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Rojas-castillo, O.a.; Kepfer-rojas, S.; Vargas, N.; Jacobsen, D.

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Forest buffer-strips mitigate the negative impact of oil palm plantations on stream communities

O.A. Rojas-Castillo a,d,⁎, S. Kepfer-Rojas b, N. Vargas c,d, D. Jacobsen a

a Freshwater Biology Section, Department of Biology, University of Copenhagen, Universitetsparken 4, third floor, 2100 Ø, CPH, Denmark
b Forest, Nature and Biomass Section, Department of Geosciences and Natural Resource Management, University of Copenhagen, Rolighedsvej, 23 1958 Frederiksberg C, Denmark
c Centro de Estudios del Mar y Acuicultura, Universidad de San Carlos, Ciudad Universitaria zona 12, Edificio T-14, Guatemala
d Escuela de Biologia, Universidad de San Carlos, Ciudad Universitaria zona 12, Edificios T-10 y T-12, Guatemala

HIGHLIGHTS

• Oil palm agriculture threatens biodiversity, but few studies address freshwaters.
• We compared stream macroinvertebrate communities in oil palm, forest, and grazing land.
• Oil palm and grazing land negatively affected the aquatic ecosystem integrity.
• Riparian forests in the plantations regulated stream temperature and substratum.
• Oil palm with riparian forests showed a forest-alike macroinvertebrate community.

ABSTRACT

The global area cultivated with oil palm has doubled in the past two decades, causing deforestation, land-use change, freshwater pollution, and species loss in tropical ecosystems worldwide. Despite the palm-oil industry being linked to severe deterioration of freshwater ecosystems, most studies have focused on terrestrial environments, while freshwaters have been significantly less studied. We evaluated these impacts by contrasting freshwater macroinvertebrate communities and habitat conditions in 19 streams from primary forests (7), grazing lands (6), and oil palm plantations (6). In each stream, we measured environmental characteristics, e.g., habitat composition, canopy cover, substrate, water temperature, and water quality; and we identified and quantified the assemblage of macroinvertebrates. Streams in oil palm plantations lacking riparian forest strips showed warmer and more variable temperatures, higher turbidity, lower silica content, and poorer macroinvertebrate taxon richness than primary forests. Grazing lands showed higher conductivity and temperature, and lower dissolved oxygen and macroinvertebrate taxon richness than primary forests. In contrast, streams in oil palm plantations that conserved a riparian forest, showed a substrate composition, temperature, and canopy cover more similar to the ones in primary forests. These habitat improvements by riparian forests in the plantations increased macroinvertebrate taxon richness and maintained a community resembling more the one in primary forests. Therefore, the conversion of grazing lands (instead of primary forests) to oil palm plantations can increase freshwater taxon richness only if riparian native forests are safeguarded.
1. Introduction

Palm oil from *Elaeis guineensis* Jacq (oil palm) is the most consumed vegetable oil in the world (Meijaard et al., 2020). Currently, it represents a total of 19.6 M ha (Descals et al., 2021), with expectations of expanding even further (Coley, 2009; Meijaard et al., 2020). As oil palm agriculture is limited to the wet tropics, much of its expansion has been at the expense of biodiversity hotspots such as tropical rainforests (Ghazoul and Sheil, 2010). A clear example is Southeast Asia, where the long history of oil palm agriculture has led to considerable environmental impacts due to the loss of natural forest areas (Fitzeberht et al., 2008; Turner et al., 2011).

In Meso-America, another biodiversity hotspot, Guatemala has experienced a recent expansion of the crop. However, the dynamic of this expansion has differed from that in Southeast Asia in that it has involved less deforestation and more conversion of other land uses (Vijay et al., 2016). The Guatemalan expansion of oil palm has come mainly at the expense of non-forested areas (Furumoto and Aide, 2017). While this expansion has provoked considerable forest loss in some regions of the country, especially in the northern state of Petén (10,296 ha, representing 90 % of the forest loss by the crop in the country), in most territories, it has expanded at the expense of grazing lands, crop lands, and other plantations (Furumoto and Aide, 2017).

The expansion of agricultural land uses such as grazing land and oil palm plantations is a major driver of biodiversity decline in tropical forested regions (Tilman et al., 2017). Furthermore, the use of agrochemicals and the removal of riparian buffers in these land-uses threatens freshwater habitats (Carpenter et al., 2011) by increasing erosion and several pollutants, and negatively affecting water quality, diversity of habitats, food, and shelters (Johannesen and Armitage, 2010; Kano et al., 2020; Luke et al., 2017; Shortle et al., 2001); the impacts of these practices, can even affect global scales events (e.g., bio-geochemical cycles of carbon, phosphorus, nitrogen, and silicon) (Derry et al., 2005; Quinton et al., 2010). Due to these consequences, agriculture has been associated with negative impacts on freshwater communities and biodiversity (Chará-Serna et al., 2015). The retention of riparian forests in tropical systems has been effective in minimizing the impacts of agriculture on water quality (Fernandes et al., 2014) and biodiversity (Deere et al., 2022); however, the degree to which riparian forests support biodiversity in these lands is highly variable and specific to location, taxa, and land use (Deere et al., 2022).

In Guatemala, the oil palm industry has been accused of causing severe deterioration of freshwater ecosystems, including water pollution (Rojas et al., 2022) and detrimental impacts on freshwater biota (Olmstead, 2018). However, so far, no study has actually quantified the impacts of oil palm plantations on freshwater biodiversity nor the mitigating effect of riparian forests in these plantations. This study is the first to assess the impact of land use conversion caused by oil palm plantations on streams and their macroinvertebrate community in the country. We assessed freshwater macroinvertebrates since these are used worldwide as bioindicators for habitat and water quality (Eriksen et al., 2021), and play crucial roles in several ecological processes (Ramírez and Gutiérrez-Fonseca, 2014). These processes include stream food webs encompassing functional feeding groups (FFG) such as shredders, scrapers, predators, collectors, and filter feeders (Ishikawa et al., 2016).

We studied streams from grazing lands, primary forests, and oil palm plantations, with and without riparian forest strips. In each stream, we measured a number of environmental characteristics and sampled benthic macroinvertebrates; we then compared these variables across the land-use types to answer our research questions. (A) How do water quality and habitat properties differ between these land uses? (B) How are these land uses and riparian forest strips affecting the macroinvertebrates (in terms of taxon richness, abundance, composition, and biomass per functional feeding groups)? (C) What are the key environmental factors driving these biotic responses?

We hypothesized that the streams with riparian forests strips in oil palm plantations would resemble more the streams from primary forests than any streams lacking these strips, in terms of habitat, water quality, and macroinvertebrates. Streams lacking forest strips will host fewer taxa and abundance, and a less homogeneous macroinvertebrate assemblage and biomass of FFGs. And finally, that these biotic responses would be associated with environmental characteristics in the streams.

2. Methodology

2.1. Study area

We conducted our study in the Lachuá region within the Northern Transversal Strip or FTN (a major road project expected to connect Belize to Mexico). This area was initially covered with tropical broadleaf forests. However, in the period from 1962 to 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). More recently, oil palm plantations started establishing in the FTN and Petén (department neighboring FTN) (MAGA, 2012); at the moment, these regions hold 59 % of the total area cultivated with oil palm in the country (GREPALMA, 2019).

The Lachuá region in Alta Verapaz, Guatemala, is part of the Petén-Ve-racruz Moist Forest ecoregion (Quezada et al., 2014). The region extending 535 Km2 is composed by the Laguna Lachuá National Park and its surrounding buffer zone (MAGA, 2012). Lachuá is found within the Usumacinta basin, and it is bordered to the north, west, and east by the Chixoy and Icbolay rivers and to the south by the Sultana mountains. The physiographic setting is contextualized by the Lacandón Fold Belt province and karstic Upper Cretaceous geology. Annual rainfall is >2500 mm, mean air humidity 91 %, mean annual temperature 25.3 °C, and there are two predominating seasons: dry (February–April) and rainy (June–October). The original vegetation is tropical rainforest, and it is considered one of the most important rainforests in the country (CONAP, 2003).

The study included seven streams in primary forest within the Laguna Lachuá National Park (F), six streams in grazing lands (G), and six in oil palm plantations: three with riparian forest strips (P RF) and three with no riparian forest strips (P) (Fig. 1). The primary forest was characterized by diverse dense vegetation composed of 76 different plant families including Orchidaceae, Arecaceae, Fabaceae, Rubiaceae, Moraceae, Melastomataceae and Meliaceae (Castañeda Cerna, 1997). The grazing lands were small-scaled (13–20 ha), with low tree cover (mean 13 %), and characterized by grasses and occasionally cattle. The oil palm plantation under study was established in 2014. It covers an area of 420 ha with secondary tropical forest strips (30–40 m wide) surrounding some of the streams. The fact that our study was performed in comparable streams, that were located close to each other, and were expected to host similar communities, reduced the effect of other confounding environmental variables, such as climate, altitude, and type of soil, that can affect big-scale studies.

2.2. Mapping and stream selection

We mapped the stream network by walking along every stream to the headwaters while recording the GPS coordinates with the track function (Garmin GPSMAP 64s). We then selected the 19 streams and their sampling spots based on their accessibility.

2.3. Environmental characteristics

At each site, we measured dissolved oxygen at two depths (near the stream bed and in the middle of the water column) employing a handheld multiprobe (Model 6000; YSI, Yellow Springs, OH, USA), turbidity with a Eutech-100 turbidity meter (Eutech, Nijkerk, Netherlands), and pH with the pH-meter ecolTest pH 2. We calculated stream discharge and current velocity by dilution gauging (White, 1978) by diluting 1 kg of salt (NaCl) in 10 L of water and adding it to the stream while monitoring the stream's electrical conductivity every 10 s (6000 YSI multiprobe) 15 m downstream of the addition site. We installed HOBO Pendant® MX2202 Temperature/Light data loggers for 30 days, and HOBO U24 conductivity data loggers and HOBO U20L-01 water level data loggers for 15 days.
Additionally, we took water samples for analyses of inorganic N, phosphates, silica, and total solids. These analyses were performed at Analytic Solutions company and LIQA (Laboratorio de Investigación Química y Ambiental), following Standard Methods (Rice et al., 2012).

We calculated the mean width of each study reach from 15 transverse transects, and the mean depth from 50 points allocated along those transects. From these transects, we characterized the different habitats in the streams based on the percentage of riffles, runs, and pools within a 100 m reach. In addition, we categorized substrate composition based on 100 measurements taken randomly within the 100 m reach, applying the following categories: gravel & pebbles, sand, mud, wood (sticks, branches, roots), leaf litter, and submerged vegetation (aquatic plants and grasses); and calculated the Shannon-Wiener Index to estimate substrate diversity.

We calculated canopy cover from 20 perpendicular photographs taken 15 m apart (from a 1 m tripod standing in the middle of the stream) within a transect of 300 m. We then employed HabitApp software to measure the percentage of cover of each photograph and calculated an average. To conduct a further description of the catchment, we calculated the area and characterized the main land uses employing Google Earth’s most recent satellite images (Google earth V7.3.4.8642, 2021).

2.4. Macroinvertebrate fauna

In each stream, we collected six Surber samples (20 × 25 cm, mesh-size 200 µm), two from each of the habitats runs, riffles, and pools, separated by at least 10 m. The material from the Surber-net was stored in plastic bags and preserved in 96 % ethanol. We counted and identified the macroinvertebrates to family or when possible, to genus level (Domínguez and Fernández, 2009; Hamada et al., 2018; Springer et al., 2010). We classified the fauna according to functional feeding group (Merritt et al., 2017; Oliveira and Nessimian, 2010; Pereira et al., 2021). After a standard drying procedure (20 s in a paper towel), we weighed the macroinvertebrates (WW) using a Mettler Toledo AE50 analytical balance.

2.5. Statistical analysis

2.5.1. Environmental characteristics

We averaged the variables measured by dataloggers. We tested differences in environmental characteristics between land uses, employing ANOVAs (for normally distributed data) or Kruskall-Wallis (KW) (for non-normal data); and an ANOSIM (Bray Curtis dissimilarity index) for substrate composition. To visualize the environmental variables, we performed a PCA (scaling = “symmetric”) to address both the relationship between the variables and the sites) to the standardized data of mean conductivity, silica, phosphates, inorganic nitrogen, near-bed dissolved oxygen, pH, turbidity, mean water temperature, canopy cover, watershed area, width, depth, current speed, water level variability, and bottom substrates.

2.5.2. Macroinvertebrate community

We tested the differences in macroinvertebrate taxon richness and abundance between land uses by performing an ANOVA on a linear mixed-effect model on each habitat (riffles, runs, and pools), and included the stream as a random effect since we used a repeated measurements design (Forman, 2019).

To explore the macroinvertebrate assemblage composition, we performed Non-metric multidimensional scaling (NMDS) based on the Jaccard index on presence/absence data per stream reach, as well as on the Bray-Curtis index from relative abundances. These were followed by Permanovas to detect differences between land uses. These similarity indexes were selected as they are the least sensitive to sampling error.
(Schroeder and Jenkins, 2018). We then performed an indicator species analysis to identify the taxa associated with each land use; we used a point biserial correlation coefficient (r.g.) as it corrects the measure for unequal group sizes (Borcard et al., 2011) (R code available in supplementary material S2).

To address the macroinvertebrate community functionality, we analyzed biomass per functional feeding group (FFG) (i.e., scrapers (SC), shredders (SH), collectors (GC), filter feeders (FC), and predators (P)). We summarized the information in a bubble graph and, to have an estimation per stream, we calculated a stream weighted average (WA) for the biomass of each FFG and the macroinvertebrate abundance. The WA was based on the proportional cover of the main habitats (riffles, runs, and pools) by following the formula:

$$WA_{stream} = \left( x \text{ in riffles} \times \text{stream A} \times \text{riffle} \times \% \text{ stream A} \right) + \left( x \text{ in pools} \times \text{stream A} \times \text{pool} \times \% \text{ stream A} \right) + \left( x \text{ in runs} \times \text{stream A} \times \text{run} \times \% \text{ stream A} \right)$$

*Where: $x =$ biomass per FFG or abundance.

We then used an ANOVA on the WA biomass per FFG to test differences between land uses.

2.5.3. Fauna in relation to environmental characteristics

We performed linear models to elucidate relationships between fauna variables that differed between land uses (macroinvertebrate taxon richness per stream, WA abundance, and WA shredders biomass) and the environmental variables. Since we did not have numerous measurements of the environmental variables (only one per stream), we performed simple models between our response biotic variables and stream substrates, water temperature, and water quality parameters. We then compared the AICs of the models selecting the best-fitting ones. Since canopy cover was highly influential on the stream habitat and environment, we selected from the best-fitting models the variables that correlated with canopy to build path analyses aimed at explaining the possible environmental effects on the fauna. Finally, to test for potential spatial autocorrelation we performed a Mantel test on the community assemblage that was not significant ($p = 0.18$).

All statistical analyses were performed in R version 4.1.3 (R Core Team, 2022) and RStudio version 2022.2.1.461 (RStudio Team, 2022) (R code and list of packages employed available in supplementary material S1).

3. Results

3.1. Environmental characteristics

Streams in different land uses did not differ significantly in terms of watershed area, habitat composition (i.e. riffle, run, pool), substrate diversity (Shannon index), width, depth, current velocity, discharge, water level variability, pH, and concentration of phosphate, inorganic nitrogen, total solids, and dissolved oxygen. However, they did differ regarding canopy cover, silica concentration, near-bed dissolved oxygen, turbidity, electrical conductivity, water temperature, light, and substrate composition (Table 1).

The PCA separated the streams with riparian forest or dense canopy (primary forest and oil palm with riparian forest) from the ones with open canopy (grazing land and oil palm with no riparian forest). Streams with riparian forest showed a positive relationship with canopy, sand, leaf litter, and dissolved oxygen; while streams with open canopy showed higher temperature, mud, and submerged vegetation (Fig. 2). The substrates from primary forest differed (ANOSIM, $p < 0.05$) from every other land use except from oil palm plantations with riparian forests; while the substrates from grazing lands were only similar to the ones in oil palm plantations lacking riparian forests.

Primary forest streams had almost twice the concentrations of silica compared to those in grazing lands and three-times more than those in oil palm plantations. Near-bed dissolved oxygen, which was low in all land uses, differed (ANOVA, $p = 0.024$) between land uses. Primary forest streams showed more than double the concentration of near-bed dissolved oxygen compared to open canopy streams while oil palm plantations with riparian forests showed intermediate values. Stream turbidity in oil palm

Table 1

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Canopy (%)</th>
<th>Shed (ha)</th>
<th>Riffles (%)</th>
<th>Pools (%)</th>
<th>Runs (%)</th>
<th>Substrate diversity (H)</th>
<th>Gravel &amp; pebbles (%)</th>
<th>Wood (%)</th>
<th>Leaf litter (%)</th>
<th>Mud (%)</th>
<th>Sand (%)</th>
<th>Submerged vegetation (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Grazing land</td>
<td>14 (17.5)</td>
<td>C</td>
<td>23 (4.0)</td>
<td>29 (10.9)</td>
<td>48 (9.8)</td>
<td>23 (9.2)</td>
<td>1.6 (0.3)</td>
<td>13 (20.9)</td>
<td>12 (5.3)</td>
<td>9 (8.8)</td>
<td>35 (6.0)</td>
<td>0 (0.0)</td>
</tr>
<tr>
<td>Oil palm</td>
<td>37 (12.3)</td>
<td>B</td>
<td>15 (6.5)</td>
<td>24 (7.7)</td>
<td>44 (13.9)</td>
<td>31 (13.9)</td>
<td>1.8 (0.1)</td>
<td>8 (11.3)</td>
<td>26 (4.6)</td>
<td>14 (5.8)</td>
<td>33 (6.9)</td>
<td>1 (0.9)</td>
</tr>
<tr>
<td>Oil palm with riparian forest</td>
<td>81 (0.9)</td>
<td>A</td>
<td>67 (31.1)</td>
<td>31 (7.7)</td>
<td>47 (6.7)</td>
<td>22 (3.8)</td>
<td>2 (0.2)</td>
<td>7 (4.1)</td>
<td>35 (3.7)</td>
<td>36 (13.9)</td>
<td>8 (4.1)</td>
<td>14 (8.0)</td>
</tr>
<tr>
<td>Forest</td>
<td>83 (2.5)</td>
<td>A</td>
<td>71 (80.3)</td>
<td>35 (8.5)</td>
<td>41 (15.6)</td>
<td>24 (11.4)</td>
<td>1.8 (0.2)</td>
<td>19 (12.9)</td>
<td>30 (3.8)</td>
<td>31 (3.3)</td>
<td>8 (5.2)</td>
<td>13 (6.3)</td>
</tr>
</tbody>
</table>

Stream’s characteristics

<table>
<thead>
<tr>
<th>Width (cm)</th>
<th>Depth (cm)</th>
<th>Current (L/s)</th>
<th>Discharge (L/s)</th>
<th>Temp °C</th>
<th>Temp max. °C</th>
<th>Temp min. °C</th>
<th>Light (lux)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Grazing land</td>
<td>168 (47)</td>
<td>17 (4.7)</td>
<td>1.9 (1const (1)</td>
<td>5 (2.8)</td>
<td>24.5 (0.7) C</td>
<td>28.8 (3.4) B</td>
<td>21.1 (0.5) B</td>
</tr>
<tr>
<td>Oil palm</td>
<td>137 (2)</td>
<td>16 (5.1)</td>
<td>1.4 (0.6)</td>
<td>3 (1.8)</td>
<td>24.6 (0.6) BC</td>
<td>28.2 (1.8) B</td>
<td>20.4 (0.5) AB</td>
</tr>
<tr>
<td>Oil palm with riparian forest</td>
<td>153 (52)</td>
<td>20 (1.7)</td>
<td>2 (0.8)</td>
<td>6 (2.5)</td>
<td>22.7 (0.5) AB</td>
<td>24.0 (0.2) AB</td>
<td>20.8 (0.7) AB</td>
</tr>
<tr>
<td>Forest</td>
<td>192 (76)</td>
<td>15 (4.7)</td>
<td>2.1 (2.9)</td>
<td>12 (25)</td>
<td>22.3 (0.4) AB</td>
<td>23.6 (0.4) A</td>
<td>19.9 (0.3) A</td>
</tr>
</tbody>
</table>

Water quality

<table>
<thead>
<tr>
<th>SiO2 (mg/L)</th>
<th>PO4 (mg/L)</th>
<th>Inorganic N (mg/L)</th>
<th>TS (mg/L)</th>
<th>DO (mg/L)</th>
<th>DO depth bed (mg/L)</th>
<th>pH</th>
<th>Turbidity (NTU)</th>
<th>Conductivity (µS/cm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Grazing land</td>
<td>21 (7.8)</td>
<td>0.02 (0.03)</td>
<td>0.39 (0.45)</td>
<td>0.13 (0.05)</td>
<td>3 (1.5)</td>
<td>1.2 (1.1) A</td>
<td>6.6 (0.3)</td>
<td>4 (2) AB</td>
</tr>
<tr>
<td>Oil palm</td>
<td>13 (2.1)</td>
<td>0.04 (0.02)</td>
<td>0.14 (0.1)</td>
<td>0.26 (0.32)</td>
<td>4 (0.8)</td>
<td>1.6 (0.9) A</td>
<td>6.7 (0.1)</td>
<td>19 (2.1) B</td>
</tr>
<tr>
<td>Oil palm with riparian forest</td>
<td>12 (1.0)</td>
<td>0.09 (0.07)</td>
<td>0.24 (0.36)</td>
<td>0.13 (0.02)</td>
<td>4 (0.8)</td>
<td>2.2 (1.2) AB</td>
<td>6.9 (0.3)</td>
<td>11 (8.1) AB</td>
</tr>
<tr>
<td>Forest</td>
<td>37 (15.4)</td>
<td>0.08 (0.11)</td>
<td>0.23 (0.17)</td>
<td>0.12 (0.04)</td>
<td>4 (1.5)</td>
<td>3.3 (1.2) B</td>
<td>6.6 (0.4)</td>
<td>5 (3.2) A</td>
</tr>
</tbody>
</table>
were the second coldest and with only a 0.8 °C
temperature were lowest in primary forest and with only a 0.7 °C
vegetation (sub_veg), mud. Streams in primary forest (F1
Fig. 2. Principal components analysis on the environmental variables in the
Fig. 3. Daily water temperature per land use. Mean water temperature represented
by line and standard error (mean +/− SE) represented by shaded area; grazing
land (G), oil palm plantations with no riparian forest (P) and with riparian forest
(P_RF), and primary forest (F).

plantsations was higher than in primary forests and grazing lands (KW, p
and water conductivity was greater (ANOVA, p = 0.046) in streams
in grazing lands than in primary forests, while streams in oil palm planta-
tions showed intermediate values (Table 1 and Fig. 2).

Open-canopy streams had higher temperature, light, and daily variabil-
ity than dense-canopy streams (ANOVA, p < 0.05) (Fig. 3). Overall, water
temperatures were lowest in primary forest and with only a 0.7 °C fluctua-
tion, while being warmest and with higher fluctuations (2.7 °C) in grazing
lands. Water temperature in oil palm plantation was intermediate. How-
ever, the plantations lacking riparian forests showed warmer temperatures and higher fluctuations (3 °C) than the ones with riparian forest, which were the second coldest and with only a 0.8 °C fluctuation (Table 1, Fig. 3).

3.2. Macroinvertebrate taxon richness and abundance

The number of taxa differed by land use (ANOVA, p < 0.05). Primary
forest showed the highest values in all habitats (pools, ripples, and runs)
followed by oil palm plantations with riparian forests and the lowest oc-
curred in grazing lands and oil palm plantations with no riparian forests (Fig. 4A). On average, primary forest streams hosted twice the number of taxa (49 taxa) than open-canopy streams (p = 21 taxa & G = 28 taxa)
but only 8 % more than oil palm plantations preserving riparian forest strips (45 taxa). Abundance differed significantly only in runs (ANOVA, p = 0.001) where open-canopy streams had fewer individuals than the dense-
canopy ones (Fig. 4B).

3.3. Macroinvertebrate community composition

The macroinvertebrate community assemblage (presence absence data)
from primary forest differed (Permanova/pairwise Adonis, p < 0.05) from
every other land uses except from oil palm plantations with riparian forests
(p = 0.08) which differed from grazing land (p = 0.035) but not from oil
palm lacking forest strip (p = 0.1) (Fig. 5A). However, when analyzing the
relative abundance data, the assemblage from primary forest differed sig-
nificantly from every other land use (Permanova/pairwise Adonis, p < 0.05), while grazing lands differed from oil palm plantations lacking ri-
parian forests (Fig. 5B). The indicator species analysis showed the highest
number of taxa (5 taxa) associated to streams in oil palm plantations with riparian forests, followed by primary forest (4 taxa), and three associated to both dense-canopy streams (Fig. 5C). It also associated grazing land to
one dipteran subfamily (Tanytarsinae) and the snail family Ampullariidae
(shared with oil palm with riparian forests); while oil palm plantation lack-
ing riparian forests was only associated with Orthocladiinae (Fig. 5C).

3.4. Macroinvertebrate biomass and functional feeding groups (FFGs)

All FFGs were found in all four stream groups, but the distribution of
biomass by FFG differed (Fig. 6). In the grazing lands and oil palm planta-
tions (without riparian forests) the biomass was dominated by scrappers
(especially gastropods such as Tarebia granifera and Ampullaridae family)
followed by predators (Aphylla). The land uses with denser canopy (pri-
mary forests and oil palm plantations with riparian forests) showed a bal-
anced representation of shredders (Tipulidae, Caridae, Phyllocus,
Triplectidae, and Anchytarsina) and predators (Hetaerina and Phyllogomphoidae). The only FFG biomass that differed sig-
nificantly (ANOVA, p < 0.05) between land uses was that of shredders,
which was greater in dense-canopy land uses, intermediate in oil palm plant-
tations with no riparian forest, and lower in grazing lands.

3.5. Key environmental factors driving macroinvertebrate community response

Our path analyses showed a positive effect of canopy on leaf litter, sand,
wood, substrate diversity, and near-bed dissolved oxygen and a negative ef-
effect of canopy on water temperature, light standard deviation, and mud
(Fig. 7). There were positive effects of leaf litter, sand, near-bed dissolved oxygen, and wood on taxon richness, and negative effects of mud and water temperature (Fig. 7A). There were positive effects of substrate diver-
sity, dissolved oxygen, and wood on macroinvertebrate abundance, and
Fig. 4. Macroinvertebrate number of taxa (A) and abundance (B). Land uses: primary forest (F), grazing land (G), oil palm with no riparian forest (P), and oil palm with riparian forests (P_RF). Each land use shows the values per habitat which includes pools (lighter colors and located to the left in each land use), riffles (intermediate colors and located in the middle of each land use), and runs (darker color and located to the right in each land use). Significant differences (ANOVA, p < 0.05) between land uses are represented by different letters in the upper part of each graph (the gray area). The mean is represented by the line inside each box, Quartile 1 and 3 by the lower and upper limits of the box, minimum and maximum values by the lower and upper vertical lines outside the box, and outliers by gray dots outside the boxes.

Fig. 5. Macroinvertebrate community assemblage: presence/absence data (A) relative abundance (B) and indicator species analysis per land use (C). Land uses: primary forest (P), grazing land (G), oil palm with no riparian forest (P), and oil palm with riparian forests (P_RF). Fig. C shows the taxa correlated (p < 0.05) with each land use (ovals) and the phi coefficient of association (r.g.) per taxon.
negative effects of light standard deviation and mud (Fig. 7B). There were positive effects of leaf litter, sand and wood on shredder biomass, and negative effects of water temperature and mud (Fig. 7C).

4. Discussion

Our results support previous findings that oil palm agriculture has negative effects on the integrity of aquatic ecosystems when primary forests are removed (Chellaiah and Yule, 2018a; Wilkinson et al., 2018a, b). This was perceived by differences in environmental characteristics in the streams and the macroinvertebrate community composition, taxon richness, and biomass of shredders between oil palm plantations lacking riparian forests and primary forests. Yet, by comparing grazing lands and oil palm plantations we found that the conversion from grazing lands to oil palm plantations can improve the stream environment, increase species richness, and, to some extent, conserve the natural aquatic community, but only if riparian forests are protected, as reported by Deere et al. (2022).

4.1. How do water quality and habitat properties differ between land uses?

In general, dense-canopy (primary forests and oil palm plantations with riparian forests) streams had more leaf litter, wood, and sand; and less mud and submerged vegetation. Riparian forests provide leaf litter input (Muto et al., 2009), habitat structure (Pusey and Arthington, 2003), and prevent soil erosion, minimizing the deposition of fine substratum (e.g., mud) into the streams (Feld et al., 2018; Jones et al., 1999). Dense-canopy streams were also cooler by 1.7 °C - 4.8 °C. This coincides with previous studies ascribing 3 °C - 7 °C warmer waters due to the increased solar radiation and reduced evapotranspiration resulting from the removal of riparian vegetation (Carlson et al., 2014; Chellaiah and Yule, 2018b). Furthermore, water temperatures in open-canopy streams fluctuated three times more than in streams with a dense-canopy, demonstrating the buffer role of riparian forests on stream temperature (Knouft et al., 2021).

All dissolved oxygen concentrations were low. Even so, near-bed dissolved oxygen differed significantly by land use. Primary forest streams
Fig. 7. Path analyses of the environmental factors driving macroinvertebrate community response. Arrows show interactions and values above arrows show linear models' slopes, p-value below arrows, and R-squared on top of boxes.
showed more than double the amount of oxygen than open-canopy streams, coinciding with Carlson et al. (2014). Oxygen might be affected by oxygen-consuming mud on the stream bottom together with the almost stagnant water (current velocity of 1–2 cm s⁻¹). This probably explains why dissolved oxygen differed only near the bed and not in the middle of the water column.

Silica seemed to be depleted in oil palm plantations, primary forests showed the greatest silica concentrations in the water, and grazing lands showed intermediate values. Forests act as sinks and sources of silicon by promoting weathering and forming phytoliths that are then reincorporated into the environment (Cornelis and Delavaux, 2016), controlling the silicon cycling and export (Derry et al., 2005; Fernandes et al., 2014). Our results concur with previous knowledge that agriculture depletes the phytolith pools (reservoirs of silicon) through crop uptake and harvesting (Koller et al., 2012) tampering the smooth release/recycle of the element and affecting its biogeochemical cycle (Derry et al., 2005). The intermediate values of silicon in grazing lands have been previously attributed to a more diverse vegetation structure when compared to the monocrop (Rojas et al., 2022). However, this seems to be different in our study since all streams in oil palm plantations showed low silica concentrations regardless of their riparian forests. Streams and rivers provide 80 % of the silicon entering the ocean (Treguer et al., 1995), changes in its cycle might have further ecological and climatic impacts since silicon interacts with the carbon in the atmosphere increasing carbon sequestration (Rojas et al., 2022; Song et al., 2012; Street-Perrott and Barker, 2008). The reduction of silica might also affect some macroinvertebrates species that consume it (e.g., tubificid oligochaetes and chironomid larvae) (Adámek and Maršílek, 2013) or are susceptible to changes in pH (Vicca et al., 2022); though, too few studies have tested the effect of silica on macroinvertebrates.

4.2. How are macroinvertebrates affected by land uses and riparian buffer strips?

4.2.1. Taxon richness and abundance

Our study showed that overall, streams in grazing lands and oil palm plantations lacking riparian forests host 43 % or 56 % (21 and 28 taxa, respectively) fewer taxa than primary forest streams. This coincides with most published studies (Savilaakso et al., 2014). On the other hand, our results also show that the streams in the plantations with riparian forests host only 8 % fewer taxa (4 taxa) than in primary forests. Studies in oil palm plantations with well-developed 30 m-wide riparian natural vegetation strips in the Amazon showed higher or similar taxon richness of larvae and adult dragonflies in the plantation compared to primary forests (Carvalho et al., 2018; Mendes et al., 2019). This indicates a positive relationship between macroinvertebrate taxon richness and riparian vegetation observed elsewhere (Bojsen and Jacobsen, 2003). However, this does not seem to be the case for all groups of macroinvertebrates; EPT (Ephemeroptera, Plecoptera, and Trichoptera) taxa and heteropterans in Brazil seem to show fewer genera in oil palm plantations with riparian forest than in primary forest (Cunha and Juen, 2017; de Paiva et al., 2017; Luiza-Andrade et al., 2017), this was not the case for mayflies and caddisflies (Ephemeroptera and Trichoptera) in our study (supplementary material S4). Regardless of these differences, riparian forests indeed appear to buffer, to some extent, the loss of taxa in the plantations (Chellaiah and Yule, 2018a; Marques et al., 2021).

Macroinvertebrate abundance response to land use seems more variable and contradictory among studies as some report higher (Harun et al., 2015; Mendes et al., 2019) and others lower abundance in oil palm plantation streams compared to primary forest (Chellaiah and Yule, 2018a; Cunha and Juen, 2017; Mercer et al., 2013), or no difference between these land uses (Carvalho et al., 2018; Shimano and Juen, 2016). These contrasting findings might depend on the dynamics and biology of the species present within and close to the plantations. Generalists tend to thrive while forest specialists decrease in the plantations, influencing the abundance of the community differently (Correa et al., 2015; Ferreira et al., 2018; Fitzherbert et al., 2008; Wilkinson et al., 2018a).

4.2.2. Community composition

All over the world, macroinvertebrate communities have been employed as bioindicators of the state of freshwater habitats (Ruaru et al., 2020); the EPT group (Ephemeroptera, Plecoptera, and Trichoptera), for example, has been the most popular tool in the tropics (Eriksen et al., 2021) due to their general high vulnerability to habitat degradation (e.g., physical habitat, substratum, water quality) (Ferreira et al., 2014; Suhaila et al., 2012). Furthermore, members of this group 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). Even though we found clear differences in the EPT assemblage between land uses (supplementary material S4); here we focused on the whole macroinvertebrate community assemblage.

The community assemblage (based on presence/absence data) in primary forest streams differed from all open-canopy streams, but not from oil palm plantations with riparian forests. Therefore, we can argue that riparian forests conserved, to some extent, the macroinvertebrate community assemblage in the streams, as reported in other studies (Amorim et al., 2011; Deere et al., 2022; Kasangaki et al., 2008; Lorion and Kennedy, 2009). Nevertheless, when based on relative abundance, the community assemblage in primary forest streams differed from all other land uses. These results suggest that the conservation of riparian forests does not translate to the conservation of the community assemblage, but rather the conservation of most of the taxa. Wider riparian buffers may be needed to fully conserve the assemblage of macroinvertebrates, as suggested by Rodrigues et al. (2016). Alternatively, more continuous ones (not fragmented) would suffice, as Lorion and Kennedy (2009) found that 15 m-wide continuous riparian forests maintained the macroinvertebrate assemblages in Costa Rica.

Primary forest streams were associated with stoneflies (Anacronoeirus), as in previous studies (Astudillo et al., 2016; Morabowen et al., 2019). Anacronoeirus tends to decrease in monocrops in the tropics due to increased temperatures (Morabowen et al., 2019) which might explain its relatively high abundance in oil palm plantations with riparian forest cover >30 %. Besides temperature, the great variety of prey types in dense-canopy streams might be benefitting Anacronoeirus, as a facultative predator (Carvalho and Uieda, 2009). Primary forest streams were also associated with dance flies (Empididae) as they tend to be more abundant in areas rich on preys (e.g., non biting midges, black flies, caddisflies) (Ivkovic et al., 2007) and with the shredders including the large crane flies (Tipulidae) (Merritt et al., 2017) and the toe-winged beetle Anchytsaurus (Chará-Serna et al., 2012), probably as forests carried more leaf litter (Cummins et al., 1989).

Oil palm plantations with riparian forests were associated with the caddisflies Triplectides and Marilia. These two caddisflies have been observed in this type of plantation in Brazil (Luiza-Andrade et al., 2017) and have shown strong associations with riparian zone preservation (Corbi et al., 2013; Henriques-Oliveira et al., 2015). The plantations with riparian forests were also associated with dragonfly Phyllogomphoides, coinciding with a Brazilian study (Mendes et al., 2019), and unexpectedly with an adult beetle (Heterelmis) which generally prefers wider riparian forests (Braun et al., 2018). Farrodes and Ulmeritoides mayflies were indicator species of dense-canopy streams. Both have been reported in dense-canopy streams in Brazil (Luiza-Andrade et al., 2017; Shimano and Juen, 2016). However, a recent study considered Farrodes as a forest specialist (Marques et al., 2021) as it avoided most croplands, even with riparian forests. The damselfly Hetaeria was also an indicator of dense-canopy streams, as observed in the Amazon, where primary forests and oil palm plantations with riparian forests strips showed high representation of the genus family (Mendes et al., 2019).

Open-canopy streams were associated with the family Ampullariidae (apple snails), probably as the family includes several tolerant/generalist species that thrive in perturbed or semi-perturbed areas (Joshi et al., 2017; Neumann and Dudgeon, 2002). The non-biting midges sub-family, Orthocladiinae, was particularly abundant in oil palm plantations lacking riparian forests; in contrast, Tanypodinae (another sub-family of non-biting midges) preferred open-canopy streams. These responses might be
in function of the shade; according to Luferov (1971), Orthocladiinae prefers high levels of light and tends to be opportunistic (Pillot, 2014). However, different genera belonging to these subfamilies tend to respond differently to riparian vegetation and agriculture (Sensole et al., 2012).

4.2.3. Functional feeding group

Total biomass (wet weight) did not differ between land uses, however, its composition did. Shredder biomass was significantly greater in dense-canopy streams than in open-canopy ones, coinciding with previous studies worldwide (Alberts et al., 2018; Casotti et al., 2015; Cummins et al., 1989; Masse et al., 2014). Shredders, as leaf eaters, tend to show strong associations with riparian forests as these provide food and shelter (Henriques-Oliveira et al., 2015). The decrease in the shredders' biomass in open-canopy streams can have far-reaching ecological impacts as the FFG play an important role in leaf decomposition (Aguiar et al., 2018; Casotti et al., 2015; Cummins et al., 1989). However, their role appears to be less important in the neotropics as shredders tend to be smaller, less diverse, and less abundant than in non-tropical areas (e.g., Europe). Therefore, their role is often replaced by benthic omnivores such as shrimps and crabs (Wantzen and Diger Wagner, 2006) and probably bacteria and fungi (Wright and Covich, 2005); all of which tend to be more abundant and diverse in forests (McGuire et al., 2015, Fig. 6) than in the plantations, which suggests a decline on litter decomposition due to land use change. The biomass composition from dense-canopy streams was also more homogeneously represented by shredders, scrapers, and predators; while scrapers dominated open-canopy streams. However, the exaggerated biomass of scrapers in oil palm plantations appears to be inflated by P1 stream which hosted several big individuals of the snail Tarebia granifera. Nevertheless, the presence of this invasive species in open-canopy streams could be associated to further ecological impacts, as this invader has produced the displacement of native tropical species in other central American countries (Gutiérrez et al., 1997) and has been associated to alterations of the N nutrient cycle in aquatic systems (Moslemi et al., 2012).

4.3. What are the environmental factors driving the biotic responses?

We found riparian forest (measured as canopy cover) to be the key environmental factor affecting macroinvertebrates, coinciding with Sargac et al. (2021). Riparian forests buffered water temperature, as extreme temperatures cause decreases in species number and densities (Stenert et al., 2008), due to the inability of some of the taxa to survive in extreme conditions, these forests might be preventing species loss (including shredders). Additionally, these forests affected the substratum by providing leaf litter, wood, and sand (Muto et al., 2009), thus facilitating a more complex microhabitat setting richer in food and shelter (Aguiar et al., 2018). This might have benefited taxon richness and shredders biomass, as it is reported for macroinvertebrate richness (Mhaka et al., 2015), fish (Glam et al., 2015), and shredders (Aguiar et al., 2018; Cummins et al., 1989). Riparian forests also reduced the amount of mud and increased near-depth dissolved oxygen which possibly increased taxon richness and abundance (Croljmans et al., 2021; Cunha and Juen, 2017; dos Reis Oliveira et al., 2019). Abundance might also be responding to the substrate diversity, which was positively affected by canopy, coinciding with Boyero (2003).

5. Conclusion

We conclude that streams in primary forests exhibit a greater complexity and ecosystem integrity than in oil palm plantations and grazing lands. However, the conservation of riparian forests within oil palm plantations mitigates the impacts of this agriculture by increasing leaf litter and wood, reducing submerged vegetation and mud, and regulating water temperature. Collectively, these effects increase taxonomic richness and abundance and lead to community assemblages and biomass composition of FFG that resemble more the ones from primary forests. Therefore, the conversion of grazing lands -instead of primary forests- to oil palm plantations can only increase freshwater taxon richness if riparian native forests are safeguarded. The conservation of riparian forests is then a tool to safeguard freshwater ecosystems, as stated by Tolkkinen et al. (2021). Moreover, riparian forests improve connectivity between terrestrial forests and increase not only aquatic but also terrestrial biodiversity (Almeida et al., 2016; Cardoso et al., 2021; Deere et al., 2022), which makes them necessary for a sound management of the plantations. But still, more focus is needed on the logistics, implementation, and restoration of effective riparian forests for the conservation of biodiversity in agricultural lands in the tropics (Deere et al., 2022).

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Appendix A. Supplementary data

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References


