Biological Assessment and Criteria

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CHAPTER 15

Development and Application of the Invertebrate Community Index (ICI)

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1.0 INTRODUCTION

Aquatic macroinvertebrates have been used widely as an indicator group for many years in pollution studies involving flowing waters. Cairns and Pratt (1993) provide a detailed account of the current and historical use of macroinvertebrates in freshwater biomonitoring. At the Ohio EPA, macroinvertebrate communities have been collected and analyzed since the agency's inception in 1973 in an effort to provide biological data to be used in the water quality monitoring and assessment process. At least one collection has been made from over 2300 locations displaying a wide variety of water quality conditions within the state. Early assessments of macroinvertebrate data depended on the individual expertise of the biologists who collected the data. With the aid of tools such as the Shannon–Wiener Diversity Index (Shannon and Wiener 1949; Wilhm 1970) and a healthy dose of "best professional judgment," numerous narrative evaluations of water quality problems were made over the years. However, the inherently subjective nature of such evaluations was often considered a major liability, especially in complicated environmental issues involving permit holders and litigation. As a result, a more objective means to assess macroinvertebrate data was sought.

As an offshoot of the 1983–84 Ohio Stream Regionalization Project, a cooperative pilot venture with USEPA/ERL-Corvallis (Whittier et al. 1987), methods were researched to develop a multimetric macroinvertebrate index patterned after the concept of the Index of Biotic Integrity (IBI) developed for fish community assemblages by Karr (1981) and refined by Fausch et al. (1984). The result was the Invertebrate Community Index (ICI), which is now used as the principal assessment tool by Ohio EPA macroinvertebrate biologists for monitoring and assessment activities in all free-flowing rivers and streams in Ohio. In 1987, numeric ecoregional biological criteria were developed and codified in the Ohio Water Quality Standards by the Ohio EPA with the ICI being one of three biological indices applied (Yoder and Rankin, Chapter 9).

2.0 METHODS SUMMARY

The primary sampling gear used by the Ohio EPA for the quantitative collection of macroinvertebrates in streams and rivers is the modified multiple-plate artificial substrate sampler (Hester and Dendy 1962). The sampler is constructed of ⅛ in. (3 mm) tempered hardboard cut into 3-in.^{2} (7.5-cm^2) plates and 1-in.^{2} (2.5-cm^2) spacers. A total of eight plates and twelve spacers are used for each sampler. The plates and spacers are placed on a 3 in. (7.5 cm) long, ¼ in. (6 mm) diameter eyebolt so that there are three single spaces, three double spaces, and one triple space between the plates. The total surface area of the sampler, excluding the eyebolt, approximates 1 ft^2 (roughly 0.1 m^2). A sampling unit consists of a composite cluster of five substrates tied to a construction block that is colonized in-stream for a 6-week period beginning no earlier than June 15 and ending no later than September 30. Detailed descriptions of the placement, collection, and processing of the artificial substrates are available (Ohio EPA 1989a). In addition to the artificial substrate sample, routine monitoring also includes a qualitative collection of macroinvertebrates that inhabit the natural substrates at the sampling location. All available habitat types are sampled and voucher specimens are retained for laboratory identification. More specific information for the collection of this sample has also been detailed (Ohio EPA 1989a). For the purpose of generating an ICI value, both a quantitative and qualitative sample must be collected at a sampling location.

The use of artificial substrates for monitoring purposes has a number of advantages. According to Rosenberg and Resh (1982), the major advantages in using artificial substrates in general are that they:

- allow collection of data from locations that cannot be sampled effectively by other means,
- permit standardized sampling,
- reduce variability compared with other types of sampling,
- require less operator skill than other methods,
- are convenient to use, and
- permit nondestructive sampling of an environment.

The authors also listed a number of disadvantages. These include:

- incompletely known colonization dynamics,
- long exposure times to obtain a sample,
- loss of fauna on retrieval,
- unforeseen losses of artificial substrates, and
- inconvenient to use and logistically awkward.

Generally, however, the authors concluded that these problems could be minimized by adhering to strict guidelines concerning sampler placement and collection and data analysis and interpretation.

Klemm et al. (1990) specifically focused on the use of modified Hester-Dendy multiple-plate artificial substrate samplers and listed the following advantages:

- are excellent for water quality monitoring,
- provide uniform substrate type,
- allow for a high level of precision,
- provide habitats of known area for a known time at a known depth.

The authors noted that colonization of macroinvertebrates should be relatively equal in similar habitats and should reflect the capacity of the water to support aquatic life. Although acknowledging that these samplers may exclude certain mollusks or worms, they concluded that a sufficient diversity of benthic fauna is collected to be useful in assessing water quality.

Thus, by exploiting the strengths and yet recognizing and controlling the liabilities, the use of multiple-plate artificial substrate samplers serves as an important component of macroinvertebrate sampling in Ohio and helps to ensure that a standardized approach to monitoring a wide variety of sites is maintained. When selecting any sampling method, it is imperative to have a clear definition of the objectives of the sampling as well as an understanding of the potential shortcomings of using that method for the collection of macroinvertebrates.

A composited set of five artificial substrate samplers of eight plates each has been used by the Ohio EPA in collecting macroinvertebrate samples since 1973. At this level of effort, it has been found that consistent, reproducible ICI values can be scored despite the collections of often highly variable numbers of individual organisms. The latter is a result of the tendency of macroinvertebrate populations to have naturally clumped (i.e., negative binomial) distributions in the environment. Results of analyzing replicate composites of five artificial substrates have shown that variability among calculated ICI values is at an acceptable level. Details of that analysis can be found elsewhere in this chapter. The reliability of the sampling unit not only depends on the fact that colonization surface areas are standard, but equally

Table 1. Current Taxonomic Keys and the Level of Taxonomy Routinely Used by the Ohio EPA for Various Macroinvertebrate Taxonomic Classifications

- Porifera: Species (Pennak 1989)
- Coelenterata: Genus (Pennak 1989)
- Platyhelminthes: Class (Pennak 1989)
- Nematomorpha: Phylum/genus (Pennak 1989)
- Ectoprocta: Genus/species (Thorp and Covich 1991)
- Entoprocta: Species (Thorp and Covich 1991)
- Annelida
  - Oligochaeta: Class (Pennak 1989)
  - Hirudinea: Species (Klemm 1982)
- Arthropoda
  - Crustacea
    - Isopoda: Genus (Pennak 1989)
    - Amphipoda: Genus (Pennak 1989)
      - *Gammarus*: Species (Holsinger 1972)
    - Decapoda
      - *Cambarus* and *Fallicambarus*: Species (Jezerinac and Thoma 1984)
      - *Orconectes* and *Procambarus*: Species (Jezerinac 1978)
      - *Palaemonetes*: Species (Pennak 1989)
  - Arachnoidea: Class (Pennak 1989)
  - Insecta
    - Ephemeroptera: Genus (Edmunds et al. 1976, Merritt and Cummins 1984)
      - Baetidae: Genus/species (Morihara and McCafferty 1979, McCafferty and Waltz 1990)
      - Heptageniidae
        - *Stenonema*: Species (Bednarik and McCafferty 1979)
      - Ephemerellidae
        - *Dannella*: Species (Allen and Edmunds 1962)
        - *Ephemerella*: Species (Allen and Edmunds 1965)
        - *Eurylophella*: Species (Allen and Edmunds 1963a)
        - *Serratella*: Species (Allen and Edmunds 1963b)
      - Baetiscidae
        - *Baetisca*: Species (Burks 1953)
      - Ephemeroidea: Species (McCafferty 1975)
    - Odonata: Family/genus (Merritt and Cummins 1984)
      - Anisoptera: Genus/species (Needham and Westfall 1955, Walker 1958, Walker and Corbett 1975)
    - Plecoptera: Genus (Stewart and Stark 1988)
      - Perlidae
        - *Acroneuria*: Species (Hitchcock 1974)
        - *Paragnetina*: Species (Hitchcock 1974)
        - *Perlinella*: Species (Kondratieff et al. 1988)
      - Perlodidae: Species (Hitchcock 1974)
    - Hemiptera: Genus (Hilsenhoff 1982b, Merritt and Cummins 1984)
    - Megaloptera: Genus (Merritt and Cummins 1984)
      - *Nigronia*: Species (Neunzig 1966)
    - Neuroptera: Genus (Merritt and Cummins 1984)
    - Trichoptera: Genus (Wiggins 1977, Merritt and Cummins 1984)
      - Philopotamidae: Species (Ross 1944)
      - Hydropsychidae
        - *Hydropsyche*: Species (Schuster and Etnier 1978)
      - Rhyacophilidae
        - *Rhyacophila*: Species (Flint 1962)
      - Leptoceridae
        - *Ceraclea*: Species (Resh 1976)
        - *Nectopsyche*: Species (Haddock 1977)
    - Lepidoptera: Genus (Merritt and Cummins 1984)
    - Coleoptera: Genus (Hilsenhoff 1982b, Merritt and Cummins 1984)
      - Dryopoidea: Genus/species (Brown 1972)
    - Diptera: Family/genus (Merritt and Cummins 1984)
      - Ceratopogonidae
        - *Atricopogon*: Species (Johannsen 1935)
      - Chironomidae: Genus/species groups (Wiederholm 1983)
        - *Ablabesmyia*: Species (Roback 1985)
        - *Labrundinia*: Species (Roback 1987)
        - *Tanypus*: Species (Roback 1977)
        - *Corynoneura*: Species (Simpson and Bode 1980, Bolton In Prep.)
        - *Eukiefferiella* and *Tvetenia*: Species groups (Bode 1983)
        - *Nanocladius*: Species (Saether 1977, Simpson and Bode 1980, Bolton In Prep.)
        - *Parakiefferiella*: Species (Bolton In Prep.)
        - *Rheocricotopus*: Species (Saether 1985)
        - *Thienemanniella*: Species (Simpson and Bode 1980, Bolton In Prep.)
          - *Chironomus*: Species groups (Oliver and Roussel 1983)
        - *Dicrotendipes*: Species (Epler 1987)
        - *Endochironomus* and *Tribelos*: Species (Grodhaus 1987)
        - *Parachironomus*: Species (Simpson and Bode 1980)
        - *Paracladopelma* and *Saetheria*: Species (Jackson 1977)
        - *Polypedilum*: Species groups/species (Maschwitz 1976, Bolton In Prep.)
      - Tanytarsini: Genus/species groups/species (Simpson and Bode 1980, Bolton In Prep.)
      - Muscidae: Species (Johannsen 1935)
- Mollusca
  - Gastropoda: Genus/species (Burch 1982)
  - Pelecypoda
    - Sphaeriidae: Genus (Burch 1982)
    - Unionidae: Species (Waters 1993)

important are the actual physical conditions under which the units are placed in the aquatic environment. It is imperative that the artificial substrates be located in a consistent fashion with particular emphasis on sustained current velocity over the set. With the exception of water quality, the amount of current tends to have the most profound effect on the types and numbers of organisms collected using artificial substrates in Ohio. For an accurate interpretation of the ICI, current speeds should be no less than 0.3 ft/s (10 cm/s) under normal summer–fall flow regimes. These conditions can usually be adequately met in

Table 2. Metrics Used to Calculate the Ohio EPA Invertebrate Community Index (ICI)

1. Total Number of Taxa	6. Percent Caddisfly Composition
2. Number of Mayfly Taxa	7. Percent Tribe Tanytarsini Midge Composition
3. Number of Caddisfly Taxa	8. Percent Other Dipteran and Non-Insect Composition
4. Number of Dipteran Taxa	9. Percent Tolerant Organisms (from Table 3)
5. Percent Mayfly Composition	10. Number of Qualitative EPT Taxa

Note: Scoring (6, 4, 2, or 0 points) for all metrics determined by basin drainage area (mi^2) at the sampling location. See Figures 1 to 10.

all sizes of perennial Ohio streams but can be a problem in small headwater streams or those streams so highly modified for drainage that dry weather flows maintain intermittent, pooled habitats only. In these situations, sampling can be conducted, but an alternative interpretation of the ICI value and/or the use of other assessment tools may be necessary.

An additional area of importance concerns the accuracy of identification of the sample organisms. The ICI has been calibrated to specified levels of taxonomy currently being used by the Ohio EPA. It is imperative that accurate identifications to those levels be accomplished. Otherwise, problems may arise in the ICI metrics dealing with the identity and/or number of taxa of a particular organism group. Inaccurate identifications can also be a problem in the ICI metric dealing with the percent abundance of pollution tolerant organisms. Table 1 lists current taxonomic keys and the level of taxonomy routinely used by the Ohio EPA for various macroinvertebrate taxonomic classifications.

3.0 DEVELOPMENT OF THE INVERTEBRATE COMMUNITY INDEX

3.1 Metric Selection

The principal measure of overall macroinvertebrate community condition used by the Ohio EPA is the Invertebrate Community Index (ICI), a measurement derived from the wealth of macroinvertebrate community data collected over the years by aquatic biologists at the Ohio EPA. The ICI consists of ten compositional and structural community metrics (Table 2), each of which receives a score of 6, 4, 2, or 0 points based on a comparison with a set of ecoregional reference sites. Metrics 1 to 9 of the ICI are generated from the artificial substrate sample data while Metric 10 is based on the qualitative sample data.

The selection of the ten metrics ultimately chosen for the ICI was facilitated by analyzing the process by which Ohio EPA biologists had subjectively judged the quality of a macroinvertebrate sample. In essence, the index and its final set of metrics effectively quantified a more subjective, narrative approach that was previously used. This allowed for a more objective and efficient level of assessment and decision making. Structural and compositional rather than functional metrics were selected because of their accepted historical use, simpler derivation, and ease of interpretation. However, a functional component is inherent in the index since watershed size at the sampling location affects metric scoring. In effect, scoring of metrics is strongly influenced by functionally based differences in the macroinvertebrates that are inhabiting the wide range of stream sizes of the reference sites used to calibrate scoring of each metric (e.g., populations predominated by collector/gatherers, scrapers, or filter feeders). The strength of the ICI lies in its ability to directly compare the biological performance of the subject stream site against performance measured at reference sites of similar watershed size and from the same ecoregion of Ohio. The ICI value, the summation of the metric scores, is a single number that reflects general biological condition, which has incorporated into it ten measurements that, with various degrees of effectiveness, can and have often been used to accomplish this task individually. It was thought that, used in the aggregate, these metrics would minimize the weaknesses and drawbacks that each has alone.

Application of an ICI-type multimetric assessment tool outside of Ohio should not be restricted by or limited to the set of metrics derived for use in Ohio. Rather, the flexibility of the multimetric approach allows for the utilization of differing collection methodologies with selection of metrics most appropriate for the diverse geographic settings and ecoregions of the United States. However, the common denominator of all applications should be the regional reference site approach and its use to calibrate scoring of the selected metrics and, ultimately, to set performance expectations and establish biocriteria.

3.2 Scoring of Metrics

The 6, 4, 2, or 0 point system is structured to score sample metrics against expectations derived from a database of least impacted regional reference sites. These sites were selected from the Ohio EPA database using guidelines developed by Hughes et al. (1986). Scoring criteria for each metric were developed through a quantitative calibration process in which reference site metric values were plotted against a log transformation of drainage area (a reflection of stream size) and scoring ranges determined after a method modified from Fausch et al. (1984). For example, six points are scored if a given metric falls in the range exhibited by exceptional stream communities, 4 points for those metric values characteristic of more typical, but good communities, 2 points for metric values moderately deviating from the expected range of good and exceptional values, and 0 points for metric values strongly deviating from the expected range of good and exceptional values. The summation of the individual metric scores, as determined by the relevant attributes of an invertebrate sample with consideration given to stream drainage area at the sampling location, results in the ICI value that ranges from 0 to 60. Four scoring categories were chosen because of the historical use by the Ohio EPA of four levels of biological community condition (i.e., exceptional, good, fair, and poor), a situation that, as defined above, is reflected by the metric score of a sample.

The four scoring categories were calibrated using data from 246 least impacted reference sites distributed across Ohio's five ecoregions as delineated by Omernik (1987) and Omernik and Gallant (1988). To determine the 6, 4, 2, or 0 values for each ICI metric, the reference site database was plotted vs. drainage area. Similar to procedures used to calibrate the IBI (Fausch et al. 1984), the scatter plot of each metric was examined by eye to determine if any sloped relationship existed with drainage area. When it was decided if a direct, inverse, combination of both, or no relationship existed, the appropriate 95th percentile line was estimated and the area beneath partitioned into four equal parts as determined by the distribution of the reference points. One difference between this procedure and that used by Fausch et al. (1984) to calibrate the IBI was the use of four scoring categories rather than three. Another difference involved some percent abundance and taxa richness scoring categories that were not equally divided since the distribution of data points showed a tendency to clump at or near zero. In these situations, a modified method was used where the zero scoring category included zero values only and the 6, 4, and 2 point categories were delineated by sequential bisections of the remaining wedge of reference data points. One final difference involved the use of drainage area as a scaling factor rather than stream order. The decision to use drainage area as an indicator of stream size rather than stream order, or a factor such as stream width used by Lyons (1992a), was based on the availability and ease of drainage area calculation and its relevance to stream size. Stream order was viewed as being too coarse (Hughes and Omernik 1981b) and stream width is simply not representative of stream size given the widespread historical modification of streams in Ohio. In other regions of the United States, these and other parameters may be appropriate as scaling factors. The ultimate decision must be determined by experts familiar with regional patterns of stream morphology.

3.3 Description of Metrics

3.3.1 Metric 1. Total Number of Taxa

The species area plot of the total taxa metric vs. drainage area is depicted in Figure 1. Taxa richness is a key component of several new-generation indices currently used to evaluate macroinvertebrate community integrity (Lenat 1988; Plafkin et al. 1989; Kerans and Karr 1994). The underlying reason is the basic ecological principal that healthy, stable biological communities in warmwater streams have high species richness and diversity. As can be seen by the species area curve, the total number of taxa collected from artificial substrates in Ohio tends to decrease in the larger rivers. This is consistent with the River Continuum Concept (RCC; Vannote et al. 1980), which predicts maximum taxa richness in midsized streams and decreased diversity in larger streams and rivers due to changes in organic inputs and plant growth. A contributing factor to the decline in taxa richness with increased watershed size is the more monotonous nature of substrate types in larger rivers. An additional consideration, however, is that even the best, larger rivers have been subjected to some degree of cultural degradation in Ohio.

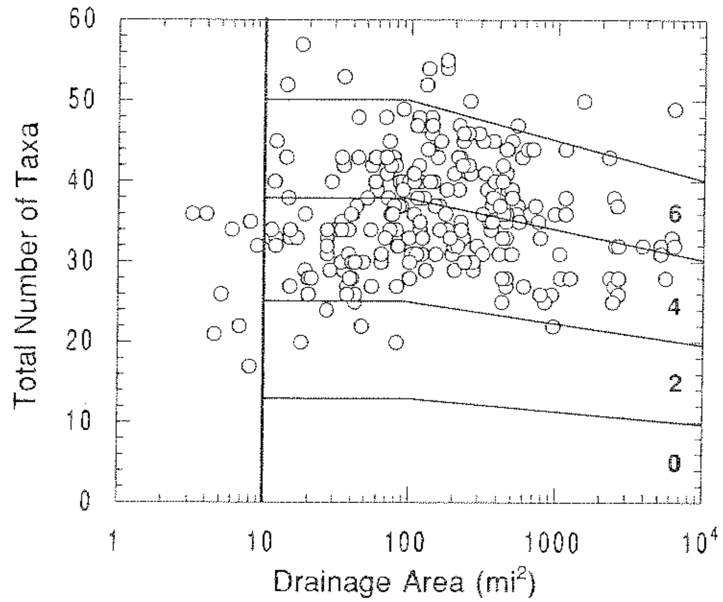

Figure 1. The relationship of ICI Metric 1, Total Number of Taxa, with the log transformation of drainage area at 246 Ohio reference sites. (An inverse relationship exists with drainage areas >100 mi^2.)

3.3.2 Metric 2. Number of Mayfly Taxa

Mayflies are an important component of an undisturbed stream macroinvertebrate fauna in Ohio. As a group, they are decidedly pollution sensitive and are often first to decline and eventually disappear from artificial substrate collections with the onset of environmental perturbation. Thus, they are a good indicator of high quality ambient conditions. Environmental requirements and pollution tolerances have been thoroughly documented for many species (Hubbard and Peters 1978). Taxa richness of mayflies is included as an individual metric of one current assessment protocol (Kerans and Karr 1994) but it most often appears as a component of an EPT (Ephemeroptera, Plecoptera, and Trichoptera) taxa richness metric (Lenat 1988; Plafkin et al. 1989). The species area plot of reference site mayfly taxa vs. drainage area is depicted in Figure 2. The general trend in mayfly diversity reflects highest variety of types in intermediate size streams with slight decreased diversity in the smaller and larger drainages. As predicted by the RCC, this is the result of the transitional nature of the intermediate streams and the corresponding increased variety of macrohabitat, microhabitat, and food sources. In effect, environmental conditions are highly diverse and support a mayfly fauna transitional between the smaller Ohio streams and the larger Ohio rivers.

3.3.3 Metric 3. Number of Caddisfly Taxa

Caddisflies are often a predominant component of the macroinvertebrate fauna collected from artificial substrates in larger, relatively unimpacted Ohio streams and rivers. Though generally thought to be slightly more pollution tolerant as a group than mayflies, they display a wide range of tolerance among genera and species (Harris and Lawrence 1978). Notwithstanding, few can tolerate heavy pollutional stress and, as such, can be good indicators of environmental conditions (Kerans and Karr 1994). The distribution of reference site caddisfly taxa vs. drainage area shows a clear, increasing trend with stream size (Figure 3). This can be explained by the predominance in Ohio of net spinning, filter feeding caddisflies of the families Hydropsychidae, Polycentropodidae, and Philopotamidae and case-making microcaddisflies of the family Hydroptilidae. Habitat preferences of the filter feeders include streams with abundant suspended organic matter while the micro-caddisflies feed mainly on periphytic diatoms and filamentous algae. These environmental conditions are best met in the larger streams and

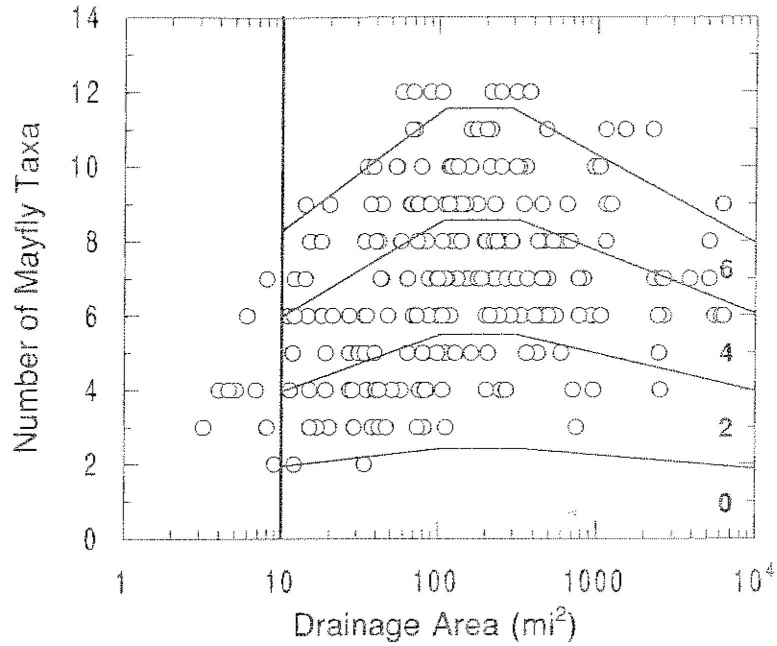

Figure 2. The relationship of ICI Metric 2, Number of Mayfly Taxa, with the log transformation of drainage area at 246 Ohio reference sites. (A direct relationship exists with drainage areas <100 mi²; an inverse relationship exists with drainage areas >300 mi².)

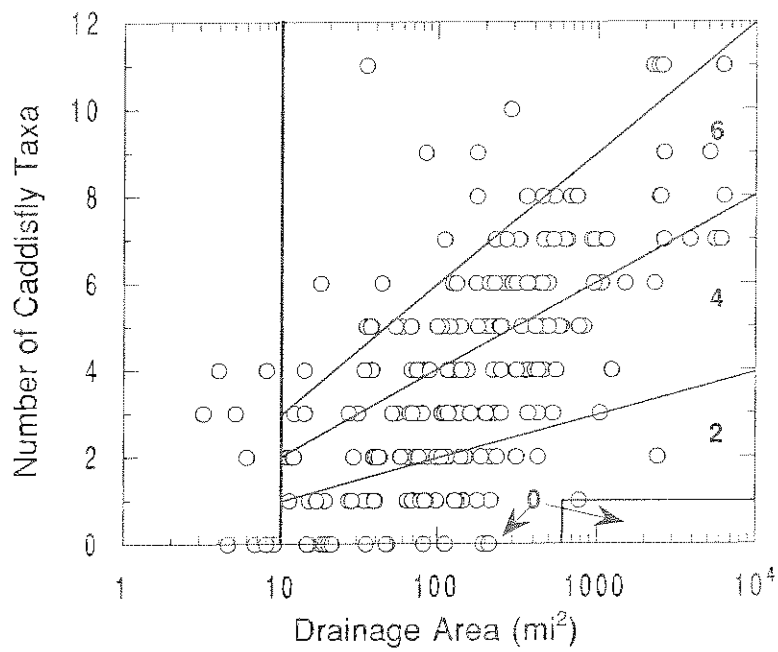

Figure 3. The relationship of ICI Metric 3, Number of Caddisfly Taxa, with the log transformation of drainage area at 246 Ohio reference sites. (A direct relationship exists with drainage area; a score of 0 for no taxa at drainage areas <600 mi²; a score of 0 for <2 taxa at drainage areas >600 mi².)

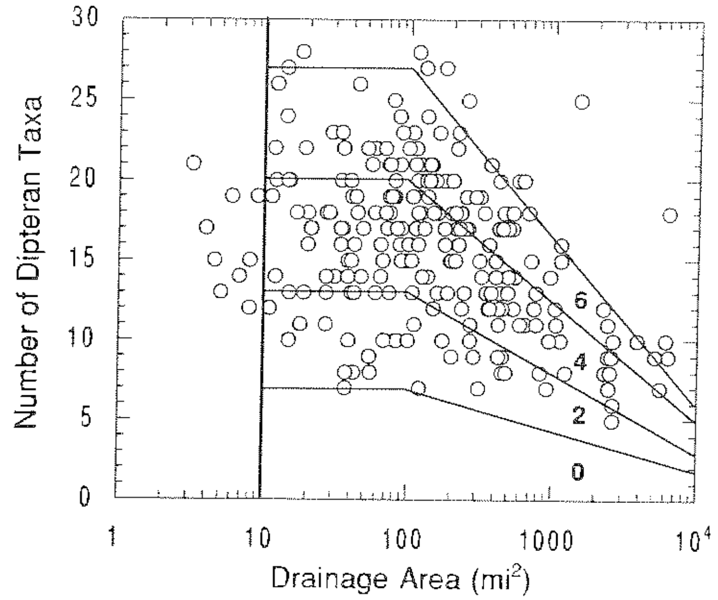

Figure 4. The relationship of ICI Metric 4, Number of Dipteran Taxa, with the log transformation of drainage area at 246 Ohio reference sites. (An inverse relationship exists with drainage areas >100 mi^2.)

rivers where the import of fine particulate organic matter is maximized and algal growth is optimal due to the availability of nutrients and more open canopies. As can he seen in Figure 3, for drainages less than 600 mi^2 (1550 km^2), zero scores occur only when no caddisfly taxa are present. In artificial substrate collections from the smaller Ohio watersheds, it is normal to collect fewer kinds of the more common caddisflies even at unpolluted sites. For drainages greater than 600 mi^2 (1550 km^2), habitat conditions at sampling sites are much more conducive to the proliferation of these caddisflies and, therefore, at least two taxa must be present to score higher than zero.

3.3.4 Metric 4. Number of Dipteran Taxa

Among the major aquatic macroinvertebrate groups, dipterans, especially midges of the family Chironomidae, rank high in faunal diversity and display a wide range of pollutional tolerances (Beck 1977; Berg and Hellenthal 1990; Lenat 1993). They are usually the major component of a macroinvertebrate collection using Ohio EPA methodologies and, under heavy pollutional stress, are often the only insects collected and, at the same time, are the predominant macroinvertebrate group. Larval taxonomy has improved greatly and, as a result, clear patterns of organism assemblages have become more distinctive under water quality conditions ranging from near pristine to heavily organic and toxic. The fact that they do not usually disappear under severe pollutional stress makes them especially valuable in evaluating water quality. The distribution of dipteran taxa vs. drainage area is shown in Figure 4. A clear, inverse relationship exists with watersheds greater than 100 mi^2 (260 km^2). In the larger rivers, there is a tendency towards increased populations of proportionately fewer dipteran taxa. This is probably the result of abundant food supplies but fewer functional niches as habitat conditions and food types become increasingly monotonous.

3.3.5 Metric 5. Percent Mayfly Composition

As with number of mayfly taxa, the percent abundance of mayflies collected in an artificial substrate sample is often readily and rapidly affected by often minor environmental disturbances. Though much more reference site variability exists in this metric compared with the taxa metric, there is a strong relationship with water quality. The range of abundances in the relatively unimpacted reference site database varies from near zero to greater than 80% (Figure 5). However, data from slightly degraded (fair) and severely degraded (poor) stream communities in Ohio indicate that mayfly abundance is reduced

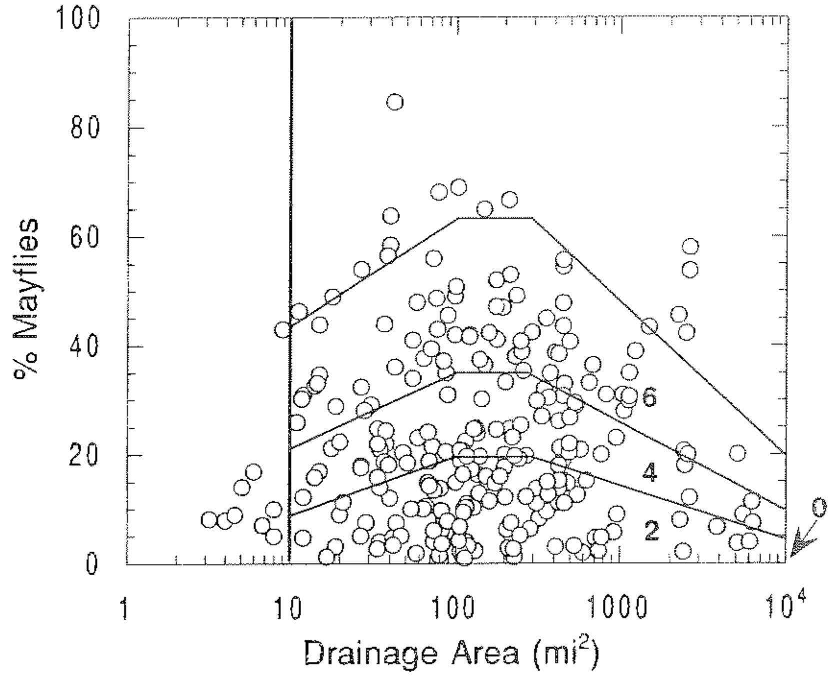

Figure 5. The relationship of ICI Metric 5, Percent Mayflies, with the log transformation of drainage area at 246 Ohio reference sites. (A direct relationship exists with drainage areas < 100 mi^2; an inverse relationship exists with drainage areas >300 mi^2.)

considerably under even slight impacts and is essentially nonexistent under more severe impacts. Thus, it was felt that even a few mayflies in low abundance should contribute a metric score of at least 2 points. Therefore, only those artificial substrate samples with no mayflies are scored zero for the metric. The distribution of reference site data points shows a trend similar to that observed for the mayfly taxa metric.

3.3.6 Metric 6. Percent Caddisfly Composition

As with the number of caddisfly taxa metric, the percent abundance of caddisflies collected from artificial substrates is strongly related to stream size (Figure 6). Again, optimal habitat and availability of appropriate food type seem to be the main considerations for stimulating large populations of the common Ohio caddisflies. As can be seen in Figure 6, caddisflies can make up a significant portion of the macroinvertebrate community, often exceeding 25% of all organisms collected. However, they are just as likely to be found in quite low numbers, at times comprising less than 1% of a sample. Because of their general disposition as an intermediate group (more tolerant than most mayflies and less tolerant than many dipterans) and because they disappear rapidly under environmental stress, zero scores are restricted to those sites with drainage areas less than 600 mi^2 (1550 km^2) where no caddisflies are collected. This scaling, similar to that used in Metric 3, is necessitated because low numbers of caddisflies are often collected from artificial substrates at the smaller stream sites even when resource disturbance is minimal. At sites with greater than 600 mi^2 (1550 km^2) of drainage, appropriate habitat conditions are much more likely to exist; therefore, individuals from the most prevalent Ohio caddisfly families should be well represented and must be present in at least minimal numbers to score greater than zero.

3.3.7 Metric 7. Percent Tribe Tanytarsini Midge Composition

The tanytarsini midges are a tribe of the chironomid subfamily Chironominae. The larvae are generally burrowers or clingers, and many species build cases out of sand, silt, and/or detritus. Many species feed on microorganisms and detritus through filtering and gathering, though a few are scrapers.

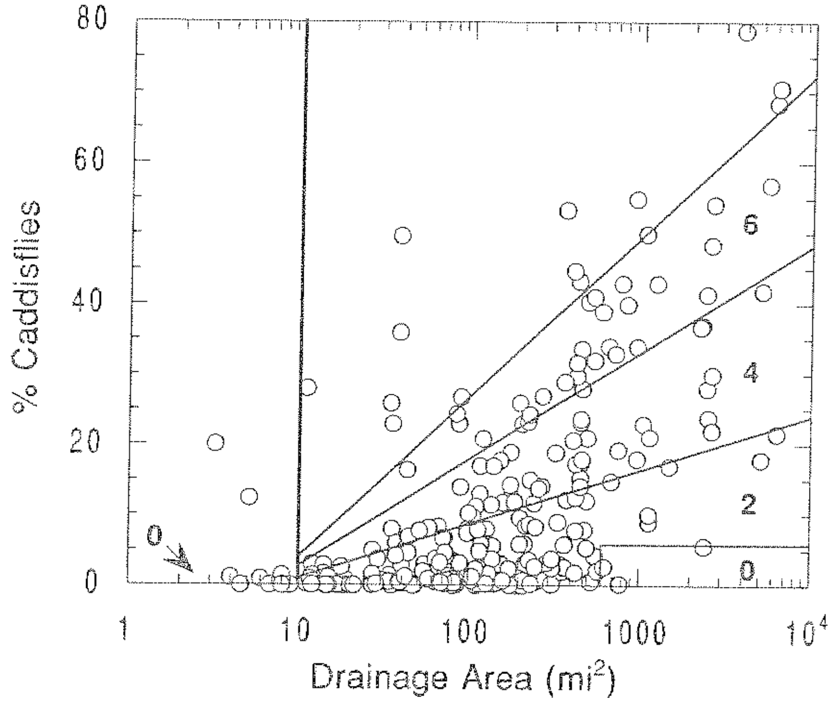

Figure 6. The relationship of ICI Metric 6, Percent Caddisflies, with the log transformation of drainage area at 246 Ohio reference sites. (A direct relationship exists with drainage area; a score of 0 for no individuals at drainage areas <600 mi^2; a score of 0 for minimal percent abundance at drainage areas >600 mi^2.)

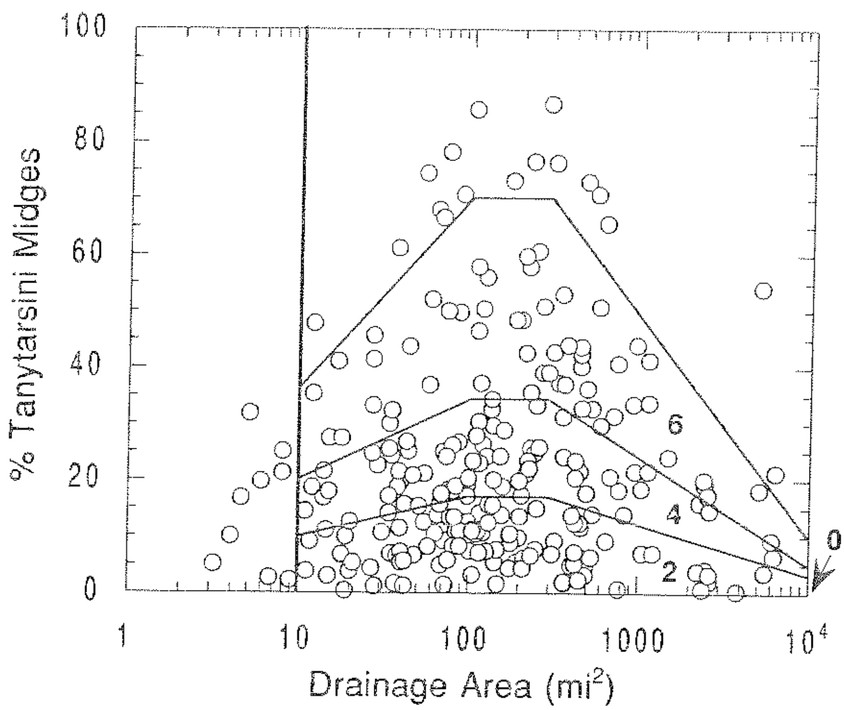

Figure 7. The relationship of ICI Metric 7, Percent Tanytarsini Midges, with the log transformation of drainage area at 246 Ohio reference sites. (A direct relationship exists with drainage areas <100 mi^2; an inverse relationship exists with drainage areas >300 mi^2).

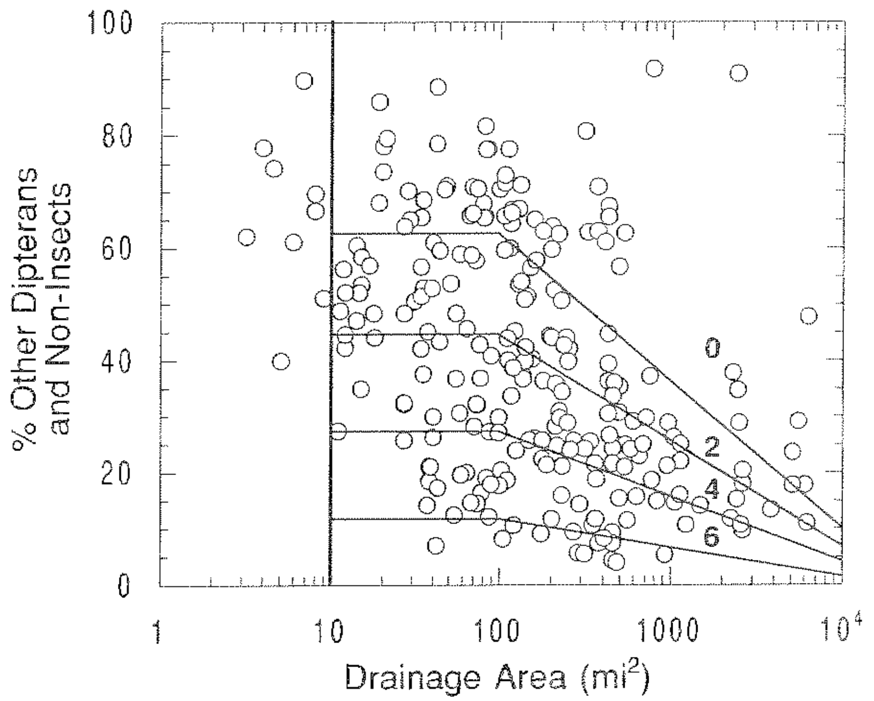

Figure 8. The relationship of ICI Metric 8, Percent Other Diptera and Non-Insects, with the log transformation of drainage area at 246 Ohio reference sites. (An inverse relationship exists with drainage areas >100 mi^2).

Eleven genera and up to 140 species occur in North America, although only eight genera and 22 distinct taxa have been collected in Ohio streams and rivers. At the relatively unimpacted Ohio reference sites, they are often the predominant midge group and can exceed 50% of the total number of organisms collected from artificial substrates. As a group, they appear to be intermediate in pollution tolerance and often disappear or decline under moderate pollutional stress. However, some genera, species groups, and species have been determined to be quite sensitive to pollution (Anderson et al. 1980; Simpson and Bode 1980; Hilsenhoff 1987; Lenat 1993). As depicted in Figure 7, populations of tanytarsini midges tend to peak at reference sites in the 100 to 300 mi^2 (260 to 775 km^2) range of watershed size. Because of their moderate intolerance to environmental disturbance, zero scores only occur when no tanytarsini midges are present.

3.3.8 Metric 8. Percent Other Dipteran and Non-Insect Composition

This metric includes the community percentage of all dipterans (excluding the midge tribe Tanytarsini) and other non-insect invertebrates such as aquatic worms, flatworms, amphipods (scuds), isopods (aquatic sow bugs), freshwater hydras, and snails. This metric is one of two negative metrics of the ICI in that an increased abundance results in a lower metric score. Taxa in these groups of macroinvertebrates, though often present as part of a healthy stream community, are those that generally tend to predominate under adverse water quality conditions (Hynes 1966; Hart and Fuller 1974). Depending on the severity of the stress, these organisms will comprise over 90% of the individuals collected in an artificial substrate sample. Figure 8 depicts the distribution of reference site data for the metric. As indicated, reference site percentages are inversely related to stream size. However, this relationship does not seem to hold for seriously impaired situations; under these circumstances, the dipterans and non-insects defined by this metric usually predominate at a high percentage regardless of stream size. In cases where conditions are so severely degraded that no or only a few organisms (<50 individuals) are collected and the percentage of dipterans and non-insects is consequently at or near zero, the metric defaults to a zero score rather than

Table 3. List of Pollution-Tolerant Macroinvertebrates Used to Determine Metric 9 of the Invertebrate Community Index (ICI)

Common name		Scientific name
Aquatic segmented worms	Annelida:	Oligochaeta
Midges	Diptera:	*Psectrotanypus dyari*
		Cricotopus (C.) bicinctus
		Cricotopus (Isocladius) sylvestris group
		Nanocladius (N.) distinctus
		Chironomus (C.) spp.
		Dicrotendipes simpsoni
		Glyptotendipes barbipes
		Parachironomus hirtalatus
		Polypedilum (Pentapedilum) tritum[a]
		Polypedilum (P.) fallax group
		Polypedilum (P.) illinoense
Limpets	Gastropoda:	*Ferrissia* spp.
Pond snails		*Physella* spp.

[a] New listing not included in original table from Ohio EPA (1987b).

a higher value. This adjustment is needed since a low number of organisms renders the proportional relationships between macroinvertebrate groups relatively meaningless.

3.3.9 Metric 9. Percent Tolerant Organisms

Values for this metric are generated using a predetermined list of organisms compiled by the Ohio EPA (1987b) and reproduced in Table 3. The list includes those organisms in Ohio that have consistently been observed to be extremely tolerant of a broad range of impacts and which tend to predominate artificial substrate collections from areas with severe perturbation. The list includes organisms tolerant to organic pollution as well as some taxa observed to withstand toxic impacts (Hynes 1966; Hart and Fuller 1974; Simpson and Bode 1980; Burch 1982). Thus, this should be a reasonable metric for the evaluation of community tolerance over a broad range of degradation types. This is a preferable difference over other measurements of community tolerance that have been developed primarily to reflect one type of pollution impact. Like Metric 8, this is a negative metric and, as such, a low number (<50 individuals) or an absence of organisms in a sample defaults to a zero score for the metric regardless of the presence or absence of the specified tolerant taxa. Figure 9 depicts the reference site tolerant organism percentages vs. drainage area. A strong inverse relationship with watershed size exists. For drainages greater than 1000 mi^2 (2600 km^2), the percentage of tolerant organisms found at reference sites becomes so low that the scoring categories are quite restrictive. In fact, at a number of the reference sites, none or fewer than 1% of these organisms were present. However, as with Metric 8, watershed size tends to have little effect when pollutional disturbances are prevalent. Sites with minor to severe degradation can have large populations of these organisms regardless of stream size.

3.3.10 Metric 10. Number of Qualitative EPT Taxa

This is the sole ICI metric that is generated by the qualitative sample taken in conjunction with the artificial substrate sampling. Since the qualitative sampling utilizes a substrate and habitat dependent method, that is, a method affected by the variation in natural substrates and habitats available in the sampling area, the metric is a measurement of both habitat quality and diversity as well as water quality. The metric consists of the taxa richness of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) and is commonly used as an index component (Lenat 1988; Plafkin et. al 1989; Resh, Chapter 12; Southerland and Stribling, Chapter 7). Since stoneflies are relatively uncommon in summer collections in Ohio, the metric is mostly dependent on the kinds of mayflies and caddisflies found. The depiction of qualitative EPT taxa vs. drainage area (Figure 10) reflects a trend similar to Metrics 2 and 5, total taxa richness and percentage of mayflies. As with Metrics 1 and 2, the higher numbers of EPT taxa occur in the midsized streams and rivers, a trend predicted by the RCC, and result from greater habitat and food type variety in the systems transitional between small streams and large rivers.

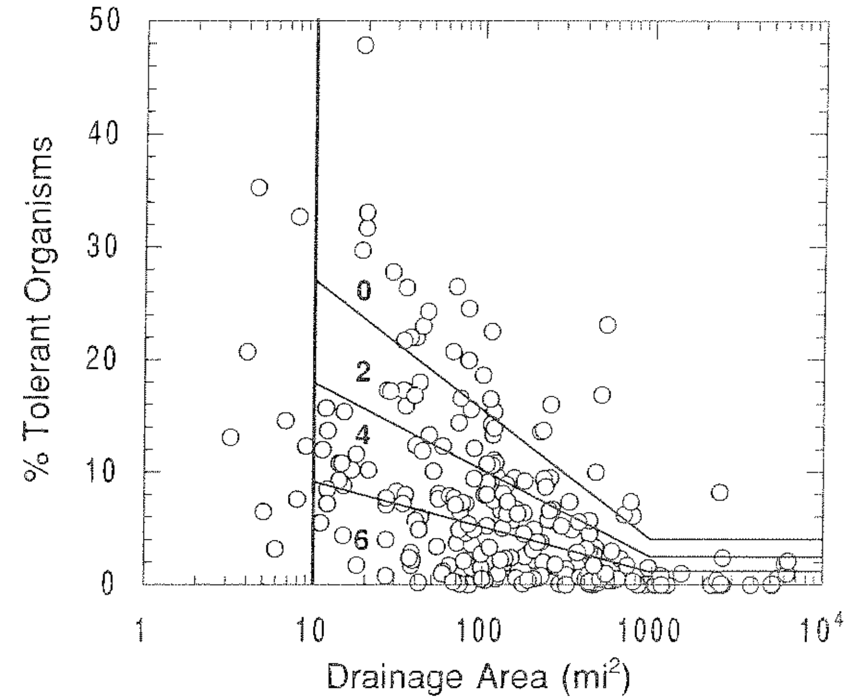

Figure 9. The relationship of ICI Metric 9, Percent Tolerant Organisms, with the log transformation of drainage area at 246 Ohio reference sites. (An inverse relationship exists with drainage areas >1000 mi^2.)

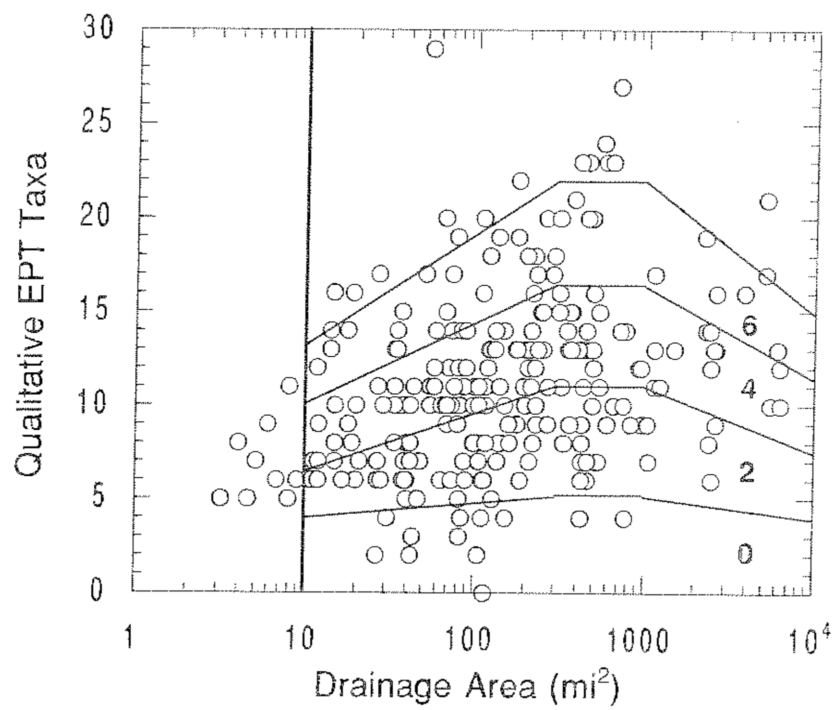

Figure 10. The relationship of ICI Metric 10, Qualitative EPT Taxa, with the log transformation of drainage area at 246 Ohio reference sites. (A direct relationship exists with drainage areas <300 mi^2; an inverse relationship exists with drainage areas >1000 mi^2.)

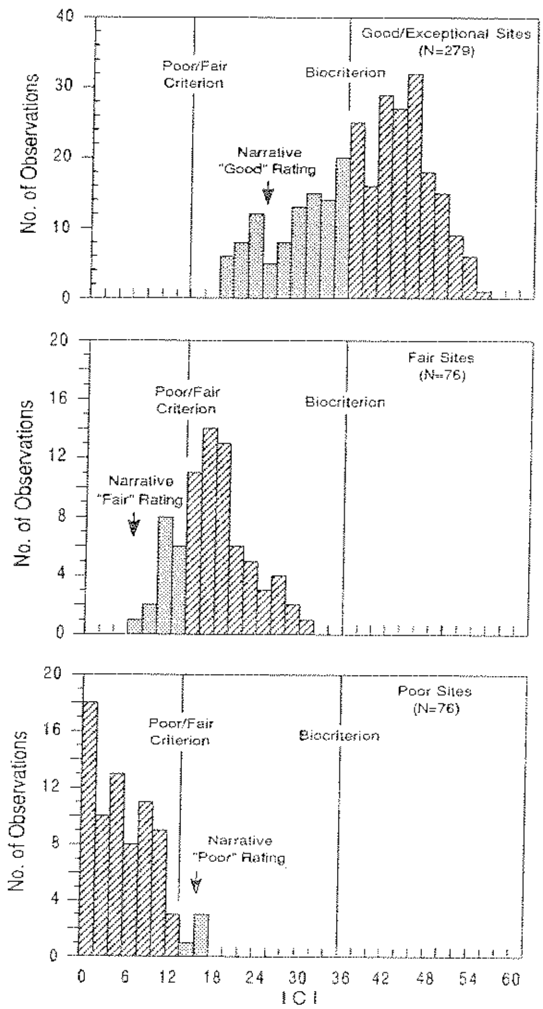

Figure 11. Frequency distribution of ICI scores for 431 sites rated as good/exceptional, fair, and poor using the original Ohio EPA narrative assessment protocol. The dotted bars are sites where narrative assessments rated differently than assessments using ICI scores and biocriteria derived from a regional reference site approach; slashed bars are sites where the two assessments were in agreement.

4.0 ICI VALIDATION AND TESTING

4.1 Comparison with Original Narrative Protocols

In an effort to determine the effect of using this more definitive, less subjective assessment methodology, evaluations using the original narrative protocols (i.e., exceptional, good, fair, or poor) from 431 sites sampled between 1981 and 1987 were compared to ICI-based biocriteria calibrated using regional reference sites. The results indicated that the original narrative approach rated a significant number of sites as being better than indicated by the calibrated ICI (Figure 11). The narrative approach rated as "good" (attaining the designated aquatic life use) 36% of sites classified by the ICI as impaired (less than the applicable ecoregional criterion and, thereby, not meeting the designated use). Additionally, 22% of the sites classified by the ICI as severely impaired (the numeric equivalent of poor) were placed in the "fair" category using the narrative approach. Conversely, only 5% of sites rated as "poor" by the narrative method were classified higher by the ICI and no ICI scores exceeded the ecoregional biocriteria at sites narratively rated as "fair." The primarily unidirectional error orientation of the narrative approach resulted in the rating of sites better than they should be when judged against the ICI score. In many cases, the tendency of the investigator was to give a stream location the benefit of the doubt and judge it acceptable when it may not have been. This is not a problem when using the more objective ICI assessment process. While it may seem premature to assume that the ICI is more accurate, the fact that it is a multimetric assessment mechanism designed to produce the essence of the narrative system, but with greater objectivity and precision, and that it extracts information directly from the regional reference sites, supports the contention that the ICI is the preferable assessment tool.

4.2 Variability Analyses

It is of critical importance in biological monitoring to collect a consistent and reproducible sample. Variation in data can be divided into sampling variation (i.e., errors in methods and techniques) and natural variation, both between sites and at a given site over time (i.e., temporal variability). Sampling error should be minimized to detect true spatial differences between sampling sites or at a site over time. Data from a special Ohio EPA methods study conducted in 1981 were used to estimate the ramifications of both natural variability at a sampling site and the inherent error of the macroinvertebrate sampling techniques and methodologies. Temporal variability of the ICI at a given site was assessed by analyzing sites in the Ohio EPA macroinvertebrate database having multiple year information. Within year variation (i.e., seasonality) of the ICI at a given location has not been thoroughly assessed; however, this should not be a significant consideration when evaluating community quality since macroinvertebrate sampling by the Ohio EPA is confined to a specified index period. Ohio EPA protocols restrict sampling to a mid-June through September index sampling period with all artificial substrate samples, in actuality, being retrieved over a six-week period from mid-August to the end of September. Thus, seasonal differences in macroinvertebrate communities at a given location over this relatively brief time frame should be minimal and not significantly affect ICI scoring as a consequence.

4.2.1 Site Variability Analysis

The 1981 study was conducted at a representative site in Big Darby Creek, a medium sized stream located in central Ohio. Big Darby Creek is a documented high quality aquatic resource and is populated by a very diverse macroinvertebrate fauna (Ohio EPA 1983). It was thought the potential for variation attributed to sampling error under these conditions would be significant and, thus, be a good test of the reliability of the ICI. Since external impacts are minimal, measured variability would be most likely due to sampling inconsistencies. Twenty-two artificial substrate sampling units were arranged in an X-shaped grid and colonized under similar conditions with regard to current velocity, water depth, and riparian canopy. Nineteen of the units were subsequently retrieved and analyzed; three units were not used in subsequent analyses because of differences in current velocity at the specific locations where these substrates were retrieved.

Initial examination of the data indicated that measured physical parameters (depth and current velocity) were relatively constant and should have had no significant effect on various biological parameters (e.g., total taxa, etc.). Similar results were found when the physical factors were compared to calculated ICI values. It seemed appropriate to assume that the same water quality conditions were affecting all the sampling units; thus, it was inferred that any variability in ICI scores was due to sampling error related to methodologies and/or natural biological processes such as predation, emigration, immigration, mortality, and natality. ICI summary statistics generated from the test data are presented in Table 4; the frequency distribution of ICI scores is depicted in Figure 12. ICI scores were reasonably consistent among the nineteen samples. The median ICI value was 36 and scores among the 19 samples ranged from 30 to 44. Though this appears to be a considerable range of ICI values, the 25th and 75th percentile scores were 36 and 38, respectively. Fourteen of nineteen (75%) and seventeen of nineteen (90%) scores were within plus or minus two and four points of the median, respectively. As such, it was determined that changes in ICI scores at test sites compared to an ecoregional biocriterion or to an upstream control station

Table 4. Invertebrate Community Index (ICI) Summary Statistics Generated from Macroinvertebrate Data Collected at the 1981 Big Darby Creek Test Site

Sample size	19
Mean ICI	36.6
Standard error	0.7
Median ICI	36
Minimum ICI	30
Maximum ICI	44
ICI quartiles	
Lower (25%)	36
Upper (75%)	38

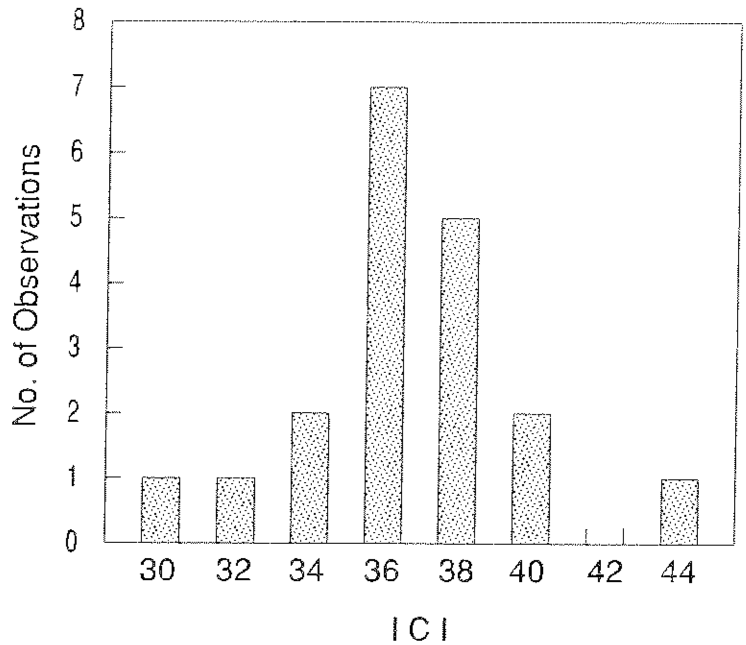

Figure 12. Frequency distribution of Invertebrate Community Index (ICI) scores derived from 19 replicate samples collected at the 1981 Big Darby Creek test site.

should be considered in a zone of insignificant departure if the ICI difference is four points or less. Based on the test data, this interval should adequately allow for the potential effects of natural variation and sampling error on the ICI value.

4.2.2 Temporal Variability Analysis

Because of the inability to predict with absolute certainty that a given site has not changed in quality over a period of years, it was difficult to assess the potential effect of natural, year-to-year variability on ICI scoring. However, there are sampling locations in the Ohio EPA database where multiple-year data are available and where little or no change in resource quality is believed to have occurred. ICI scores from three such sites are depicted in Figure 13. Out of necessity, these sites needed to be high quality and located on streams well removed from major anthropogenic pollution sources. In general, these sites varied little from year to year. For the most part, ICI scores were consistent and differed by no more than 6 points over the sampling intervals which spanned 7 to 8 years and included 3 to 6 sampling events. Most importantly, however, scores always exceeded the applicable biocriterion and the evaluation of aquatic

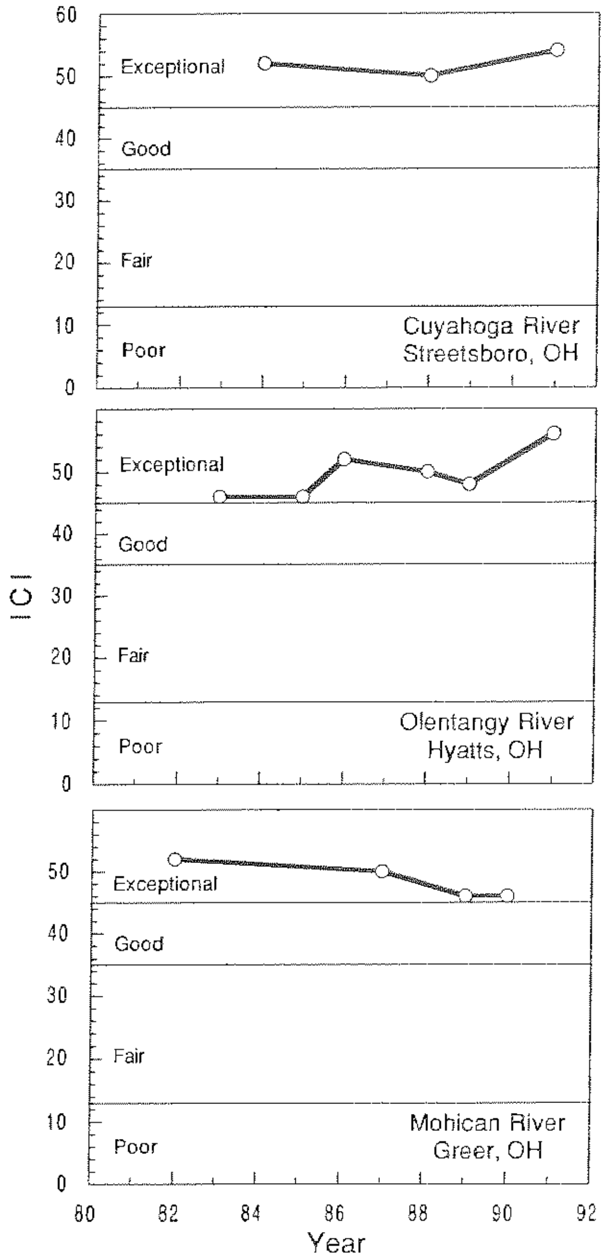

Figure 13. Invertebrate Community Index (ICI) scores from multiple-year sampling conducted at three high-quality Ohio stream sites.

life use attainment status did not change. A wide range of natural environmental conditions existed over the years at each location and included exceptionally high flow years as well as drought years with minimal stream flows and the potential for maximum biological stress. It was concluded that ICI scores are consistent at locations where little man-induced change has occurred. Thus, macroinvertebrate community condition, as reflected by the ICI, reacts minimally to natural biological processes that might otherwise be perceived as important factors which could potentially affect scoring and thereby the community assessment at a sampling site.

It is interesting to note that the Ohio EPA database also includes sites sampled over multiple years in areas of lesser quality and impacted by human activities. If degradation is not severe at these borderline quality sites, it is not unusual to see moderate fluctuations of ICI scores from year to year. However, far from indicating a "noise" problem with the ICI, this type of temporal variability can be of considerable importance in the diagnosis of pollution impacts. Oftentimes, these sites are located on stream reaches with significant nonpoint source problems or in urban/industrial influenced reaches where stream flows are nearly effluent dominated. In these cases, differing stream flow years and the corresponding effect on nonpoint source pollutant loadings and point source effluent dilution are apparently influencing the quality of the macroinvertebrate community that becomes established in a given year.

4.3 U.S. Environmental Protection Agency Evaluation

An independent evaluation of the ICI conducted by the U.S. Environmental Protection Agency (Davis and Lubin 1989) determined that the ICI, along with its associated metrics, seems to be a valid empirical indicator of macroinvertebrate community quality and is quite acceptable for its stated use. Various Statistical Analysis System (SAS Institute, Inc. 1985) procedures were use to test a number of aspects of the ICI including: (1) an evaluation of the reasonableness of the use and derivation of the invertebrate community measurements used to establish the ten metrics, (2) a determination if the drainage area relationships visually interpreted for the ten metrics were reasonable, (3) a determination if any of the ten ICI metrics are interrelated and, thereby, provide redundant information, (4) a determination if the assumption of equal weights for each metric was optimal, and (5) an evaluation of the overall accuracy of the ICI. The authors' analyses revealed no substantial faults or unnecessary redundancy in these various aspects of the ICI, and they concluded that there were no obvious changes which would significantly improve upon it.

5.0 APPLICATIONS OF THE ICI IN WATER RESOURCE ASSESSMENTS

The Ohio EPA has operated a program of intensive biological and water quality surveys of Ohio rivers and streams since 1979. In a 14-year period, over 550 different rivers and streams covering nearly 8000 miles have been assessed statewide. More than 90% of the principal pollution problem areas have been surveyed at least once. The Ohio EPA employs a multidisciplinary approach to the chemical, physical, and biological monitoring and assessment of surface waters. Biological evaluation methods include the use of the Index of Biotic Integrity (IBI) and Modified Index of Well-Being (MIwb) for fish, and the Invertebrate Community Index (ICI) for macroinvertebrates (Ohio EPA 1987b; Yoder 1989). Reference site-calibrated ecoregional biocriteria for all three indices were adopted into the Ohio Water Quality Standards and became effective in May, 1990. The rationale and procedures used in the development of Ohio's biocriteria have been reported elsewhere (Ohio EPA 1987a, 1987b, 1989a, 1989b). Besides biological monitoring, the Ohio EPA survey design also includes an assessment of physical habitat and the more traditional chemical/physical analyses of the water column and effluents. Additionally, assessments may include monitoring of toxic substances in the water column, effluents, fish tissue, and sediments as necessary. Together these data are used to support Ohio EPA program areas such as Water Quality Standards, NPDES permits, basinwide planning activities, natural resource damage assessments, and nonpoint source assessments. In 1990, the Ohio EPA initiated a five-year rotating basin approach to ambient biological and water quality monitoring and NPDES permitting. This cyclic and orderly approach to the assessment of Ohio's major watersheds not only makes the utilization of limited monitoring resources more cost effective, but assures that monitoring information will be available to support program areas when needed. Yoder (1991b) and Yoder and Rankin (Chapter 9) provide detailed discussions of the Ohio EPA's biological and water quality survey program and biocriteria applications.

Assessments of the ambient macroinvertebrate community using the ICI are primarily of two basic types: (1) intensive surveys of stream or river reaches using multiple sites in upstream to downstream longitudinal or synoptic subbasin configurations, and (2) multiple-year sampling at a specific fixed station on a stream or river. Intensive surveys are the basic design used in Ohio EPA's annual sampling program to address issues from a mainstem, subbasin, or basinwide perspective. Sampling sites are located based on the peculiarities of the stream or river and in accordance with the survey objectives. Assessments of

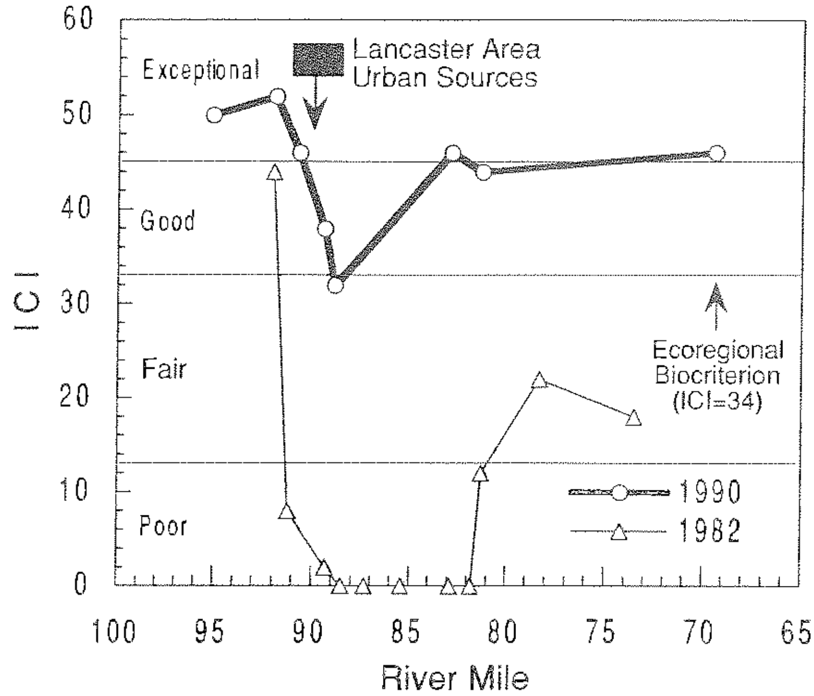

Figure 14. Longitudinal patterns and temporal trends in the Invertebrate Community Index (ICI) based on macroinvertebrate community data collected from the Hocking River within and downstream from the city of Lancaster, Ohio, 1982 and 1990.

point sources of pollution include upstream control(s) and downstream impact/recovery stations including mixing zone analyses to detect the potential for acutely toxic or rapidly lethal conditions. Additional sites are typically located within the study area to evaluate natural background conditions, to resample regional reference sites (once every ten years), or to monitor other issues such as pollution runoff from agriculture, mine drainage, urban, and construction site sources. With the advent of the five-year cycle to watershed monitoring, opportunities are arising to revisit previously sampled watersheds and to assess temporal changes in aquatic resource condition after major monetary expenditures for improvements in point source water pollution control. A similar opportunity exists over the long-term to evaluate the benefits of current efforts to implement best management practices and control nonpoint sources of pollution. In conjunction with the intensive biological surveys, macroinvertebrate sampling has also been conducted over multiple years since the early 1970s at over 30 key locations in major Ohio watersheds. Many fixed stations have had 10 or more annual samples taken over the intervening 18 years and significant long-term, mostly positive trends in ICI scores are being observed at some locations.

Results from these two distinctly different types of macroinvertebrate community monitoring are beginning to provide a uniquely comprehensive and standardized database from which changes in the quality of Ohio's water resources can be quantitatively evaluated over time. Of relevance to regulatory programs and others is the feedback provided about the success of efforts to control, reduce, and eliminate surface water pollution problems. The results of these types of analyses do more than simply answer the question of whether or not a waterbody is performing up to expectations with regards to biological integrity. Information is also derived that quantifies the degree of biological change, whether it be positive or negative, and, in the latter case, what amount of additional biological improvement will be necessary to achieve acceptable biological condition.

5.1 Use of the ICI in Intensive Surveys

An example of the use of macroinvertebrate community data in an intensive survey, including an assessment of temporal trends, is graphically presented in Figure 14. The subject waterbody is the

Hocking River, an Ohio River tributary originating in south-central Ohio, which was first surveyed and assessed in 1982 (Ohio EPA 1985a) and then again in 1990 (Ohio EPA 1991). Both surveys were focused on the river in the vicinity of the small city of Lancaster, Ohio, which for many years severely degraded the river. Inadequate wastewater treatment and raw sewage bypassing at the municipal treatment facility, combined wet and dry weather sewer overflows, and a heavy contribution of industrial effluents to the sewage collection system resulted in gross enrichment and heavy metals contamination, significant levels of instream toxicity, and periodic fish kills.

In 1982, the macroinvertebrate community was severely degraded in the vicinity of, and downstream from, Lancaster for over 20 miles (32 km). The impact of the various sources was dramatic and immediate. The ICI score decreased by 36 points between the very good quality upstream background sampling location and the first impacted site even though the sites were less than 1 mi (1.6 km) apart. The worst conditions were found in a 6 mi (10 km) stretch below the municipal treatment facility where ICI values of 0 were scored at five locations. Communities at these sites were almost exclusively composed of large numbers of tubificid worms populating extensive beds of sewage sludge throughout the river. Recovery in the macroinvertebrate community was incomplete at the farthest downstream site of the study area where only limited improvement was observed in the macroinvertebrate community.

Between the 1982 and 1990 surveys, numerous construction and operational improvements were instituted within the city's sewage system, which resulted in much better quality pretreatment of industrial effluents, higher-quality wastewater treatment plant effluent, and the virtual elimination of sewage bypass events. In 1990, macroinvertebrate communities, as assessed by ICI scores, reflected the vastly improved water quality conditions in the Hocking River as a result of these changes. Though there was still degradation in close proximity to the urban area and the municipal treatment facility, the differences between the 1982 and 1990 communities were visibly apparent. Beds of sewage sludge and tubificid worms were essentially absent and were replaced by much more diverse invertebrate populations including good numbers of mayfly and caddisfly taxa at sites where only worms had been found previously. Communities comparable to those found at upstream sites were collected only 6 mi (10 km) downstream and communities in between were not nearly as degraded as those in 1982.

Fish community assessments at comparable sites in the Hocking River reflected similar improvements between 1982 and 1990. However, changes were not nearly as dramatic as those observed in the macroinvertebrate community primarily due to a lagging and as yet incomplete recolonization process. However, complete community recovery to a level approaching that observed at reference sites appears further limited by the prevalence of severe macrohabitat degradation. Mainstem habitat in Lancaster remains in a state of recovery from prior channelization. Bank erosion is significant downstream from Lancaster where adjacent land use has encroached on the riparian zone. Although pockets of quality riffle, pool, and riparian habitat exists, much of the stream channel and riparian zone has suboptimal macrohabitat conditions, which will probably hamper the reestablishment of diverse populations of many fish species. The macroinvertebrate results, based on the artificial and natural substrate collections, indicate a strong positive response to the improved water quality and the availability of the few oases of quality macrohabitat. The net result is that continued fish community degradation precludes the full attainment of the designated beneficial aquatic life use of the Hocking River. The differential sensitivity and response shown by each organism group is advantageous when attempting to discern the effects of water quality problems in streams that also have serious habitat degradation. The use of this type of complementary biological data in assessments of streams and rivers in Ohio in conjunction with associated chemical and physical analyses has resulted in higher quality and more accurate information on which to base management decisions affecting the resource. Rankin (Chapter 13) provides a detailed discussion concerning the importance of macrohabitat integrity and its influence on the quality of Ohio rivers and streams.

5.2 Use of the ICI in Multiple-Year Fixed Station Sampling

Use of macroinvertebrate community data in multiple-year fixed station sampling are graphically presented in Figure 15. Two locations, the Black River at Elyria, Ohio, and the Cuyahoga River at Independence, Ohio, have been extensively monitored from the mid-1970s to the present. Over this interval, macroinvertebrate communities have been sampled nine times in the Black River and twelve times in the Cuyahoga River. In both cases, significant positive changes have occurred in the communities that are correlated with improvements in municipal wastewater treatment in each watershed. These rivers

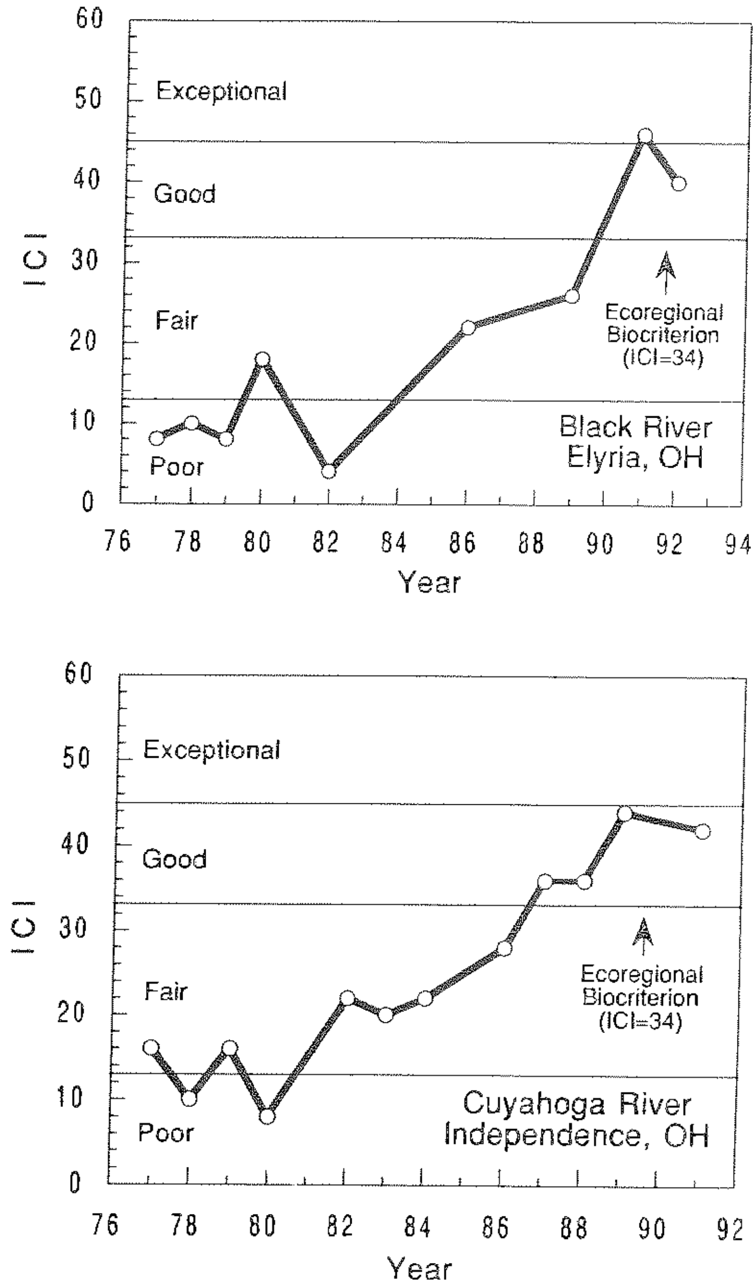

Figure 15. Multiple-year trends of the Invertebrate Community Index (ICI) based on macroinvertebrate community data collected from fixed stations located on the Black River at Elyria, Ohio, and the Cuyahoga River at Independence, Ohio, 1977–1992.

continue to be extensively investigated as both have been identified as Great Lakes Areas of Concern and have Remedial Action Plans in various stages of development.

5.2.1 Black River

The Black River is a major Lake Erie tributary originating in northeastern Ohio in the Erie–Ontario Lake Plain ecoregion. The long-term fixed station is located on the mainstem approximately 1 mi (1.6 km) downstream from the confluence of the East and West Branches and upstream from the Elyria wastewater treatment plant. This area of the river is located within the city of Elyria and, until recently, was impacted by a variety of sources including combined sewer overflows, industrial dischargers, on-site septic systems, and urban runoff. Sampling at the fixed station in the late 1970s and early 1980s resulted in low ICI scores indicating poor community performance. A biological and water quality survey of the Black River conducted in 1982 documented serious degradation throughout the mainstem (Ohio EPA 1985b). The predominant causes of the biological impairment were determined to be low instream dissolved oxygen levels and gross nutrient enrichment resulting from combined sewer releases and industrial discharges.

Since the 1982 survey and especially since 1986, significant improvements in the macroinvertebrate community at the fixed station have resulted in ICI scores achieving the biocriterion established for the Erie–Ontario Lake Plain ecoregion (Figure 15). Biological changes are correlated with improved combined sewer overflow controls and the elimination of industrial discharges due to plant closure or tie-in to the recently upgraded Elyria wastewater treatment plant. The 1988 plant expansion, coupled with the extension of a major interceptor sewer line, has helped to decrease the incidence of combined sewer discharges to the Black River within the Elyria wastewater collection system.

5.2.2 Cuyahoga River

The long-term fixed station on the Cuyahoga River is located at Independence, a small northeastern Ohio community located in the southern Cleveland metropolitan area and at the northern end of the Cuyahoga Valley National Recreation Area. The Cuyahoga River, like the Black River, is in the Erie–Ontario Lake Plain ecoregion of Ohio and is a major Lake Erie tributary. The basin is unique in that its headwaters begin near Lake Erie; the mainstem then flows south for about 50 mi (80 km) before flowing north near the City of Akron. Just north of Akron and at the southern edge of the National Recreation Area, the Akron wastewater treatment facility discharges to the river, essentially doubling river volume at baseflow conditions. The fixed station is located approximately 20 mi (32 km) downstream from the wastewater treatment plant; there are no other significant point sources in between.

Historically, the macroinvertebrate community was severely degraded at this site. From the late 1970s through the early 1980s, ICI scores were in the poor/fair performance category (Figure 15). An intensive survey in 1984 revealed biological impairment that extended from the Akron area all the way to Cleveland. The response patterns of both the fish and macroinvertebrate communities, a lack of observed exceedences of conventional chemical water quality criteria, and the fact that degradation extended for over 20 mi (32 km) downstream strongly suggested a persistent toxic impact. Bioassays of the Akron wastewater treatment plant effluent as late as 1985 confirmed the presence of acute toxicity. Additionally, combined and sanitary sewer releases were a problem upstream from the treatment facility at many overflow locations in the Akron urban area.

In 1986, acute toxicity in the treatment plant effluent essentially disappeared; chronic toxicity was not measured. Although the principal reason for this was not precisely known, a number of possibilities exist including the removal of several industrial sources to the collection system due to plant closures, additional pretreatment requirements of remaining industrial inputs to the plant, more stringent regulation of illegal "drop-in" dischargers, and major upgrades at the plant, both on-line and under construction. The latter upgrades resulted in decreased loadings of ammonia, suspended solids, biochemical oxygen demand, and, since 1988, heavy metals. Plant bypassing of raw or partially treated sewage has also decreased. As can be seen in the trend at the fixed station, a significant improvement has occurred in the quality of the macroinvertebrate community as reflected by the ICI (Figure 15). Beginning in 1987 and continuing to the present, annual sampling has documented attainment of the ICI ecoregional biocriterion at this site. Intensive surveys in the basin as recently as 1991 have documented substantial biological improvements in the entire reach of the river between Akron and Cleveland (Ohio EPA 1992d). Although the severity of degradation has decreased, biological improvement has been limited in closer proximity to the wastewater treatment plant and urban area. There remain serious problems with discharges of untreated combined and sanitary sewer overflows and frequent bypassing of partially treated secondary flows at the plant.

6.0 FUTURE CONSIDERATIONS FOR ICI DEVELOPMENT AND APPLICATION

6.1 Reference Site Resampling

Calibration of ICI metric scoring categories are based on the prevailing background conditions at 246 least impacted reference sites that were sampled across Ohio from 1981 to 1986. This follows the guidance of Hughes et al. (1986) and recognizes that attainable biological community structure and function in aquatic systems is influenced by widespread activities such as intensive land surface uses (e.g., row-crop agriculture, surface mining), natural stream channel alterations (e.g., channelization), human settlement, road and highway construction, and general land surface conversion (e.g., deforestation), all to suit socioeconomic desires. The use of least impacted reference sites is not intended to represent pristine, wilderness, or pre-Columbian conditions in Ohio but recognizes that the aforementioned factors have collectively or individually influenced the ability of watersheds to support a certain level of biological performance. This does not mean that the impacts from these land use activities are necessarily acceptable; however, metric scoring reflects what is reasonably attainable in Ohio rivers and streams. The calibration of metric scoring categories can and will change (i.e., become more stringent) if it becomes apparent that the impacts of these pervasive influences have lessened through improved nonpoint source control programs or other means.

To determine if the background reference condition has changed significantly, resampling of reference sites has become an Ohio EPA monitoring program priority. Between 1988 and 1993, over 100 macroinvertebrate reference sites were resampled. It is anticipated that enough sites will have been resampled by the late 1990s (based on a goal of resampling 10% of the reference sites each year) to reexamine the ten ICI metrics and determine if a scoring recalibration is in order. At the same time, it can be determined if refinements and advancements in macroinvertebrate taxonomy have been sufficient enough to warrant further adjustments to ICI scoring categories or the metrics themselves. Modifications of the metrics or scoring categories and any subsequent changes in ecoregional expectations will be subject to the requirements of the Ohio Water Quality Standards rule-making process and final approval by the U.S. Environmental Protection Agency.

6.2 Qualitative Community Tolerance Values

A new Ohio EPA assessment tool currently in use and a direct offshoot of the ICI is the Qualitative Community Tolerance Value (QCTV). The QCTV is envisioned as having application when a quick turnaround is needed to problem assessment or when a screening-level, less definitive technique is desired in lieu of the more complex ICI process. As such, the method utilizes the qualitative, natural substrate collection procedure, which necessitates one site visit and minimal laboratory analysis. The assessment of the qualitative data relies on taxon tolerance values that have been established for many macroinvertebrates collected in Ohio. The tolerance value of a given taxon was derived using numerical abundance data from sites around Ohio where that taxon had been collected with artificial substrates. To determine the tolerance value, ICI scores at all locations where the taxon was collected were weighted by the abundance data of that taxon. The mean of the weighted ICI scores for the taxon is its tolerance value. Thus, the tolerance value represents the relative level of tolerance of a particular macroinvertebrate taxon in terms of the 0 to 60 scale of the ICI. The most pollution-intolerant taxa, which tend to reach greatest abundance at undisturbed sites (i.e., sites with highest ICI scores), yielded high tolerance values. Conversely, the most pollution-tolerant taxa, attaining greatest abundance at highly disturbed sites with low ICI scores, resulted in lower tolerance values.

At a sampling location where only qualitative, natural substrate data has been collected, the QCTV score is determined using only those taxa for which QCTV values have been calculated. This provides a link to the ICI that cannot be calculated for natural substrate data. The QCTV is most commonly expressed as the median of the available tolerance values but can also be expressed at other frequency intervals such as the 25th and 75th percentiles. Thus examining the community performance at different percentiles of the QCTV increases the dimension of the analysis. This was recently demonstrated in an assessment of small urban and suburban streams in the greater Cincinnati, Ohio area (Ohio EPA 1992).

Currently under evaluation are interim performance criteria for the median QCTV determined by correlating QCTV scores with sites achieving and not achieving ecoregional expectations as determined by ICI biocriteria. Though still subject to revision, the QCTV procedure has shown promising potential as an additional Ohio EPA assessment tool that can be used under specified circumstances. It is currently being used to evaluate qualitative data in conjunction with other more traditional sample attributes such as total and EPT richness and overall community composition and balance. A list of tolerance values (i.e., average weighted ICIs) derived for commonly collected Ohio macroinvertebrate taxa is provided (Table 5).

ACKNOWLEDGMENTS

The author wishes to recognize and thank his fellow "bug pickers" at the Ohio EPA — Jack Freda, Mike Bolton, Chuck McKnight, Bernie Counts, and Marty Knapp. Their long years of dedication and effort contributed significantly to the success of the program. This work is dedicated to the memory of J. Pat Abrams whose years of service laid the foundation for our efforts.

Table 5. Tolerance Values for 314 Common Macroinvertebrate Taxa Derived Using the Invertebrate Community Index (ICI) Weighted by Abundance Data and Averaged Over All Sites (N ≥ 5) where Collected with Modified Hester-Dendy Multiple-Plate Artificial Substrate Samplers

Taxon	Tolerance value	N
Coelenterata		
Cordylophora lacustris	22.5	24
Hydra sp.	33.5	707
Platyhelminthes		
Turbellaria	23.0	805
Nematomorpha		
Unidentified	22.0	6
Bryozoa		
Fredericella sp.	45.2	5
Fredericella indica	40.9	6
Hyalinella punctata	43.3	20
Lophopodella carteri	42.6	11
Paludicella articulata	41.7	29
Pectinatella magnifica	25.2	6
Plumatella sp.	37.4	591
Entoprocta		
Urnatella gracilis	42.6	162
Annelida		
Oligochaeta		
Unidentified	11.9	1289
Hirudinea		
Dina sp.	28.6	8
Erpobdella punctata punctata	8.8	28
Helobdella stagnalis	10.8	44
Helobdella triserialis	12.6	60
Mooreobdella fervida	8.0	11
Mooreobdella microstoma	14.3	66
Placobdella ornata	27.1	9
Arthropoda		
Crustacea		
Isopoda		
Caecidotea sp.	18.8	231
Lirceus sp.	35.1	136
Amphipoda		
Crangonyx sp.	20.2	151
Gammarus sp.	15.8	10
Gammarus fasciatus	18.7	45
Hyalella azteca	26.2	129
Decapoda		
Orconectes sp.	30.9	58
Orconectes (*Crokerinus*) *propinquus*	35.8	6
Orconectes (*C.*) *sanbornii sanbornii*	32.2	54
Orconectes (*Procericambarus*) *rusticus*	32.3	114
Arachnoidea		
Hydracarina	38.1	437
Insecta		
Ephemeroptera		
Siphlonuridae		
Isonychia sp.	48.7	617
Baetidae		
Acerpenna macdunnoughi	53.1	9
Acerpenna pygmaeus	46.3	66
Baetis sp.	41.1	928
Baetis armillatus	50.9	5
Baetis flavistriga	42.6	105
Baetis intercalaris	43.8	165
Callibaetis sp.	26.1	27
Centroptilum sp.	43.5	98
Cloeon sp.	38.7	137
Diphetor hageni	46.0	16
Pseudocloeon sp.	49.5	55
Heptageniidae		
Heptagenia diabasia	48.3	5
Leucrocuta sp.	45.8	246
Leucrocuta hebe	43.8	18
Leucrocuta maculipennis	47.5	15
Nixe perfida	29.2	9
Stenacron sp.	41.1	1019
Stenonema exiguum	48.4	204
Stenonema femoratum	41.5	450
Stenonema mediopunctatum	48.7	99
Stenonema mexicanum integrum	44.5	307
Stenonema pulchellum	46.3	484
Stenonema terminatum	46.4	455
Stenonema vicarium	46.2	193
Leptophlebiidae		
Choroterpes sp.	35.9	8

Table 5 (continued). Tolerance Values for 314 Common Macroinvertebrate Taxa Derived Using the Invertebrate Community Index (ICI) Weighted by Abundance Data and Averaged Over All Sites (N ≥ 5) where Collected with Modified Hester-Dendy Multiple-Plate Artificial Substrate Samplers

Taxon	Tolerance value	N
Paraleptophlebia sp.	43.4	225
Ephemerellidae		
Dannella simplex	52.9	5
Ephemerella sp.	42.5	9
Eurylophella sp.	47.5	45
Serratella sp.	56.3	5
Serratella deficiens	47.4	52
Tricorythidae		
Tricorythodes sp.	43.7	690
Caenidae		
Caenis sp.	41.6	976
Potamanthidae		
Anthopotamus sp.	40.6	52
Ephemeridae		
Ephemera sp.	41.4	82
Hexagenia sp.	38.1	11
Hexagenia limbata	44.2	16
Odonata		
Calopterygidae		
Calopteryx sp.	37.9	217
Hetaerina sp.	43.0	116
Coenagrionidae		
Coenagrionidae	23.2	297
Argia sp.	29.2	728
Aeshnidae		
Boyeria vinosa	38.7	64
Gomphidae		
Gomphus sp.	44.8	10
Plathemis lydia	10.5	5
Plecoptera		
Pteronarcyidae		
Pteronarcys sp.	46.7	7
Perlidae		
Acroneuria carolinensis	45.9	6
Acroneuria evoluta	42.9	73
Acroneuria internata	47.0	10
Acroneuria lycorias	45.3	8
Agnetina sp.	49.0	50
Neoperla sp.	41.4	11
Paragnetina media	47.4	27
Perlesta sp.	32.9	16
Megaloptera		
Sialidae		
Sialis sp.	30.6	201
Corydalidae		
Corydalus cornutus	45.7	312
Nigronia fasciatus	49.0	5
Nigronia serricornis	38.5	52
Trichoptera		
Philopotamidae		
Chimarra aterrima	48.5	19
Chimarra obscura	39.3	127
Psychomyiidae		
Lype diversa	43.1	73
Psychomyia flavida	48.5	16
Polycentropodidae		
Cyrnellus fraternus	36.1	179
Neureclipsis sp.	43.8	99
Nyctiophylax sp.	45.1	72
Polycentropus sp.	39.7	144
Hydropsychidae		
Cheumatopsyche sp.	42.5	1197
Hydropsyche (*Ceratopsyche*) *morosa*	49.1	19
Hydropsyche (*C.*) *morosa* group	44.6	747
Hydropsyche (*C.*) *slossonae*	48.0	62
Hydropsyche (*C.*) *sparna*	40.6	58
Hydropsyche (*Hydropsyche*) *aerata*	44.5	27
Hydropsyche (*H.*) *bidens*	45.2	179
Hydropsyche (*H.*) *depravata* group	39.4	404
Hydropsyche (*H.*) *dicantha*	39.2	293
Hydropsyche (*H.*) *frisoni*	48.9	54
Hydropsyche (*H.*) *orris*	44.2	215
Hydropsyche (*H.*) *simulans*	45.3	304
Hydropsyche (*H.*) *valanis*	41.8	153
Hydropsyche (*H.*) *venularis*	42.3	52
Macrostemum zebratum	49.2	64
Potamyia flava	45.2	223
Hydroptilidae		
Agraylea sp.	25.6	6
Dibusa angata	44.5	15
Hydroptila sp.	40.7	488
Neotrichia sp.	38.3	9
Oxyethira sp.	28.5	7
Limnephilidae		
Pycnopsyche sp.	40.0	32
Helicopsychidae		
Helicopsyche borealis	49.1	16
Leptoceridae		
Ceraclea sp.	44.4	41
Ceraclea maculata	38.4	19
Nectopsyche sp.	45.8	20
Nectopsyche diarina	41.0	9
Nectopsyche pavida	47.1	15
Oecetis sp.	42.2	71
Lepidoptera		
Petrophila sp.	43.6	92
Coleoptera		
Gyrinidae		
Dineutus sp.	38.0	39
Gyrinus sp.	38.1	12
Haliplidae		
Peltodytes sp.	17.7	5
Dytiscidae		
Hydroporus sp.	35.6	8
Laccophilus sp.	12.3	8
Hydrophilidae		
Berosus sp.	20.0	223
Laccobius sp.	48.5	6
Psephenidae		
Ectopria nervosa	36.2	16
Psephenus herricki	38.9	41
Dryopidae		
Helichus sp.	40.7	111
Lutrochus laticeps	42.7	9
Elmidae		
Ancyronyx variegata	35.7	414
Dubiraphia sp.	31.5	226

Table 5 (continued). Tolerance Values for 314 Common Macroinvertebrate Taxa Derived Using the Invertebrate Community Index (ICI) Weighted by Abundance Data and Averaged Over All Sites ($N \geq 5$) where Collected with Modified Hester-Dendy Multiple-Plate Artificial Substrate Samplers

Taxon	Tolerance value	N
Dubiraphia bivittata	41.1	16
Dubiraphia quadrinotata	40.9	8
Dubiraphia vittata group	37.8	316
Macronychus glabratus	42.8	886
Optioservus sp.	38.3	18
Optioservus fastiditus	48.4	10
Stenelmis sp.	41.7	870
Diptera		
Tipulidae		
Antocha sp.	46.9	75
Hexatoma sp.	45.9	15
Tipula sp.	17.0	54
Tipula abdominalis	36.1	23
Psychodidae		
Pericoma sp.	26.6	8
Psychoda sp.	37.6	14
Simuliidae		
Simulium sp.	31.3	574
Ceratopogonidae		
Ceratopogonidae	25.8	307
Atrichopogon sp.	32.9	11
Atrichopogon websteri	33.7	9
Chironomidae		
Tanypodinae		
Ablabesmyia sp.	28.6	32
Ablabesmyia annulata	35.1	5
Ablabesmyia janta	30.7	7
Ablabesmyia mallochi	30.1	388
Ablabesmyia rhamphe group	21.4	90
Clinotanypus pinguis	15.0	13
Conchapelopia sp.	34.3	500
Hayesomyia senata or *Thienemannimyia norena*	32.2	706
Helopelopia sp.	36.8	285
Labrundinia sp.	36.9	112
Labrundinia pilosella	41.5	123
Larsia sp.	34.2	69
Meropelopia sp.	43.5	73
Natarsia sp.	24.6	20
Natarsia species A (sensu Roback 1978)	11.6	37
Natarsia baltimoreus	26.5	5
Nilotanypus fimbriatus	43.2	453
Paramerina fragilis	31.7	6
Pentaneura inconspicua	30.8	60
Procladius sp.	20.9	160
Psectrotanypus sp.	11.5	10
Psectrotanypus dyari	8.2	8
Rheopelopia paramaculipennis	42.8	66
Tanypus sp.	7.3	7
Tanypus neopunctipennis	14.8	8
Telopelopia okoboji	36.1	40
Thienemannimyia group	29.6	526
Zavrelimyia sp.	35.6	28
Orthocladiinae		
Brillia flavifrons group	33.2	78
Cardiocladius obscurus	46.6	28
Corynoneura sp.	38.9	114
Corynoneura n. sp 1	46.2	31
Corynoneura "celeripes" (sensu Simpson and Bode 1980)	45.9	120
Corynoneura lobata	40.0	536
Cricotopus (C.) sp.	34.4	488
Cricotopus (C.) bicinctus	19.7	569
Cricotopus (C.) tremulus group	32.4	477
Cricotopus (C.) trifascia group	19.0	33
Cricotopus (C.) vierriensis	34.6	16
Cricotopus (Isocladius) intersectus group	26.7	11
Cricotopus (I.) sylvestris group	11.9	26
Doncricotopus prob. *bicaudatus*	30.9	11
Limnophyes sp.	41.3	5
Nanocladius (N.) sp.	28.2	97
Nanocladius (N.) crassicornus	41.9	6
Nanocladius (N.) crassicornus or *N. (N.) rectinervus*	34.1	31
Nanocladius (N.) distinctus	23.1	425
Nanocladius (N.) minimus	33.2	52
Nanocladius (N.) spiniplenus	38.7	71
Nanocladius (Plecopteracoluthus) n. sp.	48.6	11
Orthocladius sp.	36.5	6
Orthocladius (O.) sp.	43.5	9
Orthocladius (O.) carlatus	48.0	5
Parakiefferiella sp.	31.0	28
Parakiefferiella n. sp 1	39.5	56
Parakiefferiella n. sp 2	35.6	46
Parametriocnemus sp.	43.0	119
Paratrichocladius sp.	48.1	7
Psectrocladius sp.	25.9	8
Rheocricotopus sp.	30.4	6
Rheocricotopus (Psilocricotopus) robacki	37.1	325
Synorthocladius semivirens	44.8	5
Thienemanniella sp.	38.0	7
Thienemanniella n. sp 1	46.9	82
Thienemanniella n. sp 3	41.6	125
Thienemanniella similis	45.6	43
Thienemanniella xena	38.2	418
Tvetenia bavarica group	46.4	48
Tvetenia discoloripes group	46.4	103
Chironominae		
Chironomini		
Chironomus (C.) sp.	11.4	98
Chironomus (C.) decorus group	12.8	230
Chironomus (C.) riparius group	5.5	77
Cryptochironomus sp.	30.7	177
Cryptotendipes sp.	34.8	8
Dicrotendipes sp.	26.6	53
Dicrotendipes modestus	24.8	12
Dicrotendipes fumidus	25.4	5
Dicrotendipes neomodestus	32.7	598
Dicrotendipes lucifer	21.9	202
Dicrotendipes simpsoni	15.8	199
Endochironomus sp.	26.2	5
Endochironomus nigricans	28.3	24
Glyptotendipes (Phytotendipes) sp.	22.5	568

Table 5 (continued). Tolerance Values for 314 Common Macroinvertebrate Taxa Derived Using the Invertebrate Community Index (ICI) Weighted by Abundance Data and Averaged Over All Sites (N ≥ 5) where Collected with Modified Hester-Dendy Multiple-Plate Artificial Substrate Samplers

Taxon	Tolerance value	N
Glyptotendipes (*G.*) *amplus*	40.8	21
Harnischia curtilamellata	38.8	7
Kiefferulus dux	28.7	11
Microtendipes "caelum" (sensu Simpson and Bode 1980)	43.7	70
Microtendipes pedellus group	39.5	289
Microtendipes rydalensis	48.0	6
Nilothauma sp.	41.1	16
Parachironomus sp.	35.2	10
Parachironomus abortivus	11.9	48
Parachironomus carinatus	28.0	39
Parachironomus directus	12.5	5
Parachironomus frequens	36.9	98
Parachironomus pectinatellae	37.9	23
Paralauterborniella nigrohalteralis	36.5	36
Paratendipes sp.	25.7	12
Paratendipes albimanus	34.0	236
Phaenopsectra obediens group	33.1	310
Phaenopsectra punctipes	39.1	11
Phaenopsectra flavipes	25.8	133
Polypedilum (*Pentapedilum*) *tritum*	13.4	6
Polypedilum (*Polypedilum*) *albicorne*	37.0	14
Polypedilum (*P.*) *aviceps*	48.4	14
Polypedilum (*P.*) *convictum*	38.6	972
Polypedilum (*P.*) *fallax group*	30.6	945
Polypedilum (*P.*) *illinoense*	18.4	449
Polypedilum (*P.*) *ophioides*	45.3	7
Polypedilum (*P.*) *n.*sp. same as *tuberculum* (*Maschwitz*, 1976)	48.4	8
Polypedilum (*P.*) *ontario*	44.4	12
Polypedilum (*Tripodura*) *halterale* group	37.2	31
Polypedilum (*Tripodura*) *scalaenum* group	26.1	871
Stenochironomus sp.	38.2	75
Stictochironomus sp.	36.1	36
Tribelos fuscicorne	29.3	67
Tribelos jucundum	26.5	79
Pseudochironomini		
Pseudochironomus sp.	31.9	35
Tanytarsini		
Cladotanytarsus sp.	33.5	76
Cladotanytarsus species *group A*	39.5	14
Cladotanytarsus mancus group	37.1	20
Cladotanytarsus vanderwulpi group 1	46.2	32
Cladotanytarsus vanderwulpi group 5	48.4	12
Micropsectra sp.	38.9	57
Paratanytarsus sp.	34.9	377
Paratanytarsus n. sp. 1	43.6	31
Rheotanytarsus sp.	39.9	41
Rheotanytarsus distinctissimus group	47.0	179
Rheotanytarsus exiguus group	43.8	1034
Stempellinella sp.	44.1	46
Stempellinella n. sp. nr. *flavidula*	40.1	24
Sublettea coffmani	47.0	45
Tanytarsus sp.	38.9	298
Tanytarsus Type 1	43.6	7
Tanytarsus Type 3	37.1	32
Tanytarsus curticornis group	44.4	126
Tanytarsus glabrescens group	40.4	818
Tanytarsus guerlus group	40.6	506
Tabanidae		
Chrysops sp.	32.6	16
Athericidae		
Atherix lantha	41.3	74
Empididae		
Empididae	39.2	1002
Mollusca		
Gastropoda		
Hydrobiidae		
Hydrobiidae	17.7	49
Pleuroceridae		
Elimia sp.	38.1	248
Pleurocera sp.	37.8	6
Lymnaeidae		
Lymnaeidae	18.4	5
Fossaria sp.	44.6	11
Physidae		
Physella sp.	14.3	536
Planorbidae		
Gyraulus (*Torquis*) *parvus*	27.0	12
Helisoma anceps anceps	24.0	16
Menetus (*Micromenetus*) *dilatatus*	11.2	32
Planorbella (*Pierosoma*) *pilsbryi*	15.4	16
Planorbella (*Pierosoma*) *trivolvis*	3.1	5
Ancylidae		
Ferrissia sp.	28.7	772
Laevapex fuscus	19.7	13
Pelecypoda		
Corbiculidae		
Corbicula fluminea	40.8	89
Sphaeriidae		
Pisidium sp.	32.5	63
Sphaerium sp.	31.5	155