

## Assessing the Status of Aquatic Life Designated Uses in Urban and Suburban Watersheds

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### 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, 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 specific chemical, physical, and biological criteria, assure the protection and restoration of aquatic life, recreational, and water supply functions and attributes. Ohio Environmental Protection Agency (EPA) employs biological, chemical, and physical monitoring and assessment techniques to assess the status of these beneficial uses and to satisfy 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.

An integrated biological, chemical, and physical monitoring and assessment approach has been used to support all relevant water quality management activities, including urban stormwater issues, within Ohio EPA during the past 18 years. The details of this process have been extensively described elsewhere (Ohio EPA 1987a,b; Ohio EPA 1989a,b; Yoder and Rankin 1995, 1998).

### Urban Watersheds

Urban watersheds in Ohio exhibit a familiar legacy of aquatic resource degradation. Few, if any, ecologically

healthy watersheds exist in the older, most extensively urbanized areas of Ohio (Yoder 1995) and no headwater streams (i.e., draining <20 mi.<sup>2</sup>) sampled by Ohio EPA during the past 18 years in these areas have exhibited full attainment of the Warmwater Habitat (WWH) use designation (Yoder and Rankin 1997).

The activities that have the greatest impacts on aquatic life in Ohio's urban watersheds include the wholesale alteration of watershed hydrology, loss and degradation of riparian habitat, direct instream habitat degradation via channelization, culverting, and interceptor sewer line placement, excessive sedimentation resulting from land disturbance activities and stream bank erosion (strongly linked to riparian encroachment), and contributions of excessive nutrients, oxygen-demanding wastes, and toxic chemical pollutants via urban runoff, point source discharges (both permitted and unpermitted), and spills and other releases. According to the 1996 Ohio Water Resource Inventory (305[b] report), urban and suburban sources are responsible for aquatic life use impairment in nearly 1000 miles of Ohio streams and rivers and more than 23,000 acres of lakes, ponds, and reservoirs (Ohio EPA 1997). These activities also threaten existing full use attainment in nearly 160 miles of streams and rivers and may pose a potential problem in more than 4380 miles of streams and rivers that have not yet been fully monitored and evaluated. These are also one of the fastest growing threats as urban and suburban development extends further into rural watersheds.

While much attention has been paid to toxic substances in urban runoff, evidence suggests that sedimentation is the most pervasive *single* cause of impairment associated with nonpoint sources in Ohio. While sediment deposition in lotic and lentic environments is a natural process, it becomes a problem when the capability of the ecosystem to

"assimilate" the sediment load is exceeded. The effects of sediment on aquatic life are the most severe in the ecoregions of Ohio where: (1) upland erosion and runoff are moderate to high, (2) clayey silts that attach to and fill the interstices between coarse substrates predominate, and (3) streams and rivers lack the ability to expel the finer grained sediments from the low-flow channel because of instream and riparian habitat degradation. Estimates of gross erosion *alone* are not consistently correlated with adverse impacts to aquatic communities, although this is a frequently used indicator for prioritizing nonpoint source management efforts (Yoder 1995).

### **Bioassessment of Urban Watersheds**

Ohio EPA uses biological criteria via a bioassessment approach in the designation and assessment of rivers and streams. Biological criteria are the principal tool for determining impairment of designated aquatic life uses and 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 1997), and watershed-specific assessments of which Ohio EPA completes from 6-12 each year. Biological criteria represent a measurable 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 source information necessary to establish linkages between stressors and the biological responses.

### **Methods And Analyses**

For bioassessments to achieve their maximum effective use in the assessment of urban streams, a watershed design to sampling and analysis should be employed. A recent example is the Cuyahoga River basin in northeastern Ohio and small, wadeable streams of the Columbus metropolitan area (Franklin County) in central Ohio. The former represents historically and extensively urbanized streams including a mix of residential, commercial, and industrial land use, streams draining recent and rapid suburban development, and larger streams which are dominated by point source effluents, principally treated municipal sewage. The latter case includes small watersheds affected mostly by residential urban land use with a wide range of intensity from older areas to recent and rapidly developed suburban areas.

### **Biological and Water Quality Assessments**

Fish and macroinvertebrates were sampled respectively, at 82 and 48 locations, in the Cuyahoga River basin in 1996, and an additional 32 locations were sampled for macroinvertebrates in 1991. Water samples were collected up to six times at 40 macroinvertebrate sampling locations and 63 fish sampling locations, and included standard field parameters (D.O., temperature, pH, conductivity), nutrient series (N and P), demand parameters (suspended solids, BOD, COD), and selected heavy metals. Drainage areas at Cuyahoga River basin stream sites ranged from approxi-

mately 2 to 700 mi<sup>2</sup>. Fish communities only were sampled in the Columbus area, at 80 stream locations with drainage areas at all sites less than 35 mi<sup>2</sup>. No water chemistry samples were collected. Macroinvertebrate community performance was evaluated using the Invertebrate Community Index (ICI; DeShon, 1995). The ICI is a multimetric index comprising ten attributes of community structure and composition. The individual metrics were scored against expectations derived from least-impacted reference sites (Ohio EPA 1987b, 1989a; DeShon 1995; Yoder and Rankin 1995). Fish communities were sampled using generator-powered, pulsed D.C. electrofishing units and a standardized methodology (Ohio EPA 1987b, 1989b). Fish community attributes were collectively measured with the Index of Biotic Integrity (IBI; Karr 1981; Karr et al., 1986) modified for Ohio streams and rivers (Yoder and Rankin 1995; Ohio EPA 1987b). 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.).

Two indicators of urbanization were developed for the Cuyahoga River basin, housing density and urban land use cover. Housing density by Census Block Group was obtained from the 1990 Census of Population (U.S. Bureau of Census, 1990). Urban land use cover was derived from Landsat Thematic Mapper satellite imagery of land cover classification (September 1994) provided by the Ohio Department of Natural Resources. The number of housing units per hectare was calculated for the subwatershed upstream from each fish and macroinvertebrate sampling point to the boundary of the watershed. The percent urban land use for subwatersheds upstream from the fish sampling locations only were similarly calculated for both the Cuyahoga Basin and Columbus area study areas.

### **Statistical Analyses**

IBI scores were regressed against chemical water quality parameters, an index of habitat quality (QHEI), and housing density. ICI scores were regressed against chemical water quality parameters and housing density. Water quality parameters were expressed as the average concentrations of phosphorus, dissolved oxygen (D.O.), nitrate+nitrite-nitrogen, ammonia-nitrogen, arsenic, lead, and cadmium (macroinvertebrates only) based on grab samples collected 6-8 times during June-October. Lead was highly intercorrelated with zinc, copper and chromium. Arsenic and cadmium were intercorrelated at fish sampling locations. Transformations used to correct departures from normality are provided in Table 1.

The relationship between different levels of urbanization, as indicated by housing density or percent urban land use (IBI only), and performance of the IBI, ICI, and selected metrics was further quantified using an analysis of variance model where quartile distributions of housing density and percent urban land use (e.g., 1st quartile ≤

Table 1. Parameter Estimates from the Regression of IBI on Water Quality Variables, Habitat Quality (QHEI) and Housing Density, and ICI on Selected Water Quality Variables and Housing Density.

Effect	Coefficient	Std Error	Index of Biotic Integrity (IBI)		Adjusted R <sup>2</sup>
			t	Squared multiple R: 0.368	
Constant	23.318	11.019	2.116	0.039	
Log <sub>10</sub> (Ar)	5.123	9.740	0.526	0.601	-0.011
Dissolved Oxygen	0.549	0.852	0.644	0.522	0.006
Log <sub>10</sub> (Pb)	3.997	5.923	0.675	0.503	0.022
1/NH <sub>3</sub>	-0.098	0.107	-0.916	0.364	-0.011
QHEI	0.091	0.095	0.952	0.346	0.071
Log <sub>10</sub> (TP)	-7.876	4.781	-1.647	0.105	0.048
Log <sub>10</sub> (NO <sub>x</sub> )	-4.484	2.053	-2.184	0.033	0.063
(House/Hectare)	-7.171	1.769	-4.053	0.000	0.274

25th percentile of housing density, etc.) were used as factor levels. Metrics of the ICI that were used as dependent variables included the number of Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa, the percent composition of mayflies, other dipterans/non-insects, and tolerant taxa. IBI metrics used included the percent composition of omnivores, tolerant fishes, sensitive fishes, and insectivores. IBI scores and metrics from a subset of samples in the Cuyahoga Basin with drainage areas less than 100 mi<sup>2</sup> were also analyzed according to percent urban land use in a similar manner to examine for potential differences due to stream and watershed size. Because sample sizes varied widely in the subsets, multiple comparisons were made using Sheffé's procedure (Neter et al., 1991). An analysis of covariance model was constructed for Columbus area streams using quartiles of percent urban land use as factor levels, QHEI as a covariate, and IBI scores, percent composition of tolerant fishes, insectivores, and omnivores, the number of darter and sculpin species, and number of sensitive species as dependent variables. Multiple comparisons were made using Tukey's procedure (Neter et al., 1991).

Because Cuyahoga River basin streams are subject to a variety of multiple stressors, fish sampling sites were qualitatively classified by predominant impact type and regressed against percent urban land use cover (log<sub>10</sub> transformed) as a comparison to the results derived by using housing density and to determine the influence of impact type on the regression function. Impact types were defined as least impacted, estate (i.e., subwatersheds with large lot-size residential homes 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 alone and with CSOs, sites with evidence of impacts by legacy pollutants, or urbanization only. Regression coefficients from a subset of least-impacted, estate, and urban-only sites with drainage areas less than 100 mi<sup>2</sup> were compared to the same subset of sites for all drainage areas. Results of an ANOVA model using quartile distribution of percent land use as a factor level effect and IBI

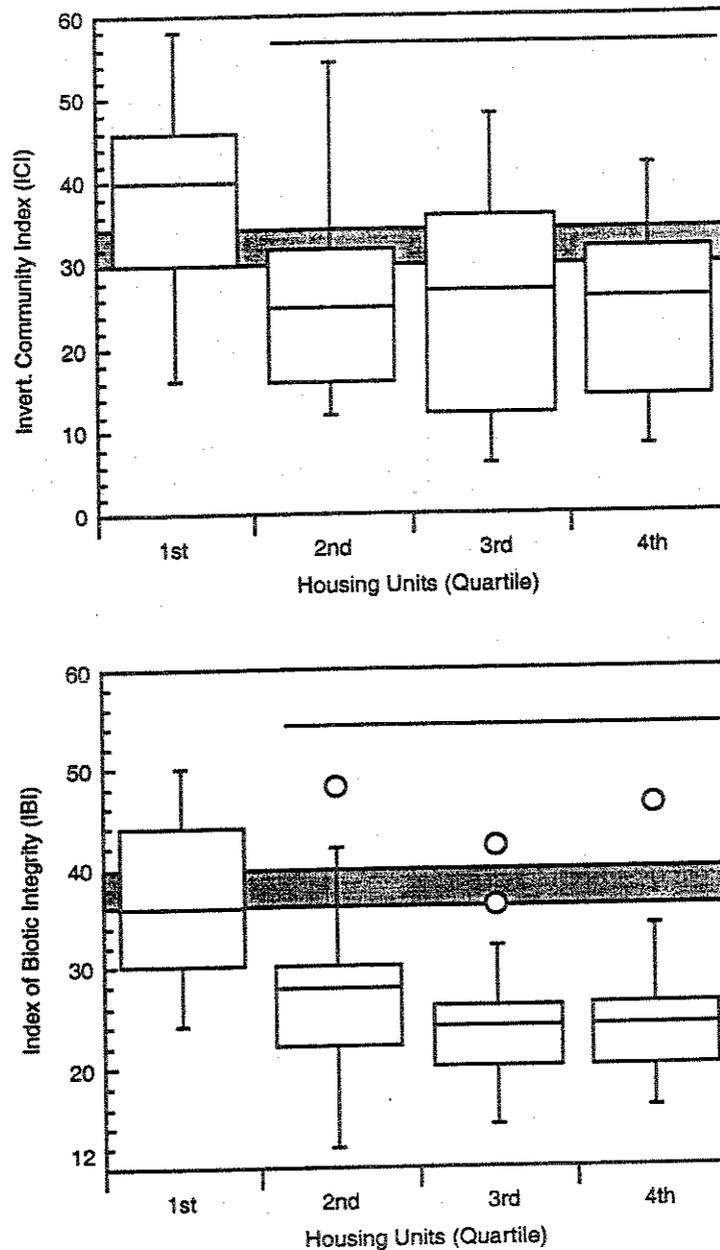
scores as independent variables were compared to those derived from the housing density model. Housing density, as an indicator of the degree of urbanization, was further evaluated by comparison with percent urban land use.

### Housing Density and Biological Performance

When paired with chemical water quality data, housing density explained approximately 27% and 59% of the variation in IBI and ICI scores in the Cuyahoga River basin (Table 1). Of the water quality variables tested, only nitrate+nitrite-nitrogen and ammonia-nitrogen explained a small, but significant proportion of the variation in IBI and ICI scores (≈3% and 1%, respectively). For all IBI and ICI scores, housing density accounted for 31% and 23% of the variation in scores. Multiple comparisons of factor levels based on quartile distribution of housing density identified a threshold level of urbanization, coinciding with 2.53 housing units per hectare, beyond which IBI or ICI scores will increasingly fail to attain the biological criteria for the warmwater habitat use designation (Figure 1).

Shifts within the macroinvertebrate community were also associated with a threshold level of urbanization (Figure 2). The number of EPT taxa were significantly higher at the lowest levels of urbanization. Conversely, the percent composition of pollution tolerant taxa collected from the artificial substrate samplers increased sharply at sites exceeding the twenty-fifth percentile of housing density. Similarly, the percent composition of other dipterans and non-insects increased with increasing urbanization. The percent composition of mayflies found on the artificial substrates did not change with increasing level of urbanization (Figure 2).

Shifts in the compositional metrics of the fish community were associated with the degree of urbanization in the Cuyahoga River basin (Table 2) and included an increase in the relative abundance of tolerant and omnivorous fish. The relative abundance of omnivorous fishes, however, tended to be highest at intermediate levels of urbanization, but differences were not statistically significant for the subset of streams with drainage areas less than 100 mi<sup>2</sup>. Insectivorous fishes were least abundant when housing density exceeded the seventy-fifth percentile threshold.



**Figure 1.** Distributions of Index of Biotic Integrity (IBI; lower) and Invertebrate Community Index (ICI; upper) scores from the Cuyahoga River basin plotted by quartiles of housing density upstream from sampling locations. The level of urbanization is given by quartiles of housing density per hectare of the subwatershed upstream from sampling locations. Horizontal lines spanning adjacent box plots indicate similar means. Levels of housing density per hectare corresponding to the 25th, 50th and 75th percentiles are 2.53, 4.45 and 7.26 units/ha, respectively. The shaded areas indicate the applicable biological criterion and the range of insignificant departure.

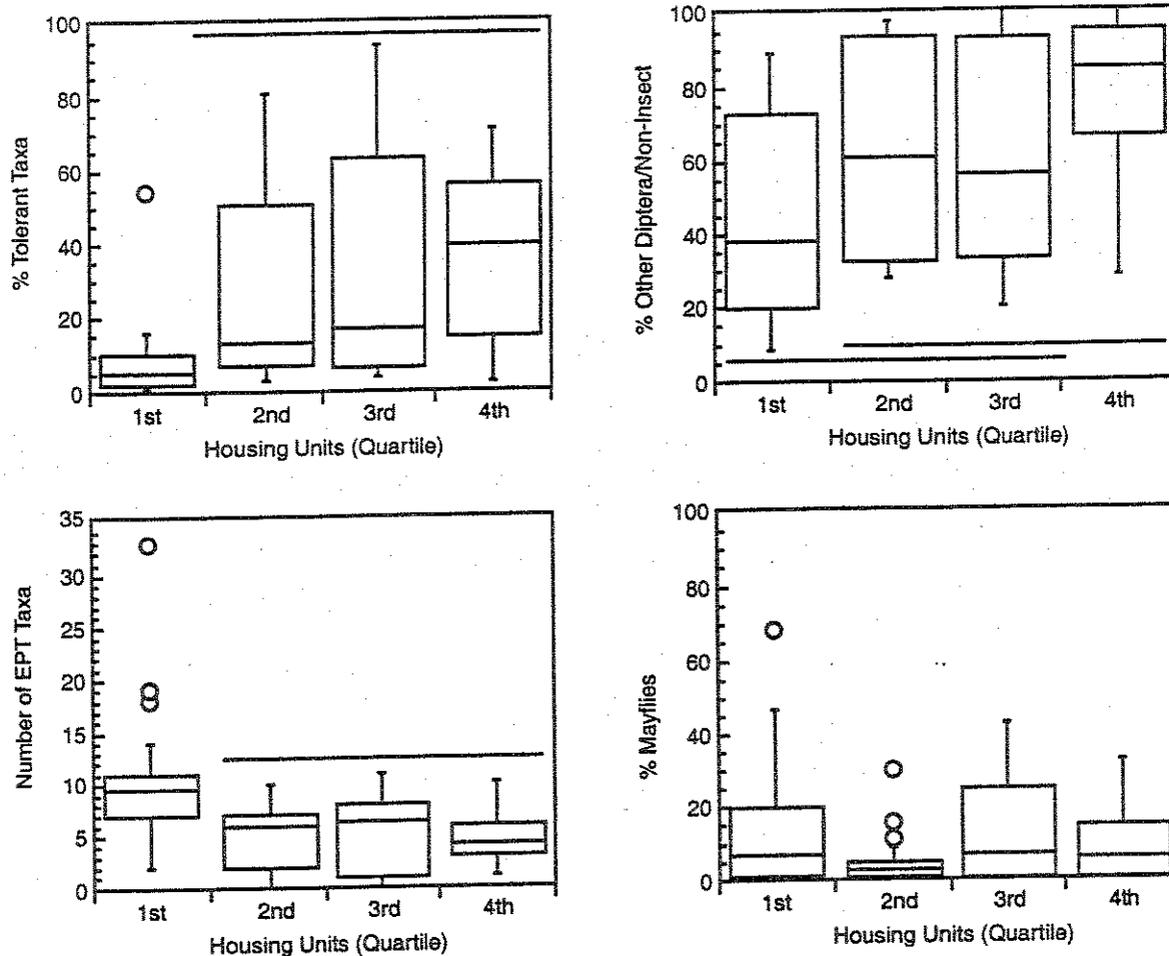


Figure 2. Performance of four invertebrate Community Index (ICI) metrics in relation to housing density for the Cuyahoga River basin. The level of urbanization is given by quartiles of housing density per hectare of the subwatershed upstream from sampling locations. Horizontal lines spanning adjacent box plots indicate similar means. Levels of housing density per hectare corresponding to the 25th, 50th and 75th percentile are 2.53, 4.45 and 7.26 units/ha, respectively.

### Urban Land Use and Biological Performance

The percentage of urban land use cover explained 26.7% of the variation in IBI scores in the Cuyahoga River basin, similar to that explained by housing density. When classified by quartile level of percent urban land use cover, the mean of IBI scores in the first quartile was significantly higher than those in the third or fourth quartile (Figure 3). However, classification by percent urban land use cover showed a more continuous decrease in mean IBI scores with an increasing level of urbanization than did housing density. Multiple comparisons of component IBI metrics classified by level of urban land use cover showed similar average responses to increasing urbanization as did classification by housing density (Table 2). However, intraquartile variation of the metric responses was greater among urban land use cover than for housing density, leading to fewer significant differences between means and reflecting the more continuous decrease in mean IBI response with respect to percent urban land use cover.

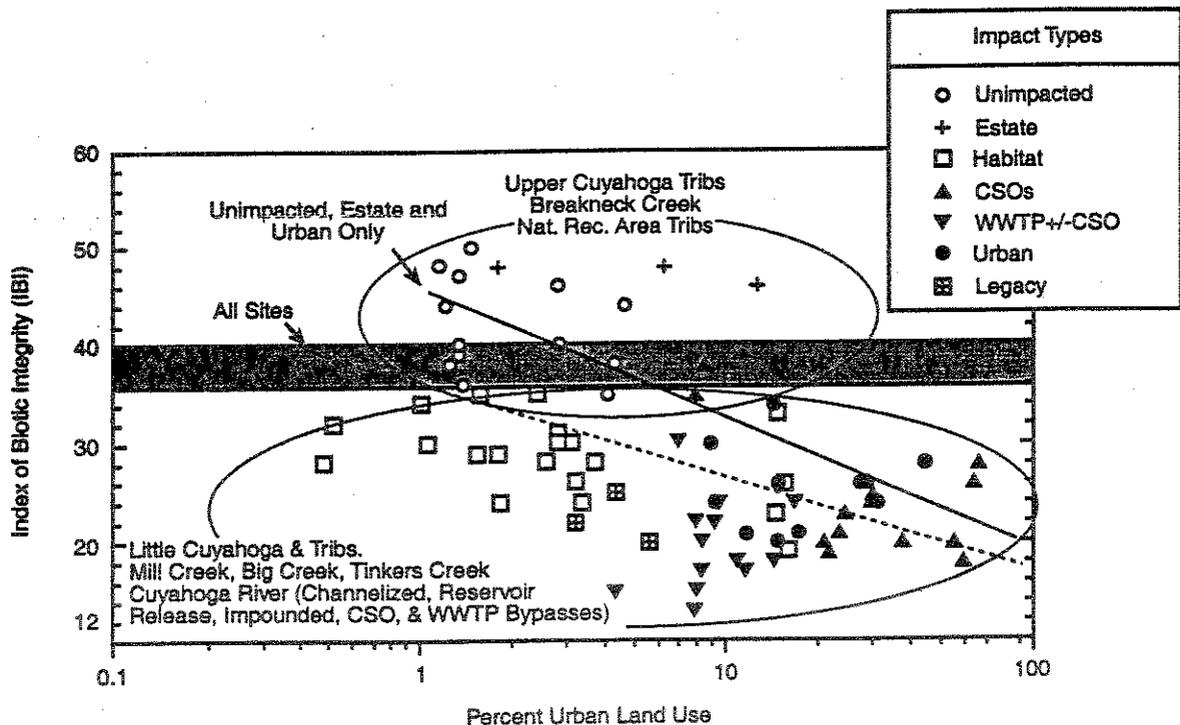
Significant differences in mean IBI scores between the levels of urban land use were also found for Columbus area streams (Figure 3). Mean IBI scores from streams with less than 3% urban land use were significantly higher than those with greater than 33% urban land use (Figure 3). Shifts in the composition of the fish community associated with increasing percent urbanization included the loss of darters, sculpins, and other pollution and habitat sensitive species, decreased abundance of insectivores, and an increase in the proportion of tolerant fishes (Table 3).

### 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). The threshold levels in our study of approximately 8% and 33% urban land use cover for the Cuyahoga River basin and Columbus area streams, as identified by analysis of variance, is in general agreement with the studies reviewed by Schuler (1994).

**Table 2.** Factor Level Means and Sheffe' Groupings for Selected Fish Community IBI Metrics Sampled in the Cuyahoga River basin in Relation to Urban Land Use Indicators. Means Sharing a Common Letter are Not Significantly Different. The Asterisks Denote where Significant Differences Between Groups were Not Detected in Multiple Comparisons for the Percent Tolerant Group from all Sites, and for the Number of Sensitive Species in Streams Less than 100 M<sup>2</sup>, The Overall F Tests Indicated a Significant (P < 0.05) Linear Relationship.

Urban Indicator (Quantile)	N	Number of Sensitive Species	Percent as Insectivores	Percent as Tolerant	Percent as Omnivores
All sites - Housing Units per Hectare					
1st	22	A	3.0	A	49.9
2nd	21	AB	2.1	A	39.2
3rd	19	CB	1.4	A	27.4
4th	21	C	0.4		10.5
All sites - Percent Urban Land Use					
1st	22	A	2.5	A	49.5
2nd	21	A	2.2	A	41.4
3rd	19	AB	1.4	B	18.6
4th	21	B	0.7	B	14.6
Drainage Area < 100 m <sup>2</sup> - Percent Urban Land Use					
1st	12	A*	2.5	A	44.8
2nd	11	A	2.0	A	40.8
3rd	9	A	0.8	B	13.2
4th	17	A	0.6	B	10.5



**Figure 3.** Distribution of Index of Biotic Integrity (IBI) values plotted by quartiles of percent urban land use cover upstream from sampling locations for all sites in the Cuyahoga River basin, Cuyahoga basin sites with drainage areas less than 100 m<sup>2</sup>, and Columbus area streams. The shaded areas indicate the applicable biological criterion and the range of insignificant departure.

Table 3. Factor Level Means and Tukey Groupings for Selected Components of Fish Communities Sampled in Columbus Area Streams. Means Sharing a Common Letter are not Significantly Different. Samples Sizes in Ascending Order by Quartile are 20, 18, 20 and 20; Two Cases in the Second Quartile had Missing Values Due to No Fish.

Percent Urban by Quartile	Number of Sensitive Species	Number of Darters/Sculpins	Percent as Insectivores	Percent as Tolerant	Percent as Omnivores
1st	A 3.3	A 3.1	A 40.8	A 54.6	A 13.0
2nd	A 3.8	AB 2.2	AB 32.0	A 56.3	A 13.9
3rd	B 0.9	CB 1.1	B 17.0	B 82.7	A 10.5
4th	B 0.6	C 0.6	B 19.7	B 78.3	A 5.1

However, the threshold level identified by regression for the Cuyahoga River basin was influenced by the presence of other stressors (e.g., CSOs, point sources, legacy pollutants). The elimination of those sites impacted by other stressors from the regression resulted in an increased threshold of urbanization (Figure 4). Although other stressors acted as covariates in a sense, these were not amenable to an analysis of covariance because each occurred in relatively discrete groupings along the continuum of increasing urbanization. Analysis of variance was better able to identify a threshold level by contrasting discrete ranges (i.e., quartiles) along the entire range of increasing urbanization (Figure 3).

Similar patterns in the effect of increasing urbanization on biological communities were evident for both the Cuyahoga River basin and Columbus area streams. Detectable differences in the number of sensitive fish species in Columbus area streams occurred at lower levels of urbanization than did IBI scores, illustrating the role of sensitive species as sentinels of urban effects. Sensitive fishes are rare in the Cuyahoga River basin as a whole due to historic, complex, and widespread anthropogenic stressors, yielding less response and higher variation associated with interquartile means compared to the Columbus area streams. However, the number of EPT taxa, a sensitive macroinvertebrate guild, similarly acted as sentinels of urbanization given that EPT abundance was significantly reduced at relatively low levels of urbanization. The abundance of mayflies, showing little correlation with the level of urbanization, did not respond in a manner similar to the number of EPT taxa. While this may reflect the difference in collection technique as percent mayflies are based on the data from artificial substrates, whereas EPT taxa are based on data collected from natural substrates, it may also be due to differing sensitivities within the EPT guild. This result, in combination with the response of the fish community, implies that substrate degradation is a major factor which limits aquatic communities at relatively low levels of urbanization.

The relative abundance of omnivores tended to be highest at intermediate levels of urbanization when all sites in the Cuyahoga Basin were included. This response was due in part to enrichment by wastewater treatment plant discharges and CSOs discharging to the Cuyahoga River mainstem. No differences were detected for the subset of streams with drainage areas less than 100 mi<sup>2</sup>, nor in the

Columbus area streams. However, the relative abundance of insectivores was negatively correlated with increasing urbanization in both study areas, suggesting a disruption within the aquatic food web. Conversely, the proportion of tolerant fishes was positively correlated with increasing urbanization. The high proportion of tolerant fishes at the highest levels of urbanization is indicative of both degraded habitat and water quality, specifically toxicity and organic enrichment. Collectively, these changes in biological communities suggest a continuous negative response to increasing urbanization starting with the loss of sensitive fish and macroinvertebrate species at comparatively low levels of urban development (<5% urban land use) due to substrate degradation, disruption within the aquatic food web at intermediate levels of development, and a response to toxicity, organic enrichment, or both at higher levels of development (>15% urban land use).

Overlaying impact types with percent urban land use (Figure 4) demonstrates that the negative effects of urbanization and associated cofactors (e.g., imperviousness, polluted runoff, altered hydrology) may be partially offset by beneficial land use practices. Biological performance at sites impacted by estate-type residential developments remained comparatively intact and attained the ecoregion biocriteria even at relatively high levels of urbanization (up to 15%). The best performing sites within those watersheds also had relatively intact stream habitat and well-vegetated, wider riparian buffers. Conversely, sites with increasingly modified habitats performed poorly and failed to attain the biocriteria regardless of the degree of urbanization. The most degraded sites were associated with either poorly treated sewage, CSOs, and/or a high degree of urbanization. These findings agree with those of Steedman (1988) who demonstrated a co-relationship between riparian zone quality and land use in terms of how each affected the fish communities of Toronto area streams. Horner et al. (1997) found the steepest rates of decline in biological functioning (in terms of the B-IBI; Kerans and Karr 1992) to occur with increases in impervious cover of as little as 1-6% in streams flowing into Puget Sound, Washington. Exceptions occurred where urban land use was mitigated by extensive riparian protection or other management interventions, but these factors ceased to be effective above 45% as impervious land cover.

Unlike the Cuyahoga River basin, the Columbus area streams were not subject to extensive CSO impacts and

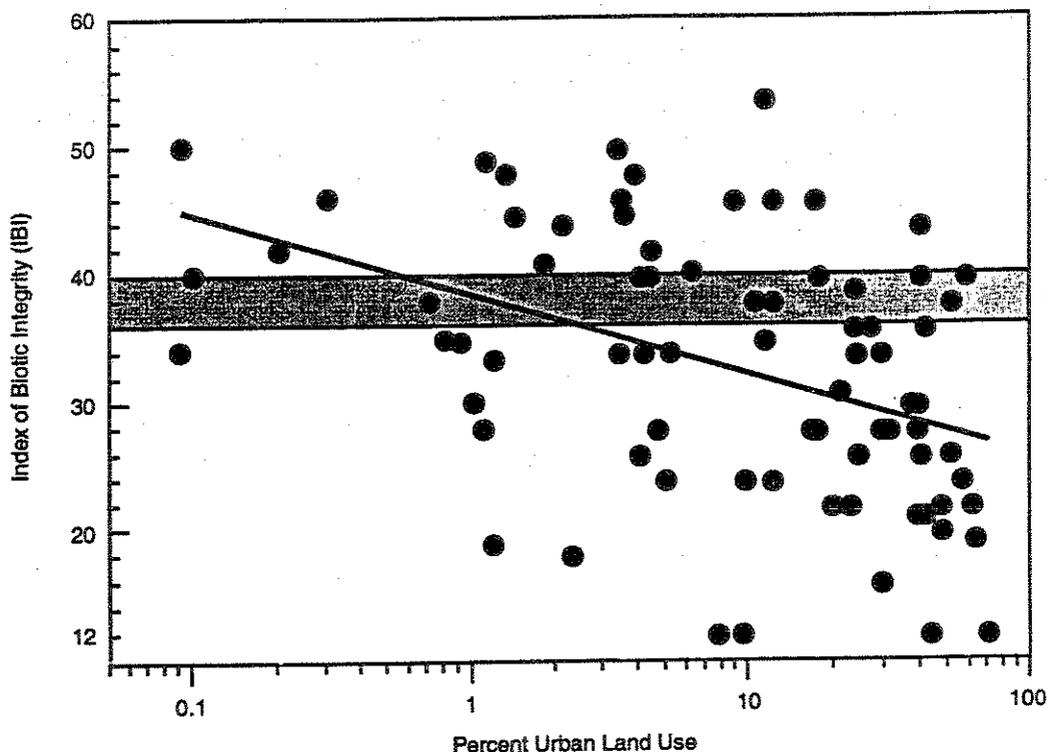


Figure 4. Index of Biotic Integrity (IBI) scores from sites sampled in the Cuyahoga River basin plotted by stressor group (symbols) against percent of urban land use for sites draining less than 100 mi<sup>2</sup>. The fitted regression lines are for all points and those lacking stressors other than urbanization. The shaded areas indicate the applicable biological criterion and the range of insignificant departure.

industrial legacy pollutants were virtually absent. Consequently, the threshold level of urbanization precluding attainment of the biological criteria was higher for the Columbus area streams (Figure 5), results which are analogous to that for sites influenced by the estate impact type in the Cuyahoga River basin. In fact there were a few sites with urban land use as high as 50% which fully attained the ecoregional biocriterion. This suggests that the type of urban development strongly influences the attainability of aquatic life uses within a watershed. Furthermore, factors such as impermeability and urbanization alone do not automatically disqualify streams from meeting designated uses based on biological criteria.

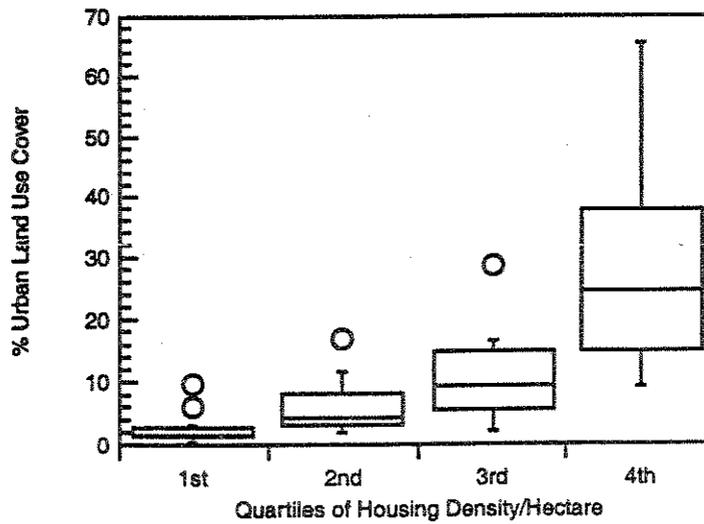
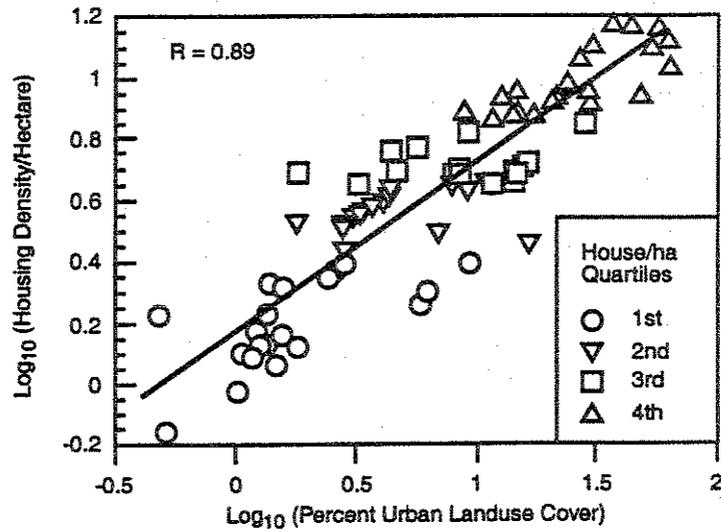
Although housing density and percent urban land use demonstrated a strong linear relationship (Figure 6), each urban indicator showed somewhat differing results. The percent of urban land use indicator, which is a more precise measure of urbanization and imperviousness, was negatively correlated with biological community performance. By comparison, the housing density indicator showed a discrete threshold between the lowest quartile and all others. The principal difference is that high-quality sites were more frequently associated with the second quartile of percent urban land use than for housing density, reflecting good IBI scores from relatively urbanized subwatersheds containing large residential lot sizes and more green space. Also, urban land use within successive quartiles of housing density apparently becomes increasingly mixed as inferred by increasing interquartile varia-

tion in percent urban land use (Figure 6). Higher levels of housing density coincided with increased industrial, commercial, and transportation related land uses. The difference in results by urban indicator underscores the importance of maintaining natural features within a watershed including instream habitat, vegetated riparian buffers of adequate width, and green space in addition to minimizing and controlling chemical impacts from wastewater treatment plants, CSOs, and other sources.

### Implications for Use Attainability

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, lower uses may be established and assigned on a case-by-case basis. Federal water quality regulations (40CFR Part 131.10[g]) generally specify three criteria for setting designated uses below "fishable/swimmable" standards as follows: 1) imposition of the criteria for a higher use would result in widespread, adverse socioeconomic impacts; 2) the criteria are not attainable due to natural background conditions; or 3) the criteria are not attainable due to irretrievable, anthropogenic impacts.

Compliance with the aquatic life uses defined in the Ohio WQS are determined primarily by the biological criteria (OAC 3745-1-07) which are stratified according to designated use, ecoregion, and stream size. As such this repre-



**Figure 5.** Index of Biotic integrity (IBI) scores from sites sampled in Columbus area streams against percent of urban land use. The fitted regression lines are for all points and those lacking stressors other than urbanization. The shaded areas indicate the applicable biological criterion and the range of insignificant departure.

Quality Gradient of Aquatic Life Uses and Narrative Descriptions of Biological Community Condition

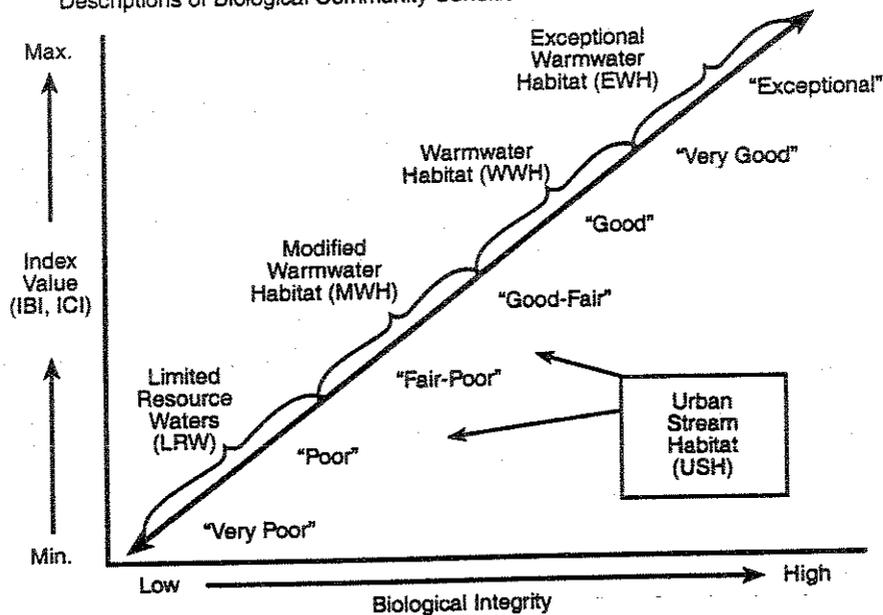


Figure 6. Relationship between housing density and percent of urban land use cover for subwatersheds upstream from fish sampling locations in the Cuyahoga River basin. The upper plot shows housing density as a function of percent urban land use cover. The lower plot shows distributions of percent urban land use cover within quartile levels of housing density per hectare.

sents a stratified system of uses and criteria that occur along a gradient of biological integrity as expressed by the biological indices which comprise the numerical biological criteria (Figure 7). For most Ohio streams the "default" expectation is attainment of the warmwater habitat (WWH) use provided the physical habitat is relatively intact and no extensive alterations are evident. Obvious anthropogenic alterations to small urban streams such as culverting, relocation, bank and channel stabilization with artificial structures, and extensive channelization are relatively easy to identify and assess. In such cases, the Limited Resource Waters (LRW) use designation is assigned which means that the minimum level of protection (i.e., prevention of lethality) afforded by the Ohio WQS applies. The difficulty is with small urban streams that exhibit adequate habitat (as defined by the QHEI score), but which fail to attain the WWH biocriteria. The recent finding that no urban headwater stream sites in the Ohio EPA database attain the WWH biocriteria (Yoder and Rankin 1997) only serves to further the notion that the degree of watershed urbanization can preclude the WWH use regardless of the site-specific habitat quality.

Recently, the imperviousness of the watershed has been used as an indicator which is correlated with use attainability. If the frequently cited threshold of 25% impermeability is used, streams in watersheds with greater than this value would be unlikely to ever attain a beneficial use regardless of site and reach factors. The results of our study suggest that there is a threshold of watershed urbanization beyond which attainment of the WWH use is increasingly unlikely. However, this threshold is different among

watersheds as evidenced by the results from the Cuyahoga Basin and Columbus area streams. Co-occurring 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 greatly influence the biological quality in the receiving streams.

While the development of indicators of watershed urbanization has merit from a management and decision-making standpoint, there are simply too many other factors, some of which are controllable and amenable to remediation, to use it as a sole determinant for aquatic life attainability. We suggest that the co-factors in addition to urban watershed indicators be better developed and tested using datasets from broader geographic areas and spanning the extremes of the urbanization gradient. One goal should be to develop, if appropriate, an urban stream habitat designation that would fit along the already existing hierarchy of aquatic life use designations in Ohio (Figure 7). We have indicated on Figure 7 where the biological criteria for this potential new designation might occur compared to the already existing hierarchy of aquatic life uses in the Ohio WQS. However, placing it on the existing quality gradient will require substantial calibration and validation with existing datasets. Having this use would satisfy the desire to afford streams with the maximum protection practicable, while recognizing the inherent limitations that urbanization imposes on stream quality.

In the meantime, simplistic regulatory and management approaches should be limited, particularly in those water-

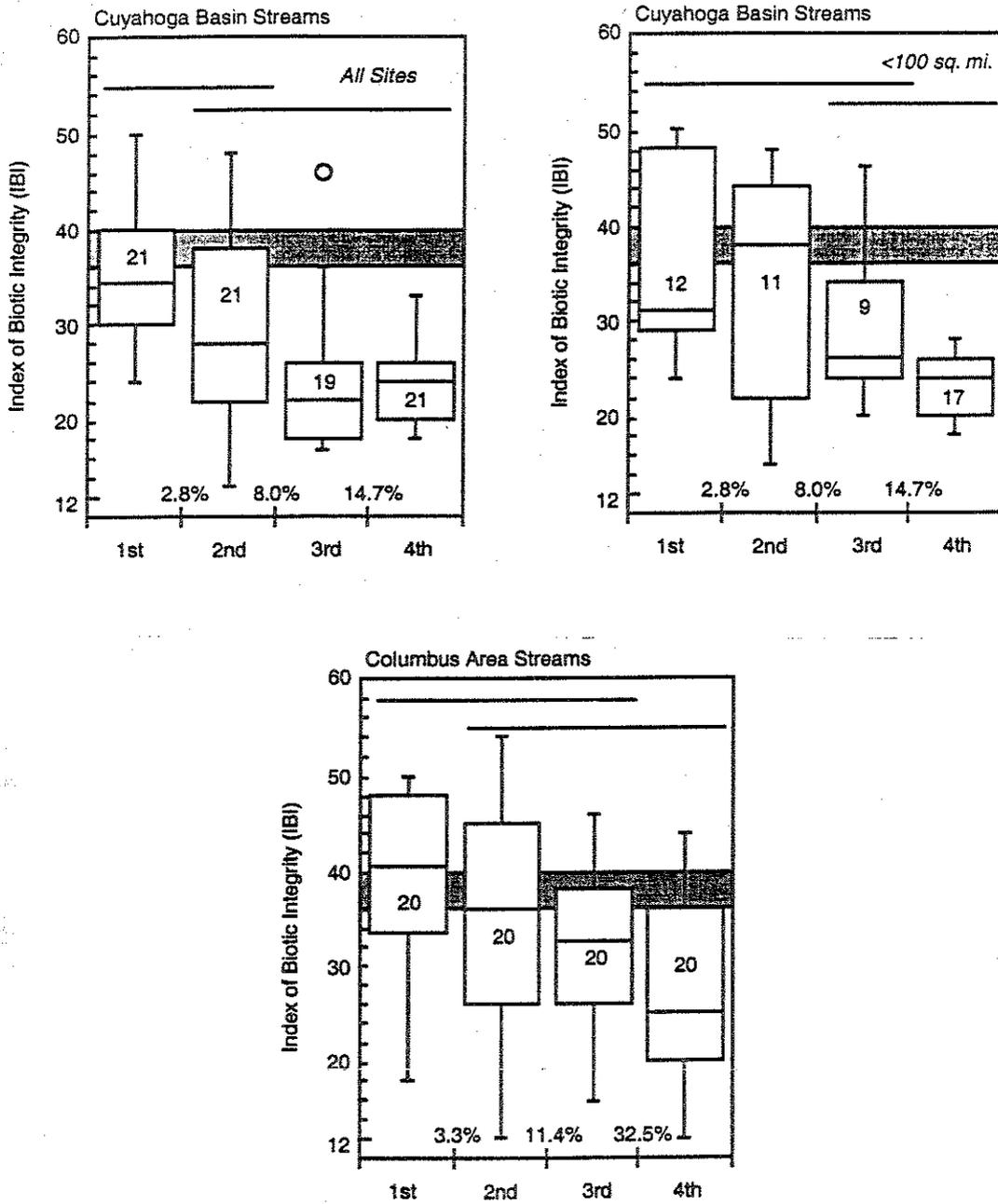


Figure 7. Index of Biotic Integrity (IBI) scores from sites sampled in the Cuyahoga River basin plotted by stressor group (symbols) against percent of urban land use for sites draining less than 100 mi<sup>2</sup>. The fitted regression lines are for all points and those lacking stressors other than urbanization. The shaded areas indicate the applicable biological criterion and the range of insignificant departure.

sheds where uncertainty about the attainability of CWA goal uses (i.e., WWH and higher) exists. For example, initial approaches such as the nine minimum controls for CSOs seem reasonable. However, proceeding beyond these requirements with long-term control plans should be done cautiously and with the aid of sufficiently robust before-and-after biological and water quality assessments.

The results of our study also point out the benefits of a regular, sustained, and robust state monitoring and assessment effort (see also Yoder and Rankin 1998). Dealing with complex water quality management issues such as CSOs, stormwater, and TMDLs in urban watersheds would be difficult at best within the confines of the traditional administrative approach to water quality management. Steedman (1988) described multimetric biological indices like the IBI and ICI as being based on simple, definable ecological relationships which is quantitative as an ordinal, if not linear, measure and which responds in an intuitively correct manner to known environmental gradients. Further, when incorporated with mapping, monitoring, and modeling information, such an approach has been shown to be valuable in determining management and restoration requirements for warmwater streams (Steedman 1988; Bennet et al., 1993). The value added by a robust bioassessment and tiered use designation framework coupled with sufficiently detailed and accurate GIS information was amply demonstrated herein.

## References

- Bennet, M.R., J.W. Kleene, and V.O. Shanholtz. 1993. Total Maximum Daily Load Nonpoint Source Allocation Pilot Project. File Report, Dept. of Agricultural Engineering, Blacksburg, VA. 44 pp.
- DeShon, J.D. 1995. Development and application of the invertebrate community index (ICI), pp. 217-243. in W.S. Davis and T. Simon (eds.). *Biological Assessment and Criteria: Tools for Risk-based Planning and Decision Making*. Lewis Publishers, Boca Raton, FL.
- Horner, R.R., D.B. Booth, A. Azous, and C.W. May. 1997. Watershed determinants of ecosystem functioning, pp. 251-274. In C. Roesner (ed.). *Effects of Watershed Development and Management on Aquatic Ecosystems*, American Society of Civil Engineers, New York, NY.
- Karr, J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6(6): 21-27.
- Karr, J.R., K.D. Fausch, P.L. Angermier, P.R. Yant, and I.J. Schlosser. 1986. Assessing biological integrity in running waters: a method and its rationale. *Illinois Natural History Survey Special Publication* 5: 28 pp.
- Kerans, B.L., and J.R. Karr. 1992. An evaluation of invertebrate attributes and a benthic index of biotic integrity for Tennessee Valley rivers, Proc. 1991 Midwest Poll. Biol. Conf., EPA 905/R-92/003.
- Neter J., W. Wasserman, and M.H. Kutner 1991. *Applied Linear Statistical Models: Regression, Analysis of Variance, and Experimental Designs*. Irwin, Homewood, IL.
- Ohio Environmental Protection Agency. 1997. *Ohio Water Resource Inventory, Volume I, Summary, Status, and Trends*. Rankin, E.T., C.O. Yoder, and D.A. Mishne, (eds.), Ohio EPA Tech. Bull. MAS/1996-10-3-1, Division of Surface Water, Columbus, OH. 190 pp.
- Ohio Environmental Protection Agency. 1990. *Ohio's nonpoint source pollution assessment*. Division of Water Quality Planning and Assessment. Columbus, OH.
- Ohio Environmental Protection Agency. 1989a. *Biological Criteria for the Protection of Aquatic Life. Volume III: Standardized Biological Field Sampling and Laboratory Methods for Assessing Fish and Macroinvertebrate Communities*, Division of Water Quality Monitoring and Assessment, Columbus, OH.
- Ohio Environmental Protection Agency. 1989b. *Addendum to Biological Criteria for the Protection of Aquatic Life. Volume II: Users Manual for Biological Field Assessment of Ohio Surface Waters*, Division of Water Quality Planning and Assessment, Surface Water Section, Columbus, OH.
- Ohio Environmental Protection Agency. 1987a. *Biological Criteria for the Protection of Aquatic Life: Volume I. The Role of Biological Data in Water Quality Assessment*. Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, OH.
- Ohio Environmental Protection Agency. 1987b. *Biological Criteria for the Protection of Aquatic Life: Volume II. Users Manual for Biological Field Assessment of Ohio Surface Waters*, Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, OH.
- Rankin, E. T. 1995. The Use of Habitat Assessments in Water Resource Management Programs, pp. 181-208. in W. Davis and T. Simon (eds.). *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*. Lewis Publishers, Boca Raton, FL.
- Rankin, E.T. 1989. *The Qualitative Habitat Evaluation Index (Qhei), Rationale, Methods, and Application*, Ohio EPA, Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.
- Schuler, T.R. 1994. The Importance of Imperviousness. *Watershed Protection Techniques* 1: 100 - 111.
- Steedman, R.J. 1988. Modification and Assessment of an Index of Biotic Integrity to Quantify Stream Quality in Southern Ontario. *Canadian Journal of Fisheries and Aquatic Sciences* 45: 492 - 501.
- U.S. Bureau of Census. 1990. *1990 Census of Population*. Government Printing Office, Washington, D.C.
- Yoder, C.O. 1995. Incorporating Ecological Concepts and Biological Criteria in the Assessment and Management of Urban Nonpoint Source Pollution, pp. 183-197. in D. Murray (ed.). *National Conference on Urban Run-off Management: Enhancing Urban Watershed Man-*

agement at the Local, County, and State Levels. EPA/625/R-95/003.

Yoder, C.O. and E.T. Rankin. 1998. The Role of Biological Indicators in a State Water Quality Management Process. *J. Env. Mon. Assess.* 51(1-2): 61-88.

Yoder, C.O. and E.T. Rankin. 1997. Assessing the Condition and Status of Aquatic Life Designated Uses in Urban and Suburban Watersheds, pp. 201-227. *in*

Roesner, L.A. (ed.). *Effects of Watershed Development and Management on Aquatic Ecosystems*, American Society of Civil Engineers, New York, NY.

Yoder, C.O. and E.T. Rankin. 1995a. Biological Criteria Program Development and Implementation in Ohio, pp. 109-144. *in* W. Davis and T. Simon (eds.). *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*. Lewis Publishers, Boca Raton, FL.