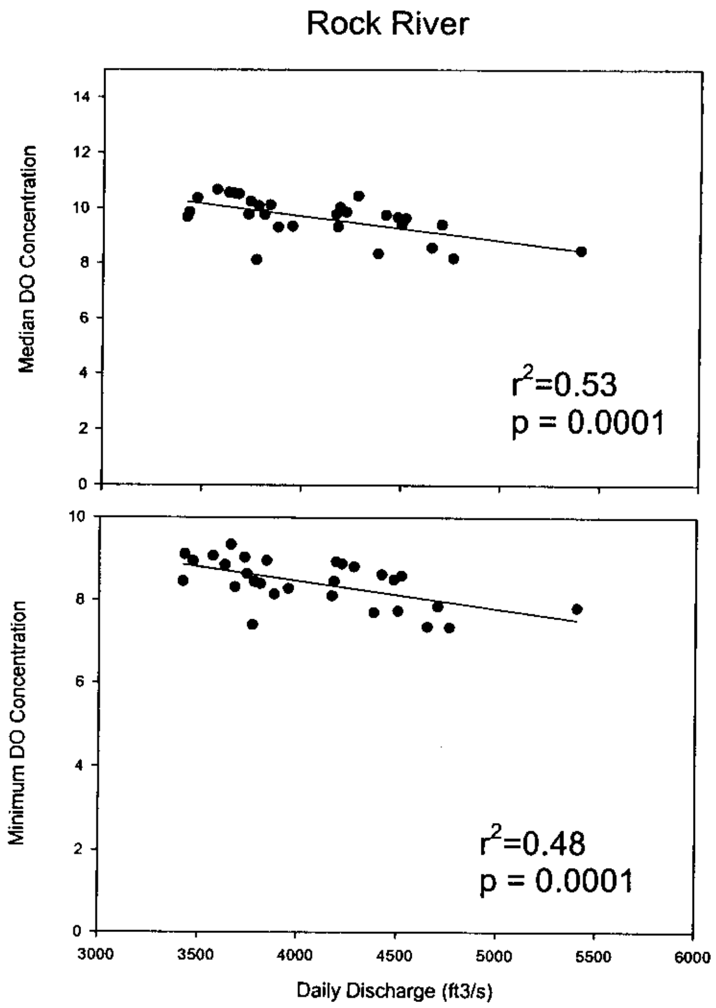


Figure 22. Daily discharge versus daily median and minimum dissolved oxygen concentration. All regressions were linear.

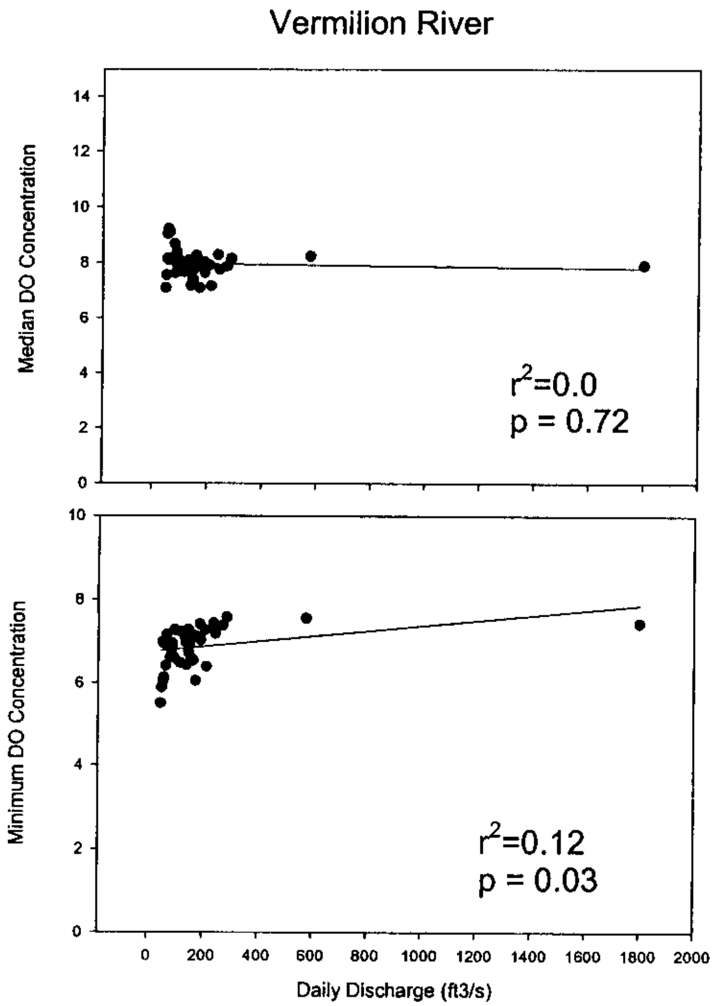


Figure 23. Daily discharge versus daily median and minimum dissolved oxygen concentration. All regressions were linear.

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IAWA DATA FROM ENHANCED SEGMENTS

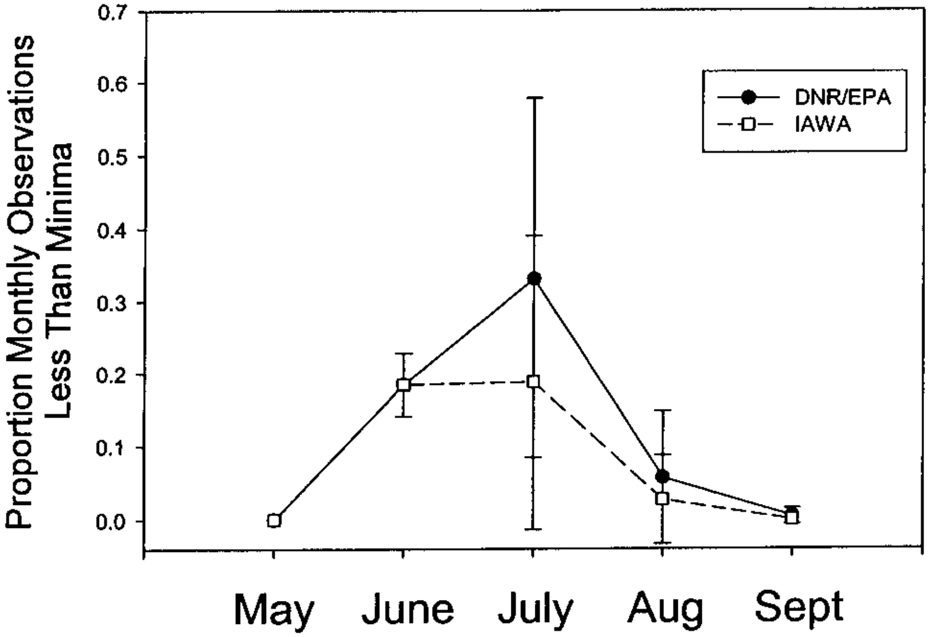

Figure 1. Mean ± standard deviation proportion of observations (averaged across sites) that exceeded the acute minimum dissolved oxygen criteria for the two proposed standards.

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Table 1. Minimum criteria. Number of dissolved oxygen concentration point estimates (mg/L) quantified at each "enhanced" river site and month that were less than 5 mg/L, 4 mg/L, and 3.5 mg/L. Proportions that were less than the minimum acute concentration proposed by DNR/EPA and IAWA are listed. DNR/EPA: Not less than 5 mg/L through July 31; IAWA: Not less than 5 mg/L through June 30. For later months, DNR/EPA: Not less than 4 mg/L; IAWA: Not less than 3.5 mg/L.

Month	River	Site	Total Obs	below 5	below 4	below 3.5	Violate DNR/EPA	Violate IAWA
May	Fox	Aurora	1630	0	0	0	0.00	0.00
		Ashland/Montgomery	1018	0	0	0	0.00	0.00
		Oswego	143	0	0	0	0.00	0.00
June	DuPage	West Branch	558	67	0	0	0.12	0.12
	Fox	Aurora	2854	619	417	328	0.22	0.22
		Ashland/Montgomery	1151	236	108	61	0.21	0.21
		Oswego	1058	208	109	65	0.20	0.20
July	DuPage	West Branch	611	202	45	17	0.33	0.03
	Fox	Aurora	560	255	184	149	0.46	0.27
		Ashland/Montgomery	1324	868	755	657	0.66	0.50
		Oswego	774	167	136	117	0.22	0.15
	Rock	Rock River	2712	0	0	0	0.00	0.00
August	DuPage	135th	635	193	49	0	0.08	0.00
		Renwick	556	44	0	0	0.00	0.00
	Kickapoo	Peoria	2917	1268	609	394	0.21	0.14
	Rock	Rock River	2729	0	0	0	0.00	0.00
	Vermilion	Danville	658	1	1	1	0.00	0.00
Sept	DuPage	135th	311	17	0	0	0.00	0.00
		DuPage	44	0	0	0	0.00	0.00
		Naperville	320	9	0	0	0.00	0.00
		Renwick	310	0	0	0	0.00	0.00
	Kickapoo	Peoria	1404	318	34	0	0.02	0.00
	Vermilion	Danville	297	0	0	0	0.00	0.00
						Average	0.12	0.08
						Proportion Site-Dates	0.48	0.39

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Table 2. Seven-day mean criteria. Number of dates where mean daily dissolved oxygen concentrations averaged across seven days were less than the proposed criteria for DNR/EPA and IAWA. DNR/EPA: Seven day averages must not be less than 6.25 mg/L through July; IAWA: Seven day averages must not be less than 6 mg/L through June. N/A = not applicable.

Month	River	Site	TOTAL DAYS	DNR < 6.25	IAWA <6	Violate DNR/EPA	Violate IAWA
May	Fox	Aurora	6	0	0	0.00	0.00
		Ashland/Montgomery	11	0	0	0.00	0.00
June	Fox	Aurora	30	0	0	0.00	0.00
		Ashland/Montgomery	19	2	1	0.11	0.05
		Oswego	21	0	0	0.00	0.00
	DuPage	West Branch	13	0	0	0.00	0.00
July	Fox	Aurora	12	1	1	0.08	N/A
		Ashland/Montgomery	28	21	19	0.75	N/A
		Oswego	19	0	0	0.00	N/A
	Rock	Rock River	23	0	0	0.00	N/A
	DuPage	West Branch	26	12	11	0.46	N/A
					Average	0.13	0.01
					Proportion of Site-Dates	0.36	0.17

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Table 3. Seven-day minimum criteria. Number of dates where minimum daily dissolved oxygen concentrations averaged across seven days were less than the proposed criteria for DNR/EPA and IAWA. DNR/EPA: Seven day averages must not be less than 4.5 mg/L after July 31; IAWA: Seven day averages must not be less than 4 mg/L after June 30. N/A = not applicable.

Month	River	Site	Total	DNR violation < 4.5	IAWA violations < 4	DNR PROP	IAWA PROP
July	DuPage	West Branch	26	12	6	N/A	0.23
	Fox	Aurora	12	12	12	N/A	1.00
		Ashland/Montgomery	28	28	28	N/A	1.00
		Oswego	19	12	11	N/A	0.58
	Rock	Rock River	23	0	0	N/A	0.00
August	DuPage	135th	2	1	0	0.50	0.00
		Renwick	1	0	0	0.00	0.00
	Kickapoo	Peoria	25	25	20	1.00	0.80
	Rock	Rock River	29	0	0	0.00	0.00
	Vermilion	Danville	22	0	0	0.00	0.00
Sept	DuPage	135th	8	0	0	0.00	0.00
		Renwick	8	0	0	0.00	0.00
	Kickapoo	Peoria	15	11	3	0.73	0.20
	Vermilion	Danville	13	0	0	0.00	0.00
					Average	0.25	0.27
					Proportion Site-Dates	0.33	0.43

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Table 4. Thirty-day mean criteria. Number of dates where mean daily dissolved oxygen concentrations averaged across thirty days were less than the proposed criteria for DNR/EPA and IAWA. DNR/EPA: Seven day averages must not be less than 6 mg/L through July; IAWA: Seven day averages must not be less than 5.5 mg/L through June. N/A = not applicable

Month	River	Site	TOTAL	DNR < 6	IAWA < 5.5	DNR	IAWA
June	Fox	Aurora	4	0	0	0.00	0.00
July	Fox	Aurora	12	0	0	0.00	N/A
		Ashland/Montgomery	18	14	11	0.78	N/A
		Oswego	12	0	0	0.00	N/A
	DuPage	West Branch	6	0	0	0.00	N/A
					Average	0.16	0
					Proportion of Site-Dates	0.2	0