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Review Article

A review of macroinvertebrate- and fish-based stream health indices

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ABSTRACT

The focus of this review is to discuss the current uses and developments of macroinvertebrate and fish indicators in riverine ecosystems. Macroinvertebrates and fish are commonly used indicators of stream heath, due to their ability to represent degradation occurring at the local or regional scales, respectively. A total of 78 macroinvertebrate and fish indices were reviewed, and the frequently used macroinvertebrate and fish indices are discussed in detail in the context of aquatic ecosystem health evaluation. This review also discusses several types of common components, or metrics, used in the creation of indices. Following this, the review will focus on the different methods used for macroinvertebrate and fish collection, in both wadeable and non-wadeable aquatic ecosystems. With the basics of macroinvertebrate and fish indices discussed, emphasis will be placed on the application of indices and the different regions for which they are developed. The final section will provide a summary of the benefits and limitations of macroinvertebrate and fish indices. In general, the majority of studies have been performed in wadeable streams; therefore, our knowledge about these indices in non-wadeable streams is limited, which should be the subject of future research.

1. Introduction

As the human population continues to grow, it can be expected that anthropogenic activities will have impacts on the environment (Walters et al., 2009; Young and Collier, 2009; Dos Santos et al., 2011; Pander and Geist, 2013). This in combination with changing climates will only amplify the impacts on stream ecosystems (Meyer et al., 1999; Ridoutt and Pfister, 2010). To determine how climate change and anthropogenic activities impact aquatic ecosystems, it has been recognized that monitoring the health of streams is required. Furthermore this helps ensure that stream systems are able to function and will be able to provide ecosystem services for future generations (USGS, 2013). Stream health can be defined as the chemical, physical, and biological condition of a stream (Karr, 1999; Maddock, 1999). This definition describes aspects of a very complex system, in which organisms interact with their surrounding and vice versa.

To evaluate stream health three components are often used, which include: chemical, physical, and biological integrity of the surface water (Karr, 1981; Karr et al., 1986; Butcher et al., 2003a). Traditionally of these three, chemical is the most commonly used to evaluate stream health; however, recently it has be recognized that the use of biological integrity can lead to a better understanding of what is occurring in the ecosystem as well as identify the cause of degradations (EPA, 2011). And with the high diversity found within aquatic ecosystems (Pander and Geist, 2013), there are many organisms, such as algae,

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amphibians, diatoms, fish, macroinvertebrates, mammals, microorganisms, periphyton, phytoplankton, plants, reptiles, and zooplankton, that can be included in the decision making process to evaluate the quality of the stream health. Another benefit to using biological indicators for evaluating stream health is that they not only take into account biological factors but also the physical and chemical characteristics of the system (Brazner et al., 2007; Pelletier et al., 2012; Leigh et al., 2013). This is because biological factors are influenced by the physical and chemical characteristics of the ecosystem. By using indicators to evaluate the biotic integrity, environmental resource managers are able to identify degraded areas and can allocate resources to restore the ecosystems with the greatest needs (Butcher et al., 2003a; Walters et al., 2009; Einheuser et al., 2012; Pelletier et al., 2012), in the most cost-effective way (Neumann et al., 2003b). The specific objectives for this study were to (1) determine the origins and applications of macroinvertebrate and fish stream health indices; (2) summarize the benefits and limitations of existing macroinvertebrate and fish stream health indices; and (3) identify the knowledge gaps within the field of biomonitoring that require additional research. This will be done by first reviewing the individual components, collection strategies, and applications of stream health indices. Following these sections the paper will explore macroinvertebrate and fish based indices as well as more detailed reviews of the major indices being used in the field.

2. Stream health indices

Stream health indices are evaluation systems that are used to assess aquatic ecosystems conditions for individual streams (Hu et al., 2007). These indices are also used to for comparison purposes among different ecoregions (Butcher et al., 2003a). In general, stream health indices are divided into three general groups: biotic indices, multi-metric indices, and multivariate methods (Ollis et al., 2006).

2.1. Biotic indices

Biotic indices or uni-metric, such as the Hilsenhoff Biotic Index (Hilsenhoff, 1977), utilize only one metric or characteristic to evaluate stream health. Originally, biotic indices focused on organism tolerances to organic pollution (Hilsenhoff, 1987; Ollis et al., 2006). This allowed for the identification of regional degradations. However there are many stressors that can impact stream health besides organic pollution. Therefore, to advance the use of biotic indices additional organisms should be selected that are sensitive to other pollutions such as nitrogen, sediment, and temperature (Smith et al., 2007; Haase and Nolte, 2008). One of the benefits of biotic index is that stream health can be determined by simple calculation of one metric. However, this approach did not take into account the combined impacts of multiple stressors within streams or the complex nature of stream ecosystems. This led to the development of more complex stream health indices such as multi-metric indices and multivariate methods.

2.2. Multi-metric indices

Multi-metric indices, such as the Index of Biotic Integrity (Karr, 1981) and the Benthic Index of Biotic Integrity (Kerans and Karr, 1994), utilize several metrics or characteristics to evaluate stream health. The development of multi-metric indices takes into account the following factors: metric selection (Stoddard et al., 2008), survey design (Hughes and Peck, 2008), sampling procedures (Hughes and Peck, 2008), organism taxonomic identification level (Waite et al., 2004; Chessman et al., 2007), number and types of sampled habitats (Chessman et al., 2007), and organism classification and identification (Cuffney et al., 2007). By accounting for the complexity of stream ecosystems a more comprehensive view of what is occurring within streams can be made (Thorne and Williams, 1997; Rakocinski, 2012). This provides decision makers and stakeholders with more detailed information about the degradation within the streams and allows them to effectively implement mitigation practices. However, with the increased complexity of multi-metric indices the calculations required to determine stream health are more complicated than those used by biotic indices.

2.3. Multivariate methods

Multivariate methods require the development of models to relate physical and chemical stream features to observed organisms (Wright et al., 1998). Several commonly used multivariate models include the River Invertebrate Prediction and Classification System (RIVPACS), Two-Way Indicator Species Analysis (TWINSPAN), Detrended Correspondence Analysis (DCA), and the Australian River Assessment Scheme (AusRivAS). After the models were developed, they can be used to evaluate stream health beyond sampling points. The data inputs to the models can be simulated from calibrated watershed models. This makes multivariate methods very useful for identifying degraded areas. However, the model development can be challenging and there is an uncertainty in their predictions. Therefore, it is recommended that multivariate methods be used in combination with multi-metric and biotic indices for evaluating the stream health (Reynoldson et al., 1997).

3. Metrics

Metrics are individual characteristics of the ecosystem used to provide information about the conditions within streams (Barbour et al., 1999; Butcher et al., 2003a). Biological metrics include species abundance and condition, species richness and composition, and trophic composition. These metrics are used to describe stream health (Van Hoey et al., 2007) through development of stream health indices.

3.1. Species abundance and condition

Metrics that are used to describe the number and condition of each species found in the rivers are known as species abundance and condition metrics. These include the number of species collected, such as the number of Ephemeroptera taxa collected per sample (Walters et al., 2009), or determining the percentage of injured individuals in a sample, such as the percentage of individuals with disease, tumors, fin damage, and skeletal anomalies (Karr et al., 1986). In many multi-metric indices, the use of abundance and condition metrics is common (Houston et al., 2002; Boyle and Fraleigh, 2003; Butcher et al., 2003a; Couceiro et al., 2012; Magbanua, 2012). Often abundance indicators are used to evaluate key or sensitive macroinvertebrate and fish families, such as the EPT (Ephemeroptera/Plechoptera/Trichoptera) index, to provide information about the conditions in the stream. In general, for identical streams, the system with more sensitive organisms is less impacted by anthropogenic activities (Johnson et al., 2013).

3.2. Species richness and composition

Metrics that fall under the category of species richness and composition are used as a qualitative measure to approximate the diversity found in the ecosystem. This not only gives an overview of what is found in the stream, but it can also indicate the stream health based on species distributions. In general, the presence of dominant species or absence of rare species indicates impacted environments (Wan et al., 2010). Furthermore, it has generally been shown that regions with high biodiversity are in better condition and show less degradation, while the opposite condition, of low biodiversity, often indicates a region with more degradation (Boyle and Fraleigh, 2003). This is calculated by recording the number of different taxa, from the highest taxonomic level, such as order, to the lowest possible level, such as genus or species, taken from a stream sample (Smith et al., 2007). In many multi-metric indices, including the Index of Biotic Integrity, the Benthic Community Index, and government indices such as the Alabama Department of Environmental Management Index, include the use of species richness and composition metrics (Houston et al., 2002; Boyle and Fraleigh, 2003; Butcher et al., 2003a; Couceiro et al., 2012; Magbanua, 2012). Another example of indices that utilize species richness metrics are the Simpson and Shannon diversity and richness indices (Keylock, 2005). The Simpson index is based on the probability that two randomly selected organisms from a set are the same species. Meanwhile, the Shannon index is the proportional abundances of each species within a set. While the Simpson and Shannon indices use different approaches, both take into account species richness and composition to provide diversity scores that can be used to describe the stream biodiversity (Keylock, 2005).

3.3. Trophic composition

Metrics that fall under the trophic composition category are used to study the transfer of energy and nutrients through the system (Cummins and Klug, 1979). Trophic levels or functional feeding groups are categories that describe an organism's role in the food web. Benthic macroinvertebrates can be classified in one or more of the following functional groups: collectors, scrapers, shredders, piercers, and predators (Cummins and Klug, 1979; Couceiro et al., 2012). While fish can be classified as herbivores, insectivores, planktivores, piscivores, and omnivores (Karr, 1981). Each functional group has a specific role in the ecosystem; collectors either filter or gather nutrients from the water, scrapers live on the rocks on the streambed and scrap off organic material to eat, shredders break down biomass such as leaves, and piercers and predators actively hunt other organisms for a food supply. Similarly herbivores feed off plant life within the streams, insectivores feed off macroinvertebrates, planktivores feed off microscopic organisms, piscivores feed off other fish, and omnivores feed off both plants and other organisms. Since macroinvertebrates and fish can be found in every functional level (Karr, 1981; Barbour et al., 1999), they can be used to develop an overall picture of the ecosystem. To use these metrics, the functional feeding group of each organism taxa is determined by classifying each taxa by its method of food acquisition for macroinvertebrates (Cummins and Klug, 1979) and trophic level for fish (Karr, 1981). This distribution of the functional feeding groups within the system is used to evaluate the status of the stream. Often changes in the functional feeding groups are driven by nutrient changes (Smith et al., 2007), which means that the use of these metrics can provide information about the chemical composition of the river system. Similar to the species richness metrics, many multi-metric indices, including the Index of Biotic Integrity, Benthic Community Index, and government indices such as the Florida Department of Environmental Protection Index, use function feeding group metrics (Karr, 1981; Houston et al., 2002; Boyle and Fraleigh, 2003; Butcher et al., 2003a; Couceiro et al., 2012).

4. Application

Studies involving macroinvertebrate and fish communities often focus on either defining stream health in a region through the development of a new index (Butcher et al., 2003a), using a previously created index (Butcher et al., 2003b), testing an index to see if it can identify a known stressor (Compin and Céréghino, 2003), comparing the results of different indices in one region (Justus et al., 2010), or testing to see if a previously created index can be applied to a new region (Muxika et al., 2005). The first type of study is preformed to provide an index that can be used for streams in the region; stakeholders and governments can use these studies to implement projects to improve the locations within the region that require it the most. Testing already known indices is performed to see if the current index can be extended to include more results about the ecosystem. If the results of the study are positive, this shows that the index can be applied to more regions and provide a more complete understanding of the environment (Compin and Céréghino, 2003). The comparison studies between different indices are very useful on several levels. First, it identifies the best index to use for stream health evaluation in the region; second, it allows generalizations to be drawn about indices and what they can determine. This was the case in the study by Justus et al.

(2010), where macroinvertebrates were not as capable as algae at detecting low concentration changes in nutrients levels. However, the macroinvertebrates were able to respond to the low nutrient concentrations better than the fish community. The final type of study was to determine if an index can be applied to a new region. This is important because it can expand the use of indices to provide information about the region without having to create a new index. One example is the AZTI Marine Biotic Index (AMBI), which was originally developed by Borja et al. (2000) but applied by Muxika et al. (2005) to six different costal sites throughout Europe with the goal of determining the suitability of the index for evaluating the health of the ecosystems. These sites ranged from the Baltic to the Mediterranean Seas. After evaluating all of the sites, it was determined that the AMBI was a suitable choice for all European coastal ecosystems. At the same time these studies have the chance of showing that the index in question cannot be applied to the region without modifications.

5. Sampling protocols

Since the majority of metrics used for indices are based on observations of macroinvertebrate and fish communities found in rivers, strategies are needed to collect samples for analysis. While individual strategies may change from study to study, such as number of samples or equipment used for sampling; all require the use of individuals, either volunteers or trained workers, to go out and take samples (Butcher et al., 2003b). Often this includes taking samples at different times of the year to determine the general condition year round (Neumann et al., 2003a). However, the actual process of collecting the samples is not uniform across all regions; this brings up the need for different monitoring strategies to capture stream network characteristics. The river continuum concept describes the predictable physical and biological patterns seen in different regions of rivers (EPA, 2014). Based on the river continuum concept, headwater organisms are dependent on interactions with the riparian zone for sources of energy and nutrients; therefore, the macroinvertebrate communities found there are primarily composed of collectors and shredders (Vannote et al., 1980). However, for large rivers, organic transport from upstream (headwaters and medium-sized streams) and algae are the major sources of nutrients and energy, completely replacing the significance of the riparian vegetation (Vannote et al., 1980). This change in primary production source also changes the community composition, for example the macroinvertebrate communities are mainly collectors (Vannote et al., 1980).

In the river continuum concept, three physically based categories are used to describe the ecological regions of a river system; headwaters (stream orders 1–3), medium-sized streams (stream orders 4–6), and large rivers (stream orders > 6) (Vannote et al., 1980). However, the type of equipment and the ease of sampling are largely dependent on the size of the rivers. Therefore, the Environmental Protection Agency (EPA) introduced the concept of wadeable and non-wadeable streams that is not ecological driven and solely based on the river size (EPA, 2006). In this concept, stream order 5 is generally used as a break point. Stream orders 1–5 generally represent wadeable streams and stream orders greater than 5 generally represent non-wadeable streams (EPA, 2006). It is important to note that the stream order concept does not always correspond to wadeability or river size. Overall the majority of wadeable streams express the patterns seen in headwaters and medium-sized streams while non-wadeable streams express the patterns seen in large rivers. Understanding the patterns described in the river continuum concept allows for the creation of indices that can accurately capture the expected community populations and detect degradation within wadeable and non-wadeable stream systems.

5.1. Wadeable waterways

Streams are classified as wadeable by the EPA when they are shallow enough to take samples from without using a boat (EPA, 2006). The EPA mainly focus on these streams since they represent about 90% of the perennial streams and river miles in the United States (EPA, 2006). For macroinvertebrate sampling, a variety of methods exist, these include: surbers, hesses, D-frame dip nets, rectangular dip nets, and kick nets (Plafkin et al., 1989). Subers are $0.3\ m \times 0.3\ m$ square frame nets that are placed on cobble substrates in shallow water ($<0.3\ m$) and are used to collect dislodged organisms (Plafkin et al., 1989). Hesses are 0.5 m diameter metal cylinders that are used similarly to subers, however they can be used in slightly deeper water ($<0.5\ m$) and prevent organisms from escaping (Plafkin et al., 1989). D-frame dip nets ($0.3\ m \times 0.3\ m$), rectangular dip nets ($0.5\ m \times 0.3\ m$), and kick nets ($1\ m \times 1\ m$) are recommended for use in stony riffles and runs with depths smaller than 1 m (Plafkin et al., 1989). The technique used to collect organisms with these three methods is similar: the stream bed is disturbed and a collection net is dragged along parallel to the disturbance, collecting the displaced macroinvertebrates (Plafkin et al., 1989; Butcher et al., 2003a; Young and Collier, 2009; Couceiro et al., 2012). Of these five recommended methods the most often used is the kick net method. For all of these methods, nets are used to collect the organisms; however, the mesh size can vary based on the goal of the project and the location of the study. For example the standard mesh size suggested by the EPA for benthic macroinvertebrate sampling is a $500\ \mu$ screen (EPA, 2012). The organisms collected in the nets are transferred to containers (Barbour et al., 1999), which are sent to labs for analysis. However it is important to note that the kick net method, while very popular, has some errors. During the collection process only those organisms residing near the stream bed and transects are caught, which leads to an incomplete community sampling (Blocksom and Flotemersch, 2005). To account for this deficiency, multiple transects per site should be performed to provide a more complete view of stream organisms (EPA, 2002).

As for sampling fish communities in wadeable streams, electrofishing is commonly used (Plafkin et al., 1989; Terra et al., 2013). In electrofishing, a direct current is applied to the water to stun nearby fish (Plafkin et al., 1989). Once stunned they can be collected with nets and placed in buckets for field identification before being released back into the stream (Plafkin et al., 1989). There are some limitations with electrofishing including misrepresentation of fish populations during seasonal migrations (Zalewski, 1983; Roset et al., 2007). This can be somewhat mitigated by taking multiple samples from the same site at different times throughout the year. Furthermore, smaller fish are less efficiently collected using the electrofishing technique.

5.2. Non-wadeable waterways

Streams are classified as non-wadeable when they are too large for an individual to take samples without the use of a boat (EPA, 2006; Rossaro et al., 2007). These water bodies include coastal regions (Muxika et al., 2005), estuaries (Puente et al., 2008), large rivers (Angradi and Jicha, 2010), and lakes (Rossaro et al., 2007; Launois et al., 2011). For macroinvertebrate sampling, wadeable techniques can be used in shallow river edges while deeper regions of the river can be sampled by using drift nets and multi-plate samplers (Blocksom and Flotemersch, 2005). Drift nets are anchored to stream beds with steel rods and trap macroinvertebrates as they drift with the current; however this method is generally recommended for depths not exceeding three meters (Lazorchak et al., 2000). Multi-plate samplers are stacks of plates with spacers between the plates that are secured to the bottom of the river and left for a few weeks before being retrieved. After collected the multi-plate samplers, macroinvertebrates are gathered from the gaps between the plates (Wisconsin DNR, 1995).

For fish sampling, electrofishing and trawling nets are used (Esselman et al., 2013; Harrison and Kelly, 2013). Electrofishing is conducted from a boat and the stunned fish collected with nets for identification and release (Esselman et al., 2013). Trawling nets are used for deep coastal regions and lakes. In this technique a net is dragged behind a boat to collect fish for identification (Harrison and Kelly, 2013).

6. Macroinvertebrate- and fish-based indices

6.1. Macroinvertebrate-based indices

One group of organisms that is often used for determining stream health are macroinvertebrates (EPA, 2013). They are useful at determining local sources of degradation due to their limited mobility within the stream channel (Kerans and Karr, 1994). Also, macroinvertebrates are sensitive to low levels of pollutants allowing for early detection of stream degradation (Compin and Céréghino, 2003). Due to the frequent use of macroinvertebrates (Flinders et al., 2008; Sharma and Rawat, 2009; Pelletier et al., 2012), many indices have been developed and are used to monitor stream health. Table A1 presents 41 macroinvertebrate indices that were reviewed in this study. The first column indicates the name of the index and its reference. The second column indicates the index that it was based on. The third column presents the changes or modifications made from the based index to create the new index. The fourth column describes specific characteristics of the index such as the number of metrics, score trends, or aspect that is evaluated. The fifth column describes the stream size in which the index is applied, with a total of three possibilities: wadeable streams, non-wadeable streams, and wadeable and non-wadeable streams. And the final column lists the metrics used for each index.

The indices presented below offer many different techniques for evaluating stream health. However, these indices generally originate from four common indices, which include Benthic Index of Biotic Integrity (B-IBI), Hilsenhoff Biotic Index (HBI), Ephemeroptera, Plecoptera, Trichoptera (EPT) Index, and the Biological Monitoring Working Party Index (BMWP). These indices can look at many aspects of the ecosystem, such as the B-IBI, or focused on one particular characteristic of the environment, such as the BMWP index. Out of the 40 macroinvertebrate indices listed in Table A1, 15 used EPT as the base index. This made EPT the most often used base index. Of the modifications made to the EPT index, the most common was the addition of metrics that evaluated other aspects of the streams, such as the presence of other organisms, for example the number of Chironomidae (Houston et al., 2002), or functional feeding groups metrics, for example the % filters (Houston et al., 2002). This allowed the new index to provide a better picture of the conditions within the stream as well as take into account local characteristics. The following sections describe the major macroinvertebrate indices into three groups according to the stream health grouping (biotic indices, multi-metric indices, and multivariate methods).

6.1.1. Macroinvertebrate-based biotic indices

6.1.1.1. Hilsenhoff Biotic Index. The Hilsenhoff Biotic Index (HBI) is a commonly used (Butcher et al., 2003a) index developed by Hilsenhoff in the 1970s (Hilsenhoff, 1977). It was based on the tolerances of each observed taxa in the river system to organic pollutants (Hilsenhoff, 1987). Therefore, HBI is used as an indicator for chemical degradation within river systems. To use this index, samples are taken from the river and used to determine the average tolerance value for the system (Hilsenhoff, 1987). After recording all of the tolerances each river segment is ranked on a scale from 0 to 10, with 0 being the best (Goetz and Fiske, 2013). This value can be compared to other sites to determine the degradations across the region. To allow for a faster analysis of the system, Hilsenhoff provided a table describing the HBI values and their corresponding stream health classification. The scores were grouped into seven water quality categories of: Excellent, Very Good, Good, Fair, Fairly Poor, Poor, and Very Poor. Each water quality category represents a different level of organic pollution based on the dissolved oxygen level (Hilsenhoff, 1987). For example, an Excellent water quality category corresponds to no apparent organic pollution and a score range of 0.00–3.50, while a Very Poor water quality category corresponds to severe organic pollution and a score range of 8.51–10.00. Continued use of the HBI has also led to the discovery that this index can also be used to identify regions with low dissolved oxygen and related temperature regimes (Butcher et al., 2003a). Additionally, the HBI has become a very useful measurement of stream heath to the point where it has been included as a metric in other multi-metric indices (Butcher et al., 2003a) to provide information about the condition of the stream with respect to organic pollutants. However, the number and type of organisms in the stream varies based on the location and size of the streams as suggested by the river continuum concept. Therefore, the organisms originally ranked for use in the HBI may not naturally occur in all rivers, so additional organisms need to be added to insure the HBI captures what is happening within the river systems (Chessman, 1995). Furthermore, the organisms used in the HBI index can be sensitive to several stressors, such as stream flow and nitrogen. This can lead to inaccurate results using HBI (Lenat, 1993; Hilsenhoff, 1998; Barbour et al., 1999). Finally, the presence of tolerant organism communities may not be indicative of a degraded system, however these organisms increase HBI scores, which can be misleading (Hilsenhoff, 1998).

Other studies have taken the concept used for the HBI and applied it to other stressors to make new indices. One example of a new index that is based on the HBI, is the Nutrient Biotic Index (NBI), which instead of considering the impacts of organic pollutants; it was developed to assess the tolerances of organisms to nutrient loading within aquatic ecosystems and in particular wadeable streams (Smith et al., 2007). To do this, two different indices were created, one for nitrogen (NBI-N) and one of phosphorous (NBI-P). Stream health is calculate with these indices by ranking organisms based on their tolerance to nitrogen and phosphorous; after this step stream samples can be taken used to determine the average nitrogen and phosphorous tolerance scores (Smith et al., 2007). These scores are used to compare between different streams and locate the optimal concentration needed of each nutrient for organism survival (Smith et al., 2007). Smith et al. (2007) identified the tolerances of 164 collected taxa and ranked them from a 0 to 10 scale where 10 indicated high tolerance and 0 low tolerance (Smith et al., 2007). This allowed for comparisons between different streams and evaluation of the nutrient loading in the study region. Using the concept of HBI to evaluate nutrient loading was also used in Haase and Nolte's study (2008). The Invertebrate Species Index (ISI) was developed to determine stream health and in particular the impacts of eutrophication in Queensland, Australia (Haase and Nolte, 2008). They scaled the sensitivity of macroinvertebrate species from 1 to 10, where a score of 10 means the species is very sensitive to pollution and a score of 1 means the species is resistant (Haase and Nolte, 2008), exactly the same as the HBI and NBI. Once all the sensitivity scores were determined an average score is calculated to represent the conditions within the stream (Haase and Nolte, 2008). In Haase and Nolte (2008), tolerances were determined for 203 species of macroinvertebrates, which were used for comparison and evaluation of the upland streams in southeast Queensland, Australia. However, due to changes in community compositions, it was noted that ISI species related scores that were calculated for the stream classifications may not be accurate in other regions (Haase and Nolte, 2008). But if the organisms are ranked again for the new region, this index would be useful for identifying nutrient based degradations within stream systems. In addition to NBI and ISI, other indices were developed for calculating stressor tolerances. A study by Meador et al. (2008) looked at organism tolerances to dissolved oxygen, nitrite plus nitrate, total phosphorus, and water temperature. This shows how versatile the concept of organism tolerances is, and the need for studies to explore organism tolerances to other stressors.

Table A1 presents the metrics used in HBI as well as the metrics used in other indices that are either based on or use the HBI for analysis. Of the original metrics listed in Table A1, the most common adjustment to the HBI was to change the stressor being evaluated. The HBI looks at organism tolerances of organic pollutants, while the indices based on the HBI look at organism tolerances to other stressors such as nutrients (NBI) or temperature (TIV).

6.1.1.2. Biological Monitoring Working Party. The Biological Monitoring Working Party Index (BMWP) was developed by the UK Biological Monitoring Working Party in 1978 to evaluate stream health in both England and Wales (Chesters, 1980; Paisley et al., 2014). Since its development, it has become a commonly used index throughout the world (Junqueira and Campos, 1998; Mustow, 2002; Monaghan and Soares, 2010; Navarro-Llácer et al., 2010; Gutiérrez-Fonseca and Lorion, 2014). To determine the stream health based on the BMWP, macroinvertebrate organic pollution tolerances were determined by relating macroinvertebrate presence to stream organic pollution levels based on dissolved oxygen (Chesters, 1980; Hawkes, 1998; Junqueira and Campos, 1998), this is similar to the technique used in the HBI (Hilsenhoff, 1987). However, the scoring system is reversed, while the HBI has tolerance rankings from 0 to 10 with 0 being the best (Goetz and Fiske, 2013); the BMWP has tolerance rankings from 0 to 10 with 10 being the best (Hawkes, 1998: Junqueira and Campos, 1998). Macroinvertebrate samples are identified to the family level (Mustow, 2002; Pander and Geist, 2013; Paisley et al., 2014) with some studies going further to the genus level for ranking pollution sensitivity (Beauger and Lair, 2008). Once all macroinvertebrate families/genera have been identified, an average stream score is calculated. These scores are categorized into different classes to allow easy comparison between different stream sites. For example, in Junqueira and Campos (1998), five stream classes were defined: class I was for streams with BMWP scores $\geq$86 and were considered to have excellent water quality, class II was for streams with BMWP scores ranging from 64 to 85 and were considered to have good water quality, class III was for streams with BMWP scores ranging from 37 to 63 and were considered to have satisfactory water quality, class IV was for streams with BMWP scores ranging from 17 to 36 and were considered to have bad water quality, and class V was for streams with BMWP scores $\leq$16 and were considered to have very bad water quality. These classes allow watershed managers to relate BMWP scores to water quality, allowing for easier identification of regions that need restoration. However, like the HBI and EPT, the BMWP is based on organism tolerances to organic pollution; and the organisms used are sensitive to more than just organic pollution (Department for International Development, 2004). This can lead to distorted BMWP scores. Furthermore, these organisms may not be naturally present in many regions, so different organisms need to be considered to insure accurate representation of river health conditions (Junqueira and Campos, 1998; Mustow, 2002; Department for International Development, 2004). Studies that applied the BMWP index without making modifications reported that it did not represent stream health accurately (Iliopoulou-Georgudaki et al., 2003). This is expected since the size and location of a stream dictates the number and type of organisms found there according to the river continuum concept. Meanwhile, studies that have modified the BMWP to include local macroinvertebrate families have accurately evaluated stream health (Junqueira and Campos, 1998; Mustow, 2002; Gutiérrez-Fonseca and Lorion, 2014).

6.1.1.3. Abundance Biomass Comparison. Abundance Biomass Comparison (ABC) index was originally introduced by Warwick et al. (1987) and used for evaluating the health of lake ecosystems by comparing macroinvertebrate biomass and macroinvertebrate species abundance k-dominance curves. If the biomass curve lies above the species abundance curve the site in question is unpolluted, if the curves are similar to each other the site is moderately polluted, and if the species abundance curve lies above the biomass curve the site is severely polluted (Warwick, 1986). Further evaluation of this index showed that the ABC was sensitive to many different types of disturbances, such as organic pollution and suspended sediment (Warwick et al., 1987).

6.1.1.4. AZTI Marine Biotic Index. AZTI Marine Biotic index was developed by Borja et al. (2000) to evaluate the health of non-wadeable, coastal regions. It categorizes macroinvertebrate species into one of five ecological groups based on their tolerance to pollutants (Borja et al., 2000). The group definitions are as follows: Group I are species that are very sensitive to organic enrichment and present in unpolluted conditions; Group II are species that are unaffected by organic enrichment; Group III are species that are tolerant to excess organic enrichment; Group IV are species that are common in moderately degraded conditions; and Group V are species that are common in highly degraded conditions (Borja et al., 2000). After sorting the organisms, a weighted biotic coefficient is calculated for each site. The weighted biotic coefficient scores range from 0 to 6 where 0 indicates an undisturbed site and 6 a heavily degraded site (Borja et al., 2000).

6.1.1.5. Number of Macroinvertebrate Families. Number of Macroinvertebrate Families (NFAM) index is a uni-metric index similar to the EPT (Sánchez-Montoya et al., 2010). However unlike the EPT, which uses three stressor sensitive taxa (Compin and Céréghino, 2003), the NFAM index uses the total number of macroinvertebrate families present in the stream to evaluate stream health (Sánchez-Montoya et al., 2010). This index assumes that the number of taxa within an ecosystem increses in healther streams (Wan et al., 2010). Therefore streams with many different macroinvertebrate taxa have higher NFAM scores and are considered less degraded, while sites dominated by few taxa have lower NFAM scores and are considered highly degraded.

6.1.2. Macroinvertebrate-based multi-metric indices

6.1.2.1. Benthic Index of Biotic Integrity. The Benthic Index of Biotic Integrity (B-IBI) is a multi-metric index developed by Kerans and Karr (1994) and is based on the Index of Biotic Integrity (IBI) developed by Karr in 1981. The B-IBI functions similarly to the IBI since it also uses organism communities to evaluate stream health; however, the major change is that the B-IBI considers macroinvertebrates instead of fish. The metrics used in the B-IBI are classified into three categories: taxa richness, taxa composition, and biological processes of the macroinvertebrate community in the aquatic ecosystem (Kerans and Karr, 1994). Kerans and Karr (1994) described these categories as follows: taxa richness metrics are the number of taxa observed within the stream, taxa composition metrics are the percentages of the total population for different taxa, such as % Ephemeroptera, and biological processes metrics describe the percentages of the total population for different functional feeding groups, such as % shredders. This allows for a detailed analysis of the system and its condition. The thirteen metrics included in this index are total taxa richness, intolerant snail and mussel species richness, mayfly richness, caddisfly richness, stonefly richness, relative abundance of *Corbicula*, *oligochaetes*, omnivores, filterers, grazers, and predators, proportion of individuals in two most abundant taxa, and total abundance. Each metric is given a score from 1 to 5 based on the observations of the stream region in comparison to a reference site that had minimal ecosystem degradation (Kerans and Karr, 1994). A higher score indicates that the metric is closer to the reference site. A reference site is defined as the attainable or undisturbed stream conditions for a particular region (Reynoldson et al., 1997; Hawkins et al., 2010). Selection of reference sites has been identified as a key step in the development and application of stream health indices (Whittier et al., 2007). To calculate stream health scores, all of the metric scores are summed. These scores can be used to evaluate the impacts of watershed management scenarios. Based on this analysis, sites that are given lower scores exhibit greater degradation and thus can be selected for restoration projects. For example, the original index scores ranged from 0 to 65 with a score of 65 representing a non-impacted ecosystem and a score of 0 representing a heavily degraded ecosystem (Kerans and Karr, 1994). Kerans and Karr's study (1994) showed that this index was effective at detecting industrial degradations by taking samples above and below industrial effluents. However, a universal B-IBI does not exist and the B-IBI components need to be adjusted for different regions to better describe the ecosystem. This was done in the study by Roy et al. (2003), where the B-IBI was modified to better represent the local condition using 11 metrics instead of the original 13 metrics. Furthermore, despite the fact that B-IBI is a great measure for evaluating stream conditions, its metrics may not clearly represent biological conditions (Henderson, 2014). Therefore, it is important to select metrics that capture local characteristics such as community compositions and land use (Rehn et al., 2008). In addition, the stressor source (natural versus anthropogenic) may not always be identified using B-IBI (Weisberg et al., 1997; Engle and Summers, 1999; Bilkovic et al., 2006).

Table A1 presents the metrics used in the B-IBI as well as the metrics of other indices originated from the B-IBI. Of the original metrics listed, the most commonly removed metrics were % grazers and intolerant snail and mussel species richness; however, no single metric was commonly added. Overall, these changes were made to better represent the local conditions and the ecosystem according to the river continuum concept in which, the number and type of organism varies based on the location and size of the streams.

6.1.2.2. Ephemeroptera, Plecoptera, and Trichoptera Index. The Ephemeroptera (E), Plecoptera (P) and Trichoptera (T) index, also known as the EPT index, is based on the observation of organisms of the Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) families (Lenat, 1988). These families are used because they are particularly sensitive to organic pollution levels within the ecosystem and therefore can be used to identify local impacted regions (Butcher et al., 2003a; Compin and Céréghino, 2003). Their sensitivity to organic pollutants also allows for early identification of problems in the ecosystem and allows subsequent actions to be taken to repair the ecosystem (Johnson et al., 2013). Couceiro et al. (2012) initially used EPT richness and abundance to evaluate stream health conditions within the Central Amazon region of Brazil for distinguishing between degraded and non-degraded sites. However, it was disregarded due to its insensitivity between the sites. In contrast, Oliveira et al. (2011) used EPT as one of the final 9 metrics for their multi-metric index with scores ranging from 0.27 to 65.90 (Oliveira et al., 2011). EPT was also part of the final list of metrics for the benthic community index developed by Butcher et al. (2003a). EPT can also be used as a standalone index. However, in the last two examples, EPT was used as a metric in a multi-metric framework, which can lead to a better understanding of the system and what is affecting it (Butcher et al., 2003a; Oliveira et al., 2011). The macroinvertebrate families used in the EPT are widespread in all streams and regions. However, there are some limitations to using EPT that include: insensitivity in Afrotropic regions due to the low diversity of Plecoptera, which makes it difficult to accurately evaluate stream health; among the EPT families some are tolerant or moderately tolerant to organic pollution, this compromises its utility as a discriminator of organic pollutants in streams (Thorne and Williams, 1997; Masese et al., 2013); and the EPT families are sensitive to other stressors, such as flow regime and stream geomorphology, this can lead to misleading index scores (Meixler and Bain, 1999; Brooks et al., 2002; Mažeika et al., 2004).

To improve applications of the EPT index, it has been modified by including invertebrates from the Coleoptera family. This modified index is known as the EPTC index (Compin and Céréghino, 2003). By adding an additional invertebrate order to the index, the sensitivity of the index to pollution is increased, which helps provide a better view of what is happening in the ecosystem. The EPTC index was used to evaluate conditions in both streams and large rivers (Compin and Céréghino, 2003). The scores from the index we grouped into five different classes, Excellent, Good, Good-fair, Fair, and Poor. The score ranges for each class depended on the type of ecosystem evaluate; for example a score of 50 or more was considered as "Excellent" for streams while for the large rivers, a score of 35 or more was considered as "Excellent". Meanwhile, a EPTC score less than 24 was considered as a poor stream condition, while a EPTC score less than two is poor for the large rivers. Distinction between streams and large rivers in the EPTC method makes it more realistic because the ecosystems found in each generally quite different. However, EPTC is recommended for evaluation of small bodies of water such as streams rather than large bodies of water such as rivers.

Table A1 presents the metrics used in EPT as well as the metrics of other indices that are either based on or use EPT for analysis. Of the original metrics listed, the most common change to the EPT was the removal of the % abundance metric. In the cases when EPT % abundance was removed, additional richness and composition metrics were added, such as Diptera taxa richness, % Coleoptera taxa, and % Oligochaeta and leech taxa (Blocksom and Johnson, 2009). Another common addition to the EPT index was functional feeding group metrics, such as % collector-filterer individuals, predator taxa richness, number of scrapers/number of gatherers, number of shredders/total number collected, and % filterers (Houston et al., 2002; Blocksom and Johnson, 2009). The addition of these metrics increases the index's ability to determine what is occurring within the ecosystem. For example, the addition of the functional feeding group metrics helps determine energy and nutrient flows, while the abundance EPT metrics identify pollution levels within the stream.

6.1.2.3. Multimetric Index for Castilla-La Mancha. Multimetric Index for Castilla-La Mancha (MCLM) index was developed by Navarro-Llácer (2006) for the Castilla-La Mancha region in Spain. The MCLM uses three metrics to describe stream health (Navarro-Llácer et al., 2010) These three metrics include: the average biological monitoring water quality, the number of families from Plecoptera and Trichoptera, and the number of families from Gasteropoda, Oligochaeta, and Diptera (Navarro-Llácer et al., 2010). For each site the individual metric scores are calcualted and averaged to obtain the stream health score. Streams with higher scores are less degraded than those with lower scores (Navarro-Llácer et al., 2010).

6.1.2.4. Yungas Biotic Index. Yungas Biotic Index was developed by Dos Santos et al. (2011) to evaluate wadeable stream heath in the Yungas Rainforest in Southern Bolivia and Northern Argentina. This index determines stream health solely based on the presence of four macroinvertebrate taxa: Elmidae, Plecoptera, Trichoptera, and Megaloptera. Using this system each stream is ranked between 0 and 4, with each value indicating the number of these taxa present at the site (Dos Santos et al., 2011). Therefore a stream site with none of the four taxa will have a score of 0 and will be considered degraded, while a stream with all four taxa present will have a score of 4 and will be considered non-degraded.

6.1.3. Macroinvertebrate-based multivariate indices

6.1.3.1. River Invertebrate Prediction and Classification System. River Invertebrate Prediction and Classification System (RIVPACS) index is a multivariate method that is based on species diversity within stream systems (Moya et al., 2011). Developed in the late 1970s, RIVPACS aimed to relate macroinvertebrate species diversity to physical and chemical features within minimally disturbed streams (Wright et al., 1998). Thirty physical and chemical features were selected and correlated to macroinvertebrate assemblages. After the development of the RIVPACS model, it was used to predict the species and number of organisms that would be expected to appear in a stream system. Comparison of these results with observed macroinvertebrate samples was used to evaluate stream condition.

6.2. Fish-based indices

Another group of organisms that is often used to evaluate stream heath are fish (Mack, 2007; Zhu and Chang, 2008; EPA, 2013; Krause et al., 2013). Karr (1981) listed seven advantages for using fish for evaluating the stream conditions, which included (1) well known life-history, (2) species found in many trophic levels (omnivores, herbivores, insectivores, planktivores, and piscivores), (3) easy identification, (4) understood by general public, (5) can be used to identify a variety of stresses, (6) are present in most water bodies, (7) can be easily connected with regulations. Points 1, 2, 5, and 6 show the usefulness of fish as indicators to determine what is occurring within the ecosystem; while points 3, 4, and 7 show that data collection and presentation is relatively easy when compared to other types of organisms. Unlike macroinvertebrates, fish move throughout entire river systems, which allows for representation of the conditions within an entire water system over a longer period of time (Karr, 1981; EPA, 2013). Another benefit of fish is that they promptly respond to changes in flow regime (Navarro-Llácer et al., 2010), which means that they can be used to evaluate the impacts of flow altering structures, such as dams, on the ecosystem. All of these factors make fish based indices very useful for stream health monitoring (EPA, 2013). Nevertheless, using fish communities for indices has its fair share of limitations as well. Limitations include sampling selectivity, fish seasonal migrations, and the cost of sampling. Table A2 presents 37 fish indices that were reviewed in this study. The first column indicates the name of the index and its reference. The second column indicates the index that it was based on. The third column presents the changes or modifications made from the based index to create the new index. The fourth column describes specific characteristics of the index such as the number of metrics, score trends, or aspect that is evaluated. The fifth column describes the stream size in which the index is applied, with a total of three possibilities: wadeable streams, non-wadeable streams, and wadeable and non-wadeable streams. And the final column lists the metrics used for each index. Out of the 34 fish indices listed in Table A2, 28 were based on the Index of Biological Integrity (IBI). This made IBI, by far, the most often used base index. Of the modifications made to the IBI index, the most common was the addition or subtraction of metrics to provide a better picture of the ecosystems by taking into account local characteristics. An example of this is the Fish Based Index for Lakes (FBIL) developed by Launois et al. (2011). To consider the differences for evaluating a lake in France; three metrics were added: number of planktivore species, total biomass of strict lithophilic individuals, and % total biomass of tolerant individuals. Meanwhile, 10 of the 12 original metrics used in the IBI were removed (Launois et al., 2011). By doing this, the FBIL was able to identify urban and local pressures that were a source of degradation for the French lakes. Of the indices listed in Table A2, few are not based on the IBI, included in this category are the Tolerance Indicator Values Index (TIVI) and the Stressor Gradients Index (SGI). The TIVI was developed by Meador and Carlisle (2007) and functions similarly to the HBI. However, instead of considering organic pollutant tolerances, it looks at the organism tolerances to dissolved oxygen, nitrite plus nitrate, total phosphorus, and water temperature (Meador and Carlisle, 2007). The scores from each river can be used to compare between different rivers as well as indicate the levels of each component identifying where there is too much or too little of each. The SGI was used by Angradi et al. (2009) to correlate stressor gradients, such as total nitrogen, sediment toxicity, and water temperature to stream health. This was unique in the sense that the stressor gradients were correlated to biological metrics in order to determine the condition within the stream. The use of the SGI was able to identify the anthropogenic impacts on the river systems of the Upper Mississippi River basin. The following sections describe the major fish indices into three groups according to the stream health grouping (biotic indices, multi-metric indices, and multivariate methods).

6.2.1. Fish-based biotic indices

6.2.1.1. Fish Response Curves. The Fish Response Curves (FRC) biotic index was developed by Zorn et al., 2012 with the purpose of identifying regions where altered stream flow has adverse impacts on fish communities in Michigan. In this technique, streams were classified based on two parameters: size (streams, small rivers, and large rivers) and temperature (cold, cold-transitional, warm-transitional, and warm) (Zorn et al., 2012). Within each stream, fish species were further classified into "characteristic" and "thriving" based on their abundance. This allows for capturing the variability in fish communities between the different types of streams. In the next step, fish assemblage response curves were developed for different levels of flow alteration within the driest month of the year (Zorn et al., 2012). Once developed the curves could be used to evaluate how much water could be removed from the stream before the fish community was adversely impacted. This technique was adopted by law makers as a guideline for water withdrawal in the state of Michigan (IWR, 2008).

6.2.1.2. Fish Species Biotic Index. Fish Species Biotic Index (FSBI) was developed by Paller et al. (1996) with the purpose of evaluating stream health for the U.S. Department of Energy facility in South Carolina. The FSBI utilizes four species richness metrics including: percentage of expected number of total species, percentage of expected number of native minnow species, percentage of expected number of piscivorous species, and percentage of expected number of madtom and darter species (Paller et al., 1996). Each metric is given a score of 1, 3, or 5 with 1 representing degraded sites and 5 representing non-degraded sites. A weighted average of the individual metric scores was used to determine the overall stream health score for each sampling site (Paller et al., 1996).

6.2.2. Fish-based multi-metric indices

6.2.2.1. Index of Biotic Integrity. The Index of Biotic Integrity (IBI) is a multi-metric index introduced by Karr in 1981. It is based on fish communities and widely used to determine the overall stream health (Karr, 1981). Karr listed three assumptions that are needed for the use of this index; (1) the fish sample is a balanced representation of the community at the site, (2) the chosen site is representative of the region in which the IBI is being applied, and (3) the personnel charged with analysis of the collected data are trained (Karr, 1981). If any of these assumptions is violated, the results of this index can be misleading. Originally, the IBI was composed of 12 metrics, which can be grouped in one of the three following classifications; (1) species richness and composition, (2) trophic composition, and (3) fish abundance and condition (Karr, 1981; Hu et al., 2007). Each of these metrics is given a score of 1, 3, or 5 based on undisturbed reference sites, or sites with as little human disturbance as possible (Stoddard et al., 2006; Whittier et al., 2007), where a score of 5 is the best. After scoring all the metrics, the individual scores are summed to provide the IBI score for each site. The IBI scores ranged from 0 to 60 and were grouped into 9 stream classes, Excellent, Excellent-Good, Good, Good-Fair, Fair, Fair-Poor, Poor, Poor-Very Poor, and Very Poor. Under this class system a stream scoring a 23 or less would be classified as Very Poor while scores of 57–60 would be considered Excellent. Even though the 9 stream classes are applicable in different regions, caution should be taken when correlating IBI scores from different regions. In order to address this issue, Karr (1981) also provided a description of what should generally be found in each stream class. This makes it easier to modify the IBI so it can be more transferable for multiregional studies of stream health. The IBI has been applied and modified in a variety of studies (Zhu and Chang, 2008; Smith and Sklarew, 2012; Krause et al., 2013). In Europe, a commonly used index of stream health based on the IBI is the Fish-Based Index (FBI) (Launois et al., 2011). In Launois et al.'s application of the FBI, 15 metrics were used with scores ranging from 0 to 100 with 100 being the best. The FBI was able to successfully identify degraded water bodies, but lacked the ability to identify individual stressors (Launois et al., 2011). This shows that the selection of metrics is vital to ensure that the expected regional characteristics and stresses are represented (Ruaro and Gubiani, 2013).

Recently, Lyons (2012) modified the IBI for use in perennial coolwater streams in Wisconsin. This required the creation of two different IBIs the Cool-Cold Transition (CCT) IBI and the Cool-Warm Transition (CWT) IBI. Each index uses five metrics to represent the ecosystems (Table A2) (Lyons, 2012). The metrics are given a score of 0, 10, or 20 based on the analysis of the sample. Next, the metric scores are summed to calculate the IBI score giving a range of scores from 0 to 100 with 100 being the best similar to the FBI (Lyons, 2012). Overall, the results showed that while both indices identified disturbed areas with low scores; the CWT index performed better than the CCT index. However, due to the wide variation in scores for similar stream sites, it was recommended that multiple samples and a mean or median score should be used to classify the systems instead of a single sample (Lyons, 2012).

A different study that utilized the IBI found that rare taxa had major impacts on the results of IBI scores (Wan et al., 2010). In Wan et al. (2010) the sensitivity of the IBI was tested and it found that the presence/removal of rare taxa, often considered an indicator of lower degradation, can lower the IBI score by 38 points. While this was a concern, this result of the study still shows that the IBI is sensitive to the conditions within the stream, and as long as the metrics are weighted correctly, the results of the index can provide accurate information about stream degradation. However, seasonal migration of fish communities can lead to incomplete community sampling which in turn leads to misleading IBI results especially at a large scale (Zalewski, 1983; Schlosser, 1990; Roset et al., 2007). In addition, using IBI may not always help in determining source of stressors (natural or anthropogenic) even though it provides overall stream health condition.

Table A2 presents the metrics used in IBI as well the metrics used in other indices that are either based on or use IBI for analysis. Of the metrics listed in the table, the most common change to the IBI was the removal of most of the original metrics such as the species richness and composition of darters, suckers, and sunfish (except green sunfish), and the proportion of green sunfish (Karr, 1981). This was done in combination with the addition of other metrics to represented local characteristics. For example, number of coolwater species, percentage tolerant species, % invertivore/piscivore individuals, and % native large river taxa (Kanno et al., 2010; Esselman et al., 2013). This also follows the river continuum concept in which, the number and type of organism varies based on the location and size of the streams. By modifying the IBI to such an extent allows for better understanding of what is occurring within the ecosystems by taking into account local characteristics.

6.2.2.2. Estuarine Multi-metric Fish Index. Estuarine Multi-metric Fish Index (EMFI) was developed by Harrison and Kelly (2013) with the purpose of evaluating Irish transitional waters. To capture the characteristics of transitional waters the EMFI uses fourteen metrics: species richness, number of introduced species, species composition, species abundance, dominance, number of daidromous species, estuarine species richness, marine migrant species richness, estuarine species abundance, marine migrant species abundance, zoobenthivore species richness, piscivore species richness, zoobenthivore abundance, and piscivore abundance (Harrison and Kelly, 2013). Each of these metrics is given a score from 1 to 5 with 1 representing degraded conditions and 5 representing non-degraded conditions. After individual metric scores are calculated, they are summed to provide site health scores, which can be used to compare between sites (Harrison and Kelly, 2013).

6.2.2.3. Fish Community Index. Fish Community Index (FCI) was developed by Jordan et al. (2010) with the purpose of evaluating estuarine environments within the Gulf of Mexico. The conditions that FCI was developed for are similar to those for the EMFI. However, the FCI only uses three metrics (Jordan et al., 2010) compared to the fourteen used for the EMFI (Harrison and Kelly, 2013). The metrics used for the FCI include: number of species, species abundance, and trophic index (Jordan et al., 2010). Each metric is given a score of 0, 1, or 2 with 0 representing degraded sites and 2 representing non-degraded sites (Jordan et al., 2010). The individual metric scores are summed to provide the health score for each site (Jordan et al., 2010). These scores were not only used to compare between sites but also between years.

6.2.2.4. Similarity Indices. Similarity Indices (SI) were developed by Navarro-Llácer et al. (2010) to evaluate stream health by relating conditions within stream sites to established reference sites. Four different metrics of the fish community (composition, relative abundance, age structure, and a global similarity value) are used to compare stream conditions (Navarro-Llácer et al., 2010). Each of these metrics is given a score from 0 to 1 with 1 representing the reference conditions (Navarro-Llácer et al., 2010). These scores allow for rapid comparison of sites and the identification of heavily degraded regions.

6.2.3. Fish-based multivariate indices

6.2.3.1. Stressor Gradients. Stressor Gradients index was developed by Angradi et al. (2009) to assess stream health by relating stressor metrics to fish communities. A variety of stressors including total nitrogen, turbidity, human disturbance, distance to upriver dam, and percent riparian wetland were used (Angradi et al., 2009). These metrics were related to a variety of fish assemblage metrics, such as number of minnow species, total number of fish species, and proportion of invertivore individuals (Angradi et al., 2009). Once these relationships were determined, stressor metrics were given a score from 0 to 1 for each site, where 1 represented non-degraded regions (Angradi et al., 2009). By using the relationships between fish communities and stressors, this index can be used to evaluate stream health in regions where fish communities have not been sampled.

7. Conclusions and recommendations

Throughout this review a variety of macroinvertebrate and fish indices were discussed, each had benefits and limitations. For macroinvertebrate indices, the B-IBI was capable of identifying industrial and chemical degradation (Kerans and Karr, 1994) as well as changes brought about by land use change such as urbanization (Roy et al., 2003). However, these indices are site specific (Kerans and Karr, 1994), which means that to insure accurate evaluation of stream health the metrics need to be fitted to the conditions of the site. The HBI, NBI, and ISI were all able to determine organism tolerances to pollutants whether organic (HBI) (Goetz and Fiske, 2013) or nutrient (NBI, ISI) (Smith et al., 2007; Haase and Nolte, 2008). The HBI also has the benefit that it can be used as a metric of other multi-metric indices (Butcher et al., 2003a) allowing for better understanding of the ecosystems. Yet again, these indices may not be applicable to other regions (Haase and Nolte, 2008) because the tolerances of species may change based on the natural conditions within different habitats. The EPT index is capable of detecting low levels of degradation due to the sensitivity of the Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) families (Goetz and Fiske, 2013). And similar to the HBI, the EPT index can all be included in other multi-metric indices (Butcher et al., 2003a). However, if the diversity of these families is low it can lead to misleading index scores of stream health (Couceiro et al., 2012). In terms of fish indices, the most commonly used and modified index is the IBI. This index allows for the evaluation of entire regions (Karr, 1981) while at the same time being easily modified to take into account different climates (Lyons, 2012). However, the selection of the metrics used in this index is vital for interpretation of the results (Wan et al., 2010; Launois et al., 2011).

7.1. Benefits

There are many reasons that a macroinvertebrate or fish index would be applied to a river system; whether it is to indicate the presence of pollutants (Karr, 1981; Johnson et al., 2013), or determine the optimal nutrient load for the system (Smith et al., 2007), or compare levels of degradation between streams (Karr, 1981; Kerans and Karr, 1994). Furthermore, some macroinvertebrates are sensitive to very low levels of degradation at local levels; therefore they can be used by stakeholders to detect and correct problems before more serious damage occurs (Barbour et al., 1999; Flinders et al., 2008). While fish indices can be used to evaluate the conditions on a regional scale, due to their mobility and lifespans (Karr, 1981). This makes them useful for watershed managers, since they can be used to identify problems found throughout the entire watershed. Another benefit to using macroinvertebrate and fish indicators, is that they are also sensitive to the development of storage structures such as dams (Navarro-Llácer et al., 2010; Marzin et al., 2012) and can be used to monitor the impact of anthropogenic changes to the flow levels in the rivers. Besides being able to be used for a variety of different stream health indices, macroinvertebrates and fish can also be used to identify the stressors causing the degradation of a site, based on the number and type of sensitive taxa present. And the wide distribution of macroinvertebrates and fish over trophic levels allows for a better understanding of what is actually happening within the system and what changes are occurring due to anthropogenic impacts. When all of this is taken into account, macroinvertebrates and fish can be seen as a very versatile indicator of stream health and the impacts humans have on the aquatic ecosystems for which they rely on for drinking water and irrigation.

In regard to the benefits of specific indices; the IBI is a comprehensive index and due to its multi-metric nature it can be used to capture broad characteristics within streams that is beneficial in regional studies. Like the IBI, B-IBI is a multi-metric index, which provides a comprehensive overview of stream condition at local levels. This index can also be modified to be sensitive to individual pollutants such as industrial effluent. The HBI and BMWP use macroinvertebrate tolerances of organic pollution to evaluate stream health. The wide distribution of ranked organisms allows this index to be applied in many locations with minimal modification. Additionally, the use of organism tolerances has been expanded to include other stressors such as nutrients and temperature. Another index that is sensitive to organic pollution is the EPT, which is composed of a group of organisms that is commonly present in streams. Therefore, the EPT is often added as a metric for multi-metric indices regardless of the location.

7.2. Limitations and future research

Macroinvertebrates and fish are useful indicators of stream health (Karr, 1981; Iliopoulou-Georgudaki et al., 2003) and a number of studies have used them to evaluate large regions (Whittier et al., 2007; Paulsen et al., 2008; Stoddard et al., 2008; Marzin et al., 2012). These regions can be as large as entire countries. For example Marzin et al. (2012) evaluated stream health for all of France; while Paulsen et al. (2008) performed a nationwide analysis on the first national assessment of the United States. Evaluating stream health on this scale allows for the comparison of scores between many different locations. However, some level of inaccuracy is expected on regional use of biological indicators due to ecological and physiographical diversity (Hering et al., 2010). This is more pronounced for fish than macroinvertebrate indices, such as the IBI. To reduce this inaccuracy, ecoregions are commonly used for regional studies (Whittier et al., 2007; Paulsen et al., 2008), this is due to the fact that ecoregions are relatively uniform in terms of biotic and abiotic characteristics (Butcher et al., 2003a).

The riverine macroinvertebrates and fish communities have been characterized by seasonal dynamics. However, seasonal fish migrations along the river continuum seriously affect community structure, both upstream and downstream (Roset et al., 2007). Those effects may be further biased by electrofishing efficiency, which in one run collects only certain fractions of the community. Additionally, this is being affected by size distribution of community (smaller fish are less efficiently collected) and for the same size of fish body shape (long and slender fish are more efficiently collected than wide-bodied). Thus, to eliminate those biases during sampling procedure the mathematical formula was elaborated toward assessment of efficiency of electrofishing on the basis of only one electrofishing run (Zalewski, 1983).

In regard to limitations of specific indices; while the IBI can capture broad characteristics within streams, its multimetric nature can make it difficult to determine the origin of the stressors (natural versus anthropogenic). Similar to the IBI, the B-IBI may be unable to identify the stressor source. The HBI, BMWP, and EPT are sensitive to organic pollution for stream health evaluation. However, the organisms used for these indices are also sensitive to other stressors. This can lead to the misidentification of the stressor impacting the system. Additionally, the organisms used for these indices may not naturally occur in different regions, this prevents the indices from accurately describing the system.

Overall, determining which index to apply to a region is challenging. Biotic indices (HBI, BMWP and EPT) while effective at determining the stream health based on a specific stressor, such as organic pollution, are insensitive to other stressors that can impact the system. Multi-metric indices (IBI and B-IBI) help solve this problem by looking at several different metrics and allowing for a wider understanding of what is occurring within the stream. However, these systems are still limited by sampling technique efficiency. In general, this can be mitigated by increasing the number of samples taken from each site in the study, but it still needs to be noted that incomplete community samples limit the usefulness of stream health indices.

Throughout this review, different aspects and applications of macroinvertebrate and fish indices have been discussed. The majority of these works were performed in wadeable streams, describing how the ecosystem responds to different stressors. However, fewer studies have been done for non-wadeable streams, which should be the focus of future research.

Conflict of interest

None declared.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecohyd.2015.04.001.

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Further reading[1]

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[1] These references can be found in the supplementary material.