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An invertebrate predictive model (NORTI) for streams and rivers: Sensitivity of the model in detecting stress gradients

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ABSTRACT

NORTI (NORThern Spain Indicators system) is a predictive model for assessing the ecological status of rivers of Northern Spain based on invertebrates. The system can be used to assign any test site to a type of river under minimal disturbed conditions. Macroinvertebrates were sampled with a multihabitat approach from 676 sites covering the variation in environmental conditions across Northern Spain, between 2000 and 2008 ($n = 1421$ samples), including a spatial network of 108 reference sites selected by the absence of significant pressures. A multinomial logistic regression was conducted using the GAAC cluster-derived groups of reference sites as response variable. Obligatory typology factors, following WFD System A, were included as forced entry terms in the model, other potential predictors were selected using a forward stepwise procedure. Ecological quality ratios (EQRs) were estimated from the observed similarity between the faunal composition of the sample of interest (test sample) and the expected median similarity for the reference community of each river type. The model predictions as EQRs responded significantly to the most important pressures: sewages inputs, eutrophication, hydromorphological alterations, and intensive and low intensity agriculture, demonstrating its accuracy in detecting impact in Northern Spanish streams and rivers.

1. Introduction

Predictive models have been used for the bioassessment of streams and rivers for nearly 30 years (Wright et al., 1984; Moss et al., 1987; Hawkins et al., 2000; Reynoldson et al., 1995, 2000; Simpson and Norris, 2000), and there is renewed interest in European countries (e.g. Kokeš et al., 2006; Feio et al., 2007) since the appearance of new water legislation such as the Water Framework Directive (WFD; 60/CE/2000). The technical basis underlying predictive modeling is substantially different from the "upstream/downstream" approach more traditionally used to evaluate the degree and magnitude of impact (Green, 1999). The approach compares impacted sites with an expected reference community. In essence, predictive modeling aims to predict the composition of biological communities based on stream environmental attributes that may influence the distribution and abundance of species. For a given site, the predicted community would represent the biotic conditions that would exist under no impact and, thus, can be used as a reference to infer the departure of such site from its natural status. It should be noted that the utility of predictive models strongly depends on the existence of a relationship between the selected environmental features and the species occurrence or/and abundance (Wright, 2000). In this sense, changes in the species habitat templet (*sensu* Southwood, 1977), influenced by natural (drought, floods) or human disturbances (climate change, organic or chemical pollution, hydromorphological alterations), or by both, can change patterns of species distribution, abundance, diversity and ecosystem functioning (Puccinelli, 2011).

Macroinvertebrates are commonly used as bioindicators of environmental stress in streams and rivers (Bennett et al., 2011), as they may respond to environmental change from a variety of disturbances ranging from nutrient enrichment to hydromorphological alteration (Johnson and Hering, 2009). This capability of macroinvertebrates should ideally be integrated in assessment systems in a way that may allow the detection of impact across multiple pressure gradients, as opposed to the selection of responses targeting a specific single stressor (i.e. organic pollution, such as

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the saprobic index (Zelinka and Marvan, 1961), or hydromorphological degradation (Lorenz *et al.*, 2004)), as it has been indicated for macrophytes (Kanninen et al., 2013), to account for synergies or antagonisms between stressors (Darling and Côte, 2008). Predictive models can detect multiple directional shifts in community composition as a response to single or multiple combined stressors, because they use the whole community in assessing disturbance, and mostly because the structure of the community is not constrained by a priori selection of biological responses to stressors causing degradation (i.e. construction of multimetrics combining biological metrics that respond individually to different stressors).

The WFD provides scientific and technical guidance documents for its implementation across Europe, establishing standardized ecological designs for biological assessment to ensure comparable environmental objectives among countries (e.g. Bennett et al., 2011; Kelly et al., 2012). A basic requirement is the division of natural aquatic ecosystems in water categories (rivers, lakes, wetlands, coastal and transitional waters) and, within each, in "types". The "type" comprises aquatic ecosystems of similar structure and function, conceptually they are characterized by type-specific chemical and hydromorphological conditions and inhabited by specific biotic communities. They need to be differentiated using abiotic descriptors (e.g. altitude, catchment area, geology), some being obligatory and some optional. It must be stressed that the biological communities cannot be used to infer typologies to avoid circularity in assessment, despite the fact that the expected unimpaired biological communities should be type specific (WFD: Directive, 2000/60/EC). Another important concept is the "ecological status", defined as the "expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters" (art. 2.21). The concept of reference condition (Reynoldson et al., 1997; Bailey et al., 1998; Reynoldson and Wright, 2000; Bailey et al., 2004) is at the core of the ecological status classification systems of the WFD used to gauge the effects of human activity (Karr and Chu, 1999). In brief, the European application of the concept used spatial networks of minimally disturbed streams and rivers *sensu* Stoddard et al. (2006) for each "type", compliant with a given set of pressure criteria thresholds. The check with pressure criteria should ensure the absence of significant impact at those sites (Pardo et al., 2012). Consequently, the type-specific derived biological reference conditions are the benchmark against which any test site belonging to the type is assessed. Finally, biological classification systems for the ecological status have to show a significant response to pressures, in line with the analytical framework for the assessment of pressures and impacts (Driving force-Pressure-State-Impact-Response; DPSIR), adopted by the European Environment Agency (1999). Moreover, they have to provide some understanding of the level of confidence and precision of the ecological status assessment, to estimate the confidence to which an individual water body can be assigned to an ecological status class (Clarke and Hering, 2006).

The aim of this study is to develop a predictive model for Northern Spain that meets the new scientific and technical requirements described in the new EU policy for water management. Within this framework, biotic and abiotic data were assembled from minimally disturbed sites or reference sites, where the absence of significant pressure criteria was verified. Here, we describe the development of the invertebrate NORTI (NORThern Spain Indicators system) predictive model for streams and rivers of Northern Spain, a fully WFD compliant method. We used stress gradients to test the invertebrate response to multiple sources of pollution and stream degradation, in order to assess the sensitivity of the predictive models. The classification system developed here is presently used by the Regional water Authorities in Northern Spain to assess the ecological status of streams and rivers.

2. Methods

2.1. Study area

The study area included most of the Northern coast of Spain, from the Western Atlantic corner (Galicia) to the beginning of the western Pyrenees (Navarra) in the East, covering an area of 38,450 km^2 (Fig. 1). The latitudinal range is small with the high mountains, where the rivers originate, very close to the Northern coast. The altitudinal range is high, from a maximum altitude of 2640 to 0 m.a.s.l. In the Western part, the longest river (River Miño) drains the largest catchment of 16,357 km^2. The dominant climate is oceanic, with abundant rainfall throughout the year (mean annual precipitation of 1500 mm [Coastal Galicia water district], mean annual precipitation of 1175 mm [Miño-Sil water district] and 1350 mm [Cantabrian water district]), and moderate variation in temperatures, with mild winters and cool summers. The geology in the area varies from mainly granitic and siliceous rocks in the West, to a dominance of carbonate rock in the Northeast. The highest human population density occurs near the coast and, in particular, in the Eastern and Western parts of the study area (total population in the study area is approximately 7 million). The main anthropogenic effects on streams and rivers are motivated by channel modification in urban areas, flow regulation for hydropower generation, point source organic and industrial pollution, and diffuse pollution from agriculture.

Rivers within the Northern catchments are short, rapid and plentiful, flowing South to North through steep valleys. The exceptions occur within the Miño-Sil Catchment, where some long rivers with multiple tributaries flow east-west through elongated and narrow valleys. Most streams and rivers are fast flowing high gradient streams, because of the proximity of their mountainous origin to the sea, their substrate is generally coarse as a consequence of large hydrological variation. Legislation requires the rivers in Northern Spain to maintain, at minimum, narrow forested riparian areas, thus, helping to maintain the hydromorphological integrity of streams and rivers, while supporting habitat and providing litterfall inputs that sustain stream functions in these oligotrophic streams (Pardo and Alvarez, 2006).

2.2. Data collection

In this study, a total of 676 sampling stations were identified spatially incorporating the wide range of existing environmental gradients, aiming to cover thoroughly the natural variability of stream types and reference conditions that exist in the studied area. Each sampling station was sampled at least once between 2000 and 2008. Most stations were sampled only once a year, in summer, except for 2006 and 2008, when 162 and 6 stations, respectively, were also visited in spring. The total number of samples collected was 1421. Sampling effort per year varied, the smallest number corresponding to 2000–2002 ($<$10 samples per year) and the highest in 2003, 2006 and 2008 ($\geq$290 per year).

2.2.1. Benthic macroinvertebrates

At each station, macroinvertebrates were sampled from 20 subsampling units collected in a 100 m reach using a D-frame dip net (width = 0.25 m; mesh size = 0.5 mm). The subsampled units were selected following the multihabitat sampling approach (adapted from Barbour et al., 1999) in which five major habitats (representing > 5% of the total area) were sampled according to their proportional distribution in the reach. Each subsampling unit covered a distance of 0.5 m, thus yielding a sampled surface area of 0.125 m^2. All subsampling units were pooled into one sample, for a total sampled surface area per site of 2.5 m^2. Samples were preserved in 96% ethanol. In the laboratory, each sample was washed