No Short-Term Effects of Riparian Harvesting on Aquatic Macroinvertebrates

A Case Study from Boreal Headwaters in Northern Sweden

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Master’s thesis • 30 credits
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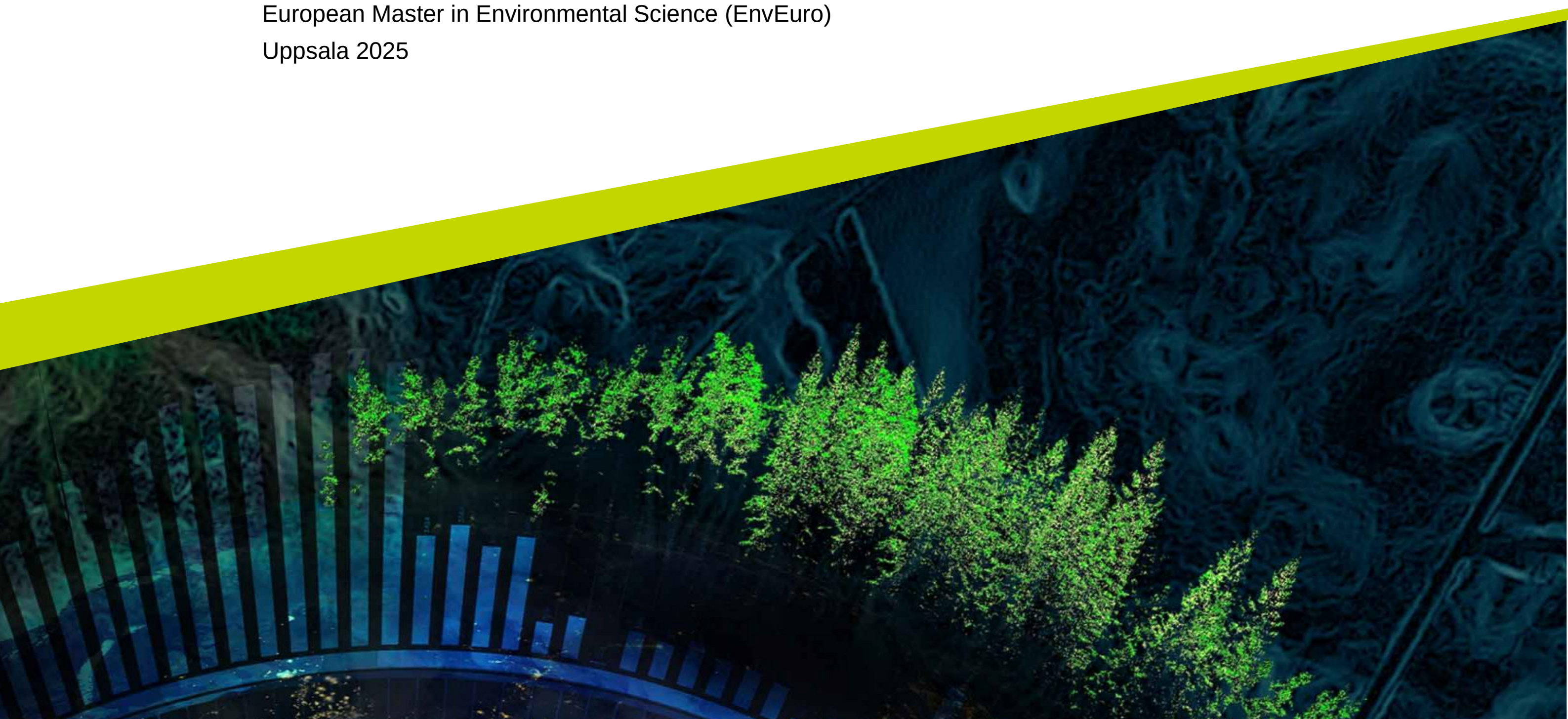
European Master in Environmental Science (EnvEuro)
Uppsala 2025



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Credits:	30 credits
Level:	Second cycle, A2E
Course title:	Master thesis in environmental science
Course code:	EX0897
Programme/education:	EnvEuro - European Master in Environmental Science
Course coordinating dept:	Department of Aquatic Science and Assessment
Place of publication:	Uppsala
Year of publication:	2025
Copyright:	All featured images are used with permission from the copyright owner.
Keywords:	buffer zones, aquatic macroinvertebrates, forest management, riparian forest, short-term ecological response, stream ecology, clear-cutting

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Abstract

Riparian forests serve as critical buffers between terrestrial and aquatic ecosystems, particularly in boreal headwaters where biodiversity is sensitive to land-use disturbances such as forest harvesting. This study investigates the short-term impacts of different riparian buffer designs on the biodiversity and community composition of aquatic macroinvertebrates in seven headwater streams in northern Sweden. Using a Before-After-Control-Impact (BACI) approach, the study compared macroinvertebrate metrics before and after riparian forest harvesting across various treatment and reference sites. Key biodiversity indicators, including total abundance, species richness, and richness and abundance of sensitive EPT taxa (Ephemeroptera, Plecoptera, Trichoptera), were assessed alongside habitat variables such as buffer width, stream substrate composition, and stream size.

The results indicated no statistically significant short-term effects of riparian harvesting on macroinvertebrate community structure across the studied sites, suggesting that the riparian buffer practices – which were distinct in design, exceeding typical standards in terms of width, structural complexity, and were specifically tailored to the characteristics of each site – may be sufficient to maintain ecological integrity in the immediate aftermath of harvesting. Furthermore, buffer width did not significantly correlate with biodiversity metrics, whereas substrate composition, particularly the presence of bryophytes and coarse substrates, showed stronger correlations with macroinvertebrate diversity. Catchment area, as an indicator of stream size, had no significant correlation with biodiversity. These findings highlight the importance of local habitat features over buffer width alone and suggest that ecological responses to forest management may take longer to manifest. Long-term monitoring is recommended to capture delayed or cumulative effects.

Keywords: buffer zones, aquatic macroinvertebrates, forest management, riparian forest, short-term ecological response, stream ecology, clear-cutting

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Abbreviations

BACI	Before-After-Control-Impact
CBP	Current best practice
CI	Confidence interval
DBH	Diameter at breast height
DEM	Digital elevation model
DOC	Dissolved organic carbon
EPT	Ephemeroptera, Plecoptera, Trichoptera
GAP	Buffer with canopy gaps
GIS	Geographic information system
IQR	Interquartile range
LMM	Linear mixed model
NMDS	Non-metric multidimensional scaling
OM	Organic matter
OS	One-sided buffer
VR	Variable retention buffer

1. Introduction

Forested headwater streams are ecologically sensitive systems shaped by interactions between land use, riparian vegetation, and in-stream conditions. This thesis explores how forest harvesting and riparian buffer design influence aquatic macroinvertebrate biodiversity and community composition in boreal Sweden, focusing on short-term ecological responses to forestry interventions and environmental gradients.

1.1 Riparian forests

Riparian forests are transition areas between land and water that occur alongside streams, rivers, and lakes. These dynamic interfaces play a vital ecological role by linking catchment-level processes with in-stream biological and physical functioning (Gundersen et al., 2010; Kominoski et al., 2011). Riparian zones are shaped by fluvial disturbance regimes such as flooding, sediment deposition, and erosion, which interact with vegetation succession and hydrological connectivity. This dynamic nature fosters high spatial and temporal heterogeneity in habitat structure, promoting biodiversity and resilience (Hylander, 2004; Yeung et al., 2017).

Riparian forests are among the most biodiverse components of boreal landscapes, supporting both terrestrial and aquatic species (Hylander, 2006) . Their structural complexity, including multilayered canopies, deadwood, and root systems, provides essential resources for a wide range of taxa. For aquatic ecosystems in particular, riparian forests maintain water quality by filtering sediments and nutrients, buffering temperature extremes through shading, and stabilising stream banks (Lowrance et al., 1984; Richardson & Béraud, 2014). Litter input from riparian vegetation also sustains detritus-based food webs, linking terrestrial productivity to aquatic consumers (Kominoski et al., 2011).

These forested riparian corridors are especially important in headwater streams, which constitute the majority of the river network in boreal regions such as Sweden. Swedish headwater streams are typically small, shallow, and shaded, with low pH and high concentrations of dissolved organic carbon (DOC), reflecting their drainage through podzolic soils and peat-rich landscapes (Jonsson et al., 2017). Despite their size, these streams play a disproportionately large role in biodiversity conservation, water purification, and nutrient cycling. Moreover, they support specialised macroinvertebrate communities that are sensitive to environmental change and therefore serve as effective indicators of ecosystem integrity (Frainer & McKie, 2015).

However, the structure and function of riparian forests are increasingly threatened by intensive land-use practices, particularly forestry. In Sweden, clear-cutting, ditching, and thinning near streams have significantly altered the composition of riparian vegetation and hydrological regimes, reducing habitat complexity and increasing stream light and nutrient inputs (Erdozain et al., 2018; Ring et al., 2023). Such changes can shift macroinvertebrate communities toward more generalist or tolerant species, reducing ecosystem functions such as litter decomposition and trophic linkages (Erdozain et al., 2019; Frainer & McKie, 2015).

1.2 Forest management

Forest clear-cutting has long been recognised as a significant contributor to biodiversity loss, particularly in the riparian zones of headwater streams, which are highly sensitive to anthropogenic disturbances. Clear-cutting can increase sedimentation, alter water quality, and disrupt habitat complexity, negatively impacting aquatic macroinvertebrate communities and other components of biodiversity (Chellaiah & Kuglerová, 2021; R. K. Johnson et al., 2017). In response to these pressures, riparian buffers are often implemented as a mitigation strategy to reduce sedimentation and preserve aquatic habitats. However, the effectiveness of these buffers is highly dependent on both their width and their management practices (Chellaiah & Kuglerová, 2021).

Research demonstrates that narrow buffers (<10 m) are often inadequate for preserving biodiversity and safeguarding stream ecosystems, whereas wider buffers (>15 m) are more likely to sustain essential ecological functions such as erosion control, nutrient retention, and the provision of shaded, complex habitats for aquatic organisms. In Sweden, national guidelines emphasise six core functions of riparian buffers, including biodiversity preservation, erosion prevention, and nutrient cycling, while not specifying the required width of the buffers. Consequently, buffers in Sweden have an average width of 7-10 m. while not prescribing how wide buffers should be. As an outcome, buffers in Sweden are on average 7-10 m wide (Kuglerová et al., 2019; Ring et al., 2023).

Long-term studies show that selective thinning near headwater streams alters light availability and water chemistry, with consequences for macroinvertebrate assemblages (Ring et al., 2023). These effects include shifts in species composition and a reduction in functional diversity, which can impact ecosystem processes such as litter decomposition (Frainer et al., 2021; Ring et al., 2023). Disturbance from forestry also changes the availability of coarse woody debris and leaf litter, influencing the basal resources available to aquatic consumers (Erdozain et al., 2019; Kominoski et al., 2011).

Despite the broad use of riparian buffers, there is a surprising lack of studies that explicitly compare the ecological outcomes of different buffer styles, such as one-sided buffers, buffers with canopy gaps, or variable-width buffers. Most existing research focuses on buffer presence or general width thresholds, rather than on structural variation within the buffer itself. Yet, emerging studies suggest that these design elements can strongly influence ecological effectiveness. For instance, Chan et al. (2004) demonstrated that variable-density riparian management enhances structural heterogeneity and habitat quality, while Johnson et al. (2022) found divergent macroinvertebrate responses depending on riparian treatment type, including shifts in dominant taxa and functional groups. These findings point to substantial—yet understudied—variation in buffer performance depending on design. Given this knowledge gap, further research is needed to explore how specific buffer configurations may influence aquatic communities.

In addition to buffer characteristics, other environmental factors such as streambed substrate and stream size are known to strongly influence macroinvertebrate communities, yet they are often overlooked in riparian management research. Substrate composition affects habitat structure, flow refugia, and resource availability, and plays a key role in determining macroinvertebrate diversity and functional group distribution (Brown & Brussock, 1991; Frainer & McKie, 2015). Streams dominated by stable, coarse substrates like pebbles or cobbles generally support more diverse and sensitive assemblages than those with high proportions of fine sediment, which can reduce oxygen availability and disrupt interstitial habitats (Sundermann et al., 2011; Yeung et al., 2017). Similarly, stream size—often approximated by catchment area—influences biodiversity via its effects on flow stability, nutrient transport, and habitat heterogeneity (Allan & Castillo, 2007; Johnson et al., 2017). Larger catchments tend to exhibit more stable hydrological regimes and greater habitat complexity, which can support richer macroinvertebrate communities. Although most riparian studies focus on buffer effects alone, there is growing recognition that local and landscape-scale factors interact to shape biodiversity outcomes (Verdonschot et al., 2012). Considering physical habitat features such as streambed structure and stream size alongside buffer design is important for a better understanding of how forest harvesting influences biodiversity in stream ecosystems.

1.3 Macroinvertebrates and the EPT Group as Bioindicators

Aquatic macroinvertebrates function as pivotal bioindicators of water quality and overall ecosystem integrity. These invertebrates are the most frequently used organisms in the biological monitoring of freshwater ecosystems globally. Their widespread distribution, relative immobility, ease of sampling, short life cycles, and broad range of tolerances to environmental stressors make them highly effective indicators (Birk et al., 2012; Bonada et al., 2006). They respond rapidly and predictably to changes in water chemistry, hydrology, nutrient enrichment, sedimentation, temperature, and physical habitat structure, making them valuable tools for detecting both acute and chronic disturbances (Bonada et al., 2006;

Frainer & McKie, 2015). Consequently, macroinvertebrates are central to ecological assessments under frameworks such as the European Union's Water Framework Directive (WFD), which incorporates biological quality elements to determine stream health (Birk et al., 2012).

Riparian zones strongly influence the diversity and structure of macroinvertebrate communities. Intact riparian forests regulate stream temperature and light, stabilise banks, and supply organic matter, all of which support complex aquatic habitats (Kominoski et al., 2011; Sargac et al., 2021). Degradation of riparian buffers through forestry practices or inadequate protection increases sediment loads and stream temperatures and reduces detrital inputs, leading to loss of sensitive species and homogenisation of community structure (Frainer et al., 2021; Ring et al., 2023).

Macroinvertebrates in intensively managed forests have shown increased reliance on terrestrial-derived carbon, reflecting changes in riparian shading and reduced autochthonous production (Erdozain et al., 2019). These trophic shifts demonstrate how land use affects stream food web structure and underscore the role of riparian buffers in maintaining aquatic–terrestrial linkages (Erdozain et al., 2018; Kominoski et al., 2011).

Among aquatic macroinvertebrates, the EPT group—Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)—is especially valuable for biomonitoring due to its high sensitivity to environmental change (Birk et al., 2012; Bonada et al., 2006). These taxa prefer well-oxygenated, clean, and cool waters and are typically the first to decline under stress from pollutants, sedimentation, or thermal load (Jonsson et al., 2017). The richness and relative abundance of EPT taxa are commonly used to evaluate stream ecological integrity and have been closely linked to riparian forest structure and land-use intensity (Sánchez-Montoya et al., 2010; Sargac et al., 2021; Frainer & McKie, 2015). Thus, changes in riparian forest condition can trigger cascading effects on invertebrate communities and associated ecosystem functions, such as leaf litter decomposition and nutrient cycling (Erdozain et al., 2019; Frainer et al., 2021).

1.4 Aims and Research Questions

Riparian forest and adjacent streams are a highly sensitive ecosystem, and their biodiversity is particularly vulnerable to anthropogenic disturbances such as forest clear-cutting. While riparian buffers are commonly implemented to mitigate such impacts, a comprehensive understanding of how different buffer designs and management practices influence aquatic macroinvertebrate communities remains lacking. This thesis aims to address this knowledge gap by analysing the effects of various riparian buffer strategies on macroinvertebrate biodiversity across multiple stream sites in northern Sweden. The main research questions were:

1. How did aquatic macroinvertebrate biodiversity and community composition change from 2022 (pre-harvest) to 2023 (post-harvest) across treatment and reference sites?

To address this question, biodiversity metrics, including total abundance, species richness, and EPT (Ephemeroptera, Plecoptera, Trichoptera) richness and abundance, as well as community composition, were evaluated across all study sites. Biodiversity indicators were derived from samples collected using Surber samplers and analysed using mixed-effect models within a Before-After-Control-Impact (BACI) framework. This design compares ecological data from before and after a disturbance at impacted sites, alongside unimpacted reference sites, to distinguish changes caused by the disturbance from natural background variation (Green, 1979).
Community structure was further explored using non-metric multidimensional scaling (NMDS) and PERMANOVA to assess changes in species assemblages over time.

2. How do biodiversity responses correspond to buffer width?

To investigate this question, biodiversity metrics were examined in relation to riparian buffer width across harvested sites. Buffer width was quantified at each research plot using high-resolution (20 cm) drone imagery and GIS-based measurements, taken from both sides of the stream and averaged per plot. This continuous variable was then compared to indicators such as total abundance, species richness, and EPT richness and abundance to assess whether wider buffers were associated with greater ecological integrity. The aim was to evaluate the extent to which buffer width alone could explain variation in macroinvertebrate biodiversity following forest harvesting.

3. How does biodiversity relate to bottom substrates and stream size before and after harvest?

This question focused on the potential role of physical habitat characteristics in shaping the composition of macroinvertebrate communities. Substrate composition was assessed from benthic samples and categorised into distinct material types. Catchment area was used as a proxy for stream size, which can influence biodiversity through mechanisms such as greater flow permanence, enhanced habitat complexity, and increased resource inputs. Associations between these environmental factors and biodiversity metrics were examined through correlation and linear modelling to evaluate their contribution to observed ecological responses across different buffer treatments.

The thesis aims to contribute to a deeper understanding of buffer effectiveness in mitigating biodiversity loss, with implications for enhancing riparian management practices, particularly in regions where headwater streams are highly vulnerable to anthropogenic disturbances.

2. Materials and methods

This thesis was conducted as part of the MUST DEFINE project, led by the thesis supervisor. The project aims to understand what riparian buffer practices best provide ecological functions, including biodiversity protection, as listed in the national guidelines. Throughout the project, various environmental variables were collected, including ground vegetation, leaf litter, soil samples, temperature, light intensity, humidity, sediment characteristics, tree inventories, dead wood, and benthic invertebrates. For the purpose of this thesis, the primary variables of interest were benthic invertebrates, substrate samples, the size of the catchment area and buffer width. Due to time limitations associated with the scope of a master's thesis, I did not participate in the fieldwork or data collection. The dataset was provided to me by the thesis supervisor. My contribution consisted of conducting the statistical analyses, interpreting the results, comparing the findings with existing literature, and preparing the written thesis.

2.1 Study area and project structure

This study was conducted in the boreal forest landscapes of northern Sweden. To assess the effects of different forest management strategies on riparian buffer zones and their adjacent aquatic ecosystems, seven study sites (see Table 1) were selected in boreal forest landscapes (Figure 1). These comprised a variety of experimental buffer treatments and control conditions, enabling robust comparisons. All treatment sites were planned in collaboration with the land owner - SCA forest company - and implemented with specific ecological objectives and with improved designs compared to the standard fixed-width 7-10 m wide buffer (Ring et al., 2023). Each treatment approach is described in detail below.

Of the total of 7, four sites underwent forest clear-cutting during the winter of 2022-2023, two served as untreated reference sites (streams in mature production forest stands), and one acted as a negative reference (a stream in a clear-cut harvested in 2020, without a riparian buffer). Sampling occurred before (summer-autumn 2022) and after (summer-autumn 2023) clear-cutting, enabling a temporal analysis of biodiversity changes. Benthic invertebrates were collected using five Surber samples per stream, with a focus on riffles or faster-flowing sections, to provide detailed data on species richness, abundance, and functional group composition.

Table 1. Overview of study sites with treatment type, harvest timing, and buffer design

Site	Treatment Type	Harvest Timing	Buffer Design & Width
GAP	Buffer with Canopy Gaps	Winter 2022–23	≥10 m on both sides with 3 canopy gaps (~20 m each)
CBP	Current Best Practice	Winter 2022–23	7–10 m variable-width buffer
OS	One-Sided Buffer	Winter 2022–23	~20.3 m average (10 m no-harvest + 10 m partial)
VR	Variable Buffer	Winter 2022–23	Variable width: some >20 m, some <5 m
Ref20	Reference (Control)	None	Intact riparian zone
Kryk Ref	Reference (Control)	None	Intact riparian zone
Kryk Neg	Negative Reference	Harvested 2020	Minimal buffer (only scattered trees)

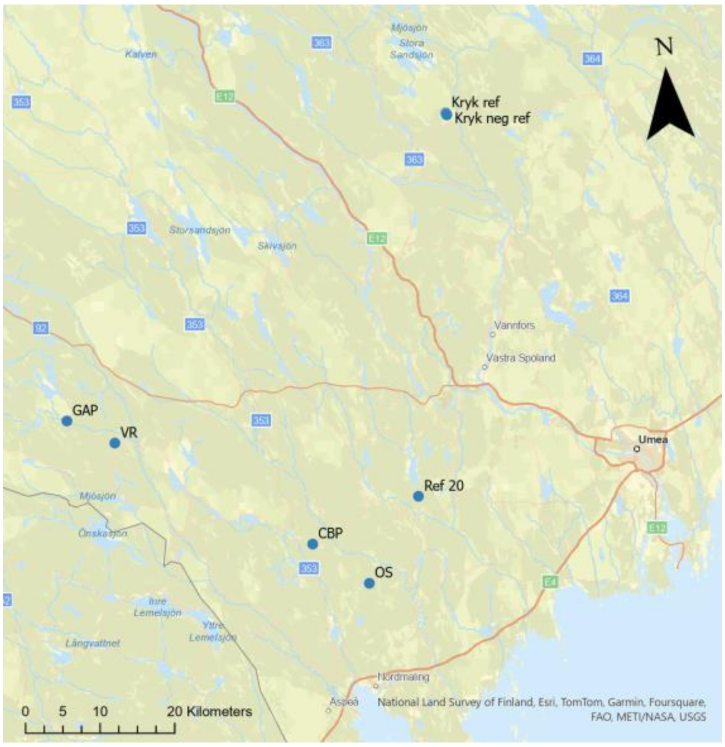

Figure 1. Overview map of all study sites in the Västerbotten region, northern Sweden. The map shows the geographic locations of all research sites included in the study. Treatment sites are marked as GAP (buffer with canopy gaps), VR (variable buffer), CBP(current best practice), and OS (one-sided buffer), while reference sites are labelled as Ref 20, Kryk ref (Kryklan reference), and Kryk neg ref (Kryklan negative reference).

Buffer with Canopy Gaps (GAP)

The site incorporated an intact riparian buffer of at least 10 metres on both sides of the stream. Within this buffer, three distinct canopy gaps, each approximately 20 metres in length, were created by removing trees down to the water's edge. The selection of trees was focused on those with a diameter exceeding 8 centimetres at breast height (DBH), predominantly spruce, to simulate localised disturbances and enhance light penetration into the riparian floor and stream channel (see Figure 2A).

Current Best Practice (CBP)

The Current Best Practice site featured a variable-width buffer ranging from 7 to 10 metres along the stream, designed to balance ecological objectives with operational and economic feasibility. Although formally labelled “Current Best Practice” within the project, the treatment reflects commonly applied riparian buffer widths in Swedish forestry, which are generally aligned with national recommendations rather than empirically validated as ecologically optimal (see Figure 2B).

One-Sided Buffer (OS)

This stream reach had previously (2018) undergone clear-cutting on its south-western bank, where only a narrow 5-metre strip of riparian vegetation remained. To compensate for these effects, an enhanced retention zone was established on the opposite bank, harvested within this project. This zone consisted of a 10-metre no-harvest buffer adjacent to the stream, followed by a 10-metre outer zone where partial harvesting was conducted. It is estimated that approximately 50% of the trees in this outer strip were removed, resulting in an average retained buffer width of 20.3 metres across the site. (see Figure 3A).

Variable Buffer (VR)

At this site, the riparian buffer was designed using a spatial soil moisture model, which informed variable-width retention based on local hydrological conditions. In areas of greater moisture and groundwater discharge, wider buffers were left intact, while harvesting operations extended closer to the stream in drier zones. In specific locations, the buffer zone exceeded 20 metres in width; in others, trees were removed from areas near the stream. In several areas along the streams, the slope of the riparian bank prevented harvesting, and there the riparian forest is intact as far as 80 m away from the stream (see Figure 3B).

Control and Reference Sites

Three reference sites were included in order to capture a gradient of riparian buffer integrity. Two of these served as control sites (Reference 20 and Kryklan reference), where no harvesting has been carried out for at least 60 years. However, these streams are situated in production stans. These sites thus provided a baseline for evaluating the ecological effects of the harvest interventions. Conversely, the third site (Kryklan negative) functioned as a negative control, representing a previously logged area (2020) with minimal riparian retention,

consisting only of single trees left. This site, which exhibited a significantly reduced buffer zone, was selected as a model to illustrate the ecological implications of worst-case-scenario buffer retention (see Figure 4B).
The following maps illustrate the geographical layout of the study sites and the spatial distribution of research plots for each buffer treatment and reference condition. Research plots, identified by numbered markers, were systematically established along each stream reach. Macroinvertebrate sampling was conducted on a subset of these plots - five plots per stream. The maps also highlight variations in buffer width and forest structure, which are critical to understanding the context of the ecological effects of each treatment. Maps showing treatment sites after harvesting can be found in Appendix 3.

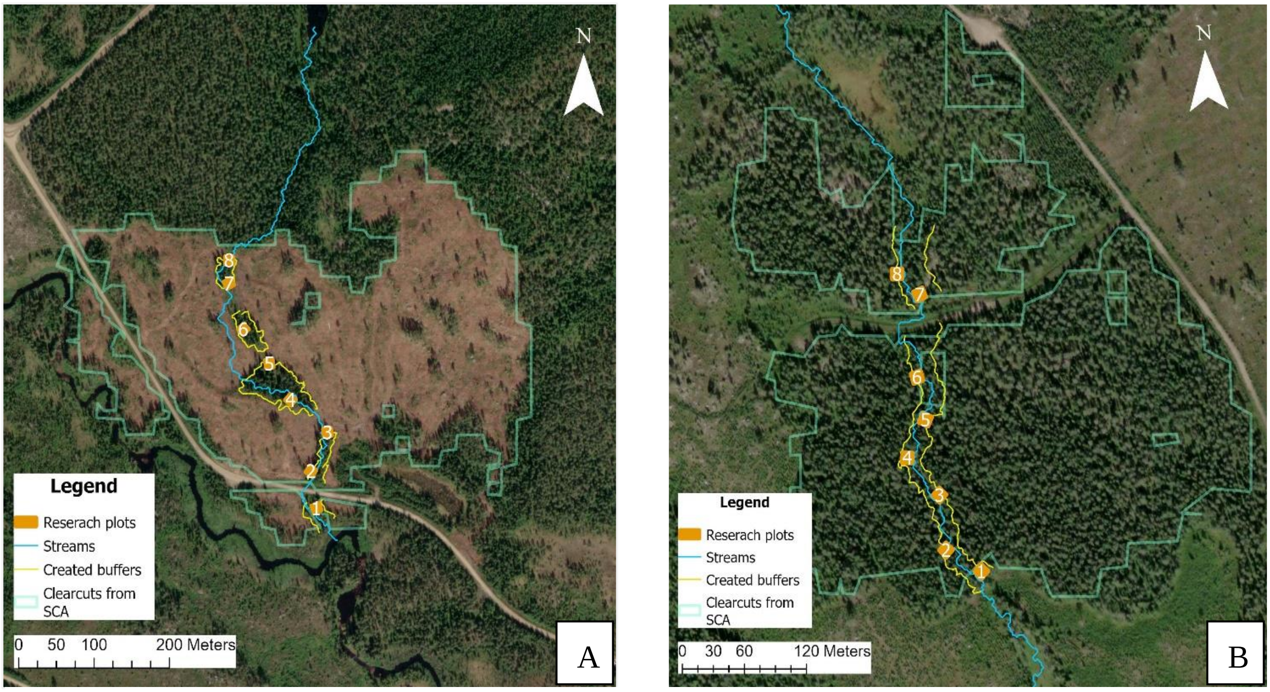

Figure 2. Location of research plots in the GAP (A) and CBP (B) treatment areas. For the CBP site (Map B), a post-harvest aerial image was not available; therefore, a pre-harvest image is shown. The forest within the green boundary labelled as "Clearcuts from SCA" has already been harvested, although this is not visible in the image (Source: Esri, Maxar, Earthstar Geographics, and the GIS User Community).

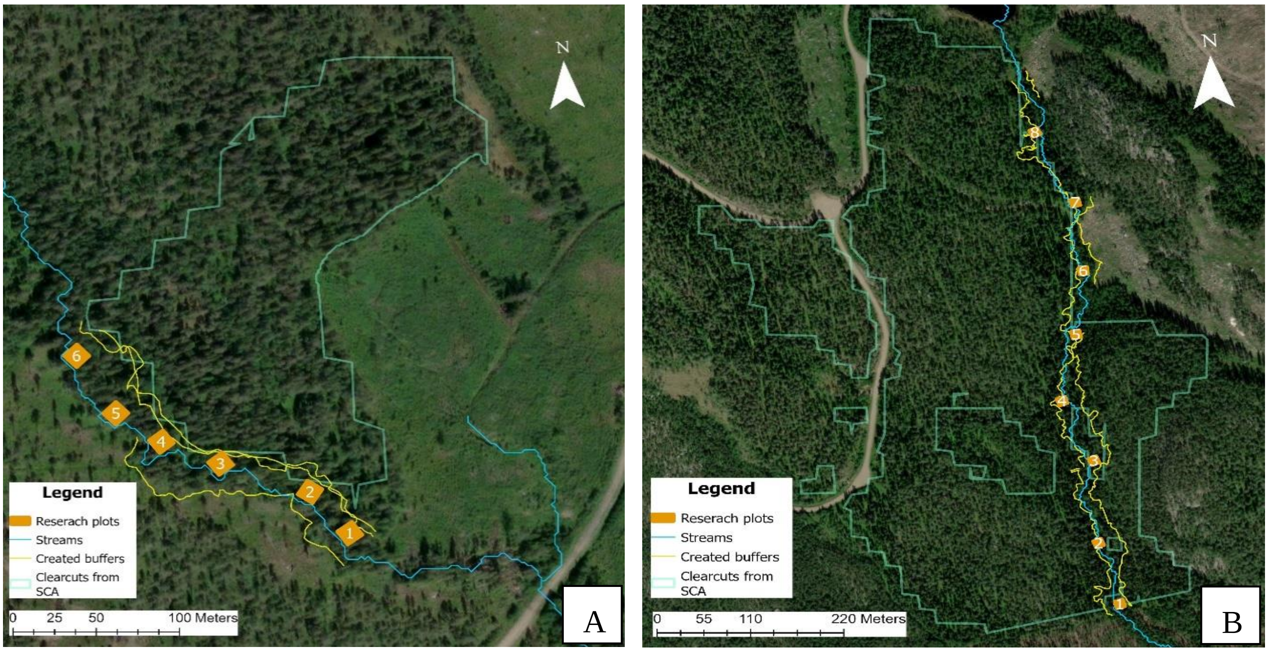

Figure 3. Location of research plots in the OS (A) and VR (B) treatment areas. For both sites, a post-harvest aerial image was not available; therefore, a pre-harvest image is shown. The forest within the green boundary labelled as "Clearcuts from SCA" has already been harvested, although this is not visible in the image. (Source: Esri, Maxar, Earthstar Geographics, and the GIS User Community)

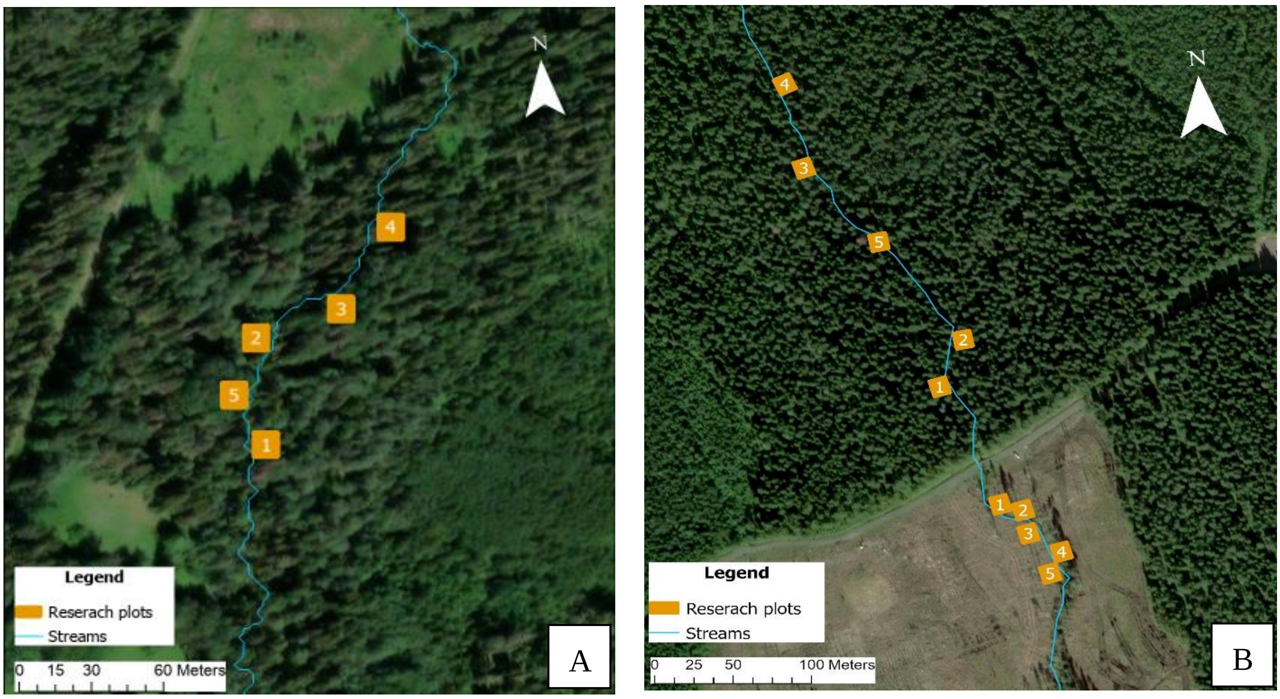

Figure 4. Reference sites used in the study. Map A shows the Reference 20 site, and map B shows the Kryklan reference site in the forested area (upper part) and the Kryklan negative reference site in the harvested area (lower part) (Source: Esri, Maxar, Earthstar Geographics, and the GIS User Community).

2.2 Field sampling

Macroinvertebrate samples were collected using the Surber sampler method (Surber, 1970) to investigate spatial and temporal changes in community structure. The Surber sampler is a quantitative sampling device designed for collecting benthic macroinvertebrates from stream bottoms, particularly in riffle areas with a strong current. It consists of a square brass frame, one foot in size, with a hinged net that captures organisms dislodged from the substrate. The sampling process involves positioning the sampler securely on the streambed, removing larger stones while rinsing them to collect adhering organisms, and allowing the current to carry displaced fauna into the net. The collected material is then transferred to a pail for further processing, including decantation and sieving, to isolate macroinvertebrates for laboratory analysis (Surber, 1970). At each stream site, five Surber samples were collected and treated as independent replicates within that site. These were used to assess within-site variability in macroinvertebrate metrics and to support site-level analyses such as linear mixed models and ordination methods.

2.3 Lab analysis

Following field collection, all samples obtained using the Surber sampler were preserved in 70% ethanol and transported to the laboratory for detailed analysis. Each sample corresponded to a specific plot and was kept isolated to preserve spatial resolution.

2.3.1 Macroinvertebrate Sorting and Identification

Identification and sorting were performed in a certified SLU lab in Uppsala. In the lab, macroinvertebrates were separated from inorganic material and detritus using sieving and decantation techniques. The samples were first washed through a mesh sieve (typically 0.5 mm) to retain organisms and coarser debris. The residue was then transferred to a Petri dish and examined under a stereomicroscope.

Macroinvertebrates were sorted manually from the debris and identified to the lowest taxonomic level possible (typically family or genus) using general identification keys by (Nilsson, 1996, 1997), complemented by more specialised literature for specific taxonomic groups (eg. Brooks et al., 2007; Edington & Hildrew, 1981)

Abundances were recorded separately for each taxon and site. Data were entered into structured datasets prepared for subsequent statistical analysis of biodiversity indices and community composition.

2.3.2 Substrate Composition Analysis

In addition to macroinvertebrate identification, the composition of substrate material retained in the Surber sampler was analysed and quantified. Each substrate sample was visually assessed and categorised into the following classes:

- Whole Leaves/Needles
- Bryophytes
- Fragments of Organic Matter
- Very Fine Organic Matter
- Very Fine Mineral Material (<0.5 mm)
- Sand and Gravel (0.5–15 mm)
- Pebbles (15–64 mm)

The material from each sample was manually separated into these classes and, if needed, dried for easier handling. For each sample, the relative abundance of each category was then estimated visually as a percentage of the total substrate volume present in the sampler. This proportion-based approach was used instead of raw counts or weights because total substrate volume varied among samples. Using percentages allowed for standardised comparisons of substrate composition across samples and sites, and these data were subsequently used to evaluate how variation in habitat structure correlates with macroinvertebrate metrics.

2.4 Buffer width analysis

Buffer width was measured using GIS drone images from post-harvest flights with a resolution of 20 cm. Drone images were overlaid with a stream layer and a research plot layer, and aligned using georeferencing if necessary. At each research plot (shown in Figure 2, Figure 3 and Figure 4) a distance measure tool in ArcGIS Pro (Esri, 2025) was used to measure the total width of the buffer across both sides of the stream. The width was measured at each side of the research plot.

For the analyses, the total buffer width from the two measurements at each plot was averaged. One plot at the VR site was excluded from this analysis because it was located on a steep riverbank where no harvesting was conducted (hence, the buffer width was not assessed).

2.5 Catchment area analysis

The catchment area was calculated using the Whitebox GAT (version 3.3) analyses. A national digital elevation model (DEM) with 2 m resolution was used to calculate flow accumulation, from which the size of the catchment area (in km2) was derived. The catchment area is used here as a proxy for stream size, as larger catchments generally collect more water and support broader hydrological regimes. Catchment size may influence macroinvertebrate diversity by affecting factors such as discharge volume, habitat heterogeneity, and organic matter input.

2.6 Statistical Analysis

The original data tables were first prepared and cleaned in Excel. Metrics were organised so that each sample was represented with the relevant variables, and the table was then saved as a .csv file and imported into R for analysis. All further data processing and statistical analyses were performed using R version 4.5.0 (R Core Team, 2024) within the RStudio environment (Posit Team, 2024).

The dataset was then filtered to create subsets for four macroinvertebrate metrics: total abundance, species richness, EPT richness (Ephemeroptera, Plecoptera, Trichoptera), and EPT abundance.

2.6.1 Changes in Biodiversity and Community Composition Before and After Harvest

To answer the first research question, linear mixed-effects models (LMMs) were applied to individual Surber samples (n = 5 per site per year), preserving the full sampling resolution. Models were constructed using the lme4 package (Bates et al., 2015) to test the effects of year, treatment type, and harvest status on macroinvertebrate metrics. Site was included as a random intercept to account for repeated measures, while year and treatment were treated as fixed effects.

Model comparisons were made using the anova () function to assess whether including interaction terms improved model fit. Model significance and effect sizes were evaluated using Type III ANOVA via the car (Fox & Weisberg, 2019) and lmerTest (Kuznetsova et al., 2017) packages. Visual inspection of results was supported by ggplot2 (Wickham, 2016) , with boxplots used to illustrate spatial and temporal trends in community metrics. To enhance visual clarity and highlight overall patterns, boxplots were generated by grouping all control and all treatment sites together.

To complement this analysis, multivariate techniques were applied to assess changes in community composition. The macroinvertebrate community data were first cleaned to exclude empty samples and then subjected to non-metric multidimensional scaling (NMDS) based on Bray-Curtis dissimilarities using the vegan package (Oksanen et al., 2022). As with the univariate models, each point in the ordination represented a single Surber sample. NMDS ordinations were carried out both at the site level and across all sites combined. Groups were defined by harvest status (before vs. after), and variation within each year was visualised using ordihull(). To statistically test for differences in community composition, PERMANOVA (adonis2) was applied with 999 permutations. The independent variable in these models was harvest status (before or after), and the models were first run for all sites combined. However, the results indicated strong variation across sites, and therefore, ordination analyses were also run separately for each site, using the five Surber samples as replicates.

2.6.2 Effects of Riparian Buffer Width on Biodiversity

To answer the second research question, separate linear mixed models (LMMs) were constructed for each macroinvertebrate metric, with buffer width as a fixed effect and site as a random intercept. The input data remained at the Surber sample level, and the analyses were carried out only for sites that were harvested—that is, the four treatment sites with buffers and the negative reference site where buffer width is zero. Results were visualised through scatterplots with linear trends superimposed using geom_smooth() in ggplot2 (Wickham, 2016). A table containing all buffer width and macroinvertebrate metric values used in this analysis is provided in Appendix 2.

2.6.3 Correlations with Substrate Composition and Catchment Area

To answer the third research question, Spearman's rank correlation coefficients were calculated to assess whether variation in the proportion of specific substrate types (e.g. bryophytes, coarse sediments) corresponded with differences in macroinvertebrate metrics. This analysis also used Surber sample-level data, with each row representing an individual sample and its associated substrate composition and macroinvertebrate metrics. Substrate categories included organic matter fragments, bryophytes, fine material, and coarse mineral material, all aggregated from raw variables using the dplyr (Wickham et al., 2023) and tidyr (Wickham & Girlich, 2023) packages. Correlations were computed separately for each site and harvest status combination using nested functions with map_dfr() from the purrr package (Henry & Wickham, 2020) to determine whether the relationship between substrate composition and biodiversity was already present before harvesting and whether it changed after the intervention.

Correlations between macroinvertebrate diversity and stream size, approximated by catchment area in square kilometres, were also assessed to complement this analysis. For this part only, the five Surber samples per site and year were aggregated into one composite data point, producing site-level summaries. Total abundance and EPT abundance were calculated by summing values across the five samples. At the same time, species richness and EPT richness were derived by merging species lists from all samples and removing duplicate taxa, ensuring each species was counted only once.

The final table (see Appendix 4) was then imported into R and transformed into a wide format, with biodiversity metrics as individual columns. The dataset was split into two subsets by year (2022 and 2023), and a custom function was written using the purrr package (Henry & Wickham, 2020) to calculate Spearman's rank correlation coefficients between each biodiversity metric and catchment area. This function reported the correlation coefficient (ρ) and the associated p-value for each metric, handling missing data.

3. Results

Results are structured to reflect key ecological dimensions: temporal changes across sites, variation in community composition, the role of substrate characteristics, and the influence of buffer width. Each section highlights observed trends and statistical outcomes, shedding light on the ecological responses to riparian buffer interventions.

3.1 Temporal Changes in Macroinvertebrate Biodiversity Across Sites

To address the first part of the first research question, temporal changes were analysed across sites using biodiversity metrics analysis. Boxplots showing the same metrics without grouped buffers (i.e., displayed by individual site) can be found in Appendix 5.

Abundance

There was no significant effect of the interaction between treatment type (control vs. treatment) and year (estimate = –10.62, t = –0.258, p = 0.796), indicating that neither control nor treatment sites exhibited a consistent change in abundance between 2022 (pre-harvest) and 2023 (post-harvest). The individual effect of year was also not statistically significant (estimate = 56.77, t = 1.351, p = 0.1772), and the effect of treatment type alone was likewise non-significant (estimate = 66.74, t = 0.718, p = 0.2349).

As shown in Figure 5, median macroinvertebrate abundance remained similar between control and treatment sites in 2022, with a visible increase in 2023, particularly at treatment sites. However, variation in abundance across sites and years, as reflected in the wide interquartile ranges and presence of outliers, suggests natural spatial and temporal heterogeneity. No clear pattern emerged that would link these differences directly to treatment application, and none of the observed differences were statistically significant.

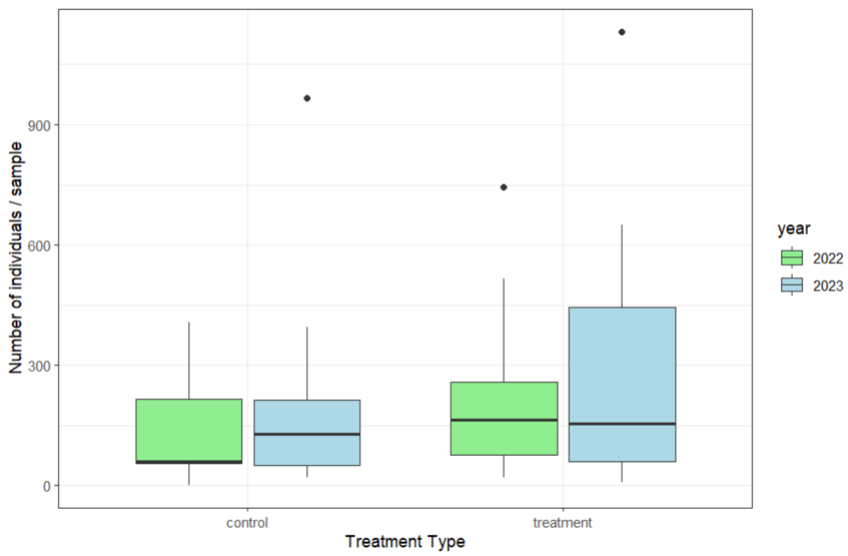

Figure 5. Boxplot showing the number of macroinvertebrate individuals per sample at treatment and control sites in 2022 (pre-harvest, green) and 2023 (post-harvest, blue). Control sites include all reference streams (Ref20, RefKryck, RefNegative), while treatment sites include those with active management interventions (CBP, GAP, OS, VR). The horizontal line within each box represents the median; boxes indicate the interquartile range (IQR), whiskers extend to the smallest and largest values within 1.5 times the IQR from the quartiles, and black dots indicate statistical outliers..

Species richness

There was no significant effect of year on species richness (estimate = 1.86, t = 1.551, p = 0.1259), and the effect of treatment was also not statistically significant (estimate = 2.55, t = 0.813, p = 0.4532), as shown in
Figure 6, median species richness was slightly higher at treatment sites than control sites in both years. A modest increase from 2022 to 2023 was visible in both treatment and control groups, though overlapping interquartile ranges and data variability suggest these trends are not statistically meaningful.

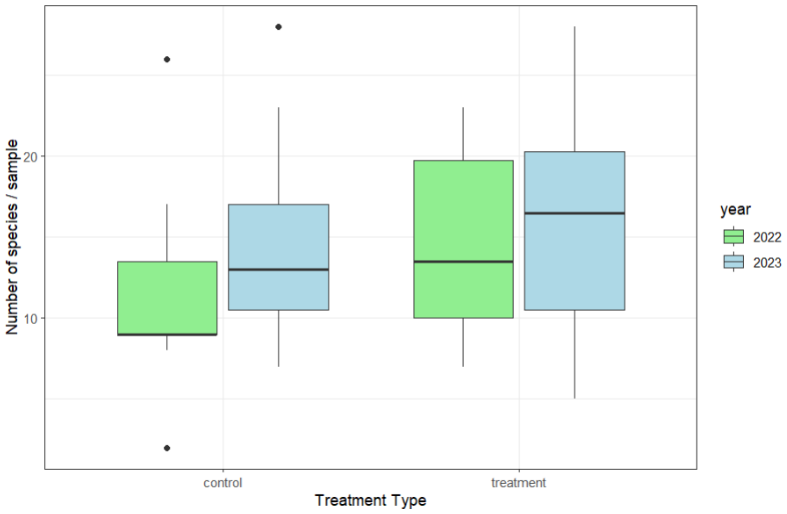

Figure 6. Boxplot showing species richness (number of macroinvertebrate taxa per sample) at control and treatment sites in 2022 (pre-harvest, green) and 2023 (post-harvest, blue). Control sites include reference streams (Ref20, RefKryck, RefNegative), and treatment sites include those where active management occurred (CBP, GAP, OS, VR). The black line within each box indicates the median, boxes represent the interquartile range (IQR), whiskers extend to values within 1.5× the IQR, and black dots denote statistical outliers.

EPT Richness

The richness of EPT taxa (Ephemeroptera, Plecoptera, Trichoptera) did not differ significantly between years or treatment types. No significant effect of treatment was found (estimate = 0.8583, t = 0.670, p = 0.5324), and the effect of year was also non-significant (estimate = 0.3143, t = 0.533, p = 0.5959). On average, samples contained 5.2 EPT taxa in 2022 and 5.5 in 2023. As shown in Figure 7 median EPT richness appeared slightly higher in treatment sites compared to control sites in both years, and a modest increase was observed from 2022 to 2023 across both site categories. However, the distribution of values, including overlapping interquartile ranges and several outliers, indicates high variability and no consistent directional trend.

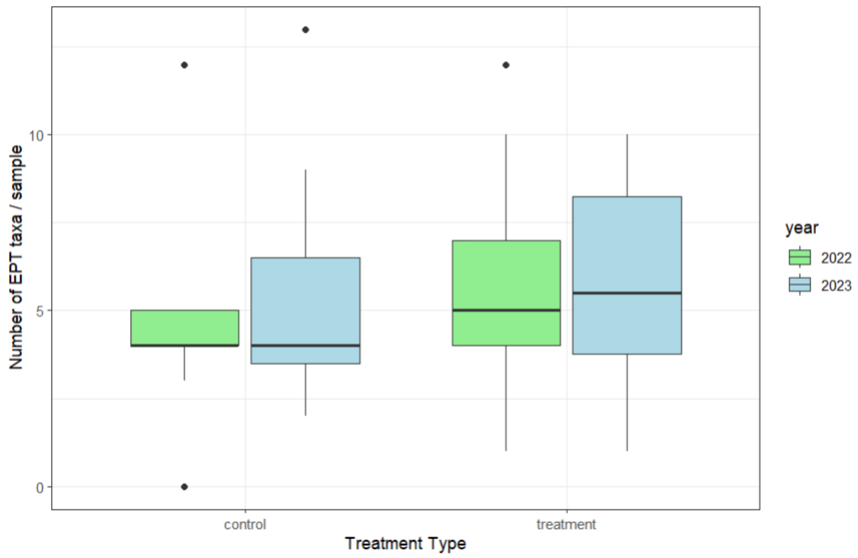

Figure 7. Boxplot showing the number of EPT taxa (Ephemeroptera, Plecoptera, Trichoptera) per sample at control and treatment sites in 2022 (pre-harvest, green) and 2023 (post-harvest, blue).Control sites include reference streams (Ref20, RefKryck, RefNegative), and treatment sites include those with active management (CBP, GAP, OS, VR).The black line within each box indicates the median, boxes represent the interquartile range (IQR), whiskers extend to values within 1.5× the IQR, and black dots denote statistical outliers.

EPT Abundance

The species richness of EPT taxa was not significantly affected by either year (estimate = 0.3143, t = 0.533, p = 0.5959) or treatment (estimate = 0.8583, t = 0.670, p = 0.5324), as shown in Figure 8. The average number of EPT taxa per sample was 5.2 in 2022 and 5.5 in 2023.

Although median values were slightly higher in treatment sites compared to control sites across both years, and a modest increase was observed from 2022 to 2023, these differences were not statistically significant.

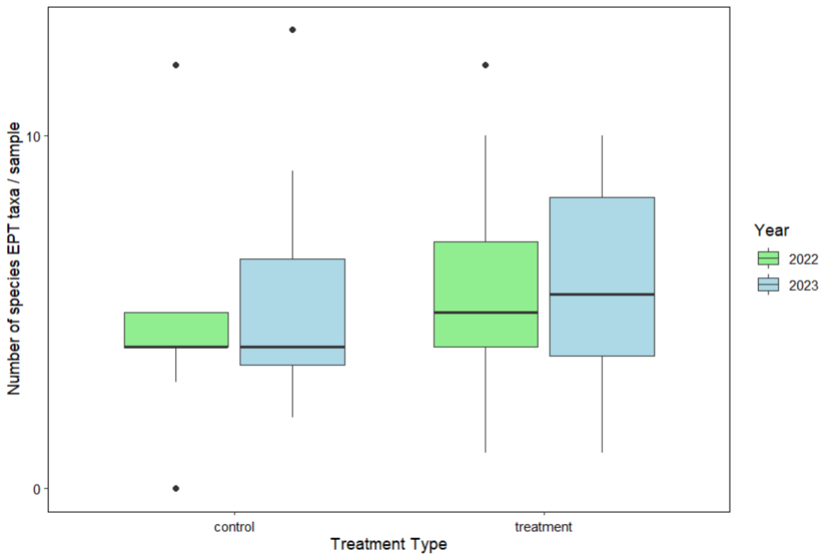

Figure 8. Boxplot showing the number of species from the EPT group (Ephemeroptera, Plecoptera, Trichoptera) per sample at control and treatment sites in 2022 (pre-harvest, green) and 2023 (post-harvest, blue). Control sites include reference streams (Ref20, RefKryck, RefNegative), and treatment sites include managed locations (CBP, GAP, OS, VR). The horizontal black line within each box marks the median; boxes represent the interquartile range (IQR), whiskers extend to values within 1.5 times the IQR, and black dots indicate statistical outliers.

3.2 Temporal Variation in Community Composition Across Sites

To address the second part of the first research question, which concerned the overall temporal variation in aquatic invertebrate communities between 2022 and 2023, non-metric multidimensional scaling (NMDS) ordination was initially performed on all study sites combined. The resulting ordination plot (Figure 9) visualises the community composition in reduced dimensional space based on Bray-Curtis dissimilarity, with ellipses enclosing samples from each year. Although the ellipses for 2022 and 2023 show some spatial divergence, their substantial overlap indicates high variability across sites, which may mask meaningful temporal patterns.

The test did not reveal a significant effect of year ($F = 0.791$, $R^2 = 0.012$, $p = 0.684$), indicating that the variation in community structure attributable to year was minimal when all sites were combined. Due to the large variation among sites and the low explanatory power of the harvest factor, the data were subsequently analysed at the individual site level.

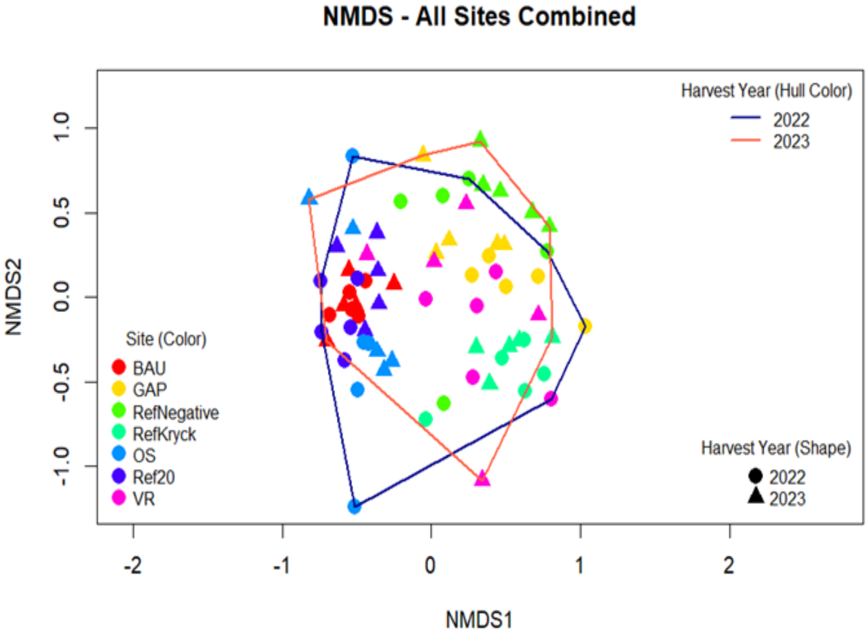

Figure 9. Non-metric Multidimensional Scaling (NMDS) ordination plot displaying the invertebrate community composition for all sites combined (GAP, BAU, OS, VR, Ref20, RefKryck, and RefNegative) across two sampling years (2022 and 2023). Each point represents a single Surber sample (n = 5 per site per year), and ellipses enclose samples from each year to illustrate temporal grouping based on Bray-Curtis dissimilarity.

Separate NMDS ordinations for treatment sites are shown in Figure 10 including GAP, BAU, OS, and VR. The ordinations revealed generally overlapping community structures between 2022 and 2023 for most sites, indicating temporal stability in macroinvertebrate composition. The ellipses representing each year largely overlapped at BAU, OS, and VR, suggesting little change in community structure between the two sampling periods. At GAP, however, the NMDS plot indicated a more precise separation between years, with distinct clusters forming for 2022 and 2023.

Despite these visual patterns, PERMANOVA results indicated that none of the observed differences were statistically significant. The test results were as follows: GAP ($F = 0.7401$, $R^2 = 0.085$, $p = 0.597$), BAU ($F = 1.5102$, $R^2 = 0.159$, $p = 0.233$), OS ($F = 0.9059$, $R^2 = 0.102$, $p = 0.459$), and VR ($F = 0.6983$, $R^2 = 0.080$, $p = 0.668$).

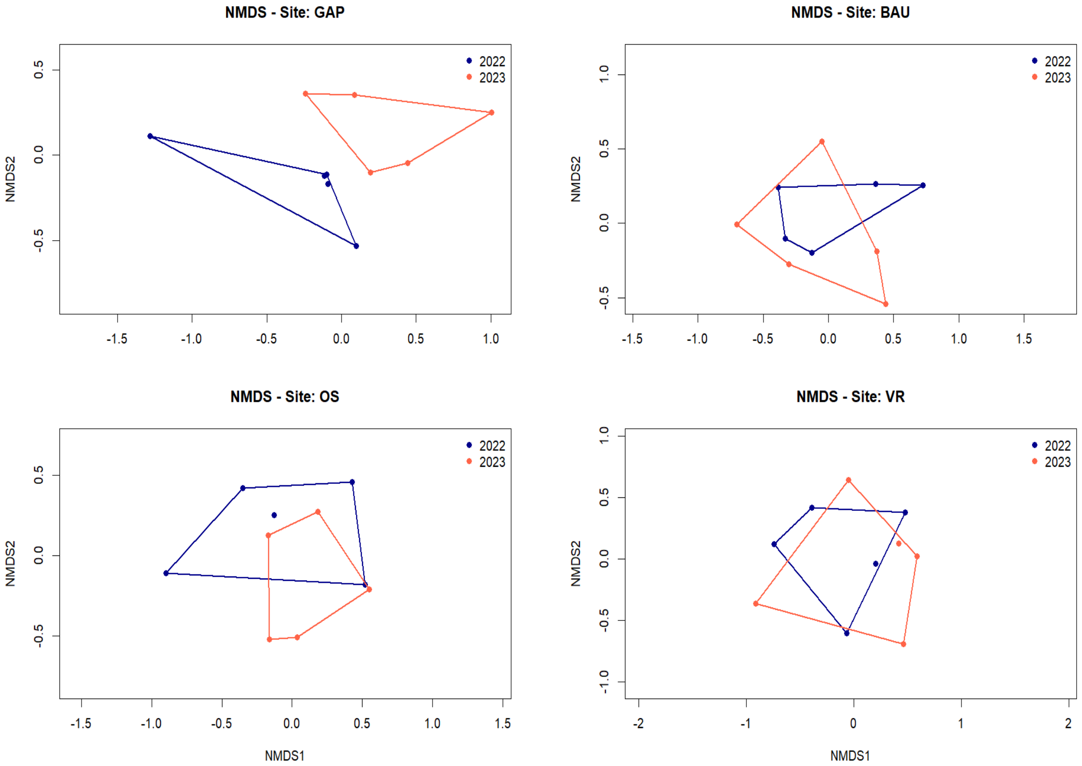

Figure 10. Non-metric Multidimensional Scaling (NMDS) ordination plots displaying the invertebrate community composition at four treatment study sites (GAP, BAU, OS, and VR) across two sampling years (2022 and 2023). Each point represents one Surber sample (five samples per site per year), and the ordinations are based on Bray–Curtis dissimilarity. Ellipses illustrate the dispersion of samples within each year to indicate potential temporal differences in community structure.

The results of the NMDS ordination for control sites are shown in Figure 11. Among these, Ref20 was the only site to show a statistically significant change in community composition between years, as indicated by PERMANOVA ($F = 2.3217$, $R^2 = 0.225$, $p = 0.048$). The NMDS plot for this site reflects the results with clearly separated year clusters, possibly indicating the influence of external environmental factors or natural variability. In contrast, RefKryck exhibited only partial overlap between years, and the difference was not statistically significant ($F = 0.9798$, $R^2 = 0.109$, $p = 0.397$). Likewise, RefNegative showed some visual separation between 2022 and 2023 samples, but PERMANOVA revealed no significant difference ($F = 0.7822$, $R^2 = 0.089$, $p = 0.587$).

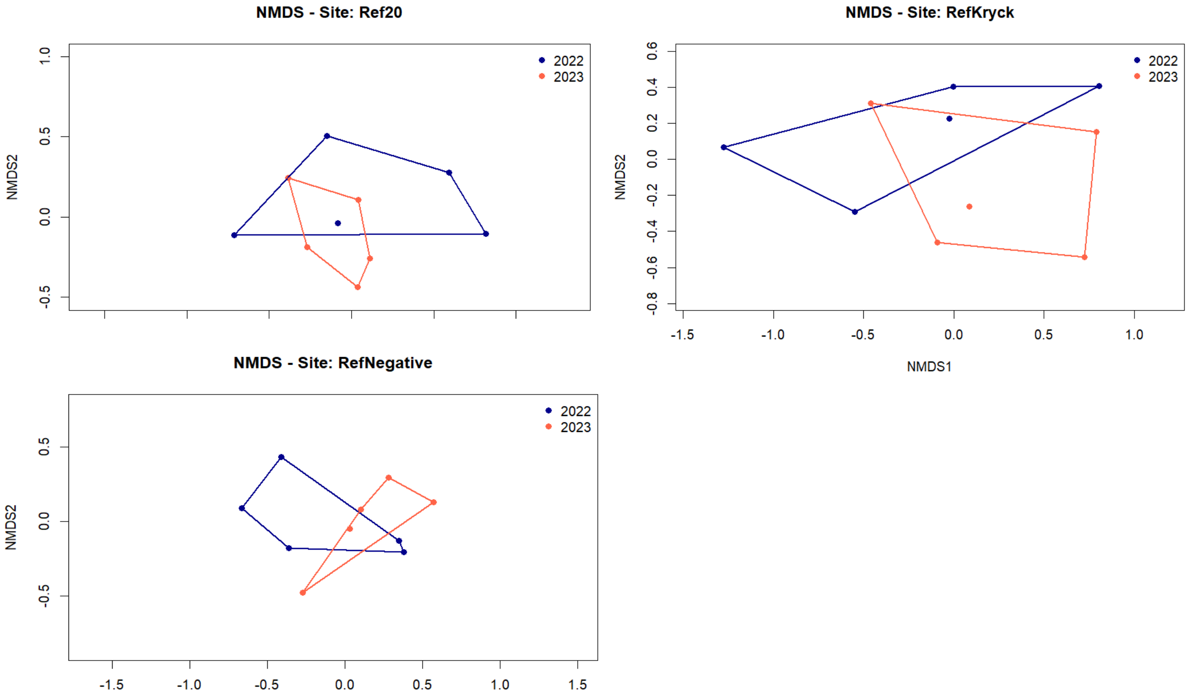

Figure 11. Non-metric Multidimensional Scaling (NMDS) ordination plots displaying the invertebrate community composition at three control study sites (Ref20, RefKryck, and RefNegative) across two sampling years (2022 and 2023). Each point represents one Surber sample (five samples per site per year), and the ordinations are based on Bray–Curtis dissimilarity. Ellipses illustrate the dispersion of samples within each year to indicate potential temporal differences in community structure.

3.3 Effects of Riparian Buffer Width on Macroinvertebrate Community Metrics

To address the second research question, which aimed to evaluate whether the width of riparian buffers is statistically associated with macroinvertebrate community metrics, a series of statistical analyses was conducted. The results consistently showed no statistically significant effect of buffer width on any of the analysed variables.

Taxonomic richness (see Figure 12 on the left) showed no statistically significant association with buffer width (estimate = 0.0097, $t = 0.059$, $p = 0.958$). Similarly, the abundance of macroinvertebrates (see Figure 12 on the right) was not significantly associated with buffer width (estimate = 0.0468, $t = 0.008$, $p = 0.994$).

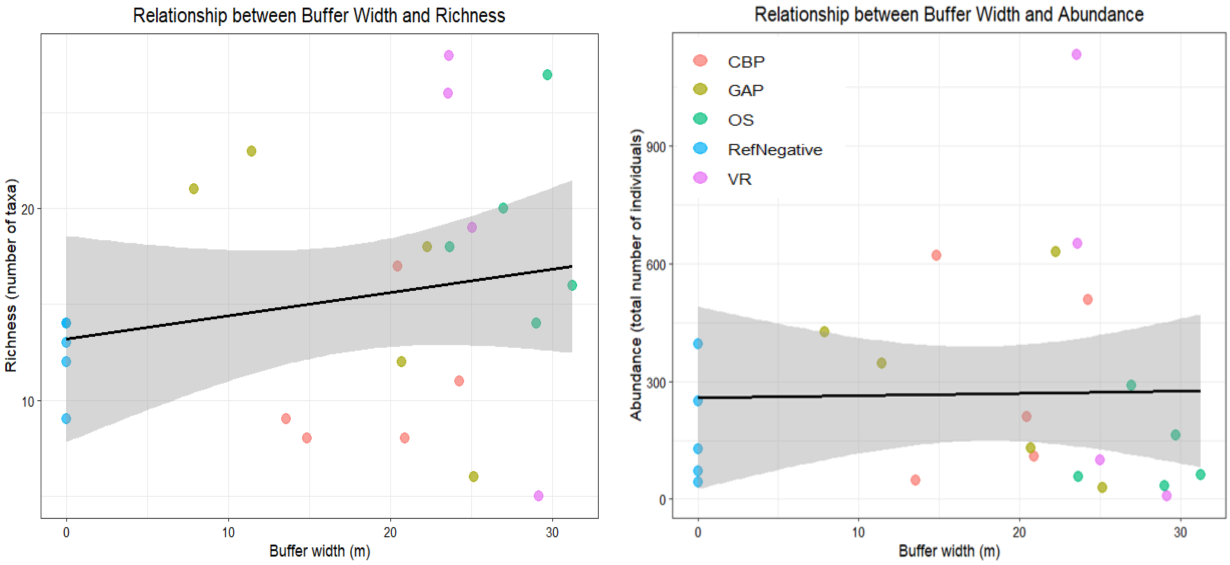

Figure 12. Relationships between buffer width and richness and abundance of macroinvertebrates in 2023 (post-harvest). Each point represents a site-level average based on five Surber samples, with buffer width measured using GIS from drone imagery. Points are coloured according to site type (CBP, GAP, OS, RefNegative, VR). A linear trend line with a 95% confidence interval is included for each metric.

Regarding the EPT taxa, richness (see Figure 13 on the left) exhibited a non-significant negative trend with buffer width (estimate = –0.1701, t = –1.855, p = 0.096), and EPT abundance (see Figure 13 on the right) also showed no statistically significant relationship (estimate = 0.5807, t = 0.394, p = 0.711).

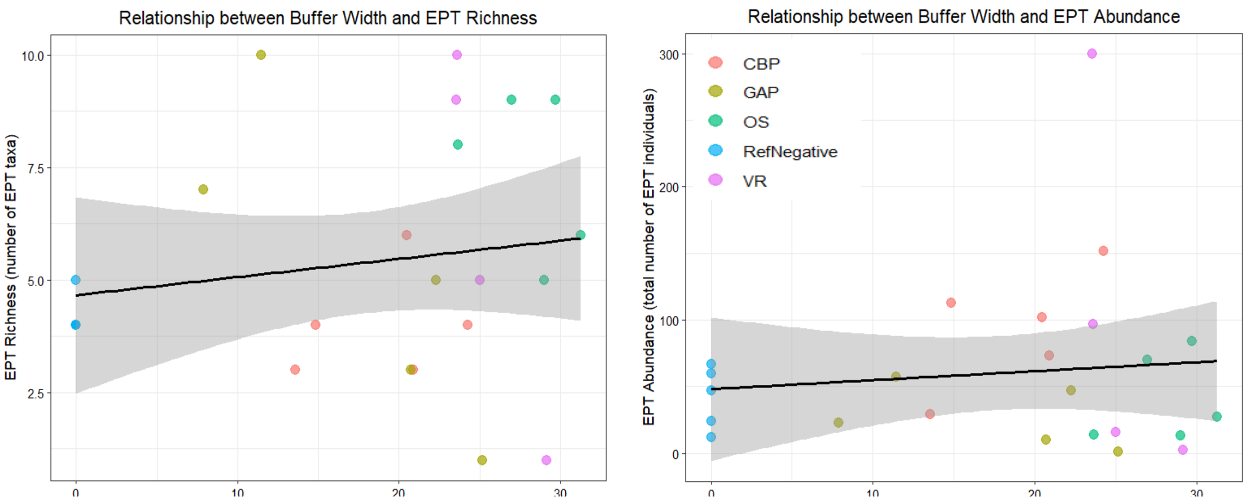

Figure 13. Relationships between buffer width and EPT richness and EPT abundance in 2023 (post-harvest). Each point represents a site-level average based on five Surber samples, with buffer width measured using GIS from drone imagery. Points are coloured according to site type (CBP, GAP, OS, RefNegative, VR). A linear trend line with a 95% confidence interval is included for each metric.

3.4 Influence of Substrate Composition on Macroinvertebrate Biodiversity

To address the first part of the third research question, which focused on the correlation between substrate materials and macroinvertebrate community metrics, the analysis revealed several statistically significant correlations ($p < 0.05$), varying by material type and site. The significant values are discussed below, grouped by substrate type, followed by a summary table (Table 2) highlighting these values. A table with all values is available in Appendix 1.

Bryophytes

There was a positive correlation between the proportion of bryophytes and macroinvertebrate diversity metrics at several sites. At the CBP site in 2022 (pre-harvest), bryophytes were positively correlated with macroinvertebrate abundance ($\rho = 0.900$, $p = 0.037$). At the GAP site in 2022, a significant positive correlation was observed for EPT richness ($\rho = 0.889$, $p = 0.044$). In 2023, a significant positive correlation between bryophytes and EPT abundance was found at the RefKryck site ($\rho = 0.947$, $p = 0.014$). Notably, positive correlations between bryophytes and macroinvertebrate metrics were not observed post-harvest at the CBP and GAP sites in 2023, with RefKryck being the only site where a significant relationship remained.

Fragments of Organic Matter

Fragments of organic matter exhibited both positive and negative correlations depending on site context. At the GAP site in 2022, strong negative correlations were observed with richness ($\rho = –0.975$, $p = 0.005$) and EPT richness ($\rho = –1.000$, $p < 0.001$). In contrast, at the VR site in 2023, fragments of organic matter were positively correlated with abundance ($\rho = 0.975$, $p = 0.005$), EPT richness ($\rho = 0.975$, $p = 0.005$), and EPT abundance ($\rho = 0.975$, $p = 0.005$).

Pebbles

The presence of pebble substrate was positively associated with macroinvertebrate presence. At the CBP site in 2023, significant positive correlations were found with both abundance ($\rho = 0.949$, $p = 0.014$) and EPT abundance ($\rho = 0.949$, $p = 0.014$).

Sand and Gravel

The sand and gravel showed predominantly negative correlations with macroinvertebrate metrics. At the OS site in 2023, a significant negative correlation was found with abundance ($\rho = –0.975$, $p = 0.005$), and at the Ref20 site in 2023, EPT richness was also negatively correlated ($\rho = –0.894$, $p = 0.041$).

Additional negative correlations with abundance were observed at the RefNegative site in 2022 ($\rho = –0.900$, $p = 0.037$) and at the VR site in 2023 ($\rho = –0.900$, $p = 0.037$). Furthermore, at the VR site, sand and gravel showed a negative correlation with EPT abundance in both 2023 ($\rho = –0.900$, $p = 0.037$) and 2022 ($\rho = –0.900$, $p = 0.037$). This pattern is interesting, as most of these negative correlations were observed in post-harvest years, except for EPT abundance at the VR site.

Very Fine Organic Matter

Very fine organic matter exhibited a significant negative correlation with invertebrate metrics at specific locations. At the RefKryck site in 2022, a strong negative correlation was found with abundance ($\rho = –1.000$, $p < 0.001$) and EPT abundance ($\rho = –0.900$, $p = 0.037$). At the RefNegative site in 2022, EPT abundance was also negatively correlated with very fine organic matter ($\rho = –0.894$, $p = 0.041$).

Whole Leaves and Needles

Whole leaves and needles exhibited both negative and positive correlations depending on the site. At the GAP site in 2022, they were strongly negatively correlated with richness ($\rho = –0.949$, $p = 0.014$) and EPT richness ($\rho = –0.973$, $p = 0.005$). Conversely, at the OS site in 2023, a positive correlation was found with EPT abundance ($\rho = 0.900$, $p = 0.037$). Additionally, at the Ref20 site in 2022, abundance was negatively correlated with whole leaves and needles ($\rho = –0.900$, $p = 0.037$), while at the RefNegative site in 2023, richness was positively correlated ($\rho = 0.900$, $p = 0.014$). At the RefNegative site in 2022, EPT abundance was positively correlated ($\rho = 0.900$, $p = 0.037$).

Table 2. Significant Spearman correlation coefficients (ρ) and p-values showing correlations between macroinvertebrate metrics and specific substrate types at various sites and years. Only statistically significant results ($p < 0.05$) are included.

			richness		*abundance*		*EPT richness*		*EPT abundance*	
site	***harvest***	***material***	***sp_rho***	***p_value***	***sp_rho2***	***p_value3***	***sp_rho4***	***p_value5***	***sp_rho6***	***p_value7***
CBP	after	pebbles	0.162	0.794	**0.949**	**0.014**	0.667	0.219	**0.949**	**0.014**
CBP	before	bryophytes	0.103	0.870	**0.900**	**0.037**	-0.224	0.718	0.700	0.188
GAP	before	leaves.needles	**-0.949**	**0.014**	-0.791	0.111	**-0.973**	**0.005**	-0.791	0.111
GAP	before	bryophytes	0.866	0.058	0.577	0.308	**0.889**	**0.044**	0.577	0.308
GAP	before	fragments.om	**-0.975**	**0.005**	-0.821	0.089	**-1.000**	**0.000**	-0.821	0.089
OS	after	leaves.needles	0.700	0.188	0.700	0.188	0.616	0.269	**0.900**	**0.037**
OS	after	sand.and.gravel	-0.667	0.219	**-0.975**	**0.005**	-0.763	0.133	-0.821	0.089
Ref20	after	sand.and.gravel	-0.860	0.061	-0.783	0.118	**-0.894**	**0.041**	-0.112	0.858
Ref20	before	leaves.needles	-0.400	0.505	**-0.900**	**0.037**	-0.671	0.215	-0.205	0.741
RefKryck	after	bryophytes	0.205	0.741	0.564	0.322	0.054	0.931	**0.947**	**0.014**
RefKryck	before	very.fine.om	-0.224	0.718	**-1.000**	**0.000**	-0.316	0.604	**-0.900**	**0.037**
RefNegative	after	leaves.needles	**0.947**	**0.014**	0.718	0.172	-0.181	0.770	0.359	0.553
RefNegative	before	leaves.needles	0.800	0.104	0.600	0.285	0.577	0.308	**0.900**	**0.037**
RefNegative	before	very.fine.om	-0.224	0.718	-0.447	0.450	0.000	1.000	**-0.894**	**0.041**
RefNegative	before	sand.and.gravel	-0.200	0.747	**-0.900**	**0.037**	0.289	0.638	-0.600	0.285
VR	after	fragments.om	0.872	0.054	**0.975**	**0.005**	**0.975**	**0.005**	**0.975**	**0.005**
VR	after	sand.and.gravel	-0.500	0.391	**-0.900**	**0.037**	-0.700	0.188	**-0.900**	**0.037**
VR	before	sand.and.gravel	-0.800	0.104	-0.800	0.104	-0.667	0.219	**-0.900**	**0.037**

3.5 Correlation Between Biodiversity Metrics and Catchment Area

To answer the second part of the third research question, which examined the relationship between macroinvertebrate community metrics and catchment area, Spearman correlation coefficients were calculated separately for the years 2022 and 2023. None of the correlations were statistically significant (all $p > 0.05$). All of the values are presented in Table 3.

Most correlations were positive but weak, except for total abundance and EPT abundance in 2022, which were weakly negative ($\rho = -0.179$ and $\rho = -0.214$, respectively). A few metrics showed moderate positive correlations ($\rho > 0.4$), such as total richness ($\rho = 0.673$), EPT abundance ($\rho = 0.464$), and EPT richness ($\rho = 0.468$) in 2023. However, due to the lack of statistical significance, no reliable correlation between catchment area and macroinvertebrate biodiversity indicators could be confirmed.

Table 3. Spearman correlation coefficients (ρ) and p-values showing the relationship between catchment area and macroinvertebrate biodiversity metrics in 2022 and 2023.

Metric	Spearman_rho	p_value	Year
total abundance	-0.179	0.702	2022
total abundance	0.393	0.383	2023
total richness	0.321	0.482	2022
total richness	0.673	0.0976	2023
total ept abundance	-0.214	0.645	2022
total ept abundance	0.464	0.294	2023
total ept richness	0.43	0.335	2022
total ept richness	0.468	0.289	2023

4. Discussion

This section evaluates the short-term ecological responses of aquatic macroinvertebrate communities to different riparian buffer designs and forest harvesting treatments in boreal headwater streams. The results are interpreted in the context of buffer type, local habitat characteristics, and prior findings from boreal forest ecosystems.

4.1 Ecological Effects of Riparian Harvesting

This study investigated the short-term ecological effects of riparian forest harvesting on macroinvertebrate communities in boreal headwater streams. By applying a BACI (Before-After-Control-Impact) design across multiple treatments, the study evaluated a range of biodiversity metrics, including total abundance, species richness, and the diversity and abundance of EPT taxa. No statistically significant post-harvest changes in macroinvertebrate community structure were detected at treatment sites, regardless of buffer design.

The observed short-term stability of aquatic macroinvertebrate communities suggests high resilience in boreal streams subjected to riparian buffer modifications. While prior research has emphasised the importance of maintaining wide riparian buffers, typically at least 30 m, to mitigate the impacts of forest harvesting (Davies & Nelson, 1994; Kreutzweiser et al., 2005), this study demonstrates that narrower but spatially complex buffers may also effectively preserve stream biodiversity, at least in the initial year following disturbance. This is especially relevant given the nature of the buffer zones used in this study. They were not simple, uniform strips, but were thoughtfully adapted to the specific topography, vegetation, and hydrology of each site. Their design varied even over short distances, including differences in width, tree species, and the presence of retention features. Such site-specific design likely helped maintain ecological functions that support invertebrate resilience. However, it is also noteworthy that in 2023, all reference sites showed a slight increase in species richness, while none of the treatment sites did. Although this trend was not statistically significant, the consistency of this difference may indicate that subtle biological effects of harvesting are beginning to emerge.

It is also important to consider the possibility of delayed responses. One year may not be enough time for macroinvertebrate communities to react to structural and environmental changes following buffer harvesting. Supporting this, complementary microclimate measurements taken at the same sites (Jentzsch, 2023) showed that shading and air temperature in the buffer zones changed significantly following harvest. However, stream water temperature—the variable most critical to aquatic insects (Bonacina et al., 2023)—remained relatively stable across all sites post-harvest. This likely contributed to the limited short-term biological response observed here. Moreover, substantial shifts in macroinvertebrate communities may require multiple years to manifest, particularly as a result of cohort-level dynamics or cumulative habitat changes. Johnson et al., (2022) for example, observed strong post-harvest changes in macroinvertebrate densities, including pulses of chironomids, that became evident only in the second year after logging. These findings underscore the importance of multi-year monitoring to detect delayed ecological responses, especially for taxa with seasonal life cycles that may only reflect new environmental conditions after a full generation has developed. Taken together, these patterns suggest that stronger or clearer responses may emerge in subsequent years.

These findings are consistent with those of Erdozain et al. (2018), who demonstrated that stream integrity can be maintained under varying forest management intensities when key riparian functions are preserved, and they further align with evidence from Northern and Central Europe indicating that the structural retention of riparian features, such as shading and organic matter inputs, effectively buffers short-term ecological impacts on aquatic ecosystems (Feld & Hering, 2007). Nevertheless, in contrast to many earlier studies, this project documented apparent shifts in buffer microclimate, specifically reduced shading and increased air temperatures, while water temperature remained stable. This pattern, also supported by Jentzsch's (2023) site-level measurements underscore the importance of evaluating thermal dynamics separately when interpreting responses of aquatic communities.

The results presented here also prompt a closer look at reference site dynamics. For example, the Kryck reference site—the least species-rich site in the entire study—had even lower richness than the adjacent harvested site just 100 m downstream (Kryck negative). This counterintuitive pattern supports the notion that riparian harvesting is not necessarily detrimental and, under certain conditions, may even enhance biological communities. Dense spruce riparian forests, known to support species-poor communities due to low light and primary production, might benefit from selective light penetration achieved through variable retention harvesting (Hasselquist et al., 2021). Such interventions could stimulate algal growth and increase food availability for grazers, potentially strengthening the entire food web (Kuglerová et al., 2017).

4.2 Buffer Width

Despite the widespread use of buffer width as a key management metric, this study found no evidence that wider buffers were associated with higher macroinvertebrate richness or abundance in the short term. However, a very weak but positive relationship between buffer width and species richness was observed, suggesting that wider buffers may still confer modest ecological benefits even within the short post-harvest timeframe. The lack of significant relationship between diversity and buffer width presented here contrasts with the findings of Richardson & Béraud (2014), whose meta-analysis supported positive biodiversity responses to wider buffer zones. One possible explanation for this discrepancy lies in the temporal scale of the current study: ecological responses to buffer loss may take multiple years to develop and manifest, particularly in slowly recovering stream systems (Frainer et al., 2021; Yeung et al., 2017). An additional contributing factor may be the limited range of buffer widths implemented in this study. Most treatments fell below the widely cited 30 m threshold (Broadmeadow & Nisbet, 2004; Sweeney & Newbold, 2014), which has often been associated with more substantial positive biodiversity outcomes. The narrow gradient may have constrained our ability to detect stronger patterns.

Moreover, buffer width alone may not capture the ecological complexity of riparian zones. Features such as vegetation structure, canopy cover, spatial heterogeneity, and local hydrology can strongly influence stream conditions, even in the absence of wide buffer zones (Kominoski et al., 2011). This view is supported by large-scale European research demonstrating that native deciduous riparian forests enhance the dispersal and diversity of sensitive macroinvertebrate taxa, such as Ephemeroptera, Plecoptera, and Trichoptera, thereby emphasising the critical role of riparian vegetation quality in sustaining stream biodiversity (Peredo Arce et al., 2023).

In this study, treatment types with adaptive or variable retention (e.g., VR and GAP) did not exhibit uniform responses, suggesting that site-specific features likely mediate ecological outcomes. Jonsson el al. (2017) similarly, highlighted that macroinvertebrate community composition in boreal streams is shaped by a combination of light conditions, geomorphology, and upstream land use, which complicates efforts to isolate the role of buffer width

4.3 Substrate Composition

In contrast to buffer characteristics, substrate composition showed consistent associations with macroinvertebrate metrics. Sites dominated by bryophytes and pebbles supported greater total abundance and EPT richness. At the same time, those with high proportions of fine sediment and very fine organic matter exhibited lower diversity and abundance. These findings support the hypothesis that physical habitat quality is a primary determinant of macroinvertebrate assemblages in headwater streams (Frainer & McKie, 2015; Ring et al., 2023).

Bryophytes and coarser substrates provide microhabitats, attachment surfaces, and flow refugia. However, the strength of the relationship between bryophytes and macroinvertebrate metrics varied from year to year. In 2022 (pre-harvest), bryophyte cover was significantly associated with higher macroinvertebrate abundance and EPT richness at several sites. These associations, however, largely disappeared in 2023, persisting only at the RefKryck site where no harvesting occurred. This pattern suggests that bryophyte–macroinvertebrate relationships may be sensitive to changes in habitat quality following forest interventions. As bryophytes often colonise stable substrates and enhance habitat complexity, their reduced association with biological metrics post-harvest may reflect altered streambed conditions or diminished substrate stability (Lehosmaa et al., 2017). The observation that the positive relationship was maintained at the undisturbed reference site further supports this interpretation and highlights the potential utility of bryophytes as indicators of physical habitat integrity (Sundermann et al., 2011).

Fine sediments, on the other hand, can reduce interstitial space and oxygen availability, leading to habitat degradation and loss of sensitive taxa (Yeung et al., 2017). The strong correlations between substrate and biological metrics suggest that even with well-designed riparian buffers, in-stream conditions may ultimately determine the integrity of the macroinvertebrate community.

Interestingly, most of the negative correlations between sand and gravel and macroinvertebrate metrics were observed in the post-harvest year, suggesting that these responses may reflect the increased transport of finer material into the stream channels following logging activities, rather than an inherent biological avoidance of sand and gravel. This distinction is important, as it implies that observed declines in abundance or richness could be a response to habitat alteration driven by sediment input. This finding is consistent with those from Austrian streams, where multimetric assessments have identified substrate quality as a key factor influencing macroinvertebrate assemblages. Stable substrates were associated with higher ecological integrity and biological diversity, emphasising the importance of physical habitat structure in supporting benthic communities (Ofenböck et al., 2004).

However, a key limitation of our approach is that substrate composition was assessed only from material retained in Surber samples. This method inherently favours the collection of finer, mobile materials and likely underrepresents larger, more stable substrates such as rocks and boulders. These coarse substrates, although ecologically important (Brown & Brussock, 1991), are too heavy or fixed to be collected by the sampler. As a result, our analysis may have underestimated the availability of preferred habitats for macroinvertebrates. Given this limitation, bryophytes may serve as an indirect proxy for the presence of stable substrates, since aquatic mosses typically grow on rocks and boulders. Thus, where bryophytes were abundant in our samples, it is likely that coarse substrates were present even if not directly captured. This link is important, as it partially compensates for the absence of direct measurement of rocks and may help explain the positive correlations observed.

4.4 Temporal Variation

While no significant treatment effects were detected, one reference site exhibited a notable temporal shift in community composition. This variation, occurring without direct disturbance, indicates that natural environmental fluctuations, such as changes in hydrology, temperature, or upstream inputs, can drive year-to-year changes in macroinvertebrate assemblages. For example, summer 2022 was exceptionally warm in Sweden, with temperatures reaching 37.2 °C (SMHI, 2022). In contrast, summer 2023 was notably colder and more unstable, with much of the country experiencing below-average temperatures and unusually high rainfall, particularly in July (SMHI, 2023). These climatic differences could have influenced stream temperature, flow regimes, and habitat availability, contributing to the observed variation in community composition. This supports the view that ecological responses in boreal streams are not solely determined by management interventions but also by underlying climatic and landscape processes (Frainer et al., 2021).

Interestingly, the negative control in this study, where no forest harvesting occurred, exhibited community changes similar in direction and magnitude to those observed at the positive control. This result may reflect background ecological variability across years rather than treatment-related effects. Alternatively, it could represent a gradual recovery process. Since the major disturbance (harvest) at this site occurred in 2020, the observed changes might indicate that sensitive macroinvertebrate taxa are slowly recolonising and re-establishing. It also raises the possibility that short-term natural dynamics, such as fluctuations in flow, temperature, or resource availability, can produce changes of scale similar to those induced by forest interventions. This highlights the need for cautious interpretation of short-term BACI data and the importance of including multiple reference sites in forest ecosystem research.

Additionally, the macroinvertebrate community may exhibit lagged responses to harvesting. Functional and trophic changes, such as shifts in resource use or feeding group dominance, may occur before detectable taxonomic shifts (Erdozain et al., 2019). As such, the absence of short-term change does not preclude longer-term ecological consequences. Continued monitoring will be necessary to assess delayed or cumulative impacts over time.

4.5 Catchment Area

The size of the catchment area was not significantly correlated with biodiversity metrics, suggesting that local fluvial conditions (e.g., flow velocity or current) may be more relevant to macroinvertebrate assemblages than broader catchment characteristics. While larger catchments may offer more stable hydrological regimes or greater habitat heterogeneity, the narrow range of catchment sizes in this study, combined with the dominance of local-scale drivers, may have limited the detectability of such effects. Jonsson et al. (2017) emphasised the importance of integrating both local and landscape-scale factors in analyses of land-use effects on aquatic ecosystems.

Similarly, a study by Verdonschot et al. (2012) demonstrated that macroinvertebrate assemblages in French rivers were more strongly influenced by reach-scale and riparian corridor pressures than catchment-scale land use, underscoring the significance of local habitat conditions over broader landscape factors.

4.6 Management Implications

These findings have implications for the management of riparian forests and the design of aquatic monitoring programs. First, the results suggest that in the short term, macroinvertebrate community composition may be relatively indifferent to buffer zone characteristics, such as type and width, including more innovative designs like variable-width configurations, gap cutting, and selective harvesting, particularly when implemented with attention to site-specific ecological conditions.

Second, the analysis highlights the potential importance of habitat-level drivers, such as substrate composition and hydrological integrity, which may serve as more immediate and sensitive indicators of ecological change than buffer characteristics alone.

Ultimately, the results should be interpreted with some caution, as ecological changes may develop over time. While macroinvertebrate richness and abundance offer valuable indicators of ecological condition, they may not fully capture delayed or functional responses to riparian forest harvesting. To gain a more comprehensive understanding of the effects of buffer zones, long-term monitoring is needed, ideally incorporating both taxonomic and functional indicators. As suggested by Kominoski et al. (2011) and Erdozain et al. (2019), a broader assessment framework that includes trophic dynamics, carbon sources, and food web structure may be necessary to evaluate the full ecological consequences of management interventions in riparian forests.

4.7 Limits and weaknesses of this thesis

One of the key limitations of this thesis is its short temporal scope. Monitoring occurred over a one-year post-harvest period, which may be insufficient to capture delayed ecological responses, particularly in systems where community-level changes and functional shifts often unfold over longer timeframes. Previous research shows that legacy effects from harvesting, such as altered litter inputs or canopy-driven microclimatic changes, can manifest in biological communities only after multiple years (Frainer & McKie, 2021; Yeung et al., 2017). Therefore, while no significant changes were observed in the short term, long-term monitoring would be necessary to evaluate the persistence or emergence of ecological impacts.

Another limitation lies in the spatial scale of the study. Although the BACI design provided a robust framework for detecting change, the relatively small number of streams and limited geographic distribution of study sites may constrain the generalizability of findings. Including a broader set of catchments with more variation in buffer types, land-use intensity, and geomorphology would improve the capacity to detect context-dependent responses and strengthen the statistical power of comparisons.

A further limitation is the lack of replication of buffer treatments across independent catchments. Each treatment type (e.g. GAP, CBP, OS, VR) was applied at a single site, meaning that treatment effects are confounded with site-specific conditions. Without replication, it is not possible to statistically separate the effects of buffer design from the unique environmental characteristics of each stream. This limits the extent to which conclusions about specific buffer strategies can be generalised beyond the study sites.

Additionally, this thesis primarily focused on biotic metrics (macroinvertebrate communities) while omitting potentially influential abiotic variables, such as microclimate, light availability, soil moisture, and temperature regimes. These factors respond rapidly to canopy disturbance and exert strong bottom-up effects on stream ecosystems, especially in buffer-edge environments (Oldén et al., 2019; Myrstener et al., 2023). Including such variables could provide a mechanistic understanding of how buffer structure translates into ecological outcomes.

5. Conclusion

This study evaluated the short-term effects of different riparian buffer designs on aquatic macroinvertebrate communities in boreal headwater streams. Across seven study sites, including both harvested treatments and reference conditions, no statistically significant differences in biodiversity metrics were observed between pre- and post-harvest years. The lack of response suggests that, under current best practices, including retention of structurally complex riparian buffers, macroinvertebrate communities can remain stable in the first year after harvest.

Although buffer width alone did not show a strong correlation with macroinvertebrate diversity, the type of streambed substrate played a more significant role in determining diversity. Habitats with bryophytes and coarse materials, such as pebbles, supported more diverse and abundant communities, whereas areas dominated by fine sediments tended to have reduced habitat quality. These findings underscore the importance of considering local habitat characteristics in conjunction with buffer design in riparian management.

Still, the absence of immediate changes does not rule out longer-term or cumulative effects. Shifts in food webs, lingering impacts from past disturbances, or changes driven by climate may only become apparent after several years. Therefore, continued monitoring and more detailed functional assessments are essential for a comprehensive understanding of the ecological effects of riparian forest practices. This study contributes to the conversation on sustainable forestry by demonstrating that stream ecosystems can respond in complex and time-dependent ways to human activities.

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Popular science summary

Forests near streams, known as riparian forests, are crucial for maintaining the health of aquatic ecosystems. They provide shade, reduce pollution, and offer food and shelter for countless small creatures. In northern Sweden, as in many forested regions, clear-cutting is a common practice in forest management. To minimise its impact, strips of forest are often left untouched along streams, forming what are known as riparian buffers.

However, how effective are these buffers at protecting the life within streams? This study focused on tiny aquatic creatures called macroinvertebrates, such as mayflies, stoneflies, and caddisflies, which are key indicators of water quality and ecosystem health.

Over a two-year period, researchers collected data before and after forest harvesting at seven sites with different types of buffer zones. The study used advanced statistical tools to compare biodiversity and the composition of macroinvertebrate communities. The big surprise? There were no significant changes after the forest was harvested, regardless of the buffer type. In other words, the macroinvertebrates seemed resilient, at least in the short term.

Interestingly, the width of the buffer zone had no clear effect on biodiversity. Instead, what mattered more was the type of material found on the streambed. Sites with moss and pebbles had richer invertebrate life, while areas with fine sediments were less diverse.

These results suggest that while current forestry practices may not cause immediate harm, it is crucial to monitor streams over the long term. The effects of harvesting may take years to become apparent, and features like streambed quality may be just as important as buffer width in protecting aquatic ecosystems. This study contributes to a growing body of knowledge aimed at balancing forest use with the need to protect Sweden’s precious freshwater biodiversity.

Acknowledgements

First and foremost, I would like to thank my supervisor, Lenka Kuglerová, for her invaluable support and guidance throughout the writing of this thesis. I am highly grateful for the opportunity to work with the data she provided, her thoughtful insights into the project, and her patience in teaching me how to use RStudio. Thanks to her kind and understanding attitude, I had a very positive experience on my research journey. I would also like to thank my co-supervisor, Gabriele Weigelhofer, for her constructive feedback and valuable suggestions, which helped me to improve the content and structure of this work. I would further like to express my gratitude to my examiner, Joachim Strengbom, for his insightful comments and helpful suggestions, which contributed meaningfully to the final version of this thesis. Finally, my sincere thanks go to Nina Mosor, my student opponent, whose thoughtful feedback and remarks also helped to refine and strengthen my work.

Appendix 1

Table A 1. Spearman correlation coefficients (ρ) and p-values showing relationships between macroinvertebrate biodiversity metrics and substrate composition across sites and sampling years.

			richness		abundance		EPT richness		EPT abundance	
site	harvest	material	spear_rho	p_value	spear_rho	p_value	spear_rho	p_value	spear_rho	p_value
CBP	after	whole.leaves.needles	-0.460	0.436	0.053	0.933	-0.028	0.965	-0.158	0.800
CBP	after	bryophytes	0.263	0.669	0.718	0.172	0.460	0.436	0.821	0.089
CBP	after	fragments.om	0.289	0.637	0.410	0.493	0.676	0.210	0.154	0.805
CBP	after	very.fine.om	0.526	0.362	0.462	0.434	0.865	0.058	0.308	0.614
CBP	after	very.fine.mm	0.406	0.498	-0.316	0.604	0.250	0.685	-0.474	0.420
CBP	after	sand.and.gravel	-0.237	0.701	-0.872	0.054	-0.649	0.236	-0.872	0.054
CBP	after	pebbles	0.162	0.794	**0.949**	**0.014**	0.667	0.219	**0.949**	**0.014**
CBP	before	whole.leaves.needles	-0.344	0.571	-0.447	0.450	0.750	0.144	-0.671	0.215
CBP	before	bryophytes	0.103	0.870	**0.900**	**0.037**	-0.224	0.718	0.700	0.188
CBP	before	fragments.om	0.342	0.573	-0.564	0.322	-0.287	0.640	-0.154	0.805
CBP	before	very.fine.om	0.363	0.548	-0.707	0.182	0.395	0.510	-0.707	0.182
CBP	before	very.fine.mm	NA	NA	NA	NA	NA	NA	NA	NA
CBP	before	sand.and.gravel	-0.237	0.701	-0.872	0.054	0.057	0.927	-0.667	0.219
CBP	before	pebbles	-0.181	0.770	-0.354	0.559	-0.395	0.510	0.000	1.000
GAP	after	whole.leaves.needles	-0.154	0.805	-0.462	0.434	-0.154	0.805	-0.410	0.493
GAP	after	bryophytes	0.667	0.219	0.872	0.054	0.667	0.219	0.872	0.054
GAP	after	fragments.om	0.205	0.741	-0.103	0.870	0.205	0.741	-0.051	0.935
GAP	after	very.fine.om	0.300	0.624	0.100	0.873	0.300	0.624	0.100	0.873
GAP	after	very.fine.mm	0.105	0.866	0.527	0.361	0.105	0.866	0.316	0.604
GAP	after	sand.and.gravel	-0.205	0.741	0.205	0.741	-0.205	0.741	0.154	0.805
GAP	after	pebbles	NA	NA	NA	NA	NA	NA	NA	NA
GAP	before	whole.leaves.needles	**-0.949**	**0.014**	-0.791	0.111	**-0.973**	**0.005**	-0.791	0.111
GAP	before	bryophytes	0.866	0.058	0.577	0.308	**0.889**	**0.044**	0.577	0.308
GAP	before	fragments.om	**-0.975**	**0.005**	-0.821	0.089	**-1.000**	**0.000**	-0.821	0.089
GAP	before	very.fine.om	-0.224	0.718	0.112	0.858	-0.229	0.710	0.112	0.858
GAP	before	very.fine.mm	NA	NA	NA	NA	NA	NA	NA	NA
GAP	before	sand.and.gravel	-0.354	0.559	-0.354	0.559	-0.363	0.548	-0.354	0.559
GAP	before	pebbles	NA	NA	NA	NA	NA	NA	NA	NA
OS	after	whole.leaves.needles	0.700	0.188	0.700	0.188	0.616	0.269	**0.900**	**0.037**
OS	after	bryophytes	0.700	0.188	0.700	0.188	0.821	0.089	0.500	0.391
OS	after	fragments.om	-0.335	0.581	-0.447	0.450	-0.516	0.373	-0.112	0.858
OS	after	very.fine.om	0.205	0.741	0.667	0.219	0.263	0.669	0.564	0.322
OS	after	very.fine.mm	0.447	0.450	0.224	0.718	0.459	0.437	0.224	0.718
OS	after	sand.and.gravel	-0.667	0.219	**-0.975**	**0.005**	-0.763	0.133	-0.821	0.089
OS	after	pebbles	NA	NA	NA	NA	NA	NA	NA	NA
OS	before	whole.leaves.needles	0.872	0.054	0.872	0.054	0.811	0.096	0.667	0.219
OS	before	bryophytes	0.667	0.219	0.667	0.219	0.811	0.096	0.872	0.054
OS	before	fragments.om	-0.100	0.873	0.200	0.747	-0.053	0.933	0.000	1.000
OS	before	very.fine.om	0.354	0.559	0.354	0.559	0.186	0.764	0.000	1.000
OS	before	very.fine.mm	-0.354	0.559	-0.707	0.182	-0.559	0.327	-0.707	0.182
OS	before	sand.and.gravel	-0.300	0.624	-0.400	0.505	-0.527	0.361	-0.700	0.188
OS	before	pebbles	-0.354	0.559	-0.707	0.182	-0.559	0.327	-0.707	0.182
Ref20	after	whole.leaves.needles	0.895	0.040	0.872	0.054	0.667	0.219	0.359	0.553
Ref20	after	bryophytes	-0.816	0.092	-0.872	0.054	-0.462	0.434	-0.616	0.269
Ref20	after	fragments.om	0.564	0.322	0.500	0.391	0.700	0.188	-0.200	0.747
Ref20	after	very.fine.om	0.592	0.293	0.577	0.308	0.866	0.058	0.289	0.638
Ref20	after	very.fine.mm	-0.057	0.927	-0.224	0.718	0.112	0.858	-0.224	0.718
Ref20	after	sand.and.gravel	-0.860	0.061	-0.783	0.118	**-0.894**	**0.041**	-0.112	0.858
Ref20	after	pebbles	NA	NA	NA	NA	NA	NA	NA	NA
Ref20	before	whole.leaves.needles	-0.400	0.505	-0.900	0.037	-0.671	0.215	-0.205	0.741
Ref20	before	bryophytes	0.447	0.450	0.783	0.118	0.500	0.391	-0.229	0.710
Ref20	before	fragments.om	-0.359	0.553	-0.821	0.089	-0.574	0.312	-0.132	0.833
Ref20	before	very.fine.om	NA	NA	NA	NA	NA	NA	NA	NA
Ref20	before	very.fine.mm	NA	NA	NA	NA	NA	NA	NA	NA
Ref20	before	sand.and.gravel	0.154	0.805	0.410	0.493	0.459	0.437	0.711	0.179
Ref20	before	pebbles	-0.707	0.182	-0.707	0.182	-0.791	0.111	-0.363	0.548

RefKryck	after	whole.leaves.needles	0.000	1.000	0.000	1.000	0.761	0.135	0.444	0.454
RefKryck	after	bryophytes	0.205	0.741	0.564	0.322	0.054	0.931	**0.947**	**0.014**
RefKryck	after	fragments.om	0.200	0.747	-0.500	0.391	0.632	0.252	-0.359	0.553
RefKryck	after	very.fine.om	-0.205	0.741	-0.359	0.553	0.703	0.185	0.000	1.000
RefKryck	after	very.fine.mm	0.224	0.718	0.224	0.718	0.825	0.086	0.516	0.373
RefKryck	after	sand.and.gravel	0.462	0.434	0.821	0.089	-0.081	0.897	0.289	0.637
RefKryck	after	pebbles	0.112	0.858	0.447	0.450	-0.059	0.925	0.000	1.000
RefKryck	before	whole.leaves.needles	0.750	0.144	0.447	0.450	0.354	0.559	0.671	0.215
RefKryck	before	bryophytes	-0.783	0.118	0.300	0.624	-0.369	0.541	0.100	0.873
RefKryck	before	fragments.om	0.459	0.437	0.872	0.054	0.649	0.236	0.616	0.269
RefKryck	before	very.fine.om	-0.224	0.718	**-1.000**	**0.000**	-0.316	0.604	**-0.900**	**0.037**
RefKryck	before	very.fine.mm	NA	NA	NA	NA	NA	NA	NA	NA
RefKryck	before	sand.and.gravel	0.177	0.776	-0.580	0.306	0.472	0.422	-0.791	0.111
RefKryck	before	pebbles	0.395	0.510	-0.354	0.559	0.559	0.327	-0.354	0.559
RefNegativ	after	whole.leaves.needles	**0.947**	**0.014**	0.718	0.172	-0.181	0.770	0.359	0.553
RefNegativ	after	bryophytes	NA	NA	NA	NA	NA	NA	NA	NA
RefNegativ	after	fragments.om	-0.526	0.362	-0.359	0.553	0.544	0.343	0.564	0.322
RefNegativ	after	very.fine.om	0.289	0.637	-0.051	0.935	-0.181	0.770	-0.667	0.219
RefNegativ	after	very.fine.mm	-0.500	0.391	-0.051	0.935	0.363	0.548	0.103	0.870
RefNegativ	after	sand.and.gravel	-0.564	0.322	-0.200	0.747	-0.354	0.559	-0.600	0.285
RefNegativ	after	pebbles	NA	NA	NA	NA	NA	NA	NA	NA
RefNegativ	before	whole.leaves.needles	0.800	0.104	0.600	0.285	0.577	0.308	**0.900**	**0.037**
RefNegativ	before	bryophytes	NA	NA	NA	NA	NA	NA	NA	NA
RefNegativ	before	fragments.om	-0.410	0.493	-0.051	0.935	-0.444	0.454	0.154	0.805
RefNegativ	before	very.fine.om	-0.224	0.718	-0.447	0.450	0.000	1.000	**-0.894**	**0.041**
RefNegativ	before	very.fine.mm	0.000	1.000	0.707	0.182	-0.408	0.495	0.707	0.182
RefNegativ	before	sand.and.gravel	-0.200	0.747	**-0.900**	**0.037**	0.289	0.638	-0.600	0.285
RefNegativ	before	pebbles	0.000	1.000	0.707	0.182	-0.408	0.495	0.707	0.182
VR	after	whole.leaves.needles	0.410	0.493	0.872	0.054	0.667	0.219	0.872	0.054
VR	after	bryophytes	0.632	0.252	0.264	0.668	0.264	0.668	0.264	0.668
VR	after	fragments.om	0.872	0.054	**0.975**	**0.005**	**0.975**	**0.005**	**0.975**	**0.005**
VR	after	very.fine.om	0.400	0.505	0.600	0.285	0.700	0.188	0.600	0.285
VR	after	very.fine.mm	-0.316	0.604	-0.791	0.111	-0.632	0.252	-0.791	0.111
VR	after	sand.and.gravel	-0.500	0.391	**-0.900**	**0.037**	-0.700	0.188	**-0.900**	**0.037**
VR	after	pebbles	NA	NA	NA	NA	NA	NA	NA	NA
VR	before	whole.leaves.needles	0.410	0.493	0.410	0.493	0.289	0.637	0.667	0.219
VR	before	bryophytes	0.112	0.858	0.112	0.858	0.229	0.710	-0.224	0.718
VR	before	fragments.om	0.100	0.873	0.100	0.873	-0.103	0.870	0.200	0.747
VR	before	very.fine.om	0.783	0.118	0.783	0.118	0.803	0.102	0.447	0.450
VR	before	very.fine.mm	NA	NA	NA	NA	NA	NA	NA	NA
VR	before	sand.and.gravel	-0.800	0.104	-0.800	0.104	-0.667	0.219	**-0.900**	**0.037**
VR	before	pebbles	-0.707	0.182	-0.707	0.182	-0.725	0.165	-0.707	0.182

Appendix 2

Table A 2 Table with values for biodiversity metrics and buffer width at treatment and reference sites (2022–2023).

site name	plot.id	treat.type	year	richness	abundance	EPT richness	EPT abundace	buffer_width (m)
CBP	bau_1	treatment	2023	11	507	4	152	24.235
CBP	bau_3	treatment	2023	9	48	3	29	13.565
CBP	bau_4	treatment	2023	8	109	3	73	20.9
CBP	bau_5	treatment	2023	8	621	4	113	14.85
CBP	bau_7	treatment	2023	17	209	6	102	20.465
VR	vr_1	treatment	2023	5	9	1	2	29.155
VR	vr_3	treatment	2023	19	99	5	16	25.03
VR	vr_6	treatment	2023	28	650	10	97	23.59
VR	vr_7	treatment	2023	26	1132	9	300	23.55
VR	vr_8	treatment	2023	12	146	6	58	80
OS	os_1	treatment	2023	14	33	5	13	29.015
OS	os_2	treatment	2023	18	58	8	14	23.665
OS	os_4	treatment	2023	20	289	9	70	26.96
OS	os_5	treatment	2023	27	163	9	84	29.715
OS	os_6	treatment	2023	16	61	6	27	31.255
GAP	gap_2	treatment	2023	12	131	3	10	20.71
GAP	gap_3	treatment	2023	21	425	7	23	7.895
GAP	gap_5	treatment	2023	6	29	1	1	25.165
GAP	gap_6	treatment	2023	18	629	5	47	22.28
GAP	gap_7	treatment	2023	23	346	10	57	11.46
RefNegativ	kryck_neg_	control	2023	9	72	4	47	0
RefNegativ	kryck_neg_	control	2023	12	44	4	12	0
RefNegativ	kryck_neg_	control	2023	14	249	4	67	0
RefNegativ	kryck_neg_	control	2023	14	396	4	24	0
RefNegativ	kryck_neg_	control	2023	13	128	5	60	0

Appendix 3

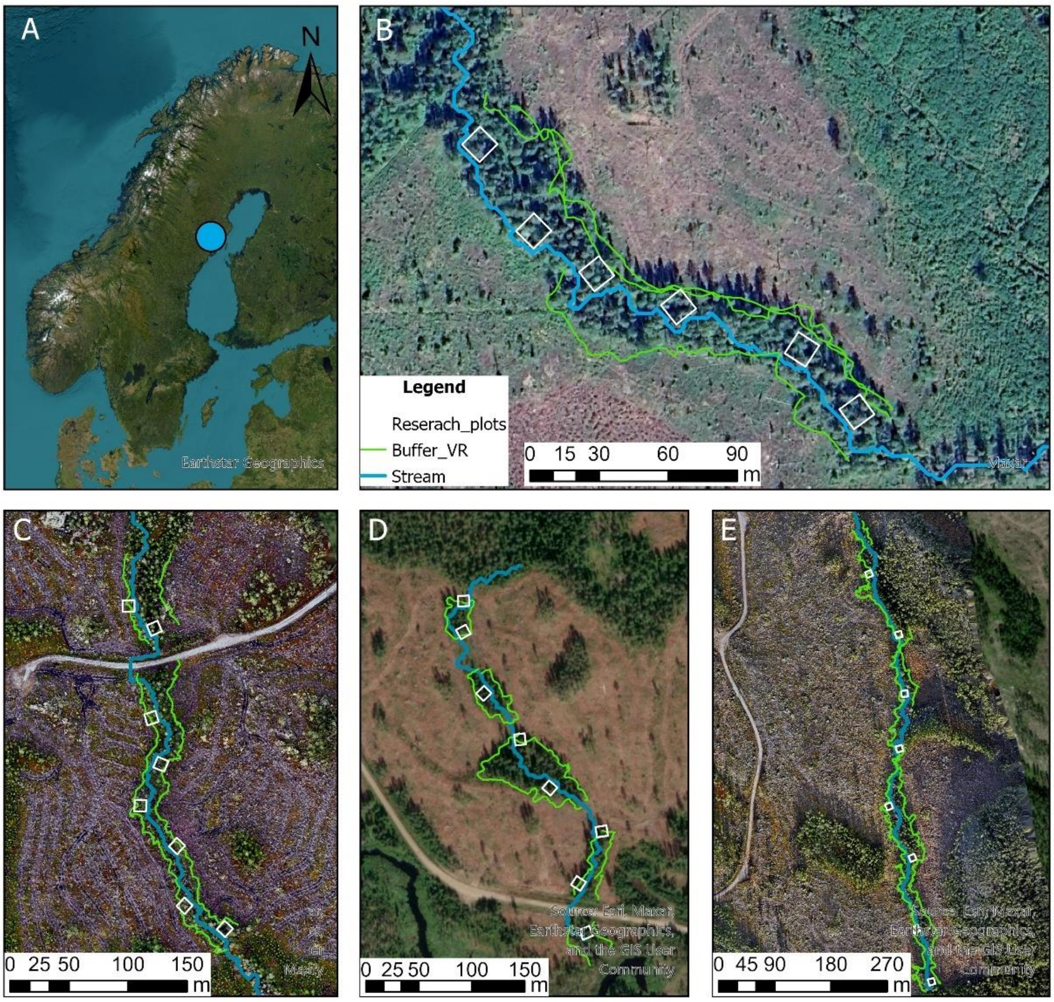

Figure 1A 1 (A) Map of northern Europe showing the geographical location of the study region (indicated by the blue dot) in central Sweden. (B–E) Maps showing the spatial arrangement of research plots (white squares) along streams (blue lines) within four different riparian buffer treatment sites, after harvest:(B) OS site, (C) CBP site, (D) GAP site, (E) VR site. (Source: Esri, Maxar, Earthstar Geographics, and the GIS User Community).

Appendix 4

Table A 3. Catchment area and corresponding biodiversity metrics across all study sites and years.

site_name	treat_type	metrics	response	year	catchment (km2)
CBP	treatment	tot_abundance	1030	2022	0.72
CBP	treatment	tot_abundance	987	2023	0.72
CBP	treatment	tot_richness	18	2022	0.72
CBP	treatment	tot_richness	19	2023	0.72
CBP	treatment	tot_ept_abundance	299	2022	0.72
CBP	treatment	tot_ept_abundance	469	2023	0.72
CBP	treatment	tot_ept_richness	8	2022	0.72
CBP	treatment	tot_ept_richness	7	2023	0.72
GAP	treatment	tot_abundance	1610	2022	0.61
GAP	treatment	tot_abundance	1560	2023	0.61
GAP	treatment	tot_richness	36	2022	0.61
GAP	treatment	tot_richness	35	2023	0.61
GAP	treatment	tot_ept_abundance	393	2022	0.61
GAP	treatment	tot_ept_abundance	138	2023	0.61
GAP	treatment	tot_ept_richness	16	2022	0.61
GAP	treatment	tot_ept_richness	18	2023	0.61
OS	treatment	tot_abundance	657	2022	2.51
OS	treatment	tot_abundance	604	2023	2.51
OS	treatment	tot_richness	33	2022	2.51
OS	treatment	tot_richness	36	2023	2.51
OS	treatment	tot_ept_abundance	130	2022	2.51
OS	treatment	tot_ept_abundance	208	2023	2.51
OS	treatment	tot_ept_richness	14	2022	2.51
OS	treatment	tot_ept_richness	15	2023	2.51
VR	treatment	tot_abundance	767	2022	1
VR	treatment	tot_abundance	2036	2023	1
VR	treatment	tot_richness	35	2022	1
VR	treatment	tot_richness	38	2023	1
VR	treatment	tot_ept_abundance	174	2022	1
VR	treatment	tot_ept_abundance	474	2023	1
VR	treatment	tot_ept_richness	15	2022	1
VR	treatment	tot_ept_richness	16	2023	1
Ref20	control	tot_abundance	613	2022	1.43
Ref20	control	tot_abundance	1719	2023	1.43
Ref20	control	tot_richness	34	2022	1.43
Ref20	control	tot_richness	38	2023	1.43
Ref20	control	tot_ept_abundance	182	2022	1.43
Ref20	control	tot_ept_abundance	427	2023	1.43
Ref20	control	tot_ept_richness	14	2022	1.43
Ref20	control	tot_ept_richness	20	2023	1.43
Ref Kryck	control	tot_abundance	208	2022	0.24
Ref Kryck	control	tot_abundance	208	2023	0.24
Ref Kryck	control	tot_richness	20	2022	0.24
Ref Kryck	control	tot_richness	20	2023	0.24
Ref Kryck	control	tot_ept_abundance	94	2022	0.24
Ref Kryck	control	tot_ept_abundance	81	2023	0.24
Ref Kryck	control	tot_ept_richness	8	2022	0.24
Ref Kryck	control	tot_ept_richness	8	2023	0.24
Ref negative	control	tot_abundance	1181	2022	0.52
Ref negative	control	tot_abundance	889	2023	0.52
Ref negative	control	tot_richness	21	2022	0.52
Ref negative	control	tot_richness	19	2023	0.52
Ref negative	control	tot_ept_abundance	530	2022	0.52
Ref negative	control	tot_ept_abundance	210	2023	0.52
Ref negative	control	tot_ept_richness	8	2022	0.52
Ref negative	control	tot_ept_richness	7	2023	0.52

Appendix 5

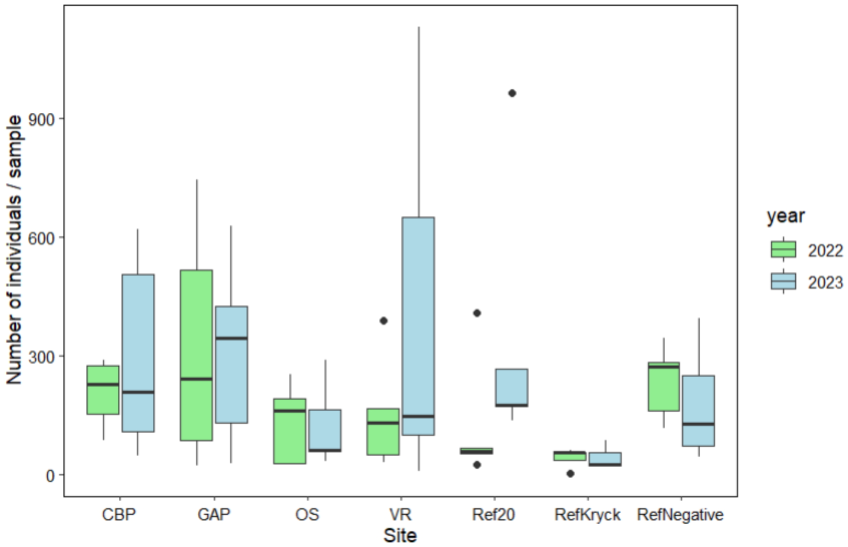

Figure 1A 2. Boxplot showing the number of individuals per sample at seven sites in 2022 (pre-harvest, green) and 2023 (post-harvest, blue). Including treatments (CBP, GAP, OS, VR) and references (Ref20, RefKryck, RefNegative) sites. Each box represents values derived from five Surber sampling points collected along the stream reach at each site. The horizontal line within each box represents the median, the box outlines indicate the interquartile range (IQR), the whiskers extend to the smallest and largest values within 1.5 times the IQR from the lower and upper quartiles, and the black dots represent statistical outliers.

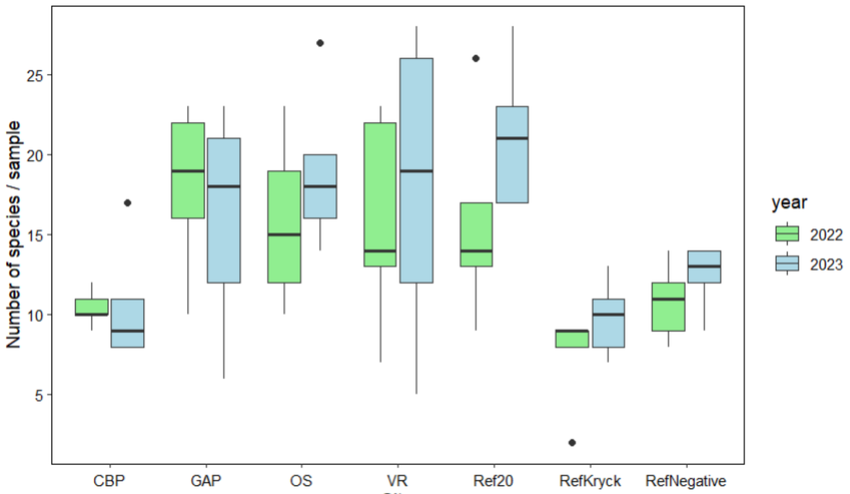

Figure 1A 3. Boxplot showing species richness per sample at seven sites in 2022 (green) and 2023 (blue), including management (CBP, GAP, OS, VR) and reference sites (Ref20, RefKryck, RefNegative). Each box represents values derived from five Surber sampling points collected along the stream reach at each site. The horizontal line within each box represents the median, the box outlines indicate the interquartile range (IQR), the whiskers extend to the smallest and largest values within 1.5 times the IQR from the lower and upper quartiles, and the black dots represent statistical outliers.

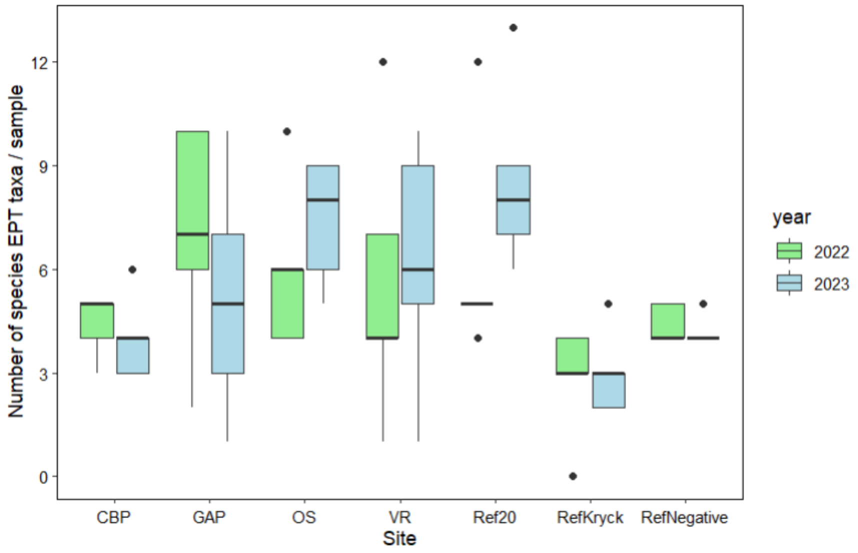

Figure 1A 4. Boxplot showing the number of EPT taxa (Ephemeroptera, Plecoptera, Trichoptera) per sample at seven sites in 2022 (green) and 2023 (blue), including treatment (CBP, GAP, OS, VR) and reference (Ref20, RefKryck, RefNegative) sites. Each box represents values derived from five Surber sampling points collected along the stream reach at each site. The horizontal line within each box represents the median, the box outlines indicate the interquartile range (IQR), the whiskers extend to the smallest and largest values within 1.5 times the IQR from the lower and upper quartiles, and the black dots represent statistical outliers.

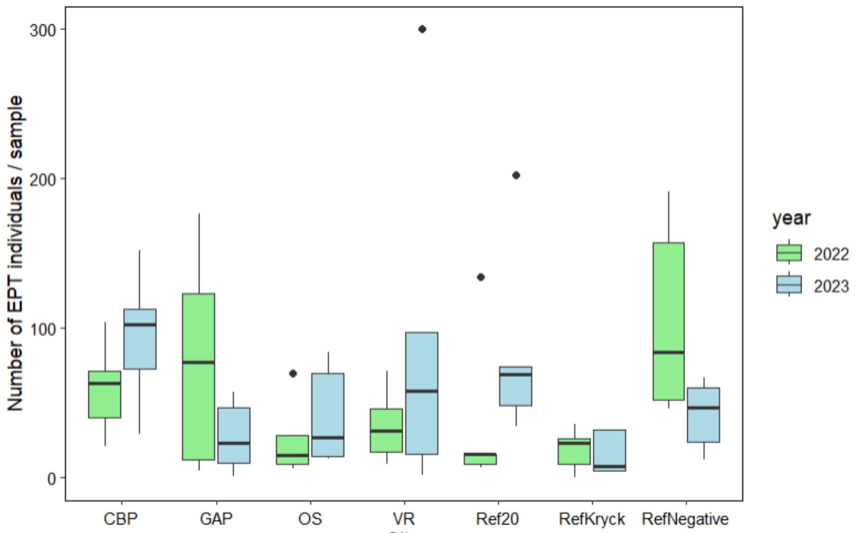

Figure 1A 5. Boxplot showing EPT (Ephemeroptera, Plecoptera, Trichoptera) individual counts per sample at seven sites in 2022 (green) and 2023 (blue), including treatment (CBP, GAP, OS, VR) and reference (Ref20, RefKryck, RefNegative) sites. Each box represents values derived from five Surber sampling points collected along the stream reach at each site. The horizontal line within each box represents the median, the box outlines indicate the interquartile range (IQR), the whiskers extend to the smallest and largest values within 1.5 times the IQR from the lower and upper quartiles, and the black dots represent statistical outliers.

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