Aquat Ecol
DOI 10.1007/s10452-007-9147-6

Spatial and temporal variability of internal and external phosphorus loads in Mona Lake, Michigan

Alan Steinman · Xuefeng Chu · Mary Ogdahl

Received: 27 February 2007 / Accepted: 9 October 2007

Abstract A study was conducted in Mona Lake, a small eutrophic lake located in western Michigan (USA) to address the temporal and spatial variability of external and internal phosphorus loading. External P load varied among subbasins, which was mostly related to discharge, but also to land use. Black Creek, which drains lands with natural cover and agriculture, accounted for the majority of flow, and total phosphorus (TP) and soluble reactive phosphorus (SRP) load, to Mona Lake. However, the relative contribution of SRP load was greater in Little Black Creek, which flows through a mostly urbanized subbasin, than in Black Creek. The relative importance of internal loading was strongly related to season, as internal TP loads contributed only ~9% of the overall P load in April 2005, but ~68–82% of the overall P load in the summer and early fall seasons. Internal TP and SRP loading was greater under anaerobic than aerobic conditions. Mean anaerobic TP release rates ranged from 0.80 to 15.56 mg P m^{-2} d^{-1}, varying with site and season. Spatial variability in both internal phosphorus loading and sediment P concentration was also evident. By taking into account the spatial and temporal variability of different loading sources, management practices can be targeted to optimize nutrient source control strategies.

A. Steinman (✉) · X. Chu · M. Ogdahl
Annis Water Resources Institute, Grand Valley State University, 740 W. Shoreline Drive, Muskegon, MI 49441, USA
e-mail: steinmaa@gvsu.edu

Keywords Eutrophication · Nonpoint source pollution · Phosphorus transport · West Michigan

Introduction

Cultural eutrophication remains a critical problem despite the scientific advances made over the past four decades (Smith et al. 2006; Schindler 2006). Impaired water quality has both environmental and economic implications; for example, excessive nutrients can result in nuisance and toxic algal blooms, hypoxic conditions can lead to fish kills, and contaminants adsorbed to bottom sediments can be biomagnified up the food chain. All of these impacts can influence local and regional economies through their effects on real estate values, commercial and recreational fisheries, ecotourism (cf. Carpenter et al. 1998a; Loomis et al. 2000; Ludwig et al. 2003; Pretty et al. 2003), and possibly, human-health related costs (Levin et al. 2002).

Although point source pollution in the United States has been addressed through the Clean Water Act, and nonpoint source pollution currently is being addressed through a variety of improved watershed management practices, ~45% of the country's water bodies are still considered impaired (U.S. Environmental Protection Agency 2000). Nonpoint sources now contribute more nutrients to water bodies in the United States than point sources (Carpenter et al. 1998b). Legacies from both point and nonpoint source pollution are present in the sediments of many lakes and, via internal loading, can be a significant source of nutrients (Boström et al. 1982; Søndergaard et al. 2001). Indeed, reductions in tributary nutrient loads have failed to improve water quality in lakes that have significant internal loads (Sas 1989; Graneli 1999).

Changing land use patterns, including increases in urbanization, drainage efficiency, and impervious surface area, have resulted in faster nutrient delivery to water bodies, with less opportunity for removal by abiotic or biotic processes (Groffman et al. 2003; Steinman et al. 2006). Attempts to limit or reduce external loading to water bodies have resulted in the development of low impact development initiatives in many municipalities. Detaining and retaining water on site, for example, through the use of constructed wetlands or rain gardens (Schueler and Holland 2000), provides a greater opportunity for pollutants to be captured before they enter local streams and lakes (Steinman and Rosen 2000). These watershed-based practices are based 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. In addition, the timing of these inputs can vary within and among years. The influence of these inputs depends, in part, on the hydraulic retention rates of the lake; in systems with high flushing rates, such as Mona Lake, nutrient inputs will have much stronger ecological impacts during periods of low flow compared to high flow, when the nutrients will be retained. In contrast, it may be more difficult to detect the ecological influence associated with seasonal variability of nutrient inputs in lakes with long retention times, as the inputs from different seasons and different sources become blended over time.

Management efforts to control sources should take into account these spatial and temporal variations in nutrient inputs (cf. Shostell and Bukaveckas 2004). Ultimately, integrated approaches that examine source control in a holistic way are needed, especially in urbanizing watersheds where multiple stressors often exist (Paul and Meyer 2001; Walsh et al. 2005; Steinman et al. 2006).

In this study, we examine the temporal and spatial variability of internal and external (i.e., tributary) phosphorus (P) loading in the Mona Lake watershed, an urbanizing region in western Michigan, United States. This watershed faces many of the environmental and socio-economic challenges common to watersheds, throughout the developed world (Steinman et al. 2006); as a consequence, the findings in this study should have application elsewhere.

Materials and methods

Study site

The Mona Lake watershed (200 km^2) is located in western Michigan. The lake connects directly to Lake Michigan through a dredged navigation channel (Fig. 1). Mona Lake has a surface area of 2.65 km^2, or 1.4% of the total watershed area, with mean and maximum depths of 6.1 and 8.3 m, respectively. Water retention times vary from 105 to 160 days during low flow periods to <35 days during high flow periods (Evans 1992). The lake is moderately eutrophic, with mean annual near-surface total phosphorus (TP) and chlorophyll *a* (chl *a*) concentrations of 68 and 15 µg l^{-1}, respectively (Steinman et al. 2006). Mona Lake is dimictic, and usually, is ice-covered from December through March. Summer stratification can be temporarily disrupted because of episodic storm events, but quickly reforms once conditions stabilize.

Four tributaries account for >95% of the annual discharge to Mona Lake under base flow conditions: Black Creek (77.2%), Little Black Creek (9.3%), Cress Creek (5.3%), and Ellis Drain (4.7%; Steinman et al. 2006). Black Creek originates as a series of agricultural ditches in the upper reaches of the watershed before flowing through forested areas, and then residential reaches on its way to Mona Lake (Fig. 2). Little Black Creek flows through heavily urbanized areas. Historically, sources of contamination and impaired water quality in this creek included: (1) a petroleum refinery site (crude oil and light hydrocarbons); (2) storm sewers from foundry and metal finishing industries (oils, grease,

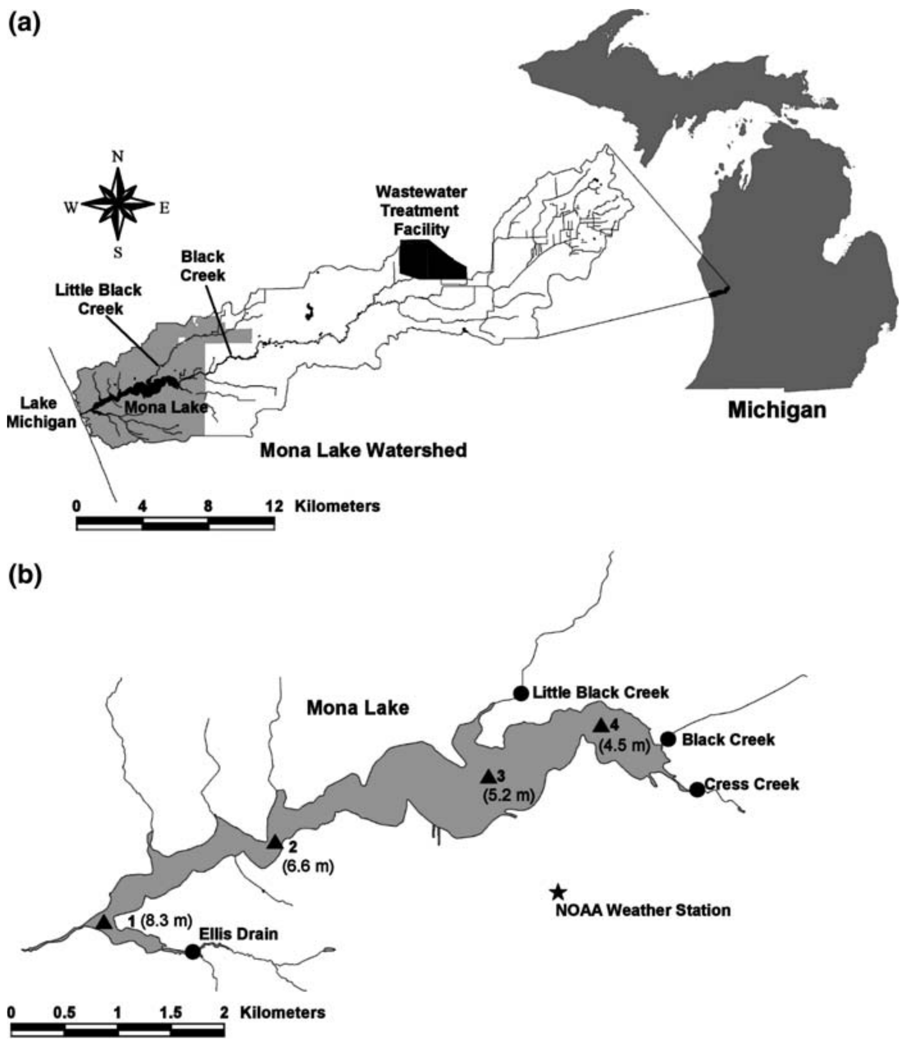


Fig. 1 (**a–b**) State of Michigan (right) and blowup of Mona Lake watershed in western Michigan (left) (**a**). The gray-shaded area within the watershed represents municipal boundaries for the urbanized region covering the cities of Muskegon, Muskegon Heights, and Norton Shores. The black-shaded area shows the spatial extent of the Muskegon County Wastewater Management System. Blowup of Mona Lake (**b**). Location of tributary (*circles*) and lake (*triangles*) sampling sites for measurement of external and internal loads to Mona Lake; location of National Oceanic Atmospheric Administration (NOAA), National Climatic Data Center weather station (*star*). For lake sites, the average depth over the sampling period is given in parentheses

PAH compounds, heavy metals, and PCBs); (3) a plating Superfund site (cadmium, chromium, copper, nickel, and zinc); (4) spills from a municipal sanitary/industrial wastewater pump station; and (5) an abandoned municipal landfill without a leachate collection system (MDEQ 2000, 2002). Although the refinery, plating, and landfill sites are no longer in operation, they are still having environmental concern because contaminated sediments and groundwater plumes continue to impair water quality. The Muskegon County Wastewater Management System handles all domestic sewage from residences adjacent to Mona Lake.

Land use/land cover categories for the entire watershed, based on 2005 data include: 46.6% natural cover, 37.8% developed, and 15.6% agricultural. The Black Creek subbasin is the largest (53.8% of total area), followed by the Little Black Creek (13.9% of total), Ellis Drain (9.2% of total), and Cress Creek (8.7% of total) subbasins. The Black Creek subbasin is primarily natural cover (54.6%) and has more area devoted to agriculture (11.2%) than any other subbasin; the other subbasins are dominated by developed land (all >66%; Table 1, Fig. 2). Most of the agricultural activity is in the upper portion of the watershed (Black Creek subbasin) where the impervious surface area is <4%. In contrast, the area immediately adjacent to Mona Lake is largely residential and has an impervious surface area of >22% (Steinman et al. 2006). Sediments throughout the watershed are primarily sand, with pockets of fine organics in wetland areas.

Table 1 Land use characterization of the Mona Lake watershed by major subbasin, based on 2005 data

Land use category	Black Creek (%)	Little Black Creek (%)	Cress Creek (%)	Ellis Drain (%)
Agricultural	11.2	0.0	4.3	1.2
Natural cover	54.6	33.1	23.9	23.7
Developed	30.3	66.2	71.7	74.4
Water	3.9	0.7	0.1	0.7

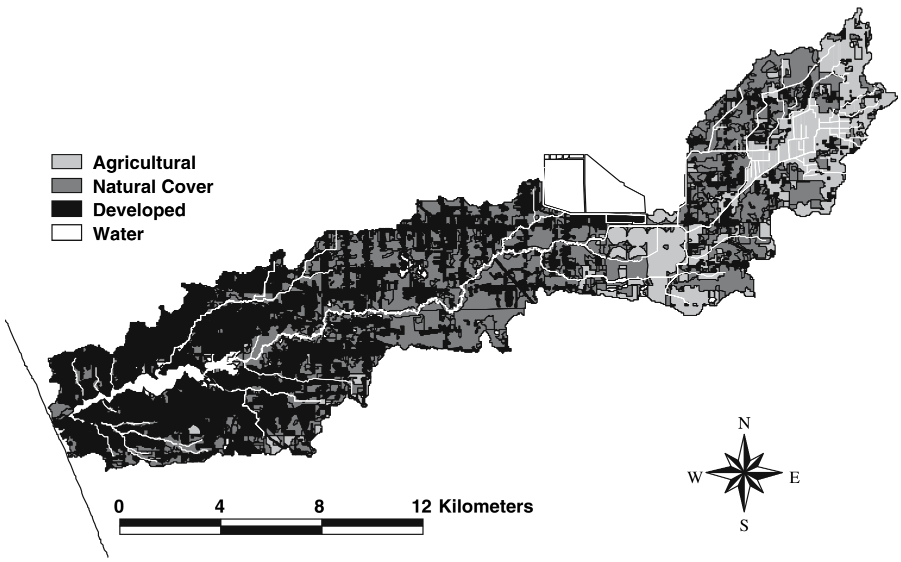


Fig. 2 Land use map of the Mona Lake watershed, based on 2005 data

External P loading

Grab samples were collected, and discharge was measured monthly near the mouth of each of the four main tributaries over the period July 2004 through July 2005 (Fig. 1). Additional tributary samples were collected during two storm events (>2 cm rainfall): 19 November 2004, and 30 June 2005. Also, to improve our estimate of external P loads, the water quality data (low-discharge TP concentrations) that we collected in 2002 and 2003 (Steinman et al. 2006) were used and compared with the new 2004–2005 data, providing a total of ~30 observations for each tributary. In spite of these efforts, we recognize that our approach provides a coarse estimation of external P loads, given the possible high variability in the relationship between stream discharge and P concentration.

Flow velocity was measured with a Marsh-McBirney flow meter (Flo-Mate Model 2000), and the HYDROL-INF software (Chu and Mariño 2006) was used for processing the measured data, and calculating discharges and other hydraulic parameters. The velocity-area method (U.S. Environmental Protection Agency 2004) was employed in the computation. Samples for TP analysis were collected in 250-ml acid-washed bottles. Water for soluble reactive phosphorus (SRP) analysis was syringe-filtered (0.45-μm membrane) immediately after collection into 20-ml scintillation vials. Samples were stored on ice until transported to the laboratory, always within 5 h of collection. Total P samples were stored at 4°C, and SRP samples were frozen until analysis. Total P and SRP concentrations were determined using the ascorbic acid method with a BRAN + LUEBBE Autoanalyzer; persulfate digestion was used to prepare TP samples for analysis (U.S. Environmental Protection Agency 1983).

An Odyssey pressure and temperature recording system was installed to continuously measure (10 min intervals) stream stage and temperature at the mouth of each of the four main tributaries (Fig. 1) (note that the stream monitoring was discontinued during the winter months from November 2004 to March 2005, when the tributaries were covered with ice). Stage discharge relationships were developed with weekly to biweekly manual discharge measures, and the observed hydrographs were then generated for all monitoring sites. In addition, average daily discharges at the mouth of each of the main four tributaries also were computed.

Due to the dissimilar sizes, land use/land covers, soil types, and other conditions, the four major tributaries have distinct hydrologic characteristics (e.g., response to rainfall events and sources of stream water), which consequently affect both temporal and spatial distribution of external P loads. Thus, the discharge-P load relationships can be complex and highly variable. Our previous water quality data, mostly collected during dry periods, did not show a significant relationship between these two variables for most tributaries. However, the storm data (high-discharge) collected in this study did suggest a systematic increase in P concentrations for all tributaries, compared with the P levels observed during low (base) discharge periods. In order to take into account this significant difference in P concentrations between high and low discharges, we developed a method for estimating P levels based on the corresponding discharge conditions. Specifically, daily loads of TP and SRP were calculated by multiplying daily water discharge by concentration (either low- or high-discharge concentration, depending on discharge conditions). Average monthly discharge was calculated and used as a threshold for determining low- and high-discharge conditions. For each daily time step, if daily discharge was less than or equal to the monthly threshold discharge, the low-discharge concentration was used in the loading computation. Otherwise, a high-discharge concentration was computed using the low-discharge concentration plus the increased rate as follows:

For $Q_i > Q_{T,j}$

$$M_i = Q_i C_{HF,j} = Q_i C_{LF,j}(1 + r/100) \quad (1)$$

For $Q_i \leq Q_{T,j}$

$$M_i = Q_i C_{LF,j}, \quad (2)$$

where M_i = load at daily time step i; Q_i = discharge at daily time step i; $C_{HF,j}$ = high-discharge concentration in month j with respect to daily time step i; $C_{LF,j}$ = low-discharge concentration in month j with respect to daily time step i; $Q_{T,j}$ = threshold discharge in month j with respect to daily time step i; and r = percent of concentration change between low and high discharges (%). In addition, the winter months from November 2004 to March 2005 were not included in our P loading computation primarily for the following reasons: (1) Mona Lake and its tributaries were frozen during the winter months, thereby affecting our estimation methods; (2) the sensor-recorded stream data were not available for the winter months; and (3) our current data were not sufficient for understanding the hydrologic processes related to melting of snow and frozen soils in the winter.

Hourly rainfall data for the study period were obtained from a National Oceanic Atmospheric Administration (NOAA), National Climatic Data Center weather station located at the Muskegon County Airport (43°10′16″N, 86°14′12″W), which is adjacent to Mona Lake (Fig. 1).

Internal P Loading

Lake sampling and sediment coring

At each site, Secchi depths were determined and vertical profiles (1-m intervals) of dissolved O_2 (O_2), pH, temperature, and chl *a* were measured with a Hydrolab DataSonde 4a equipped with a Turner Designs fluorometer. Fluorometric data were calibrated against acetone-extracted chl *a* standards measured via spectrophotometry. Near surface (just below surface layer) and near-bottom (~0.5 m above sediment layer) water samples for TP and SRP analyses were collected with a Van Dorn bottle, maintained on ice until delivery to the laboratory, and analyzed as described above. Water for SRP analysis was immediately syringe-filtered (0.45 µm membrane) into 20-ml scintillation vials.

Six sediment cores were collected at each site on four dates: 27 July 2004, 14 September 2004, 14 April 2005, and 7 June 2005 (Fig. 1). The piston corer was constructed of a graduated (1-cm) 0.6-m long polycarbonate tube (7-cm inner diameter), and a polyvinyl chloride attachment assembly for coupling to aluminum drive rods (Fisher et al. 1992; Steinman et al. 2004). The piston was advanced 20–25 cm prior to deployment to maintain a water layer on top of the core during collection. 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 core $\sim$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 on ice during transit. An additional core was collected from each site, and the top 5 cm removed for the following sediment analyses in the lab: ash free dry mass (AFDM), TP on the ashed material, and total Fe, Ca, and Mg, as described below.

Core processing and P flux

The 24 cores (six per site) were placed in a Revco environmental growth chamber, with the temperature maintained to match bottom-water conditions in Mona Lake at the time of collection. The water column in three of the cores from each site was bubbled with N_2 (with 330 ppm CO_2) to create buffered anaerobic conditions, while the remaining three were bubbled with O_2 to create aerobic conditions.

Internal TP and SRP loads were estimated with methods described in Steinman et al. (2004). Briefly, a 40-ml water sample was removed by syringe through the sampling port of each core tube at 2 h, 12 h, 1 day, 2 day, 4 day, 8 day, 12 day, 16 day, 20 day (September 2004), 24 day (April and June 2005), and 28 day (July 2004). 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. Phosphorus concentrations were determined, as described above. The 40-ml drawn from the core 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 over time.

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

$$P_{rr} = (C_t - C_0)V/A, \tag{3}$$

where P_{rr} = net P release rate or retention per unit surface area of sediments; C_t = TP or SRP concentration in the water column at time t; C_0 = TP or SRP concentration in the water column at time 0; V = volume of water in the water column; and A = planar surface area of the sediment cores. Flux calculations were based on the increase in water column concentrations of TP from the linear portion of the curve (maximum apparent release rates). Total P release rates (mg P m^{-2} d^{-1}) were converted to metric tons, and calculated for the entire lake surface area via the following steps: (1) based on dissolved oxygen concentrations measured at each site at the time of sediment core collections (cf. Table 2), either anaerobic (<1 mg l^{-1}) or aerobic (>1 mg l^{-1}) release rates from the laboratory incubations were used in the overall load calculation; (2) Mona Lake was divided into four, equally-sized geographic zones, and the appropriate release rates from each zone were extrapolated to the surface area of the zone, and summed to obtain a rate for the entire lake; we assumed that the release rates from each site were representative of that entire zone. We recognize that release rates vary spatially within and among zones, and that the zones do not have exactly equal bottom surface areas, but we believe this is a reasonable first approximation, given the available data; and (3) release rates were multiplied by the number of days in the incubation, converted to metric tons, and summed to obtain a total load for each season. It is likely that zones classified as anaerobic (based on field data) include oxic sediments (in shallow areas), even during stratified periods; therefore, the total P release rates may be overestimated.

Following the incubations, the top 5 cm of sediment was removed from each core and placed in a zipseal plastic bag. Anaerobic core sediment was immediately placed in a Labconco glove box that was previously purged of air and filled with N_2 gas; the plastic bags were left open to fill with N_2. The sediment was then homogenized, and a 5 g subsample was placed in a tightly-capped centrifuge tube for pore water analysis and sequential phosphorus fractionation (Moore and Reddy 1994). Oxidation condition was maintained during the pore water extraction and fractionation procedure by processing anaerobic core sediment in the N_2-filled glove box. The wet sediment was centrifuged to remove excess pore water, which was filtered, frozen, and analyzed for SRP concentration, as described above. Residual sediment was shaken for 17 h with 0.1 M NaOH,

Table 2 Selected limnological characteristics of sampling sites in Mona Lake

Variable	Date	Site			
		1	2	3	4
Secchi depth (m)	7/27/04	2.05	1.66	1.5	1.2
	9/14/04	0.82	0.84	0.72	0.55
	4/14/05	1.25	1.0	1.0	1.0
	6/7/05	2.4	1.95	1.6	1.3
Temperature (°C) (surface/bottom)	7/27/04	23.0/18.1	23.1/19.5	23.2/20.9	23.2/21.4
	9/14/04	21.8/19.9	22.5/20.7	22.6/21.0	23.1/21.2
	4/14/05	11.8/11.0	11.0/11.0	10.7/10.4	10.7/10.3
	6/7/05	21.2/14.2	22.0/15.2	22.8/20.6	23.7/22.5
O_2 (mg l^{-1}) (surface/bottom)	7/27/04	6.1/0.2	6.3/0.5	7.5/0.5	8.3/7.5
	9/14/04	7.8/0.3	10.1/0.3	9.7/0.3	11.4/1.8
	4/14/05	10.5/8.9	9.9/9.4	10.3/8.5	10.0/8.9
	6/7/05	6.0/0.3	7.6/0.3	6.6/3.8	6.9/5.8
Chl *a* (μg l^{-1}) (surface)	7/27/04	25.3	31.8	31.7	39.4
	9/14/04	30.2	27.9	36.3	102.2
	4/14/05	26.3	26.5	25.7	24.9
	6/7/05	11.3	11.6	7.3	5.5
TP (μg l^{-1}) (surface/bottom)	7/27/04	28/90	51/115	37/55	41/34
	9/14/04	62/58	68/59	93/105	134/103
	4/14/05	40/41	50/37	40/48	32/40
	6/7/05	39/92	60/64	55/56	54/50
SRP (μg l^{-1}) (bottom)	7/27/04	70	100	BD	BD
	9/14/04	BD	BD	BD	BD
	4/14/05	BD	BD	BD	BD
	6/7/05	19	27	15	10

centrifuged, and the supernatant filtered, frozen, and analyzed for SRP. This fraction is referred to as Al- and Fe-bound P (NaOH–P), and represents a mineral association that can become soluble under anoxic conditions (Olila et al. 1995). After this extraction, the sediment was extracted for 24 h with 0.5 M HCl, centrifuged, and the supernatant filtered, frozen, and analyzed for SRP. This fraction is referred to as Ca- and Mg-bound P (HCl–P), and represents a stable mineral association (Olila et al. 1995). Another subsample was taken from the top 5 cm of each core for total metals (Fe, Ca, and Mg) and AFDM analysis. The ashed material was analyzed for TP, as described previously. Sediment metal concentrations were determined according to method SW-846 Method 3050B (nitric acid hydrogen peroxide digestion + HCl; U.S. Environmental Protection Agency 1994) and Method 7000A (flame atomic absorption; U.S. Environmental Protection Agency 1994) using a Perkin Elmer 300 Atomic Absorption Spectrometer.

Statistical analysis

Differences in median concentrations of TP and SRP among tributaries were analyzed using a Kruskal–Wallis test. If the overall model was significant at $\alpha = 0.05$, either Tukey's multiple comparison test (equal sample sizes) or Dunn's test (unequal sample sizes) was applied to detect contrasts among categories.

Total P and SRP release rates and sediment chemistry were analyzed with a 3-way ANOVA. The main effects were sampling date ($n = 4$), site ($n = 4$), and oxidation state ($n = 2$). Data were ln-transformed where necessary. Pair wise multiple comparisons were conducted using the Holm-Sidak method. All statistical tests were undertaken with SigmaStat 3.1 (Systat Software, Point Richmond, California, USA).

The comparison of internal versus external P loads was restricted to the time periods when internal loading incubations were conducted: 27 July–16 August 2004 (late summer stratified period), 14 September–4 October 2004 (autumn mixing period), 14 April–4 May 2005 (spring mixing period), and 7–27 June 2005 (early summer stratified period). This approach permitted us to assess seasonal variability (among season and year), and spatial variability (within lake and internal versus external), over defined time periods.

Results

External P loading

During low discharge, SRP concentrations in the tributaries ranged from <10 to 22 μg l^{-1}, with peak concentrations consistently occurring in September and late winter (Fig. 3). Separate peaks were detected in March for Cress Creek, and May for Little Black Creek. Median SRP concentrations were highest in Little Black Creek, but there were no statistically significant differences in SRP concentrations among tributaries ($H = 6.63$, $P = 0.157$). Low-discharge TP concentrations in the tributaries ranged from 11 to 105 μg l^{-1} (Fig. 3). Overall, low-discharge TP concentrations were significantly different among tributaries ($H = 12.838$, $P < 0.05$), but the only significant contrast was between Black Creek and Ellis Drain ($P < 0.05$). In general, median TP concentrations were higher in Black Creek (31 μg l^{-1}) and Little Black Creek (30 μg l^{-1}) than in Cress Creek (19 μg l^{-1}) and Ellis Drain (20 μg l^{-1}).

Median SRP concentrations were nearly four times higher during the June 2005 storm event (19 μg l^{-1}) than during the November 2004 storm event (5 μg l^{-1}; Fig. 3). During both events, Little Black Creek experienced a considerably greater spike in SRP concentration (November 2004: 19 μg l^{-1}; June 2005: 116 μg l^{-1}) than all other sites (November 2004: <10 μg l^{-1}; June 2005: 18–19 μg l^{-1}). Median TP concentrations reflected a pattern similar to SRP concentrations, with values nearly four times higher

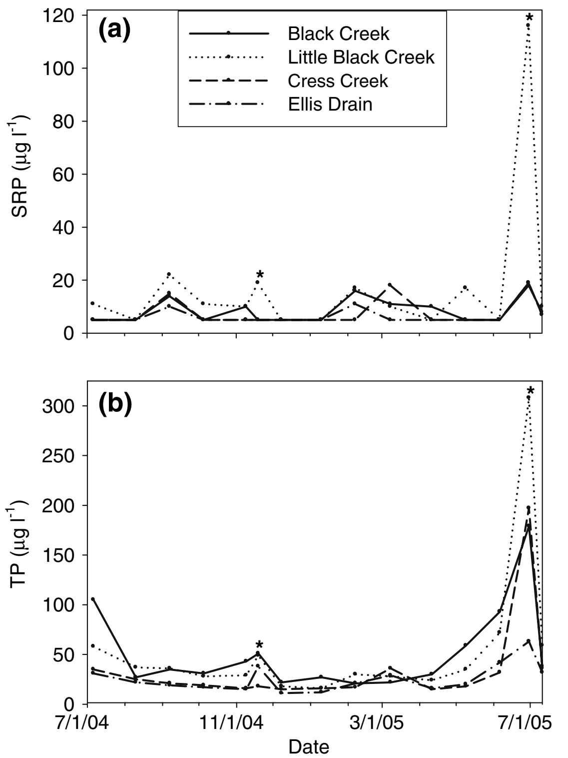


Fig. 3 Concentrations of SRP (**a**) and TP (**b**) in Mona Lake tributaries sampled between July 2004 and July 2005. Asterisks indicate samples taken during storm events

during the June 2005 event (188 μg l^{-1}) than the November 2004 event (44 μg l^{-1}; Fig. 3). Total P concentration was much lower in Cress Creek (18 μg l^{-1}) in November 2004 than in all other sites (38–51 μg l^{-1}). In June 2005, Little Black Creek had a higher TP concentration (308 μg l^{-1}) than all other sites; Black Creek and Cress Creek had similar TP concentrations (179 and 197 μg l^{-1}, respectively), and Ellis Drain had considerably less TP than all other sites (63 μg l^{-1}; Fig. 3).

Out of the four major inflows that were surveyed, Black Creek contributed the largest discharge to Mona Lake (76.3%), followed by Little Black Creek (13.5%), Cress Creek (6.0%), and Ellis Drain (4.2%; Fig. 4). Median discharge over the 2004 period of record differed significantly among tributaries ($H = 13.78$, $P < 0.01$), with Black Creek (1190 l s^{-1}) significantly greater than Ellis Drain (48 l s^{-1}, $P < 0.05$), and with Little Black Creek (240 l s^{-1}) and Cress Creek (73 l s^{-1}) having intermediate discharge. Both discharge trends among tributaries and statistical differences were similar in 2004 and 2005.

Rainfall was below average over the study period. Approximately 68 cm was recorded from July 2004 through June 2005, which compared to an annual average of ~82 cm for 1950 through 2000. Five events had total precipitation that exceeded 2 cm (Fig. 4).

When the phosphorus concentration and discharge data were combined to calculate loads, Black Creek contributed 73.3% of the external TP load, and 55.8% of the external SRP load to Mona Lake over the periods of record. Little Black Creek accounted for 18.9% and 37.2% of the TP and SRP loads, respectively. Both Cress Creek and Ellis Drain accounted for <5% of the TP and SRP loads to Mona Lake.

Median TP loads were significantly different among tributaries ($H = 27.32$, $P < 0.001$), with Black Creek significantly greater than Cress Creek and Ellis Drain ($P < 0.05$), and Little Black Creek significantly greater than Ellis Drain. On a seasonal basis (Fig. 5), the greatest external TP loads to Mona Lake occurred in July 2004, and the lowest loads occurred in October 2004. Black Creek was the dominant source of TP to Mona Lake throughout the period of record, but there were a few dates (e.g., July 2005) when Little Black Creek contributed more TP load than Black Creek (Fig. 5).

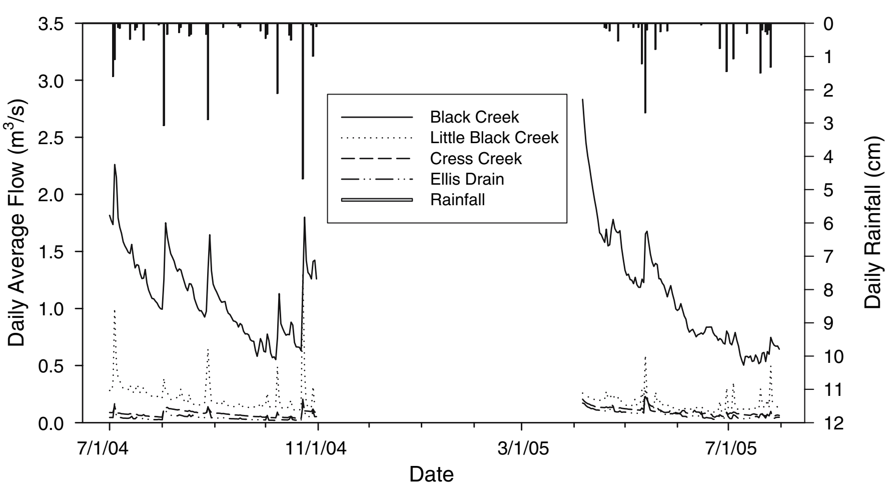
Median SRP loads were significantly different among tributaries ($H = 25.17$, $P < 0.001$), with the same statistical results among tributaries as found for TP. On a seasonal basis (Fig. 5), the greatest external SRP loads to Mona Lake occurred in April 2004, and the lowest loads occurred in June 2005. There were numerous occasions throughout the year when Little Black Creek contributed more SRP load than Black Creek (Fig. 5), despite its considerably lower discharge.

Internal P loading

Sites 1 and 2, at the western end of Mona Lake, were deeper, more transparent, and in the summer months had lower bottom water temperatures and O_2 concentrations, and higher bottom SRP concentrations, than sites 3 and 4 (Fig. 1; Table 2). Chlorophyll *a* concentrations were similar among sites on a given sampling date with the exception of Site 4 in September 2004, when an algal bloom resulted in very high concentrations (Table 2). Near-surface TP concentrations were similar among sites with the exception of higher levels at sites 3 and 4 in September 2004; bottom TP concentrations were elevated at sites 1 and 2 compared to sites 3 and 4 in July 2004, but this was reversed just two months later (Table 2). High hypolimnetic SRP concentrations at sites 1 and 2 in July 2004 coincided with the high TP levels (SRP was ~80% of TP), suggesting that internal loading was responsible for this increase. Conversely, low near-bottom SRP concentrations were measured at sites 3 and 4 two months later when TP was elevated (SRP was ~5% of TP), suggesting the high TP levels were due to sediment resuspension or biotic accumulation of P.

Phosphorus release showed generally similar patterns for all sampling dates, with significantly higher

Fig. 4 Discharge of the major tributaries (left axis) and precipitation (right axis)



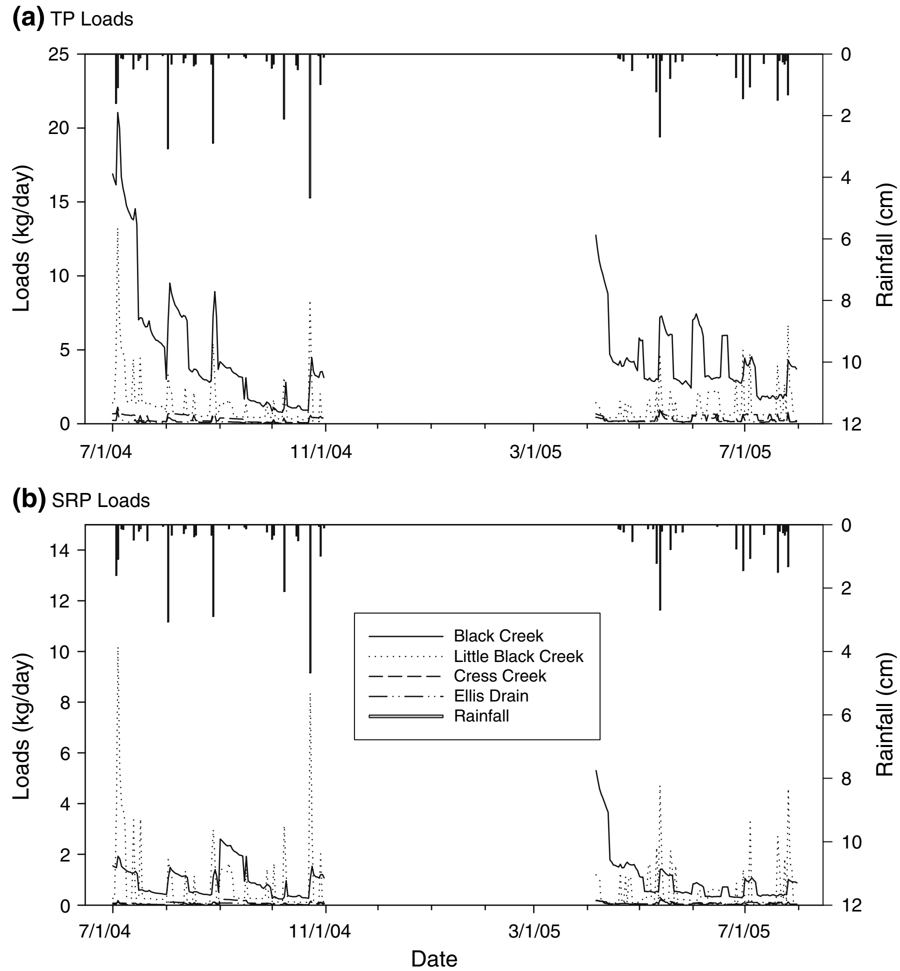


Fig. 5 Total P (**a**) and SRP (**b**) loads for the main four inflows to Mona lake (left axis) and precipitation (right axis)

rates under anaerobic than aerobic conditions ($F = 163.304$, $P < 0.001$; Table 3). Representative data are shown for the July 2004 cores (Fig. 6). Overall, mean TP release rates varied significantly by sampling date ($F = 24.665$, $P < 0.001$; Table 3), with mean release rates significantly greater in July 2004 than all other dates; release rates were next highest in June 2005, which in turn were significantly greater than in September 2004, or in April 2005. These last two dates were not significantly different from one another. The negative values in September 2004 under aerobic conditions (Table 3) could be a function of chemical adsorption of TP to sediments, or sedimentation of particles from the water column. TP release rates did not vary significantly across sites ($P = 0.353$).

The sampling date × oxidation state interaction was highly significant ($F = 15.968$, $P < 0.001$), as the oxidation state-related differences in TP release rates were more pronounced during the summer sampling dates than fall or spring sampling dates. Neither the sampling date × site ($P = 0.052$) nor the sampling site × oxidation state ($P = 0.8$) interactions were statistically significant.

SRP release rates varied significantly by sampling date ($F = 24.222$, $P < 0.001$; Table 3). Mean release rates were significantly lower in April 2005 than all other dates; the only other significant difference was greater SRP release rates in September 2004 compared to July 2004. No mean negative SRP release rates were measured throughout the study, although no net release was measured under aerobic conditions at two sites in April 2005 (Table 3). Soluble reactive P release rates were significantly greater under anaerobic than aerobic conditions ($F = 77.710$, $P < 0.001$; Table 3). SRP release rate varied

Table 3 Mean (±SD) maximum release rates (mg P m^{-2} d^{-1}) of TP and SRP from sediment cores collected from Mona Lake and incubated under anaerobic and aerobic conditions

Site	TP		SRP	
	Anaerobic	Aerobic	Anaerobic	Aerobic
I. July 2004				
1	7.06 ± 2.57	0.46 ± 0.24	4.36 ± 1.12	0.24 ± 0.10
2	9.27 ± 5.99	1.36 ± 0.73	1.19 ± 0.36	0.38 ± 0.15
3	15.56 ± 1.00	0.90 ± 0.29	1.22 ± 0.22	0.90 ± 0.22
4	13.63 ± 1.82	0.59 ± 0.41	0.83 ± 0.07	0.39 ± 0.25
II. September 2004				
1	4.48 ± 1.56	–0.66 ± 0.22	4.60 ± 1.14	0.37 ± 0.01
2	2.87 ± 0.97	–1.14 ± 0.93	1.73 ± 0.40	0.51 ± 0.32
3	3.10 ± 4.08	0.51 ± 0.13	2.34 ± 2.59	1.54 ± 0.77
4	6.46 ± 4.66	–0.79 ± 0.23	3.28 ± 2.11	1.10 ± 0.00
III. April 2005				
1	2.77 ± 1.53	0.25 ± 0.01	2.12 ± 1.01	0.66 ± 0.66
2	2.82 ± 0.83	0.26 ± 0.23	0.73 ± 0.67	0.00 ± 0.00
3	0.80 ± 0.07	0.17 ± 0.07	0.44 ± 0.38	0.44 ± 0.38
4	1.15 ± 0.71	0.12 ± 0.04	0.22 ± 0.38	0.00 ± 0.00
IV. June 2005				
1	9.28 ± 2.38	0.53 ± 0.50	4.73 ± 1.54	0.39 ± 0.19
2	7.33 ± 1.55	0.63 ± 0.08	3.15 ± 0.85	0.45 ± 0.07
3	9.56 ± 7.11	1.47 ± 0.14	1.32 ± 0.49	0.51 ± 0.12
4	4.75 ± 3.74	0.94 ± 0.10	0.42 ± 0.02	0.35 ± 0.15

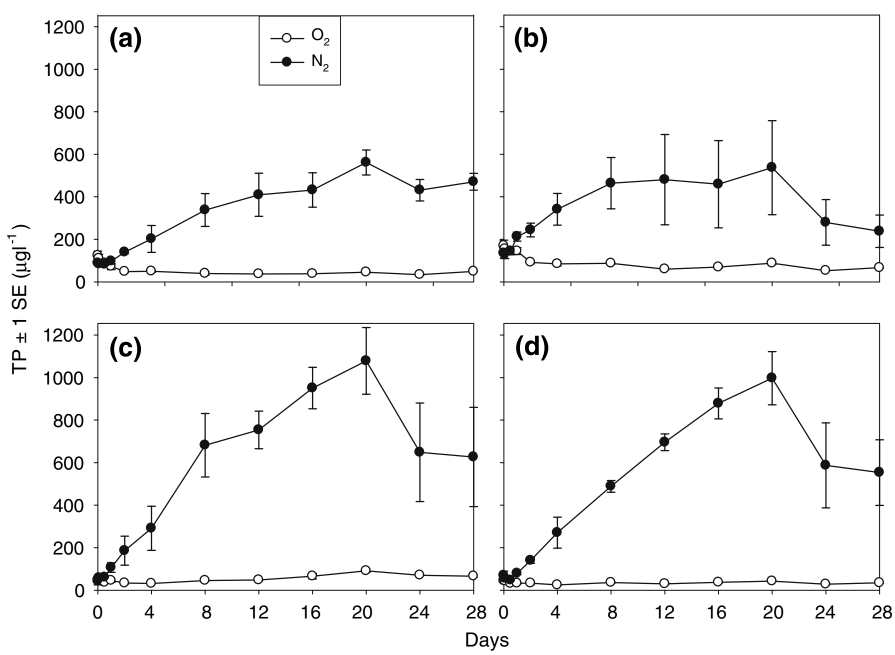


Fig. 6 Mean (±SE) TP concentrations released from sediment cores from four sites in Mona Lake sampled in July, 2004: site 1 (**a**), site 2 (**b**), site 3 (**c**), and site 4 (**d**). (N_2 = nitrogen, anaerobic condition; O_2 = oxygen, aerobic condition)

significantly also by site ($F = 8.564$, $P < 0.001$); the site × oxidation state interaction was statistically significant ($F = 11.028$, $P < 0.001$). Under anaerobic conditions, site 1 had significantly greater SRP release rates than all other sites, and site 2 had significantly greater release rates than site 4, whereas under aerobic conditions site 3 had significantly greater release rates than sites 2 and 4, but none of the other pair wise contrasts were statistically different (Table 3). Neither the sampling date × oxidation state interaction ($P = 0.854$) nor the sampling date × site × oxidation state ($P = 0.698$) interaction was statistically significant.

Soluble reactive P release accounted for anywhere from ~5 to 100% of the TP released (Table 3). In general, SRP accounted for a much larger percentage of the TP released under aerobic conditions than anaerobic conditions (Table 3). Despite the larger percentage under aerobic conditions, the absolute SRP release rates were much lower under aerobic than anaerobic conditions (Table 3).

Sediment chemistry showed spatial, but not temporal differences. Sediment TP concentration was significantly greater at site 3 than all other sites ($F = 183.301$, $P < 0.001$), irrespective of oxidation state status (Fig. 7). There were no statistically significant differences in sediment chemistry associated with sampling date, oxidation state, or any of the interaction terms (all $P > 0.05$). Sediments also were characterized based on percent solid content, with sites 3 (12.2%) and 4 (11.5%), having significantly more ($P < 0.001$) percent solids than sites 1 (8.1%) and 2 (8.3%).

Sediment fractionation yielded NaOH–P and HCl–P. On all dates, extractable P varied significantly by site ($P < 0.001$), as site 3 consistently had more NaOH–P and HCl–P than all other sites (Fig. 8). In addition, NaOH–P was significantly greater than HCl–P on all dates except April 2005. The site × extractant interaction was significant, as greater amounts of NaOH–P were evident at site 3, but not at the other sites (Fig. 8). Oxidation state had no significant effect on either NaOH–P or HCl–P.

Mean concentrations (±SD) of Fe, Mg, and Ca were 35.19 ± 11.89, 6.16 ± 2.89, and 49.78 ± 18.05 g kg^{-1}, respectively. Statistically significant differences in metal concentration were associated with both sampling date and site, although the differences were not consistent among metals. Iron was significantly greater in July 2004 and April 2005 than in June 2005, and in September 2004 versus June 2005. All pair wise contrasts among dates were significantly different for Mg, whereas the only significant difference for Ca was July 2004 being greater than June 2005. Iron was significantly greater

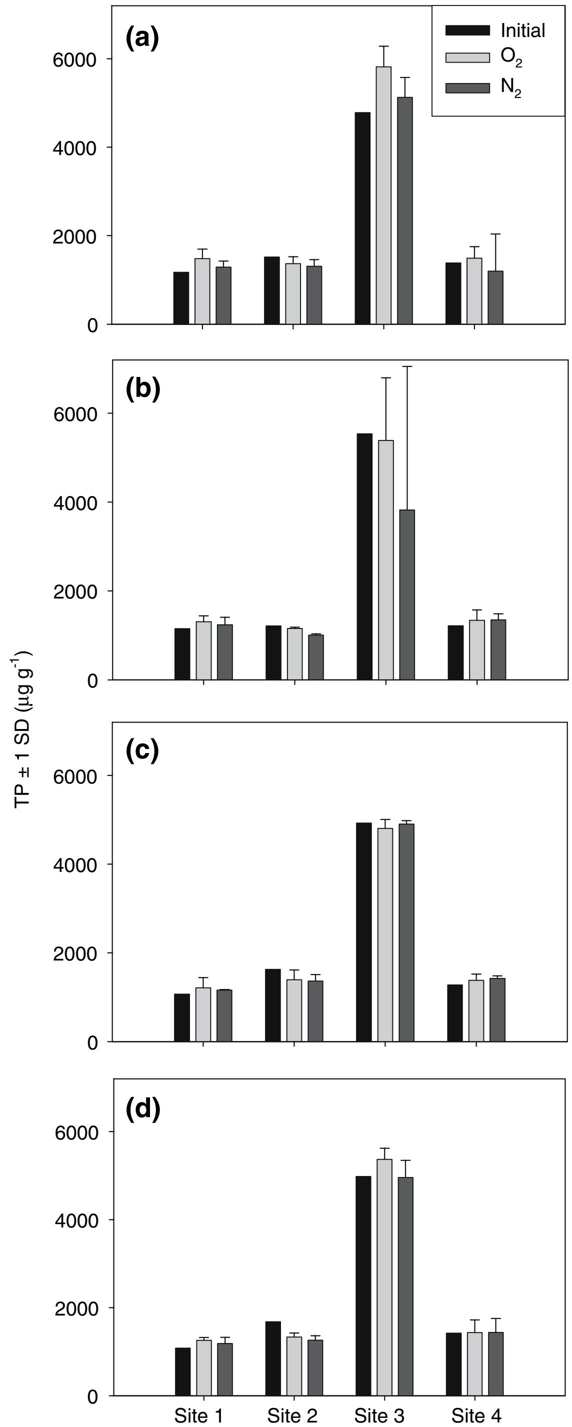


Fig. 7 Mean (±SD, except Initial values $n = 1$) TP concentration measured in ashed sediment (μg g^{-1}) from July 2004 (**a**), September 2004 (**b**), April 2005 (**c**), and June 2005 (**d**) sediment cores analyzed at the end of the laboratory incubations

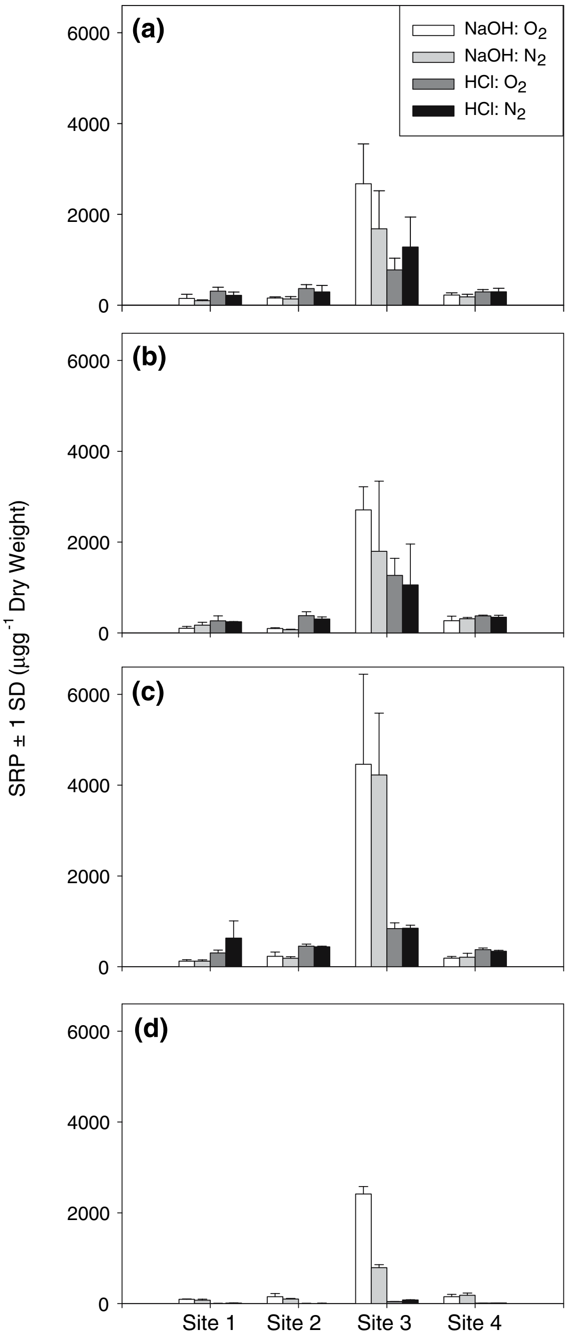


Fig. 8 Mean (±SD) NaOH–P and HCl–P (μg g^{-1} dry weight) from July 2004 (**a**), September 2004 (**b**), April 2005 (**c**), and June 2005 (**d**) extracted from sediment cores that were analyzed at the end of the laboratory incubations

at sites 2 and 4 compared to site 1, Mg was significantly greater at sites 1 and 2 compared to site 4, and greater at site 2 versus site 3, and Ca was significantly greater at site 2 than at site 4. Neither oxidation state nor the interaction terms had a significant effect on Fe, Mg, or Ca in the sediments.

Internal versus external loading analysis

The relative contribution of internal total phosphorus loading to Mona Lake under anaerobic conditions was equal to or greater than 68% during all sampling periods except April 2005, when it accounted for only 8.9% of the total load (Table 4). On a mass basis, internal loading was the dominant source of TP to Mona Lake over the sampling periods, accounting for ~68% of the total.

Discussion

Watersheds throughout the world are being exposed to an array of environmental stressors associated with population growth, changing land use practices, and increasing amounts of nonpoint source pollution (Allan 2004). Increased nutrient loading can result in both environmental and economic impairments in the watershed (Carpenter et al. 1998b; Smith 1998; Pretty et al. 2003); hence, it is critical that society gains a better understanding of where and when nutrient sources are a problem, and how to manage these sources best. In shallow lake systems where nutrient loading has taken place for an extended period, both internal and external nutrient sources can contribute to ecological impairment (Sas 1989; Reddy et al. 1996; Steinman et al. 1999).

Excessive phosphorus loading to Mona Lake is a long-standing problem (Freedman et al. 1979). Although diversion of wastewater in the 1970s to a wastewater treatment plant substantially reduced nutrient concentrations in Mona Lake (Steinman et al. 2006), the ambient levels of phosphorus are still high enough to cause impairment, as evidenced by annual cyanobacterial blooms (Hong et al. 2006; Steinman et al. 2006). The findings from the current study indicate that internal and external sources of phosphorus to Mona Lake vary across time and space. Understanding this variation may help in developing mitigation and restoration strategies.

Both TP and SRP external loading to Mona Lake were influenced by subbasin size and land use. Larger

Table 4 Internal and external TP load estimates (metric tons) compared for periods of time when internal load incubations were conducted. Internal load estimates are based on maximum release rates from anaerobic sediment cores, and therefore, are liberal estimates

Time period	Internal TP load	External TP load	Internal load contribution (% total)
27 July–16 August, 2004	0.498	0.189	72.5
14 September–4 October, 2004	0.247	0.053	82.3
14 April–4 May, 2005	0.012	0.122	8.9
7–27 June, 2005	0.278	0.131	68.0

subbasins have a greater number of potential sources and hence can contribute more nutrients to receiving water bodies, but the location and proximity of the sources to rivers also play critical roles (Alexander et al. 2000; Steinman and Denning 2005). In the Mona Lake watershed, size was related to TP load for the Black Creek subbasin, as the subbasin accounted for 54% of the total watershed area but 73% of the TP load (and 74% of the discharge) to Mona Lake. The Little Black Creek subbasin, in contrast, accounted for ~14% of total watershed area but 37% of SRP load (and 13% of the discharge) to Mona Lake. These comparisons should be viewed with caution, as our sampling methodology likely did not fully capture the peak concentrations in Black Creek during storm events (see below). When normalized per unit area, the 2004 and 2005 TP yields in the Black Creek subbasin (0.040 and 0.033 kg km^{-2} d^{-1}, respectively) were slightly less than those in the Little Black Creek subbasin (0.065 and 0.052 kg km^{-2} d^{-1}, respectively). The SRP data showed an even stronger disparity, as the 2004 and 2005 yields were much lower in the Black Creek subbasin (0.008 and 0.009 kg km^{-2} d^{-1}, respectively) than the Little Black Creek subbasin (0.030 and 0.019 kg km^{-2} d^{-1}, respectively). Thus, while subbasin size was related to absolute phosphorus loads, clearly other factors are influencing external phosphorus delivery to Mona Lake.

Land use has been shown to influence the amount of nutrients leaving the landscape (Jones et al. 2001; but see Soranno et al. 1996); agricultural and developed land uses are well-known to be major nutrient sources to water bodies (Allan 2004). In the Black Creek subbasin, where agriculture land cover is more abundant than in the other subbasins, entrainment of P-rich sediment may have accounted for the high levels of TP (cf. Daniels et al. 1998). In the more urbanized Little Black Creek subbasin, runoff has little opportunity for biological assimilation (Paul and Meyer 2001; Meyer et al. 2005). The high amount of impervious area in the Little Black Creek subbasin (22% compared to <4% in upper Black Creek subbasin; Steinman et al. 2006) results in a flashy hydrology, which in turn can lead to a quick flush of bioavailable P reaching the creek (cf. Fig. 5). This input of highly bioavailable P, in turn, may result in biological hot spots in Mona Lake, such as algal blooms (McClain et al. 2003).

This spatial variation in external P sources suggests that different management strategies may be needed within the watershed based on unique subbasin characteristics. For example, limiting sediment erosion may be an effective management practice to reduce TP in the Black Creek subbasin. However, stormwater detention, coupled with biotic assimilation such as rain gardens or constructed wetlands, may be a more effective management practice to remove SRP from the Little Black Creek subbasin.

Managing external phosphorus loading also requires an appreciation of temporal variation. External loads, in the form of tributary flow and concentration, vary within and among seasons and years (e.g., Shostell and Bukaveckas 2004). For example, P inputs from tributaries increase substantially during wet years compared to dry years (Soranno et al. 1997) because increased runoff entrains P-bound particles. In the current study, there appears to be a weak relationship between discharge and P concentration in Little Black Creek; this is likely a function of the limited retentiveness of this system, with nutrient-laden runoff quickly reaching the tributary, and being measured in our storm sampling. In contrast, peak flow can be observed more than 10 hr following storm events (based on our monitoring data) in the much larger and topographically complex Black Creek subbasin, so it is likely that our storm event sampling missed peak concentrations. Clearly, more intensive sampling, especially during storm events, would provide us with a better understanding of the temporal variability of external P loading among subbasins. Mitigation strategies for subbasins with short retention have more limited options than subbasins with long retention times; in urban subbasins with high levels of impervious surface, a higher density of smaller-sized detention devices, such as rain gardens, may be more effective than in subbasins with long retention times, where a smaller number of larger-sized detention devices, such as constructed wetlands or stormwater treatment areas, may work better (cf. Guardo et al. 1995).

Internal P loading also can be a significant source of nutrients in shallow eutrophic lakes, and can result in serious impairment to water quality (Welch and Cooke 1995; Steinman et al. 1999, 2004; Søndergaard et al. 2001; Nürnberg and LaZerte 2004). The summer TP release rates from Mona Lake under anaerobic conditions ($\sim$1–16 mg m^{-2} d^{-1}) were in the same general range as those measured in mesotrophic to eutrophic systems (Nürnberg and LaZerte 2004). The TP concentrations generated in our core tubes were an order of magnitude (or more) greater than water quality impairment thresholds (Smith 1998), indicating that internal loading can be a significant source of P in this system.

The internal loading incubations revealed the following: (1) TP release rates were significantly greater under anaerobic than aerobic conditions regardless of sampling date. This suggests that the P release in Mona Lake sediments is influenced by iron-binding properties and oxidation state (Boström et al. 1982; Petticrew and Arocena 2001), as reduced conditions will result in PO_4^{-3} desorbing from Fe(II), thereby increasing P flux. There was no apparent relationship between P release rates, and Mg or Ca sediment concentrations, suggesting that these metals had less influence on internal loading than Fe. (2) Mona Lake sediments showed the potential to serve as a P sink under aerobic conditions. Negative TP flux rates were measured in the aerobic treatments at three of the four sites during September 2004. Although the magnitude of this negative flux was relatively small, it indicates that the sediments have the capacity to bind P. (3) there was spatial variation in the release rates from Mona Lake sediments, which was related to sediment chemistry. Jensen et al. (1992) found that the Fe:TP ratio in surface sediments was a robust indicator of the adsorption capacity of oxidized sediments. They reported that SRP release from aerobic sediments was very low in lakes where the sediment Fe:P ratio (by weight) was >15. In Mona Lake, the Fe:P ratio at sites 1, 2, and 4 exceeded 15 on all sampling dates; only site 3 had ratios consistently <10. The SRP release data under aerobic conditions is consistent with the findings of Jensen et al. (1992), as overall P release rates were low at sites 1, 2, and 4, and site 3 generally had the highest release rates of TP and SRP under aerobic conditions (Table 3). However, the differences in release rates among sites were usually relatively modest ($\sim$1–2 mg P m^{-2} d^{-1}).

Spatial variability was also evident when comparing SRP and TP release rates. The highest mean SRP release rates were measured from cores sampled at site 1 in all seasons, which was not the case with TP. Lower iron concentrations at site 1 may have led to greater SRP release than TP. Spatial variability in P concentration was evident also within Mona Lake sediments. Sediment P concentrations were consistently greater at site 3 compared with other sites; this may be attributable to its location close to the mouth of Little Black Creek, although this tributary currently delivers less P to Mona Lake than Black Creek. A sewage treatment plant was located formerly on this tributary, and the legacy of those loads may be represented in the sediment P content. The substantially greater sediment P at this site translated into an internal load signal that was detectable at times: anaerobic P flux rates at this site were greater than those at the other sites in July 2004, while aerobic P flux rates were higher at this site in three out of the four sampling periods. We focused on SRP and TP, but other forms of phosphorus are likely present in Mona Lake sediments; additional studies are needed to examine the dynamics of organic P compounds, such as polyphosphates, with respect to their variability in concentration, release rate, and bioaccumulation (cf. Hupfer et al. 2004).

The spatial variability in internal loading and sediment chemistry suggests that mitigation strategies can be optimized to target specific problem areas. A similar approach has been suggested for Lake Okeechobee, Florida, where differing phosphorus release rates are related to sediment characteristics throughout this large, subtropical lake (Olila et al.

1995; Steinman et al. 1999). In Mona Lake, the area near site 3 would be a priority candidate for mitigation, such as dredging or chemical inactivation, given its high phosphorus release rates and elevated phosphorus sediment content.

Temporal variations in internal P loading under anaerobic conditions were also evident. (1) Mean TP release rates were ~2–4× greater, whereas mean SRP release rates were generally similar, in July than in September 2004. There was little seasonal difference in temperature or degree of stratification on the dates when the sediment cores were collected, so these factors are likely not responsible for the different TP release rates. (2) Mean TP and SRP release rates were ~3–4× greater in June than in April 2005. Differences in bottom water temperatures likely accounted for this difference in P flux; mean temperatures of the bottom water in Mona Lake were much lower in April than in June (Table 3). Reduced metabolic activity during colder periods results in less microbially-mediated P mineralization (Boström et al. 1982; Gächter et al. 1988; Jensen and Andersen 1992). In addition, degree of stratification may have influenced release rates at different sites in the field, although laboratory release rates would not have been affected by this factor. (3) Finally, mean summer TP release rates were ~2× greater at sites 3 and 4, and similar at sites 1 and 2, in July 2004 compared to June 2005; mean SRP release rates were similar in years at all sites. Temperatures were slightly warmer in 2004 than 2005, and stratification was more evident at site 3 (but not site 4) in 2004 than 2005, which may partially explain the differences. However, none of the collected limnological data provides an obvious explanation for the very high TP release rates at sites 3 and 4 in July 2004.

Our analysis suggests that internal loading contributes the majority of phosphorus loads to Mona Lake, and that this contribution is most evident during the summer period, when temperatures are warmer, and stratification is most developed. We suspect that external loading will be more significant in winter months, assuming the tributaries are flowing. Prior sampling indicated that the hypolimnion of Mona Lake remains oxygenated during ice cover (Steinman et al. 2006), suggesting low phosphorus release rates from sediment during the winter period. Nonetheless, internal phosphorus loading clearly is a significant process in Mona Lake, especially during summer, when algal blooms are most common (Steinman et al. 2006).

Summary

Nutrient loading is a significant problem in Mona Lake, which is situated in a rapidly developing watershed in western Michigan. External P loads appeared to be related to subbasin area and land use, and varied across time and space. Temporal and spatial variability were also observed in internal loading; the highest mean P flux rates occurred in anaerobic treatments during the summer months (up to 15.6 mg P m^{-2} d^{-1}), when internal TP loading accounted for between 73 and 82% of the TP load to Mona Lake. Aerobic treatments occasionally resulted in negative flux rates, suggesting that the sediments also can serve as a sink for P. Our study has implications for nutrient loading assessments. Remediation strategies dealing with nonpoint source pollution should adopt an integrated approach, which includes both in-lake and watershed-related sources. Assessing loading over time and space allows managers to target specific time periods or regions associated with high loading, and develop management strategies specifically tailored to the source.

Acknowledgments We are grateful to Lori Nemeth, Eric Nemeth, Brian Scull, Gail Smythe, and Jennifer Cymbola who assisted with field sampling and laboratory analyses. Constructive comments from three anonymous reviewers improved the final version of the manuscript. Funding for this project was provided by the Michigan Department of Environmental Quality.

References

Alexander RB, Smith RA, Schwarz GE (2000) Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico. Nature 403:758–761

Allan JD (2004) Landscapes and riverscapes: the influence of land use on stream ecosystems. Ann Rev Ecol Evol Syst 35:257–284

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

Carpenter SR, Bolgrien D, Lathrop RC, Stow CA, Reed T, Wilson MA (1998a) Ecological and economic analysis of lake eutrophication by nonpoint pollution. Aust J Ecol 23:68–79

Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH (1998b) Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol Appl 8: 559–568

Chu X, Mariño MA (2006) Simulation of infiltration and surface runoff: a windows-based hydrologic modeling system HYDROL-INF. In: Graham R (ed) Examining the confluence of environmental and water concerns, Proceedings of the 2006 World Environmental and Water Resources Congress. American Society of Civil Engineers, pp 1–8

Daniels TC, Sharpley AN, Lemunyon JL (1998) Agricultural phosphorus and eutrophication: a symposium overview. J Environ Qual 27:251–257

Evans ED (1992) Mona, White, and Muskegon Lakes in Muskegon County, Michigan: the 1950s to the 1980s. Michigan Department of Natural Resources, Lansing, MI. MI/DNR/SWQ-92/261

Fisher MM, Brenner M, Reddy KR (1992) A simple, inexpensive piston corer for collecting undisturbed sediment/water interface profiles. J Paleolimnol 7:157–161

Freedman P, Canale R, Auer M (1979) The impact of wastewater diversion spray irrigation on water quality in Muskegon County lakes. U.S. Environmental Protection Agency, Washington, D.C. EPA-905/9-79-006-A

Gächter R, Meyer JS, Mares A (1988) Contribution of bacteria to release and fixation of phosphorus in lake sediments. Limnol Oceanogr 33:1542–1558

Graneli W (1999) Internal phosphorus loading in Lake Ringsjon. Hydrobiologia 404:19–26

Groffman PM, Bain DJ, Band LE, Belt KT, Brush GS, Grove JM, Pouyat RV, Yesilonis IC, Zipperer WC (2003) Down by the riverside: urban riparian ecology. Front Ecol Environ 6:315–321

Guardo M, Fink L, Fontaine TD, Newman S, Chimney M, Bearzotti R, Goforth G (1995) Large-scale constructed wetlands for nutrient removal from stormwater runoff: an everglades restoration project. Environ Manage 19: 879–889

Hong Y, Steinman AD, Biddanda B, Rediske R, Fahnenstiel G (2006) Occurrence of the toxin-producing cyanobacterium *Cylindrospermopsis raciborskii* in Mona Lake, a drowned river mouth tributary of Lake Michigan. J Great Lakes Res 32:645–652

Hupfer M, Rübe B, Schmeider P (2004) Origin and diagenesis of polyphosphate in lake sediments: A ^{31}P-NMR study. Limnol Oceanogr 49:1–10

Jensen HS, Andersen FØ (1992) Importance of temperature, nitrate, and pH for phosphate release from aerobic sediments of four shallow, eutrophic lakes. Limnol Oceanogr 37:577–589

Jensen HS, Kristensen P, Jeppensen E, Skytthe A (1992) Iron:phosphorus ratio in surface sediment as an indicator of phosphate release from aerobic sediments in shallow lakes. Hydrobiologia 235/236:731–743

Jones KB, Neale AC, Nash MS, Van Remortel RD, Wickham JD, Riitters KH, O'Neill RV (2001) Predicting nutrient and sediment loadings from landscape metrics: a multiple watershed study from the United States Mid-Atlantic Region. Landscape Ecol 16:301–312

Levin RB, Epstein PR, Ford TE, Harrington W, Olson E, Reichard EG (2002) U.S. drinking water challenges in the Twenty-First century. Environ Health Perspect 110:43–52

Loomis J, Kent P, Strange L, Fausch K, Covich A (2000) Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey. Ecol Econ 33:103–117

Ludwig D, Carpenter S, Brock W (2003) Optimal phosphorus loading for a potentially eutrophic lake. Ecol Appl 13:1135–1152

McClain ME, Boyer EW, Dent CL, Gergel SE, Grimm NB, Groffman PM, Hart SC, Harvey JW, Johnston CA, Mayorga E, McDowell WH, Pinay G (2003) Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6:301–312

MDEQ (Michigan Department of Environmental Quality) (2000) A biological survey of Big Black Creek, Muskegon County. MDEQ, Lansing, MI. MI/DEQ/SWQ-00/050

MDEQ (Michigan Department of Environmental Quality). (2002) A biological and chemical assessment of Big Black Creek, Muskegon County. MDEQ, Lansing, MI. MI/DEQ/SWQ-02/030

Meyer JL, Paul MJ, Taulbee K (2005) Stream ecosystem function in urbanizing landscapes. J N Am Benthol Soc 24:602–612

Moore PA, Reddy KR (1994) Role of Eh and pH on phosphorus geochemistry in sediments of Lake Okeechobee, Florida. J Environ Qual 23:955–964

Nürnberg GK, LaZerte BD (2004) Modeling the effect of development on internal phosphorus load in nutrient-poor lakes. Water Resour Res 40, W01105, doi:10.1029/2003WR002410

Olila OG, Reddy KR, Harris WG (1995) Forms and distribution of inorganic phosphorus in sediments of two shallow eutrophic lakes in Florida. Hydrobiologia 302:147–161

Paul MJ, Meyer JL (2001) Streams in the urban landscape. Annu Rev Ecol Syst 32:333–365

Petticrew EL, Arocena JM (2001) Evaluation of iron-phosphate as a source of internal lake phosphorus loadings. Sci Total Environ 266:87–93

Pretty JN, Mason CF, Nedwell DB, Hine RE, Leaf S, Dils R (2003) Environmental costs of freshwater eutrophication in England and Wales. Environ Sci Technol 37: 201–208

Reddy KR, Fisher MM, Ivanoff D (1996) Resuspension and diffusive flux of nitrogen and phosphorus in a hypereutrophic lake. J Environ Qual 25:363–371

Sas H (1989) Lake restoration by reduction of nutrient loading: expectations, experiences and extrapolation. Academia-Verlag, Richarz

Schindler DW (2006) Recent advances in the understanding and management of eutrophication. Limnol Oceanogr 51:356–363

Schueler TR, Holland HK (2000) The practice of watershed protection. Center for watershed protection. Ellicott City, Maryland

Shostell J, Bukaveckas PA (2004) Seasonal and interannual variation in nutrient fluxes from tributary inputs, consumer recycling and algal growth in a eutrophic river impoundment. Aquat Ecol 38:359–373

Smith VH (1998) Cultural eutrophication of inland, estuarine, and coastal waters. In: Pace ML, Groffman PM (eds) Successes, limitations, and frontiers in ecosystem science. Springer, New York, pp 7–49

Smith VH, Joye SB, Howarth RW (2006) Eutrophication of freshwater and marine ecosystems. Limnol Oceanogr 51:351–355

Søndergaard M, Jensen JP, Jeppesen E (2001) Retention and internal loading of phosphorus in shallow, eutrophic lakes. The Scientific World 1:427–442

Soranno PA, Hubler SL, Carpenter SR, Lathrop RC (1996) Phosphorus loads to surface waters: a simple model to account for spatial pattern of land use. Ecol Appl 6: 865–878

Soranno PA, Carpenter SR, Lathrop RC (1997) Internal phosphorus loading in Lake Mendota: response to external loads and weather. Can J Fish Aquat Sci 54:1883–1893

Steinman AD, Denning R (2005) The role of spatial heterogeneity in the management of freshwater resources. In: Lovett GM, Jones CG, Turner MG, Weathers KC (eds) Ecosystem function in heterogenous landscapes. Springer, New York, pp. 367–387

Steinman AD, Havens KE, Aumen NG, James RT, Jin K-R, Zhang J, Rosen B (1999) Phosphorus in Lake Okeechobee: sources, sinks, and strategies. In: Reddy KR, O'Connor GA, Schelske CL (eds) Phosphorus biogeochemistry of subtropical ecosystems: Florida as a case example. CRC/Lewis Publishers, Boca Raton, FL, pp 527–544

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

Steinman AD, Rediske R, Reddy KR (2004) The reduction of internal phosphorus loading using alum in Spring Lake, Michigan. J Environ Qual 33:2040–2048

Steinman AD, Rosen BH (2000) Lotic-lentic linkages associated with Lake Okeechobee, Florida. J N Am Benthol Soc 19:733–741

U.S. Environmental Protection Agency (1983) Methods for chemical analysis of water and wastes. U.S. Environmental Protection Agency, Cincinnati, OH. EPA-600-/ 4–79-020

U.S. Environmental Protection Agency (1994) Test methods for evaluating solid waste, physical/chemical methods. Doc. No. 995-001-0000001. U.S. Government Printing Office, Washington, DC

U.S. Environmental Protection Agency (2000) The quality of our nation's waters, a summary of the national water quality inventory: 1998 Report to Congress. EPA 841-S-00-001

U.S. Environmental Protection Agency (2004) Wadeable streams assessment: field operations manual. Office of water and office of research and development. U.S. Environmental Protection Agency, Washington, DC. EPA/841-B-04-004

Walsh CJ, Roy AH, Feminella JW, Cottingham PD, Groffman PM, Morgan RP II (2005) The urban stream syndrome: current knowledge and the search for a cure. J N Am Benthol Soc 24:706–723

Welch EB, Cooke GD (1995) Internal phosphorus loading in shallow lakes: importance and control. Lake Reserv Manage 11:273–281