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RESEARCH ARTICLE

The effects of land use on downstream water quality and biodiversity in a changing tropical mosaic landscape

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Abstract

Context Land use and other human activities generally reduce downstream water quality, with consequences for ecosystem services and freshwater biodiversity. However, the relative effects of different types of land use and associated activities across large spatial scales can be difficult to understand, and thus to address, in heterogeneous landscapes.

Objectives To assess how different types of landscapes affect the water quality and aquatic biodiversity downstream. To assess if coffee agroforestry landscapes are intermediate between forests and agriculture in terms of their effects on biodiversity in streams.

Methods We measured water quality parameters and sampled stream macroinvertebrates in 46 catchments with perennial streams draining through mosaic landscapes managed by smallholder farmers in southwest Ethiopia. For each of the catchments, we estimated the proportion of each land use, including agriculture, coffee agroforestry and forests, estimated the settlement density and calculated an index representing the number of and distance to coffee washing stations.

Results We found that water quality was better (e.g. high dissolved oxygen, low turbidity and low nutrient concentration) for streams draining through landscapes with high forest cover as compared to landscapes dominated by agroforestry or agriculture.

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Furthermore, we found elevated *Escherichia coli* (*E. coli*) concentrations in most streams, especially those draining agroforestry landscapes. The composition of macroinvertebrates varied much across streams with sensitive taxa dominating clean streams and vice versa. This pattern became even clearer when we analysed the direct effects of settlement density and coffee washing stations on the abundances of families with different sensitivity to poor water quality, but there were also inconsistent responses in some groups.

Conclusion Our finding suggests that a single measurement of water chemistry alone inadequately captured land use effects on water quality, highlighting the importance of including biodiversity assessment. Since people frequently use these streams for various purposes and freshwater biodiversity is at stake, our findings highlight the urgent need of measures to reduce these negative effects. Such measures should focus on agriculture and agroforestry-dominated landscapes to primarily reduce the negative impacts of wastes from settlement areas and wet coffee washing stations.

Keywords Agroforestry · Aquatic biodiversity · Coffee washing stations · Macroinvertebrate · Tropical stream

Introduction

In human-dominated landscapes, agriculture, industry, and settlement areas generate organic waste and chemical pollutants (e.g. nutrients, pathogens, herbicides, and pesticides). Through surface runoff, these pollutants can reach streams, rivers and lakes downstream, with negative effects on biodiversity and potential harms to human health, especially where pollutants are directly deposited in streams without treatment and in the absence of riparian buffer vegetation (Hetling and Sykes 1973; Bain et al. 2014; Dodds and Smith 2016; Le Moal et al. 2019). Other human activities such as damming of water courses can affect fish migration and fishing opportunities due to changes of stream channel morphology, sediment transportation, and connectivity (Nilsson et al. 2005; Grill et al. 2015; Doretto et al. 2020; Zhang and Gu 2023) and the draining of wetlands interrupts denitrification and reduces downstream baseflows (Vitousek et al. 1997; Martínez-Espinosa et al. 2021). The understanding of the land use, water quality and aquatic biodiversity nexus has been better documented in temperate regions than in the tropics (Blann et al. 2009; Buss et al. 2014; Poikane et al. 2020; Schürings et al. 2022; Shahady and Montero-Ramírez 2023). In the tropics, few studies have examined these links in detail across multifunctional mosaic landscapes dominated by smallholder farmers, which are common in low-income countries (but see Rahayu et al. 2013). However, understanding how land use affects aquatic biodiversity and water quality is crucial to be able to develop management strategies and policies that meet the UN's sustainable development goals (SDGs) for water quality and biodiversity conservation in landscapes experiencing fast change due to modernization and climate change (UN SDGs 2015).

Variation in water chemistry and species composition in streams and rivers are often governed by complex and interacting processes such as differences in flow regimes, the location along the upstream–downstream gradient, bedrock and soil types, topography, substrate composition, riparian vegetation, and surrounding human activities (Poff et al. 1997; Allan 2004; Doretto et al. 2020). For example, forest clearing generally reduces canopy shade which thereby often leads to increased water temperatures and reduces input of terrestrial food resources for aquatic organisms (Connolly et al. 2016). Reduction of vegetation cover can further accelerate stormwater runoff due to reduced infiltration (Kumwimba et al. 2024; Nedbal et al. 2025). Increase in the cover of agricultural fields can increase the transport of pollutants, sediments and excess nutrients to streams, and lead to eutrophication and depletion of dissolved oxygen which can further deteriorate water quality (dos Reis Oliveira et al. 2019; Piffer et al. 2021). The effects can vary in agroforestry depending on natural variation in for example elevation and management practices (de Jesús Crespo et al. 2020). Furthermore, improper sewage disposal from residential areas and agricultural runoff containing animal waste often introduce *E. coli* and other pathogens into surface water (Buckerfield et al. 2020). In rural areas of Africa and Asia a large portion of drinking water sources are at risk of *E. coli* contamination (Bain et al. 2014) and detecting the presence of *E. coli* has been suggested an effective tool for monitoring water quality (Edberg et al.

2000). Poor water quality can thus manifest in many different ways including pathogen contamination or chemical and physical alterations. The many drivers involved make it difficult to disentangle the links and propose effective interventions.

Given that species have different niches, it is not surprising to find different community composition in different types of waters. Some taxa actually thrive in polluted waters, while others are very sensitive to a deteriorated quality of one or several water quality parameters (Armitage et al. 1983; Berger et al. 2018). Mayflies, stoneflies, caddisflies and true flies are among the most commonly used indicators of water quality (Molineri et al. 2020; Tubić et al. 2024). Using species composition, often in the way of a biological index, is considered as a robust measure for assessing water quality and aquatic ecosystem health in many countries (Dallas 1997; Palmer and Taylor 2004; Aschalew and Moog 2015). The benefit of such an indicator is that it can be compiled after a single visit to a field site. The assumption is that the organisms respond to the effects of surrounding land use pressures that extend beyond the type of snapshot picture that is obtained by collecting water chemistry data on a single occasion. Yet, as noted above, poor water quality can mean different things and it is unlikely that a single index even based on species composition of aquatic organisms effectively can capture all these aspects (Vadas et al. 2022).

The countryside in low-income countries is often characterized by mosaic landscapes where a large proportion of the population is highly dependent on nature for their basic needs and livelihoods including water resources of different kinds (Fedele et al. 2021). The current water use per capita might be low (Wada et al. 2011), but with rapid modernization and population growth we can expect higher competition for water and a fast degradation if appropriate measures are not taken to protect the processes that enhance, or at least maintain, water quality and aquatic biodiversity (Wang et al. 2024). Climate change poses another threat, but at the same time might open up for restoration actions. One example is that the tree cover may expand at new locations when coffee agroforestry move to higher elevations (Hylander et al. 2024). In many landscapes we need to understand how human land use and other activities affect water quality and biodiversity in order to predict the long-term effects of the projected changes on water resources.

Southwestern Ethiopia is a typical tropical mosaic landscape where the water quality and associated biodiversity are governed by the aggregated effects of the smallholder farmers, small companies and settlements. Most people are making their living from a combination of growing food crops in gardens and on open fields, coffee agroforestry, and holding a few livestock (Manlosa et al. 2019). Their cattle regularly drink water from streams and both in the countryside and in emerging urban centres, sewage treatment facilities are not well developed. One particularity of this landscape is that coffee agroforestry is widespread. This is a land use practice where coffee shrubs are grown under native shade trees and it could be considered as an intermediate habitat type between forests and open agriculture in terms of system complexity and ecological integrity (Hylander et al. 2024). In the region there is a trajectory of modernization, especially in terms of the expansion of smaller and larger urban centres, the draining of wetlands, the increase in small-scale irrigation and in the establishment of new wet coffee washing stations. A wet coffee washing station is an agro-processing facility, owned either privately or by cooperatives, and usually located in coffee-dominated landscapes close to streams, where coffee cherries are processed through soaking and fermentation in water tanks. This process removes the pulp from the coffee beans, but generates large amounts of organic waste and wastewater as by-products (Dadi et al. 2018). Climate change is projected to be a significant driver of land use change in the area and as temperatures rise, coffee agroforestry might potentially expand into areas at higher elevations that now have a too cold climate for coffee (Tollera 2020; Chemura et al. 2021; Hylander et al. 2024). Moreover, the population continues to grow, potentially putting higher pressure on water resources (Rodrigues et al. 2021).

To better understand the link between land use, water quality and biodiversity in mosaic landscapes, we selected 46 small perennial streams in southwest Ethiopia from catchments varying in upstream characteristics such as the proportion of cropland, an index representing the number and distance to wet coffee washing stations and settlement density. Hereafter we use the term ‘land use’ to encompass also settlement density and the coffee washing station index. In each stream, we sampled water quality parameters and collected data on the densities at the family level of

three orders of macroinvertebrates, namely mayflies (Ephemeroptera), true flies (Diptera), and stoneflies (Plecoptera). The families of these orders are known to show a wide difference in sensitivity to water quality deterioration and some of the true flies are vectors of diseases (Armitage et al. 1983; Adler and Courtney 2019). We asked the following questions:

(1) What are the effects of land use on various measures of water quality that may impact both biodiversity and human health in streams?
(2) What are the effects of different water quality parameters and local environmental variables on invertebrate communities in streams?
(3) Are there any direct effects of land use on invertebrate communities not captured by a snapshot measurement of water chemistry variables?

Methods

Study area

We conducted this study in Jimma Zone of Oromia region, southwest Ethiopia (36.0 – 37.0° E and 7.4 – 8.1° N) (Fig. 1) at elevations ranging from 1500 to 2220 m above sea level. This area receives main rainfall from June to September and the average annual rainfall is 1557 mm (Zignol et al. 2023), but there is variation both spatially and temporally. The study area is important in terms of its water resources and associated biodiversity, since it is located in the part of the country with the highest precipitation (Gemeda et al. 2021). It is also the head water source for transboundary rivers including Blue Nile and Omo-Gibe (Chimdessa et al. 2019). There are still some large remnant of Afromontane moist forests in this region (Fashing et al. 2022). In these forests, coffee is commonly harvested, but with minimum disturbance or management (Beche et al. 2024).

Catchments in this area are characterized by a dynamic and mixed land use system. Most people live in rural households which are distributed across the whole landscape (i.e. more than 87% of the total population in Jimma zone), but there are also scattered small villages and small towns along the main roads connecting the different districts (Tamirat and Tadele 2023). Smallholder farmers practice mixed agriculture (crop, livestock, plantation, homegarden and coffee agroforestry) where surface water is an essential resource of their livelihood strategies for both domestic and agricultural uses. Coffee is an important cash crop grown by many farmers. It is suggested that coffee agroforestry is contributing to an increase in tree cover and in protecting forests from deforestation in the landscape (Hylander et al. 2013; Ango et al. 2014, 2020; Manlosa et al. 2019). The practice of processing coffee with water instead of dry processing is increasing, and thus the number of wet processing stations is also increasing. These stations generate large amounts of organic waste loaded with solid substances, nutrients and organic substances (tannins, phenolics and alkaloids) that exhibit high biological and chemical oxygen demand (Rattan et al. 2015; Ijanu et al. 2019; Campos et al. 2021). This wet processing is active once a year during the coffee harvesting season (usually October-December). Untreated or partially treated effluents from the stations can easily enter the streams unless careful waste treatment and handling is undertaken; practices that vary among stations (Dadi et al. 2018; Chala et al. 2018).

Study design

Using a digital elevation model (DEM) with a spatial resolution of 30 m (ASTER DEM Product 2001) we generated a stream network map across our focal landscape of approximately 8000 km^2, using the hydrology tool in ArcGIS Pro 2.7 (Esri, 2020). Then, we selected 46 small perennial streams with catchment sizes ranging from 2 – 48 km^2 and marked on the map a suggestion of one sampling reach at the outlet for each selected catchment. The selection procedure of catchments aimed to capture a variation in dominant land use (forest, agroforestry and agriculture) and elevation, for which we used an early version of a land use map (see Section "Upstream land use composition" below). We used a GPS to locate the pre-selected sampling reach in the field. If the stream had a mixture of coarse substrates (sand, gravel, stones and rocks) in this place we marked a reach of 100 m of the stream for data collection (Fig. 1b). To get comparable samples, we discarded the pre-selected reach if it consisted of only fine sediments, and if so we walked along the selected stream (upstream or downstream) until the substrate conditions met our

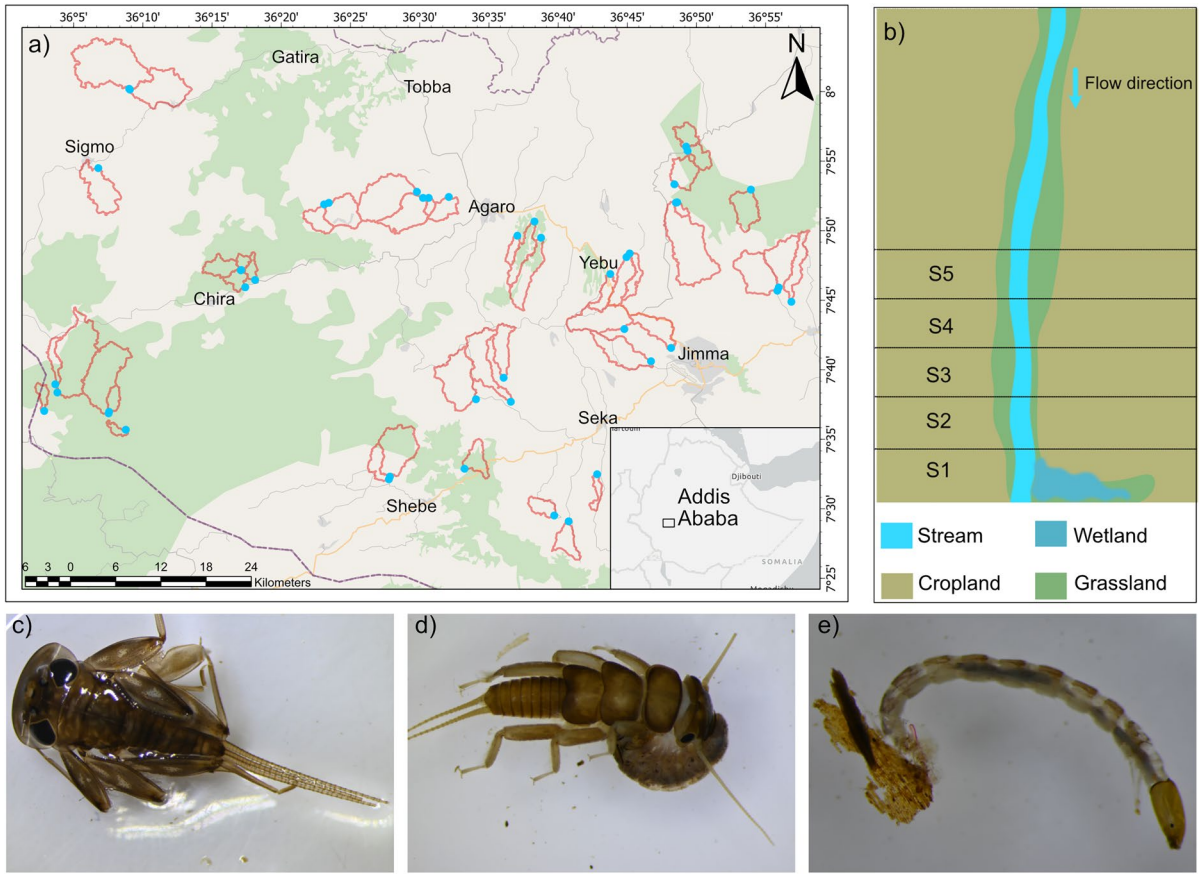

Fig. 1 Overview of the study system. **a** Location of the 46 catchments (boundaries marked by red lines) and sampling reaches (blue dots) in southwest Ethiopia (inset in the lower right corner: Map of Ethiopia with focal landscape indicated by a black rectangle). The green shaded areas represent forest dominated landscape while the remaining areas are predominantly occupied by agriculture or agroforestry (white), villages or towns (grey) or infrastructure including asphalt roads (yellow lines) or unpaved roads (grey lines). **b** Illustration of a sample reach (S1–S5 = 50 m) and the surrounding land use map sketched during fieldwork (their location is indicated by the blue dots at the outlet of the catchments in **a**. Focal invertebrate taxa **c** mayflies (Ephemeroptera: Heptageniidae), **d** stoneflies (Plecoptera: Perlidae) and **e** true flies (Diptera: Chironomidae)

criteria and marked our 100 m reach there instead. We took new coordinates at each selected reach to later be able to accurately recalculate the size and as well as the land use composition of each catchment. The lower part (50 m) of each selected reach was divided into 5 segments (S1–S5), which were used for macroinvertebrate sampling (Fig. 1b). All the field work for water quality, environmental variables and macroinvertebrate sampling was conducted from November 2022 to February 2023 (which is in the dry season with generally low precipitation).

Data on macroinvertebrates

At the middle of the five segments (Fig. 1b), macroinvertebrate samples were collected using a D-frame sampling net of 500 µm mesh (Natural History Book Service® WNB30-500D) for 2 min by disturbing all substrates to dislodge the macroinvertebrates from the stream bed (i.e. kick-sampling). We sampled each segment separately, starting with S1. Each sample was transferred to a white tray and macroinvertebrates were picked out and preserved in 96% ethanol in separate bottles for each segment and stored in a freezer for later identification. In the laboratory,

each specimen of mayflies (Ephemeroptera), stoneflies (Plecoptera) and true flies (Diptera) were identified to family level using identification keys (Nilsson 1996, 1997; Bouchard 2004). (Although Trichoptera are often included in these kinds of studies we did not have literature to be able to identify them to family). The rest of the samples were sorted to order level but not included in the analysis. We calculated an abundance weighted taxa water quality index for each site based on scorings of the different families' sensitivity. Each family was assigned a sensitivity score ranging from 1 to 10, theoretically allowing the taxa water quality index to span from 1 to 10 depending on its taxa composition. We assigned scores following a priority order: First we used the Biological Monitoring Working Party (BMWP) (Armitage et al. 1983), then for taxa not occurring in that source the biotic score system (ETHbios) (Aschalew and Moog 2015) and finally then Hindu Kush-Himalayan biotic score (HKHbios) (Ofenböck et al. 2010). For detailed description of how this was done see Supplementary file1.

Upstream land use composition

We developed a land use map using supervised classification of Sentinel-2 satellite images (with 10-m resolution) within the Google Earth Engine platform and local expert knowledge from the study area (previous knowledge of the landscape by Kristoffer Hylander). The final map has the following land use categories: forest, agriculture, agroforestry, urban, and wetland, and was evaluated using a validation dataset. For details on how the land use map was produced see Supplementary file2 in the supplementary information. We calculated the proportion of each land use type in each of the 46 study catchments. Secondly, we counted all the households in each catchment after digitizing the households from a high-resolution satellite image in Google Earth. Then we estimated the total number of households per area (in ha) of the catchment, hereafter referred to as settlement density. We used this metric of settlement density, assuming that it would better reflect household-related effects on water quality across our rural landscapes, compared to the urban class from the land use map. Thirdly, we identified all the coffee washing stations in the catchments by a combination of visual inspection of satellite images (coffee washing stations generally have a characteristic appearance) and interviews with smallholder farmers in the context of a related project (Hirko, unpublished). Then we created a simple index representing the number of and distance to coffee washing stations. We scored the catchments from 1 to 3 based on number of stations upstream of our sampling sites. We categorized the distances into two categories: stations within 1 km got a score of 2 and those beyond 1 km a score of 1. The final index was calculated by multiplying these numbers getting an index potentially ranging from 1 to 6 with a large value indicating a greater potential effect. The dominant land uses in the 46 catchments were either forests range (0 to 90%), agroforests (6 to 79%) or agriculture (0 to 89%) and their mean values are 23%, 27%, and 45% respectively. Settlement density varied from 0 to 1.4 households per ha and the coffee washing station index ranged from 0 to 4. For a summary of the land-use types and related parameters see Table S2 in Supplementary file2 and Fig S3 in Supplementary file2 for an example map of a catchment with its different land use. Note that we use the term 'land use' and 'land use composition' in many places in the text in a broad sense that encompass both the land use cover derived from the satellite image, settlement density and the coffee washing station index.

Data on water quality and other local environmental variables

At each site we first measured dissolved oxygen (DO), pH, turbidity, conductivity, and water temperature using a digital multiparameter portable meter (MultiLine® 3510 IDS). Then, at the same spot we collected two water samples for later additional analyses of nutrient and bacteria load, respectively. The samples were first stored dark in a cooling box and as soon as possible one of them was transferred to a freezer (−20 °C). Most samples were put in the freezer within 8 h, but for 13 remote sites only within 24–48 h. From the samples that had been frozen we measured total nitrogen and total phosphorus using the segmented flow analysis (SFA) method and total organic carbon as non-purgeable organic carbon (NPOC) after acidification and combustion at 800 °C at the Marine Ecology Laboratory of Stockholm University in Sweden. From the other sample (only those that reached the lab within 8 h) we incubated

a sub-sample to estimate two metrics of bacteria load (*E. coli* and coliform bacteria, respectively) using 3 M™ Petrifilm™ culture media following the standard procedure at the laboratory at Jimma University in Ethiopia (Hörman and Hänninen 2006; Bird et al. 2020). Coliforms are a group of bacteria found in soil, water and in human or animal waste. They are widespread and commonly serve as an indicator for assessing drinking water quality, while *E. coli* is one species of coliform bacteria that indicates faecal contamination. Especially *E.coli* load is a good indicator for the presence of pathogens (Canada Health 2012; Holcomb and Stewart 2020).

We recorded additional local environmental variables at each sample reach as follows. A land use map covering the 100 m sample reach and 30 m on each side was sketched by hand, from which we later calculated the stream buffer land use composition for each of the sample reaches (Fig. 1b). Canopy cover (%) was estimated from five canopy photos taken by a Canon IXUS 285 HS digital camera from the middle of the stream channel and then processed using the program ImageJ (Smith and Ramsay 2018). We visually estimated the substrate composition of the stream bed during the macroinvertebrate sampling for each of the segments and recorded it as the proportion (%) of sand, gravel, stones and rocks and later converted it to a substrate index based on the weighted average of the different classes. A water volume index was calculated by multiplication of the water depth and width as measured at the middle of the macroinvertebrate sampling segments.

Statistical analysis

For this study, we used three major data sets. For each of the 46 streams, we have: (1) land use variables at the catchment scale derived from satellite images, including settlement density and presence of coffee washing stations, (2) water quality (including both water chemistry and bacteria load of *E. coli* and coliforms), and local environmental variables namely stream bed substrate composition, canopy cover, reach buffer land use composition and elevation at the reach, and (3) macroinvertebrate biodiversity variables at the reach. All statistical analyses were performed in R 4.3.1 (R Core Team 2023). We checked multicollinearity among the explanatory variables using the *corrplot* package and then, if two explanatory variables were strongly correlated (correlation > 0.6), one was dropped. In the case of forest cover and agriculture cover, that were correlated, we included them in separate models (i.e. we ran two different models in most instances). For the piecewise structural equation modelling (pSEM) (see below), the model with agriculture showed qualitatively similar results to the model with forest (not shown). We standardized explanatory variables using the *scale* function to simplify the comparison of their importance.

Effects of land use on water quality including bacterial load

To assess the effects of upstream catchment land use on the composition of water quality parameters we modelled the multivariate matrix of eight water chemistry variables as a function of our focal land use variables (i.e. 'water chemistry composition ~ agriculture cover + agroforestry cover + wetland cover + settlement density + coffee washing station') using the multivariate method *adonis2* with argument *by* = *'margin'* in the *vegan* package (Oksanen et al. 2022). To visualize how the land use variables influenced the stream water chemistry we used a principal component analysis (PCA) of water chemistry variables overlaid with catchment land use variables using the *envfit* function in the *vegan* package. To further examine the effects of the land use composition on the water chemistry and pathogen load we ran separate generalized linear models (GLM) for each of eight water chemistry variables, *E. coli* and coliforms load with Gaussian distribution using the *MASS* package and a backward selection procedure retaining variables with a p-value < 0.1. Model assumptions were evaluated by inspecting diagnostic plots.

Effects of water quality and local environmental variables on invertebrate communities

The specimens sampled from the five segments of the sample reach were pooled and treated as one data point per site. To assess the response of the macroinvertebrate community composition to water quality and other local environmental variables we ran an *adonis2* analysis. For this we used a taxa-by-site

matrix encompassing the abundance of each taxon (family in our case) at each site as our response variable. As explanatory variables we a priori selected four water quality parameters (dissolved oxygen, turbidity, total nitrogen, total phosphorus) and three local variables (canopy cover, elevation, water volume index) to not overfit the models. To visualize how water chemistry and local environmental variables drove the composition of the invertebrate community we used non-metric multidimensional scaling (NMDS) with Bray–Curtis dissimilarity in the *vegan* package, with the water chemistry and local environmental variables overlaid using the *envfit* function.

We further tested if the selected explanatory variables could explain any variation in total abundance (all taxa pooled), taxa richness (i.e. number of families present in a sample), Shannon diversity index and taxa water quality index using separate GLMs with a Gaussian distribution. Similarly, for the 13 families recorded in more than 10 sites we ran separate GLM models with the same explanatory variables as in the previous analyses. We modelled abundance using a negative binomial error structure for the models of the families recorded in most sites and a binomial error structure using only presence/absence data for families which were absent in half of the sites (but still occurred in > 10 sites).

Direct effects of land use on invertebrate communities

The direct and indirect effects of catchment land use on macroinvertebrates were analysed using piecewise structural equation modelling (pSEM) using the piecewiseSEM package (Lefcheck 2016). We ran separate pSEMs for total abundance, taxa richness, Shannon diversity index, and taxa water quality index as well as separately for the abundance in the 13 most frequent families (same as above). The pSEM method allows to analyse complex ecological processes by breaking them into a series of paths as individual models with different distributions and then combine the models to build a causal model. In our case, several paths of linear models (water quality ~ land use; biodiversity ~ water quality + local environmental variables + land use) were combined in pSEM using GLMs with Gaussian, negative binomial or binomial distributions, depending on the data structure. To reduce the complexity of the models and due to the limited level of replication (46 sites) we selected only 8 explanatory variables in total (5 land use variables and 3 water chemistry and local environmental variables).

Results

Effects of land use on water quality in streams

The characteristic water quality profile of the 46 streams could be explained by a combination of upsteam land use parameters, where cover of agriculture and agroforestry explained the largest proportion of the variation ($r^2 = 0.39$ and $r^2 = 0.38$, respectively, adonis2, $p = 0.001$; Fig. 2a, Supplementary file1 Table S1). If forest cover, instead of proportion of agriculture, was used in the analysis the results were similar, but then forest cover explained more than the proportion of agroforestry (proportion of forest $r^2 = 0.45$ vs proportion of agroforestry $r^2 = 0.08$, adonis2, $p = 0.001$ and $p = 0.002$, respectively, Supplementary file1 Table S1).

In general, water chemistry variables indicating good water quality displayed higher values in catchments with high cover of forest. In contrast the water had a deteriorated quality (e.g. elevated turbidity, water temperature and nutrient concentration) in streams draining areas dominated by agroforestry or open agriculture (Fig. 2a, Supplementary file1 Table S1, and separate GLMs in Table S2). The highest turbidity (Fig. 2c) and total nitrogen (TotN) were in catchments with both high proportion of agroforestry and agriculture (separate GLMs in Supplementary file1 Table S2).

Although both streams draining agroforestry and agricultural land had poor water quality relative to streams draining forest covered catchments (Fig. 2a, Supplementary file1 Table S1 and S2), we found the water chemistry composition still differed in important ways between streams dominated by agroforestry and agriculture. For example, stream water temperature was more affected by agriculture (Fig. 2b, GLM, $p < 0.001$) than by agroforestry cover at the catchment scale (compare standardized coefficients in GLMs in Supplementary file1 Table S2). Moreover, total organic carbon (TOC) mostly increased in catchments with high amounts of agroforests, but was unaffected by the amount of agriculture (Supplementary file1 Table S2).

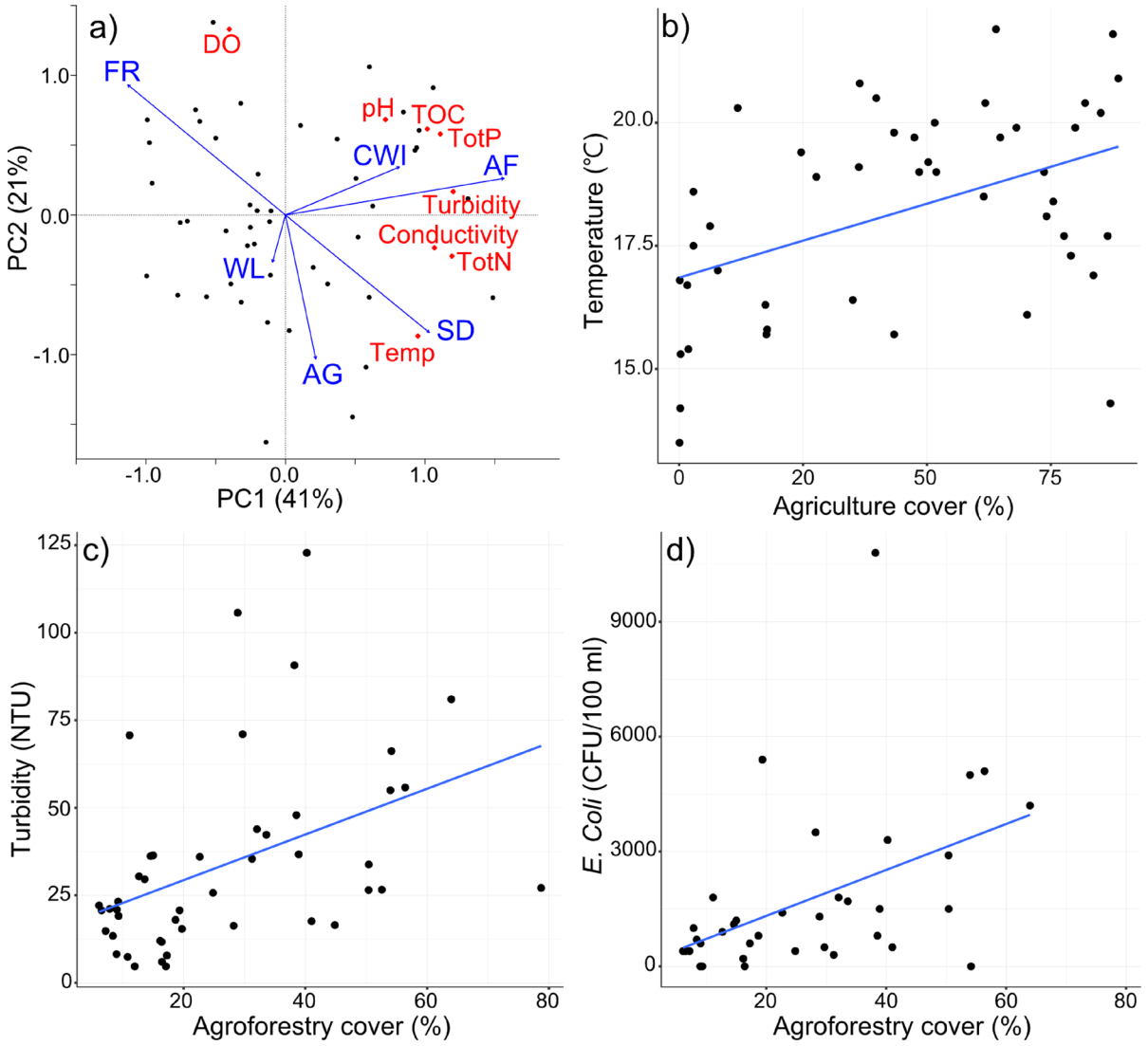

Fig. 2 Effects of catchment land use on water quality in 46 streams. **a** Principal component analysis (PCA) of water chemistry variables (in red) overlaid (fitted) with catchment land use variables (in blue) (*FR* Forest, *CWI* Coffee washing station index, *AF* Agroforestry, *WL* Wetland, *AG* Agriculture, *SD* Settlement density, *DO* dissolved oxygen, *TOC* Total organic carbon, *TotP* total phosphorous, *TotN* Total nitrogen). **b** to **d**, relationships of specific catchment land use and water quality variables namely **b** water temperature as a function of agricultural cover, and **c** turbidity as a function of agroforestry cover and **d** *E. coli* load as a function of agroforestry cover. A linear regression line is shown for the significant relationships. See Supplementary file1 Table S1 and S2 for the statistical tests

Although the proportion wetland cover is generally low in these landscapes, turbidity, temperature and nutrient concentration (e.g. nitrogen) were lower in streams with a higher proportion of wetland (Supplementary file1 Table S2) and wetlands contributed to understanding the overall water quality profile of the streams (adonis2, $r^2 = 0.12$, Supplementary file1 Table S1).

Settlement density reduced dissolved oxygen (DO, GLM, $p = 0.006$, Supplementary file1 Table S2) and in streams in catchments with a high coffee washing station index there was an increased pH (GLM, $p = 0.002$, Supplementary file1 Table S2).

Higher amount of *E. coli* and coliform bacteria in general were recorded in streams running through landscapes with a high proportion of agroforestry (Supplementary file1 Table S2, Fig. 2d). Coliform load was lower in catchments with high cover of forests and wetlands, whereas agroforestry cover had an opposite effect (Supplementary file1 Table S2).

Effects of water quality and local environmental variables on invertebrate communities in streams

In total, we sampled 31,421 specimens belonging to the three selected orders of which 75% were mayflies (Ephemeroptera), 16% true flies (Diptera) and 9% stoneflies (Plecoptera) (Supplementary file1 Table S7). The number of families ranged from 6 to 14 in the different streams. The composition of the invertebrate community of the 46 streams could be explained by a combination of water quality and local environmental variables, where dissolved oxygen (DO, $r^2 = 0.044$) and total nitrogen (TotN, $r^2 = 0.045$, Supplementary file1 Fig S3 and Table S3) explained the largest proportion of the variation.

The taxa water quality index was higher, i.e. indicating better water quality, in streams with higher dissolved oxygen. The index value ranged from 4.7 to 9.3 and the mean index for the 46 streams was 7.0 (Fig. 3a, DO, GLM, $p = 0.014$, Supplementary file1 Table S5). Total abundance had a significant negative relationship with elevation, but not with any of the other variables (GLM, elevation $p = 0.035$, Supplementary file1 Table S6). Taxa richness had a significant positive relationship with dissolved oxygen (GLM, $p = 0.021$, Supplementary file1 Table S6, Fig. 3b), whereas the stream invertebrate diversity was significantly higher in streams with higher local canopy cover ($p = 0.009$) (Supplementary file1 Table S6).

Different families responded to stream water quality and local environmental variables in different ways (GLM on each family, Supplementary file1 Table S5). For example, the abundance of Perlidae (common stoneflies) increased, whereas the abundance of Caenidae (small square-gill mayflies) decreased, with dissolved oxygen in stream water (Fig. 3c, d and Supplementary file1 Table S5).

Direct effects of land use on invertebrate communities

There were significant direct positive effects of the coffee washing station index on total abundance ($p = 0.006$) and taxa richness ($p = 0.049$) when analysed with piece-wise structural equation models (Supplementary file1 Table S4).

Land use variables repeatedly showed direct significant effects ($p < 0.05$), or trends ($p < 0.1$), on the abundance of several different families when analysed separately with piece-wise structural equation models; a pattern that was not detected via the measured water chemistry variables (Table 1, pSEM). Settlement density and coffee washing stations were the two land use variables that most often had direct effects on the abundances. For example, settlement and coffee washing stations favoured the abundance of Chironomidae (Chironomids, non-biting midges), which is considered the most tolerant family among the focal taxa (Fig. 4a), while Perlidae (common stoneflies), which is often regarded the most sensitive (Fig. 4b) were directly negatively affected by the same land use variables. However, there were also inconsistencies with for example Ephemerellidae that were positively affected by both settlements and coffee washing stations despite having a high taxa score (Table 1).

Discussion

We assessed the effects of upstream land use, including variables such as cover of agriculture, settlement density and coffee washing stations, on water quality and invertebrate communities in 46 perennial streams in southwest Ethiopia where the land use composition is heterogenous and people heavily rely on surface water for their livelihoods. In this landscape, assessing and monitoring stream water quality has both ecological and public health implications. As we expected, we found better water quality in streams draining through forest than open agriculture for most of the water chemistry variables. However, several aspects of the water quality were equally poor or even worse in streams with a high coffee agroforestry cover as compared to open agriculture, despite a higher tree cover in the former. This was also true for the *E. coli* and coliforms concentrations, which

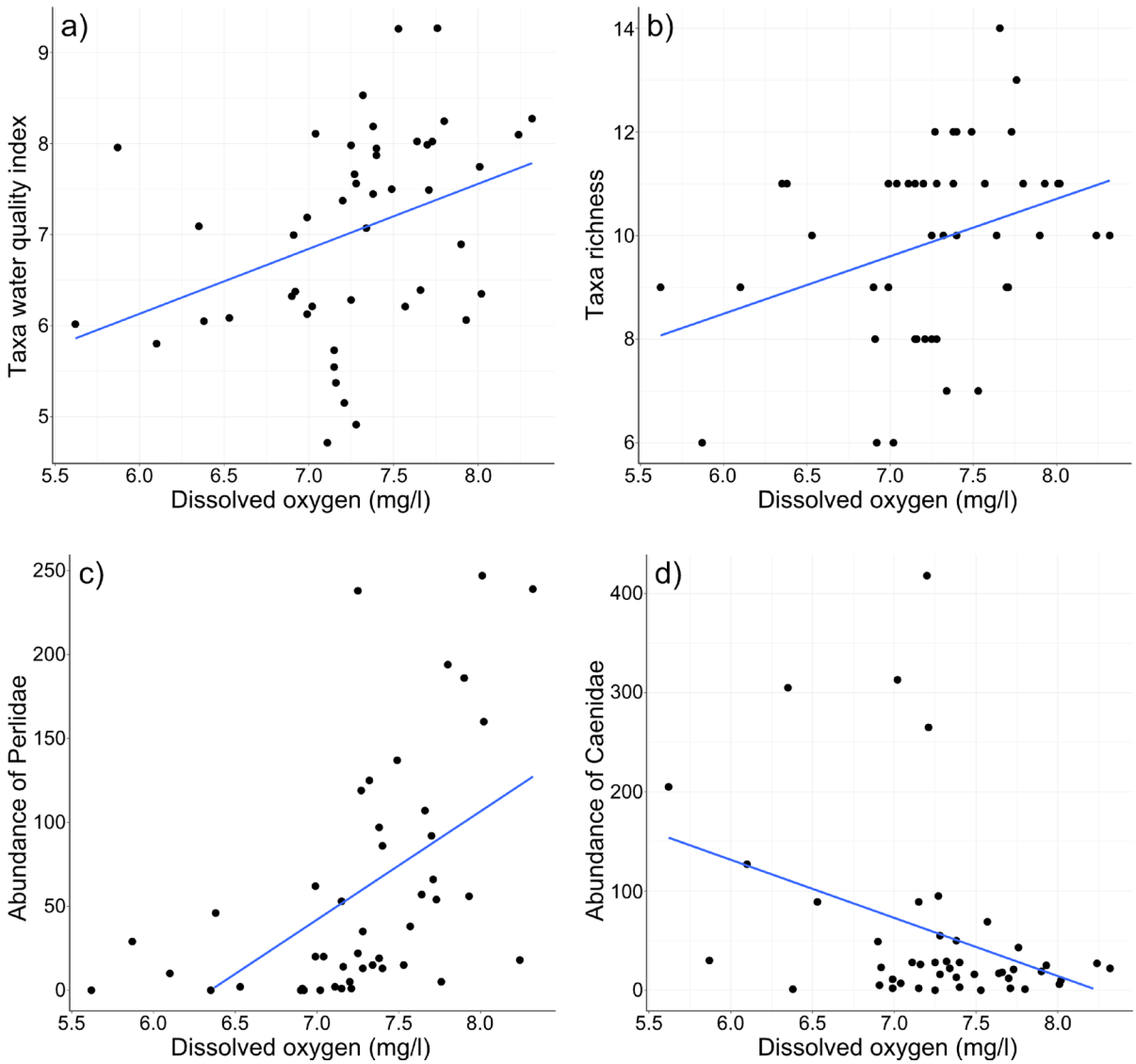

Fig. 3 The effects of water chemistry on the invertebrate community in 46 streams. Relationships between dissolved oxygen and **a** taxa water quality index, **b** taxa richness, **c** the abundance of Perlidae (stoneflies), and **d**) the abundance of Caenidae (small squaregill mayflies). A single regression line is shown for the significant relationships. See Supplementary file1 Table S5 and S6 for details of the statistical analyses from the generalized linear models

were in general high, but especially so in catchments with a high proportion of agroforestry. The composition of stream invertebrates was strongly affected by the water quality, where certain sensitive groups only had high abundances in streams with good quality (e.g. well oxygenated), while others were tolerant to various types of water quality deterioration. When exploring possible direct effects of upstream activities on the invertebrate communities we found that both positive and negative effects of settlement density and coffee washing stations were often missed in the water chemistry analyses. One reason might be that seasonal activities that negatively affect water quality and the stream biota might go unnoticed during a single (snapshot) water quality measurement. The study highlights the need of improved land use and water management practices, based on the results from both the water chemistry, bacteria load and stream

Table 1 Familywise piecewise structural equation modelling (pSEM) summary and taxa sensitivity score (high values = good water quality, low values = poor water quality).

Families	Taxa score*	Direct effects of land use	Dissolved oxygen (mg/l)	Turbidity (NTU)	Total nitrogen (µg/l)	Model
Baetidae	4					Negative binomial
Caenidae	7	Settlement density (+), Coffee washing station index +				Gaussian (on log transformed)
Ceratopogonidae	5	Coffee washing station index +				Binomial
Chironomidae	2	Settlement density (+), Coffee washing station index (+)		−		Gaussian (on log transformed)
Ephemerellidae	10	Wetland −, Settlement density +, Coffee washing station index (+)	+			Negative binomial
Heptageniidae	10					Gaussian (on log transformed)
Leptophlebiidae	10	Coffee washing station index (−)				Binomial
Limoniidae	8			−	+	Gaussian (on log transformed)
Muscidae	2	Agriculture (+), Coffee washing station index (+)				Binomial
Perlidae	10	Settlement density −, Coffee washing station index −	+	+		Negative binomial
Simuliidae	5	Wetland −			(−)	Gaussian (on log transformed)
Tabanidae	6	Agriculture +, Agroforestry (+), Wetland −, Settlement density (−)	+			Binomial
Tipulidae	5			−	+	Negative binomial

The positive and negative significant effects ($p < 0.05$, signs without parentheses) or trends ($p < 0.1$, signs in parentheses) of upstream land use are indicated by + and – signs (open agriculture land retained in the model)

*Taxa score ranges from 1 to 10. Families with high taxa scores represent sensitive taxa while those with low scores represent tolerant taxa. Each family was assigned a score using hierarchical approach. First checking BMWP (Armitage et al. 1983), then ETHbios (Aschalew and Moog 2015) and finally HKHbios (Ofenböck et al. 2010) if the taxon was not listed in the previous scoring systems

invertebrate assessments. Notably this is also true for the agroforestry landscapes, that in many other respects provide improved environmental conditions such as biodiversity conservation, carbon sequestration and various other ecosystem services (Kay et al. 2019; Udawatta and Gantzer 2022, but see, de Jesús Crespo et al. 2020).

Impact of land use on water quality in streams

The relationships between the open agriculture land use type and certain water quality variables (e.g. increased nutrient load, turbidity and water temperature) were expected results. Other studies have also reported similar results, showing that the application of fertilizers, regular soil disturbance during cultivation, overgrazing, livestock trampling, higher settlement density and waste from households trigger deterioration of water quality in human-modified landscapes (Mello et al. 2018; Xu et al. 2022; Locke 2024). Agroforestry has been highlighted as a promising strategy to mitigate agricultural pollutants by reducing excess nutrient and pesticide leaching and runoff (Pavlidis and Tsihrintzis 2018). However, surprisingly, in our study system the proportion of agroforestry in catchments had similar effects as agriculture on many of the water quality variables, such as nutrient pollution, turbidity and bacteria load, despite a higher tree cover. In other words, catchments with high cover of agroforestry could not in general be

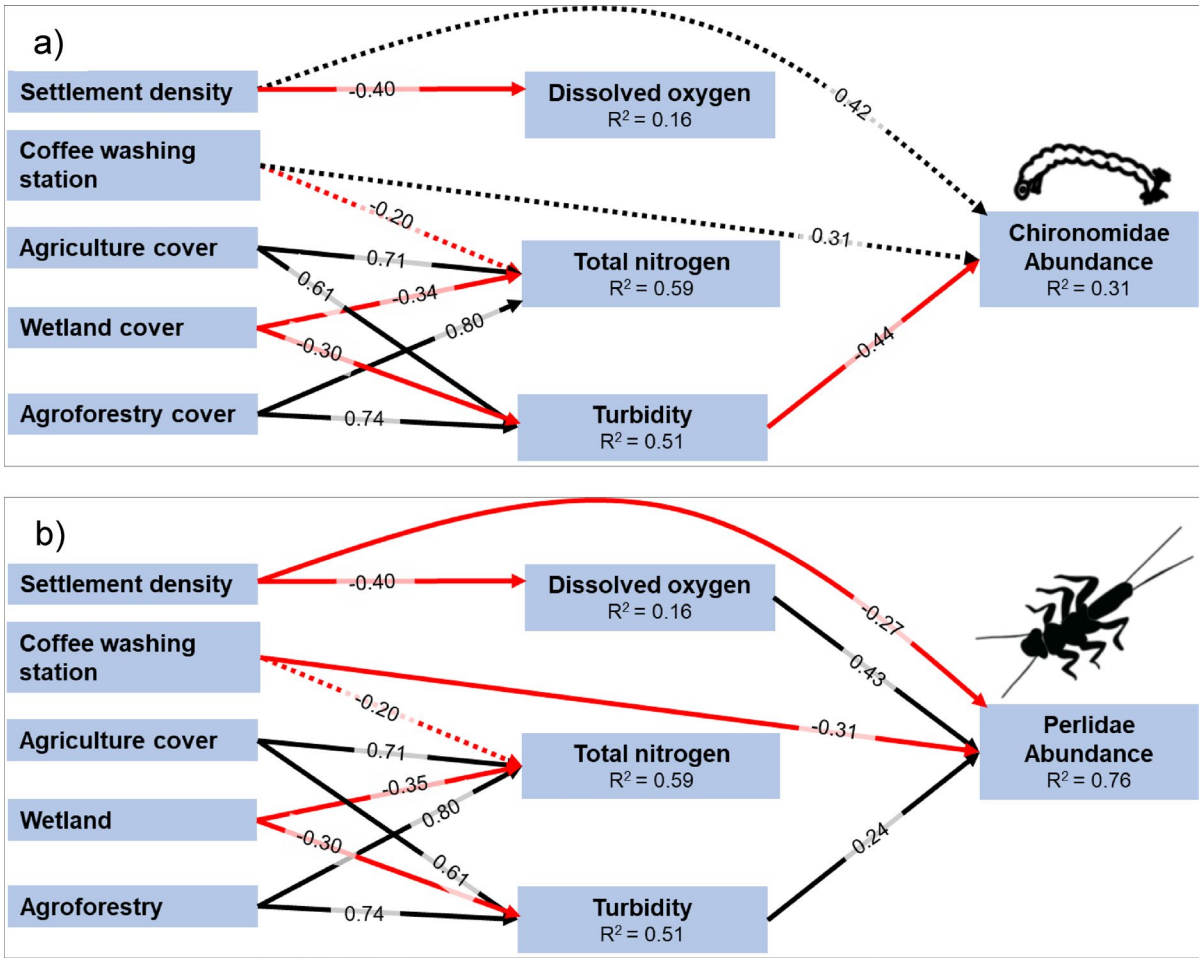

Fig. 4 Direct and indirect effects of land use and water chemistry on a) the abundance of Chironomidae (non-biting midges; true fly family) (pSEM, the model global goodness-of-fit: AIC = 937, Fisher's C = 9,83 with p-value = 0.88, df = 16) and b) the abundance of Perlidae (Common stoneflies) (pSEM, the model global goodness-of-fit: AIC = 1246, Fisher's C = 9.83 with p-value = 0.88, df = 16). Solid lines indicate significant effects ($p < 0.05$), dotted lines indicate trends ($p < 0.1$) and nonsignificant effects links were removed. Positive and negative effects are colored in black and red lines, respectively. See Table 1 for results from the pSEMs of all families

characterized as intermediate between forest and agriculture in terms of the water quality as expected based on prior studies (Kay et al. 2019; Udawatta and Gantzer 2022). The only variable that was intermediate was water temperature, that naturally is higher in open agricultural landscapes without shade. The unexpected pattern of relatively poor water quality in agroforestry landscapes might be due to that the human activities and disturbances in agriculture landscapes (application of pesticides and herbicides, soil disturbance and waste from households which is unresolved in our study) are also common in agroforestry dominated landscapes as coffee farmers also produce food crops and have livestock (Melese and Kolech 2021; Daba et al. 2023). Moreover, wastewater effluent from wet coffee washing stations and organic waste (a by-product from coffee processing) could be an additional burden in the agroforestry landscapes as most of such stations were located close to streams.

Although the responses varied among different variables, in general land use under natural forest improved water quality (e.g. reduce water temperature, nutrient load and turbidity), which many researchers also attribute to the shading effects of

trees, less human activities, higher filtering capacity (purification function) of trees in forest landscape (dos Reis Oliveira et al. 2019; Piffer et al. 2021). Similarly, despite a low proportion of wetland area in many of the catchments, wetlands also improved water quality in a similar fashion as forest cover. Previous studies also highlight that wetlands keeps the water temperature low (Kadlec 2006) and play an essential role in aquatic ecosystem purification by trapping or filtering sediment and nutrients before they reach the surface water such as streams, rivers or lakes (Wang et al. 2021; Gonzalez-Flo et al. 2023; Ferreira et al. 2023). Our study indicates that the *E.coli* loads were lower in such catchments, which is interesting since sometimes wetlands areas are associated with other types of health risks for humans, such as malaria and other waterborne diseases (Cools et al. 2013). Although there was variation in *E. coli* loads among the streams, most recorded concentrations exceeded both WHO and U.S. Environmental Protection Agency's standards (above 1000 CFU/100 ml in most study site streams, compared to the recommended below 100 CFU/100 ml for bathing and recreational water use Wiedenmann et al. 2006; Tiwari et al. 2021)), raising significant public health concerns in these landscapes. In the Southern region of Ethiopia Alemayehu et al. (2020) reported a similar finding, showing that more than 15% of water sources for domestic use are contaminated with *E. coli* to the levels considered high risk or above. We did not find a significant signal between bacteria load and the density of settlements. This could be because our measurements only represented a single event in time, but maybe also since our metric of settlement density does not capture important differences in location and distance of households to the focal measurement point. It is important to note that some water quality variables are more likely to be affected by local conditions closer to the focal sampling point, while others may reflect conditions across the entire catchment. Comparisons between sites with similar land use compositions but with different spatial configurations (i.e., where in a catchment different land use types are located) would therefore be needed in order to separate the effects in future studies.

Impact of water quality and local environmental variables on invertebrate communities

We found that the variation in the response of invertebrate communities to changes in water quality and local conditions was better reflected by variation in taxa composition rather than differences in diversity and total abundance. When we ran the analyses separately for different taxa, the general pattern was that the variation in abundance could be attributed to their different niches, which was expected and in line with others result (Armitage et al. 1983). For example, the abundance of Ephemerellidae (spiny crawler mayflies) and Perlidae (common stoneflies) were favoured in streams with high levels of dissolved oxygen, whereas the true flies (e.g. the families Ceratopogonidae and Simuliidae) were common or less affected in almost all streams, including those with higher turbidity and nutrient loads. Streams draining higher elevations or catchments with a large proportion of forest cover favoured the occurrence of common stoneflies, as stoneflies are sensitive to high water temperature. They prefer cool and well-oxygenated streams, whereas mayflies are often adapted to warmer waters, even if they also are considered sensitive (Brittain 1990). Also, the pSEM-models in general showed that many sensitive taxa were less common in streams affected by settlements and wet coffee washing stations and vice versa for tolerant taxa. However, the pattern was not totally consistent with much variation among families. For example, settlement density and coffee washing stations positively affected the abundance of Ephemerellidae, despite that they were negatively affected by low oxygen levels and are considered a sensitive family. Possible explanations for these results might be that our low taxonomic resolution (family level) possibly mask varying responses of different species within families to different niche parameters (Powell et al. 2023; Heß et al. 2024).

The poor knowledge in many countries with unmonitored streams about which freshwater species are present as well as their ecological requirements is alarming given that the threats to them are often greater than those faced by terrestrial (and better known) species (Wallace and Webster 1996; Sundar et al. 2020). It is interesting to note that even streams with rather poor water quality support abundant invertebrate taxa, indicating that high macroinvertebrate abundance alone does not necessarily signal

good water quality (Powell et al. 2023). In Germany, Heß et al. (2024) also highlighted an increase in the abundance of tolerant taxa in rivers with high pollution. The lack of correlation between diversity and water quality in our system, together with the fact that most of our streams had a taxa water quality index above 6 (which is an intermediate value at the scale from 1 to 10), would indicate that many of our investigated streams still contain a valuable macroinvertebrate fauna, even if we do not have any other reference landscape to compare with (neither with better or worse water quality). Yet, we see clear negative effects from many human activities. In the future, climate change and agricultural intensification and its indirect influence through land use changes (Hylander et al. 2024) could potentially accelerate the degradation of the investigated stream network and lead to the loss of yet unknown but ecologically important species.

Direct effects of land use on invertebrate communities in streams

By using piecewise structural equation modelling (pSEM) we revealed that settlement density and coffee washing stations likely are more important than our water chemistry data show, in terms of negatively affecting the stream water quality and its biodiversity. The observed direct effects of upstream land use implies that we might have missed measuring certain water quality variables. Another explanation could be that certain variables vary much over the year and that the peaks of bad water quality were not captured by our snapshot assessment, and that those events have a disproportional importance in the regulation of the invertebrate community. For example, coffee washing stations are active during the coffee harvesting seasons and then the flushes of organic debris and wastewater drain with the surface runoff to the streams (Beyene et al. 2012). Similarly, agronomic practices (e.g. application of fertilizer, pesticide, herbicide, ploughing, weeding) in cropland (Aliyi et al. 2018; Jirata et al. 2024) and coffee farms are also more prevalent during a certain period of the year. Hence, the response of the invertebrate taxa composition might rather reflect the water quality conditions over a longer time period, integrating temporal variation that our single event measurement likely missed. This is among the main reasons why different indices using macroinvertebrates have been developed and promoted as an effective strategy for monitoring stream water quality and aquatic ecosystem health. Examples of such indices from Africa include the Ethiopian 'ETHbios', the South African scoring system (SASS) and Namibian scoring system (Dallas 1997; Palmer and Taylor 2004; Aschalew and Moog 2015). However, even if such approaches are useful in many contexts our results suggest that it can be more complex. The simple taxon index we used, did indeed correlate with dissolved oxygen, but on the other hand the model for this index did not improve in the pSEM where settlement density and coffee washing stations were included. Moreover, many taxa responded individualistically to different water quality parameters. These findings highlight the need for caution when using invertebrate communities as the only direct proxy for water quality and suggest that a combination of biodiversity measures and regular monitoring of chemical, physical and biological parameters is important. Moreover, the variation in the biodiversity per se is important to understand and protect.

Implications for land use management to reach sustainable development goals (SDGs)

Freshwater and associated aquatic biodiversity are given great emphasis in goal 6 of the UN SDGs due to the ecological and economical importance of this vital resource. In our study system, variation in upstream land use was important in explaining the variation of water quality and invertebrate communities in streams. We have three major points that highlight the need for revisiting the existing waste management strategies and to explore the spatial arrangement of land use in stream buffer zones to maximize the potential ecosystem services that could be delivered from agroforestry landscapes and wetlands.

First, despite a low proportion of wetland cover in this landscape, our results showed that wetlands provided important ecosystem services in terms of cooling water temperature and reducing nutrient pollution, turbidity and *E. coli* load. Wetlands are continually declining in Ethiopia, due to human pressures such as crop cultivation, overgrazing, irrigation, ditching, damming, urban expansion and

waste dumping (Mereta et al. 2012; Fentaw et al. 2022). This decline is very worrying not only from a conservation of aquatic biodiversity point of view, but also since it will exacerbate the deterioration in water quality of streams and rivers used for drinking water and other domestic purposes (Wood and Armitage 1997; Arsiso et al. 2017; Tefera et al. 2023).

Second, our results suggest that agroforestry-dominated landscapes exert significant negative effects on water quality and freshwater biodiversity, in the same order of magnitude as agriculture. While this land use type has been widely promoted for biodiversity and sustainable land management in various conservation literature (Garrity 2012; De Beenhouwer et al. 2016) there are studies which have reported water quality decline in coffee agroforestry systems (Martínez et al. 2009; Vázquez et al. 2011; García-García et al. 2017), though their local context may vary from ours. With a rapid development in this area and increasing demand of coffee on the global market (Campos et al. 2021; Duguma et al. 2021), coffee agroforestry is likely to increase in the future via intensification and via expansion of coffee growing range into higher elevations. Further investigations are needed to better utilize the beneficial effects of trees in agroforestry and mimic the processes in natural forests to improve water quality. As a starting point, we suggest adopting and evaluating different stream buffer management and restoration techniques such as determining optimum stream buffer cover and width, and manipulating the spatial arrangement of coffee shade trees along streams to improve their effectiveness in trapping sediment and filtering surface runoff. Moreover, it could be interesting to look for alternative options for drinking water supplies for cattle, which may help reduce direct water contamination from cattle.

Third, better handling of wastewater from both humans and livestock in settlements and from wet coffee washing stations is likely very important in these landscapes. Deterioration of water quality due to increased turbidity and bacteria concentrations in this landscape could threaten human health. Moreover, this degradation may potentially create an additional burden for drinking water supply and treatment, especially as urban centres increasingly consider surface water (streams and rivers) as an alternative to groundwater to meet drinking water demands (Arsiso et al. 2017; Tefera et al. 2023). In the future, modernization and climate change are expected to drive intensification of coffee production and its spatial expansion into new growing regions (e.g. higher elevations) (Hylander et al. 2024), which may result in more settlements and wet coffee washing stations, resulting in increased waste generation. Therefore, as headwater streams are important for downstream ecosystem services, urgent action is needed to improve wet coffee processing wastewater treatment and handling as well as developing wastewater treatment from settlements.

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Author contributions O.H, S.H, A.T and K.H conceived the idea and designed the study. N.G and F.V.M developed the land cover map for the study area. O.H conducted the field work and identified the specimens to taxa. O.H led the analyses with support mainly from KH and led the writing of the original draft. All authors contributed critical revisions at different stages of the manuscript and gave final approval for publication.

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Declarations

Competing interests The authors declare no competing interests.

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