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A diagnostic biotic index for assessing acidity in sensitive streams in Britain

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ABSTRACT

Despite the history of freshwater biomonitoring, there is still a dearth of proven indices that allow accurate status assessment while simultaneously diagnosing the cause of impairment, particularly when stressors are multiple. Here, we present an approach to developing diagnostic indices where the sensitivity of biota is quantified using multivariate ordination. We applied the approach to the development of an index to detect acidity in British streams.

Using a 197-site calibration dataset, we quantified variation in macroinvertebrate assemblages and determined which environmental variables best described the pattern. We then ranked taxa along an acid–base gradient, having first considered the merits of factoring out confounding variation from natural environmental factors. The response of the new species-level Acid Water Indicator Community (AWIC*sp*) index to variation in base-flow and storm-flow pH and acid neutralising capacity (ANC) was quantified using independent data. Performance was also compared with existing family-level and species-level indices.

AWIC*sp* was consistently the species-level diagnostic index most clearly related to base-flow pH, storm-flow pH and ANC, accounting for 38–56% of the variation in acid conditions among the 76 test sites.

Given the need to develop bio-diagnostic indicators, these data illustrate how organisms can indicate causes of stream impairment using robust and objective procedures, and when applied to strong environmental gradients such as acid–base status. We suggest that given the necessary calibration data, this approach could be applied successfully to other widespread stressors with equally strong biological effects such as organic pollution and fine sediment deposition, particularly if used in combination with RIVPACS-type predictive bioassessment models.

1. Introduction

In freshwaters, more than any other environment, there is a strong tradition of inferring ecosystem condition from biological communities (Rosenberg and Resh, 1993; Jones et al., 2010; Geist, 2011). Historically this had mainly been focussed on the impacts of a small number of stressors, typically organic pollution (Sládeček, 1967; Armitage et al., 1983). With improvements in the treatment of organic discharges over recent decades, environmental managers are increasingly turning their attention to the impacts of other stressors such as habitat modification, nutrient enrichment, metal pollution, acidification and sedimentation, which often occur diffusely away from point-sources, and often in combination. In these circumstances, there is need not only to detect the field effects of pollutants on communities as distinct from laboratory organisms (Iwasaki and Ormerod, 2012), but also to diagnose which have greatest effects thereby guiding the most appropriate management action (Suter, 2001; Clews and Ormerod, 2009; US EPA, 2010). Unfortunately, adequate bioassessment tools are not always available to allow the reliable diagnosis of the stressor responsible for impaired water quality (Jones et al., 2010). While many diversity, community composition and multi-metric indices are available: e.g., taxon richness, relative abundance of particular groups, macroinvertebrate index of biotic condition (Barbour et al., 1995; Stoddard et al., 2008), few have been developed to diagnose particular stressors (e.g., Dahl and Johnson, 2004; Carlisle et al., 2007).

The need for reliable diagnosis is increasingly prompting improved empirical understanding of how biological communities

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respond to changes in levels of particular stressors, but more particularly how any derived biotic indices can be shown to respond directly only to the stressor of interest rather than to other stressor variables. Recent advances in the collection, storage and analysis of biological and environmental data, together with increases in the quantity of data being collected, mean that it is now possible to develop the required robust diagnostic biomonitoring tools. In this paper we present an approach to developing stressor-specific diagnostic indices where the stress-sensitivity of biota is quantified, having first considered the confounding effects of other unrelated environmental variables. We used acidification as a case study to illustrate the development and testing procedure.

There are a number of different sources of acidification in freshwaters, natural (e.g., poorly buffered peaty soils) and anthropogenic (metal mining waste) (Mason, 1996; Petrin et al., 2008). However, probably the most widespread post-industrial source has been the atmospheric deposition of sulphur and nitrogen oxides originating from the burning of fossil fuels in power stations and other industries. Substantial reductions in atmospheric acid deposition over the past two decades have resulted in the general chemical recovery of many sensitive waters (Evans et al., 2001; Skjelkvåle et al., 2005). In contrast, the evidence for biological recovery is more limited, with many locations having gained only a fraction of sensitive species that were initially lost (Gunn and Sandøy, 2003; Monteith et al., 2005; Ormerod and Durance, 2009), while widespread recovery is apparently impeded by factors that include climate, continued acid episodes and land uses that exacerbate effects locally (Yan et al., 2003; Kowalik et al., 2007; Monteith and Shilland, 2007; Ormerod and Durance, 2009). There is, therefore, a continuing need to quantify the recovery process, to monitor affected streams, to ensure their improvement towards a target state and to regulate local land-uses that might still exacerbate problems locally. For European Union member states, the Water Framework Directive (WFD; European Commission, 2000) requires that all streams are in at least 'good ecological status' by 2015. National regulatory authorities will need diagnostic tools that can both identify streams degraded by acid deposition and indicate when they have successfully recovered.

Studies over the past three decades have shown that aquatic invertebrate assemblages generally decline in diversity with decreasing pH (Sutcliffe and Carrick, 1973; Simpson et al., 1985; Dangles and Guérold, 2000), most likely reflecting a combination of increased hydrogen ion concentrations coupled with increased bioavailability of toxic elements such as labile aluminium and, locally, some heavy metals (Ormerod et al., 1987; Matschullat and Wyrobek, 1993; Lepori and Ormerod, 2005). The relative sensitivities of different invertebrate taxa to acid conditions are known to the extent that several biotic indices have been developed in different countries that, to a greater or lesser extent, claim to indicate the acid status of streams (Hämäläinen and Huttenen, 1990; Raddum, 1999). In Britain several regional studies have related macroinvertebrate community composition to acid gradients and identified those families or species that are most sensitive and tolerant to low pH (Wade et al., 1989; Rutt et al., 1990). Here, however, no proven macroinvertebrate index has ever been adopted universally by regulatory agencies and researchers.

Ecological quality targets set by the WFD for all surface water bodies mean there is now an urgent need for proven diagnostic indices that allow managers to accurately assess the status of rivers and streams. This has provided the impetus for the recent development and testing of a robust and widely applicable family-level index for use throughout Britain, the Acid Water Indicator Community (AWIC*fam*) index (Davy-Bowker et al., 2005; Ormerod et al., 2006). At the same time, however, this index shares a limitation with other routine biological monitoring of streams in Britain by using family level macroinvertebrate data. There is an opinion that some community response information is lost by not identifying specimens to a more detailed taxonomic level, particularly where families contain genera or species with contrasting environmental preferences (Lenat and Resh, 2001; Verdonschot, 2006). The species-level Raddum Index (Fjellheim and Raddum, 1990), developed in Norway, has occasionally been applied to British data (e.g., Soulsby et al., 1997), but is based on a different suite of species than those found in the British ecoregion (Illies, 1978). Ideally, a new species-based index should have a more pronounced stress–response relationship with the acidity gradient than (i) the family-level AWIC*fam* and (ii) other species-level acid indices used previously in Britain or elsewhere. We call these tests 1 and 2, respectively.

The current study describes an empirical approach to the development and testing of a diagnostic index, using as a case-study, the changes to stream macroinvertebrate communities along a gradient of increasing acidity in British streams. We also compare the performance of the new index to established species-level indices from Britain and Scandinavia.

2. Materials and methods

2.1. Training dataset

A training dataset was compiled to best represent the range of acid conditions experienced within Britain, and was derived from 197 benthic stream macroinvertebrate samples from seven different sources (see supplementary information) (Fig. 1). Macroinvertebrates were sampled at each site using comparable methods based on a 3-min kick sample of all habitats within the stream. Specimens were identified to species level where possible; otherwise to the most detailed level possible (e.g., *Lype* sp., *Baetis scambus* group). However, it was necessary to standardise taxonomic resolution where different data sources identified the same groups to different levels, resulting in the loss of some taxonomic discrimination e.g., many species- and genus-level records in the caddis fly family Limnephilidae, had to be down-graded to family-level, due to the large number of early-instar specimens that could only be identified to family level. Only genus and species-level records were then retained in the training dataset. The presence/absence of all 48 taxa (11 genera and 37 species) was recorded in each sample. Only spring samples were considered because this is the period after which stream discharge has usually been at its annual peak and the biotic community is most likely to respond to variations in acid conditions across sites (Weatherley and Ormerod, 1987; Wade et al., 1989). Only sites considered not to be impacted by other anthropogenic stressors were included in the training dataset.

Each biological sampling site was associated with at least five pH measurements taken within 3 years prior to, and 1 year after, the invertebrate sampling date. The availability of data for other acid–base parameters (e.g., alkalinity) or for associated metals (e.g., aluminium), was very inconsistent across sites and therefore only pH could be included in analysis (expressed as mean, and mean of the two lowest measurements [mean-low pH]). The elevation (m), slope (derived from 1:50,000 scale maps, m km^{-1}), distance from source (km) and 1:50,000 stream order (Strahler, 1957) were acquired for each site from the Intelligent River Network; a GIS-based application for the automated extraction of physical data from a 1:50,000-scale river network of Great Britain (Dawson et al., 2002; see supplementary data for range information). Each of the 197 samples was from a separate watercourse and together incorporated a gradient in mean pH from 4.6 to 7.0 and 4.0 to 6.6 for mean-low pH (Fig. 1, see supplementary data).