Limnetica, 40(2): 343-357 (2021). DOI: 10.23818/limn.40.23

Macroinvertebrate communities and macrophyte decomposition could be affected by land use intensification in subtropical lowland streams

Burwood, M.[1,*], Clemente, J.[1], Meerhoff, M.[1,2], Iglesias, C.[1], Goyenola, G.[1], Fosalba, C.[1], Pacheco, J.P.[1,2] and Teixeira de Mello, F.[1,*]

[1] Departamento de Ecología y Gestión Ambiental. Centro Universitario Regional Este. Universidad de la República. Tacuarembó s/n, entre Av. Artigas y Aparicio Saravia, Maldonado CP 20000, Uruguay.
[2] Department of Bioscience, Aarhus University. Nordre Ringgade 1 CP 8000 Aarhus, Denmark.

[*] Corresponding author: maiteburwood@gmail.com; frantei@fcien.edu.uy

Received: 27/06/19 Accepted: 30/07/20

ABSTRACT

Macroinvertebrate communities and macrophyte decomposition could be affected by land use intensification in subtropical lowland streams

In lowland streams with little or no riparian forest, autochthonous organic matter can be an important source of the energy supporting the aquatic trophic webs, but different kind of land use in their catchment areas can directly affect the aquatic community structure and indirectly affect the decomposition rate of coarse particulate organic matter (CPOM), among other processes. We investigated whether intensification in the land use affects the autochthonous (aquatic macrophyte) CPOM decomposition rate (k), by affecting the macroinvertebrate community (e.g. density, biomass and richness of taxa and functional groups). Stems of the emergent macrophyte *Schoenoplectus californicus* (C. A. Mey) were introduced, for 338 days, into two streams with different land use intensity (extensive pasture *vs* intensive agriculture-dairy). In our study, the stream draining the more intensified land use pasture (hereafter intensive stream) presented higher water nutrient concentrations and decreased dissolved oxygen percentage, pH and discharge. The stream in extensive pasture (hereafter extensive stream) presented a significantly higher decomposition rate ($k = -0.010 \pm 0.00068$ days^{-1}) than the intensive stream ($k = -0.006 \pm 0.0005$ days^{-1}). Ephemeroptera, Amphipoda, Chironomidae and Oligochaeta, accounted for 94 % of the total abundance of macroinvertebrates communities. Moreover, 89 % of total analysed individuals belonged to the collector-gatherers functional group, whereas scrapers dominated the total biomass (85 %). Density and biomass of macroinvertebrates increased in the last dates in both systems, being higher in the extensive stream. An increase in scraper density, biomass and richness in the last sampling dates co-occurred with a rapid loss of *S. californicus* mass in the extensive stream. Our results show that the stream with higher land use intensification presented more simple macroinvertebrates communities, what could negatively affect a key ecosystem function for these subtropical lowland streams, the decomposition rate of *Schoenoplectus californicus*.

Key words: *Schoenoplectus californicus*, decomposition rate, functional groups

RESUMEN

La comunidad de macroinvertebrados y la descomposición de macrófitas pueden ser afectados por la intensificación en el uso del suelo en arroyos subtropicales

*En arroyos de bajo orden con ausencia o con escaso bosque ripario, la materia orgánica autóctona puede ser una importante fuente de energía para soportar las redes tróficas acuáticas. Distintas actividades de uso del suelo presentes en las cuencas hidrográficas pueden modificar directamente la comunidad acuática e indirectamente la función de descomposición de materia orgánica particulada gruesa (MOPG). Nosotros investigamos si la intensificación en el uso del suelo afecta la tasa de descomposición (*k*) de la MOPG autóctona (macrófita acuática), al afectar atributos de la comunidad de macroinvertebrados asociados al proceso de descomposición (densidad, biomasa y riqueza de los diferentes taxones y grupos funcionales). Fragmentos de la planta emergente Schoenoplectus californicus (C. A. Mey) fueron introducidos en bolsas de malla plástica, durante 338 días, en dos arroyos con diferente intensidad de uso del suelo (pastoreo extensivo* vs *agricultura* *intensiva). En este estudio, el arroyo con un uso intensivo del suelo (a partir de aquí, arroyo intensivo) presentó un aumento en la concentración de nutrientes en agua y una disminución en el porcentaje de oxígeno disuelto, pH y caudal. El arroyo con pastoreo extensivo (arroyo extensivo) presentó una tasa de descomposición significativamente mayor (*k *= - 0.010 ± 0.0008 days^{-1}) en comparación con el arroyo intensivo (*k *= - 0.006 ± 0.0005 days^{-1}). Ephemeroptera, Amphipoda, Chironomidae y Oligochaeta representaron el 94 % de la abundancia total de las comunidades de macroinvertebrados. Además, el 89 % del total de individuos analizados pertenecieron al grupo funcional de colectores-recolectores, mientras que los raspadores dominaron la biomasa total (85 %). La densidad y biomasa de taxones aumentaron en las últimas etapas de la descomposición en ambos arroyos, siendo mayor en el sistema extensivo. El incremento en la densidad, biomasa y riqueza de organismos raspadores en las últimas etapas fue concordante con la rápida pérdida de masa de* S. californicus *en el arroyo extensivo. Estos resultados muestran que en el arroyo con mayor intensificación en el uso del suelo se simplifica la comunidad de macroinvertebrados, lo cual podría afectar negativamente una función ecosistémica clave para estos arroyos subtropicales, la tasa de descomposición de* Schoenoplectus californicus.

Palabras clave: Schoenoplectus californicus*, tasa de descomposición, grupos funcionales*

INTRODUCTION

An intensification in land use, such as the destruction of natural forest and grasslands to free land for intensive agricultural and dairy production, usually causes water quality to deteriorate and the loss or degradation of many ecosystem services provided by freshwater ecosystems (e.g. food production, high quality freshwater, riparian forest resources; Foley *et al.*, 2005). The most common effect of such land use intensification is eutrophication, which is an increase in the external (and often internal) nutrient loading that changes the structure and function of some key biological communities, affecting the functioning of all the ecosystem (Lampert & Sommer, 2007).

An increase in nutrients, in particular, may indirectly affect coarse particulate organic matter (CPOM) decomposition (Manning *et al.*, 2016; Bastias *et al.*, 2018), a key ecosystem process since it constitutes a link between terrestrial and aquatic carbon fluxes and nutrient cycling (Tank *et al.*, 2010). Also, CPOM is often one of the main sources of organic matter that fuels trophics webs (Graça & Canhoto, 2006). Due to its ecosystem importance, the decomposition process, is often used as a monitoring tool (Gessner & Chauvet, 2002; Tiegs *et al.*, 2019) to detect the effects of land use changes on the functioning of aquatic systems (Ferreira *et al.*, 2016a; 2018).

The closed canopy of forested streams limits available light and allochthonous organic matter is consequently the main support of aquatic trophic webs (Vannote *et al.*, 1980). In contrast, in systems with poor or completely absent riparian forest coverage, autochthonous primary producers (algal and macrophytes) may be the main source of organic matter (Cummins *et al.*, 1983). In this context, it has been suggested that macrophytes can be an important source of matter and energy, depending on riparian diversity (Vannote *et al.*, 1980, Thorp & Delong, 1994), and the frequency and intensity of flood pulses (Junk *et al.*, 1989; Humphries *et al.*, 2014).

CPOM decomposition follows several different, but complementary, steps or phases, characterized by changes at community and functional level (Graça & Canhoto, 2006; Graça *et al.*, 2015). In the first phase, the microbial community colonize and condition the detritus, and a large amount of biomass and nutrients are lost from the CPOM lixiviation. This conditioned detritus becomes available to the macroinvertebrate community, leading to the second phase, when the macroinvertebrates community transforms CPOM into fine particulate organic matter through fragmentation. Shredders and scrapers play an important role in this phase, directly fragmenting or indirectly feeding on plant tissues, respectively (Bianchini, 1999; Ferreira *et al.*, 2016b; Graça *et al.*, 2015; Tonin *et al.*, 2018). Some studies have found an effect of water nutrient enrichment in the CPOM decomposition process, both affecting the bacterial and macroinvertebrate phases of the process, resulting in a relatively low decomposition rate at both extremes of a nutrient gradient (Woodward *et al.*, 2012; Ferreira *et al.*, 2015; Manning *et al.*, 2016). High nutrient concentrations affect some macroinvertebrates functions, particularly by causing a lower diversity and abundance of shredders (Woodward *et al.*, 2012; Ashton *et al.*, 2014). Conversely, under moderate water nutrient concentrations, higher decomposition rates have been reported, especially in nutrient poor substrates (Manning *et al.*, 2016). A synergistic effect of CPOM decomposition phases tend to occur, since microbial activity modifies the substrate stoichiometry (C:N, C:P; Suberkropp & Chauvet, 1995; Cheever *et al.*, 2012; Tant *et al.*, 2013; Mehring *et al.*, 2015), and this further stimulates the colonization and activity by shredders.

Land use intensity in the watersheds can differentially affect the aquatic macroinvertebrate community through a variety of process. For example, high ammonium and nitrate concentrations can be toxic for some macroinvertebrates (Berenzen *et al.*, 2001; Lecerf *et al.*, 2006). Also, the introduction of fine sediments (Graeber *et al.*, 2017), heavy metals and pesticides and the typical decrease in dissolved oxygen (Chang *et al.*, 2014) can negatively affect the structure of macroinvertebrate communities (Piggott *et al.*, 2012; Chang *et al.*, 2014; Ferreira *et al.*, 2015; Berger *et al.*, 2016), especially resulting in the loss of shredder and scraper taxa (Lecerf *et al.*, 2006) and sensitive species.

A large area of the Uruguayan countryside (34° 52' 1" S, 56° 10' 0" W) has experienced a strong and rapid transformation of land use, from traditional, extensive free rang livestock production over native grasslands or modified pastures (from here onwards, "extensive") to industrial and intensive agriculture-dairy production (from here onwards, "intensive").

In the present study we analysed two similar low-order streams with contrasting land use intensity (extensive vs intensive), using the physico-chemical variables of water as a proxy for land use intensification. We aimed to understand whether the land use intensity indirectly affects CPOM decomposition rates by modifying macroinvertebrate community attributes (richness, density and dry biomass of main taxa and functional groups). In particular, we hypothesized that the stream on a more intense land use would: (1) have a lower water quality, (2) present a significant reduction in the density, richness and biomass of shredders and scrapers and, as a consequence, (3) show a lower CPOM decomposition rate.

METHODS

Study site

Uruguay has a vast hydrographic network that runs through extensive modified grasslands belonging to The Río de la Plata grasslands or "campos" (Paruelo *et al.*, 2010), which is the dominant landscape. Two lowland subtropical streams

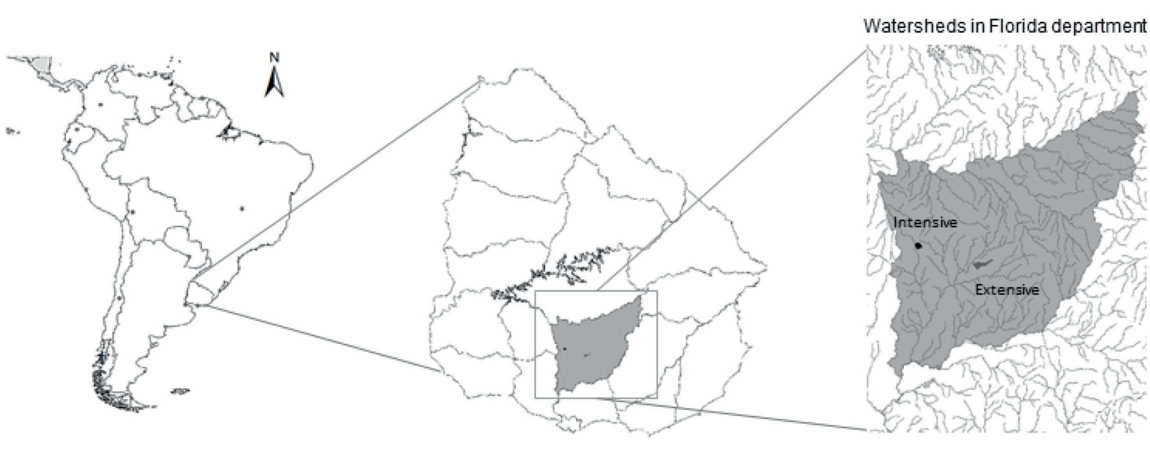

Figure 1. Location of the extensive and the intensive streams used as models in this study (Florida, Uruguay). *Localización del arroyo extensivo y arroyo intensivo utilizados como modelo en nuestro estudio (Florida, Uruguay).*

draining from microcatchments, order 3, were selected as representatives of extremes of current land use intensity in Uruguay (Fig. 1). Both systems belong to the Santa Lucía River basin (Florida, Uruguay). The "extensive" stream (-33° 54' 12.32" S, -56° 00' 22.73" W) is located in a 1880 ha basin, this stream has extensive livestock production based on natural grasslands (70 % of the total area) and a smaller proportion of agriculture (30 %). The "intensive" stream (33° 49' 31.75" S, 56° 16' 55.38" W), in an 840 ha basin, is subject to intensive agriculture-dairy land use (90 %), with a small area devoted to grasslands (7 %) and urbanization (3 %). Previous comparative studies found significant differences in water nutrient concentrations between the two streams (Goyenola *et al.*, 2015; Goyenola *et al.*, 2020).

Water variables

Water temperature, pH, conductivity, dissolved oxygen, total dissolved solids and water flow were measured in situ every 10 min during the 338 days of the experiment, with a YSI V6600 multiparameter probe. Approximately one water sample per month was taken to measure the concentration of total nitrogen (TN), dissolved total nitrogen (DTN), total phosphorus (TP), and soluble reactive phosphorus (SRP), following standard protocols (e.g. Valderrama, 1981).

Experimental design

CPOM decomposition and the associated macroinvertebrate community were compared in a field experiment conducted between June 2012 and May 2013 (338 days of exposure time). Stems of the emergent plant *Schoenoplectus californicus* (C. A. Mey) were used as an autochthonous substrate, since it is a frequent and abundant littoral macrophyte that contributes substantial CPOM, as visually confirmed in most aquatic ecosystems in Uruguay. Decomposition mass loss was estimated using the litter bag technique (Graça *et al.*, 2005), using litter bags and collecting replicates at different times. As subtropical streams are generally characterized by high densities of small fishes (Teixeira de Mello *et al.*, 2012; 2014), direct predation on macroinvertebrates was prevented using bags with 4mm knot to knot mesh (rhombus: length 7 mm and width 4 mm).

Stems of *S. californicus* were collected from surrounding streams and transported to the laboratory, where they were cut into 10-cm long fragments, oven dried at 60 °C for 48 h and weighed to obtain their initial dry weight (W_0, Villar *et al.*, 2001). Thirty-three bags, containing five pieces each, were introduced to each studied stream at the beginning of the winter. The bags were attached to ropes parallel to the shore and kept below the water surface all through the experiment. On a monthly basis (total 11), three bags were randomly removed from each stream for further analysis in the laboratory (Supplementary information, Fig. S1, available at http://www.limnetica.net/es/limnetica).

Once in the laboratory, *S. californicus* remains were carefully washed on a 500-µm mesh sieve, from where macroinvertebrates were collected and then fixed with 70 % ethanol for preservation and subsequent identification and analysis (see below). Plant fragment remains were oven dried at 60 °C for 48 h and weighed (0.0001 g) to determine their final dry mass remaining (DMr) (Fig. S1).

Decomposition was calculated, according to Petersen & Cummins (1974), as the percentage of remaining mass (% RM) from the equation:

$$\%R = \left(\frac{W_t}{W_0}\right) \times 100,$$

and as decomposition rate (k) from the exponential model,

$$k = -\left(\frac{1}{t}\right) \times ln\, ln\, \left(\frac{W_t}{W_0}\right),$$

where W_t is the dry weight at time t and W_0 is the initial dry weight.

Exponential decomposition rates (k/day) were estimated as the slope of the linear regressions of the fraction of remaining mass (ln-transformed) against time (days), with free intercept. Also, we performed the same analyses using fixed intercept and we found similar results.

Analysis of macroinvertebrates

Macroinvertebrates were classified to the lowest taxonomic possible level (genus in some cases), using taxonomic keys (Brinkhurst & Marchese, 1989; Lopretto & Tell, 1995; Trivinho- Strixino & Strixino, 1995; Domínguez & Fernandez, 2009). Individuals were counted under a binocular loupe (10×), and related to the respective weight of macrophyte remains (ind/g. DMr). Each taxon was assigned to a functional group, according to previous classifications (Merrit & Webb, 2008; Reynaga, 2009; Ramírez & Gutierrez- Fonseca, 2014; Ferrú & Fierro, 2015).

The functional groups present in our streams were: Shredders (Sh), Collector-Gatherers (CG), Collector-Filters (CF), Scrapers (Sc), Filters (Ft) and Predators (Pr). We calculated Shredders and Scrapers richness as the number of each functional group (FG) per sample (number of taxa/ sample).

The body length of every collected individual was measured from the front of the head to the last segment of the abdomen. In the case of Gastropods, the maximum carapace length was measured (Méthot *et al.*, 2012). Macroinvertebrates biomass was estimated using the length dry weight equation proposed by Meyer (1989), $DryMass = a * L^b$, where a and b are taxa specific constants and L is the measured body length. Constants a and b were assigned according to the literature (McCullough *et al.*, 1979; Smock, 1980; Benke *et al.*, 1999; Miserendino, 2001; Sabo *et al.*, 2002; Baumgartner & Rothhaupt, 2003; Methot *et al.*, 2012; Gualdoni *et al.*, 2013; Rivera-Usme *et al.*, 2014).

Data analysis

To check for differences in physico-chemical variables, paired Student´s t-tests were performed between streams, using the mean value of each parameter between samplings dates (Table 1; data were $\log_{10}$ transformed to fulfil normality requirements).

Decomposition rates were compared using a regression model, and applying a one-way Analysis of Covariance (one-way ANCOVA), with stream as fixed factor, the remaining mass (ln-transformed) as dependent variable and the number of days as covariate. We tested for differences in the slopes between streams, the interaction time*streams. The estimate and the standard error of the slope of each stream were considered as the decomposition rate. The model validation was performed using the visual QQplot and the observed vs fitted residuals values (Faraway, 2014).

Table 1. Physicochemical variables recorded in the intensive and extensive streams analysed in the period between June 2012 and February 2013. Mean, standard deviation and p value (paired Student´s t-test) are shown for each variable. *Variables fisicoquímicas analizadas para el arroyo intensivo y arroyo extensivo en el período junio 2012 – febrero 2013. Para cada variable se muestra media, desvío estándar y p valor (Test de t de Student pareado).*

Stream	Extensive		Intensive		*Student´s t test*
Variables	mean	SD	Mean	SD	*p*
T (ºC)	17.20	5.53	17.36	4.77	>0.05
Cond. (ms/cm)	424	126	384	133	>0.05
pH	7.60	0.23	7.32	0.30	0.01*
TDS (ppm)	0.28	0.08	0.24	0.09	>0.05
OD %	73.94	13.86	50.05	22.80	0.009*
TP (µg/L)	103	54	1183	485	2.4 E-10*
TN (µg/L)	838	137	3099	1958	4.4 E-07*
DTN (µg/L)	618	219	2499	1428	3.2 E-08*
SRP (µg/L)	36	24	697	252	1.1 E-07*
Discharge (L/s)	203	484	94	297	0.04*

* significant differences $p < 0.05$

The differences in the relative density and biomass of the different taxa and functional groups between the streams during the entire experimental period were assessed using non-parametric multivariate variance analysis (PERMANOVA; Anderson, 2001). To determine which taxa and functional groups explained the observed differences, we used similitude permutation analysis (SIMPER; Clarke, 1993). For these analyses, $\log_{(x+1)}$ transformation and Bray-Curtis distance was used. The potential differences in scraper or shredder density, biomass and richness between streams at each last sampling dates, were compared using Student´s t-test.

Analyses were run using used PAST3 (Hammer *et al.*, 2001) and R Studio (R version 3.5.3; R Core Team, 2013) working with a confidence level of 95 % in all cases.

RESULTS

Water variables

The intensive stream was found to have higher concentrations of TP, TN, DTN, and SRP and lower values for dissolved oxygen percentage, than the extensive stream (Table 1). Also, the intensive stream had lower values of pH and discharge, than the extensive stream (Table 1). All the other water variables (i.e. temperature, conductivity, total dissolved solids) showed no significant differences between streams.

Macrophyte decomposition

We found clear differences in CPOM decomposition between the streams, for both approaches followed. The percentage of remaining mass (% RM) of *S. californicus* over time reached lower values in the extensive stream. The 50 % RM occurred at around 163 days (Fig. 2), and after this date differences between streams were more notable. The extensive stream presented a significantly higher decomposition rate than the intensive stream (k = - 0.010 ± 0.0008 days^{-1} and k = - 0.006 ± 0.0005 days^{-1}, respectively; Fig. 2). The regression model was significant (ANCOVA, F: 98.32, $p < 0.01$), and explained 82 % of the variance. The interaction between the factors streams and time was significant (t: 3.87, $p < 0.01$).

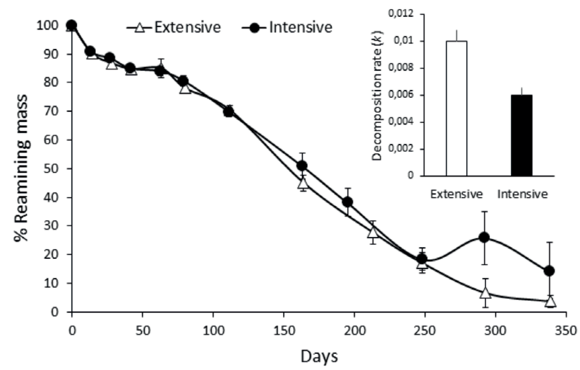

Figure 2. Decomposition of Schoenoplectus californicus mass in both streams, showing the mean and standard error of dry mass % remaining over time. Circles: intensive stream; triangles: extensive stream. Decomposition rates (mean and SE) in the extensive and intensive streams (k) are shown in the small panel. *Descomposición de Schoenoplectus californicus en ambos arroyos, se muestra media y error estándar del % de masa seca remanente en el tiempo. Círculos: arroyo intensivo; Triángulos: arroyo extensivo. Se muestra en el pequeño panel la tasa de descomposición (k) (media y ES) de los arroyos, extensivo e intensivo.*

Macroinvertebrates

A total of 4710 individuals were found, 52 % were found in the extensive stream and 48 % in the intensive stream. The CG functional group was the most abundant group, accounting for 89 % of the total density, however, the SC dominated, with 85 % of the total biomass, principally Gastropoda and Plecoptera. Ephemeroptera was the taxa with the highest representation, 37 %, with 87 % belonging to the Caenidae family. All the Amphipoda belonged to the Hyalellidae family, and they represented 22 % of the total abundance; the Chironomidae family constituted 13 %, composed for the Chironominae subfamily (78 %) and Oligochaeta represented the 18 %; with individuals only from the Naididae family. Ephemeroptera, Amphipoda, Chironomidae and Oligochaeta taxa representing 94 % of the total abundance recorded.

The density of macroinvertebrates in the stems of *S. californicus* increased over time in both systems, with more noticeable increases at the end of the sampling period (Fig. 3). In the extensive stream, the final three sampling dates had, on average, a total density of 261 ± 84 ind/g.DMr (mean and standard error) compared to an initial density of 30 ± 10 ind/g.DMr recorder for the earliest eight dates. In the intensive stream, in contrast, the average over the last three dates was 150 ± 44 ind/g.DMr compared to an initial density of 32 ± 15 ind/g.DMr for the first eight sampling dates (Fig. 3).

Differences in the relative density of the community taxa between streams were detected (Table 2). The SIMPER analysis showed an overall average dissimilarity of 66 %. Ephemeroptera accounted for 29 % of this difference, showing higher relative density in the extensive stream than in the intensive stream (56 % and 16 %, respectively). Amphipoda with 24 % (extensive: 4 %; intensive: 42 %), Oligochaeta with 19 %

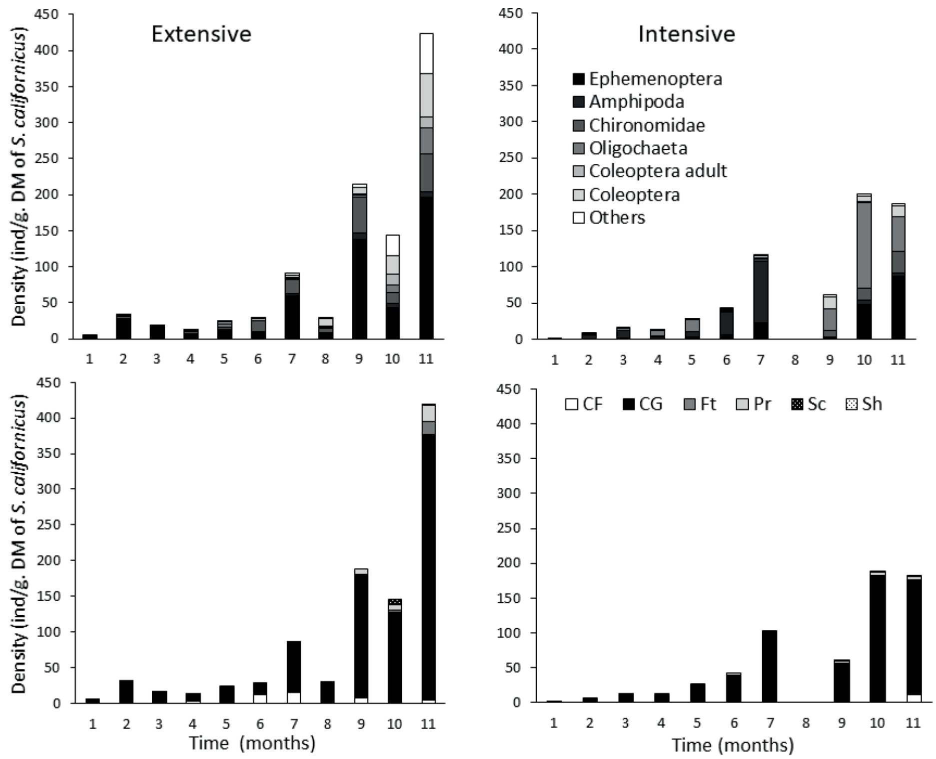

Figure 3. Density of principal macroinvertebrate taxa (above) and functional groups (FG, below), showing mean values in each system at each sampling time (months). Left = Extensive stream. Right = Intensive stream. Taxa and functional group codes are presented on the right. Please note data missing for the intensive stream in the eighth month. Codes: Sh = Shredders; Sc = Scrapers; Pr = Predators; Ft = Filterers; CG = Collector- Gatherers; CF = Collector-Filterers. *Densidad de los principales taxones (arriba) y grupos funcionales (FG, abajo) de macroinvertebrados. Se muestran valores medios para cada sistema y para cada fecha de muestreo (meses). Izquierda = Arroyo extensivo. Derecha = Arroyo intensivo, a la derecha de la gráfica referencia de cada taxón y grupo funcional. Por favor, atención a la la falta de información en el mes 8 del arroyo intensivo. Códigos: Sh = Fragmentadores; Sc = Raspadores; Pr = Depredadores; Ft = Filtradores; CG = Colectores- Recolectores; CF = Colectores- Filtradores.*

(extensive: 7 %; intensive: 30 %) and Chironomidae with 12 % (extensive: 19 %; intensive: 6 %) explained 84 % of the observed variance between streams. In the intensive stream, Oligochaeta and Amphipoda taxa dominated over time (Fig. 3). In contrast, the relative density of the FGs was similar between streams (Table 2).

The biomass of macroinvertebrates followed the same tendency as that of density, with a noticeable increase in the last three sampling dates (Fig. 4). In the extensive stream, the last three sampling dates produced an average of 485 ± 87 mg/g.DMr, while during the earliest eight dates macroinvertebrate biomass was on average 23 ± 8 mg/g (Fig. 4). In the intensive stream the last sampling dates recorded an average of 63 ±

Table 2. Comparative analysis of the extensive stream and the intensive stream, in terms of relative density and biomass of macroinvertebrates taxa and functional groups (FG), using a non-parametric multivariate variance analysis (PERMANOVA). *Análisis comparativo entre el arroyo extensivo y el arroyo intensivo, en términos de densidad y biomasa relativa de los diferentes taxones de macroinvertebrados y grupos funcionales (FG), utilizando un análisis de varianza multivariado no paramétrico (PERMANOVA).*

Variable	Factor	SS	F	*p*
Density taxa (ind/g.DM)	Stream	3.5	10.5	0.0001
Density FG (ind/g.DM)	Stream	1.1	-0.1	0.99
Biomass taxa (mg/g.DM)	Stream	3.8	5.7	0.001
Biomass FG (mg/g.DM)	Stream	1.4	0.9	0.49

25 mg/g while the macroinvertebrate biomass for the early dates was 18 ± 8 mg/g (Fig. 4).

The relative biomass of taxa also differed between streams (Table 2). The SIMPER analysis showed an overall average dissimilarity of 64 %, with Ephemeroptera accounting for 33 % of this difference (a similar proportion was found between streams; 45 % and 49 % of the total biomass, in the extensive and intensive stream, respectively). Moreover, Oligochaeta contributed to the differences with a 21 % (extensive: 1 %; intensive: 36 %), Amphipoda with 13 % (extensive: < 1 %; intensive: 7 %), and Gastropoda with 10 % (extensive: 25 %; intensive: 1 %) explained 77 % of the observed variance. No significant differences in the relative biomass of the different functional groups were found (Table 2).

Low shredder density and biomass was observed in both streams; only *Aegla* sp. (Crustacea, Aeglidae) and the genus *Cricotopus* sp. (Diptera, chironomidae) represented taxa associated with this functional group in both streams. On the other hand, the scraper functional group was represented by the families Ampullaridae, Hydrobiidae, Notoneumoridae and Polycentropodidae in the extensive stream, and Hydrobiidae and

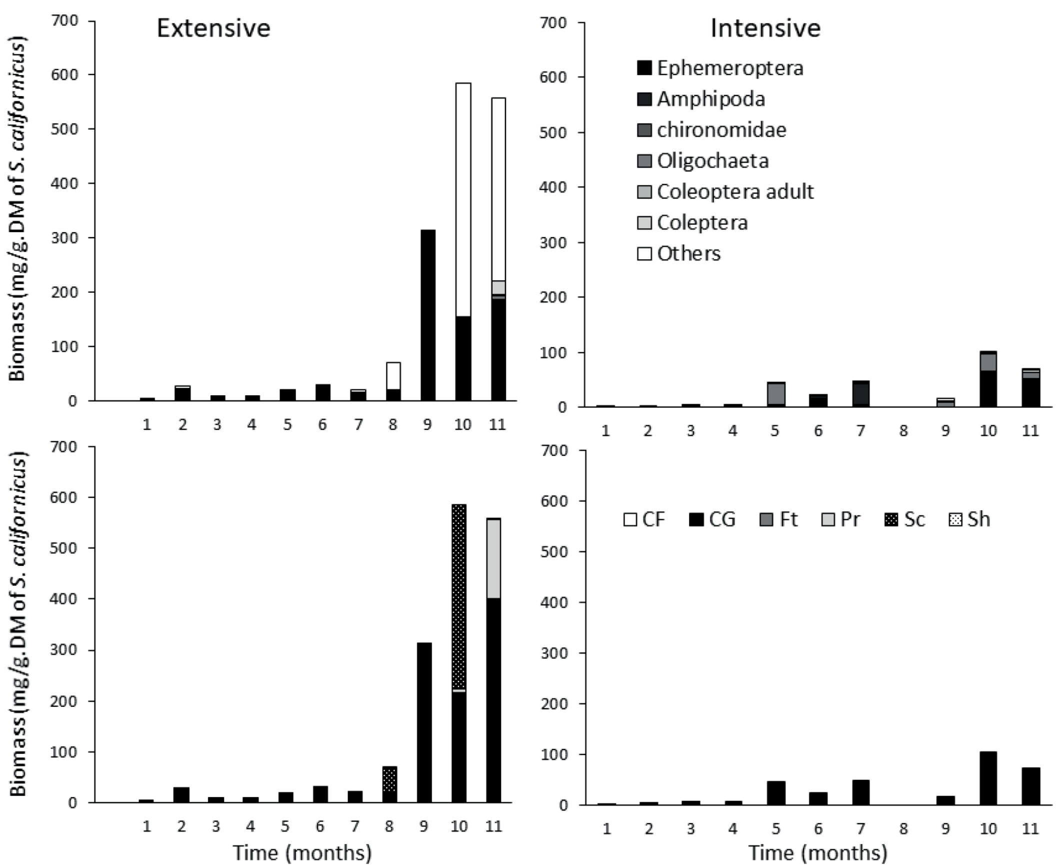

Figure 4. Biomass of principal macroinvertebrate taxa (above) and functional group ((FG), below), showing mean values for each system at each sampling time (months). Left = Extensive stream. Right = Intensive stream. Taxa and functional group coding is presented on the right. Please note data missing for the intensive stream in month eighth. Codes: Sh = Shredders; Sc = Scrapers; Pr = Predators; Ft = Filterers; CG = Collector- Gatherers; CF = Collector-Filterers. *Biomasa de los principales taxones (arriba) y grupos funcionales (FG, abajo) de macroinvertebrados, se muestran valores medios para cada sistema y para cada fecha de muestreo (meses). Izquierda = Arroyo extensivo. Derecha = Arroyo intensivo, a la derecha de la gráfica referencia de cada taxón y grupo funcional. Por favor, atención a la falta de información en el arroyo intensivo en el mes 8. Códigos: Sh = Fragmentadores; Sc = Raspadores; Pr = Depredadores; Ft = Filtradores; CG = Colectores- Recolectores; CF = Colectores- Filtradores.*

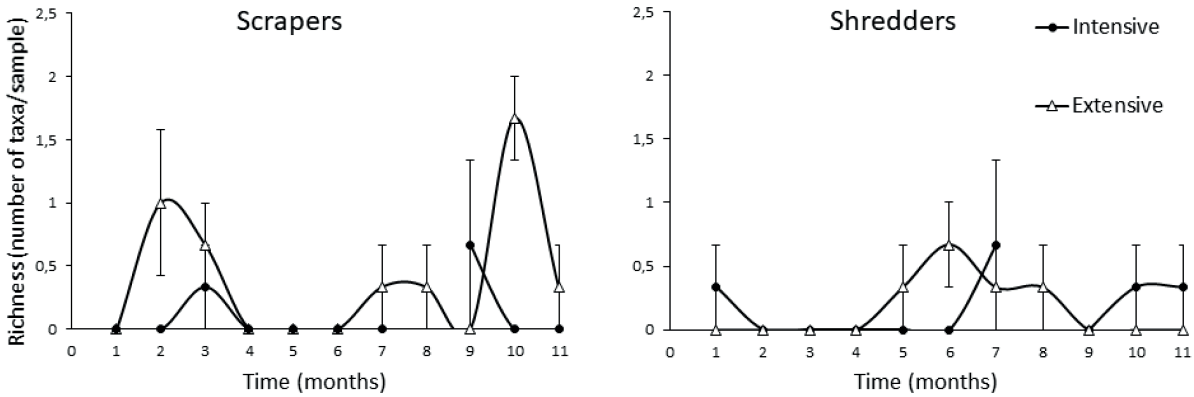

Figure 5. Richness (as number of taxa) of scraper and shredder functional groups in each system (extensive and intensive stream), showing means and SE at each sampling time (months). *Riqueza (número de taxones) de los grupos funcionales raspadores y fragmentadores en cada sistema (arroyo extensivo e intensivo), se muestran valores medios y error estándar para cada fecha de muestreo (meses).*

Succineidae in the intensive stream. Scraper density and biomass increased in the two last sampling dates, with significant differences only in the 10th sampling date. The richness of shredder and scraper groups showed lower values over time, with an apparent peak on the richness of scrapers in the extensive stream on the 10th sampling date (Fig. 5).

DISCUSSION

Through this study we have tested a series of hypotheses that were partially confirmed. Thus, the stream on a more intense land use: 1) had a lower water quality, 2) presented a simpler macroinvertebrate community with a significant reduction of density and biomass of scrapers (but not shredders), and as a consequence 3) showed a lower CPOM decomposition rate.

We found a higher relative density of Ephemeroptera, in particular Caenidae (CGs), in the extensive stream (with intermediate nutrients concentrations, higher dissolved oxygen levels, pH and discharge), whereas a higher relative density and biomass of Oligochaeta and Amphipoda occurred in the intensive stream (displaying higher nutrient concentrations and lower values for dissolved oxygen, pH and discharge). Ephemeroptera can tolerate a wide range of nutrient values (Sampaio *et al.*, 2008; Chang *et al.*, 2014; Morelli & Verdi, 2014; Berger *et al.*, 2016), and their emergence and growth depend heavily on dissolved oxygen concentrations (Connolly *et al.*, 2004). In contrast, many Oligochaeta (CGs; Tachet *et al.*, 2000; Mandaville, 2002; Marchese, 2009) and Amphipoda (CGs; Ríos-Touma & Prat, 2004; Peralta & Grosso, 2009; Ríos-Touma *et al.*, 2014) are tolerant of conditions where high nutrient values and poor ecosystem health prevail, which were the characteristics of our intensive stream. The consequences of intensive land use on water quality, such as higher nutrient concentrations, lower values of dissolved oxygen, and pH, are common stressors for the macroinvertebrate community (Ashton *et al.*, 2014; Fierro *et al.*, 2017; Solis *et al.*, 2019; Peralta *et al.*, 2020).

Macroinvertebrate communities play a key role in the organic matter decomposition process, mainly due to the presence of shredders (Graça, 2001). This group is particularly sensitive to excessive nutrient levels in the water, which can lead to their impoverishment or inhibition (Woodward *et al.*, 2012) and a lower CPOM decomposition rate. However, shredders occurred at low density in both of our streams, regardless of nutrient concentration, which is consistent with the typically low abundance and diversity of this functional group in tropical and subtropical streams (Morelli & Verdi, 2014; Rezende *et al.*, 2016; Calderón *et al.*, 2019). In contrast, the scraper group became more important (in terms of density, biomass and richness) over the last sampling dates in the extensive stream. This result suggests that the higher decomposition rate observed in the extensive stream could have been promoted by the higher biomass of scrapers, which could be functioning as shredders when scraping periphyton from substrate surfaces (Wantzen & Wagner, 2006; Tonin *et al.*, 2018).

In agreement with previous studies, the decomposition rate in our experiment was initially the same in the two streams, with a rapid loss of biomass corresponding to the loss of soluble and labile matter by leaching and microbial metabolism (Graça *et al.*, 2015; Ferreira *et al.*, 2016b). Studies of the decomposition of *S. californicus* in streams are scarce, but the rates found here were higher than those reported in the floodplain marsh of the lower Paraná River (Argentina, Villar *et al.*, 2001), suggesting that the *S. californicus* decomposition rate may respond differently under different environmental conditions.

The macrophyte *S. californicus* could be an effective tool for evaluating the potential effects of changes in land use in the watersheds, when it is subjected to prolonged exposure to the water. However, these results should be considered with caution and supported with more studies, due to the lack of replication of streams for each type of land use (extensive and intensive). Evidence provided here constitutes a first approximation in understanding how land use intensification can indirectly affect a key ecosystem function in two lowland subtropical streams. This species can be a relevant biomonitoring tool for decomposition measurement, particularly in those subtropical streams with little or no riparian forest, where the coverage of *S. californicus* is common and it can represent an autochthonous source of organic matter.

ACKNOWLEDGEMENTS

We acknowledge Carsten Brian Nielsen for technical assistance in the setup of monitoring stations in Uruguay. We are grateful for field assistance from colleagues and students, the logistic support of Daniella Agrati and family, and landowners (Mr Mendiverri & Mr O. Laturré). G. Goyenola, C. Iglesias, M. Meerhoff, J.P. Pacheco and F. Teixeira de Mello received support from the SNI (Agencia Nacional de Investigación e Innovación, ANII, Uruguay and PEDECIBA). The study was funded by the Danish Council for Independent Research, a grant by ANII-FCE (2009-2749) Uruguay, and the National L'Oreal-UNESCO award for Women in Science - Uruguay with support from DICyT granted to M. Meerhoff.

REFERENCES

ANDERSON, M. J. 2001. A new method for non-parametric multivariate analysis of variance. *Austral Ecology*, 26: 32-46. DOI: 1442-9993.2001.01070

ASHTON, M. J., R. P. MORGAN & S. STRANKO. 2014. Relations between macroinvertebrates, nutrients, and water quality criteria in wadeable streams of Maryland, USA. *Environmental Monitoring and Assessment*, 186(2): 1167–1182. DOI: 10.1007/s10661-013-3447-1

BASTIAS, E., M. RIBOT, A. M. ROMANÍ, J. MORA-GÓMEZ, F. SABATER, P. LÓPEZ. & E. MARTÍ. 2018. Responses of microbially driven leaf litter decomposition to stream nutrients depend on litter quality. *Hydrobiologia*, 806(1). DOI: 10.1007/s10750-017-3372-3

BAUMGÄRTNER, D., & K. O. ROTHHAUPT. 2003. Predictive length–dry mass regressions for freshwater invertebrates in a pre-alpine lake littoral. *International Review of Hydrobiology*, 88(5): 453–463. DOI: 10.1002/iroh.200310632

BENKE, A. C., A. D. HURYN, L. A. SMOCK, J. B. WALLACE, A. C. BENKE, A. D. HURYN, L. A. SMOCK & J. B. WALLACE. 1999. Length-Mass Relationships for Freshwater Macroinvertebrates in North America with Particular Reference to the Southeastern United States. *Journal of the North American Benthological Society*, 18(3): 308–343.

BERENZEN, N., R. SCHULZ & M. LIESS. 2001. Effects of chronic ammonium and nitrite contamination on the macroinvertebrate community in running water microcosms. *Water Research*, 35: 3478–3482. DOI: 10.1016/S0043-1354(01)00055-0

BERGER, E., P. HAASE, M. OETKEN & A. SUNDERMANN. 2016. Field data reveal low critical chemical concentrations for river benthic invertebrates. *Science of The Total Environment*, 544: 864–873. DOI: 10.1016/j.scitotenv.2015.12.006

BIANCHINI Jr., I. 1999. Aspectos do processo de decomposição nos ecossistemas aquáticos continentais. In *Perspectivas da limnologia no Brasil*. M. L. M. Pompêo (ed.): 21–43. Gráfica e Editora União. São Luís, Brazil.

BRINKHURST R. O. & M. R. MARCHESE. 1989. *Guide to freshwater Aquatic Oligochaeta of South and Central America*. Colección CLIMAX 6. Asociación ciencias naturales del Litoral. Santo Tomé, Argentina.

CALDERÓN DEL CID, C., R. S. REZENDE, A. R. CALOR, J. S. DAHORA, L. N. DE ARAGÃO, M. L. GUEDES, A. N. CAIAFA. & A. O. MEDEIROS. 2019. Temporal dynamics of organic matter, hyphomycetes and invertebrate communities in a Brazilian savanna stream. *Community Ecology*, 20(3): 301–313. DOI: 10.1556/168.2019.20.3.10

CHANG, F. H., J. E. LAWRENCE, B. RIOS-TOUMA & V. H. RESH. 2014. Tolerance values of benthic macroinvertebrates for stream biomonitoring: assessment of assumptions underlying scoring systems worldwide. *Environmental Monitoring and Assessment*, 186(4): 2135–2149. DOI: 10.1007/s10661-013-3523-6

CHEEVER, B. M., E. B. KRATZER & J. R. WEBSTER. 2012. Immobilization and mineralization of N and P by heterotrophic microbes during leaf decomposition. *Freshwater Sciences*, 31: 133–147. DOI: 10.1899/11-060.1

CLARKE, K. R. 1993. Non-parametric multivariate analysis of changes in community structure. *Australian Journal of Ecology*, 18: 117–143. DOI: 10.1111/j.1442-9993.1993.tb00438.x

CONNOLLY, N. M., M. R. CROSSLAND & R. G. PEARSON. 2004. Effect of low dissolved oxygen on survival, emergence, and drift of tropical stream macroinvertebrates. *Journal of the North American Benthological Society*, 23(2): 251–270. DOI: 10.1899/0887-3593(2004)023<0251:EOLDOO>2.0.CO;2

CUMMINS, K. W., J. R. SEDELL., F. J. SWANSON., G. W. MINSHALL., S. G. FISHER, C. E. CUSHING, R. C. PETERSEN & R. L. VANNOTE. 1983. Organic matter budgets for stream ecosystems: problems in their evaluation. In: *Stream Ecology*. J. R. Barnes & G. W. Minshall (eds.):299–353. Springer. Boston, U.S.A. DOI: 10.1007/978-1-4613-3775-1_13

DOMÍNGUEZ E. & H. R. FERNÁNDEZ. 2009. *Macroinvertebrados bentónicos sudamericanos. Sistemática y biología*. 1a ed. Tucumán: Fundación Miguel Lillo. Argentina.

FARAWAY, J. J. 2014. Linear models with R. CRC Press. Boca Raton. U.S.A.

FERREIRA, V., B. CASTAGNEYROL, J. KORICHEVA, V. GULIS, E. CHAUVET & M. A. S. GRAÇA. 2015. A meta-analysis of the effects of nutrient enrichment on litter decomposition in streams. *Biological Reviews*, 90(3): 669–688. DOI: 10.1111/brv.12125

FERREIRA, V., J. KORICHEVA, J. POZO & M. A. S. GRAÇA. 2016a. A meta-analysis on the effects of changes in the composition of native forests on litter decomposition in streams. *Forest Ecology and Management*, 364: 27–38. DOI: 10.1016/j.foreco.2016.01.002

FERREIRA, V., J. CASTELA., P. ROSA, A. M. TONIN, L. BOYERO & M. A. S. GRAÇA. 2016b. Aquatic hyphomycetes, benthic macroinvertebrates and leaf litter decomposition in streams naturally differing in riparian vegetation. *Aquatic Ecology*, 50(4): 711–725. DOI: 10.1016/j.funeco.2015.08.001

FERREIRA, V., L. BOYERO, C. CALVO, F. CORREA, R. FIGUEROA, J. F. GONÇALVES, G. GOYENOLA, M. A. S. GRAÇA, L. HEPP, S. KARIUKI, A. LÓPEZ-RODRÍGUEZ, N. MAZZEO, C. M'ERIMBA, S. MONROY, A. PEIL, J. POZO, R. REZENDE & F. TEIXEIRA DE MELLO. 2018. A global assessment of the effects of eucalyptus plantations on stream ecosystem functioning. *Ecosystems*, 20: 1–14. DOI: 10.1007/s10021-018-0292-7

FERRU, M., & P. FIERRO. 2015. Estructura de macroinvertebrados acuáticos y grupos funcionales tróficos en la cuenca del río Lluta, desierto de Atacama, Arica y Parinacota, Chile. *Idesia (Arica)*, 33(4): 47–54. DOI: 10.4067/S0718-34292015000400007

FIERRO, P., C. BERTRÁN, J. TAPIA, E. HAUENSTEIN, F. PEÑA-CORTÉS, C. VERGARA, C. CERNA, & L. VARGAS-CHACOFF. 2017. Effects of local land-use on riparian vegetation, water quality, and the functional organization of macroinvertebrate assemblages. *Science of the Total Environment*, 609: 724–734. DOI: 10.1016/j.scitotenv.2017.07.197

FOLEY, J. A., R. DEFRIES, G. P. ASNER, C. BARFORD, G. BONAN, S. R. CARPENTER, F. S. CHAPIN, M. T. COE, G. C. DAILY, H. K. GIBBS, J. H. HELKOWSKI, T. HOLLOWAY, E. A. HOWARD, C. J. KUCHARIK, C. MONFREDA, J. A. PATZ, I. C. PRENTICE, N. RAMANKUTTY & P. K. SNYDER. 2005. Global Consequences of Land Use. *Science*, 309 (5734): 570 –574. DOI: 10.1126/science.1111772

GESSNER, M. O. & E. CHAUVET. 2002. A case for using litter breakdown to assess functional stream integrity. *Ecological Applications*, 12 (2): 498–510. DOI: 10.2307/3060958

GOYENOLA, G., M. MEERHOFF, F. TEIXEIRA-DE MELLO, I. GONZÁLEZ-BERGONZONI, D. GRAEBER, C. FOSALBA. N. VIDAL, N. MAZZEO, N. B. OVESEN, E. JEPPESEN & B. KRONVANG. 2015. Monitoring strategies of stream phosphorus under contrasting climate-driven flow regimes. *Hydrology and Earth System Sciences*, 19(10): 4099–4111. DOI: 10.5194/hess-19-4099-2015

GOYENOLA, G., D. GRAEBER, M. MEERHOFF, E. JEPPESEN, F. TEIXEIRA DE MELLO, N. VIDAL, C. FOSALBA, N. B. OVESEN, J. GELBRECHT, N. MAZZEO. & B. KRONVANG. 2020. Influence of Farming Intensity and Climate on Lowland Stream Nitrogen. *Water*, 12(4): 1021. DOI: 10.3390/w12041021

GRAÇA, M. A. S. 2001. The role of invertebrates on leaf litter breakdown in a streams - a review. *International Review of Hydrobiology*, 86: 383–393. DOI: 10.1002/1522-2632(200107)86:4/5<383::AID-IROH383>3.0.CO;2-D

GRAÇA, M. A. S., F. BARLOCHER. & M. O. GESSNER. 2005. *Methods to Study Litter Decomposition: A Practical Guide*. Springer. Dordrecht, Netherlands. DOI: 10.1007/1-4020-3466-0

GRAÇA, M. A. S. & C. CANHOTO. 2006. Leaf litter processing in low order streams. *Limnetica*, 25(1–2): 1–10. DOI: 10.23818/limn.25.01

GRAÇA, M. A. S., V. FERREIRA, C. CANHOTO, A. C. ENCALADA, F. GUERRERO-BOLAÑO, K. M. WANTZEN & L. BOYERO. 2015. A conceptual model of litter breakdown in low order streams. *International Review of Hydrobiology*, 100(1): 1–12. DOI: 10.1002/iroh.201401757

GRAEBER, D., T. M. JENSEN, J. J. RASMUSSEN, T. RIIS, P. WIBERGLARSEN & A. BAATTRUP-PEDERSEN. 2017. Multiple stress response of lowland stream benthic macroinvertebrates depends on habitat type. *Science of the Total Environment*, 599–600: 1517–1523. DOI: 10.1016/j.scitotenv.2017.05.102

GUALDONI, C., P. W. FRENCH, & A. M. OBERTO. 2013. Relaciones longitud-biomasa en macroinvertebrados bentónicos de un arroyo serrano del sur de Córdoba, Argentina. *Ecología Austral*, 23: 194–201.

HAMMER, Ø., D. A. HARPER & P. D. RYAN. 2001. PAST: paleontological statistics software package for education and data analysis. *Palaeontologia Electronica*, 4(1): 1–9.

HUMPHRIES, P., H. KECKEIS & B. FINLAYSON. 2014. The river wave concept: integrating river ecosystem models. *BioScience*, 64(10): 870–882. DOI: 10.1093/biosci/biu130

JUNK, J. W., P. B. BAYLEY & R. E. SPARKS. 1989. The flood pulse concept in river floodplain systems. *Canadian Special Publication of Fisheries and Aquatic Sciences*, 106: 110–127.

LAMPERT, W & U. SOMMER. 2007. *Limnoecology: the ecology of lakes and streams*. Oxford university press. Oxford, U. K.

LECERF, A., P. USSEGLIO-POLATERA, J. Y. CHARCOSSET, D. LAMBRIGOT, B. BRACHT & E. CHAUVET. 2006. Assessment of functional integrity of eutrophic streams using litter breakdown and benthic macroinvertebrates. *Archiv für Hydrobiologie*, 165: 105–126. DOI: 10.1127/0003-9136/2006/0165-0105

LOPRETTO, E., & G. TELL. 1995. *Ecosistemas de aguas continentales. Metodologias para su estudio*. Ediciones Sur. La Plata, Argentina.

MANDAVILLE, S. M. 2002. Benthic Macroinvertebrates in Freshwaters - Taxa Tolerance Values, Metrics, and Protocols. *Soil & Water Conservation Society of Metro Halifax*. Nova Scotia, Canada.

MANNING, D. W. P., A. D. ROSEMOND, V. GULIS, J. P. BENSTEAD, J. S. KOMINOSKI & J. C. MAERZ. 2016. Convergence of detrital stoichiometry predicts thresholds of nutrient-stimulated breakdown in streams. *Ecological Applications*, 26(6): 1745–1757. DOI: 10.1890/15-1217.1

MARCHESE, M. R. 2009. Annelida Oligochaeta. In E. Domínguez & H. R. Fernández (eds.): 551-564. *Macroinvertebrados bentónicos sudamericanos: sistemática y biología*. Fundación Miguel Lillo. Tucumán, Argentina.

MCCULLOUGH, D. A., G. W. MINSHALL & C. E. CUSHING. 1979. Bioenergetics of lotic filter-feeding insects *Simulium* spp. (Diptera) and *Hydropsyche occidentalis* (Trichoptera) and their function in controlling organic transport in streams. *Ecology*, 60(3): 585-596. DOI: 10.2307/1936079

MEHRING, A. S., K. A. KUEHN, A. THOMPSON, G. VELLIDIS, C. M. PRINGLE, A. D. ROSEMOND., M. R. FIRST, R. R. LOWRANCE. & G. VELLIDIS. 2015. Leaf litter nutrient uptake in an intermittent blackwater river: influence of tree species and associated biotic and abiotic drivers. *Functional Ecology*, 29: 849- 860. DOI: 10.1111/1365-2435.12399

MERRITT, R. W., & D. W. WEBB. 2008. Aquatic diptera (part two): pupae and adults of aquatic diptera. In: *An Introduction to the Aquatic insects of North America*, R. W. Merritt, K. W. Cummins & M. B. Berg (eds.): 723–800. Kendall Hunt Publishing, Dubuque, U.S.A.

MÉTHOT, G., C. HUDON, P. GAGNON, B. PINEL-ALLOUL, A. ARMELLIN. & A. M. T. POIRIER. 2012. Macroinvertebrate size–mass relationships: how specific should they be? *Freshwater Science*, 31(3): 750–764. DOI: 10.1899/11-120.1

MEYER, E. 1989. The relationship between body length parameters and dry mass in running water invertebrates. *Archiv fur Hydrobiologie*, 117: 191–203.

MISERENDINO, M. L. 2001. Length–mass relationships for macroinvertebrates in freshwater environments of Patagonia (Argentina). *Ecología Austral*, 11: 3–8.

MORELLI, E., & A. VERDI. 2014. Diversidad de macroinvertebrados acuáticos en cursos de agua dulce con vegetación ribereña nativa de Uruguay. *Revista Mexicana de Biodiversidad*, 85(4): 1160–1170. DOI: 10.7550/rmb.45419 1161

PARUELO, J., G. PINEIRO, G. BALDI, S. BAEZA, F. LEZAMA, A. ALTESOR & M. OESTERHELD. 2010. Carbon Stocks and Fluxes in Rangelands of the Río de la Plata Basin. *Rangeland Ecology & Management*, 63: 94–108. DOI: 10.2111/08-055.1

PERALTA, M., & L. E. GROSSO. 2009. Crustacea, Syncarida, Amphipoda y Decapoda. In: *Macroinvertebrados bentónicos sudamericanos: sistemática y biología*, E. Domínguez & H. R. Fernández (eds.): 469–495. Fundación Miguel Lillo. Tucumán, Argentina.

PERALTA, E. M., L. S. BATUCAN, I. B. B. DE JESUS, E. M. C. TRIÑO, Y. UEHARA, T. ISHIDA, Y. KOBAYASHI, C-Y. KO, T. IWATA, A. S. BORJA, J. C. A. BRIONES, R. D. S, PAPA, F. S. MAGBANUA & N. OKUDA. 2020. Nutrient loadings and deforestation decrease benthic macroinvertebrate diversity in an urbanised tropical stream system. *Limnologica*, 80, 125744. DOI: 10.1016/j.limno.2019.125744

PETERSEN, R. C., & K. W. CUMMINS. 1974. Leaf processing in a woodland stream. *Freshwater Biology*, 4(4): 343–368. DOI: 10.1111/j.1365-2427.1974.tb00103.x

PIGGOTT, J. J., K. LANGE, C. R. TOWNSEND & C. D. MATTHAEI. 2012. Multiple Stressors in Agricultural Streams: A Mesocosm Study of Interactions among Raised Water

Temperature, Sediment Addition and Nutrient Enrichment. *PLOS One*, 7(11), e49873. DOI: 10.1371/journal.pone.0049873

R Core Team. 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

RAMIREZ, A. & P. E. GUTIERREZ-FONSECA. 2014. Estudios sobre macroinvertebrados acuaticos en America Latina: avances recientes y direcciones futuras. *Revista de Biologia Tropical*, 62: 9–20. DOI: 10.15517/rbt.v62i0.15775

REYNAGA, M. C. 2009. Hábitos alimentarios de larvas de Trichoptera (Insecta) de una cuenca subtropical. *Ecología Austral*, 19(3): 207–214.

REZENDE, R., M. A. S. GRAÇA, A. M. DOS SANTOS, A. O. MEDEIROS, P. F. SANTOS, Y. R. NUNES & J. F. GONÇALVES, Jr. 2016. Organic matter dynamics in a tropical gallery forest in a grassland landscape. *Biotropica*, 48(3): 301–310. DOI: 10.1111/btp.12308

RÍOS-TOUMA, B. P., & N. PRAT. 2004. Estudio de las condiciones de referencia de las cuencas de los ríos Pita, San Pedro y Machángara. Informe 2: Estado ecológico de los ocho puntos del sistema de monitoreo ambiental de la EMAAP-Q estudiados (San Pedro, Machángara y Guayllabamba). Barcelona: Universitat de Barcelona

RÍOS-TOUMA, B., R. ACOSTA & N. PRAT. 2014. The Andean biotic index (ABI): revised tolerance to pollution values for macroinvertebrate families and index performance evaluation. *Revista de Biologia Tropical*, 62: 249–273. DOI: 10.15517/rbt.v62i0.15791

RIVERA-USME, J. J., G. A. PINILLA-AGUDELO, D. L. CAMACHO-PINZÓN, M. I. CASTRO-REBOLLEDO, & J. O. RANGEL-CHURIO. 2014. Length-mass relationships in the aquatic invertebrate genera Helobdella (Hirudinea: Glossiphoniidae) and Asellus (Crustacea: Asellidae) of an Andean wetland of Colombia. *Actualidades Biológicas*, 36(100): 39–45. DOI: 10.1080/01650424.2014.1001400

SABO, J. L., J. L. BASTOW & M. E. POWER. 2002. Length–mass relationships for adult aquatic and terrestrial invertebrates in a California watershed. *Journal of the North American Benthological Society*, 21(2): 336–343. DOI: 10.2307/1468420

SAMPAIO, A., P. RODRIGUEZ-GONZALEZ, S. VARANDAS, R. M. CORTES & M. T. FERREIRA. 2008. Leaf litter decomposition in western Iberian forested wetlands: lentic versus lotic response. *Limnetica*, 27(1): 93–106. DOI: 10.23818/limn.27.08

SMOCK, L. A. 1980. Relationships between body size and biomass of aquatic insects. *Freshwater Biology*, 10: 375–383. DOI: 10.1111/j.1365-2427.1980.tb01211.x

SOLIS, M., M. ARIAS, S. FANELLI, C. BONETTO & H. MUGNI. 2019. Agrochemicals' effects on functional feeding groups of macroinvertebrates in Pampas streams. *Ecological Indicators*, 101: 373–379. DOI: 10.1016/j.ecolind.2019.01.036

SUBERKROPP, K. & E. CHAUVET. 1995. Regulation of leaf breakdown by fungi in streams: influences of water chemistry. *Ecology*, 76: 1433–1445. DOI: 10.2307/1938146

TACHET, H., P. RICHOUX, M. BOURNAUD & P. USSEGLIO-POLATERA. 2000. *Invertébrés d'Eau Douce: Systématique, Biologie, Écologie*. CNRS Éditions. Paris, France.

TANK, J. L., E. J. ROSI-MARSHALL, N. A. GRIFFITHS, S. A. ENTREKIN & M. L. STEPHEN. 2010. A review of allochthonous organic matter dynamics and metabolism in streams. *Journal of the North American Benthological Society*, 29(1): 118–146. DOI: 10.1899/08-170.1

TANT, C.J., A. D. ROSEMOND., & M. R. FIRST. 2013. Stream nutrient enrichment has a greater effect on coarse than on fine benthic organic matter. *Freshwater Science*, 32: 1111–1121. DOI: 10.1899/12-049.1

TEIXEIRA DE MELLO, F., M. MEERHOFF, A. BAATTRUP-PEDERSEN, T. MAIGAARD, P. B. KRISTENSEN, T. K. ANDERSEN, J. M. CLEMENTE, C. FOSALBA, E. A. KRISTENSEN, M. MASDEU, T. RIIS, N. MAZZEO, E. JEPPESEN. 2012. Community structure of fish in lowland streams differ substantially between subtropical and temperate climates. *Hydrobiologia*, 684: 143–160.

DOI: 10.1007/s10750-011-0979-7

TEIXEIRA DE MELLO, F., E. A. KRISTENSEN, M. MEERHOFF, I. GONZÁLEZ-BERGONZONI, A. BAATTRUP-PEDERSEN, C. IGLESIAS, P. B. KRISTENSEN, N. MAZZEO & E. JEPPESEN. 2014. Monitoring fish communities in wadeable lowland streams: comparing the efficiency of electrofishing methods at contrasting fish assemblages. *Environmental Monitoring and Assessment*, 186(3): 1665–1677. DOI: 10.1007/s10661-013-3484-9

THORP, J. H. & M. D. DELONG. 1994. The riverine productivity model: a heuristic view of carbon sources and organic processing in large river ecosystems. *Oikos*, 70(2): 305-308. DOI: 10.2307/3545642

TIEGS, S. D., D. M. COSTELLO, M. W. ISKEN, G. WOODWARD, P. B. MCINTYRE, M. O. GESSNER, E. CHAUVET, N. A. GRIFFITHS, A. S. FLECKER, V. ACUÑA, R. ALBARIÑO, D. C. ALLEN, C. ALONSO, P. ANDINO, C. ARANGO, J. AROVIITA, M. V. M. BARBOSA, L. A. BARMUTA, C. V. BAXTER, T. D. C. BELL, B. BELLINGER, L. BOYERO, et al. 2019. Global patterns and drivers of ecosystem functioning in rivers and riparian zones. *Science Advances*, 5 (1): eaav0486. DOI: 10.1126/sciadv.aav0486

TONIN, M. A., L. U. HEPP & J. F. GONÇALVES, Jr. 2018. Spatial variability of plant litter decomposition in stream networks: from litter bags to watersheds. *Ecosystems*, 21(3): 567–581. DOI: 10.1007/s10021-017-0169-1

TRIVINHO-STRIXINO, S & G. STRIXINO. 1995. Larvas de Chironomidae (Diptera) do Estado de São Paulo: guia de identificação e diagnose dos gêneros. São Carlos PPG/ERN/UFSCar, Brazil.

VALDERRAMA, J. C. 1981. The simultaneous analysis of total nitrogen and total phosphorus in natural waters. *Marine Chemistry*, 10(2): 109–122. DOI: 10.1016/0304-4203(81)90027-X

VANNOTE, R. L., G. W. MINSHALL, K. W. CUMMINS, J. R. SEDELL & C. E. CUSHING. 1980. The river continuum concept. *Canadian journal of fisheries and aquatic sciences*, 37(1): 130–137. DOI: 10.1139/cjfas-2019-0080

VILLAR, C. A., L. DE CABO, P. VAITHIYANATHAN & C. BONETTO. 2001. Litter decomposition of emergent macrophytes in a floodplain marsh of the Lower Paraná River. *Aquatic Botany*, 70(2): 105–116. DOI: 10.1016/S0304-3770(01)00149-8

WANTZEN, K. M & R. WAGNER. 2006. Detritus processing by invertebrate shredders: a neotropical–temperate comparison. *Journal of the North American Benthological Society*, 25: 216–232. DOI: 10.1899/0887-3593(2006)25[216:dpbisa]2.0.co;2

WOODWARD, G., M. O. GESSNER, P. S. GILLER, V. GULIS, S. HLADYZ, A. LECERF, B. MALMQVIST, B. G. MCKIE, S. D. TIEGS, H. CARISS, M. DOBSON, A. ELOSEGI, V. FERREIRA, M.A.S. GRAÇA, T. FLEITUCH, J. O. LACOURSIÈRE, M. NISTORESCU, J. POZO, G. RISNOVEANU, M. SCHINDLER, A. VADINEANU, L. B. VOUGHT & E. CHAUVET. 2012. Continental-Scale Effects of Nutrient Pollution on Stream Ecosystem Functioning. *Science*, 336 (6087): 1438–1440. DOI: 10.1126/science.1219534