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UNESCO-IHE Institute for Water Education

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EFFECTS OF LAND USE ON NUTRIENT CONCENTRATIONS, CARBON DYNAMICS AND STREAM METABOLISM IN RIVER RUPINGAZI, KENYA

Master of Science Thesis

by

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DECLARATION AND RECOMMENDATION

Declaration

This Thesis is my original work and has not been submitted or presented for examination in any institution.

Signature:

Dorcas Mutei Samuel Date: 24/05/2020

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Recommendation

This Thesis has been submitted with our approval as supervisors for examination according to Egerton University regulations, on behalf of partnering institutions

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DEDICATION

This work is dedicated to all those who gave me support during the research period.

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ABSTRACT

Rivers and streams store, process, and transport carbon and nutrients between terrestrial and marine ecosystems. Concentrations of these elements vary from one part of the stream to another depending on the land use change, which alters infiltration, surface runoff and evapotranspiration, thus influencing nutrient input into rivers and streams. This study was conducted in Embu, Kenya between November 2020 and January 2021 to determine how land use affects nitrogen (N), phosphorus (P) concentrations, carbon dynamics and stream respiration in River Rupingazi, since significant land use change has been observed over the years yet little has been documented about the effects. Water samples for nutrients and Dissolved Organic Carbon (DOC) analysis and sediment samples for respiration measurements were collected from ten selected sites in upstream, mid-stream and downstream of the river identified according to land use. In the laboratory, ammonium, nitrites, nitrates, Total Nitrogen, Soluble Reactive Phosphorus, Total Phosphorus and DOC were determined. Stream respiration was investigated through incubating sediment samples and measuring organic matter content afterwards. Questionnaires were administered to locals to find out the dominant land use type and main agricultural activities. Data from this study was analysed using R software version 4.0.0 and SPSS software version 25. One way ANOVA was used to check for differences in nutrient, DOC concentrations and respiration in the three land use types. Pearson correlation was used to determine the relationship between the variables. Statistical tests were done at 0.05 significance levels. Most nutrients and physicochemical variables showed significant differences (ANOVA, $p<0.05$) among the sampled sites except pH and TP. The agricultural land use had significantly higher nutrient concentrations compared to urban and forest land uses. Concentrations of DOC increased along the longitudinal continuum and ranged between 0.44 ± 0.14 mg/L in the forested section and 0.67 ± 0.30 mg/L in the agricultural section. Stream respiration rates were highest in site R5 (1.8 mg O_2/h) in the agricultural section and lowest in R10 (0.9 mg O_2/h), also in the agricultural area and did not show significant variations among the land use types. Therefore, results from this study bridge the gap between the influence of land use changes on N, P, DOC and stream metabolism and form baseline information for sound catchment management to improve the health of River Rupingazi. This study recommends that nutrients, DOC dynamics and stream respiration be studied during both the wet season and dry season to get the dynamics well.

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LIST OF ABBREVIATIONS AND ACRONYMS

ANOVA	Analysis of Variance
APHA	American Public Health Association
BOD	Biological Oxygen Demand
DOC	Dissolved Organic Carbon
DOM	Dissolved Organic Matter
FAO	Food and Agricultural Organization
GPS	Global Positioning System
KNBS	Kenya National Bureau of Statistics
N	Nitrogen
NH_4-N	Ammonium-Nitrogen
NO_2-N	Nitrite-Nitrogen
NO_3-N	Nitrate-Nitrogen
P	Phosphorus
QGIS	Quantum Geographical Information Systems
RCC	River Continuum Concept
SRP	Soluble Reactive Phosphorus
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
UV	Ultraviolet
WHO	World Health Organization

CHAPTER ONE
INTRODUCTION

1.1 Background information

Rivers are essential in carbon storage, transport and processing between terrestrial and marine environments (Aufdenkampe *et al*., 2011; Wohl *et al*., 2017). In addition, they act as both sources and recipients of Carbon by releasing CO_2 to the atmosphere and receiving water from the groundwater sources rich in CO_2 as well as heterotrophic processes in the system (Tamooh *et al*., 2013). Primary nutrients are known to limit primary production in streams and rivers where the extent and nature of their limitation varies with latitude. Concentrations, stoichiometry and molecular forms of macronutrients: N and P control primary production in most aquatic systems (Matano *et al*., 2015). Their availability controls stream and river processes like primary production and organic matter decomposition. Therefore, understanding their cycles is vital in understanding the biogeochemistry of a system (Boulton *et al*., 2008). Dodds and Smith (2016) added that availability of N and P influences autotrophic and heterotrophic processes in streams and river ecosystems.

Rivers and streams provide many social services and ecological functions (Yeakley *et al*., 2016) including water provision for community and industrial use. Therefore, these systems could easily be subjected to pollution and overexploitation by humans. Past 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). Bodmer *et al*. (2016) stated that there is need to understand how rivers, especially tropical systems, are rapidly being influenced by land use, potentially changing river nutrient characteristics. Several studies have demonstrated how rivers' nutrient contents are affected by land-use change (Aheimer & Liden, 2000; Carpenter *et al*., 1998; Zhou *et al*., 2019). This may occur through the degradation of riverine adjacent riparian land coupled with an increase in soil erosion and sediment loading into the river channel. Njuguna *et al*. (2020) found percolation of leached nutrients into rivers due to riparian encroachment increasing and polluting river waters in the Lake Victoria catchment.

Land-use change has more severe effects in tropical areas than in the temperate because tropical organic soils are rapidly mineralized and highly erodible during surface runoff (Hartemink *et al*., 2008). There is a wide belief that agriculture is the main source of nutrients in river waters in rural catchments and this threatens people's livelihoods, biodiversity, and fish (Verschuren *et al*., 2002). Several researchers (Misana *et al*., 2003; Mugisha, 2002; Olson *et al*., 2004) have noted that most land use changes in Africa are caused by agricultural activities intensification and extension. Forests are being cleared in many tropical areas and it has been approximated that 38% of East African forest land will be converted into cropland and grazing land by 2035 (Misana *et al*., 2003). This will increase soil erosion from overgrazing and uncontrolled tillage (Semalulu *et al.*, 2015).

Studies about DOC have been of great interest due to its importance in the downstream ecosystems (Pagano *et al*., 2014; Post *et al*., 2009) through modifying other chemicals and stream processes (Prairie, 2008). Changes in DOC levels affect nutrient uptake and surface water quality (Stanley *et al*., 2012). In Eastern Africa, these changes have been evident in Rwanda's Rukarara River (Rizinjirabake *et al*., 2019) and Kenya's Tana River basin (Tamooh *et al*., 2012). When terrestrial ecosystems get modified by anthropogenic activities, the sources and concentrations of DOC within streams are altered (Lu *et al*., 2014; Yamashita *et al*., 2011).

Microbial respiration of organic carbon is one component of stream metabolism and is a fundamental ecosystem process that utilizes DOC as a source of energy which is crucial in the whole carbon cycle (Demars, 2019). It takes place in the riverbed sediments which are biogeochemically active zones playing a key role in the energy flow and carbon processing in running waters (Battin *et al*., 2016). Stream respiration affects the dissolved oxygen concentration and is controlled by DOC and Dissolved Organic Matter (DOM) delivery from different aquatic and terrestrial sources such as soil into rivers and streams (Lu *et al*., 2014). The quality and quantity of DOM delivered depends on the land use in the catchment (Battin *et al*., 2008). Therefore, quantifying the spatial and temporal dynamics of stream sediments across a stream network helps to understand both carbon cycling and the microbial compartment (Mejia *et al*., 2019).

In the Mt Kenya region, land use has been transformed from forested areas to settlements, farmlands, grazing lands and urban areas (Bonareri, 2017). These activities are known to change the ecological conditions of rivers (Sponseller *et al*., 2001). For example, crop farming leads to nutrients (mainly from chemical fertilizers) input, organic pollutants and sediment input when there is surface runoff into the streams (Buck *et al*., 2004; Greig *et al*., 2005). Households release waste and sewage into the river leading to increased nitrogen and phosphorus input (Zhang *et al*., 2012). River Rupingazi drains a land use gradient area ranging from forested to agricultural and urban areas (Bonareri, 2017). However, little information has been documented on the effect of this land use change on the river water quality. Therefore, this study aimed to provide more information about the effect of different land use types (forested, urban and agricultural) on N, P, DOC concentrations and stream metabolism along the course of river Rupingazi. This information can be used to develop management strategies for the Rupingazi catchment.

1.2 Statement of the problem

The population growth in East Africa is increasing rapidly and due to unemployment, people are exploiting the natural resources for livelihood. In Kenya, population has been increasing at a rate of 3% per annum (Kenya National Bureau of Statistics). Many of the existing forests are being cleared to create land for agriculture to support the fast-growing population, some of which utilize chemical fertilizers or organic manure to boost crop productivity. This threatens water catchment areas and consequently water quality of recipient systems. Land use change in Mount Kenya area by anthropogenic activities is on the rise and consequently, it is essential to understand how these land use changes influence nutrients and carbon dynamics as they affect stream and river functioning.

Limited information has been documented on the effects of human activities on nutrient levels in River Rupingazi and its tributaries. Previous studies in this catchment have focussed mainly on the broad topic of how human activities influence water quality. Less attention has been given to macronutrient concentrations, carbon cycling, and stream metabolism which are key in establishment of a system health. Major human activities like forestry, tea, coffee and dairy farming in the upstream, human settlements and mixed farming (maize, beans, sorghum, millet, and green grams) in the downstream are on the rise yet major nutrients and carbon concentrations in the water column remain little known. This study sought to investigate how N, P, DOC concentrations and stream respiration rates change with land-use types along River Rupingazi to improve the existing knowledge for decision making regarding the sustainable management of the catchment and other similar systems in the region.

1.3 Objectives

1.3.1 General Objective

To investigate the influence of land use on nutrient concentration, organic carbon dynamics and stream respiration rates in tropical rivers of Kenya.

1.3.2 Specific Objectives

i. To determine the concentrations of N, P and DOC in the water column along the longitudinal continuum of River Rupingazi.

ii. To evaluate the differences in N, P and DOC concentrations in the water column between the different land use types.

iii. To quantify stream respiration rates in the sediments at the forested, agricultural and urban stream reaches.

1.4 Hypotheses

$H_0 1$: There is no significant change in water column N, P and DOC concentrations along the longitudinal continuum of River Rupingazi.

$H_0 2$: There is no significant difference in N, P and DOC concentrations in the water column between the forested, agricultural and urban reaches of River Rupingazi.

$H_0 3$: There is no significant difference in stream respiration rates at the forested, agricultural and urban stream reaches.

1.5 Justification

With the increasing population in East Africa and the need for increased food production, there has been a huge conversion of forests into agricultural and settlement areas yet the impacts of these land use changes on stream and river water quality remain understudied. This calls for better understanding of nutrient and carbon dynamics since they are essential elements for stream health and processes like decomposition and primary production. There is still limited knowledge about how DOC is altered by land use, yet it has been increasing in most inland surface water bodies, mostly due to changing land use and land cover. The extent to which the magnitude of stream respiration changes with the conversion of forests to agricultural and urban lands in tropical streams and rivers is poorly understood.

Therefore, this study contributes to provide more understanding and new knowledge on the influence of land use on, N, P, DOC concentrations, and stream metabolism in River Rupingazi. The results of this study will also add to knowledge about the impacts of land use on tropical stream and river ecosystems, which are crucial for the survival of people and biological communities like macroinvertebrates and algal communities. The ability to link nutrient, carbon dynamics, and respiration rates with land use types further helps to understand system dynamism and its effect on production. Results from this study provide a baseline information that can be used to recommend best measures for sustainably managing River Rupingazi catchment through control of nutrient pollution.

CHAPTER TWO

LITERATURE REVIEW

This study focused on nutrients (Nitrogen and Phosphorus), carbon (DOC) and stream metabolism (respiration rate), which are essential in the processes and functioning of aquatic ecosystems. Rivers and streams in many parts of the world receive numerous inputs of N and P (Dodds & Smith, 2016) hence causing eutrophication which calls for nutrient control in the lotic ecosystems (Dodds & Welch, 2000). Nutrient concentrations in streams and rivers are often coupled to organic carbon metabolism (Gucker & Pusch, 2006). Some studies state that stream respiration rates increase in response to increased nutrients from agricultural land use (McTammany *et al*., 2007).

2.1 Nitrogen

Nitrogen occurs in water in both organic and inorganic forms. Organic forms include decaying animal and plant matter, while inorganic forms include nitrates, nitrites and ammonium (Kumar & Puri, 2012). Nitrogen is essential for primary production in aquatic ecosystems. Allowable natural systems levels are 10 mg/L for nitrates, 1 mg/L for nitrites and 0.5 mg/L for ammonia in Kenya (WHO, 2017).

Ammonium occurs naturally in water bodies arising from the microbiological decomposition of nitrogenous compounds in organic matter. Ammonium in rivers is as a result of discharges from industrial effluents, wastewater treatment plants, municipal effluent discharges and excretion of nitrogenous waste by animals such as cows, goats, sheep among others (Kumar & Puri, 2012). It can also enter rivers through indirect means like air deposition, nitrogen fixation and run off from agricultural farms. Ammonium can also arise in waters from the decay of discharged organic waste and is known to exert a demand on oxygen in water as it is transformed to the 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 in relation 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, ionised (NH_4^+) and un-ionised (NH_3) with the latter being toxic to freshwater fish at very low concentration. The relative proportions of ionised and un-ionised ammonium in water depend on temperature and pH and to a lesser extent on salinity. The concentration of un-ionised ammonium becomes greater with increasing temperatures and pH and with decreasing salinity (Kir *et al*., 2019).

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 (Smuleak *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 & Puri, 2012). This form of nitrogen can be used as a source of nutrients for plants and its presence speeds up plant growth.

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 compound 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 (Smuleak *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 & Puri, 2012).

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 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 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 rocks 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 & 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 Dissolved Organic Carbon (DOC)

Organic carbon, the measurable component of organic matter, exists in either organic or inorganic forms (Weishar *et al*., 2003). Gonet and Debska (2006) defined Dissolved Organic Carbon (DOC) as the fraction of total organic Carbon able to pass through a filter of below 0.45 µm in size while Sabine *et al*. (2004) stated the importance of its export from terrestrial to aquatic ecosystem in the global carbon cycle. Molecules comprising of DOC in water samples have their average absorptivity at 254 nm (Weishar *et al*., 2003). Involvement of DOC in water pH buffering, precipitation of nutrients, microbial processing and its influence on stream metabolism makes it a crucial biogeochemical element in streams and rivers. Studies about DOC are important because it is a potential carbon source, provides energy for heterotrophic organisms, promotes growth of microorganisms, alters aquatic ecosystem chemistry and contributes significantly to stream ecosystem metabolism (Sabine *et al*., 2004). High concentrations of stream DOC are observed during floods than during low water levels because when rainfall increases, runoff increases, thus increasing carbon release (Lambert *et al*., 2016). Dissolved organic carbon in natural waters ranges between 0.46 and 5.75 mg/L (WHO, 2017). Its concentration is controlled by river and stream catchment properties. The quantity and quality of terrestrial DOC, once delivered to the streams and rivers can be modified by sedimentation, microbial processing, adsorption/desorption and respiration (Stanley *et al*., 2012).

According to Finlay and Kendall (2007), DOC comes from either autochthonous (produced by biological activities in the river) or allochthonous sources (DOC from detrital input during soil erosion). Dissolved organic carbon is also delivered to stream waters through leaching, riparian zone exchange, groundwater retention and waste discharge (Bouwman *et al*., 2013). It can also get into streams and rivers through exudes from instream metabolism and carbon fixation by aquatic vegetation. Dissolved organic carbon is the most mobile and important source of carbon for microorganisms and can easily reflect the effect of land use on river and stream water quality. Changes in DOM composition affects microbial processing. In pristine streams, the most prevalent dissolved organic matter is soil or plant-derived while disturbed areas are dominated by bacterial or algal matter (Lambert *et al.,* 2017; Wilson & Xenopoulos, 2009;). The DOC in river water is a highly discharge-dependent variable (Fraser *et al*., 2001; Räike *et al*., 2012; Strohmeier *et al*., 2013) for example, when discharge is high after a storm event, DOC is usually higher (Jones *et al*., 2019).

2.4 Stream respiration

Stream metabolism, defined by the balance between organic matter produced via photosynthesis and consumed through aerobic respiration, is an important property of ecosystems, with significant influences on energy fluxes and ecosystem functioning (Correa-González *et al*., 2014). It has been proposed as a functional indicator of streams' and catchments' ecological integrity of streams (Fellows *et al*., 2006). The relative importance of gross primary production (GPP) and ecosystem respiration (ER) defines the prevalence of either autotrophy or heterotrophy, influencing the role of the aquatic systems in carbon dynamics, nutrient cycling and dissolved oxygen production (Dodds & Cole, 2007). Stream respiration is normally reflected by the diurnal changes in DO concentration (Dodds *et al*., 2013).

Respiration activity in streams and rivers is regulated by factors such as light, flow regime, and seasonal variation of nutrient availability. It is further affected by land-use conversion and anthropogenic impacts, causing overall environmental degradation for example eutrophication, introduction of terrestrial DOM and biodiversity loss (Capps *et al*., 2016; Dodds *et al*., 2013). Stream respiration has multiple implications for ecological processes, for example it affects the nutritional quality of resources available for consumers in the food chains, and therefore influencing secondary productivity (Boëchat *et al*., 2011). It also plays an important role in providing ecosystem services related to drinking water quality, pollution abatement and nutrient retention (Hall & Tank, 2003; Sobota *et al*., 2012). Peaks in stream respiration are expected when energy inputs (light or leaf litter) and temperature are elevated and disturbance events (high flows) are absent (Dodds *et al*., 2013). At instances of high DOM concentrations and organic-rich particles delivery into stream and river sediments from agricultural and pasture lands, stream metabolism is stimulated.

2.5 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 on 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 towards 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 &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 watershed 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, variation of climatic conditions should be taken into consideration when studying whole nutrient dynamics in 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 intensified 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 & Xenopoulos, 2009). In agricultural lands, some of the chemicals like herbicides, pesticides and fertilisers are washed into the streams in surface runoff during floods and this increases the nutrient loads particularly nitrogen and phosphorus compounds. 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 & Tyagi, 2015). Clearing of catchments exposes streams and waters to direct sunlight thus causing temperatures to increase and chemical composition to change. 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 & 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 to reduce surface runoff. This causes retention time to lengthen thus allowing nitrogen and phosphorus uptake, break down and utilization (Löfgren, 2009). Suspended solids and dissolved nutrients are lower in forested areas with minimal disturbance compared to areas with human disturbance like agricultural and urban landscapes (Ghermandi *et al.,* 2009). This causes organic matter to be degraded faster instream and the result is increased dissolved carbon in the former.

In areas with dense riparian vegetation, the vegetation helps to stabilize banks thus preventing erosion (Clary & 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).

Unrestricted grazing of cows as well as their frequent access to the rivers also increases nutrient input when they defecate and re-suspend deposited sediments and nutrients when they trample on riverbanks and riverbeds (Conroy *et al*., 2016; Terry *et al*., 2014). Some sediments and nutrient find their way into streams through surface runoff from the surrounding grazing areas (Koci *et al*., 2020) especially when the riparian zone is bare land (Grudzinski *et al*., 2020).

Urbanization causes increased nutrients and sediment loads in streams through increased runoff from the impervious areas like pavements, roof tops and parking lots (Zhang *et al*., 2012). Impervious areas have low percolation and higher speed of runoff. Nitrate concentrations have been found to be high in urban water discharging into rivers and this is mainly attributed to wastewater discharge and runoff from non-point sources (Krause *et al*., 2008). Hydrologic pathways get altered due to impervious surfaces that bypass nitrogen retention hotspots like riparian zones thus increasing nitrate loading (Newcomer *et al*., 2012). Smith and Kaushal (2012) observed that increased DOC concentrations in urban streams and rivers increase biological uptake and denitrification of nitrogen. Similar observations had been done by Sivirichi *et al*. (2011).

2.6 Impacts of land use on DOC

Sources and concentration of DOC are altered when humans modify terrestrial ecosystems (Lu *et al*., 2014; Yamashita *et al*., 2011). Land use change affects the amount and quality of DOC produced in the soil thus affecting how it is mobilized and exported to rivers and streams (Vaughan *et al*., 2017; Williams *et al*., 2010). To deeply understand DOC dynamics in the face of climate change, it is necessary to understand how it is influenced by both forested and agricultural catchments (Ritson *et al*., 2019). Agricultural land use alters sources of fluvial DOC leading to more production of allochthonous DOC. Studies in small agricultural watersheds often note increases in DOC concentration during storm events (Royer & David, 2005; Vidon *et al*., 2008) because agricultural activities alter the source of stream water DOC leading to greater in-stream DOC production (Wilson & Xenopoulos, 2008). A study by Sebestyen *et al*. (2008) revealed that DOC concentrations in stream water are lowest when nutrient input is highest which means stream DOC level is likely to decrease with high input of fertilizer from runoff. They also stated that when nitrogen inputs increase, the rate at which DOC is released decreases because of increased metabolism and bacterial production though Kalbitz *et al*. (2000) confirmed that the results are not always consistent.

Urban land use has a substantial effect on stream hydrology (Walsh *et al*., 2005) and can result in increased DOC loading in streams (Newcomer *et al*., 2012). Increased DOC loads in urban streams during storms is mainly from wastewater inputs, human and animal waste, grass clippings from home lawns and organic material deposited onto the impervious surfaces (Sickman *et al*., 2007). Most of these DOC sources are not found in the forested areas. In addition, these sources include decomposing organic matter that accumulates in soil layers like in unaltered systems. Increased DOC export into rivers from the urban and suburban areas compared to undeveloped forest lands is attributed to increased hydrologic connectivity and greater drainage density due to gutters, installations of underground pipe networks and ditches (Kaushal & Belt, 2012). However, there is no consensus on the overall effects of urbanization on DOC concentrations and loads (Sickman *et al*., 2007). Recent studies have reported that urbanization causes DOC export into rivers to either increase (Kaushal & Belt, 2012), decrease (Newcomer *et al*., 2002) or provide a compensatory mechanism where internal production balances decrease of external inputs, resulting in no net change (Parr *et al*., 2015).

2.7 Effects of land use on stream respiration rates

Saltarelli *et al*. (2018) stated that with the multiple implications of stream respiration on ecological processes, it is still relatively little known about how it is influenced by land use in tropical streams and rivers. Anthropogenic modification of streams and their riparian zones influences stream respiration thus affecting nutrient retention and transformation as water moves downstream (Houser *et al*., 2005). Human activities like clearing of vegetation cause reduced canopy cover and shading effect while water abstraction causes stream discharge to decrease thus impacting respiration. Stream respiration is usually related to water nutrient concentration, temperature and amount of organic matter present. Human disturbance also causes temperatures, nutrient input and light availability to increase due to decreased shading where riparian vegetation has been cleared (Onyando *et al*., 2013). High nutrient inputs cause stream respiration rates to be high at night especially in urban areas where effluent discharges are received (Halliday *et al*., 2015) and eutrophic river segments where hydrology has been altered (Pinardi *et al*., 2014) or has high organic matter supply conditions like leaf inputs, substrate organic content (Pinardi *et al*., 2011). Restoration of riparian vegetation has a positive effect in driving stream metabolic conditions in the direction of pristine condition, though the effectiveness of this approach is reduced in highly impacted systems (Houser *et al*., 2005)

Stream respiration, which aids in break down and distribution of materials, energy flow in streams, regulation of dissolved organic carbon and organic matter in sediment (Cloern, *et al*., 2014) is influenced by sediments' particle sizes and structure (Santmire & Leff, 2007). Colonization of microorganisms and breakdown of organic matter takes place in the sediments (Vance & Chapin, 2001). Structure and distribution of the sediments is influenced by the land use type from which nutrient and other inputs were derived from (Qu *et al*., 2017). Shading from sparse remnant riparian vegetation influences stream respiration, but that dense riparian canopies are essential to avoid increase in the magnitude of stream metabolism (Ortega-Pieck *et al*., 2017). Conversion of tropical forests to intensive agriculture modifies the magnitude of stream metabolism. Findings of Ortega-Pieck *et al*. (2017) also show that respiration rates increase in streams draining forest-agricultural boundaries mainly due to higher light availability resulting from riparian deforestation. It is noted that the influence of agriculture on stream metabolism increases with increasing proportion of agricultural cover within a watershed (Yates *et al*., 2013). Areas with smaller agricultural land covers have lower rates of stream metabolism as compared to areas of bigger land cover.

Available information on how land use influences nutrients, carbon dynamics and stream metabolism in tropical rivers is limited. The only documented research in River Rupingazi was carried out by Dobson *et al*. (2007) on freshwater crabs distribution and abundance and by Bonareri (2017) on influence of human activities on water quality parameters. This study therefore aimed to determine how land use affects N, P, DOC concentrations and stream metabolism along the longitudinal gradient of River Rupingazi as a knowledge gap of this system.

CHAPTER THREE
MATERIALS AND METHODS

3.1 Study area

3.1.1 Description of the study area

River Rupingazi is a second order stream found in the Mount Kenya water tower in Kenya. The river flows from the peak of Mt. Kenya at an altitude of 4000 metres above sea level and is joined by River Nyamindi before draining into River Tana, the longest river in Kenya. The length of river Rupingazi is approximately 60 km and it flows for around 15 km within the Mount Kenya forest (Dobson *et al*., 2007). Its main tributaries are rivers Kapingazi, Thambana and Nyanjara (Figure 1). However, it is also fed by several other small rivers as it flows downstream. The upper catchment is about 243 km^2. This river is located in Embu County between latitude 0° 22′ and 0° 34′ S and longitude 37° 24′ and 37° 34′ E (Figure 1). Embu County borders Tharaka Nithi County to the North, Machakos County to the South, Kitui County to the East, Kirinyaga County to the West, Meru to the North West and Murang'a County to the South West (Bonareri, 2017).

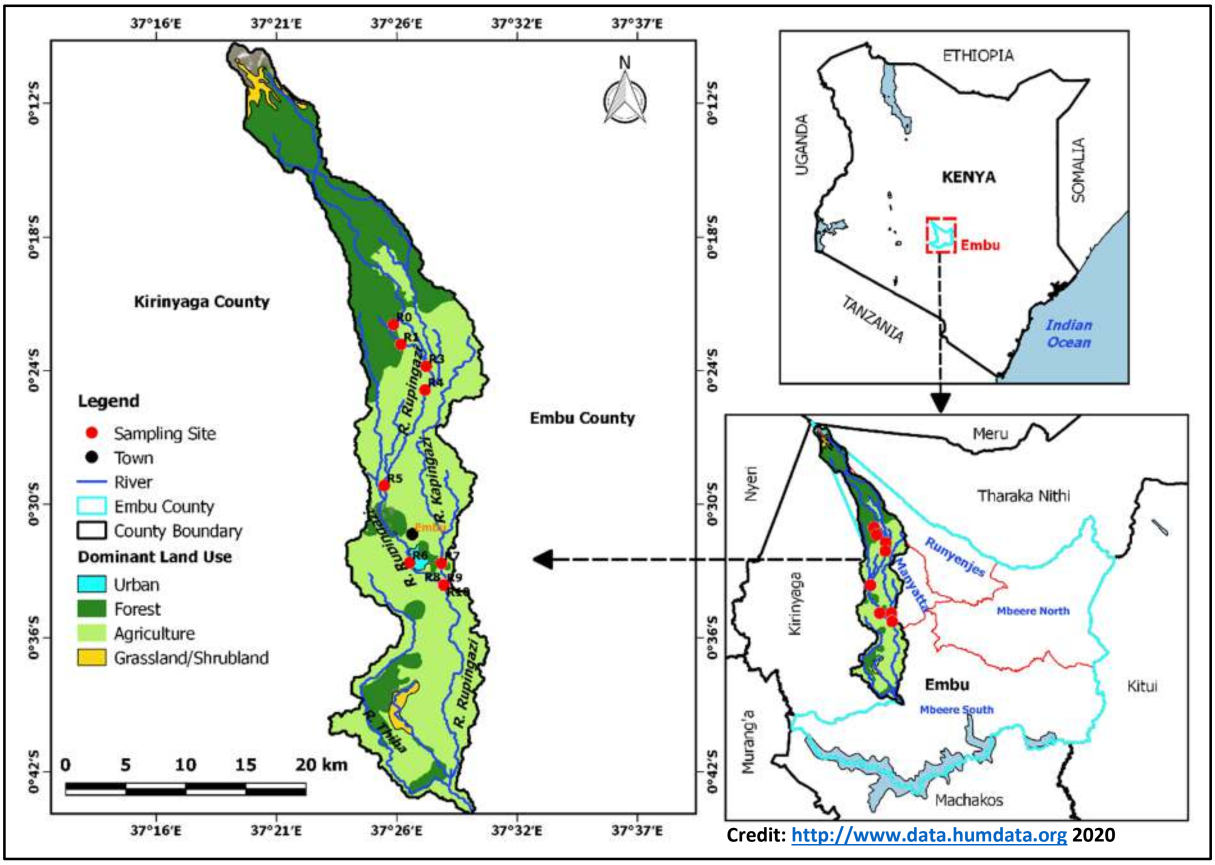

Figure 1: A map showing the sampling sites along River Rupingazi and its tributaries (Source: QGIS)

3.1.2 Climate of the study area

Rupingazi catchment experiences bimodal rainfall with long rain periods between March and May and short rains between June and September. The average rainfall received is around 1067.5 mm ranging between 640 and 1495 mm though it varies according to altitude (Bonareri, 2017). The mean temperature is 21 °C and ranges from 12 °C to 29 °C (Bonareri, 2017). The rainfall and temperatures are projected to change with global warming and climate change with time.

3.1.3 Land use activities in Rupingazi catchment

Generally, there is a variation in land use activities from upstream to downstream of River Rupingazi. The upstream is dominated by indigenous forest characterized by dense network of trees and bushes with little human disturbance as it is situated in a restricted area under the Kenya Forest Service. The Nyayo tea zones demarcate the forest from the agricultural fields mainly tea farms. Upstream riparian vegetation is least disturbed with native vegetation present on both sides of the river, intact canopy and with continuous woody vegetation along the riparian zone, dense ground cover and riverbanks in natural condition. From the edge of the forests towards midstream, the land opens up to a rich upland agricultural area of extensive and intensive farming characterized by tea, coffee and banana plantations and few human settlements. Moving downstream, mixed agricultural farming predominates together with urban land use. There are khat (*Catha edulis*) plantations extending to the riverbanks at the eighth sampling site. Embu town, the largest town in the County, is located 2 km from the most downstream sampling sites.

The main economic activity in the catchment is crop farming as the climate and soils in the area favour growing of cash crops like coffee, tea and macadamia. Maize, beans, cabbage, kales, avocado, tomatoes and french beans are grown on subsistence scales (Chimoita *et al*., 2019). The cash crops are under irrigation as well as some of the subsistence crops. Small scale fisheries and livestock keeping (cows and goats) are also practised. Businesses practised include selling fertilizers, agrochemicals, building materials and foodstuffs (Oguntoye *et al*., 2018). The major soil types in Mount Kenya and the study area mainly: Andosols, Phaeozems and Nitrosols which are volcanic in origin and are very rich in primary minerals, nutrients and montmorillonitic clay minerals. Similarly, most of Embu County has Nitrosols and Andosols soils which are most suitable for agricultural crop farming (Kisaka *et al*., 2013).

3.1.4 Population distribution in Embu County

Kenya National Bureau of Statistics (Kenya National Bureau of Statistics) estimated the population of Embu County to be 516,212 (254,303 males; 261,909 females) from the 2019 census and compared to a population of 370,178 in 2009, it is a big increase. The population density is 183 persons per square kilometre with around 131,638 households in an area of 2,818 km^2 (Chimoita *et al*., 2019). This growing population needs land for farming to produce food and this has seen forest land being converted to agricultural crop lands in the past years. The water of River Rupingazi and its tributaries is used to irrigate the cash crops and some food crops. It is also utilized in the homesteads and in the urban settlements (Bonareri, 2017).

3.1.5 Study sites

This study was conducted at ten sites (Figure 1; Table 1) along River Rupingazi and its main tributaries: Two in the forested area (to act as reference sites), six in the agricultural area and two in the urban area, selected according to dominant land use in the adjacent areas (Table 2). All sampling sites were marked using Global Positioning System (GPS) to make sure that water and sediment samples were collected from the same points throughout the sampling period. The first two sites (Plate 1) in the forested area are characterized by gravel substrate, high channel stability, overhanging vegetation, little bank erosion, riparian width of more than 50 m and they are inside a conserved area with minimal human disturbance. The sites in the midstream are characterized by boulders and slabs, moderate channel stability, moderate bank erosion, riparian width of between 5 to 10 m and moderate human disturbance (Plate 2) Natural vegetation has been replaced with eucalyptus (*Eucalyptus globulus*) trees in most of the sites. The downstream sites are characterized by sand, pools, low channel stability, severe bank erosion and riparian width of less than 5 m (Plate 3). There is less natural and planted vegetation therefore most of the sites are exposed to the sun (Table 2). Site 10, which is the confluence of River Rupingazi and Kapingazi had the widest channel (Table 1).

Table 1: Geographical location and altitude (metres above sea level) of the ten sampled sites

Site	Latitude	Longitude	Altitude (M asl)	Channel width (M)
R0	00°21’56.2’’	037°26’14.8’’	1927	3.5
R1	00°22’49.5’’	037°26’35.0’’	1816	12
R3	00°23’48.4’’	037°27’42.8’’	1732	11
R4	0°24’52.1’’	037°27’40.0’’	1697	13
R5	0°29’9.528’’	037°25’50.1’’	1503	13
R6	00°32’37.1’’	037°26’58.4’’	1260	14
R7	00°32’38.7’’	037°28’24.3’’	1299	6
R8	00°33’36.2’’	037°28’28.4’’	1241	14
R9	00°33’37.6’’	037°28’31.1’’	1242	5
R10	00°33’39.0’’	037°28’30.8’’	1243	16

Table 2: Summary of major land use activities in the selected sampling sites

Site code	Description	Land use and other description
R0	River Kanyuango-tributary	Natural forest
R1	River Rupingazi as it leaves the forest	Natural forest, Nyayo tea zones and settlement on the opposite side
R3	Rupingazi-Nyanjara confluence	Sh[illegible] planted forest (eucalyptus), nappier grass plantation, small-scale farming (maize, vegetables) and settlement
R4	Rupingazi-Thambana confluence	Nappier grass plantations, coffee farming, small-scale farming (maize, tea, bananas), planted forest (eucalyptus) and settlements
R5	Ndunda camp falls- a tourist attraction site with a waterfall and board walk	Nappier grass plantations, small-scale farming (maize, tea, bananas, vegetables), planted forest (eucalyptus), few settlements, ziplining and ca[illegible]te
R6	River Rupingazi in Embu town	Hotel business, little nappier grass, small-scale farming (maize, tea, banana), livestock keeping (cows, goats, pigs), planted forest (eucalyptus) residential houses, slum- next to the river and clothes washing in the river
R7	River Kapingazi (tributary) after town	Small-scale agriculture (vegetables), nappier grass, settlements, hotel business, tree nursery and water abstraction
R8	Rupingazi before confluence with Kapingazi	Settlement, livestock keeping (zero grazed cows and pigs), khat (mugoka) farming and small-scale farming (maize, beans, bananas)
R9	Kapingazi before confluence with Rupingazi	Settlement, livestock keeping (zero grazed cows and pigs), khat farming and small-scale farming (maize, beans, bananas, guavas)
R10	Rupingazi-Kapingazi confluence	Small scale farming (maize and vegetables) and water abstraction

Plate 1: Physical appearance of sites R0, R1, R3 and R4

Plate 2: Physical appearance of sites R5, R6, R7 and R8

Plate 3: Physical appearance of sites R9 and R10. Picture of R9 was taken on a stormy day

3.2 Study design

This study had both experimental design and social inquiry. For the experimental design, water samples for nutrient and DOC analysis were taken on a fortnight basis between November 2020 and January 2021 while sediment samples for metabolism experiment were taken once for five consecutive days in January 2021. A total of 120 water samples, 12 per site, for nutrients and DOC analysis were collected from the selected points during the entire sampling period. Additionally, a total of 100 sediment samples, two samples per site per day, were collected and incubated for measurement of oxygen consumption rate during the five days. A total of 50 water samples to act as control, 1 sample per site per day, were collected from every sampling station during the entire sampling period. For social inquiry, sixty questionnaires were used to collect information about agricultural activities carried out in the land bordering the river and also the land management practices in place.

3.2.1 Collection of samples

Water samples for nutrients and DOC analysis were collected according to APHA, (2005). Physicochemical parameters that is 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). Stream width, depth and discharge were measured using a tape measure, and flow meter (OTT MF Pro- OTT Hydrometer) respectively. This was done once at every site during a sampling session for the purpose of calculating the stream discharge using equation 1:

$$\text{Discharge (Q) } m^3/s = \text{Cross sectional area } (m^2) \times \text{Mean velocity (m/s)} \quad (1)$$

Where cross sectional area = stream width × stream depth

Water samples for nutrients and DOC concentration analysis were collected from the ten sampling sites in triplicates using 500 ml acid washed (10% H_2SO_4) plastic bottles, preserved in cooler boxes to minimize biological activity (APHA, 2005), then transported to the laboratory for analysis. Sediment samples for the respiration experiment were collected in triplicates using 50 ml falcon tubes from a 3 cm depth from sites of deposition while avoiding fine sediment fractions rich in clay and organic content. The most dominant land use activity at each sampling site (range of 100 metres) were recorded.

3.2.2 Questionnaire survey

A semi-structured questionnaire survey was done and responses were obtained face to face from the respondents. The questionnaires were first validated through face validity and a pilot test conducted during the field pre-visit. Fifteen questionnaires were randomly administered in the upstream section, thirty in the midstream and fifteen in the downstream of River Rupingazi. The target audience were crop farmers and livestock keepers whose land is closest to the river. Details of the questionnaire included size of the farm, type of crop grown, chemical usage, type of fertilizer used, time and frequency of application and farm management practices to prevent soil erosion (Appendix 1).

3.3 Laboratory analysis

Analyses of water samples for nutrients (nitrogen and phosphorus forms) and DOC were done according to APHA (2005). Water samples were filtered upon arrival in the laboratory using Whatman GF/F filters of 7 µm pore size with a 47 mm diameter, dried at 95°C for 24 hours. Standard calibration curves were prepared for all the nutrients.

3.3.1 Determination of different forms of Nitrogen

The forms of nitrogen which were determined are: Ammonium, nitrites, nitrates and TN. Ammonium-Nitrogen was determined using the hypochlorite method through 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 Sulfanilamid solution to 25 ml of filtered water sample. After 2-8 minutes 1 ml of N-Naphthyl-(1)-ethylendiamine-dihydrochloride solution was added to this mixture and gently mixed. The solution was left standing for 10 minutes after which absorbance was read from the spectrophotometer at a wavelength of 543 nm.

Nitrate-Nitrogen was determined using 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 added. The bottles were put in the oven and samples dried at a temperature of 95 °C. The resulting residue were 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-tartarate solution (prepared by dissolving 400 g NaOH in 1 litre distilled water and adding 50g K-Na-Tartarate) was added, mixed and absorbance read at a wavelength of 420 nm. Total nitrogen (TN) was determined through persulphate digestion by adding 1 ml of warm potassium persulphate 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 were analyzed using sodium-salicylate method. Concentrations of NH_4-N, NO_2-N, NO_3-N and TN were calculated from their respective equations generated from the standard calibration curves (APHA, 2005).

3.3 2 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 were mixed in ratios 2:5:2:1 and the resulting solution added to 25 ml filtered water sample in a ratio of 1:10. The prepared samples absorbance was read using a GENESIS 10uv scanning 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 ortho-phosphate using persulphate digestion. After digestion, the total reduced forms were analysed using the same procedure as for SRP. 12 g of potassium persulphate were dissolved in 100 ml of distilled water and 1 ml added to the 25 ml sample while still warm. The bottles were weighed without lids and their weights noted. The covers were put back but not closed tightly after which they were autoclaved for 90 minutes at about 120 °C and 1.2 atm. After cooling, the bottles were re-weighed and the evaporated water replaced by 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).

3.3.3 Total Suspended Solids (TSS)

TSS was estimated using gravimetric method by filtering a known volume of water samples through a pre weighed Whatman GF/C filter papers of 0.45 µm. The filter papers were then dried to constant weight for 3 hours at 95 °C.TSS weight was calculated using equation 2 (APHA, 2005).

$$TSS= ((Wc-Wf) \times 10^{6})\ V^{-1} \qquad (2)$$

Where TSS=Total Suspended Solids (mg/L), Wf=Weight of dried filter paper (g) and Wc= Constant weight of filter paper + residue in grams and V=Volume of water filtered (ml).

3.3.4 Nutrient loading rates

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

$$\text{Nutrient loading} = \text{Discharge} \times \text{nutrient concentration} \times 0.0864 \qquad (3)$$

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

3.3.5 Dissolved Organic Carbon

The filtered water samples were poured into quartz cuvettes and subjected to spectrophotometric readings using a Genesys 10 uv scanning spectrophotometer at 254 nm upon arrival at the laboratory and absorbances were recorded (APHA, 2005). This was done within 48 hours of the water sampling as guided by APHA (2005). DOC concentration in mg/L was estimated using the standard calibration equation modified from Ngari (2020) which had been prepared using the same spectrophotometer at the Egerton University laboratory in addition to Shimadzu TOC-L (Total Organic Carbon Analyzer) at Wassercluster Lunz am see. The equation used was:

$$DOC\ (mg/L) = (0.1604 \times UV254) - 0.1102 \quad (4)$$

Where UV254 is the absorbance reading obtained from the spectrophotometer at 254 nm.

3.3.6 Respiration measurements in sediments

Stream respiration experiments were done in the ten sites whereby for every site, two sediment samples (two replicates) and one insitu water sample were taken using 50 ml incubation tubes (Figure 2). The incubation tubes fitted with sensor spots (PreSens oxygen sensor spots) on the inner walls were weighed and their weight recorded (as initial weight) before filling them with sediments until there was no space left. The tubes were then closed airtight and shaken gently. The initial temperature and oxygen concentration within the chamber were recorded and the tubes were incubated in the dark at room temperature. Thereafter, temperature and dissolved oxygen concentration were measured after every sixty minutes using a PreSens polymer optical fibre (Fibox 4). After the 4-hour incubation period, the tubes were weighed again as final weight. Stream respiration rates were estimated from changes in DO concentration (Odum, 1956) in the incubation tubes over the 4-hour period using the equations below. If DO concentrations dropped lower than 2 mg, the experiment was ended for that particular incubation tube and the figures omitted in the calculations to eliminate the risk of a non-linear relationship due to limitation effects.

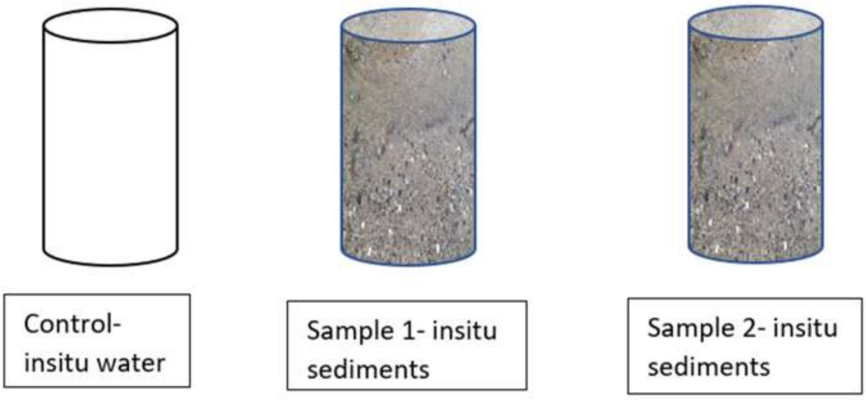

Figure 2: Experimental set up for respiration for every sampling site

Change of Oxygen concentration in the incubation tubes was approximated using the following equation as described by Odum (1956):

$$\Delta c=(Ci-Ct)/\Delta t \quad (5)$$

Where Δc change in Oxygen concentration (mg O_2/h), *Ci is* the initial oxygen, *Ct* is the final oxygen concentration, Δt is the change in time (duration of incubation) in hours.

3.3.7 Organic matter (OM) in sediments

Sediment samples collected from the field were sub sampled into pre-weighed crucibles and the weight of the crucible plus sediments taken and recorded. The initial weight of the sediment sub sample was calculated as the difference between the weight of the crucibles plus sediments and the weight of the pre-weighed crucible. The crucibles with sediment samples were then placed in the oven for 24 hours at 75°C to dry after which they were weighed again and placed in a muffle furnace for 2hrs at 500°C to remove the organic matter content. The muffle furnace was switched off and left to cool the combusted samples for twelve hours. The crucibles with the cooled samples were weighed ensuring samples did not absorb atmospheric moisture. The organic matter content was calculated as percentage weight loss using the following equation:

$$\text{OM (g)} = ((W1-W2)/W1) \times 100 \quad (6)$$

where OM is organic matter, W1 is weight of crucible and sediments after drying in the oven while W2 is the weight of crucible and sediments after combusting in the muffle furnace.

3.4 Data analysis

The data was organized using Microsoft excel. All variables physicochemical parameters, discharge, stream respiration rates, nutrient and DOC concentrations and loadings was checked for normality using Shapiro Wilks test and found to be normally distributed ($p>0.05$) thus parametric tests were done. One-way analysis of variance (ANOVA) was used to determine the difference in mean concentrations of N, P and DOC along the land use gradient while linear regression analysis was used to determine the influence of land use on nutrients and DOC. One-way analysis of variance (ANOVA) was used to determine the differences in mean respiration rate between the forested, agricultural and urban stream reaches. Tukey HSD test was done for the significantly different results to find out which site or reach was different from the other. PCA analysis was done to show the relationship between selected parameters for different sites in River Rupingazi. Regression analysis was also performed to show how various parameters changed with change in others. Statistical data analysis was done using IBM SPSS version 25 and R version 4.0.0 with all statistical tests performed at a significance level (α) of 0.05. Data from the questionnaire was computed to establish the major land use types, kind of fertilizer and chemicals used, mode of application, how often they are used and the various land management practices in place to prevent soil erosion. This was used to help explain and further discuss the results of physicochemical parameters, nutrient concentrations and loading, DOC concentrations and loading, discharge and stream metabolism rates.

CHAPTER FOUR

RESULTS

4.1 Variation of physicochemical parameters along the longitudinal continuum

Values of physicochemical parameters, nutrients, DOC concentrations and organic matter for River Rupingazi showed significant variations (ANOVA, $p<0.05$) among the sampled sites except for TN and pH.

4.1.1 Variation of insitu measurements in River Rupingazi and its tributaries

Mean water temperature was 18.32 ± 1.97 °C and ranged between 15.07 ± 0.79 °C in site R1 upstream and 20.76 ± 0.93 °C in site R9 downstream (Table 3). In the main river, Rupingazi, the highest temperature value was 19.59 ± 0.27 at site R10. Temperature values differed significantly across the sites (ANOVA, $p<0.05$) with mean temperature of site R1 being significantly lower than all other sites (Tukey HSD, $p<0.05$) except site R0 (Tukey HSD, $p>0.05$). Mean DO concentration was 8.09 ± 0.45 mg/L. The lowest concentration was recorded at site R9 with a value of 7.72 ± 0.77 mg/L while the highest was recorded at site R1 with value of 8.47 ± 0.27 mg/L (Figure 3). In the main river, Rupingazi, the lowest value for DO was 7.84 ± 0.64 mg/L at site R10 (confluence with river Kapingazi). DO values differed significantly among the sites (ANOVA, $p<0.05$). Mean DO concentration at site R1 was significantly higher than sites R0, R7, R9 and R10 (Tukey HSD, $p<0.05$).

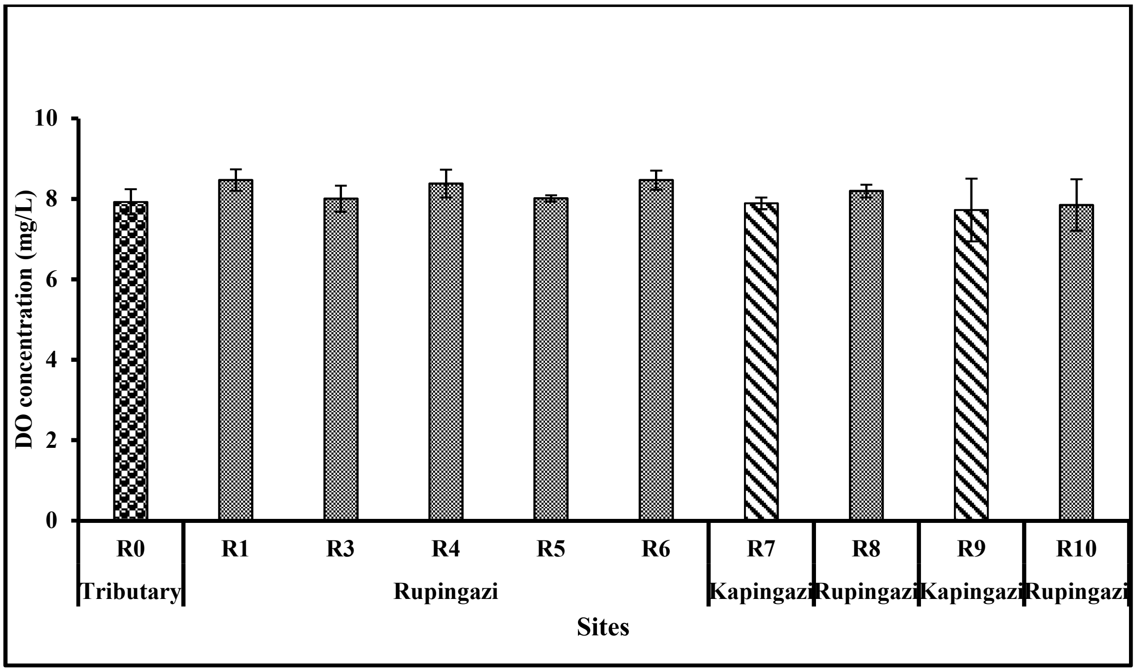

Figure 3: Variation of temperature and DO in a) River Rupingazi (small dots), b) Kanyuango-tributary (big dots) and c) Kapingazi-tributary (striped) (n=12).

pH values ranged between 6.61 at site R8 and 8.41 at site R4 (Table 3). There were no significant variations in pH among the 10 sites (ANOVA, $p<0.05$). pH values were within the neutral range of 7.0 to 7.9 except for site R0 and R4 during the second sampling done on 10th December 2020. Mean EC was 48.32 ± 12.05. EC values ranged between 28.93 ± 7.39 in site R1 and 62.68 ± 4.73 in site R9 (Table 3). In the main river, River Rupingazi, the highest EC recorded was 57.52 ± 7.38 at site R10. Generally, River Kapingazi sites had higher values compared to the main river. EC values differed significantly among the sites (ANOVA, $p<0.05$) with site R1 values being significantly lower than all other sites (Tukey HSD, $p<0.05$) except site R0 (Tukey HSD, $p>0.05$).

Turbidity and TDS showed a similar trend of increasing downstream. Mean turbidity was 29.98 ± 35.72 NTU. Turbidity was lowest in site R2 (4.31 ± 2.04 NTU) and highest in site R9 (78.2 ± 64.53 NTU) before the confluence. In the main river, Rupingazi, turbidity was highest at the confluence with River Kapingazi (50.08 ± 24.69 NTU). Mean TDS was 22.93 ± 5.374 mg/L.TDS values ranged between 14.86 ± 1.67 mg/L in site R0 and 29.87 ± 1.88 mg/L in site R9 (Table 3). In the main river, Rupingazi, the lowest TDS was 15.79 ± 1.89 mg/L at site R1 and the highest was 27.08 ± 3.64 mg/L at the confluence with River Kapingazi.

Table 3: Mean ± S.D values of physical-chemical variables at all the sampled sites (n=12)

Site	Temperature (°C)	DO (mg/L)	pH range	EC (µs/cm)	TDS (mg/L)	Turbidity (NTU)
R0	16.05 ± 1.12	7.92 ± 0.31	6.97 - 8.15	31.98 ± 4.06	14.86 ± 1.68	6.97 ± 0.87
R1	15.07 ± 0.79	8.47 ± 0.27	6.87 - 7.56	28.93 ± 7.39	15.79 ± 1.89	4.31 ± 2.04
R3	17.50 ± 1.34	8.00 ± 0.33	6.96 - 7.51	45.45 ± 5.22	21.14 ±2.52	5.83 ± 0.99
R4	17.51 ± 1.52	8.38 ± 0.35	6.81 - 8.41	46.19 ± 1.76	21.78 ± 0.75	9.82 ± 2.19
R5	18.93 ± 0.51	8.01 ± 0.08	6.65 - 7.52	46.59 ± 5.64	21.86 ± 2.65	22.78 ± 6.79
R6	18.33 ± 0.94	8.46 ± 0.24	6.78 - 7.40	49.32 ± 7.12	22.99 ± 3.46	27.78 ± 7.87
R7	20.40 ± 0.97	7.89 ± 0.15	6.69 - 7.56	58.34 ± 5.59	27.57 ± 2.92	68.71 ± 42.07
R8	19.08 ± 0.53	8.19 ± 0.16	6.61 - 7.52	56.23 ± 7.92	26.34 ± 3.83	25.29 ± 8.05
R9	20.77 ± 0.93	7.72 ± 0.78	6.90 - 7.61	62.68 ± 4.73	29.87 ± 1.88	78.20 ± 64.53
R10	19.59 ± 0.26	7.84 ± 0.64	7.27 - 7.83	57.52 ± 7.38	27.08 ± 3.64	50.08 ± 24.69

4.1.2 Relationship between selected physicochemical parameters

From the linear regression analysis, there was a significantly strong positive relationship between DO and temperature (R^2=0.31, $p<0.05$) and conductivity and TDS, (R^2=0.93, $p<0.05$) (Figure 4a and 4b). A significant positive relationship was also observed between TSS and turbidity (R^2=0.66, $p<0.05$) (Figure 4c). This means that temperature accounts for only 31% variations in DO. 93% of the changes in conductivity are attributed to TDS and 66% of the variations in turbidity are caused by TSS in the catchment.

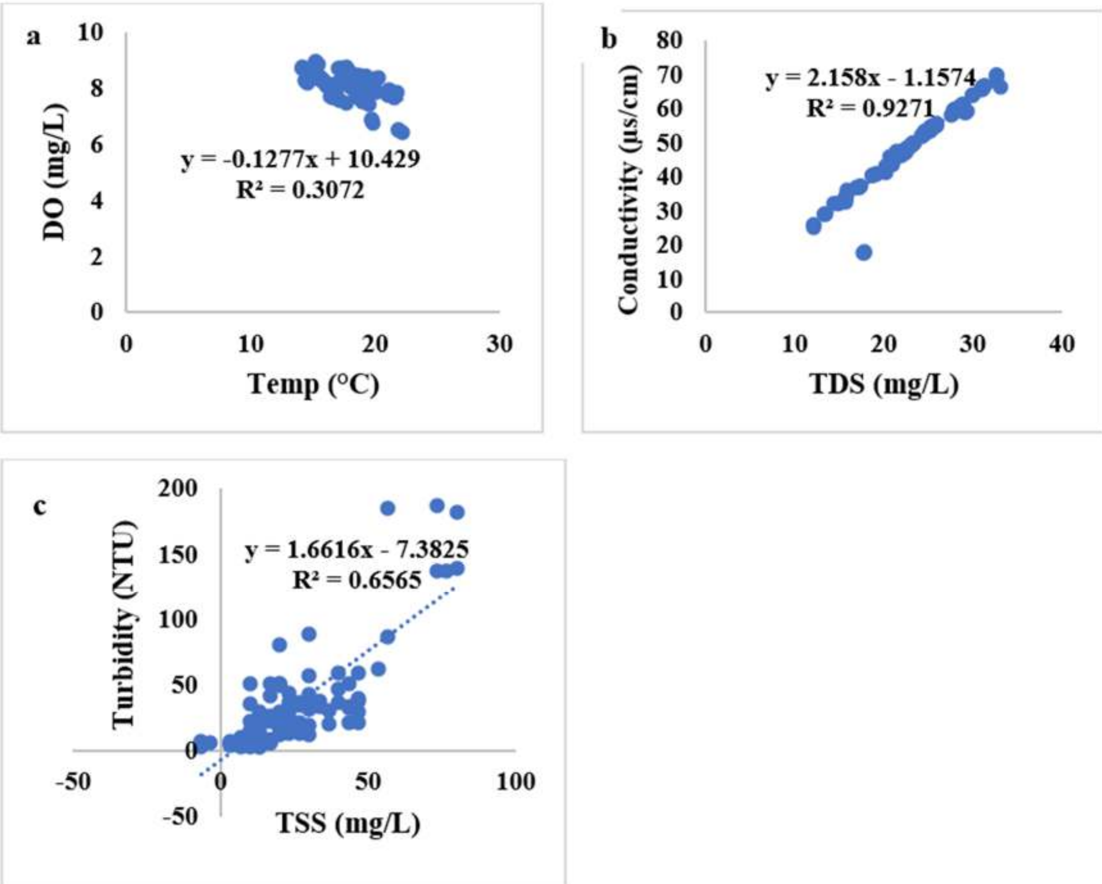

Figure 4: Regression analysis results for a) DO and temperature, b) Conductivity and TDS and c) Turbidity and TSS

4.1.3 Variation of nutrient concentrations, TSS and OM along the longitudinal continuum

There were notable variations in nutrient concentrations along the longitudinal continuum of river Rupingazi. All nutrients showed significant differences in the different sites except for Total Nitrogen which had no significant variations (ANOVA, $p<0.05$). Mean NH_4-N concentration was 0.04 ± 0.03 mg/L. The concentrations ranged between 0.02 ± 0.02 mg/L at site R6 and 0.09 ± 0.04 mg/L in site R8 (Figure 5). In the main river, Rupingazi, NH_4-N decreased from site R1 to site R6 then rose in site R8 and R10 at the confluence. Concentrations were significantly different among the sampled sites (ANOVA, $p<0.05$). Post hoc results (Tukey HSD, $p<0.05$) revealed that the concentration at site R1 was significantly lower than sites R8 and R10.

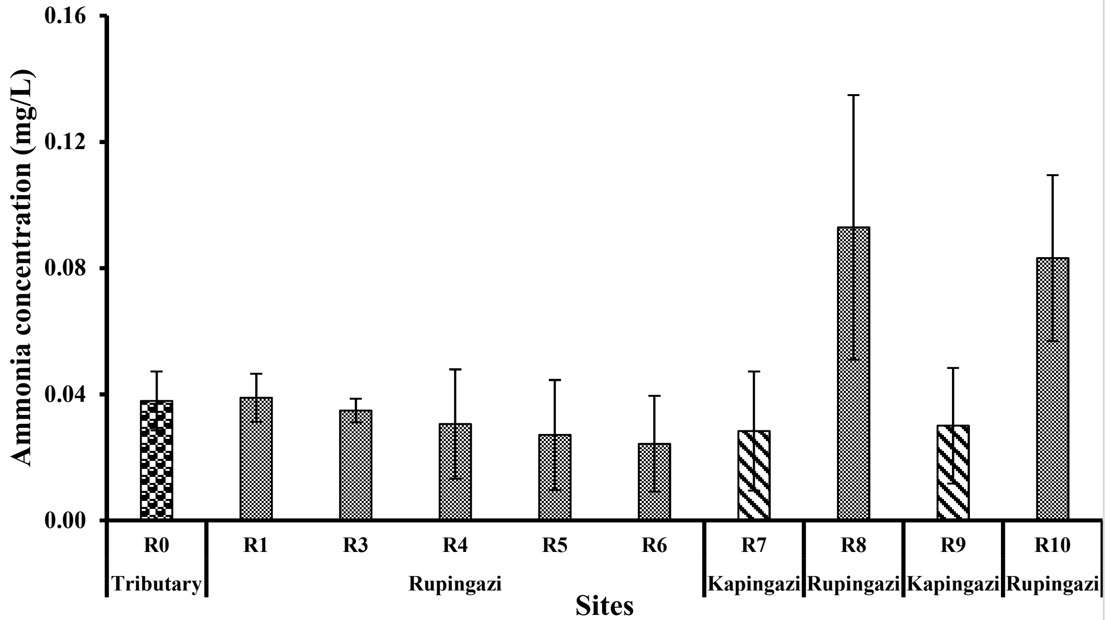

Figure 5: Variation of NH_4-N concentrations in a) River Rupingazi (small dots), b) Kanyuango-tributary (big dots) and c) Kapingazi-tributary (striped) (n=12).

Mean NO_2-N concentration was 0.002 ± 0.002 (mg/L). NO_2-N concentration ranged from 0.001 ± 0.001 mg/L (site R3) and 0.0042 ± 0.001 mg/L (site R10) as shown in Figure 6. Generally, the concentrations increased along the longitudinal continuum of the river. NO_2-N concentrations were significantly different among the ten sites (ANOVA, $p<0.05$). Post hoc results (Tukey HSD, $p<0.05$) revealed that R1 concentration was significantly lower than for sites R8, R9 and R10.

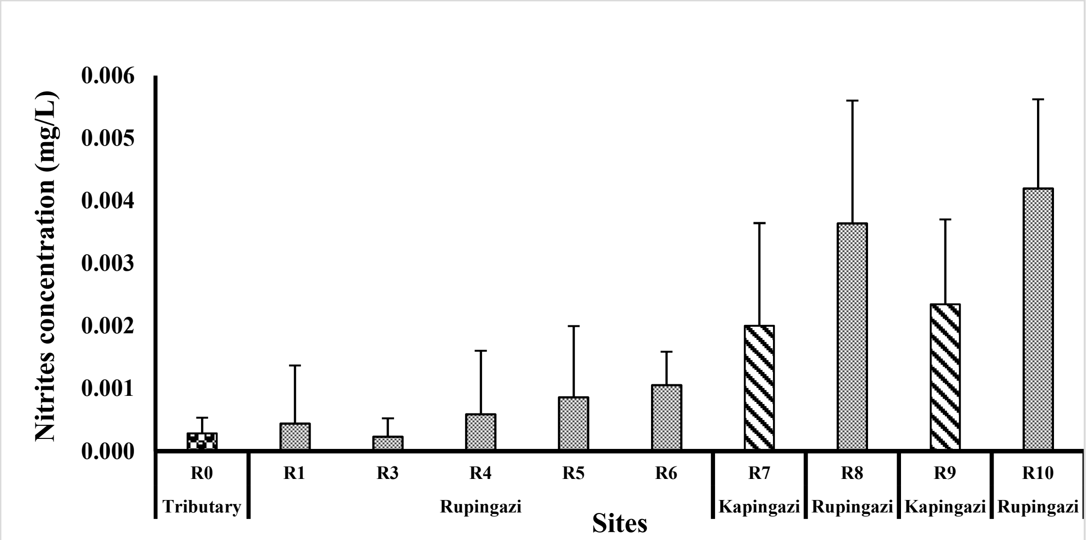

Figure 6: Variation of NO_2-N concentrations in a) River Rupingazi (small dots), b) Kanyuango-tributary (big dots) and c) Kapingazi-tributary (striped) (n=12).

The mean NO_3-N concentration was 0.53 mg/L. The highest NO_3-N concentration was 1.05 ± 0.17 mg/L at site R9 and the lowest concentration was 0.11 ± 0.05 at site R1 (Figure 7). In the main river, Rupingazi, the highest concentration was 0.74 ± 0.26 at site R8. There were significant differences (ANOVA, $p<0.05$) among all the ten sites and post hoc results revealed that concentration at site R0 was significantly different from those of sites R1, R7, R8 and R9. In addition, concentrations at site R1 concentrations were significantly lower than all other sites except site R3 and R5 (Tukey HSD, $p<0.05$).

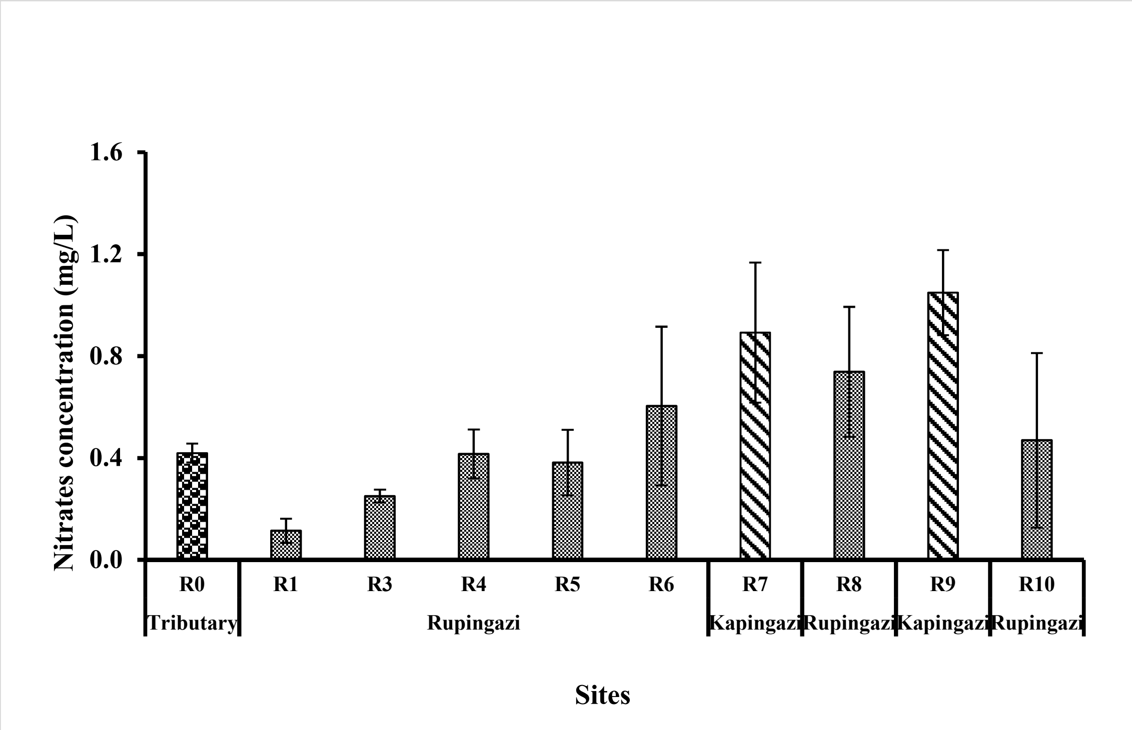

Figure 7: Variation of nitrates concentrations at a) River Rupingazi (small dots), b) Kanyuango-tributary (big dots) and c) Kapingazi-tributary (striped) (n=12)

Mean TN concentration was 2.48 ± 2.15 mg/L. In the main river, Rupingazi, the lowest TN concentration was 1.94 ± 0.88 mg/L at site R8 while the highest was 3.05 ± 2.23 mg/L at site R6 (Figure 8). Site R9 had the lowest TN concentration (1.77 ± 1.00 mg/L) among the sampled sites. There were no significant differences in TN concentrations among the ten sites (ANOVA, $p>0.05$).

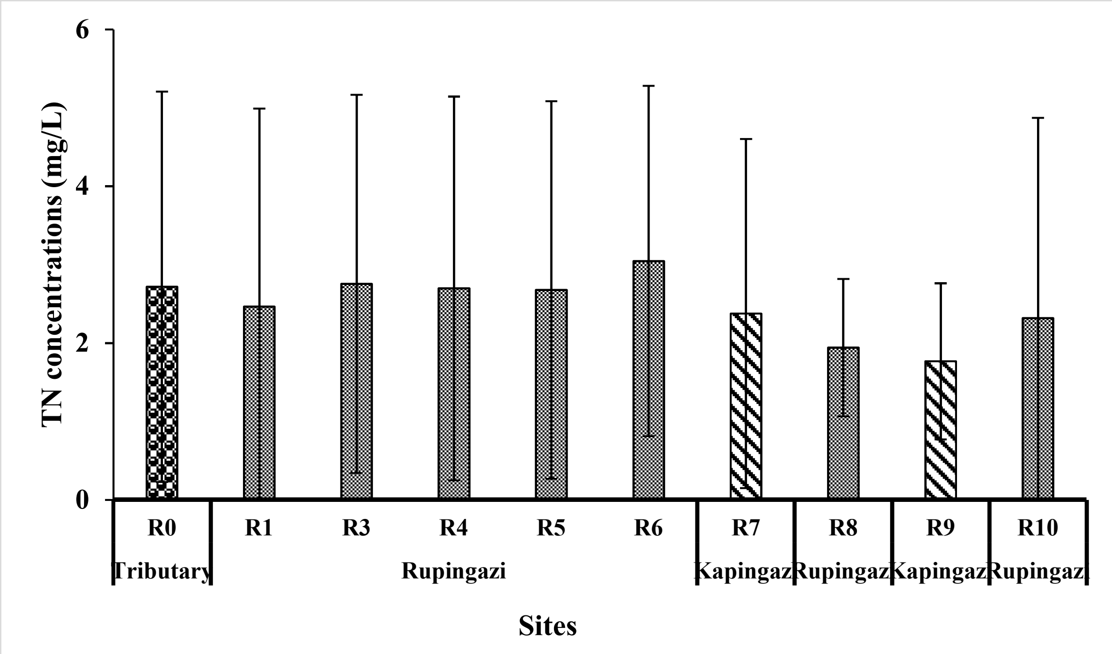

Figure 8: Variation of TN concentrations in a) River Rupingazi (small dots), b) Kanyuango-tributary (big dots) and c) Kapingazi-tributary (striped) (n=12)

Mean SRP was 0.008 ± 0.006 mg/L. SRP concentrations ranged from 0.003 ± 0.003 mg/L at site R6 and 0.014 ± 0.007 mg/L at site R1 (Figure 9). There were significant differences among the ten sites (ANOVA, $p<0.05$). Post hoc test results (Tukey HSD, $p<0.05$) revealed that site R1 had significantly higher concentrations than all other sites except site R8 and R10.

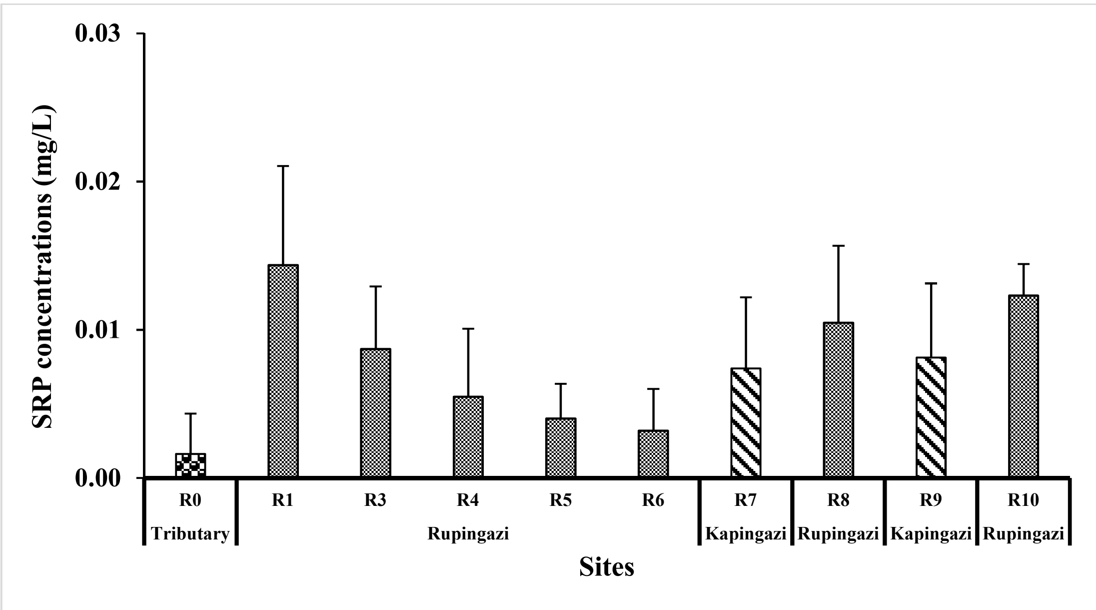

Figure 9: Variation of SRP concentrations in a) River Rupingazi (small dots), b) Kanyuango-tributary (big dots) and c) Kapingazi-tributary (striped) (n=12)

Mean TP was 0.06 ± 0.05 mg/L. Concentration of TP ranged from 0.04 ± 0.01 mg/L at site R3 and 0.10 ± 0.04 mg/L at site R10 (Figure 10). Generally, the concentrations increased along the longitudinal continuum. There were significant differences in TP concentrations among the sites (ANOVA, $p<0.05$) with that of site R1 being significantly lower than those of sites R9 and R10 (Tukey HSD, $p<0.05$).

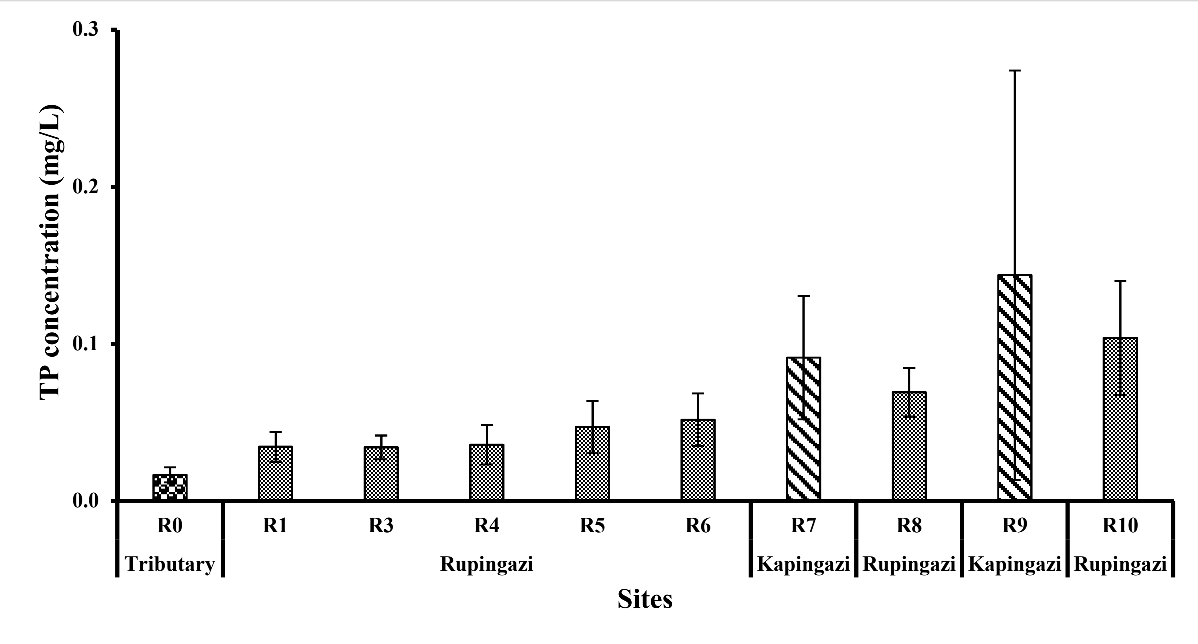

Figure 10: Variation of TP concentrations in a) River Rupingazi (small dots), b) Kanyuango-tributary (big dots) and c) Kapingazi-tributary (striped) (n=12)

The mean TSS concentration was 22.94 ± 17.13 mg/L and the values showed an increasing trend downstream with the lowest concentrations recorded at R0 (7.7 ± 3.67 mg/L) in the upstream and highest at R9 (40.56 ± 20.83 mg/L) in the downstream. The concentrations varied significantly among the ten sites with R9 being having significantly higher concentrations than sites R0, R1, R2 and R3 (Tukey HSD, $p<0.05$). The sites on River Kapingazi (R7 and R9) had the highest TSS concentrations: 36.67 ± 24.82 mg/L and 40.56 ± 20.83 mg/L respectively (Figure 11).

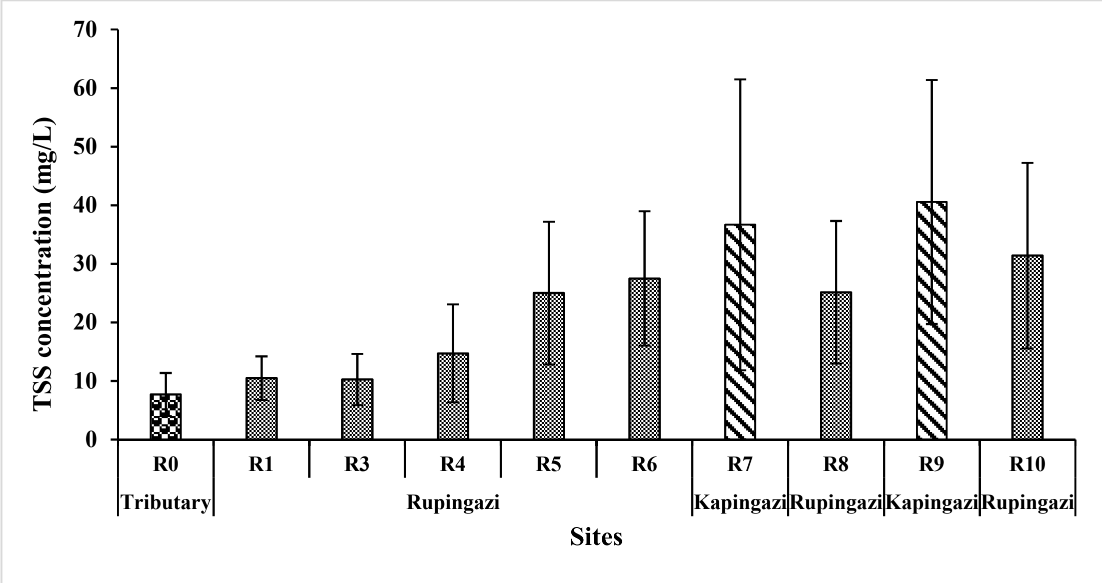

Figure 11: Variation of TSS concentrations in a) River Rupingazi (small dots), b) Kanyuango-tributary (big dots) and c) Kapingazi-tributary (striped) (n=12)

The amount of OM in the sediments was highest at site R0 (11.30 ± 3.31 g) and lowest at R3 with 4.28 ± 1.54 g). There were significant variations in amount of organic matter among the sampled sites (ANOVA, $p<0.05$) whereby OM in site R0 was significantly higher than in all the other sites (Tukey HSD, $p<0.05$). The sites in the agricultural areas had higher amounts of OM compared to those in urban sites (Figure 12).

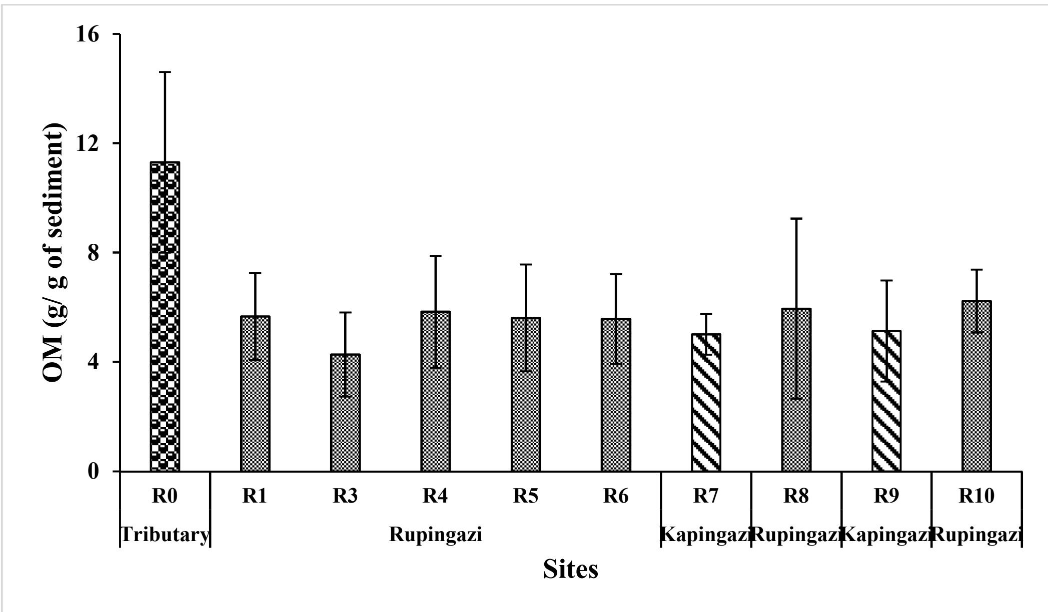

Figure 12: Variation of OM amount in a) River Rupingazi (small dots), b) Kanyuango-tributary (big dots) and c) Kapingazi-tributary (striped) (n=12)

4.1.4 Interactions between nutrients and selected parameters

Regression analysis was performed to predict NO_3 from temperature and DO (ANOVA, $F=16.531$, $p<0.005$). The results showed that the temperature was statistically significant in predicting changes in NO_3 while DO was not. Temperature accounts for 34% ($R^2=0.34$) of the variations in NO_3 while DO accounts for 4% ($R^2=0.04$) of variations of NO_3 concentrations (Figure 13a and 13b).

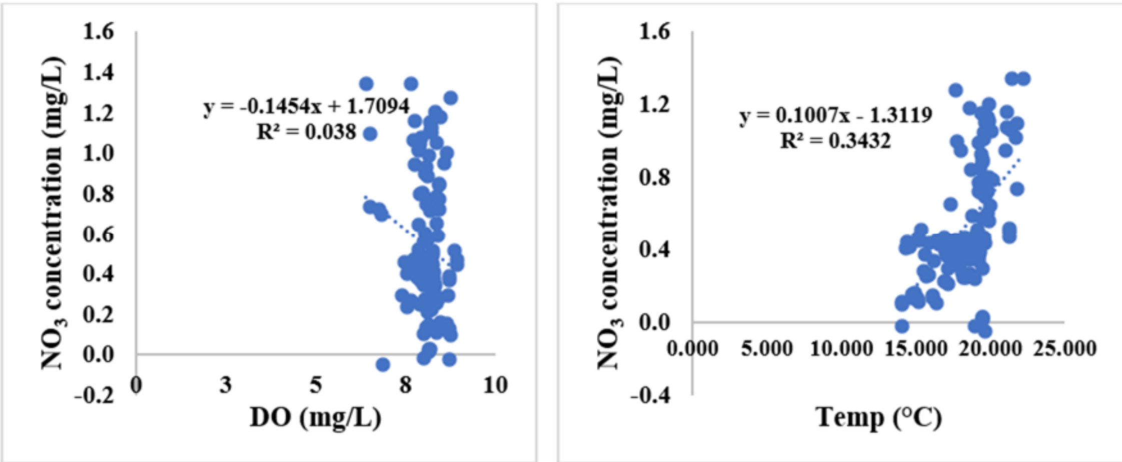

Figure 13: Relationship between a) NO_3 concentration and temperature and b) NO_3 concentration and DO

Figure 14 shows that TSS had a positive relationship with both TN and TP though not significant. The findings gave a R^2 of 0.314 (Figure 14b) which means that 31% of variations in TP were attributed to TSS in the catchment. The R^2 for TN was 0.001 (Figure 14a) which means TSS accounts for only 0.1% of TN variations.

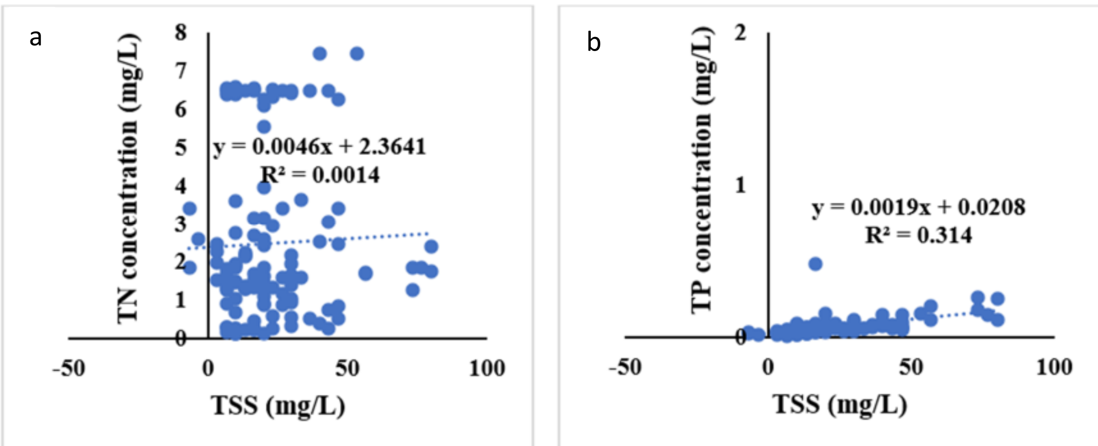

Figure 14: Regression analysis showing the relationship between a) TN concentrations and TSS and b) TP concentrations and TSS

4.1.5 Variation of discharge, TN and TP loading rates

Mean discharge was 2.50 ± 1.42 m^3/s. The lowest discharge was recorded at site R0 with 0.25 ± 0.12 m^3/s and highest was at site R10 (confluence with river Kapingazi) with 4.40 ± 0.68 m^3/s (Figure 15a). In the main river, Rupingazi, mean discharge ranged between 1.85 ± 0.93 m^3/s at site R1 and 4.40 ± 0.68 m^3/s at site R10. The sites in the sampled tributaries: Kanyuango and Kapingazi, had lower discharge compared to sites in the main river. Generally, discharge increased downstream. There were significant differences in discharge among the sampled sites (ANOVA, $p<0.05$) with that of site R1 being significantly lower than for sites R6, R8 and R10 (Tukey HSD, $p<0.05$).

Loading rates for TP showed an increasing trend downstream unlike TN which increased in site R3 and R4 then decreased only to rise again in site R6 (Figure 15a and 15b). Sites R7 and R9 had minimal input of TN into the main river, Rupingazi, hence sites R6 and R8 were the main sources of TN downstream. This is contrary TP where all the sites downstream had a significant input of TP downstream. The upstream sites (R3 and R4) together with site R6 in the midstream had higher TN loadings compared to TP. At the confluence of River Rupingazi and Kapingazi, the loading rates for both TN and TP ranged between 8785 to 11291 kg/day and 210 to 458 kg/d respectively.

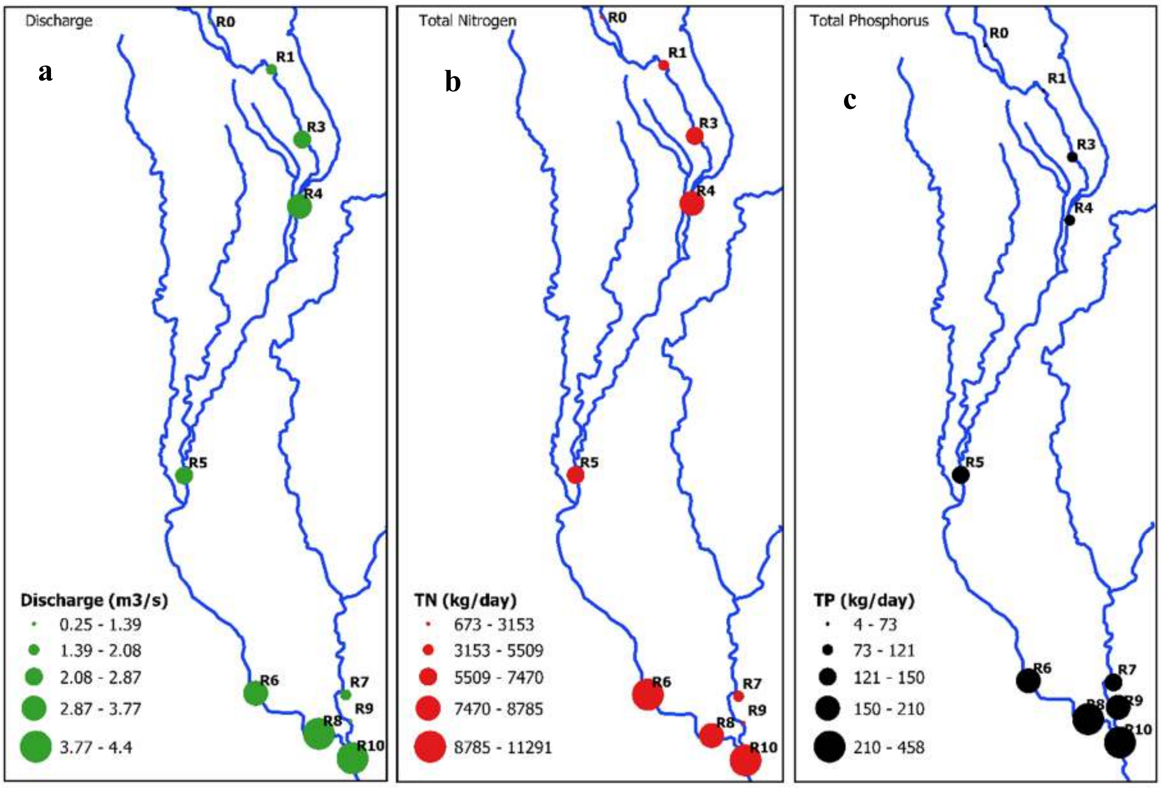

Figure 15: Variation of a) Discharge, b) Total Nitrogen loadings and c) Total Phosphorus loadings along the longitudinal continuum of River Rupingazi

4.1.6 Variation of dissolved organic carbon concentrations along the longitudinal continuum

Mean DOC concentration was 0.60 ± 0.31 mg/L. The lowest DOC concentration recorded was 0.41 ± 0.12 mg/L at site R0 and the highest was 0.85 ± 0.45 mg/L at site R9 (Figure 16). Generally, DOC concentration had an increasing trend along the longitudinal continuum. DOC values varied significantly among the sampled sites (ANOVA, $p<0.05$) whereby R9 concentrations were significantly higher than all other sites except R6, R7, R8 and R10 (Tukey HSD, $p<0.05$).

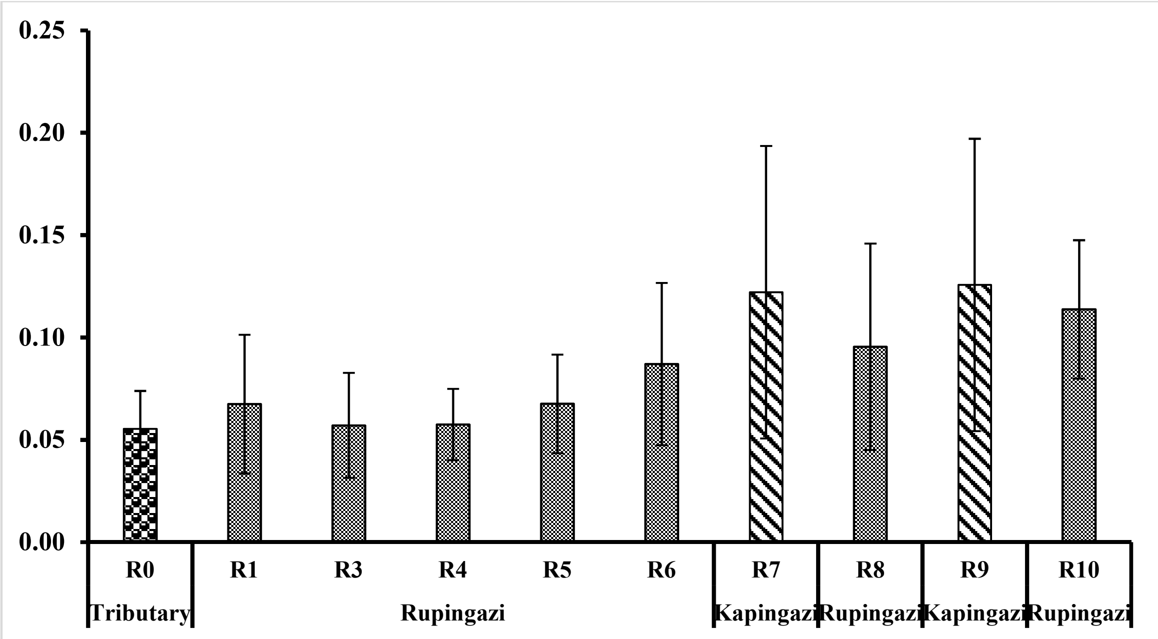

Figure 16: Variation of DOC concentrations in River Rupingazi (small dots) and its tributaries: Kanyuango (big dots) and Kapingazi (striped) (n=12)

4.1.7 Interaction between DOC, physicochemical parameters, nutrients in the sampled sites

PCA analysis was done to identify parameters that accounted for much of the variation in the data for nutrients and in situ parameters. The first three components were extracted which explained 88.7% of the variability in the sampled sites. The findings showed that TDS, TP and DOC (Figure 17) had a high positive loading on the first component with a high negative loading for TN and DO. For the second component, discharge, ammonia and SRP had a high positive loading while turbidity and temperature had a high negative loading for the second component. This means that TDS, TP, DOC, Conductivity, temperature, turbidity and NO_3 are associated with sites R7 and R9 in the downstream. In the second component, the most likely sources of NH_4 and SRP were sites R8 and R10 (agricultural sites) and accounted for most of the variations in the second component.

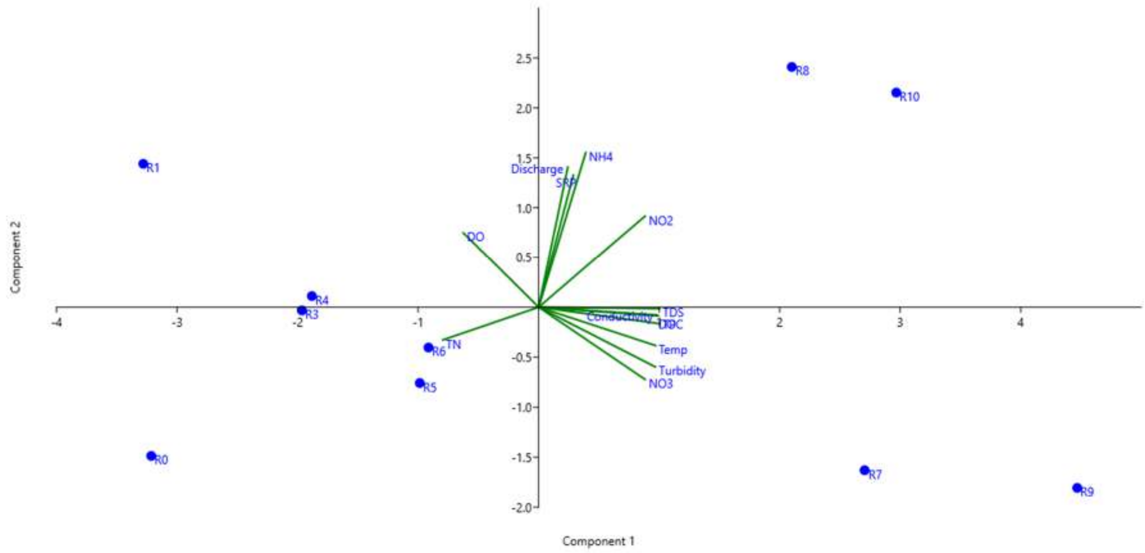

Figure 17: Principal Component Analysis for selected parameters in all the sampled sites

4.2 Variation of physicochemical parameters among the three land use types

There were marked variations in physicochemical parameters among the three land use types (ANOVA, $p<0.05$) (Table 5). However, temperature and pH did not have significant variations.

4.2.1 Changes in insitu parameters along the land use gradient

There were significant differences in temperature among the three land use types (ANOVA, $p<0.05$). It was significantly lower in the forested than in the urban section (Tukey HSD, $p<0.05$). Temperature was highest in the downstream urban section (19.36 ± 1.41 °C) < agricultural section (18.89 ± 1.48 °C) < upstream forested section (15.56 ± 1.07 °C). There were no significant differences in DO among the three land use types (ANOVA, $p>0.05$). DO concentration was highest in the forest (8.19 ± 0.39 mg/L) > urban section (8.18 ± 0.35 mg/L) > agricultural section (8.03 ± 0.49 mg/L) (Table 4).

pH ranged between 6.87 and 8.15 at the forested reach, 6.96 to 7.56 in the urban reach and 6.81 to 8.41 in the agricultural section (Table 4). There were no significant differences in pH values among the land use types (ANOVA, $p>0.05$). Conductivity was highest at the urban section (53.83 ± 7.78), followed by agricultural (52.44 ± 8.74) and lowest in the forest (30.45 ± 6.04). There were significant differences among the land use types (ANOVA, $p<0.05$) with that of the forested reach being significantly lower than the urban agricultural reaches (Tukey HSD, $p<0.05$).

TDS was highest in the urban section (25.28 ± 3.91), followed by the agricultural section (24.68 ± 4.23) and was lowest at 15.33 ± 1.81). Turbidity ranged between 5.64 ± 2.05 in the forested section and 48.25 ± 36.23 NTU in the urban section (Table 4). In the agricultural section it was 32.00 ± 37.34). There were significant differences in the two parameters among the land use types (ANOVA, $p<0.05$) with the values of the forested reach being significantly lower than both the urban and agricultural reaches (Tukey HSD, $p<0.05$).

Table 4: Variations in mean ± SD values of physicochemical parameters in the three land use types. n=24 (forest), n=24 (urban) and n=72 (agricultural)

Parameter	Forest	Urban	Agricultural
DO (mg/L)	8.19 ± 0.39	8.18 ± 0.35	8.03 ± 0.49
Temperature (°C)	15.56 ± 1.07	19.36 ± 1.41	18.89 ± 1.48
Ph range	6.87 - 8.15	6.96 - 7.56	6.81 - 8.41
Conductivity (µs/cm)	30.45 ± 6.04	53.83 ± 7.78	52.44 ± 8.74
TDS (mg/L)	15.33 ± 1.81	25.28 ± 3.91	24.68 ± 4.23
Turbidity (NTU)	5.64 ± 2.05	48.25 ± 36.23	32.00 ± 37.72

TSS concentrations varied significantly among the land use types (ANOVA, $p<0.05$) with the forested reach having significantly lower concentrations than agricultural and urban reaches (Tukey HSD, $p<0.05$). TSS was lowest in the forested reach (7.64 ± 6.40 mg/L) and highest in the urban section (32.08 ± 19.48 mg/L) as shown in Figure 18.

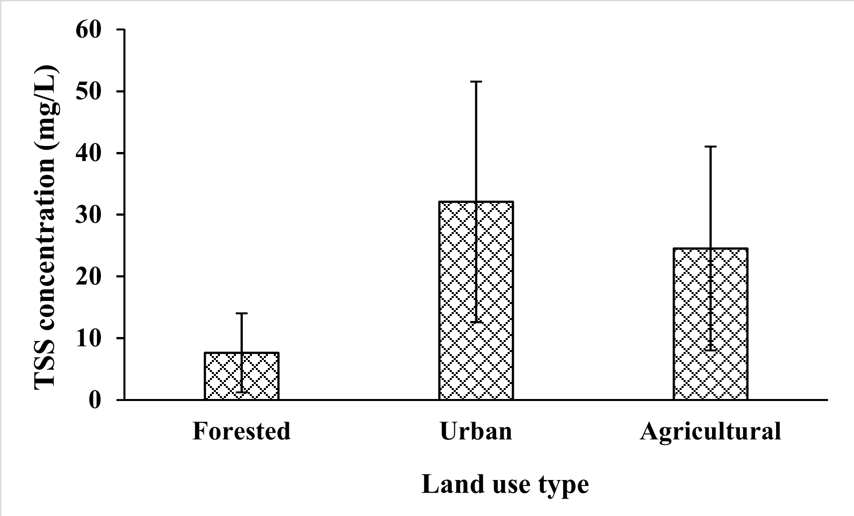

Figure 18: Variations in TSS concentrations in the forested, urban and agricultural sites. n=24 (forest), n=24 (urban) and n=72 (agricultural)

4.2.2 Discharge variation among the three land use types

Figure 19 presents stream discharge values which were lowest in the forested river reach (1.05 ± 1.04 m^3/s) < urban reach (2.57 ± 1.29 (m^3/s) < the agricultural section (2.96 ± 1.27 m^3/s). There were significant variations among the three land use types (ANOVA, $p<0.05$) with the forested reach having a significantly lower discharge compared to the urban and agricultural reach (Tukey HSD, $p<0.05$).

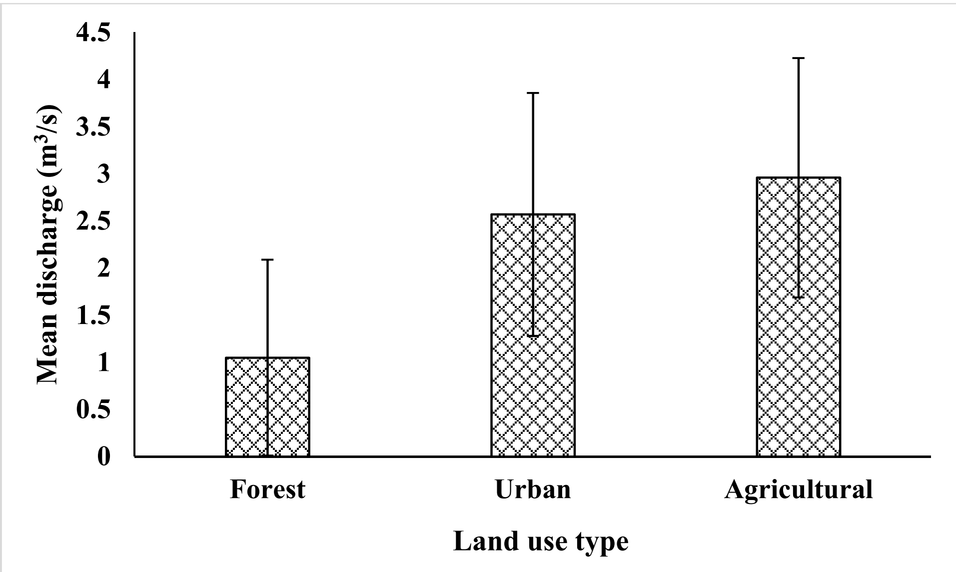

Figure 19: Variation of discharge among the forested, urban and agricultural river sections of River Rupingazi. n=24 (forest), n=24 (urban) and n=72 (agricultural)

4.2.3 Variation of nutrient concentrations among the land use types

Ammonium-Nitrogen concentration ranged between 0.03 ± 0.02 mg/L in the urban and 0.05 ± 0.04 mg/L in the agricultural reach (Figure 20a). The forest had 0.04 mg/L ± 0.01 mg/L. Nitrites was lowest in the forest 0.001 ± 0.000 and highest in the urban and agricultural (0.002 mg/L ± 0.001 mg/L) (Figure 20b). There were significant differences in nitrite-nitrogen concentrations among the three land use types (ANOVA, $p<0.05$). Post hoc test revealed that both ammonia and NO_2 concentration in the forest differed significantly with the values obtained in the urban and agricultural section (Tukey HSD, $p<0.05$).

Nitrate-Nitrogen were highest in the urban section (0.75 mg/L ± 0.32 mg/L) followed by the agricultural (0.55 mg/L± 0.33 mg/L) and lowest in the forest section (0.27 mg/L ± 0.16 mg/L) (Figure 20c). There were significant differences in nitrates concentration among the three land use types (ANOVA, $p<0.05$) and post hoc test showed that NO_3 concentration in the forest was significantly lower than that of the urban and agricultural section (Tukey HSD, $p<0.05$). Total Nitrogen concentrations ranged between 2.36 mg/L ± 2.04 mg/L in the agricultural section and 2.71 ± 2.21 mg/L in the urban section. The forested section had concentration of 2.59 mg/L ± 2.46 mg/L (Figure 17d). There were no significance variations in TN concentrations among the three land use types (ANOVA, $p>0.05$).

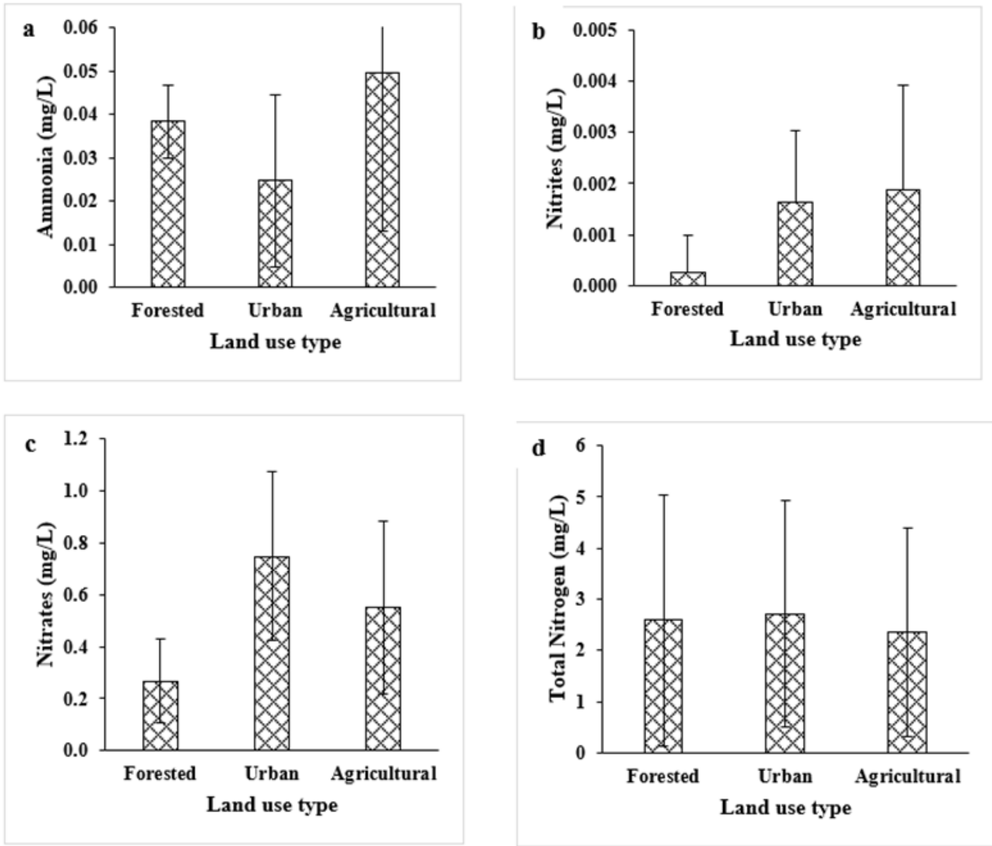

Figure 20: Variation of mean ± SD concentrations of a) Ammonia, b) Nitrites, c) Nitrates and d) TN in forested, urban and agricultural reaches of River Rupingazi. n=24 (forest), n=24 (urban) and n=72 (agricultural)

SRP was lowest in the urban section (0.005 ± 0.004 mg/L) followed by the forest (0.007 ± 0.009 mg/L) and highest in the agricultural section (0.008 ± 0.005 mg/L) (Figure 21). There were no significant differences in SRP concentration among the three land use types. TP ranged between 0.026 ± 0.012 mg/L in the forested section and 0.072 ± 0.068 mg/L in the agricultural section. The urban section had a concentration of 0.071 ± 0.036 mg/L (Figure 21). TP concentrations varied significantly among the three land use types (ANOVA, $p<0.05$). Post hoc test showed that TP concentration in the forest was significantly lower than the one in the urban and agricultural section (Tukey HSD, $p<0.05$).

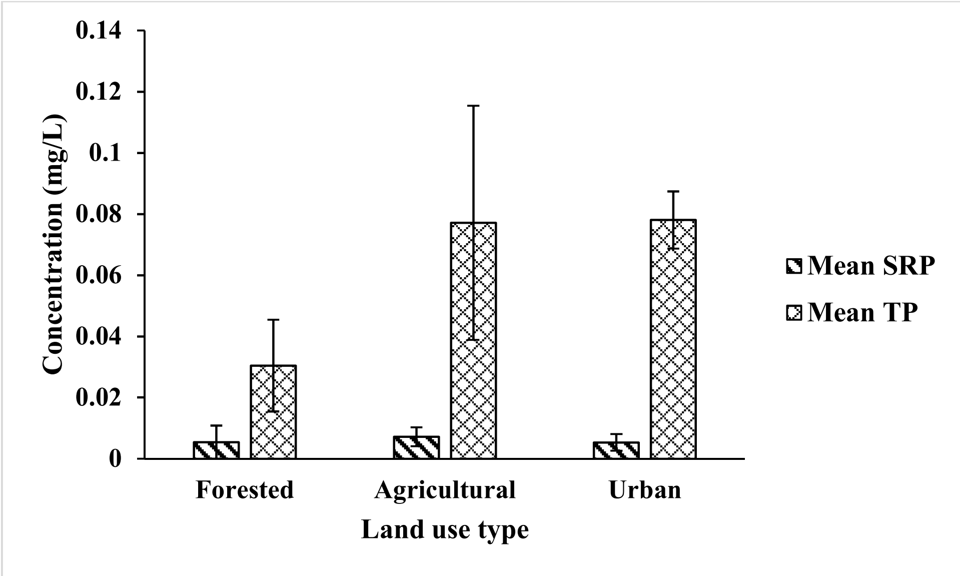

Figure 21: Variation of mean ± SD concentrations of a) SRP and b) Total Phosphorus among the three land use types. n=24 (forest), n=24 (urban) and n=72 (agricultural)

4.2.4 Changes in DOC concentration among the three land use types

DOC concentrations ranged between 0.45 ± 0.17 mg/L in the forest and 0.69 ± 0.34 mg/L in the agricultural (Figure 22). In the urban section, the mean DOC concentration was 0.47 ± 0.16 mg/L. There were significant differences in DOC concentrations among the three land use types (ANOVA, $p<0.05$) with the agricultural reach having significantly higher concentrations than both urban and forested reaches (Tukey HSD, $p<0.05$).

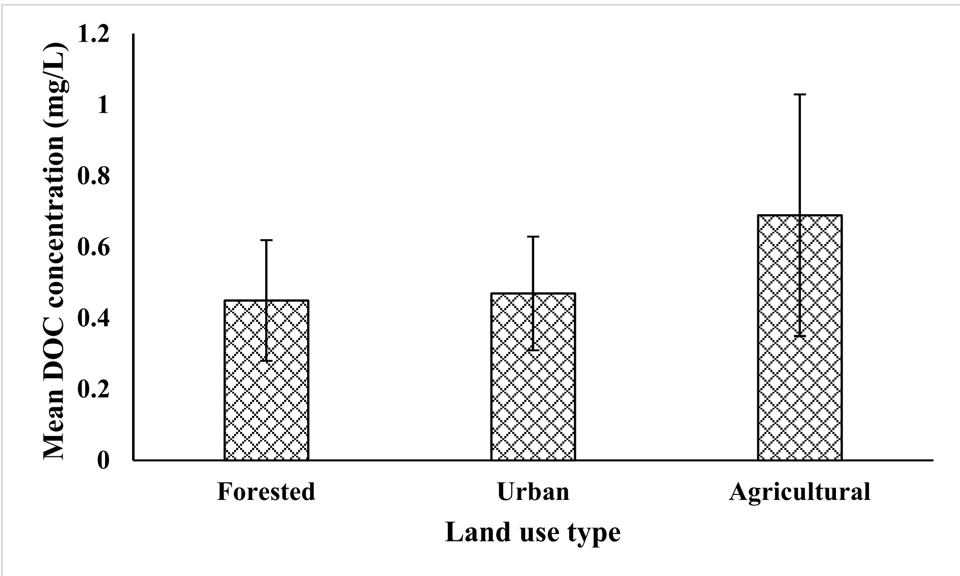

Figure 22: Variation of DOC concentration in the forested, urban and agricultural river reaches of River Rupingazi. n=24 (forest), n=24 (urban) and n=72 (agricultural)

4.2.5 Relationship between nutrients and DOC concentrations in the three land use types

The PCA analysis (Figure 23) was done using a correlation matrix developed using PAST software version 4.0.3. In these results, two components were extracted which explained variation of nutrients and DOC concentrations in the three land use types. For example, the first and second component explained 71.7% and 28.3% of the variability of the parameters respectively. Nitrites, NO_3, TP and DOC have large positive loadings on component 1 meaning they are strongly correlated. This means the sources of NO_2, TP, DOC and NO_3 can be attributed to the urban land use. Soluble Reactive Phosphorus and NH_4 had a large negative loading on component 2 meaning that as agricultural activities decrease, SRP and NH_4 also decrease. Therefore, the variations of SRP and NH_4 can be attributed to agricultural sources.

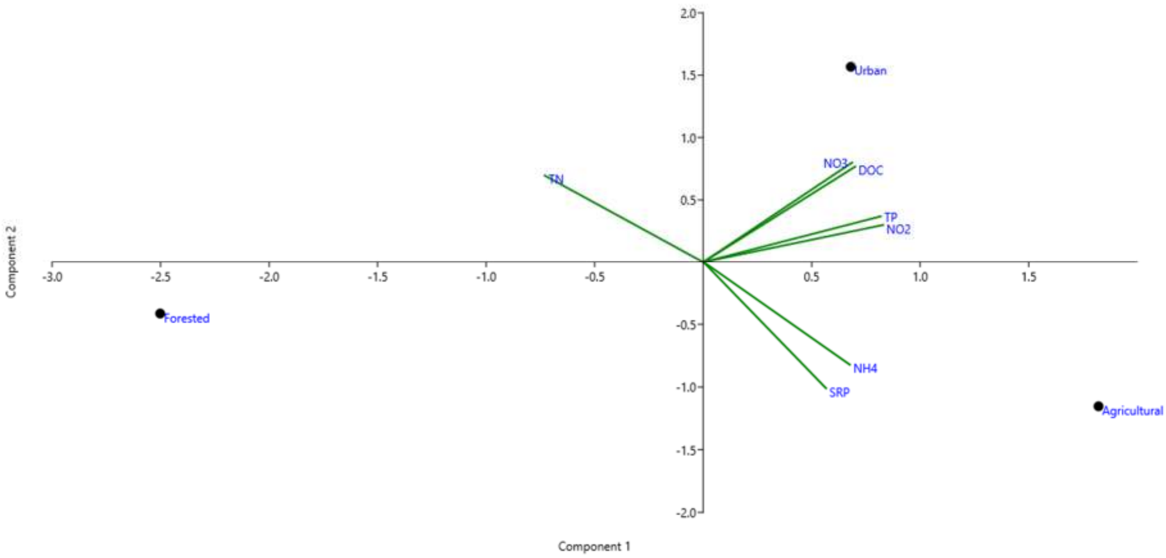

Figure 23: Principal Component Analysis (PCA) for inorganic nutrients and DOC in the forested, urban and agricultural reaches of River Rupingazi

4.2.6 Stream respiration rate along the longitudinal continuum and land use gradient

Respiration rates were calculated as change in oxygen concentration over the 4-hour incubation period ranged between 0.9 mg O_2/h at R10 and 1.8 mg O_2/h at R5 (Figure 24). In river Kapingazi, a tributary, the rate was higher at R9 (1.68 mg O_2/h) than at R7 (1.46 µg O_2/h).

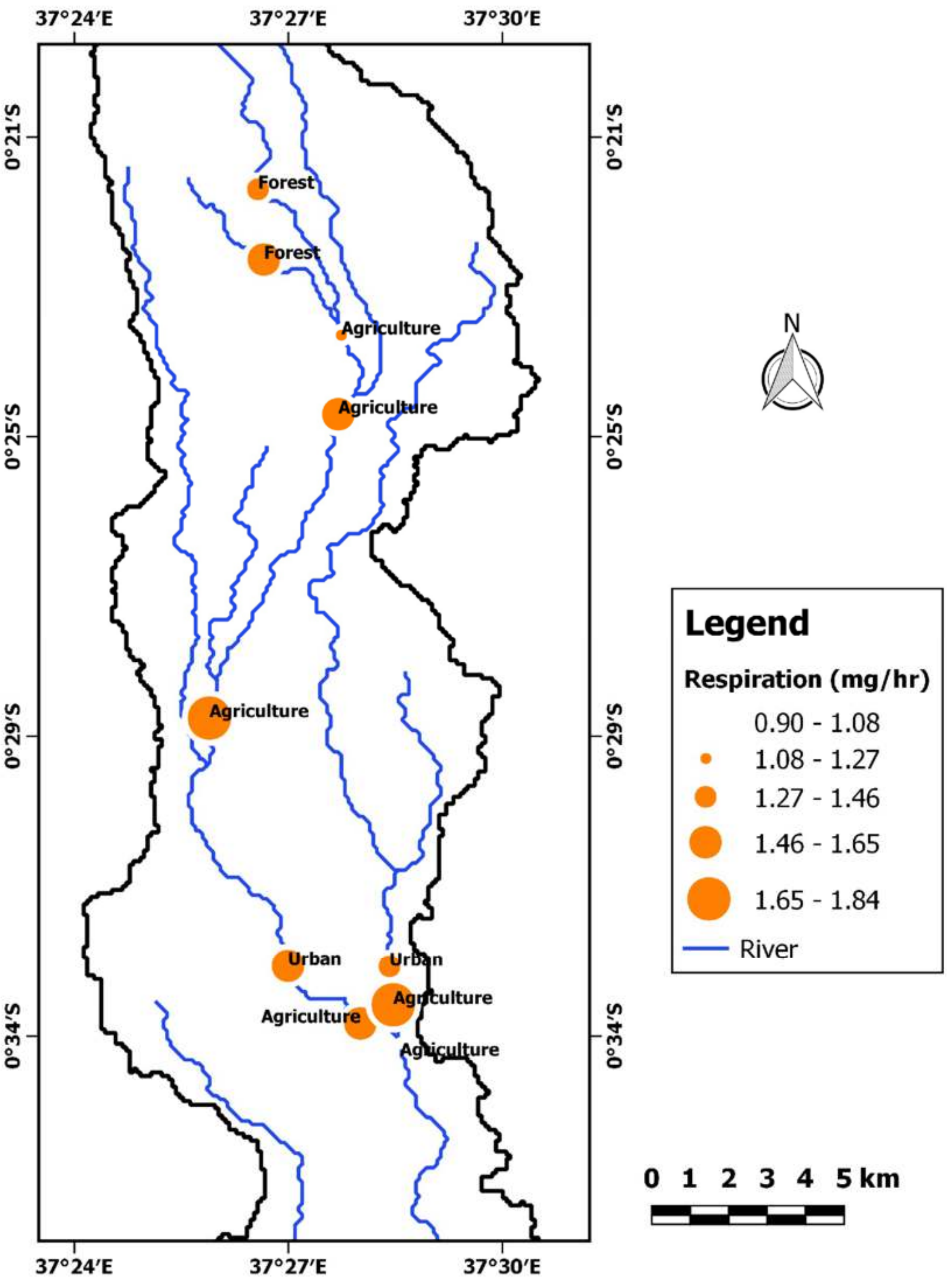

Figure 24: Change in oxygen concentration along River Rupingazi and sampled tributaries

On the other hand, respiration rates along the land use gradient ranged between 1.45 ± 0.06 mg O_2/h in the forested reach and 1.49±0.04 mg O_2/h in the urban reach (Figure 25). However, the respiration rates did not vary significantly among the three land use types (ANOVA, $p>0.05$).

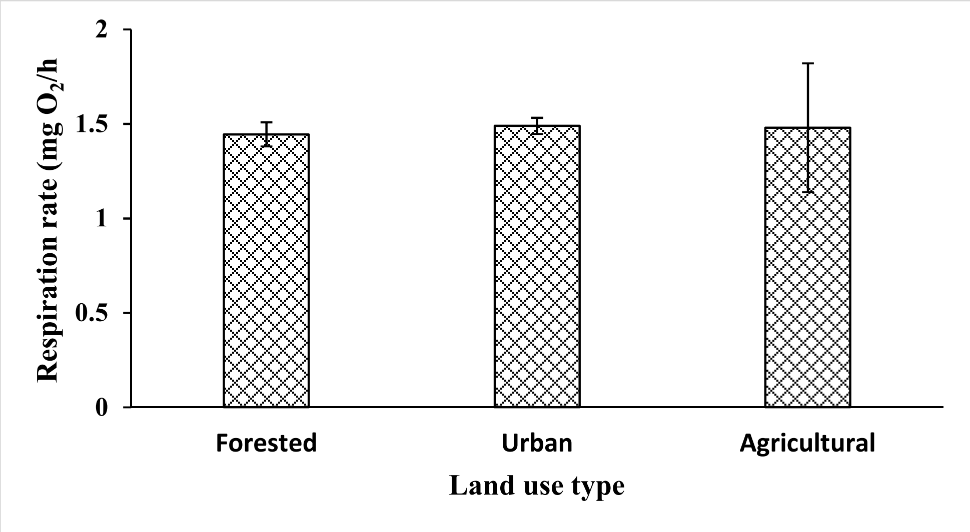

Figure 25: Variation of mean respiration rates among the three land use types. n=24 (forest), n=24 (urban) and n=72 (agricultural)

4.2.7 Relationship between respiration rates, temperature, DO, DOC and OM

Correlation analysis was performed to show how respiration rates related with temperature, dissolved oxygen, DOC and organic matter. Results presented in table 5 showed that respiration rate had a positive trend with OM (0.172) though the relationship was not significant. Respiration rate had a negative relationship with temperature (-0.117), DO (-0.12) and DOC (-0.496).

Table 5: Pearson Correlation analysis showing the relationship between respiration rate, temperature, DO, DOC and OM

	Respiration rate (mg O_2/h)	Temperature (°C)	DO (mg/L)	DOC (mg/L)	OM (g)
Respiration rate (mg O_2/h)	1				
Temperature (°C)	-0.117	1			
	0.748				
DO (mg/L)	-0.12	-0.591	1		
	0.741	0.072			
DOC (mg/L)	-0.496	0.762*	-0.446	1	
	0.145	0.01	0.197		
OM (g)	0.172	-0.43	-0.123	-0.364	1
	0.635	0.215	0.734	0.301	

* Correlation is significant at the 0.05 level (2-tailed).

4.2.8 Human activities within the agricultural reach of the river

From the 60 crop farmers and livestock keepers interviewed, 25 had tea plantations, 20 had coffee plantations and around 50 farm subsistence food crops. Tea plantations size ranged from 2 acres to 10 acres. All tea, coffee and khat farmers use chemical fertilizers, mostly thrice per year. The food crop farmers apply once or twice in one planting season. The common fertilizer used is DAP, Ammonia and NPK 2323. Around site R4, most of the farmers use organic manure. Most of the livestock keepers have between 2 to 5 cows each and the cows are zero grazed except for the area around site R6 where they are grazed near the riverbanks. Most farmers reported that they plant nappier grass for feeding the cows as well as preventing flooding in the farms when River Rupingazi is in high flows. The common soil erosion control method is terracing and contour ploughing.

CHAPTER FIVE

DISCUSSION

There were marked variations in physicochemical parameters, nutrients and DOC among the sampled sites. Respiration rates did not vary significantly though. It was evident that land use had effect on most parameters except TN and stream respiration rates.

5.1 Variation of physicochemical parameters

Physico-chemical parameters such as temperature, dissolved oxygen, pH and conductivity have a profound effect on nutrient and carbon dynamics in River Rupingazi. Mean concentrations and variability in DO were generally similar at the two least disturbed sites (site R0 and R1). Strong positive 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 also influences the amount of dissolved oxygen in the water (Cox, 2003). This is expected as saturation levels of oxygen decrease with increase in temperature. Von Sperling (2007) reported that the colder the water is, the more oxygen it can hold and this increases water pH. In the main river, the highest DO concentration was recorded at site R1 (8.47 ± 0.27 mg/L) where temperature was lowest (15.07 ± 0.79 °C) while the lowest DO concentration was recorded at site R10 (7.85 ± 0.64 mg/L) where temperature was highest (19.59 ± 0.27 °C). A linear regression model showed that 30.7% of the variations in DO could be explained by change in temperature. Pristine streams have high dissolved oxygen levels which reduces as the system becomes polluted (Lewis, 2008). Low values of DO were recorded in the agricultural sites which had high temperature. This is attributed to human influence. Most agricultural 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).

Dissolved oxygen levels can be used as a parameter to measure whether a system is organically polluted (Huang *et al*., 2017). The sites R6 and R7 which received sewage effluents from nearby hotels and households had low dissolved oxygen. The scenario of urban sewage being dumped into rivers without treatment is common in Kenya (Kerich & Fidelis, 2020). 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). Weak negative correlation between DO and DOC was observed. When more DOC enters streams and rivers through sewage, more organic matter is carried with it into the river water and more O_2 is consumed to decompose this extra input of organic matter. DO had a positive correlation with all nutrients except SRP. Results from this study are similar to those of Bonareri (2017 who got the highest DO concentration to be 8.30 ± 0.06 mg/L in the forested sites of River Rupingazi.

The lowest and highest temperature of 15.07 °C site R1 and 20.76 °C in site R9 respectively measured in this study are within the range for effective nitrification process, organic matter degradation and photosynthetic activity with an optimum temperature at 30 °C (USEPA, 2014). The temperature variation obtained could be attributed to the influence of ambient air temperature (USEPA, 2015). Temperature change can also result from change in altitude, shading and insolation (Ohmura, 2012). Embu has a high altitude and the temperatures never exceeded 20.76 °C. The lowest temperature values were found in the sites R0 and R1 with the highest latitude (1927 and 1816 metres above sea level respectively) while the highest temperatures were found at the downstream sites with lower altitude (below 1270 metres above sea level). The high temperature values in agricultural reaches can be attributed to low canopy cover hence high insolation. Site R9 which had the highest temperature values had the lowest canopy cover. The low temperatures obtained in the upstream sites: R0 and R1 can also be attributed to the high canopy cover in the forest which shades the water preventing sun rays form penetrating and heating the water. The time of sampling also has effect on the water temperature as in the morning the water is cold and during the afternoon it is already heated and temperatures are higher. This is why temperatures at site R7, sampled in the late afternoon, were high (20.4 °C) despite having higher altitude compared to R8 and R10. Site R0 and R1 were sampled in the morning when the air temperature was still cool and this explains the low temperatures obtained. Site R7 had the lowest canopy cover and had a mean temperature of 20.4 °C, very close to the highest mean temperature recorded at site R9. Temperature directly influences 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). 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 6.61 - 8.41 along the longitudinal continuum of river Rupingazi, are within the range (6 - 9) which favours 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 most turbulent flows like R4, R9 and R10, the reaeration of water is enhanced thus causing pH to rise, agreeing with study of Bhateria and Jain (2016), who documented a link between water reaeration and increased pH. In this study pH had a strong negative correlation with ammonia implying that increase in pH transforms ionized ammonia to non-ionized ammonia as observed by Camargo and Alonso (2006).

Electrical conductivity is a measure of the quantity 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 EC recorded (62.68 μS/cm) and the lowest (28.93 μS/cm) can be attributed to release of nutrients and ions into the river water column from leaching and mineralization. The release seems to be highest at site R9 which is an agricultural site and lowest at site R1, a forested site. Several studies found EC to be low in forested catchments (Githaiga *et al*., 2003; Kambwiri *et al*., 2014). High conductivity recorded in the agricultural sites may be attributed to increased human activities in the riparian area adjacent to the streams which increases loading of iron rich sediments from the fertilized farms in the catchment into the river. Forested streams on the other hand are characterized with dense riparian zone which protects the stream from much sediment loading thus lowering the EC. At the urban sites, EC was significantly high (49.32 μS/cm and 58.34 μS/cm at site R6 and R7 respectively) because of sewage load and other anthropogenic instream activities. This is similar to the results of a study by Markewitz *et al*. (2001). However, sometimes ions and nutrients are locked in sediments and not available in the water column thus resulting to low EC (WEF, 2010).

According to a study by Gichana *et al*. (2015), streams draining similar geological catchment are more likely to have similar EC thus any variation could be a likely indicator of anthropogenic impacts. Agricultural land use is associated with usage of chemical fertilizers, pesticides and acaricides for livestock which end 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 river Rupingazi. Among the agricultural sites, the highest EC values were observed in the catchments dominated by small scale agriculture, followed by commercial tea and coffee plantations. The sites in the natural forest had the lowest EC (Table 2).

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 (site R10) in the main river and lowest at site R2 in the forest. The surface runoff from the surrounding agricultural farms 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 in rainfall events (Lee *et al*., 2019) for example in the mid and downstream sites. A study by Li *et al*. (2015) stated that water turbidity strongly depends on precipitation rate which causes the 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 sites R3 and R4 is a result of several mitigation measures like vegetative buffers implemented in this area by local governments and farmers. The residents around the above-mentioned sites reported that people are instructed to leave a 5 metres buffer zone between the river and their farms coupled with inflows from upstream non- agricultural areas. There were numerous small tributaries flowing from the non-farmed surrounding areas into River Rupingazi around these sites. The turbidity differences between the forested and agricultural reaches means 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.

5.2 Discharge variation along the longitudinal continuum and land use types

Discharge kept increasing downstream even though there was water abstraction from most sections of river Rupingazi (Appendix 4). 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 the previous sampling station along the main river. Discharge was highest at the confluence with river Kapingazi, site R10 and was lowest at site R0 which is a small order 1 tributary. The decrease in discharge from site R7 to site R9 could be attributed to water abstraction for agricultural crop farming, domestic use and for hotel operations. Pumps and pipings for water abstraction were visible within this reach. It’s important to note that most sampling was done during rainy days which coincided with the time of study, explaining the high amount of water flowing from the catchment into the river thus the high discharge. This is evidenced by good relationship between discharge and variation in nutrient concentrations reported in several studies (OEPA, 2016).

Land use change affects river discharge in watersheds (Konkul *et al*., 2014). Human activities for example urbanization and agriculture can cause decrease in infiltration consequently increasing the rate and volume of runoff (Klongvessa *et al*., 2017). This explains the high discharge obtained at the agricultural and urban sites during the samplings done on rainy days. The results of this study, which showed low discharge at forested sites compared to agricultural and urban sites, agree with those of Bruijnzeel (2004) who argued argued that forests act as sponges through increasing infiltration rates and retaining soil moisture. The high discharge recorded in site R1 during the first sampling, after a heavy storm, (3.216 m^3/s) 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 increase in rainfall intensity.

5.3 Effect of land use on nutrient concentrations

There were notable variations in nutrient concentrations along the river. Some had an increasing while others decreasing trend depending on the land use. 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 through preventing dissolved and suspended nutrients from moving towards streams and rivers (Basnyat *et al*., 2000).

Generally, the ammonium levels were within the recommended range for river water quality. Ammonium was lowest at site R6 and highest at site R1. The high dissolved oxygen concentration in site R6 (8.463 mg/L) explains the low ammonium concentrations since DO favours nitrification process and inhibits ammonification. The corresponding high DO levels make the environment less conducive for existence of ammonium as the main nitrogen component because it is converted to nitrites and nitrates which, as evident in the data, are in higher concentrations than NH_4. When a watershed 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 untreated or partially treated sewage input into a water body.

Nitrites had the lowest concentrations among all the forms of nitrogen along river Rupingazi. High nitrite concentrations are associated with incomplete nitrification. The lowest concentration of nitrite was recorded at site R3 (0.0002 ± 0.0002 mg/L) and the highest was recorded at site R10 (0.0042 ± 0.0014 mg/L). There was a significant positive correlation between temperature and nitrites meaning that as the temperatures increased, degradation of organic matter by bacteria thus releasing more nitrites through mineralization (Truu *et al*., 2009) increased. The slight decrease in nitrite concentration from site R1 to R3 could be due to increase in oxygen concentrations thus enabling conversion nitrites to nitrates, a process which utilizes dissolved oxygen. The decrease could also be as a result of dilution by the incoming water from River Nyanjara, a tributary which drains into River Rupingazi just before site R3. McDonald *et al*. (2011) reported that dilution of river water brings about self-purification and that water abstraction may affect the dilution capacity of a river and consequently influence the river self-purification processes.

Nitrates concentrations were within the recommended levels of 10 mg/L by WHO (2017). The values were highest in site R9 and lowest in site R1. Nitrate is a good indicator of catchment disturbance in P-limited systems (Rao & Puttanna, 2000). Site R9 is an agricultural site and chemical fertilizers rich in nitrates like UREA, CAN and NPK 23-23 used in the nearby farms could have been washed into the river in surface runoff during the rains. According to the responses from the questionnaire, most farmers apply fertilizer twice per planting season: During planting period and topdressing. As observed in the downstream sites, most farms extended into the riverbanks. This sampling having been done in the rainy season, it is possible nitrates were washed into the river through surface runoff. High dissolved oxygen levels promote nitrification resulting to high levels of nitrates due to nitrification and mineralization of logs and woody materials on the riverbed. The decrease in nitrates from site R9 to R10 (1.05 mg/L to 0.47 mg/L) could have been caused by dilution at the Rupingazi-Kapingazi confluence since the incoming water from River Rupingazi had a lower NO_3 concentration (1.05 mg/L).

Stream water temperature has a positive effect on ammonification, nitrification and decomposition of organic matter which are source processes of nitrates and this explains the positive correlation between temperature and NO_3-N concentration. NO_3-N was highest in site R9 where temperature was highest. Riparian vegetation increases NO_3 retention (Vidon & Hill, 2004) and this is probably why site R1 which is found in the forest had low NO_3 concentrations compared to the downstream sites which had less riparian vegetation.

The total nitrogen levels were highest at site R6 and could be attributed to human input from the catchment (Wen *et al*., 2017). Site R9 had the lowest TN. The higher concentrations of TN during the rainy season in the agricultural 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 the urban areas (Biggs *et al*., 2004). From the regression analysis, TSS accounts for only 0.1% of variations in TN. This means TN may have been coming from point sources and leaf litter.

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 main river reduced in all sites from site R1 to R6 (this is attributed to dilution effect by waters from incoming tributaries) then increased in R8 and R10. The concentrations were significantly higher in the agricultural reaches than the forested and urban 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 site R8 and R10. 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) who noted a positive correlation between nutrients, TSS and discharge. The high levels could also mean that the mineralized phosphorus from the agricultural lands end up in the streams and rivers through surface runoff as noted by some studies (Liu *et al*., 2012; Wilson & Xenopoulos, 2009). Usage of chemical fertilizers: DAP and NPK 23-23 which are rich in phosphates also contributed to the high levels of TP in the agricultural land use sites. When there is rain, the fertilizers are carried in surface runoff from the farms into the river as non-point sources.

5.4 Effect of land use on DOC

A clear variation in DOC concentration was observed in the ten sampling sites. DOC reduced from site R1 (forested area) to R3 (agricultural area) then kept increasing in all the other sites in the main river. This is in agreement with the results of Wi *et al*., 2012 and Wilson & Xenopaulos (2009) who documented that agricultural tillage facilitates the mobilization of DOC which end up in the streams. These results are also consistent with Recha *et al*. (2013) who reported a 153% rise in DOC export after conversion of a tropical rainforest in western Kenya into agricultural farmland mainly due to mobilization of the DOC stored in the topsoil of the forested land. Other studies (Graeber *et al*., 2012; Singh *et al*., 2015) in the northern hemisphere also confirmed that streams flowing from catchments with large percentage of agricultural land have higher DOC concentrations compared to streams from catchments mainly dominated by forests. Wilson and Xenopoulos (2008) found out that stream water DOC concentrations are strongly related to discharge patterns. Results of this study however showed a weak negative relationship between the two variables (-0.075) meaning that DOC concentrations decreased with increase in discharge possibly due to dilution effects. Results of this study are consistent with those of Pacific *et al*. (2010) who observed that increase in stream flow quickly depletes DOC through flushing whereby the mobilizable DOC which had built up during base flow conditions is washed away.

5.5 Effects of land use on stream respiration

The highest stream respiration rate was measured in site R5 and this is influenced by high nutrient concentrations in the agricultural reach. In general respiration rates seemed to respond less to agricultural effects with the common response being an increase. This outcome is unexpected because with reduction of allochthonous organic matter received by streams running through the landscapes, reduced respiration was expected. However, the increased respiration rate could be fueled by autochthonous sources of organic matter leading to the observed pattern. Respiration rates were noted to increase with increase in water temperature and nutrient concentrations in the river.

Urban land use influences stream respiration in most instances and results of this study agree with Hall and Beaulieu (2013) who found that respiration rates increase substantially in urban land uses due to increased nutrient and carbon inputs from septic and sewer systems and frequent changes in stream flow. Carroll and Jackson (2008) observed that urbanization also reduces inputs of leaf litter and wood into streams and rivers due to deforestation and alteration of riparian land hence reducing the available energy sources for microbial respiration. This explains the low respiration rates in sites R6 and R7 compared to most agricultural sites. The increase in fine sediments and decrease in stability of stream sediments in the agricultural and urban land uses reduces sites available for microbial respiration. Results of this study are consistent with Roy *et al*. (2009) who observed elevated respiration rates in streams receiving sewage and wastewater discharge. They however stated that respiration rates do not always show clear patterns with urbanization.

CHAPTER SIX
CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

For objective 1, nutrient and DOC concentrations generally increased downstream, with significant variations among the sampled sites. 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. Therefore, the null hypothesis that there is no significant change in water column N, P and DOC concentrations along the longitudinal continuum of river Rupingazi is rejected.

For objective 2, it was concluded that nutrient and DOC concentrations differed significantly among the three land use types: forested, agricultural and urban river reaches. Concentrations of most nutrients and DOC were higher in both agricultural and urban reaches than in the forested reaches. DOC concentration ranged between 0.45 ± 0.17 mg/L in the forested reach and 0.69 ± 0.34 mg/L in the agricultural reach. The null hypothesis that there is no significant difference in N, P and DOC concentrations in the water column of the forested, agricultural and urban reaches of river Rupingazi is thus rejected.

For objective 3, stream respiration did not respond to land use meaning that even though this fundamental stream process is affected by land use. Site R5 had the highest respiration rate while R10 had the lowest. Surprisingly, they are all in the agricultural land use area. The forested reach had the highest respiration rate (1.49 ± 0.09 µg O_2/g/w) while the agricultural reach had the lowest (1.25 ± 0.5 µg O_2/g/w). However, it was concluded that there were no significant variations in stream respiration rates among the three river reaches: forested, agricultural and urban. Therefore, the null hypothesis that there is no significant difference in stream respiration rates at the forested, agricultural and urban stream reaches is not rejected.

6.2 Recommendations

From objective 1 and conclusion 1, it is recommended that restoration of riparian vegetation be done in the Rupingazi catchment by the inhabitants. The existing riparian buffer zones should be maintained to mitigate the impacts of the land-based anthropogenic activities. These preventive measures can be achieved through motivating farmers and providing incentives and guidance by the local leaders.

From objective 2 and conclusion 2, it is recommended that farmers around Rupingazi catchment adopt organic farming and reduce the usage of chemical fertilizers. This will reduce the levels of nutrients and DOC entering the river through surface runoff. Farmers should also avoid farming close to the riverbank and washing clothes in the river to minimize soil erosion and increased phosphorus concentrations.

From objective 3 and conclusion 3, it is recommended that farmers in the Rupingazi catchment should maintain the riparian vegetation so that there may be more OM entry into the river consequently increasing stream respiration and overall stream health. Dumping of partially treated and untreated sewage into River Rupingazi should be stopped so as to improve dissolved oxygen concentrations in the urban sites and therefore improve microbial respiration.

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APPENDICES

Appendix A: Questionnaire

Name of interviewer	
Date of interview	
Location	
Size of farm	
Type of crop grown	
Use of chemicals and fertilizers	(YES/NO)
Type of fertilizer used	
Time of fertilizer application	Before/after rains
How often the fertilizer is applied	
Number and kind of livestock kept	
Amount applied per season	
Farm management practices in place to prevent soil erosion	

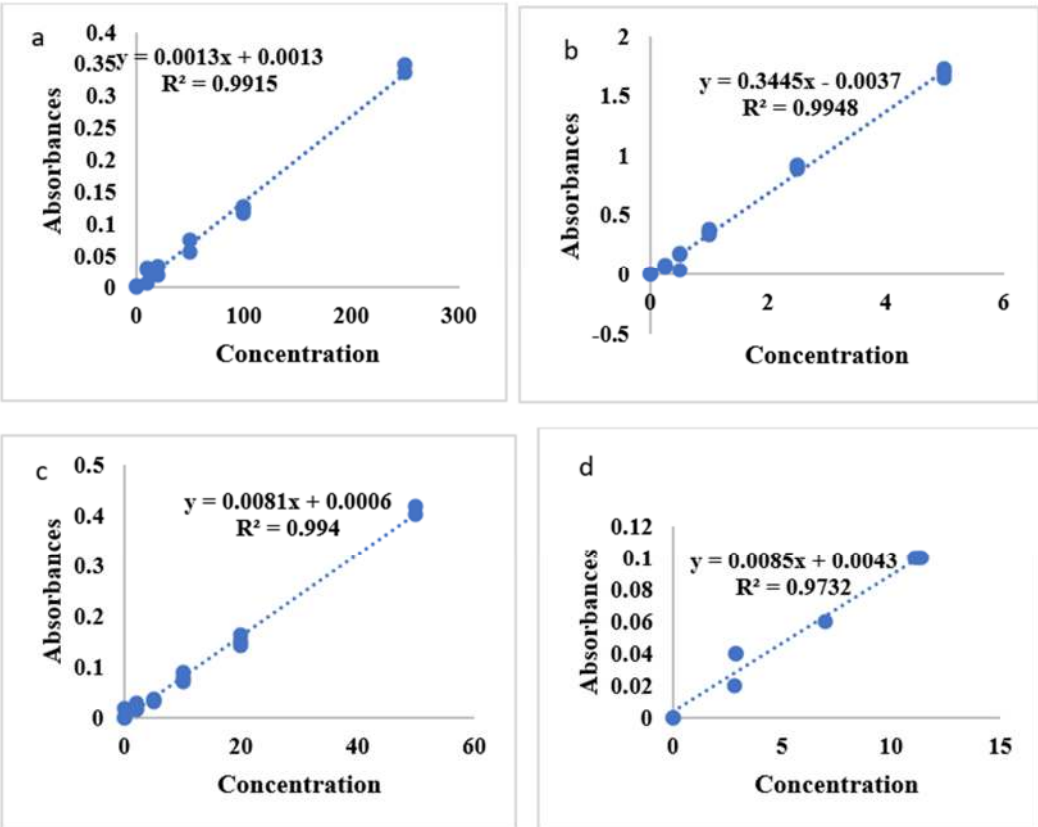
Appendix B: Correlation matrix of selected variables. ** sig. 0.01 level (2 tailed)

	DO	Saturation	Temp	Conductivity	TDS	Turbidity	Discharge	NO_3	NO2	NH_4	TN	SRP	TP	TSS
DO	1.000													
Saturation	.838**	1.000												
	0.000													
Temp	-.554**	-.293**	1.000											
	0.000	0.001												
Conductivity	-0.175	-0.032	.755**	1.000										
	0.055	0.726	0.000											
TDS	-0.137	0.002	.742**	.963**	1.000									
	0.136	0.984	0.000	0.000										
Turbidity	-0.098	-0.070	.475**	.626**	.649**	1.000								
	0.286	0.446	0.000	0.000	0.000									
Discharge	0.103	0.106	0.294	0.287	0.249	-0.061	1.000							
	0.529	0.515	0.066	0.073	0.121	0.710								
NO_3	-.195*	-0.095	.586**	.579**	.576**	.452**	-0.054	1.000						
	0.033	0.301	0.000	0.000	0.000	0.000	0.739							
NO2	-.230*	-.286**	.459**	.584**	.587**	.490**	.373*	.349**	1.000					
	0.011	0.002	0.000	0.000	0.000	0.000	0.018	0.000						
NH_4	-.212*	-.333**	0.082	0.088	0.089	0.042	.383*	0.002	.639**	1.000				
	0.020	0.000	0.371	0.337	0.334	0.649	0.015	0.981	0.000					
TN	-0.119	-.226*	-0.037	-.249**	-.308**	-0.004	0.019	-0.146	0.087	0.141	1.000			
	0.197	0.013	0.684	0.006	0.001	0.964	0.910	0.112	0.343	0.126				
SRP	0.070	-0.004	-0.029	.210*	.249**	.280**	0.278	-0.095	.402**	.403**	-0.035	1.000		
	0.449	0.962	0.757	0.021	0.006	0.002	0.082	0.300	0.000	0.000	0.701			
TP	-.247**	-.244**	.468**	.499**	.522**	.703**	0.176	.334**	.419**	0.104	0.007	.214*	1.000	
	0.007	0.007	0.000	0.000	0.000	0.000	0.278	0.000	0.000	0.257	0.937	0.019		
TSS	-0.041	-0.087	.472**	.575**	.589**	.810**	-0.037	.505**	.442**	0.006	0.037	0.177	.560**	1.000
	0.6615	0.34712	6E-08	8.0946E-12	2E-12	6E-29	0.824	4.83E-09	5E-07	0.9516	0.689	0.0542	3.4E-11	

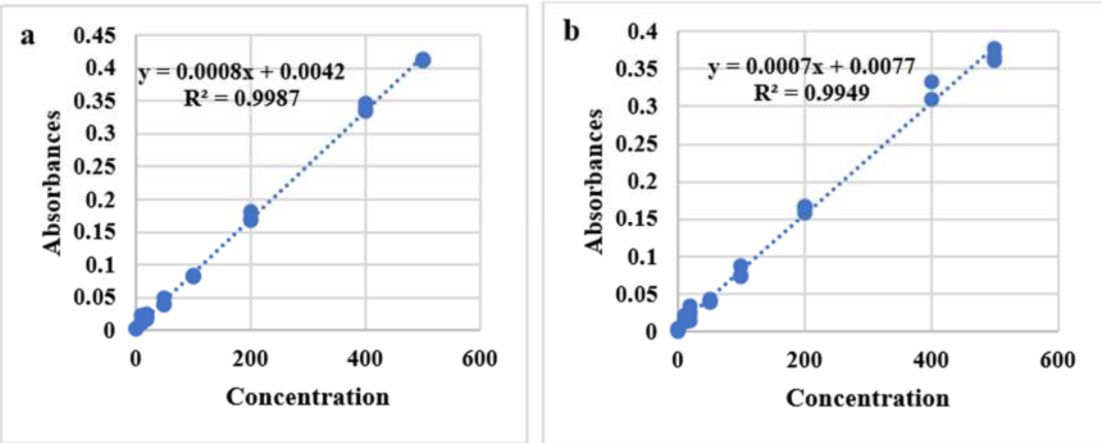
Appendix C: Human activities at some of the selected sampling sites along River Rupingazi and its tributaries

Appendix D: An oven, autoclave and muffle furnace used for nutrient analysis at the Egerton University laboratories

Appendix E: Prepared standard calibration curves for a) Ammonium, b) Nitrites, c) Nitrates and d) Total Nitrogen.



Appendix F: Standard calibrations curves prepared for a) SRP and b) Total Phosphorus.



Appendix G: Research permit

REPUBLIC OF KENYA

NACOST

THE SCIENCE, TECHNOLOGY AND INNOVATION ACT, 2013

The Grant of Research Licenses is Guided by the Science, Technology and Innovation (Research Licensing)

Regulations, 2014

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