Ambio 2019, 48:867–878
https://doi.org/10.1007/s13280-018-1122-z

RESEARCH ARTICLE

# Student monitoring of the ecological quality of neotropical urban streams

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Received: 28 February 2018 / Revised: 30 July 2018 / Accepted: 31 October 2018 / Published online: 17 November 2018

**Abstract** Most Latin American demands for drinking water are in or near urban areas. However, population growth and untreated sewage disposal degrade water quality, with negative effects for biodiversity and ecosystem services. Mobilizing society to monitor quality of urban watercourses facilitates training and allows diagnosis that may further help implement mitigation and management strategies. Therefore, our research was conducted in a $> 4000\ km^2$ metropolitan region of high human influence. Urban water body assessments were conducted by 1965 teachers and students and their consistency validated by rigorous scientific methods. The assessments revealed degradation of physical habitat, water quality, or biology in 91% of the evaluated urban stream sites. Increased knowledge concerning environmental stressors and biological responses by local citizens may increase their participation in public policy development and implementation. We conclude that participatory scientific monitoring is a viable way for improving science education, increasing social participation, and improving the ecosystem services provided by urban watercourses.

**Keywords** Aquatic ecology · Citizen involvement · Education · Ecosystem conservation · School capacity · Scientific management

## INTRODUCTION

In Latin America, most sources of water for human uses are very close to or within urban areas, where high population densities cause serious environmental problems (Hardoy et al. 2013). In addition, urban population expansion has increased the demand for water resources, but it has not been accompanied by sufficient garbage and sewage disposal (Vörösmarty et al. 2000). This has resulted in increased water pollution levels, leading to increased human morbidity and mortality and reduced water availability for transportation and recreation (Booth et al. 2016). In addition to the impacts of disorderly population growth, effects of climate change will likely worsen the already poor environmental conditions and their dire consequences for the quality of human life (Ripple et al. 2017).

Ecological monitoring is frequently used in water body-assessment programs and is indispensable in the management of urban ecosystems (Hughes et al. 2014). Such programs offer tools for defining and implementing actions to forecast and mitigate environmental damages and to rehabilitate ecological conditions (Elosegui et al. 2017). Although agency monitoring programs can lead to effective solutions to environmental problems, they are also costly and technically challenging, requiring substantial investments imposing additional burden upon already scant economic resources and qualified personnel (Lovett et al. 2007), leading to substantial monitoring shortfalls in developing countries. Alternatively, ecological monitoring conducted by local communities can offer realistic and effective approaches for environmental monitoring as well as tools for raising awareness regarding possible solutions to local and regional problems (Andrianandrasana et al. 2005). Most importantly, when associated with standardized methodologies, these monitoring efforts are efficient at

**Electronic supplementary material** The online version of this article (https://doi.org/10.1007/s13280-018-1122-z) contains supplementary material, which is available to authorized users.

large-scale environmental diagnoses (Schmeller et al. 2017). Monitoring by local communities is particularly relevant in developing countries for addressing the major threats affecting natural resources at a more affordable cost (Danielsen et al. 2008).

The establishment of water body-monitoring programs at the community school level may be a practical alternative to overcome some of these economic and logistical problems, as well as governmental incapacities for improving management of urban watersheds (Angeler et al. 2014). Student monitoring provides an economic approach where levels of accuracy and reliability can be balanced against costs (Collins et al. 2012). Schools are potential sources of guaranteed, long-lasting personnel (Stapp 1978) through the engagement of students and teachers, who can become qualified and active in the process of increasing environmental diagnosis and encouraging improved governmental actions (Conrad and Hilchey 2011). Urban water management must evolve with a focus on addressing the challenges of providing good-quality water, maintaining ecosystem services, and controlling the undesirable consequences of human activities (Walsh et al. 2016). Increased access to information and training of students and teachers by research centers and universities leads to improved engagement of children and society in nature education and appreciation (Conrad and Hilchey 2011). High-quality public education can provide effective steps that communities can take to transition to sustainability, including a long-term sustainable human population (Ripple et al. 2017).

Changes in individual behaviors and increased social awareness are ways to minimize growing environmental problems in urban areas (Angelstam et al. 2013). Such changes can promote conservation and management actions through social participation in the production of scientific knowledge as practitioners of citizen science (Conrad and Hilchey 2011). Citizen science assumes that lay people can be effectively involved in data gathering and scientific research (Bonney et al. 2016). To apply this concept to the conservation of natural resources, we should move from a science of discovery to a science of engagement (Cooper et al. 2007). Citizen engagement in environmental diagnosis activities using scientific approaches amplifies the exercise of citizenship and provides a foundation that helps sustain valuable efforts in conservation and environmental management (Pullin et al. 2004). Therefore, citizen science activities can yield better strategies for mitigating impacts and implementing environmental conservation, when based on rigorous conceptual grounds and scientific methods (Kohler and Brondizio 2016).

In this study, we evaluated the relevance of citizen monitoring activities implemented by trained elementary, middle, and high school students for assessing the ecological conditions of urban streams. Specifically, we addressed the main question: Can community school students of various educational levels be trained to use simplified monitoring methods for human influences on the ecological quality of urban streams, such that their results are consistent with those of more rigorous scientific methods? Answering this question can shed light on whether and how citizen science and participatory monitoring can be used to build capacity in social actors and target populations that may in turn affect their future orientation and participation in public policy development and implementation.

## MATERIALS AND METHODS

### Study area and target audience

Stream sampling was conducted in Belo Horizonte, the sixth largest metropolitan district in Brazil, housing c. 5 million people and with a population density of 7 thousand inhabitants per km$^2$. The district, covering 4211 km$^2$, is located in the state of Minas Gerais, southeastern Brazil, in the upper reaches of the São Francisco Basin, the largest river basin entirely in Brazil. In addition to the metropolitan region, the area includes a concentration of industries and iron mining in the Paraopeba and Velhas River Basins (Fig. 1). The target community was the school-aged (elementary, middle, and high school) population (1810 students, 9–18 years old) and 155 teachers from public ($N$ = 51) and private ($N$ = 3) schools of 12 municipalities (Tables S1 and S2).

### Community school monitoring methodologies

From 2013 to 2017, 46 urban stream sites were sampled between two and eight times by different student/teacher teams. Scientific methodologies and concepts were adapted when we trained students to match their educational levels. The community student monitoring included a simplified physical habitat protocol, water-quality sampling, and benthic macroinvertebrate sampling and assessment (Fig. 2).

The community student physical habitat protocol (adapted from Callisto et al. 2002) was scored from local habitat characteristics at sites 50 m long and 10 m distant from both the left and right margins (50 m by 20 m total; Table S3). Assessed indices included three indices and 11 metrics: % stability of riparian vegetation (stream canopy cover, riparian vegetation extent, erosion, number of trees > 30 cm diameter at breast height); % habitat homogenization (siltation, bottom substrate composition

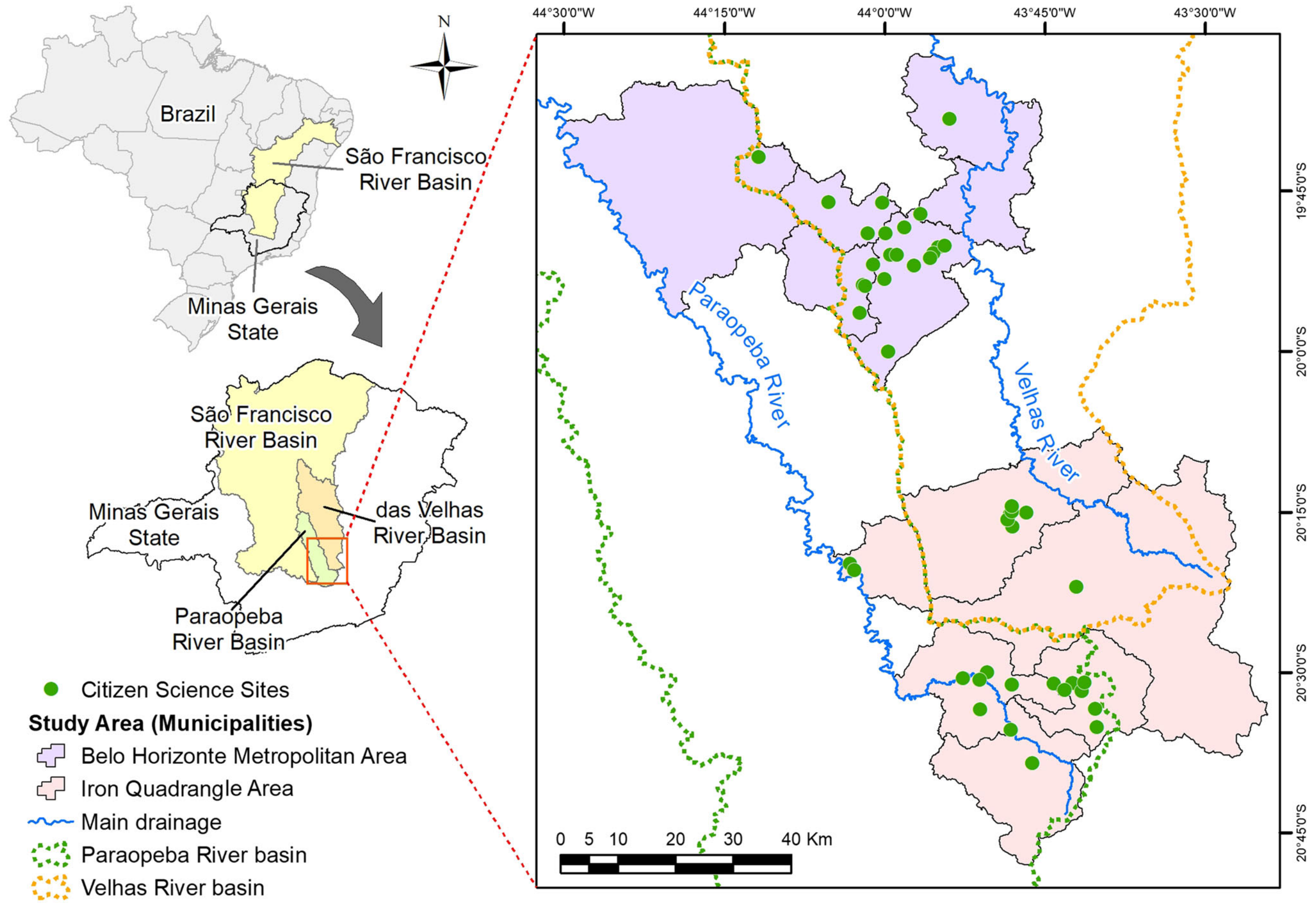


**Fig. 1** Locations of the 46 sites sampled by community school students in the São Francisco River Basin, southeastern Brazil. *Site geographical coordinates in Table S5

and diversity); and % local human impacts (garbage, water odor, untreated sewage disposal, pipes) (USEPA 2017). Each of the 11 metrics (plus an additional biological metric described below) were ranked from 0 to 8, receiving higher scores for better condition and lower scores for poorer conditions. Thus, the sum of all three index scores could vary from 0 to 88, and condition was classified subjectively into three categories: highly disturbed (0–40), moderately disturbed (41–64), or least-disturbed (> 64). Each index was scored on the average of the students' evaluations.

Water quality was measured by community students using colorimetric kits (www.alfakit.com.br) to assess temperature, pH, turbidity, dissolved oxygen, N-NH4, and P-PO4. Values were compared with the Brazilian environmental legislation (CONAMA 357/2005; Brasil 2005), which is the recommended framework in the state of Minas Gerais, Brazil (DN COPAM 001/2008; www.siam.mg.gov.br). Students considered the Class 2 limits for freshwater destined for: (1) human drinking after conventional treatment; (2) protection of freshwater biological communities; (3) water contact recreational activities; (4) irrigation; and (5) aquaculture and fisheries. The values considered in compliance with Brazilian legislation were pH (between 6 and 9), turbidity (< 100), dissolved oxygen (> 5 mg/L), N-NH4 (< 0.05), and P-PO4 (< 0.1).

Benthic macroinvertebrates were collected by 2 community students using hand nets (500-µm mesh, in a 1 $m^2$ area, for 10 min) and identified to order or phylum using field cards and the research team confirmed identifications in situ. Community students calculated the CS-BMWP (Biological Monitoring Working Party) considering sensitive taxa (score 10 for Ephemeroptera, Plecoptera, and Trichoptera), tolerant taxa (score 7 for Coleoptera and Megaloptera, and 6 for Odonata and Heteroptera), and resistant taxa (score 3 for Mollusca, 2 for Diptera, and 1 for Annelida). Students adapted the formula of Junqueira and Campos (1998) by including the total number of organisms. A site score was calculated using the formula:

$$\text{CS - BWMP} = \frac{\Sigma ni \times pi}{N},$$

where $ni$ number of individuals of taxon $i$; $pi$ pollution tolerance of taxon $i$; and $N$ total number of collected invertebrates.

**Fig. 2** Methodologies used in student monitoring: **a** simplified physical habitat protocol; **b** water-quality sampling (colorimetric method-Alfakit); **c, d** benthic macroinvertebrate sampling and data assessment

The final CS-BMWP result classification for each site was again classified into three categories: less disturbed (> 6 points, dominated by sensitive taxa); moderately disturbed (3–6 points, dominated by tolerant taxa), and highly disturbed (< 3 points, dominated by resistant taxa).

### Data analyses

The community student-collected indicators were compared against two independent metrics collected by university graduate students (Master and Ph.D. students from the Universidade Federal de Minas Gerais): population density (number per municipality area), and an integrated disturbance index and reference sites. We also evaluated the CS-BMWP and alternative biological metrics via an indicator-screening process (Stoddard et al. 2008; Silva et al. 2017).

The community-student sites were classified by human population density (source Brazilian Institute of Geography and Statistics—www.ibge.gov.br) into three disturbance categories: 11 highly urbanized sites (> 800 inhabitants per km$^2$); 16 moderately urbanized sites (> 100 to < 400 inhabitants per km$^2$); and 9 minimally urbanized sites (< 100 inhabitants per km$^2$). Ten sites (initially considered as having the best ecological conditions available, or being least-disturbed, sensu Bailey et al. 2004) occurred in urban protected areas (e.g., municipal parks, ecological stations, and private reserves).

An Integrated Disturbance Index (IDI) was calculated by combining measurements of local and buffer anthropogenic pressures (Ligeiro et al. 2013). To subjectively quantitate local disturbance (LDI), we used three types of disturbance (garbage, sewage, and odor), each scored subjectively into three classes according to high presence (0 points = 100%); average (4 points = 50%), or low or absent (8 points = 0%). We scored each of those three percentages as (0% = 0), (50% = 0.5) and (100% = 1.0). The final LDI score was the sum of the 3 measurements, ranging from 0 to 3 (low to high disturbance). To estimate a buffer disturbance, we determined a buffer disturbance index (BDI) by assessing land uses within 1 km upstream via interpretation of Google Earth images 2016 (Macedo et al. 2014). We based that distance on other research in our region that demonstrated the proximity of the local hydrologic unit or contributing area results in better water-quality predictions (Oliveira et al. 2017). The evaluation was based on three land uses (pasture/abandoned, agriculture, and urban). The BDI was calculated by the sum of land uses, weighted by the degradation potential of each on aquatic ecosystems (BDI = 4 × % urban + 2 × % agriculture + % pasture) (Rawer-Jost et al. 2004). The integrated disturbance index (IDI) was calculated from the LDI and the BDI. The IDI score for a site is the Euclidean distance between the site location and the origin of the disturbance plane formed by the LDI and BDI (standardized, Ligeiro et al. 2013); therefore, the higher the IDI score the greater the combined anthropogenic pressures on the sites. Because the BDI can range from 0 to 400 and the LDI from 0 to 3, we divided both by 75% of the maximum value each can achieve (Ligeiro et al. 2013). Therefore, we standardized the ranges of the two axes, by dividing BDI values by 300 and LDI values by 2.25, leading to:

$$\mathrm{IDI} = \left[\left(\mathrm{BDI}\ 300^{-1}\right)^2 + \left(\mathrm{LDI}\ 2.25^{-1}\right)^2\right]^{1/2},$$

where, we again classified the IDI scores into three classes (least-, moderately-, and highly disturbed) based on the break points in their IDI distribution.

We evaluated community students' scores for the physical and chemical quality variables against our IDI, BDI, and LDI scores. We first evaluated whether the protected urban areas defined a priori as having the best ecological conditions available were indeed not affected by anthropogenic impacts, considering the evaluation of physical habitat and water quality. For this purpose, we used a Pearson correlation matrix using water quality (dissolved oxygen, N-NH4, P-PO4 and turbidity) and physical habitat (% habitat homogenization and % riparian vegetation stability) variables versus the IDI, LDI, and BDI scores (Martins et al. 2017).

To evaluate the community students' selection of the CS-BMWP versus other candidate indicators, we followed the metric screening process used in multimetric index (MMI) development studies (Stoddard et al. 2008; Silva et al. 2017). We started with a set of 17 biological metrics calculated from the field data plus the CS-BMWP. We calculated CS-BMWP, richness, and abundance of benthic macroinvertebrates, % sensitive taxa, % resistant taxa, and % individuals of each insect order or phylum (Table S4). With these metrics, we sought to represent key aspects of site biological condition. The screening steps included a range test to eliminate metrics with small ranges or identical scores, a sensitivity test to eliminate metrics unrelated to site- or buffer-scale disturbance, a redundancy test to eliminate highly correlated metrics, and a stability test. We quantified the stability of an indicator by comparing its variance among sites (signal, S) with its variance between re-visits to the same sites (noise, N) (Kaufmann et al. 2014). The higher the signal-to-noise (S:N) ratio, the more stable the metric (Stoddard et al. 2008).

We also assessed the performance of the community-student CS-BMWP in two ways. First, we performed a simple linear regression between the CS-BMWP and the IDI. Finally, an ANOVA was performed between the three IDI ecological condition classes (least-, moderately-, and highly disturbed) and the CS-BMWP, followed by a Tukey Test to determine significant differences between IDI classes. The assumptions of residual normality and homogeneity of variances (Levene's Test) were verified, and data were log-transformed (log CS-BMWP) whenever necessary.

## RESULTS

### Student monitoring

The simplified physical habitat, water-quality, and biological assessments used by students scored few sites as least-disturbed. The simplified physical habitat assessment protocol scored 17 (37%) sites as highly disturbed ($< 40$ points, with 53% of the stream sites being in areas with a population density $> 800$ inhabitants/km$^2$); 21 (46%) as moderately disturbed (40–64 points, with 38% of the streams sites being in areas with 100–400 inhabitants/km$^2$); and eight (17%) as the least-disturbed ($> 64$ points, with 37.5% of the stream sites being under best ecological conditions available) (Table S5).

Evaluation of the water-quality parameters revealed that 25 sites (54%) had values that violated the limits established by the national regulation for Class 2 waters, including 40% of the sites occurring in urban protected areas. The water-quality parameter that most frequently violated the standards was P-PO4 (between 0.19 and 1.87 mg/L) in 50% of the sites. (Table S5).

The CS-BMWP classified 21 sites (46%) as highly disturbed ($< 3$ points; 43% of these were in areas with a population density of $> 800$ inhabitants/km$^2$), 19 (41%) as moderately disturbed (3-6 points with 37% of the ecosystems being in areas with 100–400 inhabitants/km$^2$) and six (13%) as least-disturbed ($> 6$ points). Those six sites included only three (33%) of the ten sites in urban protected areas (Fig. 3; Table S5).

### Data analyses

The graduate student LDI and BDI scores showed similar patterns, despite widely different score ranges. The LDI scores were high and varied widely among sites ($1.33 \pm 0.99$), with 61% of the sites having values above 1.0 and $> 50\%$ above the overall average ($> 1.3$). Likewise, BDI scores were high ($250.48 \pm 115.99$), with only six sites having BDI $< 100$ and 57% of the sites above the overall average ($> 250.0$) (Table S5). There was a low proportion of agriculture in the vicinity of the sites, with only five sites having $< 10\%$ of the area cultivated. On the other hand, pasture and abandoned areas were found in 36 sites, with six of them having $> 50\%$ pasture. The

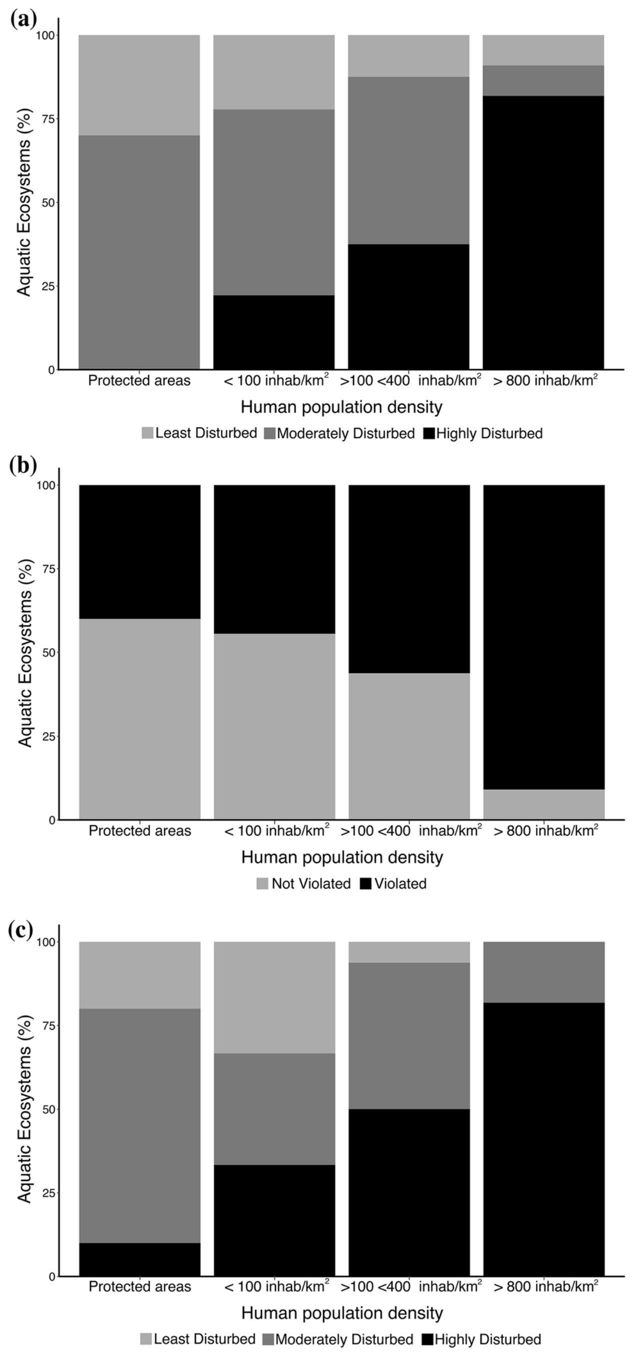


**Fig. 3** Indicators related to population density of the municipality: **a** physical habitat protocol; **b** water quality; and **c** CS-BMWP (Citizen Science-Biological Monitoring Working Party)

proportions of urban areas were considerably higher, being present in 100% of the sites with values between 2.1 and 97.1%. Most sites had > 50% urbanization (26 sites) in their vicinities, with eleven having > 90%. The Pearson correlation between the LDI and BDI was moderately significant ($r^2 = 0.42$; $p < 0.05$). Few sites occurred near the origin (reference sites) in the disturbance plane, and many sites occurred far from the origin because of high levels of anthropogenic influence at both local and buffer scales (Fig. 4a).

The large-scale estimates of levels of disturbance estimated by graduate students differed slightly from those of the community student measures of site condition. We classified 10 sites as least-disturbed (IDI < 0.50), 26 as moderately disturbed (IDI 0.51–1.45) and 10 as highly disturbed (IDI > 1.46) (Fig. 4b). Similarly, the estimates based on municipality population density classified nine sites as minimally disturbed, 16 as moderately disturbed, and 11 as highly disturbed. The student measures considered 10 sites in protected areas as being in the best ecological conditions available, but we considered only three based on the IDI distribution (Table S5). Few community student measures of local site condition correlated with our LDI, BDI, and IDI scores. However, the percentage stable riparian vegetation metric scored by the community students was significantly correlated with our LDI, BDI, and IDI scores, indicating the greater sensitivity of that metric and that those sites did not necessarily represent ideal ecological conditions (Table 1).

In the biological metric screening process, we reduced our initial set of 17 metrics to 12, excluding five metrics that had similar values for many sites (e.g., 60% sites with the value of 0) (Table S4). Next, we excluded eight metrics that failed to distinguish least- from highly disturbed sites in the responsiveness test ($t$-value < 5) and one redundant metric (Pearson $r > 0.6$). Finally, we found two metrics that were not correlated with LDI: % resistant organisms and % Mollusca. After eliminating 16 variables, our single final variable was CS-BMWP.

The signal-to-noise (S:N) test for CS-BMWP was 2.11, representing intersite variation two times greater than the sampling variation. Therefore, the selected CS-BMWP metric is considered an adequate (S:N > 1) indicator. The CS-BMWP was negatively related to the IDI ($r^2 = 0.48$, $p < 0.001$), corroborating the effectiveness of the student biological assessments (Fig. 5a). Three outliers occurred with relatively high IDI (and BDI) scores (high disturbance), but relatively high CS-BMWP scores (good biological condition). One site is an urban site that contained a few Baetidae, a tolerant taxon at the family level, but deemed a sensitive taxon (scoring 10) at the order (Ephemeroptera) level for the CS-BMWP. The other two sites were dominated by tolerant taxa (scoring 7 and 6) versus resistant taxa scoring 3, 2, or 1. Nonetheless, the CS-BMWP was significantly different between IDI categories, especially least-disturbed and highly disturbed sites ($F_{(2,43)} = 16.88$, $p < 0.001$) (Fig. 5B), further indicating the value of student biological assessments—even those based only on order—and phylum-level identifications.

**Table 1** Correlations between local disturbance index, buffer disturbance index, integrated disturbance index, and disturbance final classification scores versus physical and chemical habitat metrics at 10 sites in protected urban areas

| Physical & chemical habitat metrics | Local disturbance index | Buffer disturbance index | Integrated disturbance index | Disturbance final classification[a] |
|---|---|---|---|---|
| % Habitat homogenization | − 0.07 | 0.25 | 0.19 | 0.30 |
| % Stable riparian vegetation | − 0.58** | − 0.70* | − 0.73* | − 0.51 |
| Dissolved oxygen (mg/L) | 0.35 | 0.43 | 0.45 | 0.34 |
| Orthophosphate (mg/L) | 0.01 | 0.41 | 0.36 | 0.23 |
| Ammonia (mg/L) | 0.53 | 0.49 | 0.55 | 0.26 |
| Turbidity (NTU) | 0.42 | 0.50 | 0.52 | 0.19 |

*Correlation significant at $p < 0.05$; **correlation significant at $p < 0.1$

[a]Disturbance final classification after community student and graduate student evaluations: least-disturbed environment—all the parameters (students and academics) indicated low disturbance; moderately disturbed—some parameters (community or graduate students) indicated different levels of disturbance; highly disturbed—all parameters (community and graduate students) indicated high disturbance

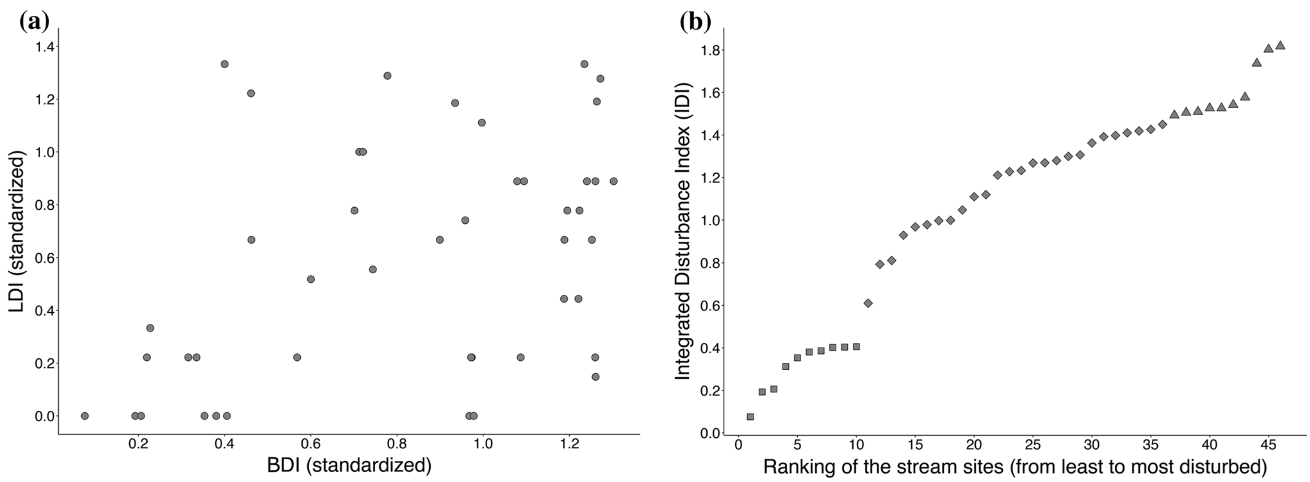


**Fig. 4** **a** distribution of sites in the disturbance plane where the axes of the local disturbance index (LDI) and the buffer disturbance index (BDI) were standardized to the same scale and, **b** disturbance gradient represented by ascending values of the integrated disturbance index (IDI). Squares: least-disturbed; diamonds: moderately disturbed; triangle: highly disturbed

## DISCUSSION

We demonstrated that student-participatory monitoring, through use of a simplified physical and chemical habitat protocol and a biological index at 46 urban stream sites was satisfactorily relevant for ecological assessment and related to the human pressures on their landscapes (Fig. 3). The students' water-quality measurements revealed that 25 sites (> 54% of total sites) violated the limits established by the Brazilian water-quality legislation, but 40 sites (87%) were deemed highly or moderately disturbed biologically. This, along with the local physical habitat results, indicates the importance of assessing physical habitat and biological condition as well as water quality (Hughes and Noss 1992; USEPA 2017). The sites initially scored as least-disturbed did not include 40% of the urban protected areas, providing strong evidence that those areas are negatively influenced by pressures from human activities in their surroundings. The validation of these simplified methodologies demonstrates that scientific research methodologies, adapted for use by community school students, can be effective in ecological assessments of urban water body quality, thereby potentially amplifying the scale in which monitoring can be undertaken (Figs. 4 and 5). Perhaps even more important socially and politically, ecological monitoring activities by young citizen scientists can help develop their knowledge of the consequences and needs for protecting and rehabilitating urban streams. From a bottom-up perspective, public participation enhances transparency and confidence in society and governance (Fraser et al. 2006).

Over the recent decades, Brazil has enacted public policies for establishing quality standards and river-monitoring programs through resolution CONAMA 357/2005 (Brasil 2005), which have contributed to improved water management. In addition, the federal government has

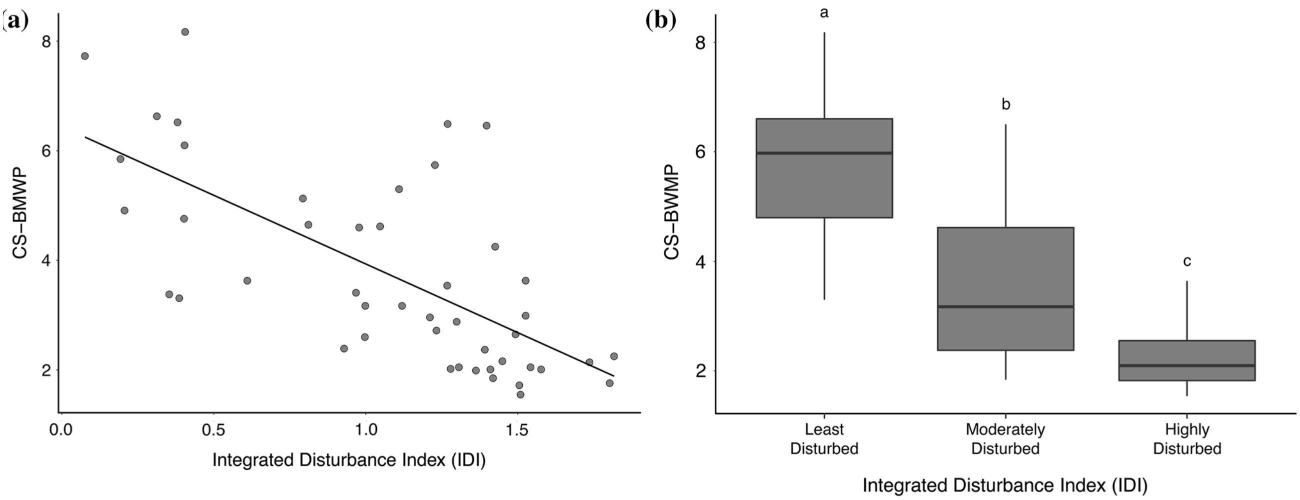


**Fig. 5** Integrated disturbance index versus CS-BMWP: **a** linear regression ($r^2 = 0.48$, $p < 0.001$) and **b** box plots representing the 25th and 75th percentiles; continuous lines within the boxes are medians; and whiskers are minimum and maximum values. Different letters indicate significant differences between all three classes (Tukey Test)

proposed to improve sewage-treatment systems throughout Brazil by 2030. However, good management policies and proposals are still not in line with the reality of current water-quality and quantity conditions. Approximately 18 million people lack access to safe water supplies, 93 million to sewage collection, and 14 million to solid-waste collection in Brazil (Razzolini and Günther 2008). The lack of execution of the proposed actions runs counter to the monitoring programs required in Brazilian environmental legislation. In addition, standard sampling and analysis methodologies, logistic strategies, training, and data dissemination are lacking, which hinder effective monitoring and assessment programs (Buss et al. 2008).

A standardized methodology and simplified information can help meet the demand for urban stream and river monitoring grounded in public participation. When student monitoring shows that over half of the monitored urban sites violate water-quality standards established by environmental legislation, it calls attention to the question of the sites' suitability as human potable water supplies. This situation demonstrates the urgency for management actions and a more participative population to improve urban watercourses in Brazil. Brazilians urgently need to popularize the perception that they can improve their quality of life through increased scientific information and public participation.

Although expected to maintain ecosystem services in urban areas, even the best ecological conditions undergo intense pressure from anthropogenic activities with high potential for ecological degradation. The community students determined that only a third of the reference sites (urban parks) used in this study are relatively free from urban pressures in their surroundings. Thus, even protected urban areas are not effective buffers of human pressures (Güneralp et al. 2015). Mcdonald et al. (2008) argued that the consequences of continued rapid urban growth on biodiversity conservation are poorly known and will alter ecoregions, rare species, and protected areas. These results indicate the difficulty of maintaining or improving the quality of ecosystem services such as providing good-quality water and maintenance of aquatic life at current treatment and protection levels (Walsh et al. 2016).

Multimetric biotic integrity indices have become important tools for ecosystem assessments since the time first proposed by Karr (1981), especially when they involve careful selection of biological metrics (Stoddard et al. 2008). They have been employed for making continental-scale aquatic ecosystem assessments (Stoddard et al. 2008) as well as for neotropical headwater streams in Bolivia (Moya et al. 2011), Brazil (e.g., Silva et al. 2017), and Chile (Fierro et al. 2018). Our selection of a simplified CS-BMWP index for urban streams explained 48% of the variation in the IDI. Ligeiro et al. (2013) found similar results in less densely populated areas with ~ 40% EPT (Ephemeroptera, Plecoptera, Trichoptera) richness, explained by an IDI. Although our results are supported by such published scientific studies, we assume that further research and improved training are needed to increase the explanatory power of the CS-BMWP and our IDI.

In addition to verifying the validity of participatory scientific monitoring, it is important to note that students can have a valuable educational experience. The implementation of new learning approaches, using practical outdoor techniques, provides pedagogical improvements needed for current and future generations (Ripple et al. 2017). These tools encourage and provide arguments for students, teachers, and their social circles to discuss and demand better management and governance of the environmental quality of urban streams by public authorities and other citizens. In this way, participatory monitoring could provide improved information for decision-making, appropriate use of resources and conflict resolution, and improved urban life (Fraser et al. 2006).

Similar participatory methodologies have been used successfully in several other areas and countries: (1) sustainability indicators (Fraser et al. 2006); (2) migratory species (Singh et al. 2014); and (3) river rehabilitation (Hughes et al. in press). Clearly, there are disadvantages and advantages of water body monitoring by students: (1) Student status is temporary, but it is balanced by the fact that annually more young people are trained. (2) Not everyone participating can be comfortable with the work, but the group-effect buffers these discomforts. (3) Data may sometimes not be reliable enough to be used effectively, but we have demonstrated otherwise. (4) We had initial resistance in some schools, but none of the 54 schools withdrew during the 5 years of the project. Therefore, we recommend applying this methodology over a wider geographic range that would involve less-disturbed (rural) sites and greater spatial balance across the basin, which would offer more robust data.

Participatory water-quality monitoring can be an initial step forward in stimulating citizens to have a greater voice in governance and public policies by reducing deficits generated by urbanization and economic growth (Angelstam et al. 2013). The main strength of our study was to show that the monitoring data collected with very simple protocols and executed by school students are relevant and can be an incentive for Brazilian cities to attain the water-quality and sanitation standards common in North America, Europe, and Australia. Participatory monitoring by community schools has made it possible to detect water-quality problems that create difficulties for water supplies for human use and demonstrate problems in basic sanitation with sufficient clarity to lead to more reasonable solutions in a timely manner. Participatory monitoring is now a reality for 54 schools, 1810 students, and 155 teachers in an area of 4211 $km^2$ of the São Francisco River Basin. We are aware that our contribution represents a first step in demonstrating that these methodologies can be efficient and effective means to help focus citizen participation in public policies and management of urban streams.

What we propose here is a simplified and flexible methodology that has proven to be scientifically valid and can be effectively used to overcome the huge governmental monitoring shortfall of water bodies in South America and other developing regions. In the future, we plan to offer reliable training to greater numbers of the school-age population. If successful, a skilled portion of the population will become aware of the importance of urban and rural streams for maintaining the quality of life. From this acquired knowledge, part of the population will be able to exercise its citizenship through social pressure on the decision makers and sanitation companies. Their actions will help us reach the 2030 target, that 100% of the Brazilian population, including megacities, have access to basic sanitation. From then on, we may, in the near future, aim at quality urban rivers that reach the goals established by law and standards similar to more developed countries. However, a more rigorous assessment of the impact of citizen science on student participation in local governance would require social science research that objectively tests the social and governmental impacts of student monitoring. We have not yet reached this step in our study, but it will be the next challenge so that we can achieve the international guidelines for basic sanitation.

## CONCLUSIONS

We demonstrated that by means of well-grounded scientific methods adapted to match their abilities, young students can be trained to evaluate environmental quality and monitor urban water body condition. Using citizen-science-adapted methods to train elementary, middle, and high school students, we demonstrated an efficient and enjoyable foundation for environmental monitoring and evaluation of urban streams. The implementation of participatory monitoring programs in schools can be an effective and economically viable tool to change social perceptions regarding environmental issues. Nonetheless, we believe that future student-participatory monitoring can be improved by employing more quantitative physical habitat metrics. For example, quantitative estimates of substrate composition, stream habitat complexity, stream canopy cover, riparian vegetation complexity, and riparian human disturbances (USEPA 2017) will contribute to more robust ecological assessments by students. Likewise, the use of more quantitative and comprehensive GIS catchment disturbance and natural indicators by graduate students will facilitate better understanding of the relationships between anthropogenic pressures and stream site responses (e.g., Ligeiro et al. 2013; Macedo et al. 2014). We believe, therefore, that citizen science activities, through the participatory monitoring with community schools demonstrated in this study, are a first step for increasing knowledge about, and improving, urban streams (Fig. 6).

**Acknowledgements** We thank the schools for their partnership, logistical support, and enthusiasm. Diego Macedo, Déborah Silva,

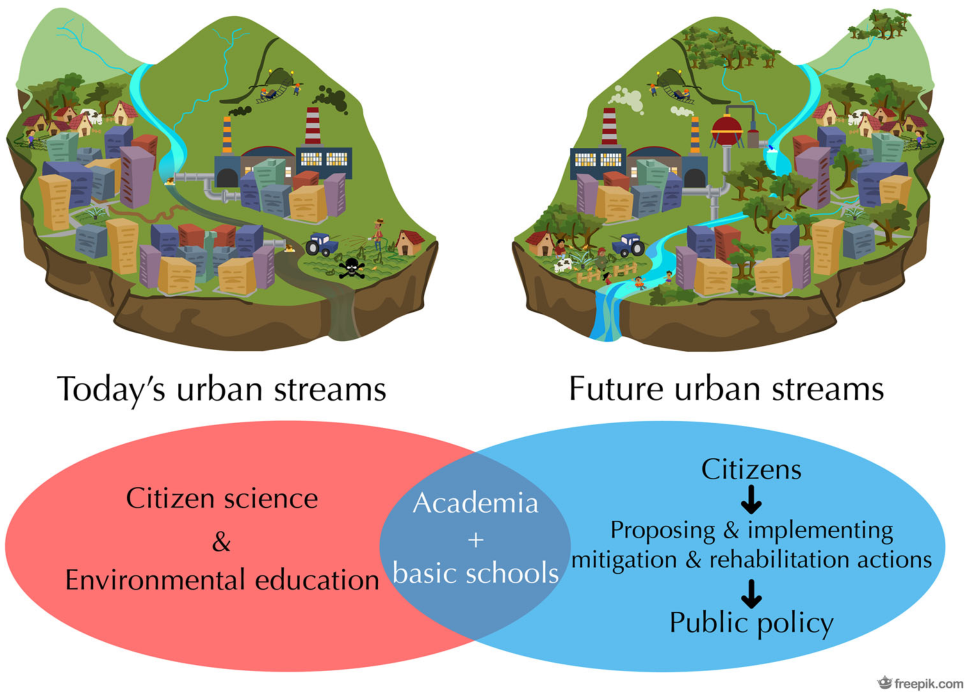


**Fig. 6** Current and potential future conditions of urban streams following citizen science activities and community participation in public policies and urban management (Credits: Stephanie2212 / Freepik)

Raphael Ligeiro, Maria Auxiliadora Drumond, and Rafael Leitão provided very useful comments and suggestions on our earlier version. The team of undergraduate and graduate students of the Laboratório de Ecologia de Bentos/ICB-UFMG supported the field activities and confirmed taxonomic identifications. This research was funded by the Ministério da Educação/Secretaria de Educação Superior (PROEXT 2013); Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES)-Support Program for Events in the Country (23038.006993/2013-24); Fundação de Apoio à Pesquisa do Estado de Minas Gerais (FAPEMIG; APQ-03402-14); Programa Gerdau Biocentro Germinar; and Programa de Pesquisa e Desenvolvimento Tecnológico do Setor de Energia Elétrica-Companhia Energética de Minas Gerais (P&D Aneel-Cemig GT-487, GT-599). J.F. was funded by a Ph.D. CAPES-DN fellowship; RMH received a Fulbright Brasil Grant; MC was awarded research productivity (CNPq 303380/2015-2), Research Project (CNPq 446155/2014-4), and Minas Gerais research (FAPEMIG PPM-IX 00525-15) Grants. We also thank Bo Söderström, Georgia Destouni and two anonymous reviewers for the excellent comments that improved this manuscript.

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