Ecological Indicators 61 (2016) 1000–1010

Contents lists available at ScienceDirect

Ecological Indicators

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

A hydrologically sensitive invertebrate community index for New Zealand rivers

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ARTICLE INFO

Article history:
Received 7 September 2015
Received in revised form 22 October 2015
Accepted 23 October 2015
Available online 21 November 2015

Keywords:
Aquatic invertebrate
Flow regime
Water velocity
Indicator
Water quality

ABSTRACT

Identifying ecological response variables sensitive to hydrological change is a key step in determining the impacts of river flow alterations on aquatic ecosystems and in setting environmental flows that sustain certain ecological values. Building on the successful use of flow regime sensitive aquatic invertebrate indices in other countries, particularly the UK based Lotic Index for Flow Evaluation (LIFE), we provide two variants of a similar index for use in New Zealand (LIFENZ and a weighted variant: LIFENZ_W). As in the original LIFE, the New Zealand indices were based on water velocity preference categories assigned to aquatic invertebrate taxa using professional judgement. To calculate the indices a lookup table is used to assign a score to each taxon based on their velocity category and abundance. For the LIFENZ_W variant an additional step down weighted the scores if the taxon has a general compared to a more specific velocity preference. The two index variants were correlated with each other and to similar environmental parameters. Across a total of 74 sites, both indices were correlated with depth-averaged water velocity. Changes in index values, both between sites and temporally within sites, were predominantly associated with changes in hydrological parameters, such as the magnitude and length of time since a recent high flow, and to a lesser degree with other physico-chemical parameters. Commonly used indices in New Zealand designed to respond to nutrient enrichment (MCI and variants) were not correlated with local water velocity, but were correlated with antecedent flow conditions and were likely influenced by effects of flow stability on algal growth. Further testing of LIFENZ and LIFENZ_W in combination with MCI is recommended, particularly in rivers subject to more extreme hydrological and water quality stresses and with regard to other physical parameters such as hydraulic habitat. However, the LIFENZ and its weighted variant (LIFENZ_W) appear to be useful tools for understanding and managing the effects of hydrological alteration on aquatic invertebrate communities in New Zealand. As LIFENZ and LIFENZ_W were strongly correlated and only showed a relatively small deviation from a slope of 1 we recommend the use of the more straightforward LIFENZ in almost all circumstances.

1. Introduction

Hydrological regimes are defining features of river ecosystems, influencing channel morphology and in-stream habitat (Arscott et al., 2002), the rates of ecosystem processes (Resh et al., 1988; Power, 1995), the composition, diversity and abundance of communities (Townsend et al., 1997; Lake, 2000) and the evolutionary adaptations and life histories of specific taxa (Lytle and Poff, 2004). Flow regulation and water abstraction are therefore major factors altering river ecosystems worldwide (e.g., Stanford et al., 1996; Nilsson et al., 2005) and are linked to a variety of ecological impacts in both aquatic and adjacent riparian ecosystems (Poff and Zimmerman, 2010). As a consequence, there is increasing demand for science-based management tools to assist in setting environmental river flows that balance anthropogenic uses with the maintenance of functioning ecosystems (Palmer et al., 2005; Poff et al., 2010).

Determining effective environmental flows requires identification of suitable biological response endpoints and quantifying their relationships with various aspects of the hydrological regime (Bunn and Arthington, 2002; Arthington et al., 2006; Poff et al., 2010). However, the complexity of the hydrological regime and ecological processes, combined with an ability to generate hundreds of possibly correlated hydrological indices (Olden and Poff, 2003), makes

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http://dx.doi.org/10.1016/j.ecolind.2015.10.058
1470-160X/

identifying the most relevant drivers of eco-hydrological relationships difficult (Monk et al., 2007). Identifying univariate measures of biological responses to river flow alterations (e.g., Armanini et al., 2011) is one tool that can help advance understanding of the eco-hydrological relationships needed for effective environmental flow setting (Arthington et al., 2006; Poff et al., 2010).

Aquatic macroinvertebrates are commonly used for the assessment and monitoring of water quality in running waters (Boothroyd and Stark, 2000) as they are ubiquitous, affected by local conditions, provide a longer-term view of in-stream conditions than water chemistry indicators and often have relatively well defined taxonomy and ecological preferences (Wallace and Webster, 1996; Boothroyd and Stark, 2000). Similarly, changes in macroinvertebrate community composition have been used to investigate the effects of alterations to the hydrological regime, such as artificially reduced flows below dams (Rehn, 2009), increased discharge due to flow restoration (Merigoux et al., 2015) and the effects of flow intermittence (Arscott et al., 2010). Indices designed to respond to changes to the hydrological regime have been developed using aquatic invertebrates in both Europe and North America (e.g., Extence et al., 1999; Armanini et al., 2011; Timm et al., 2011). Such indices can act as indicators of ecological effects of altered flow regimes and have been used to distinguish changes in flow due to hydroelectric dams (Kairo et al., 2012; Armanini et al., 2014), to assess aquatic invertebrate community composition across river classes (Monk et al., 2006) and to identify ecological responses to both high and low flow events (Monk et al., 2008).

The Lotic invertebrate Index for Flow Evaluation (LIFE), which is based on water velocity preferences of benthic macroinvertebrates, has been shown to respond to both natural and anthropogenic variations in flow in the UK (Extence et al., 1999; Clarke and Dunbar, 2005). LIFE is now commonly used in Britain to identify sites subjected to hydrological stress and in the development of river management plans (Monk et al., 2008), however, the index is not directly applicable to regions with different fauna.

In New Zealand, macroinvertebrates are used by most regulatory authorities as part of water quality biomonitoring programmes (Collier et al., 2000) and their depth, velocity and substrate preferences have been used to help set minimum flows in rivers subjected to water abstraction (Jowett and Mosley, 2004). However, New Zealand does not have an invertebrate index designed to respond to changes in the river flow regime (but see Schwendel et al., 2011 for an index designed for bed movement). Existing macroinvertebrate indices in New Zealand include the Macroinvertebrate Community Index (MCI) and its variants that are often used as proxy indicators of water quality (Stark and Maxted, 2007b). They are generally insensitive to local water velocity (Stark, 1993) but can be affected by floods and extended periods of low flow (Boothroyd and Stark, 2000), particularly in more pristine waterways (Death et al., 2009). In general, aquatic invertebrate community indices will respond to any factor that influences macroinvertebrate community composition (Boothroyd and Stark, 2000) and indices that are designed to be sensitive to specific environmental stressors often (e.g., Kairo et al., 2011), but not always (Armanini et al., 2011), show overlap in the parameters that influence their observed values. Thus, one of the common criticisms of the use of biological indices in rivers is the difficulty in determining the mechanistic causes of changes in scores due to the joint effects of hydrology, water quality and habitat on invertebrate communities (Chessman and McEvoy, 1998). However, invertebrate indices designed to be sensitive to hydrology have been shown to be better at distinguishing the effects of altered hydrological regimes than indices designed to be sensitive to water quality (Monk et al., 2006; Kairo et al., 2012). Furthermore, even when indices designed to be responsive to separate environmental stressors are correlated, using them in combination can help elucidate the drivers behind observed patterns in the aquatic invertebrate community (Clews and Ormerod, 2009).

The objectives of this work were to create a flow regime-sensitive aquatic invertebrate community index for use in New Zealand and to compare its performance with existing indices designed to respond to water quality. Ideally, an index designed to be sensitive to hydrology will respond to local water velocity, be more responsive to temporal changes in river flow than water quality or habitat conditions and be influenced by between-site differences in hydrological regime rather than variables associated with nutrient enrichment or agricultural land use. Our specific hypotheses regarding the testing of the new index were:

1. It would be responsive to changes in water velocity (H1a) while existing water quality indices would not (H1b).
2. It would be responsive to recent flow conditions and less responsive to changes in nutrient concentrations or water temperature over time at a site (H2a). In contrast, nutrient enrichment sensitive indices would be more responsive to changes in nutrient conditions than to flow conditions (H2b).
3. Differences between sites in regard to hydrological regime would be more strongly associated with the new index than parameters associated with agricultural land use or nutrient enrichment (H3a) while nutrient enrichment sensitive indices would be more strongly associated with parameters representing agricultural land use (H3b).

2. Methods

2.1. Invertebrate data

Two datasets were used to create and test the index: an extensive national dataset focussed on relatively large rivers and a regional dataset from smaller lowland rivers. The national dataset comprised 66 sites on New Zealand's National Rivers Water Quality Network (NRWQN) (Davies-Colley et al., 2011) (Fig. 1). The network was initiated in 1989 with an aim to monitor long-term trends in water quality, biology and habitat (Smith et al., 1989). The river catchments in the monitoring network together drain about half of New Zealand's total land area and are biased towards larger rivers. Mean flows observed at gauging stations in the rivers vary from 0.8 to 567 $m^3 s^{-1}$ (Smith et al., 1989; See Table S1 in the supplementary materials for more site details). Since 1989 aquatic invertebrate samples have been collected annually in late summer to early autumn and under baseflow conditions ($Q < Q_{median}$) (Scarsbrook et al., 2000). At each site seven Surber samples (0.1 m^2, 250 μm mesh) are collected from relatively shallow (∼<0.7 m deep) areas with moderate water velocities (0.2–1 $m s^{-1}$) and cobble or gravel substrate (Smith et al., 1989). The samples from seven locations at each site are pooled into one sample for analysis. We used data from 1990 to 2011 and restricted aquatic invertebrate samples to those collected between December and April (Austral summer and autumn) to reduce seasonal differences in densities and community composition of aquatic larvae. Samples were sorted in the laboratory using a full count method with sub-sampling of abundant (>100 individuals) taxonomic groups and identification to the lowest taxonomic resolution possible, most commonly genus level. Taxonomic resolution of aquatic invertebrate samples was standardised to be consistent across all years of data. The selected sites had annual data available for between 16 and 22 years. Depth averaged water velocity (at 0.6 depth) was measured at all seven Surber locations in each river and averaged to give one value per site visit. We assumed that depth-averaged water velocity could be used as