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Effects of Land Use on the Community Structure of Aquatic Invertebrate in Subtropical Streams

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Abstract: Streams constitute the water supply of the watersheds and provide the transfer of energy along the course of water, ensuring water biodiversity. Due to the different uses of the earth, the benthic community is being suppressed. The objective of this study is to investigate the effects of changes in land use on the abundance, richness, and diversity of benthic macroinvertebrates in subtropical streams, considering the physical and chemical variables of streams seasonally. In total, 144 samples were collected between February 2022 and November 2023 at 12 sites distributed in a watershed in southern Brazil, four sites for each land use. Herein, 83.520 individuals were identified, comprising 67 taxa. There was a decrease in richness along the forested–rural–urban gradient and a greater abundance in the urban environment, without the significant influence of seasonality. We conclude that the diversity of benthic macroinvertebrate in the streams is influenced not only by environmental variables but also by the spatial distribution between streams. The proximity of the streams generates a similarity in dominance. This study contributed to understanding the relationships of the environment with the benthic community and considering the spatial distribution among the streams sampled in future investigations.

Keywords: aquatic ecosystem; benthic community; richness and diversity; land use changes

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1. Introduction

It is estimated that the global fluvial area comprises 485.000 to 662.000 km^2, with rivers and streams covering approximately between 0.3 and 0.56% of the land surface [1,2]. The overall estimate of the area of rivers and streams implies lotic processes, especially the low-order ones, because the streams are essential to the constitution of the river network in a watershed [3,4]. Due to its longitudinal connectivity, it provides downstream transfers of energy, water, sediments, nutrients, organic matter, and organisms that guarantee the maintenance of water biodiversity [5–7].

Changes in land use resulting from population growth are a threat to river ecosystems around the world [7,8]. The urbanization process needs the construction of houses, an increase in agricultural areas, and water demand to meet the basic needs of the population [4,9]. Areas of forest land use, which once protected the headwaters of rivers and streams, have become deforested, resulting in a decrease in water shading, associated with increased irradiance and water temperature, as well as a lower intake of allochthone organic matter [10,11]. Due to the increase in effluent dumping and the loading of pesticides into watercourses [12,13], these changes lead to a decrease in habitat quality and, consequently, affect the diversity and richness of plants and animals, compromising the functioning of aquatic ecosystems [14,15].

Despite the advances in research in recent years, there is still a lack of information on the habitat specificity of many aquatic insects in relation to the gradient of land use change, forested–rural–urban [3,16]. Benthic macroinvertebrate communities play a key role in nitrogen (N) and phosphorus (P) cycling in streams [17], avoiding the eutrophication process [18]. Also, the diagnosis of the presence or absence of these communities can act as an indicator of the quality of the environment since the heterogeneity of the substrates provides shelter and food availability [6,19,20], in the continuous stream [21]. However, this process can be interrupted by the decrease in the diversity and/or abundance of these assemblies [17,22], considering that the multiple uses of the land alter the trophic structure of these communities [23,24].

Mitigation measures are necessary to preserve the structure of the benthic macroinvertebrates community and ensure the maintenance of functioning in the trophic chain of these ecosystems [22,25]. Quantitative data on abundance and richness can help better detect relationships between communities and the environment [26]. Efficient analyses of the taxonomic composition of aquatic insects can provide punctual information for measures and strategies of management and deliberation by the public power, with a view to the restoration of rivers and streams [27,28] and monitoring on spatial and temporal scales, which can be a support tool [14].

Considering that the streams are hydrographic in 70% of the basins and are highly vulnerable to anthropogenic disturbances [2,7,29], it is crucial to understand how changes in land use influence the functioning of these ecosystems [23,30]. In this context, the investigation of the influence of different land uses on the benthic community in the streams of the Itajaí-mirim river basin (Brazil) allowed to evaluate two main areas: (i) the composition of benthic macroinvertebrate communities present in first- and second-order streams in different seasons and (ii) local environmental variables that influence the abundance, richness, and diversity of macroinvertebrates in streams. Global patterns indicate the general structure of benthic macroinvertebrate communities in aquatic environments, while local variables are responsible for shaping their specificities [31].

Given the importance of streams in maintaining watercourses and designing the research territory responsible for the capture, treatment, and public water supply of 63% of the population of this watershed [32], we predicted the following hypotheses: (i) benthic invertebrate communities have greater abundance, richness, and taxonomic diversity on the gradient of land use of forested–rural–urban, with seasonal influence; and (ii) a cascade effect with physical and chemical changes of water, with increased temperature and nutrient concentrations along the forested–rural–urban gradient.

Based on the above hypotheses, the objective of this study is to investigate the effects of changes in land use on abundance, richness, and diversity in the structure of benthic macroinvertebrate communities throughout the seasons in the Itajaí-mirim river basin (Brazil). This line of research will serve as a reference for studies of benthic communities in streams, considering that there is no reference in the territory in question, and it will provide scientific input on future monitoring to evaluate the stressful impacts of these communities on different land uses over time.

2. Materials and Methods

2.1. Study Area

The Itajaí-mirim river basin (26°53 17.1′26″—56°05.1′48″ S, 40°57.8′48″–44° 12.4′′′ W) is located in the Itajaí Valley region, Santa Catarina State—Brazil, and is part of one of the seven sub-basins that make up the Itajaí river basin, belonging to the Hydrographic Region of the South Atlantic strand. It has a drainage territory of 1677 km^2, the Itajaí-mirim river being its main course, with an approximate extension of 170 km and average flows to the mouth of 71,368 L/s, the second largest flow of the territorial area [33,34]. Its headwaters originate in the Serra dos Faxinais at 1.000 m altitude, and the region is in the domains of the Atlantic Forest [35], with a temperate climate humid with a hot summer and no dry season (Cfa) in the Köppen-Geiger classification. Annual rainfall ranges from 1.600 mm to 1.800 mm, with more intense rainfall in the summer [29]. Currently, 636.525 thousand people live in this watershed, allocated to nine municipalities, which corresponds to 8.36% of the population of the State of Santa Catarina [36].

The streams that make up the Itajaí-mirim river basin drain through regions and sub-basins with different land uses [4]. In the lower and closest regions to the river, which are occupied by housing and industry, rural areas are occupied by rice cultivation and pastures. However, in steeper areas, where there is a predominance of hills, the preserved forest prevails [37]. In order to integrate the different land uses and expand the possibility of spatial distribution of benthic macroinvertebrates, 12 sub-basins were selected for this study, four streams—first and second order [38]—for each land use as follows: forested (F1–F4), rural (R1–R4), and urban (U1–U4), distributed in four municipalities in the Itajaí-mirim river basin, as described in Figure 1 and Table 1. The study criteria of the sample plan can be consulted in full in Bohn Vieira et al. [29].

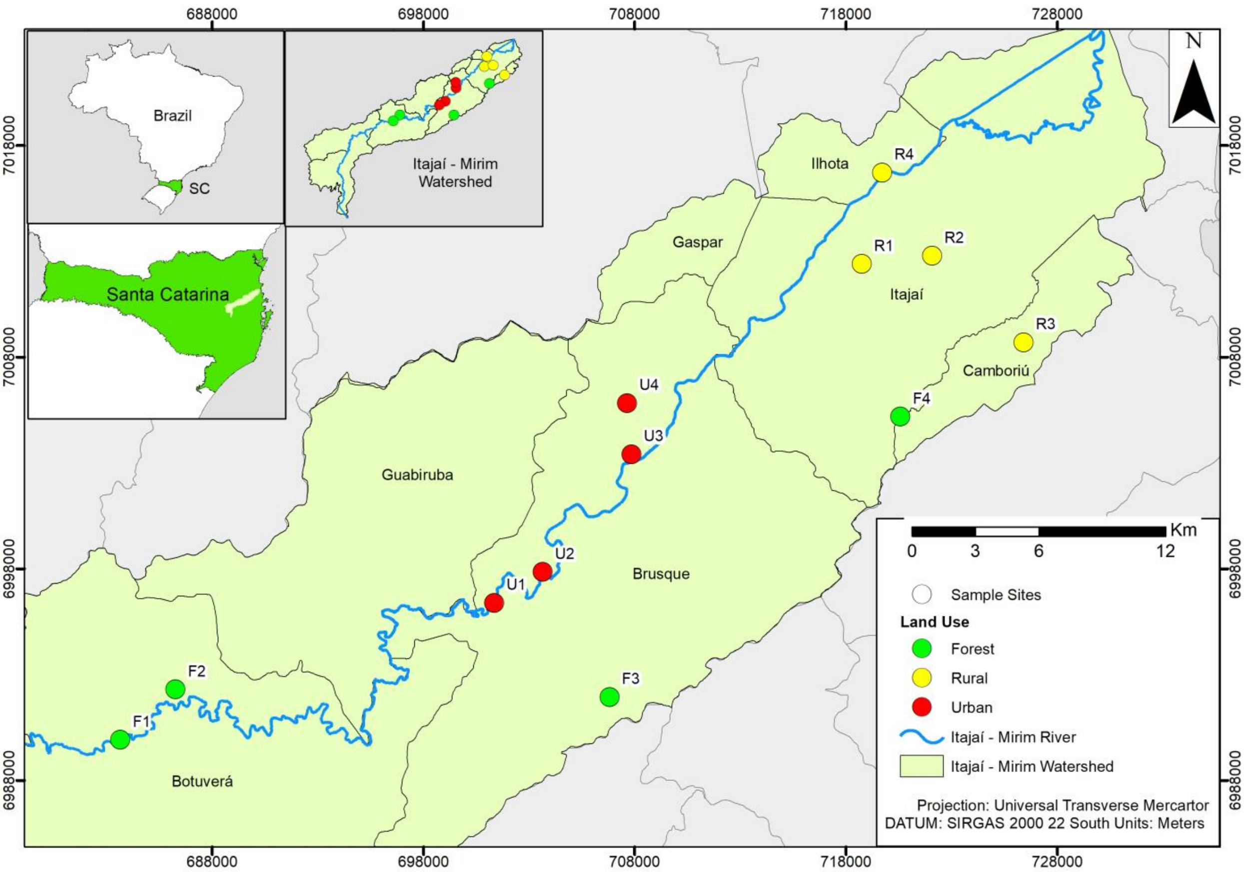

Figure 1. The Itajaí-mirim river basin and the 12 sampling sites in the study area. F1–F4 are forest land use sites; R1–R4 are rural; U1–U4 are urban.

Table 1. Geographic coordinates and descriptions of the sampling sites of the streams in the Itajaí-mirim river watershed. See Figure 1 for sampling sites.

Site	Longitude	Latitude	Site Description
F 1	49°8′48′′ O	27°11′59′′ S	Stream upstream of the basin, a shaded, lotic environment with rapids and few backwaters, near the main tributary.
F 2	49°07′12′′ O	27°10′45′′ S	Stream upstream of the watershed, partially shaded, lotic environment with rapids and backwaters, heterogeneous substrate. Located after the waterfall, on private property.

Table 1. *Cont.*

Site	Longitude	Latitude	Site Description
F 3	48°54′45″ O	27°10′47″ S	Stream in the intermediate region, between upstream and downstream of the main course of the river in the watershed, shaded and lotic environment, heterogeneous substrate, presence of downstream dwellings.
F 4	48°46′34″ O	27°03′29″ S	Stream downstream of the basin, partially shaded, lotic environment with rapids and backwaters, heterogeneous substrate, far from the main tributary, with animal breeding in the surrounding area. Located on private property.
R 1	48°47′44″ O	26°59′36″ S	Stream downstream of the basin, without shading, lentic environment, homogeneous substrate, distant from the main tributary.
R 2	48°46′03″ O	26°58′58″ S	Stream downstream of the basin, without shading, lentic environment, homogeneous substrate with the presence of periphyton, near the main tributary, surrounded by rice plantations.
R 3	48°43′05″ O	27°01′32″ S	Stream downstream of the basin, partially shaded, intermediate flow environment, heterogeneous substrate, located on private property, with animal breeding in the surrounding area.
R 4	48°47′13″ O	26°57′15″ S	Stream downstream of the basin, without shading, intermediate flow environment, homogeneous substrate, fine. Stream partially piped (rural road), with a steep slope.
U 1	48°58′06″ O	27°08′25″ S	Stream in the intermediate region, between upstream and downstream of the main course of the river, without shading, a lentic environment, and a homogeneous substrate. Water with a foul odor, the formation of supernatant oil, and pigmentation. Stream in the allotment region, with a steep slope.
U 2	48°56′38″ O	27°07′45″ S	Stream in the intermediate region, between upstream and downstream of the main course of the river, shaded, with intermediate flow and heterogeneous substrate. Presence of supernatant oil in the water, surrounded by banana plantations and partially piped.
U 3	48°90′40″ O	27°04′34″ S	Stream in the intermediate region, between upstream and downstream of the main course of the river, partially shaded, with intermediate flow and homogeneous substrate. Water with a foul odor, Chinonomidae larvae with red pigmentation in the flow, and the presence of direct sewage in the site.
U 4	48°54′23″ O	27°03′15″ S	Stream in the intermediate region, between upstream and downstream of the main course of the river, without shading, environment and intermediate flow, homogeneous substrate, with steep slope. Water with a foul odor, pigmentation, and the presence of larvae of the Hirudinea group in the flow, with direct sewage in the site

2.2. Measurements of Environmental Variables

Environmental variables were measured synchronously at each site and/or transported to the laboratory. The width of the stream (Wid) was measured as the average of three transects using a tape measure, considering the wet area. The water velocity (Vel) was measured in three cross-sections in a range of three times the value of the width of the wet area through the float method [39], being considered the mean of the values. Depth (Dep) was measured as the average of three equally spaced points along the transect of 50 m using a graduated rod. Rainfall (Pluv) was established by accumulated precipitation seven days prior to collection, with data from rainfall stations a maximum of 20 km away from the sampling site.

Water pH, dissolved oxygen (DO), electrical conductivity (Cond), and water temperature (Temp) were measured using a handheld probe model Akrom—86021 on site. However, the variables total phosphorus (TP), nitrate (NO_3), nitrite (NO_2), and ammonium nitrogen (NH_4) water samples were collected, packaged under refrigeration, forwarded for laboratory analysis for no more than 24 h, and measured according to Standard Methods [40].

2.3. Macrobenthic Invertebrate Sampling

Benthic macroinvertebrates were sampled seasonally in February and May 2022 (summer and fall) and July and November 2022 (winter and spring) in 12 sites distributed in the Itajaí-mirim river basin. A Surber sampler (opening 30 × 30 cm, mesh 250 μm) was used to collect benthic macroinvertebrates, that were stored in plastic bags, and then fixed in 70% ethanol [41].

In each site, three samples were collected along a 50 m stretch in a including ponds, backwaters, and rapids, totaling 144 samples in the studied period [27,42]. In the laboratory, the aquatic fauna was carefully washed, identified at family level in the stereomicroscope, and accounted for using identification keys [43,44]. Taxonomic sufficiency for identifying biota at the family level is justified when the aim of this study is to distinguish sites affected by human activities from those in their natural or near-natural state (different land uses) [45,46].

The following metrics/indices were calculated to evaluate the structure of benthic communities: richness (S), total abundance (N), diversity of Shannon–Weaver (H') [47] and Simpson dominance index (D) [48].

2.4. Statistical Analysis

In order to evaluate the spatial–temporal variation patterns of the benthic community descriptors, models were applied to the data, assuming as variable responses the abundance, richness, and diversity of Shannon and Simpson (separately) and as predictor variables the seasons and categories of land use (forest, rural, and urban). All models underwent validation by inspection of residuals and assumptions, and the most appropriate analytical strategy for each data set was adopted. The spatial–temporal variation of richness was evaluated with a Variance Analysis, and Shannon and Simpson diversity indices were evaluated with generalized linear models (GLMs) with a Poisson distribution. Abundance was evaluated with generalized linear models with a negative binomial distribution.

In order to evaluate the effect of environmental variables on the benthic community descriptors, multiple regression models were applied, assuming as response variables the abundance, richness, and diversity of Shannon and Simpson (separately) and as predictive variables the limnological, physical, and spatial parameters. The environmental variables were previously submitted to a correlation test in order to identify and remove colinear variables ($r > 70\%$). The analyses were performed with total phosphorus, water conductivity, nitrate, nitrite, depth, temperature, width, pH, water speed, dissolved oxygen, and rainfall. Considering that diversity and environmental variables are spatially structured in the landscape, Moran's eigenvector maps (MEMs) were calculated to incorporate the spatial component in the analysis. The first MEM axis was used as a special predictor in regression analysis. The exploratory variables were all z-score standardized (mean 0, std unit) prior to the analysis. All models underwent a backward selection process, and the best model was selected according to Akaike's criteria. For the GLM models, we calculated the deviance-based R^2 as $R^2 = 1 - \frac{Deviance}{Null\ Deviance}$. Then the Adjusted R^2 was computed as $1 - \left(\frac{n-1}{n-p}\right)1 - R^2$, where p is the number of parameters in the linear predictor and n is the sample size. All analyses were performed in the software R, version 4.2.2 [49].

3. Results

3.1. Local Environmental Variations

The physical and chemical parameters of water were measured by mean values and standard deviations between the four sampling points for the same treatment (land use), as described in Table 2. The water temperature values did not show significant differences between land uses, with higher elevation in the summer season. pH and DO show decreasing values from forested to rural and urban land use gradient. The conductivity and nutrient concentrations, TP and NH_4, showed increasing values from forested–rural–urban gradient. Nitrite had no variation between forested and rural streams, with an increase in the indicator for urban streams, and nitrate also showed higher concentrations in urban streams. These results express the increase in nutrient concentration and low oxygenation in the waters and may compromise the survival of aquatic biota.

Table 2. Mean values and standard deviation of physical and chemical water parameters by land use in the four seasons of the year in the streams of the Itajaí-mirim river.

Land Use	Season	Temp (°C)	pH (Unit)	DO (mg/L^{-1})	Cond (µS cm^{-1})	TP (mg/L^{-1})	NO_3 (mg/L^{-1})	NO_2 (mg/L^{-1})	NH_4 (mg/L^{-1})
Forest	Summer	23.11 ± 0.56	8.32 ± 0.21	9.61 ± 0.89	43.62 ± 23.99	0.01 ± 0	0.33 ± 0.09	0.05 ± 0	0.11 ± 0.01
	Fall	17.21 ± 0.64	8.12 ± 0.04	8.56 ± 0.88	36.53 ± 17.44	0.02 ± 0	0.38 ± 0.13	0.05 ± 0	0.10 ± 0
	Winter	16.95 ± 0.87	7.95 ± 0.36	8.60 ± 0.26	36.57 ± 16.93	0.02 ± 0	0.33 ± 0.12	0.10 ± 0	0.10 ± 0
	Spring	16.67 ± 0.56	7.24 ± 0.40	8.09 ± 0.34	33.58 ± 15.34	0.01 ± 0	0.45 ± 0.14	0.10 ± 0	0.10 ± 0
Rural	Summer	23.45 ± 0.50	7.67 ± 0.18	6.38 ± 2.32	61.17 ± 12.47	0.17 ± 0.29	0.21 ± 0.01	0.05 ± 0	0.18 ± 0.13
	Fall	18.03 ± 0.26	7.37 ± 0.22	6.63 ± 2.30	44.56 ± 9.04	0.10 ± 0.10	0.36 ± 0.10	0.05 ± 0	0.10 ± 0
	Winter	17.93 ± 0.59	7.20 ± 0.65	5.50 ± 1.31	55.82 ± 10.31	0.04 ± 0.02	0.28 ± 0.15	0.10 ± 0	1.01 ± 1.38
	Spring	17.41 ± 0.19	7.06 ± 0.67	6.18 ± 2.31	84.49 ± 65.87	0.10 ± 0.10	0.45 ± 0.36	0.10 ± 0	1.37 ± 1.82
Urban	Summer	23.87 ± 0.19	7.22 ± 0.33	3.82 ± 1.27	250.11 ± 70.90	0.51 ± 0.29	0.52 ± 0.57	0.09 ± 0.04	7.79 ± 2.83
	Fall	18.44 ± 0.32	7.11 ± 0.08	3.13 ± 1.39	200.29 ± 134.97	0.40 ± 0.22	0.53 ± 0.42	0.23 ± 0.26	3.90 ± 3.09
	Winter	18.32 ± 0.10	7.22 ± 0.28	4.39 ± 2.14	272.00 ± 129.01	0.53 ± 0.42	0.55 ± 0.70	0.11 ± 0.01	4.80 ± 4.12
	Spring	17.87 ± 0.42	6.45 ± 0.14	3.48 ± 2.28	196.36 ± 49.31	0.91 ± 0.83	0.63 ± 0.62	0.12 ± 0.02	5.31 ± 3.37

The habitat physical parameters are described in Table 3. Streams in a forested environment presented the widest and shallow watercourse, while in rural and urban environments, it is narrower and deeper. The rainfall difference did not change the water velocity in both treatments. Forested streams showed water speed higher than streams in rural and urban environments, ratifying the lotic environments when compared to the others.

Table 3. Mean values and standard deviation of environmental parameters of the streams of the Itajaí-mirim river by land use in the four seasons of the year.

Land Use	Season	Wid (m)	Vel (m/s)	Dep (m)	Pluv (mm)
Forest	Summer	2.64 ± 1.23	0.30 ± 0.07	0.14 ± 0.04	33.27 ± 14.76
	Fall	2.88 ± 0.42	0.37 ± 0.18	0.12 ± 0.03	144.00 ± 11.66
	Winter	1.88 ± 0.12	0.34 ± 0.10	0.12 ± 0.03	6.35 ± 3.50
	Spring	3.42 ± 1.36	0.32 ± 0.08	0.18 ± 0.06	19.99 ± 9.52
Rural	Summer	2.86 ± 1.39	0.10 ± 0.16	0.21 ± 0.13	57.08 ± 6.98
	Fall	2.42 ± 1.10	0.12 ± 0.24	0.22 ± 0.05	159.65 ± 0.10
	Winter	1.75 ± 1.35	0.19 ± 0.31	0.19 ± 0.11	2.54 ± 2.52
	Spring	1.81 ± 1.15	0.08 ± 0.16	0.21 ± 0.12	12.39 ± 6.22
Urban	Summer	2.06 ± 0.92	0.19 ± 0.16	0.23 ± 0.11	44.96 ± 9.97
	Fall	1.41 ± 0.34	0.20 ± 0.16	0.17 ± 0.03	152.24 ± 8.61
	Winter	1.44 ± 0.61	0.18 ± 0.17	0.15 ± 0.02	1.53 ± 0.29
	Spring	1.48 ± 0.47	0.13 ± 0.11	0.14 ± 0.02	25.06 ± 30.16

3.2. Changes in the Macroinvertebrates Community

During the entire sample period, 83.520 individuals were identified, comprising 67 taxa (Appendix A). Considering the interaction of abundance and richness with the gradient of land use and the seasons of the year, it was found that the greatest abundance occurred in the urban streams, with less difference between forested and rural streams, as described in Table 4, being the season not representative (Figure 2a). The highest richness, however, is visible in a decrease in the gradient of land use forested–rural–urban, with a significant influence of seasonality Figure 2b).

Table 4. Abundance, richness, and diversity indices (mean values ± SD) for each land use, throughout the sampling period.

	Abundance	Richness	Shannon (H)	Simpson (D)
Treatment	**Mean ± SD**	**Mean ± SD**	**Mean ± SD**	**Mean ± SD**
Forest	1038.06 ± 514.70	26.75 ± 1.96	4.68 ± 2.82	2.24 ± 1.91
Rural	591.81 ± 254.93	20.62 ± 3.64	5.45 ± 2.30	3.47 ± 1.80
Urban	2340.16 ± 3044.18	8.56 ± 4.51	2.21 ± 1.77	2.01 ± 113

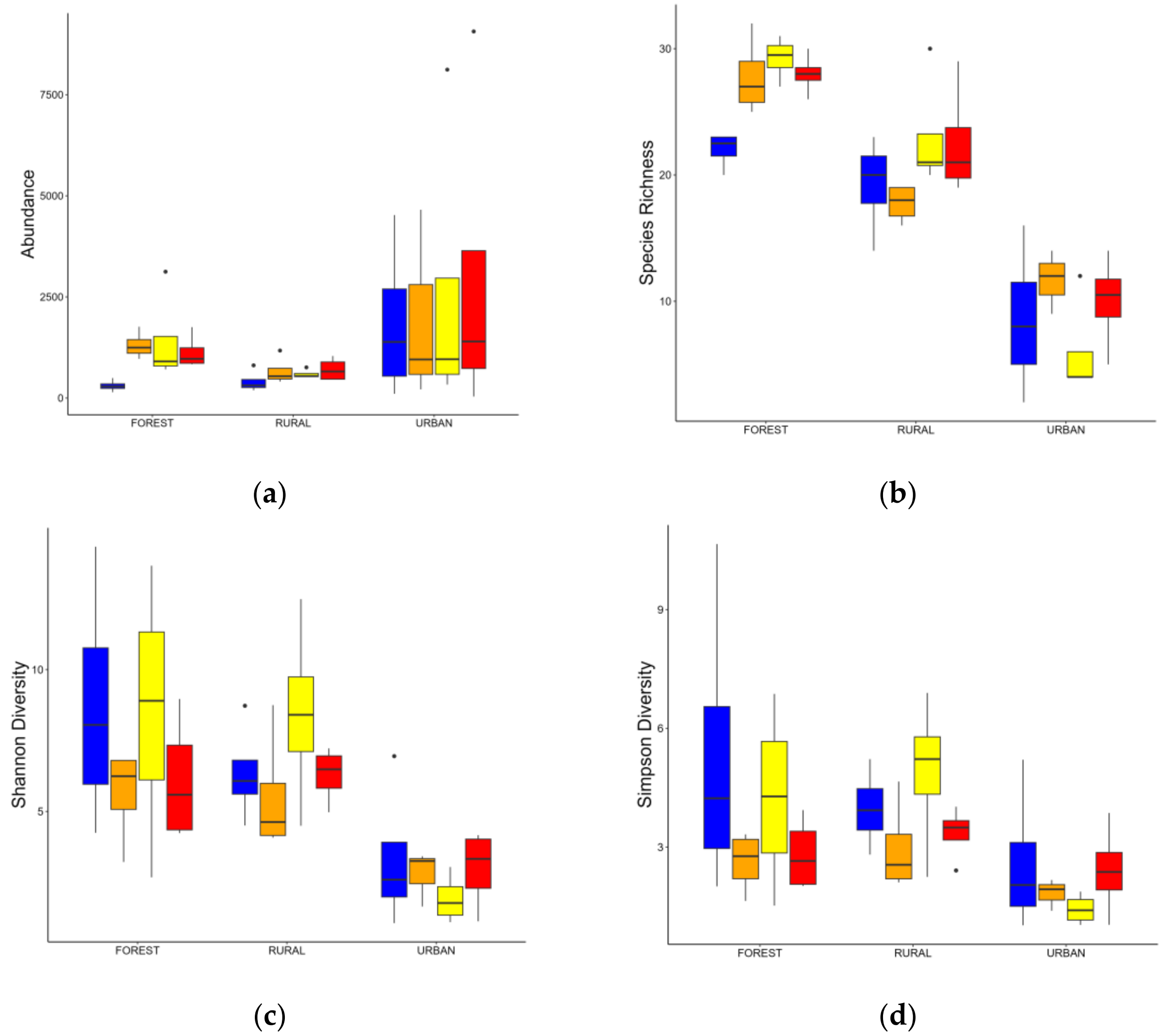

Figure 2. Benthic invertebrate community structure in the streams of the Itajaí-mirim river basin along the gradient from forest to rural and to urban land use: (**a**) abundance, (**b**) richness, (**c**) Shannon diversity, and (**d**) Simpson diversity. The colors represent the seasons (blue = winter; orange = autumn; yellow = spring, and red = summer).

The variation of richness as a function of soil use is significant (F = 101.236; $p = 8.81 \times 10^{-15}$), being more evident for the forested streams, in which there is a decrease in richness in the fall (22 ± 1.41), which is why the interaction between land use and season of the year is marginally significant (F = 2.375; $p = 0.0497$).

Regarding the Shannon diversity index, there is no significant difference among the seasons, as represented in Figure 2c. However, there is a significant difference in taxonomic diversity in relation to land use between forested and urban streams, not evidenced between forested and rural treatments, as described in Table 5. This model can be explained by the presence of 28 families, distributed in 11 orders, present in a forested environment and that are not found in the urban environment, and by the presence of only 10 families (distributed in six orders) present in the forested environment and not identified in the rural environment as follows: Blephariceridae, Crambidae, Haliplidae, Naucoriae, Calamoceratidae, Helicopsychidae, Polycentropodidae, Psychodidae, Psephenidae, and Hyalellidae.

Table 5. Regression analysis of the Shannon diversity difference between streams of different land uses. Considered significant difference: *p*-value < 0.05.

	Estimate	Std. Error	Z value	Pr(>∣z∣)
Intercept	1.978	0.092	21.289	<0.001
Rural Use	−0.083	0.134	−0.624	0.532
Urban Use	−0.969	0.181	−5.341	<0.001

For the Simpson diversity index, which weighs the highest value for dominant, there is also no significant difference among the seasons (Figure 2d) since the dominant taxa between the environments are the same: Chironomidae and Hirudinidae, regardless of seasonality. In relation to land uses, the difference is significant between forest and urban treatments that share 31 taxa in common in the environment, while in forested and rural streams, there is no significant difference, as evidenced in Table 6, because they share 49 taxa in common, presenting an equitable distribution of families between environments.

Table 6. Regression analysis of the Simpson diversity difference between streams of different land uses. Considered significant difference: *p*-value < 0.05.

Fixed Effects	Estimate	Std. Error	Z value	Pr(>∣z∣)
Intercept	1.318	0.129	10.201	<0.001
Rural Use	0.015	0.182	0.085	0.931
Urban Use	−0.587	0.220	−2.661	<0.001

It should be noted that the exclusive presence of taxa was identified for each treatment of land use. There were seven families for forested streams: Calamoceratidae, Polycentropodidae, Psychodidae, Psephenidae, Blephariceridae, Crambidae, and Naucoridae; four families for rural streams: Chaoboridae, Culicidae, Osmylidae, and Pisidiidae; and three families for urban streams: Camaenidae, Hydrobiidae, and Physidae.

We recorded evidence of a greater abundance of taxa in urban environments, while there is no variation in abundance between forested and rural streams. On the other hand, richness is significantly higher in forested streams and decreases in rural streams, presenting lower values in urban streams. The same pattern was observed for diversity indices, with a less evident trend, probably reflecting the substitution of taxa.

3.3. Relationship between Macrobenthic Invertebrates and Environment

The abundance of benthic macroinvertebrate species increases with the increase in phosphorus concentration and electrical conductivity in water, which is more evident in urban streams. Water velocity influence over abundance was significant, due to higher invertebrate abundance in some of the urban streams. It does not represent a direct influence because higher invertebrate abundance occurred in streams with similar water velocity; however, the removal of velocity from the model produced a worse model fit, and the best model is presented. The model with the three parameters explained 53% of the variation in invertebrates' abundance. The data are arranged in Table 7 and Figure 3.

Table 7. Relationship between local environmental variables and family abundance of benthic macroinvertebrate. Considered significant difference: p-value < 0.05. Adjusted R^2: 0.528.

Fixed Effects	Estimate	Std. Error	t Value	Pr(>∣t∣)
Intercept	5.859	0.228	25.672	<0.001
TP	0.922	0.425	2.168	0.030
Cond	0.003	0.001	2.465	0.013
Vel	2.691	0.677	3.975	<0.001

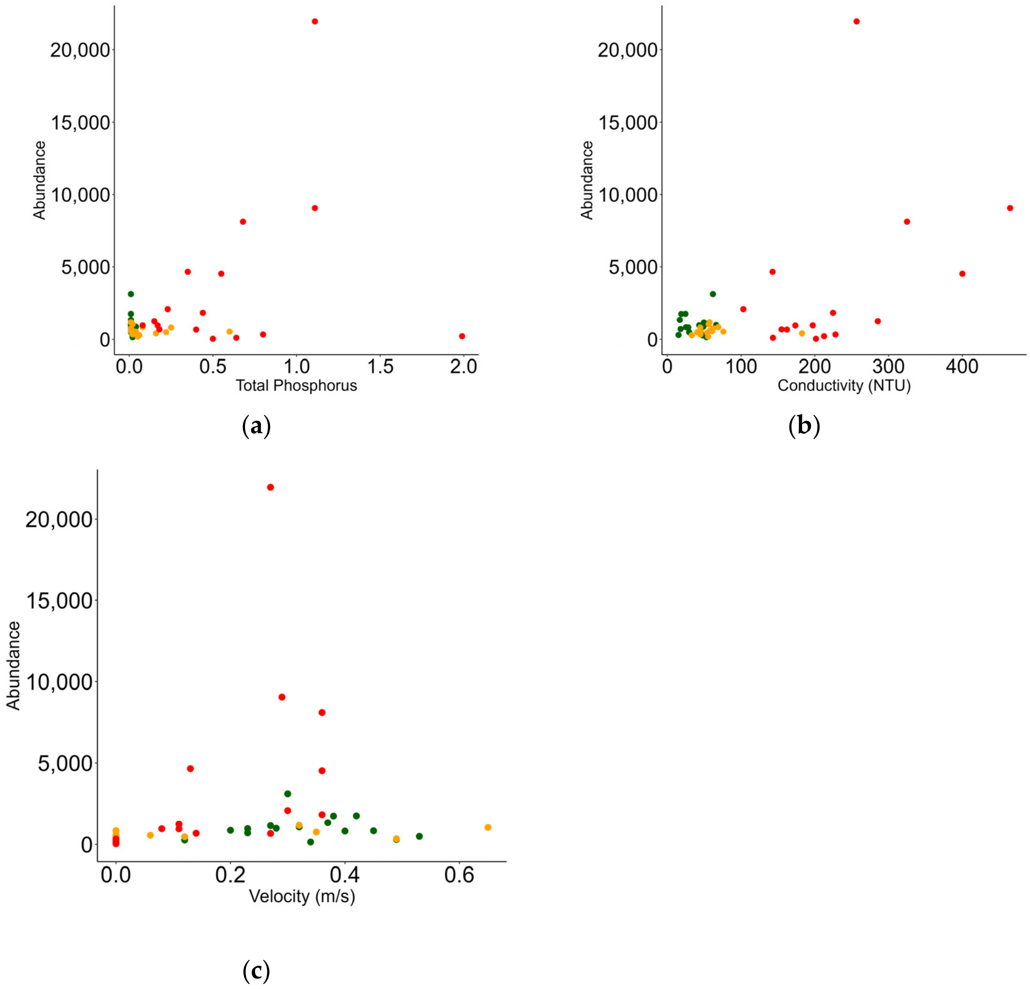

Figure 3. Environmental variables that significantly influence the abundance of benthic macroinvertebrates in the streams of the Itajaí-mirim river basin: (**a**) total phosphorous, (**b**) conductivity, and (**c**) water velocity. The colors represent land uses (green = forested, orange = rural, and red = urban).

Family richness increases with an increase in pH and dissolved oxygen and decreases with an increase in conductivity. Depth, nitrate, and pluviosity were all significantly related to family richness; however, there is no clear association with the response variable, and these predictors should be evaluated with caution. Exploratory models without these predictors produced a worse model fitting, so they were kept. The variables used explained 74% of the variability of family richness in the streams. The data are arranged in Table 8 and Figure 4.

Table 8. Relationship between local environmental variables and family richness of benthic macroinvertebrate. Considered significant difference: p-value < 0.05. Adjusted R^2: 0.74.

Fixed Effects	Estimate	Std. Error	t Value	Pr(>|t|)
Intercept	−11.338	10.450	−1.085	0.284
Cond	−0.040	0.008	−5.030	<0.001
NO_3	4.609	2.019	2.282	0.027
Dep	25.497	9.049	2.818	0.007
pH	2.878	1.410	2.041	0.047
DO	1.353	0.382	3.536	0.001
Pluv	−0.030	0.010	−2.766	0.008

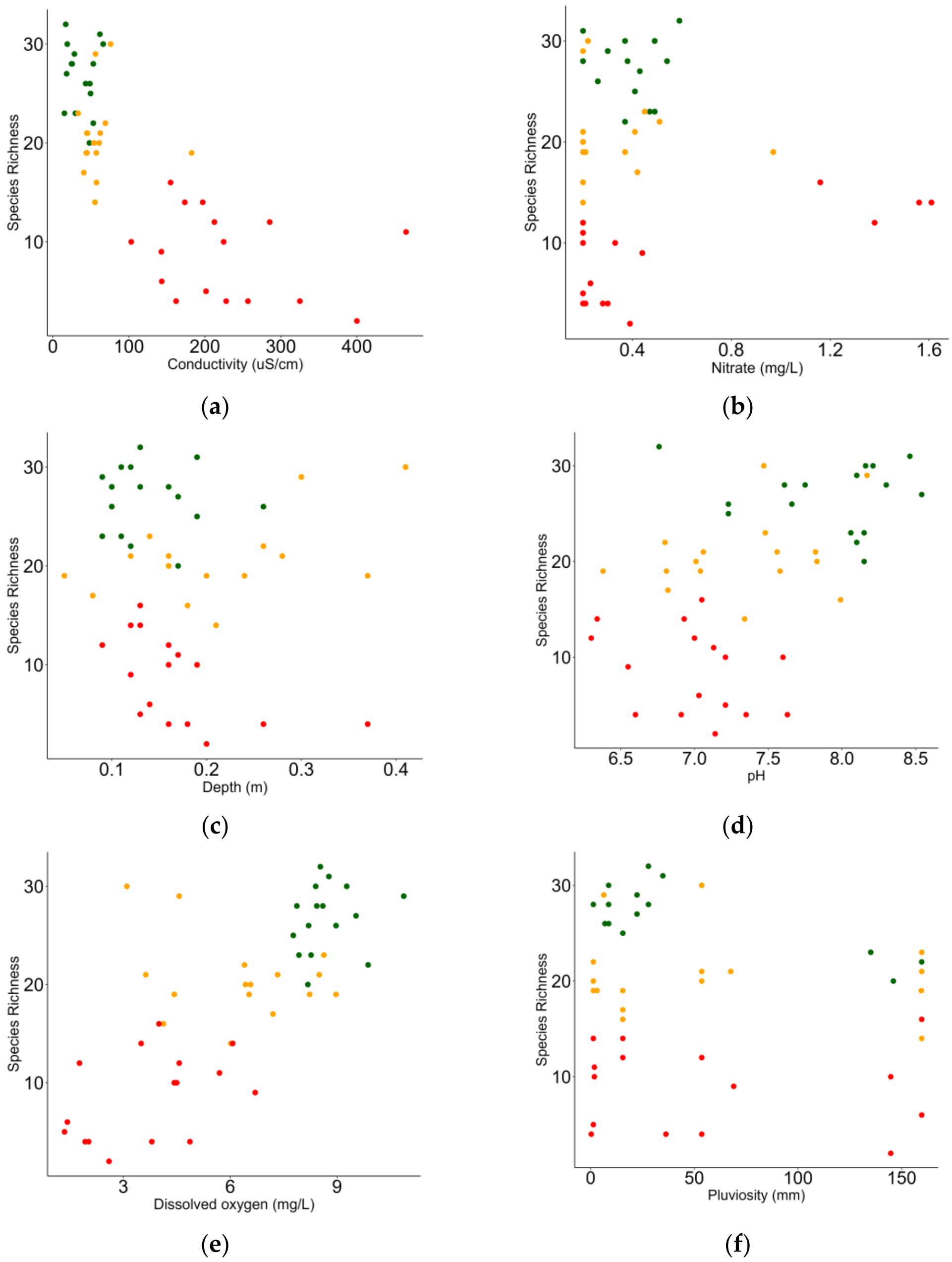

Figure 4. Environmental variables that influence the richness of benthic macroinvertebrate in the streams of the Itajaí-mirim river basin: (**a**) conductivity, (**b**) nitrate, (**c**) depth, (**d**) pH, (**e**) dissolved oxygen, and (**f**) pluviosity. The colors represent land uses (green = forested, orange = rural, and red = urban).

In total, 55% of Shannon's diversity index variation was explained by four variables. The index increases with the increase in nitrate and the width of the stream and decreases with the increase in conductivity in the water. Streams closer to each other present similar diversity, and diversity decreases in more urbanized places and in the forested streams closest to these urbanized streams (MEM1). The model explained 55% of Shannon's diversity response. The data are arranged in Table 9 and Figure 5.

Table 9. Relationship between local environmental variables and the Shannon diversity index of benthic macroinvertebrate. Considered significant difference: *p*-value < 0.05. Adjusted R^2: 0.55.

Fixed Effects	Estimate	Std. Error	t Value	Pr(>|t|)
Intercept	1.468	0.216	6.778	<0.001
Cond	−0.003	<0.001	−3.726	<0.001
NO_3	0.536	0.228	2.355	0.018
Wid	0.139	0.058	2.367	0.017
MEM1	−0.196	0.069	−2.842	0.004

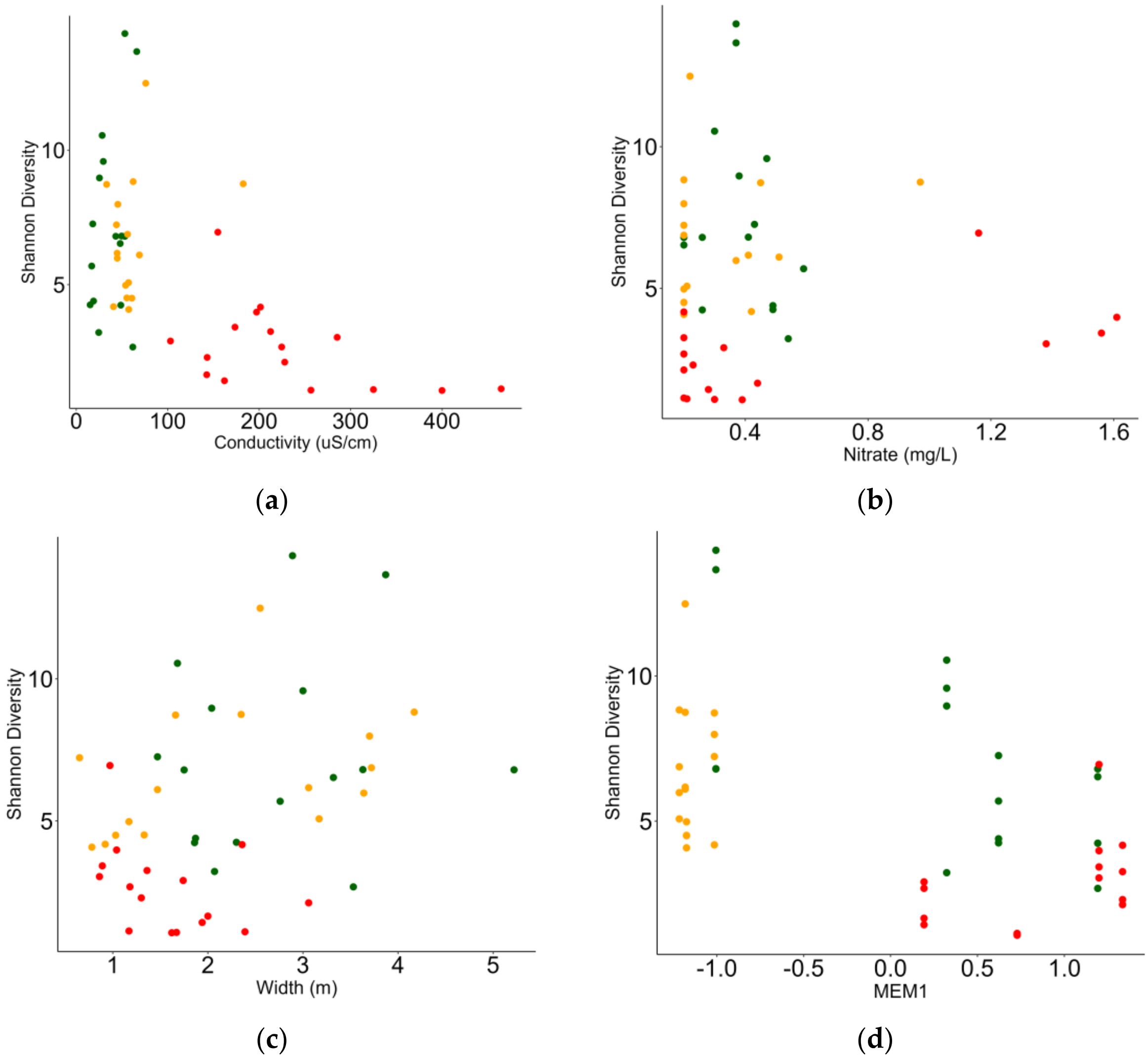

Figure 5. Environmental variables that influence the Shannon diversity index in the benthic macroinvertebrate community in the streams of the Itajaí-mirim river basin: (**a**) conductivity, (**b**) nitrate, (**c**) stream width, and (**d**) space distance between streams (MEM1). The colors represent land uses (green = forested, orange = rural, and red = urban).

Simpson's index increases with the increase in pH in water. Also, streams closer to each other present similar diversity, and diversity decreases in urbanized sites and in the forested streams closest to these urbanized streams (MEM1). The model with both parameters explained 32% of the index variation. The data are arranged in Table 10 and Figure 6.

Table 10. Relationship between local environmental variables and Simpson diversity index of benthic macroinvertebrate. Considered significant difference: p-value < 0.05. Adjusted R^2 = 0.32.

Fixed Effects	Estimate	Std. Error	t Value	Pr(>|t|)
Intercept	−1.020	1.080	−0.944	0.345
pH	0.287	0.143	2.002	0.045
MEM1	−0.259	0.083	−3.097	<0.001

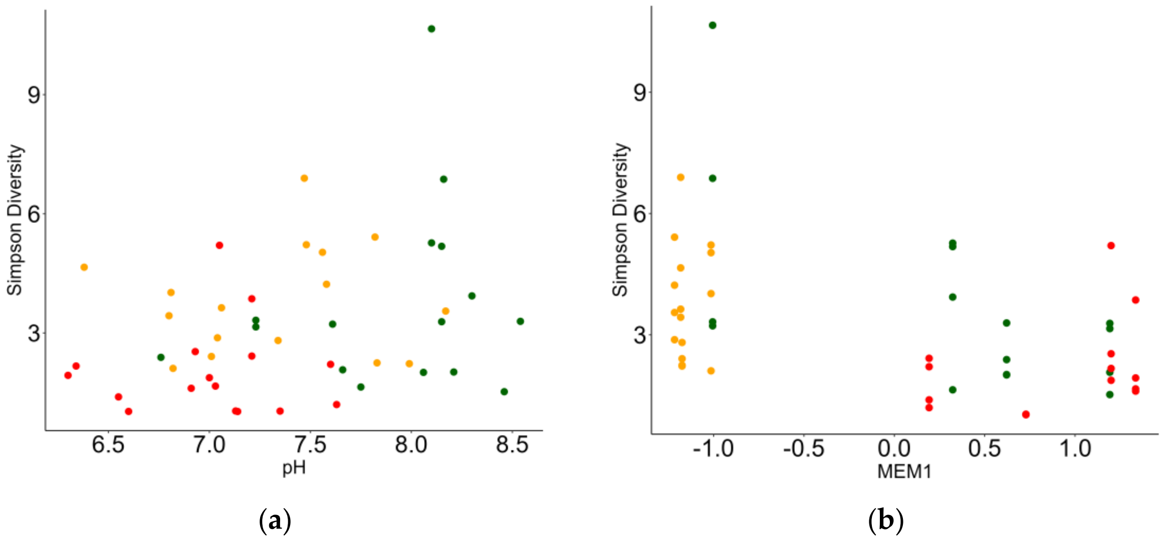

Figure 6. Environmental variables that influence the Simpson diversity index in the benthic macroinvertebrate community in the streams of the Itajaí-mirim river basin: (**a**) pH and (**b**) spatial distance between streams. The colors represent land uses (green = forested, orange = rural, and red = urban).

4. Discussion

The different land uses have a significant impact on the abundance, richness, and diversity of benthic macroinvertebrate communities in the streams of the Itajaí-mirim river basin. While the abundance seemed more affected by the urban nature of the streams, with a less pronounced difference between forested and rural streams, the family richness decreases as the gradient from the forested environment to the urban one. In addition, seasonality variation was only significant for richness. Comparing with data from the literature in studies of benthic communities in streams, it showed similarity to richness [3,6], but differed to the abundance [25,50]. This dissonance is due to local sampling conditions, since streams are heterogeneous environments [15] and because they have small size [10] are susceptible to changes in habitat patterns.

The change in landscape generates changes in local environmental conditions, which directly reflect on the biotic community [3,24]. The benthic macroinvertebrate community diversity responses were first used as bioindicators in the United States. It uses three components of community structure, namely, richness, evenness, and abundance, to describe the community's response to environmental quality [51]. The removal of natural vegetation promotes increased runoff of water, promoting erosion and brooding of streams [11,20]. A fact that corroborates with the reduction in width (wet area) and depth of streams in urban and rural areas is diagnosed in this study. The need to increase agricultural and/or urban areas to meet the demands of the population [17] reduces the physical heterogeneity of the streams and the diversity of the substrate [31]. The benthic community is very sensitive to the type and composition of the substrate, and the siltation processes tend to simplify the composition of the community, favoring the dominance of a few adapted species. The increase in nutrient concentration, electrical conductivity, and decreased oxygenation in water also selects more resistant species to the stressful conditions of the environment [3,6].

It was found that taxa from the orders Ephemeroptera (E), Plecoptera (P), and Tricoptera (T), which are sensitive to environmental degradation [3,25], suffered a decrease in diversity according to land use: forested (14 families) to rural (11 families) and urban (3 families). Also, the replacement of the families Blephariceridae (Diptera), Calamoceratidae (Trichoptera), Crambidae (Lepidoptera), Polycentropodidae (Trichoptera), Naucoridae (Hemitera), and Psephenidae (Coleoptera), present in the forested environment, which settle in a lotic flow environment with high water oxygenation because they are sensitive to environmental changes [52,53], and the Hydrobiidae and Physidae families, snails found in urban environments, adapted to the stressful conditions of a lentic, hypoxic environment with a high concentration of nutrients [3,54].

Data obtained in the Shannon and Simpson diversity index followed the gradient of rural–forested–urban land use. This can be explained by the Intermediate Disturbance Theory (IDH) proposed by Connell (1978) [55]. According to the IDH theory, at intermediate levels of disturbance, as diagnosed in the streams in rural areas of this study, diversity is maximized because species that thrive in the early and late successional stages can coexist. The exclusive presence in rural land use of the families Chaoboridae, Culicidae (Diptera), predators and from environments with lentic flow [44,56]; Osmilydae (Neuroptera), also predators, but from environments with lotic flow [57]; and Pisidiidae, a filter-feeding bivalve that tolerates environments with low oxygen availability [58], confirms the coexistence of taxa with different adaptations in environments with intermediate disturbance (rural), in which they find conditions to establish themselves.

The IDH is a nonequilibrium model used to describe the relationship between disturbance and species diversity [59]. An example of this applicability to benthic macroinvertebrate communities, analogous to the one found in this research, was carried out in the Czech Republic, which shows that the restoration of previously canalized streams in a quasi-natural environment can be colonized by macroinvertebrates, successfully promoting biodiversity over time [27]. Communities that suffer disturbance in the habitat, from gradient to the forest to rural environment, have the process of delayed succession. Local changes increase the space and the possibility of the installation of biotic communities, leading to an increase in local biodiversity for a period until the climax or dynamic balance of these communities occurs in this new environment.

The dominant in the sampled streams remained consistent over time, regardless of seasonality. Rural and forested streams presented a more homogeneous distribution, sharing the presence of 33 families in common, while the comparison of forested and urban streams showed 20 families in common. This is exemplified by the constant presence of the dominant taxa Chironomidae and Hirudinidae in all land uses, followed by the Simuliidae family, as severed in other studies [3,30,31]. Although the forested and rural streams share a substantial number of taxa in common and present an equitable distribution of family between environments [23], the urban streams have a taxonomic composition with less diversity of dominant [6]. Changes in land use lead to the dominance of tolerant generalist taxa (Chironomidae and Oligochaeta) and reduce taxonomic and functional diversity in these sites [11].

Greater abundance is quite common in urban streams [30], due to the dominance of Chironomidae [22]. In the most preserved environments, we have competition between species, which tends to increase diversity and regulate the abundance that makes up the community. Studies also signaled the Chironomidae taxon overlapping the quantitative for other benthic macroinvertebrate assemblages in impacted environments [11,13].

The exclusive presence of taxa for each land use treatment presented in this study suggests that different environments in the streams house unique families [16]. Calamoceratidae (Trichoptera) depend on the presence of plant fragments, as they feed on them and build cocoons to shelter and protect themselves from predators [44,60]. The absence of this material makes it impossible for this family to remain in the environment. This pattern was observed in the gradient of richness, forested–rural–urban, which can be a reflection of the substitution of exposed families to different environmental conditions [3,24,53]. Streams in forested environments presented water velocity higher than other land uses, characteristic of lotic environments [53], with less anthropic interference, maintaining natural characteristics [25], compared to streams in impacted environments, usually with lentic and/or intermediate flow [3,30]. Even with the difference in rainfall in sampling sites, the water velocity was not affected by the different land uses in the seasons. Dissonant factor to other studies in streams [15,61]. This confirms the importance of considering habitat heterogeneity when studying biodiversity in streams [15].

We had predicted that the abundance, richness, and diversity indices would follow the gradient of forested–rural–urban land use, with seasonal influence; however, this pattern is followed only for richness. We verified that the abundance and diversity in the benthic macroinvertebrates community depend not only on environmental variables but also on the spatial component of the distribution of streams in a watershed. The shared influence of environmental components (climatic, hydromorphological, physical, and chemical) modulates macroinvertebrate communities [15]. Streams closer to each other have similarity in family composition, which may be the result of migratory processes and/or colonization by the continuous flow of water [61,62].

We also anticipate the cascade effect with an increase in nutrients in the gradient of soil use, which differed from expected in relation to the increase in water temperature, which did not observe significant differences between treatments, even presenting five sampling sites without shading (R1; R2; R4; U1 and U4) and two partially shaded (R3 and U3). This is probably due to the sampling period that occurred in the early morning, a period of lower sun irradiance in water [53], but a result contrary to other studies carried out in streams [24,30].

Finally, human occupation of the urban areas of the Itajaí-mirim river watershed had a greater impact on the richness and diversity of the benthic macroinvertebrate community. Results are diagnosed in other studies [6,25,30]. It is considered that of the nine municipalities that make up the Itajaí-mirim river basin, 1677.2 km^2 of drainage territory, only three municipalities have a sewage collection network, a tiny 174 km (Itajaí, 112 km; Ilhota, 16 km; Camboriú, 46 km) [36]. The discharge of pollutants directly into watercourses in urbanized regions is affecting the basic conditions of the integrity of ecological processes in these landscape units. It is suggested the implementation of networks for sewage treatment in the area, as well as the need to consider land use to develop conservation and management strategies to ensure biodiversity in streams.

5. Conclusions

This study generated new knowledge about ecology in subtropical streams, asserting the influence of different land uses in benthic macroinvertebrate communities, regardless of seasonality. It was possible to conclude that the spatial distribution of the streams in a watershed promotes the similarity in the composition of the benthic biota among nearby sampling units, not only the environmental variables being the factors that interfere in the process of establishing the assemblies in the environments.

Considering that the sampling occurred in first and second stream order, these should be reference sites in ecological integrity since they are the initial drainage contribution in a watershed. The water contamination was evidenced by the results of physical and chemical variables and by the dominant presence of taxa resistant to these stressful environments in all land uses. Also, surprisingly, the significant variation of rainfall did not alter the richness and diversity of benthic macroinvertebrates in the Itajaí-mirim river basin. The region is constantly affected by floods, which signals the establishment of communities before this kind of weather.

Finally, it is raised that new studies are carried out, considering that this was a pioneering study of the benthic community in the watershed of the Itajaí-mirim river and human occupation of this territory is affecting the maintenance of biota. Knowing the ecology of the streams allows us to infer, with a view to the conservation and health of the aquatic ecosystem, maintaining the biodiversity in these ecosystems.

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Appendix A

Table A1. Abundance of benthic macroinvertebrate community collected in the streams of the Itajaí-mirim watershed.

Taxa		Forest	Rural	Urban
Megaloptera	Corydalidae	43	4	0
Odonata	Aeshnidae	21	8	1
	Dicteriadidae	23	17	3
	Calopterigydae	73	58	17
	Libellulidae	153	47	5
	Gomphidae	16	22	0
	Megapodagrionidae	53	12	0
	Coenagrionidae	6	3	0
Ephemeroptera	Caenidae	97	14	0
	Baetidae	402	572	0
	Leptohyphidae	163	74	0
	Leptophlebiidae	439	94	0
Plecoptera	Perlidae	236	13	0
	Gripopterygidae	355	25	0
Trichoptera	Hydroptilidae	124	185	0
	Hydropsychidae	387	157	90
	Philopotamidae	35	2	0
	Leptoceridae	153	15	20
	Calamoceratidae	196	0	0
	Lmnephilidae	12	20	0
	Helicopsychidae	8	0	6
	Polycentropodidae	12	0	0
Diptera	Tipulidae	124	41	21
	Tabanidae	17	12	5
	Psychodidae	4	0	0
	Empididae	227	56	60
	Chironomidae	10.298	4.537	51.803

Table A1. *Cont.*

Taxa		Forest	Rural	Urban
	Ceratopogonidae	403	201	356
	Syrphidae	3	1	0
	Simuliidae	366	819	466
	Blephariceridae	3	0	0
	Chaoboridae	0	13	0
	Culicidae	0	9	0
Coleoptera	Haliplidae	9	0	24
	Elmidae	782	147	47
	Girinidae	16	26	1
	Staphylinidae	13	2	0
	Hydrophilidae	5	2	0
	Dytiscidae	14	19	1
	Psephenidae	100	0	0
Hemiptera	Gerridae	15	25	0
	Corixidae	63	17	0
	Pleidae	7	2	0
	Mesoveliidae	63	73	4
	Belostomatidae	25	149	3
	Veliidae	76	12	7
Lepidoptera	Pyralidae	35	22	0
	Crambidae	6	0	0
Neuroptera	Osmylidae	0	8	0
Heteroptera	Naucoridae	18	0	0
Arthropoda	Collembola	123	449	22
Acari	Hydracarina	267	305	38
Crustacea	Ostracoda	19	372	0
	Macrobrachium	20	7	6
Annelida	Oligochaeta	354	352	567
	Hirudinidae	29	76	3.366
Nematoda	Nematoda	23	27	33
Basommatophora	Planorbidae	21	102	68
	Ancylidae	10	27	9
	Physidae	0	0	2
Caenogastropoda	Thiaridae	24	32	249
Architaenioglossa	Ampullariidae	0	5	39
Stylommatophora	Camaenidae	0	0	1
Veneroida	Pisidiidae	0	223	0
	Sphaeriidae	13	56	88
Amphipoda	Hyalellidae	7	0	3
Littorinimorpha	Hydrobiidae	0	0	11

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