# Assessment of Ecological Health Using Benthic Macroinvertebrates and Hydrological Alteration in Gilgel Abay River, Ethiopia

**By**

**Endalh Mekonnen Mengistie**

**A thesis Submitted to Addis Ababa University in Partial Fulfillment of the Requirements for the Degree of Master of Science in Aquatic Ecosystems and Environmental Management (AEEM)**

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# ADDIS ABABA UNIVERSITY

## SCHOOL OF GRADUATE STUDIES

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# DECLARATION

I, the undersigned, declare that this thesis was written entirely by myself and that it has not previously been presented, in whole or in part, in any application for a degree or other qualification. Except as referenced or acknowledged, the work provided in this thesis is completely mine.

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# LIST OF ABBREVIATIONS AND ACRONYMS

| | |
|---|---|
| **ASPT** | Average score per taxon |
| **ETHbios** | Ethiopia biotic score |
| **GPS** | Global positioning system |
| **HQA** | Habitat quality assessment score |
| **IHA** | Indicators of hydrologic alteration |
| **GSM** | Gravel-sand-mud |
| **MHS** | Multi habitat sampling scheme |
| **MoWIE** | Ministry of water, irrigation and electric |
| **FFG** | Functional feeding groups |

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## ABSTRACT

*Bio-monitoring and hydrological flow alterations are currently given due attention as they appropriately reflect the health of aquatic ecosystems. This study aimed to assess the ecological health of the Gilgel Abay River using benthic macroinvertebrates and hydrological flow-alteration metrics. Flow alteration analysis was monitored using the Indicators of Hydrological Alteration (IHA) with observed discharge data gathered from 1979 to 2020. Five sampling sites were selected based on major stressors including; water abstraction for irrigation, deforestation, fishing, cloth washing, bathing, swimming, spiritual purpose, livestock watering, grazing, and the accessibility of the sites for sampling. In-situ environmental variables such as dissolved oxygen, pH, temperature, and conductivity were measurement using a multiline-probe system (HQ40d) and the nutrients including, nitrate, nitrite, soluble reactive phosphorus,and total suspended solid laboratory analysis were carried out in 2023 folowing APHA, 1995 protocol. Benthic macroinvertebrate samples were collected following a multi-habitat sampling approach and identified to family level except in the case of Chironomidae which were identified to species level. Both 1-day and 90 days durations of high flow and low flow had a decreasing trend over the past 42 years (1979–2020). The environmental flow component analysis found four large floods (i.e., 1980, 1986, 1988, and 1994), but no large flood was recorded after 1995. The low flow conditions of the stream flow were equal to 15.81$m^3s^{-1}$ for 95% of the time in the flow duration curve. Highest concentrations of $NO_3^-$ (4.25 mg/l), TSS (0.065 mg/l), $NH_4^+$ (0.622 mg/l), were recorded at the Telifa site, while lowest levels of soluble reactive phosphorus (0.395±0.5 mg/l), $NO_3^-$ (1.23 mg/l), $NO_2^-$ (0.008 mg/l) and the highest concentration of dissolved oxygen (7.74 mg/l) were recorded at the reference site. Redundancy analysis indicated that dissolved oxygen, $NO_2^-$, TSS, $NO_3^-$, and pH directly impacted macroinvertebrates distribution. All environmental variables except $NH_4^+$ and SRP have significantly different among sampling sites. The overall qualitative habitat assessment values of the river was 59.3%, indicating Class D (largely modified) based on the intensity of human influence. A total of 28 family benthic macroinvertebrate taxa under 9 orders were collected. The families Caenidae, Hydropsychidae, Heptageniidae, Tricorythidae, and Naucoridae were frequently encountered in minimally impacted upstream sampling sites and considered as indicators of relatively minimally impacted sites. The family Chironomidae was dominant in highly impacted sites and can be considered as indicators of impaired conditions. Polypedilum wittei was dominant in highly agricultural activities impacted sites and can be considered as an indicator of highly polluted sites in the study river. Metrics composed of %Ephemeroptera and Trichoptera, percent dominant taxon, %Diptera individuals, Shannon-Wiener diversity index (H), total number of taxa (family), percent dominant functional group, ETHbios, and habitat quality assessment score were able to differentiate reference sites, and agriculturally impacted sites. Altered river flows in the Gilgel Abay likely harm aquatic life. Studies should focus on water requirements for the ecosystem, bank stability, and local community-centred approaches to watershed conservation structures.*



**Keywords/Phrases:** Benthic macroinvertebrate, Bioindicator, Environmental flow, Functional feeding groups, Hydrological regime

## ማጠቃለያ

*የባዮ-ክትትል እና የሃይድሮሎጂ ፍሰት ለውጦች የውሃ ሥነ ምህዳሮችን ጤና በአግባቡ ስለሚያንፀባርቁ በአሁኑ ጊዜ ተገቢ ትኩረት ተሰጥቷቸዋል ። ይህ ጥናት የግልገል አባይ ወንዝ ሥነ ምህዳራዊ ጤናን ለመገምገም የታለመ ሲሆን በውሃ  የጀርባ አጥንት የሌላቸው ነፋሳት እና የሃይድሮሎጂ ፍሰት-ለውጥ መለኪያዎችን በመጠቀም ነበር ። ከ1979 እስከ 2020 የተሰበሰቡ የተመለከቱ የፍሳሽ ማስወገጃ መረጃዎችን በመጠቀም የፍሰት ለውጥ ትንታኔ በሃይድሮሎጂካል ለውጥ ጠቋሚዎች (አይኤችኤ) ተጠቅሞ ክትትል ተደርጓል ። አምስት የናሙና አሰባሰብ ቦታዎች የተመረጡት ለመስኖ፣ ለደን መጨፍጨፍ፣ ለዓሣ ማጥመድ፣ ለጨርቅ ማጠብ፣ ለመታጠብ፣ ለመዋኘት፣ ለመንፈሳዊ ዓላማ፣ ለከብት ማጠጣት፣ ለመንጋ፣ ናሙና አሰባሰብ ቦታዎችን ተደራሽነት ጨምሮ በዋና ዋና የጭንቀት ሁኔታዎች ላይ ተመስርተው ነው። በቦታው ላይ ያሉ የአካባቢ ተለዋዋጮች ለምሳሌ የተሟሟ ኦክስጅን ፣ ፒኤች ፣ የሙቀት መጠን እና ማስተላለፊያ በበርካታ መስመሮች-መርማሪ ስርዓት (HQ40d) በመጠቀም የተለካ ሲሆን ንጥረ ነገሮችን ጨምሮ ናይትሬት ፣ ናይትሬት ፣ የሚሟሟ ምላሽ ሰጭ ፎስፈረስ እና አጠቃላይ የተንጠለጠለ ጠንካራ የላቦራቶሪ ትንተና በ 2023 ተካሂዷል ። በውሃ  የጀርባ አጥንት የሌላቸው ነፋሳት ናሙናዎች የተሰበሰቡት በበርካታ መኖሪያ አካባቢዎች ናሙና በመውሰድ ሲሆን በቤተሰብ ደረጃ የተለዩ ናቸው ፣ በኪሮኖሚድስ ጉዳይ ላይ ግን በዘር ደረጃ የተለዩ ናቸው ። ባለፉት 42 ዓመታት (1979 ⁇ 2020) የከፍተኛ ፍሰት እና ዝቅተኛ ፍሰት የ 1 ቀን እና የ 90 ቀናት ጊዜያት የመቀነስ አዝማሚያ ነበራቸው ። የአካባቢያዊ ፍሰት አካል ትንተና አራት ትላልቅ ጎርፍ (ማለትም ፣ 1980 ፣ 1986 ፣ 1988 እና 1994) አገኘ ፣ ግን ከ 1995 በኋላ ምንም ትልቅ ጎርፍ አልተመዘገበም ። የኸረት ፍሰት ዝቅተኛ ፍሰት ሁኔታዎች በፍሰት ቆይታ ኩርባ ውስጥ ለ 95% ጊዜ $15.81 m^3s^{-1}$ እኩል ነበሩ ። ከፍተኛው የ $NO_3^-$ (4,25 mg/l) ፣ TSS (0,065 mg/l) ፣ $NH_4^+$ (0,622 mg/l) ፣ በቴሊፋ ቦታ ላይ ተመዝግቧል ፣ ዝቅተኛው የሚሟሟ ምላሽ ሰጪ ፎስፈረስ (0,395±0,5 mg/l) ፣ $NO_3^-$ (1,23 mg/l) ፣ $NO_2^-$ (0,008 mg/l) እና ከፍተኛው የተሟሟ ኦክስጅን (7,74 mg/l) በመረጃ ቦታ ላይ ተመዝግቧል ። የተሟላነት ትንተና እንደሚያመለክተው የተሟላ ኦክስጅን ፣ $NO_2$ ፣ TSS ፣ $NO_3$ እና pH በቀጥታ በውሃ  የጀርባ አጥንት የሌላቸው ነፋሳት  ስርጭት ላይ ተጽዕኖ አሳድሯል ። ከኤንኤች 4+ እና ኤስአርፒ በስተቀር ሁሉም የአካባቢ ተለዋዋጮች በናሙና ጣቢያዎች መካከል ከፍተኛ ልዩነት አላቸው ። የወንዙ አጠቃላይ ጥራት ያለው የኑሮ ደረጃ አሰጣጥ እሴቶች 59,3% ነበሩ ፣ ይህም በሰው ልጅ ተጽዕኖ ጥንካሬ ላይ በመመርኮዝ D ክፍል (በከፍተኛ ሁኔታ የተቀየረ) መሆኑን ያሳያል ። በጠቅላላው 28 የቤተሰብ በውሃ  የጀርባ አጥንት የሌላቸው ነፋሳት ዝርያ በ 9 ክፍለ-መደብ  ውስጥ ተሰብስበዋል ። የኬኒዴይ፣ ሃይድሮፕሲኪዴይ፣ ሄፕታጌኒዴይ፣ ትሪኮርቲዴይ እና ናውኮሪዴይ ቤተሰቦች በአነስተኛ ተፅእኖ በተጎዱ የላይኛው የናሙና ናሙና ጣቢያዎች ውስጥ በተደጋጋሚ የተገኙ ሲሆን በአንፃራዊነት አነስተኛ ተፅእኖ ባላቸው ጣቢያዎች አመላካች ሆነው ተወስደዋል ። የካሮኖሚዳኢ ቤተሰብ በከፍተኛ ሁኔታ በተጎዱ አካባቢዎች የበላይ ነበር እናም የተበላሹ ሁኔታዎችን አመላካች ተደርጎ ሊወሰድ ይችላል ። ፖሊፔዲሊየም ዊቲ በከፍተኛ ደረጃ በግብርና እንቅስቃሴዎች በተጎዱ ቦታዎች ላይ የበላይ ነበር እናም በጥናቱ ወንዝ ውስጥ በጣም የተበከሉ ጣቢያዎች አመላካች ሆኖ ሊታይ ይችላል ። % ኤፌሜሮፕቴራ እና ትሪኮፕቴራ ፣ መቶኛ የበላይ ታክሰን ፣ % ዲፕቴራ ግለሰቦች ፣ የሻነን-ዊነር ብዝሃነት መረጃ ጠቋሚ (ኤች) ፣ አጠቃላይ የታክሶዎች ብዛት (ቤተሰብ) ፣ መቶኛ የበላይነት ያለው ተግባራዊ ቡድን ፣ ኢቲቢዮስ እና የኑሮ ጥራት ግምገማ ውጤት የመመሪያ ጣቢያዎችን እና በግብርና የተጎዱ ጣቢያዎችን መለየት ችለዋል ። በግልገል አባይ የተለወጠው የወንዝ ፍሰት የውሃ ውስጥ ህይወትን ሊጎዳ ይችላል ። ጥናቶች ለሥነ-ምህዳሩ የውሃ ፍላጎቶች ፣ የባንክ መረጋጋት እና በአካባቢው ማህበረሰብ ላይ ያተኮሩ የውሃ ማጠራቀሚያ መዋቅሮች ላይ ማተኮር አለባቸው ።*

**ቁልፍ ቃላት/ሐረጎች:** በውሃ  የጀርባ አጥንት የሌላቸው ነፋሳት, ባዮኢንዳክተር, የአካባቢ ፍሰት, ተግባራዊ የመመገብ ቡድኖች, የሃይድሮሎጂ ስርዓት

# 1. INTRODUCTION

## 1.1. Background and justification

Water has a critical role in all aspects of human and ecosystem survival and health (Martin & Alister, 1988). Rivers are among the most valuable freshwater resources. They sustain the biota and ecosystems (Karr & Chu, 1999). Historically, the availability and distribution of freshwater contained in riverine systems influenced human social, economic, and political development (Hooker, 1996). Moreover they are also use for cleaning, transporting, disposing of waste, livestock watering, industrial wastes, mine drainage waters, irrigation, bathing and fishing (Meybeck and Helmer, 1996). Without sufficient freshwater supplies, both quantity and quality, global sustainable development will not be possible (Lemma Abera *et al*., 2018).

Today, irresponsible human action is radically changing this global pattern (Richter *et al*., 1996). human disturbance have drastically altered the natural flow regime of most of the world's rivers (Nilsson *et al*., 2005). The alteration of river flow regimes is claimed to be the most serious and has confounding impact on the biota structure, biodiversity and functions of aquatic ecosystems (Richter *et al*., 1996; Poff & Zimmerman, 2010) and threat to the ecological sustainability of riverine environments (Tharme, 2003; Finn *et al*., 2009). Tropical riverine ecosystems are increasingly deteriorating as a consequence of rapidly growing human populations, land use changes, intensified agriculture, increasing urbanization and industrialization, all of which tend to associate with the natural flow regimes and degradation (Ramirez *et al*., 2008). Ethiopia encounters similar challenges in relation to human induced water resources degradation (Lemma Abera *et al*., 2018). Ethiopia's still water shortage isn't a result of a lack of water; rather, it cit is due to ineffective management and administration of the water resources that are already available (Mekete Bekele, 2022).

To protect and maintain the health of aquatic ecosystems, as well as the livelihoods and well-being of people, the natural flow regime in water bodies within the environment has to be conserved and monitored. Finding out how much water is needed for such a sustainable use is the focus of environmental flow assessments, a recent and expanding field of study. Previous studies confirmed that maintaining the minimum requirement of a river's flow is a potential management strategy for a river health and sustainable use of water resource (Mcclain, 2013).

Different groups of aquatic organism, including fish, alga, benthic invertebrates, zooplankton, and macrophytes were applied for biomonitoring purposes (Barbour *et al*., 1999; Resh, 2008; Li *et al*., 2010; Gosset *et al*., 2016). The spatial distribution of zoo benthic fauna was governed by water depth, volume and flow rate; where flow alterations cause relevant habitat changes in terms of substrate heterogeneity reduction, nutrient enrichment and physico-chemical water quality variables regime alteration (Munasinghe *et al.*, 2021). Benthic macroinvertebrates are an important component of the river biota and are used as river health indicators (Alvarez *et al.*, 2010). Because they respond to different stressors for instance organic pollution (Brabec *et al*., 2004), heavy metal contamination (Bere *et al*., 2016); and stream hydraulic **(**Wills *et al*., 2006; James *et al*., 2009; Zerega *et al*., 2021). Most of them have restricted mobility, and thus they are indicators of local environmental conditions. They can be found every where and abundant even in small water bodies (Ollis *et al*., 2006), historical conditions can be indicated (Mangadze *et al*., 2019), cost effective and provide an integrative assessment of ecosystem health status (Taylor *et al*., 2009; Aschalew Lakew & Moog, 2015a). Even though this is the case, so far there is no comprehensive study that relate the assemblages of macroinvertebrates with alteration in flow regime. This is mainly due to the lack of ecoregion based standardized biomonitoring protocols for tropical countries (Mangadze *et al.*, 2018: Dallas, 2021), and scarcity of resources and equipment's (Mangadze *et al.*, 2019; Torremorell *et al*., 2021; Amare Mezgebu, 2022). Despite of these constraints, several researchers have used diversity measures, sensitivity, abundance, and feeding functional groups of macroinvertebrate communities and habits quality assessment score to evaluate the ecological health of African aquatic ecosystems (Mbaka *et al*., 2014; Musonge *et al*., 2019; Dalu & Chauke, 2020; Sitati *et al.*, 2021; Edegbene *et al.*, 2023; El Yaagoubi *et al*., 2023).

In Ethiopia, Benthic macroinvertebrate communities have been used by scholars in recent years to assess the ecological health of streams, rivers, and natural wetlands, for instance (Melaku Getachew *et al* , 2012; Argaw Ambelu *et al.*, 2013; Assefa Wosnie & Ayalew Wondie, 2014; Getachew Beneberu *et al*., 2014; Aschalew Lakew & Moog, 2015b; Sisay Derso *et al.*, 2015; Solomon Akalu, 2018; Amare Mezgebu *et al*., 2019; Geda Kebede *et al.*, 2020; Alemayehu Negassa & Misganaw Dubie, 2021; Amelework Zewudu *et al.*, 2021; Tolera Kuma *et al*., 2022; Tesfaye Muluye *et al*., 2024). Macroinvertebrates based scoring system (ETHbios) is the sole biomonitoring tool developed in the country for assessing the ecological status of inland water ( Aschalew Lakew & Moog, 2015a).

Gilgel Abay River is one of the major socio- economic benefited and upstream fish spawning migration pathways (Dagnew Mequanent *et al*., 2021). However, the river is impacted by anthropogenic activities (Arega Mulu & Dwarakish, 2016). The previous study conducted on this river (Adane Sirage, 2006; Uhlenbrook *et al*., 2010; Yihun Taddele *et al*., 2013; Anwar Assefa. *et al*., 2014; Arega Mulu & Dwarakish, 2016; Yirga Kebede *et al*., 2016; Bitew Genet *et al*., 2019; Hanibal Lemma *et al*., 2019; Habtamu Getnet *et al*., 2020; Temesgen Gashaw *et al*., 2020; Dagnew Mequanent *et al*., 2021) has not focused on the macroinvertebrate-based biomonitoring and hydrological alteration of the Gilgel Abay River. Therefore, information available with respect to the ecological condition of the Gilgel Abay River in relation to macroinvertebrate and flow alteration was limited.

## 1.2. Statement of the problem

Gilgel Abay is the largest tributary of Lake Tana that supports social and economic development of the local community as well as the country. It is used for various purposes such as agriculture, domestic uses, recreation, drinking, livestock watering, and supports a wide range of flora and fauna including fish, and birds (Yihun Taddele *et al*., 2013). Anthropogenic activities and climate change have effects in river flow leads to impacts on the lake Tana and socio-economic conditions of the region and their ecosystem services (Yihun Taddele *et al*., 2013; Dagnew Mequanent *et al*., 2021). Because of this, there is a need for continuous follow up of water quality parametres and strict water management practices (Adane Sirage, 2006). A number of recent studies have conducted in the upper watershed characteristics and the river water quality of Gilgel Abay River, such as runoff generation in the Upper Gilgel Abay watershed which is mostly influenced by quick flow (Uhlenbrook *et al*., 2010). Climate change may have effects on a reduction of mean monthly flow volume between -40% and -50% from 2010 to 2040 (Yihun Taddele *et al*., 2013). In the two years time periods (2050s and 2080s) mean annual rainfall will be expected to rise by 2.23, and 1.89 mm respectively and the average temperature will rise by 1.05, and 1.92 $^{o}{}_{C}$ in order (Anwar Assefa. *et al*., 2014). The mean annual soil erosion from the entire watershed of Gilgel Abay River was estimated at ~32.8 t $ha^{-1}yr^{-1}$ (Temessgen Gashaw *et al.*, 2020). The estimated mean suspended sediment load was ranges from 247 to 220,028 Mg $d^{-1}$ and bedload discharge ranges from 0.7 to 427.5 Mg $d^{-1}$. The area-specific bedload flux and suspended load are also 0.6 and 22 Mg $ha^{-1}yr^{-1}$ respectively (Hanibal Lemma *et al*., 2019). The Gilgel Abay river hydrological flow increase from 7.8% (128.4 $Mm^3$) to 25.3% (432

Mm$^3$) due to the effects of land use land cover change (Arega Mulu & Dwarakish, 2016). A higher peak discharge was recorded as a result of the more intense rainfall that occurred on July 20 and 21, 2008 (Bitew Genet *et al*., 2019). The existing irrigation activities on the Gilgel Abay River have impacts on fisheries by restricting upstream spawning migration pathways (Dagnew Mequanent *et al*., 2021). The various physico-chemical water quality parameters of the Gilgel Abay River were not met the guidline of World Health Organization and Ethiopian drinking water quality standards (Yirga Kebede *et al*., 2016). Anthropogenic activities like agriculture and the discharge of municipal sewage within and around the wetlands have an impact on the physicochemical parameters and habitat quality of the wetlands in the lower part of the Gilgel Abay River (Habtamu Getnet *et al*., 2020), However, the previous studies conducted in this river did not take into account benthic macroinvertebrates and river flow alteration to assess the ecological status of the Gilgel Abay River. In Lake Tana sub-basin, on Gilgel Abay River the government has planned for construction of large dam for irrigation, so it needs establish the minimum environmental flow requirement for the preservation, restoration, and proper use of water resources for sustainable aquatic ecosystems maintenance.

This study tried to maximize the knowledge regarding the ecological status of the river, and provide scientific recommendation for stakeholders for the sustainable management of river ecosystems. In addition, since the river feeds Lake Tana which is the main source of the transboundary Nile River, it is also serve as the political corner stone for the riparian countries for implementing and coordinating management practice in this river that would help to maintain a continuous Nile River flow within and beyond the national boundary.

## 1.3. Research questions

- The current study intended to answer the following research questions.
  - How are benthic macroinvertebrate communities structured in relation to environmental variables?
  - What is the overall health of the river based on macroinvertebrate metrics and stream flow alteration?

## 1.4. Hypothesis

HO: The distribution of benthic macroinvertebrates is not affected by physico-chemical variables and within 42 years there is no significant difference in water flow changes in Gilgel Abay.

HA: The distribution of benthic macroinvertebrates is affected by physico-chemical variables and within 42 years there is a significant difference in water flow changes in Gilgel Abay.

## 1.5. Objective of the study

### 1.5.1. General objective

To investigate the ecological status of Gilgel Abay River based on macroinvertebrate metrics and hydrological alterations.

### 1.5.2. Specific objectives

- To assess the flow alterations since 1979 up to 2020 based on ecologically relevant environmental flow components and flow indices.
- To determine the relationship between benthic macroinvertebrate distribution with some selected measured environmental variabiles across sampling sites in association with stressors in the Gilgel Abay River
- To determine the diversity, abundance and distribution of the benthic macro-invertebrates in relation to stressors in the Gilgel Abay river.

## 1.6. Significance of the study

The Gilgel Abay River has numerous uses and functions, including irrigation, cattle watering, swimming, household supply, and fish production for local communities and commercial purposes. Based on the response of benthic macroinvertebrates metrics and flow alteration, this study was generate baseline scientific information on the condition of the Gilgel Abay River and update physicochemical water quality variables. The information generated from the study will assist government and non-government organizations in developing strategies for the sustainable management and utilization of the Gilgel Abay River and other similar water bodies elsewhere in the country. It also encourages researchers and academic institutions to conduct additional studies on the use of macroinvertebrates and hydrological flow alteration in Ethiopian highlands conditions.

## 1.7. Limitation of the study

The present study mainly focused on assessing the status of riverine based macroinvertebrates communities along the Gilgel Abay River. Because of the regional insecurity, it was difficult to do frequent sampling as per the schedule especially for water quality parameters and benthic macroinvertebrates. However it was managed to collect meteorological data for the last 42 years for flow alteration analysis.

# 2. LITERATURE REVIEW

## 2.1. Biological indicators to assess the ecological status of aquatic ecosystems

Bioindicators determine the presence, condition, and numbers of the types of organisms that can provide precise information about the health of specific rivers, streams, lakes, and wetlands. Most water bodies in the country are severely overexploited and negatively impacted in a variety of ways. Biomonitoring is important to assess the status of freshwater and identify threats to develop adequate restoration and protection strategies through biomonitoring.

Benthic macroinvertebrate is the widely utilized bioindicator to assess the aquatic ecosystem condition and serve as the foundation for several biotic indices; for instance, those used in the United Kingdom (Armitage *et al*, 1983), Europe (Bonada *et al*., 2006), North America (Rosenberg and Resh, 1993; Barbour *et al*., 1999), South America (Buss & Vitorino, 2010), Asia (Morse *et al*., 2007), Australia (Bruce, 2003), New Zealand (Stark & Maxted, 2007), South African (Dickens & Graham, 2002), Zambia (Dallas, 2009) and Ethiopia (Aschalew Lakew & Moog, 2015a). Macroinvertebrates have their own merits and weakness to assess the health of aquatic ecosystems.

### 2.1.1. Advantages of benthic macroinvertebrate for biomonitoring

Aquatic macroinvertebrates are tiny but numerous, have an important role in ecosystem function, and make up a significant component of freshwater biodiversity. Therefore, utilizing them in bio evaluation has various benefits; they can be found every where and abundant even in small water bodies, they make it possible to do chemical analysis when quantities are undetectable in the water resource because they accumulate (retain) dangerous chemicals (Ollis *et al*., 2006), making it possible to conduct biomonitoring in practically any type of rivers and streams (Armitage *et al*., 1983). They are a highly diverse group with different environmental needs, which classifies them as top candidates for research on biodiversity changes.

The sampling and processing methods are well-developed and taxonomic keys are available to identify major benthic macroinvertebrates, and a certain degree of mobility and the ability to actively choose environments that met their environmental needs; On the other hand, they have restricted mobility, and thus they are indicators of local environmental conditions, they have the ideal size to be easily collected, requires few people and minimal equipment, does not adversely affect other organisms, they response to increased nutrient levels in streams is have altered biological structures and functions such as species richness, composition (Hart and Robinson, 1990; Stevenson, 1997), long generation time; the group is useful to assess changes over time, short generation time; the group has rapid responses to change and quick recovery from disturbance (Mangadze *et al*., 2018). Historical conditions can be indicated, the group has fossils that can be used to reflect past conditions (Mangadze *et al*., 2018). There is a need for a shift to the use of biological indicators which are cost-effective and provide an integrative assessment of ecosystem health status (Taylor *et al*., 2009; Aschalew Lakew & Moog, 2015a).

### 2.1.2. Weakness of benthic macroinvertebrate for biomonitoring

Contrary to its merits, benthic macroinvertebrate has drawbacks, such as the fact that the scores used in certain areas need to be modified for application to other regions, not only because some macroinvertebrates may be absent from the respective area and replaced by other taxa, but also some taxa may exhibit different pollution tolerances from region to region (Buss & Salles, 2007), they were not able to discriminate between natural and human disturbance (Moya *et al.,* 2007; Tomanova *et al*., 2008). Some benthic invertebrates like Chironomidae have lack of a consistent set of metrics response. For instance, chironomid individuals performed better in reference sites than in disturbed sites in the sand-bottomed streams of the Northern Plains ecoregion of the USA; in contrast, they are ineffective in the mountain ecoregion of the same country (Whittier *et al*., 2006).

There is no common conscience at the tax identification level. For instance, species or genus is more capable of detecting impact or enabling greater resolution of different degrees of impact, than identification only to the family level (Hilsenhoff, 1988; Lenat & Resh, 2001; Schmidt-kloiber & Nijboer, 2004), However, others have claimed that family-level assessments differed from species-level or genus-level assessments in their ability to distinguish different levels of human impact (Feio *et al*., 2007). To differentiate different levels of pollution the family level identification of chironomid larvae is insufficient (Getachew Beneberu *et al.*, 2014). Identification of a family is much less expensive and faster than that of a species or genus (Thorne & Williams, 1997; Marshall *et al*., 2006; Aschalew Lakew & Moog, 2015a). Some taxa were difficult to sample, required expertise and effort for identification, few taxonomic keys were available, and short generation time did not show bioaccumulation (Bonada *et al*., 2006; Ziglio *et al*., 2006), and effects of multiple stressors are not easily detected or distinguished.

## 2.2. Overview of macroinvertebrates based biomonitoring in Ethiopia

In Ethiopia, physicochemical measurements are largely utilized and have a long history of assessing the quality of water (Temesgen Alemneh *et al*., 2019). Several bioassessment approaches to assessing surface water quality use accessible biological organisms such as fish, algae, aquatic plants, and invertebrates as bioindicators of ecosystem integrity (Seyoum Mengistou, 2006). In Ethiopia, there is currently a study of river benthic macroinvertebrates for the biological assessment of pollution and degradation of inland water bodies some of them were (Getachew Beneberu *et al*., 2014; Aschalew Lakew & Moog, 2015a; Solomon Akalu, 2018; Amare Mezgebu *et al*., 2019; Amelework Zewudu, *et al*., 2021; Misgana Dabessa *et al*., 2021).

Some of the many benthic macroinvertebrate-based assessments of water quality conducted in Ethiopian freshwater were; headwater streams and values of bio-assessment for sustainable river management in the highlands of Ethiopia (Aschalew Lakew, 2020). Adaptation of field-based ecological health assessment protocols for central and south-eastern highland rivers in Ethiopia (Aschalew Lakew, 2022). Assessment of the water quality of the upper Awash River, along the river course about 500 km (Fasil Degefu *et al.*, 2013). The ecological status of streams and rivers in Modjo, Kebena, Akaki, Chacha, Megecha, Wabe, Ghibe, Dabena, and the macroinvertebrate community index of Sebeta River (MCISR) was developed using selected metrics. Assess the water quality status of Omo-gibe, Abay, and Awash River Basins Rivers using Chironomidae (Getachew Beneberu *et al*., 2014). Assess the level of degradation of rivers and streams ecosystems around Sebeta using benthic macroinvertebrate as biomonitoring (Amare Mezgebu *et al*., 2019). The ecological health of highland rivers in Ethiopia was assessed using benthic macroinvertebrates (Aschalew Lakew & Moog, 2015b) and other similar studies in various rivers, streams of Ethiopia using benthic macroinvertebrate metrics were conducted.

Benthic macroinvertebrate based biomonitoring studies conducted in many streams and rivers of Ethiopia shows that it is a robust and sensitive organism that can be applied to evaluate the ecological condition of natural rivers, streams, lakes, reservoirs and wetlands of our nation (Seid Tiku *et al*., 2013; Aschalew Lakew, 2014; Getachew Beneberu *et al*., 2014; Tolera Kuma *et al.*, 2022). While biomonitoring is still in its toddlerhood in Ethiopia (Seyoum

Mengistou, 2006; Getachew Beneberu *et al*., 2014; Aschalew Lakew & Moog, 2015b; Amare Mezgebu *et al*., 2019). Therefore, research on river invertebrates ought to be prioritized while evaluating and monitoring pollution and the deterioration of Ethiopia's wetlands, lakes, streams, and rivers.

## 2.3. Hydrology of environmental flow

The hydrology of a river includes the system's hydrodynamics, such as stream flow, flow velocity, and surface runoff. Discharge is a hydrological parameter affected by the basin's size, shape, and geological features, as well as climatic circumstances. It is the product of velocity (flow) and channel cross-sectional area (UNEP, 1996).

Rivers are exposed to various effects, such as flood dams, and irrigation canals. Water is the most important component that must be protected along the river bed. Environmental flow is defined as the quantity, time, and quality of water flows required to sustain freshwater and estuarine ecosystems, as well as the human livelihood and well-being that rely on these ecosystems (Fang *et al*., 2010). Around 207 environmental flow assessment methods have been discovered around the world, and they have been divided into four groups: hydrologic index, hydraulic rating, habitat simulation, and holistic methods (Tharme, 2003; Acreman *et al*., 2004). Some approaches are widely used, while others are more regional. The river's environmental flow is important to safeguard ecological sustainability, the river's structure, the water cycle, and all other factors associated with rivers, such as energy generation (Tharme, 2003). Many studies have been undertaken to measure river environmental flow based on these essential advantages. Some of them were the East Rapti River (Smakhtin & Eriyagama, 2008), the La Nga` River in Vietnam (Babel *et al*., 2012), the influence of dams on the Huaihe River in China (Hu *et al*., 2008) and Gumara River (Wubneh Belete *et al*., 2020). Human activities and natural phenomena have an impact on this catchment's hydrological water balance (Arega Mulu & Dwarakish, 2016). The deterioration of river water quality and macroinvertebrate communities is accelerated by altered hydrology due to runoff from impermeable structures and a poor drainage system (Solomon Akalu, 2018). Water abstractions are planned which impacts the environment (Tadesse Alemayehu *et al*., 2010). Unmanaged pump irrigation practices, the expansion of eucalyptus trees, and sand mining on upstream of the Gumara River result in the river's flow stopping in the dry season (Wubneh Belete *et al*., 2020). The composition and diversity of vegetation, and macroinvertebrate species in the Gumara River were significantly influenced by flow conditions, water quality, and land-use change (Wubneh Belete *et al*., 2021). Machroinvertebrate assemblages was influenced by hydrological flow fluctuations (Chessman et al., 2010). Flow duration curve plays a critical role in assessing the river's health and its ecosystem, since it provides information about the variability of flow rate (Booker & Snelder, 2012) and its ecological responses (Poff & Zimmerman, 2010). Flow duration curve statistics are frequently utilized to quantify hydrologic regimes (Booker & Snelder, 2012). Information obtained in the flow duration curve can be used for various purposes; including assessing the physical habitat quality in the river ecosystem (Booker *et al*., 2004), reliability of water supply and ecosystem protection (Snelder *et al*., 2011), the river health and its ecosystem (Booker & Snelder, 2012). In cases where significant alterations like intensive irrigation projects are contemplated within a river system, indicators of hydrological alteration (IHA) advised setting the minimum environmental flow reqiurments (EFRs) at 90% during low flow and 50% during high flow periods to maintain the river eco-environmental health and the services provided by the river ecosystem (Pastor *et al*., 2014).

## 2.4. Main anthropogenic stressors of streams and rivers

Anthropogenic pollutants are substances caused by human actions, mostly resulting from land-use practices. Surface water differs from groundwater because it may contain many hazardous chemicals from human practices; as a result, it is highly contaminated (Tyagi, 2014).

### 2.4.1. Agriculture practices

Agriculture is one of the most common activities of human beings that can affect both surface and ground water. The activities of this category include fish farming, crop cultivation, pesticides, fertilizers, and livestock farming. These disturbances were have impactes soil, vegetation, rainfall, surface, and irrigation water penetration can lead to pollutants entering the ground water (Akhtar *et al*., 2021). Alteration of river flows has intensified through the expansion of irrigated agriculture. Agriculture has a significant impact on macroinvertebrate communities (Getachew Beneberu *et al.*, 2014; Aschalew Lakew & Moog, 2015a). Agricultural land use intensification was impacted the community structure and taxon richness of macroinvertebrates. Reduced agricultural land usage, on the other hand, provides for the establishment of more diversified benthic invertebrate communities (Gerth *et al*., 2017). The stressor-sensitive taxa, including the orders Ephemeroptera, Plecoptera, and

Trichoptera (EPT), as well as flies from the family Simuliidae, were found in minimal agricultural usage.

### 2.4.2. Urbanization

Urbanization is a broad type of land-use and land-cover change that is expanding quickly world wide. It involves the converting of wetlands, forest, pastures and other land cover types for cropland, commercial, industrial, residential, and transportation purposes, which results increasing of impermeable surfaces (Mueller *et al*., 2011). Therefore, impervious surfaces are measurable factors closely correlating with rising polluted runoff that decreases water resource quality (Mcgrane, 2016). The waste waters come from our every day lives and includes preparing food, washing, bathing, and toileting; moreover, grey water and black water are released from domestic dwellings, schools and health centers; all these have a significant impact on aquatic biological communities. Likewise urbanization and impervious area in watersheds have negative impact on benthic macroinvertebrates biomass and community structure such as taxon richness which are likely to have a significant impact on river ecosystem function (Allan, 2004; Wang *et al*., 2007).

# 3. Materials and methods

## 3.1. Description of the study area

Gilgel Abay is a river located in central Ethiopia found within 11°48′N and 37°7′E directions. It rises in the Gojjam mountains and flows from north to south-western Lake Tana. Tributaries of the Gilgel Abay River include the Ashar, Jamma, Kelti, and Koger. The Gilgel Abay River, with a catchment area of 5,004 $km^2$, is the largest river discharging into Lake Tana (Yihun Taddele *et al*., 2013). Moisture comes from the Atlantic and Indian oceans and moves north-south in the intertropical convergence zone, generating rainfall in the Gilgel Abay River area and the Upper Blue Nile region (Mohamed *et al*., 2005). According to Ethiopian climatic classification, the Upper Gilgel Abay River watershed has a monomodal climate concerning rainfall regimes. The study area's main rainfall season lasts from June to September and contributes for 70-90% of the annual rainfall (Tarekegn Deksyos & Tadege Abebe, 2006). The study area's watershed the average annual rainfall (1988–2018) ranged from 1230 to 2350 mm (Yeshiwas Yeneneh, 2023) and the average monthly maximum temperature in Wetet Abay (1994–2008) ranges from 24.3 to 31.3 $^{o}C$ (Anwar Assefa *et al*., 2014). The geographical location of the study area along sampling sites is depicted in Figure 1.

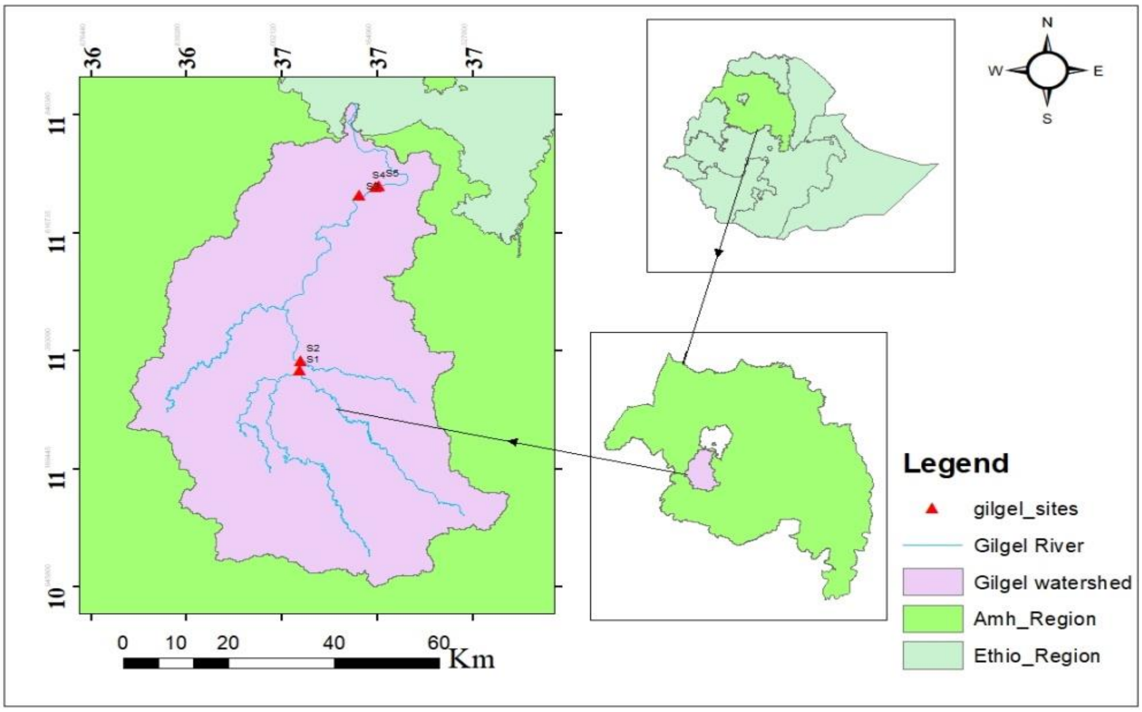



Figure 1. Map of the Gilgel Abay River showing the sampling sites

## 3.2. Sampling site selection

### 3.2.1. Site Selection and description

To explore the ecological status of the river the sample sites were selected purposively considering protected forest areas, grazing areas, hydro-morphology, and rural-agricultural areas. A relative reference site was selected during preliminary reconnaissance field sampling based on twenty prior criteria established for reference site selection as described in Appendix 1.

Totally five sampling locations reflecting various anthropogenic activities were chosen from the study river (plate1). Site 1 (S1) is located in rural areas, which is the most upper stream sites before the Jemma River joins with Gilgel Abay River. It is relatively less disturbed, characterized by better riparian vegetation cover, and utilized for drinking. Site 2 (S2) is located above the confluence of the Koga River. Site 3 (S3) was located before the main Gilgel Abay River separated into two small rivers, which are highly affected by water abstraction for irrigation, livestock watering, and deforestation. Site 4 (S4) lies above the bridge, which has high river bank instability and gully formation due to anthropogenic activities and is utilized for cattle watering, washing, and bathing. Site 5 (S5) is located below the bridge, which is highly affected by water abstraction through pumping for irrigation, livestock watering, and deforestation. The latitude, longitude, and altitude of the sampling sites were determined using a global positioning system (GPS) sensor (Table1).

Table1. Name of study sites, site code, and their respective geographic coordinates

| Local Name | Site Code | Longitude | Latitude | Altitude |
|---|---|---|---|---|
| Fendika | S1 | 37.03437 | 11.35763 | 1901.4 |
| Dibkan | S2 | 37.03535 | 11.37525 | 1879.41 |
| Telifa | S3 | 37.13622 | 11.68821 | 1801.4 |
| Chimba | S4 | 37.16348 | 11.70419 | 1798.46 |
| Tankuaber | S5 | 37.16926 | 11.70668 | 1801.47 |


***Plate 1*:** **Some of the activities carried out in the studied river; charcoal production (a), Pump for irrigation (b), Livestock watering (c &d), Cloth washing and bathing (e &f) (Source: Endalh mekonnen, 2023)**

## 3.3. Data collection

### 3.3.1. Sampling period and frequency

To evaluate the influence of anthropogenic activities, sampling should be avoided during high water levels and immediately after flooding (Barbour *et al*., 1999; Aschalew Lakew, 2014); in this regard, sampling was conducted during the pre-wet seasons in May 2023.

### 3.3.2. Stream flow

The hydrology of the river was characterized for the entire Gilgel Abay River catchment using the gauged station data. There was a lack of consistent daily recorded observable flow data because of this we used the period of recorded data available since 1979. Forty-two (42) years of daily observed flow data from 1979 to 2020 were used for analysis; from these twenty-two years of daily flow data were obtained from ministry of water, irrigation and electricity (MoWIE), Ethiopia and the rest 20 years of data were downloaded from climate explorer website.

### 3.3.3. Environmental variables

Environmental variables that were measured during the sampling period included physical, chemical, and hydro-morphological characteristics. These factors are related to the diversity and abundance of indicator organisms (BMI). To avoid disruption physicochemical parameters of the water were measured from each sampling site before macroinvertebrate samples were taken. Using a multiline-probe system (HQ40d), *in-situ* measurements of electrical conductivity (EC), dissolved oxygen (DO), pH, and temperature ($T^o$) were conducted (plate 2).


**Plate 2: Onsite measurement of physicochemical parameters**

Major nutrients; including nitrate nitrogen ($NO_3$-N), nitrite nitrogen ($No_2$-N), ammonium nitrogen ($NH_4$-N), and soluble reactive phosphorus (SRP) ($PO_4$) were analyzed by collecting two liters of water from each site, stored in the ice box, and transported to the Bahir Dar Fishery and Aquatic Life Research center (BFALRC, Bahir Dar) and analyzed using standard procedures as outlined in APHA (1995). For all nutrient variables, 150ml water samples were first filtered through glass fiber filters (GF/F) and then the filtrate water was used for the final analysis. The water samples were taken from the sub-surface of the water.

### 3.3.4. Habitat quality assessment score (HQA)

The habitat quality per site was assessed using protocol (Barbour *et al.*,1999) (Appendix 2), which were used for the evaluation of river and riparian modifications, and also all reaches were characterized according to this protocol. The criteria include substrate quality, embeddedness, velocity/depth regime, sediment deposition, channel flow status, channel alterations, frequency of riffles, bank stability, vegetative protection, and extent of riparian vegetation. Site evaluation was based on a scale of 0 to 20. The HQA was obtained by summing individual scores of the ten parameters (the maximum score for each parameter was 20 and the maximum total score at each site was 200). In addition the human activities observed at each site were recorded and every disturbance was given a score of 0.01 and then the number of disturbances was multiplied by 0.01and substract it from the sum of habitate quality score to get the actual HQA for each sampling site (Raburu *et al.*, 2009). The reason for including human disturbances in the final value of HQA was that they lower the natural habitat quality of a site and the river habitat quality classes were categorized (Table 2).

Table 2. Classes for assessment of river habitat (King *et al*., 2006)

| Class | Description | Score (%) |
|---|---|---|
| A | Unmodified, natural | 100 |
| B | Largely natural with few modifications. A small change from natural in habitats and biotas may have taken place, but the ecosystem functions are essentially unchanged | 80-99 |
| C | Moderately modified. A loss of and change from natural habitats and biotas has occurred, but the basic ecosystem functions are still predominantly unchanged | 60-79 |
| D | Largely modified. A large loss of natural habitats, biotas and basic ecosystem functions has occurred | 40-59 |
| E | The losses of natural habitats, biotas and basic ecosystem functions are extensive | 20-39 |
| F | Modifications have reached a critical level and the lotic system has been completely modified, with an almost complete loss of natural habitats and biotas. In the worst instances, basic ecosystem functions have been destroyed and changes are irreversible | 0-19 |

### 3.3.4. Benthic macroinvertebrates

#### 3.3.4.1. Field sampling

Benthic macroinvertebrates samples collection was performed using a 500 µm mesh D-frame net for the pool as well Surber sampler for riffle habitat (Elias, 2021). Multi-habitat sampling scheme (MHS) was implemented to include major habitat types in proportional representation of sampling reach (Hartmann *et al*., 2005; Moog, 2007). Sampling of benthic macro invertebrates started at the downstream of the reach against water current to avoid disturbance of the upstream sampling units. To remove bias due to spatial and heterogeneity impact, samples were collected separately from different biotopes available at each site. The biotopes include; stone, vegetation (marginal and in-water vegetation), GSM (gravel-sand-mud/silt), and bolder. Each biotope was disturbed while moving the Surber sampler at riffle habitats that were placed closely downstream towards upstream for one minute to trap the detached macroinvertebrates (Elias, 2021). Plate 3 shows how samples were taken from the riffle habitat. For the sake of inventory, rapid on-site taxa identification was performed and subsequently, the composite samples were preserved in 70% ethanol for laboratory processing, identification, and recording (Getachew Beneberu *et al.*, 2014) and all necessary information (stream name, site code, collector's name and date of collection) were labeled using water proof marker.

**Plate 3: Field sampling of benthic macroinvertebrates**

#### 3.3.4.2. Laboratory analysis

The macroinvertebrate samples collected and preserved in the field were transported to the laboratory for further analysis. The information written in the sample container was copied to a data sheet before identification. Sorting was done through the naked eye by placing the sample in a white tray. Macroinvertebrates' specimens were identified to the family level or the possible taxa level with the help of a dissecting microscope for detailed observation (plate 4). Identification was performed using these keys (Gerber & Gabriel, 2002; Bouchard, 2004&2010; Beauchene, 2021).


**Plate 4: Benthic macroinvertebrate identification (a), Chironomids in 10% KOH solution on a hot plate to make their soft tissue transparent (b) (Source: Endalh mekonnen, 2023)**

For further investigation chironomids were removed from the preservative and placed in 10% KOH solution on a hot plate for 5 minutes to make their soft tissue transparent (plate 4b). The head capsules were detached from the body, cleaned with distilled water, dehydrated, and mounted using glycerin; cover slips  by inverting upside down (Oliver & Roussel, 1983). The coverslips were sealed with nail polish. This is to fix the cover slip to make it permanent for future validation, education and research. The Chironomidae were sorting and mounting them on the slide using discting microscope and identification was made under compound microscope with the help of various identification keys (Eggermont & Verschuren, 2003a; Eggermont & Verschuren, 2003b; Eggermont & Verschuren, 2004a; Eggermont & Verschuren, 2004b; Getachew Beneberu *et al*., 2014; Cuppen & Tempelman, 2018; Cranston, 2019) and images of representative specimens were taken during identification with android phone.

## 3.5. Benthic macroinvertebrates' metrics

### 3.5.1. Abundance and diversity of macroinvertebrates

The diversity and abundance of macroinvertebrate taxa at each sampling site was determined using the Shannon diversity index (H') (Ejtehadi *et al*., 2009).

$$H' = -\Sigma\ Pi*Ln\ (Pi)$$

Where H' = the Shannon diversity index, Pi = fraction of the entire population made up of the species i.

### 3.5.2. Evenness

Evenness (E) is a measure the relative abundance of different species in a particular area. It uses species richness (S). When there are similar proportions of all species, then the evenness value is very close to one, but when the abundance is very dissimilar (some rare and some dominant species) then the value decreases. Using the same log base as with H, evenness was calculated according to Ejtehadi *et al*. (2009) as follow:

$$E = H/ \ln (S)$$

Where "S" is the total number of species, and H is the Shannon-Wiener index

### 3.5.3. Functional feeding groups

The functional feeding groups provide information on the balance of feeding strategies in the benthic assemblage. According to Barbour *et al.*(1999) and Ramirez & Gutierrez-fonseca (2014) functional feeding groups are categorized into scrapers, shredders, gatherers, filterers, and predators. The functional feeding groups of each taxon was categorized based on the information reviewed from different literature (Nessimian & Luiz, 2010; Gholizadeh & Heydarzadeh, 2020; Li et al., 2021; Ilmi *et al.,* 2023).

### 3.5.4. Calculation of biotic score (ETHbios)

ETHbios was calculated as the sum of the sensitivity score of each taxon present in a sample (Aschalew Lakew & Moog, 2015a). However, the sensitivity score values for the taxa Simuliidae, Pyralidae, Planorbidae, and Lymnaeidae were not provided by ETHbios. For this reason, we used the sensitivity-weighted values for these taxa from SASS (Dickens & Graham, 2002) and BMWP (Armitage *et al*., 1983). The river quality class was classified based on the developed ETHbios threshold values shown in Appendix III:

$$\text{ETHbios} = \sum_{i=1}^{n} \text{Scorei}$$

The Average score per taxon (ASPT) was determined as ETHbios divided by all number of taxa considered in the calculation.

$$\text{ASPT} = \sum_{i=1}^{n} \text{Scorei}/n$$

Where, the score of taxa i (Score i) and the number of taxa (n) considered in the calculation.

## 3.6. Environmental flow components and flow duration curve

The five ecologically relevant environmental flow component parameters, i.e., extreme low flows, low flows, high flow pulses, small floods, and large floods were determined using IHA software (The Conservancy, 2009). The 42 years (1979-2020) period of recorded was used to estimate the FDC of th Gilgel Abay River. The flow duration curve (FDC) was formulated as $F = (R/(n + 1)) (100)$ (Langat *et al.*, 2019), where F is the frequency of occurrence expressed as % of the time a particular stream flow value is equaled or exceeded (exceedance probability), R is the rank corresponding to stream flow and n is equal to a total number of days.

## 3.7. Data analysis

The river flow statistics (five environmental flow components, monthly flow, high and low flow and flow duration indices) that were analyzed using IHA software version 7.1 (The Conservancy, 2009). One-way ANOVA through IBM SPSS version, 23 was performed to see if there was a significant difference of physicochemical variables and macroinvertebrate community metrics among sampling sites.

To examine the relationships between metrics and measured environmental variables, person correlation was employed. To do this, the metrics and physicochemical parameters that have a significant difference in mean values across sampling sites ($P<0.05$) using SPSS comparison of mean through one sample t-test were used for correlation analysis. The metrics that differentiate sites of different water quality classes were selected based on the metric redundancy tested by Spearman correlation between all combinations of candidate metrics. Metrics with Spearman correlation values of ($r \geq 0.70$; $p<0.01$) were considered as redundant and at least one of the redundant variables was eliminated. We chose one based on a combination of the lowest P-value from the t-test and the lowest CV (Angermeier *et al*., 2000).

Hierarchical cluster analysis (HCA) using the unweighted pair group method with arithmetic mean (UPGMA) based on the Bray-Curtis similarity index was performed on macroinvertebrate species and FFG compositions to show major differences and similarities among the study sites through stastical software of  PAST version 4.04.

The multivariate analysis was applied using the CANOCO 5 program to see the relationship between the structure of macroinvertebrate community with environmental variables among sampling sites (Šmilauer & Lepš, 2015). The square root of the proportion of macroinvertebrate data and log10 transformation of environmental variables were performed prior to conducting analysis (Legendre & Gallagher, 2001). We conducted a test of unimodality and linearity. The CANOCO ordination method used to select the linear (RDA) based on data set has a gradient length of 1.95 turnover (SDs) units, indicating the linear response model is a preferable choice (Lepˇs & ˇSmilauer, 2003). Consequently, redundancy analysis (RDA) with all constrained axes test procedures following Monte Carlo 999 permutations was used to explore the predictor variables that mostly contribute to the distribution of macroinvertebrates.

# 4. RESULTS

## 4.1. Flow indices

### 4.1.1. Monthly flows

The monthly flow analysis indicated that low flow occurs from February to April, with the lowest being observed in April, with a median flow of 17.43 $m^3s^{-1}$ with a standard deviation of 4.95 $m^3s^{-1}$. Monthly high flow occurred from June to November, with the highest recorded in September, with a median flow of 268.6 $m^3s^{-1}$ and a standard deviation of 52.7 $m^3s^{-1}$ (Table 3).

Table 3. Monthly median flows ($m^3s^{-1}$) of Gilgel Abay River during the period of recorded (1979-2020)

| Months | Medians flows(Q50), $m^3s^{-1}$ | Coefficient of Disp.(Q75 Q25)/Q50 | Months | Medians flows (Q50), $m^3s^{-1}$ | Coefficient of Disp. (Q75–Q25)/Q50 |
|---|---|---|---|---|---|
| October | 196.3 | 0.5347 | April | 17.43 | 0.3176 |
| November | 79.04 | 0.576 | May | 30.42 | 1.349 |
| December | 36.69 | 0.4107 | June | 90.36 | 0.3941 |
| January | 26.20 | 0.23 | July | 126.3 | 0.2526 |
| February | 22.89 | 0.22 | August | 185.2 | 0.3413 |
| March | 19.98 | 0.25 | September | 268.6 | 0.1728 |

NB; Q25 is flow that is exceeded 25 % of the time, Q50 is the median and Q75 is flow exceeded 75% of the time.

A high coefficient of dispersion (COD) was found in November and May during the dry season a low COD was observed in September during the wet season.The table revealed the median flow (Q50) for each month, reflecting the average amount of water flowing through the river over a given period of time.

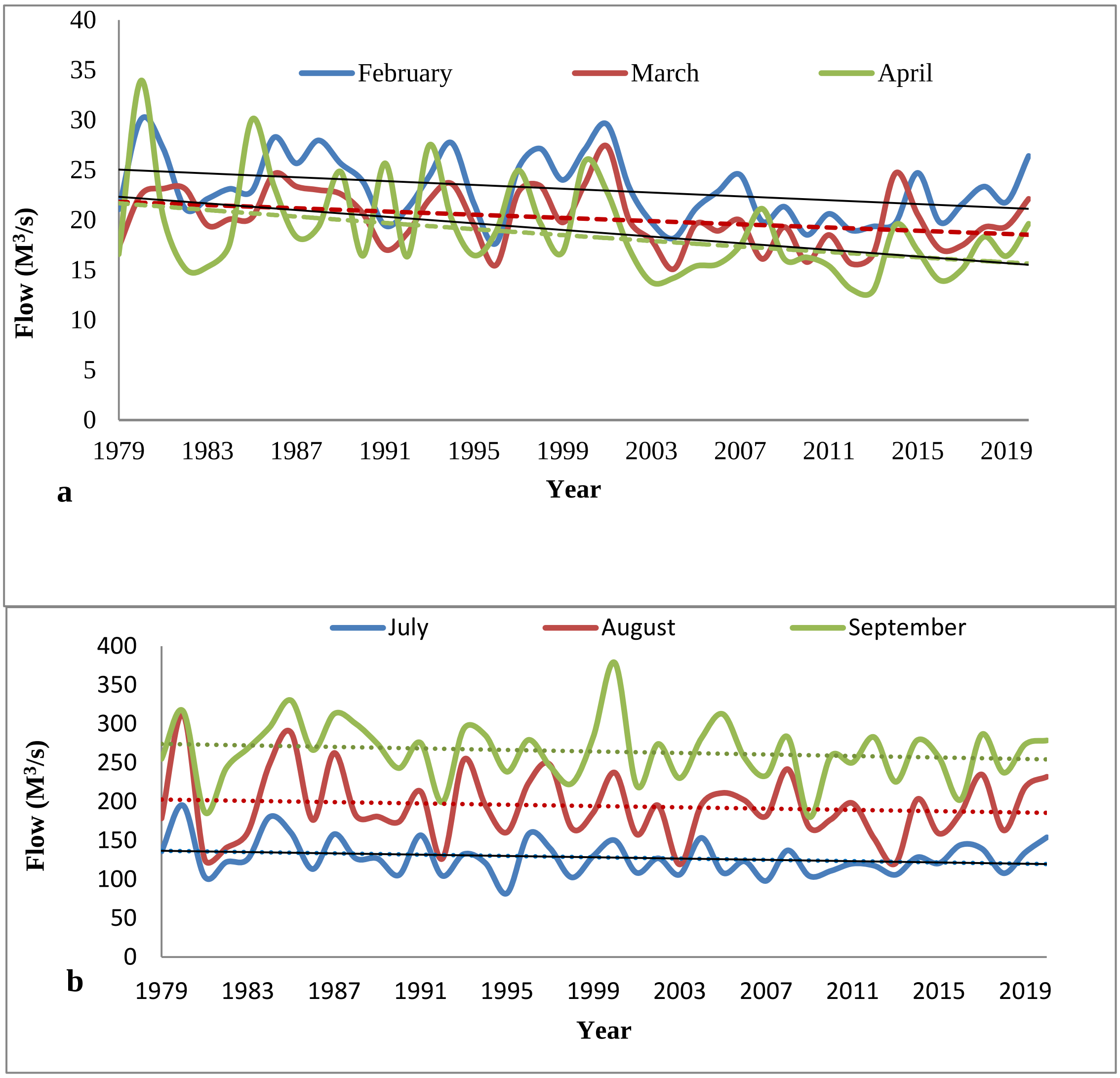



**Figure 2. Monthly flow of Gilgel Abay River 1979–2020 in (a) dry season (February, March and April) and (b) wet season (July, August and September).**

There is a downward trend in the river's flow during that particular time period (Figures 2a, b). Unnatural fluctuations in flow can upset trophic systems, aquatic habitats, and spawning cycles. They can also have an effect on water quality by affecting elements like oxygen levels, sediment transport, and the dilution of contaminants. River ecology may suffer from variations in this natural seasonal timing brought on by anthropogenic disturbances and climate change.

### 4.1.2. Low and high flow

The 1-Day, 30-Day and 90-Day low flow, there was a statistically significant decreasing trend over the study period; Mann- Kendall trend test significant at $p < 0.05$. Quantitatively, 1-Day low flow decreased from 19.41 $m^3s^{-1}$ in 1980 to 13.98 $m^3s^{-1}$ in 2020, 30- Day (monthly) from 16 $m^3s^{-1}$ 1979 to 15 $m^3s^{-1}$ 2019 and 90-Day (seasonal) low flow decreased from 18.5 $m^3s^{-1}$ in 1979 to 18 $m^3$ $s^{-1}$ in 2020. The minimum and maximum flows were indicated in Table 4.

**Table 4. The minimum and maximum flows ($m^3s^{-1}$) of Gilgel Abay River at different periods (1979-2020)**

| Duration | Medians flows (Q50), $m^3s^{-1}$ | Coefficient of Disp.(Q75–Q25)/Q50 | Duration | Medians flows (Q50), $m^3s^{-1}$ | Coefficient of Disp.(Q75–Q25)/Q50 |
|---|---|---|---|---|---|
| 1-day minimum | 13.91 | 0.3005 | 1-day maximum | 548.1 | 0.5961 |
| 3-day minimum | 14.07 | 0.28 | 3-day maximum | 472.6 | 0.5863 |
| 7-day minimum | 14.43 | 0.30 | 7-day maximum | 389 | 0.3674 |
| 30-day minimum | 15.97 | 0.27 | 30-day maximum | 289.3 | 0.2666 |
| 90-day minimum | 19.72 | 0.24 | 90-day maximum | 208.3 | 0.2367 |

The high flows, similarly to the low flows, decreased over the study periods. Quantitatively, 1-Day high flow decreased from 579.1 $m^3s^{-1}$ in 1979 to 378.4 $m^3s^{-1}$ in 2020, and 90-Day (seasonal) high flow decreased from 214.6 $m^3$ $s^{-1}$ in 1979 to 211.3 $m^3$ $s^{-1}$ in 2019, but the 90- day high flow decreasing trends were very small (Figure 3).

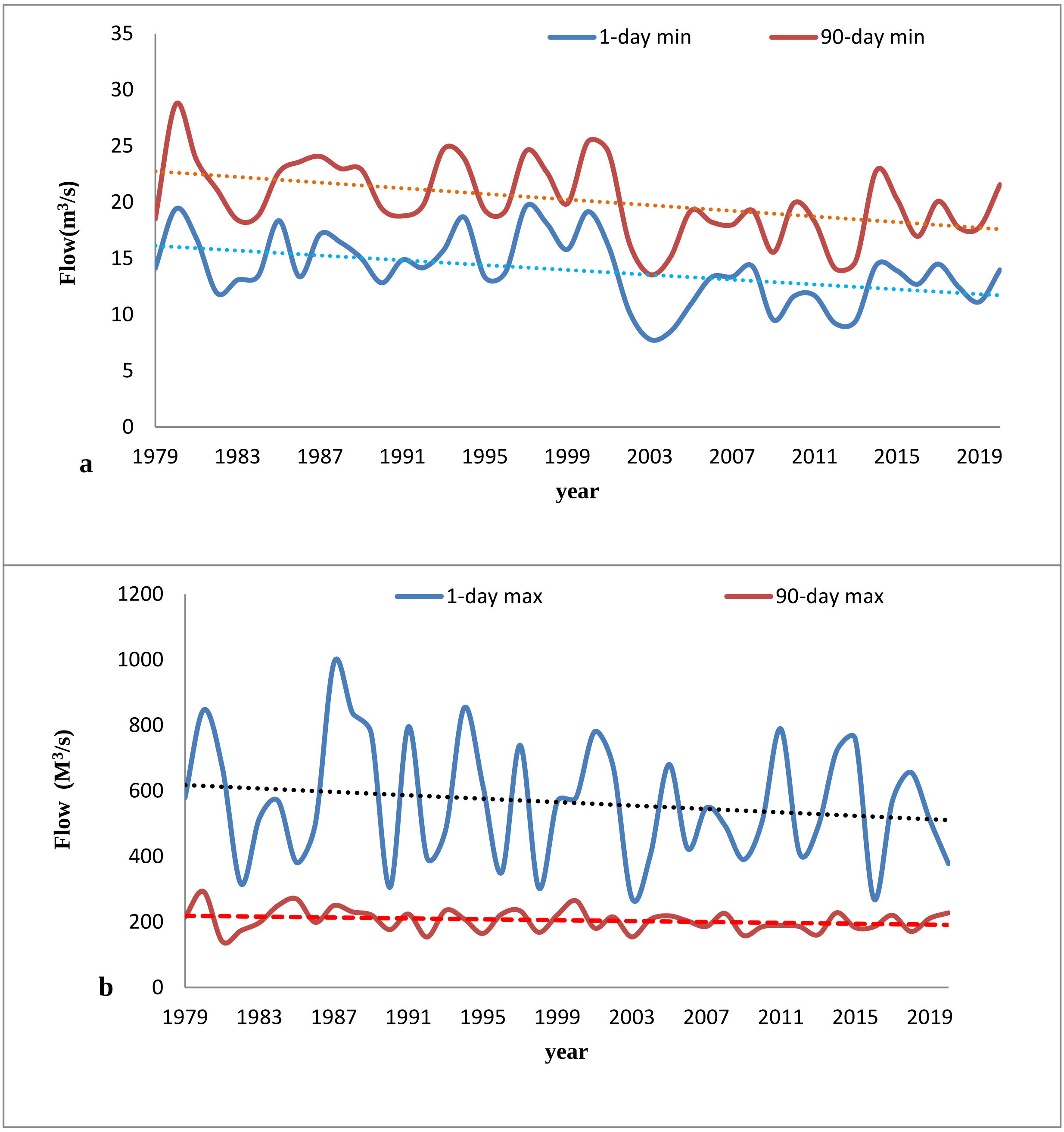



**Figure3. One-day and 90-day low (a) and high flow (b) of Gilgel Abay River (1979–2020).**

The figure depicts (a) the lowest flow from each day of the year as a 1-day duration low flow and the lowest flow of average 90 day (seasonal) duration low flow and (b) the highest flow from each day of the year as 1-day duration high flow; and highest flow of average 90 days duration high flow.

### 4.1.3. Environmental flow components (EFC)

The EFC analysis indicated four large floods events during the last 42 years, i.e., 1980, 1986, 1988, and 1994 (Figure 4); however, after 1995, no large flood was recorded. The incidence of extremely low flow has increased since 2002, and small floods were not observed after 2008. However, large floods, low flow, and small flow had similar interpretation in decreasing conditions (Figure 4).

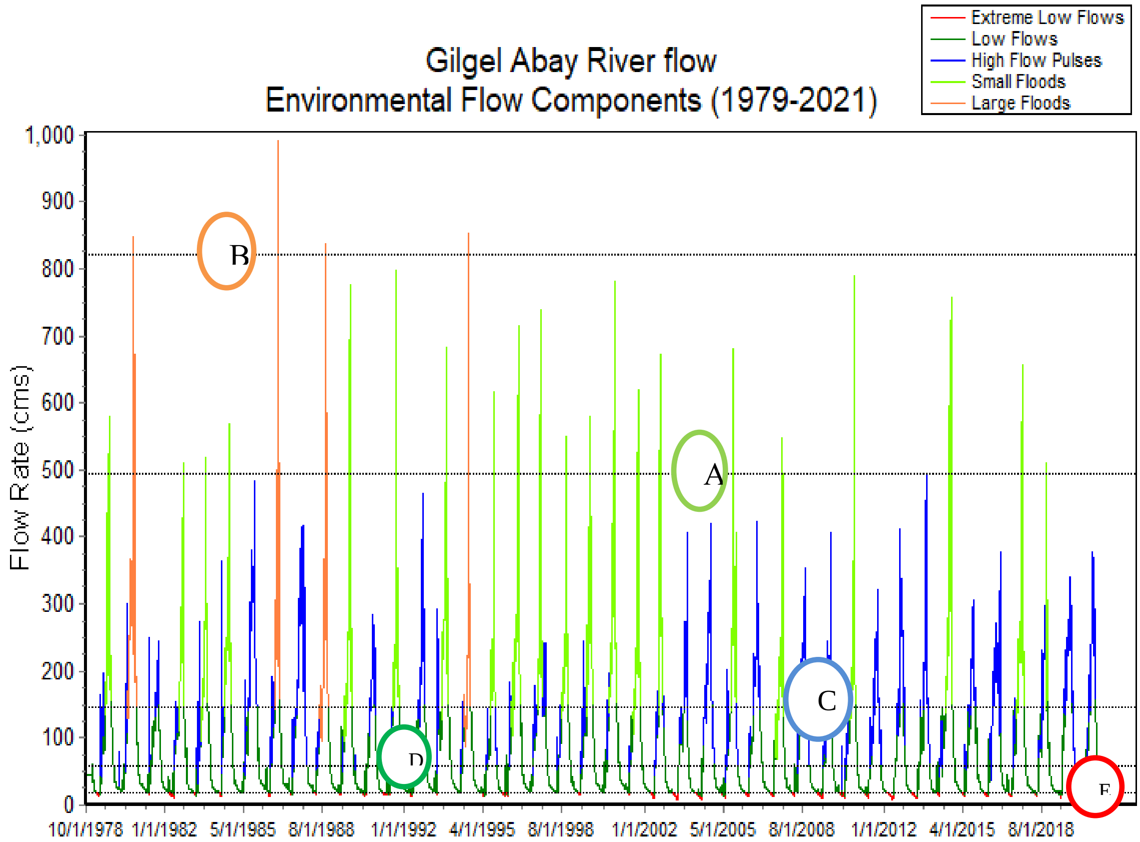



**Figure 4. The five environmental flow components of the Gilgel Abay River:**

The horizontal line (a) shows the small flood minimum peak flow and the horizontal line (b) shows the large flood minimum peak flow, high flow pulse minimum peak flow (C), low flow minimum peak flow (D) and extremely low flow minimum peak flow (E).

### 4.1.4. Flow duration curve

This flow indices is used in deciding the minimum flow requirement in maintaining the health of aquatic ecosystems or streams; i.e. like 95 percentile flow is the minimum flow of streams which is meant to be suficient in supporting life in streams or aquatic ecosystems.The flow values and exceedance probability for each month are displayed graphically alongside the FDC findings, arranged from highest to lowest (Figure 5). In this study, low flow circumstances were defined as those with a 95% flow exceedance probability (95% of the time the river receives 15.81 $m^3/s$ of discharge). The effect is evident at 100% exceedance probability, indicating that the low flow may diminish throughout the dry seasons of 1979–2020 (Figure 5b). In this season, monthly flow duration curves of May show low flow at 100 % exceedance probability (close to 100% of the probability of the occurrence of 8 $m^3/s$ of discharge) (Figure 5a).

At the low flow zone, the 1979–2020 period is equal to or exceeded 100% of the time was 8 $m^3/s$ this was the lowest-flow for this study period record, whereas the inter-annual year high flow < (Q10), which is stream flow that is equal to or exceeded less than 10% of the time was 240.36 $m^3/s$, represents the state of high flows. According to the IHA analysis, in cases where significant alterations like dam construction and intensive irrigation projects are done within this river system, the amount of discharge ranges from 18.47 to 15.81 $m^3/s$ at 90 % and 95 % of the time respectively during low flow and 50% (57.67 $m^3/s$) during high flow periods were the minimum EFRs to maintain the river functioning.

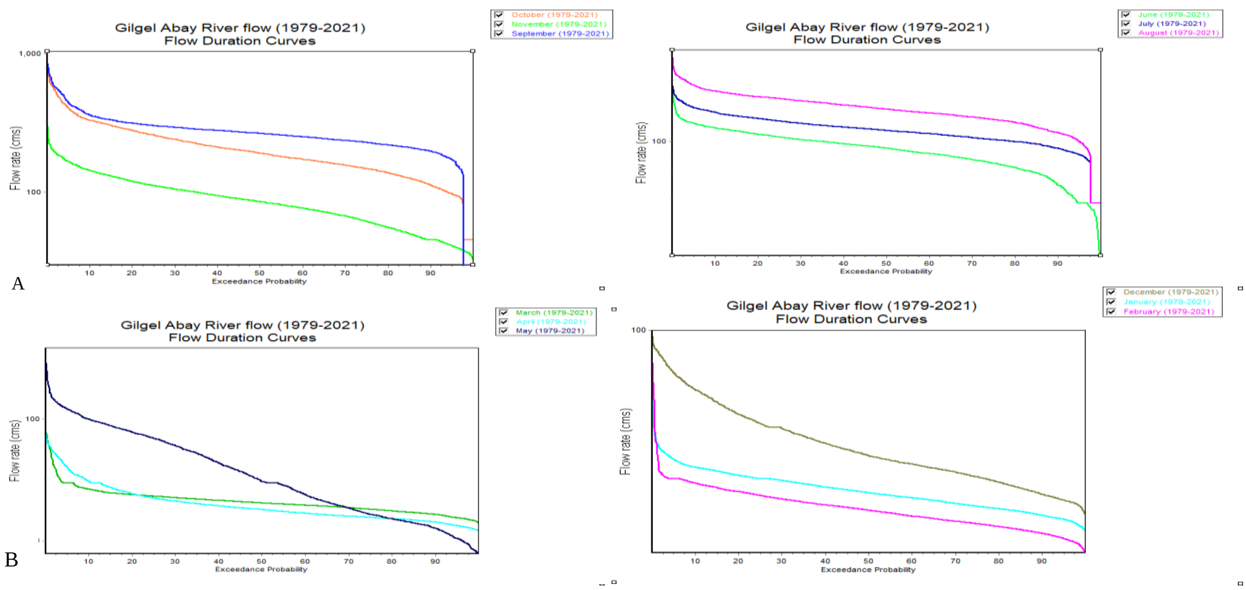



**Figure 5. Flow duration curve at the Gilgel Abay (wet =A; dry season = B)**

## 4.2. Environmental parameters

Most of the environmental variables measured in the field and laboratory showed significant differences between sampling sites ($p<0.05$) (Table 5). Excepting DO the concentration of most environmental variables was lower in the less impacted sites than the other categories for instance, the temperature at the reference (Fendka) and Chimba sites was low compared to other sampling sites, while it was higher at the Dibkan sites. The mean pH values indicated a slightly alkaline condition ranging from 8.11±0.03 (S3) to 8.60±0.04 (S2). There was a significant difference ($p < 0.05$) in dissolved oxygen concentration (with a mean value range of 7.74–5.44 mg/l) between sampling site. The highest DO was recorded relatively at reference sites (7.74 mg/l), while the lowest record was at sites with the highest agricultural activities, Telifa (S3) and Chimba (S4), with a mean value of 5.31 mg/l and 5.34 mg/l, respectively. Lowest conductivity 129.13 (µS/cm) was recorded at Dibkan (S2) site, while the highest recorded at Fendka (S1) site. There was significant difference in the amount of conductivity among sampling sites ($p < 0.05$).

The concentration of ammonium nitrogen (0.622 mg/L) at the Telifa (S3) site was high and can be attributed to animal wastes during livestock watering. Similarly, higher concentrations of SRP was recorded at Chimba (S4) and Tankua Ber (S5) with a mean value of 1.06 and 0.515 mg/l, respectively, both with intense agricultural activities. In spite of these, the lowest values (0.395 and 0.425 mg/l) of SRP have been recorded at the reference sites (S1) and Dibkan (S2) sites, respectively. The concentrations of $NO_3^-$ (mg/l) and $NO_2^-$ (mg/l) were significantly different ($p< 0.05$) among sampling sites. The highest mean value of total suspended solids (TSS) was recorded at the Telifa (S3) sampling site, while the lowest was recorded at the reference site (S1). In general the measured physicochemical parameters showed significant differences ($p<0.05$) among sampling sites, except for $NH_4^+$ and SRP during the study period.

**Table 5. Mean ± standard deviation of the main physico-chemical water quality parameters in Gilgel Abay River in 2023**

| Parameters | Fendka (S1) | Dibkan (S2) | Telifa (S3) | Chimba(S4) | Tankua Ber (S5) | p<0.05 |
|---|---|---|---|---|---|---|
| Temp (°c). | 24.18±1 | 26.50±0.94 | 25.55±0.2 | 24.48±0.15 | 25.23±1.0 | ** |
| pH | 8.50±0.1 | 8.6±0.03 | 8.11±0.0 | 8.19±0 .03 | 8.28±0.06 | ** |
| DO (mg/l) | 7.74±0. | 6.75±0.19 | 5.31±0.0 | 5.34±0.04 | 5.44±0.07 | * |
| EC (µS/cm) | 168.48±1. | 129.13±0.2 | 154.0±0.8 | 146.70±0.2 | 145.45±1.8 | * |
| $NH_4^+$(mg/l) | 0.114±0.1 | 0.108±0.0 | 0.622±0.8 | 0.024±0. | 0.084±0.0 | ns |
| $NO_3^-$(mg/l) | 1.229±0.3 | 2.640±1.2 | 4.250±0.2 | 3.740±0.0 | 3.825±0.1 | * |
| $NO_2^-$(mg/l) | 0.008±0.0 | 0.038±0.0 | 0.024±0.0 | 0.026±0. | 0.064±0.0 | * |
| SRP (mg/l) | 0.395±0.5 | 0.425±0.3 | 0.540±0.0 | 1.060±0.2 | 0.515±0.2 | ns |
| TSS (mg/l) | 0.003±0.0 | 0.015±0.01 | 0.065±0.0 | 0.01±0.0 | 0.025±0.0 | * |

(*), significant, (ns) insignificant

## 4.3. Human activities along the Gilgel Abay River

Human activities observed at each site are presented in Table 6. Telifa and Tankua Ber had the greatest degree of disturbance, with nine human activities being observed there. These two sampling sites were also highly impacted because they had large-scale water abstraction for irrigation. Livestock grazing and watering were the key activities being undertaken at every sampling site, except the Fendika site. However, the disturbance was minimal at the Fendka (S1) site, which is a reference site.

**Table 6. Human activities manifested at the five sampling sites (S1-S5) along the Gilgel Abay River, (numbers in brackets indicate the number of observed human disturbances)**

| Site ID | Local Name | Human activities |
|---|---|---|
| S1 | Upstream/Fendika | Bathing, cloth washing (2) |
| S2 | Dibikan/Above the Koga confluence | Deforestation, bathing, livestock watering and grazing (3) |
| S3 | Telifa | Deforestation, bathing, cloth washing, swimming, fishing, farming, water abstraction for irrigation, water abstraction for domestic use, livestock watering and grazing (9) |
| S4 | Chimba | Deforestation, charcoal production, bathing, fishing, swimming, and livestock watering and grazing (6) |
| S5 | Tankua Ber/ lower stream | Deforestation, fishing, cloth washing, bathing, swimming, spiritual purpose, water abstraction for irrigation, livestock watering and grazing, farming (9) |

## 4.4. Habitat quality assessment

Habitat quality assessment (HQA) values indicate the degree of diversity of a habitat's natural hydro morphological features (Kijowska-Strugała *et al*., 2021). The habitat quality assessment scores showed significant differences among the sampling sites (Df = 4, p = 0.0002). Fendika (S1) had the highest score which is 88% and the site also had the lowest human disturbances. S5 has the lowest (33%) habitat quality assessment scoring value. The scoring for various parameters per site is described in Table 7. Channel flow status, velocity, and depth regime had the highest scoring value at S1 sampling reaches, while vegetative protection and riparian width had the lowest scoring values at S4 and S5 sites.

**Table 7: Habitat quality assessment (HQA) scores for each sampling site in the Gilgel Abay River (S1= Upstream/Fendika; Dibikan/Above the Koga confluence, Telifa, Chimba/ above bridge; and Tankua Ber/ lower stream)**

| S/No | Habitat Parameters | S1 | S2 | S3 | S4 | S5 |
|---|---|---|---|---|---|---|
| 1 | Substrate Quality | 15 | 14 | 15 | 13 | 10 |
| 2 | Embeddedness | 18 | 16 | 10 | 16 | 7 |
| 3 | Velocity/Depth Regime | 19 | 18 | 18 | 16 | 15 |
| 4 | Sediment Deposition | 20 | 14 | 11 | 15 | 8 |
| 5 | Channel Flow Status | 18 | 14 | 16 | 18 | 17 |
| 6 | Channel Alterations | 20 | 13 | 10 | 18 | 15 |
| 7 | Frequency of Riffles | 19 | 10 | 11 | 13 | 0 |
| 8 | Bank Stability | 18 | 15 | 5 | 7 | 9 |
| 9 | Vegetative Protection | 17 | 16 | 13 | 5 | 4 |
| 10 | Riparian Width | 16 | 10 | 12 | 5 | 3 |
| Total | | 180 | 140 | 121 | 126 | 88 |
| HQA Score (Total/200) | | 0.9 | 0.7 | 0.605 | 0.63 | 0.44 |
| Human Activities | | 0.02 | 0.03 | 0.09 | 0.06 | 0.11 |
| Actual HQA (HQA-Human activities) | | 0.88 | 0.67 | 0.515 | 0.57 | 0.33 |
| Final HQA score (%) | | 88 | 67 | 51.5 | 57 | 33 |
| Class of RHQA (King et al., 2006) | | B | C | D | D | E |

RHQA: river habitate quality class, B: few modification, C: Moderately modification, D:largely modified, E: loss of natural habitat

## 4.5. Benthic macroinvertebrate community structure

A total of 38 benthic macroinvertebrate taxa were collected during the sampling period. Among the 38 taxa, 28 were identified at the family level, while the rest, which are under the family Chironomidae, were identified to the genus and species level. A detail of benthic macroinvertebrates identified, diversity, and the proportion of taxa occurrence across sampling sites are shown (Tables 8 and 9). High numbers of taxa (21 and 19) were recorded at less impacted sites (S1) and (S2), respectively. These sites are relatively less impaired upstream sites, while lower numbers of taxa (10 and 12) were recorded at the Telifa and Chimba sites, respectively, which were affected by anthropogenic impacts.

The family Heptageniidae and Tricorythidae were more abundant at the less impacted site (S1). Whereas the family Belostomatidae and Coenagrinoidae were more predominant at the S3 site, while the family Chironomidae was distributed throughout all sampling sites. From the pollution-intolerant group, Ephemeroptera was represented by six families, while tolerant taxa Diptera were represented by three groups. The highest proportion of Ephemeroptera (52.34% and 73.3%) were found at S1 and S4 respectively, because these sites contain suitable substrates like grass, stones, rocks, and submerged pieces of wood for those taxa to live on or in. The lowest proportion of Chironomidae was found at S1 (5.47%) and S2 (6. 13%) (Table 9). The family Hydropsychidae was found abundantly along the minimally impacted sampling site (S2). Tricorythidae and Potamanthidae were recorded only at sampling sites (S1 and S2). The families belonging to the order of Ephemeroptera and Trichoptera macroinvertebrates were not found at the impacted sites (S3 and S5) of the Gilgel Abay River.

From the Chironomidae taxa, only three subfamilies were observed (subfamilies Chironominae, Orthocladiinae, and Tanypodinae). Sub-family Chironominae were comprised of *Chironomus imicola*, *Chironomus formosipennis, Polypedilum wittei*, and *Dicrotendipes septemmaculatus*, while sub-family Tanypodinae were represented by *Procladius brevipetiolatus*, *Ablabesmyia sp1*, *Ablabesmyia sp2*, and *Ablabesmyia sp3*. The subfamily Orthocladiinae contains *Symbiocladius Kieffer* and *Cricotopus flavocinctus*. *Ablabesmyia sp2,* and *Polypedilum wittei* were found in polluted sites (S3). The distribution of *Chironomus formosipennis* was abundant at livestock watering, bathing, and cloth-washing exposure site (4).

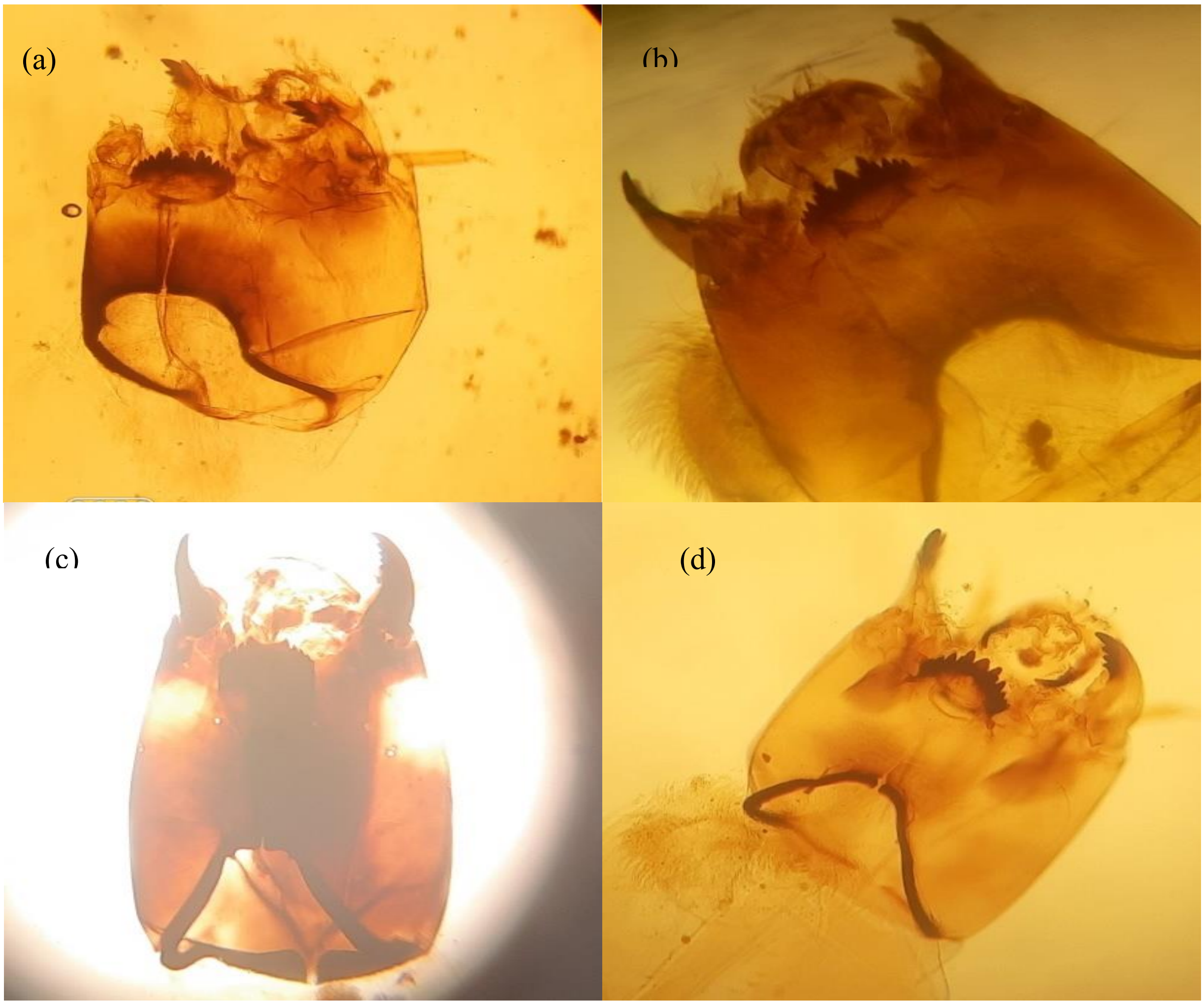

(a)
(b)
(c)
(d)


**Figure 6: Head capsules of larval Chironomidae taxa found in Gilgel Abay River**

showing the mentum and ligula: (a) *Chironomus imicola*, (b) *Chironomus formosipennis*, (c) *Symbiocladius Kieffer*, (d) *Dicrotendipes Septemmaculatus*, (e) *Procladius brevipetiolatus*, (f) *Cricotopus flavocinctus*, (g) *Polypedilum wittei*, (h) *Ablabesmyia sp1*, (i) *Ablabesmyia sp2*, ( j) *Ablabesmyia sp3*

**Table 8. Benthic macroinvertebrate score, feeding functional groups and abundance in Gilgel Abay River**

| Order | Family | ETHbios score [c] | FFG | Number of organism per sample sites | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | | S1 | S2 | S3 | S4 | S5 |
| | Elmidae | 7 | SC[e] | 0 | 8 | 0 | 1 | 1 |
| | Gyrinidae | 5 | PD[e] | 2 | 1 | 6 | 0 | 1 |
| | Hydrophilidae | 5 | PD[e] | 1 | 0 | 9 | 0 | 7 |
| Coleoptera | Dytiscidae | 5 | PD[e] | 1 | 4 | 2 | 1 | 9 |
| | Gomphidae | 6 | PD[a] | 1 | 2 | 1 | 1 | 2 |
| | Aeshnidae | 7 | PD[e] | 0 | 0 | 1 | 0 | 1 |
| | Libellulidae | 5 | PD[b] | 2 | 3 | 8 | 0 | 2 |
| Odonata | Coenagrinoidae | 5 | PD[b] | 7 | 1 | 47 | 22 | 13 |
| | Nepidae | 3 | PD[d] | 0 | 0 | 0 | 0 | 4 |
| | Notonectidae | 3 | PD[e] | 15 | 0 | 1 | 0 | 2 |
| | Belostomatidae | 3 | PD[b] | 6 | 16 | 19 | 3 | 2 |
| | Naucoridae | 6 | PD[e] | 2 | 5 | 0 | 0 | 0 |
| Hemiptera | Velidae | 5 | PD[d] | 5 | 0 | 0 | 1 | 0 |
| Oligochaeta | Earthworm | 1 | CG[d] | 0 | 0 | 0 | 0 | 1 |
| | Planorbidae | 3[f] | SC[b] | 1 | 1 | 0 | 0 | 4 |
| Mollusca | Lymnaeidae | 3 [f] | SC[b] | 10 | 2 | 0 | 0 | 4 |
| | Leptophlebiidae | 9 | CG[ad] | 1 | 0 | 0 | 0 | 0 |
| | Baetidae | 6 | CG[a] | 11 | 72 | 0 | 134 | 0 |
| | Canidae | 6 | CG/SC[d] | 15 | 6 | 0 | 6 | 0 |
| | Heptageniidae | 9 | SC[ad] | 31 | 12 | 0 | 0 | 0 |
| | Tricorythidae | 8 | CG[d] | 7 | 6 | 0 | 0 | 0 |
| Ephemeroptera | Potamanthidae | 7[g] | CF[d] | 2 | 3 | 0 | 0 | 0 |
| | Hydrosychidae | 6 | CF[ae] | 4 | 108 | 0 | 2 | 0 |
| Trichoptera | Ecnomidae | 8 | CF[e] | 1 | 0 | 0 | 6 | 0 |
| Lepidoptera | Pyralidae | 12 [f] | SH[e] | 0 | 0 | 0 | 2 | 0 |
| | Chironomidae | 1 | CG[b] | 4 | 8 | 16 | 12 | 3 |
| Diptera/True Flies | **Sub family Chironominae** | | | | | | | |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | *Chironomus formosipennis* | | | 0 | 0 | 0 | 12 | 0 |
| | *Chironomus imicola* | | | 1 | 0 | 0 | 0 | 0 |
| | *Dicrotendipes septemmaculatus* | | | 0 | 0 | 0 | 0 | 1 |
| | *Polypedilum wittei* | | | 0 | 3 | 12 | 0 | 2 |
| | **Sub family Tanypodinae** | | | | | | | |
| | *Procladius brevipetiolatus* | | | 2 | 0 | 0 | 0 | 0 |
| | *Ablabesmyia sp2* | | | 0 | 0 | 3 | 0 | 0 |
| | *Ablabesmyia sp1* | | | 0 | 3 | 0 | 0 | 0 |
| | *Symbiocladius Kieffer* | | | 0 | 1 | 0 | 0 | 0 |
| | **Sub family Orthocladiinae** | | | | | | | |
| | *Cricotopus flavocinctus* | | | 0 | 1 | 0 | 0 | 0 |
| Diptera/True Flies | Tabanidae | 6 | CG/PD[ae] | 0 | 2 | 0 | 0 | 0 |
| | Simuliidae | 5 [f] | CF[ae] | 0 | 1 | 0 | 0 | 0 |
| Total number of family | 28 | | | 21 | 19 | 10 | 12 | 15 |

[a] Gholizadeh & Heydarzadeh, 2020, [b]Ilmi *et al*., 2023, [c]Aschalew Lakew *et al*., 2015a, [d] Bouchard, 200, [e] Nessimian & Luiz, 2010; [f] Dickens & Graham, 2002; [g]Armitage *et al*., 1983

**Table 9. The proportion of taxa occurrence a cross sampling sites**

| Order | Family | S1 | S2 | S3 | S4 | S5 |
|---|---|---|---|---|---|---|
| Coleoptera | Elmidae | 0.00 | 3.07 | 0.00 | 0.52 | 1.79 |
| | Gyrinidae | 1.56 | 0.38 | 5.41 | 0.00 | 1.79 |
| | Hydrophilidae | 0.78 | 0.00 | 8.11 | 0.00 | 12.50 |
| | Dytiscidae | 0.78 | 1.53 | 1.80 | 0.52 | 16.07 |
| Odonata | Gomphidae | 0.78 | 0.77 | 0.90 | 0.52 | 3.57 |
| | Aeshnidae | 0.00 | 0.00 | 0.90 | 0.00 | 1.79 |
| | Libellulidae | 1.56 | 1.15 | 7.21 | 0.00 | 3.57 |
| | Coenagrinoidae | 5.47 | 0.38 | 42.34 | 11.52 | 23.21 |
| Hemiptera | Nepidae | 0.00 | 0.00 | 0.00 | 0.00 | 7.14 |
| | Notonectidae | 11.72 | 0.00 | 0.90 | 0.00 | 3.57 |
| | Belostomatidae | 4.69 | 6.13 | 17.12 | 1.57 | 3.57 |
| | Naucoridae | 1.56 | 1.92 | 0.00 | 0.00 | 0.00 |
| | Velidae | 3.91 | 0.00 | 0.00 | 0.52 | 0.00 |
| Oligochaeta | Earthworm | 0.00 | 0.00 | 0.00 | 0.00 | 1.79 |
| Mollusca | Planorbidae | 0.78 | 0.38 | 0.00 | 0.00 | 7.14 |
| | Lymnaeidae | 7.81 | 0.77 | 0.00 | 0.00 | 7.14 |
| Ephemeroptera | Leptophlebiidae | 0.78 | 0.00 | 0.00 | 0.00 | 0.00 |
| | Baetidae | 8.59 | 27.59 | 0.00 | 70.16 | 0.00 |
| | Canidae | 11.72 | 2.30 | 0.00 | 3.14 | 0.00 |
| | Heptageniidae | 24.22 | 4.60 | 0.00 | 0.00 | 0.00 |
| | Tricorythidae | 5.47 | 2.30 | 0.00 | 0.00 | 0.00 |
| | Potamanthidae | 1.56 | 1.15 | 0.00 | 0.00 | 0.00 |
| Trichoptera | Hydroesychidae | 3.13 | 41.38 | 0.00 | 1.05 | 0.00 |
| | Ecnomidae | 0.78 | 0.00 | 0.00 | 3.14 | 0.00 |
| Lepidoptera | Pyralidae | 0.00 | 0.00 | 0.00 | 1.05 | 0.00 |
| Diptera | Chironomidae | 3.13 | 3.07 | 14.41 | 6.28 | 5.36 |
| | Tabanidae | 0.00 | 0.77 | 0.00 | 0.00 | 0.00 |
| | Simuliidae | 0.00 | 0.38 | 0.00 | 0.00 | 0.00 |
| | Total | 100.00 | 100.00 | 100 | 100 | 100 |

## 4.6. Benthic macroinvertebrate metrics

In the present study, metrics of taxonomic abundance and composition, richness and diversity, sensitivity and tolerance, and feeding functional groups were included. Based on the comprehensive literature review of previous studies conducted in the same agro-ecological conditions (highland rivers in tropical regions), about 26 benthic macroinvertebrate metrics were selected (Table10).

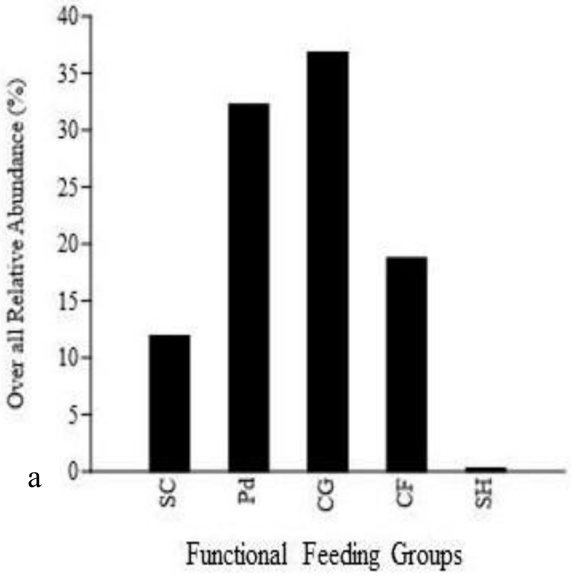
**Table10. Various metrics used in Gilgel Abay River during the study period**

| S/N | Metrics | Sampling sites | | | | |
|---|---|---|---|---|---|---|
| | | S1 | S2 | S3 | S4 | S5 |
| 1 | No. of Taxa (family) | 21 | 19 | 10 | 12 | 15 |
| 2 | No. of Ephemeroptera indi. | 67 | 99 | 0 | 140 | 0 |
| 3 | No. Trichoptera *indi.* | 5 | 108 | 0 | 8 | 0 |
| 4 | No. of ET *indi.* | 72 | 207 | 0 | 148 | 0 |
| 5 | No. of Chironomidae indi. | 4 | 8 | 16 | 12 | 3 |
| 6 | No. of Tolerant taxa indi. | 54 | 40 | 109 | 41 | 52 |
| 7 | No. of Intolerant Taxa indi. | 75 | 221 | 1 | 150 | 4 |
| 8 | %Ephemeroptera indi. | 52.34 | 37.95 | 0.0 | 73.30 | 0.0 |
| 9 | %Trichoptera indi. | 3.91 | 41.38 | 0.0 | 4.19 | 0.0 |
| 10 | % ET taxa indi. | 56.25 | 79.33 | 0.0 | 77.49 | 0.0 |
| 11 | %Diptera indi. | 5.47 | 7.33 | 19.82 | 12.57 | 8.93 |
| 12 | % Tolerant indi. | 41.22 | 16.52 | 72.19 | 21.47 | 7.14 |
| 13 | % Intolerant indi. | 58.78 | 83.48 | 27.81 | 78.53 | 92.86 |
| 14 | % Chironomids and Oligochaeta indi. | 3.13 | 3.07 | 14.41 | 6.28 | 7.15 |
| 15 | ET/Chironomidae | 18 | 26 | 0 | 12 | 0 |
| 16 | No. Chironominae taxa | 1 | 1 | 1 | 1 | 2 |
| 17 | No. of Chironominae indi. | 1 | 3 | 12 | 12 | 3 |
| 18 | No. of Tanypodinae Taxa | 2 | 2 | 2 | 0 | 0 |
| 19 | No. of Tanypodinae indi. | 2 | 4 | 3 | 0 | 0 |
| 20 | No. of Orthocladiinae indi. | 0 | 1 | 0 | 0 | 0 |
| 21 | % of Chironominae indi./ Chironomidae | 25 | 38 | 75 | 100 | 100 |
| 22 | %Total Chironomidae /total benthic indi. | 3 | 3 | 14 | 6 | 5 |
| 23 | Evenness | 0.62 | 0.34 | 0.55 | 0.27 | 0.72 |
| 24 | Shannon-Wiener index | 2.54 | 1.86 | 1.70 | 1.16 | 2.38 |
| 25 | ETHbios | 114 | 102 | 45 | 58 | 62 |
| 26 | ASPT | 5.43 | 5.37 | 4.50 | 4.83 | 4.13 |

## 4.7. Functional feeding groups of macroinvertebrates

In this study, 28 families of benthic macroinvertebrates were recorded belonging to five functional feeding groups. The five major FFGs recognized in this study were collector-gatherers (CG), collector-filterers (CF), scrapers (SC), predators (PD), and shredders (SH).

The overall functional feeding group (FFG) in the Gilgel Abay River collector-gatherer was the most abundant group (36.81%), followed by predator (32.26%), collector-filterer (18.74%), scraper (11.91%), and then shredder, which were found in very small amounts (0.27%) (Figure 7a). The spatial distribution of benthic macroinvertebrates (FFG) at each site has a different composition ( Figure 7b). The scrapers (S1 = 38.76%), collector-filterers (S2 = 45.21%), predators (S3 = 85.45% and S5 = 76.79%), and collector-gatherers (S4 = 78.01%) were the predominante FFGS found across sampling sites. There were significant differences ($p < 0.05$) in the composition of the functional feeding groups among sampling sites in the study river.



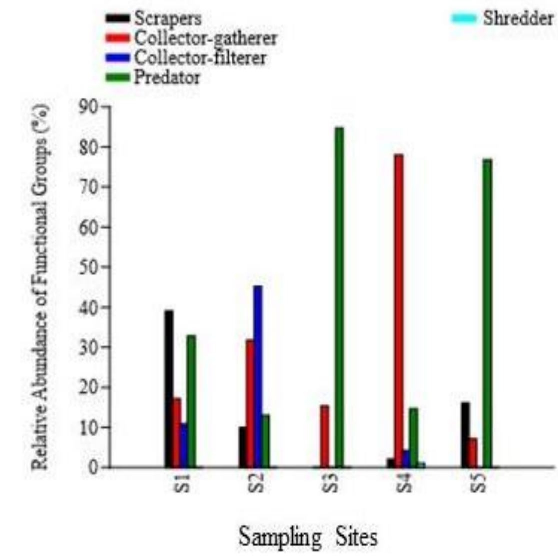
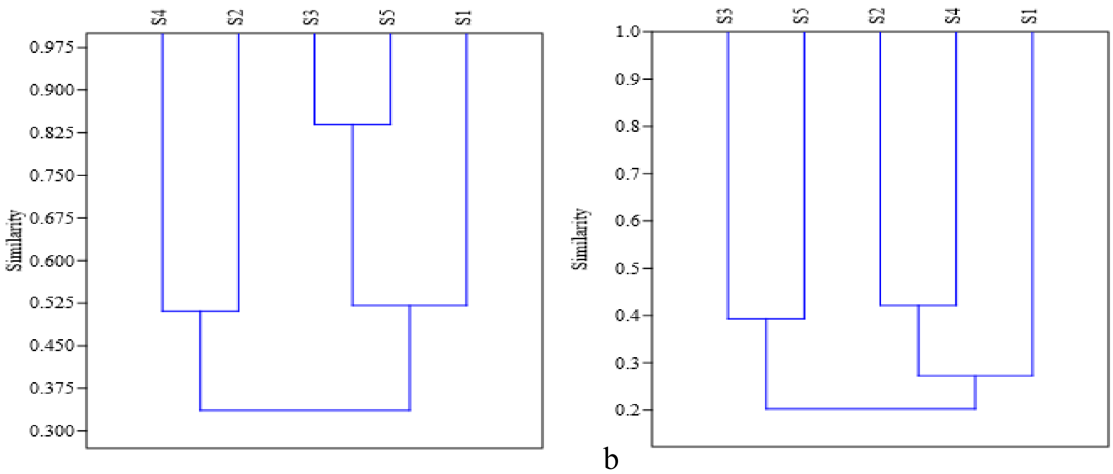




Figure7. The Overall relative abundance of feeding functional groups (a) and at sampling sites (b) of the Gilgel Abay River (CF = collector filterers, CG = collector gatherers, PD = predators, SC = scrapers, SH = shredders)

#### 4.7.1. Feeding functional groups spatial distribution among clusters

The FFG composition between sites 3 and 5 shows high similarity because they have the highest relative abundance of PD and collector-filterer, but the shredder was not present (Figure 8a). The functional feeding groups like SC, SH, and CF were not found at site 3, which is highly different from sites 1, and 2. S1 is slightly different from S3 and S5, it has high scrapers present, and PD shows very low abundance comparatively. S1 is separated into different HCA branches (Figure 8a). S2 and S4 indicated high similarity since they have a high abundance of CG and the lowest proportion of PD observed. This implies that at these sites there is an uneven distribution of functional feeding groups. While a more even distribution of FFG is found in the less impacted sites (S1), which have evenness (0.9002), while at the S4 site, which had a lower evenness value (0.4963) indicates uneven distribution of FFGs due to various human activities on the riverbanks.

On the other hand, from the taxa distribution perspective, S2 and S4 are more or less similar than S1, whereas S3 and S5 are on a different HCA branch. Since most taxa are distributed in both (S3 and S5) sites, they have a very low community composition compared to the other sites because the highly human-impacted site results in reduced taxa diversity, which may contribute to the dissimilarity of benthic macroinvertebrate distribution with other sites (Figure 8b). There were a significant difference in abundance for all FFG in the study river, which implies there was trophic structure imbalance at all study sites (p-value = 0.04<0.05) due to human impairment, indicating that the Gilgel Abay River needs a balanced trophic structure.




b

a

**Figure 8. Hierarchical cluster analysis using UPGMA on benthic macroinvertebrates assemblages based on FFG distribution**

(a) and taxa distribution at family level (Elmidae, Gyrinidae, Hydrophilidae, Dytiscidae, Gomphidae,Aeshnidae, Libellulidae, Coenagrinoidae, Nepidae, Notonectidae, Belostomatidae, Naucoridae, Velidae, Earthworm, Planorbidae, Lymnaeidae, Leptophlebiidae, Baetidae, Canidae, Heptageniidae, Tricorythidae, Potamanthidae, Hydroesychidae, Ecnomidae, Pyralidae, Chironomidae, Tabanidae, Simulidae) (b) in Gilgel Abay River

## 4.8. Correlation between metrics and environmental variables

Following the procedures stated in the data analysis section, five metrics were retained for correlation analysis. These includes number of taxa at family level, Shannon-Weiner diversity index, % of intolerant taxa individuals, % Chironomidae and Oligochaeta, and ETHbios. Based on this Pearson's correlation was applied to examine the relationship among these single metrics and five selected environmental variables (DO, TSS, EC, TO, and $No_3^-$) using mean value of variables at each sampling site. From physico-chemical parameters, phosphate ($PO_4$) and ammonium ($NH_4^+$) were omitted from the analysis as there was no significant variation among sites, likewise pH and $No_2^-$ has not strong negative and positive correlation with metrics. The number of taxa at family level (r = 98 and 0.93) has a strong positive correlation with DO and ETHbios. TSS has a strong positive correlation (r = 0.95) with % Chironomidae and Oligochaeta has positive correlation (r = 0.97). $No_3^-$ has a strong negative correlation with with DO (r= -0.98) and ETHbios (r = -0.97) (Table11).

**Table 11: Correlations between metrics and environmental variables in Gilgel Abay River**

| | No. TaxF | % ITOI | % Chir & Oli. | Sha-H | ETHbios | DO | EC | To | $No_3^-$ | TSS |
|---|---|---|---|---|---|---|---|---|---|---|
| No. TaxF | 1 | 0.54 | 0.06 | 0.22 | 0.00 | 0.02 | 0.95 | 0.99 | 0.02 | 0.17 |
| % ITOI | 0.37 | 1 | 0.20 | 0.88 | 0.68 | 0.96 | 0.33 | 0.71 | 0.81 | 0.21 |
| % Chir & Oli. | -0.86 | -0.68 | 1 | 0.61 | 0.07 | 0.18 | 0.76 | 0.95 | 0.11 | 0.01 |
| Sha-H | 0.67 | 0.10 | -0.31 | 1 | 0.34 | 0.30 | 0.53 | 0.85 | 0.31 | 0.76 |
| ETHbios | 0.98 | 0.26 | -0.85 | 0.55 | 1 | 0.01 | 0.91 | 0.95 | 0.01 | 0.18 |
| Do | 0.93 | 0.03 | -0.71 | 0.59 | 0.97 | 1 | 0.69 | 0.83 | 0.00 | 0.30 |
| EC | 0.04 | -0.56 | 0.19 | 0.37 | 0.07 | 0.25 | 1 | 0.06 | 0.64 | 0.96 |
| To | -0.01 | 0.23 | 0.04 | -0.12 | -0.04 | -0.13 | -0.87 | 1 | 0.67 | 0.63 |
| $No_3^-$ | -0.94 | -0.15 | 0.80 | -0.58 | -0.97 | -0.98 | -0.29 | 0.26 | 1 | 0.17 |
| TSS | -0.72 | -0.68 | 0.95 | -0.19 | -0.71 | -0.58 | 0.04 | 0.29 | 0.72 | 1 |

NB: No.Taxf = number of taxa at family level, %Chiro&Oli = % of Chironomidae and Oligochaeta taxa individual, %ITOI= % Intolerant individual, Sha-H = Shannon wiener diversity index, ETHbios= Ethiopia biotic score

## 4.9. Macroinvertebrate distribution in relation to environmental parameters

The multivariate analysis of RDA ordination was utilized to examine the impact of environmental variables on the distribution of benthic macroinvertebrates linked to the sampling locations. From physico-chemical parameters, phosphate ($PO_4$) and ammonium ($NH_4^+$) were omitted from this analysis as there was no significant variation among sites. The other parametres EC, pH, Tem, dissolved oxygen, $No_2^-$, $No^-_3$, and TSS were retained as the main parameters that affect the community structure of benthic macroinvertebrates. The selected environmental parameters clearly affect the distribution of macroinvertebrate (Figure 9). The first axis of the RDA eigenvalue (0.535) explained the greatest variation of macroinvertebrate distribution across sampling sites by the environmental variables. There is a strong degree of variance among species along ordination axis (eigenvalues >0.3) (Helson *et al.*, 2006). Axis 1 53.5% of the cumulative variation among taxa data along sampling sites were explained by the environmental variables, whereas the second axis explained 23.4% of the variation (Table 12). The contributions of pH (34.2%), and DO (31.4%) were higher for the variation of macroinvertebrates community structure. The first axis is primarily associated with agriculturally impacted sites (S3 and S5), and it is significantly related to EC, TSS, and $No_2^-$. Conversely, the negative side of the first axis is strongly correlated with DO and pH at minimum impaired sites (S1 and S2).

**Table 12. Summary statistics of redundancy analysis (RDA) results for species–environment relationships**

| Axes | 1 | 2 | 3 | 4 |
|---|---|---|---|---|
| Eigenvalues | 0.5354 | 0.234 | 0.1270 | 0.1035 |
| Explained variation (cumulative): | 53.54 | 76.95 | 89.65 | 100.0 |
| Pseudo-canonical correlation | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| Sum of all canonical eigenvalues: 0.999; i.e. the summation of each axis eigenvalue | | | | |

The triplot of redundancy analysis (RDA) ordination showed the distribution of benthic macroinvertebrates along with physico-chemical parameters across sampling sites. Sensitive and moderately sensitive macroinvertebrate families, including the Elmidae, Potamanthidae, Tricorythidae, Simuliidae, Heptageniidae, Hydrophilidae, Caenidae, and Naucoridae, were aligned in a river with a high concentration of dissolved oxygen. This shows that these organisms can be indicators of good water quality. Chironomidae are lined with sampling sites with high concentrations of nitrate ($NO_3^-$) and nitrite ($NO_2^-$), and can be considered an indicator of nutrient enrichment. Therefore; the dominance of these taxa can predict the presence of low dissolved oxygen tolerant taxa in the water body. Notonectidae and Lymnaeidae were dominant in river where the concentration of conductivity is high (EC) (Figure 9).

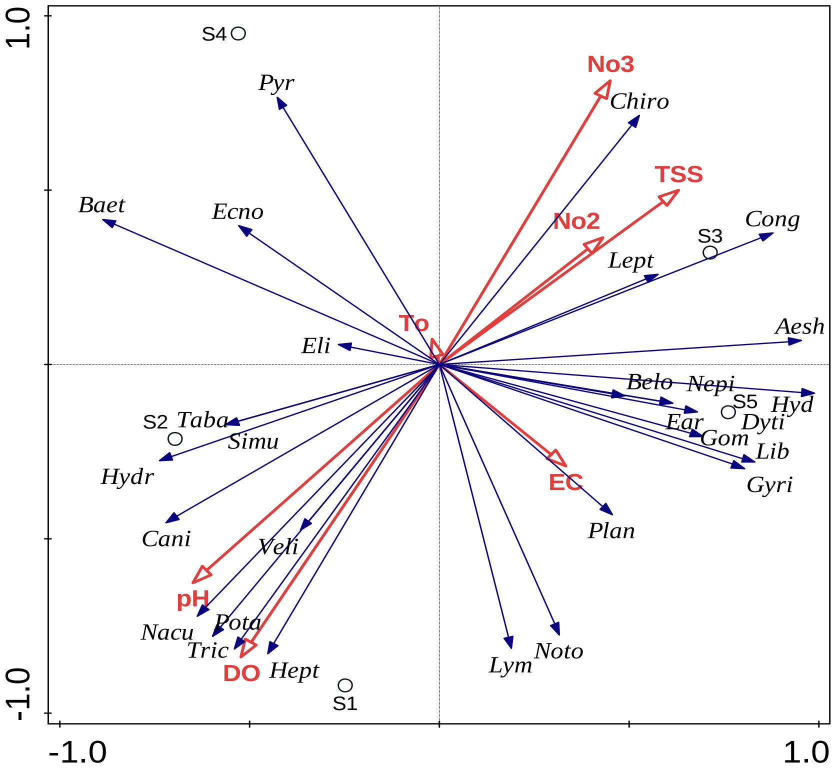



**Figure 9. Triplot redundancy analysis (RDA) showing the benthic macroinvertebrates distribution in relation with environmental variables among sampling sites**

Black circles indicate sampling sites. The red arrows show the physico-chemical parameters. Blue arrows indicate benthic taxa; dissolved oxygen (DO), temperature (To), total suspended solid (TSS), nitrite ($NO_2^-$), nitrate ($NO_3^-$), and conductivity (EC) in relation to RDA axes 1 and 2. Macroinvertebrates are abbreviated (Eli= Elmidae, Dyti= Dytiscidae, Gyri = Gyrinidae, Hydr = Hydrophilidae, Gom = Gomphidae, Aesh = Aeshnidae, Lib = Libellulidae, Cong = Coenagrinoidae, Nepi = Nepidae, Noto = Notonectidae, Belo = Belostomatidae, Nacu = Naucoridae, Veli = Velidae, Ear = Earthworm, Plan= Planorbidae, Lym = Lymnaeidae, Lept= Leptophlebiidae, Baet = Baetidae, Cani = Canidae, Hept = Heptageniidae, Tric = Tricorythidae, Pota= Potamanthidae, Hyd = Hydroesychidae, Ecno = Ecnomidae, Pyr = Pyralidae, Chir = Chironomidae, Taba = Tabanidae, Simu = Simulidae.

# 5. DISCUSSION

## 5.1. Flow indices

### 5.1.1. Monthly flow

During the dry season, the coefficients of dispersion (COD) values were higher for all durations of flow, implying greater variability in a dry season's monthly flow. Despite of this, a low coefficient of dispersion (COD) value was observed in September, indicating lower flow variability in this month (Table 3). The smallest flow regime variations can influence the distribution and abundance of specific species of fauna and flora (Wetmore *et al*., 1990). The seasonal flows in the dry season (February, March, and April), as revealed in Figure 2a and during the wet season (July, August, and September) as indicated in Figure 2b, showed a decreasing trend mainly due to increasing human-induced impacts (pump and diversion of water for irrigation), consequently, the river ecology is becoming degraded. This finding was in accordance with the study results obtained on the Gumara River ( Wubneh Belete *et al*., 2020), and also Lloyd *et al*. (2004), who reported that 45 (64%) of 70 reviewed papers examined flow alteration in relation to water abstraction or withdrawals for irrigation. The present study results revealed that the Gilgel Abay River ecology is becoming degraded due to a decreasing of ecologically relevant flows.

### 5.1.2. Low and high Flow

The magnitude and duration of flow alteration are related to increases in ecological responses (Poff *et al*., 2010). The 90-day low flow had a statistically significant decreasing trend over the study period at $p < 0.05$. This result is in agreement with the feedback received from the local people. The decrease in low flow after 2003 might be due to the consequence of increasing anthropogenic activities performed on the Gilgel Abay River, such as water abstraction through pumping and diversion for irrigation and livestock trampling and grazing. This study results revealed that the coefficients of dispersion (COD) values are greater at high flow for all durations of flow, indicating greater variability in high flow. High flow variability (i.e., high coefficient of variation of daily flows) was favored to species with generalized feeding strategies and preferences for silt (Poff & Allan, 1995), the proliferation of non-indigenous species, and the loss of longitudinal and lateral connectivity of rivers and floodplain wetland ecosystems, which affect the abundance, diversity, and composition of native species in rivers (Poff *et al*., 2010; Pardo- loaiza *et al*., 2021). Comparatively, the low flow has a lower value of coefficients of dispersion (COD) (Table 4), showing lower variability in low flow. The hydrological river flow regime's variability is triggered by various interactions between human interventions and natural variables (Min *et al.*, 2011).

As observed in this study, the volumetric flow of the Gilgel Abay River showed a decreased trend. The maximum flows, similarly to the low flows, decreased over the study periods. The 1-day maximum is a good indicator for large flood decreases as it indicates individual peaks rather than the mean of the other durations like 3- Days, 7- Days, 30-Days, and 90-Days (Table 4 and Figure 3). From these results, we found the river has a decrease in large floods and low flows during the wet and dry seasons, which is the consequence of unmanaged pump irrigation and deforestation (where a decrease in infiltration rate leads to reduced base flow) and impacts the aquatic biota. All of these will undoubtedly lower aquatic-riparian vegetation, migratory fish spawning habitat, and macroinvertebrate abundance, distribution, and diversity. In agreement with the present study, the abundance of migratory fish species were impacted in the Gumara River because of excess irrigation water abstraction in March to May (Eshete Dejen *et al*., 2013; Wubneh Belete *et al*., 2020). Flow reduction influences larvae and juvenile fishes (Bradford, 1997) and different ecological response variables (Matthaei *et al*., 2010).

### 5.1.3. Environmental flow components (EFC)

Since 2002, there has been a high frequency of extreme low flow occurrences, indicating that the inference of drought periods was greater, and river levels dropped to very low levels, which can be stressful for many organisms. Extremely low flows could have an impact on predator-prey relationships, may concentrate aquatic prey for some species (Wubneh Belete *et al*., 2020), or may be dry out low-lying floodplain areas that may result in an imbalanced aquatic community structure. Starting from 2008, the occurrence of small floods decreased. This impacted the upstream and downstream movement of organisms and resulted in the disconnectivity of floodplains or flooded wetlands to access additional habitats. As used here, the decreasing of ecologically relevant "small flood," has impacted a river's aquatic communities because it determines the amount of aquatic habitat availability. These changes have hurt the biological and hydrological services that water ecosystems provide (Brown, 2003). In addition, after 1994, large flood flow was not observed, which led to a loss of the longitudinal and lateral connectivity between river channels and floodplain wetlands ecosystems (Ward & Stanford, 1995). This loss in connectivity can affect the migratory fish spawning areas and plant seedlings with prolonged access to soil moisture (Junk *et al.*, 1989), which attributed to the loss of biodiversity and nursery and foraging areas for many fishes and different vertebrates (Ward *et al*., 1999). This study's findings are in line with the studies conducted in the Gumara River (Wubneh Belete *et al*., 2020). Poff & Zimmerman. (2010) reviewed 165 papers regarding to aquatic or riparian biota response due to flow regime alteration, from these 145 papers reported responses in relarion to population or community change. Higher flows are responsible for channel widening due to fluvial and sediment deposits (Langat *et al*., 2019).

### 5.1.4. Flow duration curve

The FDC analysis can help understanding the change in the magnitude of stream flow values due to various factors (Langat *et al*., 2019). In this study, almost the flat slope between the $15^{th}$ and $90^{th}$ percentiles of the FDC during wet seasons (Figure 5a), indicated there is little variation in the stream’s flow magnitude between these ranges of percentiles. The sharped shape of FDC during the periods of dry season (low flow) is an indication that there is a dramatically decline in flow (Figure 5b). Similar to the current finding Tote *et al*. (2019) found a sharp FDC in the tropical river in West Africa and come to conclusion that the river is vulnerable to water scarcity during the dry season. Similar phenomena is evident in Gilgel Abay River during the dry season. Likewise, the slope of FDC shows the variability of flows in the lotic system (Belete Berhanu *et al*., 2015). The flow variability during the dry season may be a result of increasing water abstraction for domestic and irrigation in the catchment. Moreover, Tote *et al*. (2019) reported changes in the shape of FDC due to alterations of land use, climate change, and infrastructure development, which can have adverse effects on aquatic ecosystems and biodiversity and affect various physical, chemical and biological characteristics of the river or ecological responses within the watershed (Poff *et al*., 2010). It is understood that FDC provided from longer periods of record data has better representation of watershed conditions. If shorter periods are used, extreme climatic conditions (e.g., much wetter or drier than average) may influence the results (Mckay & Fischenich, 2016).

## 5.2. Environmental parameters

Environmental variables are the prior consideration to assess the water quality status at the time of study only because its result fluctuates with the diurnal and seasonal variation of the weather condition and input of pollutants at a certain time. It gives current information about water bodies through determining various *in situ* field measurement and laboratory analysis parameters at a given place and time. Most of these essential parameters were affected by the geomorphology, climate, and hydrology of the catchment. Sites found in the same ecoregion mostly have similar physico-chemical characteristics unless impacted by human interference. Thus, physico-chemical measurements can show the changes even if they were a snapshot. In this study, except for soluble reactive phosphorus ($PO_4$), and ammonium ($NH_4^+$), all physico-chemical parameters and nutrients measured showed significant differences at ($p<0.05$) among sampling sites.

In this study, the mean DO values (5.31-7.74 mg/L) were within the permissible limits of EPA (2011). The highest dissolved oxygen was recorded at S1 (7.74 mg/l) and S2 (6.75 mg/l) sample sites. These sites were relatively less disturbed by anthropogenic activities, and there was less organic load at the river bank which was covered with riparian vegetation. Compared to S1 and S2 sampling sites, the lowest concentration of dissolved oxygen (5.31 mg/l) was recorded at S3 (Table 5), indicating there was a high human disturbance at the site. The variability of dissolved oxygen along sampling sites might be related to temperature and concentrations of other nutrients. The results of this study were similar to the concept confirmed by Hussain *et al*. (2010) there is opposite relationship between water temperature and dissolved oxygen. Several scholars reported different mean values of dissolved oxygen concentrations from various water bodies, for instance, ranging from 1.54 to 7.86 (Amare Mezgebu *et al.*, 2019), 1.81 to 10.04 mg/l (Tesfaye Muluye *et al.*, 2024), 6.3 to 8.2 mg/l (Habtamu Tadesse, 2022), 0.36 to 12.0 mg/l (Getachew Beneberu *et al*., 2014), 7.5 to 8.0 (Aschalew Lakew, 2020), 2.8 to 7.89 mg/l (Adekanmi *et al*., 2023) and 2.5 to 7.6 mg/l (Muntalif *et al*., 2023). In a selected sites of some rivers in Omo-gibe and the Awash basin DO was >9 mg/ l (Getachew Beneberu *et al*., 2014), and 10.04 mg/l of DO was recorded at the upper stream sampling site of Awash River (Tesfaye Muluye *et al*., 2024), which was higher compared to the records of our study.

The mean pH values of the study river water ranged from 8.11 to 8.6, which is within the permissible limits for aquatic organisms and cattle consumption range from 6.5 to 8.5 EPA (2011) and 6.0 to 9.0 WHO (2006, 2017). In this study, when we go to the lower gradient along the river flow continuum, the total suspended solids increase, and as these are decomposed or oxidized by decomposers as they breakdown organic and dead materials, the release of carbon dioxide may increase the acidity of the water, which then leads to a decrease in the pH value. In the same river (Gilgel Abay), Yirga Kebede *et al*. (2016)

reported lower pH values 7.31 and 7.54. In the streams and rivers around Sebeta the lower mean value of pH (7.3) was recorded (Amare Mezgebu *et al*., 2019). This indicates that the highest TSS transported from different sources decreases pH, which directly affects water quality as well as the distribution of benthos. Similarly, when it is more basic, it results in the accumulation of unionized ammonia ions ($NH_3$), which are very toxic to aquatic animals (Constable *et al*., 2003).

The less impacted site (S1) showed a lower temperature than other study sites. This can probably due to their location upstream at Fendika, a higher altitude area having good riparian vegetation cover and the lowest TSS with a cooling effect on the water. This environment is a preferable for some organisms like Stoneflies and Mayflies (Amanuel Aklilu, 2011). The rising of water temperature may be the possible reason for the absence of stoneflies in the present study. Harp & Sweeney (2007) justified, when the water temperature rises out of the ideal range (10-15 $^{o}$c) by climate change can negatively impacted the development, behavior, and population of this taxa. The temperature value recorded in the current study is higher than the result reported in the same river (Yirga Kebede *et al.*, 2016), which was (13.6 $^{o}$C), the values range from 13.23 to 21.57 $^{o}$C in streams and rivers around Sebeta (Amare Mezgebu *et al*., 2019), and 19.3 to 20.6 $^{o}$C in the streams of the upper Awash River (Habtamu Tadesse, 2022). This variation may happen because of climatic differences due to the altitude of the area where the rivers were located, the riparian vegetation cover (Aschalew Lakew, 2014; Amare Mezgebu, 2017).

A comparison between the five sites showed that the water from Telifa had significantly high levels of TSS, this can lead increase the water temprature. The aquatic organism's natural migrations and movements (Hickin, 1995) and their physiology and life cycle (Stewart, 2005) could also be hampered by the suspended solids.

With respect to conductivity, it was within the maximum permissible limit set by WHO (2006), which recommends EC levels of < 400 µS/cm as good for aquatic life. The conductivity of the Gilgel Abay River at the S1 sampling site was higher than others. Since there is a lower discharge results in reduced dilution effects, the concentration of a strong pair of negative (-) and positive (+) ions increases, leading to an increase in the capability of the water to conduct an electric current (Aweng-Eh *et al*., 2010). In the same river, Yirga Kebede *et al.* (2016) found 120 µS/cm of EC at the downstream site. The higher discharge cause higher dilution effect leading to lowered mean EC values (Barbera & Andreo, 2015).

There was a significant difference in concentrations of $NO_3^-$ and $NO_2^-$ ($p<0.05$) among the sampling sites, with a high concentrations at S3 (4.25 mg/l) and S5 (0.064 mg/l), but the concentrations of these two was low at S1 (1.23, 0.008 mg/l), respectively. In the same River Yirga Kebede *et al.* (2016) reported lowest concentrations of nitrate (0.038 mg/l) during the dry season, and in streams of the upper Awash River concentrations of $No_3^-$ (0.41 to 1.805) mg/l were found (Habtamu Tadesse, 2022). In this study, the increase of these nutrient concentrations might be due to the intensification of agricultural activities in the catchment, which results deterioration of the water quality and habitat destruction. Agricultural inputs used for production like fertilizer highly contribute to the increase of $NO_3^-$ and $NO_2^-$ at human-influenced sampling sites. Due to population growth, most rivers and streams are affected by human activities like washing and bathing, which may increase the nutrients level as agriculture contributes. While $NH_4^+$ and $PO_4$ did not show a significant difference ($p>0.05$), these parameters may not be a good predictor of the impacted and less degraded sites in the study river.

## 5.3. Habitat quality assessment score

A habitat assessment score would allow investigators to evaluate the quality of streams and rivers riparian habitat. The habitat quality assessment scores clearly differentiated the degradation status of sites from minimally impacted to extremely impacted sites (Figure10). The biological integrity of the river is directly affected by the availability of habitat quality. The information obtained from the habitat assessment can be used to supplement biological and physicochemical data when determining the overall health of the river. In this study, most of the HQAS obtained from field-recorded data showed significant differences among sampling sites. The highest HQAS value (88%) found at the reference sample site (S1) indicates there were fewer anthropogenic activities at this station. While the lowest HQAs were recorded at S5 due to high human factors (e.g., cattle grazing, water abstraction for irrigation, deforestation, agricultural expansion to the river bank) were evident. Human disturbanc affect habitat quality both in the riparian zone and in rivers (Mathooko & Kariuki, 2000; Mathooko, 2001). According to the classification of Kleynhans (1996), on the basis of HQAS, the Gilgel Abay river study site S1 could be placed in the category of B (a slightly modified site) where a slight change in natural habitats and biota could take place. The study site S2 was in class C (moderately modified), where a loss and change in natural habitat and biota could have occurred. S3 and S4 were categorized in class D (largely modified; highest loss of natural habitats and biota has occurred), and S5 could be placed in class E (loss of habitats, biota, and basic ecosystem functions were extensive). The overall ecological health of the study river falls under the class of D (the river was largely modified through the highest loss of natural habitat). These results were contradictory with the study results obtained from the Naro Moru, Honi, Liki, Sirimon, Mariara, and Karigu rivers (M'Erimb *et al*., 2014), which were placed in the class range from slight to moderately modified. So in conclusion, the Gilgel Abay River is more impacted as compared to these six Kenyan rivers.

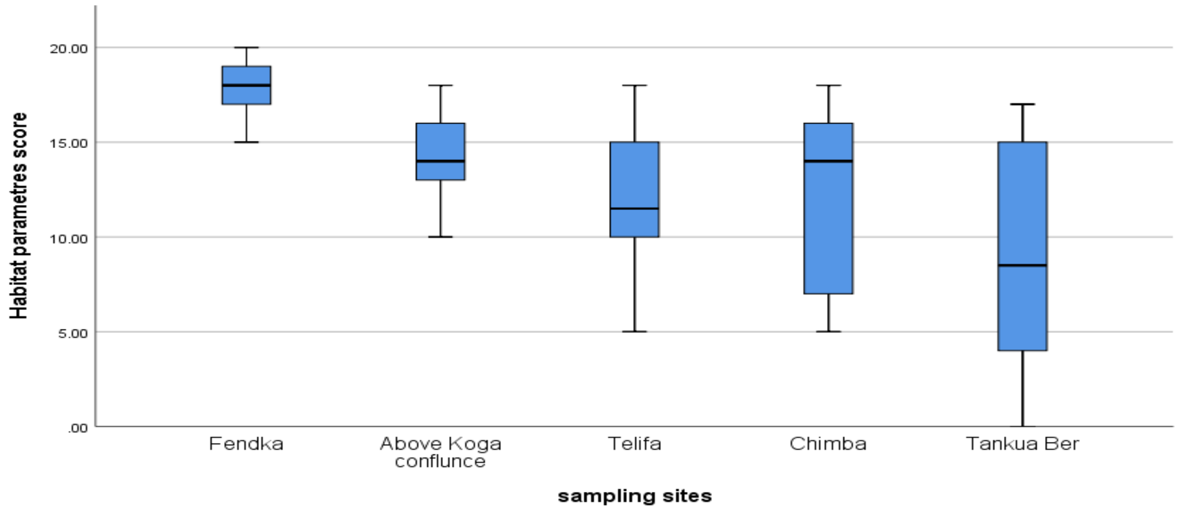



**Figure 10. Box plot illustration of HQAS metrics along different stressors**

## 5.4. Benthic macroinvertebrate community structure

The distribution of different benthic macroinvertebrates based on ecological niches implies the degree of tolerance to various anthropogenic impact, they were used as a good indicator tool for the status of river ecological health. The highest diversity, abundance and balanced trophic structure of macroinvertebrates were found at the minimum-impacted sites, but lower at the impaired sites due to various anthropogenic activities. The family Canidae, Heptageniidae, Tricorythidae, Potamanthidae, and Hydroesychidae were distributed mostly on the upstream (S1 and S2) sites. In agreement with our findings, Geda Kebede *et al.* (2020); Aschalew Lakew and Moog (2015) reported the abundant of these sensitive taxa in upper Awash River, streams and rivers around Sebeta (Amare Mezgebu *et al*., 2019) and Ryder *et al*. (2015) reported these families in rivers with low total suspended solids, and sediment loads. In the current study, the loss of the most sensitive taxa (Ephemeroptera, and Trichoptera) was also observed at impacted sites (S3, S4 and S5). The same thing was also reported at Kebena River (Worku Legesse *et al*., 2004; Baye Sitotaw, 2006), upper and middle Awash River (Tesfaye Muluye *et al*., 2024), and in Opa stream (Adedapo *et al.*, 2023). This may be because of various pollutants enrichment from the upper catchment through runoff, and habitat alteration. ET metrics have been shown to respond negatively to impairment of water and habitat quality, such as nutrient pollution, flow reduction, and sedimentation (Matthaei *et al*., 2010; Piggott *et al*., 2015; Ryder *et al*., 2015). Therefore, these organisms can be an indicator of good water quality. However, in this study, highest number of Baetidae was collected in the impacted sites (S3). Likewise, in the past, a large number of Baetidae taxa were recorded at the impacted sites (Baye Sitotaw, 2006; Aschalew Lakew & Moog, 2015a; Amare Mezgebu *et al.*, 2019). In contrast to this study result, Tesfaye Muluye *et al.* (2024) reported the highest proportion of Caenidae, Simulidae, and Notonectidae families at the stressed sites in upper Awash that were impacted by agriculture and urban pollution.

The family Coenagrinoidae, and Chironomidae have a wide range of distribution. The family Chironomidae tolerance to various disturbances, like enrichment of organic nutrients (Helson *et al.*, 2006), heavy metals (Diggins, 2000), and toxic organic compounds (Wright *et al.*, 1996). In present study, higher number of family Chironomidae was found in highly polluted sites and subsequently low dissolved oxygen, while they were less abundant at minimum disturbance sites. In agreement to this finding (Callisto & Goulart, 2001; Getachew Beneberu *et al*., 2014; Aschalew Lakew & Moog, 2015a; Tampo *et al*., 2021) reported this taxa were rare at slightly polluted sites and abundant at heavily polluted sites, since they have a unique physiological adaptation (hemoglobin molecule) that is highly efficient to extracting enough oxygen from the environment or water to maintain their metabolic processes and survive in challenging conditions (Bolshakov & Andreeva, 2012), A larger body size which helps in ventilation of larval tubes also allows chironomids to tolerate low dissolved oxygen levels (Panis *et al.,* 1996). This indicates that most of these groups possess intermediate and highest pollution tolerance and can be an indicator of bad water quality.

In the present findings, *Polypedilum wittei* has wide range of tolerance from minimum to high impacted sites with variety of disturbances, but more abundant in highly impacted sites. In line with this findings, Getachew Beneberu *et al.* (2014) reported that this species was observed at moderately polluted sites. Conversely, Amare Mezgebu (2017) reported the predominant of these species at less disturbance sites and in small proportion at highly impacted sites in streams and rivers around Sebeta .

## 5.5. Functional feeding groups

The intolerant specialized feeders, such as scrapers are more dominant at the less impacted site (S1) and are thought to be well represented in healthy streams (Cummins & Klug, 1979). The overall functional feeding group (FFG) abundance of collector gatherers in the study area was in agreement with other related study (Solomon Akalu, 2018), and predators were distributed all over across sampling sites, but collector-gatherer was the dominant group (36.81%), followed by predator (32.26%). The generalists, such as collectors-gatherer was more abundant at 54. This could be due to the fact that the good supply of fine particulate organic matter in the bottom sediments (Addo-Bediako, 2021; Cummins *et al*., 2022), However, filter feeders were also thought to be decreased significantly in low-gradient streams as the amount of sediment increased (Wallace *et al*., 1977), since deposited sediment clogging their feeding apparatus (Rabení *et al*., 2005). Likewise, in this study, they are not present at S3 and S5, which have a high sediment load and a low dissolved oxygen concentration. Less competition and the availability of prey, or food, may be the causes of the high abundance and taxonomic richness of predators throughout the river's longitudinal gradient (Ono *et al*., 2020). A high percentage of collector-gatherers indicate degradation of river habitat (Hilsenhoff, 1988; Barbour *et al*., 1999), the same thing is happing in Gilgel Abay River. Extremely high or low percentages of one or two FFGs reflect an imbalance trophic structure, possibly due to anthropogenic perturbation (Barbour *et al*., 1999). Various anthropogenic activities performed on the riverbanks, results the uneven distribution of FFGs. Therefore, metrics could be the best indicators of river health when comparing less and highly impacted sites.

## 5.6. Metrics that differentiate sites of different water quality classes

All the calculated metrics may not have equal relevance and hence the selection of metrics that enable the differentiation of sites based on various ecological conditions is essential. Therefore the following macroinvertebrates metrics were used to discriminate between the sites of the Gilgel Abay River in varying degrees of disturbance due to anthropogenic activities.

### 5.6.1. Taxa (Family)

The highest number of species can be supported by the less agriculturally impacted site (S1), but the agriculturally impacted site (S3) has a lower number of taxa occurring. Similarly, Tesfaye Muluye *et al*. (2024) reported a higher number of taxa at the reference sites (14–26) as compared to the impaired sites (6–18), and a decreased number of taxa were found at sites with depleted dissolved oxygen (Solomon Akalu *et al*., 2011; Amare Mezgebu, 2017). The number of taxa at the family level distinguishes streams impacted by anthropogenic activities. Therefore this metrics is to be a suitable metric for river ecological quality assessment tool which used to differentiate minimally impacted and highly impacted sites.

### 5.6.2. Shannon – Wiener diversity index (H)

The Shannon diversity metrics show spatial differences between sites, and the S1 (reference) site had the highest value of diversity indices and the lowest at the S3 impacted sites ( Table 10). Similarly, Tesfaye Muluye *et al*. (2024) in the Awash River, and Amare Mezgebu (2017) in Sebeta streams and rivers reported the highest diversity index at reference or less impacted sites as compared to the impacted sites. High taxonomic richness indicates a diverse habitat that supports a broader range of ecosystem functions (Elmqvist *et al*., 2003); these are characteristics of a healthy ecosystem. In this study, the reference site has a better diversity index than the impacted sites. Therefore, this index can be a better tool to differentiate between impacted and less disturbance sites in the Gilgel Abay River.

### 5.6.3. ETHbios

The upstream sites (S1 and S2) had the highest biotic score, which belonged to the good water quality class, indicating slight ecological degradation. However, the lowest biotic score was recorded at the middle and river-mouth sites of the Gilgel Abay. These study sites demonstrated moderate water quality, which implies significant ecological disturbance. Pollutants from agricultural farms most likely affect the impaired sites. Similar to the current study, Tesfaye Muluye *et al.* (2024) reported that the lowest ETHbios score found at the river mouth of the upper Awash River demonstrated severe ecological impairment, and Solomon (2018) classify the Greater Akaki and Little Akaki rivers as belonging to low water quality classes, indicating major ecological degradation. However, there was a difference in the levels of anthropogenic activities performed at sampling sites of the study river, resulting in various levels of degradation confirmed by those scoring systems. Therefore, this scoring system can be used to determine the level of river degradation from severely to minimum impacted sites.

# 6. CONCLUSION

The results of this study indicate that the low and high-flow regimes of the Gilgel Abay River have decreased over the last 42 years. The highest monthly flow variability was observed in May, with a high coefficient of dispersion (COD). Only four large floods were found during the last 42 years, while no large flood was observed after 1995. The incidence of extremely low flow has increased since 2002, and small floods were not observed after 2008. It could have negative ecological responses (reduce the abundance, composition, and diversity) to aquatic fauna and flora. The FDC in this river has steeper slope during dry season, while gentle slope during wet season over the periods of recorded. The amount of discharge range from 18.47 to 15.81 $m^3/s$ during low flow and 57.67 $m^3/s$ during high flow were the minimum environmental flow requirements to sustain the river ecosystem functioning.

Most of the environmental variables measured, except phosphate and ammonium, showed significant differences between sampling sites. Telifa and Tankua Ber sampling sites had the greatest degree of human disturbance, while minimal disturbance was recorded at the reference site (S1). Fendika (S1) sampling site has the highest (88%) HQA score, but Tankua Ber (S5) has the lowest (33%) HQA score. The study river's upstream site is slightly modified with minimal human influence, while the downstream sites experienced loss of natural habitat, biota and basic ecosystem functions due to large modifications. Higher numbers and diversity of taxa were recorded at minimally impacted sites; however, lower numbers were recorded at anthropogenically impacted sites. The stress-sensitive taxa were more abundant at minimally impacted sites in the river and can indicate good water quality. In contrast, the predominance of intolerant taxa can indicate highly impacted sites in the river. *Polypedilum wittei* and *Chironomus formosipennis* were dominant in higher $NO_3^-$ levels. The feeding functional groups of benthic macroinvertebrates in the study river reflect imbalanced trophic structures and indicate a degraded river habitat. According to ETHbios and ASPT scores, S1 and S2 belonged to good water quality classes, indicating slight ecological degradation, while S3, S4, and S5 had moderate water quality classes, revealing significant ecological disturbances.

# 7. RECOMMENDATION

If any governmental or non-governmental organization was interested in constructing a new project on this river, 18.47 to 15.81 $m^3/s$ amount of discharge during low flow and 57.67 $m^3/s$ during low flow periods should be leaft in the river to maintain the river ecosystem functions and human wellbeing and livelhoods.

The socio-economic loss of the local community due to the degraded ecosystem has not been assessed in this study, so these issues should be addressed or integrated into future studies.

Monitoring of water resource usage and the implementation of comprehensive, appropriate conservation strategies to conserve buffer zones of the river bank and the upper watershed would be necessary to maintain ecosystem services and prevent the degradation of the aquatic ecosystems in the river-wetland-lake interconnections.

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# Appendixes

**Appendix1:** Criteria for selection of reference sites for ecological river health assessment adapted for Ethiopian highland streams (Aschalew Lakew, 2017)

| Group | Criteria | Reference |
|---|---|---|
| River morphology | 1. Natural channel structure typical to the region (riffle, run, pool )<br>2. No migration barrier | Hughes (1995)<br>Barbour et al. (1996)<br>Hering et al. (2003) |
| Habitat composition | 3.Representative diversity of substrate composition appropriate to the region<br>4. Spawning habitats for the natural fish population | Hughes (1995)<br>Barbour et al. (1996)<br>Moog and Sharma (2005) |
| Hydrological Condition | 5.No alteration of the natural hydrograph and discharge regime<br>6. No water extraction, damming or diversion upstream | Barbour et al. (1996)<br>Hering et al. (2003)<br>Nijboer et al. (2004) |
| Point source Pollution | 7. No point source pollution and eutrophication*<br>8. Natural color and odor*<br>9. Total phosphorus concentration < 0.035 mg/l<br>10. Oxygen concentration > 8mg/l and Saturation range between 95 – 110%<br>11. BOD5< 2mg/l<br>12. No known cattle watering points upstream*<br>13. Minimal washing and bathing activity | Hering et al. (2003)<br>Nijboer et al. (2004)<br>Moog and Sharma (2005)<br>Moog and Sharma (2005)<br>Moog (1988)<br>Aschalew Lakew, 2017<br>Aschalew Lakew, 2017 |
| Non-point source of Pollution | 14. Absence of known or expected diffusion input<br>15. Crop farming < 20% of the catchment area<br>16. Natural land use (forest, bushes, grassland) > 60% coverage<br>17. Rural resident < 0.5% in the catchment area<br>18. No sand or gravel excavation upstream* | Hering et al. (2003)<br>Nijboer et al. (2004)<br>Hering et al. (2003)<br>Aschalew Lakew, 2017<br>Aschalew Lakew, 2017<br>Nijboer et al. (2004) |
| Riparian vegetation and wildlife | 19. Having adjacent natural vegetation appropriate to the type and geographic location of the stream<br>20. Presence of wild animals that are representative of the region and drive some support from aquatic ecosystems | Barbour et al. (1996)<br>Barbour et al. (1996 |

BOD5 Five days biological oxygen demand, * Compulsory criteria

**Appendix2:** Habitat assessment field data sheet for high gradient streams used to obtain habitat scores for each sampling site

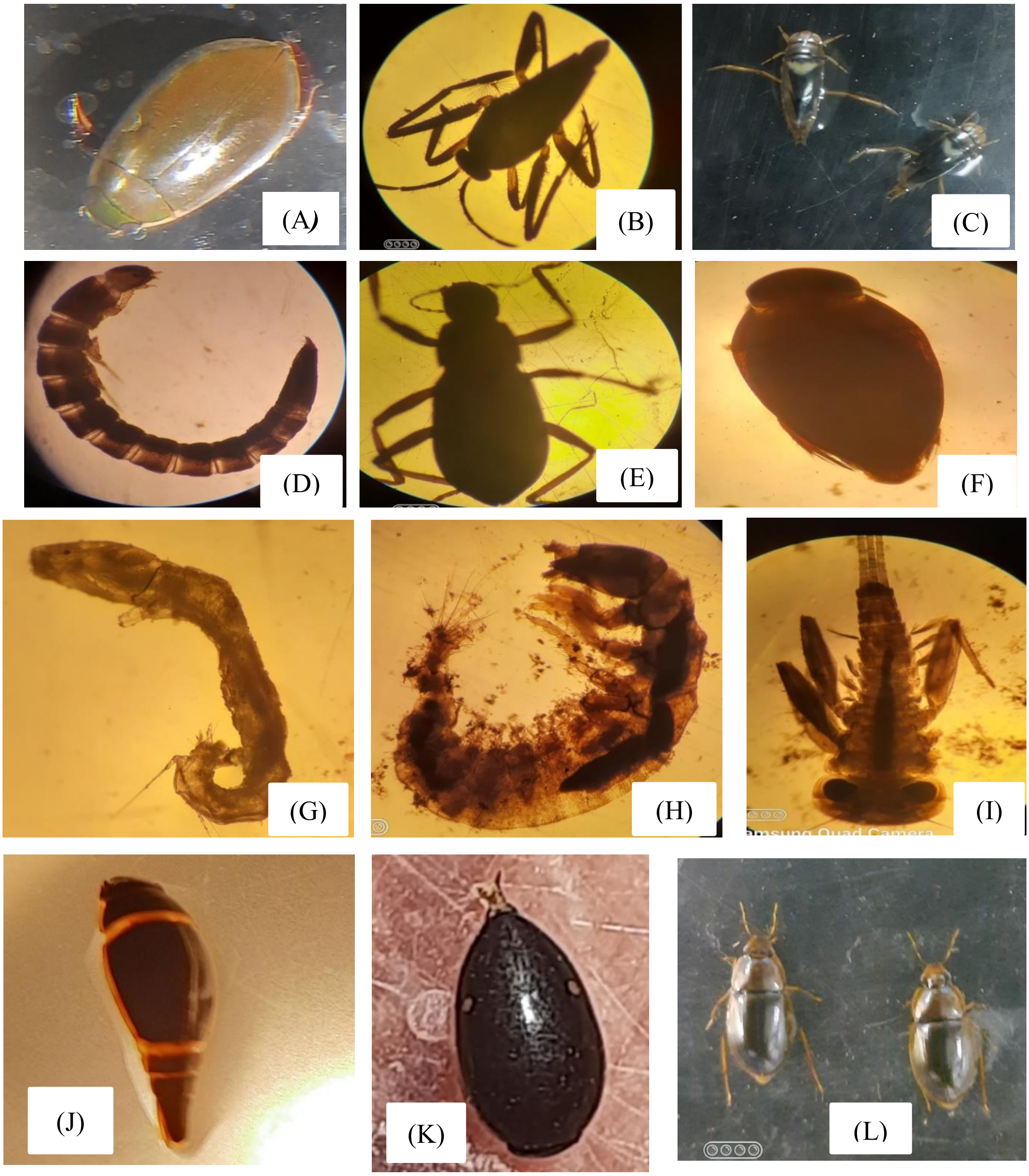
| Habitat Parameter | Condition Category | | | |
|---|---|---|---|---|
| | **Optimal** | **Suboptimal** | **Marginal** | **Poor** |
| **1. Epifaunal Substrate/ Available Cover** | Greater than 70% of substrate favorable for epifaunal colonization and fish cover; mix of snags, submerged logs, undercut banks, cobble or other stable habitat and at stage to allow full colonization potential (i.e., logs/snags that are not new fall and not transient). | 40-70% mix of stable habitat; well-suited for full colonization potential; adequate habitat for maintenance of populations; presence of additional substrate in the form of newfall, but not yet prepared for colonization (may rate at high end of scale). | 20-40% mix of stable habitat; habitat availability less than desirable; substrate frequently disturbed or removed. | Less than 20% stable habitat; lack of habitat is obvious; substrate unstable or lacking. |
| **SCORE** | 20 19 18 17 16 | 15 14 13 12 11 | 10 9 8 7 6 | 5 4 3 2 1 0 |
| **2. Embeddedness** | Gravel, cobble, and boulder particles are 0- 25% surrounded by fine sediment. Layering of cobble provides diversity of niche space. | Gravel, cobble, and boulder particles are 25- 50% surrounded by fine sediment. | Gravel, cobble, and boulder particles are 50- 75% surrounded by fine sediment. | Gravel, cobble, and boulder particles are more than 75% surrounded by fine sediment. |
| **SCORE** | 20 19 18 17 16 | 15 14 13 12 11 | 10 9 8 7 6 | 5 4 3 2 1 0 |
| **3. Velocity/Depth Regime** | All four velocity/depth regimes present (slowdeep, slow-shallow, fastdeep, fast-shallow).(Slow is < 0.3 m/s, deep is > 0.5 m.) | Only 3 of the 4 regimes present (if fast-shallow is missing, score lower than if missing other regimes). | Only 2 of the 4 habitat regimes present (if fastshallow or slow-shallow are missing, score low). | Dominated by 1 velocity/ depth regime (usually slow-deep). |
| **SCORE** | 20 19 18 17 16 | 15 14 13 12 11 | 10 9 8 7 6 | 5 4 3 2 1 0 |
| **4. Sediment Deposition** | Little or no enlargement of islands or point bars and less than 5% of the bottom affected by sediment deposition. | Some new increase in bar formation, mostly from gravel, sand or fine sediment; 5-30% of the bottom affected; slight deposition in pools. | Moderate deposition of new gravel, sand or fine sediment on old and new bars; 30-50% of the bottom affected; sediment deposits at obstructions, constrictions, and bends; moderate deposition of pools prevalent. | Heavy deposits of fine material, increased bar development; more than 50% of the bottom changing frequently; pools almost absent due to substantial sediment deposition. |
| **SCORE** | 20 19 18 17 16 | 15 14 13 12 11 | 10 9 8 7 6 | 5 4 3 2 1 0 |
| **5. Channel Flow Status** | Water reaches base of both lower banks, and minimal amount of channel substrate is exposed. | Water fills >75% of the available channel; or <25% of channel substrate is exposed. | Water fills 25-75% of the available channel, and/or riffle substrates are mostly exposed. | Very little water in channel and mostly present as standing pools. |
| **SCORE** | 20 19 18 17 16 | 15 14 13 12 11 | 10 9 8 7 6 | 5 4 3 2 1 0 |
| **6. Channel Alteration** | Channelization or dredging absent or minimal; stream with normal pattern. | Some channelization present, usually in areas of bridge abutments; evidence of past channelization, i.e., dredging, (greater than past 20 yr) may be present, but recent channelization is not present. | Channelization may be extensive; embankments or shoring structures present on both banks; and 40 to 80% of stream reach channelized and disrupted. | Banks shored with gabion or cement; over 80% of the stream reach channelized and disrupted. Instream habitat greatly altered or removed entirely. |
| **SCORE** | 20 19 18 17 16 | 15 14 13 12 11 | 10 9 8 7 6 | 5 4 3 2 1 0 |

| Habitat Parameter | Condition Category | | | |
|---|---|---|---|---|
| | **Optimal** | **Suboptimal** | **Marginal** | **Poor** |
| **7. Frequency of Riffles (or bends)** | Occurrence of riffles relatively frequent; ratio of distance between riffles divided by width of the stream <7:1 (generally 5 to 7); variety of habitat is key. In streams where riffles are continuous, placement of boulders or other large, natural obstruction is important. | Occurrence of riffles infrequent; distance between riffles divided by the width of the stream is between 7 to 15. | Occasional riffle or bend; bottom contours provide some habitat; distance between riffles divided by the width of the stream is between 15 to 25. | Generally all flat water or shallow riffles; poor habitat; distance between riffles divided by the width of the stream is a ratio of >25. |
| **SCORE** | 20 19 18 17 16 | 15 14 13 12 11 | 10 9 8 7 6 | 5 4 3 2 1 0 |
| **8. Bank Stability (score each bank) Note: determine left or right side by facing downstream.** | Banks stable; evidence of erosion or bank failure absent or minimal; little potential for future problems. <5% of bank affected. | Moderately stable; infrequent, small areas of erosion mostly healed over. 5-30% of bank in reach has areas of erosion. | Moderately unstable; 30-60% of bank in reach has areas of erosion; high erosion potential during floods. | Unstable; many eroded areas; "raw" areas frequent along straight sections and bends; obvious bank sloughing; 60-100% of bank has erosional scars. |
| **SCORE ___ (LB)** | Left Bank 10 9 | 8 7 6 | 5 4 3 | 2 1 0 |
| **SCORE ___ (RB)** | Right Bank 10 9 | 8 7 6 | 5 4 3 | 2 1 0 |
| **9. Vegetative Protection (score each bank)** | More than 90% of the streambank surfaces and immediate riparian zone covered by native vegetation, including trees, understory shrubs, or nonwoody macrophytes; vegetative disruption through grazing or mowing minimal or not evident; almost all plants allowed to grow naturally. | 70-90% of the streambank surfaces covered by native vegetation, but one class of plants is not wellrepresented; disruption evident but not affecting full plant growth potential to any great extent; more than one-half of the potential plant stubble height remaining. | 50-70% of the streambank surfaces covered by vegetation; disruption obvious; patches of bare soil or closely cropped vegetation common; less than onehalf of the potential plant stubble height remaining. | Less than 50% of the streambank surfaces covered by vegetation; disruption of streambank vegetation is very high; vegetation has been removed to 5 centimeters or less in average stubble height. |
| **SCORE ___ (LB)** | Left Bank 10 9 | 8 7 6 | 5 4 3 | 2 1 0 |
| **SCORE ___ (RB)** | Right Bank 10 9 | 8 7 6 | 5 4 3 | 2 1 0 |
| **10. Riparian Vegetative Zone Width (score each bank riparian zone)** | Width of riparian zone >18 meters; human activities (i.e., parking lots, roadbeds, clear-cuts, lawns, or crops) have not impacted zone. | Width of riparian zone 12-18 meters; human activities have impacted zone only minimally. | Width of riparian zone 6- 12 meters; human activities have impacted zone a great deal. | Width of riparian zone <6 meters: little or no riparian vegetation due to human activities. |
| **SCORE ___ (LB)** | Left Bank 10 9 | 8 7 6 | 5 4 3 | 2 1 0 |
| **SCORE ___ (RB)** | Right Bank 10 9 | 8 7 6 | 5 4 3 | 2 1 0 |
| | | **Total Score ________** | | |

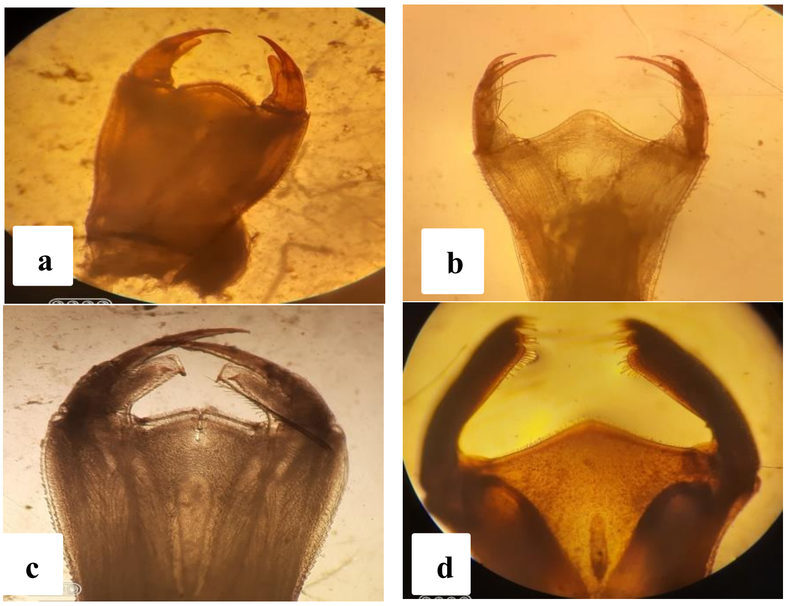
**Appendix 3**. *Suggested ETHbios threshold values (Aschalew Lakew and Moog, 2015a*)

| River quality class | Color | ETHbios score | ASPT-ETHbios | Interpretation |
|---|---|---|---|---|
| 1 | Blue | >115 | >6.5 | High water quality; low level of degradation |
| 2 | Green | 65-114 | 5.01-6.4 | Good water quality; slight ecological degradation |
| 3 | Yellow | 45-64 | 4-5 | Moderate water quality; significant ecological disturbance |
| 4 | Orange | 12-44 | 2.4-3.99 | Poor water quality; major degradation |
| 5 | Red | <12 | <2.4 | Bad water quality; heavily degraded |

Appendix4. *Some of the benthic macroinvertebrates discovered from Gilgel Abay River*




A) Dytiscidae, B) Veliidae, C) Notonectidae, D) Elmidae- Larva, E) Elmidae – adults, F) unidentified, G) Chironomidae, H) Hydropsychidae, I) Heptageniidae, J) Gyrinidae larvae, K) Gyrinidae adult, L) Hydrophilidae, Nepidae




A) Gomphidae, B) Coenagrionidae, C) Aeshnidae, D) Libellulidae