Local and downstream cumulative effects of traditional meadow management on stream-water quality and multiple riparian taxa

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HIGHLIGHTS

- Traditional meadow management in the region had little detrimental effect on in-stream water quality.
- Meadows adjacent to streams may help to attenuate downstream nutrient pollution.
- Most biological changes were associated with the effects of weirs and modified riparian zones.
- There were joint downstream effects of weirs, meadow uses and riparian-zone modifications.
- Plants, snails, crawling arthropods and flies are potentially disturbance-sensitive taxa.

GRAPHICAL ABSTRACT
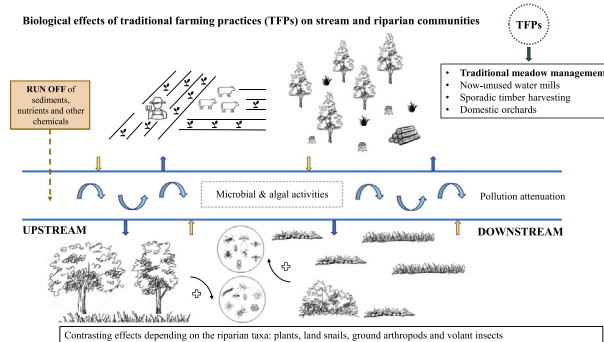
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ABSTRACT

Water quality and riparian communities are among the most affected stream components by agriculture. However, little is known about the effects of riparian management for both aquatic and terrestrial taxa at different spatial scales. Here, we surveyed aquatic (diatoms) and terrestrial taxa (bryophytes, vascular plants, litter-dwelling snails, and ground and volant arthropods), to compare the abundance and richness of riparian taxa and chemical quality between reference and exposed sites in two stream reaches each of c. 3.5 km in northwestern Spain. Impacts in exposed sites were mainly due to traditional farming practices (TFPs), which included traditional meadow management, weirs built for now-unused water mills and sporadic timber harvesting. Therefore, we measured ten covariates and predictors related to the intensification of TFPs at local and within-stream scales and explored associations with riparian and water-quality measures to study the potential effects of TFPs in more detail. Reference and exposed sites did not differ significantly in water properties (diatom-biotic indices, conductivity, total organic carbon and nitrates), but exposed sites had less concentrations of soil metals Cd, Cu, Ni and Zn and less cover and richness of riparian trees, as inferred by the index QBR. Exposed sites had more volant insect decomposers and reference sites a greater abundance or richness of snails, ground predators and decomposers. Bryophyte richness was greater in reference sites. Our inferences may inform the joint

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cumulative downstream effects of weirs, meadow uses and riparian alterations but were generally consistent with most riparian taxa benefiting from having larger forested areas. Given the contrasting responses among taxa, we argue that land snails, terrestrial flies, and centipedes may be valuable additions to current riparian assessments mostly based on plants, beetles and spiders as indicator taxa. Our study also suggests caution when inferring farming impacts on streams from the surface area of pastoral land.

1. Introduction

The use of traditional farming practices (TFPs) is becoming rare due to the movement of people from rural areas to cities and from the expansion of intensive farming (MacDonald et al., 2000; Johnson and Lichter, 2019). Consequently, there is a risk of losing the cultural heritage of human populations and the allied biodiversity that has developed in these human-modified landscapes (Gavin et al., 2015; Merçon et al., 2019). This socio-ecological situation may be reversed through direct payments to small farms to ensure the farms' continuity and ecological benefits (Guth et al., 2020). However, TFPs, if not wholly abandoned, often have incorporated some practices of farming intensification, including high livestock densities (Guillet, 2006: Leibundgut and Kohn, 2014; Renes et al., 2020). Therefore, the design of funding schemes for empowering rural areas, such as the EU's Common Agricultural Policy (Pe'er et al., 2020; De Castro et al., 2020), may benefit from studies relating different intensities of TFP to changes in biodiversity and water quality in streams, which are among the most adversely affected ecosystems by agriculture (Allan, 2004; EEA, 2012; Flávio et al., 2017).

Riparian buffers can reduce the impacts of agricultural runoff on streams, including inputs of sediments, nutrients, and other chemicals (e.g., metals) (Allan, 2004; Newbold et al., 2010; King et al., 2016; Pinay et al., 2018). However, most evidence is from experiments that have used relatively young, planted riparian strips (e.g., <40 years, Feld et al., 2018; Cole et al., 2020), and so, how inferences apply to the old-growth riparian vegetation typical of rural areas under TFPs is unclear. A common landscape in these areas is streams bordered by pastures interspersed with riparian forest strips of various widths (Guillet, 2006: Leibundgut and Kohn, 2014; Buhk et al., 2018; Renes et al., 2020). Ecological theory posits that the abundance and diversity of riparian taxa should increase with habitat size and complexity, which often is positively related to ecosystem age (Lomolino, 2000; Báldi, 2008; Allouche et al., 2012). Nevertheless, gains in biodiversity may not increase the capacity of mature riparian forests to deal with surface runoff if, for instance, forests become too shaded for understory plant development (Pinay et al., 2018). Different riparian vegetation structures are likely to complement each other in maintaining riverine biodiversity and water quality along the river length (Hynes, 1975; Vannote et al., 1980). Therefore, the ecological effects of TFPs on streams should be studied both at local and broader spatial scales because many seemingly mild upstream effects may manifest as severe downstream effects (Peterson et al., 2001; Allan, 2004; EEA, 2012).

Alongside scale-dependent effects, inferences from the ecological effects of TFPs in streams may be influenced by the group of organisms chosen as bioindicators (Barbour, 1999; Hering et al., 2006). Diatom biotic indices are widely used in monitoring schemes to inform chronic nutrient pollution (Descy and Coste, 1991; Hering et al., 2006). The structure of riparian vegetation, including bryophytes and understory vascular plants, often is used to measure hydromorphological alterations (e.g., Barbour, 1999; Munné et al., 2003; Lozanovska et al., 2020). However, it is less clear which riparian animal taxa may be the most sensitive to riverine alterations arising from TFPs, including reduced riparian cover and the presence of weirs (Cole et al., 2020; Riis et al., 2020). Sampling multiple taxa, although desirable, often is expensive, time-demanding or limited by the availability of taxonomists, and so, it may be useful for managers to identify taxa that might be used as surrogates for others (Caro and O'Doherty, 1999; Lindenmayer and Westgate, 2020).

Riparian arthropods (e.g., beetles, spiders, flies, and wasps) and litter-dwelling snails probably are good candidates for addressing the biological effects of TFPs on riparian zones because they are key players in ecological processes (Robinson et al., 2002; Ondina et al., 2004; Marshall, 2012) and in affecting the abundance of other invertebrates and vertebrates (Maisonneuve and Rioux, 2001; Baxter et al., 2005). Moreover, the abundance and diversity of terrestrial arthropods and snails are expected to be influenced by the structure of vegetation and the associated microclimatic conditions (e.g., Ondina et al., 2004; Rodrigues et al., 2014; Albacete et al., 2020). Adjacent open habitats to forests, such as meadows, may alter forest microclimate by increasing light and air temperature and by reducing humidity (Murcia, 1995; Januschke et al., 2011). Moreover, cattle in meadows may increase nutrient levels in adjacent lands due to surface runoff of excretions, which may alter abundances and composition of riparian taxa (Hadjicharalampous et al., 2002; Didham et al., 2015; Cole et al., 2020).

In this article, we used a multi-taxon approach, together with environmental data, to explore the local and downstream cumulative effects of TFPs on meadows in two stream reaches each of c. 3.5 km in northwestern Spain. By examining congruence in taxon richness among bryophytes, vascular plants, litter-dwelling snails, and major trophic guilds of ground and volant arthropods, we also aimed to identify surrogates for riparian taxa, which generally have been little studied in riverine ecology (Heino, 2010; Maceda-Veiga et al., 2019; Riis et al., 2020). The study area is a relatively well-preserved mountain region in which TFPs are the main human perturbation. TFPs mostly consisted of using meadows bordering streams for cattle pasturing and stream water to drain meadows during drought (Guillet, 2006; Buhk et al., 2018; Renes et al., 2020). Other TFPs affecting streams are domestic orchards, now-unused water mills, and sporadic timber harvesting, but downstream stream reaches mainly showed an increased gradient of meadow-surface area. Therefore, we designed a field survey to assess associations between the meadow-surface area gradient and several aquatic and riparian measures. We expected TFPs on meadows to have limited impacts on water quality if values of chemicals (e.g., nutrients, conductivity, and metals) and biotic indices (e.g., gauging from diatom assemblages) are within the legal thresholds. We expected positive associations of these chemicals and negative ones for biotic indices with the location of sites along the river length if farming effects intensify downstream. Last, we expected that the abundance and taxonomic richness of riparian taxa to be greatest in the most well-preserved riparian zones (e.g., no weirs, wide riparian areas).

2. Materials and methods

2.1. Study area

The study was conducted along 3.7 km in the Cabe stream and 3.2 km in the Louzara stream in northwestern Spain (Fig. 1). These streams were chosen because they had similar underlying geology (quartzite, limestone, slate and sandstones, 'Instituto Geológico y Minero de España'), stream-bed composition (c. 80% cobbles), similar streambed size (c. 6 m wide) and length of riparian forests (c. 1.8 km). Riparian trees such as *Alnus glutinosa*, *Fraxinus excelsior*, *Corylus avellana* and *Populus nigra* also often completely shade the two stream channels during spring and summer. The surface area of meadows also increases downstream in the two streams, but the streams differed in land use intensity. The Cabe stream
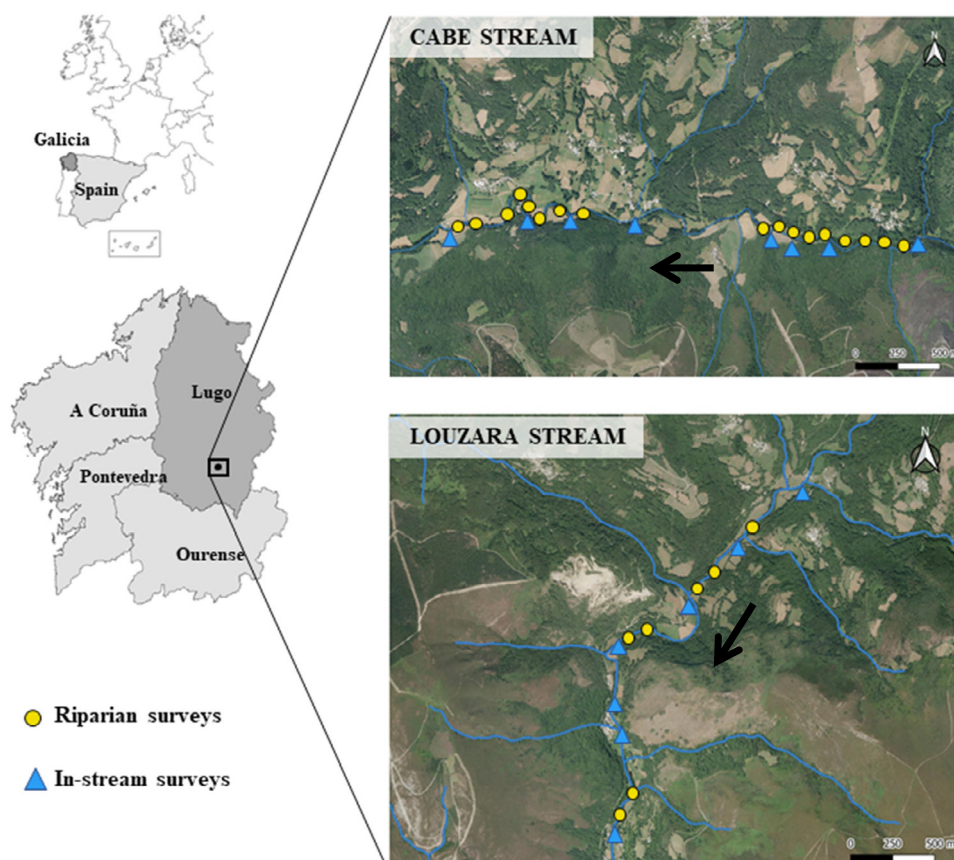
Fig. 1. Location of the riparian (n = 24) and in-stream surveys (n = 15) to explore the ecological impacts of traditional farming practices in the Cabe and Louzara streams in northwestern Spain (Galicia). The black arrow indicates flow direction.

reach had four small domestic cattle farms (c. 15–17 cattle each), whereas the Louzara stream reach had two. Moreover, all meadows in the Cabe are drained in summer, whereas this only happens in the upstream Louzara meadows. Mean water flow of streams in summer is about 0.4 m^3 s^{-1} (August 2007–2016). The flow of meadow draining was 1 L s^{-1} in one of them (AMV unpublished data), but it may be highly variable because stream water is conducted by gravity through irrigation and drainage ditches that produce a temporary surface flow over meadows until reaching the downhill ditch. Meadow flora consists of many graminoids (e.g. *Lolium perenne*), *Taraxacum officinale*, *Anthriscus sylvestris* and *Plantago lanceolata* among other flowering plants.

2.2. General sampling design of biological communities and environmental data

Sampling sites were selected, based on accessibility, to build a down-stream increased gradient of meadow surface area in the Cabe (n = 8 aquatic and 17 riparian surveys) and Louzara stream reaches (n = 7 aquatic and 7 riparian surveys) (Fig. 1). The most well-preserved upstream reaches in each stream were the reference sites for the aquatic (n = 5) and riparian study (n = 7). In each aquatic survey, we sampled diatoms and water physicochemical properties (see details in Section 2.2.1). Riparian surveys were to study soil metals and the composition of flora (bryophytes and vascular plants) and fauna (snails, ground arthropods and volant insects) (see details in Section 2.2.2). All sites were surveyed only once in August 2019 unless otherwise specified.

2.2.1. Aquatic surveys

Diatom sampling, preparation and counting followed CEN standards (CEN, 2003, 2004). Briefly, three medium-size cobbles (20 × 20 cm) were collected from a well-lit riffle section, excluding cobbles with

filamentous algae and soft sediment. The cobbles were scraped with a knife to detach the algal community (3 cm^2) and samples were rinsed in 5 mL of stream water and fixed in 4% formaldehyde until analysis. Diatom frustules were cleaned (33% hydrogen peroxide and 35% hydrochloric acid) and mounted on Naphrax® resin (Brunel Microscopes Ltd., Chippenham, UK) to be counted under light microscope at ×1000. At least 400 valves were counted and identified to calculate three widely used biotic indices, namely Specific Polluosensitivity Index (IPS, Coste, 1982), the Diatom Biologic Index (IBD, Prygiel and Coste, 2000) and the Index of the European Economic Community (CEE, Descy and Coste, 1991) using the software OMNIDIA® v. 6.0.8 (Lecointe et al., 1999).

Physicochemical water properties were measured in early and late August; we used the average value of the two sampling dates. Conductivity (μS cm^{-1}), pH and temperature (°C) were measured in the centre and at each stream border using portable Hanna® probes (HI 98127-98311). Water hardness was measured alike using the colorimetric test Visocolor® (Germany). Triplicates of water samples were collected for nitrate (μg L^{-1}) and Total Organic Carbon (TOC, mg L^{-1}) analyses. Samples were filtered in the field through Whatman GF/F glass fibre filters (Kent, U.K.), and then stored on ice and frozen at −20 °C in the laboratory until analysis. The concentration of nitrate was determined using standard colorimetric methods described in APHA (1995). TOC was measured using a high-temperature catalytic oxidation TOC analyser (Kyoto, Japan).

2.2.2. Surveys of riparian vegetation and adjacent land uses

The riparian area was defined as the area located from the edge of the high water channel and bank full to the edge of upland, characterized by the presence of a distinctive riparian vegetation and substrate type (Corti and Datry, 2014). The quality of riparian vegetation cover was assessed using the biotic index QBR (Qualitat del Bosc de Ribera)

for a 100-m riparian stream section, following the guidelines of Munné et al. (2003). The QBR index, which is widely used by Spanish water agencies, uses a 0–25 scale to score different aspects of riparian vegetation and stream hydromorphology (see details in Munné et al., 2003), providing an overall score from 0 (bad) to 100 (very good).

For riparian areas, we also recorded: (1) the total number of species of bryophytes and understory vascular plants following Lozanovska et al. (2020) and Colin et al. (2016); (2) the riparian area width (m); (3) the number of upstream weirs, which are used for meadow irrigation; and (4) three potential predictors related to adjacent land-uses to the streams (retrieved from Google Earth®): (i) the surface area of forest patches, including adjacent non-riparian forests; (ii) the surface area of meadows; and (iii) the linear distance of riparian forest upstream each sampling site along the stream length. These three potential predictors were calculated for local and within-stream scales. For instance, we measured the total meadow-surface area that was immediately adjacent to the sampling site (i.e. the local-meadow surface area) and the sum of surface areas of all meadows until the first of the studied sites upstream in each stream (i.e. the total upstream-meadow surface area). We measured the surface of forests and meadows until there were natural discontinuities (e.g. ecotone between forest and meadow patches). There was a median of 10 cattle faeces per 100 m^{-2} of meadows in the Cabe stream reach (AMV observation). We focused on the c. 3.5 km reaches in each stream because the focal reaches, either upstream or downstream, travel through a relatively deep forest area.

We also recorded the longitudinal position of each site in relation to the first sampling site in each stream to account for downstream effects (hereafter 'downstream distance'). The factor 'number of upstream weirs' was then linearly weighted by the covariate 'downstream distance' because the nearer a weir is to the sampling site, the greater its ecological effects are expected to be (Elosegi et al., 2010; Kemp and O'hanley, 2010).

2.2.3. Riparian animal surveys

Given that sampling methods for riparian invertebrates are not internationally standardized, we designed our own protocol to get comparable quantitative data. Litter-dwelling snails and ground arthropods were collected using 60-cm diameter, 20-cm depth metal cylinders placed in a number of plots proportional to the size of riparian area. We did a pilot study based on three small (1.5 m wide) and large (6 m wide) riparian patches and found that sampling 56,548 cm^3 per 1.5 m wide led to detect 80% of litter-dwelling snail species in a 6-m riparian patch surveyed more intensively (i.e. 90 min of active search for snails). Therefore, we employed this standardized sampling effort, which had been previously used in studies of riparian arthropods (e.g. Maceda-Veiga et al., 2016). Ground samples were processed in three steps: (1) the collected leaf litter and soil were placed on a tray to capture the most mobile taxa (e.g. centipedes, spiders); (2) the ground samples were wet-sieved through a 7-mm mesh and the material retained was carefully examined using a magnifying glass; (3) the same process was repeated using a 0.5-mm mesh size to detect animals or parts to obtain the sieved soil samples for metal analyses (see Section 2.2.4).

Volant insects were captured for 19 days using a modified version of the widely used beer trap (Dvorák et al., 2010; Manko et al., 2019). We hung one trap on a tree branch at c. 1.5 m from the ground and c. 40 cm from the trunk in each riparian patch and the trap content was replaced every week (see further details in Albacete et al., 2020; Matas et al., 2021). We acknowledge that no individual sampling method can sample all types of insects (Manko et al., 2019; Montgomery et al., 2020), and that beer traps mostly capture fermented liquid-feeders (e.g. Nitidulidae, Staphylinidae, Heleomyzidae, Muscidae, Sarcophagidae) and predatory wasps (Albacete et al., 2020; Maceda-Veiga et al., 2021). Our captures should be regarded as a measure of 'abundance activity' that may be affected, alongside the abundance of volant insects per se, by any factor altering the emission of volatile compounds from the traps and their perception by the insects, as occurs with dung and carrion traps in ecological studies (Braack, 1987; Gibbs and Stanton, 2001).

Snails were air-dried and identified to species. Arthropods were fixed in 70% ethanol and identified to the lowest feasible taxonomic resolution, which was to family for most individuals (Appendix S1). The use of family is warranted because most spiders were juveniles, which prevents the identification of spiders to species. We captured individuals from 20 families of flies for which expert taxonomists were not available. However, there was high congruence ($R^2 = 0.89$) in richness values of flies at the family and species levels for the 11 families studied in detail (Appendix S2). Trophic guilds (e.g. predators, decomposers) were assigned based on mobility (volant or ground), taxonomy and oral morphology in adult arthropods (Maceda-Veiga et al., 2016; Albacete et al., 2020; Matas et al., 2021).

2.2.4. Riparian soil metal analyses

Although we were not able to measure metal concentrations for all 24 riparian patches due to issues with the transport of samples, we had enough soil to analyse six metals (mg kg^{-1}) for five and four patches of the Cabe and Louzara streams, respectively. The metals cadmium, copper, chromium, nickel, lead, and zinc were measured by the certified laboratory Eurofins Agro, S.A. (www.eurofins.es). The study of soils is pertinent because metals often attach to soil particles (Macías-Vázquez and Calvo de Anta, 2009; Comas et al., 2014; Monroy et al., 2014). Therefore, if riparian areas slow surface runoff, there would be clean waters but there would be significant amounts of metals in riparian soils, particularly Zn and Cu, which are common in livestock excretions (Comas et al., 2014).

2.3. Data analyses

All statistical analyses were performed using R statistical software (R Core Team, 2014). The distributions of all variables were inspected prior to modelling and variables other than pH were log-transformed to meet statistical requirements of models (e.g. reducing severe kurtosis). We also calculated estimated taxon richness (Chao2 estimator) of snails and arthropods instead of using raw taxon richness to account for differences in species detection due to disparities in total captures (Walther and Moore, 2005).

To identify potential surrogates for monitoring riparian taxa, Kendall's τ coefficients were applied to taxonomic richness of riparian taxa and

Table 1
Descriptive statistics of biotic indices, water-quality properties and soil-metal concentrations (M, median; IQR, inter-quartile range and the minimum-maximum values) in reference and exposed sampling sites. Letters group values of streams at $P \leq 0.05$ based on model outputs in Appendix S3.

	Reference sites		Exposed sites	
	M – IQR	Min–max	M – IQR	Min–max
Biotic indices				
IPS (score)	29 – 11[a]	18–30	27 – 6[a]	22–34
CEE (score)	21 – 11[a]	13–27	22 – 4[a]	18–28
IBD (score)	30 – 12[a]	17–32	27 – 6[a]	22–34
QBR (score)	80 – 17[a]	70–100	55 – 20[b]	40–90
Water quality				
Conductivity (µS cm^{-1})	87 – 25[a]	61–104	88 – 20[a]	70–112
Nitrate (mg L^{-1})	2.3 – 0.8[a]	1.9–2.7	2.9 – 0.4[a]	2.1–3.9
TOC (mg L^{-1})	9.6 – 4.0[a]	6.7–11.8	11 – 2.6[a]	6.7–13
pH	8.1 – 0.3[a]	7.2–8.2	8.0 – 0.2[a]	7.4–8.0
Water hardness (°dH)	2.0 – 1.0[a]	2.0–3.0	3.0 – 0.8[b]	3.0–4.0
Soil quality				
Cadmium (mg kg^{-1})	0.7 – 0.39[a]	0.5–1.03	0.5 – 0[b]	0.5–0.5
Copper (mg kg^{-1})	26.1 – 10[a]	21.4–33.5	25.9 – 4[b]	23.2–27.2
Chromium (mg kg^{-1})	15.5 – 6.6[a]	12.7–22.4	20.5 – 6.1[b]	16.3–23.4
Nickel (mg kg^{-1})	29.5 – 15[a]	22.5–40.2	37.5 – 8.9[b]	29.3–48
Lead (mg kg^{-1})	19.3 – 11.7[a]	13.8–27.4	21 – 3.4[a]	17.7–21.3
Zinc (mg kg^{-1})	205 – 202[a]	109–337	169 – 80[b]	119–200

IPS, CEE and IBD were diatom-based indices; QBR is an index of riparian quality; TOC is total organic carbon (see Materials and methods for detailed information).

scores of the riparian vegetation index QBR. The non-parametric Kendall's τ does not rely on assumptions about the distributions of pairs of variables and is adequate for small sample sizes (Kendall and Gibbons, 1990). Simple pair-wise associations are the preferred method to look at congruence among diversity measures (e.g., review by Heino, 2010). However, we also explored this question by means of linear mixed models (LMMs, the R function *lmer*, Bates et al., 2019) to account for the spatial structure of our data. LMMs allowed us to include stream identity as random intercept, whereas simple pair-wise correlations do not.

To assess the local effects of TFPs, we compared values of water properties, biotic indices and the abundance and taxon richness of riparian taxa between site types (reference or exposed) using linear models (LMs). LMs included main and interactive effects between site type and stream identity (Cabe or Louzara). To assess the cumulative downstream effects of TFPs, we used the same LM design but replaced the factor 'site type' with the covariate 'downstream distance'. Permutational Analyses of Variance (PERMANOVA, the R function *adonis*; Oksanen et al., 2020) were used to explore differences in taxonomic composition of riparian taxa between site types and streams. Indicator-value analyses (the R function *multipatt*; De Cáceres et al., 2016) were then performed to identify the most characteristic taxon in reference and exposed sites at $P \le 0.05$.

Given that 'simple' classifications of sites as reference and exposed do not reflect the spatial complexity of biological and environmental data (e.g. Ter Braak and Prentice, 1988; Kreyling et al., 2018), we used multivariate regression methods to study associations between biological and environmental data in more detail. First, we standardized all potential

predictors to zero mean and unit SD. Second, we deleted potential predictors one-by-one until all variables had Variation Inflation Factor (VIF) ≤ 3, which reduces bias in regression estimates due to collinearity following Zuur et al. (2010). Where factors were excluded, we reported Kendall's τ coefficients to show the strength and direction of associations between included and excluded potential predictors. Last, we used multimodel inference (the R function *dredge*) to identify potentially important predictors in LMMs at AIC ≤ 2 (Burnham and Anderson, 2004). Initial LMMs included main effects for all potential predictors and stream identity as a random intercept to account for potential systematic differences between the two streams. Model fits were assessed by means of R^2 (Nakagawa and Schielzeth, 2013). Statistical significance of potential predictors in final models was gauged at $P \le 0.05$, and their importance was assessed by inspecting values of standardized regression coefficients.

3. Results

3.1. Differences in biological and environmental data between reference and exposed sites

Reference and exposed sites did not significantly differ in values of diatom biotic indices and most physicochemical water properties

Table 2
Linear regression equations showing the associations between the downstream spatial location of sampling sites (DLSS) and the categorical factor stream identity (ID) for all biotic indices, water-quality properties and soil metal concentrations. DLSS is the predictor that informs the potential downstream accumulated effects of pastured lands on the stated water and riparian environmental indicators. For 'good models' (i.e. R^2 with $P \le 0.05$ indicated with †), statistically significant regression coefficients are highlighted in bold.

	Equations (y = response variable in the first column)	Statistics for the linear regressions
Biotic indices		
IPS	$y = 3.49 - 0.38 \cdot$ Stream ID (Lou) $- 0.13 \cdot$ DLSS $+ 0.29 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.30$; $F_{3,11} = 1.6$; $P = 0.24$
†CEE	$y = \mathbf{3.34 - 0.54 \cdot}$ **Stream ID (Lou)** $- 0.15 \cdot$ DLSS $+ \mathbf{0.38 \cdot}$ **DLSS $\times$ Stream ID (Lou)**	$R^2 = 0.49$; $F_{3,11} = 3.6$; $P = 0.04$
IBD	$y = 3.51 - 0.42 \cdot$ Stream ID (Lou) $- 0.16 \cdot$ DLSS $+ 0.34 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.30$; $F_{3,11} = 1.6$; $P = 0.24$
†QBR	$y = 4.60 + 0.01 \cdot$ Stream ID (Lou) $- \mathbf{0.42 \cdot}$ **DLSS** $- 0.03 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.66$; $F_{3,20} = 12.9$; $P < 0.01$
Water quality		
Conductivity	$y = 4.56 - 0.30 \cdot$ Stream ID (Lou) $- 0.00 \cdot$ DLSS $+ 0.15 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.41$; $F_{3,11} = 2.5$; $P = 0.11$
Nitrate	$y = 1.24 - 0.12 \cdot$ Stream ID (Lou) $+ 0.09 \cdot$ DLSS $+ 0.12 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.34$; $F_{3,11} = 1.9$; $P = 0.18$
†TOC	$y = \mathbf{2.46 - 0.30 \cdot}$ **Stream ID (Lou)** $+ 0.07 \cdot$ DLSS $- 0.00 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.71$; $F_{3,11} = 8.3$; $P < 0.01$
pH	$y = 8.21 - 0.49 \cdot$ Stream ID (Lou) $- 0.34 \cdot$ DLSS $+ 0.47 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.26$; $F_{3,11} = 1.3$; $P = 0.31$
†Water hardness	$y = \mathbf{1.32 - 0.29 \cdot}$ **Stream ID (Lou)** $+ 0.13 \cdot$ DLSS $+ 0.21 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.77$; $F_{3,11} = 12.5$; $P < 0.01$
Soil quality		
†Cadmium	$y = \mathbf{0.73 - 0.33 \cdot}$ **Stream ID (Lou)** $- \mathbf{0.22 \cdot}$ **DLSS** $+ \mathbf{0.22 \cdot}$ **DLSS $\times$ Stream ID (Lou)**	$R^2 = 0.90$; $F_{3,5} = 115.1$; $P < 0.01$
†Copper	$y = \mathbf{3.52 - 0.49 \cdot}$ **Stream ID (Lou)** $- \mathbf{0.12 \cdot}$ **DLSS** $+ 0.21 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.86$; $F_{3,5} = 18.2$; $P < 0.01$
†Chromium	$y = \mathbf{3.00 - 0.63 \cdot}$ **Stream ID (Lou)** $+ 0.10 \cdot$ DLSS $+ 0.27 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.84$; $F_{3,5} = 15.3$; $P < 0.01$
†Nickel	$y = \mathbf{3.63 - 0.77 \cdot}$ **Stream ID (Lou)** $+ 0.08 \cdot$ DLSS $+ 0.34 \cdot$ DLSS $\times$ Stream ID (Lou)	$R^2 = 0.88$; $F_{3,5} = 20.2$; $P < 0.01$
†Lead	$y = \mathbf{3.37 - 1.08 \cdot}$ **Stream ID (Lou)** $- \mathbf{0.22 \cdot}$ **DLSS** $+ \mathbf{0.78 \cdot}$ **DLSS $\times$ Stream ID (Lou)**	$R^2 = 0.90$; $F_{3,5} = 26.8$; $P < 0.01$
†Zinc	$y = \mathbf{5.89 - 1.28 \cdot}$ **Stream ID (Lou)** $- \mathbf{0.43 \cdot}$ **DLSS** $+ \mathbf{0.57 \cdot}$ **DLSS $\times$ Stream ID (Lou)**	$R^2 = 0.96$; $F_{3,5} = 66.1$; $P < 0.01$

Stream ID (Lou) is the second level of the categorical factor Stream ID, which shows values of the Louzara stream relative to the Cabe.
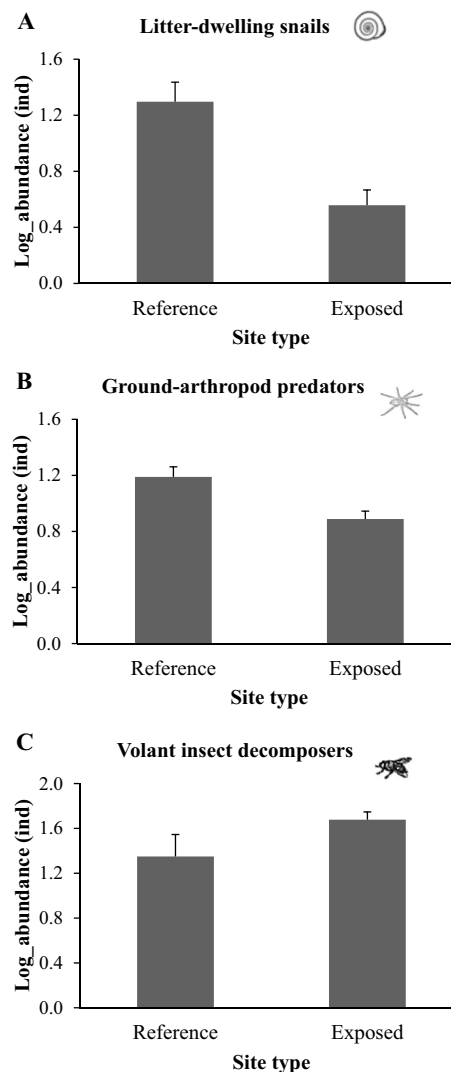

Fig. 2. Changes in mean abundances ($\pm$S.E.) of riparian fauna between reference ($n = 7$) and exposed ($n = 17$) sampling sites after having accounted for the effects of stream identity. Only taxa with significant differences between site types at $P \le 0.05$ in Appendix S4 are shown.

(Table 1, Appendix S3). The exception was water hardness, which significantly increased in exposed sites (Table 1). Conversely, exposed sites had significantly lower values in the riparian index QBR and the metals Cd, Cu, Ni and Zn than reference sites (Table 1). The opposite trend was found for Cr (Table 1).

Some metals (Cd, Cu, Pb and Zn) and the QBR index had significant inverse associations with the downstream location of sites within the stream reaches (Table 2). This association was not significant for other metals, physicochemical properties or diatom-biotic indices (Table 2).

Reference and exposed sites differed significantly in values of litter-dwelling snails, ground-arthropod predators, volant insect decomposers and bryophytes (Figs. 2 and 3). Specifically, the abundance of the first two decreased in exposed sites compared to reference values (Fig. 2; Appendix S4), as did their estimated taxonomic richness (Fig. 3). The estimated taxonomic richness of bryophytes and ground-arthropod decomposers were significantly less in exposed than in reference sites (Fig. 3; Appendix S4). However, exposed sites had more volant insect decomposers than did reference sites (Fig. 3; Appendix S4). There were also differences in taxonomic composition between reference and exposed sites for understory vascular plants and for ground predators (Appendix S4). In particular, Lithobiid centipedes and the fern *Dryopteris affinis* were significantly more common in reference sites than exposed sites (both $P \leq 0.05$). None of the other taxa had significant associations based on the indicator-value analyses.

3.2. Associations between biological and environmental data

All diversity measures of riparian taxa were included in LMMs as individual response variables because congruence in taxon-richness values between groups was only moderate (e.g. All Kendall's $\tau < 0.5$, Appendix S5). However, we only studied the factors affecting variation in the CEE index because diatom indices (CEE, IPS and IBD) all were strongly correlated (Kendall's $\tau > 0.8$). Regarding the potential predictors, we excluded four of the original ten from modelling (Appendix S6), including the total upstream-meadow surface area, due to collinearity (VIFs < 3). However, this potential predictor co-varied substantially with the number of weirs (Appendix S6), so we assumed that the latter might be a proxy for the downstream cumulative effects of meadow-surface area on riparian taxa. The overall quality of the riparian forest patch (QBR index) was omitted because of its high correlation with the number of weirs upstream of the sampling site (Kendall's $\tau = -0.66$). However, we report pair-wise correlations between QBR scores and taxonomic richness because this information may be useful for users of the riparian index QBR (Appendix S5).

In the LMM for diatoms ($R^2 = 0.81$), we found that CEE scores were positively associated with understory plant height but were negatively with the number of upstream weirs and the riparian forest width (Table 3). The inclusion of the number of upstream weirs suggested that total upstream meadow surface area (i.e. cumulative effects) was a more important factor than local meadow-surface area (i.e. local effects) to explain variation CEE scores. Therefore, we examined the associations between these two factors and the CEE scores in detail, and found that CEE scores were more strongly associated with the catchment meadow-surface areas ($R^2 = 0.43$ vs 0.30) than with the local meadow-surface areas ($R^2 = 0.11$ vs 0.02) for the Cabe and the Louzara stream reaches respectively (Fig. 4, Appendix S7).

Riparian LMMs had the highest fits (R^2) for the estimated taxonomic richness and abundance of litter-dwelling snails ($R^2 > 0.62$), ground-arthropod decomposers ($R^2 > 0.62$) and volant-insect decomposers ($R^2 > 0.75$). The LMM fit for the abundance of ground-arthropod predators was good ($R^2 = 0.61$), so we focus on these animal groups. Forest continuity, as indicated by upstream linear-forest cover and total forest-patch size, often had similar or stronger associations than did the number of upstream weirs or local meadow surface area with these faunal elements (Fig. 5). However, the number of weirs was the only potential predictor more related to traditional meadow management that explained variation in the estimated taxonomic richness and abundance of litter-dwelling snails, and the estimated taxonomic richness of volant-insect decomposers (Fig. 5). Local-meadow surface area only had statistically significant (and positive) associations with the estimated taxon richness of volant-insect decomposers (Fig. 5). Of these two potential predictors, the number of weirs had the stronger association.
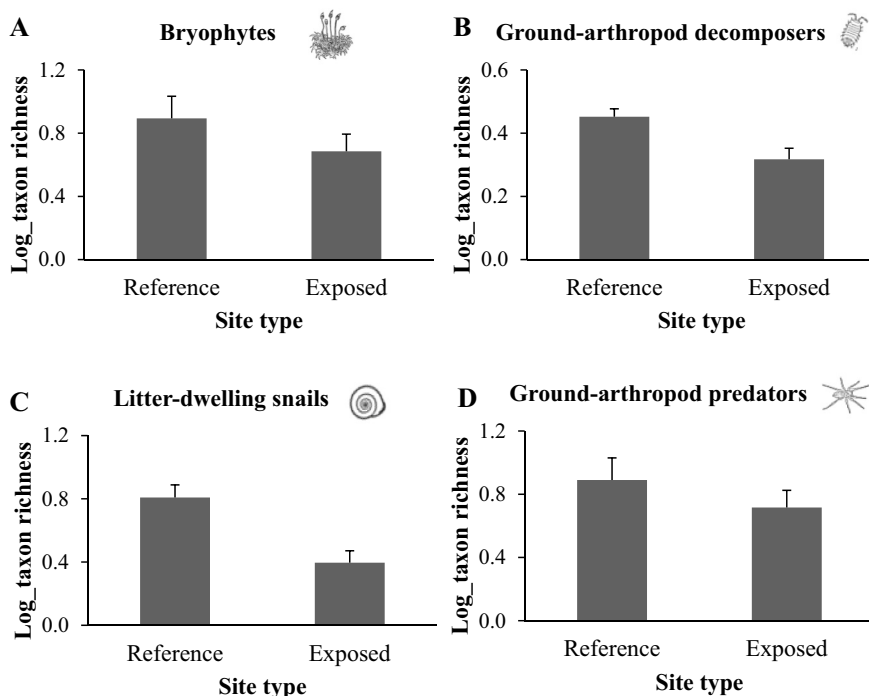

Fig. 3. Changes in mean estimated taxonomic richness ($\pm$S.E.) of riparian organisms between reference ($n = 7$) and exposed ($n = 17$) sampling sites after having accounted for the effects of stream identity. Only taxa with significant differences between site types at $P \leq 0.05$ in Appendix S4 are shown.

Table 3

Linear mixed model outputs showing the associations between the diatom-biotic index of the European Commission (CEE) and the characteristics of the Cabe and Louzara streams. β are the standardized regression coefficients, *SE* is the standard error of these coefficients, *df* are the degrees of freedom for the statistic *t*. Model included stream identity as random effect.

Model fit ($R^2 = 0.81$)	β	SE	t-Value	df	P-value	Stream-random factor
Number of weirs	−0.21	0.04	−5.59	9	<0.001	SD = 0.008|Res = 0.098
Riparian-forest width[a]	−0.10	0.03	−3.7	9	0.005	
Understory plant height	0.22	0.03	5.71	9	<0.001	

[a] Refers to the riparian area with riparian trees.

4. Discussion

Our study suggests that the conservation of traditional meadow management in two stream reaches each of c. 3.5 km in northwestern Spain may have greater impacts on stream hydromorphology and riparian communities than on in-stream water quality. However, our inferences could not differentiate among joint impacts arising from other TFPs, including weirs built for now-unused water mills and sporadic timber harvesting. Our inferences also may inform the joint cumulative downstream effects of meadow runoff and altered stream hydromorphology because the total number of upstream weirs and total upstream meadow-surface area were positively correlated.

4.1. Legal thresholds for physicochemical water properties and biotic indices

Our multitaxon inferences are based on litter-dwelling snails (12 species), ground arthropods (≥23 families), volant insects (32 families),

Fig. 4. Linear regressions of the diatom index of the European Commission (CEE) and the local and catchment meadow-surface area in the Cabe (green diamonds) and Louzara streams (orange squares) showing different slope directions between the streams.

bryophytes (33 species), understory vascular plants (40 species) and diatoms (79 taxa). For these biological elements, there are only legal thresholds for the diatom-biotic index IPS (18) and the riparian index QBR (95) in streams of the typology studied (Spanish Act RD 817/2015). For water physicochemistry and soil metals, there are for nitrates (10 mg NO_3^- L^{-1}), Cd (0.1 mg kg^{-1}), Cu (20.6 mg kg^{-1}), Cr (80.3 mg kg^{-1}), Ni (55.8 mg kg^{-1}), Pb (29.3 mg kg^{-1}) and Zn (60.3 mg kg^{-1}) (BOE RD 817/2015; DOG D 60/2009). Given that our scores for diatoms, metals and nitrates were below these values, we deduce that water quality was good in a legal sense. Although chemical analyses were snapshot measures, diatom indices are good indicators of chronic pollution by nutrients and other chemicals (Descy and Coste, 1991; Hering et al., 2006; Colin et al., 2016). Moreover, water-quality measures in the two streams overall were typical of oligotrophic and poorly mineralized headwaters (Wetzel, 1983). However, QBR values in many exposed sites (median = 55) were well below the legal threshold, primarily because there was often only lone line of trees bordering the streams.

4.2. Local and downstream effects of traditional meadow management on water

While reference and exposed sites did not differ significantly in values of nutrients, total organic carbon, conductivity and soil metals, we found that the Cabe stream had greater values of these variables than the Louzara stream reach. The two streams had similar underlying geology (see study area), so that the more intense meadow irrigation in the Cabe and higher cattle density may explain the greater metal levels, at least of Cu and Zn. Livestock food often is supplemented with these elements (Comas et al., 2014), which can reach riparian areas by runoff of stock animal deposits (Pinay et al., 2018). Nonetheless, levels of N and metals were much lower in our study than in regions with intensive farming, which often have water-nitrate levels > 50 mg L^{-1} (García-Galán et al., 2010) and soil Cu levels > 50 mg kg^{-1} (Li et al., 2020). Greater levels of Pb in the Cabe than in the Louzara stream may be related to legacy pollution of the now-prohibited leaded fuels or recreational hunting, although we have no supporting evidence for effects of these activities.

The Louzara stream reach had significantly lower scores for the diatom index CEE than the Cabe stream reach (Table 2), which is not consistent with potential greater impacts of traditional meadow management in the latter. However, the total upstream meadow-surface area was positively associated with CEE scores in the Louzara but was negatively correlated with CEE in the Cabe stream. Although we did not measure in-stream light intensity, the positive association found in the Louzara may be due to the Louzara stream reach having greater light penetration than the Cabe reach (AMV pers. observ.). Streams draining open habitats have greater in-stream photosynthetic activity, and so, a greater capacity to attenuate nutrient pollution than do more shaded stream reaches (Sabater et al., 2000). Therefore, the risk for cumulative downstream effects of chemicals from meadow runoff may be greater for the Cabe stream reach than in the Louzara stream reach. Nonetheless, our findings generally are consistent with streams in relatively unimpacted areas of different climatic ranges retaining nutrients effectively (McColl, 1974; Newbold et al., 1981; Munn and Meyer, 1990; Peterson et al., 2001; Pinay et al., 2018).
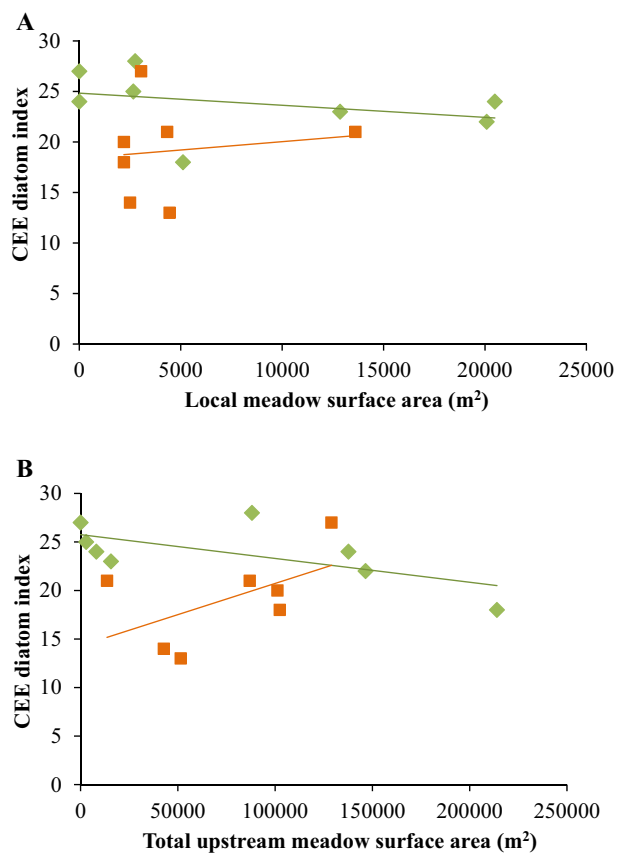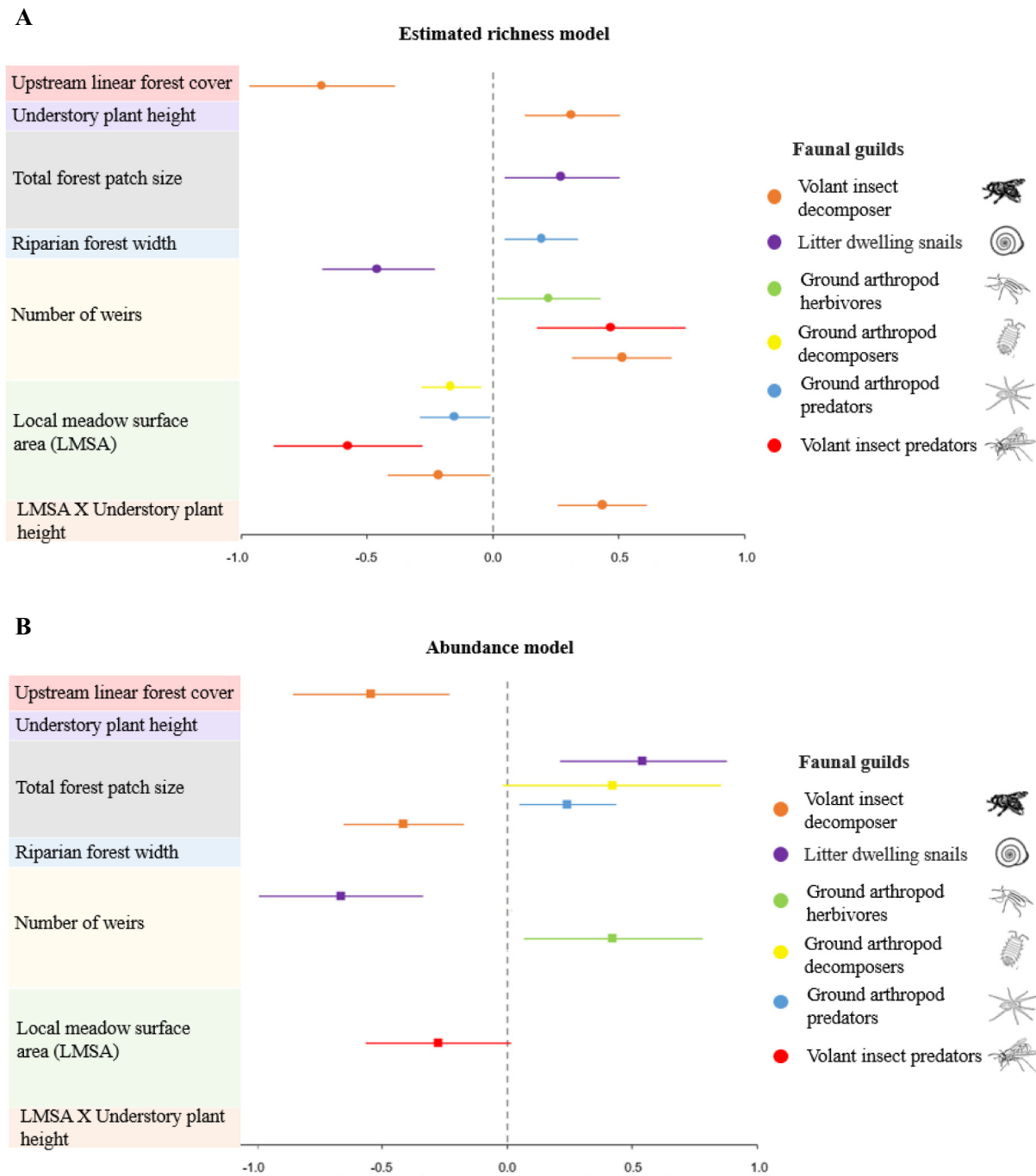
A

B

Fig. 5. Standardized regression coefficients (X-axes) showing the direction and strength of associations between (A) taxonomic richness and (B) abundance of riparian organisms and stream characteristics describing farming impacts alongside covariates. Only statistically significant predictors from Appendix S8 are shown. All models included 'stream identity' as random effect to account for spatial correlation among sampling sites within each stream.

4.3. Local and downstream effects of traditional meadow management on riparian taxa

The preceding paragraph may suggest that a manner to increase downstream pollution attenuation is to thin or reduce riparian forests. However, there are adverse side effects to such a management action, including higher stream water temperatures (Pinay et al., 2018) and forest fragmentation (Murcia, 1995; Januschke et al., 2011). We also found lower abundances and taxonomic richness of litter-dwelling snails and ground-arthropod predators in the narrow, tested forest patches. These animal groups are key players in riparian ecosystem processes, such as

litter breakdown for snails and the regulation of aquatic-terrestrial in-fluxes for ground predators (e.g. Tetragnathid spiders, carabid beetles) (Robinson et al., 2002; Baxter et al., 2005; Ramey and Richardson, 2017). Nonetheless, the ecological consequences of changes in species or individual traits are hard to predict without a priori establishing their associations with particular ecosystem functions (Von Schiller et al., 2017; Sobral, 2021). The numbers of volant-insect decomposers (e.g., Heleomyzidae, Muscidae, Drosophilidae) seemed to be greater in smaller forest patches. This may be due to well-preserved riparian forests being too shaded for the flowering plants upon which these insects feed (e.g., blackberries, Allen, 2016; Maceda-Veiga et al., 2021), to nearby

habitats acting as sources of volant insects because of their high mobility (e.g. 9000 m in Drosophilidae, Tait et al., 2018) or to flies' attraction to light (Goldsmith, 1990; Marshall, 2012). The benefits of preserving mature riparian forests for agricultural lands may be manifold, including less agricultural pests (Cole et al., 2020; Riis et al., 2020; Maceda-Veiga et al., 2021). Among the many factors influencing these ecosystem services, our study suggests forest microclimatic conditions because there was low richness of bryophytes and ground-arthropod decomposers, both of which are moisture-dependent (Hadjicharalampous et al., 2002; Maceda-Veiga et al., 2016; Lozanovska et al., 2020).

The abundance or taxonomic richness of volant decomposers and litter-dwelling snails had opposite associations with the number of upstream weirs. However, of these two, the latter may be better indicators of local conditions because they are slow-moving, microhabitat specialists (Kappes et al., 2006). Weirs may reduce the downstream deposition of dead leaves and wood upon which litter-dwelling snails feed and shelter (Mason, 1970; Kappes et al., 2006). Associations with weirs were not significant for the abundance or richness of ground arthropod predators despite spiders and crawling beetles often being regarded as vulnerable to flow regulation (e.g., Lambeets et al., 2008; Paetzold et al., 2008; Jähnig et al., 2009). However, this may be due to differences in taxonomic resolution among studies (but see Timms et al., 2013) or to the fact that most studies surveyed dry gravel banks, which our study area did not have. In any case, more complete inventories of riparian invertebrates probably are needed to discern the biological effects of riparian management, including stream hydromorphology. Our study suggests that Lithobiid centipedes, litter-dwelling snails and terrestrial families of flies (e.g., Heleomyzidae, Muscidae, and Drosophiliidae) may be particularly valuable 'new' indicator taxa.

5. Management implications

Our study addressed a major challenge for managers around the world: the conservation of traditional human cultures and natural heritages (Gavin et al., 2015). Local farmers in Spain are the main contributors to maintaining semi-natural habitats, such as the >40 year-old meadows in this study, at a time when these ecologically valuable habitats (Stebbings, 2005; Leibundgut and Kohn, 2014; Renes et al., 2020) are disappearing due to rural human abandonment and to increased cover of plantations and young forests, at least in Europe (MacDonald et al., 2000; Márquez-Castro, 2014). Our findings in one of the two streams studied suggested that the surface area of pastoral land might not necessarily be a proxy for the impacts of farming on streams. However, extant farming practices in the study area may need better management practices for the streams to be preserved, at least to protect riparian communities and natural stream hydromorphology (Palmer et al., 2005; Maceda-Veiga, 2013; Cole et al., 2020). Detailed management actions need further testing because the riparian-aquatic systems can be very dynamic and different taxa may show contrasting responses.

CRediT authorship contribution statement

Conceived the research and conducted the field work: AMV
Counted and identified the taxa: AMV, SA, NFA (diatoms), CV (bryophytes), VB (snails), MD (spiders) and JPV (wasps)
Conducted chemical analyses: JMB and FS
Analyzed data: AMV and SA.
Writing was led by AMV but all authors edited (particularly RM) and approved the final version.

Declaration of competing interest

Authors declared no conflict of interest.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.148601.

References

Albacete, S., Mac Nally, R., Carles-Tolra, M., Domenech, M., Vives, E., Espadaler, X., et al., 2020. Stream distance and vegetation structure are among the major factors affecting various groups of arthropods in non-riparian chestnut forests. For. Ecol. Manag. 460, 117860.

Allan, J.D., 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257–284.

Allen, D.C., 2016. Microclimate modification by riparian vegetation affects the structure and resource limitation of arthropod communities. Ecosphere 7 (2), e01200.

Allouche, O., Kalyuzhny, M., Moreno-Rueda, G., Pizarro, M., Kadmon, R., 2012. Area–heterogeneity tradeoff and the diversity of ecological communities. Proc. Natl. Acad. Sci. 109 (43), 17495–17500.

APHA, 1995. Standard Methods for the Examination for Water and Wastewater. 19th ed. 591 American Public Health Association, Inc., New York.

Báldi, A., 2008. Habitat heterogeneity overrides the species–area relationship. J. Biogeogr. 35 (4), 675–681.

Barbour, M.T., 1999. Rapid Bioassessment Protocols for Use in Wadeable Streams and Rivers: Periphyton, Benthic Macroinvertebrates and Fish. US Environmental Protection Agency, Office of Water.

Bates, Douglas, Maechler, Martin, Bolker, Ben, Walker, Steven, 2019. Lme4: linear mixed-effects models using 'Eigen' and S4. https://CRAN.R-project.org/package=lme4.

Baxter, C.V., Fausch, K.D., Carl Saunders, W., 2005. Tangled webs: reciprocal flows of invertebrate prey link streams and riparian zones. Freshw. Biol. 50 (2), 201–220.

Braack, L.E.O., 1987. Community dynamics of carrion-attendant arthropods in tropical African woodland. Oecologia 72 (3), 402–409.

Buhk, C., Schirmel, J., Rebekka, G., Frör, O., 2018. Traditional water meadows: a sustainable management type for the future? Irrigation in Agroecosystems. IntechOpen

Burnham, K.P., Anderson, D.R., 2004. Multimodel inference: understanding AIC and BIC in model selection. Sociol. Methods Res. 33 (2), 261–304.

Caro, T.M., O'Doherty, G., 1999. On the use of surrogate species in conservation biology. Conserv. Biol. 13 (4), 805–814.

CEN EN13946, 2003. Waterquality – Guidance Standard for the Routine Sampling and Pre-treatment of Benthic Diatoms from Rivers. Comité Européen De Normalisation, Geneva, p. 14.

CEN prEN14407, 2004. Waterquality – Guidance Standard for the Identification, Enumeration and Interpretation of Benthic Diatom Samples from Rivers. Comité Européen De Normalisation, Geneva, p. 12.

Cole, L.J., Stockan, J., Helliwell, R., 2020. Managing riparian buffer strips to optimise ecosystem services: a review. Agric. Ecosyst. Environ. 296, 106891.

Colin, N., Maceda-Veiga, A., Flor-Arnau, N., Mora, J., Fortuño, P., Vieira, C., ... De Sostoa, A., 2016. Ecological impact and recovery of a Mediterranean river after receiving the effluent from a textile dyeing industry. Ecotoxicol. Environ. Saf. 132, 295–303.

Comas, J., Domínguez, C., Salas-Vázquez, D.I., Parera, J., Díez, S., Bayona, J.M., 2014. Input and leaching potential of copper, zinc, and selenium in agricultural soil from swine slurry. Arch. Environ. Contam. Toxicol. 66 (2), 277–286.

Corti, R., Datry, T., 2014. Drying of a temperate, intermittent river has little effect on adjacent riparian arthropod communities. Freshw. Biol. 59 (4), 666–678.

Coste, M., 1982. Étude des méthodes biologiques d'appréciation quantitative de la qualité des eaux Rapport Cemagref QE Lyon-AF Bassin Rhône Méditerranée Corse.

De Cáceres, M., Jansen, F., De Caceres, M.M., 2016. Package 'indicspecies'. indicators 8 (1).

De Castro, P., Miglietta, P.P., Vecchio, Y., 2020. The Common Agricultural Policy 2021–2027: a new history for European agriculture. Ital. Rev. Agric. Econ. 75 (3), 5–12.

Descy, J.P., Coste, M., 1991. A test of methods for assessing water quality based on diatoms. Verh. Int. Ver. Theor. Angew. Limnol. 24 (4), 2112–2116.

Didham, R.K., Barker, G.M., Bartlam, S., Deakin, E.L., Denmead, L.H., Fisk, L.M., ... Schipper, L.A., 2015. Agricultural intensification exacerbates spillover effects on soil biogeochemistry in adjacent forest remnants. PLoS One 10 (1), e0116474.

Dvorák, L., Budrys, E., Cetkovic, A., Springate, S., 2010. Assemblages of social wasps in forests and open land across Europe – an ALARM-FSN Study. Chapter 2 Atlas of Biodiversity Risk, pp. 50–51.

Elosegi, A., Díez, J., Mutz, M., 2010. Effects of hydromorphological integrity on biodiversity and functioning of river ecosystems. Hydrobiologia 657 (1), 199–215.

European Environment Agency [EEA] (Ed.), 2012. European Waters — Assessment of Status 640 and Pressures. European Environment Agency, Copenhagen.

Feld, C.K., Fernandes, M.R., Ferreira, M.T., Hering, D., Ormerod, S.J., Venohr, M., et al., 2018. Evaluating riparian solutions to multiple stressor problems in river ecosystems—a conceptual study. Water Res. 139, 381–394.

Flávio, H.M., Ferreira, P., Formigo, N., Svendsen, J.C., 2017. Reconciling agriculture and stream restoration in Europe: a review relating to the EU Water Framework Directive. Sci. Total Environ. 596, 378–395.

García-Galán, M.J., Garrido, T., Fraile, J., Ginebreda, A., Díaz-Cruz, M.S., Barceló, D., 2010. Simultaneous occurrence of nitrates and sulfonamide antibiotics in two ground water bodies of Catalonia (Spain). J. Hydrol. 383 (1–2), 93–101.

Gavin, M.C., McCarter, J., Mead, A., Berkes, F., Stepp, J.R., Peterson, D., Tang, R., 2015. Defining biocultural approaches to conservation. Trends Ecol. Evol. 30 (3), 140–145.

Gibbs, J.P., Stanton, E.J., 2001. Habitat fragmentation and arthropod community change: carrion beetles, phoretic mites, and flies. Ecol. Appl. 11 (1), 79–85.

Goldsmith, T.H., 1990. Optimization, constraint, and history in the evolution of eyes. Q. Rev. Biol. 65 (3), 281–322.

Guillet, D., 2006. Rethinking irrigation efficiency: chain irrigation in northwestern Spain. Hum. Ecol. 34 (3), 305–329.

Guth, M., Smędzik-Ambroży, K., Czyżewski, B., Stępień, S., 2020. The economic sustainability of farms under common agricultural policy in the European Union countries. Agriculture 10 (2), 34.

Hadjicharalampous, E., Kalburtji, K.L., Mamolos, A.P., 2002. Soil arthropods (Coleoptera, Isopoda) in organic and conventional agroecosystems. Environ. Manag. 29 (5), 683–690.

Heino, J., 2010. Are indicator groups and cross-taxon congruence useful for predicting biodiversity in aquatic ecosystems? Ecol. Indic. 10 (2), 112–117.

Hering, D., Johnson, R.K., Kramm, S., Schmutz, S., Szoszkiewicz, K., Verdonschot, P.F., 2006. Assessment of European streams with diatoms, macrophytes, macroinvertebrates and fish: a comparative metric-based analysis of organism response to stress. Freshw. Biol. 51 (9), 1757–1785.

Real Decreto 817/2015, de 11 de septiembre, por el que se establecen los criterios de seguimiento y evaluación del estado de las aguas superficiales y las normas de calidad ambiental. Boletín Oficial del Estado, 219, de 12 de septiembre de 2015. https://www. boe.es/buscar/pdf/2015/BOE-A-2015-9806-consolidado.pdf.

Decreto 60/2009, do 26 de Febreiro, polo que se establece a relación de actividades potencialmente contaminantes do solo e os criterios e estándares para a declaración de solos contaminados. Diario Oficial de Galicia, 57, de 24 de marzo de 2009. https:// www.xunta.gal/dog/Publicados/2009/20090324/Anuncio10CC6_es.html.

Hynes, H.B.N., 1975. The stream and its valley. Verh. Int. Ver. Theor. Angew. Limnol. 19, 1–15.

Jähnig, S.C., Brunzel, S., Gacek, S., Lorenz, A.W., Hering, D., 2009. Effects of re-braiding measures on hydromorphology, floodplain vegetation, ground beetles and benthic invertebrates in mountain rivers. J. Appl. Ecol. 46 (2), 406–416.

Januschke, K., Brunzel, S., Haase, P., Hering, D., 2011. Effects of stream restorations on riparian mesohabitats, vegetation and carabid beetles. Biodivers. Conserv. 20 (13), 3147–3164.

Johnson, K.M., Lichter, D.T., 2019. Rural depopulation: growth and decline processes over the past century. Rural. Sociol. 84 (1), 3–27.

Kappes, H., Topp, W., Zach, P., Kulfan, J., 2006. Coarse woody debris, soil properties and snails (Mollusca: Gastropoda) in European primeval forests of different environmental conditions. Eur. J. Soil Biol. 42 (3), 139–146.

Kemp, P.S., O'hanley, J.R., 2010. Procedures for evaluating and prioritising the removal of fish passage barriers: a synthesis. Fish. Manag. Ecol. 17 (4), 297–322.

Kendall, M.G., Gibbons, J.D., 1990. Rank Correlation Methods. Oxford University Press, New York, NY.

King, S.E., Osmond, D.L., Smith, J., Burchell, M.R., Dukes, M., Evans, R.O., et al., 2016. Effects of riparian buffer vegetation and width: a 12-year longitudinal study. J. Environ. Qual. 45 (4), 1243–1251.

Kreyling, J., Schweiger, A.H., Bahn, M., Ineson, P., Migliavacca, M., Morel-Journel, T., ... Larsen, K.S., 2018. To replicate, or not to replicate–that is the question: how to tackle nonlinear responses in ecological experiments. Ecol. Lett. 21 (11), 1629–1638.

Lambeets, K., Hendrickx, F., Vanacker, S., Van Looy, K., Maelfait, J.P., Bonte, D., 2008. Assemblage structure and conservation value of spiders and carabid beetles from restored lowland river banks. Biodivers. Conserv. 17 (13), 3133–3148.

Lecointe, C., Coste, M., Prygiel, J., 1999. Omnidia, versión 3: notice d'utilisation CLCl Monbazillac.

Leibundgut, C., Kohn, I., 2014. European traditional irrigation in transition part II: traditional irrigation in our time—decline, rediscovery and restoration perspectives. Irrig. Drain. 63 (3), 294–314.

Li, X., Zhang, J., Gong, Y., Liu, Q., Yang, S., Ma, J., ... Hou, H., 2020. Status of copper accumulation in agricultural soils across China (1985–2016). Chemosphere 244, 125516.

Lindenmayer, D.B., Westgate, M.J., 2020. Are flagship, umbrella and keystone species useful surrogates to understand the consequences of landscape change? Curr. Landsc. Ecol. Rep. 5, 76–84.

Lomolino, M.V., 2000. Ecology's most general, yet protean pattern: the species-area relationship. J. Biogeogr. 17–26.

Lozanovska, I., Rivaes, R., Vieira, C., Ferreira, M.T., Aguiar, F.C., 2020. Streamflow regulation effects in the Mediterranean rivers: how far and to what extent are aquatic and riparian communities affected? Sci. Total Environ. 749, 141616.

MacDonald, D., Crabtree, J.R., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., et al., 2000. Agricultural abandonment in mountain areas of Europe: environmental consequences and policy response. J. Environ. Manag. 59 (1), 47–69.

Maceda-Veiga, A., 2013. Towards the conservation of freshwater fish: Iberian Rivers as an example of threats and management practices. Rev. Fish Biol. Fish. 23 (1), 1–22.

Maceda-Veiga, A., Basas, H., Lanzaco, G., Sala, M., De Sostoa, A., Serra, A., 2016. Impacts of the invader giant reed (*Arundo donax*) on riparian habitats and ground arthropod communities. Biol. Invasions 18 (3), 731–749.

Maceda-Veiga, A., Mac Nally, R., de Sostoa, A., 2019. Congruence in riverine conditions and associations between native fish and several species of amphibians in a region prone to fish invasions. Hydrobiologia 836 (1), 109–122.

Maceda-Veiga, A., Albacete, S., Carles-Tolrá, M., Pujade-Villar, J., Máca, J., Mac Nally, R., 2021. Streams and rural abandonment are related to the summer activity of the invasive pest *Drosophila suzukii* in protected European forests. For. Ecol. Manag. 485, 118942.

Macías-Vázquez, F., Calvo de Anta, R., 2009. Niveles genéricos de referencia de metales pesados y otros elementos traza en suelos de Galicia. Consellería de Medio Ambiente e Desenvolvemento Sostible. Xunta de Galicia, p. 232.

Maisonneuve, C., Rioux, S., 2001. Importance of riparian habitats for small mammal and herpetofaunal communities in agricultural landscapes of southern Québec. Agric. Ecosyst. Environ. 83 (1–2), 165–175.

Manko, P., Demková, J., Kohútová, M., Oboňa, J., 2019. Efficiency of traps in collecting selected Diptera families according to the used bait: comparison of baits and mixtures in a field experiment. Eur. J. Ecol. 4 (2), 92–99.

Márquez-Castro, B., 2014. Entre pozas y presas: el principal sistema de regadío tradicional de Galicia: morfología tradicional, lucha contra la industrialización tardía gallega y regulación comunitaria de su agua. In: Sanchis-Ibor, C., Palau-Salvador, G., Mangue Alférez, I., Martínez-Sanmartín, L.P. (Eds.), Irrigation, Society, Landscape. Tribute to Thomas F. Glick. Universitat Politècnica de València, València.

Marshall, S.A., 2012. Flies: The Natural History and Diversity of Diptera. Fire y Books.

Mason, C.F., 1970. Food, feeding rates and assimilation in woodland snails. Oecologia 4 (4), 358–373.

Matas, A., Mac Nally, R., Albacete, S., Carles-Tolrá, M., Domènech, M., Vives, E., et al., 2021. Wild boar rooting and rural abandonment may alter food-chain length in arthropod assemblages in a European forest region. For. Ecol. Manag. 479, 118583.

McColl, R.H.S., 1974. Self-purification of small freshwater streams: phosphate, nitrate, and ammonia removal. N. Z. J. Mar. Freshw. Res. 8 (2), 375–388.

Merçon, J., Vetter, S., Tengö, M., Cocks, M., Balvanera, P., Rosell, J.A., Ayala-Orozco, B., 2019. From local landscapes to international policy: contributions of the biocultural paradigm to global sustain- ability. Global Sustain. 2 (e7), 1–11.

Montgomery, G.A., Belitz, M.W., Guralnick, R.P., Tingley, M.W., 2020. Standards and best practices for monitoring and benchmarking insects. Front. Ecol. Evol. 8, 513.

Monroy, M., Maceda-Veiga, A., de Sostoa, A., 2014. Metal concentration in water, sediment and four fish species from Lake Titicaca reveals a large-scale environmental concern. Sci. Total Environ. 487, 233–244.

Munn, N.L., Meyer, J.L., 1990. Habitat-specific solute retention in two small streams: an intersite comparison. Ecology 71 (6), 2069–2082.

Munné, A., Prat, N., Solà, C., Bonada, N., Rieradevall, M.J.A.C.M., 2003. A simple field method for assessing the ecological quality of riparian habitat in rivers and streams: QBR index. Aquat. Conserv. Mar. Freshwat. Ecosyst. 13 (2), 147–163.

Murcia, C., 1995. Edge effects in fragmented forests: implications for conservation. Trends Ecol. Evol. 10 (2), 58–62.

Nakagawa, S., Schielzeth, H., 2013. A general and simple method for obtaining R^2 from generalized linear mixed-effects models. Methods Ecol. Evol. 4 (2), 133–142.

Newbold, J.D., Elwood, J.W., O'Neill, R.V., Winkle, W.V., 1981. Measuring nutrient spiralling in streams. Can. J. Fish. Aquat. Sci. 38 (7), 860–863.

Newbold, J.D., Herbert, S., Sweeney, B.W., Kiry, P., Alberts, S.J., 2010. Water quality functions of a 15-year-old riparian forest buffer system 1. JAWRA J. Am. Water Resour. Assoc. 46 (2), 299–310.

Oksanen, et al., 2020. Community Ecology Package Version 2.5-7 Author Jari Oksanen, F. Guillaume Blanchet, Michael Friendly, Roeland Kindt, Pierre Legendre, Dan McGlinn, Peter R. Minchin, R. B. O'Hara, Gavin L. Simpson, Peter Solymos, M. Henry H. Stevens, Eduard Szoecs, Helene Wagner.

Ondina, P., Hermida, J., Outeiro, A., Mato, S., 2004. Relationships between terrestrial gastropod distribution and soil properties in Galicia (NW Spain). Appl. Soil Ecol. 26 (1), 1–9.

Paetzold, A., Yoshimura, C., Tockner, K., 2008. Riparian arthropod responses to flow regulation and river channelization. J. Appl. Ecol. 45 (3), 894–903.

Palmer, M.A., Bernhardt, E.S., Allan, J.D., Lake, P.S., Alexander, G., Brooks, S., ... Sudduth, E., 2005. Standards for ecologically successful river restoration. J. Appl. Ecol. 42 (2), 208–217.

Pe'er, G., Bonn, A., Bruelheide, H., Dieker, P., Eisenhauer, N., Feindt, P.H., ... Lakner, S., 2020. Action needed for the EU Common Agricultural Policy to address sustainability challenges. People Nat. 2 (2), 305–316.

Peterson, B.J., Wollheim, W.M., Mulholland, P.J., Webster, J.R., Meyer, J.L., Tank, J.L., ... Morrall, D.D., 2001. Control of nitrogen export from watersheds by headwater streams. Science 292 (5514), 86–90.

Pinay, G., Bernal, S., Abbott, B.W., Lupon, A., Marti, E., Sabater, F., Krause, S., 2018. Riparian corridors: a new conceptual framework for assessing nitrogen buffering across biomes. Front. Environ. Sci. 6, 47.

Prygiel, J., Coste, M., 2000. Guide méthodologique pour la mise en œuvre de l'Indice Biologique Diatomées NF T 90-354.

R Core Team, 2014. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria Available at:http://www.R-project.org/.

Ramey, T.L., Richardson, J.S., 2017. Terrestrial invertebrates in the riparian zone: mechanisms underlying their unique diversity. BioScience 67 (9), 808–819.

Renes, H., Centeri, C., Eiter, S., Gaillard, B., Kruse, A., Kucera, Z., et al., 2020. Water meadows as European agricultural heritage. Adaptive Strategies for Water Heritage, p. 107.

Riis, T., Kelly-Quinn, M., Aguiar, F.C., Manolaki, P., Bruno, D., Bejarano, M.D., ... Dufour, S., 2020. Global overview of ecosystem services provided by riparian vegetation. BioScience 70 (6), 501–514.

Robinson, C.T., Tockner, K., Ward, J.V., 2002. The fauna of dynamic riverine landscapes. Freshw. Biol. 47 (4), 661–677.

Rodrigues, E.N.L., Mendonça, M.D.S., Costa-Schmidt, L.E., 2014. Spider diversity responds strongly to edge effects but weakly to vegetation structure in riparian forests of Southern Brazil. Arthropod Plant Interact. 8 (2), 123–133.

Sabater, F., Butturini, A., Martĺ, E., Muñoz, I., Romaní, A., Wray, J., Sabater, S., 2000. Effects of riparian vegetation removal on nutrient retention in a Mediterranean stream. J. N. Am. Benthol. Soc. 19 (4), 609–620.

Sobral, M., 2021. All traits are functional: an evolutionary viewpoint. Trends Plant Sci. 26 (7), 674–676.

Stebbings, R., 2005. Reptiles and amphibians in water meadows. In: Everard, M. (Ed.), Water Meadows. Living Treasures in the English Landscape. Forrest Text, Ceredigion, pp. 77–82.

Tait, G., Grassi, A., Pfab, F., Crava, C.M., Dalton, D.T., Magarey, R., et al., 2018. Large-scale spatial dynamics of *Drosophila suzukii* in Trentino, Italy. J. Pest. Sci. 91 (4), 1213–1224.

Ter Braak, C.J., Prentice, I.C., 1988. A theory of gradient analysis. Adv. Ecol. Res. 18, 271–317.

Timms, L.L., Bowden, J.J., Summerville, K.S., Buddle, C.M., 2013. Does species-level resolution matter? Taxonomic sufficiency in terrestrial arthropod biodiversity studies. Insect Conserv. Divers. 6 (4), 453–462.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., Cushing, C.E., 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37 (1), 130–137.

Von Schiller, D., Acuña, V., Aristi, I., Arroita, M., Basaguren, A., Bellin, A., ... Elosegi, A., 2017. River ecosystem processes: a synthesis of approaches, criteria of use and sensitivity to environmental stressors. Sci. Total Environ. 596, 465–480.

Walther, B.A., Moore, J.L., 2005. The concepts of bias, precision and accuracy, and their use in testing the performance of species richness estimators, with a literature review of estimator performance. Ecography 28 (6), 815–829.

Wetzel, R.G., 1983. Limnology. 2nd edition. Saunders College Publishing, Philadelphia, PA, p. 204.

Zuur, A.F., Ieno, E.N., Elphick, C.S., 2010. A protocol for data exploration to avoid common statistical problems. Methods Ecol. Evol. 1 (1), 3–14.