Development, calibration, and validation of a littoral zone plant index of biotic integrity (PIBI) for lacustrine wetlands

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Abstract

We examine lacustrine wetland plant assemblages in the Central Corn Belt Plain portion of the Lake Michigan basin and developed a multimetric plant index of biotic integrity (PIBI). Our objectives were to determine the structural and functional attributes of littoral zone plant assemblages of least-impacted lacustrine wetlands, establish and test candidate metrics, statistically test and calibrate metrics, and finally validate a PIBI along a disturbance gradient. Of 35 candidate metrics, we chose 11 metrics that were grouped into four categories: species richness and composition, species tolerance, guild structure, and vegetation abundance. Based on Spearman correlations, we identified a suite of metrics, particularly those related to species richness and tolerance that had a strong response to human-induced habitat change. The overall PIBI correlated strongly with independent measures of habitat quality ($p < 0.001$) using a qualitative habitat index developed for lacustrine habitats. We validated the lacustrine PIBI by comparing index response to various landuse, landcover, and management types. Least impacted lakes and lakes classified as recreational or undergoing ecological restoration were not statistically separable and received the highest index scores, while the lowest scores were associated with industrial and residential land use. Least-impacted sites differ significantly ($p < 0.001$) from both industrial and residential lakes.

Keywords: Plant IBI; Metrics; Aquatic plant assemblage; Lake assessment; Biological assessment

1. Introduction

Biological integrity is the “ability to support and maintain a balanced, integrated, adaptive assemblage of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region” (Karr and Dudley, 1981). The mission of the Clean Water Act (CWA) is to maintain and restore the physical, chemical, and biological integrity of the nation’s surface waters. At the same time, the Environmental Protection Agency (U.S. E.P.A.) has established a policy of no-net loss of wetlands. The combined goals of wetland quality under the CWA and wetland quantity under E.P.A.

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policy provide an opportunity to meld effective management objectives and prudent conservation measures.

Despite the national no net-loss policy, wetlands are disappearing at an alarming rate as a result of anthropogenic effects (Simon and Stewart, 2006). As wetlands are filled, drained, or tilled, in most cases, no idea of the comparative value of the destroyed wetland exists. In order to establish priorities for future wetland preservation and conservation, it would be desirable to rank wetlands according to relative “quality” or “value” at national (U.S.E.P.A., 2002; Ferreira et al., 2005), regional (Simon et al., 2001; DeKeyser et al., 2003; Rothrock and Simon, 2006), and state level (Gernes and Helgen, 1999; Mack, 2001) scales.

The U.S. E.P.A., Region 5, prescribed an Advanced Identification of Wetlands (AIDID) project to delineate quality of remaining wetlands in the Great Lakes portion of northwest Indiana. The AIDID process entailed selecting wetlands from aerial photographs and maps based on land use, land cover, and wetland heterogeneity. Wetlands that possessed intact riparian corridors as well as heterogeneous emergent and submergent aquatic plant assemblages were selected (the best remaining) for visual inspection. Once at the site, an observer inspected the wetlands, visually judging its integrity, noting the presence of exotics, determining the dominant plant forms, and if seen, noting threatened or endangered species (Stewart et al., 1999; Simon et al., 2001). The rationale was that once the very best wetlands were identified, then efforts could be made to protect them by either direct purchase or working with the property owner. As part of this process we determined that the visual inspection method was inadequate and that a more thorough method that still met the criteria of rapidity was needed. This approach became the hypotheses behind Simon et al. (2001), which developed a plant index of biotic integrity (PIBI) for aquatic plant assemblages in riverine and palustrine wetlands along the southern shore of Lake Michigan. This index used the same rationale as the original fish assemblage IBI (Simon, 1998), but modified 12 metrics to characterize and reflect attributes important to plant assemblages.

Current waterbody assessment practices focus on the development of environmental indicators that are rapid, cost-effective, precise, and repeatable (Herrick and Schaeffer, 1985). Since the development of multimetric indices, the original index of biotic integrity (IBI) has been repeatedly adapted and now is considered a family of indices (Simon, 2000). The development of lake assessment indices has remained one of the last resource types needing environmental indicator development (O’Connor et al., 2000).

Furthermore, despite the success of primary producers as indicators of water quality and hydrological modification (e.g., Seddon, 1972; Whitton, 1979; van Dam et al., 1994; Demars and Harper, 1998), development of rapid multimetric indices that emphasize primary producers lagged behind those for various animal groups. Recent efforts have begun to take advantage of a broad diversity of primary producers including algae (e.g., Hill et al., 2000; Tang et al., 2006) and bryophytes (Heino et al., 2005b), as well as vascular plants.

Vascular plants are currently the focus of rapid assessment indicators for wetlands in the United States (e.g., Davis and Simon, 1995; Stewart et al., 1999; Gernes and Helgen, 1999; Nichols et al., 2000; Simon et al., 2001; Mack, 2001; Cohen et al., 2005; Miller et al., 2006; Rothrock and Simon, 2006). These include both univariate approaches such as the floristic quality index (Swink and Wilhelm, 1994; Herman et al., 1997; Lopez and Fennessy, 2002) and multimetric approaches (Simon et al., 2001; DeKeyser et al., 2003; Miller et al., 2006). For example, Nichols et al. (2000) proposed an aquatic macrophyte community index that relies on morphoedaphic attributes, diversity indices, the relative frequencies of submersed, sensitive, and exotic species, and number of taxa. Mack (2001) tested vegetation IBIs suitable for emergent, shrub-carr, and wooded wetlands in Ohio. DeKeyser et al. (2003) developed an index for the prairie pothole region, while Miller et al. (2006) incorporated measures of floristic quality into a broader headwater index in Pennsylvania. Outside the United States plant assemblages have also proven useful in assessment of aquatic and wetland habitat (Heino et al., 2005a,b), especially those in riverine settings (Haury et al., 1996; Spencer et al., 1998; Ferreira et al., 2002; Iliopoulos-Georgoudaki et al., 2003; Dodkins et al., 2005; Ferreira et al., 2005). Especially noteworthy among these studies was the ability of macrophytes to detect habitat degradation due to changes in land use (Heino et al., 2005a,b) and
differences in index signatures attributable to elevated nitrate as opposed to siltation (Dodkins et al., 2005).

In the southern Great Lakes basin of the United States, only a small remnant of remaining wetlands persist in the Central Corn Belt Plain (Simon et al., 2001), yet they include a rich and varied flora along a measurable disturbance gradient. The high species diversity of wetland plant species (Choi, 2000; Simon et al., 2001) including the variety of life history strategies, sensitivities and tolerances (Swink and Wilhelm, 1994), and response to anthropogenic stressors (Stewart et al., 1999; Wilhelm et al., 2003) provide a strong foundation for development of wetland assessment tools (U.S.E.P.A., 2002).

The purpose of this paper is to further the development of the index of biotic integrity concept for plant assemblages. In this case we propose a PIBI suitable for lacustrine littoral zones based upon a large sample of natural lakes from the glaciated region of northwestern Indiana.

2. Methods

2.1. Study sites

Sixty-five natural lakes from northwest Indiana were chosen as study sites following the least-impacted criteria identification of the ADID project (Fig. 1). These lakes were randomly selected as a subset of the best lakes remaining in the area, but still covered a broad range of quality including industrial, residential, and recreational lakes, as well as, those undergoing ecological restoration or considered least impacted by human activity. Geologically the lakes have similar parent material since they are situated on a single massive depositional sequence composed of glacial till and outwash laid down 14–15,000 years ago by the Wisconsinan glaciation (Fleming et al., 1994). The lakes fall within several physiographic subdivisions of northwestern Indiana, including the Chicago Lake Plain, Valparaiso Moraine, and Kankakee Sand sections. Lake sizes varied from less than 1 ha to nearly 380 ha. Depending upon lake size and the number of vegetated areas available for sampling, study sites ranged from one to four per lake. The number of sites surveyed increased with increasing lake size. For example, lakes less than 100 ha had two sites sampled per lake, while lakes greater than 100 ha had three to four sites depending on the amount of natural vegetated shoreline available. A complete listing of lake sites included in this study is available at www.indiana.edu/~inbsarc/projects_files/projects_fish_files/projects_lakesofindiana.html.

2.2. Sample methodology

Qualitative plant sampling techniques were used to evaluate aquatic vascular plant assemblages. Sampling was done by surveying the site, up to a 500 m distance along the shore in all wetland vegetation zones, submergent, floating-leaved, and emergent. The emergent zone included the immediate shoreline vegetation bounded by the nearer of 4 m from the edge of the water or the transition to upland vegetation. The sampling intent was to perform a qualitative survey achievable in a single site visit, not an exhaustive census, and was targeted at biological diversity and relative abundance estimates (Simon et al., 2001).
All species of wetland obligate and facultative plants were identified and an abundance rating (1, observed; 2, rare; 3, rare/common; 4, common; 5, very common; 6, abundant) assigned to each species. Abundance categories represented the number of individuals of a plant species at a site: “observed” was assigned when only one individual of a species is found; “rare” was assigned when a plant species was found two to four times at a site; “rare/common” was assigned when the plant species was found more than four instances, but was never a common component of the community at a site; “common” species were those that were easily located at a site; “very common” species were slightly dominant at a site, and comprised up to about 25% of the community at a site; and “abundant” species were those that dominated a site, and comprised from 25% to almost 100% of the plant community.

Identifications were done in the field using appropriate floristic manuals (Fassett, 1957; Winterninger and Lopinot, 1977; Gleason and Cronquist, 1991; Swink and Wilhelm, 1994). Unknowns were carried to the laboratory for identification and deposited in the herbarium at Taylor University.

Sample lakes were assessed in two ways for habitat quality. First, based upon best professional judgment (BPJ), each lake site was ranked on a scale of 0–10 by TPS, with 10 representing highest habitat quality. Since limited water chemistry was collected and land-use outside of the immediate shoreline was not incorporated into the rapid habitat index, a qualitative aesthetic scoring was a quality assurance check for determining the completeness of the qualitative habitat index. Secondly, a modified QHEI (Qualitative Habitat Evaluation Index, T.P. Simon, unpublished data) was completed that included qualitative habitat attributes that were assumed to have the most impact on aquatic plant assemblages. These variables included: sediment quality, sediment deposition, littoral and profundal zone quality, shoreline alteration, shoreline length, bank stability, riparian vegetative quality, width of the riparian zone, littoral slope, littoral depth, and bottom morphology.

2.3. Development of metrics, data analysis, and statistics

Between-site similarity of the aquatic plant communities was evaluated by clustering and by ordination techniques (Rohlf, 1997). For clustering analysis, the species × cover matrix was converted to a distance matrix and subjected to an unweighted pair-group cluster analysis. The analysis was performed on both the entire data matrix as well as on a subset of lakes consisting of the 51 sites receiving BPJ rankings of medium to high quality. We expected that the latter approach would clearly demonstrate whether distinctive lake types were present and, therefore, whether one or multiple PIBI tools were needed for assessing lakes in the Great Lakes basin of northwestern Indiana.

The final lake PIBI had 11 metrics in four functional categories following the rationale of Simon et al. (2001). The PIBI metrics were chosen after an evaluation of over 35 characteristics of aquatic plant communities. Metric hypotheses were made a priori and examined graphically against estimates of habitat quality to determine if the patterns found fit expectations. Further quantitative testing was then performed by means of Spearman correlations. The scoring for each PIBI metric follows Karr et al. (1986). In short, each metric was scaled against lake area to detect possible factor ceiling-distributions and the data were then trisected. A score of “5” was assigned to the least impacted characteristics or those representative of reference condition lakes, “3” to the middle grouping that shows character deviation from reference conditions, and “1” to the lowest quality, most impacted characteristics of sites. The total PIBI score was obtained by summing the scores from the 11 metrics, resulting in a maximum score of 55.

The metrics relating to the concept of sensitive and tolerant species rely upon the coefficient of conservatism (CC) developed by Swink and Wilhelm (1994). In our metrics, species with CC values of 8 to the maximum of 10 were considered sensitive; conversely species with CC values of 0–2 were classified as tolerant.

2.4. Disturbance gradient and index validation

To validate the PIBIs, lakes were classified into land use and management categories of least impacted, ecological restoration, recreational, residential, and industrial (U.S.E.P.A., 2001). Lakes were placed into five classes depending on whether: (1) they represented natural “least impacted” conditions of the
region (LI), (2) shoreline development included greater than 20% industrial land use (IND), (3) residential development along the shoreline included greater than 45% development (RES), (4) the lake was managed by the Department of Natural Resources or other local or federal agency for public boating, fishing, or recreation (REC), or (5) the lake included private ownership that was undergoing ecological restoration by environmental groups such as the Sierra Club or other private or public not-for-profit groups (ER).

Least impacted (LI) lakes were classified whether they had intact riparian corridors, heterogeneous habitats including submergent, floating, and emergent vegetation zones, and were aesthetically pleasing. These LI lakes are not reference lakes, but represent the collection of best remaining and minimally disturbed lakes in the region. Ecological restoration (ER) lakes represent lakes with active programs for alien species removal, prescribed burns, and other management factors useful for promoting native species biodiversity. Recreational (REC) lakes are those managed by state, federal, or local government entities as public lakes. These lakes are managed to promote fishing, boating, and recreational opportunities for the public. Residential (RES) and industrial (IND) land use was determined using BASINS (Better Assessment Science Integrating Point and Non-point Sources) (U.S.E.P.A., 2001). Final PIBI scores were subjected to ANOVA and Tukey’s multiple range test.

3. Results and discussion

3.1. Species diversity of lacustrine wetlands

The lacustrine communities of northwestern Indiana have a diverse flora. The 65 lakes in this study supported 226 species from 55 families. This represents about 9% of the species reported for the region (Swink and Wilhelm, 1994). The largest families included Cyperaceae (23 species), Zosteraceae (14 species), and Polygonaceae (12 species). The physiognomy of these species is also broad. The obligate and facultative wetland species included over 30 submergent species, 7 floating-leaved species, and 100 emergent and 24 woody species. Given the limited extent of least-impacted lacustrine habitat remaining in Indiana, it was not surprising that these lakes support many species from Indiana’s Heritage Data Base (Indiana Division of Nature Preserves, 1996). The genera Potamogeton and Utricularia are particularly represented on this list, including P. pulcher, P. pusillus, P. richardsonii, P. robbinsii, U. geminiscapa, U. minor, and U. purpurea.

3.2. Lacustrine ordination

Before proposing PIBIs for wetland plant communities in southern Lake Michigan coastal wetlands (Simon et al., 2001), community patterns proved sharply divergent and required two separate IBI protocols, one for riverine and another for palustrine systems. In this present study of lacustrine wetlands, cluster analysis of 51 medium to high quality sites did not reveal any distinctive subcategories of lakes or

![Cluster analysis of 51 lake sites from northwestern Indiana. Only taxa found in more than two samples were used in the analysis.](image)
lacustrine communities (Fig. 2). Instead the cluster dendrogram had a “chaining” pattern. This pattern is produced when there is a broad spectrum of variant communities with minor differences between each one. We found similar chaining patterns when the entire data set was analyzed (i.e., including poor as well as medium to high quality sites) and when variant clustering techniques were applied. Within the dendrogram (Fig. 2), lakes such as Spectacle, Flint, Canada, Long, and Round Lake had lower species richness, especially of emergents, when compared with those at the opposite end of the dendrogram. Ordination of all lake sites (not shown) provided results similar to those of cluster analysis, i.e., no apparent separation of lakes into distinctive ecological types. Therefore, in this geographical area characterized by a broadly similar glacial parent material (Fleming et al., 1994), a single lake PIBI was considered an adequate assessment tool.

3.3. Candidate metric evaluation

In development of the lake PIBI, over 35 candidate metrics were statistically evaluated. We evaluated candidate metrics using a range test, skewness, signal to noise ratio, and responsiveness test to disturbance as criteria for selection (Hughes et al., 1998; McCormick et al., 2001). Based on this analysis, 11 metrics were chosen for the final lacustrine PIBI (Table 1). Metrics were divisible into four functional categories: species richness and composition, species tolerance, guild structure, and vegetation abundance. Except for the relative abundance of pioneer species (discussed below), the accepted metrics had significant correlations with both BPJ and QHEI estimates of habitat quality ($r^2$ of 0.22–0.69, $p = 0.02$ or less, Table 2). The number of sensitive species generated the highest correlation with habitat quality ($r^2 = 0.67–0.69$, $p < 0.0001$). Seven other metrics, including those based on tolerant and obligate species as well as number of species, also had highly significant responses (i.e., $p = 0.001$ or less). The PIBI, derived from summing the 11 metrics, had a highly significant correlation with our two estimates of habitat quality ($r^2 = 0.62–0.69$, $p < 0.0001$, Table 2).

Four metrics were included in the first functional category—species richness and composition (Tables 1 and 2). As with our palustrine and riverine PIBIs, these metrics captured total species richness, as well as, number of emergent and of submergent species. In the lacustrine setting, the floating-leaved guild was added to the species composition metrics since this guild is diverse in high quality habitats ($r^2 = 0.27–0.48$, $p = 0.006$ or less).

Table 1
Calibration of plant index of biotic integrity (PIBI) for lakes of northwestern Indiana

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Scoring</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1 (worst)</td>
</tr>
<tr>
<td>I. Species richness and composition</td>
<td></td>
</tr>
<tr>
<td>1. Total number of species</td>
<td>0–22</td>
</tr>
<tr>
<td>2. Number of submergent species</td>
<td>Varies with surface area (Fig. 3B)</td>
</tr>
<tr>
<td>3. Number of floating-leaved species</td>
<td>0–1</td>
</tr>
<tr>
<td>4. Number of emergent species</td>
<td>1–10</td>
</tr>
<tr>
<td>II. Species tolerance</td>
<td></td>
</tr>
<tr>
<td>1. Number of sensitive species</td>
<td>0–3</td>
</tr>
<tr>
<td>2. Percent tolerant and exotic species</td>
<td>&gt;36</td>
</tr>
<tr>
<td>III. Guild structure</td>
<td></td>
</tr>
<tr>
<td>1. Relative abundance of obligate wetland species</td>
<td>&lt;12</td>
</tr>
<tr>
<td>2. Relative abundance of pioneer species</td>
<td>&gt;30</td>
</tr>
<tr>
<td>3. Relative abundance of woody species</td>
<td>&gt;25</td>
</tr>
<tr>
<td>IV. Vegetation abundance</td>
<td></td>
</tr>
<tr>
<td>1. Average cover</td>
<td>&lt;2</td>
</tr>
<tr>
<td>2. Relative abundance of exotics</td>
<td>&gt;16</td>
</tr>
</tbody>
</table>

The ranges for the various IBI scores are based upon trisection of the metrics presented in Fig. 3.
Of the 11 metrics included in the PIBI, only the number of submergent species (Fig. 3B), showed a relationship with lake size. Small, least impacted lakes had fewer maximum species than large ones. Presumably, small lakes are shallower and naturally more eutrophic and turbid (Hough et al., 1989), more prone to broad swings in temperature and oxygen, and have more extensive winter ice damage. This metric showed a species area relationship with increasing lake size. For all other metrics, no size relationship was observed so criteria lines were drawn without a surface area relationship (Fig. 3).

Two potential metrics were rejected from the final PIBI. Both the relative abundance of submergent species ($r^2 = 0.25–0.29, p = 0.01–0.003$) and the relative abundance of floating-leaved species ($r^2 = 0.14–0.25, p = 0.16–0.01$) were significantly correlated with at least one index of habitat quality. However, in both cases, these metrics were not only among the weaker predictors of quality ($r^2 = 0.14–0.29$) but, more critically, had strong correlations ($r^2 > 0.78$) with other

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Table 2

Spearman rank correlation coefficients and significance level between proposed plant metrics and two estimates of habitat quality, qualitative ranking (best professional judgment) and QHEI

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Hypothesized change with decreasing quality</th>
<th>Quality measure</th>
<th>$r^2$</th>
<th>$p$</th>
</tr>
</thead>
<tbody>
<tr>
<td>I. Species richness and composition</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1. Total number of species</td>
<td>Decrease</td>
<td>Ranking</td>
<td>0.35</td>
<td>0.0002</td>
</tr>
<tr>
<td>2. Number of submergent species</td>
<td>Decrease</td>
<td>Ranking</td>
<td>0.49</td>
<td>&lt;0.0001</td>
</tr>
<tr>
<td>3. Number of floating-leaved species</td>
<td>Decrease</td>
<td>Ranking</td>
<td>0.54</td>
<td>&lt;0.0001</td>
</tr>
<tr>
<td>4. Number of emergent species</td>
<td>Decrease</td>
<td>Ranking</td>
<td>0.27</td>
<td>0.006</td>
</tr>
<tr>
<td>II. Species tolerance</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1. Number of sensitive species</td>
<td>Decrease</td>
<td>Ranking</td>
<td>0.69</td>
<td>&lt;0.0001</td>
</tr>
<tr>
<td>2. Percent of tolerant and exotic species</td>
<td>Increase</td>
<td>Ranking</td>
<td>−0.55</td>
<td>&lt;0.0001</td>
</tr>
<tr>
<td>III. Guild structure</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1. Relative abundance of obligate wetland species</td>
<td>Decrease</td>
<td>Ranking</td>
<td>0.33</td>
<td>0.0006</td>
</tr>
<tr>
<td>2. Relative abundance of pioneer species</td>
<td>Increase</td>
<td>Ranking</td>
<td>−0.23</td>
<td>0.07</td>
</tr>
<tr>
<td>3. Relative abundance of woody species</td>
<td>Increase</td>
<td>Ranking</td>
<td>−0.24</td>
<td>0.01</td>
</tr>
<tr>
<td>IV. Vegetation abundance</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1. Average cover</td>
<td>Decrease</td>
<td>Ranking</td>
<td>0.22</td>
<td>0.02</td>
</tr>
<tr>
<td>2. Relative abundance of exotics</td>
<td>Increase</td>
<td>Ranking</td>
<td>−0.33</td>
<td>0.0006</td>
</tr>
<tr>
<td>Total PIBI</td>
<td>Decrease</td>
<td>Ranking</td>
<td>0.62</td>
<td>&lt;0.0001</td>
</tr>
</tbody>
</table>

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metrics included in the PIBI, specifically the number of submergent and floating-leaved species.

Disturbance and poor water quality are expected to favor the pioneer species guild (that includes most annuals as well as species such as Lobelia siphilitica, Penthorum sedoides, Carex vulpinoidea, Juncus tenuis, and Leersia oryzoides) and the decline of other native herbaceous species (Ehrenfeld and Schneider, 1991). In lakes where the lacustrine border is subjected to dredging, filling, introduction of carp, herbicides, and shoreline raking to remove macrophytes, the lacustrine fringe and its layered diversity is reduced relative to the woody vegetation of the shore line (Nichols and Lathrop, 1994). As expected, the woody species guild increased along the disturbance gradient in our study sites (Table 2). The response of pioneer species, on the other hand, was less clear: the trend was significant when compared against BPJ disturbance rankings ($r^2 = -0.23$, $p = 0.02$), but not when compared against the QHEI assessment ($r^2 = -0.18$, $p = 0.07$). We have tentatively accepted this metric for use in the current lake PIBI, but this metric should be further evaluated.

Two additional metrics show no response to habitat quality ($r^2 < 0.13$, $p > 0.13$). We hypothesized that the relative abundance of annuals, as seen by Poiani and Johnson (1989), would increase with disturbance. In our lacustrine sites, the number of annual species and their abundance proved too low to provide a signal of habitat quality even though analogous metrics were of value in PIBIs for palustrine and riverine wetlands (Simon et al., 2001). Since a diversity of annual species may emerge from a persistent seedbank in some lake ecosystems (Keddy and Reznicek, 1982), their low presence in this study may reflect the stability of water level in these particular lakes. The second unresponsive metric was the relative abundance of emergent species. This lack of response was unexpected since several tolerant emergent species, i.e., Phragmites and Typha, seem to have great abundance in degraded habitats. We suggest that our result may reflect the limitation of a rapid cover estimate method that uses a qualitative, non-linear scale with a low maximum value.

Animal IBIs include an individual health and condition metric in order to assess the lowest levels of biological integrity (Karr, 1981; Karr et al., 1986). We have not seen clear application of this concept in plants of lacustrine settings. This may stem from the subtlety of queues involved (Tracy et al., 1995; Mal et al., 2002). We suggest that the plasticity of plant growth, the turnover of plant parts through new growth, and the difficulty of separating damage due to habitat degradation from that found sporadically under reference conditions may render this metric of little value in rapid PIBI techniques.

### 3.4. Validation of a lacustrine PIBI

Total PIBI scores from the 65 lake sites ranged from 11 to 51. Sites lacking in vegetation would score zero on a PIBI and were not included in our analysis. Total PIBI scores, after delimitation of integrity classes (Table 3), correspond to lake quality categories ranging from “very poor” to “very good/excellent.”

We evaluated the response of the PIBI among various lake uses, settings, and disturbance gradients. When lakes were categorized by disturbance and setting, we detected significant differences in PIBI scores (d.f. = 4, $p < 0.0001$, Fig. 4). Lake PIBI scores followed a disturbance gradient in that least impacted (LI) lakes and those experiencing ecological restoration (ER) had significantly higher scores than those with associated local watersheds with industrial (IND)
or residential (RES) development ($p < 0.05$). Least impacted lakes had a PIBI range of 39–51, indicative of integrity classes “fair” to “good/excellent”. The range of PIBI for lakes undergoing ecological restoration scores was broader with more lakes falling into the “fair” integrity class. Recreational lakes (REC) also had higher PIBI scores than those from industrial and residential settings and, in fact, were not shown to be statistically different from our least impacted lake sample (Fig. 4). The signal to noise ratio for these five classes showed that IND and RES lakes had the greatest amount of variation in the PIBI. This suggests that lakes falling into these land-categories are dependent upon active management of wetland habitats in order to promote biotic quality. PIBI scores for industrial lake classes never attained the mean of natural lakes of the region, while residential lakes were capable of attaining those levels. The restricted quality of industrial lakes mirrors the results reported for Confined Disposal Facilities (CDFs) in the Great Lakes where passive management of CDFs never matched the reference condition of natural wetland habitats (Wilhelm et al., 2003).

Finally, it is important to set the proposed PIBI protocol into the broader context of biotic integrity (Angermeier and Karr, 1994). The complexity of lacustrine systems dictates that a variety of indicators, capturing multiple organizational and spatiotemporal scales be employed (Dale and Beyeler, 2001). Recent work has documented the lack of congruence between IBI scores across taxonomic groups. The lack of congruence can result from responding to different environmental factors (Chessman et al., 1999; Paavola et al., 2003; Heino et al., 2005a), having asynchronous lag times to the same environmental stress (Findlay and Bourdages, 2000), and/or differing temporal variability (Chessman et al., 1999). In light of potential incongruence, some have proposed indices involving multiple taxonomic groups (Brooks et al., 1998; O’Connor et al., 2000; Zampella et al., 2006). We believe that, while multiple taxonomic group indices have a role in assessing ecological integrity, a carefully selected suite of individual IBI will be more sensitive to the stresses on the system in question.

4. Conclusion

The lacustrine PIBI differentiates least impacted lakes from impacted littoral zones. It has the added benefit of accumulating a species list at a site, including the identification of exotics and threatened and endangered species, and estimates of waterbody condition can be ascertained. Another benefit is that site assessment can take place in a fraction of the time that it takes for a formal assessment with plots and grids. In our experience, as the skill of the taxonomist improved, up to five zones/day could be assessed. This PIBI should prove useful to those interested in assessing and identifying the quality of sites, leading toward enhanced protection of these lacustrine wetlands. When used in conjunction with other monitoring tools for macroinvertebrates and fish, we think that a PIBI may prove particularly sensitive to changes in hydrology, provide additional information on long-term water quality, and anticipate pending change in biotic integrity due to habitat disturbance. This information may be attained without undue sacrifice of time and resources for each site. These metrics should be tested and others substituted and incorporated into additional calibrations of the PIBI for other wetland systems that need protection and conservation.
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