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Phosphorus Budget and Management Strategies for an Urban Wisconsin Lake

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ABSTRACT

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Multiple external and internal phosphorus (P) sources to an urban lake, Half Moon Lake in Wisconsin, were examined during the summer of 1999 in order to develop management strategies for effective P control and reversal of eutrophication (Trophic State Index=74). Internal recycling of P accounted for 80% of the summer P budget of the lake. Flux of P from the sediment accounted for most of the internal P loading (42% of total budget). However, decomposition of *Potamogeton crispus* and recycling of macrophyte P during the middle of the summer growing season, and P resuspension due to motor boat activity, accounted for 20% and 17% of the P budget, respectively, representing additional important sources to be controlled. In contrast, summer P loading via the watershed (storm sewers and precipitation) was much less. Using a water quality model (*Bathtub*), we found that reduction of internal P sources could substantially reduce by greater than 70% the high concentrations of algae in the lake (mean summer chlorophyll = 82 $mg \cdot m^{-3}$). Suggested internal P control measures included a sediment chemical treatment to bind P, greater harvesting of *P. crispus* to reduce the macrophyte P pool at the time of senescence, and limiting motor boat activity when the lake is weakly stratified.

Key Words: lake restoration, macrophyte senescence, motor boat activity, phosphorus budget, sediment phosphorus flux, urban lake.

Half Moon Lake is a hypereutrophic (mean summer chlorophyll = 82 $mg \cdot m^{-3}$ and Trophic State Index for chlorophyll = 74; Carlson 1977) isolated oxbow of the Chippewa River located in the heart of a small (pop. 50,000) urban city, Eau Claire, Wisconsin (Fig. 1). It has served as an important recreational oasis for swimming, fishing, and boating and is immediately adjacent to a city park that houses ball fields, picnicking, hiking trails, play equipment, and historical museums. Although motor boating is not permitted on the lake by the general public, a local water ski club has a permit to practice and present ski shows to the public five nights a week during the summer. In recent decades, the lake has exhibited dense, canopy-forming macrophyte growth (*Potamogeton crispus*; Borman 1990), covering nearly the entire lake during early summer, and severe algal bloom problems (*Ceratium hirundinella*), necessitating the closure of the swimming beach and initiation of aquatic plant harvesting to cut boat lanes (Brakke 1995; Konkel and Borman 1996).

Concerns over deteriorating water and recreational quality of the lake led to an investigation of its phosphorus (P) budget during the summer of 1999 in order to develop a management plan for P control. Our objectives were to quantify fluxes of P to the lake from external watershed sources and from internal

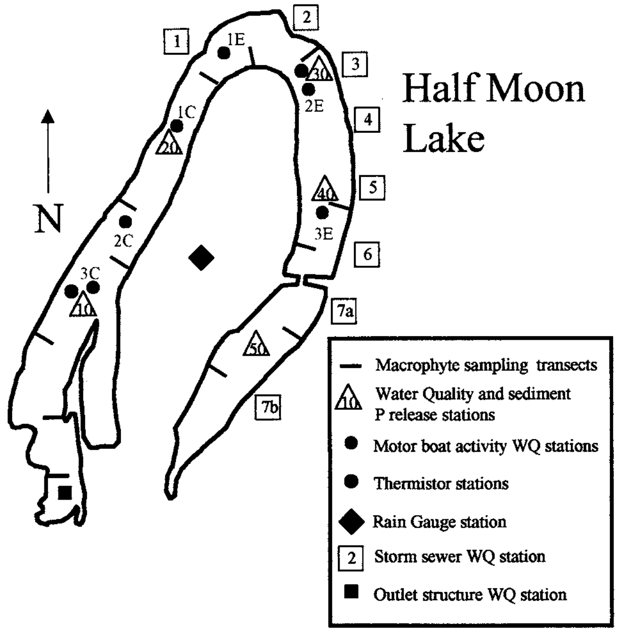

Figure 1.–Water sampling stations, sediment sampling stations, transects for macrophyte sampling, and water sampling stations for monitoring motor boat activity on Half Moon Lake.

recycling pathways that were identified as potentially important to the P economy of the lake. Internal recycling pathways included sediment P flux as a function of eH and pH, macrophyte (primarily *P. crispus*) P uptake and senescence (decomposition and leaching of sediment-derived P), and P resuspension of bottom sediment via motor boat disturbances during water skiing events. Many of these P fluxes are not commonly quantified and incorporated into P budgets. Based on our results, we offer some novel P management strategies for implementation which are applicable to shallow eutrophic lakes.

Methods

Study Site

Half Moon Lake is small (0.5 km^2 and 8.9 x 10^5 m^3), polymictic, and has a mean and maximum depth of 1.6 m and 4 m, respectively. Over fifty percent of the total watershed area (2.3 km^2) is drained directly by storm sewers that enter the lake at several locations (Fig. 1). Land use in the watershed is primarily single and multiple residential; commercial and industrial (15%), park land, and a cemetery occupy a much smaller portion (Barr Engineering 1992). Other water sources to the lake include water that is pumped through storm sewer 7b from shallow wells located at the Owen Park facility, situated southeast of the lake immediately adjacent to the Chippewa River, and direct runoff from three sub-watersheds that are not drained by the storm sewer system. Because there is net groundwater discharge from the lake, water subsidies from the Owen Park facility are required to maintain pool levels in the summer. The outlet structure is located in the southwest portion of the lake. It consists of an uncontrolled surface structure that drains water from the lake when pool elevation exceeds 234.5 m mean sea level.

Hydrology

Flows (cubic meters per second; cms) were measured (15-min intervals or less) at storm sewers 2, 5, and 7b (Fig. 1) between April and September, 1999, using recording stage height-velocity sensors (ISCO Model 750 and 4150). Flows for storm sewers 3, 4, and 7a were estimated using FLOWLINK software (ISCO, Inc) via the Manning equation using continuous (15 minute intervals) stage height measurements (ISCO model 4120) and slope and diameter characteristics of the sewers. To estimate flows contributed via the Owen Park pumps, a stage height recorder was deployed in a street manhole located upstream of influences from storm water inflow. A rating curve for pumping was generated by collecting flow information in the manhole at different pumping rates. Discharge from the outlet of Half Moon Lake was estimated using continuous measurements of pool elevation and a discharge rating curve based on the dimensions of the outlet structure that was provided by Barr Engineering (Hal Runke, Barr Engineering, St. Paul, MN, pers. comm.). Precipitation was measured at 5-minute intervals using a recording rain gauge (ISCO Model 674) deployed near the athletic field in Carson Park (Fig. 1).

A hydrological budget was developed using the following equation: Change in Lake Volume = (Storm Sewer Inflows + Owen Park Pumping + Direct Precipitation) - (Direct Discharge + Net Seepage + Evaporation). The change in lake volume was estimated from changes in pool elevation and estimates of pool volume at those elevations. To estimate hydrological contributions by storm sewers 1 and 6 (Fig. 1) that were not monitored during the study, we assumed that these flows were similar to those generated by storm sewers 5 and 4, respectively, based on similarities in sub-watershed size. Hydrological contributions to the lake by direct precipitation were calculated as the amount of rainfall over the lake's surface area. Evaporation was estimated based on estimates in Barr Engineering (1992). Net seepage was estimated by difference.

External P Loading

Water samples were collected at storm sewers 2, 3, 4, 5, 7a, and 7b at 15-min intervals during periods of rainfall using automated sampling techniques (ISCO Model 3700 and 6700). In the laboratory, samples were weighted with respect to flow and composited to represent a daily sample. Samples were collected in the outflow or at nearby station 10 (see Fig. 1) when water was spilling over the surface spillway. Water samples were collected from the manhole to monitor inputs to the lake via Owen Park pumps (not shown). Water samples collected at various inflows and the outflow were analyzed for total P and soluble reactive P (SRP; see below for analytical methods). Since we did not measure concentrations of total P in the rainwater, literature values (~0.03 mg · L^{-1}; Wetzel 1975) were used. The computer program *Flux* (Walker 1996) was used to estimate P loading from storm sewers.

Internal Loadings

Nine replicate intact sediment cores were collected from the profundal sediments of stations 10, 20, 30, and 40 (depth range = 2.3 to 2.8 m), for determination of rates of SRP release from the sediment (Fig. 1). Sediments were too flocculent for collection at station 50. Sediment cores were collected using a Wildco KB sediment core sampler (Wildco Wildlife Supply Co.) equipped with an acrylic core liner (6.5-cm ID and 50-cm length). Lake water was collected from the epilimnion for incubation with the collected sediment.

Sediment systems, constructed according to the methods of James et al. (1995), were incubated in an environmental chamber at 20 °C for 1-2 weeks. One set of three replicate sediment incubation systems was subjected to an oxic environment while the other set was subjected to an anoxic environment for each station. The oxidation-reduction environment in each system was controlled by gently bubbling either air (oxic) or nitrogen (anoxic) through an air stone placed just above the sediment surface. A third set of sediment incubation systems was subjected to an oxic environment and high pH (~8.5) by bubbling with CO_2-free air. Additional systems containing only filtered lake water were subjected to the same pH and oxidation-reduction treatments as the sediment systems and served as controls. At daily intervals, pH was monitored and water samples were collected from the overlying water of each system, filtered through a 0.45 m membrane filter, and analyzed colorimetrically for SRP using the ascorbic acid method (APHA 1992). Water removed for analysis was replaced with filtered lake water that had been preadjusted to the pH and oxygen conditions of the systems. Rates of P release from the sediment ($mg \cdot m^{-2} \cdot d^{-1}$) were calculated as linear changes in P mass in the overlying water (corrected for dilution effects due to daily replacement of lake water) divided by time and the surface area of the incubation system.

In early June (1-5 June), sampling was conducted in Half Moon Lake to quantify biomass and macrophyte P near the time of *P. crispus* senescence (mid- to late June). In early June, the lake was dominated by a near monospecific stand of *P. crispus* that had reached the surface of the lake and was forming turions. The lake was divided into six equal regions for macrophyte sampling purposes. In each region, one transect with three sampling stations was established perpendicular to the shoreline on each side of the lake for a total of two transects and nine stations per region (Fig. 1). The locations of transects and stations in each region were determined randomly.

Three replicate samples were collected at each station using a quadrat box sampler (0.56 m^2 by 1.5 m high). The box, constructed with aluminum sheet metal sides, was lowered from the boat to enclose an area of the sediment and water column, and a rake was used to carefully pull macrophytes trapped inside. Several rake passes were conducted to ensure that all the macrophytes were removed from within the quadrat sampler. The macrophyte samples were thoroughly rinsed to remove sediment and placed in a mesh bag for fresh weight analysis.

On the shore, bagged macrophyte samples were placed in a washing machine, allowed to spin-dry for a period of five minutes to remove excess water, and weighed to the nearest 1 mg for fresh weight determination. Randomly-chosen samples (about 15% of the samples) were also dried at 70 °C to determine a fresh weight:dry weight conversion factor. Fifteen randomly-chosen macrophyte samples were analyzed for tissue P content according to Allen et al. (1974).

A lakewide estimate of macophyte biomass and P content in the lake near the time of *P. crispus* senescence in June was calculated by weighting estimates with respect to sediment area represented by each transect and region. Since macrophyte harvesting was being conducted by the City of Eau Claire between May and mid-June, we also estimated the amount of macrophyte standing crop and tissue P removed from the lake via this mechanism by obtaining truck weights before and after filling the truck dump box with harvested macrophytes.

To determine *P. crispus* breakdown, plants were removed from different regions of the lake and spun down in a washing machine to remove excess moisture.

A known mass (100 g fresh weight) of macrophyte tissue was placed in replicate mesh bags (2 mm diameter mesh size). The mesh bags containing plant material were air dried for ~3 days before deployment in the lake to initiate senescence. On 14 June, replicate mesh bags were deployed in the lake near mid-water column depth (~1.2 m) at station 10 (Fig. 1). On days 2, 5, 7, 14, 30, 58, and 90, five replicate bags were removed from the lake for analysis of loss of mass and changes in P content. In the laboratory, macrophyte material was carefully washed to remove sediments and other debris, then dried at 70 °C to a constant weight. Subsamples of tissue were analyzed for P (see above methods).

On days 0, 14, and 30 of *P. crispus* breakdown, studies were conducted to determine the rate of P leaching from macrophyte tissue. Subsamples of macrophyte tissue contained in the mesh bags were placed in 1-L beakers containing filtered lakewater and incubated in a darkened environmental chamber at 20° C. At time intervals ranging from several minutes to days, water samples were collected for the determination of SRP concentration. Rates of P leaching were calculated as the change in SRP per gram dry mass of plant material per day.

The local water ski club conducted practices and presented ski shows to the community in the northeast region of the lake (the area delineated by station 1E and 3E) between ~1730 and 2030 on Sunday through Thursday throughout the summer. Several twin-engine inboard (150 HP) and outboard ski and pontoon boats were used during these periods. Maximum depths (in the thalweg) in the skiing area ranged between 2.5 and 2.8 m.

To examine P fluxes into the water column via motor boat disturbance of bottom sediment during ski shows and practices, three sampling stations each were established in control and experimental sites for water sampling purposes. Stations 2C (~2.9 m deep) and 2E (~2.8 m deep) were located in the approximate center of the control and experimental sites, respectively. Station 2E was located near the ski jump where shows and practices occurred. Stations 1E and 3E were located near the northern and southern extent of the skiing area. A similar area was delimited for establishment of control stations 1C and 3C (Fig. 1). In general, sampling was conducted prior to skiing events (~1630-1730 hours) and immediately after the skiing events (2030-2130 hours). Sampling was conducted in the control and experimental sites during 2 ski tournament practices and 4 ski shows. Sampling was also conducted during 3 dates when no ski events occurred to serve as a control. The same time bracket (i.e., 1630 hours to 2130 hours) was used for sampling during days when skiing did not take place.

At the sampling stations, measurements of temperature, dissolved oxygen, pH, and conductivity were collected using an *in situ* water quality monitor (Hydrolab Data Sonde 4, Hydrolab, Corp.) that was precalibrated with Winkler titrations (APHA 1992) and known buffer solutions. Water samples were collected at stations 2C and 2E at 0.25-m depth intervals from the surface to near the bottom (~the 2.0-m depth) using a close-interval syringe sampler as described by James et al. (1992) for total P and SRP analysis.

Recording thermistors (DataLoggers, Inc.; calibrated to the nearest 0.1 °C with a National Bureau of Standards calibrated thermometer) were deployed at 0.5-m intervals between the lake surface and near the bottom at stations 2C and 2E between late June and September to monitor stratification patterns and turbulence. Temperature measurements were collected at 15-minute intervals throughout the study period at each station. The local thermal stability of the lake in the vicinity of stations 2C and 2E was calculated as:

$$S = 1/A \int_0^{z_m} (z - z_g)(p_z - p_g)dz \qquad 1)$$

where A is the surface area (m^2), z_m is the maximum depth (m), z is the depth at stratum *z*, z_g is the depth of the center of mass or p_g, and p_z is the density of water (kg·m^{-3}) at depth *z* (Idso 1973). P_g was calculated as :

$$p_g = 1/V \quad V_z p_z dz \qquad 2)$$

where V is lake volume (m^3) and V_z is the volume at depth z. Stability (g-cm cm^{-2}) represented the amount of work (in the form of wind power, motor boat activity, etc.) required to completely mix a water body that is stratified due to vertical differences in water density. Higher stability values were indicative of strong stratification and greater work required to disrupt stratification. Conversely, lower stability values were indicative of weak stratification and less work required to disrupt stratification.

Limnological Monitoring and Modeling

Five stations (stations 10 through 50) were established along the longitudinal axis of Half Moon Lake for limnological monitoring purposes (Fig. 1). During April through October, water samples were collected biweekly at 1-m intervals from the surface to within 0.5 m from the bottom for analysis of total P, SRP, and viable chlorophyll *a*. For SRP, samples collected from anoxic water in the lake were filtered immediately without exposure to oxygen. In conjunction with the water sampling schedule, measurements of water temperature, dissolved oxygen, pH, and conductivity were also collected (see above). Secchi disk transparency was measured to the nearest 1 cm using a 20 cm diameter, alternating black and white, disk.

Water samples for analysis of SRP were filtered through a 0.45 m filter (Gelman Metricel) prior to colorimetric analysis using automated procedures (Zellweger Analytics, Lachat Div., Milwaukee, WI). Total P was analyzed using automated procedures after digestion with potassium persulfate (Ameel et al. 1993). Samples for chlorophyll *a* were extracted in dimethyl-sulfoxide-acetone (50:50) at < 0 °C for a minimum of 12 hours. Viable chlorophyll *a* was determined fluorometrically (Turner Designs; Model TD-700) according to Welschmeyer (1994).

The water quality model *Bathtub* (Walker 1996) was used as a planning tool to examine the trophic response of Half Moon Lake to reductions in P loading. Since the lake is very small, we used measurements of chlorophyll *a* and total P weighted for the entire lake over the period June through August as average summer conditions. We chose this period because it was the most active recreational period during the summer. The computer program *Profile* (Walker 1996) was used to estimate weighted summer lake concentrations for input into *Bathtub*.

Results and Discussion

Hydrological Conditions

During April through September, 1999, measurable precipitation occurred over the lake on 40 days (Fig. 2). Daily rainfall exceeding 25.4 mm (1 inch) occurred in early April, mid-May, early June, and July. Airport (Eau Claire) rainfall totals during the study period in 1999 were nearly equivalent to the 30 year (1961-1990) average total rainfall of 602 mm over the same period (23.7 inches; National Weather Service). Peaks in mean daily inflow from monitored storm sewers coincided with peaks in precipitation during the study period (Fig. 2). Overall, the Owen Park pumping station represented 54% of the measured inflow to the lake during the study period. Precipitation over the lake surface and storm sewer inflows accounted for 31% and 15% of the measured inflow, respectively.

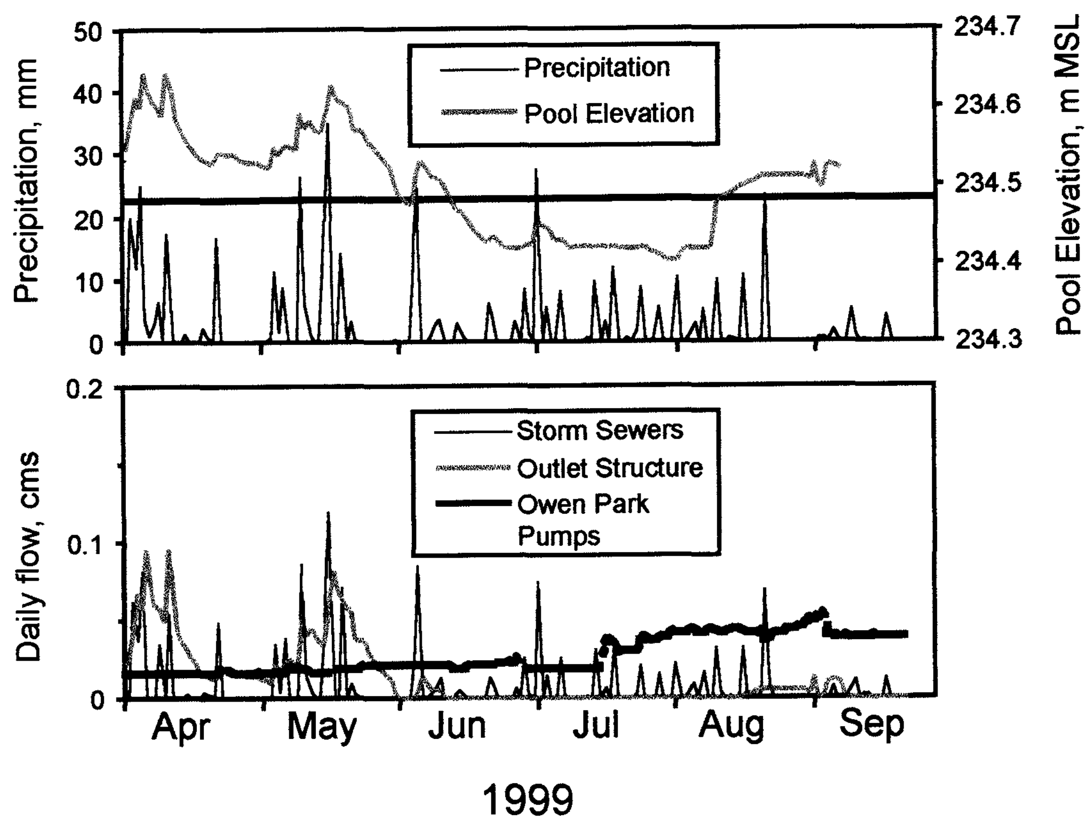

Figure 2.–Variations in daily precipitation and pool elevation (upper panel) and daily flow measured for storm sewers, pumping from the Owen Park facility, and discharges from the outlet structure (lower panel). The Owen Park facility pumps water from shallow wells adjacent to the Chippewa River into storm sewer 7b.

The surface outlet structure of Half Moon Lake accounted for 25% of the water outflow from the system. This pattern was due to extended periods (early June through July) when the pool elevation was below the mouth of the surface outlet structure (Fig. 2). Net groundwater seepage, estimated via hydrologic balance, represented the greatest water outflow from the system (50%). Evaporation, based on estimates provided in Barr Engineering (1992), accounted for 23% of the water outflow. The average residence time of the lake during the study period was 198 days.

Phosphorus Sources

Flow-weighted concentrations of total P and SRP from the Owen Park pumping station were generally low (0.074 and 0.010 mg · L^{-1}, respectively) relative to precipitation-related concentrations observed for the storm sewers (range for total P = 0.149 to 0.344 mg · L^{-1}; range for SRP = 0.038 to 0.065 mg · L^{-1}). Surprisingly, however, external P loading to the lake was not dominated by storm sewer inflows, due primarily to lower water income from this source versus other hydrologic inputs. When normalized with respect to lake surface area and the summer period used in *Bathtub* modeling (June-August), it was 0.3 mg · m^{-2} · d^{-1} (Table 1), versus 0.4 mg · m^{-2} · d^{-1} for the Owen Park pumping station and 0.5 mg · m^{-2} · d^{-1} for direct precipitation to the lake .

Rates of P release from sediments, measured in the laboratory, were substantial under anoxic conditions, ranging between a mean of 2.3 and 11.7 mg · m^{-2} · d^{-1} for the 4 lake stations. Rates of P release from sediments under oxic conditions were also high and varied linearly as a function of pH (Rate = 1.8495pH - 13.488; r^2=0.68; over a pH range of 7.3 to 8.5). At pH values near 7.3, they were less than 0.5 mg · m^{-2} · d^{-1}. At pH values near 8.5, they were substantially greater at ~2 to 3 mg · m^{-2} · d^{-1}.

In Half Moon Lake, the bottom waters at several stations exhibited anoxic conditions during June through August (Fig. 3). During periods of anoxia, the pH of the bottom waters declined below 7.5 (Fig. 3). In contrast, pH approached 10 in late April and was greater than 8 in May through June and mid-August through September. We used fluctuations in pH and oxygen measured throughout the water column at stations 10 through 50 to estimate rates of P release from sediment at different depths in the lake for each station during the summer period (Fig. 3). These rates were weighted with respect to sediment areas represented by each depth and station to calculate an overall summer lakewide rate of P release from sediment. For periods when pH exceeded our experimental value of 8.5, we assumed linearity in our results above this pH value. In general, the estimated lakewide rate of P release from sediment was greatest in July, coincident with the occurrence of bottom water anoxia (Fig. 3). However, it was also high during other summer periods due to high pH levels in the lake. The estimated lakewide rate of P release from sediment for the summer period used in *Bathtub* modeling was 2.5 mg · m^{-2} · d^{-1} (Table 1).

During the period May through mid-June, ~9000 kg standing crop (dry mass) and 30 kg macrophyte P were harvested from Half Moon Lake. The estimated macrophyte standing crop and P mass remaining in Half Moon Lake after final harvesting on 13 June was 25.4 g biomass · m^{-2} (±2.3 S.E., n=108) and 111.6 mg P · m^{-2} (±9.9 S.E.). Lakewide plant P mass available for flux to the water column (assuming complete decomposition) at that time was 60 kg. Visual observations indicated that the majority of the macrophyte standing crop in Half Moon Lake during early June was *P. crispus*.

From mesh bag decomposition experiments in

Table 1.–Lakewide summer (June-August) phosphorus loading rates for various sources to Half Moon Lake. All rates were adjusted with respect to lake surface area and the three month summer period chosen for *Bathtub* modeling.

Loading Variable	Source	mg · m^{-2} . d^{-1}	g · d^{-1}	Percent
External P Load, mg · m^{-2} · d^{-1}	Storm Sewers	0.3	163	5
	Owen Park Pumping Station	0.4	217	7
	Precipitation[1]	0.5	272	9
Internal P Load, mg · m^{-2} · d^{-1}	Sediment	2.5	1359	42
	P. crispus Decomposition	1.2	652	20
	Motor Boat Activity	1.0	544	17
Total		5.9	3207	100

[1] P concentrations were not directly measured in rainwater. Literature values were used to estimate loading from Wetzel (1975).

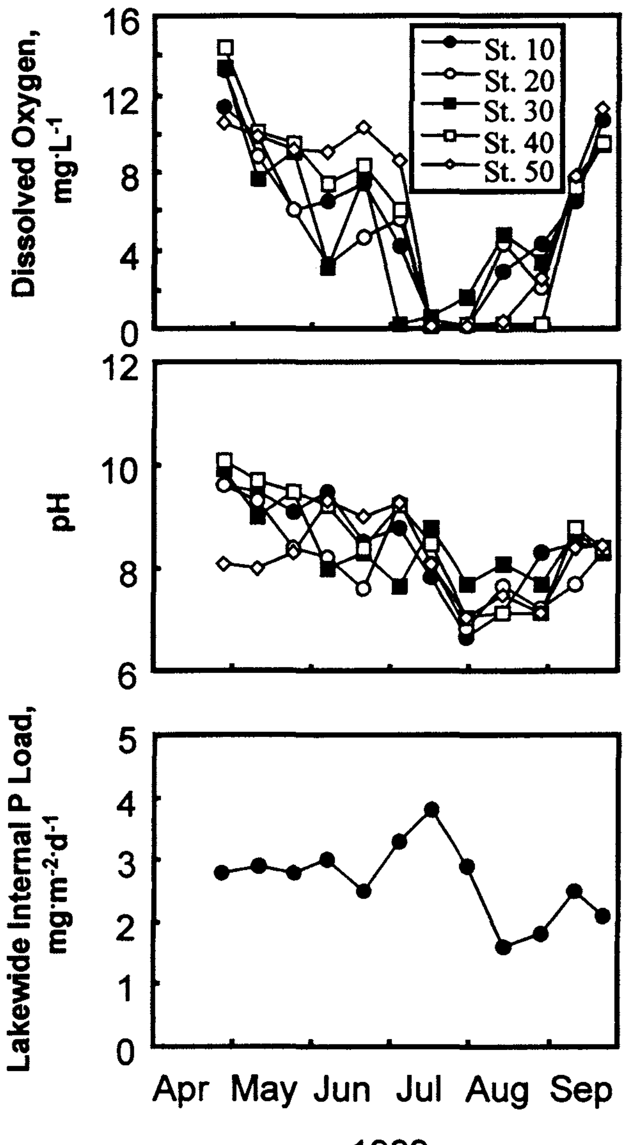

Figure 3.–Seasonal variations in near-bottom dissolved oxygen concentration, near-bottom pH, and the lakewide internal P load from sediments. The lakewide internal P load from sediments was estimated using rates of P release as a function of redox and pH and measured oxygen and pH conditions throughout the water column in the lake. This rate was weighted with respect to area for each station.

Half Moon Lake, we found that the loss of P from *P. crispus* was greatest during the first week of decomposition, with 40% of the P loss occurring during the first 2 days of decomposition (Fig. 4). Leaching of soluble P from plant tissue (i.e., autolysis) into the water was greatest during the first 24 hours of decomposition (Fig. 4). Fragmentation, breakdown, and loss of plant tissue P, which most likely occurred via microbial degradation, accounted for tissue P loss after the initial leaching period (Fig. 4). Within 30 days, nearly all of the P mass was lost from the mesh bags. Combined P breakdown and leaching rates, normalized with respect to lake surface area and the summer period used in *Bathtub* modeling, was 1.2 mg $\cdot$m^{2} $\cdot$d^{1} (Table 1).

Activities on 17 August provided an example of P flux into the water column via motor boat activity during a ski show practice. On that date, thermal stratification was observed at both control and experimental sites between 1600 and 1719 hours shortly before the start of the practice at ~1730 hours (Fig. 5). After the practice (~2000 hours), the water column was completely destratified at station 2E, and bottom water warmed by greater than 1 °C due to mixing and entrainment of warmer surface water to the bottom of the lake. Destratification was less noticeable at stations 1E

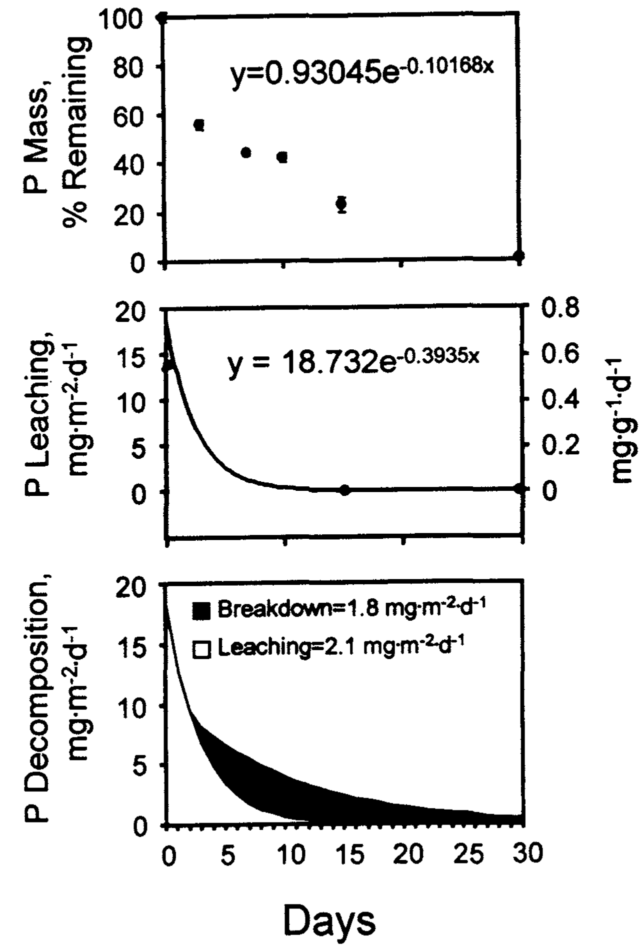

Figure 4.–Percent phosphorus (P) mass remaining in decomposing *Potamogeton crispus* contained in mesh bags at station 10 in Half Moon Lake as a function of time (upper panel); laboratory-based rates of P leaching from decomposing *P. crispus* (middle panel), and area-weighted P decomposition over a thirty day period (lower panel). The latter rate was calculated as a function of estimated *P. crispus* density in the lake near the time of senescence. Leaching is defined as autolysis of cellular contents into the water column while breakdown refers to microbial decomposition and fragmentation.

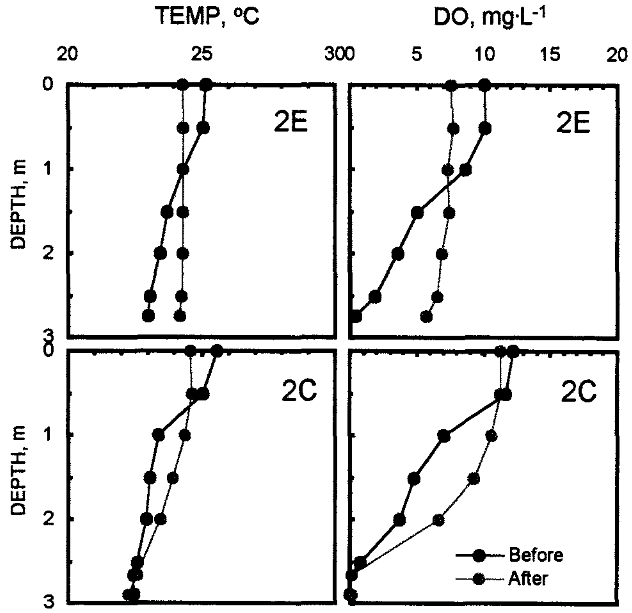

Figure 5.–Vertical variations in water temperature and dissolved oxygen (DO) before and after motor boat activity at stations 2E (experimental station) and 2C (control station) on 17 August.

and 3E, located at the northern and southern extent of the ski area, respectively (not shown). In the control area, the upper water column cooled at the 3 stations (i.e., station 1C, 2C, and 3C) between 2022 to 2106 hours as a result of evening air temperature declines (Fig 5; station 1C and 3C not shown). Unlike station 2E, bottom temperatures at the control stations did not fluctuate between 1700 and 2130 hours.

In the experimental and control areas, anoxic conditions were observed in the bottom waters at all stations prior to the start of the practice (Fig. 5) on 17 August. After the practice, dissolved oxygen in the bottom waters at station 2E increased substantially (i.e., by greater than 5 mg·L^{-1}) as a result of motor boat-induced mixing and entrainment. In contrast, the bottom waters at stations 1E and 3E remained anoxic after the practice, indicating less mixing and reaeration of the bottom waters (not shown). Boating activity was much less in the vicinity of these stations. The bottom waters were anoxic after the practice at all control stations (Fig. 5). However, dissolved oxygen increased between the 1- and 2-m depths at station 2C after the practice. This increase in dissolved oxygen may have been due to convective cooling and circulation in the upper 2-m of the water column at this station.

Prior to the start of practice on 17 August, total P exhibited uniform concentrations in the upper 1 m of the water column and lower concentrations in the hypolimnion at station 2E, while they were nearly uniform throughout the water column at station 2C (Fig. 6). Total P ranged between 0.10 and 0.15 mg·L^{-1} at both the control and experimental station before the practice. After the practice, total P concentrations increased at station 2E, primarily in the lower 1 m of the water column (Fig. 6). Concentrations at station 2C remained nearly constant before and after the practice. At station 2E, the mean (over entire water column) total P concentration of 0.140 mg·L^{-1} (±0.002 S.E.) observed after the practice was significantly different than the mean concentration of 0.123 mg·L^{-1} (±0.015 S.E.) observed before the start of the practice ($p < 0.05$; T-Test; SAS 1994). There were no significant differences in mean total P concentrations measured before (0.122 mg·L^{-1} ± 0.004 S.E) and after (0.115 mg·L^{-1} ± 0.002 S.E.) the practice at station 2C on this date. Mean concentrations were also significantly different between station 2E and 2C after the practice ($p<0.05$; T-Test; SAS 1994) versus before practice, when mean concentrations were not significantly different from each other.

An example of a period during which motor boat activity caused no apparent P resuspension and P loading to the water column occurred on 15 July. The ski club also conducted a practice on that particular date between ~1730 and 2030 hours. Before the start of the practice, a strong metalimnion was observed between the 1.5- and 2.5-m depths at stations 2C and 2E, with marked declines in dissolved oxygen in the hypolimnion and near anoxic (less than 5 mg·L^{-1}) conditions in the bottoms waters immediately above the sediment interface (Fig. 7). Shortly after the end of

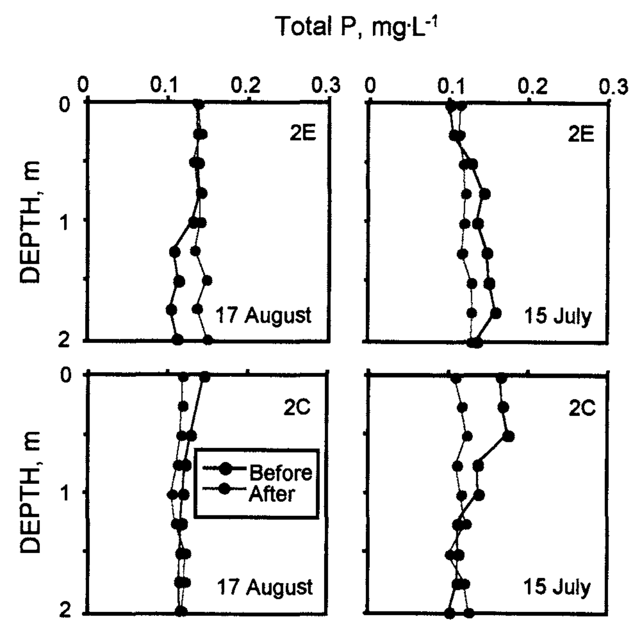

Figure 6.–Vertical variations in total phosphorus (P) before and after motor boat activity at stations 2E (experimental station) and 2C (control station) on 17 August and 15 July.

the practice, nearly complete water column destratification was observed at stations 2E. However, anoxia persisted near the sediment interface after the practice at this station, indicating that motor boat activity was not sufficient to completely mix the water column down to the sediment interface. Total P did not increase in the bottom waters at station 2E as a result of motor boat activity on 15 July (Fig. 6), indicating that P resuspension was minor. The water column remained thermally and chemically stratified at all control sites after the practice (Fig. 7) and there was a slight increase in total P in the bottom waters of that station (Fig. 6). However, there were no significant differences in mean total P over the entire water column before and after practice for either the control or experimental station. During periods when no motor boat activity occurred (i.e., control periods), control and experimental stations exhibited similar thermal and chemical stratification patterns (not shown). No resuspension of P or significant differences in mean water column P as a function of station or time were observed on these dates.

In general, significant increases in mean water column total P concentration were observed at station 2E in conjunction with motor boat activity on 2 of the 9 sampling dates; 17 August and 1 September. On 4 dates when motor boat activity occurred (i.e., 24 June, 7 July, 12 July, and 15 July), no significant changes in mean total P occurred as a result of mixing and P resuspension at station 2E. Mean total P concentrations did not change significantly in either control or experimental stations on 3 dates (i.e., 23 July, 28 August,

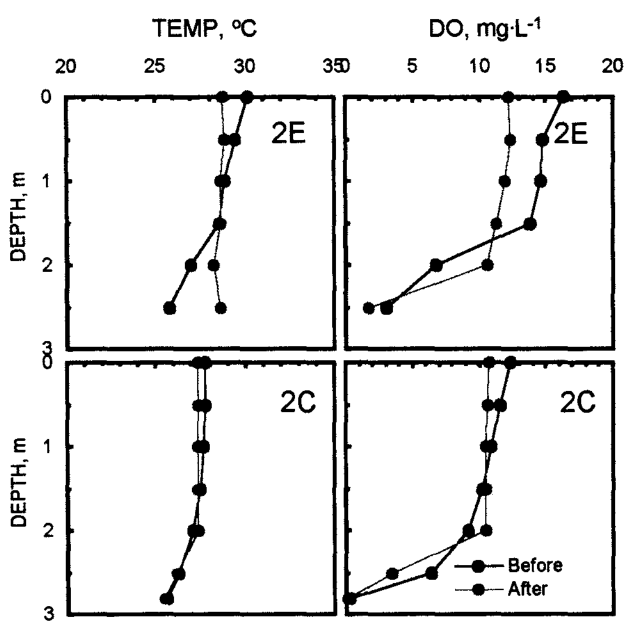

Figure 7.–Vertical variations in water temperature and dissolved oxygen (DO) before and after motor boat activity at stations 2E (experimental station) and 2C (control station) on 15 July.

and 16 September) when no motor boat activity occurred. On all dates and time periods, concentrations of SRP in the water column were very low (less than 10 g · L^{-1}) at the control and experimental station, suggesting that most of the total P was in particulate form.

We used the standard deviation (S.D.) of the mean bottom temperature, determined over the time period 1800-2000 hours (i.e., general hours of greatest ski boat operation), as a surrogate measurement of the relative extent of mixing of the bottom water and P resuspension via motor boat activity for summer dates when P profiling was not conducted. A greater S.D. indicated greater temperature fluctuation around the mean bottom temperature and, thus, greater mixing and entrainment of warmer water from above. A smaller S.D. indicated less temperature fluctuation around the mean bottom temperature and less mixing and entrainment of warmer water from above. For instance, on 17 August, when total P concentrations increased significantly in the water column in response to motor boat activity, bottom temperatures at the experimental station 2E fluctuated and warmed dramatically between 1800-2000 hours due to mixing and entrainment by motor boats (Fig. 8), resulting in a high bottom temperature S.D. of 0.38. Bottom temperatures at the control station 2C, where no motor boat activity occurred, did not fluctuate during that time period on 17 August (Fig. 8) and the bottom temperature S.D. was only 0.03. A similar pattern of high bottom temperature fluctuation (S.D. = 0.27) and P resuspension occurred at the experimental station 2E on 1 September in conjunction with motor boat activity (not shown). In contrast, the bottom temperature S.D. was generally much lower when no motor boat activity occurred, as on 28 August (i.e., S.D. = 0.03 at station 2E; Fig. 8), and during periods when mixing and entrainment via motor boats were not great enough to resuspend P, as on 15 July (i.e., S.D. = 0.22 at station 2E; Fig. 8).

Based on observed resuspension of P on 17 August and 1 September when bottom temperature S.D. exceeded 0.27 (i.e., critical S.D. threshold), we estimated that P resuspension due to motor boat activity occurred at the experimental site ~21% (i.e., on 14 dates during the evening hours) of the time in late June through early September (Fig. 9). In contrast, bottom temperature S.D. exceeded the critical S.D. on only 3 dates at the control station during the study period (Fig. 9). Our percentage represents a conservative estimate of periods of P resuspension because we could not account for possible resuspension on dates exhibiting isothermal conditions during ski shows or practices. On these particular dates (~5% of the time), bottom temperature would not be expected to fluctuate and

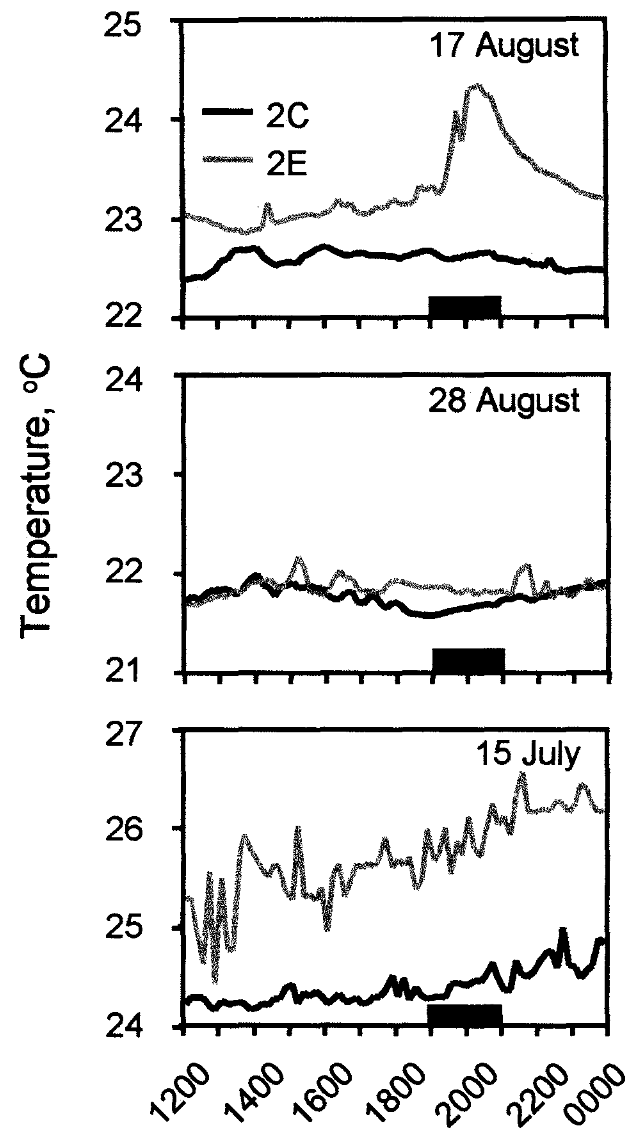

Figure 8.–Variations in bottom water temperature at station 2C (control station) and 2E (experimental station) during a period of motor boat activity on 17 August (upper panel), a period when no motor boat activity occurred on 28 August (middle panel), and a period of motor boat activity on 15 July. The black horizontal bar represents the time period of motor activity (with the exception of 28 August) and temperature averaging period used for estimation of the standard deviation (see text).

thus, the bottom temperature S.D. would be minor even though P resuspension potential was high.

A relationship between the stability of the water column at station 2E one hour before motor boat activity and the bottom temperature S.D. between 1800-2000 hours (i.e., ~duration of motor boat activity) is shown for ski shows and practices in Fig. 10. In general, when stability was greater than 30 g-cm $\cdot$ cm^{-2}, bottom temperature S.D. was usually near or below the critical S.D. of 0.27 during motor boat activity, suggesting no P resuspension. The bottom temperature S.D. increased above the critical threshold on 10 occasions in conjunction with declines in stability below 30 g-cm $\cdot$ cm^{-2}. However, on other occasions when stability was below 30 g-cm $\cdot$ cm^{-2}, bottom temperature S.D. was also below the critical S.D. during both ski shows and practices. As stability approached zero, bottom temperature S.D. also approached zero due to isothermal conditions. These results suggested that the potential for P resuspension due to motor boat activity was greatest when the stability of the water column was below 30 g-cm $\cdot$ cm^{-2}.

The average change in mean total P in the water column at station 2E for the dates 17 August and 1 September (0.021 mg $\cdot$ L^{-1}) was weighted with respect to a water volume that encompassed about two-thirds of the area between stations 1E and 3E to estimate a loading mass of ~2.8 kg P $\cdot$ d^{-1} (i.e, 0.021 mg P $\cdot$ L^{-1} x 13.4 x 10^7 L of water in ski area) as a result of motor boat activity on days when P resuspension occurred from the bottom during the evening hours (i.e., ~21% of the days). The overall estimated P loading rate via motor boat activity for the summer period used in *Bathtub* modeling was ~1.0 mg $\cdot$ m^{-2} $\cdot$ d^{-1} (i.e., 2.8 kg P $\cdot$ d^{-1} ÷ 534590 m^2 lake area x 0.21; Table 1).

Limnological Conditions, Water Quality Modeling, and P Management Scenarios

Seasonal patterns in lakewide-average concentrations of P and chlorophyll *a* are shown in Fig. 11. Total P was constant near 0.05 mg $\cdot$ L^{-1} between late April and early June and increased to greater than 0.10 mg $\cdot$ L^{-1} in mid-July. Total P remained greater than 0.10 mg $\cdot$ L^{-1} between mid-July and August and declined to less than 0.10 mg $\cdot$ L^{-1} in September. SRP was very low (less than 0.010 mg $\cdot$ L^{-1}) throughout the study (Fig. 11). Chlorophyll *a* increased to a lakewide-average concentration of 155 g $\cdot$ L^{-1} in July, coincident with peaks in total P and estimated internal P loading from the sediments (Fig. 3). Chlorophyll *a* declined to ~70 g $\cdot$ L^{-1} between August and September (Fig. 11). Although algal composition was not determined in this study, Brakke (1995) reported that the algal assemblage of Half Moon Lake was dominated by *Ceratium hirundinella* in the summer. This dinoflagellate has been shown to migrate vertically in the water column on a diel basis (Heaney and Talling 1980), providing it with the capability of directly accessing stores of P in the bottom waters of Half Moon Lake. Mean summer concentrations used for *Bathtub* modeling were 0.110 mg $\cdot$ L^{-1} for total P, 82.0 mg $\cdot$ m^{-3} for chlorophyll *a*, and 1.1 m for Secchi disk transparency.

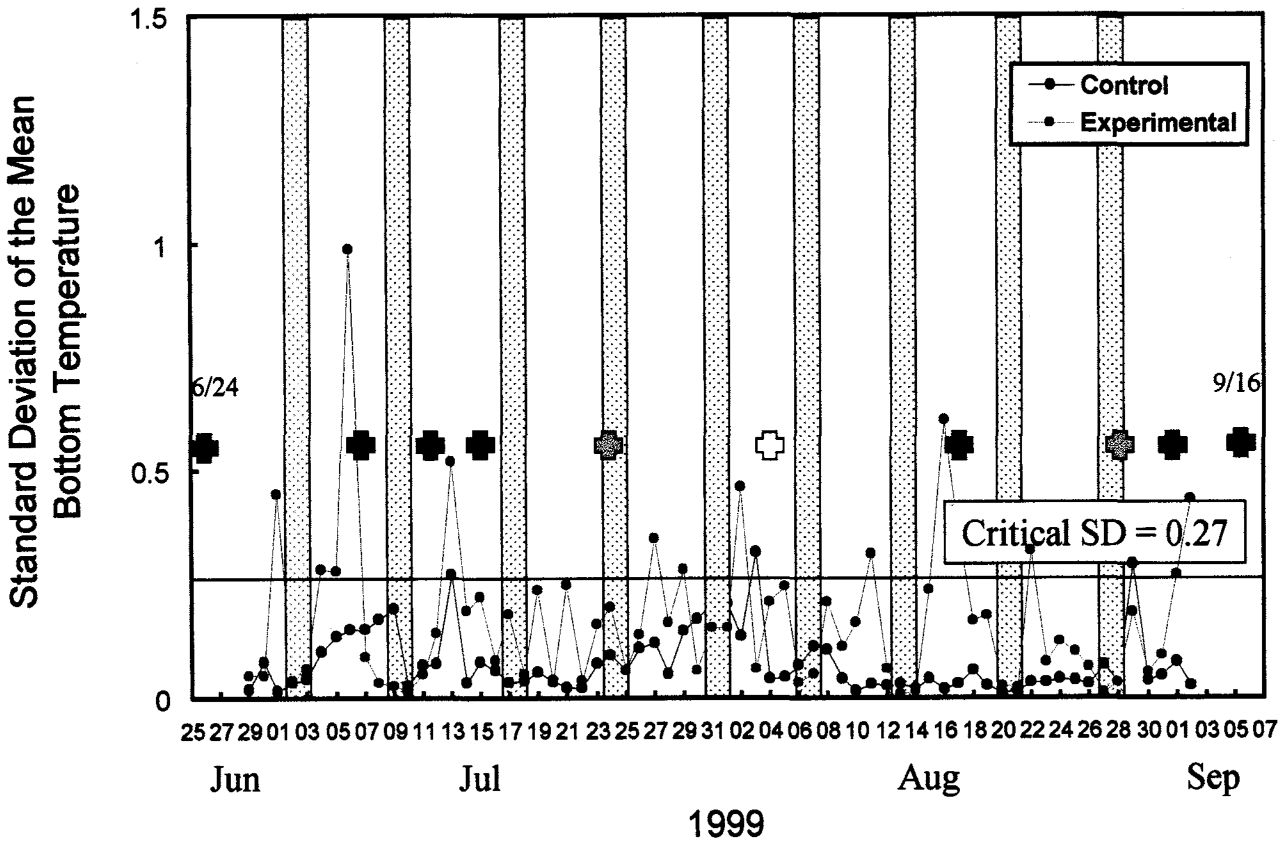

Figure 9.–Variations in the standard deviation (SD; n=9) of the mean bottom temperature for water temperature measurements collected between 1800 and 2000 hours over the summer period. Water temperature was measured at station 2C (control) and 2E (experimental) at the 2.5-m depth using thermistors and a data logger. The horizontal line represents the estimated threshold SD above which phosphorus (P) resuspension occurred. High SD and P resuspension on 17 August and 1 September were used to define the critical SD and P resuspension potential on other dates when P profiling was not conducted. Black crosses represent water sampling dates when motor boat activity occurred. Grey crosses represent water sampling dates when motor boat activity did not occur. The white cross represents a sampling date when boat failure prevented water sampling. The shaded bars represent Fridays and Saturdays when no skiing occurred.

Overall, external P sources accounted for ~20%, while internal P sources accounted for ~80%, of the measured P input to the lake (Table 1). External loading via direct P precipitation, P inflows from the Owen Park pumps, and P inflows from storm sewer inflows were approximately equal (Table 1). Internal loading via P release from the sediments represented the greatest measured source of P to the lake (42% of the total P load; Table 1). P sources to the lake via decomposition of *P. crispus* and resuspension due to motor boat activity accounted for 20% and 17% of the total P load to the lake.

We explored the following P management scenarios for Half Moon Lake: 1) reduction of storm sewer P loading by 90%, 2) reduction of P loading by motor boat activity by 90%, 3) reduction of *P. crispus* decomposition by 90% (i.e., more aggressive harvesting), 4) reduction of sediment P release by 90% through, for instance, a sediment chemical treatment, 5) reduction of both motor boat activity and sediment P release by 90%, 6) reduction of both *P. crispus* decomposition and sediment P release by 90%, 7) reduction of all measured internal P loads by 90%, and 8) reduction of all measured internal P loads and storm sewer P loads by 90%.

A reduction individually in storm sewer- or motor boat- or plant-related inputs did not change predicted summer chlorophyll *a* concentration appreciably (Fig. 12), as concentrations were still above 68 ug $\cdot$ L^{-1} after model manipulation. A 90% reduction in sediment P loading resulted in a 36% decrease in chlorophyll *a* to 53 g $\cdot$ L^{-1}, and a modest increase in Secchi Disk transparency to 1.6 m (Fig. 12). Control of both sediment P loading and plant P inputs (i.e., greater than 50% reduction in the total P load; scenario 6) resulted in an ~45% decrease in chlorophyll *a*. Control of P resuspension via motor boat activity in combination with sediment P control (scenario 5) provided similar results

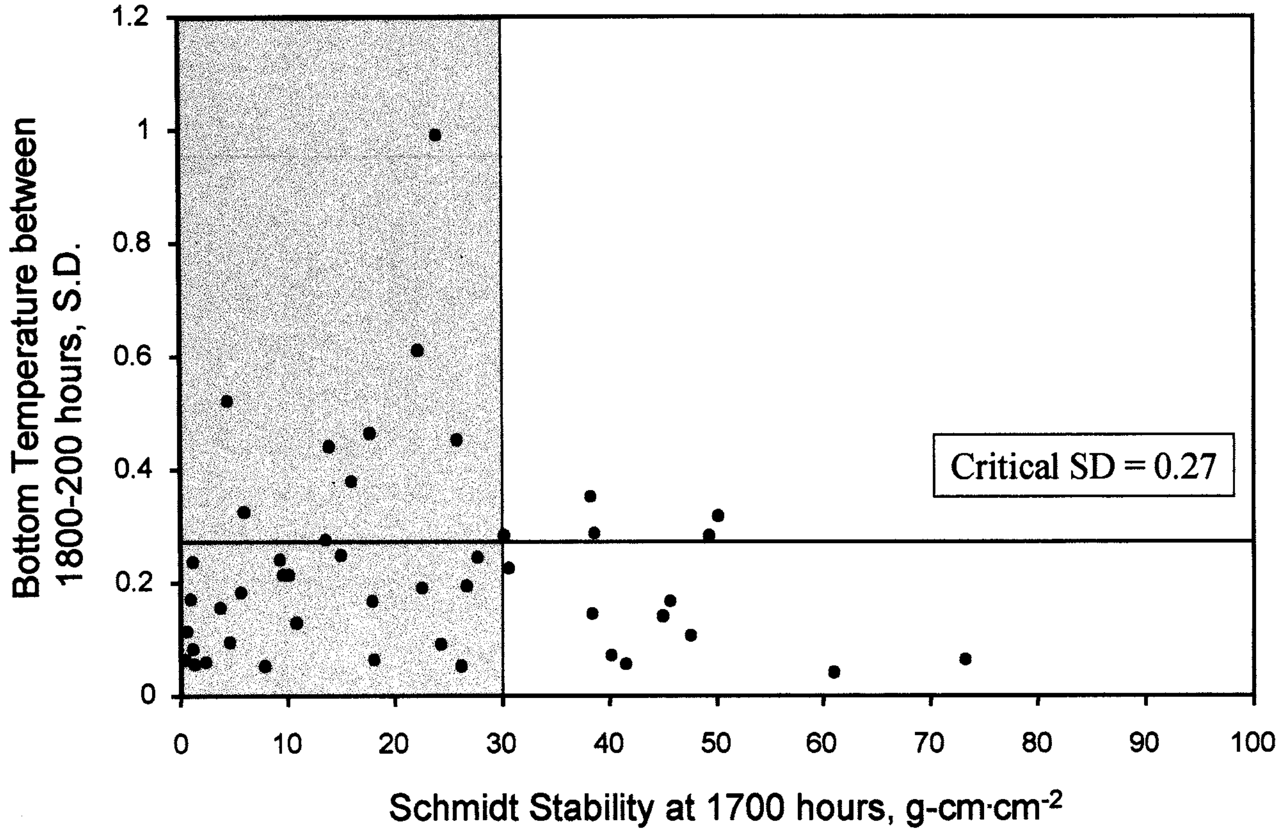

Figure 10.–Variations between the standard deviation (SD) of the mean bottom temperature for water temperature measurements collected between 1800 and 2000 hours and Schmidt stability at 1700 hours (i.e., prior to motor boat activity). Temperature data were collected at station 2E. The shaded area represents an example of setting criteria for minimizing P resuspension via motor boat activity. In this case, when Schmidt stability < 30 g-cm · cm^{2}, the potential for P resuspension is high and practice should be canceled.

as scenario 6. A reduction in all measured internal P sources (scenario 7) resulted in a substantial decrease in total P and chlorophyll *a* concentrations and an increase in Secchi disk transparency. Total P declined by nearly 60% while chlorophyll *a* declined by greater than 70%. Secchi disk transparency increased nearly 3 times (i.e., reached the bottom of the lake) as a function of internal P loading reduction. A reduction in storm sewer loading, in addition to control of internal loads (i.e., scenario 8), resulted in minor additional improvement in total P, chlorophyll *a*, and Secchi disk transparency over those observed for scenario 7.

Conclusions and Recommendations

Internal P loadings from sediments, plant decomposition, and motor boat activity were clearly important to the P economy of Half Moon Lake. There are several practical management strategies for reducing high

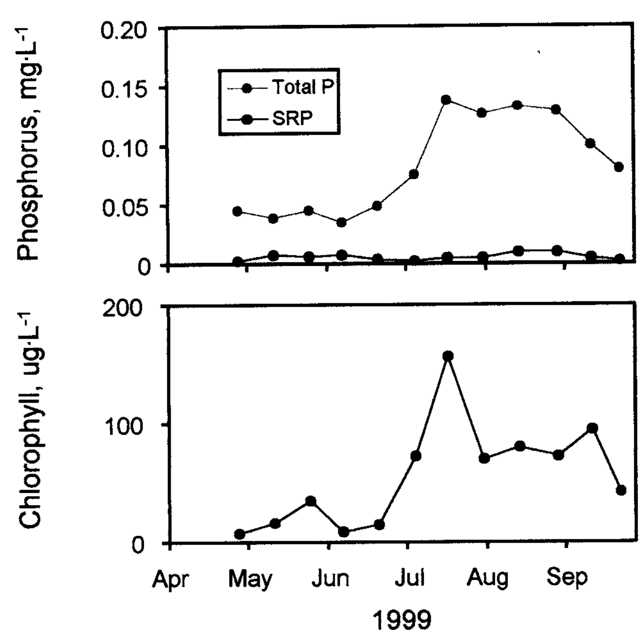

Figure 11.–Seasonal variations in mean total phosphorus (P), soluble reactive phosphorus (SRP), and chlorophyll *a* in Half Moon Lake. Concentrations were weighted with respect to depth and area of the lake using the software program *Profile* (Walker 1996).

summer algae concentrations by reducing these P incomes to the lake. Aeration of the hypolimnion and destratification to alleviate anoxic conditions may not be an effective means of controlling sediment P sources to the water column because P release from sediments appears to be regulated by pH as well as eH. Sealing the sediment from further P release via an alum application would likely improve water quality conditions in the lake. However, the lake exhibits low alkalinity (~30 mg Ca·L^{-1}), that would require extensive buffering to prevent aluminum toxicity to the biota (Cooke et al. 1993). Although P bound to alum is insensitive to redox changes, there is some evidence that high pH

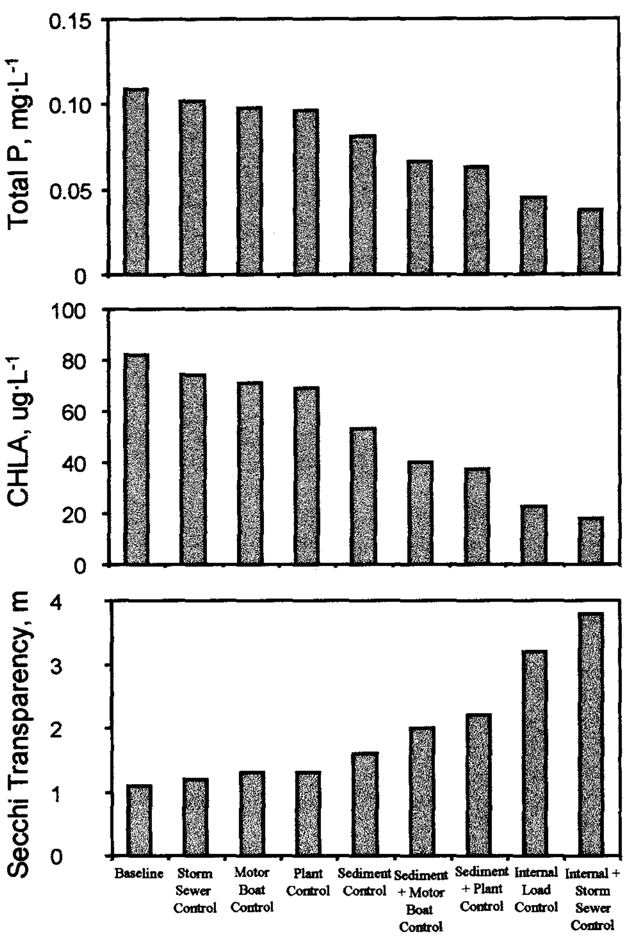

Figure 12.–Estimated changes in total phosphorus (P), chlorophyll (CHLA), and Secchi transparency as a function of various P management scenarios. Baseline = current internal and external loading conditions; storm sewer control = 90% reduction of storm sewer P loading; motor boat control = 90% reduction in P resuspension via motor boat activity; plant control = 90% reduction in plant decomposition; sediment control = 90% reduction in internal P loading from the sediments; sediment + plant control = 90% reduction in both sediment and plant P loading; internal load control = 90% reduction in loading from sediment, plants, and motor boat activity; and internal + storm sewer load control = 90% reduction in loading from sediment, plants, and motor boat activity and 90% reduction in storm sewer P loading.

(for example, due to macrophyte productivity in Half Moon Lake) may reduce the binding capacity of alum for P (Cooke et al. 1993; Welch and Cooke 1999). In addition, since rooted macrophytes obtain P primarily from sediments (Barko and Smart 1980), P recycling via *P. crispus* senescence may short circuit efforts to control sediment P via an alum treatment (Welch and Cooke 1999). Thus, removal of biomass (and macrophyte P) from the system would be recommended to reduce recycling via decomposition and to decrease productivity and elevated pH if an alum treatment were considered for Half Moon Lake. Alternatively, lime additions may be more advantageous in both controlling sediment P release and reducing macrophyte biomass if pH levels can be maintained between 8 and 10 (Prepas et al. 2001).

Phosphorus sources to the lake from decomposing *P. crispus* could be reduced by more aggressive harvesting and removal of plant material from the lake prior to anticipated *P. crispus* senescence, which typically occurs in mid- to late June. A target of 50% removal of an average biomass of ~40 g·m^{-2} by mid-June would decrease P flux from decomposing plants by 50% (i.e., 2.1 mg·m^{-2}·d^{-1} versus 1.0 mg·m^{-2}·d^{-1}; Table 2). Clipping *P. crispus* before peak biomass may also reduce the turion seed bank and its ability to persist in future years (McComas and Stuckert 2001).

Although P resuspension via motor boats did not result in increases in SRP in the water column, particulate P resuspension can still be an important nutrient source for algal growth. For instance, Asplund (1996) demonstrated that sediments in Half Moon Lake had high concentrations of NaOH-extractable P (1.8 mg P·g dry mass $sediment^{-1}$). Algae were able to use much of this sediment P for growth in laboratory assays. Others (Maceina and Soballe 1990; Hellström 1991; Søndergaard et al. 1992) have shown linkages between P resuspension, nutrient recycling, and enhanced algal growth. P resuspension via motor boat activity could be minimized by considering the thermal structure of the lake in the vicinity of the beach area prior to practices. Stronger vertical temperature-density gradients reflect greater thermal stability and require greater work in the form of motor boat disturbances to mix the water column to a uniform temperature and resuspend P. Conversely, as vertical temperature-density differences diminish, less work is required to mix the water column to a uniform temperature and the potential for bottom disturbance and P resuspension increases.

Temperature profiles collected before practice could be entered into a simple speadsheet software program to calculate local thermal stability in the vicinity of the beach area. This value could be evaluated with respect to the findings of the present study to

Table 2.–Estimated phosphorus (P) decomposition rate for *Potamogeton crispus* as a function of standing crop at the time of plant senescence.

Initial Standing Crop, $g \cdot m^{-2}$	P Decomposition Rate, $mg \cdot m^{-2} \cdot d^{-1}$
10	0.5
20	1.0
30	1.4
40	2.1
50	2.5

determine the potential for P resuspension that evening. An example of setting criteria for reducing P resuspension by motor boat activity is shown in Fig. 10. In this example, if the local thermal stability near the beach area fell below 30 g-cm · cm^{-2} (i.e., a defined critical threshold), practices could be canceled for that evening. Other options for reducing P resuspension via motor boat activity include reducing the size of motors used or eliminating all motor boat activity on the lake.

Storm sewer P loading did not appear to contribute greatly to the overall P budget of Half Moon Lake during the summer period of 1999. Nevertheless, there are many cost- effective BMPs that could be used to reduce P inputs to the lake from this source. Strategies include frequent street cleaning, city landscaping to detain and absorb more of the runoff, animal waste (geese, dogs, etc.) management, and impervious surface reduction.

Our results indicated that most of these P sources to this urban lake can be managed to improve water quality. Through the use of a budgetary analysis and identification of probable P contributors to the system we were able to examine the relative importance of each source and make decisions regarding management of that source. In the case of Half Moon Lake, our results suggested that storm sewer contributions were low relative to other P sources in the summer of 1999. Thus, it would not be cost effective at this point to spend millions of dollars for storm sewer diversion to reduce external P loading. Instead, money might be more wisely spent in the management of sediment P release, which represented the largest measured P contribution to the lake. Implementation of urban BMPs might be a more cost-effective means of reducing storm water runoff. Greater harvesting *P. crispus* from the system would appear to provide multiple water quality improvement benefits; 1) P incorporated as biomass is removed from rather than recycled within the system, 2) these nuisance, canopy-forming, aquatic plants are controlled for improved recreational use of the lake, and 3) the turion bank is being depleted, which may reduce populations in future years. Finally, our results indicated that motor boats can resuspend substantial P in this lake under certain conditions. Lake management plans will need to weigh the advantages and disadvantages of having a ski show versus ensuing water quality degradation in the form of P resuspension loading, and offer compromises to benefit both causes.

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