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A New Method to Establish Scoring Criteria of the Index of Biotic Integrity

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Shih-Hsiung Liang and Bruce W. Menzel (1997) A new method to establish scoring criteria of the Index of Biotic Integrity. *Zoological Studies* **36**(3): 240-250. Karr's Index of Biotic Integrity (IBI) was applied in north central Iowa streams to examine the technique's utility for measuring environmental quality in agriculturally-influenced streams and to develop methodological improvement. During 1988 and 1989, fish were collected in the Boone River and Lizard Creek systems, and physical and chemical measurements of stream environmental conditions were taken. A modified two-step procedure was proposed for establishing the scoring standard of the 12 IBI metrics. First, a simple linear regression relationship of the metric variable on log drainage area was calculated (p = 0.1). If a significant correlation was found, a Maximum-Species-Richness Line (MSRL) was determined by calculating the 95% Y-intercept as its intercept, and then plotted by the original regression slope. If a nonsignificant relationship was obtained, the upper and lower 5% data points were deleted to define the upper and lower boundaries of the remaining points with horizontal lines. Finally, the area below the calculated 95% MSRL and the area between the two horizontal lines were equally trisected into above average (5 point), average (3 point), and below average (1 point) categories. After the scoring criteria were established, the IBI value of each sampling site in Boone River and Lizard Creek systems was calculated.

The new scoring procedure resolved the problem of the commonly adopted "eye-fit" method through generating only a single MSRL for a given data set. In Lizard Creek, IBI results were generally consistent with evaluations of physical and chemical environmental quality. During the drought period, variations of IBI scores reflected the dynamics of environmental conditions in the sampling sites. Additionally, the IBI showed greater temporal consistency than another commonly used analytical tool, the Shannon-Weiner Diversity Index. Thus, the new scoring procedure seems to advance the effectiveness of IBI in assessing the environment quality of lotic waters. Numerical species metrics associated with species diversity were more important than proportional ones in contributing to the IBI. Drought conditions throughout the study period influenced some of the data and analytical results.

Key words: Biological indicator, Index of Biotic Integrity, Environmental assessment, Stream fishes, Community ecology.

In response to the need of a sound biological monitoring system for flowing water, Karr (1981) proposed the Index of Biotic Integrity (IBI) by using midwestern U.S. fish assemblages. Twelve attributes (metrics) in three categories — species composition, trophic composition, and health and abundance of resident fish communities — are scored by comparison with those expected in the least human disturbance (Table 1). According to findings of Fausch et al. (1984) and Karr et al. (1986), when the raw data for the various metrics are plotted as a function of the logarithm of drainage area or stream order, the plot is expected to form a triangular shape. Its hypotenuse with slope fit by eye, that includes about 95% of the data points is arbitrarily accepted as the standard for the optimal condition. Karr et al. (1986) referred to this as the Maximum-Species-Richness Line

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(MSRL) when applied to those metrics involving enumeration of species. To determine metric scoring values, the area below the MSRL is then equally trisected to establish regions defining conditions which are above equal (5 points), somewhat below (3), or far below (1) the best condition. Finally, the scores of 12 metrics of the sampling site are added together, and the site is assigned it to one of six ecological integrity classes: excellent (58 – 60), good (48 – 52), fair (40 – 44), poor (28 – 34), very poor (12 – 22), and no fish.

During the last decade, results of applications of the IBI throughout North America and worldwide have verified that it is suitable for assessing a number of concerns for water quality (Leonard and Orth 1986, Oberdorff and Hughes 1992). Furthermore, it generally has performed with greater sensitivity and consistency than other ecological measurements, such as the Shannon-Wiener Diversity Index and the Index of Well Being (Hughes and Gammon 1987, Karr et al. 1987).

Although the IBI has advantages over other techniques, some limitations have been recognized as well. These include the need for a large data base, complications associated with habitat structure (Gorman and Karr 1978, Schlosser 1982), differing efficiencies of fish collecting methods (Mahon 1980), biological factors such as seasonal migrations and recruitment of fishes (Schlosser 1985), expertise in evaluating the data base, regional variations in the suitability of the various metrics (Fausch et al. 1984, Angermeier and Karr 1986, Hughes and Gammon 1987), and establishment of different scoring criteria for stream size or drainage area (Ohio EPA 1987). However, very few suggestions based on statistical principles have been made to improve the procedure for establishing IBI's scoring criteria, particularly for determination of the MSRL. Thus, the objectives of this study were to: 1. develop and evaluate a new procedure for establishing scoring criteria of IBI metrics, and to apply it in the study area; 2. evaluate the new scoring procedure through testing the IBI's spatial sensitivity and temporal consistency in the study streams, making comparisons with measures of chemical and physical attributes of test sites and with the Shannon-Weiner Diversity Index (H'); and 3. assess the contributions of individual IBI metrics.

MATERIALS AND METHODS

Study site

The flat area of north – central Iowa, known as the Des Moines Lobe, is one of the most intensively farmed areas in the USA, with corn and soybeans being the principal crops. Two study streams, Boone River and Lizard Creek, are sub-

Table 1. Metrics used to assess fish communities in the north-central Iowa streams (modified from Karr 1981 and Fausch et al. 1984). Metric abbreviations are listed in parentheses

Category	Metrics
Species richness and composition	
	1. Total number of fish species (SP)
	2. Number of sunfish species (Centrarchidae except *Micropterus*) (SNFR)
	3. Number of darter species (Percidae: Etheostominae) (DRTR)
	4. Number of sucker species (Catostomidae) (SUKR)
	5. Number of intolerant species (INSP)
	6. Proportion of individuals as green sunfish (*Lepomis cyanellus*) (GRN)
Trophic composition	
	7. Proportion of individuals as omnivores (OMNI)
	8. Proportion of individuals as insectivorous cyprinids (INSE)
	9. Proportion of individuals as piscivores (top carnivores) (PIS)
Fish abundance and condition	
	10. Number of individuals in sample (catch per minute) (CPUE)
	11. Proportion of individual as hybrids[a] (HYBI)
	12. Proportion of individuals with disease, tumors, fin damage, and skeletal anomalies (SICK)

[a]Because there was no information for this metric, 5 points were given to each sample.

drainages of the Des Moines River (Fig. 1). In this study, the Boone River system, with a drainage area of 2319 km^2, served as the "least disturbed area" upon which to establish standard criteria for application of the IBI. Although the lower reaches of Boone River are regarded as one of the least modified rivers in Iowa (Harlan et al. 1987), most of its headwaters have been channelized for agricultural drainage.

Over most of its length, the Lizard Creek system, with a drainage area of 1122 km^2, flows through an exposed, high-banked, ditched channel and receives subsurface water from numerous tile-line outlets which drain adjacent cropland (Fig. 1). The city of Pocahontas contributes treated wastewater effluent to the main branch of the creek. Near its mouth, the creek is semi-natural, where it meanders and is bordered by woodlands and pasture.

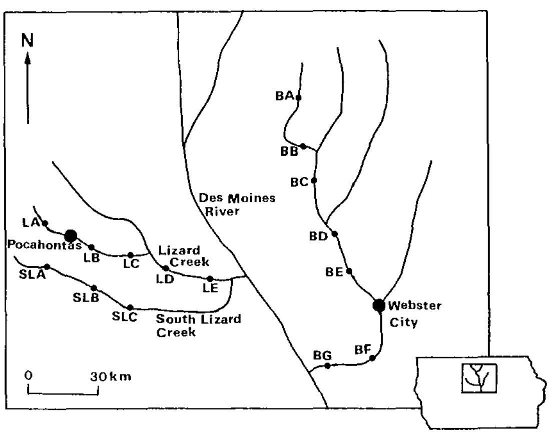

Fig. 1. Sampling area, streams and sites.

Environmental measurements

To compare general environmental characteristics of the two streams, water quality and discharge measurements were made at seven Boone River sites and eight Lizard Creek sites from June to October in both 1988 and 1989 (Fig. 1). IBI scoring criteria were based on fish samples from 21 Boone River sites. Application of the IBI was made for the eight water sampling sites in Lizard and South Lizard creeks. All Lizard Creek sites are more-or-less directly influenced by agricultural activity in their watersheds (Table 2). In the main section of Lizard Creek, sites LB and LC, immediately below the City of Pocahontas, served to test the utility of the IBI for measuring urban impacts on stream environmental quality.

Each 30-m-long sampling site was chosen to be representative of the area and included at least one riffle-pool-riffle sequence. Sampling sites were selected on the basis of a preliminary review of available data, advice from local conservationists, and field inspection. All sites were located away from habitat disturbance associated with bridges and at least 100 m away from conjoining streams.

Discharge measurements were made biweekly at three transects set at 7, 15, 22 m in each sampling site. Wetted stream width was measured to the nearest 1 cm with a fiberglass tape. Along each transect, stream depth and velocity were measured at quarter points. Depth was measured to the nearest 1 cm. Velocity (m/s) was taken as the average of two 30 second measurements made with a Weathermeasure Model F583 water current

Table 2. Environmental conditions and water quality of sampling sites in the Lizard Creek system from June to October 1988 and 1989

Site	Drainage area (km^2)	Stream substrate[a]	Buffer zone (m)[b]	Riparian vegetation[c]	Discharge (m^3/s)	Cond. ($\mu s/cm$)	Turb. (NTU)	NO_3-N (mg/l)	Total phosphate (mg/l)
				Lizard Creek					
LA	49	S	< 10	CS	0.01	535	13	15.7	0.13
LB	72	S	< 10	CS	0.05	1307	19	19.2	0.74
LC	169	S/G/C	< 10	CS	0.21	1064	31	17.1	0.44
LD	596	S/G/C	> 50	B/F	0.41	656	30	13.4	0.16
LE	666	S/G/C	> 100	B/F	1.07	533	35	12.5	0.09
				South Lizard Creek					
SLA	23	S	< 10	CS	0.03	678	71	14.1	0.17
SLB	169	S	< 10	CS	0.07	652	40	14.1	0.14
SLC	256	S	< 10	CS	0.09	559	45	14.9	0.13

[a]S: Sand, G: Gravel, C: Cobble.
[b]Distance between stream and adjacent farm land.
[c]CS: Corn and soybean, B: Bush (tree height < 2 m), F: Forest (tree height > 2 m).

meter. Discharge was calculated as the product of site average for stream width, depth, and velocity. Substrate features and riparian vegetation were visually estimated at all sampling sites.

An one-liter water sample for water quality analyses were taken at each sampling site and transported on ice to the Limnology Laboratory of the Department of Animal Ecology, Iowa State University where analyses were made within 24 h. Water conductivity was measured with a Hach conductivity meter (model: 2511). Turbidity was measured with a Hach turbidimeter (model: DR-A3321). Nitrate and total phosphate concentrations were determined with a Hach Nitrate-Nitrite Test Kit (model: N1-12) and Hach Total Phosphate Test Kit (model: PO-24), respectively.

Three problems interfered with environmental sampling and measurement taking during 1988. An analytical problem resulted in loss of some water quality data for Lizard Creek in June 1988. Bridge construction prevented sampling at site SLC in South Lizard Creek. Moreover, drought conditions prevailed in Iowa during 1988 and 1989 (National Oceanic and Atmospheric Administration 1988 1989). Because of drought conditions, water ceased flowing at site SLA in August 1988.

Fish sampling

A Coffelt model BP-4 backpack electrofisher with a maximum capacity of 600 V and 200 A was used for all fish collecting. Prior to sampling, block seines of 20-mm mesh were placed at the upper and lower ends of the site. Several electrofishing passes were then made throughout the length of the site or until fish were no longer caught. At the several downstream Boone River sites, block netting was not possible, and sampling consisted of 30 min of electrofishing time in all major habitats.

All collected fish were preserved in 10% formalin and returned to the laboratory for identification. Specimens were identified to species, examined for external signs of disease, parasites, injury, and anomalies, and measured to the nearest 0.1 cm in total length. Juveniles of total length less than 20 mm were arbitrarily excluded from tabulations.

Trophic categorization of species was determined from information in Carlander (1969 1977), Karr et al. (1986), and Pflieger (1975). Five intolerant species, hornyhead chub (*Nocomis biquttatus*), rosyface shiner (*Notropis rubellus*), smallmouth bass (*Micropterus dolomieu*), slenderhead darter (*Percina phoxocephala*), and Iowa darter (*Etheostoma exile*) were identified based on information from the literature, local fish biologists, and personal observations.

Establishment of the IBI's scoring criteria

Following recommendations of the Ohio EPA (1987), a simple linear regression relationship of the metric variable on log drainage area (mi^2) was calculated for establishing the maximum-species-richness line (MSRL). For declaring that a statistical correlation existed, the 0.1 level of significance was accepted. We assumed that the calculated regression line represents the true Boone River population line if a significant correlation was found. Theoretically, variance is the same along the length of the line, and the deviations from the group means are normally distributed (Steel and Torrie 1980). Thus, a MSRL which is predicted by the mean square error (MSE) as the population variance should parallel the calculated regression line. For application here, a MSRL was established by first determining the 95% Y-intercept by the formula:

$$V = I + t * (MSE/Y)^{1/2}$$

where V is the 95% Y-intercept value;
I is the original Y-intercept;
t is the $t_{0.05}$ value for n-2 d.f.; and
Y is the number of sampling years (2).

Then a 95% MSRL was plotted using the new intercept value (V) and the original regression slope. Finally, following the procedure of Karr et al. (1986), the area below the calculated MSRL was equally trisected to establish the three possible scores for the metric.

If a nonsignificant correlation ($p > 0.1$) between a variable and log drainage area was obtained, the metric was scored as a three-part constant (Ohio EPA 1987). The three standard values for such metrics were determined by deleting the upper and lower 5% of the data points, establishing the upper and lower boundaries of the remaining points with horizontal lines, and trisecting the area between them into above average, average, and below average categories.

After the scoring criteria was established, the IBI value of each sampling site in Boone River, Lizard and South Lizard creeks were calculated. No data were available on hybrid species, thus this metric was assigned a score of five (Karr et al. 1986).

Shannon-Wiener Diversity index

The Shannon-Wiener Diversity Index (H') is calculated as

$$H' = -\Sigma Pi \,(\ln Pi)$$

where Pi is the proportion of individual that species i contributes to the entire community. H' was calculated for each sampling site of Lizard and South Lizard creeks.

Statistical analyses

Discharge and water quality characteristics at Boone River and Lizard Creek water sampling sites were compared for spatial and temporal consistency by analysis of variance with repeated measures (ANOVARM). The ANOVARM result performed on the untransformed data set is shown because similar conclusions were derived using both the original and transformed ($\log_{10}(x + 1)$) environmental data sets. Statistical differences between samples were considered significant at the 5% level ($p = 0.05$). Environmental grouping of the 15 water sampling sites was achieved by application of principal component analysis with the correlation matrix of the 1988 and 1989 environmental data sets, respectively.

It was recognized that drought conditions might have had confounding influences upon the IBI data. A test was made, therefore, between Lizard Creek IBI values from the two study years. Kendall's tau, a nonparametric rank-correlation coefficient, was used to compare mean site differences in discharge and IBI between 1988 and 1989 (Conover 1971).

For comparisons between IBI and H', only data from the five Lizard Creek sites were used; data from the three South Lizard Creek sites were deleted because they were incomplete. Repeated measure ANOVA was used to compare results between IBI and H' among sites and over time.

To determine the consistency of the IBI, individual IBI metrics, and H' in ranking Lizard Creek fish samples through time, Kendall's nonparametric coefficient of concordance (W) was applied. Kendall's W ranges from 0 when ranks are random over time to 1 when ranks agree completely. The rank correlations between the IBI and both individual IBI metrics and H' were calculated by using Spearman's rho.

A several-step analysis was applied to determine the relative importance of individual metrics to the IBI. First, principal component analysis was applied to values calculated for 11 of the individual metrics. A series of weighting coefficients for the first two principal components was generated by this procedure. Their absolute values were interpreted as representing the relative importance of each metric to the two principal components.

RESULTS

Environmental measurements

ANOVARM results showed highly significant differences in discharge among sites within each stream (Boone River: $F_{6,18} = 15$, $p < 0.01$, Lizard Creek: $F_{7,23} = 1.3$, $p < 0.01$). Perhaps as a result of drought, no temporal differences in discharge were found in either stream. Highly significant differences existed in conductivity ($F_{7,26} = 3.8$, $p < 0.01$), nitrate-nitrogen ($F_{7,26} = 0.9$, $p < 0.05$), and total phosphate ($F_{7,26} = 18.4$, $p < 0.01$) among sites in Lizard Creek, but not for turbidity. In contrast, ANOVARM results showed that spatial concordance occurred for water quality parameters in the Boone River except for conductivity ($F_{6,18} = 3.7$, $p < 0.01$). Temporal variations were not detected in either stream.

An environmental comparison of all 15 water sampling sites was conducted separately for 1988 and 1989 by principal component analysis (PCA) based on annual means of discharge, turbidity, conductivity, nitrate-nitrogen, and total phosphate (Table 3). Similar PCA results were obtained for both years. Among the five generated principal components, the first two accounted for 79 and 82% of the correlation matrix total variance for 1988 and 1989, respectively. Conductivity, nitrate-N,

Table 3. Loadings of the first two environmental components for both 1988 and 1989

	Component 1		Component 2	
Variable	1988	1989	1988	1989
Discharge	0.053	0.088	−0.604	−0.552
Turbidity	0.332	0.286	−0.610	−0.589
Conductivity	0.731	0.704	0.089	0.056
Nitrate-nitrogen	0.561	0.584	0.121	0.174
Total phosphate	0.614	0.633	0.067	0.104
Accumulated explained variance (%)	56.7	56.9	11.9	13.7

and total phosphate were included in component 1 which contributed an average of 57% of the variance. Component 2 explained 12 and 14% of the variance for 1988 and 1989, respectively, with major loading associated with discharge and turbidity. Plots of the first and second components for both data sets are shown in Figure 2. Positive component 1 values generally indicate higher conductivity, higher nitrate-nitrogen, and higher total phosphate. Positive component 2 values tend to reflect lower discharge and higher turbidity.

These results suggest three environmental groupings of the sampling sites (Fig. 2; Table 4). Sites LB and LC, located consecutively below the city of Pocahontas, consistently had high mean values of conductivity, nitrate-N, and total phosphate, and comprise Group I, a zone of eutrophication. The downstream areas of Boone River (BE, BF, and BG) and Lizard Creek (LD and LE) had higher discharges and the overall best water quality (Group III). The remaining sites (Group II) were characterized by variable discharge and intermediate water quality when compared with the two environmentally contrasting groups.

Overall, sampling sites in Boone River generally had equal or better water quality than those in Lizard Creek. Both stream systems had better water quality and discharge regimes downstream.

IBI performance

Thirty-one fish species of five families were collected from Boone River. Thirty-two fish species in six families were collected from Lizard Creek watershed over two years. Cyprinidae represented the highest proportion of species in both streams. As detailed in Appendix I, a specific set of IBI scoring standards (linear or discrete) was established based on fish samples from Boone River. The IBI score and H' value were calculated for the eight sampling sites in Lizard and South Lizard Creek watersheds (Table 4).

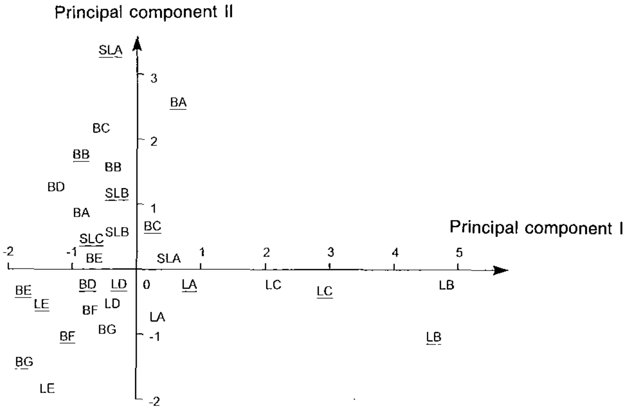

Fig. 2. Environmental relationships of sampling sites in Boone River and Lizard Creek during 1988 and 1989 warm seasons plotted relative to the first two principal components. Sites with or without underlining indicate data from 1988 or 1989, respectively.

Table 4. Scores of the Index of Biotic Integrity (IBI) and mean values of Shannon-Weiner index (H') for five sites in Lizard Creek (LA-LE) and three sites in South Lizard Creek (SLA-SLC) in June, August, and October 1988 and 1989. Environmental grouping of sites were determined from Figure 2

		IBI Score							
		1988			1989				
Site	Environmental group[a]	Jun	Aug	Oct	Jun	Aug	Oct	Mean (CV[b])	H' (CV)
LA	II	40	40	40	32	38	38	38 (8.1)	1.5 (13.3)
LB	I	40	28	30	36	36	30	33 (14.2)	1.5 (44.4)
LC	I	32	40	32	46	40	36	38 (14.2)	1.7 (29.4)
LD	III	48	50	44	34	40	36	43 (13.5)	1.6 (31.3)
LE	III	46	46	44	42	52	42	45 (8.2)	2.0 (20.0)
SLA	II	32	dry	26	36	28	28	30 (13.3)	0.9 (55.6)
SLB	II	40	38	32	34	38	40	37 (8.9)	1.4 (50.0)
SLC	II	–[c]	–	–	30	32	42	35 (18.2)	1.3 (38.5)

[a]I: moderate discharge and turbidity; high conductivity, nitrate-nitrogen, and total phosphate;
II: moderate-high turbidity; low-moderate discharge, conductivity, nitrate-nitrogen, and total phosphate;
III: moderate-high discharge; low turbidity, conductivity, nitrate-nitrogen, and total phosphate.
[b]CV = coefficient of variation (%).
[c]no data.

As previously noted, drought and low water conditions were a potential confounding factor affecting the distribution and abundance of fishes during the study period. Kendall's tau, therefore, was used to evaluate the correlation of yearly differences in mean discharge (1988 vs. 1989) with yearly differences in mean IBIs for the five Lizard Creek sites. The resulting tau value was nonsignificant ($p > 0.05$), meaning that a drought effect could not be detected.

When applied to Lizard Creek, the IBI and H' described similar patterns of spatial variation (Table 4). In South Lizard Creek, site SLB typically rated higher than site SLA (site SLC data was incomplete). In the main Lizard Creek, site LB consistently scored the lowest and site LE was usually the highest for both indices. ANOVARM confirmed that these spatial differences were significant among sites in Lizard Creek for both indices as well (IBI: $F_{4,8} = 7.3$, $p < 0.01$; H': $F_{4,8} = 8.8$, $p < 0.01$). However, temporal differences, between years or months, were not detectable in either case ($p > 0.05$).

Correlations between rankings (Spearman's rho) for the IBI and corresponding H' values were significant (Spearman's rho = 0.55, $p < 0.05$). Significant concordance (Kendall's W) of IBI (W = 0.58, $p < 0.05$) and H' (W = 0.51, $p < 0.05$) indicated that the relative ranking of sites was quite consistent over the two years of data collection. However, a trend of greater spatial sensitivity was observed in the IBI than in the H' (Table 4).

Contributions of IBI metrics

Concordance associations of the IBI metrics among Lizard Creek sites over time were significant for all involving the five numerical species metrics (Table 5). Additionally, all were equal to or greater than the H's W value of 0.51. But no significant temporal stability was declared for the proportional metrics. Similarly, Spearman's rho correlations between the individual metrics and the IBI were significant for all numerical metrics, but there was only one significant correlation involving the proportional metrics.

The first two principal components of the correlation matrices of IBI metrics accounted for 53% of the variance among metrics (Table 5). Loadings were evenly distributed among metrics in PC I, but heavier for the five numerical species metrics. In PC II, more contributions were from the total individual metric and five proportional metrics. The metric loading pattern is evidence of the relative contribution of the various metrics in accounting for variance in the IBI.

DISCUSSION

Determination of metric scoring criteria

Karr's (1981) original proposal for the IBI was based only on discrete sets of metric scoring standards, determined subjectively through experience with local fish fauna in small midwestern streams. As interest in the IBI spread, it became apparent that at least some of the metrics required linear scoring criteria, based on stream size. Fausch et al. (1984) recommended such linear standards for six of the metrics in a study involving drainages in six midwestern states. In Ohio, up to 10 of the metrics have been defined as linear variables

Table 5. Kendall's coefficient of concordance (W) (N = 6) among sites in Lizard Creek watershed (two years, three samples each year) for the 11 metrics of Index of Biotic Integrity (IBI) and the correlation (Spearman's rho (SR), N = 30) between each metric and IBI. Weighting coefficient of the 11 IBI metrics on the first two principal components for five sites in Lizard Creek are also shown. The percent variance accounted for by each component is given in the lower part of the table. Empty cells indicate that the hybrids metric was constant within a data set. The abbreviations of IBI metrics are shown in Table 1

			Principal component	
Metric	W	SR	1	2
Number				
SP	0.57*	0.65*	0.412	0.277
SNFR	0.85*	0.59*	0.440	0.023
DRTR	0.60*	0.56*	0.412	0.277
SUKR	0.65*	0.64*	0.420	0.106
INSP	0.62*	0.61*	0.358	0.170
Proportion				
GRN	0.20	0.08	–0.273	0.484
OMNI	0.12	0.17	–0.223	0.142
INSE	0.18	0.08	–0.261	0.262
PIS	0.33	0.31	0.208	–0.174
SICK	0.15	0.54*	0.129	0.486
HYBI				
Number/min.				
CPUE	0.29	0.36	–0.099	0.530
Variance (%)			36	17

* $p < 0.05$

in routine monitoring use of the IBI (Ohio EPA 1987). Variation in this important element of the construction of the IBI can depend on a number of factors such as range of drainage areas, local topographic features, and general ichthyofaunal diversity and abundance.

Until now, the MSRL defined for linear variables has typically been established by the subjective "eye-fit" procedure (Fausch et al. 1984, Karr et al. 1986). This study proposes another approach for establishing the MSRL that has some basis in statistical theory and, therefore, it should reduce some of the investigator-related variability inherent in the eye-fit approach. At the same time, the possible biases of this procedure need to be recognized as well. Like the original approach, it arbitrarily establishes the MSRL as the 95% upper limit of a plot of species richness against drainage size. It does this by establishing a 95% confidence interval line above the regression line, assuming that the regression line is the true line for the entire "population". In actuality, the regression only describes the relationship for a population sample, and the 95% confidence interval is not represented by a straight line, but rather by a parabola which is closest to the regression line near the independent variable mean (Steel and Torrie 1980). Therefore, from the standpoint of statistical validity, the procedure used here systematically overestimates the position of the MSRL around the center of the regression plot and underestimates it at both ends. The magnitude of bias is sample-dependent. Despite this violation of normal statistical procedure, we believe the approach used here has merit for general application of the IBI. With modern computational facilities, it is easy to calculate the MSRL and it can be readily conceptualized. Inspection of some actual data plots suggests that it would often describe a line quite like one established by the "eye-fit" technique; however, it would produce only a single MSRL for a given data set while eye-fit lines will inevitably vary.

Performance of the IBI calculated by the new scoring procedure

The value of a biological monitoring index depends on its usefulness in making assessments. A useful index should be consistent temporally at a site if no change in environmental quality occurs during the sample period. Alternatively, index values should increase or decrease as environmental quality at a site rises or declines, respectively.

In several small streams of Illinois and Indiana, Angermeier and Schlosser (1987) and Karr et al. (1987) reported that spatial variations in the (old) IBI, based on the "eye-fit" method, parallel known variations in habitat quality or water quality. Additionally, when watershed conditions remain relatively stable, the (old) IBI ranks sites similarly each time they are sampled.

The modified scoring procedure proposed in this study may actually advance, or at least sustain, the effectiveness of the IBI in environmental assessment. There was good agreement between the extremes of Lizard Creek sites for the (new) IBI, calculated by the modified scoring procedure, and water quality measurements (best: LD and LE, worst: LB and LC). During the extreme low-water time, the IBI for site LB declined notably for early August 1988 (28) compared to the June score (40). At the same time, the IBI scores of downstream sites generally increased. This may reflect a heightened fish community response, such as downstream migration (Paloumpis 1956, Larimore et al. 1958, Marsh and Luey 1982) to the overall poor habitat quality at this site. The relative ranking among sites in the main Lizard Creek was consistent over the two years of data collection.

During the last decade, use of indices based on fish community parameters has become increasingly popular for evaluating stream environmental quality (Munkittrick and Dixon 1990). Among a relatively limited number of analytical approaches that have been tried, the IBI and H' indices have been used most commonly. Two previous studies have demonstrated that the (old) IBI was more consistent in its assessment of individual sites than was H'; i.e., rankings of sites based on H' scores varied more among sample periods than rankings based on the (old) IBI scores (Angermeier and Schlosser 1987, Karr et al. 1987). In this study, the (new) IBI and H' were similar in their sensitivity and consistency in rating the sampled fish communities over time, although H' tended to be somewhat more variable.

Overall, the (new) IBI in this study also displays the spatial sensitivity and temporal consistency in site rankings as does the (old) IBI. The changes of (new) IBI scores reflect the variations of environment conditions during the drought period. Additionally, the (new) IBI shows greater consistency than another commonly used analytical tool, the H'. Thus, on the basis of these considerations, the modified scoring procedure proposed in this study seems to have distinct advantages over the "eye-fit" method that employs fish community data for calculating the IBI scores.

Contributions of individual IBI metrics

Although there is yet no typical pattern concerning the relative importance of individual metrics to the IBI (Angermeier and Karr 1986, Miller et al. 1988, Steedman 1988). Principal component analysis revealed a major distinction between the five numerical species metrics as a group and the proportional metrics. The numerical metrics were of more informative value to the first principal component, which included most of the accounted-for variation. Moreover, these metrics were generally quite stable over time and positively correlated with the IBI.

If it is generally true for Iowa that the numerical species metrics are most informative, then an abbreviated IBI based solely on those metrics may be useful. The utility is that less handling of specimens, fish identification expertise, and computational analysis are required. However, Angermeier and Karr (1986) found that the importance of individual metrics varied considerably in a comparison of the performance of the IBI in Illinois, Ohio, and West Virginia. For example, the metrics of green sunfish and piscivores contributed significantly to the final IBI scores in Ohio. In West Virginia, the metrics of omnivores and insectivorous cyprinids were important in determining the final IBI values. Similar inconsistency was found in the present study when compared to the results of the other three states' data sets. Thus, the amount of information conveyed by a particular metric varies regionally. We strongly caution for applying the results of an investigation regarding the relative contribution of individual metrics to the final IBI scores outside of a given study area.

Influence of drought conditions and other confounding factors

Drought conditions reduced the sensitivity of the IBI as well by reducing environmental variation, especially for water quality and quantity. For example, sites SLA, SLB, and SLC were categorized as intermediate in water quality according to principal component analysis, but their IBI scores were consistently low. In these cases, it seems likely that physical habitat features were an important element of overall environmental quality. Although physical habitat was not measured or evaluated in detail, Table 3 shows that the South Lizard Creek sites were essentially open ditches with monotonous sand bottoms, a condition which is not conducive to faunal diversity and abundance. This example is useful, however, for pointing out a caution in the use of the IBI. Although its applications often tend to be oriented toward evaluating chemical water quality, the IBI is sensitive to overall environmental conditions including water quantity and physical habitat features. Proper evaluation of IBI results, therefore, requires some knowledge of each of these classes of environmental characteristics (Karr 1981, Karr et al. 1986).

An important element of the IBI technique is identification of the most appropriate "least disturbed" area which serves as a baseline for establishing metric standards. Although there is little question that the Boone River system as a whole is an appropriate choice for the least disturbed drainage, the fact is that many stream areas within its watershed are of poor environmental quality because of agricultural perturbations. This is demonstrated by calculated IBI values for the Boone River sites which were most commonly in the poor to fair condition classes and none in the excellent class (Liang 1990). In consideration of the drought conditions, it is possible that these calculated metric standards are based largely on unstable fish communities. Increasing the number of sampling sites in the least disturbed drainage or relying more on use of historical data bases may compensate for such a situation. However, it must be recognized that in areas of intensive agriculture such as Iowa or other areas under heavy human disturbance, near-pristine stream environments and fish communities may be extremely rare or non-existent. In that case, comparisons with IBI results obtained for nearby areas may be useful, but application of metric standards derived for such areas are probably inappropriate.

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Appendix 1. Scoring criteria of the Index of Biotic Integrity's metrics in north-central Iowa streams. The abbreviations of IBI metric are shown in Table 1

	Score				
Metric	5		3		1
Number					
SP		$Y = 5.20 + 2.24X^a$		$Y = 2.60 + 1.12X$	
SNFR		$Y = 0.15 + 0.50X$		$Y = 0.07 + 0.25X$	
DRTR	> 2		2		0-1
SUKR		$Y = 0.03 + 0.84X$		$Y = 0.02 + 0.42X$	
INSP		$Y = 0.41 + 0.45X$		$Y = 0.21 + 0.22X$	
Proportion (%)					
GRN		$Y = 0.01 + 0.63X$		$Y = 0.03 + 1.25X$	
OMNI	< 12		12-22		> 22
INSE		$Y = 69.75 - 8.91X$		$Y = 34.88 - 4.45X$	
PIS		$Y = -0.85 + 2.10X$		$Y = -0.43 + 1.05X$	
SICK	< 3		3-6		> 6
Number/Minute					
CPUE		$Y = 19.48 - 3.37X$		$Y = 9.74 - 1.84X$	

[a] $X = \log_{10}$ (drainage area in mi^2)

一種建立生物整合指標評分標準之新方法

梁世雄[1] Bruce W. Menzel[2]

生物整合指標 (Index of Biotic Integrity, IBI) 是綜合魚種歧異度、食性群與健康等 12 個魚類群聚特徵，以評估溪流環境之生物評估系統。本研究之目的在改進計算 IBI 給分標準之程序，並以改良之 IBI 使用於愛荷華州中北部流域，以測試其評估溪流環境品質之功能。我們在 Boone River 選擇 21 個採集點，於 1988 和 1989 年七月，以電魚法採集魚類樣本，作爲建立 IBI 計分標準的基本資料。另於受到城市廢水和農業影響的 Lizard Creek 流域選擇 8 個採樣點，於 1988 和 1989 年六月、八月和十月以電魚法進行魚類採集，以探討 IBI 之環境評估功能。

IBI 的計分標準，由原來以目測法決定 Maximum-Species-Richness Line (MSRL) 再三分其下限區域之方法，改爲如果群聚變數與流域面積間有統計上顯著的直線關係存在時，則 MSRL 由 Y 軸截距的 95% 上限和原有斜率決定，再三分其下區域，以建立高於平均值 (5 分) 、類似於平均值 (3 分) ，與低於平均值 (1 分) 的區域以爲 IBI 的計分標準，若無直線迴歸關係存在，則先移除上下 5% 的採集點建立界限，再將剩餘區域均衡三分。

根據我們的改良計分法，一組資料間將只會產生一條 MSRL，這可解決目測法因研究人員目視差異而有同組資料產生多條 MSRL 之缺點，這個新的計分法也可能促進 IBI 評估溪流環境之能力，因爲與環境因子之測量結果比對，利用改良方法計算的 IBI (1)可分辨 Lizard Creek 中採集點間的環境品質差異，(2)對枯水期造成的溪流環境變化可迅速反應，(3)同時季節變動時，分辨空間環境差異的敏感度仍維持穩定，且優於 Shannon-Weiner Diversity index。在本研究中，IBI 分數之變化主要受到和魚種歧異度有關的群聚變數影響。

關鍵詞：生物指標, 生物整合指標(IBI), 環境評估, 溪流魚類, 群聚生態。

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