Acta Limnologica Brasiliensia

Publication of the Brazilian Association of Limnology

Thematic Section: Neotropical Zooplankton Symposium
Acta Limnologica Brasiliensia, 2025, vol. 37, e103
https://doi.org/10.1590/S2179-975X3224
ISSN 2179-975X on-line version

Effects of urban pollution on zooplankton diversity along the Almada River (Bahia, Brazil)

Efeitos da poluição urbana sobre a diversidade de zooplâncton no Rio Almada (Bahia, Brasil)

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Cite as: Santos, J.F., Nascimento, J.R. and Simões, N.R. Effects of urban pollution on zooplankton diversity along the Almada River (Bahia, Brazil). *Acta Limnologica Brasiliensia*, 2025, vol. 37, e103. https://doi.org/10.1590/S2179-975X3224

Abstract: Aim: This study aimed to analyze the influence of small cities on the diversity of the zooplankton community along the Almada River, Bahia. **Methods:** The samples were collected at points upstream (Clean Waters - CW) and downstream (Active Decomposition - DA) of the urban area of three cities: Almadina, Coaraci and Itajuípe, between the years 2020 and 2023. **Results:** Among the physical and chemical variables, only dissolved oxygen and water temperature varied significantly between CW and DA. 90 taxa were identified, of which: 60 from Rotifera, 17 from Cladocera and 13 from Copepoda. It was possible to verify the presence of dominant taxa, characteristic of eutrophic environments, in points downstream of urban areas, namely: *Lecane bulla bulla*, Bdelloidea, *Testudinella patina* and *Platyias quadricornis*. The community attributes with significant variation between CW and DA were: abundance, evenness and the Shannon diversity index. The Jaccard dissimilarity between the CW and DA zones was high, indicating a low rate of species sharing between the CW and DA zones. In the BIOENV analysis, the variables of dissolved oxygen and electrical conductivity associated with variation in community structure were chosen. **Conclusions:** pollution from urbanized regions in the Almada River reduces the evenness and diversity index of shannon, and increases the abundance of the zooplankton community.

Keywords: aquatic biodiversity; eutrophication; anthropogenic impacts; urbanization.

Resumo: Objetivo: Esse estudo teve como objetivo, analisar a influência da poluição oriunda de pequenas cidades sobre a diversidade da comunidade zooplanctônica ao longo do Rio Almada, Bahia. **Métodos:** As amostras foram coletadas em pontos a montante (Águas limpas-AL) e a jusante (Decomposição ativa-DA) da área urbana de três cidades: Almadina, Coaraci e Itajuípe, entre os anos de 2020 e 2023. **Resultados:** Dentre as variáveis físicas e químicas, somente o oxigênio dissolvido e a temperatura da água variaram significativamente entre AL e DA. Um total de 90 táxons foram identificados, sendo: 60 de Rotifera, 17 de Cladocera e 13 de Copepoda. Foi possível constatar a presença de táxons dominantes, característicos de ambientes eutrofizados, nos pontos a jusante das áreas urbanas, sendo elas: *Lecane bulla bulla,* Bdelloidea, *Testudinella patina* e *Platyias quadricornis*. Os atributos da comunidade com variação significativa entre AL e DA foram: abundância, equitabilidade e o índice de diversidade de Shannon. A dissimilaridade de Jaccard entre as zonas de AL e DA foi alta, indicando um baixo índice de compartilhamento de espécies entre as zonas de AL e DA. Na análise BIOENV, as variáveis de oxigênio dissolvido e condutividade elétrica foram associadas a variação da estrutura da comunidade. **Conclusões:** a poluição oriunda das regiões urbanizadas no Rio Almada diminui a equitabilidade e o índice de diversidade de Shannon e aumentam a abundância da comunidade zooplanctônica.

Palavras-chave: biodiversidade aquática; eutrofização; impactos antrópicos; urbanização.

1. Introduction

Rivers are lotic ecosystems which are essential for human development (Liang et al., 2019). However, the excessive discharge of pollutants into receiving water bodies due to urbanization makes the quality of their surface waters deteriorate (Habib et al., 2020). This occurs because the pollution of water bodies alters their physical, chemical and biological parameters, causing, for example, a loss of biodiversity and eutrophication (Adbarzi et al., 2020; Soni et al., 2022). A consequence of this is an alteration in the structure and the functions of these ecosystems (Borgwardt et al., 2019; Tóth et al., 2019; Webb et al., 2020; Jafarabadi et al., 2021), as well as negative impacts on the river's ecosystem services of self-cleaning, supplying drinking water and even providing aesthetic beauty. This environmental problem is becoming a challenge, especially for developing countries (Blettler et al., 2019).

In Brazil, around 48% of sewage water is not treated and is discharged into the environment (ITB, 2024), with rivers as the main receivers of this form of environmental degradation, affecting human health and water biodiversity. Water biodiversity has a fundamental role in the dynamics of aquatic ecosystems, especially in nutrient cycling and energy flow (Wang et al., 2021; Jiang et al., 2024). Among the biological communities that inhabit these ecosystems, zooplankton is one of the indicators of environmental change, as they provide a complete view of the state of the ecosystem (Majeed et al., 2022).

The zooplankton community is composed of microscopic invertebrate organisms that drift freely in water, among them Rotifers, Cladocera and Copepoda (Kour et al., 2022). They are important environmental indicators due to their short life cycle and their sensitivity to environmental gradients (Nascimento et al., 2023; Palmer et al., 2013). The composition of zooplankton species in the community and their abundance can be altered by pollution and can give an indication of the quality of a freshwater source (Suliman et al., 2019). As well as pollution, which is a combination of harmful alterations to water quality, the zooplankton community is affected by the concentration of nutrients, light, temperature, water transparency, parasitism, predation and competition (Schoener, 1986; Sommer et al., 2012; Zhang et al., 2019).

Despite its many water resources, Brazil has many regional differences and challenges when managing its surface waters. Among these, knowledge of the impact pollution has on zooplankton biodiversity will contribute to estimating the effects of a loss of diversity for each trophic level, strengthening conservation strategies for less impacted rivers (Jeppesen et al., 2011), and proposing restoration methods for the most impacted environments (Louette et al., 2008; Palmer et al., 2013).

In this context, the present study analyzes the effect of urban pollution on the zooplankton community of the Almada River, in the state of Bahia, Brazil. The main hypothesis is that urban pollution affects different aspects of the zooplankton biodiversity along river stretch. Therefore, we expect: (i) less species richness in active decomposition zones (a way of identifying polluted sites) due to a restricted number of species that can live in the environmental conditions of these sites; (ii) a higher overall abundance of zooplankton at the most active decomposition zones due to the opportunism of the species which are more resistant to the polluted environment; (iii) less species evenness in the affected sections due to more pollution-resistant species dominating; (iv) a decrease in the Shannon diversity index of zooplankton between clean water zone and active decomposition zones; (v) a high dissimilarity rate between the zones with clean water and those with active decomposition; (vi) the association of the community with pollution indicators.

2. Methods

Our research was carried out in the South of the state of Bahia, in the Almada River Basin (ARB) (Figure 1). The region is characterized as a tropical rainforest and the climate is classified as tropical according to the Köppen-Geiger classification. The ARB covers a drainage area of approximately 1,572.46 km^2, with a perimeter of 332.4 km. The river spans a total length of 188 km, from its source to its mouth at the Atlantic Ocean. The ARB experiences an annual mean precipitation of 1,780 mm and an average annual temperature of 22.9 °C (Gomes et al., 2012). Its resources

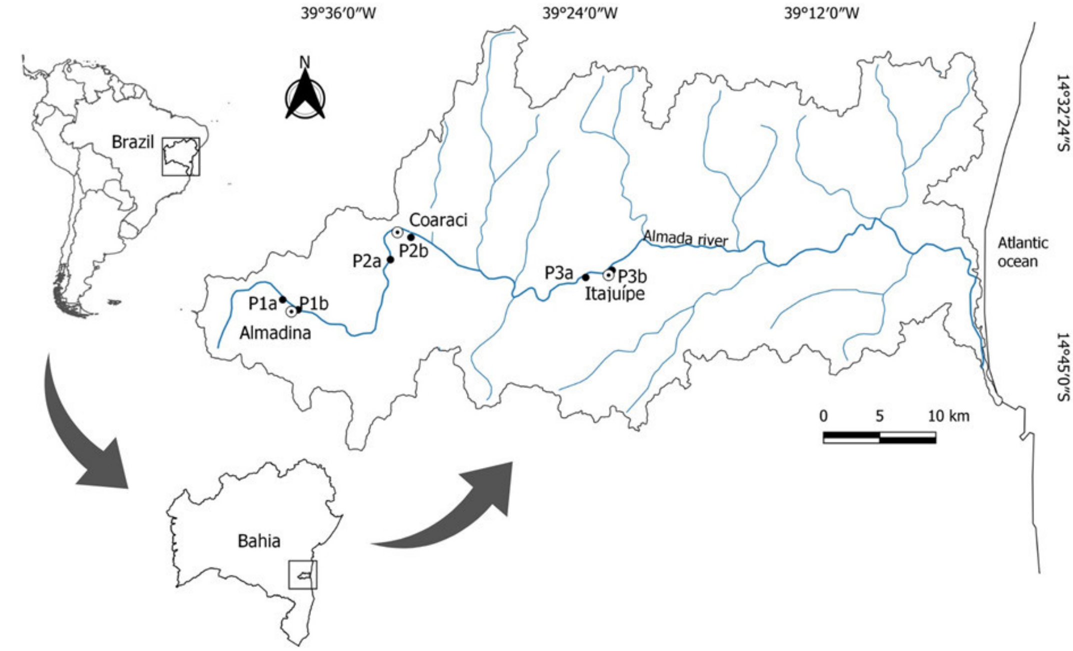

Figure 1. Study area showing the Almada River Basin (ARB) and its respective sampling points for each urban area.

subsidize activities such as fishing, irrigation and public distribution. Along its course, the Almada River exhibits different stream orders. In the city of Almadina, the river is classified as a 4th-order stream, while in Coaraci and Itajuípe, it reaches 6th order, reflecting the increased contribution of tributaries and water volume. The final order of the river, at its mouth, is classified as 7th order, according to Strahler's stream classification system. The sampling sites along the river showed an average width of 5.84 ± 4.5 m and an average depth of 0.39 ± 0.17 m. These measurements vary between different sampling points, reflecting local geomorphological and hydrological characteristics, as well as the influence of anthropogenic disturbances. The sampling points therefore included areas of lesser and greater depth and width along its course.

The following cities were analyzed: Almadina (population of 5,218 and an area of 245.236 km^2), Coaraci (population of 17,333 and an area of 274.500 km^2), and Itajuípe (population of 18,781 and an area of 270.752 km^2) (IBGE, 2022). None of these cities have a sewage treatment system in place, resulting in 0% sewage treatment coverage for their respective populations.

Six data sampling points, spread upstream (clean water zones – CW) situated 1 to 5 km before entering the city limits, representing areas with minimal human impact, and downstream (zones impacted by the effects of urban pollution, representing the zone of Active Decomposition of organic matter – AD) located 1 to 2 km after passing through of three urban areas

The average distance between CW and AD was 3.07 ± 1.81 km. These were sampled annually from 2020 to 2023. The monthly mean discharge data were obtained from the fluviometric station located in the Itajuípe city river section (P3b). During the study (2020 to 2023) of the ARB varied from 1.0 m3/s to 71.4 m3/s, with the period from October to December seeing the highest discharge values, and from January to September the lowest values (Figure 2) (ANA, 2024).

We collected zooplankton using horizontal trawls of a 68 µm plankton net. The average filtered volume per sample was 636 ± 13 L. The filtered volume (V^f) was calculated using the Formula 1:

$$V^f = \pi * r^2 * d \tag{1}$$

where V^f represents the filtered volume, r is the radius of the net mouth, and d is the distance traveled during each tow.

The samples were fixed with 4% formaldehyde buffered with calcium carbonate. Subsequently, the samples were taken to the laboratory to be identified to the species level using specialized literature (Koste, 1978; Reid, 1985; Elmoor-Loureiro, 1997; Sousa & Elmoor-Loureiro, 2019). The species count was performed using an optical microscope with 2.5 ml chambers. Our count effort was of 3 complete

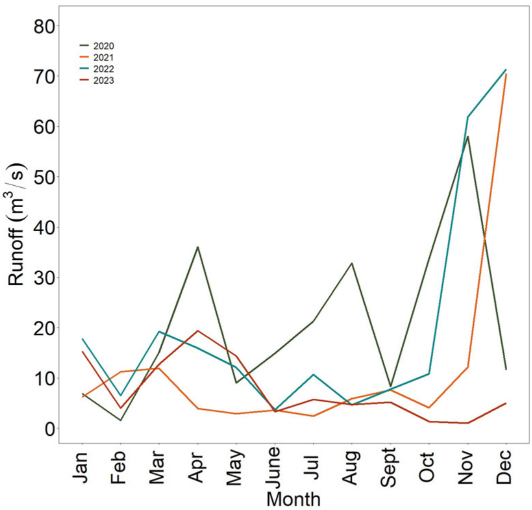

Figure 2. Runoff monthly means at the mouth of the Almada River for the years between 2020 and 2023 **Source:** ANA (2024).

chambers for samples with many individuals (more than 80 in 3 chambers), and of 10 complete chambers for samples with few individuals (less than 80 in 3 chambers). We also measured the pH, water temperature (WT), dissolved oxygen (DO) and electric conductivity (EC) *in situ*, with a Hanna 9828 multiparameter probe.

We estimated the alpha diversity of the community by measuring the following attributes: species richness, abundance, evenness, and Shannon index. Abundance (A) was calculated using the Formula 2:

$$A = (ni * 1) / V^f \quad (2)$$

where ni is the number of individuals per sample and V^f is the filtered volume. The Shannon diversity index (H') was determined using the Formula 3:

$$H' = -\sum_{i=1}^{S} pi * ln\, pi \quad (3)$$

where S is the total number of species and pi represents the relative abundance of each species.

We conducted Student's t and Mann-Whitney (W) tests to evaluate the differences in the diversity attributes and the physical and chemical variables between the CW and AD sampling points.

We used the Jaccard dissimilarity (Legendre & Legendre, 1998) to evaluate the difference in the composition of species between each CW and its corresponding AD zones, in other words, before and after each urban area. The Jaccard dissimilarity value varies from 0 to 1. When the value is close to 0, this means the CW and AD zones share many of the same species. When the value is close to 1, there is a high rate of species replacement between the sites.

We conducted a BIOENV analysis (Clarke & Ainsworth, 1993) to verify the relationship between the environmental variables and the variations in the community's structure. During this procedure, the community's abundance matrix was converted to a dissimilarity matrix based on the Bray-Curtis index. Simultaneously, the environmental data matrix was standardized. The BIOENV analysis seeks the best subset of variables that maximizes the correlation of the community's dissimilarity matrix (Clarke & Ainsworth, 1993). To test the correlation's significance, a Mantel test was carried out (Legendre & Legendre, 1998).

We evaluated the indicator species using the Indicator Value (IndVal) (Dufrêne & Legendre, 1997). This method uses and combines the relative species abundance with the relative frequency at which the species in different habitats occur; in this case, the CW and AD zones. The IndVal organizes the species into groups and provides values between 0 and 1. Species with a significance of ($p<0.1$) were considered as indicators (Dufrêne & Legendre, 1997; Cáceres & Legendre, 2009).

All statistical analyses were performed using R software (R Core Team, 2015), with the BiodiversityR v.2.15.4 (Kindt & Coe, 2005), Vegan v.2.4.3 (Oksanen et al., 2017), labdsv v.2.1-0 (Roberts, 2023), and ggplot2 v.3.4.4 (Wickham, 2016) packages.

3. Results

The Almada River water's physical and chemical parameters displayed a high variability between 2020 and 2023. The pH ranged from 5.90 to 9.36 and was not different between the clean water (CW) and active decomposition (AD) zones. Similarly, the electric conductivity (EC) also presented a high variability, with values between 66 $\mu S.cm^{-1}$ and 1033 $\mu S.cm^{-1}$, and did not differ between the CW and AD zones. This lack of distinction between zones can be attributed to the substantial seasonal variability in both pH and EC across the sampling periods, which likely masked any potential differences between the zones under investigation.

On the other hand, water temperature and the dissolved oxygen (DO) concentration did vary significantly between the CW and AD zones. The mean DO concentration was

6.43 ± 1.27 mg.L^{-1} in the CW zones and 4.45 ± 2.18 mg.L^{-1} in the AD zones. The mean temperature was 25.7 ± 2.41 °C in the CW zones and 26.6 ± 2.37 °C in the AD zones.

We identified 90 Rotifer, Cladocera and Copepoda taxa in the Almada River. The Rotifers had the highest richness of species (60 taxa) and the Lecanidae family was the richest (22 taxa). Seventeen Cladocera taxa were recorded, Chydoridae family the most representative (10 taxa). We recorded 13 Copepod taxa and the Cyclopoida order had the highest value (12 taxa) (Table 1).

The species richness varied between 5 and 33 taxa and did not show a significant difference between the CW and AD (Figure 3a; Table 2). On the other hand, the total abundance of the community differed between the CW and AD zones: on average, 176.8 ± 161.3 ind.m^{-3} were found in the CW zones and 3095 ± 6516 ind.m^{-3} in the AD zones (Figure 3b; Table 2). Furthermore, the evenness differed significantly between the CW and AD zones, varying on average 0.79 ± 0.1 in the CW zones and 0.49 ± 0.2 in the AD zones (Figure 3c; Table 2). The Shannon diversity index was also different, varying on average 2.7 ± 1.0 bits.ind^{-1} in the CW zones and 1.8 ± 0.9 bits.ind^{-1} in the AD zones (Figure 3d; Table 2).

The value of the Jaccard dissimilarity between the CW zones and their respective AD zones was high: it varied from 0.43 to 1.00. The value for the first quartile was 0.68, the median 0.78, and the third quartile 0.9. These results indicate a low rate of species sharing between the CW and AD zones, in other words, the regions above and below the urban areas.

The BIOENV analysis selected the DO and EC variables, showing a 0.26 correlation between the Euclidean distance matrix of the EC and DO and the dissimilarity matrix of the community. The Mantel test indicated that the 0.26 correlation

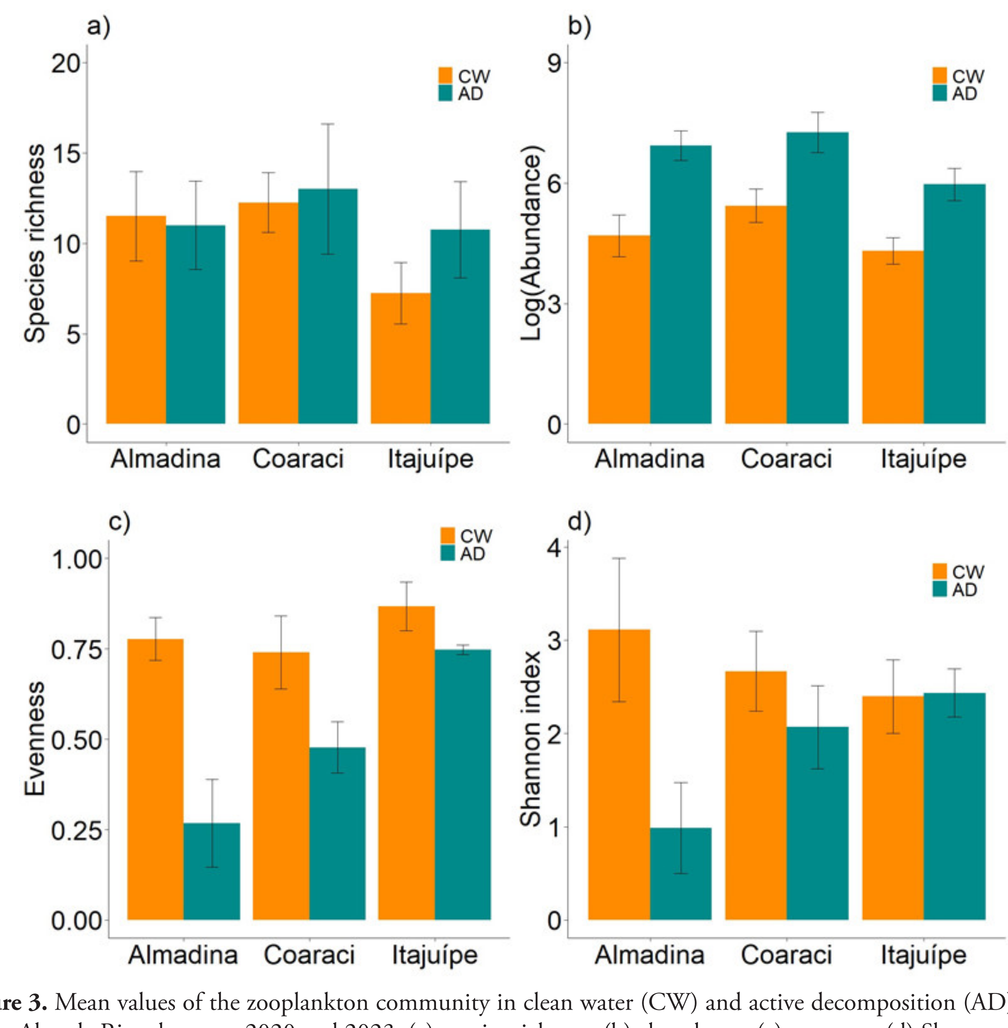

Figure 3. Mean values of the zooplankton community in clean water (CW) and active decomposition (AD) zones of the Almada River between 2020 and 2023. (a) species richness; (b) abundance; (c) evenness; (d) Shannon index.

Table 1. Zooplankton species composition of the Almada River (Bahia) from 2020 to 2023.

ROTIFERA	
Asplanchnidae	
Asplanchna sp (Gosse, 1850)	*Asplanchna priodonta* (Gosse, 1850)
Bdelloida	
Bdelloidea sp (Hudson, 1884)	
Brachionidae	
Brachionus havanaensis (Rousselet, 1911)	*Plationus patulus macracanthus* (Daday, 1786)
Brachionus quadridentatus (Hermann, 1783)	*Platyias quadricornis* (Ehrenberg, 1832)
Brachionus falcatus (Zacharias, 1898)	*Keratella americana* (Carlin, 1943)
Brachionus sp (Pallas, 1766)	*Keratella cochlearis* (Gosse, 1851)
Plationus patulus patulus (Müller, 1786)	*Keratella tropica* (Apstein, 1907)
Dicranophoridae	
Dicranophorus sp (Nitzsch, 1827)	*Dicranophorus claviger* (Hauer, 1965)
Euchlanidae	
Dipleuchlanis sp (Beauchamp, 1910)	*Euchlanis dilatata* (Ehrenberg, 1832)
Dipleuchlanis propatula (Gosse, 1886)	*Euchlanis incisa* (Carlin, 1939)
Euchlanis sp (Ehrenberg, 1832)	
Lecanidae	
Lecane papuana (Murray, 1913)	*Lecane ludwigii* (Eckstein 1883)
Lecane luna (Müller, 1776)	*Lecane bulla* (Gosse, 1851)
Lecane lunaris (Ehrenberg, 1832)	*Lecane sola* (Hauer, 1936)
Lecane robertsonae (Segers, 1993)	*Lecane tabida (Harring & Myers, 1926)*
Lecane cornuta (Müller, 1786*)*	*Lecane stenroosi* (Meissner, 1908)
Lecane curvicornis (Murray, 1913)	*Lecane sp* (Remane, 1933)
Lecane hamata (Stokes, 1896)	*Lecane pyriformis* (Daday, 1905)
Lecane clasterocerca (Schmarda, 1859)	*Lecane quadridentata* (Ehrenberg, 1830)
Lecane signifera (Jennings, 1893)	*Lecane thalera* (Harring & Myers, 1926)
Lecane thienemane (Hauer, 1938)	*Lecane furcata* (Murray, 1913)
Lecane leontina (Turner, 1892)	*Lecane asymetrica* (Murray, 1913)
Lepadellidae	
Colurella obtusa (Gosse, 1886)	*Lepadela acuminata acuminata* (Ehrenberg, 1834)
Lepadella sp (Bory de St.Vincent, 1826)	*Lepadella benjamini benjamini* (Harring, 1916)
Lepadela obtusa (Wang, 1961)	*Lepadella dactyliseta* (Stenroos, 1898)
Lepadella patella (Müller, 1773)	*Lepadella bejamini brasilienses* (Koste, 1972)
Lepadella ovalis (Müller, 1786)	*Squatinella sp* (Bory de St.Vincent, 1826)
Mytilinidae	
Mytilina ventralis (Ehrenberg, 1830)	*Mytilina mucronata* (Müller, 1773)
Scaridiidae	
Scaridium sp (Ehrenberg, 1830)	*Scaridium longicaudum* (Müller, 1786)
Synchaetidae	
Polyarthra dolichoptera (Idelson, 1925)	
Testudinellidae	
Testudinella patina (Hermann, 1783)	*Testudinella mucronata* (Gosse, 1886)
Trichocercidae	
Trichocerca sp (Lamarck, 1801)	
CLADOCERA	
Chydoridae	
Anthalona neotropica (Elmoor-Loureiro & Debastiani-Júnior, 2015)	*Flavalona iheringula* (Kotov & Sinev, 2004)
*Anthalona sp (*Van Damme, Sinev & Dumont, 2011)	*Flavalona margipluma* (Sousa, Santos, Güntzel, Diniz, Melo Junior & Elmoor-Loureiro, 2015).
Chydorus nitidulus (Sars, 1901)	*Karualona muelleri* (Richard, 1897)
Chydorus eurinotus (Sars, 1901)	*Magnospina dentifera* (Sars, 1901)
Dunhevedia odontoplax (Sars, 1901)	*Nicsmirnovius paggii* (Sousa & Elmoor-Loureiro, 2017)
Daphniidae	
Ceriodaphnia cornuta (Sars, 1885)	
Ilyocryptidae	
Ilyocryptus sordidus (Liévin, 1848)	
Macrothricidae	
Macrothrix squamosa (Sars, 1900)	*Macrothrix laticornis* (Jurine, 1820)
Macrothrix triserialis (Brady, 1886)	
Moinidae	
Moina minuta (Hansen, 1899)	
Sididae	
Diaphanosoma birgei (Korínek, 1981)	

Table 1. Continued...

ROTIFERA	
COPEPODA	
Cyclopidae	
Eucyclops n. neumani (Pesta, 1927)	*Microcyclops ceibaensis* (Marsh, 1919)
Ectocyclops bromelicola (Kiefer, 1935)	*Microcyclops finitimus* (Dussart, 1984)
Ectocyclops rubescens (Brady, 1904)	*Thermocyclops decipiens* (Kiefer, 1929)
Mesocyclops ellipticus (Kiefer, 1936)	*Paracyclops andinus* (Kiefer, 1957)
Mesocyclops meridianus (Kiefer, 1926)	*Tropocyclops prasinus* (Fischer, 1860)
Microcyclops alius (Kiefer, 1935)	*Paracyclops fimbriatus* (Fischer, 1853)
Harpacticoida	
Harparticoida (Sars, 1903)	

Table 2. T test values for the attributes of the zooplankton communities between the upstream and downstream zones of the cities of Almadina, Coaraci and Itajuípe along the Almada River, measured between 2020 and 2023.

Attribute	T test	p-value
Richness	-1.44	0.18
Evenness	4.19	<0.01
Shannon index	67	0.03
Abundance	0.1659	<0.01

Table 3. Indicator Values (IndVal) of the zooplankton species found in the Almada River between 2020 and 2023 (p-value ≤ 0.1).

Taxa	Site	IndVal	p-value
Lecane Lunaris	Clean water	0.33	0.08
Bdelloidea	Active decomposition	0.82	0.05
Testudinella patina	Active decomposition	0.60	0.08
Colurella obtusa	Active decomposition	0.57	0.03
Platyias quadricornis	Active decomposition	0.56	0.05
Lecane curvicornis	Active decomposition	0.39	0.10
Lecane hamata	Active decomposition	0.33	0.10
Lecane ludwigii	Active decomposition	0.33	0.09

was significant (p = 0.003). Therefore, there is a link between the environmental and the community structure's variations.

The Indicator Value (IndVal) significantly implied the presence of eight species (p-value ≤ 0.1), with seven species in the AD zones and only one in the CW zones. The Bdelloidea family and the *Testudinella patina* species presented the highest indicator values (Table 3).

4. Discussion

Pollution is one of the main causes of biodiversity depletion in freshwater sources during the Anthropocene (Dudgeon, 2019). In Brazil, this pollution has many causes, but urbanization is one of the central ones and is a source of organic waste, fecal coliforms, nitrogen and phosphorus (Mello et al., 2020). This happens because of the low rate of sewage treatment in Brazilian cities. As a response to the effects of this type of pollution, our results show that the zooplankton fauna of the Almada River (Brazil) responds to the pollution of three cities along its course.

The qualitative urban pollution indicators in the AD zones of the Almada River were the presence of solid residues, animals along the riverbank, exposed banks and the presence of aquatic macrophytes of the *Eicchornia, Polygonum* and *Pistia* genera. We also measured lower concentrations of DO in the AD regions. This is associated with the high concentrations of organic matter caused by the discharge of domestic effluent from urban zones. The decay of the organic matter is accelerated by aerobic bacteria and, therefore, reduces the DO concentration of the water (Blume et al., 2010).

The species richness did not differ between the CW and AD zones. The percentage of species they have in common was low, indicating a change in the river's fauna when it receives a discharge of organic pollutants. Eutrophication often changes the composition of the zooplankton community (Jeppesen et al., 2001; Hsieh et al., 2011; Rogalski et al., 2017; He et al., 2020). This occurs due to the selective conditions of eutrophication, which lead to a change in the proportion of species in each zooplankton taxonomic group and a change in the quantitative indicators (Derevenskaia et al., 2021).

Among the different responses, we verified a qualitative (species richness), quantitative (density of individuals) and quali-quantitive (represented by the IndVal) increase in the species of rotifers in the AD zones. Environments with eutrophication tend to have a higher proportion of rotifers in relation to crustaceans (Ejsmont-Karabin & Karabin 2013; Haberman & Haldna, 2014; Adamczuk et al., 2015; Ochocka & Pasztaleniec, 2016). Consequently, the abundance of zooplankton was higher in the AD zones in comparison to the CW zones, due to the high density of rotifers. Small zooplankton species are favored by pollution, making them dominate in environments with eutrophication (Shao et al., 2010; Jiang et al., 2017). These facts are associated with rotifers' versatility in inhabiting different aquatic environments and occupying rapidly opening niches (Liang et al., 2020). Furthermore, their high tolerance to environmental changes makes them good at recolonizing aquatic environments after strong disturbances (Segers, 2008). According to Wang et al. (2023), the rotifera phylum is one of the main contributors to the increase in zooplankton biomass and abundance, due to a proportional increase in tolerant species and to the species' dominance index in eutrophication conditions.

In this study, we confirmed the presence of the following rotifer taxa, abundant in the AD zones: *Bdelloidea*, *Lecane bulla*, *Platyias quadricornis* and *Testudinella patina*. These results, coupled with those obtained in the IndVal analysis, occur because these organisms are r-strategists. They have a high capacity to adapt, are small, have feeding plasticity, high reproductive rates, reproduce asexually, can produce resistant eggs, have phenotypic variability and a short life cycle (Allan, 1976; Neves et al., 2003).

Evenness was different for the CW and AD zones. This parameter refers to how uniform the relative abundance distribution of different species in the ecosystem or community (Hillebrand et al., 2008). Pollution can increase restrictive conditions for many species, altering which ones make up the environment (von Sperling, 1996). In this situation, species which tolerate change end up dominating polluted water bodies (Derevenskaia et al., 2021). Consequently, this tends to decrease the species evenness in AD zones near urban areas. Ecosystems with a high evenness are often considered more resilient to change, as they do not depend as much on one single species (Ricklefs & Miller, 2000). On the other hand, a low evenness indicates a community dominated by few species, which can make an ecosystem more susceptible to environmental changes and disturbances (Begon et al., 2007). Therefore, AD areas and sections downstream from cities are sites with more biodiversity vulnerability.

Higher Shannon index values for the CW zones indicate a more diverse community, considering both species richness and evenness (Gotelli, 2011; Magurran, 2011). The Shannon diversity index is a positive indication that an environment is ecologically more balanced, with a larger number of species more evenly distributed throughout the ecological niche (Margalef, 1974). In comparison, lower values of the index for the AD zones reveal a less diverse and even community (Gaston & Spicer, 2004; Molles Junior, 2015), which can be caused by environmental disturbances (such as pollution), or even a degradation of the habitat due to other anthropic actions (Cain et al., 2011).

In this study, we use the term 'pollution' in a broad manner to denote environmental deterioration along the Almada River. Excluding species richness, all the other community attributes presented lower values in the AD zones, indicating the negative effect pollution has on the zooplankton community. The variation in the DO concentration and EC (parameters which indicate pollution) were associated with a variation in the structure of the zooplankton community. This suggests that the changes in the structure of the community happened due to the environmental changes in the Almada River. Souza et al. (2022) also demonstrated the negative effect of the presence of cities along the Almada River, by comparing the diversity of aquatic insects upstream and downstream of these cities.

5. Conclusion

We observed variation in the diversity indices of the zooplankton community between the CW and AD zones along the Almada River. These zones exhibited high abundance, low diversity and low evenness of the zooplankton community. Therefore, this study provides information about the zooplankton community's response to anthropic environmental stress factors, showing the viability of using the community as an environmental indicator and for monitoring water quality in future studies.

Acknowledgements

We thank: Agência Nacional de Águas (ANA) for the data provided from the HIDROWEB platform; JFS thanks the Universidade Federal do Sul da Bahia (UFSB) for the undergraduate research scholarship; JRN thanks the Fundação de Amparo à Pesquisa do Estado da Bahia (Fapesb) for the master's scholarship. NRS was grateful to the CNPq for the research productivity fellowships and FAPESB for financial support to the project.

Data availability

The dataset analyzed/produced in this study can be requested from the corresponding author. However, it is part of ongoing research and cannot be made publicly available at this stage to ensure the integrity and originality of subsequent studies derived from it.

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Received: 14 April 2024
Accepted: 15 January 2025

Associate Editors: Cláudia Bonecker, Gilmar Perbiche Neves, Maria Stela Maioli Castilho Noll.