Using Biological Criteria to Assess and Classify Urban Streams and Develop Improved Landscape Indicators

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Abstract

This study consisted of a quantitative analysis of the relationship between the Index of Biotic Integrity (IBI), an indicator of urban land use, and a qualitative analysis of overlying stressors in six of the major metropolitan areas of Ohio. A database consisting of 267 sampling locations was extracted from the Ohio EPA statewide biological and habitat database. Most of these sites were sampled between 1990 and 1998 and contained watershed areas less than 50 $mi.^2$, with most draining less than 20 $mi.^2$. A negative relationship between IBI and urban land use was observed in four of the six areas, whereas little or no relationship was seen in two areas. For each area, the highest percentage of urban land use that corresponded to minimum attainment of the applicable warmwater habitat IBI biocriterion ranged from 1% (Cleveland/Akron) to 12% (Dayton) for the regression line, and 15% (Cleveland/Akron) to 58% (Columbus) as the highest %urban land use where the IBI biocriterion was attained at any given site. No significant linear relationship was found in either the Toledo or Youngstown areas, and only a weak relationship was visually apparent for the Toledo streams. The lack of association was due to the strong presence of overlying stressors (e.g., legacy pollutants, sewage discharges, combined sewer overflows) that resulted in very low IBI values at sites with lower levels of urbanization. The percentage of urban land use explained approximately 35% of the variation in IBI scores in the regression model when these impact types were excluded (compared to 11% when included). The maximum %urban land use that commonly corresponded to attainment of the warmwater habitat IBI biocriterion based on inspection of the scatter plot was approximately 26%. Only a very few sites exhibited attainment at urban land uses between 40-60% and none occurred above 60%. These former sites had either an intact, wooded riparian zone, a continuous influx of groundwater, and/or the relatively recent onset of urbanization. These results indicate that it might be possible to mitigate the negative effects of urbanization by preserving or enhancing near and instream habitats, particularly the quality of the riparian buffer zone. The results also suggest that there is a threshold of watershed urbanization (e.g., >60%) beyond which attainment of warmwater habitat is unlikely. This threshold is not the same in all watersheds and it can occupy a rather wide range. It is affected by co-factors such as pollutant loadings, watershed development history, chemical stressors, and watershed scale influences such as the quality of the riparian buffer and the mosaic of different types of land use. Thus, single-dimension urban land use indicators, such as watershed imperviousness, are not sufficiently precise or robust as a single indicator of use attainability. The further development and refinement of multiple indicators of watershed urbanization has merit from a management and decision-making standpoint. We suggest that co-factors, in addition to more refined urban land use indicators, be better developed. More precise definitions of different urban land uses are also needed to better understand and respond to the water quality management challenges posed in existing and developing urban areas.

Introduction

The health and well-being of the aquatic biota in surface waters is an important barometer of how effectively we are achieving the goals of the Clean Water Act (CWA); namely the maintenance and restoration of biological integrity, and the basic intent of water quality standards. States designate water bodies for beneficial uses (termed designated uses) that, along with chemical, physical, and biological criteria, assure the protection and restoration of aquatic life, recreational, and water supply functions and attributes. Biological criteria are the principal tool for determining impairment of designated aquatic life uses as defined by the Ohio WQS (Ohio Administrative Code 3745-l). As such, bioassessments play a central role in the Ohio Nonpoint Source Assessment (Ohio EPA 1990; 1991), the biennial Ohio Water Resource Inventory (305b Report; Ohio EPA 1998), and watershed-specific assessments, of which Ohio EPA completes between 6 and 12 each year. Biological criteria represent a measurable and tangible goal, against which the effectiveness of pollution control and other water quality management efforts can be judged. However, biological assessments must be accompanied by appropriate chemical/physical measures, land use characterization, and pollution source information necessary to establish linkages between stressors and the biological responses (Yoder and Rankin 1998). Biological criteria in the Ohio WQS also supports the determination of appropriate aquatic life use designations for individual water bodies, provides for a "reality check" on the application of surrogate indicators, assesses cumulative impacts, extends anti-degradation concerns to nonpoint sources and habitat influences, defines high quality waters, and serves as a meaningful indicator in the management of regulatory programs for environmental results. This provides a means to incorporate the broader concept of water resource integrity (Karr et al. 1986) in policy and planning while preserving the appropriate roles of the traditional chemical/physical and toxicological approaches developed over the past three decades.

We, and others at Ohio EPA, have previously described the status of Ohio's streams and rivers as affected by watershed urbanization (Yoder et al. 1999; Yoder and Rankin 1997; Yoder 1995). Small watersheds are especially impacted, as illustrated by Yoder and Rankin (1997), where no headwater streams in established urban settings throughout Ohio attained the minimum CWA benchmark use designation of warmwater habitat. This finding has led to the perception that the impairment of beneficial aquatic life uses in these small watersheds is intractable, at least within the constraints of current land use policies, restoration technologies, and funding levels. Together, these factors present potentially significant barriers to the objective of fully restoring degraded watersheds or upgrading urban streams that are presently designated for less than fishable and swimmable uses.

Headwater streams are critical to watershed functioning in that they serve as the principal interface between runoff from land use and receiving streams. The ability of a headwater stream to physically filter and biologically assimilate the primary and secondary effects of pollutants is a function of habitat quality and the structure of the biological system. A healthy headwater stream ecosystem is characterized by good habitat and a well balanced assemblage of aquatic organismsand plants, one which processes external inputs in a manner which promotes high quality downstream exports. These exports include good quality water and high value biomass, both of which positively impact the ability of downstream waters to deliver quality goods and services (e.g., water supply, recreation, waste assimilation, water retention, ecological values). A degraded headwater stream ecosystem is characterized by poor habitat and an assemblage of aquatic organisms and plants that processes external inputs in a manner which promotes low quality downstream exports. Thus in this latter scenario, the effects from urban runoff can accumulate in a downstream direction and adversely affect water quality and ecosystem goods and services in larger water bodies. In Ohio, more than 78% of stream miles drain less than 20 $mi.^2$ and are classified as headwater streams. While these may individually seem less significant than larger water bodies, they are collectively the most numerous and perhaps important stream type. In many ways, and in a collective sense, headwater streams are analogous to the capillaries of the human circulatory system where essential product transport and waste assimilation functions are accomplished. Certainly the finding that a high proportion of headwater streams fail to meet CWA goals in Ohio urban areas translates to the potential for undesirable impacts in downstream waters and obvious consequences for the overall health of the "patient".

There is concern that the attainment of CWA goal uses (e.g., warmwater habitat in Ohio) within small urban watersheds may be precluded by the legacy of urbanization. If this is true, how is this determined and what are the protection endpoints to guide water quality management? Federal water quality standards regulations (40CFR, Part 131) allow for the establishment of a use that is less than the CWA fishable and swimmable goals when it is precluded by the following:

1) the degraded conditions are naturally occurring;

2) restoring the degraded conditions would result in widespread adverse socioeconomic impacts;

3) the degraded conditions are irretrievable and human induced.

Such uses are established on a waterbody-specific basis and are supported by a use attainability analysis. In Ohio, such analyses are routinely conducted as a result of the five-year basin approach to monitoring, assessment, and water quality management. One purpose of this paper is to advance the development of the tools and indicators needed to make use attainability decisions in urban watersheds.

Ohio EPA routinely conducts biological and water quality surveys, or "biosurveys", on a systematic basis statewide. A biosurvey is an interdisciplinary monitoring effort coordinated on a waterbody-specific or watershed scale. Such efforts may be relatively simple, focusing on one or two small streams, one or two principal stressors, and a handful of sampling sites; or much more complex, including entire drainage basins, multiple and overlapping stressors, and tens of sites. Each year, Ohio EPA conducts biosurveys in 1 O-l 5 different study areas with an aggregate total of 350-450 sampling sites. Biological, chemical, and physical monitoring and assessment techniques are employed in biosurveys in order to meet three major objectives: 1) determine the extent to which use designations assigned in the Ohio Water Quality Standards (WQS) are either attained or not attained; 2) determine if use designations assigned to a given water body are appropriate and attainable; and 3) determine if any changes in key ambient biological, chemical, or physical indicators have taken place over time, particularly before and after the implementation of point source pollution controls or best management practices. The data gathered by a biosurvey is processed, evaluated, and synthesized in a biological and water quality report. The findings and conclusions of each biological and water quality study may factor into regulatory actions taken by Ohio EPA and are incorporated into Water Quality Permit Support Documents (WQPSDs), State Water Quality Management Plans, the Ohio Nonpoint Source Assessment, and the Ohio Water Resource Inventory (305[b] Report).

In 1990, the Ohio EPA initiated an organized, sequential approach to monitoring and assessment, termed the Five-Year Basin Approach. One of the principal objectives of this new approach was to better coordinate the collection of ambient monitoring data so that information and reports would be available in time to support water quality management activities such as the reissuance of NPDES permits and periodic revision of the Ohio Water Quality Standards (WQS). Ohio EPA's approach to surface water monitoring and water quality management via the Five-Year Basin Approach essentially serves as an environmental feedback process taking "cues" from environmental indicators to effect needed changes or adjustments within water quality management. The environmental indicators used in this process are categorized as stressor, exposure, and response indicators (Yoder and Rankin 1998). *Stressor* indicators generally include activities that impact, but which may or may not degrade the environment. This includes point and nonpoint source loadings, land use changes, and other broad-scale influences that generally result from anthropogenicactivities. *Exposure* indicators include chemical-specific, whole effluent toxicity, tissue residues, and biomarkers, each of which suggests or provides evidence of biological exposure to stressor agents. Responseindicators include the direct measures of the status of use designations. For aquatic life uses, the community and population response parameters that are represented by the biological indices that comprise Ohio EPA's biological criteria are the principal response indicators.

Previously, our analyses examined the water quality and biological assessment database from watersheds in and near existing and developing urban and suburban areas of Ohio. Yoder and Rankin (1997) compiled their analyses based on sampling conducted at more than 100 stream sampling locations. Yoder et al. (1999) examined more detailed land use and stressor relationships with the Index of Biotic Integrity (IBI), based on fish assemblage data, and the Invertebrate Community Index (ICI), based on macroinvertebrate assemblage data, within two major Ohio urban areas (Akron/Cleveland and Columbus). This study consisted of a quantitative analysis of the relationship between the IBI, an indicator of urban land use, and a qualitative analysis of other stressors influencing this relationship using available data from all six of the major metropolitan areas within Ohio. The importance of understanding these relationships is heightened by contemporary water quality management issues such as combined sewers and stormwater permitting. One challenge we faced was in attempting to separate the influences of these multiple stressors on aquatic life attainment status. Could we sufficiently understand the baseline influence of urbanization apart from these other and better understood stressors?

The principal analysis conducted in this study examined the relationship between urban land cover and the IBI, both visually and by statistical analysis. Some goals were to determine the extent to which biological performance (as expressed by the IBI) was correlated with urban land use, thresholds at which this occurred, and the overlying effects of other stressors.

Met hods

A database consisting of 267 sampling locations from the six major metropolitan areas of Ohio was extracted from the Ohio EPA statewide biological and habitat database. Most of these sites were sampled between 1990 and 1998 and contained watershed areas less than 50 mi.^{2}, with most draining less than 20 mi.^{2}. As such, the database represents a collection of discrete watershed units where land uses may have a significant effect on the composition and quality of the instream habitat and biological communities. Urban land use effects have been much more apparent in these smaller watersheds as evidenced by the higher proportion of impaired stream miles compared to larger streams and rivers in Ohio (Yoder 1995; Yoder and Rankin 1997).

Fish communities were sampled using generator-powered, pulsed D.C. electrofishing units and a standardized methodology (Ohio EPA 1987a,b, 1989a,b; Yoder and Smith 1999). Fish community attributes were collectively expressed by the Index of Biotic Integrity (IBI; Karr 1981; Karr et al. 1986), as modified for Ohio streams and rivers (Yoder and Rankin 1995; Ohio EPA 1987b, 1989b). Habitat was assessed at all fish sampling locations using the Qualitative Habitat Evaluation Index (QHEI; Rankin 1989, 1995). The QHEI is a qualitative, visual assessment of the functional aspects of stream macrohabitats (e.g., amount and type of cover, substrate quality and condition, riparian quality and width, siltation, channel morphology, etc.). Ohio EPA also collected macroinvertebrate assemblage data at some of these sites, but it was not included in this study because of the partial coverage and the extensive use of the qualitative method was not always compatible with regression analysis. Some of the analyses in our earlier studies (Yoder et al. 1999) included macroinvertebrate data.

The urban land use indicator was derived from Landsat Thematic Mapper satellite imagery of land cover classification (September 1994) provided by the Ohio Department of Natural Resources. The percentage of land use in the urban classification was calculated for the subwatershed upstream from each fish sampling location to the boundary of the watershed. Because many of the sites included in the statewide data set are subjected to a variety of stressors, each site was qualitatively classified by predominant impact type. Impact types included least impacted sites, estate sites (i.e., subwatersheds with large lot sizes or green space provided by parks), sites reflecting gross instream habitat alterations (i.e., channel modifications or impoundment), sites impacted directly by discharges from combined sewer overflows (CSOs), sites impacted by wastewater treatment plant discharges, sites impacted by instream sewer line placement and construction (Cincinnati area only), sites with evidence of impacts by legacy pollutants, or sites affected by general urbanization only. This latter category included urban land uses not containing any of the other impact types and usually consisted of residential development.

Results

The relationship between the IBI and urban land use was initially characterized by regressing IBI scores against percent urban land use (log,, transformed) and QHEI scores using a database of 267 sites for all of the six major metropolitan areas of Ohio (Figure 1). Diagnostic plots (e.g., residuals, normal probability) indicated nonconstancyof error variance. To provide insights into whether the results varied substantially between each metropolitan area, scatter plots of the relationship between urban land use and IBI in each of the six metro areas were also made (Figure 2). A negative relationship between IBI and urban land use was observed in four of the six areas, whereas little or no relationship was seen in two areas. For each area, the highest percentage of urban land use that corresponded to minimum attainment of the WWH IBI biocriterion was determined by inspection of the scatter plot and the intersection of the regression line and the WWH IBI biocriterion were determined (Figure 2). This ranged from 1% (Cleveland/Akron) to 12% (Dayton) for the regression line, and 15% (Cleveland/Akron) to 58% (Columbus) as the highest %urban land use where WWH was attained in each area at a given sampling location. No significant linear relationship was found in either the Toledo or

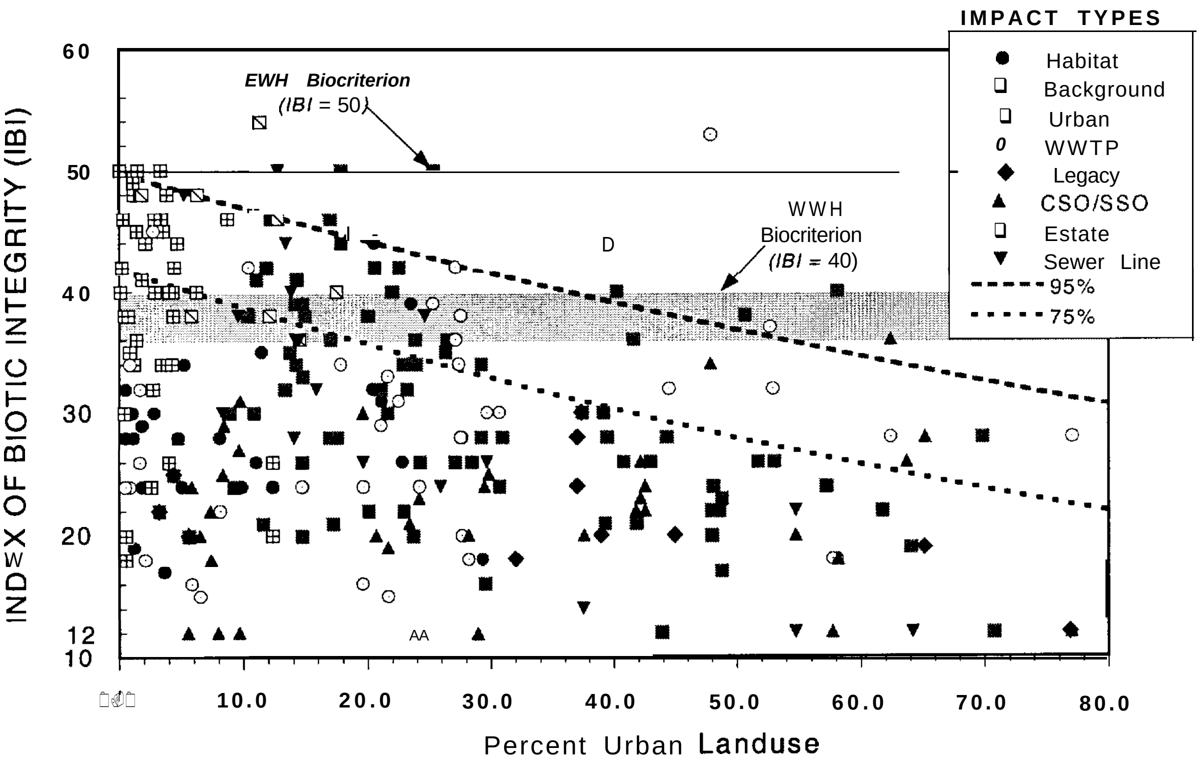

Figure 1. Scatter plot of Index of Biotic Integrity (IBI) scores against percentage of watershed upstream from the site in urban land use at 267 small (<50 mi.^{2}) sampling sites.

Youngstown areas, and only a weak relationship was visually apparent for the Toledo streams. All of the sites in the Youngstown area were impaired, and so severely that no land use relationship was evident (Figure 2). In the Toledo area, the highest urban land use corresponding to WWH attainment was 28%. However, the WWH IBI biocriterion in the Huron/Erie Lake Plain ecoregion is the lowest in the state and almost all of the small streams in the Toledo area have been channel modified to some degree. It was apparent that the lack of a stronger association between IBI and urban land use was due to overlying stressors (e.g., legacy pollutants, WWTPs, CSOs/SSOs), particularly those that resulted in very low IBI values at sites with low levels of urbanization. While some threshold relationships were evident in these results, the resulting variability in IBI scores led to only weak or non-existent linear relationships.

Some of the impact types had a strong effect on the IBI regardless of the effect of urban land use. The IBI results were examined by impact type across all six metro areas (Figure 3). The legacy, CSO/SSO, habitat, and WWTP impact types had the strongest negative effects on the IBI, respectively, and this was independent of the urban land use indicator. While these impact types are common to urban areas, they were removed from the remaining statistical analyses (elimination of these impact types reduced the sample size to 123 sites) to better develop the IBI/urban land use relationship. The entire Toledo and Youngstown datasets were also removed since they are comprised entirely of these impact types. This resulted in a better regression model fit, and diagnostics consistent with regression model assumptions (Neter et al. 1990). This also allowed us to discern the threshold of urbanization at which WWH attainment is lost with greater precision and in the absence of potentially confounding impacts, which was a major objective of our study. The relationship between different levels of urbanization and biotic integrity was further quantified with an analysis of variance model where quartiles of percent urban land use determined factor level (e.g., all sites within the 1 st quartile of percent urban land use were coded as factor level 1). Similarly, an analysis of covariance model using QHEI as the covariate was employed to test for further refinements. Multiple comparisons of factor level mean differences were made using Tukey's method (Neter et al., 1990).

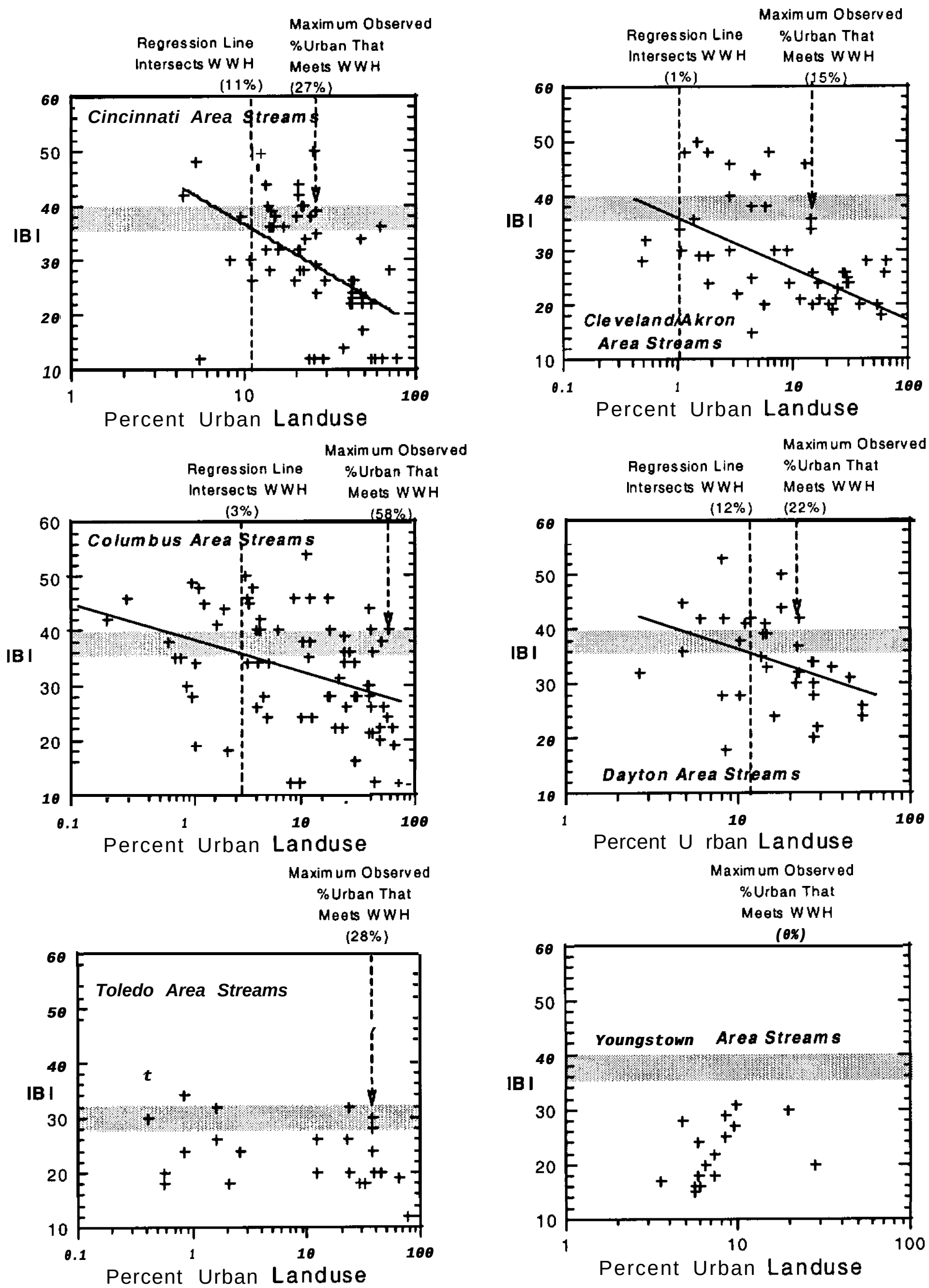

Figure 2. Scatter plots of Index of Biotic Integrity (IBI) scores against percentage of watershed upstream from the site in urban land use at small stream (<50 mi.^{2}) sampling sites in six of the major metropolitan areas of Ohio. Predominant impact types are indicated for each site (see Figure 1) along with the regression line. The warmwater habitat and exceptional warmwater habitat biological criteria for the IBI are also indicated.

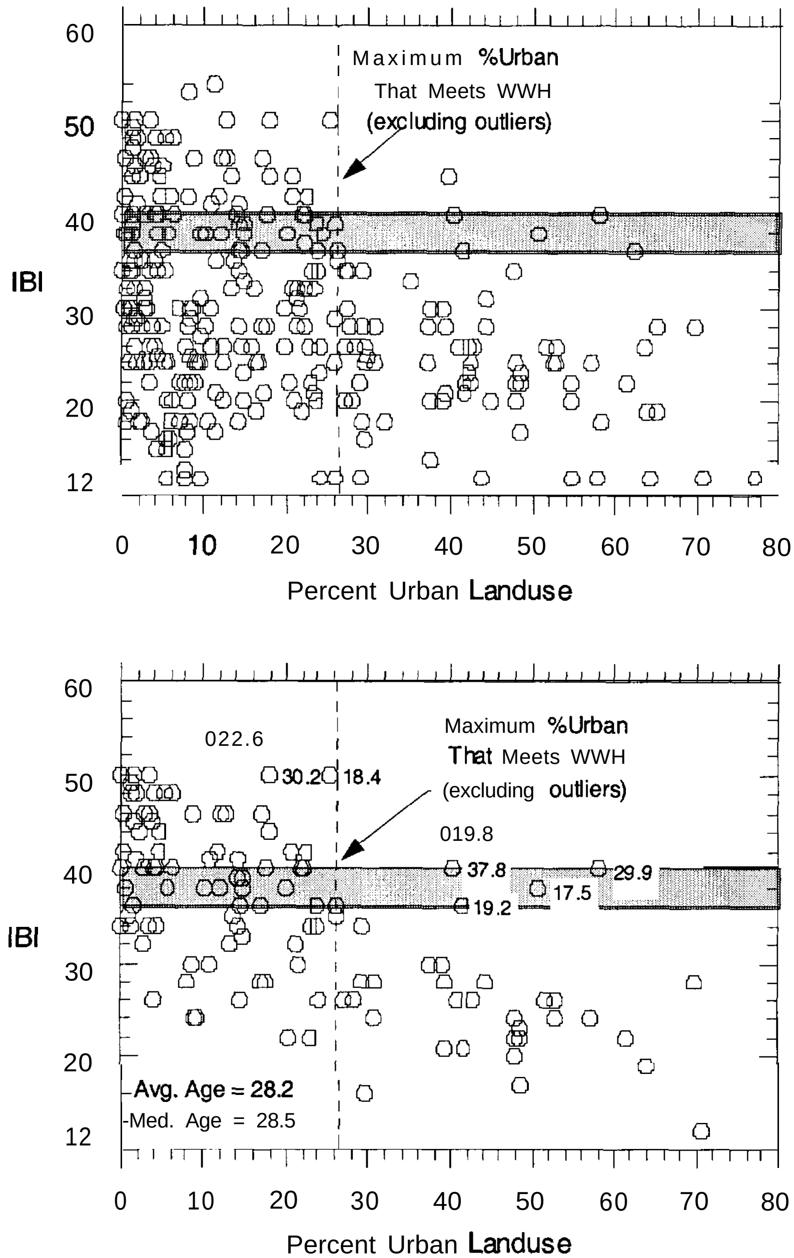

Figure 3. Box-and-whisker plots of Index of Biotic Integrity (IBI) scores by each of the major impact types used in Figures 1 and 2. The warmwater habitat and exceptional warmwater habitat biological criteria for the IBI are indicated.

The percentage of urban land use explained approximately 35% of the variation in IBI scores in the regression model when the other impact types were excluded. Local habitat quality (as measured by the QHEI) explained an additional 7% of the variation (Table 1). The ANOVA model showed that there were significant differences in mean IBI scores between quartile level of percent urbanization, with sites exceeding 29 % urban land cover having lower IBI scores on average than sites with less urban land cover (Figure 4). Sites characterized by less than 4% urban land cover had higher IBI scores than sites with urban land use exceeding 15% The ANCOVA model provided a slightly better fit, but the additional variation explained was marginal (Table 2), and the results of pairwise comparisons were similar between models (Figure 4).

Table **1.** Regression Results for the Model Ibl = Log_{10}(Percent Urban Land Use+l)+qhei for All Sites and the Removal of Selected Impact Types.

Effect	**Coefficient**	**SE**	**t**	**P(2 Tail)**	**Adj. R-Squared**
All Sites					
CONSTANT	21.9333	2.9450	7.4477	0.0000	
Urban	-6.8323	1.1370	-6.0092	0.0000	0.1179
QHEI	0.2676	0.0418	6.4102	0.0001	0.2388
Impact Types Removed					
CONSTANT	32.4069	4.2184	7.6822	0.0000	
Urban	-11.1496	1.3102	-8.5096	0.0000	0.3500
QHEI	0.2390	0.0605	3.9493	0.0001	0.4199

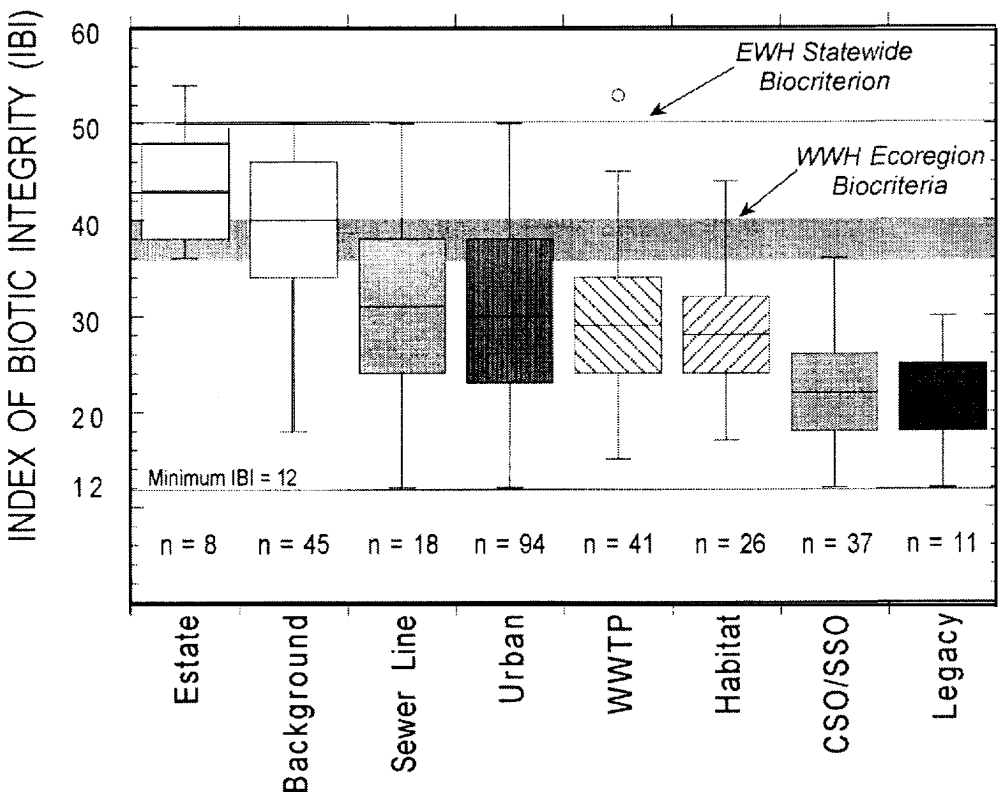

Figure 4. Distributions of Index of Biotic Integrity (IBI) from small streams (<50 mi.^{2}) in the six major metropolitan areas of Ohio plotted by quartiles of percent of urbanization upstream from sampling locations. Horizontal lines spanning adjacent box plots indicate similar means. Levels of percent of urbanization corresponding to the 25th, 50th and 75th percentile are indicated. The shaded areas indicate the applicable warmwater habitat biological criterion and the range of insignificant departure for the IBI.

Table 2. Analysis of Variance Results for the Anova Model, and the Ancova Model Using Qhei as a Covariate.

ANOVA

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P	R-Squared
Urban	4248.53	3	1416.18	27.50	0.0000	0.4094
Error	6129.10	119	51.51			

ANCOVA

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P	
Urban	4020.66	3	1340.22	28.81	0.0000	
QHEI	640.71	1	640.71	13.78	0.0003	0.4711
Error	5488.39	118	46.51			

In an attempt to better visualize where attainment of warmwater habitat occurs along the urban land use gradient, the IBI results were plotted against percent of urban land use for all sites used in this study and with the other impact types excluded (Figure 5). The elimination of the other impact types provided for a more precise statistical relationship between urban land use and the IBI (i.e., lower error of regression estimates). For example, the R^2 was higher with the removal of the other impact types and the slope of the regression was steeper, both of which suggest a more meaningful relationship between the IBI and urban land use (Table 1). However, the percent of urban land use that corresponded to attainment of the warmwater habitat IBI biocriterion based on inspection of the scatter plot was the same (approximately 26%) in both plots.

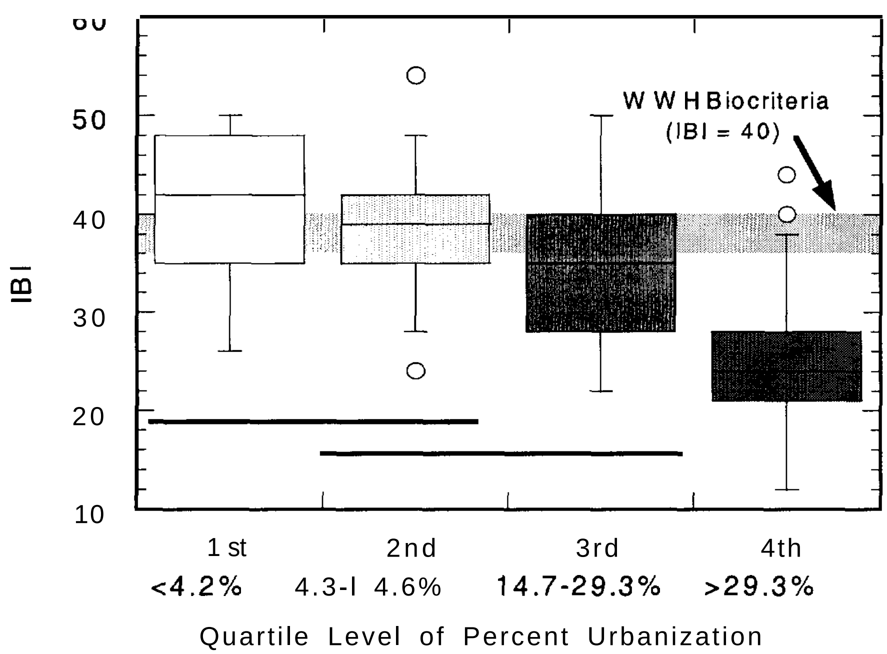

Figure 5. Scatter plots of Index of Biotic Integrity (IBI) scores against percentage of watershed upstream from the site in urban land use at small stream (<50 mi.^{2}) sampling sites in six of the major metropolitan areas of Ohio for all sites (upper) and a subset with non-urban impact types removed (lower). The age of the urbanized area is indicated for selected sites and the mean and median age for entire dataset. The warmwater habitat biological criterion and the range of insignificant departure for the IBI are indicated.

Also apparent in both plots was the occurrence of "outliers" where IBI scores above the warmwater habitat biocriterion occurred at sites with 40% to 60% urban land use. These sites had either an intact, wooded riparian zone, a continuous influx of groundwater, and/or the relatively recent onset of urbanization. Intact riparian buffers can mitigate the effects of urban land use up to a point (Steedman 1988; Horner et al. 1997) and local hydrology can strongly influence the quality of the fish assemblage (Poff and Allen 1995). The three sites with the relatively recent onset of urbanization (all <20 years) may not yet have accrued the types of negative effects that are readily apparent in some of the older urbanized areas of Ohio.

Discussion

Threshold levels of urbanization beyond which biological communities are likely to be impaired have previously been identified in the range of 8% to 20% impervious cover within a watershed (Schuler 1994). Our previous analyses (Yoder et al. 1999) produced results of approximately 8% and 33% urban land use cover for the Cuyahoga River basin and Columbus area streams, as identified by analysis of variance. We also concluded that the threshold level identified by regression for the Cuyahoga River basin was lowered by the presence of other stressors (e.g., CSOs, point sources, legacy pollutants). The elimination of those sites impacted by these other stressorsfrom the regression analysis resulted in a higher threshold of urbanization. Our expanded study seemed to confirm this phenomenon, as the elimination of the other impact types helped clarify the urban land use/IBI relationship in a broader array of urban influenced streams throughout Ohio (Figure 5). The upper threshold of urbanization which corresponded to a loss of warmwater habitat attainment was in the 25-30% range. However, our results show that non-attainment also occurs at lower thresholds of urbanization (Figure 5) due primarily to the co-occurrence of other stressors. This makes both the linear and visual derivation of sufficiently precise indicator thresholds such as percentage of impervious surfaces more difficult.

In terms of understanding the potential effect of urbanization on aquatic life use attainment, the most meaningful results of our analyses are the upper thresholds at which attainment of CWA goal uses are mostly lost (e.g., 25%) and that beyond which it never occurs (>60%). Only a very few sites exhibited full attainment of the warmwater habitat biocriteria at urban land uses between 40-60% (Figure 5). A closer examination of these sites and the watersheds showed the presence of high quality riparian zones, an influx of flow augmenting groundwater, and/or development of the urban land use occurring within the past 20 years. For the latter, we hypothesized that the full effect of negative impacts in an urban setting may take time to accumulate and may not be immediately manifest in the form of instream impairments. This could account for the higher-than-expected urban land use (i.e., 40-60%) correlating with full attainment of the biocriteria. If this is true, then we might expect these sites to exhibit declines in IBI scores over the next one or two decades. It also suggests that newly urbanizing watersheds should be developed with an emphasis on determining which attributes (e.g., riparian zones, wetlands, flow regime) need to be maintained and preserved in order to protect and maintain instream habitat and biological quality.

The results of this study indicate that it might be possible to partially mitigate the negative effects of urbanization by preserving or enhancing near and instream habitats, particularly the quality of the riparian buffer zone. The "outlier" sites that exhibited full attainment of the warmwater habitat biocriteria had more extensive and higher quality riparian zones and good to excellent instream habitat quality. Some streams were nestled in small valleys which were not amenable to development and the accompanying encroachment of urban land uses. This generally agrees with the findings of Steedman (1988) who demonstrated a co-relationship between riparian zone quality and land use in terms of how each affected the fish communities and IBI values of Toronto area streams. Horner et al. (1997) also found that the negative effects of urban land use were mitigated by riparian protection and other management interventions. However, in both studies the quality and extent of the riparian zone ceased to be effective above 45-60% impervious land cover, which generally corresponds to the thresholds identified by our study. Until we better understand the effect of the "age" of the urban effect, it seems prudent to advocate policies that preserve existing riparian zones rather than responding with post-urbanization retrofits.

Yoder et al. (1999) discussed the implications of their findings on the designation of aquatic life uses in state water quality standards, particularly to the use attainability analysis process. Uses designated for specific water bodies are done so with the expectation that the criteria associated with the use are reasonably attainable. If CWA goal uses (e.g., warmwater habitat in Ohio) are found to be unattainable, then lower quality uses may be established and assigned on a case-by-case basis (40CFR, Part 131 .10[g]). Recently, the imperviousness of the watershed has been suggested as an indicator that is correlated with use attainability. If the frequently cited threshold of 25% impermeability is used, streams in watersheds with greater than this value could be considered unlikely to ever attain a beneficial use regardless of site- and reach-specific factors. This assumes that the negative effects of urbanization cannot be remediated, which has yet to be extensively tested. However, the results of our study suggest that there is a threshold of watershed urbanization (e.g., >60%) beyond which attainment of the WWH use becomes increasingly unlikely, at least as affected by contemporary practices. This threshold is not the same in all watersheds, as evidenced by the results from the six Ohio metropolitan areas, and it can occupy a rather wide range. In addition, co-factors such as pollutant loadings, watershed development history, chemical stressors, and watershed scale influences such as the quality of the riparian buffer and the mosaic of different types of land use, also act singly and in combination to determine to the resultant biological quality in the receiving streams. Thus, single dimensional urban land use indicators, such as watershed imperviousness, is not sufficiently precise or reliable as a single indicator of use attainability.

The further development and refinement of multiple indicators of watershed urbanization has merit from a management and decision-making standpoint. Because of the many co-factors involved (e.g., water quality, habitat quality, hydrologic regime, etc.), some of which are controllable and amenable to reasonable remediation, this will be a complex undertaking. We suggest that these co-factors, in addition to more refined urban land use indicators, be developed and tested using datasets from broad geographic areas spanning the extremes of the urbanization gradient. Urban land use and its analogs (e.g., % imperviousness) are coarse approximations of the cumulative effect of all negative influences within a watershed. Thus co-factors and more precise definitions of different urban land uses need to be defined in order to better understand and respond to the water quality management challenges posed in existing and developing urban areas.

A management outgrowth of such an effort could be the development of an urban stream habitat use designation. Yoder et al. (1999) previously indicated where the biological criteria for this potential new use designation might occur compared to the already existing hierarchy of aquatic life uses in the Ohio WQS (Figure 6). This designated use would

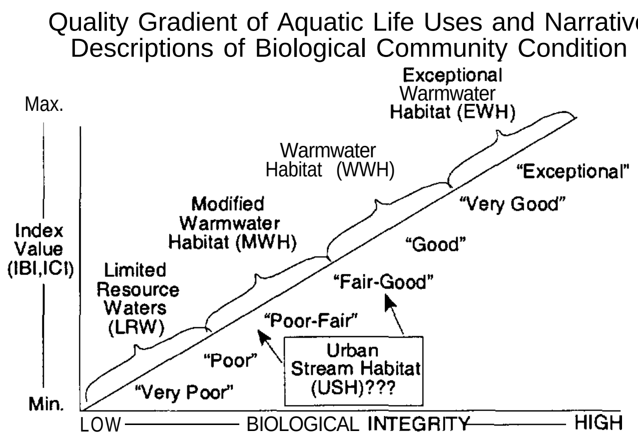

Figure 6. Relationship between the tiered aquatic life uses in the Ohio WQS and narrative evaluations of biological community performance and how this corresponds to a qualitative scale of biological integrity and of the biological indices that comprise the Ohio biological criteria. The position of a potential new Urban Stream Habitat (USH) use designation is indicated (after Yoder et al. 1999).

satisfy the desire to afford urban streams the maximum protection practicable, while recognizing the inherent limitations that the irretrievable effects of urbanization may impose on stream quality. In the meantime, simplistic regulatory and management approaches should be avoided, particularly in those watersheds where uncertainty about the attainability of CWA goal uses (i.e., WWH and higher) exists. For example, a single indicator of urban development (e.g., proportion of impermeable surfaces) is alone insufficient to drive this process. We envision that more refined, multiple indicators of urban development will provide the necessary sophistication to more appropriately define when this less than CWA goal use should be applied. In the meantime, management strategies such as the nine minimum controls for CSOs seem reasonable analogies for the management of urban watersheds and stormwater runoff. However, proceeding beyond such minimum requirements with long-term remediation plans should be done with deference to the use attainability issues and with the aid of sufficiently robust before-and-after biological and water quality assessments.

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