Org Divers Evol (2012) 12:251–259
DOI 10.1007/s13127-012-0104-4

REVIEW

ORGANISMS
DIVERSITY &
EVOLUTION

Using dragonflies to monitor and prioritize lotic systems: a South African perspective

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Received: 30 October 2011 / Accepted: 16 February 2012 / Published online: 17 August 2012

Abstract The ever-worsening condition of streams due to local, regional, and global demands on water has resulted in the development of increasingly streamlined, rapid assessment methods using macroinvertebrates. Biotic indices in particular are versatile and robust, although not always easy to use. For example, the family-level South African Scoring System is an effective water quality measure, but is time-consuming and requires high-level expert training. The index could be used alongside the species-level Dragonfly Biotic Index (DBI), originally developed for monitoring habitat integrity, with which it is significantly and strongly correlated. We review here the relevant biotic indices in stream biomonitoring and their advantages and disadvantages, and present a new extension of the DBI, the Habitat Condition Scale (HCS). The HCS enables comparison and ranking of sites in terms of their habitat condition. Indeed, the DBI is a very flexible index, having been used in site selection and prioritization for conservation, as well as the measurement of habitat recovery. The theoretical framework behind the index demonstrates the potential of the index to track biotic changes due to climate change. The index could also be easily adapted for use in other biogeographical regions, given that species distributions, threat levels and sensitivities are well-known, and that there is an adequate number of endemic species. However, like all benthic macroinvertebrate indices, the DBI cannot always identify exactly which in-water impacts have an effect and to what extent. The real power of the DBI lies in being able to quantify community response to known physical changes on the riverscape and across the region.

Keywords Mike May Festschrift · Aquatic insects · Freshwater · Conservation · Biomonitoring · Odonata · Dragonflies · Dragonfly Biotic Index

This is a contribution to the Festschrift for Michael L. May

Electronic supplementary material The online version of this article (doi:10.1007/s13127-012-0104-4) contains supplementary material, which is available to authorized users.

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Introduction

River ecosystems are the most threatened ecosystems of all (Abell 2002). Indeed, declines in biodiversity are estimated to be up to five times greater in some rivers than in the most degraded terrestrial ecosystems (Dudgeon et al. 2006). The worst impacts on rivers include the introduction of alien organisms, dam construction, habitat modification, chemical pollution, and resource over-extraction (Malmqvist and Rundle 2002; Nel et al. 2007). The extent and magnitude of these pressures, and the limits on time, funding, and personnel, call for rapid and reliable assessment methods that rely on surrogate approaches as a key tool for conservationists (Kati et al. 2004; Lawler et al. 2003). Basic requirements when using invertebrates for monitoring a system include the existence of good taxonomic and biological information, knowledge of species conservation status, and responses to habitat quality (McGeoch et al. 2011).

Indeed, a good biological indicator: (i) readily reflects the state of an environment, (ii) represents the impact of environmental change at a variety of scales, or (iii) is a useful surrogate or umbrella of other taxa (McGeoch 1998). Bioindicators can be used for measuring any of the three indicator categories: biological diversity, environmental health, and ecological condition. The function of biodiversity indicators is for use in the estimation of the diversity of taxa in a particular area and to monitor changes in that diversity. Environmental indicators are used to detect and monitor changes in environmental states, which can be readily measured using abiotic characteristics of the environment, such as pH or conductivity. Ecological indicators demonstrate the impact of a stressor, whether environmental or ecological, on biota and monitor the long-term stressor-induced changes in biota (e.g., habitat disturbance, climate change).

Against this background, we review the recent advances in monitoring and prioritizing riverine habitats for conservation purposes, using a selection of biotic indices that have gained popularity in recent years. We focus here on South African river systems which, in many areas, are highly threatened by invasive alien plants and high water demands. Furthermore, many of these rivers, especially in the Cape Floristic Region, are rich in localized endemics, making the conservation of ecological integrity of major significance (Wishart and Day 2002).

Comparing groups and assemblages

The measurement of the number and composition of species is important in monitoring for change or assessing conservation value. This may range from simple species counts and measures of taxonomic distinctness to the use of biotic indices. The latter may use a comprehensive set of variables that include species biodiversity metrics (e.g., geographic distribution, rarity, functional traits).

Species richness

Species richness is a commonly used biodiversity measure (Jennings et al. 2008). However, species counts cause significant biases in that they are (i) dependent on sampling effort, (ii) do not directly reflect phylogenetic diversity, (iii) cannot be compared against an absolute standard, (iv) may actually increase under moderate levels of disturbance (Wilkinson 1999), (v) will differ with different habitat types (Warwick and Clarke 2001), and (vi) when conditions change species may change identities, yet species richness can remain the same.

Taxonomic distinctness

Taxonomic distinctness relies on the hypothesis that disturbed assemblages are composed of more closely related species than unperturbed assemblages (Warwick and Clarke 2001). Because the most taxonomically varied assemblage will be the more diverse, taxonomic distinctness will be higher and indicate the more natural condition. However, in freshwater studies on aquatic beetles (Abellán et al. 2006), fish (Bhat and Magurran 2006), and stream macroinvertebrates (Heino et al. 2007; Simaika and Samways 2011) taxonomic distinctness do not show any response to habitat degradation. Abellán et al. (2006) suggest, therefore, that disturbed assemblages may not always be composed of more closely related species than natural assemblages. Indeed, species that are representative of the more species-rich taxa may disappear, causing taxonomic distinctness to increase. However, a study testing a dataset of aquatic insects along a water quality gradient found an inverse linear relationship of taxonomic distinctness to water quality (Marchant 2007). The author attributes the success of the method using a large dataset with a high number of species or higher taxonomic level identifications, as well as robustness of the method to longitudinal changes in species composition.

Biotic indices

Biotic indices are perhaps the most reliable and adaptable measures, and a large number have been developed for freshwater habitats across the world using different combinations of criteria (Boon and Pringle 2009). Biotic indices are typically based on the identification of invertebrates to the lowest taxonomic level that can be achieved in a reasonable amount of time (Morse et al. 2007). The indices rely on specific weightings of certain criteria at a chosen taxonomic level (e.g., family, genus, or species), for example sensitivity or tolerance to water pollution (water quality) or habitat disturbance (ecological integrity). Some biotic indices can be composed of several sub-indices, as does the Dragonfly Biotic Index (DBI) (see below), which may include a measure of geographic distribution (e.g., endemism), threat [International Union for Conservation of Nature and Natural Resources (IUCN) Red List categories; IUCN 2008] and sensitivity to pollution or disturbance (Simaika and Samways 2009a). Indeed, most scoring methods are based on species rarity, using criteria such as the IUCN Red List categories. For example, a Species Quality Score (SQS) based on the Red List has been developed by Foster et al. (1989) using water beetles to determine the conservation value of streams in the UK. Another index of note that uses the Red List is the Community Conservation Index (CCI), also in use in the UK, and which assesses the conservation value of macroinvertebrate communities (Chadd and Extence 2004). Scoring methods that do not make use of the Red Lists have also been developed, and may use a combination of local rarity, regional responsibility, and habitat vulnerability measures (Gauthier et al. 2010), or the number of source populations in an area (Angermeier and Winston 1997).

Invertebrates as bioindicators

Algae, fish, and macroinvertebrates are especially sensitive to changes in water quality and are, therefore, most commonly used in investigations of water quality or ecological integrity (Morse et al. 2007). However, algae can be more difficult to identify than macroinvertebrates and they bloom and die more rapidly in response to nutrient inputs, so the evidence of random pollution events may not be detected by periodic monitoring. Fish have relatively low numbers of species and low densities in streams, making them less useful for statistical monitoring. In addition, their greater mobility allows them to escape pollution events by swimming into unaffected areas. Furthermore, fish are not easily sampled in rapid assessments and protection laws make it difficult to sample them in many areas.

The macroinvertebrates are, thus, the most useful as bioindicators. Biotic indices often comprise only the Plecoptera, Ephemeroptera, and Trichoptera, but can include many more taxa. On occasion, single taxa may be used (see McGeoch 2007 and references therein). Interestingly, chironomids have been used both as indicators of water quality (i.e. pollution) and habitat quality (disturbance) (McGeoch 2007). The DBI (see below) was originally developed as an index for measuring habitat integrity and may also have potential as a water quality measure (Simaika and Samways 2011).

Biotic indices for measuring water quality and ecological integrity

Water quality

There is a variety of community-based water quality indices, which have been implemented in many countries, such as the Australian River Assessment System (Smith et al. 1999), the River Invertebrate Prediction Classification System in the UK (Wright et al. 2000), and the South Africa Scoring System (SASS) (Dallas and Day 1993) (see below). Rosenberg and Resh (1993) discuss the various earlier advances in the field, with special reference to North America.

The SASS involves sampling of aquatic macroinvertebrates in riffles, glides, and deposition zones. Scores are assigned to each taxon at the family (or higher) level according to the taxon's sensitivity or tolerance to disturbance or pollution, based on pre-determined research findings (Dallas 2000; Revenga et al. 2005). High sensitivity scores are allocated to the most sensitive benthic macroinvertebrate taxa and the lowest scores to those which are most tolerant. The sum of the individual scores is the macroinvertebrate (SASS Version 5) score, which gives a preliminary index of water condition. However, the Average Score Per Taxon (ASPT) is the most standardized measure and is calculated by dividing the macroinvertebrate score by the number of sampled taxa. The SASS5 scores and ASPT scores are then compared against ecological categories for SASS5 (Dallas 2007; Dallas and Day 2007) (see below). This final classification aids in the interpretation of the water quality of a particular freshwater system.

Ecological integrity

A variety of ecological integrity indices exist. Noteworthy ones include the European Odonata Habitat Index (OHI), and the South African DBI. What distinguishes these indices from the majority of water quality measures is that they are based on weighting representatives of a single Order, rather than a variety of them. Chovanec and Waringer's (2001) OHI is a weighted measure based on habitat type (i.e., the spread of species in different habitat types), abundance, and indication (i.e., weighted specificity to identify sensitivity of species). They also have the advantage that they operate at the more sensitive species level rather than at the more gross higher taxonomic level.

The DBI provides a measure of ecological integrity for both lotic and lentic freshwater systems (Simaika and Samways 2009a). The index is a weighted measure based on the quantitative assessment of three sub-indices of species distribution, threat status (i.e., Red List assessment), and sensitivity to disturbance. The total DBI of a wetland (stream, river, or pool) reflects the total dragonfly assemblage, thus enabling water bodies to be compared and restoration success to be monitored (Magoba and Samways 2010; Samways and Sharratt 2010; Simaika and Samways 2008).

Initially developed for aquatic biomonitoring, the DBI has been applied successfully to measure habitat recovery, and to select and prioritize sites for conservation. Previous work has shown a strong correlation between adult dragonfly scores and macroinvertebrate scores (Smith et al. 2007). An advantage of the DBI over conventional macroinvertebrate indices is that the DBI's operation at the level of identified species level means that it is highly sensitive to habitat condition, as species are lost and, sometimes, gained with changing conditions. Indeed, at the local scale, the DBI outperforms the SASS and in site selection gives better results than algorithm-based methods (Simaika and Samways 2011, 2009b). Therefore, it has good potential for environmental assessment and monitoring freshwater biodiversity and quality alongside the SASS. The relatively low field effort (compared to SASS) required to obtain a DBI score for a site, makes this a low-cost and readily-applied method. However, it is not exclusive of SASS as the DBI/Site value correlates strongly with the ASPT of the SASS method.

Given the variation in rivers within South Africa, it is also important that variation among regions, both geographically and longitudinally, be taken into account when interpreting aquatic data (Dallas 2004). Therefore, we present here the newly developed Habitat Condition Scale (HCS) as an interpretation guideline for using the DBI scheme. Based on the same principle as the SASS system's ecological categories (Dallas and Day 2007), the HCS is a sensitive measure of the condition of a stream habitat (Table 1). The HCS incorporates percent canopy cover (shade) with biotope diversity (i.e., structural condition of the habitat). In South Africa, the presence of invasive alien trees reduces the diversity and natural value of biotopes and amount of solar energy needed for dragonflies, particularly the endemic species, to go about their daily activities (Magoba and Samways 2010; Remsburg et al. 2008; Samways and Taylor 2004). The effect is the local extirpation of endemics and other species at the point localities where this key threat is great. Here we apply the HCS and also compare it to the SASS5 ecological categories on a dataset of 20 river sites of Tsitsikamma, South Africa (Table 2). Information on the exact site locations, collection times, and methods used in collating the dataset is presented in detail in Simaika and Samways (2011).

The HCS can be used in conjunction with the DBI/Site which is the total DBI score of a site divided by the total number of dragonfly species at that site. To maximize the comparability with the SASS5 approach of assigning ecological categories, the same method as in Dallas (2007) was applied. Dallas (2007) derived the highest SASS5 ecological category (A) by calculating the 90th percentile so that this category represented the top 10% of sites. The remaining categories (B–E/F) occurred with equal 22.5% differences between categories using the 67.5th, 45th, 36th, and 22.5th percentile. The SASS5 score was then plotted as a function of the ASPT score (Fig. 1) and the DBI as a function of the DBI/Site score (Fig. 2).

According to the SASS5 ecological categories, the 20 sites fall into all five categories of ecological condition (Table 1). The majority are B (11 sites), then C (4 sites), D and E/F (2 sites respectively), and A (1 site). Similarly, using the HCS, the majority of sites (eight) fall into the second highest category HM, five into category HH, four into MM, two into LL, and one into ML (see Table 2 for category abbreviations). It is noteworthy that the MAT site, and the related BUF(L) and BUF(U) sites are genuinely biologically impoverished, as is reflected by the SASS5 and ASPT scores (Table 2). The riparian area of these sites is natural, but the sites are well-shaded (50–70% shade). However, the STR(U) site, which is 90% shaded scores high using the SASS and ASPT. Only two dragonfly species are present at the STR(U) site and, because these are endemics, the DBI/Site is unusually high (7.00 score). The Upper Storms river is a fully natural site. The riparian area is natural and pristine, as are the aquatic conditions, as the SASS5 and ASPT scores demonstrate (Table 2). However, the high percent shade reduces the biotope quality and solar energy input, and therefore also significantly reduces the diversity of Odonata species at the site. When using either of the biotic indices in interpreting site values, it is therefore important to consider the coverage of the riparian zone in decision making. Nevertheless, we recognize that this HCS, as it stands, applies only to South African conditions where shade from trees, especially alien trees, is a highly threatening process. Elsewhere, particularly in the tropics, shade from forest trees is a required feature for many species (Cordero Rivera 2006) and in some areas, such as Mayotte, alien trees provide the required conditions, even for localized endemics (Samways 2003). Thus, we visualize the HCS being adapted and tailored according to the specific threats in a region.

Table 1 Ecological categories for use with the South African Scoring System (SASS) [combined category values as modeled by Dallas (Dallas 2007) for the south-eastern coastal belt ecoregion of South Africa] and the Habitat Condition Scale for use with the Dragonfly Biotic Index [scale modeled after methods used by Dallas (2007), based on 29 reference sites in the ecoregion]

SASS5 ecological category			
SASS5	ASPT	Abbrev.	Description
>226	>8.1	A	Unimpaired; high taxa diversity with numerous sensitive taxa
191–226	7.5–8.1	B	Slightly impaired; high taxa diversity but with fewer sensitive taxa
156–190	7.0–7.4	C	Moderately impaired; moderate diversity of taxa
121–155	6.2–6.9	D	Considerably impaired; most tolerant taxa present
<121	Variable	E/F	Severely/critically impaired; only tolerant/few tolerant taxa present
Habitat condition scale			
DBI	DBI/Site	Abbrev.	Description
>46.5	>5.4	HH	High biotope diversity; 0–25% canopy cover
37.9–46.5	4.4–5.4	MH	Moderate to high biotope diversity; 25–75% canopy cover
29.6–37.8	3.6–4.3	MM	Moderate biotope diversity; 25–75% canopy cover
26.1–29.5	2.8–3.5	ML	Moderate-to-low biotope diversity; 75–100% closed habitat
<26.0	Variable	LL	Low biotope diversity; typically 100% closed canopy

SASS5 cumulative sensitivity score of macrobenthic invertebrate families, *ASPT* Average Score Per Taxon, *Abbrev.* Abbreviation, *DBI (Dragonfly Biotic Index)* total DBI of dragonfly species, *DBI/Site* Dragonfly Biotic Index score averaged per sampling site (see text for calculation)

Table 2 Interpretation of habitat and water quality based on the Habitat Condition Scale and the SASS ecological categories as modeled for the south-eastern coastal belt of South Africa (Dallas 2007). Site: see names, co-ordinates and elevations in Supplementary Table S1

Sites	SASS5	ASPT	Water	DBI	DBI/Site	Habitat
BLU(L)	179	7.675	B	44	4.4	MH
BLU(U)	186.75	7.4875	C	55.99	5.09	HH
BOB	156.75	7.735	B	40	4	MH
BUF(L)	158.5	6.4075	C	24.01	3.43	ML
BUF(U)	127.5	6.15	D	22	2.75	LL
ELL(L)	192.5	7.635	B	28.98	4.83	MH
ELL(U)	156	7.525	B	52.95	3.53	HH
ELW(L)	96.75	5.6	E/F	30	3.75	MM
ELW(U)	138.5	7.6675	B	22.02	3.67	MM
GRE(L)	159.25	7.05	C	33.02	2.54	MM
GRE(U)	181	7.625	B	40	5	MH
GRW(L)	147	7.125	C	44.94	3.21	MH
GRW(U)	169.75	7.5875	B	37.98	6.33	HH
LOT(L)	202	8.275	A	31.02	5.17	MH
LOT(U)	186	7.8125	B	35	5	MH
MAT	114	6.05	E/F	12	3	LL
SLT(L)	149.5	6.825	D	38	4.75	MH
SLT(U)	198.25	7.8	B	37	3.7	MM
STR(L)	190	7.62	B	54	3.6	HH
STR(U)	136.25	7.675	B	14	7	HH

SASS5 South African Scoring System, *ASPT*, Average Score Per Taxon, *Habitat*, Habitat Condition Scale abbreviations from Table 1; *Water* SASS5 ecological categories abbreviations from Table 1, *DBI (Dragonfly Biotic Index)* total DBI of dragonfly species, *DBI* Dragonfly Biotic Index, *DBI/SITE* Dragonfly Biotic Index per site (see text for calculation)

Beyond aquatic biomonitoring

Bioindication methods may also have the additional potential for application as measures for habitat recovery in prioritizing sites for conservation or for monitoring climate change. As an example, the DBI has already been applied to all but the last of these cases, where application of the index has potential.

Measuring habitat recovery

In South Africa, there is an ongoing and massive project restoring rivers known as the Working for Water Programme (Richardson and van Wilgen 2004). The primary aim of the project is to remove alien invasive trees, thereby hydrologically restoring rivers while also creating jobs for local people. Although not a primary aim of the restoration project, endemic riverine biodiversity has benefitted enormously from this restoration activity (Samways et al. 2011). We show in Simaika and Samways (2008) that restoration success can be measured easily by calculating the ratio of the sum of the DBI scores pre- and post-clearance of alien trees. If, for example, the sum of the DBIs after clearance is 50, yet before it was 25, then the biotic recovery is 2. Converted into a percentage Biodiversity Recovery Score (BRS), this is 200%. The great advantage of this BRS is that streams which had previously lost their narrow-range and sensitive specialists and have now been restored as a result of removal of the alien trees have high scores (Simaika and Samways 2008). We demonstrate high values of biodiversity recovery from three examples in the Western Cape Province, site of the Cape Floristic Region biodiversity hotspot: a massive BRS of 464% for Disa Stream on Table Mountain, a BRS of 379% for DuToit's River at Franschhoek Pass (379%), and a 370% BRS for the White River in Bainskloof Pass.

Prioritizing sites for conservation

By using species identity information, the DBI incorporates important biodiversity information, which is not only useful in measuring the ecological integrity of a site, but also its conservation value. Indeed, Simaika and Samways (2009b) used the DBI for selection and prioritization of sites of conservation value on a national scale in South Africa. We tested the DBI against reserve selection algorithms (rarity–complementarity). We found that in the south west Cape, only four catchments were selected by the algorithms, while the DBI selected 12 catchments for the same region. Thus, using the algorithms, few areas were selected for Red Listed species, affecting their viability. Furthermore, we found that of the globally Red Listed fauna in South Africa, the algorithm represented all 22 species, while the DBI only selected 16. Although the DBI may appear inefficient at first, from a conservation perspective adequate representation of globally Red Listed species must take precedence over nationally Red Listed species, these over other nationally Red Listed endemics and non-threatened endemics, and so on, until one

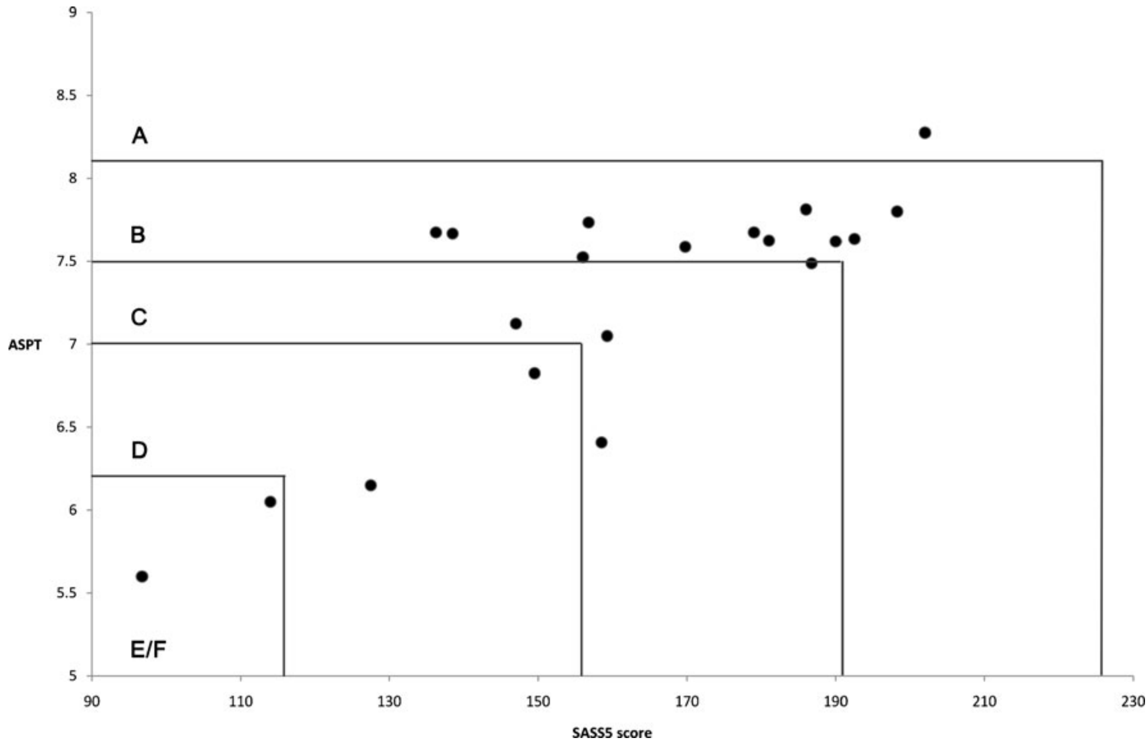

Fig. 1 Plot of South Africa Scoring System (SASS)5 scores as a function of Average Score Per Taxon (ASPT). Ecological categories (A–E/F) are explained in Table 1, and SASS5 and ASPT scores are given in Table 2

reaches the most common and geographically most represented species (Simaika and Samways 2009b). The DBI is designed to target rare, endemic, or threatened (i.e., Red Listed) taxa, and/or species that are sensitive to habitat disturbance. Red Listed

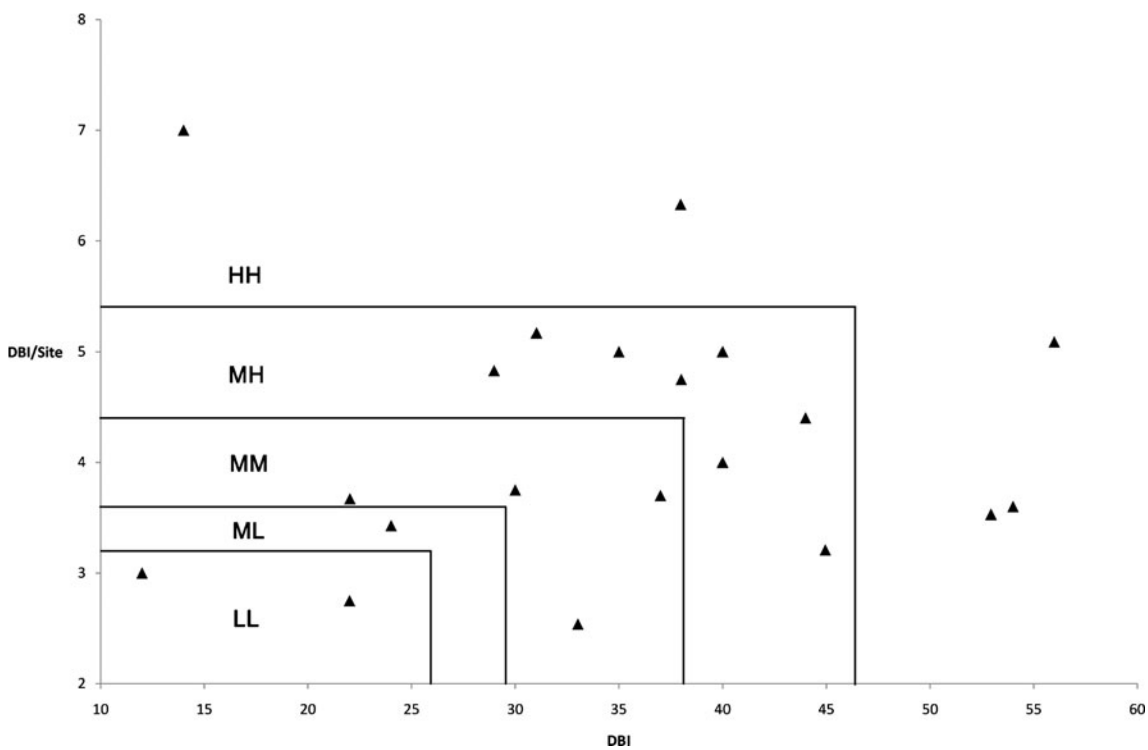

Fig. 2 Plot of Dragonfly Biotic Index (DBI) scores as a function of DBI/Site. Habitat condition categories (HH–LL) are explained in Table 1; DBI and DBI/Site scores are given in Table 2.

species are thus given conservation priority. This is in line with IUCN policy on species conservation and explains why species that fall in such categories are Red Listed, while common, widespread species are not. Of course, these also have conservation value, but the idea of using the DBI as opposed to an algorithm is that the DBI gives conservationists a picture not only of the content, but also the value of the site for conservation.

Monitoring for climate change

The DBI also has value in monitoring for climate change, whether in terms of rising temperature or in terms of changing precipitation regime. There are, however, some essential qualities of the dragonfly assemblage that first need to be considered. The dragonflies of South Africa can be divided into three groups based on their biogeography relative to the physiognomy of the region. The fauna of the east coast is essentially an extension of the tropical fauna to the north, and partly driven by high rainfall and the warming effects of the southward flowing Agulhas current down the coast. This area is also subject to the effects of the El Niño Southern Oscillation (ENSO) and, therefore, cycles of drought and high precipitation. This results in a dragonfly fauna that is highly vagile and opportunistic, seeking suitable water conditions according to the prevailing weather (Samways 2010). There are very few localized endemics in this region (Samways 2008a). By contrast, in the southern, and especially the southwestern, Cape [Cape Floristic Region (CFR)] there is a more predictable cycling of dry summers and wet winters, and a large number of highly localized endemics, many of which are threatened (Samways 2006). Inland there is a general mix of higher elevation localized endemics and, at lower elevations, of savanna species, which tend to be widespread and opportunistic generalists (Van Huyssteen and Samways 2009). While there is the common impact of invasive alien plants across the whole region (Samways and Grant 2006), other issues, such as water abstraction of mountain rivers, is really only a major problem in the water-scarce CFR. These adverse impacts are often synergistic with climate change, especially global warming, which is a serious concern for some of the CFR endemics of upland streams (Samways 2008b).

These results depend on having thoroughly tested the DBI under various climatic and weather conditions, and knowing its variation. Having a correct DBI value depends, firstly, on good and thorough identification of all the species. It then depends on sampling all those species at a particular site, which can be done by 2–3, or preferably more, site visits on relatively windless and sunny days so that a species accumulation curve asymptote is reached. Some supplementary sampling of crepuscular species is also necessary. In the Western Cape, this procedure must be repeated in both spring and fall, so as to cover the seasonal spectrum of species in that region. Further north, in the summer rainfall areas, this is not necessary as there are overlapping generations with the flight period reaching a peak mid-summer, the ideal time for sampling (Samways and Niba 2010).

The DBI can be used to quantify these various changes, whether shorter-term, decadal cycling of wet and dry conditions, or the longer-term effects of global climate change. Shifts in assemblage structure can be detected as a change in the DBI and quantified further by identifying which species are responsible for the changes. Cycling events will see a drop in the DBI in dry years and an increase in wet years. When placed in a comparative spatial context, the DBI can also be used to quantify the value of dry-year refugia (Samways 2010), although this would pertain more to still than running waters. In regions rich in endemics, such as the CFR, it can also be used to quantify shifts along river systems in the 'quality' (i.e., species richness in endemics versus richness in generalists) as the climate changes. Spatial data in this regard will then identify the adaptability of the endemic species to these climate changes.

Conclusions

The DBI is a remarkably resilient index. Besides being easy to use, with adult dragonflies much easier to sample than aquatic benthic macroinvertebrates, this index can be used for a variety of objectives. It can be used to assess impacts of invasive alien riparian trees, the success of restoration projects, and prioritization of sites for action at both the local and the regional scales. Indeed, using the DBI, conservationists can easily track community response to change. The HCS can be used to interpret the naturalness of the overall habitat, which, for South African conditions, not only includes biotope diversity, but also shade from invasive alien trees, a key threat to most of the local Red Listed species. However, like all benthic macroinvertebrate indices, it cannot necessarily identify which impacts are having which effect and to what extent. This is particularly so in terms of identifying which particular chemical pollutant may be causing a problem. Also, its power lies in knowing the habitat range of the focal species, having some at least that are threatened, and preferably some endemic species. This means that it would have to be adapted for Palearctic conditions and require more fundamental research in geographical areas where the dragonfly fauna is poorly known. Similarly, the HCS would also have to be modified according to the key threats identified in any specific region under study. In short, both the DBI and the HCS are tested, workable, and feasible frameworks that have proven to be of great value in South Africa and could readily be adapted for other goals and conditions elsewhere.

Acknowledgments We would like thank our colleagues from the Department of Water and Agricultural Affairs (DWAF) for their work on SASS. This project was funded in part by Nature's Valley Trust, the RUBICODE Coordination Action Project (Rationalising Biodiversity Conservation in Dynamic Ecosystems) funded under the Sixth Framework Programme of the European Commission (contract no. 036890), and by a Natural Sciences and Engineering Research Council of Canada (NSERC) PGS-D3 scholarship to John Simaika.

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