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DEFINING AND TESTING TARGETS FOR THE RECOVERY OF TROPICAL STREAMS BASED ON MACROINVERTEBRATE COMMUNITIES AND ABIOTIC CONDITIONS

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ABSTRACT

Here, we set target values to measure the ecological improvement of streams, based on invertebrate communities, riparian vegetation, instream habitat conditions and water chemistry. The study area is a large tropical catchment (Rio das Velhas, Minas Gerais, Brazil) affected by pastures, mining areas and a large urbanized area but also includes natural protected areas. Two stream types were found in the catchment, based on stream size, elevation, climate and geology with significantly different macroinvertebrate communities. In spite of a marked wet/dry seasons' climatic pattern, that does not result in the segregation of communities. Four classes of global degradation (IV—bad to I—good condition) were defined based on the available abiotic information, corresponding to a gradient in structure and biotic metrics of macroinvertebrate communities, matching the current knowledge on taxa sensitivity to pollution and general disturbance. Class I corresponds to target conditions to be achieved under restoration programmes. Using this approach, we were able to detect an improvement of abiotic conditions in four urban streams that benefited from enhancement measures in 2007–2008. However, invertebrate communities improved clearly in only one site (biotic metrics and community structure). Our study highlighted that good water quality alone is not enough and that only the combined effect of water quality, riparian vegetation and instream habitat condition enhancement resulted in the improvement of invertebrate communities. An important limiting factor for macroinvertebrate communities' recovery may be the distance to source populations. We concluded that the combined use of biological and abiotic target values for measuring the recovery of streams is needed to fully achieve an ecological restoration. This approach can also be valuable in the regular monitoring of streams to assess stream degradation. Target values based on other biological elements, such as fishes and algae, and functional processes could also contribute to define more global and realistic goals. Copyright © 2013 John Wiley & Sons, Ltd.

KEY WORDS: streams; tropical; recovery; macroinvertebrates; urban streams

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INTRODUCTION

Land use, biotic invasions, climate change and their interactions are causing a fast decline in riverine biodiversity (Sala *et al*., 2000). Anthropogenic disturbances are particularly severe in tropical areas because of the highly populated cities with poor sewage and waste treatment, aquifer overexploitation, the cutting of native vegetation for large-scale pastures and crops, and large mining areas (Gladwell and Sim, 1993).

It is thus urgent to prevent further degradation and improve the quality of already degraded ecosystems. In response to that need, there was, in the last decade, an exponential increase of restoration projects in temperate regions of North America, Europe, Japan and Australasia (Bernhardt *et al*., 2005; Lake *et al*., 2007; Miller *et al*., 2010). River restoration should consider various aspects of the ecosystem, such as the water quality, water and sediment transportation, channel morphology and dynamics, hydrological regime, and the aquatic fauna and flora (Tánago and Jalón, 2001). Projects on ecological restoration are also based on the paradigm that increasing habitat heterogeneity promotes restoration of biodiversity through the increase of diversity, density and/or biomass of aquatic organisms (Purcell *et al*., 2002; Bernhardt *et al*., 2007; Miller *et al*., 2010; Palmer *et al*., 2010). The improvement of the habitat heterogeneity can been made through addition of boulders or large woody debris or channel reconfiguration to add meanders and artificial riffles (Palmer *et al*., 2010).

Among streams needing restoration, urban streams are a particular case. These streams are located in highly urbanized areas, which are profoundly impacted by human activities and dense infrastructures (e.g. roads and sewer lines) that usually transport high sediment and pollutant loads, and imperviousness is often their main problem (Walsh

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et al., 2005a; Bernhardt and Palmer 2007). The restoration of these streams is thus more expensive and more difficult than that in less populated catchments, and they will probably never return to an initial state before human intervention (Wade *et al.*, 1998). It is therefore important to consider the restoration of these streams in a realistic way and integrate it in a broader catchment management strategy (Bernhardt *et al.*, 2007).

To assess the effectiveness of restoration measures on a given water body, restoration should have measurable goals, and post-project evaluation must be incorporated into the initial design (Kondolf and Micheli, 1995; Palmer *et al.*, 2005). Yet, only ≈15–30% of restoration projects include post-project monitoring (Bernhardt *et al.*, 2005, 2007; Miller *et al.*, 2010), and only a small part includes the response of macroinvertebrates (e.g. Larson *et al.*, 2001; Pretty *et al.*, 2003; Thompson, 2006; Selvakumar *et al.*, 2010), as initially those projects targeted mainly the improvement of fish communities (Miller *et al.*, 2010). The improvement of macroinvertebrate communities' condition should however be considered an essential element in post-project evaluation, as they are important elements of the ecosystem, with a high diversity and abundance, found in all river sections, with key roles in the ecosystem functioning (e.g. decomposition, productivity, nutrient cycling and energy transference; Kenney *et al.*, 2009) and have a long history as bioindicators, as they can integrate and reflect environmental changes during their life cycles (De Pauw and Vanhooren, 1983; Barbour *et al.*, 1999).

Contrary to temperate areas, it is difficult to find literature on ecological restoration of streams and rivers of tropical regions, even though some of those regions encompass large urbanized areas and plantations and simultaneously important hotspots for biodiversity. Tropical freshwater ecosystems are different from the temperate ones in several aspects such as seasonal variation (only two seasons: dry and wet), community structure (e.g. lower proportion of shredders and higher proportion of predators; Moulton and Magalhães, 2003; Gonçalves *et al.*, 2006; Wantzen and Wagner, 2006) or functional processes (e.g. decomposition; Gonçalves *et al.*, 2006). Therefore, differences in the restoration planning and results are also expectable. On the other hand, the process of restoration of tropical urban streams may not be very different from that of temperate streams as the 'urban syndrome' should overcome the natural differences in climate, geology and others (Bernhardt *et al.*, 2007). To clarify these issues, more studies on restoration of tropical streams are necessary and timely.

Here, we propose the use of the reference condition approach (RCA; Reynoldson *et al.*, 1997; Stoddard *et al.*, 2006) to measure the improvement of the ecological quality of tropical urban streams to which enhancement measures were applied. The RCA has been used as the basis for assessing the quality status of streams and rivers by comparing the actually observed values of biological communities with those expected from physically comparable reference sites (in minimally or least disturbed conditions). We based our system in the hypothesis that a progressive reduction of disturbance should result in a progressive improvement of abiotic conditions (water quality and hydromorphology), which should be reflected in the improvement of macroinvertebrate communities (i.e. higher richness and diversity, and presence of sensitive taxa).

Therefore, here, we have two main goals: (i) the development a system for the evaluation of ecological improvement of streams of a tropical catchment (Rio das Velhas, Minas Gerais, Brazil) based on abiotic conditions (water chemistry, hydromorphology, riparian vegetation and land use) and macroinvertebrate communities, and (ii) test the approach by evaluating the effect of enhancement measures taken in 2007–2008 in four urban streams (that included water decontamination, bank clearance from human constructions and decrease of sediments runoff; Macedo and Magalhães, 2010). Additionally, we tested the influence of tropical seasonal patterns (dry/wet seasons) in the macroinvertebrate communities in order to determine if different seasons should be considered in such approach.

METHODS

Study area

The catchment of Rio das Velhas (Figure 1), state of Minas Gerais (southeast Brazil: 17°15′–20°25′S and 43°25′–44° 50′W), is the largest sub-catchment of river S. Francisco with a length of 761 km covering 29173 km^2 (Pompeu *et al.*, 2005; Moreno *et al.*, 2010). The climate is tropical humid with dry winters and wet summers and mild temperatures all year round (18–24 °C). There are climatic differences over the catchment, with lower precipitation and higher temperatures in the north than in the mountainous regions in the south and east (Conselho Nacional de Meio-Ambiente, 1992).

Rio das Velhas is the most polluted large river of Minas Gerais state (Pompeu *et al.*, 2005; Maillard and Santos, 2008). In its upper section, the main impacts are mining (with exploitations since the 16th century) and deposit areas of iron ore. Belo Horizonte Metropolitan area (≈4.5 million inhabitants; Ferreira *et al.*, 2011) is also located in the upper section of the basin. In the middle section, the main impacts are cement and lime industries and pastures. However, there are also preserved areas such as the national park of Serra de Cipó. Finally, in the lower section, aquaculture, pastures, row crops, irrigated land and small urban centres are the main sources of disturbance (Pompeu *et al.*, 2005; Moreno *et al.*, 2010).

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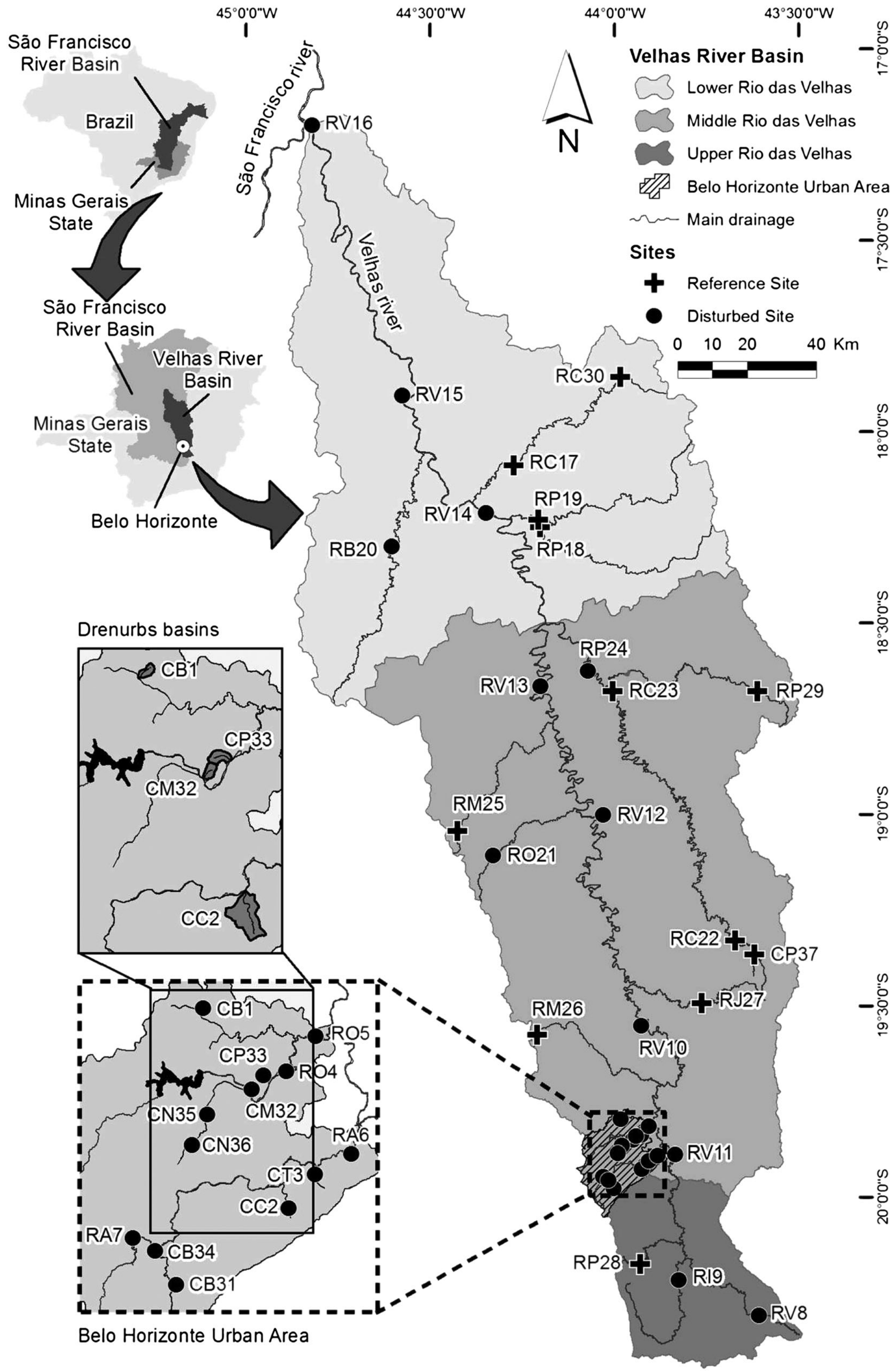

Figure 1. Map of Rio das Velhas catchment with the reference and disturbed study sites and respective codes. The square indicates the Belo Horizonte metropolitan region and respective streams

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Study sites and environmental characterization

The 37 study sites selected are distributed throughout the catchment of Rio das Velhas (17 in the upper, 13 in the middle and 8 in the lower sections), covering its natural variability (Figure 1). Twelve sites are considered as reference (least disturbed) based on previous abiotic and biotic information (e.g. Conselho Nacional de Meio-Ambiente, 2005; Paz *et al*., 2008; Moreno *et al*., 2010; Ferreira *et al*., 2011; Figure 1). These sites are located in protected areas or in low impact areas. The other 25 sites have different levels and sources of disturbance such as cutting of riparian vegetation, introduction of exotic species, water abstractions and low water quality, and changes in natural land use in the banks (pastures and other) and in the drainage areas. The worse conditions are found among the 13 sites located in urban streams of Belo Horizonte (Moreno *et al*., 2010). These sites have been affected by litter discharge, constructions in the banks, untreated sewage discharge, cutting of riparian vegetation, sand extraction and instream bed alterations (e.g. high sedimentation and bed canalization, and small weirs). Among those, there are four urban streams subjected to enhancement interventions between 2007 and 2008, which were also monitored before and after that period.

Fifteen environmental variables data were measured for each site (Table 1). Among those, eight are typological variables, independent from anthropogenic activities (e.g. geology, elevation and climate; obtained from Geographic Information Systems),

Table I. Environmental variables measured for each study site and respective units, description and ranges found in the study sites

Variables	Descriptions, units and transformations	Range
Typological variables		
Stream order	Strahler system (Strahler, 1951)	1–8
Distance to source	m (log_{10})	461–802736 m
Air temperature	°C (intervals of annual means; log_{10})	(<19)–(21–22) °C
Precipitation	mm (intervals of annual means)	(1000–1200)–(1200–1500) mm
Elevation	m	489–1240 m
Slope	% (to 500 m upstream; sqrt)	0–15%
Geology	Site-dominant geology	e.g. granite, schist, quartzite and sand deposits
Geomorphology	Large geomorphological formations that include the site	e.g. S. Francisco depression, Complex of Bação and Espinhaço Mountain Range
Disturbance variables		
Conductivity	µS/cm (measured *in situ* with portable YSI, Yellow Springs, Ohio; log_{10})	µS/cm
Total P	mg/L (measured in laboratory; Strickland and Parsons, 1960; $log_{10}+1$)	0.001–5.400 mg/L
Total N	mg/L (measured in laboratory; Mackereth *et al*., 1989; $log_{10}+1$)	0.028–141.600 mg/L
O_2	mg/L (measured in laboratory; Wetzel and Likens, 1991; sqrt)	0.0–15.5 mg/L
TDS	mg/L (measured *in situ* with portable YSI, Yellow Springs, Ohio; log_{10})	0.0–2671.0 mg/L
Turbidity	UT (measured *in situ* with portable YSI, Yellow Springs, Ohio)	0.2–1105.0 UT
Land use in the site banks	Dominant land use in the banks at the sampling site ($log_{10}+1$)	Scores: 4 = natural vegetation; 2 = pasture, agriculture, monoculture, reflorestation; 0 = urban, industrial, commercial areas
Riparian vegetation integrity	Integrity of the riparian vegetation at the site ($log_{10}+1$)	Scores: 5 = >90% native vegetation; no evidence of deforestation; 3 = 70–90% native vegetation; minor deforestation; 2 = 50–70% native vegetation; obvious deforestation; 0 = <50% native vegetation; strong deforestation
Habitat global score	Total score (obtained from the application of the habitat protocol described in Callisto *et al*. (2002) and adapted from EPA (1987) and Hannaford *et al*. (1997); final scores ranging 0–100; log_{10})	6.0–90.8

TDS, Total Dissolved Solids.

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and nine are disturbance variables, as they may translate the human disturbance over the sites (e.g. dissolved oxygen, total nitrogen, riparian vegetation integrity, land use in the banks and a habitat index; Table 1). The habitat index (Callisto *et al.*, 2002) is an adaptation of the Environmental Protection Agency of Ohio (EPA (Environmental Protection Agency), 1987) and Hannaford *et al.* (1997) protocols, and is based on 22 variables traducing habitat diversity and physical and hydromorphological features.

Enhancement measures

The DRENUBS programme (Programa de Recuperação Ambiental de Belo Horizonte) aimed to improve the environmental quality of the urban streams located in Belo Horizonte city through (i) water decontamination through the sewage treatment, (ii) reduction of floods through bank clearance from human constructions, and (iii) sediment runoff through control of erosion and bank revegetation (Macedo and Magalhães, 2010). These enhancement measures were taken in 2007–2008 in the four urban streams (hereafter called DRENUBS streams) analysed in the present study. A network of urban sewage collectors, retention basins and treatment plants was built to improve water quality and covered all studied sites. The banks and channel were left as natural as possible, with reconstruction of the riparian vegetation and bank reinforcement only where it was needed. The stabilization of banks was made whenever possible with permeable and natural materials, such as rocks and stones (Macedo, 2009). No alterations were carried out at the level of instream morphology and habitats.

Biological samples

The 37 stream sites were sampled for benthic macroinvertebrates between end of 2003 and 2010 (16 sites) or between beginning of 2004 and 2010 (21 sites). Sampling was carried out four times a year, in the beginning (October/November) and end (February/March) of the wet season and beginning (May/June) and end (August/September) of the dry season, avoiding major floods or lower water periods. At each site, three Surber samples (0.09 m^2, 0.25 mm mesh) were collected in the most representative habitats. A total of 2567 Surber samples were finally analysed in this study.

The animals were preserved in formalin 10% after collection and in ethanol 70% after sorting. They were counted and identified mainly to family level (Pérez, 1988; Merritt and Cummins, 1996; Wiggins, 1996; Pés *et al.*, 2005; Mugnai *et al.*, 2009). This taxonomic resolution is considered enough in large-scale monitoring and is the most common taxonomic level used in Europe, Australia, USA and other countries also for practical reasons (faster, less errors and easier to use an homogenous level of identification) even though some authors have found that a higher taxonomic resolution such as genus or species increases the level of sensitivity to human disturbance and is especially relevant to detect small quality changes (e.g. Bailey *et al.*, 2001; Waite *et al.*, 2004; Feio *et al.*, 2006). For the study region, family level is also the safer approach, as the aquatic invertebrate taxonomy is not identically well studied for all groups, and for such large database, the use of family level avoids potential mismatches in identifications over time. Data were finally converted into number of individuals per square meter for data analyses.

Data analysis

The mean monthly precipitation and temperature in the sampling periods, at each site, was plotted to confirm the seasonal pattern of dry and wet seasons' periods (historical data provided by Brazilian National Water Agency, Agência Nacional de Águas, 2011, and Brazilian National Meteorological Institute, Instituto Nacional de Metereologia, 2011). To check for identical biological patterns (similar distances between sites, or similar groups of sites or segregation patterns), a multidimensional scaling analysis (MDS) was performed based on macroinvertebrate data of reference sites (to exclude the interference of anthropogenic impacts) averaged by dry and wet seasons (fourth root transformation; Bray–Curtis coefficient), and an analysis of similarities (ANOSIM, PRIMER 6, Primer-E Ldt, Plymouth, U.K.) was used to check for statistical differences and within groups variability.

Because natural environmental conditions, such as climate, geology and elevation, vary throughout the catchment and are expected to structure macroinvertebrate communities (e.g. Poff and Ward, 1990; Petts, 2000; Reynoldson *et al.*, 2001; Chessman, 2004; Feio *et al.*, 2009; Hawkins *et al.*, 2010), our approach should also take those factors in consideration. Therefore, to establish appropriate targets for recovery, abiotic stream types (groups of sites with similar abiotic characteristics) were defined through a hierarchical agglomerative clustering analysis with group-average linking (Euclidean distance) of all sites, based on stream order, distance to source, air temperature, precipitation, elevation, slope, geology and geomorphology (Table 1). To characterize the natural environmental characteristics of the types and set guideline values for future sites, we calculated the average values (± standard deviation, SD) for each environmental variable used in the cluster analysis.

The typological division of the sites was verified for its biological relevance, with reference sites only: macroinvertebrate composition was tested for significant differences with a PERMANOVA test (fourth root transformation; Bray–Curtis similarity; unrestricted permutation).

To analyse the degradation gradient among the study area, we performed a PCA analysis (variables normalized) based on all study site disturbance data, after removing the data of post-DRENUBS enhancement measures for sites CB1, CC2, CM32 and CP33 (Table 2). The gradient was further divided into four intervals based on PCA best

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Table II. Urban stream abiotic characterization (pre/post-enhancement measure values for disturbance variables)

Variables	CB1	CC2	CM32	CP33
Stream order	1	2	3	1
Distance to source (m)	517	916	2135	416
Air temperature (°C)	19–21	19–21	19–21	19–21
Precipitation (mm)	1200–1500	>1500	1200–1500	>1500
Elevation (m)	804	892	991	795
Slope (%)	15	11.6	3.2	4.2
Geology	Gneiss	Schist	Schist	Gneiss
Geomorphology	BH complex	BH complex	BH complex	BH complex
Conductivity (μS/cm)	575/519	439/426	419/373	556/332
Total P (mg/L)	2.661/0.078	1.587/0.235	0.314/0.067	2.201/0.065
Total N (mg/L)	24.070/0.084	14.930/0.375	5.033/0.151	23.600/0.165
O_2 (mg/L)	3.8/6.8	3.01/6.6	3.0/6.1	1.2/7.7
TDS (mg/L)	506/284	459/231	476/209	476.0/178.7
Turbidity (UT)	165.0/2.1	125.0/7.8	36.0/30.6	76.0/27.7
Land use in the site banks	4/4	4/4	3/4	4/4
Riparian vegetation integrity	4/3	4/4	4/4	4/4
Habitat global score	31.7/45.3	35.0/57.0	42.5/39.3	31.0/39.7

explicative axis. These intervals corresponded to the four global degradation classes, from the worse (IV) to the maximum condition (I). Classification systems with the class equivalent to reference condition and three degradation classes are widely used by several assessment methods based on invertebrate communities (e.g. Reynoldson *et al*., 1995; Simpson and Norris, 2000; Feio *et al*., 2007). Here, the first class, good condition (I), includes all reference sites and was used to define target values for the full recovery of streams. The remaining classes have approximately equivalent intervals on the PC1 axis. From here, the mean (± SD) for each disturbance variable was calculated for each class, based on the sites contained in each interval. These values (mean + SD–mean − SD) set the limits for each class.

Based on the sites included in each PCA class, we performed a SIMPER analysis (similarity/distance percentages; fourth root transformation) to determine the most representative taxa (up to 90% of cumulative percentage) within degradation class, for each river type. The SIMPER examines the contribution of each taxa to average resemblances (Bray–Curtis similarity) between sample groups weighting both frequency and abundance. If the average abundance of a taxon is below 1, this taxon was not considered in future determinations of degradation class for new samples.

Additionally, we determined for each class, the mean value (± SD) for the metrics used in the multimetric index currently applied in the catchment (Ferreira *et al*., 2011), as these values could be also a simple indicator of stream recovery. The metrics included are as follows: total family richness as an indicator of biodiversity; % Oligochaeta, % Chironomidae, % Chironomidae + Oligochaeta (CHOL) as indicators of tolerant taxa; % Ephemeroptera, Plecoptera and Trichoptera (% EPT), the most sensitive taxa; % Collectors-Gatherers; and the total score of the biotic index BMWP-CETEC. This index is an adaptation of the original Biological Monitoring Working Party (Armitage *et al*., 1983) to Rio das Velhas catchment (Junqueira *et al*., 2000) based on the tolerance of invertebrate families to organic contamination.

Finally, for the urban streams (sites CB1, CC2, CM32 and CP33), we analysed the effect of restoration measures in the macroinvertebrate communities by determining their position in the degradation classes before and 2 years after the application of recovery measures (2010) in the three aspects. For (i) abiotic values, the degradation class was calculated by determining the closer interval (mean ± SD) and the worse situation in case of superimposition of values, for each chemical (average of the positions for each variable, e.g. total N, O_2,), land use, habitat and riparian vegetation variable, and, finally, averaging those classifications. For (ii) communities' composition (taxa and average abundance): the degradation class of a site was given by the Bray–Curtis similarity between the communities defining each class and the communities of that site. Finally, using (iii) biological metrics, the degradation class attributed had the closest to the mean values for a given urban site and the worse situation in case of superimposition of values.

All multivariate data analyses were performed with the software Primer 6 and PERMANOVA.

RESULTS

Ninety-one different taxa were found in our database. Besides the Oligochaeta, Decapoda, Isopoda, Hydracarina and Tricladia, not discriminated at a lower level, there were

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7 families of Ephemeroptera, 2 of Plecoptera, 13 of Trichoptera, 15 of Diptera, 12 of Coleoptera, 9 of Odonata, 4 of Heteroptera, 1 of Megaloptera, 6 of Hemiptera, 9 of Gastropoda and 2 of Lepidoptera.

Monthly precipitation during the period 2003–2010 ranged from ≈0 (dry season) to 757 mm (in wet season), with average annual amplitudes for the study period between 619 (site with highest variation) and 372 mm (site with lowest variation) at the study sites that corresponds to the expected pattern for a tropical climate. However, this pattern is not accompanied by shifts in macroinvertebrate communities, judged from MDS ordination (not shown) and ANOSIM test (Global $R = 0.053$; $p >> 0.05$; 999 permutations).

The hierarchical agglomerative clustering analysis (group-average linking; Euclidean distance; Primer 6) revealed two stream types (type 1 with 18 sites and type 2 with 19 sites; Figure 2 and Table 3). Type 1 were larger streams (order 6.3 ± 1.5) located in the lower and middle catchment sections, with lower elevation (596 ± 87 m; slope 0.86 ± 0.78%), higher mean annual temperature (22.0 ± 1.2 °C) and lower precipitation (1122 mm ± 100) than type 2 streams. The dominant geology is the siltstone and shale. Type 2 streams were smaller (stream order 3.4 ± 1.9), located mainly in the upper section of the catchment, at higher elevations (866 ± 126 m; slope 4.71 ± 3.98%), with lower mean annual temperature (20.5 ± 0.9 °C) and higher precipitation (1342 mm ± 154). The type 2 streams run over a mixture of geological elements, such as schist, metarenite, gneiss and quartzite. The two types were significantly different (PERMANOVA: Pseudo-*F*: 2.783; *p* (perm): 0.005; 998 permutations) in terms of macroinvertebrate composition.

The first axis of the PCA based on disturbance data explained alone 80.7% of the variation, while together, axis 1 and axis 2 explained 89.2% (Figure 3). Sites are distributed over PC1, and all disturbance variables (except for turbidity) contributed similarly to this distribution, with a clear gradient from reference sites (e.g. CP37 and Rp29) to most disturbed sites, including the urban streams (e.g. CP33, CB1 and CT3). Table 4 indicates the mean value of each disturbance variable for each degradation class (D).

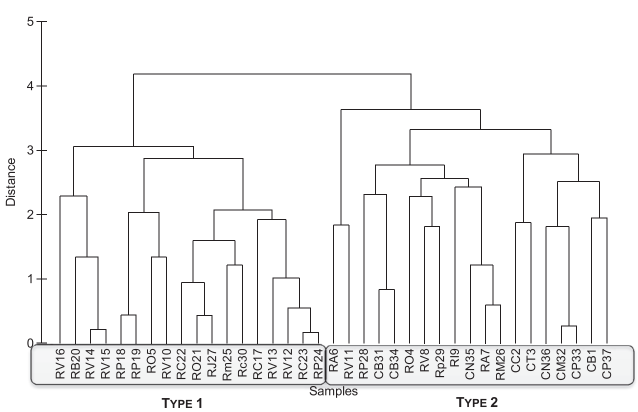

Figure 2. Hierarchical agglomerative clustering analysis with group-average linking (Euclidean distance; Primer 6), showing two main abiotic stream types within the catchment

Table III. Study site distribution by type, reference versus non-reference and degradation class attributed after PCA analysis

Sites	Type	Ref = 1/non ref = 2	Level
RC17	1	1	I
RP18	1	1	I
RP19	1	1	I
RC22	1	1	I
RC23	1	1	I
Rm25	1	1	I
RJ27	1	1	I
Rc30	1	1	I
RV8	2	1	I
RM26	2	1	I
RP28	2	1	I
Rp29	2	1	I
CP37	2	1	I
RV14	1	2	II
RV15	1	2	II
RV16	1	2	II
RB20	1	2	II
RO21	1	2	II
RP24	1	2	II
RI9	2	2	II
RV11	2	2	II
RV10	1	2	III
RV12	1	2	III
RV13	1	2	III
CB31	2	2	III
RO5	1	2	IV
CT3	2	2	IV
RO4	2	2	IV
RA6	2	2	IV
RA7	2	2	IV
CB34	2	2	IV
CN35	2	2	IV
CN36	2	2	IV
CB1	2	2	IV[a]
CC2	2	2	IV[a]
CM32	2	2	IV[a]
CP33	2	2	IV[a]

[a]Urban streams that suffered enhancement measures (DRENUBS programme): the degradation level refers to the period before interventions.

Twenty-three taxa of type 1 streams and 14 taxa in type 2 streams were relevant in the distinction between degradation classes (Table 5). In both types, some of those taxa are absent from D IV (as, for example, Polymitarcyidae, Isotomidae, Philopotamidae, Dryopidae and Leptophyidae); some of these taxa reappear in D III (e.g. Notonectidae and Isotomidae in type 1). Among the most tolerant groups (e.g. Chironomidae and Oligochaeta), maximum abundance is reached in D III, while under poorest quality conditions

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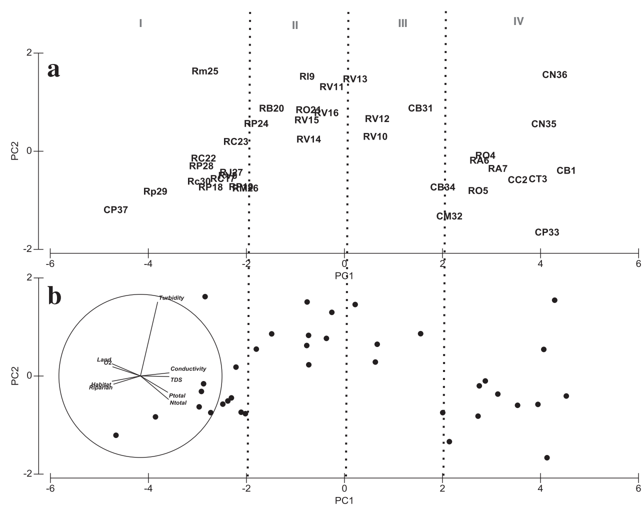

Figure 3. Principal components analysis based on disturbance data (Table 1) for all study sites: (a) distribution of sites with respective code; (b) factor vectors and distribution of sites (represented by black circles). Dashed lines divide the gradient of PC1 into four degradation classes (I–IV)

(D IV), they actually decrease in abundance. Their minimum abundances were found in D I.

In general, for both stream types, there is a clear gradient in the metric values, from D IV to D I, with increase in family richness, BMWP-CETEC and % of EPT, and decrease in % of Collectors-Gatherers and most tolerant taxa (% Olichochaeta, % of Chironomidae + Oligochaeta) (Table 6). For both types, EPT are absent at D IV.

Regarding the four urban streams under DRENUBS programme, there was in all cases a global improvement in abiotic conditions, from D IV to D III during the study period (Table 7). However, while water chemistry had a strong improvement and instream habitat conditions have improved in three sites, land use and riparian vegetation had only a clear improvement in one site each. Only two sites, CB1 and CM32, have improved from D IV to D III considering the taxonomic composition and two (CB1 and CC2) considering metrics (Figure 4). The improvements were due to the decrease in abundance of Oligochaeta and Collectors-Gatherers and to the increase in richness and % of EPT (which nevertheless happened in only one site). At sites CM32 and CP33, there was a decrease in taxa richness and BMWP-CETEC scores, in spite of some improvement regarding % Oligochaeta and Collectors-Gatherers that resulted in the maintenance of the overall position based on metrics (D III). Overall, the enhancement measures were more successful at site CB1 and least successful in CP33.

DISCUSSION

The findings of this study can be useful at both scientific and applied levels, especially for tropical systems where experience in the area of stream restoration is scarce compared with temperate systems. Furthermore, we concluded that (i) for such large-scale/catchment study and using invertebrates data at family level, the effect of seasonal variability (rain/dry seasons) in the invertebrate communities is not relevant; (ii) using a typological approach is important to define appropriate reference conditions and targets for recovery; (iii) urbanization leads to a strong impoverishment in invertebrate communities; (iv) the use of abundance data and not only richness is important to detect biological improvement; (v) the improvement of abiotic conditions is not necessarily accompanied by the biological recovery of invertebrate communities; and (vi) the biological recovery of streams only occurs when several factors are combined, such as the improvement of water quality, habitat heterogeneity and integrity of riparian vegetation. Finally, the holistic multimetric system that was here developed to assess the ecological recovery of all kinds of streams in the catchment of Rio das Velhas, which can also be used in regular monitoring.

The fact that rain seasonality did not result in shifts in the structure and composition of macroinvertebrate communities in Rio das Velhas catchment is consistent with other reports for Brazilian rivers (Melo and Froehlich, 2001) but not for karst Brazilian streams, which can be justified by

Table IV. Mean – SD and mean + SD values for each disturbance variable, within degradation class

	IV	III	II	I
Conductivity	360.8–>621.8	195.3–209.4	56.4–165.8	<21.0–72.9
Total P	0.703–>2.026	0.219–0.463	0.037–0.114	<0.001–0.063
Total N	4.749–>18.798	1.399–3.525	0.259–0.918	<0.195–0.469
O_2	<1.121–3.496	3.38–5.65	6.09–7.27	6.81–>7.54
TDS	297.6–>505.3	142.1–187.0	44.1–145.8	<19.5–64.7
Turbidity	67.2–>188.2	125.0–183.7	99.2–160.7	<2.4–97.0
Habitat global score	<27.9–35.2	36.9–49.9	43.0–59.2	67.5–>83.0
Land use in the site banks	0.0–0.5	1.1–2.1	1.5–2.4	2.8–>3.9
Riparian vegetation integrity	0–0.1	0.2–1.1	0.8–1.6	2.0–>4.3

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Table V. Taxa contributing to the dissimilarity between communities of each degradation class, from the most (IV) to the least (I) disturbed condition, for both stream types, and with their mean abundance (fourth root transformation)

Taxa*	IV	III	II	I
Type 1				
Polymitarcyidae	0.00	0.00	8.59	26.52
Notonectidae	0.00	0.05	3.83	25.58
Isopoda	0.00	0.10	11.77	19.95
Isotomidae	0.00	0.10	13.05	16.73
Philopotamidae	0.00	0.17	20.62	148.02
Naucoridae	0.00	0.44	4.80	12.25
Dryopidae	0.00	0.69	22.52	15.82
Gomphidae	0.00	1.37	3.41	7.13
Elmidae	0.00	2.53	62.05	90.07
Simuliidae	0.00	6.38	49.29	243.25
Leptohyphidae	0.00	8.62	44.79	85.30
Leptophlebiidae	0.00	10.76	44.99	77.67
Bivalvia	0.00	46.25	14.96	13.69
Baetidae	0.00	77.96	97.72	170.03
Hydropsychidae	0.00	504.97	61.56	80.98
Gastropoda	0.57	9.48	21.85	5.78
Hirudinea	0.74	156.75	13.08	7.80
Tipulidae	4.27	0.30	2.51	7.59
Ceratopogonidae	6.46	98.73	4.09	18.13
Stratiomyidae	31.05	0.32	0.05	0.00
Psychodidae	40.59	1.63	0.12	0.32
Chironomidae	283.01	827.49	294.2	837.50
Oligochaeta	2209.55	5772.46	101.7	66.39
Type 2				
Polymitarcyidae	0.00	0.00	0.00	47.14
Dryopidae	0.00	0.00	0.52	15.04
Leptophlebiidae	0.00	0.00	0.81	22.35
Leptohyphidae	0.03	0.00	18.67	61.90
Hydropsychidae	0.07	0.00	145.34	41.48
Baetidae	0.19	0.16	115.17	128.46
Elmidae	0.23	0.00	5.03	119.95
Physidae	0.57	0.16	116.58	0.00
Simuliidae	1.04	0.81	6.52	549.28
Hirudinea	1.24	0.39	35.16	5.02
Ceratopogonidae	9.02	9.32	25.48	12.26
Psychodidae	215.3	70.70	13.24	1.24
Chironomidae	853.54	7312.32	3075.12	790.81
Oligochaeta	934.26	1523.24	1721.31	155.60

rapid increases of flow after rains that might lead to the removal of benthic fauna (Righi-Cavallaro *et al*., 2010). Indeed, the existing literature from Brazil is not coherent on this topic. Natural differences between regions may explain some differences in results. Additionally, often only particular aspects of the community were analysed instead of the multivariate community composition as carried out here, which difficults generalizations. For example, Silva (2011) and Ribeiro and Uieda (2005) found a higher total abundance of invertebrates in dry season because of the smaller water velocity. Callisto and Goulart (2005) found also differences in drifting invertebrates between seasons but a higher richness and diversity in rainy period, which can be explained by the differences in discharge. In wetlands of Pantanal, Heckman (1998) found marked changes in richness between seasons. In our study, the scale of the study may have also influenced the results, as at the catchment scale, differences between sites may mask the variability between seasons for each site. So, even though there might be some differences between seasons within sites, they are not relevant for the aims of this study. This finding may have implications for biomonitoring programmes of these regions, as the effort needed (costs and time) may be considerably reduced if only one season is sampled.

Our study highlights also the importance of considering the natural variability, as not all taxa are expected in good

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Table VI. Metric mean values (family richness, biotic index BMWP-CETEC total score; % of Oligochaeta, % of Chironomidae + Oligochaeta (CHOL), % of Ephemeroptera, Plecoptera and Trichoptera (EPT) and % of Collectors-Gatherers) for all degradation classes (IV–I) and for rivers type 1 and type 2

Metrics	IV	III	II	I
Type 1				
Richness	2.4	3.5	5.3	8.5
BMWP-CETEC	4.2	8.5	22.0	40.7
% Oligoch.	58.8	40.2	11.2	4.2
%CHOL	77.7	80.9	39.2	41.3
% EPT	0.0	2.0	15.5	24.3
% Collectors-Gatherers	79.0	66.11	50.7	49.4
Type 2				
Richness	3.03	2.61	5.25	9.44
BMWP-CETEC	5.06	5.10	17.92	43.09
% Oligoch.	26.14	15.46	24.88	6.70
%CHOL	56.80	76.24	64.58	41.61
% EPT	0.00	0.48	11.38	20.32
% Collectors-Gatherers	81.61	78.53	57.06	49.21

Table VII. Degradation classes attributed to DRENUBS sites (CB1, CC2, CM32 and CM33), before (2003–2006) and after enhancement measures (2010), according to abiotic data, most representative taxa and metrics

	Degradation level		
	Abiotic	Taxa (% B–C sim)	Biotic metrics
CB1			
2003–2006	IV	IV (79%)	III
–2010	III	III (77%)	II
CC2			
2003–2006	IV	IV (81%)	IV
2010	III	IV (55%)	III
CM32			
2003–2006	IV	IV (83%)	III
2010	III	III (61%)	III
CP33			
2003–2006	IV	IV (74% B–C sim)	III
2010	III	IV (51% B–C sim)	III

B-C represent the Bray–Curtis similarity between the samples of a given period and the most similar communities within degradation classes.

conditions in all types of streams. Also, the same taxa seem to exhibit small differences in sensitivities to disturbance in different stream types (e.g. Polymitarcyidae and Hydropsychidae). This was also described by Kiffney and Clements (1993, 1996), who found differences in the response of mayflies to metals depending on the community they were included in and on the place of collection (small high-elevation vs large low-elevation streams). Indeed, the variables used here to design a typology are known to shape the habitat of freshwater benthic invertebrate communities. The geology of the riverbed influences the available instream microhabitats and refuges for invertebrates through substrate size and form (Cummins and Lauff, 1969; Williams and Mundle, 1978). Stream size, elevation, slope and precipitation determine current velocity and flow type, which are major factors in structuring invertebrates (Statzner and Higler, 1986). Climate (temperature and precipitation patterns) influences growth rates (e.g. Becker, 1973; Mackey, 1977), reproduction cycles and emergency periods (e.g. Sweeney and Schnack, 1977; Sweeney and Vannote, 1978).

The degradation gradient found in our study sites reflected the major problems in the catchment that increase from reference to disturbed sites (see PCA), such as the increase in nutrient and organic enrichment, dissolved solids and conductivity caused by urban sewages and agriculture, and the changes in land use and cuts of riparian vegetation due to pastures, urban areas and industries (Pompeu *et al.*, 2005; Moreno *et al.*, 2010). Simultaneously, the metric evolution

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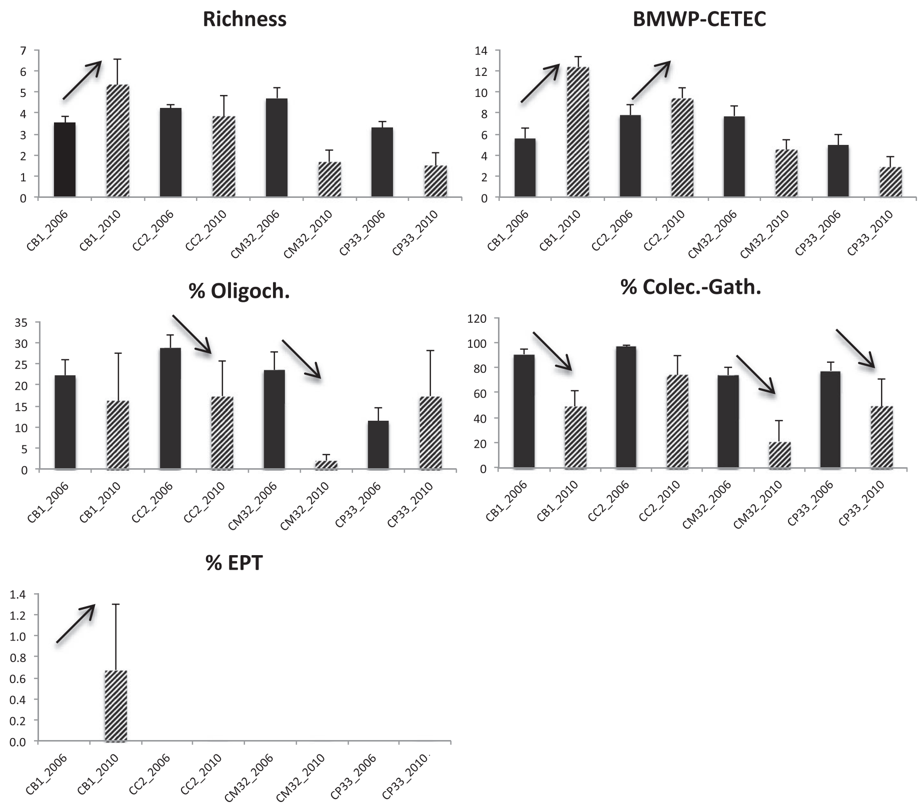

Figure 4. Comparison of metric values for the period before (2003–2006) and 2 years later (2010) enhancement measures for the sites CB1, CC2, CM32 and CP33. The arrows indicate improvement

over classes matches the current knowledge on macroinvertebrate taxa sensitivity to anthropogenic disturbance. For example, the IBMWP index (Alba-Tercedor *et al*., 2004), which scores the tolerance of organisms to organic contamination from 1 (high tolerance) to 10 (low tolerance), scores 8 the Philopotamidae, which, in our study, appear with high abundances in D I, considerably less in D II and absent from D III and D IV. Hydropsychidae, associated with mildly disturbed sites (scores 5), reach their maximum abundance in L II for type 1 and L III for type 2. Chironomidae and Oligochaeta are among the most tolerant taxa to organic contamination (scoring 2 and 1) and appear in all classes.

The major difference between classes I and II is not the presence or absence of certain taxa but in the higher abundance for almost all taxa in class I. In terms of metrics, the proportion of certain groups is also relevant across classes. This finding indicates that quantitative data rather than only a measure of species richness or loss of biodiversity are important to accompany restoration projects. Suren and McMurtrie (2005) also found changes in abundance of certain taxa after enhancement activities, and Mazor *et al*. (2006) concluded that the use of abundance allows the detection of moderate disturbance. We recommend therefore the use of abundance data when assessing the recovery of communities. As an additional metric, evenness could also be useful, as it translates the distribution of the total of number of individuals in a sample by the existent taxa. In our case, sites where the quality is degraded and with almost only Chironomidae and Oligochaeta would have a much lower evenness than sites where several taxa are represented by a similar number of individuals (sites in good condition).

The significant effect of urbanization in invertebrate community degradation depending on its intensity, urban drainage and previous land use was referred in many other studies (e.g. Walsh *et al*., 2001, 2005b; Suren and McMurtrie, 2005; Alberti *et al*., 2007; Cuffney *et al*., 2010). In our urban streams, Chironomidae and Oligochaeta dominate, and other taxa were almost completely absent, which denotes an extremely high disturbance comparable with that described for other regions (e.g. Walsh *et al*., 2001; Suren and McMurtrie, 2005).

The enhancement measures taken under the DRENUBS programme resulted in the upgrading of abiotic conditions. Yet, at only one site, the biological quality has consistently improved. The implementation of sewage treatment plants resulted in a strong improvement in water quality. On the contrary, the habitat quality improved only slightly in two streams (CB1 and CP33), after the enhancement measures.

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Increase in habitat heterogeneity was previously reported to have positive effects on macroinvertebrate richness (Miller *et al.*, 2010). This may have been one factor, along with water quality and riparian vegetation enhancements, contributing to the improvement of biological quality in CB1. In CP33, the biological improvement may have been prevented by the poor condition of the riparian corridors, as riparian vegetation has important functions in stream integrity such as promoting lateral connectivity, bank stabilization, shading, temperature regulation, runoff control and increase of instream habitat diversity (Naiman and Décamps, 1997; Tabacchi *et al.*, 1998, 2000; Kiffney *et al.*, 2004; Gurnell *et al.*, 2005). Therefore, we believe that the urban streams of Rio das Velhas could benefit from further interventions focused on the improvement in habitat diversity and in the recovery of riparian corridors.

In our urban streams, we found small improvement of biological condition 2 years after the enhancement measures. Miller *et al.* (2010) and Friberg *et al.* (1998) found recoveries within 1 and 2 years from restoration measures. However, Charbonneau and Resh (1992) found improvement only 4 years after restoration in Californian streams. On the other hand, no response was found after several years in a small stream by Fuchs and Statzner (1990) or in urban streams by Suren and McMurtrie (2005). Even though the comparability between the various restoration studies is difficult, the variety of responses does not assure us that the maximum potential for recovery in our urban streams was already achieved with the measures taken. Therefore, the continuous monitoring of these streams will be important to clarify this issue.

Nevertheless, our urban headwater streams are probably unable to a biological structural recovery, because of the socio-economic (i.e. integrated in wide urban areas) and geographical (no source populations). Specially for the insects Ephemeroptera, Plecoptera and Trichoptera, the large urban area probably acted as a 'barrier of colonization' (Bond and Lake, 2003), as the distance to the closest reference streams is ≈35 km and aerial active dispersal distances of insects such as the *Baetis* are of approx. 1 km (Hershey *et al.*, 1993). However, it is maybe even more important that they can reach an acceptable level of ecological functioning, in order to avoid a major interference of such streams in the functioning of downstream areas, considering, for example, organic matter dynamics (Miller and Boulton, 2005). In agreement, Lake *et al.* (2007) underlined that restoration projects should be based on general ecological theory and the need to understand the ecosystems processes in order to maintain them. So, we think that future research should concentrate in investigating the potential for ecological functional recovery in urban streams.

In spite of the natural differences, this study shows that restoration of tropical urban streams follows the current knowledge and experience on temperate river restoration and that a mixed multimetric system composed of both biotic and abiotic metrics is useful to follow the recovery of ecological quality of streams where enhancement measures are being implemented.

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