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Rehabilitation of tropical urban streams improves their structure and functioning

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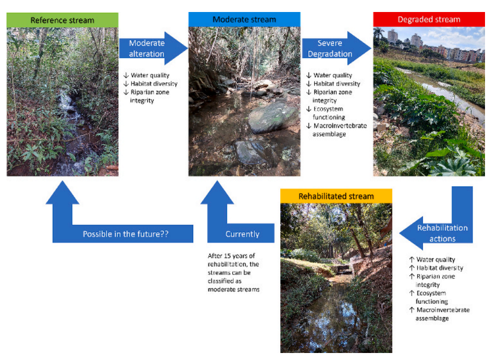
HIGHLIGHTS

- Urban streams with distinct degradation levels were compared in Brazil.
- Rehabilitation of streams improves their structural and functional indicators.
- Rehabilitated streams (15y) have lower ecological condition than reference streams.

GRAPHICAL ABSTRACT



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ABSTRACT

Urban streams are affected by a complex combination of stressors, which modify physical habitat structure, flow regime, water quality, biological community composition, and ecosystem processes and services, thereby altering ecosystem structure and functioning. Rehabilitation projects have been undertaken in several countries to rehabilitate urban streams. However, stream rehabilitation is still rarely reported for neotropical regions. In addition, most studies focus on structural aspects, such as water quality, sediment control, and flood events, without considering ecosystem function indicators. Here, we evaluated the structure and functioning of three 15-y old rehabilitated urban stream sites in comparison with three stream sites in the best available ecological condition (reference), three sites with moderate habitat alteration, and three severely degraded sites. Compared to degraded streams, rehabilitated streams had higher habitat diversity, sensitive macroinvertebrate taxa richness, and biotic index scores, and lower biochemical oxygen demand, primary production, sediment deposition, and siltation. However, rehabilitated streams had higher primary production than moderate and reference streams, and lower canopy cover, habitat diversity, sensitive macroinvertebrate taxa richness, and biotic index scores than reference streams. These results indicate that rehabilitated streams have better structural and functional condition than degraded streams, but do not strongly differ from moderately altered streams, nor have they reached reference stream condition. Nonetheless, we conclude that rehabilitation is effective in removing

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streams from a degraded state by improving ecosystem structure and functioning. Furthermore, the combined use of functional and structural indicators facilitated an integrative assessment of stream ecological condition and distinguished stream conditions beyond those based on water quality indicators.

1. Introduction

Urban freshwater ecosystems provide important goods and services for human populations, including regulatory services such as mitigating urban heat islands by regulating microclimates (Xiao et al., 2023) and mitigating floods (Yang and Zhang, 2011). Urban streams also constitute an important biodiversity reservoir by providing habitat and resources for many species, including birds, reptiles, mammals, insects, fishes, aquatic plants, and algae (Lepczyk et al., 2017).

However, urban streams are affected by a complex combination of human disturbances and commonly express degraded physical, chemical, and biological conditions that have been collectively termed “the urban stream syndrome” (Walsh et al., 2005; Hughes et al., 2014; Booth et al., 2016). To allow the growth of cities, streams are forced to fit into the urban design, regardless of their natural characteristics (Paul and Meyer, 2008). For instance, drainage works are carried out to constrain streams and natural flood pulses, with the channelization or burying of streams leading to changes in channel hydromorphology, water velocity, and removal of natural obstacles that accumulate sediments and organic matter (Paul and Meyer, 2008; Wantzen et al., 2019). Also, riparian forest is cleared to provide space for human activities, causing increased soil exposure, susceptibility to erosion, and water temperature, and decreased allochthonous plant inputs (Carvalho-Santos et al., 2016; Wantzen et al., 2019). In addition, poor sewage and garbage collection make urban streams common destinations for waste disposal, resulting in water pollution and consequent increases in the concentrations of nutrients, heavy metals, and other contaminants (Paul and Meyer, 2008). These changes alter stream physical habitat structure (Wantzen et al., 2019), water quality (Bakure et al., 2020), and biological community composition. The latter is typically indicated by decreased biotic richness and sensitive species abundances (Birk et al., 2020; Firmiano et al., 2021), and increased non-native species abundances (Gaertner et al., 2017).

Urbanization also affects ecosystem functioning (e.g., primary and secondary production, organic matter decomposition, and stream metabolism), with consequences on the flow of energy and nutrients through the food web, and to downstream reaches (von Schiller et al., 2008; Elosegi and Sabater, 2013; Pereda et al., 2019). In streams, the decomposition of organic matter is directly affected by the input of pollutants, and modification of the riparian forest, which reduce the abundance and activity of microbial decomposers and detritivores (Yule et al., 2015; Ferreira et al., 2020; Tagliaferro et al., 2020). In addition, primary production, cyanobacteria growth, and eutrophication are often stimulated by increased nutrient and light availability, both of which are enhanced by environmental degradation in urban streams (Feio et al., 2010; Mamun and An, 2019; Vincent et al., 2022).

Stream rehabilitation can improve the ecological condition of urban streams and ensure the provision of goods and services for human populations (Wantzen et al., 2019; Feio et al., 2021). Rehabilitation interventions aim to improve stream status by bringing them closer to their pre-disturbance condition (Brierley and Fryirs, 2000). In general, rehabilitation includes recovery of water quality, channel physical complexity, and bank stability (Hughes et al., 2014). Rehabilitation also includes measures to improve aquatic and riparian biodiversity by recovering riparian vegetation, creating environmental reserves, and protecting springs (Feio et al., 2021). In addition, rehabilitation projects may include the creation of linear parks or urban green areas for human recreation and environmental education activities (Porfiriev et al., 2017; Hunter et al., 2019; Macedo et al., 2022). Although stream rehabilitation projects can facilitate ecosystem functioning and services, this often is not the objective of the interventions (Feio et al., 2021; Ranta et al., 2021).

Stream rehabilitation projects have been developed in several temperate countries (Feio et al., 2021). However, urban stream rehabilitation remains rare in tropical regions (Macedo et al., 2011; Wantzen et al., 2019; Feio et al., 2021). In addition, most studies evaluating urban stream rehabilitation efficacy focus on water quality, sediment control, and flood control (e.g., da Cruz e Sousa and Ríos-Touma, 2018; Macedo et al., 2022), without considering ecosystem function indicators (Wantzen et al., 2019). However, because ecosystem structure and function are not always related (McKie and Malmqvist, 2009; Feckler and Bundschuh, 2020), both attributes must be considered for a holistic assessment of stream ecological condition (Feio et al., 2010; Ranta et al., 2021; Brosed et al., 2022). In fact, functional approaches including ecological processes such as primary production, organic matter decomposition, and ecosystem metabolism can provide important information about stream functional condition (Feio et al., 2010; Ferreira et al., 2020, 2021). Together, structural and functional indicators contribute to an integrative ecosystem assessment (von Schiller et al., 2008; Pereda et al., 2019; Ranta et al., 2021; Costa et al., 2023).

We assessed the effects of rehabilitation on the structure and functioning of 12 urban stream sites. To do so, we compared conditions at three streams in the best ecological conditions available (reference), three rehabilitated streams, three streams with moderate habitat alteration that were not rehabilitated (moderate), and three severely degraded streams (degraded), in terms of structural and functional indicators. We sought to determine the degree to which stream rehabilitation improved the structural and functional indicators of previously degraded streams. We hypothesized that the structural and functional conditions of rehabilitated streams would not differ significantly from moderate streams but would differ significantly from degraded streams. As rehabilitation interventions improve the structure and functioning of urban streams, we anticipate that rehabilitated streams will assume a moderate level of integrity on the stream alteration gradient (degraded $<$ rehabilitated $\sim$ moderate $<$ reference) across all variables studied (Table 1).

2. Methods

2.1. Study streams

We studied 12 stream sites located within parks in the metropolitan region of Belo Horizonte (Fig. 1, Table S1), the third largest Brazilian metropolis (ca. 5.5 million inhabitants), during the dry season (May to September 2022). The parks (0.014–39.400 km^2; Table 2), which are public properties and are managed by the municipality, aim to preserve green areas within the city and are used mainly for leisure by the local population. The streams were classified in four categories (three streams each), according to the environmental conditions of the park where they are located: reference, moderate, rehabilitated, and degraded.

Rehabilitated streams are heavily degraded streams that underwent intervention to promote ecological rehabilitation, and include: (i) Primeiro de Maio stream (first order, 0.207 km^2 of drainage area), located in the Primeiro de Maio Ecological Park (0.034 km^2); (ii) Baleares stream (first order, 0.145 km^2 of drainage area), located in the José Lopes dos Reis Baleares Municipal Park (0.014 km^2); and (iii) Nossa Senhora da Piedade stream (first order, 0.191 km^2 of drainage area), located in the Nossa Senhora da Piedade Municipal Park (0.058 km^2) (Fig. 1). The intervention in the rehabilitated streams was carried out by the “DRENURBS Program” of the Municipality of Belo Horizonte, with

Table 1
Mechanisms that support the prediction about the effects of rehabilitation of urban streams on structural and functional parameters.

Parameter	Mechanism	Rationale	Reference
Water quality	Dissolved oxygen concentration and canopy cover will be higher and biochemical oxygen demand, total dissolved solids, total nitrogen and total phosphorus concentrations and turbidity will be lower in reference, moderate, rehabilitated, and lastly degraded streams.	Rehabilitation actions interrupt the entry of polluting agents into the stream (sewage, garbage) and improve water quality indicators.	Macedo et al. (2022)
Habitat diversity	Habitat diversity will be higher in reference, moderate, rehabilitated, and lastly degraded streams.	The rehabilitation process recovers the riparian zone and the natural characteristics of the stream, increasing habitat diversity and complexity.	Macedo et al. (2022)
Macroinvertebrate assemblage structure and composition	Richness and diversity of total taxa, richness and abundance of sensitive taxa and biotic indices scores will be higher and abundance of total taxa, richness and abundance of resistant taxa will be lower in reference, moderate, followed by rehabilitated, and lastly degraded streams.	Rehabilitation of streams minimizes the presence of anthropogenic stressors that would select organisms with traits that confer resistance to environmental degradation, increasing taxa richness and diversity and the presence of sensitive taxa.	Castro et al. (2018); Al-Zankana et al. (2020); Firmiano et al. (2021)
Primary production	Chlorophyll a concentration will be lower and autotrophic index will be higher in reference, moderate, followed rehabilitated, and lastly degraded streams.	High primary production rate occurs mainly in streams with excessive nutrient input and lower canopy cover.	Feio et al. (2010)
Organic matter decomposition	Remaining organic matter mass and decomposition rates will be changed in rehabilitated and degraded streams compared to reference streams.	The improvement in organic matter decomposition efficiency is due to the decrease in sedimentation and increase in pH, oxygenation and habitat heterogeneity that promote the diversity and activity of decomposers.	Ferreira et al. (2021)
Sediment deposition	Sediment deposition will be less in reference, moderate, followed by rehabilitated, and lastly degraded streams.	The entry of fine sediments into the streams is mainly due to the erosion of the banks, associated with the suppression of the riparian vegetation. In the rehabilitated streams, works were carried out to contain the banks and recover the riparian zone.	von Bertrab et al. (2013)

financial support from the Inter-American Development Bank (US$ 8.718 million; BID, 2008), between 2006 and 2008. Before the rehabilitation interventions, there was human occupation of the streambanks and floodplains, untreated sewage discharges into natural channels, garbage dumping, absence of riparian vegetation, erosion on the banks, and siltation of streambeds. Linear parks were created, and several actions were implemented locally to improve water quality (Macedo et al., 2022). Interventions included: (i) sewage collection and sewage treatment networks; (ii) bank control and stabilization through artificial structures such as gabions, walls, and geotextiles; (iii) improvement of rainwater drainage systems; (iv) streambed stabilization using fixed clusters of boulders; (v) flood control systems that use detention basins; (vi) revegetation of riparian zones with woody tree species, but grass areas are maintained for public uses; (vii) removal of houses from floodplains and streambanks; and (viii) installation of trails and structures for recreational activities (BID, 2008; PBH, 2012). Rehabilitated streams still have sparse riparian woody vegetation, with a predominance of grasses, anthropogenic alterations upstream of the study stream site (channelization and residential occupation), free and easy access for visitors, and local gentrification (Macedo et al., 2022; Golgher et al., 2023).

The reference ("least disturbed") streams correspond to the best ecological state possible in the metropolitan region for water quality, habitat diversity, and riparian zone condition, and include: (i) Taboões stream (third order, 2.804 km^2 of drainage area), located in the Serra do Rola Moça State Park (39.400 km^2), where water is collected for human consumption; (ii) Serra stream (second order, 1.418 km^2 of drainage area), located in the Mangabeiras Municipal Park (2.400 km^2), an environmental protection area with 59 springs and extensive native vegetation; and (iii) Clemente stream (second order, 1.816 km^2 of drainage area), located in the Roberto Burle Marx Municipal Park (0.176 km^2), an urban conservation unit with extensive native vegetation (Fig. 1). Although parks have recreational structures, the areas where the reference streams are located have restricted human access. The streams originate in conservation areas and flow directly into the park, without interference from urban areas upstream of the study sites.

The moderate streams are in urban residential areas, with a moderate degree of habitat alteration, but have not been affected by sewage and garbage contamination or riparian deforestation. Therefore, these streams are in a moderate condition and include: (i) Bonsucesso stream (first order, 0.059 km^2 of drainage area), located in Jacques Cousteau Municipal Park (0.335 km^2), which has an extensive area of natural vegetation at the stream site, but large deposits of fine sediments in the streambed; (ii) Nado stream (first order, 0.237 km^2 of drainage area), located in Lagoa do Nado Municipal Park (0.331 km^2), which has narrow riparian vegetation, bank erosion, and large sand deposits; and (iii) Ponte Queimada stream (second order, 1.160 km^2 of drainage area), located in the Aggeo Pio Sobrinho Municipal Park (0.600 km^2), which has extensive natural vegetation and restricted visitor access (Fig. 1).

Degraded streams are severely degraded by urban activities and have severely compromised environmental quality, and include: (i) Mergulhão stream (second order, 1.103 km^2 of drainage area), located in the Belo Horizonte Technological Park (0.544 km^2), which is an area with recurrent diesel oil spill accidents, garbage deposits and sewage discharge into the stream, and nearby roads (at <5 m); (ii) Bom Jesus stream (second order, 0.704 km^2 of drainage area), located in the Belo Horizonte Zoobotanic Garden (1.234 km^2), is an area with domestic sewage discharge into the stream, sugar cane monoculture on the streambanks, and channelization; and (iii) Lagoinha stream (first order, 1231 km^2 of drainage area), located in Vilarinho Park (0.042 km^2), receives domestic sewage discharge and garbage inputs into the stream, human occupation of the streambanks and floodplains, and free horses and pigs roaming in the stream (Fig. 1).

2.2. Abiotic characterization

Land use and occupation at each stream site was analyzed for the drainage sub-basin within a radius of 500 m upstream of the study site (Fig. S1). The image was acquired using Google Earth Engine and classification of land uses and quantification of land cover categories were done using object-oriented classification (Macedo et al., 2014) (Fig. S1).

Water quality variables were measured in situ, monthly for five months during the dry season (May to September 2022), for each stream site, and included temperature (°C), dissolved oxygen concentration (mg/L) and saturation (%) (YSI ProSolo model), pH and oxi-redox potential (mV) (Digimed DM-2P), turbidity (NTU; Digimed DM-TU), electric conductivity (μS/cm), total dissolved solids (mg/L), and resistivity (KΩ/cm) (Digimed DM-3P). On the same occasions, 500 mL of water were collected in glass bottles for determination of biochemical oxygen demand (BOD; mg/L) and 250 mL of water were collected in amber plastic bottles for determination of total phosphorus and total nitrogen concentrations (mg/L) (APHA, 2012).

Stream depth (m; average of three random measurements), wet width (m; average of three random measurements), flow velocity (m/s; average of three random measurements), and canopy cover (average of three random measurements, using free Canopy Capture® software) were measured on the same occasions, at each stream site. The stations were randomized from the initial collection point, with a distance of 5 m between them, according to Macedo et al. (2014).

The Habitat Diversity Rapid Assessment Protocol (Callisto et al., 2002) was applied in May 2022 to assess the physical condition of streams and their riparian areas. This protocol is based on the assessment of 22 visual environmental parameters related to the use and occupation of the riparian zone and apparent characteristics of the water (e.g., vegetation cover in the riparian zone, presence of anthropogenic changes, presence of erosion on the streambanks, streambed siltation), scored from zero to four, and 11 visual environmental parameters related to flow conditions and substrate (e.g., types of stream bottom, and water flow characteristics), scored from zero to five. The protocol is synthesized into a final score that reflects the conservation level of each site, where a score of 0–40 indicates degraded sites, 41–60 indicates altered sites, and > 60 indicates reference sites (Callisto et al., 2002; Macedo et al., 2022).

2.3. Benthic macroinvertebrates

We collected three benthic subsamples in each of the 12 stream sites with a Surber sampler (30 cm aperture, 0.09 m^2 area, 0.250 mm mesh) in September 2022, preferably one subsample in coarse sediment (pebbles or gravels), another in fine sediments (sand and silt), and a third in leaf deposits to capture the greatest habitat heterogeneity (Ligeiro et al., 2020; Callisto et al., 2021). The subsamples were individually placed in plastic bags and fixed with 70 % ethanol. In the laboratory, the samples were washed over sieves (0.250 mm mesh) in running water, and organisms were sorted and identified to family level, except for the suborder Hydracarina, class Bivalvia, and subclass Oligochaeta, under a magnifying glass using taxonomic keys (Mugnai et al., 2010; Hamada et al., 2014; Hamada et al., 2018) (Table S2). Benthic macroinvertebrates were classified according to their tolerance to environmental degradation, as resistant, tolerant or sensitive, following Junqueira et al. (2000, 2018). The three subsamples for each stream were composited and analyzed as a single sample.

2.4. Functional variables

2.4.1. Biofilm growth on artificial substrates

Biofilm growth rate, chlorophyll *a* (Chla) concentration, and the autotrophic index were used as primary production indicators (APHA, 2012). Three clean slate stones (8 × 8 cm, 64 cm^2) were incubated in

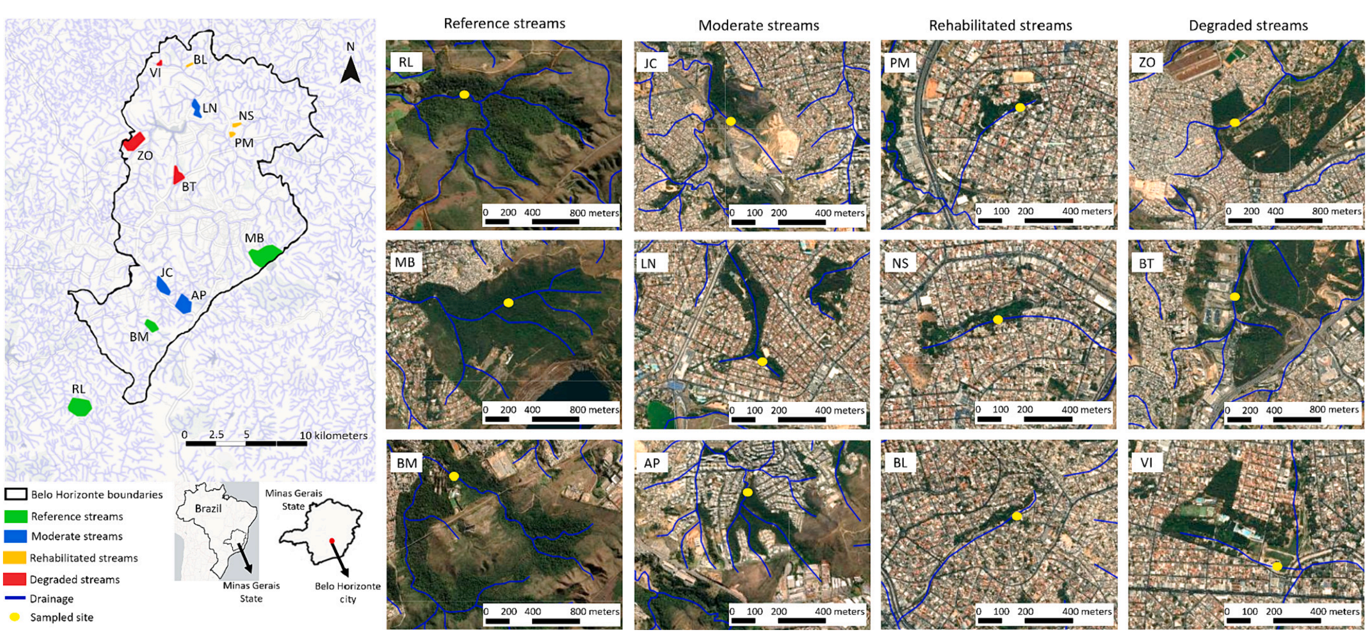

Fig. 1. Location of reference, moderate, rehabilitated, and degraded stream sites: RL = Serra do Rola Moça State Park, MB = Mangabeiras Municipal Park, BM = Roberto Burle Marx Municipal Park, JC = Jacques Cousteau Municipal Park, LN = Lagoa do Nado Municipal Park, AP = Aggeo Pio Sobrinho Municipal Park, PM = May Day Ecological Park, BL = José Lopes dos Reis Baleares Municipal Park, NS = Nossa Senhora da Piedade Municipal Park, BT = Belo Horizonte Technological Park, ZO = Belo Horizonte Zoobotanic Garden, VI = Vilarinho Park.

each stream site for four consecutive 30-day periods (May to August 2022). After each incubation period, the stones were collected, enclosed individually in aluminum foil envelopes, and taken to the laboratory, in a dark and refrigerated box.

The biofilm developing in the upper surface of each stone was removed with a cell scraper. The biofilm content in half of the scraped surface area (32 cm^2) was transferred to a pre-weighed porcelain crucible, oven-dried (60 °C, 48 h), weighed (± 0.01 mg) to determine dry mass (DM), incinerated (550 °C, 4 h), and weighed again to determine the ash mass. The biofilm ash-free dry mass (AFDM; mg) was estimated as the difference between DM and ash mass. The biofilm growth rate (Gr) was calculated as: Gr = biofilm AFDM/a/t, where a is the scraped area of the stone (m^2) and t is the incubation time (days), and results were expressed as mg AFDM/m^2/d.

The biofilm content in the second half of the scraped surface area (32 cm^2) was added to 100 mL of distilled water, vacuum filtered through a glass fiber filter, soaked in 10 mL acetone solution (90 %) for ~20 h in a refrigerator, and macerated. The Chla concentration was determined spectrophotometrically (Thermo GENESYS™ 10UV UV–Vis) by reading the absorbance at 664, 665, and 750 nm (APHA, 2012), and results were expressed as mg Chla/m^2. The relative importance of autotrophs versus heterotrophs in the biofilm was calculated as the autotrophic index (AI): AI = biofilm AFDM (mg)/Chla (mg).

2.4.2. Sediment deposition

Artificial grass mats (10 cm × 15 cm × 3 cm, 2 mm thick filaments and 42 filaments per cm^2) were used to estimate sediment deposition (von Bertrab et al., 2013). Three mats were fixed to the stream bed of each stream site with clamps and iron rods, in pool areas, for four consecutive 30-day periods (May to September 2022). After each incubation period, the mats were carefully removed, enclosed individually in plastic bags, sealed, and taken to the laboratory. In the laboratory, the mats were washed under running water, and the water and residues were collected in aluminum trays. The trays were left intact to allow the sediment to settle until the supernatant water became clear, which was later removed by suction, leaving only the wet sediment in the tray. The wet sediment was oven-dried (60 °C, 4 days) and weighed (±0.01 mg) to determine the total sediment deposition per area, and results were expressed as g/cm^2.

2.4.3. Organic matter decomposition

Senescent leaves of *Bauhinia forficata* Link (Fabaceae) were collected at the Universidade Federal de Minas Gerais with nets installed below the canopies, in autumn (February – April 2022), and allowed to air dry at room temperature. *Bauhinia forficata* is a native, deciduous tree species that is abundant in the study region (Lorenzi, 2002; Vaz et al., 2010). Because it is a pioneer fast-growing plant, it is commonly used for recovering the riparian vegetation (Lorenzi, 2002; Vaz et al., 2010). Individuals of *B. forficata* were visually identified in 10 of the 12 parks studied, including the three rehabilitated parks.

Batches (3 g, ± 0.01 mg) of air-dried leaves were placed in fine-mesh bags (FM; 17 × 10 cm, 0.5 mm mesh) to determine microbial-mediated leaf litter decomposition, and in coarse-mesh bags (CM; 10 × 14 cm, 1 cm mesh) to determine leaf litter decomposition mediated by both macroinvertebrates and microorganisms (i.e., total leaf litter decomposition) (Graça et al., 2005). Three leaf litter bags of each type (3 FM and 3 CM) were deployed at each stream site on June 2022 and allowed to decompose for 60 days. After the incubation period, all litter bags were recovered, enclosed individually in plastic bags, and transported to the laboratory in a cooler. Leaf litter was removed from the mesh bags, carefully cleaned of sediments and associated organisms with running tap water on top of a sieve (0.5 mm mesh) to retain small leaf fragments, oven-dried (60 °C, 48 h), weighed (± 0.01 mg) to determine DM, incinerated (550 °C, 4 h), and weighed again to determine ash mass. Leaf litter AFDM remaining was estimated as the difference between DM and ash mass, and the percentage of AFDM remaining was estimated as: % AFDM remaining = AFDM remaining (g)/initial AFDM (g) × 100. The initial AFDM was estimated by multiplying the initial air-dried mass by a correction factor determined from an extra set of 12 litter bags, which were prepared as the experimental litter bags, but used on day 0 to estimate the initial AFDM (as described above).

2.4.4. Data analyses

Because samples taken over the 5 dry-season months were pseudoreplicates, and there was no variation among months, we performed statistical analyses on the average across the months for each stream, with the three streams within each stream category considered as replicates.

Water quality, physical habitat structure, habitat diversity, and functional variables, except AFDM, were compared among stream

Table 2
Stream order, park and drainage area, land use in the drainage area (min–max) and water physical and chemical variables in reference, moderate, rehabilitated and degraded streams (mean ± SD, $n = 3$ streams). Different letters indicate statistical differences among stream categories (one-way ANOVA followed by Tukey's test, $p < 0.05$).

Variables	Reference	Moderate	Rehabilitated	Degraded	*p*-Value
Stream order (Strahler system)	2nd–3rd	1st–2nd	1st	1st–2nd	
Park area (km^2)	0.176–39.400	0.331–0.600	0.014–0.058	0.042–1.234	
Drainage area (km^2)	1.42–2.8	0.06–1.16	0.15–0.21	0.70–1.23	
Forest cover (%)	74.1–100.0	3.2–54.0	1.7–9.7	3.4–37.6	
Urban cover (%)	0.0–25.9	46.0–96.8	90.3–98.3	62.4–96.6	
Electric conductivity (μS/cm)	36.2 ± 30.7 a	119.5 ± 79.8 a	338.0 ± 62.6 b	318.6 ± 49.8 b	<0.01
Dissolved oxygen concentration (mg/L)	8.4 ± 0.1 a	7.4 ± 1.0 ab	7.2 ± 0.4 ab	5.0 ± 2.1 b	0.04
Dissolved oxygen saturation (%)	101.5 ± 3.0 a	88.9 ± 10.3 ab	87.6 ± 2.2 ab	61.0 ± 25.1 b	0.04
Biochemical oxygen demand (mg/L)	0.5 ± 0.1 a	1.0 ± 0.4 a	1.2 ± 0.6 a	3.1 ± 0.8 b	<0.01
pH	7.67 ± 0.30 a	7.57 ± 0.08 a	7.79 ± 0.35 a	7.68 ± 0.30 a	0.81
Oxi-redox potential (mV)	153.4 ± 19.4 a	114.1 ± 9.8 a	138.2 ± 29.8 a	130.4 ± 15.9 a	0.19
Resistivity (KΩ/cm)[a]	40.9 ± 37.9 a	10.5 ± 3.1 ab	3.8 ± 1.3 b	3.7 ± 0.9 b	<0.01
Water temperature (°C)	18.9 ± 1.4 a	19.2 ± 1.7 a	21.1 ± 1.9 a	20.3 ± 0.7 a	0.31
Total dissolved solids (mg/L)	24.1 ± 18.4 a	72.6 ± 43.0 a	210.4 ± 55.8 b	191.4 ± 29.9 b	<0.01
Total nitrogen (mg/L)	6.97 ± 2.49 a	8.44 ± 3.26 a	7.97 ± 3.22 a	12.35 ± 3.94 a	0.87
Total phosphorus (mg/L)	0.103 ± 0.055 a	0.056 ± 0.062 a	0.069 ± 0.090 a	0.051 ± 0.046 a	0.65
Turbidity	1.0 ± 0.9 a	3.3 ± 2.2 a	3.3 ± 3.3 a	8.7 ± 10.5 a	0.43
Canopy cover (%)	83.7 ± 2.6 a	78.6 ± 0.4 ab	71.6 ± 4.3 b	48.2 ± 41.9 b	0.01
Depth (m)	0.096 ± 0.043 a	0.103 ± 0.067 a	0.124 ± 0.041 a	0.075 ± 0.025 a	0.66
Flow velocity (m/s)	0.32 ± 0.22 a	0.04 ± 0.03 a	0.06 ± 0.05 a	0.63 ± 0.45 a	0.06
Wet width (m)	2.54 ± 1.02 a	3.32 ± 0.67a	2.06 ± 0.25 a	2.51 ± 0.32 a	0.20

[a] Log(x)-transformed for the analysis.

categories using one-way ANOVA, followed by Tukey's post-hoc tests to assess statistical differences among stream categories (reference, moderate, rehabilitated, and degraded). AFDM was compared among stream categories, mesh size, and their interaction using two-way ANOVA, followed by Tukey's post-hoc test. Physical and chemical water variables and functional variables were compared among stream categories using Principal Component Analyses (PCA) to determine the distance between streams categories and understand the relative importance of each variable.

The benthic macroinvertebrate assemblages were compared among stream categories for three different types of widely used indicators. (i) Assemblage structure indicators included total taxa richness, total individual abundance, density of organisms per m^2, Shannon-Wiener diversity index, Simpson diversity index, and Pielou evenness index. (ii) Assemblage composition indicators included taxonomic composition, % Ephemeroptera, Plecoptera, and Trichoptera (EPT) individuals, EPT/Chironomidae individuals ratio, EPT/(Chironomidae + Oligochaeta) individuals ratio, % Chironomidae individuals, % Chironomidae + Oligochaeta individuals, % sensitive individuals abundance, % tolerant individuals abundance, % resistant individuals abundance, % sensitive taxa richness, % tolerant taxa richness, and % resistant taxa richness. (iii) We calculated five macroinvertebrate biotic indices previously used for Cerrado streams: Biological Monitoring Working Party (BMWP), Average Score Per Taxon (ASPT) (Junqueira et al., 2000, 2018), Benthic Multimetric Index (BMI) (Ferreira et al., 2011), Macroinvertebrate Multimetric Index (MMI 1) (Macedo et al., 2016), and MMI 2 (Silva et al., 2017). Taxonomic composition was compared among stream categories by ANOSIM, followed by pairwise comparisons with Wilcoxon's test. Other macroinvertebrate indicators were compared among stream categories by one-way ANOVA, followed by Tukey's post-hoc test

Fig. 2. Principal components analysis (PCA) showing the grouping patterns of the different categories of streams in relation to: A) water physical and chemical variables and B) functional variables. Cond = conductivity, TSD = total dissolved solids, temp = temperature, OBD = biochemical oxygen demand, turb = turbidity, width = average width, speed = average speed, pH = water pH, Nitro = dissolved nitrogen, Phosp = dissolved phosphorus, redox = redox potential, depth = average depth, resist = resistivity, canopy = canopy cover, do_mg = dissolved oxygen concentration, and do_por = dissolved oxygen saturation. Sediment = total amount of sediment deposited per area, Chla = chlorophyll *a* concentration, Growth = biofilm growth, AFDM = ash-free dry mass remaining of *Bauhinia forficata* leaves, Auto/heterotrophy = ratio of autotrophy to heterotrophy. The directions of the arrows indicate the direction of the correlation, while the length of the vector shows the strength of the correlation. For stream names refer to Fig. 1.

when needed. Pearson's correlation index was calculated for contrasting macroinvertebrates indicators and habitat diversity.

All statistical tests were performed using R software (R Core Team 4.2.1). Data were transformed with log(x) whenever necessary as indicated in the statistics tables. Normality was assessed using Shapiro-Wilk's test and homogeneity of variances using Levene's test. Significance was established at $p < 0.05$.

3. Results

3.1. Land use

The result of mapping land use and cover for the upstream site sub-basins showed two main types of use: forest and urban, with the forest areas corresponding to park area (Table S1, Fig. S1). For the rehabilitated streams, land use was predominantly urban, covering 90 to 98 % of the total area, with the forest area varying from 1 to 5 %. For the reference streams, preserved natural vegetation covered 74 to 100 % of the total area. For moderate and degraded streams, forest cover varied greatly, ranging from 3 to 54 % of the total area for moderate streams and 3 to 37 % of the total area for degraded streams.

3.2. Abiotic characterization

The PCA for water physical and chemical variables (Fig. 2A) showed clear differences between reference and degraded streams, but not others. Dimensions one and two explained 28.9 % and 11.4 % of the environmental variability, respectively. The variables most strongly associated with the differences between stream types were canopy cover, dissolved oxygen, BOD, turbidity, conductivity, and total dissolved solids. Stream categories differed significantly in seven out of 16 physical and chemical variables (one-way ANOVA, $p \leq 0.04$; Table S3, Table 2). Rehabilitated streams had lower BOD than degraded streams, but higher conductivity and total dissolved solids, and lower resistivity and canopy cover than reference streams (Table 2). Dissolved oxygen concentration and saturation, resistivity, and canopy cover were higher, and conductivity, BOD, and total dissolved solids were lower in reference than in the degraded streams (Table 2; Fig. 2). Moderate streams did not differ from reference streams (Fig. 2) but had lower conductivity and total dissolved solids than degraded and rehabilitated streams and lower BOD than degraded streams (Table 2).

Habitat diversity scores differed significantly among stream categories (one-way ANOVA, $p < 0.01$; Table S3). Reference streams had the highest habitat diversity scores, followed by moderate and rehabilitated streams that did not differ from each other, while degraded streams had the lowest scores (Fig. 3).

3.3. Benthic macroinvertebrates

The macroinvertebrate assemblage structure indicators did not differ significantly among stream categories (one-way ANOVA, $p \geq 0.06$), except for total taxa richness ($p = 0.01$), which was higher in reference than in degraded streams (Fig. 4A, Table S3). Taxonomic composition differed significantly between degraded streams and other stream categories (ANOSIM, $p < 0.01$; Table S3). The assemblage composition indicators did not differ significantly among stream categories (one-way ANOVA, $p \geq 0.07$), except for sensitive and resistant taxa richness and tolerant individuals' abundance ($p \leq 0.03$) (Table S3). Percentage sensitive taxa richness was highest in reference streams, followed by moderate and rehabilitated streams that did not differ, and lowest in degraded streams (Fig. 4B). Percentage resistant taxa richness was lower in reference than in degraded streams (Fig. 4C). The abundance of tolerant individuals was higher in reference streams than in degraded streams (Table S6).

The biotic indices significantly differed among stream categories (one-way ANOVA, $p \leq 0.02$), except for the BMI ($p = 0.07$) (Table S3). In

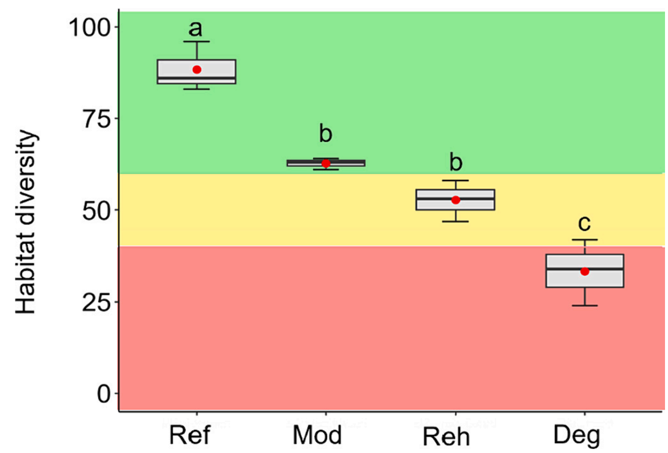

Fig. 3. Final scores of the Habitat Diversity Rapid Assessment Protocol (Callisto et al., 2002). Values <40 indicate degraded streams (red band), values between 41 and 60 indicate altered streams (yellow band), and values >60 indicate streams in reference conditions (green band). In the boxplot, the central line represents the median, the rectangle represents the 50 % confidence interval, the vertical bars represent the data dispersion, and the red dot represents the mean. Different letters indicate statistical differences among stream categories (one-way ANOVA followed by Tukey's test, $p < 0.05$). Ref = Reference streams, Mod = Moderate streams, Reh = Rehabilitated streams, and Deg = Degraded streams.

all biotic indices, except for the BMI, values were higher in reference than in degraded streams (Fig. 4D–G). BMWP values were also higher in rehabilitated and moderate than in degraded streams and lower in rehabilitated than in reference streams (Fig. 4D). MMI2 values were also higher in rehabilitated than in degraded streams (Fig. 4G).

Strong positive correlations (Pearson's, $r \geq 0.65$ and $p \leq 0.02$) were found between habitat diversity and total taxa richness, Shannon-Wiener, Simpson, Pielou evenness, sensitive taxa richness, tolerant individuals' abundance, BMWP, ASPT, MMI1, and MMI2 scores (Table 3). Strong negative correlations (Pearson's, $r \leq -0.63$ and $p = 0.03$) were found between habitat diversity and resistant taxa richness, and resistant individuals' abundance (Table 3).

3.4. Functional variables

The PCA for functional variables (Fig. 2B) showed clear functional differences between the degraded streams and the other three stream categories. Dimensions one and two were highly explanatory (81 % and 13.4 %, respectively), with sedimentation, Chla concentration, AFDM, and biofilm growth being most strongly associated with the PCA axes.

Biofilm growth rates and Chla concentration significantly differed among all stream categories (one-way ANOVA, $p < 0.01$; Table S3), with the highest values in degraded streams followed by rehabilitated, moderate, and reference streams (Fig. 5A and B). The autotrophic index significantly differed among stream categories (one-way ANOVA, $p < 0.01$; Table S3), being higher in reference and moderate streams than in rehabilitated and degraded streams; it was also higher in rehabilitated than in degraded streams (Fig. 5C).

Total sediment deposition significantly differed among stream categories (one-way ANOVA, $p < 0.01$; Table S3), being higher in degraded streams than in the other stream categories, which did not differ (Fig. 5D).

Percentage AFDM remaining of *B. forficata* leaf litter after 60 days incubation in the streams significantly differed only between coarse mesh in degraded and fine mesh in reference streams (two-way ANOVA, $p = 0.04$), but was not affected by mesh size or the interaction between stream type and mesh size ($p > 0.18$) (Table S3, Fig. 5E).

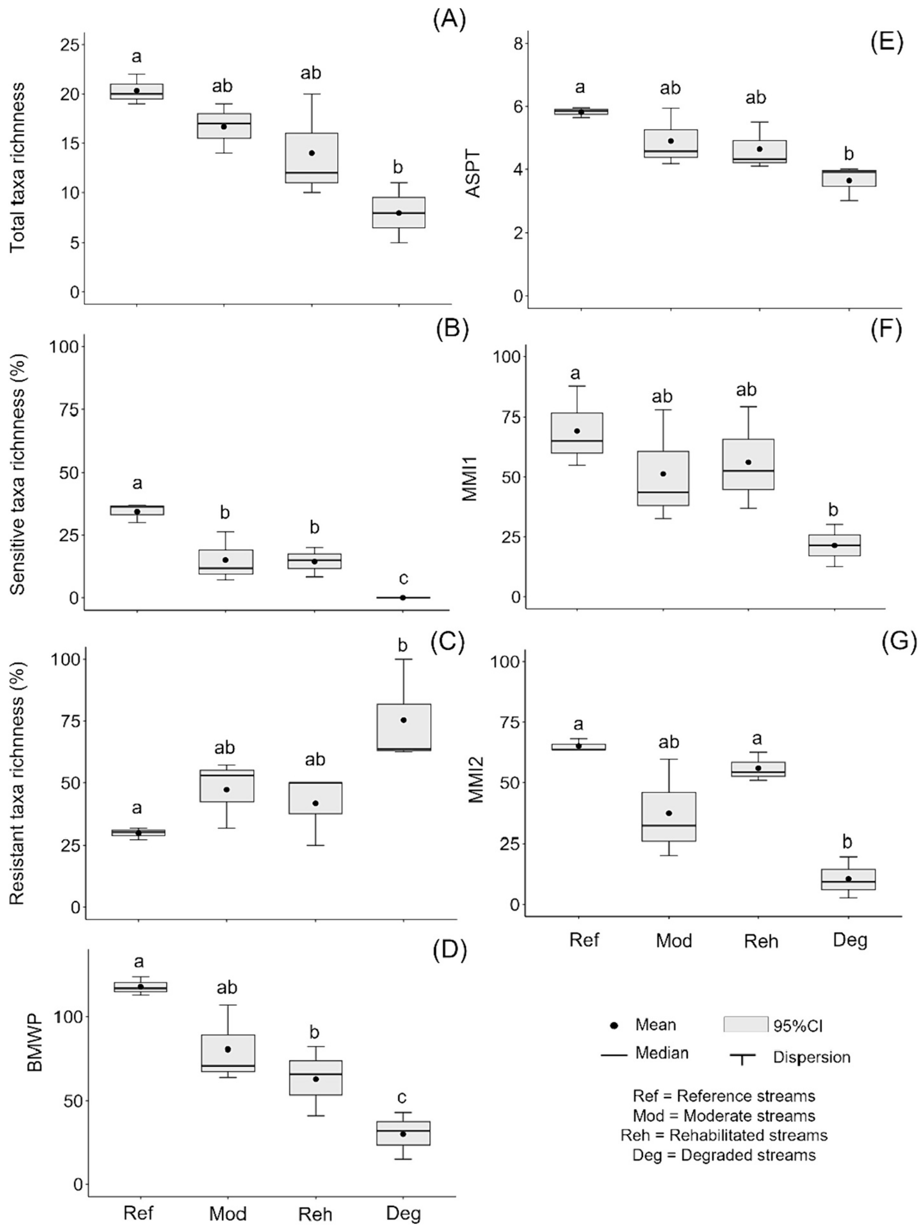

Fig. 4. Benthic macroinvertebrate metrics: A) Total taxa richness, B) % sensitive taxa richness, C) % resistant taxa richness, D) Biological Monitoring Working Party (BMWP), (E) Average Score Per Taxa (ASPT), (F) Macroinvertebrate Multimetric Index (MMI1) by Macedo et al. (2016), and (G) Macroinvertebrate Multimetric Index (MMI2) by Silva et al. (2017). Different letters indicate statistical differences among stream categories (one-way ANOVA followed by Tukey's test, $p < 0.05$).

4. Discussion

We confirmed the hypothesis that there would be a gradient of structural and functional conditions, going from reference to degraded streams. We found significant improvements in habitat diversity, primary production, sediment deposition, and percentage of sensitive macroinvertebrate taxa richness in rehabilitated compared with degraded streams. These results show the effectiveness of local rehabilitation actions in improving stream ecological condition, moving them from a degraded to a moderate condition. However, rehabilitated streams did not attain reference conditions, indicating no recovery to natural conditions, corroborating the idea that the return to a natural state (i.e., restored) in an urbanized area is an unrealistic goal (Hughes et al., 2014; da Silva and Porto, 2021).

In this sense, it is important to align ecological expectations with reality (Loflen et al., 2016). The impacts of urban areas on streams result from multiple, complex, point and diffuse sources combined with natural variability. Rehabilitation restores some structural and functional aspects, but at a small scale, as in this case, and it may not be enough to reverse the degradation state of an urban stream. Furthermore, without interventions upstream of the parks, streams continue to be exposed to stressors that may influence their structure and functioning.

4.1. Rehabilitation improves habitat structure

We expected that there would be a difference in water quality among stream categories (Ramírez et al., 2014; Macedo et al., 2022). However, few variables differed significantly, and mostly between reference and degraded streams. The absence of stronger water quality differences among stream categories may result from the continued effects of the urbanized landscape (Hughes et al., 2014). Urban effects such as surface runoff, groundwater contamination, and recreation continue to alter water quality (Fausch et al., 2002; Wiens, 2002; Paul and Meyer, 2008;

Table 3
Pearson's correlations between habitat diversity and macroinvertebrate variables. The expected sign of the correlations is indicated: +, indicates a positive correlation; −, indicates a negative correlation. Bold indicates statistical differences among stream categories ($p < 0.05$).

Metrics	r	p-Value	Expected
Total individuals' abundance	−0.27	0.39	−
Total taxa richness	**0.73**	**0.01**	+
Density of organisms per m^2	−0.27	0.39	−
Shannon-Winer diversity	**0.74**	**0.01**	+
Simpson diversity	**0.65**	**0.02**	+
Pielou Evenness	**0.67**	**0.02**	+
% EPT individuals	0.47	0.12	+
% Chironomidae individuals	−0.15	0.65	−
% Chironomidae + Oligochaeta individuals	−0.47	0.12	−
EPT/Chironomidae individuals	0.55	0.06	+
EPT/(Chironomidae + Oligochaeta) individuals	0.55	0.06	+
Sensitive individuals' abundance	0.49	0.10	+
Sensitive taxa richness	**0.89**	**<0.01**	+
Tolerant individuals' abundance	**0.70**	**0.01**	−
Tolerant taxa richness	0.10	0.76	−
Resistant individuals' abundance	**−0.63**	**0.03**	−
Resistant taxa richness	**−0.64**	**0.03**	−
BMWP	**0.76**	**<0.01**	+
ASPT	**0.65**	**0.02**	+
BMI	0.52	0.08	+
MMI1	**0.68**	**0.01**	+
MMI2	**0.76**	**<0.01**	+

EPT = Ephemeroptera + Plecoptera + Trichoptera.
BMWP = Biological Monitoring Working Party.
ASPT = Average Score per Taxon.
BMI = Benthic Multimetric Index.
MMI = Macroinvertebrate Multimetric Index.

Loflen et al., 2016). The rehabilitation programs, although integrating measures to control the entry of pollutants directly into streams (mainly domestic untreated sewage), are not capable of controlling indirect discharge, garbage entry, and surface runoff (Roni et al., 2008; Ríos-Touma et al., 2015; von Haefen et al., 2023). Also, rehabilitation at local scales may not be enough to improve water quality in a short stream reach; e.g., nutrient uptake requires longer distances when dissolved concentrations are high as in urban streams (Gibson et al., 2015). Furthermore, the forested area in rehabilitated streams was likely not enough to meet the minimum amount of vegetation needed to protect the streams (Brasil, 2012; Azevedo-Santos et al., 2019). In addition, small sample sizes and among-stream variability within our four stream categories hinder distinguishing clear water quality differences.

We expected to find a gradient in habitat diversity from reference to degraded streams. Our results showed significant differences among reference, rehabilitated, and degraded streams, while there were no differences between the rehabilitated and moderate streams, confirming our prediction. This indicates that local rehabilitation projects may achieve some improvements in rehabilitated stream habitat structure compared with degraded streams, although they do not equate with ecosystem recovery as differences between rehabilitated and reference streams are still found (Paul and Meyer, 2008; Wantzen et al., 2019). The full recovery of urbanized aquatic ecosystems is not achievable because the land use has changed and it is not possible to revert urban landscapes to forest landscapes.

Habitat diversity was significantly correlated with taxa richness, diversity, assemblage composition, and multimetric index scores. This indicates that improvements in habitat diversity, physical habitat structure, and riparian vegetation can lead to positive changes in benthic macroinvertebrate assemblages. This is because organic matter inputs (e.g., leaves, flowers, fruits, and wood) (Linares et al., 2021), channel complexity (Heino et al., 2018), and flow variations (Bouckaert and Davis, 1998; Calderon et al., 2023) influence the presence and absence of different organisms. Therefore, greater environmental heterogeneity facilitates the coexistence of more organisms with different preferences (McCreadie and Bedwell, 2013; Agra et al., 2021). Increasing the width and diversity of riparian vegetation cover along rehabilitated streams could further improve habitat diversity (all study streams obtained a minimum score in these parameters; Table S4) (Fonseca et al., 2021; Linares et al., 2021).

Although our visual assessment of habitat diversity was useful for monitoring streams and qualitatively assessing their structural conditions, it may have been insufficient to quantitatively measure changes in substrate and habitat (Kondolf, 1997; Kondolf and Lisle, 2016; Rubin et al., 2017). Therefore, we recognize that direct responses such as pebble counts and flow profiles should be assessed for a more complete evaluation of the rehabilitation efforts.

4.2. *Rehabilitation improves macroinvertebrate assemblage structure and composition*

The hypothesis that benthic macroinvertebrate assemblages would show better structure and composition in the reference streams, followed by the moderate and rehabilitated streams compared to the degraded streams, was partially supported. Biological rehabilitation occurred to a lesser extent than expected. Although the rehabilitated streams, in general, did not differ from the reference and moderate streams, they also did not differ from the degraded streams, except for sensitive taxa richness, and BMWP and MMI2 indices, indicating that they are in a moderate situation between degradation and reference.

Previous studies by Palmer et al. (2014) and Kail et al. (2015) showed that biological differences between environmental conservation categories are not always detected by richness and diversity indices, but rather by relative abundances or taxonomic composition. In our study, differences were only detected between the rehabilitated and degraded streams for the percentage of sensitive taxa richness. The increase in percentage of sensitive taxa richness is related to the improvement of environmental quality (Feio et al., 2015; Sterling et al., 2016). This result shows the importance of rehabilitation in recovering animal biodiversity, even if to a small degree. Similar results were found in urban regeneration projects in countries in the Global North, which showed mixed biological outcomes, with only 5 %–20 % showing significant biological improvements (Al-Zankana et al., 2020).

Of all biological indices tested, only BMWP and MMI2 detected differences between rehabilitated and degraded streams. These two indices, which quantify and combine several measures into a single indicator provide useful and easily communicated information as a basis for protecting and rehabilitating degraded environments (Statzner and Beche, 2010; Martins et al., 2022; Vadas Jr et al., 2022).

The Surber samples, although quantitative and carried out in three distinct habitat types (leaf deposits, coarse sediment, and fine sediment), are restricted and may not have been sufficient to consider the full diversity of stream habitats and the diversity of invertebrates that they may host (Miller et al., 2010). Furthermore, the lack of differences between reference, rehabilitated, and degraded streams may be linked to the legacy effect of urban disturbances that reverberate for a long time in communities, hindering their resilience (Allan, 2004; Camana et al., 2020; Linares et al., 2023). In addition, urban streams are not connected with other nearby reference streams that could provide potential colonizers. Thus, stream physical condition may have improved, but the absence of nearby sources of colonizers hinders substantial biological community recovery (Korsu, 2004; Parkyn and Smith, 2011; Zerega et al., 2021).

4.3. *Rehabilitation improves ecosystem functioning*

As expected, we found large differences between stream categories for chlorophyll *a* concentration, biofilm growth rate, and the autotrophic index. Previous studies found a relationship between increased periphyton biomass and increased nutrient input (Bourassa and Cattaneo, 2000; Feio et al., 2010). In addition, decreases in channel shading

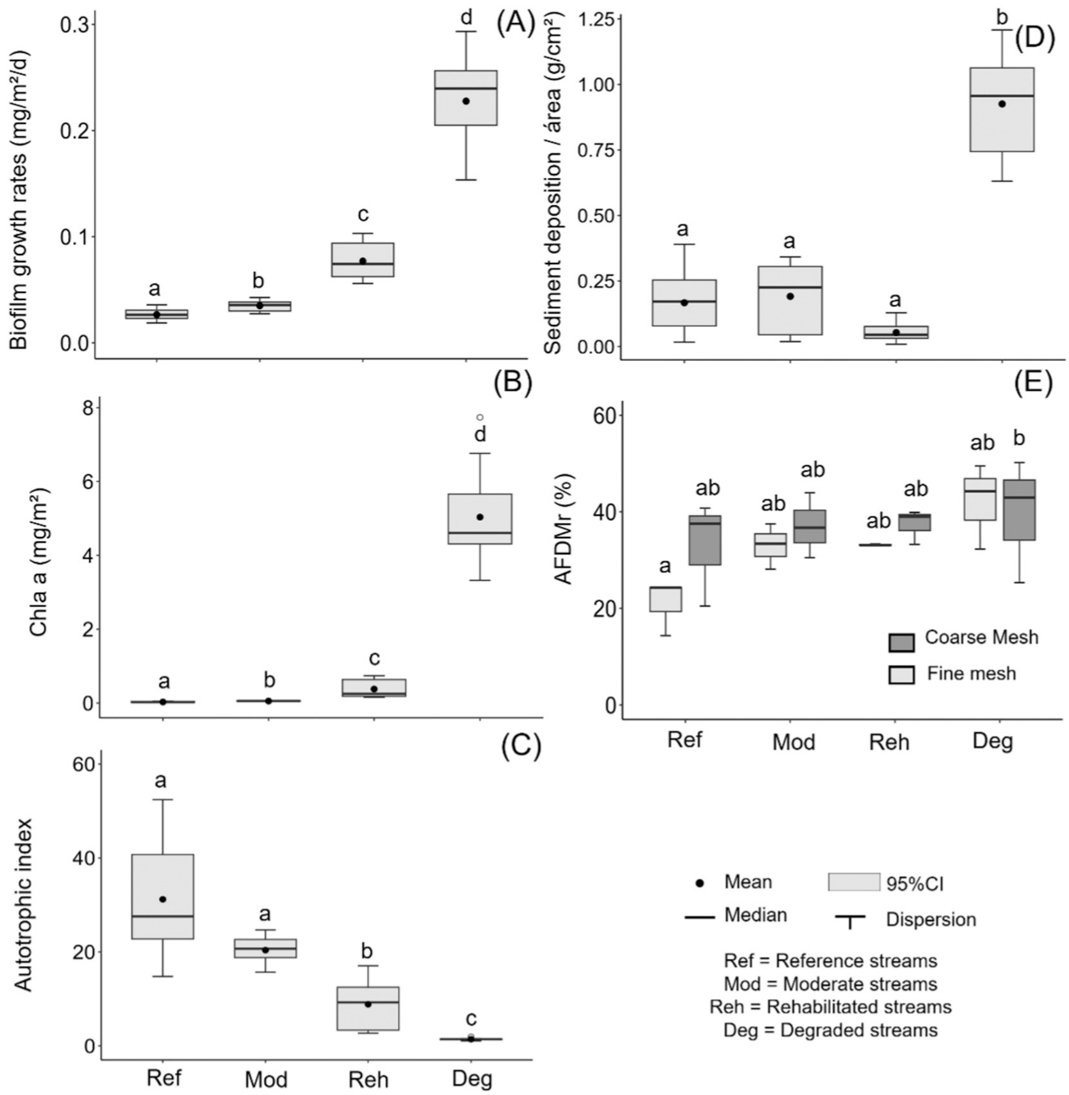

Fig. 5. Functional variables: A) Biofilm growth rate, B) Chlorophyll a concentration, C) Autotrophic index, D) Total sediment mass deposited per area, and E) Ash-free dry mass remaining (AFDMr) of *Bauhinia forficata* leaf litter in coarse- and fine-mesh bags after 60 days of stream incubation. Different letters indicate statistical differences among stream categories (one-way ANOVA (or two-way ANOVA in the case of panel E) followed by Tukey's test, $p < 0.05$).

often lead to increases in chlorophyll *a* concentration (Reisinger et al., 2019), which facilitates increased biofilm nutrient absorption (Burrell et al., 2014). We found the highest values of chlorophyll *a* and biomass in degraded streams, followed by rehabilitated and moderate streams, and the lowest values in reference streams. This is likely because of the greater nutrient enrichment, reduced vegetation cover, and increased substrate exposure to light in degraded streams.

We expected to find a significant increase in sediment deposition in degraded streams, followed by rehabilitated, moderate streams, and little sedimentation in reference streams. Previous studies have found a relationship between land use and the deposition and composition of the deposited sediment (Rosi-Marshall et al., 2016; dos Reis Oliveira et al., 2020). Interestingly, the rehabilitated streams had the least sediment deposition, possibly because the containment and stabilization works on the banks and in the catchments (Mount et al., 2002; Florsheim and Mount, 2003).

We expected to find less organic matter mass remaining (i.e., faster decomposition) in the reference streams, followed by moderate and rehabilitated, and more mass remaining in the degraded streams. Decomposition of organic matter is generally promoted by moderate nutrient enrichment (Woodward et al., 2012; Rosemond et al., 2015), presence of microbial decomposers (Gulis et al., 2019) and invertebrate shredders (Ferreira et al., 2006), and can be inhibited by water acidification (Ferreira and Guérold, 2017). Our results showed significant differences in mass remaining between reference and degraded streams. This may be associated with the reduced presence of shredders, which include taxa that are very sensitive to environmental disturbances, such as the increased nutrient load, metals, and channel homogenization, in the degraded streams (Piscart et al., 2009). However, we did not detect differences between rehabilitated or moderate streams and the other stream categories. Organic matter decomposition may not show clear responses in moderately altered environments if other confounding factors are at play (Ferreira et al., 2020), and identical decomposition rates can be observed at very different levels of nutrient loading (Woodward et al., 2012).

5. Conclusions

We found that after 15 years of intervention the rehabilitated streams were in better structural, biological, and functional conditions than the degraded streams. However, they did not differ significantly from the moderately altered streams, nor did their water quality differ significantly from the degraded streams. We conclude that rehabilitation is effective in improving sites from degraded status by improving ecosystem structure and function. Furthermore, the combined use of functional and structural indicators allowed an integrated assessment of stream ecological condition and distinguished differences between stream categories not detected by water quality indicators.

This study contributes data and critical multi-tool information about the relevance of environmental rehabilitation of urban streams in the third most populous Brazilian metropolis and offers more appropriate options for managing tropical urban streams. Future studies should focus on the rehabilitated streams to determine possible different recovery trajectories and evaluate seasonal effects of rehabilitation.

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CRediT authorship contribution statement

Karoline H. Madureira: Writing – review & editing, Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Verónica Ferreira:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Conceptualization. **Marcos Callisto:** Writing – review & editing, Writing – original draft, Supervision, Project administration, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.171935.

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