Ann. Limnol. - Int. J. Lim. 49 (2013) 129–138
© EDP Sciences, 2013
DOI: 10.1051/limn/2013048

Available online at:
www.limnology-journal.org

Concordance between Ephemeroptera and Trichoptera assemblage in streams from Cerrado – Amazonia transition

Leandro Juen[1]*, Denis S. Nogueira[2], Yulie Shimano[3], Ludgero C. Galli Vieira[4] and Helena S. R. Cabette[5]

[1] Instituto de Ciências Biológicas, Universidade Federal do Pará, Rua Augusto Correia, no 1 Bairro Guamá, 66.075-110 Belém, PA, Brazil
[2] Programa de Pós-Graduação em Ecologia e Evolução, Universidade Federal de Goiás, Caixa Postal 24.241, 74.690-970 Goiânia, GO, Brazil
[3] Programa de Pós-Graduação em Zoologia, Universidade Federal do Pará, Rua Augusto Correia, no 1 Bairro Guamá, 66.075-110 Belém, PA, Brazil
[4] Universidade de Brasília, Campus de Planaltina (FUP), Área Universitária 1, Vila Nossa Senhora de Fátima, 73.340-710 Planaltina, DF, Brazil
[5] Departamento de Ciências Biológicas, Universidade do Estado de Mato Grosso, Caixa Postal 08, 78.690-000 Nova Xavantina, Mato Grosso, Brazil

Received 11 December 2012; Accepted 22 April 2013

Abstract – We evaluated the concordance between assemblages of Ephemeroptera and Trichoptera to verify if they respond similarly to environmental gradients in the basin of Suiá-Missu river in Mato Grosso, central Brazil. We tested the predictions that: (i) the distributional pattern of mayfly and caddisfly larvae is concordant along the spatial and (ii) environmental variation along the basin, and if (iii) taxa are concordant between themselves along the seasons and with the environmental gradients disregarding the seasons of the year. We found a concordance between species composition of mayfly and caddisfly in fall-water and rainy period, when analyzed separately by each season. The concordance between environmental variables and the two taxa analyzed separately also was concordant, but only on the fall-water season. Finally, we found congruence when both analyzed groups disregarded the temporal effect, but it was less representative than when we consider the seasons variation. Our results suggest that the hydrological cycle could be a driver of changes in species composition of mayflies and caddisflies.

Key words: Biotic patterns / macroinvertebrates / aquatic insects / temporal variations / environmental gradient

Introduction

The major goal in ecology is to understand how parameters of biological communities (abundance, richness, and composition) respond to environmental variations (De Chaine and Martin, 2005; Bini *et al.*, 2007). Studies on organism's distributions in nature have revealed recurring patterns along environmental gradients, such as changes in community structures and species composition; latitudinal congruence of diversity can arise even in unrelated groups from all biogeographical regions around the world (Heino, 2002; Bini *et al.*, 2007).

Similar spatial and temporal pattern among two or more taxa in relation to environmental gradient is called community concordance (Bowman *et al.*, 2008). Concordant patterns may also represent common preferences and adaptations, biotic interactions, covariance of assemblage responses to determinant environmental properties or even historical factors related to habitat colonization and evolution (Bini *et al.*, 2008). Thus, studies of community concordance can be used to evaluate patterns in biological communities, biodiversity, as a measure of responses to environmental changes and the placement of different taxa in a single surrogate group (Cameron *et al.*, 2008). Some authors found concordance between aquatic organisms with similar dispersal ability, similar sizes and habits requirements (Heino, 2010), among

---

*Corresponding author: `leandrojuen@ufpa.br`

macroinvertebrates, fish and algae (Monaghan and Soares, 2010), between fish and birds (Paszkowski and Tonn, 2000) and among phytoplankton, zooplankton, macroinvertebrates, and fish (Bowman *et al.*, 2008).

The most important theories about energetic dynamics of rivers suggest that river sizes can produce concordance among several taxa, according to their functional adjusts along the River *Continuum* (Vannote *et al.*, 1980). This information increases the power of hypothesis generalization and effectiveness in model building (Heino, 2010). In addition, certain groups may be surrogate for biodiversity assessment when analyzing responses to environmental changes in local and landscape scale and they can be used in monitoring programs to minimize costs when two or more groups respond similarly to environmental changes (Faith, 2003; Heino and Mykrä, 2006). However, there is evidence suggesting that the choice of indicators or surrogate groups may have limited value for studies of freshwater biodiversity due to divergent patterns or low concordance between communities (Heino *et al.*, 2005; Heino, 2010), which often reflects differences in evolutive history among clades of aquatic organisms.

The insect orders Ephemeroptera (mayflies) and Trichoptera (caddisflies) occupy a wide range of habitats in lotic ecosystems, inhabiting several meso and microhabitats from small streams to large rivers (Wiggins and Mackay, 1978; Wiggins, 1996; Domínguez *et al.*, 2006). The distribution of these insects is related to food availability, environmental variables and energy input along a watershed continuum (Vannote *et al.*, 1980). In freshwater environments, mayflies and caddisflies are important links in food webs, also have similar body size range, and show high local species richness and abundance. These groups are widely studied and used in biomonitoring programs due to high sensitivity to pollutants and other environmental disturbances (Rosenberg and Resh, 1993).

Our goal was to evaluate the concordance between mayflies and caddisflies assemblages along an environmental gradient in Suiá-Missu river tributaries, Mato Grosso, Central Brazil. Specifically, following predictions of similar traits among several groups of Ephemeroptera and Trichoptera on tropical ecosystems we tested if (i) distributional pattern of mayfly and caddisfly larvae are concordant along the spatial and (ii) environmental variation along the basin, and if (iii) taxa are concordant between themselves and with the environmental variation along the seasons of the year.

Methods

Study site

Suiá-Missu river is the main tributary of the Xingu river, located in the transition zone between Cerrado and Amazon Forest, in Mato Grosso, central Brazil (between 11°15' to 13°40'S and 53°15' to 51°15'W). Climate is seasonal tropical, with a dry (May–October) and rainy season (November–April). In Köppen climate classification, the major part of the region is Savanna (*Aw*), with microregions with monsoon subtype (*Am*), and tropical rainy (*A*). The rainfall annual mean is 1.370 mm, and the monthly mean temperature varies between 32.7 and 17.0 °C (Ratter *et al.*, 1978).

The predominant economic activities are wood exploitation, rice and soybean monocultures, and extensive pasture for cattle (Riva *et al.*, 2007). Recently, dams construction project to provide energy building have generated interest conflicts between conservationists and government priorities in middle Xingu River, in Pará State, despite their low-energy supply potential and probability of high sedimentation (Carvalho *et al.*, 2004). The impact of dam construction on regional biodiversity is still understudied, but it is especially dangerous for aquatic organisms.

For evaluation of concordance among Ephemeroptera and Trichoptera, we selected 12 sampling sites encompassing several types of Cerrado vegetation in Suiá-Missu river and its tributaries (Fig. 1). The sampled streams ranged from 1st to 6th order, following Strahler (1957), and they varied in channel width, water flow, vegetation in riparian zone, and conservation status (see Fig. 1 for abbreviations). The substrata found in the sampling sites ranged from roots on banks, submerged vegetation, leaves, stems and twigs in pools and rapids, rocks, gravel and sand in lotic environments; and submerged macrophytes, roots and fine particulate organic matter, and silt in marshy environments (for details see Shimano *et al.*, 2010; Nogueira *et al.*, 2011).

Data sampling

We sampled insect larvae in dry (September 2007), rainy (December 2007), and fall-water (May 2008) seasons. We established 100 m transects in each one of 12 sampling sites, subdivided into 20 segments of 5 m following the methodology adapted from Juen and De Marco (2011). We sampled portions of substratum within each segment with a dip netting of 18 cm diameter and 0.025 mm mesh. By this method, the pseudo-sample unit is each 5 m segments of the 100 m transect, which are used to estimate species richness for each site. Taxonomic compositions are assessed by sum of species abundance in all subsamples (see details below).

We identified specimens using genus keys for Ephemeroptera (Domínguez *et al.*, 2006) and Trichoptera (Angrisano and Korob, 2001; Pes *et al.*, 2005). Afterwards, the species were morphospecied using phenotypic characteristics to distinguish the last larval stages within genus. The specimens were preserved in 85% alcohol and housed at the Coleção Zoobotânica "James Alexander Ratter" (CZNX), Universidade do Estado de Mato Grosso, Brazil.

Physicochemical data were sampled in three seasons, in the same transects used by insect larvae sampling. We recorded water temperature, pH, turbidity, electrical conductivity, dissolved oxygen, depth, flow, total soluble solids, and redox potential using a multiparameter probe.

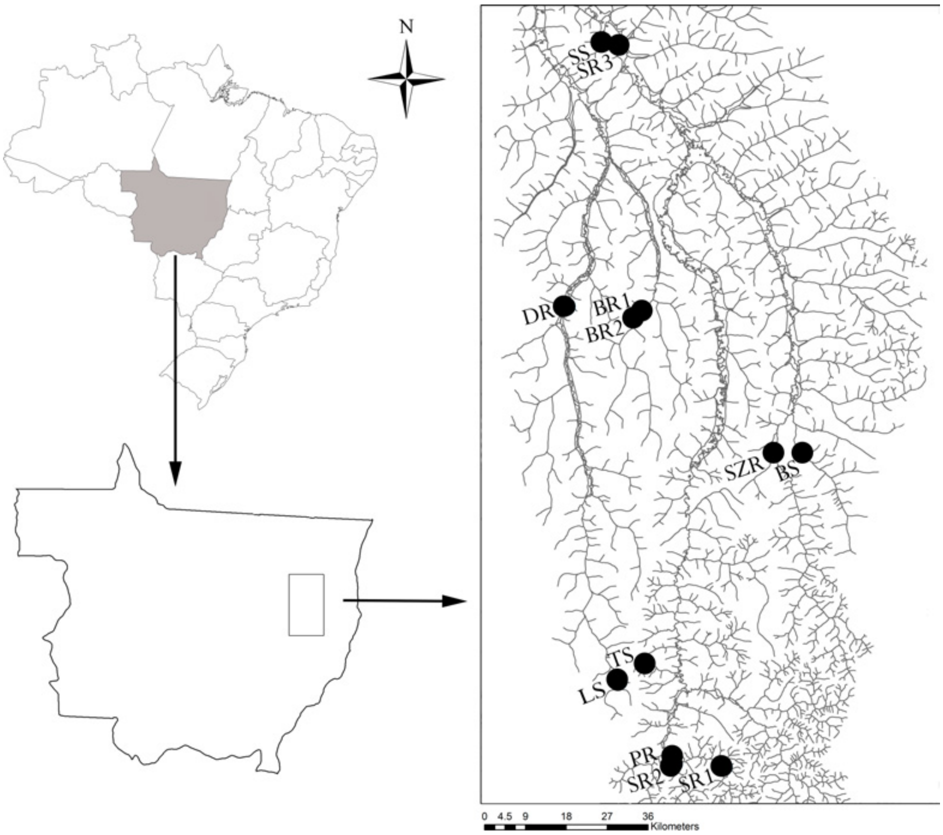

Fig. 1. Sampling sites of mayfly and caddisfly larvae in the Basin of Rio Suiá-Missu, Mato Grosso (SR1 = Suiá-Missu River; SR2 = Suiá-Missu River; SR3 = Suiá-Missu River; SS = Sucuri Stream; DR = Darro River; BR1 = Betis River; BR2 = Betis River; SZR = Suiazinho River; TS = Transição Stream; PR = Piabanha River; LS = Lúcio Stream; and BS = Brejão Stream).

We measured concentrations of ammonium nitrate, nitrite, and phosphate ions with a portable spectrophotometer, and total hardness, Ca^{++} and Mg^{++} with 0.0002 M EDTA. Streams and rivers widths were measured with a tape-measure. We evaluated environmental integrity with Habitat Integrity Index-HII (Nessimian *et al*., 2008). This index is based on measurement of a series of parameters visually assessed related to land use, riparian zone conservation, streambed characteristics, and stream channel morphology related to margin and streamlet bed structure.

Data analysis

We used eight variables that were selected after we tested multicolinearity by correlation matrix (water temperature, pH, turbidity, electrical conductivity, dissolved oxygen, depth, width, and HII). To reduce dimensionality in abiotic data, we used principal component analysis (PCA) based on the correlation matrix of environmental variables after log transformation (except for pH) and standardization (Olden *et al*., 2001). Posteriorly, we used the Broken stick method (Jackson and Harvey, 1993) to select the number of axes for interpretation. Analysis was designed in R environment by using functions of vegan package (Oksanen *et al*., 2011). To test the significance of the groups (seasons) formed in PCA, we run an ANOVA for the two first PCA axis.

We estimated species richness for each insect order and for both orders combined with the nonparametric estimator first order jackknife by using EstimateS v.7.5.0 (Colwell, 2005). Then, we have an estimative for species richness for each order in studied seasons and for a combination of both orders, which were regressed against environmental gradients summarized in PCA axes. In addition, we tested species richness and abundance along seasons using non-parametric repeated measures Friedman ANOVA and pairwise comparisons for significant results were evaluated by using Wilcoxon matched pairs test for dependent samples (Zar, 1999).

We analyzed the structure of mayfly and caddisfly assemblages in relation to abiotic data using an indirect gradient analysis (Gauch, 1982). We used a detrended correspondence analysis (DCA) to summarize species composition (abundance) in different environments. Correspondence analysis was run with data of all streams sampled, after log transformation.

We evaluated the degree of concordance between species composition matrix and abiotic data using a Procrustean routine (Jackson, 1995). We performed this analysis with the scores of the first two axes of each ordination (DCA and PCA), since they contain most of the variability of original data. We perform this procedure

Table 1. Environmental variables, loadings, eigenvalues and Broken-stick values of the first two axes of PCA. (Mean and SD in 36 measurements, three in each stream site).

	Dry season		Rainy season		Fall-water season		Loadings	
	Mean	SD	Mean	SD	Mean	SD	PCA 1	PCA 2
HII	0.63	0.1	0.63	0.1	0.63	0.1	– 0.41	– 0.36
pH	5.5	0.41	5.03	0.2	4.89	0.47	– 0.2	0.08
Electrical conductivity	0.72	0.21	0.78	0.26	0.68	0.37	0.03	0.32
Turbidity	7.93	8.46	5.97	6.33	62.02	27.56	**0.63**	– **0.77**
Water temperature	26.95	3.36	27.34	1.41	24.13	0.98	– **0.61**	0.17
Dissolved oxygen	5.97	2.26	6.23	1.8	7.53	2.58	– 0.11	– 0.38
Channel width	33.51	42.85	33.48	43.15	34.58	44.06	– **0.89**	– 0.41
Depth	1.46	1.4	1.45	1.69	1.28	1.11	– **0.57**	– **0.53**

in different ways to account for all environmental variability in our dataset. Firstly, we evaluated spatial concordance of the order within each season (dry, rainy, and fall-water seasons). Afterwards, we analyzed the concordance between a dataset composed by both orders against environmental seasonality in three separate seasons. Finally, we analyzed the temporal concordance using seasons as replicate. The test statistic (m^2) measures how divergent ordination patterns of the two datasets are, and the significance is then tested after 10 000 permutations. The Procrustes analysis was performed in R software (version 2.10.1) (R Development Core Team, 2011).

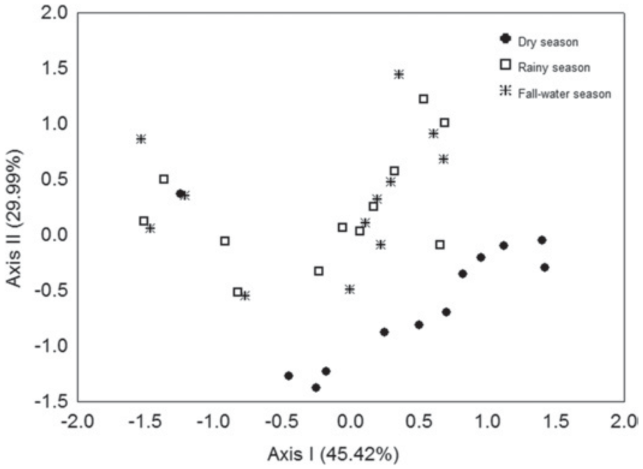

Fig. 2. PCA showing similarity of sampling sites according to stream physicochemistry and HII in the basin of Rio Suiá-Missu, along three seasons (2007–2008), Mato Grosso.

Results

Environmental variables

The first two PCA axes explain about 75.41% of the variation in environmental data (Table 1). PC1 explained 45.42% of environmental variation and was negatively correlated with water temperature, channel width and depth, and positively related to turbidity. PC2 explained 29.99% of environmental data and was negatively correlated with turbidity. There was a cluster of samples from fall-water season (at the bottom of the figure), whereas those of dry and rainy season did not form a cluster (Fig. 2). However, the ANOVA showed that there was a difference between fall-water samples and the remaining ones in the 2nd axis ($F_{(2,\ 33)} = 9.878$; $P < 0.001$).

Assemblage structure

We sampled an overall of 2223 larvae, 1356 were mayflies (Appendix 1) and 867 caddisflies (Appendix 2). The mayflies sampled belonged to seven families, 31 genera, and 42 species/morphospecies. The highest abundance was recorded at fall-water, with 529 larvae, with 44.1 ± 31.6 (mean ± SD) per sample (Table 2). In dry season, we found 464 larvae (38.7 ± 34.7) and 363 larvae (30.3 ± 28.0) in rainy season. Estimated richness was higher in dry season (14.1 ± 8.4; mean ± confidence interval), followed by fall-water (12.2 ± 5.8), and rainy seasons (11.1 ± 8.4). There was significant difference in mayflies abundance with relation to season (ANOVA repeated measure ($N = 12$, d.f. $= 2$) $= 7.277$; $P = 0.026$), restricted between fall-water and rainy (Wilcoxon = 6.500; $z = 2.356$; and $P = 0.018$) (Fig. 3). Mayflies estimated richness have shown no difference among seasons (ANOVA repeated measure ($N = 12$, d.f. $= 2$) $= 0.933$; $P = 0.627$).

Caddisfly larvae belonged to seven families, 17 genera, and 45 species/morphospecies. Abundance was higher in average in fall-water (3.1 ± 1.1), followed by dry (2.5 ± 1.1), and rainy seasons (1.6 ± 1.2). There was significant differences in abundance and estimated richness among seasons (ANOVA repeated measure ($N = 12$, d.f. $= 2$) $= 8.979$; $P = 0.011$, and ANOVA repeated measure ($N = 12$, d.f. $= 2$) $= 12.046$; $P = 0.002$, respectively). We identify that fall-water was more abundant than rainy season (Wilcoxon = 4.000; $z = 2.746$; $P = 0.006$), while fall-water and dry seasons were richer than rainy season (Wilcoxon = 0.000; $z = 2.934$; $P = 0.003$ and Wilcoxon = 1.500; $z = 2.310$; $P = 0.021$, respectively) (Fig. 3).

Accordingly, there were significant relationships among total species richness and the first PCA axes, showing decrease in species number with enhancement of channel width, depth, and increase of water temperature and decrease of turbidity (Fig. 4). Caddisflies species richness showed the same pattern observed for the general community ($r^2 = 0.128$; $P = 0.031$; $y = 4.889 - 1.912{*}x$),

Table 2. Degrees of concordance between DCA scores (for each taxon alone and combined dataset) and PCA scores (for environmental variables) calculated for the environmental variables. m^2 statistic is below and significance is above main diagonal.

	Ephemeroptera	Trichoptera	General	Environment
Ephemeroptera		**0.011**	–	0.072
Trichoptera	**0.387**		–	0.115
General	–	–		**0.001**
Environment	0.309	0.275	**0.434**	

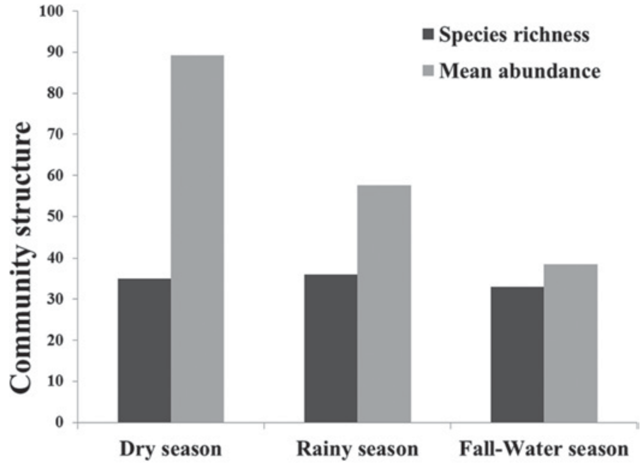

Fig. 3. Seasonal distribution of general species richness and abundance of Trichoptera and Ephemeroptera along three seasons (2007–2008), in Suiá-Missu River basin, Mato Grosso.

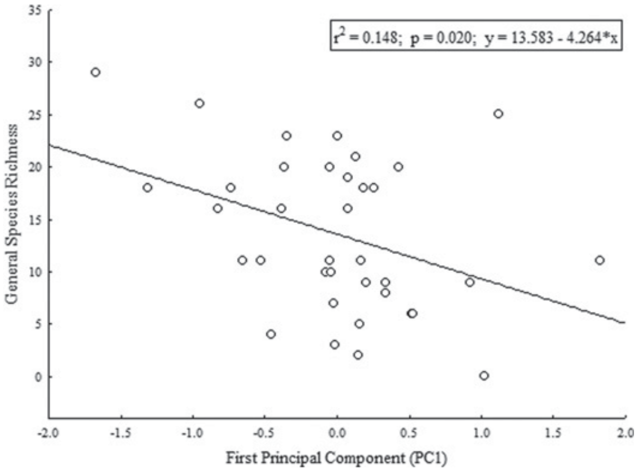

Fig. 4. Relationships of general species richness and the first Principal Component of environmental variation in three seasons (2007–2008), in Suiá-Missu River basin, Mato Grosso.

whereas mayflies did not have relationships with these environmental gradients ($P > 0.05$).

DCA did not indicate a clear separation of sampling sites according to seasons, in the first or second axis (Fig. 5). The ANOVA results corroborate these results, indicating no differences in species composition among the three seasons both the first (ANOVA: $F_{(2,32)} = 0.300$; $P = 0.743$) as well as in the second axis (ANOVA: $F_{(2,32)} = 0.036$; $P = 0.965$).

Concordance

Procrustes showed a high concordance between assemblages of mayflies and caddisflies during fall-water and rainy seasons ($m^2 = 0.66$, $P < 0.01$; $m^2 = 0.56$, $P < 0.05$, respectively). Contrasting our expectative, in dry season there was no concordance between the assemblages ($m^2 = 0.39$, $P = 0.25$). We found a concordance between environmental variables and the two taxa analyzed separately only during fall-water (Fig. 6). We also found high congruence when both groups were analyzed jointly against environmental seasonality ($m^2 = 0.43$, $P < 0.05$, Fig. 6). When we analyzed the temporal concordance using each seasons as replicate there were both concordance between caddisflies and mayflies, and between the species composition of both orders and the environment variables (Table 2). Results for concordance without seasonal variation were less representative than when we consider the seasons separately, showing the importance in considering demographic fluctuation and seasonality for considerations about concordance pattern.

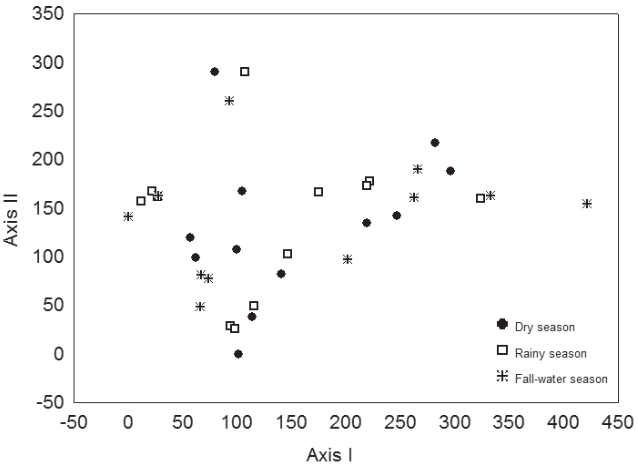

Fig. 5. DCA sampling sites according to species composition along three seasons (2007–2008), basin of Rio Suiá-Missu, Mato Grosso.

Discussion

Seasonal rainfall is a major determinant of changes in several environmental variables in freshwater ecosystems in Brazilian Cerrado, for example, pH, water temperature (Melo and Froehlich, 2001; Bini *et al.*, 2007), turbidity, conductivity, water flow, and discharge can be affected by

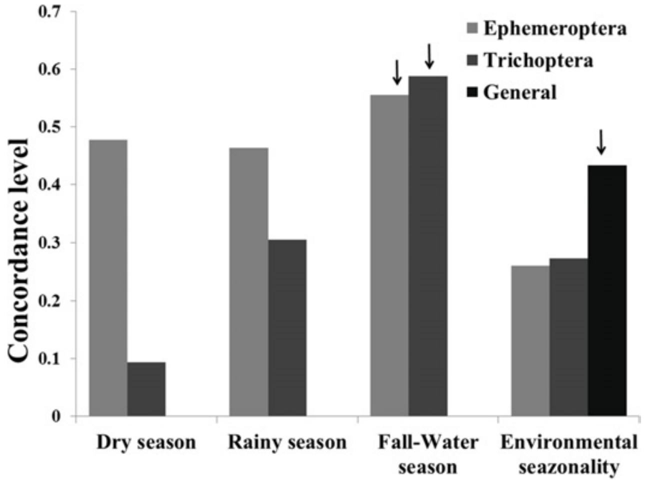

Fig. 6. Concordance patterns for caddisflies and mayflies and the environment along three seasons and environmental seasonality; concordance for composite community dataset (caddsflies and mayflies) and environmental seasonality along three seasons (2007–2008), in Suiá-Missu River basin, Mato Grosso. The black arrows indicate significant concordance according to a *P*-level of 0.05.

rain. During our study an unusual period of drought in December caused by delay of rainfall, reflected in low variation in some environmental variables.

The concordance found between Ephemeroptera and Trichoptera assemblages in fall-water and rainy seasons can reflect the sensibility of both orders to rain action. Although they present adaptations to fix themselves in the substratum, like dorsoventrally flattened body and claws with denticles and setae in Ephemeroptera (Domínguez *et al.*, 2006), and line mooring construction and silk refuges in Trichoptera (Wiggins, 1996), they are sensible to rains due to the increase and turbulence of water, that often carrys them downstream (Bispo *et al.*, 2001; Melo *et al.*, 2003). Disturbance effects caused by increase of flow have been shown in experiments in streams communities of New Zealand (Melo *et al.*, 2003). The authors showed decrease of species density and diversity in the first weeks after disturbance. According to the Flood Pulse Concept, proposed by Junk *et al.* (1989), high flow caused by water of the rains can cause catastrophic events for many organisms, being like a periodical reset in physics and biotic environment, reflecting in the homogenization of available substrata and fauna in this season of years, reflecting also in dominant species distribution.

Another factor that probably contributed to the concordance of these orders is that aquatic insects present behavioral responses that allow them to escape from negative effects of disturbance, through lateral migrations to search refuge preceding the first rains (Hynes, 1970; Junk *et al.*, 1989). Moreover, reproduction is adjusted to rainy season, in which they emerge, reproduce and oviposit, preferring substrata protected from fast flow (Brittain, 1982), and running from the period of greater hydrological instability. These kinds of behaviors happening in synchrony between both orders probably contributed to our results. Heino *et al.* (2003) agree that concordant patterns verified in fall-water season can be related to morphological, physiological adaptations, and especially, behavioral adaptations of these insects to hydrological instability.

In a previous study, Heino (2002) found a strong relationship between both species richness of dragonflies, macrophytes and fish and species richness and richness of the community as a whole. However, same author reports a negative correlation between species richness of Plecoptera and other organisms, because stoneflies have different requirements for water temperature and dissolved oxygen. In a second study, Heino *et al.* (2003) found low concordance in species richness of larvae from four groups in headwater streams (Ephemeroptera, Trichoptera, Plecoptera, and Chironomidae) in Finland, other studies often found low concordance in the structure and species richness in freshwater ecosystems, especially at small spatial scales (*e.g.* within basin Allen *et al.*, 1999; Paavola *et al.*, 2003; Tolonen *et al.*, 2005). Low concordance can arise between similar-sized organisms, such as different groups of aquatic insects, because of specific requirements to use of the environment. For instance, this phenomenon can be caused by distinct dietary and physiological requirements, differences in substrate choice, and resistance to flow disturbance (Heino *et al.*, 2003).

Environmental factors have been suggested as the main factor responsible for structuring freshwater communities (Landeiro *et al.*, 2012; Roque *et al.*, 2012). The biological processes that influence taxa establishment and their persistence in these environments are, often, manifested most strongly at the regional scale, for instance because hydrology of the catchments and inherent geomorphology are pointed out to be the main controlling factor of environmental disturbance at local scale. Considering the seasonal variations, only in the fall-water season Ephemeroptera and Trichoptera were congruent with environmental variables.

Despite evidences suggesting that deterministic and stochastic processes drive variations in freshwater community composition and congruence (Siqueira *et al.*, 2012), evaluations of the prevalence of these processes along temporal and environmental gradients including the effects of disturbances could have promising information, and need to be explicitly incorporated into the partitions variance analysis.

However, other characteristics of community seem to contribute with greater concordances in diversity patterns, as reported for taxa with similar body size (Allen *et al.*, 1999), more narrow taxonomic (niche conservatism), and within functional groups (Siqueira *et al.*, 2012).

In the same way, according to Grenouillet *et al.* (2008), the concordance between macroinvertebrates, diatomaces and fish can be explained by trophic interactions and convergences in organism histories. Once, non-related organisms with similar response to the environment can have similar patterns of distribution on aquatic ecosystems. However, we cannot affirm that environmental variables can be replaced by mayfly and caddisfly sampling once the correlation was low ($r = 0.434$, strong correlations > 0.70, obtained in Heino, 2010).

It shows one more time, that independently of each other, both orders are sensible to changes in water quality and highlight the importance of these groups to conservation planning and water quality biomonitoring (Landeiro *et al.*, 2012). Corroborating our result, Couceiro *et al.* (2010) analyzing indicator species, found that four taxa of Ephemeroptera and ten of Thichoptera were associated with non-impacted streams, and no taxa indicated the impacted ones. The greater concordance between both taxa and environmental variables may indicate that they are complementary and need to be sampled together if environmental changes have to be detected. Nevertheless, we cannot affirm that only one order could be sampled due to the greater results found when we used both orders in analysis. If indicator groups and surrogates are relevant to freshwater biodiversity conservation more investments, deep and wide studies focusing on temporal and spatial variation have to be made, mainly addressing reduction of costs by use of high-order taxonomic resolution and cross-taxon congruence (Heino, 2010).

We conclude that environmental variations in streams resulted from temporal changes (season of the year) directly influence the patterns of mayflies and caddisflies distributions, though they have adaptations for rain effect. The differences in concordance between Ephemeroptera and Trichoptera found according to the seasons of the years, the fact that both orders agree with environmental variables only in the fall-water season, and less representativeness of our results when seasons were analyzed together make us support the fact that the hydrological cycle could be a driver of changes in species composition, as predicted by the Flood Pulse Concept.

Acknowledgements. We thank Frederico Falcão Salles and Ana Maria Oliveira Pes for identification of mayfly and caddisfly larvae, respectively. Ully M. Pozobom Costa did the limnological analysis. We thank all staff of Laboratório de Entomologia of Nova Xavantina, University of State of Mato Grosso – UNEMAT, for laboratory and field help. We also thank support from PROCAD/CAPES no. 109/2007 and Conselho Nacional de Pesquisa e Desenvolvimento – CNPq (no. 520268/2005–9) received by the project's coordinator Ladislau A. Skorupa (EMBRAP-Meio Ambiente). DSN received a fellowship of Coordenação de Aperfeiçoamento de Pessoa de Nível Superior – CAPES and YS of Fundo de Amparo à Pesquisa do Pará – FAPESPA. We also thank Dr. Frederico Falcão Salles and anonymous referees for their helpful comments on an earlier version of the manuscript.

References

Allen A.P., Whittier T.R., Kaufmann P.R., Larsen D.P., O'Connor R.J., Hughes R.M., Stemberger R.S., Dixit S.S., Brinkhurst R.O., Herlihy A.T. and Paulsen S.G., 1999. Concordance of taxonomic richness patterns across multiple assemblages in lakes of the northeastern United States. *Can. J. Fish Aquat. Sci.*, *56*, 739–747.

Angrisano E.B. and Korob P.G., 2001. Trichoptera. *In*: Fernández H.R. and Domínguez E. (eds.), Guia para la identificacion de los artrópodos bentônicos sudamericanos, Editorial Universitaria de Tucumám, Tucamám, 55–92.

Bini L.M., Vieira L.C.G., Machado J. and Velho L.F.M., 2007. Concordance of species composition patterns among microcrustaceans, rotifers and testate amoebae in a shallow pond. *Int. Rev. Hydrobiol.*, *92*, 9–22.

Bini L.M., Silva L.C.F., Velho L.F.M., Bonecker C.C. and Lansac-Toha F.A., 2008. Zooplankton assemblage concordance patterns in Brazilian reservoirs. *Hydrobiologia*, *598*, 247–255.

Bispo P.C., Oliveira L.G., Crisci-Bispo V.L. and Silva M.M., 2001. A pluviosidade como fator de alteração da entomofauna bentônica (Ephemeroptera, Plecoptera e Trichoptera) em córregos do Planalto Central do Brasil. *Acta Limnol. Bras.*, *13*, 1–9.

Bowman M.F., Ingram R., Reid R.A., Somers K.M., Yan N.D., Paterson A.M., Morgan G.E. and Gunn J.M., 2008. Temporal and spacial concordance in community composition of phytoplakton, zooplakton, macroinvertebrate, crayfish, and fish on Precambrian Shield. *Can. J. Fish Aquat. Sci.*, *65*, 919–932.

Brittain J.E., 1982. Biology of mayflies. *Annu. Rev. Entomol.*, *27*, 119–147.

Cameron S.E., Williams K.J. and Mitchell D.K., 2008. Efficiency and concordance of alternative methods for minimizing opportunity costs in conservation planning. *Conserv. Biol.*, *22*, 886–896.

Carvalho N.D.O., Cafe F.A., Mota G.D.O., Franco H.C.D.B. and Braga A., 2004. Assessment of the sedimentation in the reservoirs of the Belo Monte hydroelectric complex, Xingu River, Brazil. *In*: Proceedings of the Ninth International Symposium on River Sedimentation, Vols 1–4, 233–242.

Colwell R.K., 2005. EstimateS: statistical estimation of species richness and shared species from samples. Version 7.5.

Couceiro S.R.M., Hamada N., Forsberg B.R. and Padovesi-Fonseca C., 2010. Effects of anthropogenic silt on aquatic macroinvertebrates and abiotic variables in streams in the Brazilian Amazon. *J. Soils Sediments*, *10*, 89–103.

DeChaine E.G. and Martin A.P., 2005. Historical biogeography of two alpine butterflies in the Rocky Mountains: broad-scale concordance and local-scale discordance. *J. Biogeogr.*, *32*, 1943–1956.

Domínguez E., Molineri C., Pescador M.L., Hubbard M. and Nieto C., 2006. Ephemeroptera of South America, Pensoft Publ., Moscow, 646 p.

Faith D.P., 2003. Environmental diversity (ED) as surrogate information for species-level biodiversity. *Ecography*, *26*, 374–379.

Gauch H.G., 1982. Multivariate Analysis in Community Ecology, Cambridge University Press, Cambridge, 312 p.

Grenouillet G., Brosse S., Tudesque L., Lek S., Baraillé Y. and Loot G., 2008. Concordance among stream assemblages and spatial autocorrelation along a fragmented gradient. *Divers Distrib.*, *14*, 592–603.

Heino J., 2002. Concordance of species richness patterns among multiple freshwater taxa: a regional perspective. *Biodivers. Conserv.*, *11*, 137–147.

Heino J., 2010. Are indicator groups and cross-taxon congruence useful for predicting biodiversity in aquatic ecosystems?*Ecol. Indic.*, *10*, 112–117.

Heino J. and Mykrä H., 2006. Assessing physical surrogates for biodiversity: Do tributary and stream type classifications reflect macroinvertebrate assemblage diversity in running waters?*Biodivers. Conserv.*, *129*, 418–426.

Heino J., Muotka T. and Paavola R., 2003. Determinants of macroinvertebrate diversity in headwater streams: regional and local influences. *J. Animal Ecol.*, *72*, 425–434.

Heino J., Paavola R., Virtanen R. and Muotka T., 2005. Searching for biodiversity indicators in running waters: do bryophytes, macroinvertebrates, and fish show congruent diversity patterns? *Biodivers. Conserv.*, *14*, 415–428.

Hynes H.B.N., 1970. The ecology of stream insects. *Annu. Rev. Entomol.*, *15*, 25–42.

Jackson D.A., 1995. Protest – a Procrustean randomization test of community environment concordance. *Ecoscience*, *2*, 297–303.

Jackson D.A. and Harvey H.H., 1993. Fish and benthic invertebrates: community concordance and community-environment relationships. *Can. J. Fish Aquat. Sci.*, *50*, 2641–2651.

Juen L. and De Marco P., 2011. Odonata biodiversity in terra-firme forest streamlets in Central Amazonia: on the relative effects of neutral and niche drivers at small geographical extents. *Insect. Conserv. Diver.*, *4*, 265–274.

Junk W.J., Bayley P.B. and Sparks R.E., 1989. The flood pulse concept in river-floodplain systems. *In*: Dodge D.P. (ed.), Proceedings of the International Large River Symposium, 110–127.

Landeiro V.L., Bini L.M., Melo A.S., Pes A.M.O. and Magnusson W.E., 2012. The roles of dispersal limitation and environmental conditions in controlling caddisfly (Trichoptera) assemblages. *Freshwater Biology*, *57*, 1554–1564.

Melo A.S. and Froehlich C.G., 2001. Macroinvertebrates in neotropical streams: richness patterns along a catchment and assemblage structure between 2 seasons. *J. N. Am. Benthol. Soc.*, *20*, 1–16.

Melo A.S., Niyogi D.K., Matthaei C.D. and Townsend C.R., 2003. Resistance, resilience, and patchiness of invertebrate assemblages in native tussock and pasture streams in New Zealand after a hydrological disturbance. *Can. J. Fish Aquat. Sci.*, *60*, 731–739.

Monaghan K.A. and Soares A.M.V.M., 2010. The bioassessment of fish and macroinvertebrates in a Mediterranean–Atlantic climate: habitat assessment and concordance between contrasting ecological samples. *Ecol. Indic.*, *10*, 184–191.

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

Nogueira D.S., Cabette H.S.R. and Juen L., 2011. Estrutura e composição da comunidade de Trichoptera (Insecta) de rios e áreas alagadas da Bacia do rio Suiá-Missu, Mato Grosso, Brasil, *Iheringia. Sér. Zool.*, *101*, 173–180.

Oksanen J., Blanchet F.G., Kindt R., Legendre P., Minchin P.R., O'Hara R.B., Simpson G.L., Solymos P., Stevens M.H.H. and Wagner H., 2011. Vegan: Community Ecology Package. R package version 2.0-2. http://CRAN.R-project.org/package = vegan.

Olden J.D., Jackson D.A. and Peres-Neto P.R., 2001. Spatial isolation and fish communities in drainage lakes. *Oecologia*, *127*, 572–585.

Paavola R., Muotka T., Virtanen R., Heino J. and Kreivi P., 2003. Are biological classifications of headwater streams concordant across multiple taxonomic groups?*Freshwater Biol.*, *48*, 1912–1923.

Paszkowski C.A. and Tonn W.M., 2000. Community concordance between the fish and aquatic birds of lakes in northern Alberta, Canada: the relative importance of environmental and biotic factors. *Freswater Biol.*, *43*, 421–437.

Pes A.M.O., Hamada N. and Nessimian J.L., 2005. Chaves de identificação de larvas para famílias e gêneros de Trichoptera (Insecta) da Amazônia Central, Brasil. *Revta Bras. Ent.*, *49*, 181–204.

Ratter J.A., Askew G.P., Montgomery R.F. and Gifford D.R., 1978. Observations on the vegetation of northeastern Mato Grosso II. Forest and Soils of the Rio Suiá-Missu area. *R. Soc. Lond.*, *293*, 191–208.

R Development Core Team, 2011. R: A Language and Environment for Statistical Computing, R Foundation for Statistical Computing, Vienna, Austria.

Riva A.L.M., Fonseca L.F.L. and Hasenclever L., 2007. Instrumentos econômicos e financeiros para a conservação ambiental no Brasil: Uma análise do estado da arte no Brasil e no Mato Grosso, desafios e perspectivas, Instituto Socioambiental – ISA, São Paulo, 138 p.

Roque F.O., Lima D.V.M., Siqueira T., Vieira L.J.S., Stefanes M. and Trivino-Strixino S., 2012. Concordance between macroinvertebrates communities and the typological classification of White and clear-water streams in western Brazilian Amazonia. *Biota Neotropical*, 12, 83–92.

Rosenberg D.M. and Resh V.H., 1993. Introduction to freshwater biomonitoring and benthic macroinvertebrates. *In*: Rosenberg D.M. and Resh V.H. (eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates, Chapman and Hall, New York, 1–9.

Shimano Y., Cabette H.S.R., Salles F.F. and Juen L., 2010. Composição e distribuição da fauna de Ephemeroptera (Insecta) em área de transição Cerrado-Amazônia, Brasil. *Iheringia*, *100*, 301–308.

Siqueira T., Bini L.M., Roque F.O. and Correnie K., 2012. A metacommunity framework for enhancing the effectiveness of biological monitoring strategies. *Plos One*, 7, 1–12.

Strahler H.N., 1957. Quantitative analysis of watershed geomorphology. *Am. Geophys. Union Trans.*, *38*, 913–920.

Tolonen K.T., Holopainen I.J., Hamalainen H., Rahkola-Sorsa M., Ylostalo P., Mikkonen K. and Karjalainen J., 2005. Littoral species diversity and biomass: concordance among organism groups and the effects of environmental variables. *Biodivers. Conserv.*, *14*, 961–980.

Vannote R.L., Minshall G.W., Cummins K.W., Sedell J.R. and Cushing C.E., 1980. The river continuum concept. *Can. J. Fish Aquat. Sci.*, *37*, 130–137.

Wiggins G.B., 1996. Larvae of the North American Caddisfly Genera (Trichoptera), University of Toronto Press, Toronto, 457 p.

Wiggins G.B. and Mackay R.J., 1978. Some relationships between systematics and trophic ecology in Neartic aquatic insects, with special references to Trichoptera. *Ecology*, *59*, 1211–1220.

Zar J.H., 1999. Biostatistical Analysis, Prentice–Hall, Englewood Cliffs, NJ, 960 p.

Appendix 1. Mean and standard deviation for species abundance of mayflies per stream site, during the sampling period (2007–2008) in the basin of Rio Suiá-Missu, Mato Grosso.[colcnt = 7].

	Fall-water season		Dry season		Rainy season	
Especies	Mean	SD	Mean	SD	Mean	SD
Amanahyphes saguassu (Salles & Molineri, 2006)	1.583	2.539	0.417	0.669	0.417	1.443
Americabaetis alphus (Lugo-Ortiz & McCafferty, 1995)	0.250	0.866	0.333	0.651	0.417	1.443
Apobaetis fuizai (Salles & Lugo-Ortiz, 2002)	0.000	0.000	0.000	0.000	0.083	0.289
Askola sp.	0.083	0.289	0.500	1.000	0.167	0.389
Asthenopus cf. *curtus* (Hagen, 1861)	4.667	8.532	0.500	1.732	0.417	0.793
Aturbina sp.n.	0.500	1.000	0.417	0.793	0.667	1.155
Aturbina georgei (Lugo-Ortiz & McCafferty, 1996)	0.167	0.389	0.667	1.155	0.833	2.290
Brasilocaenis irmeli (Puthz, 1975)	0.417	1.165	0.000	0.000	0.000	0.000
Brasilocaenis sp.	1.833	4.840	0.083	0.289	0.167	0.577
Caenis cuniana (Froehlich, 1969)	3.167	5.060	1.500	2.876	8.250	17.705
Caenis fitkaui (Malzacher, 1986)	0.500	1.446	0.500	1.243	1.583	4.010
Caenis pfluggelderi (Malzacher, 1990)	0.917	1.311	0.333	0.651	0.000	0.000
Callibaetis sp. 1	3.750	6.468	1.667	2.640	3.167	4.648
Callibaetis sp. 2	0.750	1.865	1.083	3.753	1.000	3.162
Campsurus spp.	4.333	6.199	2.250	2.864	3.333	6.257
Campylocia sp.	0.583	1.730	0.583	1.443	0.167	0.389
Cloeodes auwe (Salles & Batista, 2004)	0.167	0.577	0.167	0.577	0.583	2.021
Cloeodes sp.	0.000	0.000	0.083	0.289	0.000	0.000
Coryphorus aquilus (Peters, 1981)	0.417	0.900	0.333	0.492	0.417	0.793
Cryptonympha copiosa (Lugo-Ortiz & McCafferty, 1998)	1.750	3.441	0.667	1.231	1.333	1.923
Farrodes sp.	0.750	2.301	1.417	2.906	0.167	0.389
Fittkaulus sp.	4.250	7.829	3.500	4.442	1.167	2.329
Hagenulopsis sp.	0.083	0.289	0.083	0.289	0.000	0.000
Harpagobaetis gulosus (Mol, 1986)	0.083	0.289	0.000	0.000	0.000	0.000
Hydrosmilodon gilliesae (Thomas & Peru, 2004)	0.167	0.389	1.000	2.296	0.083	0.289
Michophebia surinamenses (Savage & Peters, 1983)	0.667	2.015	0.167	0.389	0.167	0.389
Microphlebia sp.	1.917	3.528	3.250	6.538	0.167	0.577
Miroculis sp. 1	0.917	2.875	2.417	5.931	0.500	1.168
Miroculis sp. 2	0.917	1.379	2.667	4.638	0.417	0.900
Paracloeodes binodulus (Lugo-Ortiz & McCafferty, 1996)	0.000	0.000	0.000	0.000	0.167	0.577
Paramaka convexa (Spieth, 1943)	0.000	0.000	0.083	0.289	0.000	0.000
Simothraulopsis sp.	3.500	4.719	3.667	5.466	1.583	2.429
Thraulodes cochunaensis (Dominguez, 1987)	0.000	0.000	0.500	1.243	0.000	0.000
Traverhyphes sp. 1	0.167	0.389	0.250	0.622	0.083	0.289
Traverhyphes sp. 2	0.250	0.866	0.000	0.000	0.000	0.000
Tricorythodes hiemalis (Molineri, 2001)	1.250	3.720	1.667	4.008	0.417	1.165
Tricorythodes sp. 1	0.083	0.289	0.167	0.389	0.250	0.622
Tricorythodes sp. 2	0.000	0.000	0.000	0.000	0.083	0.289
Tricorythopsis sp.	0.083	0.289	0.083	0.289	0.000	0.000
Ulmeritoides flavopedes (Spieth, 1943)	2.250	3.980	2.833	4.282	1.583	3.260
Waltzoyphius roberti (Thomas & Peru, 2002)	0.917	1.564	0.500	1.000	0.333	0.492
Zelusia principalis (Lugo-Ortiz & McCafferty, 1998)	0.000	0.000	2.333	7.463	0.083	0.289

Appendix 2. Mean and standard deviation for species abundance of caddisflies during the sampling period (2007–2008) in the basin of Rio Suiá-Missu, Mato Grosso.

	Fall-water season		Dry season		Rainy season	
Especies	Mean	SD	Mean	SD	Mean	SD
Amazonatolica hamadea (Holzenthal & Pes, 2004)	0.250	0.866	0.167	0.577	0.083	0.289
Amphoropsyche sp.1	0.250	0.622	0.000	0.000	0.000	0.000
Amphoropsyche sp.2	0.083	0.289	0.167	0.389	0.000	0.000
Amphoropsyche sp.3	0.083	0.289	0.000	0.000	0.000	0.000
Cernotina sp.1	0.000	0.000	0.083	0.289	0.000	0.000
Cernotina sp.2	1.000	1.706	0.917	1.730	0.500	0.905
Chimarra sp.1	0.083	0.289	1.083	2.021	0.000	0.000
Chimarra sp.2	0.000	0.000	0.333	0.778	0.000	0.000
Cyrnellus sp.1	0.250	0.622	0.333	0.651	0.000	0.000
Cyrnellus sp.2	0.333	1.155	0.083	0.289	0.333	0.888
Cyrnellus sp.3	0.000	0.000	0.083	0.289	0.000	0.000
Leptonema amazonense (Flint, 1978)	0.083	0.289	0.000	0.000	0.000	0.000
L. maculatum (Mosely, 1933)	1.000	2.486	3.583	8.372	0.000	0.000
L. sparsum (Ulmer, 1905)	22.333	63.066	4.333	5.990	4.167	8.922
Macronema spp.	4.083	4.358	2.500	2.844	1.083	2.109
Macrostemum hyalinum (Pictet, 1836)	0.000	0.000	0.083	0.289	0.000	0.000
Macrostemum sp.1	0.167	0.577	0.667	2.309	0.083	0.289
Macrostemum sp.2	5.833	18.115	0.667	1.557	0.500	1.732
Marilia sp.1	0.417	1.443	0.250	0.866	0.000	0.000
Marilia sp.3	0.167	0.389	0.000	0.000	0.083	0.289
Nectopsyche sp.1	0.000	0.000	0.083	0.289	0.000	0.000
Nectopsyche sp.2	0.167	0.389	0.500	0.798	0.083	0.289
Nectopsyche sp.3	0.083	0.289	0.167	0.389	0.000	0.000
Nectopsyche sp.4	0.083	0.289	0.083	0.289	0.000	0.000
Nectopsyche sp.5	0.083	0.289	0.000	0.000	0.000	0.000
Nectopsyche sp.6	0.083	0.289	0.000	0.000	0.000	0.000
Oecetis sp.1	0.417	0.669	0.000	0.000	0.083	0.289
Oecetis sp.2	1.083	0.793	0.000	0.000	0.000	0.000
Oecetis sp.3	1.417	2.503	0.167	0.389	0.083	0.289
Oecetis sp.4	0.750	1.055	0.083	0.289	0.250	0.622
Oecetis sp.5	1.583	5.178	0.000	0.000	0.000	0.000
Oecetis sp.6	0.167	0.389	0.000	0.000	0.000	0.000
Oxyethira sp.1	0.083	0.289	0.000	0.000	0.000	0.000
Phylloicus sp.1	0.083	0.289	0.000	0.000	0.000	0.000
Polyplectropus sp.1	0.833	2.887	0.167	0.577	0.000	0.000
Polyplectropus sp.2	0.083	0.289	0.000	0.000	0.000	0.000
Polyplectropus sp.3	0.083	0.289	0.000	0.000	0.083	0.289
Smicridea (*R.*) sp.1	0.500	1.732	0.000	0.000	0.083	0.289
Smicridea (*R.*) sp.2	0.083	0.289	0.000	0.000	0.000	0.000
Smicridea (*S.*) *truncate* (Flint, 1974)	0.083	0.289	1.083	3.147	0.167	0.577
Smicridea (*S.*) sp.2	0.667	1.775	0.583	1.730	0.000	0.000
Smicridea (*S.*) sp.3	0.000	0.000	0.333	1.155	0.417	1.443
Smicridea (*S.*) sp.4	0.083	0.289	0.000	0.000	0.000	0.000
Smicridea (*S.*) sp.7	0.167	0.577	0.000	0.000	0.000	0.000
Triplectides sp.1	0.000	0.000	0.167	0.577	0.167	0.577
Wormaldia sp.1	0.000	0.000	0.083	0.289	0.000	0.000