Original Article

ASSESSMENT OF ECOLOGICAL WATER QUALITY ALONG A RURAL TO URBAN LAND USE GRADIENT USING BENTHIC MACROINVERTEBRATE-BASED INDEXES

AVALIAÇÃO DA QUALIDADE ECOLÓGICA DA ÁGUA AO LONGO DE UM GRADIENTE RURAL-URBANO UTILIZANDO ÍNDICES BASEADOS EM MACROINVERTEBRADOS BENTÔNICOS.

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ABSTRACT: Agricultural practices such as livestock grazing and tilling can result in soil erosion and runoff of fine sediments, nutrients (e.g. nitrogen, phosphorus, potassium) and pesticides, leading to degradation of aquatic environments. Urbanization is also responsible for a variety of impacts on fluvial ecosystems, including pollution by heavy metals, oil, domestic sewage and garbage. In this study, we evaluate the impact of land use on stream health of the Uberabinha river catchment. Overall, rural streams presented better ecological conditions than urban streams. Both species composition and abundance of benthic communities showed significant differences between rural and urban streams. Urban streams presented a higher dominance of Oligochaeta, Hirudinea and Gastropoda, bioindicators of poor water quality. Rural streams presented significantly greater richness and diversity. Compared to urban streams, rural streams presented a significantly higher number of Ephemeroptera, Plecoptera, Trichoptera, Odonata and Hemiptera taxa. Our analyses also showed congruence (high correlation) among the classical biodiversity metrics (Shannon-Wiener index – H', Pielou's measure of eveness – J) and monitoring parameters (% Ephemeroptera, Plecoptera and Trichoptera – EPT, Biological Monitoring Work Party – BMWP, bioindicator approach and Rapid Assessment Protocol – RAP, a habitat-based approach). Five from seven rural streams presented good water quality according to both BMWP and RAP and none of the urban streams presented good water quality. Our results show that the urban streams of Uberlândia municipality are poor ecosystems, and require improved management actions by environmental authorities. We also encourage that the riparian forest restoration and management carried out in the upper portion of Uberabinha River catchment to be extended to the urban area of the municipality.

KEYWORDS: Aquatic degradation. Species diversity. EPT. BMWP.

INTRODUCTION

Aquatic ecosystems have been impacted by numerous human activities, derived from both agricultural and urban activities (MANGADZE et al., 2013). Agricultural practices such as livestock grazing and tilling can result in soil erosion and runoff of large amounts of fine sediments, nutrients (e.g. nitrogen, phosphorus) and pesticides, leading to degradation of aquatic environments. Urbanization is also responsible for a variety of impacts including pollution by heavy metals, oil, domestic sewage and garbage (STEPENUCK et al., 2002). Additionally, the increase of catchment imperviousness enhances stormwater runoff and channel erosion, both contributing to river quality degradation (WALSH et al., 2001; MOORE; PALMER, 2005). These alterations in stream quality affect aquatic assemblages (ALLAN et al., 1997) by reducing overall diversity, increasing the number of tolerant organisms and decreasing the number of sensitive organisms, leading to the homogenization of communities (SMITH; LAMP, 2008; HEPP et al., 2010).

The high population density in urban areas typically results in larger modifications to the environment with high concentrations of pollutants (GODRON; FORMAN, 1983). Rural environments, which are sparsely populated, exhibit less built-up areas and lower concentrations of pollution (GODRON; FORMAN, 1983, MCDONNEL et al., 1997). Therefore, it is expected that aquatic ecosystems located in urban areas are more impacted by water pollution than aquatic ecosystems located in rural areas (MCDONNEL et al., 1997). Some studies have demonstrated that urbanization changes benthic macroinvertebrate community composition and abundance (PEDERSEN; PERKINS, 1986). RESH; JACKSON (1993) observed that impacts from human activities particularly affect aquatic insects of the Ephemeroptera, Plecoptera and Trichoptera (EPT) orders, often recognized as bioindicators due to their high sensitivity to organic pollution (e.g.

DOLÉDEC et al., 2006), whilst other organisms such as Oligochaeta, Mollusca and some Chironomidae genera (Insecta: Diptera) are acknowledged for their tolerance to organic waste (CALLISTO et al., 2001, ROLDÁN-PÉREZ, 2003).

Benthic macroinvertebrates have been widely used in biomonitoring programs (see ALBA-TERCEDO, 1996, CALLISTO et al., 2000). Some characteristics, such as high abundance in streams, small body size, short life cycle, and a wide range of responses to pollution favor their use in aquatic ecosystem monitoring (ROSENBERG; RESH, 1993, COUCEIRO et al., 2012). Their potential to assess water quality has been widely used through the application of different biomonitoring protocols, based on the premise that pollution tolerance differs among taxonomic groups (RESH et al., 1996). For instance, the Biological Monitoring Working Party (hereafter BMWP), a simple and adaptable index, has been widely implemented worldwide and adapted to regional peculiarities (e.g. ARMITAGE et al., 1983 – United Kingdom; JUNQUEIRA; CAMPOS, 1998 – Brazil; WYŻGA et al., 2013 – Poland; GUTIÉRREZ-FONSECA; LORION, 2014 – Costa Rica). In the BMWP, system families are assigned a score between 1 and 10 according to their sensitivity to pollution. The BMWP score is the sum of the values for all families present in the stream sample. The higher the score, the better the ecological condition of the stream (MASON, 2002). The macroinvertebrate sampling facilities, the index calculation and also its direct interpretation of the perturbation status of a particular stream makes BMWP easily manageable (ZAMORA-MUÑOZ et al., 1995).

Another convenient approach consists of habitat-based protocols – a visual inspection of structural habitat features. These are based on the assumption that certain river parameters such as land use, riparian vegetation, bank structure, water color and smell, among others, reflect the environmental quality of the fluvial habitat (i.e. river). In the habitat-based protocol approach, parameters are assigned a score according to stream ecological conditions. The higher the score, the better the stream's conservation status (CALLISTO et al., 2001). Besides the use of different protocols, it is worth noting that classical biological variables such as richness, abundance and evenness also provide relevant information about the status of water quality. Generally, impacted environments show low diversity values with a limited number of resistant species (MELO; HEPP, 2008).

The aim of this study was to compare the impact of rural and urban activities on stream health of the Uberabinha river catchment, using macroinvertebrates as water quality bioindicators. Commencing in 2009, the upper portion of the Uberabinha River catchment has been the subject of ongoing riparian forest restoration and management (rural zone of Uberlândia municipality) through the Buriti Program for Recovery of Riparian Forests, established by Uberlândia's Water and Sewerage Department (ANA, 2015). As a result, we expect that the conservation status of sites in the rural area to be better than those in the urban perimeter of the city. We tested the hypothesis that macroinvertebrate composition and abundance is different between urban and agricultural streams within the study area. We expect that rural streams, which we assumed to be less impacted by human activities, particularly by domestic and industrial discharge, have higher diversity index values (richness, H', J', higher percentage of EPT), lower abundance, lower percentage of Chironomidae and Oligochaeta, higher BMWP scores and also present better habitat conditions.

MATERIAL AND METHODS

The study was undertaken in the Uberabinha River catchment which is located in the geographical region of Triângulo Mineiro, Minas Gerais, Southeastern Brazil. It integrates the Paraná river catchment, represented by the Mesozoic age lithologies: sandstones of Botucatu Formation, basalts of Serra Geral Formation and rocks of the Bauru Group (NISHIYAMA, 1989). The local climate is tropical and classed as AW, megatermic, under the Köppen climate classification, with summer rains and winter drought (ROSA et al., 1991; GUIMARÃES-SOUTO et al., 2009). The Uberabinha catchment drainage is subject to different land uses. At the upper and lower course, land use is predominantly agricultural (BRANDÃO, 2002). At the middle course, land use is predominantly urban and includes Uberlândia, a city of about 600,000 inhabitants (IBGE, 2010).

We surveyed 12 streams in different regions of the Uberabinha river catchment, grouped according to main land use: five streams were located within the urban area and seven streams located within agricultural landscapes (Figure 1). The selected streams are subject to different levels of perturbation (BRANDÃO, 2002). The upper portion of the Uberabinha catchment comprises the rural streams Beija-Flor (hereafter BjFl), Rancharia (Ranch) and Bom-Jardim (BJd). Although surrounded by an agricultural matrix, they have been relatively preserved due to the presence of

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riparian forest and other typical vegetation types (native grass). Cabeceira do Lageado (Laged) is located in a relatively pristine ecological reserve (Reserva do Clube de Caça e Pesca), approximately 10 km from Uberlândia city. The streams Liso (Liso), Óleo (Oleo), Lagoinha (Lago), Buritizinho (Buri) and Fundo (Fund) are located at the urban perimeter of Uberlândia municipality. These streams are surrounded by built-up areas and subject to

various kinds of anthropogenic impacts, primarily due to the loss of vegetation along the stream banks and inflow of domestic sewage and garbage (BORGES et al., 2006; GUIMARÃES-SOUTO et al., 2011). At the lower portion of the Uberabinha catchment are located the rural streams Machados (Mach), Gordura (Gord) and Rio das Pedras (Pedr). These streams are surrounded by agriculture, pasture and riparian forest.
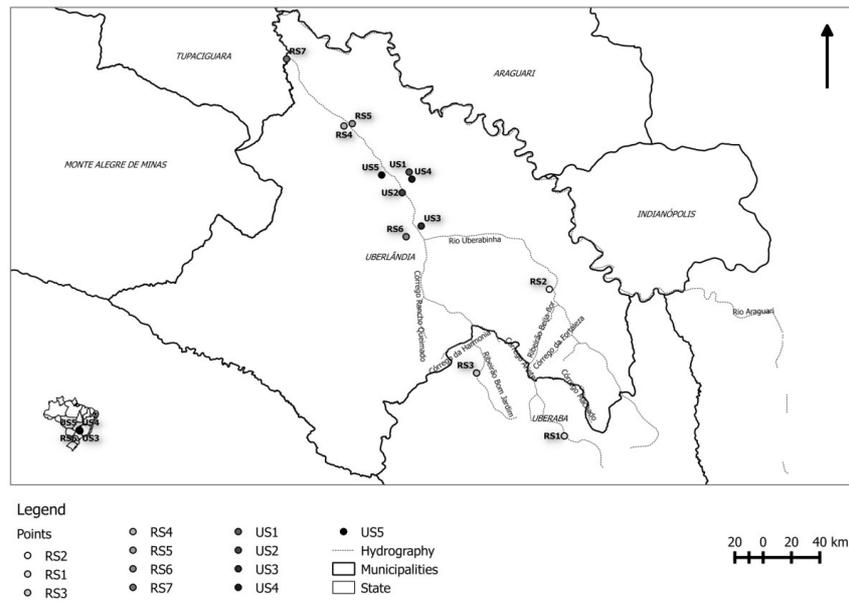

Figure 1. Uberabinha river catchment and the twelve sampling sites. Open circles – Rural Samples (RS); Black circles – Urban Samples (US). RS1 – Beija-Flor stream; RS2 –Rancharia stream; RS3 – Bom Jardim stream; RS4 – Machados stream; RS5 – Gordura stream; RS6 – Cabeceira do Lageado stream; RS7 – Rio das Pedras stream; US1 –Liso stream; US2 – Óleo stream; US3 – Lagoinha stream; US4 – Buritizinho stream; US5 –Fundo stream.

Benthic macroinvertebrate surveys were performed during the dry season (August and September/2010). At each stream, data was collected at the middle course, along a 100 m section of wadeable, running water. Three points were selected within this section and three substrate sub-samples were randomly collected using a Surber collector of 900 cm^2 with 0.25 mm mesh size. The three sub-samples of each point were pooled as one representative sample (see GUIMARÃES-SOUTO et al., 2011). Sampling was performed by disturbing the Surber delimited sediment area for one minute. Samples were bottled, labeled and fixed in a 10% formalin solution. At the laboratory, benthic macroinvertebrates were identified to the family level, except for Collembola (Order), Anellida (Subclass), Mollusca (Class) and Nematoda (Phylum). Specific identification keys were used (e.g. MERRIT; CUMMINS, 1984) and when

necessary, expertise confirmation was sought. For both the description of macroinvertebrate community structure and biomonitoring purposes, identifying macroinvertebrates to family level has been demonstrated to be as informative as genera-level data (WARWICK, 1988; BOWMAN; BAILEY, 1998; WAITE et al., 2004; JONES, 2008).

For each sampling site, different biodiversity indexes were calculated, including taxa richness, Shannon-Wiener diversity index (H') and the percentage of Ephemeroptera, Plecoptera and Trichoptera (EPT), Oligochaeta and Chironomidae (for details see Appendices). In order to compare a bioindicator-based framework to a habitat-based one, a BMWP index was calculated (Biological Monitoring Working Party – ALBA-TERCEDOR 1996), adapted for the Cerrado biome. Water quality classes were determined for different BMWP score

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intervals (very bad: ≤ 16; bad: 17 - 36; satisfactory: 37 - 63; good: 64 - 85; excellent: ≥ 86) (JUNQUEIRA; CAMPOS, 1998). A stream habitat approach was also used, also known as 'Rapid Assessment Protocol' (hereafter RAP). This protocol is a rapid and qualitative habitat assessment developed to describe overall quality of the physical habitat, and is based on a visual inspection of the site by incorporating several habitat attributes which are assigned numerical scores along a gradient of optimal to poor (EPA, 1987; HANNAFORD et al., 1997). The total score of the attributes results in three habitat classes (impacted: 0 - 40; altered: 41 - 60; natural: ≥ 61) (CALLISTO et al., 2002).

To test the hypothesis that macroinvertebrate composition and abundance were different among urban and agricultural streams, a Non-Parametric Multivariate Analysis of Variance was performed (PERMANOVA). Presence-absence records were used to assess effects on composition data. Abundance data was calculated as the average number of individuals at the three substrate sampling units and further log transformed. The rationale for transforming abundance data was to better estimate the contribution of rare and common families (ANDERSON, 2001). A Principal Component Analysis was also performed (PCA) to inspect collinearity between the diversity parameters and to examine associations between the diversity parameters and the surveyed streams. Analyses were carried out in the "vegan" package of the statistical environment R (R DEVELOPMENT CORE TEAM, 2008).

RESULTS

A total of 191,947 individuals, belonging to 50 taxa were sampled: Insecta (43), Gastropoda (3), Annelida (2), Collembola (1) and Plathyhelminthes (1). The most abundant families in both rural and urban streams were Chironomidae and Simuliidae. Ephemeroptera and Plecoptera were prevalent in rural streams while Gastropods were prevalent in urban streams (for further details see Appendices). Both the composition and abundance of benthic communities showed significant differences between rural and urban streams (PERMANOVA, F-ratio= 3.491, p=0.044, df=10, and F-ratio= 3.332, p=0.049, df=10, respectively for species composition and abundance).

According to BMWP biological scores, almost all rural streams presented "excellent" water quality, except the Machados stream, which presented "good" water quality, and Gordura, which presented "satisfactory" water quality. Urban streams differed widely according to the BMWP. Fundo stream presented "good" water quality; Liso, Óleo and Lagoinha streams were classified as "satisfactory", while Buritizinho was classified as "bad" according to the biological score (Figure 2).
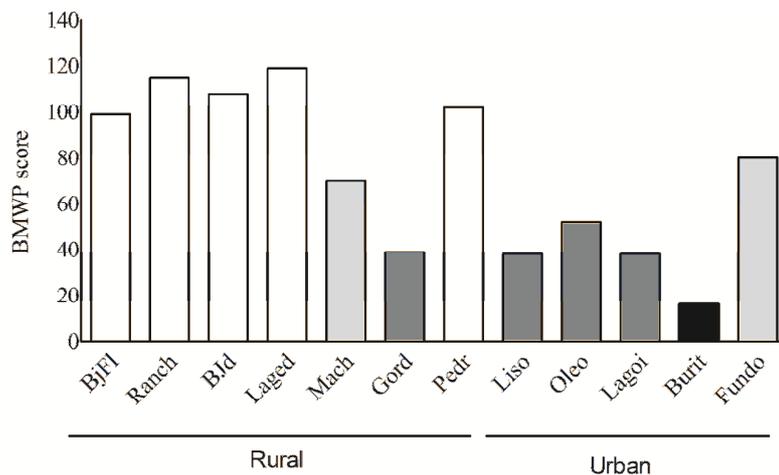

Figure 2. Assessment of environmental quality according to BMWP (bioindicator-based approach). Different colors represent different water quality according to BMWP criteria: White – excellent; Light-grey – good; Dark-grey – satisfactory; black – bad.

The application of RAP showed that in general, rural streams presented "natural" conditions, except for Machados, which was classified as "altered" and Gordura, which was classified as "impacted". Urban streams presented worse habitat conditions than rural ones. Liso, Lagoinha and Fundo streams were classified as

"altered". Óleo and Buritizinho were classified as   "impacted" (Figure 3).
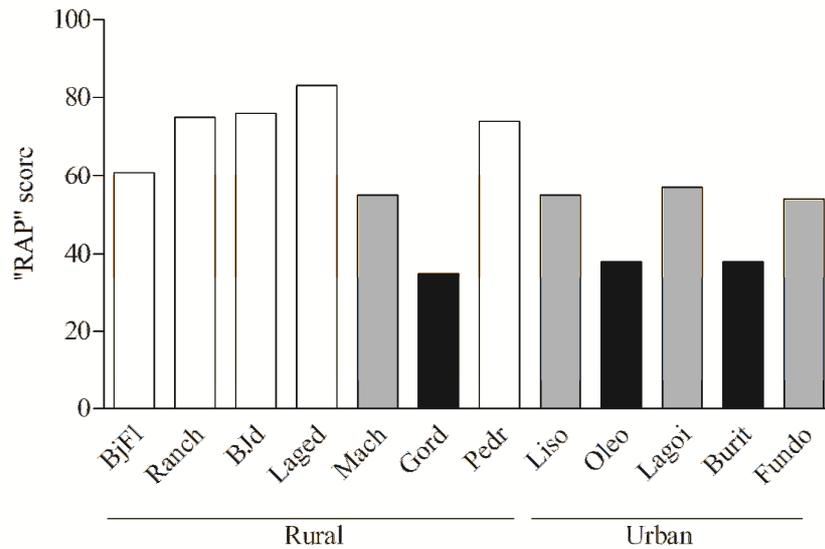

Figure 3. Assessment of environmental quality according to Rapid Assessment Protocol (habitat-based approach). Different colors represent different water quality, according to RAP criteria: White – natural streams; Grey – altered streams; Black – impacted streams.

The Principal Component Analysis (PCA) showed high correlation among the traditional diversity indices H' and J, the sensitive groups – % EPT – and the two biomonitoring approaches (BMWP and RAP) in axes construction (Table 2). In the first axis, the most important metrics were Oligochaeta percentage, with negative loading, and Shannon-Wiener diversity (H'), with positive loading. In the second axis, % Chironomidae was the most important variable, followed by J' (Table 1). Generally, rural streams were related to H', J', EPT, BMWP, Chironomidae and RAP, while urban streams were related to % Oligochaeta and abundance (Figure. 3). The first two axes explained 60.69 and 19.94% of the total variation respectively.

Table 1. Component loadings for the first and second axis of the Principal Component Analysis. The largest component loadings in each axis are in bold.

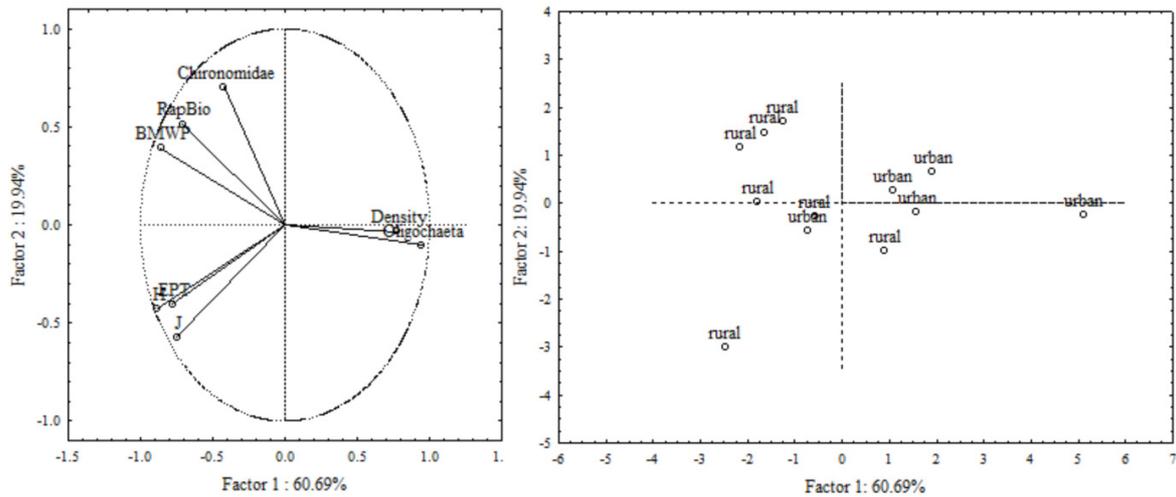| Variables	PC1	PC2
H'	**0.389**	0.338
J'	0.339	**0.453**
Abundance	-0.353	0.027
BMWP	0.389	-0.309
% Chironomidae	0.192	**-0.556**
% Oligochaeta	**-0.429**	0.082
% EPT	0.352	0.320
RAP	0.321	-0.407

Figure 3. Ordination diagram of the biodiversity metrics (left) and surveyed streams (right) by Principal Component Analysis (PCA).

DISCUSSION

Overall, our results showed that both community composition and abundance of macroinvertebrates were significantly different between rural and urban streams in the Uberabinha river catchment. In addition, rural streams showed higher BMWP and RAP scores, higher percentage of EPT and Chironomidae, higher richness, eveness and Shannon-Wiener diversity (richness, J', H').

Streams affected by both urban and agricultural activities undergo changes which influence the quality and availability of resources, as well as their ecological integrity, resulting in significant modifications to the structure and composition of the benthic community (BUSS et al., 2002; HEPP; SANTOS, 2009; MILESI et al., 2009). Urbanization may be more detrimental to aquatic ecosystems than agricultural practices as the high population density in urban areas typically results in high concentrations of pollution (STEPENUCK et al., 2002) from multiple sources, such as stormwater drainage, domestic and industrial sources (WALSH et al., 2001; MOORE; PALMER, 2005). Moreover, the absence of riparian vegetation, more frequent in the urban streams of the Uberabinha river catchment (BORGES et al., 2006; GUIMARÃES-SOUTO et al., 2011), increases vulnerability to runoff and excessive loading of nutrients and sediments, and reduces dissolved oxygen concentration (OMETTO et al., 2004). These impacts reduce the spectrum of conditions that most of the taxa are able to live within (RESH et al., 1988), reducing taxa richness to a few tolerant and generalist groups (COUCEIRO et al., 2007) and produces shifts in abundance patterns (BRASIL et al., 2014).

Benthic macroinvertebrates have been recognized as indicators of water quality because there is substantial variation in taxa response to a gradient of environmental conditions within the group (e.g. ARMITAGE et al., 1983). Sensitive groups such as EPT are usually prevalent in habitats with better environmental quality, as most EPT species are sensitive to water pollution (e.g. DOLÉDEC et al., 2006; BACEY; SPURLOCK, 2007; HEPP; SANTOS, 2009). In accordance with our hypothesis, we found a prevalence of Ephemeroptera and Plecoptera in rural streams. The high richness of EPT is related to the availability of habitats, which may be the situation experienced by rural streams less affected by anthropogenic impacts (MCDONNEL et al., 1997). The low percentage of EPT in some of the urban streams (e.g. Lagoinha and Buritizinho) indicates that these places are disturbed as EPT tend to decrease or even disappear in areas where sources of urban pollution are present (HEPP et al., 2010). In addition, there was a high dominance of Oligochaeta in some urban streams such as Buritizinho and Lagoinha, which are assigned poor water quality (LENAT; CRAWFORD, 1994). Oligochaeta is well adapted to these environments, since they feed on organic matter and can tolerate hypoxic conditions (GIERE et al., 1999; GUIMARÃES-SOUTO et al., 2009).

Some Chironomidae families are usually associated with impacted environments due to their capacity to tolerate very low levels of oxygen (NESSIMIAN, 1995; CALLISTO et al., 2001) and their rapid growth rate (JACOBSEN; ENCALADA, 1998). Thus, we expected a comparatively higher prevalence of Chironomidae in more impacted urban streams than rural ones. Contrary to our

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expectations, chironomids were ubiquitous in both urban and rural streams. This could be explained by their high abundance and diversity in almost every freshwater habitat, comprising species with numerous ecological niches and also differing in their behaviors, habitat use and feeding preferences (CALLISTO et al., 2001). In a comparative analysis of fine versus coarse taxonomic resolution in chironomid responses to environmental predictors, GREFFARD et al. (2011) showed that there was substantial variation in ecological response even among finely resolved taxa, suggesting a loss of ecological information in coarser taxonomic analysis such as family resolution, as used in our study. As different chironomid species vary in their sensitivities to environmental stressors, they may be more useful at finer taxonomic resolution, such as genera or species (CAREW et al., 2007).

However, the choice of taxonomic resolution for biomonitoring involves a compromise between the difficulty of identifying organisms at higher taxonomic resolutions and the loss of information at lower resolutions (MARSHALL et al., 2006). With the exception of Chironomidae, our overall results suggest that identification to the coarse taxonomic level is acceptable in bioassessment protocols (ARSCOTT et al., 2006; CHESSMAN et al., 2007). Even at the family level, it was possible to detect significant differences among rural and urban stream macroinvertebrate assemblages. This is in accordance with previous studies that have shown that the use of family level is sufficient in clarifying differences in streams exposed to different impact levels or distinct environmental conditions (BOWMAN; BAILEY, 1998; WAITE et al., 2004; MELO, 2005; HEINO; SOININEN, 2007).

Biodiversity indexes, such as richness and abundance, are influenced by environmental and biotic factors (BISPO et al., 2006). The present study showed differences in these indexes, presumably due to differences in habitat conditions caused by changes in land use. The highest taxonomic richness (see Appendix) was found in rural streams, possibly due to the presence of native riparian vegetation providing higher habitat heterogeneity (HEPP et al., 2010) and allochthonous matter input (LANGHANS et al., 2006). Otherwise, urbanization frequently causes changes in the water quality due to the removal of marginal vegetation, which allows an input increment of sediment, nutrients and pollutants (NESSIMIAN et al., 2008). As a consequence, intolerant taxa suffer extinction, leading to richness decreases and abundance increase of tolerant taxa (SMITH; LAMP 2008).

The estimation of biodiversity indexes relying on coarse taxonomic levels is problematic, especially for Shannon-Wiener (WU, 1982). The less precise the identification, the lower the diversity index values (JONES, 2008). Although high taxonomic resolution is desirable for biodiversity estimates, the ability to identify macroinvertebrate organisms from the Uberabinha river catchment to a lower taxonomic level than family would be very time consuming due to the required sampling effort. Furthermore, some studies have shown a strong correlation between species richness and richness based on both genus and family levels among macroinvertebrates (FURSE et al., 1984; HEWLETT, 2000; GUEROLD, 2000; MARSHALL et al., 2006). In a study about taxonomic resolution of freshwater macroinvertebrate samples from an Australian dryland catchment, Marshall et al. (2006) showed that identification to family level is highly correlated to species level identification (>0.90%), suggesting that estimates of richness based on family resolution may adequately reflect species richness. Thus, we assume that some level of information was lost due to sample identification to family level. However, considering our results and previous studies, we believe that the taxonomic resolution chosen was sufficient for comparison purposes between rural and urban streams.

In this study, community composition and abundance were significantly different between rural and urban streams, reflecting different ecological conditions related to land use. In addition, there is a convergence of the bioindicator-based approach (BMWP), the habitat-based approach (RAP) and the biodiversity metrics (H', J', % EPT), showing that rural streams are higher quality habitats than urban streams in the Uberabinha river catchment. Also, the high density of Oligochaeta in some urban streams may be evidence of organic enrichment (SCHENKOVÁ; HELEŠIC 2006) possibly caused by waste disposal. In spite of regular water and sewage provisions servicing the urban zone of Uberlândia municipality, our results show a disturbing picture regarding the environmental quality of Uberlândia´s urban streams. Therefore, we strongly encourage stakeholders (politicians, local scientists, engineers and water users) to partake in joint discussions to rethink management actions for the urban part as well the whole of the Uberabinha river catchment.

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RESUMO: Atividades agrícolas, como pecuária e cultivo de lavouras, podem levar a degradação dos ambientes aquáticos vizinhos, provocando erosão do solo e o carreamento de sedimentos finos, nutrientes (por exemplo, nitrogênio, fósforo, potássio) e pesticidas para os leitos dos rios. A urbanização também é responsável por uma variedade de impactos nos sistemas fluviais, incluindo a poluição por metais pesados, óleos, esgoto doméstico e lixo. Neste estudo, nós avaliamos o impacto do uso da terra na saúde da bacia hidrográfica do rio Uberabinha, utilizando macroinvertebrados bentônicos como bioindicadores da qualidade da água. Em geral, os córregos rurais apresentaram melhores condições ecológicas do que os córregos urbanos. Tanto a composição de espécies como a abundância da comunidade bentônica mostraram diferenças significativas entre os dois grupos. Os córregos urbanos apresentaram uma maior dominância de Oligochaeta, Hirudinea e Gastropoda, organismos indicadores de baixa qualidade ambiental. Os córregos rurais apresentaram maior riqueza e diversidade de grupos taxonômicos, tais como Ephemeroptera, Plecoptera, Trichoptera, Odonata e Hemiptera. Nossas análises mostraram congruência (alta correlação) entre os índices tradicionais de diversidade (índice de Shannon-Wiener - H ', equitabilidade de Pielou - J) e as métricas de biomonitoramento (% Ephemeroptera, Plecoptera e Trichoptera – EPT, Biological Monitoring Work Party – BMWP , índice baseado na composição taxonômica das comunidades e Rapid Assessment Protocol – RAP, abordagem baseada em características físicas do habitat). Cinco dos sete córregos rurais analisados apresentaram boa qualidade da água, de acordo com BMWP e RAP. Nenhum dos córregos urbanos apresentaram boa qualidade ambiental. Nossos resultados mostraram que córregos urbanos do município de Uberlândia possuem má qualidade ambiental. Nós encorajamos que o programa de recomposição manejo das matas ciliares dos córregos rurais do rio Uberabinha seja estendido aos córregos urbanos do município.

PALAVRAS-CHAVE: Degradação aquática. Diversidade de espécies. EPT. BMWP.

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Appendix

Table 1. Taxonomic composition and abundance, richness, Shannon-Wiener diversity (H') and Pielou's evenness (J') indexes of the surveyed streams. Seven rural streams are presented at the left side of the table. The urban streams are presented on the right side of the table (grey). BjFl - Beija-Flor stream; Ranch - Rancharia stream; BJd - Bom-Jardim stream; Laged - Cabeceira do Lageado stream; Mach - Machados stream; Gord - Gordura stream; Pedr - Rio das Pedras stream; Liso - Liso stream; Oleo - Óleo stream; Lago - Lagoinha stream; Buri - Buritizinho stream; Fund - Fundo stream. Abundance was calculated as the average number of individuals at the three substrate samples.

	Rural streams							Urban streams				
	BjFl	Ranch	BJd	Laged	Mach	Gord	Pedr	Liso	Oleo	Lago	Buri	Fund
Diptera												
Chironomidae	368	312	146.33	108.67	20.66	76.33	545.67	292	1142.7	2661.3	4000.3	44.66
Simuliidae	1560	125.66	20.33	0	18	0.66	20.33	159.66	33.66	84.33	4.33	9.66
Ceratopogonidae	1	7.33	4	0.33	0	0	3	1.66	2	0.33	0	0.33
Tipulidae	2.33	4	1.33	1.66	2	0	2.66	0.33	0.33	0	0.33	0
Empididae	0.33	1.66	0	0	0	0	2.66	7	54	0.33	0	0
Tabanidae	0	1	0	0	0	0	0	0	0	0	0	0
Sciomyzidae	0	0	0	0	0	0	0	0	0	0.33	0	0
Culicidae	0	0	0	0	0	0	0	0.33	0	0	0	0
Stratiomidae	0	0	0	0	0	0	0	0.33	0	0	0	0
Hemiptera												
Naucoridae	0.66	0.33	19.33	0	1.66	0	0.33	0	0	0	0	0
Helotrephidae	0.33	1	0.33	0.33	0	0	0	0	0	0	0	0
Veliidae	0	0	0	0	0.33	0	0	0	0	0.66	0	0
Gerridae	0	0.33	0	0	0	0	0	0	0	0	0	0
Belostomatidae	2	0	0	2	0	0	0	0	0	0	0	0
Pleidae	0	0	0	1	0	0	0	0	0	0	0	0
Coleoptera												
Elmidae	15	19.66	88	30	4	0.33	34.33	0	0.33	0	0	73.66
Psephenidae	0	0	0	0	0	0	0.66	0	0	0	0	0
Nototeridae	0	0	0.33	0	0	0	0	0	0	0	0	0

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Hydrophilidae	0	0.33	0	0	0.66	0.33	0	15	1	0.33	0.33	0
Trichoptera												
Hydropsychidae	3.33	3.33	6.33	32.66	5.33	0	36	0.33	127.67	0.33	0	16
Hydroptilidae	3.33	2.33	10.66	3	0	0.66	20	0.66	5	0	0	0.66
Polycentropodidae	1.66	1.33	2	2	0	0	0.33	0	2.66	0	0	0.33
Limnephilidae	0	1	0	0.66	0	0	0	0	0	0	0	0
Xiphocentronidae	0	0	0	0	0	0	1	0	0	0	0	0
Glossosomatidae	0	0	0.66	0	0	0	0	0	0	0	0	0
Leptoceridae	0	0	0.66	0	0	0	0	0	0	0	0	0
Anomalopsychidae	8.66	0	0	1	0.66	0	0	0	0	0	0	0
Philopotamidae	0	0	0	0.33	3	0	0.33	0	0	0	0	0
Hydrobiosidae	0	0	0	1	0	0	0	0	0	0	0	0
Sericostomatidae	0	0	0	0.66	0	0	0	0	0	0	0	0
Odontoceridae	0	0	0	0.33	0.33	0	0	0	0	0	0	0
Ephemeroptera												
Baetidae	464.33	19.66	23.33	2.66	44.33	1.66	72	0	0	0.66	0	1.33
Leptophlebiidae	49	3.66	33.33	3.33	0	0	28	0	0	0	0	1
Leptohyphidae	98.33	2.66	5.66	2.66	0	0	10.33	0	0	0	0	0.66
Euthyplocidae	0	0.66	0	1.33	0	0	1.66	0	0	0	0	0.33
Ephemeridae	0	0	0.66	0	0	0	0	0	0	0	0	0
Plecoptera												
Perlidae	42.66	1.66	0.33	0.66	1.66	0	1.33	0	0	0	0	11.33
Odonata												
Coenagrionidae	0	4.33	0	0.66	0	0	0	0	0	0	0	0
Calopterygidae	0	0.66	0.66	0.66	0	0	0	0	0	0	0	0.33
Libellulidae	2	1	0.66	0.66	0.66	0	0	0	0.33	0	0	1.66
Gomphidae	0	1.33	0.33	0	0.33	0.66	0	0	0	0	0	0
Corduliidae	5	0	1.66	0.33	1	2.66	2	0	0.66	0	0	0

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Lepidoptera												
Pyralidae	1	0	0	1.66	0	0	7.66	0	0	0	0	0
Collembola	0	0	0	0.33	0	0	0	0	0	0.66	0	0
Anellida												
Oligochaeta	42.33	39	66	1.33	10	16.33	4	420.66	313	4139.7	45238	1.66
Hirudinea	1.33	0	2.33	0	0	1.66	0	8.66	45	0.33	20.66	0
Platyhelminthes												
Planaridae	0	0	0	0	0	3	0	0	0	0	0	0
Gastropoda												
Planorbidae	0	0	0	0	0	34.66	0	3	34	0	6	0
Lymnaeidae	0	0	0	0	0	3	0	64	21	0	1.33	0
Hydrobiidae	0	0	0	0	0	0	0	0	0.33	0	0	0
Richness	21	25	24	28	17	13	21	14	17	12	8	15
H'	1.25	1.42	1.74	1.5	2.41	1.4	1.36	1.03	1.01	0.72	0.28	1.43
J'	0.43	0.49	0.67	0.56	0.97	0.66	0.5	0.45	0.48	0.4	0.16	0.62