Aquatic Macrophyte Diversity and Habitat Characterization of the Cuyahoga River Watershed in Northeastern Ohio[1]

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Abstract. We surveyed aquatic macrophyte diversity and abundance at 20 sites along the main channel of the Cuyahoga River and its tributaries. These sites included 12 sites in the watershed at which an Index of Biological Integrity (IBI) for fish communities deviated significantly from a value predicted by a statistical model of landscape urbanization and stream habitat quality. These sites were classified as Best of the Best, Worst of the Best, Best of the Worst and Worst of the Worst among 164 sites within the Cuyahoga basin. In order to characterize a site, we collected data on the physical features of the stream and quantified the species abundance of aquatic macrophytes in a 100 m transect. Within each transect, measurements of stream width, bankfull width, stream depth, bankfull depth, and canopy cover were recorded every 10 m. Nitrate, phosphate, and ammonia content of water samples were also assayed. The quality of stream habitat for each site was quantified using the Qualitative Habitat Evaluation Index (QHEI). Strong significant correlations between the measure of stream depth and stream width as well as canopy coverage and bankfull width were observed. A weak significant correlation was found between IBI and QHEI scores. Additional analysis showed that water chemistry did not influence QHEI or IBI scores. An analysis of variance indicated that the IBI scores significantly differed among site types. Macrophytes were discovered at seven of the 20 sites with an overall richness of 11 species among all sites. The most common aquatic macrophytes were: *Elodea canadensis* L., *Sparganium americanum* Nutt., and *Sagittaria latifolia* Willd. Results demonstrate that physical stream characteristics are strong indicators of fish population integrity, but are not necessarily indicative of aquatic macrophyte assemblages. Storms severely impacted many streams during the survey, possibly altering macrophyte assemblages. Further surveys should be undertaken at additional sites within the Cuyahoga River watershed for a comprehensive assessment of aquatic macrophytes.

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INTRODUCTION

Urbanization is among the most significant threats to stream and river systems of the United States and elsewhere (House and others 1993; Paul and Meyer 2001). Currently, over 130,000 km of streams and rivers are impaired in some way by urbanization in the United States (US EPA 2000; Paul and Meyer 2001) and the frequency and intensity of these impacts are likely to increase in the future. Currently, over 77% of the population of the United States lives in cities, and that number is expected to increase to nearly 85% within the next thirty years (US Census Bureau 2001). Within the decade, half or more of the world's population is expected to live in cities (United Nations 2002; Cohen 2003).

Small, headwater streams (for example, <52 km^2 basin area) are the components of stream/river networks that are most threatened by urbanization. In general, 80% or more of the length of stream/river networks are composed of streams within this size range (Horton 1945; Naiman 1983). As primary loci for sediment retention and organic nutrient processing, small stream segments have major significance for overall ecological function of the system, as well as being important determinants of the quality of receiving waters (Peterson and others 2001). Further, small streams are important as reproductive habitats for stream biota and as reservoirs for biotic recolonization following disturbance events. Yet, these segments also interdigitate most extensively within the surrounding landscape and are, therefore, strongly influenced by urbanization. Among the impacts of urbanization on small streams are the loss of stream length and complexity of the dendritic stream network through filling and covering, reduction in length of natural channels and the increase in length of artificial channels through culverting and channelization, hydrological impacts due to alteration of soils and amount of impervious surfaces in surrounding landscapes, channel incision, increased flow extremes, increased temperature extremes, and increased and varied pollutant loads (Paul and Meyer 2001). Hence, measures of the ecological value and function of small streams are an important focus of ecological restoration research in urban settings. However, studies of small urban streams remain relatively few in general and understanding of their ecology is especially limited (Paul and Meyer 2001).

Among the least understood and least studied components of urban stream biota are aquatic macrophytes. We consider this unfortunate, since changes in macrophyte communities may be especially indicative of major categories of urban stress, for example, nutrient run-off, hydrologic regime, and invasion by exotic species (Haury 1996; King and Buckney 2000; Suren 2000; US EPA 2003a). Aquatic macrophytes provide not only

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important structural supports in stream and river habitats (Small and others 1996), but also provide primary food production, nutrients, and habitat for a wide-range of macro and microorganisms living in and around lotic sites. Thus, the health and structure of macrophyte communities are likely to be important determinants of quality of habitat available for other stream organisms (Gregg and Rose 1982). In addition, macrophytes also have potential use in bioremediation and biological monitoring and have been shown to respond to and accumulate heavy metals and other contaminants (Kapitonova 2002; Mal and others 2002; Samecka-Cymerman and Kempers 2002). Indeed, the US EPA (2002) considers macrophytes to be "...excellent indicators of watershed health" due to their remarkable response to environmental factors and ease of sampling.

Nevertheless, macrophytes are seldom used in the United States for water quality monitoring or biological assessment. Aquatic macrophytes have been surveyed within North America, although most studies have been focused on pond and lake macrophyte communities. In Europe, however, indices based upon macrophytes have been created to evaluate both lotic and lentic aquatic environments (Newbold and Holmes 1987; Thiebaut and others 2002). Hence, one of the long-term goals of our research study is to explore the possibility of using macrophytes as indicators of stream health and for monitoring biological integrity in urbanized streams of North America. Here, we report on a preliminary survey of macrophytes in tributaries of the Cuyahoga River, which drains a heavily urbanized and suburbanized landscape in northeastern Ohio, USA.

The Cuyahoga River basin may be an especially appropriate location for studies developing a macrophyte indicator. Aquatic macrophytes in the Cuyahoga River watershed were documented 35 years ago (Simpson and others 1969). This earlier study provides a basis for comparison to the current study and an opportunity to compare macrophyte communities before and after enactment of the Clean Water Act, which may provide insight into the effectiveness of three decades of remediation activities (US EPA 2003b). However, the previous survey was of a qualitative nature and it is imperative that we collect quantitative information on macrophyte diversity in order to develop useful bioindicators (Dale and Beyeler 2001). In addition, site selection for this study was guided by a rich data set collected by the Ohio Environmental Protection Agency regarding water quality and biological integrity of streams in the region. Walton and others (in review) have performed an analysis of these data to evaluate the value of land use/land cover features, population and housing density, and in-stream habitat quality as predictors of an index of biological integrity (IBI) based on fish communities. The fish-based IBI is a standard tool used by the Ohio EPA and other agencies to assess aquatic life use impairment (Lyons and others 2001; Weigel and others 2002).

Hence, an additional objective of this project was to determine the extent to which macrophyte community differences among sites reflect differences identified through the more standard approach based on IBI. Specifically, we selected several sites that represent the extremes of IBI scores for NE Ohio streams and have quantified macrophyte communities within these sites. Our general hypothesis is that macrophyte diversity and abundance will differ among sites in accordance with their IBI scores. We view support of this hypothesis as a basic criterion for proceeding with a more detailed survey and future development of a macrophyte index of ecological integrity.

MATERIALS AND METHODS

Study sites were located in the Cuyahoga River watershed in Northeast Ohio. The Cuyahoga River watershed drains into Lake Erie, includes thirty-seven named tributaries, and occupies 2500 km^2. In its upper reaches, the Cuyahoga River consists of an East and West Branch, which eventually meet to form the main channel and which subsequently empties into Lake Erie (Cuyahoga County Board of Health 2003; Fig. 1).

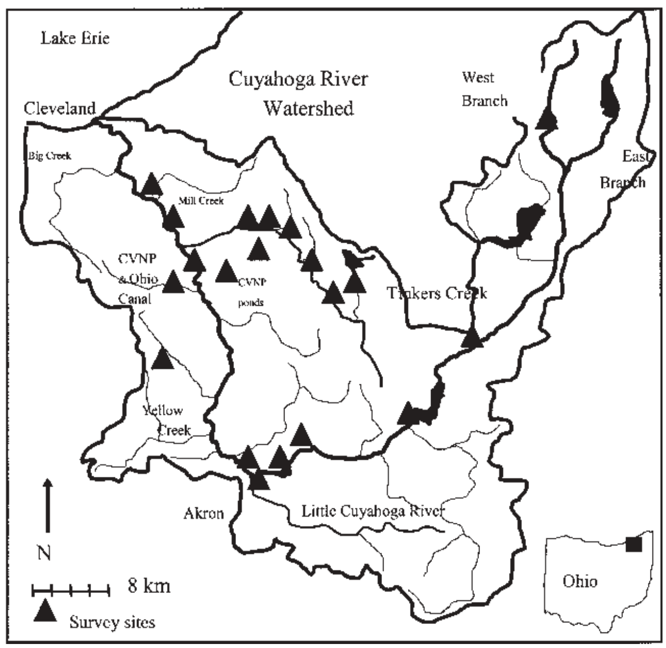

FIGURE 1. Map of the Cuyahoga River watershed, indicating major tributaries and locations of 20 survey sites.

This region is heavily urbanized with two large cities, Cleveland and Akron, within the basin. Urbanized and agricultural areas have been causing many environmental problems and surround many tributaries of the Cuyahoga River and its main channel, resulting in increased impervious surface cover and runoff carrying a variety of pollutants. The occurrence of metals, alkylphenol and alkylphenol ethoxylates (APEs), Polychlorobiphenyls (PCBs), and polynuclear aromatic hydrocarbons (PAHs), has been documented in different fish species (carp, brown bullhead), water, and sediment from the Cuyahoga River (Baumann and others 1991; Smith and others 1994; Lesko and others 1996; Lin and others 2001; Rice and others 2003; Yang and others 2003).

Site Selection

One objective of this research is to develop indicators that are sensitive to urbanization impacts, but also may be sensitive to variability not explained by standard measures of ecological integrity. Hence, the sites examined in this study were locations that have biological integrity values, as measured by a fish-based IBI, that are significantly better or worse than that predicted by the level of urbanization within the catchment. Site selected in this way relied on an analysis by Walton and others (in review) that related IBI data available from the Ohio EPA to measures of land use/land cover (Landsat Thematic Mapper), human population demography and housing density (US Census Bureau 2001), and habitat quality in streams (Ohio EPA data set for Qualitative Habitat Evaluation Index). Predictor variables describing land use/land cover, demographic, habitat quality, as well as basin area and distance from site from stream terminus were used to identify the best multivariate predictor of IBI for 164 sites within the Cuyahoga River basin. This multivariate descriptor of overall urbanization is termed the *urbanization gradient* here. Residuals of the linear regression of IBI on the urbanization gradient were then used to identify 20 sites that exhibited IBI scores that were either exceptionally high or exceptionally low (±2 standard deviations) relative to that predicted by the urbanization gradient (Fig. 2). Sites were then divided into four categories (ellipses in Fig. 2) based upon the following criteria: Best of the Best sites (BOB), IBI >2 standard deviations above predicted and urbanization gradient score greater than mean (1.47 on $\log_{10}$-scale); Worst of the Best sites (WOB), IBI <2 standard deviations below predicted and urbanization gradient score >mean (1.47 on $\log_{10}$-scale); Best of the Worst (BOW), IBI >2 standard deviations above predicted and urbanization gradient score < (1.47); and Worst of the Worst (WOW), IBI <2 standard deviations below predicted and urbanization gradient score < mean (1.47 on $\log_{10}$-scale). Three sites were chosen at random from each of the four site categories. The additional 8 sites were chosen randomly for macrophyte diversity assessment to represent the stretch of the main channel of the Cuyahoga River.

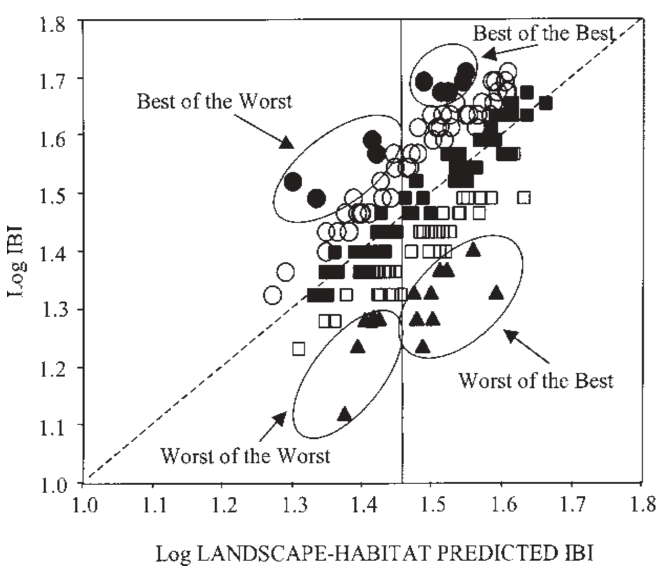

FIGURE 2. Plot of log IBI vs log landscape-habitat predicted IBI identifying four different site types (see Materials and Methods section for details).

Macrophyte Quantification

At each site, aquatic macrophytes were quantified for the number of species, number shoots for each species along a 100 m transect through the center of the stream channel. Also, percent cover of each species within a 10 m long rectangular subplot was quantified (Small and others 1996; Scott and others 2002). The width of the subplot was equal to the width of the stream at each end. To sample submerged aquatic vegetation in deep and turbid water, a benthic grab sampler was used. We quantified only in-stream macrophytes and excluded those that occupied banks and were partially or fully submerged following storm events. Areas of the streams covered by water 85% of the time or greater were considered in-stream (Thiebaut and others 2002). In accessing the stream and river sites, a wading technique for sampling shallow bodies of water was used (Capers 2000). Every 10 m within each transect, physical stream characteristics were quantified for bankfull width, stream width, stream depth, bankfull depth, and canopy cover. Canopy cover was quantified using a densiometer (Robert E. Lemmon Forest Densiometer Model-C). Plant identification was undertaken using the key constructed by Crow and Hellquist (2000).

Dissolved oxygen and pH levels were measured in the field using a YSI Model 85® handheld oxygen meter and an Orion 250A® portable pH meter, respectively. Water samples were collected on 30-31 July 2003 from 11 sample sites. Samples were collected in acid-rinsed 500 mL Nalgene® bottles and triple-rinsed with water from the site before each sample was taken. All samples were kept on ice and analyzed within 24 hours for orthophosphate (PO_4), nitrate (NO_3), and ammonia (NH_3) concentrations (Haury 1996). Nutrient concentrations were measured by spectrophotometry using a Thermospectronic Aquamate® and HACH® chemical reagents, according to US EPA approved *Standard Methods for the Examination of Water and Wastewater* (APHA 1999).

A Qualitative Habitat Evaluation Index (QHEI) was also calculated for each site. The QHEI was developed for the Ohio EPA as a means for rapidly assessing the habitat quality of a stream for fish and invertebrate communities (Rankin 1989). QHEI was calculated using several site parameters including substrate, in-stream cover, riparian vegetation, and floodplain quality, as well as stream profile measurements that included stream depth, stream width, bankfull width, and bankfull depth (Rankin 1989). Swamps and backwater areas were avoided because of the tendency for striking changes in species composition and abundance (Small and others 1996).

We conducted one-way ANOVAs with Tukey post-hoc tests on NH_4, NO_3, and PO_4 concentrations for testing the effects of site and site type. A regression analysis was also conducted between the index of biotic integrity and qualitative habitat evaluation index. Statistical analyses were conducted using SYSTAT statistical program (Wilkinson 1998).

RESULTS

Physical-Chemical Characterization

Mean stream width, stream depth, and canopy cover of all sampled sites were 11.51 ± 1.17 m, 0.45 ± 0.04 m, and 35.65 ± 4.96%, respectively, and varied among sites (Table 1). ANOVA indicated that nutrients differed significantly (P <0.05) among sites and site types (Tables 2,3; Fig. 3). ANOVAs with Tukey posthoc tests indicated that NO_3 concentrations at BOW sites were significantly lower than at BOB, WOB, and WOW sites (Fig. 3). Phosphate concentrations at BOW sites were also significantly lower than at WOB and WOW sites. Phosphate concentrations at BOW sites were not significantly different from those at BOB sites (Fig. 3). There were no significant differences in NH_3 concentrations among sites.

IBI scores differed significantly among site types (Table 4; Fig. 4) as would be expected since site type designation was based on IBI score. The mean IBI score for BOB sites was 47.33 ± 0.67. A significant dip in mean IBI score was noted between BOB and BOW, WOB and WOW sites. Interestingly, the mean IBI score is very similar between BOW and WOB sites, with no significant difference between them. WOW sites faired far worse with very low IBI (Fig. 4). A similar analysis was performed on QHEI score and site type (Table 4; Fig. 5). QHEI did not vary significantly among site types. Strikingly, BOB and WOW QHEI scores did not differ significantly (Fig. 5). QHEI was positively related to IBI among the 12 sites, but only weakly so (Fig. 6).

A strong correlation was found between stream depth and stream width (stream depth = 0.212 + 0.021 stream width; R^2 = 0.314; P = 0.016). These correlations are logical because streams with greater depth will have greater width. The same was true for canopy

TABLE 1

Mean stream width, depth and canopy cover for each site surveyed (CR: Cuyahoga River; SR: State Route)

Site Name	Latitude/ Longitude	Stream width (m) ± SE	Stream depth (m) ± SE	Canopy cover (% cover ± SE)
Indian Creek	41.2992/-81.5177	4.519 ± 0.266	0.233 ± 0.059	20.989 ± 3.027
Pond Brook	41.36417/-81.4027	6.337 ± 0.668	0.310 ± 0.021	50.787 ± 3.801
Tinkers Creek site 1	41.37623/-81.5452	12.131 ± 0.512	0.350 ± 0.010	49.447 ± 6.160
Deer Lick Run	41.37641/-81.4921	3.268 ± 0.274	0.174 ± 0.011	27.513 ± 10.893
Tributary to Furnace Run	41.26849/-81.6413	2.891 ± 0.347	0.195 ± 0.075	4.916 ± 0.942
Beaver Meadow Creek	41.36147/-81.4685	6.239 ± 0.640	0.442 ± 0.101	45.382 ± 5.439
Tinkers Creek site 2	41.33344/-81.4027	12.741 ± 0.568	0.547 ± 0.062	38.764 ± 10.248
Tinkers Creek site 3	41.37178/-81.4806	14.941 ± 0.643	0.433 ± 0.033	66.749 ± 5.220
Tinkers Creek site 4	41.3842/-81.5115	19.836 ± 0.449	n/a	60.131 ± 4.922
Chippewa Creek	41.3163/-81.5904	6.294 ± 0.857	n/a	68.356 ± 8.834
Little Cuyahoga	41.095/-81.5228	21.378 ± 2.096	0.578 ± 0.060	62.873 ± 7.321
West Branch of CR	41.4865/-81.1757	9.803 ± 1.328	0.942 ± 0.060	53.324 ± 7.803
CR by Lake Rockwell	41.22071/-81.3016	11.584 ± 0.370	0.390 ± 0.020	31.956 ± 3.859
CR at SR 43 and SR 59	41.15328/-81.3603	14.246 ± 0.430	0.675 ± 0.029	13.331 ± 2.269
CR at Cuyahoga Street	41.12096/-81.533162	10.405 ± 0.409	0.305 ± 0.020	58.145 ± 5.949
CR above Monroe Falls	41.11739/-81.52319	13.870 ± 0.462	0.448 ± 0.041	21.840 ± 3.076
CR below Monroe Falls	41.11746/-81.52319	14.429 ± 0.552	0.555 ± 0.029	6.524 ± 0.730
CR at Lock 39/Rockside	41.391631/-81.62739	15.515 ± 0.391	0.519 ± 0.044	11.724 ± 1.829
CR at Route 303	41.243975/-81.5523	15.635 ± 0.288	0.511 ± 0.037	9.549 ± 2.432
CR North of Granger Road	41.41757/-81.64586	14.105 ± 0.425	0.445 ± 0.023	10.778 ± 1.797

TABLE 2

Results of ANOVA testing the difference in concentration of NH_4, NO_3, and PO_4 among all surveyed sites.

Dependent variable	df	Mean-square	*F* ratio	*P*
NH_4 Concentration	10	0.052	207.026	<0.0001
Error	22	0.000		
NO_3 Concentration	10	0.864	2365.410	<0.0001
Error	22	0.000		
PO_4 Concentration	10	0.086	829.267	<0.0001
Error	22	0.000		

coverage (canopy cover = 7.891 + 2.63 bankfull width; $R^2 = 0.415$; $P = 0.044$). Wider streams, in terms of bankfull width, tended to have more canopy coverage over the center of the stream. Impact categories, for example, BOB, BOW, WOB, and WOW sites did not differ with respect to stream depth ($P = 0.726$), stream width ($P = 0.491$), or canopy coverage ($P = 0.255$), according to one-way ANOVA.

Macrophytes

Aquatic macrophytes were found at 7 of the 20 sites. A total of 11 species were found; one of which was an aquatic bryophyte (moss), one floating, two submerged, and six emergent aquatic macrophytes. Nine of the ten flowering plant species surveyed were native; one was non-native, *Potamogeton crispus* L. (Table 5).

BOB and BOW sites had the greatest species diversity and shoot abundance (Table 5; Fig. 7). *Elodea canadensis* Michx., *Iris versicolor* L., and *Pontederia cordata* L. exhibited the greatest shoot abundance of all species (Table 5). BOB and BOW sites contained the greatest number of shoots of aquatic macrophyte species, while WOB sites had relatively low numbers of shoots. WOW sites had none at all (Fig. 7).

TABLE 3

Results of ANOVA testing the effects of site type on NH_4, NO_3, and PO_4 concentrations.

Dependent variable	r^2	df	Mean-square	*F* ratio	P
NH_4 Concentration	0.145	3	0.025	1.643	0.201
Error		29	0.015		
NO_3 Concentration	0.414	3	1.193	6.82	0.001
Error		29	0.175		
PO_4 Concentration	0.457	3	0.132	8.130	0.000
Error		29	0.016		

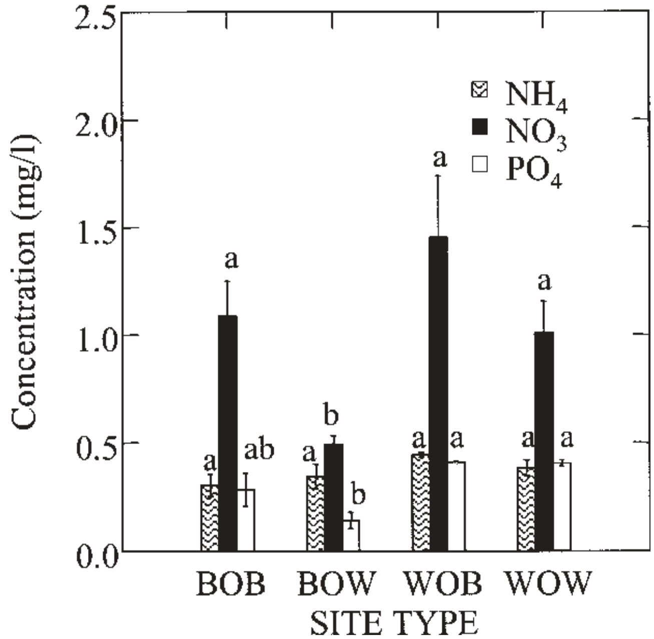

FIGURE 3. Nitrate (NO_3), ammonia (NH_4), and phosphate (PO_4) concentrations in four site types. Different letters above the bars show significant differences in concentrations of a particular nutrient among site types.

TABLE 4

Results of ANOVA testing the effects of site type on IBI and QHEI scores.

Dependent variable	r^2	df	Mean-square	*F* ratio	P
IBI	0.834	3	481.222	13.367	0.002
Error		8	36.000		
QHEI	0.239	3	158.833	0.839	0.509
Error		8	189.250		

The most common species found, in terms of shoot abundance, was *E. canadensis*. It is a submerged macrophyte that often occurred in large assemblages. One of the assemblages surveyed contained over 1600 shoots. *Sparganium americanum* Nutt. was widely distributed in one of the sites. Another very interesting find was *Fontinalis sphagnifolia* (Mull. Hal) Wijk & Margad, an aquatic moss that has been found mostly in Ohio. In fact, more than 38% of records of aquatic bryophytes are from Ohio (New York Botanical Garden 2003).

DISCUSSION

The main purpose of this study was to assess the potential for using macrophytes to distinguish sites differing in urban impacts and, therefore, evaluate the potential for further development of a macrophytic index of biological integrity for NE Ohio streams. From this

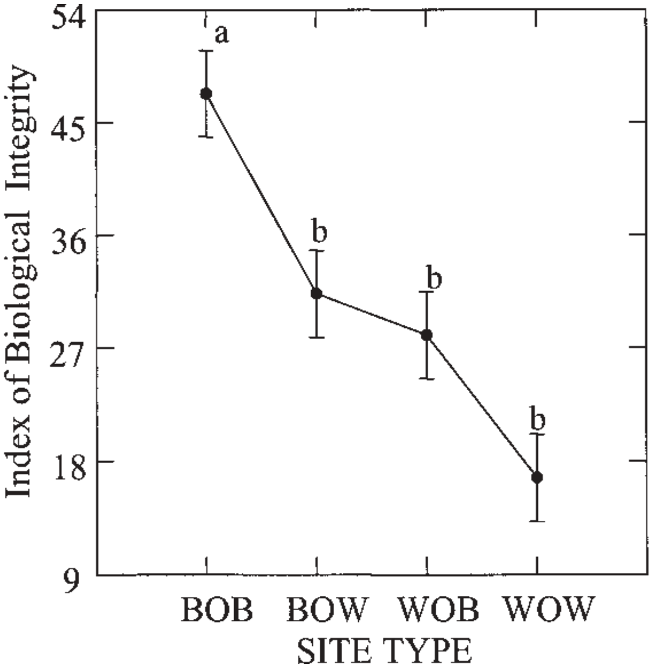

FIGURE 4. Index of Biological Integrity (IBI) scores by site types (least square means). Different letters show significant differences in IBI among site types.

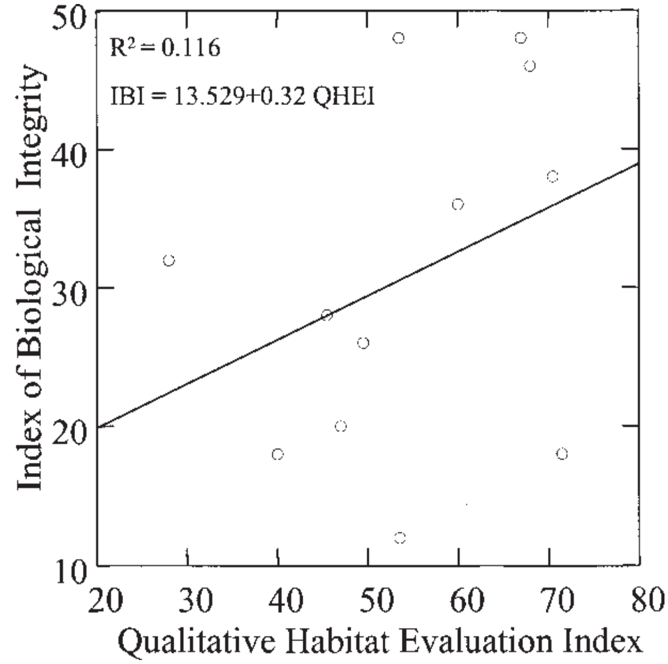

FIGURE 6. Relationship between Index of Biological Integrity (IBI) and Qualitative Habitat Evaluation Index (QHEI).

perspective, this study was successful generally. Species richness and abundance of shoots varied essentially as predicted among impact categories. Sites categorized as "Best of the Best" (BOB sites) in terms of fish-community IBI scores had the highest number of species and greatest abundance of shoots. In contrast, the sites with the poorest IBI scores in the region relative to their landscape (WOW sites) had no macrophytes at all, and sites with intermediate IBI scores (BOW and WOB sites) also had intermediate macrophyte species richness and shoot abundance. Hence, the results of this small, preliminary study indicate that additional work to develop a macrophytic index is warranted.

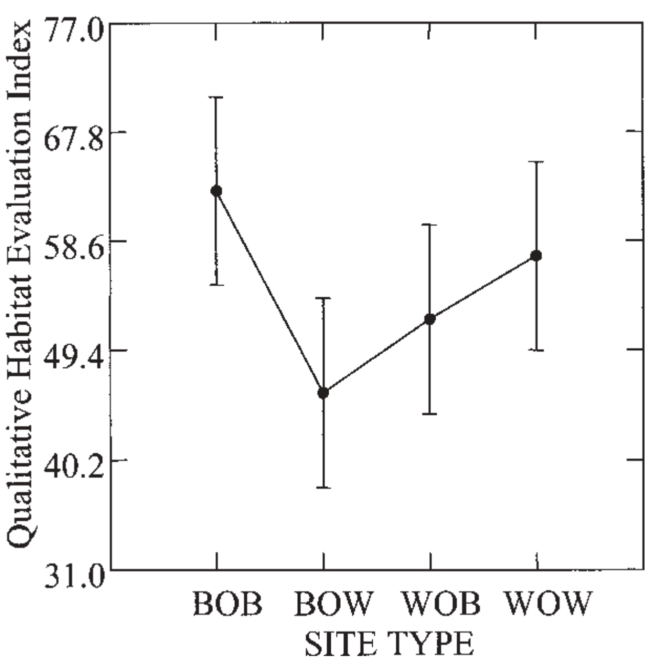

FIGURE 5. Qualitative Habitat Evaluation Index (QHEI) scores by site types (least square means). No significant differences were observed in QHEI among site types.

The results of the current study, albeit preliminary, suggest some other interesting potential properties of the macrophyte index. For example, there was no apparent correspondence between nutrient content of the streams and the degree of impact on the macrophyte communities. Similarly, stream habitat quality for fish communities, as scored by QHEI, also did not differ among sites nor did QHEI covary with macrophyte community properties. Nor were physical characteristics of streams useful predictors of the occurrence of aquatic macrophytes at different sites. This could be due, at least in part, to the fact that a long legacy of urbanization within the region has already reduced macrophyte diversity and selected for those species that are somewhat resistant to nutrient effects and in-stream degradation. Indeed, this may be a general phenomenon in urban areas. In the Cuyahoga River basin, riparian and broader scale land uses may be just as important, if not more so, as predictors of fish community integrity, IBI, than QHEI (Walton and others in review). Indeed, although QHEI and IBI were weakly correlated among the 12 sites examined here, there was no difference in QHEI between sites that were among the best in the region (BOB) and those that were the worst (WOW) with regard to fish community integrity.

Also, the complete loss of macrophyte communities at the lowest quality WOW sites suggests the potential for a threshold response to urban impacts by macrophyte communities. Similarly, fish and invertebrate indicators of ecological integrity often exhibit sharp declines after some critical level of urbanization impact, although the exact nature of this threshold is likely to

TABLE 5

Abundance of aquatic macrophytes in different sites in the Cuyahoga River (CR) watershed (SR: State Route).

Species	Site Name	Mean number of shoots per 100 m linear transect ± SE
Elodea canadensis L.	Indian Creek	1.389
	West Branch of CR	79.629 ± 48.621
	CR by Lake Rockwell	20.165 ± 9.066
	CR above Monroe Falls	3.524
	CR North of Granger Road	11.051 ± 4.122
Iris versicolor L.	Indian Creek	10.748
Fontinalis sphagnifolia (Mull. Hal.) Wijk & Margard	Beaver Meadow Creek	1.008 ± 0.477
Peltandra virginica L. (Schott & Endl.)	West Branch of CR	1.040
	CR by Lake Rockwell	1.187
	CR at SR 43 and SR 59	0.309
Pontederia cordata L.	West Branch of CR	5.417
Sagittaria latifolia Willd.	West Branch of CR	1.210 ± 0.328
	CR by Lake Rockwell	1.067 ± 0.199
	CR North of Granger Road	1.067 ± 0.199
Sparganium americanum Nutt.	West Branch of CR	1.049 ± 0.482
	CR by Lake Rockwell	0.822 ± 0.104
	CR at SR 43 and SR 59	0.625 ± 0.224
	CR above Monroe Falls	0.990
	CR North of Granger Road	0.558 ± 0.035
Potamogeton crispus L.	West Branch of CR	0.312
	CR at SR 43 and SR 59	1.539
Nymphaea odorata Aiton	West Branch of CR	0.104
	CR by Lake Rockwell	2.225 ± 0.849
Alisma subcordatum Raf.	West Branch of CR	1.318
Lemna minor L.	CR at SR 43 and SR 59	23.689

vary among urban regions (Yoder and others 1999, 2000) and, perhaps, with the particular variable chosen to score urbanization.

These are issues that must be investigated in order to fully develop a macrophyte index, yet we consider the prospects encouraging. Previous surveys of macrophytes have shown that macrophytes can be key components for biomonitoring of aquatic habitats (Small and others 1996). Macrophytes could also be used in evaluating the functional typology of a watershed or streams (Haury 1996). More importantly, anatomical features of macrophytes could be used as an indicator of pollution hot spots (Kapitonova 2002). The assessment of diversity of aquatic macrophytes in conjunction with a study of their normal and pollutant-induced anatomical variation can be a powerful predictive tool for monitoring aquatic ecosystem health (Newbold and Holmes 1987; Lovett-Doust and other 1994; Thiebaut and others 2002).

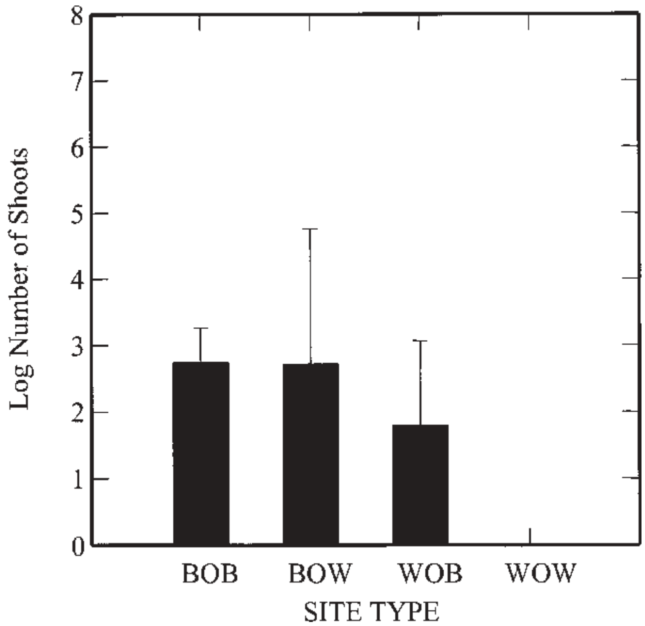

FIGURE 7. Total number of shoots of all aquatic macrophytes in four different site types.

Future studies will assess more sites and sites that encompass finer resolution of urbanization gradients within the region. We will also examine the relationships among macrophyte diversity and water chemistry, canopy coverage, turbidity, velocity and order of streams among other environmental factors. In addition, we will seek to characterize the occurrence of particular species to evaluate which are especially resistant or sensitive to urbanization impacts. In addition, we will develop models similar to those we have used to evaluate landscape predictors of IBI scores, to determine which features of the urban landscape are most likely to affect macrophyte communities.

Surveys of macrophytes are vital for identifying both the diversity and integrity of the ecological systems operating within a watershed (Angermeier and Karr 1994). The survey of aquatic macrophytes adds to the body of knowledge regarding the watershed and may be useful to other scientists and resource managers within the region. The overlap in species surveyed between this study and that of Simpson and others (1969) indicates that macrophyte richness has not deteriorated in the Cuyahoga River watershed, at least since that earlier survey. Sites (CR above Monroe Falls, CR below Monroe Falls, CR by Lake Rockwell) shared many of the same species in both surveys. Further, at the Cuyahoga River north of Granger Rd., three macrophyte species were quantified in the present study while in 1969 no macrophyte species were observed (Simpson and other 1969). This outcome suggests that water quality improvements driven by the Clean Water Act (US EPA 2003b) may have sustained, and even improved at some sites, macrophyte species richness over time. However, this is not the case throughout the watershed; one site surveyed in 1969 (CR at Cuyahoga St.) contained five macrophyte species, while none were observed in the current study.

Although only one non-native macrophyte species was identified (*Potamogeton crispus*), the finding of this species at two sites within the watershed may be significant nonetheless. For example, *Lythrum salicaria* (purple loosestrife) often forms extensive monocultures in North American wetland habitats (Mal and others 2002). Such colonization is often associated with declining diversity of native species (Mal and others 1992, 1997a,b). Similarly, *Potamogeton crispus* has the ability to spread rapidly and hamper the growth of native aquatic macrophytes (ODNR 2003). The species is of increasing concern in Ohio and throughout the United States (ODNR 2003).

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