

# Impact of agricultural land use on aquatic insect assemblages in the Garonne river catchment (SW France)

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**Abstract** The impact of agricultural land use on the composition and structure of aquatic insect assemblages (i.e., taxa of Ephemeroptera, Plecoptera, Trichoptera, and Coleoptera (EPTC)) was investigated in tributary streams of the Garonne river basin, southern France. The self-organizing map (SOM) method was applied to compare both instream environmental conditions and EPTC assemblages between forest and agricultural streams. According to the SOM model, the study sites were classified into three main clusters corresponding to distinct EPTC assemblages. The SOM cluster associated with most of the agricultural sites had lower EPTC species richness and diversity. This cluster was also characterized by high levels of total dissolved solids, nitrate ( $\text{NO}_3$ ), and chemical oxygen demand. Overall, our study shows that agricultural streams when compared

with forest streams had lower biological integrity. In accordance with the European Water Framework Directive, our results indicate that the sites most impacted by agricultural land use should be restored and that the least-impacted forest sites could serve as reference conditions.

**Keywords** Environmental change · Macroinvertebrates · Diversity · Riparian forest · SOM

## Introduction

Five interacting categories of human-induced perturbations have been reported to threaten freshwater biodiversity: degradation and destruction of habitats, water pollution, overexploitation, flow modifications, and species invasions (see review by Dudgeon et al. 2006). In particular, the global transition from undisturbed areas to human-dominated landscapes has strongly impacted on the physical features of lotic ecosystems by (i) increasing inputs of sediments, nutrients, organic matter, and contaminants (i.e., pesticides), (ii) clearing riparian vegetation and opening canopy, and (iii) altering flows and reducing habitat heterogeneity (Allan and Flecker 1993; Harding et al. 1998; Townsend et al. 2003; Allan 2004). This has resulted in drastic changes in the

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composition of stream communities and ecosystem functioning (Lenat and Crawford 1994; DeLong and Brusven 1998; Allan 2004).

The Garonne river catchment, located in south-western France, has a long history of agricultural development (Fortuné 1988). Forest areas were gradually replaced by agriculture land in the Garonne river valley along with its main tributaries in the eighteenth and nineteenth centuries (Chauvet and Décamps 1989). Due to severe agricultural practices, the areas of the riparian forest surrounding streams were reduced to a few meters or even disappeared. Deforestation can influence the physicochemical conditions of riverine ecosystems by (i) decreasing evapotranspiration, (ii) increasing rates of run-off, sediment yield, and nutrients, and (iii) affecting hydrology and discharge regimes (e.g., Chapman and Chapman 2002; Kasangaki et al. 2008). Changes in the composition of stream communities responding to anthropogenic disturbances have frequently been indicated by the groups of aquatic insects since they are sensitive indicators of long-term environmental changes in water and habitat quality (Johnson et al. 1993). In particular, Ephemeroptera, Plecoptera, and Trichoptera species are well documented as being good biological indicators in stream ecosystems (Rosenberg and Resh 1993). Despite their limited use in stream biomonitoring, the aquatic Coleoptera species, especially those belonging to Elmidae, have also been recognized as good water quality indicators (Hilsenhoff 1977; García-Criado and Fernández-Aláez 1995; Sánchez-Fernández et al. 2006). Especially, species of Coleoptera relying on a bubble or plastron for breathing have been shown to be sensitive to increase in sediment and organic pollution (e.g., Hauer and Resh 1996). Overall, considering the four insect orders might enhance the accuracy of water quality assessments (Compin and Céréghino 2003). For instance, a decrease in both the number of Coleoptera and Ephemeroptera, Plecoptera, and Trichoptera (EPT) species richness in human-impacted streams has been clearly related to changes in water quality and habitat suitability (e.g., Barbour et al. 1996; Wallace et al. 1996; Compin and Céréghino 2003).

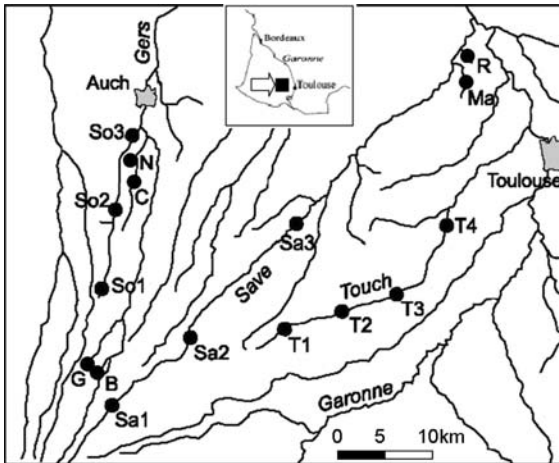
Given this context, we expected that the structure and composition of Ephemeroptera, Plecoptera, Trichoptera, and Coleoptera (EPTC) assemblages will have been substantially affected by agricultural

practices being carried out around the Garonne river basin. The aims of this study were therefore (i) to assess how agricultural land use influences EPTC assemblages in tributary streams in the Garonne river catchment and (ii) to discuss management implications and conservation issues. To do so, we compared the structure and composition of EPTC assemblages between agricultural streams (i.e., streambanks composed of croplands) and streams draining forest areas (i.e., streambanks composed of woody and grassy vegetation). In this study, we applied self-organizing mapping (SOM; Kohonen 1989, 2001) to analyze the patterns of EPTC assemblages and instream environmental conditions, because non-linearity is generally embedded in ecological data resulting from complex interactions between environmental variables and communities (Legendre and Legendre 1998; Lek and Guégan 2000; Adriaenssens et al. 2007). SOM is an unsupervised neural network that has been implemented in various ecological studies (e.g., Chon et al. 1996; Lek and Guégan 2000; Park et al. 2001; Adriaenssens et al. 2007) and provides an alternative to traditional ordination and classification methods (Lek and Guégan 2000; Adriaenssens et al. 2007).

## Materials and methods

### Study sites

The survey area is located in the basin of the Garonne river in southern France (between the Pyrenees and the Garonne valley near Toulouse; Fig. 1), which is the third largest of the large French Rivers. We investigated three tributaries of the Garonne river that mainly drain through agriculture lands: the Save stream (length ( $L$ ) 143 km; catchment area (CA) 1,150 km<sup>2</sup>), the Touch stream ( $L$  60 km; CA 515 km<sup>2</sup>), and the Sousson stream ( $L$  53 km; CA 120 km<sup>2</sup>). Four sites were selected in the forest environment (G, T1, SA1, and R), while the eight other sampling sites (see Fig. 1) were surrounded by cropland. All the sites are located between 137 and 361 m in altitude above the sea level and measure 25 m in length. The proportion (%) of different land covers in the basin surface area of each sampling site was extracted from the CORINE Land Cover (CLC) map (<http://www.ifen.fr>) through ArcGIS 9.0. CLC is a map of the European environmental landscape



**Fig. 1** Location of the sampling sites in tributary streams of the Garonne river basin, France. *Black circles* represent the sampling sites

intended for use by policy makers as well as scientists. Based on interpretation of satellite images, CLC provides comparable digital maps of land cover for each country for much of Europe. It is based on a simple three-level hierarchy classification system consisting of 44 land cover classes. We used three classes in the first level of CLC categories: urbanized artificial surface, agricultural land, and forest area (Table 1).

**Macroinvertebrate sampling**

Benthic macroinvertebrates were sampled three times at each site from February to July 2004 and in two habitat zones. One sample of sediment was collected in the riffle zone using a Surber sampler (30 cm × 30 cm, 200 μm) and the other (10 cm depth) was collected in the pool zone using a plastic bottle (volume 400 ml). Samples were preserved in 70% ethanol and pooled for subsequent analyses. EPTC were identified to the species (or genus) level. EPTC taxa richness and the Shannon–Weaver diversity index were determined for each site.

**Environmental variables**

For each site, eight environmental variables were measured along with the collection of benthic macroinvertebrates. Water samples were additionally collected in the riffle zone for the measurement of chemical oxygen demand (COD), nitrate (NO<sub>3</sub>), and total dissolved solids (TDS) according to the standard method (APHA, AWWA, WPCF 1985). Dissolved Oxygen (DO) was additionally obtained using a portable meter on site. Current velocity, mean width, and mean depth were quantified using the transect method. This method consists of delimiting each site by equidistant cross-stream transects, along which

**Table 1** The proportion (%) of three classes of land cover in the basin surface area of each sampling site

Stream	Site name	Forest area (%)	Agricultural land (%)	Urbanized artificial surface (%)
Touch	T1	42.9	57.1	0
Touch	T2	47.1	49.7	0
Touch	T3	9.9	89.8	0
Touch	T4	12.3	86.3	0.9
Save	SA1	31.6	66.5	1.9
Save	SA2	23.2	75.9	0.9
Save	SA3	11.6	87.6	0.7
Rieutord	R	55.7	44.3	0
Marguestaud	Ma	15.1	84.9	0
Sousson	SO1	34.2	65.8	0
Sousson	SO2	22.9	77.1	0
Sousson	SO3	18.5	81.2	0.3
Naroques	N	0	100	0
Cedon	C	10	90	0
Geze	G	56.2	43.8	0
Badet	B	13.1	86.9	0

See [Materials and methods](#) for more details

each variable was measured at regularly spaced intervals. Current velocity was measured by observing the horizontal displacement of a float over a calibrated distance. The sites were also classified semi-quantitatively by the degree of riparian forest development: 1 (no riparian forest), 2 (0–10 m width of riparian forest), and 3 (forest area; >10 m width of riparian forest).

### Data analysis

Firstly, a taxa abundance data set was arranged as a matrix of 36 rows (i.e., the 12 sites sampled on three occasions) and 76 columns (i.e., the EPTC taxa). Each of the 36 samples of the data set can be considered as a vector of 76 dimensions.

Then SOM was applied to classify the sample sites according to their composition of EPTC taxa abundance. The SOM preserves the neighborhood so that samples with close taxa composition are grouped together on the map, whereas samples with very different taxa composition are far from each other. The SOM consists of two layers that are composed of neurons in the form of computational units: input and output layers connected with weight vectors (i.e., connection intensity). When an input vector (abundance of each taxa)  $x$  is sent to the input layer of the network, each neuron  $k$  of the network computes the distance between the weight vector  $w$  and the input vector  $x$ . The output layer consists of  $D$  output neurons (i.e., computational units,  $24 = 6 \times 4$  in this study), which are arranged into a two-dimensional grid. The best arrangement for the output layer is a hexagonal lattice, because it does not favor horizontal and vertical directions as much as the rectangular array (Kohonen 2001). Among all  $D$  output neurons, the best matching unit (BMU) that has the minimum distance between weight and input vectors is the winner. For the BMU and its neighboring neurons, the weight vectors  $w$  are updated by the SOM learning rule. Because the map size (number of output units) of the SOM is critical for accommodating hierarchical levels in community classification (Park et al. 2004), we trained the SOM with different map sizes and chose the optimum map size based on low topographic and quantization errors (Kohonen 2001). To analyze the contribution of EPTC taxa to the cluster structures of the SOM, the weight vector of each input variable (i.e., taxa abundance) calculated during the

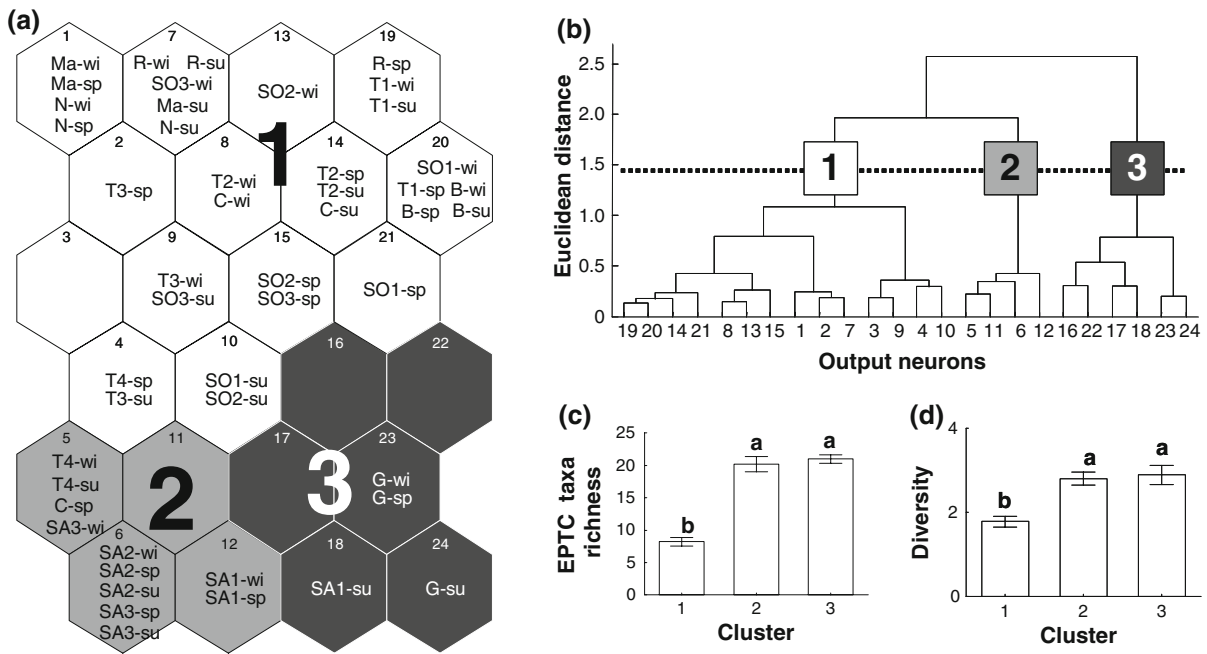
training process was visualized in each output neuron on the trained SOM in a gray scale.

Finally, we looked at whether relevant groups of samples characterized distinct EPTC assemblages by doing a hierarchical cluster analysis (Ward's linkage and Euclidean distance) on the similarity between SOM cells. To do this, we used a new matrix ( $24 \times 76$ , output neurons  $\times$  taxa) of the connection intensity values (i.e., vector weights) estimated by the SOM (Park et al. 2003). Between-cluster differences in instream environmental conditions and community parameters (i.e., EPTC taxa richness and Shannon–Weaver diversity index) were evaluated using the Kruskal–Wallis test, a non-parametric analysis of variance, followed by a Mann–Whitney test to identify specific differences.

As a preprocessing procedure before the application of the SOM, an abundance of 76 EPTC taxa was transformed with natural logarithm (i.e.,  $\log(x+1)$ ). The transformed values were also rescaled in the range of the minimum and maximum values (i.e., 0–1) before training the model. We used the functions implemented in the SOM toolbox (Alhoniemi et al. 2000) for MATLAB (The Mathworks Inc 2001), developed by the Laboratory of Information and Computer Science at the Helsinki University of Technology. The detailed algorithm of the SOM can be found in Kohonen (2001) for theoretical considerations and in Chon et al. (1996) and Park et al. (2003) for ecological applications.

### Results

According to the composition and abundance of EPTC taxa, the samples were first classified by the SOM in the 24 output units, so that each unit included samples with similar EPTC assemblages (Fig. 2a). The samples were mainly grouped according to the geographic location of the sample sites (Fig. 2a). Then, a hierarchical cluster analysis classified the samples into three main clusters by using the output matrix of the SOM (Fig. 2b). The samples from the Géze stream (G) were located in the lower-right part of the SOM map (cluster 3), while the samples (SA1–SA3) coming from the Save stream were mainly grouped in the lower-left part of the SOM map (cluster 2). The samples from the other streams (e.g., Touch (T1–T4), Sousson (SO1–SO3), Marguestaud



**Fig. 2** Classification of the sample sites on the self-organizing map (SOM). **a** The SOM map showing the classification of the sample sites according to their composition in EPTC taxa. Abbreviations in each output neuron of the SOM map correspond to the site name (see Table 1) that were sampled on different season (wi: winter; sp: spring; su: summer). The numbers (1, 2 and 3) associated to a gray scale correspond to the three clusters defined by the hierarchical cluster analysis

(Ma), and Rieutord (R)) were mainly grouped in the upper part of the SOM map (cluster 1). Most of the samples from the Touch and Sousson streams belonged to this former cluster.

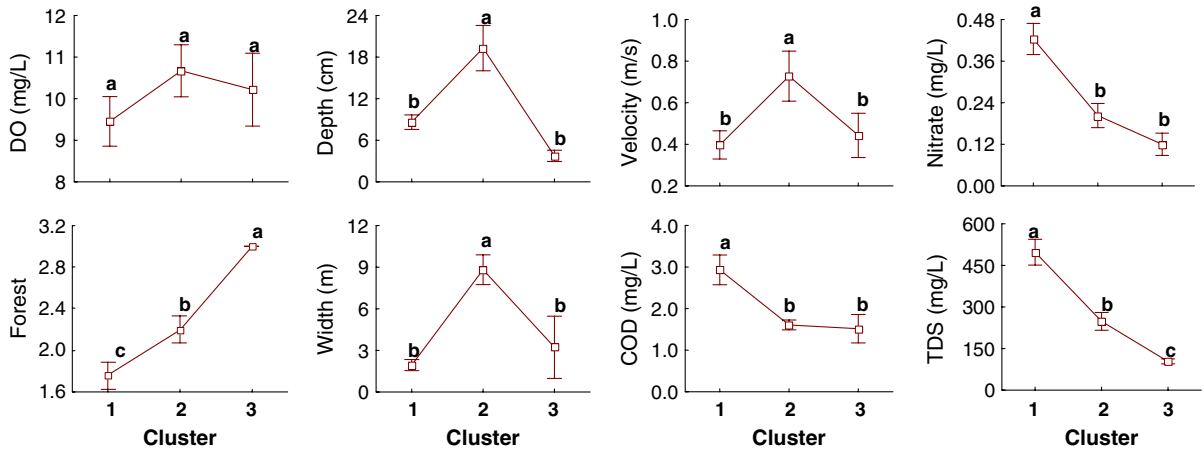
The number of EPTC taxa and the Shannon–Weaver diversity index differed significantly between the three clusters (Kruskall–Wallis test:  $P < 0.001$ ). A Mann–Whitney test revealed that the number of EPTC taxa and the Shannon–Weaver diversity index were significantly lower in cluster 1 compared with clusters 2 and 3 ( $P < 0.001$ ; Fig. 2c, d). By contrast, no significant difference in the number of EPTC taxa and the Shannon–Weaver diversity index was found between the clusters 2 and 3.

Except for the DO, all the environmental variables differed significantly between the three clusters (Kruskall–Wallis test:  $P < 0.001$ ; Fig. 3). The degree of riparian forest development was significantly higher in cluster 3 compared with clusters 2 and 3, whereas higher concentrations of  $\text{NO}_3$ , COD, and TDS were found in cluster 1 (Mann–Whitney test:

(see below). **b** Hierarchical cluster analysis applied to the output neurons of the SOM map. The numbers (1, 2 and 3) associated to a gray scale correspond to the three clusters. **c** Mean and SE of EPTC taxa richness in the three clusters. The same letters indicate no significant difference based on Mann–Whitney tests. **d** Mean and SE of the Shannon–Weaver diversity index in the three clusters defined by the SOM. The same letters indicate no significant difference

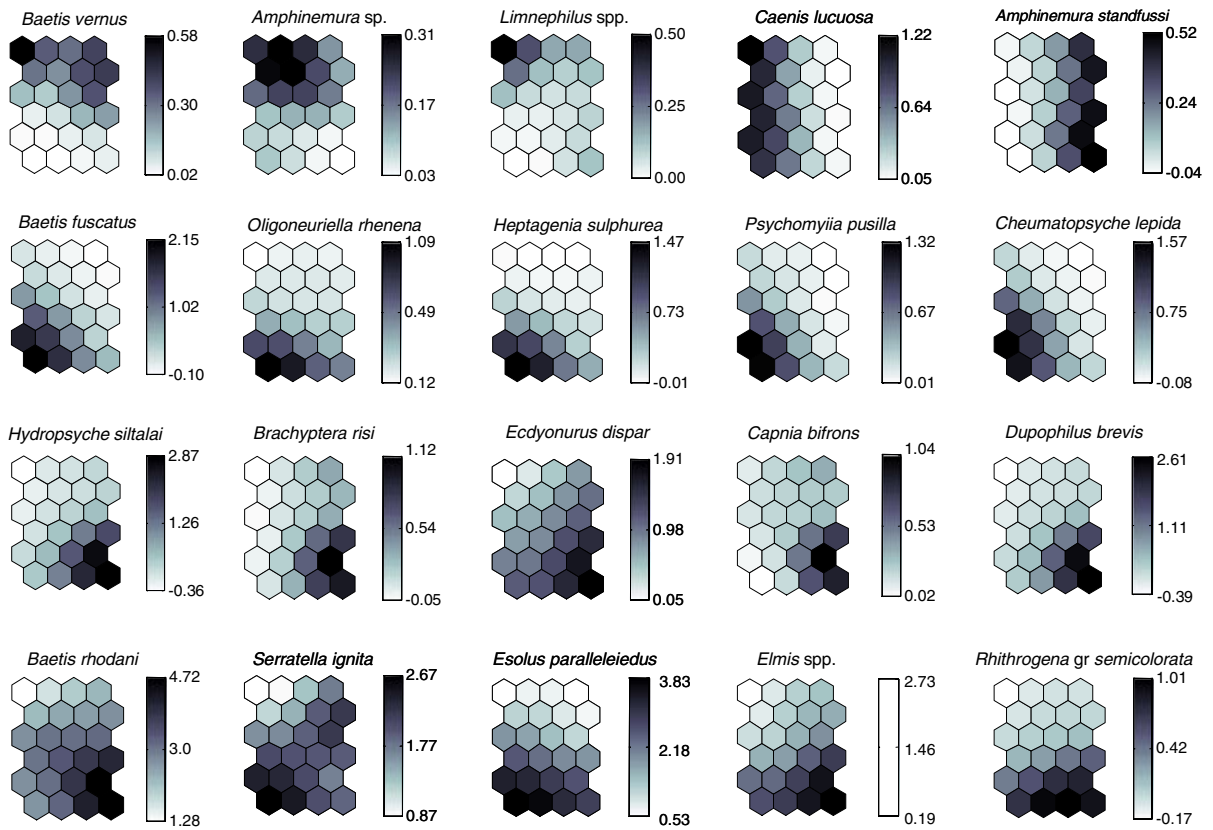
$P < 0.0001$ ). Finally, the environmental variables related to the size of the stream (i.e., depth, velocity, and width) displayed the highest values in cluster 2 (Mann–Whitney test:  $P < 0.001$ ). No significant differences in depth, velocity, and width were found between the clusters 1 and 3 (Mann–Whitney test:  $P > 0.05$ ).

According to the training process of the SOM, the abundance profiles of EPTC taxa were estimated and projected on the SOM map in a gray scale (Fig. 4). This allowed us to characterize three groups of EPTC taxa according to the abundance of each EPTC taxa in the three clusters defined on the SOM map (see Appendix). For example, *Baetis vernus*, *Amphinemura* sp., and *Limnephilus* spp. were most abundant in samples characterizing cluster 1. By contrast, *Baetis rhodani*, *Hydropsyche siltalai*, *Brachyptera risi*, *Dupophilus brevis*, and *Capnia bifrons* were most abundant in cluster 3. Cluster 2 was characterized by a high abundance of *Baetis fuscatus*, *Oligoneuriella rhenana* and *Heptagenia sulphurea* (Fig. 4).



**Fig. 3** Environmental variables associated with the three clusters defined by the SOM analysis. Forest indicates the degree of riparian forest development. Error bars indicate

standard error. The same letters indicate no significant difference based on Mann–Whitney tests



**Fig. 4** Abundance patterns of some species on the SOM map. Scale bars indicate the weight vector of each taxa (i.e., species abundance) in corresponding SOM units. Dark represents high species abundance, and light indicates low species abundance

## Discussion

In this study, the identification of EPTC was mostly carried out at the species level to assess small-scale changes in community composition between sites and dates in response to anthropogenic disturbances (Lenat and Resh 2001). In addition, community parameters such as diversity index and taxa richness can be considerably under-evaluated at the family and genus level, especially in the nonimpacted streams (Guerold 2000). Species data are also useful in providing specific information for the conservation of rare (e.g., endemic) species.

The impact of agricultural land use on water quality and aquatic assemblages has been widely documented (e.g., Genito et al. 2002; Lenat and Crawford 1994; Riley et al. 2003; Allan 2004; Probst et al. 2005). Overall, our results are consistent with those obtained in previous studies. Most of the agricultural sites (i.e., cluster 1) were characterized by high levels of TDS, COD, and  $\text{NO}_3$  that most likely result from high loads of organic and inorganic suspended materials in the increased runoff from agricultural lands. This may explain why most of the agricultural sites (i.e., the sites grouped in cluster 1) showed lower EPTC taxa richness and diversity than forest sites (i.e., cluster 3). Indeed, high sedimentation rates often lead to the decline of benthic aquatic insects (i.e., Ephemeroptera, Plecoptera, and Trichoptera) that are sensitive to mud on their integument and gills and to the filling of interstitial habitats due to accumulations of silt (e.g., Chutter 1968; Allan 2004).

Compared with the agriculture-impacted sites belonging to cluster 1, the forest sites (e.g., Géze and SA1) were inhabited by several sensitive taxa such as *H. siltalai* and *D. brevis* that prefer high current velocity and low temperature (Thomas and Berthélemy 1991; Stuijzand et al. 1999; Stubauer and Moog 2000; Urbanič et al. 2005). We also noticed that numerous sensitive species that were absent in the agriculture-impacted sites were historically common to the Pyrenees and the Garonne river. For example, *B. risi* and *C. bifrons* (Plecoptera) were only sampled in forest sites characterizing cluster 3, whereas these species were historically widely distributed in the study area (Berthélemy 1966). Loss of sensitive species in deforested streams due to the alteration of habitats and water quality (Quinn 2000; Allan 2004) has also been reported in many studies

(e.g., Collier 1995; Lenat and Crawford 1994; Wang et al. 1997).

Although we showed that some forest sites were characterized by distinct EPTC assemblages (i.e., the sites belonging to cluster 3), we noticed that the forest sites T1 and R were grouped with the agriculture-impacted sites in cluster 1. Indeed, the site R was characterized by low flow conditions and was dominated by small pool habitats during the sampling period compared with the other sites localized in forest areas. These harsh environmental conditions most likely favored EPTC assemblages that are similar to those occurring in the agriculture-impacted sites since stream drying required high tolerance on the part of aquatic insects (Williams 1996; Meyer et al. 2003). In addition, the spring upstream of T1 was exposed to agricultural land use along 1 km, which explains the high level of organic matter and nutrients observed at this site. This underlines that the features of the whole catchment area are also important when predicting the impacts of anthropogenic activities on stream macroinvertebrate communities (Allan et al. 1997; Vondracek et al. 2005).

Compared with clusters 1 and 3, the sites belonging to cluster 2 were characterized by high depth, width, and current velocity. These sites were also inhabited by diverse EPTC taxa. Among them, *B. fuscatus*, *O. rhenana*, and *H. sulphurea* are sensitive species and prefer fast currents (Beketov 2004; Moog et al. 1997). Surprisingly, this cluster was composed of sites surrounded by cropland (i.e., T4, C, SA2, and SA3). We suspect that the impact of agricultural land use on these sites was masked by the variability of the natural hydrology and the high flows that make instream habitats more heterogeneous and hence more diverse in macroinvertebrates (Allan 2004). This highlights that the covariation between natural landscape features and anthropogenic factors can make assessing the biotic integrity of stream ecosystems difficult (e.g., Richards et al. 1997; Fitzpatrick et al. 2001).

The longitudinal nature of rivers and streams has been long recognized as a major force in structuring lotic communities (Hynes 1970). Absence of a longitudinal gradient has been often reported in streams dominated by agricultural land use (e.g., Delong and Brusven 1998; Allan 2004). Indeed, streams impacted by agricultural land use are generally characterized by low diverse assemblages of macroinvertebrates that are dominated by ubiquitous

species capable of tolerating long-term sources of pollution (Allan 2004). Our results are consistent with these previous studies as both upstream and downstream sites of agricultural streams were classified in the same cluster defined on the SOM map (e.g., the sites of the Touch and Sousson streams).

Overall, our study revealed that forest streams had higher biological integrity than agricultural streams. In a recent synthesis, Vondracek et al. (2005) showed that the composition and diversity of macroinvertebrates can be enhanced by increasing the amount of riparian vegetation along streams and in upland areas. We therefore recommend that the streams impacted by agricultural land use in the study area should be restored by planting grass or/and woody vegetation in continuous blocks 10–20 m wide. This width of riparian vegetation is commonly accepted as a standard method to reduce nutrients and organic matter carried by surface runoff (e.g., Vought et al. 1994). In addition, a buffer width of 10–20 m is the minimum necessary for the development of sustainable indigenous vegetation with minimal weed control and to achieve many aquatic functions (Parkin et al. 2000). This restoration project could be initiated only if the least impacted streams (streams draining forest areas) are preserved. Indeed, according to the European Union Water Framework Directive (WFD), reference situations are necessary to assess both the biotic integrity of human-impacted streams and their recovery after restoration. The WFD that provides generic descriptions of five ecological status classes for

European surface waters has two main ecological objectives: preventing deterioration in ecological status and restoring to good ecological status by 2015 (Logan and Furse 2002; Wharton and Gilvear 2006). The WFD requires that a variety of taxonomic groups, including macroinvertebrates, be studied to assess the ecological status of rivers (including the reference conditions). To our knowledge, very few studies have assessed the impacts of anthropogenic activities on macroinvertebrate communities in the Garonne river basin (see Compin and Céréghino 2007). Such a lack of baseline data therefore limits the promotion of river restoration projects. Overall, we encourage future studies to concentrate their efforts on the assessment of the ecological status of rivers at the whole Garonne river basin scale.

To conclude, in this paper, we have only dealt with the impacts of agricultural land use at the reach scale. Instructive and complementary results could be obtained by comparing several catchments with different land use practices. This would allow the testing of whether landscape indicators at multiple spatial scales (i.e., reach, catchment) are more efficient for assessing the impact of human activities on riverine ecosystems (see review by Gergel et al. 2002).

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## Appendix

List of dominant EPTC taxa in each cluster defined by the SOM analysis

Species	Cluster	Species	Cluster
Ephemeroptera		Trichoptera	
<i>Baetis vernus</i>	1	<i>Brachycentrus subnubilus</i>	2
<i>B. cf lutheri</i>	2	<i>Agapetus</i> sp.	2
<i>B. fuscatus</i>	2	<i>Goera pilosa</i>	2
<i>B. rhodani</i>	3	<i>Goera</i> sp.	2
<i>Centroptilum luteolum</i>	1	<i>Cheumatopsyche lepida</i>	2
<i>Cloeon cognatum</i>	2	<i>Hydropsyche contubernalis</i>	1
<i>Procloeon</i> sp.	1	<i>Hydropsyche cf exocellata</i>	1
<i>P. bifidum</i>	1	<i>Hydropsyche siltalai</i>	3
<i>Caenis luctuosa</i>	1	<i>Hydroptila</i> sp.	1
<i>C. pusilla</i>	2	<i>Lepidostomatidae</i> sp.	2
<i>Serratella ignita</i>	2	<i>Athripodes bilineatus</i>	1
<i>Ephemera lineata</i>	2	<i>Athripodes</i> sp.	1

## Appendix continued

Species	Cluster	Species	Cluster
<i>E. danica</i>	3	<i>Mystacides azurea</i>	2
<i>Ecdyonurus insignis</i>	2	<i>Limnephilus</i> spp.	1
<i>E. dispar</i>	3	<i>Odontocerum</i> sp.	3
<i>E. venosus</i>	3	<i>Chimarra marginata</i>	2
<i>Electrogena</i> sp.	3	<i>Philopotamus montanus</i>	1
<i>Epeorus torrentium</i>	3	<i>Polycentropodus kingi</i>	2
<i>Heptagenia sulphurea</i>	2	<i>Psychomyia pusilla</i>	2
<i>Rhithrogena gr semicolorata</i>	3	<i>Rhyacophila</i> sp1	1
<i>Habroleptoides confusa</i>	3	<i>Rhyacophila dorsalis</i>	2
<i>Habrophlebia</i> sp.	3	<i>Rhyacophila fasciata</i>	2
<i>Paraleptophlebia submarginata</i>	2	<i>Rhyacophila</i> sp2	3
<i>Oligoneuriella rhenana</i>	2	<i>Sericostoma personatum</i>	1
<i>Ephoron virgo</i>	2	Coleoptera	
<i>Potamanthus luteus</i>	2	<i>Dryops</i> sp.	1
<i>Siphonurus aestivalis</i>	1	<i>Dytiscidae</i> sp.	1
Plecoptera		<i>Dupophilus brevis</i>	3
<i>Capnia bifrons</i>	3	<i>Elmis</i> spp.	3
<i>Euleuctra geniculata</i>	2	<i>Esolus parallelepipedus</i>	2
<i>Leuctra fusca</i>	2	<i>Limnius</i> sp.	2
<i>Amphinemura</i> sp.	1	<i>Limnius opacus</i>	2
<i>Amphinemura standfussi</i>	3	<i>Oulimnius troglodytes</i>	2
<i>Nemoura</i> sp.	1	<i>Riolus cupreus</i>	1
<i>Nemoura cinerea</i>	1	<i>Stenelmis canaliculata</i>	2
<i>Protonemura intricata</i>	3	<i>Orectochilus villosus</i>	2
<i>Isoperla gr grammatica</i>	1	<i>Helophoridae</i> sp.	3
<i>Brachyptera risi</i>	3	<i>Hydraenidae</i> sp.	3
		<i>Hydrophilidae</i> sp.	1
		<i>Helodes</i> sp.	3

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