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RESEARCH ARTICLE

Effect of land use on water quality of tropical headwater streams: comparison across rainforest, cropland, pastureland, and oil palm plantation

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Abstract

Oil palm plantations in Guatemala's northern lowlands have been expanding rapidly over the past two decades, driven by large-scale conversion of tropical forests and croplands. This extensive land use change to oil palm has placed enormous pressure on the region's freshwater ecosystems. The extent to which various land uses impact the quality of tropical stream water remains poorly understood. In this study, we evaluated the effect of different land uses on aquatic macroinvertebrates and stream water quality in the northern lowlands of Guatemala. We assessed 19 sampling sites, including at least four replicated headwater streams for each of the four dominant land uses: tropical forest, cropland, pastureland, and oil palm plantation. In each stream, we measured and analyzed physicochemical water quality parameters in both rainy and dry seasons and sampled benthic macroinvertebrates during the dry season. Subsequently, we used the resulting data to calculate two water quality indices and assess the ecological status of the sampling sites: the Water Quality Index (WQI_{simp}), based on physicochemical parameters, and the Biological Monitoring Working Party for Costa Rica (BMWP-CR), based on the aquatic macroinvertebrate community. Water quality scores differed among sampling sites, with sites draining forested catchments exhibiting the highest scores and those draining pastureland and oil palm plantations exhibiting the lowest. In contrast, the WQI_{simp} results did not differ significantly among land uses and were largely influenced by seasonal variations. Overall, water quality scores declined as the proportion of tropical forest within the watershed decreased. Oil palm plantation sites were characterized by the lowest benthic macroinvertebrates species richness and diversity, alongside higher abundance of contamination-tolerant families. These findings reveal significant negative impacts of oil palm plantations on water quality and aquatic macroinvertebrate communities, particularly when compared with forested areas and basic grains croplands. Therefore, safeguarding primary forests and basic grains cropland from further conversion to oil palm is crucial to preserving biodiversity, maintaining ecosystem functions and services, and securing food provision.

Keywords Water quality · Water quality index · Benthic macroinvertebrates · Oil palm plantation · *Elaeis guineensis* · Watershed scale · Tropical streams · Guatemala

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Introduction

Land use is the primary driver of anthropogenic activity at the catchment scale, as landscape patterns control several processes within the watershed (Ekka et al. 2020). For example, land use change can modify whether a terrestrial ecosystem is a source or sink of carbon, which has an impact on the global climate via the carbon cycle (Geist et al. 2008). Furthermore, land use change is a major driver of alterations in freshwater quality (Calijuri et al. 2015), as its effects on rivers result from complex interactions among physical, chemical, and biological processes occurring across spatial and temporal scales (Viza et al. 2024). River networks are hierarchically organized systems, in which local variability is deeply affected by catchment-scale features (Uriarte et al. 2011). Aquatic ecosystems have been altered globally by broad-scale changes in catchment land use such as forestry, agriculture, settlements, and infrastructure developments (Allan 2004; Reid et al. 2019), modifying the character of watersheds, degrading rivers, and resulting in loss of biodiversity and ecosystem functioning (Carpenter et al. 2011; Tickner et al. 2020). Directly assessing the impact of land use change on water quality is necessary to evaluate its effects on receiving aquatic ecosystems and to implement effective actions to protect surface-water bodies (Gorgoglione et al. 2020).

The expansion of intensive agriculture, which has accelerated since the 1960s, has significantly increased crop production in many regions of the world. However, such an increase has emerged as a major driver of biodiversity loss, greenhouse gas emissions, reductions in freshwater quality and quantity, and the disruption of essential ecosystem services vital to the livelihoods of forest-dependent rural communities (Cabrera et al. 2023; Carpenter et al. 2011). Across the tropics, between 1980 and 2000, about 55% of new agricultural land replaced intact forests while 28% replaced disturbed forests (Gibbs et al. 2010). Oil palm plantations alone have expanded globally by approximately 0.7 million hectares per year, mainly at the expense of tropical rainforests (Meijaard et al. 2020), placing substantial pressure on freshwater ecosystems by altering water physicochemical conditions, increasing organic matter inputs, and reducing riparian vegetation (Cabrera et al. 2023; Luiza-Andrade et al. 2017). Agricultural catchments typically exhibit higher stream temperatures (Macedo et al. 2013), elevated sediment loads following forest clearing (Bruijnzeel 2004), and increased nutrient concentrations near cultivated areas (Calijuri et al. 2015; Gorgoglione et al. 2020).

In Mesoamerica, the recent expansion of oil palm has occurred primarily on nonforested lands, particularly former pastures and croplands (Furumo and Aide 2017; Vijay et al. 2016). In northern Guatemala, between 2010 and 2019, half of the new oil palm plantations were established on basic grain farmland (primarily smallholders), 19% on fallow land, and up to 15% on land previously occupied by tropical forests (Hervas 2021). Regardless, oil palm plantations can exert significant impacts on water quality and biota of freshwater ecosystems even when they replace smallholder agricultural areas rather than rainforests (Rojas-Castillo et al. 2023). For example, changes in macroinvertebrate assemblages have been reported in areas of oil palm plantation expansion, attributed to degradation in riparian zones and stream conditions (e.g., alterations in organic matter, changes in stream banks, and increased sediments) that influence macroinvertebrate distribution and occurrences (Chellaiah and Yule 2018b; Rojas-Castillo et al. 2024). Furthermore, activities related to oil palm plantation establishment and exploitation, including road construction and the use of fertilizers and pesticides, affect water flow and nutrient conditions, with significant effects on downstream freshwater ecosystems, including eutrophication (Comte et al. 2015). Consequently, understanding the differentiated impacts of land use, such as oil palm plantations, croplands, and forests, on freshwater ecosystems and services is critically important, particularly in tropical regions where land use conversion is rapidly accelerating (Hervas and Isakson 2020).

Water quality monitoring studies in Guatemala are scarce due to limited economic investment and insufficient analytical capabilities in remote regions. Nevertheless, two particularly relevant studies investigated the impact of land use conversion associated with oil palm plantations on stream water quality and macroinvertebrate communities in lowland Guatemala (Rojas-Castillo et al. 2022, 2023). Findings revealed that streams adjacent to oil palm plantations without riparian forest buffers exhibited warmer and more variable temperatures, higher turbidity, and reduced richness of macroinvertebrate taxa compared with those in primary forests. In contrast, streams in oil palm plantations that conserved riparian forests showed similar characteristics to those in primary forests (Rojas-Castillo et al. 2023). Although the studies provide valuable insights into the contrasting impacts of oil palm plantations and forested areas on water quality and macroinvertebrate communities, it is equally important to examine the effects of oil palm expansion into less intensive land uses, such as smallholder croplands. This trend is becoming increasingly common in Guatemala, and understanding its implications is crucial for safeguarding ecosystem services across affected catchments. Addressing this research gap is essential to evaluate the impacts on stream water quality and to evaluate ways to mitigate the effects of land use conversion on tropical aquatic ecosystems.

Our study aimed to assess the impact of different land uses on stream water quality and macroinvertebrate communities in an area undergoing dramatic expansion of oil palm plantations in Guatemala. We evaluated water quality through physicochemical parameters, as direct measurements for assessing water quality year-round (Tyagi et al. 2013), and freshwater macroinvertebrates, which are used as bioindicators for habitat and water quality (Eriksen et al. 2021), playing crucial roles in several ecological processes (Ceneviva-Bastos et al. 2017). Specifically, we studied 19 streams draining catchments with forest, oil palm plantation, pastureland, and cropland land use within the Northern Humid Lowlands located in the northeast of Guatemala. In each stream, we measured and analyzed physicochemical water quality parameters in both rainy and dry seasons and sampled benthic macroinvertebrates during the dry season. We tested the following hypotheses: (i) water quality differs significantly among land uses; (ii) decreases in tropical forest cover led to lower water-quality scores; and (iii) land use types have effects on aquatic macroinvertebrates (in terms of taxon richness, abundance, and diversity) as well as on key physicochemical metrics.

Materials and methods

Study area

We conducted our study in the Northern Humid Lowlands (below 1000 m altitude) of Guatemala (Mendez-Paiz and Serech-Van Haute 2018), between 15°44’ and 16°45’ N latitude and 89°31’ and 90°32’ W longitude, enclosing a large area including the so-called Northern Transverse Strip (FTN) and the southern part of Petén province (Fig. 1). The total area of the study extent is approximately 80 km^2, with sampling sites ranging from 3 km to as far as 167 km apart. The study area is one of the regions with the highest rainfall in Mesoamerica where the average annual precipitation can reach more than 2500 mm, the annual mean temperature is 25 °C, mean air humidity of 91% and present two predominating seasons (Fig. 2): dry (Nov–April) and rainy (May–October) (Camacho-Valdez et al. 2022). The study catchments are part of nine sub-basins and 20 micro-basins that comprise the four main watersheds: Chixoy, La Pasion, and Xaclbal, which drain to the Gulf of Mexico, and Sarstún, which drains to the Caribbean Sea. At the national level, these catchments and sub-catchments have the highest water availability, with an average availability of 2000 million m^3

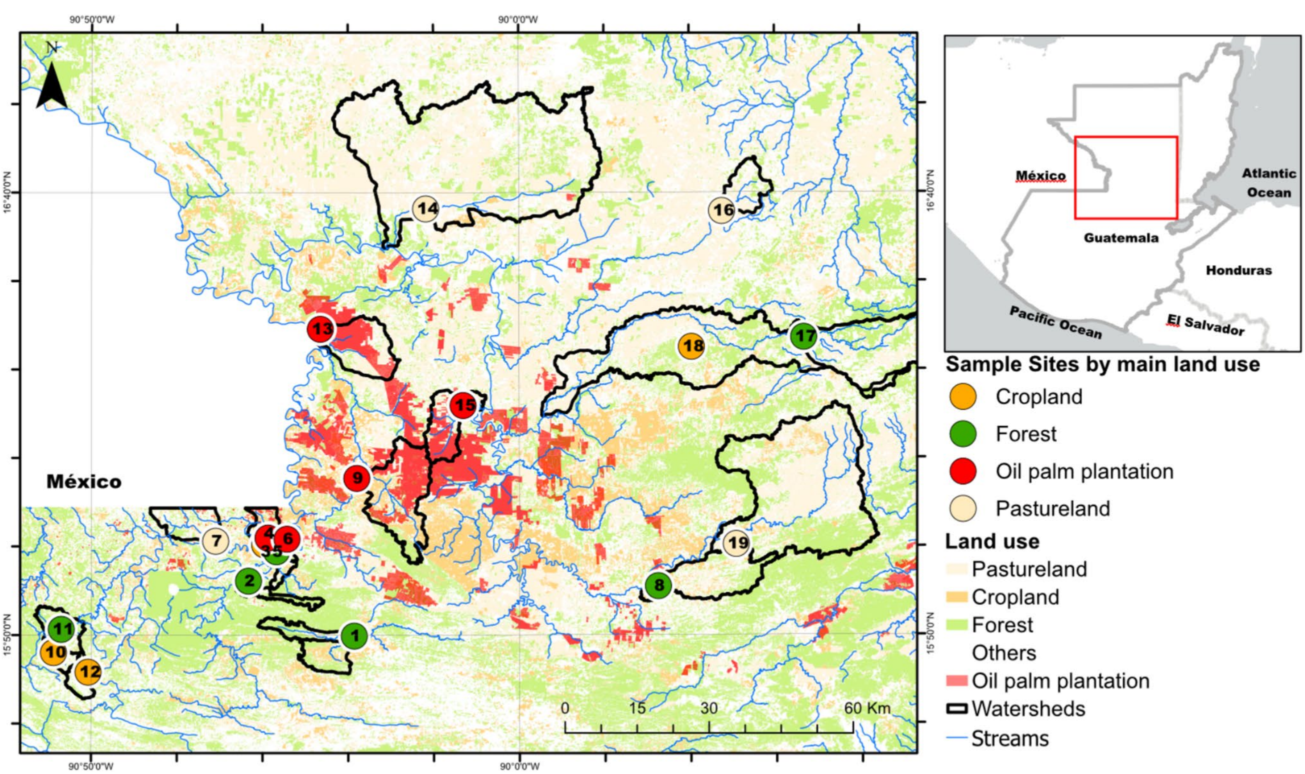

Fig. 1 The selected study area, catchments, and sampling sites in Guatemala (right). Different colors represent different land use categories, and colored circles indicate sampling sites that match the corresponding land use category within the catchment. Delineated areas are the different study catchments

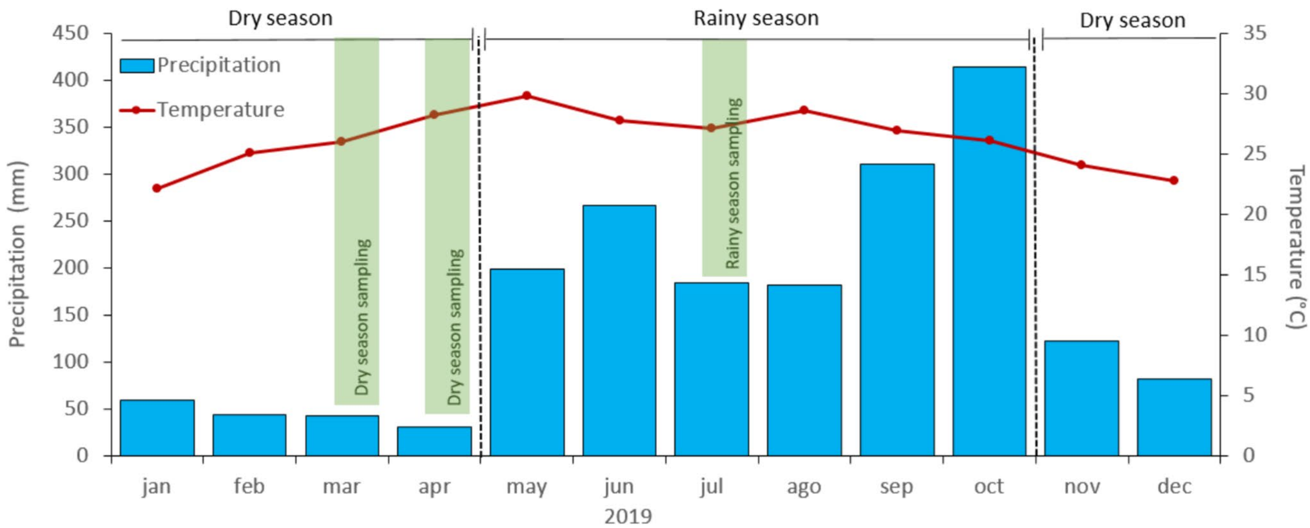

Fig. 2 Monthly mean temperature and monthly rainfall for the study area (2019). Rainfall and temperature data are derived from CHIRPS (Funk et al. 2015) and ERA5 (Copernicus Climate Change Service (C3S), 2017), respectively

year^{-1}, which exceeds the national average of 1780 million m^3 year^{-1} (Carrera & Mosquera, 2023). Therefore, the study catchments were selected for their noted water availability and growing incidence of land use changes (Quezada et al. 2014).

A total of 637,500 people inhabit the region, with 46% living in urban areas and 54% living in rural areas. Most of the population is Maya (75%), with the Q'eqchi' ethnic group being the predominant one. The original vegetation in the area is tropical rainforest; however, between 1962 and 2011, at least 55% of the forest was replaced by human settlements, roads, annual crops, and the establishment of cattle pastures (Quezada et al. 2014). Specifically, between 2003 and 2010, approximately 67,500 ha were converted to oil palm plantations, and between 2010 and 2019, an additional 88,000 ha were converted in Guatemala (Hervas 2021), resulting in the country holding 59% of the total area cultivated with oil palm (Rojas-Castillo et al. 2023). Most recently, oil palm plantations have been established in the area on previously used basic grain farmland, fallow land, and tropical forest (Hervas 2021).

Land use categories and sampling sites

To identify land use categories in the study area, we used the Guatemalan National Land Use Map for 2012 and afterwards updated the land use classification for each sub-catchment with the 2020 National Land Use Map from the Ministry of Agriculture (MAGA 2021), by extracting the land use in all catchment areas using ArcGIS 10 software. We found that the five main land uses were shrubby vegetation (31%), oil palm plantation (28%), pastureland (15%), annual cropland (15%), and forests (5%). Shrubby vegetation, although with a high percentage coverage, is highly sparse in the area; hence, we decided not to consider this land use. Here, forest classification includes broadleaf, coniferous, and mixed forests; cropland includes basic grains (corn and beans) and other vegetables (potatoes, onions, cabbage, and tomatoes); and pastureland is considered areas for livestock.

Guatemala's northern lowlands comprise a land use mosaic in which the four main land types analyzed in this study are intricately interwoven, making it challenging to find or physically reach a stream outflow draining a catchment with only one targeted land use. Therefore, we calculated the area coverage (%) of this main land use for all sub-catchments in the study area and used hierarchical cluster analysis (HCA) and k-means to identify the land use similarities of the sampling sites (Gorgoglione et al. 2020; Itoh et al. 2023) using the R-package vegan (Oksanen et al. 2019) with standardized parameters. This way, we could cluster sub-catchments with similar land use characteristics along a land use gradient between oil palm plantations and forests. Afterwards, at least four replicated headwater streams (1st–3rd order, using Strahler's stream order method) were chosen draining different sub-catchments within each land use category (Chellaiah and Yule 2018a): forest (6 sampling sites), oil palm plantation (5 sampling sites), cropland (4 sampling sites), and pastureland (4 sampling sites), with a total of 19 sampling sites (Supplementary Table S1). We selected sampling sites at the outlet of each sub-catchment in every catchment. Moreover, when choosing the streams and sampling sites, we considered community consent and accessibility to the sampling site. Furthermore, to minimize the impact of other confounding environmental variables, such as climate, altitude, and soil type, our study was conducted in comparable streams located close to each other, with similar physicochemical characteristics and host communities (Rojas-Castillo et al. 2023). Finally, to best describe the effects of each land use category, we only chose streams that did not drain clustered urban areas.

Stream sampling

Water physicochemical parameters and macroinvertebrate samples were collected in all 19 sampling sites. Physicochemical samples were taken in two sampling occasions, during dry season (i.e., second half of March and April, respectively) and rainy season (i.e., second half of July) of 2019, while macroinvertebrates were collected in the two sampling occasions only during dry season (Fig. 2) due to high water levels in rainy season (i.e., high-flow periods). The classification of rainy and dry seasons was based on daily rainfall data extracted from the global climate datasets from the Climate Hazards Group Infrared Precipitation with Station data (CHIRPS) (Funk et al. 2015). In total, we collected 57 samples for water physicochemical analysis and 38 samples for aquatic macroinvertebrate analysis. For macroinvertebrates, we conducted kick-sampling of 100 m reaches in each stream using a D-shaped net (250 µm) to collect organic and inorganic material, including attached macroinvertebrates, for a total collection time of 10 min, accounting for the different microhabitats at each site. Samples were preserved in 70% ethanol and transported to the laboratory, where separation, sorting, and individual identification to the lowest possible taxonomic level (i.e., family) were done. Regional guides (Roldán, 1996; Springer et al. 2010) supplemented by the North American Guide (Merritt et al. 2008) were used to identify macroinvertebrates. Macroinvertebrate community metrics in terms of total macroinvertebrate abundance, taxonomic richness and diversity, and contribution of mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera) (EPT) to total abundance (i.e., %EPT relative abundance) were calculated.

At each sampling site, dissolved oxygen (DO), pH, temperature (temp), electrical conductivity (EC), and total dissolved solids (TDS) were measured. In addition, we collected two water samples from each stream in acid-washed high-density polyethylene (HDPE) 125 ml bottles, fully inverting and submerging them to a depth of 0.3 m below the water surface (USEPA 1996). Immediately after the sample collection, one set was filtered (0.45 µm Millipore) on site. Within 24 h, all samples were analyzed for chemical oxygen demand (COD), biological oxygen demand (BOD), and turbidity (turb) in a mobile laboratory. Afterwards, all samples were transported frozen (–4 °C) to the laboratory and analyzed within 7 days. All samples were analyzed for total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP). Collection of samples, their preservation, and physicochemical analyses were performed following the methodology recommended by Standard Methods for the Examination of Water and Wastewater (APHA 2017). Specifically, COD was performed following the closed reflux, colorimetric method (Standard Methods Committee 2017a), TSS following the filtration method (Standard Methods Committee 2017b), TP following the acid digestion–ascorbic acid method (Standard Methods Committee 2017d), TN following the Valderrama method (Valderrama 1981), and turbidity following the nephelometric method (Standard Methods Committee 2017c). Finally, all samples were processed at Rafael Landívar University's Environmental Analytical Laboratory (LAA).

Water quality indices

BMWP-CR index

The Biological Monitoring Working Party (BMWP) for Costa Rica (CR) is a quick and easy method to evaluate water quality using aquatic macroinvertebrates as bioindicators, adapted to the macroinvertebrate families present and specific to the environmental characteristics of the region's ecosystems. The BMWP-CR is a modification of the BMWP index developed in England in 1970 and subsequently adapted for Colombia (Roldán, 2003) and Costa Rica (Stein et al. 2008). This index classifies each sampling site into different levels of water quality on the basis of an identification of organisms to the family level. Each family is assigned a sensitivity value reflecting its tolerance to pollution, based on knowledge of distribution and abundance (Roldán, 2003), ranging from 1 (for tolerant species) to 10 (for sensitive species). The values for each family are summed up in an index score, regardless of abundance and generic diversity (Stein et al. 2008). Afterwards, the index assigns each site to one of six categories on the basis of the index score: Excellent water quality (> 120), good water quality (101–120), regular water quality with some contamination (61- 100), bad water quality (36–60), bad water quality with a high level of contamination (16–35), and very bad water quality (< 15) (Gutiérrez-Fonseca and Lorion 2014).

WQI_{simp}

A water quality index (WQI) is a valuable and unique rating that summarizes overall water quality status by aggregating data, enabling the conversion of extensive water quality data into a single value (Uddin et al. 2021). In this study, we used the Simplified Water Quality Index (WQI_{simp}), which shows trends similar to those of more complex indices but at lower analytical costs (De Bustamante 1989; Losada Benavides et al. 2020). In WQI_{simp}, all parameters have equal importance; therefore, no weighting factor is applied. The index is calculated as $WQI_{simp} = T\,(A + B + C + D)$, where, after normalization (Table 1), T is a function of the river water temperature, measured in degrees Celsius. A is a function of oxidizability and corresponds to the oxygen consumed in oxidation with MnO_4K at boiling point in an acidic medium (i.e., COD). It varies from 0 to 30. B is a function of suspended matter that can be separated by filtration (i.e., TSS); this parameter varies between 0 and 25. C is a function of oxygen dissolved in water and varies from 0 to 25; D is a function of electrical conductivity at 18 °C and varies between 0 and 20 (De Bustamante 1989; Martínez-Graña et al. 2014). Finally, each water sample can be attributed to five different categories of water quality according to the WQI_{simp} value obtained: very bad (0–30), bad (30–45), medium (45–60), good (60–85), and excellent (85–100) (Sánchez et al. 2007).

Statistical analysis

We used a linear mixed-effects model (LMM) to analyze the differences in invertebrate community metrics, including total abundance, taxonomic richness, and diversity (i.e., Shannon diversity Index), BMWP-CR and WQIsimp scores, between the different land use categories. The analysis was performed using the lme model from the R-package lme4 (Bates et al. 2021). The LMM provided a parametric approach to explain variability in the response variables using fixed effects (factors included in the study design) and random effects, which accounted for factors not part of the study design but that could affect water quality index variability across land use groups. The fixed effects considered in this study were land use (i.e., forest, cropland, pastureland, and oil palm plantation) and sampling time (i.e., month). The random effects included were catchment ID. LMM models were fitted using the restricted maximum likelihood (REML) method in the lme function from the nlme package (Pinheiro et al., 2022), and the model structure selection was based on the lowest AIC (Akaike's Information Criterion). We evaluated model performance using a Type III analysis of variance with the Kenward–Roger approximation. F-statistics were interpreted as the ratio of explained to unexplained variance, and effects were considered statistically significant at $p < 0.05$. Finally, we evaluated the assumption of a Gaussian distribution of errors by inspecting residuals and quantile distributions. Differences among treatment means were determined using Tukey's HSD post hoc tests with the emmeans function from the emmeans R-package (Lenth et al. 2023). In addition, individual physicochemical variables (i.e., all physicochemical parameters) could not be transformed to meet the normality assumption; thus, these variables were instead analyzed with a Kruskal–Wallis nonparametric rank sum test and using Fisher's least significant difference (LSD) for the post hoc nonparametric test in the agricolae package in R (Mendiburu and Muhammad 2020).

We used ordinary least-squares regression to test the linear relationships between the percentage of forest present in each catchment and the BMWP-CR scores, as well as the relationship between the two water quality index scores. Furthermore, the relationship between the land use variable and water quality parameters was analyzed using redundancy analysis (RDA) from the R-package vegan (Oksanen et al. 2019). The RDA enabled us to simultaneously examine the influence of land use variables on all water-quality parameters (Ding et al. 2015). An analysis of variance test was used to assess significance. The land use catchment characteristics used in the RDA are shown in Supplementary Table S5. Finally, all figures in this study were produced using the R package ggplot2 (Wickham, 2016), and all statistical analyses were conducted using the freely available software R (R Core Team 2022).

Results

Grouping of sampling sites according to land use

According to the cluster analysis (Fig. 3), the 19 sampling sites were grouped into four clusters on the basis of land use percentages in the watershed. Each group was classified into one of the main land use categories on the basis of the group's higher average land use coverage. The highest number of sites was classified under "forested" group, with six sampling sites, followed by the

Table 1 Parameters considered for WQI_{simp} calculations

Parameter	Normalization function
Temperature (T)	If $T < 20$; $T = 1$; If $T > 20$; $T = 1 - (T–20) \times 0.0125$
COD (A)	If COD < 10; $A = 30 -$ COD; If 10 < COD < 60; $A = 21 - (0.35 \times$ COD); If COD > 60; $A = 0$
TSS (B)	If TSS < 100; $B = 25 - (0.15 \times$ TSS); If 250 < TSS > 100; $B = 16.67 - 0.0667$; If $B > 250$; $B = 0$
Dissolved oxygen (C)	If $O_2 < 10$; $C = 2.5 \times O_2$; If $O_2 > 10$; $C = 25$
Conductivity (E)	If conductivity < 4000; $D = (3.6 -$ log conductivity) × 15.4; If > 4000; $D = 0$

Values in milligrams per liter (mg L^{-1}), temperatures in centigrade, turbidity in NTU, and conductivity in micro siemens per centimeter (µS cm^{-1})

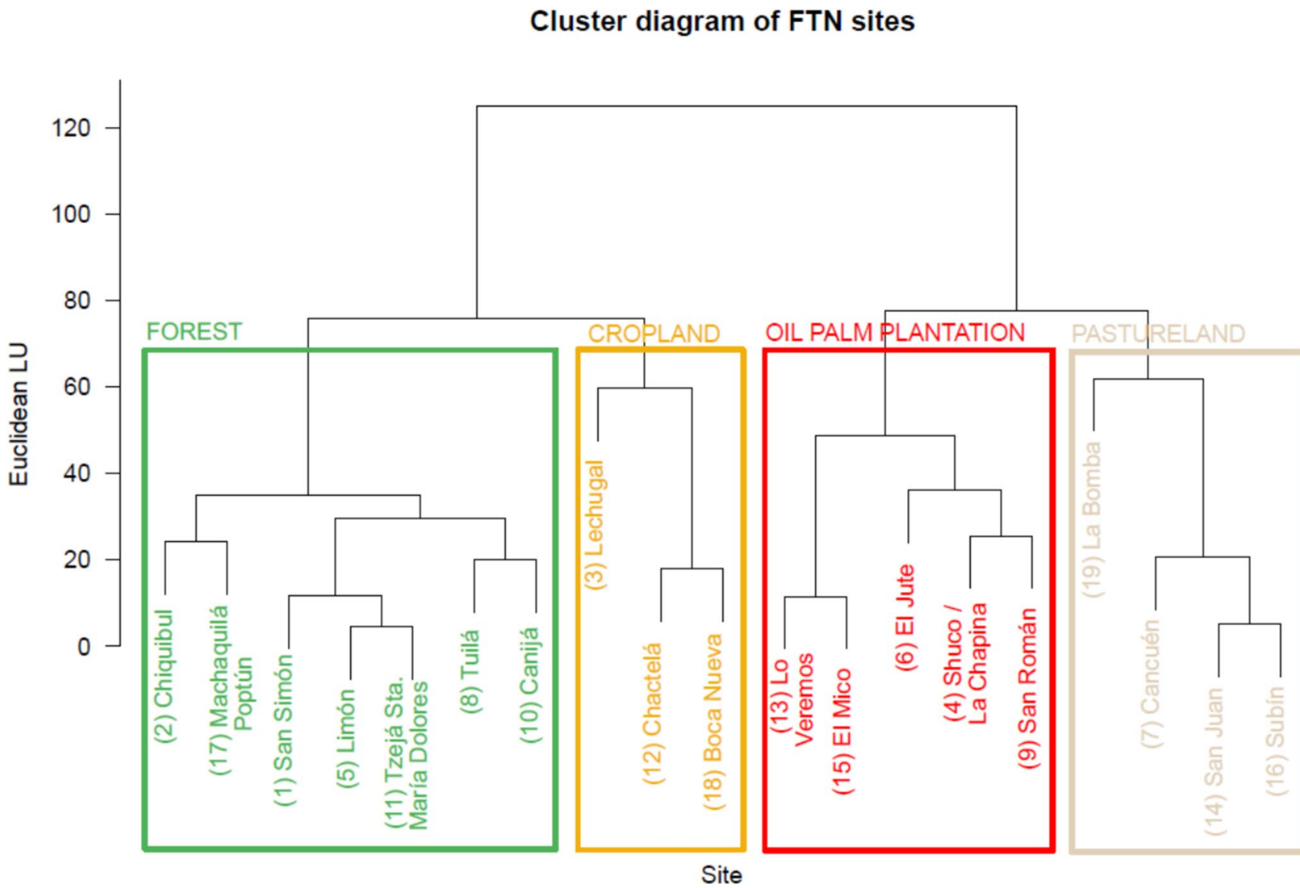

Fig. 3 Dendrogram based on the hierarchical clustering analysis according to the Ward linkage method using Euclidean distance, showing the different clusters from the 19 sampling sites

"oil palm plantation" group, with five sampling sites, and the "pastureland" group, with four sampling sites. In the forest category, sampling site 10 (i.e., Canijá) was grouped together with the cropland group, because the stream flows mainly through annual cropland fields; hence, the "cropland" group included four sampling sites. The watersheds classified as forest presented an average of 51% forest coverage, while watersheds in the pastureland category had 40% pastureland coverage. Watersheds categorized as cropland and oil palm plantation had 30% and 32% coverage, respectively. Within each land use group, the highest land use cover was 64% for forest, 46% for pastureland, 42% for oil palm plantations, and 34% for cropland. The watersheds classified within the cropland category were smaller in extent, while those classified under the pastureland category had the greatest extent, averaging 56 km^2 and 572 km^2, respectively. Finally, although the catchment was classified into a specific land use group, it is important to mention that all catchments included other land uses, including urban areas, shrubby vegetation, and others (Supplementary Table S1).

Land use effect on stream environmental quality using aquatic macroinvertebrates

A total of 3349 invertebrates collected from the 19 streams belonging to 55 families were identified (Supplementary Table S2). The order Diptera (22%), followed by Ephemeroptera (21%) and Hemiptera (18%), numerically dominated the macroinvertebrate assemblage in the study area (Fig. 4). Specifically, the oil palm plantation sites exhibited the highest abundance of *Chironomidae* (abundance across sites = 5–563; 1–96%), followed by pastureland sites (abundance across sites = 4–19; 2–12%), in total constituting 22% of the total macroinvertebrate assemblage. *Veliidae* dominated in two forested sites (abundance across sites = 136–232), comprising 15% of the total assemblage. Finally, the third most common family was Leptophlebiidae of the order Ephemeroptera (8%), which dominated across the cropland catchments (abundance across sites: 9–80; 2–14%) and constituted 8% of the macroinvertebrate assemblage. Forested sites had a total of 19 families, and cropland sites had 14 families recorded, while pastureland

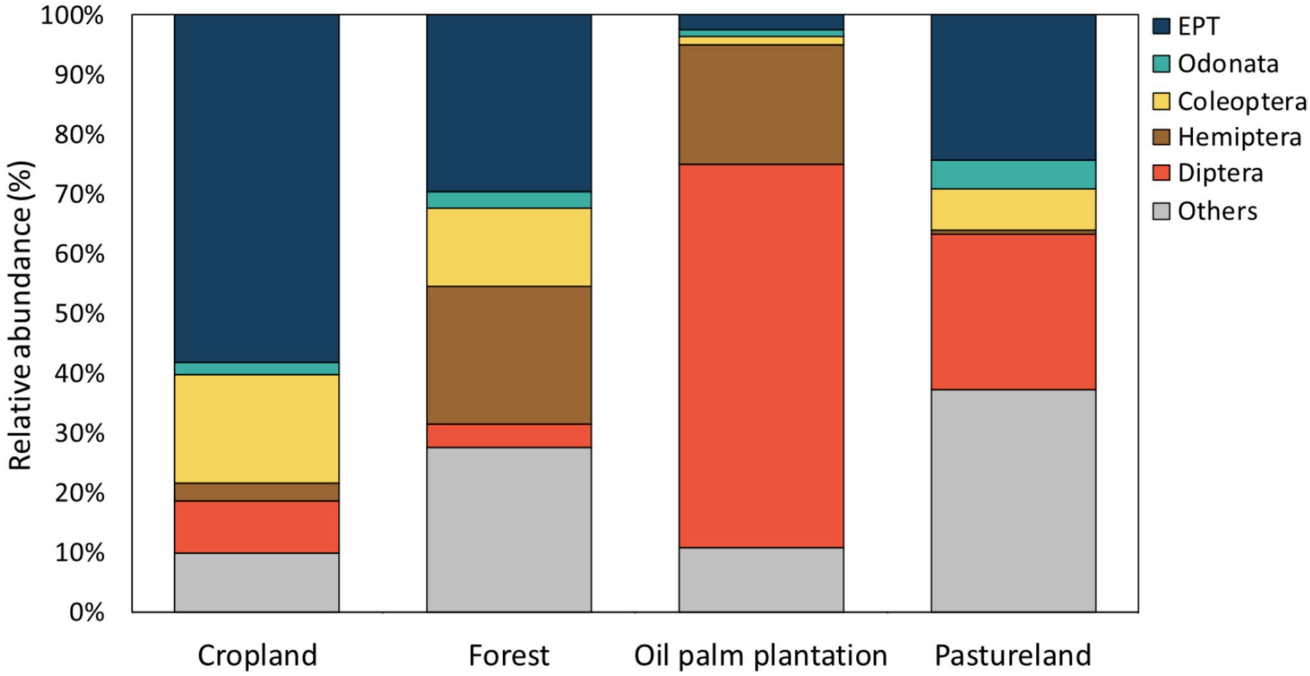

Fig. 4 Macroinvertebrate metrics of relative taxon abundance across different land use categories

and oil palm plantation sites had 11 and 10 orders recorded, respectively. Orders Ephemeroptera (26%), Diptera (19%), Hemiptera (16%), and Coleoptera (10%) dominated at all land use categories. Diptera dominated oil palm plantation sites (64%) and pastureland (26%) in terms of relative abundance, while Ephemeroptera + Plecoptera + Trichoptera (EPT) dominated cropland (58%) and forest sites (30%) in terms of relative abundance (Fig. 4).

Macroinvertebrate assemblage structure differed between land use categories (Fig. 5, Supplementary Table S3 and Table S4) and across sites, with organism abundance varying from 8 to 587 (Fig. 5a). Forested sites showed the highest mean abundance (187; range 84–278), followed by oil palm plantations (132; range 8–587), the latter influenced by the high abundance of Chironomidae (587) present in catchment 16 (Supplementary Table S1). Pastureland sites had the lowest mean abundance (32; range 15–64). Despite the marked difference in abundance between land use categories, we did not find the difference significant ($p > 0.05$; Supplementary Table S3a). Taxa richness across sites ranged from 4 to 22 (Fig. 5b) and was significantly higher in forested sites compared with oil palm plantations and pastureland ($p < 0.05$; Supplementary Table S4a). Mean richness decreased in the following order: forested (mean 16; 9–22), cropland (mean 11; 8–15), pastureland (mean 7; 5–10), and oil palm plantations (mean 7; 4–12). Diversity differed significantly only between forested and oil palm plantation sites ($p < 0.05$; Supplementary Table S4b; Fig. 5c), with forested sites having the highest mean diversity (1.95) and oil palm plantations the lowest (1.20), including the minimum value (0.22) observed in catchment 16.

The BMWP-CR index showed a significant difference ($p < 0.05$; Supplementary Table S3 and Supplementary Table S4) in water quality across the different land uses. Specifically oil palm and pastureland scored the lowest, revealing the poorest water quality, while, forest land use scored the highest, exhibiting the best water quality among the different land uses (Fig. 6a). Forested catchments had the highest BMWP-CR scores (mean 74; range 44–112), spanning water quality classifications from polluted to excellent. Oil palm catchments showed much lower scores (mean 30; range 14–49), corresponding to highly polluted to poor-quality water, and pastureland catchments likewise presented low scores (17–73). Cropland catchments had intermediate values (mean 52; range 40–66), but their scores did not differ significantly from any other land use category. We found a significant positive correlation ($p < 0.05$) between forest cover and BMWP-CR scores, indicating better water quality with increasing forest percentage (Fig. 6b). Notably, the highest BMWP-CR score occurred at the Machaquilá Poptún site, despite its forest cover being only 41%.

Variation in water quality among land use groups using physicochemical parameters

To help interpret the physicochemical data, WQI_{simp} scores for dry and rainy seasons per land use group are shown (Fig. 7). Our study showed that the water quality in the

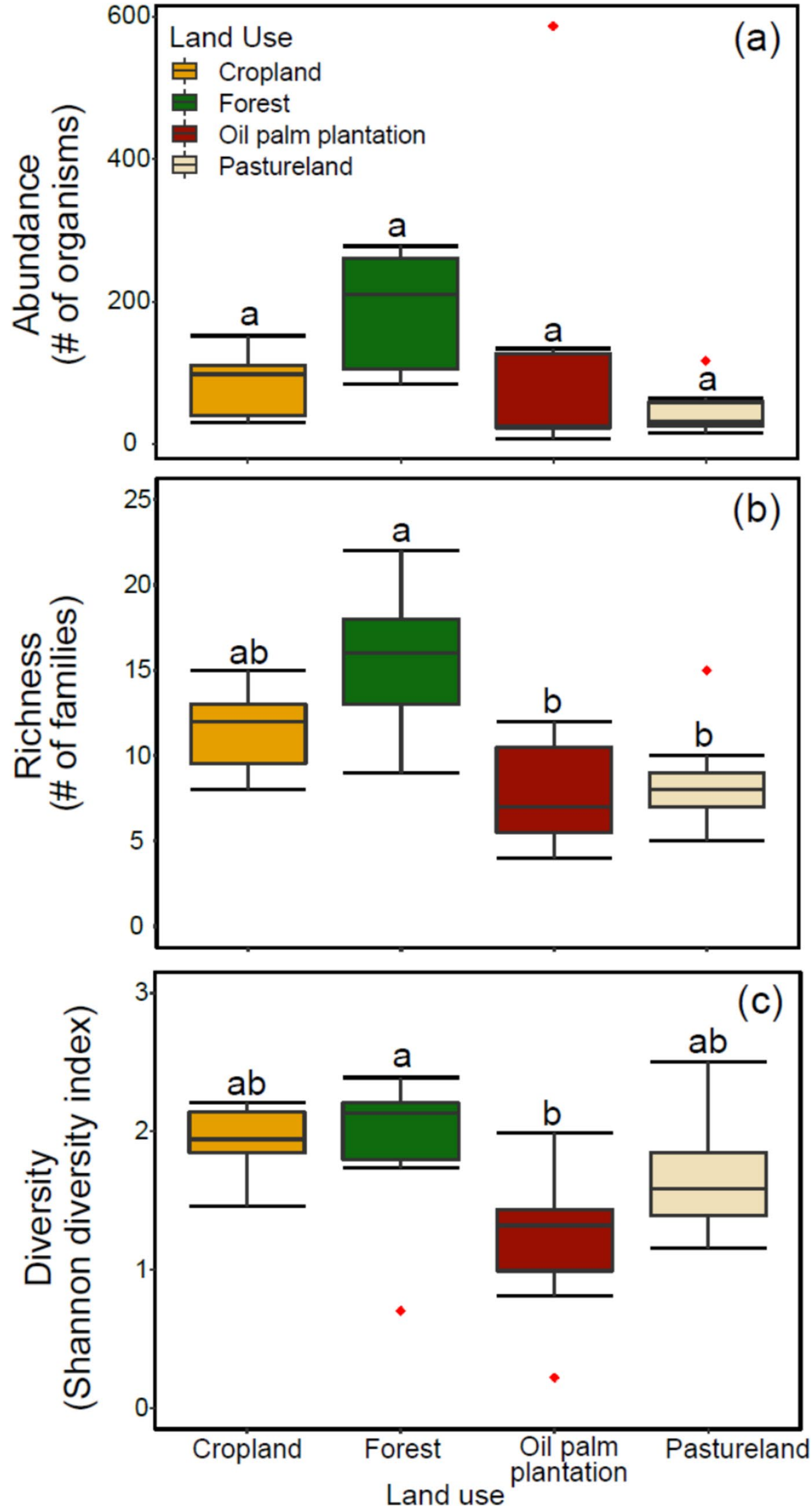

Fig. 5 Assemblage structure indicator including **a** abundance of organisms, **b** richness of families and **c** diversity of the Shannon diversity index. Any two values sharing a common lower-case letter are not significantly different ($\alpha = 0.05$) between land uses (LMM). Solid line in box plots is the median value, and box extents are the interquartile range (IQR)

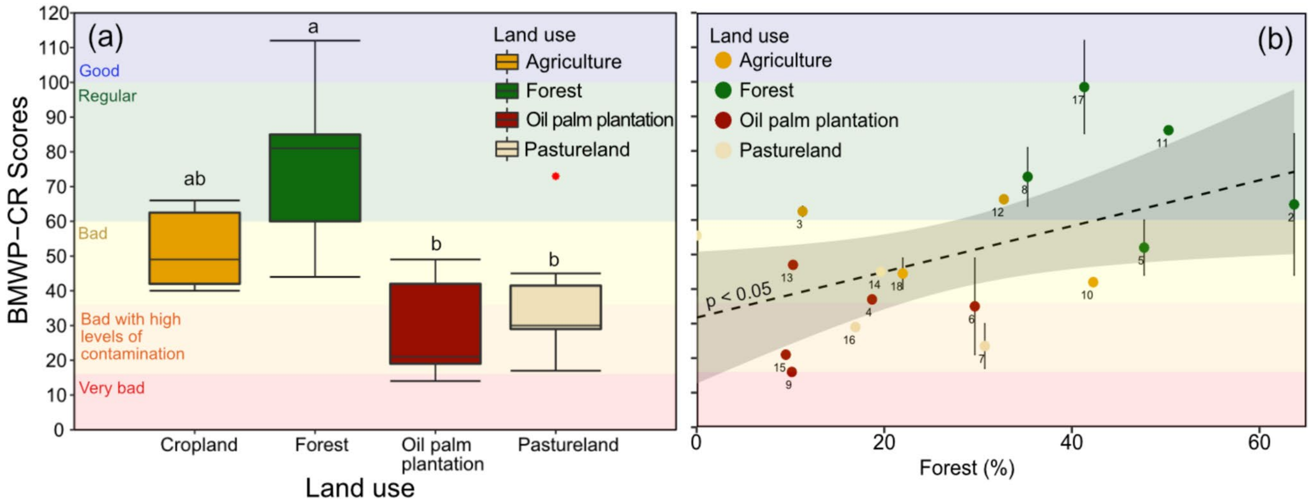

Fig. 6 Water quality of the different land uses using the BMWP-CR index (**a**) BMWP-CR scores for different catchment land uses. Any two values sharing a common lower-case letter are not significantly different ($\alpha = 0.05$) in land use BMWP-CR scores (LMM). Solid line in box plots is the median value, box extents are the interquartile range (IQR), red asterisks indicate outliers, and colors indicate the different land uses. **b** Linear correlation between BMWP-CR scores and forest coverage (%) in the catchments of the study area. Black dotted line is the significant linear regression with ± 1SE (in light black). Colored circles represent the different land uses analyzed in the study area with ± 1SE between the two sampling occasions during dry season, and numbers indicate site ID (D = dry season, R = rainy season). Background colors delimitate BMWP-CR ecological water quality levels

outflow from the oil palm plantation catchments ranked the lowest among all the land use categories, regardless of whether it was the rainy or dry season; however, we did not find a statistical difference ($p > 0.05$) between any land use WQI_{simp} scores (Fig. 7a and b). Specifically, streams discharging from oil palm plantation catchments scored on average 51 and 69 in dry and rainy seasons, respectively. Conversely, during the dry season, the cropland catchments showed on average the highest score (63), followed by forested and pastureland catchments (60). In contrast, during the rainy season, the forested catchments showed on average the highest score (79), followed by cropland and pastureland catchments (76). Overall, the lowest score was 44 (i.e., catchment 6; El Jute) within the oil palm plantation category during the dry season, while the highest score was 89 (i.e., catchment 18; Boca Nueva) within the cropland catchments during the rainy season. The rainy season showed better water quality across all streams, with an average overall score of 75 compared with 59 during the dry season. We did not find a significant correlation ($p > 0.05$) between the two water quality index scores, as WQI_{simp} values did not vary consistently with BMWP-CR scores, and some sites showed different water quality levels depending on the index used. For example, catchment 7 (i.e., Cancuen) within the pastureland land use category scored 78 using the WQI_{simp} index, indicating good water quality, while using the BMWP-CR, the site scored 17, classified as having bad water quality with high levels of contamination.

Although we did not find a significant difference between land uses using the WQI_{simp} scores, we did find significant differences ($p < 0.05$) between land uses for individual water quality parameters. During rainy season, DO, pH, and turbidity differed among land uses, and TP differed in both seasons (Supplementary Fig. S1). Oil palm plantation and pastureland catchments had the lowest rainy season DO levels (5 and 6 mg L^{-1}, respectively), while cropland and forested catchments had the highest (8 and 7 mg L^{-1}, respectively; Supplementary Fig. S1c). Turbidity peaked in the rainy season in the streams from the catchments with cropland (53 mg L^{-1}) and oil palm plantation (42 mg L^{-1}), with significantly higher turbidity in oil-palm streams than in forested or pastureland streams (Supplementary Fig. S1h). Oil palm catchments also had significantly higher TP during the rainy season (0.10 mg L^{-1}) compared with forested (0.04 mg L^{-1}) and pastureland (0.04 mg L^{-1}) catchments, and higher TP in the dry season (0.07 mg L^{-1}) compared with pastureland (0.02 mg L^{-1}; Supplementary Fig. S1f). Finally, all physicochemical parameters except EC and DO differ significantly between seasons (Supplementary Fig. S1).

Relationship between land use and water quality

We performed an RDA to assess whether land use is associated with physicochemical variables across watershed categories. The RDA results showed that all the canonical axes significantly ($p < 0.05$) accounted for 34% of the water quality variations in the dry season and 36%

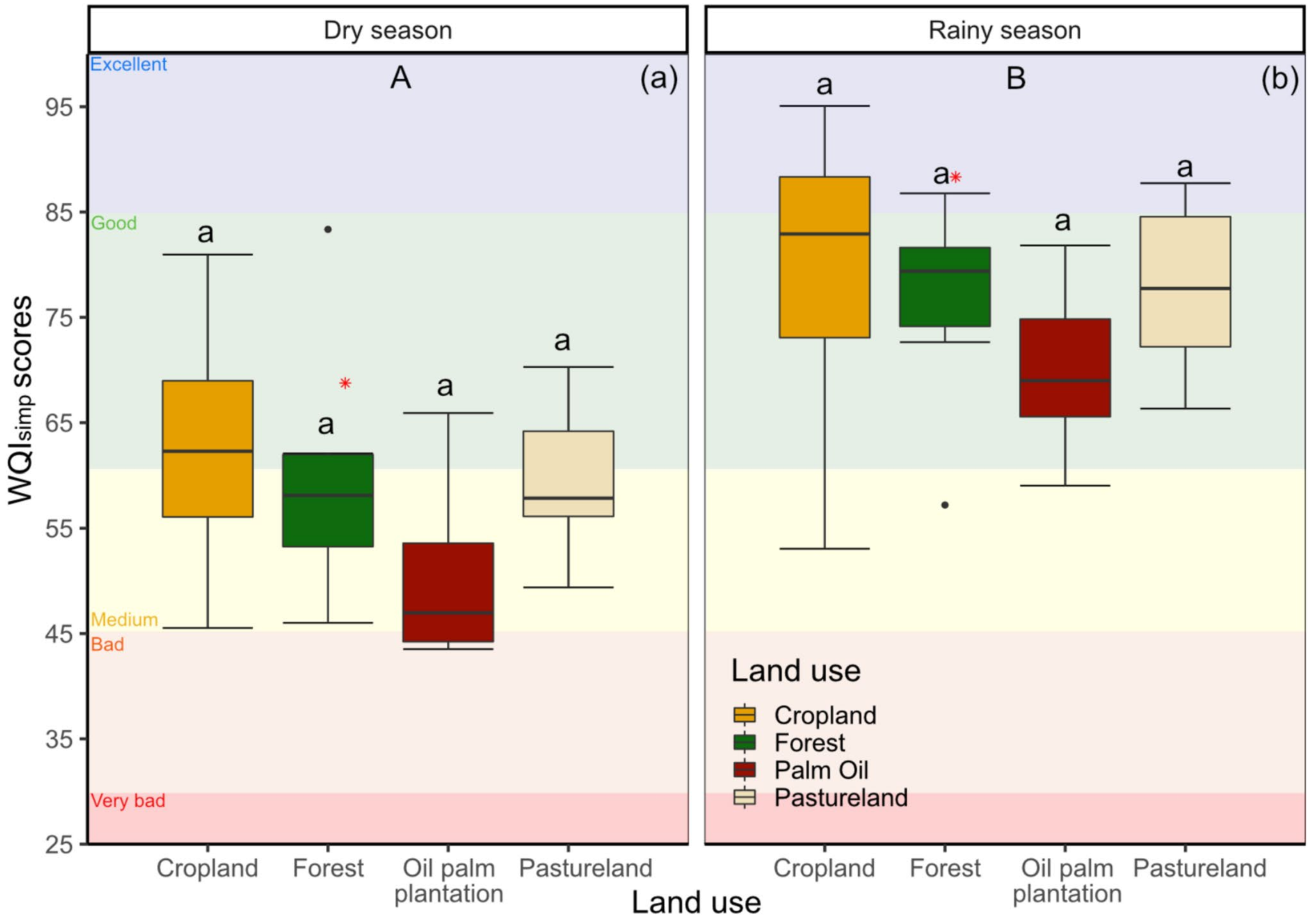

Fig. 7 Water quality of the different land uses using the WQI_{simp} index **a** WQI_{simp} scores for different catchment land uses during rainy season and **b** during dry season. Any two values sharing a common lower-case/uppercase letter are not significantly different ($\alpha = 0.05$) between WQI_{simp} scores (Kruskal–Wallis), lower-case letters show difference between land uses, and uppercase letters show difference between seasons. Solid line in box plots indicates the median value, box indicates interquartile range (IQR), red asterisks indicate outliers, and colors indicate the different land uses. Background colors delineate WQI_{simp} levels

($p < 0.05$) in the rainy season. The first axis significantly explained 25% ($p < 0.05$) of the variation during rainy season, positively correlated with the proportion of oil palm (0.88) and pastureland (0.64), and negatively correlated with the proportion of forest (−0.67) and cropland (–0.60) (Supplementary Table S5; Fig. 8a). During dry season, axis one significantly explained 20% ($p < 0.05$) of the variation with strong positive correlation with the proportion of oil palm plantation (0.96) and pastureland (0.23), while negatively correlated with the proportion of forest (−0.34) and cropland (−0.27) (Supplementary Table S5; Fig. 8b). Moreover, axis one negative variation is driven by TN (−0.37) during dry season and by pH (−0.78), DO (−0.58), and EC (−0.33) during rainy season, while positive variation is driven by TP (0.84) and BOD_5 (0.78) during dry season and TN (0.82) and BOD_5 (0.72), during rainy season (Fig. 8a). Finally, the second axis accounted for 9% and 7% during dry and rainy seasons, respectively; however, nonsignificant ($p > 0.05$) and was mainly determined by cropland land use.

Discussion

How are different land uses affecting stream water quality?

Our results support previous findings that oil palm plantations have negative effects on stream water quality, and the reduction of forest cover negatively affects water quality (Mello et al. 2018), both using physicochemical indices (Itoh et al. 2023) and macroinvertebrate communities as bioindicators (Cabrera et al. 2023). Both indices showed the lowest scores for oil palm plantation land use, while the highest scores were shown for the forested catchments; however, the WQI_{simp} results were not statistically different.

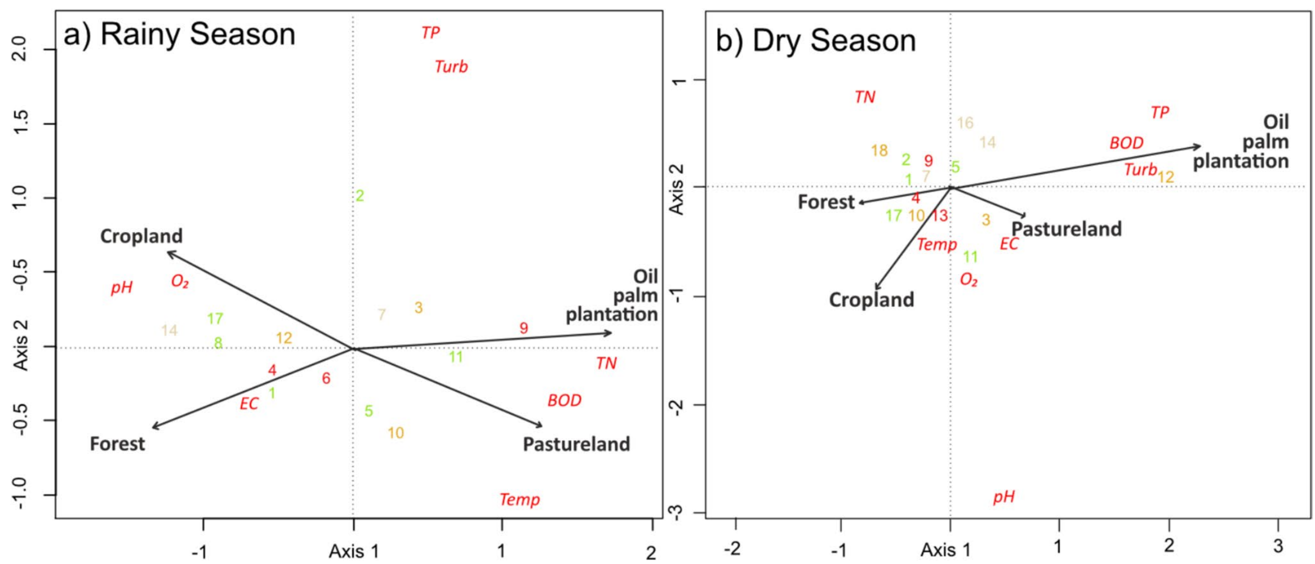

Fig. 8 Tri-plot of redundancy analysis: water quality parameters and land use variables in **a** the dry season and **b** the rainy season. Numbers represent different sites, and different colors represent different land use categories that match the corresponding land use category within the catchment

Overall, anthropogenic disturbances such as land use activities related to extensive agriculture (i.e., oil palm plantation and pastureland), including deforestation, agricultural practices, unpaved road construction, and settlements have been found to influence the quantity of runoff, organic matter, nutrients and sediments that enter recipient water bodies (Lubanga et al. 2021; Masese et al. 2017; Wantzen 2006). Specifically, oil palm plantations are often associated with decreased water quality owing to soil erosion, the use of agrochemicals (e.g., commercial fertilizers, herbicides, insecticides, and fungicides), the removal of native riparian vegetation, alterations to streambank and bed dynamics, as well as changes in detrital input (Luke et al. 2017; Mercer et al. 2014; Rojas-Castillo et al. 2022). Overall, the altered macroinvertebrate composition in streams draining oil palm plantations likely reflects the combined influence of multiple hydrological and physicochemical changes and their interactions within the stream environment (Chappell et al. 2005). For example, in our study, these catchments showed elevated turbidity during the rainy season, probably driven by increased soil erosion associated with limited riparian buffers, ploughing, and dense networks of unpaved roads (Pak et al. 2021; Ziegler et al. 2004). Such erosion is known to reduce sensitive EPT taxa by smothering gills or clogging filtering structures with fine sediments (Wantzen 2006). Yet, erosion levels may vary among oil palm catchments depending on plantation age (Carlson et al. 2014).

Our results showed that TP concentration was 2.5-fold higher in streams draining oil palm plantations than in streams draining forest and pastureland during the rainy season, and during dry season, 3.5-fold higher compared with pastureland. Increase in turbidity and TP may indicate greater levels of soil erosion in these watersheds relative to the forested catchments (Uriarte et al. 2011). Another potential explanation for the increase in stream TP concentration is the theoretical overuse of fertilizer in oil palm plantation areas (Mercer et al. 2014); however, we do not have access to the fertilization practices in our study area. Yet, our results are in line with other studies that have also shown a 2- to 3-fold increase in TP in streams draining oil palm plantations due to the heavy use of fertilizer that flows into waterways (Chellaiah and Yule 2018a; Luke et al. 2017; Mercer et al. 2014; Rojas-Castillo et al. 2024). Although further research is required to identify harmful nutrient levels for macroinvertebrates, higher P concentrations have been associated with declines in sensitive taxa and a reduction in overall macroinvertebrate diversity (Chellaiah and Yule 2018b). Contrary to our expectations, TN concentrations did not differ significantly across land uses. These results appear to be an effect of the efficient utilization of fertilized N in plantation areas with a high potential for oil palm nutrient uptake (Itoh et al. 2023).

We report higher BOD and lower DO in streams draining oil palm plantations during the rainy season, likely driven by increased sediment organic matter content (dos Reis Oliveira et al. 2019), which enhances respiration rates and consequently increases DO consumption (Wantzen et al. 2008). The influence of oil palm plantations on stream organic matter and dissolved oxygen is also evident in the RDA analysis, where BOD is directly related to oil palm plantations, presumably linked to increased organic matter, and DO is inversely related owing to increased DO consumption.

DO levels can help explain responses in macroinvertebrate richness and water quality scores, as low DO levels have been shown to affect macroinvertebrate taxonomic richness negatively (Croijmans et al. 2021). Unexpectedly, our results indicated that forested catchments had lower abundances of EPT-sensitive taxa than croplands during the dry season, likely reflecting combined hydrological and biogeochemical effects on stream DO. In forested catchments, increased evapotranspiration and reduced runoff may decrease physical aeration (Pak et al. 2021), and canopy shading can limit primary production, while organic matter inputs from riparian vegetation can elevate respiration and further deplete DO (Young & Huryn 1999), all of which align with the lower DO levels we recorded during the dry season in streams draining the forested catchments. Specifically, DO levels can potentially influence the abundance of pollution-sensitive EPT taxa, as these organisms rely on DO for respiration via gills or other aquatic respiratory structures (Verberk et al. 2016).

Pastureland showed better water quality than oil palm plantations using the WQI_{simp} index; however, using the BMWP-CR index, pastureland sites showed scores that were almost as low as those of oil palm plantation sites. Here, in addition to the degradation of physical habitat, removal of nutrient-demanding vegetation (e.g. forest) and riparian vegetation acting as buffers, pasturelands could be subject to overgrazing, which could lead to undecomposed livestock excrements and accelerate soil loosening and erosion (Lubanga et al. 2021; Monaghan et al. 2007), resulting in the worsening of water quality. Yet, our results may also be due to the extensive pasture lands in the area, which cover 30–50% of the catchment, exceeding the typical threshold above which streams in agricultural catchments often fail to maintain good conditions (Karlsen et al. 2019; Uriarte et al. 2011). Cropland, which in our study refers to basic grains (corn and beans) and other vegetables (potatoes, onions, cabbage, and tomatoes), exhibited water quality scores for both water quality indices that were either as high as those of forested catchments or above those of pastureland and oil palm plantation sites. These results may be a consequence of the small-scale agricultural practices in this area, where stream water quality is less degraded by nonpoint pollutant inputs, as riparian buffers and stream channel habitats are maintained, and therefore flow is less altered (Allan 2004). Finally, although all catchments were classified into a specific land use group, they all had a percentage of forest cover.

BMWP-CR scores indicated that streams draining forested catchments had the highest water quality. Moreover, scores were positively correlated with forest cover, suggesting that higher forest percentages are associated with higher scores and, consequently, better water quality. Our results are in line with many that have demonstrated that tropical forests provide a better water quality than watersheds with other land uses (dos Reis Oliveira et al. 2025; Mello et al. 2018) and that water quality parameters have a strong positive correlation with the proportion of forest cover (Zhou et al. 2016). Overall, tropical forests, particularly in riparian zones, improve stream water quality by buffering nutrient and sediment inputs (Chua et al. 2019), reducing organic matter (Heartsill Scalley et al. 2012), enhancing soil infiltration (Lozano-Baez et al. 2019), and regulating light and temperature through shading (Goss et al. 2014). These conditions benefit macroinvertebrate metrics such as abundance and richness; for instance, temperature buffering prevents declines in species density (Mazzoni et al. 2023). Forest cover also influences stream substratum by adding leaf litter and wood (Muto et al. 2009), creating complex habitats with more food and shelter (Aguiar et al., 2018), which support higher diversity (Mbaka et al. 2015). Reduced sediment export may increase near-depth dissolved oxygen, enhancing richness and abundance (dos Reis Oliveira et al. 2019). However, as mentioned before, we observed the lowest DO in forested catchments during dry season, likely owing to reduced turbulence and mixing at low discharge and/or high organic content and temperature (Pak et al. 2021).

We used two water quality indices to encompass multiple parameters and assess overall stream water quality variation, since land use effects on stream's environmental quality are not caused by a single factor but rather a combination of water quality (i.e., physicochemical) and their interactions with physical and environmental conditions (Lee et al. 2023). Specifically, the index using macroinvertebrates as bioindicators reflects the combined influence of historical and ecological factors, such as temperature, substrate type, and hydraulic conditions, as well as physicochemical parameters, including organic pollution, nutrients, pH, and dissolved oxygen (Eriksen et al. 2021; Leunda et al. 2009). In contrast, the index using only physicochemical parameters is valuable for summarizing these characteristics in a single term (Tyagi et al. 2013); however, it offers limited insight into freshwater ecosystem deterioration, as it reflects only sampling-time conditions and may miss pollutant events or long-term effects (Assie et al. 2024; Holt and Miller 2011). Thus, the BMWP-CR score likely offers a broader classification of land use impacts on freshwater ecosystems than the WQI_{simp} score, as several influential variables, such as sediments and agrochemicals, were excluded (Carlson et al. 2014). For instance, insecticide use in plantations may affect sensitive macroinvertebrates, with coleopterans and hemipterans often absent where insecticides are applied to control the Asiatic rhinoceros beetle (*Oryctes rhinoceros*), a major oil palm pest (Mercer et al. 2014; Rojas-Castillo et al. 2024). In addition, stream size, soil type, and bedrock could also influence temperature, sediment, and nutrient levels (Carlson et al. 2014). Nevertheless, in both water quality indices, forested catchments had the best water quality, followed by cropland, pastureland, and oil palm plantations, highlighting the negative effects of oil palm plantations and the importance of tropical forests on stream water quality.

How are macroinvertebrates affected by land use?

Different land uses are responsible for modifications in the taxonomic composition of macroinvertebrate assemblages (Rojas-Castillo et al. 2024). Sampling sites located in oil palm plantations had the lowest taxon richness and diversity, whereas those in forested areas showed the highest. Our results are consistent with previous studies in tropical South America, which report low macroinvertebrate richness and abundance in rivers flowing through oil palm plantations (Cabrera et al. 2023; Luiza-Andrade et al. 2017; Mercer et al. 2014). The decrease in macroinvertebrate diversity and richness has been primarily attributed to altered environmental conditions resulting from changes in catchment land use, such as increased temperatures, reduced shading, microhabitat loss, worsened water quality, and reduced food and shelter availability (Chellaiah and Yule 2018b; Comte et al. 2015). The low Shannon diversity index values (median = 1.3) in the oil palm plantation and pastureland sites indicate widespread degradation affecting macroinvertebrate communities, as these sites showed communities with high dominance of a few taxa (mainly Diptera and Hemiptera). Specifically, the Diptera and Hemiptera taxa are not highly affected by human-mediated environmental and ecological changes (Masese and Mcclain 2012). Macroinvertebrate abundance tended to be higher in forested catchments; however, we did not find a statistical difference among land use categories. The absence of an organism's abundance response to land use seems to be a result of species asynchrony, where the density of one species decreases but is compensated by the increase of another (Rojas-Castillo et al. 2024). For example, generalist macroinvertebrates thrive in streams draining oil palm plantations, while forest specialists decrease (Rojas-Castillo et al. 2023).

Several studies have reported a decline in both taxonomic richness and the relative abundance of pollution-sensitive orders, such as EPT, accompanied by an increase in tolerant groups, such as Diptera, in response to rising pollution levels and deteriorating river conditions (Lubanga et al. 2021; Masese et al. 2017; Masese and Mcclain 2012). Our results show similar trends, particularly, organisms in the EPT orders were predominant in forest and cropland sites, while almost absent in oil palm plantation sites, therefore showing a better state of freshwater habitats in streams draining catchments with forest and croplands as main land use compared with oil palm plantation (Eriksen et al. 2021). Specifically, the EPT group is vulnerable to changes in physical habitat, substratum, and water quality (Suhaila et al., 2012), particularly to changes in water temperature and pH, as well as the amount of organic debris and vegetation cover in the streams (Luiza-Andrade et al. 2017). Important to mention that organisms of these orders are prominent among tropical stream biota and are associated with several ecological processes (e.g., litter decomposition and trophic network) (Ceneviva-Bastos et al. 2017). Conversely, the highly contamination-tolerant Chironomidae family, part of the Diptera order, was found in high abundance in oil palm plantation sites, explained by its capacity to tolerate high organic pollution levels, low oxygen concentrations and its multivoltine reproductive characteristics, which allow rapid recolonization from water bodies with less fluctuant environmental conditions (Cabrera et al. 2023; Mercer et al. 2014).

How do physicochemical parameters differ seasonally?

Land use in watersheds influenced all water quality parameters; however, these effects were specific to each parameter and seasonally dependent. For example, the WQI_{simp} index showed better overall water quality, independent of land use, during rainy season. Our results align with those of Zhou et al. (2016), who found that catchment land use had a greater influence on physicochemical water quality during the rainy season, mainly because nonpoint source pollution depends on surface runoff, which predominates during the wet season. Conversely, Rodríguez-Romero et al. (2018) observed significant temporal variation, with the dry-cold season exhibiting the highest water quality. Our results showed that watershed land use influenced stream water temperature with oil palm plantation catchments having higher water temperature, followed by pastureland, cropland, and forested sites. Stream temperature is inversely correlated with water quality scores, with higher scores associated with lower stream temperatures. This pattern reflects the role of riparian vegetation, evapotranspiration, and upland shading in regulating stream thermal regimes (Chellaiah & Yule 2018a; Rojas-Castillo et al. 2024).

The RDA analysis revealed opposite physicochemical characteristics between land use groups: oil palm plantations and pastureland were associated at one extreme of axis 1, and forest and cropland at the other, during both dry and rainy seasons, resembling the behavior observed with the WQI_{simp} and the BMWP-CR. In line with our results, Cabrera et al. (2023) found a decrease in water pH and an increase in temperature, EC, turbidity, and nutrients in agricultural areas compared with natural areas, while we found that during rainy season, oil palm plantations and pasturelands were associated with BOD, TN and water temperature, and forests and croplands were associated with pH and DO. Yet, only oil palm plantations showed significant correlations with parameters of temperature, pH, turbidity, and phosphate. Mello et al. (2018) found that the degraded catchments were characterized by higher values of total suspended solids (TSS) and TP, whereas forested catchments were associated with high values of DO and the difference between groups seems to be attributed to differences in soil management practices, forest cover, and riparian vegetation.

Catchments with land use mosaics as a limitation to our study

Land use changes in the Guatemalan humid lowlands have a complex array of proximate causes and underlying driving forces (Quezada et al. 2014) that have created an area with a land use mosaic intertwined among the four main land uses analyzed in this study. Not even oil palm plantation expansion has reached the whole catchment scale, yet. Therefore, it was almost impossible to find or reach a stream outflow draining a catchment with only one targeted land use, generating sources of uncertainty in our results. However, by using a cluster analysis, we attempted to group the catchment by its main land use, considering that other land uses could also influence water quality. In addition, our study grouped all oil palm plantation stages and practices into a single oil palm plantation group, preventing us from distinguishing the specific characteristics of different stages. For example, it is well established that if riparian buffers are established in oil palm plantations the effect on macroinvertebrate community and physicochemical parameters by land conversion is mitigated (Chellaiah and Yule 2018b; Rojas-Castillo et al. 2023), the difference in age of the oil palm plantation might affect the sediment exports (Carlson et al. 2014) and differences in fertilization protocols might have different effects in water quality (Comte et al. 2015; Itoh et al. 2023). Regardless, the establishment and practices of corporately controlled oil palm plantations in the area have followed very similar patterns followed very similar patterns (Dürr 2017; Hervas 2019), thereby decreasing differences between plantations. Finally, all catchments have sparse human settlements that does not reach the category of urban areas but can still directly or indirectly influence the water quality in the streams. To mitigate this effect, we attempted to minimize the likelihood of detecting an island-wide effect of human settlements on a specific water quality parameter by excluding catchments with highly spatially clustered human settlement distributions (Uriarte et al. 2011).

Implications for conservation and land management

There is a lack of studies on tropical streams, with even fewer assessing the impact of land use change in these systems, despite alarming rates of tropical forest deforestation (Gibbs et al. 2010; Pendrill et al. 2022). Specifically, over the last two decades, Guatemala has been part of the rapid expansion of oil palm plantations in the tropics, which often involves deforestation (Hervas 2021) and potential alterations in water quality, macroinvertebrate communities, and stream ecosystem functioning (Rojas-Castillo et al. 2023). However, despite the need for sustainable land management in areas with higher oil palm plantation expansion, a significant research gap remains in understanding how this land use affects water quality and aquatic communities, with potential implications for stream ecological processes and ecosystem services. Our findings have important implications for land management, aquatic ecosystem conservation, and the protection of water resources for human consumption in tropical regions. We show apparent differences in water quality between forested and oil palm plantation catchments, with oil palm plantations negatively affecting aquatic macroinvertebrates and, consequently, freshwater ecosystem health (Ramírez & Gutiérrez-Fonseca 2014). Macroinvertebrates are food for fish, amphibians, and wildlife and are important contributors to energy and nutrient processing (Suter & Cormier 2014) and, therefore, good bioindicators for habitat and water quality (Eriksen et al. 2021). Considering the well-established link between land use, water quality degradation, and the decline of sensitive aquatic taxa, it is clear that effective freshwater ecosystem conservation requires integrated management of both terrestrial and aquatic systems. Our study emphasizes the importance of protecting remaining forests to conserve stream ecological processes and ecosystem services. Yet, different stream management practices, such as riparian buffers in oil palm plantations and pasturelands, could also be implemented to maintain cool water temperatures, increase canopy cover, increase leaf litter and wood, reduce submerged vegetation and mud, and safeguard freshwater ecosystems (Rojas-Castillo et al. 2023).

Conclusions

Land use change from forest to oil palm plantations is an increasing concern in tropical regions, particularly where conversion is accelerating. Our study, one of the few conducted in the Guatemalan humid lowlands, shows that oil palm plantations and pasturelands most strongly degrade water quality, while forested catchments provide the highest water quality based on both macroinvertebrates and physicochemical parameters. Notably, significant differences among land uses were only detected when using macroinvertebrates as bioindicators, underscoring the need to consider historical and ecological context when assessing water quality. These findings could potentially contribute to the development of mitigation strategies to protect freshwater ecosystems and the services they provide, emphasizing the importance of conserving and restoring forest cover at the catchment scale and regulating further expansion of oil palm plantations into primary forest and basic grains cropland. Our study grouped all stages and management practices of oil palm plantations into a single category; therefore, future work should examine how specific plantation characteristics influence water quality. Finally, despite limitations, our results may be applied to similar tropical catchments undergoing rapid conversion to industrial monocultures, to anticipate how land use change affects stream water quality and aquatic ecosystem functioning.

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Author contributions V.M. together with R.M. determined the research goals and secured funding. V.M, J.G., and E.R collected and analyzed all the water quality and macroinvertrbrate data. V.M. wrote the paper with important contributions from J.C, P.P., and C.G.

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Data availability Data will be provided upon request to authors.

Declarations

Conflicts of interest The authors declare no competing interests.

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