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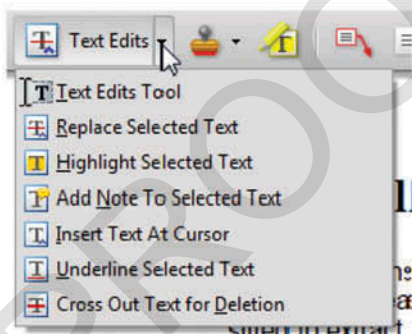
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Influence of riparian quality on macroinvertebrate assemblages in subtropical mountain streams

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The type and extent of the riparian vegetation is known to have a significant influence on macroinvertebrate communities. The objective of this work was to assess the influence of the quality of a subtropical riparian forest on the structure and composition of macroinvertebrate assemblages. Lower diversity and richness of invertebrates has been found in degraded reaches in comparison with native forested sites. Richness, diversity and density of collector-gatherers are sensitive to the quality of the riparian vegetation. Native riparian species at sites of good riparian quality contribute to greater habitat complexity and higher quantities of palatable food, determining the differences in the structure of macroinvertebrate assemblages between sites. This work suggests the use of the IBY-4 index in future management studies of Yungas streams because of its sensitivity to changes in riparian conditions.

Keywords: biotic index; ecological condition; macroinvertebrate diversity; northwestern Argentina; subtropical forest

Introduction

The structure and functioning of stream macroinvertebrate communities are intimately linked with riparian vegetation and land use (Hynes 1975; Allan 2004). The type and extent of riparian vegetation regulates the supply of food resources, can affect adult population dynamics and determine habitat structure and quality (water temperature, light levels, channel form, stream hydrology). It is not surprising, therefore, that anthropogenic impacts on the riparian zone have had a dramatic effect on macroinvertebrate structure and composition (Jansen and Robertson 2001; Townsend et al. 2004), producing declines in macroinvertebrate diversity and a shift in relative abundance among functional feeding groups (Dudgeon 2006; Wantzen and Wagner 2006).

Tropical forests are increasingly threatened by accelerating rates of forest conversion and degradation (Kasangaki et al. 2008; Lorion and Kennedy 2009). In northwestern Argentina, subtropical forests of the Yungas phytogeographical province have been significantly degraded by years of land-use practices (Brown et al. 2002, 2006). Pasture conversion, agriculture, urbanization, exotic plantations and deforestation are some threats that affect the ecological integrity of this zone (Grau and Aragón 2000; Brown et al. 2006; Sirombra and Mesa 2012). Unrestricted stream access of livestock has degraded native riparian vegetation, reducing

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vegetation cover, increasing bare ground, depleting highly palatable or disturbance-sensitive species and allowing the introduction of exotic species (Grau and Aragón 2000; Grau et al. 2008; Sirombra and Mesa 2010). To assess the effect of these anthropogenic changes and gain knowledge for the management of the riparian zone of Yungas streams, an adaptation of the original QBR (*qualitat del bosc de ribera*, Munné et al. 2003) had been proposed (QBRy index, Sirombra and Mesa 2012). The QBRy index represents a useful tool to identify sites where riparian vegetation is severely impaired or pristine, and those where conservation effort should be directed (Sirombra and 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). Some studies have demonstrated changes in macroinvertebrate composition as a result of changes in riparian vegetation (Kasangaki et al. 2008; Arnaiz et al. 2011), and modifications in the riparian zone produced by cattle (Herbst et al. 2012), but studies in subtropical streams are scarce (Lorion and Kennedy 2009). Consequently, the aim of this work was to assess the influence of the quality of the riparian forest at the reach level on macroinvertebrate assemblages of subtropical streams of northwestern Argentina. The hypothesis of this work was that richness and diversity are greater at reaches with good riparian quality, and that sites of higher riparian quality have higher-value biotic indices based on macroinvertebrates.

Material and methods

Study area

This study was conducted in the Lules river basin, and Andean basin located in Tucumán province in northwestern Argentina (Figure 1). The maximum altitude of this basin is around 4488 m above sea level (a.s.l.), decreasing toward the piedmont (408 m a.s.l.) (Figure 1). The region is characterized by a humid climate, with annual rainfall exceeding 1500 mm, concentrated mostly in the summer period (November–March).

The Yungas phytogeographical province extends through Argentina between 22° and 28° S with an area of 3,900,000 hectares. This phytogeographical province represents less than 2% of Argentina, but includes 50% of its biodiversity (Brown et al. 2006). Some common native riparian species included in the Lules river basin are *Solanum riparium*, *Tipuana tipu*, *Carica quercifolia*, *Celtis iguanaea*, *Juglans australis*, *Pisoniella arborescens*, *Baccharis salicifolia*, *Cinnamomum porphyrium*, *Eugenia uniflora* trees and *Cestrum parqui*, *Baccharis salicifolia* and *Phenax laevigatus* shrubs (Sirombra and Mesa 2010). Fifteen exotic species occurred in the riparian zone of degraded sites, seven of which are extremely invasive (*Morus alba*, *Gleditsia triacanthos*, *Ligustrum lucidum*, *Psidium guayaba*, *Acacia macracantha* trees, *Pyracantha angustifolia* shrubs and *Arundo donax* reed), representing a real threat for native riparian species (Sirombra and Mesa 2010, 2012).

The QBRy index has proved to be sensitive to changes in the composition of riparian vegetation and changes in stream channel due to anthropogenic alterations,

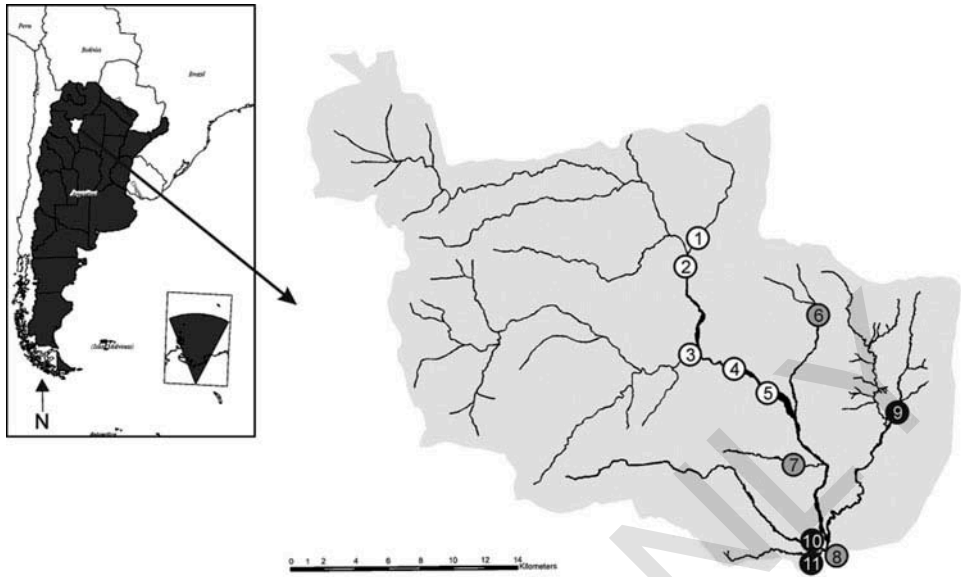


Figure 1. Localization of the study area and distribution of reaches of good (white dots, 1–5), poor (grey dots, 6–8) and bad (black dots, 9–11) quality of their riparian zone.

decreasing the values of each section in accordance with these factors (Sirombra and Mesa 2012). This index includes four sections: total riparian cover, cover structure, cover quality and channel alteration. Sites of good riparian quality (QBRy = 75–90) have good vegetation cover, good cover structure, high cover quality composed only by native species and minimum alterations in their channels. Riparian forest at sites of poor quality (QBRy = 30–50) and bad quality (QBRy ≤ 25) has been degraded by anthropogenic impacts, reducing vegetation cover, decreasing the quality of the structure and cover of the riparian vegetation by introduction of exotic species, and showing significant modifications in their channels (Sirombra and Mesa 2012).

Sampling design

Eleven sampling reaches of the Lules River basin were selected in accordance with their different riparian conditions: five sites of good quality (1–5, QBRy = 90), three poor (6–8, QBRy ≤ 50) and three of bad quality (9–11; QBRy < 30) of their riparian zone in accordance with the values of the QBRy index (Sirombra and Mesa 2012) (Figure 1). The stream order of the studied streams varied from five to six, and the altitudinal difference between reaches of different riparian conditions was ≤ 400. In addition, physical characteristics were similar between streams (Table 1).

The selected sites were sampled during September 2005 and September 2006 in spring (low-water season). Three Surber samples (mesh size 300 μm, 0.09-m² area) were taken in each riffle to collect the benthic fauna and particulate organic matter. In the field, samples were preserved immediately in 4% formaldehyde

Table 1. Altitude, and mean (\pm SD) of environmental variables relative to reaches of good, poor and bad riparian quality.

| Variables | Good | Poor | Bad |
|--|-------------|------------|------------|
| Altitude (m) | 1145 (160) | 898 (170) | 740 (92) |
| Depth (m) | 0.3 (0.1) | 0.3 (0.1) | 0.2 (0.1) |
| Discharge ($\text{m}^3 \cdot \text{s}^{-1}$) | 0.53 (0.45) | 0.31 (0.4) | 0.11 (0.1) |
| Wet channel width (m) | 6.9 (4.5) | 6.1 (5.0) | 6.0 (2.6) |
| Dry channel width (m)* | 24.6 (10.1) | 17.5 (9.4) | 9.6 (5.4) |
| Bank-full height (m) | 1.4 (0.3) | 1.7 (0.3) | 2.0 (0.6) |
| Nitrate ($\text{mg} \cdot \text{l}^{-1}$) | 2.5 (0.3) | 2.8 (0.5) | 3.2 (0.9) |
| Conductivity ($\mu\text{S} \cdot \text{cm}^{-1}$)*** | 124 (26) | 321 (174) | 601 (252) |
| pH | 7.1 (0.9) | 7.7 (1.0) | 7.9 (1.1) |
| Dissolved oxygen ($\text{mg} \cdot \text{l}^{-1}$) | 8.9 (0.2) | 8.7 (0.4) | 8.7 (0.3) |
| Water temperature ($^{\circ}\text{C}$)* | 17.2 (1.9) | 18.8 (0.7) | 19.3 (1.7) |
| Algae ($\text{g} \cdot \text{m}^{-2}$) | 0.3 (0.5) | 0.4 (0.6) | 0.4 (0.3) |
| FPOM ($\text{g} \cdot \text{m}^{-2}$) | 0.2 (0.2) | 0.2 (0.1) | 0.3 (0.6) |
| CPOM ($\text{g} \cdot \text{m}^{-2}$) | 0.3 (0.3) | 0.2 (0.2) | 0.8 (1.7) |

Note: * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$.

CPOM, coarse particulate organic matter; FPOM, fine particulate organic matter.

and packed for examination in the laboratory. Invertebrates were identified to the lowest possible taxonomic level (mostly genus) using the available keys (Fernández and Domínguez 2001; Domínguez and Fernández 2009).

Water chemistry and physical measures were assessed on each sampling date at each site. These measures included temperature, pH, oxygen concentration, conductivity, NO_3 , PO_4 and benthic organic matter. Water samples were placed on ice and taken to the laboratory for immediate analyses using standard methods (APHA 1992). Current velocities were measured at a cross-section using a flow meter (Global Water Flow meter). Organic matter was separated from benthic samples, cleaned of sediment and divided into fine (300 μm to 1 mm) and coarse (> 1 mm) particulate fractions [fine (FPOM) and coarse (CPOM) particulate organic matter, respectively]. CPOM was separated into wood, leaves and “others” (seeds, fragments of roots, fruits and flowers). All fractions were dried (100°C for 24 h) and weighed on an electronic balance to ± 0.01 mg.

Data analyses

Macroinvertebrates were assigned to functional feeding groups according to Reynaga (2009), Reynaga and Rueda Martín (2010), Reynaga and Dos Santos (2012). Environmental variables, assemblage structure measures and relative density (%) of each functional feeding group were compared among reaches of different riparian quality (good, poor, bad) with simple analysis of variance (ANOVA). Assemblage structure measures included taxon richness, Shannon diversity and total density (individuals $\cdot \text{m}^{-2}$). The values of taxon richness were first corrected using the rarefaction method (Krebs 1989).

AQ1 The sum of the tolerance scores of all families recognized in a sample adapted
 for the region [biological monitoring working party (BMWP) index, Armitage
 et al. 1983 modified by Dominguez and Fernández 1998] and the number of
 species/morpho-species included in Ephemeroptera, Plecoptera and Trichoptera
 orders (EPT index, Klemm et al. 1990) are widely used indices in the bioassess- 135
 ment of Yungas streams (Fernández et al. 2002; Von Ellenrieder 2007; Dos
 Santos et al. 2011). Recent work (Dos Santos et al. 2011) proposed a new index
 AQ2 for Yungas streams based on the occurrence of Elmidae, Plecoptera, Trichoptera
 and Megaloptera (IBY-4 index, Dos Santos et al. 2011). These indices were
 selected to evaluate their sensitivity to different riparian conditions of the studied 140
 reaches.

Analysis of similarities (ANOSIM, R program, package Vegan) was used to
 determine differences in macroinvertebrate assemblages between reaches of differ- 145
 ent riparian quality. Redundancy analysis (RDA) was used to assess the relation-
 ship between the riparian quality, the environmental characteristics and
 macroinvertebrate data. This analysis was performed using R program, package
 Vegan (Oksanen et al. 2006). Before running the RDA, taxon densities were
 Hellinger-transformed, as suggested by Legendre and Gallagher (2001). The
 Hellinger transformation preserves the Euclidean distance among the rows, and 150
 therefore allows the use of Euclidean-based ordination methods. It also offers the
 advantage of under-weighting the rare taxa (Legendre and Gallagher 2001). The
 environmental variables were standardized (Legendre and Legendre 1998, R pro-
 gram, package *pls*).

Results

A total of 112 taxa was identified in this survey. Sites of good riparian quality were 155
 dominated by Orthocladiinae (32%), followed by *Austrelmis* sp. (larvae) (15%),
 whereas Oligochaeta (> 39%) followed by Orthocladiinae (> 20%) were common in
 sites of bad and poor quality (Appendix 1).

Among the environmental variables measured, water temperature and conduc- 160
 tivity differed significantly among reach types, being higher in sites of bad riparian
 quality (ANOVA, $P < 0.05$, Table 1). In addition, dry channel width was sig-
 nificantly higher in sites of good riparian quality (ANOVA, $P < 0.05$, Table 1).
 No significant differences were observed between types of particulate organic
 matter among reach types (Table 1). In addition, significant differences in taxa
 richness and diversity of macroinvertebrates were found among reach types, being 165
 higher in sites of good riparian quality and lower in those of poor quality
 (Table 2).

Among the biotic indices analyzed, only IBY-4 differed significantly among
 reaches of different riparian quality, being significantly higher in sites of good
 riparian quality (Table 2). Percentage density of scrapers and predators, richness 170
 and diversity of collector-gatherers were significantly higher in reaches of good
 riparian quality, whereas the density of collector-gatherers increased in sites of bad
 quality (Table 2).

ANOSIM revealed that assemblage composition differed significantly among
 reaches of different riparian quality ($R = 0.23$, $P < 0.006$). This pattern was also 175
 shown by RDA, determining the separation of sites in the ordination space in

Table 2. Mean (\pm SD) of assemblage metrics, values of ecological indexes, and relative density (%), richness and diversity of each functional feeding group of reaches of good, poor and bad riparian condition.

| Parameters | Good | Poor | Bad | <i>P</i> |
|--|---------------|-----------------|-----------------|---------------|
| Taxa richness | 23.5 (2.4) | 15.5 (2.4) | 18.2 (4.7) | 0.0003 |
| Shannon diversity | 1.0 (0.1) | 0.7 (0.1) | 0.8 (0.2) | 0.004 |
| Total density (individuals.m ⁻²) | 14,115 (7310) | 34,699 (30,732) | 33,444 (32,701) | n.s. |
| Ecological condition | | | | |
| IBY-4 | 4.0 (0.3) | 3.5 (0.5) | 3.0 (0.9) | 0.01 |
| EPT | 17.3 (2.6) | 14.8 (3.5) | 15.5 (2.1) | n.s. |
| BMWP | 210 (16.1) | 197.5 (38.8) | 197.7 (33.4) | n.s. |
| Functional feeding groups | | | | |
| Collector-gatherers (%) | 85.9 (2.5) | 90.0 (8.8) | 93.4 (3.7) | 0.01 |
| Collector-filterers (%) | 4.3 (3.8) | 3.0 (4.4) | 2.4 (2.5) | n.s. |
| Scrapers (%) | 1.5 (1.2) | 0.4 (0.6) | 0.2 (0.1) | 0.01 |
| Predators (%) | 8.3 (2.5) | 6.6 (9.7) | 3.9 (2.2) | 0.01 |
| Shredders (%) | 0.01 (0.01) | 0.02 (0.04) | 0.01 (0.01) | n.s. |
| <i>R</i> Collector-gatherers | 15.0 (2.1) | 10.1 (1.9) | 12.2 (2.3) | 0.002 |
| H' Collector-gatherers | 0.8 (0.1) | 0.6 (0.1) | 0.7 (0.2) | 0.016 |
| <i>R</i> Predators | 6.4 (1.3) | 6.3 (1.8) | 6.9 (1.3) | n.s. |
| H' Predators | 0.8 (0.1) | 0.8 (0.3) | 0.9 (0.1) | n.s. |
| <i>R</i> Scrapers | 0.2 (0.2) | 0.1 (0.1) | 0.03 (0.07) | n.s. |
| H' Scrapers | 0.4 (0.8) | 0.1 (0.2) | 0.1 (0.3) | n.s. |
| <i>R</i> Collector-filterers | 0.7 (0.5) | 0.8 (0.4) | 0.7 (0.5) | n.s. |

Note: *R* = richness, H' = Shannon diversity, n.s. = not significant.

accordance with the quality of their riparian zone. The first two axes of the RDA accounted for 42% of the total variance in the species data and 64% of the cumulative percentage variance of the species–environment relationship. The first RDA axis (explained variation = 30%) described a gradient from reaches of good riparian quality to sites of bad riparian quality (Figure 2). Sites of bad riparian quality generally had higher nitrate, conductivity and water temperature, whereas those with native riparian composition had higher dry channel width. The second axis (explained variation = 13%) represented a gradient of biomass of fine particulate organic matter and leaves, and differences in bank-full height among sites. Within the first axis, *Tricorythodes popayanicus*, *Caenis* sp. (ephemeropteran), *Hydrodroma* sp. (acarina), *Heterelmis* sp. (larvae), Lutrochidae (coleopteran) and Oligochaeta were associated with sites of bad riparian quality on the positive side of this axis, whereas *Anacronuria* sp. (plecopteran), *Americabaetis* sp., *Camelobaetidium penai* (ephemeropteran), *Austrelmis* sp. (adult) (coleopteran), *Atopsyche* sp. Hydrobiosidae, glossosomatidae *Mortoniella* sp. (trichopteran), Empididae and Tipulidae (dipteran) were related to sites of good riparian quality on the negative side of this axis. Staphilinidae (adult) (coleopteran), Pyralidae (megalopteran), *Leptohyphes eximius* (ephemeropteran) and Orthoclaadiinae (dipteran) were positively related to axis 2, whereas Ceratopogonidae (dipteran) was negatively related to this axis.

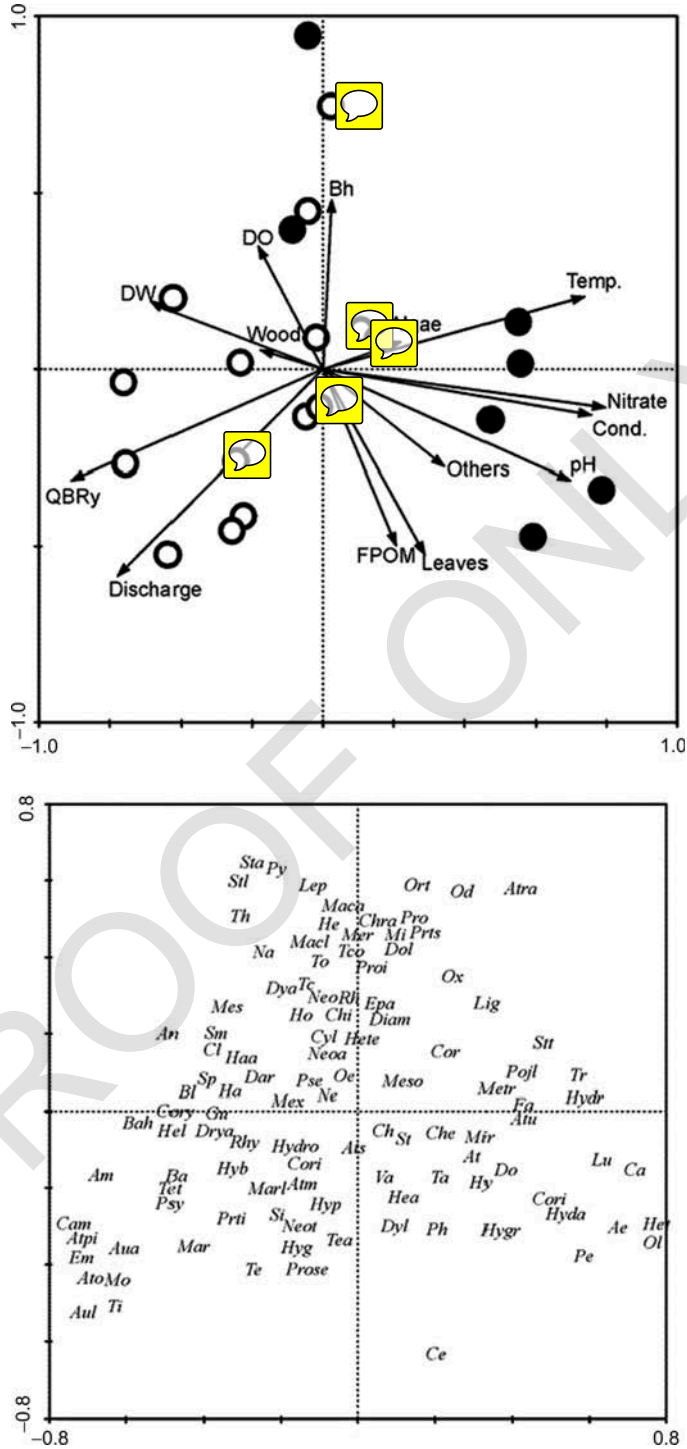


Figure 2. Redundancy analysis (RDA) of macroinvertebrate density in sites of good (white dots), poor (grey dots) and bad (black dots) riparian quality according with the QBRy index. References: DO = dissolved oxygen; Bh = bank-full height; DW = dry channel width; Temp = water temperature; Cond = conductivity; FPOM = fine particulate organic matter.

Discussion

This study has found significant relationships between the quality of the riparian forest and macroinvertebrate metrics. Degraded reaches had lower diversity and richness of invertebrates in comparison with native forested sites. This finding was consistent with other studies related to tropical streams (Kasangaki et al. 2008; Arnaiz et al. 2011) and neotropical streams (Roque and Trivinho-Strixino 2000; Roque et al. 2003; Couceiro et al. 2007). Arnaiz et al. (2011) studied 12 reaches of southeastern Australia finding that those sites of better riparian condition had higher taxonomic richness. Kasangaki et al. (2008) found that the removal of riparian vegetation in a high-altitude equatorial rain-forest negatively affects benthic macroinvertebrate assemblages through a reduction in the number of sensitive taxa and dominance by the intolerant taxa. Roque and Trivinho-Strixino (2000) and Roque et al. (2003) compared forested and non-forested streams of Brazil, finding the highest values for taxonomic richness in stream sections with intact riparian zones. Harrison and Harris (2002) associated reductions in macroinvertebrate abundance and richness with reductions in available bank and littoral habitats caused by livestock activity. Native riparian species at sites of good quality would contribute to greater habitat complexity (Robertson and Rowling 2000; Arnaiz et al. 2011) and higher quantities of palatable food (Bunn et al. 1999; Robertson and Milner 2001) in comparison with those species structuring degraded sites. Accordingly, the lower richness and diversity of degraded sites could be related to the lower cover of vegetation and the replacement of native vegetation by exotic plants, resulting in a reduction of allochthonous material from native species and the input of organic matter from exotic species in the stream. Whole leaves and woody debris of exotic species such as *Citrus* and *Acacia* observed in benthic samples of these sites confirm this statement (Correa pers. comm.), determining a change in the type of resource available for macroinvertebrates (Robertson and Rowling 2000).

As was shown in the ordination analysis, degraded sites had significantly different assemblage composition in comparison with native forested sites, and this was in agreement with reports from similar studies (Thompson and Townsend 2004; Compin and Céréghino 2007; Encalada et al. 2010; Miserendino and Masi 2010; Arnaiz et al. 2011). The crane flies (Tipulidae) are shredders and are known to be important in organic matter breakdown in streams. Their occurrence in high numbers at native forested sites may be attributed to the availability of the plant material on which they feed. Several studies have documented large increases in generalist taxa such as Trichorythidae and Caenidae in streams affected by canopy removal (Benstead et al. 2003; Kasangaki et al. 2008). Degraded environmental conditions would facilitate the dominance of some tolerant groups such as Oligochaeta (Karr and Morishita-Rossano 2001; Couceiro et al. 2007). In addition, many works have reported the sensitivity of Elmidae to anthropogenic changes and the necessity of its inclusion in biological indices (Fossati et al. 2001; Von Ellenrieder 2007; Encalada et al. 2010).

Water temperature was generally higher in sites of poor and bad quality conditions. This finding was consistent with other works performed in tropical streams (Reintal et al. 2003; Kasangaki et al. 2008). The higher temperature in degraded streams is a result of the reduction of the total cover and poor cover structure of the riparian vegetation in these reaches in comparison with native forested sites. In addition, the higher values of conductivity in degraded sites could not be attributed to anthropogenic impacts. The main factor influencing the conductivity at these sites is the nature of the underlying geology.

Density, richness and diversity of collector-gatherers differed significantly among sites of different riparian quality, and this result was in accordance with other studies (Danger and Robson 2004; Thompson and Townsend 2004; Compin and C  r  ghino 2007; Miserendino and Masi 2010) showing the sensitivity of this functional feeding group to anthropogenic changes. Degraded sites are situated at lower altitude, determining a higher deposition of detritus and, consequently, higher food source for this group, increasing its density. Furthermore, leaves and woody debris of exotic species can provide shelter (habitat/refuge) for this group.

The density of scrapers was higher in sites of good riparian quality. This result could be related to the higher dry channel width in these sites, that would determine a higher insolation of the stream and an increase in primary production. Channel width was also higher in these sites, despite the lack of a significant difference as a consequence of the inclusion of site 1, a tributary with a narrow channel in comparison with those including in the main channel of the studied basin. In addition, shredders did not show significant differences in relation to the quality of riparian vegetation, and this could be related to the scarcity of this group in these streams, in accordance with other reports of tropical streams (Dobson et al. 2002; Mathuriau and Chauvet 2002; Wantzen and Wagner 2006).

In contrast with other studies (Rios and Bailey 2006; Walsh et al. 2007; Raymond and Vondracek 2011), the EPT index was not sensitive to anthropogenic changes in riparian conditions. This study found a higher sensitivity of the IBY-4 index in comparison with the EPT and BMWP indices to changes in riparian quality of Yungas streams. In accordance with others studies (Dom  nguez and Fern  ndez 1998; Baptista et al. 2001; Dos Santos et al. 2011), the presence of some tolerant families of Ephemeroptera such as Baetidae and Caenidae, would determine the lower sensitivity of the EPT index to anthropogenic changes in Yungas streams.

In this study, riparian degradation has been linked to a decrease of macroinvertebrate richness and diversity, supporting the use of invertebrates as indicators of riparian condition of Yungas streams. In addition, this work highlights the importance of riparian vegetation in the maintenance of functional organization of macroinvertebrates in subtropical streams. The higher sensitivity of the IBY-4 index to changes in riparian characteristics suggests its usefulness in future management studies in Yungas streams. The conservation of good riparian quality conditions will sustain the ecological integrity of stream communities.

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Appendix 1

List and mean density (\pm SD) of macroinvertebrate taxa of sites of different riparian quality. Abbreviations of taxa relative to the RDA were also shown.

| Taxa | Abbreviation | Good | Bad | Poor |
|--|--------------|------------------|---------------|-------------|
| <i>Aegla</i> sp. | Ae | 0 | 0.6 (1.5) | 11.1 (22) |
| <i>Anacroneuria</i> sp. | An | 99.6 (81) | 97.5 (117) | 8.6 (9.6) |
| <i>Americabaetis alphus</i> | Am | 349.4 (262) | 41.4 (59) | 235.2 (392) |
| <i>Baetodes</i> sp. | Ba | 123.5 (190) | 49.4 (67) | 60.5 (137) |
| <i>Baetodes huaico</i> | Bah | 1264 (983) | 1025.3 (1358) | 2018 (3295) |
| <i>Caenis</i> sp. | Ca | 0.74 (2.3) | 25.9 (34) | 492.3 (652) |
| <i>Camelobaetidius penai</i> | Cam | 474.1 (353) | 8 (10.3) | 12.7 (14) |
| <i>Farrodes</i> sp. | Fa | 0.4 (1.2) | 0 | 4.3 (4.9) |
| <i>Haplohyphes baritu</i> | Ha | 19.1 (20.0) | 9.9 (12.5) | 0.6 (1.5) |
| <i>Leptohyphes eximius</i> | Lep | 55 (48) | 158.6 (187) | 622.8 (878) |
| <i>Nanomis galera</i> | Na | 147.2 (145) | 57.4 (60.1) | 373.5 (495) |
| <i>Thraulodes</i> sp. | Th | 168.7 (208) | 96.9 (112) | 141.7 (134) |
| <i>Thraulodes</i> <i>cochunaensis</i> | Tco | 23.1 (71) | 1.9 (4.5) | 0.6 (1.5) |
| <i>Thraulodes consortis</i> | Tc | 12.9 (30) | 0 | 0.6 (1.5) |
| <i>Tricorythodes</i> <i>popayanicus</i> | Tr | 0.7 (1.6) | 9.9 (9.8) | 380.6 (576) |
| <i>Varipes</i> sp. | Va | 0 | 0 | 0 |
| <i>Atopsyche</i> sp. | Ato | 44.8 (44) | 3.7 (6.2) | 1.9 (4.5) |
| <i>Atopsyche maxi</i> | Atm | 0 | 0.6 (1.5) | 0 |
| <i>Atopsyche spinosa</i> | Atpi | 14.2 (12) | 0 | 0 |
| <i>Chimarra</i> sp. | Chi | 0 | 0 | 1.2 (3) |
| Halipidae (adult) | Haa | 0.4 (1.2) | 0 | 0 |
| <i>Helicopsyche</i> sp. | Hel | 103.9 (171) | 32.7 (49) | 8.6 (19) |
| <i>Hydroptila</i> sp. | Hy | 0.55 (1.7) | 56.2 (87) | 41.4 (30) |
| <i>Marilia</i> sp. (larvae) | Marl | 0 | 1.9 (4.5) | 0 |
| <i>Metrichia</i> sp. (larvae) | Metr | 45.2 (49.5) | 636.4 (554) | 376 (286) |
| <i>Mortoniella</i> sp. (larvae) | Mo | 113.7 (163) | 208.6 (378) | 25.6 (37) |
| <i>Nectopsyche</i> sp. | Ne | 2.0 (3.7) | 0 | 0.6 (1.5) |
| <i>Neotrichia</i> sp. | Neot | 0.4 (1.2) | 0 | 0 |
| <i>Mexitrichia</i> sp. | Mex | 1.1 (3.5) | 0 | 0 |
| <i>Oecetis</i> sp. | Oe | 1.5 (2.5) | 4.3 (11) | 2.8 (6.8) |
| <i>Oxyethira</i> sp. | Ox | 1.1 (3.5) | 0.6 (1.5) | 16.7 (21) |
| <i>Polycentropus</i> <i>joergenseni</i> | Pojl | 0.4 (1.2) | 0 | 4.9 (12) |
| <i>Protoptila</i> sp. | Prti | 0 | 0 | 0 |
| <i>Smicridea</i> sp. | Sm | 196.3 (236) | 383.3 (309) | 197.8 (313) |
| Chironominae | Ch | 1216.8 (783) | 2739 (2198) | 4356 (6500) |
| Orthoclaadiinae | Ort | 4570 (3677) | 8481.5 (6534) | 7058 (3865) |
| Tanypodinae | Ta | 235 (209) | 111.7 (78) | 1405 (2107) |
| Blephariceridae | Bl | 2.2 (3.1) | 0 | 0 |
| Ceratopogonidae | Ce | 56.5 (63.1) | 55.6 (46) | 191.4 (256) |

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(Continued).

| Taxa | Abbreviation | Good | Bad | Poor |
|--------------------------------------|--------------|-------------------|------------------|------------------|
| Psychodidae | Psy | 399.4 (601) | 2.5 (3) | 2.5 (6) |
| <i>Simulium</i> sp. | Si | 592 (474) | 1825.3 (3806) | 1094.4 (2087) |
| Stratiomyidae | St | 0.7 (1.5) | 1.2 (2.0) | 3.1 (4.3) |
| Tipulidae | Ti | 313.3 (297) | 9.3 (9.3) | 9.9 (22) |
| Dolichopodidae | Dol | 0.4 (1.2) | 0 | 0 |
| Empididae | Em | 76.1 (62.3) | 35.8 (35) | 15.7 (22) |
| Oligochaeta | Ol | 314 (456) | 16,421 (19121) | 13,062 (16864) |
| <i>Maruina</i> sp. | Mar | 98.9 (266) | 0.6 (1.5) | 10.5 (14) |
| <i>Austrelmis</i> sp. (adult) | Aua | 76.3 (62.6) | 36.4 (37) | 1.2 (2) |
| <i>Austrelmis</i> sp. (larvae) | Aul | 2139 (2920) | 1616.7 (2253) | 429 (725) |
| Chrysomelidae (adult) | Chra | 0.7 (2.3) | 0 | 0 |
| <i>Cylloepus</i> sp. (adult) | Cyl | 2.4 (4.1) | 0.6 (1.5) | 0 |
| <i>Dryops</i> sp. (adult) | Drya | 1.1 (2.5) | 0 | 0 |
| Dyticidae (larvae) | Dyl | 1.1 (3.5) | 1.9 (2) | 4.3 (4) |
| Dyticidae (adult) | Dya | 0.4 (1.2) | 0.6 (1.5) | 1.2 (2) |
| Ephydriidae (adult) | Epa | 0.7 (2.3) | 1.9 (4.5) | 0 |
| <i>Heterelmis</i> sp. (adult) | Hea | 0 | 0 | 0 |
| <i>Heterelmis</i> sp. (larvae) | Het | 0 | 1.9 (4.5) | 3.4 (4) |
| Hydrophilidae (adult) | Hyda | 0 | 0 | 0.6 (1.5) |
| Hydrophilidae (larvae) | Hydro | 8 (20) | 2.5 (6) | 2.5 (6) |
| Lutrochidae | Lu | 0 | 1.9 (2) | 6.8 (13) |
| <i>Macrelmis</i> sp. (adult) | Maca | 4.1 (7.9) | 4.3 (8.9) | 1.2 (3) |
| <i>Macrelmis</i> sp. (larvae) | Macl | 134.2 (279) | 37.7 (30) | 42 (85) |
| <i>Neelmis</i> sp. (adult) | Neoa | 43.5 (65) | 10.5 (13) | 1.9 (4.5) |
| Staphilinidae (larvae) | Stl | 120.9 (283) | 2.5 (1.9) | 74.1 (149) |
| Staphilinidae (adult) | Sta | 66.8 (149) | 3.1 (4.3) | 104.9 (205) |
| <i>Phanocerus</i> sp. | Ph | 0.9 (2) | 14.8 (23) | 0.9 (2.3) |
| Psephenus sp. | Pse | 91.9 (125) | 44.4 (62) | 39.2 (43) |
| <i>Aturus</i> sp. | Atu | 0 | 1.9 (4.5) | 6.2 (15) |
| <i>Atractides</i> sp. | Atra | 4.1 (6.6) | 44.4 (43) | 53.1 (37) |
| <i>Atractides sinuatipes</i> | Ats | 0 | 1.9 (4) | 0 |
| <i>Atractidella</i> sp. | At | 0 | 0.6 (1.5) | 0 |
| <i>Clathrosperchon punctatus</i> | Cl | 8.1 (10) | 3.1 (3.6) | 4.9 (4.5) |
| <i>Corticacarus</i> sp. | Cor | 11.3 (14.4) | 43.8 (51) | 31.2 (23) |
| <i>Diamphidaxona yungasa</i> | Diam | 0 | 0 | 1.9 (4) |
| <i>Dodecabates dodecaporus</i> | Do | 2.6 (3.5) | 42 (61) | 109.3 (109) |
| <i>Hygrobates</i> sp. | Hygr | 2.6 (3.5) | 3.1 (6) | 31.2 (47) |
| <i>Hygrobates plebejus</i> | Hyp | 0.4 (1.2) | 1.2 (1.9) | 0 |
| <i>Hygrobatella</i> sp. | Hyb | 0 | 0 | 0 |
| <i>Hygrobatella multiacetabulata</i> | Hyg | 0.7 (2.3) | 1.2 (3) | 0 |
| <i>Hydrodroma</i> sp. | Hydr | 1.1 (3.5) | 0.6 (1.5) | 13 (23) |
| <i>Letaxanella argentinensis</i> | Let | 0 | 0 | 0 |

(Continued)

(Continued).

| Taxa | Abbreviation | Good | Bad | Poor |
|--|--------------|-------------|-------------|------------------|
| <i>Meramecia</i> sp. | Mer | 0.4 (1.2) | 0 | 0 |
| <i>Miraxonides</i> sp. | Mir | 0 | 1.2 (3) | 0 |
| <i>Neomamersa</i> sp. | Neo | 0 | 0 | 0.6 (1.5) |
| <i>Protolimnesia interstitialis</i> | Proi | 0.4 (1.2) | 0.6 (1.5) | 0.6 (1.5) |
| <i>Protolimnesia setifera</i> | Prose | 27 (40.5) | 3.1 (2.8) | 44.4 (78) |
| <i>Rhycolimnochares expansisetata</i> | Rhy | 13.5 (16) | 12.3 (13) | 0 |
| <i>Sperchon neotropicus</i> | Sp | 0.4 (1.2) | 0 | 0 |
| <i>Stygaliella tucumanensis</i> | Stt | 1.1 (2.5) | 1.9 (2) | 4.9 (5) |
| <i>Tetrahygrobatella</i> sp. | Tet | 13 (17) | 1.2 (2) | 0.9 (2) |
| <i>Tetrahygrobatella argentinensis</i> | Tea | 2.2 (4.7) | 0.6 (1.5) | 0.6 (1.5) |
| <i>Tetrahygrobatella bovala</i> | Te | 0.7 (1.6) | 0.6 (1.5) | 0 |
| <i>Torrenticola eolombiana</i> | To | 153.5 (251) | 101.9 (104) | 85.5 (61) |
| <i>Protolimnesia</i> sp. | Pro | 0.4 (1.2) | 0 | 0 |
| <i>Protolimesella</i> sp. | Prts | 0.7 (2.3) | 0 | 0 |
| Pyralidae | Py | 6.5 (6.2) | 29 (22) | 53.1 (78) |
| <i>Corydalis</i> sp. | Cory | 6.7 (3.8) | 3.7 (5.0) | 1.2 (2) |
| <i>Chepuvelia</i> sp. | Che | 0 | 0 | 0 |
| Corixidae | Cori | 0 | 0 | 0.6 (1.5) |
| <i>Darwinivelia</i> sp. | Dar | 2.6 (7) | 0 | 1.2 (3) |
| Guerridae | Gu | 0.4 (1.2) | 0 | 0 |
| <i>Hebrus</i> sp. | He | 1.5 (3.6) | 0.6 (1.5) | 0.6 (1.5) |
| <i>Heterocorixa</i> sp. | Hete | 0 | 1.9 (4.5) | 2.5 (6) |
| Horvatina sp. | Ho | 0.4 (1.2) | 0 | 0.6 (1) |
| Ligomorphus sp. | Lig | 1.1 (2.5) | 2.5 (4.5) | 13.6 (26) |
| <i>Mesoveliea</i> sp. | Mes | 6.3 (20) | 3.1 (4.3) | 0.6 (1.5) |
| Mesoveloidea | Meso | 0 | 1.2 (3) | 0 |
| <i>Microvelia</i> sp. | Mi | 2.6 (8.2) | 0 | 0 |
| <i>Petrophila</i> sp. | Pe | 0 | 0 | 1.5 (2.5) |
| <i>Rhagovelia</i> sp. | Rh | 0.4 (1.2) | 9.3 (16) | 0 |
| Odonata | Od | 0 | 2.5 (3) | 17.3 (12) |