



Evaluating land use impacts on water quality: perspectives for watershed management

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Abstract

Human activities threaten the integrity of watersheds. We aimed to investigate the impact of land use on water quality, adopting a multiscale approach. We collected water samples from twelve streams in Southern Brazil and conducted limnological analyses (physical, chemical, and biological) during the dry season. We used the water quality index based on the quality standards of Canada and Brazil. Land use percentage was measured in two groups (local scale and network scale). Environmental variables were summarized through Principal Component Analysis, and we organized them into Linear Models, integrating the percentage of land use classes and terrain slope in the Multifit formula. Statistical analyses were performed using the R software. Results indicated contamination by lead, chromium, copper, nitrogen, and *Escherichia coli* in water samples. The Canadian Water Quality Guidelines for the Protection of Aquatic Life resulted in an index ranging from 23.3 to 47.3, compared to the Brazilian Resolution No. 357/2005 for Class 2, which had an index ranging from 47.5 to 100. This disparity is attributed to the more rigorous and sensitive monitoring approach adopted by the Canadian guidelines. Riparian forests which are up to 50 m wide are associated with improved water quality. Agricultural and urban activities were the main contributors to water quality degradation in an area extending up to 1000 m from the watershed. We emphasize the importance of a multiscale approach in watershed management and public policies, considering not only riparian forest preservation, but also human activities throughout the watershed. It is crucial to prioritize science-based environmental public policies and strengthen enforcement to prevent increasingly pronounced environmental collapses. We have identified the urgency to reformulate CONAMA Resolution No. 357/2005 with a more conservationist and ecosystem-oriented approach, as well as to propose modifications to the Brazilian Forest Code, particularly regarding the buffer zones of permanent preservation areas. Thus, this study can provide insights, such as incorporating the “effect scale,” to enhance water resource management in landscapes heavily influenced by human action, contributing to the advancement of future research in freshwater ecosystems.

Keywords Environmental impacts · Water quality index · Aquatic ecosystems · Streams · Geoprocessing · Environmental legislation

Abbreviations

WQI	Water Quality Index	CCME 2007	Canadian Water Quality Guidelines for the Protection of Aquatic Life
GIS	Geographic Information System	CONAMA 357/2005	Resolution CONAMA No. 357, dated March 17, 2005
WQI-CCME	Canadian Council of Ministers of the Environment Water Quality Index	SRTM	Shuttle Radar Topography Mission
CONAMA	National Council for the Environment	DEM	Digital Elevation Model
		PCA	Principal Component Analysis
		AIC	Akaike Information Criterion
		F_1	Scope
		F_2	Frequency
		F_3	Amplitude
		nse	Sum of excursions
		Temp	Temperature
		Turb	Turbidity

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Extended author information available on the last page of the article

Cond	Conductivity
TDS	Total dissolved solids
ORP	Oxidation/reduction potential
TN	Total nitrogen
TP	Total phosphorus
DO	Dissolved oxygen
BOD	Biochemical Oxygen Demand
COD	Chemical Oxygen Demand
Cd	Cadmium
Pb	Lead
Cr	Chromium
Zn	Zinc
Ni	Nickel
Cu	Copper
TC	Total coliforms
EC	<i>Escherichia coli</i>
<i>E. coli</i>	<i>Escherichia coli</i>

Introduction

Economic expansion and increased demand for consumer goods are determining factors for environmental degradation (Hughes et al. 2023). This phenomenon has particularly adverse effects on aquatic ecosystems, as watersheds serve as connecting pathways for landscape-wide impacts (Cionek et al. 2021; Vieira et al. 2022). Urban areas that dump untreated wastewater into rivers and streams cause metals and organic substances to build up artificially, which leads to eutrophication and bioaccumulation in the ecosystem's food webs (Singh et al. 2017a; Vieira et al. 2017; Yu et al. 2020; Eissa et al. 2022; Kumar et al. 2022). The practice of agriculture and cattle farming has been found to have a significant impact on nutrient concentrations in soil, which can subsequently leak into waterways. This process ultimately results in a drop in the amount of dissolved oxygen in water, primarily owing to heterotrophic metabolism. This reduction in dissolved oxygen levels creates less favorable conditions for the majority of aquatic organisms. These findings have been supported by studies conducted by Leip et al. (2015) and Mills et al. (2017). Furthermore, the combined impact of deforestation and climate change has significant implications for the hydrologic cycle of watersheds. This is evidenced by research conducted by Kumar et al. (2022), which highlights the increased presence of impermeable surfaces and the resulting structural changes in waterways, including canalization, burial, and rectification. As a result, waterscapes may encounter either a shortage of water or instances of excessive runoff (Singh and Panda 2017; Xue et al. 2017).

The aforementioned studies conducted by Mello et al. (2018), Vieira et al. (2019a,b), Ramião et al. (2020), and Ahmad et al. (2021) suggest that the implementation of

significant alterations is likely to result in a deterioration of water quality in streams. Consequently, there is a range of endeavors underway to evaluate the caliber of aquatic habitats and their impacts on aquatic organisms (Alexandre et al. 2010; Cunico and Gubiani 2017; Alvarenga et al. 2021; Raji and Packialakshmi 2022; Souza et al. 2023). The utilization of water quality indices (WQI) and the integration of remote sensing and Geographic Information System (GIS) tools are imperative for the surveillance of these ecosystems (Srivastava et al. 2012; Batbayar et al. 2019; Sharma et al. 2019; Gonino et al. 2020; Kheswa et al. 2021; Singh et al. 2021; Cicilinski and Virgens Filho 2022; Nguyen et al. 2022; Vieira et al. 2022). The Water Quality Index (WQI-CCME) developed by the Canadian Council of Ministers of the Environment is recognized as one of the most extensively utilized indices on a global scale. Its notable attributes include its adaptability to diverse local circumstances, as highlighted by several studies (Yan et al. 2016; Wagh et al. 2017; INEA 2019; Olanrewaju et al. 2021).

Water resource management policies are subject to regulation through the implementation of specific laws and recommendations in various countries (Silva et al. 2019). Several nations, including Australia, New Zealand, Canada, the United States, and the European Union, have established noteworthy water quality standards with the objective of safeguarding aquatic ecosystems (Nugegoda and Kibria 2013). Canada is widely acknowledged as a frontrunner in the field of water management, with its Canadian Water Quality Guidelines for the Protection of Aquatic Life (CCME 2007) being highly regarded as valuable instruments for evaluating aquatic ecosystems (Rosemond et al. 2009; Theodoro et al. 2016). The primary entity tasked with the regulation and management of water resources in Brazil is the National Council for the Environment (CONAMA) (OECD 2021). Water bodies are categorized based on their intended functions, as outlined in Resolution CONAMA No. 357, issued on March 17, 2005 (CONAMA 357/2005) (BRASIL 2005). Nevertheless, it is important to note that this particular methodology has the potential to expose deficiencies in the overall management and preservation of aquatic ecosystems across the nation (Silva et al. 2018; Padovesi-Fonseca and Faria 2022).

The impact of landscape composition and structure on water quality (Shi et al. 2017) and the structure of aquatic biota can manifest across several scales, encompassing both localized stream segments and large watersheds (Garofolo and Rodriguez 2022). Nevertheless, it is well acknowledged that the most noteworthy consequences typically manifest themselves within a distinct scale referred to as the "effect scale" (Jackson and Fahrig 2012; Miguet et al. 2016; Fletcher and Fortin 2018; Huais 2018). The escalating disagreements surrounding these features show that the relationship between biological response and the landscape has been a subject of

growing contention within the field of ecology (Miguet et al. 2016; Huais 2018; With 2019; Tomchenko et al. 2022).

According to Shi et al. (2017), the existence of riparian forests adjacent to waterways can serve as a filtration mechanism, leading to a decrease in surface runoff, the retention of sediments, and the processing of nutrients, ultimately resulting in an enhancement of water quality. Hence, the extent of the vegetation strip adjacent to riverbanks assumes a pivotal role in the conservation of aquatic ecosystems. Recent studies by Wang et al. (2020) and Shi et al. (2022) have found evidence that suggests a positive correlation between the width of this strip and its capacity to support conservation efforts. Due to the prevalence of landscape alteration in many places, there is significant variation in riparian width and length, as well as their ability to mitigate any adverse impacts resulting from changes in land use. According to Guidotti et al. (2020), it has been shown that narrow riparian buffers do not provide significant protection against erosion and the entry of silt into waterways. This deficiency has a detrimental impact on the physical, chemical, and biological attributes of streams.

One way to make sure that riparian forests and human land use activities can live together peacefully is to protect these forests within buffer zones that are big enough to protect and keep the aquatic environment working properly (Hilary et al. 2021). Several countries, including Brazil, Mexico, the United States, Germany, and Australia, have implemented legislation with the objective of safeguarding riverbanks within designated areas. Various economic interests frequently have an impact on the design of these zones (McDermott et al. 2009; Miguel and Velho 2013; Chiavari and Lopes 2017). Consequently, such circumstances may give rise to distortions and insufficient interpretations among decision-makers (Monte et al. 2021).

Regardless of significant efforts to comprehend the effect of the landscape on water quality, there are several voids in identifying the most appropriate criteria for monitoring and preserving watershed quality. Jackson and Fahrig (2012), Holland and Yang (2016), Miguet et al. (2016), to name a few, have all examined the influence of scale on biological communities in terrestrial ecosystems. In addition to employing a limited number of spatial scales, however, the effects of a multiscale approach on water quality are rarely examined in a majority of studies. Consequently, the findings of this study have considerable potential to guide and encourage the revision of public policies. Incorporating the scale of effect to prevent pollution impacts and assure the structure and function of neotropical aquatic ecosystems, particularly at the headwaters of watersheds, is one example of the information provided by these findings. These areas provide essential ecosystem services to human populations and are home to numerous species that, by carrying out their functions in the ecosystem (i.e., feeding and nutrient cycling), contribute to the provision of water of adequate quality and quantity

(Sá et al. 2013; Hilary et al. 2021). Thus, it is feasible to prevent future mass extinctions and water crises. This work contributes directly to the four Sustainable Development Goals (SDGs) established by the United Nations: (11) Sustainable Cities and Communities, (13) Climate Action, (14) Life Below Water, and (15) Life on Land.

The main goal of this research is to examine the impact of land utilization on water quality using a multiscale methodology. The objectives outlined are as follows: (i) to evaluate and compare the efficacy of various quality standards in the assessment of stream conditions, (ii) to examine the correlation between landscape characteristics and water quality, and (iii) to analyze the impact of several spatial scales on water quality. The paper posits the following hypotheses: (i) the efficacy of the CCME, 2007 guidelines in safeguarding aquatic ecosystems surpasses that of Resolution CONAMA 357/2005; (ii) the existence of forest vegetation within stream ecosystems exhibits a positive correlation with water quality, whereas areas characterized by agricultural and urban activities demonstrate a negative association with water quality; and (iii) the local scale exerts a greater influence on the enhancement of water quality.

Materials and methods

Study area

The study was conducted in 12 first- and second-order streams (Strahler 1952) that are part of the Pirapó and Ivaí River basins (Fig. 1). These streams are located in the urban and peri-urban areas of Maringá municipality, in the north-west region of the state of Paraná, South Brazil (Fig. 1). Maringá city is situated in the Third Paraná Plateau, between the coordinates 23° 25' S and 51° 57' W, with an average altitude of 555 m (IBGE 2021). With a population of just over 400,000 inhabitants and an area of 487,012 km² (IBGE 2021), Maringá stands out as the third largest city in the state of Paraná in terms of urbanization and demographic growth and has been recognized as one of the main agricultural centers in the country (Rodrigues 2004; Macedo 2011). According to Carfan et al. (2005), the area has a subtropical climate with an average annual precipitation of more than 1500 mm and an average annual temperature of between 18 and 22 °C. The local vegetation belongs to the Atlantic Forest biome, specifically the Seasonal Semideciduous Forest (IAT 2022a). According to Ghisi et al. (2016), the Pirapó River basin supplies the Maringá region and is in danger of deforestation because of the growth of agriculture and urban development (Rigon and Passos 2014; Cunico and Gubiani 2017). Similarly, the Ivaí River basin encompasses various anthropogenic landscapes and is primarily affected by increased land

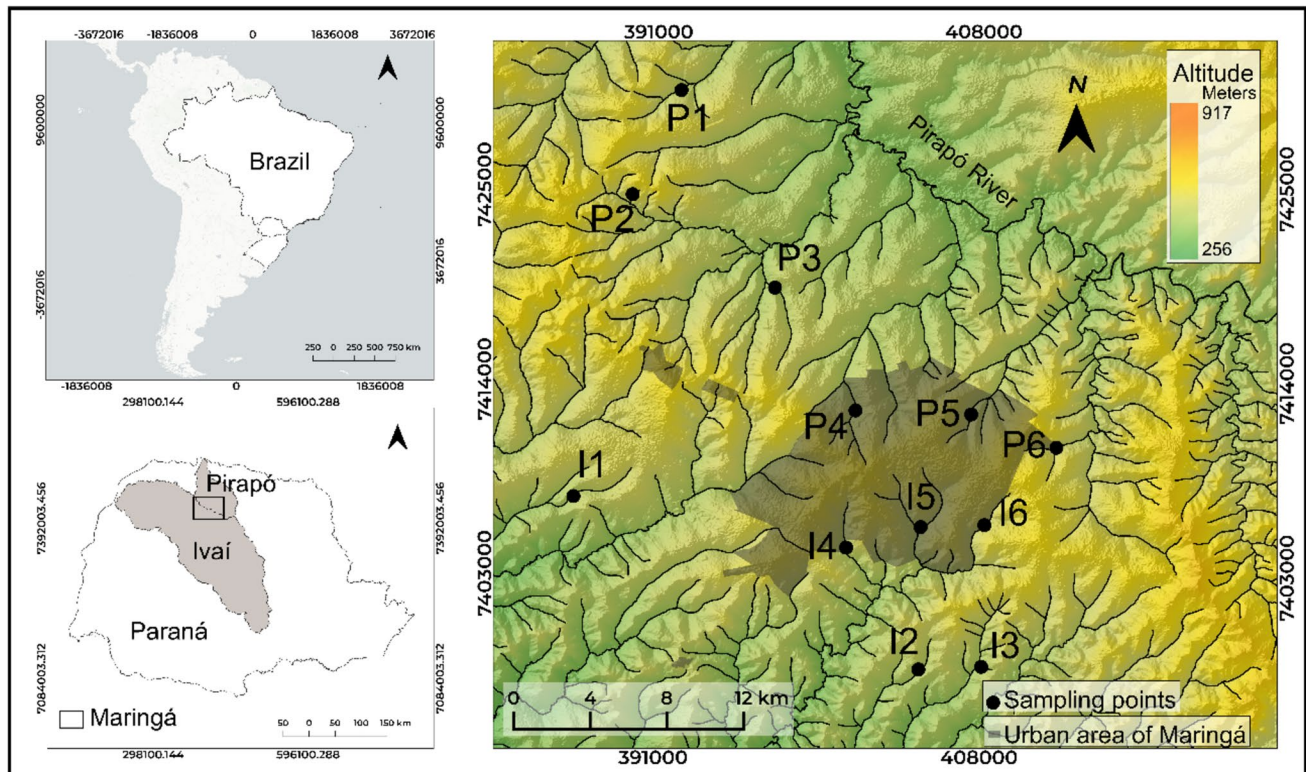


Fig. 1 Location of sampling points in the study area, covering the Pirapó (North) and Ivaí (South) River basins. Legend: (P_n): Pirapó basin streams, and (I_n): Ivaí basin streams. **P1**—Alto Alegre (23° 13' 55.2" S; 52° 03' 13.0" W); **P2**—Jaborandi (23° 17' 06.6" S; 52° 04' 42.8" W); **P3**—Atlântico (23° 19' 58.3" S; 52° 00' 20.9" W); **P4**—Maringá (23° 23' 44.3" S; 51° 57' 52.8" W); **P5**—Morangueira (23° 23' 51.6" S; 51° 54' 19.7" W); **P6**—Guaiapó (23° 24' 53.0" S;

51° 51' 43.0" W); **I1**—Colombo (23° 26' 21.7" S; 52° 06' 31.6" W); **I2**—Jaçanã (23° 31' 35.8" S; 51° 54' 01.5" W); **I3**—Jaguaruna (23° 31' 40.5" S; 51° 55' 57.0" W); **I4**—Borba Gato (23° 27' 56.4" S; 51° 58' 10.2" W); **I5**—Moscados (23° 27' 18.7" S; 51° 55' 52.8" W); **I6**—Pinguim (23° 27' 15.2" S; 51° 53' 55.6" W). Digital Elevation Model—NASA DEM, EPSG: 4674—SIRGAS 2000 DATUM, Transverse Mercator Projection

occupation, with agriculture being the main economic activity (Meurer et al. 2010; IAT 2022b).

Sampling procedures and laboratory analysis

Field sampling was conducted during the dry season in September 2021 for 6 days. During this period, we recorded 37.20 mm of accumulated precipitation (SIMEPAR 2021). This period was selected to minimize interferences caused by variations in precipitation and water flow during the sampling activities (Rosa et al. 2020). In each of the 12 streams (Figure 1), we selected a 50-m section (hereafter, referred to as the “sampling site”), where three measurements of each limnological variable were obtained. From these measurements, a representative average was calculated for each parameter. The variables of temperature (°C), pH, electrical conductivity ($\mu\text{S}/\text{cm}$), turbidity (NTU), ORP—Oxidation/Reduction Potential (mV), and dissolved oxygen (% and mg/L) were collected at a sampling point using a HORIBA

multiparameter probe (model U-50). Subsurface water samples (at a depth of 20 cm) were collected manually with (i) polyethylene bottles of 500 ml, (ii) falcon tubes of 50 ml, and (iii) previously labeled, sterilized, and preserved glass bottles of 200 ml. Samples were kept below 6 °C in thermal boxes and sent to the LASAM Laboratories at the State University of Maringá (UEM, Maringá/PR) and MERIEUX - NutriSciences (Curitiba/PR) for analysis.

Thirteen water quality parameters were selected as they are widely recognized and employed as indicators of pollution, resulting from agricultural, industrial, and urban activities practiced in the Pirapó and Ivaí river basins (Alves et al. 2008; Santos et al. 2008; Freire et al. 2012). The analyses of Total Nitrogen (TN), Total Phosphorus (TP), Biochemical Oxygen Demand (BOD), Chemical Oxygen Demand (COD), Cadmium (Cd), Lead (Pb), Chromium (Cr), Zinc (Zn), Nickel (Ni), Copper (Cu), and Total Dissolved Solids (TDS) were conducted at the MERIEUX laboratory. The concentrations of Total Coliforms (TC) and

Escherichia coli (EC) were determined at the LASAM laboratory. All analyses followed the methodologies described in the Standard Methods for Examination of Water and Wastewater, 22nd and 23rd edition—AWWA/APHA/WEF, EPA—Environmental Protection Agency methods (SW 846 series and others), NBR standards—Brazilian Association of Technical Standards (ABNT), and methods from the Environmental Company of the State of São Paulo (CETESB).

Water quality index (WQI-CCME)

We used the Water Quality Index (WQI-CCME) developed specifically by the Canadian Council of Ministers of the Environment (CCME) and adopted by the Water Resources Management Division of the Department of Environment and Climate Change of Newfoundland & Labrador (CWQI 2022). The WQI-CCME index numerically classifies the quality of a specific water body by comparing the analysis results with established limits for each parameter. For this study, we were interested in analyzing the outcomes of water quality results from two guidelines: (i) the Canadian Water Quality Guidelines for the Protection of Aquatic Life (CCME 2007) for freshwater and (ii) the Resolution CONAMA No. 357/2005 for freshwater/Class 2, which encompasses various water uses such as drinking water supply, protection of aquatic communities, recreation, irrigation, aquaculture, and fishing (CONAMA 2005). The classification procedure provides a response that reflects the environmental condition of the assessed environment (CWQI 2022). To calculate the WQI-CCME, three main factors were considered (F_1 , F_2 , F_3):

Scope F_1 represents the percentage of parameters that failed to meet the objectives of the limnological parameters during the period of interest, relative to the total number of parameters evaluated (CCME 2017):

$$F_1 = \left(\frac{\text{Number of failed parameters}}{\text{Total number of parameters}} \right) * 100. \quad (1)$$

Frequency F_2 represents the percentage of individual tests that did not meet the objectives, relative to the total number of tests (CCME 2017):

$$F_2 = \left(\frac{\text{Number of failed tests}}{\text{Total number of tests}} \right) * 100. \quad (2)$$

Amplitude F_3 is the extent to which the values of failed tests did not meet the objectives of the parameters. F_3 was calculated in three steps (CCME 2017):

- (i) The number of times an individual concentration exceeded (or fell below, when the objective is a

minimum) the target is called an “excursion” and is expressed as follows:

When the test value exceeded the parameter objective:

$$\text{Excursion}_i = \left(\frac{\text{Failed test value}}{\text{objective}} \right) - 1. \quad (3a)$$

For cases where the test value was below the parameter objective:

$$\text{Excursion}_i = \left(\frac{\text{Objective}}{\text{Failed test value}} \right) - 1. \quad (3b)$$

- (ii) The collective value by which individual tests did not meet the objectives, which is calculated as the sum of excursions of individual tests divided by the total number of tests (including both those that meet and those that do not meet these objectives), is referred to as the Normalized Sum of Excursions (nse) and was calculated as follows:

$$\text{nse} = \frac{\sum_{i=1}^n \text{excursion}_i}{\neq \text{ of tests}}. \quad (4)$$

- (iii) The F_3 amplitude was calculated using an asymptotic function that scales the nse to provide a range between 0 and 100.

$$F_3 = \left(\frac{\text{nse}}{0.01\text{nse} + 0.01} \right). \quad (5)$$

Finally, the WQI-CCME was obtained (CCME 2017):

$$\text{WQI} - \text{CCME} = 100 - \left(\frac{\sqrt{F_1^2 + F_2^2 + F_3^2}}{1.732} \right). \quad (6)$$

This index provides a numerical classification of water quality on a scale of 0 to 100, where 0 represents the poorest quality and 100 represents the best quality. This scale is divided into five classification categories, each associated with a specific color (red, orange, yellow, green, and blue), as presented in Table 1.

The WQI-CCME index used in this study incorporated 14 parameters (pH, Turbidity, Dissolved Oxygen, Nitrogen, Phosphorus, Biochemical Oxygen Demand, *E. coli*, Cadmium, Lead, Chromium, Zinc, Nickel, Copper, and Total Dissolved Solids). The Ivaí and Pirapó river basins were classified as Class 2 according to SUREHMA Ordinances No. 019/92 and 004/91.

Table 1 Water quality classification categories (WQI-CCME)

Categories	Index WQI-CCME	Description
Excellent	95–100	Water quality is protected with virtually no threat or harm; conditions very close to natural or pristine levels
Good	80–94	Water quality is protected only with a lesser degree of threat or impairment; conditions rarely deviate from natural or desirable levels
Fair	65–79	Water quality is generally protected but occasionally threatened or impaired; conditions sometimes deviate from natural or desirable levels
Marginal	45–64	Water quality is frequently threatened or impaired; conditions often deviate from natural or desirable levels
Poor	0–44	The condition of water quality is almost always threatened or impaired; conditions generally deviate from natural or desirable levels

Fonte: CCME (2017)

Landscape analysis

The Geographic Information System (GIS) was used for the manipulation and processing of geographic data. Through GIS, we mapped the water network and delineated the watersheds, then calculated the average percentage of terrain slope. Topographic information was obtained from the Shuttle Radar Topography Mission (SRTM) Digital Elevation Model (DEM) with a resolution of 30 m, using the Global Mapper software (Global Mapper 2017). Land use classes were delineated and processed using QGIS (QGIS Development Team 2020). High-resolution georeferenced images from 2018 were obtained from the BING Satellite (2020) and downloaded from the SAS. Planet software (2019) with a resolution of 2.5 meters. The technical land use manual from IBGE (2013) was used as a reference to classify land use types in the study area as forest, farming, and urban areas.

The average percentage of area for the three land use categories was calculated at five distinct scales (i.e., 30-, 50-, 100-, 200-, and 500-m buffers). These scales were defined based on the sampled stream reach (local scale). The average percentage of land use was also determined along the drainage basin upstream of the sampling point (network scale). In this case, six buffers of 30, 50, 100, 200, 500, and 1000 m were adopted (see explanation below, Fig. 2).

The buffers were initiated at 30 m, following the environmental protection limit established by the Brazilian Forest Code, Federal Law No. 12.651/2012 (BRASIL 2012), for waterways with a width smaller than 10 m. For the local scale, we established a buffer of up to 500 m in order to avoid exceeding the boundaries of the watersheds. For the network scale, on the other hand, we used a buffer of up to 1000 m, considering the upstream extension of the watershed from the collection point. The local slope was determined within the 500-m buffer, exclusively considering the geographical boundaries of the drainage basin. In turn, the slope of the basin was evaluated within a 1000-m buffer area.

Data analysis

To summarize the environmental variables (Appendix 1), Principal Component Analysis (PCA) was employed using the “prcomp” function. The variables were transformed to have zero mean and unit variance using the “decostand” function with the *standardize* method. Next, the significant PCA axes were selected using the broken stick method. These axes were used as dependent variables, while the percentage of land use classes and terrain slope (i.e., local and network) were used as independent variables in the Multifit function formula (Huais 2018). This function allows for the simultaneous execution of multiple statistical models to analyze a biological response in relation to different spatial scales, automating the multiscale analysis process (Huais 2018).

All independent variables and their interactions were considered for each scale group separately (i.e., local and network). Stepwise model selection was performed using backward and forward methods, and the model with the lowest Akaike Information Criterion (AIC) value that was statistically significant ($p < 0.05$) was selected. Model validation was conducted using the *gvlma* package (Pena and Slate 2019). The statistical analyses were conducted using R software, version 4.0.5 (R Core Team 2021), with *vegan* package (Oksanen et al. 2022) and *MASS* package (Venables and Ripley 2002).

Results

Water quality of streams

The investigated streams showed turbidity (Turb) above the limit established by the CCME 2007 (Table 2; Appendix 1). The concentrations of total nitrogen (TN) and total phosphorus (TP) exceeded the reference limits set by the CCME, 2007 and CONAMA 357/2005. High biochemical oxygen demand (BOD) was observed in the streams, surpassing the values established by the CCME 2007 and CONAMA 357/2005.

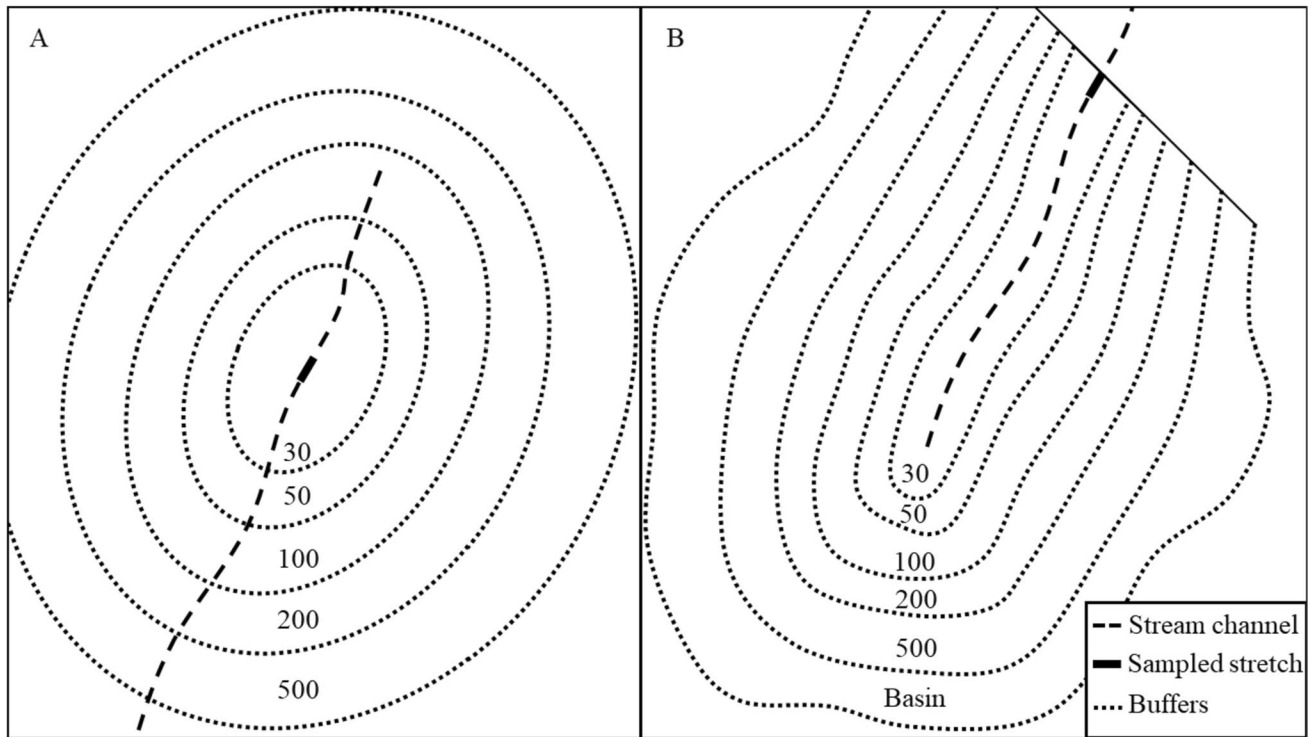


Fig. 2 Spatial scales generated from the study area grouped as follows: Local Scale (A), 30, 50, 100, 200, and 500-m buffers from the sampled stream reach, and Network Scale (B), 30, 50, 100, 200, 500,

and 1000-m buffers corresponding to extensions created from the upstream drainage basin of the sampling points

Table 2 Minimum (Min), maximum (Max), mean/standard deviation (SD), Canadian Water Quality Guidelines of the Protection of Aquatic Life (CCME, 2007), and Resolution CONAMA N° 357/2005 (CONAMA 357/2005) values for the evaluated limnological parameters in neotropical streams

Parameters	Unit	Min–Max	Mean ± SD	CCME 2007	CONAMA 357/2005
Temp	°C	22.19–26.31	24.02 ± 1.33	–	–
Turb	NTU	1.80–51.67	14.83 ± 15.30	5	100
Cond	µS/cm ²	0.03–0.25	0.14 ± 0.08	–	–
TDS	mg/L	0.02–0.17	0.09 ± 0.05	500	500
ORP	mV	152.67–312.67	255.53 ± 41.79	–	–
pH	–	6.86–7.82	7.40 ± 0.24	6.5–9.0	6.0–9.0
TN	mg/L	0.92–10.30	3.94 ± 2.85	0.1	2.18
TP	mg/L	0.00–0.56	0.13 ± 0.14	0.1	0.1
DO	mg/L	8.15–10.96	9.41 ± 0.90	> 5.5	> 5
BOD	mg/L	0.00–31.00	4.37 ± 8.65	4	5
COD	mg/L	3.10–107.00	15.30 ± 29.20	–	–
Cd	mg/L	0.00–0.00	0.00 ± 0.00	0.00002	0.001
Pb	mg/L	0.00–0.03	0.00 ± 0.01	0.001	0.01
Cr	mg/L	0.00–0.09	0.01 ± 0.03	0.002	0.05
Zn	mg/L	0.00–0.16	0.03 ± 0.04	0.03	0.18
Ni	mg/L	0.00–0.03	0.00 ± 0.01	0.025	0.025
Cu	mg/L	0.00–0.32	0.03 ± 0.09	0.002	0.009
TC	NMP/ml	> 2419.6	2419.6 ± 0	–	–
EC	NMP/ml	66.30–2419.60	817.43 ± 896.87	0	1.000

Temperature (Temp), Turbidity (Turb), Conductivity (Cond), Total Dissolved Solids (TDS), Oxidation/Reduction Potential (ORP), pH, Total Nitrogen (TN), Total Phosphorus (TP), Dissolved Oxygen (DO), Biochemical Oxygen Demand (BOD), Chemical Oxygen Demand (COD), Cadmium (Cd), Lead (Pb), Chromium (Cr), Zinc (Zn), Nickel (Ni), Copper (Cu), Total Coliforms (TC), Escherichia coli (EC)

The levels of total coliforms (TC) showed high concentrations exceeding 2419.6 MPN/mL. The presence of *Escherichia coli* (EC) bacteria was identified in all the analyzed streams. As a result, the streams do not comply with the limits set by both the CCME 2007 and CONAMA 357/2005.

Cadmium (Cd) was not detected in the streams. Zinc (Zn) (0.16 mg/l) exceeded only the limits of the CCME 2007. The maximum concentrations of lead (Pb) (0.03 mg/l), chromium (Cr) (0.09 mg/l), and nickel (Ni) (0.16 mg/l) exceeded both regulation limits. Copper (Cu) reached up to 0.32 mg/L.

Application and evaluation of the water quality index (WQI-CCME)

Considering the CCME 2007 guidelines, the water quality index ranged from 23.3 to 47.3 (Table 3; Fig. 3). Eleven streams were classified with “poor” water quality (23.3–43.5), and one stream with “marginal” water quality

(47.3). The concentration parameters of NT, Pb, Cr, Cu, and *Escherichia coli* were the ones that most influenced the IQA-CCME, with failures in the tests that exceeded the objective by more than 25 times. Scope and frequency were the same for each sample and did not vary. However, amplitude was consistently higher compared to other factors (Table 3).

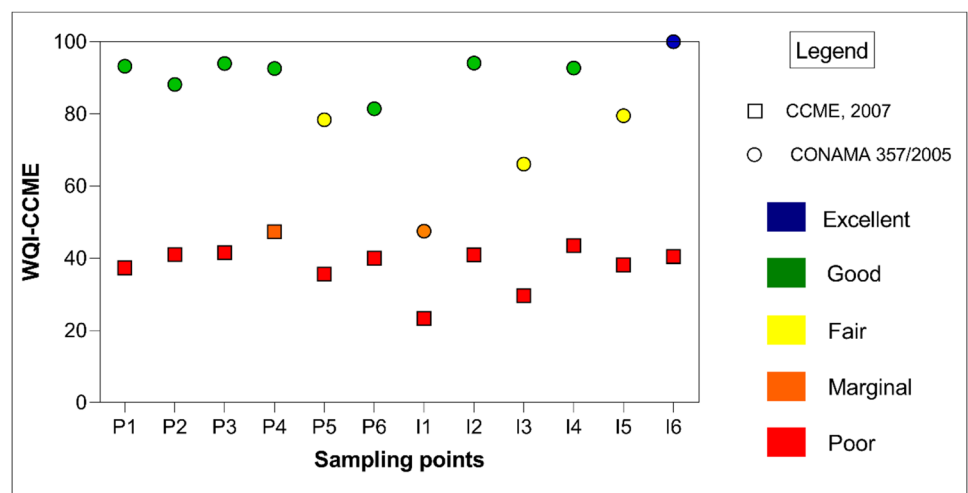
Considering the CONAMA 357/2005 guidelines, the water quality index ranged from 47.5 to 100 (Table 3 and Fig. 3), with seven streams classified with “good” water quality (81.4–94.1), one stream with “excellent” water quality (100), three streams with “fair” water quality (66.1–79.5), and one stream with “marginal” water quality (47.5) (Table 1). Scope and frequency were equal, while amplitude was frequently higher, with only copper exerting the greatest influence in decreasing the index (Table 3). Lastly, stream I1 (Colombo; Fig. 1) exhibited the most critical levels in the index for both CCME 2007 and CONAMA 357/2005 standards.

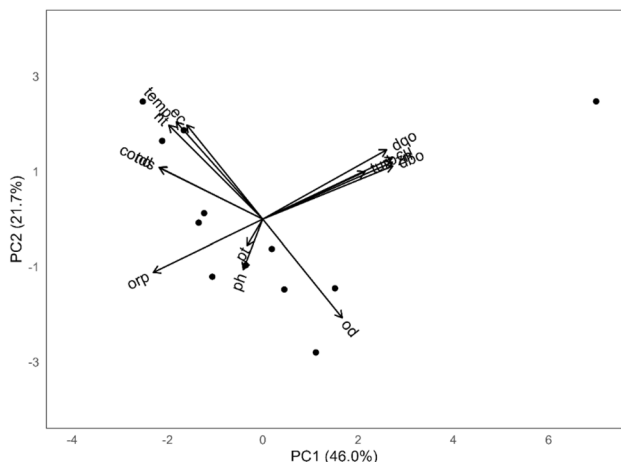
Table 3 Scope (F₁), Frequency (F₂), Amplitude (F₃), and WQI-CCME index values for neotropical streams according to the Canadian Water Quality Guidelines of the Protection of Aquatic Life (CCME 2007) and CONAMA Resolution No. 357/2005 (CONAMA 357/2005)

Points	CCME 2007				CONAMA 357/2005			
	F ₁	F ₂	F ₃	WQI-CCME	F ₁	F ₂	F ₃	WQI-CCME
P1	42.9	42.9	90.2	37.3	7.1	7.1	6.0	93.2
P2	28.6	28.6	93.9	41.0	14.3	14.3	3.3	88.2
P3	28.6	28.6	93.0	41.5	7.1	7.1	2.3	94.0
P4	14.3	14.3	89.0	47.3	7.1	7.1	7.9	92.6
P5	35.7	35.7	99.4	35.6	21.4	21.4	22.0	78.4
P6	21.4	21.4	99.3	40.0	14.3	14.3	25.2	81.4
I1	64.3	64.3	96.8	23.3	35.7	35.7	75.5	47.5
I2	28.6	28.6	94.0	40.9	7.1	7.1	1.3	94.1
I3	50.0	50.0	99.1	29.7	28.6	28.6	42.6	66.1
I4	21.4	21.4	93.1	43.5	7.1	7.1	7.7	92.7
I5	28.6	28.6	99.3	38.1	21.4	21.4	18.6	79.5
I6	21.4	21.4	98.6	40.4	0.0	0.0	0.0	100.0

Pn = streams in the Pirapó River basin; In = streams in the Ivaí River basin

Fig. 3 Water Quality Index (WQI-CCME) according to the Canadian Water Quality Guidelines of the Protection of Aquatic Life (CCME, 2007) and CONAMA Resolution No. 357/2005 (CONAMA 357/2005)





Graph 1 Biplot graph (PCA) based on environmental variables (temp = temperature; turb = turbidity; cond = conductivity; tds = total dissolved solids; orp = oxidation–reduction potential; ph; nt = total nitrogen; pt = total phosphorus; od = dissolved oxygen; dbo = biochemical oxygen demand; dco = chemical oxygen demand; zn = zinc; cu = copper; ec = *Escherichia coli*) in 12 neotropical streams (symbols in black)

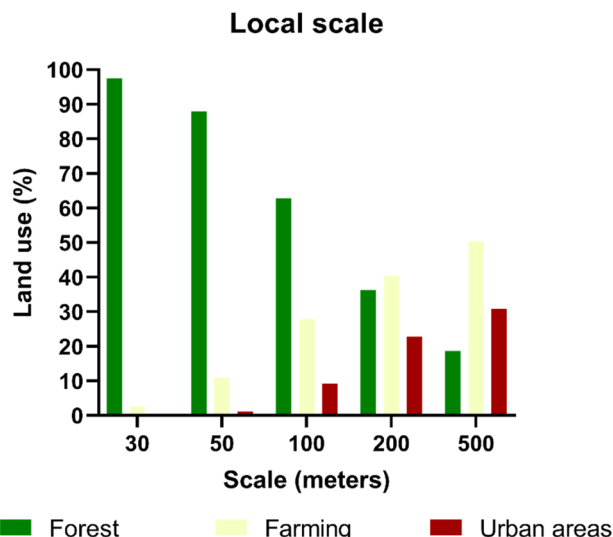
Limnological variables

The first two principal component analysis (PCA) axes were retained for interpretation, explaining 67.7% of the variation in the limnological data (Graph 1). The first axis (PC1) explained 46% of the total variation.

On the negative side, the variables that contributed the most were oxidation-reduction potential (ORP), total dissolved solids (TDS), and electrical conductivity (Cond). On the positive side, the significant contributors were chemical oxygen demand (COD), biochemical oxygen demand (BOD), and zinc level (Zn). The second axis (PC2) explained more than 21.7% of the variation, and the variables that most contributed to the variability of the streams were the levels of dissolved oxygen (OD), ORP, and pH on the negative side, and total nitrogen (TN), *Escherichia coli* (EC), and temperature (Temp) on the positive side.

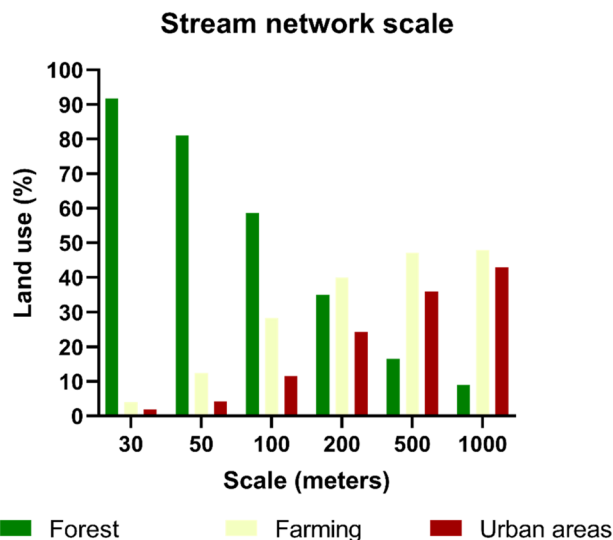
Analysis of land use at multiple scales

The local scale buffer with the narrowest widths (30, 50, and 100 m) were those with higher forest cover (up to 97.43%) (Graph 2; Figs. 4 and 5; Appendix 2). In all buffers, there was a negative relationship between the percentage of forest and the PCA axes. This means that streams with higher forest cover had higher values for variables such as ORP, TDS, Cond, OD, and pH. The effect of forest cover on the PC1 and PC2 axes was significant in the 50-m buffer ($p = 0.002$) and 100-m buffer ($p = 0.01$) (Appendices 3 and 4).



Graph 2 Average percentage (%) of land use in streams measured at the local scale (i.e., buffers of 30, 50, 100, 200, and 500 m)

The local scale buffer with the wider buffers (200 and 500 m) presented higher occupation of farming, reaching up to 50.31%. All buffers showed a positive relationship between the PCA axes and the percentage of agriculture. This indicates that streams with higher agricultural use had higher values of DQO, DBO, Zn, NT, EC, and temperature. The effect of agriculture on the PC1 and PC2 axes was significant in the 50-meter buffer ($p = 0.004$) and 100-meter buffer ($p = 0.02$). The average percentage of slope ranged from 7.08 to 9.67% for the local scale (Appendix 2).



Graph 3 Average percentage (%) of land use in streams measured at the stream network scale (i.e., buffers of 30, 50, 100, 200, 500, and 1000 m)

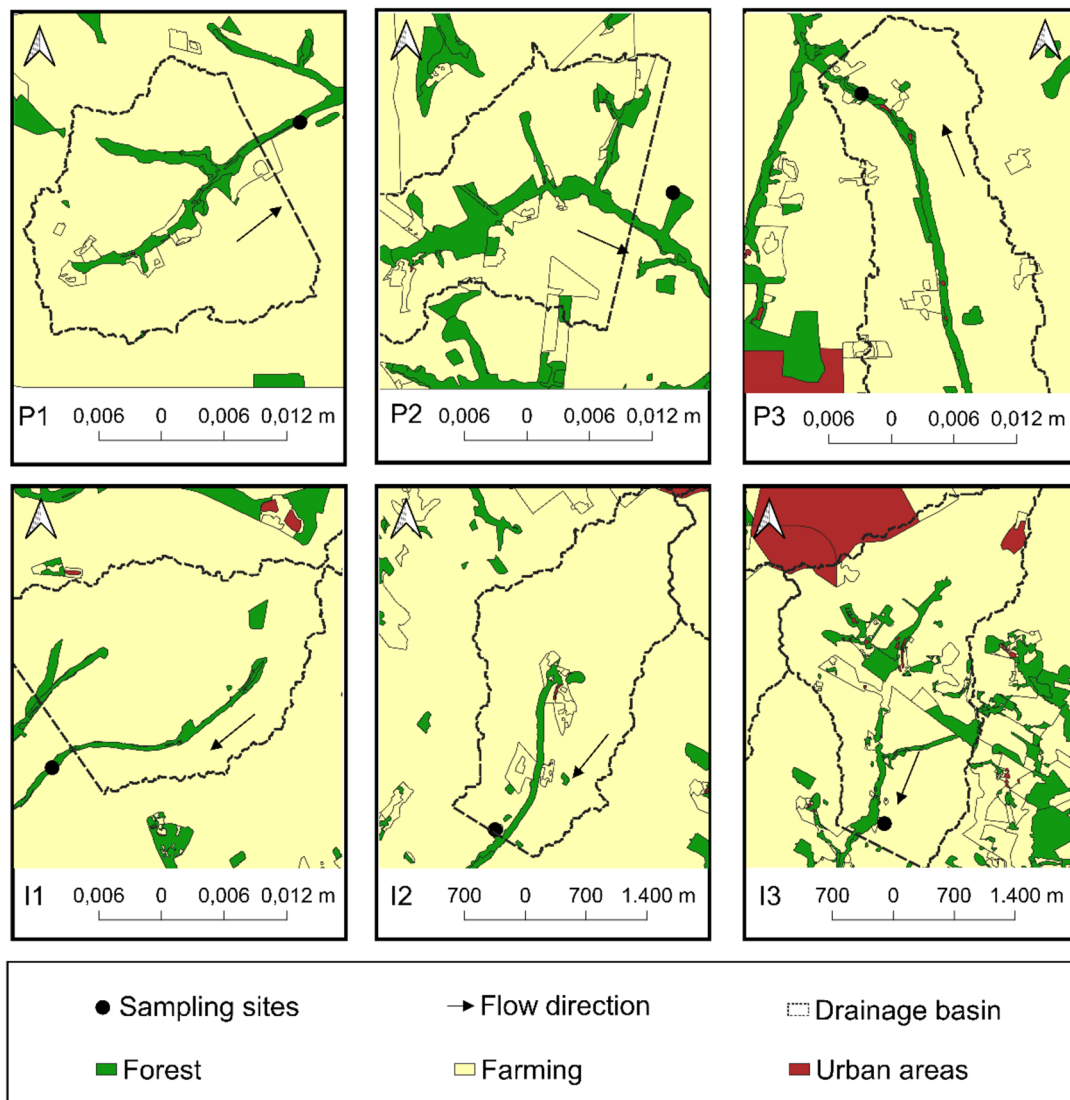


Fig. 4 Land use in the watershed of six peri-urban neotropical streams

On the stream network scale, wider buffers (500 and 1000 m) were mostly occupied by farming, reaching up to 47.95% (Graph 3; Figs. 4 and 5; Appendix 2). In all buffers, there was a positive relationship between PC1 and the percentage of agricultural land use, indicating that streams with higher agricultural use had higher values of DQO, DBO, and Zn. The effect of agricultural land use on PC1 was considered significant in the 500-m buffer ($p = 0.01$). However, all buffers showed a negative relationship between PC2 and the percentage of agricultural land use, indicating that streams with higher agricultural use had higher values of OD, ORP, and pH. The effect of agricultural land use on PC2 was considered significant in the 30-m buffer ($p = 0.0003$) (Appendices 3 and 4).

The wider buffers (500 and 1000 m) were also mainly occupied by urban areas, reaching 42.87%. In all buffers, there was a negative relationship between PC1 and the percentage of urban areas, indicating that streams with higher urban usage had higher values of ORP, TDS, and conductivity. The effect of urban areas on PC1 was significant in the 1000-m buffer ($p = 0.01$). However, all buffers showed a positive relationship between PC2 and the percentage of urban areas, indicating that streams with higher urban usage had higher values of NT, EC, and temperature. The effect of urban areas on PC2 was significant in the 500-m buffer ($p = 0.04$). The average percentage of slope ranged from 3.98 to 7.81% for the water network scale (Appendix 2).

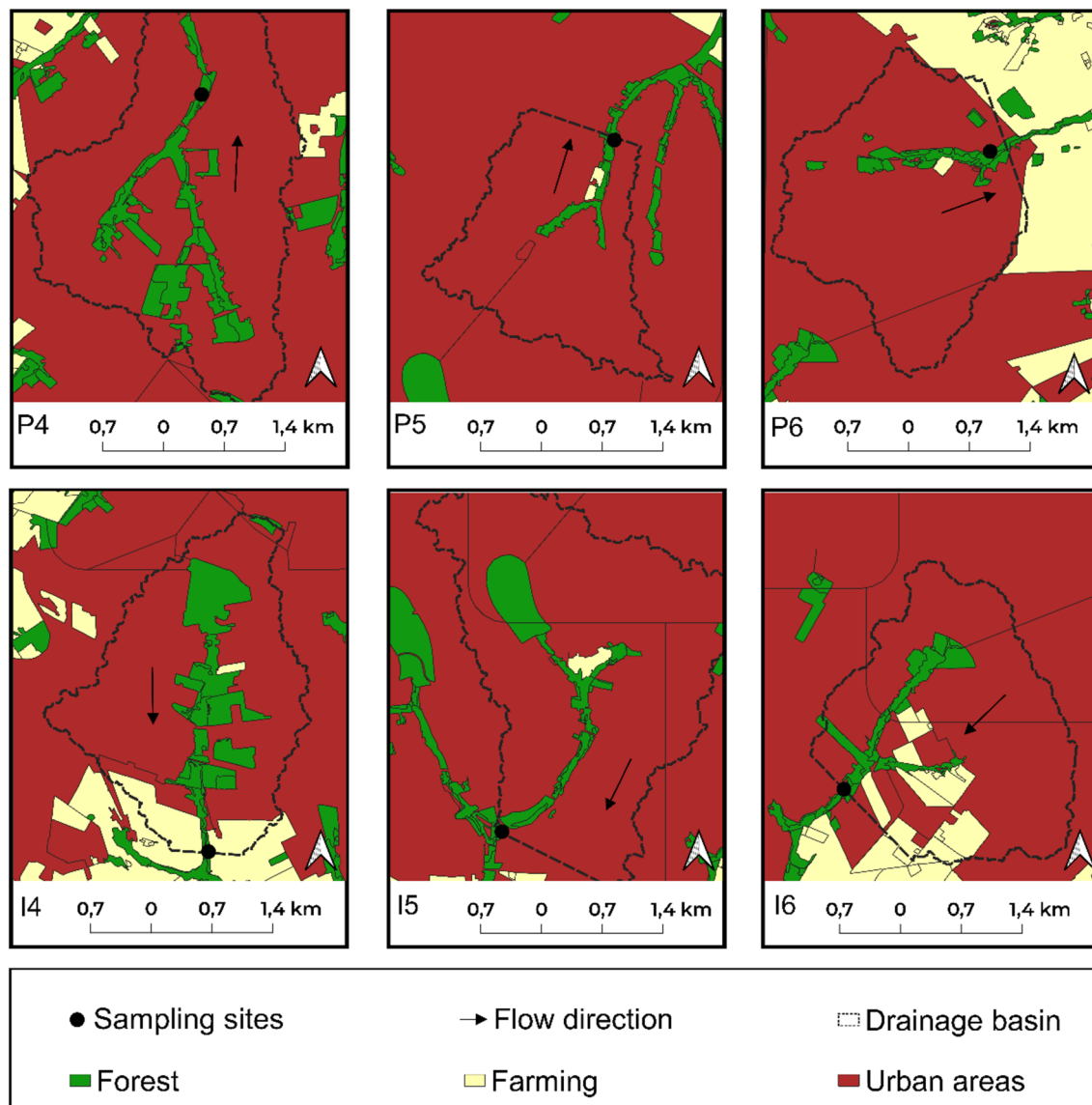


Fig. 5 Land use in the watershed of six urban neotropical streams

Discussion

The alteration of land use has exerted a substantial impact on the water quality of streams across several levels of analysis (Singh et al. 2017b). The inclusion of forested areas in small-scale buffer zones plays a significant role in preserving and enhancing water quality; whereas, agricultural activities have adverse effects on both local-scale environments and the overall water network. In contrast, it was shown that metropolitan areas had a negative influence solely on the watershed area. The water quality within a given region exhibits variations based on factors such as geographical location, temporal dynamics, weather patterns, degree of urban development, agricultural practices, types

of pollution sources, and the implementation of treatment protocols (Camara et al. 2019). Numerous studies have demonstrated that alterations at the catchment scale exert a more substantial influence on water quality (Mello et al. 2018; Puczko and Jekatierynczuk-Rudczyk 2020). The impact of land use and land cover (LULC) on water quality exhibits a scale-dependent nature, as observed in previous studies conducted by Pratt and Chang (2012) and Wilson (2015). According to Ngoye and Machiwa (2004), the utilization of agricultural land has been found to have detrimental impacts on the quality of river water due to the extensive application of fertilizers and irrigation practices. In their study, Bhattarai and Parajuli (2023) conducted an assessment to evaluate the efficacy of ponds, wetlands, riparian buffers, and their combined implementation as best management

practices (BMPs). The researchers determined that ponds and wetlands are effective in reducing total suspended solids, while riparian buffers are effective in reducing mineral phosphorus levels. In their study, Wang et al. (2023) examined the influence of seasonal variations on water quality. Their findings indicated that forested and grassland areas exhibited a positive effect on water quality; whereas, farmland and developed land had a negative impact on water quality in different seasons within the Huai River region of China. According to Wang et al. (2023), the primary sources of anthropogenic pollution are agricultural inputs and point sources. The activities occurring in upland areas play a significant role in influencing the water quality within their corresponding catchment areas and downstream streams (Li et al. 2023). According to Zou et al. (2023), the effects of severe events, such as floods and droughts, on river water quality exhibit variations. Specifically, floods are found to have a more pronounced adverse influence on water quality compared to droughts. In urbanized river basins, the pollutant load during the dry season is elevated as a result of the diminished dilution effect, as noted by Liu et al. (2017).

As an overall pattern for these urban and periurban streams, the water quality of the streams did not comply with the parameters established by the Canadian standards, indicating interference in the quality and considerable damage to aquatic ecosystems, fauna, and human populations. The initial hypothesis was confirmed with the finding that the CCME 2007 methodology is more rigorous and sensitive, leading to a classification of “poor and marginal.” The CONAMA Resolution 357/2005 proved to be more flexible and tolerant, resulting mainly in classifications of “good” and “fair,” indicating that the water is still considered suitable for the conservation of the aquatic ecosystem.

Water quality standards

The water quality of the streams has been determined to be impaired based on the criteria outlined in the Canadian Council of Ministers of the Environment (CCME) standards of 2007. This compromised water quality poses a significant risk to the survival of aquatic biota. The presence of metals is a significant contributing element to the decline in water quality index, as it signifies the bioavailability of these compounds (Souza et al. 2013). Multiple studies have demonstrated that the existence of metallic elements has played a role in the decline of the water quality index (WQI-CCME), particularly in relation to the preservation of aquatic organisms (Lumb et al. 2006; Al-Janabi et al. 2015). This issue has been notably apparent in watersheds that are subject to significant human-induced disturbances. Furthermore, it was noted that the amounts of lead and chromium were beyond the permissible thresholds as outlined in the Canadian standard. The presence of these metals has been found to have adverse impacts on the regulatory and

metabolic processes in fish, with a particular emphasis on their effects on embryonic development and reproductive capabilities (Jeziarska et al. 2009).

The findings of Pekey et al. (2004) suggest that the elevated amounts of metals in the streams are indicative of a direct influx of pollutants from several sources. Streams play a significant role in the formation of watersheds and subsequent surface runoff, hence impacting the water quality of bigger rivers. The potential for bioaccumulation and biomagnification in aquatic ecosystems allows these elements to have harmful effects across long distances from their emission source (Weber et al. 2013; Lee et al. 2019). Hence, the presence of elevated levels of metals in aquatic environments has emerged as a significant concern for both aquatic organisms and human communities in contemporary times (Souza-Araujo et al. 2016; Collin et al. 2022; Noor et al. 2023).

The study conducted by Vieira et al. (2022) examined water quality in an agricultural watershed in Paraná and reported an elevation in the amounts of nitrogen and *E. coli*. These findings were shown to be significant factors contributing to the increase in the Water Quality Index-Canadian Council of Ministers of the Environment (WQI-CCME). The aforementioned studies conducted by Rovani et al. (2019) and Martíni et al. (2021) provide further evidence to support the notion that intensive farming practices are having a detrimental impact on the water quality within watersheds located in the southern region of Brazil. Research undertaken in regions characterized by elevated nitrogen concentrations and a significant presence of *E. coli* bacteria has yielded empirical evidence indicating detrimental impacts on the overall well-being of aquatic ecosystems. The observed consequences encompass a reduction in the variety of macroinvertebrates and aquatic microbial communities, alongside the discovery of indicator species that exhibit tolerance towards these unfavorable circumstances (Paruch et al. 2019; Rico-Sánchez et al. 2022).

The findings of the quantitative and qualitative assessments indicate that the water quality in the majority of the streams adheres to the requirements outlined in CONAMA Resolution 357/2005. This suggests a minimal level of risk and the capacity to safeguard aquatic organisms. The observation indicated that the sole presence of copper was the causal factor for the reduction in the index (WQI-CCME), a finding that was further validated in the study conducted in accordance with the Canadian requirements. The results presented in this study align with the research undertaken by Alves et al. (2013), wherein an evaluation of the water quality of Ribeirão Preto stream revealed that the levels of metals detected were in accordance with the guidelines set forth by CONAMA Resolution 357/2005, despite the notable impact of human activities. In a study conducted by Godoy et al. (2021), it was found that the Piquiri River basin maintained satisfactory water quality levels, as defined by CONAMA Resolution 357/2005, despite the

presence of extensive agricultural practices across a monitoring period spanning two decades.

The adoption of a human-centered perspective as an environmental indicator in CONAMA Resolution 357/2005 imposes limitations on the evaluation of aquatic ecotoxicology, as discussed by Odum (1988) and Bertolotti (2012). The aforementioned restrictions are readily apparent in the diverse applications of classes 2 through 4. Moreover, it is imperative to acknowledge that a restricted range of chemical, physical, and biological indicators fails to provide a thorough evaluation of water quality (Silva et al. 2018; Padovesi-Fonseca 2022). From this perspective, adopting a solely anthropocentric strategy may lead to ecological imbalance and inadequate monitoring of aquatic ecosystems.

While it is feasible to utilize the CCME 2007 guidelines in Brazil for the purpose of achieving a more effective analysis and comprehensive understanding of aquatic ecosystems, it is crucial to bear in mind that the nation possesses the largest reserve of freshwater globally. Moreover, Brazil encompasses diverse biomes, each characterized by distinct features pertaining to biodiversity, soil composition, vegetation, climate patterns, and water resources (Passos et al. 2018). The comprehensive evaluation of water quality should not exclusively consider human needs, but should also incorporate the distinct characteristics of individual aquatic ecosystems and their interactions with the surrounding environment to facilitate the implementation of efficient conservation strategies. Hence, it is imperative to align the quality objectives by including pertinent local studies to effectively preserve this natural asset and ensure its sustainable utilization (CCME 2007).

Landscape dynamics in water quality: a multiscale analysis

As the scale expanded, there was an observed decline in forest cover over time. The observed phenomenon was concomitant with a rise in anthropogenic activity, such as agriculture and urbanization, suggesting a tendency towards landscape homogenization (Ribeiro et al. 2021). Deforestation of the Atlantic Forest in the state of Paraná has significantly escalated in recent decades, leading to the substantial depletion of forested regions. This phenomenon may be primarily attributed to the increase of agricultural activities and animal husbandry (Mohebalian et al. 2022). The potential consequences of environmental degradation outlined in this forecast pose a significant threat to water quality, aquatic ecosystems, and the availability of water resources for human populations (Mello et al. 2020).

Multiple studies have provided evidence regarding the significance of forests in relation to water quality in streams, with a specific focus on their role in sustaining appropriate amounts of dissolved oxygen (Wang et al. 1997; Fernandes et al. 2014; Shen et al. 2015; Mello et al. 2018). The findings of this study

provide further support for the notion that there exists a clear correlation between an expansion in forested areas and elevated concentrations of dissolved oxygen in aquatic environments. This link is of utmost importance as dissolved oxygen plays a critical role in facilitating the metabolic and respiratory processes of aquatic organisms, as highlighted in the works of Oliveira et al. (2019) and Piffer et al. (2021). Nonetheless, the observed elevation in many parameters including oxidation-reduction potential (ORP), pH, conductivity, and total dissolved solids (TDS) signifies the existence of ions within the water, originating from the leaching of nutrients from vegetation (Arcos et al. 2022). Forest harvesting, in conjunction with drainage, has been found to have significant effects on various scientific disciplines such as chemistry, physics, and ecology. These activities contribute to an increase in the accumulation of organic carbon, leading to the phenomenon known as browning, and, subsequently, causing detrimental changes in aquatic ecosystems (Härkönen et al. 2023). According to Härkönen et al. (2023), the process of browning in water bodies has detrimental effects on their recreational value and increases the costs associated with treating drinking water.

The principal driver of environmental change has been attributed to human activities (Ding et al. 2016; Mello et al. 2018, 2020; Shi et al. 2022). The results obtained in our study provide support for the notion that water quality degradation is mostly attributed to the presence of agriculture and urban areas. Nevertheless, the issue can be further worsened by agricultural practices, as the findings suggest a correlation between these activities and a decline in oxygen levels and an increase in organic matter in the water. This is supported by the observed rise in biochemical oxygen demand (BOD) and chemical oxygen demand (COD). The exacerbation of this situation is attributed to the escalation of pollutants, namely nitrogen and zinc, which originate from the excessive application of fertilizers and pesticides, as well as the outflow of sewage (McGrane 2016; Xiaojing et al. 2021). Moreover, the escalation in *Escherichia coli* can be attributed to regions without adequate sanitary infrastructure, as well as the presence of livestock-derived manure or its application in agricultural settings (Hubbard et al. 2004; Lim et al. 2022). A positive correlation was identified between the decrease in native vegetation cover and the increase in water temperature. The aforementioned discovery aligns with the outcomes reported by Santos et al. (2017). The investigation reveals a significant correlation between the various interactions and the inclusion of local terrain slope as a covariate. Slopes with a higher gradient possess a heightened capacity to detrimentally affect water quality as a result of amplified water flow velocity. In the given circumstances, water possesses the capability to convey greater quantities of sediment, nutrients, and contaminants (Schmidt et al. 2019; Liu et al. 2021; Lei et al. 2021).

The study revealed a positive correlation between the existence of forests within a local area spanning 50–100 m and a

notable enhancement in water quality. Conversely, agricultural activities at the same scale were found to have a detrimental effect on water quality. The findings of Turunen et al. (2021) and Shah et al. (2022) underscore the susceptibility of the aquatic ecosystem to land use disputes. Another significant element that contributes to the deterioration of water quality is the growth of urban and agricultural regions within the watershed. The observable impacts of this expansion can be detected within a radius of 500–1000 m in the river network scale. The aforementioned results are consistent with the findings of a recent investigation conducted by Shi et al. (2022). This study revealed that extensive land use practices, such as deforestation and urbanization, have a detrimental impact on water quality within a specific watershed in China. Conversely, the presence of vegetation cover at smaller scales, particularly along individual streams, was found to facilitate improvements in water quality. In their study, Cheng et al. (2023) found that landscape metrics associated with urban areas were the primary factor contributing to the deterioration of water quality in both dry and wet seasons. This degradation was mostly due to the expansion of urban land. Additionally, metrics connected to farmland were identified as a secondary contributor to water quality degradation in the agricultural basin of the Longxi River basin in western China. Irrespective of their spatial and temporal heterogeneity, the landscape metrics, namely patch density, landscape form index, and splitting index, exerted a notable influence on the quality of the river water, as demonstrated by Li et al. (2023). The significance of landscape patterns on water quality is essential, both in terms of the riparian zone and the sub-basin, as highlighted by Xu et al. (2023). According to Xu et al. (2023), the landscape metrics provide a significant ability to forecast the variability in total nitrogen in relation to total phosphorus.

The incorporation of a multiscale approach in watershed management and public policy is of significant importance, as it encompasses not only the conservation of local forests but also the comprehensive consideration of human activities across the entire watershed. This perspective is substantiated by a range of investigations, including those undertaken by Buck et al. (2004), Pratt and Chang (2012), Ding et al. (2016), and Shi et al. (2022).

Conclusions

The present study investigated the primary impacts of land use on the quality of water in neotropical streams. The streams exhibit significant contamination from organic nutrients and fecal coliforms, specifically nitrogen and *Escherichia coli*. Furthermore, the presence of heavy metals such as lead, chromium, and copper poses a considerable threat to both water quality and the long-term viability of the ecosystem (Yuan et al. 2021). The streams are classified as having “poor” and “marginal”

quality according to the CCME index and 2007 standards. The aforementioned methods showed efficacy in the realm of stream monitoring, hence underscoring the imperative for a more comprehensive evaluation of water resources. According to Silva et al. (2018), the results indicate the need for a revision of Resolution CONAMA No. 357/2005 in order to adopt a more conservation-oriented and comprehensive approach.

In order to protect the integrity of the local water quality, the study proposes the implementation of a conservation strategy targeting riparian woodlands with a minimum width of 50 m. Furthermore, the study identified agriculture and urban areas as the primary factors contributing to the decline of water quality within a radius of 1000 m from the watershed. The aforementioned research findings underscore the necessity of implementing public policies and making amendments to the Brazilian Forest Code with regards to permanent preservation zones. These measures are crucial for safeguarding aquatic ecosystems through the maintenance of vegetation cover and taking into account the impacts of human activities (Chaves et al. 2023).

Notwithstanding the limited number of streams and the assessment of only one seasonal period, the outcomes of this study are in alignment with previous studies. Therefore, we propose conducting multiscale investigations in order to enhance comprehension of water quality and enhance the efficacy of management strategies. This research has the potential to enhance water resource management in anthropogenically impacted environments, hence enhancing the field of freshwater ecosystem studies.

The degradation of stream water quality poses a significant threat to both natural and human populations who are reliant on river ecosystems. The performance of the CCME index and the CCME, 2007 recommendation in this analysis was satisfactory, emphasizing the necessity for a more rigorous approach to evaluating water quality. The findings additionally propose a revision of CONAMA Resolution No. 357/2005, with an emphasis on conservation and a comprehensive strategy. According to the research findings, the preservation of forests plays a crucial role in maintaining the quality of local water resources, while the presence of agricultural and urban areas might have detrimental effects on the overall water quality within a given basin. The aforementioned findings underscore the imperative of implementing public policies and making adjustments to the Brazilian Forest Code in order to preserve aquatic ecosystems. This can be achieved through the protection of plant cover and the inclusion of human influences in decision-making processes. The implementation of these measures will enhance water quality and promote environmental conservation, ensuring the sustainability of future generations. In order to enhance comprehension of water quality and effectively manage and preserve aquatic ecosystems, it is imperative to complement the findings with multiscale investigations.

Appendix 1: Limnological analysis of physicochemical and biological parameters evaluated in neotropical streams. Pn = streams from the Pirapó basin; In = streams from the Ivaí basin.

Parameters	P1	P2	P3	P4	P5	P6	I1	I2	I3	I4	I5	I6
Temperature	22.92	23.89	22.32	25.13	24.85	26.31	23	23.66	23.48	22.19	25.69	24.75
Turbidity	6.567	6.100	26.900	1.800	35.067	5.133	51.667	10.067	10.167	6.633	2.900	14.900
Conductivity	0.033	0.086	0.052	0.247	0.204	0.201	0.058	0.144	0.076	0.253	0.198	0.188
TDS	0.022	0.056	0.034	0.161	0.132	0.130	0.038	0.094	0.049	0.165	0.129	0.122
ORP	206.667	244.000	312.667	287.000	259.000	255.000	152.667	270.000	263.333	249.667	281.000	285.333
pH	6.863	7.473	7.553	7.417	7.297	7.230	7.310	7.590	7.543	7.823	7.297	7.350
Total Nitrogen	1.380	2.350	0.920	4.800	7.460	10.300	1.500	2.570	3.000	4.740	6.130	2.100
Total Phosphorus	0.190	0.060	0.080	0.070	0.110	0.070	0.000	0.090	0.560	0.070	0.140	0.070
Dissolved Oxygen	10.070	10.193	10.960	10.013	8.687	8.233	9.997	9.140	8.900	9.950	8.153	8.660
BOD	1	7.02	1.3	2.95	0	0	31	2.27	4.56	0	2.39	0
COD	3.2	3.8	3.1	10.4	5	5.7	107	7.5	17.7	4.4	11.5	4.3
Cadmium	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Lead	0.000	0.000	0.000	0.000	0.000	0.000	0.032	0.000	0.000	0.000	0.000	0.000
Chromium	0.000	0.000	0.000	0.000	0.000	0.000	0.087	0.000	0.021	0.000	0.000	0.000
Zinc	0.038	0.000	0.020	0.028	0.000	0.016	0.156	0.012	0.024	0.000	0.017	0.010
Nickel	0.000	0.000	0.000	0.000	0.000	0.000	0.033	0.000	0.000	0.000	0.000	0.000
Copper	0.006	0.000	0.012	0.000	0.006	0.000	0.322	0.006	0.054	0.000	0.005	0.000
Total Coliforms	> 2419.6	> 2419.6	> 2419.6	> 2419.6	> 2419.6	> 2419.6	> 2419.6	> 2419.6	> 2419.6	> 2419.6	> 2419.6	> 2419.6
<i>Escherichia coli</i>	112.4	191.8	167.4	66.3	2419.6	1986.3	149.1	193.5	1413.6	142.5	1986.3	980.4

Appendix 2: Percentage (%) of land use measured in the scale groups: Local Scale (L), River Network Scale (R), and average percentage of terrain slope (Dec.) at Local Scale (L—500 m buffer) and Network Scale (R—drainage basin) included in forest use. Pn = streams in the Pirapó basin; In = streams in the Ivaí basin

Points	Forest (%)												
	30		50		100		200		500		1000	Dec	
	L	R	L	R	L	R	L	R	L	R	R	L	R
P1	100.00	97.02	87.98	89.19	51.81	59.26	27.15	31.48	13.68	12.62	7.50	7.81	8.39
P2	100.00	91.31	100.00	83.27	66.47	64.15	32.67	40.27	22.38	20.83	15.15	9.67	9.5
P3	90.06	89.77	61.07	71.17	44.23	44.19	29.75	24.61	20.55	10.23	7.12	8.39	8.09
P4	99.98	96.73	93.16	91.31	70.49	67.76	35.71	43.16	13.81	22.37	15.97	8.76	9.84
P5	100.00	80.91	100.00	71.98	83.34	44.49	45.21	22.15	16.20	8.43	4.02	8.08	7.89
P6	100.00	96.50	100.00	91.20	85.60	78.45	52.35	48.39	26.88	18.59	5.72	9.17	11.32
I1	94.86	97.60	76.16	76.79	38.39	43.49	20.05	22.82	9.73	8.53	5.70	7.08	8.01
I2	84.60	80.62	73.56	69.16	59.21	46.84	36.38	28.28	19.48	15.12	9.19	8.36	10.67
I3	100.00	94.90	85.50	70.26	51.45	37.41	28.47	19.68	11.23	8.31	3.48	7.21	7.03
I4	99.66	97.45	84.41	93.01	47.28	74.71	24.48	52.68	14.04	32.14	16.03	8.92	9.01
I5	100.00	84.45	100.00	76.82	77.78	61.26	53.94	46.01	31.89	20.92	9.04	9.48	12.89
I6	100.00	93.24	94.11	88.31	76.80	81.90	49.37	40.40	23.77	20.75	8.38	7.13	9.92

Farming (%)

Points	30		50		100		200		500		1000
	L	R	L	R	L	R	L	R	L	R	R
P1	0.00	2.75	12.02	10.67	48.19	40.67	72.85	68.49	86.32	87.37	92.49
P2	0.00	8.22	0.00	16.45	33.53	35.71	67.33	59.64	77.62	79.13	84.82
P3	9.94	7.72	38.93	27.14	55.77	54.87	70.25	74.93	79.25	89.59	91.63
P4	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15
P5	0.00	0.37	0.00	1.69	0.00	4.32	0.00	4.57	1.80	1.74	0.83
P6	0.00	0.68	0.00	1.77	0.00	3.14	0.00	3.79	18.76	1.79	1.58
I1	5.14	2.40	23.84	23.21	61.61	56.51	79.95	77.18	90.27	91.47	94.30
I2	15.40	16.31	26.44	28.24	40.79	51.21	63.62	70.74	80.52	83.96	87.14
I3	0.00	4.92	14.50	29.63	48.55	62.31	71.53	79.86	88.77	91.53	96.34
I4	0.34	0.22	15.21	3.01	45.68	11.95	57.21	16.95	69.75	19.33	12.36
I5	0.00	0.60	0.00	2.16	0.00	4.60	0.00	5.17	0.35	2.18	0.90
I6	0.00	3.24	0.00	5.43	0.00	14.66	1.21	17.79	10.41	17.94	12.87

Urban areas (%)

Points	30		50		100		200		500		1000
	L	R	L	R	L	R	L	R	L	R	R
P1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
P2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
P3	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.12
P4	0.02	3.27	6.84	8.69	29.51	32.24	64.29	56.84	86.19	77.63	83.87
P5	0.00	8.25	0.00	16.16	16.66	45.03	54.79	70.27	82.00	88.68	94.61
P6	0.00	2.32	0.00	5.52	14.40	16.79	47.65	47.01	53.95	79.32	92.62
I1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
I2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.49	2.24
I3	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11
I4	0.00	2.33	0.39	3.98	7.04	13.34	18.31	30.37	16.21	48.53	71.60
I5	0.00	5.05	0.00	11.54	20.12	28.58	38.08	45.93	65.51	75.64	89.62
I6	0.00	1.93	5.89	5.30	23.20	2.84	49.42	41.58	65.83	61.23	78.71

See Figs. 4 and 5.

Appendix 3: Estimates of the slopes of the relationship between the percentage of land use and the axes (PC1 and PC2) for each group of scales: Local (local scale) and Network (Water network scale), as well as the overall statistics of the models (AIC, R2, and P)

Forest						
Axis	Buffer	Group	Estimate	AIC	R ²	P
PC1	30	Local	-8.58	46.8	0.78	0.002
		Rede	0.97	59.68	0.37	0.09
	50	Local	-2.00	45.2	0.81	0.002
		Rede	-0.46	63.48	0.14	0.42
	100	Local	-0.71	47.2	0.77	0.02
		Rede	-0.36	59.80	0.37	0.18

Forest						
Axis	Buffer	Group	Estimate	AIC	R ²	P
	200	Local	-0.82	48.8	0.74	0.04
		Rede	-0.60	58.04	0.45	0.09
	500	Local	-1.62	55.8	0.54	0.09
		Rede	-1.08	58.16	0.45	0.07
	1000	Rede	-1.59	63.54	0.14	0.37

Forest						
Axis	Buffer	Group	Estimate	AIC	R ²	P
PC2	30	Local	-4.39	50.3	0.38	0.09
		Rede	0.03	49.06	0.45	0.91
	50	Local	-0.74	48.3	0.48	0.19
		Rede	0.11	48.23	0.49	0.71
	100	Local	-0.63	40.8	0.72	0.01
		Rede	-0.07	47.69	0.51	0.61
	200	Local	-0.65	45.5	0.59	0.06
		Rede	-0.005	47.29	0.52	0.98
	500	Local	-1.03	52.2	0.28	0.19
		Rede	-0.07	48.29	0.48	0.83
	1000	Rede	-0.68	47.97	0.50	0.46
	Farming					
Axis	Buffer	Group	Estimate	AIC	R ²	P
PC1	30	Local	8.58	46.8	0.78	0.002
		Rede	1.26	62.9	0.18	0.41
	50	Local	1.70	46.1	0.79	0.004
		Rede	0.69	52.63	0.65	0.02
	100	Local	0.34	50.3	0.71	0.10
		Rede	0.31	48.4	0.75	0.01
	200	Local	0.19	52.1	0.66	0.19
		Rede	0.21	46.9	0.78	0.01
	500	Local	0.17	54.0	0.60	0.25
		Rede	0.16	46.1	0.79	0.01
	1000	Rede	0.15	46.4	0.79	0.01
	PC2	30	Local	4.39	50.39	0.38
Rede			-2.08	28.09	0.90	0.0003
50		Local	0.74	48.63	0.47	0.15
		Rede	-0.13	45.67	0.58	0.46
100		Local	0.37	42.34	0.68	0.02
		Rede	-0.01	45.43	0.59	0.84
200		Local	0.20	45.26	0.60	0.08
		Rede	-0.01	45.60	0.59	0.80
500		Local	0.16	47.19	0.53	0.15
		Rede	-0.01	45.67	0.58	0.77
1000		Rede	-0.01	46.08	0.57	0.76

Urban areas						
Axis	Buffer	Group	Estimate	AIC	R ²	P
PC1	30	Local	-41.54	59.54	0.27	0.75
		Rede	-0.09	59.44	0.38	0.95
	50	Local	-3.02	58.37	0.44	0.30
		Rede	-0.06	57.34	0.48	0.92
	100	Local	-0.59	55.33	0.56	0.29
		Rede	-0.16	58.66	0.42	0.47
	200	Local	-0.24	54.51	0.59	0.31
		Rede	-0.20	50.62	0.70	0.05
	500	Local	-0.18	54.85	0.58	0.31
		Rede	-0.16	47.31	0.77	0.02
	1000	Rede	-0.14	46.05	0.79	0.01
	PC2	30	Local	-1.05	54.25	0.004
Rede			0.13	49.5	0.20	0.13
50		Local	0.02	56.25	0.005	0.99
		Rede	0.10	48.2	0.28	0.07
100		Local	-0.47	51.14	0.35	0.31
		Rede	0.05	48.47	0.27	0.08
200		Local	-0.23	48.29	0.48	0.21
		Rede	0.03	47.76	0.31	0.05
500		Local	-0.16	48.68	0.47	0.24
		Rede	0.02	47.38	0.33	0.04
1000		Rede	0.02	48.08	0.29	0.06

Bold highlights emphasize models that show a significant impact at the chosen scale

Appendix 4: Estimates of parameters for models selected using backward and forward methods, based on the lowest AIC value, for each group of scales: Local Scale (L), Network Scale (R)

Forest						
	Coefficients	Estimate	Pr(> t)	F-statistic	p-value	Adjusted R-squared
PC1 (L)	(Intercept)	190.53	0.001	11.65 (3.8)	0.002	0.74
	floresta_50	-2.00	0.002			
	paisagem_local\$slope_500	-21.91	0.002			
	floresta_50:paisagem_local\$slope_500	0.22	0.003			
PC1 (R)	(Intercept)	21.63	0.09	2.24 (3.8)	0.16	0.25
	floresta_200	-0.60	0.09			
	paisagem_montante\$slope_bacia	-3.11	0.15			
	floresta_200:paisagem_montante\$slope_bacia	0.08	0.15			
PC2 (L)	(Intercept)	45.5	0.009	7.0 (3.8)	0.01	0.62
	floresta_100	-0.63	0.01			
	paisagem_local\$slope_500	-6.11	0.006			
	floresta_100:paisagem_local\$slope_500	0.08	0.008			
PC2 (R)	(Intercept)	5.36	0.01	8.16(1.10)	0.01	0.39
	paisagem_montante\$slope_bacia	-0.90	0.01			
Farming						
	Coefficients	Estimate	Pr(> t)	F-statistic	p-value	Adjusted R-squared
PC1 (L)	(Intercept)	-4.21	0.43	10.6 (3.8)	0.003	0.72
	agropecuaria_50	1.70	0.004			
	paisagem_local\$slope_500	0.32	0.6			
	agropecuaria_50:paisagem_local\$slope_500	-0.19	0.005			
PC1 (R)	(Intercept)	-4.51	0.22	10.52 (3.8)	0.003	0.72
	agropecuaria_500	0.16	0.01			
	paisagem_montante\$slope_bacia	0.45	0.47			
	agropecuaria_500:paisagem_montante\$slope_bacia	-0.02	0.04			
PC2 (L)	(Intercept)	-5.80	0.28	5.874 (3.8)	0.02	0.57
	agropecuaria_100	0.37	0.02			
	paisagem_local\$slope_500	0.81	0.20			
	agropecuaria_100:paisagem_local\$slope_500	-0.05	0.01			
PC2 (R)	(Intercept)	10.68	5.45	25.35 (3.8)	0.0001	0.86
	agropecuaria_30	-2.08	0.0003			
	paisagem_montante\$slope_bacia	-1.63	8.80			
	agropecuaria_30:paisagem_montante\$slope_bacia	0.27	0.0003			
Urban areas						
	Coefficients	Estimate	Pr(> t)	F-statistic	p-value	Adjusted R-squared
PC1 (L)	(Intercept)	10.52	0.07	5.713 (2.9)	0.025	0.46
	urbano_200	-0.05	0.03			
	paisagem_local\$slope_500	-1.11	0.11			

Urban areas

	Coefficients	Estimate	Pr(> t)	F-statistic	p-value	Adjusted R-squared
PC1 (R)	(Intercept)	10.79	0.003	10.66 (3.8)	0.003	0.72
	urbano_bacia	-0.14	0.01			
	paisagem_montante\$slope_bacia	-1.45	0.007			
	urbano_bacia:paisagem_montante\$slope_bacia	0.01	0.06			
PC2 (L)	(Intercept)	7.82	0.18	2.539 (3.8)	0.12	0.29
	urbano_200	-0.23	0.21			
	paisagem_local\$slope_500	-1.06	0.14			
PC2 (R)	(Intercept)	-0.94	0.14	5.086 (1.10)	0.04	0.27
	urbano_200:paisagem_local\$slope_500	0.03	0.15			
	urbano_500	0.02	0.04			

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Data availability The datasets generated during and/or analyzed during the current study are available from the corresponding author on reasonable request.

Declarations

Conflict of interest All the authors declare that they have no conflict of interest.

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