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Land Use Land Cover Change and Its Impact on Major Nutrients And Macroinvertebrate Along Wabe-Megecha Rivers Sub-Catchments, Gibe-Omo Catchment, Ethiopia

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Thesis Submitted in Partial Fulfilment for the Requirement for Masters of Science Degree in Aquatic Ecosystems and Environmental Management (AEEM)

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Master of Science Thesis

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Declaration

I, the undersigned, hereby declare that this thesis entitled ‘‘Land Use Land Cover Change and Its Impact on Major Nutrients and Macroinvertebrate along Wabe-Megecha Rivers Sub-Catchments, Gibe-Omo Catchment, Ethiopia’’ is my original work and that all sources of materials used for the thesis have been duly acknowledged.

Name: Abel Feyisa Woldeyohanis

Signature ------------------------------------ Date---------------------------------

Recommendation

I the principal advisor/supervisor of this thesis, hereby certify that I have closely advised/supervised the student while developing this thesis and read the daft thesis entitled "Land Use Land Cover Change and Its Impact on Major Nutrients and Macroinvertebrate along Wabe-Megecha Rivers Sub-Catchments, Gibe-Omo Catchment, Ethiopia'' prepared under my guidance by Abel Feyisa Woldeyohanis. Therefore, I recommend submitting the thesis to the department for further review and evaluation.

Tadesse Fetahi (PhD.) ________________________ ________________

Major advisor/supervisor Signature Date

Approval of board of reviewers

We, the undersigned, members of the Board of Reviewers of the thesis open defence by Abel Feyisa Woldeyohanis have read and evaluated the thesis/dissertation thesis entitled "Land Use Land Cover Change and Its Impact on Major Nutrients and Macroinvertebrate along Wabe-Megecha Rivers Sub-Catchments, Gibe-Omo Catchment Ethiopia" and assessed the understanding of the candidate about the research. This is, therefore, to certify that the thesis is accepted and we recommend the implementation of the thesis.

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List of abbreviations and acronyms

AGR	Agricultural dominated catchment
ANOVA	Analysis of variance
DO	Dissolved Oxygen
EC	Electrical Conductivity
EPT	Ephemeroptera, Plecoptera, and Trichoptera
FAO	Food and Agriculture Organisation
FFGs	Functional Feeding Groups
FR	Forest dominated catchment
CCA	Canonical Correspondence Analysis
SPSS	Statistical Package for social science
POM	Particulate Organic Matter
TDS	Total Dissolved Solids
TSS	Total Suspended Solids
UAGL	The Urban, Agricultural, and Livestock dominated catchment
UAGR	The Urban and Agricultural dominated catchment

Abstract

Rivers are a major aquatic ecosystems that are rich in the biological community and characterized by nutrient dynamics playing a major role in connecting terrestrial and other aquatic ecosystems. These ecosystems provide crucial services including fish production, water provisioning, recreation, water purification, water retention, and climate regulation. However, these life-sustaining ecosystem are deteriorating due to anthropogenic activities including pollution, overexploitation and land use land cover (LULC) change. The latter affects the nutrient contents and biodiversity such as macroinvertebrate community structure, abundance and function. These changes affect the integrity and health of an ecosystem, which is evident in Ethiopia such as the Wabe-Megecha river catchment. Correspondingly, this study aims to investigate the pattern of LULC, the nutrient concentrations (N and P), and macroinvertebrate change with LULC types along the Wabe-Megecha River sub-catchment. The study was conducted at 12 sampling sites of four streams from May to June 2022. The global land use land cover dataset includes a 10m high-resolution Sentinel image for the years 2018 to 2021 was used for the change detection. Water and macroinvertebrate sampling was done for the nutrient analysis, and macroinvertebrate composition, respectively, along the different land uses, from 4 selected sites. The LULC change data was reduced and calculated to the specified study area using Arc-GIS software version 10.4. Statistical tests were done at 0.05 significance levels using SPSS V.26, Past4Project, and Rx64 v. 4.1.2 in R-studio. The result shows that there has been a dynamic of LULC between 2018 and 2021. The expansion of agricultural land and settlement was notable. While the conversion of natural riverine forests to Eucalyptus plantations is stabilizing the forest cover, but degrading the natural environment and thus biodiversity. Most nutrients and physicochemical variables showed significant differences (ANOVA, $p < 0.05$) among the sampled sites. The urban, agriculture and livestock (UAGL) impacted site had significantly ($p < 0.05$) higher nutrient concentrations compared to the other land uses. A total of 6,009 specimens from 9 orders and 29 families were sampled at 12 sites in 4 streams within the Wabe-Megecha catchment. Order Ephemeroptera, Diptera, and Odonata had the highest number of families each representing a total of six families. The highest number of individuals was recorded in Agriculture dominated (AGR) site while the lowest abundance was recorded in the site dominated by the combined effect of urban sewage, agricultural activities, and livestock (UAGL). Order Diptera was the most abundant order among sites which is recoded in the same number in all land uses except Urban, agriculture, and livestock (UAGL) which is highly polluted from all kinds of impacts. Land use change from forest to agriculture was seen to be a major driver of changes in physicochemical water parameters and habitat quality, which also significantly influenced the diversity and distribution of macroinvertebrate taxa. This work highlights the need to conserve forested areas and riparian zones to preserve the ecological integrity and functioning of the Wabe-Megecha River. Thus, efforts are needed to manage the catchment through a well-integrated catchment management for the sustainable utilization of the resources.

Keywords: Land use, satellite image, nutrient, macroinvertebrate, Wabe-Megecha River

1. Introduction

1.1. Background and Justification

Aquatic ecosystems (streams, rivers, lakes, groundwater coastal waters, seas) support the delivery of crucial ecosystem services, such as fish production, water provisioning, and recreation. Water purification, water retention, and climate regulation are also key ecosystem services, which are also connected to the hydrological cycle of the river catchment. These life-sustaining ecosystem services could easily be adversely affected by anthropogenic activities including LULC change, pollution, and overexploitation. Anthropogenic modifications and pollution in aquatic systems have resulted in unanticipated and unprecedented changes in the physical, chemical, and biotic shifts in the system (Cooper, 2013). Agriculture is regarded as the major cause of surface water quality degradation and has attracted growing public concern (Eriksson, 2019). Studies have shown evidence of high levels of nutrients in catchments dominated by human settlement and agriculture (Bodmer *et al*., 2016; Matano *et al*., 2015). The degradation of catchment increases soil erosion, sediment, and nutrients loading into the river channel (Aheimer and Liden, 2000; Carpenter *et al*., 1998; Zhou *et al*., 2019; Njuguna *et al*., 2020) causing eutrophication in many freshwater ecosystems around the world. Thus, LULC change affects the nutrient contents of a river

LULC change also affects biodiversity such as macroinvertebrate community structure, abundance, and function. Because the food webs of benthic invertebrates in the stream and river systems are based on leaf litter inputs from the catchments (such as forests). Thus, changes in the quantity, quality, and seasonality of the inputs have been shown to alter the structure and functioning of stream communities (Wallace *et al*., 1997; Molinero and Pozo, 2004). The effects of anthropogenic LULC change have been acknowledged as one of the most important drivers of the degradation of freshwater ecosystems. Indeed, the physicochemical and biological characteristics of rivers and lakes are strongly correlated with the surrounding LULC types and changes. For example, an increased percentage of agriculture and urbanization cause nutrient enrichment, habitat degradation, and biodiversity loss (Meng *et al*., 2018; Nielsen *et al*., 2012Meier *et al*., 2015; Twardochleb and Olden, 2016).

In tropical areas, LULC change has more severe effects than in the temperate region because tropical organic soils are rapidly mineralized and highly erodible during surface runoff

(Hartemink *et al.*, 2008). Several researchers (Misana *et al*., 2003; Mugisha, 2002; Olson *et al*., 2004) have noted that most LULC changes in Africa are caused by agricultural intensification and extension, population pressure, and resettlement programs. Forests are being cleared in many tropical areas and it is estimated that 38% of East African forest land will be converted into cropland and grazing land by 2035 (Misana *et al*., 2003. These will increase soil erosion and nutrient enrichment from overgrazing and uncontrolled tillage (Semalulu *et al*., 2015) but also affect the leaf litter types and the macroinvertebrate diversity in the system. In Ethiopia, agriculture has intensified and chemical fertilizers and pesticides have increased over the past decade. Similarly, industrialization, urbanization, infrastructure, and house construction have increased at a startling rate since the early 2000s. All these anthropogenic activities, however, produce large volumes of solid and liquid wastes that tremendously affect the water quality of freshwaters upon their introduction into the ecosystem. The problem is common to many Ethiopian river catchments and is severe for rivers nearby towns and cities. Wabe-Megecha sub-catchments a significant tributary river system for the Omo-Gibe river catchment, which is close to Wolkite town. The mountainous landscape of the catchments and its surroundings is undergoing a variety of human-induced changes, including deforestation and the degradation of riparian forests for agricultural purposes, the replacement of natural riverine forests with plantations, and the degradation of rivers due to increased sedimentation brought on by poor agricultural practices and the clearing of riparian forests, as well as urbanization and the untreated discharge of industrial residential effluents into rivers. These activities are known to change the ecological conditions of rivers (Sponseller *et al*., 2001).

Benthic macroinvertebrates are valuable indicators of ecosystem health (Barbour *et al*., 1999) and their function is strongly influenced by the availability of habitats in aquatic ecosystems. However, habitats in streams are threatened by LULC change through activities such as deforestation and agricultural activities that can directly influence stream benthic macroinvertebrates through changes in resource availability, habitat quality, and hydrological alterations (Tanaka and De Santos, 2017). Therefore, understanding the trend of LULC change and its impact on the limiting nutrients cycle and macroinvertebrates is vital in understanding the biogeochemistry and function of a river system (Boulton *et al*., 2008).

Rivers and streams provide many social services and ecological functions (Yeakley *et al*., 2016) including water and food provision for communities, a means of livelihood in developing countries. In Ethiopia, these ecosystems have received less attention, and few authors have studied fish and macroinvertebrate assemblage, diversity, and water quality assessment using macroinvertebrates as a biotic index (Amare Mezgebu *et al*., 2019; Aschalew Lakew and Moog, 2015a, b; Getachew Beneberu *et al*., 2014; Getachew Beneberu and Seyoum Mengistou 2010; Yimer Desta and Seyoum Mengistou, 2009; Seyoum Mengistou, 2006; Baye Sitotaw, 2006). However, a large portion of the country's river systems are not well addressed, and the Wabe-Mgecha sub-catchment is no exception. Furthermore, a few studies assessed the status of water quality of the major Ethiopian lakes and categorized them from eutrophic to hypertrophic conditions, in which rivers were not considered (Tadesse Fetahi, 2019; Yirga Kebede, 2016; Elizabeth Kebede *et al*., 1994; Zinabu Gebremariam *et al*., 2002). Nevertheless, these ecosystems are facing rapid deterioration and pollution primarily related to LULC change that changes river nutrient characteristics and biotic integrity.

Therefore, this study aimed to provide empirical data about the trend of LULC change (forested, urban and agricultural) in the river catchments and its effect on nutrients (N and P concentrations). Physicochemical and nutrients were analyzed to comprehend the habitats. Moreover, the macroinvertebrate assemblage along the course of the Wabe-Megecha sub-catchment of the Omo-Gibe river catchment was addressed. The information generated from this study will be used to develop management strategies for the sustainable utilization of the sub-catchment.

1.2. Statement of the problem

Ethiopia's population is currently one of the fastest-growing in the world, with a growth rate of 3.02% per year. If Ethiopia follows its current rate of growth, its population will double in the next 30 years, hitting 210 million by 2060 (http://esa.un.org/wpp/). Correspondingly, many of the existing forests will be cleared to create land for agriculture (which utilizes primary chemical fertilizers), industry, and settlements to support the fast-growing population. This LULC change affects the water quality of the recipient river systems and the biotic community mainly the macroinvertebrates. Studies indicated that a large part of the forest and wetlands in Ethiopia are converted to agricultural land to boost agricultural production and productivity (Gebissa Yigezu, 2021; Hagerenesh Sankura *et al*., 2014).

Land use land cover change in Ethiopian sub-catchments including Gilgel Abay (Minale Amare, 2013), Gumera sub-catchment (Gashaw Chakilu and Mamaru Moges, 2017; Anteneh Wubie *et al*. 2016), Ribb (Estifanos Abera, 2014; Moges Desalew and Bhat, 2018; Halefom

Afera *et al*., 2019) witnessed for conversion of forest and wetlands to agrarian and other land uses affecting aquatic biodiversity.

Aquatic biodiversity serves a major role in the functioning of streams due to their roles in organic matter processing, facilitating food web interactions, and pollution control. Benthic macroinvertebrates are valuable indicators of ecosystem health due to their benthic and sedentary nature, which renders them unable to migrate away from environmental stress (Barbour *et al*., 1999). Many species with different adaptations enable them to react differently to environmental stressors such as pollution, sediment loading, and habitat changes. Their use as biological indicators is also complemented by their ability to detect the dynamic changes in water quality and thus giving a cumulative effect of environmental stress, unlike the use of physicochemical parameters, which only reflect immediate effects of environmental disturbance (Kibichii *et al*., 2007). These organisms are, however, strongly influenced by the availability of habitats in the aquatic ecosystems which governs their utility as indicators of water quality and ecological health. Habitats in streams are threatened by LULC through activities such as deforestation, and agricultural activities that can directly influence stream benthic macroinvertebrates through changes in resource availability, habitat quality, and hydrological alterations (Tanaka and De Santos, 2017). There is a wide belief that agriculture is the main source of nutrients in river waters in rural catchments and this threatens people's biodiversity, fish, and livelihoods, (Verschuren *et al*., 2002).

LULC change in the Wabe sub-catchment, the present focus of study, is evident due to the traditional agricultural practices upstream, human settlements, and industrial and urban sprawling downstream. Furthermore, major human activities in the sub-catchment are on the rise indicating the level of modification, water pollution, and impact on the macroinvertebrate. Consequently, it is essential to understand how the LULC change influences nutrient concentration, trophic state (water quality index), and the benthic macroinvertebrate assemblages along the river. So far, there is no systematic water quality assessment program in Ethiopia, particularly for the river system. A few studies focus on rivers and they have used macroinvertebrates as water quality indicators (Amare Mezgebu *et al*., 2019; Aschalew Lakew and Moog, 2015a,b; Getachew Beneberu *et al*., 2014; Getachew Beneberu and Seyoum Mengistou, 2010; Yimer Desta and Seyoum Mengistou, 2009; Seyoum Mengistou, 2006; Baye Sitotaw, 2006). The Wabe River sub-catchment has not been investigated, even though the water quality in the sub-catchment is deteriorating due to multiple stressors on the catchments.

Thus, this study aims to investigate the pattern of LULC, the nutrient concentrations (N and P) in the river, and macroinvertebrate change with LULC types along the Wabe River sub-catchment. The study documents baseline knowledge and empirical information for policymakers and the public and will be an input for the sustainable management of the catchment maintaining the integrity of the riverine ecosystem in the study area.

1.3. Objectives

1.3.1. General Objective

- ❖ To assess the impact of land use land cover change on macro-nutrient concentration and macroinvertebrates assemblages in the Wabe-Megecha river sub-catchment, Omo-Gibe basin.

1.3.2. Specific objectives

- ❖ To assess the land use and land cover change of the past three years and identify the driving forces in the Wabe-Megecha river sub-catchment.
- ❖ To quantify the differences in the macro nutrients (N and P) along the current land use and pollution gradient.
- ❖ To determine the macroinvertebrate assemblage along the LULC change and pollution gradients

1.4. Hypotheses

H0: There is no significant difference in LULC, macronutrients, and macroinvertebrate pattern in the Wabe-Megecha rivers sub-catchment

2. Literature review

2.1. Land use influences on stream habitat quality and availability

As integrators of the effects of land use practices within their catchments, streams, and rivers can help in the diagnosis of the environmental health of the landscapes that they drain (Dallas, 2007). Changes anywhere on the landscape that influence rivers are reflected in the composition of the resident biota. Their functional and structural composition varies, both spatially and temporally, to environmental factors (Karr, 1999). These factors include discharge, substrate type, dissolved substances, turbidity, riparian vegetation, land use, temperature, altitude, and latitude. However, human activities influence the effects of these factors, which in turn affect the composition and distribution of macroinvertebrates.

Agricultural activities along rivers and streams have increased the degradation of forests in recent decades (Dallas, 2007). Land use is often associated with disturbances that lead to soil erosion, sedimentation, nutrient enrichment, and the input of toxic substances into aquatic habitats and biological communities (González *et al*., 2001). Riparian zones are very important for the maintenance and regulation of the aquatic environment. Riparian vegetation influences the structure and functioning of stream macroinvertebrate communities through the provision of organic matter and by shading the stream (Allan, 2004). Anthropogenic changes in the riparian corridor may subsequently alter the functional feeding group composition of macroinvertebrates by modifying the supply of food resources and producing changes in habitat structure and quality (Dudgeon, 2006). The presence of riparian vegetation acts as a barrier to sediment input and thus performing a hydrological role and assisting in water quality maintenance.

Forest cover reduces runoff of water, sediments, and nutrients, maintains stable flows, water temperature, and channel morphologies, and supplies coarse organic material and debris to provide food and habitat for aquatic life. Anthropogenic disturbance of the riparian vegetation, on the other hand, increases runoff; destabilizes flow, temperature, and channel morphology, and reduces the supply of coarse organic material (Wang *et al*., 1997). Deforestation can directly influence stream macroinvertebrates through changes in resource availability, habitat quality, and hydrological alterations (Tanaka and Dos Santos, 2017). Hydrological alterations can interact with LULC to determine community dynamics at local scales. LULC s for agriculture has large impacts on rural landscapes, with a strong influence on stream ecosystem functioning, and water quality (Allan, 2004).

Sedimentation in streams and rivers is a function of land use shift from the dominance of riparian vegetation to agriculture and is often accompanied by deterioration in water quality, reduced light penetration, and the filling of interstitial spaces in benthic substrates and integrity of their physical environments (Karr *et al*., 1999; Graf, 2005). Sedimentation or siltation is widely acknowledged as a major cause of degradation of instream habitats (Wood *et al*., 2005) with increased sediment delivery loads that have been widely documented as a major cause of degradation to freshwater ecosystems. Clearing of riparian vegetation for tilling increases the vulnerability to surface runoff and this can lead to high concentrations of nutrients and explosive increases in algal growth in aquatic ecosystems. When natural riparian vegetation is removed for agricultural uses, the water temperature, nutrient concentration, and sediment input tend to increase in streams, causing negative effects on the ecological integrity of aquatic ecosystems (Allan, 2004).

2.2. Nutrient dynamics in Aquatic Ecosystem

2.2.1. Nitrogen

Both inorganic and organic forms of Nitrogen occur in water in the form of nitrates, nitrites, ammonium, and decaying animal and plant matter respectively (Kumar and Puri, 2012). Nitrogen is essential for primary production in aquatic ecosystems.

Nitrate is the form of nitrogen commonly found in natural waters and is the most oxidized and stable form of nitrogen in a water body. It results from the complete oxidation of nitrogen compounds and is the primary form of nitrogen used by plants as a nutrient to stimulate growth. Excessive amounts of nitrates may result in phytoplankton or macrophyte proliferations (Smulac *et al*., 2017). Nitrates get into natural waters through percolation from decaying plant and animal material, domestic sewage, and agricultural fertilizers (Helard *et al*., 2020). In streams and rivers, they originate from domestic effluents, excessive use of agricultural fertilizers, wastewater discharge, overflow of septic tanks, and decay of plants (Kumar and Puri, 2012).

Nitrite is an intermediate in the oxidation of ammonium to nitrate. Nitrites find their way into streams and rivers through wastewater, which is either untreated or partially treated. Many effluents, including sewage, are rich in ammonium, which in turn can lead to increased nitrite concentrations in receiving waters (Helard *et al*., 2020). In surface waters, even at low levels, their presence is an indicator of sewage pollution (Smuleac *et al*., 2017). Nitrite is also toxic to aquatic life at relatively low concentrations. In unpolluted waters, nitrite levels are generally low. Upon entering an aerobic area, nitrites are oxidized to nitrates; however, they are reduced to nitrites by microorganisms and plants. The ion, however, gets quickly oxidized back to nitrate upon re-entering a water body (Kumar and Puri, 2012). This form of nitrogen can be used as a source of nutrients for plants and its presence speeds up plant growth.

Ammonium occurs naturally in water bodies arising from the micro-biological decomposition of nitrogenous compounds in organic matter. Ammonium in rivers is a result of discharges from industrial effluents, wastewater treatment plants, municipal effluent discharges, and excretion of nitrogenous waste by animals such as cows, goats, and sheep among others (Kumar and Puri, 2012). It can also enter rivers through indirect means like air deposition, nitrogen fixation, and runoff from agricultural farms. Ammonium can also arise in waters from the decay of discharged organic waste and is known to exert a demand for oxygen in the water as it is transformed into oxidized forms of nitrogen. It is also an important nitrogenous fertilizer for aquatic plants therefore can cause eutrophication and indirectly reduce the dissolved oxygen due to increased BOD (Zhang *et al*., 2017).

In addition, ammonium is toxic to aquatic life at certain concentrations to salinity, pH, and water temperature (Kir *et al*., 2019). For example, ammonium hydroxide in water is extremely toxic to fish and aquatic life at elevated pH levels. It can also cause river water pollution. Ammonium exists in aqueous solutions in two forms, ionized (NH^+) and un-ionized (NH) with the latter being toxic to freshwater fish at very low concentrations. The relative proportions of ionized and un-ionized ammonium in water depend on temperature and pH and to a lesser extent on salinity. The concentration of un-ionized ammonium becomes greater with increasing temperatures and pH and with decreasing salinity (Kir *et al*., 2019). Total Nitrogen is a measure of all forms of nitrogen (organic and inorganic). The importance of nitrogen in the aquatic environment varies according to the relative amounts of the forms of nitrogen present be it ammonium, nitrite, nitrate, or organic nitrogen (Poikane *et al*., 2021).

2.2.2. Phosphorus

Phosphorus is generally considered to be the limiting nutrient for plant growth in freshwater with small quantities occurring naturally mainly from geological sources (Correll, 1998). As a macronutrient, P plays a significant role in biota growth and development. P in natural waters and wastewater is either in inorganic (including orthophosphates and condensed phosphates) or organic form (organically bound phosphates). Orthophosphate is the most readily available form for uptake during photosynthesis. Total Phosphorus is a measure of both inorganic and organic forms of phosphorus. The common forms of phosphorus commonly found in the river and stream water are soluble reactive phosphorus (SRP), organic-bound phosphorus, and particle-bound phosphorus. According to Dzombak and Sheldon (2020), P comes from weathering rocks and chemical fertilizers and detergents. It enters surface water sources through runoff of manure, industrial effluents, domestic waste waters containing detergents, agricultural fertilizers, human and animal wastes, organic waste in sewage, and sludge. Bank erosion also adds P in stream waters. It is also a significant pollutant in flowing waters when in excess supply (Khan *et al*., 2010). When in high levels, P causes eutrophication and increases BOD thereby reducing dissolved oxygen (Kumar and Puri, 2012). To avoid these adverse effects on water quality, P concentrations in natural waters should be less than 0.03 mg/L (WHO, 2017).

2.3. Effects of different land use types on nutrients: N and P Concentrations

Many streams and rivers of the world have been strongly impacted by anthropogenic activities (Smith *et al*., 1999) which has seen nutrient concentrations increase in most of them (Alan, 2004). Different kinds of land use impact aquatic ecosystems differently (WWAP, 2017). Forest, urban, and agricultural land uses affect nutrient concentrations within the water column and stream sediments in different ways. Land use and land cover types can either transform nutrients or bar them from moving toward streams and rivers as dissolved or suspended nutrients. Human activities in the catchment can alter the water chemistry of streams and rivers through nutrient and pollution addition. When primary production, sediment deposition, infiltration rates, and biogeochemical processes are altered by land management practices, the stream water chemistry is also altered (Kilonzo *et al*., 2014; Maloney and Weller, 2011). This causes water quality, nutrient cycling, and ecological functioning of streams to be affected.

According to Onyando *et al*. (2013), nutrient concentrations in rivers are also influenced by the land use and land cover in the riparian areas hence catchment managers should focus on that crucial area. Williams *et al*. (2014) and Zampella *et al*. (2007) linked land use to total nitrogen (TN) and total Phosphorus (TP) using regression models and found a strong significant relationship between the nutrient concentrations and land use. A study by Aheimer and Liden (2000) argued that most interaction of land use and stream water quality occurs during the wet season and thus a study undertaken during this season provides sufficient information. They also found out that agriculture has got a strong influence on nitrogen concentrations. However, variations in climatic conditions should be taken into consideration when studying whole nutrient dynamics in the stream and river systems unless the study is focussing on a specific season, otherwise, the results may be misleading (Tasdighi *et al*., 2017).

FAO (2013) reported that with the shift from conventional to intensify agriculture, many chemicals are being used to boost food production for the increasing human population thus affecting the health of most aquatic ecosystems through excessive nutrient input. A study by Jones *et al*. (2019) in the Cerrado biome, Brazil confirmed that SRP concentrations are high in tropical streams that have been impacted by agricultural activities. There is a positive relationship between total nitrogen and continuous cropping (Wilson and Xenopoulos, 2009). In agricultural lands, some the chemicals like herbicides, pesticides, and fertilizers are washed into the streams in surface runoff during floods and this increases the nutrient loads, particularly nitrogen and phosphorus compounds. The use of livestock manure to boost crop production accelerates nitrates concentrations when they are washed into rivers by surface runoff during rains (Jones *et al*., 2019).

Clearing of the forests to pave way for agricultural crop farming causes surface runoff to increase and this accelerates sediment delivery into streams. Erosion of riverbanks also increases thus more sediments enter the stream increasing phosphorus concentrations and downstream sediment loadings (Khatri and Tyagi, 2015). Clearing of catchments exposes streams and waters to direct sunlight thus causing temperatures to increase and chemical composition to change. The solubility of gases like Oxygen and Carbon dioxide is reduced hence reducing the concentrations held by the system and this further impacts processes like decomposition and metabolism (Khatri and Tyagi, 2015).

Changes in land use can either increase stream nutrient concentrations through inflows or reduce them due to dilution (Neill *et al.*, 2011). There is more nutrient delivery in rivers passing through agricultural areas compared to those passing through forested areas as reported by Bodmer *et al*. (2016) which is comparable to Carpenter *et al*. (1998) results. FAO (2013) carried out a survey and observed that the effective root network and leaf litter in forested areas controls soil erosion as well as filtering terrestrial pollutants and reducing surface runoff. This causes retention time to lengthen thus allowing nitrogen and phosphorus uptake, breakdown, and utilization (Löfgren, 2009). In areas with dense riparian vegetation, the vegetation helps to stabilize banks thus preventing erosion (Clary and Kinney, 2002) as well as trapping overland runoff with contaminants and pollutants during rains hence preventing excessive sedimentation (Enanga *et al*., 2011).

The vegetation filters nutrients like nitrogen, phosphorus, and carbon from the inflowing water (Enanga *et al*., 2011) through sedimentation and bio uptake. This zone can be altered by human activities like agriculture, wastewater discharge from sewers and runoff, deforestation, and urbanization thus affecting the nutrient concentration and water quality (Shivoga *et al.*, 2007).

2.4. Benthic macroinvertebrates as indicators of stream systems' integrity

Benthic macroinvertebrate communities have been popularized in their bio-indicative role in streams as they are sensitive to environmental disturbance, integrating as well as reflecting the effects of stress, both natural and human- induced, over extended periods (Barbour *et al*., 1999). The distribution, composition, and abundance of these groups of organisms have an indicative role in the changing ecological conditions that alter ecosystem functioning and degrade water quality and overall ecological integrity. Benthic macroinvertebrate communities have been long used as bio-indicators of stream conditions and environmental impacts due to the capacity of these macroinvertebrate communities to exhibit clear responses to human disturbances (Buss *et al*., 2002) and present a large diversity of traits to these conditions as well as coping up with differential resource availability. Most of these organisms exist in a narrow range of hydraulic and physical conditions and thus their capacity to act as ecosystem surrogates was limited if their habitats interfered with. Gore (1978) documents large shifts in benthic invertebrate abundance, composition, and distribution associated with small changes in their habitats. With these arguments, benthic macroinvertebrates have been established to be good indicators of assessing the effects of environmental stressors including the LULC change on aquatic ecosystems. This has been achieved through careful analysis of biological and ecological responses along gradients of human disturbance, the identification of indicator assemblages and species among assemblages with known responses to human alterations, and the identification of driving variables acting on aquatic ecosystems (Dubois *et al*., 2017).

2.5. Adaptations of benthic macroinvertebrates to environmental conditions

Benthic macroinvertebrates in streams and rivers are either sessile, move around actively, or are passively moved around by current (Tanaka and Dos Santos, 2017). Stream metrics, biotic and abiotic factors, water quality, and the riparian environment do affect the diversity, richness, and distribution of macroinvertebrates communities (Dallas, 2007). Stream habitats are often associated with specific flow velocities. Differences within meso-habitat communities are determined by flow fluctuations producing physical changes and by the type of substrate present which often produces seasonal shifts in species composition (Townsend, 1997).

River flows (floods, pulses, and base-flows) serve different functions in molding the available physical habitat and therefore dictate the life history stages of riverine organisms (Bunn and Arningthon, 2002). Habitat preferences of mayflies can be based on their feeding strategies during their larva stages while grazers and scrapers prefer feeding on materials attached to biofilms. Shredders and gatherers on the other hand occur in substrates containing decomposing coarse and fine particulate organic matter (Masese *et al*., 2014).

Pardo and Armitage (1996) found collector-gathering organisms burrowing in sediments under low flow conditions while streamside vegetation with silted beds was inhabited by burrowers and sediment-surface organisms, collector gatherers, and shredders. Dallas (2007) also noted that in South African rivers, stone biotopes had a higher number of taxa relative to vegetation and sand samples.

This suggests that changes in the availability of different biotopes might influence community structure and informs the need for this study which will focus on investigating the community structure of macroinvertebrate communities with changing habitats along land use, altitude, and longitudinal gradient.

Based on this objective, the study seeks to contribute to the discussion on the trend and impact of land use dynamics on nutrient concentration and macroinvertebrate community in the river system. The study also presents cases of human influence on habitat availability and suitability for some specific macroinvertebrate taxa. This can be incorporated into ecological explanations of prevailing water quality conditions in riverine ecosystems.

3. Methodology

3.1. Description of the study area

The Wabe-Megecha river catchment is the sub-catchment of the Omo-Gibe River basin mainly in the Gurage highlands. It is located between $08^{\circ}10'00''$ and $08^{\circ}30'00''$ N, and $38^{\circ}20'00''$ and $37^{\circ}40'00''$ E at altitudes from 1,014 to 3611 m above sea level (Figure *1*Figure *1*). The highest altitudinal area of the Wabe-Megecha River catchment is located on the Mount Zebider side, while the lowest altitude is located at the fringe of Gibe Sheleko (Valley) National Park.

The catchment's maximum temperature has ranged from $20^{o}C$ (during the wet season) to $39^{o}C$ (during the dry season), while the minimum temperature has ranged from $0^{o}C$ to $19^{o}C$ with average annual temperature of $18^{o}C$. The average annual precipitation is between 1200 and 1320 mm (NMA, 2016). The predominant type of soil is public vertisols (Mesfin Sahle and Kumelachew Yeshitela, 2018). Sampling sites are located strategically in Wabe and

Megcha River sub-catchment accompanied by distinct land use types. It is the main tributary of Ethiopia's Omo-Gibe River Catchment, the largest supply of freshwater for Kenya's saline Lake Turkana.

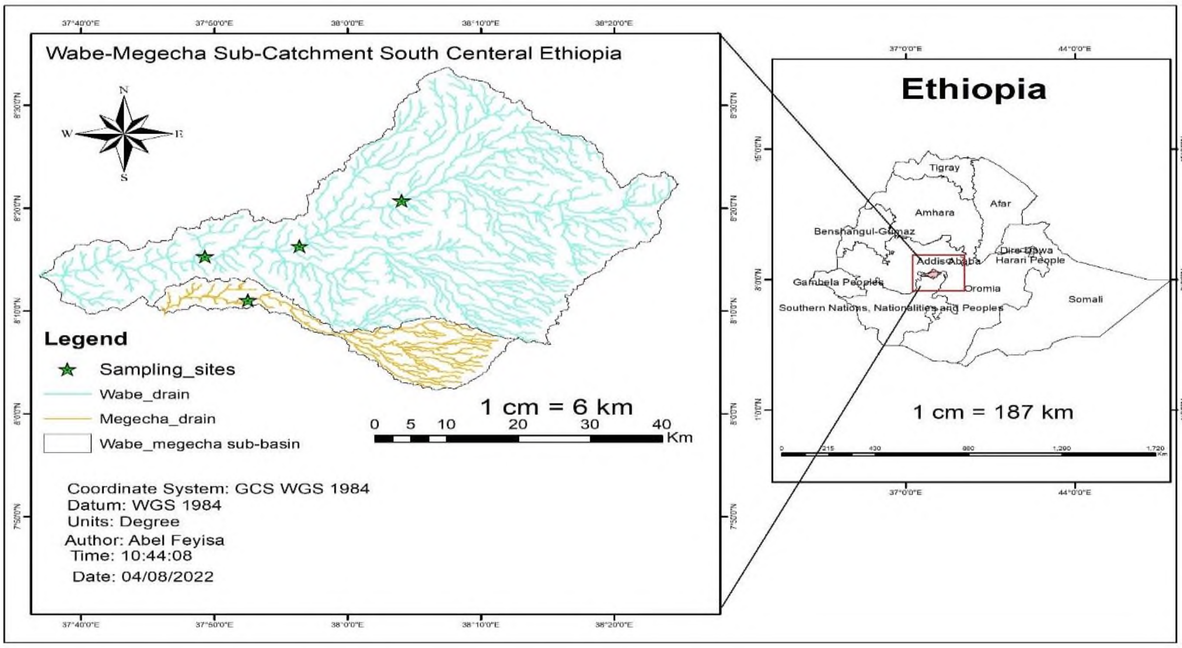

Figure 1: Map of the study area and sampling sites

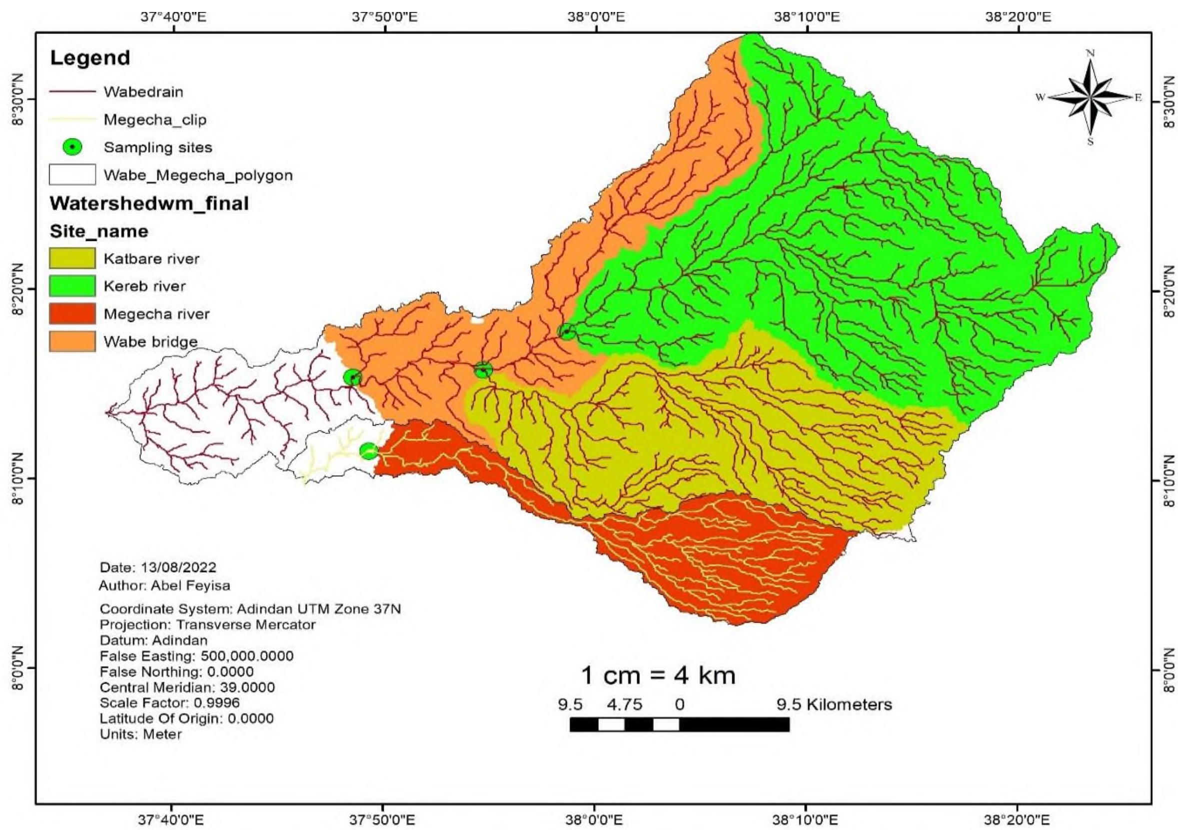

Figure 2: Classification of the 2021 land use with based on major land use land cover classes.

Four sampling sites representing different anthropogenic activities were selected as shown in Table 1.

Table 1: Sites, coordinates, altitude and dominant substrate types of the study area.

Site Name	Latitude	Longitude	Elevation (masl)	Dominant substrate
Kereb river site	8.23845	38.07312	2231	85% Macrolithal
Katbare river site	8.24565	37.79311	1685	65% Macrolithal
Wabe bridge site	8.24764	37.78711	1664	70% Macrolithal
Megecha River Site	8.18955	37.79827	1887	55% Microlithal

Site one: Kereb river forest (Reference Site)

The Kereb riverine forest having several streams, was represented by Krb is located in the upper sub catchment of the Wabe River and was used as the reference site. These sites are relatively less disturbed and characterised by undisturbed flow channel. The local population use river water for domestic use. Moreover, the nearby Abune Zena Markos and Mhur Eyesus monasteries of Ethiopian Orthodox Tewahdo Church use the streams as a source holy water to treat illness (Fig.3).


Figure 3: Kereb river forest (Reference Site)

Site Two: Katbare river site

The middle sub-catchment of the river called the Katbare site was represented by Ktb which is mainly characterized by intensive agriculture and livestock watering and this was used as the second sampling site. In response to the food demand of the growing human population, deforestation and micro-irrigation expansion were observed (Fig. 4).

Figure 4: Katbare river site

Site three: Wabe bridge site

The third sampling site is characterized by the impact of urbanization and mining activities taken place. It is in the lower part of the Wabe river (Fig.5).

Figure 5: Wabe bridge site

Site four: Megecha River Site (Most polluted site)

The fourth site is found in Gubre town site where the impact of Urbanization is high. The industrial sewage from Tinaw flower farm, sewage from Wolkite University, the nearby market, car wash, livestock watering, and other impacts. This site was considered to be the most polluted.

Figure 6: Megecha River Site

3.2. Method

3.2.1. The Trend of Land Use Land Cover change

The Digital elevation model (DEM) which is used for catchment delineation was downloaded from (https://earthexplorer.usgs.gov/). The Wabe-Megecha catchment was delineated using DEM in ArcGIS 10.4 software. The Sentinel-2 10m land cover time series of the world was downloaded from the global land use land cover dataset created by Impact Observatory, Microsoft, and Esri (https://livingatlas.arcgis.com/landcover/), which is the land cover time series of the entire planet for the years 2017 to 2021. This layer shows a 10m-resolution global map of land use/land cover (LULC) obtained from ESA Sentinel-2 images. Each year, the National Geographic Society develops a vast training dataset of billions of human-labelled image pixels for Impact Observatory's deep-learning AI land classification model. Applying this model to the Sentinel-2 scene collection on Microsoft's Planetary Computer, which processes over 400,000 Earth observations annually, produced worldwide maps (Karra K. 2021).

This study opted to use the more comprehensive and high resolutions dataset since the years 2018–2021 are based on a more full set of imagery. Global land use and land cover maps include details on, among other things, food security, hydrologic modelling, and conservation planning. Anywhere on Earth, this dataset can be used to see how much land is used or covered (Karra K. 2021).

Table 2: Description of the identified LULC in the Wabe-Megecha River catchment (Source: Modified from Karra K. 2021)

LULC class	Description
Water	Areas where water was predominantly present throughout the year; may not cover areas with sporadic or ephemeral water; contains little to no sparse vegetation, no rock outcrop nor built-up features like docks; examples: rivers, ponds, lakes.
Trees/Natural and plantation forest areas	Any significant clustering of tall (~15 feet or higher) dense vegetation, typically with a closed or dense canopy; examples: wooded vegetation, clusters of dense tall vegetation within savannas, plantations, swamp or dense/tall vegetation with ephemeral water, or canopy too thick to detect water underneath).
Crops/Agriculture	Humans planted/plotted cereals, grasses, and crops not at tree height; examples: Enset, corn, wheat, soy, and fallow plots of structured land.
Built Area (Settlement)	Human-made structures; major road and rail networks; large homogenous impervious surfaces including parking structures, office buildings, and residential housing; examples: houses, dense villages/towns/cities, paved roads, and asphalt.
Bare land	Areas of rock or soil with very sparse to no vegetation for the entire year; large areas of sand and deserts with no to little vegetation; examples: exposed rock or soil, desert, and dunes, dry salt flats/pans, dried lake beds, mines.

Accuracy assessment

An accuracy assessment was carried out to what extent the produced classification is compatible with what exists on the ground (Congalton, 1991). The reference data (latitude and longitude), was collected for each LULC type of 2022 classified images from corresponding Google earth images. Accordingly, a total of 300 validation points were taken for six land use land cover classes (50 validation points for each sampling area). Consequently, the classified images were compared with reference images by using error matrices.

Table 3: Accuracy assessment matrix for the classified images of 2018 (a), 2019 (b), 2020 (c), and 2021 (d).

a)

LULC	Water	Forest	Agriculture	Settlement	Bare land	Grazing land	UA
Water	46	0	0	0	0	0	0.92
Forest	0	44	0	0	0	0	0.88
Agriculture	0	2	40	0	2	0	0.91
Settlement	0	0	0	48	0	0	0.96
Bare land	4	0	4	2	48	3	0.79
Grazing land	0	4	6	0	0	47	0.82
PA	0.92	0.88	0.8	0.96	0.96	0.94	0.91

b)

LULC	Water	Forest	Agriculture	Settlement	Bare land	Grazing land	UA
Water	46	1	0	0	0	0	0.98
Forest	0	44	0	1	0	0	0.98
Agriculture	0	2	40	0	6	3	0.78
Settlement	0	0	0	46	0	0	1.00
Bare land	4	0	4	3	42	2	0.76
Grazing land	0	3	6	0	2	45	0.80
PA	0.92	0.88	0.8	0.92	0.84	0.9	0.88

c)

LULC	Water	Forest	Agriculture	Settlement	Bare land	Grazing land	UA
Water	49	0	0	0	0	0	0.98
Forest	1	50	0	0	0	0	0.98
Agriculture	0	0	48	0	1	3	0.92
Settlement	0	0	0	48	0	0	0.96
Bare land	0	0	1	2	49	0	0.94

Grazing land	0	0	1	0	0	47	0.98
PA	0.98	1	0.96	0.96	0.98	0.94	0.97

d)

LULC	Water	Forest	Agriculture	Settlement	Bare land	Grazing land	UA
Water	49	0	0	0	0	0	0.98
Forest	0	48	0	0	0	0	0.96
Agriculture	0	1	47	0	1	0	0.96
Settlement	0	0	0	48	0	0	1.00
Bare land	1	0	1	2	49	2	0.89
Grazing land	0	1	2	0	0	48	0.94
PA	0.98	0.96	0.94	0.96	0.98	0.96	0.96

Over all accuracy = 91% Kappa coefficient= 0.94.

Over all accuracy = 87% Kappa coefficient = 0.84.

Over all accuracy = 97% Kappa coefficient = 0.92.

Over all accuracy = 96.3% Kappa coefficient = 0.93.

Image classification

The Global land use land cover map of the study years was imported into ArcGIS (version 10.4) for processing. The LULC maps were produced for the years 2018, 2019, 2020, and 2021. Thus, the final year data was used to examine the impact of LULC on water quality parameters and macroinvertebrate assemblage.

LULC change detection

The clipped and classified images of the years were compared from two periods, 2018–2019, and 2020-2021 to detect the annual rate of land use land cover change in the study area. Change statistics were computed by comparing the image values of one data set to the corresponding value of the second data set for each period. The change detection technique was performed using a tabular area of zonal statistics tools in ArcGIS 10.4 software. A cross-tabulation analysis was conducted to determine the quantitative conversions from a particular category to another land cover category and their corresponding area over the evaluated period on a pixel-by-pixel basis. To map the persistence, losses, and gains of each LULC, the Land Use Change Modular (LCM) extension to the ArcGIS model was used.

$Percentage \frac{LU}{LC} Change = \frac{Area\ of\ final\ year - Area\ of\ initial\ year}{Area\ of\ intial\ year}$ ... ...Equation 1: Area calculation for the catchment LULCC

Positive values indicate an increase whereas negative values show a decrease in the extent of LULC.

Questionnaire survey

To understand the underlining causes of the catchment change, a face-to-face interview with the 48 local elders was conducted (12 from each sampling site). Additionally, the literature and unpublished reports from the adjacent Woredas were referred to and used as secondary data sources.

3.2.2. Nutrient dynamics (N and P)

A total of 12 water samples, 3 per site, for nutrients analysis was collected on a fortnight basis between May to June 2022. This is considered to be the dry-wet interface season in the study area where the infiltration capacity of the soil ends and the small runoff to the rivers starts.

Physicochemical and nutrient analysis

Physicochemical parameters such as temperature, pH, dissolved oxygen, total dissolved solids, and conductivity were measured in situ at all the sites using a HACH 40d Multi-meter probe (HQ40D). Turbidity was measured using a turbidity meter (HACH-2100Q Multi-meter probe).

River Morphometry

Stream width, depth, and discharge were measured using a tape measure, and floating method respectively. This was done once at every site during a sampling session to calculate the stream discharge using equation 2. Floating method: Measure the time of a floating object (e.g. Oranges, cubes) to travel a certain distance; travel time should be at least 20 sec; $V_{surface} = L / t$; Where L is the length of the distance and t is the time. Correct for V mean as follows (the correction has to be made because the floating method only measures the surface velocity which is higher than mean velocity): V mean = $V_{surface} * 0.85$.

Discharge (Q) m^3/s = Cross sectional area (m^2) × Mean velocity (m/s) ……….....Equation 2: Stream discharge

Where cross-sectional area = stream width × stream depth

Water samples for nutrients analysis were collected in triplicates using 500 ml acid-washed (10% H_2SO_4) plastic bottles, and preserved in cooler boxes to minimize biological activity (APHA, 2005). The sample was then transported to Seyoum Mengistou Limnology laboratory of Addis Ababa University for analysis.

Laboratory analysis

Water samples were filtered (except for TP, and TN) upon arrival in the laboratory using Whatman GF/F filters of 7 µm pore size with a 47 mm diameter, and dried at 95°C for 24 hours. Standard calibration curves were prepared for all the nutrients.

Determination of different forms of Nitrogen

Ammonium-Nitrogen was determined using the hypochlorite method by adding 2.5 ml of sodium salicylate solution to 25 ml of the sample followed immediately by 2.5 ml of hypochlorite solution to act as a catalyst. The sample was then placed in a water bath at 25 °C in the dark for 90 minutes. Absorbance was then read using a spectrophotometer at a wavelength of 665 nm. Nitrite-Nitrogen was determined using the N-Naphthyl method by adding 1 ml of Sulfanilamide solution to 25 ml of the filtered water sample. After 2-8 minutes 1 ml of N-Naphthyl-(1)-the ethylendiamine-dihydrochloride solution was added to this mixture and gently mixed. The solution that left standing for 10 minutes after which absorbance was read from the spectrophotometer at a wavelength of 543 nm.

Nitrate-Nitrogen was determined using the Sodium-salicylate method (APHA, 2005) with standard solutions of nitrate prepared for the standard calibration curve. A filtered water sample of 20ml was placed in an evaporation bottle and 1 ml of sodium salicylate solution was added. The bottles were put in the oven and samples were dried at a temperature of 95 °C. The resulting residue was dissolved by adding 1 ml of conc. H_2SO_4 then the bottles were swirled carefully while still warm before adding 40 ml of distilled water and mixing. To the treated sample, 7 ml of potassium- sodium hydroxide-tartrate solution (prepared by dissolving 400 g NaOH in 1 liter distilled water and adding 50g K-Na-Tartarate) was added, mixed and absorbance read at a wavelength of 420 nm.

Figure 7: Determination of NO_3

Total nitrogen (TN) was determined through per sulphate digestion by adding 1 ml of warm potassium per sulphate to 25 ml of unfiltered water sample to convert the nitrogen forms into nitrates. The mixture was then autoclaved for 90 minutes at 120 ºC and 1.2 atm. After digestion, the reduced nitrogen forms into nitrate was analysed using sodium- the salicylate method. Concentrations of NH_4-N, NO_2-N, NO_3-N, and TN were calculated and their respective equations were generated from the standard calibration curves (APHA, 2005).

Determination of different forms of phosphorus

Soluble reactive phosphorus (SRP) was determined using the ascorbic acid method. Ammonium molybdate solution, concentrated sulphuric acid, ascorbic acid, and potassium- Antimonyltartarate solution was mixed in ratios 2:5:2:1 and the resulting solution was added to 25ml filtered water sample in a ratio of 1:10. The prepared samples absorbance was read using a spectrophotometer after 15 minutes of adding reagents at a wavelength of 885 nm. Total phosphorus (TP) was determined by first digesting and reducing the forms of phosphorus present in the water into free orthophosphate using per sulphate digestion. After digestion, the total reduced forms were analyzed using the same procedure as for SRP. 12 g of potassium per-sulphate was dissolved in 100 ml of distilled water and 1 ml was added to the 25 ml sample while still warm. The bottles were weighed without lids and their weights were noted. The covers were put back but not closed tightly after which it was autoclaved for 90 minutes at about 120°C and 1.2 atm. After cooling, the bottles were re- weighed and the evaporated water was replaced by the addition of distilled water. TP was analyzed as SRP using the ascorbic acid method. Concentrations of SRP and TP were then calculated against their prepared standard curves (APHA, 2005).

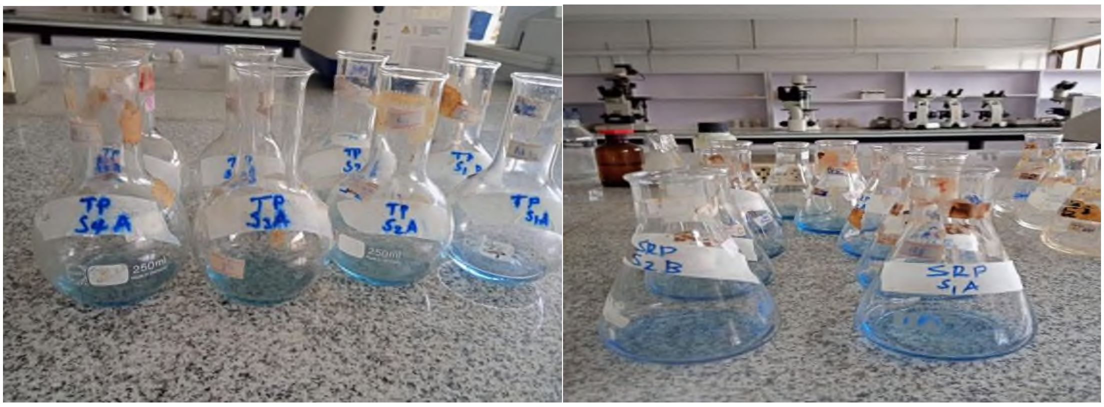

Figure 8: Determination of TP and SRP

Nutrient loading rates

Nutrient loading rates were calculated using equation 3 (Kitaka, 2000):

Nutrient loading=Discharge × nutrient concentration × 0.0864(3)…….Equation 3: Nutrient loading rate

Where nutrient loading (kg/day), discharge (L/S), and 0.0864 is the concentration time conversion factor from mg/s to kg/day.

3.2.3. Macroinvertebrate assessment

Sampling period

The frequency and time of benthic macroinvertebrate sampling depend on climatic conditions and the objective of the study. The variability of discharge volume in rivers, because of seasonal changes, affects the distribution pattern of benthic macroinvertebrates, and the rainy season limit accessibility to sampling sites. It is usually recommended to avoid sampling during high water levels and shortly after flooding if the study is assessing the impact of anthropogenic activities. In the present study, monthly sampling was conducted during the dry-wet interface season from May to the end of June 2022.

Field sampling

Benthic macroinvertebrate samples were collected using a D-frame hand net with a frame width of 30 × 25 cm, 55cm net depth, and a mesh size of 500 µm. The multi-habitat sampling scheme (MHS) was implemented to include major habitat types in pools, riffles, vegetation, and bare rocks (Aschalew Lakew and Moog, 2015). From each sampling site, 20 samples, referred to here as sampling units, were collected with at least a 5% share of each major habitat type. The samples collected from each sampling unit were then pooled into one container to represent that particular site, which was then preserved in 4% formaldehyde for further identification in Seyoum Mengistou Limnology laboratory of Addis Ababa University.

Similarity

Similarity Indices (S_S): Similarity indices measure the degree to which the species compositions of the different system are alike. Many measures exist for the assessment of similarity or dissimilarity between vegetation samples or quadrats. The Sorensen similarity coefficient is applied to qualitative data and is widely used because it gives more weight to the species that are common to the samples rather than to those that only occur in either sample (M. Kent and P. Coker, 1992). The Sorensen coefficient of similarity (S_S) is given by the following formula

$$Ss = \frac{2a}{2a+b+c}$$

Where Ss; is the Sorensen similarity coefficient, a; is the number of species common to all sites, b is the number of species in site 1, c is the number of species in site 2.

Macroinvertebrate identification

In the laboratory, a composite macroinvertebrate sample collected from each site was passed through a set of sieves (2000, 1000, 750, 500, and 250 µm mesh size) to separate macroinvertebrates from debris and to wash away the preservatives using tap water. The sample was placed in a white tray and the taxa were sorted with the naked eye. Organisms trapped in the smaller fraction of the sieve were sorted with the help of a light microscope. Subsampling was applied according to Barbour *et al*. (1999) and identifications were performed using the Aquatic Invertebrates of South African Rivers field guide (Gerber, 2002).

Figure 9: Macroinvertebrate identification

3.2.3. Statistical analysis

The benthic macroinvertebrate indices of diversity, abundance, composition, equitability, and dominance were determined by enumeration, a number of taxa, and the total number of individuals using various indices (Shannon-Weaver diversity index, Simpson's diversity index, and Margalef richness index). Composition measures like %EPT were calculated as:

$\% EPT = [(E + P + T) * 100]/N$………………….Equation 4: %EPT

Where, E = the number of Ephemeroptera, P =the number of Plecoptera, T= the number of Trichoptera, and N=Total number of individuals in a station.

Water quality parameters were analyzed by using descriptive statistics and presented as mean, and Standard Error (mean ± SE). Pearson correlation coefficient was carried out to determine the relationships between each water quality parameter. One-way analysis of variance (ANOVA) was used to evaluate the differences in the means of water quality parameters and abundance of macroinvertebrates across different land use types at a 95% level of significance and followed by Post hoc Turkey's honest significance differences (HSD) tests. Canonical correspondence analysis (CCA) was also applied to evaluate the relationship between the benthic macroinvertebrate community and Physico-chemical water parameters. Meanwhile, the abundance data of macroinvertebrates were transformed using log (x+1) before Canonical correspondence analysis was carried out to check the statistical normality. Microsoft Excel® and PAST v4.02 (Hammer *et al*., 2001) software were also used for the analysis.

The distribution of benthic macroinvertebrate taxa with the sampled environmental variables was analyzed using a principal correspondence analysis (DCA), followed by redundancy analysis (RDA). The results of DCA determined whether linear or unimodal ordination is recommended indicating a linear species response (Leps and Smilauer 2003). The RDA was run with log (x + 1) transformation of macroinvertebrate data and without a forward selection of environmental variables. The redundant physicochemical parameters that have a high inflection factor and multi-collinearity with other variables were removed for further analysis in RDA. The non-parametric Kruskal-Wallis independent sampling test was performed, followed by pairwise comparisons to realize the significant difference between each sampling site. All statistical analyses were carried out in the statistical software packages SPSS V.26, Past4project, and Rx64 v. 4.1.2 in R-studio.

4. Results and Discussion

4.1. Accuracy assessment of the classified LULC maps

*The overall accuracy of the classified images from 2018, 2019, 2020, and 2021 were 91%, 87%, 97%, and 96.3% respectively (*Table 3: Accuracy assessment matrix for the classified images of 2018 (a), 2019 (b), 2020 (c), and 2021 (d).

). The kappa coefficient, which shows a high level of agreement, was 0.94 in 2018, 0.84 in 2019, 0.92 in 2020, and 0.93 in 2021. Producer's accuracy (PA) of the individual classes of the four classified maps ranged from 80% (Agriculture in 2018, and 2019) to 100% (Forest in 2019), respectively. The user’s accuracy (UA) was lowest for the Bare land (76%) in 2019 and the highest for the Settlement classification (100%) in 2021. where UA = user's accuracy and PA = producer's accuracy. The columns represent the actual location of samples on the ground, while rows display classified data showing the location of samples in the classified images. The diagonal numbers shown in bold are the correct classifications. The off-diagonal numbers in rows and columns are misclassifications or errors. The settlement class had a 100% producer accuracy because before an accuracy assessment was conducted, the area was masked and replaced by correctly classified shape files.

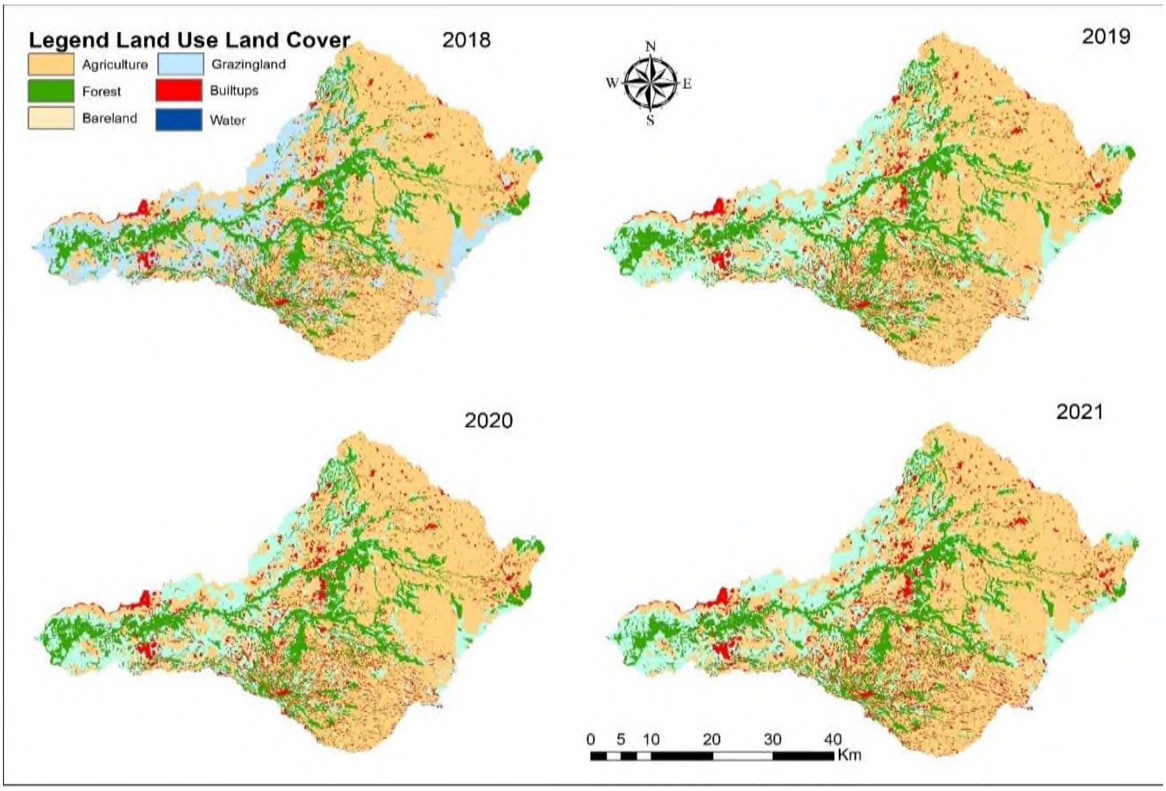

Figure 10: Spatial distribution of LULC during four periods in the Wabe-Megecha River catchment.

4.2. States of land use land cover

The coverage of the LULC for the four consecutive year periods is presented in Table 5 and the spatial distribution is in Fig. 11. The LULC of 2018 that was obtained from the global land cover map shows that forest covered 17.5% of the total study area. The settlement and agricultural land occupied about 4.90% and 52.97%, respectively. Only 111 ha (0.05%) of the catchment area was occupied by settlement. Only 6.45% of the catchment was urbanized in 2019, forests covered 17.34%, agriculture 52.28%, and bare land 0.20%. Between 2018 and 2019 the area cover of the forest, Agriculture, and grazing lands was reduced in the catchment, whereas the area cover of the other land uses increased within the catchment.

In 2020, the area covered by agriculture accounted for 117,638 ha (51.9%). Forest and bare land covered 41,286 ha (18.21%) and 106 ha (0.05%), respectively. The settlement area covered 16896 ha (7.45%), water 140 ha (0.06%), and grazing land 50,644 ha (22.34%).

In 2021, the area covered by Agriculture accounted for 122,514 ha (54.04%). Forest and bare land covered 38,766 ha (18.21%) and 101 ha (0.04%) respectively. The settlement and grazing land area covered 17,870 ha (7.89%), and water coverage didn’t show a very slight fluctuation with the previous year's 135 ha (0.06%).

4.3. LULC change for the period 2018-2019

The LULC conversion for the years 2018, 2019, 2020, and 2021 is depicted in Table 4: Land use/land cover classes and percentage., and **Error! Reference source not found.**.

4.3.1. Forests

The forest cover of the study area had shown a gradual decline during the study periods (2019 - 2021) except 2019 - 2020. In 2018, the forest cover was 39,745 ha (17.53%) of the catchment and decreased to 39,323 ha (17.34%) in 2019. A large percent of forest coverage decreased between 2020 and 2021 than between 2018 and 2019. A slight (5%, 1965 ha) increment in forest coverage was witnessed from the year 2019 to 2020. In general, for the whole study period, there was a fluctuating annual rate of forest coverage. This change was occurred due to losses and gains. The major losses were due to the conversion of forest cover to grazing land (8%) and agricultural land (4%) in 2018. The dynamic was more than the period between 2020 and 2021. From the field observation, the natural forest of the area is being converted to Eucalyptus plantations and small-scale irrigation fields. The extent of remnant forest covers exists in the riverine section of the catchment and in the lower catchment of the Wabe-Megecha River that is under the protection of Gibe Sheleko National Park. The conversion rate of the forest cover was triggered due to the human-induced fire for the conversion of the forest into commercial tree plantation. The conversion in the last four years looks to fluctuate due to the gain and loss of forest.

4.3.2. Grazing land

Grazing land decreased from 55,522 ha (24.49 %) in 2018 to 53,663 ha (23.67 %) in 2019. Moreover, it decreased from 50,644 ha (22.34 %) in 2020 to 47,324 (20.87 %) in 2019. The observed decrease during the first period (2018 - 2019) was due to the conversion of 1859 ha (3.35%) of grazing land to agricultural and bare lands. Ninety percent of the grazing land area remained unconverted between 2018 and 2019. Between 2020 and 2021, 6.56 % of grazing land was converted to Agricultural land. The grazing land area gained was from the conversion of forests.

4.3.3. Agricultural land

This land use is the dominant LULC in the catchment and occupied about 120,083 ha (52.97%) in 2018, 118,530 ha (52.28 %) in 2019, 117,638 ha (51.90 %) in 2020, and 122,514 ha (54.04%) in 2021. There was a slight decrement in agricultural land from the years 2018 to 2020 perhaps due to settlement expansion. However, the agricultural land coverage has increased. It took the decrement mainly from the aggravating expansion of settlements (built-up areas) at the expense of Agricultural areas.

4.3.4. Settlement (Built area)

The 2018 LULC analysis shows that 11,120 ha (4.90 %) of the catchment was used by built-up areas within the Wabe-Megecha catchment. The area increased to 14,616 ha (6.45 %) in 2018 and 17,870 ha (7.89 %) in 2021. This indicated that settlement areas have been expanding over the last 4 years. Encompassing grazing and agricultural fields.

4.3.5. Bare land

The area coverage of the bare land was 129 ha (0.06 %) of the total area in 2018 and 454 ha (0.20 %) in 2019, showing a significant increase. In 2020, the area coverage of bare land decreased to 106 ha (0.05 %), and 101 ha (0.04%). This is because most of the area was converted to Agricultural land. Some part of the bare land was converted to grazing, shrub, and forest land. However, still, 0.04% of the catchment is bare land and most of the area is degraded which is mainly attributed to the mining activities in the catchment.

4.3.6. Water bodies

As the topography of the Wabe-Megecha River catchment consists of steep terrain, the probability of finding a large area of natural lakes, and wetlands was rear. Small-sized lakes and wetlands can however be found in a scattered manner within the catchment, but it was difficult to detect. This study only detects a few small lakes, riverine water, and small wetlands. The total water coverage of the water bodies was 450 ha (0.2 %) in 2018, 445 ha (0.2%) in 2019, and 345 ha (0.06%) in 2021. The lake exists in the northeast part of the catchment. The area coverage of water bodies decreased to 450 ha in 2018 and 345 ha in 2021. Which is mainly due to the conversion of water bodies to settlement and agriculture.

In brief, the resulting decrease in the size of forests, grazing land, and water bodies have been decreasing and the increase in agricultural land area of the Wabe-Megecha River catchment which was confirmed through the interview, and group discussion of experts. This result is in agreement with the results of other studies conducted in Ethiopia (Mesfin and Kumelachew, 2018; Miheretu and Yimer, 2017; Wubie *et al*., 2016; Kindu *et al*., 2013; Shiferaw, 2011; Dessie and Kleman, 2007; Tegene, 2002; Zeleke and Hurni, 2001; Tekle and Hedlund, 2000).

Table 4: Land use/land cover classes and percentage.

	2018		2019		2020		2021	
LU Type	Area	%	Area	%	Area	%	Area	%
Forest	39,745	17.53	39,321	17.34	41,286	18.21	38,766	17.09
Water	450	0.20	445	0.20	440	0.194	345	0.06

Agriculture	120,083	52.97	118,530	52.28	117,638	51.9	122,514	54.04
Settlement	11,120	**4.90**	14,616	6.45	16,896	7.45	17,870	7.89
Bare land	129	0.06	454	0.20	106	0.05	101	0.04
Grazing land	55,522	24.49	53,663	23.67	50,644	22.34	47,324	20.87

Table 5: Land use/land cover change during (2018 -2019, 2019-2020, 2020-2021)

	2018-2019		2019-2020		2020-2021	
LU Type	Area	%	Area	%	Area	%
Forest	-424	-1.07	1965	5.00	-2520	-6.10
Water	-5	-1.11	-5	-1.12	-95	-21.59
Agriculture	-1553	-1.29	-892	-0.75	4876	4.14
Built ups	3496	31.44	2280	15.60	974	5.76
Bare land	325	251.94	-348	-76.65	-5	-4.72
Grazing land	-1859	-3.35	-3019	-5.63	-3320	-6.56

NB: Positive values indicate an increase whereas negative values show a decrease in the extent of LULC.

Table 6: Land use matrix for the study years (A; 2018-2019, B; 2020-2021, C; 2018-2021)

A		**Land class 2019**						
LU Type		Waterbodies	Forest	Agriculture	Settlement	Bare land	Grazing land	Grand total
Land class 2018	Waterbodies	106.24	8.17	0.91	0.01	0	3.32	118.64
	Forest	6.50	36071.59	796.35	326.86	27.04	2515.69	39744.03
	Agriculture	2.05	1168.00	109229.91	2342.04	181.51	7154.43	120077.94
	Settlement	0.00	48.64	523.88	10281.11	57.64	207.30	11118.56
	Bare land	0.00	2.94	22.78	45.18	29.89	13.01	113.80
	Grazing land	10.72	2020.64	7952.27	1619.17	151.97	43762.44	55517.22
	Grand total	125.51	39319.98	118526.10	14614.36	448.06	53656.19	226690.20

B		Land class 2021						
LU Type		**Waterbodies**	**Forest**	**Agriculture**	**Settlement**	**Bare land**	**Grazing Land**	**Grand total**
Land class 2020	**Waterbodies**	118.45	3.83	0.36	0.01	0.00	2.86	125.51
	Forest	10.74	37025.46	779.94	208.39	1.89	1292.98	39319.39
	Agriculture	2.26	1174.27	108524.95	2317.87	34.51	6471.00	118524.86
	Settlement	0.00	156.37	1029.66	13115.37	8.17	305.35	14614.92
	Bare land	0.00	31.54	166.44	145.38	32.92	71.77	448.05
	Grazing land	8.97	2893.19	7133.61	1106.84	19.01	42496.32	53657.94
	Grand total	140.43	41284.66	117634.96	16893.85	96.50	50640.27	226690.67

C		Land class 2021						
LU Type		Waterbodies	Forest	Agriculture	Settlement	Bare land	Grazing land	Grand Total
Land class 2018	Waterbodies	107.67	8.11	1.20	0.00	0.00	1.66	118.64
	Forest	9.92	34669.8	1435.87	543.43	0.47	3084.42	39743.92
	Agriculture	3.98	1606.83	107386.52	4908.89	3.94	6166.34	120076.49
	Settlement	0.01	66.49	819.18	9925.05	0.22	307.60	11118.55
	Bare land	0.00	2.48	26.25	73.17	0.09	11.81	113.79
	Grazing land	13.27	2411.46	12880.76	2445.63	5.14	37747.41	55503.67
	Grand total	134.85	38765.17	122549.78	17896.17	9.86	47319.22	226675.05

4.3.7. Underlining Drivers of LULC changes in the Wabe-Megecha River Catchment

Among the 48 local elders asked about the major causes of the land use change, 83% of the respondents agreed that the expansion of agricultural practices (which was justified that it is to satisfy the need and aspirations of the growing population), expansion of settlement at the expense of forest land. The demand for construction materials were also among the major causes of the land use change in the study catchment. The respondents consider however, the livestock population grazing pressure on the pasture lands, prescribed fire, and illegal firewood collection in the riverine areas of the forest were mentioned as the least causes of the land use change. From the field observation perspective, mining, conversion of natural vegetation including the riverine forest to mining, eucalyptus plantation, and small-scale irrigation areas) has been seen as a significant challenge for land use in the study area. Similar findings were reported by Zerga (2016) in the Ezha district of the Gurage zone which is part of the Wabe-Megecha River catchment. On the other hand, the Green legacy initiative has helped the restoration practices of degraded lands in Wabe-Megecha river catchment.

Figure 11: Nursery seedlings prepared for green legacy at Wabe-Bridge site (Photo: Solomon A. 2022)

Similar studies from the Wabe-river catchment identified the possible direct and indirect drivers to change the LULC. The possible indirect drivers are demographic changes, economic activities, poverty, and policy, and institutional changes. While improper agricultural practices, climate variability, and change and demand for firewood and construction materials are the main direct drivers (Mesfin and Kumelachew, 2018).

4.4. Macronutrients along the land use and pollution gradients of the Wabe-Megecha catchment

4.4.1. Relationship between selected physicochemical parameters

From the linear regression analysis, there was a negative relationship between DO and temperature (R^2 = 0.45, F = 28.02, $p < 0.05$) and a perfect positive relationship between conductivity and TDS, (R^2 =0.999, $p < 0.05$) (Figure 4a and 4b). A significant positive relationship was also observed between TSS and turbidity (R^2 = 0.48, F = 31.828, $p < 0.05$) (Figure 4c). This means that temperature accounts for only 45% of variations in DO. Nearly 100% of the changes in conductivity are attributed to TDS and 48% of the variations in turbidity are caused by TSS in the Wabe-Megecha river catchment.

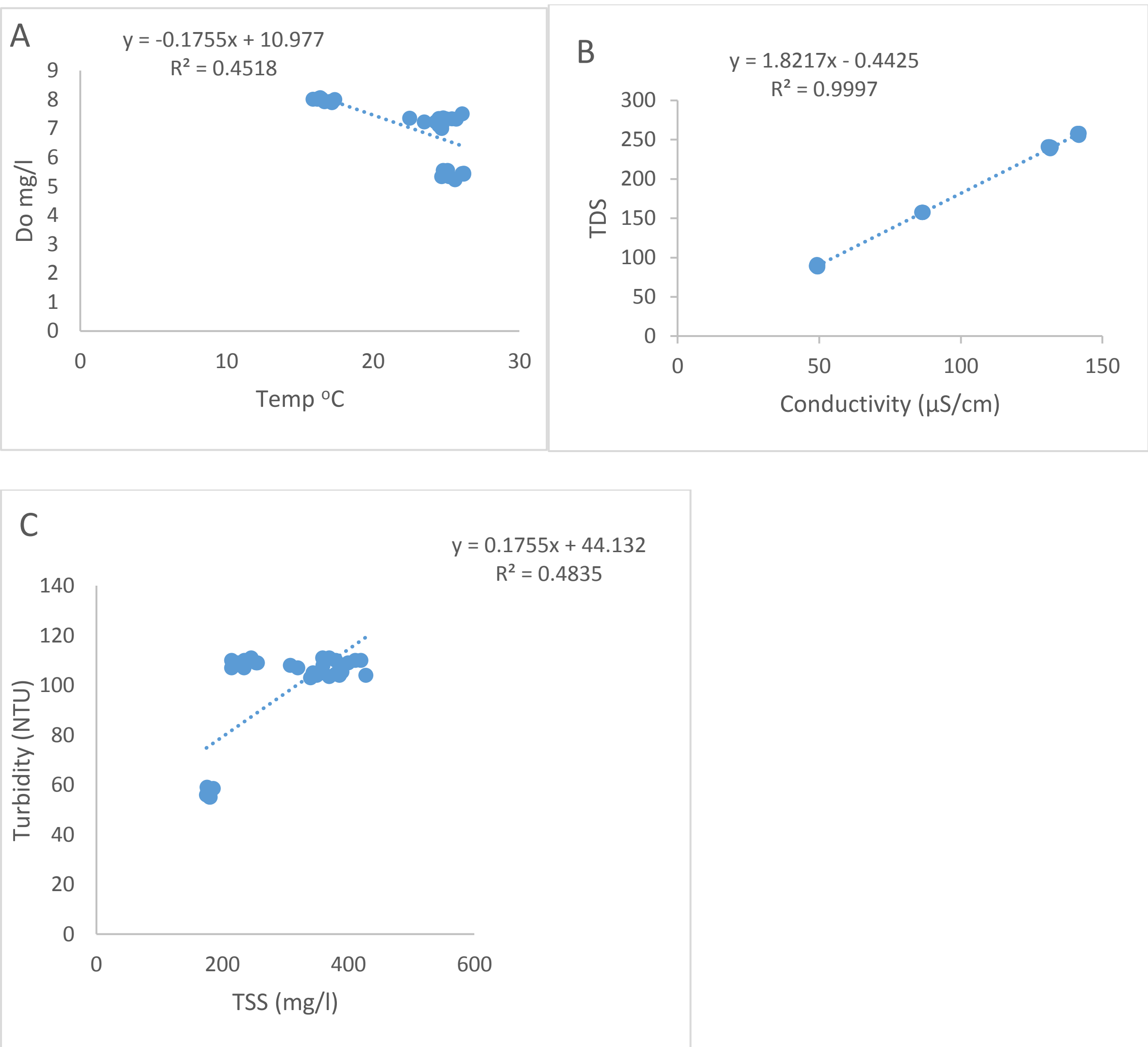

Figure 12: Relationship between a) DO concentration and Temperature, b) TDS and Conductivity C) Turbidity and TDS

4.4.2. Interactions between nutrients and selected parameters

The regression analysis was performed to predict NO_3 from temperature and DO (ANOVA, $F = 18.531$, $p < 0.05$). The results showed that the DO was statistically significant in predicting changes in NO_3 while temperature was not. Temperature accounts for 39% (R^2=0.392) of the variations in NO_3 while DO accounts only for 9% (R^2 = 0.097) of variations in NO_3 concentrations (**Error! Reference source not found.**).

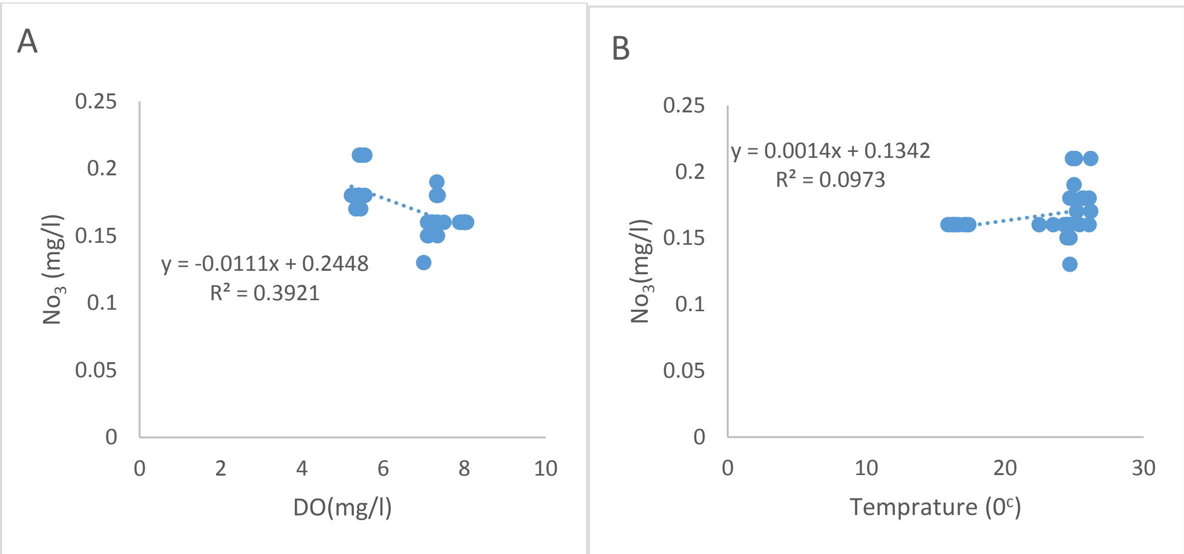

Figure 13: Relationship between a) NO_3 concentration and DO and b) NO_3

Figure 15 shows that TSS had a positive relationship with both TN and TP though not significant. The findings gave an R^2 of 0.39 (Figure 14a) which means that 31% of variations in TN were attributed to TSS in the catchment. The R^2 for TP was 0.33 (Figure 15) which means TSS accounts for 33% of TP variations.

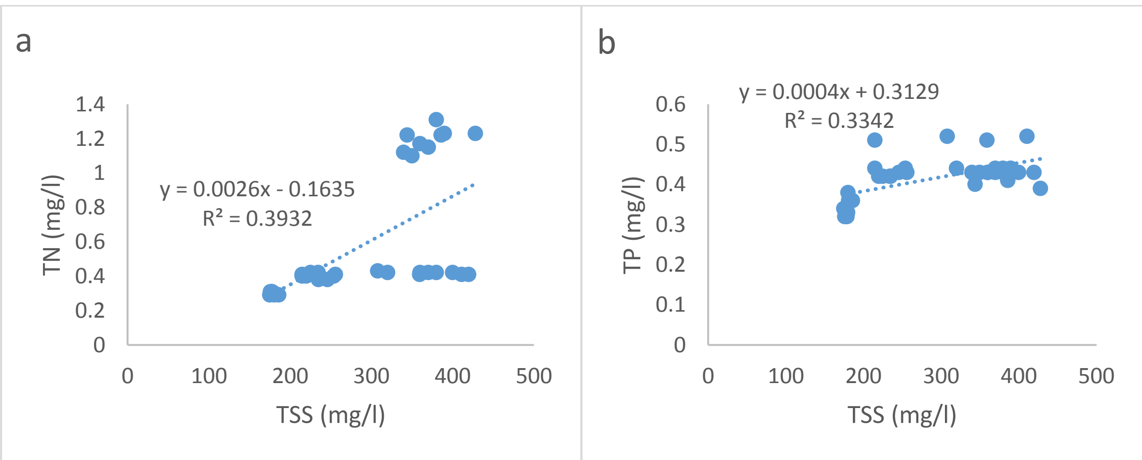

Figure 14: Regression analysis showing the relationship between a) TN concentrations and b) TP concentrations and TSS.

4.4.3. Variation of physicochemical parameters among the four land use types

There were marked variations in physicochemical parameters among the four land use types (FR, AGR, UAGR, and UAGL) (ANOVA, $p < 0.05$) (Table 5).

Changes in in-situ parameters along the land-use gradient in Wabe-Megecha River catchment

There were significant differences in temperature among the four land use types (ANOVA, $p < 0.05$). It was significantly lower in the forested than in the others land use types (AGR, UAGR, and UAGL sections) (Tukey HSD, $p < 0.05$). The AGR land use section was significantly lower than the forest and non-significantly higher than the rest of the land uses i.e. UAGR, UAGL section. The temperature was highest in the downstream AGR section (25.17 ± 0.47) > UAGR section (24.63 ± 0.12) > UAGL section (21.17 ± 0.06 °C) > FR (16.70 ± 0.50).

There were no significant differences in DO among the four land use types (ANOVA, $p > 0.05$). DO concentration was highest in the FR section (7.94 ± 0.06 mg/L) followed by (7.34 ± 0.02), while the lowest DO was recorded in UAGL dominated site (5.48 ± 0.07 mg/L) (Table 4). pH ranged between 8.56 to 8.63 at the FR reach, and 8.74 to 8.78 at the AGR reach. There were no significant differences in pH values among the four land use types (ANOVA, $p > 0.05$) (Table 4). Conductivity was the highest at the UAGR section (257.33 ± 0.58), followed by AGR (239.33 ± 0.58). There were significant differences among the land use types (ANOVA, $p < 0.05$) with that the FR reach being significantly lower than the other reaches (Tukey HSD, $p < 0.05$).

TDS was highest in the UAGR-dominated section (141.54 ± 0.22), followed by the AGR section (131.49 ± 0.35), and was lowest in the FR-dominated section 49.35 ± 0.18). The highest turbidity (109.67 ± 1.53 NTU) was recorded in the UAGR-dominated sites. While the lowest (56.57 ± 1.70 NTU) was recorded at the FR-dominated sites (Table 4). There were significant differences in the turbidity among the four land use types (ANOVA, $p < 0.05$) with the values of the FR reach being significantly lower than the rest of the reaches (Tukey HSD, $p < 0.05$).

Table 7: Variation of in-situ measurements in WMRSC (Mean ± SD), and range of pH

Sampling sites	T^o	DO	%saturation	Conductivity	pH range	Turbidity

FR	16.70±0.50	7.94±0.06	131.50±45.81	89.70±0.40	8.56-8.63	56.57±1.70
AGR	25.17±0.47	7.34±0.02	106.07±4.04	239.33±0.58	8.74-8.78	108.67±1.53
UAGR	24.63±0.12	7.13±0.04	104.33±0.15	257.33±0.58	8.97-9.02	109.67±1.53
UAGL	21.17±0.06	5.48±0.07	77.47±2.02	157.23±0.06	8.62-8.70	104.33±0.58

Where, FR, Forested sites; AGR, Agricultural sites; UAGR, Urban, and Agricultural impacted sites; UAGL, Urban, agricultural, and livestock impacted sites.

TSS concentrations varied significantly among the land use types (ANOVA, $p < 0.05$) with the

FR reach has significantly lower concentrations than AGR, UAGR, and UAGL reaches (Tukey HSD, $p < 0.05$). TSS was lowest in the FR reach (179.17 ± 3.28 mg/L) and highest in the UAGL section (372.00 ± 27.75 mg/L) as shown in Figure 16.

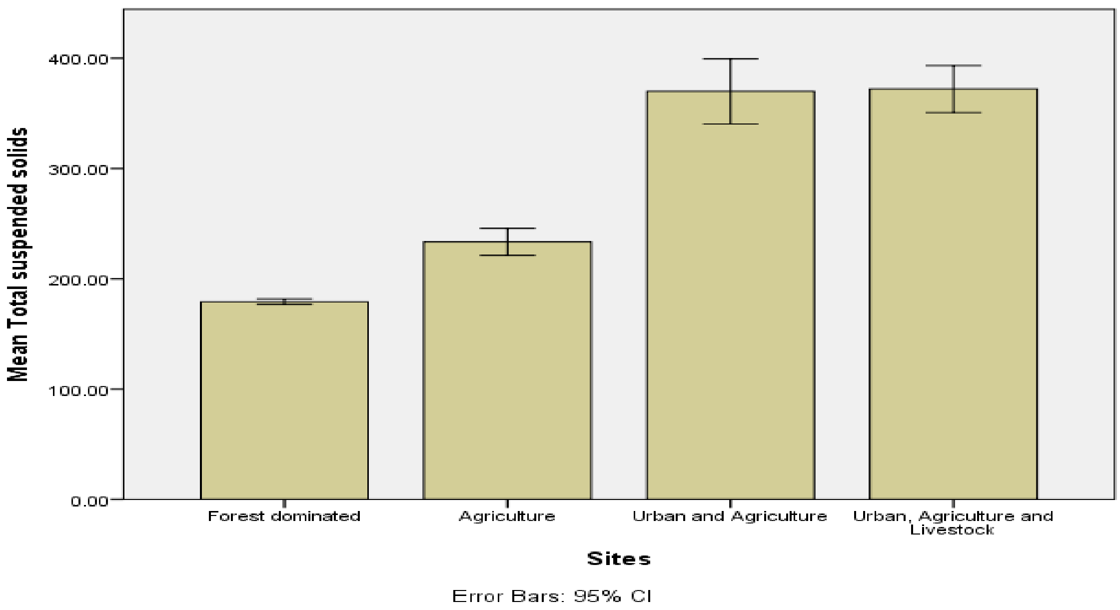

Figure 15: Variations in TSS concentrations in FR, AGR, UAGR, and UAGL

4.4.4. Discharge variation among the four land use types

Figure 17 presents stream discharge values which were lowest in the FR-dominated river reach (0.1567 ± 0.12 m^3/s) < the UAGL section (0.67 ± 0.41 m^3/s) < the AGR section (1.95 ± 1.03 m^3/s) < the UAGR section (6.53 ± 2.17 m^3/s). There were significant variations among the four land use types (ANOVA, $p < 0.05$) with the FR reach having a significantly lower discharge compared to the other reaches (Tukey HSD, $p<0.05$).

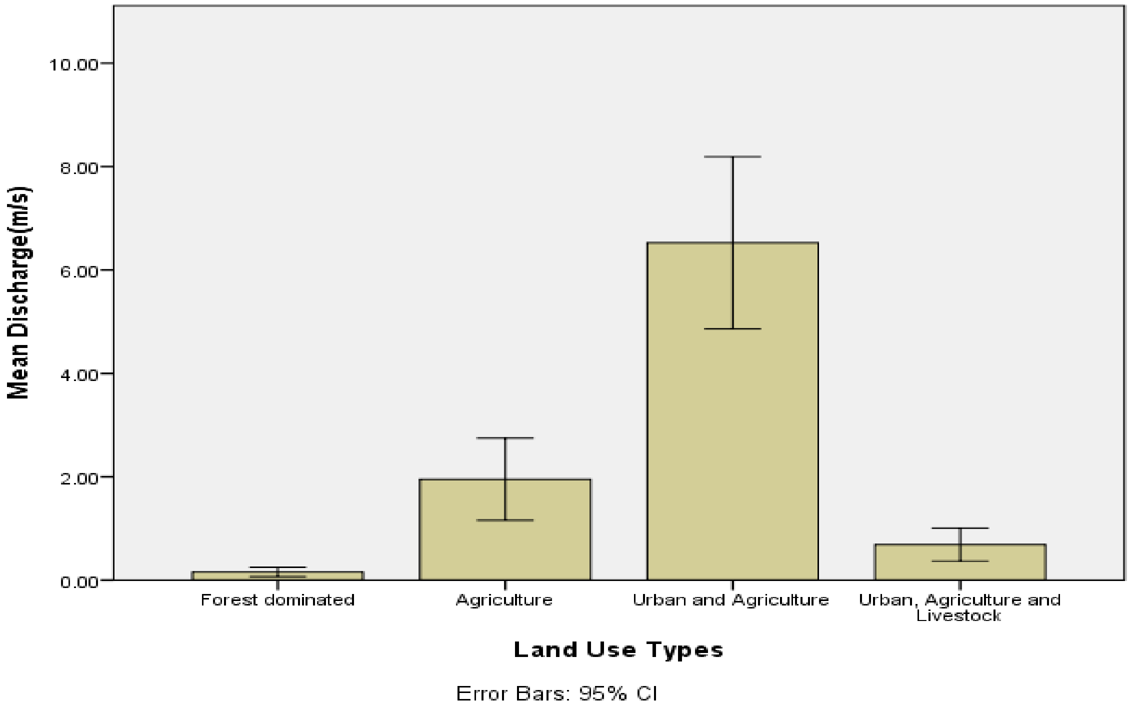

Figure 16: Variation of discharge among the FR, AGR, UAGR, and UAGL river sections of the Wabe-Megecha River sub catchment.

Variation of discharge among the FR, AGR, UAGR, and UAGL river sections of the Wabe-Megecha River sub catchment.

4.4.5. Variation of nutrient concentrations among the land use types

Ammonia-Nitrogen concentration ranged between 0.116 ± 0. 00394 mg/L in the FR and 0.966 ± 0.062 mg/L in the UAGL reach (Figure 17a). The AGR had 0.21 ± 0.012 mg/L. Nitrites was lowest in the FR (0.025 ± 0.00) and highest in the UAGL 0.037 mg/L ± 0.00 mg/L) (Figure 17b). There were significant differences in nitrite-nitrogen concentrations among the four land use types (ANOVA, $p < 0.05$). The post hoc test revealed that both ammonia and NO_2 concentration in the FR differed significantly from the values obtained in the other land use types (Tukey HSD, $p < 0.05$).

Nitrate-Nitrogen was highest in the UAGL section (0.19 mg/L ± 0.018 mg/L) followed by the AGR (0.17 mg/L ± 0.014 mg/L) and lowest in the FR section (0.16 mg/L ± 0.001 mg/L) (Figure 20c). There were significant differences in nitrates concentration among the four land use types (ANOVA, $p < 0.05$), and the post hoc test showed that NO_3 concentration in the FR and UAGR was significantly lower than that of the AGR and UAGL (Tukey HSD, $p < 0.05$).

TN concentrations ranged between 0.30 mg/L ± 0.01 mg/L in the FR section and 1.19 ± 0.07 mg/L in the UAGL section. The AGR section had a concentration of 0.40 mg/L ± 0.02 mg/L

(Figure 17d). Except for AGR and UAGR, there were significant variations in TN concentrations among the four land use types (Tukey HSD, $p < 0.05$).

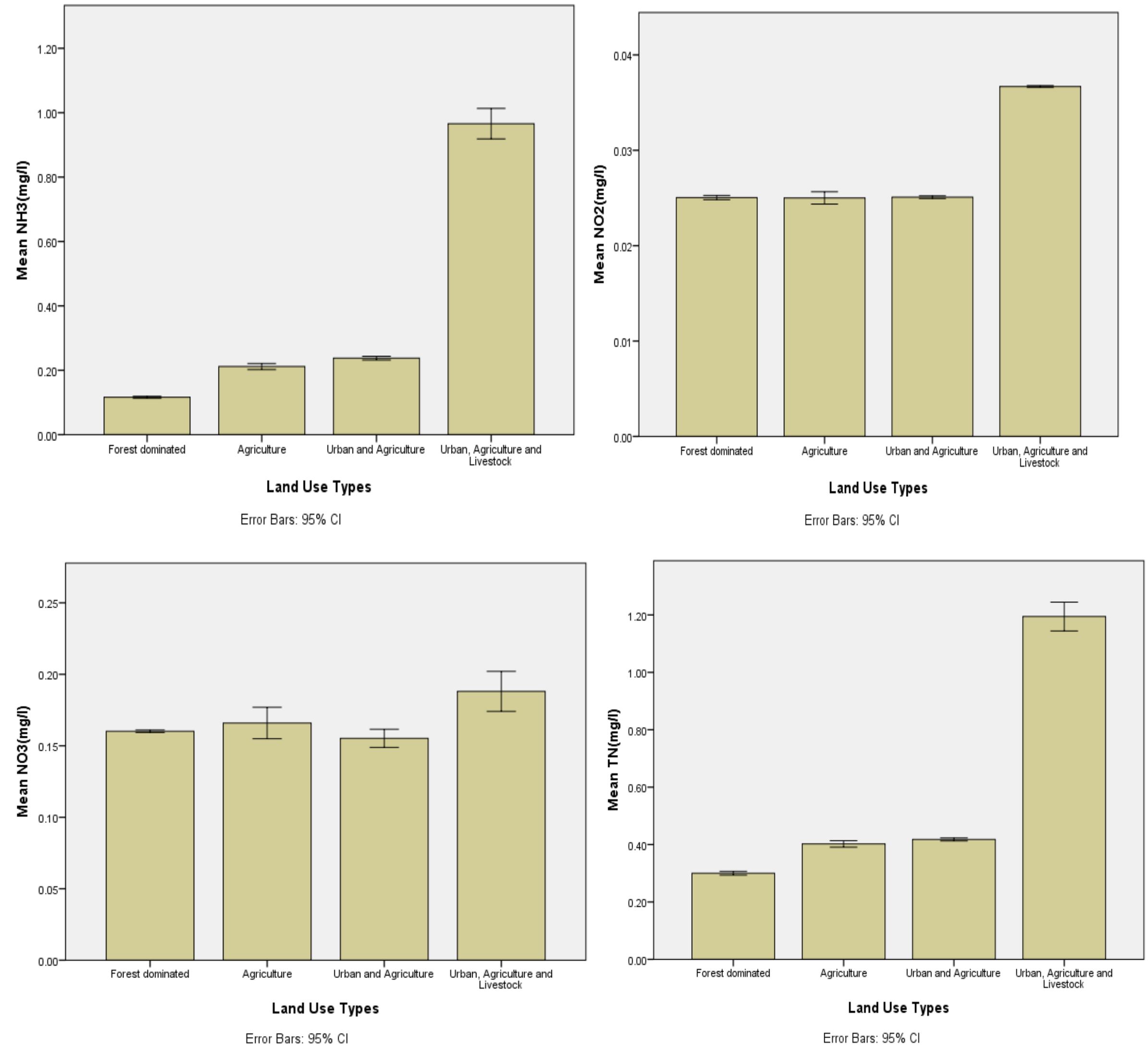

Figure 17: Variation of mean ± SD concentrations of a) Ammonia, b) Nitrites, c) Nitrates, and d) TN in FR, AGR, UAGR, and UAGRL reaches of Wabe-Megecha River.

SRP was lowest in the FR section (0.32 ± 0.007 mg/L) followed by the UAGL (0.42 ± 0.02 mg/L) and highest in the AGR and UAGR section (0.43 ± 0.05 mg/L, and 0.43 ± 0.02 mg/L respectively). There were significant differences in SRP concentration among the four land use types. This is mainly attributed to the AGR and UAGR uses chemical fertilizer and other effluents from the catchment.

TP ranged between 0.34 ± 0.02 mg/L in the FR section and 0.46 ± 0.04 mg/L in the UAGR section. The UAGL and AGR sections had a concentration of 0.42 ± 0.02 mg/L and 0.44 ± 0.03 mg/L respectively (Figure 18). TP concentrations varied significantly among the four land use types (ANOVA, $p < 0.05$). The Post hoc test showed that TP concentration in the FR was significantly lower than in the rest of the land uses (Tukey HSD, $p < 0.05$). While the AGR was not significantly varied with UAGR, and UAGL (Tukey HSD, $p > 0.05$).

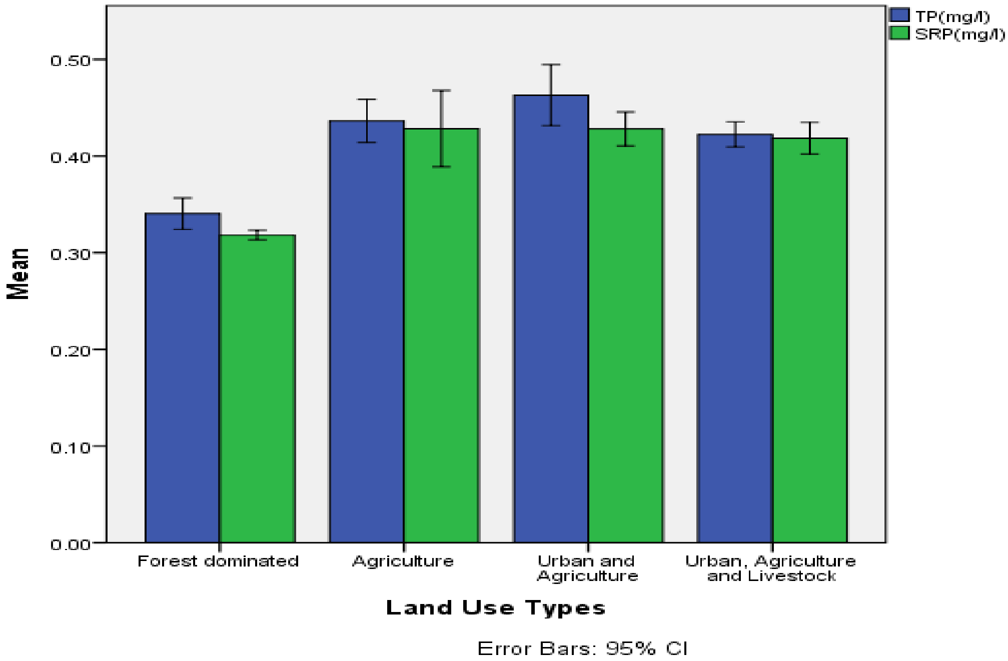

Figure 18: Variation of mean ± SD concentrations of TP and SRP.

4.4.6. Relationship between nutrients concentrations in the four land use types

The PCA analysis (Figure 23) was done using a correlation matrix developed using PAST software version 4.03. In these results, two components were extracted which explained the variation in nutrient concentration in the four land use types. For example, the first and second components explained 67.7% and 30.5% of the variability of the parameters respectively. Nitrites, NO_3, TN, and NH_3 have large negative loadings on component 2 meaning they are strongly correlated. This means the sources of NO_2, NO_3, TN, and NH_3 can be attributed to the UAGL reaches of the land use. TP and SRP had a large positive loading on component 1 meaning that as AGR, and UAGR activities increase, TP and SRP also increase. Therefore, the variations of SRP and NH4 can be attributed to AGR and UAGR activities in the catchment.

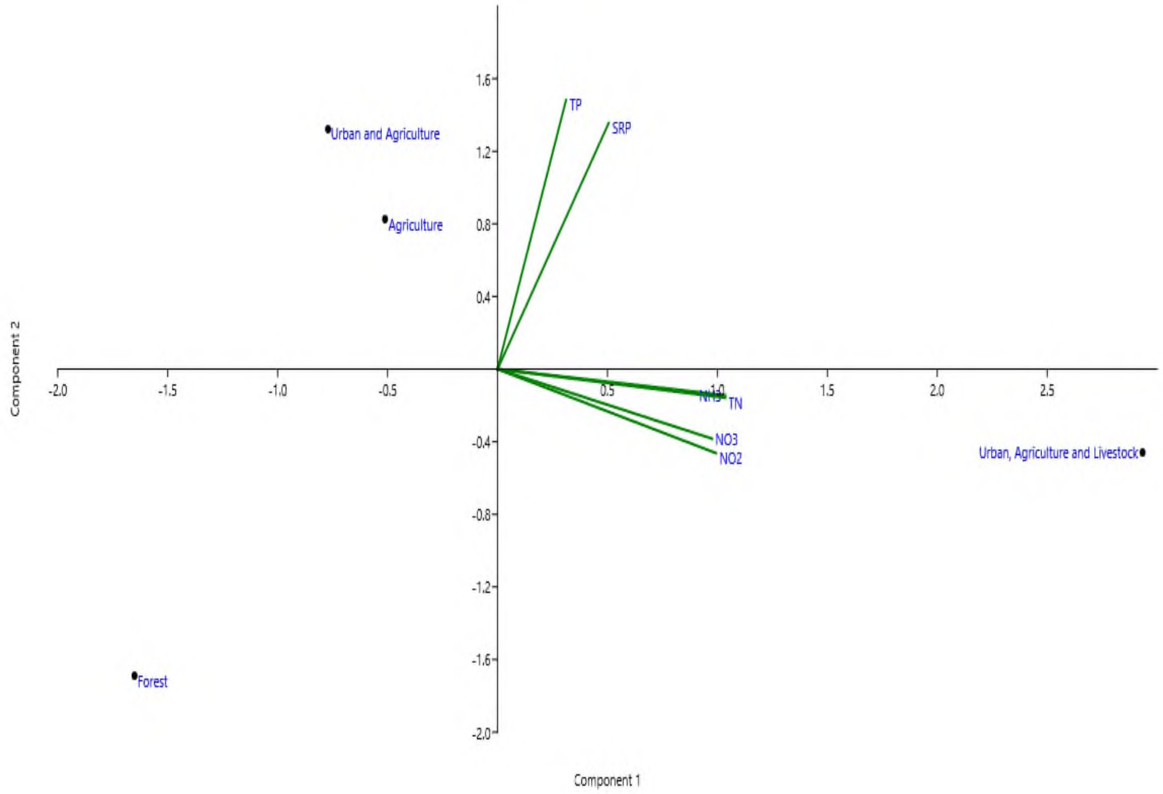

Figure 19: Principal Component Analysis (PCA) for inorganic nutrients in the FR, AGR, UAGR, and UAGL reaches of Wabe-Megecha River Catchment

4.4.7. Correlation analysis showing the relationship between physicochemical characteristics (temperature, DO, EC, pH, and TDS) of the Wabe-Megecha River Catchment

Correlation analysis was performed to show how the physico-chemical measurements are related to one another. The correlation analysis was done on temperature, DO, EC, pH, and TDS. Results presented in Table 9 showed that TDS had a perfect positive correlation with EC (1) followed by the significant positive correlation between TDS and pH (0.83), and pH with EC (0.82). While DO had a significant negative relationship with temperature (-0.672).

Table 9 Pearson Correlation analysis showing the relationship between physicochemical characteristics (temperature, DO, EC, pH, and TDS)

Parameters	To	DO (mg/l)	EC(µs/cm)	pH	TDS
To	1				
DO	-0.672**	1			
EC	0.762**	-0.085	1		
pH	0.506**	0.032	.829**	1	

TDS	0.761**	-0.085	1.00**	0.83**	1
	36	36	36	36	36

**. Correlation is significant at the 0.01 level (2-tailed).

4.4.8. Variation of physicochemical parameters

Physico-chemical parameters such as temperature, dissolved oxygen, pH, and conductivity have a significant effect on nutrient concentration in Wabe-Megecha River, mean concentrations and variability in DO were generally similar at the AGR and UAGR sites. Strong negative correlations were observed between DO and water temperature. Naturally, rivers have high dissolved oxygen, but it varies depending on organic matter content in the water, temperature conditions, and re-aeration processes (Effendi, 2016). Temperature influences the amount of dissolved oxygen in the water as saturation levels of oxygen decrease with an increase in temperature (Cox, 2003). In the main river, the highest DO concentration was recorded at FR sites (7.94 ± 0.06 mg/L) where the temperature was lowest (16.70 ± 0.50°C) while the lowest DO concentration was recorded at UAGL dominated site (5.48 ± 0.07 mg/L) where temperature (21.17 ± 0.06 °C) was the highest (19.59 ± 0.27 °C). This is attributed to human influence in and around the riverine reaches of the site. Most UAGL sites had an open canopy exposing the river channel to the sun. This explains the high water temperature obtained in this reach due to direct insolation. Warmer water has lower gas solubility and combined with high metabolic rates, can lead to low dissolved oxygen (Lewis, 2008).

A linear regression model showed that 45.2% of the variations in DO could be explained by temperature change. Pristine streams have high dissolved oxygen levels which reduce as the system becomes polluted (Lewis, 2008).

Dissolved oxygen levels can be used as a parameter to measure whether a system is organically polluted (Huang *et al*., 2017). The sites in UAGL domination received sewage effluents from nearby hotels, industries, and homes with low dissolved oxygen. Decomposition of the introduced waste causes high biological oxygen demand (BOD) thus reducing oxygen levels in the stream water. Therefore, Dissolved Oxygen has been used as a good indicator of sewage loading, rates of production, and consumption of organic matter in aquatic ecosystems (Huang *et al*., 2017). In Ethiopia, it is a common practice to dump urban sewage in to rivers without treatment **(**Hamere Yohannes and Eyasu Elias, 2017).

The temperature variation obtained could be attributed to the influence of ambient air temperature (USEPA, 2015). The temperature change can also result from the change in altitude, shading, and insolation (Ohmura, 2012). The forest site has the highest altitude and the temperatures never exceeded 20.02 °C.

The highest temperature values in AGR reaches can be attributed to low canopy cover hence high insolation. The low temperatures obtained in the upstream FR sites can also be attributed to the high canopy cover in the forest which shades the water preventing sun rays from penetrating and heating the water. The time of sampling also affects the water temperature as in the morning the water is cold and during the afternoon it is already heated and temperatures are higher. Temperature directly influences the decomposition of organic matter in streams. Increased temperatures increase the rate of leaf litter decomposition thereby enhancing microbial growth and metabolism (Saltarelli *et al*., 2018) which is resulted in depletion of DO. The results of this study are similar to those of Ontumbi *et al*. (2015) who found that temperatures in river Sosiani in Kenya ranged between 13.1 °C and 25 °C during the rainy season.

The pH values obtained ranging from 8.56 - 9.02 along the pollution gradient of the Wabe-Megecha river catchment, are within the range (6 - 9) which favors the survival and activity of most bacteria. If the pH variation range is below or above the optimum range of 6 to 9 nutrient cycling is inhibited (WHO, 2017). In the sites with the most turbulent flows like AGR, and UAGR sites the reaeration of water is enhanced thus causing pH to rise, agreeing with the study of Bhateria and Jain (2016), who documented a link between water reaeration and increased pH.

Electrical conductivity is a measure of the number of ions and salts in water and thus acts as an indicator of dissolved elements in the streams. High conductivity levels indicate high levels of dissolved ions in the stream water. The highest (257.33 μS/cm) and the lowest (89.70 μS/cm) EC recorded can be attributed to the release of nutrients and ions into the river water
column from leaching and mineralization. The release seems to be highest at the UAGR site which has the highest urban and agricultural influence and lowest at the FR site, a forest-dominated environment. Several studies found EC to be low in forested catchments (Githaiga *et al*., 2003; Kambwiri *et al*., 2014).

High conductivity recorded in the UAGR sites may be attributed to increased human activities in the riparian area adjacent to the streams which increase the loading of iron-rich sediments from the fertilized farms in the catchment into the river. Forested streams, on the other hand, are characterized by the dense riparian zone which protects the stream from much sediment loading thus lowering the EC. But due to the upstream agriculture, the forested area also exhibited higher EC. At the UAGL sites, EC was significantly higher than FR sites (89.70 μS/cm and 157.23 μS/cm) because of sewage load, livestock waste while drinking water, and other anthropogenic instream activities. This is similar to the results of a study by Markewitz *et al*. (2001).

AGR land use is associated with the usage of chemical fertilizers, pesticides, and acaricides for livestock which ends up in streams and rivers during runoff events. This can increase the ionic concentrations of water leading to high EC levels as observed in the downstream sites of the Wabe-Megecha River catchment in Ethiopia. Among the UAGR sites, the highest EC values were observed in the areas dominated by small-scale irrigation, followed by mining. The sites in the FR had the lowest EC.

Turbidity is a measure of the amount of light scattered or absorbed by a sample of water and is the most used metric for quantifying suspended sediment in streams (Markewitz *et al*., 2001). Turbidity is expressed in nephelometric turbidity units (NTU). Turbidity value was highest at the confluence UAGR site. The surface runoff from the surrounding agricultural farms and the sewage from the home, car wash, and floriculture contributed to the high turbidity in the downstream sites. Most of the farms in the downstream sites extend to the riverbanks thus whenever there is rainfall, the topsoil is washed into the river.

Land use change is considered a primary factor dominating soil erosion (Symeonakis *et al*., 2007) with anthropogenic activities like tillage in the agricultural areas aggravating soil loss (West *et al*., 2015) thus causing high turbidity in river water during rainfall events (Lee *et al*., 2019). A study by Li *et al*. (2015) stated that water turbidity strongly depends on the precipitation rate which causes erosion runoff. This surface runoff causes increased turbidity and its positive relationship with TSS concentration facilitates the estimation of TSS (Daphne *et al*., 2011).

The low turbidity in the forested sites is a result of intact riverine forest, relatively constant watering points for the livestock and several mitigation measures like vegetative buffers implemented in this area by local governments especially the plantation of *Acacia meanssi* as a measure of soil and water conservation.

The turbidity differences between the forested and agricultural reaches mean that turbid water mainly originates from agricultural activities (Lee, 2008). Lee *et al*. *(*2019) suggested a modified and sustainable agriculture system to reduce the soil loss in upstream agricultural areas and prevent its runoff as turbid stream water.

4.4.9. Discharge variation along the four land use types

Discharge kept increasing downstream. Rising discharge along the longitudinal continuum is caused by the tributaries which drain into the main river as it flows downstream. Wherever there was a confluence, discharge was higher than at the previous sampling station along the Wabe main river. Discharge was highest at the confluence with the Wabe Bridge site and was lowest at site FR site which has smaller tributaries. It's important to note that the sampling was done during dry-wet interface season which coincided with the time of the study, explaining there was a considerable amount of water flowing from the catchment into the river thus the increased discharge.

This is evidenced by the good relationship between discharge and variation in nutrient concentrations reported in several studies (OEPA, 2016). Land use change affects river discharge in catchments (Konkul *et al*., 2014). Human activities for example urbanization and agriculture can cause a decrease in infiltration consequently increasing the rate and volume of runoff (Klongvessa *et al*., 2017).

This explains the high discharge obtained at the AGR, UAGR, and UAGL sites during the samplings. The results of this study, which showed low discharge at FR sites compared to other land uses, agree with those of Bruijnzeel (2004) who argued that forests act as sponges by increasing infiltration rates and retaining soil moisture. The high discharge recorded in UAGR indicates that at high rainfall intensities, the impacts of forest cover in reducing stream flows are masked (Guzha *et al*., 2018). Romero *et al*. (2016) concluded that the impact of forest cover on peak discharges becomes insignificant with an increase in rainfall intensity.

4.4.10. Effect of land use on nutrient concentrations

There were notable variations in nutrient concentrations along the river. A clear understanding of the relationship between land use and water quality aids in identifying the primary threats to water quality (Ding *et al*., 2015). Both land use and land cover types can act to transform nutrients or bar them by preventing dissolved and suspended nutrients from moving towards streams and rivers (Basnyat *et al*., 2000). Ammonia was the lowest at the FR site and highest at the UAGL site. The high dissolved oxygen concentration in the FR site (7.94 ± 0.06 mg/L) explains the low ammonia concentrations since DO inhibits ammonification and favours the nitrification process. The corresponding high DO levels make the environment less conducive to the existence of ammonium as the main nitrogen component because it is converted to nitrites and nitrates. When a catchment has many point source inputs, there will be more ammonia (Li *et al*., 2014; Pernet-Coudrier *et al*., 2012). This is mainly due to the low dissolved oxygen levels that are created by high concentrations of organic matter in existing point source inputs thus causing ammonia to persist as a major component (Pernet-Coudrier *et al*., 2012). Ammonia is commonly found in high concentrations in urban rivers with high BOD mainly from untreated sewage and other sources of organic pollutants. Airsien *et al*. (2003) explained that the presence of high ammonia levels in surface water may mean there is un-treated or partially treated sewage input into a water body.

Nitrites had the lowest concentrations among all the forms of nitrogen along the Wabe-Megecha River. High nitrite concentrations are associated with incomplete nitrification. The lowest concentration of nitrite was recorded at the FR site (0.0250 ± 0.00029 mg/L) and the highest was recorded at site UAGL (0.0367 ± 0.00013 mg/L). There was a significant positive correlation between temperature and nitrites meaning that as the temperatures increased, the degradation of organic matter by bacteria thus releasing more nitrites through mineralization increased (Truu *et al*., 2009). Nitrate concentrations were at the base of the recommended levels (10 mg/L) by WHO (2017). The values were highest in UAGL and lowest in the FR site.

Nitrate is a good indicator of catchment disturbance in P-limited systems (Rao and Puttanna, 2000). Site UAGL is an area where the cumulative impact of urban effluent, agricultural activities (which use chemical fertilizers rich in nitrates like UREA), and the direct livestock faeces could have been washed into the river in surface runoff during the rains. According to the responses from the questionnaire, most farmers apply fertilizer per ploughing season (During the planting period and top dressing). As observed in the downstream sites, most farms extended into the riverbanks. This sampling having been done in the dry-wet interface season, it is possible nitrates were washed into the river through surface runoff. High dissolved oxygen levels promote nitrification resulting in high levels of nitrates due to the nitrification and mineralization of logs and woody materials on the riverbed.

Stream water temperature has a positive effect on ammonification, nitrification and decomposition of organic matter which is source process of nitrates and this explains the positive correlation between temperature and NO_3-N concentration. NO_3-N was highest in the UAGL site where the temperature was high.

Riparian vegetation increases NO_3 retention (Vidon and Hill, 2004) and this is probably why site FR which is found in the forest-dominated areas had the lowest (0.1601 ± 0.00113) NO_3 concentrations compared to the downstream sites which had less riparian vegetation.

The total nitrogen levels were highest at the UAGL site and could be attributed to human input
from the catchment (Wen *et al*., 2017). Site FR had the lowest TN. The higher concentrations of TN during the dry-wet interface season in the AGR stream reaches are consistent with the results from Ribeiro *et al*. (2014). There is evidence that TN increases with direct inputs from humans and cattle, especially near urban areas (Biggs *et al*., 2004).

Land use change and especially reduction of forest cover is known to affect phosphorus concentrations and export (Harris, 2001). Use of chemical fertilizers in agricultural farms increases SRP and TP levels in the nearby streams and rivers due to surface runoff (Harris, 2001). Concentrations of SRP in the Wabe-Megecha River have exhibited a slight increment. The concentrations were higher in the AGR, followed by UAGR reaches than the FR reaches during the entire sampling period. Bank erosion and resuspension of sediments during the storm events could also have been the cause of the high concentrations in the human-dominated and affected sites. This is in agreement with the study by Githumbi *et al*. (2021) who confirmed that SRP concentrations rise due to sediment resuspension following storm events.

Total phosphorus showed an increasing trend in all the sampled sites. The high TP levels downstream are attributed to the increased discharge and TSS, similar to studies by Crespo *et al*. (2011) and Markewitz *et al*. (2001) noted a positive correlation between nutrients, TSS, and discharge. The high levels could also mean that the mineralized phosphorus from the agricultural lands ends up in the streams and rivers through surface runoff as noted by some studies (Liu *et al*., 2012; Wilson and Xenopoulos, 2009). Usage of chemical fertilizers: AREA and DAP which are rich in phosphates also contributed to the high levels of TP in the AGR and UAGR sites. When there is rain, the fertilizers are carried in surface runoff from the farms into the river as non-point sources.

4.5. Macroinvertebrates Distribution, Abundance, and Composition

A total of 6,009 specimens from 9 orders and 29 families were sampled at 12 sites in 4 streams within the Wabe-Megecha catchment. Order Ephemeroptera, Diptera, and Odonata had the highest number of families each representing a total of six families. Hemiptera had three families. Trichoptera, Annelida, and Gastropoda were represented by two families. The lowest taxa in the study sites were Coleoptera (Elmidae), and Plecoptera (Perlidae) which were represented by only one family. Diptera was the dominant taxonomic group during this study period, which could be because of its ability to tolerate a wide range of water quality.

Spatially the highest relative abundance of individuals (38%) was found in AGR reach and the lowest (9.14%) was recorded in the FR site which is characterised by the intact forest**.** The highest percentage in the AGR site is mainly due to the increased input of organic nutrients from the catchment agricultural activities, which could have possibly given the benthic macroinvertebrates communities to increase in number as well as their tolerance ability to the existing pollution. A similar study was done by Deborde *et al.* (2016), who observed that the highest abundance of benthic macroinvertebrates was found in agricultural and mixed land uses. Bartlett-Healy *et al.* (2012), were also reported that most of the Dipteran families such as Chironomidae, Dixidae, and Culicidae tolerate a wide range of water qualities, especially in polluted waters by using atmospheric oxygen.

4.5.1. Macroinvertebrate community structure (Abundance and richness of macroinvertebrate taxa along the four land use types)

The highest number of individuals was recorded in AGR while the lowest abundance was recorded in UAGL. Order Diptera was the most abundant order among sites which is recorded in the same number in all land uses except UAGL which is highly polluted from all kinds of impacts i.e. urban effluents, agricultural chemicals, and livestock dung. Order Ephemeroptera, Trichoptera, and Diptera had higher taxa richness across the sites. Order Ephemeroptera and Diptera were the dominant taxa across the sites. Relative taxa richness across sites was dominated by Diptera, Ephemeroptera, and Trichoptera. Sites in AGR areas recorded higher taxa followed by FR, UAGR, and UAGL

4.5.2. Substrate composition along the Land use types of the Wabe-Megecha River catchment

Substrate distribution among the four land use types showed the dominance of Megalithal substrate in streams in forested areas while Macrolithal substrate dominated in streams within agricultural areas and Urban and Agricultural dominated sites of the Wabe-Megecha river sub-catchment. Akal (gravel) and Psammal (sand) substrate types were higher in Megecha river sites.

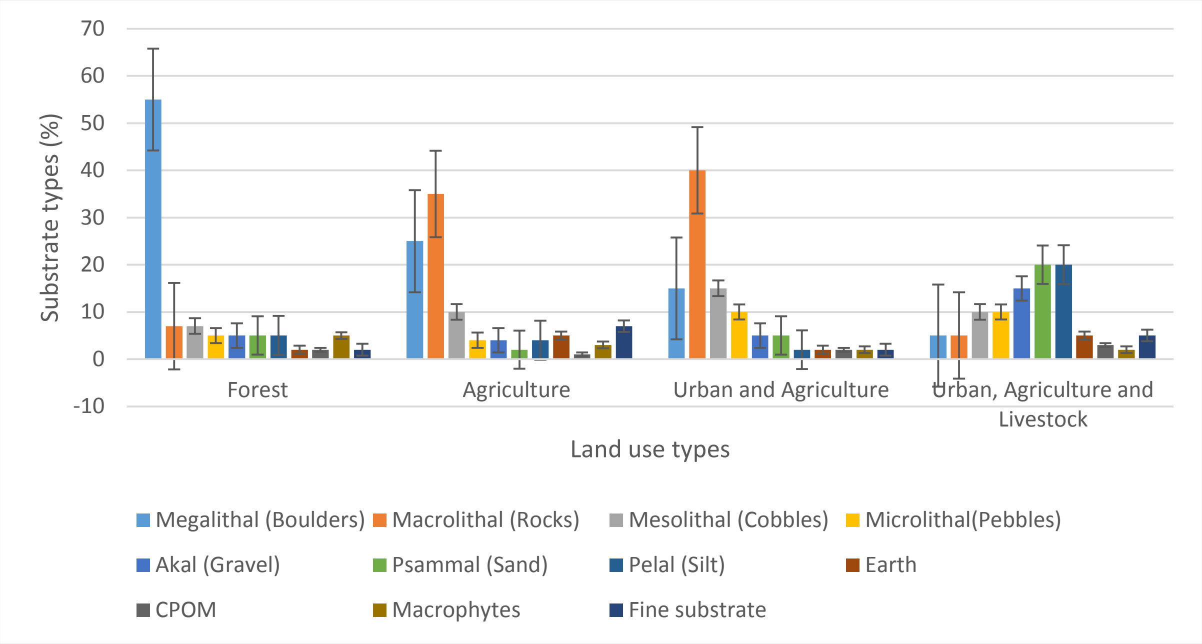

Figure 20: Mean Percentage of substrates per land use (FR, AGR, UAGR, and UAGRL). Error bars represent a 95% confidence interval

Table 8: Macroinvertebrate taxa composition, functional feeding types, and abundance in the Wabe-Megecha river catchment streams.

Taxa	Sampling sites				
	FR	AGR	UAGR	UAGL	FFG
Baetidae	924	120	127	29	CG
Leptophlebiidae	87	103	0	0	SCR/CG
Tricorythidae	95	369	98	0	CG
Caenidae	0	144	151	112	CG
Teloganodidae	0	223	84	0	C
Heptageniidae	603	680	254	0	SCR
Hydropsychidae	36	231	107	0	CF
Sericostomatidae	0	12	10	0	SHR
Elmidae	12	30	24	18	CG
Libellulidae	17	0	30	0	PRD
Gomphidae	0	110	42	0	PRD
Chlorolestidae	0	0	12	0	PRD
Coenagrionidae	0	12	6	12	PRD
Chlorocyphidae	0	68	60	0	PRD
Aeshnidae	6	0	12	0	PRD
Simulidae	37	63	31	24	FT
Chironomidae	24	90	108	78	CG/FT/PR
Tabanidae	0	0	5	0	PRD
Culicidae pupa	18	30	6	18	FT/CG
Tipulidae	6	1	0	0	SHR/CG
Athericidae	30	36	6	0	PRD
Perlidae	18	0	0	0	PRD
Corixidae	6	0	0	18	Pr/Sc
Belostomatidae	0	0	6	6	Pr
Notonectidae	0	0	0	10	Pr
Oligochaeta	6	0	0	0	CG
HIRUDINAE	0	0	22	212	Pr/Par
Planorbidae	0	0	6	0	SCR
Physidae	0	0	6	12	SCR

Where: FR; Forest dominated site AGR; Agriculture dominated site, UAGR; urban plus agriculture dominated, UAGL; the cumulative effect of Urbanization, Agricultural activities and livestock

4.5.3. Diversity of Macroinvertebrates in Wabe-Megecha River Catchment

The diversity of macroinvertebrates along with the four land use types in the Wabe-Megecha River catchment during the study period is indicated in (**Error! Reference source not found.**). Thus, based on the value of the Shannon- Weiner diversity index (H) the four land use types varied from 1.5 to 2.6. The highest value (2.6) was observed at the UAGR reach followed by 2.3 at the AGR reach. Whereas, the lowest (1.5) was found at the FR reach. This variation among the land uses could be the availability of quality and quantity of food sources, multi-habitat, the trophic structure, and the level of environmental stress for each site. The present result agreed with Morphin-Kani and Murugesan (2014), who suggested that the high macroinvertebrate diversity could be an indication of a stressful- environment and very low diversity showing the environment is under some lack of habitat availability. However, Egler *et al*. (2012), and Lozupone *et al*. (2012), stated that stable environments have high diversity, while unstable environments have low diversity.

Besides, the Shannon-Wiener index, diversity within the macroinvertebrate community was also described using Simpson’s diversity index (1-D). According to Mandeville (2002), the Simpson Index (1-D), with values ranging from 0 to 1. The values 0, indicate a low level of diversity, and 1 for a high level of diversity. Simpson’s diversity index in the Wabe-Megecha River catchment varied from 0.7 (FR) to 0.9 (AGR and UAGR). The highest (0.9) was observed in AGR, and UAGR where the diverse nutrients come from the catchment and the lowest (0.7) was in FR sites where there is relatively high water quality. This recorded value in the Wabe-Megecha catchment according to the given range indicated the presence of almost a high level of diversity. The Shannon-Wiener Index (H′) and Simpson’s diversity index (1-D) show the same trend for each land use type. However, they had an inverse relation to Dominance (D). This finding was also agreed with Magurran (2013), who stated that Shannon’s diversity and Simpson’s diversity indices differ in their theoretical foundation and interpretation, but they have strong correlations with each other.

Margalef richness index in the Wabe-Megecha River was observed in the range of 2.0 and 3.1. Relatively the highest Margalef richness index was 3.1 followed by 2.1, 2.0, and 1.7 which were found at the river reaches of UAGR, AGR, FR, and UAGL respectively. The variation among the riverine reaches of different land uses might be due to the level of environmental stress in the area via increased human activities. This idea was verified by Andem *et al*. (2012), who suggested that low taxa richness may indicate the environment is seriously degraded with various anthropogenic activities. This also affected the benthic macroinvertebrate community.

Table 9: Macroinvertebrate taxa composition, functional feeding types, and abundance in the Wabe-Megecha river catchment streams.

Category	Land use types			
Richness measures	Forest	Agriculture	Urban and Agriculture	Urban, Agriculture and Livestock
Total no. of individuals /all sample	1925	2322	1213	549
Total no. Of taxa	16	17	23	12
No. of EPT taxa	6	8	7	2
No. Ephemeroptera taxa	4	6	5	2
No. Trichoptera taxa	2	2	2	0
No. Hemiptera taxa	1	0	1	2
No. Coleoptera taxa	1	1	1	1
No. Diptera taxa	5	5	5	3
No. Odonata taxa	2	3	5	1
No. Oligochaeta taxa	2	2	1	1
Composition measures				
% EPT	89.7	70.6	58.9	25.7
% Diptera	6.0	9.5	12.9	21.9
% Ephemeroptera	88.8	70.6	58.9	25.7
% Trichoptera	1.9	10.5	9.6	0.0
% Hemiptera	0.3	0.0	0.5	6.2
% Coleoptera	0.6	1.3	2.0	3.3
% Diptera	6.0	9.5	12.9	21.9
% Odonata	1.2	8.2	13.4	2.2
% Oligochaeta	0.3	0.0	1.8	38.6
Dominance and diversity				
Dominance_d	0.3	0.1	0.1	0.2
Simpson_1-d	0.7	0.9	0.9	0.8
Shannon_h	1.5	2.3	2.6	1.9
Evenness_e^h/s	0.3	0.6	0.6	0.5
Margalef	2	2.1	3.1	1.7
Equitability_j	0.5	0.8	0.8	0.8

Diversity of Macroinvertebrates in Wabe-Megecha River Catchment

The range of evenness also varied from 0.3 to 0.6. The highest evenness (0.6) was recorded at both the riverine reaches of the AGR, and UAGR sites. This might be due to the status of water quality. This result was agreed with Dipankar and Jayanta (2015), and Upen and Sarada (2015), who observed high evenness showing poor water quality and lack of available food. The recorded value indicates that the evenness index and richness index had the same trends from all types of land use types. This idea disagreed with Chrisoula *et al*. (2011), who suggested that the increasing value of the species richness index was responsible for the reduced value of the evenness index.

The equitability of benthic macroinvertebrates along sampling sites was found in the range of 0.5 and 0.8. The maximum value (0.8) was observed at all sampling reaches of land use types except at the FR site, which has a value of 0.5. It had similar trends with the Margalef richness index at each site.

4.5.4. Functional feeding groups (FFGs) among the four land use types

Predators (PRD) and collectors and grazers (CG) were the most dominant FFGs distributed across all sites (Figure 21). The functional feeding groups belonging to SHR/CG and Pr/Sc were the least abundant and they were only found in FR sites. The highest FFG abundance and richness were recorded in UAGR, followed by AGR and FR. While the lowest was in UAGL. This is mainly because of the degradation of water quality due to the cumulative impact of multi-anthropogenic factors including the sewage from the agro-industries, flower farms, house sewage, and livestock.

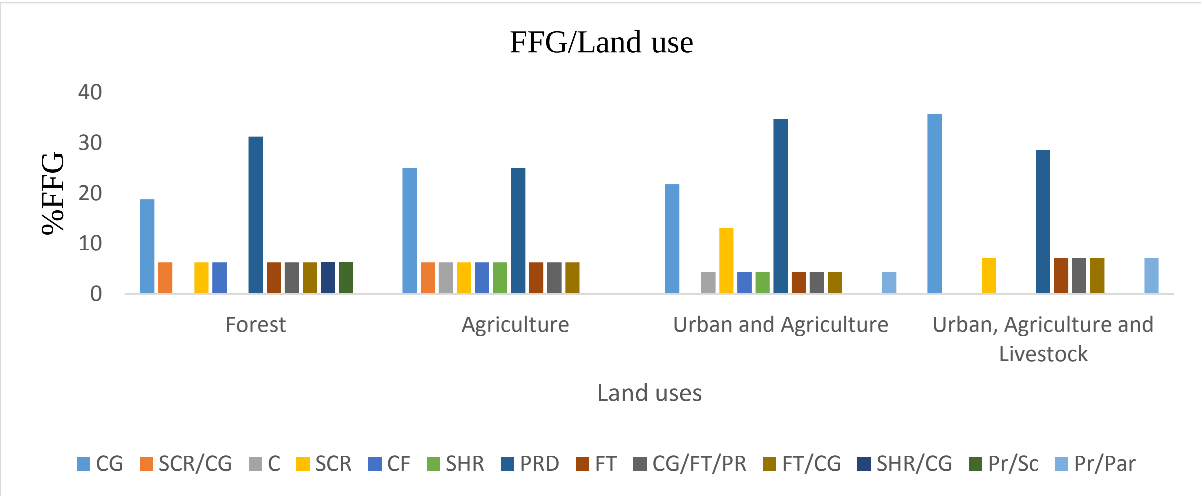

Figure 21: Functional feeding groups (FFGs) among the four land use types: Functional feeding groups (FFGs) among the four land use type

4.5.5. Impact of land use types, and physical water quality on macroinvertebrate

The relationship between the physicochemical parameter and macroinvertebrates communities is illustrated in (Figure 22). The first and the second canonical axes explained 58.066 % (eigenvalue of 10.4519) and 37.462 % (eigenvalue of 6.74325) of the variation in the macroinvertebrates data respectively. Based on (Figure 3) the CCA ordination showed that variations in benthic macroinvertebrate communities was related the land use and physicochemical characteristics including temperature, conductivity, turbidity, dissolved oxygen, pH, saturation, and so on. In the first axis, UAGL is positively correlated to Oligochaete and Hemiptera. While the AGR, and UAGR are characterized by high discharge, TSS, conductivity, TDS, and pH, had a positive correlation with Trichoptera and Odonata. Benthic macroinvertebrates require varied optimal temperature to survive (Singh and Sharma, 2014; Prommi and Payakka, 2015). In contrast, the temperature was negatively correlated with those macroinvertebrates. A similar result was observed in the study by Maneechan and Prommi (2015).

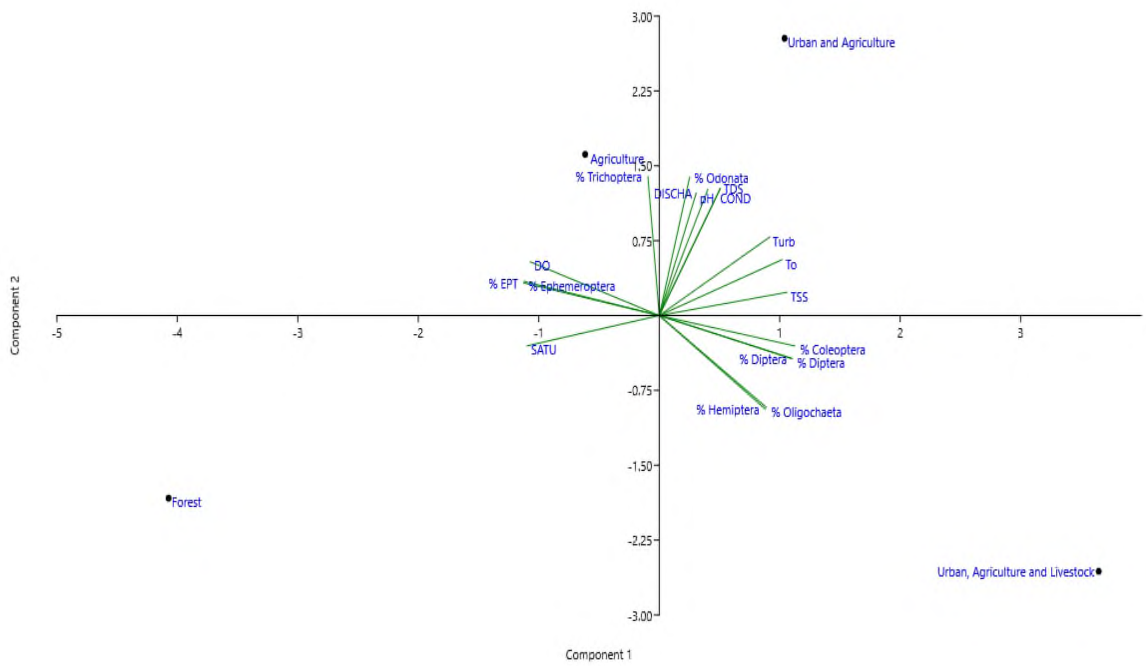

Figure 22: Relationship between the physicochemical parameter and macroinvertebrates communities

4.5.6. Impact of land use types, and nutrients on macroinvertebrate

The relationship among the land use types, the macronutrients, and macroinvertebrates communities is illustrated in (Figure 23). The first and the second canonical axes explained 73% (eigenvalue of 11.6953) and 23.4 % (eigenvalue of 3.74766) of the variation in the macroinvertebrates data respectively.

Based on (Figure 23) the CCA ordination showed that variations in benthic macroinvertebrates communities were related to the UAGL where high nutrients mainly the nitrogen compounds including the nitrate, nitrite dissolved oxygen, ammonia, and other variables. In the first axis nitrite, nitrate, total nitrogen, and ammonia were positively correlated with Hemiptera and Oligochaeta. While, UAGR which is characterized by phosphores compounds SRP and TP positively correlated with Trichoptera and Odonata. Benthic macroinvertebrates require nitrate, salinity and soluble reactive phosphate (SRP) were negatively correlated with those macroinvertebrates. Ephemeroptera were exhibited a negative correlation with Phosphates. A similar result was observed in the study by Maneechan and Prommi (2015), who stated that Baetidae negatively correlated with the concentration of phosphate.

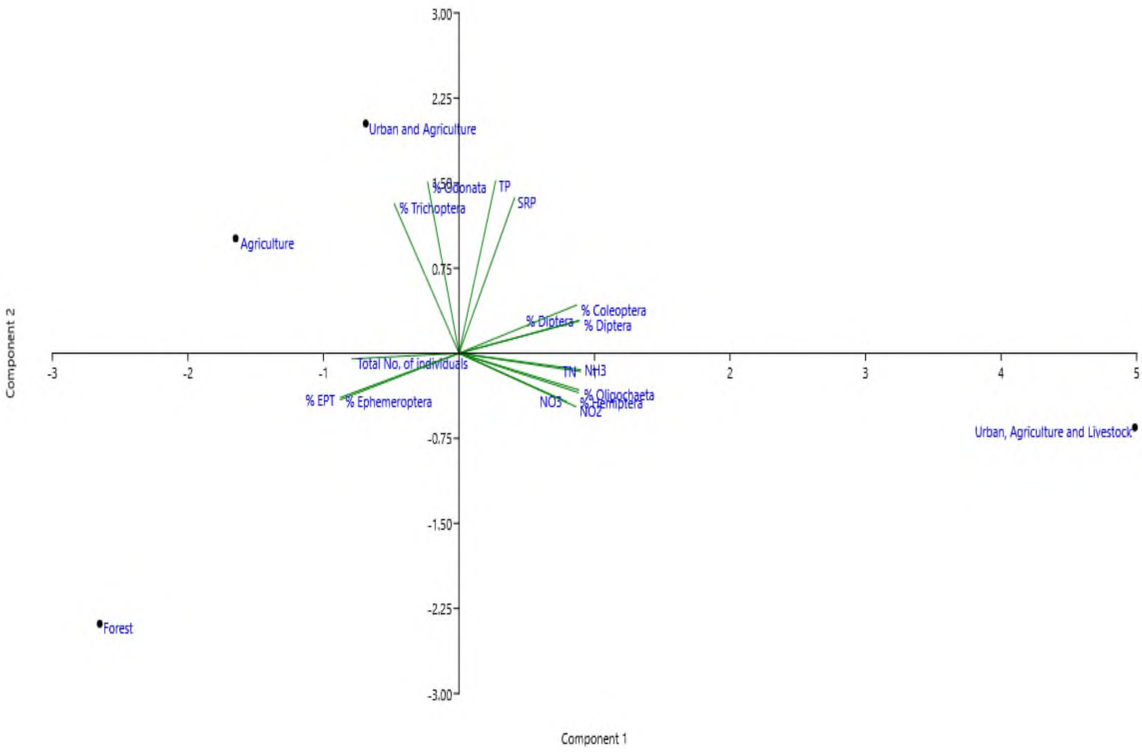

Figure 23: Relationship among the land use types, the macronutrients and macroinvertebrates communities

4.5.7. The similarity of macroinvertebrate communities along the four land use types

Among the four land uses types, the highest (75%) macroinvertebrate taxa similarity was recorded between the AGR and UAGR. While the lowest (42.9%) was recorded between the FR and UAGL. The highest similarity between AGR, and UAGL is mainly because these land uses share relatively similar nutrient inputs from the catchment and share similar habitats. The water quality of the two sites is nearly similar. While the lowest taxa similarity between the FR and UAGL is mainly attributed to the difference in water quality and habitat types.

Table 10: Similarity of macroinvertebrate communities along the land use types

	Forest	Agriculture	Urban and Agriculture	Urban, Agriculture and Livestock
Forest	1			
Agriculture	0.667	1		
Urban and Agriculture	0.513	0.750	1	
Urban, Agriculture, and Livestock	0.429	0.483	0.571	1

5. Conclusion and recommendations

In the last 4 years, land use/land cover changes have undergone considerable change in the study area. Similar to other catchments in Ethiopia, the Wabe-Megecha River catchment showed the expansion of AGR while FR showed a stabilized pattern. The undergoing green initiative has contributed to the stabilization and future increment.

Nutrient concentration generally increased downstream, with significant variations among the land use types; FR, AGR, UAGR, and UAGL. Most nutrients were highest in the downstream sites and lowest in the upstream except SRP and TN which were low in the midstream and downstream respectively.

The land use change along the Wabe-Megecha River Catchment has significantly affected the abundance, composition, distribution, and diversity of benthic macroinvertebrates fauna of the study area. It was mainly due to the deterioration of the natural habitats and environmental stress.

Sustainable land use management and natural forest conservation should be practiced, and the undergoing green legacy efforts should be done seriously to maintain the ecosystem services, and biodiversity in the study area.

The existing riparian buffer zones should be maintained to mitigate the impacts of land-based anthropogenic activities. These preventive measures can be achieved by motivating farmers and providing incentives and guidance by local leaders.

Moreover, the farmers around the reach of Wabe-Megecha River catchment should adopt organic farming and reduce the usage of chemical fertilizers. This will reduce the levels of nutrients entering the river through surface runoff. Farmers should also avoid riverbank farming and washing clothes in the river to minimize soil erosion and increased phosphorus concentrations.

Generally, this study can be used as a baseline for further studies in the area and provides information for the management and improvement of the Wabe-Megecha River Catchment.

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