2006

An Analysis of Internal Phosphorus Loading in White Lake

Alan Steinman
Annis Water Resources Institute

Mary Ogdahl
Annis Water Resources Institute

An Analysis of Internal Phosphorus Loading in White Lake

Submitted by:

Alan Steinman, Ph.D. and Mary Ogdahl

Annis Water Resources Institute
Grand Valley State University
Lake Michigan Center
740 W. Shoreline Dr.
Muskegon, MI 49441

EXECUTIVE SUMMARY

An analysis of internal phosphorus loading in White Lake, MI was conducted during the summer, 2006. Sediment cores were removed from 4 sites in White Lake and incubated in the laboratory under aerobic (with oxygen) and anaerobic (without oxygen) conditions. Phosphorus flux from the sediments into the overlaying water column was measured over a 27-day period and compared to rates measured from sediment cores collected previously from Mona and Spring Lakes.

Internal phosphorus loading in White Lake sediments ranged from 1.55 to 7.78 mg $TP/m^2/d$ in anaerobic conditions and from -0.18 to 0.14 $mg/m^2/d$ in aerobic conditions. The negative value suggests that the sediments in some areas of White Lake could act as a sink for TP during aerobic periods. These internal loading rates were generally similar to summer measurements made in Mona Lake in previous years but considerably lower than those measured under anaerobic conditions in Spring Lake in 2003.

Internal total loading contributed 1.24 tons of phosphorus based on our laboratory study, with about half coming from the eastern-most basin in White Lake. Compared to an estimated external total phosphorus load of 15.48 tons/yr (Mark Luttenton, GVSU, unpublished data), internal loading of TP accounts for ~7.4% of the total TP load entering White Lake. This percent is much lower than what has been measured in Spring and Mona Lakes, suggesting internal phosphorus loading is a less important process in White Lake than nutrients entering from groundwater or surface inflows. These data suggest that management strategies should be focused, at least initially, on reducing external phosphorus loads to White Lake.

INTRODUCTION

White Lake, one of several coastal lake systems unique to the West Michigan region, has been listed as a Great Lakes Area of Concern (AOC). The White Lake Remedial Action Plan lists 8 beneficial use impairments (BUIs) for this system, including eutrophication and undesirable algae. Over the past several decades, land use changes within the watersheds providing drainage to Great Lakes drowned river mouth systems have accelerated nutrient loading to these systems. In the White Lake watershed, nutrient loading from the White River was documented as early as the 1960s, when it was estimated that 98% of total phosphate and 96% of total nitrogen entered White Lake via the White River (Freedman et al. 1979). Furthermore, White Lake retained 66–76% of phosphate and between 5–51% of nitrogen received from the White River (Freedman et al. 1979). These data have generally led to the conclusion that most nutrient-based water quality problems in White Lake are due to nutrient loading from the White River.

However, recent changes in White Lake have initiated a reevaluation of the environmental factors and processes that have primary influence over various biological components within White Lake. For example, some improvement in water quality was noted following the diversion of sewage and industrial discharges to the Whitehall treatment system (RAP 1995). In addition, zebra mussels, which reached significant population densities during the early to mid-1990s, have reduced suspended material and increased water clarity. However, lakeshore development has increased and has moved from seasonal residential use to year-round use, resulting in more pressures on the system. Finally, there has been more than a two-fold increase in the maximum biomass of aquatic plants in White Lake between the mid-1970s and 1995 (Luttenton 1996), with much of the increase probably occurring during the 1990s. In

combination, these changes suggest that the system has experienced significant modification during the past 2 decades. These apparent changes, coupled with the fact that White Lake is a Great Lakes Area of Concern, are compelling reasons to reevaluate the nutrient conditions and determine the sources of nutrient loading in the lake.

Attempts to limit or reduce external loading to water bodies have resulted in smart growth and low impact initiatives in many municipalities. By detaining and retaining water on site, there is a greater opportunity for pollutants to be captured before they enter local streams and lakes (Steinman and Rosen 2000). These watershed-based practices are founded on the premise that external nutrient loads are the primary source of nutrients entering surface water bodies, and their control will result in improved water quality. However, in systems where internal nutrient loads are important, nutrient control efforts that are focused solely at the watershed-level will solve only part of the problem. Increased development, changing land uses, and heavy upstream agricultural activity all contribute to heavier nutrient loads to White Lake; retention of these nutrients in the sediments could lead to high rates of internal nutrient loading in White Lake. Ultimately, an integrated approach that examines source control in a holistic way is needed, especially in urbanizing systems where multiple stressors exist (Paul and Meyer 2001, Walsh et al. 2005, Steinman et al. 2006).

In this study, we examine the spatial variability of internal phosphorus loading in White Lake. This watershed faces many of the environmental and socio-economic challenges common to watersheds throughout the developed world (Steinman et al. 2006). This study complements a separate analysis of external nutrient loading conducted by Dr. Mark Luttenton.

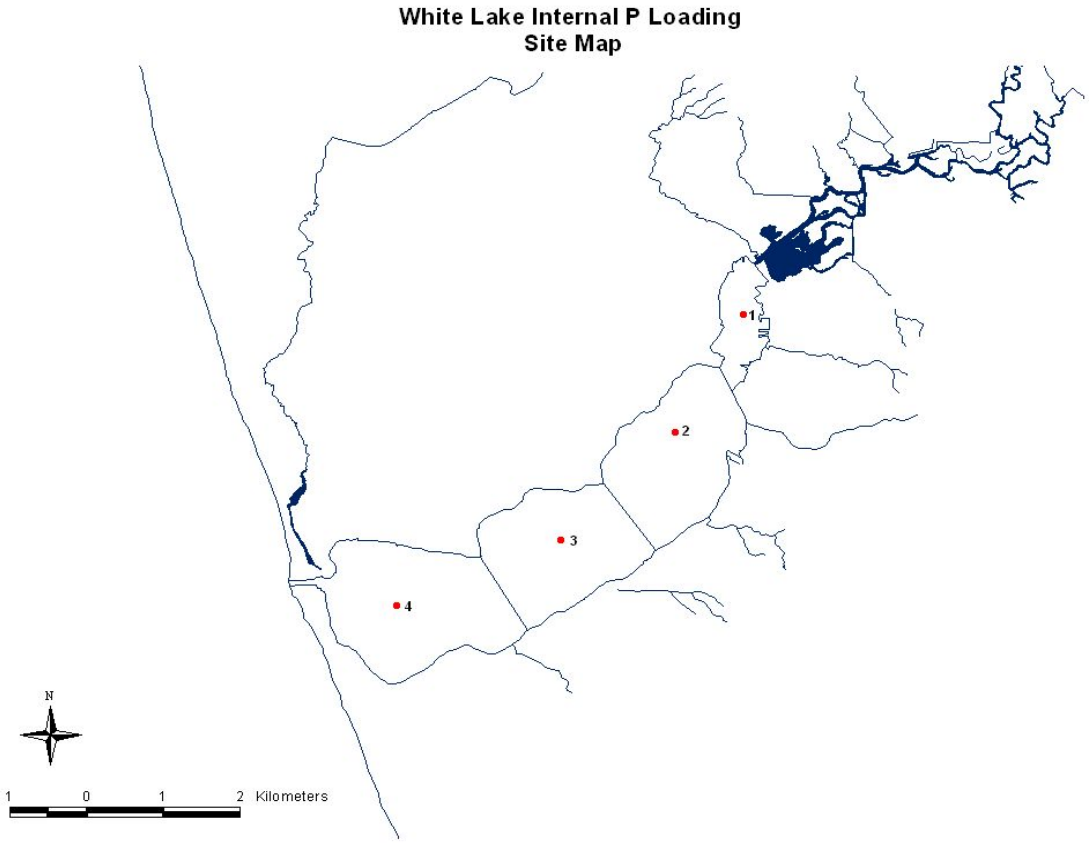
METHODS

Field Methods: Sediment cores were collected in June, 2006 from 4 sites (Fig. 1). At each site, dissolved oxygen, pH, temperature, specific conductance, total dissolved solids, and chlorophyll *a* were measured at 3 depths (top, middle, and bottom) using a Hydrolab DataSonde 4a equipped with a Turner Designs fluorometer. Water samples for phosphorus analysis were collected with a Van Dorn bottle. Water for soluble reactive phosphorus (SRP) analysis was syringe-filtered immediately through 0.45-μm membrane filters into scintillation vials. Samples were stored on ice until transported to the laboratory, always within 5 h of collection. TP samples were stored at 4°C and SRP samples were frozen until analysis. SRP and TP were analyzed on a BRAN+LUEBBE Autoanalyzer (US EPA 1983). SRP values below detection were calculated as ½ the detection limit (5 μg/L).

Sediment core sampling and laboratory incubation were conducted in summer, 2006. Sediment cores were collected from 4 sites (Fig. 1). Six sediment cores were collected from each site using a piston corer (Fisher et al. 1992, Steinman et al. 2004). The corer was 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 that was ~20 cm in length, with a 25-cm overlaying water column. The piston was then bolted to the top of the core tube to keep it stationary during transit. Core tubes were placed in a vertical rack and maintained at ambient temperature during

transit. An additional core was collected from each site and the top 5 cm removed for the following sediment analyses in the lab: bulk density, AFDM, TP on the ashed material, and metals (Fe, Ca, Mg).

Figure 1. Map of White Lake, showing sampling locations (1-4) of the sediment cores. White Lake is divided into four basins based on depth distributions. The area of each basin was used as a weighting function when summing the total internal phosphorus load in White Lake.



Laboratory methods. The 24 cores (6/site) were placed in a Revco environmental growth chamber, with the temperature maintained to match ambient bottom water conditions in White Lake at the time of collection. The water column in 3 of the 6 cores from each site was bubbled with N_2 (with 330 ppm CO_2) to create buffered anaerobic conditions, while the remaining 3 were bubbled with oxygen to create aerobic conditions.

Internal load estimates were made using the methods outlined in Moore et al. (1998), with minor modifications (Steinman et al. 2004). Briefly, a 40-mL water sample was removed by syringe through the sampling port of each core tube at time 0, and at 2 h, 12 h, 1 d, 2 d, 4 d, 8 d, 12 d, 16 d, 20 d, 24 d, and 27 d after time 0. The 40-mL subsample was replaced with filtered water collected (at the same time as the cores were removed) from the corresponding site in the lake; this maintained the original volume in the core tubes. 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 soluble reactive P (SRP). P was analyzed as described previously.

Flux (P release rate) calculations were based on the increase in water column TP or SRP using the following equation (Steinman et al. 2004):

$$P_{rr} = (C_t - C_0)\, V/A$$

where, P_{rr} is the net P release rate or retention per unit surface area of sediments, C_t is the TP or SRP concentration in the water column at time t, C_0 is the TP or SRP 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 release rate was calculated from the time when P concentrations stabilized in the water column (~day 1) until the time when an asymptote was reached. For this report, only the TP internal loading data are presented; patterns for TP and SRP were similar, although the magnitude of P release was much lower for SRP than TP.

Following the incubations, the top 5 cm of sediment was removed from each core. The sediment was homogenized and subsampled for metals (Fe, Ca, Mg) analysis and AFDM. The ashed material was analyzed for TP as described previously. Another subsample (5 g) from the wet sediment was centrifuged to remove excess porewater and sequentially fractionated (Moore

and Reddy 1994) to determine the fraction of phosphorus bound to iron and calcium minerals in the sediments. Porewater was filtered, frozen, and analyzed for SRP as described previously. Residual sediment was shaken for 17 h with 0.1M NaOH and centrifuged. The supernatant was drawn off, filtered, and frozen, and subsequently analyzed for SRP. This fraction is referred to Al- and Fe-bound phosphorus and represents a mineral association that can become soluble under anoxic conditions. After this extraction, the sediment was shaken for 24 h with 0.5M HCl, centrifuged, and the supernatant filtered, frozen, and subsequently analyzed for SRP. This fraction is referred to as Ca- and Mg-bound phosphorus and represents a stable mineral association.

Total internal load calculations were made by 1) averaging the internal load at each site under anaerobic and aerobic conditions, 2) extrapolating the mean areal rate of the cores to the area of each subbasin, and 3) summing the fluxes in each of the four subbasins. External load calculations were made as part of a separate study conducted by Dr. Mark Luttenton of AWRI-GVSU, and include loads from the White River (the major tributary to White Lake) and groundwater inflow directly to White Lake.

RESULTS

There was a distinct depth gradient in the sampling sites. Site 1, located farthest to the east (Fig. 1), was the shallowest and also had the lowest secchi depth (i.e., least transparent). Sites became progressively deeper and more transparent as one moved west (Table 1). Stratification of the water column, based on temperature and dissolved oxygen concentration, was evident at sites 3 and 4, incipient at site 2, and not evident at site 1, consistent with its shallow depth (Table 1). Elevated total phosphorus concentrations were measured near the water column bottom at sites 3 and 4, where hypoxic conditions were present (Table 1). At sites 1 and

2, where near bottom DO concentrations were still relatively high, TP concentrations were very similar between the water surface and bottom. This suggests that iron remained in the oxidized form at sites 1 and 2, where it was bound to phosphorus in the sediment and prevented the P from diffusing into the water column (cf. Bostrom et al. 1982).

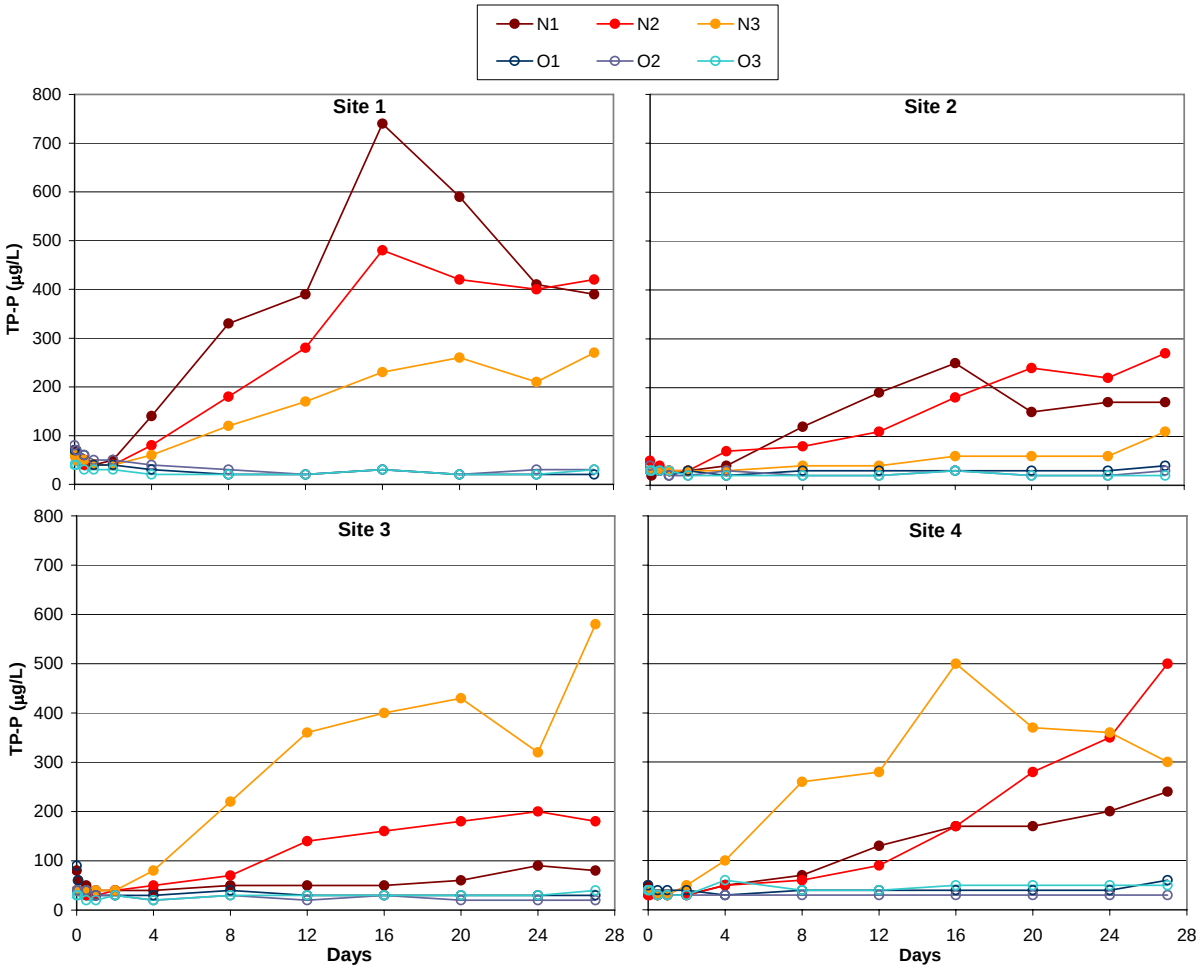
Table 1. Selected limnological characteristics of sampling sites in White Lake. ND = no data.

Parameter		**Site**			
		1	2	3	4
Depth (m)		3.2	10.2	13.5	16.1
Secchi Depth (m)		1.6	2.2	2.25	2.6
Chl a (µg L^{-1})	Surface	4.5	8.1	7.8	6.4
DO (mg L^{-1})	Surface	7.40	7.84	8.09	8.09
	Bottom	6.72	2.07	0.89	1.11
Temperature (°C)	Surface	21.89	23.14	23.3	23.29
	Bottom	20.62	18.22	15.24	14.47
TP (µg L^{-1})	Surface	30	30	20	20
	Bottom	30	20	420	80
SRP (µg L^{-1})	Surface	9	6	16	5
	Bottom	9	7	ND	21

Redox state had a statistically significant influence on TP flux, with TP flux significantly greater in the anaerobic treatments than in the aerobic treatments (2-way ANOVA: $F = 70.324$; $P < 0.001$). Site had no significant affect on TP flux ($P > 0.18$). The interaction term between site and redox state was significant ($F = 4.021$; $P < 0.026$), as the influence of redox was more evident at sites 1, 3, and 4 than at site 2 (Fig. 2).

In the anaerobic treatments, TP concentrations peaked between days 16 and 27 (Fig. 2), and averaged 480, 180, 280, and 350 µg/L at sites 1-4, respectively. In the aerobic treatments, TP concentrations stayed very low, and declined slightly over time in several of the core tubes (Fig. 2). The mean TP concentrations on day 27 at sites 1-4 were 30, 30, 30, and 35 µg/L, respectively.

Figure 2. TP concentrations released from sediment cores from 4 sites in White Lake sampled in summer, 2006. The first number in the legend refers to redox state (N = nitrogen, anaerobic condition; O = oxygen, aerobic condition); the second number refers to replicate number (1-3).



Mean diffusive flux rates of TP ranged from 1.55 to 7.78 mg TP/m^2/d in anaerobic conditions and from -0.18 to 0.14 mg/m^2/d in aerobic conditions (Table 2). The negative value at site 1 suggests that the sediments could act as a sink for TP during aerobic periods. Flux rates were generally similar to summer measurements made in Mona Lake in previous years but considerably lower than those measured under anaerobic conditions in Spring Lake in 2003 (Table 2).

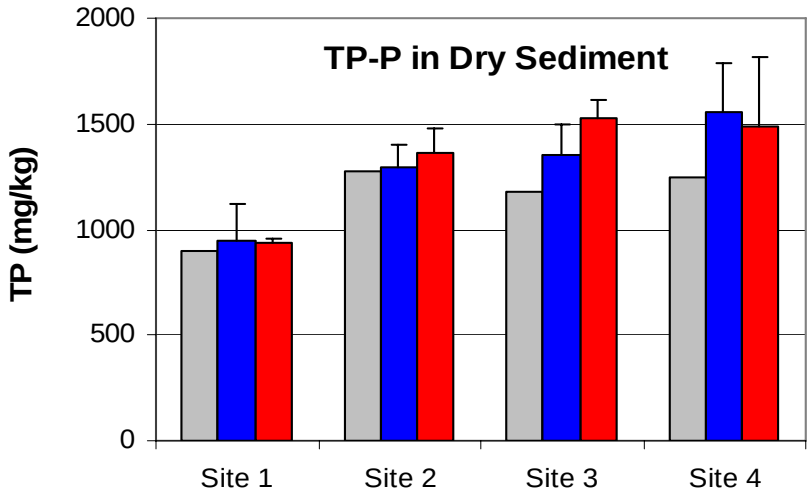
Table 2. Mean flux rates of TP from sediment cores collected from White Lake during summer, 2006 compared to Mona Lake (summer, 2004 and 2005).

TP Flux Rate (mg P/m^2/d)		
Site	**Anaerobic**	**Aerobic**
	White Lake (this study)	
1	7.78	-0.18
2	1.55	0.07
3	2.46	0.03
4	3.21	0.14
	Mona Lake (2004)	
1	6.46	-1.84
2	5.38	-2.41
3	13.63	0.99
4	12.82	0.30
	Mona Lake (2005)	
1	4.16	0.22
2	5.57	0.21
3	4.85	0.52
4	3.19	0.19
	Spring Lake (2003)	
1	26.71	0.40
2	16.02	-2.00
3	9.04	0.16
4	10.64	-1.04

The TP concentration in the sediment cores (as function of dry weight) prior to incubation ranged from 899 (Site 1) to 1280 mg/kg (Site 2; Fig. 3). No inferential statistics were applied to these data as replicate cores were not sampled at each site. These numbers are similar to mean concentrations measured in Spring Lake (1282 mg/kg) and Mona Lake (1394 mg/kg, excluding the site near the mouth of Little Black Creek, which had a TP concentration of 4893 mg/kg). At the end of the incubation, site-specific differences in TP in White Lake were still evident and highly significant (2-way ANOVA: $F = 12.852$, $P < 0.001$), with Site 1 having a lower mean percentage compared to all other sites (Fig. 3). Redox status had no statistical significance on the %TP concentration of the sediment analyzed at the end of the laboratory

incubation (F = 0.291; P = 0.597). The interaction between site and redox was not statistically significant (F = 0.530, P = 0.668).

Figure 3. TP concentration in dry sediment (mg/kg) from summer 2006 sediment cores. Gray bar = initial sample (prior to incubation); blue bar = end of incubation under aerobic conditions; red bar = end of incubation under anaerobic conditions.

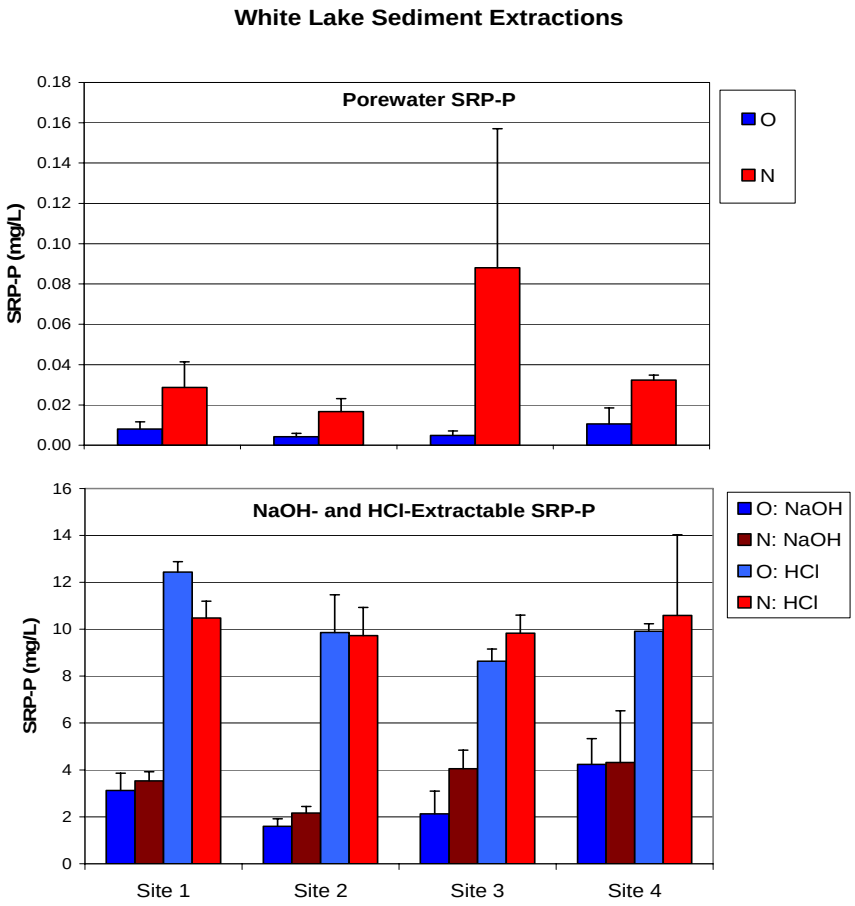


SRP in the porewater at the end of the incubations ranged from below detection to an anomalously high concentration of 0.167 mg/L in one core from Site 3 (Fig. 4). Overall, site had no statistically significant effect on SRP porewater concentration; the high variance in the data at Site 3 prevented statistical differences, despite the apparently higher mean concentrations at that site. Redox was statistically significant, however; porewater SRP was greater under anaerobic than aerobic conditions (2-way ANOVA: F = 44.817, P < 0.001; Fig. 4 top panel). The interaction term between site and redox was not statistically significant (F = 1.818, P = 0.184).

There were clear differences between the NaOH-extractable and HCl-extractable fractions of SRP, regardless of site (Fig. 4 bottom panel), with the SRP from the NaOH extraction significantly lower than the SRP from the HCl extraction (3-way ANOVA: F = 363.652, P < 0.001). The mean NaOH-extractable SRP fraction was 3.14 mg/L, while the mean HCl-extractable SRP fraction was 10.2 mg/L. Site also was statistically significant (3-way

ANOVA: F = 4.470, P = 0.01), with Sites 1 and 4 having significantly more extractable SRP than Site 2 (Fig. 4). The interaction term between site and extraction type was marginally significant (F = 2.377, P = 0.088), as the effect of site was influenced by the type of extraction; the significant effect of site appeared to be much more prominent in the NaOH fraction than the HCl fraction (Fig. 4). Redox had no significant effect on extractable SRP (F = 0.869, P = 0.358), and the redox x extract and site x redox x extract interactions terms were non-significant, as well.

Figure 4. Top panel = Porewater SRP concentrations in sediment cores at the end of the incubation period. Blue bar = aerobic incubation conditions; red bar = anaerobic incubation conditions. Bottom panel = NaOH- and HCl-extractable SRP concentrations from sediment cores at the end of the incubation period. Blue-toned bars = aerobic incubation conditions; red-toned bars = anaerobic incubation conditions.



Lake-wide Internal Loading:

Sediment cores from the eastern-most subbasin contributed potentially 2.5 to 5X more TP than any of the other subbasins (Table 3) based on the anaerobic laboratory results. However, in nature, the shallow depth in this subbasin results in a well-mixed water column, keeping the sediments frequently aerobic, which likely minimizes the release of phosphorus that would otherwise be disassociated from reduced iron. Therefore, the estimates in Table 3 should be viewed as maximum potential rates, and are likely overestimates compared to natural conditions. More dissolved oxygen data are needed, on both spatial and temporal (seasonal and diel) bases, to reduce the uncertainty in these flux estimates. The internal loads from sub-basin 4 were the second highest, but it is unclear if the released P actually reaches the epilimnion (upper waters) of the lake given the depth at this site (and presumably deeper point of stratification of the column). If the released P stays in the hypolimnion, then it may not cause ecological impairments, such as algal blooms. Again, more information is needed on the hydrodynamics of White Lake, to determine if bottom circulation patterns move the water from deeper reaches to shallower sub-basins, where it can then be used by algae. Internal TP loads were generally very low under aerobic conditions (Table 3), as expected when the phosphorus dynamics are being driven by iron biogeochemistry.

Table 3. Potential mean internal TP load estimates (tons) by site for summer, 2006 in White Lake. See methods for description of calculation.

Sub-basin	Anaerobic	Aerobic
1	0.640	-0.014
2	0.128	0.006
3	0.203	0.003
4	0.264	0.011
Sum	**1.235**	**0.006**

Internal vs. External Loading Analysis:

In a separate analysis conducted by Dr. Mark Luttenton of AWRI-GVSU, estimates of external TP loads were made for the White River and minor tributaries flowing directly into White Lake (13.05 tons/yr), as well as groundwater (based on well data from both south and north sides of White Lake: 2.43 tons/yr). Results from this current study revealed a potential internal TP load of 1.24 tons/yr. Summing all these sources resulted in a total potential TP load of 16.72 tons/yr. Based on these data, internal loading of TP accounts for ~ 7.4% of the total TP load entering White Lake (Table 4).

Table 4. Internal and external TP load estimates (tons/yr).

Potential Internal TP load	External TP load	Internal load (% of total)
1.24	15.48	7.4%

DISCUSSION

Internal P loading can be a significant source of nutrients in shallow, eutrophic lakes, and can result in serious impairment to water quality (Welch and Cooke 1995, 1999; Steinman et al. 1999, 2004; Søndergaard et al. 2001; Nürnberg and LaZerte 2004). This process has both ecological and societal implications; internal loading rates can be sufficiently great that reductions in external loading fail to improve water quality. If these reductions in external loading require the investment of money and create expectations of success, the resulting disappointment (or worse) from stakeholders at the failure to improve lake conditions can set back future restoration activities and harm the reputation and credibility of natural resource managers.

Water column TP concentrations were lower in White Lake compared to other local lakes. TP levels measured near the water surface in White Lake were 20 to 30 µg/L, compared to summer surface concentrations of 30 to 60 µg/L in Mona Lake (Steinman et al. submitted), and 60 to 120 µg/L in Spring Lake (Steinman et al. 2004). However, near-bottom TP concentrations in White Lake were similar to those measured in the other lakes; levels ranged from 20 to 420 µg/L in White Lake (although only one of the 4 sites had severely elevated concentrations), whereas near-bottom TP levels were 30 to 420 µg/L in Mona Lake and 40 to 80 µg/L in Spring Lake.

The summer TP release rates from White Lake under anaerobic conditions (1.6-7.8 mg P m^{-2} d^{-1}) were in the same general range as those measured in mesotrophic systems (~3 to 7 mg m^{-2} d^{-1}; Nürnberg and LaZerte 2004). However, these release rates are lower than what we have measured in other phosphorus-impacted lakes (Table 5). For example, median summer TP release rates from Spring Lake and Mona Lake were ~2 to 5X greater than the measured rates in White Lake. The overall amount of TP potentially released from sediments in White Lake during the summer (1.24 tons) is very similar to the summer estimates in Mona Lake (1.09 tons) but is considerably less than the estimate in Spring Lake (2.7 to 6.4 tons). The Spring Lake estimate is inflated slightly because it also includes spring internal loading, but those fluxes were very small compared to the summer (Steinman et al. 2004).

Table 5. Comparison of mean TP release rates (mg P m^{-2} d^{-1}) from coastal west Michigan lakes under anaerobic conditions.

Lake	Date of Core Collection	Minimum Release Rate	Maximum Release Rate	Median Release Rate
White Lake	June, 2006	1.55	7.78	2.84
Spring Lake	June-July, 2003	1.64	29.54	12.50
Mona Lake	July, 2004	5.38	13.63	9.64
Mona Lake	June, 2005	3.19	5.57	4.51

The lower TP release rates in White Lake also result in internal loading accounting for a much lower percentage of total TP load in White Lake compared to either Mona or Spring Lakes. Internal loading during summer months was estimated to account for 55 – 67% of the total TP load in Spring Lake (Steinman et al. 2004) and 67 – 85% of the total TP load in Mona Lake (Steinman et al. submitted), whereas it accounted for only ~7% of the total TP load in White Lake. Furthermore, the 7% value may be an overestimate, as external load estimates included only minimal storm event sampling (M. Luttenton, pers. comm.), when most of the nutrients are transported downstream.

Several reasons could account for the lower TP release rates in White Lake. First, the total phosphorus sediment concentrations are lower than in Mona and Spring Lakes. This suggests that there may be less phosphorus available for internal loading to the water column in White Lake. However, TP can be a poor predictor of internal loading because it includes both highly mobile and immobile phosphorus fractions. Analyzing the sediment by different fractions allows one to determine the relative abundance of mobile vs immobile phosphorus. The NaOH-extractable SRP fraction measures a relatively mobile phosphorus fraction: the Al- and Fe-bound phosphorus that can become soluble under anoxic conditions. In contrast, the HCl-extractable SRP fraction measures a relatively stable phosphorus fraction: the Ca- and Mg-bound phosphorus.

The mean NaOH-extractable SRP fraction in White Lake sediment was ~½ the amount in Spring Lake (3.14 mg/L vs 6.75 mg/L, respectively) and about the same as Sites 1, 2, and 4 in Mona Lake (2.9 mg/L) but considerably lower than the amount at Site 3 in Mona Lake (49 mg/L). The mean concentration of the less mobile HCl-extractable SRP fraction in White Lake sediment was ~¼ the amount in Spring Lake (10.2 vs 13.9 mg/L, respectively) and a bit greater

than at Sites 1, 2, and 4 in Mona Lake (8.9 mg/L) but again, considerably less than at Mona Lake Site 3 (63 mg/L). These data suggest that the lower levels of mobile P in White Lake sediments account, at least in part, for the lower internal P loading rates in this system.

Second, the morphometry of White Lake is very different than Mona or Spring Lakes; White Lake is deeper than the other two, so even if P flux was occurring from the sediments to the water column during periods of low DO, it is unclear how much of that P would make its way from the hypolimnion to the epilimnion, where the algae could take advantage of it. A better understanding of White Lake's hydrodynamics is needed, to know if water currents in the hypolimnion may transport P-rich bottom water from the deep sites in the western basins toward shallow sites in the eastern basin, or if this P-rich water flows out into Lake Michigan.

Our analyses included several assumptions that also need caveats. It was assumed that release rates from sediments in the core tubes were representative of sediments and conditions in White Lake. Our sampling strategy was designed to cover as much of the geographic range in White Lake as possible, given the study's constraints. However, there is likely considerable sediment variation within each sub-basin; sampling only one site per sub-basin does not allow us to estimate the importance of this variation. In addition, internal loading varies on an annual basis (Steinman et al. submitted); our analysis in summer 2006 should be viewed as a snapshot, and not as definitive.

The second assumption was that the incubation conditions were representative of natural conditions. Although the laboratory conditions mimicked the ambient temperature and light regime, clearly the hydrodynamics were altered. It is likely that the laboratory set-up for the anaerobic water column represented an optimal situation for release of P (constant anaerobic conditions) compared to natural conditions. Hence, our estimates of internal loading are likely

higher than what is occurring in nature. However, because our measurements only cover one season, the estimated internal load of 1.24 tons in White Lake probably underestimates the true annual load.

Even taking these caveats into account, the P release rates measured under anaerobic conditions indicate that internal loading is a distinct and measurable phenomenon in White Lake, albeit a relatively minor source overall of P to White Lake. Its relative importance to the overall phosphorus budget in White Lake is much less than in other regional drowned-river mouth lakes, such as Mona and Spring Lakes, which have much greater developmental pressures along their shorelines. While internal P loading should not be ignored as a source of phosphorus to White Lake, it is recommended that management strategies, at least initially, should focus on external loads (e.g. riparian buffer strips, on-farm comprehensive nutrient management plans, stormwater retention areas, and constructed wetlands/detention areas), which likely will have the greatest influence on reducing phosphorus loads to White Lake.

ACKNOWLEDGEMENTS

We are grateful to Brian Scull, Jennifer Cymbola, and Kelly Wessell for their help with field sampling, Jennifer Cymbola who also assisted with laboratory sampling, and Gail Smythe for assistance with laboratory analysis. Mark Luttenton graciously shared his external phosphorus loading data with us. We are grateful to Jim Duncan, Dave Farhat, and GVSU for helping to fund this project.

REFERENCES

Boström, B., M. Janson, and C. Forsberg. 1982. Phosphorus release from lake sediments. Arch. Hydrobiol. Beih. Ergebn. Limnol. 18: 5-59.

Fisher, M.M., M. Brenner, and K.R. Reddy. 1992. A simple, inexpensive piston corer for collecting undisturbed sediment/water interface profiles. J. Paleolimnol. 7:157-161.

Freedman, P. L., R. P. Canale, and M. T. Auer. 1979. Applicability of land treatment of wastewater in the Great Lakes area basin: Impact of wastewater diversion, spray irrigation on water quality in the Muskegon County, Michigan, lakes. U.S. EPA Report, EPA-905/9-79-006-A.

Luttenton, M. 1996. Distribution and abundance of aquatic macrophytes in White Lake. Project Report, submitted to, Muskegon County Soil Conservation District.

Moore, P.A., K.R. Reddy, and M.M. Fisher. 1998. Phosphorus flux between sediment and overlying water in Lake Okeechobee, Florida: spatial and temporal variations. J. Environ. Qual. 27: 1428-1439.

Moore, P.A., and K. R. Reddy. 1994. Role of Eh and pH on phosphorus geochemistry in sediments of Lake Okeechobee, Florida. J. Environ. Qual. 23: 955-964.

Nürnberg, G.K., and B.D. LaZerte. 2004. Modeling the effect of development on internal phosphorus load in nutrient-poor lakes. Water Resour. Res. 40:W01105, doi:10.1029/2003WR002410.

Paul, M.J., and Meyer, J.L. 2001. Streams in the urban landscape. Annu. Rev. Ecol. Syst. 32: 333-365.

RAP 1995. White Lake Area of Concern Remedial Action Plan: 1995 Update. Public Sector Consultants.

Søndergaard, M., J.P. Jensen, and E. Jeppesen. 2001. Retention and internal loading of phosphorus in shallow, eutrophic lakes. TheScientificWorld 1: 427–442.

Steinman, A.D., X. Chu, and M. Ogdahl. Submitted. Spatial and temporal variability of internal and external phosphorus loads in an urbanized watershed. Limnology and Oceanography.

Steinman, A.D., K.E., Havens, N.G. Aumen, R.T. James, K.-R. Jin, J. Zhang, and B. Rosen. 1999. Phosphorus in Lake Okeechobee: sources, sinks, and strategies. p. 527-544. *In* K.R. Reddy, G.A. O'Connor, and C.L. Schelske (ed.) Phosphorus Biogeochemistry of Subtropical Ecosystems: Florida as a case example. CRC/Lewis Publ., New York.

Steinman, A.D., R. Rediske, and K.R. Reddy. 2004. The reduction of internal phosphorus loading using alum in Spring Lake, Michigan. J. Environ. Qual. 33: 2040-2048.

Steinman, A.D., R. Rediske, X. Chu, R. Denning, L. Nemeth, D. Uzarski, B. Biddanda, and M. Luttenton. 2006. An environmental assessment of an impacted, urbanized watershed: the Mona Lake Watershed, Michigan. Arch. Hydrobiol. 166: 117-144.

Steinman, A.D. and B.H. Rosen. 2000. Lotic-lentic linkages associated with Lake Okeechobee, Florida. J. N. Am. Benth. Soc. 19: 733-741.

U.S. EPA. (United States Environmental Protection Agency). 1983. Methods for Chemical Analysis of Water and Wastes. Washington, D.C., USA.

Walsh, C.J., A.H. Roy, J.W. Feminella, P.D. Cottingham, P.M. Groffman, and R.P. Morgan, II. 2005. The urban stream syndrome: current knowledge and the search for a cure. J. N. Am. Benthol. Soc. 24: 706-723.

Welch, E.B., and G.D. Cooke. 1995. Internal phosphorus loading in shallow lakes: importance and control. Lake Reserv. Manage. 11: 273-281.

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