Ecological Status of Gumara-Maksegnit River as reflected by Macroinvertebrate Assemblage

By:

Alebachew Mulat

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)

Advisors: Seyoum Mengistou (Professor)

Ayenew Gize (PhD)

June, 2024

Addis Ababa, Ethiopia

DECLARATIONS

I hereby declare that the work presented in this thesis entitled " **Ecological Status of Gumara-Maksegnit River as reflected by Macroinvertebrate Assemblage** " is my original work and has been carried out at Addis Ababa University as part of the fulfillment of the requirements for the Master of Science in Aquatic Ecosystems and Environmental Management. This thesis has not been presented to any other university for the award of a degree. Unless otherwise stated or acknowledged, all the content presented in this thesis is solely my own.

Name **Signature**

Alebachew Mulat (Candidate Graduate) ____________________

DEDICATION

I dedicate this work to my beloved families and to all Aquatic Ecosystems and Environmental Management instructors and students.

ACKNOWLEDGMENTS

I would like to express my sincere gratitude to the Almighty God and St. Mary for granting me life and the strength to overcome the challenges I have faced.

First and foremost, I would like to extend my deepest appreciation to my advisors, Professor Seyoum Mengistou and Dr. Ayenew Gize, for their unwavering support, expert guidance, and invaluable feedback throughout my research journey. Their encouragement and advice have played a crucial role in shaping this thesis.

Furthermore, I am thankful to Prof. Tadesse Fetahi for his willingness to assist me in all aspects of my study, and to all my instructors for their guidance during the learning process.

I would also like to express my heartfelt gratitude to Maseresha Birara, Melkamu Aginchew, and Muluken Melkamu for their invaluable contributions during the field and laboratory work. Their dedication and expertise have been essential to the success of my research.

Special thanks are also due to the Australian Development Cooperation (ADC) for funding this research study and enabling me to pursue my master's program. Their support has helped me realize my dreams of academic achievement.

I am deeply grateful to Addis Ababa University, Bahir Dar University, the Ethiopian Institute of Agricultural Research (EIAR), and Egerton University for providing the necessary resources, facilities, and intellectually stimulating environment that have made this journey possible.

I would like to express my appreciation to my classmates and friends for their camaraderie and moral support during challenging times. Our constructive discussions and collaborative spirit have been instrumental in completing this work.

I am also thankful to Bahir Dar University and Bahir Dar Fisheries and Other Aquatic Life Research Center (BFALRC) for their cooperation in providing logistical support and provision of necessary laboratory facilities.

Lastly, I am indebted to my family for their constant support and encouragement. To my parents, your unwavering belief in me has been my driving force.

Thank you all for your contributions and support.

TABLE OF CONTENTS

LIST OF TABLES

List of figures

List of Plates

List of Annexes

Acronyms and Abbreviations

CPOM	Coarse Particulate Organic Matter
DO	Dissolved Oxygen
EC	Electrical Conductivity
EPA	Environmental Protection Authority of Ethiopia
FFG	Functional feeding group
FPOM	Fine Particulate Organic Matter
GPS	Global Positioning System
NMSA	National Meteorological Service Agency
NTU	Nephelometric Turbidity Unit
RDA	Redundancy Analysis
SASS	South Africa Scoring System
SPSS	Statistical Package for Social Sciences
TDS	Total Dissolved Solids
USEPA	The United States Environmental Protection Agency
WHO	World Health Organization

ABSTRACT

Like other tropical rivers, Gumara-Maksegnit River in the Lake Tana sub-basin faces considerable challenges due to agricultural runoff, pollutants, and agro-pastoral activities. Despite these influences, there remains a lack of information regarding the relationship between aquatic macroinvertebrate distribution and environmental parameters within the Gumara Maksegnit watershed. This study, conducted on May 2023 during the pre-rainy season, aimed to address this gap by assessing the influence of water and habitat quality on the distribution and feeding traits of benthic macroinvertebrates across eight sampling sites. Water quality parameters such as temperature, pH, dissolved oxygen, total dissolved solid, conductivity was taken in-situ measurements while ammonia, nitrate, nitrite and phosphate analyzed in the laboratory. Benthic macroinvertebrates were collected using a D-frame kick net and habitat quality was evaluated by using Gitonga method. Statistical analyses, including Pearson's correlation, ANOVA, and Canonical Correspondence Analysis were employed. The ranges for physicochemical parameters were 23.13 - 27.77 °C, 8.4- 9.10 ,5.99-8.23 mg L^{-1}, 289- 472.33 µS cm^{-1}, and 0.033 -0.62 $mgL^{-}1$ for temperature, pH, DO, conductivity and ammonia respectively. The river water quality parameters were determined to be within acceptable standards for aquatic life. A total of 831 macroinvertebrate individuals from 7 orders and 19 families were identified and counted in this study. The most dominant taxonomic group was Coleoptera, accounting for 58.47% of the total, followed by Mollusca (9.65%). Due to scarcity of stony substrates the presence of EPT taxa was relatively low in the study sites. In terms of abundance of functional feeding groups Predators (Coleoptera, Hemiptera and Odonata) were the most abundant trophic group at most of the sites, with a proportion of 84.71 %, followed by Scrapers (13.83%) and Collector-gatherers (1.32%) while Collector-filters had the lowest representation across all sites with 0.72 %. Gumara Makesegnit River was considered as a moderately polluted as shown by macroinvertebrate communities' composition and environmental variables. Further research across different seasons and locations along the river is recommended to comprehensively assess its water quality status.

Key words: Biomonitoring, Functional Feeding Groups, Gumara, Macroinvertebrates, Water Quality

ረቂቅ

እንደሌሎች ሞቃታማ ወንዞች ሁሉ በጣና ሀይቅ ንዑስ ተፋሰስ የሚገኘው የጉማራ - ማክሰኝት ወንዝ በእርሻ ፍሳሽ ፣በቆሻሻ እና በግብርና አርብቶ አደር እንቅስቃሴ ምክንያት ትልቅ ፈተና ይገጥመዋል። እነዚህ ተፅዕኖዎች እንዳሉ ሆኖ በጉማራ ማክሰኝት ተፋሰስ ውስጥ በውሃ የጀርባ አጥንት የሌላቸው ነፋሳት ስርጭት እና የአካባቢ መለኪያዎች መካከል ያለውን ግንኙነት በተመለከተ የመረጃ እጥረት አለ። ይህ ጥናት በግንቦት 2023 በቅድመ ዝናባማ ወቅት የተካሄደ ሲሆን ይህንን ክፍተት ለመቅረፍ ያለመ የውሃ እና የመኖሪያ አካባቢ ጥራት በስምንት የናሙና ጣቢያዎች በሚገኙ የቤንቲክ ማክሮ vertebrates ስርጭት እና የአመጋገብ ባህሪያት ላይ ያለውን ተፅእኖ በመገምገም ነው። ። እንደ ሙቀት፣ ፒኤች፣ የተሟሟ ኦክሲጅን፣ አጠቃላይ የተሟሟ ጠጣር እና ኮንዳክሽን ያሉ የውሃ ጥራት መለኪያዎች በቦታው ላይ መለኪያዎች ሲወሰዱ አሞኒያ፣ ናይትሬት፣ ናይትሬት እና ፎስፌት በቤተ ሙከራ ውስጥ ተተነተኑ። በውሃ የጀርባ አጥንት የሌላቸው ነፋሳት የተሰበሰቡት በዲ-ፍሬም ኪክ መረብ በመጠቀም ሲሆን የመኖሪያ ጥራት ደግሞ የጊቶንጋ ዘዴን በመጠቀም ተገምግሟል። የፒርሰን ትስስር፣ አኖቫ እና ቀኖናዊ የመልእክት ልውውጥ ትንታኔን ጨምሮ ስታቲስቲካዊ ትንታኔዎች ጥቅም ላይ ውለዋል። የፊዚዮኬሚካላዊ መለኪያዎች ክልሎች 23.13 - 27.77 °C, 8.4-9.10,5.99-8.23 mg L-1, 289-472.33 µS ሴሜ-1, እና 0.033 -0.62 mgL-1 ለሙቀት, ፒኤች, የተሟሟ ኦክሲጅን, ማዕበል አስተላላፊነት እና አሞኒያ በቅደም ተከተል. የወንዙ የውሃ ጥራት መለኪያዎች በውሃ ውስጥ ህይወት ተቀባይነት ባለው መስፈርት ውስጥ ተወስነዋል. በዚህ ጥናት ውስጥ ከ 7 ክፍለ-መደብ እና 19 ቤተሰቦች የተውጣጡ 831 በውሃ የጀርባ አጥንት የሌላቸው ነፍሳት ዝርያ ተለይተው ተቆጥረዋል. በጣም የበላይ የሆነው የታክሶኖሚክ ቡድን ኮሌፕቴራ ሲሆን ከጠቅላላው 58.47% ይሸፍናል, ከዚያም ሞለስካ (9.65%). በድንጋያማ ንጣፎች እጥረት ምክንያት የኢፒቲ ታክሳ መኖር በጥናት ቦታዎች በአንጻራዊ ሁኔታ ዝቅተኛ ነበር። ከተግባራዊ የአመጋገብ ቡድኖች ብዛት አንፃር አዳኞች (ኮሊዮፕተራ, ሄሚፕቴራ, እና ኦዶናታ) በአብዛኛዎቹ ጣቢያዎች በጣም የተትረፈረፈ የትሮፊክ ቡድን ነበሩ, በ 84.71 %, ከዚያም አተንፎች (13.83%) እና አከማችቶች (1.32). %) ኣጣሪ አከማችቶች በሁሉም ጣቢያዎች ዝቅተኛው ውክልና 0.72 % ነበራቸው። የጉማራ - ማክሰኝት ወንዝ በውሃ የጀርባ አጥንት የሌላቸው ነፋሳት ማህበረሰቦች' ስብጥር እና የአካባቢ ተለዋዋጮች እንደሚታየው በመጠኑ የተበከለ ተደርጎ ተወስዷል። በወንዙ ዳር በተለያዩ ወቅቶች እና ቦታዎች ላይ ተጨማሪ ምርምር የውሃ ጥራት ሁኔታን በጥልቀት ለመገምገም ይመከራል።

ቁልፍ ቃላት: *ባዮሞኒተሪንግ፣ ተግባራዊ የምግብ ቡድኖች፣ ጉማራ፣ በውሃ የጀርባ አጥንት የሌላቸው ነፍሳት፣ ውሃ ጥራት*

1. INTRODUCTION

1.1. Background of the study

Freshwater ecosystems provide indispensable contributions to human existence on Earth (Aylward *et al*., 2005). They support the livelihood of billions of people worldwide. Freshwaters represent surface waters, such as rivers, wetlands, lakes, reservoirs, and ponds compounded with groundwater. Freshwater accounts for 2.5 % of aquatic ecosystems on Earth. Surface water comprised only the remaining 0.26%. Rivers are significant ecosystems with great ecological value (Nguyen *et al*., 2018), and their health is critical for the human communities that rely on them (Dickens *et al*., 2018). Rivers are a significant source of renewable water supply for humans and freshwater ecosystems (Vorosmarty *et al*., 2010) and provide a variety of products and services, including residential usage, navigation, recreational activities, nursing grounds, and food for many creatures (Berger *et al.,* 2016). Despite their huge significance to life on Earth, surface waters are the most degraded natural ecosystems (Carpenter *et al*., 2011), and human activities pose a growing threat to them. Streams and rivers are the most vulnerable freshwater ecosystems to anthropogenic stressors (Allan, 2004; Best and Darby, 2020).

Freshwater ecosystems have experienced significant negative effects due to human actions, including sedimentation, mining, removal of riparian vegetation, water extraction, dam construction, and grazing (Matthews, 2016; Sabater *et al*., 2018). Pollution linked to rapid population growth and climate change has worsened the degradation of freshwater environments and the loss of biodiversity (Woodward *et al*., 2010; Peters *et al*., 2013; Wen *et al.*, 2017). These human-induced threats negatively affect the biodiversity and the physical and chemical components of the river ecosystem. This interaction impacts the ecological processes within the rivers and leads to the loss of river ecosystem services. Deforestation of the riparian area resulted in canopy cover removal. This, in turn, increases the exposure of the riparian region to intensive solar radiation, soil erosion, and siltation (Mbaka,2010).

Biomonitoring is the most commonly used tool to monitor the integrity of freshwater ecosystems. Biomonitoring can provide a movie picture of an ecosystem condition, while physicochemical parameters provide snapshot information. Several biotic entities have been used to analyze the ecological integrity of river ecosystems. Macroinvertebrate assemblages were the most frequently used biotic entities. Geographical variation, human activity, seasonal variation, water quality, geological formation, and physical habitat conditions affect the diversity, composition, and abundance of macroinvertebrate communities in rivers (Wohl *et al*., 2006). As the macroinvertebrate composition of an intact river ecosystem has been studied, variation in assemblage can be explained by natural factors such as longitudinal differences. The main differences in macroinvertebrate composition are also ascribed to the distance from the source, longitude, and stream width (Helen and Jenny, 2007). This suggests that without anthropogenic interference, the composition of macroinvertebrate assemblages could vary with natural factors, such as the hydrological regime, geology formation, and drainage morphometry of the watershed. Drainage morphometry helps to understand the hydrological and morphological characteristics of the watershed of a river ecosystem (Asfaw & Workineh (2019); Mahala, 2020). River morphology, such as riffles and pools, perennial and intermittent rivers, narrow and wider rivers, and shallow and deep rivers, affect the composition of characteristic macroinvertebrate communities (Argaw Ambelu, 2010). Some of these natural variations can also contribute to variations in habitat conditions and water chemistry, which are strong predictors of the composition of macroinvertebrate communities in river ecosystems (Maul *et al*., 2004). Therefore, investigating the influence of natural variations, physical habitat conditions, and anthropogenic disturbances on the abundance, structure, and composition of macroinvertebrates could provide plausible information for informed management decisions. In addition, macroinvertebrates link the lower and higher trophic levels in the aquatic ecosystem food chain. They also demonstrated a wide range of tolerances for human perturbations.

Macroinvertebrate functional feeding groups serve as valuable proxies for ecosystem attributes and provide insights into the overall environmental conditions. This approach relies on assessing the relative abundance of different invertebrate functional groups to gauge the state of the ecosystem. For instance, the importance of autotrophy and heterotrophy in changing the aquatic food chain in rivers can be determined by analyzing the relative importance of these functional groups (Cummins *et al*., 2005; Ramírez and Gutiérrez-Fonseca, 2014). Directly measuring ecosystem attributes can be challenging and timeconsuming because it requires integrating measurements over a season and accounting for spatial heterogeneity. Additionally, the application of this approach to tropical rivers and streams is hindered by limited information on the functional composition of macroinvertebrate communities (Boyero *et al*., 2009). Importantly, most rivers and streams in Ethiopia are understudied and there is limited knowledge of ecological health to enable proper management and provide a systematic overall picture of the status of these riverine environments. Hence, assessing the effects of different types of stressors on the water quality of streams and rivers using various bioassessment methods is of utmost importance for scientists, river managers and policy makers to determine an appropriate river use strategy. There is lack of empirical studies regard to river health assessment in Gumara-Makesegnit Watershed. Therefore, understanding the functional composition of invertebrates in tropical streams and rivers is crucial for understanding processes, such as organic matter processing, energy flow, trophic relationships, and implementing management strategies to minimize disruptions to ecosystem functioning (Boyero *et al.,* 2011a; Fereira *et al.,* 2012). Moreover, addressing the relationship of human disturbance, habitat condition, and natural variation along the altitudinal gradients influence macroinvertebrate assemblage and their feeding traits could have significant contribution in river ecosystem management and conservation. Thus, the results of this study will provide scientifically plausible information that can use as an input in designing policy compounded with river ecosystem health monitoring and assessment tools.

1.2. Statement of the problem

Understanding the responses of stream macroinvertebrate communities to multiple changes in human activities in natural environments is crucial due to the presence of various stressors in the ecosystem. The structure of benthic macroinvertebrates and their role in stream ecosystems are influenced by riparian vegetation, which provides organic matter and canopy cover in streams (Masese *et al.,* 2014b). The quality and integrity of biotic stream ecosystems are affected by various factors such as the expansion of agricultural and urban areas (Allan, 2004), overgrazing (Kibichi *et al*., 2008; Leip *et al.,* 2015), and the introduction of exotic species (Masese *et al.,* 2017).

Human-made changes to the riparian corridor impact the functional feeding groups of macroinvertebrates by altering food availability and causing shifts in habitat structure and quality (Dudgeon, 2006; Wantzen & Wagner, 2006). Since assessing the effects of different types of stressors on the water quality of streams and rivers using various bioassessment methods is of utmost importance for scientists, river managers and policy makers to determine an appropriate river use strategy. However, Little studies on benthic macroinvertebrates have been carried out in some streams and rivers across Ethiopia, demonstrating their effectiveness in assessing water quality and ecological health (Tesfaye Berhe, 1988; Getachew Beneberu and Seyoum Mengistou, 2010; Argaw Ambelu *et al.,*2010; Solomon Akalu *et al.,* 2011; Aschalew Lakew, 2012; Getachew Beneberu, 2013; Aschalew Lakew and Moog, 2015). In addition, Gizachew Teshome *et al*. (2024) studied Variation in macroinvertebrate assemblages and water quality as a test for different levels of ecological impairment across an Ethiopian highland river system in case of Megech river which one of perennial river that drain to Lake Tana same to Gumara Maksegnit River. Yet there has been no studies regard to river health assessment in Gumara-Makesegnit watershed which flows from upstream to downstream via various natural habitats and human perturbations. Therefore, addressing the relationship of human disturbance, habitat condition, and natural variation along the altitudinal gradients influence macroinvertebrate assemblage and their feeding traits could have significant contribution in river ecosystem management and conservation around Lake Tana sub-basin, Ethiopia.

1.3. Objectives

1.3.1. General objective

The purpose of this study was to assess ecological status of Gumara river using macroinvertebrate assemblage.

1.3.2. Specific objectives

- Identify effect of habitat quality on macroinvertebrate assemblages
- To determine the relationship of physicochemical parameters with macroinvertebrate assemblage
- To see the relationship between the functional feeding groups (FFG) and other parameters in the river

1.4. Significance of the study

For the river to be managed properly and for its biological value, it is crucial to increase our understanding of the ecological factors that determine whether the river is maintaining the desired aquatic species in the area. The present study would generate baseline scientific information on the status of Gumara Maksegnit river based on the response of macroinvertebrates metrics. Moreover, it is feasible to create bioassessment instruments to determine the health of river ecosystems. The criteria used to measure a river's water quality would also improve the river's capacity to support a rich and thriving biological community. Because of this, other organizations engaged in related fields of research can benefit from the study's results about macroinvertebrates. Most importantly, the knowledge gained from this study would improve our understanding of how to develop methods that are suitable for evaluating the river's water quality. A relevant scientific basis would then be established with these data for developing sustainable management, conservation practices, and a reliable biomonitoring program. The results of this study could also contribute to a better understanding of the spatial distribution of freshwater macroinvertebrates in poorly explored Ethiopian regions.

1.5. Conceptual framework

Human activities, variation of altitude, river morphology, water quality, drainage morphometric parameters and habitat condition affects the diversity and composition of macroinvertebrate assemblages. Drainage morphometry helps to understand the hydrological and morphological characteristics of a watershed (Daniel Asfaw & Getachew Workineh 2019; Mahala 2020). River morphology such as riffles and pools, perennial and intermittent rivers, narrow and wider rivers, shallow and deep rivers have their own characteristic macroinvertebrate community composition (Argaw Ambelu ,2010). Macroinvertebrates have a wide range to tolerance for various variables, and they link the higher and lower trophic levels. There diversity and composition are an important indicator of state of biodiversity in river ecosystems. The ability of benthic macroinvertebrates to react to a variety of ecological perturbations makes them suitable indicators to track stream conditions in this regard (Resh *et al.,* 1988). Dive deeper to underhand the details of an ecosystem are necessary before attempting to respond in the form of restoration. Macroinvertebrate assemblage and their functional feeding groups considered as representative groups of organisms that can show the state of biodiversity in the aquatic ecosystem. They make up the majority of the ecology in a natural highland stream and contain both generalists and specialists. The higher diversity and composition of macroinvertebrate assemblage, the healthier the river ecosystem. This implies in turn the healthier the watershed is the better the biodiversity in a river ecosystem. Figure 1 below shows the framework of the interplays of different variables in the river ecosystems.

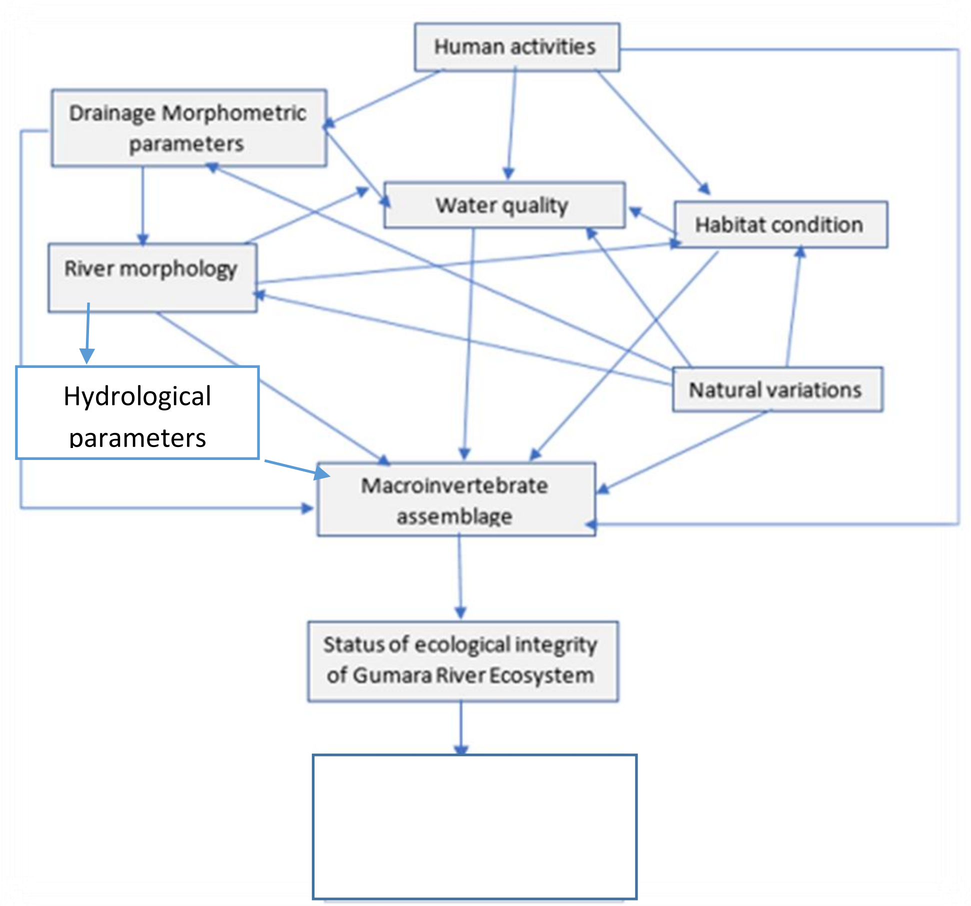

Figure 1:Research conceptual framework

1.6. Scope of the study

The study was focused on analyses of selected physicochemical parameters (temperature, DO, pH, conductivity, TDS, ammonia, nitrite, nitrate and phosphorus), habitat quality, their effect on macroinvertebrates assemblages and assess the ecological status of Gumara - Maksegnit River.

1.7. Limitation of the study

- The sampling was carried out during only one season impacting the observed presence and diversity of macroinvertebrates and necessitating seasonal studies to comprehend fluctuations and temporal variations.
- Habitat quality scoring was performed by a single person.

- Unseasonal rainfall before sampling led to water turbidity and flooding, potentially resulting in the washing away of macroinvertebrates.
- Multivariate analysis was not extensively analyzed due to the limited data sets.
- The use of temperate taxa feeding traits for the identification of macroinvertebrates' functional feeding groups was due to the unavailability of tropical ecosystems feeding traits database.
- The absence of historical data or baseline conditions in the Gumara River made it challenging to select sampling sites and compare ecological changes or deterioration.

2. LITERATURE REVIEW

2.1. Water Quality in Rivers

Water quality is the chemical, physical, biological, and radiological properties of water. These physical, chemical, and biological characteristics of water change with the season, the watershed's natural environment, the pattern of land use, and, to a considerable extent, human activity (Haque, 2018). Table 2.1 the most frequently used water quality parameters that fall under several categories. The condition of water in relation to the needs of one or more biotic species and to any human need or purpose is measured by the water quality. It is most usually used in reference to a set of standards that may be used to measure compliance, which is typically attained through water treatment. Drinking water, protecting the environment, industry, agriculture, recreation, and aquatic ecosystem habitat are just a few of the many uses for high-quality water. But one of the main problems people are dealing with is the rapidly declining quality of water in many places. Both anthropogenic and natural factors, such as excessive human exploitation of water resources in urban settlements, industrial areas, and agricultural operations, are the primary causes of the deteriorations (Bhateria & Abdullah, 2015; Selemani *et al*.,2018). As a result, the study will examine physical, chemical, and biological characteristics in relation to the quality of the water. This offers information about the Gumara River basin's surface water quality.

Table 1: Water quality parameters (Source: Haque, (2018))

Classification	Water quality parameters
General, physical and chemical	Temperature, Dissolved Oxygen (DO), pH, Conductivity, Alkalinity, Suspended Solids
Nutrients	NH4-N2 NO2-N, NO3-N, PO4 3-
Biological	Macrobenthos

2.2. Water quality and anthropogenic impacts

According to Dudgeon *et al*. (2006), freshwater biodiversity offers a wide range of valued products and services to human society, some of which are irreplaceable. Yet, human activities have always had an impact on aquatic ecosystems. Because of anthropogenic impacts, rivers are extremely susceptible to alteration, and their flow is frequently altered to supply water for human use (Bredenhand & Samways, 2009). Globally, due to human activity, freshwater ecosystem biodiversity is quickly declining (Dahl *et al*., 2004). Five main kinds of threats to freshwater biodiversity have been identified by Dudgeon *et al*. (2006): overexploitation, water pollution, flow modification, habitat destruction or degradation, and invasion by foreign species. The aquatic biodiversity found in these systems may be further jeopardized by human demand on water resources in the next decades (Strayer, 2006). In many places, notably in the African region, there is already a serious issue with overusing rivers and aquifers for irrigation. This activity may result in physical and chemical changes, drought, and the elimination of inland aquatic habitats (Abellán *et al*., 2006; Belmar *et al*., 2010). (Velasco *et al*., 2006). Pollution caused by various pollutants, such as fertilizers, sewage, heavy metals, or pesticides, is a major issue everywhere in the globe.

River water quality is impacted by several non-point sources of contamination created by growing cities and industrialization (Beasley & Kneale, 2003). The detrimental effects of various contaminants on aquatic biota, which cause biodiversity loss and poor water quality, have been the subject of several research (Beasley & Kneale, 2003, 2004; Benetti & Garrido, 2010; Fernández-Díaz et al., 2008; Garrido *et al*., 1998; Harper & Peckarsky, 2005; Hirst *et al*., 2002; Lytle & Peckarsky, 2001; Smolders et al., 2003; Song *et al*., 2009). One of the most significant changes to rivers is the development of dams (Belmar *et al*., 2010). The overall result is a change from dynamic patterns to static, comparatively steady ones with lower fluxes (Baeza *et al*., 2003; Benejam *et al*., 2010; Stanford & Ward, 1979). As a result of the loss of some species' microhabitats and the formation of new ones, changes in flow velocity and marginal vegetation may result in changes in the composition of aquatic assemblages (Fulan et al., 2010; Lessard & Hayes, 2003; Sarr, 2011). Another negative effect of humans on freshwaters is the widespread introduction and invasion of exotic species.

Usually, due to competition, predation, and biotic homogenization, this results in the extinction of native species (Raehl, 2002).

2.3. Macroinvertebrates Composition, Abundance and Diversity

To determine how the diversity of macroinvertebrates in nearby streams changes as a result of relative human activities like urbanization, it is possible to assess the diversity of these organisms and compare its patterns to the information (Roy *et al*., 2001). Since the majority of these species are reliable indicator species, seeing them can be very helpful in determining the stream quality. There are numerous factors that make the majority of macroinvertebrates effective indicator species. A pollutant entering the stream or some other harmful disturbance will undoubtedly have an impact on them because they are crucial components of the food web of a stream ecosystem. The most frequently, they are too little to escape these human disturbances. Accordingly, if they are an intermediately tolerant species, they will live, but they will be so exhausted from trying to survive that they will finally become extinct. They will prosper in the absence of competition, though, if they are more accommodating. Also, they are rather simple to sample and identify (Herrmann, 1991; Karr & Chu, 1997).

For instance, if a bio-survey revealed that stoneflies were absent from areas where they had existed, one may speculate that the stream's dissolved oxygen content has dropped (Roy *et al*., 2001). Yet, this raises the main drawback of this kind of biological evaluation. While studying these organisms allows us to pretty confidently determine whether a stream is healthy or unhealthy, it does not tell us why certain species are present or absent. In other words, it doesn't explain why the oxygen levels have dropped, and it's also possible that there's something else entirely affecting the population of stoneflies. Roy *et al*. (2001) revealed that as urbanisation increased, the variety of macroinvertebrates declined in a study spanning 30 streams in the piedmont of Northern Georgia. Their main findings were that macroinvertebrate diversity is a good predictor of stream health and the effects of development, and that restricting development in a stream catchment is essential to maintaining the integrity of the stream.

Changes in the composition and abundance of the macroinvertebrate community can result from a variety of factors. For example, species able to adapt to unstable habitats, such as chironomids and oligochaetes, are favoured by high frequency and prolonged floods as well as changed sediment transport dynamics. Invertebrates are especially vulnerable to drift because too much sedimentation reduces refugial space (Sarkar *et al.,*2002). In order to confirm these findings, this study will focus on how macroinvertebrate nutrition levels and water quality respond.

2.4. Introduction to bio monitoring /bio assessment

Biological assessment is the assessment of the biological status of an ecosystem based on studies of the structural and functional organization of the community of resident biota (Karr *et al.,* 1986; Barbour *et al*., 1999; USEPA, 2013). It is now acknowledged as one of the most beneficial tools for water resource managers. This is mostly as a result of its contribution to understanding the link between species and their environments (Gobiet *et al*., 2013). Beyond chemical changes, human influences on water resources also include hydro-morphological changes, riparian loss, substrate degradation, and climatic change, which frequently interact with one another. Since aquatic life provides an integrated perspective on the consequences of human actions, it has long been recognized as the most direct and effective indicator of the integrity of water resources (Karr, 2006). However, it should be considered an advantageous component to physical and chemical monitoring rather than an alternative. The response of organisms that is affected by their natural cycle renders strength of the bio monitoring.

2.5. Aquatic Invertebrates as Bioindicators

Aquatic macroinvertebrates are the most frequently used taxa for monitoring habitat quality (Hellawell, 1986), with a well-developed body of information for lotic systems (Braukmann, 2001; Lotufu, 2001). Most research involves IBI is or similar multi-metric rapid assessments used to compare quantitative indices based on numbers and types of species for disturbed versus reference sites, and monitoring programs employing these techniques are underway in the USA and United Kingdom (Resh *et al*. 1995; Reynoldson *et al*., 1997; Karr & Chu, 1997; USEPA 2002a, b). IBI have mainly been developed for specific habitats (i.e., small streams) and pollutants; more research is needed regarding other aquatic habitats (i.e., wetlands) and non-point source pollutants (Rosenburg & Resh 1996).

Macroinvertebrates are often used as biomonitoring tools (Dallas & Mosepele, 2007). Biomonitoring is based on the principle that organisms are the ultimate indicators of the health of the environment they are within (USEPA, 2002). Biomonitoring offers the advantage of detecting the cumulative physical, chemical, and biological effects of detrimental actions on an aquatic system. Aquatic macroinvertebrates are frequently used for biomonitoring due to the following three factors: First, they are not very mobile, therefore they are indicative of the place from whence they are collected, secondly, they have relatively short life cycles and therefore can reflect environmental changes quickly through changes in their community composition and finally they respond to pollutants in both water column and sediments (Reece & Richardson, 2000).

2.6. Macroinvertebrates Functional Feeding Groups

Functional feeding groups (FFGs) are a classification system that categorizes macroinvertebrates based on their morphological mechanisms and behavioral characteristics for acquiring food, rather than their taxonomic group (Cummins, 2016). This approach is used to assess environmental conditions and variables, and it has significant implications for ecosystem functioning (Uwadiae, 2010). The composition of functional feeding groups within macroinvertebrate communities can vary, and this variation has important consequences for the overall functioning of ecosystems. Unfortunately, due to land use changes and human activities, many regions have experienced a loss of diversity and major shifts in the structure and function of macroinvertebrates in rivers (Allan *et al.,* 2015). However, by studying the functional feeding groups of macroinvertebrates, we can gain valuable insights into the status of the environment.

This approach relies on the relative abundance of different functional groups of invertebrates as indicators of ecosystem conditions. For instance, the balance between autotrophy and heterotrophy in the aquatic food chain can be used to assess the health of rivers (Ramírez & Gutiérrez-Fonseca, 2014). Understanding the functional composition of invertebrates in tropical streams and rivers is crucial for comprehending organic matter processing, energy flow, trophic relationships, and implementing management strategies to minimize the negative impact on ecosystem functioning (Boyero *et al*., 2011). The River Continuum Concept (Vannote *et al*. 1980) and other relevant literature(Brasil *et al*., 2014) support the hypothesis that there is a longitudinal zonation of macroinvertebrate functional feeding groups along the profile of a river, influenced by the distribution of energy inputs and matter transfers, riparian conditions and the availability of leaf litter play significant roles in the distribution and abundance of macroinvertebrates, especially the shredders, and the ratios of the various FFGs can be used as indicators of ecosystem attributes to assess the ecological condition of river.

3. MATERIAL AND METHODS

3.1. Study area description

The Gumara-Maksegnit watershed is located in the Lake Tana sub- basin, on the northeast side of the lake, crossing Gondar Zuriya and East Dembya Woreda. It covers an area of 37,051 hectares and extends between 12°17'.06'' to 12°30'.53'' Latitude and 37°25'.07'' to 37°41'.54'' Longitude. The altitude in the area ranges from 1785 meters at Lake Tana (outlet) to 2848 meters a.s.l. at the headwaters in the western escarpment of Gondar Zurya Woreda (Figure 2). This river originates from the high plateau in the east, which is known for its mountainous, rugged, and dissected topography, and then flows into the plain in the west, characterized by flat to gentle slopes. The lower part of this river is a high-risk area for flooding in East Dembya Woreda, primarily due to water overflowing from the river banks and backflow into Lake Tana during heavy rainy seasons. The Gumara-Maksegnit watershed experiences a range of agro-climatic zones, from low-altitude sub-tropical (Woyna Dega) at elevations of 1785-2300 m.a.s.l to mid-altitude temperate (Dega) at elevations of 2300-2848 m.a.s.l. Data from the meteorological station at Maksegnit, obtained from NMSA Bahir Dar branch, shows that the average annual rainfall in the watershed ranges from 792.1mm to 1129.4mm. The high rainfall season in the watershed occurs from June to September, with short rainy seasons in the autumn. The Gumara River is a perennial river that flows into Lake Tana. The Lake Tana watershed benefits greatly from the ecological and socioeconomic contributions of the Gumara River. Passing through two agroecological zones, the river encompasses the Dega and Woyna Dega ecosystems. Human activities have posed a threat to the mountain ecosystem located at the eastern escarpment of Gondar Zuria woreda and its surrounding areas.

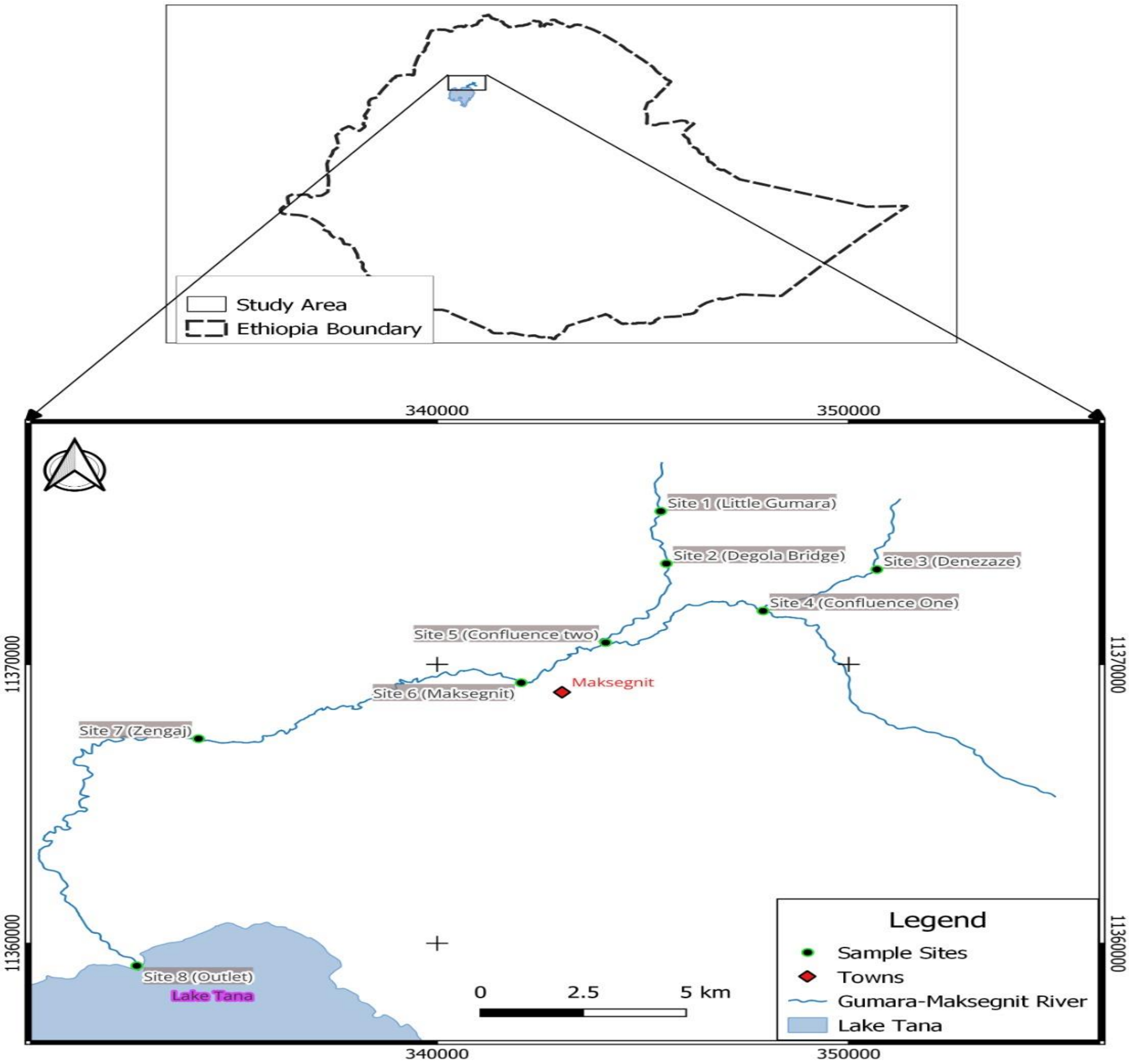

Figure 2: Map of Gumara Makesegnit River

3.2 Sampling Design

Before field sampling, a reconnaissance survey was carried out to designate the suitable sampling station. The selection of sampling locations along the river for collecting water samples and macroinvertebrates was purposeful and based on their intended use, accessibility, physical proximity, and riparian land uses. Each sampling station was marked with a Geographical Positioning System (GPS) to ensure consistent sampling from the same location at every sampling period and divided into the four biotopes of riffle, pool, run, and marginal vegetation. A multi-habitat macroinvertebrate sampling approach was used to collect macroinvertebrates from riffle, run, pool, and marginal vegetation biotopes at each station. Habitat quality parameters were evaluated through visual assessment. A 100-meter stretch upstream of the river was the sampling unit for macroinvertebrates and habitat quality assessment at each station.

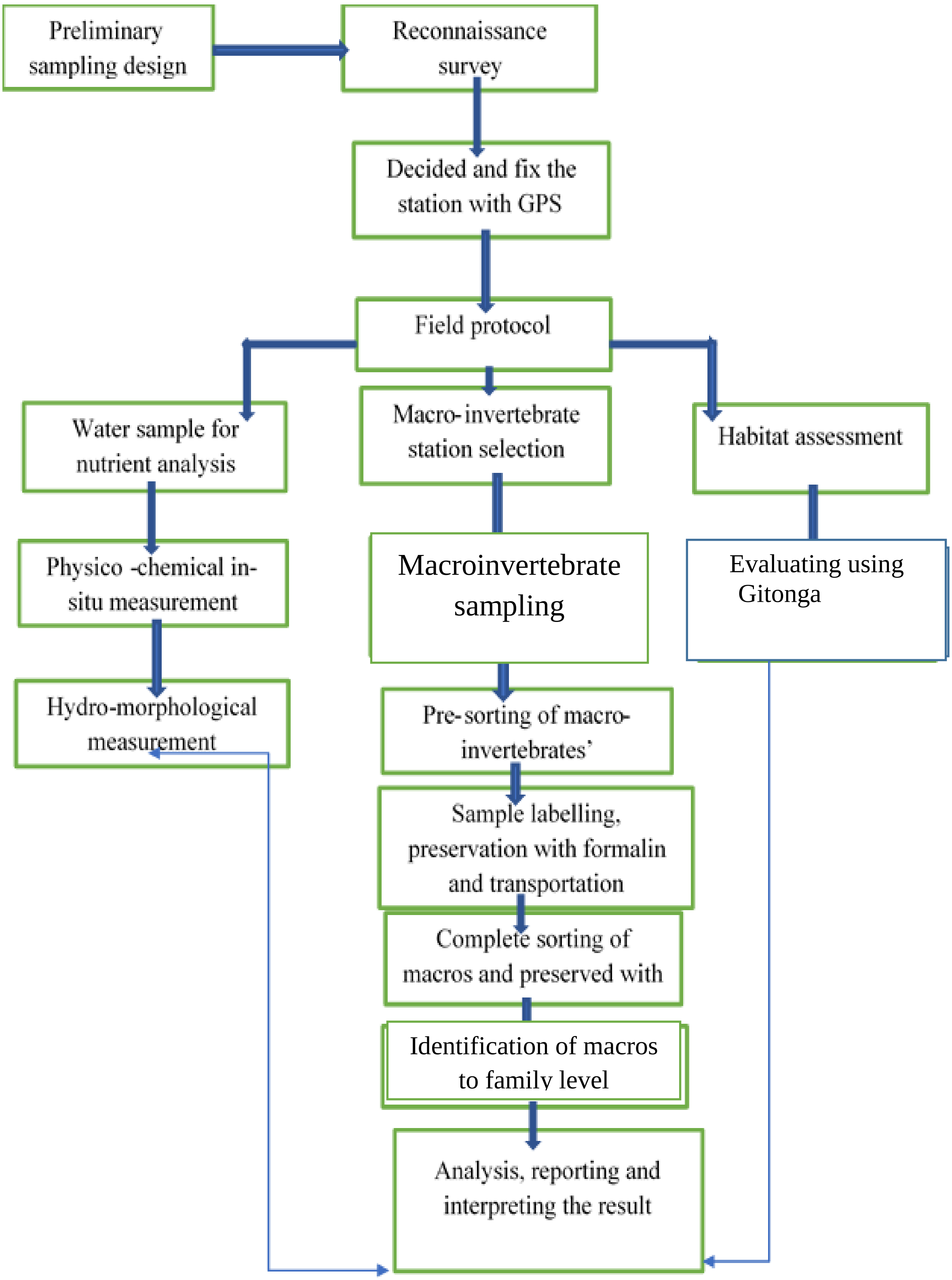

Figure 3:Sampling design and procedure modified from (Veronica, 2010)

3.2.1. Sampling sites

The study area's sampling sites were chosen purposefully, taking into account different habitat conditions, proximity, land uses, riparian land use, and human activity. Gumara River serves as a water source for institutions, homes, urban centres, animal watering, and washing activities. The upper stream comprises two tributaries known as Little Gumara and Jhoana. Similarly, the middle stream is located at the confluence of these two tributaries, just before the river crosses Maksegnit Town. Downstream sites are situated where the river crosses both the town and Gumara Bridge on the main road from Bahir Dar to Gondar. A total of eight sampling sites were selected for conducting physicochemical measurements, assessing habitat quality, and sampling macroinvertebrates.

Table 2: Description of Sampling sites

Sites	Latitudes N	Longitudes E	**Major features and human activities**
Site one (Little Gumara)	12°43´88.35´´	37°57´79.08´´	Mostly agricultural activities. Poor bank stability and the riparian vegetation
Site two	12°25´18.55´´	37°34´45.30´´	Bridge, crop farming and Sand mining
Site three (Denezaze)	12°42´02.06´´	37°62´63.23´´	Little anthropogenic activities, near reference site
Site four (Confulence one)	12°40´66.33´´	37°60´09.45´´	Activities such as agriculture, cattle grazing and washing clothes
Site five (confluence two)	12°39´62.00´´	37°56´58.19´´	Sand mining, vegetation clearing, water abstraction
Site six	12 ° 38´30.66´´	37 ° 54´70.12´´	wastes from town households, hospital and slaughters houses
Site seven (Zengaj)	12 °36´45.32´´	37 °47´49.86´´	Intense agricultural activities and unstable river bank
Site Eight(outlets) of the river	12 °29´08.37´´	37 °46´16.43´´	cattle grazing, crop farming and water abstraction

Sampling site 1

Sampling site 2

Bank of sampling site 3

Human activities at sampling site 4

The bank instability of sampling site 5

Bank stability of site 6

Banks of sampling site 7

Plate 1: sampling sites of the study area

3.3.2. Field Sampling

3.3.2.1. Measurement of Environmental Variables

Environmental variables were assessed in both field and in the laboratory to determine their impact on benthic macroinvertebrate communities. Three main categories of environmental factors were considered: regional factors impacting the watershed, physical characteristics within the stream, and components of water quality. Regarding regional variables influencing watershed characteristics, altitude and types of land use were taken into consideration. Field surveys involved measuring the following physical in-stream characteristics: (i)stream width which is the dimension from bank to bank at a transect representative of the stream channel, (ii) water depth which is the vertical dimension from the water surface to the stream bottom; and (iii) current velocity at riffles and gliding runs. The hydro-morphological variables of river velocity, width, and depth were measured using a tape measure and flow meter. Depth was assessed at all sampling locations across the river, while width was measured three times at each sampling station. The river's discharge was estimated using the velocity-area method described by Gordon et al. (1993), and calculated accordingly.

Q = A* V (1) Where Q = stream discharge ($m^3 s^{-1}$).) s -1 A = cross sectional area (m^2) V = average velocity ($m s^{-1}$). Cross sectional area (A) can be estimated = wetted width*average depth.

3.3.2.2. Physicochemical Parameters

Before collecting macroinvertebrates, in-situ physicochemical parameters were measured. Using a portable multi-parameter meter model 900p, water temperature, pH, Dissolved oxygen, conductivity, and total dissolved solids were measured in triplicate at sub-surface water at all sampling sites. These parameters were selected due to their widespread use and their ability to easily assess water quality in various field conditions.

3.3.2.3. Nutrients

The study utilized nitrate, phosphorus, ammonia, and nitrite due to their extensive use in aquatic ecology. Subsurface water samples were gathered from a depth of 20 to 30 cm in the river and placed in two clean plastic bottles, each with a capacity of 0.5 liters, totaling 1 liter.

To maintain sample purity, water quality samples were collected prior to obtaining macroinvertebrates. The plastic bottles underwent thorough cleaning and rinsing with deionized (distilled) water, and the field-preserved samples were treated with four drops of 10% concentrated sulfuric acid. These samples were then packed, transported in a cooling box, and stored in a refrigerator at 4°C until analysis. Measurements were carried out in the laboratory to analyze ammonia, nitrate, nitrite, and phosphorus according to the Standard Methods (APHA, 2005) at the limnology laboratories of the Bahir Dar fishery and other aquatic life research center.

3.3.1.4. Benthic Macro invertebrates

Sampling along the Gumara River took place once at every sampling location before the rainy season in May 2023. A D-frame kick net measuring 1 m^2 with a 500 μm mesh size was used to sample benthic macroinvertebrates. The net was pulled along the riverbank up to 1 m from each sampling station, against the water flow, as per Dickens & Graham, 2002. Various types of habitats were sampled in relative proportion to their coverage of the total macroinvertebrate habitat in the reach. According to the South African Scoring System (SASS) Version 5, Stone (S) biotopes, marginal vegetation, and Gravel, Sand & Mud (GSM) biotopes were quantitatively sampled at each site. Stones were disturbed using kicks for a duration of 3 minutes in the stone biotopes, while marginal vegetation was swept for a total of 2 minutes. Gravel, sand, and mud were stirred and swept for a combined 1 minute. Small stone substrate areas were disrupted for the final three minutes, covering riffle, run, pool, and marginal vegetation microhabitats, each within a 1 m2 area. Samples from different habitats were combined to create a single uniform sample. To ensure accurate information, sample bottles were labeled twice, both inside and outside, with the sampling code, time, date, and other important details.

3.3.1.5. Habitat Quality Assessment

The methodology outlined in Gitonga (2021) was used to calculate the habitat quality index. This approach involves utilizing 9 metrics to assess HQI: available instream cover, bottom substrate, dimension of largest pool, number of riffles, water level, channel sinuosity, bank stability, riparian buffer vegetation, and aesthetics of reach. Each station's total HQI was determined by summing the scores and categorized as exceptional, high, intermediate, limited, or minimal integrity index for 26 – 31, 20 – 25, 14 – 19, 8 – 13, and < 7, respectively.

Table 3: Habitat characteristics scoring criteria (Gitonga, 2021).

Habitat parameter	Scoring criteria			
Available instream cover	Abundant>50% of substrate that favors colonization and fish cover, virtuous mix of several stable cover types such as snags, wood debris, holes in the bank, macrophytes	Common30 – 50% of substrate supports firm habitat; sufficient habitat for maintenance of populations; may be limited in the number of various habitat types.	Rare 10 – 29.9% of substrate ropes firm habitat; habitat accessibility less than desirable; substrate frequently troubled or removed	Absent< 10% of substrate supports firm habitat; absence of habitat is obvious; substrate unstable or absent.
Score	4	3	2	1
Bottom substrate	Stable>50% gravel or larger substrate; gravel, sarsens; dominant substrate type is grit or larger	Moderately Stable30 – 50% gravel or larger substrate; prevailing substrate type is mix of grit with some finer sediments	Moderately Unstable10 – 29.9% grit or bigger substrate; prevailing substrate type is finer than gravel, but may still be a mix of sizes.	Unstable< 10% grit or bigger substrate; substrate is uniform sand, silt, clay, or bedrock
Score	4	3	2	1
Dimension of largest pool	Large Pool occupies more than 50% of the	Moderate Pool occupies roughly 50% or slightly less of the channel width; maximum depth is 0.5 – 1 meter	Small Pool occupies	Absent Pools are absent, only

	channel width; maximum depth is >1 meter		roughly 25% of the channel width; maximum depth is < 0.5 meter	shallow auxiliary pockets.
Score	4	3	2	1
Number of Riffles To be reckoned, riffles must extend > 50% the width of the channel and be at least as long as the channel width	Abundant >5 riffles	Common 2 – 4 riffles	Rare 1 riffle	Absent No riffles
Score	4	3	2	1
Water level	High Water reaches the base of both lower banks; <5% of channel substrate is uncovered	Moderate Water fills > 75% of the channel; or < 25% of channel substrate is uncovered.	Low Water fills 25-75% of the existing channel or riffle substrates are mostly exposed.	No flow Very little water in the channel and mostly extant in standing pools, or stream is dry Absent No riffles.
Score	4	3	2	1

Channel sinuosity	High ≥ 2 well-defined bends with deep outside areas -cut banks and shallow inside areas-point bars present	Moderate 1 well defined bend or ≥ 3 moderately defined bends existing.	Low< 3 moderately defined bends or only poorly-defined bends existing	None Straight channel ; might be Channeli zed
Score	4	3	2	1
Bank stability	Stable Little proof (<10%) of erosion or bank letdown; bank angles average <30°	Moderately Stable Some proof (10 – 29.9%) of erosion or bank letdown; small areas of erosion mostly healed over; bank angles average 30 – 39.9°	Moderately Unstable Proof of erosion or bank letdown is common (30 – 50%); high possibility of erosion during flooding; bank angles average 40 – 60	Unstable Large and frequent proof (> 50%) of erosion or bank letdown; raw areas recurrent along steep banks; bank angles average >60
Score	4	3	2	1
Riparian Buffer Vegetation	Extensive Width of natural buffer is greater than	Wide Width of natural buffer is 10.1 to 20 meters	Moderate Width of natural buffer is 5 to 10	Narrow Width of natural buffer is less

Score	20 meters		meters	than 5 meters
	4	3	2	1
Aesthetics of Reach	Wilderness	Natural Area	Common Setting	Offensive
	Exceptional natural beauty; usually wooded or unpastured area; no apparent indications of anthropogenic activity	Trees or native flora is common; some development evident - from fields, pastures, natural dwellings little proof of human activity	Not offensive; area is developed, but uncluttered such as in an urban park	Stream does not augment the aesthetics of the area; cluttered; Highly developed; may be a dumping area
Score	4	3	2	1

3.4. Laboratory Analysis

3.4.1. Nutrients

Conventional colorimetric methods were employed in the laboratory to analyze the nutrients present in the water column (APHA, 2005). Phosphate analysis was conducted on unfiltered water samples, while ammonia, nitrate, and nitrite analyses were carried out on filtered water samples.

3.4.2. Macroinvertebrates

The samples were placed in a white plastic tray and then transferred to vials in the laboratory after being washed with tap water through a 300 µm mesh size sieve. Any visible organisms found on the substrate were carefully collected with forceps and placed in specimen vials.

The Samples were composted on location and preserved in 95% ethanol for subsequent sorting and identification. The benthic macroinvertebrates in the sample were then counted to assess their relative abundance, diversity, composition, and the percentage of functional feeding groups present along the river. Identification down to the family level was carried out using a dissecting microscope and standard identification keys. The field guide "Aquatic Invertebrates of South African Rivers" by Gerber (2002) was used as a reference for the identifications.

3.4.3 Classification of macroinvertebrates into functional feeding groups

The functional feeding groups were categorized according to the criteria and classification methods described by Cummins (1975), Merritt and Cummins 1996, Rempel *et al*. (2000), Mandeville (2002), and Arimoro (2007). The analysis method of functional feeding groups links to essential aquatic food resource categories: coarse particulate organic matter (CPOM, particles > 1mm), and fine particulate organic matter (FPOM, particles < 1 mm and > 0.45 μm). The main functional feeding groups consist of:

a. Scrapers/grazers which consume algae and associated materials.

b. Shredders, which consume leaf litter or other CPOM including wood.

c. Collector-gatherers, which collect FPOM from the stream bottom

d. Collector-filterers, which collect FPOM from the water column using a variety of filterers.

e. Predators, feed on other consumers, piercer or engulfers.

3.5. Data Analysis

3.5.1. Physical-chemical Parameters, Morphological Variables and Nutrients

The data for physical-chemical parameters, morphological factors, and nutrients was analyzed using descriptive statistics to calculate mean and standard error (mean±SE). The Pearson correlation coefficient was utilized to examine the relationship between each parameter and the river's nutrients. Since the research data exhibited non-normal distribution, the Kruskal-Wallis Test in SPSS was used to evaluate variations in the means of physicochemical parameters and nutrients at each station. Upon identifying significant differences between sampling sites, pairwise differences were indicated through Kruskal-Wallis significance differences tests. Furthermore, the results for each sample station were presented using tables and bar graphs. Throughout the analysis, the significance level was set at 95%.

3.5.3. The Benthic Macroinvertebrate Assemblages

The benthic macroinvertebrates were identified to family level. Then different indices and metrics were calculated: Shannon Diversity Index (SDI) and composition measures

The benthic macroinvertebrate diversity, richness, composition and abundance were determined from each sampling station and sampling occasion.

The Shannon-Weaver diversity index (Shannon & Weaver, 1949) was calculated as follows:

H’= $-\sum PiIn(Pi)$ (11) where, H’ = the Shannon-Weaver Diversity Index Pi = the relative abundance of each group of organism’s ln = natural logarithm

The taxa richness was determined and were identified to detect their suitability for M-IBI metrics for assessment of ecological status based on the macroinvertebrates data.

- ✓ Total No. of taxa
- ✓ No. of EO taxa (Ephemeroptera and Odonata taxa)
- ✓ No. of EOC taxa (Ephemeroptera, Odonata and Coleoptera taxa)
- ✓ The No. of Ephemeroptera
- ✓ The No. of Hemiptera
- ✓ The No. of Coleoptera

3.5.4. Functional feeding group ratios used as indicators of stream ecosystem attributes

The FFG ratios serve as indicators of stream ecological attributes. Table 1, taken from Cummins et al. (2005), displays the calculated ratios along with their corresponding general criteria ratio levels. The balance between autotrophy and heterotrophy (Production/Respiration) index was determined by the ratio of Scrapers to (Shredders + total collectors [Collector-filters + Collector-gatherers]); while the linkage between riparian inputs and stream food webs (CPOM/FPOM) was calculated using the ratio of Shredders to total collectors (Collector-filters + Collector-gatherers). Since Shredder macroinvertebrates were not collected in this study, the remaining four functional feeding groups ratios were utilized as indicators of stream ecosystem attributes.

Table 4 : Functional feeding group ratios as indicators of stream ecosystem attributes (Cummins *et al*. 2005)

Ecosystem attributes	Symbols	Functional feeding group ratios for attributes	General criteria ratio levels
Autotrophy to heterotrophy index	P/R	Scrapers to Shredders + total Collectors	Autotrophic > 0.75
Predator-prey ratio	P/P	Predators to the total of all other functional groups	< 0.15 indicates a normal predator/prey ratio
FPOM in transport (suspended) to FPOM storage in sediments (deposited in benthos)	TFPOM/BFPOM	Collector-filters to Collector-gatherers	FPOM transport (in suspension) enriched unusual particulate loading) > 0.50
Substrate (Channel) stability	Channel Stability	Scrapers + Collector-filters to Shredders + Collector-gatherers	Stable substrates (e.g. cobbles, boulders, large woody debris, rooted vascular plants) plentiful > 0.50

3.5.5. The Relationship between Water Quality and Benthic Macroinvertebrates

The correlation between macroinvertebrate structural and functional composition, water quality variables, nutrients, and stream size factors were examined using the (RDA) method across the various disturbance gradients

3.5.6. The Habitat Quality Determination and their Relation with Benthic Macroinvertebrates

The habitat was classified into different management classes - excellent, good, fair, or poor - based on the final score of the habitat criteria. A Kruskal Wallis test was conducted to compare the mean rank of habitat scores across sampling sites. Redundancy analysis (RDA) was utilized to examine the interactions between macroinvertebrates and habitat quality measures. Descriptive statistics and inferential statistics such as ANOVA were performed using SPSS. PAST software and CANOCO 5 were employed for all analyses, while Sigma Plot and Microsoft Office Excel (2021) were used to generate all figures.

4. RESULTS

4.1 Water Quality in Gumara River

4.1.1 Physico-chemical Parameters and Nutrients

The table presents the data for the physico-chemical parameters of water samples collected from various locations along the river (Table 4.1). Water temperature ranged from 23.13±0.25 to 27.77±1.21 °C. The highest temperature was observed at site five, while the lowest was recorded at site Eight. Notably, the temperature increased from the upstream to the middle of the river, but then decreased after site six. These temperature variations were determined to have statistical significance ($P < 0.05$) across the different sampling stations.

The levels of dissolved oxygen (DO) varied greatly across the sampling sites, with measurements ranging from 5.99 ± 0.21 to 8.23 ± 0.26 mg/L. Site three exhibited the highest DO value, whereas the lowest was measured at site eight. Conversely, there were no notable discrepancies in pH values among the sampling stations ($P > 0.05$), with values ranging from 8.4 ± 0.15 to 9.10 ± 0.20. The highest pH value was identified at site six, while the lowest was recorded at site eight.

The range of electrical conductivity values was from 289±6.93 to 472.33±7.51 ($\mu S\ cm^{-1}$), with site six recording the highest conductivity and site eight the lowest. Significant differences in conductivity were observed between the sampling points ($P < 0.05$), similar to temperature. The total dissolved solids (TDS) varied significantly, ranging from 142.9±0.23 to 236.00±3.46 mg L^{-1}. Site six had the highest TDS value, while site eight had the lowest, with significant differences between the sampling sites ($P < 0.05$). Ammonia (NH_3) concentration in the water samples ranged from a maximum of 0.62±0.06 mg L^{-1} at site two to a minimum of 0.033±0.005 mg L^{-1} at site eight, revealing significant differences between the sampling stations ($P < 0.05$). The nitrate levels varied from 0.18±0.002 to 1.15±0.06 mg L-1, with the highest concentration recorded at sampling point three upstream of the river.
In general, there were observable fluctuations in the physicochemical characteristics of the water samples at various sampling locations along the river.

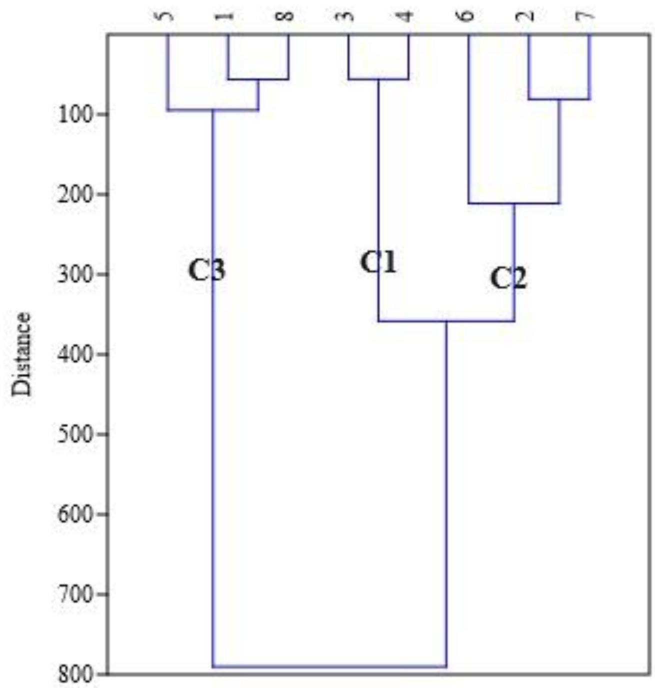

Figure 4: Clustered sites of Gumara-Maksegnit River

The sampling sites were grouped into three categories based on measured physicochemical parameters. Category C1 included sites three and four, which were near reference sites. Category C2 consisted of sites two, six, and seven, which were moderately impacted. Category C3 comprised sites one, five, and eight, which were classified as highly polluted sites.

Table SEQ Table * ARABIC 5 : Means of water quality parameters for different stations (DO = dissolved oxygen **in** mg L-1, pH = power of hydrogen **in** scale, conductivity **in** µS cm-1 and nutrients in mg L-1

	Sites								
Parameters	1	2	3	4	5	6	7	8	P.value
DO (mg L^{-1})	8.00±0.17	6.92±0.19	8.23±0.26	6.91±0.20	6.92±0.45	7.49±0.03	6.5±0.15	5.99±0.21	0.004
Temp (℃)	26.00±0.51	27.30±0.10	24.73±1.00	27.13±0.15	27.77±1.21	26.83±0.59	25.00±1.40	23.13±0.25	0.012
pH	8.70±0.20	8.57±0.15	8.63±0.15	8.63±0.31	8.70±0.20	9.10±0.20	8.93±0.25	8.4±0.15	0.058
EC (µS cm^{-1})	317.67±11.72	325.33±1.53	429.33±1.53	446.00±3.61	386.67±1.53	472.33±7.51	355.67±0.58	289±6.93	0.003
TDS (mg L^{-1})	160.67±9.37	177.57±2.32	214.67±0.58	222.00±3.61	192.60±0.00	236.00±3.46	177.73±0.23	142.9±0.23	0.002
NH_3(mg L^{-1})	0.15±0.005	0.62±0.06	0.62±0.02	0.11±0.01	0.093±0.004	0.07±0.003	0.11±0.004	0.033±0.005	0.043
PO_4(mg L^{-1})	0.53±0.021	0.42±0.04	1.09±0.05	0.83±0.05	1.02±0.06	1.58±0.54	1.18±0.08	0.54±0.04	0.046
NO_3(mg L^{-1})	0.6±0.03	0.2±0.01	1.15±0.06	0.64±0.04	0.82±0.03	0.56±0.02	0.9±0.02	0.18±0.002	0.045
NO_2(mg L^{-1})	0.02±0.01	0.004±0.001	0.022±0.013	0.3±0.01	0.021±0.003	0.4±0.02	0.34±0.037	0.003±0.0003	0.051
Turbidity (NTU)	920.4±1.45	439.3±2.2	266.5±2.72	343.5±2.6	965.1±3.4	684.9±3.2	549.7±2.68	992.4±3.6	0.038

4.1.4 Hydro-morphological Variables

The average depth of the river differed at the sampling locations, with the greatest average depth of 0.38± 0.08 m recorded at site eight and the lowest average depth of 0.07±0.012 m recorded at site two. Stations one, four, and five all had the same measurement. A significant difference in depth among the sampling stations was detected (ANOVA, $P < 0.05$). Similarly, the river's width exhibited variation among the sites, ranging from 7.33±0.58 m. Site five had the widest width, while site seven had the narrowest. The differences in width between the sampling sites were also deemed statistically significant ($P < 0.05$). Notably, the river's width increased after the confluence of the two streams. The velocity varied from 0.5±0.1 to 0.05±0.002 m/s. The site with the highest velocity was site five, where the streams converged to form the river, while the lowest velocity was observed at site eight, the outlets of the river. During the study period, the river discharge was highest (1.2±0.25 m3 /s) at site five (confluence river), and the lowest was (0.1±0.003 m3 /s) at station two (upstream of the river), as depicted in (4.1d). These measurements offer valuable insights into the physical characteristics of the river and emphasize the importance of considering these factors in environmental assessments.

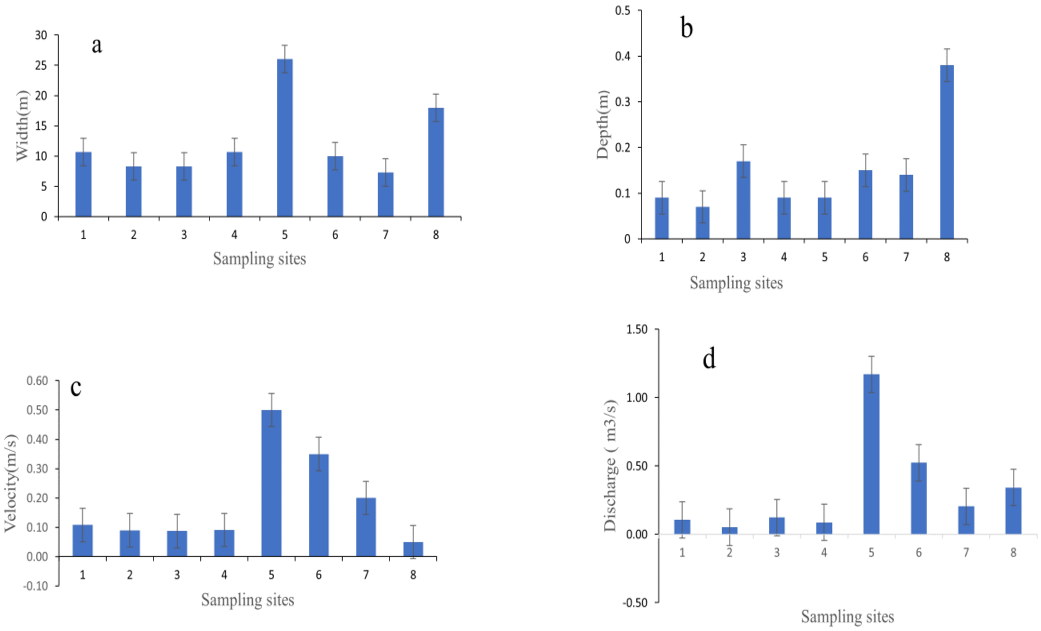

Figure 5 : Measured values of hydro-morphological parameters at sampling stations (a=width (m), b= depth (m), c = velocity (m/s), d = discharge (m3 /s).

4.2. Benthic Macroinvertebrates Assemblages

4.2.1 Macroinvertebrates Composition and Abundance

In this study, a total of 831 macroinvertebrate individuals from 6 orders, 1 class and 19 families were identified and counted (Table 4.3). Among the taxonomic groups, the most prominent one was Coleoptera, with 4 families and a total of 642 individuals, accounting for 58.47% of the total. This was followed by Mollusca with 106 individuals (9.65%), Odonata with 33 individuals (3.00%), Hemiptera with 21 individuals (1.91%), Diptera with 10 individuals (0.91%), Annelida with 3 individuals (0.27%), and Ephemeroptera with 4 individuals (0.36%).

In terms of taxa richness, site two displayed the highest diversity with 9 taxa, while site one exhibited the lowest with only 3 (Table 4.4). However, regarding individual abundance, site six recorded the highest with 261 individuals, whereas site eight had the lowest with only 7 individuals. Spatially, sampling station six, situated below the town, showed the highest relative abundance of Diptera, representing 79% of the total. Throughout the study duration, the order Coleoptera (family Dysticidae) was present in all sampling sites, followed by the Mollusca (family Planorbidae), which was found in all sites except site one.

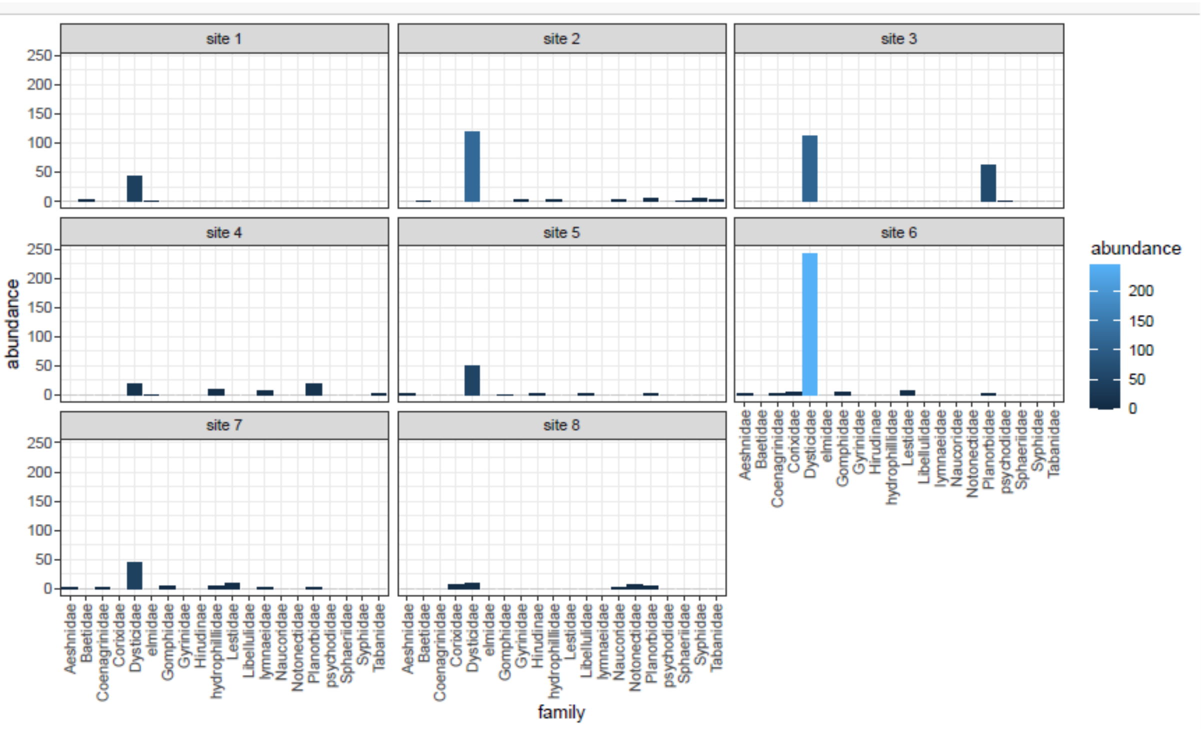

Figure 6: Abundance of macroinvertebrates along sampling site

Table 6: List of macroinvertebrates identified among sampling sites

Order	Family	sampling sites							
		1	2	3	4	5	6	7	8
Coloptera	Dytiscidae	42	119	112	18	50	243	44	8
	Elmidae	1	0	0	1	0	0	0	0
	Gyrinidae	0	2	0	0	0	0	0	0
	Hydrophilllidae	0	2	0	9	0	0	3	0
Mollusca	Planorbidae	0	6	61	18	2	2	1	5
	Sphaeridae	0	1	0	0	0	0	0	0
	Lymnaeidae	0	0	0	8	0	0	2	0
Diptera	Tabanidae	0	3	0	2	0	0	0	0
	Syrphidae	0	4	0	0	0	0	0	0
	Psychodidae	0	0	1	0	0	0	0	0
Ephemeroptera	Baetidae	3	1	0	0	0	0	0	0
Annelida	Hirudinae	0	0	0	0	3	0	0	0
Odonata	Aeshnidae	0	0	0	0	2	3	1	0
	Libellulidae	0	0	0	0	2	0	0	0
	Gomphidae	0	0	0	0	1	4	3	0
	Coenagrionidae	0	0	0	0	0	2	1	0
	Lestidae	0	0	0	0	0	6	8	0
Hemiptera	Naucoridae	0	3	0	0	0	0	0	2
	Notonectidae	0	0	0	0	0	0	0	6
	Corixidae	0	0	0	0	0	4	0	6
Total		46	141	174	56	60	264	63	27

4.3. Diversity of Macroinvertebrates

The diversity of macroinvertebrates of Gumara River is indicated in Table 4.4. The value of Shannon-Wiener diversity index in the sampling station varied from 0.3443 to 1.53. The trend of evenness along the river was also high in site 8 and low in site 6.

Table 7 Diversity and Richness Measures

Diversity indices	*site 1*	*site 2*	*site 3*	*site 4*	*site 5*	*site 6*	*site 7*	*site 8*
Taxa_S	3	9	3	6	6	7	8	5
Shannon_H	0.34	0.73	0.68	1.49	0.71	0.41	1.11	1.53
Evenness_e^H/S	0.47	0.23	0.66	0.74	0.34	0.22	0.38	0.93

Table 8: Composition measures

	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Total number of individuals	50	163	239	97	75	291	89	54
Total No. of taxa	3	9	3	6	6	7	8	5
No. of EOC Taxa	3	4	1	3	4	4	6	1
No. Ephemeroptera Taxa	1	1						
No. Hemiptera Taxa		1						3
No. Coleoptera Taxa	2	3	1	3	1	1	1	1
No. Diptera Taxa		2	1	1				
No. Odonata Taxa					3	4	4	

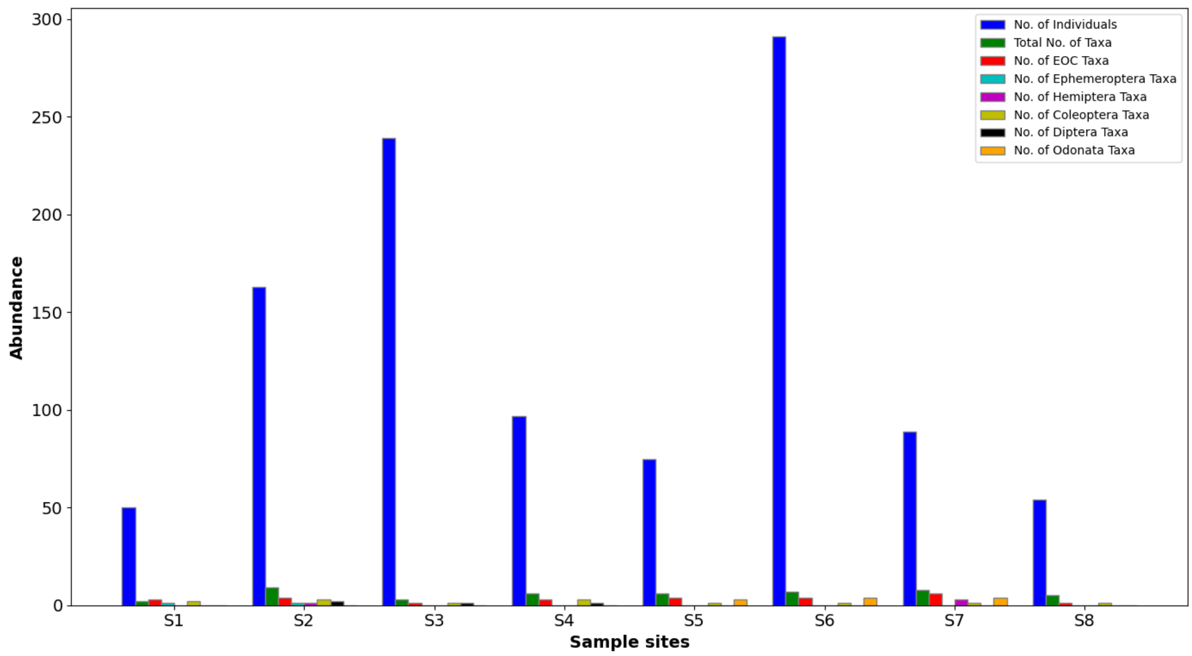

Figure 7: Composition measures of macroinvertebrates across sites

4.4. Proportion and distribution of functional feeding groups

A total of 831 specimens were collected from the following orders of aquatic macroinvertebrates: coleoptera (n= 642), Mollusca (n=106), Odonata (n=33), Hemiptera (n=21), Diptera (n=10), Annelida (n= 3), and Ephemeroptera (n= 4). Four functional feeding groups (FFGs) were recognized in this study. These include collectors-gatherers (c-g), predators (prd), collectors-filterers (c-f), scrapers (scr). The aquatic macroinvertebrates obtained from the Eight stations were listed as, Predators (n = 704), Scrapers (115) Collector-gatherers (n = 10) and Collector filters (n =2). Predators were the most common category in the entire study area with an important abundance in S2 and S6. Scrapers were the second most common group amongst the sampled sites, with a high proportion in S3, followed by Collector-gatherers and Collector filters with a comparable abundance. Meanwhile, Table 2 shows the abundance of the main functional feeding groups at their respective sites. During sampling season, predators were numerically dominant in the selected sites, accounting for 84. 71 % of the total assemblage, followed by Scrapers (13.83%) and Collector-gatherers (1.32%). Group Collector -filters were extremely low. The average relative abundance of Collector-gatherers was absent in downstream, favoring the occurrence of Predators. The relative abundance of Scrapers decreased drastically at S5 in favor of Collector-gatherers and

Collector-filters. The proportion of Collector-gatherers decreased downstream, with S2 representing the highest proportion. Predators were numerically well-represented in the upstream rivers, with less or almost equal proportions in the inundated and downstream sites

Table 9 : Taxa recorded from Gumara River with allocation to functional feeding groups

Order	Family	FFG
Coloptera	Dysticidae	Predator
	Elmidae	Collectors-Gatherers
	Gyrinidae	Predator
	Hydrophilllidae	Predator(L) /Collectors-gatherers(A)
Mollusca	Planorbidae	Scraper
	Sphaeridae	Filtering collectors
	Lymnaeidae	Scrapers
Diptera	Tabanidae	Predator
	Syrphidae	Filtering collectors
	Psychodidae	Collectors-Gatherers
Ephemeroptera	Baetidae	Collectors-Gatherers
Annelida	Hirudinae	Predator
Odonata	Aeshnidae	Predator
	Libellulidae	Predator
	Gomphidae	Predator
	Coenagrinidae	Predator
	Lestidae	Predator
Hemiptera	Naucoridae	Predator
	Notonectidae	Predator
	Corixidae	Scrapers

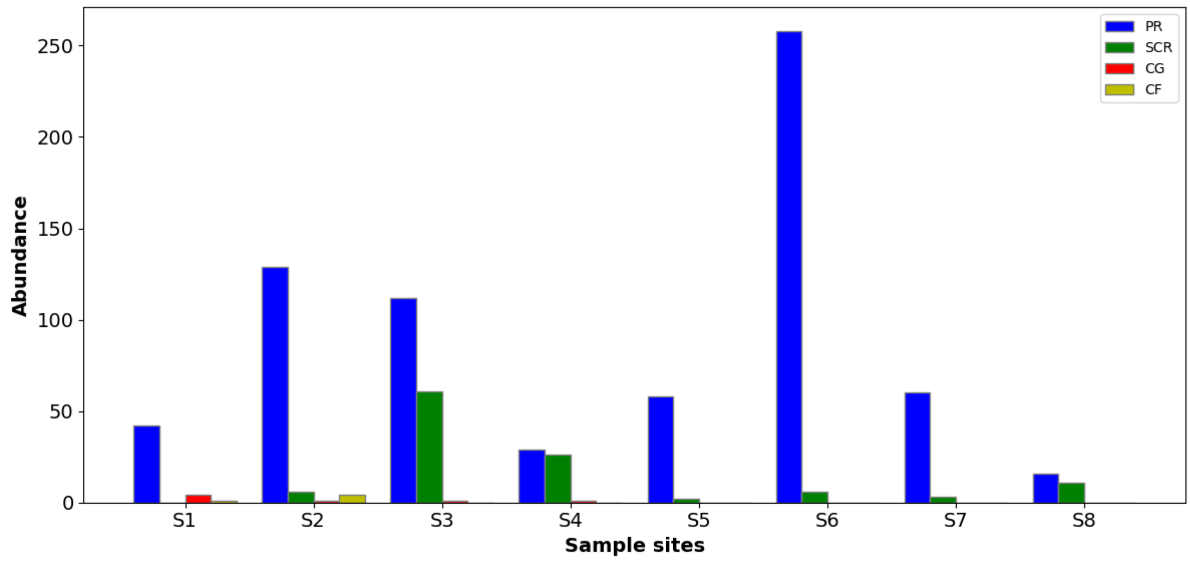

Figure 8: Abundance of functional feeding groups (FFGs) along investigated sites.

Keys: CF: Collector Filterers, CG: Collector Gatherer SCR: Scrapper, PR: Predator

Table 10: Ecosystem attributes based on the ratio of FFG (functional feeding group)

Ecosystem attributes	1	2	3	4	5	6	7	8
P/R	5	1.2	61	26	2	6	3	11
P/P	8.2	11.72	1.8	1.07	29	43	20	1.45
TFPOM/BFPOM	0.25	4	1	1	0	0	0	0
Channel Stability	0.25	10	61	26	2	6	3	11

The numerical value of the production to respiration ratio (P/R) indicated that all sampling stations were autotrophic (P/R > 0.75), demonstrating the presence of functioning riparian areas across all sites. With the exception of site one, the habitat stability index exceeded the threshold value of 0.5 at all sampling sites, suggesting the availability of stable substrates suitable for scrappers and filters. Conversely, site one exhibited a habitat stability index below 0.5, indicating an insufficiently stable habitat. During a study conducted in Gumara River, sampling sites two, three, and four displayed abundant loading of fine particulate organic matter for filters (TFPOM (suspended)/BFPOM (sediment) > 0.50) based on the proposed threshold values. In terms of predator-prey dynamics, the river's top-down predator control at all sampling stations was overwhelmed by an excessive number of predators (P/P >

0.2), contributing to an overall predator burden throughout the river. Site six exhibited the highest predator-prey ratio, followed by site seven, site two, and site one, while the remaining sampling sites displayed moderate predator-prey ratios.

4.5. Habitat Quality Assessment

Based on habitat quality index using the methodology described in Gitonga (2021), habitat measurement characteristics shown in Table 4.6. As a result, habitat parameters like Available instream cover Bottom Substrate, Channel sinuosity and Riparian buffer vegetation were relatively maximum at sampling site 3 than others. Whereas, the Number of riffles were maximum in site 4. Generally, the highest (28) total habitat scoring was found in site 3 where the upstream of the main river in mountainous area and the lowest (18) was in site 7. Except site seven and site two the other sampling sites are categorized in excellent habitat quality ranges (26-31). Most habitat parameters had significant differences among sampling stations (ANOVA, $P < 0.05$) except embeddedness, bank stability and total habitat scoring.

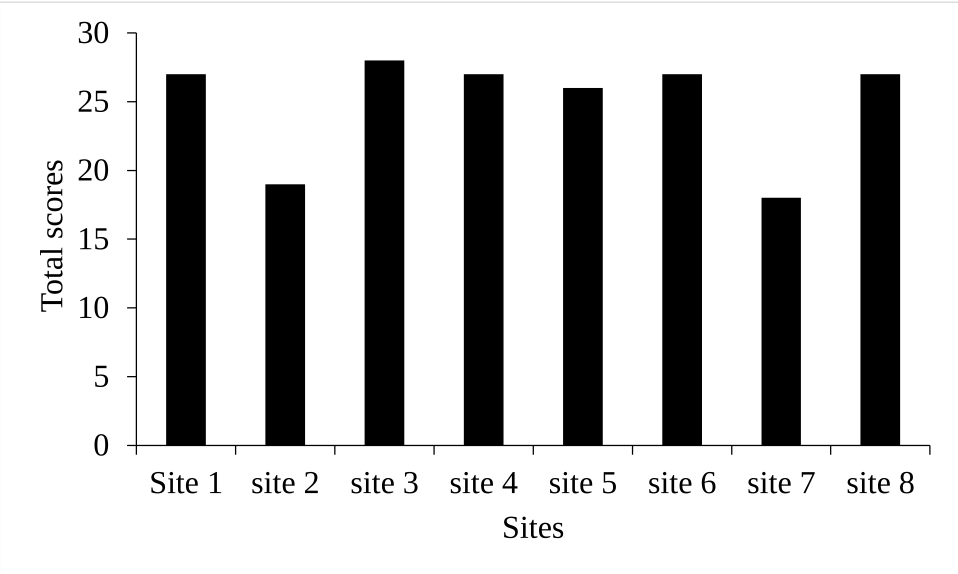

Figure 9 :Habitat Quality index in the different sites of the river.

4.6 Relationships of macroinvertebrates and environmental variables

Moderately sensitive macroinvertebrate taxa (Gomphidae, Aeshnidae and high tolerance taxa Lestidae and Coenagrinidae) were related with Po4, No2, pH, Temperature and velocity on sampling site 6 and highly influenced by axis 1. In addition, moderately sensitive and tolerant taxa Planorbidae, Psychodidae, Tabanidae, and Gyrinidae was associated with axis 2 and strongly correlated with HQ, DO, TDS, and NH_3 on sites 2 and 3. moreover turbidity and width are influenced at site 7 axis 1 affecting Elmidae, Hirudinae and Notonectidae.

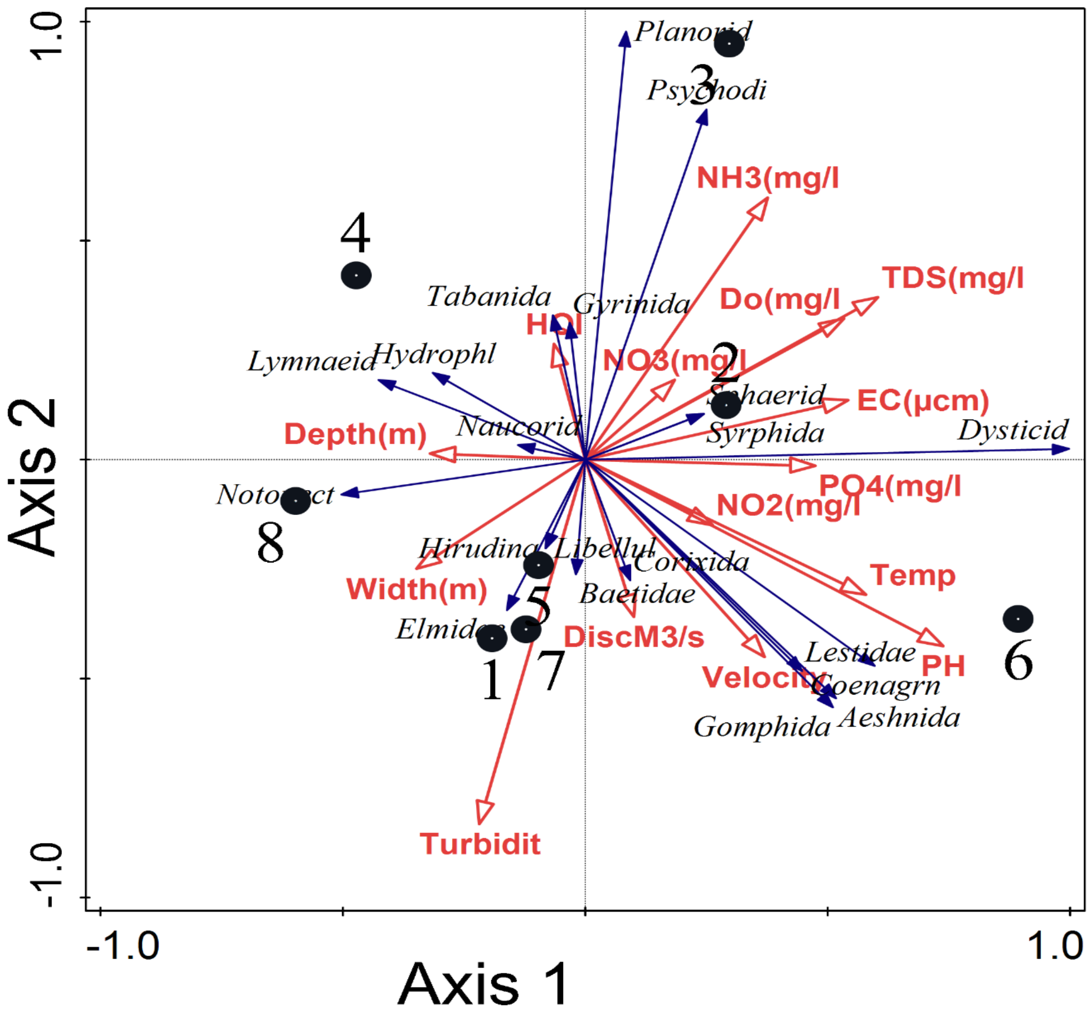

Figure 10: Triplot Redundancy Analysis (RDA), Macroinvertebrates environmental variables and sampling sites

Circles indicate sampling sites. The arrow shows the major physicochemical parameters that structure the benthic community: DO=dissolved oxygen, Turb=Turbidity, PO4=phosphate, NH3 =ammonia, No2=Nitrate, No3= Nitrite, EC= Electric conductivity, Temp= Temperature,HQ=habitat quality, Disc=discharge. Macroinvertebrates are abbreviated as (Baet=Baetidae,Noto=Notonectidae,Nauco=Naucoridae,Corix=Corixidae,Dysti=Dystiscidae, Gyri=Gyrinidae,Hydrophi=Hydrophilidae, Lym=Lymnaeidae, Syrphi= Syrphidae, Psychodi=Psychodidae,Lesti =Lestidae, Gomphi=Gomphidae, Coenagrn= Coenagrionidae, Aesh= Aeshnidae, Elmi= Elmidae, Libellul= Libellulidae, Hirudi= Hirudinae, Sphaeri= Sphaeridae, Tabani= Tabanidae, Planorbd= Planorbidae).

4.6.1 The Relationship between Habitat Quality and Benthic Macroinvertebrates

As illustrated in Figure 4.4 the habitat quality parameters such as Aesthetics of reach, Riparian buffer vegetation, water level, Dimension of largest pool and Bank stability combination had a significant positive relationship with family Notonectidae and Corixidae. Boottom Substrate and Number of riffles had correlation with Tabanidae, lymnaeidae, Syrphidae, Hydrophilllidae, Psychodidae, Baetidae, Elmidae while habitat parameter Channel Channel sinuosity had positive relationship with Gyrinidae, Sphaeridae, Planorbidae and Naucoridae. Moreover, Available instream cover habitat parameter had a correlation with Dysticidae, Hirudinae, Aeshnidae, Libellulidae, Gomphidae, Coenagrinidae, and Lestidae.

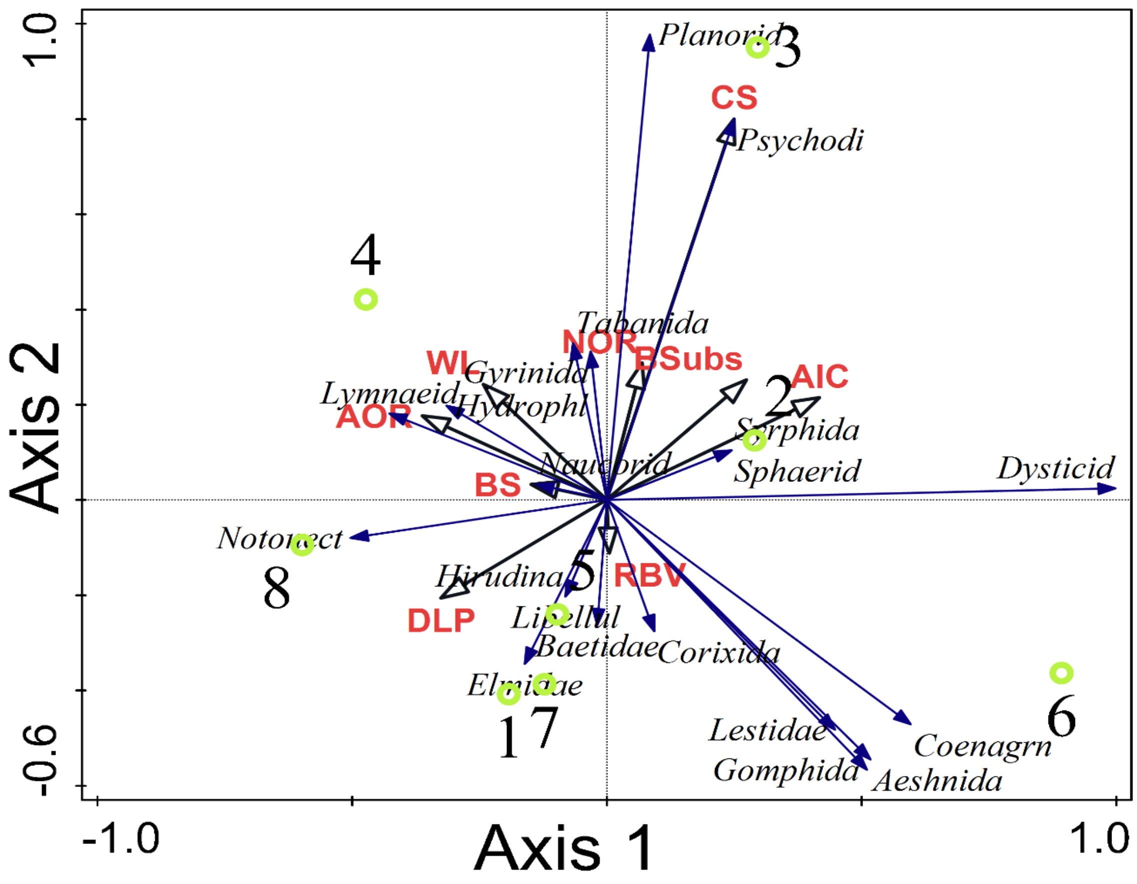

Figure 11: The redundancy analysis (RDA) triplot of the benthic macroinvertebrates in relation to the habitat quality parameters.

Representation where: AIC (Available instream cover), BS (Bottom Substrate), DLP (Dimension of largest pool), NOR (Number of riffles), WL (water level), CS (Channel sinuosity), BS (bank stability), RBV (Riparian buffer vegetation) and AOR (Aeshthetics of reach).

5. DISCUSSION

5.1 Water Quality in Gumara River

5.1.1 Physico-chemical Parameters

Physicochemical parameters provide valuable insights into the condition of a water body during the sampling period. These parameters are influenced by various factors such as geomorphology, climate, and hydrology of the catchment area. Generally, sites within the same ecoregion exhibit similar physicochemical characteristics, unless they are impacted by human activities. Therefore, by measuring physicochemical properties, even in a snapshot manner, it is possible to detect changes over time. The present study reveals significant differences in most of the physicochemical parameters measured both in the field and laboratory across the sampling sites. The surface water temperature ranged from 23.13±0.25 to 27.77±1.21 °C. Variations in mean temperature were observed across different sites, might be influenced by factors such as impact of forest cover/shading, human activities in the watershed, water flow and water depth. The highest temperature was recorded at site five, where the main river is formed by the convergence of two major streams near Maksegnit town. Stream water's optimal temperature ranges from 11-25°C (EPA, 2003) and 12-25°C (WHO, 2008), with Gumara river falling within this range. Temperature plays a crucial role in regulating biological activities, metabolic rates, and the growth of aquatic organisms, leading to variations in biotic features (Zabinski *et al.,* 2009). It also affects the solubility of gases in water, with higher temperatures resulting in lower dissolved oxygen levels in streams (Allan, 1995). Temperature changes impact biogeochemical processes in aquatic ecosystems, such as decreasing gas solubility, increasing organism metabolic rates, and accelerating organic matter decomposition, ultimately depleting oxygen levels and harming aquatic life (Aschalew Lakew, 2012). Site eight exhibited lower temperatures compared to other study sites, likely due to its location downstream and with riparian vegetation that has a cooling effect on the water (Aschalew Lakew, 2014). 1. In addition, Lower elevations in downstream sections can lead to a more stable water flow and reduced exposure to direct solar heating compared to the faster-flowing upstream sections.

The average pH values of the study sites ranged from 8.4±0.15 to 9.10±0.20. The pH values indicate that the medium is basic, as they are greater than seven. These pH values fall within the acceptable range set by the EPA (2003) for surface water, which is 6.0-9.0. Notably, the site located below Maksegnit town, site six, has a relatively higher pH value compared to the other study sites. This observation clearly suggests that there is a significant increase in pH as one moves from upstream to downstream sites and probably due to effluent from the hospital and households of the town. It is important to note that low pH levels can enhance the solubility of heavy metals, which are known to be carcinogenic to aquatic life. Conversely, high pH levels can harm the gills and skin of aquatic organisms, potentially leading to their death (Aschalew Lakew, 2012). Additionally, when the pH exceeds 9, nitrogen compounds undergo conversion into ammonia (NH_3), which is highly detrimental to aquatic life. In the present study, there was no statistically significant difference ($p>0.05$) in pH among the sampling sites.

The concentration levels of dissolved oxygen in the water of Gumara River varied between 5.99 ± 0.21 and 8.23 ± 0.26 (mg/L). The forested area, specifically site three, had the highest levels of dissolved oxygen, while site eight had the lowest. A significant difference was observed among the different sites (ANOVA, $P < 0.05$). Dissolved oxygen is influenced by various factors such as temperature, turbulence, solute pressure, photosynthesis, respiration, organic matter availability, and the presence of decomposer biota (Aschalew Lakew, 2012). As temperature and salinity increase, dissolved oxygen levels decrease. Additionally, high organic and nutrient loads lead to lower dissolved oxygen concentrations due to increased decomposer activities (Deshu Mamo, 2004). Similar findings were reported by Zang *et al*. (2011), who found that as pH decreases, dissolved oxygen also decreases. Kuligiewicz *et al.* (2015) stated that dissolved oxygen levels in the river fluctuate due to temperature-dependent biological processes such as photosynthesis, respiration, and decomposition of organic matter. The survival, development, and mobility of aquatic organisms heavily rely on the availability of suitable dissolved oxygen levels. According to Davies & Cornwell (2017), based on the US EPA standard, the minimum acceptable level of dissolved oxygen in water should not be below 5.5 mg/L. Therefore, based on the mentioned value, the concentration of dissolved oxygen in Gumara River meets the required level for aquatic organisms.

The range of electrical conductivity values varied from 289±6.93 to 472.33±7.51 (μS.cm-1)), with the highest conductivity observed at site 6 and the lowest at site 8. The standard value set by EPA (2003) for EC was 1000 µs/cm for surface water. Gumara river's conductivity fell within the permissible limit compared to these standards. Megech river had a maximum EC of 928 µS.cm-1 (Gizachew Teshome *et al*.,2024), higher than Gumara river. The discharge of fertilizers, municipal sewage, and domestic waste into Gumara river may have contributed to its conductivity. The variation in conductivity levels is linked to the total dissolved solids in water (Bauder *et al*., 2003). Conductivity measures the ability of an aqueous solution to conduct electricity, influenced by ions, their concentration, mobility, valence, and temperature. Increasing conductivity and cations result from the breakdown and mineralization of organic materials (Abida, 2008). Significant differences were found between sites (ANOVA, $P < 0.05$). Electrical conductivity levels at all sample stations were below the WHO standards for household water (1000 S/cm).

The average TDS value ranged from 142.9±0.23 to 236.00±3.46 mg/L, with the highest value observed at site six. Similar to conductivity, the increase in TDS levels could be attributed to disturbances in the watershed, runoff of organic waste, and the use of fertilizers. The presence of TDS in the river can have an impact on water clarity, temperature, interstitial space, and the accumulation of harmful pollutants like heavy metals (USGS, 2003, Murphy, 2005). This study revealed varying amounts of total dissolved solids across all sites. However, according to the Food and Agricultural Organization (FAO) recommendation, the acceptable range for TDS in livestock drinking water is 100-1,500 mg/L. EC is highly sensitive to and correlated with total dissolved solids and major ions. In natural water bodies, TDS can be estimated by dividing EC by two. Therefore, the quantity of total solids entering the river may be the primary factor contributing to the increase in conductivity, temperature, and oxygen depletion. The TDS concentration in drinking water should not exceed 1000 mg/L (WHO, 2008).

Turbidity is a parameter used to measure the cloudiness of a body of water, which is typically caused by suspended or dissolved particles. These particles can include clay, silt, organic and inorganic matter, colored organic compounds, algae, and other microscopic organisms

(Minnesota Pollution Control Agency, 2008). The average turbidity values ranged from 992.4 to 266.5 NTU, with the highest value observed at site eight (Table 3). Site three had a lower mean turbidity value of 266.5 compared to the other sites. The World Health Organization (WHO) sets the turbidity standard at 5 NTU (2008), and all sites exceeded this level. This suggests that human activities, hospital and household discharges, agricultural practices, inadequate sanitation for riparian communities, and runoff have contributed to the high turbidity levels in the river. The continuous accumulation of suspended and dissolved solids in water bodies reduces transparency, impacting primary productivity and benthic macroinvertebrates (Emere and Narisu, 2007).

The ammonia levels varied from a maximum of 0.62±0.06 mg/L at site two to a minimum of 0.033±0.005 mg/L at site 8 (downstream of the river). Significant differences were observed among the sampling sites in terms of ammonia concentration (ANOVA, $P < 0.05$). The presence of ammonia in the study area could be attributed to the use of fertilizers in the watershed, domestic sewage discharge, and the biological decomposition of manure (WHO, 2011). Ammonia is a harmful pollutant commonly found in sewage, liquid manure, and liquid organic waste. It also occurs naturally in water bodies due to the breakdown of nitrogenous waste in soil and water, biota excretion, nitrogen gas reduction by microorganisms in water, and gas exchange with the atmosphere. Therefore, it can serve as an indicator of the health of natural water bodies such as rivers (Deepa *et al.*, 2016). The ammonia levels exceeded the WHO standards for residential water at site two (0.5 mg/L), indicating potential toxicity to aquatic organisms.

The average nitrite concentration was high at site six (0.4 ± 0.02 mg/L), while it was low at site two (0.004 ± 0.0001 mg/L) and site one (0.02 ± 0.01 mg/L) (Table 3). The nitrite-nitrogen levels measured at all sampling sites in the study area were below the recommended limit. This could be due to the absence of microbial communities responsible for converting nitrite (e.g., Nitrosamines bacteria) from nitrogen-containing compounds, lack of oxygen for chemical processes, and high organic load. The standard concentration for nitrite-nitrogen is 3 mg/L (WHO, 2008).

Phosphate levels varied among the study sites, with the lowest average concentration recorded at site two (0.42 ± 0.04 mg/L) and the highest at site six (1.58 ± 0.54 mg/L). This variation can be attributed to different land use practices and human activities in the area. The slight increase in phosphate levels at these sites may be a consequence of diffuse pollution from agricultural activities in the watershed, including the application of fertilizers and pesticides, as well as other human-related factors such as cleaning practices and waste disposal. The significantly higher concentration of phosphate at site six could be associated with the release of phosphate-containing products like detergents and soaps into the river water. Phosphorus is an essential nutrient in aquatic ecosystems, supporting autotrophs and providing food and habitat for aquatic organisms through primary productivity. However, excessive phosphate levels can lead to eutrophication, which has negative impacts on aquatic systems. The phosphate concentrations observed in this study exceeded the permissible value established by the WHO (0.4 mg/L in 2008) and may have detrimental effects on benthic macroinvertebrates.

In this study, the physicochemical water quality parameters indicated good or moderate ecological quality at the upstream locations of the study rivers. These were within the range recommended for irrigation, aquatic life, and livestock consumption (FAO 1985; WHO 2008; US EPA 2022). This study findings agree with those of Aschalew Lakew. (2020) and Assegide *et al*. (2022) that Ethiopian upstream river locations are often of better ecological quality than those downstream.

5.1.2 Hydro-morphological Variables

According to the result in Figure 4.1(a), the highest depth value (0.38± 0.08 m) was observed in a station eight. While, the lowest (0.07±0.012 m) was in station five. There were statistically significant variations in depths between sampling sites. The major cause for depth variation could be the availability of canopy cover to the topography, the bank stability, riparian vegetation protection, gradient of the area and types of substrate composition which is found. This is true in station eight which has higher canopy cover and bank stabilities than others. This corresponds with Cunningham & Schalk (2011), who indicated that the low water depth might be related to significant water evaporation and low water input from rain and runoffs. The highest width value (26 m) was measured in confluence of the river (site five) and the lowest (2.71 m) in headwater (site two and three). Generally downstream sites have higher width than upstream sampling sites. This result consistent with river continuum concept. This variation possibly might be owing to the status of channel stability, bank vegetation protection, human activities, and vulnerability to sedimentation deposition, slope differences and the contribution of other tributaries in the watersheds. The maximum (0.5 m/s) velocity value was measured at site five and minimum (0.05 m/s) in site eight. There were significant statistical variations between sampling sites (ANOVA, $P < 0.05$). Differences in velocity can be due to the shape of the channels, slope, and the width of channels and the composition of the substrates. For example, the highest velocity found where the area had a steep slope and in narrow channels. However, the lowest velocity occurred in the gentle slope and wide channels. This idea also confirmed by the river continuum concept (RCC), which states that the velocity of water decreased from the upper reaches (narrow channel) to lower reaches (wide channel). Dietz & Clausen (2008) also moted that stations that had sufficient boulder and gravels substrates results in high water velocity. Silt and sand substrates, on the other hand can have low water flow. Likewise, the flow rate (discharge) of the river fluctuated between 1.2 m^3/s and 0. 1 m^3/s. The highest value was recorded in the lower reaches of the river and the lowest in the upper reaches. This may be because of the area, depth, width, gradient, channel size, channel stability, type of substrates, and other stream attributes that contributes to increasing flow downstream of the river. This result was consistent with the concept of (RCC), which stated that channel size, cross-sectional area, nutrients (mineral), and width increased from top to bottom.

The interaction between the physical environment of a stream and benthic macroinvertebrates has received a considerable attention in scientific literature. benthic invertebrates' community structure is a conservative indicator of water quality (Leland & Fend 1998). In this study, the majority of macroinvertebrates had significant associations with hydromorphological variables, indicating good markers of hydro geomorphological stress. Discharge (channel width, depth, and flow velocity) is thought to play a dominant role in regulating of benthic species in general (Hart & Finelli 1999). The dispersal of macroinvertebrates in flowing water is mainly determined by channel width, conductivity and pH (Soininen & Kononen 2004).

5.2 Macroinvertebrates Abundance, Composition, Diversity and Functional Feeding Groups

5.2.1 Diversity of Macroinvertebrates

In this study 6 orders 1 class and 19 families were identified. The highest taxa richness (5 families) was recorded at site 6, the site faced minimal human impacts. In the current study, the macro-invertebrate communities' composition was higher when compared to findings by Gurmessa Tessema & Agumassie Tesfahun (2018); that found 6 orders and 11 families at Teltele Stream, Ambo West Showa, Ethiopia. However, the macro-invertebrate communities composition in this study were lowest when compared to related findings, such as 10 orders and 37 families in the spring and stream sites of the upper Awash River (Negero *et al*., 2017), 10 orders and 34 families in Cheffa wetland from Borkena Valley (Getachew *et al.,* 2012), 9 orders and 34 families in Wedech River in Debrezeit (Tamiru *et al*., 2017), 12 orders and 33 families in a highland stream in Northern Ethiopia (Teferi *et al.,* 2017), 7 orders and 20 families in Northern highland of Ethiopia (Wolemariam *et al*., 2018) , 7 orders and 20 families in Enda Gabr stream in Mekele Northern Ethiopia (Tesfay *et al*., 2016), 20 families in Northern highland of Ethiopia (Wolemariam *et al*., 2018) and 9 orders and 36 families in Southwestern Ethiopia (Worku and Ambelu, 2018) were recorded .Moreover the results of study were lower than those of study by Gizachew Teshome *et al,*(2024) 33 families and nine orders were found on the Megech River . The difference in diversity of benthic macroinvertebrate communities in this study area compared to other rivers could be due to differences in substrate type (e.g., sand, gravel, rock), water flow, presence of aquatic vegetation, difference levels of pollution, and different anthropogenic activities, hydromorphological variables, Land use practices and length of study period. The majority of families identified in this study had moderate tolerance levels, particularly the family Dysticidae. This suggested that River was moderately polluted compared to many Ethiopian rivers.

The values of Shannon-Wiener diversity index (H′) in the sampling stations ranged from 0.34 to 1.53, as shown in Table 4.4. This value is lower than that of Mbaka *et al,* (2014) specified value. According to Gencer and Nilgun (2010), most values measured by Shannon diversity index are between 1.5 and 3.5 and rarely exceed 4.5. Value above 3.0 indicate a stable and balanced habitat structure, while values below 1.0 indicate pollution and degradation of the habitat structure. Based on these criteria, the other sampling sites in Gumara, with exception of site eight, did not exceed the Shannon diversity index value of 1.5. Similarly, sites one, two, three, five, and six had Shannon diversity index values below one, indicating increased levels of pollution and degradation of the habitat structure. The highest value of 1.53 was observed at site eight, followed by 1.49 at station four. The lowest value of 0.34 was measured at site one. There were minimal differences in Shannon Wiener diversity among sampling sites. This could be due to factors such as the availability of food sources in quality and quantity, trophic structure and the level of environmental stress at each location. This result is consistent with the findings of Morphin-Kani & Murugesan (2014), who indicated that high macroinvertebrate diversity indicates a good environment, while low diversity indicates a lack of habitat availability. In addition to the Shannon-Wiener index, the Simpson diversity index (1-D) was also applied to depict the diversity within the macroinvertebrate community. According to Mandeville (2002), the Simpson Index (1-D) ranges from 0 to 1, with 0 indicating low diversity and 1 indicating high diversity. In Gumara River, the Simpson's diversity index varied between 0.15 (site six) and 0.77 (site eight), with the highest value observed at site eight and the lowest value at site six. The limited variety of macrohabitats observed at site six, pollution from the city of, and its susceptibility to invasion due to being openness are likely reasons for this. The recorded value in Gumara River is below the specified range, indicating low diversity. The Shannon-Wiener Index (H′) and Simpson diversity index (1-D) show a similar trend at each sampling station, but they have an inverse relationship with Dominance (D). This finding is consistent with Magurran's (2013) statement that although Shannon and Simpson's diversity indices have different theoretical foundations and interpretations, they are essentially interrelated. Furthermore, highest taxa richness was found at site two (9), followed by site seven (8), while the lowest (3) were found at sites one and three. The difference between sites may be due to the level of environmental pollution in the area and increased human activities. For example, at site two there is a minimal human activity. This result is consistent with the assumption of Andem *et al*., (2012) agreed that low taxa richness indicates severe environmental degradation from various anthropogenic activities, which in turn impacts the benthic macroinvertebrate community. Evenness ranges from 0.22 to 0.93, with the highest value (0.93) at site eight and the lowest (0.22) at site seven. This can be attributed to the state of water quality. The results were consistent with the research of Dipankar & Jayanta (2015) and Upen & Sarada (2015), which highlighted that high evenness resulted in poor water quality and limited food availability. The data showed that the evenness index and richness index did not follow the same patterns at all sampling locations. This is consistent with the study by Chrisoula et al. (2011), who suggested that an increase in the species richness index led to a decrease in the evenness index.

5.3 The Relationship between physicochemical parameters and Benthic Macroinvertebrates

Benthic macroinvertebrates exhibit sensitivity to changes in the environment (Bazzanti *et al.,* 2017) and demonstrate rapid responses to various types of environmental changes, including alterations in physical, chemical, and biological conditions within aquatic ecosystems (Odume *et al*., 2012; Poikane *et al.,* 2016). Likewise, the present study shows that dissolved oxygen (DO), temperature, NH3 and NO3 have a significant positive correlation with numerous macroinvertebrates, thereby influencing their distribution. The diversity, richness and spatial distribution of benthic macroinvertebrates are influenced by physicochemical parameters. In particular, the levels of physicochemical parameters such as oxygen content, pH and other physical parameters appear to support the survival of most benthic invertebrate communities in the Gumara River. Martinez-Sanz et al. (2014) observed a decrease in the number of Ephemeroptera, Plecoptera, and Trichoptera taxa in wadeable streams with increasing nutrient concentration, indicating a negative correlation. Similarly, this study shows that as nutrient concentrations increase, sensitive taxa are eliminated, further supporting a negative association (Martínez-Sanz et al., 2014). In this study, Ephemeroptera were found only at the upstream site, while Plecoptera and Trichoptera were completely absent at all sites.

The result on Redundancy analysis (RDA) showed the relationship between benthic macroinvertebrates taxa (biological indexes) and water quality parameters.

The result of Redundancy analysis (RDA) illustrated the correlation between the taxa of benthic macroinvertebrates and water quality parameters. This demonstrated the role of macroinvertebrates as bioindicators. Moreover, Ephemeroptera were present in site one as well as site two. This could be attributed to the existence of stony substrates in both locations, which provide a suitable habitat for highly sensitive macroinvertebrates. As per Belmar *et al*, (2013) Ephemeroptera (EPT) are known for their preference for stable habitats in swiftly flowing water upstream. On the other hand, Odonata, Coleoptera, Diptera, and similar taxa are recognized for their ability to thrive in disturbed environments within stagnant water.

The abundance of Diptera and Odonata is high in degraded sites, as indicated by Lyimo (2012). Furthermore, there is a significant association between this order and temperature, salinity, nitrate, and phosphorus. Benthic macroinvertebrates need different optimal temperatures for their survival, growth, and reproduction, as noted by Singh & Sharma (2014), Prommi & Payakka (2015). On the contrary, Gyrinidae, hydrophilidae, lymnaeidae, and Naucoridae show a negative relationship with nitrate and phosphorus. Maneechan & Prommi (2015) reported a similar finding, suggesting that Baetidae also has a negative correlation with phosphate. Therefore, these taxa, which are limited to clean, oxygenated water and are vulnerable to pollution, can serve as valuable indicators for assessing water quality. In this study, benthic macroinvertebrates were influenced by the sampling site's location (upstream or downstream), as well as the sources of anthropogenic activities or land use.

The study area is currently facing erosion of streambanks and disturbances caused by human activities, mainly due to settlements, agriculture, and grazing, leading to a mixture of terrestrial soil and water. As a result, the river has become a suitable habitat for semi-aquatic fauna such as Diving beetles (Dytiscidae) and Hydrophilidae, which thrive in slow-flowing water directly in contact with terrestrial soil. Some sampling points along the river are situated near preserved upland forests, creating an optimal environment for species adapted to turbulent water, rocky substrate, and leaf litter accumulation. Consequently, these sites have supported the majority of Ephemeroptera (Baetidae), Diptera (Tabanidae), Coleoptera (Hydrophilidae), and Gyrinidae taxa (Gerth, Li &Giannico, 2017).

Upstream sites were found to harbor sensitive and moderately sensitive taxa (such as Baetidae, Elmidae, and Tabanidae) that prefer improved water conditions. Meanwhile, tolerant taxa like Corixidae, Gyrinidae, and Notonectidae were recorded in intermittent sites. Additionally, downstream sites were dominated by taxa preferring slow-moving water with the availability of pools and hosts (Gomphidae, Dytiscidae Lestidae.). These observations in Gumara Maksegnit headwater streams are in line with reports from tropical and temperate regions (Gerth, Li, & Giannico, 2017; Keke *et al.,* 2017).

5.4 Benthic Macroinvertebrates Relationship with Habitat Quality

At site three (near reference site), the highest total habitat scores of 28 were recorded, followed by 27 at station site one, six, and eight. Conversely, the lowest scores were documented at site seven (18) and two (19) as shown in Table 4.7. The higher scores at site three can be linked to fewer human activities in that area. Common human activities such as sand mining, animal husbandry, washing, and agricultural activities were observed at station site seven and two. Hence, these human activities have the potential to diminish habitat quality in both riparian and river ecosystems. The presence or absence of benthic macroinvertebrates in aquatic environments is significantly influenced by habitat quality and physicochemical water quality characteristics (Akasaka et al., 2010). The families of Tabanidae, lymnaeidae, hydrophilllidae, Gyrinidae, and Naucoridae at site four demonstrated a positive association with habitat quality parameters such as bottom substrate, water level, number of riffles, and aesthetics of reach. This correlation can be attributed to the availability of suitable bottom substrate and adequate water levels along the river's banks, which provide abundant food resources, egg-laying sites, and refuge zones from predators. Conversely, the families Corixidae, Lestidae, Gomphidae, Coenagrinidae, and Aeshnidae exhibited a negative relationship with these habitat parameters. Similar findings were reported in Southeast Asian tropical streams by Maneechan & Prommi (2015).

In general, these results align with earlier research conducted by Brown *et al* (2011) & Masikini *et al* (2018). These studies emphasized the strong connection between the abundance and composition structure of aquatic insect communities and physical habitat attributes, substrate type, as well as biological factors like dispersal, competition, and predation.

5.5. Functional Feeding traits of Macroinvertebrates and Ecosystem Attributes

The Gumara River contained four functional feeding groups (FFGs): gathering collectors, filter collectors, predators, and scrapers. This study aligns with the conclusions drawn by Boyero *et al* (2011), Brasil *et al* (2014), and Masese *et al* (2014), who observed diverse feeding groups in many tropical rivers. The presence of various feeding groups can be attributed to the uneven distribution of energy inputs and changes in river morphology over time, such as variations in channel characteristics (e.g., presence of rapids, riffles, plant cover, and water flow), leading to a diverse range of substrates and microhabitats. Consequently, this diversity influences the arrangement of FFGs in lotic environments (Brasil *et al.,* 2014).

The study's findings revealed that the Gumara Makesegnit River had a prevalence of predators, followed by scrappers and filterers, while there was a lack of shredder-feeding groups. Figure 4.5 shows that downstream (site six) had the highest composition of predators (36.64%), whereas the outlet of the river (station eight) had the lowest (2.27%). Discrepancies in predator distribution between sites may be influenced by prey availability and the presence or absence of riparian vegetation. Notably, certain predators like Odonata rely on vegetation for hunting and resting, especially for less mobile species (Koneri et al., 2017). In accordance with the river continuum concept, the presence of predators may be influenced by the availability of prey, and conversely, the abundance of predators can also impact prey populations. Favretto et al. (2014) found that the predator functional group is highly abundant in human-altered environments. The forested area (site three) had the highest percentage of scrapers (53.04%), while scrapers were absent at sampling site one and least abundant at sampling site five. Site one exhibited the highest percentage of gatherers

(57.14%), which are known to feed on small particles collected on the stream bed. These fine particles are produced from the breakdown of organic matter by shredders, indicating that the presence of shredders determined the abundance of gatherers. The highest scraper feeder population was observed in the forested site (site three), while the lowest was found in the agricultural area. This could be attributed to the reduced periphyton productivity and the absence of macrophytes as food sources, likely resulting from increased depth and turbidity in agricultural areas. This is because scrapers typically feed on macrophytes attached to bedrock, stones, and vegetation (Oliveira & Nessimian, 2010). Barbee (2005) also found similar results, suggesting that scraper densities are influenced by the presence or absence of algal biomass and production. Lymnaeidae, Planorbidae, and Corixidae were the common scraper families observed in the river during the study period, as reported by the study.

The collector-filterer was discovered exclusively at locations one and two and was not present at the other sampling sites. Collector-gatherers and filter collectors exhibited the lowest ratios of total abundance. The limited presence of collectors in this river is believed to be linked to the accelerated decomposition of litter and leaves by microbial activity in tropical regions, due to high temperatures. As a result, there is insufficient plant debris available for filtering, and the river's high turbidity leads to a low presence of algae, contributing to their low proportion. The inadequate presence of collector gatherers and collector filterers at stations one and two, as well as their absence from all other sampling locations, may be linked to the irregularities in seston supply and limited water circulation in the muddy sediment (Uwadiae, 2009). The variations observed among the sites could be attributed to water speed and the level of disturbances. Parker *et al* (2013) validated this concept: an increase in water velocity leads to a higher presence of filter feeders, indicating that water velocity plays a role in aiding filtration. The main families categorized as filter feeders in the research area were Syrphidae and Sphaeriidae. The composition of benthic macroinvertebrates and the distribution of functional feeding groups differed across the various sampling areas, likely due to environmental variations, human activities, energy input distribution, and changes in river morphology, including channel characteristics such as rapids, riffles, plant cover, stable substrates, food availability, and water flow (Brasil *et al*., 2014; Azhar *et al*., 2015; Merritt *et al*., 2017; Atkinson *et al*., 2018).

The ratios of counted benthic macroinvertebrate functional groups were used to calculate surrogates for four ecosystem attributes (P/R, TFPOM/BFTOM, habitat stability, and P/P) following a summarized protocol in Table 2.3.

In Table 4.6, it was observed that the production to respiration (P/R) ratio ranged from 1.2 to 61. Based on this numerical data, all sampling stations exhibited autotrophic characteristics (P/R>0.75). The absence of shredders in this study possibly contributed to this result, which contradicts findings from other studies. Previous works by different authors have suggested that most flowing aquatic ecosystems, particularly in the tropics, are predominantly heterotrophic (Cummins *et al*., 2005). This finding supports the widespread perception that shredders are not as prevalent in tropical regions (Dobson *et al*., 2002, Dudgeon & Wu 1999; Wantzen & Junk 2000; Wantzen *et al*., 2002). Numerous studies have demonstrated the diminished presence of shredders in tropical regions. The absence of shredders may be a result of the decrease in riparian forests, which are necessary to provide enough organic matter such as woody vegetation and various types of riparian plants. When native vegetation is removed for agricultural and other activities, it reduces the external resources available to the river, leading to a decline in shredder populations (Minaya *et al*., 2013). In tropical water bodies, a diverse range of riparian plants with both physical and chemical defense mechanisms, rapid release of nutrients, and high levels of predation can hinder the success of shredders (Wantzen *et al.,* 2002). Agricultural practices, such as crop cultivation, are widespread along the Gumara River at most sites, with the exception of site three, and may contribute to the ineffectiveness of the riparian area. Sites one and three have sufficient stable substrates such as bedrock, boulders, cobbles, and debris to support filter feeding and scraping, leading to the high frequency of filter feeding groups at site one. However, the calculated value indicated that sites five, six, seven, and eight had a habitat value lower than the threshold value (< 0.5), suggesting that these sites did not provide adequate and stable habitat for functional feeding groups of macroinvertebrates.

With the exception of site eight (outlet), most sites were overwhelmed with predators, contributing to an overall surplus of predators in the entire river. In contrast, at Site four, the top-down predator control over prey was considered moderate. Throughout this study, Odonata, Hemiptera, and Diptera were identified as the common predators.

The FFG ratios indicate widespread human influences in the Gumara River, such as vegetation removal, livestock grazing, washing activities, and crop farming, suggesting impairment of the river ecosystem's function. The FFG ratios demonstrate significant human impacts in the sampled location, serving as sensitive indicators of the overall health of the ecosystem. Utilizing different macroinvertebrate functional group ratios can contribute valuable information with minimal effort by substituting for distinct ecosystem characteristics. The study results suggest that human activities have affected the physical and chemical properties, as well as the biological aspects, of the river. The changes in land use have led to the decline of riparian vegetation, negatively impacting the physical, chemical, and biological aspects of aquatic ecosystems. This is particularly significant in countries like Ethiopia, where the common practice of replacing indigenous forests with deforestation, urban expansion, and agricultural/pastoral activities prevails. Macroinvertebrate communities have been effectively used to assess the ecological impacts of different land uses. The sampling sites in the partially forested area and the reference location exhibited a higher diversity of macroinvertebrates and the presence of sensitive species. Conversely, the sites situated in agricultural area showed reduced macroinvertebrate diversity and the absence of specific sensitive taxa. Each location offers unique insights into the current ecological condition through the analysis of different factors. Therefore, it is crucial to combine these metrics with biological and physicochemical assessments to obtain a comprehensive understanding of both aquatic and terrestrial habitats.

6. CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

This study showed that most water quality parameters in Gumara River were within the acceptable limits for aquatic life. The river's environmental factors had directly or indirectly affected macroinvertebrate assemblages, showing that macroinvertebrates were useful indicators of water quality in Gumara River. Less sensitive Macroinvertebrate taxa richness and diversity decreased downstream indicated better water quality (lower water temperature, TDS, pH and high DO), as well as more favorable conditions for macroinvertebrate communities in the upper sites. A variety and abundance of macroinvertebrates were recorded from the Gumara River with the Coleoptera dominating, followed by the mollusks. The abundance of less-sensitive organisms and the total absence of the sensitive order Plecoptera could be an indication of increasing organic pollution even in the upper sites of the Gumara River and low presence of EPT taxa could be scarcity of stony substrates and lack of riparian vegetation in the river. Most of macroinvertebrates identified in this study were moderately tolerant except Mollusca, this indicates the river is moderately stressed. However, there were few pollution-sensitive and some moderately-sensitive families which imply that the upper sites were not as polluted as the downstream.

The results of this study concluded that sampling site two and site seven in Gumara River had a high macroinvertebrates taxa richness as compared to that of the other sites. The most abundant macroinvertebrate taxa groups were Coleoptera, followed by Mollusca in site three and four, respectively. On the other hand, order Odonata presence in downstream sampling sites and absent in the upper stream sites. site six had higher number of macroinvertebrates that other sites. The diversity (Shannon-Wiener values and Simpson's) was high at site eight and six, but lower in site one upstream. The results have shown also that there was diversity of FFGs namely: predators, gathering-collectors, filtering- collectors, and scrapers. Predators were the most dominant particularly in agricultural sites. However, shredders were not present. Functional feeding traits such as predators and scrapers were dominated in such turbid rivers, limiting the use of macroinvertebrates as indicators of ecological stress. Most macroinvertebrates are good indicators of hydro geomorphological stresses. Thus, this study also concluded that the composition of benthic macroinvertebrates functional feeding groups and ecosystem attributes were affected by the human activities near the river such as agriculture, grazing, deforestation and washing activities which lead to natural habitat quality deterioration and soil erosion. In addition, variations in natural environmental factors, such as the flow of water and the composition of the riverbed, can have distinct effects on the communities of macroinvertebrates. This is evident in the smaller number of EPT taxa, which is attributed to the limited presence of rocky substrates and sparse riparian vegetation.

According to the habitat quality index using the methodology described in Gitonga (2021) total habitat quality scores for all stations were generally classified as moderately modified, this suggests that there may have been relatively minimal loss and alteration of natural habitat and biota. Environmental variables in the river had a direct and/or indirect impact on macroinvertebrate assemblages, indicating that macroinvertebrates might be used as water quality indicators in the Gumara River. In the same way, habitat degradation negatively impacts macroinvertebrate communities like high sedimentation problems. Functional feeding traits of macroinvertebrates has implication about the ecology of the rivers.

6.2 Recommendations

Based on study findings, it is recommended that:

- ✓ Further studies should be conducted along the Gumara River, taking into account seasonal variations, in order to assess the water quality during different seasons and establish its status.
- ✓ Developing a multimetric index recommended to assess the overall health of ecosystems in a single value important for future research due to the varied nature of these indices
- ✓ It is recommended to identify macroinvertebrate taxa to genus/species level for better bioindicator value, as family-level qualitative data does not provide important information on the status of a water body. Finer taxonomic identification can provide a more accurate representation of ecosystem health (Li *et al*. 2010).

7. REFERENCES

Abebe Beyene, Taffere Addis,Demeke Kifle,Worku Legesse., Kloos, H., & Triest, L. (2009). Comparative study of diatoms and macroinvertebrates as indicators of severe water pollution: Case study of the Kebena and Akaki rivers in Addis Ababa, Ethiopia. *Ecological indicators*, *9*(2), 381-392.

Allan, E., Manning, P., Alt, F., Binkenstein, J., Blaser, S., Blüthgen, N., ... & Fischer, M. (2015). Land use intensification alters ecosystem multifunctionality via loss of biodiversity and changes to functional composition. *Ecology letters*, *18*(8), 834-843.

Allan, J. D. (2004). Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annu. Rev. Ecol. Evol. Syst.*, *35*, 257-284.

American Public Health Association (APHA). Standard Methods for the Examination of Water and Wastewater, 21st ed.; APHA: Washington, DC, USA, 2001.

Andem, A. B., Okorafor, K. A., Udofia, U., Okete, K. A., & Ugwumba, A. A. A. (2012). Composition, distribution and diversity of benthic macroinvertebrates of Ona River, Southwest, Nigeria. *European Journal of Zoological Research*, 1 (2), 47-53.

Argaw Ambelu, 2009. Biological monitoring based on macroinvertebrates for decision support of water management in Ethiopia. PhD thesis, Ghent University, Gent, Belgium.

Arimoro, F. O., & Muller, W. J. (2010). Mayfly (Insecta: Ephemeroptera) community structure as an indicator of the ecological status of a stream in the Niger Delta area of Nigeria. *Environmental monitoring and assessment*, *166*, 581-594.

Aschalew Lakew (2012). Applicability of bioassessment methods using benthic macroinvertebrates to evaluate the ecological status of highland streams in Ethiopia. MSc thesis. UNESCO- IHE Institute for Water Education, Delft, the Netherlands.

Aschalew Lakew & Moog, O. (2015). A multimetric index based on benthic macroinvertebrates for assessing the ecological status of streams and rivers in central and southeast highlands of Ethiopia. Hydrobiologia 751, Atique, U., & An, K. G. (2018). Stream health evaluation using a combined approach of multi-metric chemical pollution and biological integrity models. Water, 10(5), 661. https:// doi.org/10.3390/w10050661

Atkinson, C. L., Encalada, A. C., Rugenski, A. T., Thomas, S. A., Landeira-Dabarca, A., Poff, N. L., & Flecker, A. S. (2018). Determinants of food resource assimilation by stream insects along a tropical elevation gradient. *Oecologia*, *187*, 731-744.

Aylward, B., Bandyopadhyay, J., Belausteguigotia, J. C., Borkey, P., Cassar, A. Z., Meadors, L., ... & Bauer, C. (2005). Freshwater ecosystem services. *Ecosystems and human well-being: policy responses*, *3*, 213-256

Azhar, S. C., Aris, A. Z., Yusoff, M. K., Ramli, M. F., & Juahir, H. (2015). Classification of river water quality using multivariate analysis. *Procedia Environmental Sciences*, *30*, 79-84.

Barbee, N. C. (2005). Grazing insects reduce algal biomass in a neotropical stream. Hydrobiologia, 532, 153-165. https://doi.org/10.1007/s10750-004-9527-z

Barbour, M. T. (1999). *Rapid bioassessment protocols for use in wadeable streams and rivers: periphyton, benthic macroinvertebrates and fish*. US Environmental Protection Agency, Office of Water.

Baye Sitotaw (2006). Assessment of Benthic macroinvertebrate structures in relation to Environmental degradation in some Ethiopian Rivers. M.Sc Thesis, School of Graduate Studies, Addis Ababa University.

Bazzanti, M., Mastrantuono, L., & Pilotto, F. (2017). Depth-related response of macroinvertebrates to the reversal of eutrophication in a Mediterranean lake: Implications for ecological assessment. *Science of the Total Environment*, *579*, 456-465.

Beasley, G., & Kneale, P. E. (2003). Investigating the influence of heavy metals on macro-invertebrate assemblages using Partial Cononical Correspondence Analysis (pCCA). *Hydrology and earth system sciences*, *7*(2), 221-233.

Belmar, O., Velasco, J., Gutiérrez‐Cánovas, C., Mellado‐Díaz, A., Millán, A., & Wood, P. J. (2013). The influence of natural flow regimes on macroinvertebrate assemblages in a semiarid Mediterranean basin. *Ecohydrology*, *6*(3), 363-379.

Berger, E., Haase, P., Oetken, M., & Sundermann, A. (2016). Field data reveal low critical chemical concentrations for river benthic invertebrates. *Science of the total environment*, *544*, 864-873.

Best, J., & Darby, S. E. (2020). The pace of human-induced change in large rivers: stresses, resilience, and vulnerability to extreme events. *One Earth*, *2*(6), 510-514.

Bhateria, R., & Abdullah, A. (2015). Analyzing uncertainties in lake water: A review. *International Journal of Environmental Sciences*, *6*(1), 155-168.

Booth, D. B., & Jackson, C. R. (1997). Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation 1. *JAWRA Journal of the American Water Resources Association*, *33*(5), 1077-1090.

Boyero, L., Pearson, R.G., Dudgeon, D., Graça, M.A.S., Gessner, M.O., & Albariño, R.J., et al. (2011). Global distribution of a key trophic guild contrasts with common latitudinal diversity patterns. Ecology, 92 (9), 1839-1848. https://doi.org/10.1890/10-2244.1

Brasil, L.S., Juen, L., Batista, J.D, Pavan, M.G, & Cabette, H.S.R. (2014). Longitudinal Distribution of the Functional Feeding Groups of Aquatic Insects in Streams of the Brazilian Cerrado Savanna. Neotropical Entomology, 43, 421–428.

Bredenhand, E., & Samways, M. J. (2009). Impact of a dam on benthic macroinvertebrates in a small river in a biodiversity hotspot: Cape Floristic Region, South Africa. *Journal of Insect Conservation*, *13*, 297-307.

Buss, D. F., & Vitorino, A. S. (2010). Rapid bioassessment protocols using benthic macroinvertebrates in Brazil: evaluation of taxonomic sufficiency. *Journal of the North*

Carpenter, S. R., Stanley, E. H., & Vander Zanden, M. J. (2011). State of the world's freshwater ecosystems: physical, chemical, and biological changes. *Annual review of Environment and Resources*, *36*, 75-99.

Chrisoula, B.P., Vasiliki, K. G., & Savvas, C. T. (2011). Abundance, diversity, and distribution of macrophyte communities in neighboring lakes of different trophic states and morphology in north-central GREECE. Archives of Biological Science, Belgrade, 63 (3), 125– 142. https://doi.org/10.2298/ABS1103763P

Cummins, K. W., Merritt, R. W., & Andrade, P. C. (2005). The use of invertebrate functional groups to characterize ecosystem attributes in selected streams and rivers in south Brazil. *Studies on Neotropical Fauna and Environment*, *40*(1), 69-89.

Cunningham, W.L, & Schalk, C.W. (2011). Ground water technical procedures of the U.S Geological Survey- Measuring water levels by use of a graduated steel tape: U.S Geological Survey Techniques and methods, 1–A1, 151 p.

Dallas, H. F., & Day, J. A. (2007). Natural variation in macroinvertebrate assemblages and the development of a biological banding system for interpreting bioassessment data—a preliminary evaluation using data from upland sites in the south-western Cape, South Africa. *Hydrobiologia*, *575*, 231-244.

Daniel Asfaw & Getachew Workineh (2019). Quantitative analysis of morphometry on Ribb and Gumara watersheds: Implications for soil and water conservation International Soil and Water Conservation Research 7: 150–157

Davies, M. L., & D. A. Cornwell (2017). Introduction to Environmental Engineering 5th Edition. New York: McGraw Hill Education India. Delta, Nigeria. Environmentalist, 28, 85–98.

Deepa, P., Raveen, R., Venkatesan, P., Arivoli, S., & Samuel, T. (2016). Seasonal variations of physicochemical parameters of Korattur Lake, Chennai, Tamil Nadu, India. International Journal of Chemical Studies, 4 (3), 116-123.

Deng C., Zhang G., Liu Y., Nie X., Liu J., & Zhu D. (2021): Advantages and disadvantages of terracing: A comprehensive review. International Soil and Water Conservation Research 9: 344 -359

Deshu Mamo (2004). *An assessment of downstream pollution profile of Sebeta River and its surrounding* (Doctoral dissertation, M. Sc. Thesis, Addis Ababa University).

Dickens, C. W., & Graham, P. M. (2002). The South African Scoring System (SASS) version 5 rapid bioassessment method for rivers. *African Journal of Aquatic Science*, *27*(1), 1-10.

Dietz, M. E., & Clausen, J. C. (2008). Stormwater runoff and export changes with development in a traditional and low impact subdivision. *Journal of environmental management*, *87*(4), 560-566.

Dobson, M., Magana, A., Mathooko, J. M., & Ndegwa, F. K. (2002). Detritivores in Kenyan highland streams: more evidence for the paucity of shredders in the tropics? *Freshwater biology*, *47*(5), 909-919.

Downes, B.J., Barmuta, L.A., Fairweather, P.G., Faith, D.P., Keough, M.J., Lake, P.S., Mapstone, B.D. & Quinn, G.P. (2002) Monitoring Ecological Impacts. Concepts and Practice in Flowing Waters. Cambridge University Press, Cambridge. https://doi.org/10.1017/CBO9780511542015

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z. I., Knowler, D. J., Lévêque, C., & Sullivan, C. A. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological reviews*, *81*(2), 163-182.

Elbag, M.A. (2006) Impact of Surrounding Land Uses on Surface Water Quality. Unpublished MSc Thesis, Worcester Polytechnic Institute, Worcester, MA.

Emere MC, & Narisu CE. 2007. Macroinvertebrates as indicators of water quality of an urbanised stream in Kaduna, Nigeria. Journal of Fisheries International 2: 152–157.

Erdogan H. E., Havlicek E., C. Dazzi , Montanarella L., Van Liedekerke M., _Caj B. Vr_s, Krasilnikov P., Khasankhanova G. & Vargas R. (2021). Soil conservation and sustainable development goals (SDGs) achievement in Europe and central Asia: Which role for the European soil partnership? International Soil and Water Conservation Research 9: 360-369

Favretto, M. A., Orlandin, E., Santos, B., & Piovesan, M. (2014). Insetos aquáticos em um lago artificial no sul do Brasil. Biota Amazônia, 4 (4), 113-116. http://dx.doi.org/10.18561/2179-5746

Gencer, T., & Nilgun, K., 2010. Applications of Various Diversity Indices to Benthic Macroinvertebrate Assemblages in Streams of a Natural Park in Turkey. Hacettepe University, Science Faculty, Biology Department, Hydrobiology Section, Ankara, Turkey.

Gerber A. 2002. Aquatic invertebrates of South African rivers. Pretoria, South Africa: Institute for Water Quality Studies, Department of Water Affairs and Forestry.

Gerth, W. J., Li, J., & Giannico, G. R. (2017). Agricultural land use and macroinvertebrate assemblages in lowland temporary streams of the Willamette Valley, Oregon, USA. *Agriculture, Ecosystems & Environment*, *236*, 154-165.

Getachew Beneberu (2013). The family Chironomidae (Insecta: Diptera) as indicators of environmental stress, and macro-invertebrate based multimetric index development in some selected rivers in Ethiopia. PhD Thesis, School of Graduate Studies, Addis Ababa University.

Getachew Beneberu & Seyoum, Mengistou. (2010). Benthic macroinvertebrate metrics relation to physicochemical parameters in selected rivers of Ethiopia. *Management of shallow waterbodies for improved productivity and people's livelihoods in Ethiopia*, 161-171.

Ghosh, D., & Biswas, J. K. (2015). Biomonitoring macrophytes diversity and abundance for rating aquatic health of an oxbow lake ecosystem in Ganga River Basin. *American Journal of Phytomedicine and Clinical Therapeutics*, *3*(10), 602-621.

Gizachew Teshome, Abebe Getahun, Seyoum Mengistou, Minwyelet Mingist, Fasil Taddese, & Glenn Wilson, G. (2024). Variation in macroinvertebrate assemblages and water quality as a test for different levels of ecological impairment across an Ethiopian highland river system. *Inland Waters*, (just-accepted), 1-44.

Gobiet, A., Kotlarski, S., Beniston, M., Heinrich, G., Rajczak, J., & Stoffel, M. (2013). 21st century climate change in the European Alps - A review. Science of the Total Environment, 493:1138–115

Griffith, M. B., Hill, B., McCormick, H., Kaufmann, R., Herlihy, T. & Selle, A. R. (2005). Comparative application of indices of biotic integrity based on periphyton, macroinvertebrates, and fish to southern R Mountain streams. In: , 117- 136

Gurmessa Tessema., & Agumassie Tesfahun, (2018). Assessment of benthic macroinvertebrate communities in relation to water quality in Teltele stream, Ambo West Showa, Ethiopia. *Journal of Environmental Management and Public Safety*, *7*(3), 043-052.

Hammer MJ. Water and Wastewater Technology. 7th ed. Upper Saddle River: Pearson education; 2011

Jones, N.E. (2011). Electrofishing Rivers: Nearshore Community. Sampling Methodologies for Ontario's Flowing Waters. Ontario Ministry of Natural Resources, Aquatic Research and Development Section, River and Stream Ecology Lab, Aquatic Research Series, 6, 11-23.

Karr, J. R., Fausch, K. D., Angermeier, P. L., Yant, P. R., & Schlosser, I. J. (1986). Assessing biological integrity in running waters. *A method and its rationale. Illinois Natural History Survey, Champaign, Special Publication*, *5*, 1-28.

Karr, J. R. (1999). Defining and measuring river health. *Freshwater biology*, *41*(2), 221-234.

Karr, J. R. (2006). Seven Foundations of Biological Monitoring and Assessment. Biologia Ambientale, 20: 7–18

Karr, J. R., & Chu, E. W. (1997). Biological monitoring and assessment: using multimetric indexes effectively.

Keke, U. N., Omoigberale, M. O., Ezenwa, I., Yusuf, A., Biose, E., Nweke, N., ... & Arimoro, F. O. (2021). Macroinvertebrate communities and physicochemical characteristics along an anthropogenic stress gradient in a southern Nigeria stream: Implications for ecological restoration. *Environmental and Sustainability Indicators*, *12*, 100157.

Keke, U., Arimoro, F.O., Iliya, A.Y., & Ayanwale, V., (2017). Temporal and spatial variability in macroinvertebrate community structure in relation to environmental variables in Gbako River, Niger State, Nigeria Temporal and spatial variability in macroinvertebrate community structure in relation to environmental. Trends Ecol. Evol. 58, 229–240

Koneri, R., Nangoy, M.J., & Saroyo, T.E.T. (2017). Diversity and community composition of dragonfly (Insecta: Odonata) in Tangkoko Nature Reserve North Sulawesi, Indonesia. Bioscience Research, 14 (1), 01–08.

Li, Z., Heino, J., Liu, Z., Meng, X., Chen, X., Ge, Y., & Xie, Z. (2021). The drivers of multiple dimensions of stream macroinvertebrate beta diversity across a large montane landscape. *Limnology and Oceanography*, *66*(1), 226-236.

Lyimo, E. (2012). *Amphibian and benthic macroinvertebrate response to physical and chemical properties of Themi River, Arusha, Tanzania* (Master's thesis).

Magurran, A. E. (2013). Measuring biological diversity. John Wiley and Sons, New York, USA, 285-286pp

Mahala A.(2020). The significance of morphometric analysis to understand the hydrological and morphological characteristics in two different morpho-climatic settings. Applied Water Science 10:33. https://doi.org/10.1007/s13201-019-1118-2

Mandeville, S.M., (2002). Benthic Macroinvertebrates in Freshwaters-Taxa Tolerance Values, Metrics, and Protocols. Soil & Water Conservation Society of Metro Halifax, Nova Scotia, Canada.

Maneechan, W., & Prommi, T. O. (2015). Diversity and distribution of aquatic insects in stream of the Mae Klong watershed, western Thailand. A Journal of Entomology, 2, 1-7. https://doi.org/10.1155/2015/912451

Martínez-Sanz, C., Puente-García, S. M., Rebolledo, E. R., & Jiménez-Prado, P. (2014). Macroinvertebrate richness importance in coastal tropical streams of Esmeraldas (Ecuador) and its use and implications in environmental management procedures. *International Journal of Ecology*, *2014*(1), 253134.

Masese, F. O., Kitaka, N., Kipkemboi, J., Gettel, G. M., Irvine, K., & McClain, M. E. (2014). Litter processing and shredder distribution as indicators of riparian and catchment influences on ecological health of tropical streams. Ecological Indicators, 46, 23-37.

Masese, F. O., Omukoto, J. O., & Nyakeya, K. (2013). Biomonitoring as a prerequisite for sustainable water resources: a review of current status, opportunities and challenges to scaling up in East Africa. *Ecohydrology & Hydrobiology*, *13*(3), 173-191.

Masikini, R., Kaaya, L. T., & Chicharo, L. (2018). Evaluation of ecohydrological variables in relation to spatial and temporal variability of macroinvertebrate assemblages along the Zigi River–Tanzania. *Ecohydrology & Hydrobiology*, *18*(2), 130-141

Mayers, J. (2009). *Water Ecosystem Services and Poverty Under Climate Change: Key Issues and Research Priorities: Report of a Scoping Exercise to Help Develop a Research Programme for the UK Department for International Development* (No. 17).

Mekonen Teferi, Tsegazeabe Hadush Haileselasie, Tsehaye Asmelash , Haile G. Selasie, Alem G, Amare S, Weldegerima K, Tesfay S, Kiros & S, Gebru (2013). Influence of water quality on the diversity and distribution of macroinvertebrates in highland stream, Northern Ethiopia. Scholarly Journal of Agricultural Science, 2(2): 17- 25. 30.

Melaku Getachew, Argaw Ambelu, Seid Tiku, Worku Legesse, Aynalem Adugna, & Kloos, H. (2012). Ecological assessment of Cheffa Wetland in the Borkena Valley, northeast Ethiopia: Macroinvertebrate and bird communities. *Ecological Indicators*, *15*(1), 63-71.

Mengistu Asmamaw.,Seid Teku Mereta., & Aragaw Ambelu,. (2021). Response of macroinvertebrates to changes in stream flow and habitat conditions in Dinki watershed, central highlands of Ethiopia. *Ecological Indicators*, *133*, 108448.

Mengistu Welemariam., Fassil Kebede ., Boobe Bedadi, B., & Emiru Birhane,. (2018). The effect of community-based soil and water conservation practices on abundance and diversity of soil macroinvertebrates in the northern highlands of Ethiopia. *Agronomy*, *8*(4), 56.

M'Erimba, C. M., Mathooko, J. M., & Leichtfried, M. (2006). Variations in coarse particulate organic matter in relation to anthropogenic trampling on the banks of the Njoro River, Kenya. *African Journal of Ecology*, *44*(2), 282-285.

Merritt, R. W., Cummins, K. W., & Berg, M. B. (2017). Trophic relationships of macroinvertebrates. In *Methods in stream ecology, volume 1* (pp. 413-433). Academic Press.

Minnesota Pollution Control Agency (2008). Turbidity: Description, Impact on Water Quality, Sources, Measures (available at https://www.pca.state.mn.us/sites/default/files/wqiw3-21.pdf).

Morphin-Kani, K., & Murugesan, A.G. (2014). Assessment of River Water Quality Using Macroinvertebrate Organisms as Pollution Indicators of Tamirabarani River Basin, Tamil Nadu, India. International journal of Environmental Protection, 4, 1-14.

Negero, Aschalew Lakew , Prabha D & Wolfram G (2017). Macro Invertebrate Communities in the spring and Stream Sites of Upper Awash River at Chilimo, Ethiopia. Innovative Techniques in Agriculture, 1(3): 141-151

Odume, O. N., Muller, W. J., Arimoro, F. O., & Palmer, C. G. (2012). The impact of water quality deterioration on macroinvertebrate communities in the Swartkops River, South Africa: a multimetric approach. *African Journal of Aquatic Science*, *37*(2), 191-200.

Oliveira, A. L. H., & Nessimian, J. L. (2010). Spatial distribution and functional feeding groups of aquatic insect communities in Serra da Bocaina streams, southeastern Brazil. Acta Limnologica Brasiliensia, 22 (4), 424-441. https://doi.org/10.4322/actalb.2011.007.

Parker, L.M., Ross, P.M., O'Connor, W.A., Pörtner, H.O., Scanes, E., & Wright, J.M. (2013). Predicting the Response of Molluscs to the Impact of Ocean Acidification. Biology, 2, 651–692. https://doi.org/10.3390/biology2020651

Poikane, S., Johnson, R. K., Sandin, L., Schartau, A. K., Solimini, A. G., Urbanič, G., ... & Böhmer, J. (2016). Benthic macroinvertebrates in lake ecological assessment: A review of methods, intercalibration and practical recommendations. *Science of the total environment*, *543*, 123-134.

Prommi, T., & Payakka, A. (2015). Aquatic insect biodiversity and water quality parameters of streams in Northern Thailand. *Sains Malaysiana*, *44*(5), 707-717.

Rajwa-Kuligiewicz, A., Bialik, R. J., & Rowiński, P. M. (2015). Dissolved oxygen and water temperature dynamics in lowland rivers over various timescales. *Journal of Hydrology and Hydromechanics*, *63*(4), 353-363. https://doi.org/10.1515/johh-2015-0041

Rosser, Z. C., & Pearson, R. G. (2018). Hydrology, hydraulics and scale influence macroinvertebrate responses to disturbance in tropical streams. *Journal of freshwater ecology*, *33*(1), 1-17.

Roy, A., Rosemond, A. D., Leigh, D. S., Paul, M. J., & Wallace, J. B. (2001). Effects of changing land use on macroinvertebrate integrity: identifying indicators of water quality impairment. Georgia Institute of Technology.

Sánchez-Fernández, D., Abellán, P., Mellado, A., Velasco, J., & Millán, A. (2006). Are water beetles good indicators of biodiversity in Mediterranean aquatic ecosystems? The case of the Segura River basin (SE Spain). *Biodiversity & Conservation*, *15*, 4507-4520.

Sarkar, S. K., Bhattacharya, B., Debnath, S., Bandopadhaya, G., & Giri, S. (2002). Heavy metals in biota from Sundarban Wetland Ecosystem, India: Implications to monitoring and environmental assessment. *Aquatic Ecosystem Health & Management*, *5*(4), 467-472.

Selemani, J. R., Zhang, J., Muzuka, A. N. N., Njau, K. N., Zhang, G., Mzuza, M. K., & Maggid, A. (2018). Nutrients' distribution and their impact on Pangani River Basin's ecosystem–Tanzania. *Environmental technology*, *39*(6), 702-716.

Shelly, S. Y., Mirza, Z. B., & Bashir, S. (2011). Comparative ecological study of aquatic macroinvertebrates of Mangla Dam and Chashma Barrage wetland areas.

Singh V, Singh UC (2011) Basin morphometry of Maingra River, district Gwalior, Madhya Pradesh, India. Int J Geomat Geosci 1(4):891–902

Sisay Misganaw Tamiru., Seyoum Leta Asfaw., & Seyoum Mengistu Yilma, (2017). Correlation study of some physico-chemical parameters and benthic macroinvertebrates metrics on the ecological impacts of floriculture industries along Wedecha River, Debrezeit, Ethiopia.

Solomon Akalu, Seyoum Mengistou & Seyoum Leta (2011). Assessing human impacts on the Greater Akaki River, Ethiopia using macroinvertebrates. SINET: Ethiop.J.Sci. 34, 89-98.

Solomon Tesfay., Mekonen Teferi., Shicur Aragaw., &Weleterifael Abrha, (2016). Assessment of Macroinvertebrate Communities in Enda Gabr Stream, in Mekelle, Northern Ethiopia. Journal of Natural Sciences Research, 6(19):115:125.

Strayer, D. L. (2006). Challenges for freshwater invertebrate conservation. *Journal of the North American Benthological Society*, *25*(2), 271-287.

Stubbington, R., Bogan, M. T., Bonada, N., Boulton, A. J., Datry, T., Leigh, C., & Vander Vorste, R. (2017). The biota of intermittent rivers and ephemeral streams: aquatic invertebrates. In *Intermittent rivers and ephemeral streams* (pp. 217-243). Academic Press.

Sultana, R., & Kala, D. S. (2012). Water body quality analysis by benthic macro invertebrates. *International Journal of Pharmacy and Biological Sciences*, *2*(1), 269-279.

Uwadiae, R. E. (2009). *An ecological study on the macrobenthic invertebrate community of Epe lagoon, Lagos* (Doctoral dissertation, University of Lagos (Nigeria)).

Uwadiae, R.F. (2010). Macroinvertebrates functional feeding groups as indices of biological assessment in a tropical aquatic ecosystem: implications for ecosystem functions. New York Science Journal, 3, 8-17.

Vannote, R. L., Minshall, G. W., Cummins, K. W., Sedell, J. R., & Cushing, C. E. (1980). The river continuum concept. *Canadian journal of fisheries and aquatic sciences*, *37*(1), 130-137.

Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., ... & Davies, P. M. (2010). Global threats to human water security and river biodiversity. *nature*, *467*(7315), 555-561.

Walsh, C. J. (2006). Biological indicators of stream health using macroinvertebrate assemblage composition: a comparison of sensitivity to an urban gradient. *Marine and Freshwater Research*, *57*(1), 37-47.

Wantzen, K. M., Wagner, R., Suetfeld, R., & Junk, W. J. (2002). How do plant-herbivore interactions of trees influence coarse detritus processing by shredders in aquatic ecosystems of different latitudes? *Internationale Vereinigung für theoretische und angewandte Limnologie: Verhandlungen*, *28*(2), 815-821.

WHO (2008). Guidelines for drinking water quality. World Health Organization, Recommendations Geneva, Switzerland.

WHO (2011). Nitrate and nitrite in drinking-water. Background document for development of WHO guidelines for drinking-water quality, WHO/SDE/WSH/07.01/16/Rev/1.

Yared Worku., & Aragaw Ambelu, (2018). Evaluation of the Impacts of Eucalyptus Plantation on Macro-Invertebrate Assemblage in the Aquatic Environment, South Western Ethiopia. *Int. J. Environ. Sci. Nat Res*, *9*(1), 001-012.

Zang, C., Huang, S., Wu, M., Du, S., Scholz, M., Gao, F., ... & Dong, Y. (2011). Comparison of relationships between pH, dissolved oxygen and chlorophyll a for aquaculture and non-aquaculture waters. *Water, Air, & Soil Pollution*, *219*, 157-174.

8. Annexes

Annexes 1: Physiochemical data

Sites	Do(mg/l)	TEMP	PH	EC(µcm)	TDS (mg/l)	depth(cm)	width(m)	dicharge(m/s)
1	8.2	27	8.9	331	171.4	5	12	27.38
	7.89	26.3	8.7	313	156.5	10	8	28.92
	7.92	26	8.5	309	154.1	12	12	27.4
2	7.08	27.3	8.4	324	175	8	9	47.29
	6.71	27.4	8.7	327	179.5	6	9	29.1
	6.98	27.2	8.6	325	178.2	6	7	27.97
3	8.5	23.6	8.5	428	215	25	9	43
	8.2	25.5	8.6	429	215	15	9	31
	7.98	25.1	8.8	431	214	10	7	32
4	6.72	27.3	8.7	440	225	10	8	24
	7.12	27	8.3	444	223	8	11	34
	6.89	27.1	8.9	454	218	9	13	49
5	6.41	26.5	8.9	388	192.6	5	27	8
	7.08	28.9	8.5	387	192.6	12	25	6
	7.27	27.9	8.7	385	192.6	10	26	5
6	7.51	27.5	9.1	468	234	15	10.5	8
	7.46	26.4	9.3	481	240	15	9.4	9
	7.49	26.6	8.9	468	234	15	10	9
7	6.64	26.6	8.7	356	178	16	8	14
	6.52	24.4	9.2	356	177.6	15	7	19
	6.34	24	8.9	355	177.6	12	7	14
8	5.8	23.4	8.3	297	143.1	30	16	62
	5.96	22.9	8.6	285	142.7	40	18	65
	6.22	23.1	8.4	285	143.1	45	20	67

Annexes 3: habitat quality assessment scoring data

Habitat Parameter	Site 1	Site2	site 3	site 4	site 5	site 6	site 7	site 8
Available instream cover	3	2	4	3	3	4	2	3
Boottom Substrate	4	4	3	4	2	3	2	1
Dimension of largest pool	2	2	2	3	4	3	2	4
Number of riffles	3	3	3	4	3	3	3	2
water level	2	2	3	3	3	3	2	4
Channel sinuosity	3	3	4	3	3	3	3	3
Bank stability	4	1	3	3	3	3	1	3
Riparian buffer vegetation	3	1	3	2	3	3	2	3
Aeshthetics of reach	3	1	3	2	2	2	1	4
Total	27	19	28	27	26	27	18	27

Annexes 4: Macroinvertebrates abundance in sampling sites of the study area and their functional feeding groups

Order	Family	sampling sites								FFG
		1	2	3	4	5	6	7	8	
Coloptera	Dytiscidae	42	119	112	18	50	243	44	8	Predator
	Elmidae	1	0	0	1	0	0	0	0	Collecrs-Gatherers
	Gyrinidae	0	2	0	0	0	0	0	0	Predator
	Hydrophilllidae	0	2	0	9	0	0	3	0	Predator(L) /Collectors-gatherers(A)
Mollusca	Planorbidae	0	6	61	18	2	2	1	5	Scraper
	Sphaeridae	0	1	0	0	0	0	0	0	Filtering collectors
	Lymnaeidae	0	0	0	8	0	0	2	0	Scrapers
Diptera	Tabanidae	0	3	0	2	0	0	0	0	Predator
	Syrphidae	0	4	0	0	0	0	0	0	Filtering collectors
	Psychodidae	0	0	1	0	0	0	0	0	Collectors-Gatherers
Ephemeroptera	Baetidae	3	1	0	0	0	0	0	0	Collectors-Gatherers
Annelida	Hirudinae	0	0	0	0	3	0	0	0	Predator
Odonata	Aeshnidae	0	0	0	0	2	3	1	0	Predator
	Libellulidae	0	0	0	0	2	0	0	0	Predator
	Gomphidae	0	0	0	0	1	4	3	0	Predator
	Coenagrionidae	0	0	0	0	0	2	1	0	Predator
	Lestidae	0	0	0	0	0	6	8	0	Predator
Hemiptera	Naucoridae	0	3	0	0	0	0	0	2	Predator
	Notonectidae	0	0	0	0	0	0	0	6	Predator
	Corixidae	0	0	0	0	0	4	0	6	Scrapers
Total		46	141	174	56	60	264	63	27	

Annexes 5: photos showing sampling sites of the river

Sampling station one

Sampling station two

Sampling station three

Cattles at sampling station four

Sampling station five

Sampling site six

Sampling site seven

Annexes 6: field sampling activities and laboratory analysis

In situ physicochemical measurement

Macroinvettbartes sampling

Macroinvertebrates sorting and identification at Bahir Dar university Limnology laboratory

Annexes 6: Some of identified macroinvertebrates

Dysticidae(diving betteles)

Gomphidae (Clubtails)

Coenagrinidae (Sprites and blues)

Corixidae (Water boatmen)

Baetidae (Small minnow flies)

Planorbidae (Orb snails)