Dissolved Oxygen Fluctuation Regimes in Streams of the Western Corn Belt Plains Ecoregion

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INTRODUCTION

As we consume more of our natural resources, it becomes increasingly apparent that we must do everything possible to insure that these resources will still be viable in the future. In building upon the Clean Water Act of 1972, the U.S. government developed and issued the Clean Water Action Plan (CWAP) in 1998. The CWAP was designed to protect public health from polluted waters, improve the control of polluted land runoff, and promote the protection of water quality on a watershed basis. In response to the CWAP, the U.S. Environmental Protection Agency (USEPA) developed the National Strategy for the Development of Regional Nutrient Criteria (NSDNC). The NSDNC outlines a framework for developing water body-specific, technical guidance used to assess nutrient status and develop regional nutrient criteria (USEPA 1998). Within this framework, the USEPA identified several areas in which to focus research. These areas include grey literature on nutrient data, studies on instream chlorophyll levels, and diel dissolved oxygen fluctuation related to instream nutrient concentrations. The USEPA formed Regional Technical Assistance Groups (RTAG) to address these topics throughout the country. The Central Plains Center for BioAssessment (CPCB) at the Kansas Biological Survey is a member of the USEPA Region 7 RTAG and is working with other members from states and tribes in Iowa, Kansas, Missouri, and Nebraska on various aspects of the research topics. CPCB has led this study that attempts to address several of the identified research needs and focuses on how instream nutrient concentrations (and other factors) may affect diel dissolved oxygen fluctuation and impact primary production.

The established and quantifiable relationships commonly observed in lakes and reservoirs, for example total phosphorus and hypolimnetic dissolved oxygen deficit (Welch

1992), do not exist for streams and rivers (Dodds *et al.* 1998). However, it is believed that these types of relationships could be developed for streams and rivers (EPA 2000). Incomplete and "snap shot" measures of temporal and diel dissolved oxygen (DO) patterns is one reason that statistical analysis of instream variables and dissolved oxygen yield poor relationships (Uehlinger and Naegeli 1998). Dissolved oxygen levels fluctuate throughout the day and nighttime periods (i.e., diel flux) by as much as 10 mg/L. Many past projects that attempted to draw relationships among dissolved oxygen and other instream variables have used a single dissolved oxygen measure for each stream that may have been taken at any point on the diel oxygen cycle and during any time of the year. The CPCB has compiled a database of over 25,000 records from streams in EPA Region 7 from a number of sources (i.e. U.S. Army Corps of Engineers, Iowa Department of Natural Resources, Kansas Department of Health and Environment, Missouri Department of Natural Resources, Nebraska Department of Environmental Quality, USEPA, U.S. Geological Survey) in which measures for dissolved oxygen are reported as a single value for each site. Because oxygen levels exhibit patterns that reflect the results of exogenous and endogenous factors operating at differing time scales that cannot be discerned by single point-in-time measures, it is little wonder that few if any relationships have been quantified. By examining continuous measurements of dissolved oxygen, it is hypothesized that the relationships between the watershed, water chemistry, nutrients, stream biota and dissolved oxygen will become more apparent. This study focuses on identifying and defining potential causal relationships among stream nutrient concentration, dissolved oxygen levels and daily oxygen flux during summer growing periods.

Nutrients provide vital sustenance for both terrestrial and aquatic plants. In aquatic ecosystems, excessive concentrations of nutrients can lead to declines in water quality as well as undesirable changes in the ecology of water bodies. Evidence from a wide array of studies showed that lotic systems are sensitive to anthropogenic inputs of nitrogen and phosphorus, the two macro-nutrients essential for plant growth (Smith *et al.* 1999).

Cultural eutrophication is a leading cause of impairment in U.S. surface waters (USEPA 1996). Eutrophication is often a normal part of the natural aging process of a lentic body of waters. Cultural eutrophication occurs when nutrient loads and concentrations increase due to anthropogenic activities and factors often associated with the watershed or airshed of a waterbody. Nitrogen and phosphorus are the primary nutrients limiting eutrophication in nearly all freshwater ecosystems except for some waterbodies in which nitrogen is the limiting factor in aquatic plant production (USEPA 2000).

Nutrient impaired waters can cause problems ranging from recreational annoyances, to serious public health concerns, to adverse effects on the ecology of the aquatic ecosystem (Dodds and Welch 2000). High levels of algae and macrophytes can develop quickly in response to excessive nutrient inputs. Eutrophication can restrict water use for fisheries and recreation (Sharpley *et al.* 2000). Although autotrophic biomass is important in many streams as a food source for organisms (Lamberti 1996), algal blooms can produce toxins such as microcystin and trihalomethanes (THMs) that affect livestock and human health (Palmstrom *et al.* 1988, Kotak *et al.* 1994, USEPA 1998, USEPA 2000). Stimulated by nutrients, harmful algal blooms such as brown tides and *Pfiesteria* have lead to major fish kills in costal areas (Palmstrom *et al.* 1988, Kotak *et al.* 1994). High concentrations of ammonia are toxic to fish and nutrients like nitrate can have direct detrimental effects when ingested by humans (Matson *et al.* 1997). Increased nutrient enrichment can directly and indirectly affect the ecology of the aquatic ecosystem by degrading water quality, altering instream habitat and changing primary producer communities and production. High algal and macrophyte biomass may be associated with severe swings in dissolved oxygen (USEPA 1998, Sharpley *et al.* 2000). Low dissolved oxygen concentrations can increase the availability of toxic substances (e.g. ammonia), thereby reducing the amount of quality habitat for aquatic organisms.

Aside from changes in the structure and function of aquatic ecosystems, nutrient impairment can affect the quality of drinking water resulting in increased treatment expenses (USEPA 2000). Increases in algae and macrophytes from nutrient impairments often leads to clogged intakes for water treatment plants. The increased biomass in water sources requires additional chemicals and longer settling times to attain acceptable water quality.

Several factors regulate the instream concentration of dissolved oxygen and these are compounded by eutrophication caused by macronutrients nitrogen and phosphorus. Drainage accrual, diffusion of oxygen across the air/water interface, and stream biota control the level of instream oxygen (Odum 1956).

The input of oxygen rich or oxygen poor waters by drainage accrual results in changes to dissolved oxygen concentrations. Drainage accrual is attributed to the upwelling of groundwater into the stream flow (Uehlinger and Naegeli 1998), runoff from the surrounding watershed, inputs from agricultural drainage tiles, or through municipal wastewater discharges. Since these waters are typically low in dissolved oxygen concentrations, their introduction lowers the concentration of dissolved oxygen in the stream (Naiman and Bilby 1998). However, the addition of waters by groundwater or vadose water accrual may stimulate the uptake of dissolved oxygen through community respiration and thus enhance stream community respiration.

Diffusion of oxygen across the air-water interface tends to moderate changes in dissolved oxygen concentrations in streams. For streams that have a high surface area to volume ratio, like the small, relatively shallow streams examined in this study, diffusion becomes a major component in the regulation of instream oxygen concentrations. As dissolved oxygen levels depart from saturation levels, oxygen will diffuse into or out of the stream in order to maintain an oxygen concentration near saturation (Wilcock *et al.* 1995). Changes in dissolved oxygen concentrations due to photosynthetic production or respiration will only be detected when these biotic changes in oxygen concentration exceed the rate of change due to diffusion.

The third component affecting dissolved oxygen concentrations in streams is stream biota. The metabolic activity of stream organisms controls the level of productivity in the stream. Viable macrophyte and algae populations produce oxygen as a byproduct of photosynthesis thus increasing the concentration of dissolved oxygen while whole system respiration demands use oxygen. The respiratory activity of fish, benthic invertebrates and bacteria continuously utilizes dissolved oxygen while plant respiration is limited to dark photoperiods. Bacterial decomposition of autochthonous organic mater from decaying primary producers and stream organisms along with the decomposition of allochthonous inputs of organic matter consume dissolved oxygen. Biological modification of the oxygen regime leads to a more productive environment and greater fluctuations in concentrations during the day (Horne and Goldman 1994). This production of oxygen during the day and continuous consumption of oxygen throughout the day/night periods drives the fluctuating diel cycle of dissolved oxygen.

In situations where drainage accrual is negligible and rates of diffusion are known or can be estimated, the relationship between changes in dissolved oxygen concentrations and the metabolic activity of stream biota can be used to calculate values of production and respiration. Odum (1956) presented the diurnal oxygen curve method, which estimates daily gross primary production and community respiration from changes in measured stream dissolved oxygen concentrations over time. Since this method measures the entire stream community it should provide more representative results than chamber measurements that neglect many of the dynamic factors that influence whole stream metabolism (Marzolf *et al.* 1994). When reaeration coefficients can be accurately estimated, Odum's diurnal curve method provides an accurate measure of gross primary productivity (Bott *et al.* 1978).

Primary productivity patterns in lotic ecosystems have been shown to respond to downstream changes in ecological processes and stream morphology (Naiman and Bilby 1998). Vannote and co-researchers describe these trends in stream function in the river continuum theory introduced in 1980 (Vannote *et al.* 1980). This theory views river systems as a continuum of geophysical and hydrological changes with an associated gradient of biotic communities in which structure and function changes. Community metabolism is a main part of the theory (Fleituch 1999). The theory proposes that small headwater streams are generally strongly influenced by riparian vegetation, which reduces primary production through shading and inputs large amounts of organic detritus. This results in gross primary production to respiration ratios (P/R) of less than one. Minshall (1978) points out that this concept is modeled on natural, undisturbed stream ecosystems. The longitudinal profile of productivity in agriculturally dominated prairie streams, like those selected in this study, has been shown to deviate from the current theory (Wiley *et al.* 1990). Anthropogenic disturbances like those found in the agriculturally dominated watersheds of Western Corn Belt Plains streams could increase P/R ratios.

The streams in this study are located in the Western Corn Belt Plains (WCBP) of the U.S. and were selected because of their agriculturally dominated watersheds, relatively open canopied stream channels and high nutrient concentrations. Nutrient inputs from fertilizers and alteration of riparian vegetation by farming, grazing, or channelization may have a significant impact on the metabolic activities of the stream biota. The sparse canopy structure of many of the riparian corridors in the WCBP provides little shading. The high light intensities, combined with high nutrient levels are expected to result in high rates of production that cause large fluctuations in dissolved oxygen concentrations and potentially low DO levels during all or parts of the diel period.

Study Objectives

The overall goal of this study was to quantify DO values and DO flux in small streams under normal or low flow conditions associated with late summer and investigate possible causal relationships among DO and a number of stream and watershed factors. This study focused on wadeable streams of the Western Corn Belt Plains of the Central Plains region because their watershed are dominated by cultivated cropland and thus receive high inputs of nutrients. It was anticipated that these streams were highly productive and therefore susceptible to large diel swings in oxygen concentrations and low DO concentrations. It was hypothesized that relationships among DO and stream nutrients and other variables could be quantified by continuously monitoring instream dissolved oxygen concentrations through daily or weekly time periods and comparing these values with grab sample chemistry.

The objectives of this study were:

1. Quantify the diel fluctuations of dissolved oxygen in wadeable streams within the Western Corn Belt Plains.
2. Quantify stream production and respiration in wadeable streams within the Western Corn Belt Plains.
3. Examine the possible relationships among dissolved oxygen, stream productivity, and instream and watershed factors.

STUDY DESIGN AND METHODS

Ecoregional Approach and Study Sites

The Western Corn Belt Plains (WCBP) of the central U.S. (Figure 1) ecoregion was selected as the area of study because lotic ecosystems in this region are typically low gradient streams with fine (e.g. sand, silt) bottom substrates. In mid- to late summer water temperatures often exceed twenty-eight degrees Celsius and flows are generally at or near yearly low flow averages. In addition, high inputs of nutrients are the result of the rather intense agricultural land use associated with nearly all of the ecoregion. Streams within the Western Corn Belt Plains (WCBP) were thought to be highly susceptible to dissolved oxygen (DO) depletion as a potential result of the cumulative effects of the factors mentioned above. These streams reflect the nature of the landscape they drain. This regional approach reduced the effects among stream sites of spatial differences (i.e. soils, geology, land use practices, etc.) that could bias measurements if regional boundaries were ignored. Omernik's (1987) ecoregion classification scheme was employed because it was designed to aid in understanding the regional patterns of aquatic and terrestrial resources and their attainable quality. This ecoregion framework was based on climate, mineral availability, vegetation, and physiographic factors.

The WCBP ecoregion forms part of the landscape in the states of Iowa, Kansas, Minnesota, Missouri, and Nebraska, with the majority of the region occurring within Iowa. With over ninety percent of the Western Corn Belt Plains used for agriculture, the WCBP is well known for its highly productive cropland (Chapman *et al.* 2001). Crops such as alfalfa, corn, sorghum, soybeans, and wheat dominate this ecoregion that was once covered by tall grass prairies. Topography varies from level alluvial plains to gently rolling glaciated till to hilly loess plains. Warm, fertile, moist soils cover the region and on average receive fifty-eight to eighty-nine centimeters of precipitation annually, mainly during the harvest season. The high concentration of cropland and livestock creates major environmental concern with regard to surface and groundwater contamination.

During the summers of 1999 and 2000, thirty-six streams in the WCBP ecoregion were monitored and analyzed to address basic questions regarding DO flux and concentrations, and possible relationships with other abiotic and biotic factors (Figure 1). Sixteen of these streams and watersheds had been previously studied for two and a half-years as part of a grant funded by the USEPA and so were selected for study in 1999. In 2000, twenty additional streams were added to the study to provide better spatial coverage of the ecoregion and to examine stream systems with a history of either limited or high nutrient concentrations. Watershed drainage areas ranged from approximately 3,000 to 55,000 hectares. Variation between stream sizes was controlled by limiting study streams to fourth and fifth order streams or smaller. Also, limiting the study to low order streams provided

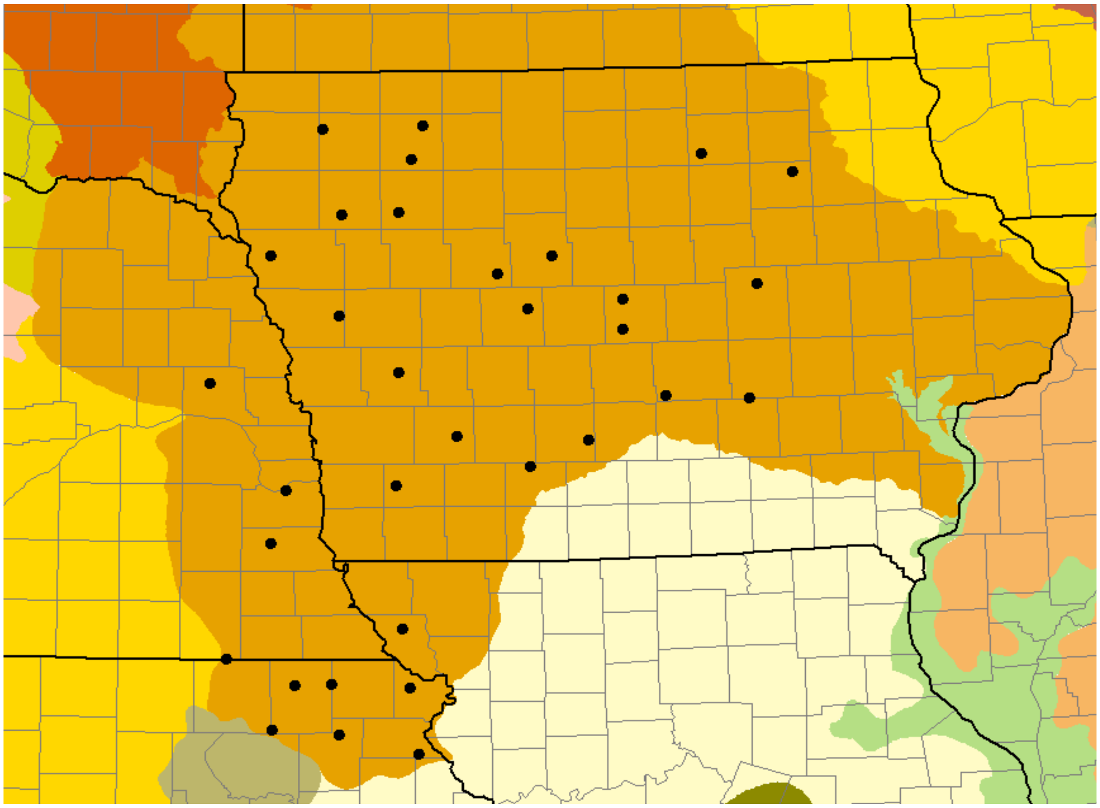


Figure 1. Map of the streams sampled during the summer of 1999 and 2000. The dark tan-colored region is the Western Corn Belt Plains ecoregion that occurs mainly in Iowa.

less complex watersheds that both have a higher potential for isolating the variables of interest and represent ecosystems with maximum terrestrial/aquatic interface. Unlike higher order streams that are affected by the many separate watersheds that make up their drainage basins, low order streams are more affected by their immediately surrounding watersheds because of the high land/water interface. Additionally, because large watersheds are typically aggregates of smaller watersheds, identifying and resolving relationships among variables within the more manageable smaller watersheds directly contributes to large watershed management. This allowed for the exploration of water quality interactions across spatial scales by examining the connection between the composition of the watershed and water quality measurements made at the stream site. Furthermore, the low order streams and their smaller watersheds are less costly and easier to study.

Seasonal Variation

Many of the physical, biological and chemical characteristics of lotic systems vary temporally as they respond to seasonal changes in climate and landscape conditions. Much of the temporal variation in the data was reduced by focusing the monitoring effort into brief sampling episodes with the late summer period that is characterized by high stream temperatures and minimal flows. It is hypothesized that the stream ecosystem experiences high levels of anthropogenic stress during the low flow period of late summer. Two predominate climate variables that can have critical effects on the stream ecosystem are naturally limited precipitation and high temperatures. Decreases in precipitation contribute to the reduction in stream flows from the normal baseflow regime to lower, more stable flow conditions. Thus during low flow periods, stream flow tends to be stable and with reduced stream power, instream habitat differences are often more pronounced with pools that have near zero velocity and reservoir-like conditions, and the heating capacity of the stream is elevated. High temperatures of late summer increase stream temperatures, facilitating high algal productivity and metabolic rates, while stable stream conditions promote accumulation of algae biomass (USEPA 2000). Algal populations thrive under these conditions, which can lead to increases in the frequency, duration and extent of algal blooms and "die offs." Large increases in algal production and biomass contribute to both high and low levels of DO and large diel fluctuations of instream DO concentrations as a result of gross production and community respiration demands. Collectively these factors result in DO-induced ecosystem stress and negative impacts to stream biota.

Daily Variation

Rainfall events occurring in the study stream watersheds during the study period may have directly and/or indirectly affected short-term primary production, DO, and water temperature regimes for these streams. Rainfall events of sufficient intensity and duration can result in near immediate surface runoff and increased stream flows. These elevated flows from runoff can result in increased turbidity, increased or decreased water temperatures, and increased nutrients and/or oxygen-demanding substances. In addition, runoff conditions can result in the physical scouring of periphyton. Associated increases in cloud cover may also significantly reduce in solar irradiation and air and water temperatures. Moderate to large changes in any one of these factors or conditions as a result of single or multiple events can affect DO values, fluctuations in these values, and/or primary production of these streams during a sampling period. Similarly, larger scale temporal (e.g., shorter day length, cooler temperatures) and spatial (e.g., reduced riparian shading, cultivation) changes affect both the rate and patterns of primary production and community respiration. In order to account for the potential effects of short-term climate phenomena (e.g., rainfall, air temperature, and solar irradiation) on stream measurements and estimated stream processes during the sampling period, daily precipitation data was obtained from the National Climatic Data Center (NCDC). The NCDC's Daily Surface Data series provided by state the station names and identification numbers along with daily precipitation values (measured to hundredths on an inch) and location coordinates. Each state's monitoring network comprised 300 to 450 weather stations that record precipitation.

To associate DO study sites with local precipitation stations, both the NCDC precipitation stations and DO stream study sites were mapped and displayed in ArcView. A 24-kilometer radius circle centered on each study site was delineated to help identify precipitation stations recording rainfalls that may have influenced the site's stream flow or climate conditions. Precipitation data within each 24-kilometer circle were then examined to quantify precipitation conditions just prior to and during the sampling time for each selected site. Any day (24-hour period) with precipitation over one third of an inch (10.8 cm) was flagged for possible significance as a potential indicator of surface runoff within the watershed draining to the site. Since the precipitation stations are not located exactly where the study sites are, it can be debated as to whether or not any rain actually fell at the study site or in the watershed. Thus, each recorded precipitation event (i.e. single or multiple days) was compared with graphs of the raw DO and water temperature data.

The corresponding segments of the raw DO and water temperature graphs were inspected for anomalies that might be attributed to a change in local weather and/or stream flows resulting from surface runoff. It was hypothesized that if the precipitation event resulted in significant changes then it would cause a visual anomaly in the water temperature or DO graphs. These visual responses most likely would be observed as changes in the shapes and values of individual diel curves that correspond to the precipitation period when compared to the complete series of diel curves generated during the study period. When a graphed DO or temperature anomaly (i.e. stream disturbance) and precipitation event of one third of an inch or greater coincided, it was assumed that a significant precipitation event was responsible for the noted changes in the DO and temperature graphs. In this case, the corresponding DO and temperature data were not used to calculate the average DO flux and production for normal (non-runoff) days. Furthermore, if graphed anomalies did not coincide exactly with the precipitation event, but appeared to be somewhat advanced or delayed (12-36 hours), it was assumed that the precipitation event was still responsible for the noted anomalies and that direct time correlations were affected by the spatial position of the reporting precipitation stations relative to the study site and/or time-related flow responses. In these cases, the offset associated diel data were grouped with data affected by climate and removed from the analyses of non-runoff data.

Occasionally, a precipitation event of one third of an inch or less of precipitation was recorded but no corresponding discernable disturbances or anomalies in the graphs were observed. In these few instances, it was concluded that the precipitation event did not cause sufficient changes in local weather (e.g. cloud cover, air temperature change) or stream flow to alter the DO or temperature patterns in the monitored stream segment. Thus, the DO flux data in these cases were retained and used to estimate values for non-runoff periods. In practice, it was found that the majority of precipitation events over one-half inch were correlated to a disturbance in the raw data graphs.

<u>Procedures and Study Variables</u>

Site Selection

Stream sampling events were targeted for the low flow, high temperature period of late summer starting in 1999 and continuing through the late summer periods of 2000 and 2001. Sampling sites were located between 50 and 100 meters upstream of bridge crossings except where stream access was limited, in which case best professional judgment was used to choose the site. In an effort to reduce instream variation associated with stream macrohabitats (e.g. velocity, depth, mixing), only runs were sampled. A single Aqua 2002 Dissolved Oxygen and Temperature Data Logger® (BioDevices Corp, of Ames, Iowa, 1999) was deployed at each stream site to continuously monitor stream temperature and DO. At the sampling site, steel fencing T-posts were positioned in the thalweg of a run and driven into the streambed at a downstream angle. Data Loggers were then attached to the backside of the fence posts with cable ties and padlocked to the posts to reduce thief and vandalism. The sensor heads of the Data Loggers were positioned approximately four centimeters above the bottom substrate. The Data Loggers were positioned close to the streambed in the thalweg so that under reduced flow conditions they would remain submerged and functioning properly. Following the procedures outlined in the Data Logger manual, the Data Loggers were calibrated as a group under the same aeration conditions prior to and during the study period. Measurements of dissolved oxygen and water temperature were taken at ten minutes intervals throughout the two-week long deployment period. At the end of this period, the logger was removed and the data was downloaded to a laptop computer.

Water Chemistry

At the start and end of each logger deployment period a series of *in situ* water quality measurements (Table 1) were taken using a Horiba U-10 Water Checker U10. In addition, a water sample was collected in a one liter amber glass jar during installation and during removal of the Data Logger. Samples were stored in coolers and packed with ice until delivered within four days to the laboratory for analysis using standard analytical procedures (Table 2). In addition to the parameters analyzed for in Table 2, several variables were

Table 1. *In situ* measured parameters, associated methods and instrument detection limits.

Parameter	Method Citation	Method Detection Limit
Air Temperature	Horiba, 1991; APHA, 1995; 2550 B	0.1 C
Water Temperature	Horiba, 1991; APHA, 1995; 2550 B	0.1 C
pH	Horiba, 1991; APHA, 1995; 4500-H A	0.1
Conductivity	Horiba, 1991; APHA, 1995; 2510 A-B	1 mS cm-1
Turbidity	Horiba, 1991; APHA, 1995; 2130 B	1.0 NTU
Dissolved Oxygen	Horiba, 1991; APHA, 1995; 4500-O G	0.1 mg/L

Table 2. Water chemistry parameters, associated equipment, laboratory methods and detection limits.

Parameter	Equipment/Method	Method Citation	Method Detection Limit
Nitrate-N (NO_3-N); Nitrite-N (NO_2-N)	Lachat QuikChem 4200 Flow Injection Analyzer	APHA, 1995 4500-NO_3/NO_2 G	10.0 µg/L
Ammonia (NH_3)	Lachat QuikChem 4200 Flow Injection Analyzer	Ebina *et al*., 1983	1.0 µg/L
Ammonium-N (NH_4-N)	Lachat QuikChem 4200 Flow Injection Analyzer	APHA, 1995 4500-NH_3 G	1.0 µg/L
Total Nitrogen	Lachat 48 Place Digestor Lachat QuikChem 4200 Flow Injection Analyzer	Ebina *et al*., 1983	10.0 µg/L
Phosphate (PO_4)	Lachat QuikChem 4200 Flow Injection Analyzer	APHA, 1995 4500 P	1.0 µg/L
Dissolved P (PO_4-P)	Lachat QuikChem 4200 Flow Injection Analyzer	APHA, 1995 4500 P	1.0 µg/L
Total Phosphorous	Lachat 48 Place Digestor Lachat QuikChem 4200 Flow Injection Analyzer	Ebina *et al*., 1983	5.0 µg/L
Chlorophyll *a*	Optical Tech. Devices, Ratio-2 System Filter Fluorometer	APHA, 1995 10200 H	1.0 µg/L
Phaeophytin *a*	Optical Tech. Devices, Ratio-3 System Filter Fluorometer	APHA, 1996 10201 H	1.0 µg/L
Chlorophyll *a* and Phaeophytin *a* from Periphyton	Filtered, extracted, centrifuged; Optical Tech. Devices, Ratio-2 System Filter Fluorometer	APHA, 1996 10201 H	1.0 µg/L

calculated from these data. Organic nitrogen was taken as total Kjeldahl nitrogen minus ammonia while organic phosphorus was calculated as total phosphorus minus dissolved phosphate. Nitrogen to phosphorus ratios were calculated using total nitrogen and total phosphorus. Also at the start and end of the logger deployment period, three to five periphyton samples were collected using the techniques outlined in Bouchard and Anderson (2001, unpublished manuscript) and returned to the laboratory for chlorophyll and phaeophytin analysis.

Velocity and Depth

At the start and end of each logger deployment period velocity and depth were measured along a series of five transects located at 20 to 30 meter intervals upstream from the logger at each study site. Measurements were made using a Swoffer Model 2100 Current Velocity Meter as outlined in the Swoffer manual. Eight or more measures were taken across each transect. Weighted averages for velocity and depth values were calculated for each transect and then averaged across the entire stream segment to produce a single estimate of velocity and depth for the stream segment upstream of the logger site.

This approach was thought to provide better representation of stream flow characteristics that could effect the reaeration potential at the logger (sampling) site than could a single set of measurements from 1 transect at the logger site. Stream discharge was calculated from cross-sectional transect data (i.e. velocity and depth measurements) using the methods of Maidment (1993).

Landuse/Landcover

Land use/land cover (LULC) data were compiled using geographic information systems and were derived from the EPA's Multi-Resolution Land Characteristics (MRLC) dataset. The MRLC project was established to provide multi-resolution land cover data of the conterminous United States from local to regional scales (Bara 1994). This LULC dataset was chosen because MRLC data was available in a seamless coverage for the entire study region and utilized a high number of LULC classes that provided a detailed assessment of the region.

Watersheds were defined as all of the land that drains to the sampling site. Based on the coordinates of the Data Logger at each site, the watersheds were mapped using ESRI's ArcView 3.2 software and displayed in ArcView with the River Reach 1 stream coverage for the WCBP and 1:24,000 scale USGS digital raster graphics (DRG) topographic maps of drainage areas. Utilizing the contour lines on the DRGs, the watersheds were delineated by digitizing the ridgeline that defined the area of land that drains surface water to the sampling site. The stream coverage aided in defining the extent of the watershed, while the sampling coordinates defined the lowest point of the watershed. Once the watersheds were delineated, they were re-projected as necessary to match the map projection of the MRLC image and overlaid on the MRLC image. Using ArcView's Spatial Analyst extension package, the area (m^2) of each LULC class occurring within each watershed was computed and divided by the total area of the watershed to determine the percent composition for each class. This was done so that LULC comparisons could be made among watersheds using standardized variables.

Estimates of gross primary production and community respiration were calculated from diel oxygen and temperature data curves following the widely used and generally accepted method of Odum (1956). To estimate production, a computer program was created from Odum's methods in Microsoft Excel. This productivity calculator is explained in detail in the following section.

The DO and temperature data was graphed over time to check the flux patterns for anomalies or irregularities (see Daily Variation section) that might have been due to precipitation, siltation, or a malfunction of the logger itself. If data anomalies were identified, those twenty-four hour cycles containing anomalies were removed from the datasets. Next, the DO and temperature data were prepared for graphical and statistical analysis. First, percent saturation values were calculated from the DO and temperature data to allow standard comparisons to be made between watersheds (APHA, 1998). Next, a series of statistical variables were calculated using both the raw DO and the percent saturation DO values. The calculated variables included daily average, median, mode, standard deviation, maximum, minimum, and the difference between the maximum and minimum values. Averages for the sampling period (five to fourteen days) were calculated for each sampling site from the daily averages.

Instream Productivity

Odum (1956) productivity methods were incorporated into the Productivity Calculator developed for this study. The rate of change of dissolved oxygen (Q) is affected by four main factors that include the rate of gross primary production (P), the rate of community respiration (R), the rate of oxygen diffusion (D), and the rate of drainage accrual (A) (Odum 1956). The majority of studies conducted on estimates of production derived from diel oxygen data follow the method set forth by Odum (1956).

The rate of change of dissolved oxygen can be determined by subtracting both the rate of community respiration (R) and oxygen diffusion (D) from the rate of gross primary production (P) plus the rate of drainage accrual (A) as shown in Equation 1.

$$Q = P - R - D + A \qquad (1)$$

The fluctuation of dissolved oxygen levels in the stream due to drainage accrual is assumed negligible relative to the other factors, but as a precaution, monitoring days in which dissolved oxygen readings could have been impacted by rainfall should be removed from the study dataset. This simplifies Equation 1 to Equation 2, the basic equation used to compute gross production.

$$Q = P - R - D \qquad (2)$$

There are several other factors that also affect the concentration of dissolved oxygen in streams and must be examined in order to provide more accurate estimates of production. These factors are reaeration, temperature, salinity, and pressure.

Reaeration of streams is caused by two factors: entrainment of oxygen due to turbulent flow and the replacement of oxygen due to a deficit from saturation caused by the combustion of organic matter. Reaeration was originally defined in the Streeter-Phelps (1925) equation as the reoxygenation (k_2) of streams. Today it is largely understood that the effects of the hydraulic properties of water (i.e. turbulent flow) are expressed as the coefficient of reaeration, k_2 (Langbein and Durum 1967).

The single most important factor regulating the concentration of dissolved oxygen in water is temperature (Horne and Goldman 1994). The concentration of oxygen in water is inversely proportionally to water temperature. Cold waters contain higher oxygen concentrations than the same volume of warm water at a given pressure.

Changes in barometric pressure alter the concentration of dissolved oxygen, since all gases are more soluble at higher pressures. This same principle is directly applicable to increases in altitude. In these instances where pressures are less due to increased altitude, concentrations of dissolved oxygen are reduced.

Salinity has a minor effect on dissolved oxygen concentrations in fresh waters when compared to the other constituents. Increases in dissolved salts reduce the intermolecular space with in the water molecules available to oxygen. Salinities must be high for increases in salt concentrations to effect dissolved oxygen concentrations. Conductivity was used in the calculator as an alternative input variable to salinity, because of its relationship to salinity and greater commonality of measure.

An understanding the influencing factors leads back to Equation 2. The first component of Equation 2 that needs to be solved for is D, the rate of oxygen diffusion. There are several aspects that affect the rate of oxygen diffusion (D). The reaeration coefficient $k_{2,20}$, in units of 1/day, is one of the first variables to define. Several authors, Owens *et al.* (1964) and O'Connor and Dobbins (1958) among others, have developed simple predictive equations for the estimation of $k_{2,20}$. Wilcock (1982) has provided an overview of some of the most widely used reaeration equations and the stream conditions for which the equations are most viable. The vast majority of these equations are of the form:

$$k_{2,20} = aU^{b} \cdot z^{c} \tag{3}$$

Where U is the mean stream velocity (m/s), z is the mean stream depth (m), and a, b, and c are constants. Once $k_{2,20}$ is computed, it must be corrected for temperature. This conversion is accomplished using the Elmore and West (1961) equation.

$$k_{2,T} = k_{2,20} \cdot 1.024^{T_C - 20} \tag{4}$$

A new temperature corrected reaeration coefficient, $k_{2,T}$, must be calculated for each recorded measurement of dissolved oxygen since the temperature is also recorded at the same time and does fluctuate throughout the day and night.

After calculating $k_{2,T}$, the concentration of dissolved oxygen at saturation for each recorded temperature must be calculated. During these computations corrections for salinity and pressure will be addressed. In order to calculate the dissolved oxygen concentration in mg/L at the standard pressure of one atmosphere, C_p, all temperature values are converted from Celsius to Kelvin, Equation 5.

$$T_K = T_C + 273.15 \tag{5}$$

The temperature in Kelvin, T_K, is used to calculate the dissolved oxygen concentration in mg/L at the standard pressure of one atmosphere, C_p.

$$C_p = e^{\left(-139.34411 + (157{,}570.1/T_K) - \left(66{,}423{,}080/T_K^2\right) + \left(1.2438\times10^{10}/T_K^3\right) - \left(8.621949\times10^{11}/T_K^4\right)\right)} \tag{6}$$

In order to utilize the user's conductivity value as the correction of salinity, several steps must first take place. The conductivity units must be converted from the user entered units mS/cm to µS/cm, Equation 7.

$$cond_2 = cond_1 \times 1000 \tag{7}$$

Then the conductivity correction factor for salinity is computed, Equation 8.

$$f_{cond} = -0.000003 \times cond_2 + 1.0002 \tag{8}$$

Once calculated, the correction factor is multiplied by C_p to correct the dissolved oxygen concentration at saturation and standard pressure for salinity, Equation 9.

$$C_{p,sal} = f_{cond} \times C_p \tag{9}$$

Having solved for, $C_{p,sal}$, we will calculate the nonstandard air pressure at the sampling site. This is accomplished using a measure of the site altitude as a surrogate for air pressure. The altitude is entered into the calculator in meters (A_m) and then converted to feet, Equation 10.

$$A_{ft} = A_m \times 3.280839895 \tag{10}$$

The equation that converts altitude to nonstandard pressure, P, in atmospheres was derived from a table relating pressure and altitude created by Cole-Palmer Instrument Co. (http://www.coleparmer.com/techinfo/techinfo.asp?htmlfile=PEquationsTables.htm). This conversion is calculated using Equation 11.

$$P = -3 \times 10^5 A_{ft} + 0.996 \tag{11}$$

Now the partial pressure of water vapor, P_{wv}, in atmospheres, can be computed. Equation 12 uses the measured water temperature, in Kevin.

$$P_{wv} = e^{\left(11.8571 - \left(3{,}840.7/T_K\right) - \left(216{,}961/T_K^2\right)\right)} \tag{12}$$

The final variable required for the calculation of the dissolved oxygen concentration at saturation for nonstandard pressure corrected for salinity and pressure is theta, θ. Theta is a temperature adjustment needed to calculate the final corrected concentration at saturation, Equation 13.

$$\theta = 9.75 \times 10^{-4} - \left(1.426 \times 10^{-5} T_C\right) + \left(6.436 \times 10^{-8} T_C^2\right) \tag{13}$$

Having calculated all the necessary variables, the dissolved oxygen concentration at saturation for nonstandard pressure corrected for salinity and pressure in mg/L, C_s , can be computed, Equation 14.

$$C_s = C_{p,sal} P \left[\frac{(1 - P_{wv}/P)(1 - \theta P)}{(1 - P_{wv})(1 - \theta)} \right] \tag{14}$$

These calculations result in a corrected value of dissolved oxygen at saturation for every measure of temperature logged.

The next portion of the procedure involves calculating the reaeration exchange rate, r . The reaeration exchange rate incorporates all the corrections previously calculated, salinity, pressure, reaeration, and temperature, into the determination of gross production. This begins with the calculation of the dissolved oxygen deficit in mg/L, C_d , Equation 15.

$$C_d = C_s - C_t \tag{15}$$

The dissolved oxygen deficit, C_d , is the difference between the corrected dissolved oxygen concentration, C_s , and the recorded dissolved oxygen concentration at some time, t . The dissolved oxygen deficit is then multiplied by the temperature corrected reaeration coefficient, $k_{2,T}$, and divided by the number of recording intervals per day, I_{sd} , resulting in the reaeration exchange rate, Equation 16.

$$r = k_{2,T} \cdot C_d / I_{sd} \tag{16}$$

The number of recording intervals per day, I_{sd} , is calculated from the logging interval, l_i , selected by the user in the *Stream Variables* worksheet.

From here the uncorrected change in dissolved oxygen concentration over time, ${\Delta C}/{\Delta t}_{Uncor}$, is computed. This is found by subtracting the current measure of dissolved oxygen, C_t, from the next measure, C_{t+1}, Equation 17.

$$ {\Delta C}/{\Delta t}_{Uncor} = C_{t+1} - C_t \qquad (17) $$

Since the program is calculating a rate, a change in concentration over unit time, the units for ${\Delta C}/{\Delta t}_{Uncor}$ are in mg/L/l_i, where l_i, the selected logging interval,

With the reaeration exchange rate and the uncorrected change in dissolved oxygen concentration over time calculations completed, we can correct the change in dissolved oxygen concentration over time for salinity, pressure, reaeration, and temperature by subtracting the reaeration exchange rate, Equation 18.

$$ {\Delta C}/{\Delta t}_{Cor} = {\Delta C}/{\Delta t}_{Uncor} - r \qquad (18) $$

With all the preliminary calculations completed, it is now possible to begin calculating estimates of production and respiration. The first step is the estimation of respiration, R. The calculated respiration rate is basically an estimated 24-hour community respiration rate that includes all stream respiration including that from fish, macroinvertebrates and microorganisms. Since photosynthesis occurs only in the presence of light, during the nighttime period the only community respiration should be occurring. Using this concept, the program determines the average change in dissolved oxygen concentrations at night and then extrapolates the value over the entire day to generate a daily rate of respiration. The average nighttime ${\Delta C}/{\Delta t}_{Cor}$ is calculated by summing all ${\Delta C}/{\Delta t}_{Cor}$ values that occur before sunrise and after sunset. The values for sunrise and sunset are input by the user. All individual nighttime values are summed then divided by the number of nighttime recording intervals to obtain the average nighttime corrected change in dissolved oxygen concentration over time, ${\Delta C}/{\Delta t_{\bar{n}}}$. To extrapolate this nighttime value over the entire day ${\Delta C}/{\Delta t_{\bar{n}}}$ is multiplied by the number of recording intervals per day, I_{sd}. And finally, everything is multiplied by depth, z, to convert the units from volumetric to areal. The resulting Equation 19 has units of g O_2/m^2/day.

$$R = {\Delta C}/{\Delta t_{\bar{n}}} \cdot I_{sd} \cdot z \qquad (19)$$

By reporting oxygen rates as respiration it is understood that an oxygen deficit exists therefore respiration values should be reported as positive numbers. For this purpose, the absolute value of R is reported.

The next estimated component is net primary productivity. Net primary productivity, P'_N, is the sum of the corrected change in dissolved oxygen concentration over time multiplied by depth to produce an areal measure, Equation 20 with units g O_2/m^2/day.

$$P'_N = \sum {\Delta C}/{\Delta t_{Cor}} \cdot z \qquad (20)$$

The final production estimate is gross primary production, P'_G. Gross primary production, units of g O_2/m^2/day, is computed from the addition of respiration and net primary productivity, Equation 21. As with R and P'_N gross primary production is has the units of g O_2/m^2/day.

$$P'_G = P'_N + R \qquad (21)$$

RESULTS AND DISCUSSION

Precipitation and Watershed Characteristics

The low flow and limited precipitation conditions normally associated with late summer in the WCBP were observed during the late summer sampling periods in this study. Approximately 147 centimeters of rain fell at the sampling sites, or within the streams' watersheds, during the sampling periods of 1999, 2000, and 2001. Almost half of the rain impacted only four stream sites. This highly excessive amount of rain lead to questions about the accuracy and validity of the DO and temperature data logged at the four sites. After examining the graphs of the DO and temperature data, three of the four sites were removed from the study based on the precipitation events' extreme degree of impact on the diel curves. The precipitation event at the fourth site occurred during the last few days of the sampling period. This meant that the days prior to the event were not impacted by the precipitation. Thus, by removing the precipitation-affected days at the end of the sampling period the majority of the DO and temperature data could be utilized. The remainder of the sampling sites appeared to have yielded strong datasets in which days that were affected by precipitation events could be readily accounted for and removed.

As expected, agriculture was the dominant land use/land cover (LULC) type in the study watersheds of the WCBP (Table 3). The average study watershed was about 13,300 hectares in size and was dominated by cropland, which covered 76% of the average watershed area. Row crops such as corn and soybeans comprised 99% of the total cropland of the average watershed. With the majority of watersheds located in Iowa, a state with 86.1% of the land area in agriculture (1997 Census of Agriculture, http://www.nass.usda.gov/census/), it was expected that cropland would be the dominant the

LULC. The inclusion of more reference quality stream sites than impacted sites in this study was the most likely cause for the lower cropland percentages found in some of the study watersheds. Pasture was the next largest LULC class with an average extent of 15 percent within a typical study watershed. No other single LULC class compromised more than 5% of the average watershed with grassland/herbaceous and forest categories accounting for a little less than 3.0 to 3.5 percent of the total area.

Table 5. LULC components of the average watershed associated with study streams.

LULC Class	Description	Hectares	Percent
11	Open Water	23.17	0.17
12	Perennial Ice/Snow	0.00	0.00
21	Low Intensity Residential	34.45	0.26
22	High Intensity Residential	5.38	0.04
23	Commercial/Industrial/Transportation	99.44	0.75
31	Bare Rock/Sand/Clay	0.01	0.00
32	Quarries/Strip Mines/Gravel Pits	0.17	0.00
33	Transitional	0.00	0.00
41	Deciduous Forest	370.94	2.78
42	Evergreen Forest	0.11	0.00
43	Mixed Forest	23.58	0.18
51	Shrubland	3.20	0.02
61	Orchards/Vineyards/Other	0.00	0.00
71	Grasslands/Herbaceous	457.51	3.43
81	Pasture/Hay	2,030.95	15.24
82	Row Crops	9,926.51	74.51
83	Small Grains	214.72	1.61
84	Fallow	0.00	0.00
85	Urban/Recreational Grasses (parklands)	17.08	0.13
91	Woody Wetlands	61.28	0.46
92	Emergent Herbaceous Wetlands	53.76	0.40
	Average Watershed Size (ha)	13,322.26	100

Small extents of urban, shrubland and wetland areas made up the rest of an average watershed (mean size = 13,322 ha), but individual watersheds varied both in size and in the extent and composition of LULC classes that created each landscape. Watershed sizes ranged from 2,849 to over 56,000 hectares. While land uses within these watersheds displayed high degrees of variation, cropland areas were always present and never comprised less than 17% of the total landscape. The largest proportion of row crop (e.g. corn, soybeans) within a single watershed was 93%. While row crops dominated the land used for agricultural purposes, small grains were not a major component of the watersheds. The maximum amount of land used for small grains was 14%, some watersheds had no small grain crop land. Pasture and hay accounted for 45% the watershed area in some watersheds to as little as 3% in others. The grassland/ herbaceous class was also highly variable, varying from 30% to near zero in some watersheds.

Stream Characteristics

Physical Measures

The physical measurements of each stream site were representative of stream systems in the WCBP. The vast majority of the sites were dominated by soft substrates composed primarily of sands, while some soft silts, clays, and gravels were also present in the streambeds. The majority of the habitats upstream of the DO loggers were the upstream continuation of the runs in which the loggers were positioned. Riffles and pools comprised approximately one forth of the upstream habitats. Typical of Midwestern streams, the riparian canopy of the study sites was sparse, on average just over 29% cover. While some stream sites had 100% canopy cover, the median site measure was 15%.

Measurements of the flow characteristics observed during the sampling periods of the study are summarized in Table 4. These measurements were obtained during low or normal flow periods and exclude rain induced runoff events. These are typical physical and hydrological characteristics of small streams located in the Central Plains region during late summer, low flow periods. Stream discharges were also low with 0.109 m^3/s the mean during this study. The high correlation of the low discharge values to average stream depths and velocities indicated that none of these streams appeared to be heavily influenced by ground water inflows.

Table 4. Mean values for selected physical and hydrology variables for study streams

Physical Measures	Mean	Median	Range
Depth (m)	0.181	0.178	0.038–0.341
Velocity (m/s)	0.105	0.086	0.007–0.324
Discharge (m^3/s)	0.109	0.073	0.002–0.699
% Canopy	29.4	15.0	0-100

Water Quality

Water quality conditions measured during the study were typical of values associated with small streams of the WCBP ecoregion. The Central Plains Center for BioAssessment (CPCB) has conducted several studies on reference and non-reference quality streams in the region in which they measured several of the same water quality variables collected for this project. Measures of total nitrogen (TN), total phosphorus (TP), turbidity, and chlorophyll *a* for other streams located in WCBP ecoregion were queried from the CPCB reference stream database and compared to the values found in this study. The CPCB also has an extensive database of stream data gathered from the USEPA, state and federal agencies and academic sources. This larger data set contains information on all types of streams sampled in the WCPB ecoregion and was included so that water quality conditions for the DO flux streams could be contrasted with water quality associated with the larger stream population that comprised this more comprehensive data set (Table 5). The mean values listed in Table 5 were calculated from measurements taken only during the summer months (i.e. July, August, and September). This reduced seasonally induced variations in water quality values due to natural climate and land use changes. Mean site values recorded in this DO study were the lowest mean values for any stream group in Table 5.

Table 5. Mean values for selected water quality parameters sampled during summer low or normal flows for streams of the Western Corn Belt Plains ecoregions.

Stream Dataset	Total N (mg/L)	Total P (µg/L)	Turbidity (NTU)	Chl *a* (µg/L)
All streams (USEPA and others)	6.2	185.4	67.3	10.9
Reference streams (CPCB only)	5.2	360.9	72.8	29.8
DO study streams	4.7	246.0	54.7	16.6

Differences between the mean values found in the DO study and the two data sets were likely the result of several factors. In part, the lower means of this study may be a reflection of the restricted sampling period and the stable, low flow conditions associated with the sampling regime. Reference streams were thought to drain higher quality watersheds and have less disturbances associated with them, which may account for their reduced nutrient, turbidity and chlorophyll values when compared to the values from the larger “all stream data set” that contained streams varying in size, hydrology and water quality. These factors probably contributed to their relatively higher mean values.

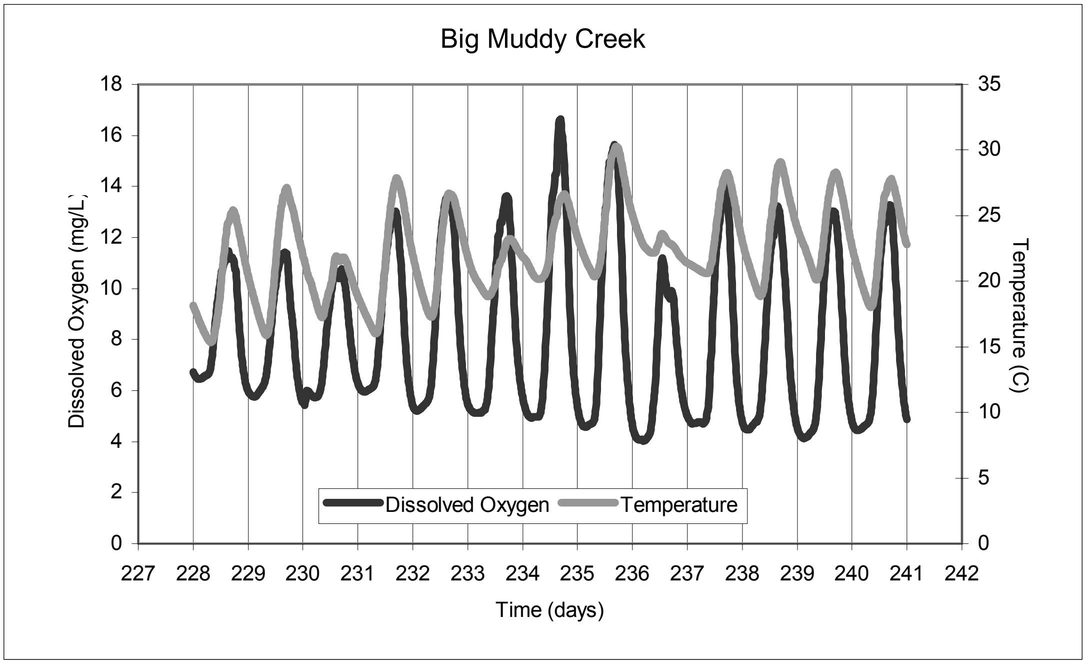
A much larger suite of water quality parameters were measured during this study than those shown in the comparisons presented in Table 5. The general descriptive statistics for these variables are shown in Table 6 and a number of observations are noteworthy. Although all the streams included in the study were drained by watersheds occurring entirely within the WCBP ecoregion, rather large variations in stream water quality was found to exist between stream sites. Omernik's (1987) ecoregion framework was established in part to allow comparative studies of streams draining similar landscapes, but as shown in the following table large variation in water quality can exist even within the same ecoregion.

Table 6. Descriptive statistics for water quality parameters measured in all streams in this study.

Parameter	Mean	Median	Minimum	Maximum
Air temperature (C°)	23.0	23.3	14.5	30.0
Water temperature (C°)	23.2	22.8	15.9	31.9
pH	8.4	8.4	7.8	8.9
Conductivity (mS/cm)	0.58	0.53	0.37	1.35
Turbidity (NTU)	54.7	37.5	6.0	274.5
NO_2 (mg-N/L)	0.0	0.0	0.0	0.3
NO_3-NO_2 (mg-N/L)	3.9	3.2	0.0	13.3
NH_3 (µg-N/L)	96.5	39.9	12.3	1774.4
Organic N (mg-N/L)	0.7	0.4	0.1	4.6
Total N (mg-N/L)	4.7	3.7	0.4	14.1
PO_4 (µg-P/L)	145.4	65.9	10.2	1022.6
Organic P (µg-P/L)	101.9	58.8	13.2	925.7
Total P (µg-P/L)	246.0	138.6	40.2	1856.2
N/P ratio	42.1	21.4	1.0	238.3
Seston chlorophyll *a* (µg/L)	16.6	7.1	1.2	160.1
Seston phaeophytin *a* (µg/L)	8.0	5.1	1.2	65.0
Periphyton chlorophyll *a* (µg/m^2)	32486.2	30239.3	7143.2	75924.5
Periphyton phaeophytin *a* (µg/m^2)	12242.8	8488.2	3430.6	35990.4

Dissolved Oxygen Fluctuation

The DO measurements taken during this study demonstrated three facets of variation: diel (within a day), daily, and spatial variation (stream to stream). Dissolved oxygen flux between streams often displayed great variation, fluctuating along a diel cycle with the higher values occurring in mid afternoon and lows occurring just after midnight of a typical sampling day (Figure 2). The stream fluxes illustrated in Figure 2 show very different values and flux amplitudes even though these streams were sampled under very similar climatic and flow conditions, suggesting that other factors were contributing to the observed DO levels and flux patterns. The mean DO flux amplitude (range) value for Big Muddy Creek was 8.18 mg/L, which was considerably higher than that the mean amplitude for the West Nishnabotna River (2.63 mg/L). The DO flux graphs of Figure 2 in many ways typify the variations of DO flux noted throughout this study and among the different stream systems. However, some streams displayed anoxic and hypoxic conditions (DO levels zero or > 2.0 mg/L, respectively) and variations among DO values, flux patterns, productivity and respiration were apparent when examining the larger set of DO stream sites and values.



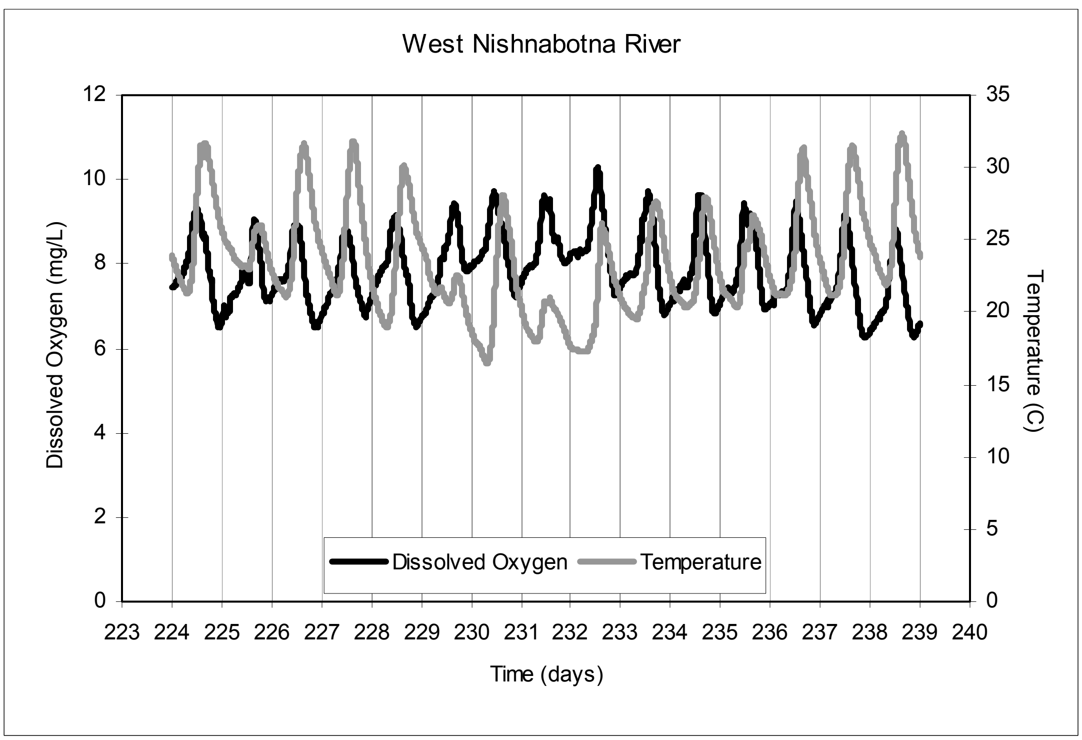


Figure 6. Diel DO flux recorded for Muddy Creek (Clay Co., IA from August 16-28, 2001) and the West Nishnabotna River (Shelby Co., IA from August 12-26, 2000).

Significant differences among stream variations in DO concentrations were associated with the DO fluctuations monitored during the deployment periods of individual streams within the WCBP ecoregion (Table 7). The daily minimums and maximums ranged from early morning DO minimums of 2.9 to 8.6 mg/L to afternoon DO maximums of between 6.5 to 17.6 mg/L. At some stream sites, daily fluxes in DO of nearly 10 mg/L were recorded, although amplitudes of this magnitude were infrequent. The average daily change in DO between afternoon maximums and nighttime minimums for all the stream sites was 4.3 mg/L. Diel variation at each stream site sometimes displayed daily changes in the flux pattern of DO and could have very different daily averages even under apparently stable flows. DO levels exhibited an average daily minimum of 6.0 mg/L during the early morning hours and increased in concentration to a maximum of 10.3 mg/L in the early to mid afternoon hours. The average daily DO concentration over the entire study was 7.7 mg/L, with the range of mean stream site concentrations falling between 4.7 and 11.2 mg/L.

Table 7. Mean, median and range values for DO concentration and saturation values for all streams monitored during this study.

	DO (mg/L)			DO (% Saturation)		
Daily DO flux values	Mean	Median	Range	Mean	Median	Range
Maximum	10.3	9.7	6.5 – 17.6	125	118	77 – 228
Minimum	6.0	6.4	2.9 – 8.6	69	71	30 – 97
Mean	7.7	7.9	4.7 – 11.2	89	90	54 – 137
Flux (amplitude)	4.3	3.8	0.9 –10.2	56	49	9 – 138

Because stream temperatures affect DO levels, oxygen saturation values were examined. Use of saturation values removed the effect of temperature and allowed between-stream comparisons to be made on a standardized reporting variable. Just as DO concentration values varied significantly, so did the percent saturation of dissolved oxygen.

The average saturation value for the entire study was 89%. The mean stream site DO value for percent saturation fluctuated from 54% to 137%. During the daytime, daily DO saturation values rose to an average maximum of 125%. While at night, the average minimum for the study was 69% saturation. Individual stream sites displayed average maximum saturation values from 77% up to a very high 228% of saturation. Like the other mean stream site variables, the mean stream site minimums displayed a similar degree of variability ranging from 30% to 97%.

Stream Production

The Productivity Calculator that is based on Odum's (1956) diurnal DO curve method was used to calculate estimates of net and gross production and stream respiration. The mean gross production for this study based on all measures for all study streams in the WCBP was 4.21 g O_2/m^2/day (Table 8). Mean gross production value for streams ranged from near zero (0.01 g O_2/m^2/day) to an exceptionally high gross production value of 18 g O_2/m^2/day.

Table 8. Summary of net and gross production (g O_2/m^2/day), respiration (g O_2/m^2/day), and production/respiration (P/R) ratio values of the study streams.

	Mean	Range	Median
Net production	-2.30	-14.75 – 14.01	-2.44
Gross production	4.21	0.01 – 18.04	3.70
Respiration	6.50	0.82 – 17.57	5.37
P/R	0.95	0.06 – 10.27	0.53

Individual stream estimates of gross production indicated that even within fairly stable and uniform temporal periods, stream values varied greatly. Mean values for community respiration also varied greatly among streams sites ranging from values below 1.0 to almost 18 g O_2/m^2/day, however the study median was just of 5 O_2/m^2/day. Estimates for net productivity based on all data using either the mean or median values resulted in negative net productivity values of more than 2 g O_2/m^2/day. Consequently, the P/R ratios based on either mean or median values for respiration and gross production were below 1.00 (0.95 – 0.53) indicating that in general these streams were autotrophic in nature. While individual site means for P/R values ranged from 0.06 to 10.27, the vast majority (71%) of streams were autotrophic and all but two sites had mean P/R values above 2.00.

DO and Production Relationships

Watershed Characteristics

A large number of potential relationships between LULC classes and different measures of DO, DO flux, and production were examined using correlation techniques. Of the 17 measured or derived variables of DO and production, and 18 LULC class variables recorded as occurring in the watersheds, very few significant correlations ($p < 0.05$) were found between the two groups. The only DO/productivity variables to show significant correlations with these original LULC class variables were the estimates of production and respiration. It appeared that the number of original LULC classes fragmented the watershed percentages into too many sub-classes, because analytical attempts to relate DO variables to a large number of small LULC classes resulted in diminished power for correlation tests. To explore more general LULC concepts, similar LULC classes were combined to create four composite classes representing urban areas (high human habitation), grass and shrub lands (more complete ground cover), forests (multi-dimensional ground cover and canopy), and cultivated lands (highly developed agricultural lands). As examples, combining similar classes such as small grains with row crop, or deciduous with coniferous and mixed forests, formed the new LULC classes - total crop and forest, respectively. However, these composite classes did not show any stronger or different correlation with stream variables than many of the original variables (Table 9).

Table 9. Significant correlations among large LULC classes and primary production variables. Alpha = 0.05.

		Production Estimates - g O_2/m^2/day		
LULC Class Number	Description	Net	Respiration	Gross
21 + 22 + 23	Urban	-0.3483	0.3316	–
41	Deciduous forest	–	–	-0.3298
41 + 42 + 43	Forest	–	–	-0.3335
81	Pasture/Hay	0.3424	–	–
82	Row crop	-0.3228	0.3112	–
82 + 83	Total crop	-0.3259	–	–

The r values associated with the significant correlations shown in Table 9 indicted weak relationships between land use and stream production.. However, most of these relationships appeared to define plausible causal-based relationships. Certainly, the positive relationship between the urban LULC class and respiration might be expected in streams draining urban landscapes. More organic material (e.g. BOD, biological oxygen demand) may be introduced to these stream systems as a result of urban runoff and sewage treatment plant effluent, thus increasing community respiration as a result of bacterial decomposition of the allochthonous organic matter. As increasing respiration rates consume more DO, an associated decline in net production may be anticipated. Net production did have an inverse correlation with the percentage of urban LULC in the watersheds. Both forest classes were found to have inverse correlations with gross production and one hypothesis is that forest canopy (stream shade) might be a limiting factor of instream gross productivity. Several assumptions would have to be made to accept this hypothesis and even then causality cannot be determined from these correlations. The first assumption is that the amount of forest in the watershed is an indicator of the extent of riparian forest along the stream. Secondly, this riparian forest results in increased canopy cover that causes a decrease in the amount of sunlight reaching the stream. This decrease in sunlight in turn limits photosynthetic production. (i.e. gross production).

Relationships between agriculturally developed lands and production exist as well. Although the positive correlation between pasture/hay and net production seems counterintuitive, an argument can be made in support of this relationship. Depending on management practices, manure from grazing animals can lead to increased stream loads of available phosphorous. Lotic systems in agricultural watershed often receive much of their phosphorus from manure originating from animal confinement areas and pasturelands. As nearly all of these streams appear to be phosphorous limited (high N: P ratios), increases in pasture lands and the potential increase in TP could lead to elevated net production. While the pasture/hay class showed a positive correlation with net production, the percent row crop was negatively correlated with net production and positively correlated with respiration. A possible hypothesis for these relationships is that row crops contributed more organic material and sediment. While row crop did not show a significant relationship with gross production, there may have been some limited relationship related to increased suspended sediment, decreased light penetration and lower gross production which would result in decreased net production even if the respiration rate remained the same (which it would not based on the observed relationship in Table 9). It is more likely that the potential increased transport of organic material to the stream related to increased row cropping resulted in higher respiration rates and subsequently lower net production. All proposed causal relationship were speculative and the true relationships made be only correlative at best.

Physical Characteristics

Few statistically significant relationships among stream discharge, instream habitat, substrate composition, temperature and DO/productivity variables were observed using correlation analyses. Again, the productivity variables tended to better correlate with measures of physical stream properties than DO and DO flux variables.

No significant relationships were observed among DO/productivity and stream macro habitat types (i.e. pool, riffle, run), air and water temperatures, and discharge. One might expect that upstream macrohabitat plays a role in determining DO concentrations associated with down stream reaches. Our hypothesis was that during low, stable flow periods, large upstream pools functioning as highly productive run-of-the-river reservoirs or upstream riffles with their rich periphyton growths contributed large amounts of photosynthically produced DO that resulted in increases in downstream DO concentrations or flux. However, no significant correlations between habitat and DO/productivity variables were found.

It would also be expected that surrounding air and water temperatures plays a role in determining the state of stream oxygen concentrations, and that increases in both air and water temperature correlates to increases in maximum DO concentration, DO flux, and/or gross productivity. But again, no such relationships were found between the variables. This could be a due to sampling procedures that limited air and water temperature measurements to a two-sample average recorded during the installation of the logger and its removal. Thus, the observed relationship is between continuous oxygen measurements and essentially "point-in-time" measures of temperature. This temporal discrepancy is the most likely source for the absence of a significant correlation between temperature and the DO factors.

Discharge was also considered as a factor affecting instream DO, but in this study it too was revealed no relationship. It was hypothesized that a flushing effect due to increased discharge would wash out in stream primary producers leading to decreased oxygen content and lowered production levels. By examining the design of this study, which attempted to eliminate the impact of runoff events, it should be no surprise that discharge played no visible role in affecting DO concentrations or productivity.

In examining the significant relationships, substrate was the only physical measure to correlate with the DO variables. The stream substrate was classified as bedrock, cobble, gravel, sand, or clay/mud and coded 1 through 5, respectively. The correlations in Table 10 suggest that stream sites with stable substrates (larger mass and weight) also have higher maximum DO values, higher difference in the maximum and minimum values, and larger standard deviations in minimum/maximum values. Conversely, sites with bottom substrates of mostly small substrate sizes (e.g. sand, clay) tended to have lower maximum DO concentrations and lower DO difference in maximum and minimum daily values. These data support the premise that large substrates provide stable, higher quality substrate for greater algae growth and production, which in turn contribute to high DO levels and larger flux.

Table 10. Significant correlations between DO variables and stream substrate. Substrate size classes varied from 1 (bedrock), to 2 (cobble) on down to 5 representing clay/mud. Alpha = 0.05.

	DO variables (daily means for deployment periods)					
	Maximum	Difference	Std Dev	% Max	% Difference	% Std Dev
Substrate class	-0.3282	-0.3512	-0.3828	-0.3835	-0.3519	-0.3745

Several physical variables displayed significant correlations with production estimates (Table 11). The correlations with gross productivity seemed to follow a logical pattern. Just as the measures of DO showed an inverse relationship with substrate, so too did gross production. Sites with larger, more stable substrate (and surface area) promoted higher gross production in by facilitating greater biomass accumulation. Finer bottom substrates are less stable than large substrates easily mobilized along the bottom or in the water column. Algae associated with these mobilized particles are physically scoured, often damaged and are transported downstream, thus limiting algal growth and production at a particular stream site or reach.

Table 11. Significant correlations among stream site productivity and physical and hydrological stream variables. Alpha = 0.05.

	Production Estimates			
Description	Net	Respiration	Gross	P/R
Substrate	–	–	-0.3717	–
% Canopy	–	–	-0.3064	-0.3324
Mean Depth	–	–	-0.3057	–
Mean Velocity	–	0.3323	–	–

As expected, increases in percent of riparian canopy were related to decreased gross production and P/R ratios. Gross production and average stream depth also had a negative relationship. Reduced gross productivity with increased stream depth and canopy cover suggests that light limitations were the probable indirect factor limiting gross production.

Water Quality Characteristics

Nutrients along plant pigment concentrations and other water quality measures were correlated with measures of DO and production. These correlations yielded some interesting relationships with the majority of significant correlations occurring between the water quality measures and the DO values. A number of basic statistics (e.g. mean, median) for the raw DO values and percent saturation DO values were compared to the water quality variables. Because no mode values were calculated for the percent saturation values, the mode values calculated for the raw DO values were not included the correlation analysis. The raw DO statistics and the percent DO saturation statistics yielded very similar correlations with the same water quality parameters. Thirty-two significant correlations were observed for the raw DO variables along with 34 correlations for the percent saturation variables. Out of these significant correlations, 29 were between the same water quality variable and the matching statistical measure for the raw DO and percent saturation variables, suggesting that either the variables based on raw DO values or saturation values could be used to examine relationships between DO and other water quality variables as illustrated in Table 12. This table shows that the correlation values and sign relationships between raw and saturation derived variables and nitrate/nitrite, N:P ratios and chlorophyll *a* concentrations are nearly identical. In addition, the examination of correlation matrix for raw and percent saturation variables showed that there were virtually no differences between these variables as they were highly correlated. The mean difference between correlation pairs (raw verses % saturation) for the 29 common DO variables was only 0.005. Therefore, in order to eliminate some of the redundancy in the analyses, only the correlations between the water quality variables and the raw DO statistics were examined in any detail.

Table 12. Comparative correlation table (r values) between raw DO and percent saturation DO variables and selected water quality parameters. Significant correlations are shown in italics. Alpha = 0.05.

	Raw DO Statistics					
	Mean	Maximum	Minimum	Difference	Median	Std Dev
NO_3-NO_2	*0.357*	0.079	*0.441*	*-0.295*	*0.402*	-0.234
N/P	*0.411*	0.237	*0.362*	-0.078	*0.412*	-0.005
Chl *a*	0.019	0.199	-0.115	*0.359*	-0.015	*0.305*
	Percent DO Saturation Statistics					
	Mean	Maximum	Minimum	Difference	Median	Std Dev
NO_3-NO_2	*0.322*	-0.015	*0.443*	*-0.290*	*0.401*	-0.243
N/P	*0.338*	0.120	*0.337*	-0.104	*0.353*	-0.033
Chl *a*	0.077	0.276	-0.179	*0.397*	0.032	*0.325*

Nitrate/nitrite values showed significant, positive correlations with the mean, median and minimum DO, indicating that increases in nitrate (nitrite levels in most streams are extremely low) were related to increased levels of stream DO, including higher minimum values. These relationships might be the result of increased stream production or algal biomass stimulated by higher nitrate values. The N:P ratio was also positively correlated with the same DO variables. Chlorophyll *a* concentrations were significantly correlated with the difference (r = 0.359) between the mean daily high and low DO values. This might have been the result of increased maximum values (r = 0.199) due to photosynthesis, which increased the spread or difference in high/low values. Most raw DO variables were found to be correlated with conductivity but not pH, turbidity or the Horiba DO measurements (Table 13). These water quality parameters were sampled at the beginning and end of the DO logger deployment period using the Horiba U-10 Water Quality Checker. The limited temporal variation captured in the mean of water quality variables based on two samples may have prevented the identification of significant correlations with the DO variables that were measured at both a daily (diel) and longer time sequence. The single daily measures of DO were predominantly taken during the afternoon hours when oxygen production is at its highest. This most likely resulted in the significant positive correlation with maximum DO and none of the other DO variables. During periods of increased oxygen production, streams will become less acidic as primary producers remove carbon from the water for photosynthesis. This is shown by the significant positive correlation between pH and the DO flux variables. The fact that pH correlated better with the measures of DO flux was because pH follows a cyclic pattern that was expressed by these variables. Conductivity exhibited significant negative correlations with the DO variables. Conductivity is closely correlated with total dissolved solids, which are typically comprised of organic substances, a potential source of BOD. The BOD may have then reduced DO concentrations. However, the positive relationship between DO and stream conductivity is purely conjecture and based upon several sound but unsubstantiated assumptions.

Table 13. Correlation table (r values) between DO and water quality variables. Significant correlations are shown in italics. Alpha = 0.05.

	Raw DO Statistics					
	Mean	Maximum	Minimum	Differences	Median	Std Dev
pH	-0.016	0.206	-0.191	*0.362*	-0.114	*0.325*
Conductivity	*-0.352*	*-0.300*	*-0.367*	-0.027	*-0.361*	-0.087
Turbidity	-0.103	-0.066	0.020	-0.044	-0.059	-0.103
DO (single daily measure)	0.281	*0.332*	0.191	0.222	0.251	0.269

Examination of significant correlations between DO and the measures of nitrogen suggested that nitrogen does not affect DO flux. Most nitrogen species except for ammonia were positively correlated with minimum, mean and median DO (Table 14). Increases in nitrate as well as inorganic N and TN would raise the level of minimum DO but not maximum, thus increasing the mean and median DO (see Tables 13 and14). Also, the rise in only the minimum DO was the apparent cause for the maximum/minimum difference to display a negative relationship with nitrate. The correlations with the inorganic component of nitrogen followed the same pattern as those with nitrate largely because the dominant constituent of inorganic nitrogen is nitrate. Total nitrogen also showed a positive relationship with the minimum and median values of DO. The median relationship was no doubt a byproduct of the increase in the minimum DO measure. Ammonia was negatively correlated with minimum, mean and median DO levels; the strong correlation (r=-0.467) was between mean DO and ammonia. These negative relations might be the result of several instream processes. As Peterson et al. (2001) showed, ammonia is removed from the stream and assimilated by photosynthetic plants and heterotrophic organisms. In addition, microbial organisms living in anoxic conditions within the streambed or in deep pool reaches may produce ammonia through the reduction of nitrogen.

Table 14. Correlation table (r values) between DO and nitrogen variables. Significant correlations are shown in italics. Alpha = 0.05.

	Raw DO Statistics					
	Mean	Maximum	Minimum	Differences	Median	Std Dev
NO_3-NO_2	*0.357*	0.079	*0.441*	*-0.295*	*0.402*	-0.234
NO_2	0.072	0.033	0.109	-0.120	0.097	-0.106
NH_3	*-0.467*	-0.253	*-0.430*	0.035	*-0.455*	-0.010
Inorganic N	*0.338*	0.073	*0.417*	*-0.292*	*0.381*	-0.235
Organic N	0.118	0.141	0.038	0.161	0.105	0.157
Total N	0.267	0.023	*0.359*	-0.270	*0.316*	-0.242

Few correlations were found between phosphorus variables and the various measures of DO (Table 15). All significant correlations with phosphorus were negative. Increases in $P0_4$ concentrations resulted in lower maximum, mean and median DO levels. The relationships with the median and average DO values were most likely a consequence of the negative relationship with maximum DO. Total phosphorus also exhibited a negative relationships with mean DO levels. These data suggest that increases in stream phosphorus values could lead to lower stream DO levels. This idea is supported by the observed correlations between nitrogen and phosphorus ratios (N/P) and mean, minimum and median DO.

Table 15. Correlation table (r values) between DO and phosphorus variables and the N:P ratio. Significant correlations are shown in italics. Alpha = 0.05.

	Raw DO Statistics					
	Mean	Maximum	Minimum	Difference	Median	Std Dev
N:P	*0.411*	0.237	*0.362*	-0.078	*0.412*	-0.005
PO_4	*-0.390*	*-0.400*	-0.203	-0.220	*-0.346*	-0.259
Organic P	-0.059	-0.022	-0.000	0.052	-0.027	-0.033
Total P	*-0.294*	-0.273	-0.132	-0.130	-0.244	-0.184

All significant correlations between N:P and the DO statistics were positive indicating that as the N:P ratio increased so did the average, median, and minimum DO statistics (Table 15). However, the most important relationship with the N:P ratio is the positive relationship with minimum DO values. High N:P values indicates that the system is likely to be phosphorus limited, meaning that in these systems as N:P increase, DO values increase and phosphorus decreases. What becomes apparent after examining these correlations between DO and phosphorus is that limiting, phosphorus is not controlling the increase or fluctuation of DO, but appears to be related to decreases in DO.

Few significant correlations between water quality variables and estimates of stream production and respiration were observed in this study. Those few water quality variables that did show a significant correlation with at least one stream productivity variable are included in Table 16. The average of the two instantaneous measures of DO (single point in time measures) taken with the Horiba U-10 Water Quality Checker at the beginning and end of the logger deployment period correlated fairly well with gross production. This relationship might have been the result of the fact that high productivity rates (gross production) often occur in mid- to late afternoon with increased solar radiation and water temperatures, which coincided with the typical mid-day measurement of stream DO with the Horiba. Since gross production is a major component in the calculation of both net production and the P/R ratio, it is reasonable to assume that correlations between these estimates and instantaneous DO were a result of the relationship between gross productivity and the DO variable. The positive relationship between pH and gross production was assumed to be related to the uptake of carbon mostly from CO_2 as a result of photosynthesis and thus a reduction in carbonic acid.

Table 16. Correlations (r values) between the water quality variables and estimates of stream production, respiration and the P/R ration. Significant correlations are shown in italics. Alpha = 0.05.

	Production and Respiration Variables			
	Respiration	Gross production	Net production	P/R ratio
pH	0.131	*0.333*	0.118	0.209
Conductivity	*0.372*	0.084	-0.269	-0.191
Turbidity	-0.209	-0.109	0.129	0.116
DO (single point in time)	-0.182	*0.447*	*0.313*	*0.391*
NH_3	*0.391*	-0.186	*-0.503*	*-0.402*

Ammonia was the only nutrient variable to significantly correlate with the estimates of production and respiration. It appears that increases in stream ammonia are related to increases stream respiration, but it is unclear what this relationship represents. A similar relationship was observed between ammonia and minimum stream DO (Table 14) where increases in ammonia were correlated to decreases in DO. These two relationships suggest that increased ammonia values are related to increased respiration that leads to lower instream DO level (al least lower minimums). One explanation might be that increases in heterotrophic activity results in increased respiration rates, and ammonia levels rise as a result of the uptake and conversion of nitrate to ammonia by the same heterotrophic community.

Seston and Benthic Chlorophyll

No benthic chlorophyll or phaeophytin variables were significantly correlated with any of the DO variables, suggesting that periphyton biomass was not related to DO levels in these streams. This does not mean that benthic chlorophyll does not contribute to instream oxygen levels, but rather that no relationships were observed during these low-flow sampling events. Additionally, there were few significant correlations found between sestonic chlorophyll concentrations and stream DO (Table 17). Sestonic chlorophyll, phaeophytin and the chlorophyll/phaeophytin ratio were positively correlated with one or both of the measures of DO flux amplitude (e.g.the difference and standard deviation variables). Overall, these rather weak relationships suggest that as sestonic algal biomass increases so does DO flux, probably because of increased photosynthetic processes. Phaeophytin, a breakdown product of chlorophyll, reflect episodes of high algal growth. When the proportion of chlorophyll to phaeophytin is high, algal populations experience high growth rates or low senescence. The positive correlation between the difference variable and the chlorophyll/phaeophytin ratio indicates that growth in suspended algae populations lead to an increase in DO fluctuations.

Table 17. Correlation table (r values) between DO and benthic algal pigment variables. Significant correlations are shown in italics. Alpha = 0.05.

	Raw DO Statistics					
	Mean	Maximum	Minimum	Difference	Median	Std Dev
Chlorophyll *a*	0.019	0.199	-0.115	*0.359*	-0.015	*0.305*
Phaeophytin *a*	0.029	0.169	-0.106	*0.285*	-0.009	0.245
Chlorophyll/Phaeophytin Ratio	0.012	0.209	-0.084	*0.386*	-0.012	*0.341*

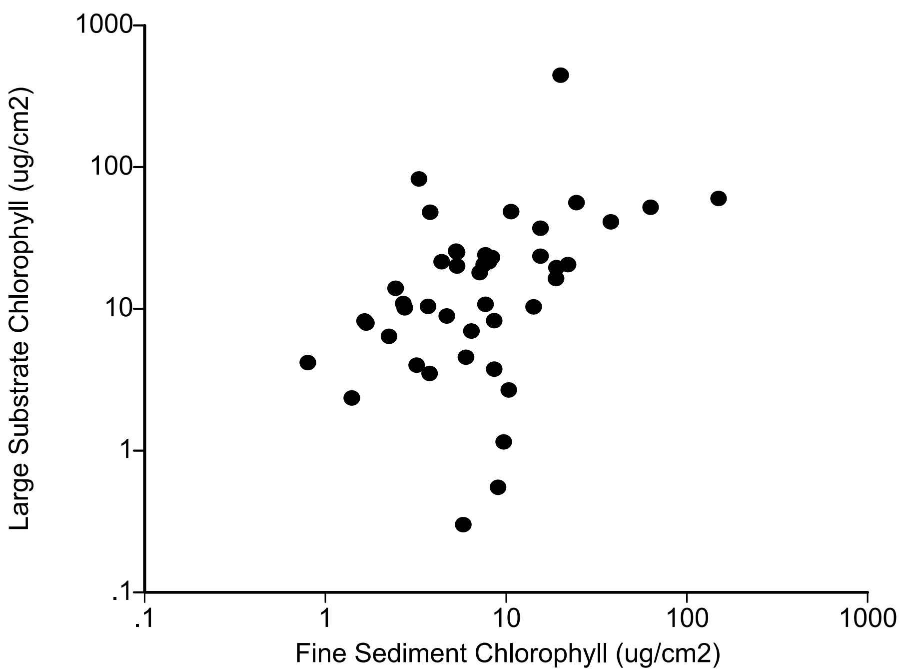


Figure 2. Scatter plot of chlorophyll a concentration obtained from samples taken from large (e.g. cobble, logs) and fine (e.g. sand, silt) substrates within the same stream sites (n = 44)

Co-collected benthic algal collections were analyzed to determine if periphyton communities colonizing different substrates were similar or different. If substrate size did promote the establishment of different periphyton communities then comparisons of sites with different bottom substrates would have to be evaluated with care. Data collected by Iowa Department of Natural Resources (IDNR) and provided for this effort were analyzed to evaluate periphyton relationships between communities occurring on large and fine substrates found at the same sites. A scatter plot (Figure 17) of these data suggests a weak positive relationship between benthic chlorophyll *a* concentrations on large verses small substrates. However, both regression and correlation analyses indicated that there was no statistically significant relationship between these periphyton concentrations.

Macroinvertebrates

Again using IDNR productivity, DO and macroinvertebrate data, a number of analyses were preformed. Examination of the Spearman correlation matrix for these data revealed only two significant correlations when examining macroinvertebrate variables and DO and productivity measures. No significant correlations (Spearman-rank correlation, alpha = 0.05) were observed between percent EPT (Ephemeroptera, Plecoptera, Trichoptera) and DO or productivity variables. However, both mean daily respiration and consequently net production were correlated with IDNR's modification of the Hilsenhoff Biotic Index (Hilsenhoff 1987). A significant positive correlation (r = 0.546) was found between the modified Hilsenhoff Biotic Index scores (possible range of 1 to 10) and respiration indicating that increases in stream respiration rates negatively affected the macroinvertebrate community. A scatter plot with both a regression line and smoothed fit line was constructed and the results are shown in Figure3. A threshold affect was apparent between a respiration rate of 9 to 10 g O_2/m^2/day, above which increased respiration rates do not appear to influence the Hilsenhoff Biotic Index score; that is the macroinvertebrate community condition does not continue to degrade.

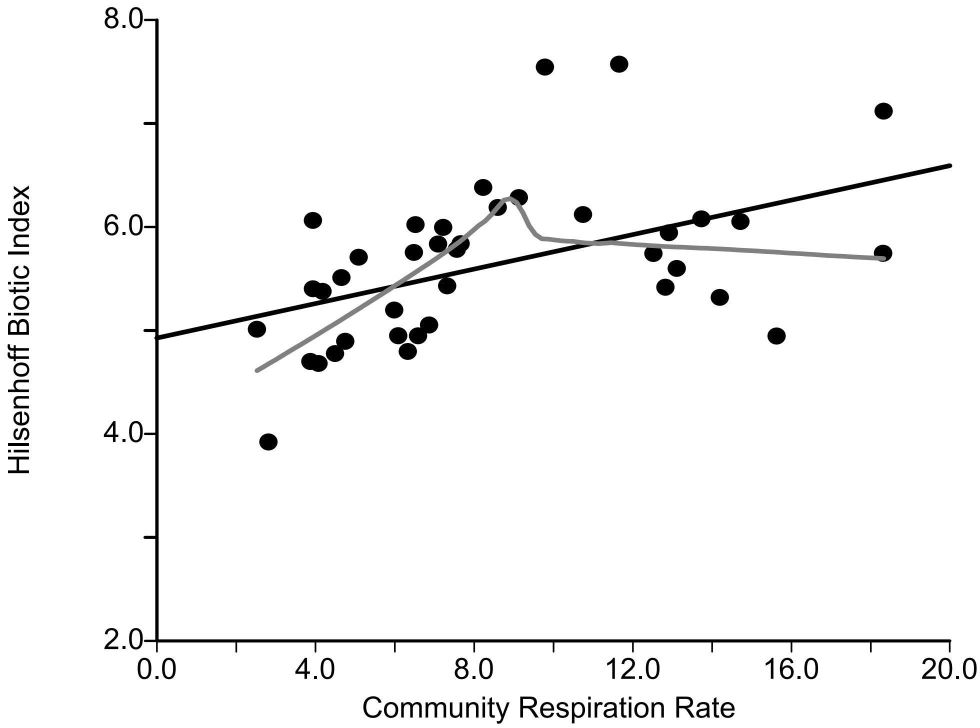


Figure 3. Regression (dashed line = leasted-squares trend line) and smoothed (solid grey line = LOWESS) fit relationships between the Modified Hilsenhoff Biotic Index and community respiration rates (g O_2/m^2/day).

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