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An assessment of long-term post-restoration water quality trends in a shallow, subtropical, urban hypereutrophic lake

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Abstract

City Park Lake is a shallow urban hypereutrophic lake located in Baton Rouge, Louisiana, with a surface area of 0.23 km^2 and a mean depth of 1.2 m. By the late 1970s, the lake had become highly eutrophic and suffered from frequent and severe algal blooms and fish kills. A major restoration effort was undertaken in 1983 that consisted of dredging and the repair of sewage infrastructure. Immediate improvements in water quality were observed following restoration; algal blooms and fish kills were virtually eliminated for nearly a decade. However, large floating mats of filamentous algae periodically occurred during the early 1990s. Results of a water quality sampling program conducted in 2000 and 2001 indicated that phosphorus has once again reached pre-restoration levels, and nitrogen levels have decreased well below those observed during pre-restoration years. Whereas phosphorus-limited conditions predominated in the years preceding the 1983 restoration, results of the 2000–2001 sampling program indicate that the lake has become nitrogen-limited with respect to photosynthetic activity. This trend in nutrient levels has likely influenced the recent predominance of filamentous over unicellular species of algae observed during the last decade. Nearly 4 years of drought-like conditions beginning in 1998 have resulted in an overall increase in the hydraulic retention time of the lake. This condition has resulted in organic staining of the lake waters, or the development of a tea-like color due to the decomposition of organic compounds. This phenomenon has played a major role in inhibiting the sunlight available for filamentous algal growth since 1998 and the absence of filamentous algae during the 2000–2001 sampling program.

Keywords: Filamentous algae; Lake restoration; Eutrophication; Assessment; Subtropical; Nutrients; Organic color

1. Introduction

The establishment of the Clean Lakes Program (Section 314 of the Federal Water Pollution Control Act) in the USA in 1972 signaled a commitment at the federal level to our nation's lakes and reservoirs. While the initial scope of the program was narrow, primarily focused on the water bodies themselves in contrast to today's focus on entire watersheds, the program succeeded in focusing attention on the level of 'cultural eutrophication' that had taken place in US lakes. Since 1976, more than $145 million in funds have been awarded for the restoration of severely

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eutrophic and deteriorated lakes (USEPA, 2001). Although program funding aided the restoration of lakes and reservoirs in 49 states, the majority of research programs and assessments of restoration effectiveness have focused on deep lakes located in northern, temperate climates.

More than two decades have passed since the initial lake restoration projects, but little documentation is available detailing the long-term effectiveness of these efforts on the water quality of individual lakes. In contrast to today's restoration programs that not only focus on the water body, but on the management of the surrounding watershed, many of the restoration projects initially funded by the Clean Lakes Program were viewed as one-time, problem-solving efforts. Presently, more data and knowledge are available to establish the relationship between restoration techniques, watershed management, and the water quality of lakes. Still, relatively few studies have focused on the restoration of shallow, subtropical, hypereutrophic lake systems.

The focus of this paper is to present the assessment of the long-term, post-restoration water quality trends and the effects of limited implementation of watershed management strategies on a shallow, subtropical, urban hypereutrophic lake. The specific objectives of this study were: (1) the collection and analysis of City Park Lake water quality data during 2000–2001; (2) an assessment of the current status of the lake with respect to trophic state; and (3) an assessment of the post-restoration water quality trends based on available historic data.

2. Study site

Located in Baton Rouge, Louisiana, City Park Lake is approximately three kilometers (km) east of the Mississippi River and 105 km north of the Gulf of Mexico. Baton Rouge is located in the subtropical climate zone at an approximate elevation of 9 m. The city's 30-year normal precipitation is 155 cm, and rainfall is most frequent during July and August. Severe storms are most frequent during the spring months. The average normal temperature (based on a 30-year period) is 19.8 °C with an average low of 9.9 °C and an average high of 27.9 °C (NOAA, 2001). Baton Rouge is subject to polar fronts during the winter months and winds are normally light, averaging less than 16 km/h on an annual basis.

City Park Lake is one of six lakes that comprise the University Lakes System. It was constructed during the early 1930s on land that was originally characterized as cypress swamp. Its construction coincided with the establishment of Louisiana State University (LSU) and the City Park Lake golf course. The primary purposes of the lake's construction were the elimination of stagnant mosquito-breeding areas and the establishment of a recreational and educational resource for the community.

As a shallow lake with an average depth of 1.2 m and an approximate surface area of 0.23 km^2, City Park Lake has a mean hydraulic retention time (HRT) of 56 days (Malone et al., 1991). The fetch of the lake is oriented from north to south (Fig. 1). Bayou Duplantier, located at the north end of the lake, serves as the primary source of storm water flows to the lake. A sharp-crested weir serves as the lake's principle spillway and is located on the south end of the lake. The lake is located on relatively recent alluvial deposits of the Mississippi River and at the transition between the nearly level silty soils of the Prairie formation and the gently sloping silty soils of the Montgomery formation (SCS, 1968). The Montgomery formation is characterized by ridges that average approximately 6–12 m in elevation above those of the Prairie formation soils. The original foundation of the lake was formerly the site of a cypress swamp and is comprised of slowly permeable organic clays and silty clays (City-Parish of Baton Rouge, 1977). Lake sediments are primarily silty in texture and relatively unconsolidated. The watershed is comprised of Loring silt loam (LoA, LoB, LoC2) with level to five percent slopes and Olivier silt loams (O1A, O1B) with level to three percent slopes. The Loring and Olivier silt loam soils belong to the Alfisols Order. As such, Alfisols are high in bases and have a B-horizon that is rich in clay (SCS, 1968). Terraced escarpments are common along the northern perimeter of the lake watershed. As silt loams, these soils range from

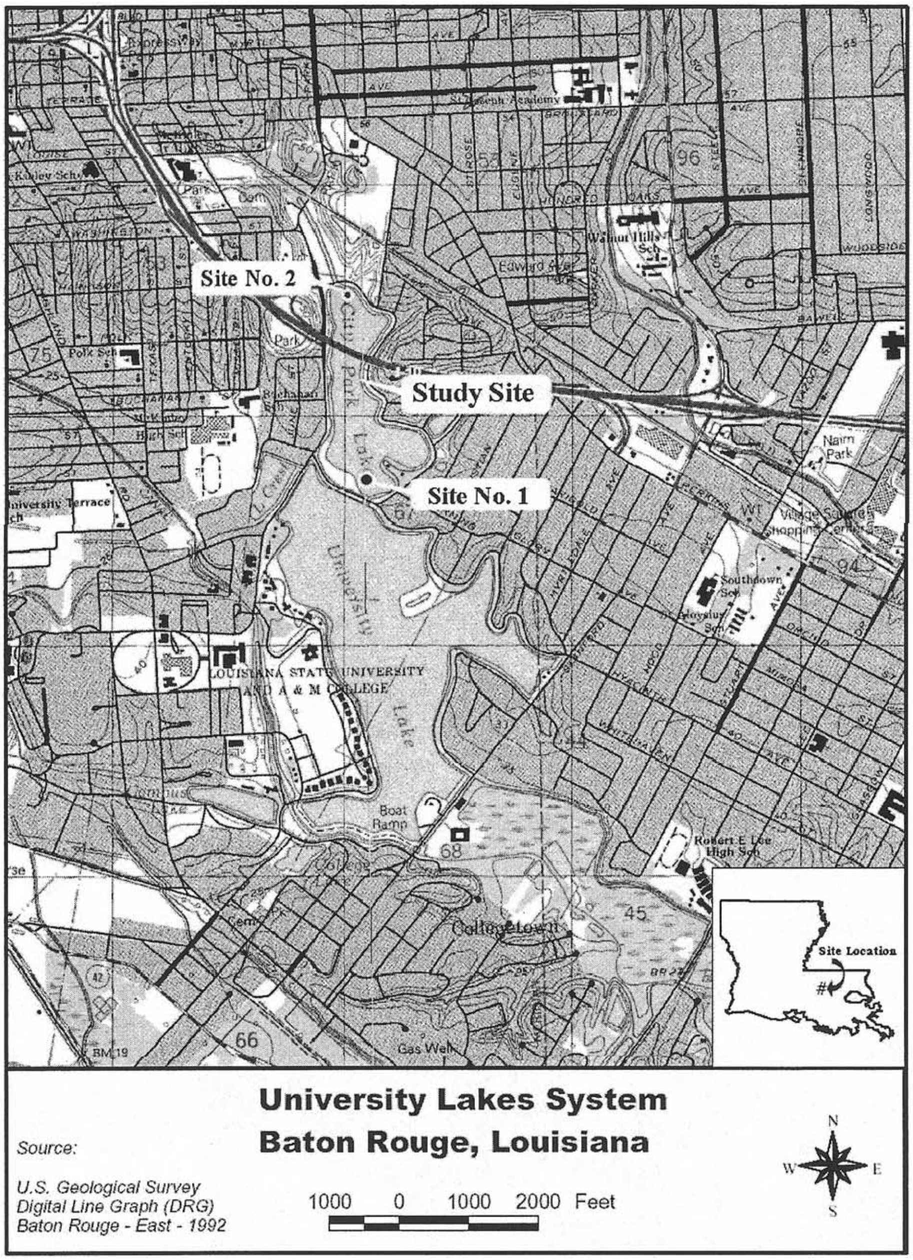

Fig. 1. Map of study area and University Lakes System.

poorly to moderately well drained and slowly to moderately permeable.

The lake watershed is mostly urban, with a total drainage area of 1.922 km^2. The majority of the watershed is comprised of single-family housing (44.2%), with City Park Lake golf course being the second largest land use (Table 1). The vegetated portion of the watershed is dominated by oak and pine trees and bermuda and bahaia grasses. The lake is traversed by Interstate-10 and a heavily-traveled railroad, which serve as potential sources of anthropogenic pollutants in the lake. The lake

Table 1
Land uses in the City Park Lake watershed

Land use	Area (km^2)	% of total watershed area
Streets[a]	0.219	11.4
Interstate-10[a]	0.049	2.5
Railroad[a]	0.012	0.6
Single family housing[a]	0.850	44.2
Apt./multi. family housing[a]	0.012	0.6
Institutional[a]	0.045	2.3
Commercial[a]	0.045	2.3
Repair/manufacturing[a]	0.004	0.2
Park/open[a]	0.401	20.9
Other open[a]	0.057	3.0
Lake[b]	0.230	12.0
Total	1.922	100.0

[a] Reich Associates, 1991.
[b] Malone et al., 1991.

itself accounts for 12.0% of the total watershed. Land uses within the watershed have changed very little over the past 30 years.

In the decades that followed its construction, City Park Lake served as a recreational resource for fishing, boating and bird watching. Historically, largemouth bass, crappie and other sunfish were stocked and fishing was common (City-Parish of Baton Rouge, 1977). As water quality deteriorated in the 1950s and 1960s, fish species that are commonly referred to as 'trash' fish, mainly threadfin and shad, became more abundant. In 1951, water lilies had to be removed from the lakes (Reich Associates, 1991). Swimming was banned in 1957, reportedly due to sewage contamination from neighboring residences. Between 1957 and 1978, frequent fish kills were observed on the University Lakes primarily due to oxygen depletion from algal decay.

3. Overview of restoration activities

3.1. Restoration of City Park Lake

The main objectives of the 1983 restoration were to increase the overall depth of the lake, to remove phosphorus-laden sediments through dredging, and to reduce nutrient loads through the repair of leaky sewer pipelines (City-Parish of Baton Rouge, 1977). By deepening the lake, researchers and local officials anticipated that the eutrophic symptoms of the lake would be minimized or eliminated by reducing the nutrients available to primary producers, creating a profundal zone where organic materials would concentrate and decay, and increasing the oxygen storage capacity of the lakes, thereby resulting in less frequent fish kills (City-Parish of Baton Rouge, 1977). Initial water quality and biological assessments were conducted in conjunction with researchers at Louisiana State University, and bathymetric surveys were performed in each lake. Funding for the restoration was approved in 1978 in the total amount of US $3 million as part of the Clean Lakes Program, with equal funding from the City-Parish of Baton Rouge and the US Environmental Protection Agency. Ancillary funding on the part of local agencies was estimated at $1.5 million (Knaus and Malone, 1984). Dredging of City Park Lake occurred between February and June of 1983. Routine water quality sampling occurred continuously between 1979 and 1984, with brief interruptions during dredging.

Due to concerns over contamination from heavy-metals in City Park Lake's sediments, a 'dredge-skim' method was used to reduce potential impacts in spoil disposal areas. Accordingly, upper layer sediments that were more likely to be contaminated were pushed to one side of the lake, and deeper, less contaminated sediments were excavated. Contaminated sediments were then deposited in the deeper recesses and covered with less contaminated sediments. The remaining sediments were used to form a beach area on the southern end of the lake. In total, approximately 100 000 m^3 of sediment were dredged, resulting in a net increase in the mean depth of approximately 0.3 m (Knaus and Malone, 1984). A comparison of pre- and post-restoration lake characteristics is shown in 2.

The 1983 restoration resulted in immediate improvements in the oxygen storage capacity of the lake and improved transparency in the water column (Gremillion et al., 1985). With the exception of a large filamentous algal bloom that

Table 2
Summary of pre- and post-restoration characteristics of City Park Lake

Characteristic	Pre-restoration	Post-restoration
Surface area (km^2)	0.24	0.23[a]
Average depth (m)	0.9	1.2
Hydraulic retention time (days)	47	56

[a] Decrease in surface area due to in-lake spoil disposal. (Source: Knaus and Malone, 1984).

occurred the year following the completion of the restoration project, the frequency of algal blooms was diminished during the remainder of the 1980s (Gremillion et al., 1985; Malone et al., 1991). However, beginning in the 1990's, several large outbreaks of floating filamentous algae provided indication that the water quality of City Park Lake had degraded since restoration. Public concern over thick mats of floating filamentous algae resulted in local news coverage when the algae were observed along large stretches of the City Park Lake shoreline in September of 1993 (Shinkle, 1993). During this notable event, floating algae were approximated to extend 90 meters from the shoreline. Unlike unicellular algae, which spend the majority of their lifecycle suspended in the water column, filamentous algae originate in the sediment surface where nutrients are in great abundance. These algae are buoyed to the surface of the lake by decompositional gases such as methane (CH_4) and hydrogen sulfide (H_2S). In subtropical climates, mats of filamentous algae often exist on the lake surface for the duration of the growing season.

3.2. Previous assessments of restoration effects on water quality for City Park Lake and other lakes in the University Lakes System

Previous studies of the University Lakes System primarily focused on the short-term effectiveness of dredging and sewage diversion in improving water quality (Knaus and Malone, 1984; Gremillion et al., 1985). In all cases, the effectiveness of the restoration was evaluated based on data collected up to 1984. In 1986 a Lakes Commission was established to address continued management of the University Lakes, and a City-Park/University Lake Management Plan was ultimately developed (Reich Associates, 1991). An extensive water quality monitoring program was implemented in 1990–1991 to evaluate the status of the University Lakes system nearly one decade following restoration (Malone et al., 1991). Malone et al. (1991) reported that total phosphorus (TP) levels fell slightly between pre-restoration and 1991, and algal density dropped dramatically. Overall, nutrient levels were found to remain high. The control of mass loadings from Bayou Duplantier was suggested as a potential focal point for reducing the phosphorus levels in City Park Lake. In addition, fecal coliform counts were very high due to continued problems with sewage leakage. This source of contamination suggests that sewage leaks might continue to serve as yet another source of phosphorus.

Since restoration, few or no improvements in watershed and lake management have occurred with respect to City Park Lake or the other lakes in the University Lakes System. A small retention pond and strategic erosion control structures were constructed during the late 1990s on the City Park Lake golf course as an attempt to minimize the impact of runoff to the lake.

4. Methods and materials

The 2000–2001 sampling and analysis program was designed to be consistent with sampling and laboratory methods used by Malone et al. (1988, 1991). Exceptions include the collection of data at two sites versus one and the measurements of *in situ* and analytical parameters using modern equipment and recent *Standard Methods* (APHA, 1998). Two sampling sites, one near the outlet end of the lake (Site 1) and one near the inlet end of the lake (Site 2), were designated for the collection of routine water quality samples (Fig. 1). The northing and easting positions for each sampling site were recorded with the aid of a hand-held Rockwell Global Positioning System (GPS). The depths at each site were measured using a fiberglass engineering survey rod. Based on the observation of seasonal thermal stratification in many southern lakes and on the previous, documented water quality analyses conducted in 1990 (Malone et al., 1991), it was determined that separate *in situ* measurements and samplings may still be required to account for vertical heterogeneity. Water samples were collected at each site at depths of 0.3 m below the water surface (Top) and 0.3 m above the sediment (Bottom). Bottom samples were obtained using a fabricated PVC bailer.

Top and Bottom *in situ* measurements and analytical data were collected for Sites 1 and 2 in City Park Lake between June 2000 and June 2001. The sites were sampled twice-per month from March to October, and once-per-month from November to February. *In situ* measurements of air temperature, water temperature and dissolved oxygen (DO), pH, and transparency were performed with a YSI 95 hand-held dissolved oxygen and temperature meter, an Orion pH meter, and a 25 cm diameter Secchi disk, respectively.

Water samples were analyzed in LSU's Department of Civil and Environmental Engineering's Water Quality Lab for total phosphorus (TP) [Method 4500-P E], ortho-phosphate (OP) [Method 4500-P E], total Kjeldahl nitrogen (TKN) [Method 4500-N_{org} C], total ammonia nitrogen (TAN) [Method 4500-NH_3 F], nitrite + nitrate–nitrogen ($NO_2 + NO_3$–N) [Method 4500-NO_3 E], Chlorophyll-*a* (Chl-*a*) [Method 10200 H], total suspended solids (TSS) [Method 2540 D], and volatile suspended solids (VSS) [Method 2540 E]. All water quality analyses were performed in triplicate and in accordance with *Standard Methods* (APHA, 1998). TSS, VSS, TKN and TP analyses were performed on unfiltered samples. TAN and $NO_2 + NO_3$–N analyses were performed on samples that were filtered through 1.2 μm glass fiber filters. Chl-*a* samples were filtered through 0.7 μm glass fiber filters.

The statistical analyses of *in situ* and analytical data and data correlations were performed using Microsoft Excel and Statistical Analysis System (SAS) software (Version 8). Arithmetic means, standard error, ranges and the number of samples were calculated for the 2000–2001 *in situ* and analytical data. Geometric means of trophic state variables were calculated for the 2000–2001 sample period in order to compare the variables to those reported by the Organization for Economic Cooperation and Development (OECD) (Table 5). The significance of data correlations between TP and Chl-*a* concentrations and Secchi depths and Chl-*a* concentrations were evaluated for the pre- and post-restoration periods. Log transforms of the data were calculated for consistency with methods used by other researchers (Dillon and Rigler, 1974; Rast and Lee, 1978; Bartsch and Gakstatter, 1978; Smith and Shapiro, 1981). The Pearson product moment correlation coefficient (r), the linear regression equation, and the number of samples were calculated for each correlation. Probability (P) values were calculated and used as a basis for evaluating the significance of the data correlations.

5. Results and discussion

5.1. General observations and characteristics of City Park Lake

During the summers of 2000 and 2001, no floating filamentous algae was observed on the surface of City Park Lake. Filamentous algae were observed, however, growing in the sediment along very shallow portions of the lake, particularly on the north and south shores. Coincidental to the absence of floating algae, the lake water was observed to be rich in organic color, in contrast to relatively clear water conditions in previous years. The darker water color in City Park Lake and the absence of floating algae during 2000 and 2001 was attributed to below normal precipitation amounts, which resulted in reduced dilution and increased retention of decompositional products.

In 2000, Baton Rouge experienced 97 cm of precipitation (Louisiana Office of State Climatology, 2001). In contrast, the normal annual precipitation for Baton Rouge, based on 30 years of precipitation data, is approximately 155 cm (NOAA, 2001) (Fig. 2). The precipitation deficiency in 2000 indicated a continuation of drought-like conditions that began in 1998. During each of the years since the beginning of the drought, floating filamentous algae has been absent from City Park Lake. The mean HRT for City Park Lake increased to approximately 89 days, or 1.6 times the normal HRT, based on the 2000 precipitation amount. The floating filamentous algae that occurred during the early 1990s coincided with normal to above-normal annual precipitation amounts.

Extensive phytoplankton surveys were conducted in 1977, 1979 and 1980, presumably during periods of relatively large algal growth in City Park Lake. The principal phytoplankton divisions identified in City Park Lake during the 1977 growing season included, in order from most abundant to least abundant, cyanophytes, chlorophytes, chrysophytes and euglenophytes (City-Parish of Baton Rouge, 1977). Predominant algal taxa documented during 1980 included *Spirulina* sp. (cyanophyceae), *Microcystis* sp. (cyanophyceae), *Scenedesmus quadricauda* (chlorophyceae), *Lyngbya* (cyanophyceae), Coccoid Green Cells, Centric diatoma, *Micractinium pusillum* var. *elegans* (chlorophyceae), *Nitzscia* spp. (bacillariophyceae), *Merismopedia tenuissima* (cyanophyceae), *Anaebaena spiroides* var. *contracta* (cyanophyceae), and *Golenkina radiata* var. *brevispina* (chlorophyceae) (Malone *et al.*, 1998). Three to four fish kills were documented to have occurred per year due to the collapse and decay of large algal populations in years immediately preceding restoration (Gremillion et al., 1985). During each year for which floating algae were documented (1977, 1979 and 1980), precipitation amounts exceeded the 30-year normal precipitation. Due to discontinuous records of algal blooms and taxa between 1977 and 2000, the historic relationship between rainfall amounts and the occurrence of floating algae remains inconclusive. However, years for which algal populations were documented in City Park Lake either coincided with or were preceded by normal to above-normal precipitation amounts.

Based on the small quantities of filamentous algae below the water surface during the 2000 and 2001 growing seasons and relatively high phosphorus concentrations, growth of the algae was likely limited more by light absorbance in the water column than by nutrient limitation. As is typical of most biological reactors, increased HRT results in a longer contact time between microorganisms and substrates. Since 1998, City Park Lake has acted much like a fed-batch reactor; i.e. storm flows through the lake are less continuous than in normal years. Due to the longer retention time, the biological decomposition of substances in storm flows and detritus in the lake can continue into digestion stages that are not commonly observed during normal precipitation years. The products of respiration and photosynthesis are, therefore, flushed less frequently in years for which precipitation amounts are below normal, resulting in 'chemical perturbation' (Stumm and Morgan, 1981). Chemical perturbation occurs when the ratio of photosynthetic production to respiration is less than one, thereby resulting in the accelerated growth of heterotrophic organisms and the subsequent release of biologically degradable organic substances. The decomposition of biological particles such as algae, plants, leaf litter and bark in aquatic environments results in the release of humic substances, such as tannins, which are characterized by complex formations, adsorbability and color (Stumm and Morgan, 1981). Like other natural organics, tannins are slowly degradable and can remain for long periods of time in retention basins such as lakes. Specifically, tannins contribute to the 'true color' of the lake water and, therefore, cannot be removed through filtration. The presence of tannins reduces the amount of sunlight that is available for photosynthetic activity. This reduction in sunlight is especially critical

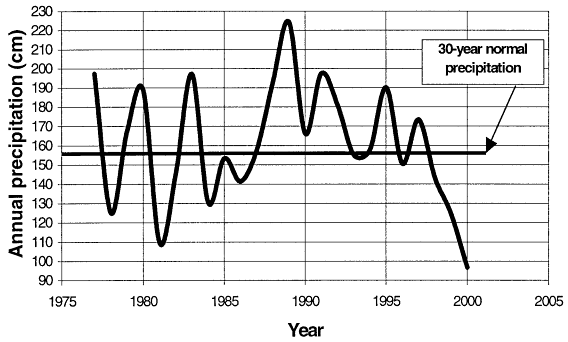

Fig. 2. Trends in annual precipitation amounts for Baton Rouge, Louisiana (Source: NOAA, 2001).

for filamentous algae, which begins its lifecycle in lake sediments.

5.2. Comparison of 2000–2001 monitoring program and pre-restorations results

A summary of results for the 2000–2001 water quality sampling program is presented in Tables 3 and 4. The tables are comprised of: (1) site specific arithmetic means for each parameter; (2) the arithmetic means for both sites according to the location in the water column (top or bottom); (3) the overall arithmetic mean for both sites and both locations in the water column; (4) range of the overall means; (5) the standard error of the overall mean; and (6) the number of samples comprising the overall mean.

In order to gage the trophic conditions of City Park Lake to other lakes, the Organization for OECD's trophic-state classification system was selected (Table 5). Trophic-state classifications provide an indication of nutrient enrichment and the biological productivity of lakes. The annual geometric means for pre-restoration (June 1979–February 1983), post-restoration (June 1983–Sept. 1984), March 1990–June 1991 and June 2000–June 2001 data were superimposed on the classification table. Following restoration, the lake transitioned from hypereutrophic to eutrophic conditions. Based on the phosphorus and Chl-*a* concentrations measured during the 2000–2001 sampling program, hypereutrophic trophic status is indicated. Relatively low total nitrogen (TN, $TKN+NO_3+NO_2$) concentrations in comparison to the other trophic-state variables during 2000–2001 indicate a possible correlation between TP concentrations and increases in algal biomass, as measured by Chl-*a*. The use of Secchi depth data for classification purposes without consideration for the organic staining of the lake water during 2000–2001 may lead to incorrect conclusions concerning the source of light attenuation. The relatively low Secchi depths measured during the 2000–2001 sampling period are influenced more by organic staining than by the presence of phytoplankton.

Gremillion et al. (1985) compiled summer (June through August) TP, TKN, TSS concentration and Secchi depth means for years that corresponded with pre- and post-restoration periods. Presentation of this data and summer means for the 1990 and 2000 sampling programs clearly demonstrate the effects that dredging and the repair of leaky sewer pipelines had on the water quality in City Park Lake (Fig. 3). Dramatic decreases in mean summer TP, TKN, and Chl-*a* concentrations were accompanied by improved water clarity immediately following restoration activities in 1983. Gremillion et al. (1985) documented the occurrence of 'new problems not historically observed in this lake system' in the summer of 1984. The most apparent problem was the occurrence of filamentous *Spirogyra* along approximately 80% of the shallow City Park Lake shoreline. This first post-restoration outbreak of filamentous algae was attributed primarily to a decline in turbidity following dredging activities, elimination of phytoplanktonic shading, and the transport of interstitial nutrients from deep to shallow sediments following dredging (Gremillion et al., 1985). Macrophytes were largely absent throughout the previous history of the University Lakes System and during the first post-restoration year.

In the decades that followed restoration, TP concentrations steadily increased (Fig. 3). Following 1984, TKN concentrations peaked during the summer of 1990 (Fig. 3). Due to the absence of monitoring data between 1991 and 2000, no conclusion could be made as to whether the decline in TKN concentrations between 1991 and 2000 was an indication of a long-term trend or the result of drought conditions prior to and during the 2000 sampling program. The contrast in nutrient trends during post-restoration was likely influenced by sediment phosphorus release and reduced storm water flows, especially after 1998. McKenna (1987) previously determined that sediment phosphorus release contributed significantly to the overall phosphorus budget of City Park Lake, especially during summer months when water temperatures peak. Furthermore, other researchers have recognized that sediment phosphorus release can delay the recovery of lake systems after the reduction of external phosphorus loadings (Brezonik and Pollman, 1999). Phospho-

Table 3
Summary of 2000–2001 in situ measurements for City Park Lake

	Water depth (m)	Secchi depth (m)	Air temp. (°C)	Water temp. (°C)	pH (SU)	DO (mg/l)
Site 1 mean[a]:	1.04	0.39	20.14	23.6	7.42	4.75
Site 2 mean[a]:	1.37	0.45	20.20	24.1	7.43	4.42
Top mean[a]:	–	–	–	24.0	7.45	4.74
Bottom mean[a]:	–	–	–	23.8	7.40	4.42
Overall mean[a]	1.19	0.42	20.17	23.9	7.42	4.58
Range[b]:	0.82–1.65	0.20–1.16	4.7–30.4	8.6–32.0	6.53–8.26	0.64–7.88
S.E.[b]:	0.0337	0.0380	0.9822	0.6970	0.7755	0.1950
n[c]:	46	44	46	90	92	90

Note: All in situ measurements performed between 06:50 and 09:45 h.
[a] Arithmetic mean for 1-year sample period.
[b] Range and standard error based on overall mean.
[c] n = number of samples comprising overall mean.

rus that is released from lake sediments is retained in the lake in either organic or inorganic forms until it is flushed or settles from the lake water. For this reason, shallow lake systems such as City Park Lake are generally modeled as a two-compartment system with respect to phosphorus; one compartment consisting of lake water and one compartment lake sediment. In contrast, nitrogen that is released from the sediments may enter the atmosphere. Measurable nitrogen in the lake primarily exists in organic form due to biological assimilation or in inorganic forms due to storm water loadings. In years of relatively low precipitation and high organic staining of lake water, nitrogen levels can be expected to decrease while phosphorus levels can be expected to increase.

Since restoration, Chl-*a* concentrations and Secchi depths gradually increased, but remained below pre-restoration summer concentrations (Fig. 3). Since peaking in 1984 when the first post-restoration outbreak of filamentous algae occurred, Secchi depths were observed to return

Table 4
Summary of 2000–2001 analytical results for City Park Lake

	Chl-*a* (μg/l)	TP (mg/l)	PO_4-P (mg/l)	TKN (mg/l)	TAN (mg/l)	NO_2+NO_3–N (mg/l)	TSS (mg/l)	VSS (mg/l)
Site 1 mean[a]:	42.79	0.368	0.113	0.690	0.114	0.102	30.23	15.83
Site 2 mean[a]:	39.67	0.375	0.129	0.682	0.108	0.114	32.23	17.41
Top mean[a]:	41.51	0.350	0.116	0.695	0.118	0.115	28.80	15.75
Bot. Mean[a]:	40.95	0.393	0.126	0.678	0.103	0.100	30.85	15.46
Overall mean[a]:	41.23	0.371	0.121	0.686	0.111	0.108	29.80	15.61
Range[b]:	1.54–95.44	0.081–0.712	0.011–0.474	0.162–1.809	0.001–0.413	$<$0.010–0.341	8.67–58.75	2.67–36.89
S.E.[b]:	3.660	0.018	0.008	0.036	0.012	0.015	1.66	1.18
n[c]:	60	88	87	75	71	72	82	78

[a] Arithmetic mean for 1-year sample period.
[b] Range and standard error based on overall mean.
[c] n = number of samples comprising overall mean.

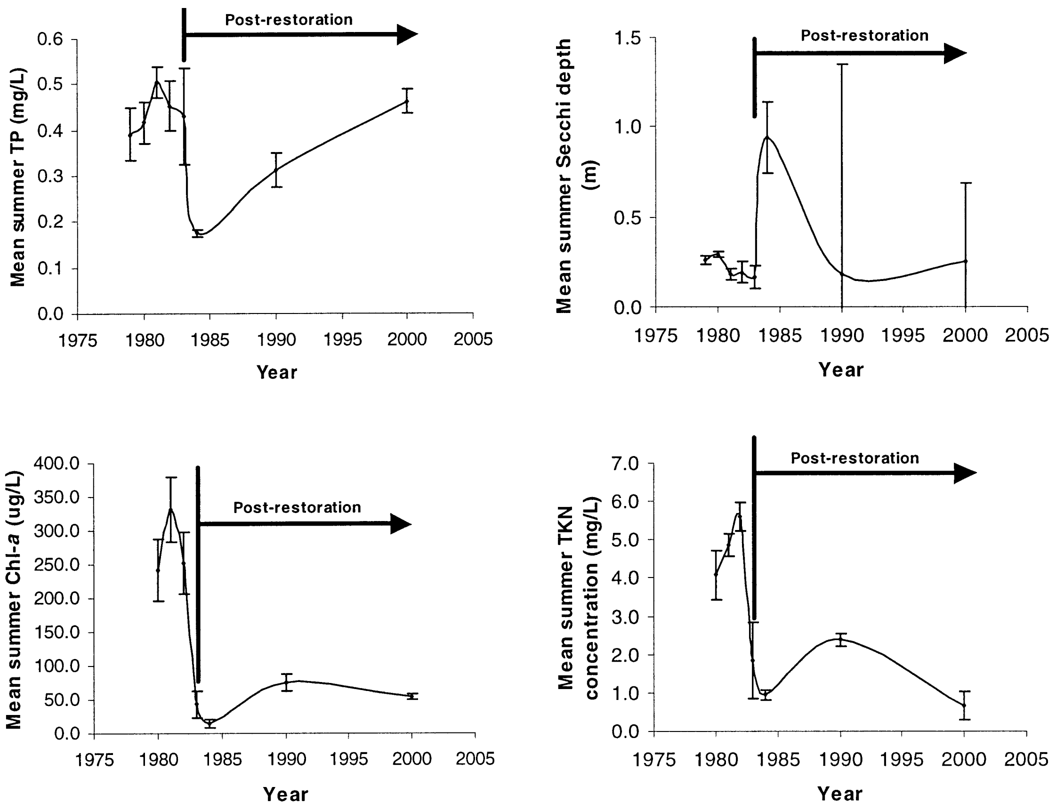

Fig. 3. Timetrace of mean summer TP, TKN and Chl-*a* concentrations and Secchi depths in City Park Lake.

to pre-restoration levels. This transition is due in part to the presence of organic substances during the 2000–2001 sampling program.

Higher Chl-*a* concentrations generally coincided with higher TP concentrations during summer months for pre-restoration years (Fig. 4). Chl-*a* as a percentage of TP concentration for each of the pre-restoration years for which data were collected (1980, 1981 and 1982) averaged 59.8%. Data shown for 1983 reflect release of phosphorus from dredged sediments. Low Chl-*a* concentrations for the summer of 1983 when relatively high TP concentrations were observed is likely due to substantial sediment suspension in the months that following dredging activities (Gremillion et al., 1985). Therefore, data for 1983 are not representative of post-restoration conditions. On the other hand, data for 1984 through 2000 show a post-restoration trend toward TP concentrations that approximate pre-restoration summer concentrations. Summer Chl-*a* as a percentage of summer TP for 1984, 1990 and 2000 averaged 14.6%. This trend demonstrates the limitation of algal growth due to parameters other than phosphorus. The observation of organic staining in City Park Lake indicates that algal growth was likely inhibited due to the preferential absorption of light by the 'true color' of the water column, as influenced by the presence of tannins.

The impact of light limitation on algal growth was evaluated by comparing summer Secchi depths to summer Chl-*a* concentrations (Fig. 4). Pre-restoration Secchi depths approximated those measured during post-restoration years (1984, 1990 and 2000). The data presented for 1984 provide indication of a critical shift in the predominant types of phytoplankton that occurred during the post-restoration period. Although Chl-*a* provides a good measurement of suspended biomass concentrations, lower Chl-*a* concentra-

Table 5
OECD ranges of trophic state variables based on scientists' opinions (after Vollenweider and Kerekes, 1980)

Trophic-state variable	Trophic State			City Park Lake		
	Oligotrophic	Mesotrophic	Eutrophic	1979–1983	1983–1984	2000–2001
TP (μg/l)						
Geometric mean	8	27	84	332	196	330
Range (*n*)	3–18 (21)	11–96 (19)	16–390 (71)			
TN (μg/l)						
Geometric mean	660	750	1900	3074	997	682
Range (*n*)	310–1600 (11)	360–1400 (8)	390–6100 (37)			
Chl-a (μg/l)						
Geometric mean	1.7	4.7	14	191.1	2.7	35.1
Range (*n*)	0.3–4.5 (22)	3–11 (16)	2.7–78 (70)			
Peak Chl-a (μg/l)						
Geometric mean	4.2	16	43	793.8[a]	63.3[a]	95.4[a]
Range (*n*)	1.3–11 (6)	5–50 (12)	10–280 (46)			
Secchi depth (m)						
Geometric mean	9.9	4.2	2.4	0.2	0.3	0.4
Range (*n*)	5.4–28 (13)	1.5–8.1 (20)	0.8–7.0 (70)			

Note: Table was adapted from USEPA Nutrient Guidance for Lakes and Reservoirs, 2000 (USEPA, 2000).
[a] Peak concentration for sampling period.

tions reported for post-restoration years do not accurately reflect the occurrence of filamentous algae or vascular plant growth. By comparison to pre-restoration years; sufficient phosphorus was available for the production of algae in line with what occurred during pre-restoration years, however, the presence of organic acids during a drought period is likely responsible for the relatively low Chl-*a* concentrations and low Secchi depths observed during the summer of 2000.

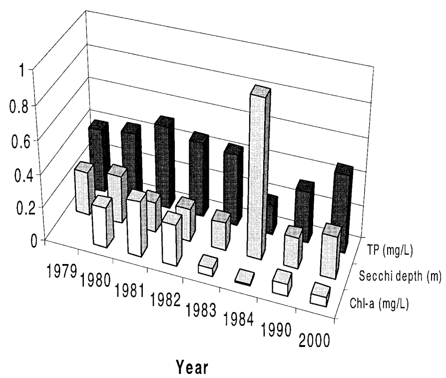

Fig. 4. Comparison of mean summer TP and Chl-*a* concentrations and Secchi depths in City Park Lake.

5.3. Nutrient limitation trends and implications

Limnologists and engineers refer to the nitrogen to phosphorus (N:P) ratio of a lake to describe lake status. Lakes for which N:P ratios exceed 7.2 are generally termed 'phosphorus-limited' (Chapra, 1997). This number is derived from Redfield's stoichiometry and is the approximate ratio of nitrogen mass to phosphorus mass in suspended algal biomass. The limitation of a particular nutrient may result in the overall limitation of algal growth if light and temperature levels are also unfavorable for algal growth; however, nutrient limitation may also promote certain types of algae over others. For example, algae that fix atmospheric nitrogen, such as cyanobacteria, are more commonly observed in fresh surface waters with low N:P ratios (<4) (Thomann and Mueller, 1987).

The ratio of TN to TP was computed for each set of available data from pre- and post-restoration sampling programs. A review of the annual TN:TP ratios from pre-restoration to present demonstrates a trend from phosphorus-limited conditions to nitrogen-limited conditions (Fig. 5). This trend indicates that phosphorus accumulation, likely due to sediment phosphorus release, has exceeded that of nitrogen over the past 20 years. Although 'nitrogen limitation' implies a deficiency in nitrogen with respect to phosphorus, elevated phosphorus levels rather than deficiencies in nitrogen are responsible for nitrogen limitation in City Park Lake.

Analysis of the 2000–2001 data resulted in an annual mean TN:TP ratio of 2.14 (S.E. = 0.21, $n = 75$) and a mean summer TN:TP ratio (June, July and August) of 1.48 (S.E. = 0.14, $n = 23$). These data indicate that lake conditions are strongly nitrogen-limited. A comparison of the annual versus summer TN:TP ratio supports the magnitude and impact of sediment phosphorus release under higher summer temperatures and lower DO conditions on nutrient concentrations in the lake water.

Because TN and TP account for forms of each nutrient that are not immediately available for uptake (i.e. organic nitrogen and phosphorus), some researchers have found that nutrient limitation assessments falsely predicted that phosphorus was the limiting nutrient in eutrophic and hypereutrophic Florida lakes (Schelske et al., 1999). TN:TP ratios for pre-restoration and 2000–2001 sampling periods were compared with total inorganic nitrogen (TIN):soluble reactive phosphorus (SRP) ratios for the same periods. Whereas phosphorus limited conditions (11.14) were indicated for the pre-restoration year (1979–1980) using a TN:TP ratio, nitrogen limited conditions were indicated using a TIN:SRP (5.21) ratio. The nutrient limitation for the 2000–2001 sampling period remained virtually unchanged (TN:TP = 2.60 and TIN:SRP = 2.05). The data presented indicate that phosphorus loadings to the lake from internal sources such as sediments have outpaced net nitrogen loadings. Although nitrogen loadings from sediments were not measured as part of this study, the impact of nitrogen mass in the lake water is mitigated by nitrification/denitrification processes, whereas phosphorus that is released from sediments is primarily exported from the lake through hydraulic flushing.

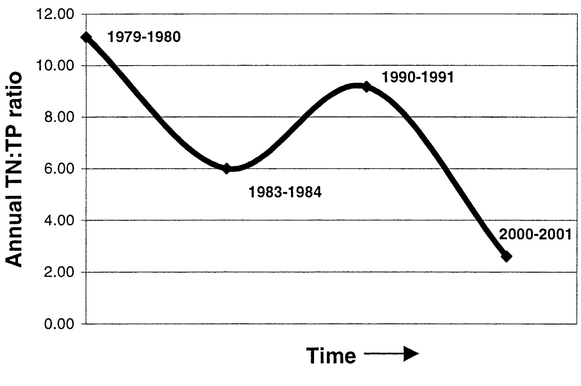

Fig. 5. Trends in annual TN:TP ratios in City Park Lake.

As lakes become more eutrophic, the diversification of phytoplankton decreases, ultimately resulting in the dominance of cyanobacteria (formerly referred to as blue–green algae) (Dokulil and Teubner, 2000). Shallow lake depths have been observed to promote the predominance of filamentous cyanobacteria, while deep lakes are commonly predominated by colony-forming types (Schreurs, 1992). The transition from marginally nitrogen-limited conditions to extremely nitrogen-limited conditions can result in the predominance of algal species with the ability to 'fix' atmospheric nitrogen. Although lakes with low TN:TP ratios generally favor nitrogen fixing algal species, Teubner et al. (1997) suggest that the timing of critical ratios influences dominance of one cyanobacteria species versus another. In addition, filamentous algae, like cyanobacteria, originate in sediments and therefore have access to large reservoirs of phosphorus, thus gaining the competitive edge during early growth stages, over suspended, non-filamentous algal species. Although no formal phytoplankton surveys have been conducted in recent years, the transition to extremely nitrogen-limited conditions presumably played a role in the occurrence of recent filamentous algal blooms. The interrelation between TN, TP and soluble reactive silica also has been documented to promote specific species of phytoplankton over others

(Dokulil and Teubner, 2000). For example, based on research conducted on a deep alpine lake, Dokulil and Teubner (2000) found that higher TN:TP ratios and low silica to nitrogen (Si:N) and silica to phosphorus (Si:P) ratios resulted in the predominance of cyanobacteria, whereas phytoplankton surveys conducted on the same lake during the following year revealed that high soluble reactive silica (SRSi) concentrations and low TN:TP ratios resulted in the predominance of diatomaceous algae. Thus, although the elimination of phytoplankton in hypereutrophic lakes is improbable, management of nutrient loadings can result in nutrient limitations that favor one phytoplankton over another.

The use of nutrient ratios as a prediction tool for estimating the potential for phytoplankton blooms or the potential for predominance by a specific species of phytoplankton should be used with caution. For example, nutrient ratios provide no indication of the effects of temperature or light climate on the pre-dominance of particular phytoplankton species. Also, lake morphology characteristics such as depth and fetch can influence whether or not algal growth occurs and the survival of specific species versus others. Additionally, nutrient limitations do not account for the role of luxury uptake by algae. Luxury uptake occurs when algae consume SRP when its quantity in the environment exceeds the metabolic capacity to assimilate (Chapra, 1997). Under these conditions, algae can store SRP as polyphosphate and utilize the stored phosphorus when phosphorus levels decline in the water environment. Thus, when luxury uptake occurs, nutrient ratios cannot accurately account for biologically integrated forms of phosphorus that may be used for luxury uptake.

5.4. Post-restoration trends in phytoplankton biomass correlations

Empirical phosphorus-to-chlorophyll-*a* correlations are commonly found in literature (Dillon and Rigler, 1974; Rast and Lee, 1978; Bartsch and Gakstatter, 1978; Smith and Shapiro, 1981). In general, many of the reported phosphorus-to-chlorophyll-*a* correlations were based on empirical data from northern and northeastern United States lakes that varied widely in depths, mixing conditions and nutrient concentrations. Although the correlations provide a simple tool for estimating primary productivity of lake systems, they only provide an indication of productivity as a function of the relationship under study and should only be applied to lakes represented by the sample population. For example, phosphorus-to-Chl-*a* relationships can provide useful information about the degree of primary productivity based on phosphorus concentrations. The degree of scatter and location of the plotted data in relation to linear correlations can also provide an indication of growth-limiting light conditions. Due to the nature of sample collection and analyses procedures, Chl-*a* concentrations are closely related to the amount of phytoplankton biomass that occupies volume within the water column. Chl-*a* concentrations, therefore, can provide a good measurement of suspended algae.

Phosphorus and Chl-*a* concentrations and Secchi depths were measured during several critical stages in the history of City Park Lake. These stages include pre-restoration years (1980, 1981, and 1982) and post-restoration years (1983–1984, 1990–1991, and 2000–2001). The determination of TP-to-Chl-*a* and Secchi depth-to-Chl-*a* correlations for various historical periods can provide a critical link between historical phosphorus trends, light conditions, and algal growth. Consistent with methods used by other researchers (Dillon and Rigler, 1974; Rast and Lee, 1978; Bartsch and Gakstatter, 1978; Smith and Shapiro, 1981), the correlations between TP and Chl-*a* concentrations, and between Secchi depths and Chl-*a* concentrations were evaluated. Statistical analyses of the log transformations for each variable are presented in Table 6. The results of these analyses indicate relatively strong correlations between TP and Chl-*a* in pre-restoration years (1980–1982). During this period, weaker correlations between Secchi depths and Chl-*a* were indicated by the reported data. During the post-restoration period, overall trends in data correlations were not obvious. Very strong correlations resulted between Secchi depths and Chl-*a* concentrations during the 1983–1984 and 2000–2001 sampling programs.

Table 6
Comparison of statistical analyses for City Park Lake data correlations for various pre- and post-restoration years

Parameter	TP vs. Chl-*a*	Chl-*a* vs. Secchi depth
1980 (Jan.–Dec.) n = 15		
r:	0.7138	0.5816
$y = ax + b$[a]	$y = 1.24x - 0.82$	$y = -1.88x + 1.18$
P value	0.0028	0.0229
1981 (Jan.–Dec.) n = 14		
r:	0.6724	0.0416
$y = ax + b$[a]	$y = 0.85x + 0.20$	$y = -0.03x + 2.37$
P value:	0.0084	0.8877
1982 (Jan.–Dec.) n = 15 (For Secchi depth vs. Chl-a, n = 13)		
r:	0.7172	0.5466
$y = ax + b$[a]	$y = 0.80x + 0.35$	$y = -0.68x + 1.87$
P value:	0.0026	0.0532
1983–1984 (Sept.–Sept.) n = 16		
r:	0.445	0.8262
$y = ax + b$[a]	$y = 1.92x - 3.05$	$y = -1.50x + 0.64$
P value:	0.1108	0.00027
1990–1991(Mar.–Feb.) n = 15		
r:	0.3607	0.0128
$y = ax + b$[a]	$y = 1.06x - 1.22$	$y = 0.02x + 1.38$
P value:	0.1865	0.9638
2000–2001(June–May) n = 28		
r:	0.6628	0.8186
$y = ax + b$[a]	$y = 1.10x - 1.24$	$y = -1.44x + 0.84$
P value:	0.0001	0.0000

Note: Correlation between variables within 95% confidence interval indicated for P values < 0.05.

[a] y = Chl-*a* concentration; x = TP concentration or Secchi depth.

However, no significant correlations between any variables were indicated for the 1990–1991 data. Furthermore, the apparently strong correlation between water clarity and algal biomass for the 2000–2001 data was complicated by the presence of organic color in the lake water.

The dredging of City Park Lake in 1983 resulted in a large decrease in the amount of phosphorus (Fig. 3) that was available for algal growth, specifically the growth of suspended unicellular algae. Although filamentous algae such as cyanophytes and chlorophytes were documented during pre-restoration phytoplankton surveys, large outbreaks of filamentous algae have predominated post-restoration algae observations (R. Malone, personal communication, 2001). Filamentous algae (*Spirogyra*) were documented for the 1984 growing season by Gremillion et al. (1985). Data shown for the 1990–1991 and 2000–2001 sampling periods represent years for which no substantial growth of filamentous algae was documented. Correlations for samples collected during the 1983–1984 mark a shift in the relationship between phosphorus and the concentration of algal biomass suspended the water column and a marked improvement in water clarity following dredging activities. This shift may be explained by the predominance of floating filamentous algae under improved water clarity conditions in the summer of 1984 as opposed to the suspended unicellular algae that was documented in pre-restoration years. The apparent shift in the predominant phytoplanktonic species was further encouraged by the trend toward nitrogen-limited conditions during the post-restoration period.

6. Conclusions

Several important trends in the water quality of City Park Lake can be inferred from the water quality data presented in this study.

- Restoration efforts resulted in a dramatic reduction in the frequency of algal blooms and fish kills; however, outbreaks of filamentous algae occurred less than one decade following restoration;
- Current phosphorus levels approach, and in some cases surpass, phosphorus levels measured during the pre-restoration period;
- Sediment phosphorus release and recent deficiencies in precipitation have influenced the increasing trend in TP concentrations and the decreasing trend in TN concentrations during post-restoration years;
- A shift in nutrient limitation conditions has likely played a role in recent observations of predominance by filamentous species;
- Organic color in lake waters has resulted in a deficiency of light that can be used for photosynthesis by plants and phytoplankton located in the hypolimnion;

- Trends in phosphorus-to-Chl-*a* correlations between pre- and post-restoration periods underscore the influence of internal light conditions on the growth of phytoplankton and indicate a possible shift from unicellular, suspended algae species to filamentous forms.

The long-term water quality trends indicate that the restoration of City Park Lake was successful in drastically improving water quality conditions in City Park Lake. Without the implementation of long-term watershed and in-lake nutrient management strategies, however, the beneficial effects and expense of lake restoration is severely undermined. Nitrogen-limited conditions and the predominant filamentous types of algae can be expected to continue as long as phosphorus concentrations remain in excess. The observation of filamentous algae in small quantities along the shoreline of City Park Lake during the 2000 and 2001 growing seasons indicates that nutrient levels were indeed sufficient for algal growth. Even with the implementation of phosphorus management strategies in the lake watershed, internal phosphorus loadings from lake sediments will continue to play an important role in the recovery of this shallow lake system.

The occurrence and implications of organic color in lakes such as City Park Lake are difficult to predict. The concept of photobleaching of sunlight due to tannin levels in the lake water and its effects on lake quality is currently the subject of research in other lakes in the Mid-west and in Florida (Reche et al., 1999; Crisman et al., 1998), and is not well understood. As observed in City Park Lake, lake hydrology cannot be neglected as an important factor controlling the balance between photosynthesis and respiration.

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