Limnological Analysis of Half Moon Lake, Wisconsin

by

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PREFACE

This study was conducted in response to a request from the City of Eau Claire, WI, and the State of Wisconsin Department of Natural Resources (WI-DNR) to the U.S. Army Engineer District (USAED), St. Paul, for planning assistance under Section 22 of the Water Resources Development Act (Public Law 93-251). Funding was provided by the City of Eau Claire, WI-DNR, and USAED, St. Paul. The study coordinator for the City of Eau Claire was Mr. Ken Van Ese. The study coordinator for WI-DNR was Mr. Patrick Sorge. The Section 22 coordinator for the USAED, St. Paul, was Mr. Terry Engel.

This study was conducted and the report written by Mr. William F. James, Dr. John W. Barko, and Mr. Harry L. Eakin of the Eau Galle Aquatic Ecology Laboratory (EGAEL) of the Environmental Processes and Engineering Division (EPED) of the Environmental Laboratory (EL), Engineer Research and Development Center. We gratefully acknowledge Mss. Laura Blegen, Alyssa Boock, Susan Fox, and Michele Huppert, and Messrs. Dale Dressel, Alan Lamphere, and Matthew Pommier of the EGAEL for analytical and field support on this project. We gratefully acknowledge Mr. Brian Amundson, Mr. Patrick Oldenburg, Mr.Patrick Sorge, and Mr. Daniel Wilcox for reviewing this manuscript.

SUMMARY

We examined external constituent loadings from storm sewer inflows and outlet structure discharges from Half Moon Lake, internal phosphorus (P) fluxes from profundal sediments, decaying *Potamogeton crispus*, and motor boat activity, and water quality conditions in Half Moon Lake, Wisconsin in 1999. Water samples for external loadings and discharge determinations were collected from storm sewers and the outlet structure of Half Moon Lake. Five stations were established in the lake for limnological profiling during April through September. Sediment cores collected in the lake (total = 36) were incubated in the laboratory for determination of P release under oxic and anoxic conditions and high pH. A macrophyte biomass survey was conducted in early June, prior to *P. crispus* senescence, for determination of macrophyte P available for decomposition. Rates of leaching and breakdown of macrophyte P were measured using laboratory incubation and mesh bag techniques. P resuspension via motor boat activity was examined by measuring changes in bottom temperature and concentrations of total P during ski shows and practices.

In the lake, stratified conditions were observed in mid-May, mid-June, and most of July. During periods of stratification, oxygen depletion occurred in the bottom waters and anoxia developed during July through early August. The pH at station 10 was > 9.0 in the surface waters between April and early July.

Lakewide average total P was constant near 0.05 mg/L between late April and early June and increased to > 0.10 mg/L in mid-July. It remained > 0.10 mg/L between mid-July and August and declined to < 0.10 mg/L in September. Lakewide average soluble

reactive P (SRP) was very low throughout the study. Lakewide average total nitrogen (N) exhibited a peak in concentration in July, and declined steadily between mid-July and September. Lakewide average ammonium-N (NH_3-N) was low in concentration throughout the study. With the exception of a minor peak in mid-May, lakewide average chlorophyll was < 20 μg/L and nearly constant between April and mid-June. It increased to an average concentration of 155 μg/L in July coincident with peaks in lakewide average total N and total P.

In general, mean concentrations of total P(~ 0.109 mg/L), chlorophyll (~82 μg/L) and Secchi disk transparency (~1.1 m) values were similar at 4 of 5 sampling stations during the summer. Station 5 exhibited lower total P and chlorophyll, and a higher Secchi disk transparency, than the other stations due most likely to dilution and flushing influences of groundwater inputs via the Owen Park pumping facility. Overall, the Carlson and Wisconsin trophic state indices were high for all stations and indicative of very eutrophic conditions.

There were a number of rain storms and high inflow periods to the lake during the study. The groundwater pumping at Owen Park and net seepage from the lake accounted for the greatest inflow (~54%) to and outflow (~50%) from the lake, respectively. Storm sewer inflows accounted for 15% of the measured inflow to the lake. The average residence time of the lake during summer was 198 days. Half Moon Lake was a sink for seston, total N, and total P, capturing 97, 78, and 91% of the measured loads, respectively. Measured external P loading to the lake during the summer was 1.2 mg m^{-2} d^{-1}.

Internal P loading (i.e., from sediments, plant decomposition, and motor boat activity) accounted for 80% while P loading from external sources (i.e., storm sewers, Owen Park pumping facility, and precipitation) accounted for the remainder of the total P load to the lake during the summer. Storm runoff into sewers, the Owen Park pumping facility, and direct precipitation each accounted for ~ 33% of the external P load during the summer.

Sediment represented the greatest internal P load to the lake, accounting for 53% of the total internal P load. Rates of P release from sediments were high under anoxic conditions (range=2.3 to 11.7 mg m^{-2} d^{-1}) and comparable to those rates measured for other eutrophic systems. Anoxia was observed in the bottom waters at several stations intermittantly during July through August, suggesting that reducing conditions in the hypolimnion were important in the regulation of P release from the bottom sediments during these periods. High pH enhanced the rate of P release from sediments under oxic conditions, as there was a strong positive and linear relationship between pH and the rate of P release. In general, it appeared that high pH and oxic conditions primarily regulated P release from sediments during the summer in the lake. Photosynthetic activities by algae and aquatic macrophytes can drive pH up to high values. The lakewide rate of P release from sediments for the summer was high at 2.5 mg m^{-2} d^{-1} and indicative of eutrophic conditions. These results suggested that aeration techniques to reduce anoxia may not be an effective means of controlling sediment P sources to the water column because P release from sediments is high under oxic conditions due to high pH.

During the period May through mid-June, ~ 9000 kg biomass (dry mass) and 30 kg macrophyte P were mechanically harvested from Half Moon Lake. The estimated macrophyte biomass in Half Moon Lake after final harvesting on 13 June was 13,580 kg. Lakewide plant P mass available for flux to the water column (assuming complete decomposition) at that time was 60 kg, or about twice that removed by harvesting. Loss of P from *P. crispus* contained in mesh bags was greatest during the first week of decomposition, with 40% of the P loss occurring during the first 2 days. Within 30 days, nearly all of the P mass was lost from the mesh bags. Leaching of soluble P from plant tissue (i.e., autolysis) into the water, measured using laboratory incubation systems, was greatest during the first 24 hours of decomposition. In general, leaching of P directly into the water column dominated the decomposition process during the early stages, while breakdown of P via bacterial degradation and sloughing dominated the later stages of decomposition. Combined P breakdown and leaching rates for the summer period was 1.2 mg m^{-2} d^{-1} .

Motor boat activity generally occurred during the evening hours (~1730-2030 hours) on Sunday through Friday. We estimated that P resuspension during motor boat activity occurred at the experimental site ~21% (i.e., on 14 dates during the evening hours) of the time during the summer (June-August) for an average lakewide rate of ~ 1.0 mg m^{-2} d^{-1}. During two periods of motor boat activity (17 August and 1 September, 1999) the estimated mass of P resuspended was ~ 2.8 kg. In general, stronger thermal stratification (measured as stability) prior to motor boat activity was associated with minimal P resuspension, whereas weaker stratification prior to motor boat activity was associated with more occurrences of P resuspension.

Several P management scenarios were explored for Half Moon Lake using *Bathtub*: These scenarios included 1) reduction of storm sewer P loading, 2) reduction of P loading by motor boat activity, 3) reduction of *P. crispus* decomposition by more aggressive harvesting, 4) reduction of sediment P release through, for instance, an alum treatment, 5) reduction of both motor boat activity and sediment P release 6) reduction of both *P. crispus* decomposition and sediment P release, 7) reduction of all measured internal P loads, and 8) reduction of all measured internal P loads and storm sewer P loads. We did not include in any management scenario exploration a reduction in P inputs via the Owen Park pumping facility, because this hydrological source is important in control of pool elevation in the lake.

Bathtub modeling results indicated that control or management of more than one P source is required in order to achieve a significant reduction in algal biomass. Modeling results suggested that internal loads should be targeted for P control and algal bloom reduction. Practical management strategies include an alum treatment to reduce sediment P release, more aggressive harvesting of *P. crispus* communities prior to mid-June to remove macrophyte P from the system before plant senescence, and minimizing P resuspension by motor boat activity by considering the thermal structure of the lake prior to ski practices.

INTRODUCTION

The overall objectives of these investigations were to examine water quality conditions and constituent fluxes in Half Moon Lake, Wisconsin. In particular, the relative importance of various internal and external nutrient (primarily phosphorus or P) loadings were evaluated in relation to water quality conditions and phytoplankton biomass (chlorophyll) in the lake. Primary internal sources of P included the profundal sediment, decaying *Potamogeton crispus*, and P resuspension generated by motor boat activity during ski show events. Predicted impacts of P loading reduction on viable chlorophyll *a* concentrations in the lake were examined using the model *Bathtub* (Walker 1996).

METHODS

Study Site

Half Moon Lake is a small oxbow of the Chippewa River located in Eau Claire, WI. Morphological and watershed characteristics are reported in Barr Engineering (1992). A summary of information from that report pertinent to the present study is presented in Table 1.

Eight storm sewers enter the lake in several locations. Other water sources to the lake include ground water that is pumped through storm sewer 7b from wells located at the Owen Park facility, situated southeast of the lake on the Chippewa River, and direct runoff from 3 sub-watersheds that are not drained by the storm sewer system. 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 (MSL). The lake is currently eutrophic and exhibits high algae and aquatic macrophyte growth (Borman 1990; Brakke 1995; Konkel and Borman 1996)

Hydrology

Flows (cubic meters per second; cms) were measured continuously (15-min intervals or less) at storm sewer 2 between June and September using a recording stage height-velocity sensor (ISCO Model 750). Flows were estimated from stage height measurements recorded between April and June (15-min intervals) using stage height-flow relationships developed from the summer period.

For storm sewer 5, daily flows were estimated indirectly using relationships between daily flow and daily precipitation measured during the month of June (n=5 storm inflows; r^2=0.97). Flows were measured continuously in June at this station using a recording stage height-velocity sensor (ISCO Model 750). Measurements during other periods could not be conducted due to repeated equipment failure at this station.

We estimated flows generated from storms independently from those generated via groundwater pumping from the Owen Park facility for storm sewer 7b. Flows during storms were measured continuously (15-min intervals or less) at this station between June and September using a recording stage height-velocity sensor (ISCO Model 750). Storm-related daily flows occurring prior to June were estimated using relationships between daily flow and daily precipitation (n=20 storms; r^2 = 0.90). To estimate flows contributed via groundwater pumps, a stage height recorder was deployed in the manhole located at 4th Street and Niagra Ave between June and September. Our original intention was to estimate flows via groundwater pumps at storm sewer 7b during periods of no precipitation for use in the development of a hydrological rating curve. However, lake level intrusion into storm sewer 7b during the summer biased flow measurements collected in 1999. We instead developed a rating curve for groundwater pumping by collecting flow information in the manhole at 4^{th} Ave. and Niagra St. at different pumping rates in June 2000.

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.

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, personal communication). 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 (cms) was developed using the following equation:

1) _ Lake Volume = (Storm Sewer Inflows + Owen Park Pumping + Direct Precipitation)

- (Direct Discharge + Net Seepage)

The change in lake volume (_ Lake Volume) was estimated from changes in pool elevation and estimates of pool volume at those elevations. We used the pool volume of 722.3 acre-feet at a pool elevation of 769.05 ft mean sea level (msl) as the nominal volume estimate (Barr Engineering 1992), and a lake surface area of 132.1 acres (Barr

Engineering 1992) to calculate incremental volume increases and decreases as a function of pool elevation. To estimate hydrological contributions by storm sewers that were not monitored during the study, we assumed that flows from storm sewers 1 and 6 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. Net seepage was estimated by difference using equation 1. We could not measure direct runoff from the land and did not include estimates for this source in our hydrological budget. Thus, inflows to the lake may be underestimated.

External 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.
As with flow measurements, samples for water quality analysis collected in 1999 at the mouth of storm sewer 7b during non-precipitation periods for characterization of groundwater pumping inputs were biased by frequent lake level intrusions into the storm sewer. These data had to be discarded. Additional water samples were collected from the manhole located on 4th St. and Niagra Ave. in 2000 to monitor inputs to the lake via groundwater pumps. We assumed that concentrations were similar in 1999 and 2000.

Water samples collected at various inflows and the outflow were analyzed for the variables listed in Table 2. For total suspended sediment (TSS) analyses, suspended material retained on a precombusted glass fiber filter (Gelman (A/E) was dried to a constant weight at 105^{o} C (APHA 1992). Samples for total nitrogen (N) and phosphorus (P) were predigested with potassium persulfate according to Ameel et al. (1993) before analysis. Total N and P were measured colorimetrically on a Lachat QuikChem automated system (Lachat Methods 10-107-04-1-A and 10-115-01-1-A; Zellweger Analytics, Lachat Div., Milwaukee, WI). Soluble reactive P (SRP; Lachat Method 10-115-01-1-A) and ammonium-N (NH_3-N; Lachat Method 10-107-04-01-A) were analyzed colorimetrically using automated procedures (Zellweger Analytics, Lachat Div., Milwaukee, WI).

Loadings by various external sources were estimated using the computer model *Flux* (Walker 1996). Loadings were estimated via either weighting concentrations with respect to flow (Method 2) or via a regression algorithm (Method 6).

Internal Loadings

Sediment. In August of 1999, nine replicate intact sediment cores were collected from the profundal sediments of stations 10, 20, 30, and 40 (Fig. 1), for determination of rates of SRP release from the sediment. Sediments were too flocculent for collection at station 5. 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). Additional lake water was collected from the epilimnion for incubation with the collected sediment. Overall, a total of 36 sediment cores were collected for examination of P release from sediments in Half Moon Lake.

Sediment systems, constructed according to the methods of James et al. (1995), were incubated in an environmental chamber at 20^{o} C for 1-2 weeks. One set of 3 replicate sediment incubation systems was subjected to an oxic environment while the other set (3 replicates) 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. Bubbling action insured complete mixing of the water column but did not disrupt or resuspend the sediment. A third set of replicate sediment incubation systems was subjected to an oxic environment and high pH (~9.0) by bubbling with CO_2-free air. Water samples were collected daily from the overlying water of each sediment system, filtered through a 0.45 μm membrane filter, and analyzed colorimetrically for SRP using the ascorbic acid method (APHA 1992). Rates of P release from the sediment (mg m^{-2} 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 (0.00332 m^2).

Potamogeton crispus decay. 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*. The lake was divided into six equal regions for macrophyte sampling purposes. In each region, one transect with 3 sampling stations was established perpendicular to the shoreline on each side of the lake total of 2 transects and 9 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 sampler. Thus, a total of 9 samples were collected along each transect. Overall, a total of 108 samples were collected throughout the lake. The quadrat sampler consisted of a sheet metal enclosure measuring 0.75 m by 0.75 m. The length of the sampler was 1.5 m. The sampler was lowered using a winch to enclose a sediment area. A rake was used to carefully pull macrophytes from the area enclosed by the sampler. 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 and allowed to spin-dry for a period of 5 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^{o} 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. Since intensive 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 dia. 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 at 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^{o} C to a constant weight. Subsamples of tissue were analyzed for P (see above methods). Turions were excluded from all mass and tissue P determinations because they are resistant to decomposition.

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 a 1-L beakers containing filtered lakewater and incubated in an environmental chamber at 20^{o} 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.

Motor boat disturbances. To examine P fluxes into the water column via motor boat disturbance during Ski Sprite ski shows and practices, three sampling stations were established in control and experimental sites for water sampling purposes (Fig. 1). Stations 2C and 2E were located in the approximate center of the control and experimental sites. Station 2E was located near the ski jump. 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. 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 rehearsals and 4 ski shows. Sampling was also conducted during 3 periods when no ski event 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, i*n situ* 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 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 using a close-interval syringe sampler as described by James et al. (1992). Total P was analyzed using automated techniques after digestion with potassium persulfate (APHA 1992). Turbidity was determined using a Hach Nephelometer (Model 2100N).

Recording thermistors 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:

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 :

where V is lake volume (m^3) and V_z is the volume at depth z. Areas and volumes for stations 2C and 2E were determined using a graphics tablet (SummaSketch II, SummaGraphics, Inc.) . Stability (g-cm cm^{-2}) represents 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 are indicative of strong stratification and greater work required to disrupt stratification. Conversely, lower stability values are indicative of weak stratification and less work required to disrupt stratification. In shallow aquatic systems such as Half Moon Lake, disruption of stratification (i.e., declines in water column stability) can lead to water column mixing and the vertical transport of nutrients from the bottom waters to the euphotic zone for uptake by algae.

Attempts to quantify water movements directly by deploying a fluorescent dye on the bottom of the lake failed and will not be reported here. The primary difficulty encountered was in tracking the dye after its deployment on the lake bottom. It appeared that advective dye movement along the lake bottom was very rapid, resulting in horizontal movement of dye from the study areas (primarily the control area) within hours. This pattern may be attributed to near-bottom water currents and needs to be pursued in future studies.

Limnological Monitoring

Five stations (stations 10 through 50) were established along the longitudinal axis of Half Moon Lake for limnological monitoring purposes (Fig. 1). During the ice-free period, water samples were collected biweekly at 1-m intervals from the surface (i.e., 0.1 m) to within 0.5 m from the bottom for analysis of the variable listed in Table 2. For soluble constituents (i.e.,SRP), samples collected from anoxic water in the lake were filtered immediately without exposure to oxygen. Water samples for analysis of soluble constituents were filtered through a 0.45 μm filter (Gelman Metricel) prior to analysis. SRP (Lachat Method 10-115-01-1-A) and NH_3-N (Lachat Method 10-107-04-01-A) were analyzed colorimetrically using automated procedures (Zellweger Analytics, Lachat Div., Milwaukee, WI). Total N and P were analyzed using automated procedures after digestion with potassium persulfate (Ameel et al. 1993). Turbidity (APHA 1992; Method 2130 B.) was measured using a Hach nephelometric turbidimeter (Model 2100N). Samples for chlorophyll were extracted in dimethyl-sulfoxide (DMSO)-acetone (50:50) at $< 0^{o}$ C for a minimum of 12 hours. Viable chlorophyll *a* was determined fluorometrically (Turner Designs; Model TD-700) according to Welschmeyer (1994). In conjunction with the water sampling schedule, measurements of water temperature, dissolved oxygen, pH, and conductivity were collected using a Hydrolab Data Sonde 4 that was precalibrated against Winkler titrations (APHA 1992) and buffer solutions. The Carlson Trophic State Index (Carlson 1977) was estimated using the computer program *Bathtub* (Walker 1996) using Secchi transparency values and total P and viable chlorophyll *a* concentrations determined over the upper 2 m of the lake. In addition, the Wisconsin Trophic State Index was estimated using equations described in Lillie et al. (1993).

Bathtub Modeling

The computer 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 and total P weighted for the entire lake over the period June through August as average summer conditions. The computer program *Profile* (Walker 1996) was used to estimate weighted summer concentrations for input into *Bathtub*.

RESULTS

Hydrological Conditions

During April through September, measurable precipitation occurred on 40 days (Fig. 2). Daily rainfall exceeding 1 inch occurred in early April, mid-May, and early June and July. In general, peaks in mean daily inflow from monitored storm sewers coincided with peaks in precipitation during the study period (Fig. 2). Storm sewer 3 exhibited the greatest storm-related mean flow during April through September, followed by storm sewer 7b and 2 (Table 3). In contrast, storm sewers 4, 5, and 7a exhibited much lower mean flows (Table 3). When normalized with respect to watershed area, however, a different pattern emerged. Runoff (inches/y) from the small watershed draining into storm sewer 4 was high and comparable to runoff from the much larger watersheds draining into storm sewers 2 and 7a and b (Table 3). The watershed draining into storm sewer 3 exhibited the greatest runoff during the study period (Table 3).

Overall, the greatest measured water income to Half Moon Lake during the study period was provided by the groundwater pumping station at Owen Park (Fig. 2; Table 4). Flows from this source varied in June through late-July as an apparent result of operation of one pump (i.e., as in early June) versus two pumps (i.e., as in late July; Fig. 2). Flows from this source were greatest in August through September as a result of apparent operation of two pumps (Fig. 2). Direct precipitation provided the second greatest water income to Half Moon Lake followed by storms sewers (Table 4).

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. 3). Net groundwater seepage, estimated via hydrologic balance, represented the greatest water outflow from the system (Table 4). Evaporation, based on estimates provided in Barr Engineering (1992), accounted for 23% of the water outflow (Table 4). The average residence time of the lake in 1999 was 198 days.

Pool elevation exceeded the nominal level of the surface outlet structure (234.4 m MSL) in April through early June, coincident with high precipitation (Fig. 3). Declines in precipitation in mid-June through late July (Fig. 3), coupled with a lower groundwater pumping rate (Fig. 2), were associated with a decrease in pool elevation below the nominal pool elevation of 234.4 m MSL (Fig. 3). High precipitation in late August and higher groundwater pumping rates between mid-July and September resulted in an increase in pool elevation above the nominal level in late August through mid-September.

Sediment and Nutrient Sources and Sinks

External Loadings.

Concentrations of N (i.e., total N and NH_3-N) and P (i.e., total P and SRP) for the groundwater pumping sources were generally low relative to precipitation-related concentrations observed for the storm sewers (Table 5). However, loadings of these constituents by the Owen Park facility were greatest during April through September due to higher overall flow rates (Table 5). Storm sewers 2, 3, and 7b (not including groundwater pumping) dominated precipitation-related loading of seston and nutrients to Half Moon Lake (Table 5, Fig. 4). Similar to trends in runoff coefficients, storm sewer 3 exhibited the greatest watershed export coefficients for N and P (Table 6). Half Moon Lake was a sink for seston, total N and total P, capturing 97, 78, and 91% of the measured loadings, respectively (Table 7). External P loading from all measured storm sewers, normalized with respect to lake surface area and the summer period used in *Bathtub* modeling (June-August), was 0.3 mg m^{-2} d^{-1} (Table 8). External P loading from groundwater pumping sources during the same period was 0.4 mg m^{-2} d^{-1}. Estimated contributions via direct precipitation represented a rate of 0.5 mg m^{-2} d^{-1}.

Internal Loadings.

Profundal sediments. 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 in-lake stations (Fig. 5). Rates of P release under oxic conditions were also high and varied linearly as a function of pH (Fig. 6). At pH values near 7.0, rates of

P release under oxic conditions were < 0.5 mg m^{-2} d^{-1}. At pH values near 8.5, rates of P release under oxic conditions 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 September (Fig. 7). During periods of anoxia, the pH of the bottom waters declined below 7.5 (Fig. 7). In contrast, pH approached 10.0 in late April and was > than 8.0 in May through June and mid-August through September. We used fluctuations in pH and the occurrence of anoxia to estimate a lakewide rate of P release from sediment during the summer period (Fig. 7). In general, the estimated lakewide rate of P release from sediment was greatest in July, coincident with the occurrence of bottom water anoxia (Fig. 7). However, it was also high during other summer periods when oxic conditions existed 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 (June through August) was 2.5 mg m^{-2} d^{-1} (Table 8).

Potamogeton crispus biomass and decomposition. During the period May through mid-June, ~ 9000 kg standing crop (dry mass) and 30 kg macrophyte P were harvested from Half Moon Lake (Fig. 8). The estimated macrophyte standing crop and P mass remaining in Half Moon Lake after final harvesting on 13 June was 25.4 g/m^2 (± 2.3 S.E., n=108) and 111.6 mg/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*. The occurrence of other macrophyte species, as well as *P. crispus*, was only observed in the southwest embayment of the lake.

From mesh bag decomposition experiments in 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. 9). Leaching of soluble P from plant tissue (i.e., autolysis) into the water, measured using laboratory incubation systems, was greatest during the first 24 hours of decomposition (Fig. 10). On day 14 and 30 of decomposition, leaching of P from plant tissue was minor (Fig. 10), but fragmentation, breakdown, and loss of plant tissue P, which most likely occurred via microbial degradation, continued between day 14 and day 30 of decomposition (Fig. 9). Within 30 days, nearly all of the P mass was lost from the mesh bags (Fig. 9). In general, leaching of P directly into the water column dominated the decomposition process during the early stages, while breakdown of P via bacterial degradation and sloughing dominated the later stages of decomposition (Fig. 11). Combined P breakdown and leaching rates, normalized with respect to lake surface area and the summer period used in *Bathtub* modeling (June-August), was 1.2 mg m^{-2} d^{-1} (Table 8).

Phosphorus resuspension via motor boat activity. Activities on 17 August provided an example of P flux into the water column via motor boat activity during a tournament rehearsal. On that date, thermal stratification was observed at both control and experimental sites between 1600 and 1719 hours shortly before the start of the rehearsal at ~ 1730 hours (Fig. 12). After the rehearsal (~2000 hours), the water column was completely destratified at station 2E, and bottom water warmed by > 1^{o} C due to mixing and entrainment of warmer surface water to the bottom of the lake. Destratification was less noticeable at stations 1E and 3E, located at the northern and southern extent of the ski area, respectively. 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 evening air temperature declines. Unlike station 2E, bottom temperatures at all 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 rehearsal (Fig. 13). After the rehearsal, dissolved oxygen in the bottom waters at station 2E increased dramatically (i.e., by > 5 mg/L) as a result of motor boat-induced mixing and entrainment. In contrast, the bottom waters at stations 1E and 3E remained anoxic after the rehearsal, indicating less mixing and reaeration of the bottom waters. The bottom waters were anoxic after the rehearsal at all control stations (Fig. 13).

Prior to the start of rehearsal on 17 August, turbidity and total P concentrations were similar at stations 2C and 2E (Figs. 14 and 15). Turbidity ranged between 10 and 20 NTU while total P ranged between 0.10 and 0.15 mg/L at both the control and experimental station before the rehearsal. After the rehearsal, both turbidity and total P concentrations increased at station 2E. Concentration increases at station 2E were observed primarily in the lower 1 m of the water column (Figs. 14 and 15). In contrast, concentrations at station 2C remained nearly constant before and after the rehearsal. At station 2E, the mean total P concentration of 0.140 mg/L (±0.002 S.E.) observed after the rehearsal was significantly different than the mean concentration of 0.123 mg/L (± 0.015 S.E.) observed before the start of the rehearsal (T-Test; SAS 1994; Table 9). There were no significant differences in total P concentrations measured before (0.122 mg/l ± 0.004 S.E) and after (0.115 mg/L ± 0.002 S.E.) the rehearsal at station 2C.

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 Sprites also conducted a tournament rehearsal on that particular date between ~ 1730 and 2030 hours. Before the start of the rehearsal, a metalimnion was observed between the 1.0- and 2.5-m depths at station 1E, 2E, and 3E (Fig. 16). Within the metalimnetic region at these stations, water temperature variation ranged between 2 and 3^{o} C (Fig. 16). Similarly, a metalimnion located between the 2- and 2.5-m depths was observed at all control stations before the start of the rehearsal (Fig. 16). Shortly after the end of the rehearsal, nearly complete water column destratification was observed at stations 1E and 2E. While the epilimnion deepened at station 3E coincident with motor boat activity, stratified conditions were observed in the bottom waters at this station after the rehearsal. In contrast, the water column remained stratified at all control sites after the rehearsal (Fig. 16).

Before the tournament rehearsal, dissolved oxygen concentrations were > 10 mg/L in the epilimnion at both the control and experimental stations (Fig. 17). All stations exhibited marked declines in dissolved oxygen in the hypolimnion and near anoxic (< 5 mg/L) conditions in the bottoms waters immediately above the sediment interface shortly before the start of rehearsal. After rehearsal, dissolved oxygen profiles at the control stations were similar to those observed between 1700 and 1800 hours, and anoxia persisted in the bottom waters immediately above the sediment interface (Fig. 17).

Deepening of the epilimnion at station 2E as a result of motor boat activity was accompanied by an increase in dissolved oxygen concentrations at the 2-m depth (Fig. 17). Unlike patterns observed on 17 August, however, anoxia persisted near the sediment interface after the rehearsal at station 2E. In addition, dissolved oxygen concentrations declined in the upper 1.75-m of the water column at station 2E after the rehearsal. The dissolved oxygen profile did not change at station 3E in conjunction with motor boat activity. At station 1E, dissolved oxygen increased in the bottom waters immediately above the sediment interface, suggesting some entrainment.

Unlike patterns observed on 17 August, turbidity exhibited only a slight increase in the bottom waters at station 2E as a result of motor boat activity on 15 July (Fig. 18), suggesting that sediment resuspension was minor. Concentrations of total P were high (i.e., > 0.1 mg/L) at both the control and experimental station before the rehearsal (Fig. 19). Total P declined slightly at depths > 0.75 m at station 2E after the rehearsal. The control station also exhibited a decline in total P in the upper 1-m of the water column after the rehearsal. Overall, there were no significant changes in mean total P as a result of motor boat activity in either the control or experimental station on 15 July (Table 9).

An example of thermal and chemical stratification patterns during a control period when no ski show or tournament rehearsal occurred is shown in Figs. 20 through 23 for 28 August. On that date, both control and experimental stations exhibited stratification during the period of ~1600 and 1730 hours (Fig. 20). A distinct metalimnion was observed between the 1- and 2-m depths during those hours at both control and experimental stations. Thermally-stratified conditions persisted during the period ~2000 and 2100 hours, and bottom temperatures remained constant during both time periods at all stations.

Dissolved oxygen exhibited strong stratification at all stations, with high concentrations in the surface waters and anoxic conditions in the bottom waters during the period 1600-1700 hours (Fig. 21). Approximately 4 hours later (i.e., 2000-2100 hours), dissolved oxygen stratification remained strong with anoxic conditions occurring in the bottom waters at all stations. Increases in dissolved oxygen concentration were observed in the metalimnetic region (i.e., 1-2 m), at many of the stations during the 2000-2100 time period (Fig. 21).

Turbidity and total P concentrations were nearly uniform in the water column during both the 1600-1700 and 2000-2100 hour periods at the control and experimental stations (Figs. 22 and 23). Total P concentrations increased slightly between the 1.5- and 2-m depths during the 2000-2100 period at the control station, relative to total P concentrations measured during the 1600-1700 period (Fig. 23). Mean concentrations of total P were not significantly different for the periods 1600-1700 hours and 2000-2100 hours for the control and experimental station (Table 9).

In general, significant increases in total P concentration were observed at station 2E in conjunction with motor boat activity on 2 of the 8 sampling dates; 17 August and 1 September (Table 9). On 4 dates when motor boat activity occurred (i.e., 24 June, 7 July, 12 July, and 15 July), no significant changes in total P as a result of mixing and P resuspension were observed in the water column of station 2E (Table 9). Total P concentrations did not change significantly in either control or experimental stations on 3 dates (i.e., 23 July, 28 August, and 16 September) when no motor boat activity occurred (Table 9). On all dates and time periods, concentrations of SRP in the water column were very low (<10 μg/L) at the control and experimental station, suggesting that most of the total P was in particulate form.

We used the standard deviation (S.D.; i.e., the dispersion of values around the mean) of the mean bottom temperature, determined over the time period 1800-2000 hours (i.e., general hours of ski boat operation), as a surrogate measurement of the relative extent of mixing of the bottom water. 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 (Table 9), bottom temperatures at the experimental station 2E fluctuated and warmed dramatically between 1800-2000 hours as a result of mixing and entrainment (Fig. 24), resulting in a high bottom temperature S.D. of 0.38 (Table 9). In contrast, bottom temperature at the control station 2C did not fluctuate during that time period (Fig. 24) and the bottom temperature S.D. was only 0.03 (Table 9). A similar pattern in bottom temperature fluctuation occurred on 1 September (S.D. = 0.27) when total P concentrations increased significantly in the water column at the experimental station 2E (Table 9).

The bottom temperature S.D. was generally much lower in conjunction with no motor boat activity, as on 28 August (i.e., S.D. = 0.03; Table 9), and during periods when mixing and entrainment were not great enough to resuspend P, as on 15 July (i.e., S.D. = 0.22; Table 9). In contrast to patterns observed on 17 August (Fig. 24) and 1 September, bottom temperatures at the experimental site did not fluctuate abruptly on 15 July and 28 August (Figs. 25 and 26).

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. 27). In contrast, bottom temperature S.D. exceeded the critical S.D. on only 3 dates at the control station during the study period (Fig. 27). Our percentage represents a conservative estimate of periods of P resuspension because we could not account for possible resuspension during 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 thus, the bottom temperature S.D. would be minor even though P resuspension potential was high.

The stability of the water column (i.e., the amount of work required to mix a stratified water column) shortly before the start of motor boat activity (i.e., 1700 hours), also tended to be lower at the experimental station on 17 August and 1 September in conjunction with P resuspension on those dates (Table 9). In contrast, high water column stability values at the experimental station on 7, 12, and 15 July were associated with minimal P resuspension, suggesting that the strength of stratification prior to motor boat activity had an impact on P resuspension.

A relationship between the stability of the water column at the experimental station shortly 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. 28. In general, when stability was > 25 g- cm/cm^2, bottom temperature S.D. was near or below the critical S.D. of 0.27 during motor boat activity, suggesting no P resuspension. During practices, the bottom temperature S.D. increased above the critical threshold on

7 occasions in conjunction with declines in stability below ~ 25 g-cm/cm^2. However, on other occasions when stability was below ~ 25 g-cm/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 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 25 g-cm/cm^2. Below this stability value, potential P resuspension appeared to be greater for practices than for the ski shows.

The average change in total P concentration in the water column at station 2E for the dates 17 August and 1 September (0.021 mg/L; Table 9) 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/d (i.e, 0.021 mg P/L 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 (June-August) was ~ 1.0 mg m^{-2} d^{-1} (i.e., 2.8 kg P/d ÷534590 m^2 lake area x 0.2; Table 8).

Limnological Conditions in the Lake

<u>Vertical and Seasonal Water Quality Patterns at Station 10</u>. Station 10 reflected general

vertical and seasonal limnological patterns in Half Moon Lake. At this station, stratified conditions were observed in mid-May, mid-June, and most of July (Fig. 29). During periods of stratification, oxygen depletion occurred in the bottom waters and anoxia developed during July through early August (Fig. 30). Concentrations of dissolved oxygen were also < 5 mg/L in the surface waters during the same period (Fig. 30) in conjunction with very high chlorophyll concentrations (> 100 μg/L; see Fig. 32). Since station 10 was visited in the mornings (~1000 hours), low dissolved oxygen in the surface waters may represent the outcome of nighttime respiration activities at that station. The pH at station 10 was > 9.0 in the surface waters between April and early July (Fig. 31). The pH declined in the bottom waters in late July in conjunction with hypolimnetic anoxia. Lower pH (< 9.0) throughout the water column was observed between late July and September.

Chlorophyll at station 10 was homogeneous in the water column and near 40 μg/L between April and mid-June (Fig. 32). Concentrations increased markedly at this station in July, exceeding 150 μg/L throughout the water column in mid-July (Fig. 32). Chlorophyll declined to ~ 80 μg/L in August.

Total P was very high at station 10 throughout the study period. Total P exhibited a concentration peak (> 0.10 mg/L) throughout the water column in July (Fig. 33), coincident with high chlorophyll concentrations during that period. This pattern suggested that much of the P was incorporated into algae biomass. A peak in total P was also observed near the sediment interface in late July which coincided with a period of hypolimnetic anoxia (Fig. 33), indicating P release from sediments under anoxic conditions. Total P was high and homogeneous in the water column between August and September. In contrast, SRP was near the detection limit throughout the study period (Fig. 34).

Total N exhibited a peak in concentration throughout the water column of station 10 in late June through early July, in conjunction with peaks in chlorophyll concentration (Fig. 35). In late July through August, NH_3-N concentrations increased slightly throughout the water column (Fig. 36). NH_3-N declined to near zero in September (Fig. 36).

<u>Lakewide Patterns in Water Quality.</u> Seasonal patterns in lakewide-average concentrations of P, N, and chlorophyll are shown in Fig. 37. Total P was constant near 0.05 mg/L between late April and early June and increased to > 0.10 mg/L in mid-July. Total P remained > 0.10 mg/L between mid-July and August and declined to < 0.10 mg/L in September. SRP was very low throughout the study (Fig. 37). Total N exhibited a similar peak in concentration in July, and declined steadily between mid-July and September (Fig. 37). NH_3-N was low in concentration throughout the study. With the exception of a minor peak in mid-May, chlorophyll was < 20 μg/L and nearly constant between April and mid-June (Fig. 37). Chlorophyll increased to a lakewide-average concentration of 155 μg/L in July coincident with peaks in total N and total P. Chlorophyll declined to ~ 70 μg/L between August and September (Fig. 37).

Mean summer (June-August) concentrations of total P, chlorophyll, and Secchi disk transparency values used for *Bathtub* modeling are shown in Table 10. In general, concentrations of total P(~ 0.109 mg/L) and chlorophyll (~82 μg/L) and Secchi disk transparency (~1.1 m) values were similar for stations 1 through 4. Station 5 exhibited lower total P and chlorophyll, and a higher Secchi disk transparency, than the other stations due most likely to dilution and flushing influences of groundwater inputs via the Owen Park pumping facility. Overall, the Carlson and Wisconsin trophic state indices were high and indicative of very eutrophic conditions (Table 10).

Bathtub Modeling for the summer period of 1999

Budgetary Analysis. External P loadings and discharges, calculated for the period June through August using the program *Flux*, were used as input for the model *Bathtub*. Overall, external P sources accounted for ~20%, while internal P sources accounted for ~ 80%, of the measured P input to the lake. External loading via direct P precipitation, P inflows from the Owen Park pumping facility, and P inflows from storm sewer inflows were approximately equal (Table 8). 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 8). P sources to the lake via decomposition of *P. crispus* and resuspension due to motor boat activity accounted for 20% and 17% of the measured internal P load to the lake.

Bathtub Modeling. 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, an alum 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%.

We did not include in any management scenario exploration a reduction in P inputs via the Owen Park pumping facility, because this hydrological source is important in control of pool elevation in the lake.

The models chosen for various management scenario explorations, and their coefficients, are described in Table 11. Under current limnological conditions and unmodified P inputs, the model predicted a very high summer total P (0.109 mg/L) and chlorophyll (82 μg/L) concentration, and low Secchi Disk transparency (1.1 m; Figure 38), which were exactly the same as those values observed for Half Moon Lake during the period June through August.

A 90% reduction individually in storm sewer- or motor boat- or plant-related inputs did not change predicted chlorophyll concentration appreciably (Figure 38), as concentrations were still above 68 ug/L after model manipulation. A 90% reduction in sediment P loading resulted in an ~ 36% decrease in chlorophyll to 53 μg/L, and a modest increase in Secchi Disk transparency to 1.6 m (Figure 38). Control of both sediment P loading and plant P inputs (i.e., > 40% reduction in total P load; scenario 6) resulted in an ~45% decrease in chlorophyll. Control of P resuspension via motor boat activity in combination with sediment P control (scenario 5) provided similar results as scenario 6. A 90% reduction in all measured internal P sources (scenario 7) resulted in a substantial decrease in total P and chlorophyll concentrations and an increase in Secchi disk transparency. Total P declined by nearly 60% while chlorophyll declined by > 70%. Secchi disk transparency increased nearly 3X (i.e., reached the bottom of the lake) as a function of internal loading reduction. A 90% 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, and Secchi disk transparency over those observed for scenario 7.

Overall, estimated algal bloom frequency and the concentration level of bloom was very high under current P loading (i.e., baseline) conditions (Fig. 39). The model estimated that algal blooms exhibiting chlorophyll concentrations > 30 μg/L occurred over 80% of the time during the summer period, while algal blooms exhibiting chlorophyll concentrations > 60 μg/L occurred nearly 60% of the time during the summer period. Individual reduction of P loading sources from storm sewers, motor boat activity or plant decomposition did little to improve the algal bloom frequency. With control of P loading from the sediments, algal blooms exhibiting a chlorophyll concentrations > 30 μg/L occurred 74% of the time while algal blooms exhibiting chlorophyll concentration > 60 μg/L occurred only 32% of the time. Control of both sediment P release and plant P inputs or motor boat activity (scenarios 5 and 6) resulted in a reduction in bloom frequency for chlorophyll concentrations in excess of 60 μg/L to ~14% occurrence during the summer. Control of all measured internal P loads

(Scenario 7) resulted in substantial reduction in the algal bloom frequency and severity of the bloom.

Similar to patterns observed for algal bloom frequency, TSI values exhibited only minor declines as a result of individual control of storm sewer loading, motor boat activity, or plant decomposition (Table 12). Declines in TSI to near 50 (i.e., approaching mesotrophy) occurred with control of all internal P loadings and with combined control of internal P loadings and storm sewer inputs (Table 12).

We also explored the impacts of incremental increases and decreases in all measured P loads on water quality in Half Moon Lake during the summer (Fig. 40). Fifty percent reduction in summer P loading (i.e., all measured external and internal P loads) resulted in an estimated 31% reduction in total P concentration, 42% reduction in chlorophyll, and a 60% increase in Secchi Disk transparency in the lake (Fig. 41). Conversely, an incremental increase in measured P loading to 150% of 1999 summer P loading levels resulted in a 25% increase in total P concentration, a 37% increase in chlorophyll, and a 27% decline in Secchi Disk transparency in the lake (Fig. 41). The estimated bloom frequency occurrence improved with a 50% P loading reduction and worsened with a 50% P loading increase (Fig. 42).

DISCUSSION

Hydrological inputs from the Owen Park pumping facility and outputs via net groundwater seepage appeared to play an important role in the hydrological budget of Half Moon Lake, as they each accounted for ~50% or more of the total inflow and outflow from the lake, respectively. Nevertheless, the residence time of water in the lake was ~6 months, indicating a low flushing potential for P sources stored in the lake. The high residence time was also associated with a high retention capacity for watershed-derived TSS and nutrients important for algal growth.

Precipitation-related storm sewer runoff accounted for a surprisingly small portion of the hydrological input and total P load to the lake even though rainfall in 1999 of 18.25 inches during the period April through September was only slightly lower than the average of ~ 21 inches. The sub-watersheds drained by storm sewers 2, 3, and 7b provided the greatest mass of TSS, total P, and other nutrients to the lake, which was most likely attributed to the larger areas of these sub-watersheds. However, when normalized with respect to watershed area, runoff (inches/y) was greatest from the smaller sub-watersheds 3 and 4, which contained the greatest estimated impervious areas (Barr Engineering 1992). In contrast, sub-watershed 5 had a relatively low runoff rate, even though its total and impervious areas were comparable to that of sub-watershed 2.

Internal P loadings from sediments, plant decomposition, and motor boat activity were clearly important to the P economy of the lake, as these sources accounted for 80% of the total (i.e., external + internal) measured P load to the lake during the summer. Sediment represented the greatest internal P load to the lake, accounting for 53% of the total internal P load. Rates of P release from sediments were high under anoxic conditions and comparable to those rates measured for other eutrophic systems

(Nürnberg et al. 1986). Anoxia was observed in the bottom waters at several stations during July through August, suggesting that reducing conditions in the hypolimnion were important in the regulation P release of the bottom sediments during these periods. One mechanism of P release under reducing conditions is iron-phosphorus disassociation (Mortimer 1971). Under oxidized conditions, iron has a high binding affinity for P (Lijklema 1977). However, P bound to iron hydroxides can desorb and diffuse into the sediment porewater and the water column as iron compounds are reduced from Fe^{+3} to Fe^{+2} under conditions of hypolimnetic anoxia. Asplund (1996) measured high concentrations of sediment iron and iron-aluminum bound sediment P for sediments collected in Half Moon Lake, suggesting that iron-phosphorus interactions are important and likely contribute to sediment P release in this lake.

The pH also appeared to play an important role in effecting the rate of P release from sediments under oxic conditions, as there was a strong positive and linear relationship between pH and the rate of P release under oxic conditions. Others have demonstrated this relationship between P release from sediments and pH (Boers 1991; James et al. 1996). Enhanced P release at high pH (and high hydroxyl ion (OH^-) concentration) is thought to occur via ligand exchange and replacement of PO_4^- with an OH^- ion on oxidized Fe compounds (Drake and Heaney 1987). Photosynthesis by aquatic plants and algae in Half Moon Lake, which is primarily responsible for driving pH up in aquatic systems, appeared to provide a mechanism of enhancing P release from sediments under oxic conditions.

Evidence that P sources from the sediment contributed to algal growth in Half Moon Lake is indirect based on information collected in 1999. We did not observe the occurrence of strong vertical gradients in total P or SRP near the sediment interface, which can occur as the result of sediment-water interactions in aquatic systems that are permanently stratified during the summer (James et al. 1992). SRP concentrations were also typically low in Half Moon Lake throughout the water column during the summer. This pattern may be attributed to several factors including rapid uptake of sediment-derived P by algae and nighttime destratification which would tend to dissipate any P gradients that might have developed during the day. However, during periods of daytime stratification in mid- to late July, anoxic conditions were observed in the bottom meter of the lake at nearly all the stations (see Figs. 7 and 30). Associated with these anoxic periods were modest increases in total P near the sediment interface, suggesting evidence of P loading due to sediment release. Chlorophyll concentrations were also very uniform throughout the water column during the summer indicating that algae had direct access to P sources derived from the sediment. Chlorophyll also exhibited a peak in concentration at the onset of the development of anoxia in mid-July, which coincided with the development of higher total P concentrations in the bottom waters. These trends suggested a link between sediment-derived P sources and algal growth.

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. In particular, James et al. (1992) showed that this dinoflagellate migrated as much as 4 m at night into hypolimnetic P gradients for uptake and growth in nearby Eau Galle Reservoir, Wisconsin. The ability of *C. hirundinella* to scavenge P from the bottom waters via vertical migration may explain the lack of P gradients, particularly SRP, above the sediment interface in Half Moon Lake.

P. crispus decomposition provided another important source of P to the lake, accounting for ~26% of the total internal P load during the summer. Even at a relatively low biomass level of 25 g/m^2 near the time of plant decomposition, the internal P loading rate via decomposition was high at 1.2 mg m^{-2} d^{-1}. Since it appeared that most of the P flux occurred within 2 weeks of plant senescence, based on loss of tissue P in mesh bags and leaching studies, we believe that impacts to the P budget via this mechanism were confined primarily to the month of June, when plant senescence occurred. Harvesting was important in reducing the biomass and P content level near the time of plant senescence, as it removed an estimated 16 g/m^2 biomass and 56 mg/m^2 tissue P. This removal of biomass represented ~ 30% of the overall biomass and plant P in the lake available for decomposition in early June. Had harvesting and removal of plant tissue P from the system not occurred prior to plant senescence, the predicted rate of internal P loading via plant decomposition would have been nearly 2X greater at 2.2 mg m^{-2} d^{-1}.

Impacts of motor boat activity on internal P loading to the lake were the most difficult rates to quantify, due to the complexity and variation in thermal stratification in the lake and to variations in motor boat activity as a function of rehearsals versus shows. Our estimated overall internal P loading rate via motor boat activity of ~1 mg m^{-2} d^{-1} accounted for 21% of the internal P load to the lake and 17% of the total P load to the lake during the summer. During periods of thermal stratification, the magnitude of variation in bottom temperature at the experimental versus the control station during motor boat activity appeared to be a good predictive surrogate of bottom mixing and P resuspension. In general, stronger thermal stratification prior to motor boat activity was associated with minimal P resuspension and lower bottom temperature S.D., whereas weaker stratification prior to motor boat activity was associated with a higher bottom temperature S.D. and more occurrences of P resuspension. Since thermal stratification is the result of differences in the density of water and therefore layering of warmer water (i.e., less dense) over cooler water (i.e., denser), more energy is required to mix and homogenize a water column under conditions of stronger thermal stratification than under conditions of weaker thermal stratification. Thus, although we observed extensive thermocline erosion and increases in hypolimnetic dissolved oxygen during motor boat activity on several occasions, P resuspension did not generally occur until sufficient energy to resuspend P in the form of turbulence and mixing reached the bottom of the lake just above the sediment surface. It was only during these periods of lower thermal stability that we observed a substantial increase in the S.D. of the mean bottom temperatures and increases in P in the bottom waters of the experimental station.

Although P resuspension via motor boat 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 with P bound to iron-aluminum complexes can be utilized by algae for growth. Others (Maceina and Soballe 1990; Hellström 1991; Søndergaard et al. 1992) have shown linkages between P resuspension and nutrient recycling and enhanced algal growth.

PHOSPHORUS MANAGEMENT RECOMMENDATIONS

There are several practical management strategies for reducing high summer algae by reducing P income 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 and is high under oxic conditions. Since P release from sediments, measured in the laboratory, was substantial under both oxic and anoxic conditions, an aluminum sulfate (alum) treatment to bind P and seal the sediment from further release would likely improve water quality conditions in the lake, as suggested by the *Bathtub* model response to a reduction in sediment internal P loading. It appears that Half Moon Lake would be a good candidate for an alum treatment because laboratory-based rates of P release from sediment were high and represented a dominant P source to the lake that needs to be controlled.

Alum treatments have been successful in reducing P and algal blooms for many years in other shallow systems (Cooke et al. 1993; Welch and Shrieve 1994). Welsh and Shrieve (1994) suggested that alum treatment may be more effective in reducing the internal P load available for algal growth in shallow lakes because sediment sources are usually directly available to the euphotic zone for algal uptake, due to shallower lake depth and higher P entrainment potential.

The impacts of macrophytes, which are prevalent in Half Moon Lake, on alum effectiveness needs to be considered. Dense macrophyte beds may impair efforts to spread a uniform alum layer over the sediment (Welch and Shrieve 1994). Thus, application should be restricted to a period when biomass is low (spring). Macrophytes also play an important role in the regulation of pH in the water, which may impact alum treatment effectiveness by inducing ligand exchange at high pH (Cooke et al. 1993). Although P bound to alum is insensitive to redox changes, there is some evidence that high pH may reduce the binding capacity of alum for P (Cooke et al. 1993; Welch and Shrieve 1994) In addition, since rooted macrophytes obtain P primarily from sediments (Barko and Smart 1980), P recycling via macrophyte senescence may short circuit efforts to control sediment P via an alum treatment (Welch and Shrieve 1994). More intensive macrophyte harvesting and removal of biomass (and macrophyte P) from the system would be recommended if an alum treatment were considered for Half Moon Lake.

Motor boat activity and possible resuspension of the alum floc during practices and shows needs to be considered. Loss of alum from the lake due to resuspension of the floc into the water column and discharge would likely be minor to negligible due to the high nutrient and sediment retention capacity exhibited by the lake. In addition, alum is typically denser than sediment, resulting in sinking of the alum layer up to several cm into the sediment (Welsh and Cooke 1999), and thereby increasing the turbulent energy required to resuspend the alum layer. Garrison and Knauer (1984) indicated that resuspension of the alum floc can lead to its movement from shallow to deeper regions of a lake (i.e., sediment focusing), thus diluting its effectiveness in shallow locations. An analysis of the distribution of sediment in the Half Moon Lake basin would be useful in evaluating the potential for sediment focusing and areas of sediment erosion in the lake. However, the river channel morphometry (i.e., steep banks and a fairly uniform bottom; ~ 50% of the lake's sediment area is ≥ the mean depth of 5.5 ft; Barr Engineering 1992) of the lake suggests that sediment focusing may not be as important in impacting shallow sediments as it would be in other lakes that have a greater proportion of shallow sediments. Otherwise, little is currently known about impacts of motor boat activity on alum treatment effectiveness and longevity.

If an alum treatment is considered for Half Moon Lake, we suggest that the following publications be evaluated for development of dose determinations and application techniques; Kennedy and Cooke (1982), Eberhardt (1990) Cooke et al. (1993), Rydin and Welch (1999), Welch and Cooke (1999), and James et al. (2000). In particular, Welch and Cooke (1999) suggest that treatment effectiveness and longevity is higher in lakes that were treated with an alum dosage sufficient enough to result in an Al/Fe ratio > 1 in the sediment. Effectiveness and longevity at higher Al:Fe ratios may be attributed to more complete inactivation of P associated with Fe compounds. Rydin and Welch (1999) found that a high alum dosage (Al:P ratio up to 100:1) was required to inactivate iron-bound P, which is much higher than an Al:P ratio of 1 that was used to calculate dosage for Eau Galle Reservoir (Kennedy et al. 1987). Cooke et al. (1993) evaluated many case histories and found that failures in alum treatment were usually caused by low doses, focusing of the alum layer, Al toxicity, or inadequate reductions of external nutrient loading. Although there are many options for determining alum dosage (i.e., jar test, internal P loading rate, sediment P concentration, etc: Cooke et al. 1993), we suggest that higher dosages should be considered if alum treatment is implemented as a management technique.

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. In 1999, ~ 30% of the lakewide biomass and tissue P was removed by mid-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 13).

Information on impacts of harvesting on P flux from decomposing *P. crispus* is shown for 1999 biomass levels in Table 13.

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 uniform a 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 (Table 14). This value could be evaluated with respect to the findings of the present study to determine the potential for P resuspension that evening. An example of setting criteria for reducing P resuspension by motor boat activity is shown in Figure 43. In this example, if the local thermal stability near the beach area fell below 25 g-cm/cm^2 (i.e., a defined critical threshold), practices could be canceled for that evening. Additional temperature and P data could be collected at relatively low cost to evaluate the effectiveness of the critical threshold on reducing P resuspension by motor boat activity. For instance, at randomly selected times, an integrated sample (i.e., representing the entire water column) could be collected at a control and experimental station before and after practice, etc. (i.e., 4 samples) for total P analysis and compared with the thermal structure and stability of the lake before the start of motor boat activity at those same stations . Based on additional results, the critical threshold could be renegotiated and modified by interested stakeholders at a later date. Alternatively, restrictions on the amount of motor boat activity, the seasonal timing of boat use, and/or the size and configuration of boats might be considered to reduce P resuspension in the lake.

Barr Engineering (1992) provided many practical recommendations for reducing P inputs via storm sewers using various BMP's and diversion strategies. These strategies should be considered in overall plans to reduce P loading to Half Moon Lake. Our measured external P loading rates were lower than those estimated using published export coefficients for different land use categories by Barr Engineering (1992). While disparities may be partly attributed to differences in estimated P concentrations used for loading determination (i.e., measured versus estimated concentrations) our estimates were based on seasonal results (i.e., summer months), whereas estimates presented by Barr Engineering (1992) represented annual loads. We also probably underestimated external P loading during the summer months also because we could not measure P runoff from watersheds associated with Half Moon Lake that were not drained by storm sewers and did not include this source in our calculations. In contrast, this source of P was estimated in the P loading budget of Barr Engineering (1992).

One surprising result of our study was the high P contribution from the Owen Park pumping facility. We were, unfortunately, unable to obtain accurate total P concentration information from this source in 1999 due to lake water intrusion into the storm sewer and contamination of samples. However, information collected in 2000 indicated that total P concentrations were high and ranged between 0.024 and 0.107 mg/L. Because Owen Park is an important hydrological source to the water balance of Half Moon Lake and compensates for water losses due to net seepage, eliminating this source for P control purposes would most likely lead to undesired drops in pool elevation during the summer. Since this source contributes only 7% of the measured P load to the lake, efforts would probably be better spent reducing the more substantial internal P loads first.

REFERENCES

Allen, S.E., H.M. Grimshaw, J.A. Parkinson, and C. Quarmby. 1974. Chemical analysis of ecological materials. Wiley Press, New York, NY. 565pp.

Ameel, J.J., R.P. Axler, and C.J. Owen. 1993. Persulfate digestion for determination of total nitrogen and phosphorus in low nutrient waters. Am. Environ. Lab. (October 1993) pages 8-10.

APHA (American Public Health Association). 1992. Standard methods for the examination of water and wastewater. 18th ed.

Asplund, T.R. 1996. Impacts of motor boats on water quality in Wisconsin lakes. A final report to the Bureau of Water Resources Management - Lakes Management Section. Wisconsin Department of Natural Resources, Bureau of Research, Water Resources Section, 1350 Femrite Dr., Monona, WI.

Barko, J.W., and R.M. Smart. 1980. Mobilization of sediment phosphorus by submersed freshwater macrophytes. Freshwat. Biol. 10:229-238.

Barr Engineering. 1992. Management alternatives report on the diagnostic-feasibility study of Half Moon Lake water quality problems. Rport prepared for the City of Eau Claire, WI. Procurement No. 9063. March. 1992. Barr Engineering Company, Minneapolis, MN

Boers, P.C.M. 1991. The influence of pH on phosphate release from lake sediments. Wat. Res. 25:309-311.

Borman, S. 1990. Wisconsin aquatic plant control - Reconnaissance report. Wisconsin Department of Natural Resources - Western District. Eau Claire, WI.

Brakke, D. 1995. An evaluation of thermal structure, trophic status and the potential impact of boating on nutrient concentrations in Half Moon Lake, Wisconsin. Department of Biology, University of Wisconsin at Eau Claire, Eau Claire, WI.

Carlson, R.E. 1977. A trophic state index for lakes. Limnol. Oceanogr. 22:361-369.

Cooke, G.D., E.B. Welch, S.A. Peterson, and P.R. Newroth. 1993. Restoration and management of lakes and reservoirs. 2nd edition. CRC Press, Lewis Publishers.

Drake, J.C., and S.I. Heaney. 1987. The occurrence of phosphorus and its potential remobilization in the littoral sediments of a productive English lake. Freshwat. Biol. 17:513-523.

Eberhardt, T. 1990. Alum dose calculations for lake sediment phosphorus inactivation based upon phosphorus release rates and a call for related research. Presentation, North American Lake Management Society Annual Meeting, Springfield, MA.

Garrison, P.J., and D.R. Knauer. 1984. Long-term evaluation of three alum treated lakes. p 513-517. *In:* Lake and Reservoir Management. EPA 440/5-84-001. Environmental Protection Agency.

Heaney, S.I., and T.I. Talling. 1980. Dynamic aspects of dinoflagellate distribution in a small productive lake. J. Ecol. 68:75-94.

Hellström, T. 1991. The effect of resuspension on algal production in a shallow lake. Hydrobiologia 213:183-190.

Idso, S.B. 1973. On the concept of lake stability. Limnol. Oceanogr. 18:681-683.

James, W.F., J.W. Barko, and W.D. Taylor. 1992. Production and vertical migration of *Ceratium hirundinella* in relation to phosphorus supply in a north-temperate reservoir. Hydrobiologia 215:231-241.

James, W.F., J.W. Barko, and H.L. Eakin. 1995. Internal phosphorus loading in Lake Pepin, Upper Mississippi River. J. Freshwat. Ecol. 10:269-276.

James, W.F., J.W. Barko., and S.J. Field. 1996. Phosphorus mobilization from littoral sediments of an inlet region in Lake Delavan, Wisconsin. Arch. Hydrobiol. 138:245-257.

James, W.F., J.W. Barko, H.L. Eakin, and D.R. Helsel. 2000. Distribution of sediment phosphorus pools and fluxes in relation to alum treatment. J. Am. Water Res. Assoc. 36:647-656.

Kennedy, R.H., and G.D. Cooke. 1982. Control of lake phosphorus with aluminum sulfate. Dose determination and application techniques. Wat. Res. Bull. 18:389-395.

Kennedy, R.H., J.W. Barko., W.F. James, W.D. Taylor, and G.L. Godshalk. 1987. Aluminum sulfate treatment of a eutrophic reservoir: Rationale, application methods, and preliminary results. Lake Res. Manage. 3:85-90.

Konkel, D., and S. Borman. 1996. The aquatic plant community of Half Moon Lake, Eau Claire County, Wisconsin. Wisconsin Department of Natural Resources - Western District. Eau Claire, WI.

Lijklema, L. 1977. Interaction of orthophosphate with iron (III) and aluminum hydroxides. Environ. Sci. Tech. 5:537-541.

Lillie, R.A., S. Graham, and P. Rasmussen 1993. Trophic state index equations and regional predictive equations for Wisconsin Lakes. Research Management Findings, No. 35. Bureau of Research - Wisconsin Department of Natural Resources Publication, Madison, Wisconsin.

Maceina, M. J. and D. M. Soballe. 1990. Wind-related limnological variation in Lake Okeechobee, FL. Lake and Reservoir Manage. 6: 93-100.

Mortimer, C.H. 1971. Chemical exchanges between sediments and water in the Great Lakes - speculations on probable regulatory mechanisms. Limnol. Oceanogr. 16:387-404.

Nürnberg, G.K., M. Shaw, P.J. Dillon, and D.J. McQueen. 1986. Internal phosphorus loading in an oligotrophic precambrian shield lake with an anoxic hypolimnion. Can. J. fish. Aquat. Sci. 44:960-966.

Rydin, E., and E.B. Welch. 1999. Dosing alum to Wisconsin lake sediments based on possible in vitro formation of aluminum bound phosphate. Lake Res. Manage. 15:324-331.

SAS (Statistical Analysis System). 1994. SAS/STAT User's Guide. Ver. 6. Fourth ed. SAS Institute, Cary, NC.

Søndergaard, M., P. Kristensen, and E. Jeppesen. 1992. Phosphorus release from resuspended sediment in the shallow and wind-exposed Lake Arresø, Denmark. Hydrobiologia 228:91-99.

Walker, W.W. 1996. Simplified Procedures for eutrophication assessment and prediction: User Manual. Instruction Report W-96-2. US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Welch, E.B., and G.D. Shrive. 1994. Alum effectiveness and longevity in shallow lakes. Hydrobiologia 275/276:423-431

Welch, E.B., and G.D. Cooke. 1999. Effectiveness and longevity of phosphorus inactivation with alum. Lake Reserv. Manage. 15:5-27.

Welschmeyer, N.A. 1994. Fluorometric analysis of chlorophyll a in the presence of chlorophyll b and pheopigments. Limnol. Oceanogr. 39:1985-1992.

TABLES

Table 1. (upper) Morphological characteristics of Half Moon Lake (Barr Engineering 1992). (lower) Watershed areas draining into storm sewers 1 through 7.

Morphological Characteristics		
	Metric	English
Surface Area	534590 m^2	132.1 acres
Volume	890596 m^3	722.3 acre-ft
Maximum Depth	4.3 m	14 ft
Mean Depth	1.7 m	5.6 ft

Sub-watershed	Area, m^2	Area, acres
1	153862	38
2	209871	52
3	148763	37
4	12869	3.2
5	201049	50
6	26750	6.6
7	419863	104

Table 2. Water quality variable list.

In Situ variables[1]	
	Temperature, C
	Dissolved Oxygen, mg/L
	pH
	Conductivity, S/cm2
	Secchi Transparency, cm
Water Chemistry	
	Turbidity, NTU
	Suspended Seston, mg/L
	Total phosphorus, mg/L
	Soluble reactive phosphorus, mg/L
	Total nitrogen, mg/L
	Ammonia-nitrogen, mg/L
	Viable chlorophyll a, ug/L1

[1] In-lake sampling stations only

Table 3. Average flow (cms, cubic meters per second) and runoff normalized with respect to watershed area for storm sewers monitored during the period April through September. Flow from storm sewer 7b does not include flow contributions from the groundwater pumps.

Storm Sewer	Mean Flow, cms	Runoff, inches/y
2	0.001296	7.7
3	0.003825	31.9
4	0.000095	9.2
5	0.000271	1.8
7a	0.000231	
7b	0.001949	6.5[1]

[1]Represents combined runoff into storm sewers 7a and 7b

Table 4. Hydrologic balance for Half Moon Lake during the period April through September. Values represent the mean flow expressed as cubic meters per second (cms). Included in the means storm sewer flow is an estimate of flow from unmeasured storm sewers 1 and 6. Values in parentheses represent 1 standard deviation.

	Hydrologic Balance	Mean CMS
Inflows		
	Storm Sewers	0.008 (0.019)
	Pumps	0.028 (0.011)
	Precipitation	0.016 (0.036)
Outflows		
	Outlet Structure	0.013 (0.021)
	Net Seepage[1]	0.026
	Evaporation[2]	0.012
	Storage Volume Change	0.001 (0.085)

[1]Estimated via hydrologic balance (see methods)
[2]BarrEngineering (1992)

Table 5. Summary statistics for external loads to and discharges from Half Moon Lake for the period April through September, 1999. CV represents the coefficient of variation.

Station	Seston			Total N			NH3-N			Total P			SRP		
	LOAD kg	CONC. mg L^{-1}	CV	LOAD kg	CONC. mg L^{-1}	CV	LOAD kg	CONC. mg L^{-1}	CV	LOAD kg	CONC. mg L^{-1}	CV	LOAD kg	CONC. mg L^{-1}	CV
Storm Sewer 2	4324	218	0.152	39	1.988	0.063	4.7	0.236	0.126	6.0	0.301	0.063	0.9	0.045	0.111
Storm Sewer 3	2960	51	0.135	117	2.000	0.054	19.4	0.331	0.127	8.7	0.149	0.059	2.6	0.044	0.208
Storm Sewer 4	56	385	0.139	2.6	1.815	0.087	2.6	1.823	0.083	0.2	0.132	0.167	0.1	0.040	0.196
Storm Sewer 5	243	59	0.088	7.3	1.773	0.159	1.1	0.276	0.165	0.6	0.157	0.225	0.2	0.058	0.255
Storm Sewer 7a	115	325	0.154	6.1	1.720	0.091	0.7	0.206	0.125	0.7	0.187	0.143	0.2	0.065	0.149
Storm Sewer 7b (not including G.W.)	3975	135	0.234	59.9	2.010	0.127	5.2	0.175	0.149	10.3	0.344	0.219	1.1	0.038	0.127
Groundwater Pumps	736	171	0.156	230	0.537	0.051	127	0.144	0.030	31.8	0.074	0.219	5.9	0.010	0.067
Outflow	280	14.1	0.145	107	0.536	0.167	7.9	0.040	0.054	5.1	0.026	0.249	1.2	0.006	0.130

Table 6. Export coefficients for nitrogen and phosphorus during April through September, 1999.

Storm Sewer	N Export, kg km^{-2} summer^{-1}	P Export, kg km^{-2} summer^{-1}
2	187	29
3	786	58
4	202	16
5	37	3
7*	160	25

Table 7. Measured external loads, discharges, and net retention of external loads of seston, total N, and total P for the period April through September, 1999. Net retention of external loads was calculated as the percentage of the following equation: 1-(discharge/external load). We assumed in our calculation of net retention of external loads that internal nutrient loadings did not leave the system and, thus, contribute to discharge concentration or load.

	Seston	Total N	Total P
Measured Inputs, kg	12636	473	60
Measured Discharge, kg	280	106	5
Change in Lake, kg	6075	187	27
Net Retention, %	97	78	91

Table 8. 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 3 month summer period chosen for BATHTUB modeling.

Loading Variable	Source	mg m^{-2} d^{-1}	g/d	Percent
External P Load, mg m^{-2} d^{-1}	Storm Sewers	0.3	163	5
	Pumps	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 this source (Wetzel 1975)

Table 9. A comparison of mean (± 1 S.D.) total P (TP) immediately before and after motor boat activity in the control and experimental area of Half Moon Lake. During ski show and tournament rehearsals, motor boats were confined to the experimental area (see Fig. 1) only. Thus, the control area (no motor boat activity) represented conditions with no motor boat activity. Activities designated as "no show" represented dates when motor boat activity did not take place and served as a control representing conditions with no motor boat activity. TEMP S.D. represents the standard deviation of the mean bottom temperature between 1800 and 2000 hours at stations 2C and 2E. A high TEMP S.D., as on 17 August and 1 September, reflected mixing and bottom turbulence. Stability represents the amount of work required to completely mix the water column. A higher value indicates more work. Stability was calculated for the period 1700 hours, typically immediately before the start of motor boat activity on Half Moon Lake.

Date	Activity	Experimental Station 2E				Control Station 2C			
		TP Before	TP After	Temp S.D.	Stability, g-cm/cm^2	TP Before	TP After	Temp S.D.	Stability, g-cm/cm^2
24 June	Tournament Rehearsal	0.061 (0.010)	0.061 (0.004)	No Data	No Data	0.077 (0.027)	0.075 (0.043)	No Data	No Data
7 July	Ski Show	0.082 (0.019)	0.085 (0.013)	0.09	24.4	0.085 (0.032)	0.091 (0.015)	0.15	33.7
12 July	Ski Show	0.092 (0.016)	0.107 (0.041)	0.14	38.6	0.129 (0.005)	0.103 (0.016)	0.07	20.7
15 July	Tournament Rehearsal	0.133 (0.019)	0.121 (0.006)	0.22	30.8	0.135 (0.028)	0.116 (0.008)	0.08	24.7
23 July	No Show	0.123 (0.120)	0.120 (0.064)	0.16	53.9	0.123 (0.086)	0.098 (0.030)	0.08	55.1
4 August	Ski Show	No Data (Boat Problems)							
17 August	Tournament Rehearsal	0.123 (0.015)	0.140 (0.006)[a]	0.38	16.1	0.122 (0.011)	0.115 (0.005)	0.03	31.0
28 August	No Show	0.123 (0.021)	0.117 (0.026)	0.03	44.4	0.109 (0.007)	0.117 (0.018)	0.04	43.60
1 September	Ski Show	0.102 (0.007)	0.127 (0.007)[a]	0.27	13.6	0.119 (0.013)	0.111 (0.005)	0.08	13.3
16 September	No Show	0.098 (0.005)	0.091 (0.009)	No Data	No Data	0.095 (0.002)	0.099 (0.007)	No Data	No Data

[a] Before and after total P concentrations were significantly different ($p < 0.05$; Comparison of Means; SAS 1990)

Table 10. Estimates of Carlson and Wisconsin Trophic State Index (TSI) values for Stations 1 through 5 in Half Moon Lake. Concentrations of chlorophyll *a* and total phosphorus (TP) and Secchi transparency represent means (CV) over the upper 2-m water column for the period June through August.

					Carlson TSI			WI TSI	
Station	Secchi, m	Chla, μg/L	TP, μg/L	TSI_{SD}	TSI_{chla}	TSI_{TP}	$WTSI_{SD}$	$WTSI_{chla}$	$WTSI_{TP}$
10	1.1 (0.34)	85.2 (0.24)	117 (0.15)	58.6	72.5	71.7	58.7	67.1	64.4
20	1.1 (0.33)	108.1 (0.28)	123 (0.12)	58.6	74	71.7	58.7	68.3	64.4
30	0.9 (0.24)	89.5 (0.20)	114 (0.14)	61.5	72.7	70.9	61.5	67.2	64
40	1.1 (0.29)	92.1 (0.29)	116 (0.18)	58.6	73.6	71.3	58.7	67.9	64.2
50	1.4 (0.17)	32.9 (0.24)	71 (0.14)	55.1	63.9	64.4	55.3	60.5	60.5
Lake	1.1 (0.27)	81.5 (0.24)	109 (0.14)	58.6	72.0	70.3	58.7	63.6	63.6

Table 11. Models and coefficients used in BATHTUB.

Variable	Model[1]	Coefficient
Phosphorus	2nd order decay	0.92
Chlorophyll	Jones and Bachman	1.07
Secchi Transparency	Chlorophyll and turbidity	1

Table 12. Response of the Carlson Trophic State Indices to various management scenarios. P = phosphorus, CHLA = chlorophyll, SD = Secchi Disk Transparency.

Carlson TSI	Baseline	Storm Sewers (scenario 1)	Motor Boats (scenario 2)	Plants (scenario 3)	Sediments (scenario 4)	Sediments + Motor Boats (scenario 5)	Sediments + plants (scenario 6)	All internal loads (scenario 7)	All internal loads + storm sewers (scenario 8)
TSI-P	72	71	70	70	68	65	64	59	57
TSI-CHLA	74	73	72	72	70	68	66	61	59
TSI-SD	59	57	57	56	53	50	49	43	41

Table 13. Estimated P decomposition rate for *Potamogeton crispus* as a function of standing crop at the time of plant senescence.

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

Table 14. Spreadsheet example of calculation of Schmidt stability (g-cm/cm^2) using field-collected temperature information. Critical stability values may be defined to minimize P resuspension.

FIGURE CAPTIONS

Fig. 1. Map of Half Moon Lake showing routine water sampling stations, sediment sampling stations, transects for macrophyte sampling, and water sampling stations for monitoring motor boat activity. Storm sewers 1 and 6 were not sampled and are not represented on the map.

Fig. 2. Daily precipitation and flows measured for storm sewers 2, 3, 4, 5, 7a, 7b, and groundwater pumping from the Owen Park facility. The Owen Park facility pumps groundwater into storm sewer 7b.

Fig. 3. Variations in daily precipitation, measured inflow from storm sewers, measured outflow from the outlet structure, and pool elevation for the period April through September. The horizontal line represents a nominal pool elevation of 234.49 m NGVD.

Fig. 4. Contributions of seston, total P, soluble reactive P (SRP), total N, and ammonium-N (NH_3-N) loading for various storm sewers and the Owen Park pumping facility during the period April through September.

Fig. 5. Mean (± 1 S.E.) rates of P release from sediments under anoxic conditions. Rates were determined in laboratory incubation systems at 20^{o} C.

Fig. 6. Variations in rates of P release from sediments under oxic conditions as a function of pH. Rates were determined in laboratory incubation systems at 20^{o} C.

Fig. 7. Seasonal variations in near-bottom dissolved oxygen concentration, near-bottom pH, the estimated internal P load from sediments, and the lakewide internal P load from sediments. The 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 in the lake. The lakewide rate of internal P loading from the sediment was estimated by weighting rates with respect to area for each station.

Fig. 8. Estimated kg of macrophyte P harvested from Half Moon Lake in May and June.

Fig. 9. Percent P mass remaining in decomposing *Potamogeton crispus* contained in mesh bags at station 10 in Half Moon Lake as a function of time.

Fig. 10. Rates of P leaching from *Potamogeton crispus*, measured in laboratory systems, as a function of time. Macrophyte tissue contained in mesh bags were removed from the lake at days 0, 15, and 30. Samples were incubated in laboratory systems containing filtered lake water at a constant temperature. Changes in soluble reactive P through time in the systems were used to determine rates of P leaching.

Fig. 11. Variations in *Potamogeton crispus* decomposition via leaching and breakdown. Leaching is defined as autolysis of cellular contents into the water column while breakdown refers to microbial decomposition and fragmentation.

Fig. 12. Vertical variations in water temperature before (black circles) and after (grey circles) motor boat activity at stations 1E (experimental station), 2E, and 3E and 1C (control station), 2C, and 3C on 17 August.

Fig. 13. Vertical variations in dissolved oxygen before (black circles) and after (grey circles) motor boat activity at stations 1E (experimental station), 2E, and 3E and 1C (control station), 2C, and 3C on 17 August.

Fig. 14. Vertical variations in turbidity before and after motor boat activity at stations 2E (experimental station) and 2C (control station) on 17 August. See Fig. 13 for sampling times at these stations.

Fig. 15. Vertical variations in total P before and after motor boat activity at stations 2E (experimental station) and 2C (control station) on 17 August. See Fig. 13 for sampling times at these stations.

Fig. 16. Vertical variations in water temperature before (black circles) and after (grey circles) motor boat activity at stations 1E (experimental station), 2E, and 3E and 1C (control station), 2C, and 3C on 15 July.

Fig. 17. Vertical variations in dissolved oxygen before (black circles) and after (grey circles) motor boat activity at stations 1E (experimental station), 2E, and 3E and 1C (control station), 2C, and 3C on 15 July.

Fig. 18. Vertical variations in turbidity before and after motor boat activity at stations 2E (experimental station) and 2C (control station) on 15 July. See Fig. 16 for sampling times at these stations.

Fig. 19. Vertical variations in total P before and after motor boat activity at stations 2E (experimental station) and 2C (control station) on 15 July. See Fig. 16 for sampling times at these stations.

Fig. 20. Vertical variations in water temperature before (black circles) and after (grey circles) motor boat activity at stations 1E (experimental station), 2E, and 3E and 1C (control station), 2C, and 3C on 28 August.

Fig. 21. Vertical variations in dissolved oxygen before (black circles) and after (grey circles) motor boat activity at stations 1E (experimental station), 2E, and 3E and 1C (control station), 2C, and 3C on 28 August.

Fig. 22. Vertical variations in turbidity before and after motor boat activity at stations 2E (experimental station) and 2C (control station) on 28 August. See Fig. 20 for sampling times at these stations.

Fig. 23. Vertical variations in total P before and after motor boat activity at stations 2E (experimental station) and 2C (control station) on 28 August. See Fig. 20 for sampling times at these stations.

Fig. 24. Variations in water temperature in the control (station 2C) and experimental (station 2E) station at different depths as a function of time on 17 August.

Fig. 25. Variations in water temperature in the control (station 2C) and experimental (station 2E) station at different depths as a function of time on 115 July.

Fig. 26. Variations in water temperature in the control (station 2C) and experimental (station 2E) station at different depths as a function of time on 28 August.

Fig. 27. Variations in the standard deviation (SD) of the mean bottom temperature for water temperature measurements collected between 1800 and 2000 hours. Water temperature was measured at the control (station 2C) and experimental (station 2E) station at the 2.5-m depth using thermistors and a data logger. The horizontal black line represents the estimated threshold SD above which P resuspension occurs. P resuspension on 17 August and 1 September were used to define the critical SD. 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.

Fig. 28. 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.

Fig. 29. Contour plot of seasonal and vertical variations in water temperature at station 10 in Half Moon Lake.

Fig. 30. Contour plot of seasonal and vertical variations in dissolved oxygen at station 10 in Half Moon Lake. The shaded area represents periods when dissolved oxygen concentrations were < 2 mg/L.

Fig. 31. Contour plot of seasonal and vertical variations in pH at station 10 in Half Moon Lake.

Fig. 32. Contour plot of seasonal and vertical variations in viable chlorophyll *a* at station 10 in Half Moon Lake.

Fig. 33. Contour plot of seasonal and vertical variations in total P at station 10 in Half Moon Lake.

Fig. 34. Contour plot of seasonal and vertical variations in soluble reactive phosphorus at station 10 in Half Moon Lake.

Fig. 35. Contour plot of seasonal and vertical variations in total nitrogen at station 10 in Half Moon Lake.

Fig. 36. Contour plot of seasonal and vertical variations in ammonium-N at station 10 in Half Moon Lake.

Fig. 37. Seasonal variations in phosphorus, nitrogen, and chlorophyll in Half Moon Lake. Concentrations were weighted with respect to depth and area of the lake.

Fig. 38. Estimated changes in total phosphorus (P), chlorophyll, 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.

Fig. 39. Estimated changes in the frequency of algal bloom occurrence of different concentrations of chlorophyll 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.

Fig. 40. Incremental increases and decreases in P loading (i.e., the sum of all measured P loads to Half Moon Lake) as a function of the percentage of current P loadings during the summer (June-August, 1999). 100% represents 1999 summer loading conditions.

Fig. 41. Estimated changes in total phosphorus (P), chlorophyll, and Secchi transparency as a function of incremental increases and decreases in P loading (i.e., the sum of all measured P loads to Half Moon Lake) . 100% represents 1999 summer loading conditions.

Fig. 42. Estimated changes in the frequency of algal bloom occurrence of different concentrations of chlorophyll as a function of incremental increases and decreases in P loading (i.e., the sum of all measured P loads to Half Moon Lake) . 100% represents 1999 summer loading conditions.

Fig. 43. Variations in the standard deviation (SD) of the mean bottom temperature for water temperature measurements collected between 1800 and 2000 hours. Water temperature was measured at the control (station 2C) and experimental (station 2E) station at the 2.5-m depth using thermistors and a data logger. The horizontal black line represents the estimated threshold SD above which P resuspension occurs (see Fig. 27). The shaded are represents an example of setting criteria for minimizing P resuspension via motor boat activity. In this case, when Schmidt stability < 25 g-cm/cm^2, the potential for P resuspension is high and practice should be canceled.

APPENDICES

A1. Table of acronyms

Acronym	Description
P	Phosphorus
SRP	Soluble Reactive Phosphorus
N	Nitrogen
NH_3-N	Ammonium-Nitrogen
cms	Cubic Meters per Second

A2. Table of conversion factors

1 Inch	= 0.0254 m (meter)
1 Foot	= 0.3048 m
1 Mile	= 1.609 km
1 Acre	= 4046.86 m^2
1 acre-foot	= 1233.489 m^3
1 cfs (cubic feet per second)	= 0.028317 cms (cubic meters per second)
1 Pound	= 0.4536 kg (kilograms)
1 Ton	=1016.05 kg