Phosphorus Budget and Loading Reduction Analysis of Mead Lake, West-Central Wisconsin

Aerial digital photograph of Mead Lake courtesy of the U.S. Geological Survey on-line map acquisition webpage (http://geography.usgs.gov/partners/viewonline.html.)

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PREFACE

This research was conducted in response to a request from 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 WI-DNR and USAED, St. Paul. The study coordinator for WI-DNR was Mr. Patrick Sorge and the Section 22 coordinator for the USAED, St. Paul, was Mr. Roland O. Hamborg.

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ABSTRACT

Suspended solids and tributary nutrient (nitrogen and phosphorus) loading, internal P fluxes from aquatic sediment, and water quality conditions were examined for Mead Lake, Wisconsin, in 2002-2004. Mean annual and summer flows and constituent loads for the South Fork of the Eau Claire River were greater in 2002 versus 2003, reflecting variations in annual and summer precipitation patterns. Mean total P concentrations for the Eau Claire River ranged between 0.115 and 0.123 mg/L and soluble reactive P accounted for 51% to 54% of the total P load to Mead Lake. Total P export for the Eau Claire River watershed was moderate compared to Wisconsin watersheds and values fell within the range observed for watersheds exhibiting greater than 50% agricultural land use.

Laboratory-derived rates of internal P loading from profundal sediments were very high under anoxic conditions (range = 16 to 38 mg m^{-2} d^{-1}) suggesting potential for substantial P flux from bottom sediments. Rates of P release from sediments under oxic conditions were less than 1 mg m^{-2} d^{-1}. However, they varied linearly as a function of pH (Rate = 0.5921pH – 4.3859; r^2 = 0.67) over a pH range of 7.6 to 8.6, indicating that modest increases in rates of P release could occur at higher pH values induced by aquatic plant and algal photosynthesis.

During the summer of 2002, the average water residence time for Mead Lake was only 10.5 days due to frequent precipitation and elevated flow. In 2003, the water residence time was nearly 18 days during the summer as a result of extended periods of minimal precipitation and lower flows. Stratified conditions were observed at the deepest sampling station established in the lake during the summer months of both years. However, storms and high inflow resulted in partial destratification in August, 2002. Anoxia developed within the hypolimnion of the lake by early June in conjunction with stratified conditions during both summers. Hypolimnetic anoxia was generally confined to between 2.5 m and the bottom of the lake. Total P concentrations in the bottom waters increased markedly in 2003 in conjunction with higher residence time and lower flushing, versus 2002. Concentrations of total P in the surface waters were greater than

0.10 mg L^{-1}; soluble reactive P accounted for 28% to 40% of the total P, indicating abundant readily available P for algal uptake and growth.

Chlorophyll concentrations were high during both summers, averaging 51 mg m^{-3} in 2002 and 76 mg m^{-3} in 2003. Higher concentrations in 2003 were probably due to higher water residence time. Conversely, frequent storm inflows in 2002 reduced water residence time and promoted flushing of algal cells from the lake. Peaks in chlorophyll concentrations were generally associated with peaks in total P, suggesting incorporation of total P in the surface waters as algal biomass. Mean values for total P, chlorophyll, and Secchi transparency for the surface waters of Mead Lake during the summers of 2002 and 2003 translated into very high TSI values (i.e., > 60), indicating eutrophic conditions.

Overall, tributary P loading provided the largest input to Mead Lake, accounting for 87% and 78% of the measured P load in 2002 and 2003, respectively. In contrast, internal P loading from sediment accounted for 12.5% in 2002 and 21.5% in 2003 of the measured P input. Differences in the relative contributions of tributary versus internal P loads were due primarily to precipitation and runoff, as lake-wide internal P loading was nearly constant for both summer periods (3.0 to 3.5 mg m^{-2} d^{-1}).

Simulated decreases in tributary P loading from the Eau Claire River and the ungauged portion of the watershed, using the computer model *Bathtub*, resulted in predicted decreases in the average summer concentration of total P and chlorophyll of the surface waters and increases in Secchi transparency for both years. A 50% reduction in summer tributary P loading resulted in a predicted 40% and 37% decrease in total P and a 52% and 49% decrease in chlorophyll concentrations in 2002 and 2003, respectively. Simulated total P loading increases due, for instance, to increased P runoff in the watershed, resulted in substantial predicted increases in total P and chlorophyll concentrations and decreases in Secchi transparency in Mead Lake. *Bathtub* modeling also suggested lower nuisance algal bloom occurrence with simulated tributary P loading reductions. *Bathtub* P loading reduction simulations suggested that P management for water quality improvement should focus on reductions in tributary P loading, as this source accounts for the majority of the P budget of Mead Lake.

INTRODUCTION

Eutrophication of receiving waters is strongly linked to the erosion and transport of particulate and soluble nutrients (primarily phosphorus) derived from the watershed landscape. In agriculturally-managed watersheds, amendment of soils with fertilizers and manure is usually based on crop nitrogen (N) requirements rather than phosphorus (P) to obtain optimal yield. This practice can result in the buildup of soil P levels to excessive concentrations that can be transported to receiving waters during storms via runoff (Lemunyon and Gilbert 1993; Sharpley et al. 1994; Sharpley 1995). Reservoirs are particularly susceptible to accelerated eutrophication rates due to excessive loading by large tributary inflows draining sizable watersheds. Watershed P inputs can contribute directly to algal productivity in soluble form and indirectly in particulate form through sedimentation of adsorbed P and later recycling via eH and pH reactions at the sediment-water interface (James et al. 1995, 1996). The objectives of this research were to examine water quality conditions and algal growth in relation to fluxes of P from tributary inflows and bottom sediments, and to simulate water quality response to reductions (or increases) in tributary P loading for Mead Lake, Wisconsin.

SITE DESCRIPTION AND METHODS

Watershed and lake characteristics

Mead Lake is a eutrophic (Trophic State Index for Secchi transparency is 63) impoundment located in Clark County, Wisconsin (Figure 1). The lake has a surface area of 1.3 km^2, a volume of 1.9 hm^3, and mean and maximum depths of 1.5 m and 5 m, respectively. The Osgood index (Osgood 1988) for the lake, calculated as the mean depth (m) divided by the square root of the lake's surface area (km2; $z/\sqrt{Ao}$) is 1.3. Values less than 6 are often associated with lakes exhibiting intermittent mixing during the summer and potential for vertical transport of internally recycled P from bottom sediments. Values greater than 6 are usually associated with strongly stratified lakes that exhibit minimal vertical P transport to the lake surface.

The major tributary inflow to the lake is the South Fork of the Eau Claire River, which drains a predominantly agricultural watershed (Figure 2). Row crop production agriculture and grass pasture account for 54% and 12%, respectively, of the land use coverage in the watershed (Figure 3 and Table 1). Forested land use areas occupy 23% of the watershed.

Precipitation, flow monitoring, and tributary loading determinations

Precipitation measurements were obtained from the National Weather Service (Chanhassen, Minnesota) for nearby Eau Claire, Wisconsin. A tributary flow monitoring station was established on the South Fork of the Eau Claire River at County MM in Clark County, Wisconsin. Stage elevations of the river were monitored at 15-minute intervals using a stage height recorder equipped with a pressure transducer (ISCO Model 4120, ISCO Incorporated, Lincoln, Nebraska). Mean daily stage elevations were converted to volumetric flow (cubic meters per second; cms) using stage-discharge relationships generated under different flow regimes. Flow gauging was conducted between April, 2002, and February, 2004.

Grab samples were collected from mid-stream at biweekly intervals using an integrating water column sampler (Model 5200 or 5250; Scientific Instruments). Samples were analyzed for total suspended solids (TSS), total N, total P, and soluble reactive P. For TSS, suspended material retained on a precombusted glass fiber filter (Gelman (A/E) was dried to a constant weight at 105 ^{o}C (APHA 1998). Samples for total N and P were predigested with potassium persulfate according to Ameel et al. (1993) before analysis. Water samples for analysis of soluble reactive P were filtered through a 0.45 µm filter (Millipore MF). N and P analyses were determined colorimetrically on a Lachat QuikChem automated water chemistry system (Hach Company, Loveland, CO). Annual and seasonal (i.e., May through September) loadings were estimated using the computer model *Flux* (Walker 1996). P export coefficients were used to estimate P loadings for ungauged portions of the watershed (i.e., Rocky Run and the watershed area below County MM, which represented 22% of the total watershed area). P export coefficients published in Shultz et al. (1996), Brakke (1997), Corsi et al. (1997), and Robertson et al. (2002) for row crops (0.61 kg/ha), forest (0.10 kg/ha), grassland (0.30 kg/ha), and wetlands (0.30 kg/ha) were weighted with respect to different land uses in these ungauged areas in order to estimate P loadings.

Internal phosphorus loading from lake sediments

In July, 2002, nine replicate sediment cores were collected at stations 10, 15, and 20 for determination of rates of P release from the sediment (Figure 1). For each station, three sediment cores were analyzed for P release under oxic conditions at pH ~7.5, another three cores were analyzed for P release under oxic conditions at pH ~ 8.5, and a final set of three cores were analyzed for P release under anoxic conditions.

A Wildco KB sediment core sampler (Wildco Wildlife Supply Co.), equipped with an acrylic core liner (6.5-cm ID and 50-cm length), was used to collect sediment. The core liners, containing both sediment and overlying water, were immediately sealed using stoppers and stored in a protective box until analysis. Additional lake water was collected for incubation with the collected sediment.

In the laboratory, sediment cores were carefully drained of overlying water and the upper 10 cm of sediment were transferred intact to a smaller acrylic core liner (6.5-cm dia and 20-cm ht) using a core remover tool. Lake water was filtered through a glass fiber filter (Gelman A-E), and 300 mL was siphoned onto the sediment contained in the small acrylic core liner without causing sediment resuspension. Sediment incubation systems consisted of the upper 10-cm of sediment and filtered overlying water contained in acrylic core liners that are sealed with rubber stoppers. The sediment incubation systems were placed in a darkened environmental chamber and incubated at a constant temperature (20 ^{o}C). The incubation temperature was set to a standard temperature for all stations for comparative purposes.

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. Higher pH was adjusted by bubbling sediment core systems with carbon dioxide-free air. Bubbling action ensured complete mixing of the water column but did not disrupt the sediment. Control systems, containing only lake water, were incubated under the same redox and pH conditions.

Water samples for soluble reactive phosphorus (SRP) were collected daily from the center of each sediment incubation system using an acid-washed syringe and filtered through a 0.45 µm membrane syringe filter (Millipore Millex HV). The water volume removed from each system during sampling was replaced by addition of filtered lake water preadjusted to the proper oxidation-reduction condition. SRP was determined colorimetrically using the ascorbic acid method (APHA 1998). Rates of phosphorus release from the sediment (mg m^{-2} d^{-1}) were calculated as the linear change in concentration (corrected for dilution effects) in the overlying water divided by time and the area of the incubation core liner.

Lake monitoring

Sampling stations were established in the deep basin near the dam (station 10; maximum depth = ~ 4.5 m), a shallow littoral area near the north shore (station 15; maximum depth = ~ 0.75 m), and in the shallow upstream portion of the lake (station 20; maximum depth = ~ 2 m) for biweekly sampling during the ice-free period of 2002 and 2003 (Figure 1). In situ profiles of temperature, dissolved oxygen, pH, and conductivity were collected at ~1-m intervals at each station using a Hydrolab Quanta monitor calibrated against known standards. Secchi disk transparency was measured using an alternating black and white, 10-cm, disk. Water samples were collected at 1-m depth intervals for analysis of total N and P, soluble reactive P, and viable chlorophyll. Chlorophyll was determined via a fluorometric technique following extraction in a 1:1 solution of acetone and dimethyl sulfoxide (Welschmeyer 1994).

Schmidt stability (kg-m m^{-2} s^{-2}), or 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, was calculated according to Idso (1973). Higher stability values were indicative of strong stratification and greater work required to disrupt stratification. Conversely, lower stability values were indicative of weak stratification and less work required disrupting stratification and promoting mixing.

The Carlson Trophic State Index (TSI; Carlson 1977) was estimated for Mead Lake using the computer program *Bathtub* (Walker 1996). Mean Secchi transparency values and mean surface concentrations of total P and chlorophyll estimated over the period May through September of both years were used in the calculation. In addition, the Wisconsin TSI was estimated using equations described in Lillie et al. (1993).

A monitoring station was established near station 15 for examination of diel and seasonal variations in dissolved oxygen and pH in the littoral region of the lake that might impact rates of P release from shallow sediments. YSI 6000 in situ water quality monitors (YSI Incorporated, Yellow Springs, Ohio) were deployed 0.25 m above the sediment and collected measurements at 4-hour intervals between May and September of 2002 (monitoring did not occur in 2003). Deployed monitors were exchanged every two weeks with a new, calibrated monitor. Both before and after deployment, the monitors were pre- and post-calibrated using Winkler titrations and pH buffers (APHA 1998). Before each deployment, the monitors and probes were cleaned and dissolved oxygen probes were serviced with new membranes.

Trophic response modeling and loading reduction scenarios

The computer model *Bathtub* (Walker 1996) was used as a management tool to forecast the trophic response of Mead Lake to reductions and increases in tributary P loading. Surface concentrations of chlorophyll and total P averaged for the three stations over the period May through September of 2002 and 2003 were used as summer conditions for input into the model. Since hydrological and loading conditions for 2002 and 2003 were not similar, calibration coefficients were developed for each year to examine trophic response. Tributary P loading for ungauged regions of the watershed (i.e., downstream of County MM) were estimated indirectly using published P export coefficients. Tributary P loadings were increased and decreased incrementally by nearly 100% of P loading conditions observed in 2002 and 2003 to examine total P, chlorophyll, and Secchi transparency response for Mead Lake.

RESULTS AND DISCUSSION

Tributary flow and loading patterns

Annual and summer precipitation contrasted markedly over the study period; it was above average (32.12 inches for Eau Claire, Wisconsin) in 2002 and below average in 2003 (Table 2). Mean annual and summer flows and constituent loads for the Eau Claire River were greater in 2002 versus 2003, reflecting variations in annual and summer precipitation patterns. The Eau Claire River exhibited elevated flows during storms or snowmelt over most of the ice-free period in 2002 (Figure 4). The greatest flows in 2002 occurred in conjunction with storms in mid-August. In contrast, flows were much lower during the summer of 2003 due to low precipitation and an extended period of dry conditions between mid-June and ice formation in November.

Flow-weighted total P concentrations for the Eau Claire River ranged between 0.115 mg L^{-1} and 0.123 mg L^{-1} and soluble reactive P accounted for 51% to 54% of the total P load (Table 2). Overall, total P export for the Eau Claire River watershed was moderate compared to southern and southeastern Wisconsin watersheds and values fell within the range observed for watersheds exhibiting greater than 50% agricultural land use (Table 3; Panuska and Lille 1995). On an annual basis, between 30 and 50% of the tributary P load was retained in the lake.

Internal phosphorus loading from lake sediments

Laboratory-derived rates of internal P loading from profundal sediments for various stations in Mead Lake are shown in Table 4. Under laboratory conditions of anoxia, rates of P release were very high for all stations (> 5 mg m^{-2} d^{-1} is considered eutrophic; Nürnberg 1987; Effler 1996), suggesting the potential for substantial P flux from bottom sediments. In contrast, rates of P release from sediments under oxic conditions were less than 1 mg m^{-2} d^{-1}. Rates of sediment P release under oxic conditions varied linearly as a function of pH (Rate = 0.5921pH – 4.3859; r^2 = 0.67) over a pH range of 7.6 to 8.6. This pattern indicated that increases in rates of P release could occur, particularly in littoral sediments, at higher pH values induced by aquatic plant and algal photosynthesis.

Lake water quality

The theoretical water residence time (i.e., the length of time it takes to completely refill the lake) of Mead Lake varied in conjunction with differences in precipitation and flow patterns over the two year period. During the summer of 2002, the average water residence time was only 10.5 days due to frequent precipitation and elevated inflow. In 2003, the average water residence time was nearly 18 days during the summer as a result of extended periods of minimal precipitation and lower inflows.

Stratified conditions were observed at station 10 during the summer months of both years (Figure 5). However, storms and high inflows resulted in partial destratification in August, 2002. During the lower summer flow conditions of 2003, stratification was observed between early June and late August. Lake stability values were generally low (< 15 kg-m m^{-2} s^{-2}) during both summers, suggesting potential for mixing and disruption of stratification (Figure 6), particularly during periods of strong wind activity. The storm inflow in August, 2002, was associated with a decrease in lake stability, suggesting potential mixing. The average metalimnetic depth during both summers was ~ 1.2 m to 1.5 m. Autumnal overturn occurred in late September of both years.

Anoxia developed within the hypolimnion of the lake by early June in conjunction with stratified conditions during both summers at station 10 (Figure 7). Hypolimnetic anoxia was generally confined to between 2.5 m and the bottom of the lake. During August, 2002, some reoxygenation of the upper hypolimnion occurred due to high inflows and disruption of stratification. Hypolimnetic anoxia occurred at station 10 for ~ 100 days each summer. The anoxic factor (i.e., the number of days per summer that a sediment area equivalent to the lake's surface area is anoxic; Nürnberg 1995) was moderate at 11 d in 2002 and 6 d in 2003.

Like temperature and dissolved oxygen, pH at station 10 exhibited vertical stratification during the summer (Figure 8). It approached 9.5 in the surface waters in conjunction with net photosynthesis by algae and aquatic plants located in the euphotic zone. It was near 7 in the bottom waters due to bacterial respiration.

At the shallow littoral station 15, dissolved oxygen and pH exhibited diel fluctuations near the lake bottom throughout the summer of 2002 (Figure 9). In general, both variables increased in magnitude during the day due to net photosynthesis and declined at night due to respiration. Oxygenated conditions were observed in the bottom waters of this station between May and July. Periods of anoxia (D.O. < 1 mg L^{-1}) in the bottom waters of station 15 occurred at night on several dates in August (i.e., ~ 13% of the time in August). pH increased steadily from mid-June through mid-July and fluctuated between 7 and > 9 from mid-July through August, 2002.

In 2002, chlorophyll concentrations (Figure 10) and lake-wide chlorophyll mass (Figure 11) exhibited peaks in late July through early August and in late September. These maxima were confined to the upper 2 m of the water column and concentrations exceeded 100 mg m^{-3} during the first peak and 40 mg m^{-3} during the second peak in 2002. Chlorophyll concentrations and mass declined in mid August, 2002, in conjunction with a period of high storm inflow and low water residence time, suggesting removal and dilution of algae via flushing and washout during that period. A sustained algal bloom that exceeded a chlorophyll concentration of 100 mg m^{-3} occurred between late July and early September of 2003.

Elevated total P concentrations in the bottom waters of station 10 (Figure 12) and modest increases in lake-wide total P mass (Figure 11) were observed in early June, 2002, that coincided with the development of hypolimnetic anoxia. Total P declined abruptly in the bottom waters and concentrations were nearly uniform throughout the water column in August, 2002, due to the large storm inflow that rapidly flushed the lake. In 2003, accumulation of total P in the anoxic bottom waters was much greater throughout the summer and concentrations exceeded 1 mg P L^{-1} above the sediment interface in late July and early August, suggesting internal P loading from bottom sediments. Greater total P mass in the hypolimnion in 2003 versus 2002 coincided with lower inflows from the Eau Claire River and greater hydraulic residence time. During both years, total P mass was greater in the epilimnion than in the hypolimnion, and concentrations often exceeded 0.10 mg L^{-1} in the surface waters. Elevated lake-wide total P mass and concentrations in the epilimnion during August, 2002, and during August through early September, 2003, were associated peak lake-wide chlorophyll mass and concentrations, suggesting incorporation of total P in the surface waters as algal biomass. High total P in the surface waters was also observed during early September of both years which coincided with the onset of epilimnetic cooling and autumnal overturn, suggesting probable vertical entrainment of total P. Overall, soluble reactive P accounted for 40% (± 9.8 S.E.) and 28% (± 3.7 S.E.) of the lake-wide total P mass during the summer of 2002 and 2003, respectively. Higher soluble reactive phosphorus in 2002 was probably related to greater precipitation, runoff, and tributary P loading from the Eau Claire River.

Secchi transparency exceeded 1.5 m in June, 2002, then declined and remained nearly constant at ~ 0.5 m from July through September (Figure 13). In 2003, Secchi transparency was greater than 1 m in June. It declined steadily between late June and September in conjunction with increases in chlorophyll in the surface waters.

Mean values for total P, chlorophyll, and Secchi transparency for the surface waters of Mead Lake during the summers of 2002 and 2003 translated into very high TSI values (Table 5). They were greater than 60 on both the Carlson and Wisconsin TSI, indicating eutrophic conditions.

Summer phosphorus budget and loading reduction scenarios

From the above observations, a summer P budget was constructed for 2002 and 2003 (Table 6). Inputs to the lake included phosphorus from direct precipitation over the lake ($P_{precipitation}$), gauged and ungauged tributary P loading ($P_{tributaries}$), and phosphorus release from sediments and movement into the hypolimnetic water column ($P_{sediment}$). The measured output from the lake was phosphorus discharge from the outlet structure ($P_{discharge}$). Groundwater P fluxes into and out of the lake were not measured and could not be included in the P budget. Using a mass balance approach, sedimentation of particulate phosphorus ($P_{deposition}$) in the lake was calculated via substraction after substitution of known P sources into the equation:

$$\Delta P_{lake} = (P_{precipitation} + P_{tributaries} + P_{sediment}) - (P_{deposition} + P_{discharge}) \qquad 1)$$

where ΔP_{lake} was the change in P mass in the water column. All fluxes were normalized with respect to a 153 day summer period (i.e., May through September) and the surface area of the lake.

$P_{sediment}$ was calculated using rates derived from laboratory incubations under differing conditions of redox and pH and vertical profiles of dissolved oxygen and pH measured at 1-m intervals at station 10. Profile data obtained from this station were extrapolated to the sides of the basin to estimate conditions above the sediment. Anoxic conditions were assumed to occur when dissolved oxygen decreased below 1 mg L^{-1}. Laboratory-derived rates of P release, adjusted for redox and pH conditions from in situ profiles, were weighted with respect to sediment area at each depth stratum and summed in order to estimate a lake-wide $P_{sediment}$ rate. An independently-derived estimate of $P_{sediment}$ was determined via budgetary analysis for the summer of 2003, when marked total P accumulation occurred in the hypolimnion (Figure 11f), for comparision with the laboratory estimate of $P_{sediment}$. The mass balance estimate of 3.2 mg m^{-2} d^{-1} was very similar to the laboratory-derived estimate of 3.5 mg m^{-2} d^{-1} determined for the summer of 2003.

Overall, $P_{tributaries}$ provided the largest summer input to Mead Lake, accounting for 87% and 78% of the measured P load in 2002 and 2003, respectively (Table 6). In contrast, $P_{sediment}$ accounted for 12.5% in 2002 and 21.5% in 2003 of the measured P input. Differences in the relative contributions of tributary versus internal P loads were due primarily to precipitation and runoff, as $P_{sediment}$ was nearly constant for both summer periods.

Littoral (i.e., shallow, shoreline regions) P flux from the shallow sediments (i.e., sediments occupying the upper 1-m of the water column), was estimated using laboratory-derived rates of P release from sediments under differing redox and pH conditions and in situ diel variations in dissolved oxygen and pH measured near the sediment interface at station 15 in 2002. As with $P_{sediment}$, anoxic conditions were assumed to occur when dissolved oxygen fell below 1 mg L^{-1}. Mean littoral rates of sediment P flux over the summer period were high at 1 mg m^{-2} d^{-1} (± 0.2 S.E.). When normalized for the ratio of lake surface area to sediment surface area encompassing the 1-m depth, it was 0.6 mg m^{-2} d^{-1} (i.e., 1 mg m^{-2} d^{-1} · 0.61).

Bathtub models used for the prediction of total P, chlorophyll, and Secchi transparency in Mead Lake under differing tributary P loading scenarios are shown in Table 7. Model coefficients were calibrated against data collected during the summers of 2002 and 2003. In general, P model coefficients were usually near 1.0 but fell within the range of 0.5 to 2.5 found for reservoirs in the United States (Walker 1996). Model coefficients for chlorophyll were usually less than 1.0 for the lakes examined using *Bathtub*. This pattern may be attributed to differences in the biological availability of P loads in the Mead Lake watershed versus model projections based on other reservoirs.

Simulated Mead Lake water quality responses to variations in summer tributary P loading in 2002 and 2003 are shown in Figures 14 and 15. Simulated decreases in tributary P loading from the Eau Claire River and ungauged runoff resulted in predicted decreases in the average summer concentration of total P and chlorophyll of the surface waters and increases in Secchi transparency for both years. For instance, a 50% reduction in summer tributary P loading resulted in a predicted 40% and 37% decrease in total P and a 52% and 49% decrease in chlorophyll concentrations in 2002 and 2003, respectively. In contrast, simulated total P loading increases due, for instance, to increased P runoff in the watershed, resulted in substantial predicted increases in total P and chlorophyll concentrations and decreases in Secchi transparency in Mead Lake. A simulated increase in external P loading of 150% over P loading conditions observed in 2002 and 2003 resulted in a 34% and 31% increase in total P and a 54% and 49% increase in chlorophyll, respectively.

Bathtub modeling was also used to examine changes in the bloom frequency of algal populations in the lake under conditions of simulated reductions or increases in external P loading during both summer periods (Figures 16 and 17). Under tributary P loading conditions observed in 2002 and 2003, the model suggested that nuisance blooms on the order of 30 mg m^{-3} (i.e., visible to the eye and considered an aesthetic problem) occurred between 71% and 88% of the time during the summer. A simulated reduction in tributary P loading over 2002 conditions by 50% resulted in a predicted decrease in the 30 mg m^{-3} chlorophyll bloom frequency to only ~26% of the time in the summer. Conversely, a simulated 50% increase in external P loading over 2002 conditions resulted in a predicted 30 mg m^{-3} chlorophyll bloom frequency of ~89% of the time during the summer. For tributary loading conditions in 2003, modeled bloom frequency response was lesser in magnitude than for 2002. For instance, the 30 mg/m^3 bloom frequency occurrence was much higher at 54% of the time in 2003, even with a 50% reduction in tributary P loading. Differences between the 2 summers in bloom frequency occurrence response to decreases in tributary P loading may reflect differences in the lake hydraulic residence time and the relatively greater importance of internal P loading to the lake under lower flow conditions.

CONCLUSIONS

Mead Lake is currently very eutrophic and exhibits high concentrations of P and chlorophyll in the surface waters during the summer months. Because it impounds a large portion of the agriculturally-dominated South Fork of the Eau Claire River watershed, it receives substantial P loads that overwhelmingly contribute to poor water quality conditions. P export rates are comparable to other Wisconsin watersheds dominated by agricultural land use and approximately 50% of this load is in the form of soluble reactive P, which can be directly used by algae for growth. Concentrations of soluble reactive P are high in the surface waters of the lake during the summer, reflecting high tributary loadings of this form of P.

Sediments deposited in Mead Lake can contribute P to the water column via recycling under conditions of anoxia or high pH. Under anoxic conditions, P associated with metal hydroxides is readily desorbed and diffuses into the sediment porewater and overlying water where it can accumulate within the hypolimnion (Mortimer 1971). Under oxygenated conditions, high pH in the water column induced by net photosynthesis can enhance desorption of P from sediments through ligand exchange (i.e., competition of hydroxyl ions with P for binding sites on sediments; Drake and Heaney 1987). Lake-wide rates of internal P loading for Mead Lake were high and indicative of eutrophic conditions. Accumulated P in the bottom waters over the summer, as in 2003, may become a source of P to algal growth in the surface waters through vertical transport mechanisms. Wind-induced mixing can promote the upward transport of P through vertical eddy diffusion and entrainment of P-rich hypolimnetic water (Stauffer and Lee 1973; Imoden and Emerson 1978). Motile algae (i.e., algae with flagella or buoyancy regulation capabilities) can access P from the hypolimnion directly for growth (James et al. 1992). The importance of vertical P transport to the P budget of the surface waters is difficult to assess and currently not known for Mead Lake. However, other studies have suggested that 10-30% of the internally-derived P can become transported to the surface via mechanisms described above (Auer et al. 1993).

Water residence time, in addition to P inputs, appeared to play a role in the regulation of algal concentrations in Mead Lake. During the high flow summer of 2002, mean water residence time was less than 2 weeks and periods of high inflow resulted in water residence times on the order of days. During these periods, chlorophyll concentrations declined in the surface waters, suggesting flushing and dilution of algal cells. Lower water residence times during storm inflows approached reproductive doubling times for algae (i.e., 5-15 days), suggesting population growth could not compensate for cellular losses as a result of rapid flushing. In 2003, mean summer chlorophyll was much higher and bloom frequencies for concentrations greater than 30 mg m^{-3} were greater than for 2002. A higher mean water residence time (i.e., greater than theoretical doubling times for algae) coupled with high total and soluble P in the surface waters were correlated with higher mean concentrations of chlorophyll, suggesting algal populations were not P limited in growth and could increase faster than the flushing rate.

Bathtub modeling for the summers of 2002 and 2003 suggested that water quality indicators were responsive to simulated P loading reductions from tributary inflows, suggesting water quality improvement with decreases in P loading. Conversely, increases in P loading could result in exacerbation of current water quality conditions. Lake response and water quality improvement to simulated decreases in tributary P loading were less in 2003, versus 2002. This difference was likely due to increased residence time during dry summers and possible contributions of P from the sediments stored in the lake.

P management for water quality improvement should focus on reductions in tributary P loading, as this source accounts for the majority of the P budget of Mead Lake. Transport management strategies such as such as tilling nutrient amendments into the soil shortly after application, avoiding winter applications of manure, constructing buffer strips adjacent to critical source areas, planting perennial cover in hydrologically-sensitive areas, contouring, conservation tillage, and wetland rehabilitation would be beneficial in reducing erosion of particulate P and slow its runoff to Mead Lake by promoting interception and trapping (Sharpley et al. 1999, Gburek et al. 2000). Additionally, equilibrium processes between TSS and aqueous phases can result in very high SRP concentrations in the runoff due to P desorption from particles for agriculturally-managed watersheds like the South Fork of the Eau Claire River. Thus, both source and transport control practices that achieve overall reductions in soil P concentration (i.e., modifying or abating P amendments and allowing crop uptake to deplete concentrations over a period of years) in critical source areas would be needed in order to reduce soluble P concentrations in the runoff. Watershed models would be useful for simulating the impacts of various source and transport control mechanisms on tributary P loading reduction to Mead Lake and for goal-setting to achieve realistic water quality improvements.

Management of internal P loading from the sediment should not be attempted in Mead Lake until significant tributary P loading reduction has been achieved through Best Management Practices. Otherwise, in-lake treatments such as an alum application will likely be negated due to a readily available tributary P source and retention of new P-rich sediment in the lake that can be recycled at a later date.

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Table 1. Areal land use practice for the entire watershed above Mead Lake, the area draining the South Fork of the Eau Claire River above County MM, and the Rocky Run watershed located downstream of the Eau Claire River gauging Station.

Land Use Category	Entire Watershed (ha)	Eau Claire River (ha)	Rocky Run (ha)
Water	136	18	2
Grassland and Pasture	2966	2194	600
Row Crop Agriculture	13962	11612	1922
Forest	5946	3911	1075
Wetlands	2440	2003	220

Table 2. Annual and summer precipitation, Mead Lake water residence time, mean flow for the South Fork of the Eau Claire River, and loading of total suspended solids, total phosphorus, soluble reactive phosphorus, and total nitrogen for the Eau Claire River in 2002 and 2003. CV = coefficient of variation.

Eau Claire River Annual	Precip[1]	Flow	Res. Time	Total Suspended Solids			Total Phosphorus			Soluble Reactive Phosphorus			Total Nitrogen		
	(inches)	(cms)	(d^{-1})	(kg/y)	(mg/L)	(CV)	(kg/y)	(mg/L)	(CV)	(kg/y)	(mg/L)	(CV)	(kg/y)	(mg/L)	(CV)
2002	40.8	1.76	12.5	701987	12.7	0.20	6682	0.120	0.15	3475	0.063	0.15	109739	1.978	0.08
2003	26.0	1.30	16.8	552893	13.4	0.22	4931	0.120	0.15	2574	0.063	0.15	82283	2.003	0.08

Eau Claire River May-Sep	Precip[1]	Flow	Res. Time	Total Suspended Solids			Total Phosphorus			Soluble Reactive Phosphorus			Total Nitrogen		
	(inches)	(cms)	(d^{-1})	(kg/su)	(mg/L)	(CV)	(kg/su)	(mg/L)	(CV)	(kg/su)	(mg/L)	(CV)	(kg/su)	(mg/L)	(CV)
2002	25.5	2.09	10.5	388516	14.1	0.21	3397	0.123	0.16	1732	0.063	0.16	57867	2.093	0.08
2003	13.4	1.24	17.8	171839	10.5	0.19	1872	0.115	0.13	1024	0.063	0.13	28584	1.750	0.07

Table 3. A comparison between phosphorus export coefficients determined for the South Fork of the Eau Claire River watershed in 2002 and 2003 and the range of coefficients observed for agricultural watersheds in Wisconsin.

Watersheds with > 50% agricultural land use	(kg ha^{-1} y^{-1})
Low[1]	0.16
Most Likely[1]	0.56
High[1]	2.35
S. Fork Eau Claire River 2002	0.34
S. Fork Eau Claire River 2003	0.25

[1] Panuska and Lillie (1995)

Table 4. Mean rates of phosphorus release from sediments as a function of redox (i.e., oxic or anoxic) and pH. The standard error of the mean is in parentheses.

Lake station location	Rates of phosphorus release (mg m^{-2} d^{-1})		
	Oxic pH ~7.7	Oxic pH 8.5	Anoxic
Station 10	0.12 (0.03)	0.53 (0.03)	16.25 (2.20)
Station 15	0.22 (0.05)	0.44 (0.12)	25.34 (3.32)
Station 20	0.22 (0.12)	0.96 (0.29)	38.66 (2.38)

Table 5. Summer (May – September) mean values for Secchi transparency, viable chlorophyll (Chla), total phosphorus (TP) and trophic state index (TSI) values for the surface waters of Mead Lake.

				Wisconsin Trophic State Index			Carlson Trophic State Index		
Year	Secchi (m)	Chla (μg L^{-1})	TP (mg L^{-1})	TSI_{SD}	TSI_{CHLA}	TSI_{TP}	TSI_{SD}	TSI_{CHLA}	TSI_{TP}
2002	0.52	50.8	0.130	69.2	64.5	65.8	69.4	69.1	74.4
2003	0.70	76.2	0.125	65.0	67.6	65.5	65.1	73.1	73.8

Table 6. Summer phosphorus (P) fluxes to Mead Lake. Percentage of the measured P input or output for each flux is shown in parentheses.

Estimated Summer P Flux to Mead Lake (mg m^{-2} d^{-1})		2002	2003
Change in Lake P Storage		0.30	0.67
P inputs	Gauged and Ungauged Tributary P Loading	21.1 (87.4)	11.7 (78.4)
	Direct P Precipitation	0.03 (0.1)	0.02 (0.1)
	Internal P Loading (mg m^{-2} d^{-1})	3.0 (12.5)	3.5 (21.5)
P outputs	P Discharge	12.0 (50.5)	6.9 (47.1)
	P Deposition	11.8 (49.5)	7.7 (52.9)

Table 7. Model algorithms and coefficients used for Bathtub load reduction modeling.

Variable	2002		2003	
	Coef	Model	Coef	Model
Phosphorus	0.72	2^{nd} order, available P	1.19	2^{nd} order, available P
Chlorophyll	0.51	P, Jones and Bachman	0.82	P, Jones and Bachman
Secchi Transparency	1.7	CHLA and turbidity	1.0	CHLA and turbidity

Mead Lake, Wisconsin

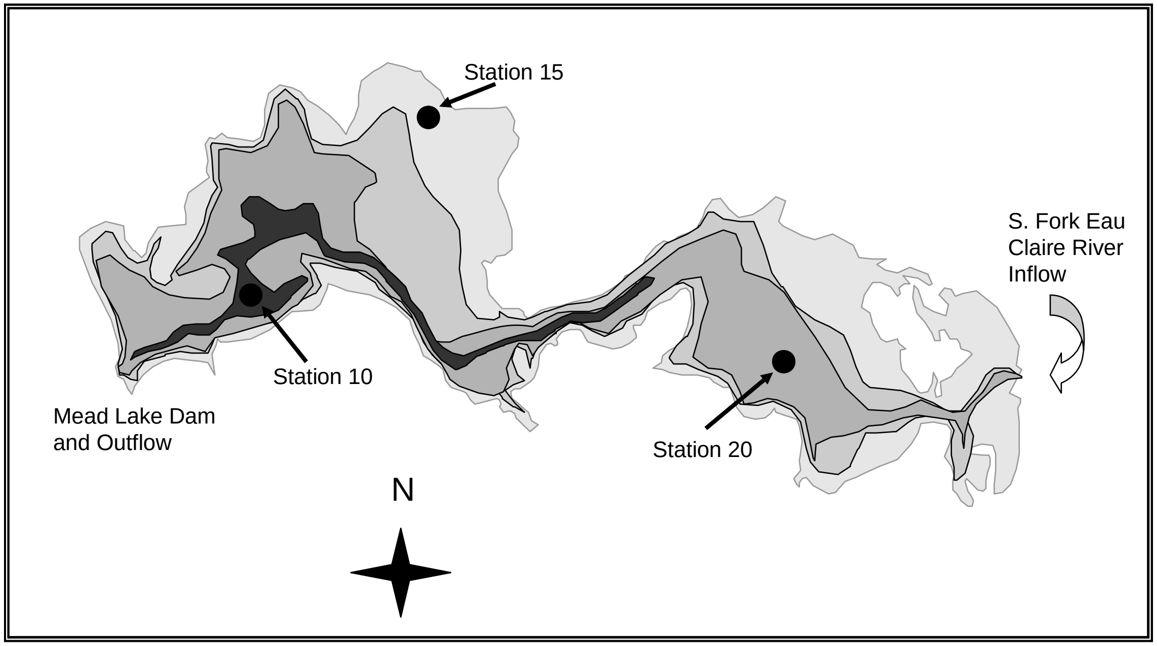

Figure 1. Bathymetric map of Mead Lake showing water quality and sediment core sampling station locations.

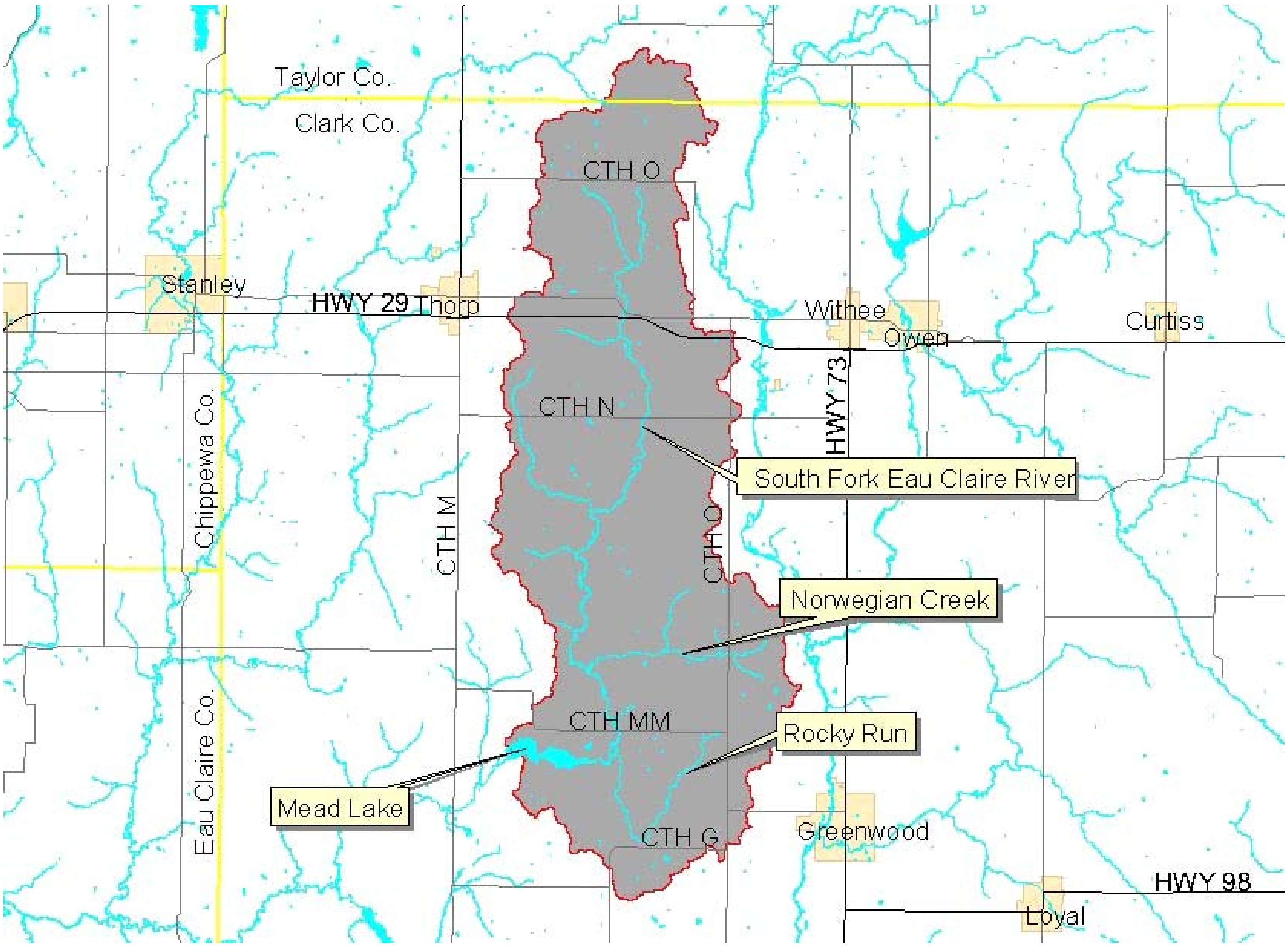

Figure 2. Mead Lake watershed.

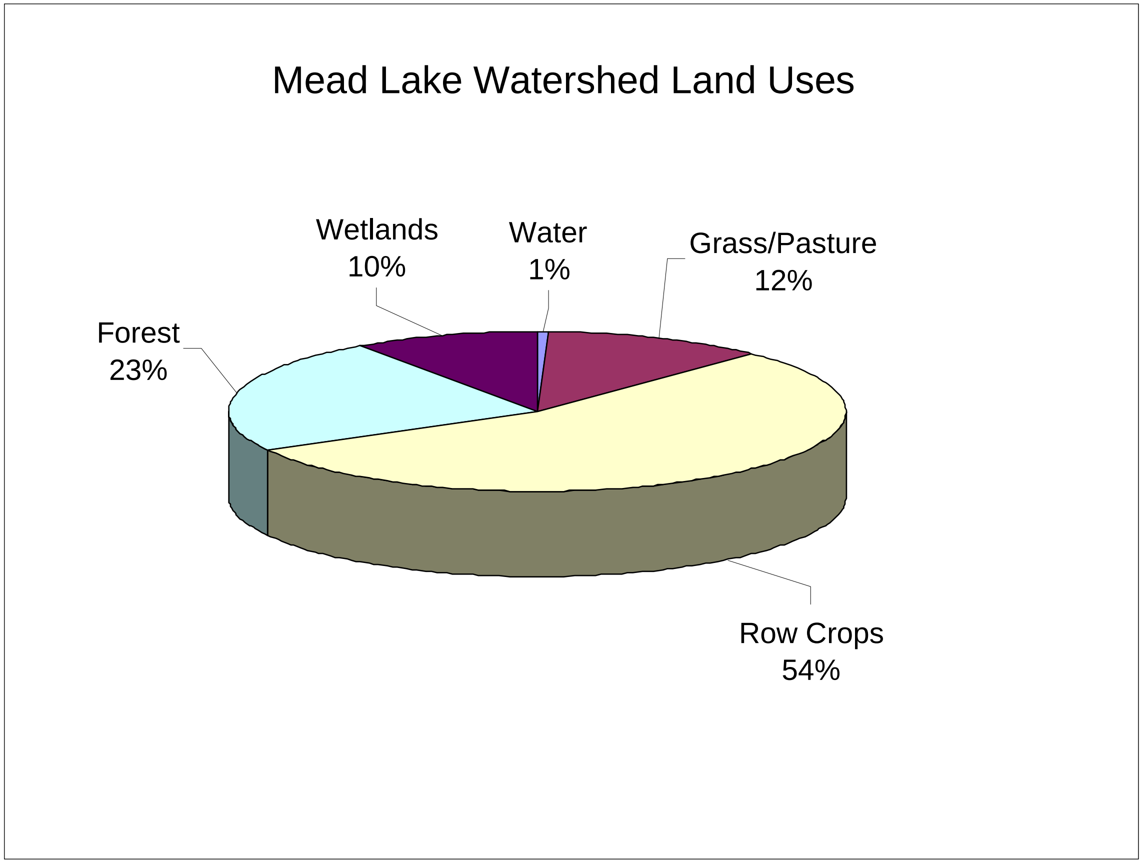

Figure 3. Distribution of land-use practices in the Mead Lake watershed.

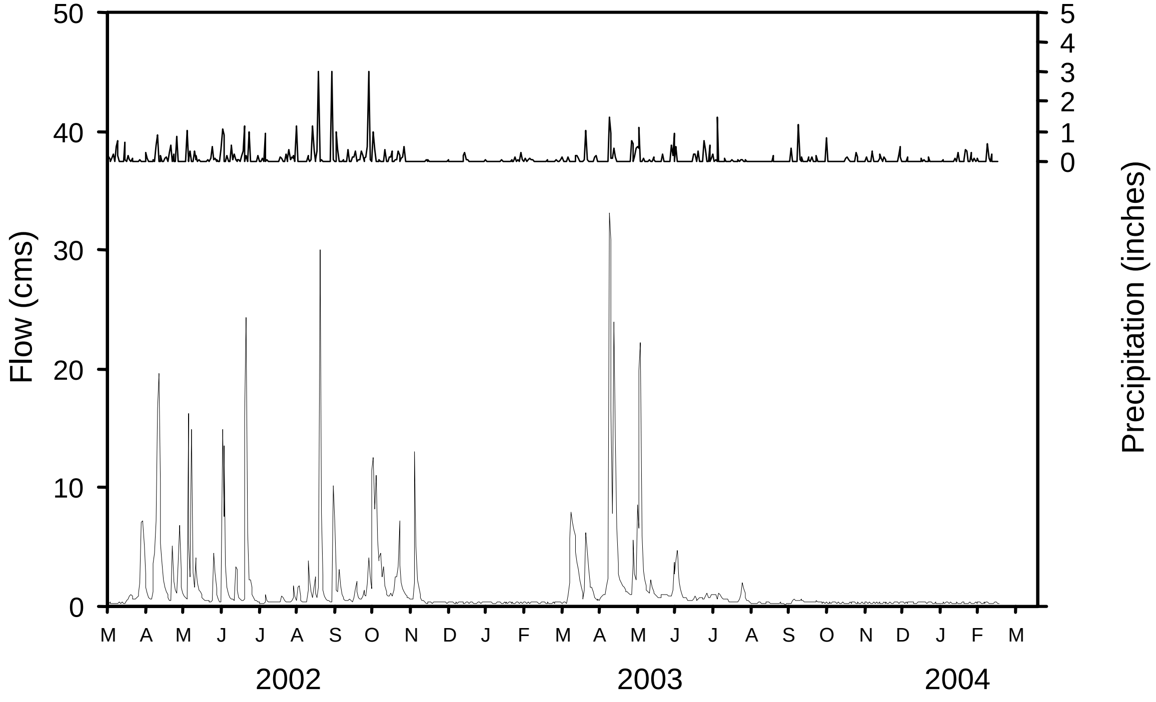

Figure 4. Seasonal variations in daily precipitation (upper) and mean daily flow of the South Fork of the Eau Claire River (lower).

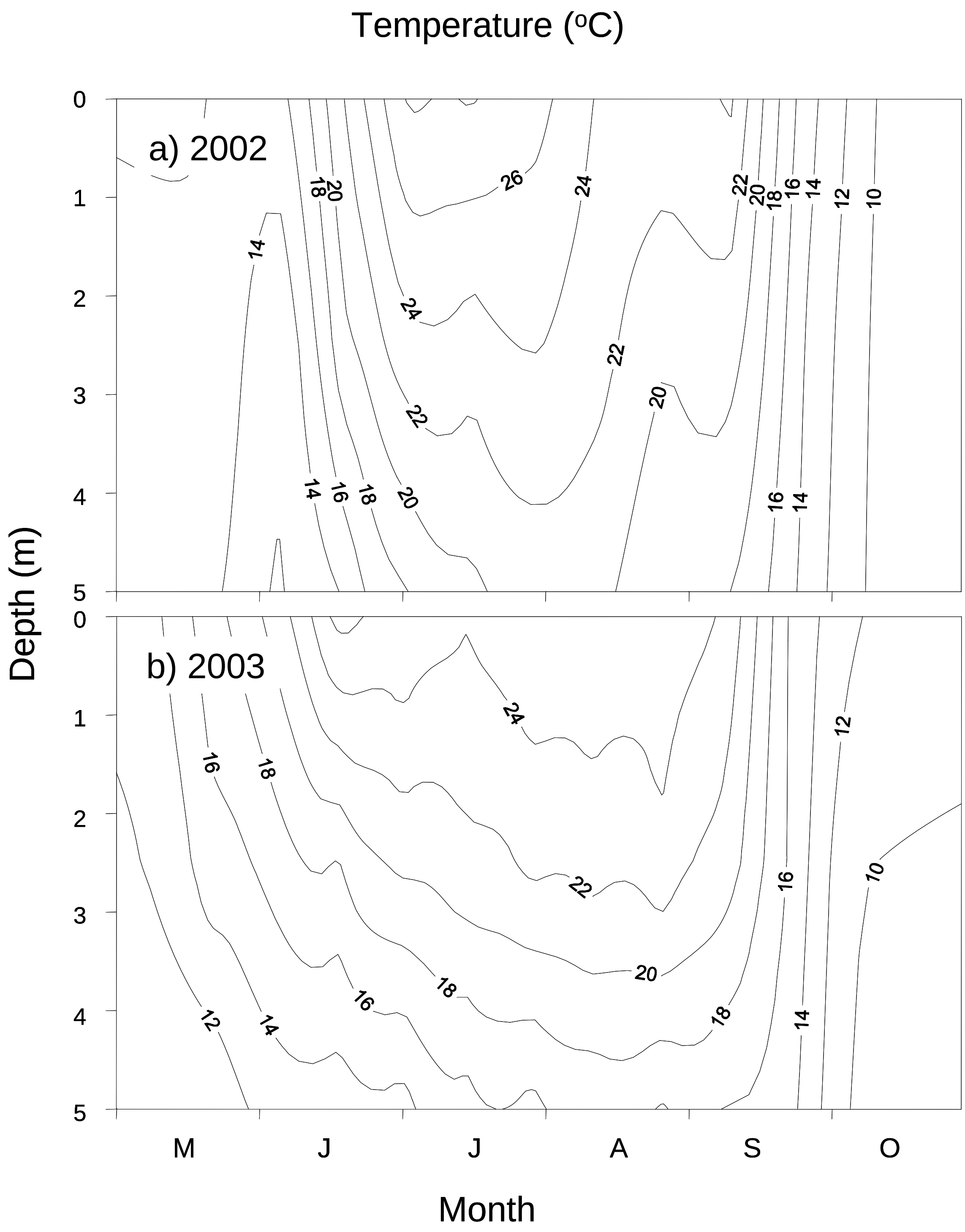

Figure 5. Seasonal and vertical temperature (^0C) contours at station 10.

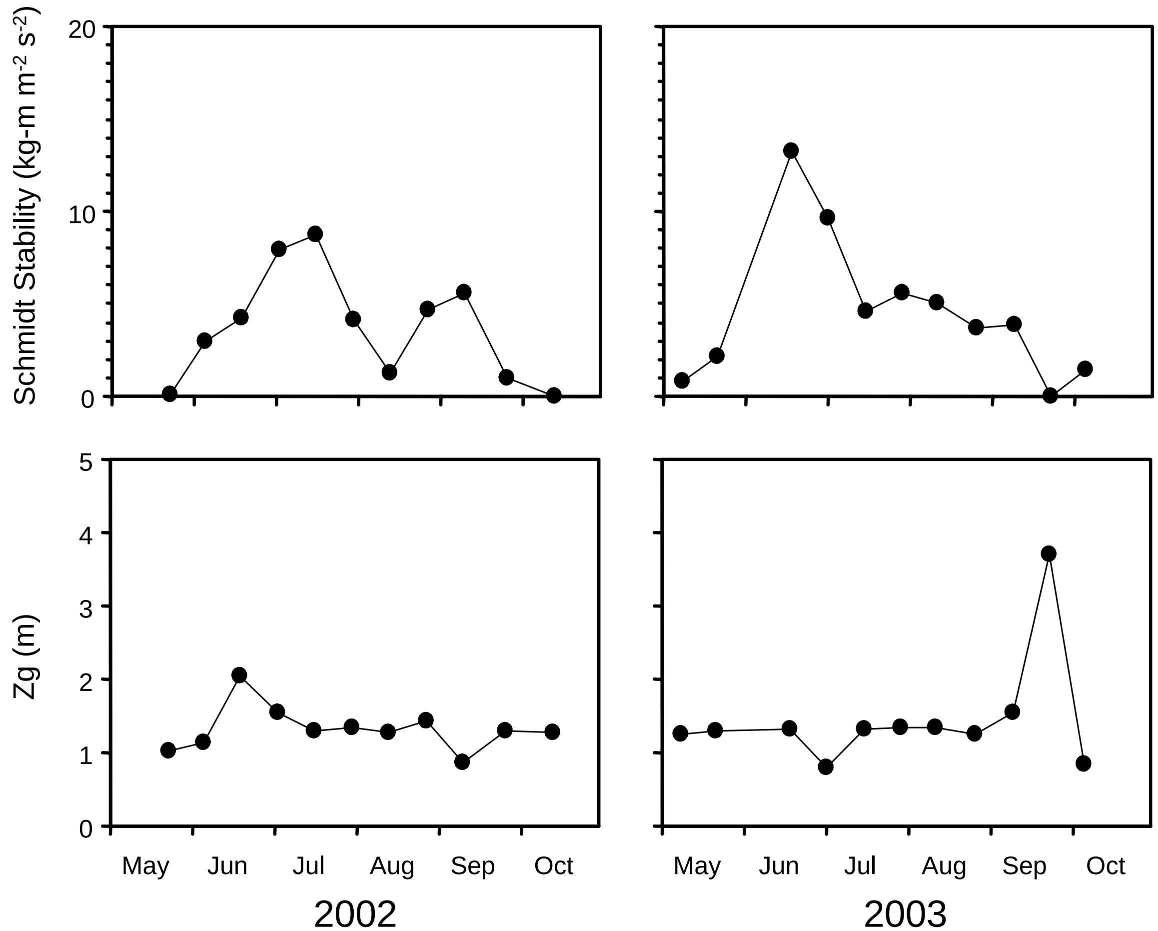

Figure 6. Variations in lake stability (upper) and the approximate metalimnetic depth (lower).

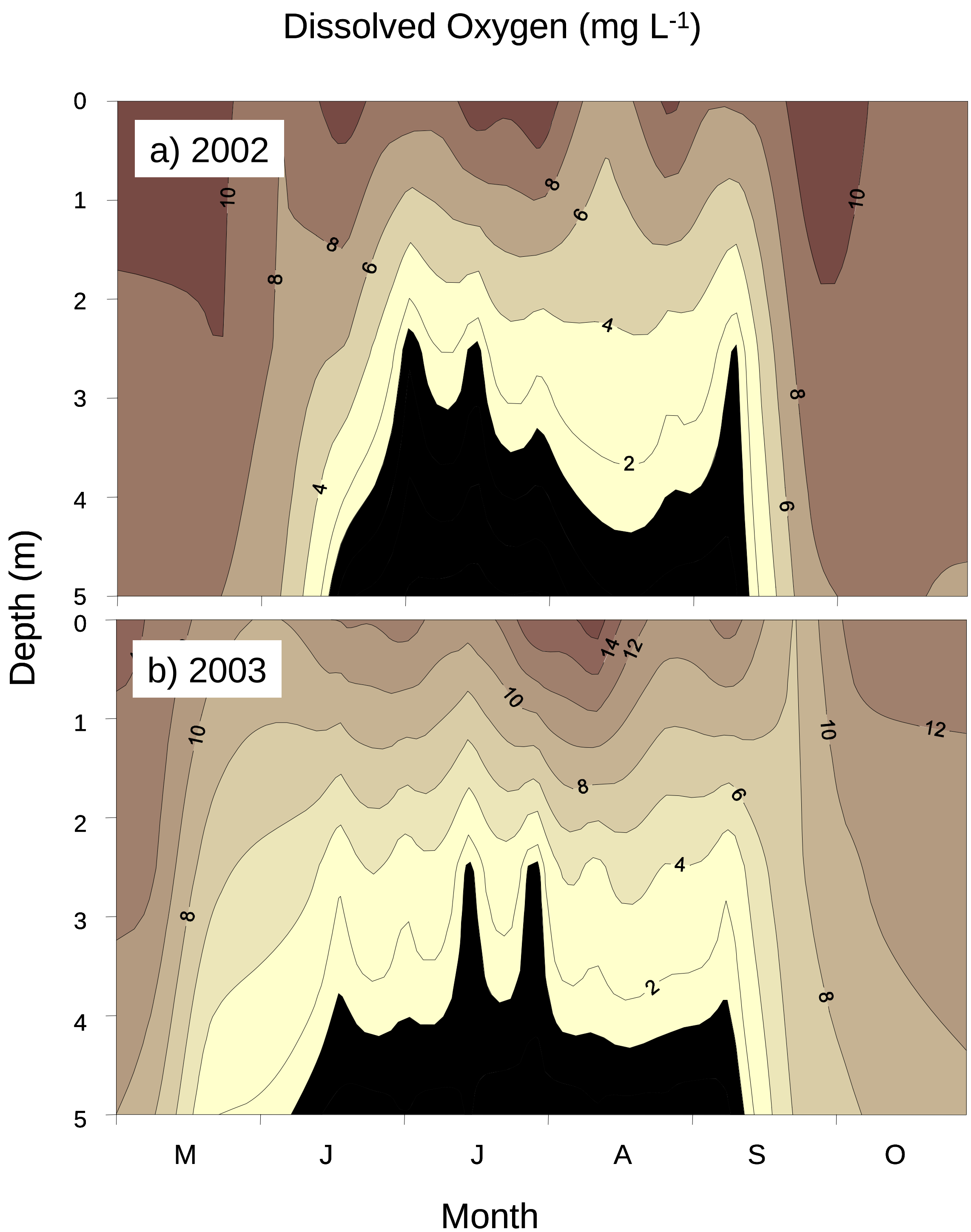

Figure 7. Seasonal and vertical dissolved oxygen (mg L^{-1}) contours at station 10. Black shaded area represents concentrations < 1 mg L^{-1}.

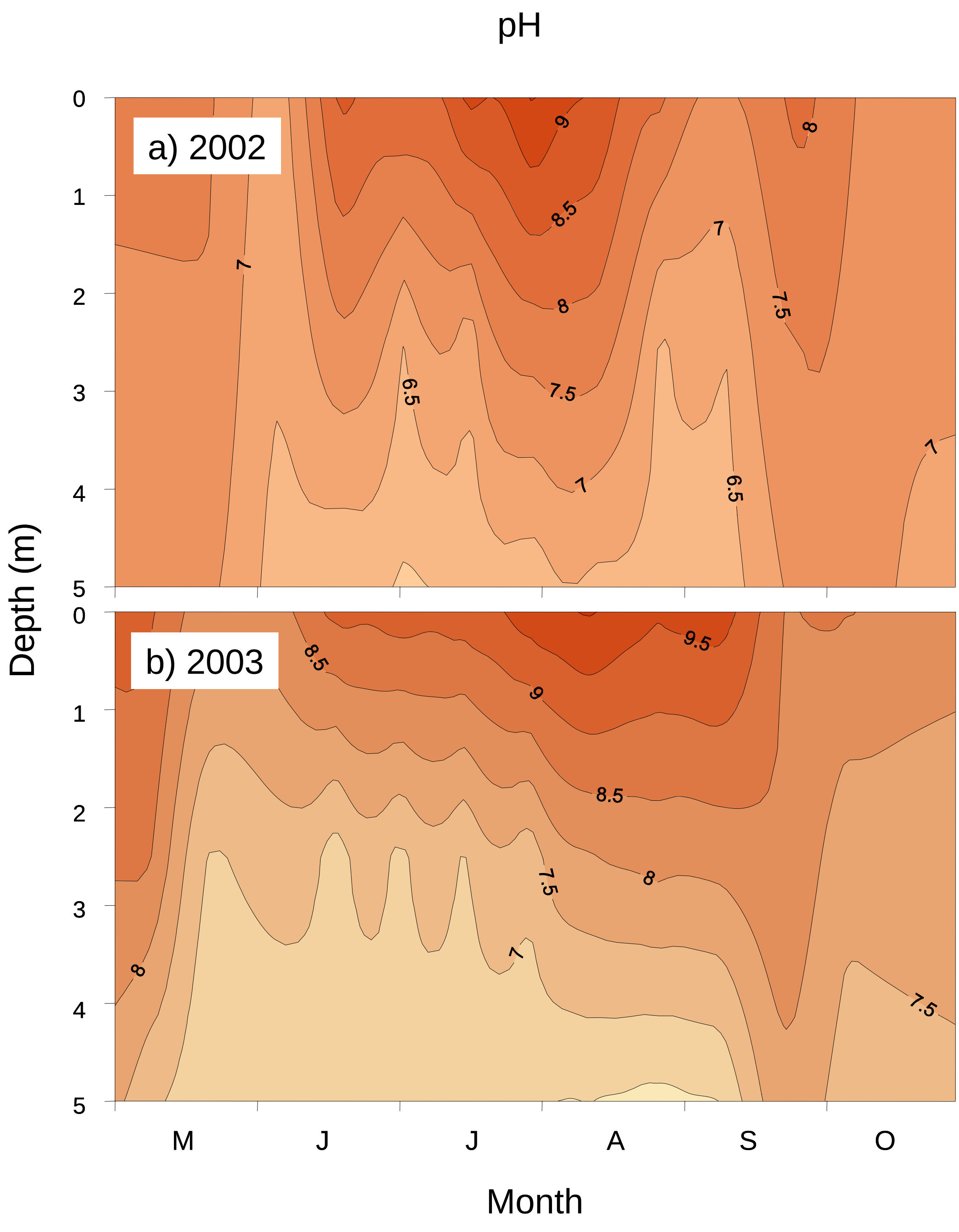

Figure 8. Seasonal and vertical pH contours at station 10.

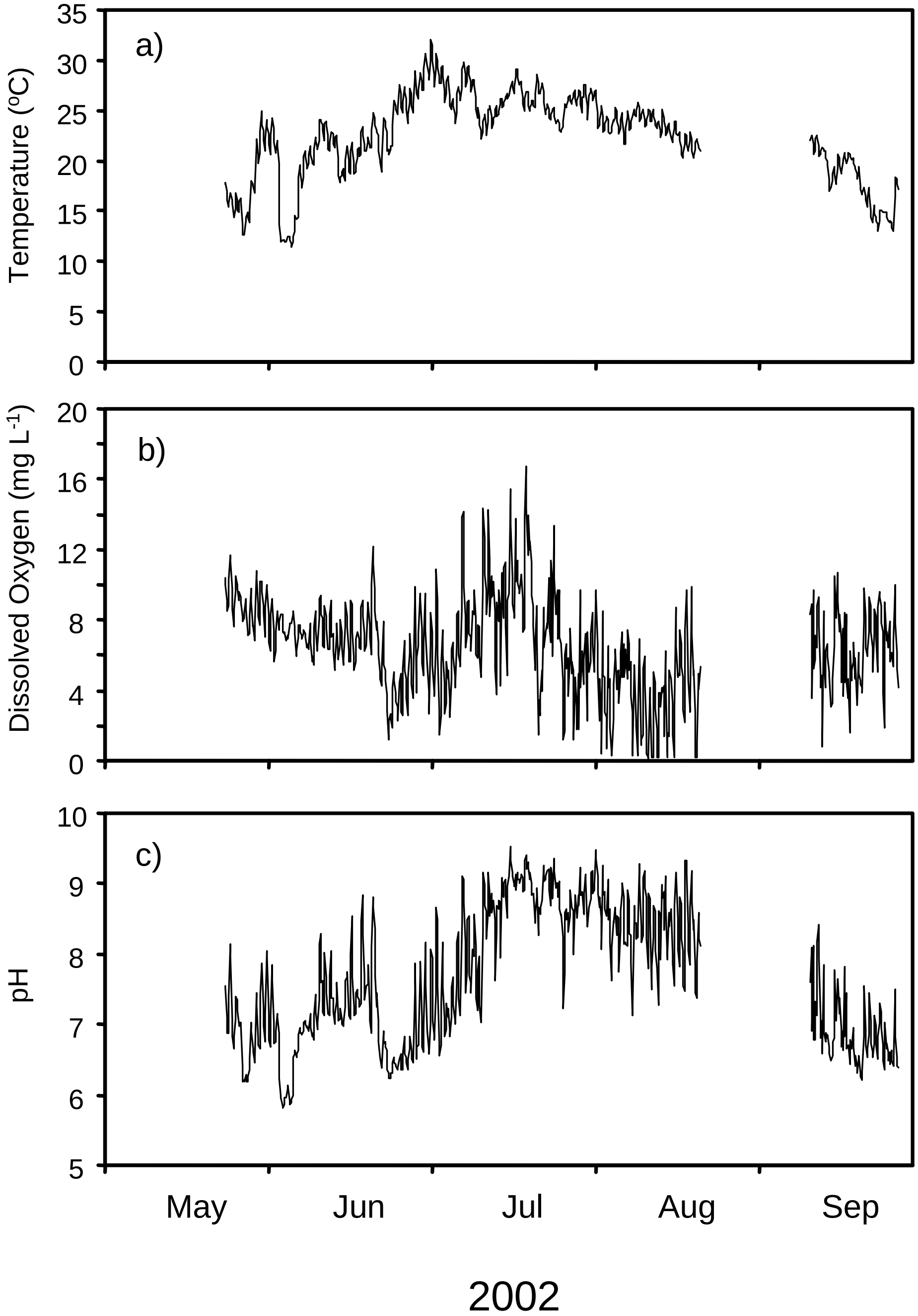

Figure 9. Diel and seasonal variations in temperature (a), dissolved oxygen (b), and pH (c) above the sediment interface at station 15.

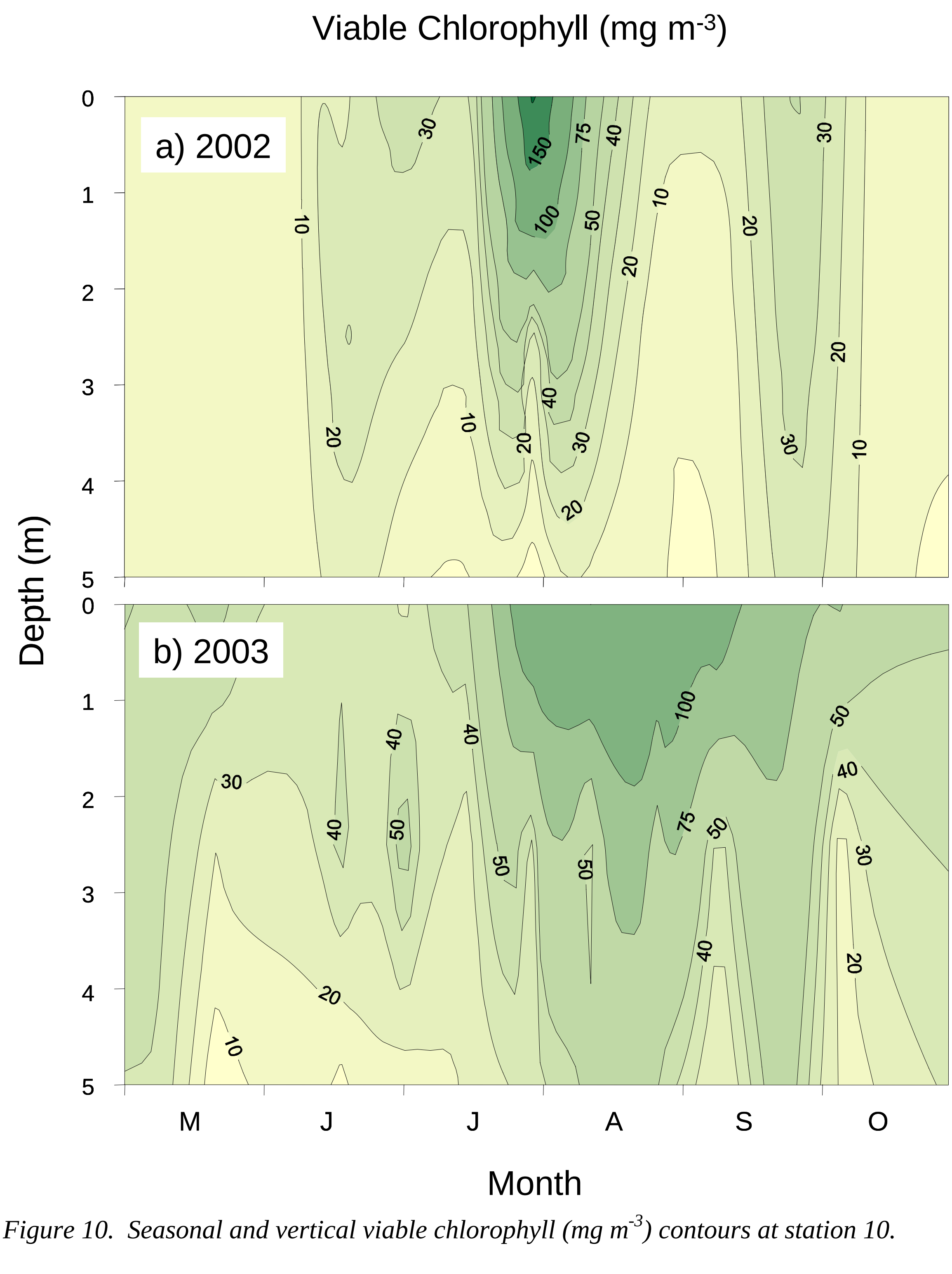

Figure 10. Seasonal and vertical viable chlorophyll (mg m^{-3}) contours at station 10.

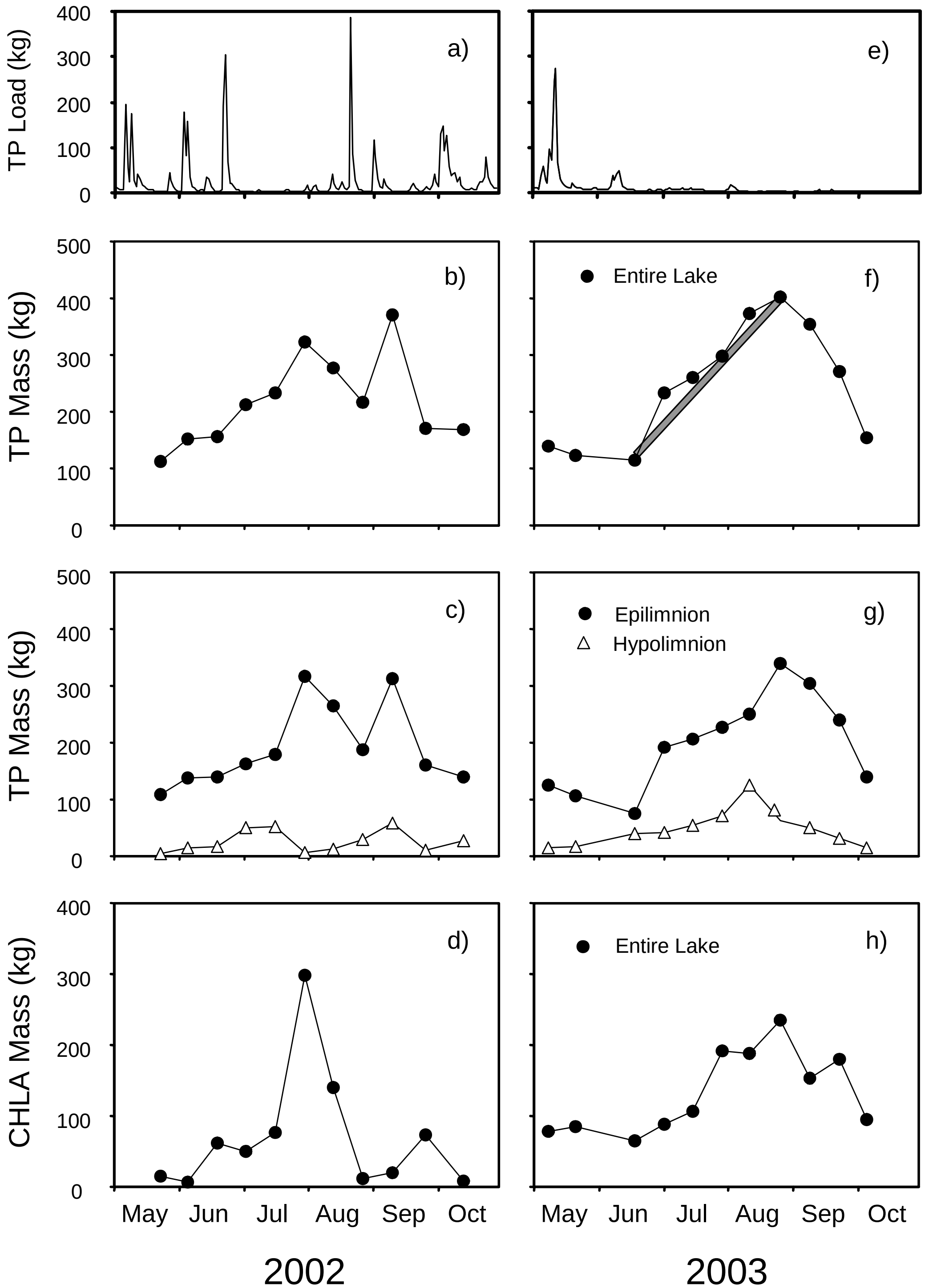

Figure 11. Seasonal variations in tributary total phosphorus (TP) loading from the Eau Claire River (a and e), lake-wide TP mass (b and f), lake-wide TP mass in the epilimnion and hypolimnion (c and g), and lake-wide chlorophyll (CHLA) mass (d and h).

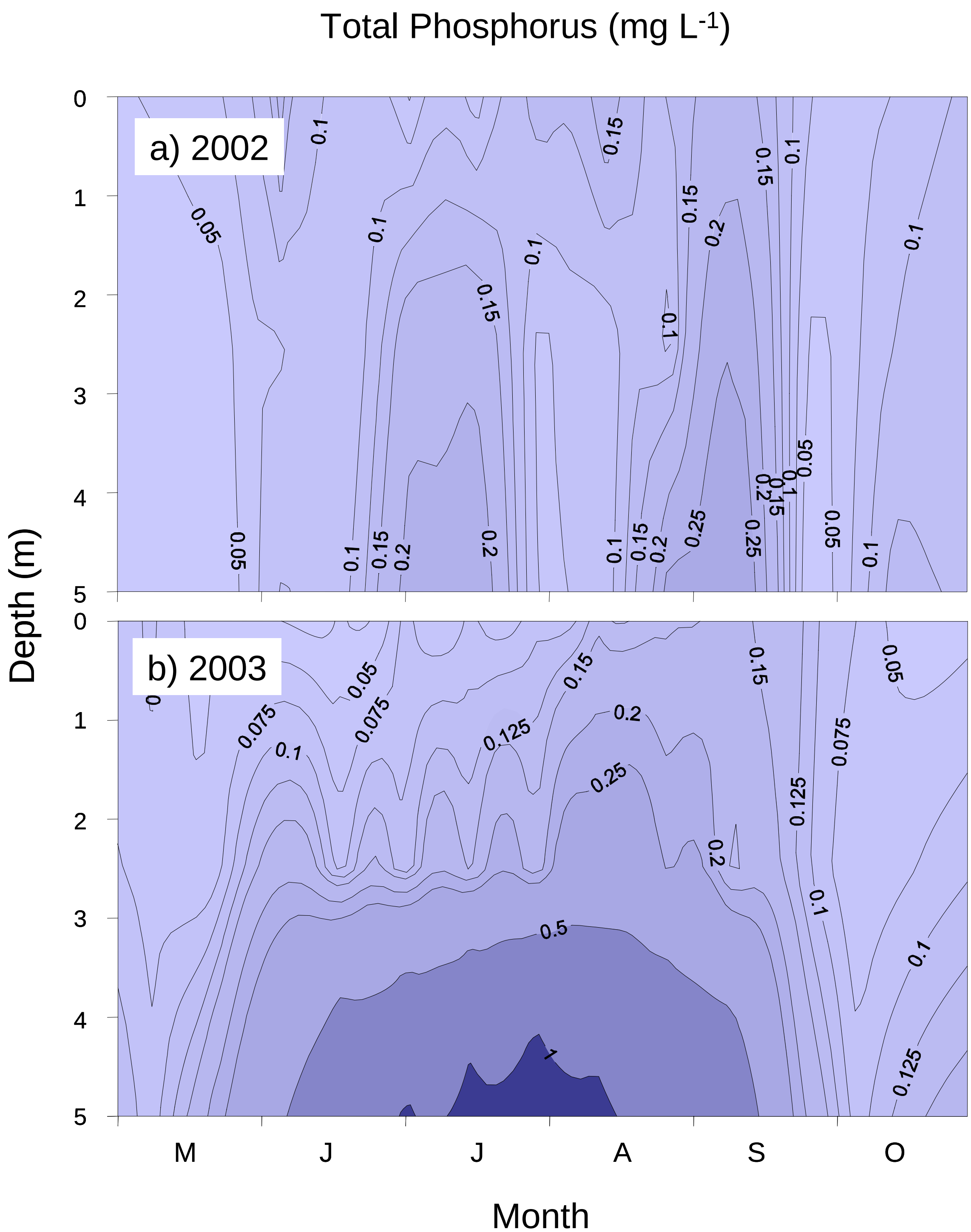

Figure 12. Seasonal and vertical total phosphorus (mg L^{-1}) contours at station 10.

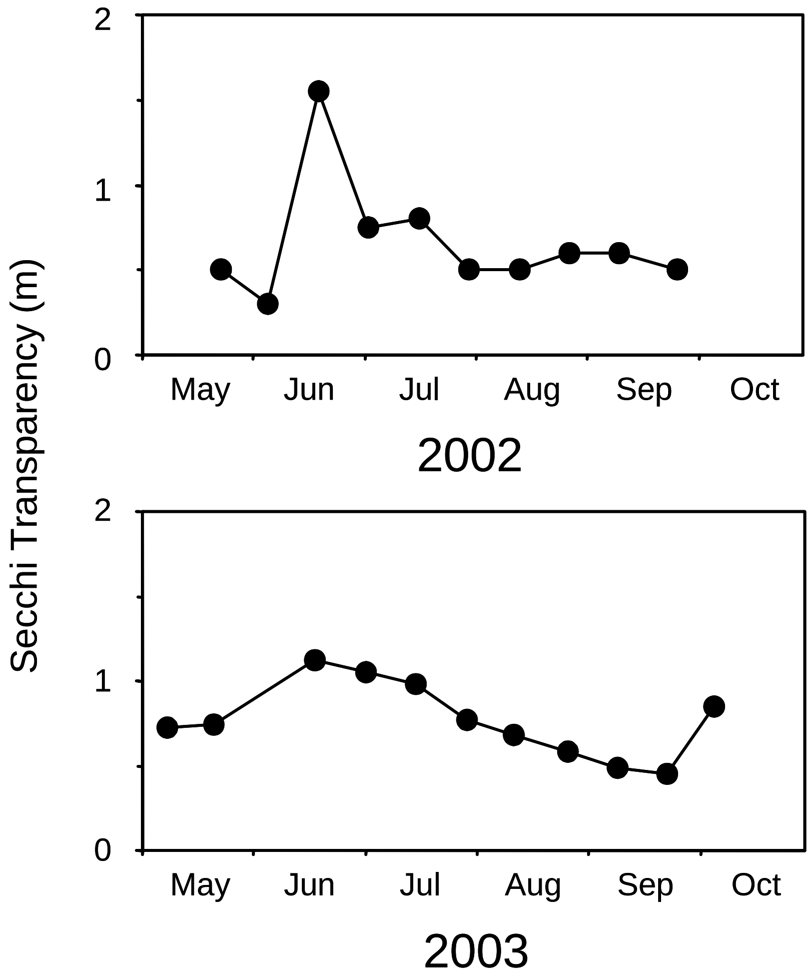

Figure 13. Seasonal variations in Secchi transparency at station 10.

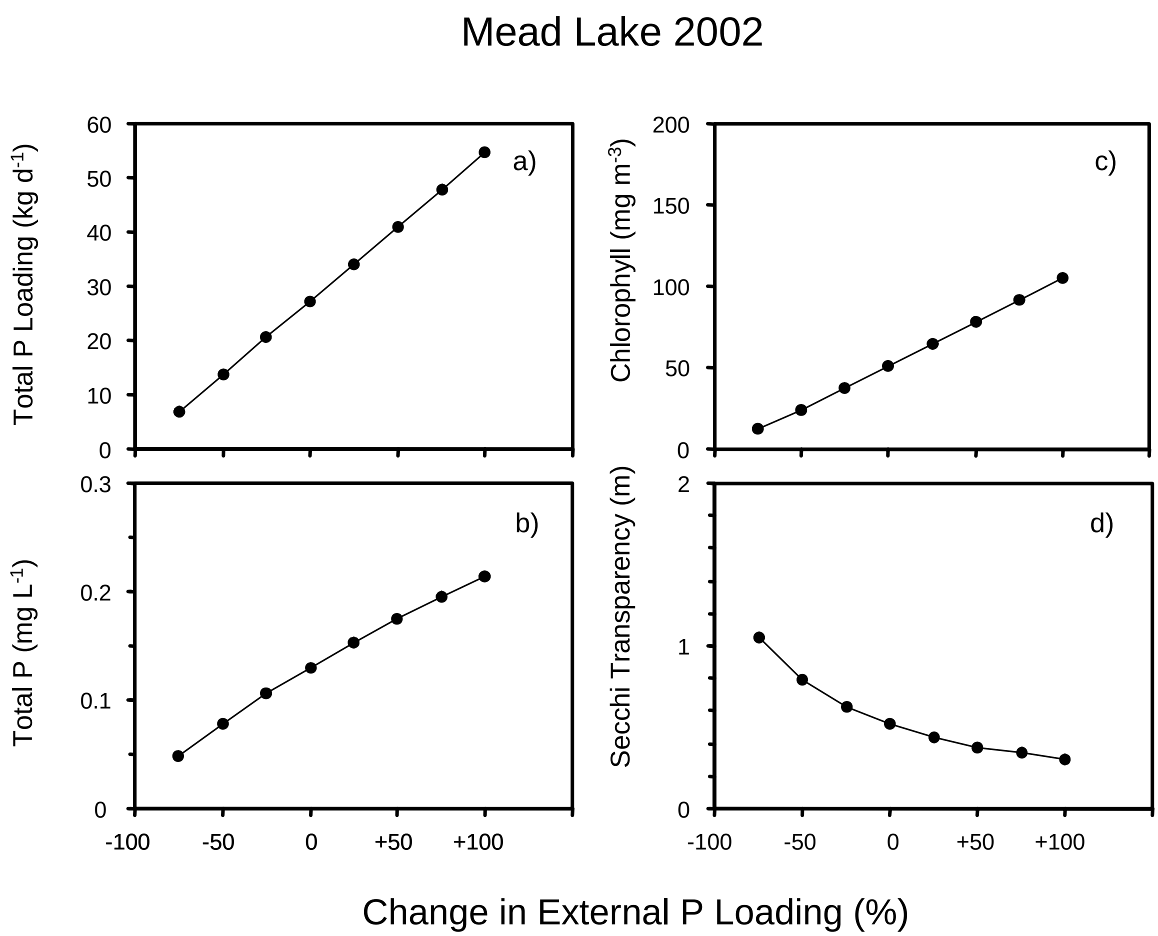

Figure 14. Bathtub model output of predicted changes in total phosphorus (P), chlorophyll, and Secchi transparency as a function of increases or decreases in 2002 P loading conditions to Mead Lake.

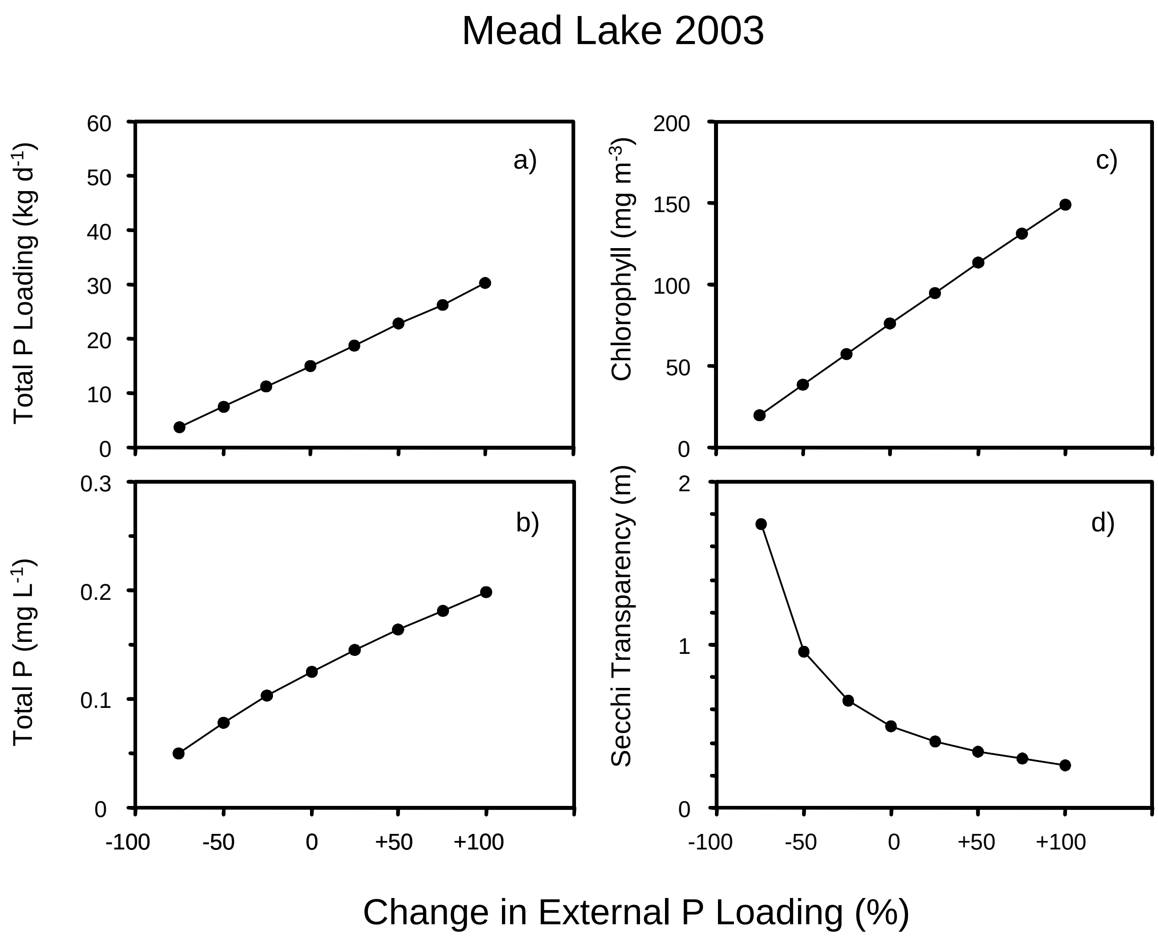

Figure 15. Bathtub model output of predicted changes in total phosphorus (P), chlorophyll, and Secchi transparency as a function of increases or decreases in 2003 P loading conditions to Mead Lake.

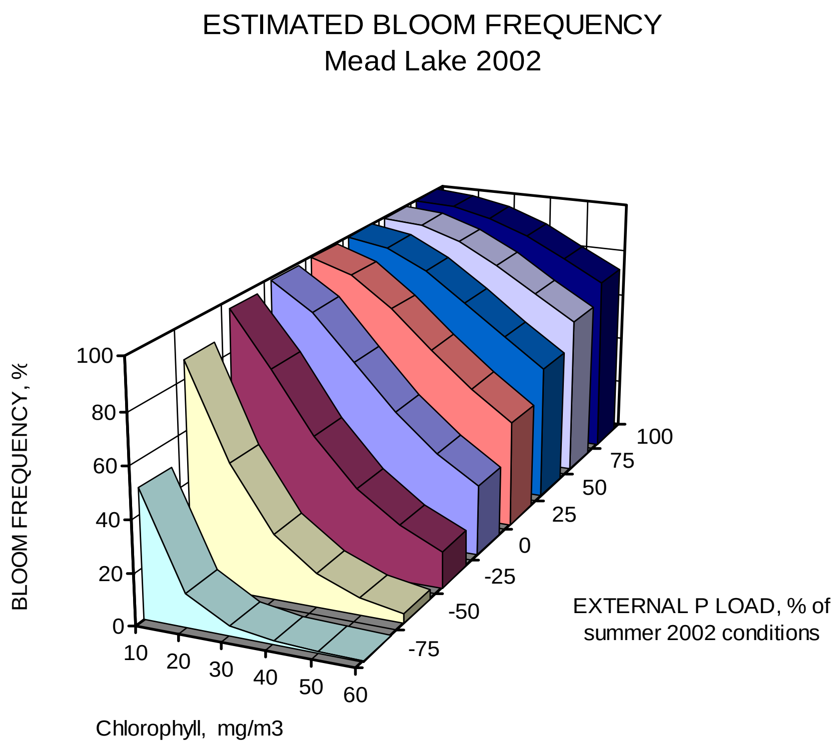

Figure 16. Bathtub model output of predicted changes in algal bloom frequency occurrence (as chlorophyll) as a function of increases (i.e., > 100 %) or decreases (i.e., < 100 %) in 2002 tributary total P loading conditions to Mead Lake.

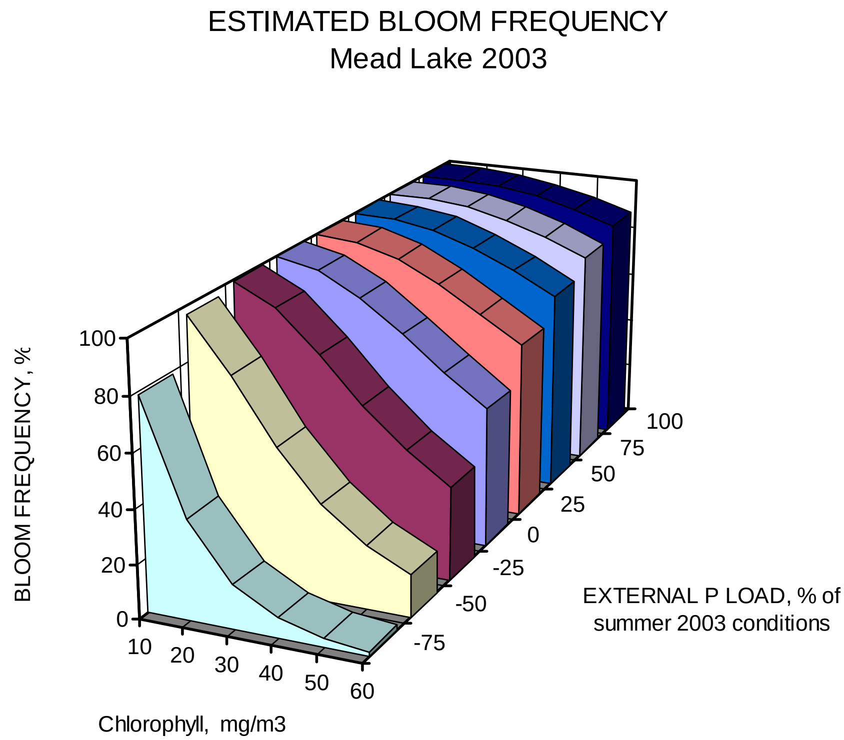

Figure 17. Bathtub model output of predicted changes in algal bloom frequency occurrence (as chlorophyll) as a function of increases (i.e., > 100 %) or decreases (i.e., < 100 %) in 2003 tributary total P loading conditions to Mead Lake.