Effects of anthropogenic impacts on benthic macroinvertebrates assemblages in subtropical mountain streams

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ABSTRACT. The nature of the riparian and surrounding landscape has been modified by anthropogenic activities, which may subsequently alter the composition and functional structure of macroinvertebrate assemblages. The effect of these changes on function of benthic fauna is difficult to assess due to the scarce knowledge on functional structures in tropical streams. In this study we evaluate whether sites impacted and unimpacted by anthropogenic alterations differed in assemblage composition and density, richness and diversity of each functional feeding group. The selection of the sites was related to their distinct riparian characteristics, following the QBRy riparian quality index. Collector-gatherer was the dominant functional feeding group, comprising 91% of total density, whereas the proportion of shredders was very low, representing less of 0.5% of total density. Asemblage composition of macroinvertebrates differed between impacted and unimpacted sites. Predators were dominant in taxa number, representing about 60% of total taxa richness. In addition, the diversity and richness of collector-gatherers differed significantly between degraded and unimpacted sites, reflecting the sensitivity of this group to environmental changes and the utility to be used in the assessment of anthropogenic modifications. The results of this study reinforce the idea that riparian corridor management is critical for the distribution of macroinvertebrate assemblages as well as functional organization of lotic streams.

KEYWORDS. Anthropogenic changes, Argentina, lotic streams, mountain basin, riparian quality index.

RESUMEN. Efecto de los impactos antrópicos sobre los ensambles de macroinvertebrados bentónicos en ríos subtropicales de montaña. Las actividades antrópicas han producido modificaciones en el paisaje ripario y el sistema terrestre adyacente, las cuales causan alteraciones en la composición y estructura funcional de los ensambles de macroinvertebrados. Asimismo, el efecto de estos cambios sobre la estructura funcional de macroinvertebrados es difícil de evaluar debido al escaso conocimiento de este parámetro en ríos tropicales. En este estudio se evaluó si sitios degradados y no degradados por impactos antrópicos diferían en la composición de los ensambles y en la densidad, riqueza y diversidad de cada grupo funcional. La selección de los sitios se relacionó con su calidad riparia, de acuerdo con el índice QBRy. Colectores-recolectores fue el grupo funcional dominante, comprendiendo el 91% de la densidad total, mientras que la proporción de trituradores fue muy baja, representando menos del 0.5% de la densidad total. La composición de los ensambles de macroinvertebrados difirió significativamente entre sitios degradados y de buena calidad riparia. Los predadores fueron dominantes en número de taxones, representando alrededor del 60% de la riqueza de taxones. La diversidad y riqueza de collectores-recolectores fue menor en sitios degradados, reflejando la sensibilidad de este grupo frente a cambios antrópicos. Los resultados de este estudio destacan la importancia del manejo sostenible del corredor ripario debido a su estrecha relación con la distribución de los ensambles de invertebrados y su estructura funcional.

PALABRAS-CLAVE. Argentina, cambios antrópicos, cuenca de montaña, índice de calidad riparia, ríos lóticos.

Riparian vegetation influences the structure and functioning of stream macroinvertebrate communities through the provision of organic matter and by shading the stream (Hynes, 1975; Allan, 2004). Anthropogenic changes in the riparian corridor may subsequently alter the functional feeding group composition of macroinvertebrates by modifying the supply of food resources, and producing changes in habitat structure and quality (Dudgeon, 2006; Wantzen & Wagner, 2006).

Theories of macroinvertebrate function have typically been derived from detailed studies of temperate systems (Merrit & Cummins, 1996; Cummins *et al*., 2005), and consideration of tropical systems has been either excluded, or included as an interesting anomaly, despite the tropics occuping the largest land area of the world's climatic regions (Dobson *et al*., 2002). Information on factors affecting the functional structure of macroinvertebrates is not only vital for basic ecological understanding and biodiversity conservation, but also as a model for monitoring, restoring and maintaining the quality of stream ecosystems (Rosenberg & Resh, 1993; Palmer *et al*., 1997).

In the Northwestern Argentina, the ecological integrity of streams is being threatened by different anthropogenic changes. The expansion of agricultural and urban areas, overgrazing and the introduction of exotic species are some impacts that affect the riparian quality and integrity of biotic stream communities in this area (Fernández *et al*., 2008; Sirombra & Mesa, 2012). Although degradation of the riparian zone has significantly increased in tropical streams in the last few decades (Kasangaki *et al*., 2008), the consequences of these changes on the structure of macroinvertebrate assemblages are scarcely known (Nessimian *et al*., 2008). According to that, we asignate macroinvertebrate taxa of subtropical streams to functional feeding groups in order to assess whether sites of constrasting riparian quality differ in density, richness and diversity of functional feeding groups and the environmental variables related to these changes. We hypothesized that sites impacted and unimpacted by anthropogenic changes differ in assemblage composition and richness and diversity of functional feeding groups.

MATERIALS AND METHODS

Study area. We studied streams included in a basin of Tucumán province, located in northwestern of Argentina (26°36'S; 65°45'W) (Fig. 1). This region is characterized by the monsoons, with an annual precipitation of 1000 mm, falling mostly during summer (80%). High water period extends from November to April, with maximum precipitation values during January (250 mm approximately). The dry period extends from April to October with minimum rainfall values (11 mm).

The Yungas Phytogeographical province extends in the northwestern of Argentina from the frontier of Bolivia (23°) to the north of Catamarca (29°), with an area of 52.000 km^2. Not only are these forests and their streams extremely important for the provision of water for irrigation, generation of electricity and human consumption throughout most of the Andean region, but they are also an extremely diverse and threatened ecosystems (Grau & Aragón, 2000; Brown *et al*., 2006). They are undergoing rapid change as a result of increasing human population with growing demands for water and food (Brown *et al*., 2006). The conversion of forests to pasture for cattle rearing is common and probably the most destructive change in the riparian zone of the Yungas streams, due to the radical modification of the vegetation, the reduced opportunities for regeneration of native species and because this activity creates the opportunity for the invasion of exotic species (Sirombra & Mesa, 2012).

Eleven sites within the studied basin were selected, taking into account their contrasting riparian conditions (QBRy index, Sirombra & Mesa, 2012): five sites of good riparian quality (QBRy≥90), and six sites of poor and bad quality of their riparian zone (QBRy≤60) (Fig. 1). Riparian forests of the first group of sites were composed by native species and had minimal anthropogenic alterations. These sites had high values of total riparian cover, cover structure and cover quality (QBRy, Sirombra & Mesa, 2012). Native shrubs such as *Verbesina suncho* (Griseb.) S.F. Blake, *Eupatorium lasiophtalmun* Griseb., and *Acacia praecox* Griseb., *Celtis iguanaea* (Jacq.) Sarg., *Juglans australis* Griseb., *Myrsine laetevirens* (Mez) Arechav., *Parapiptadenia*

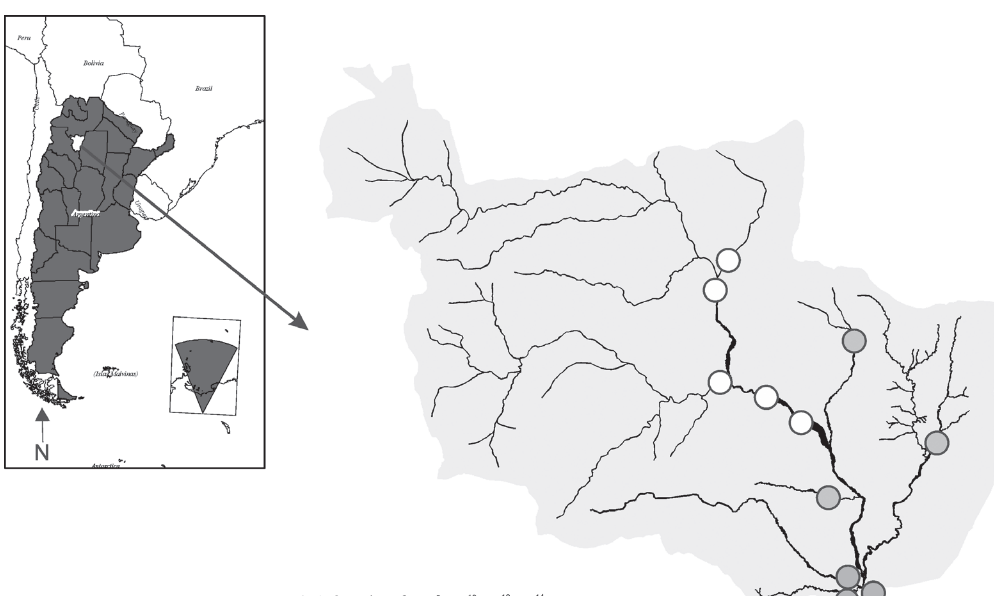

Fig. 1. Localization of the study area and distribution of reaches of good (white dots), and poor/bad quality (grey dots) of their riparian zone, Tucumán province, northwestern of Argentina.

excelsa (Griseb.) Burkart, *Cinnamomum porphyrium* (Griseb.) Kosterm, *Eugenia uniflora* L., *Zanthoxilum fagara* (L.) Sarg. native trees were common in these sites (Sirombra & Mesa, 2010). In sites of the second group, cover quality and structure of the riparian zone had been degraded by anthropogenic impacts. Exotic species such as *Acacia macracantha* Humb. & Bonpl. ex Willd., *Citrus aurantium* L., *Ligustrum lucidum* W.T. Aiton, *Gleditsia triacanthos* L., *Morus alba* L., *Pinus taeda* L., *Pyracantha angustifolia* (Franch.) C.K. Schneid and *Ricinus communis* L. were introduced as a result of cattle impact, agriculture expansion, urbanization and recreation (Sirombra & Mesa, 2010; Sirombra & Mesa, 2012).

Field methods. The selected sites were sampled during September 2005 and September 2006 (low water season). A Surber sampler (300 µm, 0.09 m^2) was used to collect the benthic samples. In the field, samples were preserved in 4% formaldehyde and packed for examination in the laboratory. Invertebrates were identified to the lowest possible taxonomic level (mostly genus) using the available keys (Domínguez & Fernández, 2009). Organic detritus were separated from benthic samples, cleaned of sediment and divided into fine (250 µm – 1 mm) and coarse (>1 mm) particulate fractions (FPOM and CPOM, respectively). CPOM was separated into wood, leafs, and 'others' (seeds, fragments of roots, fruits and flowers). All fractions were dried (100 °C for 24 h) and weighed on an electronic scale to ± 0.01 mg.

Discharge was measured each sample date with a velocity meter (Global Water Flow meter). Water samples were analyzed in the laboratory for nutrient (nitrate) and dissolved oxygen following APHA (1992). Water temperature, pH (Methrom 704) and conductivity (Methrom E587) were measured *in situ* in each sampled site and date.

Assignation of taxa to functional feeding groups. Aquatic insects have been assigned to feeding categories based on the kind of ingested food item obtained through gut content analysis, the behavioral strategy oriented to obtain resources and the nature of the morphological structure for alimentary acquisition. Gut contents of 1-5 individuals of each genus were subjected to examination of gut contents. The diet was analysed by removing the foregut and midgut content using ventral dissection of thorax (Peckarsky, 1996). The description and identification of ingested items were made under a microscope (200 X). The gut content of each specimen was mounted with glycerine in a customized slide with a central squared receptacle of uniform depth. The receptacle (20 mm x 20 mm) was divided into 625 equal-sized grid cells (0.64 mm^2), from which 15 were randomly selected. At each cell, the point interception method was employed for surveying diet composition. The scale bar of the microscope eye piece was used as a transect. Ten points were identified along the transect with a constant interval between them (Reynaga & Rueda Martin, 2010). Five categories of food resource were recognized: sediment particles, fine particulate organic matter (<1 mm), coarse particulate organic matter (>1 mm), microalgae and rests of aquatic invertebrates. The allocation of each taxon to the functional feeding group (FFG) depended mainly on the size, type and proportion of food items in gut contents (Cummins, 1973). Our own observations of feeding behavior during sampling and the observations of the mouthpart morphology in the laboratory were also helpful for FFG determination. Previous FFG allocations from literature were also consulted (Merritt & Cummins, 1996; Cummins *et al*., 2005; Tomanova *et al*., 2006; 2007; Reynaga, 2009; Reynaga & Dos Santos, 2012)

Statistical analyses. Functional structure was described in terms of density (ind.m^{-2}), species richness (using Rarefaction), diversity (Shannon diversity index) and evenness of the different FFGs.

Data of macroinvertebrates density (transformed at $\log_{10}$ (x+1)) were explored using non-metric multidimensional scaling (NMDS) with Bray-Curtis distances (R program, package Vegan, Oksanen *et al*., 2006). This ordination was performed to assess if sites impacted and unimpacted by anthopogenic changes differed in assemblage composition. We correlated the density of each taxon with ordination axes in order to determine which best accounted for separation of sites in the ordination space using the Spearman correlation. One way similarities analysis was used to compare the variability in assemblage composition among impacted and unimpacted sites using the ANOSIM statistic test at $\alpha = 0.05$ (R program, package Vegan, Oksanen *et al*., 2006).

Nonparametric Mann-Whitney test (Infostat) was used to evaluate the differences in physical and chemical variables, functional structure and density of dominant taxa of each FFG between impacted and unimpacted sites.

RESULTS

Mean values of environmental characteristics relative to impacted and unimpacted sites were shown in Table I. Dry channel width was significantly higher in unimpacted sites (Mann-Whitney test, $p<0.05$), whereas water temperature and conductivity were higher in degraded sites (Mann-Whitnet test, $p<0.05$) (Tab. I).

A total of 105 macroinvertebrate taxa were collected in this study: 56 were assigned as predators, 38 collector-gatherers, 5 scrapers, 4 shredders and 2 collector-filterers (Appendix 1). Collector-gatherers were the most abundant FFG, comprising 91% of total density, whereas predators (4%), collector filterers (4%), scrapers (0.5%) and shredders (0.5%) represent a small proportion of density (Appendix 1). Tipulidae dominated among predators (mean density = 147

Tab. I. Mean values (SD) of environmental variables relative to impacted and unimpacted sites, and results of Mann–Whitney tests used to compare these characteristics among the groups of sites, Tucumán province, northwestern of Argentina. Significant differences are showed in bold (n.s, no significant).

	Unimpacted (n=10)	Impacted (n=12)	P
Dry channel width (m)	24.6 (10.5)	13.5 (8.4)	**0.005**
Wet channel width (m)	6.0 (1.2)	6.3 (1.1)	n.s
Discharge ($m^3.s^{-1}$)	0.5 (0.4)	0.2 (0.3)	n.s
Dissolved oxygen ($mg.l^{-1}$)	8.9 (0.2)	8.7 (0.3)	n.s
Nitrate ($mg.l^{-1}$)	2.5 (0.3)	3.0 (0.4)	n.s
Water temperature (°C)	16.7 (2.3)	19.1 (1.3)	**0.02**
Conductivity ($\mu S.cm^{-1}$)	124 (26)	461.6 (237)	**0.0003**
pH	7.1 (1.0)	7.8 (1.0)	n.s
Algae ($g.m^{-2}$)	3.5 (6.3)	4.3 (5.1)	n.s
FPOM ($g.m^{-2}$)	2.3 (1.7)	2.9 (4.6)	n.s
Leafs ($g.m^{-2}$)	0.9 (1.1)	1.5 (2.6)	n.s
Wood ($g.m^{-2}$)	1.2 (3.5)	0.2 (0.7)	n.s
Others ($g.m^{-2}$)	0.8 (0.7)	5.7 (16.5)	n.s

Tab. II. Mean values (SD) of density ($ind.m^2$), richness (Rarefaction), diversity (Shannon-Wiener index) and evenness of each functional feeding group in sites impacted and unimpacted by anthropogenic alterations, Tucumán province, northwestern of Argentina and results of Mann–Whitney test used to compare these parameters among the two groups of sites. Significant differences are showed in bold (n.s, no significant).

	Impacted (n=12)	Unimpacted (n=10)	P
FFG density			
Predators	30.8 (18.3)	36.5 (20.7)	n.s
Collector-gatherers	790.3(694)	305.7 (161.9)	n.s
Scrapers	10.4 (9.1)	33.1 (40.3)	n.s
Shredders	2.6 (4.9)	0.5 (0.7)	n.s
Collector-filterers	730.2 (1475)	296 (236.8)	n.s
FFG richness			
Predators	6.64 (1.5)	6.4 (1.3)	n.s
Collector-gatherers	11.16 (2.7)	15.06 (2.14)	**0.001**
Scrapers	0.25 (0.62)	0.58 (0.93)	n.s
Shredders	0.2 (0.4)	0.3 (0.2)	n.s
Collector-filterers	0.1 (0.2)	0.15 (0.3)	n.s
FFG diversity			
Predators	0.86 (0.19)	0.83 (0.1)	n.s
Collector-gatherers	0.65 (0.16)	0.84 (0.12)	**0.007**
Scrapers	0.2 (0.15)	0.23 (0.2)	n.s
Shredders	0.15 (0.3)	0.17 (0.4)	n.s
Collector-filterers	0.18 (0.4)	0.18 (0.3)	n.s
FFG evenness			
Predators	0.68 (0.13)	0.66 (0.09)	n.s
Collector-gatherers	0.49 (0.12)	0.62 (0.1)	**0.01**
Scrapers	0.3 (0.01)	0.35 (0.14)	n.s
Shredders	0.24 (0.2)	0.24 (0.15)	n.s
Collector-filterers	0.15 (0.0)	0.16 (0.3)	n.s
Dominant taxa			
Tipulidae	9.6 (16.4)	313.3 (297.5)	**0.0002**
Oligochaeta	14741.3 (17278)	313.9 (456)	**0.004**
Helicopsyche sp.	20.6 (37.8)	103.9 (171)	n.s
Phanocerus sp.	7.9 (17.4)	0.9 (2)	n.s
Simulium sp.	1459.9 (2951)	592 (473.7)	n.s

$ind.m^{-2}$), Oligochaeta was highly abundant between collector-gatherers (8183 $ind.m^{-2}$), *Helicopsyche* sp. (58 $ind.m^{-2}$) among scrapers, *Phanocerus* sp. (4 $ind.m^{-2}$) among shredders, whereas *Simulium* sp. (1065 $ind.m^{-2}$) was the most abundant collector-filterer (Tab. II).

The NMDS ordination converged in a stable, two-dimensional solution (stress=22.39, final instability=0.0004) (Fig. 2). Sites were differentiated according with land use, showing a significant influence of this variable on assemblage composition. *Corticacarus* sp., *Atractides* sp., *Stygalbiella tucumanensis* (Cook, 1980), *Hydrodroma* sp. acarina, *Petrophila* sp. lepidopteran, *Oecetis* sp., *Metrichia* sp. trichopteran, Ceratopogonidae, Orthocladiinae, Tanypodinae, Chironominae dipteran, Oligochaeta, Lutrochidae, *Phanocerus* sp. and *Heterelmis* sp. (larvae) coleopteran were positively related with axis 1 (R>0.40, $p<0.05$). In addition, algae, fine particulate organic matter, leaf biomass, nitrate, water temperature, conductivity and pH were significantly positive related with axis 1 (R>0.40, $p<0.05$), whereas bankfull width and discharge were negatively related with this axis (R>-0.33, $p<0.05$). ANOSIM revealed that the assemblage composition differed significantly among impacted and unimpacted sites (R=0.25; $p=0.01$) reinforcing the result of the NMDS analysis.

Total abundance, richness, diversity and evenness of each FFG were similar between sites, except for collector-gatherers. Richness, diversity and evenness of collector-gatherers were significantly higher in unimpacted than impacted sites (Mann Whitney test, $p<0.01$) (Tab. II). In addition, density of Tipulidae was significantly higher in unimpacted sites, whereas Oligochaeta was dominant in degraded sites (Tab. II).

DISCUSSION

The dominance of collector-gatherers could represent an adaptation to highly disturbed enviroments, as suggested by other tropical and subtropical studies (Callisto *et al*., 2001; Buss *et al*., 2002; Boyero *et al*., 2006; Tomanova *et al*., 2006; Uieda & Motta, 2007). In disturbed streams in terms of discharge, the supply and persistence of a particular food item is very variable. Hence the ability to exploit changing resources may potentially maintain population stability against natural fluctuations (Dobson *et al*., 2002). Organic matter is a non-limited resource in tropical streams. Lotic streams receive large quantities of particulate organic matter from riparian zone by erosion of banks and by drift from upper zones. In addition, leaf decomposition to fine detritus in warm tropical streams is fast (Mathuriau & Chauvet, 2002; Dobson *et al*., 2002) and continuous throughout the year. Hence, the occurrence of this food item in the guts contents would be mainly due to its high availability in the habitat (Henriques-Oliveira *et al*., 2003) rather than to feeding specialization. Indeed, a community dominated by taxa with high affinity

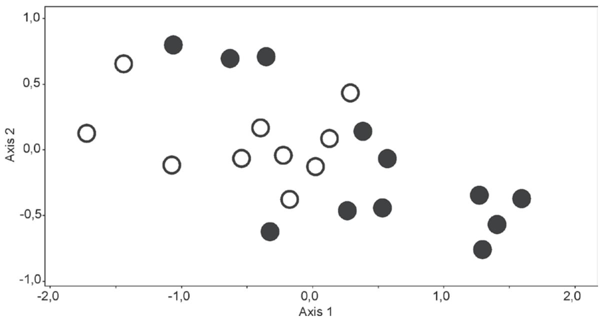

Fig. 2. Non-metric multidimensional scaling ordination based on density of benthic macroinvertebrates collected in impacted (black circles) and unimpacted sites (white circles), Tucumán province, northwestern of Argentina.

to the same food resource implies less inter-specific competition for the resource, and hence a relatively high resource availability.

Predators were highly dominant in taxa number, representing about 60% of total taxa richness. This finding coincides with other studies in tropical streams (CALLISTO *et al.*, 2001; CHESHIRE *et al.*, 2005), showing the importance of this FFG for the biodiversity of these lotic systems. On the other hand, shredders were scarce in the studied streams, comprising about less than 0.5% of total density of taxa. This pattern was in acordance with other reports relating with the scarcity of shredders in tropical streams (DOBSON *et al.*, 2002; MATHURIAU & CHAUVET, 2002; WRIGHT & COVICH, 2005; WANTZEN & WAGNER, 2006). It has been suggested that shredding may be less important in tropical systems because there are alternative decomposition pathways for leafs, such as faster microbial processing due to higher temperatures (IRONS *et al.*, 1994). In addition, the low quality of tropical leafs in terms of roughness and high tannin and low nutrient contents may also explain the low numbers of this group in comparison to temperate systems (STOUT, 1989; WANTZEN *et al.*, 2002). Shredders tend to have long life-cycles and are slow colonizers (JACOBSEN & ENCALADA, 1998), life traits that make them unsuitable for living in frequently disturbed streams.

In relation to the effects of land use, the higher value of nitrate in degraded streams could be associated with livestock and agricultural activities, factors of increasing expansion in the riparian zone of the studied streams (BROWN *et al.*, 2002; SIROMBRA & MESA, 2012). Modifications in cover structure and quality of the riparian vegetation, factors associated with the remotion and trampling of native species and the introduction of exotic vegetation, could increase the amount and intensity of light reaching the stream surface, increasing water temperature. The increase of light reaching the stream surface would increase the biomass of algae in impacted sites. In addition, a higher leaf biomass characterized degraded sites. According with our observations, whole leafs of *Citrus* and *Acacia* appeared in most samples of impacted reaches, constituting a significant proportion of the total organic matter biomass in these sites. Leafs of these exotic species have high cellulose, low nitrogen content and chemical inhibitors, factors that negatively affect microbial invasion and subsequent processing and degradation (ALONSO *et al.*, 2010). In addition, these leafs could be used mainly as substrate and indirect source of food for several macroinvertebrates, such as collectors.

This study found that sites of different riparian quality differed in the assemblage composition of macroinvertebrates, and this result was in accordance with other studies (MISERENDINO & MASI, 2010; ARNAIZ *et al.*, 2011). In addition, richness, diversity and evenness of collector-gatherers differed among undegraded and degraded sites. This result was in accordance with other works (RAWER-JOST *et al.*, 2000; STEPENUCK *et al.*, 2002; COMPIN & CÉRÉGHINO, 2007), showing the utility of the structure of functional feeding groups of macroinvertebrates as indicators of anthropogenic impacts. Native forested sites having good ecological conditions supported more diversity and richness of collector-gatherers, whereas sites degraded by anthropogenic impacts showed a significant decrease in these parameters. This could be related with that 33 of a total 40 taxa relative to this FFG corresponded to ephemeropteran, trichopteran and coleopteran species. A great proportion of these groups are composed by taxa intolerant to anthropogenic changes, such as increase of nitrate and water temperature, determining their absence and consequently, the lower richness of collector-gatherers in degraded sites. In contrast, Oligochaeta had a higher density in impacted streams, and this was related with the tolerance of this group to degraded environmental conditions (ROSENBERG & RESH, 1993).

The results of this study showed the utility of collector-gatherers as a tool in the assessment of anthropogenic changes in subtropical streams. In addition, this work reinforces the idea that riparian corridor management is critical for the distribution of macroinvertebrate assemblages as well as functional organization of lotic streams. The maintenance of good conditions of this corridor makes possible the availability of habitats necessaries for maintain a high biodiversity in stream communities.

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REFERENCES

ALLAN, J. D. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. **Annual Review of Ecology, Evolution, and Systematics 35**:257-284.

ALONSO, A.; GONZÁLEZ-MUÑOZ, N. & CASTRO-DÍEZ, P. 2010. Comparison of leaf decomposition and macroinvertebrate colonization between exotic and native trees in a freshwater ecosystem. **Ecological Research 25**(3):647-653.

APHA. 1992. **Standard Methods for the Examination of Water and Wastewater**. 18th edition. American Public Health Association, Washington, D.C. USA. 70 p.

Arnaiz, O. L.; Wilson, A. L.; Watts, R. J. & Stevens, M. M. 2011. Influence of riparian condition on aquatic macroinvertebrate communities in an agricultural catchment in south-eastern Australia. **Ecological Research 26**(1):123-131.

Boyero, L.; Pearson, R. G. & Camacho, R. 2006. Leaf breakdown in tropical streams: the role of different species in ecosystem functioning. **Archiv für Hydrobiologie 166**(14):453-466.

Brown, A. D.; Grau, A.; Lomáscolo, T. & Gasparri, N. I. 2002. Una estrategia de conservación para las selvas subtropicales de montana (Yungas) de Argentina. **Ecotropicos 15**(2):147-159.

Brown, A. D.; Malizia, L. & Lomáscolo, T. 2006. **Reserva de la Biosfera de las Yungas: armando el rompecabezas entre todos**. Secretaria Programa sobre El Hombre y la Biosfera Artículo para el Libro sobre Reservas de la Biosfera de países que integran la Red Iberomab. 15p.

Buss, D. F.; Baptista, D. F.; Silveira, M. P.; Nessimian, J. L. & Dorville, L. F. M. 2002. Influence of water chemistry and environmental degradation on macroinvertebrate assemblages in a river basin in south-east Brazil. **Hydrobiologia 481**(1-3):125-136.

Callisto, M.; Moreno, C. E. & Barbosa, F. A. R. 2001. Habitat diversity and benthic functional trophic groups at Serra do Cipó, southeast Brazil. **Revista Brasileira de Biologia 61**(2):259-266.

Cheshire, K.; Boyero, L.; & Pearson, R. G. 2005. Food webs in tropical Australian streams: shredders are not scarce. **Freshwater Biology 50**:748-769.

Compin, A. & Céréghino, R. 2007. Spatial patterns of macroinvertebrate functional feeding groups in streams in relation to physical variables and land-cover in Southwestern France. **Landscape Ecology 22**:1215-1225.

Cummins, K. W. 1973. Trophic relations of aquatic insects. **Annual Review of Entomology 18**:183-206.

Cummins, K. W.; Merritt, R. W. & Andrade, P. C. N. 2005. The use of invertebrate functional groups to characterize ecosystem attributes in selected streams and rivers in south Brazil. **Studies on Neotropical Fauna and Environment 40**(1):69-89.

Dobson, M.; Mathooko, J. M.; Magana, A. & Ndegwa, F. K. 2002. Macroinvertebrate assemblages and detritus processing in Kenyan highland streams: more evidence for the paucity of shredders in the tropics? **Freshwater Biology 47**(5):909-919.

Domínguez, E. & Fernández, H. R. 2009. **Macroinvertebrados bentónicos Sudamericanos: Sistemática y biología**. Fundación Miguel Lillo, Tucumán. 654 p.

Dudgeon, D. 2006. The impacts of human disturbance on stream benthic invertebrates and their drift in North Sulawesi, Indonesia. **Freshwater Biology 51**:1710-1729.

Fernández, H. R.; Romero, F. & Dominguez, E. 2008. Intermountain basins use in subtropical regions and their influences on benthic fauna. **River Research and Applications 25**(2):181-193.

Grau, H. R. & Aragón, R. 2000. **Ecología de árboles exóticos en las Yungas Argentinas**. Laboratorio de Investigaciones Ecológicas de las Yungas, Universidad Nacional de Tucumán, Tucumán. p. 5-20.

Henriques-Oliveira, A. L.; Nessimian, J. L. & Dorvillé, L. F. M. 2003. Feeding habits of Chironomid larvae (Insecta: Diptera) from a stream in the Floresta da Tijuca, Rio de Janeiro, Brazil. **Brazilian Journal of Biology 63**(2):269-281.

Hynes, H. B. N. 1975. The stream and its valley. **Verhenlungen der Internationale Vereinigung für Theoretische und Angewandte Limnologie 19**:1-15.

Irons, J. G.; Oswood, M. W.; Stout, R. J. & Pringle, C. M. 1994. Latitudinal patterns in leaf litter breakdown: is temperature really important? **Freshwater Biology 32**(2):401-411.

Jacobsen, D. & Encalada, A. 1998. The macroinvertebrate fauna of Ecuadorian highland streams in the wet and dry season. **Archiv für Hydrobiology 142**(1):53-70.

Kasangaki, A.; Chapman, L. J. & Balirwa, J. 2008. Land use and the ecology of benthic macroinvertebrate assemblages of high-altitude rainforest streams in Uganda. **Freshwater Biology 53**(4):681-697.

Mathuriau, C. & Chauvet, E. 2002. Breakdown of leaf litter in a neotropical stream. **Journal of the North American Benthological Society 21**(3):384-396.

Merritt, R. W. & Cummins, K. W. 1996. **An introduction to the aquatic insects of North America**. Kendall-Hunt, Dubuque.

Miserendino, M. L. & Masi, C. I. 2010. The effects of land use on environmental features and functional organization of macroinvertebrate communities in Patagonian low order streams **Ecological Indicators 10**(2):311-319.

Nessimian, J. L.; Venticinque, E. M.; Zuanon, J.; De Marco Jr., P.; Gordo, M.; Fidelis, L.; D'arc Batista, J. & Juen, L. 2008. Land use, habitat integrity, and aquatic insect assemblages in Central Amazonian streams. **Hydrobiologia 614**:117-131.

Oksanen, J.; Kindt, R.; Legendre, P. & O'Hara, R. B. 2006: **Vegan: Community Ecology Package version 1.17– 4**. Available at <http:// cran.r project.org/web/ packages/vegan/index.html.>. Accessed on August 2013.

Palmer, M. A.; Ambrose, R. F. & Poff, N. L. 1997. Ecological theory and community restoration ecology. **Restoration Ecology 5**(4):291-300.

Peckarsky, B. L. 1996. Predator–prey interactions. *In*: Hauer, F. R. & Lamberti, G. A. eds. **Methods in Stream Ecology**. San Diego, Academic Press, p.431-451.

Rawer-Jost, C.; Böhmer, J.; Blank, J.; Rahmann, H. 2000. Macroinvertebrate functional feeding group methods in ecological assessment. **Hydrobiologia 422/423**:225-232.

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

Reynaga, M. C. & Rueda Martin, P. 2010. Trophic analysis of two species of Atopsyche (Trichoptera: Hydrobiosidae). **Limnologica 40**:61-66.

Reynaga, M. C. & Dos Santos, D. A. 2012. Rasgos biológicos de macroinvertebrados de ríos subtropicales: patrones de variación a lo largo de gradientes ambientales espacio-temporales. **Ecología Austral 22**:112-120.

Rosenberg, D. M. & Resh, V. H. 1993. **Freshwater Biomonitoring and Benthic Macroinvertebrates**. New York, Chapman and Hall. 488p.

Sirombra, M. G. & Mesa, L. M. 2010. Composición florística y distribución de los bosques ribereños subtropicales andinos del Río Lules, Tucumán, Argentina. **Revista de Biologia Tropical 58**(1):499-510.

_____. 2012. A method for assessing the ecological quality of riparian forests in subtropical Andean streams: QBRy index. **Ecological Indicators 20**:324-331.

Stepenuck, K. F.; Crunkilton, R. L. & Wang, L. 2002. Impacts of urban landuse on macroinvertebrate communities in southeastern Wisconsin streams. **Journal of the American Water Resources Association 38**(4):1041-1051.

Stout, R. J. 1989. Effects of condensed tannins on leaf processing in mid-latitude and tropical streams: a theoretical approach. **Canadian Journal of Fisheries and Aquatic Sciences 46**(7):1097-1106.

Tomanova, S.; Goitia, E. & Helesic, J. 2006. Trophic levels and functional feeding groups of macroinvertebrates in neotropical streams. **Hydrobiologia 556**(1):251-264

Tomanova, S.; Tedesco, P. A.; Campero, M.; Van Damme, P.; Moya, N. & Oberdorff, T. 2007. Longitudinal and altitudinal changes of macroinvertebrate functional feeding groups in neotropical streams: a test of the River Continuum Concept. Fundamental and Applied Limnology. **Archiv für Hydrobiology 170**(3):233-241.

Uieda, A. V. S. & Motta, R. L. 2007. Trophic organization and food web structure of southeastern Brazilian streams: a review. **Acta Limnologica Brasiliensis 19**(1):15-30.

Wantzen, K. M. & Wagner, R. 2006. Detritus processing by invertebrate shredders: a neotropical – temperate comparison. **Journal of the North American Benthological Society 25**(1):216-232.

Wantzen, K. M.; Wagner, R.; Suetfel, R. & Junk, W. J. 2002. How do plant-herbivore interactions of trees influence coarse detritus processing by shredders in aquatic ecosystems of different latitudes? **Verhandlungen Internationale Vereiningung für Theoretische und Limnology 28**:815-821.

Wright, M. S. & Covich, A. P. 2005. The effect of macroinvertebrate exclusion on leaf breakdown rates in a tropical headwater stream. **Biotropica 37**(3):403-408.

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Appendix 1. Mean density of taxa (SD) of each functional feeding group (FFG) in impacted and unimpacted sites, Tucumán province, northwestern of Argentina (P, predator; CG, collector-gatherer; SCR, scraper; SHR, shredder).

Taxa	FFG	Impacted	Unimpacted
Aegla sp.	P	5.9 (1.4)	0
Anacroneuria sp.	P	53.1 (8.3)	99.6 (7.3)
Corydalus sp.	P	2.5 (0.3)	6.7 (0.3)
Petrophila sp.	P	0.8 (0.2)	0.00
Pyralidae	P	41.0 (5.1)	6.5 (0.6)
Odonata	P	9.9 (1.0)	0.00
Atopsyche spp.	P	2.8 (0.5)	44.8 (3.9)
Atopsyche maxi	P	0.31 (0.1)	0.0
Atopsyche spinosa	P	0.0	14.2 (1.1)
Oecetis sp.	P	3.5 (0.8)	1.5 (0.2)
Polycentropus joergenseni	P	2.5 (0.8)	0.4 (0.1)
Smicridea sp.	P	290.6 (28.1)	196.3 (21.3)
Atractides sp.	P	48.8 (3.4)	4.1 (0.6)
Atractides sinuatipes	P	0.9 (0.3)	0.0
Atractidella sp.	P	0.3 (0.1)	0.0
Aturus sp.	P	4.0 (1.0)	0.0
Clathrosperchon punctatus	P	4.0 (0.4)	8.1 (0.9)
Corticacarus sp.	P	37.5 (3.4)	11.3 (1.3)
Diamphidaxona yungasa	P	0.9 (0.3)	0.0
Dodecabates dodecaporus	P	75.6 (8.2)	2.6 (0.3)
Hydrodroma sp.	P	6.8 (1.5)	1.1 (0.3)
Hygrobates sp.	P	17.1 (3.2)	2.6 (0.3)
Hygrobates plebejus	P	0.6 (0.1)	0.4 (0.1)
Hygrobatella multiacetabulata	P	0.6 (0.2)	0.7 (0.2)
Meramecia sp.	P	0.00	0.4 (0.1)
Miraxonides sp.	P	0.6 (0.2)	0.00
Neomamersa sp.	P	0.3 (0.1)	0.00
Protolimnesia setifera	P	23.7 (5.1)	27.0 (3.6)
Protolimnesia interstitialis	P	0.6 (0.1)	0.4 (0.1)
Rhycholimnochares expansiseta	P	6.2 (1.0)	13.5 (1.5)
Sperchon neotropicus	P	0.00	0.4 (0.1)
Stygalbiella tucumanensis	P	3.4 (0.4)	1.1 (0.2)
Tetrahygrobatella sp.	P	1.1 (0.2)	12.9 (1.5)
Tetrahygrobatella argentinensis	P	0.6 (0.1)	2.2 (0.4)
Tetrahygrobatella bovala	P	0.3 (0.1)	0.7 (0.1)
Torrenticola columbiana	P	93.7 (7.4)	153.5 (22.6)
Protolimnesella sp.	P	0.00	0.7 (0.2)
Protolimnesia sp.	P	0.00	0.4 (0.1)
Corixidae sp.	P	0.3 (0.1)	0.00
Darwinivelia sp.	P	0.6 (0.2)	2.6 (0.6)
Guerridae	P	0.00	0.4 (0.1)
Hebrus sp.	P	0.6 (0.1)	1.5 (0.3)
Heterocorixa sp.	P	2.2 (0.5)	0.00
Horvatinia sp.	P	0.3 (0.1)	0.4 (0.1)
Ligomorphus sp.	P	8.0 (1.7)	1.1 (0.2)
Mesovelia sp.	P	1.8 (0.3)	6.3 (1.8)
Mesoveloidea	P	0.6 (0.2)	0.00
Microvelia sp.	P	0.00	2.6 (0.7)
Rhagovelia sp.	P	4.6 (1.1)	0.4 (0.1)
Ceratopogonidae	P	123.4 (17)	56.5 (5.7)
Dolychopodidae	P	0.00	0.4 (0.1)
Empididae	P	25.8 (2.7)	76.1 (5.6)
Tipulidae	P	9.6 (1.5)	313.3 (26)
Dytiscidae (larvae)	P	3.1 (0.3)	1.1 (0.3)
Dytiscidae (adult)	P	0.9 (0.1)	0.4 (0.1)
Hydrophilidae	P	2.5 (0.5)	7.9 (1.8)
Americabaetis alphus	CG	138.3 (25.7)	349.4 (23)

Appendix 1. (Continue).

Taxa	FFG	Impacted	Unimpacted
Baetodes sp.	CG	54.9 (9.3)	123.5 (17)
Baetodes huaico	CG	1521.4 (221)	1264.4 (88.5)
Caenis sp.	CG	259.1 (45.3)	0.7 (0.2)
Camelobaetidius penai	CG	10.4 (1.1)	474.1 (31)
Farrodes sp.	CG	2.2 (0.4)	0.4 (0.1)
Haplohyphes sp.	CG	5.2 (0.9)	19.1 (1.8)
Leptohyphes eximius	CG	390.7 (58)	55 (4.3)
Nanomis galera	CG	215.4 (33.7)	147.2 (13)
Thaulodes sp.	CG	119.3 (11)	168.7 (18.7)
Thraulodes consortis	CG	0.31 (0.1)	12.9 (2.6)
Thraulodes cochunaensis	CG	1.2 (0.3)	23.1 (6.4)
Tricorythodes popayanicus	CG	195.2 (39)	0.7 (0.1)
Austrelmis sp. (larvae)	CG	1022.8 (154)	2138.9 (263)
Austrelmis sp. (adult)	CG	18.8 (2.8)	76.3 (5.6)
Cylloepus sp. (adult)	CG	0.3 (0.1)	2.4 (0.4)
Ephydridae (adult)	CG	0.9 (0.3)	0.7 (0.2)
Hydrophilidae (adult)	CG	0.3 (0.1)	0.00
Lutrochidae	CG	4.3 (0.8)	0.00
Macrelmis sp. (larvae)	CG	39.7 (5.5)	134.2 (25)
Macrelmis sp. (adult)	CG	2.8 (0.6)	4.1 (0.7)
Neoelmis sp. (adult)	CG	6.2 (0.9)	43.5 (5.9)
Staphylinidae (larvae)	CG	38.3 (9.7)	120.9 (25)
Staphylinidae (adult)	CG	54.0 (13.3)	66.8 (13.4)
Stratyomyidae	CG	2.2 (0.3)	0.7 (0.1)
Hydroptila sp.	CG	48.8 (5.6)	0.5 (0.2)
Marilia sp. (larvae)	CG	0.9 (0.3)	0.00
Metrichia sp. (larvae)	CG	506 (40)	45.2 (4.5)
Mortoniella sp.	CG	117.1 (25)	113.7 (15)
Nectopsyche sp.	CG	0.31 (0.1)	2.0 (0.3)
Oxyethira sp.	CG	8.6 (1.5)	1.1 (0.3)
Orthocladiinae	CG	7769 (465)	4570.5 (330)
Tanypodinae	CG	758 (141)	235 (19)
Chironominae	CG	3547.5 (423)	1216.8 (70.5)
Blephaceridae	CG	0.00	2.2 (0.3)
Maruina sp.	CG	5.5 (1.0)	98.9 (23.9)
Psychodidae	CG	2.5 (0.4)	399.4 (54)
Oligochaeta	CG	14741.3 (1555)	313.9 (41)
Dryops sp. (adult)	SCR	0.00	1.1 (0.2)
Helicopsyche sp.	SCR	20.7 (3.4)	103.9 (15.4)
Mexitrichia sp.	SCR	0.00	1.1 (0.3)
Neotrichia sp.	SCR	0.00	0.4 (0.1)
Psephenus sp.	SCR	41.8 (4.6)	91.8 (11.3)
Phanocerus sp.	SHR	7.9 (1.6)	0.9 (0.2)
Chrysomelidae (adult)	SHR	0.00	0.7 (0.2)
Haliplidae (adult)	SHR	0.00	0.4 (0.1)
Heterelmis sp. (larvae)	SHR	2.6 (0.4)	0.00
Chimarra sp.	CF	0.6 (0.2)	0.00
Simulium sp.	CF	1459.9 (265)	592 (42)