Does land use impact on groundwater invertebrate diversity and functionality in alluvial wetlands?

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Abstract

Land use change, especially the transformation of riparian forest to agricultural fields, plays an important role in groundwater quality. Yet little is known about the effects of land use change on groundwater invertebrate assemblages and diversity. This study assesses for the first time the effect of land use (agricultural *vs*. forest use) on the groundwater invertebrate community of four river floodplains representative of variable conditions across SW Europe (the Garonne River in France, and the Bidasoa, Ebro and Tajo Rivers in Spain). Groundwater invertebrate and water samples were collected quarterly in 8 to 12 piezometers located in each floodplain over a year. Taxonomic and functional diversity indices and ecosystem services (i.e. biogeochemical filtration and particulate organic matter breakdown) were calculated. The taxonomic and functional diversity of groundwater invertebrate communities increased linearly with the increasing surface occupied by riparian forests and decreased under intensified agriculture use. Moreover, the provision of key ecosystem services related to the biogeochemical filtration and particulate organic matter breakdown also increased 2-fold under more natural forest land. According to regression models, this is likely due to the pressure of agricultural practices on groundwater quality, with high concentrations of nitrates and sulphates exerting a negative influence over groundwater invertebrate diversity and their associated ecosystem services. The results of this study have important management implications, and suggest that the presence of large riparian corridors enhances groundwater invertebrate diversity and functionality in floodplains deeply disturbed by agricultural practices.

Highlights

- We study the effect of land use on groundwater invertebrate community.
- Groundwater invertebrate community change between land use types.
- Invertebrate diversity and ecosystem services increases with forest occupancy.
- Agriculture has a negative influence on groundwater invertebrate communities.
- Water quality controls groundwater invertebrate diversity and ecosystem services.

Keywords

Biogeochemical filtration; biological traits; ecosystem services; particulate organic matter breakdown; floodplain; stygofauna.

1. Introduction

Subterranean habitats have been traditionally considered as a biological desert, only interesting as a major source of water resources (Gibert et al., 1994). However, recent studies have demonstrated that groundwater systems are home to a great variety of cryptic and endemic organisms (Avramov, 2014; Deharveng et al., 2009; Sket, 1999), which provide important ecosystem services ranging from water purification and bioremediation to water infiltration and transport (Boulton et al., 2008). Unfortunately, the spread of groundwater habitats has also favoured their degradation, mainly because of intensification of agriculture, forest clearance, industrial activities, groundwater extraction, river flow regulation, and waste water discharge (Boulton et al., 2010; Danielopol et al., 2003; Griebler et al., 2010; Tockner et al., 2008). These pressures reduce the hydrological connection between ground and surface waters, and favour the accumulation of pollutants that impair the characteristic biological structure and functionality of groundwaters (Danielopol et al., 2003; Korbel et al., 2013).

Since the early 2000s, the importance of groundwater as a living ecosystem has increased. Scientists and environmental managers are working to develop guidelines and strategies for the conservation of groundwater ecosystems, taking into account their biota and functionality (Gibert and Culver, 2009; Gibert et al., 2009; Gibert and Deharveng, 2002). The majority of studies focus on describing groundwater invertebrate (also called stygofauna) distribution patterns, such as the recent European PASCALIS project (http://pascalis.univ-lyon1.fr/; Castellarini et al., 2007; Gibert et al., 2009). Results from this project have identified temperate areas as hot-spots of groundwater biodiversity, and have suggested that the full potential of groundwater ecosystems as sources of biodiversity and ecosystem services is still to be fully comprehended (Deharveng et al., 2009). For this reason, researchers have proposed the use of groundwater invertebrates as bioindicators of ecological groundwater health in the same way that they are used in surface waters (Boulton et al., 2008; Iepure et al., 2013). The distribution of groundwater organisms indeed responds to hydrogeological and environmental factors, such as floods, dissolved organic carbon, and the concentration of nutrients and pollutants (Castellarini et al., 2007; Danielopol et al., 1994; Di Lorenzo et al., 2014; Iepure et al., 2014, 2013;), which makes them suitable candidates to reflect the ecological status of subterranean habitats. Furthermore, groundwater invertebrates contribute to the functioning of groundwater ecosystems, and consequently to the provision of groundwater ecosystem services used by humankind. For example, amphipods and isopods are keystone taxa groups that enhance the decomposition of organic matter, which in turn stimulates bacterial growth and activity, supporting the biodegradation of contaminants (Tomlinson and Boulton, 2010; Ward et al., 1998). These crustaceans also recycle nutrients through the excretion of faecal pellets, and increase the flux of oxygenated water through burrowing (Hakenkamp and Palmer, 2000). However, such functional processes have been poorly studied, preventing the development of consistent ecological criteria for assessing groundwater ecological status. More research is thus needed to ascertain the response of subterranean biodiversity patterns and functionality to both natural and human-induced changes (Culver and Sket, 2000; Deharveng et al., 2009; Hancock et al., 2005; Korbel et al., 2013).

One of the most interesting subterranean habitats is the hyporheic zone, which is the area of mixing between surface and subsurface water (White, 1993; Wondzell, 2011). As surface water contains rich oxygen and organic matter and groundwater contains abundant nutriment elements, the water mix between those two systems has a significant impact on water quality, ecosystems and biogeochemistry cycling (Boulton et al., 1998; Brunke and Gonser, 1997; Krause et al., 2013; Marmonier et al., 2012; Sanchez-Pérez et al., 2003; Sánchez-Pérez and Trémolières, 2003; Vervier et al., 2009). These ecosystems support important agricultural activities, and, consequently, groundwater in alluvial plains often suffer from nitrate pollution (Arrate et al., 1997; Almasri and Kaluarachchi, 2007; Liu et al., 2005; Sánchez-Pérez et al., 2003a). Several studies show that the hyporheic zone (i.e. surface-groundwater interface) contributes to nitrogen retention and/or transformation of the land-surface water continuum (Sabater et al., 2003; Weng et al., 2003). This zone supports the purification of water by its ability to eliminate nitrates during their infiltration through the vegetation-soil system to groundwater, but also through diffusion from groundwater to surface water (Sanchez-Perez et al., 1991a, 1991b; Takatert et al., 1999). However, little is known about the groundwater biota and its role in these purification processes, and, definitely in the general ecosystem functioning.

To this end, this study evaluates for the first time the effect of land use on the groundwater invertebrate community of the hyporheic zone in four river floodplains located across SW Europe. First, we hypothesized that forest areas would show higher groundwater invertebrate diversity than agricultural areas, since forests provide high heterogeneity and quality in terms of resources (Brunke and Gonser, 1997; Datry et al., 2005). Moreover, we expected the groundwater invertebrate community to change along a land use gradient, showing varying characteristic taxa and functions. For example, the vegetation of forest areas adds debris to the system, increasing the organic matter content in soils and groundwater. Consequently, the presence of detritivores and omnivores, which are essential for organic matter decomposition and nutrient cycling in groundwater ecosystems, is favoured (Avramov, 2014; Boulton et al., 2008; Vannote et al., 1980). In contrast, pollution-tolerant taxa are likely to dominate agricultural areas because of the high levels of fertilizers and manures employed, as was observed by Dumas and Lescher-Moutoué (2001). Thus, our study allows us to evaluate the overall role of riparian corridors as natural biofilters of pollutants from agricultural areas. Ultimately, this study is essential for understanding the local variation in groundwater assemblages and their response to human pressures, contributing to the development of suitable conservation and management policies.

2. Material and methods

2.1 Study area

For this study, we selected four river floodplains located in South-western Europe: Garonne, Ebro, Bidasoa and Tajo. Floodplains were selected in areas characterized by a combination of agricultural occupation and patches of natural riparian forest that might favour the potential degradation of agricultural pollutants. The study covers a wide gradient of climatic and environmental conditions, from floodplains with little forest use to others with a high forest surface. To our knowledge, no studies on groundwater invertebrates have been performed in any of the four selected floodplains.

The Garonne study area was located 30 km north of Toulouse in South-West France (UTM 355758/4861267; Figure 1, Table 1). This floodplain site is dominated by agriculture, mainly corn, sunflower and poplar plantations (Table 1). Its riparian forest is dominated by poplar (*Populus* spp.), ash (*Fraxinus* spp.) and oak (*Quercus* spp.). The river flow at the study site is very variable, peaking in winter and spring (2880 $m^3 \cdot s^{-1}$, during the study period that corresponds to a 10 years return period) and declining in summer (47 $m^3 \cdot s^{-1}$).

The Ebro study site was located 12 km downstream of Zaragoza, Spain (UTM 686179/4607010; Figure 1, Table 1). It is mostly subjected to agricultural land use; mainly corn and irrigated cereals, but it shows an extensive riparian corridor dominated by white poplar (*Populus alba*), European black poplar (*Populus nigra*), white willow (*Salix alba*) and salt cedar (*Tamarix* spp.) (González et al., 2010). River discharge varied from 205 to 1450 $m^3 \cdot s^{-1}$ during the study period.

The Bidasoa study area was located 7 km upstream of Irun, Spain (UTM 602190/4797250, Figure 1). Agriculture (mainly corn and pasture) and livestock practically cover the entire floodplain surface, leaving a tight riparian corridor along the river shore (Table 1). White willow (*Salix alba*), alder (*Alnus glutinosa*) and ash (*Fraxinus excelsior*) dominate this riparian corridor, and the invasive species American pokeweed (*Phytolacca Americana*) is also abundant. River discharge varied between 9 and 630 $m^3 \cdot s^{-1}$ during the studied period.

The Tajo study site was located 30 km downstream of Toledo, Spain (UTM 380482/4410115; Figure 1, Table 1). This floodplain site is under extensive agriculture; mainly irrigated cereals, corn and pasture (Table 1). Only some isolated patches of riparian forest remain, which are dominated by white poplar (*Populus alba*) and salt cedar (*Tamarix* spp.). The river is highly regulated by dams and water derivations, with a discharge that hardly varied during the study period (10 to 12 $m^3 \cdot s^{-1}$).

The piezometric network in each floodplain was designed considering the floodplain surface and land use. For this study, we sampled 11 piezometers in the Garonne River, 12 piezometers in the Ebro River, 7 piezometers in the Bidasoa River, and 10 piezometers in the Tajo River, covering different percentages of land use surface (see Figure 1). More detailed information about the piezometric network design, environmental conditions, and hydrodynamics of each studied area can be found in Antiguedad et al. (to be submitted in the same special issue) and Bernard-Jannin et al. (to be submitted in the same special issue).

We calculated the surface occupied by each type of land use within a buffer of 50 m around each piezometer using QGIS v2.6.1. For the buffer selection, we took into account the groundwater velocity in the alluvial aquifers (i.e. hydraulic conductivity). For the hydraulic conductivity range (10^{-3} to 10^{-5} $m^3 \cdot s^{-1}$) in our studied areas, we estimated that a particle (e.g. nutrients, pollutants) would move between 1 and 100 metres per day. It is for this reason that we selected an average surface of influence (i.e. buffer) of 50 metres. Each piezometer was classified as agriculture-dominated when this land use represented ≥ 40% of the buffer surface and otherwise as forest-dominated.

2.2 Taxonomic and functional composition of the invertebrate community

One groundwater invertebrate sample was collected at each piezometer every three months for one year: (i.e. April/May 2013, July/August 2013, October/November 2013 and January/February 2014). This meant that 90-100 L of groundwater were extracted at each piezometer with a manual pump (based on Malard et al., 2002) and filtered through a net with a mesh of 65 µm. Samples were preserved in situ in 70% ethanol. Groundwater invertebrate samples were sorted and identified in the laboratory at least to order level (Table A1, Appendix).

At the taxonomic level, three metrics were calculated: the total abundance of individuals, the total richness of taxa, and the Shannon-Wiener diversity index. To characterize the functional composition of the macroinvertebrate community, we used feeding habits, the biogeochemical filtration capacity, and the particulate organic matter (POM) breakdown capacity. In terms of feeding habits, we used affinity scores from zero to five for each of the following habits: absorber, deposit feeder, shredder, scraper, filter-feeder, piercer, predator, and parasite. We used a fuzzy coding approach based on scores published in Tachet et al. (2002). For those taxa not included in this reference guide, we classified organisms using information from the scientific literature as well as expert consultation (Table A2, Appendix). A score of zero indicated no affinity, and a score of five indicated the highest affinity of the taxon to a particular feeding habit. For taxa identified at higher taxonomic levels than genera, the most frequent score across all taxa belonging to a particular taxonomic group was selected. This may have resulted in a certain underestimation of functional diversity, although according to Dolédec et al. (2000) the overall functional structure of the invertebrate communities is conserved. The matrix containing sites per taxa abundance was multiplied by the taxa per feeding habit matrix to calculate three functional diversity metrics: abundance of individuals showing affinity for each feeding habitat (i.e. functional abundance), total richness of feeding habits (i.e. functional richness), and the Shannon-Wiener diversity of feeding habits (i.e. functional Shannon diversity index). To assess the biogeochemical filtration and POM breakdown capacities of the community, we used the functional scores defined by Boulton et al. (2008). Thus, for each taxon, a biogeochemical filtration and POM breakdown efficiency score was assigned: 0: no or unknown direct role, 1: minor role, 2: moderate role, and 3: major role (Table A2, Appendix). We expressed the biogeochemical filtration capacity of the groundwater community as the product of the absolute abundance of each taxon by its efficiency score.

Taxonomic and functional diversity metrics were computed with the ‘vegan’ (Oksanen et al., 2013) and ‘ade4’ (Thioulouse et al., 1997) packages of R software, 2.15.3 (R Core Team, 2013).

2.3 Environmental variables

Water samples were collected at each piezometer with a submersible pump. To ensure that the water sample corresponded to the aquifer and not stagnant water accumulated in the piezometer, we previously extracted groundwater until conductivity and oxygen values were constant (Griebler et al., 2010; Sánchez-Pérez, 1992). Water samples were transferred into a 1.5 L PVC bottle previously washed in acid (ClH 0.1 N), and placed on ice for transportation to the laboratory (see total number of samples in Table 1).

The alkalinity of unfiltered water samples was estimated within 4 h of collection by automatic titration with H_2SO_4 0.04 N (APHA, 1989). Total suspended solids, total dissolved solids and organic matter content were determined by the gravimetric method, i.e. filtering of samples through pre-combusted (450ºC, 4 h) Whatman GF/F glass-fibre filters following standard protocols (APHA, 1989). Filtered water aliquots were stored at -20 ºC and used within one month for the following analyses. Ion chromatography (Metrohm 861 Advanced Compact IC; APHA, 1989) was applied to determine ammonia (NH_4^+), nitrate (NO_3^-) and sulphate (SO_4^{2-}) concentration. Modified Berthelot reaction using salicytate and dichloroisocyanurate was used to determine the amount of ammonia (NH_4^+) by the colorimetric method (Krom, 1980) for Bidasoa River samples. Soluble reactive phosphorus (SRP) was measured by the ascorbic acid method and determined by the colorimetric method (APHA, 1989). Total dissolved phosphorus (TDP) was also estimated by the ascorbic acid method, but potassium persulphate digestion was performed beforehand (90 min, 115 °C) (APHA, 1989). Finally, water temperature, pH, conductivity, dissolved oxygen, and oxidation-reduction potential (ORP) were recorded in situ with portable probes. The depth of the water table was also measured with a sound piezometric probe.

2.4 Statistical analysis

All statistical analyses were based on log-transformer data (with the exception of water pH, left untransformed) to normalize distributions and linearize relationships. However, water physicochemical parameters and diversity metrics still showed a non-normal distribution (Kolmogorov-Smirnoff test, $P < 0.05$). Thus, non-parametric Mann-Whitney *U* (for two samples) and Kruskal-Wallis (for k samples) tests were applied to identify significant differences in diversity metrics between rivers, land use types within each river, and seasons.

The response of diversity indices (both taxonomic and functional) and ecosystem services (biogeochemical filtration and POM breakdown capacities) to increasing forest surface was plotted and evaluated by simple linear regression after checking that no increase in fitness was achieved when quadratic or logistic functions were used.

To evaluate changes in the composition of groundwater invertebrate assemblages in response to land use types, a similarity test was used. This test allows detection of the most characteristic taxa and functions associated to each land use, as well as those that better explain the dissimilarity between land use types. Similarity tests were performed for each floodplain separately by means of two-way SIMPER analysis with squared-root transformed abundances, Bray-Curtis similarity distance, and a cut-off for low contributions set at 90%. Taxa and feeding traits that contributed more to dissimilarity (i.e. with values higher than one of the quotient between average dissimilarity and standard deviation [Av. Dissimilarity/SD > 1]), were considered significant in terms of discriminating land use types (Clarke and Warwick, 2001; Ruhí et al., 2012).

Linear mixed-effect models (LME, Laird and Ware, 1982) were used to identify the environmental characteristics that controlled taxonomic and functional diversity. This statistical technique was used to avoid the co-dependence effect introduced by repeated measurements over time and across the four riparian areas (Demidenko, 2004). Physicochemical parameters (non-correlated, Spearman rank test $P < 0.6$) and forest surface percentage were included as fixed effects in LME models. Sampling season and floodplain identity were included as random factors. Model selection followed an automatic stepwise forward regression selection of predictors, based on the lowest Akaike Information Criteria (AIC) that quantify the goodness of fit of multiple alternative models.

Non-parametric analyses of variance (Mann-Whitney *U* test and Kruskal-Wallis test) and linear regressions were performed with SPSS version 18.0 (©SPSS, Inc., Chicago, IL). SIMPER tests were performed with PRIMER v. 6.0 for Windows. LME models were computed with the 'nlme' (Pinheiro et al., 2013), 'MuMIn' (Barton, 2013), 'MASS' (Venables and Ripley, 2002), 'lme4' (Bates et al. 2013), and 'effects' (Fox, 2003) packages in R version 2.15.3 (R Core Team, 2013).

3. Results

3.1 Groundwater invertebrate community and associated ecosystem services

Diversity indices and the associated ecosystem services (i.e. biogeochemical filtration and POM breakdown) showed a linear positive response with increasing forest land use. Moreover, the percentage of forest land use was able to explain between 14 and 52% of the diversity indices and associated ecosystem services of groundwater communities (Figures 2, 3, and A1, Appendix).

In general, the taxonomic diversity and composition of groundwater invertebrate communities showed differences between river and land use types (Table 1, Figures 4 and A2, Appendix). Particularly, more than 30% of the community in forest areas was composed of crustacea (mainly Copepoda and Amphipoda), and Oligochaeta was one of the most abundant taxa in agricultural areas. Moreover, SIMPER analysis showed that Amphipoda was one of the taxa that contributed most to the dissimilarity between land use types in three out of the four studied floodplains (Garonne, Ebro, and Bidasoa). The abundance of Ostracoda was particularly important in the dissimilarity between land uses in the Tajo River floodplain (Table 3, and A3, Appendix). At the functional level, the most abundant (ca. 50%) functional feeding group corresponded to deposit feeders (Figure 5), which also contributed most to the dissimilarity between land use types in all of the studied floodplains (SIMPER analysis, Table 3 and A4, Appendix). The proportion of shredders, scrapers, and filter-feeders was generally higher in forest than in agricultural areas (Figure 5). Diversity indices and the associated ecosystem services remained practically unchanged between sampling campaigns (i.e. seasons, Figure A3, Appendix).

3.2 Environmental factors controlling diversity indices and ecosystem services

In general, concentration values of dissolved oxygen and nitrate in piezometers located in agricultural areas were almost double those in forest areas, whereas conductivity and DOC increased under forest land use (Table 2).

According to LME, the percentage of forest area and water quality (mainly, nitrate, sulphate, phosphate, and DOC concentrations) explained between 51 and 71% of the response of diversity indices and the associated ecosystem services (Table 4; Figure A4, Appendix). Diversity indices and ecosystem services had a positive response to the increase in the forest use surface and the groundwater content of nitrates, sulphates, and DOC. Likewise, phosphate concentration had a negative influence on diversity indices (Table 4; Figure A4, Appendix).

4. Discussion

In this study we demonstrate that the diversity and functionality of groundwater invertebrates increase with the area occupied by riparian corridors in floodplains affected by intensive agricultural development. This is probably because riparian forests provide greater availability of quality resources (such as organic matter) to groundwater organisms and unpolluted groundwater. More importantly, riparian corridors supported organisms that provide important ecosystem services, such as biogeochemical filtration and POM breakdown. Similar results were observed in other studies (Boulton et al., 2008; Korbel and Hose, 2011), where changes in the composition, abundance, and richness of the groundwater fauna were directly related to ecosystem functions. Our study therefore highlights the need to conserve or even increase the forest area in floodplains affected by intense agricultural use as a means to create buffer areas that favour groundwater biodiversity and, consequently, the biogeochemical degradation of pollutants coming from agricultural areas.

4.1 How does land use affect groundwater biodiversity and functionality?

Groundwater invertebrate assemblages showed notable changes between land use types, at both taxonomic and functional levels. For instance, Amphipoda and Copepoda were the most abundant taxa in the forest areas, whereas Oligochaeta and Ostracoda were more frequent in agricultural areas. In the Bidasoa floodplain, where agricultural and livestock activities are intensively developed, we mainly found species associated with poor water quality, such as Oligochaeta, Diptera and Nematoda (see Figure 4). In contrast, Ostracoda, often associated with low groundwater level fluctuations (Malard et al., 1996), was the most abundant taxon in the Tajo River, which is consistent with the low river discharge fluctuations registered during the study period (10 to 12 $m^3 \cdot s^{-1}$). Similar results were found by Danielopol (et al., 2003), who suggested that river regulation combined with the negative effect of organic pollution strongly alters groundwater habitats. This is partly explained through river regulation which reduces water energy, thereby increasing the transport large amounts of fine sediments (e.g. silt and clay) that are accumulated in the aquifer, favouring its siltation. This siltation reduces the porosity of alluvial aquifers and, consequently, reduces the space to host fauna and reduces water and energy exchange between rivers and alluvial aquifers. Moreover, the high content of organic pollution together with the characteristic low oxygen concentration of alluvial aquifers favours the creation of chemical-reducing conditions that also eliminate groundwater fauna. Consequently, free-moving crustaceans such as stygobiotic copepods and isopods are replaced by epigean animals (i.e. surface animals), mostly nematodes and oligochaetes, which are able to live under these conditions (Danielopol et al., 2003).

Unlike agricultural use, forest use does not add pollutants to groundwater, favouring the provision of better-quality habitats and resources. Forest areas can thus act as local hotspots of food resources for groundwater invertebrates, thanks mainly to the organic matter provided by the vegetation (Boulton and Foster, 1998). More particularly, organic matter is degraded by surface fauna, increasing dissolved organic carbon content in soils and infiltrated water. Consequently, dissolved organic carbon content increases in groundwater under forest areas and is used as a food resource for many taxa, such as Amphipoda and Isopoda. The increase of this kind of taxon also favours better ecosystem functioning. For example, shredder crustaceans are keystone species for nutrient cycling in groundwater ecosystems (Avramov, 2014; Boulton et al., 2008) because of their role in bioturbation and the compaction of fine sediments into faecal pellets (Boulton et al., 2008). Such biological processes boost the substrate availability for bacteria, which in turn enhance denitrification processes and the transformation of micropollutants in groundwaters, thereby contributing to healthy waters (Gibert and Deharveng, 2002; Tomlinson and Boulton 2010; Ward et al., 1998; Yao et al., submitted). Consequently, floodplains dominated by forest land use, such as the Ebro and Garonne, showed the greatest biogeochemical filtration and POM breakdown capacities. It thus follows that the loss of groundwater invertebrates in agricultural areas, particularly of keystone species such as amphipods and isopods, can have serious repercussions for the functioning of groundwater ecosystems (Boulton et al., 2008; Korbel et al., 2013) and the associated supporting and provisioning ecosystem services (e.g. purification of water, sustainability of surface ecosystems, provision of fit/clean water for agricultural use) (Danielopol et al., 2003).

On the other hand, increase of nitrates, phosphates, and sulphates as a consequence of agricultural fertilization and pesticides application influenced the groundwater invertebrate community (Tables 2 and 4). These compounds can inhibit the development of key taxa (mainly amphipods and isopods) at high concentrations (Fakher et al., 1998), and favour the presence of other pollution-tolerant taxa such as Oligochaeta. Other studies (Di Lorenzo et al., 2014; Iepure et al., 2013; Yao et al., submitted) also observed a negative influence of human use (e.g. agricultural, urban or industrial) on groundwater invertebrate communities that was mainly associated with the deterioration of water quality and ecosystem functioning. In contrast, other authors (Boulton et al., 1998; Castellarini et al., 2007; Di Lorenzo et al., 2005; Dole-Olivier et al., 2009; Dumas and Lescher-Moutoué, 2001) found little evidence that land use and the associated changes in water chemistry affect the composition of stygobiotic assemblages. These authors emphasized the role of hydrological (e.g. hydraulic conductivity) and geomorphological (e.g. pore size) factors instead. This discordant result can be explained by the strong agricultural pressure in our studied areas, characterized by long periods of low discharges that usually coincide with fertilization and irrigation periods. Diffuse pollution from agricultural areas may thus override the effects of other hydrological and geomorphological determinants. However, the role of hydrological factors on groundwater invertebrate communities should not be disregarded, especially given its importance for surface waters (Español et al., 2015; Gallardo et al., 2008, 2009).

4.2 Conclusions

The results of this study demonstrate that land use impacts on groundwater biota and their associated ecosystem services. We observed a negative influence of agricultural land use on invertebrate diversity and associated ecosystem services, which is attributed to a high concentration of nitrates and sulphates. In contrast, riparian corridors provide food resources (dissolved organic matter) and better water quality (low nitrate and sulphate concentration) that enhance biota development and, consequently, the associated ecosystem services (e.g. POM breakdown and biogeochemical filtration capacities). Therefore, management measures to protect groundwater ecosystems should include the conservation, creation, and eventually expansion of riparian forest corridors in floodplains occupied by intensive agriculture. In addition, the rationalization of agricultural fertilization activities and pesticides application are needed to reduce diffuse contamination of subterranean habitats.

In terms of biomonitoring, our results suggest that groundwater invertebrate abundance, taxonomic richness, POM breakdown capacity, and biogeochemical filtration capacity are appropriate indicators for comparing the composition and functionality of alluvial aquifers and assessing the effect of human pressures. However, more detailed regional long-term studies are needed for better understanding of the structure and function of groundwater ecosystems, which remain poorly investigated. Such basic studies are essential for developing the ecological criteria proposed by the European standard (EU-GWD, 2006) for assessing groundwater ecosystem status at the same level as surface aquatic ecosystems.

5. Acknowledgements

This study was performed as part of the EU Interreg SUDOE IVB programme (ATTENAGUA - SOE3/P2/F558 project, http://www.attenagua-sudoe.eu) and funded by ERDF. The main author was funded by the Aragon government (Research Group E61 on Ecological Restoration and B079/09 pre-doctoral grant) and AISECO (student grant). This work has been made possible thanks to the work of the rest of

ATTENAGUA’s team: S. Teisser, L. Bernard-Jannin, E. Pinelli, O.Chamsi, X. Sun, A. Barcos, E. Navarro, J.J. Jiménez, M. Garcia, R. Sorando, R. Neves, D. Brito, J. Bodoque, I. Antiguedad, and E. Ruiz.

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Figure 1. Study area location and piezometric network of each studied river floodplain. The study area is composed by four river floodplains located in South-western Europe (Garonne, Ebro, Bidasoa and Tajo Rivers). Samples were collected in the piezometric network (8 to 12 piezometers) of each floodplain.

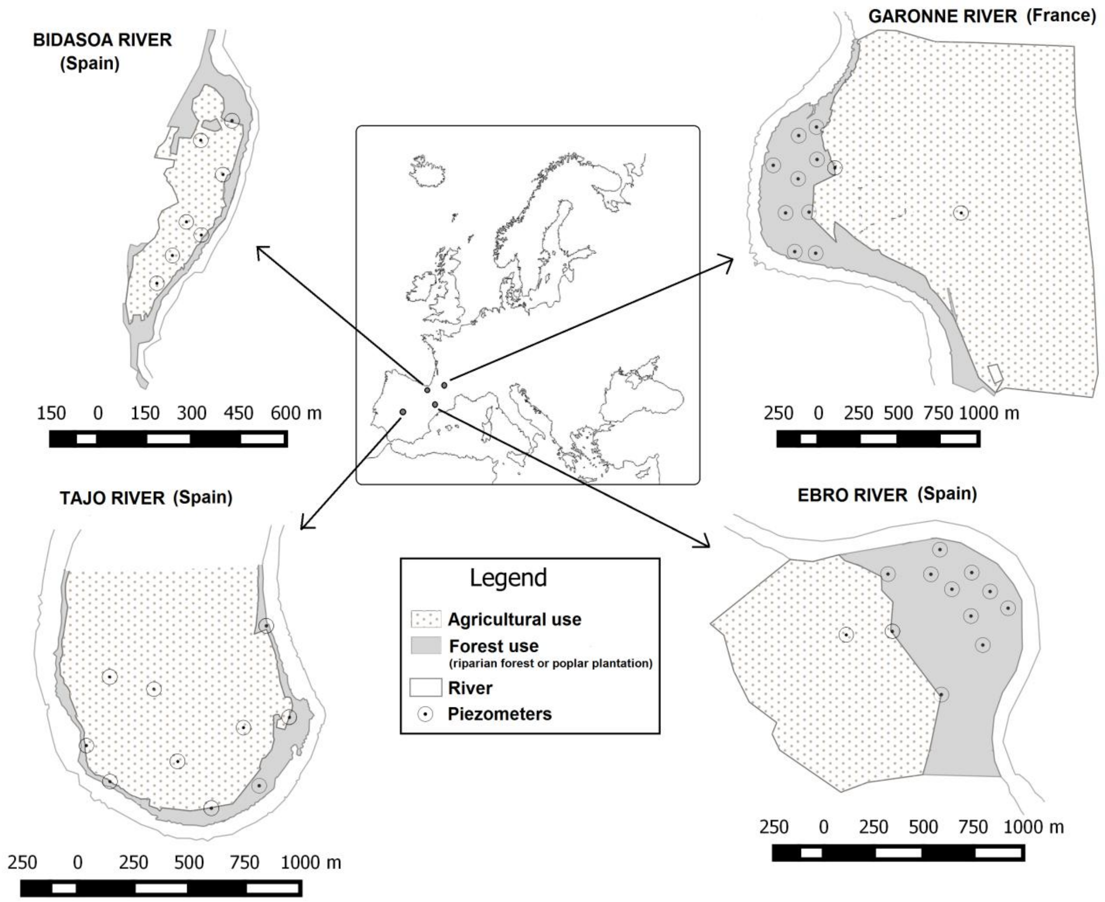

Figure 2. Relationship between taxonomic and functional diversity indices and the percentage of surface occupied by forest (buffer 50 m) in the floodplain of four rivers in South-western Europe.

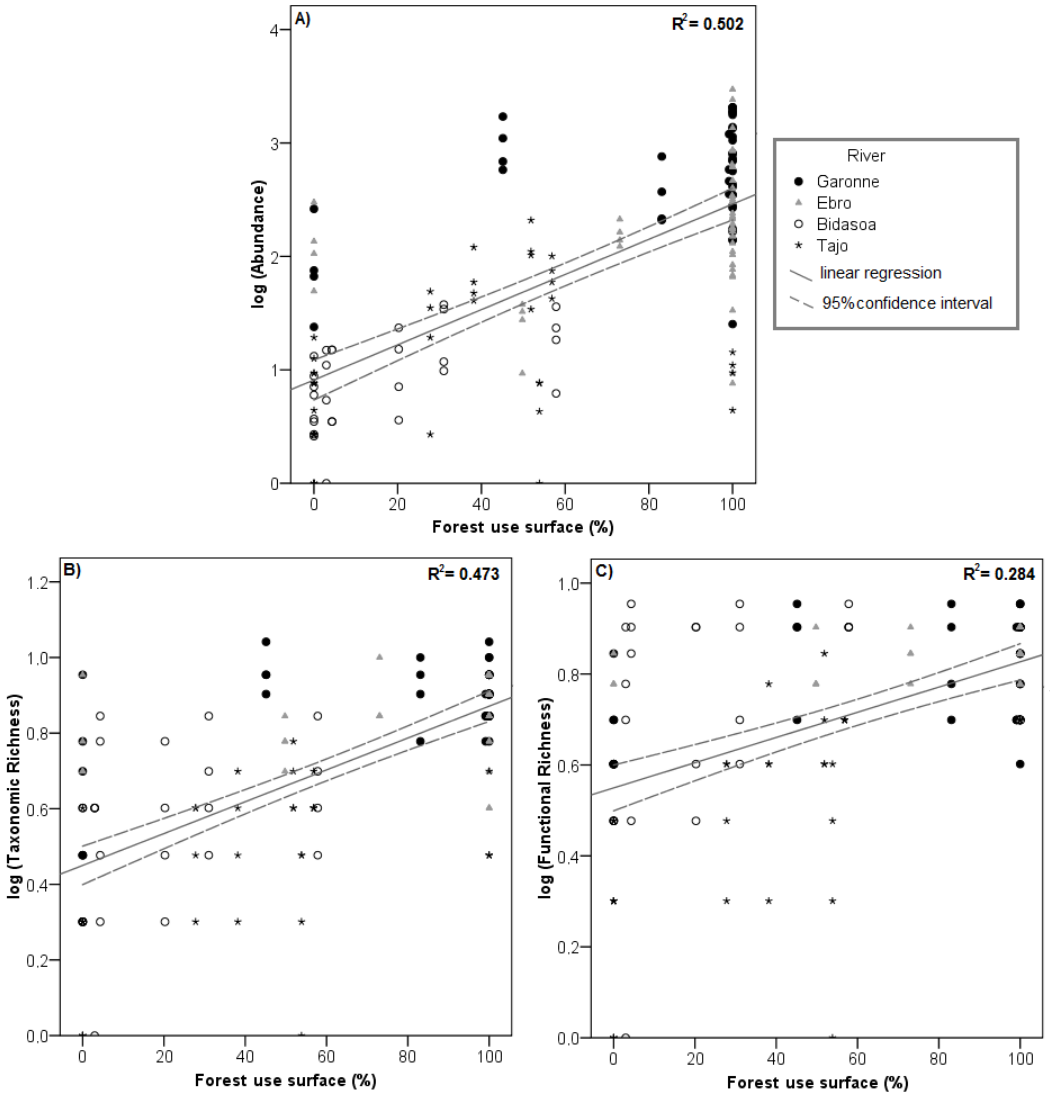

Figure 3. Relationship between ecosystem services (i.e. biogeochemical filtration and POM breakdown capacities) and the percentage of surface occupied by forest (buffer 50 m) in four river floodplains in South-western Europe.

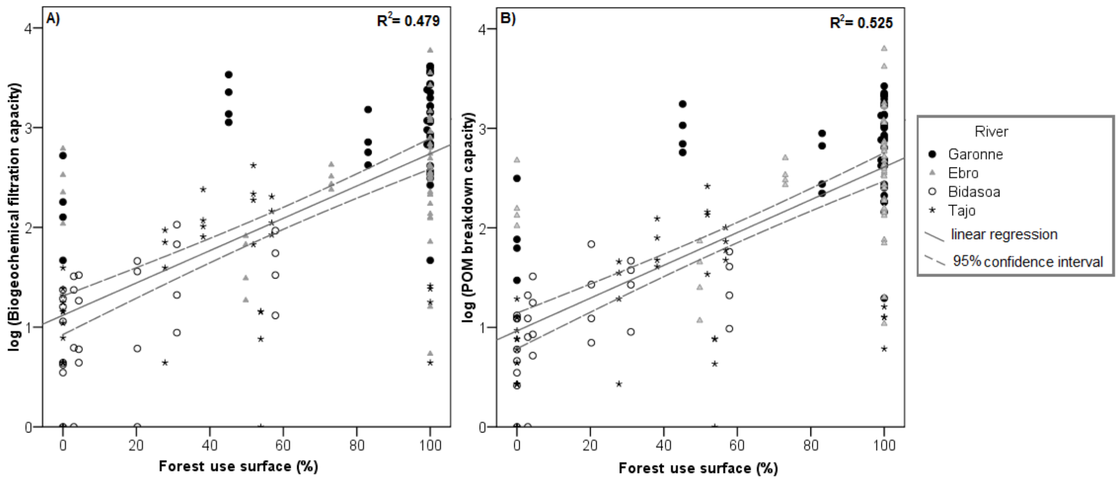

Figure 4. Taxonomic composition of the groundwater invertebrate community of four river floodplains according to their dominant land use type. Bars represent the relative abundance (average percentage) of each taxon in the community for each land use type. Significant differences between land use types are indicated by * (P < 0.05, Mann-Whitney U test) and ** (P < 0.01).

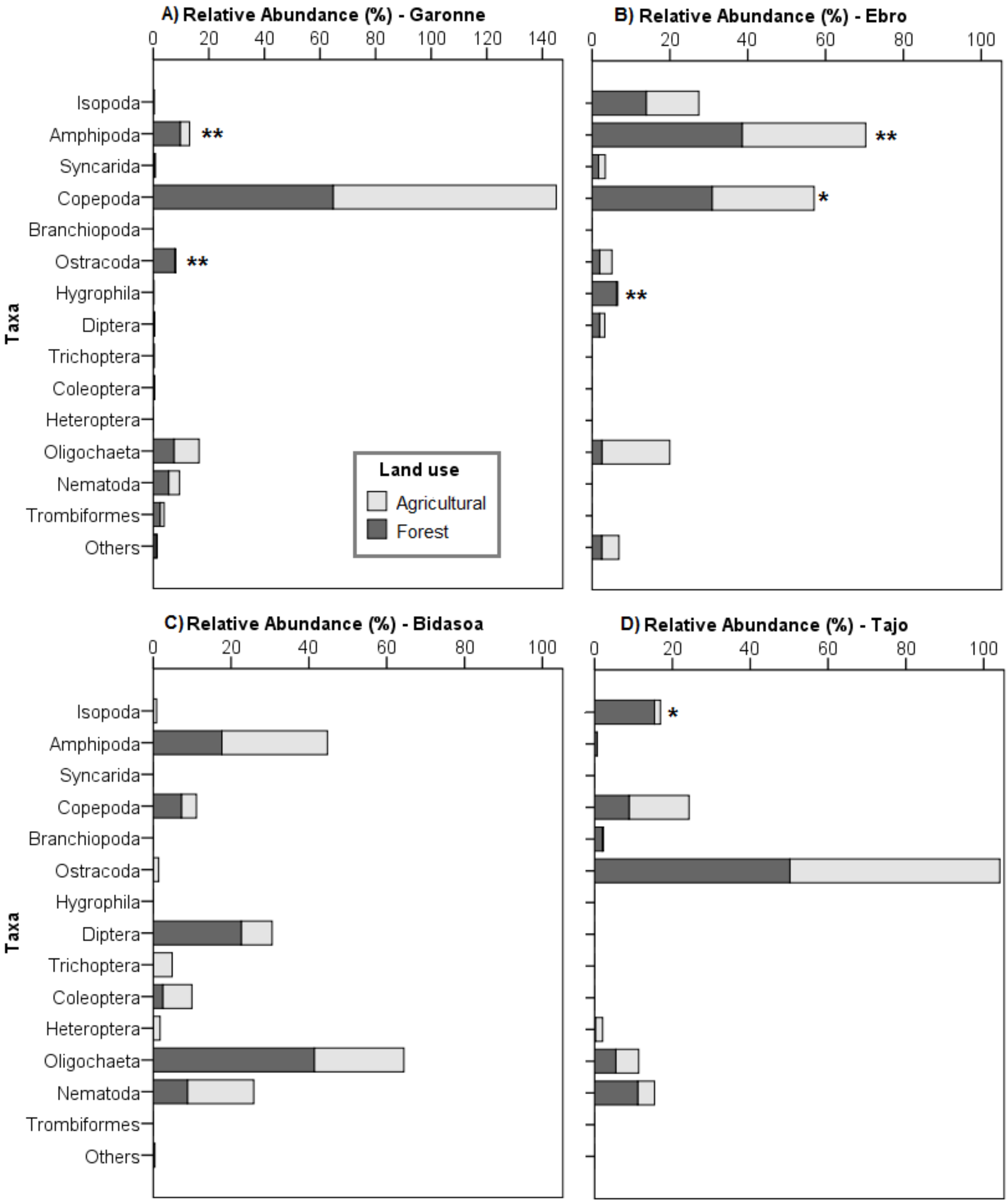

Figure 5. Functional feeding habits of groundwater invertebrates in four river floodplains according to their land use. Bars represent relative abundance (average percentage). Significant differences between land use types are indicated by * (P < 0.05, Mann-Whitney U test) and ** (P < 0.01).

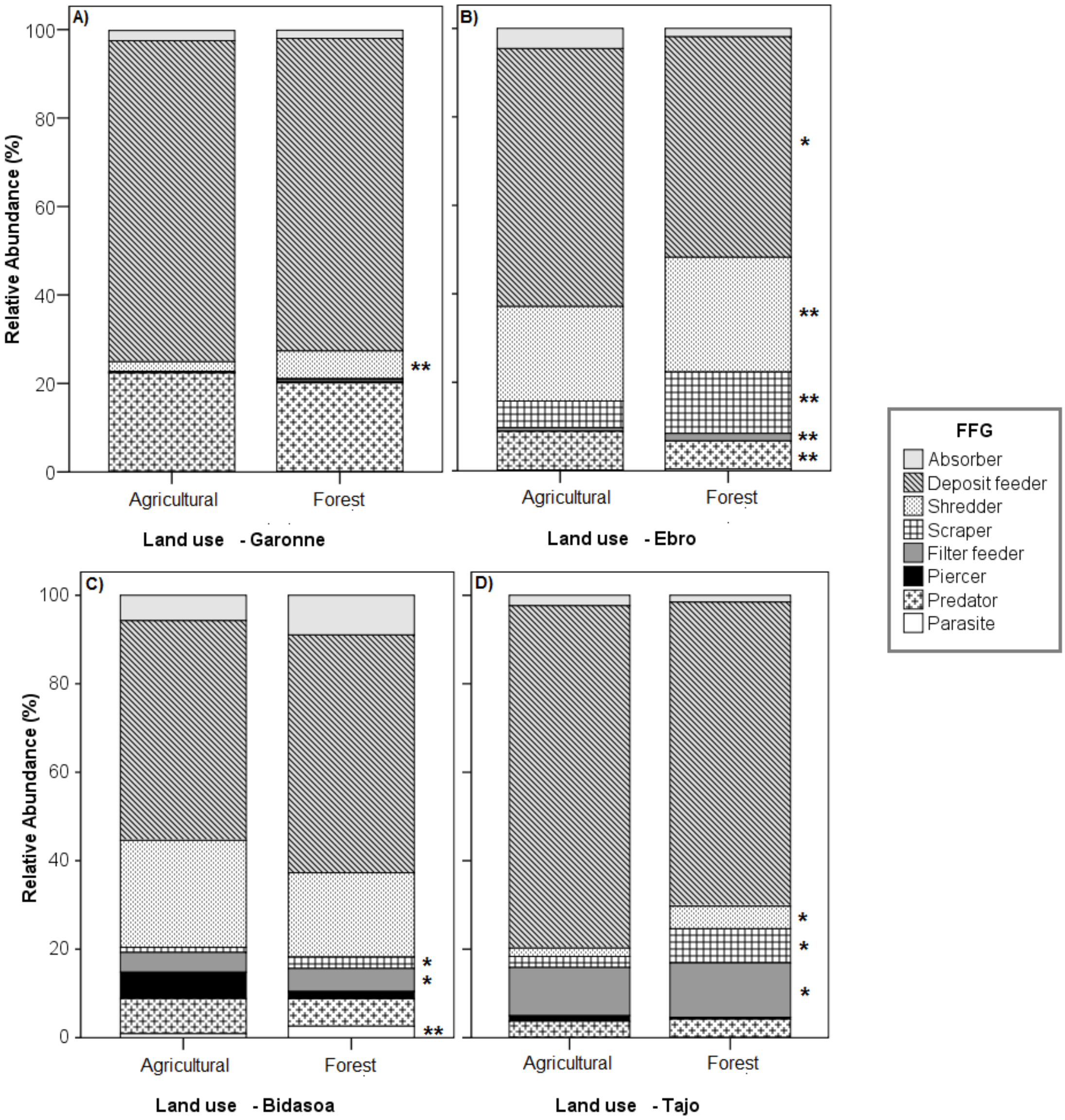

Table 1. Study site characteristics, dominating land use type, diversity metrics (mean ± SD), and provided ecosystem services (mean ± SD) in four river floodplains (Garonne, Ebro, Bidasoa and Tajo Rivers). N = sample size. Significant differences between river floodplains are indicated by ** (P < 0.05, Kruskal-Wallis test).

	Units	Garonne (N = 44)	Ebro (N = 48)	Bidasoa (N = 28)	Tajo (N = 40)
Piezometers	nº	11	12	7	10
Total Floodplain Area	Ha	1200	80	17	62
Habitat					
Agricultural area	Ha	960	30	15	52
Forest area	Ha	240	50	2	8
Hydrological conditions					
Hydraulic conductivity (aquifer)	m/day	86	1 - 3	50 - 100	8-9
River discharge (average)	m^3/s	195	250	25	11
Specific flow (average)	$m^3/s \cdot km^2$	$1.4 \cdot 10^{-2}$	$5.3 \cdot 10^{-3}$	$3.7 \cdot 10^{-2}$	$4.3 \cdot 10^{-4}$
Diversity indices					
Abundance *	nºindiv/100L	776 ± 645	354 ± 560	11 ± 10	30 ± 44
Taxonomic richness *	nº taxa	7 ± 1	6 ± 1	3 ± 2	2 ± 1
Taxonomic Shannon diversity index *		0.83 ± 0.39	1.13 ± 0.25	0.70 ± 0.52	0.38 ± 0.36
Functional richness *	nº categories	5 ± 2	6 ± 1	5 ± 2	2 ± 1
Functional Shannon diversity index *		0.77 ± 0.16	1.24 ± 0.10	1.07 ± 0.40	0.52 ± 0.41
Ecosystem services					
Biogeochemical filtration capacity *		1539 ± 1272	639 ± 1014	24 ± 27	60 ± 88
POM breakdown capacity *		873 ± 717	635 ± 1077	18 ± 17	34 ± 53

Table 2. Groundwater physicochemical features (mean ± SD) of four floodplains located in the Garonne, Ebro, Bidasoa and Tajo Rivers under different land use types. N = sample size. Significant differences between land use types in each river floodplain are indicated by * ($P<0.05$, Mann-Whitney *U* test).

		Garonne		Ebro		Bidasoa		Tajo	
Physicochemical features (Abbr.)	Units	Agricultural use (N = 8)	Forest use (N = 36)	Agricultural use (N = 12)	Forest use (N = 36)	Agricultural use (N = 24)	Forest use (N = 4)	Agricultural use (N = 24)	Forest use (N = 16)
Depth (Depth)	m	2.26 ± 0.77 *	3.28 ± 1.28 *	3.90 ± 0.71*	2.49 ± 0.45*	4.17 ± 0.76	3.99 ± 0.73	3.53 ± 1.20*	2.13 ± 0.73*
Temperature (Temp)	ºC	14.2 ± 1.7	13.79 ± 1.23	16.3 ± 1.4*	14.0 ± 1.8*	15.1 ± 2.0	14.1 ± 1.3	17.5 ± 1.3	17.6 ± 2.4
pH (pH)		7.02 ± 0.14	6.97 ± 0.12	6.92 ± 0.49*	7.36 ± 0.28*	6.49 ± 0.28	6.49 ± 0.13	7.14 ± 0.12	7.07 ± 0.15
Dissolved oxygen (O2)	%	65.60 ± 10.53 *	29.95 ± 29.20 *	24.06 ± 19.74	16.72 ± 15.94	37.66 ± 17.80	38.00 ± 8.94	42.34 ± 32.67*	26.10 ± 29.96*
Conductivity (EC)	µS/cm	945 ± 115	901 ± 225	3464 ± 646*	1618 ± 959*	367 ± 101	397 ± 31	2234 ± 410*	2493 ± 439*
Oxidation-reduction potential (ORP)	mV	252 ± 35	195 ± 91	104 ± 21*	68 ± 34*	98 ± 66	163 ± 50	118 ± 36	116 ± 39
Dissolved organic carbon (DOC)	mg/L	0.69 ± 0.45	1.22 ± 0.98	13.95 ± 3.47*	8.58 ± 5.58*	3.31 ± 1.22	3.08 ± 0.85	1.61 ± 0.60	1.99 ± 0.87
Alkalinity (Alk)	meq/L	5.34 ± 0.35 *	6.02 ± 1.13 *	6.39 ± 0.44*	5.32 ± 0.93*	3.15 ± 1.01	3.23 ± 0.32	5.78 ± 1.01	5.98 ± 1.31
Phosphate (PO4)	µg/L	9.28 ± 8.53	9.70 ± 9.59	2.68 ± 1.89*	16.47 ± 20.48*	5.13 ± 3.33	3.65 ± 1.52	182.08 ± 150.79	160.00 ± 165.81
Ammonium (NH4)	µg/L	7.90 ± 4.21	106.48 ± 269.98	0.004 ± 0.014*	10.50 ± 38.73*	3212.68 ± 9516.56	176.04 ± 161.71	199.94 ± 362.97	239.68 ± 350.90
Nitrate (NO3)	mg/L	76.88 ± 15.96 *	47.26 ± 38.73 *	34.68 ± 4.85*	10.10 ± 8.25*	9.23 ± 8.84	8.71 ± 3.36	32.25 ± 12.58	38.66 ± 16.16
Sulphate (SO4)	mg/L	67.38 ± 10.79	64.81 ± 29.66	1410.96 ± 154.57*	451.18 ± 421.78*	21.96 ± 12.33	32.16 ± 4.22	924.13 ± 371.81	1009.03 ± 384.17
Silica oxide (SiO2)	mg/L	13.01 ± 1.00	11.43 ± 3.97	18.66 ± 4.06*	7.75 ± 4.96*	9.36 ± 2.92	12.98 ± 4.33	15.23 ± 3.16	14.81 ±3.50

Table 3. Taxa and feeding traits that contributed the most to dissimilarity (Av. Diss/SD>1) between land use types (agricultural use areas vs. forest use areas). Statistics obtained through a two-way SIMPER analysis with factors date and land use.

	GARONNE		**EBRO**		**BIDASOA**		**TAJO**	
Taxonomic level	(average dissimilarity 49.3%)		(average dissimilarity 52.7%)		(average dissimilarity 74.1%)		(average dissimilarity 71.1%)	
	Taxa	**Diss/SD**	**Taxa**	**Diss/SD**	**Taxa**	**Diss/SD**	**Taxa**	**Diss/SD**
	Copepoda	1.46	Amphipoda	1.39	Oligochaeta	1.67	Ostracoda	1.44
	Amphipoda	1.31	Copepoda	1.22	Amphipoda	1.02		
Functional level	(average dissimilarity 38.4%)		(average dissimilarity 37.1%)		(average dissimilarity 41.1%)		(average dissimilarity 53.7%)	
	FFG	**Diss/SD**	**FFG**	**Diss/SD**	**FFG**	**Diss/SD**	**FFG**	**Diss/SD**
	Deposit feeder	1.60	Deposit feeder	1.44	Deposit feeder	1.38	Deposit feeder	1.54
	Predator	1.52	Shredder	1.43	Shredder	1.24	Scraper	1.09
	Shredder	1.25	Scraper	1.36	Absorber	1.33	Predator	1.04
			Predator	1.34	Filter feeder	1.57		
			Absorber	1.34	Predator	1.47		
					Parasite	1.41		
					Scraper	1.37		

Table 4. Results from LME models linking physicochemical variables with taxonomic and functional diversity indices and ecosystem services. All selected explanatory variables were statistically significant at $P < 0.05$. Total sample size = 160. d^2 = variance of the random intercept; α = variance of the fixed intercept; ρ = Spearman correlation coefficient between observed and predicted values, used as a measure of goodness of fit.

	Explanatory variables	Coefficient	Intercept	Spearman correlation test
Taxonomic diversity indices				
Abundance	% Forest area	0.30	$d^2 = 1.10^2$	$\rho = 0.71$
	NO_3	0.28	$\alpha = 15.00$	$P < 0.01$
	Temp	- 4.55		
Taxonomic richness	% Forest area	0.09	$d^2 = 0.33^2$	$\rho = 0.57$
	SO_4	0.06	$\alpha = 3.21$	$P < 0.01$
	Temp	- 0.79		
Taxonomic Shannon diversity index	DOC	0.07	$d^2 = 0.10^2$	$\rho = 0.58$
	% Forest area	0.04	$\alpha = 1.50$	$P < 0.01$
	O_2	- 0.05		
	Temp	- 0.37		
Functional diversity indices				
Functional richness	% Forest area	0.08	$d^2 = 0.16^2$	$\rho = 0.51$
	PO_4	- 0.03	$\alpha = 2.99$	$P < 0.01$
	Temp	- 0.54		
Functional Shannon diversity index	DOC	0.07	$d^2 = 0.10^2$	$\rho = 0.61$
	EC	- 0.07	$\alpha = 0.91$	$P < 0.01$
	% Forest area	0.03		
Ecosystem services				
Biogeochemical filtration capacity	% Forest area	0.30	$d^2 = 1.13^2$	$\rho = 0.69$
	NO_3	0.31	$\alpha = 15.38$	$P < 0.01$
	Temp	- 4.50		
POM breakdown capacity	% Forest area	0.34	$d^2 = 1.11^2$	$\rho = 0.70$
	NO_3	0.28	$\alpha = 13.86$	$P < 0.01$
	Temp	- 4.11		

Supplementary Material

Click here to download Supplementary Material: Appendix_att4.doc