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A landscape-based predictive approach for running water quality assessment: A Mediterranean case study

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ABSTRACT

The ecological integrity of lotic ecosystems is intimately linked to the quality of their catchments. Environmental protection efforts should therefore be implemented at the relevant catchment-scale in order to support river biodiversity and associated ecosystem services. Therefore, freshwater management can greatly benefit from tools that enable spatially-explicit predictions of water quality across heterogeneous landscapes. To this end, we investigated the relationships between the biological quality of running waters (expressed by the Italian Extended Biotic Index, EBI) and surrounding topographic and land cover matrices in central Italy. To do so, we adopted a spatial approach based on generalized linear models (GLM) that allowed us to identify the catchment features influencing the EBI values, and to produce a map of expected water quality in the region. A gap-analysis was then performed to assess the overlap between the existing regional protected areas and the modelled water quality map. Our findings provide suitable support for the identification of certain land uses in order to preserve riverine biodiversity and hopefully restore lotic ecosystems in Mediterranean catchments.

1. Introduction

Anthropogenic transformation of the landscape is arguably among the most important driver of environmental degradation and biodiversity loss (Pimm & Raven 2000). Ecological effects, however, are usually monitored at the local scale and quantification at regional and national scales is therefore challenging (Vitousek, Mooney, Lubchenko, & Melillo, 1997). Moreover, the relationship between key ecological parameters such as biodiversity distribution and associated environmental drivers must be examined at multiple spatial scales in order to develop a mechanistic understanding (Forman & Godron 1986; Larsen, Scalici, & Tancioni, 2015).

Freshwater ecosystems are highly heterogeneous in terms of structure and functions and are increasingly altered by changes in land use at different spatial scales (e.g., Klein, 1979; Tong & Chen, 2002; Lee, Hwang, Lee, Hwang, & Sung, 2009). Lotic ecosystems, in particular, mirror the quality of the catchment they drain (Cummins, 1974; Harding, Young, Hayes, Shearer, & Stark, 1999), with numerous studies identifying the surrounding landscape as the key determinant of the composition and diversity of aquatic communities (Allan, Erickson, & Fay, 1997; Townsend, Arbuckle, Crowl, & Scarsbrook, 1997; Townsend & Riley 1999; Strayer, Beighley, Thompson, Brooks, & Nilsson, 2003; Townsend, Dolédec, Norris, Peacock, & Arbuckle, 2003). For instance, the distribution of aquatic insects appeared equally influenced by in-stream features as well as riparian and upland land use (Ormerod, Rundle, Lloyd, & Douglas, 1993; Traversetti, Scalici, Ginepri, Manfrin, & Ceschin, 2014, Traversetti, Ceschin, Manfrin, & Scalici, 2015).

Although previous studies have focused on the relative proportion of land use types (e.g., natural woodland, urban, and agricultural areas) within catchments to explain variations in local water quality (Klein, 1979; Uuemaa, Roosaare, & Mander, 2005; Alberti et al., 2007), our understanding of the underlying mechanisms is still limited. Indeed, different land uses may alter both water chemical-physical parameters (Malmqvist & Rundle, 2002; Horn, Rueda, Hörmann, & Fohrer, 2004) and biological assemblages (Townsend et al., 2003; Tran, Bode, Smith, & Klepper, 2010). Klein (1979) initially proposed a model to predict and quantify changes

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Table 1
Land cover categories considered in the analyses (Bossard et al., 2000).

Variables	Description
WOO	Natural wood areas with vegetation formation composed principally of trees, including shrub and bush understories represented by broadleaved and coniferous species
URB	Urban areas include: (i) continuous urban fabric where most of the land is covered by buildings, roads and artificially surfaced area; (ii) discontinuous urban fabric where most of the land is covered by structures buildings, roads and artificially surfaced areas associated with vegetated areas and bare soil; (iii) industrial or commercial units where artificially surfaced areas (with concrete, asphalt, or stabilised, e.g., beaten earth) devoid of vegetation, occupy most of the area. Furthermore in the variable are included road and railways networks, port areas, airports, mine, dump and construction sites
FRU	Parcels planted with fruit trees or shrubs: single or mixed fruit species, fruit trees associated with permanently grassed surfaces. Includes also chestnut and walnut groves
FIR	Areas affected by recent fires, still mainly black
OLI	Areas planted with olive trees, including mixed occurrence of olive trees and vines on the same parcel
GRA	Dense, predominantly graminoid grass cover, of floral composition, not under a rotation system. Mainly used for grazing, but the fodder may be harvested mechanically. Includes areas with hedges (bocage). The category includes also low productivity grassland. Often situated in areas of rough uneven ground
AGR	Arable land with non- and permanently-irrigated lands including cereals, legumes, fodder crops, root crops, fallow land, flower and tree (nurseries) cultivation, vegetables, aromatic, medicinal and culinary plants. Rice and crops cultivations are also included
WAT	Inland wetlands, inland marshes, peatbogs, coastal wetlands, Salt marshes, salines and íntertidal flats
SAN	Beaches, dunes and expanses of sand or pebbles in coastal or continental, including beds of stream channels with torrential regime
VIN	Areas planted with vines

in water quality and macroinvertebrate diversity as a function of anthropogenic land use in the catchment. To date, several models have been developed (Hawkins, Norris, Hogue, & Feminella, 2000; Tong & Chen 2002; Tran et al., 2010). In particular, Tran et al. (2010) showed that catchment land use exerts a direct influence on the biological water quality of running waters as reflected in biotic indices. Biotic indices are a widely recognized tool in freshwater management, which are used to cost-effectively assess ecosystem integrity. The benthic macroinvertebrate diversity, in particular, is a widely used bio-indicator (Lenat, 1993; Stark, 1998; Klemm et al., 2003) and in Italy it has been officially monitored for decades (e.g., Ghetti et al., 1997; Manfrin, Larsen, Traversetti, Pace, & Scalici, 2013; Traversetti & Scalici, 2014). In particular, The Extended Biotic Index (EBI; Ghetti, 1997) was legally adopted by the Italian Legislative Degree 99/152 for the assessment of rivers' water quality until 2010, when the implementations of EU directives (WFD) led to the introduction of the new Intercalibration Common Metric Index (ICMi, AQEM consortium, 2002).

The aim of the present work is threefold: (1) to contribute to the understanding of how land use influences stream water quality across highly heterogeneous Mediterranean catchments; (2) to propose a new model to predict the expected biological quality of water courses based on the identification of the most important landscape parameters (see below); (3) to perform a first gap-analysis comparing rivers' water quality with the regional network of protected areas (Habitat Directive sites and National Protected Areas—CEC, 1992; Italia, 2010), in order to evaluate the effectiveness of the established reserves for the conservation of freshwater ecosystems.

2. Material and methods

2.1. Biotic Index

In this study, we used the Extended Biotic Index (EBI) (the only official index at the time of data collection—Supplementary material 1) calculated for the development of the Biological Quality Map of Latium watercourses (Mancini & Arcà, 2000). In the computation of the EBI, taxon richness of macroinvertebrate assemblages and presence/absence of the most sensitive groups (i.e., taxa belonging to certain families of mayflies, caddisflies, and stoneflies) are taken into account. Macroinvertebrate specimens were collected in 189 sampling sites belonging to 95 different streams throughout the Latium region, in central Italy, between 1998 and 2000. Macroinvertebrates were collected by kick-sampling with a hand net ($25 \times 9 \times 40$ cm; mesh = 0.9 mm) from one bank to the other for 3 min, paying attention to cover all microhabitats present. Macroinvertebrates were fixed in alcohol (70%) and identified in the laboratory to the taxonomic level required for the EBI computation (e.g., the genus for mayflies and stoneflies; the family for caddis flies and coleopterans) (see Supplementary material 1 for a detailed description of the method).

2.2. Environmental variables

Fourteen landscape level variables were taken into account. Specifically, we calculated the percentage of 10 different land cover categories (Table 1) in 1×1 km cells from the land use thematic map of the Latium Region (Regione Lazio, 2000). This map was realized by interpretation of satellite images and based on the categories of the Corine Land Cover dataset (Bossard, Feranec, & Otahel, 2000; Krynitz 2000). We calculated, for each cell, the mean elevation (MEAN_DEM) and the mean slope (MEAN_SLOPE) for describing the land morphology from a Digital Elevation Model (ISPRA, 2012). As a descriptor of the level of anthropogenic modification in each cell, we calculated the Anthropic Index (AI) by using the following equation:

$$\mathrm{AI} = \sum k_i \mathrm{p_i}$$

where k_i is a specific anthropic coefficient for the ith land use category (1, woodlands; 2, pastures, meadows, bush areas, scrubs; 3, agricultural areas; 4, urban and industrial areas) and p_i is the relative frequency of the same land use category in the cell (Larsen, Sorace, & Mancini, 2010). Precipitation and temperature data were retrieved from the Latium Regional Agrometereological Information System (ARSIAL) database (www.arsial.regione.lazio.it/portalearsial/agrometeo) from 2004 to 2010. Using precipitation and temperature grid cell means, we obtained the De Martonne index (DMI) (De Martonne, 1940), a climatic classification based on the duration of the aridity over the year. The De Martonne Aridity Index (DMI) is defined as the ratio of the mean annual precipitation (P) in mm and annual mean temperature (T) in °C +10. The DMI index is given by the following equation:

$$\mathrm{DMI} = P/(T + 10)$$

All these variables were obtained at the spatial resolution of 1 km in ArcGis 9.3.

2.3. Data analysis

In order to eliminate the effect of correlation between explanatory variables and to ensure that our set of predictors is not saturated (Blanchet, Legendre, & Borcard, 2008), we calculated the variance inflation factor (VIF; Belsley, Kuh, & Welsch, 1980) in R (version 2.12.1, R Development Core Team, 2010). We removed variables until all VIFs were below 5 (Belsley et al., 1980; Marx & Smith, 1990).

In order to take into account the non-independence of EBI data due to the clustering of sites from different water courses, we adopted a resampling technique for stratifying data across water courses. More specifically, we generated 10 random subsets with only one site per water course. This number of repetitions is sufficient to have all data represented in the subsets with no more than one record per water course. For each subset, from the initial full model we selected the significant explanatory variables to be retained in the final model by applying a stepwise procedure based on the Akaike's Information Criterion (AIC; Akaike, 1974; Herrmann, Babbitt, Baber, & Congalton, 2005). This procedure works both backward and forward, by excluding and adding variables one by one from an initial full model, and comparing the resulting AIC values. The AIC score quantifies the fit of the models to the data (based on the likelihood) (Burnham & Anderson, 2002). The variables selected by the stepwise procedure were then used as predictors in generalized linear models (GLM; McCullagh & Nelder, 1989). We adopted a modeling approach for quantifying the effect of each landscape variable on the observed EBI and calculating the expected water quality across the study area. The quantification of such taxa-environment relationships represents the core of predictive modeling in spatial ecology (Guisan & Zimmermann, 2001). We fitted Gaussian GLMs on EBI values of sites in each subset and projected the expected values across the study area. Generalized linear models are mathematical extensions of linear models that allow for non-linearity and non-constant variance structures in the data. Generalized linear model (GLM) is a classical regression approach that has been extensively tested elsewhere and has proven robust in a number of independent situations (e.g., Pearce & Ferrier, 2000). We validated each model by correlating the expected EBI values with those observed in the sites not included in the considered subset. For producing a single output, we generated a consensus model by calculating the average of the subset models.

3. Results

The proportion of woodland (WOO) and the proportion of urban land cover (URB) were the most important factors in all final models (10/10), followed by the cover of orchards (FRU) (8/10) and by other five variables retained in a smaller number of repetitions (see Table 2). Moreover, four land cover variables were never retained in any repetition (see Tables 1 and 2).

The average coefficient values of the WOO and URB land use variables from the 10 subsets were used for producing a final consensus model with which the expected EBI values were obtained.

Predicted EBI values from all subset models were significantly correlated with the observed EBI values at sites not included in the subset ($R \pm \mathrm{SD} = 0.356 \pm 0.069$; $P \pm \mathrm{SD} = 0.015 \pm 0.016$) indicating a good model performance.

The predicted EBI values obtained from the GLM models were implemented into the GIS program, producing the EBI prediction map (Fig. 1—see also Supplementary materials 2 and 3), which shows the pattern of potential EBI at the regional scale. Overall, mountain areas have generally high EBI values. In particular, class I of EBI is predicted in the Appennines, the Sabini, and the Simbruini mountains. Along the seacoast the potential quality value decreases. A fragmented area is identified along the median zone of the Latium region that correspond with an area characterized both by natural areas like Tolfa and Ausoni-Aurunci mountain systems and patches affected by agricultural land use.

Two areas along the River Sacco, at the base of the Simbruini and Ernici mountain system, were identified as supporting particularly high biological water quality.

Within the low-quality matrix of the Tyrrhenian coast, two natural protected areas stand out with high-EBI values: Circeo National Park and Castelporziano.

Results from the gap analysis carried out by overlapping the map of predicted EBI values with the location of protected areas in the Latium region is shown in Fig. 1. Important overlaps were observed. In particular, the Castelli Romani and Monte Rufeno Regional Parks show a clear perimetral overlapping. Other visible cases of positive overlap are evident in the Monti Simbruini, Monti Lucretili and Monti Aurunci Regional Parks. It is interesting to observe that the Natural Areas within and around the city of Rome do not appear to support higher biological water quality.

4. Discussion

Most environmental management activities (e.g., forestry practices, regional planning, and natural resource development) involve decisions that can potentially modify the landscape matrix (Forman, 1986, 1995). This work contributed to the understanding of how natural and anthropogenic land use can influence rivers' integrity over heterogeneous catchments.

There is an extensive literature exploring the effects of landscape pressures on river systems at multiple scales, from the entire catchment to the individual channel units (e.g., Allan et al., 1997; O'Driscoll, Clinton, Jefferson, Manda, & McMillan, 2010; Clapcott et al., 2012; Manfrin et al., 2013). The majority of these studies focused on the evaluation of water quality by means of chemical-physical parameters (e.g., Hawkins et al., 2000; Lee et al., 2009), and only few studies have considered water quality estimated from the application of biotic indices (Allen, 2004; Tran et al., 2010). Townsend et al. (2003), for instance, investigated the influence of both geomorphological and landscape variables on river ecosystems at multiple spatial scales and showed that benthic assemblages were influenced mostly by reach scale characteristics. Nonetheless, the relative importance of local and catchment scale drivers for benthic assemblages is clearly context-dependent and strongly contingent upon the extent and resolution of the study as well as the underlying environmental gradients (Mykra, Heino, & Muotka, 2007).

Macroinvertebrate metrics are, in fact, sensitive to land use pressures at multiple scales, and are widely used as bio-assessment tools (Clapcot et al., 2012). Additionally, use of biotic metrics also represent a diachronic assessment, since the composition of assemblages reflect both current and historic effects of different stressors. The predicted EBI model at the regional scale was strongly affected by two landscape variables: urbanization and forest cover, which were always selected in the best models. The negative effects of urbanization are widely documented and include direct river pollution and hydro-morphological modifications. Sewage and drainage systems, for instance, can flow directly into water courses causing a significant deterioration of physico-chemical water quality (Mancini et al., 2005). Pollution from urban runoff also modifies macroinvertebrates community structure, often leading to a relative increase of generalist species and a loss of specialised and rarer taxa (Sonneman, Walsh, Breen, & Sharpe, 2001; Walsh, Sharpe, Breen, & Sonneman, 2001). Additionally, urbanization is often associated with hydro-morphological alterations, such as channelization and hydric regimentation, with consequent loss of riparian habitats and increases of fine sediments in the channel (e.g., Owens et al., 2005; O'Driscoll et al., 2010).

Table 2
Variable factors (VAR) in the final models for each independent repetition (REP 1–10) and total number of repetitions that retained each variable. For the detailed explanation of the analyzed variables, see Table 1; MEAN-DEM = mean elevation of each 1 × 1 km cell from the DEM (ISPRA 2012); MEAN SLOPE = mean slope of each 1 × 1 km cell from the DEM (ISPRA 2012); N_MOD = number of repetitions retaining the variable. Variables not retained in any repetition have been omitted.

VAR	REP 1	REP 2	REP 3	REP 4	REP 5	REP 6	REP 7	REP 8	REP 9	REP 10	N_MOD
WOO	2.91	3.18	2.56	2.54	2.74	3.13	2.84	2.05	2.84	2.683	10
URB	−3.87	−3.21	−3.57	−3.48	−2.3	−2.56	−3.2	−2.3	−3.31	−3.08	10
FRU	–	−8.77	−6.29	−6.01	−4.9	–	−6.6	−7.9	−4.18	−6.09	8
FIR	22.3	25.6	20	–	–	21	–	18	–	–	5
OLI	–	4.36	–	–	4.49	–	–	2.78	–	–	3
GRA	–	−12.2	–	–	–	–	–	−13	–	–	2
MEAN_DEM	–	0	–	–	–	–	0	0	–	–	3
MEAN_SLOPE	–	−0.18	–	–	–	–	−0.1	–	–	–	2

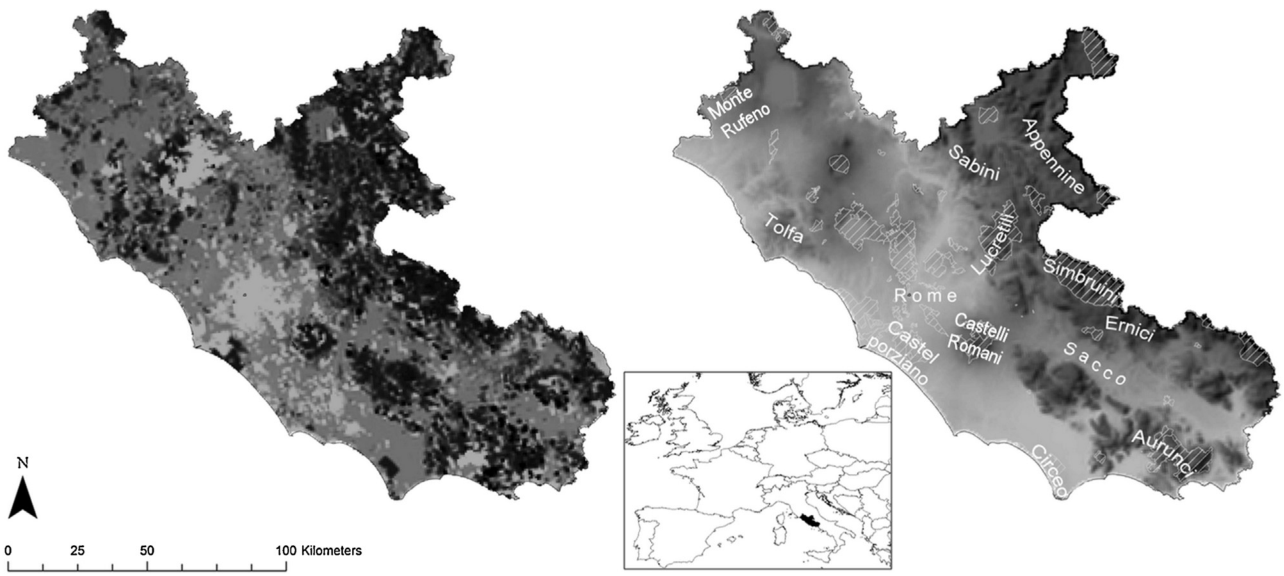

Fig 1. On the left, the potential freshwater quality map in the Latium region: the darkness of the pixels is proportional to the predicted Extended Biotic Index (EBI) class, ranging from 1 (very good quality, black) to 5 (very bad quality, pale grey). On the right, the map shows the potential freshwater quality overlapped with Latium region protected areas (barred polygons) and other important areas and localities further reported in the text.

Woodland vegetation in the catchment and in riparian zones, conversely, can positively influence the EBI values, by directly improving water quality and in-stream habitat availability (e.g., higher bank stability associated with an increase in sediment trapping, input of litter and wood, and retention of nutrients) (Allan, 2004).

If on one hand the majority of regional protected areas supported watercourses with high EBI values and minimally impacted river habitats, on the other hand lower EBI values were observed within the water network of protected areas surrounding Rome. This finding indicates how the proximity to a large metropolis as well as the small size and fragmentation of these nature reserves can outweigh any benefit associated with their protection status. This clearly demonstrates the importance of the surrounding landscape matrix for river ecosystems. It is possible, however, that the observed low water quality might be due to negligent reserve enforcement and management, rather than to the location of the actual reserves.

Results from landscape ecological studies strongly suggest that a broad-scale perspective, which explicitly acknowledges spatial relationships, is necessary in land use planning. The long-term maintenance of biological diversity may require management strategies that place regional biogeography and landscape patterns (Lindenmayer et al., 2008). Moreover, the proposed procedure can be generalized and extended to larger scale with important applications in both national and European conservation and management planning.

Our findings confirmed the importance of assessing landscape pressures in order to propose suitable management plans, as highlighted by Mörtberg, Balfors, and Knol (2007). In this regard, the use of biotic indices allowed us to identify landscape pressures by evaluating their effects on the biotic components and the associated abiotic environment. We presented an easily implemented approach that allows the evaluation not only of stream ecosystems within protected areas, but also the influence of the landscape context in which each reserve exists. There remains a tremendous potential (and a necessity) for truly interdisciplinary cooperation among ecologists, geographers, landscape planners, and resource managers to develop an integrated approach to landscape management.

The proposed interpolation-based EBI model can be suitable in this context because it is cost-effective and is relatively easy to perform and consult. The model could thus provide an important support to both scientists and conservationists in the region and beyond.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.jnc.2016.01.002.

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Further reading

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