ADDIS ABABA UNIVERSITY SCHOOL OF GRADUATE STUDIES

ADVANCING BIOASSESSMENT OF WATER QUALITY IN WADEABLE RIVERS AND STREAMS IN ETHIOPIA

By

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Advancing BIOASSESSMENT of Water Quality in Wadeable Rivers and Streams in Ethiopia

By

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After consideration of the dissertation of Melaku Getachew Tadesse entitled **“Advancing Bioassessment of water quality in wadeable rivers and streams in Ethiopia”**, members of the doctoral advisory committee approved by their signatures below the dissertation is ready for defense subject to the following criteria:

Criteria	Acceptable	Unacceptable
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Declaration

I, the undersigned, declare that this dissertation is my original work; it has not been presented in other universities, colleges, or institutes for a degree or other purpose. All sources of the materials used in this dissertation have been fully acknowledged.

Name: Melaku Getachew signature: date: 25 December 2023

This work has been done under my supervision.

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Dedication

I dedicate this dissertation to my parents, Senait Birhane and Getachew Tadesse, whose love and support have remained with me. This piece is also dedicated to my beloved brother, Ameha Getachew.

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List of abbreviations

GWH	Giga Watt Hour
BMC	Billion meter cube
APHA	American Public Health Association
DO	Dissolved Oxygen
DPSIR	Driver-Pressure-State-Impact-Response
EC	Electrical Conductivity
MDS	Non-metric Multidimensional Scaling
TP	Total Phosphorous
USEPA	United States Environmental Protection Agency
APHA	American Public Health Association
ASPT	Average Score per Taxon
BOD	Biochemical Oxygen Demand
BMWP	Biological Monitoring Working Party
CCA	Canonical Correspondence Analysis
DCA	Detrended Correspondence Analysis
EPT	Ephemeroptera, Plecoptera, Trichoptera
FBI	Family Biotic Index
UCD	University College Dublin
TDS	Total Dissolved Solids
TS	Total Solids
TSS	Total Suspended Solids

List of published and unpublished papers

This dissertation is prepared based on the following three published articles and one unpublished but submitted manuscript as mentioned below.

Published articles

1. Getachew, M., Mulat, W. L., Mereta, S. T., Gebrie, G. S., & Kelly-Quinn, M. (2020). Challenges for water quality protection in the greater metropolitan area of Addis Ababa and the upper Awash Basin, Ethiopia – time to take stock. Environmental Reviews, 29(1), 87-99. DOI: 10.1139/er-2020-0042

2. Getachew, M., Mulat, W. L., Mereta, S. T., Gebrie, G. S., & Kelly-Quinn, M. (2022). Refining benthic macroinvertebrate kick sampling protocol for wadeable rivers and streams in Ethiopia. Environmental monitoring and assessment, 194(3), 196. DOI: 10.1007/s10661-021-09594-x

3. Getachew M, Mereta ST, Gebrie GS, Mulat WL, Kelly-Quinn M. Impacts of the Koka hydropower dam on macroinvertebrate assemblages in the Awash River Basin in Ethiopia. *J. Limnol.2023;82:2153*. DOI: 10.4081/jlimnol.2023.2153.

Unpublished manuscript

1. Getachew M, Mereta ST, Gebrie GS, Mulat WL, Kelly-Quinn M. Factors Affecting Macroinvertebrate Community Composition in Three River Basins of Ethiopia: A Variation Partitioning Analysis (Under publication in the Journal of Limnology).

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ADVANCING BIOASSESSMENT OF WATER QUALITY IN WADEABLE RIVERS AND STREAMS IN ETHIOPIA

Chapter 1: General Introduction

Freshwater ecosystem conservation is generally weak and declining worldwide (Dudgeon *et al.*, 2006; Butchart *et al.*, 2010). Among others, rivers and wetlands are the most threatened freshwater ecosystems because of various anthropogenic actions that modify the ecosystems' integrity and functions (Ravera, 2001; Malmqvist and Rundle, 2002; Assessment, 2005; Board, 2005; Sendzimir and Schmutz, 2018). A river is an open system with strong directionality and strong interactions with its drainage basin (Timmerman *et al.*, 1996; Beyene *et al.*, 2009a). Rivers are complex and dynamic ecosystems whose abiotic and biotic components change slowly and continuously from their source to the mouth (Gopal and Chauhan, 2013). According to Ward (1989), there are four dimensions in environmental relationships between the river and surrounding landscape, namely longitudinal, lateral, and vertical gradients and a temporal dimension.

Distinct longitudinal gradients from the source to the downstream river mouth area result from dynamics in hydrology and morphology, together with spatial differences in geology, relief, and soil in the catchment area (Ward, 1989; Wetzel, 2001), for example, increasing river bed width and depth, decreasing stream velocity and substrate grain size, and increasing enrichment by nutrients in the downstream direction (Ward, 1989; Rosgen, 1994; Newbury and Bates, 2007). From their origin to the river mouth, these longitudinal gradients of the abiotic components determine characteristics that lead to an ecological zonation of communities, both structural and functional (Vannote and Sweeney, 1980; Ward, 1989).

The transverse or lateral gradient in natural streams and rivers can be characterised in a way that the aquatic zone of a riverine ecosystem is interlinked with the riparian zone and the terrestrial zone/floodplains (Ward, 1989). Abiotic determining factors like erosion and sedimentation patterns and stream velocity differ greatly between stream beds, banks, floodplains, inner and outer curves, and others (Ward, 1989; Baillie and Neary, 2015). A third gradient/dimension is the vertical relationship between the river sediment and the underlying groundwater system (Ward, 1989; Rosgen, 1994; Newbury and Bates, 2007). Lastly, a temporal dimension can be considered in the duration of certain natural events like floods and other changes in water levels (Ward, 1989; Dallas, 2002). The aforementioned gradients may lead to differences in habitats in and along natural rivers (Ward, 1989; Newbury and Bates, 2007; Pander and Geist, 2013).

Interactions among geology, climate, and geomorphic elements determine a river system's overall characteristics (Gopal and Chauhan, 2013). According to Rutherfurd *et al*. (2000), there are five major components of the river ecosystem, namely: physical habitat (fluvial geomorphology), flow regime, water quality, biological diversity, riparian zone (including the floodplain), and physical habitat (fluvial geomorphology).

1.1. Rivers and concepts of river health

A river or stream is a natural flow of running water that follows a well-defined, permanent path, usually within a valley (Dickens *et al.*, 2021). Although some rivers are larger than others, size is not a distinguishing factor. The origin of a river or stream is called its source (Heino *et al.*, 2003; Dickens et al., 2021). If its source consists of many smaller streams coming from the same region, they are called headwaters (Dickens et al., 2021). Its channel is the path along which it flows, and its banks are its boundaries, the sloping land along each edge between which the water flows. The point where a stream or river empties into a lake, a larger river, or an ocean is its mouth. When one stream or river flows into another, usually larger stream or river, and adds its flow, it is considered a tributary of the larger river.

Many tributaries make up a river system (Stoddard *et al.*, 2005; Dickens et al., 2021). The following definition for aquatic ecosystems is adapted from the Millennium Ecosystem Assessment (MEA, 2005): “A water-related ecosystem is a dynamic complex of plant, animal, and microorganism communities and the nonliving environment dominated by the presence of flowing (lotic) or still (lentic) water, interacting as a functional unit.”

The concept of what constitutes health is inherent in all people, and this also holds true for ecosystems. Health is a concept that comprehensively integrates a wide range of difficulties, just as the status of any other system or human being. "Aquatic system is the ability of that environment to support and maintain key ecological processes and a community of organisms with a species composition, diversity, and functional organization as comparable as possible to that of undisturbed habitats within the region" (Schofield and Davies, 1996). According to Burkhard *et al.* (2008), the concept of ecosystem health integrates natural conditions and knowledge about human impacts on the environment to support sustainable resource management. The 1972 US Water Pollution Control Act Amendments, now called the Clean Water Act (CWA, 1972) set a standard for defining river health. According to this act, river health is the maintenance of the chemical, physical, and biological integrity of the water resources.

Integrity is a concept that refers to a condition in which the natural structure and function of ecosystems are maintained (Karr, 1999). This definition describes aspects of a very complex system in which organisms interact with their surroundings (Karr, 1999). It has been documented that bioassessment of the health of rivers is required to determine how climate change and anthropogenic activities influence aquatic ecosystems (USGS, 2013). In addition, monitoring helps ensure that stream systems can function and will be able to provide ecosystem services for future generations (USGS, 2013), which has a direct or indirect link with internationally agreed sustainable development goals and targets (Le Blanc, 2015).

The recent emphasis on surface water protection and management has created an urgent need to develop assessment tools, establish and monitor human impacts on water resource ecosystems, prioritize conservation and rehabilitation actions, and monitor the effects of these actions (Barbour *et al.*, 1999; Cole, 2006). Physical, chemical, biological, and hydromorphological factors affect the ecological integrity of surface water ecosystems (Ollis *et al.*, 2006) such as streams and rivers. Scholars (Karr, 1999; Karr and Chu, 2000), grouped these factors into five classes, such as water quality parameters, flow regime conditions, habitat structures, biotic interactions, and energy sources, as illustrated in Figure 1-1. However, it is impractical and probably impossible to measure and monitor all the factors contributing to the ecological integrity of a river system (Ollis et al., 2006). It is, therefore, recommended to generally use a limited number of indicators representative of anthropogenic impacts (Ollis et al., 2006) to determine the ecological status of a river.

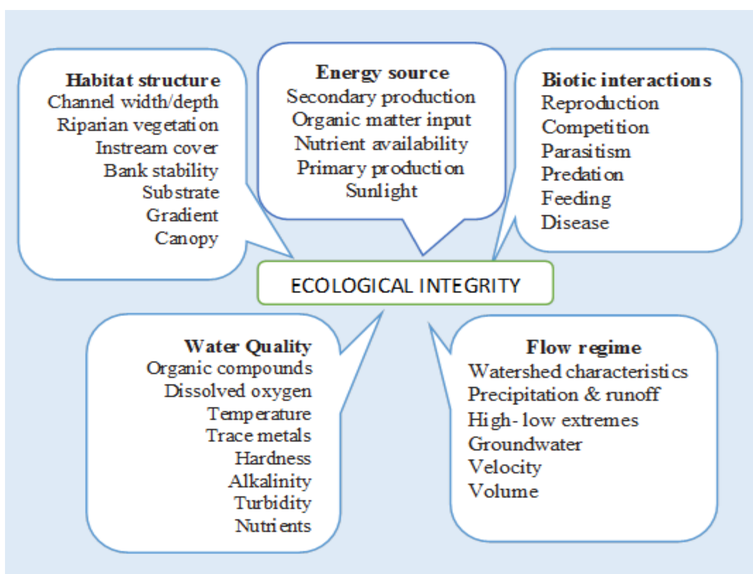

Figure 1-1: Factors affecting the ecological integrity of river ecosystems: (Karr et al., 1986; Ollis et al., 2006)

The biological, physicochemical, hydrological, and morphological components are generally the preferred options for inclusion in representative frameworks as river health assessment indicators (Dickens et al., 2021).

1.2. River Health indicators

Indicators are measures of features that provide clues to matters of greater significance or enable the perception of trends or phenomena that may not be immediately detectable (Dickens et al., 2021). Thus, their significance extends beyond what is actually measured to larger phenomena of interest (Hamouda *et al.*, 2009; Dickens et al., 2021). Therefore, ecological indicators provide insights into ecosystem conditions (Dickens et al., 2021). For example, measuring the concentrations of N or P in a waterbody not only provides information on their current levels but, when elevated, can signal a change in trophic state or eutrophication, which has a cascade of effects on the state of biological communities, water quality, and even human health. Thus, the choice of indicators used by an assessment framework is crucial to its success (Burkhard et al., 2008; Dickens et al., 2021). They are central to ensuring the robustness of the information obtained, while flexibility in the choice of indicators is key to enabling the flexibility and scalability of the frameworks themselves, i.e., their application in different contexts and at variable scales. There are thus several conditions to consider when choosing indicators (Burkhard et al., 2008).

First, consider the types of indicators to use. The OECD (1993) defined three categories of indicators, namely measures of environmental pressures or drivers, state, and societal responses (i.e., DSR or PSR). This is further refined by Smeets and Weterings (1999) into five indicator classes, known as the DPSIR framework (Figure 2-1), used by the European Environment Agency. This distinguishes between indicators of driving forces (e.g., indirect social and economic developments that result in pressures on the environment), pressures (forces acting on the environment, e.g., pollution discharges, land-use changes, etc.), state (changes in the environment, e.g., the conditions of biological communities, water quality, habitat, etc.), impacts (of the environmental state on other features, e.g., human health, ecological systems, or materials), and responses (societal reactions to changes in the environmental state, e.g., policy change)

(Kristensen, 2004; Hamouda et al., 2009; Dickens et al., 2021). Most of the ecosystem health frameworks include a combination of state and pressure indicators, where state indicators form the core of 'freshwater ecosystem health' assessment, as this is fundamentally a measure of 'state'. Most frameworks also include some aspects of pressure indicators, as these enable the interpretation of changes in the ecological state. Only the Incident Threat Indices (ITI) focus entirely on the drivers and pressures, as their objective was to determine the threat to water resources (not measure the state) (Dickens et al., 2021). Some go further to show the link between changes in the state and further impacts, such as on human health, recreation, or ecosystem services. Response indicators are the form of governance and stakeholder value indicators in an attempt to show the role that government and societal attitudes play in influencing the ecological state. Therefore, the choice of indicator class(es) used by a framework is heavily influenced by its definition of "freshwater health," which itself depends on the management objectives of the assessment, so the sound consideration of these aspects during framework development is fundamental to influencing the choice of indicators (Dickens et al., 2021).

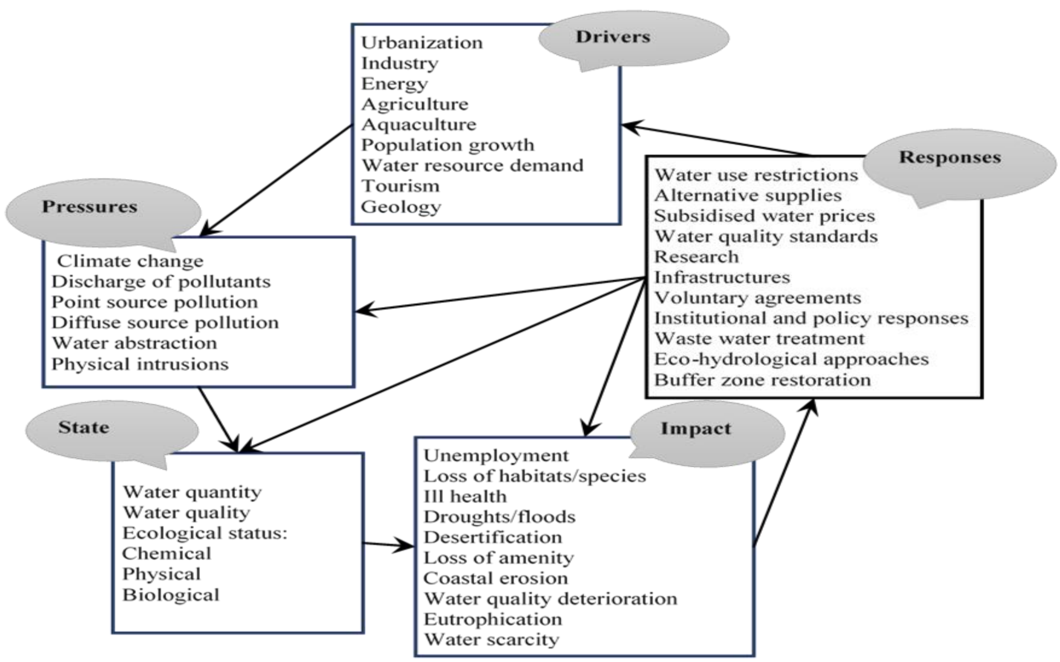

Figure 2-1:A generic DPSIR framework of environmental indicators, consisting of driving forces, pressures, states, impacts, and responses for water (Kristensen, 2004; Hamouda et al., 2009; Dickens et al., 2021)

1.3. Classification of river systems

Flowing water, the main agent of the erosion process and the creation of channel paths and physical habitat in a river system (Padmalal and Maya, 2014), creates, destroys, and re-creates the landforms, channel patterns, and habitat of the biotic community (Kamboj *et al.*, 2020). The river system can be classified according to sources, stream order, and age of the rivers, geomorphological features, biotic zone classification, and whitewater classification techniques. The following characteristics are generally used to classify the rivers (Figure 3-1).

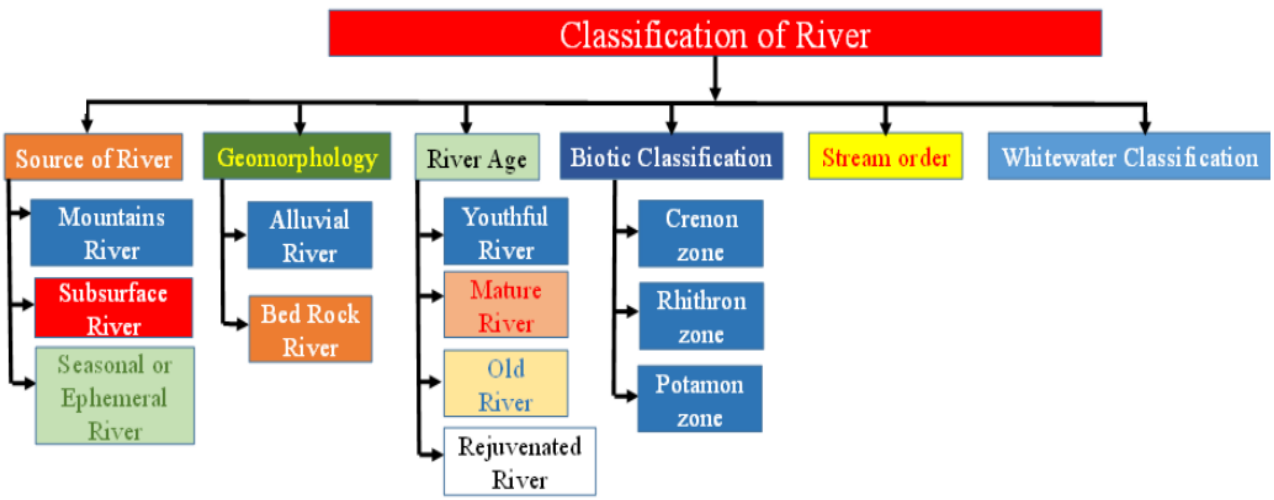

Figure 3-1: Classification of river systems.Source: Kamboj et al. (2020)

1.3.1. On the basis of source

Sources refer to the beginning point or path of the river, also called a course. The ending point of the river is called the mouth. The river is divided into two- parts on the basis of water flow, such as upstream (direction of flowing water towards the source) and downstream (direction towards the mouth of the river). On the basis of source, rivers are classified into three types.

a. Mountain Rivers: The Mountain Rivers refer to those rivers that start from the mountains, also called glacier-fed rivers and perennial rivers.

b. Subsurface streams: The subsurface streams refer to the river that flows underground in caves, glaciers, and ice sheets. These rivers are mostly found in the limestone-geological formation region.

c. Seasonal or ephemeral rivers: seasonal or ephemeral rivers depending on the precipitation or rainfall of that area.

1.3.2. On the basis of geomorphology

Around the globe, rivers are essential for moving loads of sand, gravel, and boulders, from one location to another, which are among the geomorphology components that make up the riverbed (Kamboj et al., 2020). Based on features of their geomorphology, the rivers are classified as follows:

a. Alluvial rivers: The alluvial rivers refer to floodplain areas that are created, destroyed, and recreated by weakly consolidated sediments. Channel pattern refers to the path of the watercourse, and floodplain area refers to the nearby area of the river, which consists of sediments (Ward and Stanford, 1995). On the basis of the sinuosity index calculated as: $\text{Sinuosity index} = (\text{Length of water course})/(\text{Straight length of the river})$ (Miall, 1985), alluvial channel is classified into three types such as straight channel (sinuosity index value is less than 1.05), sinuous channel (sinuosity index value lies between 1.05 and 1.50), and meandering channel (sinuosity index is greater than 1.50). The meandering channel is composed of riffles and pools and provides habitat for aquatic organisms (Kamboj et al., 2020). On the basis of geomorphological characteristics, alluvial rivers are divided into two types, such as braided river channels and anabranching river channels. The braided river channel indicates the separation of the main channel in many channels due to the permanent and temporary bars and islands. The formation of a braided channel in a river system is due to the high sediment load, high water flow, high meandering pattern, weak bank, and steeper slope (Schumm and Khan, 1972). Similarly, the Anabranching river channel mostly looks like a braided river channel. It consists of a number of channels that are separated by vegetation; there is an alluvial island in the active channel, not in the floodplain area (Nanson and Knighton, 1996). The formation of these types of rivers is due to the low and high water flow and banks that are stable sediments.

b. Bedrock Rivers: The formation of bedrock rivers occurs when the flow of water passes through the new level of sediments where bedrock is below. These types of rivers are commonly found in upland and mountainous rivers where the earth's surface is shifting upwards. The formation of these types of rivers is due to their alluvial nature, weak sediments and banks, and the high flow of water.

1.3.3. On the basis of river age

The study of river age, also known as the chronological classification of rivers, is based on a study of the changing river path and pattern of erosion. On the basis of a cycle of erosion developed by William Morris Davis, the rivers are classified into four types: mature rivers, old rivers, rejuvenated rivers, and young channels. This theory is rejected by the geomorphologist because it does not produce a testable hypothesis, and this theory is nonscientific (Castree *et al.*, 2005).

1.3.4. On the basis of biotic zones

Rivers are an example of a lotic ecosystem. It refers to the flowing water in a landscape, including the biotic community such as producers (the plankton community), consumers (fishes), and decomposers (microorganisms), as well as abiotic factors (Angelier, 2019). In a river system, the zones are divided by the river bed gradient, the velocity of the current, and other abiotic factors such as temperature. Illies and Botosaneanu (1963) divided the rivers into three primary zones according to the presence of abiotic and biotic factors:

a) Crenon zone: an area that is near the source of the river or where the river gets its start. The crenon zone is divided into two zones, namely the eucrenon zone and the hypocrenon zone. The eucrenon zone is the spring zone, and the hypocrenon zone is the headstream zone. In these areas, the temperature of the water, oxygen level, and flow of water are low.

b) Rhithron zone: A rhithron zone is the upstream area of a river characterized by the presence of the following conditions or factors: steep gradients, narrow and shallow rapids, riffles, and deeper, flatter pools. In this zone, the flow of current is higher in riffles than in pools. In riffles, vegetation is attached to boulders, stones, and rocks. In pools, vegetation is attached to the fine material and some other rotting vegetation. In the rhithron zone, the temperature is low and the dissolved oxygen is high. In the Rhithron Zone, the main biotic community consists of plankton, periphyton, nekton, and a variety of benthos. The rhithron zone is absent in tropical rivers and found to be relatively lengthy in temperate rivers.

c) Potamon zone: A potamon zone is the downstream area of a river with a flat gradient, lower speed, and lower oxygen content; the temperature of the water is warmer than the rhithron zone; and the river bed is sandy. In the Potamon zone, the presence of habitats depends on the meanders of the channel as well as the main channel and floodplain areas. The Potamon zone is more complex and differs environmentally from the Rhithron zone based on the following aspects: geomorphology of the channel, submerged vegetation, quality of water, biotic community, and bacterial density.

1.3.5. On the basis of stream order

Stream order is a method based on a number that shows how a number of tributaries link to a stream network. Most of the rivers are first- and second-order rivers. The concept of the stream order method was proposed by a scientist, namely Strahler (Hughes *et al.*, 2011; Kamboj et al., 2020). Strahler describes stream order as the linking of tributaries in a stream network. According to this method, all the rivers start in the first order (Kamboj et al., 2020). When two first-order tributaries are linked together with a point after this, the stream order is two. If the two-order stream is linked with another two-order stream, then it creates the third-order stream (Figure 4-1). This is the most common method of stream classification based on their orders (Tarboton *et al.*, 1991).

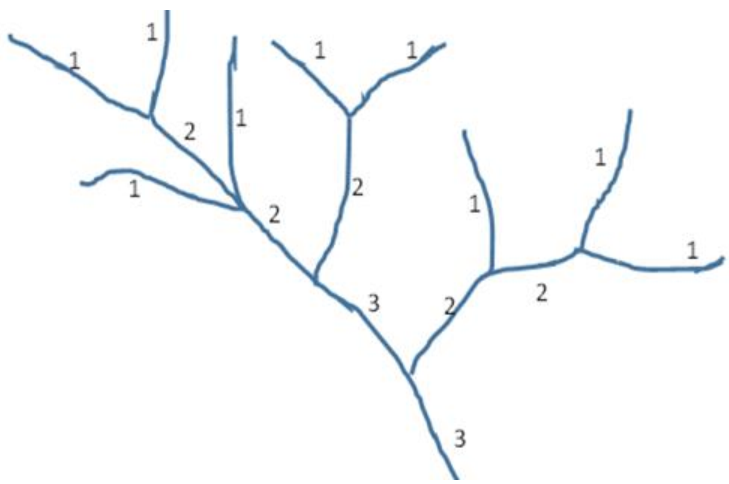

Figure 4-1: Diagrammatic map of Strahler stream

1.3.6. On the basis of whitewater classification

Whitewater classification is the classification method based on the navigation challenges, recreational activities, and healthy ecosystem of a freshwater river (Table 1-1). The International Scale of River Difficulty divided the whitewater classification into six classes on the basis of some conditions. Class I is the easiest, and Class VI is the hardest (Walbridge and Singleton, 2005).

Table 1-1: Classes of whitewater classification: (ISRD, 2005)

Class name	Water flow/Swimming condition for peddlers	Conditions according to rapids and waves
Class I	Easy	Fast flow, small waves, easy to swim
Class II	Novice	Rivers are wide, waves are medium-sized,
Class III	Intermediate	Waves are irregular and intermediate the difficulty level
Class IV	Advanced	Rapids are intense and powerful but predictable, unavoidable waves, a good skill for swimming, risk of a swimmer is moderate to high
Class V	Expert	Rapids are extremely long, violent and obstructed; waves are large, unavoidable and hole or steep; scouting is recommended but swimming is dangerous, a good skill for swimming
Class VI	Extreme and exploratory rapids	Rapids/ run in this class are rarely attempted, very difficult and dangerous, swimming is done for expert only with good equipment

1.4. Water resources in Ethiopia

The Ministry of Water and Energy (MoWE, 2013) states that dry land makes up 99.3% of Ethiopia's overall land area, with water bodies covering just 0.7% of the country's land area. Figure 5-1 shows that there are twelve large lakes, twelve important basins, and a variety of smaller water bodies in the country. Nevertheless, three of the larger basins are dry since there is no stream flow in them. Notwithstanding the need for a more recent and targeted study, the nation's surface water potential is 122 billion cubic meters (BCM), as calculated and estimated in a number of integrated river basin master plans (Awulachew *et al.*, 2007). The majority of Ethiopia's projected annual stream flow flows transboundary, meaning that 97% of it exits the country into neighboring states (Berhanu *et al.*, 2014).

Ethiopia is designated the “Water Towers of Northeast Africa” due to the existence of several river resources that drain from the highlands to the lowlands and neighboring countries (Alemayehu, 2006; Awulachew et al., 2007; Belete, 2013). However, when we calculate total surface and ground water volume (152 BMC) with the current population size, we have less than 1500 cubic meters per person per year, which is by far less than the global water availability that is 7000 cubic meters per person per year (Weiss, 2006). A country with a per capita water availability of 1700 cubic meters per person per year is classified as water-stressed, and less than 1000 cubic meters per person per year is classified as water-scarce (Weiss, 2006). So, Ethiopia as a water tower is a bit controversial.

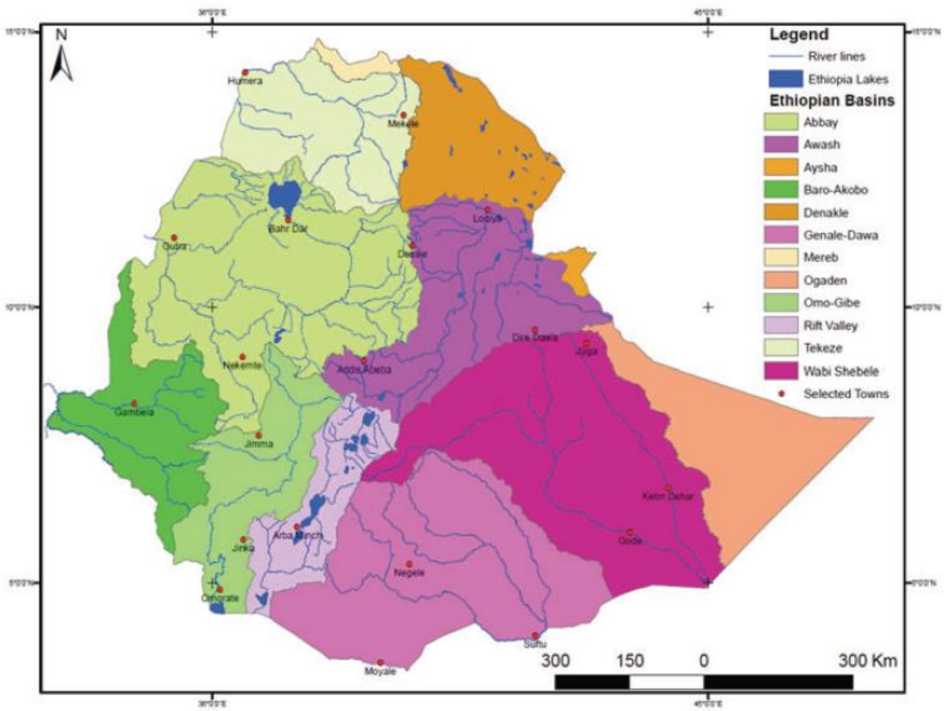

Figure 5-1: Surface Water and Groundwater Resources of Ethiopia. Source:Berhanu et al. (2014)

The water bodies in the country tend to be rich in fishery resources and diverse vegetation and have been used for various purposes including irrigation development, electric power generation, drinking water supply, mineral extraction, fishery development transportation (Awulachew et al., 2007; Tesfaye and Wolff, 2014). Despite all these uses and benefits, the water availability, quality, and suitability to aquatic habitats have been deteriorating as a result of climate change (Demissie *et al.*, 2013), land use and management change (Lenat and Crawford, 1994), and direct human impact on water resources (Malmqvist and Simon, 2002).

Water-course exploitation such as damming, channelization, or abstraction of water (for agricultural, domestic, or industrial purposes) directly alters the passage of water through river channels (Friedl and Wuest, 2002). Often the quantity and timing of run-off as well as siltation rates within river channels are affected indirectly due to habitat alteration (Harding, 2009). Surface water resource degradation can also result from land-use alterations such as deforestation and overgrazing (Malmqvist and Simon, 2002).

The climate in Ethiopia is geographically quite diverse, due to its equatorial positioning and varied topography (Berhanu et al., 2014) and densely populated lowlands like the Upper Awash valley (Tumebo, 2008). Surface water pollution in Ethiopia is also increasing from various sources. Studies have indicated that anthropogenic impacts on water quality in developing countries are predominantly from point-source pollution due to minimal sewage treatment (Beyene *et al.*, 2009b).

Table 2-1: Catchment area, runoff, irrigable land, and the electric potential of major rivers (Awulachew et al., 2007)

River Basin	Area (Km^2)	Runoff (BMC)	Potential Irrigable Land (ha)	Hydro-electric potential Gwh/year
Tekeze	82,350	8.2	83,368	5,980
Abbay	199,812	54.8	815,581	78,820
Baro-Akobo	75,912	23.6	1,019,523	13,765
Omo-Ghibe	79,000	16.6	67,928	36,560
Rift Valley	52,739	5.6	139,300	800
Mereb	5,900	0.65	67,560	-
Afar /Denakil	74,002	0.86	158,776	-
Awash	112,696	4.9	134,121	4,470
Aysha	2,223	-	-	-
Ogaden	77,121	-	-	-
Wabi-Shebelle	202,697	3.16	237,905	5,440
Genale-Dawa	171,042	5.88	1,074,720	9,270
Total	1,135,494	122.25	3,798,782	155,102

In recent years, heavy metal pollution of surface water is also of great concern because of the human health implications (Aschale *et al.*, 2016c, 2017). In developing countries such as Ethiopia, industries produce huge volumes of effluent that contain heavy metals and other toxicants that are discharged to surface waters without any treatment (Aschale *et al.*, 2016a, 2017). This pollution affects the aquatic environment and the poor people who depend on rivers to supply water for domestic and agricultural needs as well as fishing and other day-to-day activities (Melaku *et al.*, 2007; Gutiérrez-Fonseca and Lorion, 2014). Aquatic pollution by heavy metals pollution is less visible compared to other types of pollutants; however, its effects on the ecosystem and humans can be intensive and extensive (Chen *et al.*, 2016) causing severe health effects due to bioaccumulation and biomagnification in living organisms, particularly in humans (Xiao *et al.*, 2014).

Water pollution in developing countries, Ethiopia included, is also related to law enforcements difficulties (Akele, 2011; Cabrera and Custodio, 2014), poor management and monitoring of surface water resources (Bonada *et al.*, 2006), and absence of waste treatment facilities (by the factories, hospitals, municipals, and other organizations) (Akele, 2011). On the other hand, the use of bioassessment tools for the evaluation of surface water resources is very limited in developing countries, unlike developed nations like Europe, America, and Australia (Dallas *et al.*, 2010a; Elias *et al.*, 2014). Therefore, the advancing bioassessment methods for Ethiopia would have a substantial role for the monitoring of surface water resources in the country.

1.5. Bioassessment and Biomonitoring

In many nations, it is now customary to monitor the chemistry of the water after centuries of human influence on aquatic resources. Many developing and transitional nations lack biological monitoring instruments, and the basic scientific principles of sustainable water management remain mostly unclear (Bere and Nyamupingidza, 2014). Efforts to create monitoring methods have been accelerating in the last few decades, and the results have produced aquatic resource biomonitoring and bioassessment methods that are reliable enough to be incorporated into monitoring programs (Karr, 1997; Barbour and Yoder, 2000; Buss and Borges, 2008; Oliveira et al., 2011).

1.5.1. Bioassessment

Bioassessment is an evaluation of the condition of a waterbody using biological surveys and other direct measurements of the resident biota in surface waters (Barbour et al., 1999). It allows us to understand more about the processes occurring in watersheds by determining what organisms are found in a river and comparing them to what organisms are expected to be present (Karr and Chu, 1999). Although bioassessment of freshwaters is in its infancy stage in developing countries, including Ethiopia, it dates back to the Saprobic System developed (Kolkwitz and Marsson, 1908, 1909). The saprobic system, which stems from the research work of Kolkwitz and Marsson in German rivers in the early 1900s, is generally considered to be the first biological scoring system for the assessment of water quality in river ecosystems (Ollis et al., 2006; Dallas et al., 2010a). The saprobic system as a method has been developed as a biological determination of water quality and levels of organic waste (pollution) in rivers and streams (Elias et al., 2014). The saprobic system, a non-chemical analysis method, was based on patterns of abundance and distribution of various biological species from several different groups and trophic levels (mainly bacteria, algae, protozoans, and rotifers, but including some benthic invertebrates and fish), for which the tolerances to organic pollution have been established (Metcalfe, 1989). At that time, Kolkwitz and Marsson (1908, 1909) examined the biological patterns of approximately 800 species of water plants and invertebrates in their studies and produced a saprobic index of four zones defining levels of water quality.

Other methods were also developed following the Saprobic System in different parts of the world. For instance, in the United Kingdom (Armitage *et al.*, 1983; Wright *et al.*, 1984; Wright, 1995), North America (Hilsenhoff, 1987), the United States of America (Barbour et al., 1999), Canada (Rosenberg and Plan, 1999), Australia (Simpson and Norris, 2000), Thailand (Mustow, 2002), Brazil (Baptista *et al.*, 2007), Bolivia (Jacobsen and Marín, 2008), Tanzania (Kaaya, 2014), and South Africa (Dickens and Graham, 2002b). There are three main biotic indexes used to determine the level of water quality within a water body, namely: the Trent Biotic Index (TBI) (Hilsenhoff, 1987), the Biological Monitoring Working Party (BMWP) score (Armitage et al., 1983), and the Chandler Biotic Index (CBI) (David, 2011), which are calculated by the number of invertebrates present and their sensitivity levels to pollution.

Since then, new approaches to bioassessment based on macroinvertebrates have been developed based on predictive modeling such as the River Invertebrate Prediction and Classification System (RIVPACS), used for assessing water quality in freshwater rivers based on the macroinvertebrate species in the United Kingdom (Clarke *et al.*, 2003); the Australian River Assessment System (AUSRIVAS), a prediction system used to assess the biological health of Australian rivers (Wright, 1995); and BEnthic Assessment of SedimenT (BEAST) in Canada (Reynoldson *et al.*, 1995), which have been utilized in their monitoring programs.

In the United States, scientists have also developed multimetric indices using fish, periphyton, and macroinvertebrate communities based on a priori site classification (Barbour *et al.*, 1995; Barbour and Gerritsen, 1996; Gibson *et al.*, 1996). All these bioassessment methods are based on the establishment of reference conditions at unimpaired or minimally impaired sites and data comparison with test-impaired sites (Dolph *et al.*, 2010). Similarly, in Europe, an integrated assessment system for the ecological quality of streams and rivers using benthic macroinvertebrates (AQEM) has been developed (Hering *et al.*, 2004) in response to the requirements of the European Community Water Framework Directive (EWFD, 2000).

1.5.2. Biomonitoring

Biomonitoring is defined as the systematic use of biological responses to evaluate changes in the environment with the intent to use this information in a quality-control program (Dallas et al., 2010a). Most monitoring activity over the past few decades has focused on chemical monitoring, with an emphasis on meeting human health goals. Unfortunately, the emphasis on chemical monitoring has not led to clean water or healthy rivers. Chemical monitoring only provides a snapshot of the water quality at the time of sampling, and most often, water quality is assessed in terms of its relevance to humans and can underestimate degradation in overall ecosystem health (Karr and Chu, 1999) as well as missing intermittent pollutant inputs. Biomonitoring, on the other hand, provides insight into a stream's ability to provide a healthy place for aquatic organisms to survive and engage in ecosystem processes that also maintain the flow of ecosystem goods and services to humans (Turley *et al.*, 2015).

Effective biomonitoring procedures have been successfully developed and are currently being implemented in all countries in Europe, the United States, Australia, and Canada (Dallas *et al.*, 2010b), unlike those in developing countries. With the high diversity found within aquatic ecosystems, many organisms can be included in bioassessment metrics to evaluate the quality of stream and river health (Birk *et al.*, 2012; Pander and Geist, 2013). Bioindices based on benthic macroinvertebrates are best developed and well-established worldwide (Carter and Resh, 2013).

1.6. Macroinvertebrates as indicators of water quality

Some general observations of surface water resources can be drawn from an understanding of the pattern of ecological integrity that resulted from the legacy of human uses within the water basin (Quigley *et al.*, 2001). An ecological integrity assessment is held to be useful for measuring and monitoring biological diversity as well as ecosystem integrity (Brown and Williams, 2016). As indicated by Andreasen *et al.* (2001), ecological integrity encompasses ecosystem health, biodiversity, stability, naturalness, wildness, and beauty. Ecological integrity is defined by Tierney *et al.* (2009) as “a measure of the composition, structure, and function of an ecosystem about the system’s natural or historical range of variation, as well as perturbations caused by natural or anthropogenic agents of change.”

Assessment tools have been developed using various organisms as ecological integrity indicators of aquatic ecosystems including bacteria, protozoans, diatoms, algae, macrophytes, macroinvertebrates, and fish (Barbour et al., 1999). Of these indicators, benthic macroinvertebrates are the most widely applied and documented in different parts of the world (Dallas, 2022). These organisms are invertebrates without backbones that inhabit the bottom substrates (benthic) for at least part of their life cycle (Resh and McElravy, 1993; Barbour et al., 1995; Resh *et al.*, 1995).

The effectiveness of an indicator organism should be characterized by its relevance, consistency, and measurability (Oliveira *et al.*, 2005). Macroinvertebrates have a long history as indicator organisms due to their ease of collection, and their immediate and measurable response to impairments (Boyle and Fraleigh, 2003). A comparison made by Marzin *et al.* (2012) with four biological quality element metrics of macrophytes, macroinvertebrates, diatoms, and fish showed that fish and macroinvertebrate metrics were very sensitive to morphological degradations whereas diatom and macrophytes metrics did not show strong responses to the impairment changes. As explained by Duncan (2013) macroinvertebrates are relatively easy to identify at the family level and abundant in most surface waters. They have sensitive life stages which respond quickly to stress, integrate the effects of short-term environmental variations, and are well-suited for assessing site-specific impacts (David, 2011). They also provide information on the community structure (Boyle and Fraleigh, 2003; Effert, 2015) and can be influenced by environmental factors

(Washington, 1984). In addition, their pollution sensitivity differences make them excellent indicators of surface water quality status (Baptista et al., 2007).

Macroinvertebrates are used to identify the stressors causing the degradation of a site, based on the number and type of sensitive taxa present (Allan, 2004). The wide distribution of them over trophic levels allows for a better understanding of what is happening within the system (Barbour et al., 1999). They may also show the cumulative impacts of multiple stressors (David, 2011), like habitat loss, which are not always detected by the traditional water quality assessments using physicochemical measurements (Menetrey *et al.*, 2011). They not only take into account biological factors but also the physical and chemical characteristics of the system (Herman and Nejadhashemi, 2015) since the indicators are influenced by all these characteristics in the ecosystem. By using indicators to evaluate biotic integrity, managers can identify degraded areas and can allocate resources to restore the ecosystems with the greatest needs in the most cost-effective way (Butcher *et al.*, 2003; Walters *et al.*, 2009). In general, benthic macroinvertebrates are the best indicators to monitor surface water resources as described by scientists (Black; Barbour et al., 1999; Anderson *et al.*, 2006; Agboola *et al.*, 2019).

- Macroinvertebrates are essential component of freshwater ecosystems
- They serve as food for other organisms (fish, amphibians, and waterfowl)
- They are essential to the breakdown and cycling of organic matter, and nutrients
- Macroinvertebrates are vital to a properly functioning ecosystem because of their diversity
- They are good bioindicators as they are reliable indicators of water quality
- Macroinvertebrates spend all or most of their lives in water
- They are easy to collect
- They differ in their tolerance to pollution
- They respond to human disturbance in fairly predictable ways
- Macroinvertebrates are relatively easy to identify in the laboratory
- Macroinvertebrates are reliable indicators because many species show sedentary habits

1.7. Common indices used for bioassessment of river ecosystems

The selection of an appropriate technique depends on the issues being addressed and available resources (Mwedzi *et al.*, 2020; Ochieng *et al.*, 2020). Potential bioassessment metrics include diversity indices (Shannon and Weaver, 1949; Etemi *et al.*, 2020), biotic indices (Chessman and McEvoy, 1997; Abbasi and Abbasi, 2011; Arslan *et al.*, 2016), functional feeding groups (FFGs) (Statzner *et al.*, 2005; Mwedzi et al., 2020) and multiple biological traits as body size, shape, exoskeleton hardness, type of respiration, locomotion, feeding habit, and specific adaptations to flow constraint (Statzner et al., 2005; Carter *et al.*, 2017). Metrics may be individually reported (biotic or diversity indices approach) or incorporated into multimetric indices (Reynoldson *et al.*, 1997; Barbour and Yoder, 2000). In contrast, multivariate approaches use species composition and structure data to develop predictive models (Reynoldson et al., 1997; Chapman and Underwood, 1999).

1.7.1. Diversity Indices

The measure of diversity has been used by several authors for monitoring the pollution level of the environment (Hewitt, 1991; Ravera, 2001; Fitzgerald *et al.*, 2018; Etemi et al., 2020). As stated by Tolkamp (1985) many diversity indices have been developed to describe the responses of a community to environmental variation, combining the three components of community composition vis-à-vis (a) richness (number of species present), (b) evenness (uniformity in the distribution of individuals among the species) and (c) abundance (total number of individuals present). Some examples of these indices are the Shannon-Wiener Index (Shannon and Weaver, 1949), Simpson Index (Simpson, 1949), and Margalef Index (Margalef, 1951). The assumption is that undisturbed environments are characterized by high diversity or richness, an even distribution of individuals among the species, and moderate to high counts of individuals (Simpson, 1949; Ravera, 2001). The best use of diversity-related indices in river and stream monitoring is probably as an indicator of changes in species composition when comparing impacted and reference assemblages (Stevenson, 1984). Many criticisms have been made against the usefulness of diversity indices when employed separately in the assessment of river systems (Metcalfe, 1989), and now these indices are recommended to be used together with other metrics (Barbour and Yoder, 2000; Gabriels *et al.*, 2010; Melo *et al.*, 2015).

1.7.2. Biotic Indices

The biotic index approach, as defined by Tolkamp (1985), combines the relative abundance of certain taxonomic groups or taxa with their pollution sensitivities or tolerances into a single index or score. The sensitivity and tolerance of indicator assemblages to several environmental characteristics, such as organic pollution, heavy metals, pesticides, eutrophication, and pH, are known to differ among species. Therefore, these species-specific pollution indications can be used to infer environmental conditions in a habitat (Li *et al.*, 2010). Biotic indices based on macroinvertebrates and periphyton are widely used in developed countries in Europe, America, and Australia (Beck and Hatch, 2009; Carter et al., 2017; Etemi et al., 2020). Commonly used macroinvertebrate biotic indices include the Trent Biotic Index (TBI) (Pineda-Pineda *et al.*, 2018) and Extended Biotic Index (EBI), Chandler's Score System, Biological Monitoring Working Party Score System (BMWP), and ASPT (Average Score per Taxon), Hilsenhoff's Biotic Index (HBI) (Hilsenhoff, 1987) and others. Among these indices, BMWP and its derivative, IBMWP, are widely used in the European Union for Water Framework Directive (WFD) assessments. Within a site, the variance in the biotic index depends on the spatial distribution of benthic invertebrates and the effects of temporal fluctuations on the composition of the benthic community (Jones *et al.*, 1981). In the developing world such as in Africa, several biotic indices have been developed. For example, the South African Scoring System (SASS) (Dickens and Graham, 2002a), ETHbios developed for Ethiopia (Lakew and Moog, 2015), and the Tanzania River Scoring System (TARISS) (Kaaya *et al.*, 2015).

The major sources of variation in the biotic index among sites have been attributed to nutrient enrichment (Jones et al., 1981; Teodoru and Wehrli, 2005; Bussi *et al.*, 2021). However, in recent studies, major sources of variation in the biotic index could be specific pollutants such as heavy metals, organic waste emissions (Oliveira et al., 2005), and hydromorphological changes (Friberg *et al.*, 2009; Capela and Munné, 2015).

1.7.3. Multimetric Approaches

The multimetric approach, as noted above, combines several metrics associated with biological attributes like functional feeding groups, species composition, pollution tolerance, and trophic structure metrics into a single index (Mwedzi et al., 2020). Thus, multimetric indices represent a means to integrate a set of variables or metrics, which represent various structural and functional attributes of an ecosystem (such as taxa richness, relative abundance, dominance, functional feeding groups, pollution tolerance, life history strategies, disease, and density), and therefore provide robust and sensitive insights into the responses of an assemblage to natural and anthropogenic stressors (Karr, 1981; Barbour et al., 1995; Barbour *et al.*, 2000). Since Karr (1981) first introduced the Index of Biotic Integrity (IBI) based on fish assemblages, similar indices have been developed for benthic macroinvertebrates (Miller *et al.*, 1988; Barbour *et al.*, 1996; Fore *et al.*, 1996). Multimetric approaches for benthic macroinvertebrates have been the most widely used approach for river bioassessment in the USA (Barbour and Yoder, 2000) and other parts of the world as well (Vlek *et al.*, 2004).

1.7.4. Multivariate Approaches

Multivariate approaches were initially introduced to assess the biological status of rivers within the UK, with the development of RIVPACS (River Invertebrate Prediction and Classification System) (Wright *et al.*, 2000). Multivariate approaches adopt statistical analyses to predict site-specific fauna patterns, which are expected in the absence of major environmental stress; the biological evaluations are then performed by comparing the observed fauna at the site with the expected fauna (Norris and Hawkins, 2000; Niemi and McDonald, 2004). Multivariate approaches have been proven effective for biomonitoring. Several predictive models using multivariate techniques are widely used, such as RIVPACS and its derivative, AusRivAS (Australian Rivers Assessment System) (Simpson and Norris, 2000), and BEAST (Benthic Assessment Sediment) (Reynoldson et al., 1995; Rosenberg *et al.*, 2000).

1.7.5. Macroinvertebrate functional feeding groups approach

Functional feeding groups (FFG) are a classification approach that is based on morphobehavioral mechanisms of food acquisition rather than taxonomic groups (Oliveira et al., 2011). The same general morphobehavioral mechanisms in different species can result in the ingestion of a wide range of food items (Merritt *et al.*, 2002; Merritt *et al.*, 2006). The benefit of this method is that instead of hundreds of different taxa to be studied, a small number of groups of organisms can be studied collectively based on the way they function and process energy in the stream ecosystem (Metcalfe, 1989; Merritt et al., 2006). Individuals are categorized based on their mechanisms for obtaining food and the particle size of the food, not specifically on what they are eating (Merritt et al., 2006).Thus, the functional feeding group method of analysis avoids the relatively non-informative necessity to classify the majority of aquatic insect taxa as omnivores and it establishes linkages to basic aquatic food resource categories (coarse particulate organic matter [CPOM, particles >1mm], fine particulate organic matter [FPOM, particles <1 mm and >0.45 µm], periphyton, and prey) requiring different adaptations for their exploitation (Rawer-Jost *et al.*, 2000; Merritt et al., 2006; Fu *et al.*, 2016).

The major functional feeding groups are: 1) scrapers/grazers which consume algae and associated material; 2) shredders, which consume leaf litter or other CPOM, including wood; 3) collector-gatherers, which collect FPOM from the stream bottom; 4) collector-filterers, which collect FPOM from the water column using a variety of filters; and 5) predators, which feed on other consumers (Naiman et al., 2000; Kantor, 2001). A sixth category, other, includes species that are omnivores, or simply do not fit neatly into the other categories (Naiman et al., 2000).

Functional feeding group analyses support the notion that linkages exist in riparian-dominated headwater streams between CPOM and shredders, and FPOM and collectors, and between primary production (e.g., periphyton in midsized rivers) and scrapers (Naiman et al., 2000; Rawer-Jost et al., 2000). The feeding of shredders on riparian litter affects detrital processing in aquatic systems (Naiman et al., 2000). According to Tongayi *et al.* (2020), about 30% of the conversion of CPOM leaf litter to FPOM has been attributed to shredder feeding, and this can affect the growth of FPOM feeding collectors. In addition, shredder feeding enhances the release of dissolved organic matter (Benke and Meyer, 1988; Bohman and Tranvik, 2001). Such analyses link the balance between food resource categories and the predictable response of aquatic insect assemblages (Bohman and Tranvik, 2001).

1.8. Factors affecting the aquatic ecosystem

The definition of "water quality" includes the concentration of different constituents in water, for example, dissolved oxygen, nutrients, organic matter, inorganic sediment, temperature, as well as toxic elements such as Cd, Hg, Pb, and others in the water (Nilsson and Malm, 2008; Apitz, 2012) that are influenced by anthropogenic activities in the catchment. The major processes governing water quality within a catchment are related to transport, retention, and processing (e.g., decomposition of organic matter) (Nilsson and Malm, 2008; Ferrier and Jenkins, 2009). The variation in the flow of water also has a considerable impact on the physical as well as chemical quality aspects of water (Nilsson and Malm, 2008), in addition to hydromorphological conditions. As the human population continues to grow, it can be expected that anthropogenic activities will have impacts on the environment (Walters et al., 2009; Dos Santos *et al.*, 2011; Pander and Geist, 2013). This, in combination with changing climates, will amplify the impacts on streams, rivers, and other surface water ecosystems (Cabrera and Custodio, 2014). Malmqvist and Simon (2002) categorised the proximate causes of water pollution as urbanization, industry/mining, land use/agriculture, and watercourse alterations (Figure 6-1).

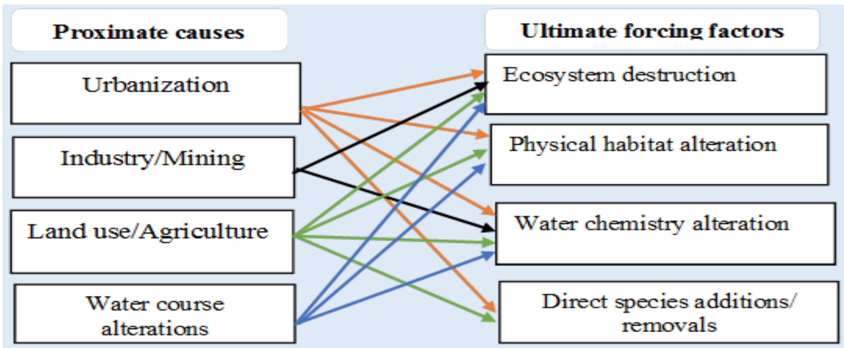

Figure 6-1: The four main proximate causes of ecosystem change in rivers and their link with factors that ultimately lead to change (Malmqvist and Simon, 2002).

Currently, these factors are alarmingly affecting the water quality of rivers and streams in developing countries, including Ethiopia (Table 3-1).

Table 3-1: Summary of anthropogenic factors affecting surface water resources in Ethiopia

Anthropogenic pressures	Reference
Urbanization	(Akalu *et al.*, 2011; Mazhindu *et al.*, 2012)
Industrial and domestic sewage	(Beyene *et al.*, 2012; Mazhindu et al., 2012; Mereta *et al.*, 2012)
Deforestation	(Teketay, 1997; Assefa and Bork, 2014)
Industrial processes	(Beyene *et al.*, 2013; Aschale *et al.*, 2016b)
Mining	(Mereta et al., 2012)
Intensive livestock grazing	(Kuru, 1978; Getachew *et al.*, 2012; Mereta et al., 2012; Ambelu *et al.*, 2013)
Agricultural cultivation	(Westphal *et al.*, 1975; Mereta et al., 2012; Meshesha *et al.*, 2014; Effert, 2015)
Removal of riparian vegetation	(Mereta et al., 2012; Effert, 2015), ,
Drainage	(Getachew et al., 2012; Mereta et al., 2012),
Nutrient input	(Ambelu et al., 2013)
Invasive species	(Tewabe *et al.*, 2016; Seid and Getenet, 2017)
Cultivation	(Meshesha et al., 2014)
Damming	(Vijverberg *et al.*, 2009; Ambelu et al., 2013)
Fuel wood	(Effert, 2015)

In general, the water flow regime, channel morphology, water chemistry, riparian vegetation, and the overall habitat quality (aquatic and terrestrial) are all affected by the aforementioned anthropogenic and natural impacts (Shilling et al., 2005; Ollis et al., 2006). This causes an imbalance in the natural aquatic ecosystems (Stevenson, 2011).

1.9. Management objectives of water resources

The environment is getting more unpredictable as a result of factors like climate change and human activity, making it harder to manage water resources (Jacob, 2020). Consequently, collaboration between the public and private sectors is necessary to devise strategies for the overall management of the water cycle. Water resource management makes it feasible to manage water resources effectively across disciplines, boundaries, and all water uses (Cabrera and Custodio, 2014; Jacob, 2020).

The empirical concept of water resources management (WRM) was developed from practitioners' practical experience (UN, 2008). The Global Water Partnership (GWP) provided a summary of these ideas. "WRM recognizes that water is a natural resource, a social and economic good, and an essential component of the ecosystem, whose quantity and quality determine the nature of its utilization (Jacob, 2020)." It is focused on the equitable, efficient, and sustainable WRM (UN, 2008). Meanwhile, the World Bank defines WRM as the "process of planning, developing, and managing water resources, in terms of both water quantity and quality, across all water uses." It includes the institutions, infrastructure, incentives, and information systems that support and guide water management (Jacob, 2020). According to the FAO (2006), WRM seeks to harness the benefits of water by ensuring there is sufficient water of adequate quality for drinking and sanitation services, food production, energy generation, inland water transport, and water-based recreational activities, as well as sustaining healthy water-dependent ecosystems and protecting the aesthetic and spiritual values of lakes, rivers, and estuaries .

The WRM also includes the control of risks associated with water, such as contaminants, droughts, and floods (Walther, 2010; Reid *et al.*, 2019). Because of the intricate interactions that exist between water and families, economies, and ecosystems, integrated management is necessary to take into consideration the many benefits and trade-offs that come with using and valuing water (Cabrera and Custodio, 2014). Water security is one of the key objectives of WRM (Altieri, 2016; Jacob, 2020). For the world's constantly urbanizing and expanding population, there is no single path to water security that can be "predicted and planned." Building capability, adaptability, and resilience for the planning and management of water resources in the future is necessary to support the strengthening of water security (Falkenmark, 2000).

Essentially, the objective of water resource management is to integrate several groups from various fields to develop comprehensive plans for future water use (Kasbohm *et al.*, 2009; Jacob, 2020). The objectives of WRM might change depending on the area and the state of the water resources at the time, as well as on policy and execution. Nonetheless, WRM goals frequently involve fostering circumstances that support the economical, environmentally friendly, and fairly distributed use of water resources (Kasbohm et al., 2009; Jacob, 2020). The objective shared by all projects is the integration of policy approaches with other sectoral policies in a broader domain (Watkins, 2006; Kasbohm et al., 2009). Creating administrative, technical, and social instruments for managing water resources is frequently a part of this. According to GWP (2000), the three main pillars of the RWM approach are:

- Creating an environment that is conducive to the development and management of sustainable water resources through appropriate policies, strategies, and legislation;
- Establishing the institutional framework that will allow these institutions to carry out their mandates
- Establishing the management tools that these institutions need to function.

1.10. Research gaps and objectives:

The significance of biodiversity for human wellbeing was recognized with the formation of the Convention on Biological Diversity (CBD), an intergovernmental agreement among 193 countries to support the conservation of biological diversity, the sustainable use of its components, and the fair and equal sharing of benefits (Cardinale *et al.*, 2012). Despite this agreement, evidence showed that biodiversity loss at the global scale was continuing (Butchart et al., 2010), often at alarming rates (Cardinale et al., 2012). The gaps in both science and policy need attention if future ecosystems are to provide the range of ecosystem services required to support more people sustainably (Butchart et al., 2010; Cardinale et al., 2012). To date, these gaps have been reported in Ethiopia (Awoke *et al.*, 2016). For example, law enforcement difficulties (Akele, 2011; Awoke et al., 2016) and poor management and monitoring of surface water ecosystems (Tewabe et al., 2016) have been reported as major gaps in water resources management in the country.

With the increasing anthropogenic impact on rivers and streams in Ethiopia, the scarcity of methods for monitoring and assessment of surface water resources was the impetus to work on advancing the bioassessment of water quality in wadeable rivers and streams in Ethiopia. Several bioassessment tools and protocols have been developed and used by developed countries for the assessment, management, restoration, and conservation of surface water resources (Barbour and Gerritsen, 1996; Baillie and Neary, 2015; Carter et al., 2017; Etemi et al., 2020; Ochieng et al., 2020) by determining or developing optimal macroinvertebrate kick-sampling protocol. In Ethiopia, both multimetric and biotic indices have been developed for river bioassessment (Lakew and Moog, 2015, 2015b; Alemneh *et al.*, 2019) and wetland assessment (Mereta *et al.*, 2013). However, these indices used kick-sampling methods developed elsewhere in different ecoregions and may not apply to the bioassessment of streams and rivers in Ethiopia based on the conclusions and suggestions of many studies (Frost *et al.*, 1971; Feeley *et al.*, 2012; Buss *et* *al.,* 2015). Consequently, the main objective of the present work was to advance the bioassessment of water quality in wadeable rivers and streams in Ethiopia.

1.11. The specific objectives

i. Identify the challenges for water quality protection in the greater metropolitan area of Addis Ababa and the upper awash basin.
ii. Refine/develop a benthic macroinvertebrate kick sampling protocol for wadeable rivers and streams in Ethiopia
iii. Investigate the Impacts of the Koka hydropower dam on macroinvertebrate assemblages in the Awash River Basin in Ethiopia using a developed kick-sampling protocol
iv. Assess the factors affecting macroinvertebrate community composition in three river basins of Ethiopia: a variation partitioning analysis

1.12. Research Questions?

For the objectives set above, the corresponding research questions have been established as follows:

i. What are the major challenges for water quality protection in the greater metropolitan area of Addis Ababa and the upper awash basin in Ethiopia?
ii. Which kick sampling method(s) best recruits adequate macroinvertebrate taxa in Ethiopia without compromising its representativeness to explain the quality of the river and streams?
iii. Can the hydropower dam affect the macroinvertebrate assemblages?
iv. Which factors significantly affect macroinvertebrate community assemblages in the three river basins of Ethiopia?

1.13. Organization of the chapters

The thesis is organised into six chapters:

Chapter 1 gives a general background of river health concepts, river and major components of the river ecosystem, bioassessment, biomonitoring, and types of monitoring tools to assess water quality. The chapter further explains the classification of rivers and major factors affecting the aquatic ecosystem.

Chapter 2 deals with challenges for water quality protection in the greater metropolitan area of Addis Ababa and the upper awash basin, Ethiopia – presenting the driver-pressure-state-impact-response (DPSIR) framework used in water vulnerability assessment. This allows decision-makers to identify major anthropogenic drivers of water quality degradation, the contribution of the drivers to pollution pressures, major impacts of water pollution, and the research gaps that would be conducted in the future, highlighting the need for bioassessment of water quality.

Chapter 3 deals with refining the benthic macroinvertebrate kick sampling protocol for wadeable rivers and streams in Ethiopia. It determines the optimum macroinvertebrate kick sampling method which enables stakeholders to carry out a rapid but accurate assessment of water quality in terms of time and habitat type in Ethiopia.

Chapter 4 uses the sampling method developed in Chapter 3 to assess the effects of a hydropower dam on the structure and composition of macroinvertebrates in the Awash River. It will help stakeholders consider and minimise the impact of existing and proposed hydropower dams on aquatic ecology in rivers and streams downstream of the hydropower dam in Ethiopia and elsewhere.

Chapter 5 identifies factors that affect macroinvertebrate community composition in three river basins of Ethiopia using variation partitioning analysis. It helps to determine unique and shared effects of environment, land use, and space on the composition of macroinvertebrates in streams.

Chapter 6 is a synthesis. It is a general discussion of all chapters and makes recommendations for future water quality monitoring of wadeable streams in Ethiopia.

Chapter Two: Challenges for Water Quality Management **in the Greater Metropolitan Area of Addis Ababa and the Upper Awash Basin, Ethiopia – a Time to Take Stock**

Adapted from:

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Chapter 2: Challenges for Water Quality Management in the Greater Metropolitan Area of Addis Ababa and the Upper Awash Basin, Ethiopia – A Time to Take Stock

2.1. Abstract

Ethiopia, the second-most populous country in Africa after Nigeria, has more than one hundred million people and is one of the world's fastest-growing countries in terms of economy. It has 12 major river basins with an annual renewable flow of 122 billion m^3. The country is facing increasing pressure on water resources in terms of both quantity and quality. The objective of this review was to synthesise the key results of research to date on the water quality in the environs of Addis Ababa and use that information to highlight management gaps, challenges, and future research needs. The general search was conducted on Google and Google Scholar web-based search engines and PubMed databases using relevant keywords. In addition, manual searches of other relevant peer-reviewed journals, policy documents, standards, online materials, reports, and other documents (published or unpublished) were conducted. ArcGIS version 10.5 was used to show the locations of rivers, lakes, land use, and land cover maps. A framework, driver-pressures-states-impacts-responses (DPSIR), was used to illustrate a chain of causal links. According to the studies reviewed, water pollution pressures result from rapid urbanisation and industrial expansion without adequate solid waste management and wastewater treatment facilities, and agricultural activities compounded by law enforcement difficulties. Trace metal contamination of rivers, streams, and reservoirs and their bioaccumulation in vegetables highlight the urgency of addressing water pollution in the upper Awash River catchment. Hence, effective pollution detection, mitigation measures, and monitoring including the development of biomonitoring tools, together with cost-effective management measures are urgently required to reverse the decline in water quality in Ethiopia in general and in the greater metropolitan area of Addis Ababa and the upper Awash River basin in particular.

Keywords: Addis Ababa, metropolitan, waste management, upper Awash River basin, water quality.

2.2. Introduction

People are at the center of sustainable development; however, continued rapid population growth presents challenges for it (UN, 2019). The world population was estimated to be 7.7 billion people in 2019 and could grow to 9.7 billion in 2050 (UN, 2019; Leridon, 2020). According to the UN (2019) report, countries of sub-Saharan Africa, Ethiopia included, could account for more than half of the growth of the world's population between 2019 and 2050. Ethiopia is the second-most populous country in eastern Africa after Nigeria with more than one hundred million people (Failler *et al.*, 2016) where the pollution of surface water is strongly related to rapid urbanization (Tsutsumi and Bendewald, 2010).

The environmental policy review by Kimball (2011) highlighted that Ethiopia faces several environmental challenges such as deforestation, soil erosion, loss of biodiversity, declines in soil fertility, and water quality degradation that pose significant risks for people. The country has 12 major river basins (Awulachew et al., 2007) with aggregate annual renewable flows of 122 billion m^3 (Adeba *et al.*, 2015). Addis Ababa, the capital of Ethiopia, sits within the headwaters of the Awash River basin that covers a total land area of 110,000 km^2 (Degefu *et al.*, 2013) with an estimated 18.3 million people in six planning sub-catchments (Awash-Basin-Authority, 2017). The upper Awash basin (Figure 7-2) has an area of about 10,841 km^2 (Yitbarek *et al.*, 2012). The greater metropolitan Addis Ababa has grown from scattered settlements to 530 km^2 (Mafuta *et al.*, 2011). It is a fast-growing city overwhelmed by urban poverty, unemployment, inadequate housing, overcrowding, high rural-urban migration, and underdeveloped infrastructure in its catchment (Tsutsumi and Bendewald, 2010).

The population of the city increases rapidly. For example, in the Central Statistics Authority (CSA) report in 2012, the population was 3.14 million with a growth rate of 3% per year, and it is estimated to be 4.37 and 5.27 million by 2020 and 2030, respectively (CSA, 2012). As a result of the rapid population and industrial growth, the water demand for domestic, commercial, agricultural, and industrial sectors continues to rise from time to time whereas water availability continues to decline because of competing uses, environmental degradation, and climate change (Shewaye and Adam, 1999; Worku, 2017).

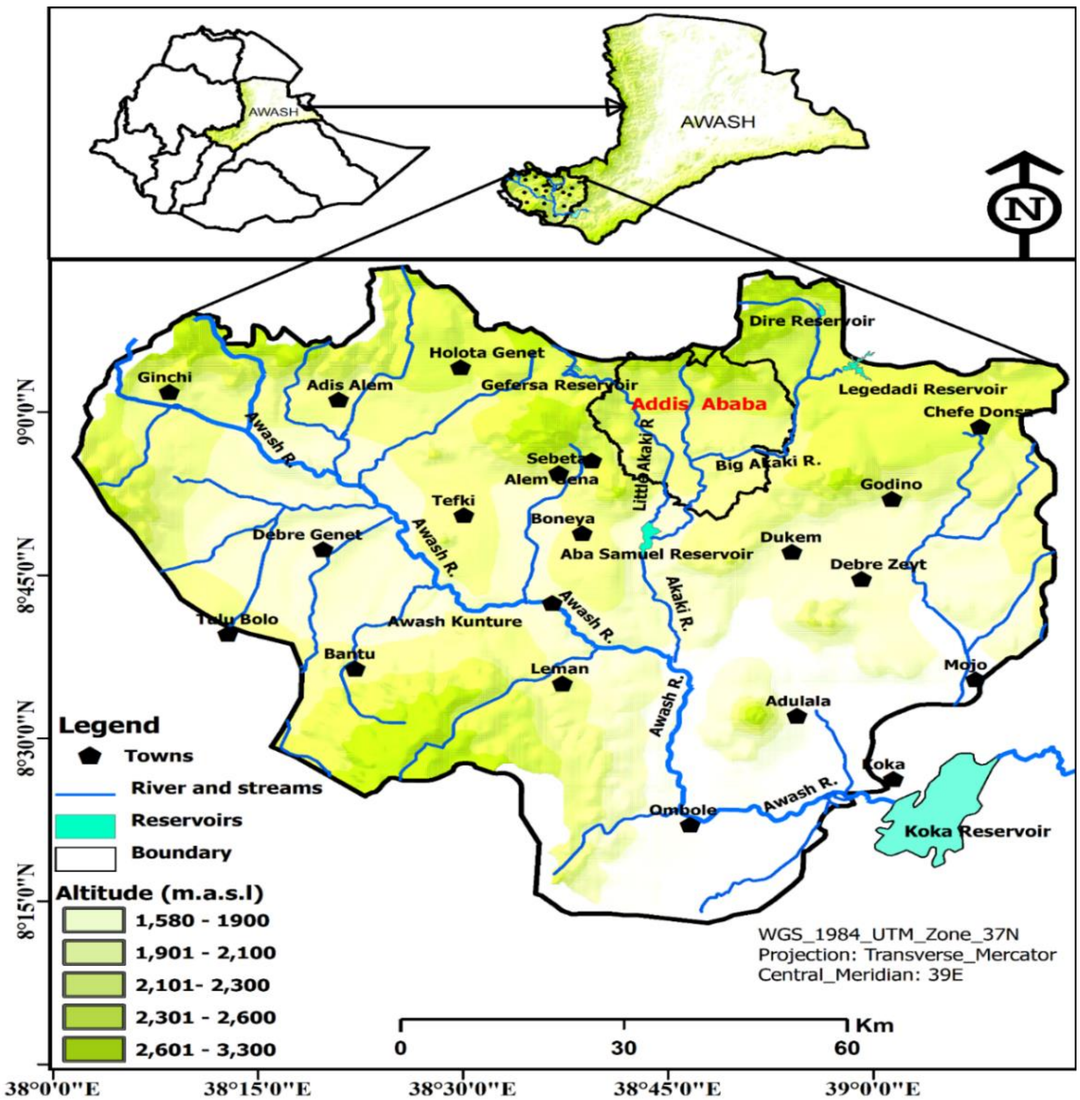

Figure 7-2: The greater metropolitan area of Addis Ababa and the Upper Awash basin and its reservoirs, rivers, and towns. Source: ArcGIS 10.5

For example, the population increased from 3.87 million in 2015 to 4.37 million in 2020 whereas the water demand increased from 670,000 m^3/d in 2015 to 1.1 million m^3/d in 2020 (Semie, 1998; Worku, 2017) (Figure 8-2). From Figure 8-2, it is evident that the water supply for the city was far below the demand.

For instance, the overall estimated production for 2015 from all reservoirs and wells was just 464,000 m^3/d representing only 69% of the water demand of the city for that year (AAWSA, 2011). Besides, a large volume of water loss has aggravated the problem. According to Worku (2017), the water loss obtained from the difference between the water volume produced and the billed water volume was about 37% on average and the amount of water distributed serves only to meet about 50% of the demand (Figure 8-2).

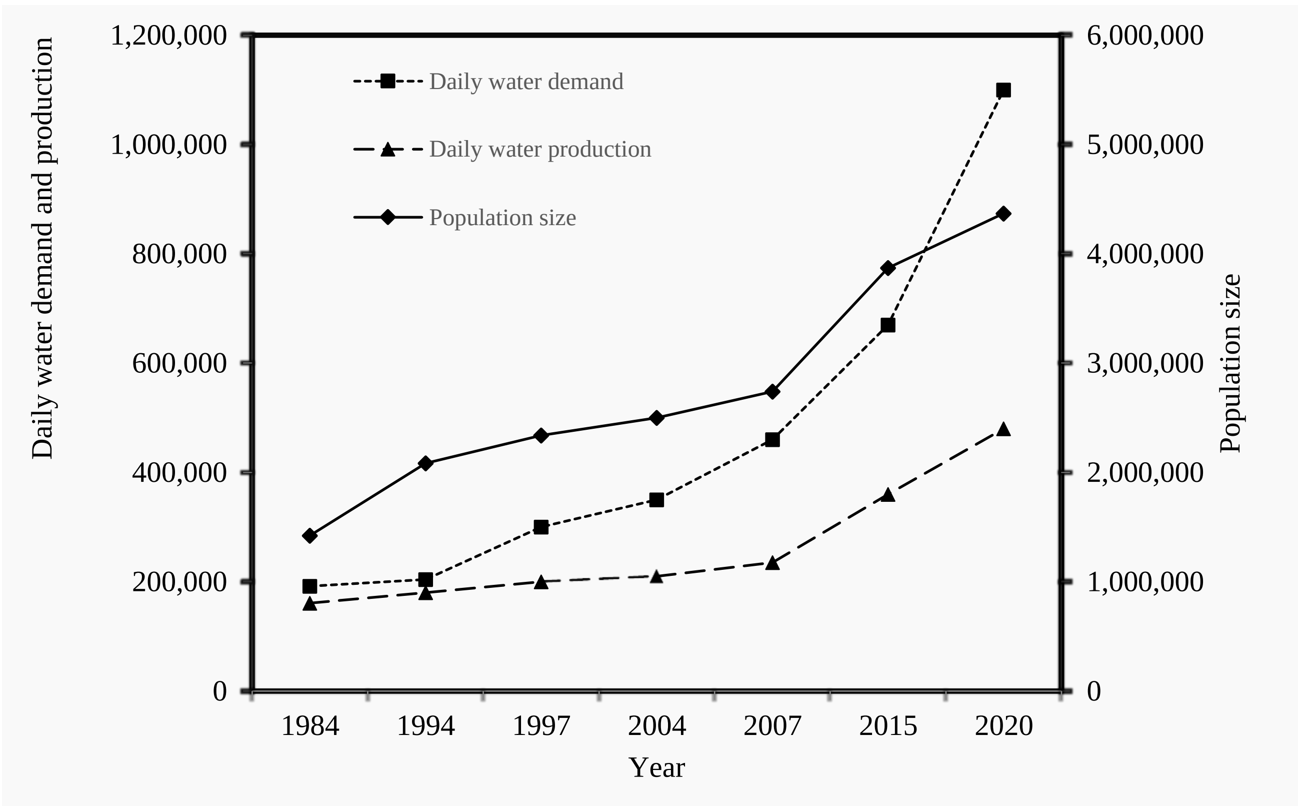

Figure 8-2: The population growth, daily water demand, and production of Addis Ababa City (1984-2020). Source: Own conversion based on Semie (1998) and Worku (2017)

Addis Ababa city and its surroundings in the upper Awash catchment include highly urbanised and industrialised areas (Figure 7-2) such as Debrezeyt, Dukem, Sebeta, and other towns (Awash-Basin-Authority, 2017). Several rivers and streams, for example, the Big and Little Akaki Rivers originate from the "Entoto" mountains and cross Addis Ababa city and other towns to the confluence with the Awash River upstream of Lake Koka (Figure 7-2). Besides, there are several reservoirs namely Legedadi, Gefersa, Dire, and Aba-Samuel that represent the main surface water resources for the catchment's population (Alemayehu, 2001; Worku, 2017).

The presence of large farmlands and its proximity to large cities makes the Awash River basin highly valuable for investment (Awash-Basin-Authority, 2017). According to the VividEconomics (2016) report, the Awash Basin water resources are the most intensively exploited of any of Ethiopia's major river basins. Understanding the sources of water quality problems, as well as the nature and consequences of water pollution and current management practices, is vital to identify opportunities to mitigate the impacts. Therefore, the objective of this review was to synthesise the key results of research to date on the water quality of rivers flowing through Addis Ababa and the upper Awash River catchment and use that information to highlight management gaps and future research needs.

2.3. Approaches

This review work was based on a range of studies on the extent and causes of surface water pollution in Addis Ababa city and the rest of the upper Awash River catchment. The general search was conducted on Google and Google Scholar web-based search engines and PubMed databases using keywords such as "Challenges for water quality, sources of water pollution, and impacts of polluted water." In addition, manual searches of other relevant peer-reviewed journals, policy documents, standards, online materials, reports, and other documents (published or unpublished) were conducted. ArcGIS version 10.5 was used to show the locations of rivers, lakes, land use, and land cover maps.

The Hamouda et al. (2009) framework was used to illustrate a chain of causal links starting with "driving forces" (human activities) through "pressures" (emissions, wastes) to "states" (physical, chemical, and biological) and "impacts" on (ecosystems, human health, and functions), eventually leading to political "responses" (prioritization, target setting, indicators) (Figure 9-2). It is common practice to break down complex tasks, like describing the causal chain from driving forces to impacts and responses, into smaller, more manageable ones. One such example is the analysis of the pressure-state relationship, which is shown in Figure 9-2 and whose details are given in Figure 2-1.

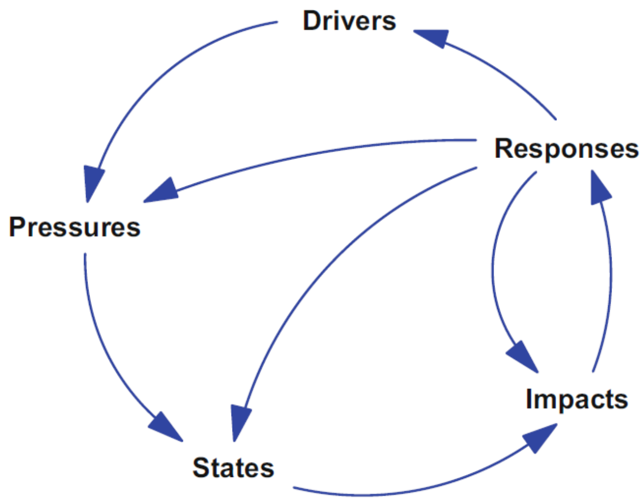

Figure 9-2: A generic DPSIR framework of environmental indicators, consisting of driving forces, pressures, states, impacts, and responses for water (Kristensen, 2004; Hamouda et al., 2009; Dickens et al., 2021)

Finally, the findings of this review work are presented in terms of drivers, pressures, and impacts. The available data have been summarized in Tables 4-2 to 9-2.

2.4. Findings

Because of the fast population growth, the demand for natural resources has increased significantly, leading to degradation including water resources (Worku, 2017). According to Adugnaw (2014), the causes of environmental degradation were grouped into proximate causes (which refers to inappropriate resource management practices) and underlying causes (which constitute initial conditions in the human-environment interaction) (Figure 10-2). The availability of clean water was one of the main reasons for moving Ethiopia's capital from "Entoto" to "Filwuha", the present downtown Addis Ababa, in 1886. Today, however, water pollution in the city, including the surrounding catchments, presents a significant threat to health and development (Mulu *et al.*, 2013; Gebre *et al.*, 2016; Yimer and Geberkidan, 2020). For example, Yimer and Geberkidan (2020) indicated that the area upstream of the Koka reservoir is the principal contributor to the deterioration of the Awash River because of the existence of several risk factors in the catchment.

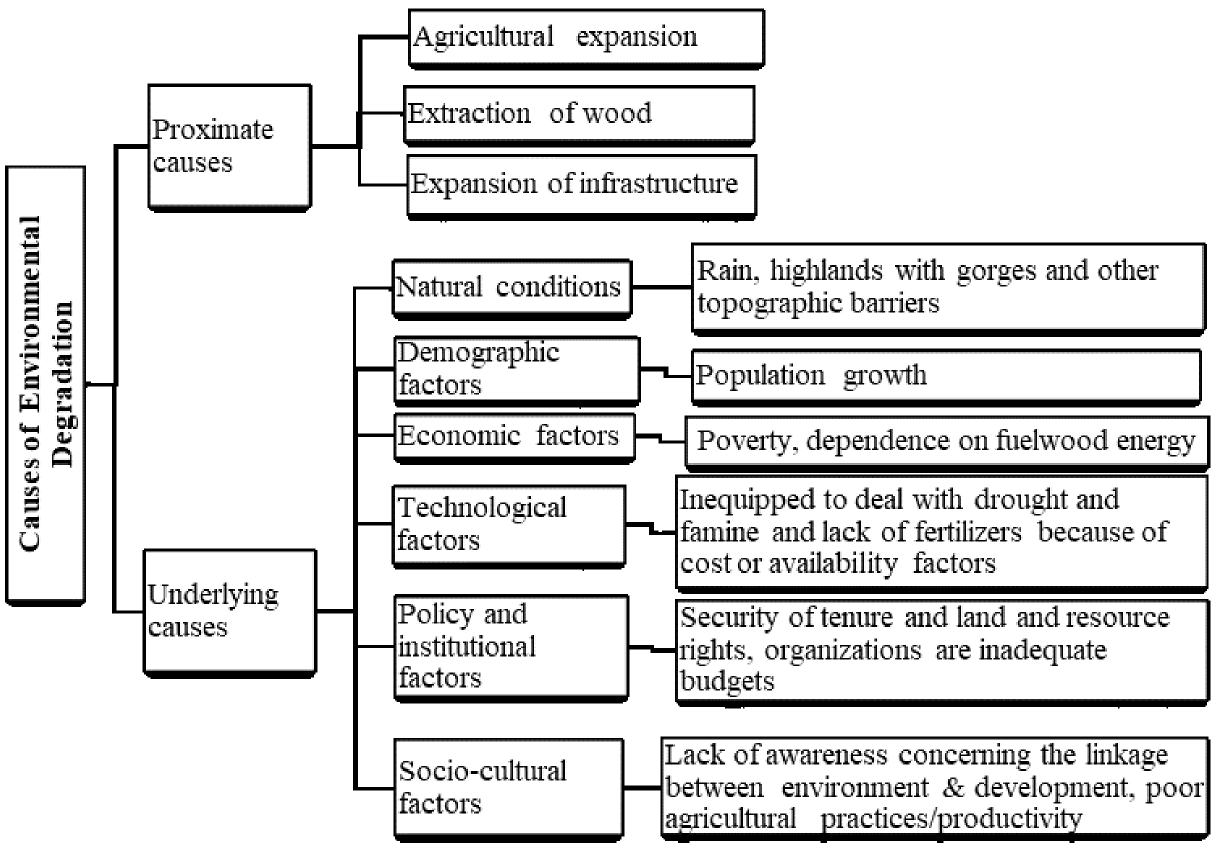

Figure 10-2: Causes of environmental degradation in Ethiopia. Source: Own modification based on Adugnaw (2014)

2.4.1. Major anthropogenic drivers of water quality degradation in the greater metropolitan area of Addis Ababa and the Upper Awash basin

In the Environmental Policy Review of Ethiopia by Graichen (2011), the major drivers of water quality degradation in the greater metropolitan area of Addis Ababa and the Upper Awash basin are categorised into urbanisation, industry, agriculture, and deforestation.

2.4.1.1. Urbanisation

Urbanisation is a complex socio-economic process that transforms the built environment, converting formerly rural into urban settlements, while also shifting the spatial distribution of a population from rural to urban areas (Golfieri *et al.*, 2018). According to Pauleit *et al.* (2019), the rate of urban expansion increase is predicted to be highest in Africa until 2030 and will be concentrated in five regions of the continent of which "the greater Addis Ababa" region in Ethiopia is one. As the continent's rapid urbanisation outpaced its capacity to provide essential water services, the number of people without access to adequate water facilities has increased quickly in recent decades (Gashaye, 2020) and the problem is serious in regions where the poverty levels are high. The level of environmental degradation is mainly determined by the region's demographic characteristics such as urbanization (Wang and Dong, 2019). According to Tsutsumi and Bendewald (2010), the pollution of surface water in Ethiopia is strongly related to rapid urbanisation. In 2018, the population of Addis Ababa was 4.4 million (Figure 2-2) and it is expected to increase to 9 million in 2035 by the next 15 years (Golfieri et al., 2018). Recent studies highlighted that the population increase puts pressure on local resources and services including water supply, sanitation, and solid waste management for the community in the greater metropolitan area of Addis Ababa and the upper Awash basin (Eriksson and Sigvant, 2019; Mohamed *et al.*, 2019; Gashaye, 2020).

a. Water supply services (Water resource demand)

The world's most valuable natural resource is water and one of the most challenging global environmental issues is the contamination of water resources (Eriksson and Sigvant, 2019). According to UN-Habitat (2017), water supply service coverage in Addis Ababa has been improved significantly; however, the demand still exceeds the supply (Figure 8-2) mainly due to the scarcity of surface water for treatment plants. By indicating that service infrastructures in Addis Ababa are unable to provide the residents with the means to lead healthy and productive lives, Worku (2017) observed that the city has already outgrown its local water resources and will, shortly, need to convey freshwater over long distances or from deep aquifers. Also, the work by Birhanu et al. (2018), using a dynamically integrated model tested on the city water supply, explained that both surface water and groundwater supply sources are being stressed. This has stimulated renewed interest in water infrastructure development for the city. For example, the expansion of existing water supplies and planning for new water supply dam construction projects in the neighboring Abay Basin (e.g. Sibilu and Gerbi Dams–30 km north of Addis Ababa) has been intensified (Birhanu et al., 2018). Furthermore, water supplies are more likely to be highly stressed in the future because of pollution (Eriksson and Sigvant, 2019) and climate change (Birhanu et al., 2018). The report by the Federal Environmental Protection Authority FEPA (2005), and other studies (Eriksson and Sigvant, 2019; Tariq et al., 2020; Yimer and Geberkidan, 2020) also identified that the water resources in this catchment have been contaminated with different organic and inorganic pollutants, discussed in more detail later.

b. Sanitation services and facilities

According to the United Nations Water Quality Sanitation Report (2011) cited in Yimer and Geberkidan (2020), over 2 million tons of sewage and other effluents drain into the World's surface water. The wastewater from Addis Ababa City drains toward the Akaki River, and, finally, to the Awash River and Koka Reservoir (Worku, 2017; Yimer and Geberkidan, 2020). With only 16% of the city connected to the sewage system, human feces inevitably end up in the river (Eriksson and Sigvant, 2019). According to CSA (2012), an estimated 72.3% of the city's residents are without access to adequate toilet facilities. Furthermore, unregulated discharges from slums (Adane *et al.*, 2017) and other sources reduce the quality of water, raise the cost of treating the water, and thus reduce its effective availability for the catchment population (VividEconomics, 2016).

According to the FEPA (2005), report, the first centralised sewage collection system in Addis Ababa was constructed with a capacity of 7,500 m^3/day to serve only 200,000 inhabitants. Similarly, a faecal sludge management plant was established at Kotebe and has a capacity of 30,000 m^3/day that serves 800,000 residents (FEPA, 2005). Even these facilities are inefficient in treating the wastewater and sludge to an acceptable standard. Furthermore, Mohammed and Elias (2017) reported that wastes have become more complex and the quantity has also increased from time to time, particularly in the greater metropolitan area of Addis Ababa and downstream of the city.

c. Solid waste management system

Solid waste continues to receive a great deal of attention as cities and countries deal with the lack of available space to dispose of household garbage and municipal solid wastes (Pimentel *et al.*, 2004). Despite generating large amounts of solid waste from domestic and several other activities in Addis Ababa, Mafuta et al. (2011) showed that the city did not have adequate waste management facilities. Currently, the system of waste removal in the city relies on the municipality which is expected to provide the full range of solid waste management at the "Rephi" open disposal site (Mohammed and Elias, 2017). Besides its unsightliness, the unregulated waste disposal activity in the city has affected both surface and groundwater and soil through the leaching of contaminants (Haile and Abiye, 2012).

Of the 750 tonnes of solid waste generated each day in the city (Mafuta et al., 2011; Mohammed and Elias, 2017), only 65% is collected and transported to the open disposal site (Rephi) and the remaining 35% is left unattended and often washed off into rivers (Mafuta et al., 2011; Worku, 2017). Besides, Woldeyohans *et al.* (2014) and Worku (2017) have indicated that commercial, domestic, and industrial solid wastes dumped anywhere in the city caused severe pollution problems for water resources in Addis Ababa and the downstream catchment.

2.4.1.2. Industry

Gebre and Van Rooijen (2009) and recently Menbere and Menbere (2019) highlighted that accelerated water quality change due to industrial pollution is one of the major environmental concerns globally. Although significant progress has been made in many developed nations to reduce direct discharges of pollutants from water bodies, more than 70% of industrial wastes in developing countries are dumped untreated into surface waters (Ademe, 2014). The United Nations Environment Programme (UNEP, 2010) cited in Ademe (2014) estimated that each year 300-400 million tons of heavy metals, solvents, toxic sludge, and other wastes from the industrial sector are dumped into water resources each year worldwide. Similarly, the industry sector has been identified as the largest source of water pollution in Addis Ababa both in terms of quantity and quality (Worku, 2017).

Addis Ababa and its nearby towns and cities in the upper Awash basin are known for their high concentration of industrial establishments (Awash-Basin-Authority, 2017), possibly selected for the proximity of rivers to dispose of discharges of wastewater (Degefu et al., 2013). This adds significantly to the extent and severity of the pollution in the catchment (Adugnaw, 2014). For example, out of 2,500 industries in Addis Ababa city, 90-96% lack onsite treatment facilities (McKee, 2007; Gebre and Van Rooijen, 2009; Menbere and Menbere, 2019) and directly discharge raw wastes to surface water (Adeba et al., 2015). The report by Leta *et al.* (2016) estimated that 4,877,362 m^3/year of wastewater is discharged from industries into the rivers and streams (Table 4-2).

The majority of the pollution comes from the textile and food industries (Ademe, 2014). Textile industries consume large volumes of water and chemicals at different stages of the wet processing phases. For example, one textile plant can use as many as 2,000 different chemicals (Khan and Malik, 2014) and close to 2270 liters of water to complete the production of fewer than 10 meters of fabrics (Islam *et al.*, 2011). Adeba et al. (2015) also reported that the highest amount of wastewater discharge (88.9%) was generated largely from three industry types (textile, food and beverage, and leather and footwear) (Table 4-2). According to Addis Ababa City Administration (2017) cited in Worku (2017) industry is growing at the rate of 20% per annum in the city worsening the degradation of the aquatic environment and increasing the constant danger of water-related diseases. A recent study by Yimer and Geberkidan (2020) revealed that the pollution status of the upper Awash River is highly related to the deterioration of its physicochemical, toxic metals, and organic nutrients.

Table 4-2: The volume of wastewater discharged from different industries in Addis Ababa, Leta et al. (2016)

Type of industry	Quantity (m^3/Year)	Percentage
Soap and detergents	1,089	0.02
Non-ferrous metals	2,217	0.05
Petrochemicals	11,421	0.23
Tobacco	31,080	0.64
Paper and Printing	45,967	0.94
Wood	47,805	0.98
Pharmaceuticals	50,089	1.03
Iron and steel	146,239	3.00
Rubber	205,746	4.22
Leather and Footwear	**547,860**	**11.23**
Food and Beverages	**1,795,252**	**36.81**
Textiles	**1,992,597**	**40.85**
Total	4,877,362	100.00

Note: Texts in bold show industries with a large volume of wastewater

2.4.1.3. Agriculture

The world's most valuable natural resource is water, and one of the most pressing global environmental issues is the contamination of water resources (Eriksson and Sigvant, 2019). Recent studies suggest that agriculture, which accounts for 70% of water abstractions worldwide, is among the leading causes of water pollution globally (Mateo-Sagasta *et al.*, 2018; Evans *et al.*, 2019). The use of wastewater in agriculture is a common practice around the world (Gashaye, 2020). In sub-Saharan Africa, less than 5% of the wastewater produced in the region is being treated (Janeiro *et al.*, 2020). Nevertheless, untreated wastewater is widely being used for agricultural purposes (Eriksson and Sigvant, 2019; Gashaye, 2020; Janeiro et al., 2020) due to population increase as a driving force to satisfy their food requirements (Evans et al., 2019; Tariq et al., 2020). According to Woldetsadik *et al.* (2017) and Aschale *et al.* (2019), urban farmers in Addis Ababa have cultivated vegetables for the last 60 years, using the Akaki River systems as the main source of irrigation water. Currently, about 60% of the city's vegetables are irrigated with contaminated river water (Gashaye, 2020). Because of increasing urban farming and other anthropogenic activities, the city of Addis Ababa is affected by serious surface water pollution (Eriksson and Sigvant, 2019).

The common cultivation practices outside of the city in the upper Awash catchment also range from horticulture to animal husbandry and the growing of field crops usually on a large scale with formal and informal irrigation (Van Rooijen *et al.*, 2010). For example, in 2008, 814 flower-growing farms covered an area of about 1400 hectares clustered in four districts at Bishoftu, Sabbata, Managesha, and Zeway (Schewel, 2018). The first three clusters are located in the Upper Awash basin at a distance of 30 to 50 km from Addis Ababa city. The report by Schewel (2018) also noted the Dutch-owned Agriflora Sher corporation in the southeast of Addis Ababa owns Zeway, Adami Tulu, and Koka greenhouses that cover some 650 hectares and represent the largest rose farm in the world. Consequently, there is widespread use of fertilizers and pesticides, and the dumping of liquid and solid waste to surface water resources from these activities with the reported impacts on ecological quality (Degefu et al., 2013). According to Evans et al. (2019), water pollution due to agriculture is predicted to increase in the future as the population grows and the demand for food increases.

2.4.1.4. Demand on forests for urban expansion, agricultural land, woodworking, and firewood (Deforestation)

Forests are instrumental in controlling soil erosion, land degradation, and desertification (Gebru, 2016). However, deforestation appears to have reached its climax (Gebru, 2016) putting humanity at risk of survival across the world including in Ethiopia (Zegeye, 2017). For example, a study by Assefa and Bork (2014) quantified a 23% decline in forest cover from 1972 to 2006 in the country. A recent study by Deribew and Dalacho (2019) also indicated that forest land decreased by 39.79% in six decades (1957–2017) whereas agricultural land area increased by 36.7% in the Addis Ababa city catchment. There are several drivers for the reduction of forest cover. Several researchers agree that the major drivers of deforestation include agricultural expansion, urbanisation, fire incidence, exotic species, population growth, and demand for fuel wood and building material (Tsutsumi and Bendewald, 2010; Gebru, 2016; Zegeye, 2017; Deribew and Dalacho, 2019), all of which have been happening in the upper Awash catchment (Gebru, 2016; Deribew and Dalacho, 2019).

Deforestation includes river and stream buffer strips, a vegetated area along with river water bodies (Izzati *et al.*, 2019) and it is severe in urban areas (Izzati et al., 2019) such as the metropolitan Addis Ababa and the upper Awash basin affecting environmental, social and economic problems in Ethiopia (Zegeye, 2017). It has led to reduced quantity and quality of surface and groundwater (Assefa and Bork, 2014; Gebru, 2016; Zegeye, 2017). Deforestation due to urbanisation further increases the water demand (Deribew and Dalacho, 2019) and plays a key role in water quality degradation (Zegeye, 2017), and affects biodiversity and ecosystem services (Gebru, 2016; Deribew and Dalacho, 2019). Protecting buffer riverside strip forests is not in Ethiopian environmental policy except for the recent exceptional Riverside Project in Addis Ababa that will stretch along the city's two biggest rivers (Terrefe, 2020). To reconfigure Addis Ababa as a hub for urban tourism, the grand riverside project (*Beautifying Sheger*) has been started (Terrefe, 2020). This experience should be shared with the other affected localities in the country and studies on buffer riverside strips should be conducted in the future.

2.4.2. The Contribution of the Drivers to Pollution Pressure

Pollution pressures originating from the drivers outlined in the above sections include organic matter, heavy metals, nutrients, and associated eutrophication, sediment, pesticides and herbicides, and medical wastes.

2.4.2.1. Organic pollution

Industries (Ademe, 2014) and untreated effluents from slaughterhouses (Mulu et al., 2013) are the main contributors of organic wastes to surface water in the greater metropolitan area of Addis Ababa city and its downstream catchment. The physicochemical characteristics of wastewater in various locations and the leachate from the solid waste disposal site in the catchment assessed by several researchers are presented in Table 5-2. This highlight the likely organic pollution with extremely high chemical oxygen demand (COD) and biochemical oxygen demand (BOD) ranging from 481.7 to 14702 mg/L O_2 and 177.3 to 7367 mg/L O_2 for COD and BOD, respectively compared to the standards of 20 mg/L for BOD and 200 mg/L for COD (WHO, 2006).

The water with a BOD concentration greater than 100 mg/L indicates very poor water quality that contains organic wastes and bacteria that deplete dissolved oxygen (DO) due to decomposition (Yimer and Geberkidan, 2020). Recently, Yimer and Geberkidan (2020) found that DO concentrations ranged from 0 to 2.5 mg/L from samples of water in the upper Awash basin indicating high organic pollution. This indicates the water pollution in the catchment is severe. The DO level below 3 mg/L affects most aquatic organisms and may result in death caused by suffocation (Yimer and Geberkidan, 2020).

Table 5-2: Average physiochemical characteristics of leachate from a solid waste disposal site and wastewater from different sources in the environs of Addis Ababa

Sites	EC (µS/cm)	TDS (mg/L)	PO_4^{3-} (mg/L)	TSS (mg/L)	NO_3-N (mg/L)	COD (mg/L O_2)	BOD_5 (mg/L O_2)	References
Kera abattoir (AA)	**1614.7**	NS	**67.3**	**3835**	**1450**	**11547**	**3980**	Mulu et al. (2013)
Luna abattoir (Mojo)	**3850**	NS	28.3	**125.6**	13.7	**431.7**	**177.3**	Mulu et al. (2013)
Paint Factories (AA)	**229771**	**1161.2**	NS	**467**	NS	**951.7**	NS	Berihun and Solomon (2017)
Rephi Leachate (DS)	NS	**19220**	NS	60	NS	**6581.5**	**684.7**	Woldeyohans et al. (2014)
Rephi Leachate WS)	419.5	**20910**	NS	**102**	NS	**7845**	**720**	Woldeyohans et al. (2014)
Rephi Leachate (AA)	NS	**8883.3**	NS	**309**	6.9	**14643**	**7367**	Haile and Abiye (2012)
Industrial WW (AA)	NS	NS	NS	**1563**	NS	**14702**	**4475**	Tsutsumi and Bendewald (2010)
Guideline (industry)	10-1000	2000	30*	100	10*	200	20	WHO (2006)

*Note: Exceedances are in bold, COD-Chemical Oxygen Demand, NS-Not Studied, AA - Addis Ababa, WW-Wastewater, EC-Electrical Conductivity, TDS - Total Dissolved Solids, TSS - Total Suspended Solid, BOD-Biochemical Oxygen Demand, DS-Dry Season, WS-Wet Season * Source: Al-Jasser (2011)*

Generally, most determinands presented in Table 5-2 showed values much higher than the recommended standards.

2.4.2.2. Heavy metals

Trace elements include more than 60 substances that are usually available in low concentrations in the environment and mammalian tissues, and many are considered essential minerals in humans. However, in higher amounts, many of these trace elements such as iron (Fe), copper (Cu), zinc (Zn), arsenic (As), lead (Pb), cadmium (Cd), chromium (Cr), nickel (Ni), and others have been linked to instances of acute or chronic liver injury. Heavily contaminated water resources with heavy metals in the Awash River Basin were identified in the downstream areas of the major towns (Dirbaba *et al.*, 2018). The pollution originating from the industries in the greater metropolitan area of Addis Ababa and other towns in the Upper Awash catchment contains elevated concentrations of several trace elements (Table 6-2).

Table 6-2: Mean concentrations of some trace elements (µg/L) from samples of wastewater, rivers, and groundwater in the environs of Addis Ababa, Ethiopia

River/Lake/Reservoir	Cr	Mn	Ni	Cu	Zn	Cd	Pb	Reference
Kera River	7.4	**1690**	8.9	39	193	<1.0	**33**	Itanna (2002)
Borehole water, AA	0.97	NS	0.61	0.85	56.6	0	**20.3**	Alemayehu (2006)
Modjo River, Dry season	**269**	**2358**	**185**	90.9	419	0.41	44.2	Masresha *et al.* (2011)
Awash River, Dry season	5.2	209	9.7	5.1	10.6	0.03	1	Masresha et al. (2011)
Awash River-inlet, wet season	**239**	**3139**	**180**	89.5	481	0.37	**50.9**	Masresha et al. (2011)
Koka reservoir, wet season	**50.9**	**422**	39.4	20.8	98.4	0.06	8.5	Masresha et al. (2011)
Awash River	NS	NS	NS	NS	107.6	**32.2**	**6091**	Degefu et al. (2013)
LAR	**67**	**1540**	6.66	5.61	25.5	0.06	3.13	Aschale et al. (2016a)
LAR	38.6	**1415.2**	10.4	128.35	174.8	0	**10.2**	Akele *et al.* (2016)
Akaki River	**56.48**	**1150**	NS	NS	140	1.02	6.92	Derso *et al.* (2017)
Kadisco paint WW	**460**	NS	NS	NS	515.5	**813.4**	**786.1**	Berihun and Solomon (2017)
Zemilli paint WW	**600.5**	NS	NS	NS	704.3	**701.5**	**500.2**	Berihun and Solomon (2017)
Rain bow WW	**567.7**	NS	NS	NS	400	**900.1**	**765**	Berihun and Solomon (2017)
Gast solar WW	**689.4**	NS	NS	NS	621.3	**690**	**1674.3**	Berihun and Solomon (2017)
Nifas silk WW	**802.5**	NS	NS	NS	807.6	**798**	**1842.5**	Berihun and Solomon (2017)
Modern building industry WW	**467.9**	NS	NS	NS	306.6	**599**	**934**	Berihun and Solomon (2017)
Aba Samuel Reservoir (ASR)	**77.5**	**1127.5**	**210**	10	187.5	NS	**111.5**	Kassegne *et al.* (2019)
BAR upstream of ASR	**60**	**1330**	45	10	245	NS	NS	Kassegne et al. (2019)
LAR upstream of ASR	**220**	**1335**	60	10	260	NS	NS	Kassegne et al. (2019)
LAR near Gefersa Reservoir	**70**	20	50	ND	210	NS	NS	Kassegne et al. (2019)
Kebena River- Near Entoto	**90**	30	**110**	NS	310	NS	NS	Kassegne et al. (2019)
Akaki River- below ASR	**80**	**1760**	**70**	ND	250	NS	NS	Kassegne et al. (2019)
LAR at Burayu site, AA	2.4	38	3.1	5.75	13	0.06	NS	Aschale et al. (2019)
LAR at Kolfea site, AA	255	**2000**	4.9	3.3	21	0.06	NS	Aschale et al. (2019)
LAR at Kera site, AA	2.35	**1150**	5.15	5.55	21.5	0.06	NS	Aschale et al. (2019)
LAR at Gofa site, AA	3.85	**1300**	3.9	6.55	10.9	0.04	NS	Aschale et al. (2019)
LAR at Akaki site, AA	46	**1400**	6.5	6.35	22.5	0.06	NS	Aschale et al. (2019)
WHO Standard	50.0	400.0	70.0	2000.0	3000.0	3.0	10.0	WHO (2011)

Note: Exceedances are in bold, NS-Not Studied, AA-Addis Ababa, US-upstream, LAR-Little Akaki River, BAR-Big Akaki River

In the studies by Masresha et al. (2011), Derso et al. (2017), Kassegne et al. (2019), and Aschale et al. (2019), high concentrations of Cr, Mn, Ni, Cd, and Pb (Table 6-2) in the unfiltered water samples were linked to industrial pollution. The concentrations of most trace elements in these studies exceeded the permissible limits set by WHO (2011). Similarly, Berihun and Solomon (2017) also found the concentration of Cd from heavy metal concentration from effluents of the paint industry in Addis Ababa to be above the permissible limit of WHO (2011) (Table 6-2). Similarly, Dirbaba et al. (2018) found that Cd, Cu, Ni, Zn, and Cr were the most contaminating metals and threat to aquatic life in the surface sediments of Awash River Basin (Table 6-2) and this study highlighted that the mean enrichment values of heavy metals increased in the order of Hg < As < Pb < Ni < Cu < Cr < Zn < Cd.

According to Mohan *et al.* (1996), the heavy metal pollution index of water varies between 0 and 100 indicating excellent and poor water quality, respectively. The study by Derso et al. (2017) showed that the Akaki River, which crosses Addis Ababa city, recorded a heavy metal pollution index ranging between 91.6 (upstream) and 24,100 (downstream). This study clearly illustrated that the river is unsuitable even for agricultural irrigation. Several other studies also reported that water from polluted rivers has been used for crop irrigation in farms such as Burayu, Kolfe, Kera, Goffa, Akaki and others in the catchment to produce a variety of crops and vegetables for both market and home consumption (Gebre and Van Rooijen, 2009; Van Rooijen et al., 2010). Approximately 60% of all vegetables (Golfieri et al., 2018) and 90% of leafy vegetables in the city come from urban farms, many of which are irrigated with contaminated river water (Weldesilassie *et al.*, 2011). Example, Aschale *et al.* (2015) indicated the agricultural products from these farms contain elevated concentrations of trace metals compared to the WHO standards (Table 7-2).

Table 7-2: Metal concentrations (mg/kg) in leafy vegetables from farms in the environs of Addis Ababa, Ethiopia

Vegetable type	Farms	Element (mg/Kg)									
		As	Cd	Co	Cr	Cu	Fe	Mn	Ni	Pb	Zn
Ethiopia kale	Burayu[a]	0.09	0.06	0.05	1.52	4.23	241	31.13	1.18	0.23	31.16
	Kolfea[a]	0.06	0.03	0.05	1.6	3.74	162	22.01	0.55	0.21	26.74
	Kera[a]	0.07	0.09	0.07	1.78	5.17	148	30.55	0.29	**1.11**	35.03
	Gofa[a]	0.07	0.05	0.03	1.39	5.14	185	29.6	0.38	**0.61**	44.81
	Akaki[a]	0.08	0.17	0.08	1.73	5.74	153	27.65	0.75	**1.37**	49.01
Lettuce	Kera[b]	**1.04**	0.13	0.76	**9.74**	6.62	**1345**	106	1.86	**1.59**	48.03
	Peacock	0.31	0.08	0.17	1.21	6.24	351	54	0.71	**0.39**	47.8
	Burayu[b]	0.11	0.12	0.25	1.99	9.76	248	51.65	2.63	**0.62**	40.11
	Kolfe[a]	0.13	0.07	0.43	**3.99**	16.11	**712**	43.94	1.88	**0.5**	56.19
	Kera[a]	0.05	0.07	0.07	1.39	5.46	183	23.23	0.76	**0.69**	36.88
	Akaki[a]	0.11	0.13	0.41	**2.89**	13.67	**795**	51.31	1.99	**1.27**	63.85
Cabbage	Kera[b]	0.13	0.02	0.06	0.89	3.03	73	29	0.8	0.21	31.8
	Peacock	0.11	0.01	0.13	1.61	3.3	173	25	0.91	0.29	31.81
	Akaki[a]	0.08	0.04	0.21	1.66	3.26	357.5	29.31	1.26	**0.32**	24.2
Swiss chard	Kera[b]	**1.21**	0.08	0.68	2.05	8.06	**527**	37.5	2.1	**1.79**	56.19
	Peacock	0.34	0.04	0.32	1.04	7.88	**461**	67	**89**	**0.61**	48.91
	Burayu[a]	0.13	**0.21**	0.44	2.04	14.31	**557**	229.8	2.76	**0.38**	83.27
	Kolfea[b]	0.12	0.05	0.42	**2.46**	11.58	369	143.3	1.47	**0.85**	47.11
	Kera[b]	0.08	0.09	0.24	1.24	10.35	209	112.09	0.93	**0.76**	58.07
	Gofa[b]	0.14	0.06	0.26	2.11	10.1	315	103.34	0.73	**3.99**	42.62
Potato	Burayu[a]	0.03	0.01	0.07	1.05	3.19	75.5	3.78	0.53	**2598**	7.54
	Akakai[a]	0.03	0.06	0.19	1.81	7.28	128	7.02	0.57	**3997**	21.72
Carrot	Burayu[a]	0.06	0.05	0.21	2.13	5.94	291	21.12	1.42	**4500**	21.92
	Kolfea[a]	0.05	0.02	0.15	2.24	6.56	178	11.81	0.74	**716**	21.22
	Akakai[a]	0.04	0.07	0.09	**2.32**	9.5	175	9.23	0.84	**2314**	25.96
Guideline Value[c]		0.43	0.2	50	2.3	73.3	425.5	500	67.9	0.3	99.4

Sources: [a]Aschale et al. (2015). [b]Itanna (2002). [c]Ewers (1991). Exceedances are in bold.

2.4.2.3. Nutrients and Eutrophication

According to Tegegn (2014) and Rahman *et al.* (2020), the higher occurrence of nitrate (NO_3-N) in groundwater was likely due to anthropogenic activities, e.g. excessive use of agrochemicals, unmanaged irrigation systems, sewage effluents, and unplanned disposal of solid waste in densely populated areas. Similarly, the sources of phosphate (PO_4^-P) are the discharges of sewage, defective septic tank effluents, chemical fertilizers, and consumer products such as detergents are major sources of phosphate (Tegegn, 2014) all of which observed in the upper Awash catchment including rivers in Addis Ababa city. For example, high concentrations of NO_3-N (1450 mg/l) and PO_4^-P (67.3 mg/l) were detected in wastewater from the Kera abattoir and disposed of in Akaki Rivers in the city (Mulu et al., 2013) (Table 8-2). According to Wagh *et al.* (2020), groundwater samples in terms of nitrate concentration are classified into three categories: no risk (<10 mg/l), moderate risk (10-45 mg/l), and high risk (>45 mg/l) (Table 8-2).

Table 8-2: The mean concentration of nutrients (NO3-N and phosphate, mg/L) in water samples from different springs, streams, and rivers in the environs of Addis Ababa, Ethiopia

Rivers/streams/spring	**Nitrate-nitrogen (NO_3-N)**	**Phosphate (mg/L)**	**Reference**
Ras Mekonen spring	**481.41**	**0.12**	Alemayehu (2001)
Yeka North spring	**55.34**	**0.11**	Alemayehu (2001)
Kolfe stream	39.77	**2.27**	Alemayehu (2001)
Kebena stream	**242.70**	**2.09**	Alemayehu (2001)
Kechene stream	42.14	**2.49**	Alemayehu (2001)
Little Akaki River	4.64	**3.95**	Melaku et al. (2007)
Little Akaki River	**123.55**	**12.19**	Mulu et al. (2013)
Modjo River	25.86	**12.15**	Mulu et al. (2013)
Upper Awash River	22.70	**51.70**	Degefu et al. (2013)
Little Akaki River (wet season)	30.00	**1.30**	Tegegn (2014)
Little Akaki River (Dry season)	**189.00**	**8.00**	Tegegn (2014)
Big Akaki River (wet season)	**116.00**	**0.80**	Tegegn (2014)
Big Akaki River (Dry season)	8.00	**7.00**	Tegegn (2014)
Modjo River	3.87	**2.41**	Gebre et al. (2016)
Little & Big Akaki	44.74	**0.28**	Derso et al. (2017)
Akaki River	0.293	**7.35**	Eriksson and Sigvant (2019)
Akaki River at Trunesh Bejing Hospital	**52.63**	**2.46**	Yimer and Geberkidan (2020)
Awash River Zeway road	11.9	**1.21**	Yimer and Geberkidan (2020)
Modjo River downstream of the factory	**215.13**	**1.50**	Yimer and Geberkidan (2020)
Koka Reservoir	1.4	**0.18**	Yimer and Geberkidan (2020)
Awash River at Wonji Bridge	2.32	**0.97**	Yimer and Geberkidan (2020)
WHO limit	50.00	0.10	Deborah (1996)

Note: Exceedances are in bold.

Similarly, the phosphate concentration presented in Table 2-5 ranges from 0.18 to 51.7 mg/l which is also beyond the guideline value (0.1 mg/l) (Deborah, 1996). As a result of excessive loads of these nutrients, the downstream water bodies show eutrophication symptoms (Kimball, 2011; Tegegn, 2014; Eriksson and Sigvant, 2019) and have produced cyanobacteria (Willen *et al.*, 2011). For example, Willen et al. (2011) found the highest cyanobacteria cell count in the Koka reservoir that represents a high level of risk for the catchment, for example, the local people have complained about stomach problems after drinking untreated water from the reservoir probably due to the presence of toxic cyanobacteria.

2.4.2.4. Sediment

Several studies have used sediment analyses to determine the spatial and temporal distributions of various contaminants in rivers in Addis Ababa city and other areas of the upper Awash River catchment (Table 9-2). The high levels of toxic elements in the sediment samples could also act as secondary sources of pollution to the overlying water column in the rivers. Compared to Burton (2002) sediment quality guideline (SDG), most sediment samples from the Akaki River had a high concentration of trace metals (Table 9-2). Similarly, Dirbaba et al. (2018) generalised that there was a high accumulation of heavy metals in the sediments of rivers and streams of the Awash River Basin perhaps because of urban and agricultural runoff and less-treated effluents from point sources like different industries.

Table 9-2: Comparison of heavy metal concentrations (mg/kg) in sediments of rivers in the environs of Addis Ababa, Ethiopia

Sampled river	Cd	Cr	Cu	Pb	Ni	Zn	Reference
Awash River	**2.60**	**121.00**	**79.43**	13.53	**89.46**	**282.73**	Dirbaba et al. (2018)
Akaki River	NS	37.00	8.30	**38.00**	15.00	**280.00**	Mekonnen *et al.* (2014)
Akaki River	0.22	**162.11**	32.02	**45.36**	**31.20**	**108.60**	Aschale et al. (2016a)
Akaki River	NS	**71.10**	**59.20**	**1441.60**	**48.90**	**228.50**	(Akele et al., 2016)
TEL^{SQG}	0.60	37.30	35.70	35.00	18.00	123.00	Burton (2002)

Exceedances are in bold. NS-Not Studied. TEL- Threshold Effect Level. SQG- Sediment Quality Guidelines

Hence, the high concentrations of trace metals in river sediments pose a high risk to the health of the community (Aschale et al., 2016a) and the ecosystem (Dirbaba et al., 2018). None of the studies has considered the physical impacts of excessive sediment inputs on the ecological health of various rivers.

2.4.2.5. Pesticides and herbicides

Pesticide is an agent used to kill or control undesired insects, weeds, rodents, fungi, bacteria, or other organisms. Pimentel (1995) had estimated that less than 0.1% of the applied pesticide reaches the target pest, leaving 99.9% as a pollutant in the environment, including soil, air, and water, or on nearby vegetation. According to Pimentel et al. (2004), no pesticide may legally be sold or used unless the chemical's label bears an Environmental Protection Authority (EPA) registration number. However, the pollution concerns have been growing because of several anthropogenic activities that use fungicides (Assefa, 2008) pesticides, and herbicides (Willen et al., 2011) in the upper Awash basin, including Addis Ababa city. As Assefa (2008) showed, all farmers applied two or more pesticide types together depending on the pest type and disease pressure. Particularly, floriculture requires intensive use of pesticides and also needs larger amounts of water than conventional farming (Getu, 2009). This study indicated that flower farms and factories located on the banks of the Awash River and its tributaries (Figure 2-1) heavily pollute the Koka reservoir. However, research regarding the use, type, legality, or magnitude of pesticide pollution has not been carried out in the catchment.

2.4.2.6. Medical Wastes

Medical waste refers to waste materials generated by healthcare activities, including a broad range of materials from used needles and syringes to soiled dressing, body parts, diagnostic samples, blood, chemicals, and pharmaceuticals (Debere *et al.*, 2013). According to the 2009 health indicators cited inDebere et al. (2013), there were 41 hospitals in Addis Ababa. Also, the 2012/2013 report prepared by the Ministry of Health in Ethiopia cited in UN-Habitat (2017) documented 62 health centers, 226 pharmacies, and 190 drug stores in the city. These institutions and other health care units produce a high volume of waste including hazardous ones. For example, the FEPA (2009) found that 430.7 tons of infectious wastes were generated by the 29 evaluated hospitals in the city. The conclusion by Debere et al. (2013) was that waste separation and treatment practices of the hospitals were generally poor and even some hospitals dispose of their waste off-site. These wastes can find their way into the nearby water resources (FEPA, 2009) compromising the water quality and ecosystem degradation.

2.4.3. Major impacts of water pollution in the greater metropolitan area of Addis Ababa and the Upper Awash basin

According to the United Nations World Water Assessment Programme, WWAP (2017), the consequences of water pollution can be classified into three groups: adverse human health effects associated with reduced water quality; negative environmental effects due to the degradation of water bodies and ecosystems; and potential effects on economic activities.

2.4.3.1. Human health impact

Drinking water quality is directly related to human health; however, consumption of contaminated water causes several health-related complications particularly in developing countries (Wagh et al., 2020). According to WHO (2014), an estimated 842,000 deaths in the middle-and low-income countries were caused by contaminated drinking water, inadequate hand washing facilities, and inadequate sanitation services in 2012. Wastewater-related diseases remain widespread in countries where sanitation services are poor, where informal use of untreated wastewater for food production is high, and where there is a reliance on contaminated surface water for drinking and recreation (WWAP, 2017), all common in Ethiopia.

Eriksson and Sigvant (2019) highlighted that bad river water quality, in combination with the usage for irrigation induces a health risk both for farmers and consumers in the catchment. Several studies concluded that rivers and streams in the Addis Ababa and the upper Awash basin are considered environmental health hazards due to the aforenoted high concentrations of chemicals (Derso et al., 2017; Aschale et al., 2019; Eriksson and Sigvant, 2019) and biological pollution from wastewater (Derso et al., 2017; Bedada *et al.*, 2019). Bedada et al. (2019) concluded that the detection of total coliphages in all sub-cities of Addis Ababa indicated the possibility that the pollution of urban rivers and hospital wastewaters may be a source for pathogenic viral infections. Despite this, nearly half of the urban and all the rural population are forced to use these water resources for domestic use, and in the worst cases, for drinking (Degefu et al., 2013; Dadi *et al.*, 2017).

The FEPA reported that all people in the city who use the Akaki River water in the city were affected by various communicable diseases (FEPA, 2005). According to Nifas Silk Health Centre 2008 Annual Report in Addis Ababa city, cited in Mazhindu et al. (2012), out of 320,000 people, 79,608 (24.8%) were infected with various types of sanitation-related diseases such as typhoid, dysentery, cholera, and faecal-oral transmitted diseases. Failler et al. (2016) also stated that water-based, water-born, and water-related diseases and water quality deterioration are the observed impacts of water pollution in the catchment. Besides, heavy metal toxicity from the vegetables grown using Akaki wastewater has been reported (Itanna, 2002; Weldegebriel *et al.*, 2012; Aschale et al., 2019). In this situation, there is a clear agreement from the studies to date that water from the polluted rivers crossing the city and further downstream should not be used for human consumption or irrigation (Masresha et al., 2011; Weldegebriel et al., 2012; Derso et al., 2017; Aschale et al., 2019). Especially, the downstream water users are critically suffering from pollution-induced impacts (Worku, 2017) due to microbes, organic, and inorganic pollutants from various sources (Bedada et al., 2019; Eriksson and Sigvant, 2019; Evans et al., 2019; Kassegne et al., 2019; Yimer and Geberkidan, 2020). For example, the high concentrations of NO_3-N beyond the WHO (2011) limit (Table 5) can pose a considerable risk to human and ecosystem health.

2.4.3.2. Economic impacts

The economic support offered by plentiful and high-quality surface waters includes agricultural irrigation, process and cooling waters for power plants, and chemical, steel, lumber, mining, and other industrial operations (Pimentel et al., 2004). The loss of income from tourism because of pollution and the restrictions imposed on food products has resulted in substantial economic losses (WWAP, 2017). The main economic impacts of water pollution arise from decreased labour productivity, as well as increased health care costs, loss of productivity through ecosystem damage, all leading to gross domestic product (GDP) loss which has been estimated to be 0.1% in some Intergovernmental Authority on Development (IGAD) countries (Failler et al., 2016). In reality, the costs of using contaminated or polluted water for production also relate to the decrease in both the quality and quantity of products. Thus, the economic effects of pollution are key considerations in Ethiopia's development plan. However, there is a paucity of studies on the economic impacts of water pollution in the metropolitan area of Addis Ababa and the upper Awash River catchment. For example, Evans et al. (2019) showed that linkages between water pollution and its impacts are not well understood and monetary values have not been assigned in less developed countries. The most devastating economic fallout from water pollution, according to the U.S. Environmental Protection Agency (USEPA, 2017), occurs in four main areas: (1) Cost of treating drinking water, (2) Losses to tourism (swimming, snorkeling, boating), (3) Damage to commercial fishing and shellfish harvests (4) Lower real estate values.

2.4.3.3. Ecological/environmental impacts

The continued use of chemical fertilizers and pesticides destroys beneficial organisms including the nitrogen-fixing bacteria in the soil affecting aquatic and terrestrial ecology in Addis Ababa mainly the downstream catchments (Getu, 2009). The environmental implication of high BOD_5 in wastewater is associated with the removal of dissolved oxygen (DO), which is essential for aquatic ecosystems (Dadi et al., 2017) affecting, for example, the fish population in both rivers and lakes. The surveyed literature also showed that the severity of pollution in the catchment would affect surface water ecology (Tegegn, 2014), but there are few studies on aquatic ecology impacts.

2.4.4. Responses options to water quality management problems

2.4.4.1. Environmental policy and institutions

The Constitution of the Federal Democratic Republic of Ethiopia Constitution (1994) is the most important source of environmental law since the basis for the Environmental Policy of Ethiopia are Articles 92.1 and 92.2 under Environmental Objectives (Cheever, 2011). (a) Article 92.1 states "Government shall endeavour to ensure that all Ethiopians live in a clean and healthy environment"; (b) Article 92.2 states "Government and citizens shall have the duty to protect the environment".

In Ethiopia, the key environmental body is the Environmental Protection Authority (EPA), whereas, other governing bodies such as the Ministry of Agriculture, the Ministry of Mines, the Ministry of Energy, and the Ministry of Water Resources are implementing the policy that is consistent with the Federal Democratic Republic of Ethiopia's Constitution (Cheever, 2011; Teklu, 2012). However, there are significant gaps between policies and their implementation (McKee, 2007; Eriksson and Sigvant, 2019). As explained by Kefauver (2011), the implementation gap has meant that many policies that appear "good on paper" have resulted in few tangible environmental outcomes on the ground. For example, Failler et al. (2016) outlined that most national and regional policies are broad and intended for protecting the environment and human health in Ethiopia, but lack specific components for regulation, and hence are difficult to enforce.

Environmental laws without enforcement are just words on paper (Pimentel et al., 2004).

The drawbacks of poor implementation capacity of laws and regulations have been mentioned by several researchers. According to Kefauver (2011), the implementation capacity of government organisations, environmental policies, and institutions remains weak partly due to inadequate budgets, lack of expertise, and inadequate facilities to test environmental conditions. Besides, weak legislative and institutional arrangements, weak support for private sector participation (Mafuta et al., 2011), lack of law enforcement mechanisms (Akele, 2011), the mismatch between environmental enforcement and investment policies (Zinabu *et al.*, 2018), weak implementation and lack of accountability, and bad collaboration between authorities (Eriksson and Sigvant, 2019) were identified as major areas to be improved. In addition, Awoke et al. (2016) also argued that the authorities do not have the right strategy for implementation because of a lack of knowledge and policies might be too weak to address such a water quality problem as the one found in the metropolitan area of Addis Ababa and the upper Awash basin.

Studies (Sharma *et al.*, 2014; Reid et al., 2019) suggested that the following measures (responses) can be taken to address water pollution:

(1) Tightening rules on industrial and agricultural activities, making sure waste is managed properly, and lowering the amount of pollutants released into water bodies,

(2) Sewage can be efficiently treated and purified before being released back into the environment by investing in state-of-the-art wastewater treatment infrastructure and facilities,

(3) Encouraging environmentally friendly farming methods, such as precision irrigation and organic farming, can minimize agricultural runoff and cut back on the usage of dangerous pesticides

(4) In order to develop a sense of environmental responsibility in both individuals and communities, it is imperative to raise awareness about the significance of water conservation, pollution avoidance, and responsible water consumption.

2.4.5. Research Gaps

Research to date has been focused on wastewater discharges, particularly organic and heavy metal contaminants, and implications for human health and the environment. No studies have adequately addressed the magnitude of pesticide pollution probably due to the high cost or the absence of laboratory facilities. Similarly, there is an insufficient study on the economic impacts of water pollution in the upper Awash River catchment including Addis Ababa city. Studies are also needed to identify cost-effective waste treatment approaches. Although several studies alluded to effects on aquatic ecosystems, there has been no concerted effort to assess the ecological health of the rivers draining the metropolitan area of Addis Ababa and the upper Awash River catchment.

2.5. Conclusion and Recommendation

2.5.1. Conclusion

From this extensive review, the following conclusions can be drawn:

(i) The surface waters in the metropolitan area of Addis Ababa and the upper Awash catchment are highly impacted particularly due to urbanisation, population growth, industrialisation without proper waste management facilities, unregulated agriculture practiced near the river banks, and several floriculture farms that use large quantities of fertilizers and pesticides, coupled with deforestation.

(ii) The poor waste management practices by the industries, health institutions, residents, and other sectors together with poor law enforcement and other factors such as poverty pose significant challenges in addressing water pollution problems in the catchment.

(iii) Heavy metal contamination of rivers and their bioaccumulation in vegetables grown using contaminated water highlight the urgency of addressing water pollution in the catchment.

2.5.2. Recommendation

From the foregoing findings of this review, it is clear that an integrated approach is needed to reduce challenges to water quality protection in the metropolitan area of Addis Ababa and the Upper Awash Basin. Based on the evaluation of scientific papers and reports, the following recommendations are forwarded:

(i) Decreasing point and non-point source pollution, increase the capacity of waste management, buffer zone demarcation, fund river studies, and support environmental charities

(ii) Never pour harmful substance such as used motor oil or antifreeze down a storm drain, onto the soil, into a waterway, or into the sanitary sewer affect the natural life of waterway

(iii) Mitigation and rehabilitation measures should be instigated to minimize the impacts on water and soil, and there should be planned afforestation of impacted sites to reduce erosion and associated water pollution.

(iv) Agriculture in the upper Awash River catchment particularly urban farming should be controlled as rivers and streams are a mainstay of unregulated irrigation, and a buffer area to protect the rivers should be zoned.

(v) Funding streams need to be established to enable further research works on the impacts and solutions to water pollution in the catchment.

Chapter Three: Refining Benthic Macroinvertebrate Kick Sampling Protocol for Wadeable Rivers and Streams in Ethiopia

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Chapter 3: Refining Benthic Macroinvertebrate Kick Sampling Protocol for Wadeable Rivers and Streams in Ethiopia

3.1. Abstract

Streams and rivers cover a larger proportion of the Earth's surface but they are highly affected by human pressures. Conversely, bioassessment methods are in their infancy in developing countries such as Ethiopia. The purpose of this study was to refine a kick-sampling protocol using macroinvertebrate data by considering riffle and multihabitat samples. The performance of each method was evaluated using benthic macroinvertebrate metrics and diversity indices using several statistical methods such as Analysis of Similarity (ANOSIM), non-metric multidimensional scaling (MDS), and the index of multivariate dispersion (IMD), the RELATE routine in PRIMER statistical software. The Kruskal-Wallis test and classification strength-sampling method comparability (CS-SMC) were also used in the analysis. The Kruskal-Wallis test in this study showed no significant differences among methods tested in minimally impacted streams in Ethiopia and generally performed equally irrespective of the methods employed except for total abundances and Ephemeroptera abundances. Furthermore, multivariate analysis of the relative abundances of macroinvertebrate communities using ANOSIM, RELATE, MDS, and CS-SMC indicated a high similarity in the macroinvertebrate communities recorded among all methods employed in this study area. However, the IMD test showed variations in the relative abundances of macroinvertebrate communities among the methods. In summary, if the focus is not on rare taxa and the required information is not dependent on additional evidence provided by the use of lower taxonomic levels of identification (genus and species), the results of the present study support the use of the shorter 2-minute riffle habitat kick sampling method for the biomonitoring of wadeable rivers and streams in Ethiopia. However, further study should be conducted across pollution gradients.

Keywords: Bioassessment. Fixed sampling time. Single and multihabitat sampling. Macroinvertebrates. Ethiopia

3.2. Introduction

Highland streams in Ethiopia have been reported to show deteriorated water quality from severe anthropogenic impacts (Alemneh *et al.*, 2017) including poor waste management practices, urbanisation, population growth, industrialisation, unregulated agriculture, floriculture farms that use large quantities of fertilizers and pesticides as well as deforestation (Getachew *et al.*, 2020b). A sampling strategy to monitor water quality must be capable of distinguishing anthropogenic impacts from natural variations (Callanan *et al.*, 2008). Natural variation in biological community structure occurs in the river continuum especially across ecoregions in response to changes in altitude (Wang *et al.*, 2012) and seasonal weather changes (Gratwicke, 1998; Alemneh et al., 2019). At the same time, the delivery of pollutants to rivers and streams may be from a point or diffuse sources as chronic or acute inputs. According to Buss et al. (2015), several factors also dictate the limitations of adopting standardised biomonitoring protocols. For example, climatic differences among countries create differences in season, month, or flow condition for sampling in all regions; the lack of taxonomic expertise to identify specimens to genus or species level; and the fact that most biological indicators and indices have been developed for specific geographic regions, states, or countries, as well as for specific pressures (Buss et al., 2015).

Inferring water quality and in particular ecological water quality from biological communities is well established (Resh and Rosenberg, 1993; Geist, 2011). Biological sampling protocols have been established for most developed regions such as America, Europe, and Australia (Barbour et al., 1999; Stark *et al.*, 2001; Koester and Gergs, 2017) based on their ecological contexts but are often not consistently applied in local, national, or continental scales, even in their home countries (Buss et al., 2015). The protocols follow several levels of sampling effort that can be categorised into three groups (Buss et al., 2015) namely: fixed sample number, fixed sampling length/area, and fixed sampling time.

Benthic macroinvertebrates are among the most widely used globally for the assessment of water quality as they reflect prevailing conditions, respond rapidly to environmental stresses, and are often the first ecological indicators to react to changes in the environment Barbour et al. (1999). For example, Carlisle *et al.* (2008) showed that the macroinvertebrate model was the most precise compared to the diatom and fish models. However, the use of macroinvertebrates has been criticised due to long-time requirements and expensive costs of sampling, sorting, counting, and identifying them (De Bikuna *et al.*, 2015) and this is likely to affect the effectiveness of biomonitoring protocols using macroinvertebrate communities (Pinna *et al.*, 2014).

Kick sampling is one of the most commonly applied methods for bioassessment using macroinvertebrates. However, the approach to kick sampling is variable especially in terms of timing, examples include 2-minute (Mandaville, 2002), 3-minute (Larsen *et al.*, 2012), 4-minute (Haase *et al.*, 2004), 5-minute (Cheshmedjiev *et al.*, 2011; Luo *et al.*, 2018) to collect samples of macroinvertebrate communities at each location. Conversely, there is often little attention given to the duration of the 'kicking' that should be conducted to take a sample (Bradley and Ormerod, 2002; Clarke *et al.*, 2002). Furthermore, there is variability in terms of the mesohabitats (e.g. riffle, run/glide, pool) targeted for sampling. Both single-habitat (Blocksom *et al.*, 2008) and multihabitat (Barbour *et al.*, 2006) approaches have been adopted. However, most developing countries including Ethiopia have no agreed macroinvertebrate kick sampling protocols. So far, several bioassessment studies conducted in Ethiopia have used various kick sampling methods, for example, 3-minute (Akalu et al., 2011; Getachew et al., 2012; Alemneh et al., 2017) (most frequently used method), 5-minute (Ambelu et al., 2013), and even 10-minute (Mereta et al., 2012; Mereta et al., 2013) in different parts of the country. There is no consensus on whether a standardised approach should be adopted for monitoring wadeable rivers and streams in the country in terms of the kick sampling method. This was a major gap identified as a priority for the current project.

In identifying the method that might be most appropriate both processing time and costs were considered. As the kick time increases, the cost of sample processing also increases unless sub-sampling is used. Researchers explained that sorting is often the most time-consuming aspect of a bioassessment study that leads to either economic or deliverability consequences (Haase et al., 2004). A reduction in time spent on sample processing while not decreasing sampling effectiveness will increase productivity and reduce costs. For example, Feeley et al. (2012) compared two multihabitat (MH) kick sampling methods for streams and found that a kick sampling method as short as 20-second was as effective as 60-second to produce a reasonable representation of the taxa present. Similarly, Mykra *et al.* (2006) found that there was no indication of differences in community structure among different sample sets collected for 1-, 2-, 3-, and 4-minutes, and suggested that even 1-minute kick samples can adequately represent macroinvertebrate community structure in streams for that particular ecoregion.

Reducing effort and cost in developing a protocol is not the only aim of a sampling method. It should also maintain accuracy to answer research or monitoring questions (Barbour and Gerritsen, 1996). Thus, effective yet rapid sampling and processing methods are required. However, there is a scarcity of studies in developing countries such as Ethiopia that have evaluated different kick-sampling methods in rivers and streams. As previously explained, studies in the country have used kick sampling methods varying in duration from 3 to 10 minutes, all of which have been developed elsewhere in different ecoregions such as non-tropical countries (Barbour and Gerritsen, 1996; Barbour et al., 1999; Ostermiller and Hawkins, 2004; Gabriels et al., 2010) without determining the habitat type to sample. The 2-minute kick sampling method has not been used in previous bioassessment studies in Ethiopia; however, it has been effectively used in different ecoregions (Bradley and Ormerod, 2002; Heino *et al.*, 2002; Heino et al., 2003). Therefore, the objective of this study was to determine whether there are differences in four sampling approaches; MH kick samples collected for 2- and 3-minute and riffle habitat (RH) kick samples collected for 2- and 3-

minute at minimally impacted sites (Table 2). The study tested the hypothesis that "no differences occur in terms of a range of macroinvertebrate metrics as well as community structure between samples collected using the four methods".

3.3. Methods and materials

3.3.1. Study areas

Ethiopia has a diverse climate due to its equatorial positioning and varied topography (Block, 2008) ranging from a semi-arid desert to a humid and warm (temperate) type in southwest Ethiopia (Beyene and Meissner, 2010) where this study was conducted. Traditionally climatic conditions in Ethiopia are classified into five major agroecological zones based on the altitude and temperature variations (MoA, 1998; Berhanu et al., 2014) (Table 10-3). The study sites for this study are located in the "Kola" and "Weynadega" agroecological zones which have nearly 80% share of the country's surface area (MoA, 1998).

Table 10-3: Traditional climatic zones of Ethiopia and their physical characteristics. Source: (MoA, 1998) and (Berhanu et al., 2014)

Agroecological zones	Altitude (m)	Mean annual rainfall (mm)	Mean annual temperature (ºC)	Area share (%)
Wurch (cold to moist)	>3200	900–2,200	<11.5	0.98
Dega (cool to humid)	2300-3200	900–1,200	11.5–17.5	9.94
Weynadega (sub humid)	1500-2300	800–1,200	17.5–20.0	26.75
Kola (warm semiarid)	500-1500	200–800	20.0–27.5	52.94
Bereha (hot arid)	<500	Below 200	> 27.5	9.39

Ethiopia has twelve major river basins (Awulachew et al., 2007). The study sites were located in the Abay and Omo-Gibe river basins (Figure 11-3) based on their physical characteristics (e.g. slope, soil type, stream order, and land use types) and known ecological conditions (Lakew, 2017; Alemneh et al., 2019) to reflect typical Ethiopian wadeable rivers and streams.

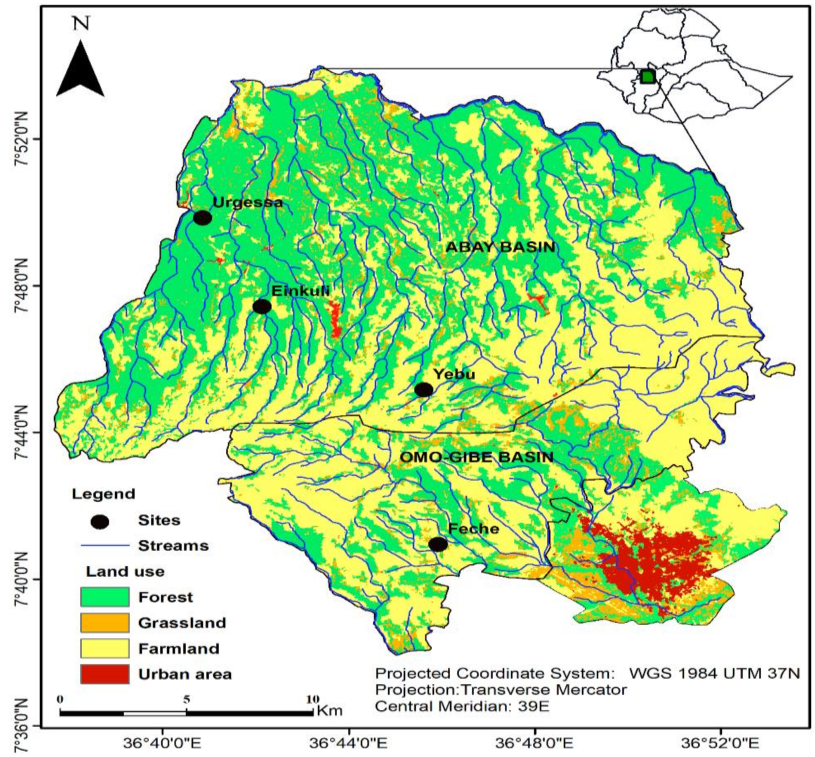

Figure 11-3: Land use types and location of study sites in the Abay and Omo-Gibe River basins of Ethiopia. Source: ArcGIS 10.5

Twenty reaches on four reference streams (Urgessa, Yebu, Enkulu, and Feche) with minimal anthropogenic pressures were selected using the standards representing minimally impacted waters (Table 3-2). The collection of macroinvertebrates samples within designated index periods is critical for repeated assessments (KDOW, 2015) and to minimize seasonal variation (Montana, 2012). The dry season was selected as an index period to collect the samples when water levels are sufficiently low to enable sampling. All samples were collected in the same year (2019) in one month (May). Previous bioassessment studies in Ethiopia have also been conducted during this season (mid-September to mid-June) (Getachew et al., 2012; Ambelu et al., 2013). Prior to field sampling background information on land use, geology, and soils where available was compiled (Table 11-3).

The coordinates and altitudes were recorded using a global positioning system (GERMIN-72H). Dissolved oxygen (mg/L), pH, water temperature (°C), conductivity (µS/cm @ 25^0C), and total dissolved solids (mg/L) were measured in-situ using a digital handheld portable multi-parameter (Hach HQD) probe. Habitat assessment (representation of flow mesohabitats and benthic substrates) was also conducted using the rapid bioassessment protocol set for wadeable streams and rivers (Barbour et al., 1999). Nutrients such as nitrate (mg/L NO_3-N) and phosphate (mg/L) were measured using a screen touch spectrophotometer in the Department of Environmental Health Sciences and Technology Laboratory of Jimma University.

Table 11-3: Details of streams selected for the collection of macroinvertebrate samples in southwestern Ethiopia, 2019

	Streams					
Variables	Urgessa	Yebu	Enkulu	Feche	Standard	References
Longitude	36.86	36.72	36.682	36.78	-	-
Latitude	7.7	7.77	7.80	7.68	-	-
Altitude (m.a.s.l)	1487	2231	1710	1723	-	-
Stream order	2	2	2	3	-	-
Soil type (Nitrosols-Nitr)	Nitr	Nitr	Nitr	Nitr	-	-
Point source pollution	None	None	None	None	None	Sánchez-Montoya *et al.* (2012)
%Urbanization	None	None	None	None	<0.4%	Lakew (2017)
Sand/gravel extraction	None	None	None	None	None	Sánchez-Montoya et al. (2012)
Intensive use of grazing	None	None	None	None	None	Sánchez-Montoya et al. (2012)
Water diversion and dam	None	None	None	None	None	Alemneh et al. (2019)
Dry land farming	10%	5%	10%	15%	<20%	Sánchez-Montoya et al. (2012)
Tree removal	5%	None	5%	8%	<10%	Alemneh et al. (2017)
Conductivity (µS/cm)	65	64	59	68	45-265	Lakew (2017)
Total diss. solids (mg/L)	46	45	41	48	<500	Oluyemi *et al.* (2010)
pH	7.49	7.65	6.55	6.85	7.3-8.2	Sánchez-Montoya et al. (2012)
Dissolved oxygen (mg/L)	8.41	7.73	7.45	7.26	6.8-9.4	Lakew (2017)
% DO saturation	98	98	99	95	95-120	Lakew (2017)
Phosphate (mg/L)	0.013	0.021	0.017	0.024	0.01-0.03	Fadiran *et al.* (2008)
Nitrate-nitrogen (mg/L)	0.35	0.67	0.42	0.78	2.0	Camargo *et al.* (2005)

3.3.2. Kick sampling method evaluation

The typical sampling method for streams and rivers is called kick sampling and involves disturbing the material on the bed of the river or stream and collecting the freed organisms in a net (Anderson *et al.*, 2014). An accessible length of the stream was first selected at each site to locate five 50m reaches (250 m in total on each river) for sampling after Feeley et al. (2012) and Mabidi *et al.* (2017). From these stream reaches, 15 samples for each 2- and 3-minute MH method were collected from all mesohabitats including riffles, run/glides, and pools in proportion to their representation in each of the 50m streams reaches. Similarly, 5 samples of 2- and 3-minute duration were collected from riffle habitats only within similar reaches where the MH samples were collected. There was no overlap in the sampling area covered by each sampling method. For all methods, sampling was started from downstream and continued upstream to avoid habitat disturbance prior to sampling. A total of 120 MH replicates (60 for each method) and 40 RH replicates (20 for each method) were collected using a D-frame pond net (0.5mm mesh). Immediately after the collection of benthic macroinvertebrates, large leaves, sticks, and stones were individually rinsed and inspected for organisms and discarded at the site.

The entire sample was then preserved in 96% ethanol alcohol after Barbour et al. (1999) and transported to the laboratory for sorting and identification to the family level and counted the entire sample in the laboratory. Likewise, Wright et al. (2000) highlighted that family-level data are adequate for rapid bioassessment of water quality. Moreover, Barbour et al. (1999) showed that family-level identifications provide a high degree of precision among samples, require less expertise to perform, and accelerate the production of results. However, the necessary level of taxonomic resolution is determined depending on the purpose(s) of a study (Bailey *et al.*, 2001). Fundamentally, most biomonitoring programs acknowledged a tradeoff between efficiency and sensitivity for large-scale monitoring, thus many strategies for reducing processing time and cost have been implemented (Buss et al., 2015).

3.3.3. Data and Statistical Analysis

The non-parametric Kruskal-Wallis test was used to test for significant differences in a range of metrics among the four kick sampling methods (2- and 3-minute RH and 2- and 3-minute MH samples). The analysed metrics include taxonomic richness at the family level, total abundance, Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness, %EPT, Ephemeroptera richness, Ephemeroptera abundance, family biotic index (FBI), the Biological Monitoring Working Party (BMWP) score, Average Score Per Taxon (ASPT) (Hawkes, 1998), benthic macroinvertebrates based on the biotic score (ETHbios) and ASPT calculated from ETHbios (Aschalew and Moog, 2015). Also, the Shannon-Wiener index, evenness, Simpson's index, %scrappers, %shredders, %collector-gatherers, %collectors-filterer, and %predators were used to evaluate the four kick sampling methods. The post *hoc* Mann-Whitney test was calculated to evaluate pairwise differences between the groups. The Bonferroni corrected P-values were calculated in PAST statistical software to reduce Type I error, or false rejection of the null hypothesis (i.e. there is no difference between the kick sampling methods). The bar charts with the standard error of the mean (Cumming *et al.*, 2007) were also displayed for selected metrics (richness, total abundance, and Ephemeroptera abundance).

The relative abundances of macroinvertebrate communities were used to carry out the multivariate tests to control for the differences in kick net, sampling times, and habitat differences. Analysis of similarity (ANOSIM), a randomization test (Clarke and Green, 1988), was performed on the resemblance matrix based on the Bray-Curtis similarity coefficient (Armitage *et al.*, 1987; Friberg *et al.*, 2006). This was carried out in PRIMER 6, based on log(x+1) transformed macroinvertebrate community relative abundance data using 999 permutations. It allows a test of the null hypothesis that there are no macroinvertebrate assemblage differences among groups of replicated samples collected by different kick sampling methods (Friberg et al., 2006). The differences among the groups were measured by the global R statistic, which is calculated as $R = \frac{\mathbf{4}(\bar{\boldsymbol{r}}_{\boldsymbol{B}}-\bar{\boldsymbol{r}}_{\boldsymbol{W}})}{\boldsymbol{n}(\boldsymbol{n}-\mathbf{1})}$, where $\bar{\boldsymbol{r}}_{\boldsymbol{B}}$ and $\bar{\boldsymbol{r}}_{\boldsymbol{W}}$ are the mean rank between-group similarity and within-group similarity, respectively, and *n* is the total number of samples (Cao *et al.*, 2005).

The ANOSIM calculates a test statistic R that ranges from -1 to 1 (Chapman and Underwood, 1999). An R-value close to 1 indicates good separation or differences exist among groups and a value close to 0 indicates weak separation or complete random grouping (Chapman and Underwood, 1999; Friberg et al., 2006) and the negative values occur when the samples within a group are less similar to one another than to the samples of other groups, probably due to inappropriate sampling designs (Chapman and Underwood, 1999). ANOSIM provides associated P-values to R-statistics which highlights the significance level of the test (Friberg et al., 2006).

The two-dimensional MDS ordination plot which is a visual representation of differences in macroinvertebrate structures among the methods was also performed using a relative abundance matrix. An MDS is a rank-based approach where the relative abundance data are substituted with ranks (Buttigieg and Ramette, 2014). An ordination plot with stress values equal to or below 0.05 indicated a good fit and values >0.2 are arbitrary or more of a random pattern indicating that there is little explainable pattern in the MDS plot (Clarke, 1993). To elucidate the differences in macroinvertebrate distribution in the MDS plot, an index of multivariate dispersion (MVDISP) test was then employed to statistically measure the relative rank dissimilarity between the samples within each method. Correspondingly, the within-group similarity was also determined for samples within each method using SIMPER analysis according to the method described in (Friberg et al., 2006).

Furthermore, to compare the patterns of macroinvertebrate community structure among different kick sampling methods, the RELATE routine was used in PRIMER 6 (Friberg et al., 2006) which compares the relationship between two different resemblance matrices or MDS plots by calculating Spearman's rank correlation coefficients ρ(rho) from relative abundances of macroinvertebrates based on 999 permutations to test the (dis)similarity of the methods. A Spearman's rank correlation coefficient ($\rho = 1$) indicates a perfect similarity between method pairs.

Finally, the classification strength-sampling method comparability (CS-SMC) (Cao et al., 2005) was computed based on the formula proposed by several authors (Van Sickle, 1997; Cao et al., 2005; Cao and Hawkins, 2011). The CS-SMC allows the direct comparison of benthic macroinvertebrate community structure similarity obtained by kick sampling methods (Cao et al., 2005). All six pairs of the sampling methods were compared from the log(x+1) transformed Bray–Curtis similarities of taxonomic assemblages (SIMPER) to calculate the CS-SMC using the following equation (Van Sickle, 1997; Cao et al., 2005): $CS - SMC = \frac{2\bar{S}_b}{\bar{S}_{w1}+\bar{S}_{w2}} \times 100$, where $\bar{S}_b$ denotes the mean between-group similarity, $\bar{S}_{w1}$ and $\bar{S}_{w2}$ denotes the mean within-group similarity for any possible pairs of macroinvertebrate kick sampling methods from MH and RH samples. The CS-SMC measures similarity between methods relative to that between replicates within a method Cao et al. (2005) and indicates that methods are similar if the average within-method similarity is equal to the average within-method similarity of the other method. The CS-SMC is a useful measure of comparability as it is independent of site differences and sampling effort (Cao et al., 2005; Feeley et al., 2012).

3.3.4. Quality Assurance and Quality Control

In addition to the specific requirements of the sampling and any supplementary site-specific procedures, quality assurance, and quality control measures have been carried out. Some of these steps are mentioned below. Additional macroinvertebrate sorting and identification training was provided for the principal investigator with the sponsorship of University College Dublin (UCD) in collaboration with the Ethiopian Institute of Water Resources (EIWR), Addis Ababa University (Certificate annexed). The data collection activities were checked by the principal investigator, the sites were visited and some of the data collection procedures were guided and supported by the supervisors (Associate professors Mary Kelly-Quinn and Seid Tiku).

Furthermore, the calibration and maintenance of equipment were checked before data collection according to the guidelines specified in the manu405060facturer manuals. For example, pH buffer solutions, donated from UCD were used to calibrate the pH meter (Donated materials are annexed). The identified macroinvertebrates were also checked by the supervisor (Associate professor Mary Kelly-Quinn) (Images of macroinvertebrates are annexed). All data collectors were also experienced experts in macroinvertebrate data collection and additional training was given to the collectors. To avoid differences in macroinvertebrate data collection efforts, kick sampling was done by the principal investigator. Macroinvertebrate sorting and identification were done in the laboratory using a headlight, dissection microscope, and hand lenses to avoid sorting and identification errors.

3.4. Results

A total of 56,930 macroinvertebrates belonging to 78 families and 15 orders were collected using the four methods. Ephemeroptera 29,817 (52.37%), Coleoptera 6,243 (10.97%), Trichoptera 6150 (10.80%) and Diptera 5,405 (9.47%) were the four most abundant orders of macroinvertebrates present. The various metrics calculated are given in Table 12-3 below.

Figure 12-3: Descriptive summary, Mean±SE, mean ranks (MR), and Kruskal-Wallis (K-W) test with its P-value of macroinvertebrate metrics for each kick sampling method. SE=standard errors, H'=Shannon Index, D=Simpson Index, ASPTBMWP =Average score per taxon from BM

Metrics	Descriptive summary for each method				MR for each method and K-W test (H) and P-values					
	2-min RH (n=20)	2-min MH (n=60)	3-min RH (n=20)	3-min MH (n=60)	2-min RH	2-min MH	3-min RH	3-min MH	H (DF =3)	P-value
Richness (family)	32.05±0.94	33.17±0.65	33.9±1.09	32.07±0.53	77.28	94.93	82.87	74.4	3.25	0.355
Abundance	347.2±4.42	334.4±3.08	392.55±6.4	367.88±4.55	72.33	126.58	55.38	92.98	42.4	0.001*
EPT richness	6.8±0.2	7.55±0.18	7.05±0.21	7.42±0.15	60.28	71.83	86.43	84.2	6.33	0.097
EPT (%)	66.9±1.04	66.5±0.85	66.7±1.14	67.48±0.72	78.3	76.58	78.83	84.21	0.65	0.885
Eph. richness	4.05±0.05	4.1±0.04	4.1±0.07	4.1±0.04	77	81	81	81	0.51	0.916
Eph. abundance	181.75±5	175.4±3.05	205.7±6.33	192.4±4.13	75.13	111.1	63.36	89.23	19.34	0.001*
BMWP	150.05±4.1	158.2±2.86	161.95±4.9	153.95±2.58	70.4	96.6	84.12	74.88	4.62	0.202
$ASPT_{BMWP}$	4.69±0.05	4.79±0.04	4.79±0.04	4.81±0.03	65.33	82.28	79.26	86.21	3.13	0.372
Evenness	0.77±0.00	0.78±0.01	0.77±0.00	0.77±0.00	77.13	78.28	82.89	79.98	0.32	0.956
Shannon index	2.66±0.03	2.71±0.03	2.71±0.03	2.67±0.02	74.83	89.73	82.97	76.85	1.64	0.651
Simpson index	0.90±0.00	0.90±0.00	0.91±0.00	0.90±0.00	77.58	91.03	78.78	79.69	1.21	0.75
ETHbios	129.3±4.63	130±2.72	136.9±4.8	127.53±2.34	82.6	97.5	79.62	75.02	3.6	0.308
$ASPT_{ETHbios}$	4.03±0.06	3.93±0.05	4.04±0.06	3.98±0.03	84.58	91.45	75.67	80.33	1.93	0.588
FBI	4.76±0.04	4.78±0.03	4.72±0.03	4.70±0.02	84.95	73.25	90	71.93	5.25	0.155
Predators (%)	17.35±0.65	19.22±0.53	18.66±0.66	19.32±0.58	57.95	76.85	83.13	86.61	6.1	0.107
Scrapers (%)	21.44±0.49	20.7±0.48	21.44±0.38	21.48±0.47	88.3	86.93	72.64	83.62	2.95	0.4
Collectors (%)	47.17±0.68	46.23±0.72	46.31±0.68	44.94±0.59	94.33	84.65	82.43	72.58	3.8	0.284
Filterer (%)	13.57±0.32	13.24±0.33	13.12±0.35	13.76±0.39	88.58	77.03	75.66	83.81	1.68	0.641
Shredders (%)	0.48±0.1	0.64±0.08	0.48±0.11	0.50±0.06	77.3	71.93	87.8	77.13	2.64	0.45

*Significant differences at $p < 0.05$. DF- Degree of freedom

The Kruskal-Wallis test did not show any significant difference (Figure 12-2, Table 12-3) among the methods for all metrics except total abundance (H = 42.4, p = 0.001, DF = 3) and Ephemeroptera abundance (H = 9.34, p = 0.001, DF = 3) (Figure 3-3, Table 3-3). For example, the mean (± standard error) for richness (Figure 13-3) did not show any difference among the methods.

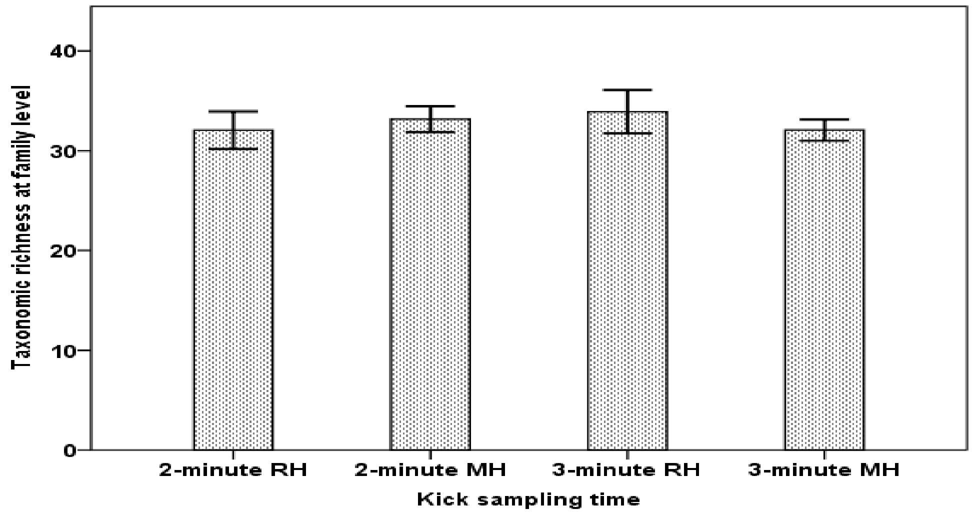

Figure 13-3: Comparison of the mean (± standard error) of selected benthic macroinvertebrate metric (family level taxonomic richness) for the four kick sampling methods using bar charts and standard error bars examined in the dry season, 2019

For significantly different metrics, the pairwise *post hoc* test indicated significant differences in total abundance among all method pairs except for 2-minute RH and the 2-minute MH pair (Figure 14-3, Table 12-3).

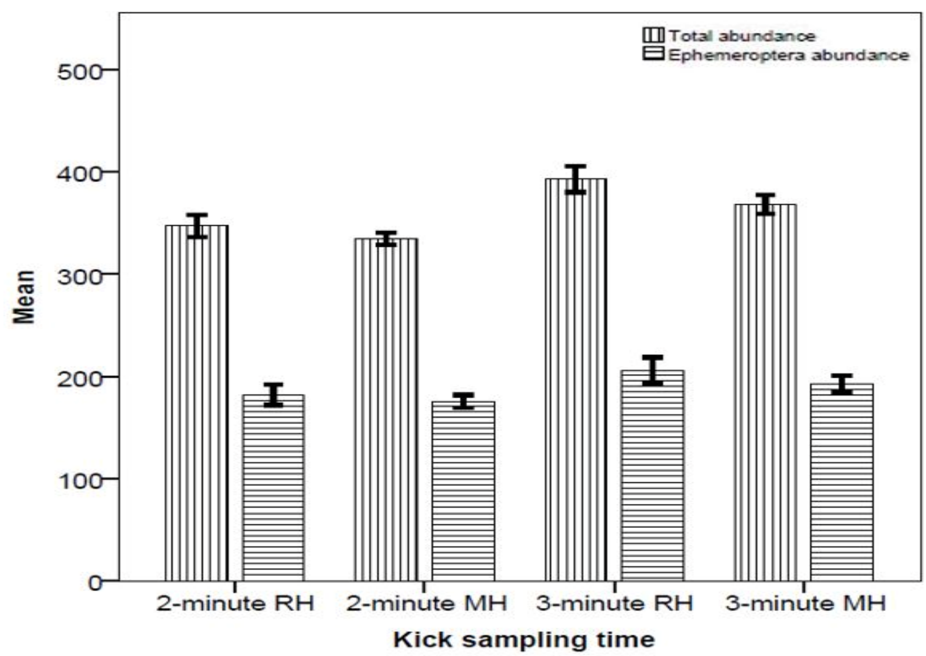

Figure 14-3: Comparison of the mean (± standard error) of total abundance and Ephemeroptera abundance for the four kick sampling methods using bar charts and standard error bars examined in the dry season, 2019

Similarly, the post *hoc* test on Ephemeroptera abundance showed significant differences between 2- and 3-minute RH, 2-minute MH and 3-minute RH, and 2- and 3-minute MH (Figure 14-3, Table 12-3).

Table 12-3: Values in the pairwise post hoc test calculated for macroinvertebrate total abundance (lower left) and Ephemeroptera abundance (upper right) with significant differences in the Kruskal-Wallis test

		2-min RH	2-min MH	3-min RH	3-min MH		
Total abundance	2-min RH	-	0.241	0.006**	0.22	2-min RH	Ephemeroptera abundance
	2-min MH	0.051	-	<0.001**	0.003**	2-min MH	
	3-min RH	<0.001*	<0.001*	-	0.081	3-min RH	
	3-min MH	0.033*	<0.001*	0.002*	-	3-min MH	

**Significant differences in total abundance; and **significant differences in Ephemeroptera abundance*

The ANOSIM test based on the relative abundance of macroinvertebrate communities also showed no significant difference among the four kick sampling methods (R = 0.048, P = 0.972, ANOSIM). Furthermore, the two-way ANOSIM test, sites versus methods, did not show any significant difference (p = 0.214) although the global R was low (R = 0.022; ANOSIM). Similarly, the MDS ordination plot (Figure 15-3) from the relative abundance of macroinvertebrates with a high-stress value of 0.24 (Clarke, 1993) as well showed no separation of the samples based on the method applied.

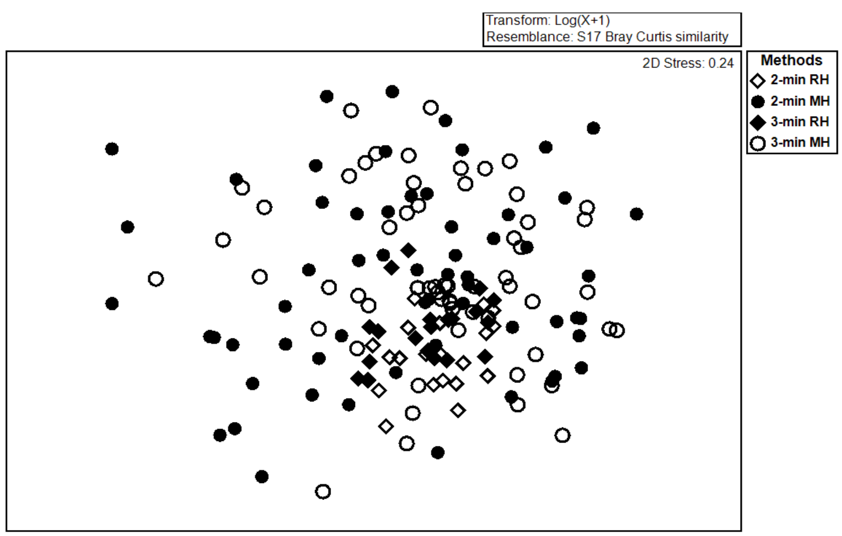

Figure 15-3: MDS plot of the macroinvertebrate community structure of the samples from the various sampling methods based on log(x+1) transformed Bray-Curtis similarity matrix

However, the MVDISP (index of multivariate dispersion) given in Table 13-3 showed variations in the relative abundances of macroinvertebrate communities among the methods. The dispersion sequence of 0.34, 0.341, 0.945, and 1.197 for 3-min RH, 2-min RH, 3-min MH, and 2-min MH, respectively, showed that the average rank dissimilarity is almost 3.5 times higher within 2-min MH samples than for 2- and 3-min RH samples. The 2-and 3-minute RH replicates were almost similar and the least dispersed compared to both 2- and 3-minute multihabitat replicates. Likewise, the similarity percentages were similar for 2- and 3-minute riffle habitats (Table 13-3). The lower the MVDISP value, the less dispersed the samples within the factor, and the greater the SIMPER, the more similar the samples within the factor (Wasserman *et al.*, 2015).

Table 13-3: MVDISP (index of multivariate dispersion) and SIMPER test results, respectively, represent the relative dispersion and similarity of samples within each method (factor)

Method	MVDISP	SIMPER (%)
3-min RH	0.34	82.43
2-min RH	0.341	82.21
3-min MH	0.945	77.51
2-min MH	1.197	75.42

The RELATE analysis computed from the relative abundance of macroinvertebrate communities further indicated a similarity in macroinvertebrate community relative abundances recorded between all pairs of methods with a Spearman rank correlation coefficient (ρ) of greater than 0.5 and $P = 0.001$. Amongst all pairs, 2- and 3-minute RH samples and 2- and 3-minute MH samples showed the highest similarity in macroinvertebrate community structures recorded (Table 14-3).

Table 14-3: The RELATE analysis results indicate the Spearman's rank correlation coefficients (ρ) between different biotic resemblance matrices

Time (t)	2-min RH	3-min RH	2-min MH	3-min MH
2-min RH	—			
3-min RH	0.76 (P = 0.001)	—		
2-min MH	0.50(P=0.001)	0.61 (P=0.001)	—	
3-min MH	0.52 (P = 0.001)	0.63 (p = 0.001)	0.80 (p =0.001)	—

Finally, the analysis of similarities between benthic macroinvertebrate community structure had a classification strength-sampling method comparability (CS-SMC) close to 100% for all pairs of methods (Figure 16-3) which highlighted that the similarity of samples between methods was very high. The trend line in Figure 16-3 illustrated that method pairs such as 2- and 3-minute RH, 2-minute RH, and 2-minute MH, and 2-minute MH and 3-minute RH had better CS-SMC scores (close to 100%) compared to the other method pairs. Method pairs between 2-min MH and 3-min MH showed a CS-SMC value greater than 100% (Figure 16-3) indicating a greater similarity between the methods than within the methods.

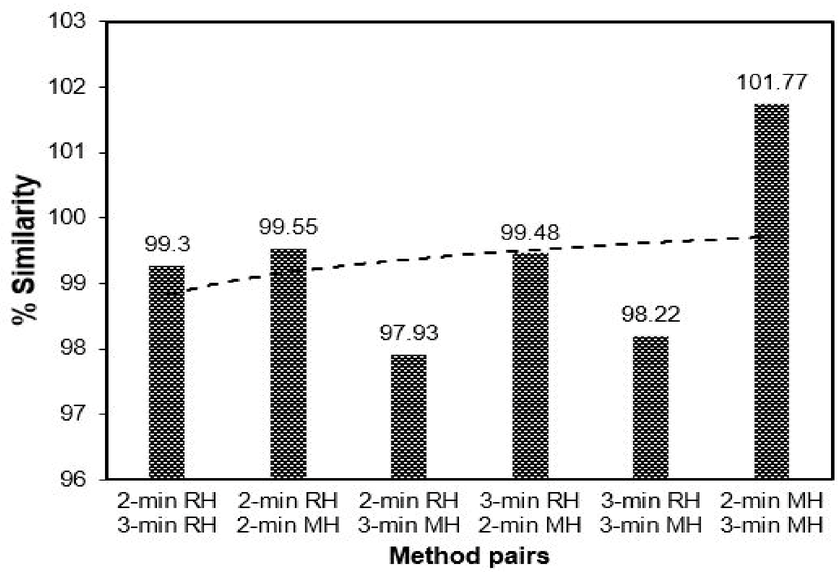

Figure 16-3: The similarity between replicate samples within each method relative to the similarity of samples between methods as calculated by CS-SM

3.5. Discussion

This study set out to develop a fixed-time sampling protocol as a method for bioassessment of streams and rivers using macroinvertebrates in Ethiopia. Comparisons among methods can be made at several levels of data organization: taxonomic composition, relative abundances, metrics, indices, and bioassessment endpoints (e.g., good-fair-poor) (Blocksom et al., 2008). Four kick sampling methods (2- and 3-minute RH, 2- and 3-minute MH) were selected and compared using various statistical analysis methods. Based on the Kruskal Wallis test (H), the majority of metrics compared were similar probably because of some reasons. According to Blocksom et al. (2008), the predominant habitat tends to be riffles in higher gradient streams and a sample that is collected using the single habitat method that focuses on riffle habitats should be very similar to the sample that is collected using the multiple habitat method, in which habitats are sampled according to their proportional representation in the stream. In addition, the lack of differences among the methods could be related to inadequate sampling and the level of taxonomic resolution. The only exceptions included total abundance and Ephemeroptera abundance, which differed significantly between methods. The differences may probably be due to the larger total number of macroinvertebrates and Ephemeroptera families in the 3-minute RH and 3-minute MH samples.

Similarly, diversity indices, biotic scores, and functional feeding group attributes evaluated for variability are in support of future biomonitoring using these 4 methods. This finding was in agreement with Friberg et al. (2006) who found that a sampling method with a lower sampling effort, such as the 2-minute, achieves an equal sampling effort as 3-minute in terms of macroinvertebrate community structure. Furthermore, Mykra et al. (2006) indicated similarities in community structure among different sample sets collected for 1-, 2-, 3-, and 4-minutes showing that even 1-minute samples would yield adequate and comparable taxonomic structures. Similarly, Feeley et al. (2012) found that 20- and 60-second MH kick sampling methods displayed a reasonable representation of the taxa in terms of various metrics tested. Likewise, the ANOSIM test did not show any overall difference among the four methods and highlighted the high similarity in macroinvertebrate community structure across all sites sampled using the methods tested. In contrast, a related study (Feeley et al., 2012) found a high similarity but significant differences between 20- and 60-second kick sampling methods.

The RELATE analysis on relative abundances of macroinvertebrate communities showed that samples from similar habitats but collected by different kick sampling methods had higher correlation coefficients than those sampled from different habitats by either different or similar kick sampling methods. For example, 2- and 3-minute RH pair matrices and 2- and 3-minute MH pair matrices had Pearson correlation coefficients of 0.76 and 0.80, respectively. This probably highlights that more similar macroinvertebrate taxa had been collected from similar habitats regardless of the different methods used. This relationship has been well described by Khudhair *et al.* (2019) where sediment type, water flow, and presence of aquatic vegetation types directly affect the assemblage structure of benthic macroinvertebrates. The other reason for this high similarity among the methods may be related to the family-level identification used. Although the family-level identification is adequate for bioassessment of water quality determination (Corkum, 1989; Chessman, 1995; Wright et al., 2000), Bailey et al. (2001) concluded that organisms identified to genus or species level provide a significantly more informative description of conditions in the stream than higher taxonomic levels.

The CS-SMC measure also showed high similarity scores (close to 100%) among the method pairs indicating that all kick sampling methods represented almost identical macroinvertebrate community structures from MH and RH samples. This finding was in contradiction to Cao *et al.* (1998) and Cao *et al.* (2002) who found that sample size is important but they identified some taxa to lower levels (species, genus, and family levels). They noted that increasing sample size can affect the relative abundance of macroinvertebrates in a sample and thus the analysis of community structure. However, Mykra et al. (2006), who sampled only riffle habitats, showed that sample size is more important to detect rare and endangered macroinvertebrate taxa and concluded that the 2-minute samples are sufficient for most biodiversity purposes.

In the current study, several analytical tests, for example, the Kruskal-Wallis, ANOSIM, RELATE, and CS-SMC showed no significant differences among the four methods indicating that the low sampling effort, such as those of the 2-minute RH and 2-minute MH method, perform equally and often better, in terms of the numbers of taxa and abundances recorded compared to that of methods with a higher sampling effort (3-minute RH and 3-minute MH). However, although the stress value of the MDS ordination plot among the four methods was higher compared to the Clarke (1993) cutoff value, it appears that the sample replicates using the MH methods (2 and 3-minute) were highly dispersed compared to the sample replicates using the RH methods (2- and 3-minute). The differences were well reflected by the numerical analyses of MVDISP and SIMPER in terms of dispersion and similarity percentage of macroinvertebrate samples. The MVDISP analysis showed that the 2- and 3-minute RH have an equal and low index of multivariate dispersion (IMD) and high within similarity (SIMPER). In contrast, high IMD and lower within similarity (SIMPER) were observed in multihabitat replicate samples collected by 2-and 3-minute (Table 3-5). According to Warwick and Clarke (1993), there are two potential sources of increased variability among replicate samples: (1) because of an increase in the variability of abundances of the same set of taxa, and (2) due to changes in species identities.

As detected in MDS and from the MVDISP analysis, it appears that the 2- and 3-minute RH kick samples were much less variable and this would enhance biomonitoring by being better able to detect impacted sites as opposed to the highly variable 2- and 3-minute MH samples. Based on the findings in this study, 2- and 3-minute RH kick sampling methods in terms of time and habitat type for the collection of macroinvertebrate data would be key considerations in the country. However, the extra time spent for kicking such as 3 minutes in RH was an effort spent increasing the abundances of macroinvertebrate taxa already sampled. This unnecessary increase in the abundance of macroinvertebrates collected by the extra time for each sample will increase the identification effort (Feeley et al., 2012) and costs which are directly related to the amount of material and the number of individuals collected (Barbour and Gerritsen, 1996). Although not examined in this study, the taxonomic resolution also plays a major role in the costs associated with sample processing (Vlek *et al.*, 2006).

In summary, if the focus is not on rare taxa and the required information is not dependent on additional evidence provided by the use of lower taxonomic levels of identification, the results in this study support the use of the shorter 2-minute RH kick samples, also used by other studies (Heino et al., 2003; Paavola *et al.*, 2003; Mykra et al., 2006), for bioassessment of wadeable rivers and streams in Ethiopia. Future studies should explore the effectiveness of lower sampling time, such as 1-minute kick samples across a wider range of wadeable stream and river types, and also the effect of variable levels of taxonomic resolution, not just for bioassessment of water quality but also for aquatic invertebrate biodiversity assessment, an issue that is becoming increasingly important with alarming losses of freshwater biodiversity in many parts of the world (Reid et al., 2019). Furthermore, as a suggestion, future studies would include impacted sites to see how the methods perform there as well.

3.6. Conclusions

A method that takes the minimum duration of time and effort but produces an equitable representation of the taxa available would be eventually adopted in the assessment of streams and rivers (Mykra et al., 2006;

Buss and Borges, 2008; Feeley et al., 2012) based on the ecological contexts of the region. The unnecessary increase in abundance, as found with 3-minute RH kick samples, will considerably increase the identification effort for each sample. Sampling for a longer time duration increases effort and fatigue (Mykra et al., 2006; Feeley et al., 2012) and reduces concentration and the productivity of the operators, in turn potentially affecting the results (Feeley et al., 2012). Consequently, the optimum macroinvertebrate kick sampling method, which enables rapid but accurate bioassessment of water quality in terms of time and habitat type, is an important consideration in Ethiopia. Based on the results in the present study, the shorter 2-minute RH kick sampling method could be a good candidate for the bioassessment of wadeable rivers and streams in Ethiopia but further testing across a pollution gradient is recommended.

Chapter Four: Impacts of the Koka Hydropower Dam on Macroinvertebrate Assemblages in the Awash River Basin in Ethiopia

Adapted from:

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Chapter 4: Impacts of the Koka Hydropower Dam on Macroinvertebrate Assemblages in the Awash River Basin in Ethiopia

4.1. Abstract

The Koka Hydropower Dam is one of the oldest large dams in Ethiopia. Damming is one of the anthropogenic activities impacting the distribution of aquatic life forms. However, to date, little attention has been focused on the dam's impacts on the river macroinvertebrate assemblages in Ethiopia. The objective of this study was, therefore, to assess the impacts of the Koka Hydropower Dam on macroinvertebrate assemblages in the Awash River basin in Ethiopia. In the three river reaches on the Awash River (upstream near the source of the river, midstream above the dam, and downstream below the dam), a total of 15 sites were selected for sampling. The statistical analysis tested the null hypothesis that there are no differences in macroinvertebrate assemblage patterns or a range of univariate metrics between the three river reaches. Additional analyses involved the identification of taxa responsible for significant differences in macroinvertebrate structure and an exploration of the variables that structure macroinvertebrates (e.g., CCA). In the upstream, midstream, and downstream reaches of the Awash River, we recorded a total of 73 taxa belonging to 43 families and 12 orders. Trichoptera was the dominant order in the upstream river reach, whereas Diptera dominated the midstream and downstream river reaches. The diversity of macroinvertebrates decreased from upstream to midstream and downstream. The three river reaches differed significantly in Shannon and Simpson diversity indices, % EPT, EPT abundance, total richness, evenness index, %collectors, and %scrapers. In this study, it was observed that macroinvertebrate assemblage differences and spatial patterns were significantly associated with values of river flow changes, phosphate concentration, and substrate index. The findings of this study have broad implications for the assessment of the impacts of dam construction on rivers of the studied region in the future.

Keywords: macroinvertebrates, environmental factors, substrate index, taxonomy-based metrics, damming, flow regimes

4.2. Introduction

Dams are structures that restrict water flow and create reservoirs to provide specific human needs (Schmutz and Moog, 2018; Winton *et al.*, 2019) such as easy navigation, lessen flooding, and water supply (hydropower generation and irrigation, and potable water) (Schmutz and Moog, 2018; Zhang and Gu, 2023). Hydropower dams are among the most detrimental anthropologic activities in river basins by altering the physiography of watersheds and flow regimes, presenting barriers to fish migration and sediment movement, and impacting other aquatic species and their habitats (Schmutz and Moog, 2018; Englmaier *et al.*, 2020). Zhang and Gu (2023) noted that dams are among the earliest types of anthropogenic infrastructure and have made significant contributions to economic progress throughout human history. According to Schmutz and Moog (2018), there are currently more than 58,400 large dams being constructed, operated, or planned worldwide. The World Commission on Dams (WCD, 2000) defined large dams as those that are "at least 15 meters high from the lowest foundation to the crest, or a dam between 5 and 15 meters high impounding more than 3 million cubic meters".

Since the 1970s, dam construction has decreased in the vast majority of developed countries such as North America, Europe, and Oceania, whereas it has increased in developing countries in Africa, Asia, and South America (Zhang and Gu, 2023). Emerging nations, such as China and India, have tremendous hydropower potential as well (Gernaat *et al.*, 2017). WCD (2000) estimated that 160–320 new large dams are constructed annually around the world. Ethiopia, located in Africa, has the second-largest population after Nigeria (Hagos *et al.*, 2022) and depends on rain-fed agriculture with limited coverage of electricity for many communities (Hagos et al., 2022). To address electricity and water demands, the Ethiopian government is building dams on major rivers (Hagos et al., 2022). The construction of the Grand Ethiopian Renaissance Dam, Africa's largest, on the Nile River Basin is one of approximately 200 places in the country where hydropower development is feasible (Degefu *et al.*, 2015; Hagos et al., 2022). The Koka Hydropower Dam, abbreviated KHD, was built in the 1960s on the Awash River and is one of Ethiopia's oldest dams (Degefu et al., 2015; Bussi et al., 2021).

Natural variations in river flow regimes related to climatic conditions, geology, and geography of the watershed are important for the long-term ecological integrity of the river ecosystems (Poff *et al.*, 1997). However, alterations to natural flow regimes have far-reaching ecological as well as social and geopolitical consequences (WCD, 2000). According to Poff et al. (1997), changes in the natural flow regimes are the key driver of river ecosystem structure and influence sediment transport, capturing 25–30% of pre-disturbance discharge (Schmutz and Moog, 2018). Schmutz and Moog (2018) also identified that damming affects 48% of rivers worldwide, and the impacts might even trickle far downstream and compromise the health of the river ecosystem (Degefu et al., 2015). In developing nations such as China, the impact has increased rapidly (Wang and Chen, 2010). Overall, dams are of concern as they modify the natural flow regimes and morphodynamical patterns of rivers, adversely affecting productivity, biodiversity, and ecosystem services in the downstream river and also in the associated reservoir (Nilsson *et al.*, 2005; Mbaka and Wanjiru, 2015; Ko *et al.*, 2020). Dams are also criticised because of the large area that the reservoirs cover and are hence responsible for the displacement of human communities from their original places. These problems are more severe in developing countries such as Ethiopia because of the lack of enough financial resources to equitably resettle people displaced by the projects (Reis *et al.*, 2011; Degefu et al., 2015).

Using biological metrics derived from aquatic organisms, it is possible to assess the ecological health of rivers (McRae *et al.*, 2017; Deinet *et al.*, 2020), particularly to explore human-induced disturbances such as dams (Poff et al., 1997). Macroinvertebrates represent a diverse group of relatively long-living taxa with limited mobility that react strongly and often predictably to human influences on aquatic systems (Cairns and Prall, 1993). As mentioned, many hydropower dams have already been built in Ethiopia, and many more are being planned (Degefu et al., 2015). However, limited studies in the country have investigated the impact of hydropower dams on benthic macroinvertebrate assemblages. The goal of this study was, therefore, to assess the impacts of the KHD on macroinvertebrate assemblages along the Awash River in Ethiopia. To achieve this goal, (1) we assessed the responses of macroinvertebrate assemblages to river damming; (2) we identified the key environmental factors that significantly affect macroinvertebrate assemblages along the Awash River. In view of the above objectives, we hypothesized that there are no macroinvertebrate assemblage differences between river reaches upstream, midstream, and downstream of the KHD.

4.3. Materials and methods

4.3.1. Study area and sampling sites

The Awash River basin, which covers an area of 112,696 square kilometers, is the most important and industrialized catchment in Ethiopia (Englmaier et al., 2020). The Awash River in this basin originates near Entoto Mountain (Bussi et al., 2021) and drains into Lake Abe on the Ethiopia-Djibouti border by crossing 1,250 kilometers northeast across the KHD (Englmaier et al., 2020). The KHD, built in the 1960s, is 42 meters high and has 42 megawatts of installed capacity (Degefu et al., 2013). Three sampling sections were defined with 5 sites in each: upstream (US1-US5, near the source of the river), midstream (MS1-MS5, above the KHD), and downstream (DS1-DS5, below the KHD) river reaches (Figure 17-4).

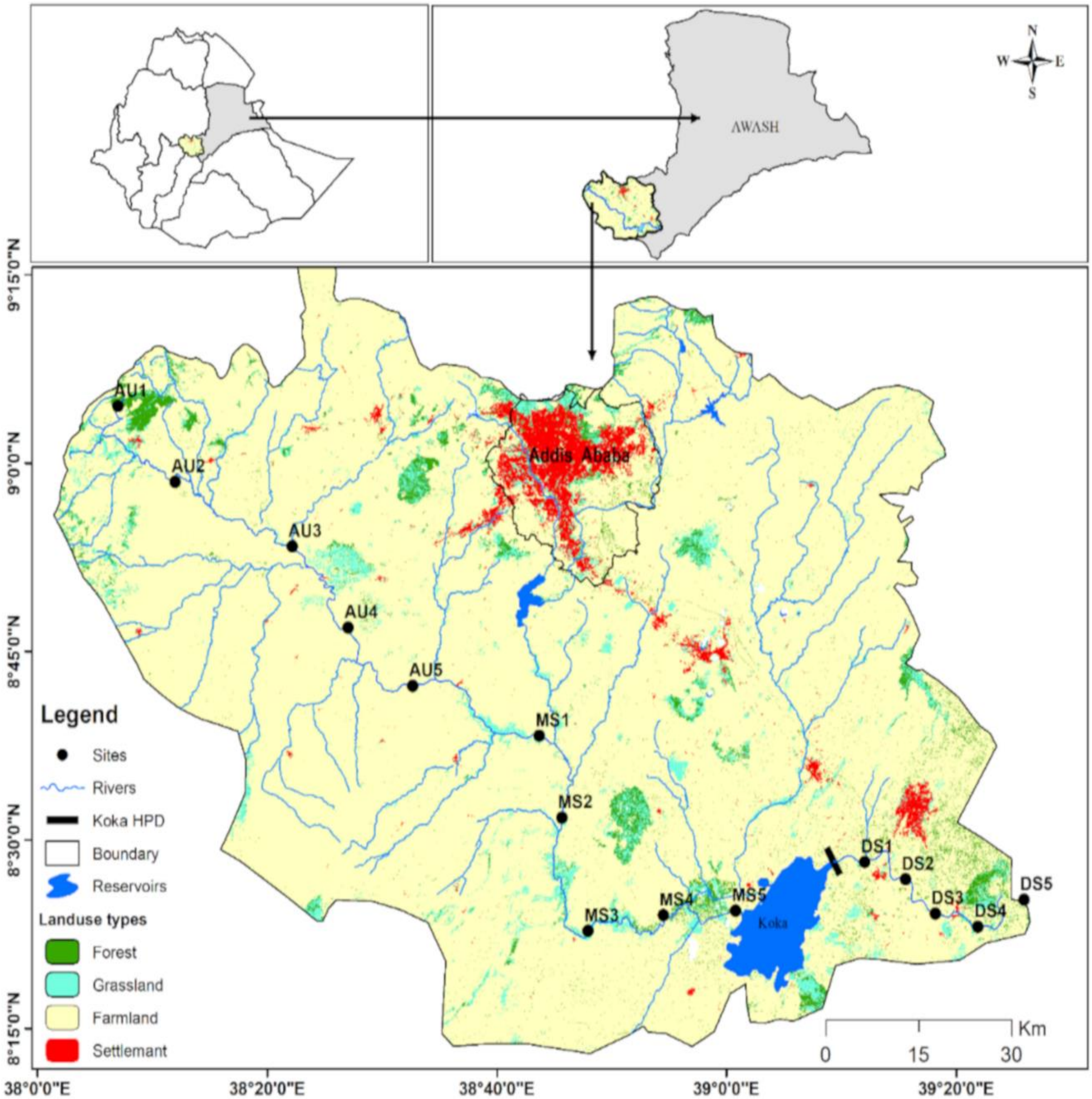

Figure 17-4: The locations of the study area, sampling sites, Koka Reservoir, KHD, rivers, and the main land use categories along the upper Awash River basin in central Ethiopia (GIS 10.5)

4.3.2. Macroinvertebrate data collection, sampling, and identification

At each of the 15 sites, macroinvertebrate samples were collected in the dry season (from the middle of March to the middle of April), as this is a period when water levels are sufficiently low to enable sampling (Montana, 2012). According to Barbour et al. (1999)_ENREF_4, the most effective sampling season of the year is an important consideration for selecting an index period. In Ethiopia, there is a wet season (June to September) and a dry season (October to May), and almost all previous studies in Ethiopia sampled surface water during the dry season (Degefu et al., 2013; Englmaier et al., 2020). Samples of macroinvertebrates were collected using a rectangular frame pond net (20 x 30 cm) with a mesh size of 0.5mm (Barbour et al., 1999). A 2-minute kick sample was collected along a 10-meter stretch of river at each site (Getachew *et al.*, 2022b). The samples, preserved in 80% ethanol in plastic bags, were taken to the laboratory, where they were rinsed through 0.5 mm mesh sieves. Macroinvertebrates were sorted into broad taxonomic groups for later identification and stored in 80% ethanol. Each macroinvertebrate specimen was then identified at the lowest possible taxonomic level (usually genus or species). Based on similar climatic conditions relative to other non-tropical regions, we used identification keys developed for South Africa (Harrison, 2009; Lowe, 2009; Schael, 2010).

4.3.3. Environmental data collection

The river depth (m) and river velocity (m s^{-1}) were measured at each site. Out of the three depth measurements recorded at equal intervals across the wetted river bank width, these were used to determine the maximum river depth for use in the statistical analysis. Surface water velocity was measured using the techniques for estimating stream velocity as described in the United States Environmental Protection Agency (USEPA) protocol (USEPA, 2017). The spatial locations and altitudes of the study sites were recorded using a global positioning system (GPS). At each site, the percentage compositions of the benthic substrates were estimated using the following particle size scales: sand (0.06-2 mm in diameter), fine gravel (2–10 mm), gravel (10–64 mm), cobble (64–256 mm), boulder (>256 mm), and bedrock (solid rock surfaces) (Jowett and Richardson, 1990). Substrate measurements were then transformed into a single variable by summing weighted substrate percentages to form a substrate index (SI). SI = 0.08% bedrock + 0.07% boulder + 0.06% cobble + 0.05% gravel + 0.04% fine gravel + 0.03% sand. The optimum substrate index for many invertebrates was just under cobble size (SI = 6) (Jowett and Richardson, 1990). Dissolved oxygen (DO), pH, temperature, and electrical conductivity (EC) were measured in situ on the same date using a digital handheld portable multi-parameter (HACH HQ40D probe). The turbidity of water was measured using a turbidity meter. In the field, 1.0 liter of the water sample was collected from each site using acid-cleaned polyethylene containers at a depth of 20–30 cm and fixed with 3 drops of concentrated sulfuric acid. The samples were then transported to the laboratory for analysis. In the laboratory, nitrate-nitrogen (mg L^{-1}) and phosphate (mg L^{-1}) concentrations and biochemical oxygen demand (BOD_5) were analysed using the standard methods (APHA, 2005). To avoid disturbance of sites before sampling, measurements were undertaken from the downstream to the upstream direction.

4.3.4. Metric selection

Metrics that are relevant to the ecology of rivers within the region were selected to address several attributes of macroinvertebrate assemblages after Barbour et al. (1999). These included diversity measures such as evenness, Simpson, and Shannon indices (Nathan *et al.*, 2017), composition measures, including total taxon richness, EPT richness, and %EPT abundances (Lenat and Penrose, 1996), and functional approaches that reflect ecological integrity based on the information on both the structure and function of aquatic ecosystems (Cummins, 1974; Sumudumali and Jayawardana, 2021). We determined the functional traits for each taxon using the functional feeding groups in Dudgeon (1999), Merritt et al. (2002), and Merritt et al. (2006) (Table 15-4).

Table 15-4: Selected metrics used to assess the water quality of the Awash River and the impacts of the KHD and their description

Metrics	Description	Metric type	References
% EPT	% EPT families in an ecosystem	Composition	Lenat and Penrose (1996)
EPT taxa	Total number of families belonging to the EPT orders	Composition	Lenat and Penrose (1996)
Total taxa	Number of different taxa represented in an ecological community	Composition	Nathan et al. (2017)
Evenness	The similarity of frequencies among units making up a population	Diversity	Nathan et al. (2017)
Simpson Index	A metric accounting for species richness and their relative abundance	Diversity	Nathan et al. (2017)
Shannon Index	A metric accounting for the total number of species and their distribution	Diversity	Nathan et al. (2017)
% Collectors	% of organisms that collect FPOM from the stream bottom	Function	(Merritt et al., 2002)
% Predators	% of organisms that feed on other consumers or capture live prey	Function	(Merritt et al., 2002)
% Shredders	% of animals feed on coarse terrestrial plant litter or aquatic macrophytes	Function	(Merritt et al., 2002)
% Filterers	% of organisms feed on FPOM from the water column by a variety of filters	Function	(Merritt et al., 2002)
% Scrapers	% of organisms feed on live plants & particulates from substrate surfaces	Function	(Merritt et al., 2002)

EPT: Ephemeroptera, Plecoptera, and Trichoptera

4.3.5. Data analysis

4.3.5.1. Biological data analysis

Univariate and multivariate tests were carried out using data on the composition, diversity, and functional measures of the macroinvertebrate assemblages. Under the assumption that the samples were selected at random and the observations were independent of one another, the Kruskal-Wallis test was used to determine whether there were significant differences among the three river reaches in terms of the univariate metrics (Table 15-4). The *post hoc* analysis, using the Mann-Whitney U test in SPSS version 27, was performed on statistically significant ($p < 0.05$) metrics from the Kruskal-Wallis test.

We used a one-way analysis of similarity (ANOSIM) to test the null hypothesis that there were no macroinvertebrate assemblage differences at the locations of the upstream, midstream, and downstream river reaches. Macroinvertebrate data were transformed to log (x+1) to increase homoscedasticity, normality, and linearity (Clarke *et al.*, 2008). The degree of group separation for those samples represented by the similarity matrix can be assessed using the global R statistic (Clarke et al., 2008). When R is closer to 1, there is good separation among the groups, and a value closer to zero shows weak separation of the groups (Clarke and Gorley, 2006). Similarity percentage (SIMPER) was used to identify the taxa that accounted for the most dissimilarity among the groups when significant differences in macroinvertebrate assemblages were found. Non-metric multidimensional scaling (NMDS) was used to visualise the separation of the sites in the three river reaches. According to Clarke and Gorley (2006), an NMDS ordination with a stress value of 0.1 is generally regarded as fair, and values below this indicate an excellent match. Furthermore, hierarchical cluster analysis of samples, known as a linkage tree, was carried out based on the macroinvertebrate assemblage data (Clarke et al., 2008). Cluster analysis explores groups that naturally occur within a data point and does not require the organization of data points into any predefined groups, which we also utilised to depict the groupings of the sites throughout the river reaches (Clarke et al., 2008). PRIMER software was used to implement all of these multivariate analyses (Clarke and Gorley, 2006).

4.3.5.1. Environmental data analysis

We used the Kruskal-Wallis test to detect differences in the environmental variables between the three reaches to obtain comparable results with biological data. Finally, the relationships between environmental and biological variables were assessed using CANOCO 4.5. After preliminary Detrended Correspondence Analysis (DCA), we determined Canonical correspondence analysis (CCA) to be the most appropriate analysis based on the gradient length (Ter Braak and Wiertz, 1994). Except for pH, all environmental variables were square-root transformed (Clarke and Gorley, 2006). To determine whether the variables had a significant influence on the distribution of macroinvertebrates, forward selections of environmental factors were used in the CANOCO model with 499 Monte Carlo permutations. The statistical significance of the eigenvalues (λ) and the taxon-environment correlations generated by the CCA were tested using Monte Carlo permutations with a p-value of <0.05.

4.4. Results

4.4.1. Environmental variables

The values of measured environmental variables and their statistical significance across the three river reaches are presented in Table 16-4. The midstream reach had the deepest water depths (minimum and maximum), whereas water velocity was, as expected, highest in the upper reaches, progressively decreasing in a downstream direction. The same applied to conductivity as well as water temperature, which varied from an average of 16.4 ^oC in the upper reach to 23.7 ^oC downstream of the dam. Oxygen concentrations were relatively low (mean value < 8 mg L^{-1}) throughout the sites, with average values decreasing from the upstream to midstream and downstream reaches. BOD was low (2.78±0.55 mg O_2 L^{-1}) in the upper reaches, but values in the midstream and downstream reaches were 13.44±3.20 and 13.94±1.31 mg O_2 L^{-1}, respectively. The mean concentration values of phosphate and nitrate at midstream were higher than downstream of KHD, which might be due to their being absorbed in the reservoir (Table 16-4).

Table 16-4: The minimum (Min), maximum (Max), mean ± standard error (SE) of measured environmental variables, and mean rank and the Kruskal-Wallis test (K-W H test) among the upstream, midstream, and downstream river reaches on the Awash River in Ethiopia

Variable	**Upstream**			**Midstream**			**Downstream**			**Mean Rank**			**K-W test**
	Min	Max	Mean±SE	Min	Max	Mean±SE	Min	Max	Mean±SE	US	MS	DS	
River depth (m)	0.33	0.37	0.35±0.01	0.39	0.42	0.40±0.00	0.29	0.32	0.31±0.01	13.00	8.00	3.00	**12.50***
Velocity (m s^{-1})	0.40	0.46	0.42±0.01	0.29	0.38	0.34±0.02	0.15	0.22	0.18±0.01	13.00	8.00	3.00	**12.50***
SI	5.79	7.33	6.62±0.26	5.42	5.68	5.60±0.05	3.54	4.02	3.83±0.11	13.00	8.00	3.00	**12.52***
Temperature (^{O}C)	15.00	17.30	16.40±0.43	17.64	21.62	18.88±0.72	21.6	26	23.70±0.73	3.00	8.10	12.90	**12.28***
EC (μS cm^{-1})	32.00	154.0	101.20±20.6	193.0	564.0	405.20±83.3	505	563	532.20±12	3.00	10.10	10.90	**9.47***
Turbidity (NTU)	19.00	78.00	46.60±10.7	97.0	428.0	278.00±73.4	291	356	318.20±14	3.00	11.00	10.00	**9.50***
DO (mg L^{-1})	7.00	8.10	7.58±0.19	6.22	7.10	6.64±0.15	4.28	6.02	5.27±0.35	12.80	8.20	3.00	**12.02***
BOD_5 (mg L^{-1})	1.30	4.30	2.78±0.55	5.20	21.00	13.44±3.20	10	16.30	13.94±1.31	3.00	10.70	10.30	**9.41***
pH	6.80	7.50	7.12±0.12	7.30	8.38	7.88±0.22	7.7	8.08	7.83±0.07	3.40	10.50	10.10	**7.98***
NO_3-N (mg L^{-1})	0.86	4.01	2.21±0.59	4.21	7.88	5.58±0.71	0.7	3.25	1.77±0.47	6.10	13.00	4.90	**9.57***
Phosphate (mg L^{-1})	0.03	0.29	0.11±0.05	0.63	1.76	1.10±0.18	0.05	0.01	0.03±0.01	7.40	13.00	3.60	**11.18***

4.4.2. Macroinvertebrate assemblages

A total of 2,305 individuals belonging to 73 species or genera, 43 families, and 12 orders were collected (Table 17-4 and Table 4-S1). The upstream sites showed the highest total taxon richness (61 at Site US1), between 19 and 26 EPT taxa richness, and 56.98 to 71.10% EPT abundance. Sites in the midstream section had values between upstream and downstream river reaches. Conversely, the downstream river reach had the lowest total taxa richness (21-28), EPT taxa richness (7-8), and % EPT abundance (21.25-31.65%) (Table 17-4). Regarding the macroinvertebrate functional feeding groups (FFGs), %Collectors generally increased from the upstream sites to the downstream direction. On the other hand, other FFGs showed variable values at all river reaches. Among the 12 orders identified from the three river reaches, Trichoptera dominated at all sites of the upstream river reach, whereas Diptera dominated at the midstream and downstream sections (Table 17-4).

Table 17-4: Composition, diversity, functional measures, and relative abundances of orders of macroinvertebrate taxa (genera or species) at each sampling site in the upstream, midstream, and downstream river reaches of KHD in the Awash River basin in Ethiopia

Metric type	Metrics	Upstream reach					Midstream reach					Downstream reach				
		US1	US2	US3	US4	US5	MS1	MS2	MS3	MS4	MS5	DS1	DS2	DS3	DS4	DS5
Composition measures	Total taxa richness	61.00	53.00	46.00	46.00	48.00	43.00	40.00	36.00	34.00	33.00	25.00	21.00	23.00	27.00	28.00
	EPT richness	26.00	22.00	19.00	20.00	21.00	15.00	15.00	12.00	13.00	12.00	8.00	7.00	7.00	7.00	8.00
	%EPT abundances	66.12	65.36	71.10	61.26	56.98	51.39	44.36	46.49	40.19	32.32	21.25	31.65	27.38	22.77	31.50
Diversity measures	Evenness	0.92	0.92	0.89	0.92	0.94	0.93	0.94	0.95	0.95	0.94	0.95	0.96	0.96	0.95	0.95
	Shannon Index	1.64	1.58	1.48	1.54	1.59	1.52	1.51	1.48	1.45	1.43	1.33	1.27	1.31	1.37	1.38
	Simpson Index	0.97	0.97	0.95	0.96	0.97	0.96	0.96	0.96	0.96	0.95	0.95	0.94	0.95	0.95	0.95
Functional measures	%Collectors	26.02	23.21	19.72	33.51	31.28	25.00	30.08	28.95	33.64	39.39	45.00	44.30	41.67	38.61	34.65
	%Predators	19.78	19.64	20.18	10.99	11.17	20.14	14.29	20.18	15.89	9.09	7.50	5.06	10.71	20.79	17.32
	%Shredders	1.90	1.43	0.46	0.00	2.79	3.47	2.26	0.00	3.74	2.02	2.50	1.27	0.00	0.99	0.79
	%Filterers	33.33	36.79	42.66	40.31	35.75	39.58	42.11	44.74	39.25	37.37	41.25	49.37	38.10	30.69	41.73
	%Scrapers	18.97	18.93	16.97	15.18	18.99	11.81	11.28	6.14	7.48	12.12	3.75	0.00	9.52	8.91	5.51
Relative abundance of orders	Ephemeroptera	19.78	21.07	23.85	24.61	21.23	20.83	14.29	17.54	15.89	13.13	3.75	10.13	7.14	6.93	10.24
	Odonata	5.15	5.71	3.67	5.24	1.68	6.94	0.75	4.39	3.74	0.00	0.00	0.00	2.38	0.99	1.57
	Diptera	8.94	11.79	10.09	13.61	15.64	23.61	30.08	32.46	38.32	49.49	53.75	50.63	41.67	40.59	38.58
	Hemiptera	5.15	3.57	5.96	2.62	4.47	6.25	7.52	4.39	4.67	2.02	1.25	0.00	0.00	12.87	12.60
	Lepidoptera	0.00	0.00	0.00	0.00	2.79	2.78	2.26	0.00	3.74	2.02	0.00	0.00	0.00	0.00	0.00
	Veneroida	1.08	1.07	0.00	3.14	2.79	1.39	3.01	2.63	0.93	2.02	12.50	12.66	8.33	9.90	2.36
	Coleoptera	10.57	8.93	6.42	12.04	8.38	4.86	6.02	7.02	1.87	3.03	5.00	1.27	2.38	1.98	1.57
	Trichoptera	40.38	38.57	40.83	36.65	35.75	30.56	30.08	28.95	24.30	19.19	17.50	21.52	20.24	15.84	21.26
	Annelida	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.88	0.93	1.01	0.00	0.00	2.38	1.98	0.79
	Plecoptera	5.96	5.71	6.42	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Basommatophora	2.98	3.57	1.83	1.05	5.59	1.39	4.51	0.00	1.87	6.06	3.75	0.00	9.52	8.91	5.51
	Unionoida	0.00	0.00	0.92	1.05	1.68	1.39	1.50	1.75	3.74	2.02	2.50	3.80	5.95	0.00	5.51

The result of the Kruskal-Wallis test showed that there was no significant difference in %Predators, %Shredders, and %Filterers between the river reaches ($p > 0.05$). However, there were significant differences in other metrics such as %Collectors, %Scrapers, the Shannon, Simpson, and Evenness indices, total and EPT taxa richness, and %EPT abundances ($p < 0.05$) (Table 18-4).

Table 18-4: The mean ± standard error (SE), mean ranks, and Kruskal-Wallis H tests (K-W-H) for macroinvertebrate metrics in the upstream (US), midstream (MS), and downstream (DS) river reaches of KHD

Metrics	Mean±SE			Mean Rank			K-W-H test	
	US	MS	DS	US	MS	DS	K-W	**P-value**
	Composition measures							
% EPT	64.8±2.34	45.7±3.3	29.30±1.9	13.00	8.00	3.00	12.50	**0.002***
EPT taxa richness	21.6±1.21	13.4±0.7	7.4±0.2	13.00	8.00	3.00	12.66	**0.002***
Total taxa richness	50.8±2.85	37.2±1.9	24.8±1.3	13.00	8.00	3.00	12.52	**0.002***
	Diversity measures							
Evenness Index	0.92±0.01	0.9±0.004	0.96±0.002	3.40	7.60	13.00	11.58	**0.003***
Simpson Index	0.97±0.001	0.96±0.01	0.95±0.002	10.40	10.20	3.40	7.94	**0.019***
Shannon Index	1.7±0.03	1.5±0.02	1.3±0.02	12.60	8.40	3.00	11.58	**0.003***
	Functional measures							
% Collectors	26.3±2.5	29.9±2.5	39.5±1.8	4.60	6.80	12.60	8.54	**0.014***
% Predators	16.1±2.13	15.1±2.5	11.9±2.8	10.00	7.80	6.20	1.82	0.40
% Shredders	1.3±0.49	2.2±0.6	1.1±0.4	7.20	10.40	6.40	2.26	0.32
% Filterers	38.9±1.9	43±3.1	42.2±3.0	5.40	10.00	8.60	2.78	0.25
% Scrapers	17.49±0.8	9.3±1.2	5.4±1.7	13.00	7.20	3.80	10.82	**0.004***

**Metrics that have significant differences*

The *post hoc* analysis of all metrics also highlighted significant differences ($p < 0.05$) between the upstream-midstream, upstream-downstream, and midstream-downstream river reach pairs except for the

Simpson Index and %Collectors in the upstream-midstream and %Scrapers in the midstream-downstream river reach pairs (Table 19-4).

Table 19-4 The pairwise post hoc test calculated for macroinvertebrate metrics having significant differences in the Kruskal-Wallis test

Metrics	Upstream-midstream reach pairs				Upstream-downstream reach pairs				Midstream-downstream reach pairs			
	MR-US	MR-MS	M-W U	P-value	MR-US	MR-DS	M-W U	P-value	MR-MS	MR-DS	M-W U	P-value
%EPT	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	**0.009***
EPT taxa richness	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.008*	8.00	3.00	0.00	**0.008***
Total taxa richness	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	**0.009***
Evenness Index	3.40	7.60	2.00	0.028*	3.00	8.00	0.00	0.009*	3.00	8.00	0.00	**0.009***
Simpson Index	5.80	5.20	11.00	0.754	7.60	3.40	2.00	0.028*	8.00	3.00	0.00	**0.009***
Shannon Index	7.60	3.40	2.00	0.028*	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	**0.009***
%Collectors	4.60	6.40	8.00	0.347	3.00	8.00	0.00	0.009*	3.40	7.60	2.00	**0.028***
% Scrapers	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.009*	7.20	3.80	4.00	0.076

*MR: Mean rank, M-W U: Mann-Whitney U test; *Significant differences between river reach pairs with 2-sided test and a significance level of $P<0.05$*

Cluster analysis based on macroinvertebrate assemblage data was effective in grouping sites according to their level of anthropogenic impacts. At 65% Bray–Curtis Similarity Index (Figure 18-4), three groups of sites were clustered: (i) upstream river reach sites relatively less impacted by human activities; (ii) midstream river reach sites upstream of the reservoir, which is impacted by urbanisation and agricultural activities; and (iii) the downstream river reach sites that received all impacts from the upstream catchment. The ANOSIM test demonstrated significant differences in macroinvertebrate assemblages between the three river reaches (Global R = 0.968, p = 0.001). SIMPER analysis also showed significant differences between all pairs of river reaches, with the highest separation between the upstream-downstream river reach pairs (65.93%) with *Amphipsyche senegalensis* (7.11%) and *Cheumatopsyche sexfasciata* (5.43%) *(*Hydropsychidae), *Heptagenia* (4.85%) and *Leucrocuta* (4.48%) (Heptageniidae), and *Neoperla* (3.95%) (Perlidae) as the taxa with the highest contributions to the dissimilarity.

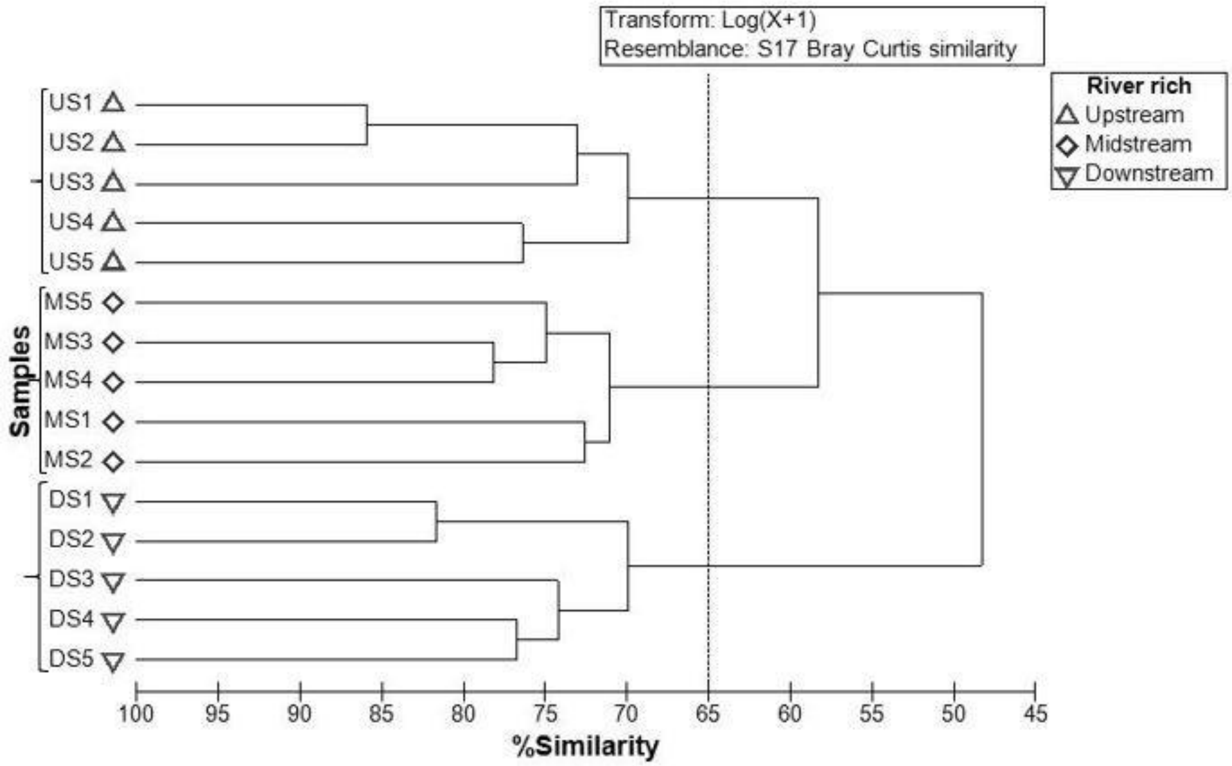

Figure 18-4: A dendrogram (similarity cluster analysis) based on results of group average macroinvertebrate log(x+1) transformed data

The mean dissimilarity between upstream and midstream groups was 51.33%. Here the taxa contributing to the dissimilarity were *Amphipsyche senegalensis* (6.44%) and *Cheumatopsyche sexfasciata* (6.23%) *(*Hydropsychidae), *Neoperla* (4.94%) (Perlidae), *Leucrocuta* (4.38%) and *Heptagenia* (3.73%) *(*Heptageniidae). Similarly, the mean dissimilarity between midstream and downstream groups was 48.35% with *Hydropsyche abyssinica* (4.92%) and *Amphipsyche senegalensis* (4.69%) *(*Hydropsychidae), and the genera *Orthocladius* (4.06%) *(*Chironomidae), *Heptagenia* (4.05%) (Heptageniidae), and *Hemerodromia* (3.87) (Empididae) as the taxa with the highest contributions to the dissimilarity. The NMDS ordination plot also demonstrated a clear separation of the three river reaches, each of which is represented by five sampling locations (Figure 19-4).

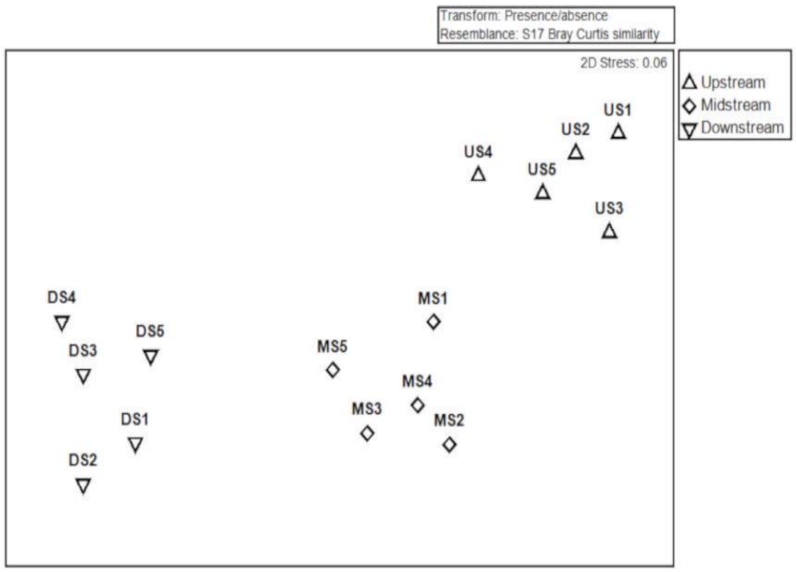

Figure 19-4: The NMDS ordination plots of the macroinvertebrate assemblages at the three river reaches. The ordination plot was produced using PRIMER 6 software. Triangles, circles, and diamonds serve as symbols for the US, MS, and DS reaches, respectively

Figure 20-4 depicts the ordination triplot diagram for the environmental variables, macroinvertebrate assemblages-, and site locations within the three river reaches. Each CCA axis has an eigenvalue (λ) that represents the maximum dispersion of taxonomic scores along the axes.

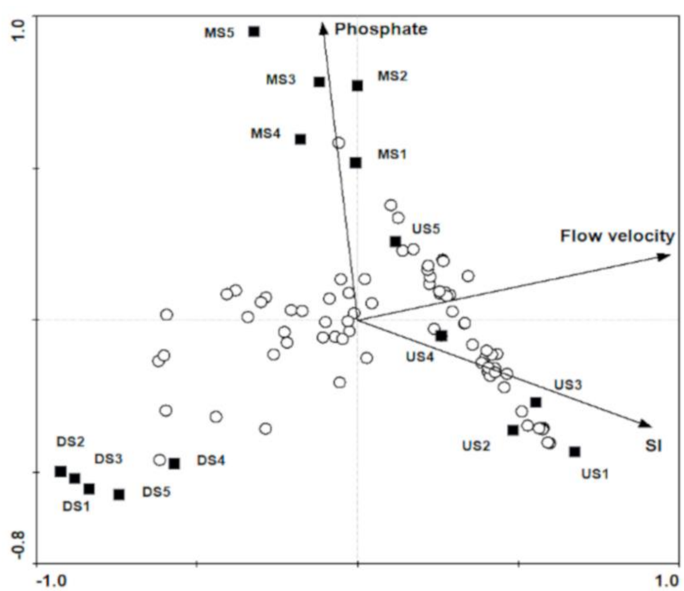

Figure 20-4: The CCA ordination triplot illustrates the relationships between significant environmental variables at $p < 0.05$ (arrows), macroinvertebrate taxa (circles), and sampling locations (squares) in the upstream, midstream, and downstream river reaches of the *KHD*

The cumulative percentage variance of the taxa-environment relationship explained by Axis-1 and Axis-2 was λ (1+ 2) = 0.38, with an eigenvalue of > 0.30 indicating strong gradients. The CCA model showed that the variation in macroinvertebrate assemblages was significantly correlated with river velocity ($F = 7.19$, $p = 0.002$), phosphate concentration ($F = 3.05$, $p = 0.002$), and substrate index (SI) ($F = 1.8$, $p = 0.016$) (Table 20-4). These three environmental variables explained 63.77% (λ = 0.44, CCA) of the total variation in macroinvertebrate assemblages. The contributions of the other variables ranged from 1.45% to 5.8% but were not significant (Table 20-4).

Table 20-4: The CCA results representing the independent and cumulative contribution of environmental factors in explaining the variation in the assemblages of macroinvertebrates in the three reaches of the Awash River

Environmental factors	Eigenvalue (λ) for each factor)	% Explained by each factor	Cumulative Eigenvalue (λ)	Cumulative percentage	F-ratio	p-value
Velocity ($m\ s^{-1}$)	0.28	40.58%	0.28	40.58	7.19	**0.002***
Phosphate ($mg\ L^{-1}$)	0.10	14.49%	0.38	55.07	3.05	**0.002***
SI	0.06	8.70%	0.44	63.77	1.8	**0.016***
DO ($mg\ O_2\ L^{-1}$)	0.04	5.80%	0.48	69.57	1.47	0.106
River depth (m)	0.04	5.80%	0.52	75.37	1.39	0.136
NO_3-N ($mg\ L^{-1}$)	0.04	5.80%	0.56	81.17	1.35	0.196
EC ($\mu S\ cm^{-1}$)	0.03	4.35%	0.59	85.52	1.19	0.342
Temperature (^{O}C)	0.03	4.35%	0.62	89.87	0.89	0.515
BOD_5 ($mg\ L^{-1}$)	0.03	4.35%	0.65	94.22	0.89	0.514
pH	0.03	4.35%	0.68	98.57	1.17	0.318
Turbidity (NTU)	0.01	1.45%	0.69	100.00	0.45	0.778

NTU: NTU: Nephelometric Turbidity Unit

4.5. Discussion

The objective of our study was to assess the impacts of KHD on macroinvertebrate assemblage patterns based on their spatial locations. The ANOSIM test highlighted significant differences (Global R = 0.968, p = 0.001) in macroinvertebrate assemblages between river reaches. This finding was consistent with previous studies conducted on dam impacts on the distribution of macroinvertebrates (Siziba, 2017; Ko et al., 2020). We observed the lowest diversity of macroinvertebrate taxa in the downstream sites relative to those in the upstream and even midstream river reaches. More pollution-sensitive Trichoptera and Ephemeroptera taxa dominated upstream sites, whereas pollution-tolerant taxa in the Diptera order such as *Chironomus* (Chironomidae), *Aedes* (Culicidae), *Musca* (Muscidae), *Simulium and Prosimulium* (Simuliidae), *and Chrysogaster* (Syrphidae) were found in the downstream river reach. The results of earlier studies demonstrated similar changes in macroinvertebrate assemblages downstream of a dam in terms of changes in either the abundance or diversity of taxa and species richness (Sharma *et al.*, 2005; Ko et al., 2020; Wang *et al.*, 2020).

All macroinvertebrate composition, diversity, and some functional measures (% collectors and %Scrapers) also showed significant differences between the three river reaches, consistent with an earlier study (Hauer and Resh, 2017). The results of the multivariate analysis also revealed considerable differences in the macroinvertebrate assemblages between the river reaches. The SIMPER analysis, for instance, indicated significant differences between all pairs of river reaches, with the upstream-downstream river reach pairs showing the greatest divergence and emphasising changes in macroinvertebrates. Several studies have highlighted the impacts of dams (Ko et al., 2020; Bussi et al., 2021; Mersha *et al.*, 2021). For example, Bussi et al. (2021) and Mersha et al. (2021) showed that variations in macroinvertebrate assemblages may be brought on by alterations in the physical characteristics of the river reaches as well as alterations in land use and water quality. The variations in these metrics might be related to several factors, such as the retention of sediments and nutrients in the reservoir, as demonstrated in the previous studies (Schmutz and Moog, 2018; Winton et al., 2019; Cattaneo *et al.*, 2021).

Earlier studies showed that the Awash River basin is subjected to high climate variability and experiences intensive anthropogenic activities (Mersha et al., 2021), such as water abstraction for various purposes and inputs of pollutants (Getachew et al., 2020b; Mersha et al., 2021). For example, Mersha et al. (2021) estimated the abstraction of 1200 million cubic meters of water from the total annual surface water resource potential of 4600 million cubic meters in the Awash basin (Adeba et al., 2015). Indeed, the cause of the variation might be associated with urbanization since the Awash basin is located in areas where several cities, including the capital Addis Ababa, are situated and many industries are concentrated (Englmaier et al., 2020; Getachew et al., 2020b; Mersha et al., 2021).

In our study, only three significant environmental factors—river flow velocity, phosphate concentration, and substrate index (SI)—significantly explained 63.8% of the variation in macroinvertebrate assemblages. Earlier studies also supported this finding (Winton et al., 2019; Ko et al., 2020). The river flow velocity accounted for the highest percentage (40.6%) of the variation in macroinvertebrate assemblages between the three river reaches ($F = 9.5$, $P = 0.002$). In this regard, the development of the dam might decrease the downstream river flow as a result of impoundment, diversions for irrigation, or drought that limits the amount of suitable habitat for aquatic organisms. Petts and Maddock (1994) and Extence *et al.* (1999) highlighted that alterations in community composition may occur as a direct consequence of varying flow patterns or indirectly through associated habitat change. Ko et al. (2020) also showed that dams negatively alter the natural flow regime in terms of magnitude, frequency, and duration in the downstream river reach. Several other studies demonstrated that flow velocity may well contribute the most to macroinvertebrate assemblage change, as velocity affected nutrient transfer, the mobility of drifting species, and the morphology of benthic substrates (Matthaei *et al.*, 1997; Nelson and Lieberman, 2002; Brooks *et al.*, 2005; Wolmarans *et al.*, 2017). The sedentary behavior of the species may further limit their ability to disperse downstream of the dam (Cairns and Prall, 1993).

The average river velocities in the downstream, midstream, and upstream portions were 0.18, 0.33, and 0.45 m s^{-1}, respectively, highlighting that the downstream river velocity is unsuitable for most macroinvertebrate taxa. According to Xu *et al.* (2014), unsuitable flow velocities are either < 0.3 or >0.8 m s^{-1}, whereas suitable flow rates fall between 0.3 and 0.8 m s^{-1}. On the other hand, Extence et al. (1999) allocated commonly identified freshwater species into one of the six groups based on recognized river flow associations as rapid (>1.0 m s^{-1}), moderate to fast (0.2-1.0 m s^{-1}), slow or sluggish flows (<0.2 m s^{-1}), and the rest three groups that are related to standing waters and drought-impacted sites. Jowett and Richardson (1990) also showed that the velocity preferences of macroinvertebrates may change with size or life stage. However, some taxa exist exclusively in very turbulent, high-velocity waters where they use sucker discs, hooks, or silk to remain attached to the substratum (Hauer and Lamberti, 2011; Hauer and Resh, 2017). For example, many Tipulidae, Trichoptera, and Plecoptera species that feed on coarse aquatic macrophytes (Shredders) (Merritt et al., 2002) occur in riffles (Hauer and Lamberti, 2011). Li *et al.* (2009) determined that the optimum river velocity for various macroinvertebrate taxa, such as *Baetis*, is 0.4 m s^{-1}. Our study did not include flow sensitivity as LIFE index and we would like to recommend future studies include it in similar research on dam impacts.

The ranges of upstream and midstream river reach velocities were 0.4-0.46 m s^{-1} and 0.29-0.38 m s^{-1}, which are closer to the optimum value and have a better composition of macroinvertebrates, while the downstream river reach velocities range from 0.15 to 0.22 m s^{-1}. However, the more diverse macroinvertebrates, particularly in the upstream reach, maybe not only attributable to river velocity but also because of the presence of other conducive environmental characteristics such as lower temperatures, higher concentrations of DO, and more varied and coarse substrate compositions.

The other environmental parameter that explained 14.5% of the variation in macroinvertebrate distribution between the three river reach sections was differences in phosphate concentration. Earlier studies have reached similar conclusions (Ko et al., 2020; Wang et al., 2020). Particularly, large dams can significantly trap nutrients (Poff and Hart, 2002; Cattaneo et al., 2021). For instance, Wang et al. (2020) reported that one sign of the effect of large dams on macroinvertebrate assemblages was a greater decline in nutrient levels at downstream sites, which has also been demonstrated in our study. The concentration of phosphate increased from the upstream to the midstream direction, probably because of differences in land uses such as agriculture and urbanisation, as also highlighted in various earlier studies (Getachew et al., 2020b; Mersha et al., 2021). However, the concentration of phosphate decreased in the downstream reach, which might partly be related to dam impacts, as supported by earlier studies. For example, Kunz *et al.* (2011) reported that more than 90% of the phosphorus was trapped in Kariba Dam on the Zambezi River.

The canonical correspondence analysis also showed that the changes in SI alone significantly explained 8.7% of the variation in the distribution of macroinvertebrates. According to Wang et al. (2020), damming disturbs the availability of suitable habitats for macroinvertebrates, and sediment trapping within reservoirs may have a significant impact. Wang et al. (2020) highlighted that the causes of macroinvertebrate richness reductions downstream of the dam were mainly attributed to changes in downstream substrate composition, i.e., from coarse substrates (high SI) to fine substrates (low SI). Fine substrates can cause an accumulation on the fine and fleshy body parts of macroinvertebrates (such as gills and filter-feeding apparatus), making it difficult for aquatic residents to breathe and feed (Jones *et al.*, 2012).

Substrate preferences of macroinvertebrates vary from taxa to taxon. For example, stoneflies, cased caddisflies, and Diptera showed a preference for a substrate index of more than 6 (boulder/cobble) and the substrate index preference of beetles is about 5.6 (gravel/cobble) (Jowett and Richardson, 1990). The optimum substrate index for the majority of macroinvertebrates was just under cobble size (SI = 6) (Jowett and Richardson, 1990). The calculated mean SI for the upstream, midstream, and downstream sites were 6.63, 5.6, and 3.83, respectively, indicating that the downstream sites had substrates between sand (SI = 3) and gravel (SI = 4) which is far below the optimum SI. The absence of pollution-sensitive taxa such as *Heptagenia* and *Leucrocuta* (Heptageniidae), *Neoperla* (Perlidae*), Ecnomus similis* (Ecnomidae), and the dominance of pollution-tolerant taxa such as *Chironomus* (Chironomidae) in the downstream river reach might partly be associated with a reduction of SI.

The overall view is that the impacts of dams could be seen as eco-deficits (a condition related to insufficient water to meet the needs of the ecosystem) or eco-surplus (a condition related to water exceeding what is needed by the ecosystem) (Serfas, 2012). Several authors showed that dams alter the quantity, quality, and availability of water in relation to the natural environment (Gracey and Verones, 2016; Winton et al., 2019). Our research has identified important environmental factors such as flow regulation, which might be related to changes in river flow velocity, changes in SI, and variations in nutrient loads to the downstream river reach that have caused changes in macroinvertebrate assemblage patterns between the reaches upstream and downstream of the KHD. However, the lower reaches of this river also carry the impact of pollution from further upstream, thus confounding efforts to attribute impacts directly to the presence of the dam.

4.6. Conclusion

Although dams often deliver economic services such as hydropower, flood risk alleviation, water supply, recreation, and more, there is evidence that dams have impaired the key functions of rivers in providing diverse habitats and maintaining ecosystem integrity. A combination of environmental and hydromorphological variables in this study explained why macroinvertebrate assemblages differed across the three river reaches above and below the KHD. Our study demonstrated that there is a significant difference between the three river reaches in terms of macroinvertebrate compositions, diversity, and some functional metrics. Changes in river flow regime due to changes in water velocity, variations in nutrient loads (phosphate concentration), and SI were highlighted as key variables determining the assemblage patterns of macroinvertebrates. Although other possible mechanisms for the observed differences may exist, the findings of this study are useful for the assessment of hydropower dam impacts on rivers in this region.

Chapter Five: Factors Affecting Macroinvertebrate Community Composition in Three River Basins of Ethiopia: A Variation Partitioning Analysis

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Chapter 5: Factors Affecting Macroinvertebrate Community Composition in Three River Basins of Ethiopia: A Variation Partitioning Analysis

5.1. Abstract

Community ecology is the study of interactions between species in communities at different geographical and temporal scales, including composition, structure, abundance, demography, and interactions between coexisting populations. Several studies examined the effects of environmental and land use individually and together on communities of benthic macroinvertebrate composition. However, there is a lack of evidence that evaluates the unique and shared effects of environment, land use, and space on the composition of macroinvertebrates in streams. Examining the effects of environmental, land use, and spatial factors on the composition of macroinvertebrate communities was the objective of the current study. We used variation partitioning analysis to evaluate the unique and shared roles of the three sets of predictor variables. In this study, we found that 51% of the variation in the compositions of macroinvertebrates could be attributed to environmental, land use, and spatial factors. The findings of this study demonstrated the shared variation in macroinvertebrate composition due to the three sets of predictors were higher than their unique effects. The result highlights the three sets of predictors were more correlated and that environmental factors are spatially structured and influenced, directly or indirectly, by land use factors, Thus, land use planning could be an important strategy to improve stream water quality, thereby increasing macroinvertebrate composition and enhancing the ecosystem services obtained from freshwater resources.

Keywords: river basin, stream reach, variation partitioning, environment, land use, space

5.2. Introduction

The composition of biological communities in streams varies across time and space in response to biotic and abiotic factors (Verberk *et al.*, 2016). By investigating the variables that affect the patterns of ecological communities, environmental experts can enhance management decisions, forecast the scope and direction of ecological change, and understand the overall effects of anthropogenic influences on the environment (Mikulyuk *et al.*, 2011). However, many predictors interact with one another often enough to have cascading effects on ecological communities (Wagner and Fortin, 2005). Investigating and interpreting the interactions between environmental and land use factors and spatial location on the composition of biological communities is one of the ecologists' main objectives (Hepp *et al.*, 2012).

In Ethiopia, rapid urbanization and agricultural development with primitive practices, cause water resource pollution that, directly or indirectly, affects aquatic life at different spatial locations (Matomela *et al.*, 2021). However, the unique and shared effects of these predictors are not examined. Ecological patterns and processes are spatially variable, and the scale that researchers use to depict spatial variation can greatly affect the results of analysis (Wagner and Fortin, 2005). Furthermore, the various benthic macroinvertebrate taxa can show distinct environmental associations and vary in their response to environmental conditions on multiple scales (Tongayi et al., 2020). Fortunately, multiscale methods exist that allow comprehensive investigations of the influence of spatial variation on biological community composition (Burrough, 1983).

A greater understanding of the predictors affecting variations in macroinvertebrate taxa composition is important to assess species compositions and their responses to a changing environment (Constable, 1999). Some researchers, for example, Rezende *et al.* (2014), Gitau *et al.* (2016), and Garcia-Giron *et al.* (2022) have examined the relative effects of environmental, land use, and spatial factors on the composition of macroinvertebrates; however, the evidence that evaluates the unique and shared roles of these factors in Ethiopia from ecological context is limited. The goal of the study was to evaluate the relative roles of environmental, land use, and spatial variables on the composition of benthic macroinvertebrates in the Abay, Omo-Gibe, and Awash River basins, Ethiopia. Specifically, the next questions were addressed: 1) what are the relative roles of environmental, land use, and spatial variables on the composition of benthic macroinvertebrates? 2) Which variable(s) are the most important in determining macroinvertebrate assemblage compositions in selected rivers?

5.3. Materials and methods

5.3.1. Study Area

This study was conducted in the Abay (Blue Nile), Omo-Gibe, and Awash River basins in Ethiopia. Based on the Strahler stream ordering (Pradhan and Ghose, 2012), the streams selected for this study were second-order to fourth-order and located in different agroecological zones based on several criteria set for Ethiopia (Alemneh et al., 2017; Lakew, 2017). The details are shown in Table 21-5.

Table 21-5: Stream ID, stream and basin names, site locations, and land use type of study sites in Ethiopia

ID	Stream name	Basin name	Longitude	Latitude	Altitude	Major land use type(s)
S1U	Urgessa1	Abay	7º46'12"	36º39'36"	2083	Forest
S2U	Urgessa2	Abay	7º50'48"	36º40'12"	1777	Forest
S3U	Urgessa3	Abay	7º 53'48"	36º41'24"	1501	Forest, grazing
S4Y	Yebu1	Abay	7º 47'30"	36º45'36"	1947	Forest
S5Y	Yebu2	Abay	7º 49'36"	36º46'34"	1856	Forest, grazing
S6Y	Yebu3	Abay	7º 51'00"	36º46'30"	1711	Forest, grazing
S7F	Feche1	Omo-Gibe	7º 44'34"	36º42'36"	2139	Forest, grazing
S8F	Feche2	Omo-Gibe	7º42'18"	36º45'47"	1899	Grazing, agriculture
S9F	Feche3	Omo-Gibe	7º40'23"	36º47'35"	1731	Grazing, agriculture
S10GM	Gumi1	Omo-Gibe	7º13' 47"	36º22'48"	2268	Forest
S11GM	Gumi2	Omo-Gibe	7º14'17"	36º20'10"	1894	Forest
S12GM	Gumi3	Omo-Gibe	7º13'58"	36º18'14"	1695	Forest, agriculture
S13GN	Gunji1	Omo-Gibe	7º24'49"	36º11'49"	1715	Forest
S14GN	Gunji2	Omo-Gibe	7º21'40"	36º12'04"	1620	Forest, agriculture
S15GN	Gunji3	Omo-Gibe	7º17'48"	36º13'12"	1617	Forest, agriculture
S16GL	Gulufa1	Omo-Gibe	7º34'32"	36º37'52"	1922	Grazing, agriculture
S17GL	Gulufa2	Omo-Gibe	7º35'37"	36º39'32"	1912	Grazing, agriculture
S18GL	Gulufa3	Omo-Gibe	7º34'10"	36º40'37"	1836	Grazing, agriculture
S19D	Demu1	Omo-Gibe	7º32'20"	36º53'13"	1902	Grazing, agriculture
S20D	Demu2	Omo-Gibe	7º33'56"	36º55'23"	1737	Grazing, agriculture
S21D	Demu3	Omo-Gibe	7º35'44"	36º56'31"	1707	Grazing, agriculture
S22S	Sebeta1	Awash	8º55'12"	38º39'22"	2262	Urbanisation
S23S	Sebeta2	Awash	8º53'55"	38º37'08"	2080	Urbanisation
S24S	Sebeta3	Awash	8º51'11"	38º34'59"	2061	Urbanisation, agriculture
S25J	Jemijem1	Awash	9º04'43"	38º14'53"	2406	Forest, agriculture
S26J	Jemijem2	Awash	9º12'01"	38º14'13"	2173	Grazing, agriculture
S27J	Jemijem3	Awash	8º58'58"	38º13'59"	2123	Urbanisation, agriculture
S28A	Awash1	Awash	9º18'59"	38º06'54"	2257	Forest, agriculture
S29A	Awash2	Awash	9º00'00"	38º09'00"	2167	Grazing, agriculture
S30A	Awash3	Awash	8º55'54"	38º18'14"	2076	Urbanisation, agriculture

We selected a sampling reach on each stream length following Frissell *et al.* (1986) and Flotemersch *et al.* (2006). In the context of hierarchy, Frissell et al. (1986) defined "reach" as the distance between breaks of a stream in the channel slope, the width of the valley floor, bank material, and riparian vegetation. Thus, "stream reach" is the smallest physically discrete unit in the hierarchy, but a very useful scale for describing medium and long-term effects of human activities on streams (Frissell et al., 1986; Flotemersch et al., 2006). Accordingly, three sites on each stream were selected (Table 21-5, Figure 21-5) based on a gradient of visible human disturbance and ecological factors (Herlihy *et al.*, 2000).

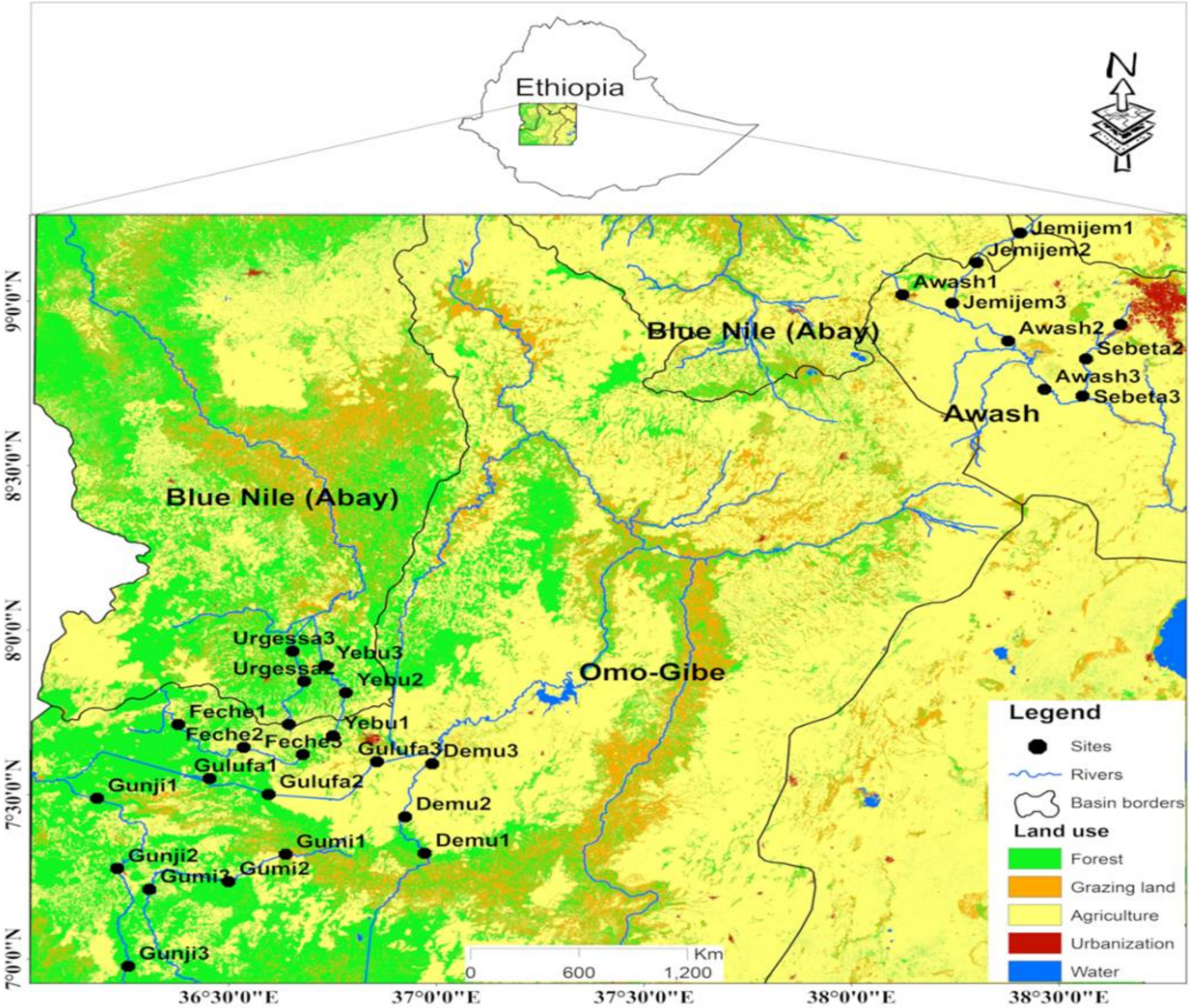

Figure 21-5: The land use map and location of the sampling points in the Omo-Gibe, Abay, and Awash River basins in Ethiopia. The land use composition was obtained from the 2018 sentinel2 raster database at EthioGIS-3, the Water and Land Resources Information System (WALRIS) at https://www.wlrc-eth.org/, or directly from Map Server Ethiopia at https://www.ethiogis-mapserver.org/

5.3.2. Environmental conditions

Physicochemical parameters such as dissolved oxygen (mg L^{-1}), pH, water temperature (°C), total dissolved solids (TDS), and electrical conductivity (μS cm^{-1} @ 25 °C) were measured in situ in the field using a multi-probe meter (HQ30d Single-Input Multi-Parameter Digital Meter, Hach Company, USA). The five-day biochemical oxygen demand (BOD_5), total organic nitrogen (TON), and total phosphorous (TP) contents were measured using 500 mL of unfiltered water. Using a cold box, water samples were brought to the laboratory analysis. TP samples were digested in a block digester using ammonium persulfate and sulfuric acid reagent (APHA, 2005) and analysed using the stannous chloride method (APHA, 2005). BOD_5 was measured using a standard method (APHA, 2005). TON samples were digested and measured with photometric kits (Hach LANGE) using a Hach DR5000 spectrophotometer. To prevent disturbing the next sampling sites, samples were collected starting from the downstream site to its upstream direction. Samples of water were collected first and then samples of macroinvertebrates were taken at each location.

5.3.3. Land use and land cover change data

The percent coverage of each land use type at each sampling site was calculated within a radius of 1000 meters following Fu et al. (2016) using ArcGIS version 10.5 with spatial visualisation and statistics.

5.3.4. Spatial Variables

To model the spatial relations of community composition among sampling sites at multiple scales, the principal coordinates of neighbor matrices (PCNM) approach was used to provide spatial factors based on overland distances among sampling sites for further analysis (Li *et al.*, 2019). This approach has been widely used for modeling spatial structures in biological communities (Legendre and Legendre, 2012). Spatial variables were produced using PCNM from the geographic coordinates of observation sites after Borcard and Legendre (2002). The latitude and longitude coordinates were used to create Euclidean distance matrices, which were then trimmed to the shortest distance that maintained connectivity between all sites in a single network. To extract eigenvectors linked to positive eigenvalues for use as explanatory variables in the next statistical studies, the shortened Euclidean distance matrices were employed in a principal coordinate analysis (PCoA) (Dray *et al.*, 2006). We retained only those associated with significant and positive eigenvalues since they represent a potential positive autocorrelation between spatial points at different scales (Dray et al., 2006).

5.3.5. Macroinvertebrate data collection and identification

From mid-February to mid-May 2022, we conducted surveys at 30 locations along 10 streams. A rectangular frame pond net (20 x 30 cm) with a mesh size of 0.5mm was used to collect macroinvertebrates at each sampling location (Barbour, 1999). From each sampling location that covers a distance of 10 meters, 2-minute samples (Getachew et al., 2022b) from each riffle, run, and pool microhabitat (Kuriqi *et al.*, 2020) were collected. The samples were then kept in plastic bags with ethanol (80%). We used consecutive sieves having different pore sizes and sorted macroinvertebrates according to their size in the laboratory. Macroinvertebrates were grouped into broad taxonomic categories and kept in 80% ethanol for eventual identification (Barbour, 1999). Identification was done to the genus level using the available macroinvertebrate keys developed in South Africa (Harrison, 2009; Lowe, 2009; Schael, 2010).

5.3.6. Data Analysis

The biotic metrics in community ecology have benefits and drawbacks. Diversity indices show the combined impact of environmental variables on species richness, evenness, and abundance (Sumudumali and Jayawardana, 2021). However, the uses of these indices have been strongly criticised as they provide no information about the real species composition of a community (Andem *et al.*, 2013). Consequently, we included selected metrics that address diversity, composition, and functional measures. Using the PRIMER program, we compute aggregate metrics for the log-transformed total taxonomic richness, evenness index, %Ephemeroptera-Plecoptera-Trichoptera (%EPT), EPT taxa richness, and Shannon and Simpson indices. The values of the aggregated macroinvertebrate metrics were then compared using a one-way ANOVA. Similarly, using one-way ANOVA in PAST software, differences in land use characteristics and in-stream water parameters across the three river basins were examined (Hammer *et al.*, 2001).

We employed variation partitioning analyses to identify the unique and shared contributions of environmental, land-use, and spatial factors to the variation in macroinvertebrate taxon composition (van den Wollenberg, 1977). The method of variation partitioning is used when two or more complementary sets of hypotheses can be used to explain the variance of ecological response variables. Using this technique, the total amount of variation that a statistical model can account for can be divided into the unique and shared contributions of various sets of explanatory variables (Peres-Neto *et al.*, 2006; Borcard *et al.*, 2011). The redundancy analysis (RDA) was used to examine important combinations of predictor variables to evaluate the overall influence of environmental, land use, and spatial factors on the composition of benthic macroinvertebrates (van den Wollenberg, 1977). Then, using forward selection based on the adjusted R^2 double-stopping criteria (the standard alpha significance level and the adjusted coefficient of multiple determination, R^2) computed using all explanatory variables, significant variables within each set of predictors were identified (Blanchet *et al.*, 2008).

A variation partitioning analysis based on RDA was then used to determine the contributions of significant sets of explanatory factors to the changes in the composition of macroinvertebrate communities. To measure the variation explained, we computed the adjusted coefficient of determination (R^2 adjusted). R^2 is biased and increases with the number of explanatory factors, even when they are random, whereas R^2adjusted provides unbiased estimates of the variance of the response data defined by groups of explanatory variables (Peres-Neto et al., 2006; Borcard et al., 2011). After the normality test, all environmental and land use variables were then square-root transformed except for pH. In the analysis of biological data, the compositions of macroinvertebrate taxa were used. The overall significance of the ordination as well as that of each major fraction of variations was tested using the Monte-Carlo test with

499 permutations. All statistical analyses were conducted in R version 4.2.1 using the PCNM, RDA, and varpart functions of the vegan library (Oksanen *et al.*, 2012).

5.4. Results

5.4.1. Macroinvertebrate community assessment

A total of 8031 macroinvertebrate individuals grouped into 15 orders, 72 families, and 103 species/genera were identified from 30 sampling sites distributed over three major river basins of Ethiopia (Table 5.S1 and Table 22-5). The computed macroinvertebrate metrics such as richness, evenness, Shannon index, Simpson index, %EPT, and EPT richness revealed significant differences between the river basins (ANOVA, $P < 0.01$). The ANOVA test also indicated significant differences in the assemblages of individual taxa belonging to each order between the three river basins, except for Annelida, Diptera, and Basommatophora (Table 22-5).

The most abundant out of 15 orders was the Ephemeroptera in all three river basins. The highest composition of Ephemeroptera was observed in the Abay river basin consisting of 14 genera and 8 families. Trichoptera was the second most prevalent order in the Abay river basin, with 10 families and 17 genera (Table 5-S1, annexed).

Figure 22-5: The Mean±SE and the ANOVA test results for metrics and orders of macroinvertebrate communities in the three major river basins of Ethiopia

Metrics	Mean±SE			ANOVA		
	Abay	Omo-Gibe	Awash	SSW (df=27)	SSB (df=2)	F
Richness	72.4±3.0	51.9±3.4	51.9±5.57	4272.7	3034	9.60*
Evenness	0.98+0.002	0.98±0.001	0.99±0.001	0.0001	0.0001	5.50*
Shannon Index	1.67±0.02	1.6±0.03	1.6±0.03	0.25	0.15	8.30*
Simpson Index	0.97±0.001	0.97±0.001	0.96±0.001	0.00	0.01	98.16*
%EPT	59.50±1.1	40.9±3.83	42.3±2.96	1840.05	2320	17.02*
EPT richness	25.70±0.8	14±1.76	12.9±1.53	473.56	1081.1	30.82*
BMWP	191.5±5.6	141.6±10.1	139.2±11.7	21420.8	18833.4	11.90*
ETHbios	142.7±3.1	106.7±7.2	108.3±8.4	10020.9	8946.6	12.05*
Orders						
Araneae	2.33±0.6	0.33±0.24	0.33±0.24	54.67	28.8	7.11*
Basommatophora	2.42±0.71	3.33±0.78	3.44±0.77	153.14	6.86	0.60ns
Annelida	2.50±0.42	2.56±0.65	1.78±0.6	78.78	3.52	0.60ns
Coleoptera	39.25±5.02	9.67±2.71	10.22±3.1	4537.8	6184.9	18.4*
Diptera	37.75±6.22	36.11±4.04	34.8±7.35	10180.7	46.27	0.06ns
Ephemeroptera	140.5±18.1	37.89±7.35	42.11±10.2	54482.8	72802	18.04*
Hemiptera	38.6±6.74	14.11±2.18	16.22±3.13	7047.36	3968.1	7.60*
Lepidoptera	0.50±0.19	1.89±0.54	2.22±0.36	35.44	17.92	6.83*
Megaloptera	5.42±0.93	0.56±0.29	0.89±0.56	144.03	159.17	14.92*
Odonata	42.83±6.44	15.56±1.95	16.11±3.46	6610.78	5250.19	10.72*
Plecoptera	30.83±4.23	1.44±0.78	0.67±0.47	2421.89	6387.08	35.60*
Trichoptera	83.75±38.4	25.56±4.9	27±5.95	20496.5	23791.4	15.67*
Decapoda	4.75±0.76	0.44±0.18	1.11±0.59	103.36	115.61	15.10*
Unionoida	2.67±0.56	1±0.37	1.44±0.38	60.89	15.91	3.53*
Veneroida	0.5±0.23	1.67±0.47	2.44±0.50	41.22	20.14	6.60*

*SSW: Sum of Squares within Groups, SSB: Sum of Squares between Groups, SE Standard Error, DF: Degree of Freedom, * and ns: Significant at $P < 0.05$ and non-significant, respectively*

5.4.2. Environmental and land use parameters

Except for pH, DO, and water depth, all land use and environmental factors showed significant differences in the ANOVA test (Table 22-5).

Table 22-5: The Mean±SE and the ANOVA test results of environmental and land use variables in the three major river basins of Ethiopia

		Mean±SE among the three river basins			ANOVA test		
Variables	Unit	Abay	Omo-Gibe	Awash	SSW (df=27)	SSB (df=2)	F
pH	-	7.63±0.03	7.08±0.18	7.69±0.31	9.26	2.09	3.05^{ns}
Temperature	°C	18.5±0.72	21.7±0.55	18.2±0.78	133.8	72.32	7.30*
DO	mg L^{-1}	7.39±0.30	6.47±0.35	6.51±0.41	32.38	5.88	2.45^{ns}
Turbidity	NTU	5.29±0.51	9.13±0.90	16.2±3.67	1061.9	616.05	7.83*
BOD	mg L^{-1}	0.87±0.04	1.57±0.11	2.02±0.33	8.95	7.05	10.64*
TON	mg L^{-1}	0.03±0.00	0.53±0.11	0.77±0.21	3.96	3.08	10.51*
TDS	mg L^{-1}	78.7±1.03	138.8±5.7	168±17.9	25508	43963	23.27*
TP	mg L^{-1}	0.14±0.01	0.95±0.21	1.73±0.28	8.82	12.98	19.86*
EC	µS cm^{-1}	45.89±7.2	256±34.3	164±23.3	130272	232525	24.1*
Water depth	m	0.34±0.02	0.33±0.02	0.41±0.03	0.15	0.03	2.98^{ns}
Velocity	m s^{-1}	0.46±0.02	0.31±0.04	0.2±0.01	0.19	0.35	25.06*
Agriculture	%	6.48±0.3	12.9±1.8	16.21±2.1	547.2	519	12.81*
Forest	%	27.86±4.4	11.86±1.91	10.98±3.9	3971.3	2336.5	7.94*
Grazing	%	7.04±0.24	16.46±1.6	18.74±2.9	809.82	826.6	13.78*
Urbanisation	%	3.62±0.24	13.28±2.9	14.84±2.6	1105.3	795.05	9.71*

SSW: Sum of Squares within Groups, SSB: Sum of Squares between Groups, SE Standard Error, DF: Degree of Freedom, * and ^{ns}: Significant at $P < 0.05$ and non-significant, respectively

5.4.3. Roles of environment, land use, and space on the composition of macroinvertebrates

The RDA results showed that environmental, land use, and spatial variables significantly contributed (51%) to the compositional variation of macroinvertebrate communities (Table 23-5). The remaining unexplained variation (49%) could be attributed to unmeasured habitat-specific environmental variables.

Table 23-5: Results of RDA separately testing the unique and shared effects of the environment, land use, and space on the community composition of benthic macroinvertebrates

Variables	Df	R^2	R^2Adj.	Variance fraction	F	P	Selected variables*
Environment	12	0.53	0.42	0.178	4.36	0.001***	pH, EC, Temp, DO, TON, Depth
Land use	4	0.44	0.35	0.146	4.87	0.001***	Agriculture, Forest, Urbanization, Grazing
Space	5	0.39	0.32	0.129	4.79	0.001***	PCNM1, PCNM2, PCNM3, PCNM4
Individual fractions (X1 = Environment, X2 = Land use, X3 = Space), a, b, c fraction of R^2 adjusted)							
[a]=X1| X2+X3	6		0.08	0.08	2.20	0.001***	The unique role of the environment in controlling land use and space
[b]=X2| X1+X3	4		0.04	0.052	2.06	0.001***	The unique role of land use in controlling environment and space
[c]=X3|X1+X2	5		0.03	0.036	1.47	0.003**	The unique role of space in controlling environment and land use
Total variation	14	0.74	0.51	0.248	3.08	0.001***	Total variation of macroinvertebrate composition due to the three sets of factors

The forward selection method was used for selecting the explanatory variables following (Blanchet et al., 2008). PCNM: Principal Component Neighbour Matrices. The significance level is indicated by an asterisk ($p < 0.05$, ** $p < 0.01$, *** $p < 0.001$).*

RDA was used to describe the statistically significant ($p < 0.05$) influences of environmental, land use, and spatial sets of predictors on macroinvertebrate community composition (a triplot of all significant variables is shown in Figure 23-5). Different groups of major environmental factors that affect the composition of benthic macroinvertebrates were found by forward selection. The selection yielded six environmental factors that were statistically significant predictors of macroinvertebrate community composition, including pH, dissolved oxygen (DO), total organic nitrogen (TON), electrical conductivity (EC), temperature, and water depth. Similarly, the four significant land use variables selected using the forward selection were agriculture, urbanisation, grazing, and forest cover.

Environmental factors have a detrimental impact on the majority of macroinvertebrate taxa that are sensitive to pollution (Figure 23-5). In the Awash River basin, the majority of sites in the Jemijem, Awash, and Sebeta Streams have poor compositions of pollution-sensitive taxa. The vast majority of the taxa found in this basin were pollution-tolerant (see the left circle in Figure 23-5). In contrast, the majority of the locations in the Abay River basin (right circle in Figure 23-5) supported a diverse range of macroinvertebrate taxa that are sensitive to pollution. The forward selection additionally demonstrated other groups of significant spatial descriptors with a wider geographical dimension which are created from their coordinates using Euclidean distance matrices. According to Borcard and Legendre (2002), the first larger spatial eigenvalues indicate broad-scale spatial relations between locations, and the last spatial vectors show smaller-scale variation among sites. For constrained ordination models, we obtained four significant positive spatial descriptors (PCNM1, PCNM2, PCNM3, and PCNM4) (Figure 23-5) in the R environment using the vegan package (Oksanen *et al.*, 2013).

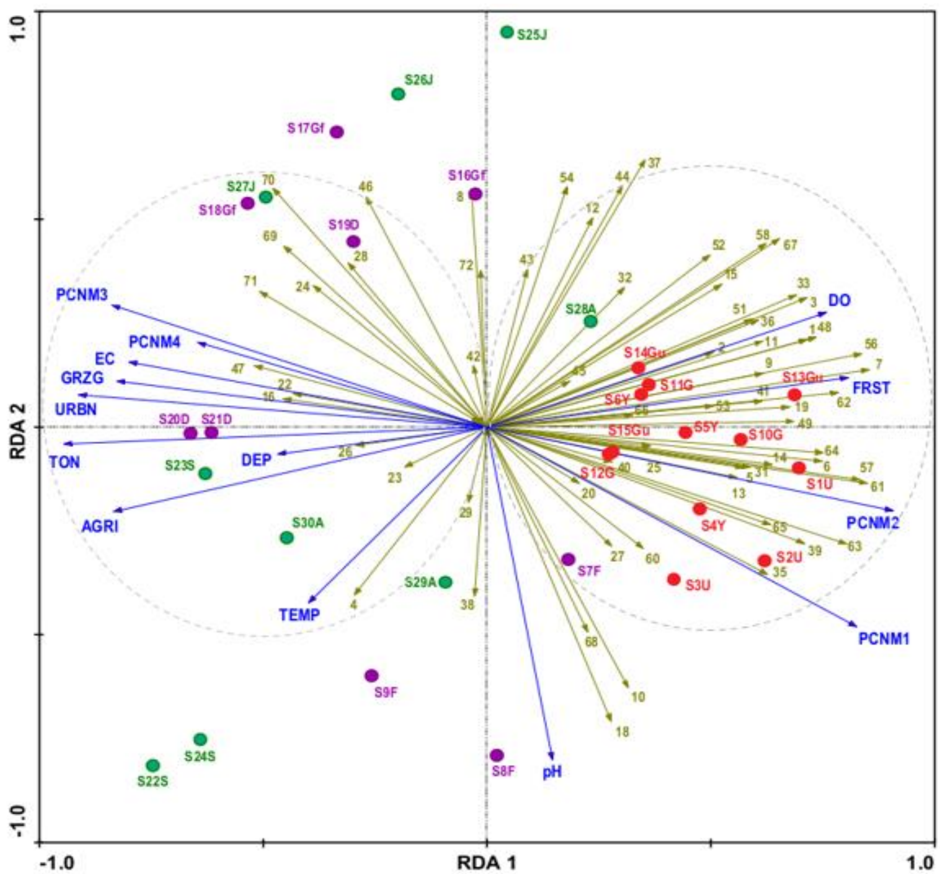

Figure 23-5: Ordination plots of the RDA of macroinvertebrates and environmental variables distributed over different sampling locations. The dark yellow arrows indicate the different genera of benthic macroinvertebrate communities, and the numbers in front of these arrows correspond with the names of the taxa indicated in Table S1 in the supplementary material. Significant variables from each set of predictor variables are represented by the blue arrows. Circes with green, red, and purple represent sampling sites in the Awash, Abay, and Omo-Gibe river basins, respectively. Broken circles encompass the majority of pollution-tolerant taxa (left) and the majority of pollution-intolerant taxa (right). AGRI: agriculture; URBN: urbanisation; GRZG: grazing; FRST: forest

The variation partitioning analysis used in this study revealed that the unique and shared effects of environmental, land use, and spatial factors can both be used to explain differences in the composition of the macroinvertebrate communities among the 30 sites. The RDA test showed that each testable fraction of variation explained a significant portion of compositional variation ($p < 0.05$). Overall, the combined model, including all variables and their conditional effects, was able to account for 51% (R^2 adjusted) of the total variation in the composition of macroinvertebrate communities. Based on the variation partitioning analysis, environmental variables appear to be notably more relevant (42%) than land use (35%), or space (32%) variables in explaining compositional variation in macroinvertebrate communities (Figure 23-5). Furthermore, compared to land use and space, environmental factors alone had a substantial unique contribution of 8%. Although the unique effects of space and land use were smaller than environmental factors, both sets of variables had significant effects on the variation of macroinvertebrate community compositions (Figure 23-5). The shared effect of the three sets of predictors was 22%, demonstrating that their shared effects on the composition of macroinvertebrate communities are more significant (Figure 23-5).

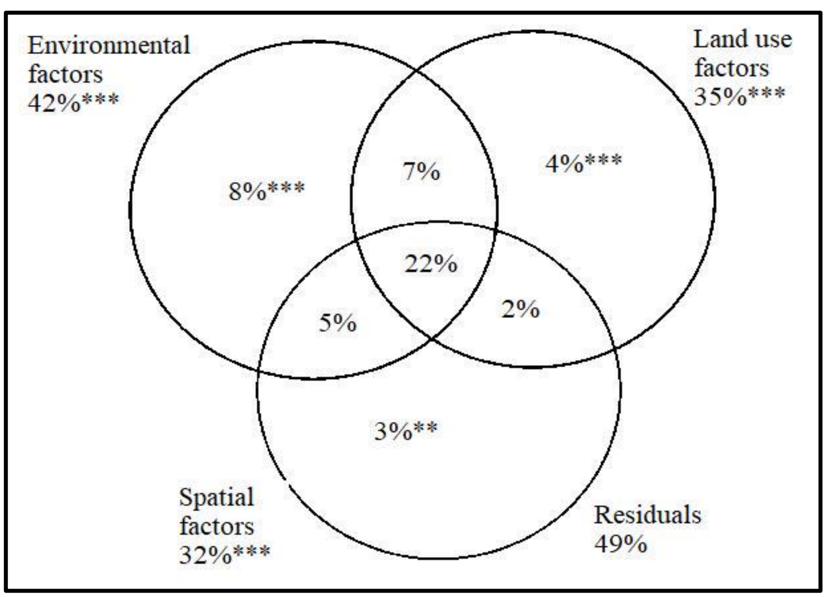

Figure 24-5: The results of a variation partitioning analysis based on the composition of the entire macroinvertebrate community against three correlated predictors (environment, land use, and space). The adjusted R2 is represented in the diagram by percentages.

5.5. Discussion

The overall results of this study demonstrated that the shared effects of environment, land use, and space were substantially responsible for the variation in macroinvertebrate community composition. The selected sets of factors explained only 51% (R^2 adjusted) of the variation in the composition of the macroinvertebrates. This reduced variation in the composition of taxa (51%) due to selected sets of factors is a common phenomenon in ecological studies due to the missing predictor variables (Hepp et al., 2012). Comparatively, the amount of shared variation between environmental and land use factors was greater than the shared variation between environmental and spatial factors, demonstrating the presence of more correlation between environmental and land use factors in explaining macroinvertebrate compositions. The variance in the response data that can be explained by the correlation of the predictors included is represented by the shared fractions. The amount of multicollinearity in the model increases with the shared proportion (Peres-Neto et al., 2006; Borcard et al., 2011).

In the diagnosis of their unique effects, environmental factors were mostly responsible for the variation in the macroinvertebrate composition. For example, TON, EC, temperature, pH, water depth, and DO were found to have made the most significant contributions to the compositional patterns of macroinvertebrates in the study catchments. This result is similar to a previous study investigating how key water quality characteristics determine the macroinvertebrate compositions in streams (Azis and Abas, 2021; Du *et al.*, 2021). For instance, Du et al. (2021) and Azis and Abas (2021) showed the majority of macroinvertebrate groups are highly reliant on environmental elements such as temperature, DO, NH_3-N, and others for surviving in egg-laying regions and providing ideal conditions for the larvae to develop.

The finding in this study also revealed that environmental factors have a more significant unique effect on the composition of macroinvertebrate communities. However, a significant proportion of the contribution from these components was also shared by spatial and land use factors. This could be as a result of differences in the spatial arrangements of environmental variables and the direct and indirect effects of land use factors on the stream water quality characteristics as also highlighted in other studies (Jun *et al.*, 2016; Garcia-Giron et al., 2022). Feld and Hering (2007) also showed that the spatial organisation of environmental factors may be the root source of the shared variance between space and environment. For example, the percentage of explained variation shared by environmental and spatial factors in this study was 5% highlighting that environmental effects are correlated in space. Bae *et al.* (2011) showed that a variety of environmental factors acting at multiple spatial scales control the composition patterns of stream macroinvertebrate assemblages in an exclusive or synergistic fashion. However, Rezende et al. (2014) found that the percentages of explained variation shared by spatial and environmental variables were 0.2% (local) and 2% (regional) and indicated that environmental effects are uncorrelated in space. Yet, the relative weight of environmental factors and spatial processes in determining macroinvertebrate community composition may vary depending on the size of the spatial scale (Mykrä *et al.*, 2007). The highest extents of spatial variation were associated with latitudinal gradients, whereas the possible process generating geographical patterns within drainage systems was spatial autocorrelation between nearby streams (Leszczynska *et al.*, 2017).

On the other hand, Azis and Abas (2021) recommended that water quality and biodiversity management and restoration must address land use processes at larger scales (e.g., river basin). Our study found that agriculture, urbanisation, deforestation, and were found to be severe in the Omo-Gibe and Awash river basins, contrary to the Abay river basin where the percentage of forest cover is high (27.86%), all of which resulted in a change in the quality of water and macroinvertebrate composition of streams. This might be linked to variations in population growth in the catchments that could increase the need for crop cultivation grazing, and urban expansion (Beyene *et al.*, 2018; Getachew *et al.*, 2020a). For example, Beyene et al. (2018) identified that agricultural land area increased by 39.55% at the expense of other land cover classes while forest area decreased by 42.76% in the Awash River basin in less than 15 years (1986-2000). As stated by Gaston (2005), the strain on tropical forests caused by the growing human population is one of the primary causes of biodiversity change and loss.

In our study, the indirect effects of land use on changes in environmental in-stream characteristics are most often represented by the shared explained variation between land use and environmental factors. A similar study showed that intensive agriculture might alter the physicochemical characteristics of the water and potentially impair the ecosystem (Galeti *et al.*, 2020). The ANOVA test in this study demonstrated significant differences in macroinvertebrate metrics such as the Shannon Index, Simpson Index, total taxa richness, %EPT, EPT richness, and evenness between the three river basins highlighting variations in the water quality of the streams caused by unique and shared effects the three sets of predictors. This finding was in agreement with previous studies observing strong links between macroinvertebrate composition and increased percentages of land use patterns via direct and indirect influences on streams (Meier *et al.*, 2014; Du et al., 2021). Authors also demonstrated that increased land use impacts on streams can result in higher nutrient inputs (e.g., TON and phosphate), EC, and temperature which can negatively affect the composition of pollution-intolerant macroinvertebrate taxa in streams (Fu et al., 2016; Leszczynska et al., 2017; Garcia-Giron et al., 2022). We observed that urbanisation, agriculture, grazing, and deforestation were predominant land use types at many sites selected in the Awash and Omo-Gibe river basin. Earlier studies in these basins also reported similar findings (Degefu et al., 2013; Getachew et al., 2020a). These extensive land use types particularly urbanization have the potential to drastically affect macroinvertebrate ecosystems and stream water quality (Wu, 2008). In comparison, more pollution-sensitive taxa were identified in the Abay river basin than Awash or Omo-Gibe river basins possibly because of its greater forest covers in the study streams which is in line with an earlier study (Fu et al., 2016). This highlights that streams in the forested catchment supported better composition of pollution-sensitive macroinvertebrates. This might be related to the availability of a conducive environment for biological communities (Wu, 2008; Oluyinka Christopher, 2020). For instance, Wu (2008), demonstrated that forest helps macroinvertebrate communities by providing essential habitat, absorbing carbon dioxide from the atmosphere, intercepting precipitation, reducing surface runoff, and avoiding and/or reducing flooding and soil erosion. The unique role of spatial predictors in this study contributed significantly but played a minor role in explaining the composition of the macroinvertebrate community, which is perhaps connected to the magnitude of the geographical scales of the studied area. Although species distribution is determined at the local or habitat scale, Jun et al. (2016) demonstrated that regional influences are important predictors of large-scale patterns in biodiversity.

5.6. Conclusions

The findings of this study showed that environmental, land use, and spatial factors explained more than half of the overall variation in macroinvertebrate composition. The findings of this study demonstrated the shared variation in macroinvertebrate composition due to the three sets of predictors were higher than their unique effects. The result highlights the three sets of predictors were more correlated and environmental factors are spatially structured and influenced, directly or indirectly, by land use factors. Thus, considering spatial scales and land use planning could be key strategies to improve the environmental conditions of streams, thereby enhancing the ecosystem services obtained from freshwater resources.

Chapter 6: Synthesis of research findings, limitations, and recommendations for further research

6.1. Introduction

This chapter is a summary of the main research findings according to the objectives of this thesis. The importance of this study to freshwater bioassessment in Ethiopia has been explained and concluded in each chapter of the thesis. This section provides a general synthesis of the research findings. Deterioration of water quality and biotic integrity by anthropogenic activities has been threatening freshwater ecosystem health and sustainability, as well as human health and socio-economic development (Mezgebu, 2020; Tavengwa *et al.*, 2022; Yardy *et al.*, 2022), as outlined in this study (Getachew et al., 2020b) using the driver-pressure-state-impact-response (DPSIR) framework to water vulnerability assessment (Hamouda et al., 2009). To carefully think about the challenges to water quality protection and to make a decision about what to do next, decision-makers (stakeholders) need to know the problem and act accordingly.

The rapid population growth, industrialisation, and its associated urbanisation, agricultural intensification, and habitat loss have increased pressure on the integrity of water resources in the metropolitan area of Addis Ababa and the upper Awash River basin (Getachew et al., 2020b). This trend is also common in other localities in Ethiopia, where population growth and the parallel development of industries without any waste treatment facilities and urbanisation without any service facilities (sanitation and water supply) are increasing. For example, Jimma (Kebede *et al.*, 2010; Mekonnen and Aticho, 2011), Hawassa (Worako, 2015; Lencha *et al.*, 2021), Kombolcka (Beyene et al., 2009a), Bahirdar (Mehari *et al.*, 2015), Dessie (Beyene et al., 2009a), and other towns faced similar problems in Ethiopia.

Several studies also indicated that water pollution is a general threat on a global scale nowadays (Barlund *et al.*, 2016; Vaughan and Gotelli, 2019; Mwedzi et al., 2020; Ochieng et al., 2020) and contradicts some of the proposed sustainable development goals (SDG) that should be achieved in the period between 2015 and 2030 (Le Blanc, 2015). For example, SDG 6 states that “by 2030, improve water quality by reducing pollution, eliminating dumping and minimising release of hazardous chemicals and materials, halving the proportion of untreated wastewater, and increasing recycling and safe reuse globally.” Contrary to this goal, the worst degradation is occurring in rivers, streams, lakes, and other surface and groundwater resources in Ethiopia (Taddese *et al.*, 2009; Degefu et al., 2013; Awoke et al., 2016; Keraga, 2019; Mersha et al., 2021) and other parts of the world (Schwarzenbach *et al.*, 2010; Barlund et al., 2016).

The degradation of aquatic ecosystems has provoked concerns because it has social, economic, and ecosystem health consequences (Halcrow and Ltd., 1989; Edossa *et al.*, 2010; Degefu et al., 2013; Englmaier et al., 2020; Getachew et al., 2020b). These degradations are due to various pressures and associated pollutant inputs (Anderson et al., 2006; Akalu et al., 2011; Alexandridis *et al.*, 2018; Agboola et al., 2019; Belmar *et al.*, 2019; Johnson *et al.*, 2019), contradicting the SDGs striving for the protection, restoration, and promotion of the sustainable use of ecosystems and forests, and combating environmental degradation to halt biodiversity loss (SDG-15) (Fraisl *et al.*, 2020; UNEP, 2022).

Bioassessment is a prerequisite for sustainable water resource use (Masese *et al.*, 2013). To carry out bioassessment of rivers and streams, method development using indicators such as macroinvertebrates is a priority. Based on the review carried out by Getachew et al. (2020b) (1st objective, chapter 2) and recommendations from other research works, the objective to refine the macroinvertebrate kick sampling protocol (Getachew *et al.*, 2022a) (2nd objective, chapter 3) for wadeable streams and rivers in Ethiopia has been developed. According to Velk et al (2006), kick sampling can be quite labor-intensive because of the large volume of substrates and organic material collected (Vlek et al., 2006), and consequently, several studies have tested the effectiveness of kick sampling duration for rivers and streams (Frost et al., 1971; Bradley and Ormerod, 2002; Mykra et al., 2006; Feeley et al., 2012). However, Ethiopia lacked testing of such a kick sampling method based on macroinvertebrates. Consequently, many studies, for example, (Ambelu *et al.*, 2010; Getachew et al., 2012; Mereta et al., 2012; Mereta et al., 2013; Derso *et al.*, 2015), used protocols developed for other ecoregions. Furthermore, some bioassessment metrics, for example, the biotic index (Lakew and Moog, 2015), multimetric indices (Lakew and Moog, 2015b; Alemneh et al., 2019) for rivers assessment, and a multimetric index (Mereta et al., 2013) for wetland assessment were developed in Ethiopia. However, all the kick sampling time applied to collect macroinvertebrates to develop these metrics did not use a method developed in the context of Ethiopian ecoregions. Instead, the kick sampling time used was refined or developed for other ecoregions in developed countries.

The current study determined the optimum kick sampling method for wadeable streams and rivers in Ethiopia. The results of this study found that the 2-minute riffle/run kick samples recruited macroinvertebrate taxa as efficiently as 3-minute riffle/run kick samples but required less sampling effort. Therefore, the shorter 2-minute kick-sampling time was recommended as a method for the ecological assessment of wadeable rivers and streams in the country. It allowed more study sites to be visited, thereby reducing costs (Hannaford and Resh, 1995) and the operators' fatigue in macroinvertebrate data collection, sorting, and identification which increases the efficiency of the biomonitoring program without compromising the results. The developed method would be useful for researchers, institutions, and decision-makers in the Bioassessment of rivers and streams, for example during the design and development of river water projects like hydropower dams. The method was developed using the macroinvertebrate data collected from minimally impaired sites and then it was tested along pollution gradients in the Awash River course during the investigation of the impacts of the Koka Hydropower

Dam on the structure and composition of macroinvertebrates. Based on various macroinvertebrate metrics computed from macroinvertebrate data collected by the 2-minute kick sampling method, study sites showed a significant difference between the study reaches indicating that the method also works along the pollution gradients in wadeable rivers and streams in Ethiopia.

Most of the direct drivers of surface water quality degradations currently remain constant or are growing in intensity in most ecosystems (Malmqvist and Rundle, 2002; Assessment, 2005; Board, 2005; Gabbud and Lane, 2016; Sendzimir and Schmutz, 2018). One of these challenges is damming (Friedl and Wuest, 2002; Santos *et al.*, 2017). A dam is a structure built across a stream or river to hold water back (Beck *et al.,* 2012) for different purposes and people have used different materials to build dams over the centuries. Globally, studies showed that thousands of dams are spatially concentrated along major river basins in Asia, North America, South America, and Europe (Zhang *et al.*, 2018). Similarly, several thousands of dams are constructed in Africa (Kibret *et al.*, 2015). While most developed countries in North America, Europe, and Oceania have witnessed a decline in dam construction since the 1970s, developing countries in Africa, Asia, and South America are experiencing a continued increase in the number of dams currently planned or under construction (Zhang et al., 2018). The continent of Africa in the last decade (2010-2020) has seen a renaissance in the construction of massive dams to supply more regulated water and generate hydroelectricity. Most of these dams have been located along the Nile and more specifically in Ethiopia at its source in the Abay River catchment (Abdelazim *et al.*, 2020). The Grand Ethiopian Renaissance Dam (GERD), with a capacity of 6450 MW, is a recent huge project in that basin (Abdelazim et al., 2020).

The economic benefits of dams have been assumed to outweigh the costs, thus providing the rationale for the construction of dams around the world (Concepcion and Nelson, 1999; Collier *et al.*, 2000; Beck et al., 2012). However, the development of these structures can be accompanied by negative biophysical, socio-economic, and geopolitical impacts; often through the loss of ecosystem services provided by fully-functioning aquatic systems (Beck et al., 2012; Hastings, 2014; Cooper *et al.*, 2017). Moreover, the impacts of dams can be involuntarily imposed on marginalised peoples whose livelihoods are dependent on riverine resources (Beck et al., 2012).

The present study investigated the impact of the Koka Hydropower dam on the composition and structure of macroinvertebrate communities on the Awash River. Although the assessment of the impact of the hydropower dam on the macroinvertebrate community structure and composition in this study was partly complicated by the water quality problems that originate upstream of the dam, the results showed that there were significant differences among the three study reaches in terms of several metrics addressing community structure, composition, and functional feeding attributes of macroinvertebrates that relate, directly or indirectly, to flow regime modifications. More specifically, even in the presence of several anthropogenic disturbances (Asfaw *et al.*, 2014; Getachew et al., 2020b; Mersha et al., 2021), the assemblage structures of the upstream and midstream reaches were far better than the downstream reach indicating impact differences among the reaches similar to other studies (Newbury and Bates, 2007; Gabbud and Lane, 2016; Ko et al., 2020).

The ecological consequences of dams could include changes in physical habitat and biotic composition (Bunn and Arthington, 2002). This study provides a fundamental understanding of the potential ecological impacts of hydropower dams based on macroinvertebrates. This would help decision-makers conduct environmental flow assessments (EFAs) which are the primary measures in the integration of water resource management, energy, and ecosystem conservation (Kuriqi et al., 2020).

Understanding the contributions of factors to explaining variation in the response variable (such as macroinvertebrates) is a challenge in approaches with many predictor variables (Rasch and Masata, 2006). This is particularly true in multicollinearity situations where predictor variables are linearly correlated with one another (Rasch and Masata, 2006; Yoo *et al.*, 2014). In redundancy analysis (RDA) and other multivariate analyses, we can use variance partitioning to break down the variance explained by predictor variables (Rasch and Masata, 2006). This can be used to determine the amount of variance explained by individual predictor variables (unique roles) as well as their overlap (shared roles) with other variables in the model (Yoo et al., 2014).

We can better understand the effects of predictor factors on the response variable using variance partitioning (Yoo et al., 2014). Variance partitioning estimates variance explained by variables in partial models (Borcard and Legendre, 2002; Yoo et al., 2014). Partial models are produced by employing a subset of the predictor variables from the entire RDA/CCA models (Dray et al., 2006; Rasch and Masata, 2006). The fractions of variance explained by these partial models can then be used to estimate the fraction of variation explained by each variable (Rasch and Masata, 2006).

The overall findings of this study (chapter 5) indicated that the shared effects of environment, land use, and space were significant contributors to variation in macroinvertebrate community composition. The amount of shared variation between environmental and land use factors was greater than the amount of shared variation between environmental and spatial factors, indicating that environmental and land use factors have more correlation in explaining macroinvertebrate community compositions.

6.2. Summary

Numerous international bilateral, plurilateral, and multilateral treaties and conventions were signed between countries to protect human health and environmental pollution. However, most of the signatory countries fail to implement the treaties or conventions (Willis, 2012). For example, though Ethiopia has endorsed or ratified the Basel, Stockholm, and Rotterdam agreements, its legislations and policies for environmentally sound management of hazardous chemicals and wastes are still in infancy and are inefficient in preventing the illegal dumping of waste as well as contamination of water, soil and air resources (Tadesse, 2009).

This study reviewed the details of problems of surface water pollution in the metropolitan area of Addis Ababa and the upper Awash River catchment. Urbanisation, population growth, industrialisation without proper waste management facilities, unregulated agricultural practices, uncontrolled liquid and solid wastes, and law enforcement difficulties were the major gaps identified. Furthermore, there are significant gaps limiting efforts to address the magnitude of pesticide pollution in the catchment probably due to the shortage of laboratory facilities and the costs associated with the chemicals and equipment. This study forwarded important conclusions and recommendations to the respective stakeholders. To help the bioassessment of wadeable rivers and streams in Ethiopia, a macroinvertebrate kick sampling protocol has been developed using data from minimally impacted sites and testing it along the pollution gradients. Based on the results, the 2-minute RH kick sampling method is the recommended candidate for the bioassessment of wadeable rivers and streams in the country.

Using the developed macroinvertebrate kick sampling protocol in this study, the impacts of the Koka Hydropower dam on benthic macroinvertebrate structure and composition were further investigated. Hydropower has been used for generations to provide reliable, fossil-fuel-free electricity (source of clean energy), water supply for domestic uses and agricultural irrigation, recreational opportunities, several flood-control benefits, as well as a stable system for navigation. However, several disadvantages have also been recognised because of the construction of hydropower dams. It is particularly severe in large dams where there has been extinction of the many fish and other aquatic species, huge losses of forest, the disappearance of birds in floodplains, erosion of deltas, loss of wetland and farmland, and many other irreversible impacts. These impacts can have economic and social (e.g. Community displacement) health implications (consequences). This in turn points towards the need to seek a balance between pros and cons associated with dam construction.

Currently, most public and policymakers perceive only the advantages (pros) of the hydropower dam and the impacts mentioned above are largely neglected. To create awareness among stakeholders and to recommend possible solutions to the problem, it was very crucial to conduct the ecological impacts of hydropower dams on the macroinvertebrate communities that include aquatic insects, crustaceans, annelids, mollusks, nematodes, planarians, and other invertebrates as these organisms play a critical role in the transfer of energy from basal resources (bottom layer) (e.g., algae, detritus, and associated microbes) to vertebrate consumers in aquatic food webs, and they serve as the primary food resource for many commercially and economically important fish species. The investigation of the hydropower dam on the structure and composition of macroinvertebrate communities showed that the downstream reach was severely impacted by water pollution compared to the midstream and upstream reaches of the Awash River, however, the results were partly complicated by the nutrient pollution, particularly from the midstream reach. The major factor, amongst others affecting the downstream reach, the water flow change (river velocity) was highlighted as a key variable structuring the macroinvertebrate communities and a variable that can be altered by damming.

Similarly, variation partitioning analysis was used to analyze the unique and shared effects of the three sets of predictor variables (environmental, land use, and spatial factors). The findings revealed that the three sets of predictors' shared variation in macroinvertebrate community composition was greater than their contributions. To put it bluntly, anthropogenic activities have a negative global influence on aquatic habitats. As a result, surface water biomonitoring is becoming increasingly crucial in estimating ecosystem deterioration and achieving environmental sustainability. Therefore, the development of methods kick sampling time) based on Ethiopian ecoregions to employ in the bioassessment of surface water quality has been mandatory. This would contribute to the assessment of water quality in wadeable rivers and streams in Ethiopia.

6.3. Limitations

This study did not include the impacts of toxic metals and pesticides on the macroinvertebrate communities due to constraints such as financial, logistical, and time constraints. The study generally used data collected in the dry season for several reasons. For example, most of the assessment studies carried out in Ethiopia were conducted in the dry season; this dry season is long (9 months/year), is the safe season for entering rivers in Ethiopia (index period), turbidity is low in the dry season, and macroinvertebrates can be effectively sampled. It is not generally recommended to sample in high-flow conditions. Apart from health and safety issues, many invertebrates move deeper into benthic substrates during high flow and are not effectively captured by kick sampling. However, wet season data might provide some additional information on biological responses to pollution as well as knowledge of the life cycle and other ecological information. Any sampling carried out in the wet season would have to be carried out between periods of high flow.

6.4. Recommendations

This study has highlighted several knowledge gaps to be addressed, many of which have been highlighted in the various chapters. Studies on the impact of toxic metals and pesticides on the structure and composition of macroinvertebrate communities are required to address significant gaps in this area. This will help refine the biological indicators and metrics to detect various types of pollution, particularly in a multi-stressor environment. Similarly, studies on macroinvertebrate communities should also include the wet season, bearing in mind the points raised above, particularly in the investigation of hydropower dam impacts and evaluation of mitigation measures.

This study used the macroinvertebrate identification key developed for South Africa with the concept that the ecoregions of this country are more similar to Ethiopia than those found in non-tropical regions. That means Ethiopia lacks a macroinvertebrate identification key based on the taxa identified in the country. Therefore, efforts should be made to develop macroinvertebrate identification keys in the country.

Developed regions use biotic metrics integrated with physicochemical and hydromorphological parameters in their surface water monitoring programs. Conversely, the monitoring of surface water resources in Ethiopia is based on only selected physicochemical parameters selected by decision-makers such as the Environmental Protection Authority at the federal level and the respective stakeholders at regional state governments in the country. Therefore, the government should work towards establishing biomonitoring programs for the assessment and monitoring of surface water resources. This will require upskilling or the recruitment of staff with the required expertise.

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8. Appendices

Annex 8.1. Recommended equipment for field sampling

Ethanol (96%)

Sample containers, sample container labels

Pencils, clipboard

First aid kit

Rubber gloves (arm-length)

Digital camera

Global Positioning System (GPS) Unit

5-gallon bucket

D-frame net or Surber sampler

Small hand net or aquarium net

Wash bottles, 250 mL

Round metal sieve with 500µm mesh

Entomological forceps or fine tweezers

Small hand trowel or 3-pronged rake

Small brush

Waders (chest-high or hip boots or Knee boots)

Shallow white plastic tray

Field data sheets

Hand lens

Macroinvertebrate identification field guide

Annex 8.2. Data collection tools

The physical and chemical variables collected encompass the factors that influence the distribution of macroinvertebrates on a catchment, reach, and individual habitat scale. These factors are geographical position, riparian vegetation, channel morphology, water chemistry, habitat composition, habitat characteristics, organic substratum, inorganic substratum, and hydrology (Parsons et al., 2002) and are mentioned below.

Physical and chemical variables are commonly measured in this study. Compiled from Parsons et al., 2002

Geographical position

- Altitude__________________masl
- Latitude__________________dms
- Longitude_________________ dms
- Distance from source__________km
- Channel slope________________%

Riparian vegetation

- Width of riparian zone______m
- Cover of riparian zone by trees, shrubs, grasses ____________%
- Canopy cover of river _______%

Channel morphology and discharge

- Reach length___________m
- River width____________m
- River depth____________m
- Bank width____________m
- Bank height___________ m
- Water velocity__________m/s
- Discharge_____________m^3/s

Water chemistry

- Temperature___________ºC
- Conductivity___________µS/cm
- pH___________________unit
- Dissolved oxygen_______mg/L
- Oxygen saturation ________%
- Turbidity______________NTU
- Total dissolved solids_________mg/L

Chemicals analyzed in the lab

- BOD5_______________________mg/L
- Nitrate-nitrogen_______________mg/L
- Total phosphate_______________mg/L
- Suspended solids______________mg/L

Riffle/channel/sand bed habitat characteristics (%)

- Bedrock_________________________
- Boulder_________________________
- Cobble_________________________
- Pebble_________________________
- Gravel_________________________
- Sand_________________________
- Silt/clay_______________________
- **Substrate index________________**

Site observations

- Local point source and non-point source pollution______ (Yes/No)
- Odor __________(Yes/No)
- Flow restrictions/Dams____________(Yes/No)
- Local bank and catchment erosion____(Yes/No)
- Land use type:
- Urban________________________(Yes/No)
- Agriculture____________________(Yes/No)
- Forest________________________(Yes/No)
- Other (Explain)_________________________

Habitat composition

- Riffle________________%
- Run/glide_____________%
- Pool_________________%

Habitat assessment field data sheet (1)

High gradient streams

Date______________________ Recorders Name___________________________

Site #_____________________ River and Location_________________________

Habitat parameter	Condition category			
	Excellent	Good	Fair	Poor
1. Epifaunal (bottom) substrate/ available cover	Greater than 70% of substrate favorable for epifaunal colonization and fish cover; the mix of snags, submerged logs, undercut banks, cobble, or other stable habitat and at the stage to allow full colonization potential (i.e. logs/snags that are not new fall and not transient).	40-70% mix of stable habitat; well suited for full colonization potential; adequate habitat for maintenance of populations; the presence of additional substrate in the form of snowfall, but not prepared for colonization (may rate at high end of scale).	20-40% mix of stable habitat; habitat availability less than desirable; substrate frequently disturbed or removed.	Less than 20% stable habitat; lack of habitat is obvious; substrate unstable or lacking.
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
2. Embeddedness	Gravel, cobble and boulder particles are 0-25% surrounded by fine sediment. Layering of cobble provides diversity of niche space.	Gravel, cobble and boulder particles are 25-50% surrounded by fine sediment.	Gravel, cobble and boulder particles are 50-75% surrounded by fine sediment.	Gravel, cobble and boulder particles are more than 75% surrounded by fine sediment.
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
3. Velocity / depth regime	All four velocity/depth regimes present (slow-deep, slow shallow, fast-deep, fast-shallow). Slow is <0.3m/s, deep is >0.5m).	Only 3 of the 4 regimes present (if fast-shallow is missing, score lower than if missing other regimes).	Only 2 of the 4 habitat regimes present (if fast-shallow or slow-shallow are missing, score low).	Dominated by 1 velocity/depth regime (usually slow-deep).
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
4. Sediment deposition	Little or no enlargement of islands or point bars and less than 5% of the bottom affected by sediment deposition.	Some new increase in bar formation, mostly from gravel, sand or fine sediment; 5-30% of the bottom affected; slight deposition in pools.	Moderate deposition of new gravel, sand or fine sediment on old and new bars; 30-50% of the bottom affected; sediment deposits at obstructions, constrictions and bends; moderate deposition in pools prevalent.	Heavy deposits of fine material, increased bar development; more than 50% of the bottom changing frequently; pools almost absent due to substantial sediment deposition.
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
5. Channel flow status	Water reaches base of both lower banks, and minimal amount of channel substrate is exposed.	Water fills >75% of the available channel; or <25% of channel substrate is exposed.	Water fills 25-75% of the available channel, and/or riffle substrates are mostly exposed.	Very little water in channel and mostly present as standing pools.

SCORE	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0

Habitat assessment field data sheet (2)

Habitat parameter	Condition category			
	Excellent	Good	Fair	Poor
6. Channel alteration	Channelization or dredging absent or minimal; stream with normal pattern.	Some channelization present, usually in areas of bridge abutments; evidence of past channelization, i.e. dredging (greater than 20 yr.) may be present, but recent channelization is not present.	Channelization may be extensive; embankments or shoring structures present on both banks; and 40 to 80% of stream reach channelized and disrupted.	Banks shored with gabion or cement; over 80% of the stream reach channelized and disrupted. In stream habitat greatly altered or removed entirely.
SCORE	20 19 18 17 16	15 14 13 12 11 10	9 8 7 6 5 4	3 2 1 0
7.Frequency of riffles (or bends)	Occurrence of riffles relatively frequent; ratio of distance between riffles divided by width of the stream <7:1 (generally 5 to 7); variety of habitat is key. In streams where riffles are continuous, placement of boulders or other large, natural obstruction is important.	Occurrence of riffles infrequent; distance between riffles divided by the width of the stream is between 7 and 15.	Occasional riffle or bend; bottom contours provide some habitat; distance between riffles divided by the width of the stream is between 15 and 25.	Generally all flat water or shallow riffles; poor habitat; distance between riffles divided by the width of the stream is a ratio of >25.
SCORE	20 19 18 17 16	15 14 13 12 11 10	9 8 7 6 5 4	3 2 1 0
8.Bank stability (score each bank)	Banks stable; evidence of erosion or bank failure absent or minimal; little potential for future problems. <5% of bank affected.	Moderately stable; infrequent, small areas of erosion mostly healed over. 5-30% of bank in reach has areas of erosion.	Moderately unstable; 30-60% of bank in reach has areas of erosion; high erosion potential during floods.	Unstable; many eroded areas; 'raw' areas frequent along straight sections and bends; obvious bank sloughing; 60-100% of bank has erosional scars.
SCORE	20 19 18 17 16	15 14 13 12 11 10	9 8 7 6 5 4	3 2 1 0
9. Vegetative protection (score each bank)	More than 90% of the stream bank surfaces and immediate riparian zone covered by native vegetation, including trees, understory shrubs, or non-woody macrophytes; vegetative disruption through grazing or mowing minimal or not evident; almost all plants allowed to grow naturally.	70-90% of the stream bank surfaces covered by native vegetation, but one class of plants is not well-represented; disruption evident but not affecting full plant growth potential to any great extent; more than one half of the potential plant stubble height remaining.	50-70% of the stream bank surfaces covered by vegetation; disruption obvious; patches of bare soil or closely cropped vegetation common; less than one-half of the potential plant stubble height remaining.	Less than 50% of the stream bank surfaces covered by vegetation; disruption of stream bank vegetation is very high; vegetation has been removed to 5 centimeters or less in average stubble height.
SCORE	20 19 18 17 16	15 14 13 12 11 10	9 8 7 6 5 4	3 2 1 0
10.Riparian zone score(score each bank)	Width of riparian zone >18 meters; human activities (i.e. roads, lawns, crops etc.) have not influenced the riparian zone.	Width of riparian zone 12-18 meters; human activities have influenced riparian zone only minimally.	Width of riparian zone 6-12 meters; human activities have influenced zone a great deal.	Width of riparian zone <6 meters; little or no riparian vegetation is present because of human activities.
SCORE	20 19 18 17 16	15 14 13 12 11 10	9 8 7 6 5 4	3 2 1 0

Total habitat score _________________

Annex 8.3. Supplementary materials

Table 4.S1: The genus and species of macroinvertebrate presence/absence data collected from the Awash River above and below the Koka Hydropower Dam and categorised by their order and family (Harrison, 2009; Lowe, 2009; Schael, 2010; Morse *et al.*, 2020)

Order	Family	*Genus/species*	US1	US2	US3	US4	US5	MS1	MS2	MS3	MS4	MS5	DS1	DS2	DS3	DS4	DS5
Annelida	Oligochaeta	*Oligochaeta*								x	x	x			x	x	x
Basommatophora	Physidae	*Physa*	x	x			x	X	x			x	x		x	x	x
	Planorbidae	*Helisoma*	x	x	x	x	x		x		x	x			x	x	x
Coleoptera	Dytiscidae	*Agabus*	x	x	x												
		Copelatus	x	x	x	x	x	X	x	x	x	x	x		x		x
	Elmidae	*Ancyronyx*	x	x		x	x										
		Dubiraphia	x	x	x	x											
		Macronychus	x	x		x	x										
		Microcylloepus	x		x				x								
		Optioservus	x	x	x	x	x	X	x								
		Stenelmis	x		x	x	x										
	Gyrinidae	*Dineutus*	x	x								x					
		Gyrinus			x	x	x		x	x	x						
	Haliplidae	*Peltodytes*			x			X					x	x		x	x
	Hydrophilidae	*Berosus*	x							x						x	
		Tropisternus	x	x			x	X									
	Psephenidae	*Psephenus*	x	x	x												
Diptera	Athericidae	*Atherix*	x	x	x		x	X									
	Ceratopogonidae	*Ceratopogon*	x	x	x		x	X			x						
		Dasyhelea				x		X	x	x	x	x	x	x	x	x	
	Chironomidae	*Chironomus*	x	x	x	x	x	X	x	x		x	x	x	x	x	x
		Orthocladius	x	x	x	x	x	X		x	x	x	x	x	x		x
		Tanytarsus	x			x	x	X	x	x	x	x	x				
	Culicidae	*Aedes*	x	x		x	x	X	x	x	x	x	x	x	x	x	x
	Empididae	*Hemerodromia*		x					x	x			x	x	x	x	x
	Muscidae	*Musca*	x	x		x	x	X	x	x	x	x	x	x	x	x	x
	Simuliidae	*Prosimulium*	x	x	x	x	x	X	x	x	x	x	x	x	x	x	x
		Simulium	x	x	x	x		X	x	x	x	x	x	x		x	x
	Syrphidae	*Chrysogaster*						X		x		x	x	x	x	x	x
	Tabanidae	*Tabanus*	x	x	x	x	x	X	x	x	x	x					
	Tipulidae	*Tipula*	x	x	x		x	X	x	x	x	x					
Ephemeroptera	Baetidae	*Heterocloeon*	x	x	x	x	x	X	x	x	x	x	x	x	x	x	x
		Baetis	x	x	x	x	x	X	x	x	x	x		x	x	x	x
	Caenidae	*Caenis*	x	x	x	x	x	X	x	x	x	x	x	x	x	x	x
	Heptageniidae	*Heptagenia*	x	x	x	x	x	X	x	x	x	x					
		Leucrocuta	x	x	x	x	x	X	x		x	x					
		Stenacron	x	x	x	x	x	X		x	x	x					
	Leptohyphidae	*Tricorythodes*	x	x	x	x	x										
Hemiptera	Belostomatidae	*Belostoma*	x	x	x	x	x	X	x			x					x
	Corixidae	*Hesperocorixa*	x	x	x			X		x		x					
	Gerridae	*Gerris*	x	x					x	x	x					x	
	Naucoridae	*Pelocoris*	x		x			X	x								
	Nepidae	*Curicta*							x	x			x			x	x
	Veliidae	*Microvelia*	x	x	x	x	x	X	x	x	x						
		Rhagovelia	x		x		x									x	x

Table 4.S1: The genus and species of macroinvertebrate presence/absence data collected from the Awash River above and below the Koka Hydropower Dam and categorised by their order and family (continued)

Order	Family	*Genus/Species*	US1	US2	US3	US4	US5	MS1	MS2	MS3	MS4	MS5	DS1	DS2	DS3	DS4	DS5
Lepidoptera	Crambidae	*Crambinae*					x	x	x		x	x					
Odonata	Aeshnidae	*Boyeria*	x	x		x											
	Gomphidae	*Lanthus*	x	x	x	x	x	x							x	x	x
		Ophiomorphic	x	x	x	x	x	x	x	x	x						
	Libellulidae	*Libellula*	x	x	x	x		x		x	x						
Plecoptera	Perlidae	*Neoperla*	x	x	x												
Trichoptera	Ecnomidae	*Ecnomus similis*	x	x													
		Ecnomus thomasseti	x	x					x								
	Glossosomatidae	*Glossosoma*	x		x		x	x									
	Hydropsychidae	*Amphipsyche senegalensis*	x	x	x	x	x	x	x	x	x	x	x	x			x
		Cheumatopsyche afra	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
		Cheumatopsyche columnata	x	x	x	x	x	x	x	x	x		x	x	x	x	x
		Cheumatopsyche falcifera	x	x	x	x	x	x	x	x	x	x	x		x	x	x
		Cheumatopsyche massa	x	x	x	x	x	x	x	x	x	x					
		Cheumatopsyche sexfasciata	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
		Hydropsyche abyssinica	x	x	x	x	x	x	x	x	x	x	x				
	Hydroptilidae	*Hydroptila cruciata*	x	x		x	x										
		Orthotrichia thariel	x	x	x	x	x		x								
	Lepidostomatidae	*Lepidostoma scotti*	x	x													
	Leptoceridae	*Athripsodes fissus*	x	x	x	x	x										
		Oecetis armaros	x				x		x								
		Oecetis reticulatella	x	x	x	x	x										
		Oecetis tripunctata	x				x										
		Tagalopsyche aethiopica				x											
		Triaenodes serratus	x			x		x									
Unionoida	Unionidae	*Elliptio*			x	x	x	x	x	x	x	x	x	x	x		x
Veneroida	Cyrenidae	*Cyreninae*	x	x		x	x	x	x	x	x	x	x	x	x	x	x
	Sphaeriidae	*Pisidium*				x							x	x	x	x	

Table 5.S1: The genus of macroinvertebrate presence/absence data collected from thirty sampling site of Abay, Omo-Gibe, and Awash River basins in Ethiopia and categorised by their order and family

Order	Family (Fa)	Fa ID	Genus	S1U	S2U	S3U	S4Y	S5Y	S6Y	S7F	S8F	S9F	S10G	S11G	S12G	S13Gu	S14Gu	S15Gu	S16Gf	S17Gf	S18Gf	S19D	S20D	S21D	S22S	S23S	S24S	S25J	S26J	S27J	S28A	S29A	S30A
Araneae	Pisauridae	1	Archipirata	x	x	x	x	x	x				x			x	x											x					
Basommatophora	Thiaridae	2	Melanoides																														
Annelida	Glossiphoniidae	3	Placobdella	x	x	x	x	x	x	x	x		x	x					x	x								x	x	x	x		
	Oligochaeta	4	Oligochaeta		x	x				x	x	x						x				x	x	x	x	x	x	x			x	x	x
Coleoptera	Dytiscidae	5	Agabus	x	x					x	x	x																	x		x	x	x
			Copelatus				x	x	x				x	x	x	x	x												x				
	Curculionidae	6	Curculio					x	x	x			x	x	x	x	x	x															
	Hydraenidae	7	Hydraena	x	x	x	x	x	x	x			x	x	x	x	x	x										x	x				
	Scirtidae	8	Elodes					x					x						x	x								x	x	x			
	Elmidae	9	Ancyronyx	x	x	x	x	x	x		x		x	x	x	x	x	x	x			x	x	x				x	x	x	x	x	
			Dubiraphia	x	x		x	x	x	x	x	x		x		x	x	x										x			x		
			Macronychus				x	x		x		x	x	x		x	x																
			Microcylloepus	x										x	x																		
			Optioservus				x	x					x																				
			Stenelmis	x	x		x		x	x				x		x	x											x					
	Gyrinidae	10	Dineutus	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x				x	x	x	x	x	x			x	x	x	x
			Gyrinus	x						x		x	x	x																		x	x
	Haliplidae	11	Peltodytes	x	x	x	x	x	x	x			x	x	x	x	x	x	x	x								x		x			
	Hydrophilidae	12	Berosus					x	x				x	x	x							x	x							x	x	x	
			Tropisternus	x	x		x							x	x				x	x		x						x	x	x		x	x
	Psephenidae	13	Psephenus	x	x	x	x	x	x	x		x	x		x	x	x	x				x			x			x	x			x	x

Table 5.S1: The genus of macroinvertebrate presence/absence data collected from thirty sampling site of Abay, Omo-Gibe, and Awash River basins in Ethiopia and categorised by their order and family (continued)

Order	Family (Fa)	Fa. ID	*Genus/ species*	S1U	S2U	S3U	S4Y	S5Y	S6Y	S7F	S8F	S9F	S10G	S11G	S12G	S13Gu	S14Gu	S15Gu	S16Gf	S17Gf	S18Gf	S19D	S20D	S21D	S22S	S23S	S24S	S25J	S26J	S27J	S28A	S29A	S30A
Diptera	Athericidae	14	*Atherix*	x	x	x	x	x	x		x		x	x		x	x	x	x													x	
	Ceratopogonidae	15	*Probezzia*	x	x		x	x	x				x	x	x	x	x	x		x								x	x	x		x	x
			Dasyhelea				x		x				x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Ephydridae	16	*Ochthera*					x								x	x		x		x	x	x	x	x	x	x	x	x	x	x	x	X
	Chironomidae	17	*Chironomus*							x	x	x			x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
			Orthocladius	x	x	x	x	x	x	x	x	x		x	x	x	x	x	x		x	x	x	x				x	x	x	x	x	x
			Tanytarsus	x	x	x	x	x	x	x		x	x	x	x	x	x	x				x	x	x	x	x	x				x	x	x
	Culicidae	18	*Aedes*	x	x	x	x	x	x		x	x	x	x	x	x	x	x					x	x	x	x	x				x	x	x
	Dixidae	19	*Dixa*	x	x								x																				
	Empididae	20	*Hemerodromia*																x	x										x			
			Ephydridae							x	x	x				x	x	x				x	x	x							x	x	x
	Limoniidae	21	*Antocha*	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x										x			x	x	x
			Hexatoma	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Muscidae	22	*Musca*	x	x	x	x	x	x	x	x	x				x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Pediciidae	23	*Dicranota*						x	x	x											x	x	x	x		x	x	x	x	x	x	x
	Psychodidae	24	*Pericomaina*	x	x		x	x	x		x		x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x		x
	Tabanidae	25	*Tabanus*	x	x	x	x	x	x	x	x		x	x	x	x	x	x		x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Simuliidae	26	*Prosimulium*				x				x	x			x	x	x	x		x	x	x	x	x	x	x	x	x	x	x	x	x	x
			Simulium	x	x	x	x	x	x	x	x	x			x	x	x	x				x	x	x	x	x	x			x	x	x	x
	Stratiomyidae	27	*Odontomyia*	x	x		x		x	x	x	x			x	x	x	x				x	x			x					x	x	x
	Syrphidae	28	*Chrysogaster*			x									x	x	x	x	x	x		x	x	x			x	x	x	x			x
	Tipulidae	29	*Tipula*	x	x		x	x		x	x	x	x	x	x	x	x	x	x		x	x	x	x	x	x	x	x	x	x	x	x	x

Table 5.S1: The genus of macroinvertebrate presence/absence data collected from thirty sampling site of Abay, Omo-Gibe, and Awash River basins in Ethiopia and categorised by their order and family (continued)

Order	Family (Fa)	Fa. ID	Genus/ species	S1U	S2U	S3U	S4Y	S5Y	S6Y	S7F	S8F	S9F	S10G	S11G	S12G	S13Gu	S14Gu	S15Gu	S16Gf	S17Gf	S18Gf	S19D	S20D	S21D	S22S	S23S	S24S	S25J	S26J	S27J	S28A	S29A	S30A
Ephemeroptera	Baetidae	30	Baetis	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
			Heterocloeon	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Baetiscidae	31	Baetisca	x	x	x	x	x	x				x	x																			
	Caenidae	32	Caenis	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x		x	x	x	x	x	x	x	x
	Ephemerellidae	33	Attenella	x	x	x	x	x	x	x	x	x	x	x	x																		
			Ephemerella	x	x	x	x	x	x	x	x		x	x	x	x	x														x		
			Serratella				x	x	x				x	x	x				x	x	x							x	x	x			
	Heptageniidae	34	Heptagenia	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
			Leucrocuta	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x				x	x	x	x	x		x	x	x	x	x	x
			Stenacron	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x				x	x								x	x	x
	Leptohyphidae	35	Tricorythodes	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x				x	x			x					x	x	
	Prosopistomatidae	36	Prosopistoma	x	x		x	x	x	x			x	x														x	x	x			
	Leptophlebiidae	37	Habrophlebia				x	x	x				x	x	x	x	x	x				x	x	x		x		x	x	x	x	x	x
			Leptophlebia		x	x	x	x	x	x			x	x	x	x	x	x	x	x	x	x	x					x		x	x	x	x
Hemiptera	Belostomatidae	38	Belostoma	x	x		x			x	x	x	x	x	x	x	x	x				x	x	x	x	x	x	x		x	x	x	x
	Pleidae	39	Neoplea	x		x	x	x	x	x	x		x	x	x	x	x	x				x						x		x		x	x
	Corixidae	40	Hesperocorixa	x	x	x	x	x	x	x	x	x	x	x	x				x	x	x	x	x	x	x	x	x	x	x	x	x	x	
	Gerridae	41	Gerris	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x				x	x	x	x	x	x
	Naucoridae	42	Pelocoris	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Nepidae	43	Curicta										x					x	x	x	x	x	x	x						x	x		x
	Notonectidae	44	Notonecta	x	x	x	x	x						x	x	x	x	x	x	x	x	x	x	x				x	x	x	x	x	x
	Veliidae	45	Microvelia	x	x	x	x	x	x	x	x	x				x	x	x	x	x	x	x	x	x	x	x	x	x	x		x		x
			Rhagovelia	x	x	x	x	x					x	x	x	x	x		x	x	x	x	x	x		x	x		x	x	x	x	
	Mesoveliidae	46	Mesovelia				x	x	x				x	x	x	x	x	x	x	x	x	x	x	x				x	x	x	x	x	x

Table 5.S1: The genus of macroinvertebrate presence/absence data collected from thirty sampling site of Abay, Omo-Gibe, and Awash River basins in Ethiopia and categorised by their order and family (continued)

Order	Family (Fa)	Fa. ID	Genus/ species	S1U	S2U	S3U	S4Y	S5Y	S6Y	S7F	S8F	S9F	S10G	S11G	S12G	S13Gu	S14Gu	S15Gu	S16Gf	S17Gf	S18Gf	S19D	S20D	S21D	S22S	S23S	S24S	S25T	S26T	S27T	S28A	S29A	S30A
Lepidoptera	Crambidae	47	Parapoynx		x	x	x	x					x	x		x	x	x	x	x	x	x	x	x	x	x	x		x	x	x	x	x
Megaloptera	Corydalidae	48	Nigronia	x	x	x	x	x	x				x	x	x	x	x	x										x	x				
	Sialidae	49	Sialis	x	x		x	x	x	x	x	x	x	x	x	x	x	x										x	x	x		x	
Odonata	Aeshnidae	50	Boyeria	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Calopterygidae	51	Calopteryx	x	x	x	x	x	x	x	x	x	x	x	x	x	x		x	x		x	x			x		x	x	x	x	x	
	Coenagrionidae	52	Argia	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x				x	x	x	x	x	x
			Enallagma	x	x	x	x	x	x				x	x	x	x	x	x	x	x	x	x	x	x				x	x	x	x	x	
			Ischnura	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x				x	x	x	x	x	
	Gomphidae	53	Lanthus	x	x	x	x	x	x	x	x	x	x	x		x	x	x			x	x		x				x	x	x	x	x	
			Ophiogomphus	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x				x	x	x			x		x	x	x	x	x
	Lestidae	54	Lestes	x	x	x	x						x			x	x	x	x	x	x	x	x			x		x	x	x	x		
	Libellulidae	55	Libellula	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Plecoptera	Perlidae	56	Neoperla	x	x	x	x	x	x	x	x		x	x	x	x	x	x				x							x				
Trichoptera	Brachycentridae	57	Brachycentrus	x	x	x	x	x	x				x	x	x	x	x	x										x	x			x	
			Micrasema	x	x	x	x	x	x	x	x	x	x	x	x	x	x															x	
	Glossosomatidae	58	Glossosoma	x	x	x	x	x	x	x	x	x	x	x		x	x		x	x	x	x						x	x	x		x	
	Hydropsychidae	59	Arctopsyche	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x				x	x	x	x		x
			Cheumatopsyche	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x						x	x	x	x	x	x
			Hydropsyche	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
			Macrostemum	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x

Table 5.S1: The genus of macroinvertebrate presence/absence data collected from thirty sampling site of Abay, Omo-Gibe, and Awash River basins

Order	Family (Fa)	Fa. ID	Genus	S1T	S2T	S3T	S4Y	S5Y	S6Y	S7F	S8F	S9F	S10G	S11G	S12G	S13Gu	S14Gu	S15Gu	S16Gf	S17Gf	S18Gf	S19D	S20D	S21D	S22S	S23S	S24S	S25J	S26J	S27J	S28A	S29A	S30A
	Hydroptilidae	60	Hydroptila	x	x	x	x	x	x	x	x	x	x	x		x	x	x				x	x	x	x	x	x	x			x	x	x
	Lepidostomatidae	61	Lepidostoma	x	x	x	x	x	x	x	x		x	x	x	x	x	x											x				
	Leptoceridae	62	Ceraclea	x			x	x	x	x			x			x	x	x															
			Nectopsyche				x	x		x						x	x															x	
			Oecetis	x	x		x	x	x				x	x	x	x	x	x										x					
	Limnephilidae	63	Ironoquia	x	x	x	x	x	x				x	x	x	x	x	x															
			Platycentropus					x	x	x	x	x																					
	Molannidae	64	Molanna	x	x	x	x	x	x				x	x																	x		
	Odontoceridae	65	Psilotreta	x	x				x	x	x	x																					
	Sericostomatidae	66	Agarodes										x	x	x																		
Decapoda	Potamonautidae	67	Potamonautes	x	x	x	x	x	x				x	x	x	x	x		x	x		x	x					x	x	x	x		
Unionoida	Unionidae	68	Elliptio	x	x	x	x	x		x	x	x										x	x		x	x	x		x		x	X	
Veneroida	Corbiculidae	69	Corbicula												x				x	x	x	x		x	x		x		x	x			
	Sphaeriidae	70	Pisidium				x									x	x	x	x	x	x	x	x	x		x		x	x		x		x
Basommatophora	Physidae	71	Physa				x	x					x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x		x	x	
	Planorbidae	72	Helisoma				x	x	x	x	x		x	x	x	x	x	x	x	x	x	x						x	x		x		x

Annex 8.4. Training and International presentation certificates

UCD School of Biology and Environmental Science

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Science Centre - West
University College Dublin
Belfield, Dublin 4, Ireland
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Ionad Eolaíochta – Thiar
An Coláiste Ollscoile, Baile Átha Cliath
Belfield, Baile Átha Cliath 4, Éire
BiolandEnv@ucd.ie

Certificate of completion

This is to certify that Mr Melaku Getachew, PhD candidate in Water and Health, Ethiopian Institute of Water Resources, Addis Ababa University, successfully completed a training course in macroinvertebrate identification (July 29th to August 26th 2017) in the School of Biology and Environmental Science, University College Dublin, Ireland.

Signed

Mary Kelly-Quinn

Associate Professor Mary Kelly-Quinn

b) International presentation's certificates

SIL 2018 Nanjing China

International Society of Limnology

XXXIV Congress

Nanjing China August 19-24 2018

Certificate of Attendance

We confirm that <u>Melaku Getachew Tadesse</u> participated at XXXIV Congress of International Society of Limonology held in Nanjing, China, 19-24 August, 2018.

He gave an oral presentation on <u>"Challenges for water quality protection in the environs of Addis Ababa, Ethiopia"</u>.

On behalf of the Local organizing committee:

Zhengwen Liu

Zhengwen Liu
Co-chair
Secretariat of SIL 2018
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SIL 2018 Nanjing China

International Society of Limnology
XXXIV Congress
Nanjing China
August 19-24 2018

Certificate of Attendance

We confirm that Melaku Getachew Tadesse

participated at XXXIV Congress of International Society of Limnology held in Nanjing, China, from19 to 24 August, 2018.

On behalf of the Local organizing committee:

Zhengwen Liu

Zhengwen Liu
Co-chair
Secretariat of SIL 2018
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Annex 8.5. Macroinvertebrate sorting and identification processes in the laboratory


LIMNOLOGY

Annex 8.6. Abstracts of published articles and unpublished manuscript

Article I:

Getachew, M., Mulat, W. L., Mereta, S. T., Gebrie, G. S., & Kelly-Quinn, M. (2020). Challenges for water quality protection in the greater metropolitan area of Addis Ababa and the Upper Awash basin, Ethiopia – time to take stock. Environmental Reviews, 29(1), 87-99. https://doi.org/10.1139/er-2020-0042

NRC Research Press

REVIEW

Challenges for water quality protection in the greater metropolitan area of Addis Ababa and the upper Awash basin, Ethiopia – time to take stock

Melaku Getachew, Worku Legesse Mulat, Seid Tiku Mereta, Geremew Sahilu Gebrie, and Mary Kelly-Quinn

Abstract: Ethiopia, the second-most populous country in Africa after Nigeria, has more than one hundred million people and is one of the world's fastest-growing countries in terms of economy. It has 12 major river basins with an annual renewable flow of 122 billion m^3. The country is facing increasing pressures on water resources both in terms of quantity and quality. Many researchers have highlighted that water pollution is severe and increasing particularly in the environs of Addis Ababa because of complex anthropogenic factors. The objective of this review was to synthesize the key results of research to date on the water quality in the environs of Addis Ababa and use that information to highlight management gaps, challenges, and future research needs. According to the studies reviewed, water pollution pressures result from rapid urbanization and industrial expansion without adequate solid waste management and wastewater treatment facilities, and agricultural activities. The problems are compounded by law enforcement difficulties. Trace metal contamination of rivers, streams, reservoirs, and their bioaccumulation in vegetables highlight the urgency of addressing water pollution in the upper Awash catchment. Most studies agreed that water from reservoirs, rivers, and streams in the environs of Addis Ababa is unfit for human consumption as it contains a wide range of pollutants that could affect community health. Hence effective pollution detection, mitigation measures, and monitoring including the development of bioassessment tools, together with cost-effective management measures are urgently required to reverse the decline in water quality in Ethiopia in general and in the greater metropolitan area of Addis Ababa and the upper Awash basin in particular.

Key words: Addis Ababa, metropolitan, waste management, upper Awash basin, water quality.

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Article II:

Getachew, M., Mulat, W. L., Mereta, S. T., Gebrie, G. S., & Kelly-Quinn, M. (2022). Refining benthic macroinvertebrate kick sampling protocol for wadeable rivers and streams in Ethiopia. Environmental monitoring and assessment, 194(3), 196. https://doi.org/10.1007/s10661-021-09594-x

Environ Monit Assess (2022) 194:196
https://doi.org/10.1007/s10661-021-09594-x

Refining benthic macroinvertebrate kick sampling protocol for wadeable rivers and streams in Ethiopia

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Seid Tiku Mereta · Geremew Sahilu Gebrie ·
Mary Kelly-Quinn

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Abstract Streams and rivers cover a larger proportion of the Earth's surface but are highly affected by human pressures. Conversely, bioassessment methods are in their infancy in developing countries such as Ethiopia. In this study, we compared 2- and 3-min macroinvertebrate kick samples at multiple locations for both riffle habitat (RH) and multihabitat (MH) approaches. The performance of each method was evaluated statistically using benthic macroinvertebrate metrics and diversity indices. Results of the Kruskal–Wallis analysis in this study showed no significant differences among methods tested in minimally impacted streams in Ethiopia and generally performed equally irrespective of the methods employed except for total abundances and Ephemeroptera abundances. Furthermore, multivariate analysis of the relative abundances of macroinvertebrate communities using analysis of similarity (ANOSIM), RELATE, non-metric multidimensional scaling (MDS), and classification strength-sampling method comparability (CS-SMC) indicated a high similarity in the macroinvertebrate communities recorded among all methods employed in this study area. However, the index of multivariate dispersion (IMD) test showed variations in relative abundances of macroinvertebrate communities among the methods. In summary, if the focus is not on rare taxa and the required information is not dependent on additional evidence provided by the use of lower taxonomic levels of identification (genus and species), the results of the present study support the use of the shorter 2-min RH kick sampling method for the bioassessment of wadeable rivers and streams in Ethiopia.

Keywords Bioassessment of streams and rivers · Fixed sampling time · Single and multihabitat sampling · Macroinvertebrates · Ethiopia

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Introduction

Highland streams in Ethiopia have been reported to show deteriorated water quality from severe

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Article III:

ORIGINAL ARTICLE

Impacts of the Koka hydropower dam on macroinvertebrate assemblages in the Awash River Basin in Ethiopia

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Key words: macroinvertebrates; environmental factors; substrate index; taxonomy-based metrics; damming; flow regimes.

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ABSTRACT

The Koka hydropower dam is one of the oldest large dams in Ethiopia. Damming is one of the anthropogenic activities impacting the distribution of aquatic life forms. However, to date, little attention has been focused on the dam's impacts on the river macroinvertebrate assemblages in Ethiopia. The objective of this study was, therefore, to assess the impacts of the Koka hydropower dam on macroinvertebrate assemblages in the Awash River basin in Ethiopia. In the three river reaches on the Awash River (upstream near the source of the river, midstream above the dam, and downstream below the dam), a total of 15 sites were selected for sampling. The statistical analysis tested the null hypothesis that there are no differences in macroinvertebrate assemblage patterns or a range of univariate metrics between the three river reaches. Additional analyses involved the identification of taxa responsible for significant differences in macroinvertebrate structure (*e.g.*, percentage similarity) and an exploration of the variables that structure macroinvertebrates (*e.g.*, canonical correspondence analysis). In the upstream, midstream, and downstream reaches of the Awash River, we recorded a total of 73 taxa belonging to 43 families and 12 orders. Trichoptera was the dominant order in the upstream river reach, whereas Diptera dominated the midstream and downstream river reaches. The diversity of macroinvertebrates decreased from upstream to midstream and downstream. The three river reaches differed significantly in Shannon and Simpson diversity indices, % EPT, EPT taxa abundance, total taxa richness, evenness index, % collectors, and % scrapers. In this study, we observed that macroinvertebrate assemblage differences and spatial patterns were significantly associated with values of river flow changes (velocity), phosphate concentration, and substrate index. The findings of this study have broad implications for the assessment of the impacts of dam construction on the rivers of the studied region in the future.

INTRODUCTION

Dams are structures that restrict water flow and create reservoirs to provide specific human needs (Schmutz and Moog, 2018; Winton *et al.*, 2019) such as easy navigation, lessen flooding, and water supply (hydropower generation and irrigation, and potable water) (Schmutz and Moog, 2018; Zhang and Gu, 2023). Hydropower dams are among the most detrimental anthropologic activities in

Submitted manuscript

Factors Affecting Macroinvertebrate Community Composition in Three River Basins of Ethiopia: A Variation Partitioning Analysis

Abstract

Community ecology is the study of interactions between species in communities at different geographical and temporal scales, including composition, structure, abundance, demography, and interactions between coexisting populations. Several studies examined the effects of environmental and land use individually and together on communities of benthic macroinvertebrate composition. However, there is a lack evidence that evaluated unique and shared effects of environment, land use, and space on the composition of macroinvertebrates in streams. Examining the effects of environmental, land use, and spatial factors on the composition of macroinvertebrate communities was the objective of the current study. We used variation partitioning analysis to evaluate the unique and shared roles of the three sets of predictor variables. In this study, we found that 51% of the variation in the compositions of macroinvertebrates could be attributed to environmental, land use, and spatial factors. The findings of this study demonstrated the shared variation in macroinvertebrate composition due to the three sets of predictors were higher than their unique effects. The result highlights the three sets of predictors were more correlated and that environmental factors are spatially structured and influenced, directly or indirectly, by land use factors, Thus, land use planning could be an important strategy to improve stream water quality, thereby increasing macroinvertebrate composition and enhancing the ecosystem services obtained from freshwater resources.

Keywords: river basin, stream reach, variation partitioning, environment, land use, space

Annex 8.7. Curriculum Vitae

Melaku Getachew Tadesse

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I. Personal data

Name: Melaku Getachew Tadesse

Date of Birth: 24 May 1975

Marital status: Married

Nationality: Ethiopian

Sex: Male

Place of Birth: Jamma Woreda, South Wollo, Ethiopia

Language: Amharic: Mother tongue

English: Proficient

Religion: Orthodox Christian

II. Educational Background

A. High school education

Senior Secondary Education ***1988-1992*** *(Grades 9-11)*

Senior Secondary Education ***1994*** *(Grade 12)*

B. University Education

- Diploma in Sanitary Science **1995-1997** (Jimma institute of health sciences, school of environmental health sciences) (GPA = 3.08)
- BSc in Environmental Health Science 2001-2004 (Jimma university, public health faculty, school of environmental health Sciences) (GPA = 3.16)
- MSc in Environmental Science and Technology 2008-2010 (Jimma university, public health faculty, Department of environmental health sciences) (GPA = 3.96)
- PhD candidate in Addis Ababa University in water and public since 2016.

III. Employment/Experience

February 2012-2022, lecturer at Wollo University

April 2011-2012. Head of the department of water resources development in Oromia Zone, Amhara National Regional State.

June 2010 – April 2011. HIV/AIDS and STI prevention and control **Program officer** in Oromia Zone Health Department in Amhara National Regional State, Kemise.

April 2007-Sep., 2009, Head of Oromia Zone Health Department in Amhara National Regional State, Ethiopia, Kemise.

January 2006-February 2007. Head of Jille Timuga Health Office in Oromia Zone, Amhara National Regional State, Ethiopia, Senbete.

October 2005 –December 2006, Health service **program coordinator** including hygiene and sanitation, MCH services, malaria prevention and control at Jille Timuga Health Office in Oromia Zone, Senbetie.

April 1997- July 2002, Senior environmental hygiene and sanitation expert in Artuma Fursi Woreda Health Office in Oromia Zone, Amhara National Regional State, Ethiopia, Cheffa Robit.

IV. Training

- Epidemiological surveillance and epidemic management at the district level
- Behaviour Change and Communication
- (BCC)in the Context of Essential Nutrition Actions (ENA)Approach
- Health Communication Process and Principles Training
- Ethiopian Contraceptive Logistic System Training
- Democracy and democratic integrity in Ethiopia, rural development, urban development, education, and capacity-building strategy.
- HIV/AIDS District coordinators' course
- Certificate of merit for research from Jimma University.
- Training on Leadership Strategic Information (LSI)

V. Other relevant training

- Diploma in computers Basic training (MS word, excel, PowerPoint, SPSS)
- Certificate of Higher Diploma Program (HDP) in education