



## Non-biting midge (Diptera: Chironomidae) assemblages as indicators of change in mountain grassland streams under pine forestry management

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**ABSTRACT.** Knowledge about chironomids as bioindicators of environmental change remains limited. This study evaluated how afforestation with pine trees (*Pinus elliotii* Engelm.) and subsequent logging influence chironomid communities, emphasizing taxonomic resolution at the genus level. Over four years (2008, 2012, 2014, and 2015), samples were collected from three forested streams (subsequently logged) and three reference streams in grasslands in the Santa Rosa sub-basin, Córdoba, Argentina. Using stereoscopic and compound microscopes, 20 taxa with different community compositions were identified according to land use. Multivariate analysis and PERMANOVA revealed significant differences between riparian vegetation types. Correspondence analysis separated the communities of standing pine plantations from those of felled pines and native grasslands. Grassland streams harbored more indicator species, particularly *Rheotanytarsus* Thienemann & Bause and *Pseudochironomus* Malloch, while pine plantations were associated with Pentaneurini and Macropelopiini. Precise taxonomic identification enabled the detection of ecological differences between riparian vegetation types, highlighting the bioindicator potential of Chironomidae and the importance of local studies for effective ecosystem monitoring.

**KEYWORDS.** Benthos. Changes in land use. Indicator value. Taxonomic resolution.

### Ensamblajes de quironómidos (Diptera: Chironomidae) como indicadores de cambio en arroyos de pastizales de montaña bajo manejo forestal de pino

**RESUMEN.** El conocimiento sobre los quironómidos como bioindicadores del cambio medioambiental sigue siendo limitado. Este estudio evaluó cómo la forestación con pinos (*Pinus elliotii* Engelm.) y la posterior tala influyen en las comunidades de quironómidos, haciendo hincapié en la resolución taxonómica a nivel de género. Durante cuatro años (2008, 2012, 2014 y 2015), se tomaron muestras de tres arroyos forestados (posteriormente talados) y tres arroyos de referencia en pastizales en la subcuenca de Santa Rosa, Córdoba, Argentina. Utilizando microscopios estereoscópicos y compuestos, se identificaron 20 taxones con composiciones de comunidades distintas según el uso del suelo. El análisis multivariante y PERMANOVA revelaron diferencias significativas entre los tipos de vegetación ribereña. El análisis de correspondencias separó las comunidades de las plantaciones de pinos en pie de las de los pinos talados y los pastizales nativos. Los arroyos de pastizales albergaban más especies indicadoras, en particular *Rheotanytarsus* Thienemann & Bause y *Pseudochironomus* Malloch, mientras que las plantaciones de pinos estaban relacionadas con Pentaneurini y Macropelopiini. La identificación taxonómica precisa permitió detectar cambios ecológicos entre tipos de vegetación ribereña, lo que subraya el potencial bioindicador de Chironomidae y la importancia de los estudios locales para un monitoreo efectivo del ecosistema.

**PALABRAS CLAVE.** Bentos. Cambios en el uso del suelo. Resolución taxonómica. Valor Indicador.

## INTRODUCTION

As human populations grow, climate change becomes more evident and the degradation of natural resources continues to increase (Weiskopf et al., 2020). A wide range of factors including mining, dam construction, deforestation and changes in land use, in particular, affects freshwater ecosystems. The impacts of any of these factors on freshwater ecosystems may have the potential to trigger cascading effects leading to significant and irreversible degradation of this resource (Castello & Macedo, 2016). In fact, the degradation and loss of its biodiversity occurs faster than in any other ecosystem (Paredes et al., 2019; Reid et al., 2019).

In the central region of Argentina, numerous studies have reported that land use changes for productive purposes have led to biodiversity deterioration, habitat loss and the consequent alteration of ecosystem structure and function (e.g. Principe et al., 2015; Licursi et al., 2016; Gómez et al., 2017; Rodríguez et al., 2025). One notable modification is the conversion of natural grasslands into exotic tree plantations, driven by the commercial exploitation of timber resources. This practice results in significant changes to rivers and streams, altering their geological and biological structure (Studinski et al., 2012; Jobbágy et al., 2013; Cibils-Martina et al., 2017) since the establishment of fast-growing perennial species leads to substantial shading of the streambed and an increased input and accumulation of coarse particulate organic matter (Giller & ÓHalloran, 2004; Richardson, 2008). As a result, these streams undergo a shift toward heterotrophic conditions, which in turn affects the biota, particularly seen in algae and aquatic macroinvertebrates communities (Cibils et al., 2015; Montilla et al., 2022; Márquez et al., 2025).

Benthic macroinvertebrates are widely recognized as effective bioindicators of biological water quality due to their distinct ecological and biological characteristics (Rocha et al., 2020; Fenoglio & Doretto, 2021; Mir et al., 2021; Espinar-Herranz et al., 2025). Numerous biotic indices summarize the information provided by this fauna to assess the quality of aquatic environments (Mir et al., 2021). These indices typically use the entire community and often require identification at higher taxonomic levels (e.g., family), making them a relatively quick and simple measure. However, some dominant and species-rich freshwater taxa are often grouped together because identifying them at lower taxonomic levels can be difficult. This is the case for the dipteran family Chironomidae, commonly known as non-biting midges, which represent the most abundant insect group in freshwater ecosystems (Armitage et al., 1995; Ferrington 2008; Pero et al., 2023).

The Chironomidae family includes a large number of sensitive species, as well as several species groups that are tolerant to human-impacted ecosystems (Roque et al., 2010;

Paisley et al., 2014; Egan & Ferrington et al., 2019; Ni et al., 2020). In the last decade, the use of non-biting midges as bioindicators has increased (Nicacio & Juen, 2015; Smaoune et al., 2025), despite the taxonomic challenges and the lack of ecological information still available for this group (Ferrington, 2008; Lencioni et al., 2018). In this regard, Molineri et al. (2020) have emphasized that Chironomidae requires identification at lower taxonomic levels to be effectively used as bioindicators of disturbance; otherwise, they are not useful for distinguishing between polluted and non-polluted environments.

In our study area, Montilla et al. (2022) analyzed the effects of a temporal sequence in the harvest management of a *Pinus elliotii* Engelm plantation on the aquatic macroinvertebrate community. The analysis, which included resolution at the subfamily level for the Chironomidae fauna and the application of biotic indices, revealed slight modifications in water quality in both forested and harvested streams. In this context, we aim to evaluate the effect of harvest management in a pine plantation located in mountain grasslands on the Chironomidae fauna of headwater streams, focusing on taxonomic resolution primarily at the genus level. For this purpose, we compared the fauna of non-biting midges between streams draining afforested and harvested pine plantations, with that in streams flowing through natural grasslands.

## MATERIALS AND METHODS

### Study area

The study was carried out in six first-order streams of the upper Ctlamochita River basin, Córdoba, Argentina (Fig. 1). The lithology is dominated by granitic rocks but localized patches of metamorphic rocks (gneiss, schist and migmatite) can also be found. Annual precipitation in the region reaches 725 mm occurring mostly between spring and the end of summer (October-March) (Cabido et al., 2003) whereas maximum temperature reaches 34 °C in summer (December-March) and decreases up to -5 °C in winter (June-September). The basal flow of these high-altitude streams also varies between seasons, reaching minimum values of up to 4 l/s and maximum values greater than 20 l/s (Jobbágy et al., 2013).

The study streams are located within the Comechingones biogeographic province, as defined by the recent regionalization of high-altitude grasslands (Arana et al., 2017). At this elevation, the grasslands are predominantly composed of native species such as *Festuca hieronymi* Hack, *Nassella filiculmis* (Delile) Barkworth, *Schizachyrium condensatum* (Kunth) Nees, and *Eragrostis airoides* Nees (Oggero & Arana, 2012). Grassland areas are primarily used for extensive livestock grazing and other human activities, such as the extraction of medicinal and aromatic herbs, which

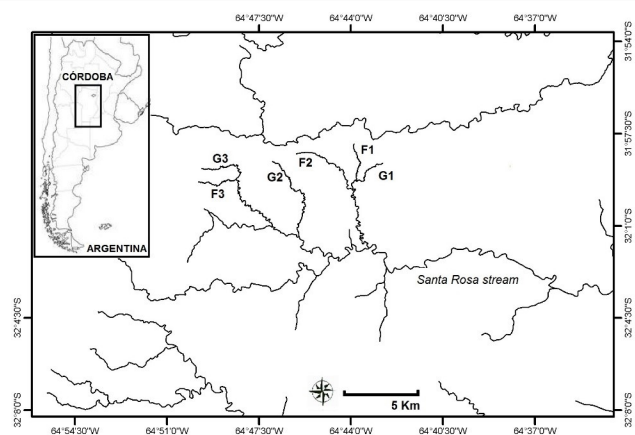
contribute to landscape modification (Cabido et al., 2003). Approximately 36,000 hectares in the area were afforested with pines between 900 and 1,600 m a.s.l., covering 15 % of the upland grasslands in the region (Plevich et al., 2002). However, this area has since decreased due to harvesting and fire. The dominant species in the plantations are *Pinus elliottii*, followed by *Pinus radiata* D. Don and *Pinus taeda* L. all of which are exotic species.

Three streams, draining natural grasslands (control streams, G), were studied alongside three adjacent streams that initially (in 2008) drained *Pinus elliottii* plantations (afforested streams, A). Harvesting of the plantations began in 2011 without any specific resource management plan. The streams selected for sampling were similar in terms of slope, exposure, and altitude (m a.s.l.): G1: 1130, F1: 1159; G2: 1157, F2: 1121; G3: 1175, F3: 1144. Additionally, their drainage areas were comparable (ha): G1: 31, F1: 27; G2: 88, F2: 142; G3: 87, F3: 89.

### Field and laboratory methods

The study was conducted during the Austral winter, corresponding to the low-flow period (July-August). In each year, streams were sampled over multiple days within this period, ensuring comparable hydrological conditions among sites. Samples were collected at the same sites within each stream over four different years: i) 2008 (pre-harvest for all afforested streams), ii) 2012 (1-year post-harvest for F2 - Fig. 1), iii) 2014 (3 years post-harvest for F2, 2 years post-harvest for F3 and 1 year and a half post-harvest for F1) and iv) 2015 (4 years post-harvest for F2 and 3 years post-harvest for F3). On each sampling occasion, hydraulic and physicochemical variables were measured. Water depth and velocity were measured at each sampling site using a digital water velocity meter (FP101 Global Flux Probe). Temperature was recorded with a mercury thermometer, while pH and conductivity were measured using a multiparameter PCSTester 35 (Eutech - Oakton). Dissolved oxygen was measured with a portable YSI DO200 sensor, but only in 2014 and 2015.

Larval Chironomidae samples were collected in riffle habitats using a Surber sampler (0.09 m<sup>2</sup>, 300 µm mesh size) and were immediately fixed in the field with 4 % formaldehyde. In the laboratory, Chironomidae were identified and counted to the lowest possible taxonomic level using a stereoscopic microscope and specific identification keys (Paggi, 2009; Paggi et al., 2023). When it was necessary to examine the details of the characters in the head capsule under the microscope, it was removed and mounted on a microscope slide with glycerin. Most of the specimens were identified to the genus level, with a few exceptions within the subfamily Tanypodinae, which were treated as a tribe (Macropelopiini and some Pentaneurini).



**Figure 1. Streams belonging to Santa Rosa stream sub-basin (Citalamochita River basin). Three streams drain grasslands (G1, G2, G3) and the other three streams drain pine plantations (F1, F2, F3).**

### Data analyses

To evaluate changes in taxonomic composition between stream periods with different vegetation types, abundance rank curves (Magurran, 1988) were plotted considering only taxa with a relative abundance greater than or equal to 5 %. Additionally, temporal variation in chironomid assemblage composition was explored using heat maps based on mean relative abundances of genera across sampling years. Heat maps were constructed separately for each system in order to visualize temporal patterns in assemblage composition, using color intensity to represent the relative contribution of each genus.

With the intention of exploring differences in the Chironomidae assemblages between different stream vegetation types, a Correspondence Analysis (CA) was performed. Then, a one-way permutational multivariate analysis of variance (PERMANOVA, Anderson, 2001) was conducted to statistically test differences between groups, using 9,999 permutations. The statistical software PAST version 3.13 was used to perform these analyses (Hammer et al., 2001).

In order to identify the characteristic taxa of streams with different land uses, the indicator value (IndVal) method proposed by Dufrêne & Legendre (1997) was used. The IndVal method identifies indicator species as those "characteristic" of a particular habitat. It combines measures of the degree of specificity of a species with respect to an ecological state, e.g., habitat type, and its fidelity within that state. Species with high specificity and high fidelity within a habitat will have a high indicator value. In this case, species with a significant indicator value greater than 70 % were considered indicators of the different environments (grassland stream, forested stream, harvested stream).

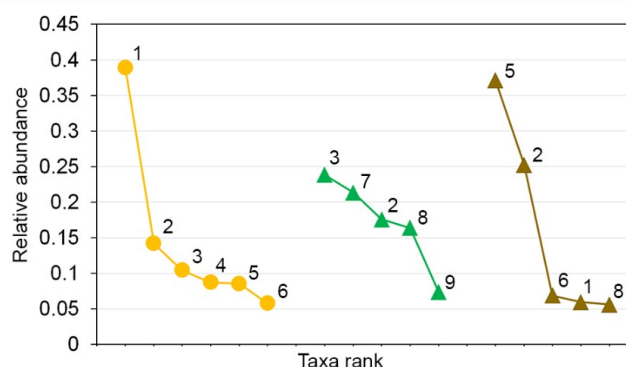
## RESULTS

Environmental variables characterizing the different stream types are shown in Table I.

A total of 20 taxa of Chironomidae were found in this study (Table II). No taxa were exclusive to grassland or forested streams. The most abundant taxa were *Corynoneura* Winnertz in planted pine forests, *Cricotopus* Wulp in harvested forests and *Rheotanytarsus* in grassland streams (Fig. 2). *Parametrioctenemus* spp. Goetghebeur was second in order of dominance for grassland and harvest streams, while Macropelopiini was second in order of dominance for forested streams (Fig. 2).

**Table I. Physicochemical variables measured in streams with different riparian vegetation.**

|                           | MEAN   | SE    | MIN    | MAX    |
|---------------------------|--------|-------|--------|--------|
| <b>GRASSLAND STREAMS</b>  |        |       |        |        |
| Water temperature (°C)    | 12.85  | 1.18  | 3.30   | 17.50  |
| Water velocity (m/s)      | 0.25   | 0.03  | 0.15   | 0.37   |
| Depth (m)                 | 0.07   | 0.01  | 0.04   | 0.13   |
| pH                        | 8.33   | 0.10  | 7.60   | 8.70   |
| Conductivity (µS/cm)      | 126.38 | 27.80 | 71.00  | 280.00 |
| Dissolved oxygen (ppm)    | 11.33  | 0.70  | 9.70   | 14.00  |
| <b>AFFORESTED STREAMS</b> |        |       |        |        |
| Water Temperature (°C)    | 10.74  | 0.63  | 9.50   | 12.50  |
| Water velocity (m/s)      | 0.26   | 0.10  | 0.00   | 0.60   |
| Depth (m)                 | 0.06   | 0.01  | 0.03   | 0.09   |
| pH                        | 8.14   | 0.07  | 8.00   | 8.40   |
| Conductivity (µS/cm)      | 163.00 | 37.31 | 104.00 | 222.00 |
| Dissolved oxygen (ppm)    | -      | -     | -      | -      |
| <b>HARVESTED STREAMS</b>  |        |       |        |        |
| Water Temperature (°C)    | 11.12  | 1.11  | 7.20   | 14.30  |
| Water velocity (m/s)      | 0.30   | 0.07  | 0.00   | 0.52   |
| Depth (m)                 | 0.06   | 0.01  | 0.04   | 0.10   |
| pH                        | 7.93   | 0.11  | 7.50   | 8.30   |
| Conductivity (µS/cm)      | 151.83 | 19.94 | 94.00  | 226.00 |
| Dissolved oxygen (ppm)    | 10.75  | 0.64  | 9.47   | 13.00  |



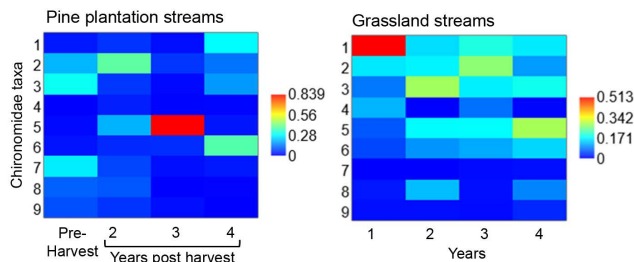
**Figure 2. Rank-abundance curves from grassland (yellow circles), pine (green triangles) and harvested (brown triangles) streams. Only taxa with relative abundance > 5 % were included in the curves. References= 1: *Rheotanytarsus* sp., 2: *Parametrioctenemus* sp., 3: *Corynoneura* sp., 4: *Podonomus* mfsp1, 5: *Cricotopus* sp., 6: *Pentaneura* sp., 7: Macropelopiini, 8: *Podonomus* mfsp2, 9: *Tanytarsus* sp.**

The heat map based on mean relative abundances of chironomids revealed clear temporal changes in community composition associated with the forest harvest gradient (Fig. 3). In afforested streams, the assemblage under pine cover was dominated by genera such as Macropelopiini and *Corynoneura*. Two years after harvest, an increase in the relative contribution of opportunistic taxa (e.g., *Parametrioctenemus*, *Cricotopus*) was observed. The third year post-harvest showed the highest compositional shift, characterized by a marked dominance of *Cricotopus*, whereas four years after harvest the assemblage exhibited a partial reorganization, with increased representation of genera such as *Pentaneura* and *Rheotanytarsus*. In grassland streams, although overall compositional patterns remained relatively stable through time, a marked increase in

**Table II. Total abundance and relative frequency of Chironomidae in streams draining grasslands, afforested, and harvested pine plantations in the Comechingones Mountains. Total abundance is presented as mean±standard error.**

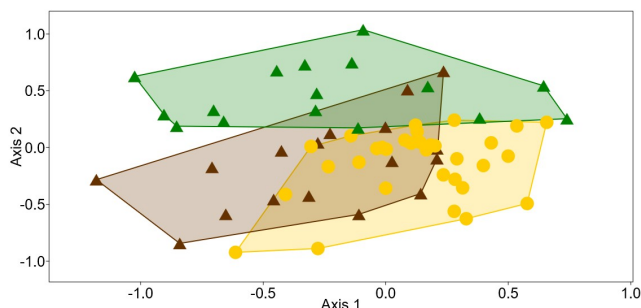
|                               | Grassland streams |      | Afforested streams |      | Harvested streams |      |
|-------------------------------|-------------------|------|--------------------|------|-------------------|------|
|                               | Abund             | Freq | Abund              | Freq | Abund             | Freq |
| <b>Subfam. Podonominae</b>    |                   |      |                    |      |                   |      |
| <i>Podonomus</i> mfsp1        | 635±383           | 0.39 | 1±1                | 0.06 | 35±21             | 0.53 |
| <i>Podonomus</i> mfsp2        | 218±101           | 0.58 | 701±306            | 0.75 | 144±140           | 0.18 |
| <i>Parochlus</i> sp.          | 3±3               | 0.03 | 0                  | 0    | 0                 | 0    |
| <b>Subfam. Tanipodinae</b>    |                   |      |                    |      |                   |      |
| Macropelopiini                | 7±3               | 0.21 | 913±766            | 0.63 | 120±56            | 0.65 |
| <i>Labrundinia</i> sp.        | 2±1               | 0.15 | 0                  | 0    | 0                 | 0    |
| <i>Larsia</i> sp.             | 10±5              | 0.18 | 11±8               | 0.13 | 12±5              | 0.29 |
| <i>Pentaneura</i> sp.         | 421±122           | 0.88 | 64±52              | 0.38 | 176±55            | 0.94 |
| <i>Thienemannimyia</i> group  | 19±8              | 0.24 | 0                  | 0    | 6±3               | 0.29 |
| <i>Pentaneurini</i> indet.    | 36±13             | 0.36 | 204±97             | 0.88 | 41±16             | 0.41 |
| <b>Subfam. Chironominae</b>   |                   |      |                    |      |                   |      |
| <i>Polypedilum</i> sp.        | 65±30             | 0.45 | 10±10              | 0.06 | 20±9              | 0.47 |
| <i>Pseudochironomus</i> sp.   | 183±79            | 0.73 | 51±23              | 0.50 | 14±11             | 0.24 |
| <i>Rheotanytarsus</i> sp.     | 2833±1790         | 0.88 | 92±81              | 0.38 | 153±62            | 0.65 |
| <i>Tanytarsus</i> sp.         | 71±28             | 0.64 | 315±182            | 0.63 | 83±29             | 0.59 |
| <b>Subfam. Orthocladinae</b>  |                   |      |                    |      |                   |      |
| <i>Corynoneura</i> sp.        | 759±215           | 0.94 | 1018±448           | 0.94 | 110±33            | 0.76 |
| <i>Cricotopus</i> sp.         | 622±113           | 0.94 | 117±56             | 0.63 | 950±409           | 0.76 |
| <i>Nanocladius</i> sp.        | 59±37             | 0.30 | 8±6                | 0.13 | 3±3               | 0.06 |
| <i>Onconeura</i> sp.          | 37±26             | 0.21 | 6±3                | 0.19 | 9±4               | 0.24 |
| <i>Orthocladus</i> sp.        | 245±240           | 0.12 | 7±7                | 0.06 | 33±29             | 0.12 |
| <i>Parametrioctenemus</i> sp. | 1034±554          | 0.97 | 752±300            | 1.00 | 646±429           | 0.94 |
| <i>Thienemanniella</i> sp.    | 24±7              | 0.45 | 3±2                | 0.13 | 5±3               | 0.12 |

the relative abundance of *Rheotanytarsus* was observed in one sampling year, indicating temporal fluctuations in dominant taxa without a clear successional trend (Fig. 3).



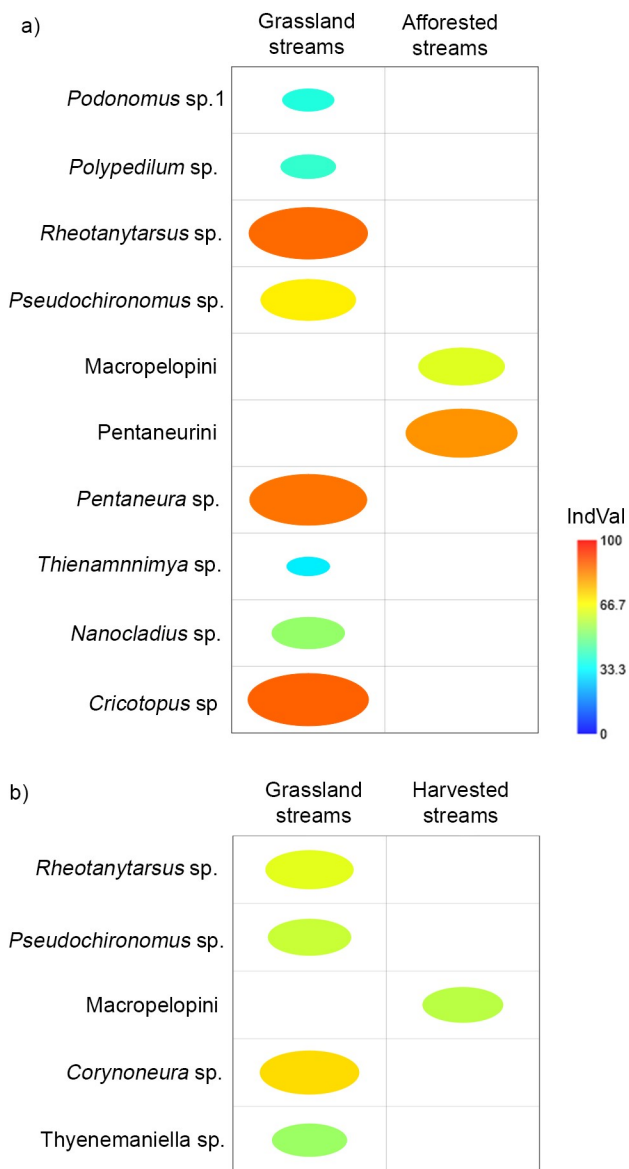
**Figure 3.** Heat maps showing the mean relative abundance of Chironomidae taxa across sampling years in pine plantation streams and grassland streams. In pine plantation streams, assemblages are shown for pre-harvest conditions and successive years post-harvest (2-4 years). In grassland streams, assemblages are shown for equivalent sampling years. Taxa numbers correspond to those shown in Fig. 2.

The ordination plot (Fig. 4) revealed clear patterns in Chironomidae assemblage composition across land-use types. The first four axes accounted for 54.9 % of the total variation in the data (Eigenvalues, Axis 1: 0.172, Axis 2: 0.142, Axis 3: 0.122 and Axis 4: 0.076). Axis 2 of the correspondence analysis clearly separated the assemblages from streams draining standing pine plantations from those in harvested pine and native grassland streams. Additionally, there was a tendency for assemblages from grassland streams to plot on the right side of Axis 1, in contrast to those from both standing and harvested pine plantations, which were primarily positioned on the left. These ordination patterns were statistically supported by a PERMANOVA test, which detected significant differences in Chironomidae assemblage composition among land-use types ( $F = 6.247$ ,  $p = 0.0001$ , 9999 permutations).



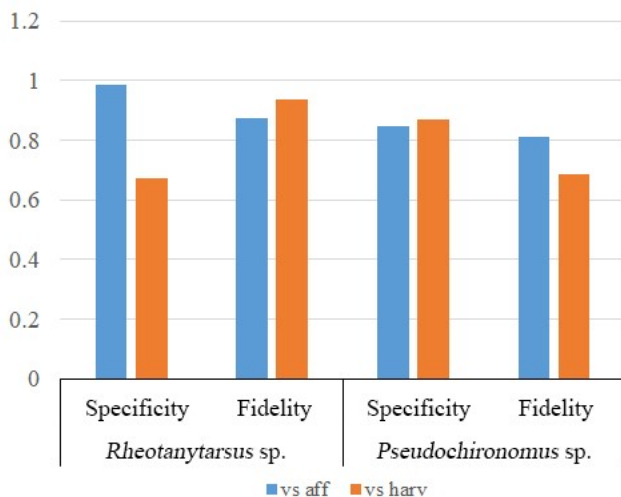
**Figure 4.** CA ordination of chironomid samples from grassland (yellow circles), pine (green triangles) and harvested (brown triangles) streams.

Grassland streams supported the highest number of indicator species, both when compared with streams flowing through standing pine plantations and those draining recently harvested pine areas (Fig. 5). *Cricotopus*, *Pentaneura* Phillip, and *Rheotanytarsus* showed the highest indicator values for grassland sites, whereas the tribe Pentaneurini (excluding *Pentaneura*) was most strongly associated with standing pine



**Figure 5.** Graphical representation of the results of the indicator species analysis (IndVal method) for Chironomidae at the genus level ( $p > 0.05$ ). a. comparison between grassland and afforested stream assemblages. b. comparison between grassland and harvested plantation stream assemblages.

plantations. *Rheotanytarsus* and *Pseudochironomus* were consistently identified as indicators of grassland streams in both comparisons, although their indicator values declined when grasslands were contrasted with harvested pine areas. This decline was linked to reduced specificity in *Rheotanytarsus* and lower fidelity in *Pseudochironomus* (Fig. 6). The tribe Macropelopiini was associated with both standing and harvested pine conditions. Within Pentaneurini, a clear pattern emerged: *Pentaneura* was restricted to grasslands, while other members of the tribe were indicative of pine plantations.



**Figure 6.** Bar plot comparing the specificity and fidelity of *Rheotanytarsus* sp. and *Pseudochironomus* sp. for grassland streams when contrasted with afforested (aff) and harvested (harv) streams.

## DISCUSSION

This study shows that the identification of Chironomidae at a lower taxonomic level is effective for distinguishing between streams with different types of riparian vegetation. This finding is consistent with prior studies suggesting that increased taxonomic resolution within this family enhances the capacity to detect ecological responses to environmental disturbances (Nicacio & Juen, 2015; Serra et al., 2017; Molineri et al., 2020). Knowledge of this family is still fragmentary; however, the number of regional identification keys is increasing (e.g., Ruiz-Moreno et al., 2000; Prat et al., 2011; Zanutto Arpellino et al., 2022). The difficulty of identification and some remaining knowledge gaps are often the reason why other taxa are selected to evaluate the ecological state of river systems (Spies et al., 2009; Anderson & Ferrington Jr., 2013), leaving aside identification at lower taxonomic levels of Chironomidae despite the valuable information that may be overlooked.

Several genera among the most abundant, such as *Rheotanytarsus*, *Parametriocnemus*, *Cricotopus* and *Pentaneura*, were shared between harvested and grassland streams. In contrast, forested streams showed a different pattern, with taxa such as *Tanytarsus* and members of the tribe Macropelopiini among the most abundant, although these were not as prevalent in the other two stream types. These findings reflect differences in community composition among stream types. Pine plantations alter several stream environmental characteristics, such as the increased input of leaf litter from pine plantations (Molinero & Pozo, 2004) might influence the distribution of functional groups across stream types (Martinez et al., 2013; Larrañaga et al., 2023). However, most of the genera observed in the rank-abundance curves correspond to collector-gatherers or

collector-filterers (Merritt & Cummins, 2008; Principe et al., 2010; Galizzi et al., 2012), without clear differences between grassland and forested environments. The accumulation of plant debris in forested or harvested streams reduces streamflow (Jobbágy et al., 2013). In this context, several members of the tribe Pentaneurini, identified as indicators of forested streams, are known to inhabit slower-flowing river sections (Paggi, 2003).

The temporal patterns observed in chironomid community composition suggest that assemblages respond differently over time depending on land-cover type and land-use history, consistent with previous studies (Saulino et al., 2021; Milner et al., 2024) and supporting their use as sensitive bioindicators of environmental change. In forested streams, the initial dominance of Macropelopiini and *Corynoneura* under pine cover shifted, two to three years after harvesting, toward a higher relative contribution of opportunistic taxa such as *Cricotopus*, commonly associated with disturbed conditions and high colonization capacity (Félix dos Anjos & Takeda, 2005). This compositional change appeared to intensify over time rather than immediately after harvesting, likely reflecting progressive changes in resource availability, light and thermal regimes, and habitat structure (Dan Moore et al., 2005; Griffith & Kiffney, 2022). Four years post-harvest, assemblages showed partial reorganization, with increased representation of *Pentaneura* and *Rheotanytarsus*, suggesting an incipient recovery of functional complexity without full return to pre-harvest conditions. In contrast, grassland streams exhibited relatively stable composition with interannual fluctuations in dominant taxa such as *Rheotanytarsus*, a pattern consistent with systems lacking strong directional disturbances, where chironomid communities vary mainly in response to natural environmental variability (Garay et al., 2020; Pero et al., 2023). This contrast reinforces the interpretation that changes in forested streams are primarily associated with post-harvest dynamics rather than natural environmental variation.

The observed differences in taxonomic composition among the different stream types are consistent with the results of the multivariate analyses, which showed that assemblages from pine forests were separated from those in grassland streams, while those from harvested pine streams partially overlapped with grassland assemblages. A possible explanation for these patterns lies in the pronounced microclimatic alterations in streams under pine plantations, where light incidence can decrease by up to 75 %, significantly reducing aquatic plant richness (Márquez et al., 2025) and periphyton biomass (Cibils-Martina et al., 2017). These changes in habitat structure and resource availability likely contribute to the differences observed in Chironomidae community composition.

Numerous taxa were identified as indicators of grassland streams, where representatives of all four families present in

the study area and at least three different feeding strategies were recorded (filtering collector, predator and gathering collector) (Principe et al., 2010; Galizzi et al., 2012). In contrast, forested and harvested streams showed a reduced number of indicator taxa, mostly belonging to the subfamily Tanytopodinae, whose members are predominantly predators (Principe et al., 2010; Galizzi et al., 2012). Interestingly, and contrary to expectations, despite the increased input of organic matter caused by pine plantations, many taxa that rely on this resource, both in its fine and coarse particulate forms, were found to be more abundant and even served as indicators of grassland streams. Some members of the subfamily Chironominae have even been reported as facultative shredders (Galizzi et al., 2012). This pattern may be explained by the fact that the dominant input from pine plantations-pine needles-has a very low decomposition rate (Webster & Benfield, 1986), due to their thick cuticle and the presence of resins, which can act as inhibitory substances for decomposers and detritivores (Bärlocher & Oertli, 1978a, b). Additionally, experimental studies conducted in the study area have shown that pine needles do not function as a food resource but can serve as a novel habitat (Márquez et al., 2017).

Identifying which taxon or taxa are good indicators, either because they are abundant, ecologically key, or ecosystem engineers in a particular stream type, can facilitate the evaluation of ecosystem recovery success (Sundermann et al., 2011; Feijoó, 2021). In this regard, our study highlights two taxa of particular importance for monitoring streams in the study area: *Rheotanytarsus* and *Pseudochironomus*, both of which were strongly associated with highland grassland streams. Furthermore, when evaluating their indicator performance between pine plantation and harvested conditions, their indicator values declined when grasslands were contrasted with harvested pine areas. This decline was linked to reduced specificity in *Rheotanytarsus* and lower fidelity in *Pseudochironomus*. These changes may reflect ecological alterations following harvest, such as increased habitat heterogeneity or transitional conditions.

It is important to stress the importance of conducting local studies on the impact of disturbances on aquatic systems, given the important role that water plays in many societal processes, enabling the agricultural, industrial and urban development of human populations (Aguirre et al., 2023). It is also important to mention that this type of study at the local scale is necessary not only to observe the current state of aquatic biodiversity, but also as a tool for resource management and monitoring over time (Bagla, 2014; Brito & Magalhães, 2017; Castrillo & García, 2020).

Finally, it is important to acknowledge that similar research questions applied to the same taxonomic group in other regions of the world may yield different results from those presented here (e.g., Puntí et al., 2009; Lencioni et al., 2012; Rossaro et al., 2022). This may be due to differences in the

resilience of river systems, the intensity and nature of disturbances, or various regional environmental factors (Heino et al., 2015). These considerations highlight the importance of developing site-specific knowledge of intervention contexts, in order to obtain accurate interpretations of ecological responses and to design monitoring tools that are both relevant and effective for the local setting avoiding the loss of critical information in the process.

## COMPETING INTERESTS

The authors have declared that no competing interests exist.

## AUTHORS CONTRIBUTIONS

All authors contributed equally to all stages of the research, including fieldwork, laboratory analyses, data interpretation, and the preparation of the manuscript.

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