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An environmental assessment of an impacted, urbanized watershed: the Mona Lake Watershed, Michigan

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With 8 figures and 7 tables

Abstract: The ecological health and integrity of watersheds throughout the world are being threatened by a variety of stressors. Often, restoration practices focus on single problems whereas comprehensive, multidisciplinary approaches are needed to address both the symptoms and underlying causes of impairment. A comprehensive assessment of a small, urbanized watershed in west Michigan, USA was conducted to evaluate the major stressors in the system. This assessment approach for the Mona Lake watershed included analyses of land use/land cover change, water quality in both the major surface inflows and the receiving water body, and toxic inputs into a major inflow. Because these issues are common to many watersheds, we developed a conceptual model that spatially links these stressors and predicted impacts, allowing us to assess them in a comprehensive manner. Based on our results, we generated a set of recommendations targeted for specific source or problem areas. This approach can be applied to other watersheds.

Key words: eutrophication, nonpoint source pollution, watershed management, contaminated sediments, Mona Lake, Michigan.

Introduction

Pressures associated with human population growth and land development are resulting in greater strains on our natural resources. Water bodies are particularly vulnerable to these pressures, as they integrate pollutants over space and

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time, and ultimately serve as indicators of landscape processes (Naiman et al. 2002, Steinman & Denning 2005). Many of the stressors result from land use activity associated with either agricultural practices or urbanization in the watershed. Urbanized watersheds face a variety of potential environmental problems, including 1) unnatural hydrology associated with stormwater discharge; 2) elevated nonpoint source pollutants, especially nutrients and sediment, related to increased impervious surfaces; 3) habitat impairment; and 4) toxic sediments associated with past or present industrial activities (Baer & Pringle 2000, Paul & Meyer 2001, Morley & Karr 2002, Groffman et al. 2003, Hatt et al. 2004, Walsh et al. 2005). Developing management plans that adequately address these concerns requires a comprehensive approach that is multifaceted, based on sound science, and accounts for the potential linkages among the stressors in the system.

The Mona Lake Watershed, located in west Michigan (Fig. 1), faces many of the environmental and socio-economic challenges common to watersheds throughout the world, including industrial and wastewater discharges (Freedman et al. 1979), nonpoint source pollution, mid-summer cyanobacterial blooms, and a wide gap in the social and economic demographics of the residents within the basin (Delta Institute 2003). The present study presents a multidisciplinary approach for addressing these stressors that can be applied to other watersheds. The approach involves analyzing changes in both land use/ land cover and water quality, evaluating nutrient loads by sub-basins within the watershed, and examining the relationship between contaminated sediments and biotic community structure. Ultimately, the data are synthesized into a conceptual model and used to generate recommendations that can be applied to restore the ecological health and integrity of the watershed under consideration.

Study site

The Mona Lake Watershed covers approximately 200 km^2, and is located almost entirely within Muskegon County, Michigan (Fig. 1). The watershed consists of three major hydrologic components: (1) Mona Lake, (2) Black Creek, and (3) Little Black Creek. There also are a number of smaller tributaries and storm drains that enter from the north and south sides of Mona Lake (Fig. 2). Each of the three main water bodies suffers from chemical and biological degradation, as described in more detail below.

Mona Lake: Mona Lake has a surface area of approximately 2.65 km^2, which is about 1.4 % of the total watershed area. Hydraulic retention times vary from 105 to 160 days during low flow periods to less than 35 days during high flow periods (Evans 1992). Documented impacts on biota in Mona Lake

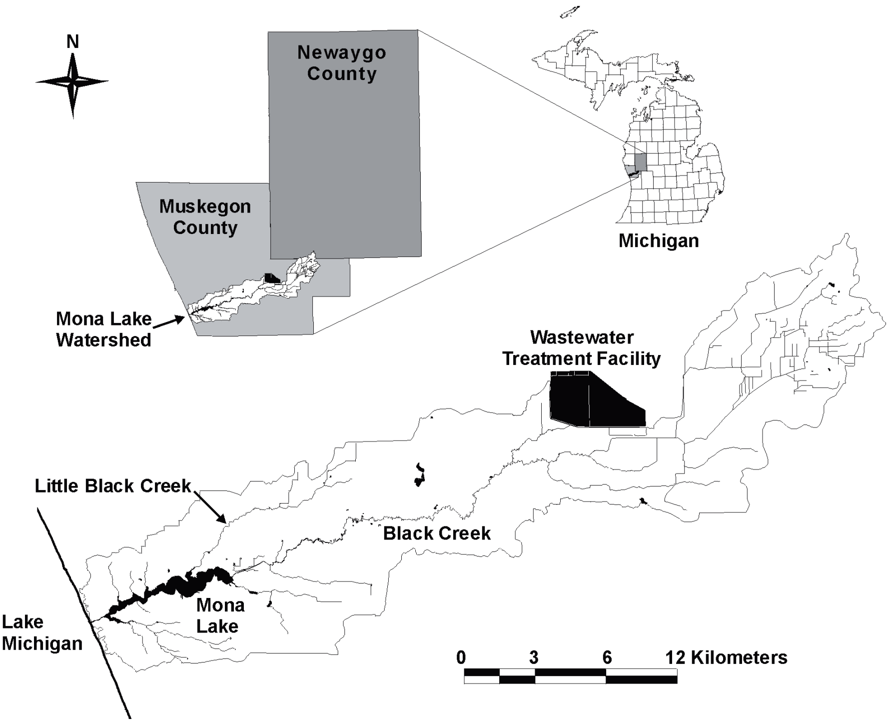

Fig. 1. Location of the Mona Lake Watershed in Muskegon and Newaygo Counties, Michigan (top). Hydrography of the Mona Lake Watershed; the major features include Black Creek, Little Black Creek, and Mona Lake (bottom). Most of the effluent from the wastewater treatment facility flows north to an adjacent watershed.

extend back to 1956, when frequent fish kills due to low dissolved oxygen concentrations were reported (Evans 1992).

Black Creek: Black Creek is the major tributary to Mona Lake. Two U.S. EPA superfund sites are located adjacent to this stream; each facility has a groundwater capture and treatment system to process volatile organic compounds and aromatic amines (MDEQ 2002). A fish consumption advisory was issued for Black Creek based on PCB concentrations in carp and white sucker that were collected in 1987. The last native trout documented in Black Creek was in the early to mid-1960 s (MDEQ 2000 a). The creek was placed on the U.S. EPA section 303(d) list, indicating it does not meet water quality standards, because of a fish community rated as "poor" and insufficient numbers of individual fish. The Michigan Department of Environmental Quality (MDEQ) suggested that the primary reason for the poor fish community was excessive sand bedload in the channel.

Table 1. Characteristics of all major inflows to Mona Lake. Refer to Fig. 2 for site locations. Range of flow is for low flow conditions only. S = south side of lake; N = north side of lake; T = tributary; D = storm drain.

Site	Inflow Type	Dominant Substrate	Flow Range (m^3/s)	Unique Features
ST1	tributary	sand & organics	0.024–0.137	flows through small wetland; residential area
ST2	tributary	sand & organics	0.001–0.003	residential area
ST4	tributary	sand & organics	0.001–0.002	residential area
ST5	drain	N/A	0.003*	drains road runoff behind airport directly to lake
ST6	tributary	sand & rock	0.050–0.163	flows through small wetland
Black Creek	tributary	sand	0.482–2.690	drains residential and agricultural areas
Little Black Creek	tributary	sand & organics	0.040–0.442	high levels of contaminants; urban area
ND1	drain	N/A	0.047**	pumped from a greenhouse drainage pond
ND2	drain	N/A	0.001–0.003	flowed continuously; drain source unknown
ND3	drain	N/A	0.001–0.008	flowed continuously; drain source unknown
NT1	tributary	sand	0.003–0.017	residential area
NT2	tributary	sand & organics	0.004–0.019	residential area
NT3	tributary	sand & organics	0.010–0.032	flows through small wetland; residential area
Channel	outlet to Lake Michigan	sand	N/A	bidirectional flow to and from Lake Michigan

* ST5 had detectable flow on only 1 date (March 17, 2003).
** ND1 had detectable flow on only 2 dates (July 22, 2002, June 4, 2003).

Little Black Creek: Little Black Creek is a second order stream that flows through heavily urbanized areas, including portions of the cities of Muskegon and Muskegon Heights. A number of industries are located adjacent to this waterway and currently discharge stormwater directly into the creek from 19 outfalls. Historically, sources of contamination and impaired water quality included: (1) a petroleum refinery site (crude oil and light hydrocarbons); (2) storm sewers from foundry and metal finishing industries (oils, grease, PAH compounds, heavy metals, PCBs); (3) a plating Superfund site (cadmium, chromium, copper, nickel, zinc); (4) spills from a municipal sanitary/industrial wastewater pump station; and (5) an abandoned municipal landfill without a leachate collection system (MDEQ 2000 a, 2002). The refinery, plating, and landfill sites are no longer in operation, but they remain sites of environmental concern because contaminated sediments and groundwater plumes continue to impair water quality. Based on surveys conducted in 1996 and 2001, the sediments throughout Little Black Creek are contaminated with a number of metals and organic chemicals (MDEQ 2000 a, 2002). Similar to Black Creek, Little Black Creek was placed on the U.S. EPA section 303(d) list because it does not support coldwater fish and macroinvertebrates.

Methods

Land use/land cover analysis

Land cover analyses were based on the Michigan Resource Information System (MiRIS) land-use database, which was developed by the Michigan Department of Natural Resources. This database contains land use/cover to Anderson Level III (Anderson et al. 1976) for areas larger than 2.5 acres. Land use/cover was interpreted from 1:24000 scale color infrared, leaf-on photography from 1978. The land use/cover inventory completed for this project was based on June, 1997 aerial slides provided by the U.S. Department of Agriculture (Farm Service Agency, Newaygo Service Center, Fremont, MI). These slides are at a scale of 1:15840, are true color, and show leaf-on conditions. For the current inventory, the same classification system and minimum mapping unit standard were utilized. To complete the inventory, we used manual photo-interpretation methods to identify and update land use and cover conditions. The 1978 digital files were then updated using ArcView 3.3 GIS (ESRI) editing tools to reflect the 1997 conditions. Summary statistics by grouped land use/cover category were created using ArcView GIS (ESRI). The percent impervious map was created by applying impervious coefficients of various land use types (USDA 1975).

Inflow sampling: water quality analysis

All inflows with detectable flow during low flow periods (14 total) were selected for monitoring (Table 1, Fig. 2). Inflow sampling occurred monthly from June 2002 through August 2003, between 9:00 and 15:00 hours each day. A Hydrolab Data-Sonde 4 a was used to measure dissolved oxygen (DO), redox, turbidity, pH, temperature, specific conductance, and total dissolved solids. Current velocity was measured at a depth of 0.6D with a Marsh-McBirney Flo-Mate Model 2000 flowmeter. The velocity-area method was used to compute water discharge. For storm drains, where a flowmeter could not be used, we estimated flow by measuring the time needed to fill a bottle to a known volume. This measurement was repeated 3–5 times per site.

Grab samples for nutrients were collected in acid-washed 1-liter bottles. Samples were stored on ice until transported to the laboratory, always within 5 hr of collection. Subsamples were filtered immediately in the laboratory for soluble reactive phosphorus (SRP), nitrate, sulfate, and chloride. Samples were stored at 4 °C until analysis. Ammonium, total kjeldahl nitrogen (TKN), SRP, and total phosphorus (TP) were analyzed on a BRAN+LUEBBE Autoanalyzer (U.S. EPA 1983). Nitrate, sulfate, and chloride were analyzed by ion chromatography on a Dionex DX500 (APHA 1999). Fecal coliforms were measured by membrane filtration following Standard Methods (APHA 1999). Chemical constituents with concentrations below detection limits were assigned a value of one-half the detection limit. QA/QC procedures followed method

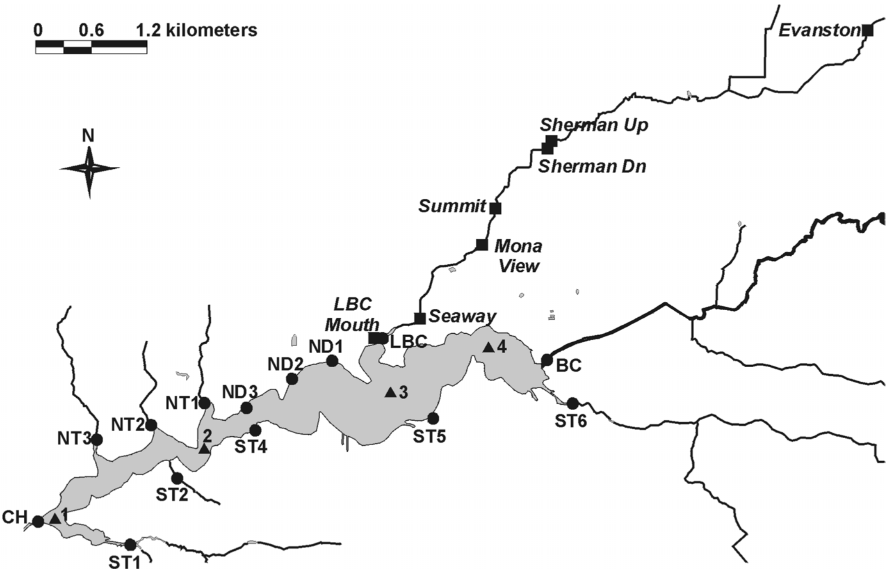

Fig. 2. Sampling locations of (1) inflows to and outflow (channel) from Mona Lake (circles); (2) lake sites (triangles); and (3) Little Black Creek (sediment only: squares). See Table 1 for description of inflows and outflow. N = north side of lake; S = south side of lake; T = tributary; D = storm drain.

guidelines including 10 % method blanks and 10 % matrix spikes/matrix spike duplicates (± 15 % limits for precision and accuracy).

Differences in nutrient concentrations among inflows were determined using nonparametric statistical analyses. For each parameter, the inflows were ranked from highest to lowest for each sampling event month; this approach normalized for seasonal variation, which could otherwise overwhelm the among-inflow differences. Inflow ST5, which had detectable flow on only one occasion, was omitted from this analysis. A Kruskal-Wallis test was used to analyze for overall significant differences. Pairwise contrasts were conducted using Mann-Whitney U tests to identify if inflows were significantly different from each other. This approach assumed that the monthly water column samples were independent of one another; this was a reasonable assumption given the flow regimes in these tributaries.

Estimation of nutrient loads under low flow conditions

Based on the observed stream flow and nutrient concentration data, nutrient loads during low flow conditions were estimated from all tributaries of Mona Lake. We define low flow conditions as those when baseflow is the dominant contributor of flow to the system. Estimates were made for five major nutrients: nitrate (NO_3-N), ammonium (NH_4-N), TKN, SRP, and TP. The analysis period ranged from June 2002 to June 2003. The average low flow discharge and the total volume of low flow were first calculated by using the measured stream flow data for each tributary. Then, loads of the

Table 2. Percent change in major land use/land cover categories and subcategories (all with >50 acres of net change) in Mona Lake watershed from 1978 to 1997.

Category	Acres: 1978	% of Total: 1978	Acres: 1997	% of Total: 1997	Net Change (acres)	Net Change (%)
Natural Cover	22,320	49.0	23,601	51.8	1,281	5.7
Agricultural	11,104	24.4	7,503	16.5	−3,601	−32.4
Developed	12,146	26.6	14,466	31.7	2,320	19.2
Total	45,570		45,570			

five major nutrients were estimated by using the computed annual low flow and the mean concentrations of the nutrients. This calculation does not capture loads during peak flows, when nutrient transport can be significant (e. g. Brett et al. 2005); thus, these data represent an underestimate of the true loads reaching Mona Lake. However, the data are still valuable in comparing the relative contributions from the different subwatersheds, and this approach avoids the problem of estimating peak flow loads from low flow conditions, when there are insufficient data to justify the extrapolation (cf. Robertson 2003).

Differences in nutrient loads per inflow were determined statistically using the same approach outlined above for nutrient concentrations. After ranking inflows, a Kruskal-Wallis test was used to analyze for overall significant differences. Pairwise contrasts were conducted using Mann-Whitney U tests to identify if inflows were significantly different from each other.

Little Black Creek sampling: sediment analysis

Seven locations were sampled for the assessment of contaminated sediments in Little Black Creek; Evanston served as the background site because it is located upstream of known contamination (Fig. 2). Site selection was based on two criteria: presence of organic sediments and known locations of past industrial discharges. Surficial sediment samples (0–10 cm depth) were collected using a stainless steel trowel. Eight samples were collected at each site, composited in the field, and transferred to a pre-cleaned 4-L glass jar. Samples were collected in depositional areas containing silts and organic sediments. Cadmium, lead, and chromium were measured via inductively coupled plasma atomic emission spectroscopy, and PCBs and semivolative compounds were measured via GC/MS, according to U.S. EPA protocols (U.S. EPA 1999).

Little Black Creek Sampling: macroinvertebrate analysis

Macroinvertebrate samples were collected from the same sites where contaminated sediments were collected in Little Black Creek, with the exception of upstream Sherman (Fig. 2). At least three replicate samples were collected from each site using a standard D-frame dip net with 0.5-mm mesh netting. Dip net sampling included sweeps just below the surface of the water, at mid-depths, and at the sediment surface in each available habitat type. In an attempt to remove variation due to habitat type, substrata sampled consisted of macrophytes, sand, large woody debris, and detritus at each site. Samples were picked for 30 person-minutes or until 150 invertebrates were collected, whichever came first. Invertebrates were preserved in a 70 % ethanol solution and returned to the laboratory, where they were sorted to lowest operational taxonomic unit (genus or species for insects, and phylum, order, or family for other organisms) (Thorp & Covich 1991, Merritt & Cummins 1996). Count data of invertebrate taxa were analyzed using correspondence analysis (CA) (Steinman et al. 2003 a). This multivariate ordination method evaluates correspondence between rows (e. g. sites) and columns (e. g. taxa) of a matrix of categorical data. CA is a form of indirect gradient analysis that maximizes the variation between row points (sites) using the columns (taxa). CA includes species scores with site scores, thereby identifying those taxa responsible for ordinating sites.

Lake Sampling: water quality analysis

Sampling of Mona Lake took place monthly at four sites between May 2002 and October 2002, in February 2003 (through the ice), and again monthly between April 2002 and August 2003. Sites were selected to cover the geographic gradient in the lake (Fig. 2). Similar to tributary sampling, all measurements occurred between 9:00 and 15:00 hours each day. A Hydrolab DataSonde 4 a equipped with a Turner Designs fluorometer was used to measure depth, DO, redox, turbidity, pH, temperature, and specific conductance. In addition, a Secchi disk was used to measure water clarity and a Li-Cor quantum sensor and data logger were used to measure incident and underwater irradiance. Water samples for nutrient analysis were collected 1 m below the water surface and ~ 1 m from the lake bottom with a van Dorn bottle, placed in acid-washed bottles, and stored on ice until delivery to the laboratory, always within 5 hr of collection. Nutrient analyses and quality controls were performed as described for tributary sampling. Chlorophyll-*a* was analyzed according to standard methods (APHA 1999).

Bioassays

Nutrient limitation of the lake plankton was assessed during the summer and fall of 2003 using bioassays (cf. Havens et al. 1996). Approximately 200 L of surface water was collected from the middle of Mona Lake (to be representative of the entire lake), brought back to the lab in acid-washed containers, and pooled into a 250 L barrel. Under constant mixing, the integrated water sample was dispensed into 12 acid-cleaned 10 L clear, polycarbonate carboys. We used 4 treatments: nitrogen only (N: 10 × ambient), phosphorus only (P: 10X ambient), nitrogen + phosphorus (N + P: each 10 × ambient), and control (C), with three replicates for each treatment. Potassium nitrate and potassium phosphate were added to the treatment carboys to achieve the appropriate nutrient concentrations. The carboys were attached to the sides of a floating rack held in place by a float and anchor assembly, with the mid-depth of the carboy approximately 0.5 m below the water surface, and incubated for 4 days. Chlorophyll-*a* was analyzed as described above.

Results

Land use / land cover

Between 1978 and 1997, the percent of watershed under: 1) natural cover increased from 49.0 % to 51.8 % (5.7 % increase), 2) agricultural use decreased from 24.4 % to 16.5 % (32.4 % decrease), and 3) developed use increased from 26.6 % to 31.7 % (15.6 % increase) (Table 2). Overall, in terms of acreage, natural cover increased by 1282 acres from 1978 to 1997 (Table 2). Developed-use land cover experienced an overall increase of 2320 acres, while there was a net decline of 3601 acres in agricultural land use from 1978 to 1997 (Table 2). Imperviousness ranged from 2 % in the upper reaches of the watershed to 32 % impervious in the subbasins surrounding Mona Lake, including the Little Black Creek subbasin that drains through the city of Muskegon Heights (Fig. 3).

Inflows and outflow: water quality

The inflows to Mona Lake included tributaries flowing through urban, suburban residential and agricultural land uses, as well as storm drains that discharged directly to the lake (Table 1). Annual median water temperatures ranged from 9.5 °C in Little Black Creek to 14.5 °C at the outflow in the chan-

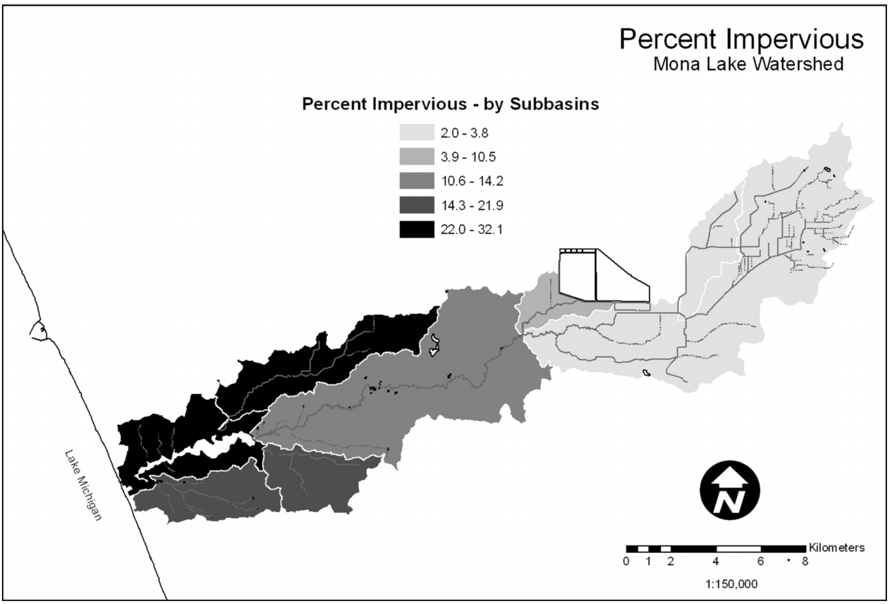

Fig. 3. Percent impervious cover within the Mona Lake watershed.

Table 3. Median (and range) of selected variables measured in all major inflows to Mona Lake from June 2002 to August 2003 under low flow conditions. Refer to Fig. 2 for site locations. FC = fecal coliform as geometric means; S = south side of lake; N = north side of lake; T = tributary; D = storm drain. Number of observations for all parameters is 15 unless otherwise noted.

Site	Temp. (˚C)	DO (mg/L)	Spec. cond. (µS/cm)	Cl (mg/L)	Nitrate (mg/L)	TKN (mg/L)	SRP (µg/L)	TP (µg/L)	FC colonies (#/100 mL)
ST1	12.8 (0.1–26.5)	10.9 (8.4–14.1)	415 (353–478)	57 (23–90)	1.1 (0.6–2.5)	0.5 (0.2–2.1)	5 (5–30)	40 (20–200)	183 (16–980)
ST2	11.1 (−0.2–21.6)	9.4 (7.4–14.3)	430 (125–524)	71 (37–150)	1.9 (1.3–2.5)	0.6 (0.1–2.0)	25 (5–50)	55 (20–330)	173 (16–1000)
ST4	11.2 (0.2–22.5)	8.0 (4.0–14.4)	410 (300–446)	45 (20–94)	0.6 (0.3–1.2)	0.7 (0.3–1.0)	5 (5–70)	65 (30–130)	290 (50–3200)
ST5*	12.2 (N/A)	11.6 (N/A)	715 (N/A)	210 (N/A)	0.3 (N/A)	2.3 (N/A)	60 (N/A)	820 (N/A)	16 (N/A)
ST6	10.9 (1.1–17.6)	10.5 (8.5–13.9)	587 (513–678)	102 (60–148)	1.4 (0.9–2.1)	0.3 (0.2–0.7)	5 (5–30)	30 (20–60)	170 (16–2690)
Black Creek	9.5 (−0.2–22.0)	10.9 (8.5–14.8)	438 (336–525)	46 (19–100)	0.7 (0.4–1.4)	0.6 (0.3–1.3)	10 (5–40)	60 (30–100)	280 (16–2600)
Little Black Creek	11.2 (−0.2–20.9)	9.0 (6.2–12.1)	747 (498–1042)	125 (31–270)	0.9 (0.5–1.5)	0.4 (0.1–0.4)	5 (5–30)	45 (30–100)	410 (16–5800)
ND1**	12.0 (0.5–18.7)	7.8 (5.9–8.0)	754 (721–874)	114 (70–140)	1.3 (0.8–1.9)	0.6 (0.4–0.7)	30 (5–40)	60 (40–100)	455 (17–680)
ND2	11.6 (2.8–22.5)	10.1 (7.6–12.4)	1180 (863–1426)	210 (39–340)	2.9 (0.1–3.4)	0.8 (0.1–1.1)	5 (5–20)	10 (10–50)	16 (10–2200)
ND3	10.7 (5.3–25.2)	9.2 (6.6–14.1)	1487 (926–4755)	295 (70–1300)	3.0 (0.3–4.9)	0.5 (0.2–2.1)	15 (5–60)	30 (20–260)	253 (16–2500)
NT1	12.4 (−0.7–20.5)	10.1 (8.1–13.7)	542 (136–1247)	74 (49–160)	1.6 (1.1–1.7)	0.4 (0.2–0.9)	5 (5–40)	25 (10–150)	420 (16–2400)
NT2	11.3 (−0.4–24.0)	10.1 (7.1–13.0)	409 (229–460)	34 (26–92)	1.1 (0.2–1.2)	0.7 (0.4–1.8)	5 (5–20)	50 (30–80)	230 (33–2100)
NT3	13.2 (−0.2–22.3)	10.7 (7.8–13.7)	529 (464–630)	71 (42–127)	2.1 (1.8–2.8)	0.4 (0.2–1.2)	20 (5–30)	35 (30–100)	188 (16–1400)
Channel***	14.5 (0.2–25.9)	11.7 (5.5–16.0)	418 (355–514)	46 (26–85)	0.2 (0.0–0.9)	0.7 (0.3–2.1)	5 (5–20)	50 (30–100)	16 (16–67)

* ST5 had detectable flow on only 1 date (March 17, 2003).
** ND1 had detectable flow on only 3 dates (July 22, 2002, February 11, 2003, June 4, 2003).
*** Measured at surface only.

Table 4. Absolute (kg/y) and relative (%) loads of nitrate, TKN, SRP, and TP, and relative (%) discharge, from June 2002 to June 2003, under low flow conditions at all measurable inflows to Mona Lake. Site abbreviations as in Table 3.

Site	Variable				Low Flow Discharge (%)
	TKN kg/y (%)	Nitrate kg/y (%)	SRP kg/y (%)	TP kg/y (%)	
ST1	1,319 (5.2)	2,585 (6.4)	23.1 (3.8)	116.5 (4.3)	4.7
ST2	28 (0.1)	79 (0.2)	0.9 (0.2)	3.5 (0.1)	<0.1
ST4	23 (0.1)	22 (0.1)	0.5 (0.1)	2.6 (0.1)	<0.1
ST5*	15 (0.1)	2 (0.01)	0.4 (0.1)	5.5 (0.2)	<0.1
ST6	862 (3.4)	3,779 (9.3)	25.1 (4.1)	79.3 (2.9)	5.3
Black Creek	20,865 (81.9)	26,770 (66.0)	492.1 (80.9)	2,221.4 (82.1)	77.2
Little Black Creek	1,486 (5.8)	4,490 (11.1)	42.4 (7.0)	205.3 (7.6)	9.3
ND1**	142 (0.6)	323 (0.8)	6.5 (1.1)	16.1 (0.6)	0.5
ND2	32 (0.1)	120 (0.3)	0.3 (0.1)	0.9 (0.1)	0.1
ND3	62 (0.2)	340 (0.8)	1.5 (0.3)	5.2 (0.2)	0.2
NT1	85 (0.3)	304 (0.8)	2.4 (0.4)	6.7 (0.3)	0.4
NT2	252 (1.0)	328 (0.8)	2.5 (0.4)	16.0 (0.6)	0.7
NT3	312 (1.2)	1,397 (3.5)	10.7 (1.8)	28.4 (1.1)	1.4

* ST5 had detectable flow on only 1 date (March 17, 2003).
** ND1 had detectable flow on only 3 dates (July 22, 2002, February 11, 2003, June 4, 2003).

nel (Table 3). Water temperature was not significantly different among sites ($P > 0.07$), although notable findings included: 1) a relatively narrow range for ST6, suggesting a groundwater influence; and 2) warmer median temperatures at the two stormwater discharge sites that were sampled year-round (ND2 and ND3), mainly because of warmer winter temperatures at these sites compared to the other inflows (Steinman et al. 2003 b).

Median dissolved oxygen (DO) concentrations ranged from 7.8 (ND1) to 11.7 (Channel) mg/L (Table 3). In general, minimum DO concentrations exceeded the water quality standard of 5 mg/L (Table 3). No significant differences in DO were found among sites ($P > 0.05$). Specific conductance and chloride concentrations closely tracked each other (Table 3); median specific conductance values ranged from 409 (NT2) to 1487 (ND3) µS/cm, and median chloride concentrations ranged from 34 (NT2) to 295 (ND3) mg/L. Significant differences were detected among inflows for both parameters ($P < 0.001$). Pairwise comparisons revealed that stormwater drains and Little Black Creek had significantly greater values than all other sites. The highest concentrations were measured in February, presumably due to road salt application (data not shown).

Annual median nitrate concentration ranged from 0.2 (channel) to 3.0 (ND3) mg/L (Table 3). Concentrations were significantly different among inflows ($P < 0.001$), with significantly greater nitrate at stormwater drains ND2 and ND3 than at all tributary sites. Median TKN concentrations ranged from 0.3 (ST6) to 2.3 mg/L at ST5 (Table 3), and were significantly greater at ST2, ST4, ND2, and NT2 than at either ST6 or Little Black Creek. Median ammonium concentrations ranged from 0.03 (several sites) to 0.24 mg/L (ST5; data not shown), and were significantly greater at stormwater drains ND1, ND2, and tributaries ST4 and Little Black Creek than at NT1, NT2, ST2, and ST6.

Median SRP concentrations ranged from 5 to 30 μg/L except for the 60 μg/L value at ST5 (Table 3). The only statistically significant pairwise contrast revealed greater SRP at ST2 than at ND2. Median TP concentrations ranged from 10 to 65 μg/L (with the exception of ST5 at 820 μg/L; Table 3). NT1 and ST6 had significantly lower TP concentrations than ND1, ST2, ST4, NT2, Black Creek, and Little Black Creek. In addition, the TP concentration at ND2 was significantly lower than the TP at the six inflows listed above, as well as at ST1 and NT3. Median fecal coliform counts ranged from < 20 (ST5 and Channel sites) to > 300 (Little Black Creek, ND1, and NT1; Table 3). Pairwise contrasts revealed that only ND2 was significantly different among sites, having lower coliform counts than all sites except ST2 and Black Creek.

Nutrient load

Four of the 13 sources of water to Mona Lake accounted for 96.5 % of the total inflow under low flow conditions: Black Creek, Little Black Creek, ST6, and ST1 (Table 4). Black Creek accounted for the majority of the nutrient load into Mona Lake during low flow, ranging from 66 % of the nitrate to > 80 % of the TKN, SRP, and TP (Table 4). The relative load of nitrate entering Mona Lake from the stormwater drains was 1.6 to 4 times greater than relative discharge from these drains, although the absolute amount of nitrate from these sources was small (Table 4).

Table 5. Absolute (kg km^{-2} yr^{-1}) and relative (%) yields of nitrate, TKN, SRP, and TP from June 2002 to June 2003 under low flow conditions, for major sub-basins (based on discharge) in the Mona Lake Watershed. Site abbreviations as in Table 3.

Sub-basin	Variable				
	Area (km^2) (%)	Nitrate kg km^{-2} yr^{-1} (%)	TKN kg km^{-2} yr^{-1} (%)	SRP kg km^{-2} yr^{-1} (%)	TP kg km^{-2} yr^{-1}
ST1	11.7	221.9 (18.2)	113.2 (22.5)	2.0 (16.2)	10.0 (19.3)
ST6	11.0	343.2 (28.1)	78.3 (15.6)	2.3 (18.7)	7.2 (13.9)
Black Creek	126.0	212.5 (17.4)	165.6 (33.0)	3.9 (31.9)	17.6 (34.0)
Little Black Creek	18.0	250.0 (20.5)	82.8 (16.5)	2.4 (19.3)	11.4 (22.0)
Other inflows (NT1–3, ND1–3, ST 2,4,5)*	15.2	192.4 (15.8)	62.7 (12.5)	1.7 (13.9)	5.6 (10.8)

* ST5 had detectable flow on only 1 date (March 17, 2003).

Table 6. Median (and range) of selected variables measured at 4 sampling locations in Mona Lake. Site numbers are ordered from western end (Site 1) to eastern end (Site 4). Number of observations for all parameters is 13 unless otherwise noted.

Site	Temp. (°C)	DO (mg/L)	Spec. cond. (μS/cm)	Chl-a (μg/L)	Nitrate (mg/L)	Ammonium (mg/L)	SRP (μg/L)	TP (μg/L)	TN : TP	Secchi Disk (m)
1–top	20.2 (0.1–25.4)	9.3 (5.8–13.7)	411 (387–509)	11 (3–45)	0.04 (0.01–0.78)	0.08 (0.01–0.35)	5 (5–10)	60 (4–90)	29.2	0.71 (0.4–1.2)
1–bottom	15.5 (1.5–21.5)	0.4 (0.0–12.1)	435 (359–585)	5 (0–41)	0.07 (0.01–0.67)	0.38 (0.05–3.01)	10 (5–220)	70 (40–420)	22.3	
2–top	20.5 (0.6–26.0)	9.6 (5.8–15.3)	421 (398–510)	14 (3–46)	0.04 (0.01–0.82)	0.09 (0.01–0.35)	5 (5–40)	60 (30–110)	28.1	0.56 (0.25–0.95)
2–bottom	16.4 (1.4–21.6)	0.9 (0.0–13.8)	447 (424–600)	5 (0–57)	0.05 (0.01–0.66)	0.39 (0.01–2.56)	10 (5–300)	110 (30–400)	20.1	
3–top	21.4 (0.0–27.4)	9.5 (6.7–14.0)	425 (393–500)	14 (7–41)	0.04 (0.01–0.93)	0.05 (0.01–0.33)	5 (5–20)	70 (30–110)	31.6	0.70 (0.35–1.2)
3–bottom	17.4 (1.5–23.0)	1.9 (0.0–13.0)	447 (427–644)	10 (0–54)	0.06 (0.01–0.72)	0.32 (0.01–1.52)	5 (5–220)	100 (30–380)	23.2	
4–top*	20.1 (0.1–26.9)	10.7 (5.8–13.3)	422 (391–485)	20 (2–45)	0.03 (0.01–0.87)	0.08 (0.04–0.35)	5 (5–20)	80 (30–130)	31.3	0.54 (0.4–1.0)
4–bottom	17.6 (1.5–24.2)	6.3 (1.5–24.2)	433 (415–501)	20 (0–74)	0.04 (0.01–0.78)	0.18 (0.02–0.41)	5 (5–100)	85 (30–230)	23.9	

* Missing data on August 2, 2002.

Nutrient yields were calculated based on the area of five sub-basins, as derived from GIS data. Under low flow conditions, nutrient yield from the Black Creek sub-basin was substantial, but its influence was much less compared to load (Table 5). For example, the Black Creek sub-basin contributed only 17 % of the total nitrate yield compared to 66 % of the nitrate load, and its relative contribution of TKN, SRP, and TP declined from approximately 80 % of the loads (Table 4) to approximately 30 % of their respective yields (Table 5). In contrast, the ST1, ST6, and Little Black Creek sub-basins had relatively high nutrient yields compared to their loads (Table 4).

Mona Lake: water quality

Median surface temperatures in Mona Lake were similar at all sites but median bottom temperatures became progressively warmer from west to east (Table 6), presumably due to shallower depths at the lake's eastern end and a greater influence of colder Lake Michigan water advected from the west. Surface DO concentrations were usually near saturation; bottom DO levels were much lower than at the surface and declined below 1 mg/L at all sites in mid-summer (data not shown). Median specific conductance values were similar at all sites, with specific conductance typically 10–25 µS/cm greater at the bottom than at the surface (Table 6). Chlorophyll-*a* concentrations ranged from 0 to 74 µg/L, with median surface values much higher than median bottom values at the deeper sites (Table 6). The highest chl-*a* concentrations were found in spring and fall (data not shown); mid-summer concentrations likely declined because of algicide treatments on Mona Lake. Median Secchi disk depths generally showed an inverse relationship with chlorophyll *a* concentrations (Table 6), with the western end (Site 1) having low chlorophyll and a high Secchi depth, and the eastern end (Site 4) showing the reverse trend (Table 6). Median Secchi depths were < 1 m at all sites.

Median nitrate values ranged from 0.03 to 0.07 mg/L (Table 6) and were very similar among sites and depths, but with distinctly higher concentrations in the winter-spring (0.60 to 0.93 mg/L) compared to other seasons. In contrast, ammonium concentrations were substantially higher at the bottom compared to the surface at all sites (Table 6). Although elevated bottom concentrations were measured throughout the year, the difference between bottom and surface ammonium levels was much greater during the summer (1.76 mg/L on June 9, 2002) than the winter (0.02 mg/L on February 17, 2003). SRP and TP concentrations followed the same general pattern as ammonium, with substantially greater concentrations at the bottom than at the surface (compare ranges of values in Table 6), and with the bottom concentrations highest during the summer months (data not shown). There was a strong inverse relationship between DO and phosphorus concentration (Fig. 4), suggesting that anoxic re-

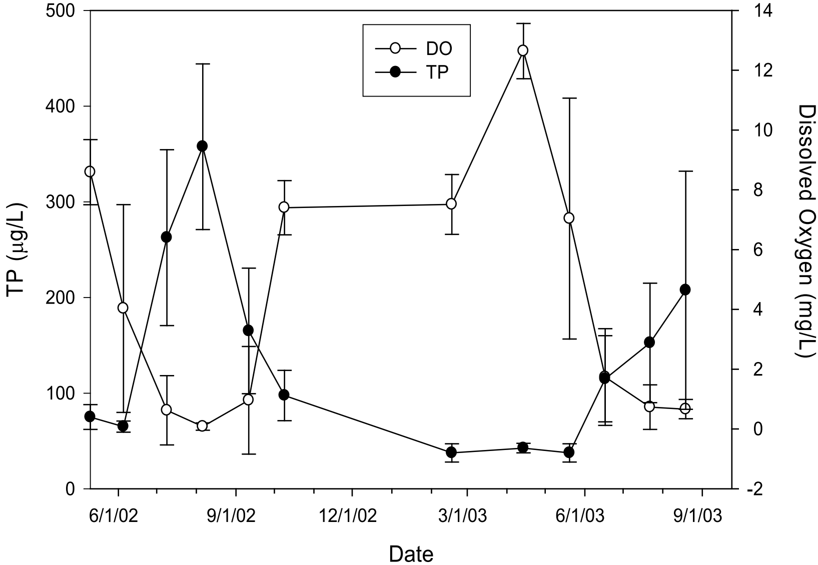

Fig. 4. Seasonal variation in mean (± 1SD) bottom concentrations of total phosphorus (μg/L) and dissolved oxygen (mg/L) in Mona Lake from all four sampling sites. Note the inverse relationship between phosphorus and oxygen.

lease of iron-bound phosphorus was occurring in Mona Lake (cf. Mortimer 1941, 1942). Mean molar TN:TP in the lake was ~30 at the surface and ~22 at the bottom (Table 6).

Bioassays

Mean chl-*a* concentrations were significantly greater ($P < 0.05$) in the N and N + P than the control (C) and P treatments during the summer bioassay experiment (Fig. 5). However, the mean chl-*a* values for each treatment were not statistically different from one another during fall (Fig. 5).

Little Black Creek: sediment analysis

With the exceptions of the reference site (Evanston) and the wetland area at LBC mouth (Fig. 2), all locations exceeded the probable effect concentrations (PEC) for cadmium and lead, and most locations exceeded the PEC for chromium (Table 7). PEC values signify the level at which there is a 75 % probability of adverse ecological effects (MacDonald et al. 2000). The downstream distribution of metals was generally similar, with elevated concentrations at upstream Sherman, and a secondary spike at the Mona View wetland site before declining again before reaching Mona Lake (Table 7). The highest

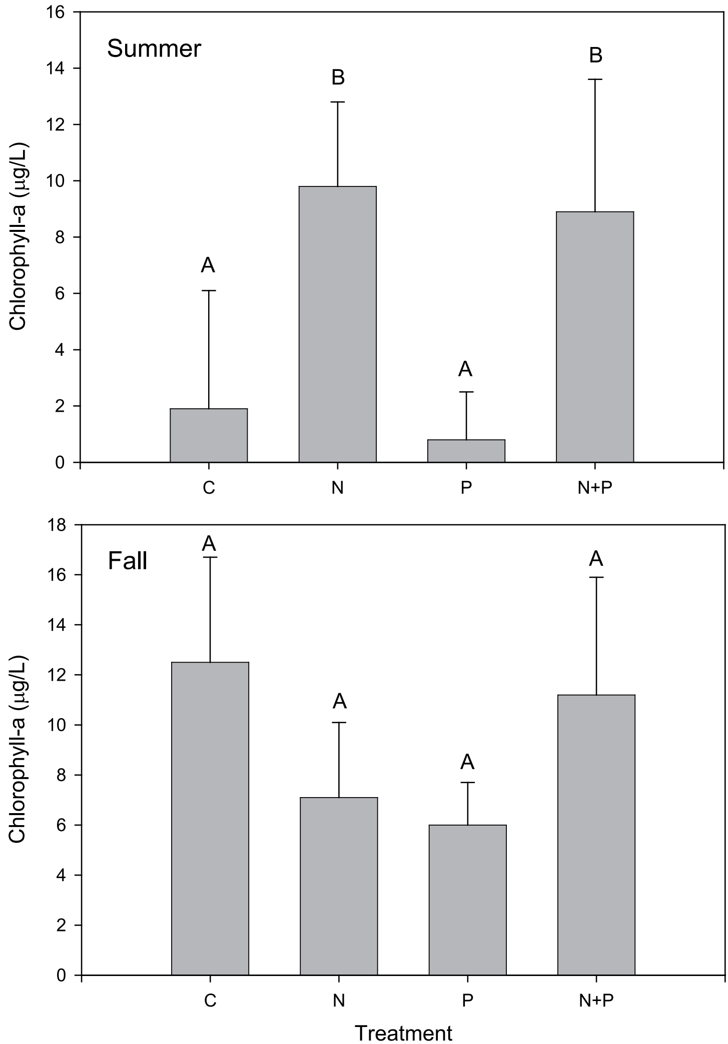

Fig. 5. Top panel: net change in chlorophyll-*a* concentrations (µg/L) from days 0 to 4 during summer, 2003. Bottom panel: net change in chlorophyll-*a* concentrations (µg/L) from days 0 to 4 during fall, 2003. Error bars = 1 SD. Treatments: C = control; N = nitrogen addition; P = phosphorus addition; N + P = nitrogen plus phosphorus addition. Different letters above bars indicate significant differences among treatments ($P < 0.05$).

concentration of Cd occurred at the upstream Sherman site, adjacent to the plating Superfund site. Cr concentrations, however, were most elevated at the wetland site, suggesting a source downstream of the plating site.

Table 7. Concentrations of metals and organic chemicals in sediments at 7 sites in Little Black Creek. Sites are arranged from upstream (Evanston) to downstream (LBC mouth). Asterisks signify those values that exceed the probable effect concentration (PEC) for each contaminant (given in bottom row), which is the level at which there is a 75 % probability of adverse ecological effects (MacDonald et al. 2000). All results are in units of mg/kg (dry weight basis).

Site	Metals			Organics		
	Cadmium	Chromium	Lead	Total PAH	Benzo(a)-pyrene	PCBs
Evanston (background)	<0.1	5	11	0.2	0	0.3
Sherman (upstream)	940*	180*	220*	63*	4.6*	1.9
Sherman (downstream)	150*	140*	230*	83*	5.6*	0.8
Summit	39*	57	170*	43*	3.0*	8.9*
Mona View	67*	400*	370*	56*	4.4*	6.3*
Seaway	11*	160*	410*	29*	2.0*	1.0
LBC mouth	2.6	26	50	3	0.3	0.5
PEC	4.98	111	128	22	1.4	2.0

With the exceptions of Evanston and LBC mouth, all locations exceeded the PEC values for total PAH compounds and benzo(a)pyrene (Table 7). Total PAH compounds and benzo(a)pyrene were highest in concentration at both Sherman stations, declined with distance downstream, and then increased at the Mona View wetland. This wetland may function as a sink for organic chemicals. PCB concentrations followed a different pattern as the highest concentrations were observed at Summit and the Mona View wetlands (Table 7). The Mona View wetland again appears to be a sink for contaminants as concentrations show only a small decrease from the upstream location at Summit.

Little Black Creek: macroinvertebrates

The two wetland sites (Mona View and LBC mouth) were removed from the CA of macroinvertebrate community structure because their inclusion forced all of the remaining sites into one overlapping group, revealing differences only between wetland and lotic sites. The resulting CA, involving only the lotic sites, explained 47 % of the variation in the dataset in the first dimension (Fig. 6). This dimension represented a split in habitat type between vegetation to the left (Summit) and sand to the right. The Summit site had the most unique community, which accounted for its separation from the other three sites along the x-axis. The second dimension explained an additional 29 % of the variation but because this dimension is often a quadratic distortion of the

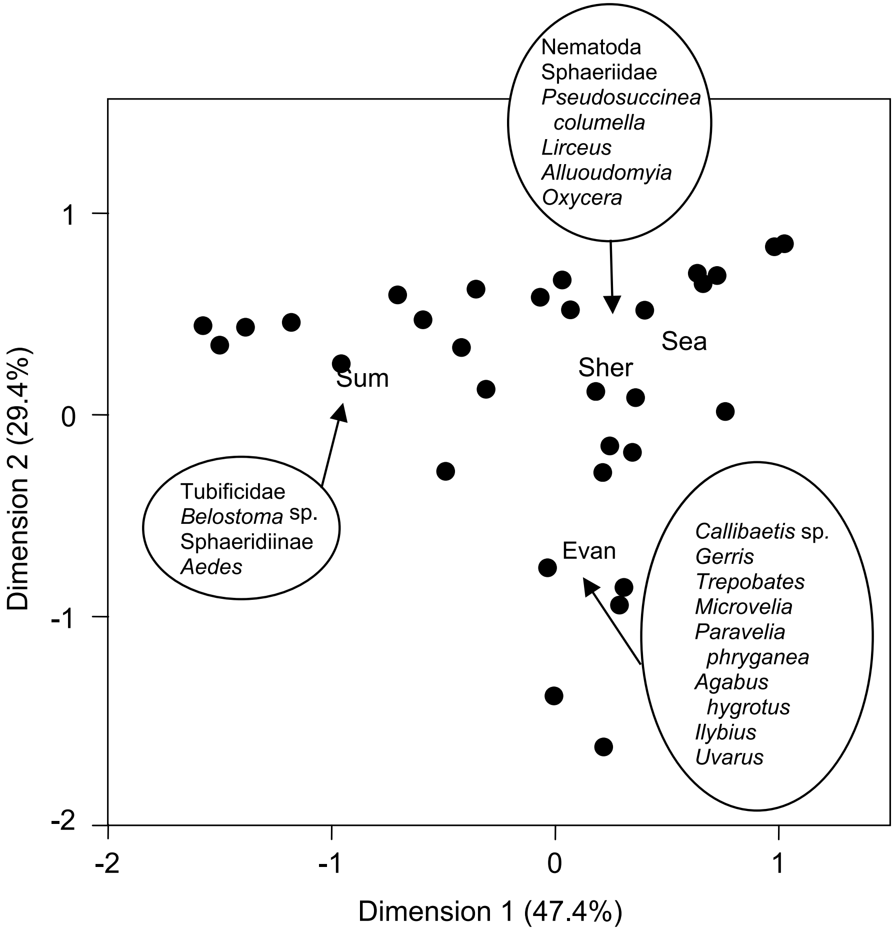

Fig. 6. Correspondence analysis of macroinvertebrate taxa collected from Little Black Creek in 2002. Sites are presented in larger font size (Sum = Summit; Sher = Sherman; Sea = Seaway; Evan = Evanston); arrows attached to ovals identify locations where the clusters of overlapping taxa are located. The overlapping taxa are expanded in corresponding ovals. Dimension 1 is related to habitat impairment; dimension 2 is related to water quality impairment.

first dimension in CA, this percentage is likely inflated. The second dimension separated sites based on differences in water quality, with the reference site (Evanston) tending to have a more insect-dominated community than the other sites. The invertebrate communities at the other three sites (top) suggested lower water quality than Evanston, and were characterized by Nematoda, Turbellaria, Hirudinea, Naididae, Tubificidae, and Sphaeriidae.

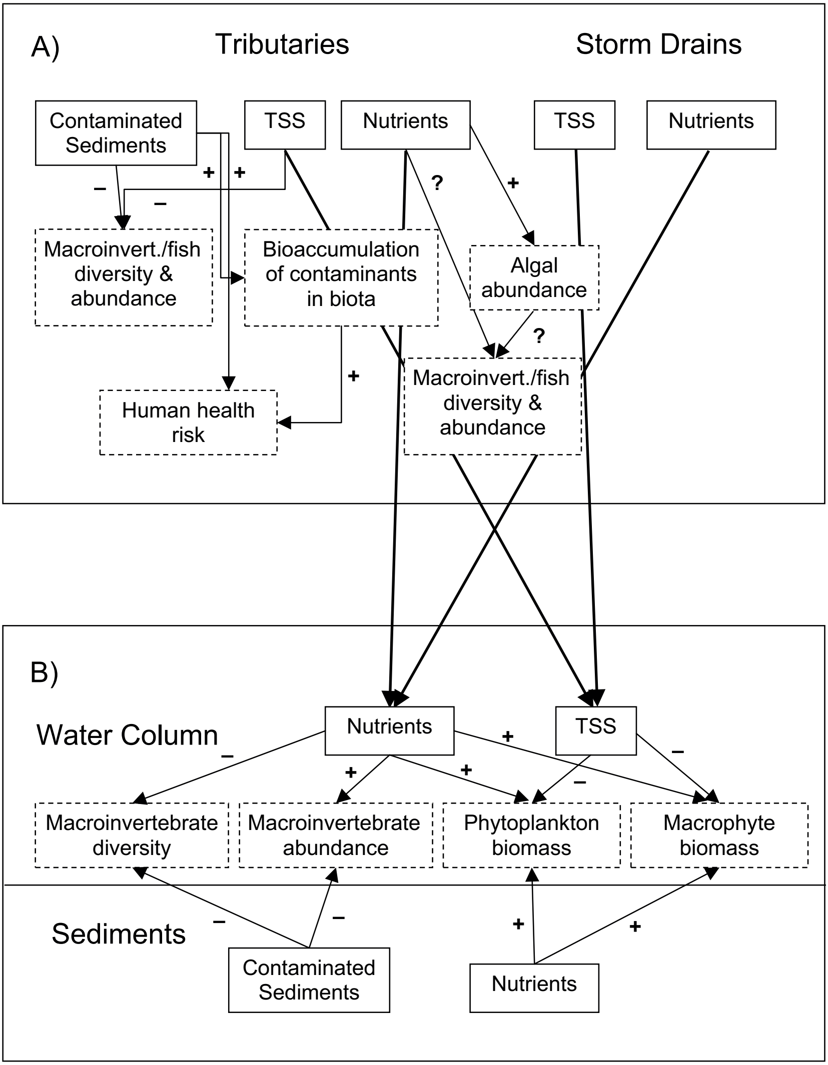

Fig. 7. Conceptual model showing the relationship of the major stressors in the Mona Lake watershed, partitioned into inflow **(A)** and lake **(B)** subsystems. Stressors are identified in solid boxes and ecological variables by dashed boxes. Plus signs indicate the ecological variable increases due to the stressor; negative signs indicate the variable decreases due to the stressor. Inputs from inflows to the lake (i.e., top panel to bottom panel) are shown in bold arrows.

Discussion

Watersheds throughout the world are facing increasing environmental stress associated with population growth, changing land use practices, and increasing amounts of nonpoint source pollution (Allan 2004). Understanding these impacts, and identifying appropriate recommendations to protect or restore impaired systems, is a critical need. For urbanized watersheds, where the stressors are multifaceted and often interacting, approaches that address problems in a comprehensive fashion improve the probability that restoration recommendations will focus on the underlying causes, and not just the symptoms of impairment.

The comprehensive approach used in the current study can serve as a model for other small, urbanized watersheds. These systems can be conceptually modeled by subdividing the watershed into the inflow (Fig. 7 a) and lake (Fig. 7 b) compartments, with the lake being further subdivided into water column and sediment. Below, the key stressors in each of these spatial elements are addressed, using the Mona Lake Watershed as a model.

The tributaries serve as sources of nonpoint source pollutants, including contaminated sediments, suspended solids, and nutrients (Fig. 7 a). Contaminated sediments result in negative impacts to higher trophic level diversity and abundance (Canfield et al. 1996), consistent with the findings in Little Black Creek. However, the design of our study did not allow us to separate the relative importance of contaminants, nutrients, and habitat impairment. The Little Black Creek macroinvertebrate community had relatively low numbers overall and was dominated by pollution-tolerant taxa. The correspondence analysis suggested that both habitat impairment and water quality impairment were important in the system. Sand was the dominant substrate type in Little Black Creek, and the separation of the Summit site likely reflects a higher abundance of aquatic macrophytes than the other sites. Understanding the underlying mechanisms behind impairment is critical to the process of restoring watershed health; linking controlled experiments to these observational studies is an important step in this process.

Highly contaminated sediments in areas with public access pose a potentially significant human health risk from either direct contact or ingestion of contaminated fish (Fig. 7 a). Recent evidence indicating that cadmium can be biomagnified in freshwater systems (Croteau et al. 2005) further illustrates the potential hazards in Little Black Creek. Most of the sediment samples had contaminant concentrations that exceeded the PECs for protection of aquatic life (MacDonald et al. 2000). In addition, concentrations of lead, benzo(a)-pyrene, and PCBs were at levels that exceed human health criteria for long-term direct contact (ATSDR 1999, 2000, MDEQ 2000 b). These data suggest the contaminants are being transported downstream within Little Black Creek and accumulating in wetlands. In systems where the exposure pathways are dynamic, the human and ecological health risks require further analysis because of the uncertainties associated with contaminant fate and transport.

Total suspended solids (TSS) in the tributaries also can negatively impact fish and invertebrates (Waters 1995, Newcombe & Jensen 1996; Fig. 7 a), whereas elevated nutrients can positively affect algal biomass (Dodds et al. 1997; Fig. 7 a). This, in turn, can have either positive or negative impacts on higher trophic levels depending upon the type and amount of algae that develop (Steinman 1996). Controlled experiments are needed to determine the impacts of TSS and nutrients in these tributaries.

Unlike tributaries, where in-stream uptake of nutrients is possible, stormwater drains provide a direct conduit to receiving water bodies. These direct discharges to the lake decrease or eliminate the detention of pollutant-laden runoff, thereby eliminating the opportunity for soils and biota to assimilate contaminants, and have been cited as one of the most important negative impacts associated with urbanization (Paul & Meyer 2001). As a consequence, nutrient impairment in urban watersheds is strongly related not only to percent impervious surface, but also to the degree of drain connectedness (Hatt et al. 2004). In the Mona Lake watershed, the highest concentrations of nitrogen species flowing into Mona Lake came from the stormwater drains (Fig. 7 a, b). Stormwater drains also contained the highest specific conductance and chloride readings, with strong peaks in winter. For example, in February 2003, Cl concentrations in ND2 and ND3 were 270 and 1300 mg/L, respectively, compared to a mean of 100 mg/L in all other inflows [$n = 10$; $SD = 63$]. In July 2003, Cl concentrations in ND2 and ND3 were 219 and 163 mg/L, respectively, compared to a mean of 73 mg/L in all other inflows [$n = 10$; $SD = 31$]). These elevated winter concentrations are likely a function of runoff containing high amounts of road salt. Indeed, increased salinization of fresh water is a growing ecological and public health problem (Kaushal et al. 2005). Fecal coliforms were occasionally high in storm drain discharge, but sustained differences among inflows were not detected. Although our lake sampling was not of sufficient spatial resolution to determine if impairments occurred where storm drains emptied into Mona Lake, it would be of interest to determine if biotic signals could be detected at these inputs (sensu hot spots; McClain et al. 2003). Storm drains serve as major conduits of contaminants during storm events, so our sampling during low flow conditions likely underestimated their relative importance as sources of nutrient loads in the watershed.

Nutrient loading data from individual sub-basins are critical for identifying the responsible source areas (Fig. 7 a, b). The Black Creek sub-basin was responsible for most of the nutrients entering Mona Lake under low flow conditions, which is a function of this sub-basin's agricultural land use and its relative size in the watershed. When normalized per unit area, the Black Creek sub-basin still accounted for most of the nutrients entering Mona Lake, although its relative importance declined. Based on these findings, efforts to reduce nutrient mass entering Mona Lake should be directed toward the Black Creek sub-basin. We suspect that inclusion of peak flows in our load analysis would result in larger load contributions from the Black Creek sub-basin, especially for phosphorus, because of its greater agricultural land use (cf. Jordan et al. 1997). Nutrient yields from the small sub-basins adjacent to Mona Lake were high relative to their respective loads. Although urbanizing watersheds have been shown to retain large amounts of nitrogen (Groffman et al. 2004), the combination of short flowpaths and elevated fertilizer use associated with the high density of residences near the lake may have limited the opportunity for nutrient retention in this sub-basin (Meyer et al. 2005).

In the lake water column, increased nutrients from the tributaries and storm drains result in positive growth for autotrophs, but TSS can result in decreased autotrophic biomass because of increased light extinction (Fig. 7b). Excessive phosphorus entering Mona Lake has been identified as a long-standing problem. The diversion of wastewater in the 1970 s to the Muskegon Wastewater Management System reduced nutrient loads to Mona Lake (Freedman et al. 1979), which is reflected in the substantial reduction in nutrients and algal biomass over the past 30 years (Fig. 8). Despite these reductions, both phosphorus and nitrogen concentrations in the lake still are at levels that can cause impairment (Dodds 2002). This may be a result of increased nonpoint source inputs over time or internal nutrient loading from the sediments (see below).

In the lake sediment compartment (Fig. 7b), biotic impairments may result from the transport of contaminants out of Little Black Creek, leading to toxicity of the lake fauna, as well as nutrient release during anaerobic periods, which can stimulate algal blooms (Fig. 7b; Boström et al. 1982). Our observational data suggest that internal loading may be a significant source of phosphorus to Mona Lake (Fig. 4). It is likely that a significant reservoir of phosphorus has accumulated in the Mona Lake sediments and that internal loading is influencing the nutrient dynamics of the lake, especially during anaerobic conditions (Welch & Cooke 1995, Søndergaard et al. 2003, Steinman et al. 2004).

TN:TP ratios can be used to assess potential nutrient limitation, and have certain advantages over inorganic nutrient ratios (cf. Dodds 2003). The mean ratios in the surface and bottom waters of Mona Lake averaged near 30 : 1 and 22 : 1, respectively, suggesting nitrogen was in plentiful supply relative to phosphorus (Smith 1982). Thus, it was surprising that the bioassay results suggested N-limitation for biomass in the summer. The fall data indicated that neither N nor P was limiting algal growth; the shallow Secchi disk depths suggest that light may be the limiting resource in Mona Lake. Resource limitation apparently varies through the year in Mona Lake, which may be because of seasonally variable loads from the inflows, variable nutrient demands associated with a changing algal community structure, or differential grazing pressure from zooplankton. Algal bioassays conducted in 1972, using a cultured alga, found the strongest growth response in N-amended media (U.S. EPA 1975). However, the TN:TP ratios in Mona Lake at that time averaged about

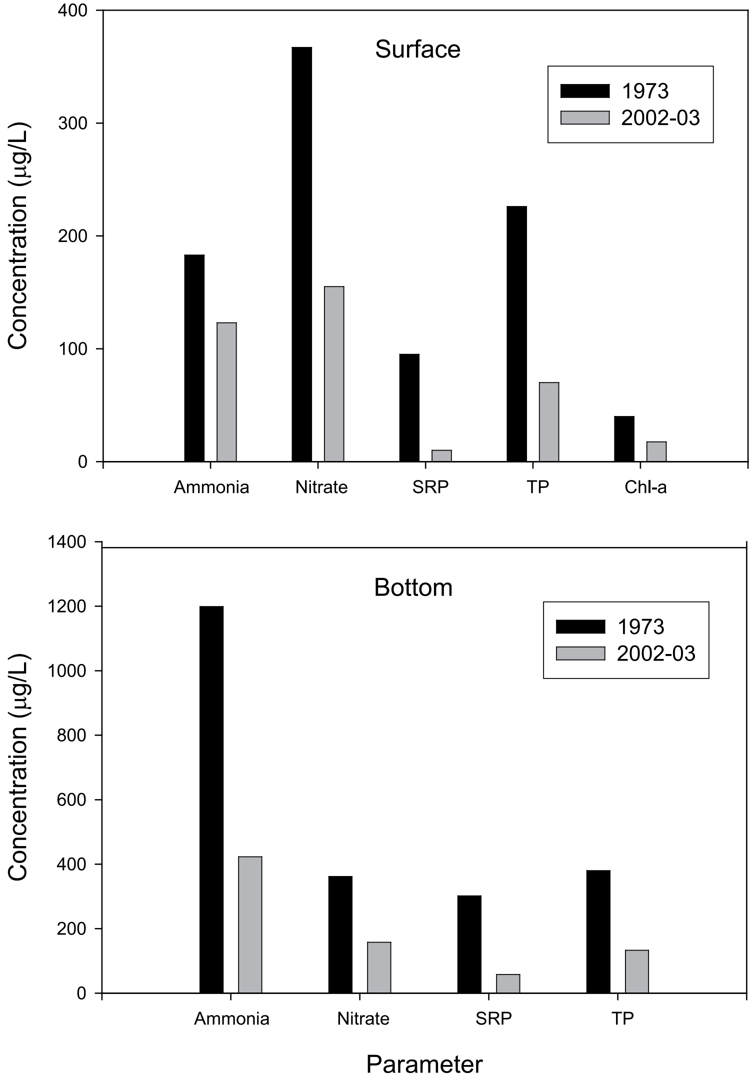

Fig. 8. Mean values of select limnological variables from the surface and bottom waters of Mona Lake. Data are averages from various stations and seasons within 1973 and 2002–2003, although samples collected after known applications of algicide were excluded from the analysis.

4 : 1, so N-limitation would be expected, compared to the 30 : 1 ratio measured in the current study.

Throughout west Michigan and the Midwestern U.S., agricultural land use is being converted to more urbanized land uses (Pijanowski et al. 2002, Radeloff et al. 2005), and changes in the Mona Lake watershed are consistent with that pattern. A continued decline in cropland should result in less nutrient inputs over time (cf. CARPENTER et al. 1998, SHARPLEY & TUNNEY 2000). However, increases in impervious surface and the degree of drainage connections will continue to provide significant inputs of nonpoint source pollutants in urbanized watersheds (e. g. HATT et al. 2004). Ecological impairments are likely when percent impervious cover of a watershed exceeds 10–15 % (CARLSON 2004), which is the case in the lower Mona Lake watershed.

The comprehensive approach used in this study provides a model for future work in urbanizing watersheds. Others have recognized that coordinated, watershed-scale efforts have the best chance of success in restoring impaired systems (cf. BOHN & KERSHNER 2002, PALMER et al. 2005, WALSH et al. 2005). We have provided an approach and conceptual model, focusing specifically on land use, nutrient loadings and source areas, and toxic contaminants, which allowed us to pinpoint the most problematic regions in the watershed. This, in turn, helped identify those areas where mitigation and restoration efforts will be most cost-effective. Based on our analyses, we have generated a set of recommendations. Although these are specific to the findings in the Mona Lake watershed, they are generally applicable to other watersheds facing similar problems:

1) Land use change: determine the sub-basins or regions where land use change is most dramatic to identify potential source areas for stressors. Work with elected officials, planning and zoning officials, and homeowners to educate them about low impact development; identify funding opportunities and strategies for the protection and/or acquisition of green infrastructure.

2) External nutrient loading: nutrient load reduction strategies should focus on the sub-basin which is contributing the most nutrient mass to the impaired water body. This would target the Black Creek sub-basin in the Mona Lake watershed. Specific strategies include implementation of agricultural BMPs, in-stream restoration activities, stabilization of erodable banks, and assessing the feasibility of converting a portion of the creek's flowpath to a flow-through marsh (cf. MITSCH & WANG 2000).

3) Internal nutrient loading: determine the importance of internal nutrient loading relative to external nutrient loading (e. g. STEINMAN et al. 2004) and if found to be significant, assess possible mitigation strategies, such as alum application or dredging.

4) Suspended solids: implement BMPs through the existing regulatory and non-regulatory pathways. Regulatory mechanisms may include Phase II Stormwater Management permits (U.S. EPA 2004), whereas non-regulatory mechanisms may involve vegetated buffer strips or no-tillage management for agricultural inputs.

5) Contaminated sediments: conduct a systematic survey of the contaminated area to identify the boundaries of contaminated areas, perform appropri-

ate toxicity tests, and remove contaminated sediments that present human or ecological health risks.

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