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WATER QUALITY ASSESSMENT OF PRAIRIE CREEK RESERVOIR IN DELAWARE COUNTY, INDIANA

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ABSTRACT. Prairie Creek Reservoir, located in an agricultural watershed in east-central Indiana, serves as a secondary drinking water supply through water releases to the White River and offers various recreational activities including fisheries. Monitoring of the reservoir and knowledge of its water quality has been very limited. Therefore, this study monitored water quality at seven locations to assess the current status, observe spatial and temporal differences, and to initiate long-term monitoring that would support future management decisions. The results of 2005–2006 monitoring demonstrate that the reservoir is an eutrophic, dimictic water body characterized by weak summer stratification (max. $\Delta T < 1.3°C$), low water transparency (<100 cm), anoxic conditions (DO <1 mg/L) in the hypolimnion from June to September, and high concentration of orthophosphates. Concentrations of chlorophyll *a* were lower than expected, possibly due to growth of macrophytes. Furthermore, the internal load of phosphorus in the reservoir might play an important role in nutrient cycling and eutrophication of this water body.

Keywords: Eutrophication, nutrients, monitoring, reservoir

In 2000, 45% of the lakes and reservoirs in the United States were classified as impaired for one or more uses; nutrients and agriculture were identified as the leading cause and source of those impairments (U.S. EPA 2000). Fertilizers designed to increase the biological productivity of agricultural soils may also increase biological productivity of waters draining these soils and contribute to lake and reservoir eutrophication (Jørgensen et. al. 2005). A persistent load of nutrients to a water body can cause algal blooms, overabundance of macrophytes, and eventually lead to depletion of dissolved oxygen from water, causing severe stress or even death to aquatic organisms (Carpenter et al. 1998; U.S. EPA 2000; Marshall et al. 2006; Mau 2002; Weld et al. 2002). Eutrophication can also negatively affect recreation, industrial use, water supplies, and fisheries due to development of undesirable taste and odor of drinking water supplies, increased cost of drinking water treatment, summer fish kills, and increased rates of sediment accumulation (Carpenter et al. 1998; U.S. EPA 2000; Jørgensen et al. 2005; Sharpley et al. 1995; Smith 2003; Weld et al. 2002).

Monitoring activities have been widely utilized to assess the existing water quality and the effectiveness of land management practices on receiving water bodies (U.S. EPA 2006a; Mau 2002). In the State of Indiana, the Department of Environmental Management (IDEM) performs watershed monitoring, including monitoring of publicly-owned lakes and reservoirs, with a goal to evaluate suitability of water resources to support their beneficial uses such as aquatic life, water supply, recreation, and fishing (IDEM 2004, 2006). With about 64% of land in Indiana used for agriculture, nutrients were identified as the major cause of pollution in the state's reservoirs (IDEM 2000). Based on monitoring between 2000 and 2004, 19% of the sampled lakes and reservoirs in Indiana were classified as oligotrophic, 46% mesotrophic, 25% eutrophic, and 9% as hypereutrophic (IDEM 2006).

In Delaware County, Indiana, privately-owned Prairie Creek Reservoir serves as a secondary water supply for the City of Muncie through water releases into the White River during the dry seasons. However, the reservoir also provides a venue for recreational activities, such as fishing, camping, and boating. It is owned by the Indiana-American Water Company but leased and maintained by the City of Muncie, Department of Parks and Recreation (Cescon 1997; Delaware Muncie-Metropolitan Plan Commission 2007). Currently, the future of development and land management in the reservoir watershed is uncertain as the lease to the City of Muncie expires in 2021. As a result, Delaware Muncie-Metropolitan Plan Commission developed the Prairie Creek Reservoir Master Plan to address the future of land development in the watershed (Delaware Muncie-Metropolitan Plan Commission 2007).

The water quality of the reservoir and its watershed is of great interest and concern to the public. Seventy-three percent of public survey respondents ($n = 300$) valued this reservoir as a unique feature in the county and 93.9% of respondents viewed the reservoir as a positive asset to the community (Delaware Muncie-Metropolitan Plan Commission 2007). To maintain the services and the value of this reservoir, it is necessary to maintain its good water quality. However, a limited number of studies have addressed this issue (Weaver 1964; Haman 1964; Cescon 1997).

In 1964 Weaver performed the first survey of the Prairie Creek Reservoir for the purpose of fisheries management (unpublished report by the Indiana Department of Conservation). At the time of the survey, 73% of land was used by agriculture, which was identified as the most significant threat to the population of sport fish. Today, agriculture is still a predominant land use in this watershed (Table 1). Haman (1964) performed the first ecological evaluation of the reservoir and reported that spring and fall turnover, and summer and winter stratification were occurring as would be expected for a temperate reservoir. Additionally, Haman (1964) found the highest mass of plankton to occur in spring. The zooplankton community was found to be typical of a medium-size, shallow, eutrophic North American lake (Cescon 1997).

In 2001, the White River Watershed Project (WRWP) began an investigation of the effects of non-point source pollution on three White River sub-watersheds, including the Prairie Creek watershed (DCSWCD 2004; Goward 2004). Increased concentrations of nitrates, orthophosphates, Atrazine®, and *Escherichia coli* were found at several locations in the reservoir watershed and were attributed to agricultural runoff and failing septic systems. However, only very limited monitoring was performed at the reservoir (DCSWCD 2004; Goward 2004). The final WRWP report recommended continuous monitoring of the Prairie Creek reservoir and development of land management practices to reduce non-point source pollution in the watershed.

Consequently, the goal of the present study was to gain knowledge about the reservoir water quality that would support future management decisions. Specific objectives were: (1) to determine the current water quality of the reservoir and acquire baseline data, (2) to observe spatial and temporal changes in water quality, and (3) to initiate long-term monitoring.

METHODS

Study location.—Prairie Creek Reservoir is a stream-fed reservoir located in Delaware County, Indiana. It is situated in the glaciated area of the Eastern Corn Belt Plains (Ecoregion 55) with extensive corn, soybean, and livestock production (U.S. EPA 2002). The reservoir was formed by construction of a dam across Prairie Creek in 1958 to provide fisheries (Cescon 1997). The east side of the reservoir is managed for recreation, such as boating, picnicking, and camping, while the west side is managed for green space, horse trails, and an "all-terrain vehicles" area in the southwest (Fig. 1).

The reservoir surface area is 507 ha (1252 acres) and the estimated water retention time is 5.27 years (assuming only the reservoir surface area and its outflow while neglecting any evaporation and groundwater seepage) (Table 1). The reservoir is located at the lowest topographical point of its watershed, where 72% of land is used for agricultural purposes. Several tributaries drain the watershed collecting water from the surrounding land via runoff and tile drains. The reservoir output drains into the White River on its northwest side (Fig. 1).

Sampling and analysis.—Due to lack of any historical water quality information, seven sites, located in open water, were selected for monitoring (Fig. 1). Holdren and Souza (2004) recommended that monitoring be conducted at sites that characterize the effects of inflow and outflow, multiple deep basins, or additional bays. In 2005, the surface water at the selected sites was monitored weekly for pH, dissolved oxygen, temperature, conductivity, transparency, nitrates, orthophosphates, and chlorophyll *a*. A portable digital meter sensION156 (Hach, Inc., Loveland, Colorado) was used to measure pH, conductivity, dissolved oxygen concentration, and temperature *in-situ*. Transparency was measured *in-situ* with a Secchi disk according to the standard method (U.S. EPA 2007).

Surface samples for nutrient analysis were collected in plastic bottles by a grab procedure. Samples for chlorophyll *a* were collected in one liter amber plastic bottles that were prepared

Table 1.—Characteristics of the Prairie Creek reservoir and its watershed.

Watershed	Prairie Creek/Cunningham (HU 5120201010110)
Watershed surface area	44 km^2
Land use	Agriculture 72.2% Green space 18.2% Residential 6.4% Transportation 2.3% Other 1.8%
Maximum depth	11 m
Average depth	5.5 m
Maximum length	5.95 km
Maximum width	1.4 km
Major axis	north – south
Surface area of water	5.1 km^2
Length of shoreline	25.7 km
Reservoir elevation at normal pool	301.7 m
Capacity	27.2 billion m^3

according to standard methods (APHA 1998). All samples were then stored on ice and transported in a cooler to the laboratory for analysis. Analysis for nitrates was performed by use of the cadmium reduction method 4500-NO3-E (APHA 1998), total reactive orthophosphates were analyzed following the U.S. EPA-approved ascorbic acid method 4500-P-E (APHA 1998; U.S. EPA 1997), and ammonia level was determined using salicylate method. The final concentrations were determined with a HACH DR/2400 spectrophotometer (Hach, Inc., Loveland, Colorado).

Samples for chlorophyll *a* analysis were filtered through a Whatman GF/C glass-fiber filter immediately after their arrival at the laboratory, and subsequent analysis was performed by acetone extraction according to the standard method (APHA 1998). The final chlorophyll *a* concentration was determined using visible spectrophotometry and calculated according to the trichromatic method (APHA

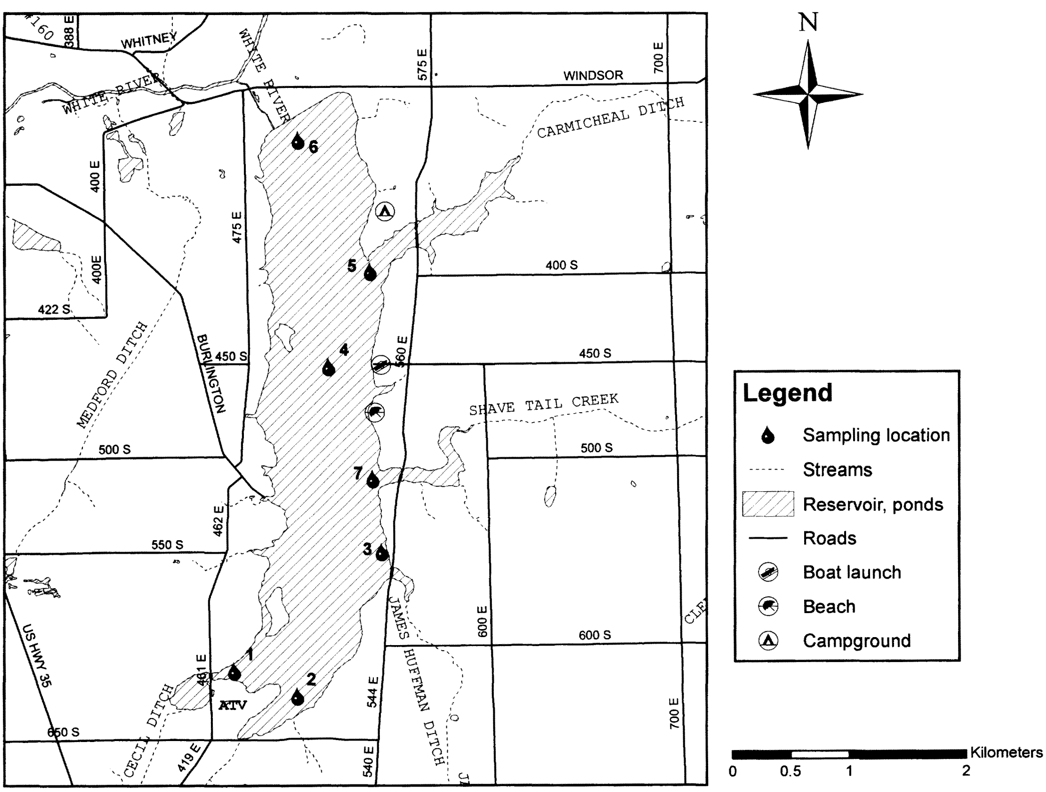

Figure 1.—Prairie Creek Reservoir and the monitored locations.

1998). One field duplicate, one laboratory blank, and one laboratory fortified sample were analyzed for each sampling event to verify the accuracy and precision of analytical procedures.

In 2006, monitoring of a vertical profile was conducted and the measurements of dissolved oxygen, pH, temperature, conductivity, chlorophyll *a*, and turbidity were performed *in-situ* with a Hydrolab Sonde DS5 (Hydrolab Inc., Austin, Texas) at all seven locations. The measurements were taken at 0.5 m intervals from the water surface to the bottom. The equipment was calibrated prior to each trip according to the manufacturer's recommendations, and its accuracy was checked in the field by measuring the values of standard solutions. Additionally, grab samples from the hypolimnion were collected using a horizontal Beta sampler, transferred to plastic bottles, and stored on ice together with the surface water samples for transportation to the laboratory. Water samples from the hypolimnion were analyzed for nitrates, orthophosphates, and ammonia following the procedures described above.

All statistical analyses were performed by SPSS software (SPSS Inc., Chicago, Illinois). The effects of location, month, and year (fixed factors) for each water quality parameter (dependent variable) were determined using a general linear model and univariate analysis. Data were log-transformed to approximate normal distribution. The level of statistical significance was set at 0.05.

RESULTS AND DISCUSSION

Thermal regime.—Surface water temperatures did not differ significantly among monitored locations. However, the effect of the month was statistically significant. The annual (May – November) mean temperature of surface water was 23.4 °C in both 2005 and 2006 (Table 2). The maximum temperatures of surface water at all locations were recorded on 9 August 2005 and on 17 July 2006. The measurements of vertical temperature profiles revealed summer stratification and periods of turnover in the spring and fall indicating that the reservoir is a warm dimictic water body (Fig. 2). Thermal stratification, defined as the minimum change of 1 °C in water temperature per 1 m of depth (Wetzel 1983), developed in mid-June at all seven locations (Fig. 2).

Maximum temperature difference (ΔT) between the epilimnion and the hypolimnion was attained on 15 June 2006, at all but the deepest location (#6), where maximum ΔT of 1.3 °C/m was achieved on 17 July 2006 (Fig. 2). The vertical temperature profile at this location exhibited weak stratification ($\Delta T < 3$ °C) that began to establish in early June, although the three distinctive, thermally stratified layers never developed (Fig. 2). The thermocline boundary changed throughout the season, usually forming in the lower 50% of the water column. Such weak stratification of this reservoir can most likely be attributed to the effects of wind and its shallow depth. Shallow lakes and reservoirs do not stratify or stratify only for a few days at a time during particularly hot and calm weather conditions (Jørgensen et. al. 2005). Additionally, shallow reservoirs undergo strong diurnal cycles in thermal stratification and mixing (Condie & Webster 2002), but diurnal changes were not investigated in this study. While the wind speed was not monitored in this study, field observations suggested strong north-east winds that might play an important role in lateral and vertical mixing (Herb & Stefan 2004).

Thermal stratification in a lake or reservoir can cause significant variations in dissolved oxygen concentrations, and therefore, affect nutrient release rates (Nowlin et al. 2005). The presence of weak stratification and the resultant vertical mixing during summer months might have significant consequences on the distribution of nutrients within the reservoir. There is a potential for release of nutrients from benthic sediments into the water column that encourage algal growth. This internal loading of nutrients in the reservoir may become an even more significant contributor than the external inputs (Gislason et al. 2004).

Dissolved oxygen.—Mean annual concentration of dissolved oxygen (DO) in the surface water was 8.3 mg/L in 2005 and 9.6 mg/L in 2006 (Table 2), with minimum values of less than 5 mg/L measured at shallow locations 1 and 2 on 27 July 2005 and 20 July 2005, respectively. In 2006, the lowest DO concentrations in epilimnion were found at locations 1, 6, and 7 in August, and at location 6 on 7 September. While there were no significant differences in epilimnetic concentration of DO among monitored locations, July, August and September were significantly different from the other monitored months ($P < 0.001$).

Table 2.—Surface water quality of the Prairie Creek Reservoir in 2005 and 2006 (April through November monitoring period). An abbreviation "BDL" signifies the level below detection limit.

		n	Mean	Standard deviation	Minimum	Maximum
Temperature (°C)	2005	156	23.38	5.04	11.10	30.50
	2006	91	23.42	4.35	11.06	28.87
	Total	247	23.39	4.79	11.06	30.50
Dissolved oxygen (mg/L)	2005	155	8.26	2.27	4.05	13.28
	2006	91	9.60	2.54	3.14	15.15
	Total	246	8.76	2.45	3.14	15.15
Dissolved oxygen (%)	2005	155	98.9	25.2	51.6	178.8
	2006	91	116.4	33.0	40.3	193.3
	Total	246	105.4	29.5	40.3	193.3
pH	2005	142	8.34	0.71	4.50	11.50
	2006	91	8.42	0.28	7.73	8.91
	Total	233	8.38	0.58	4.50	11.50
Specific conductivity (µS/cm)	2005	156	350	51	302	563
	2006	91	342	25	320	537
	Total	247	347	43	302	563
Secchi disk (cm)	2005	146	86	18	50	130
	2006	93	70	15	40	110
	Total	239	80	19	40	130
Nitrates_N (mg/L)	2005	156	0.45	0.39	BDL	2.29
	2006	93	0.28	0.23	BDL	1.20
	Total	249	0.38	0.35	BDL	2.29
Orthophosphates_P (mg/L)	2005	156	0.054	0.063	BDL	0.401
	2006	93	0.058	0.077	BDL	0.482
	Total	249	0.056	0.068	BDL	0.482
Chlorophyll *a* (ug/L)	2005	52	11.5	7.3	2.0	26.2
	2006	89	6.1	2.6	2.1	15.3
	Total	141	8.1	5.5	2.0	26.2

DO concentrations in the hypolimnion ranged from 0.24 mg/L (August 2006) to 8.9 mg/L (May 2006) (Table 4). On 24 August 2006, the concentration of DO at the lake bottom reached the annual minimum for the reservoir. These anoxic conditions began to form in early June when about 30% and 27% of bottom waters at location 4 and 6, respectively, had DO concentrations <1 mg/L, which lasted through September (Table 4). The maximum oxygen depletion at the location 4 was recorded on 30 June 2006, when 42% of water near the bottom had DO concentration <1 mg/L, while the location 6 reached the maximum depletion of 43% of depth on 17 July 2006. Conversely, the most shallow locations of the reservoir (1 and 2) reached such low DO concentrations only a few times during 2006. Location 1 had hypolimnetic DO <1 mg/L only in June and August, and location 2 reached such low levels only once in July (Table 4). These shallow waters are more susceptible to wind-induced water mixing, and therefore, increased diffusion of atmospheric oxygen into water (Jørgensen et. al. 2005). Additionally, waters in these shallow regions have shorter residence times and are thus less susceptible to oxygen depletion due to the effects of tributaries (Fig. 1). More stagnant waters in deeper portions of the reservoir are more likely to experience prolonged periods of oxygen depletion due to higher retention time and lower influence of tributaries. Herb & Stefan (2004) concluded that lateral transfer processes induced by wind are important in shallow lakes with non-uniform macrophyte coverage and in reservoirs with significant inflows and outflows.

Dissolved oxygen is an essential indicator of ecosystem health in lakes and reservoirs, and its depletion is often a sign of eutrophication

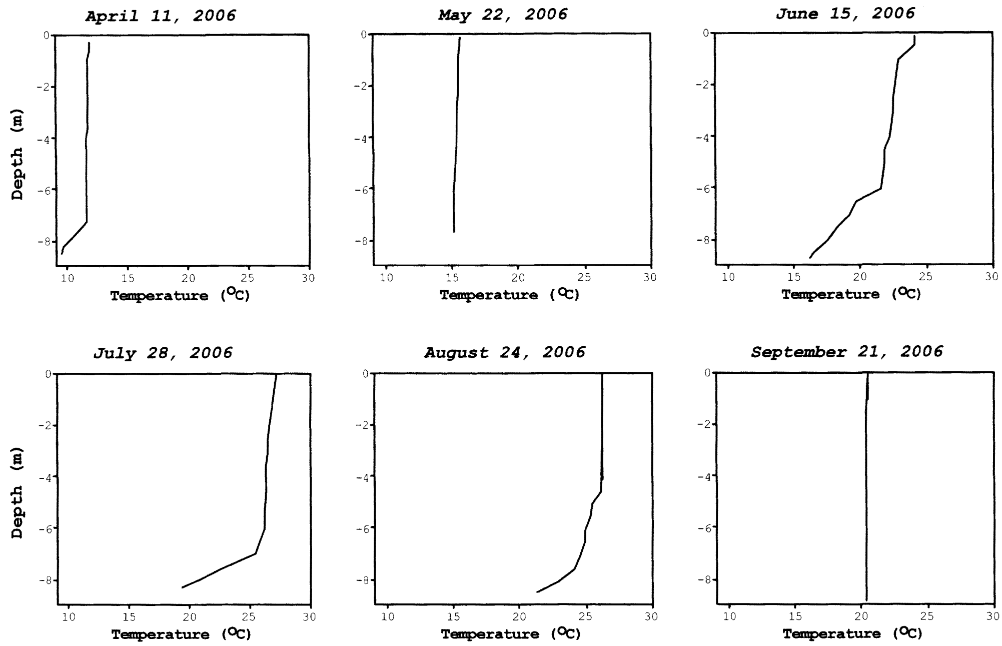

Figure 2.—Vertical temperature profiles at the deepest location (#6) of the Prairie Creek reservoir measured in 2006.

associated with excessive load of nutrients, increased primary production, and subsequent decomposition of organic matter (Carlson & Simpson 1996; Wetzel 1983). Anoxic conditions, such as those that persisted from June through September at this reservoir, can cause many significant changes in a reservoir chemistry and biology (Carlson & Simpson 1996). Aside from the loss of hypolimnetic and benthic species of plants and animals, oxygen depletion facilitates the release of sediment-bound phosphorus and ammonia into the upper layers of the water column, which increases concentration of bio-available nutrients and further stimulates eutrophication (Carlson & Simpson 1996; Nürnberg & Peters 1984). Poor stratification of this shallow reservoir may exacerbate this situation and negatively affect the reservoir recreational and water supply uses.

Conductivity and pH.—The surface water pH did not differ significantly either among the monitored locations or between the two monitoring years (Table 2). The mean pH value was 8.37 in 2005 and 8.42 in 2006, which is characteristic of an eutrophic water body (Stumm 2004). The pH decreased with depth, ranging from 6.5 to 8.9 in the hypolimnion. The minimum was reached during the time of anoxic conditions when DO concentrations were below 1 mg/L.

The specific conductivity of the surface waters ranged from 350 to 563 µS/cm in 2005, and from 342 to 537 µS/cm in 2006 (Tables 2, 4). The highest values were recorded on 29 September 2005, at all seven locations. However, no significant differences were found among locations, water depth, months, or years.

Studies of fresh waters indicate that streams supporting good quality, mixed fisheries have conductivity in the range of 150 to 500 µS/cm, while waters with conductivity outside this range may not be suitable for certain species of fish or macroinvertebrates. The measured values of specific conductance at the reservoir exceeded this guideline only on few occasions in hypolimnion. Additionally, the results were well below the standard of 1200 µS/cm set by the Indiana Administrative Code (IAC 2000); therefore, the conductivity of water is not of concern at this reservoir.

Secchi disk transparency.—The mean summer values for water transparency (Z_S) were 94 cm in 2005 and 88 cm in 2006 (Table 2). The transparency at shallow locations 1 and 2 was significantly different from other sites (*P*

Table 3.—Physicochemical characteristics of the Prairie Creek Reservoir measured from May to November 2006. Abbreviation "SD" signifies standard deviation.

	pH		Specific conductivity (μS/cm)		Water Temperature (°C)		Dissolved oxygen (mg/L)		Turbidity (NTU)	
	Surface	Bottom	Surface	Bottom	Surface	Bottom	Surface	Bottom	Surface	Bottom
n	84	84	84	84	84	84	83	83	74	74
Mean	8.45	7.69	339	389	24.04	21.39	9.63	7.23	8.20	68.79
SD	0.27	0.51	25	85	3.89	3.84	2.63	13.53	13.45	128.30

	Orthophosphates-P (mg/L)		Nitrates-N (mg/L)		Total ammonia-N (mg/L)	
	Surface	Bottom	Surface	Bottom	Surface	Bottom
n	77	74	77	71	63	58
Mean	0.058	0.135	0.28	0.20	0.05	0.11
SD	0.077	0.136	0.23	0.17	0.07	0.11

$<$0.05), which can be attributed to shallow water depths. The lowest mean transparency for both monitoring years at the reservoir was measured in September during fall turnover (Fig. 2), which is usually characterized by increased turbidity due to mixing.

According to the Indiana Administrative Code (IAC 2000), water transparency of less than 1.8 m is a sign of eutrophication. While very low transparency of this reservoir is characteristic of a eutrophic or even hypereutrophic water body, it is slightly higher than at other lakes in Ecoregion 55 (Table 5). Lakes and reservoirs in this ecoregion have the shallowest Z_s among all Indiana lakes, with a mean value of 0.7 m (SPEA 2006). Abiotic turbidity, such as suspended sediment solids, was attributed to decreased water transparency in Indiana lakes and reservoirs (SPEA 2006). The causes of this abiotic turbidity at the reservoir could be shoreline erosion, recirculation of bottom sediment from heavy motorboat activities, strong wind/wave action, and sediment re-suspension due to carp (dominant in this reservoir). This observation is further supported by the results of Mayer et al. (1999) who concluded that diffusion, advection, and resuspension of benthic sediments can be intensified by the wind/wave action, gas liberation, and the feeding and spawning of carp.

Total reactive orthopohosphate.—The mean concentration of total reactive orthophosphate (TRP) was 0.05 mg/L in 2005 and 0.06 mg/L in 2006 (Table 2). The highest monthly means were recorded in June 2005 (0.11 mg/L) and in September 2006 (0.13 mg/L). The effects of the location, year and month were not significant. The recommended guidelines on the level of orthophosphate in lakes and reservoirs differ. Vollenweider (1975) concluded that the total phosphorus (TP) concentration greater than 0.1 mg/L can encourage algal growth. However, TP concentration of more then 0.03 mg/L is commonly used as a lower level at which algal growth and eutrophication occur (SPEA 2006). Consequently, high orthophosphate concentrations that comprise a fraction of total phosphorus are well within the eutrophic range for this water body.

During the 2006 monitoring period, the mean annual TRP concentration in the epilimnion was significantly lower than in hypolimnion (P = 0.004) (Table 3), which was most likely caused by anoxic conditions (DO $<$1 mg/L) that persisted from early June until September, especially at deeper locations (Table 4). Such conditions trigger the release of sediment-bound phosphorus into overlying waters, exacerbate eutrophication, and stimulate growth of algae and aquatic plants particularly in weakly stratified lakes and reservoirs (Carlson & Simpson 1996; U.S. EPA 2007; Nürnberg & Peters 1984). Internal nutrient loads from the benthos may become more significant in the regulation of annual nutrient concentrations within an open water body than external loads from its watershed or the atmosphere (U.S. EPA 2007; Gislason et al. 2004). In shallow eutrophic systems, such as Prairie Creek Reservoir, these releases are most prominent

Table 4.—Concentrations of dissolved oxygen in the hypolimnion at seven locations of the Prairie Creek Reservoir monitored from May through November 2006.

Date/Location	1	2	3	4	5	6	7
5/22/2006	8.57	13.04	10.72	8.87	8.69	7.34	8.33
6/05/2006	2.33	10.46	1.38	0.27	0.28	0.28	0.29
6/15/2006	0.46	5.61	0.42	0.33	0.32	0.33	5.45
6/30/2006	0.31	7.08	7.64	0.25	0.70	0.68	0.35
7/17/2006	7.24	3.76	3.80	0.25	0.39	0.27	2.27
7/28/2006	2.38	0.20	7.26	0.56	0.30	0.27	
8/11/2006	6.30	6.65	4.55	3.34	0.25	0.29	4.32
8/18/2006	1.39	2.47	3.73	0.26	1.92	0.26	3.41
8/24/2006	0.24	4.82		0.61	6.14	0.24	0.68
9/07/2006	8.61	10.77	7.75	0.75	7.09	0.58	7.83
9/21/2006		1.28	5.20	6.25	6.94	0.29	
10/10/2006		9.57	5.96	3.21	5.30	3.26	7.54

in summer months (U.S. EPA 2007; Moore & Reddy 1994; Riley & Prepas 1984). In other investigations, about 30–100% of phosphorus loading was attributed to internal release (Auer at al. 1993; Jacoby et al. 1982; Nürnberg 1988).

The results for TRP suggest that internal phosphorus loading may be a factor in eutrophication of this reservoir and may exacerbate this process in the future even if external sources of phosphorus in the watershed are reduced.

Total ammonia.—Concentrations of ammonia (NH_3-N) in the epilimnion ranged from 0.01 to 0.34 mg/L, reaching their maximum in September 2006 (Fig. 3). Concentrations found in hypolimnion were significantly higher (*P* <0.001), especially from May through the end of August. This finding can be attributed to ammonification and/or denitrification occurring in the benthos during prolonged anoxic conditions (Mayer et al. 1999; Nowlin et al. 2005; Stumm 2004) (Table 4). However, NH_3-N was well distributed throughout the entire water column in late summer and early fall as a result of increased mixing, disappearance of stratification (Fig. 2), and the effect of wind.

The concentration of NH_3-N in hypolimnion exceeded the IAC recommended level in 26% of the analyzed samples. Such high NH_3-N concentrations may negatively affect or become toxic to benthic organisms (Jørgensen et. al. 2005).

Nitrates.—The annual mean concentration of nitrates (NO_3-N) in the epilimnion was 0.45 mg/L in 2005 and 0.28 mg/L in 2006 (Table 2). The concentration did not differ significantly among seven monitored locations, although a significant difference was observed between the monitoring years (*P* = 0.006) and among months (*P* <0.05). June, July and August were different from September and October. However, no significant differences were observed between NO_3-N concentrations found in hypolimnion and epilimnion (*P* = 0.36) suggesting a uniform NO_3-N distribution in this shallow reservoir due to weak stratification and wind effect.

In general, NO_3-N concentrations in water bodies are below 1 mg/L. The maximum recommended level for modified warm water habitat is 1.6 mg/L (Ohio EPA 1999), and it can become toxic to humans at levels >10 mg/L (U.S. EPA 2006b). The highest monthly means in epilimnion were recorded in October 2005 (0.69 mg/L) and in May 2006 (0.35 mg/L), during the reservoir turnover. Although neither maximum exceeded the water quality guidelines, summer NO_3-N levels were higher than the ones measured in lakes and reservoirs of the same Indiana ecoregion (Table 5). Regardless, the concentrations of NO_3-N in the Prairie Creek Reservoir were well below recommended minimum levels to pose any negative effects.

Chlorophyll *a*.—The results for chlorophyll *a* (CHA) (Table 2) showed that the concentrations did not differ significantly among the locations (*P* >0.05). However, a statistically significant difference (*P* <0.05) was recorded for the months of June, July, and August. The depth at which CHA reached its maximum varied with sampling date and location. At deep locations, the maximum summer CHA concentrations reached 9.7 µg/L (location 4) and 9.51 µg/L (location 6), while at shallow locations the maximum values reached 32.11 µg/L at location 1 and 15.11 µg/L at location 2.

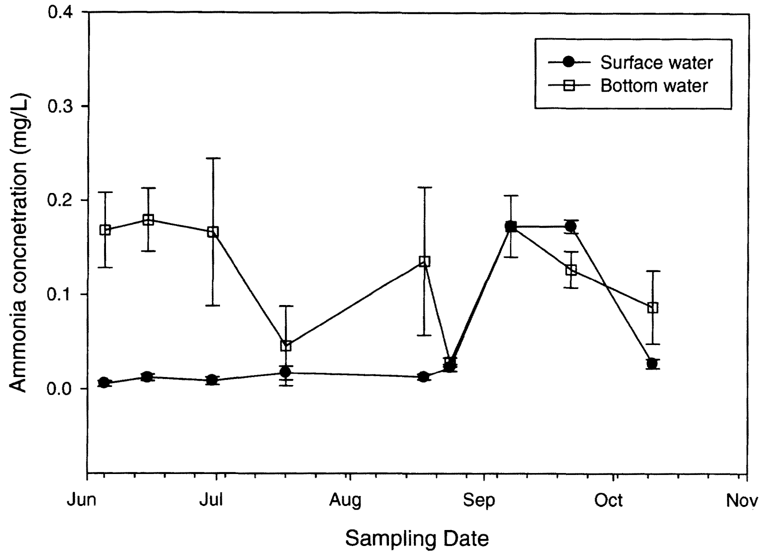

Figure 3.—Spatial and temporal trend of ammonia-N concentrations recorded at Prairie Creek Reservoir from June through October 2006. Symbols identify means, and bars signify standard error.

Carlson (1977) has indicated that eutrophic lakes are characterized by summer CHA concentrations in the range of 10 to 30 µg/L, thus suggesting a eutrophic state for this reservoir. In addition, the median CHA concentration measured in July and August in the photic zone was greater than the summer median value measured in other lakes of the Indiana Ecoregion 55 (Table 5) (SPEA 2006).

Haven & Nürnberg (2004) concluded that CHA concentration is positively correlated with concentration of total phosphorus (TP). However, high concentrations of orthophosphate measured at this reservoir did not correspond to equally high CHA levels. A similar trend was observed also at other Indiana lakes and reservoirs in the Ecoregion 55, and was attributed to the effects of turbidity caused by suspended inorganic matter and macrophyte growth in those water bodies (SPEA 2006). While macrophytic biomass was not investigated in this study, the field observations of extensive growth of aquatic plants suggest that concentrations of CHA could be affected by macrophytes that reduce availability of phosphorus for algal growth, and decrease the validity of total phosphorus as a predictor of CHA concentration (Rooney & Kalff 2003).

Trophic status.—The trophic character of a lake or reservoir, its productivity, as well as its utility for fishing or swimming is determined by

Table 5.—Comparison of the Prairie Creek Reservoir water quality with the lakes and reservoirs in the Indiana Ecoregion 55. An asterisk identifies the median values measured only in epilimnion from July to August 2005–2005. A "†" identifies median of epilimnion and hypolimnion data collected in July and August for Indiana lakes and reservoirs in 1999–2003 (SPEA 2006). The symbol "††" identifies median value for the epilimnion and hypolimnion data collected in July and August 2006.

	Prairie Creek Reservoir		Indiana Ecoregion 55†
	2005*	2006††	1999–2003
Secchi disk transparency (m)	0.94	0.88	0.7
Orthophophates-P (mg/L)	0.06	0.04	-
Nitrates-N (mg/L)	0.20	0.25	0.024
Ammonia-N (mg/L)	-	0.015	0.163
Chlorophyll *a* (µg/L)	7.92	7.24	5.34
% water column with DO > 1 mg/L	-	74%	58%

calculations of the Carlson Trophic State Index (TSI) (Carlson 1977). This empirical index is based on three in-lake variables, e.g., average summer CHA, summer Z_s, and summer TP to obtain scores in the range of zero (oligotrophic status) to 100 (hypereutrophic and very productive) (Carlson 1977). TSI for the Prairie Creek Reservoir based on the average summer Z_s was 62 in 2005 and 64 in 2006, while the TSI based on CHA was 50 in 2005 and 49 in 2006. The score for Z_s was well within the eutrophic range (SPEA 2006); however, the TSI based on CHA measurements was at the boundary of mesotrophy and eutrophy (Carlson 1977; Carlson & Simpson 1996). This discrepancy in TSI scores is in agreement with the findings for other Indiana lakes and reservoirs that were found to produce less CHA than would be expected from their phosphorus concentration. Additionally, TSI transparency scores were lower than chlorophyll scores due to turbidity caused by suspended inorganic matter (SPEA 2006). Carlson (1977) also found that lakes with high turbidity caused by suspended inorganic matter do not produce as much algae as their phosphorus concentration might suggest. Moreover, higher concentration of macrophytes in a lake will result in lower concentrations of CHA than would be expected from measured phosphorus concentration (Canfield et al. 1983; Rooney & Kalff 2003). In macrophyte-dominated reservoirs the trophic state of a waterbody might be underestimated when only data based on CHA are considered (Canfield et al. 1983). Field observations at the Prairie Creek Reservoir confirm extensive growth of aquatic plants; and, therefore, the assessment of macrophyte biomass should be performed in future investigations to gain an understanding of the relationship between CHA and TP.

Management implications.—This study provides the first comprehensive assessment of water quality at Prairie Creek Reservoir. The results demonstrate that Prairie Creek Reservoir is an eutrophic, dimictic reservoir with weak stratification, low water transparency, anoxic conditions spanning from June to September, and high concentration of orthophosphate. Additionally, water quality is uniform throughout the reservoir since no significant differences (with the exception of results for water transparency) were found among seven monitored locations. Therefore, future studies should focus on the deepest locations that represent water quality of the entire reservoir. The eutrophic status of this reservoir is caused by high phosphorus concentrations, with internal phosphorus loading in the reservoir possibly playing a significant role in eutrophication. The current management of pollutant runoff in the reservoir watershed is limited to no-till agriculture and some buffer strips. This and any future management approaches might not be sufficient to prevent the progression of eutrophication if internal loading of phosphorus in the reservoir is significant. To preserve and maintain the current reservoir uses, appropriate in-lake as well as watershed management approaches will need to consider both internal nutrient loading in the reservoir and external nutrient inputs from its watershed. Watershed management was shown to be costly and ineffective in improving water quality of lakes and reservoirs with high internal loading of nutrients (Osgood 1989; Welch & Jacoby 2001). Consequently, determination of the importance of internal and external nutrient loading is necessary for development and implementation of successful management strategies in the future.

Finally, because a minimum of 8–10 years of monitoring is usually required to detect water quality trends in lakes and reservoirs (Smeltzer et al. 1989), monitoring of the reservoir should be continued annually. Future studies should also examine the role of macrophytes in the process of eutrophication at this reservoir.

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LITERATURE CITED

Auer, M.T., N.A. Johnson, M.R. Penn & S.W. Effler. 1993. Measurement and verification of rates of sediment phosphorus release for hypereutrophic urban lake. Hydrobiologia 253:301–309.

APHA (American Public Health Association). 1998. Standard methods for the examination of water and wastewater. (Lenore S. Clesceri, A.E. Greenberg & A.D. Eaton, eds.), 20th ed.

Canfield, D.E. Jr., K.A. Langeland, M.J. Maceina, W.T. Haller, J.V. Shireman & J.R. Jones. 1983. Trophic state classification of lakes with aquatic macrophytes. Canadian Journal of Fisheries and Aquatic Sciences 40:1713–1718.

Carlson, R.E. 1977. A trophic state index for lakes. Limnology and Oceanography 22:361–369.

Carlson, R.E. & J. Simpson. 1996. A Coordinator's Guide to Volunteer Lake Monitoring Methods. North American Lake Management Society. 96.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley & V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8:559–568.

Cescon, C.T. 1997. The 1996 Population dynamics of microcrustacean zooplankton at Prairie Creek Reservoir in Delaware County, Indiana. Master's Thesis. Ball State University, Muncie, Indiana.

Condie, S.A. & I.T. Webster. 2002. Stratification and circulation in a shallow turbid waterbody. Environmental Fluid Mechanics 2:177–196.

DCSWD (Delaware County Soil and Water Conservation District). 2004. White River watershed project management plan. http://www.whiteriverwatershedproject.org; verified October 31, 2007.

Delaware Muncie-Metropolitan Plan Commision. 2007. Prairie Creek Reservoir Master Plan.

Gislason, S.R., E.S. Eiriksdottie & J.S. Olafsson. 2004. Chemical composition of interstitial water and diffusive fluxes within the diatomaceous sediment in Lake Myvatn, Iceland. Aquatic Ecology 38: 163–175.

Goward, K.J. 2004. Relationship of nutrients and pesticides to land use characteristics in three subwatersheds of the Upper White River, Indiana. Master's thesis, Ball State University, Muncie, Indiana.

Haman, R.L. 1964. A general ecological study of the Prairie Creek Reservoir. Master's Thesis, Ball State University, Muncie, Indiana.

Haven, K. & G. Nürnberg. 2004. The phosphorus-chlorophyll relationship in lakes: Potential influences of color and mixing regime. Lake and Reservoir Management 20:188–196.

Herb, W.R. & H.G. Stefan. 2004. Temperature stratification and mixing dynamics in a shallow lake with submerged macrophytes. Lake and Reservoir Management 20:296–308.

IAC (Indiana Administrative Code). 2000. Article 2, Water Quality Standards. http://www.in.gov/legislative/iac/title327.html.

IDEM (Indiana Department of Environmental Management). 2000. 2000 Indiana State of the Environment Report: Water Quality.

IDEM (Indiana Department of Environmental Management). 2004. Indiana Integrated Water Quality Monitoring and Assessment Report.

IDEM (Indiana Department of Environmental Management). 2006. Indiana Integrated Water Monitoring and Assessment Report 2006. http://www.state.in.us/idem/programs/water/305b/index.html.

Holdren, G.C. & S.J. Souza. 2004. Which monitoring program? Balancing cost, needs and accuracy. Lakeline 24:10–13.

Jacoby, J.M., D.D. Lynch, E.B. Welch & M.A. Perkins. 1982. Internal phosphorus loading in shallow eutrophic lake. Water Res. 16:911–919.

Jørgensen, S.E., H. Loffler, W. Rast & M. Straskraba. 2005. Lake and Reservoir Mismanagement. Pp. 43–106. *In* Lake and Reservoir Management. Elsevier, B.V. Netherlands.

Marshall, D.W., M. Otto, J.C. Panuska, S.R. Jaeger, D. Sefton & T.R. Baumberger. 2006. Effects of hypolimnetic releases on two impoundments and their receiving streams in southwest Wisconsin. Lake and Reservoir Management 22:223–232.

Mau, D.P. 2002. United States Geological Survey Water-Resources Investigations Report 02-4073. Sediment deposition and selected water-quality characteristic in Cedar Lake and Lake Olathe, Northeast Kansas, 2000. Retrieved 8 October 2005. http://ks.water.usgs.gov/Kansas/pubs/reports/wrir.02-4073.html.

Mayer, T., C. Ptacek & L. Zanini. 1999. Sediments as a source of nutrient to hypereutrophic marches of Point Pelee, Ontario, Canada. Water Research 33: 1460–1470.

Moore, P.A. Jr. & K.R. Reddy. 1994. Role of Eh and pH on phosphorus geochemistry in sediments of Lake Okeechobee, Florida. Journal of Environmental Quality 23:955–964.

Nowlin, W.H., J.L. Evarts & M.J. Vanni. 2005. Release rates and potential fates of nitrogen and phosphorus from sediments in a eutrophic reservoir. Freshwater Biology 50:301–322.

Nürnberg, G.K. & R.H. Peters. 1984. Biological availability of soluble reactive phosphorus in anoxic and oxic freshwaters. Canadian Journal of Fisheries and Aquatic Sciences 41:757–765.

Nürnberg, G.K. 1988. Prediction of phosphorus release rates from total and reductant phosphorus in anoxic lake sediments. Canadian Journal of Fisheries and Aquatic Sciences 45:453–462.

Osgood, R. 1989. An evaluation of the effects of watershed treatment systems on the summertime phosphorus concentration in Metropolitan Area lakes. Natural Resources Div., Metropolitan Council, St. Paul, Minnesota.

Ohio EPA (Environmental Protection Agency). 1999. Association between nutrients, habitat, and the aquatic biota in Ohio rivers and streams. Ohio EPA Technical Bulletin MAS/1999-1-1, Columbus, Ohio.

Riley, E.T. & E.E. Prepas. 1984. Role of internal phosphorus loading in two shallow productive lakes in Alberta, Canada. Canadian Journal of Fisheries and Aquatic Sciences 41:845–855.

Rooney, N. & J. Kalff. 2003. Submerged macrophyte-bed effects on water column phosphorus, chlorophyll *a*, and bacterial production. Ecosystems 6:797–807.

Sharpley, A.N., M.J. Hedley, E. Sibbesen, A. Hillbricht-Ilkowska, W.A. House & L. Ryszkowski. 1995. Phosphorus transfer from terrestrial to aquatic ecosystems. *In* Phosphorus in the Global Environment (H. Tiessen, ed.). John Wiley & Sons, Ltd.

Smeltzer, E., V. Garrison & W.W. Walker, Jr. 1988. Eleven years of lake eutophication monitoring in Vermont: A critical evaluation. *In* Proceedings of National Conference on Enhancing States' Lake Management Programs. Chicago, Illinois. 53–62.

Smith, V.H. 2003. Eutrophication of freshwater and coastal marine ecosystems: a global problem. Environmental Science and Pollution Research International 10:126–139.

SPEA (School of Public and Environmental Affairs). 2006. Indiana Lake Water Quality Assessment Report for 1999–2003.

Stumm, W. 2004. Chemical processes regulating the composition of lake waters. Pp. 79–106. *In* Lakes Handbook (P.E. O'Sullivan & C.S. Reynolds, eds.). Blackwell Science Ltd.

U.S. EPA (Environmental Protection Agency). 1997. Monitoring and Assessing Water Quality. Volunteer Stream monitoring: A methods manual http://www.epa.gov/volunteer/stream/index.html; Retrieved 23 March 2007.

U.S. EPA (Environmental Protection Agency). 2000. 2000 National Water Quality Inventory: Chapter 3. Lakes, reservoirs and ponds. http://www.epa.gov/305b/2000report/; Retrieved 9 February 2007.

U.S. EPA (Environmental Protection Agency). 2002. Level III ecoregions of the United States. National Health and Environmental Effects Research Laboratory, Corvalis, Oregon. http://dcnr.nv.gov/nrp01/ecoregions_us.pdf; Retrieved 7 November 2007.

U.S. EPA (Environmental Protection Agency). 2006a. Monitoring and assessing water quality. http://www.epa.gov/owow/monitoring/monintr.html.

U.S. EPA (Environmental Protection Agency). 2006b. Drinking water contaminants. http://www.epa.gov/safewater/contaminants/index.html.

U.S. EPA (Environmental Protection Agency). 2007. Monitoring and assessing water quality. Volunteer Lake Monitoring: A methods manual. http://www.epa.gov/owow/monitoring/lakevm.html; Retrieved 23 March 2007.

Vollenweider, R.A. 1975. Input-output models with special reference to the phosphorus loading concept in limnology. Schweizerische Zeitschrift fur Hydrologie 37:53–84.

Weld, J.L., R.L. Parsons, D.B. Beegle, A.N. Sharpley, W.J. Gburek, W.J. & W.R. Clouser. 2002. Evaluation of phosphorus-based nutrient management strategies in Pennsylvania. Journal of Soil and Water Conservation 57:448–454.

Welch, E.B. & J.M. Jacoby. 2001. On determining the principal source of phosphorus causing summer algal blooms in Western Washington Lakes. Journal of Lakes and Resource Management 17:55–65.

Wetzel, R.G. 1983. Limnology. Second ed. Saunders Publishing, Philadelphia, Pennsylvania. 753.

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