

Eucalypt plantations reduce the diversity of macroinvertebrates in small forested streams

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Abstract

Eucalypt plantations reduce the diversity of macroinvertebrates in small forested streams.— Land use patterns of a river basin have a significant effect on the structure and function of river ecosystems. Changes in the composition of riparian plant communities modify the quantity, quality and seasonality of leaf–litter inputs, determining changes in macroinvertebrate colonization and activity. The main goal of this study was to test the effect of land–use modifications, and particularly the impact of eucalypt plantations, on the macroinvertebrate communities of sixteen headwater streams. Macroinvertebrates were counted and identified to family level. Land uses were classified in five categories using aerial photography: native forest, eucalypt plantations, agricultural land, shrubland, and urban areas. We found that macroinvertebrate diversity increased with basin size and with the proportion of basin covered by native forest. This variable correlated negatively with the land occupied by eucalypt plantations. Macroinvertebrate richness diminished with the increase of land surface covered by eucalypt plantations, and a similar tendency was observed with diversity. Furthermore, streams whose drainage basin was mainly covered by *Eucalyptus* were more likely to dry up in summer. This observation adds to evidence from previous studies that concluded that fast–growing tree plantations affect hydric resources, an important ecosystem service in the context of global warming. To minimize the impact of industrial silviculture, we suggest that maintaining and/or restoring riparian forests could mitigate the effects of intensive eucalypt monocultures.

Key words: *Eucalyptus globulus*, Biodiversity, River ecosystems, Land uses, Forest, Tree plantations

Resumen

Las plantaciones de eucaliptos reducen la diversidad de macroinvertebrados en pequeños arroyos forestales.— Los usos del suelo de una cuenca hidrológica ejercen un efecto importante en la estructura y el funcionamiento de sus ecosistemas fluviales. Los cambios en la composición de las comunidades de plantas ripícolas modifican la cantidad, calidad y estacionalidad de las entradas de materia y energía a los ríos, lo que afecta a la colonización y actividad de sus comunidades de macroinvertebrados. El principal objetivo de este estudio es analizar los efectos de los cambios en el uso del suelo y, en particular, de las plantaciones de eucalipto, en las comunidades de macroinvertebrados de 16 arroyos de cabecera. Se contaron macroinvertebrados y se identificaron hasta el nivel de familia. Los usos del suelo se clasificaron en cinco categorías utilizando fotografías aéreas: bosque autóctono, eucaliptal, zona agrícola, matorral y zona urbana. Observamos que la diversidad de macroinvertebrados aumentó con el tamaño de la cuenca y con la proporción de superficie de la cuenca cubierta por bosque autóctono, lo cual resultó estar inversamente correlacionado con la superficie ocupada por eucaliptales. La riqueza de macroinvertebrados disminuyó a medida que aumentaba el suelo ocupado por eucaliptales y se produjo una tendencia similar con la diversidad. Además, nuestras observaciones indican que los arroyos cuyas áreas de captación están cubiertas principalmente por eucaliptales presentan una mayor probabilidad de secarse completamente en verano. Esta observación añade un nuevo indicio concordante con otros estudios que concluyen que las plantaciones de árboles de rápido crecimiento afectan a los recursos hídricos, que constituyen un servicio ecosistémico importante en el contexto del calentamiento de la Tierra. Con vistas a minimizar los efectos de la silvicultura industrial, se sugiere que mantener o recuperar bosques ribereños podría mitigar las repercusiones de los monocultivos intensivos de eucaliptos.

Palabras clave: *Eucalyptus globulus*, Biodiversidad, Ecosistemas fluviales, Usos de suelo, Bosque, Plantaciones de árboles

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Introduction

Rivers are closely related to the terrestrial ecosystems of their drainage basins, to the point that modifications in land use patterns determine changes in the physico–chemical properties of rivers, and as a consequence, in their biotic structure (Margalef, 1983; Allan, 2004a; Sandin, 2009). In effect, land use changes are an integrator of many human activities that have a negative impact on stream ecosystems (Allan & Castillo, 2007). Degradation of stream ecosystems derived from modifications in land use patterns is likely manifested in changes to flows and water temperature, bank erosion and silt deposition, thus affecting benthic habitat conditions (Hickey & Doran, 2004). In regions that naturally have riparian vegetation, trees strongly influence energy pathways by changing the availability of light and input of organic matter (Gregory et al., 1991).

The impact of invasive exotic species on native communities is widely known, and is considered an important component of global change (Sakai et al., 2001), particularly in relation to the loss of biodiversity (Calviño–Cancela et al., 2012, 2013; Calviño–Cancela, 2013). Furthermore, the management of invasive species is one of the greatest challenges facing conservation in Europe in this millennium (Genovesi & Shine, 2003). In the Iberian peninsula, vast areas of potentially deciduous forest have been colonised by plantations of the Australian tree *Eucalyptus globulus* Labill, which is clearly invasive in the North of Iberia (Dana et al., 2003; Calviño–Cancela & Rubido–Bará, 2013). Other species, such as Australian beetles (Cordero Rivera et al., 1999) and fungi (Díez, 2005), become invasive through their association with eucalypts. The spread of *Eucalyptus* is partially explained by the economic benefits of fast–growing tree plantations whose wood is used by the paper industry (see review in Canhoto et al., 2004), but also because, as a pyrophytic species, eucalypt are favoured by wildfires (Gutián Rivera & Cordero–Rivera, 2007).

In small forested streams in which food–webs are based on detrital inputs from surrounding forest (Wallace et al., 1997), modifications in watershed vegetation also alter the quantity, quality and seasonality of leaf–litter inputs (Abelho & Graça, 1996; Molinero & Pozo, 2004). This variation in detritus quality may influence the colonisation and activity of decomposers (Kearns & Bärlocher, 2008). In particular, the replacement of native mixed deciduous forests by evergreen monospecific plantations of eucalypts is known to produce structural and functional modifications on river ecosystems (Graça et al., 2002). *Eucalyptus* leaves are a resource of lower quality for aquatic organisms than those of native species, such as *Alnus glutinosa* (Canhoto & Graça, 1995; Santiago et al., 2011), with a lower amount of nitrogenous and phosphorous and a higher quantity of compounds of difficult degradation (lignin, oils, tannins and other phenolic compounds) (Canhoto & Graça, 1996; Molinero & Pozo, 2004). The substitution of native riparian vegetation by eucalypt plantations also determines changes in light and temperature regimes, as well as alterations of the substrate and habitats, due to the frequent deposition

of large particulate organic matter and soil particles (Graça et al., 2002). These changes also increase soil hydrophobicity. The presence of eucalypt plantations in riverine habitats thus modifies hydrologic regimes, particularly when clear cutting operations take place in the basin (Fernández et al., 2006). The resultant hydrophobicity affects the infiltration rate of water in the soil, increasing surface run–off, promoting erosion, and diminishing subterranean water reservoirs. Another side effect of this hydrophobicity is the possible summer dry out of streams whose basins are covered by eucalypt plantations (Graça et al., 2002; Jackson et al., 2005).

Given the above considerations, it is surprising that empirical data on the effects of eucalypt plantations on native biota are scarce (but see Calviño–Cancela & Neumann, 2015), particularly in small streams, which are particularly relevant to maintain biodiversity (Finn et al., 2011). The main goal of this study was to test whether the observed effect of *Eucalyptus* plantations on stream macroinvertebrate communities are generalizable to the situation in NW Spain, where the numbers of eucalypt plantations have increased dramatically in the last decades (Cordero–Rivera, 2012), and where the eucalypts clearly show invasive behaviour after fire (Gutián Rivera & Cordero–Rivera, 2007). We analysed the effect of land use patterns, and particularly the effect of eucalypt plantations, on the macroinvertebrate communities of headwater streams. We hypothesized that streams whose basins were mainly covered by eucalypt plantations would have less richness and diversity and would more likely dry out in summer than comparable streams running through native riparian forests.

Material and methods

Study area

For this study, we sampled 16 stream tributaries of the Lézé River in Pontevedra province (NW Spain) (fig. 1) in 2011. Streams were selected for their accessibility, size and land use patterns, covering the variability of land uses and vegetation types observed in the drainage (table 1). The average stream basin size is 2.66 ± 0.86 km², ranging between 0.05 km² in A Ceira and 12.93 km² in Os Calvos (table 1). In agreement with the granitic geology of the region, water is slightly acid (mean \pm SE pH = 6.01 ± 0.01) and has a low ionic content (mean conductivity = 66.58 ± 7.46 μ S/cm; table 1; see also Membiela et al., 1991, for a review of typical values of rivers of the region). Most of the streams are on a southern slope (fig. 1), and thus have less flow during summer months, with some remaining as isolated pools for several weeks. To avoid the dry season, we performed the sampling at the beginning of spring (March–April) and the end of spring (May).

Sampling methodology

Macroinvertebrates were collected using a Surber net with a mesh size of 250 μ m and a sampling area of 0.1 m². In one of the streams, As Pozas (table 1),

Table 1. Land uses and physical–chemical characteristics of the 16 streams included in this study. See figure 1 for their location: WT. Water temperature (°C); WC. Water conductivity ($\mu\text{S}/\text{cm}$). Ba. Basin area (km^2). Land uses: F. Native forest; S. Shrubland; E. Eucalypt plantations; Ag. Agricultural; U. Urban.

Tabla 1. Usos del suelo y características físicoquímicas de los 16 arroyos incluidos en este estudio. Véase la figura 1 para conocer su ubicación: WT. Temperatura del agua (°C); WC. Conductividad del agua ($\mu\text{S}/\text{cm}$); Ba. Superficie de la cuenca (km^2). Usos del suelo: F. Bosque autóctono; S. Matorral; E. Eucaliptal; Ag. Zona agrícola; U. Zona urbana.

Stream	pH	WT	WC	BA	Land use cover (%)					Predominant land use category
					F	S	E	Ag	U	
A Ceira	5.6	13.2	34.9	0.05	33.99	66.01	0.00	0.00	0.00	Native forest–shrubs
As Pozas	4.9	14.6	106.9	0.22	0.00	25.75	74.25	0.00	0.00	Eucalypt–shrub
Barranq. de Lixó	6.3	14.0	35.8	0.28	0.00	44.26	55.74	0.00	0.00	Eucalypt–shrub
Rego 1	6.3	17.2	78.8	0.38	1.29	53.80	44.90	0.00	0.00	Eucalypt–shrub
As Penizas	6.4	17.4	95.4	0.39	0.00	54.71	45.29	0.00	0.00	Eucalypt–shrub
As Ladeiras	5.9	13.8	28.9	0.76	88.28	11.72	0.00	0.00	0.00	Native forest
Grande 1	5.7	15.8	55.3	1.26	12.18	29.02	50.10	5.93	2.77	Eucalypt
Fonte Seixiña	5.7	14.3	32.8	1.55	18.15	67.64	0.00	12.85	1.36	Shrubs
O Cambado	6.3	15.1	57.2	1.56	13.04	70.99	10.48	5.49	0.00	Shrubs
Os Ladróns	6.5	14.5	66.1	1.69	45.09	9.36	3.54	33.33	8.68	Native forest–Agricultural
O Moído	5.9	14.0	65.9	2.12	25.81	26.95	20.54	23.38	3.32	Mixed
Acevedo	5.7	14.6	116.7	2.91	4.13	45.06	0.00	50.82	0.00	Agricultural–shrubs
Grande 2	6.1	17.0	27.7	2.96	7.00	22.44	60.57	8.77	1.22	Eucalypt
Os Cabaleiros	6.2	13.9	97.5	6.02	49.19	50.81	0.00	0.00	0.00	Native forest–shrubs
Os Maneses	6.4	13.7	98.8	7.44	57.52	20.64	9.87	9.40	2.57	Native forest
Os Calvos	6.3	14.1	66.6	12.93	30.96	40.19	15.75	12.10	1.00	Native forest–shrubs

Data analysis

In each sampling period (early and late spring), we studied the structure of communities in each stream by calculating the number of Families (a surrogate for richness), the number of individuals (Abundance), and diversity (Shannon index). In the analyses, we used the average values of both sampling periods and the number of families per stream. This allowed inclusion of the stream Barranqueira de Lixó, which was completely dry during the second sampling. The diversity index was calculated using a Box–Cox transformation to meet the assumptions of normality.

This study was a sample survey, and strictly speaking, there were no treatments (Shaffer & Johnson, 2008), because we could not manipulate basin cover or size of our streams. We nevertheless calculated statistical relationships to test a priori ideas derived from ecological theory, which is a powerful way to identify possible cause–effect relationships. To explore the relationship between the variables describing macroinvertebrate communities and the variables describing land uses and river characteristics (*i.e.*

proportion of catchment area covered by each land use, basin size, pH and water temperature and conductivity), first, we used standard Pearson correlation analysis. Proportions (p) were transformed before the analysis ($\arcsin \sqrt{p}$).

The response variables were analysed by GLM assuming normal errors and identity link (diversity) or Poisson with log link (number of families). We used the Akaike Information Criterion to control over–fitting in statistical modelling and thus avoid the use of frequentist methods and their associated statistical tests, which may be misleading in observational studies (Burnham et al., 2011). Agricultural and urban land cover was low in all streams, and no correlation between these variables and any measure of community structure was detected in exploratory analyses (see table 2). Land cover types made up 100%; therefore, to avoid problems of multicollinearity we included only forest and eucalypt land cover, and basin size in the models, and no interactions were fitted because we had only 16 streams. Analyses were done using xlStat2013 (www.xlstat.com) and Genstat 18th edition (GenStat, 2015).

Table 2. Pearson correlations between variables (below the diagonal) and the p -values (above the diagonal). Diversity was Box–Cox transformed and the angular transformation was applied to all land cover variables to meet the assumptions of normality: T. Temperature; C. Conductivity; B. Basin size; Ab. Abundance; F. Families; D. Diversity; F. Forest; S. Shrubland; E. Eucalypts; Ag. Agricultural; U. Urban. (Values in bold are different from 0 with a significance level $\alpha = 0.05$.)

Tabla 2. Correlaciones de Pearson entre las variables (debajo de la diagonal) y valores de probabilidad asociados (encima de la diagonal). Para cumplir los supuestos de normalidad, la diversidad se transformó mediante el método Box–Cox y se aplicó la transformación angular a todas las variables de cobertura del territorio: T. Temperatura; C. Conductividad; B. Tamaño de la Cuenca; Ab. Abundancia; F. Familias; D. Diversidad; F. Bosque; S. Matorral; E. Eucaliptal; Ag. Agrícola; U. Urbano. (Los valores en negrita son distintos de cero con un nivel de significación $\alpha = 0,05$.)

Variables	pH	T	C	B	Ab	Fa	D	Fo	S	E	Ag	U
pH		0.422	0.825	0.206	0.015	0.823	0.278	0.497	0.971	0.743	0.779	0.386
T	0.216		0.738	0.317	0.306	0.120	0.278	0.018	0.732	0.014	0.639	0.752
C	-0.060	0.091		0.385	0.164	0.626	0.428	0.395	0.950	0.835	0.510	0.647
B	0.334	-0.267	0.233		0.724	0.049	0.019	0.183	0.697	0.463	0.306	0.405
Ab	-0.597	-0.273	0.366	0.096		0.083	0.962	0.742	0.901	0.806	0.332	0.935
Fa	0.061	-0.405	-0.132	0.499	0.447		0.000	0.041	0.768	0.018	0.065	0.141
D	0.289	-0.289	-0.213	0.579	-0.013	0.777		0.005	0.648	0.093	0.236	0.096
Fo	0.183	-0.581	-0.229	0.351	-0.089	0.515	0.664		0.133	0.003	0.728	0.246
S	-0.010	0.093	-0.017	-0.106	-0.034	0.080	-0.124	-0.393		0.536	0.375	0.026
E	-0.089	0.598	0.057	-0.198	-0.067	-0.583	-0.435	-0.697	-0.167		0.244	0.917
Ag	0.076	-0.127	0.178	0.273	0.259	0.472	0.314	0.095	-0.238	-0.309		0.013
U	0.233	-0.086	-0.124	0.223	0.022	0.385	0.430	0.308	-0.552	-0.028	0.603	

Finally, to test whether predominant land uses were related to changes in the structure of macroinvertebrate communities, we calculated two ordinations with non-metric multidimensional scaling (MDS). The first ordination was calculated using the average density of macroinvertebrates over the two sampling periods, and the second ordination was calculated using presence/absence data. These ordinations, obtained using the similarity matrix based on the Bray–Curtis and Jaccard index, respectively. They allow a visual representation of the relationship between land use categories and macroinvertebrate communities of each stream, which were compared using an index of similarity (SIMPER, similarity percentages). This procedure examines the contribution of each family of macroinvertebrates and identifies the average similarity and dissimilarity between two groups of samples (land use categories). This analysis is therefore restricted to land use categories with at least two streams (*i.e.* forest, shrubland and eucalypt plantations). MDS and SIMPER analyses were performed using PRIMER v.6 (Clarke & Warwick, 2001). Means are presented with their standard errors and sample size (mean \pm SE (N)).

Results

We found a total of 56 families of macroinvertebrates in the 16 streams. Table 2 shows the pair-wise correlations between variables. In agreement with ecological theory, the diversity of macroinvertebrates was positively correlated with catchment area ($R = 0.58$, $P = 0.019$), but up to a limit, thus describing a power function (fig. 2A). Diversity also increased with the proportion of the basin covered by native forest ($R = 0.66$, $P = 0.005$), but in this case in the form of a hump-shaped curve (fig. 2B). As expected given that native forest and eucalypt plantations are the main land uses, these two categories were negatively correlated ($r = -0.70$; $P = 0.003$). Therefore, as the eucalypt plantation cover increased, both macroinvertebrate diversity ($R = -0.43$, $P = 0.093$) and richness ($R = -0.58$, $P = 0.018$) were negatively affected, although only the latter value was significant, and both seem to follow a non-linear trend (fig. 3). The proportion of shrubland, agricultural and urban land on the catchment did not show any relationship with macroinvertebrate diversity or richness (table 1).

The abundance of macroinvertebrates showed a negative correlation with pH ($R = -0.60$; $P = 0.015$;

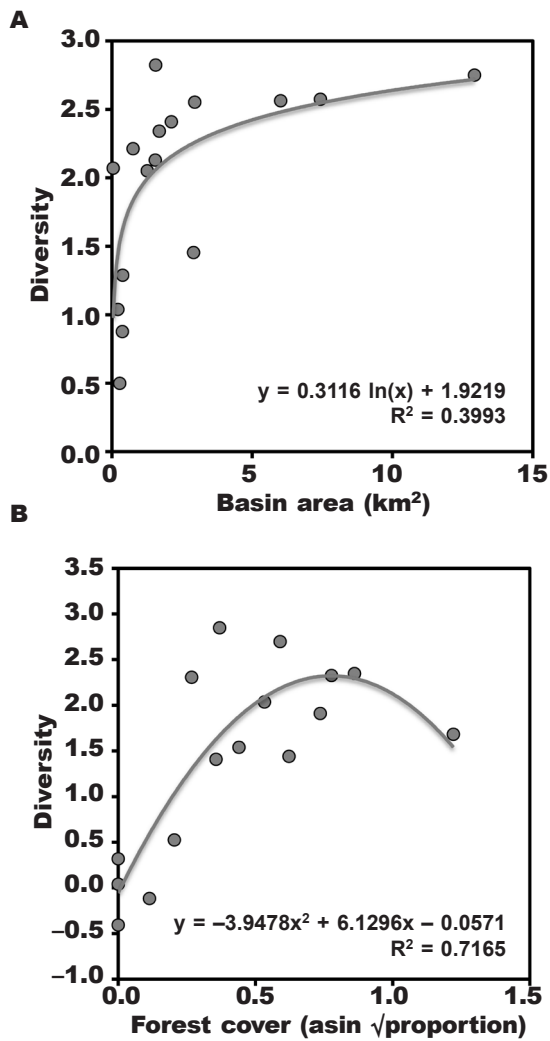


Fig. 2. The relationship between basin size (A) and the proportion of native forest in the basin (B) and the average macroinvertebrate diversity (Shannon index, Box–Cox transformed) found in 16 streams of the River Lérez basin. Equations refer to the adjusted curves.

Fig. 2. Relación entre el tamaño de la cuenca (A) y la proporción de bosque nativo en la cuenca (B) y la diversidad media (índice de Shannon, con la transformación Box–Cox) de los macroinvertebrados hallados en 16 arroyos de la cuenca del río Lérez. Las ecuaciones se refieren a las curvas ajustadas.

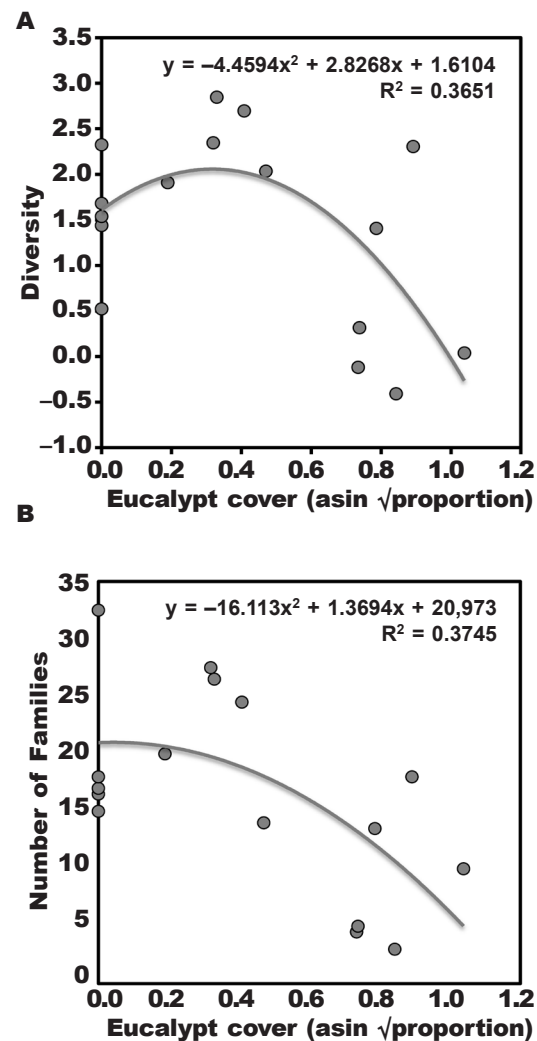


Fig. 3. The relationship between the proportion of eucalypt plantations in the basin and the average macroinvertebrate diversity (Shannon index, Box–Cox transformed) (A) and richness (number of Families) (B) of macroinvertebrate communities found on 16 streams of the River Lérez basin.

Fig. 3. Relación entre la proporción de la cuenca ocupada por plantaciones de eucalipto y la diversidad media (índice de Shannon, con la transformación Box–Cox) (A) y la riqueza (número de Familias) (B) de las comunidades de macroinvertebrados encontradas en 16 arroyos de la cuenca del río Lérez.

table 2). Nevertheless, this relationship was due to the extraordinary density of Asellidae in the source of the most acid stream, As Pozas. Excluding this datapoint, a clear outlier, the relationship was not significant ($R = -0.23$, $P = 0.402$). No other significant correlation was found between abundance and the explanatory environmental variables.

Water temperature was positively correlated with the proportion of catchment area covered by eucalypt plantations ($R = 0.60$; $P = 0.014$) (fig. 4A). We observed an opposite pattern for the native forests ($R = -0.58$; $P = 0.018$) (fig. 4B). Mean water temperature for the 6 streams included under the eucalypt and eucalypt–shrub categories was approximately 2°C

higher than the temperature measured in the 6 streams with highest forest cover ($16.0 \pm 0.6^\circ\text{C}$ and $13.8 \pm 0.2^\circ\text{C}$, respectively) (table 1). In agreement with this observation, the streams with a large eucalypt cover showed dramatic changes in water flow between the two sampling periods. One of them, Barranqueira de Lixó, completely dried out on the second sampling date, and another, As Pozas, had water remaining only at the source, forming a single pool. In contrast, the smallest stream, A Ceira, which had mainly native forest in its catchment area, had a similar flow on sampling days in both periods. The saturated model (including all three explanatory variables; *i.e.* basin size, eucalypt and forest cover) explained 57.8% of variance, and was therefore a good starting model (table 3). The most supported model to explain variability in macroinvertebrate diversity included basin size and forest cover, both with a positive effect, but with a much larger effect size for forest cover (table 3). Two other models were within 2 units of AIC, including the saturated model, suggesting that all three variables are of relevance. The saturated model explains 45.6% of variance in richness (number of Families; table 3).

The most supported model to describe variability in richness includes basin size and eucalypt cover, the former having a positive effect and the latter having a negative effect. The next models, again close to the first, included only eucalypt cover (explaining alone 31.5% of variance) and all three variables (table 3), suggesting that all variables are of relevance. Multivariate ordinations also supported the observed effect of land use and catchment size on macroinvertebrate communities. Basins with eucalypt plantations (and a percentage of the stream drainage covered by native forest below 20%), were the most differentiated communities (fig. 5). Based on SIMPER analysis, the proportion of similarity between the macroinvertebrate communities of catchments dominated by native forests and by eucalypt plantations was only 33.3%. Furthermore, we observed that the macroinvertebrates that are typical of streams with large areas of native forest belong to several families of mayflies (*e.g.*, Heptageniidae), stoneflies (*e.g.*, Chloroperlidae) and caddisflies (Sericostrimatidae, Hydropsychidae, Brachycentridae), taxa that are rare on the streams dominated by eucalypt plantations, where filter feeders simuliids and limoniids were the most characteristic taxa.

Discussion

Our results show that the structure of macroinvertebrate communities follows the expected trends derived from ecological theory: higher complexity relates positively to basin size and native forest cover and negatively to eucalypt cover (figs. 2, 3). In a general sense, running waters are one of the most impacted ecosystems on the planet as they have been the focus for human settlement and are heavily exploited for water supplies, irrigation, electricity generation, and waste disposal (Malmqvist & Rundle, 2002). But besides these direct anthropogenic effects, streams are also beginning to show the pressure of global climate change through

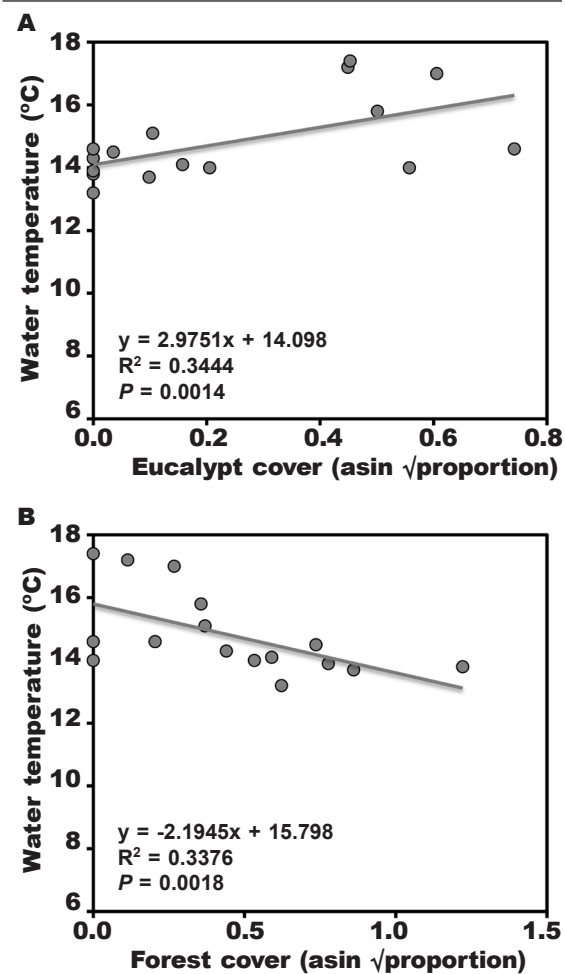


Fig. 4. The relationship between the proportions of basin covered by eucalypts (A) and native forest (B) and water temperature in 16 streams of the River Lézé basin.

Fig. 4. Relación entre la proporción de la cuenca ocupada por plantaciones de eucalipto (A) y por bosque nativo (B) y la temperatura del agua en 16 arroyos de la cuenca del río Lézé.

alterations in hydrology, thermal regimes and riparian vegetation (Meyer & Pulliam, 1992), which will directly affect the quantity and quality of the leaf litter that is the source of the detritus-based food webs. Within this context, the most relevant result of this study is the positive association between the proportion of the basin covered by native forests and macroinvertebrate diversity. This relation seems non-linear (fig. 2B), and a close scrutiny suggests that even low values of 15–20% of forest cover in the basin might provide enough resources for stream macroinvertebrate communities. The take-home message is clear: if forests cover more than one third of the basin, stream communities are expected to retain high diversity, particularly if riparian vegetation remains intact.

Table 3. Summary of model selection, with the estimated effect, SE and Akaike Information Criterion (AIC). The explanatory variables were the area of the catchment (BASIN), the percentage of the catchment covered by eucalypts (EUCALYPT) and native forests (FOREST). Models are ordered by increasing AIC. R^2 refers to the proportion of variance explained.

Tabla 3. Resumen de la selección de modelos, con el efecto estimado, la desviación estándar (SE) y el criterio de información de Akaike (AIC). Las variables explicativas fueron el área de la cuenca (BASIN) y el porcentaje de la cuenca cubierto por eucaliptos (EUCALYPT) y por bosques autóctonos (FOREST). Los modelos están ordenados de menor a mayor AIC. R^2 indica la proporción de varianza explicada.

Response variable					
Model	Estimate	SE	AIC	Δ AIC	R^2
Diversity					
BASIN + FOREST	0.120	0.059	18.006	0.000	57.79
	1.574	0.576			
FOREST	1.988	0.598	19.892	1.886	44.13
	0.120	0.061	20.000	1.994	57.81
	0.057	0.739			
	1.617	0.822			
EUCALYPT + FOREST	0.158	0.814	21.846	3.840	44.29
	2.105	0.864			
BASIN + EUCALYPT	0.156	0.064	21.874	3.868	44.20
	-0.938	0.595			
BASIN	0.176	0.066	22.907	4.901	33.53
EUCALYPT	-1.224	0.678	27.075	9.069	18.88
Number of families (richness)					
BASIN + EUCALYPT	0.053	0.028	18.056	0.000	45.35
	-0.793	0.326			
EUCALYPT	-0.839	0.034	19.110	1.054	31.50
BASIN + EUCALYPT + FOREST	0.051	0.030	20.000	1.944	45.60
	-0.731	0.431			
	0.104	0.443			
EUCALYPT + FOREST	-0.659	0.453	20.648	2.592	33.60
	0.283	0.442			
BASIN + FOREST	0.045	0.031	20.795	2.739	32.93
	0.592	0.360			
FOREST	0.713	0.034	20.963	2.907	23.10
BASIN	0.061	0.031	21.736	3.680	19.60

Eucalypt plantations are the dominant vegetation in large areas of the river Lézé basin, and they are the main land use near many small rivers. As previously shown (see for example Larrañaga et al., 2009a), we observed that streams flowing through eucalypt plantations had lower taxon richness and less diversity of macroinvertebrates than those flowing through native forests (fig. 3). In fact, the percentage of the stream drainage covered by forest was negatively correlated

with eucalypt plantations cover. Moreover, as observed in our study, previous studies in Spain and Portugal on the effects of *Eucalyptus* plantations in streams have shown changes in macroinvertebrate communities (Abelho & Graça, 1996; Larrañaga et al., 2009b).

More precisely, although we did not analyze changes on feeding groups, we observed that the trophic structure of streams under native vegetation versus altered sites pointed to a change from shredder

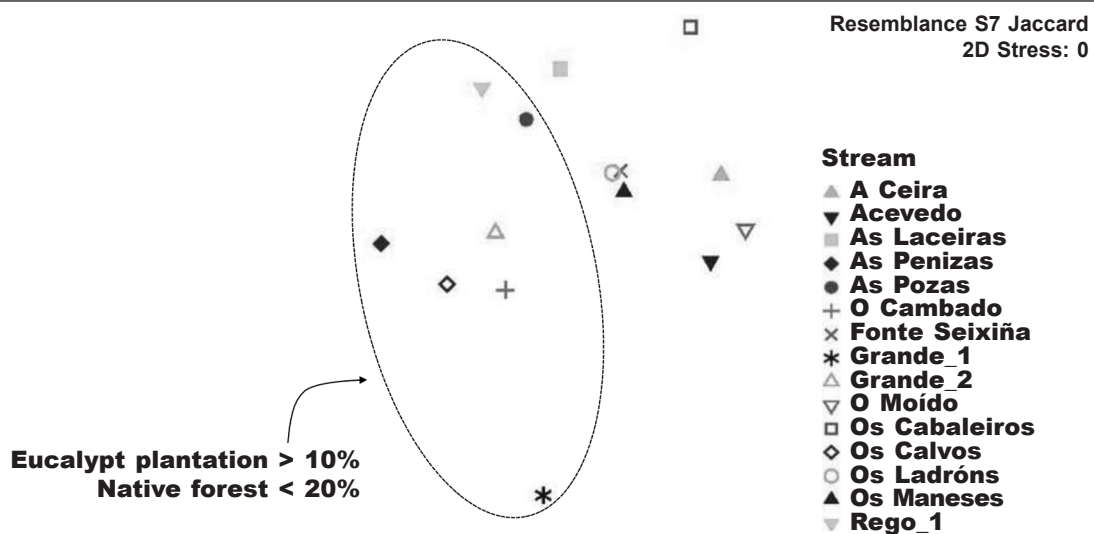


Fig. 5. Bidimensional ordination (MDS) of all the sampled streams using presence/absence data of the macroinvertebrate community.

Fig. 5. Ordenación bidimensional (MDS) de todos los arroyos muestreados usando datos de presencia y ausencia de la comunidad de macroinvertebrados.

dominance (relying on native litter–fall inputs, such as the caddisflies of the families Sericostomatidae and Brachycentridae) to collector dominance (detritivores of fine organic matter, such as simuliids or limonids) at sites under *Eucalyptus* plantations. Álvarez et al. (2001) also observed comparable tendencies in temporary Mediterranean streams. Similarly, in a litter exclusion experimental study assessing the impact of eucalypt plantations on benthic macroinvertebrate communities, Larrañaga et al. (2006) found that shredders were less abundant at sites with eucalypt leaves than at sites with leaves collected from deciduous forests.

Besides the potential effect of changes in vegetation inputs on stream biota, land use changes have a direct effect on hydrology through their links with the evapotranspiration regime. Temperature is also of major importance for poikilothermic aquatic organisms due to its effects on physiology and behaviour. As shown elsewhere (see for example Allan, 2004b), the results of our study suggest that land use patterns also affect stream water temperature. In effect, the streams with a larger proportion of their drainage basin covered by forests were colder than those dominated by eucalypt plantations, probably as a result of the reduced flow and higher evaporation rates of the latter (Ferreira et al., 2006), and the decrease in shading due to the vertical leaf orientation of eucalypt (James & Bell, 2000). In agreement with the general hydrological consequences of fast-growing tree plantations (see for example Jackson et al., 2005; Oyarzún et al., 2005), we observed that in the second sampling period, water flow was reduced or had ceased altogether in some streams running through basins with a high proportion of eucalypt

trees. However, given the small number of streams surveyed in this study, our data are not conclusive. Nevertheless, they agree with numerous examples in other regions. For instance, in the pampas grasslands of Argentina, Engel et al. (2005) observed that plantations of *Eucalyptus camaldulensis* (a phreatophytic species commonly grown in the region) often caused localized drawdowns of water tables, as a consequence of the species transpiration demand during dry periods. In China, in *r Eucalyptus* and *Pinus* plantations, it has been shown that erosion increased and water storage diminished in comparison with native vegetation (Hou et al., 2010). Similarly, a global analysis of 504 annual catchment observations (Jackson et al., 2005) showed that afforestation of grasslands, shrublands, or croplands with eucalypt and *Pinus* plantations decreased stream flow by 52% per year (227 mm) and that 13% of streams dried up for at least one year. In NW Spain, large areas have been deforested for centuries (Gutián Rivera & Cordero–Rivera, 2007) and even if eucalypt plantations do not usually substitute native forests that disappeared long time ago (but see Teixeira et al., 2010), eucalypts have been introduced over shrub vegetation (mainly *Ulex* sp. and *Erica* sp.) and have spread invasively (Calviño–Cancela & Rubido–Bará, 2013), particularly after wildfires (Gutián Rivera & Cordero–Rivera, 2007). Therefore, in a climate change scenario, eucalypt plantations can be detrimental to water availability and rapidly accelerate drying of wet soils (Montoya, 1995). Another finding of interest is that in the same region of NW Spain, when eucalypts are defoliated by beetles, annual stream water can increase by 22% (Fernández et al., 2006).

Besides the effect of changes in basin land uses, our results also indicate that macroinvertebrate communities were affected by basin size and land use patterns. Large areas are known to harbour more species due to the increase in resources and diversity of habitats (Shafer, 1990). A similar relationship usually occurs for river systems (Margalef, 1983). Accordingly, we observed that macroinvertebrate diversity increased with basin size, probably as the result of a combination of higher physical diversity (microhabitats) and more resources (more species of riverine trees, greater complexity of shoreline vegetation, and so on). Nevertheless, as shown in figure 2A, even streams draining very small basins, around 2–3 km², were able to harbour most of the families of macroinvertebrates of the regional fauna. This is in agreement with recent studies showing the relevance of headwaters for riverine biodiversity conservation (Finn et al., 2011).

Our findings add evidence to previous studies that concluded that fast-growing tree plantations affect hydric resources and support the need to maintain and/or restore riparian forests to minimize the impacts of intensive industrial silviculture on aquatic communities. In conclusion, tree plantations cannot be used as substitutes of all ecosystem properties of forests, especially when plantations are established with exotic species (Cordero–Rivera, 2011, 2012).

References

- Abelho, M. & Graça, M. A. S., 1996. Effects of *Eucalyptus* afforestation on leaf litter dynamics and macroinvertebrate community structure of streams in Central Portugal. *Hydrobiologia*, 324: 195–204.
- Allan, J. D., 2004a. Influence of land use and landscape setting on the ecological status of rivers. *Limnética*, 23: 187–198.
- 2004b. Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annual Review of Ecology, Evolution, and Systematics*, 35: 257–284.
- Allan, J. D. & Castillo, M. M., 2007. *Stream ecology: structure and function of running waters*. 2nd edition. Chapman and Hall, New York.
- Álvarez, M., Pardo, I., Moyá, G., Ramón, G. & Martínez–Taberner, A., 2001. Invertebrate communities in temporary streams of the island of Majorca: A comparison of catchments with different land use. *Limnética*, 20: 255–266.
- Burnham, K. P., Anderson, D. R. & Huyvaert, K. P., 2011. AIC model selection and multimodel inference in behavioral ecology: some background, observations, and comparisons. *Behavioral Ecology and Sociobiology*, 65: 23–35.
- Calviño–Cancela, M., 2013. Effectiveness of eucalypt plantations as a surrogate habitat for birds. *Forest Ecology and Management*, 310: 692–699.
- Calviño–Cancela, M., López de Silanes, M.E., Rubido–Bará, M. & Uribarri, J., 2013. The potential role of tree plantations in providing habitat for lichen epiphytes. *Forest Ecology and Management*, 291: 386–395.
- Calviño–Cancela, M. & Neumann, M., 2015. Ecological integration of eucalypts in Europe: Interactions with flower-visiting birds. *Forest Ecology and Management*, 358: 174–179.
- Calviño–Cancela, M. & Rubido–Bará, M., 2013. Invasive potential of *Eucalyptus globulus*: Seed dispersal, seedling recruitment and survival in habitats surrounding plantations. *Forest Ecology and Management*, 305: 129–137.
- Calviño–Cancela, M., Rubido–Bará, M. & Etten, E. J. B. van., 2012. Do eucalypt plantations provide habitat for native forest biodiversity? *Forest Ecology and Management*, 270: 153–162.
- Canhoto, C., Abelho, M. & Graça, M. A., 2004. Efeitos das plantações de *Eucalyptus globulus* nos ribeiros de Portugal. *Recursos Hídricos*, 25: 59–65.
- Canhoto, C. & Graça, M. A., 1995. Food value of introduced eucalypt leaves for a Mediterranean stream detritivore: *Tipula lateralis*. *Freshwater Biology*, 34: 209–214.
- 1996. Decomposition of *Eucalyptus globulus* leaves and three native leaf species (*Alnus glutinosa*, *Castanea sativa* and *Quercus faginea*) in a Portuguese low order stream. *Hydrobiologia*, 333: 79–85.
- Clarke, K. R. & Gorley, R. N., 2006. *PRIMER v6: user manual–tutorial*. Plymouth Marine Laboratory, Plymouth.
- Cordero–Rivera, A., 2011. Cuando los árboles no dejan ver el bosque: efectos de los monocultivos forestales en la conservación de la biodiversidad. *Acta Biológica Colombiana*, 16: 247–268.
- 2012. Bosques e plantaciones forestais: dois ecossistemas claramente diferentes. *Recursos Rurais Serie Cursos*, 6: 7–17.
- Cordero Rivera, A., Santolamazza–Carbone, S. & Andrés, J. A., 1999. Life cycle and biological control of the *Eucalyptus* snout beetle (Coleoptera, Curculionidae) by *Anaphes nitens* (Hymenoptera, Mymaridae) in north–west Spain. *Agricultural and Forest Entomology*, 1: 103–109.
- Dana, E., Sobrino, E. & Sanz–Elorza, M., 2003. Plantas invasoras en España: un nuevo problema en las estrategias de conservación. In: *Atlas y Libro Rojo de la flora vascular amenazada de España: 1010–1029* (Á. Bañares, G. Blanca, J. Güemes, J. C. Moreno & S. Ortiz, Eds.). Dirección Nacional de Conservación de la Naturaleza. Ministerio de Medio Ambiente, Madrid.
- Díez, J., 2005. Invasion biology of Australian ectomycorrhizal fungi introduced with eucalypt plantations into the Iberian Peninsula. *Biological Invasions*, 7: 3–15.
- Engel, V., Jobbágy, E. G., Stieglitz, M., Williams, M. & Jackson, R. B., 2005. Hydrological consequences of *Eucalyptus* afforestation in the Argentine Pampas. *Water Resources Research*, 41: 1–14.
- Fernández, C., Vega, J. A., Gras, J. M. & Fonturbel, T., 2006. Changes in water yield after a sequence of perturbations and forest management practices in an *Eucalyptus globulus* Labill. watershed in Northern Spain. *Forest Ecology and Management*, 234: 275–281.
- Ferreira, V., Eloşegi, A., Gulis, V., Pozo, J. & Graça, M. A. S., 2006. *Eucalyptus* plantations affect fungal communities associated with leaf–litter decomposition in Iberian streams. *Archiv fur Hydrobiologie*,

- 166: 467–490.
- Finn, D. S., Bonada, N., Múrria, C. & Hughes, J. M., 2011. Small but mighty: headwaters are vital to stream network biodiversity at two levels of organization. *Journal of the North American Benthological Society*, 30: 963–980.
- Genovesi, P. & Shine, C., 2003. *European strategy on invasive alien species. Convention on the Conservation of European Wildlife and Natural Habitats*. Council of Europe, Strasbourg.
- GenStat, 2015. *GenStat for Windows*, 18th Edition. VSN International Ltd., Oxford.
- Graça, M. A., Pozo, J., Canhoto, C. & Elozegi, A., 2002. Effects of *Eucalyptus* plantations on detritus, decomposers, and detritivores in streams. *The Scientific World*, 2: 1173–1185.
- Gregory, S. V., Swanson, F. J., McKee, W. A. & Cummins, K. W., 1991. An ecosystem perspective of riparian zones. Focus on links between land and water. *BioScience*, 41: 540–551.
- Gutián Rivera, L. & Cordero-Rivera, A., 2007. Bosques e plantacións forestais. In: *Proxecto Galicia, Ecoloxía*, vol. XLIV: 430–467 (A. Cordero Rivera, Ed.). Hércules de Ediciones, A Coruña.
- Hickey, M. B. C. & Doran, B., 2004. A review of the efficiency of buffer strips for the maintenance and enhancement of riparian ecosystems. *Water Quality Research Journal of Canada*, 39: 311–317.
- Hou, X., Duan, C., Tang, C. Q. & Fu, D., 2010. Nutrient relocation, hydrological functions, and soil chemistry in plantations as compared to natural forests in central Yunnan, China. *Ecological Research*, 25: 139–148.
- Jackson, R. B., Jobbagy, E. G., Avissar, R., Roy, S. B., Barrett, D. J., Cook, C. W., Farley, K. A., Maire, D. C., McCarl, B. A. & Murray, B. C., 2005. Trading water for Carbon with biological Carbon sequestration. *Science*, 310: 1944–1947.
- James, S. A. & Bell, D. T., 2000. Leaf orientation, light interception and stomatal conductance of *Eucalyptus globulus* ssp. *globulus* leaves. *Tree Physiology*, 20: 815–823.
- Kearns, S. G. & Bärlocher, F., 2008. Leaf surface roughness influences colonization success of aquatic hyphomycete conidia. *Fungal Ecology*, 1: 13–18.
- Larrañaga, A., Basaguren, A., Elozegi, A. & Pozo, J., 2009a. Impacts of *Eucalyptus globulus* plantations on Atlantic streams: changes in invertebrate density and shredder traits. *Fundamental and Applied Limnology / Archiv für Hydrobiologie*, 175: 151–160.
- Larrañaga, A., Basaguren, A. & Pozo, J., 2009b. Impacts of *Eucalyptus globulus* plantations on physiology and population densities of invertebrates inhabiting Iberian Atlantic streams. *International Review of Hydrobiology*, 94: 497–511.
- Larrañaga, A., Larrañaga, S., Basaguren, A., Elozegi, A. & Pozo, J., 2006. Assessing impact of *Eucalyptus* plantations on benthic macroinvertebrate communities by a litter exclusion experiment. *Annales de Limnologie-International Journal of Limnology*, 42: 1–8.
- Malmqvist, B. & Rundle, S., 2002. Threats to the running water ecosystems of the world. *Environmental Conservation*, 29: 134–153.
- Margalef, R., 1983. *Limnología*. Omega, Barcelona.
- Membiola, P., Montes, C. & Martínez Ansemil, E., 1991. Características hidroquímicas de los ríos de Galicia (NW península Ibérica). *Limnética*, 7: 163–174.
- Meyer, J. L. & Pulliam, W. M., 1992. Modifications of terrestrial-aquatic interactions by a changing climate. In: *Global climate change and freshwater ecosystems: 177–191* (P. Firth & S. G. Fisher, Eds.). Springer-Verlag, New York.
- Molinero, J. & Pozo, J., 2004. Impact of *Eucalyptus* plantation on the nutrient content and dynamics of coarse particulate organic matter in a small stream. *Hydrobiologia*, 528: 143–165.
- Montoya, J. M., 1995. *El eucalipto*. Mundi-Prensa, Madrid.
- Oyarzún, C. E., Nahuelhual, L. & Núñez, D., 2005. Los servicios ecosistémicos del bosque templado lluvioso: producción de agua y su valoración económica. *Ambiente y Desarrollo*, 20–21: 88–95.
- Río Barja, F. J. & Rodríguez Lestegás, F., 1992. *Os ríos galegos. Morfoloxía e réxime*. Consello da Cultura Galega, Santiago de Compostela.
- Sakai, A. K., Allendorf, F. W., Holt, J. S., Lodge, D. M., Molofsky, J., With, K. A., Baughman, S., Cabib, R. J., Cohen, J. E., Ellstrand, N. C., Mccauley, D. E., O'Neill, P. O., Parker, I. M., Thompson, J. N. & Wemmer, C., 2001. The population biology of invasive species. *Annual Review of Ecology and Systematics*, 32: 305–322.
- Sandin, L., 2009. The effects of catchment land-use, near-stream vegetation, and river hydromorphology on benthic macroinvertebrate communities in a south-Swedish catchment. *Fundamental and Applied Limnology / Archiv für Hydrobiologie*, 174: 75–87.
- Santiago, J., Molinero, J. & Pozo, J., 2011. Impact of timber harvesting on litterfall inputs and benthic coarse particulate organic matter (CPOM) storage in a small stream draining a *Eucalyptus* plantation. *Forest Ecology and Management*, 262: 1146–1156.
- Shafer, C. L., 1990. *Nature reserves*. Smithsonian Institution Press, Washington.
- Shaffer, T. L. & Johnson, D. H., 2008. Ways of learning: observational studies versus experiments. *Journal of Wildlife Management*, 72: 4–13.
- Teixido, A. L., Quintanilla, L. G., Carreño, F. & Gutiérrez, D., 2010. Impacts of changes in land use and fragmentation patterns on Atlantic coastal forests in northern Spain. *Journal of Environmental Management*, 91: 879–886.
- Torralla Burrial, A. & Ocharan, F. J., 2007. Comparación del muestreo de macroinvertebrados bentónicos fluviales con muestreador surber y con red manual en ríos de Aragón (NE Península Ibérica). *Limnética*, 26: 13–24.
- Wallace, J. B., Eggert, S. L., Meyer, J. L. & Webster, J. R., 1997. Multiple trophic levels of a forest stream linked to terrestrial litter inputs. *Science*, 277: 102–104.

