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Development and Validation of an Index of Biotic Integrity for Coldwater Streams in Wisconsin

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Abstract.—The index of biotic integrity (IBI), developed from information on the structure, composition, and functional organization of fish assemblages, is used to assess the health of aquatic ecosystems. We analyzed two large statewide data sets on stream fish assemblages to develop and test a version of the IBI for application to Wisconsin coldwater streams (maximum daily mean water temperature usually <22°C). This new IBI is needed because fish assemblages in Wisconsin coldwater streams differ significantly from those in warmwater streams (maximum daily mean temperature >24°C), for which an IBI already exists. High-quality coldwater streams have few species, with salmonids and cottids dominating, and lack many of the taxonomic groups that are important in high-quality warmwater streams. In contrast, high-quality warmwater streams have numerous species, and cyprinids, catostomids, centrarchids, and percids typically dominate. Environmental degradation often causes an increase in species richness in coldwater fish assemblages, the opposite of what occurs in warmwater assemblages, as a small number of coldwater species are replaced by a larger number of more tolerant eurythermal and warmwater species. The new coldwater IBI has five metrics: (1) number of intolerant species, (2) percent of all individuals that are tolerant species, (3) percent of all individuals that are top carnivore species, (4) percent of all individuals that are native or exotic stenothermal coldwater or coolwater species, and (5) percent of salmonid individuals that are brook trout *Salvelinus fontinalis*. No regional or stream-size adjustments in metric scoring criteria are needed. Relative coldwater IBI scores and ratings of stream sites throughout Wisconsin closely match independent rankings of environmental quality on the basis of physical habitat and water quality of the sites. Variation in IBI scores within and among years is generally low. The new coldwater IBI is not appropriate for coolwater streams (typical maximum summer daily mean temperature 22–24°C).

Over the past 25 years, numerous studies have demonstrated that attributes of fish communities accurately reflect the overall biotic integrity and environmental health of flowing-water ecosystems (reviewed in Fausch et al. 1990). Biotic integrity in this context has been most commonly defined as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of the natural habitat of the region" (Karr and Dudley 1981). Environmental assessment or "biomonitoring" of streams and rivers based on ambient fish assemblage characteristics has become a widely used and effective tool for managing aquatic resources in North America. The most common and arguably the best approach to this type of biomonitoring involves the family of related indices known collectively as the index of biotic integrity or IBI (Fausch et al. 1990; Simon and Lyons 1995). The IBI is a multimetric index that rates the existing structure, composition, and functional organization of the fish assemblage with regional and habitat-specific expectations derived from comparable high-quality ecosystems. The IBI was originally developed during the late 1970s and early 1980s for small streams in Illinois and Indiana by Karr and colleagues at the University of Illinois (Karr 1981; Karr et al. 1986). It has since been modified successfully for use in many different types of streams and rivers throughout North America, and more recently in Europe and Asia (Fausch et al. 1984, 1990; Miller et al. 1988; Lyons et al. 1995; Simon and Lyons 1995).

However, none of the previously developed IBIs appeared to be appropriate for use in the low-gradient, coldwater (maximum daily mean temperatures usually <22°C) streams of Wisconsin. Most applications of the IBI have involved "warmwater" (maximum daily mean temperatures >24°C) streams and rivers, which tend to have relatively high fish species richness. Lyons (1992a) provided evidence that these earlier IBIs are ineffective in characterizing the biotic integrity of Wisconsin coldwater streams, which have low fish species richness. In addition, the few coldwater versions of the IBI developed for other regions of North America (summarized in Simon and Lyons 1995) also appeared to be inappropriate. Most were developed for high-gradient streams in the mountainous regions of eastern and western North America. Only one, a combined coldwater–warmwater version (Steedman 1988) was designed for low-gradient streams similar to those in Wisconsin. However, Steedman's version did not effectively characterize the relative biotic integrity of Wisconsin coldwater streams, because many of the metrics that made up his version were insensitive to differences in environmental quality among Wisconsin streams (Lyons, unpublished data).

Coldwater streams are common in Wisconsin and parts of adjacent states, as well as in eastern and western North America. They support important recreational fisheries and are a highly valued resource. Thus, an effective IBI would be an important management tool for Wisconsin coldwater streams. Such an index would be valuable for identifying streams with degraded ecosystems that warrant restoration, designating particularly high-quality streams for additional protection, classifying streams on the basis of biotic integrity, monitoring streams for changes in ecosystem quality over time, and assessing responses of streams to management and other human activities. A coldwater IBI for Wisconsin would likely also be useful in adjacent states with similar coldwater ecosystems, and perhaps, with some modification, in other parts of North America as well.

We describe a new version of the index of biotic integrity designed for use in Wisconsin coldwater streams. We first quantify the differences between warmwater and coldwater streams in Wisconsin to demonstrate that a new coldwater version of the IBI is needed. We then use statewide stream fish assemblage data collected during the 1970s to identify appropriate metrics and scoring criteria and to develop this new index. Finally, we use a different set of statewide fish community data collected during the 1990s to test and validate the index.

Methods

Sources of data.—Most of the data used in this paper were collected as part of statewide surveys of stream fish assemblages and habitat characteristics conducted during the late 1970s and early 1990s. We also searched the literature and the Wisconsin Department of Natural Resources (WDNR) computer data files for stream sites with information about changes in fish assemblages associated with improvements or declines in environmental quality.

The 1970s data were collected from 1974 to 1979 as part of the WDNR fish distribution survey, in which numerous streams in the southern and western thirds of Wisconsin were sampled (Fago 1988, 1992). During the survey, efforts were made to capture all species of fish at each sampling site, and all captured fish were identified and counted. However, if more than 99 individuals of a species were captured at a site, the count was stopped for that species.

From the 1970s data set, we selected for analyses 77 sites on 65 streams. Each stream site was 90–180 m long and had been sampled once by a fisheries biologist who waded upstream (single pass, no block nets) with a backpack or "stream" (tow barge) electroshocker (Reynolds 1983; Fago 1988; Lyons and Kanehl 1993) during June, July, or August. Of the 77 sites, 23 (21 streams) were on "least impacted" (based on criteria in Hughes et al. 1986, 1990) coldwater streams that represented some of the best water and habitat quality throughout the sampled areas of the state and that were formally designated as "exceptional" or "outstanding" aquatic resources (Wisconsin Administrative Code, Natural Resources Chapter 102). The 23 least impacted sites were on first- to fourth-order streams and had mean widths from 0.7 to 16 m. This size range encompasses nearly all of Wisconsin's coldwater streams (Threinen and Poff 1963; WDNR 1995). Fish assemblage characteristics for these 23 sites were used to define a coldwater stream with high biotic integrity and to select and calibrate IBI metrics.

The remaining 54 sites were from the Driftless Area ecoregion (33 sites; 30 streams) in southwestern Wisconsin and the Northern Lakes and Forests ecoregion (21 sites; 14 streams) in northwestern Wisconsin (Omernik and Gallant 1988), and were all on either second- or third-order streams with mean widths from 0.7 to 13 m. Of these 54 sites, 19 were on marginal coldwater streams (on the basis of summer water temperatures and habitat and water quality) and 35 were on high-quality, least-impacted warmwater streams. Of the 19 marginal coldwater sites, 14 were from the Driftless Area ecoregion and were severely degraded by siltation, loss of bank-side vegetation, barnyard runoff, and other forms of non-point source pollution from intensive agriculture in their watersheds. Each of these sites was potentially a high-quality coldwater stream if agricultural non-point source pollution were to be eliminated. Conversely, the remaining five marginal coldwater sites from the Northern Lakes and Forests ecoregion suffered from little environmental degradation and were perhaps best classified as natural, high-quality, "coolwater" sites. Lyons (1992a) defined coolwater streams as having maximum summer daily mean temperatures of 22–24°C and fish assemblages dominated by designated coolwater species. Fish community data from the marginal coldwater and high-quality warmwater sites were contrasted with data from the high-quality coldwater sites to help identify those IBI metrics that best distinguished between coldwater streams with high and low biotic integrity.

The 1990s data came from an ongoing statewide WDNR study of the relations between watershed land-use practices and stream ecosystems. For this paper, data from 61 sites on 34 different coldwater streams were used. These sites differed from those sampled in the 1970s and encompassed a wide range of environmental conditions from high-quality "least impacted" (23 sites, 17 streams; all formally designated as exceptional or outstanding aquatic resources) to moderately degraded (10 sites, 7 streams) to highly degraded (22 sites, 8 streams; note that 3 streams had multiple sites with different quality levels). Six sites on five streams were high-quality coolwater. Many of the sites were sampled more than once between 1990 and 1994 for a total of 117 data points. Sites were located throughout Wisconsin except in the southeast where coldwater streams are rare (Threinen and Poff 1963; WDNR 1980, 1995), and were on second- to fifth-order streams and ranged in mean width from 1.5 to 16 m. Sites were 100–360 m long and were sampled with either two backpack electroshockers or a single stream electroshocker. Sampling consisted of a single upstream pass without block nets; previous studies had indicated that this approach gave a representative sample of the fish community (Simonson and Lyons 1995). We attempted to collect all fish observed during sampling, and all captured fish were identified and counted. Sites were sampled from April through November, but most were sampled in June, July, or August. Data on instream habitat quality, water quality, and watershed land use were also collected from each site within 2 d of fish sampling. These data were used to classify each site as "good," "fair," or "poor" in environmental quality, or as "good quality coolwater" on the basis of physical and chemical characteristics (Lyons 1992a; Simonson et al. 1994; Lyons et al. 1995).

We also examined the response of coldwater fish assemblages to changes in stream environmental quality. Although there are extensive data on responses of Wisconsin salmonid populations to environmental degradation or restoration (e.g., Hunt 1988), we found suitable information on fish assemblage responses for only one coldwater stream, Timber Coulee Creek (Vernon County) in the Driftless Area ecoregion. Fish species data for Timber Coulee Creek had been collected in 1966, 1976, 1980, and 1994 with the same electrofishing procedures as for the 1970s and 1990s data sets (unpublished WDNR file summaries). Timber Coulee Creek ranges from second to fourth order and 3–9 m in mean width. Once a high-quality coldwater stream known for good brown trout fishing (see Appendix for scientific names), by the mid-1960s Timber Coulee Creek had been degraded by agriculture in its watershed, and conditions were such that only a marginal fishery of stocked trout could be supported (Vetrano 1988). Beginning in the 1960s and continuing to the present, this stream and its tributaries have been sub-

TABLE 1.—Metrics considered for inclusion in the coldwater biotic integrity index.

Number of species metrics	
Native species	Native stenothermal coldwater
Darters (Percidae)	Native stenothermal coolwater
Suckers (Catostomidae)	Native coolwater and coldwater
Sunfish (Centrarchidae)	Native and exotic coolwater and coldwater
Intolerants	
Percent of total individuals metrics	
Intolerants	Native stenothermal coldwater
Tolerants	Native stenothermal coolwater
Invertivorous feeders	Native coolwater and coldwater
Omnivorous feeders	Native and exotic coldwater
Top carnivorous feeders	Native and exotic coolwater and coldwater
Simple lithophilic spawners	
Other metrics	
Percent of salmonid individuals that are brook trout	
Total fish catch per unit effort	

ject to intensive efforts by WDNR and local groups to improve instream habitat quality and riparian land-use practices. These efforts have resulted in gradual but ultimately major improvements in the stream ecosystem, and now maximum summer water temperatures are much lower and habitat and water quality are considered good to excellent (Vetrano 1988, and personal communication).

Data analyses and IBI development and testing.—All statistical analyses were done with SAS software (SAS Institute 1990), and differences were considered significant if a test for equality yielded a $P < 0.05$. The 1970s data were used to develop and calibrate the coldwater IBI. We considered 22 potential metrics (Table 1), 11 of which were from the Wisconsin warmwater IBI (Lyons 1992a) and the rest from existing coolwater or coldwater versions of the IBI or suggested by initial analyses.

We classified species into taxonomic, tolerance, feeding, spawning, thermal, and origin categories based on Karr et al. (1986) and Lyons (1992a). Intolerant species are those that are sensitive to many types of environmental stress and tend to be absent in the presence of environmental degradation. Tolerant species are just the opposite, able to tolerate a wide range of environmental conditions and often common in highly degraded environments. Invertivores eat primarily benthic and drifting macroinvertebrates; top carnivores often eat (as adults) other vertebrates and crayfish; and omnivores eat a mixture of plant and animal material (at least 25% by volume of each type). Simple lithophilous spawners lay their eggs on clean gravel or rubble without preparing a nest and do not provide parental care for eggs or young. Exotic species are those that have entered Wisconsin waters within the last 150 years because of human activities. In some streams, exotic salmonids and cyprinids have established self-sustaining, naturally reproducing populations, whereas in other streams, exotic salmonid populations are maintained by stocking. Native salmonids have been present in Wisconsin since before European settlement; the brook trout is the only native salmonid currently found in Wisconsin streams (Becker 1983). We also grouped species by summer temperature preferences as eurythermal (wide preference, although many are not found in the coldest waters) or as stenothermal (narrow preference) coldwater (maximum <22°C), coolwater (22–24°C), or warmwater (>24°C) on the basis of available laboratory and field data (Coutant 1977; Becker 1983; Eaton et al. 1995); some of these thermal designations differed from those in Lyons (1992a). For the 1970s data, we could not distinguish stocked from naturally reproduced (in the stream) salmonids, so all salmonids captured were included in the analyses.

Values for each of the 22 metrics were compared among the three site groupings—high-quality coldwater, marginal coldwater, and high-quality warmwater streams—with an analysis of variance (ANOVA) and a Tukey studentized range multiple-comparison test. To account for potential regional differences in metric values, a two-way ANOVA was used, with site grouping and ecoregion as main effects. When appropriate, metric values were transformed to approximate normality before analyses. Only sites from the Driftless Area and the Northern Lakes and Forests ecoregions were included because of insufficient sample sizes from other ecoregions. The Driftless Area and the Northern Lakes and Forests ecoregions have different warmwater stream fish communities (Lyons 1989, 1992a; Poff and Allan 1995). To minimize confounding effects of differing stream sizes on results, only second- and third-order stream sites were analyzed.

The four-step process used to identify the final IBI metrics was based on criteria in Karr et al. (1986), Lyons (1992a), and Simon and Lyons (1995). Our goal was to have metrics that clearly and consistently distinguished between high-quality and low-quality coldwater streams, and that were little influenced by stream location in the state or stream size. The first three steps involved all of the 1970s sites. First, we retained only metrics for which high-quality coldwater stream sites differed significantly from the other two site groupings in both the Driftless Area and the Northern Lakes and Forests ecoregions. Second, for each of these retained metrics, if differences between site groupings were small or not biologically meaningful (e.g., a difference in mean species number between groups of less than one), or if differences resulted from one or two "outlier" sites, then the metric was eliminated. Third, for each of the remaining metrics, if the differences among site groupings were not consistent between the two ecoregions (i.e., significant ecoregion main effect or site grouping × ecoregion interaction effect in the ANOVA), then that metric was dropped. The fourth and final step was based only on data from the 23 high-quality coldwater streams sites. Each remaining metric was correlated (Pearson product-moment correlation) with mean stream width and stream order, and if there was a significant relation with either variable, the metric was dropped.

Scoring criteria for the final coldwater IBI metrics were derived from the 23 high-quality coldwater sites by using standard procedures described in Karr et al. (1986), Ohio Environmental Protection Agency (1988), and Simon and Lyons (1995). For each metric, a frequency distribution of values was generated, and the values of the 95th percentile (for metrics in which high values indicated high quality) or the 5th percentile (for metrics in which low values indicated high quality) were identified. The 95% of the frequency distribution below or above this value was then divided into thirds (i.e., the values for the 63rd and 32nd percentiles were determined). Values that were greater than or equal to the 63rd percentile value were given a score of 20 (good) for metrics in which high values indicated high quality and a score of 0 (poor) for metrics in which high values indicated low quality. Values between the 63rd and 32nd percentile values were given a score of 10 (fair), and values less than or equal to the value for the 32nd percentile were given scores of either 0 or 20, depending on the relation between metric values and environmental quality.

All 77 of the 1970s sites were then scored for each of the final metrics. Scores for each metric were summed to give coldwater IBI scores, which were then compared among each of the three site groupings with a one-way ANOVA and Tukey multiple comparison. On the basis of fish assemblage characteristics and IBI scores for each group, we developed preliminary qualitative ratings and narrative interpretations of biotic integrity for five different scoring ranges (excellent to very poor).

The 1990s data were used to test the coldwater IBI and to explore its variability over time at individual sites. Index of biotic integrity scores were calculated for each site and sampling period. We classified all captured salmonids as either stocked or naturally reproduced on the basis of their appearance and the recent stocking history of the study streams, and included only naturally reproduced salmonids in calculations. The four 1990s site groupings—good, fair, poor, and coolwater—were compared with a one-way ANOVA and Tukey multiple comparison. For this analysis, we used only the first sampling period per site to avoid potential bias from including multiple samples from some sites but not from others. Subsequently, for those sites with multiple samples, the range and standard deviation of IBI scores for all samples were calculated so that patterns of temporal variation could be examined. Physical and chemical characteristics fluctuated from sample to sample at all sites during the study period, but only those sites showing no major trends in physical or chemical variables were included in this part of the analysis. At two sites, Dunlap Creek and Wendt Creek (Dane County) in the Driftless Area ecoregion, seven monthly samples were collected during the open-water period between May and November 1992 and two additional monthly samples were collected in May and June 1993, providing information on within-year variation in IBI scores. At 17 sites (11 streams), samples were collected during the same month in two or more consecutive years, providing information on between-year variation. The range and standard error of scores over time within each site was correlated (Pearson product-moment) with that site's mean IBI score to estimate whether the amount of temporal variation in biotic integrity was related to the relative level of biotic integrity. The range and standard error of scores over time were also compared among poor, fair, and good quality sites by correlation analysis.

Results

Differences between Coldwater and Warmwater Stream Fish Assemblages

Overall, our data and analyses indicate two major differences between coldwater and warmwater stream fish assemblages. First, high-quality coldwater streams have lower species richness than comparable high-quality warmwater streams, and many of the taxonomic groups that are important in the warmwater streams are rare or absent in the

TABLE 2.—Numbers of species in five species groups captured from three types of streams in the Driftless Area (DRT) and the Northern Lakes and Forests (NLF) ecoregions. Within each species group, mean values that are followed by the same letter are not significantly different from each other ($P > 0.05$).

Species group and ecoregion	High-quality coldwater Mean	High-quality coldwater SD	Marginal coldwater Mean	Marginal coldwater SD	High-quality warmwater Mean	High-quality warmwater SD
Total natives						
DRT	4.8 z	2.3	7.4 y	3.2	13.3 v	3.6
NLF	5.4 z	4.5	9.0 x	4.2	11.1 w	5.1
Darters						
DRT	0.3 z	0.5	1.2 y	1.0	2.0 w	0.8
NLF	0.4 z	1.0	1.4 yx	1.7	2.1 xw	1.5
Suckers						
DRT	0.5 z	0.6	1.2 y	0.9	1.4 y	0.8
NLF	0.4 z	0.5	1.2 y	0.5	1.0 y	0.6
Sunfishes						
DRT	0.0 z		0.1 zy	0.4	0.6 x	1.0
NLF	0.0 z		0.0 z		0.4 yx	0.7
Intolerants						
DRT	1.5 yx	0.8	0.8 z	1.0	1.5 yx	1.6
NLF	1.9 x	0.3	0.6 z	0.6	1.2 y	1.0

coldwater streams. Our ANOVAs indicated substantial differences in species richness between warmwater and coldwater stream sites in both the Driftless Area and the Northern Lakes and Forests ecoregions. The numbers of total native, darter, sucker, and sunfish species were significantly higher at the high-quality warmwater sites than at the high-quality coldwater sites for both ecoregions, but the number of intolerant species was not significantly different (Table 2). For each of the five species groups, numbers of species at high-quality coldwater sites did not differ between the Driftless Area and the Northern Lakes and Forests ecoregions.

Second, coldwater and warmwater streams respond differently to environmental degradation. From the literature, it is well documented that most types of degradation (e.g., habitat destruction, water pollution, flow modification) cause a decline in species richness in warmwater streams, and that as the degradation increases, the decline in number of species becomes larger (Karr et al. 1986; Fausch et al. 1990). Conversely, our analyses suggest that environmental degradation often results in an increase in species richness in Wisconsin coldwater streams, and that only the severest types of degradation consistently cause a decline in number of species. Our ANOVAs demonstrated that mean numbers of total native, darter, and sucker species were significantly higher at marginal coldwater sites than at high-quality coldwater sites (Table 2). Sunfish species were generally absent from both types of sites, and the mean number of intolerant species was significantly greater at high-quality coldwater sites. Mean numbers in each of the five species groups at marginal coldwater sites were either similar to (suckers in both ecoregions, darters in Northern Lakes and Forests ecoregion) or significantly lower (remaining species group–ecoregion combinations) than numbers at high-quality warmwater sites.

Data from Timber Coulee Creek indicate that *improvements in habitat and water quality have* been associated with a decline in overall species richness (Figure 1). In 1966, the stream contained large numbers of eight species of warmwater or eurythermal minnows, plus the eurythermal white suckers and northern hog suckers (see Appendix for tolerance, feeding, and temperature classifications), the stenothermal coolwater brook stickleback, the eurythermal fantail and johnny darters, the stenothermal coldwater brown trout (mostly stocked individuals), and sporadic occurrences of the stenothermal coolwater American brook lamprey and the stenothermal coldwater slimy sculpin. By 1994, after major improvements in environmental quality, all eight species of minnows, northern hog suckers, and johnny darters had become rare or disappeared, white sucker occurrence had been reduced, American brook lampreys and slimy sculpins were more widespread, and naturally reproduced brown trout had become abundant. Whereas 9–13 species per site had been the norm in 1966, only 4–6 species per site were observed in 1994.

Development of a Coldwater IBI for Wisconsin Streams

Our data and analyses provided the following qualitative description of a coldwater fish assemblage with high biotic integrity. This description helped guide metric selection and scoring criteria development for our coldwater IBI. High-quality coldwater streams have few species relative to comparable high-quality warmwater streams. The dominant species are salmonids, which are top carnivores. The highest integrity sites have mostly brook trout, an intolerant native species; slightly lower integrity sites have mainly brown trout, a more tolerant exotic species, or other exotic salmonid species. In addition to salmonids, high-quality coldwater sites usually have other native stenothermal coldwater or coolwater species, especially the intolerant slimy or mottled sculpins.

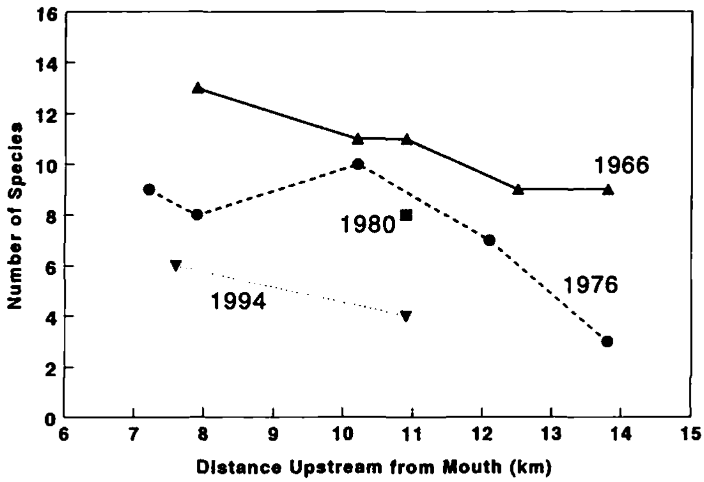

FIGURE 1.—Changes in total species richness with improved environmental quality at several sites in Timber Coulee Creek, Vernon County, Wisconsin, between 1966 and 1994; modified from Lyons (1992a).

A few native, tolerant, eurythermal species, particularly the blacknose dace, creek chub, or white sucker, may be present, but they rarely exceed 20–25% of total fish numbers. Compared to warmwater stream fish assemblages (Lyons 1992a), coldwater stream fish assemblages with high biotic integrity vary relatively little among different ecological regions of the state or among different sizes of streams.

We have identified five metrics that best characterize the attributes of fish assemblages with high biotic integrity and the responses of these assemblages to environmental degradation:

TABLE 3.—Criteria for calculating the coldwater biotic integrity index. Stocked trout should not be included in any of the metric calculations. For metric 5, if no salmonids are captured, then the score is zero.

Metric	Criteria for assigning scores of: 20 (good)	10 (fair)	0 (poor)
(1) Number of intolerant species	≥2	1	0
(2) Percent of all individuals that are tolerant species	0–5	6–22	23–100
(3) Percent of all individuals that are top carnivore species	46–100	15–45	0–14
(4) Percent of all individuals that are stenothermal coolwater and coldwater species (native and exotic)	86–100	43–85	0–42
(5) Percent of salmonid individuals that are brook trout	96–100	5–95	0–4

(1) number of intolerant species,
(2) percent of all individuals that are tolerant species,
(3) percent of all individuals that are top carnivore species,
(4) percent of all individuals that are native or exotic stenothermal coldwater or coolwater species, and
(5) percent of salmonid individuals that are brook trout.

The other 17 potential metrics that we considered either did not show a consistent difference (most metrics) between high-quality and low-quality sites or showed a difference much less dramatic (number of native stenothermal coldwater species; number of native stenothermal coldwater and coolwater species; and percent of total individuals as intolerant species, omnivorous species, native stenothermal coldwater species, and native stenothermal coldwater and coolwater species) than the differences found for the final five metrics. Scoring criteria for each of the five metrics selected are given in Table 3; the sum of the scores for the five metrics is the overall IBI score. The maximum possible overall IBI score is 100, indicating excellent biotic integrity, and the minimum is 0, indicating very poor biotic integrity. Narrative guidelines for interpreting IBI scores are provided in Table 4.

The coldwater IBI accurately reflected differences among the three site groupings for the 1970s data. High-quality coldwater sites had significantly higher IBI scores (mean, 67.5) than either marginal

TABLE 4.—Guidelines for interpreting coldwater biotic integrity index (IBI) scores, modified from Karr et al. (1986) and Lyons (1992a).

IBI score	Integrity rating	Interpretation and fish community attributes
100–90	Excellent	Comparable to the best situations with the least human disturbance: mottled or slimy sculpins are usually common; intolerant, native stenothermal coolwater species such as lampreys or redside dace may also be present; brook trout are the primary top carnivores and are present in good numbers; exotic salmonids are absent or uncommon; tolerant species may be present in low to moderate numbers
80–60	Good	Evidence for some environmental degradation and reduction in biotic integrity: either brook trout or sculpins may be uncommon or absent; exotic salmonids often dominate, keeping the abundance of top carnivores high; tolerant species may be common but do not dominate
50–30	Fair	The stream reach has experienced moderate environmental degradation, and biotic integrity has been significantly reduced: total species richness is often relatively high, but intolerant and native stenothermal coldwater species are uncommon or absent; native stenothermal coolwater species and exotic salmonids may be moderately common, but tolerant eurythermal species or warmwater species or both are usually more abundant
20–10	Poor	Major environmental degradation has occurred, and biotic integrity has been severely reduced: total species richness may be relatively high, but intolerant species, top carnivores, and salmonids are absent; a few native stenothermal coolwater species such as brassy minnows or brook sticklebacks may persist in low numbers; tolerant eurythermal species or warmwater species or both dominate
0 or no score	Very poor	Human disturbance and environmental degradation have decimated the natural coldwater fish assemblage of the reach: either only warmwater and tolerant species remain, or fish abundance is so low (<25 individuals captured) that the IBI cannot be calculated

coldwater (mean, 16.7) or high-quality warmwater sites (mean, 15.9) (F = 44.91; P < 0.0001). No differences in scores were evident between sites from the Driftless Area ecoregion and sites from the Northern Lakes and Forests ecoregion (F = 0.68; P = 0.4174).

Validation of the Coldwater IBI with 1990s Data

For the 1990s sites, all four of the different environmental quality groupings (good, fair, and poor coldwater, good coolwater) had significantly different IBI scores (F = 86.15, P < 0.0001; also different in Tukey paired tests; Figure 2). Estimates of relative biotic integrity closely matched the relative rankings of environmental quality on the basis of physical habitat and water quality for the three groups of coldwater sites but not for the single group of coolwater sites. The coldwater sites that were classified as having good environmental quality had the highest mean IBI score, 67, with a biotic integrity rating of good based on criteria in Table 4. Sites in this group that scored less than 100 points tended to lose points because of less than optimal relative abundances of brook trout or of top carnivores (which includes brook trout), or a relatively high abundance of tolerant species. Coldwater sites classified as having fair environmental quality had the next highest mean IBI score, 52, with a biotic integrity rating of fair to good. Sites in this group tended to lose points because of an absence or low relative abundance of stenothermal coldwater and coolwater species and top carnivores, coupled with a high relative abundance of tolerant species. Coldwater sites classified as having poor environmental quality had the lowest mean IBI score, 6, with a biotic integrity rating of very poor. The five sites classified as high-quality coolwater had a mean IBI score of 28, with a biotic integrity rating of only

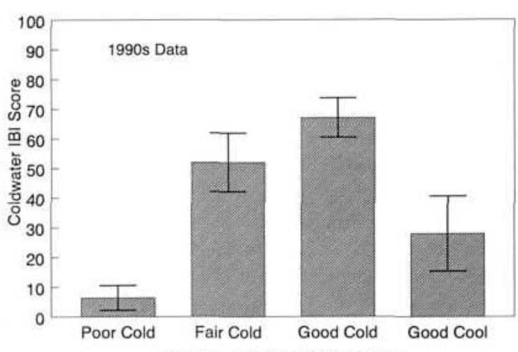

FIGURE 2.—Mean IBI scores and 95% confidence intervals for the four environmental quality groupings for the sixty-one 1990s sites. All four groupings were significantly different from each other.

TABLE 5.—Variation among years in index of biotic integrity (IBI) scores for 17 individual stream sites from the 1990s data set. Numbers in parentheses are our site numbers that distinguish among multiple sites on the same stream. A blank indicates no data for that year.

Stream	County	IBI score by year					Maximum difference between years
		1990	1991	1992	1993	1994	
Bohris Creek (1)	Buffalo	0	20	20	30	20	30
Bohris Creek (1a)	Buffalo	10	20				10
Dunlap Creek	Dane			20	20		0
Eagle Creek (1)	Buffalo			0	0	0	0
Eagle Creek (3)	Buffalo	0	0	0	10	0	10
Gill Creek	Green				20	20	0
Joos Creek (1)	Buffalo		0		0	0	0
Joos Creek (3)	Buffalo			10	0	10	10
Joos Creek (4)	Buffalo			0	0	0	0
Spring Creek	Rock				40	0	40
Story Creek	Dane				70	50	20
Timber Coulee Creek	Vernon				80	60	20
Trout Run Creek	Trempealeau	10	20	20	20	0	20
Wendt Creek	Dane			0	0		0
Widow Green Creek (1)	Adams			60	50		10
Widow Green Creek (2)	Adams			60	60		0
Widow Green Creek (3)	Adams			0	0		0

poor to fair. The coolwater sites received relatively low IBI scores largely because of a high abundance of tolerant species, a scarcity of top carnivores, and a low relative abundance of stenothermal coldwater and coolwater species. All five coolwater sites had intolerant species, and three had small numbers of brook trout.

Annual variation in IBI scores was low to moderate within individual sites. Among the 17 sites with more than 1 year of data, 8 showed no change in IBI scores among years, 4 showed a maximum difference of 10 points among years, 3 had a maximum difference of 20 points, 1 had a maximum difference of 30 points, and 1 had a maximum difference of 40 points (Table 5). For the 2 sites with differences of 30 or more points and 2 of the 3 sites with differences of 20 points, either the highest or the lowest score occurred during 1993, a year with unusually high precipitation and streamflows for the entire summer. Most fluctuations in scores were the result of changes in the relative abundance of tolerant species or the presence or absence of intolerant species. Mean scores among years for the 17 sites were not significantly correlated with the range (r = 0.24, P = 0.3443) or the standard error (r = 0.37, P = 0.1395) of the scores.

The two sites for which we had multiple samples within a year also showed relatively low variation in scores among sampling dates. Wendt Creek, which had an environmental quality rating of poor, had an IBI score of 0 and a biotic integrity rating of very poor for all nine of the monthly samples in 1992 and 1993. Dunlap Creek, which had an environmental quality rating of fair, had scores of 20 in May, June, and July 1992, for a biotic integrity rating of poor, scores of 30 in August, September, and October 1992, for a rating of fair, a score of 40 in November 1992, again for a rating of fair, and scores of 20 in May and June 1993, for a rating of poor. The fluctuations in scores were caused by changes in the relative abundances of tolerant species and stenothermal coldwater and coolwater species.

Discussion

Need for Coldwater Index of Biotic Integrity

We have documented substantial differences between the fish assemblages of coldwater and warmwater streams that make the Wisconsin warmwater IBI (Lyons 1992a) inappropriate for use in Wisconsin coldwater streams. Several of the warmwater IBI metrics are based on taxa or functional groups that are rare or absent in high-quality coldwater streams (darters, suckers, sunfishes), or that show little relation with the relative environmental quality of coldwater streams (invertivorous feeders, simple lithophilic spawners). Scoring of the five species richness metrics in the warmwater IBI (total native, darters, suckers, sunfishes, intolerants) is based on the assumption that species richness generally declines with decreasing environmental quality. This assumption of a positive relation between species richness and biotic integrity underpins all existing versions of the IBI (Fausch et al. 1990; Simon and Lyons 1995). However, this assumption does not appear to be valid in Wisconsin coldwater streams for the total number of native species or for the number of species in several large taxonomic groups, although it is valid for the number of intolerant species. Thus, a different IBI is needed for Wisconsin coldwater streams.

We believe that the inherent dissimilarity between coldwater and warmwater streams in Wisconsin lies in the different thermal preferences of Wisconsin fishes. Most Wisconsin fishes are not adapted to thrive in the cold summer water temperatures that characterize high-quality coldwater streams (Becker 1983; Lyons 1992a). From the perspective of the entire Wisconsin fish fauna, coldwater streams, although common, must be viewed as harsh environments where only a handful of species can live. Species-rich families such as the catostomids, centrarchids, and percids have few or no members adapted for the bioenergetic and reproductive thermal challenges of coldwater streams (Hynes 1970). As a result, coldwater streams have a depauperate fish fauna and lack many of the taxonomic groups that are important in the much more species-rich warmwater streams.

The reasons for the differences in fish assemblage responses to environmental degradation between Wisconsin coldwater and warmwater streams are related to the changes in thermal regions of these streams after degradation. High-quality coldwater streams in Wisconsin have relatively low and fairly stable summer water temperatures, whereas warmwater streams have warmer and usually more variable summer temperatures (Threinen and Poff 1963). Most types of environmental degradation, be they industrial or municipal discharges, channelization, impoundment, or agricultural or urban development of the watershed, directly or indirectly increase both the mean and the variability of summer temperatures in streams, thus moving degraded streams toward a more warmwater thermal regime (Hynes 1960, 1970; Warren 1971; Karr and Schlosser 1978). Degradation of coldwater streams typically makes summer temperatures less suitable for the few coldwater species that are present, but more suitable for a larger number of relatively tolerant eurythermal and warmwater species. The number of coldwater species in the stream may thus decline, but the number of colonizing eurythermal and warmwater species will typically increase by a greater amount, leading to a net increase in species richness. Only when the degradation becomes so severe that even tolerant eurythermal and warmwater species are lost does total species richness decline (e.g., Bryniidson and Mason 1975). Conversely, in high-quality warmwater streams, many tolerant and intolerant eurythermal and warmwater species are already present, and temperature increases caused by environmental degradation do not open the stream to colonization by a different thermal guild of fishes. Instead, intolerant and relatively sensitive warmwater species are eliminated by the degradation and not replaced, leading to a net decline in species richness.

Unless Wisconsin coldwater streams prove to be unusual, the patterns that we have documented are likely to hold in other regions as well. It seems reasonable to assume that the characteristics of Wisconsin coldwater streams are representative of coldwater streams in adjacent areas of northern Michigan, northern, central, and southeastern Minnesota, and northeastern Iowa that are part of the same ecoregions as Wisconsin (Omernik and Gallant 1988). Previous studies have shown strong similarities in coldwater fish assemblages within and among these ecoregions in Wisconsin and Minnesota (Lyons 1989; Poff and Allan 1995). On a broader geographic scale, the fish assemblages of high-quality coldwater streams in Wisconsin are probably comparable to those of high-quality coldwater streams in other parts of North America and Europe. Interregional comparisons have shown remarkable levels of taxonomic and ecological similarity in coldwater stream fish faunas between eastern North America (including Wisconsin), western North America, and Europe (Moyle and Herbold 1987). Although the particular species may differ, coldwater stream fish assemblages throughout North America and Europe are almost identical at the family level, having a relatively species-poor fish assemblage dominated by one or a few salmonids, one to three sculpin species, perhaps a stickleback, and a few cyprinids or catostomids.

Although the assemblage structure of Wisconsin coldwater streams may be broadly representative, it is unclear whether the apparent response of Wisconsin coldwater streams to degradation holds true for coldwater streams in other regions. The increase in species richness that follows degradation in Wisconsin coldwater streams occurs because of the presence of a diverse fauna of relatively tolerant, stream-dwelling, warmwater species that are able to easily colonize degraded streams. In many parts of western and northern North America, the warmwater fish fauna is much less diverse than in Wisconsin (Hocutt and Wiley 1986; Moyle and Herbold 1987), and many fewer eurythermal and warmwater species are available to replace lost coldwater species, possibly resulting in no net increase in species richness with degradation. In California coldwater streams, degradation often led to an increase in overall species richness, but only because of an increase in the number of exotic species as the number of native species declined (Leidy and Fielder 1985; Moyle et al. 1986). Most of the colonizing exotics were warmwater species introduced from eastern North America. Also, in more mountainous regions where colonization from downstream reaches is hindered by rapids or waterfalls, warmwater species might not replace coldwater species, and species richness might not increase after degradation. For example, Leonard and Orth (1986) did not find an increase in species richness with degradation for high-gradient, coolwater streams in the upper New River basin of West Virginia. Although West Virginia has a diverse warmwater fish fauna, waterfalls isolate the upper New River, resulting in a relatively low number of warmwater fish species available to colonize degraded streams.

Throughout North America, more data are needed on the responses of entire fish assemblages in coldwater streams to environmental change. Most previous studies have focused almost solely on salmonids, with relatively little consideration of associated nongame species. As an initial hypothesis, we propose that increased fish species richness in response to degradation is most likely to occur in coldwater streams with low to moderate gradients (≤10 m/km; 1%) that are located in areas with a relatively rich warmwater fish fauna. In addition to the upper midwestern areas of Wisconsin, Michigan, eastern Minnesota, and northeastern Iowa, these two criteria are met in parts of the northeastern U.S. and in southern Ontario, Canada. Documentation and analyses of changes in the overall species richness and species composition of coldwater streams after degradation or restoration would be particularly valuable in these geographic areas. Steedman (1988) did not report an increase in species richness with degradation in his study of southern Ontario streams in the Toronto area, but most of his highest-quality streams were coldwater and most of his lowest-quality streams were warmwater and severely degraded by urban land uses, so the relation between species richness and degradation was complicated by inherent differences in thermal regime and extreme differences in environmental quality.

Validity of the Coldwater Index of Biotic Integrity

The Wisconsin coldwater IBI appears to be a valid and useful tool for assessing the biotic integrity and environmental health of Wisconsin coldwater streams. The biotic integrity ratings derived from IBI scores match independent ratings of environmental quality based on physical and chemical conditions for the 1970s data set, from which the IBI was derived, and more importantly, for the independent 1990s data set. Coldwater streams that have good or excellent physical habitat and water quality tend to have high IBI scores, whereas streams suffering from severe non-point source pollution and habitat degradation usually have low IBI scores. Streams intermediate in environmental quality tend to have intermediate IBI scores. Temporal variation in IBI scores is generally low in the absence of significant environmental change, because IBI scores varied by 10 units or less from month to month or year to year for most 1990s sampling stations. Fluctuations in scores are undoubtedly caused by a combination of natural environmental fluctuations and sampling errors (Lyons 1992a).

Temporal variation in IBI scores for Wisconsin coldwater streams appears to be independent of the relative level of environmental health or biotic integrity, as there was no relation between mean IBI score or environmental quality rating and the range or standard error in IBI scores over time. This finding differs from results from warmwater streams, where temporal variation in IBI scores is often higher at sites with lower biotic integrity and poorer environmental quality (Karr et al. 1987; Rankin and Yoder 1990). Whether this contrast reflects a real ecological difference between coldwater and warmwater streams or merely a statistical difference between the two IBIs is unknown.

The Wisconsin coldwater IBI is simpler than the previously developed warmwater version (Lyons 1992a), reflecting the simpler nature of undegraded coldwater fish communities in Wisconsin. The coldwater IBI has 5 metrics versus 12 in the warmwater IBI, and unlike the warmwater metrics, the coldwater metrics do not require stream size or regional adjustments in scoring criteria. Despite having only five metrics, the coldwater IBI meets all of the criteria for a valid index of biotic integrity (Fausch et al. 1990; Simon and Lyons 1995): it has metrics that reflect different attributes of fish community structure (number of intolerant species), composition (percent tolerant species, cool and coldwater species, and brook trout), and functional organization (percent top carnivores); it has empirically derived, quantitative, metric expectations and scoring criteria that precisely delineate what constitutes a healthy coldwater stream fish assemblage; and it accurately portrays differences in ecosystem health among stream sites throughout Wisconsin. However, we suspect that, because it uses only five metrics, the coldwater IBI is probably less effective than the warmwater IBI in distinguishing small differences in biotic integrity.

Comparisons of the contributions of the five metrics to coldwater IBI scores for streams from different environmental quality groupings suggest that the metrics have different sensitivities to overall levels of environmental degradation. This is typical of most versions of the IBI and is considered a strength of the IBI approach to environmental assessment (Karr et al. 1986, 1987; Fausch et al. 1990; Simon and Lyons 1995). The most sensitive metric is the relative abundance of brook trout, an intolerant, stenothermal coldwater species. This metric appears to be useful for distinguishing between fair- to good-quality coldwater sites and rarely scores above zero for poor-quality sites. The relative abundance of top carnivores metric, which typically is based mostly on stenothermal coldwater salmonids, usually also best contrasts between fair- and good-quality sites. The relative abundance of all stenothermal coldwater and coolwater species and the presence or absence of intolerant species are metrics of intermediate sensitivity and largely distinguish between poor- and fair-quality sites. The metric for relative abundance of tolerant species appears to be the least sensitive and does not consistently discriminate between any of the environmental quality groupings. This metric is also the most variable over time. The relative sensitivities of the coldwater metrics are generally similar to those of comparable metrics from warmwater versions of the IBI, although in warmwater versions the metric for the number of intolerant species is usually highly rather than moderately sensitive (Karr et al. 1986). We do not yet have enough information to determine whether any of the five metrics are more or less sensitive to different types, as opposed to levels, of environmental degradation (e.g., water quality versus habitat quality changes; see Yoder and Rankin 1995).

Application of the Coldwater Index of Biotic Integrity

The coldwater IBI is appropriate for use in all types of coldwater streams throughout Wisconsin. Successful application hinges on an accurate and representative sample of the entire fish assemblage from the stream reach of interest. A previous study indicates that such a sample can be obtained by a single careful electrofishing upstream pass (Simonson and Lyons 1995). One should try to collect all fish longer than 25 mm total length, and all captured fish should be identified and counted. No block nets are needed to isolate the stream reach. Ideally, the reach should be at least 35 times the mean stream width (Lyons 1992b), although a reach as short as 18–20 times the mean width is usually adequate (Lyons, unpublished data). However, the minimum length of stream sampled should never be less than 100 m. Our limited analyses plus results from other studies suggest that sampling should occur during the summer when the stream is at base flow, as interpretation of spring and fall samples may be complicated by seasonal fish movements and variation in recruitment (Angermeier and Karr 1986; Lyons 1992a; Meyers et al. 1992).

When a very small number of fish is captured from a site, the IBI may behave erratically and not accurately reflect biotic integrity and ecosystem health. We advise not to calculate the IBI if fewer than 25 individuals are captured, and instead to tentatively set the biotic integrity rating at "very poor" pending additional biomonitoring (Karr et al. 1986; Lyons 1992a).

Stocked salmonids should not be included in any of the coldwater IBI metric calculations. The abundance of stocked salmonids largely reflects the efficiency of hatchery and distribution systems rather than the biotic integrity of the study site, and inclusion of stocked salmonids will usually inflate IBI scores. Identification of stocked salmonids is based on physical appearance and size distribution in concert with stocking records but is often difficult. Bear in mind that the coldwater IBI does not measure the quality of the coldwater fishery in a stream; a stream with many stocked fish may support an excellent fishery but may or may not have high biotic integrity. Indeed, the replacement of slow-growing native brook trout with faster-growing exotic brown trout in many Wisconsin streams may have improved fishing while reducing biotic integrity.

The coldwater IBI is not appropriate for use in either warmwater and coolwater streams. High-quality warmwater and coolwater streams both scored low and were rated only fair to poor with the coldwater IBI. Thus, it is important that the right version of the IBI be used for the right type of stream. At present, no IBI version exists for Wisconsin coolwater streams, which are common in many parts of Wisconsin, particularly the north.

Management Implications

Our results indicate that a single version of the IBI will not encompass all types of wadable streams in Wisconsin. Differences in fish assemblages among coldwater, coolwater, and warmwater streams are significant, and a separate version of the IBI is needed for each. Although some of these differences among stream types may be specific to Wisconsin, others are likely to be more widespread. Managers and researchers in other regions with coldwater or coolwater streams must determine whether existing versions of the IBI, which have been developed largely from and for warmwater streams, are appropriate for their coolwater and coldwater resources. The Wisconsin coldwater IBI, either in its current form or with appropriate modification, may well prove useful for biomonitoring in other parts of North America, but testing and validation of the index outside of Wisconsin is necessary before it can be applied widely.

A common public perception about ecosystems, created no doubt by misconceptions about the laudable worldwide effort to preserve natural biodiversity, is that higher biodiversity always equals higher biotic integrity and thus better environmental health. However, our results from Wisconsin coldwater streams demonstrate that sometimes lower biodiversity indicates better environmental health. Thus, a management goal of maximizing fish biodiversity in a Wisconsin coldwater stream would be misguided, and if implemented, could reduce biotic integrity. We agree with Angermeier and Karr (1994) and Lyons et al. (1995) that the goal of ecosystem management should be to maximize biotic integrity rather than biotic diversity. By definition, promoting biotic integrity will protect natural levels of biotic diversity, which, as in the case of coldwater streams, may be inherently low, or, as in the case of warmwater streams, may be inherently high. A key to effective ecosystem management is a detailed understanding of what a particular ecosystem or community type should look like in the absence of significant human impacts. Perhaps the greatest value of the index of biotic integrity to management is that it quantifies and formalizes this understanding for the particular ecosystem in question.

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Appendix: Physiological Preference Groups of Selected Wisconsin Fishes

TABLE A.1.—Classification of selected Wisconsin fish species into tolerance, feeding, and temperature preference groups. For brevity, only stenothermal coolwater and coldwater and eurythermal species that are likely to be encountered in Wisconsin coldwater streams are listed; many stenothermal warmwater species and lake or large river species are excluded. For a complete list see Lyons (1992a). Abbreviations for temperature types are as follows: ECD, exotic stenothermal coldwater; ECL, exotic stenothermal coolwater; EEU, exotic eurythermal; NCD, native stenothermal coldwater; NCL, native stenothermal coolwater; NEU, native eurythermal.

Common name	Scientific name	Tolerance[a]	Feeding[b]	Temperature
	Lampreys—Petromyzontidae			
Chestnut lamprey	*Ichthyomyzon castaneus*	I	–	NEU
Northern brook lamprey	*Ichthyomyzon fossor*	I	–	NCL
Southern brook lamprey	*Ichthyomyzon gagei*	I	–	NCL
Silver lamprey	*Ichthyomyzon unicuspis*	I	–	NEU
American brook lamprey	*Lampetra appendix*	I	–	NCL
Sea lamprey	*Petromyzon marinus*	I	–	ECL
	Minnows—Cyprinidae			
Central stoneroller	*Campostoma anomalum*	O	–	NEU
Largescale stoneroller	*Campostoma oligolepis*	O	–	NEU
Redside dace	*Clinostomus elongatus*	I	–	NCL
Lake chub	*Couesius plumbeus*	O	–	NCL
Common carp	*Cyprinus carpio*	T	–	EEU
Brassy minnow	*Hybognathus hankinsoni*	O	–	NCL
Common shiner	*Luxilus cornutus*	O	–	NEU
Pearl dace	*Margariscus margarita*	O	–	NCL
Hornyhead chub	*Nocomis biguttatus*	O	–	NEU
Golden shiner	*Notemigonus crysoleucas*	T	–	NEU
Emerald shiner	*Notropis atherinoides*	O	–	NEU
Blackchin shiner	*Notropis heterodon*	I	–	NEU
Blacknose shiner	*Notropis heterolepis*	I	–	NEU
Spottail shiner	*Notropis hudsonius*	I	–	NEU
Rosyface shiner	*Notropis rubellus*	I	–	NEU
Mimic shiner	*Notropis volucellus*	O	–	NEU
Northern redbelly dace	*Phoxinus eos*	O	–	NCL
Southern redbelly dace	*Phoxinus erythrogaster*	O	–	NEU
Finescale dace	*Phoxinus neogaeus*	O	–	NCL
Bluntnose minnow	*Pimephales notatus*	T	–	NEU
Fathead minnow	*Pimephales promelas*	T	–	NEU

TABLE A.1.—Continued.

Common name	Scientific name	Tolerance[a]	Feeding[b]	Temperature
Blacknose dace	*Rhinichthys atratulus*	T	–	NEU
Longnose dace	*Rhinichthys cataractae*	O	–	NEU
Creek chub	*Semotilus atromaculatus*	T	–	NEU
	Suckers—Catostomidae			
Longnose sucker	*Catostomus catostomus*	O	–	NCD
White sucker	*Catostomus commersoni*	T	–	NEU
Northern hog sucker	*Hypentelium nigricans*	I	–	NEU
Silver redhorse	*Moxostoma anisurum*	O	–	NEU
Golden redhorse	*Moxostoma erythrurum*	O	–	NEU
Shorthead redhorse	*Moxostoma macrolepidotum*	O	–	NEU
Greater redhorse	*Moxostoma valenciennesi*	I	–	NEU
	Bullhead catfishes—Ictaluridae			
Black bullhead	*Ameiurus melas*	O	–	NEU
Yellow bullhead	*Ameiurus natalis*	T	–	NEU
Stonecat	*Noturus flavus*	O	–	NEU
Tadpole madtom	*Noturus gyrinus*	O	–	NEU
	Pikes—Esocidae			
Northern pike	*Esox lucius*	O	TC	NEU
Muskellunge	*Esox masquinongy*	I	TC	NCL
	Mudminnows—Umbridae			
Central mudminnow	*Umbra limi*	T	–	NEU
	Trouts—Salmonidae			
Pink salmon	*Oncorhynchus gorbuscha*	O	TC	ECD
Coho salmon	*Oncorhynchus kisutch*	O	TC	ECD
Rainbow trout	*Oncorhynchus mykiss*	O	TC	ECD
Chinook salmon	*Oncorhynchus tshawytscha*	O	TC	ECD
Brown trout	*Salmo trutta*	O	TC	ECD
Brook trout	*Salvelinus fontinalis*	I	TC	NCD
	Trout-perches—Percopsidae			
Trout-perch	*Percopsis omiscomaycus*	O	–	NEU
	Cods—Gadidae			
Burbot	*Lota lota*	O	TC	NCL
	Sticklebacks—Gasterosteidae			
Brook stickleback	*Culaea inconstans*	O	–	NCL
	Sculpins—Cottidae			
Mottled sculpin	*Cottus bairdi*	I	–	NCL
Slimy sculpin	*Cottus cognatus*	I	–	NCD
	Sunfishes—Centrarchidae			
Rock bass	*Ambloplites rupestris*	I	TC	NEU
Green sunfish	*Lepomis cyanellus*	T	–	NEU
Pumpkinseed	*Lepomis gibbosus*	O	–	NEU
Bluegill	*Lepomis macrochirus*	O	–	NEU
Smallmouth bass	*Micropterus dolomieu*	I	TC	NEU
Largemouth bass	*Micropterus salmoides*	O	TC	NEU
	Perches—Percidae			
Rainbow darter	*Etheostoma caeruleum*	I	–	NEU
Iowa darter	*Etheostoma exile*	I	–	NEU
Fantail darter	*Etheostoma flabellare*	O	–	NEU
Least darter	*Etheostoma microperca*	I	–	NEU
Johnny darter	*Etheostoma nigrum*	O	–	NEU
Banded darter	*Etheostoma zonale*	I	–	NEU
Yellow perch	*Perca flavescens*	O	–	NEU
Logperch	*Percina caprodes*	O	–	NEU
Blackside darter	*Percina maculata*	O	–	NEU
Walleye	*Stizostedion vitreum*	O	TC	NEU

[a] Tolerance types: tolerant (T); intolerant (I); other (O).
[b] Feeding types: top carnivore (TC); not top carnivore (–).