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TMDL reevaluation: reconciling internal phosphorus load reductions in a eutrophic lake

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Abstract

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Total maximum daily loads (TMDLs) are assigned to impaired waterbodies and provide a prescription for recovery. It is implicit that both the existing conditions and targets identified in a TMDL are based on credible and accurate information, ensuring that management actions directed to recovery are appropriate and effective. We evaluated the TMDL for phosphorus in Bear Lake, Michigan, to assess if the prescribed reduction was appropriately based on existing annual internal loads. We developed 5 different annual internal load scenarios based on phosphorus release rates from laboratory-based sediment incubations and diel dissolved oxygen measurements, ranging from very conservative to very liberal estimates of phosphorus release. The most realistic scenarios indicate that the previously established TMDL for internal phosphorus loading is 3–7 times greater than what actually occurs in Bear Lake. Based on our assessment, Bear Lake is likely already meeting the TMDL internal loading target, without the implementation of any management action. Thus, management efforts aimed at reducing the water column phosphorus concentration to reach the TMDL target should instead be directed at controlling the external phosphorus load.

Key words: Great Lakes, internal phosphorus loading, polymictic lake, TMDL, west Michigan

Approximately 67% of the assessed lakes, reservoirs, and ponds in the United States are currently listed as impaired according to the US Environmental Protection Agency (EPA) online watershed assessment database (USEPA 2013). Nutrients are the second most common cause of impairment to these lentic systems (following mercury). In accordance with the US Clean Water Act, a total maximum daily load (TMDL) is assigned to impaired waterbodies to facilitate a recovery process so the waterbodies can once again provide their designated uses. Technically, a TMDL identifies the maximum amount of a pollutant that a waterbody can receive and still attain water quality standards and allocates that amount to the pollutant's sources. Metaphorically, one can think of a TMDL as a “diet plan” for a waterbody that identifies a target for weight reduction but does not prescribe the exercise regime or menu.

Critical elements in the process of establishing a TMDL include defining existing conditions, calculating the pollutant

target, and identifying its source(s). For lake phosphorus (P) TMDLs, both internal and external P loads must be considered (Havens and Schelske 2001, Havens and James 2005).

Bear Lake (MI), a small, polymictic, eutrophic lake, was issued a TMDL in 2008 for elevated total phosphorus (TP) concentrations and excess algal growth. The lake is also part of the Muskegon Lake Area of Concern (AOC), one of 43 AOCs originally listed in the Laurentian Great Lakes because of their environmentally degraded status (Steinman et al. 2008). Of the 14 potential Beneficial Use Impairments (BUIs) associated with AOCs, the Muskegon Lake AOC has 9, with eutrophication/undesirable algae being especially problematic in Bear Lake.

The TMDL estimated a current annual P load to the lake of 1536 kg (3387 lbs) per year, contributed by both external (54%) and internal (46%) sources (MDEQ 2008). The TMDL calls for a total reduction of 56% in annual P loads, achieved through a 50% reduction in external load and a 79% reduction in internal load, to attain a target growing season (Apr–Sep) water column TP concentration of 30 $\mu\text{g/L}$ (from current average of 44 $\mu\text{g/L}$) in Bear Lake.

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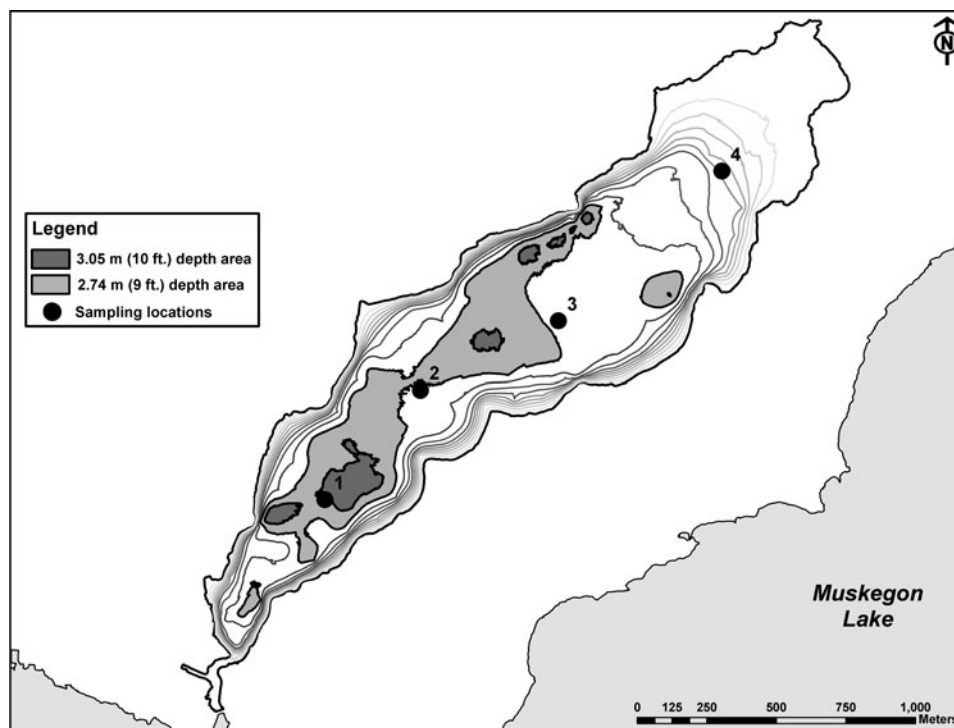


Figure 1. Sampling locations and bathymetry of Bear Lake. Light and dark gray shading show depths >2.7 m and >3 m, respectively.

An underlying assumption in the TMDL process is that the existing conditions, which ultimately determine restoration targets and necessary reductions, are based on accurate information. Although external load estimates used in the phosphorus TMDL for Bear Lake were determined using measured loads, internal load estimates were derived using an indirect modeling approach (MDEQ 2008). However, the shallow depths and frequent mixing that characterize Bear Lake may result in poor model performance and magnify the uncertainty in the internal P load estimates, given that redox-driven release of P under anoxic conditions (Boström et al. 1982) may not apply to Bear Lake. Consequently, there was concern on the part of stakeholders, regulators, and scientists that the substantial reduction in the internal P load to Bear Lake specified by the TMDL may be inappropriate. The goal of this project was to empirically determine internal P loading rates and to compare those numbers to the indirectly derived estimates on which the TMDL was based.

Materials and methods

Study site

Located in Muskegon County, Michigan, the 77 km² (29 mi²) Bear Creek/Bear Lake watershed includes forested ar-

eas (44%), urban/residential development (22%), agricultural lands (6%), wetlands (1%), and other land uses (27%; Cadmus and AWRI 2007). Bear Creek is the major tributary in the watershed and flows into Bear Lake at its northern end, which feeds into Muskegon Lake and eventually Lake Michigan. Bear Lake is a 1.66 km² (410 ac) drowned river mouth lake, with a mean depth of 2.1 m and a maximum depth of 3.6 m (Wilson et al. 2005). It experiences frequent mixing due to its shallow depth and heavy recreational boat traffic. The average hydraulic residence time is 30 d (MDEQ 2008).

Bear Lake is eutrophic, characterized by elevated TP concentrations and heavy summer blooms of cyanobacteria (Cadmus and AWRI 2007, MDEQ 2008, Xie et al. 2011). Despite the high TP concentrations, soluble reactive phosphorus (SRP) is often below detection in the summer, presumably because of active uptake by cyanobacteria (Xie et al. 2011). Based on nitrogen to phosphorus (N:P) molar ratios, the system seems to be P-limited, further validating that P control is essential to prevent even more bloom formation (Xie et al. 2011).

Water quality

Lake water quality was measured at the 4 coring locations (Fig. 1) at the time of core collection. Dissolved oxygen

(DO), pH, temperature, specific conductance, chlorophyll *a*, and total dissolved solids were measured at the surface, middle, and bottom of the water column using a YSI 6600 sonde. Photosynthetically active radiation (PAR) profiles were measured with a LiCor Li-193SA spherical quantum sensor and used to calculate the light extinction coefficient (K_d). Secchi disk depth was measured at each site to estimate water clarity. Water samples for P analysis were collected at the near-surface and near-bottom with a Niskin bottle. Water for SRP analysis was immediately syringe-filtered through 0.45 μm membrane filters into scintillation vials and stored on ice until transported to the laboratory, within 5 h of collection. TP samples were stored at 4 C, and SRP samples were frozen until analysis. Water quality sampling occurred between 09:00 and 15:00 h during each sampling event.

Diel dissolved oxygen

Measurements of diel DO concentration provided information on the redox status in Bear Lake; we used these data in the annual internal P loading models described later. Water column DO concentrations were measured overnight, from late afternoon until mid-morning, to characterize diel fluctuations in DO. YSI 6600 sondes were suspended from an anchored buoy at the near-surface and near-bottom of the lake at Sites 2 and 4. These sites were chosen because they were representative of relatively deep (3.5 m: Site 2) and shallow (2.5 m: Site 4) sediment coring locations in Bear Lake (Fig. 1). The sondes were programmed to log data every 10 min. Diel DO concentrations were measured 4 times over the study period: 16 August 2011, 14 May 2012, 27 June 2012, and 29 August 2012. Wind speed data for all diel DO events were downloaded from the Muskegon Lake Buoy Observatory (AWRI 2012), located ~ 0.5 km south of the sampling sites.

Phosphorus release rate determination

Sediment cores were collected from 4 locations in Bear Lake on 21 April 2011, 4 August 2011, 27 October 2011, and 11 July 2012 (Fig. 1). Sediment core sampling and laboratory incubation followed the procedures of Steinman et al. (2004). Six sediment cores were collected from each site using a piston corer (Fisher et al. 1992, Steinman et al. 2004) constructed of a graduated 0.6 m long polycarbonate core tube (7 cm inner diameter) and a polyvinyl chloride (PVC) attachment assembly for coupling to aluminum drive rods. The piston was advanced 20 to 25 cm prior to deployment to maintain a water layer on top of the core during collection. The corer was positioned vertically at the sediment–water interface and pushed downward with the piston cable remaining stationary. After collection, the core was brought to the surface and the bottom was sealed with a rubber stopper

prior to removal from the water, resulting in an intact sediment core ~ 20 cm in length, with a 25 cm overlying water column. The piston was then bolted to the top of the core tube to keep it stationary during transit (Ogdahl et al. 2014). Core tubes were placed in a vertical rack and maintained at ambient temperature during transit. An additional core was collected from each site for sediment chemistry analysis; the top 5 cm was removed for the analysis of TP, iron (Fe), and porewater SRP in the lab.

The 24 sediment cores (6 per site) were placed in a Revco environmental growth chamber with the temperature maintained to match ambient near-bottom conditions in Bear Lake at the time of collection. The water column in 3 of the cores from each site was bubbled with N_2 (with 330 ppm CO_2) to create buffered anoxic conditions, while the remaining 3 were bubbled with oxygen to create oxic conditions.

P release rates were estimated using the methods outlined in Moore et al. (1998), with minor modifications (Steinman et al. 2004, 2009a). Briefly, a 40 mL water sample was removed by syringe through the sampling port of each core tube at 0 and 12 h and 1, 2, 4, 6, 8, 12, 16, 20, 24, and 27 d (October 2011 samples were collected at 25 d, but not at 24 and 27 d). Immediately after removal, a 20 mL subsample was refrigerated for analysis of TP, and a 20 mL subsample was filtered through a 0.45 μm membrane filter and frozen for analysis of SRP. SRP and TP were analyzed on a SEAL AQ2 discrete automated analyzer (USEPA 1983). P values below detection were calculated as one-half the detection limit (5 $\mu\text{g/L}$). The 40 mL sample was replaced with filtered water collected from the corresponding site in the lake to maintain the original volume in the core tubes.

Flux (P release rate) calculations were based on the change in water column P using the following equation:

$$P_{\text{tr}} = (C_t - C_0)V/A, \quad (1)$$

where P_{tr} is the net P release rate (positive values) or retention rate (negative values) per unit surface area of sediments, C_t is the P concentration in the water column at time t , C_0 is the P concentration in the water column at time 0, V is the volume of water in the water column, and A is the planar surface area of the sediment cores. P_{tr} was calculated over the time period with the maximum apparent release rate, with the caveat that the shortest time interval for the calculation included a minimum of 3 consecutive sampling days to avoid potential short-term bias. Given that these release rates are laboratory-based, we refer to them as “apparent” to distinguish them from field data.

Sediment core chemistry

Immediately following the sediment core incubations, the overlying water was drawn off with a peristaltic pump. The

top 5 cm of sediment was then removed from each core, homogenized, and subsampled for analysis of TP, Fe, and porewater SRP. Fe content was analyzed using inductively coupled plasma-atomic emission spectrometry according to EPA method 6010B (USEPA 2007). TP (on ashed material) was analyzed according to APHA (1992). Sediment Fe:P ratios were determined by weight using dry weight-normalized TP concentrations.

Annual internal phosphorus loading

Using the sediment P release rates from our laboratory incubations, combined with different redox and water mixing scenarios, we generated 5 different annual internal P load estimates (Table 1). We purposefully included conservative and liberal internal load estimates to provide boundaries for plausible low and high annual internal P loads, respectively.

Annual internal P load (L_{int} ; kg/yr) was calculated for Scenarios 1, 2, 3, and 5 (Table 1) according to:

$$L_{int} = ([RX_{spring} + RX_{summer} + RX_{fall} + RX_{winter}] \times d \times A_X) + ([RO_{spring} + RO_{summer} + RO_{fall} + RO_{winter}] \times d \times A_O), \quad (2)$$

where L_{int} is the annual internal P load, RX is the anoxic release rate, RO is the oxic release rate, d is the number of days per season (91.25), A_X is the anoxic lake area, and A_O is the oxic lake area. Winter release rates were assumed to be zero (Nürnberg 2009, Nürnberg et al. 2013), and summer release rates were calculated as the average of rates measured in August 2011 and July 2012.

Scenario 4 was based on the model by Nürnberg et al. (2012) for the active sediment release area and time (AA), which represents the length of time (days per growing season) that an area similar to the lake surface area is actively releasing P (Table 1). This model was developed to calculate the number of anoxic days that a polymictic lake may experience during the growing season (i.e., summer and fall; Nürnberg et al. 2012). A hybrid approach was taken that modified the Nürnberg et al. (2012) model to include spring internal P load based on the method employed in Scenario 2 (Table 1); winter load was assumed to be zero (Nürnberg 2009, Nürnberg et al. 2013).

Annual internal P load for polymictic lakes was estimated in a series of 4 equations:

$$AA = -36.2 + 50.1 \log(P_{summer/fall}) + 0.762z/A^{0.5}, \quad (3)$$

where P is the average water column TP concentration during summer and fall, z is mean depth, and A is lake surface area. AA values (days per growing season) were used

to calculate growing season internal P load ($L_{int_summer/fall}$) according to:

$$L_{int_summer/fall} = (AA \times RX_{summer/fall}) \times A, \quad (4)$$

where AA is the number of anoxic days per growing season, $RX_{summer/fall}$ is the mean anoxic release rate for summer/fall as determined by laboratory incubations, and A is lake surface area.

We explicitly accounted for spring internal P load by incorporating a spring anoxic factor (AX_{spring} ; see Nürnberg et al. 2012). AX_{spring} was based on the approach from Scenario 2 and assumed that only the deepest areas of Bear Lake experienced hypoxia (Table 1). Spring internal P load was calculated as:

$$L_{int_spring} = RX_{spring} \times AX_{spring} \times d, \quad (5)$$

where RX_{spring} is the anoxic release rate during spring as determined by laboratory incubations, AX_{spring} is the area of Bear Lake >3 m deep, and d is the number of days during the season (91.25). The oxic release rate was zero during spring and was therefore excluded from the calculation.

Annual internal P load was calculated as the sum of $L_{int_summer/fall}$ and L_{int_spring} :

$$L_{int} = L_{int_summer/fall} + L_{int_spring}. \quad (6)$$

Statistical analysis

The effects of site and sampling date on maximum apparent P release rates and sediment TP concentration were evaluated using either a one-way analysis of variance (ANOVA) or Kruskal-Wallis one way ANOVA on ranks, if assumptions of normality and/or equal variance were not met. Multiple comparison tests were performed for all significant contrasts ($P < 0.05$) using the Holm-Sidak method (ANOVA) or Tukey test (Kruskal-Wallis). We used the t-test (or Mann-Whitney U-test) to test oxic vs anoxic differences for each incubation period.

Results

Water quality

Water depth was shallowest at Site 4 and deepest at Sites 1 and 2 (Table 2), and Secchi depth was <1 m during all sampling events. There was a seasonality in water clarity, with reduced transmissivity during the summer months, as evidenced by the lower Secchi depths, higher mean light extinction coefficients, and higher turbidity values (Table 2). Water temperature showed no sign of thermal stratification on any of the dates or at any of the locations that we sampled, with near-bottom temperatures usually within 1 C of

Table 1. Scenarios used to calculate a range of annual internal P load estimates for Bear Lake, ordered from most conservative (Scenario 1) to most liberal (Scenario 5) estimates.

| Scenario | Hypoxic Surface Area of Lake Bottom | General Explanation | Rationale |
|--|-------------------------------------|--|--|
| 1. Oxic: entire lake | 0 km ² | Assumes entire lake remains oxic throughout year; most conservative internal loading estimate | Lack of thermal stratification and presence of constant mixing precludes hypoxia |
| 2. Hypoxic: depths >3.0 m; rest of lake oxic | 0.047 km ² | Assumes lake becomes hypoxic at deepest depths; very conservative estimate | Water column mixing is less likely to introduce oxygenated water at deepest depths, allowing hypoxia to develop during calm periods |
| 3. Hypoxic: depths >2.7 m; rest of lake oxic | 0.296 km ² | Assumes lake becomes hypoxic at relatively deep locations; generally conservative estimate | Similar to Scenario 2, but this scenario allows a greater surface area of lake bottom to go hypoxic than Scenario 2, potentially resulting in more sediment P release during calm periods |
| 4. Polymictic lake estimate (modified from Nürnberg et al. 2012) | N/A | Combines modeled value for the number of anoxic days in summer/fall in polymictic lakes with spring release at the deepest depths; generally conservative estimate | Water column mixing is a temporal phenomenon in polymictic lakes; this model estimates the number of days a polymictic lake experiences anoxia, which is then used to calculate redox-specific loading |
| 5. Hypoxic: entire lake | 1.518 km ² | Assumes entire lake becomes hypoxic; most liberal estimate | This scenario does not happen under current conditions, but it provides an extreme upper limit for how much internal loading theoretically could occur in Bear Lake |

the near-surface temperatures (Table 2). Mean DO concentrations were 11–12 mg/L during the April and October 2011 sampling events. Although DO was lower during the August 2011 and July 2012 events, the lowest near-bottom concentration measured was 6.06 mg/L (Site 2, Aug 2011; Table 2), well above potentially hypoxic conditions. Mean chlorophyll *a* concentrations were lowest in April 2011 and highest in October 2011, when they ranged from 19 to 24 µg/L (Table 2). Water column TP was similar among sites and between near-surface and near-bottom depths. TP was lowest in April and October 2011 (30–40 µg/L) and highest in August 2011 and July 2012 (50–70 µg/L; Table 2), with a grand mean of 48.5 µg/L.

Diel dissolved oxygen

Although mid-day DO measurements revealed no hypoxia in Bear Lake, stratification and short-term hypoxic conditions could possibly develop on a diel basis. Diel DO measurements were made on 4 occasions at sites representing deeper (Site 2) and shallower (Site 4) areas in Bear Lake. There was no evidence of hypoxia at Site 4 on any sampling date (Fig. 2); DO concentrations remained between 7 and 12 mg/L at this site, even at the near-bottom station.

In contrast, some evidence of hypoxia (<2 mg/L) was present near the bottom of Site 2 during the night and early morning (Fig. 2) and was especially pronounced during the midsummer measurements (August 2011 and 2012). During the late spring diel measurement (mid-May 2012), DO concentrations never fell below 5 mg/L (Fig. 2). There was even less evidence for hypoxia in late June 2012, when near-bottom DO concentrations at Site 2 never fell below 7 mg/L, most likely due to wind-induced mixing (Fig. 2). Wind speeds of 15–20 knots were measured at the Muskegon Lake Buoy Observatory beginning at 11:00 h on 27 June and continued until 03:00 h on 28 June (Fig. 2). Wind speeds during the other diel measurements were <10 knots (Fig. 2).

Phosphorus release rates

SRP concentrations in the water column overlying sediment cores were below detection (Apr 2011 and Oct 2011) and/or very low (Aug 2011 and Jul 2012); therefore all P release analyses were performed using TP concentrations. TP concentrations generally followed the same pattern within each incubation period, with an initial decline in TP followed by (1) some degree of increase in TP in the anoxic treatments or (2) a stabilization or slowing of the TP decline in the oxic treatments (Fig. 3).

Table 2. Selected water quality variables for primary coring locations in Bear Lake. All measurements were taken between 09:00 and 15:00 h. July 2012 data unavailable for chlorophyll *a* (Chl-*a*) due to sensor malfunction.

| Variable | Date | Site 1 | | Site 2 | | Site 3 | | Site 4 | | Mean (SD) | |
|--|-------------|-------------|------------|-------------|------------|-------------|------------|-------------|------------|-------------------|------------------|
| Bottom depth (m) | 2 Apr 2011 | 2.85 | | 3.15 | | 3.20 | | 2.20 | | 2.85 | (0.46) |
| | 4 Aug 2011 | 3.65 | | 3.80 | | 3.30 | | 2.60 | | 3.34 | (0.53) |
| | 27 Oct 2011 | 3.60 | | 3.60 | | 3.40 | | 2.70 | | 3.33 | (0.43) |
| | 11 Jul 2012 | 3.55 | | 3.40 | | 3.00 | | 2.40 | | 3.09 | (0.51) |
| Secchi depth (m) | 2 Apr 2011 | 0.75 | | 0.60 | | 0.75 | | 0.75 | | 0.71 | (0.08) |
| | 4 Aug 2011 | 0.50 | | 0.50 | | 0.50 | | 0.50 | | 0.50 | (0.00) |
| | 27 Oct 2011 | 0.75 | | 0.90 | | 0.75 | | 0.75 | | 0.79 | (0.08) |
| | 11 Jul 2012 | 0.40 | | 0.35 | | 0.40 | | 0.50 | | 0.41 | (0.06) |
| Light extinction coefficient (K _d) | 2 Apr 2011 | 1.72 | | 1.43 | | 1.50 | | 1.75 | | 1.60 | (0.16) |
| | 4 Aug 2011 | 2.55 | | 2.81 | | 2.81 | | 3.37 | | 2.89 | (0.34) |
| | 27 Oct 2011 | 2.58 | | 2.08 | | 1.90 | | 2.11 | | 2.17 | (0.29) |
| | 11 Jul 2012 | 2.62 | | 2.41 | | 2.59 | | 2.85 | | 2.62 | (0.18) |
| Temp (C) | 2 Apr 2011 | Surface 8.2 | Bottom 7.7 | Surface 7.4 | Bottom 7.3 | Surface 7.3 | Bottom 7.2 | Surface 7.9 | Bottom 6.7 | Surface 7.7 (0.5) | Bottom 7.2 (0.4) |
| | 4 Aug 2011 | 28.1 | 27.4 | 27.6 | 27.1 | 27.4 | 26.9 | 26.7 | 26.6 | 27.5 (0.6) | 27.0 (0.3) |
| | 27 Oct 2011 | 9.9 | 9.5 | 9.6 | 9.6 | 9.6 | 9.7 | 9.6 | 9.4 | 9.7 (0.2) | 9.6 (0.1) |
| | 11 Jul 2012 | 27.8 | 26.6 | 27.9 | 26.7 | 27.5 | 26.7 | 27.2 | 26.3 | 27.6 (0.3) | 26.6 (0.2) |
| DO (mg/L) | 2 Apr 2011 | 12.0 | 12.0 | 12.1 | 12.0 | 11.9 | 11.8 | 11.5 | 11.8 | 11.9 (0.2) | 11.9 (0.1) |
| | 4 Aug 2011 | 9.5 | 7.3 | 8.9 | 6.1 | 7.7 | 6.5 | 6.9 | 6.5 | 8.2 (1.2) | 6.6 (0.5) |
| | 27 Oct 2011 | 11.8 | 11.2 | 11.4 | 10.8 | 11.1 | 11.0 | 11.1 | 10.7 | 11.3 (0.4) | 10.9 (0.2) |
| | 11 Jul 2012 | 9.5 | 7.1 | 10.0 | 7.6 | 9.9 | 8.0 | 9.5 | 9.3 | 9.7 (0.3) | 8.0 (1.0) |
| Turbidity (NTU) | 2 Apr 2011 | 5.7 | 6.0 | 5.8 | 5.9 | 5.7 | 5.8 | 5.2 | 6.2 | 5.6 (0.3) | 6.0 (0.2) |
| | 4 Aug 2011 | 19.6 | 20.6 | 19.4 | 28.1 | 19.6 | 21.7 | 17.3 | 17.3 | 19.0 (1.1) | 21.9 (4.5) |
| | 27 Oct 2011 | 5.5 | 5.8 | 5.2 | 11.7 | 10.1 | 6.5 | 5.6 | 10.6 | 6.6 (2.3) | 8.7 (2.9) |
| | 11 Jul 2012 | 15.6 | 19.8 | 22.3 | 18.3 | 14.9 | 17.6 | 14.9 | 22.1 | 16.9 (3.6) | 19.5 (2.0) |
| Chl- <i>a</i> (μg/L) | 2 Apr 2011 | 5.6 | 10.9 | 7.9 | 11.4 | 2.6 | 11.5 | 5.5 | 12.1 | 5.4 (2.2) | 11.5 (0.5) |
| | 4 Aug 2011 | 5.3 | 13.4 | 14.2 | 16.4 | 11.9 | 14.4 | 12.1 | 11.7 | 10.9 (3.9) | 14.0 (2.0) |
| | 27 Oct 2011 | 18.8 | 22.5 | 24.0 | 22.1 | 24.3 | 22.3 | 23.5 | 18.1 | 22.7 (2.6) | 21.3 (2.1) |
| TP (μg/L) | 2 Apr 2011 | 30 | 30 | 30 | 30 | 30 | 30 | 30 | 40 | 30 (0) | 33 (5) |
| | 4 Aug 2011 | 60 | 70 | 60 | 60 | 60 | 70 | 50 | 60 | 58 (5) | 65 (6) |
| | 27 Oct 2011 | 30 | 30 | 30 | 30 | 30 | 30 | 30 | 40 | 30 (0) | 33 (5) |
| | 11 Jul 2012 | 70 | 70 | 60 | 70 | 70 | 70 | 60 | 40 | 65 (6) | 63 (15) |

Internal P loading and a TMDL

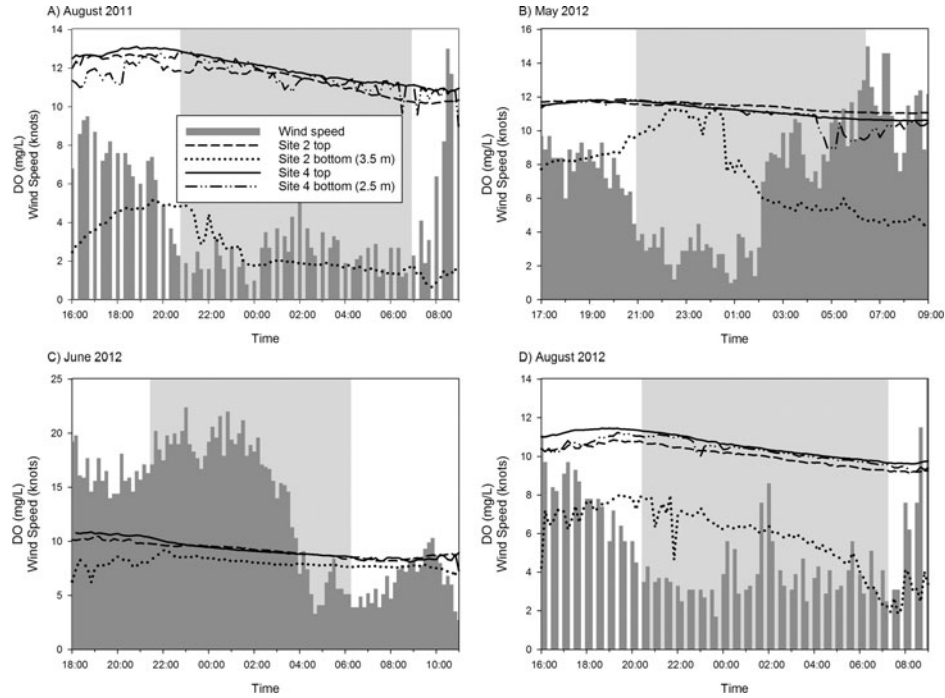


Figure 2. Diel dissolved oxygen (DO) data measured 16–17 August 2011 (upper left), 14–15 May 2012 (upper right), 27–28 June 2012 (lower left), and 29–30 August 2012 (lower right), at a shallow (Site 4) and deep (Site 2) primary coring locations in Bear Lake. Wind speed shown in dark gray, nighttime shown in light gray. Note the y-axis scale varies among panels. Wind data from AWRI (2012).

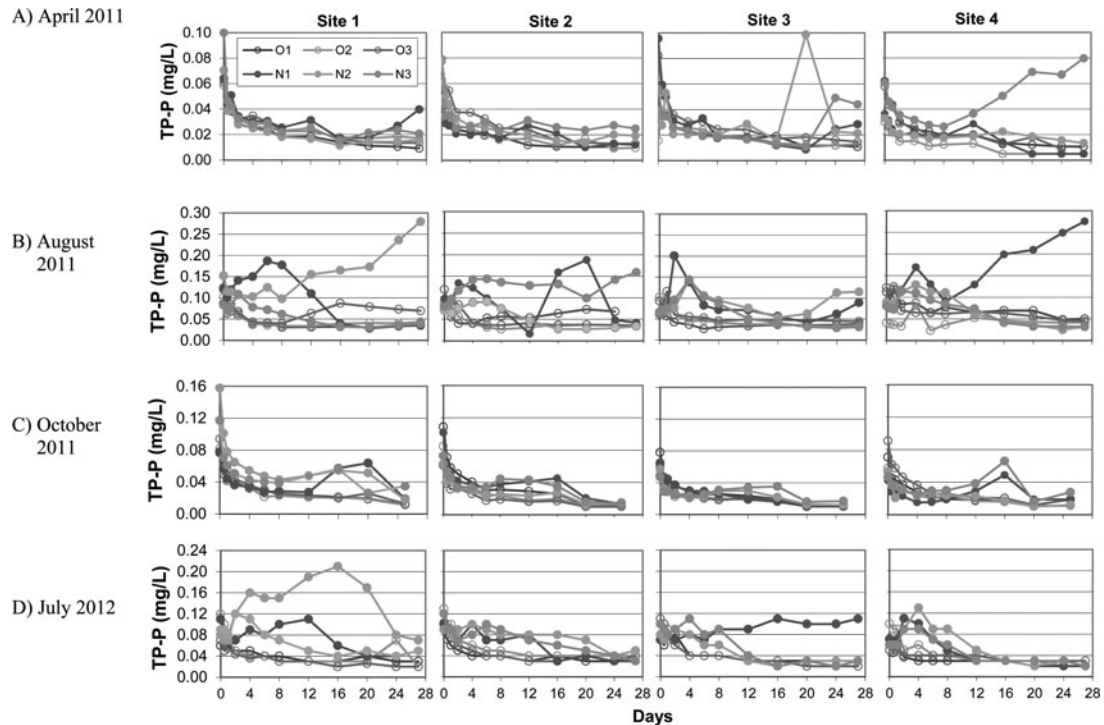


Figure 3. Total phosphorus (TP) concentrations in the water overlying sediment cores collected from 4 Bear Lake sites in 2011 and 2012. The letter in the legend refers to redox state (N = nitrogen, anoxic treatment; O = oxygen, oxalic treatment); the number refers to replicate number (1–3).

Table 3. Maximum apparent TP release rates (mg/m²/day) from sediment cores collected in Bear Lake and incubated under anoxic and oxic conditions.

| Date | Site | Anoxic Mean \pm SD | Oxic Mean \pm SD |
|-----------------------|------------|-------------------------|-----------------------|
| April 2011 | 1 | 0.38 \pm 0.38 | 0.00 \pm 0.00 |
| | 2 | 0.34 \pm 0.07 | 0.00 \pm 0.00 |
| | 3 | 0.69 \pm 0.66 | 0.00 \pm 0.00 |
| | 4 | 0.23 \pm 0.41 | 0.00 \pm 0.00 |
| | Grand Mean | 0.41 \pm 0.20 | 0.00 \pm 0.00 |
| August 2011 | 1 | 5.15 \pm 1.62 | 0.78 \pm 0.75 |
| | 2 | 3.49 \pm 1.83 | 0.33 \pm 0.33 |
| | 3 | 11.40 \pm 9.88 | 0.23 \pm 0.20 |
| | 4 | 6.72 \pm 1.34 | 1.75 \pm 2.28 |
| | Grand Mean | 6.69 \pm 3.41 | 0.77 \pm 0.70 |
| October/November 2011 | 1 | 0.66 \pm 0.33 | 0.11 \pm 0.19 |
| | 2 | 0.48 \pm 0.17 | 0.00 \pm 0.00 |
| | 3 | 0.37 \pm 0.34 | 0.00 \pm 0.00 |
| | 4 | 0.94 \pm 0.40 | 0.00 \pm 0.00 |
| | Grand Mean | 0.61 \pm 0.25 | 0.03 \pm 0.05 |
| July 2012 | 1 | 3.95 \pm 3.48 | 0.55 \pm 0.69 |
| | 2 | 1.79 \pm 0.16 | 0.24 \pm 0.21 |
| | 3 | 3.22 \pm 0.51 | 0.35 \pm 0.61 |
| | 4 | 6.72 \pm 1.83 | 0.00 \pm 0.00 |
| | Grand Mean | 3.92 \pm 2.07 | 0.28 \pm 0.51 |

During each core incubation period, maximum apparent anoxic release rates were significantly greater than oxic release rates (All: $P < 0.001$, $n = 12$; April 2011: $T = 198.0$; August 2011: $T = 218.0$; July 2012: $T = 221.5$; October 2012: $T = 214.5$; Table 3). Under anoxic conditions, the TP release rates were significantly greater during the summer incubations ($P < 0.001$, $H = 36.352$, $df = 3$), with grand means of 6.69 mg/m²/d in August 2011 and 3.92 mg/m²/d in July 2012 (Table 3) compared to the other seasons. There were no statistically significant differences in anoxic TP release rates among sites. Oxic TP release rates were low, ranging from grand means of 0 mg/m²/d in April 2011 to 0.77 mg/m²/d in August 2011, with no significant differences among site or season (Table 3).

Sediment chemistry

Sediment TP was not significantly affected by redox treatment, and therefore data analysis was performed on grand means from each site for each sampling date ($n = 6$ cores per site). Post-incubation sediment TP varied both among sites and sampling dates (Fig. 4). Among sites, Site 4 had the lowest mean sediment TP (as a function of dry weight) during all sampling events; among dates, sediment TP was significantly lower in July 2012 than any other sampling event (Fig. 4). Mean sediment Fe:P ratios ranged from 45 to 80, except at Sites 1 and 2 in April 2011, which had average Fe:P ratios of 22 and 25, respectively (Fig. 5).

Annual internal phosphorus loading

Estimates of annual internal P loading ranged from the most conservative value of 77 kg/yr (169 lbs/yr) to the most liberal value of 876 kg/yr (1931 lbs/yr) (Table 4). The lower bound of 77 kg/yr is likely overly conservative given that it assumes a completely oxic environment and our diel DO measurements indicated periodic hypoxic conditions. Conversely, the highest estimate, which assumes a completely hypoxic environment, is an implausible scenario given that our DO data clearly indicate oxic conditions through much

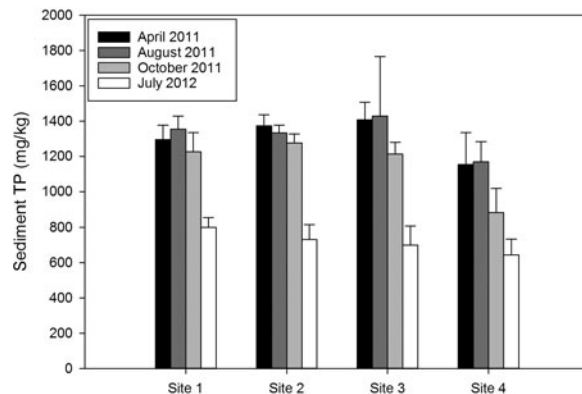


Figure 4. Mean (\pm SD) total phosphorus (TP) content of dry sediment in sediment cores collected in 2011 and 2012. Mean values include oxic and anoxic treatments ($n = 6$).

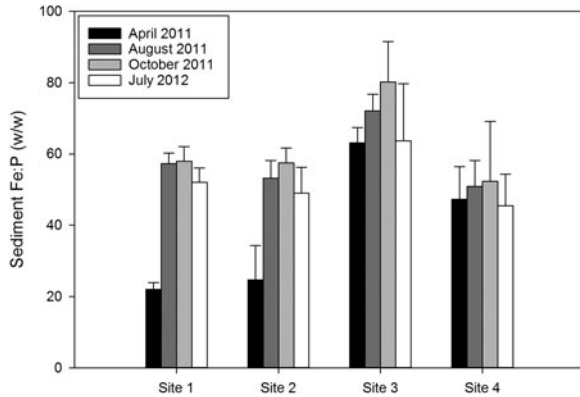


Figure 5. Mean (\pm SD) Fe:P mass ratios in sediment cores collected from Bear Lake, measured at the end of incubation. Mean values include oxic and anoxic treatments ($n = 6$).

of the lake. The annual internal P loading estimate from the Bear Lake TMDL (MDEQ 2008) is ~ 7 times greater than our load estimates based on only the deepest areas (>3 m) becoming hypoxic, ~ 3 times greater than our load estimates based on areas >2.7 m becoming hypoxic, and ~ 3 times greater than based on the polymictic estimate (Table 4).

Discussion

Implicit in the development and approval of TMDLs is that they are based on credible and accurate data. While efforts are being directed at developing a quantifiable pollutant load crediting program for meeting TMDL targets (Grismer 2013), less attention is focused on whether the original targets, and the data used to develop them, are appropriate.

In the current study, the 2008 TMDL for P in Bear Lake states that, on an annual basis, internal P loads account for 46%

Table 4. Annual internal TP load estimates in kg/yr (lbs/yr) for Bear Lake under 5 different redox and mixing scenarios.

| Scenario | Annual Internal TP Load |
|---|-------------------------|
| 1. Oxic: entire lake | 77 (169) |
| 2. Hypoxic: depths > 3.0 m; rest of lake oxic | 102 (224) |
| 3. Hypoxic: depths > 2.7 m; rest of lake oxic | 233 (513) |
| 4. Polymictic lake estimate (modified from Nürnberg et al., 2012) | 217 (478) |
| 5. Hypoxic: entire lake | 876 (1931) |
| TMDL estimate (MDEQ 2008) | 702 (1548) |

of the entire TP load to Bear Lake (i.e., 702 of 1536 kg/yr [1548 of 3387 lbs/yr]). To reduce the water column TP concentration from an average of $44 \mu\text{g/L}$ to the goal of $30 \mu\text{g/L}$ called for in the TMDL, internal P loading would need to be reduced by 79%, from 702 to 146 kg/yr (1548 to 322 lb/yr). While there are numerous lake management strategies that can be employed to reduce internal loading (Cooke et al. 2005), it is prudent to ensure the internal loading estimates used in the TMDL are accurate given the cost and potential ecological disruption associated with these strategies.

In the 2008 TMDL, the existing condition for internal P load was calculated using the Nürnberg (1988) regression equation based on measured sediment TP concentrations. The TMDL assumes that internal loading occurs from May through August at the rate of $0.0034 \text{ g/m}^2/\text{d}$ (derived from Nürnberg 1988) because summer months have a high potential sediment P release due to warmer temperatures that stimulate enzymatic activities (Holdren and Armstrong 1980, Steinman et al. 2009a, Nürnberg et al. 2012). The TMDL acknowledges uncertainty in the calculation given the possibility of diel DO cycles that could stimulate P release and that sediments in shallow Bear Lake are susceptible to resuspension to the overlying water column via recreational activities (boat propellers) or wind-induced mixing, which also can contribute to internal loading (Steinman et al. 2006, Søndergaard et al. 2013, Tammeorg et al. 2013). Consequently, the TMDL states that more accurate estimates of internal loading could be obtained by conducting internal loading analysis on sediment cores incubated under both oxic and anoxic conditions and comparing these results to estimated rates using Nürnberg's regression equation. This observation was the impetus for the current study.

Our experimental incubations indicated that Bear Lake sediments have the potential to release P at rates comparable to those in mesotrophic to eutrophic lakes (Nürnberg and LaZerte 2004, Christophoridis and Fytianos 2006, Haggard et al. 2012); however, anoxic conditions are necessary for this release to occur. Anoxia can result in diffusion of dissolved P from the sediments, as P bound to Fe oxides and oxyhydroxides ($\text{Fe} \sim \text{PO}_4$) under oxic conditions is released when the Fe is reduced (i.e., $\text{Fe}^{3+} \rightarrow \text{Fe}^{2+}$) under anoxic or hypoxic conditions (Boström et al. 1982). If Bear Lake was entirely anoxic throughout the year (Scenario 5), we estimate the annual internal P load would exceed the TMDL-derived internal load by $\sim 20\%$ (876 vs. 702 kg/yr [1931 vs. 1548 lbs/yr]). However, both our daytime and diel measurements clearly indicate that Bear Lake is mostly oxic throughout the water column, at least to the near-bottom, which is as close to the sediment–water interface that our probes could be deployed without disturbing the sediment layer. Hence, we view the annual internal P load estimate under Scenario 5 (entirely anoxic) as being unrealistic; however,

this estimate provides a theoretical upper boundary in Bear Lake.

At the other end of the spectrum is the most conservative annual internal P loading estimate (Scenario 1), whereby Bear Lake remains entirely oxic throughout the year. In this case, oxidizing conditions favor phosphate remaining bound to Fe (or Al) oxides or oxyhydroxides, resulting in an annual internal P loading estimate of only 77 kg/yr (169 lbs/yr), or about 89% less than the TMDL estimate (Table 4). While this scenario is more plausible than Scenario 5, it too is unlikely because our diel DO measurements revealed periodic deep water hypoxia. Furthermore, other studies have found that even under oxygenated water column conditions, hypoxia often develops at the sediment–water interface or within the sediments themselves (Holdren and Armstrong 1980, Nürnberg 2009, Nürnberg et al. 2013).

Rather, we believe that the actual amount of internal P loading in Bear Lake falls somewhere within the range of Scenarios 2 to 4. Scenario 2 assumes only 3% of Bear Lake becomes hypoxic, whereas Scenario 3 assumes almost 20% of Bear Lake becomes hypoxic. Even adjusting these estimates upward by including the margin of safety used in the TMDL (i.e., 10%), they still reflect significantly lower internal P loads than presented in the TMDL. In addition, these 2 scenarios likely overestimate internal P loading in Bear Lake because they assume continuous hypoxia in the deeper locations, and our data indicate that the hypoxia is episodic.

The annual internal load estimate in Scenario 4 is based on an approach developed by Nürnberg et al. (2012) for a lake in Finland. The Finnish lake was shallow and polymictic, similar to Bear Lake, but much larger in size and with lower water column P concentrations. Nürnberg et al. (2012) reasoned that even though a lake may be well-mixed, the sediment–water interface will still approach hypoxic conditions at times, despite oxic conditions in the overlying water column (Holdren and Armstrong 1980, Nürnberg 2009). This approach uses lake morphometry as a proxy for estimating the lake area that is actively releasing P from the sediment. Even with our modifications to the Nürnberg et al. (2012) approach, which result in greater load estimates by adding in spring release rates, the internal load estimate with this approach for Bear Lake was only 217 kg/yr, approximately the same as in Scenario 3.

Given that 4 of our 5 scenarios (Table 4) were lower than the annual internal P load estimate presented in the TMDL, and the one scenario that exceeded the TMDL is not realistic, we conclude the TMDL estimate for annual internal P load in Bear Lake is too high. A more reasonable estimate for the annual internal P load in Bear Lake falls between 102

and 233 kg/yr (224 and 513 lbs/yr), a range that addresses uncertainty in an ecologically meaningful way.

There are several possibilities why internal P loading is not more problematic in Bear Lake, given that the sediments have relatively high TP concentrations. First, our lab incubations show the sediments in Bear Lake release P at relatively low rates compared to those in other west Michigan lakes (Table 5); the P release rates in Bear were similar to those in White Lake, where internal loading was found to be a relatively minor source to the lake's overall P budget (Steinman et al. 2009b). Second, the well-mixed water column seems to keep the lake relatively free of hypoxic conditions. Although we did observe hypoxia during our diel DO measurements, it was episodic and restricted to the deepest part of the lake. We acknowledge that anoxic conditions may exist at the sediment surface when quiescent conditions (e.g., early morning) form, stimulating P release from the sediment even though the overlying water contains DO. However, the return of oxic conditions at the sediment–water interface will result in the binding of phosphate to Fe and aluminum oxides and oxyhydroxides. Third, Bear Lake sediments are Fe-rich, which can help keep phosphate bound in the sediment. The release of SRP from oxic sediments, like those found in Bear Lake, is low in lakes with Fe:TP ratios (by weight) >15 (Jensen et al. 1992). Rydin et al. (2000) cautioned that this ratio should be applied with care because total Fe and TP includes forms of each that may not be available for adsorption. Forms of mobile P (Pilgrim et al. 2007), consisting of the loosely sorbed and Fe-P redox sensitive fractions, are most likely to contribute to internal P loading. Nonetheless, the mean total Fe:TP ratios in Bear Lake ranged from 22 to 80, easily exceeding the 15 threshold. The relatively high ratios found in Bear Lake suggest that Fe in Bear Lake sediments is sufficient to deter the release of P from oxic sediments.

Our results are subject to several caveats. Our diel oxygen and P release analyses are based on relatively few sites and dates. Although we sampled different seasons and over multiple years, it would be preferable to have continuous DO data (Smith et al. 2011) to provide detailed redox chemistry at highly resolved time periods and depths at the sediment–water interface. High rates of primary production from abundant algal communities can result in elevated daytime DO due to photosynthesis, but then respiration can cause depleted nighttime and early morning DO, especially under calm conditions, which in turn could lead to short-term release of P from the sediments (Premazzi and Provini 1985, Nürnberg 2009, Nürnberg et al. 2013). Hence, the episodic oxygenation of the hypolimnion via water column mixing and diel DO fluctuations has implications for internal P loading rates in Bear Lake, both in terms of redox-driven

Table 5. Maximum apparent TP release rates (mg/m²/day) measured during summer from sediment cores collected from Bear Lake and three other west Michigan lakes: Mona Lake, Spring Lake, and White Lake.

| Lake | Anoxic | Oxic | Source |
|------------------------|------------------|-----------------|-----------------------|
| | Mean \pm SD | Mean \pm SD | |
| Bear Lake (2011) | 6.69 \pm 3.41 | 0.77 \pm 0.70 | This study |
| Bear Lake (2012) | 3.92 \pm 2.20 | 0.28 \pm 0.51 | This study |
| Mona Lake (2006, 2007) | 11.38 \pm 3.90 | 0.83 \pm 0.40 | Steinman et al. 2009a |
| Spring Lake (2003) | 17.97 \pm 8.07 | 0.03 \pm 0.44 | Steinman et al. 2004 |
| White Lake (2007) | 3.75 \pm 2.77 | 0.02 \pm 0.14 | Steinman et al. 2009b |

and resuspension-driven P flux. In addition, our P release rate measurements do not account for sediment resuspension because the sediment cores remain intact during the incubations. It is unlikely that porewater would be a major source of P given the low SRP concentrations in the Bear Lake porewater (usually < 20 μ g/L; data not shown). However, phosphate ions desorbing from sediment, once in suspension, could possibly be a significant source of P (Søndergaard et al. 2003, Istvánovics et al. 2004, Steinman et al. 2006), although prior studies also have shown that the effect of sediment resuspension is extremely variable in space and time (Cyr et al. 2009, Niemistö et al. 2012).

In conclusion, because the existing internal P load condition used to establish the Bear Lake TMDL was an overestimate, the 79% reduction in internal P loading prescribed in the TMDL is likely not justified. It is also plausible that other sources of internal P loading, such as sediment resuspension and diel migrating phytoplankton that translocate P to and from the sediment (Barbiero and Welch 1992), may be important. Because these sources were not measured, however, the degree of overestimation of the TMDL allocation for internal loading is uncertain. Our scenario analysis indicates that the internal P load is already meeting the reduction goal, without implementing any management action; therefore, management efforts aimed at reducing the water column P concentration to reach the TMDL target should instead be directed at controlling the external P load.

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