The Great Lakes Entomologist

Volume 27
Number 3 - Fall 1994 *Number 3 - Fall 1994*

Article 1

October 1994

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EXTRACTING ADDITIONAL INFORMATION FROM BIOTIC INDEX SAMPLES

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ABSTRACT

Macroinvertebrates were collected from a small midwestern stream over a 3-year period as part of a non-point source pollution study. Temporal and spatial variability in standard biotic index values (BIs) were computed and compared with variability expressed by a series of additional community measurements, including the mean tolerance value of all taxa present in a sample, irrespective of the numerical abundance of individual taxa. The mean tolerance value exhibited lower spatial and temporal variability than the standard BI; therefore, mean tolerance values may be useful in estimating a stream's long-term ambient water quality and its recovery potential. Computations of additional BI metrics are easily accomplished with no additional lab work required, and comparisons of mean tolerance values with standard BIs should aid investigators in interpreting changes in water quality.

Macroinvertebrates are an important component of the rapid bioassessment protocols for water quality assessment presented by the U.S. Environmental Protection Agency (Plafkin et al. 1989). The recommended protocols include several indices that are based on species richness, diversity, or community composition of benthic macroinvertebrates. The Hilsenhoff Biotic Index (HBI)(Hilsenhoff 1977, 1982, 1987), a modification of Chutter's (1972) biotic index, has proven particularly popular and reliable in detecting impacts of organic pollution on water quality. Essentially, the HBI represents the average pollution tolerance of a randomly-selected subset of more than 100 macroinvertebrate organisms (arthropods) collected from riffles or runs in a particular river or stream. The HBI, or modification thereof, is a principal method of rapid bioassessment protocols II and III of the U.S. EPA (Plafkin et al. 1989). Rapid bioassessment protocol III requires macroinvertebrates be identified to either genus or species level (where possible). The degree of environmental degradation at a site is based on relative comparison with complementary data from a nearby reference site (Plafkin et al. 1989). If reference data are lacking, replication provides an estimate of variability in HBI values, thereby permitting statistical comparisons among other stations or dates. Because this level of analysis is labor intensive, it is desirable to extract as much information as possible from the resultant data. In this paper, we present a method to extract supplemental information from HBI samples without requiring that additional labwork be performed. A new index, repre-

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senting the mean pollution tolerance value of all taxa present in an HBI sample, irrespective of the number of individuals represented by each taxon, offers promise as a complement to the HBI. The mean tolerance value is compared with the standard HBI using data collected over a 3-yr period from a small southwestern Wisconsin stream.

METHODS

Rattlesnake Creek is a medium-sized, warmwater stream located in southwestern Wisconsin with a recent history of fishkills (Mason et al. 1991). Periodic episodes of depressed dissolved oxygen concentrations during summer rainstorms have been documented in Rattlesnake Creek (Graczyk and Sonzogni 1991), and these storm-related events are believed to have had an adverse impact on stream biota (Graczyk 1993a). This paper is based on benthic surveys conducted over a 3 yr period during an intensive non-point source pollution survey of Rattlesnake Creek (Graczyk 1993a). The hydrologic regime during the period that macroinvertebrate surveys were conducted was relatively stable, coinciding with a period of extreme drought. No major run-off events or extended periods of depressed dissolved oxygen concentrations were observed.

Benthic samples were collected by two independent teams of investigators using different sampling strategies. One team collected three replicates from a riffle adjacent to a United States Geological Survey (USGS) gaging station on six dates—fall 1987, spring 1988, fall 1988, spring 1989, fall 1989, and spring 1990. These samples were intended to correspond with water quality data collected by automated monitoring equipment at the gaging station. Another team collected benthic samples from six riffle sites on three dates—fall 1987, spring 1988, and fall 1989. The latter set of samples, spaced at irregular intervals, was intended to monitor water quality in stream reaches of Rattlesnake Creek concurrent with fisheries investigations. Both teams collected field samples in accordance with standard kick-net procedures (Hilsenhoff 1987). Macroinvertebrate samples were preserved in 95% ethanol and returned to the laboratory for processing. Samples collected by the first team were processed at the University of Wisconsin-Stevens Point, and samples collected by the second team were processed by the WDNR. Both sets of samples were processed following procedures established by Hilsenhoff (1987); chironomids were identified to genus only. Standard biotic index values were computed for all data sets based on the number and corresponding tolerance value of all individuals present in a random subsample of at least 100 individuals (Hilsenhoff 1987). These values are commonly referred to as Hilsenhoff Biotic Index values (HBIs). Additionally, the mean tolerance value of each HBI data set was computed as follows:

Mean Tolerance Value = SUM t_i/ T
where t_i represents the assigned pollution tolerance value for each taxon, and T represents the number of taxa in the sample.

The mean tolerance value gives equal weight to each taxon in a sample irrespective of its numerical abundance in the sample and, therefore, rare taxa are more important in calculating the mean tolerance value than in calculating the HBI, which is dependent upon the numerically dominant taxa. In streams of poor water quality, the mean tolerance value places increased emphasis on the intolerant forms, which generally are less abundant than tolerant forms in organically enriched streams. The patterns exhibited by HBI and mean toler-

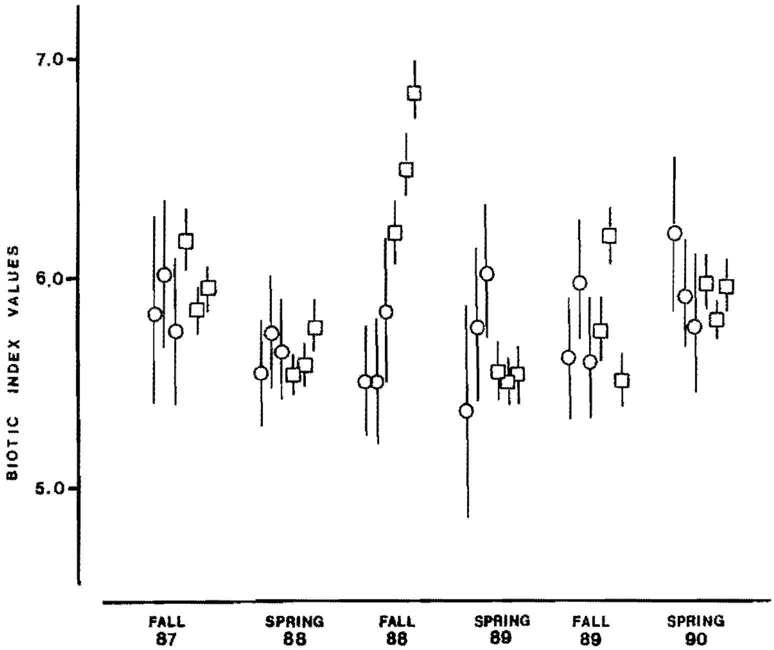

Figure 1. Temporal variations in biotic index values (squares represent HBIs, circles represent mean tolerance values) at the USGS gaging station (3 replicates each date). Vertical lines represent ± 1 SE based on distribution of pollution tolerance values of individual taxa or all organisms in each sample.

ance value data were examined visually to identify outliers and irregularities in distribution.

RESULTS AND DISCUSSION

The period of macroinvertebrate sampling on Rattlesnake Creek, October 1987 to May 1989, coincided with a severe drought in the upper Midwest. Both hydrologic and sediment loadings were much reduced. Dissolved oxygen concentrations never dropped below 1 mg/L in Rattlesnake Creek during this period (Graczyk 1993b). Other biological measurements, including total taxa richness, Ephemeroptera-Plecoptera-Trichoptera taxa richness, and abundances indicated either stable or steadily improving water quality (Lillie and Schlesser 1993). Biotic index values also were quite stable, except for high HBIs displayed in the fall 1988 samples (Fig. 1). This abrupt increase in HBIs, which suggested that a decline in water quality had occurred, was not accompanied by a corresponding increase in mean tolerance values. HBIs were substantially higher than corresponding mean tolerance values by an average 0.90 units. All samples were collected within the recommended time window for sampling warmwater streams (Hilsenhoff 1988), so no seasonal adjust-

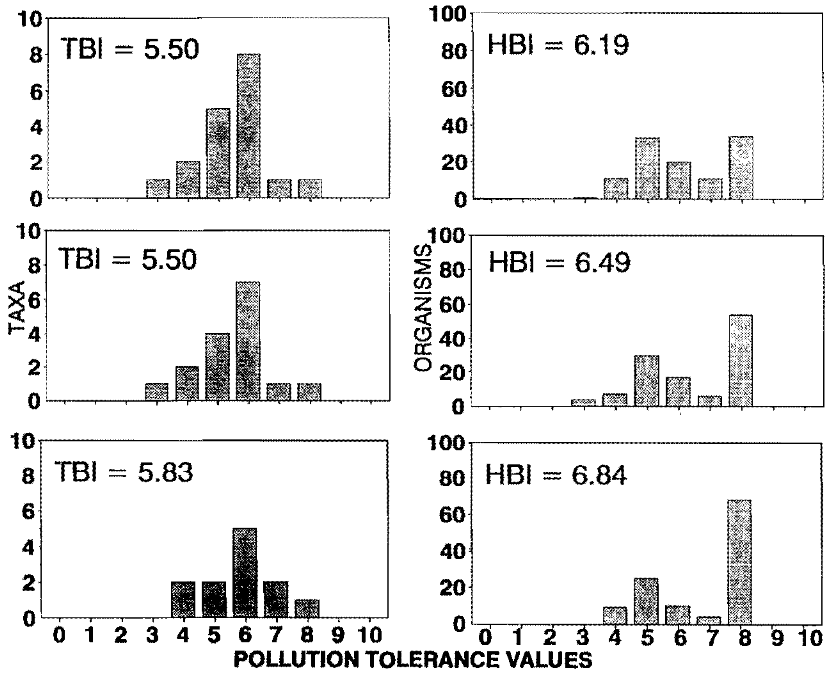

Figure 2. Histograms illustrating the distribution of pollution tolerance values among taxa (mean tolerance values or TBIs, left-hand set) and individuals (HBIs, right-hand set) comprising fall 1988 biotic index samples.

ments of the data were warranted. Closer examination of the data revealed large numbers of the isopod *Caecidotea intermedia* (Forbes) (=*Asellus intermedius* per Jass and Klausmeier 1990) were dominant on this date in all 3 replicates (Fig. 2). *Caecidotea intermedia*, with a tolerance value of 8, had a major influence on the HBI. However, *C. intermedia* was the only taxon in the samples with a tolerance value of 8; the mean tolerance values were dominated by taxa with tolerance values of 5 and 6. Consequently, mean tolerance values were substantially lower than HBIs on this date. HBIs and mean tolerance values (means of the 3 replicates) were not statistically different on the remaining 5 dates. Excluding the fall 1988 data, the average difference between matching sample HBIs and mean tolerance values was ± 0.19 units (N=15). The average discrepancy among dates was ± 0.09 units, with a net bias of −0.03 units for the 5 dates (i.e. mean tolerance values were slightly higher than HBIs by an average of 0.03 units).

The question arises, therefore, as to which measurement more closely represents true water quality conditions. Most biologists would agree that the occurrence of large numbers of isopods in a sample is indicative of generally poor water quality. However, the high degree of skewedness exhibited in the pattern of tolerance values in the HBI (Fig. 2), combined with lower HBIs in spring 1988 and 1989 samples, and stability or improvements in other water quality measures in Rattlesnake Creek during the same time period (Lillie and Schlesser 1993), suggest that the fall 1988 HBI values were unduly influenced by the large numbers of *C. intermedia* present in the sample. Additionally, HBIs from concurrent studies of three nearby Wisconsin streams did not exhibit similar rises during the fall of 1988 (Lillie and Schlesser 1993). This seems to rule out any seasonal or climatic influence on the fall 1988 data in Rattlesnake Creek. Without knowledge of other water quality data, biologists likely would have concluded, perhaps wrongly, that water quality declined from spring to fall 1988 in Rattlesnake Creek. Conversely, perhaps the HBI is more sensitive to certain forms of organic pollution than are other measured water quality indicators, and water quality of Rattlesnake Creek did indeed experience some form of event between spring 1988 and fall 1988. Certainly, the high abundance of isopods in the fall 1988 samples should signify something. Perhaps the increase in isopod abundance reflected some change in physical habitat rather than a change in water quality (i.e. Lenat 1988). If so, the mean tolerance value was not responsive to the change. Under the circumstances, it appears that biologists should examine the patterns displayed by tolerance values in HBI samples for skewedness or other abnormalities in distribution and, if detected, consider the corresponding mean tolerance value as more representative of long-term water quality. Similarly, a bimodal distribution pattern of tolerance values may suggest a confluence of two streams of differing water quality or influences of side tributaries. We do not suggest that the mean tolerance value be used in lieu of, or as a substitute for, the HBI, but rather that the mean tolerance value should be used in conjunction with existing HBI data in the interpretation of water quality.

Spatial Comparisons: Another example of the possible utility of mean tolerance values is described using data to examine spatial trends in Rattlesnake Creek (Fig. 3). Trends exhibited by mean tolerance values and HBIs were generally similar to one another on each date. However, mean tolerance values were substantially lower than corresponding HBIs on two of three dates. Mean tolerance values were an average of 0.92 and 0.70 units lower than corresponding HBIs during fall 1987 and spring 1988, respectively. Differences between mean tolerance values and HBIs were relatively consistent among all six sampling sites on these two dates. We can offer no explanation for these deviations, other than to note that tolerance value patterns of HBIs were highly skewed towards individuals with high tolerance values.

Mean tolerance values were similar to HBIs during fall 1989 (average discrepancy ± 0.28 units; average or net bias, mean tolerance values were 0.10 units lower than HBIs; N=6). The maximum discrepancy occurred at site F where the mean tolerance value was 0.78 units lower than the HBI. The occurrence of large numbers of the isopod *Caecidotea intermedia* was again the cause for the disparity. In comparison with the histograms exhibited on the other two sampling dates, the fall 1989 HBI patterns exhibited less skewedness. The greater similarity between mean tolerance values and HBIs in the fall 1989 may indicate greater instream stability associated with the prolonged drought that continued throughout the study period.

Again, there is some question as to which attribute more closely represents true water quality. The average of the six mean tolerance values in this data set (Fig. 3) compares more closely with the average of the three mean tolerance values at the gaging station (compare with data in Fig. 1), than does the average HBIs between the two data sets compare. The two data sets were collected within 4 to 13 days of one another. During a four-day period in fall 87, the HBI increased by 0.42 units and the mean tolerance value decreased by 0.41 units at site F (the site closest to the USGS gaging station). HBIs increased substantially at the same site during 12–13 day spans in spring 88 and fall 89 (+1.25 and + 1.13 units, respectively). The average daily rate of

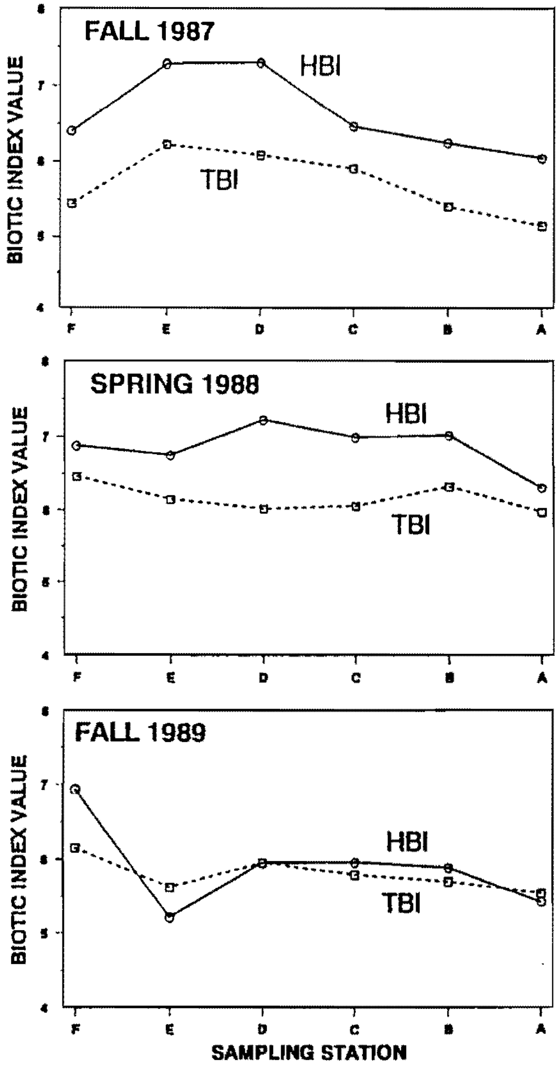

Figure 3. Comparison of spatial trends in biotic index values (HBIs) and mean tolerance values (TBIs) in Rattlesnake Creek on three sampling dates. Sampling stations are ordered from upstream (Site F) to downstream (Site A).

change in HBIs (+ 0.09–0.10 units/day) was similar on all three dates. The rate of change in mean tolerance values was smaller but inconsistent. Therefore, in our opinion, the mean tolerance values depicted in Figure 3 were more representative of true water quality conditions in fall 1987 and spring 1988 samples than HBIs. Clements (1991) suggested that because the number of taxa present at a given site may be less variable than abundance of individual taxa, numbers of taxa may have some advantages in monitoring invertebrate communities. The mean tolerance value supports this suggestion. Furthermore, it has been suggested that the disappearance of intolerant taxa may be more significant than changes in tolerant forms (Fausch et al. 1990). Tolerant taxa generally have a wider distribution ranges than intolerant forms. Although a tolerant taxa, with a tolerance value of 8 for example, may be present in abundant numbers, that same taxa may occur in waters with water quality equivalent to 5 or 6. The fact that several less tolerant taxa also are present at a site may be more indicative of the true water quality than the presence of one tolerant form with a wide range in pollution tolerance. Therefore, the mean tolerance values associated with biotic index samples may be a useful accessory metric in interpreting changes in water quality.

CONCLUSIONS

A companion metric to the standard HBI, the mean tolerance value, exhibits less temporal variability than the HBI. The mean tolerance value should be used in conjunction with the HBI to evaluate changes in community structure resulting from organic pollution. The mean tolerance value is not presented as a substitute for the HBI but, rather, is offered as a companion metric. The mean tolerance value gives equal weight to rare and dominant taxa and, consequently, may be less susceptible to short-term changes than the HBI. Thus, the mean tolerance value may have some advantages in long-term trend detection.

Some sudden, short-term changes in HBIs were observed in this study. Large population fluctuations in certain relatively ubiquitous taxa with assigned high pollution tolerance values may have had undue influence on HBIs. We suggest that some consideration be given to modifying assigned tolerance values of *Caecidotea intermedia* similar to that provided for *Simulium vittatum* (Hilsenhoff 1987).

Examination of the histogram patterns depicted by HBI data may prove useful in pollution studies. Bimodal patterns (i.e. many tolerant and intolerant taxa present with few intermediate taxa) may indicate junctions between streams of vastly different water quality. The extent and direction of skewedness in the patterns may provide clues as to the stability or biotic integrity of an invertebrate community at a particular site. The width of the distribution pattern, as measured by standard measures of variability (i.e. standard deviation. standard error, and coefficient of variation) may also provide useful information. In cases where replication of samples is lacking or not affordable, these conventional statistical measures may provide some indication of the representativeness of a particular set of biotic index data.

ACKNOWLEDGMENTS

This study was supported in part by funds from the Federal Aid in Sport Fish Restoration Act, Project F-83-R-23 and the Wisconsin Department of Natural Resources. Field assistance was provided by G. Wegner, G. Quinn, E.

Brick, J. Mason, and I. Marx-Olson. The authors thank W. Hilsenhoff, the editor, and several anonymous reviewers for their constructive suggestions.

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