Ecological Indicators 10 (2010) 565–571

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Development of dual fish multi-metric indices of biological condition for streams with characteristic thermal gradients and low species richness

Y. Kanno [a], J.C. Vokoun [a,*], M. Beauchene [b]

[a] University of Connecticut, Department of Natural Resources and the Environment, Storrs, CT 06269, USA
[b] Connecticut Department of Environmental Protection, Bureau of Water Protection and Land Reuse, Hartford, CT 06106, USA

ARTICLE INFO

Article history:
Received 19 February 2009
Received in revised form 20 September 2009
Accepted 24 September 2009

Keywords:
Multi-metric indices (MMI)
Fish
Streams
Thermal gradients
Biomonitoring
Landuse
Index of biotic integrity (IBI)
Southern New England

ABSTRACT

Biological indicators based on fish assemblage characteristics are used to assess stream condition worldwide. Fish-based bioassessment poses challenges in Southern New England, the USA, due to the effects of within-watershed thermal gradients on fish assemblage types, low regional species richness, and lack of minimally disturbed sites. Dual multi-metric indices (MMI) of biological condition were developed for wadeable streams based on fish assemblage characteristics sampled across watershed landscapes with varying levels of human disturbance. A coldwater MMI was developed using streams with drainage area of $\leq$15 km^2, and a mixed-water MMI for streams with drainage areas of >15 km^2. For each MMI development, candidate metrics represented by ecological classes were sequentially tested by metric range, within-year precision, correlation with stream size, responsiveness to landscape-level human disturbances, and redundancy. Resultant coldwater and mixed-water MMI were composed of 5 and 7 metrics, respectively. Stream sites tended to score similarly when the two MMI were applied to transitional sites, i.e., drainage areas of 5–40 km^2. However, some sites received high scores from the mixed-water MMI and intermediate scores from the coldwater MMI. It was thus difficult to ascertain high-quality mixed-water streams from potential coldwater streams which currently support mixed-water assemblages due to ecological degradation. High-quality coldwater streams were restricted to stream sites with drainage areas $\leq$15 km^2. The newly developed fish-based MMI will serve as a useful management tool and the dual-MMI development approach may be applicable to other regions with thermal gradients that transition from coldwater to warmwater within watersheds.

1. Introduction

Bioassessment has become a standard tool used by resource management agencies worldwide to monitor and assess environmental conditions of many freshwater ecosystems (USEPA, 2006; Roset et al., 2007). Since an index of biotic integrity (IBI) was proposed to assess biological condition of warmwater streams in the Midwestern USA based on fish assemblage characteristics (Karr, 1981), multi-metric indices (MMI) have been developed for other regions and other lotic and lentic systems (Miller et al., 1988; Simon, 1999; Stoddard et al., 2008). The multi-metric approach has also been expanded to other taxa such as benthic macroinvertebrates (Kerans and Karr, 1994; Klemm et al., 2003) and periphyton (Fore, 2003).

The Southern New England region, located in the Northeastern USA, encompasses a dense network of wadeable streams, but the fish-based MMI approach has faced several challenges in this region (Jacobson, 1994). First, because the region was recently glaciated, it is characterized by low freshwater species richness (Whitworth, 1996). Previous MMI developed in depauperate regions often resulted in fewer metrics than those from more speciose regions (Langdon, 2001; Whittier et al., 2007; Akre Matzen and Berge, 2008). Second, data on "pristine" stream conditions are lacking in this extensively modified landscape. In addition to current land development, extensive deforestation occurred in late 1800s across the region (Foster, 1992). Past landuse activities probably have lasting effects on contemporary lotic environments (Maloney et al., 2008; Wenger et al., 2008), but the magnitude of these legacies is unknown. Finally, the region harbors a mixture of coldwater, coolwater, and warmwater streams, characteristically occurring along an upstream-downstream gradient within a watershed. Kanno and Vokoun (2008) characterized the fish assemblages across the Southern New England watersheds and documented that stream size was an important factor (with temperature) describing the distributions among three fluvial assemblages.

Despite these challenges, recent work on MMI indicates promise for Southern New England. More MMI have been

* Corresponding author. Tel.: +1 860 486 0141; fax: +1 860 486 5408.
E-mail address: jason.vokoun@uconn.edu (J.C. Vokoun).

1470-160X/$ – see front matter © 2009 Elsevier Ltd. All rights reserved.
doi:10.1016/j.ecolind.2009.09.004

developed and applied in depauperate systems (Lyons et al., 1996; Mundahl and Simon, 1999; Langdon, 2001; Hughes et al., 2004; Kanno and MacMillan, 2004; Bramblett et al., 2005; Whittier et al., 2007; Akre Matzen and Berge, 2008). The MMI approach taken in the State of Vermont (Northern New England) is particularly relevant (Langdon, 2001; VTDEC, 2004). Vermont employs a coldwater IBI for streams with 2–5 species and a mixed-water IBI for streams containing ≥5 species. This classification resembles that identified by Kanno and Vokoun (2008), where fluvial assemblages transitioned from headwater brook trout *Salvelinus fontinalis* dominated streams to blacknose dace *Rhinichthys atratulus*—creek chub *Semotilus atromaculatus* dominated streams, and finally to more diverse assemblages including species such as white sucker *Catostomus commersonii* and fallfish *Semotilus corporalis* in larger wadeable streams. Since Vermont is located in a different ecoregion than Southern New England (i.e., Northeastern Highlands/Northern Appalachian Plateau and Uplands vs. Northeastern Coastal Zone; Omernik, 1987), we believed it was necessary to develop new MMI for the region.

The purpose of this study was to develop coldwater and mixed-water MMI using an extensive dataset collected in the State of Connecticut, located in Southern New England. First, we determined a threshold drainage area size to divide the data into coldwater (smaller stream) and mixed-water (larger stream) sub-datasets. Second, stream sites were classified along a human disturbance gradient created with watershed-level landscape variables known to affect biotic condition. Third, candidate MMI metrics were tested to develop coldwater and mixed-water MMI. Finally, we explored the performance of the two MMI at transitional sites near the cutoff criteria between coldwater and mixed-water.

2. Materials and methods

2.1. Data sources and field methods

Fish assemblage data were collected from wadeable streams across the entire state of Connecticut between 1999 and 2007. Collection locations were selected to support aquatic life use support assessments required under section 305(b) of the Federal Clean Water Act and Connecticut's water quality standards (CTDEP, 2002a). Fish were sampled during base-flow periods (June-September) from stream segments primarily characterized as high to moderate gradient riffle/run habitat. Backpack or tote-barge electrofishers were used depending on stream size. Stream lengths sampled differed among sites; sampling effort was approximately 10 times the average stream width for 3rd and 4th order streams, and between 100 and 150 m for 1st and 2nd order streams. The field crew conducted single-pass electrofishing, proceeding upstream by sampling all available habitats. Fish were enumerated, identified to species, measured for total length, and released (CTDEP, 2002b).

Fish data were screened prior to the MMI development process. Stocked salmonids, hybrid, and unidentified individuals were deleted from the dataset. Stocked salmonids were typically recognizable in the field due to size and external characteristics. Although natural reproduction is common in the study area, brown trout *Salmo trutta* were removed from the dataset in entirety because the state augments natural reproduction with a fry stocking program (CTDEP, 2007). Stream sites with ≥30 fish individuals and ≥2 native species were retained in the dataset after initial screening was completed. As a result, the dataset included fish assemblage data collected from 348 stream reaches across Connecticut spanning drainage areas of 1.16–347.20 km^2.

2.2. Division of dataset for developing coldwater and mixed-water MMI

The dataset contained fish assemblages represented by more than one thermal guild. Using another, larger dataset (collected by three-pass depletion sampling), Kanno and Vokoun (2008) identified four assemblage types; a brook trout-dominated assemblage (Assemblage A), a blacknose dace-creek chub dominated assemblage (Assemblage B), a relatively diverse fluvial-species assemblage characterized by species such as fallfish, white sucker, and longnose dace *Rhinichthys cataractae* (Assemblage C), and a relatively diverse assemblage containing macro-habitat generalists including pumpkinseed *Lepomis gibbosus*, brown bullhead *Ameiurus nebulosus*, and golden shiner *Notemigonus crysoleucas* (Assemblage D). Slight taxonomic differences for Assemblages B, C and D were observed between the eastern and western portions of the study area, but ecological similarities were obvious. A discriminant function based on proportional abundance of species at sites in the larger dataset (Kanno and Vokoun, 2008) was applied to the current dataset to classify stream sites as one of the four assemblage types. Discriminant function analysis was run with arc-sign square root transformed data in program SAS (version 9.1, SAS Institute Inc., Cary, NC). The distribution of assemblage types was then plotted against drainage area to identify a threshold value for division of the parent dataset into coldwater and mixed-water subsets.

2.3. Landscape-level human disturbance and reference condition

The magnitude of human disturbances at a stream site was quantified at the watershed scale. Land cover and use at the watershed scale has been repeatedly demonstrated to affect stream biota (Wang et al., 2001; Stanfield and Kilgour, 2006; Stranko et al., 2008; Wenger et al., 2008). For each stream site, upstream drainage area was delineated based on the 30-m resolution National Elevation Dataset using ArcGIS version 9.2 and Arc Hydro version 1.2 (ESRI, Redlands, CA). The land cover variables calculated for each drainage area included: (1) percent of impervious surface, (2) percent of forested land, (3) road density, (4) road crossing density, (5) population density, (6) dam density, and (7) density of known water quality issues (e.g., industrial discharge permits and leachate reports). Percent of impervious surface and forested land was based on the 2001 National Land Cover Dataset (Homer et al., 2007). Population density was calculated from the 2000 population census (http://www2.census.gov/census_2000/), and road density and road crossing density were derived from a 1:100,000-scale road atlas (http://seamless.usgs.gov/). Locations of dams and known water quality issues were obtained from the Connecticut Department of Environmental Protection records.

A synthetic human disturbance gradient was derived from the seven landscape variables using Principal Component Analysis (PCA) to reduce the dimensionality of the original variables (McCune and Grace, 2002). Percent of impervious surface and forested land were arcsine square root transformed, and dam density and density of known water quality issues were log transformed. The remaining variables were not transformed because data transformation did not improve data normality. PCA was executed on a correlation matrix among the landscape variables in program PC-ORD (version 5, MjM Software, Gleneden Beach, OR).

The resulting human disturbance gradient was then used to classify stream sites into three categories; least, moderately, and most disturbed sites. It was not possible to locate streams which had received minimal human disturbance in the extensively modified landscape, and our "reference condition" represented the least-disturbed condition among available streams (Stoddard et al., 2006). Highly disturbed sites were also under-represented in the dataset. Therefore, rather than a strict trisection of the data, more sites were designated as moderately disturbed than either of least or most disturbed. To ensure that the slight taxonomic differences between the eastern and western portions of the study area do not hamper the regional application of MMI, the least and most-disturbed sites were selected evenly from both the east and west. In addition, the least and most-disturbed sites were selected from the full gradient of drainage areas, so as to minimize differences in stream size among the three stream categories (see Whittier et al., 2006).

2.4. MMI metric selection

A sequence of nearly identical steps was applied to the two data subsets to develop coldwater and mixed-water MMI. Ecological characteristics of fish species were based on regional references (Whitworth, 1996; Halliwell et al., 1999; Armstrong et al., 2001) (online version only). Candidate metrics to be included in MMI were selected or modified from previous studies (Karr, 1981; Miller et al., 1988; Jacobson, 1994; Lyons et al., 1996; Mundahl and Simon, 1999; Langdon, 2001; VTDEC, 2004; Akre Matzen and Berge, 2008). Forty and fifty-five candidate metrics were complied for coldwater and mixed-water MMI, respectively (online version only). The candidate metrics were then divided into 8 ecological classes in each subset, and a single best metric was selected from each ecological class (Whittier et al., 2007). For each MMI development, the following sequence of screening was applied for each metric: range, signal-to-noise, correlation with stream size, responsiveness to landscape-level human disturbances, and redundancy.

2.4.1. Range

Metrics with small ranges of values are unlikely to discriminate discrepancies among streams (McCormick et al., 2001; Klemm et al., 2003; Bramblett et al., 2005; Whittier et al., 2007). Taxonomic richness metrics were eliminated if their range was $\leq$3 species. Any metric failed to pass the range test if $>$70% of values were zero.

2.4.2. Signal-to-noise

Good metrics possess repeatability of results. Within-year variation was tested by comparing the among-reach variance (i.e., signal) to within-year revisitation (i.e., noise). This step was applied only to the mixed-water MMI dataset as low revisitation of smaller streams precluded signal-to-noise evaluation in the coldwater MMI development; this was the only difference in development methodology between coldwater and mixed-water MMI. Seventeen sites (9% of the mixed-water MMI data) were sampled twice within the same summer. Metrics were rejected if their signal-to-noise ratio was $<$2.

2.4.3. Correlation with stream size

Stream size exerts a major influence on the longitudinal shift in fish assemblages (Vannote et al., 1980; Kanno and Vokoun, 2008). To avoid confounding influences from anthropogenic effects, only least disturbed sites were used to derive a linear regression line between drainage area and each metric. Stream size correction was deemed necessary if 95% prediction intervals of the resulting regression lines had overlapping values at both ends of the stream size gradient and if visual inspection of plotted data confirmed that these intervals were not due to a few influential points. For metrics which required stream-size correction, residuals from regression lines were calculated, and the stream-size-corrected metrics replaced the original metrics.

2.4.4. Responsiveness to human disturbances

One-way analyses of variance (ANOVA) were run to test the ability of each candidate metric to differentiate between the least and most disturbed sites. The resulting *F*-statistics were used to select the single best metric from each ecological class. Specifically, the metric with the greatest *F*-value was first incorporated into the MMI, then the metric with the next greatest *F*-value was selected from among other metric classes. This process continued until one metric was selected from each ecological class, so long as the selected metrics were not redundant (see next subsection) and the *F*-values were statistically significant at $\alpha = 0.05$. Therefore, the maximum numbers of metrics included in each MMI were bound by the numbers of ecological classes (8 classes in each MMI). It was also possible for no metrics from a given class to be selected. In addition to the *F*-value comparison, box plots of selected metrics for the least, moderately, and most disturbed categories were visually inspected to ensure that metric scores for the moderately disturbed sites were generally between those for the least and most disturbed sites.

2.4.5. Redundancy

Statistically redundant metrics add little new information. Two metrics were judged to be redundant if their Spearman correlation coefficients were $>$0.70. When a metric pair in different ecological classes was redundant, the metric selected for inclusion first (i.e., greater *F*-value) was retained and the other metric was replaced with a non-redundant metric in its class with the next greatest *F*-value.

2.5. MMI metric scoring

Each metric was scored on a continuous scale that ranged 0–100. Floor and ceiling values for each metric were bound as the 95th and 5th percentiles, respectively, of all sites. Metric scores were calculated as (value − floor)/(ceiling − floor) $\times$ 100 for positive metrics (i.e., values are higher in the least disturbed sites), and (floor − value)/(floor − ceiling) $\times$ 100 for negative metrics (i.e., values are higher in the most disturbed sites) (Blocksom and Johnson, 2009). Total MMI scores were the averages of their composite metric scores, with a potential range of 0–100.

2.6. MMI application to transitional sites

Although stream size influenced the distribution of assemblage types and a cutoff criteria was used to separate the data into coldwater and mixed-water subsets, the transition between the two was not predicted to be distinct. It was likely that a small proportion of sites, especially near the cutoff, were miscategorized in the two subsets (i.e., a true mixed-water site placed in the coldwater MMI dataset, and vice versa). Both MMI were applied to transitional sites, defined conservatively as streams with drainage areas of 5–40 km^2, and the performance was examined with respect to drainage area and assemblage type. Plots of coldwater vs. mixed-water MMI scores and MMI scores vs. drainage area were constructed.

3. Results

3.1. Fish assemblage characteristics

A total of 43 species (29 native, 14 non-native) were collected from the 348 study sites. Species richness increased with stream size, and distributions of the fish assemblage types were generally influenced by stream size (Fig. 1). Similar to a previous finding from an independent dataset (Kanno and Vokoun, 2008), the brook trout-dominated assemblage (Assemblage A: 32 sites) occupied

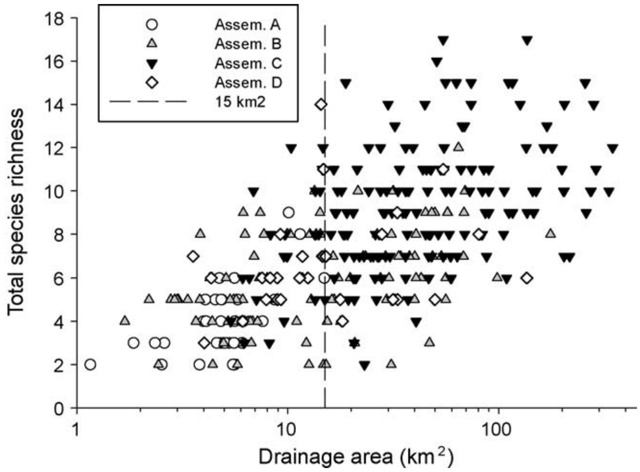

Fig. 1. Relationships between drainage area (km^2) and total species richness in 348 wadeable streams in Connecticut. Assemblage type designation is based on a discriminant function derived from Kanno and Vokoun (2008). Drainage area is on a logarithmic scale.

the headwater streams, the blacknose dace-creek chub assemblage (Assemblage B: 104 sites) were common in streams of intermediate size, and a relatively diverse fluvial assemblage (Assemblage C: 179 sites) dominated our larger wadeable streams. The distribution of the macro-habitat generalist assemblage (Assemblage D: 30 sites) appeared to be independent of stream size.

A cutoff of 15 km^2 was used to cleave the data and define the coldwater MMI-development dataset (138 stream sites $\leq$15 km^2) and the mixed-water MMI-development dataset (210 stream sites >15 km^2). This threshold drainage size was used because the characteristically coldwater Assemblage A was limited to headwater sites with a drainage area of $\leq$15 km^2 and the more diverse Assemblage C became numerically dominant in larger streams. The coldwater MMI subset included 34 species (25 native, 9 non-native), and species richness averaged 6 species (range: 2–14) per stream site. Blacknose dace (113 sites), brook trout (98 sites), white sucker (81 sites), American eel *Anguilla rostrata* (61 sites), and pumpkinseed (47 sites) were the five most common species in terms of occurrence among the 138 sites. The mixed-water MMI subset included 39 species (26 native, 13 non-native), and species richness averaged 9 species (2–17) per stream site. The five most common species were blacknose dace (184 sites), white sucker (180 sites), and American eel (143 sites), tessellated darter *Etheostoma olmstedi* (136 sites), and longnose dace (125 sites) among the 210 sites.

3.2. Human disturbance gradient

Most of the seven landscape variables were correlated to each other and their structure was represented by a single dominant gradient in the PCA. Sixty-one percent of variance was represented by the first PCA axis and it was the only statistically significant axis based on a Monte Carlo randomization test (999 permutation runs, $p = 0.001$). All variables, except dam density, were highly correlated with the first PCA axis ($|r| > 0.50$).

The first PCA axis was used as a synthetic human disturbance gradient. The PCA scores were used to classify stream sites into three categories of human disturbances. In the coldwater MMI subset, 32, 86, and 20 sites were designated as the least, moderately, and most-disturbed sites, respectively. Mixed-water sites included 45 least, 122 moderately, and 43 most-disturbed sites.

3.3. MMI development

The coldwater and mixed-water MMI was composed of 5 and 7 metrics, respectively (Tables 1 and 2). No metrics related to non-native species were selected for either MMI. Mean scores among disturbance categories were significantly different in both MMI (ANOVA; $p < 0.001$) (Fig. 2). Mean MMI scores were different between the most and least-disturbed sites, and the least disturbed sites also differed from moderately disturbed sites in both MMI (Tukey's HSD test; $p < 0.05$). Coldwater MMI scores ranged from 5 to 95 (mean = 48), and mixed-water MMI scores were between 6 and 79 (mean = 47).

The coldwater MMI distinguished Assemblage A from other assemblages, and high scores were assigned to streams in which brook trout were numerically most abundant (Fig. 3). Assemblages B and C were generally scored relative to the degree of human disturbance. Similar to the coldwater MMI, the mixed-water MMI discriminated streams by the degree of human disturbance within Assemblages B and C (Fig. 3). Assemblage D received low scores from both MMI.

3.4. MMI application to transitional sites

Streams tended to be scored similarly when the two MMI were applied to transitional sites (i.e., drainage areas of 5–40 km^2)

Table 1
Metrics included in the coldwater MMI developed for wadeable streams in southern New England (drainage area $\leq$15 km^2).

Metric	Ecological class	*F*-value	Ceiling	Floor
# Brook trout individuals per 100 m^2	Brook trout population	38.9	60.6	0
% Fluvial dependent individuals	Stream flow	19.1	0	71.7
# Warmwater species (stream-size-corrected)	Richness	9.1	-2.39	3.06
% Warmwater individuals	Thermal	8.7	0	87.5
% Brook trout individuals	Indicator species and composition	6.3	86.3	0

Table 2
Metrics included in the mixed-water MMI developed for wadeable streams in southern New England (drainage area >15 km^2).

Metric	Ecological class	*F*-value	Ceiling	Floor
% White sucker individuals	Indicator species	19.6	0	43.9
% Cyprinidae individuals	Composition	15.8	93.7	0.2
% Fluvial-specialist individuals, except blacknose dace	Stream flow	14.7	64.7	0
% Non-tolerant general feeder individuals	Trophic	8.7	51.6	0
% Native warmwater individuals	Thermal	6.3	0	67.9
% Intolerant individuals	Tolerance	5.1	38.1	0
# Fluvial specialist species	Richness	5.1	5	1

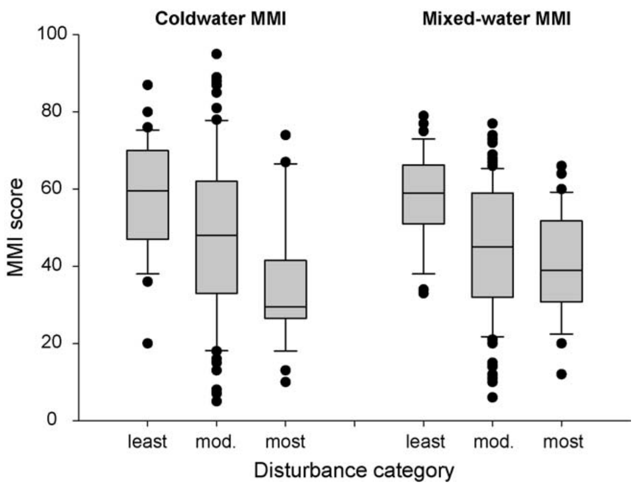

Fig. 2. Coldwater and mixed-water scores for the least disturbed, moderately disturbed, and most disturbed sites. Boxes show the interquartile ranges with medians represented by horizontal lines. Whiskers indicate the 10th and 90th percentiles, and circles represent outliers.

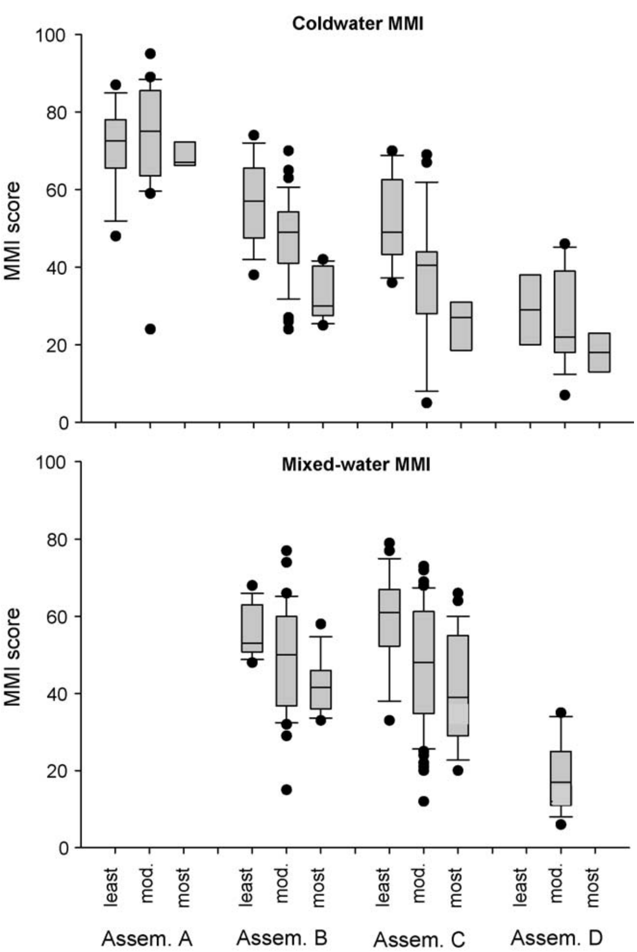

Fig. 3. Coldwater and mixed-water MMI scores by assemblage type. Boxes show the interquartile ranges with medians represented by horizontal lines. Whiskers indicate the 10th and 90th percentiles, and circles represent outliers.

(Fig. 4). However, many higher-scoring mixed-water streams received intermediate scores for the coldwater MMI. Most of these streams were represented by assemblage C. High scores of the coldwater MMI were limited to drainage areas with $\leq$15 km^2 and were dominated by assemblage A (Fig. 5).

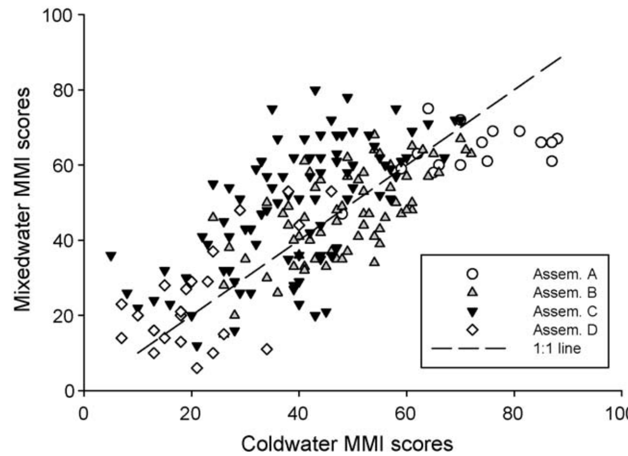

Fig. 4. Comparison of coldwater and mixed-water MMI scores for transitional sites (drainage area: 5–40 km^2) by assemblage types.

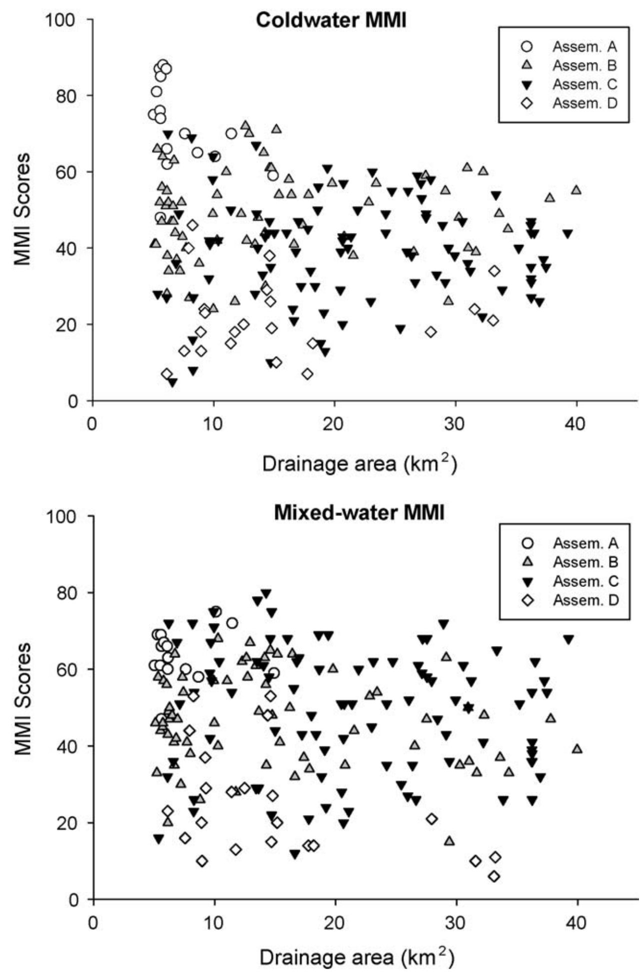

Fig. 5. Relationships between drainage area (km^2) and scores of coldwater and mixed-water MMI for transitional sites (drainage area: 5–40 km^2).

4. Discussion

The structured approach for MMI development (Whittier et al., 2007; Stoddard et al., 2008) systematized and streamlined an objective method of metric selection. The proposed MMI successfully characterized fish assemblage changes in response to landscape-level human disturbances. The small numbers of metrics included in the two MMI were due partly to the low species richness in Southern New England, but also resulted from the metric selection methodology. Certainly, the number of ecological classes chosen for testing restricted the maximum number of potential metrics to be included in MMI.

The composite metrics selected indicated patterns of fish assemblage degradation with increasing human disturbances in coldwater and mixed-water habitats. The pattern of coldwater fish assemblage degradation in our study (i.e., replacement of coldwater species by warmwater species) was similar to those reported in other regions (Lyons et al., 1996; Mundahl and Simon, 1999; Langdon, 2001; Breine et al., 2004; Kanno and MacMillan, 2004; Akre Matzen and Berge, 2008). Two of the five coldwater MMI metrics were based on brook trout (total and relative abundance), with each metric contributing statistically non-redundant information to the final MMI score. It highlighted the importance of this species as an indicator of stream condition in its native range (Lyons et al., 1996; Mundahl and Simon, 1999; Langdon, 2001; Kanno and MacMillan, 2004). Since naturally producing brown trout were not included in the dataset due to the fry stocking program (see Section 1), the coldwater MMI identified high-scoring coldwater resources conservatively. Still, our results confirm that these high-scoring coldwater streams are currently confined to small headwater streams ($\leq$15 km^2) in Southern New England, which has also been reported by Hudy et al. (2006). The signal-to-noise ratio was not examined in the coldwater MMI development, and future work is warranted to explore the temporal variability of fish assemblages in the species-poor headwater streams (Roset et al., 2007).

High mixed-water MMI scores were given to streams with an "equitable" coolwater fish assemblage. These sites typically included a higher proportion of fluvial-specialist and/or intolerant individuals, and increased human-disturbance co-occurred with numerical dominance by habitat generalist individuals (e.g., tolerant and warmwater species) (Miller et al., 1988; Ganasan and Hughes, 1998; Harris and Silveira, 1999; McCormick et al., 2001). Previous MMI developed from more speciose regions typically included several metrics related to taxonomic richness (Miller et al., 1988; McCormick et al., 2001; Oberdorff et al., 2002), but most metrics in this ecological class performed poorly. Instead, proportional abundance was the more important element in our MMI, and this appears to be a shared characteristic among MMI developed for species-poor regions (VTDEC, 2004; Akre Matzen and Berge, 2008). None of the metrics related to non-native species entered into the proposed MMI and their distributions did not reflect the watershed-scale disturbances, similar to the findings of McCormick et al. (2001) and Whittier et al. (2007).

The MMI development in our study was based on human disturbances which were quantified at the watershed landscape level. The study area was located in an extensively modified region, both historically and presently (Foster, 1992), therefore the best MMI scores obtained represent the best available conditions found in the current landscape. A qualitative interpretation of MMI scores (i.e., impaired or degraded vs. reference or high-quality) would be useful, but would need to be done cautiously in such a human-dominated landscape (Stoddard et al., 2006). Plus, fish assemblages also respond to local (Gorman and Karr, 1978) and riparian (Barton et al., 1985) conditions. Integration of human disturbances at multiple spatial scales would more precisely characterize human disturbances and offer a useful insight into management options (Wang et al., 2006; Weigel et al., 2006).

Watershed size was interpreted as a criterion variable which could guide the application of the two MMI. This is due to coldwater assemblages being predominantly limited to small headwater streams in the current landscape. Streams of drainage area with >15 km^2 would be accurately assessed with the mixed-water MMI, irrespective of their assemblage types. However, there remains a challenge for streams of drainage areas $\leq$15 km^2, as some small streams received high mixed-water MMI scores and intermediate scores for the coldwater MMI. Given a presumed temporal pattern of coldwater fish assemblage degradation in the region, potentially due to an interaction of human development and climate change (Nelson and Palmer, 2007), these sites might either be naturally mixed-water small stream habitats or degraded coldwater habitats currently supporting mixed-water fish assemblages. Also, the Southern New England region has undergone extensive deforestation shortly after European settlement followed by a period of agriculture, then land abandonment and stream damming during the industrial revolution and reforestation leading to the current trend of conversion to suburban landscapes and associated increases in impervious surfaces (Foster, 1992), such that landuse legacies are complex and difficult to separate. Future work is warranted to classify the potential for restoration of these small streams. For example, potential coldwater habitat may be modeled and located from landscape variables that are less affected by human activities such as elevation, bedrock geology, and stream size (Seelbach et al., 2006).

5. Conclusions

The development of dual-MMI in Connecticut recognizes the inherent natural gradient of fish assemblages based on stream size related to thermal variability. By applying different MMI to stream sites within the same watershed (in contrast to application among geographic eco- or faunal regions), we believe this approach will be helpful for water resource management in not only the study region but in other places with overriding shifts in fish assemblages driven by strong gradients within watersheds. Connecticut relies principally on benthic macroinvertebrates to satisfy the biological monitoring requirements of the U.S. Clean Water Act. The addition of fish-based monitoring may strengthen capability in this regard, especially if fish and macroinvertebrate assemblages respond to different aspects and scales of human disturbances (Berkman et al., 1986; Flinders et al., 2008).

Acknowledgments

This research was made possible by the Storrs Agricultural Experiment Station through the Hatch Act, the State Wildlife Grants Program and the University of Connecticut. Jason Parent of the Center for Landuse Education and Research at the University of Connecticut provided assistance with GIS programming. The Connecticut Department of Environmental Protection, Bureau of Natural Resources, Inland Fisheries Division and Bureau of Water Protection and Land Reuse collected and provided data. Justin Davis and two anonymous reviewers provided comments that improved this manuscript.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecolind.2009.09.004.

References

Armstrong, D.S., Richards, T.A., Parker, G.W., 2001. Assessment of Habitat, Fish Communities and Streamflow Requirements for Habitat Protection, Ipswich River, Massachusetts 1998–1999. Water Resources Investigations Report No. 01-4161. United States Geological Survey, Northborough, MA.

Akre Matzen, D., Berge, H.B., 2008. Assessing small-stream biotic integrity using fish assemblages across an urban landscape in the Puget Sound lowlands of western Washington. Trans. Am. Fish. Soc. 137, 677–689.

Barton, D.R., Taylor, W.D., Biette, R.M., 1985. Dimensions of riparian buffer strips required to maintain trout habitat in southern Ontario streams. N. Am. J. Fish. Manage. 5, 364–378.
Berkman, H.E., Rabeni, C.F., Boyle, T.P., 1986. Biomonitors of stream quality in agricultural areas: fish versus invertebrates. Environ. Manage. 10, 413–419.
Blocksom, K.A., Johnson, B.R., 2009. Development of a regional macroinvertebrate index for large river assessment. Ecol. Indicator 9, 313–328.
Bramblett, R.G., Johnson, T.R., Zale, A.V., Heggem, D.G., 2005. Development and evaluation of a fish assemblage index of biotic integrity for northwestern Great Plains streams. Trans. Am. Fish. Soc. 134, 624–640.
Breine, J., Simoens, I., Goethals, P., Quataert, P., Ercken, D., Van Liefferinghe, C., Belpaire, C., 2004. A fish-based index of biotic integrity for upstream brooks in Flanders (Belgium). Hydrobiologia 522, 133–148.
Connecticut Department of Environmental Protection (CTDEP), 2002a. Water Quality Standards. Bureau of Water Management, Hartford, CT.
CTDEP, 2002b. Ambient Biological Monitoring-Fish Community Structure Quality Assurance Project Plan. Bureau of Water Protection and Land Reuse, Hartford, CT.
CTDEP, 2007. Connecticut Fish Distribution Report 2007. Bureau of Natural Resources, Hartford, CT.
Flinders, C.A., Horwitz, R.J., Belton, T., 2008. Relationship of fish and macroinvertebrate communities in the mid-Atlantic uplands: implications for integrated assessments. Ecol. Indicator 8, 588–598.
Fore, L.S., 2003. Response of diatom assemblages to human disturbance: development and testing of a multimetric index for the mid-Atlantic region (USA). In: Simon, T.P. (Ed.), Biological Response Signatures: Indicator Patterns Using Aquatic Communities. CRC Press, Boca Raton, FL, pp. 445–480.
Foster, D.R., 1992. Land-use history (1730–1990) and vegetation dynamics in central New England, USA. J. Ecol. 80, 753–772.
Ganasan, V., Hughes, R.M., 1998. Application of an index of biological integrity (IBI) to fish assemblages of the rivers Khan and Kshipra (Madhya Pradesh), India. Freshwater Biol. 40, 367–383.
Gorman, O.T., Karr, J.R., 1978. Habitat structure and stream fish communities. Ecology 59, 507–515.
Halliwell, D.B., Langdon, R.W., Daniels, R.A., Kurtenbach, J.P., Jacobson, R.A., 1999. Classification of freshwater fish species of the northeastern United States for use in the development of indices of biological integrity, with regional applications. In: Simon, T.P. (Ed.), Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities. CRC Press, Boca Raton, FL, pp. 301–337.
Harris, J.H., Silveira, R., 1999. Large-scale assessments of river health using an index of biotic integrity with low-diversity fish communities. Freshwater Biol. 41, 235–252.
Homer, C.C., Dewitz, J., Fry, J., Coan, M., Hossain, N., Larson, C., Herold, N., McKerrow, A., VanDriel, J.N., Wickham, 2007. Completion of the 2001 National Land Cover Database for the conterminous United States. Photogramm. Eng. Rem. S. 73, 337–341.
Hudy, M., Thieling, T.M., Gillespie, N., Smith, E.P., 2006. Distribution, Status, and Perturbations to Brook Trout within Eastern Untied States. Final Report to the Eastern Brook Trout Join Venture, International Association of Fish and Wildlife Agencies, Washington, DC.
Hughes, R.M., Howlin, S., Kaufmann, P.R., 2004. A biointegrity index (IBI) for coldwater streams of western Oregon and Washington. Trans. Am. Fish. Soc. 133, 1497–1515.
Jacobson, R.A., 1994. Application of the Index of Biotic Integrity to Small Connecticut Streams. M.Sc. Thesis, University of Connecticut, Storrs, CT.
Kanno, Y., MacMillan, J.L., 2004. Developing an index of sustainable coldwater streams using fish community attributes in River Philip, Nova Scotia. Proc. N. S. Inst. Sci. 42, 319–338.
Kanno, Y., Vokoun, J.C., 2008. Biogeography of stream fishes in Connecticut: defining faunal regions and assemblage types. Northeast. Nat. 15, 557–576.
Karr, J.R., 1981. Assessment of biotic integrity using fish communities. Fisheries 6 (6), 21–27.
Kerans, B.L., Karr, J.R., 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecol. Appl. 4, 768–785.
Klemm, D.J., Blocksom, K.A., Fulk, F.A., Herlihy, A.T., Hughes, R.M., Kaufmann, P.R., Peck, D.V., Stoddard, J.L., Thoeny, W.T., Griffith, M.B., 2003. Development and evaluation of a macroinvertebrate biotic integrity index (MBII) for regionally assessing Mid-Atlantic Highlands streams. Environ. Manage. 31, 656–669.
Langdon, R.W., 2001. A preliminary index of biological integrity for fish assemblages of small coldwater streams in Vermont. Northeast. Nat. 219–232.
Lyons, J., Wang, L., Simonson, T.D., 1996. Development and validation of an index of biotic integrity for coldwater streams in Wisconsin. N. Am. J. Fish. Manage. 16, 241–256.
Maloney, K.O., Feminella, J.W., Mitchell, R.M., Miller, S.A., Mulholland, P.J., Houser, J.N., 2008. Landuse legacies and small streams: identifying relationships between historical land use and contemporary stream conditions. J. N. Am. Benthol. Soc. 27, 280–294.
McCormick, F.H., Hughes, R.M., Kaufmann, P.R., Peck, D.V., Stoddard, J.L., Herlihy, A.T., 2001. Development of an index of biotic integrity for the Mid-Atlantic Highlands region. Trans. Am. Fish. Soc. 130, 857–877.
McCune, B., Grace, J.B., 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, OR, p. 300.
Miller, D.L., Leonard, P.M., Hughes, R.M., Karr, J.R., Moyle, P.B., Schrader, L.H., Thompson, B.A., Daniels, R.A., Fausch, K.D., Fitzhugh, G.A., Gammon, J.R., Halliwell, D.B., Angermeier, P.L., Orth, D.J., 1988. Regional applications of an index of biotic integrity for use in water resource management. Fisheries 13 (5), 12–20.
Mundahl, N.D., Simon, T.P., 1999. Development and application of an index of biotic integrity for coldwater streams of the upper midwestern United States. In: Simon, T.P. (Ed.), Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities. CRC Press, Boca Raton, FL, pp. 383–415.
Nelson, K.C., Palmer, M.A., 2007. Stream temperature surges under urbanization and climate change: data, models, and responses. J. Am. Water Resour. As. 43, 440–452.
Oberdorff, T., Pont, D., Hugueny, B., Porcher, J.-P., 2002. Development and validation of a fish-based index for the assessment of 'river health' in France. Freshwater Biol. 47, 1720–1734.
Omernik, J.M., 1987. Ecoregions of the conterminous United States. Ann. Assoc. Am. Geogr. 77, 118–125.
Roset, N., Grenouillet, G., Goffaux, D., Pont, D., Kestemont, P., 2007. A review of existing fish assemblage indicators and methodologies. Fisheries Manage. Ecol. 14, 393–405.
Seelbach, P.W., Wiley, M.J., Baker, M.E., Wehrly, K.E., 2006. Initial classification of river valley segments across Michigan's lower peninsula. In: Hughes, R.M., Wang, L., Seelbach, P.W. (Eds.), Landscape Influences on Stream Habitats and Biological Assemblages, American Fisheries Society, Symposium 48, Bethesda, MD, pp. 25–48.
Simon, T.P. (Ed.), 1999. Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities. CRC Press, Boca Raton, FL, p. 671.
Stanfield, L.W., Kilgour, B.W., 2006. Effects of percent impervious cover on fish and benthos assemblages and instream habitats in Lake Ontario tributaries. In: Hughes, R.M., Wang, L., Seelbach, P.W. (Eds.), Landscape Influences on Stream Habitats and Biological Assemblages, American Fisheries Society, Symposium 48, Bethesda, MD, pp. 577–599.
Stranko, S.A., Hilderbrand, R.H., Morgan II, R.P., Staley, M.W., Becker, A.J., Roseberry-Lincoln, A., Perry, E.S., Jacobson, P.T., 2008. Brook trout declines with land cover and temperature changes in Maryland. N. Am. J. Fish. Manage. 28, 1223–1232.
Stoddard, J.L., Larsen, D.P., Hawkins, C.P., Johnson, R.K., Norris, R.H., 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecol. Appl. 16, 1267–1276.
Stoddard, J.L., Herlihy, A.T., Peck, D.V., Hughes, R.M., Whittier, T.R., Tarquinio, E., 2008. A process for creating multimetric indices for large-scale aquatic surveys. J. N. Am. Benthol. Soc. 27, 878–891.
United States Environmental Protection Agency (USEPA), 2006. Wadeable Streams Assessment: A Collaborative Survey of the Nation's Streams, EPA 841-B-06-002. U.S. Environmental Protection Agency, Office of Research and Development, Office of Water, Washington, DC.
Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., Cushing, C.E., 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37, 130–137.
Vermont Department of Environmental Conservation (VTDEC), 2004. Biocriteria for Fish and Macroinvertebrate Assemblages in Vermont Wadeable Streams and Rivers–Development Phase. Available online at http://www.anr.state.vt.us/DEC/waterq/bass/htm/bs_biomon.htm. Accessed November 22, 2008.
Wang, L., Lyons, J., Kanehl, P., 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. Environ. Manage. 28, 255–266.
Wang, L., Seelbach, P.W., Lyons, 2006. Effects of levels of human disturbance on the influence of catchment, riparian, and reach-scale factors on fish assemblages. In: Hughes, R.M., Wang, L., Seelbach, P.W. (Eds.), Landscape Influences on Stream Habitats and Biological Assemblages, American Fisheries Society, Symposium 48, Bethesda, MD, pp. 641–664.
Weigel, B.M., Lyons, J., Rasmussen, P.W., Wang, L., 2006. Relative influence of environmental variables at multiple spatial scales on fishes in Wisconsin's warmwater nonwadeable rivers. In: Hughes, R.M., Wang, L., Seelbach, P.W. (Eds.), Landscape Influences on Stream Habitats and Biological Assemblages. American Fisheries Society Symposium, 48, Bethesda, MD, pp. 493–511.
Wenger, S.L., Peterson, J.T., Freeman, M.C., Freeman, B.J., Homans, D.D., 2008. Stream fish occurrence in response to impervious cover, historic land use, and hydrogeomorphic factors. Can. J. Fish. Aquat. Sci. 65, 1250–1264.
Whittier, T.R., Hughes, R.M., Stoddard, J.L., Lomnicky, G.A., Peck, D.V., Herlihy, A.T., 2007. A structured approach for developing indices of biotic integrity: three examples from streams and rivers in the western USA. Trans. Am. Fish. Soc. 136, 718–735.
Whittier, T.R., Hughes, R.M., Stoddard, J.L., Lomnicky, G.A., 2006. Associations among catchment- and site-scale disturbance indicators and biological assemblages at least- and most-disturbed stream and river sites in the western United States. In: Hughes, R.M., Wang, L., Seelbach, P.W. (Eds.), Landscape Influences on Stream Habitats and Biological Assemblages. American Fisheries Society symposium, 48, Bethesda, MD, pp. 641–664.
Whitworth, W.R., 1996. Freshwater Fishes of Connecticut. Second edition. Bulletin 114. State Geological and Natural History Survey of Connecticut, and Connecticut Department of Environmental Protection, Hartford, CT, pp. 243.