THE POTENTIAL FOR LATE-SUMMER HYPOLIMNETIC WITHDRAWALS TO REDUCE SEDIMENT PHOSPHORUS CONCENTRATIONS IN DEVIL'S LAKE

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The Potential for Late-Summer Hypolimnetic Withdrawals to Reduce Sediment Phosphorus Concentrations in Devil's Lake

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Introduction

Devil's Lake is located approximately 65 km northwest of Madison in Wisconsin's most heavily used state park -- Devil's Lake State Park[1]. Beginning in the late 1970s, concerns have been raised about deteriorating water clarity in the lake (Lillie and Mason 1986). As a response to these concerns, the Wisconsin Department of Natural Resources, Bureau of Research (WDNR) conducted a detailed limnological study during 1986-87 that addressed all facets of the lake's water management problems (WDNR 1988).

From that study and research on other lakes (Nürnberg 1985), internal recycling of phosphorus (P) from the lake's extensive profundal (deep-water) sediments was identified as the major cause of the blue-green algal blooms that lead to water clarity declines during late summer. When dissolved oxygen (D.O.) concentrations near the bottom sediments are depleted by late July, iron (Fe) in insoluble hydrous-Fe-oxides in the sediments is reduced and the compounds are solubilized. The P bound to these compounds is then released, leading to a rapid increase in P (and Fe) concentrations in the overlying anoxic hypolimnetic water. P continues to build up in the hypolimnion until destratification causes these waters to mix with the lake surface waters where an algal bloom develops from the P enrichment.

Although external P loadings are now much lower than in previous years when the lake was surrounded with cottages and resorts[2], internal P recycling continues to be a significant problem because the lake has no outlet for dilutional flushing. Consequently, three management options were proposed for reducing internal P recycling in Devil's Lake: (1) aeration, (2) alum treatment, and (3) hypolimnetic withdrawal (WDNR 1988). Lake aeration was considered impractical because it would not permanently reduce P recycling rates and because the bottom area needing aeration is very large. Alum treatment, while having some success as a lake rehabilitation technique (Cooke et al. 1993), has not been accepted because local environmentalists and park managers are opposed to adding chemicals to a lake perceived as one of Wisconsin's "natural jewels."

Hypolimnetic withdrawal -- the selective removal of anoxic P-enriched water from the lake's deepest region -- was proposed because lake levels were very high during the 1986-87 study period and lowered levels would prevent ice damage to the park shoreline and associated buildings. Furthermore, a review by Nürnberg (1987) indicated that hypolimnetic withdrawal had produced positive water quality improvements in lakes where the technique had been tried (mostly in Europe). In a majority of those lakes, both hypolimnetic and epilimnetic (surface) P concentrations decreased over time, particularly in lakes with long-term withdrawals. The depth and duration of hypolimnetic anoxia also decreased in some cases. Lakes where external P inputs remained high showed the least water quality improvements.

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[1] Over a million visitor-days have been recorded annually at Devil's Lake State Park since the early 1950s (Lange and Tuttle 1975); as many as 1.7 million visitor-days have been recorded in recent years (K. Lange, Park Naturalist, pers. comm.)

[2] Devil's Lake at one time was surrounded with as many as 4 resorts and over 60 private cottages. By the late 1960s, all private facilities except for 4 cottages were gone as the state expanded its parkland surrounding the lake.

However, hypolimnetic withdrawal, which has mostly been applied as a design modification to a lake's outlet structure, is not suitable for Devil's Lake because the lake has no natural surface water outlet. Hypolimnetic water could be pumped or siphoned from the lake, but this creates concerns about lowered lake levels because the lake has no major surface water inflows to replace the discharged water. One relatively new technique that could be applied to Devil's Lake is to recycle the hypolimnetic water after removing P in a shore-based flocculation/filtration process using Fe (or aluminum) salts. This treatment process is successfully being used to remove P from inflowing waters to three lakes in Germany (Bernhardt 1980, Clasen and Bernhardt 1987, Chorus and Wessler 1988, Heinzmann and Chorus 1994). Another technique that could be used is to replace the hypolimnetic water pumped from the lake in late summer/early fall with snowmelt runoff from a nearby stream the following spring. The hypolimnetic water treatment technique would be much more costly, but would eliminate concerns about the poor quality of the hypolimnetic water being discharged downstream, an issue frequently raised for hypolimnetic withdrawal (Nürnberg 1987, Cooke et al. 1993).

In 1991 we began evaluating whether hypolimnetic withdrawal would reduce sediment P concentrations and concomitant sediment P release rates in Devil's Lake. This evaluation was deemed essential before proceeding further with any lake rehabilitation program. We measured Fe-bound P (Fe-P) concentrations in sediments collected throughout the lake's profundal zone prior to summer hypolimnetic anoxia to determine zones with a high potential for P release. We also measured P concentrations in deep-water sediments collected before and after anoxia to estimate the amount of P released into the overlying water each season. In addition, we performed a long-term laboratory column experiment to simulate the effect of multi-year hypolimnetic withdrawals on reducing sediment P concentrations and release rates. At the same time, the U.S. Geological Survey (USGS) was funded to begin a lake level and water budget modeling study to determine the impact on lake levels from multi-year hypolimnetic withdrawals. The results of that study are presented elsewhere (Krohelski and Batten 1994).

The results from the lake sediment analyses and the column experiment are the main focus of this report. Much of the limnological data that were included in previous studies (i.e., Lillie and Mason 1986, WDNR 1988) is not repeated. We have included water clarity, and lake P, Fe, and chlorophyll-*a* concentration data that were collected during 1986-93 as part of the WDNR's basic lake sampling program to facilitate an understanding of the lake's water quality problems. In addition, new insights on long-term water clarity trends for Devil's Lake are presented to better document the lake's problems. Finally, metric hydrographic maps that were prepared from soundings recorded in feet during January 1985 are also given in order to facilitate our sediment P analyses and future rehabilitation work.

Lake Description

Devil's Lake is surrounded by ancient quartzite bluffs to the east, west, and the south (Attig et al. 1990). Originally, the pre-glacial Wisconsin River flowed through a gap between these bluffs in the South Range of the Baraboo Hills. At the end of the last ice age, Devil's Lake was formed between terminal moraines deposited at both the north and southeast ends of the gap; the Wisconsin River was diverted to the east around the Baraboo Hills.

Devil's Lake has a unique "bath-tub" hydrography (Fig. 1). It has a surface area of 149 ha and a maximum depth of approximately 14.3 m, although its maximum depth can range anywhere between 13 and 15 m. Its mean depth averages about 9.3 m. Because the east and west shorelines drop off rapidly from the surrounding steep bluffs, the lake's littoral zone is mostly confined to the north and south ends of the lake. Much of the lake's profundal zone is a broad flat plain.

The lake has a relatively small watershed (6.86 km^2), most of which is forested (Lillie and Mason 1986). Its watershed to lake area ratio is only 4.6. The lake has no outlet and only one minor surface water inlet that drains through a small wetland. Groundwater and direct precipitation constitute a major portion of the lake's annual water inflows (Krohelski and Batten 1994). Consequently, long-term lake levels mirror precipitation trends, while seasonal lake level declines reflect high summer evaporation rates and discharge to the groundwater. Levels recorded during 1980-93 demonstrate the long-term and seasonal fluctuations that can occur in Devil's Lake (Fig. 2). Because of the major flood in 1993, levels fluctuated by as much as 2.5 m during those years.

Methods

Lake Sampling

Lake sampling in 1986-93 was conducted biweekly beginning in the spring or early summer months and continuing through November. All sampling was done at a buoy positioned each year near the lake's deepest location (Fig. 1), although in 1988-91 the buoy was positioned more to the center of the lake and approximately 0.5 m shallower than the deepest location. Sampling consisted of vertical temperature and D.O. profiles using the modified Winkler method, Secchi disk transparency readings using a 20 cm black/white disk, and total P (TP) analyses of water collected from 0 m and 4 m. Chlorophyll-*a* analyses using the Trichromatic technique (APHA et al. 1985) were performed on a 0-4 m composite sample. After anoxia developed in the hypolimnion, TP was measured in most years at intervals of 0.2 m, 0.5 m, 1.0 m, 1.5 m, etc. from the bottom of the lake until the water was oxygenated. TP was also measured at 0.5 m from the bottom on other dates to help define the periods when hypolimnetic P concentrations were the greatest. In all years except for 1988-90, total Fe was measured in the anoxic hypolimnetic waters. All hypolimnetic sampling was conducted with a peristaltic pump to ensure that the same water depth relative to the bottom was sampled on each date in a given year. Samples for P and Fe analyses were preserved immediately after collection with sulfuric and nitric acids, respectively. The P and Fe analyses were conducted at the State Laboratory of Hygiene in Madison using U.S. Environmental Protection Agency (EPA) prescribed procedures.

Sediment Coring

Sediment cores were taken along a north-south transect in Devil's Lake on 12 June 1991 by a scuba diver pushing a 6.7 cm diameter plastic tube into the bottom sediments (Fig. 3). The central "deep-hole" core was taken at a 0.5 m shallower water depth to the north of the deepest location in the lake (sampling station labelled as 0.5 N in Fig. 3). Additional cores were taken along the transect at 1.0 m depth intervals from the central

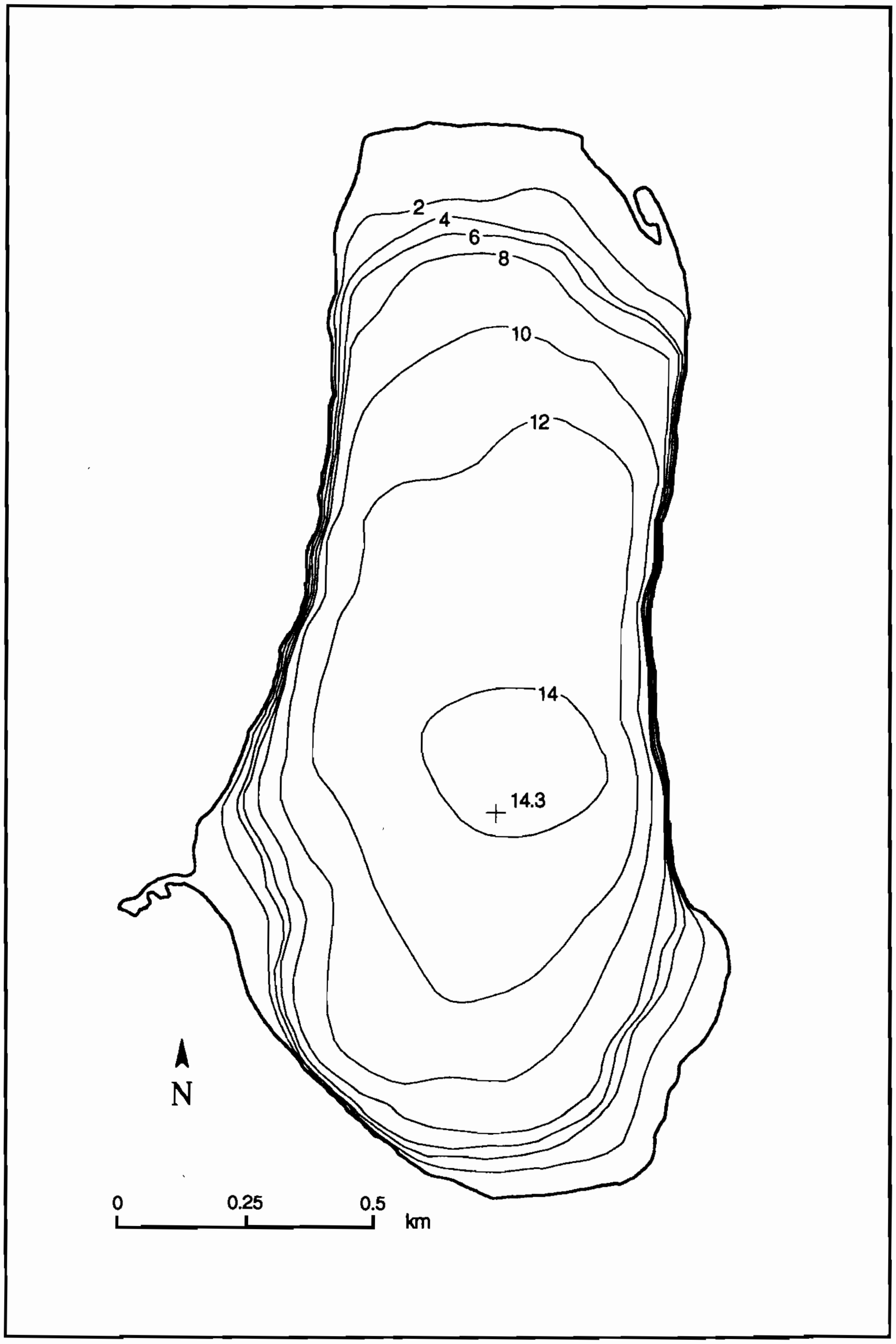

Figure 1. Hydrographic map of Devil's Lake with depth contours in meters. Depth soundings made in January 1985 when lake level was about 963 feet above mean sea level.

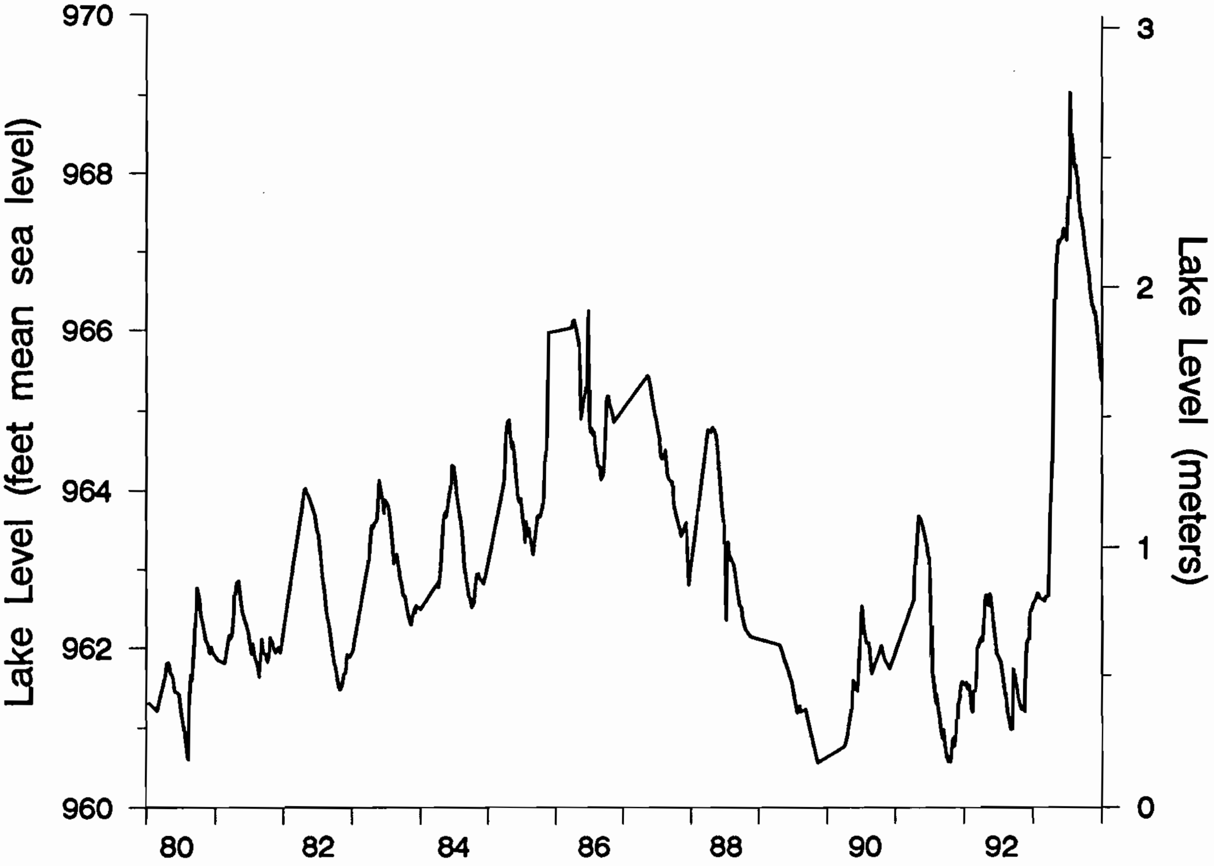

Figure 2. Surface water levels in Devil's Lake, 1980-93.

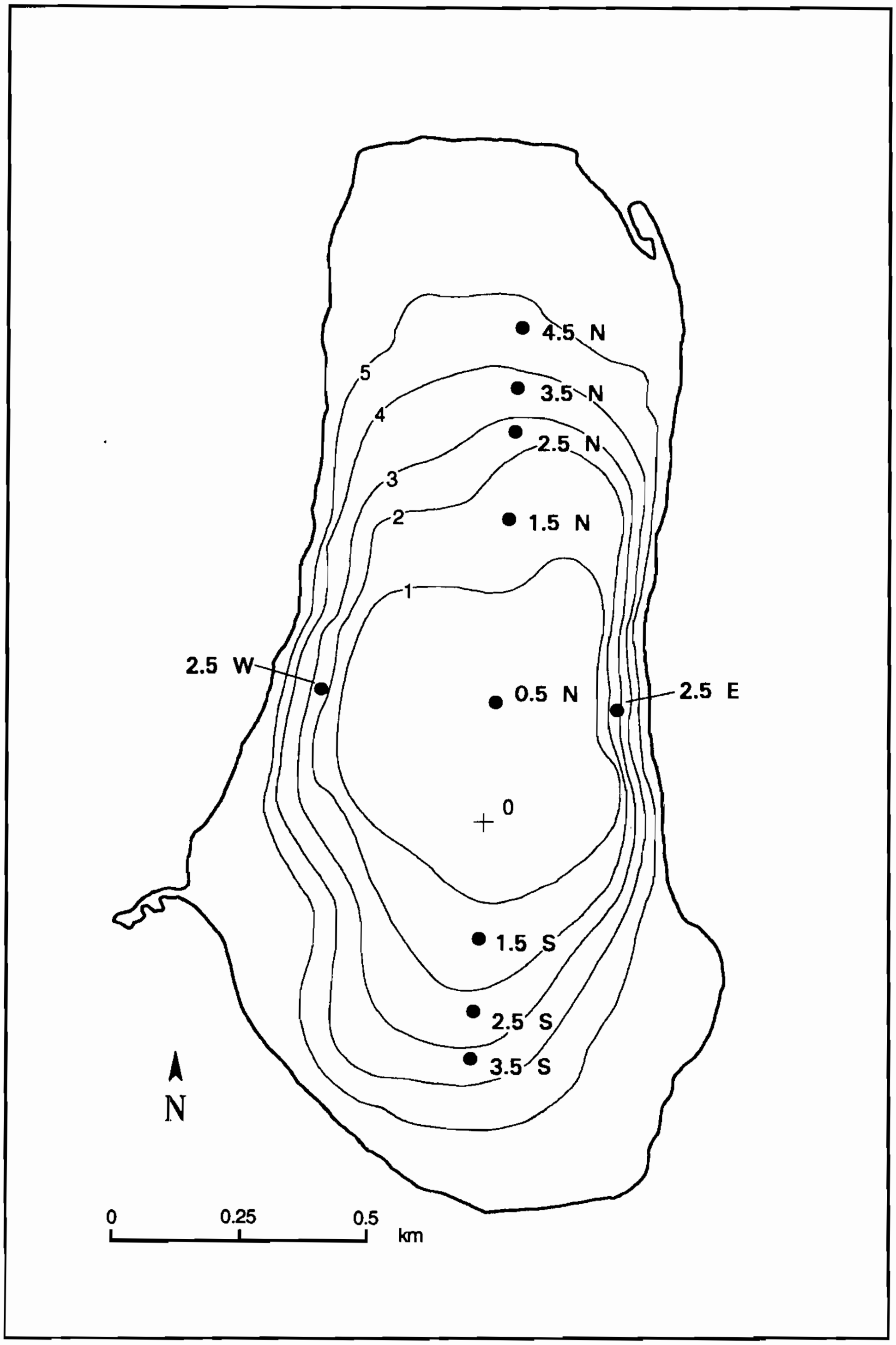

Figure 3. Hypolimnetic depth contours in meters from the deepest spot in Devil's Lake, and location of sediment cores (solid circles) taken in 1991.

core (stations 1.5 N to 4.5 N and 1.5 S to 3.5 S). Two cores were also taken at a depth 2.0 m shallower than the central core to the east and west main transect (stations 2.5 E and 2.5 W). In order to detect changes in the sediment P concentrations after an extended period of anoxia in Devil's Lake, two cores were taken at stations 0.5 N and 1.5 N on 26 September 1991 prior to complete turnover of the water column.

All cores were sectioned immediately in the boat into the following depth intervals: 1-cm intervals for 0-10 cm sediment depths, 2-cm intervals for 10-20 cm depths, and 5-cm intervals for sediment depths >20 cm. Total sediment depths sampled in each core were approximately 25-30 cm. All sediment sections were stored in plastic bags and refrigerated at 4°C. Sediments were later freeze-dried for P analyses.

Column Experiment

Four 10.2-cm diameter cores each containing 30-35 cm of sediment were taken from the central area of Devil's Lake on 17 July 1991. Back at the lab, the sediments were pushed to the top of each core barrel (50 cm) and the top 5 cm of sediment were removed from two cores. All four cores were next attached to 250-cm length tubes and filled with low-P surface lake water. Sediment/water columns A and B constituted the intact sediment cores, while columns C and D were the cores with the top 5 cm of sediment removed. Column temperatures were maintained at 10-12°C by a circulating cooling system comprised of a water chiller, pump, and copper tubing wound around the entire length of each column, which was then wrapped with insulation. Each column had water sampling ports positioned at various intervals above the sediment/water interface. Initial sediment P concentrations were analyzed from an additional 14.3-cm diameter core taken on the same date as the four cores used in the columns.

TP concentrations in water sampled at 20 cm above the sediment-water interface in each column were analyzed weekly for a period of 8 weeks. At the end of eight weeks, each column was drained into a bucket for a composite TP sample (a "flushing") and refilled with new lake water. Ten flushings were conducted to simulate multiple withdrawals from the lake. In order to consistently develop anoxic conditions in the columns, the lake water used to fill the columns was deoxygenated by bubbling with N_2 gas to D.O. <1 mg/L. The first time the columns were filled, the N_2 bubbling was done in the columns between weeks 1 and 2. For subsequent fillings of the columns, the water was deoxygenated prior to filling; a slow bubbling in the columns was also conducted to remove the last traces of D.O. Efforts were made to not disturb the column sediments during filling or N_2 bubbling.

Sediment P release rates in each column were calculated from the composite TP concentration for each 8-week period in order to compare rate changes after multiple flushings. (The concentration obtained at the end of 10 weeks was used in the first flushing experiment, because the column water was not deoxygenated until just before week 2.) At the end of the 10th flushing, the top 10 cm of sediments in columns A and B and the top 5 cm in columns C and D were sectioned in 1-cm intervals for sediment P analyses.

Sediment Chemistry Procedures

Perchloric acid digestion was used for sediment TP analysis (Sommers and Nelson 1972). Approximately 0.2 g of freeze-dried sediment was added to a digestion tube followed by a mixture of 5 mL HNO_3 (70%) and 3.5 mL $HClO_4$ (60%). Tubes were placed in an aluminum digestion block and heated on a hot plate at 130-140°C until the HNO_3 evaporated (2-3 hours). The temperature was then raised to 203°C for 75 minutes; pyrex funnels were placed on the tubes to prevent the samples from boiling dry. Following the digestion, the samples were allowed to cool and then diluted to 50 mL with milli-Q water. Samples were stoppered, inverted several times, and allowed to stand overnight. The digestate was transferred to 60 mL polyethylene bottles until analyzed colorimetrically for orthophosphate using the Ascorbic Acid Method (APHA et. al 1985). Immediately prior to analyses, the samples were neutralized with 6N NaOH using phenolphthalein as an indicator. Color was discharged with 5N H_2SO_4.

Sediment inorganic P considered a non-apatite, largely Fe-associated fraction that would be released into interstitial and hypolimnetic waters under anoxic conditions was also extracted from the freeze-dried sediments. This procedure entailed mixing sediments and a reagent solution of 0.1N NaOH and 1N NaCl at a sediment to solution ratio of 1:1000 (45 mg sediment per 45 mL of reagent) and mechanically shaking for 18 hours (Armstrong et al. 1979). Following extraction, the samples were centrifuged, filtered (0.45 μm Nucleopore filter), and analyzed for orthophosphate. Prior to colorimetric analysis, samples were neutralized with 5N H_2SO_4 using phenolphthalein as an indicator. We refer to this NaOH-extractable fraction throughout this report as Fe-bound P, or simply Fe-P.

Results

Lake Water Sampling

Secchi disk readings, and epilimnetic (0-4 m) TP and chlorophyll-*a* concentrations all showed similar patterns during the open water months of 1986-93 with a few exceptions (Fig. 4). Secchi readings were generally ≥7 m during June and July before declining to a seasonal minimum of 2-3 m by September or early October. This decline in water clarity corresponded to an increase in TP and chlorophyll-*a*, which was indicative of an increase in algal biomass. However, Secchi readings in 1993 remained relatively high through September until a minor decline occurred in October.

The late-summer decline in water clarity observed during 1986-93 was consistent with the findings of Lillie and Mason (1986) for the late 1970s through 1985. However, when they compared earlier Secchi readings with readings recorded through 1985, they felt that long-term trends for the summer months (average of Secchi readings recorded in July through September) were less obvious. Based on the raw Secchi disk data compiled by Lillie and Mason (1986), additional unpublished data that were found for the early 1900s, and data from the WDNR lake sampling program conducted in 1986-93, a decline in water clarity during both August and September occurred around 1977 (Fig. 5). Although early data are sketchy, this decline was most striking in September.

Hypolimnetic D.O., total Fe and TP concentrations in water depths ≤0.5 m from the bottom sediments (near the deepest location) during 1986-93 demonstrated the interrelationship between these three parameters (Fig. 4). As hypolimnetic D.O. concentrations declined to around 0.3 mg/L, Fe was reduced in the lake sediments and released into the overlying water. In hypolimnetic waters where D.O. concentrations were

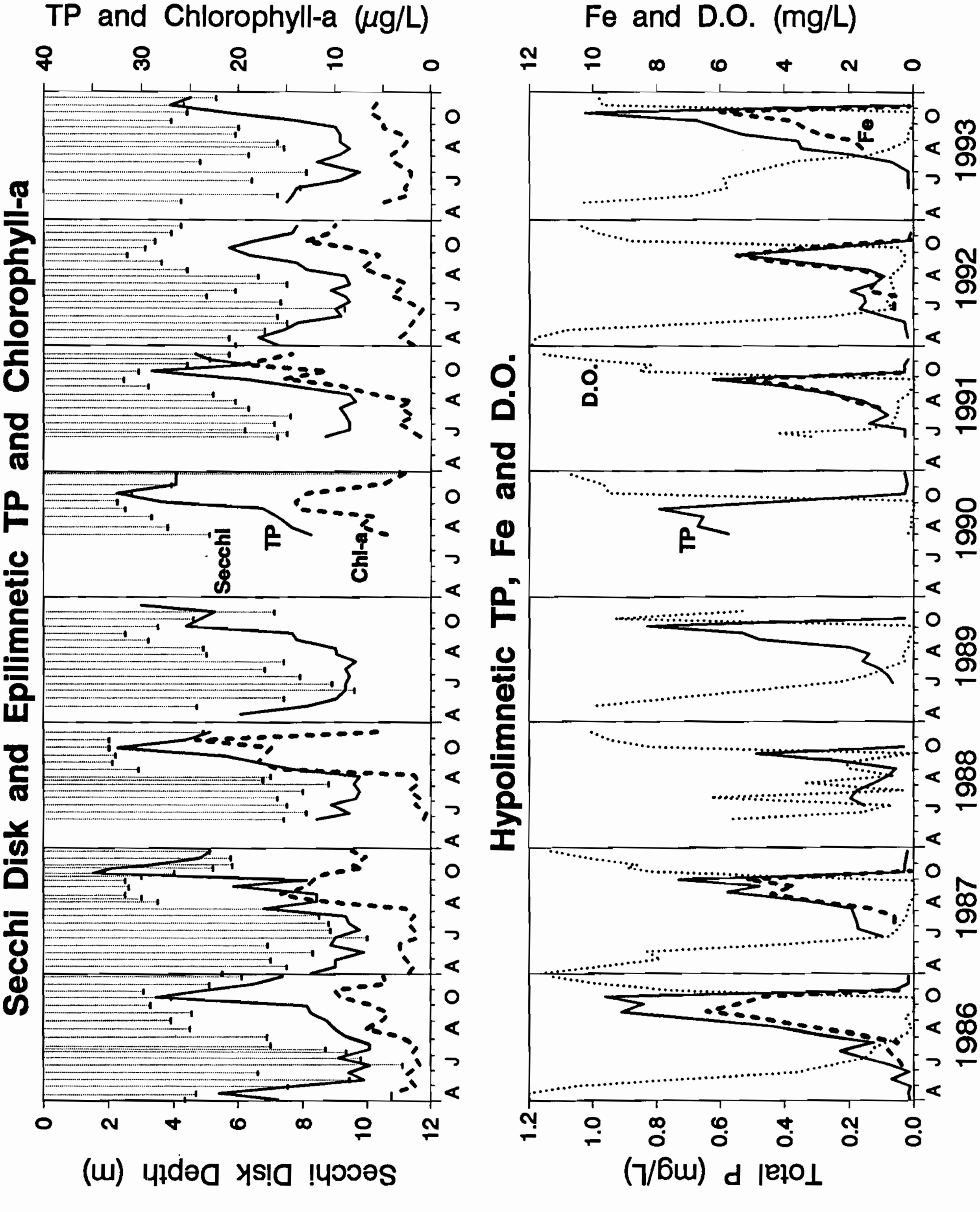

Figure 4. Secchi disk readings, epilimnetic (0-4 m) TP and chlorophyll-*a* concentrations, and hypolimnetic (bottom 0.5 m) TP, total Fe, and D.O. concentrations during April to November 1986-93 in Devil's Lake. Secchi (dash marker and thin vertical dotted line); TP (solid line), Chlorophyll-*a* and Fe (heavy dashed line); D.O. (dotted line)

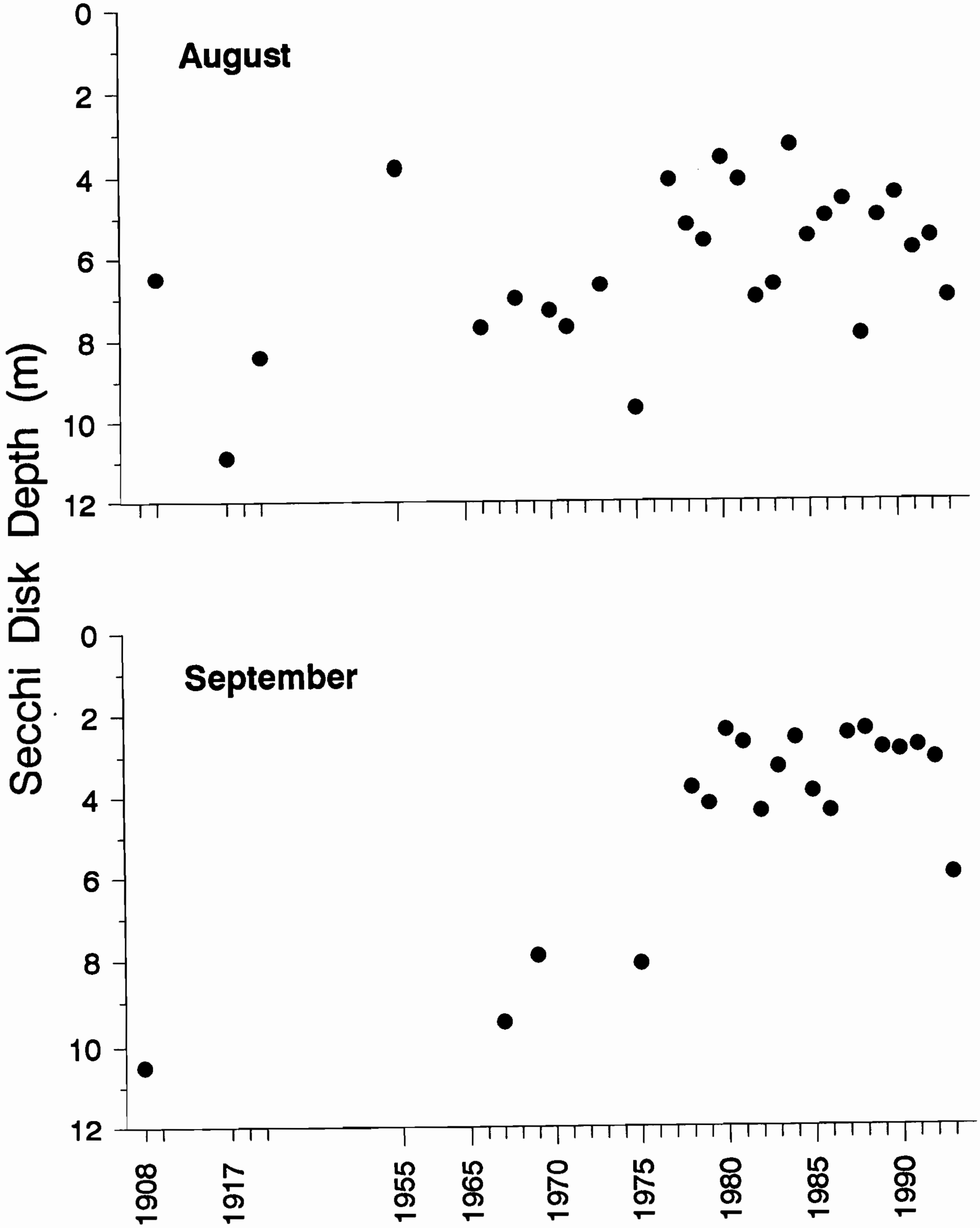

Figure 5. Average Secchi disk readings for both August and September for selected years during 1908-93 in Devil's Lake.

greater than 0.5 mg/L, Fe concentrations did not build up. (This occurred both in hypolimnetic waters with slightly higher D.O. concentrations and in the deepest water prior to complete anoxia.) The reduction in Fe allowed the Fe-bound P to also be released into the water column; TP concentrations increased rapidly in waters with reduced Fe. In hypolimnetic waters with higher D.O. concentrations, P did not build up.

In most years, hypolimnetic P concentrations near the bottom sediments in late summer were generally between 0.5 and 1.0 mg/L (Fig. 4). However, in 1988 and 1992, hypolimnetic TP concentrations were lower because D.O. concentrations were not completely depleted near the lake sediments. Fluctuations in hypolimnetic D.O. in 1988 could not be conclusively explained from the available data. Oxygen may have been added from deep-water algal production (photosynthesis) or from wind-induced mixing, although hypolimnetic water temperatures were relatively stable all summer.

Fall turnover generally occurred around early October as evidenced from the large increase in bottom D.O. concentrations (Fig. 4). Thermocline erosion prior to turnover caused epilimnetic P concentrations to increase, resulting in increased algal biomass (chlorophyll-*a*) and decreased water clarity. However, in 1993, turnover did not occur until near the end of October, which allowed hypolimnetic P concentrations to build up to over 1.0 mg/L. As a result, epilimnetic P concentrations did not increase until late fall. This delay may have prevented an early fall blue-green algal bloom as water clarity declines in 1993 were not as dramatic. Reasons for the delayed P enrichment of the lake's surface waters may have been because of unique weather conditions that year. An extremely late spring ice break-up occurred that caused the lake to stratify with unusually cold bottom water temperatures. Also, extraordinary amounts of spring and summer rainfall resulted in record water levels in Devil's Lake (Fig. 2). Both conditions probably caused greater water column stability during late summer thus delaying the period of thermocline erosion.

Sediment P Analyses

Fe-bound P (Fe-P) concentrations were much higher in the upper few centimeters of sediment analyzed from a transect of cores taken in early summer before the overlying hypolimnetic water became anoxic (Fig. 6). Because of sediment focusing, Fe-P concentrations were highest in the cores from the deep-water regions. In addition, Fe-P concentrations tended to be higher on the north end than on the south end of the lake when comparing sediments collected from the same water depth (e.g., core stations 1.5 N versus 1.5 S).

A substantial decrease in Fe-P concentrations in the top few centimeters of sediment from deep-water cores (0.5 N and 1.5 N) occurred after a prolonged period of hypolimnetic anoxia (Fig. 7). The more refractory portion of the P in the sediments, as estimated as the difference between TP and Fe-P concentrations, exhibited a smaller drop in concentrations after anoxia. The sediment P concentration decline that resulted from hypolimnetic anoxia represented a substantial loss (release) of P to the overlying water. Combining the P concentration data with sediment mass bulk density data that were obtained from a different deep-water core, we calculated the P mass decline in the top 0-2 cm of sediment to be 40-42% for Fe-P and 33-39% for TP (Table 1).

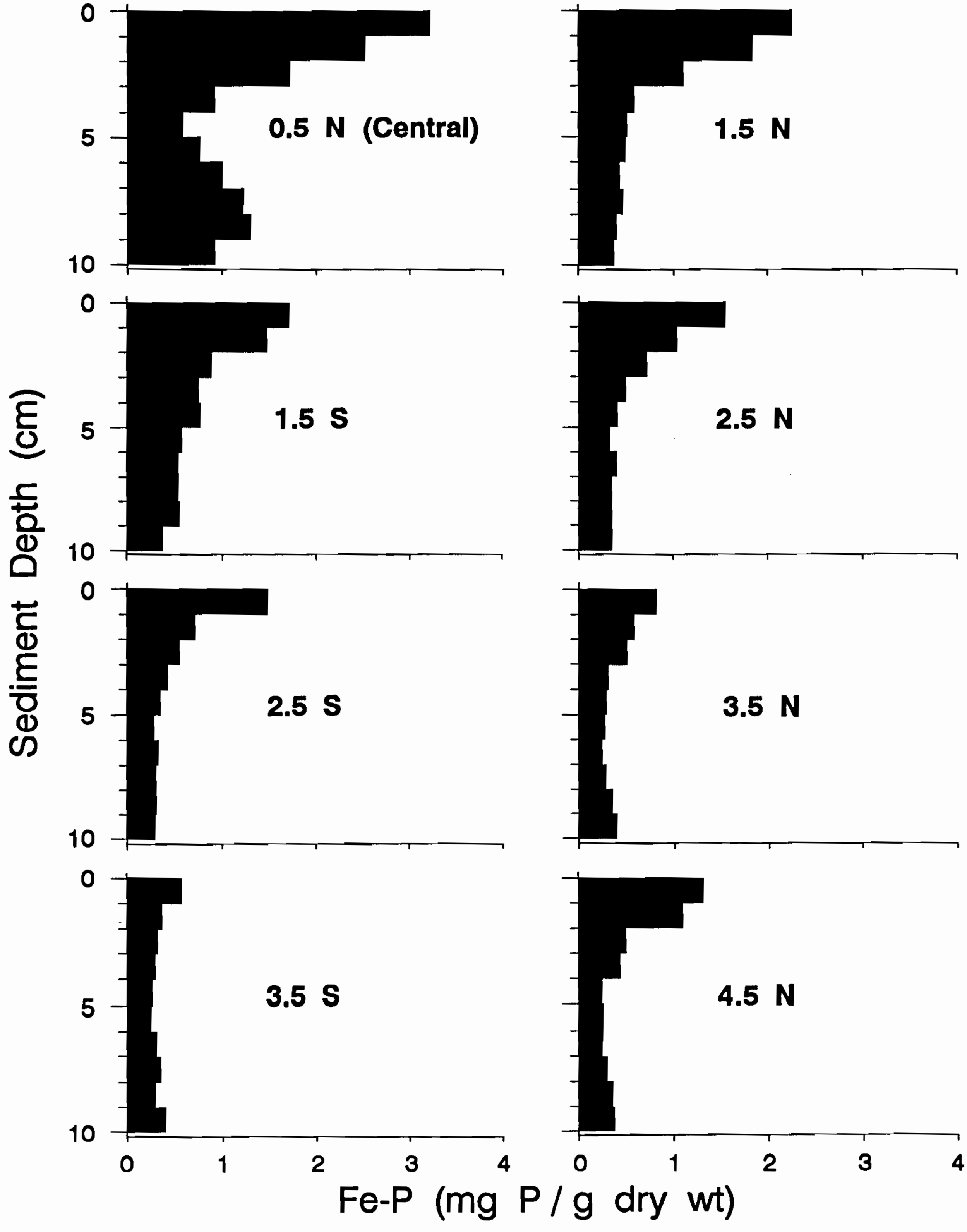

Figure 6. Fe-P concentrations from sediment cores taken along a north-south transect on 12 June 1991 in Devil's Lake. Core locations given in Fig. 3.

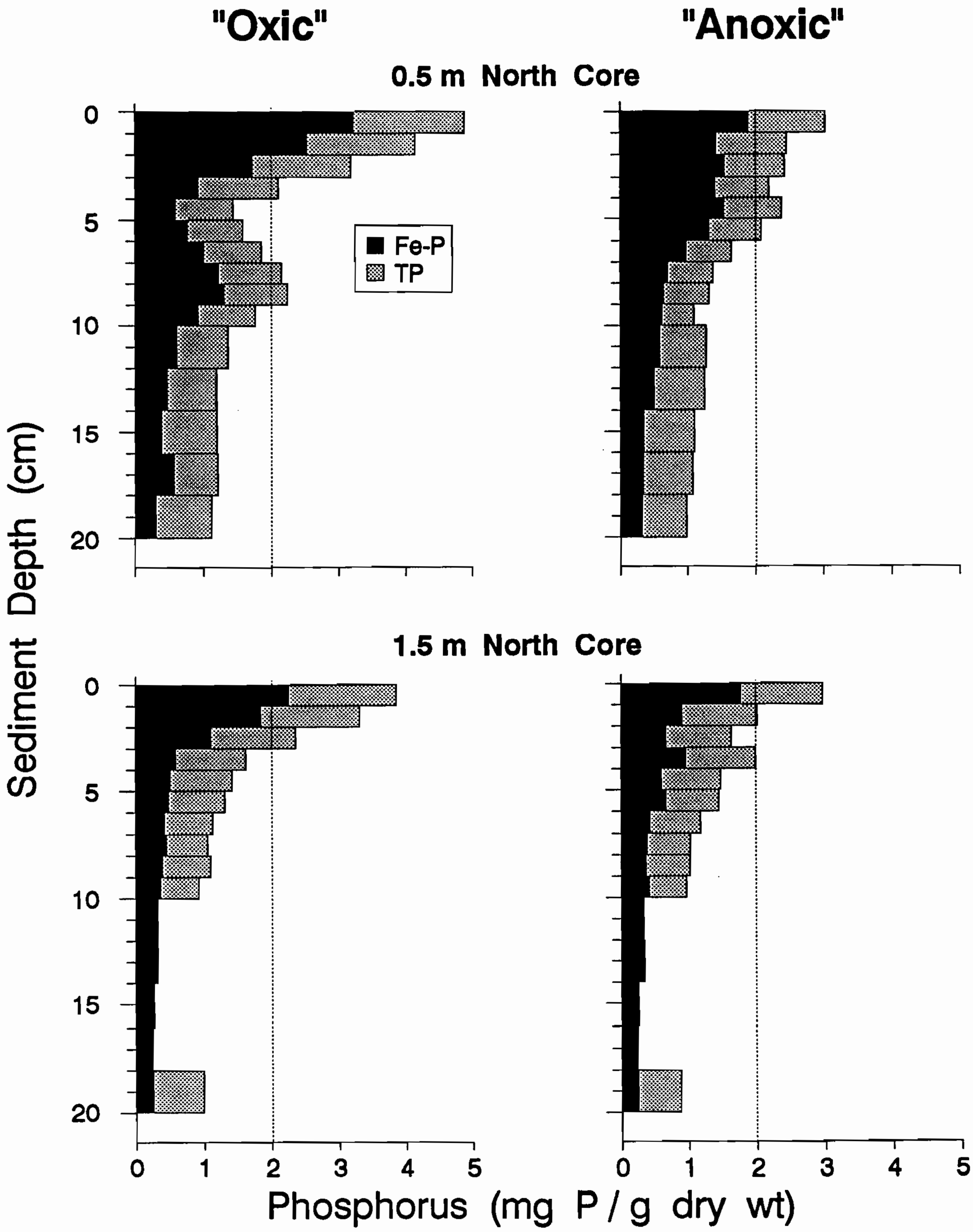

Figure 7. Fe-P and TP concentrations in sediment cores obtained when the hypolimnion was oxic (12 June 1991) and anoxic (26 September 1991) in Devil's Lake.

Table 1. Fe-P and TP masses in the top 2 cm of sediments from deep-water cores taken before and after a prolonged period of hypolimnetic anoxia[1].

	Fe-P			TP		
Core Location	Oxic (g/m^2)	Anoxic (g/m^2)		Oxic (g/m^2)	Anoxic (g/m^2)	
0.5 m N	4.03	2.33	(-42.2%)	6.42	3.89	(-39.4%)
1.5 m N	2.91	1.74	(-40.2%)	5.12	3.41	(-33.4%)

[1] Sediment cores for P analyses taken at deep-water locations (see Fig. 3) on 12 June 1991 (oxic) and 26 September 1991 (anoxic). Sediment mass bulk density data used to calculate P masses were from a core taken at the deepest location on 3 October 1993.

Column Experiment

P concentrations increased at a relatively constant rate in the columns during each 8-week period between flushings (e.g., column A data depicted in Fig. 8). However, overall sediment P release rates in the four columns declined steadily as the number of flushings increased (Fig. 9). At the end of the 10th flushing, release rates were 57% less than initial rates for the two intact sediment cores. In the two cores with the top 5 cm of sediments removed, initial and final rates were 9% and 30% lower than respective rates in the intact cores.

Fe-P concentrations in the column sediments at the end of the 80-week experiment indicated that a major decline in concentrations had occurred in the sediments nearest the sediment water interface (Fig. 10). Concentrations in deeper sediments did not exhibit major declines.

We were not able to calculate the exact mass of P released from the sediments in each column because sediment mass bulk density data were not obtained on all 4 column sediments and on the core collected for initial P concentrations. However, using bulk density data from another deep-water core, a rough estimate of the sediment TP loss in columns A and B (intact cores) was made. The decline in TP mass for the top 0-2 cm of sediment in both columns represented approximately 45-60% of the TP mass released into the water at the end of all 10 flushings. The corresponding Fe-P mass decline in the column sediments represented 36-44% of the P mass released into the water.

The decline in sediment P release rates observed in the column experiment may have been even more substantial had the sediment cores been obtained earlier in the summer (before July 17, 1991). Hypolimnetic P concentrations near the bottom sediments, while still relatively low compared to concentrations observed near the end of summer, were still higher than what was observed in mid-June (Fig. 4). Thus, some release of Fe-P from the surface sediments into the overlying water may have already occurred.

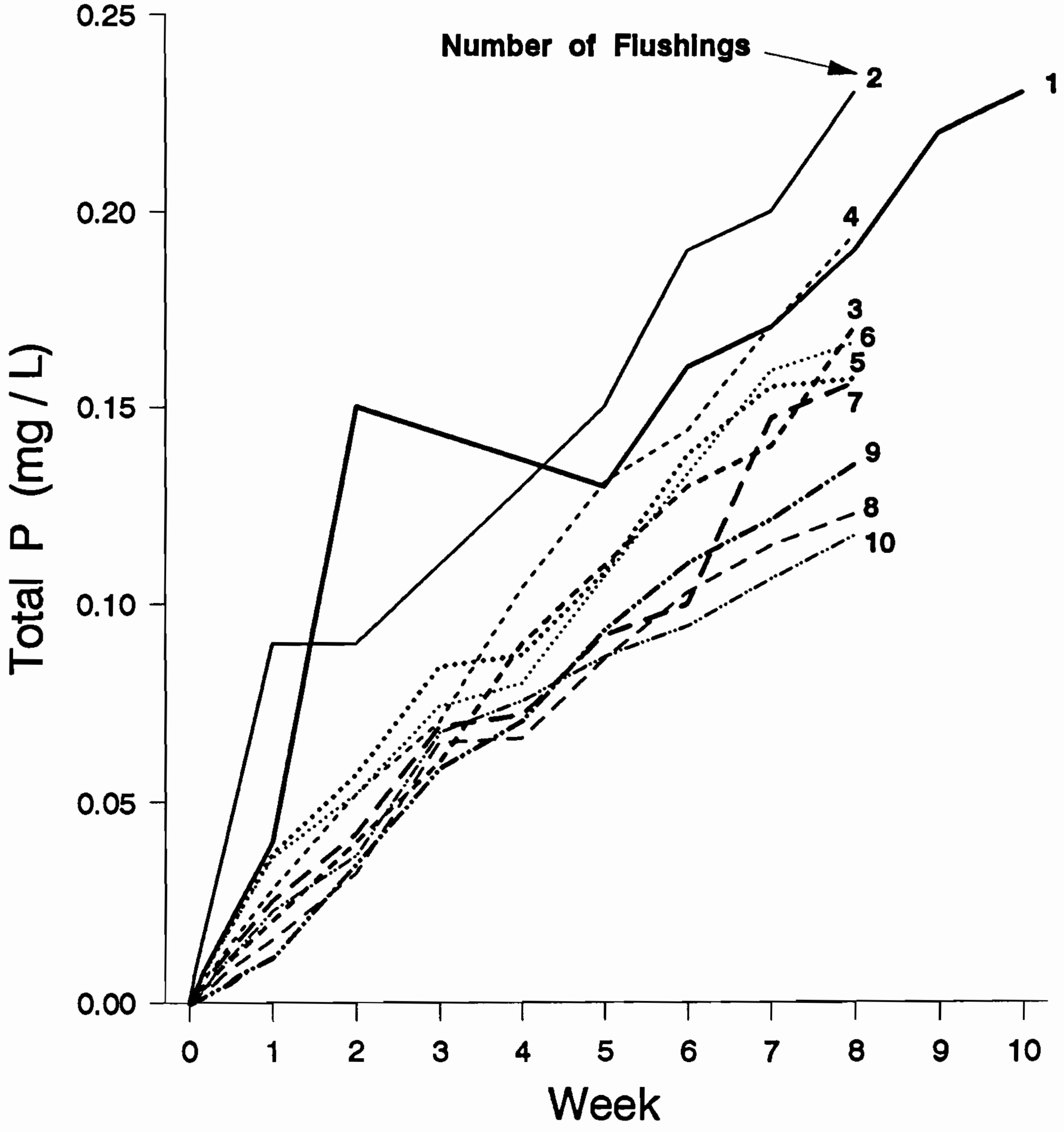

Figure 8. Weekly TP concentrations measured at 20 cm above the sediment-water interface in column A. At the end of 8 weeks, the column water was emptied and replaced with anoxic low-P Devil's Lake water. The number of flushings refers to number of times that the column was emptied at the end of each 8-week run (10 weeks for the first flushing).

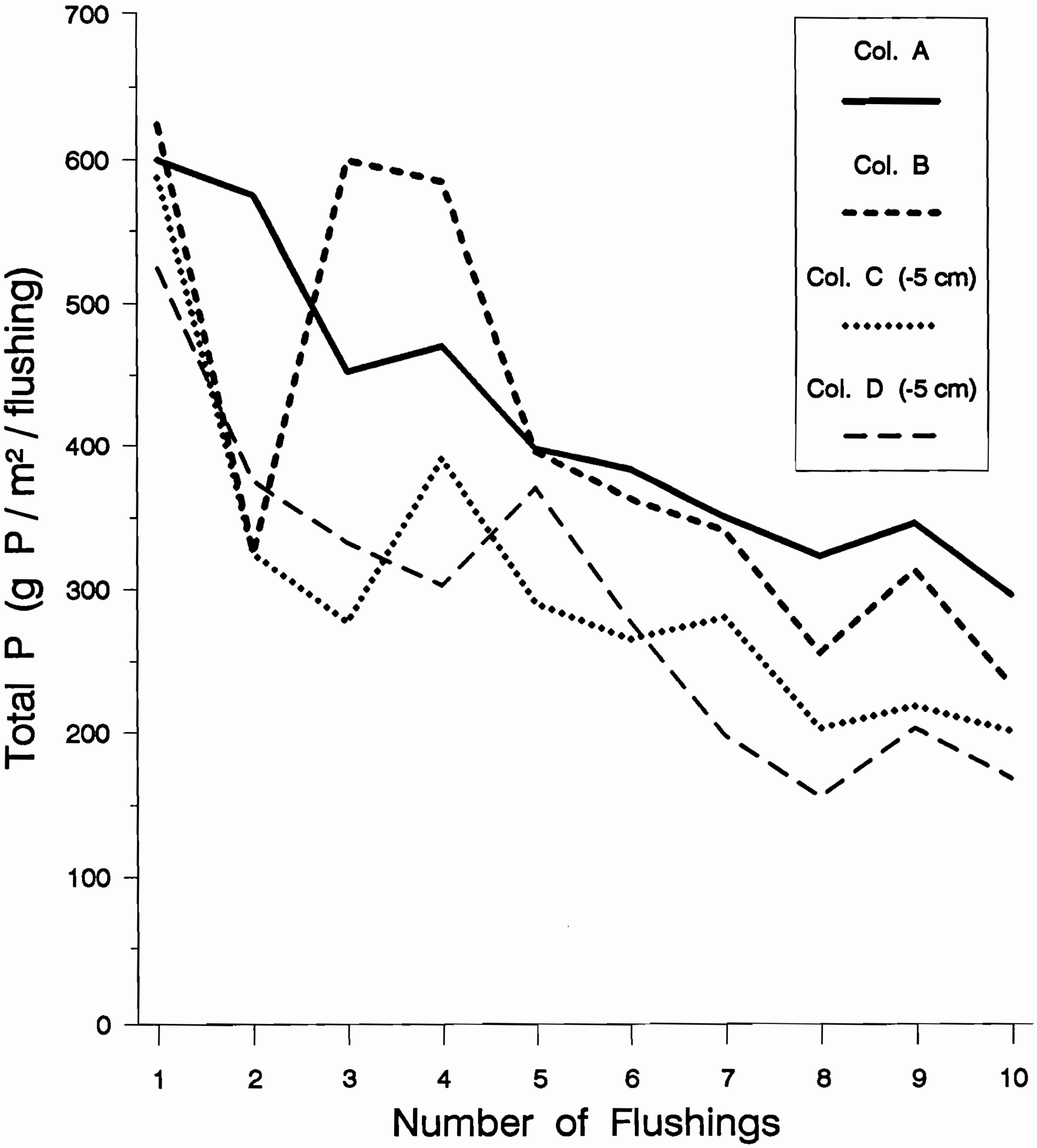

Figure 9. TP release rates from Devil's Lake sediments in the 4 columns at the end of each 8-week period when the columns were flushed (10 weeks for the first run). Columns A and B were intact cores, while columns C and D had the top 5 cm of sediment removed before the experiment began. The number of flushings refers to the number of times the columns were emptied.

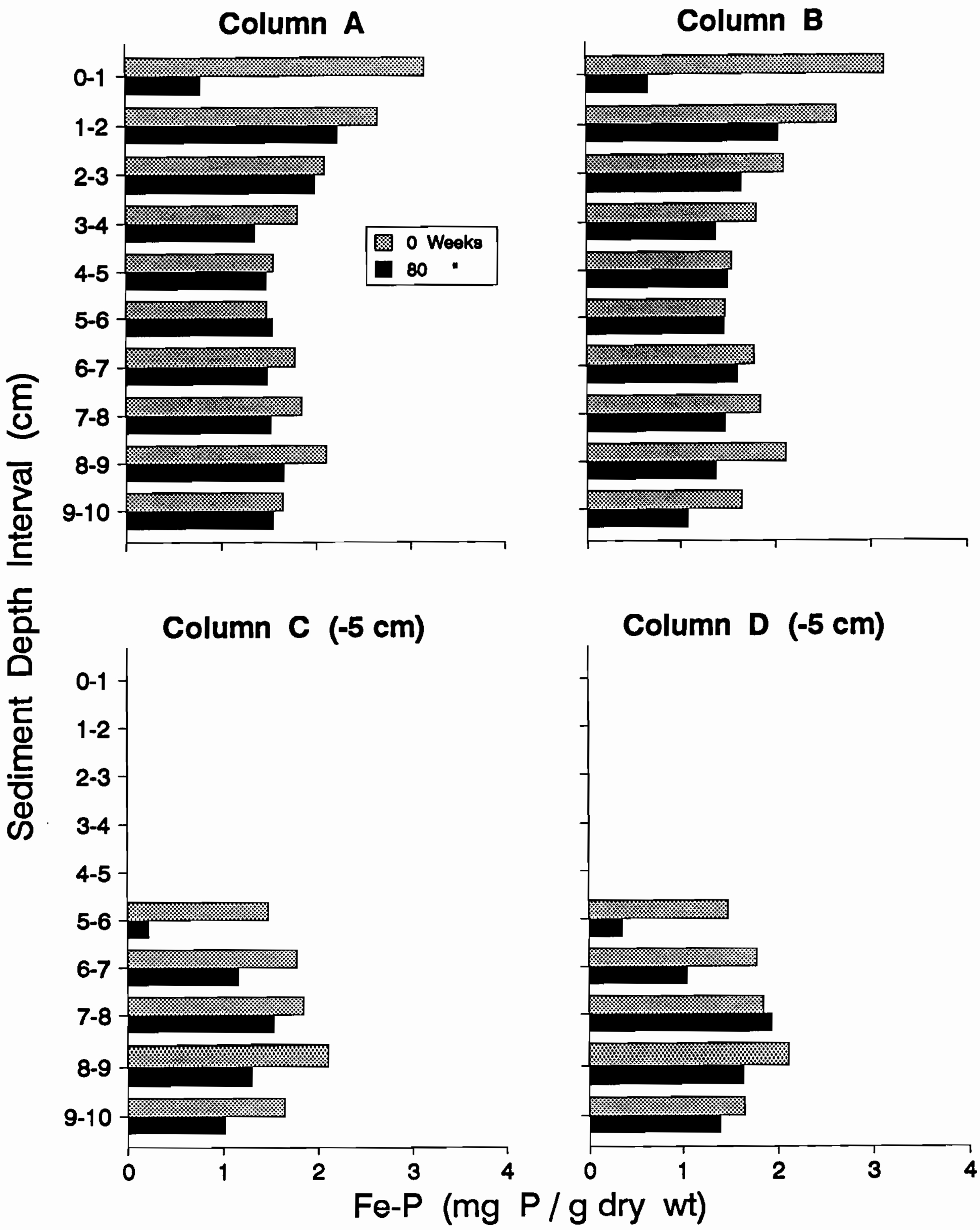

Figure 10. Sediment Fe-P concentrations in the four columns at the end of the 10-flushing experiment (solid bars) and in a core taken at the same time the cores were obtained for the column experiment on 17 July 1991.

Discussion

The water clarity and lake chemistry data collected during 1986-93 confirm that Devil's Lake continues to experience eutrophic (fertile) conditions during late summer and early fall. Each summer as the lake's hypolimnetic water becomes anoxic and sediment Fe is reduced, P is released from the bottom sediments and builds up in the hypolimnion until the water is mixed with surface waters during destratification prior to fall turnover. This increase in surface P concentrations produces an increase in algal biomass that results in a major decrease in water clarity. A dramatic change in the lake's trophic state thus occurs. TP and chlorophyll-*a* concentrations and Secchi disk readings during June and July are symptomatic of oligotrophic (infertile) lakes, whereas conditions in September and early October are symptomatic of meso- to eutrophic lakes.

The sediment analyses demonstrate the enormous potential for release of sediment P into the water column of Devil's Lake during the stratified anoxic period. TP concentrations in the upper few centimeters of sediment from the lake's deepest region are higher than sediment P concentrations measured at the same location in many other Wisconsin lakes (Lathrop et al. 1989). However, the most important indicators of the P release potential in Devil's Lake are the high Fe-P concentrations found in the surface sediment layers throughout the entire profundal zone and particularly in the deeper regions. Because Fe-P is a measure of the most labile (exchangeable) fraction of sediment P forms, high Fe-P concentrations in the sediments near the sediment-water interface are very likely to be released into the overlying water during hypolimnetic anoxia. The loss of Fe-P that we measured in deep-water sediment cores taken at the end of an extended period of anoxia when compared to cores taken before anoxia confirms the role of these surface sediment layers in the lake's P dynamics. In addition, the decline in Fe-P concentrations from the top few centimeters of sediments used in the column leaching experiments also indicates that these layers are very active in releasing P to the overlying water.

Results from the sediment column leaching experiment indicate that sediment P release rates can be significantly reduced by multi-year withdrawals of hypolimnetic water from Devil's Lake. This experiment supports Nürnberg's (1987) finding that hypolimnetic withdrawal can be an effective lake rehabilitation technique for reducing internal P recycling rates in lakes.

Reducing the rate of P release through rehabilitation could help break the annual cycle of P release from the bottom sediments and thereby reduce the problem of algal blooms in Devil's Lake. If surface water algal blooms can be reduced, then less organic matter will reach the bottom sediments where bacterial decomposition occurs. With less organic matter, bacterial respiration rates should therefore decline and hypolimnetic D.O. depletion rates should also decrease. With a shortened period of hypolimnetic anoxia, less build up of hypolimnetic P should occur. While this could decrease the efficiency of P removal by hypolimnetic withdrawal in future years, blue-green algal blooms should be less severe during late-summer and water clarity should improve when compared to current conditions.

In summary, Devil's Lake should realize improved water clarity from a lake rehabilitation program that would use hypolimnetic withdrawal coupled with either water treatment or with spring runoff water replacement. Research that will be conducted during

1995-96 should help answer questions about the cost effectiveness of both techniques, such that lake rehabilitation efforts can proceed to the final planning stages.

Acknowledgments

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