# CONTRIBUTIONS OF ANTHROPOGENIC ACTIVITIES TO LEAF LITTER DECOMPOSITION IN RIVER KAPINGAZI, EMBU COUNTY, KENYA

**NELIAH MORAA NYAMBANE**

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# DECLARATION

This thesis is my original work and has not been presented elsewhere for a degree or any other award.

Signature ……………………… Date ………………………

**Neliah Moraa Nyambane**

Department of Land and Water Management

A509/1089/2016

This thesis has been submitted for examination with our approval as the University supervisors.

Signature …………………….. Date ……………………

**Dr. John Gichimu Mbaka**

Department of Environmental Science

Machakos University

Signature ………………………. Date ………………………

**Dr. Justin M. Nyaga**

Department of Biological Sciences

University of Embu

# DEDICATION

To My Loving Father Dr. Charles Onyari, Mom Elizabeth Ongeri, my Sister Faith and Brother Daniel

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## TABLE OF CONTENTS

# LIST OF TABLES

# LIST OF FIGURES

# LIST OF PLATES

# LIST OF ABBREVIATIONS AND ACRONYMS

| | |
|---|---|
| **AFDM** | Ash Free Dry Mass |
| **CPOM** | Coarse Particulate Organic Matter |
| **DM** | Dry Mass |
| **FPOM** | Fine Particulate Organic Matter |
| **TP** | Total Phosphorus |

# ABSTRACT

Human activities have potential adverse effects on freshwater ecosystems, which sustain a wide range of ecosystem services such as the provision of potable water, nutrient cycling, and a habitat for organisms such as decomposer macro-invertebrates. The effect of these activities with regard to water quality and invertebrates has been established and well documented. However, information on the effect of human activities on leaf litter breakdown as an indicator of river ecosystem integrity is still limited in Kenya. The objectives of the study were: To determine effect of anthropogenic activity on litter decomposition rates of *Lantana camara, Eucalyptus grandis* and *Psidium guajava;* to compare leaf litter decomposition rates of the three species and To examine relationships between leaf litter decomposition rates macro invertebrate shredders and water quality parameters. The study employed the litterbag method where dry leaves enclosed in mesh bags were incubated in the river to measure mass loss decomposition and associated macro invertebrates. Data was collected from September to December 2018 The Physico-chemical characteristics of water measured *in-situ* included electrical conductivity (EC), pH, total dissolved solids, dissolved oxygen (DO) concentration and temperature. Further, water discharge, velocity, depth and width were determined. Laboratory analyses were conducted to determine concentration of nitrates, nitrites, total phosphorus and phosphates. Daily mass loss of the collected bags was determined to assess litter decomposition rates. Macro invertebrates identified were classified up to family level and grouped into functional feeding groups. Data was analyzed using R statistical packages. Findings show that site and plant species had a significant effect ($p$=0.05) on leaf litter decomposition rates. Of the three species, *Lantana camara* had the fastest decomposition rate while *Psidium guajava* had the slowest rate of decomposition. Water physico-chemical parameters had both positive and negative correlation with leaf litter decomposition rates and no macro invertebrate shredders were observed. It is recommended that future studies should evaluate other human activities (for example, mining and waste water effluents) and effects of leaf litter mixtures on the decomposition process.

# CHAPTER ONE

# INTRODUCTION

## 1.1 Background Information

River ecosystems provide a wide range of goods and services for human well-being (Gilvear *et al*., 2013). The goods include food, fiber, and water for human consumption and such as nutrient cycling, maintenance of water quality and habitats for diverse groups of organisms. Provision of many of these goods and services depends on riparian litter supply from terrestrial zones and decomposition in the aquatic zone (Graça *et al.*, 2015). Leaf litter breakdown is one of the major ecological mechanisms that maintain river ecosystem integrity and food webs by sustaining the provision of benefits such as nutrient cycling (Von *et al.,* 2017). However, anthropogenic activities continue to cause disturbance on river ecosystems through activities such as change in flow regime (Aristi *et al.*, 2014), pollution (Protano *et al.,* 2014), introduction of invasive species (Gallardo *et al.*, 2016) and climate change (Haddeland *et al*., 2014).

Globally, the effect of anthropogenic disturbances on rivers is evident (Nyairo *et al.*, 2015; Campbell *et al*., 2018; Tumusiime *et al*., 2019). For instance, flow regulation and pollution by nutrient enrichment were observed to disrupt the ecosystem integrity of St. John River in Canada by altering the abundance and distribution of aquatic fungi that influence leaf litter decomposition (Noel *et al.,* 2016). Eutrophication due to anthropogenic addition of nutrients in the streams of Ave River basin in Portugal led to decreased leaf breakdown attributable to decreased decomposer taxa richness that resulted from toxicity (Dunck *et al*., 2015). Decomposition slowed down by up to 50.7% in the streams of Sierra Nevada region in Spain due to flow regulation by weirs and water diversions (Mollá *et al.*, 2017). Construction of dams on the Henares River Basin in Spain has caused decreased temperatures and nutrients in the lower reaches (Benitez-Mora & camargo, 2014). Urbanization in Malaysia has affected ecosystem integrity of the Ampang River (Yule *et al*., 2015). The authors found 68% faunal abundance in urban sites as compared to 33% in forested sites and breakdown rates were higher in urban areas than in forested areas. This could be attributed to the anthropogenic activities that altered physicochemical properties of the river ecosystem. However, in another case study, logging in forested streams was found to have little impact on litter decomposition in Cameroon (Ibrahima *et al.,* 2016). The authors suggested that the insignificant effect was due to the fact that 6 or 7 years after logging, the ecological and biological functions were greatly restored.

In Africa, hydrocarbon pollution in the Niger River delta resulted in decreased leaf litter decomposition rates of red, white and black mangrove leaves (Numbere & Camillo, 2017). The authors also found significant differences among the decomposition rates of the three studied leaf litters. Studies have cited the bioaccumulation of aluminum and iron from coal mining in South Africa, as a reason for water quality concerns in the Olifants River with potential effects on nutrient cycling (Dabrowski *et al.,* 2014). Similarly, increase in human settlements and agricultural activities in the upper Manyame river watershed in Zimbabwe have increased eutrophication rates from 130 kg/day to 376 kg/day for nitrates (Kibena *et al.,* 2014). Such human-related changes have potential impact on leaf litter decomposition. This is because litter decomposition is modulated by factors such as physicochemical variables like nutrients and temperature which influence biotic communities (microorganisms and invertebrates) which in turn regulate the decomposition process (Silva-Junior *et al.*, 2014).

In Kenya, the release of wastewater from Nairobi city to the Athi River led to a reduction of between 36 and 67% dissolved oxygen (DO) saturation levels in the river (Marwick *et al*., 2014). Human settlements and agricultural activities have been reported to affect the water quality for domestic use through nutrient and heavy metal pollution of the Nyangores and Amala tributaries of the Mara River (Nyairo *et al.,* 2015). Tsisiche *et al* (2019) studied the effect of change of land use in the streams of Upper Mara and found that leaf litter decomposition rates increased in agriculturally-drained stream as compared to streams in indigenous forests. Additionally, they found low abundance of shredders as compared to other functional feeding groups and attributed this to the poor quality of litter from the exotic Eucalyptus plantations around the studied streams.

Various studies have demonstrated differences in decomposition rates of leaf species (Dobson *et al.,* 2004; Tenkiano & Chauvet, 2018; Given *et al*., 2019). The different studies cite differences in leaf traits as the reason for discrepancies in leaf litter breakdown rates. However, no generalization can be made. For instance, Dobson *et al* (2004), demonstrated differences in decomposition rates of five riparian species (*Vanguera spp, Syzygium spp,Pitosporum spp, Rhus spp.)* along River Njoro with *Syzygium spp*. having the highest rate of decomposition while *Dombeya spp.* had the least decomposed by the end of their study period. Others have investigated differences in the breakdown of leaf litter and found variations in the number of macro invertebrates found on the leaf material of *Rhamus frangula* and *Cornus amomum* during experimentation (Given *et al.*, 2019). Additionally, they found significant differences between the breakdown rates of the two species. Indeed, Ferreira *et al* (2016) found that riparian species diversity, leaf litter decomposition rates, macroinvertebrate richness and fungi species richness were positively correlated. This finding suggests the importance of riparian plant species in influencing the dynamics of nutrient cycling and food webs in river ecosystems. Canhoto *et al* (1996) found that the leaf litter breakdown rates of four studied plant species differed (alder > chestnut > eucalyptus > oak). Other authors have shown the effect of plant species on leaf litter decomposition (Chiba *et al.*, 2015; Mbaka & Schäfer, 2016; Raposeiro *et al.*, 2018). However, the effect of plant species on leaf litter decomposition is understudied for Kenyan rivers and information is lacking for river Kapingazi.

Leaf litter breakdown forms a significant source of energy in river ecosystems since leaves form the largest proportion of biomass (Benfield *et al.*, 2017). Riverine vegetation serves as a primary derivation of organic matter input into river and as such, litter decomposition and nutrient cycling around the river ecosystem is dependent on the type of vegetation around the river (Benfield *et al.*, 2017).

The process of decomposition of leaves has been used as an indicator of river ecosystem health (for example, Silva-Junior *et al.*, 2014; Paudel *et al.*, 2015). Previous studies have primarily focused on water quality and macro-invertebrates (Masese, 2014; Masese *et al.*, 2015; Jacobs's *et al.*, 2017). There is a paucity of information of how anthropogenic activities and plant species influence the process of leaf breakdown in Kenyan streams. This study aimed to assess the influence of anthropogenic activities on leaf litter decomposition rates of *Lantana camara, Psidium guajava* and *Eucalyptus grandis* in Kapingazi River in Embu, Kenya. The hypothesis was that anthropogenic activities around the study sites would influence physicochemical factors of the water and benthic macro invertebrates and therefore influence leaf-litter decomposition rates.

## 1.2 Statement of the Problem

Anthropogenic activities such as settlements, clearing of riparian vegetation, diversion of water for commercial and domestic use, improper disposal of wastes and watering of livestock in the Kapingazi watershed, have been on the rise in the recent past. Alterations due to these activities has the potential to change the Kapingazi river ecosystem changing the physicochemical characteristics of the river water as a habitat for organisms and a source of potable water for the local residents. The changes can alter the physico-chemistry and composition of invertebrate communities, and in the long run lead to changes in associated ecological processes such as leaf litter breakdown. Alterations of leaf litter decomposition, an essential source of external organic matter for nutrient cycling for the food webs in the river ecosystem, may affect the stream health. In the Kenyan context, information on the effect of anthropogenic activities on leaf litter breakdown as a mark of stream health is still scarce. Further, information on leaf litter decomposition of the selected plant species along River Kapingazi is lacking. This study will establish a better understanding on the effects of anthropogenic activities on leaf litter decomposition in Kenyan rivers by studying the decomposition of *Lantana camara*, *Psidium guajava* and *Eucalyptus grandis* along River Kapingazi. The study also aimed to establish the relationship between the effect of human activities on leaf litter decomposition of the aforementioned plant species by measuring the physical and chemical properties of water, involved macro invertebrates at a reach impacted by a small weir, in Kapingazi River.

## 1.3 Justification

This study will add to the current body of knowledge on the relationship between breakdown rates of leaf litter and associated aquatic macro invertebrates in Kenyan rivers. Specifically, leaf litter breakdown rates of *Lantana camara, Eucalyptus grandis,* and *Psidium guajava* were recorded. The associated macro invertebrates were identified and their characterization gave an understanding of the existing ecological conditions of the studied reach of River Kapingazi. The results from this study can therefore be helpful as a basis for biological assessment of the studied reach of River Kapingazi for the sustainable management of the River as a source of water and a habitat for organisms.

### 1.4 Research Questions

a) How do human activities affect the leaf litter decomposition rates of *Lantana camara, Eucalyptus grandis,* and *Psidium guajava* along Kapingazi River?
b) How do leaf litter decomposition rates of *Lantana camara*, *Eucalyptus grandis*, and *Psidium guajava* compare in Kapingazi River?
c) How do leaf litter decomposition rates of *Lantana camara*, *Eucalyptus grandis*, and *Psidium guajava* relate to macro invertebrate shredders and physicochemical variables of water in Kapingazi River?

### 1.5 Objectives

#### 1.5.1 Broad objective

The main objective of this study was to investigate the effect of human activities and plant species on leaf litter decomposition and how these relate with the physic-chemical properties of water and macro invertebrates in River Kapingazi.

#### 1.5.2 Specific objectives

a) To assess the effect of selected human activities on leaf litter decomposition rates of *Lantana camara, Eucalyptus grandis*, and *Psidium guajava* along Kapingazi River.
b) To compare leaf litter decomposition rates of *Lantana camara, Eucalyptus .grandis*, and *Psidium guajava* in Kapingazi River.
c) To examine the relationship between leaf litters decomposition rates of *Lantana camara, Eucalyptus grandis*, and *Psidium guajava* abundance of macro-invertebrate shredders and physical-chemical properties of water in Kapingazi River.

### 1.6 Definition of operational terms

**Allochthonous sources of organic matter:** These are the sources of organic matter derived from outside the aquatic system i.e. the terrestrial system and include leaves, twigs and wood debris.

**Abundance:** The abundance of a species refers to the number of individuals per sample.

**Canopy cover:** This refers to the vertical projection of plant foliage onto a horizontal surface.

**Decomposition:** Decomposition refers to the process by which organic matter comprising of leaves, wood debris, barks and others, convert to fine particulate organic matter and are utilized as energy along food webs.

**Disturbance:** Refers to something that causes a community or ecosystem characteristics, such as species diversity, nutrients output, vertical or horizontal structure to exceed or drop below its common (homeostatic) range of variation.

**Diversity:** This refers to the number of different species in a particular area weighted by some measure of abundances such as the number of individuals.

**Riparian vegetation:** Refers to plant communities along the river margins are called riparian vegetation. Riparian vegetation may be natural or engineered for soil stabilization or restoration.

**Shredders:** These are the aquatic invertebrates that feed on leaf litter after fungi and bacteria, converting it to fine particulate matter that can then be utilized by various organisms in the aquatic food webs, have colonized.

# CHAPTER TWO

# LITERATURE REVIEW

## 2.1 Litter decomposition and the energy cycle in stream ecology

Litter decomposition is the process of nutrient cycling organic matter breaks down to inorganic forms that can be easily utilized as energy in food webs (Alvim *et al.,* 2015).Nutrient cycling implies that nutrients pass across different components of cell, population, community or ecosystem and can be reutilized by some of these components in a continuous cycle. In stream ecology, nutrient cycling entails processes of uptake of inorganic, and sometimes organic forms of matter, transfer from one organism to another in food chains and release back to the stream environment in available forms (Benfield *et al*., 2017). The uptake of inorganic compounds occurs through assimilation of nutrients by primary consumers in available forms such as Phosphates, Nitrites and Nitrates, .

In stream ecology, primary consumers include macro invertebrates such as filterer-collectors that feed on fine particulate unprocessed material dissolved in water (Figure 2.1). For example, mussels feed through sieving available nutrients (Nitrates and Phosphates) in the water (Hoellein *et al*., 2017). The aforementioned authors studied the contribution of mussels (Unionidae) to nutrient cycling at the DuPage River in the USA. They found significant retention of N and P in the tissues and shells of the mussels and suggested that Unionid tissues and shells are potentially significant nutrient sinks in the river. Moreover, by acquiring N and P from the water column, the mussels encouraged denitrification. In a previous study, it had been demonstrated that freshwater mussels can play an important role in nutrient dynamics through nutrient retention in tissues and shells and regeneration of nutrients through excretion and breakdown of shells (Atkinston & Vaughn, 2015).

Small fish and crustaceans also serve as primary consumers feeding on coarse particulate unprocessed material present in water such as leaves, twigs and logs (Peckarsky & Lamberti, 2017). Feeding on coarse particulate organic matter turns it into fine particulate organic matter which is then fed on by other organisms in the water. The small fish and crustaceans are classified as shredders in stream ecology as they turn coarse particulate organic matter (such as vascular plants) to fine particulate organic matter through a process known as fragmentation (Benfield *et al*., 2017). Fish and crustaceans (snails) have an important role in nutrient cycling especially in nutrient poor aquatic ecosystems where they create biogeochemical spots through excretion (Peckarsky & Lamberti, 2017; Bracken. *et al*, 2018; Mormul *et al*., 2018). They also influence nutrient cycling directly through ingestion (Vanni, 2002; Hopper *et al*., 2018). For instance, snails of the *Elimis spp*. were shown to process Nitrogen through growth, excretion and assimilation in the West Fork of the Walker Branch in the USA (Hill & Griffith, 2017). Leaves were the main substrate in the study. The authors found that the snails presented more Nitrogen into the ecosystem through excretion than they accumulated in body biomass.

Shredders are fed on by bigger fish and other organisms known as secondary consumers (Vanni, 2002). Secondary consumers include predator macro invertebrates that feed on primary consumer macro invertebrates such as scrapers and filtering-collectors (Vanni, 2002). Macro invertebrates predators include members of Odonata, Plecoptera, Megaloptera, Trichoptera and Diptera. The predators feed on prey such as simuulidae, Chironomidae and Mayflies (Vanni, 2002). For instance, shrimps were shown to feed on macro invertebrate shredders including chironomids, in Corrego da Andorinha stream in Brazil (Andrade *et al*., 2017). In a study to differentiate the role of shrimps and insect macro invertebrates in leaf litter decomposition, the authors ascertained the role of shrimps as predators of insect shredders. The authors suggested that the shrimps could potentially change the ecological structure of foodwebs in the stream. Predators contribute to nutrient cycling directly through feeding on vascular plants or indirectly through preying on other macro consumers (Carvalho *et al*., 2016). In this study, an invasive crayfish (*Procambarus clarkia*) was demonstrated to not only prey on invertebrate shredders but also to feed on leaf litter. Given that the crayfish *spp*. is invasive, it was suggested that management of *Procambarus clarkia* is important as it has potential to alter the ecological structure of foodwebs in Asia, Europe and Africa due to its increasing density and biomass (Carvalho *et al*., 2016).

Elsewhere, the shrimp, *Neocaridina denticulate* enhanced decomposition directly through fragmentation of *Typha angustifolia* leaf litter in China (Kong *et al*., 2019). The authors demonstrated the role of the shrimp in nutrient cycling by reporting an 81.7% enhanced breakdown of litter attributed to the shrimp's shredding effect. Further, they reported 18.5% increased breakdown rate due to excretion which enriched the experimental environment with P.

Microbes also take part in nutrient cycling through mineralization and as primary producers (such as autotrophic bacteria). The nutrients presented by consumers through excretion are available in both organic and inorganic forms for utilization by microbes (Kong *et al*., 2019). Hence, the rates of excretion and mineralization are critical for microbial activity. In fact, a study by Callisto *et al* (2015) demonstrated that low nutrient concentration in the Indaiá stream of Brazil limited microbial activity and consequently, leaf litter decomposition. Microbial activity involves accumulation of fungi and bacteria on the surface of leaf litter in a process known as microbial conditioning (Yule *et al*., 2016). This process is necessary in the initial stages of litter breakdown since it initiates breakdown by increasing palatability of the leaves through increasing nutritional content to shredders, whose action avails food resources for other consumers in the stream ecosystem (Benfield *et al*., 2017).

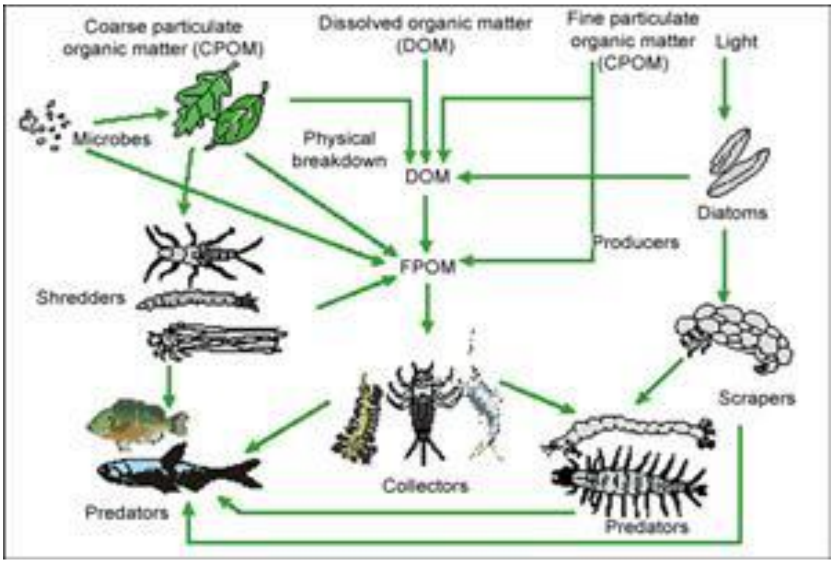


**Figure 1:** Image from web illustrating litter breakdown as part of nutrient cycling in stream ecology

Litter in river ecosystems ranges from leaves, wood debris, twigs, barks, logs, fruits and flowers (Benfield *et al*., 2017). Of these, leaves are, by density, the largest source of allochthonous organic matter for river ecosystems. Leaf wastes provide substrates, organic Carbon and nutrients (including N and P) to stream microbial communities thereby contributing to stream metabolism and nutrient cycling (Benfield *et al*., 2017). Leaf litter decomposition plays a critical role in Carbon and nutrient cycling in river and stream ecosystems (Chauvet *et al*., 2016; Tonin *et al*., 2017). When leaf litter enters the stream, it starts to breakdown in phases (Yule *et al*., 2015).

First, it will undergo leaching where it rapidly loses a significant proportion of its mass in the first 24 to 72 hours (Graça *et al*., 2001; Biasi *et al*., 2013; Bärlocher, 2016). The leaching phase of litter breakdown is where soluble compounds are released into the environment (Biasi *et al*., 2013). Although it is often overlooked when estimating breakdown rates, leaching is important since its exclusion may lead to overestimating microbial and detritivores-mediated decomposition rates (Edwartz, 2018). This author determined a leaching phase that lasted about 48 hours for Roobois tea and 72 hours for green tea. Roobois tea lost about 20% while green tea lost about 40% of its initial weight. The leaching phase is indicated by a more stable phase a few days after rapid loss of mass of the litter has occurred (Ligeiro *et al*., 2010; Pouyat *et al*., 2017; Edwartz, 2018).

The phase of leaching precedes microbial conditioning where microorganisms, accumulate on the leaf surface (Santoja *et al*., 2018). The process of microbial conditioning involves accumulation of bacteria and fungi on the surface of the leaves which makes them more palatable for macro invertebrate shredders (Abelho & Canhoto, 2020). This is attributed to N immobilization in microbial biomass (Bärlocher, 2016). As a result, nutritional quality of the litter increases due to high concentration of N which then improves palatability (Biasi *et al*., 2013). In this study, the authors attributed the rapid breakdown of *S.brasiliensis* to fungi colonization.

Fungi constitute the larger proportion (63%) of total microbial biomass associated with decomposing leaf litter (Abelho & Canhoto, 2020). It has been suggested that fungi colonization is higher in soft, N-rich leaves (Carvalho *et al*., 2016; Abelho & Canhoto, 2020). As a result, consumption of colonized leaves by macro invertebrate shredders increases. Consequently, the effects of other factors of decomposition such as physical abrasion are more pronounced (Contrares *et al.*, 2017; Abelho & Canhoto, 2020). In the study, the authors observed significantly low dry mass remaining in alder as compared to Eucalyptus and Oak (Abelho & Canhoto, 2020). They attributed this to high N-content in Alder which encouraged fungi colonization leading to mass loss due to fungal degradation.

The process of fragmentation is the last phase of decomposition that results in conversion of coarse particulate organic matter (> 1mm) to fine particulate matter (< 1mm) (Garcia-Palatios *et al*., 2016). It is characterized by detritivores which feed on leaf litter by the action of shredding or mining (Graça *et al.*, 2001). Detritivores have been shown to play an important role in leaf litter decomposition (Dobson *et al*., 2002; Raposeiro *et al*., 2018; Tenkiano *et al*., 2018). However, their role seems to be more important in temperate than in tropical aquatic ecosystems where microbial decomposition thrives (Garcia-Palacios *et al*., 2016).

The resulting fine particulate organic matter from litter shredding is availed to other forms of macro invertebrate consumers (Boyero *et al*., 2016). For instance, collectors and gatherers feed on fine particulate organic matter that is deposited in the streambed while filterers utilize the portions of fine particulate matter presented in the water current (Garcia-Palatios *et al*., 2016). Studies have reported a dominance of collector-gatherers and collector-filterers in proportion of macro invertebrates associated with leaf litter decomposition (Gholizadeh & Heydarzadeh, 2017; Fogelman *et al*., 2018; Tsisiche *et al*., 2019). After the organic matter is taken up by the different groups of macro invertebrates, it becomes available to larger consumers (predators) such as fish which feed on the aforementioned groups of macro invertebrates (Carvalho *et al*., 2016; Andrade *et al*., 2017; Bracken *et al*., 2018).

The process of leaf litter breakdown in streams and rivers is influenced by temperature (Fenoy *et al*., 2016; Amani *et al*., 2019), litter quality and stochiometry (Zhang *et al*., 2019) human disturbances around the river/stream, physical abrasion, macro invertebrate shredders and microbes (Boyero *et al*., 2016). Temperature affects leaf litter decomposition in rivers by determining the activity of microbes such as fungi (Boyero *et al.*, 2016). In this regard, some studies have reported increased leaf litter decomposition rates associated with increased temperature changes (e.g Fenoy *et al.*, 2016; Amani *et al.*, 2019; Landeira-Dabarca *et al.*, 2019).

Human disturbances such as clearing of vegetation along the river or stream influences decomposition by reducing litter input into the river or stream due to reduced canopy cover (Boyero *et al.*, 2016). Additionally, clearing of vegetation has been associated with reduced shredders in the river or stream hence reduced fragmentation of organic matter (Mlambo *et al.*, 2019). Further, human disturbance along a river or stream can influence water chemistry which indirectly affects the decomposition process through change in temperature and nutrient concentrations (Boyero *et al*., 2016).

The rate at which leaf litter breaks down depends on the litter quality, natural factors such as temperature and the activity of decomposer communities (Amani *et al*., 2019). The interactions between these factors of decomposition, makes the difference between breakdown rates. Change in environmental factors has been shown to override the other factors of decomposition. For instance, increased levels of atmospheric temperature have been shown to increase the rate of leaf litter rotting in streams across Europe regardless of their chemical characteristics or the action of decomposer communities (Amani *et al*., 2019).

Further, these authors found that elevated atmospheric temperatures increased rates of leaf litter decomposition for the studied stream sites across Europe for both laboratory studies (e.g Taylor & Chauvet, 2014) and field experiments (e.g Martínez *et al*., 2014). It is suggested that rising levels of temperature in the atmosphere over the years has the potential to stimulate microbial activities in stream ecosystems through increasing consumption rates of food resources and breakdown rates of consumed resources by both microbes and detritivores decomposers (Amani *et al*., 2019).

Notably, total decomposition (involving both detritivores and microbe decomposers) is more responsive to elevated temperature levels in the atmosphere than sole decomposition driven by microbes (Amani *et al*., 2019). Additionally, fast rotting litter types are noted to be more responsive to elevating temperatures as compared to slower-decomposing litter types whose breakdown rates remain comparatively slow due to higher degree of recalcitrant material that makes them nutrient-poor for detritivores. Another effect of elevated atmospheric temperatures could be stimulation of leaching out of soluble recalcitrant materials which then hastens the process of litter breakdown (Canhoto *et al.*, 2013).

However, some studies have shown that the effect of elevated atmospheric temperatures on leaf litter decomposition rates depends on the local conditions of the streams. For instance, while increased temperature may stimulate the rate of leaf litter rotting in cold months, it may not necessary be the case for warmer months (Canhoto *et al.*, 2015). Additionally, the effect of temperature on microbial an detritivores activity may only be effective up to a certain optimum beyond which increasing temperatures retard microbial activity hence slowing down waste breakdown (Graça *et al.*, 2015; Amani *et al.*, 2019).

Litter quality and stochiometry influences the process of breakdown in a river or stream by determining the rate of breakdown, and the affinity for macro invertebrate shredders (Leite-Rossi *et al.*, 2016; Ferreira *et al.*, 2016; Zhang *et al.*, 2019). For instance, it has been demonstrated shredder chironomidae larvae are attracted to leaf species with less lignin and secondary compounds (Leite-Rossi *et al.*, 2016). Macro invertebrate shredders influence the process of leaf litter decomposition through their abundance, density and species diversity of shredders available for the process of fragmentation (Leite-Rossi et al., 2016; Ferreira *et al.*, 2016). Microbes on the other hand determine shredder affinity to organic matter (Leite-Rossi *et al.*, 2016; Ferreira *et al.*, 2016; Wootton *et al.*, 2019).

The initial litter quality plays an essential role in the breakdown of leaf wastes in streams. The importance of litter chemistry is such that high nutrient content (N and P) in litter promotes growth of microorganisms and detritivores in stream ecosystems (Tenkiano *et al.*, 2018). By acting as a food resource, high nutrient content encourages microbial and detritivores-driven litter breakdown. In their study in Guinea, West Africa, the aforementioned authors found that the leaf litters of interest (i.e *Alchornea* and *Pterocarpus* spp.) both had high levels of Nitrogen and Phosphorus levels. This attributed to the fast decomposing nature of the species in all the studied streams. However, it was noted that *Alchornea* spp. decomposed twice as fast as *Pterocarpus spp.* This observation was attributed to the significantly higher (4-fold) levels of Calcium in *Alchornea spp.*

Micronutrients such as Magnesium and Calcium have been proved to be of great importance in stimulating litter decomposition across biomes (García-Palacios *et al.*, 2016). In fact, the micronutrients; Ca and Mg, have been termed as more important traits for leaf litter decomposition as compared to initial leaf carbon to Nitrogen and Lignin to Nitrogen ratios (García-Palacios *et al.*, 2016). Calcium encourages the growth of fungi which is necessary for initializing the process of decomposition through microbial-conditioning of leaf litter. Magnesium on the other hand, has been strongly correlated with leaf litter decomposition as it is a major constituent of invertebrate diets (Santoja *et al*., 2019). By mediating the process of priming and detritivores growth and activity, Ca and Mg stimulate the process of leaf litter rotting.

The type of decomposition in stream ecology depends on the agent associated with litter breakdown. For instance, microbial litter breakdown occurs where microbes such as fungi and bacteria mediate the process of decomposition. Decomposition can also be macro-invertebrate mediated where shredders cause fragmentation of organic matter (Pascoal & Cássio, 2004). Decomposition can also be in the form of physical abrasion, caused by flow velocity of the water (Hubai *et al*., 2017). The friction caused by water waves also causes fragmentation of litter present in the stream. Decomposition then occurs because of microbial activity, macro-invertebrate intervention or by physical abrasion (Benfield *et al.*, 2017). However, the process of decomposition is a sum of all of the driving factors and that while one factor may be more important in one region it may be insignificant in another (Benfield *et al*., 2017).

## 2.2 Influence of anthropogenic activities on leaf litter decomposition

Anthropogenic activities in rivers refer to those activities that are carried out by human beings in and around rivers (Crook *et al.*, 2015). For instance, construction of dams, infrastructure, clearance of riparian vegetation, introduction of exotic species, water abstraction, cultivation in the associated watersheds, trampling by livestock, fishing among others (Dodds *et al.,* 2013). These human activities have impacts on the river's biodiversity and ecosystem services and causes pollution, flow modification, water stress among others. The effect of these impacts is that they compromise the ability of rivers to supply the ecosystem services (Dodds *et al.,* 2013). Anthropogenic activities have also caused degradation that threatens river ecosystems through loss of habitat, altered hydrology, invasive species, climate change and alteration of energy cycling in the long run (Crook *et al*., 2015).

The extent of a freshwater ecosystem includes the catchment area from which water and organic and inorganic material are drawn (Dudgeon *et al.,* 2006). The positioning of rivers in the landscape, therefore, makes them receptors of wastes, sediments, and pollutants in runoff. Anthropogenic activities, therefore, have the potential to influence litter decomposition by causing changes in environmental conditions of river ecosystems. For instance, heavy metal contamination of rivers in watersheds where mining activities were carried out was found to inhibit litter decomposition in studies conducted between 1978 and 2014 based on a meta-analysis (Ferreira *et al*., 2016). This was due to the adverse effect on invertebrate shredders. Litter decomposition rates were seen to reduce by 40% in fungicide-polluted streams in Spain (Fernández *et al*., 2015).

In a study to assess the effect of replacement of deciduous forest plantation with *Eucalyptus grandis* plantations on Iberian Peninsula stream ecosystems, it was established that the rate of leaf litter breakdown was slower in eucalyptus streams than in deciduous forest streams (Ferreira *et al.,* 2015). This was attributed to the poor quality of *Eucalyptus grandis* leaf waste, which resulted in decreased affinity for shredder invertebrates. It was concluded that this is a sign of poor stream health (Ferreira *et al.,* 2015).

Eutrophication due to an addition of nutrients in streams from human activities such as industrial, domestic and agricultural activities in and around streams has been seen to affect leaf litter decomposition (Gulis *et al.,* 2006). These researchers observed that moderate eutrophication stimulation led to faster decomposition rates in affected streams as compared to the reference streams, by about 2.3% to 2.7%. In another study to determine the effect of increased nutrient concentration on leaf litter breakdown, it was demonstrated that decomposition rates were higher in enriched streams in comparison to unaffected streams (Tant *et al*., 2015). This was attributed the observation to the significant increased colonization of fungi due to nutrient enrichment in the experimental streams. The result of this was increased palatability of the litter by shredders which then accelerated the rate of breakdown.

Human Activities around rivers cause changes in water chemistry that alter the structure, diversity, and activity of biological communities that control decomposition (Martins *et al*., 2015). For instance, Aluminum was found to increase the pH of streams affected by acidification hence inhibited litter decomposition (Ferreira & Guerold, 2017). Additionally, different locations of a river have different characteristics due to differences in the influence of the physico-chemistry by human activities (Benfield *et al*., 2017).

In a study to determine decomposition of five leaf litter species in River Danube in Hungary, significantly higher differences in the rate of mass loss was experienced for litter bags in a depositional zone as compared to those in an erosional zone of the river (Agoston-Szabo *et al.,* 2016). The disparities were attributed to the differences in habitat conditions mainly the substrates of the sites. Further, it was reported that the chironomidae larvae involved in decomposition process differed significantly in both sites (Agoston-Szabo *et al.,* 2016). The findings of the study showed how differences in site location can result in habitat conditions which then influence the process of leaf wastes breakdown by determining dynamics of decomposer communities.

## 2.3 The relationship between plant species and leaf litter decomposition

The process of leaf litter decomposition has been described as a sequential mechanism initiated by the leaching of soluble compounds, followed by colonization by fungi or bacteria which then facilitates invertebrate shredders' degradation (Garcia-Palacios *et al.,* 2016). However, decomposition rates differ due to the influence of species-specific plant traits (Alvim *et al*., 2015). These traits include leaf roughness, phenolic content, and existence of chemical compounds such as waxes and oils (Garcia-Palacios *et al.*, 2016). For instance, the presence of chemical compounds such as oils secreted by some leaf species such as *Blepharocalyx cruckshanksii,* are important as they provide hydrophobicity to the leaf affecting the ability of microbial colonizers to penetrate the leaf tissue (Garcia-Palacios *et al.*, 2016). The oils, therefore, slow down or inhibit the litter decomposition process.

Ontoya and Igueroa (2017) studied the decomposition of a native (*Pinus radiate*, *Blepharocalyx cruckshanksii)* and a non-native species (*Eucalyptus globulus*) in a freshwater wetland in Chile. The leaves of *Eucalyptus globulus* had the highest decomposition rate while those of Temu *(Blepharocalyx cruckshanksii)* had the lowest decomposition rate. This was explained by the difference in leaf toughness, which was greater in the native species as compared to the exotic species. It would be expected that *Eucalyptus globulus* would have the slowest decomposition rate since it has phenolic compounds whose anaerobic degradation can initiate increment of compounds such as quinones that are highly toxic for decomposer communities in aerobic systems. However, this was not the case given the low oxygen concentration in forested wetlands. The greatest decomposition were during the first 15 days for both native and non-native species which coincides with the liberation phase for soluble compounds which is the phase where the greatest weight loss occurs in foliar decomposition (Ontoya & Igueroa, 2017).

In an evaluation on the decomposition rate of cottonwood, a native species and Russian olive, a non-native species on an inundation gradient in the Rio Grande, USA, the authors found no significant difference between the decomposition rates of the leaf litter (Harner *et al.,* 2009). This was in contrast with the study of Kuglerova *et al* (2017) who studied five native and five non-native species and found that the three fastest decomposing species were the Himalayan balsam, Russian olive, and purple loosestrife all of which are invasive species. The slowest decomposition rates were seen in the salal (non-native species) Japanese Knotweed (native species) and sword fern (native species). These differences were driven by discrepancies in litter constitution that is Nitrogen, lignin content, and structural carbohydrates.

Elsewhere, it was established that the leaf litter properties of invasive species are what slows their decomposition rate in comparison to that of native plant species of the

Mediterranean ecosystems in Spain (Godoy *et al.*, 2010). These researchers found that alien species decomposed slightly slower than native species due to large initial lignin and Calcium content. Dobson *et al* (2004) found that the fastest decomposition rate (Vanguera *spp.*) was five times greater than the least (Dombeya *spp.*) decomposition rate of the major riparian tree species in their study at Njoro River in Kenya. They demonstrated high microbial activity in the river and concluded that the low biomass of invertebrate shredders in the river cannot be due to the poor quality of leaf litter that enters them.

## 2.4 Leaf litter decomposition in relation to water physicochemical variables, and macro invertebrates

Leaf litter decomposition is governed by factors such as the availability and properties of litter (Liu *et al.*, 2017), abundance and diversity of decomposing organisms such as shredders and microorganisms (Grace *et al.*, 2015) and physicochemical variables of water such as temperature, acidity, velocity and nutrients (Boyero *et al.*, 2016; Ferreira & Guerold, 2017). Physical abrasion influences the process of leaf litter decomposition by enhancing the process of fragmentation hence accelerating breakdown rate (Wootton *et al.*, 2019). The type of riparian vegetation around a river, determines the type of leaf litter available for decomposition in the river ecosystem (Benfield *et al.*, 2017). Leaf traits such as roughness and presence of secondary chemicals will determine the process of microbial conditioning and hence the palatability of litter to invertebrates (Garcia-Palacios *et al.*, 2016).

Microbial conditioning or priming refers to the initial process of litter decomposition where microorganisms accumulate on the surface of the leaf (colonize) shredders feed on them, causing the process of fragmentation (Benfield *et al.*, 2017). In this process, microbial decomposers such as hyphomycete, convert litter into a more palatable form for invertebrate detritivores.

Normally, the invertebrates will prefer to feed on conditioned leaves than those that are freshly fallen (Garcia-Palacios *et al.*, 2016). Indeed, other authors established that fungal conditioning is a more important determinant of the selective feeding habits of millipedes than the other traits of the leaves such as toughness or chemical properties

(Harrop-Archibaid *et al.*, 2016). Leaf litter in aquatic ecosystems does not only serve as a food substrate but it also serves as habitat for aquatic communities (Kennedy, 2016). Clearance of vegetation along the river or stream would therefore impact on the macroinvertebrate communities in it (Kennedy, 2016).

The physcochemical properties of water have also been known to influence leaf litter decomposition (Kennedy, 2016). These properties include the water temperature, conductivity and dissolved Oxygen. Temperature affects leaf litter decomposition by way of stimulating fungal or microbial activity. This is because high temperature increases growth, abundance, emergence and sex ratios of microbes which then enhance the mechanism of priming of leaf organic matter increasing invertebrate driven leaf litter decomposition (Ferreira *et al.*, 2014; Graca *et a*l., 2016; Kennedy, 2016).

It is suggested that the mean annual water temperatures have been on the rise and this could potentially increase leaf litter decomposition rates in some streams (Kaushal *et al.*, 2010; Boyero *et al.*, 2012). This is already happening as Follstad *et al* (2017), affirmed that increased stream temperature increased leaf litter decomposition rates in previously published studies. Related studies have shown that the effect of water temperature interacts with other factors influencing leaf litter decomposition. For instance, ambient water temperature influences detritivores densities consequently affecting decomposer macro invertebrate-mediated leaf litter decomposition (Griffiths & Tiegs, 2016).

However, change in environmental factors has been shown to override the other factors of decomposition. For instance, increased levels of atmospheric temperature have been shown to increase the rate of leaf litter rotting in streams across Europe regardless of their chemical characteristics or the action of decomposer communities (Amani *et al.*, 2019).

Further, these authors found that elevated atmospheric temperatures increased rates of leaf litter decomposition for the studied stream sites across Europe for both laboratory studies (e.g Taylor & Chauvet, 2014) and field experiments (e.g Martínez *et al.*, 2014). It is suggested that rising levels of temperature in the atmosphere over the years has the potential to stimulate microbial activities in stream ecosystems through increasing consumption rates of food resources and breakdown rates of consumed resources by both microbes and detritivores decomposers (Amani *et al.*, 2019).

Notably, total decomposition (involving both detritivores and microbe decomposers) is more responsive to elevated temperature levels in the atmosphere than sole decomposition driven by microbes (Amani *et al.*, 2019). Additionally, fast rotting litter types are noted to be more responsive to elevating temperatures as compared to slower-decomposing litter types whose breakdown rates remain comparatively slow due to higher degree of recalcitrant material that makes them nutrient-poor for detritivores. Another effect of elevated atmospheric temperatures could be stimulation of leaching out of soluble recalcitrant materials which then hastens the process of litter breakdown (Canhoto *et al.*, 2013).

However, some studies have shown that the effect of elevated atmospheric temperatures on leaf litter decomposition rates depends on the local conditions of the streams. For instance, while increased temperature may stimulate the rate of leaf litter rotting in cold months, it may not necessary be the case for warmer months (Canhoto *et al.*, 2015). Additionally, the effect of temperature on microbial an detritivores activity may only be effective up to a certain optimum beyond which increasing temperatures retard microbial activity hence slowing down waste breakdown (Graça *et al.*, 2015; Amani *et al.*, 2019).

Besides, water temperature will influence microbial-mediated leaf litter decomposition in a stream through its influence on the adaptation of microorganisms (Strickland *et al.*, 2015). Water temperature has also been seen to interact with litter quality in influencing decomposition. For example, Makkonen *et al* (2012) studied leaf litter decomposition for sites in the subarctic, temperate, Mediterranean and moist tropical biomes. They found consistency in the breakdown rates of the studied leaf litters only differentiated by their quality parameters such as lignin content and C:N ratios. Interestingly, the breakdown rates for leaf litters in the tropical region site were consistently low as considered to all the other biomes considered in the study.

Following the previously noted interactions of temperature with other factors, Follstad *et al* (2017), asserted that leaf litter decomposition mediated by detritivores is more sensitive when it was coarse mesh (allows shredder activities) used as compared to when fine mesh (represents microbial-mediated decomposition) is used. Further, it was observed that latitude not altitude, predicted temperature sensitivity for detritivores-mediated decomposition. It was noted that leaf litter decomposition rates were generally higher at higher latitudes. This is possibly due to the reported high diversity and density of detritivores at temperate regions as compared to tropical regions where the role of detritivores in litter decomposition has been noted to be less important (Johnson *et al*., 2017). Additionally, litter quality predicted water temperature sensitivity, where litter that was low in Carbon and high in recalcitrant material such as polyphenols had higher breakdown rates as compared to litter high in C and low in recalcitrant materials (Follstad *et al*., 2017).

Low pH deteriorates the process of decomposition due to decreased microbial activity. Low pH inhibits decomposition by excluding pH-sensitive macro invertebrate decomposers (Kennedy, 2016). Moreover, Ferreira and Guérold (2017) demonstrated that decreased pH due to fungicide toxicity led to decreased fungal biomass and led to a decrease of up to 40% in microbial leaf litter decomposition for affected sites as compared to unaffected sites in the study. Besides, these studies found that bacterial biomass increased in the study and attributed this to increased nutrient concentration in the studied streams.

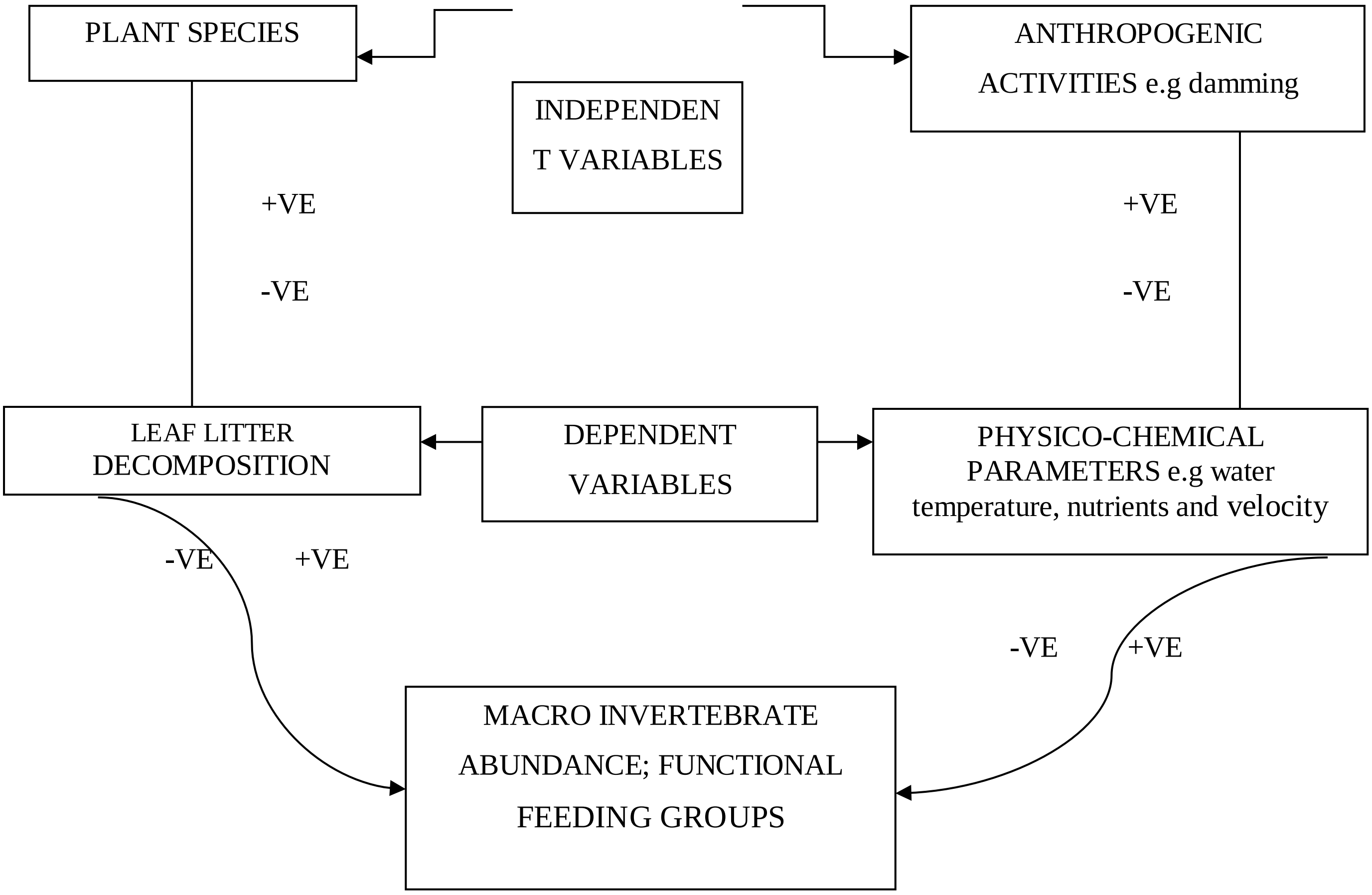
Furthermore, fungal biomass is more important in microbial decomposition of leaf litter (Duarte *et al*., 2010; Pascoal & Cássio, 2004). High conductivity of the water facilitates optimal enzyme activity and increased shredder biomass which then increases decomposition. Low dissolved Oxygen reduces litter decomposition by reducing fungal biomass therefore reducing microbial driven decomposition (Canhoto *et al*., 2013). It has previously been established that low dissolved Oxygen levels induce death of particular invertebrate species. For instance, the mayflies' species were found at risk of mortality at saturation levels of 15-48% while chironomidae *Tanytarsus dissi-milis* tended to die at saturation levels below 6.43% with increased drift at saturation levels below 10% for all taxa studied (Connolly *et al*., 2004). It has been suggested that reduced Oxygen concentration due to increased fine sediments can slow down the process of leaf litter decomposition which inhibited fungal activity (Bruder *et al.*, 2016). Further, it has previously been demonstrated that aquatic invertebrates in the tropics require more dissolved Oxygen to survive due to the higher temperature as compared to temperate regions (Kennedy, 2004).

The diversity and abundance of decomposer communities in a river are largely determined by the type of Riverine vegetation around the river (Benfield *et al.*, 2017). Alteration in riparian vegetation will, ultimately, alter the decomposition process in a river due to the effect on invertebrate shredders and other decomposer communities (Ferreira & Guerold, 2017; Kuglerova *et al.*, 2017). For example, the change in native tree species of forests caused 40% reduction in leaf litter decomposition for associated streams in São Miguel island, Azores archipelago (Ferreira *et al.*, 2017). Similarly, the effect of litter diversity on decomposition has been reported to vary with abundance and biomass of shredders (Sanpera *et al.*, 2009).

Shredders are the decomposer communities that convert coarse particulate organic matter to fine particulate organic matter (Moulton *et al.*, 2010). Studies in temperate regions have well determined that indeed shredders play a significant role in leaf litter decomposition (Graca *et al.*, 2015). However, in tropical regions, other studies have reported a paucity of shredders and hence suggest that microorganisms and other factors such as litter quality play a bigger role in leaf litter decomposition as compared to macro invertebrate shredder activity (Buyero *et al.*, 2012; Leite-Rossi *et al.*, 2016). Studies in the tropics find dominance of small collectors of the subfamily chironomidae and consistent presence of a high relative abundance of predators (Dudgeon & Wu 1999; Dobson *et al.*, 2012; Wantzen *et al.*, 2008). This suggests that macro invertebrates may not only use leaves as food but as a suitable habitat especially for collectors since leaves act as particulate organic matter.

Other authors have investigated the effect of impoundments on the breakdown of alder leaves in Palatinate Forest in the South East of Germany (Mbaka & Schäfer, 2016). This study found that leaf litter decomposition differed significantly for immediate upstream sites from those immediate downstream, further upstream and further downstream sites. They further observed that physicochemical variables of the study sites were not significantly correlated to leaf litter decomposition ($p$ = 0.06). They however found that the immediate upstream sites had greater phosphate concentrations in comparison to other sites. With regards to macro invertebrates, Mbaka and Schäfer (2016) found that shredders were dominant (56.2%) of the functional feeding groups encountered. Further, the shredders were positively correlated (r= 0.53) to the rates of leaf litter decomposition.

In another study to determine the influence of small-sized macro invertebrates in River Danube in Hungary, the authors found that larvae from the chironomidae family were the most abundant (100%) of the macro invertebrates encountered (Agoston Szabo *et al.*, 2016). The authors found positive correlation between breakdown rates of the studied leaf species and the chironomidae abundance. Chironomidae larvae have been described as shredders which means they cause fragmentation of coarse particulate organic matter by feeding on leaves that have been conditioned my microorganisms. The results from the aforementioned authors show the essential role played by macro invertebrate shredders in leaf litter decomposition as they accelerate the breakdown of organic matter hence influence nutrient cycling dynamics.



Legend: -ve –negative +ve - positive

**Figure 2: Conceptual framework**

The above framework shows the relationship between te depenedenmt and independent variables of the study. The points of observation are representative of the human activities' effect. The rates of leaf litter decomposition are dependent on the water quality parameters and the composition and structure of macro invertebrates,

# CHAPTER THREE
# MATERIALS AND METHOD

## 3.1 Description of study location

The study was carried out in River Kapingazi. The river is on the South East of Mount Kenya. It is the major tributary of Rupingazi River that drains into the Tana River, which is Kenya’s largest river (Balana *et al.*, 2011). Kapingazi river is found at the upper catchment region of the Tana River basin where human-induced ecosystem degradation issues such as deforestation, poor cultivation, overgrazing, increased subsistence farming activities, erosion of river banks and planting of *Eucalyptus grandis* trees on the river banks have been observed (Balana *et al.*, 2011).

### 3.1.1 Description of Climate, Soils and Drainage of the Area

The river catchment covers an area of 61.23 square kilometers and is 27 km long. It is found between the altitudes of 1200 and 2100m above sea level. The average annual rainfall of the river catchment area is between 1200mm and 1800 mm and mean annual temperatures vary from a mean maximum of 27.1 °C to a mean minimum of 21.2 °C. The catchment area has a bimodal rainfall pattern with heavy rains between March and May and the dry period from June to September. The major soil types in the area are Eutric Astrosols and Nitisols (Gachimbi, 2002).

## 3.2 Site Characterization

Five study sites were randomly selected along the river on an approximate 10km section of the river that had varied human disturbance near the University (Figure 3). The commonly observed human activities and effects around the study sites were cloth-washing, trampling on the river bed, bathing and drawing water for domestic use, clearing of vegetation on the riverbanks, waste disposal, construction of residential buildings, watering points and farming activities such as maize and banana planting as well as tree nurseries. These sites were selected during the preliminary visit to the study area.

Figure 3: A map of the study sites along River Kapingazi; Google Maps, 2019

Legend: FUS-Further upstream site, IUS-Immediate upstream site, Weir-Weir area, IDS-Immediate downstream site, FDS-Further downstream site

### 3.2.1 Weir area

A constructed weir (WA) (00° 29' 57.1" S – 037° 27' 41.0" E) was the defining human activity at this site (plate 1). Besides, there were maize, banana and sedge grass growing on either side of the river the left side had a student hostel. In-stream, the weir was characterized by 97% silt throughout the study period. This site was 100% pool and an estimated canopy coverof 40%.

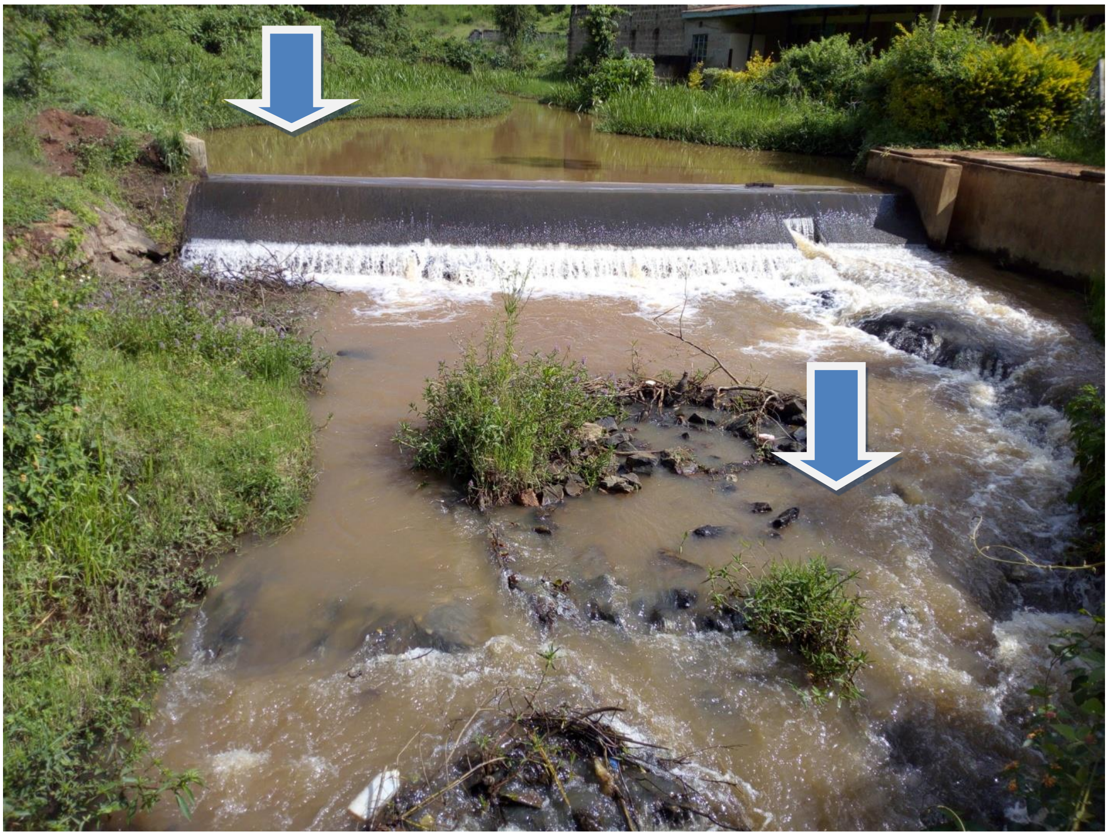


**Plate 1:** Weir site and immediate downstream sites along River Kapingazi. The arrows show the points of leaf packs' installation for the weir and downstream sites respectively

### 3.2.2 The immediate downstream site

The immediate downstream site (IDS) (00° 30' 17.8" S – 037° 27' 46.8" E) was selected right after the weir to observe effects of the weir. The distance between the point of leaf pack installation at this site and that at the weir area was 0.37 km (see plate 1 above). The study site was next to a bridge.

### 3.2.3 The immediate upstream site

The immediate upstream site (IUS) (00° 29' 57.1" S – 037° 27' 41.0" E) is the area immediately upstream of the weir site. It was 1.6 km from the weir point where part of the litter was placed. On the left side was a nursery for vegetables and a small scale farm of kales and spinach. *Lantana camara* was the commonest species observed along the bank on this site. Beyond the banks, was a students' hostel (next to the weir).

There was a dumping site observed just next to the vegetables' farm. Throughout the study period, fetching water and washing clothes along the river bank was observed. The right bank was characterized by sedge grasses, *Psidium guajava* trees, and a Napier grass plantation. There was also a stand of *Croton megalocorpus* and *Grevellia robusta*. Among these was a plantation of maize and bananas. The area of installation of the experimental bags was a shallow pool near the leaf bank and was surrounded by bedrock (see plate 2).

**Plate 2:** Immediate Upstream site: the arrow shows the point of leaf packs' installation

### 3.2.4 The further downstream area of the weir

The further downstream site (FDS) (00° 30' 22.2" S – 037° 27' 48.9") site was about 2.17 km away from the weir point of leaf installation. It was characterized by a large *Eucalyptus grandis* plantation on the right bank. Other species observed included the bougainvillea and calliandra . The left bank activities included a tree nursery where white Easter flowers were planted. There was a tree stand for *Gravillea robusta*.

The area where the bags were installed was right below a crossing bridge. The area was a small pool next to the right bank surrounded by rocks (see plate 3).

Plate 3: Further downstream site: the arrow shows the point of leaf packs' installation

### 3.2.5 The Further upstream site

The further upstream site (FUS) (00° 30' 15.2" S – 037° 27' 42.6") was the farthest site from the weir area site. The distance between the point of installation at this site and that at the weir area was 4.53Km. The bag installation point was a pool surrounded by large boulders that slowed down water movement from upstream. It is along Meru-Embu highway with a bridge nearby where the river crosses. The site was not shaded by surrounding canopy. On both sides of the river bank was a thicket formed by *Lantana camara* was observed. *Psidium guajava* trees were also observed on both sides of the river and on the left bank was sedge grass. Throughout the period of the study people were observed to wash clothes. From time to time, animals (including cows and sheep) came to the site for watering.

The experimental bags were placed in a pool created by surrounding bedrock beyond which was riffles (see plate 4).

Plate 4: Further upstream site: the arrow shows the point of installation of the leaf packs

### 3.3 Environmental variables

The variables measured included the dissolved oxygen, nutrient content (Nitrates, nitrites, total phosphorus and Phosphates), total dissolved solids, conductivity, dissolved Oxygen, pH of water, water discharge (Q) and water velocity. The experiment was carried out during the expected short rains period (between September and October, 2018), and the dry period (between November and December, 2018).

### 3.4 Preparation and placement of leaf litter

Fresh leaves of *Lantana camara, Eucalyptus grandis* and *Psidium guajava* were collected directly from the trees at the various study sites following methods by Uieda and Carvalho (2015). Whole and undamaged leaves were collected from one tree to minimize interspecies variations. The leaves were air dried for 5 days until they reached a constant dry mass (DM) and stored in a dark room. They were laid on the floor, spread out and turned over severally to ensure proper drying to a constant mass. Portions of the dried leaves weighing five grams (± 0.05 g) were put in leaf packs after weighing atop an analytical weighing balance

Packing bags (measuring 15 cm by 15 cm) were then be constructed from nylon mesh wire (1mm). The size of the mesh openings ensured that they were large enough to allow aerobic conditions and access of medium sized-invertebrates to the leaves but at the same time reduced loss of leaf fractions during handling and transportation to the study sites. Additionally, the mesh was preferred due to the small size of leaves of *Lantana camara*. On 9th September and on 27th November 2018, twelve (12) packs of each leaf species enclosed in the mesh bags were prepared for each site. An extra set of 45 bags (3 bags x 3 species x 5 sites) wasprepared to correct for initial mass losses due to handling and leaching in the first 48 hours of deployment. All the packs were then wetted using de-ionized water in order to prevent fragmentation losses. The leaves were then transported to the stream in a cooler box. A total of 180 leaf packs were deployed in the chosen sites (i.e 3 bags x 3 leaf species x 5 sites x 4 sampling dates) at each time (i.e September/October and November/December). The leaf packs were installed by tying them onto wooden strips which were then secured safely above the streambed by rocks at the sites (plate 5). The packs were anchored in shallow ends near the river bank where they would not be shifted by high flows experienced in the middle reaches of streams and rivers. The extra set of packs was transported back to the laboratory after 48 hours and reweighed to account for handling and transportation weight losses.

**Plate 5: Leaf packs installed at the further downstream site**

The incubated leaf packs were retrieved in sets of three every seven days from the stream by placing a 250-micrometer mesh dip net under the leaf packs and transferring the contents of the packs and net into a labeled zip-lock plastic bags. The experiment lasted for 28 days in both times of experimentation (September, November/ December). The retrieved litterbags were returned to the laboratory in a cooler box. The contents from each bag were rinsed thoroughly using distilled water on a 250-micrometer sieve, to remove sediments and associated macro-invertebrates. The litter was then oven dried for 24hours at 60 °C and weighed to determine leaf dry mass to the nearest 0.1g. The rest of the material was combusted at (500 °C) for 12h and weighed to determine ash-free dry mass (AFDM).

The collected fauna was preserved in 70% ethanol for later analyses to determine the functional feeding groups' abundance Abundance was expressed as the number of invertebrates present in each litterbag. The macro invertebrates were identified using X10 magnification using dissecting microscopes. The invertebrates were identified to the family level and allocated to appropriate functional feeding groups (gatherers, collectors, filterers, shredders) according to method in Tachet *et al* (2003).

### 3.5 Leaf litter decomposition

Using the dry masses obtained from section 3.4 above, the mass based leaf litter decompositionrates (g/day) was determined by calculating the difference in mass between leaf litter consecutive observed masses and dividing by the initial mass for that period. The overall rate of decomposition was computed from the masses at the start and at the end of the observation time. For both cases the mean of mass from the three leaf litter samples per plant species and per site (representing human activity) were calculated for use. The data was collected every seven days for four weeks. Other means of leaf litter mass and hence decomposition rate determined were for overall for the times of observation for each site (irrespective of plant species) and for each plant species (irrespective of site). The synergetic effects of the two variables of study (human activity and plant species) were also tabulated to show the interaction on the output variable i.e decomposition rate of the leaves. The results were presented in tables. Leaf litter decomposition was presented as percentage dry mass remaining per day (DMd$^{-1}$.

### 3.6 Determination of nutrients and physicochemical variables

Sampling of the water physic-chemical parameters was done on the first day of installation and on every time of retrieval (i.e every seven days). During sampling of water for quality analysis, water was collected from different points of each site to get a composite sample. Three replicates were collected on each site. The water samples were collected by immersing a bottle pre-cleaned with distilled water and then transferred into pre-cleaned plastic sampling bottles. The samples were then transported to the laboratory and preserved at 4 °C before being sent to the Water Resources Authority (WARMA) laboratory for analyses. In the laboratory, the samples underwent calorimetric analysis for determination of $PO_4^{3-}$, $NO_2^-$, and $NO_3^-$. Standard spectrophotometric methods were used as described by Apha (2005).

Nitrate-Nitrogen ($NO_3^-$) was determined using the sodium-salicylate method (Apha, 2005). Nitrite-Nitrogen ($NO_2^-$) determination was carried out using the reaction between sulphanilamide and N-Naphthyl-(1)-ethylenediamine-dihydrochloride to give an intense pink color of nitrite which was determined spectrophotometrically at 545nm wavelength.

Total Phosphorus was determined using the ascorbic acid method where 90-minute persulphate digestion of unfiltered water samples before analysis preceded the Total Phosphorus analysis. The ascorbic acid reduction method, which is based on the formation of phosphor-molybdate complex, was adopted for analysis of soluble reactive phosphorus ($P0_4^{3-}$-P) and total phosphorus (TP) (Apha, 2005). Readings from a spectrophotometer gave absorbance at a wavelength of 650nm for respective nutrient determinations. $P0_4^{3-}$ -P was determined using molybdate ascorbic acid method that results in a formation of an intense blue color and measured at a wavelength of 880nm (Apha, 2005).

Physicochemical variables of the river water were measured *in-situ* using portable meters. The measured variables were: electrical conductivity, pH, TDS, dissolved oxygen concentration and temperature. Temperature (°C) of the water was measured using a combined TDS/temperature meter, Jenways model 4075 (Jenway Essex, UK) with the reading of TDS corrected to 25 °C. Dissolved oxygen (mg/l), pH and conductivity ($\mu Scm^{-1}$) readings were taken using a calibrated Jenway 3405 electrochemical analyser (Barloword Scientific Ltd, Dunmow, Essex, UK). Average water depth at the various sites was computed from measurements, taken on a transect across the river channel where the litter bags were placed for observation. Current velocity was estimated by timing a float over a distance of 5metres (Mbaka & Schäfer, 2016). A stopwatch was used to measure the time taken by the float to travel the distance. The stream velocity was estimated by dividing the distance by the interval time (Hauer & Lamberti, 2011). Five replicate trials following this method were conducted and averaged for each flow velocity estimated. Water discharge (Q) was calculated from the product of velocity and cross-sectional area of the river by timing the flow of buoyant sticks over a 5meter stretch and discharge calculated using the formula

$Q=V/t = (L\ xA)/t$…………………………………………………………………(i)

Where:

$Q$ = river discharge ($m^3/s$)

$V$ = volume of water passing the river section where the measurement(s) was made (m)

$t$ = time recorded for the float to cover the distance (s)

$L$ = length covered by the float (m)

$A$ – Cross section area of the river section where the readings were taken ($m^3$)

The same float was used for all sites throughout the study period. This ensured uniformity in water velocity measurement, thereby reducing subjectivity for this variable in the study. Benthic substrates were also assessed visually at each study site and categorized as “boulders” >250mm, “gravel” >10-64mm, “sand and silt” <0.06-2mm (Mbaka & Schäfer, 2016). Canopy cover was also estimated visually (Mbaka & Schäfer, 2016).

### 3.7 Invertebrate analysis

The collected fauna was conserved in 70% ethanol for analyses to determine abundance (Monroy *et al.*, 2016; Benfield *et al.*, 2017). The invertebrates were identified under X10 magnification light and dissecting microscopes. To ensure that the process was fast, with less errors and easier use of an homogenous level of identification, the invertebrates were classified to the family level (Monroy *et al.*, 2016; Tonin *et al.*, 2018), even though some authors have found that a higher taxonomic resolution such as genus or species increases the level of sensitivity to human disturbance and is especially relevant to detect small quality changes (e.g.

Bailey *et al*., 2001; Waite *et al*., 2004; Feio *et al*., 2006**)**. Moreover, River Kapingazi is not identically studied for aquatic invertebrates and therefore it was the safer approach in order to avoid potential mismatch in identification over time. The identification keys of Hauer and Resh (2017) were used and the invertebrates were grouped into functional feeding groups (scrapers, gatherers, collectors, predators, shredders) according to Tachet *et al* (2003).

### 3.8 Data analysis and presentation

The data was managed in excel data sheets and analyzed in R statistical package (Version 4.0.1). Mean comparison was done using ANOVA while mean separation were done using Tukey test. The Pearson correlation test was used to determine the relationship among physico-chemical parameters and between leaf litter decomposition rates and physico-chemical parameters. The results were presented in tables and figures to infer trends and variations among sites, plant species and times of observation.

# CHAPTER FOUR

# RESULTS

## 4.1 Leaf litter decomposition

### 4.1.1 Leaf Litter Mass Loss for Septemeber and December 2018

The residual mass of the leaf litter decreased over time in both months (Figure 4), implying that the litter decoposed progressively over time.The decompositin rate of the leaf litter was generally higher during the month of December 2018 compared to the month of September 2018. This shows that the prevaining weather conditions had an influence on the rate of decomposion, with the wet conditions in December promoting relatively faster decay rates.

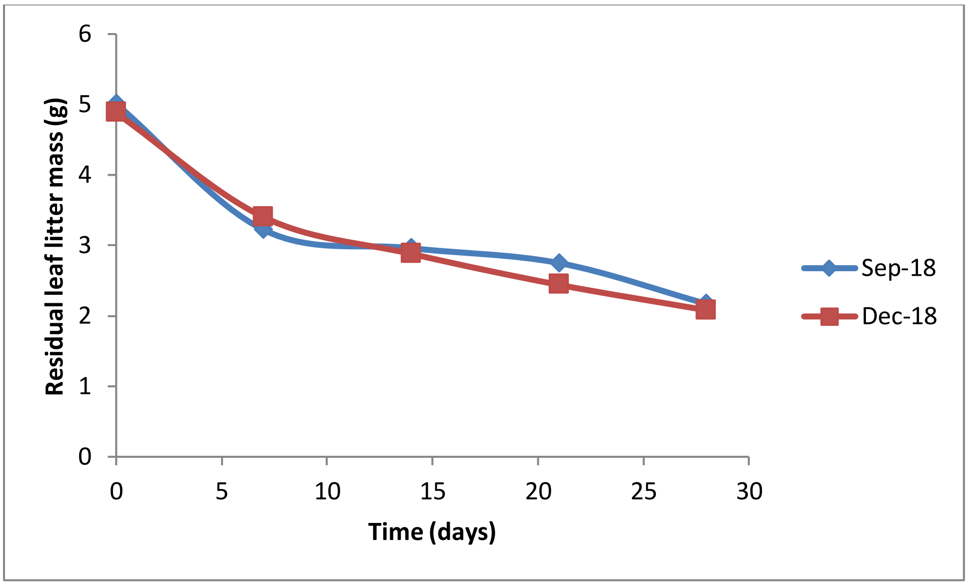


**Figure 4: Residual mass loss over the study period in September and December 2018**

The results showed that the mean leaf litter decomposition rate was highest in all the sites for all three plant species at day 7 (Table 1). The rate of leaf litter decomposition was observed to decline on the 14th and 28th day but increased on the 21st day generally for most of the plant species an observation sites. The exception was at the immediate upstream site where the rate of leaf litter decomposition for *Lantana camara* and

*Psidium guajava* declined at day 21 and slightly picked at day 28. The decomposition rate picked at day 21 and declined towards day 28 for *Eucalyptus grandis*.

At day 7, for *Lantana camara,* the highest mean (0.31± 0.05 g/day) rate was recorded at the immediate upstream site while the least (0.20 ± 0.02 g/day) leaf litter decomposition rate was recorded at the further upstream site (table 1). For *Psidium guajava* the highest leaf litter decomposition rate was recorded at the immediate upstream site (0.25 ± 0.02 g/day) while the least (0.03 ± 0.01 g/day) leaf litter decomposition rate was recorded at the immediate downstream site. For *Eucalyptus grandis*, the immediate upstream site recorded the highest rate of leaf litter decomposition (0.17 ± 0.03 g/day) while the immediate downstream site (0.07 ± 0.01 g/day) and the further downstream site (0.07± 0.02 g/day) had the least leaf litter decomposition rates.

At day 14, leaf litter decomposition was comparatively similar in all the sites (0.02 g/day) for *Lantana camara*. For *Psidium guajava* the highest leaf litter decomposition rate was recorded at the immediate upstream site (0.01 ± 0.0002 g/day) while the least decomposition rate (0.0007 ± 0.02 g/day) was recorded at the immediate downstream site (Table 1). For *Eucalyptus grandis,* the highest leaf litter decomposition rate was recorded at the immediate upstream site (0.007 ± 0.0001 g/day) while the least was at the weir area (0.002 ± 0.01 g/day) and the further downstream site (0.002 ± 0.001 g/day).

**Table 1:** Mean (±SE) leaf litter decomposition rates (gd$^{-1}$) of leaf litter at different sites after 7, 14, 21 and 28 days of incubation in Kapingazi River. Sites ordered from downstream to upstream.

| | *Lantana camara* | | | | *Psidium guajava* | | | | *Eucalyptus grandis* | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Site* | 7 | 14 | 21 | 28 | 7 | 14 | 21 | 28 | 7 | 14 | 21 | 28 |
| Further downstream | 0.25 | 0.02 | 0.09 | 0.06 | 0.048 | 0.0009 | 0.04 | 0.02 | 0.07 | 0.002 | 0.06 | 0.04 |
| | (0.02) | (0.0002) | (0.004) | (0.002) | (0.02) | (0.0005) | (0.0076) | (0.002) | (0.02) | (0.001) | (0.006) | (0.003) |
| Immediate downstream | 0.27 | 0.02 | 0.10 | 0.08 | 0.03 | 0.0007 | 0.02 | 0.005 | 0.07 | 0.005 | 0.07 | 0.05 |
| | (0.03) | (0.002) | (0.005) | (0.01) | (0.01) | (0.02) | (0.005) | (0.005) | (0.01) | (0.002) | (0.01) | (0.006) |
| Weir area | 0.28 | 0.02 | 0.15 | 0.08 | 0.051 | 0.001 | 0.03 | 0.01 | 0.1 | 0.002 | 0.06 | 0.05 |
| | (0.01) | (0.0002) | (0.02) | (0.01) | (0.02) | (0.03) | (0.01) | (0.05) | (0.03) | (0.01) | (0.007) | (0.007) |
| Immediate upstream | 0.31 | 0.02 | 0.11 | 0.12 | 0.25 | 0.01 | 0.04 | 0.05 | 0.17 | 0.007 | 0.06 | 0.05 |
| | (0.05) | (0.003) | (0.01) | (0.02) | (0.11) | (0.0002) | (0.01) | (0.003) | (0.03) | (0.0001) | (0.004) | (0.004) |
| Further upstream | 0.20 | 0.02 | 0.10 | 0.09 | 0.14 | 0.002 | 0.04 | 0.02 | 0.13 | 0.006 | 0.045 | 0.05 |
| | (0.02) | (0.0004) | (0.02) | (0.01) | (0.05) | (0.0009) | (0.007) | (0.005) | (0.02) | (0.001) | (0.006) | (0.01) |
| Mean | 0.260 | 0.020 | 0.110 | 0.050 | 0.100 | 0.003 | 0.034 | 0.021 | 0.108 | 0.0004 | 0.059 | 0.048 |

***p*=0.25**

At day 21, leaf litter decomposition was highest at the weir area (0.15 ± 0.02) while the least recorded rate of leaf litter decomposition was recorded at the further downstream site (0.09 ± 0.004) for *Lantana camara* (Table 1). For *Psidium guajava the highest leaf litter decomposition rate was at the further downstream site* (0.04 ± 0.0076 g/day), the immediate upstream site (0.04 ± 0.01 g/day) and the further upstream site (0.04 ± 0.007 g/day). The lowest leaf litter decomposition rate was recorded at the immediate downstream site (0.02 ± 0.005 g/day). For *Eucalyptus grandis,* the highest recorded rate of leaf litter decomposition was at the immediate downstream site (0.07 ± 0.01 g/day) while the least decomposition rate was at the further upstream site (0.045 ± 0.006 g/day).

At day 28, the highest rate of leaf litter decomposition rate was recorded at the immediate upstream site (0. 12 ± 0.02 g/day) while the least leaf litter decomposition rate was recorded at the further downstream site (0.06 ± 0.002 g/day) for *Lantana camara* (Table 1). For *Psidium guajava* the immediate upstream site recorded the highest rate of decomposition *(0.05* ± 0.003 g/day) while the least recorded rate of leaf litter decomposition was at the weir area (0.01 ± 0.05 g/day). For *Eucalyptus grandis* the least rate of decomposition was recorded at the further downstream site while the highest rate of decomposition was recorded at the rest of the sites (0.05 g).

Further statistical analysis showed that site (human activity) had a significant effect on leaf litter decomposition in River Kapingazi (df = 4; F-value = 27.74; *p*-value = 0.004) (Table 2). Plant species had a statistically significant effect on leaf litter decomposition rate at day 7 (df=2; sum square = 0.12; F-value = 160.9; *p*-value = 0.004). Both site and leaf species had a statistically significant effect on decomposition rate at day 7 (df = 8, F-value = 9.0, *p*-value = 0.004). Tukey contrasts showed that at day 7, leaf litter decomposition rates only differed significantly between immediate upstream site and the further downstream site and between the immediate upstream site and the immediate downstream site ($p < 0.05$) (Table 2).

The results also revealed that site had a significant effect on litter decomposition rates on day 14 of the study (df = 4, F-value = 2.77, $p = 0.09$) (Table 2). Plant species had a significant effect on leaf litter decomposition rate (df = 2, F-value = 228.5, $p=0.004$). Site and leaf species interaction had a significant effect on leaf litter decomposition rate (df = 8, F-value = 2.81, $p = 0.04$). Turkey contrasts showed that there was no significant difference in leaf litter decomposition rates among the sites at day 14 ($p<0.05$) (Table 2).

The findings also indicated that site had no significant effect on leaf litter decomposition rates on day 21 (df = 4, f-value = 1.99, p = 0.12) (Table 2). Leaf species had a significant effect on leaf litter decomposition rates on day 21 (df = 2, f-value = 102.61, p-value = 0.004). The site and leaf species interaction term showed that there was a significant effect on leaf litter decomposition rates (df = 8, f-value = 3.36, p = 0.02). Tukey contrasts showed that there was no significant difference in leaf litter decomposition rates between sites on day 21 (p-values > 0.05).

Site had a significant effect on leaf litter decomposition rates at day 28 (df = 4, F-value = 6.06, p-value = 0.004) (Table 2). Leaf species had a significant effect on leaf litter decomposition rates on day 28 (df = 2, F-value = 79.39, p-value = 0.004). Site and leaf species had no significant effect on leaf litter decomposition rates on day 28 (df = 8, F-value = 1.09, p-value = 0.39). Tukey contrasts showed that there was no significant difference in leaf litter decomposition rates among the sites at day 28 (p > 0.05).

**Table 2:** summary of T-values and p-values after Tukey comparisons for leaf litter decomposition rates for the different sites and times of observation

| | Day 7 | | Day 14 | | Day 21 | | Day 28 | |
|---|---|---|---|---|---|---|---|---|
| Site | t-value | p-value | t-value | p-value | t-value | p-value | t-value | p-value |
| Weir area versus further downstream | 1.68 | 0.46 | -0.14 | 1.0 | -0.14 | 1.0 | 0.64 | 0.97 |
| Further upstream versus further upstream | 2.77 | 0.07 | -1.98 | 0.30 | -1.98 | 0.30 | 0.6 | 0.97 |
| Immediate downstream versus further downstream | 0.28 | 0.99 | 1.20 | 0.75 | 1.20 | 0.75 | 0.73 | 0.95 |
| Immediate upstream versus further downstream | 4.37 | 0.001 | 2.14 | 0.23 | 2.14 | 0.23 | 0.98 | 0.86 |
| Further upstream versus weir area | 1.09 | 0.81 | 2.12 | 0.24 | 2.12 | 0.24 | -0.03 | 1.0 |
| Immediate downstream versus weir area | -1.39 | 0.63 | 1.34 | 0.67 | 1.34 | 0.67 | 0.09 | 1.0 |
| Immediate upstream versus weir area | 2.69 | 0.08 | 2.28 | 0.18 | 2.28 | 0.18 | 0.34 | 0.99 |
| Immediate downstream versus further upstream | -2.48 | 0.12 | -0.78 | 0.94 | -0.78 | 0.94 | 0.12 | 1.0 |
| Immediate upstream versus further upstream | 1.60 | 0.50 | 0.16 | 1.0 | 0.16 | 1.0 | 0.38 | 0.99 |
| Immediate upstream versus immediate downstream | 4.09 | 0.003 | 0.94 | 0.88 | 0.94 | 0.88 | 0.26 | 0.99 |

### 4.1.2 Effect of Site on Leaf Litter Decomposition

The decomposition rates (g/day) was computed as the seasonal decomposition rate (SDR) for the month and the wekly decomposition rate (WDR) over the period of observation (Table 3). The findings showed that the overall leaf litter decompositi was relatively the same in Sepetember 2018 (0.101 g/day) and December 2018 (0.100 g/day). It was observed that site significantly affected seasonal decomposition rate (SDR) in September 2018 but not in December 2018 at $P < 0.05$. In the dry month of September, the immediate downstream site results had significantly higher leaf decomposition rates (0.126 g/day) compared to the weir area (0.085 g/day). The other sites had comparable effects on the decomposition rates.

It was observed that the leaf litter decomposition rates were highest in the first week in September 2018 (0.256 g/day) and December 2018 (0.213 g/day) before reducing in the subsequent weeks though without common trend (Table 3). The lowest leaf litter decomposition rates were recorded during the weeks of 14 to 21 days in September 2018 (0.300 g/day) and 7 to 14 as well as 14 to 21 days in December 2018 (0.053 g/day), respectively. The findings showed that site had significant effects ($P<0.05$) on leaf litter decomposition only in the week of 14 to 21 days in the month of September 2018 but had no effect in all other periods. The leaf litter decomposition rate in the immediate downstream site differred significantly from that of the firther downstream site. The other sites caused similar effects on the decomposition of the plant species. In December 2018, site had no effect on the decomposition rates at any one period and the rates ranged from 0.085 to 0.106 g/day under the further downstream and immediate upstream sites, respectively.

**Table 3:** Weekly decomposition rates (g/day) of leaf litter as affcted by site and plant species **Variables of decomposition (g/day)**

| | | **Monthly** | **Weekly decomposition rate (g/day)** | | | |
|---|---|---|---|---|---|---|
| **Season** | **Factor** | **(g/day)** | **(0-7)** | **(7-14)** | **(14-21)** | **(21-28)** |
| **Sept 2018** | Immediate downstream | 0.126a | 0.288a | 0.056a | 0.058a | 0.102a |
| | Further downstream | 0.104ab | 0.254a | 0.029a | 0.029ab | 0.102a |
| | Weir area | 0.085b | 0.277a | 0.022a | 0.038ab | 0.045a |
| | Immediate upstream | 0.090ab | 0.203a | 0.046a | 0.042ab | 0.071a |
| | Further upstream | 0.102ab | 0.260a | 0.075a | 0.018b | 0.091a |
| | Lsd | 0.0364 | 0.1165 | 0.1013 | 0.0708 | 0.0634 |
| | *Lantana camara* | 0.150a | 0.412a | 0.037a | 0.037a | 0.113a |
| | *Eucalyptus glandis* | 0.070b | 0.182b | 0.032a | 0.004a | 0.064a |
| | *Psidium guajava* | 0.084b | 0.176b | 0.042a | 0.048a | 0.071a |
| | Lsd | 0.0282 | 0.0903 | 0.0785 | 0.0549 | 0.0491 |
| | Mean | 0.101 | 0.256 | 0.037 | 0.300 | 0.082 |
| **Dec 2018** | Immediate downstream | 0.107a | 0.236a | 0.071a | 0.082a | 0.039a |
| | Further downstream | 0.091a | 0.213a | 0.084a | 0.005a | 0.062a |
| | Weir area | 0.097a | 0.199a | 0.072a | 0.053a | 0.065a |
| | Immediate upstream | 0.106a | 0.214a | 0.089a | 0.075a | 0.048a |
| | Further upstream | 0.100a | 0.204a | 0.098a | 0.051a | 0.048a |
| | Lsd | 0.0232 | 0.0515 | 0.0394 | 0.1267 | 0.0533 |
| | *Lantana camara* | 0.153a | 0.320a | 0.132a | 0.066a | 0.095a |
| | *Eucalyptus glandis* | 0.064c | 0.122c | 0.052b | 0.052a | 0.038b |
| | *Psidium guajava* | 0.084b | 0.196b | 0.064b | 0.058a | 0.025b |
| | Lsd | 0.0179 | 0.0399 | 0.0305 | 0.0981 | 0.0413 |
| | Mean | 0.100 | 0.213 | 0.083 | 0.053 | 0.053 |

SDR – Combined decomposition rate; (0-7) – decomposition rate between 0 and 7 days

### 4.1.3 Effect of plant species on leaf litter decomposition rates

The plant species significantly ($P<0.05$) affected the SDR of the leaf litter in both the dry and wet months. In the dry month of September 2018, *Lantana camara* (0.150 g/day) decomposed fastest compared to the other two plant species which experienced statistically similar decomposition rates at $P>0.05$ (Table 3). In the wet month of December 2018, the rate of leaf litter decomposition for *Lantana camara* was higher

(0.153 g/day) than *Psydium guajava* (0.084 g/day) and *Eucalyptus glandis* (0.064 g/day), respectively.

The effects of plant species on weekly leaf litter decomposition differed significantly only the first week of observation in September 2018 but throughout the period of observation in December 2018 except in the week of 14 to 21 days after start of the experiment (Table 2). In all the weeks, *Lantana camara* had the highest leaf litter decomposition in compared to the *Eucalyptus glandis* and *Psidium guajava* in both September 2018 and December 2018. In December 2018 during the week covering 7 to 14 and 21 to 28 days, the leaf litter decomposition rates were statistically similar ($P<0.05$).

## 4.2 Physico-chemical variables

The water physico-chemical parameters measured at the study sites were: conductivity, water temperature, pH, total dissolved solids, velocity of the water, phosphates, nitrates and nitrites and Oxygen concentration (Table 3). The highest conductivity was recorded at the weir area (46.2 ± 3.6 µS/cm) while the lowest was recorded at the immediate upstream site (41.7 ± 1.4 µS/cm). The water temperature was highest at the weir area (21.5 ± 0.7 °C) while the least recorded water temperature was at the further downstream site (19.1 ± 3.3 °C). The highest pH was recorded at the immediate downstream site (7.8 ± 0.1) while the least was recorded at the further upstream site (6.9 ± 0.3). For the total dissolved solids, the highest concentration was recorded at the further downstream site (41.4 ± 25.1 mgL$^{-1}$) while the least concentration was recorded at the weir area (26.7 ± 2.5 mgL$^{-1}$ ). Phosphate concentration in the water was highest at the further downstream (1.8 ±1.8 mgL$^{-1}$) and the further upstream site (1.8 7 ± 0.5 mgL$^{-1}$) while the least recorded phosphate concentration was at the immediate downstream (1.3 ± 0.1 mgL$^{-1}$) and the weir area (1.3± 0.3 mgL$^{-1}$ ). Nitrite concentration of the water was highest at the immediate upstream site (2.4 ± 0.7 mgL$^{-1}$) while the least nitrite concentration was recorded at the immediate downstream (0.2 ± 0.08 mgL$^{-1}$). For the nitrates, the highest recorded concentration was at the weir area (3.6 ± 1.6 mgL$^{-1}$) while the least recorded nitrate concentration was at the immediate downstream site (2.3 ± 0.5 mgL$^{-1}$). Dissolved oxygen concentration was highest at the weir area (8.9 ± 0.3 $mgL^{-1}$) while the lowest concentration of oxygen was recorded at the further upstream site (7 ± 0.6 $mgL^{-1}$).

**Table 4:** Mean (±SE) physico-chemical parameters measured at the Kapingazi River sites. TDS refers to total dissolved solids

| Site | Conductivity (µS/cm) | Temperature (°C) | pH | TDS (mgL$^{-1}$) | Velocity (ms$^{-1}$) | Phosphates (mgL$^{-1}$) | Nitrites (mgL$^{-1}$) | Nitrates (mgL$^{-1}$) | Oxygen (mgL$^{-1}$) |
|---|---|---|---|---|---|---|---|---|---|
| Further downstream | 45.1 | 19.1 | 7.7 | 41.4 | 0.5 | 1.8 | 0.8 | 3.6 | 7.8 |
| | (7.2) | (3.3) | (0.3) | (25.1) | (0.2) | (1.8) | (0.8) | (1.6) | (0.3) |
| Immediate downstream | 43.8 | 20.1 | 7.8 | 37.6 | 0.7 | 1.3 | 0.2 | 2.3 | 8 |
| | (6.9) | (2.5) | (0.1) | (4.4) | (0.04) | (0.1) | (0.08) | (0.5) | (0.4) |
| Weir area | 46.2 | 21.5 | 7.3 | 26.7 | 0.2 | 1.3 | 0.3 | 3.4 | 8.9 |
| | (3.6) | (0.7) | (0.2) | (2.5) | (0.02) | (0.3) | (0.09) | (0.4) | (0.3) |
| Immediate upstream | 41.7 | 20.3 | 7.5 | 30.2 | 0.6 | 1.5 | 2.4 | 2.4 | 7.9 |
| | (1.4) | (0.8) | (0.3) | (1.8) | (0.03) | (0.3) | (0.7) | (0.3) | (0.4) |
| Further upstream | 42.7 | 20.9 | 6.9 | 32.6 | 0.5 | 1.8 | 0.4 | 3.3 | 7 |
| | (3.1) | (0.6) | (0.3) | (3.6) | (0.04) | (0.5) | (0.2) | (0.3) | (0.6) |

Most of the physico-chemical variables were correlated (Table 4). There was a positive correlation (r = 0.53) between dissolved Oxygen and total phosphorus. A negative correlation (r = -0.49) was observed between pH of the water and total Phosphorus concentration. There was a negative (r = -0.47) correlation between dissolved oxygen and total dissolved solids. It was also observed that there was a positive correlation between total phosphorus and nitrites (r = 0.61) and total phosphorus and phosphates (r = 0.77). The concentration of Phosphates and the velocity of the water depicted a negative correlation (r = -0.50).

**Table 5:** Correlation indices between physico-chemical parameters

| | Conductivity (µS/cm) | Nitrates (mgL$^{-1}$) | Nitrites (mgL$^{-1}$) | Oxygen (mgL$^{-1}$) | pH | Phosphates (mgL$^{-1}$) | TDS (mgL$^{-1}$) | Temperature ($^{o}$C) | Total Phosphorus (mgL$^{-1}$) | Velocity |
|---|---|---|---|---|---|---|---|---|---|---|
| Conductivity | 1 | -0.09 | -0.32 | 0.29 | -0.09 | -0.28 | -0.08 | 0.05 | -0.35 | -0.08 |
| Nitrates | -0.09 | 1 | 0.21 | -0.03 | -0.29 | 0.29 | 0.09 | -0.027 | 0.26 | -0.28 |
| Nitrites | -0.32 | 0.21 | 1 | -0.31 | -0.38 | 0.32 | 0.22 | 0.16 | 0.62 | -0.28 |
| Oxygen | 0.29 | -0.03 | -0.31 | 1 | 0.17 | -0.48 | -0.47 | 0.06 | -0.53 | 0.08 |
| pH | -0.09 | -0.29 | -0.38 | 0.17 | 1 | -0.49 | -0.36 | -0.27 | -0.48 | 0.17 |
| Phosphates | -0.28 | 0.29 | 0.32 | -0.48 | -0.50 | 1 | 0.45 | -0.01 | 0.77 | -0.05 |
| TDS | -0.08 | 0.092 | 0.22 | -0.47 | -0.36 | 0.45 | 1 | 0.09 | 0.46 | 0.12 |
| Temperature | 0.047 | -0.027 | 0.16 | 0.06 | -0.27 | -0.02 | 0.09 | 1 | 0.15 | -0.16 |
| Total Phosphorus | -0.35 | 0.26 | 0.61 | -0.53 | -0.49 | 0.77 | 0.46 | 0.15 | 1 | 0.05 |
| Velocity | -0.08 | -0.28 | -0.03 | -0.08 | 0.17 | -0.05 | 0.12 | -0.02 | -0.05 | 1 |

Substrate characterization and canopy cover were estimated at all the sites of study at River Kapingazi (Table 6). The substrate at the further downstream site was such that 60% was bedrock, 10% was boulders, 15% was gravel, 15% was sand and 5% was silt. 90% was a riffle, while 5% was a run and 5% was a pool. The site was surrounded by 40% canopy cover. The immediate downstream site had 20% bedrock, 20% boulders, 30% gravel, 10% sand and 20% silt. 55% of the immediate downstream site was described as a riffle, 40% as a run and 5% as a pool. The canopy cover was estimated at 5%. The weir area was described as a 100% pool with 100% sand. The immediate upstream site was described as 90% bedrock, 2% boulders, 2% sand, and 6% silt. The site was described as a 40% riffle, 40% run and 20% pool. The canopy cover was estimated at 30%. The further upstream site was characterized by 40% bedrock, 10% boulders, 15% gravel, 15% sand, and 30% silt. The site was described as 80% riffle, 5% run and 15% pool. The canopy cover at the further upstream site was estimated at 30%.

**Table 6:** Substrate characterization and canopy cover estimates (%) for the different sites studied at River Kapingazi

| | Substrate characterization | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Site | Bedrock | Boulders | Gravel | Sand | Silt | Riffle | Run | Pool | Canopy Cover |
| Further downstream | 60 | 10 | 15 | 15 | 5 | 90 | 5 | 5 | 40 |
| Immediate downstream | 20 | 20 | 30 | 10 | 20 | 55 | 40 | 5 | 5 |
| Weir area | 0 | 0 | 0 | 100 | 0 | 0 | 0 | 100 | 10 |
| Immediate upstream | 90 | 2 | 0 | 2 | 6 | 40 | 40 | 20 | 30 |
| Further upstream | 40 | 10 | 15 | 15 | 30 | 80 | 5 | 15 | 30 |

## 4.3 Macro-invertebrates

A total of 2606 invertebrates belonging to 22 taxa were found in the 135 litter bags analyzed (Table 7). Out of the total invertebrates, 1077 were associated with Lantana spp. litter, whereas 954 and 575 invertebrates were associated with Eucalyptus spp. and Guava spp. respectively. The Lantana spp. leaf litter were dominated (67.5 ± 422.02) by Chironomidae larvae at the immediate downstream site. Other dominant invertebrate taxa associated with *Lantana* spp. leaf litter included Ceratopogonidae, Ecnomidae, Hirudinae and Caenidae. The Eucalyptus sp. leaf litter was also dominated by Chironomidae larvae at the weir area (54.75 ± 18.90) individuals per bag. Other major taxa that were associated with Eucalyptus sp. leaf litter included Chironomidae, Ceratopogonidae, Baetidae, Ecnomidae, and Elmidae. The Guava leaf litter was also dominated (19 ± 7.62) by Chironomidae larvae at the weir area. Other major taxa included: Ceratopogonidae, Caenidae, Baetidae and Ecnomidae.

**Table 7: Mean abundance (± SE) Macro Invertebrates Identified at the Different Sites in River Kapingazi, Embu taxa**

| | Further downstream | Immediate downstream | Weir Area | Immediate Upstream | Further upstream | Further downstream | Immediate downstream | Weir Area | Immediate Upstream | Further upstream | Further downstream | Immediate downstream | Weir Area | Immediate Upstream | Further upstream |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *Lantana camara* | | | | | *Eucalyptus grandis* | | | | | *Psidium guajava* | | | | |
| Chironomidae | 19.25 (8.83) | 67.5 (42.3) | 39.5 (25.76) | 47.5 (23.7) | 25.75 (13.19) | 22.5 (6.85) | 46.75 (24.65) | 54.75 (18.9) | 20.75 (8.34) | 22.5 (6.86) | 11.5 (5.3) | 17 (5.46) | 19 (7.63) | 21.75 (9.54) | 17.75 (6.88) |
| Ecnomidae | 6.33 (5.06) | 3.75 (1.38) | 6.75 (2.14) | 5 (2.5) | 4.25 (2.53) | 0.5 (0.5) | 1.25 (0.75) | 2 (0.82) | 9.5 (9.17) | 0.5 (0.5) | 0.5 (0.5) | 0.5 (0.5) | 2.25 (1.3) | 3.25 (2.92) | 0.5 (0.29) |
| Baetidae | 1(0) | 3.5 (1.06) | 2 (0) | 1.5 (0.75) | 2.75 (1.25) | 4 (2.34) | 4.25 (2.98) | 0.75 (0.75) | 2.25 (1.03) | 4 (2.34) | 1 (1) | 3.25(0.85) | 1.5 (0.65) | 2.75 (1.49) | 3.5 (2.36) |
| Caenidae | 0 | 2.33 (0.76) | 4.07 (1.04) | 2.25 (1.12) | 3.5 (1.19) | 5 (2.34) | 2.25 (2.25) | 0.50 (0.29) | 1 (0.58) | 5 (2.34) | 2.25 (1.1) | 3 (1.29) | 2.25( 1.03) | 3.25 (1.7) | 2.25 (1.31) |
| Ceratopogonodae | 0 | 9 (0) | 4.75 (1.70) | 3 (1.5) | 2.5 (0.65) | 1.5 (0.96) | 5.75 (2.06) | 1.75 (0.63) | 2.25 (1.31) | 1.5 (0.96) | 2.5 (1.04) | 3.25 (2.59) | 3 (1.58) | 1 (1) | 0 |
| Perlidae | 0 | 1(0) | 0 | 0 | 0.25 (0.25) | 0 | 0 | 0 | 0 | 0 | 0 | 1.75 (1.75) | 0.25( 0.2)5) | 0 | 0.75 (0.75) |
| Oligochaeta | 1.67 (0.76) | 1 (0) | 1 (0) | 0 | 0.5 (0.5) | 0 | 0.29 (2.25) | 0.25 (0.25) | 0.25 (0.25) | 0 | 0 | 0 | 0 | 0 | 0 |
| Aeshnidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.5 (0.28) | 0 | 0 |
| Hirudinae | 0 | 3 (1.41) | 2 (0) | 0.75(0.375) | 1.25 (0.75) | 0.5 (0.29) | 1.5 (0.95) | 1.5 (0.5) | 0.5 (0.5) | 0.5 (0.29) | 1 (0.58) | 0.5 (0.28) | 0.5 (0.5) | 1.25 (0.63) | 0.5 (0.5) |
| Tubellaria | 0 | 0 | 1(0) | 0 | 0.5 (0.5) | 1.25 (1.25) | 0 | 1 (1) | 0 | 1.25(1.25 | 0.75 (1.5) | 0 | 1.25( 1.25) | 0 | 0 |
| Calopterygenidae | 0 | 0 | 0 | 0 | 0.5 (0.5) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Physicodidae | 0 | 0 | 5 (0.7) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.25 (0.25) | 0 | 0 | 0 |
| Heptagenidae | 0.5 (0.35) | 0 | 1(0) | 0 | 0.75 (0.75) | 0.5 (2.89) | 0 | 0 | 0 | 0.5 (2.89) | 0 | 0.25 (0.25) | 0 | 0.58 (0.29) | 0 |
| Corixidae | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Elmidae | 1 (0) | 0 | 2 (0) | 0 | 0 | 0.5 (0.5) | 0 | 0.5 (0.29) | 4.05 (4.25) | 0.5 (0.5) | 1.25 (2.5) | 0 | 0.25 (0.25) | 0.75 (0.75) | 0 |
| Libellulidae | 1 (0) | 0 | 1 (0) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.25( 0.25) | 0 | 0 |
| Hydrophylidae | 0 | 0 | 1 (0) | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0.5 (1) | 0.25 (0.25) | 0 | 0 | 0.25 (0.25) |
| Corydalidae | 0 | 1 (0) | 0 | 0 | 0 | 0 | 0.5 (0.5) | 0 | 0 | 0 | | | | | |
| Leptophrebridae | 0 | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.5 (0.5) |
| Amphipodae | 0 | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | | | | | |
| Athericidae | 0 | | 0 | 0.25 (0.12) | 0 | 0.5 (0.289) | 0 | 0 | 0.75 (0.25) | 0.5 (0.29) | 0.25 (0.25) | 0 | 0 | 0.75 (0.47) | 0 |
| Hydropysicidae | 0 | | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Sum total** | | | **1077** | | | | | **954** | | | | | **575** | | |

With regard to invertebrate functional feeding groups: shredders were absent whereas collector-gatherers, scrapers, predators and filtering-collectors were present (Table 8). The leaf litter was dominated by collector-gatherer and filtering collectors, whereas scrapers and predators had lower mean abundances. Shredders feed on the vascular plant tissues while collectors feed on detritus particles found floating on the water column. Scrapers on the other hand feed on algae attached to the riverbed while predators feed on live prey (other insects).

**Table 8:** Mean (±SE) macro invertebrate abundance of functional feeding groups of macro invertebrates observed at River Kapingazi, Embu

| | **Further downstream** | | | **Immediate downstream** | | | **Weir area** | | | **Immediate upstream** | | | **Further upstream** | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Taxa | Lantana sp | Eucalyptus sp. | Guava sp. | Lantana sp. | Eucalyptus sp. | Guava sp. | Lantana sp | Eucalyptus sp. | Guava sp. | Lantana sp | Eucalyptus sp. | Guava sp. | Lantana sp | Eucalyptus sp. | Guava sp. |
| Shredders | | | | | | | | | | | | | | | |
| Collector-gatheres | 3.71 (5.9) | 5 (3.47) | 4.25 (2.48) | 19.96 (15.94) | 11.05 (8.98) | 5.87 (3.77) | 7.7 (4.62) | 9.95 (8.96) | 6.37 (4.22) | 17.58 (14.95) | 6.06 (4.91) | 8.67 (6.57) | 6.55 (4.83) | 7.37 (4.16) | 6.83 (5.48) |
| Filtering-collectors | 6.33 (0) | 0.25 (0) | 0.5 (0) | 3.75 (0) | 1.25 (0) | 0.5 (0) | 6.75 (0) | 1.5 (0.5) | 2.25 (0) | 5 (0) | 9.5 (0) | 3.25 (0) | 4.25 (0) | 0.5 (0) | 0.5(0) |
| Scrapers | - | 1.5 (1.25) | 1.13 (0.13) | 3.5 (0) | 4.25 (0) | 1.75 (1.5) | 3 (1.53) | 0.63 (0.13) | 0.88 (0.63) | 1.5 (0) | 3.25 (1) | 1.33 (0.71) | 1.75 (1) | 2.25 (1.75) | 3.5 (0) |
| Predators | 1.22 (0.22) | 0.75 (0.5) | 0.58 (0.22) | 2.33 (0.67) | 0.87 (0.37) | 0.83 (0.46) | 2(0) | 1.25 (0) | 0.33 (0.08) | 0.5(0.25) | 0.63 (0.13) | 1.25 (0) | 0.67 (0.30) | 0.5 (0.03) | 0.5(028) |

## 4.4 Relationship between leaf litter decomposition rates and physico-chemical parameters in River Kapingazi, Embu

Correlation indices were calculated in R statistical package (R Core Team, 2018), to determine the relationship between leaf litter decomposition rates and physico-chemical parameters measured in the studied sites at River Kapingazi (Table 9). It was observed that at day 7, leaf litter decomposition rates were negatively correlated with conductivity (r = -0.41). There was negative correlation between temperature and leaf litter decomposition (r = -0.10). It was observed that pH had a positive correlation with leaf litter decomposition rates at day 7 (r = 0.17. Additionally, positive correlation between dissolved oxygen in the water and leaf litter decomposition rates (r = 0.28). Water velocity also showed a positive correlation with leaf litter decomposition rates (r = 0.21).

**Table 9:** Pearson's correlation (r) between leaf litter decomposition rates and physico-chemical parameters in River Kapingazi, Embu

| **Decomposition rates** | **Conductivity** | **Temperature** | **pH** | **Oxygen** | **Velocity** | **Nitrates** | **Nitrites** | **Tds** | **Phosphates** | **Total Phosphorus** |
|---|---|---|---|---|---|---|---|---|---|---|
| Decomposition_7 | -0.41 | -0.10 | 0.17 | 0.28 | 0.21 | -0.17 | -0.11 | -0.33 | -0.24 | -0.24 |
| Decomposition_14 | -0.26 | -0.17 | 0.21 | **0.46** | 0.09 | -0.07 | -0.22 | **-0.45** | -0.4 | -0.4 |
| Decomposition_21 | -0.39 | -0.15 | 0.16 | 0.37 | -0.08 | -0.08 | -0.18 | -0.34 | -0.25 | -0.25 |
| Decomposition_28 | -0.27 | -0.2 | 0.25 | 0.42 | 0.019 | 0.08 | -0.09 | **-0.53** | -0.23 | -0.23 |

Legend: Decomposition_7 -mean decomposition rates at day 7

Decomposition_14 -mean decomposition rates at day 14

Decomposition_21 -mean decomposition rates at day 21

Decomposition_28 - mean decomposition rates at day 28

Nitrates and nitrites had a negative correlation with leaf litter decomposition rates at day 7, being (r = -0.17) and (r = -0.11) respectively. The concentration of total dissolved solids in the water was found to have a weak downhill relationship (r = -0.33) with leaf litter decomposition rates at day 7. Both Phosphates and Total Phosphorus concentration had a weak negative linear relationship with leaf litter decomposition rates at day 7, being (r = -0.24). At day 14, there was weak negative linear relationship between conductivity and leaf litter decomposition rates (r = -0.26) (Table 9). Similarly, temperature had a weak negative linear relationship (r = -0.17) with leaf litter decomposition rates. A weak positive linear relationship (r = 0.21) between pH and decomposition rates at day 14 was observed.

The relationship between dissolved Oxygen concentration and leaf litter decomposition rates at day 14 was a moderate positive one (r = 0.46). Contrastingly, a weak positive linear relationship was observed between leaf litter decomposition rates and water velocity at day 14 (r = 0.09). The concentration of nitrates in the water and leaf litter decomposition rate had a weak negative linear relationship (r = -0.07).

Similarly, nitrite concentration and leaf litter decomposition rates showed a weak negative linear relationship (r = -0.22). Total dissolved solids and leaf litter decomposition rates at day 14 showed a moderate negative linear relationship (r = -0.45). There was moderately negative linear relationship between Phosphate and Total Phosphorus concentration and leaf litter decomposition rates at day 14 (r = -0.4).

At day 21, there was a weak linear relationship (r = -0.39) between conductivity and leaf litter decomposition rates (Table 6). Similarly, there was a weak negative (r = -0.15) linear relationship between water temperature and leaf litter decomposition rates at day 21. Contrastingly, there was a positive but weak linear relationship between pH and leaf litter decomposition rates (r = 0.16). Similarly, dissolved Oxygen and leaf litter decomposition rates showed a weak positive linear relationship (r = 0.37). On the other hand, the water velocity and nitrate concentration at the study sites both showed a weak negative linear relationship with the leaf litter decomposition rates at day 21 (r = -0.08).

The concentration of nitrites also showed a weak negative correlation index with leaf litter decomposition rates at day 21 (r = -0.18) (Table 6). Similarly, the concentration of total dissolved solids in the water at the study sites had a weak negative correlation index (r = -0.34) with leaf litter decomposition rates at day 21. Lastly, the concentration of Phosphates and Total Phosphorus had a weak negative linear relationship (r = -0.25) with leaf litter decomposition rates.

At day 28, conductivity had a weak negative linear relationship (r = -0.27) with leaf litter decomposition rates (Table 9). Similarly, temperature had a weak negative correlation (r = -0.2) with leaf litter decomposition rates at day 28. Contrastingly, pH had a weak positive linear relationship (r = 0.25) with leaf litter decomposition rates at day 28.

Dissolved Oxygen correlated moderately positive (r = 0.42) with leaf litter decomposition rates (Table 9). On the other hand, velocity had a weak positive linear relationship (r = 0.019) with leaf litter decomposition rates at day 28. Nitrates also had a weak positive linear relationship (r = 0.08) with leaf litter decomposition rates. In contrast, nitrite concentration had a weak negative linear relationship (r = -0.09) with leaf litter decomposition rates.

Total dissolved solids had a moderate negative linear relationship (r = -0.53) with leaf litter decomposition rates. Besides, the concentration of Phosphates and Total Phosphorus had a weak negative linear relationship (r = -0.23) with leaf litter decomposition rates.

# CHAPTER FIVE
# DISCUSSION, CONCLUSION AND RECOMMENDATIONS

## 5.1 Litter decomposition

Leaf litter breakdown rate was highest for the first 14 days of the experiment. The possible explanation for this is that this is the period which coincides with liberation phase for soluble compounds in foliar decomposition (Gómez *et al.*, 2017). This observation corroborates the work of (Ontoya & Igueroa, 2017) who found that both native (*Blepharocalyx cruckshanksii* and *Myrceugenia exsucca*) and exotic (*Pinus radiata* and *Eucalyptus globulus*) plants that they studied experienced the greatest loss of mass during the first 15 days after installation of leaf packs in the stream.

The rate of litter breakdown was significantly higher ($p < 0.05$) at the immediate upstream site in comparison to both the immediate downstream site and the further downstream site. This observation shows that the weir influenced leaf litter decomposition. This observation is in line with other researchers' work for example, Martínez *et al.*, (2017) and Mollá *et al* (2017) who demonstrated that leaf litter decomposition decreased downstream of sites whose main human activity was a small weir or reservoir. These authors attributed the decrease in leaf litter decomposition rates to low abundance and biomass of shredders caused by altered in-stream habitat conditions and riparian vegetation downstream of the point of flow regulation.

Whether or not this was the case in the current study requires further assessment since no shredders were found to be associated with the leaf litter. This finding however contrasts with that of Mbaka and Schäfer (2016) who studied the effects of small weirs and reservoirs on leaf litter breakdown and found that the sites immediate upstream of the point of regulation had significantly lower leaf litter decomposition rates as compared to the sites downstream of the point of regulation.

In the current study, *Lantana camara* had the fastest rate of breakdown, followed by *Eucalyptus grandis* while *Psidium guajava* had the slowest rate of breakdown. This means that *Lantana camara* was more readily utilized as a source of nutrients as compared to both *Eucalyptus grandis* and *Psidium guajava*. This may be explained by differences in litter quality (Amani *et al*., 2019). For example, high lignin content in *Psidium guajava* as reported by Camarena-Tello *et al* (2015) (32.65%) would have caused its low breakdown rate. *Lantana camara* reportedly has low lignin content (11%) (Gachengo *et al*., 2004) and this would explain its fast rate of breakdown as compared to the rest of the plant species studied. Lignin content for Eucalyptus *spp*. is reported to be (28.9 % – 29.6 %) (Vaz *et al*., 2019).

Decomposer communities also play a role in determining the rate of leaf litter decomposition. However, in the current study, it appears that decomposer shredders did not play a role in leaf litter decomposition since shredders were lacking. Nevertheless, the presence of other macro invertebrate groups may infer sporadic feeding on the leaves influencing their breakdown. Hence, the breakdown rate order (*Lantana camara* > *Eucalyptus grandis* > *Psidium guajava*) may be explained by the order of predominance of macro invertebrates in litter bags (*Lantana camara* > *Eucalyptus grandis* > *Psidium guajava*).

The abundance of macro invertebrates associated with the leaves could also explain the differences in their breakdown rates. The order of abundance was such that *Lantana camara* attracted the most macro invertebrates while *Psidium guajava* attracted the east number of macro invertebrates. Although the observed macro invertebrates are not specialized for fragmentation of organic matter, there is evidence that they can sometimes feed directly on litter (Merrit *et al*., 2017; Medeiros *et al*., 2018; Tsisiche *et al*., 2019). More specifically, Chironomids were the most dominant group associated with the leaf detritus. Their order of predominance (*Lantana camara* > *Eucalyptus grandis* > *Psidium guajava*) could explain the differences in breakdown rates of the leaves.

Further, microbial decomposer communities, through microbial conditioning, may have attributed to leaf litter decomposition rates differences in the current study. That *Lantana camara* had the highest number of macro invertebrates associated with it implies higher microbial conditioning. This means that microbial processing determined differences in breakdown rates of the leaves. Besides, the lack of shredders may imply that fungi played a bigger role in decomposition and overcompensated for the lack of shredders in this study.

Additionally, differences in leaf toughness may have also attributed to the differences in breakdown rates of the litters. This is because thinner cuticles are easier to breakdown and facilitate dissolution of hydrophilic compounds (Tonin *et al*., 2014). The cuticle of *Lantana camara* is thinner than that of either *Psidium guajava* or *Eucalyptus grandis* leaves. This may have caused the higher breakdown rates of *Lantana camara* as compared to *Psidium guajava* and *Eucalyptus grandis*. This finding corroborates with the case of Tenkiano *et al* (2018), who found that microbial decomposition dominated in both of their sites in Guinean savannah streams.

Leaf litter decomposition rates differed significantly between the two months of experimentation. This finding in line with that of other researchers who have found significant differences in seasons (Cowan & Anderson, 2019). For example, a study in Colombia assessed leaf litter decomposition rates' differences in two hydrological seasons (Rueda-Delgado et al., 2006). The findings revealed significantly higher decay rates in the rainy period (March to April) than in the drier period (September to November) of the study.

## 5.2 Relationship between leaf litter decomposition rates and water physico-chemical variables

All of the parameters of water at the study sites of River Kapingazi showed a correlation with leaf litter decomposition rates. Water conductivity gave a negative correlation with leaf litter decomposition rates throughout the study period. The weak correlation indicates that the relationship between conductivity and decomposition was a negligible one.

Dissolved Oxygen correlated positively with leaf litter decomposition at all the sites meaning the concentration of Oxygen favored the breakdown of litter. It has been established that dissolved Oxygen encourages harboring of a myriad of organisms in the water including fungi and other decomposer communities and consequently, promoting litter breakdown (Martins *et al*., 2015). Although decomposer macro invertebrates were not experienced in this study, it is possible that leaf litter decomposition was stimulated indirectly by microbial priming which then explains the positive correlation indices between leaf litter decomposition rates and dissolved oxygen.

The relationship between dissolved Oxygen and leaf litter decomposition was a weak one which shows that perhaps dissolved Oxygen did not play a big role in influencing breakdown rates in the current study. Besides, it has been suggested that Oxygen may not be important in the process of leaf litter decomposition in small streams such as River Kapingazi (Benfield *et al.*, 2017).

High Leaf litter decomposition rates have been shown to be a consequence of reduced leaf toughness due to microbial activity enhanced by high dissolved oxygen (Medeiros *et al.*, 2009). Whether or not dissolved Oxygen influenced leaf toughness through the mediation of fungi associated with leaf litter decomposition is not certain and requires further validation.

Temperature was negatively correlated with leaf litter decomposition rates, which indicate that an increase in the water temperature would cause a decrease in the leaf litter decomposition rates at River Kapingazi. However since the correlation was a weak one, it shows that the relationship between water temperature and decomposition rates was small.

The pH of water positively correlated with leaf litter decomposition rates at the studied sites at River Kapingazi. This result is in accordance with that of Thompson and Bärlocher, (1989) who studied stream sites whose range of pH was between 5 and 7. The pH of the water in the current study ranged between 6.9 and 7.7 and is therefore comparable with the study of the aforementioned authors. The pH range seems to have favored macro invertebrate communities influencing litter decomposition to a certain extent. However, since the correlation presented a weak relationship, pH cannot be the only variable influencing leaf litter breakdown at River Kapingazi.

Water velocity showed a positive correlation with leaf litter decomposition rates at the study sites in River Kapingazi only for day 7 and day 14. This observation can be attributed to the influence of abrasion on mass loss due to initial leaching losses. Water velocity showed a weak negative correlation with leaf litter decomposition rates at day 21 and at day 28. This shows that the influence of abrasion on mass loss was no longer significant after day 14 and that other factors were more important in determining the mass loss based decomposition.

The positive correlation between leaf litter decomposition rates at day 7 and 14, and water velocity could be explained by the fact that water velocity accelerates breakdown up to a point beyond which it ceases to be a limiting fact and other factors such as the litter quality and presence of dissolved solids, comes into play. In this case, it could be that increasing water flow rate increased the breakdown of the leaf litter at day 7 and 14 since this period ideally coincides with the stage of microbial conditioning where leaf toughness is reduced by hyphomycetes producing degrading enzymes thereby aiding leaf litter breakdown (Quinn *et al.*, 2000). This result is in accordance with that of Bastios *et al* (2019) who found a positive correlation between water velocity and *Ginkgo biloba L.* leaf litter decomposition rates at a headwater stream in Eastern Spain.

Water velocity is a major factor in determining both stream metabolism and nutrient cycling (Bastios *et al.*, 2019). This was not true past day 14 where there was a negative correlation between water velocity and leaf litter decomposition rates. Besides, the linear correlations at day 7 and day 14 depicted weak relationships. It goes to show that perhaps water velocity was not the overriding factor influencing breakdown in this study. Perhaps, factors such as presence of recalcitrant material in the leaf species played a more important role in the breakdown of leaf litter. Perhaps a more conclusive perspective could have been reached if the decomposition of the leaves was observed for longer.

All of the nutrients considered in the current study ($NO_3^-$, $NO_2^-$, $PO_4^{3-}$, TP) all portrayed a negative correlation with leaf litter decomposition rates. This result contrasts with the findings of previous studies in the review by Ferreira *et al* (2015) that found that additional nutrient concentrations in the streams led to acceleration of litter decomposition.

The discrepancy in the current study could be attributed to the fact that the studied sites were only moderately affected by human activities with the potential to increase nutrient load. For instance, farming around the areas was done on small scale therefore nutrient enrichment is assumed to be minimal. The fact that the linear relationship between leaf litter decomposition rates was weak shows that perhaps any nutrient enrichment caused by the activities around the study sites was probably insignificant. This is further supported by the range of macro invertebrates observed which point to moderate pollution levels of the water.

This is presumably the case for the current study where nutrient additions had a negative effect on leaf litter decomposition rates. Further, no detritivores were found in the current study. It is therefore assumed that other factors other than the nutrients were of greater importance in the decomposition of the leaf litters at River Kapingazi.

### 5.3 Macro invertebrates

It was observed that macro invertebrates from the Chironomidae family dominated in all of the sites. This is in accordance with other studies who have found predominance of chironomids (Rosser & Pearson, 2018; Shabani *et al*., 2019). Chironomidae larvae have a diverse feeding system in that some are grouped as shredders while others are grouped as gatherers and collectors. It is for this reason that they were found to be the most dominant invertebrate community associated with breakdown of matter. Moreover, they are more adaptable to most waters and usually account for most of the macro invertebrates in aquatic ecosystems. The other reason as to why they are the dominant macro invertebrate community is that leaf litter serves not only as a source of food for them but also as a habitat (Ágoston-Szabó *et al*., 2016; Fierro *et al*., 2017).

That chironomids dominated all the sites shows that most of the macro invertebrates found in the experimental litter bags utilized litter as refuge and not necessarily contributing to decomposition. The dominance of the Chironomidae larvae in this study indicates ecological disturbance caused by human activities around the river as demonstrated in similar studies (Kaboré *et al*., 2016; Sarker *et al*., 2018

It is largely observed that Chironomids account for the larger portion of aquatic communities especially in naturally poor waters (Serra *et al*., 2017; Molineri *et al*., 2020). Their preeminence in all of the study sites in the current study indicates moderate levels of human disturbance in the studied area in River Kapingazi. Additionally their dominance may indicate their importance in decomposition since leaves were the main substrate in this study. Chironomids cause leaf decomposition by scraping, mining and shredding.

The observation that the macro invertebrates dominated the weir area and the area immediate downstream of the weir shows that these two study sites were a good habitat for the macro invertebrates due to high sediment load. One possible explanation for the high relative abundance of macro invertebrates at the weir area is the habitat condition of the weir which was a pool. By virtue of its habitat conditions, the weir had the highest sediment load which is a favorable habitat for decomposer macro invertebrates and other organisms (Martins *et al.*, 2015). It could also be due to the high concentration of nutrients as the weir acts as a sink for materials from upstream. This is because nutrient concentration has been associated with increased numbers of aquatic macro invertebrates (Johnson *et al*., 2017).

For the site immediately downstream of the weir, high velocity of water could be the possible explanation for the recorded high abundance of macro invertebrates. This is because at high velocity, high oxygenation encourages harboring of a myriad of organisms some of which form part of the decomposer communities (Martins *et al.*, 2015).

With regard to functional feeding groups, it was observed that collector-gatherers were the most abundant followed by filtering-collectors and predators were the least while shredders were absent. This finding is in line with the work of Fu *et al* (2016) who found that collector-gatherers dominated human affected sites. The observed feeding groups in the study site could be due to the fact that the area of river studied was human-impacted. These conditions would have deterred the survival of shredder macro invertebrates and enabled collector-gatherers to thrive.

The dominance of collector-gatherers could also be a consequence of availability of fine particulate organic matter (Masese *et al.,* 2014; Mangadze *et al.,* 2019). That collector-gatherers and filtering-collectors dominated in the current study therefore shows that human activities have indeed altered structure of the aquatic macro invertebrates in River Kapingazi which has then altered ecosystem functioning.

Moreover, functional feeding groups are indicative of the nature of habitat (Martins *et al.,* 2015). Collector-gatherers and filtering-collectors are indicative of pools while scrapers and predators are indicative of rock and gravel substrate. This is the other probable reason as to why the collector-gatherers and filtering-collectors were dominant in the study as the sites were mainly characteristic of pools and rock and gravel substrates. The observation that the gatherer-collector group dominated the study sites also shows that the study area had an abundance of fine particulate organic matter which then translates to less or no shredders in the area. This observation is consistent with the river continuum concept that suggests that local unprocessed material are less important in head water streams as supported by the lack of shredders (Johnson *et al.,* 2017).

Nonetheless, the lack of shredders in the litterbags of the current study does not necessarily mean that the area lacks shredders. It is possible that the quality of the litter in the mesh bags was not palatable for the shredders given that the studied plant species were all exotic (see the case of Tsisiche *at al.,* 2019). It is also possible that there was abundant high quality feed in the stream that did not warrant the shredders aggregating in the litterbags, as suggested in the case for upper streams of Mara, in Kenya (Tsisiche *at al.,* 2019).

Additionally, a diverse guild of shredders is reported for the streams of Kenyan highlands during the rainy season. This could explain the lack of shredders in the current study as the study period covered relatively dry months. It is also possible that the microbes that colonized the studied leaves during priming stage were not preferable to shredders hence none were found in the litter bags. As well, the time of colonization may not have been enough for the leaves to reach the level of quality palatable for the shredder community at the study sites (Casotti *et al.,* 2019).

This study adds to the current body of knowledge that shows that tropical rivers, and in particular, the Kenyan highlands, are void of shredders (Dobson *et al.*, 2002).

## 5.4 Conclusion and Recommendations

### 5.4.1 Conclusions

a) The effect of site (human activity) on leaf litter decomposition was significant with the weir area and its immediate upstream and downstream sites causing comparatively higher decomposition rates than those further upstream and further downstream.
*b)* There effect of plant species on leaf litter decomposition was significantly different with *Lantana camara* exhibiting the greatest rates compared to *Eucalyptus grandis* and *Psidium guajava*.
c) There were significant correlations between the rates of leaf litter decomposition and the physico-chemical properties.

### 5.4.2 Recommendations

a) Impoundments and areas surrounding them should be considered carefully since they influence leaf litter decomposition and have the potential to alter riparian ecosystems
b) The local government and community should strive to maintain a diverse plant species along the riparian area of Kapingazi River so that varying leaf litter decomposition can be maintained for sustainable nutrient cycling so as to guarantee habitat quality.
c) The users of the riparian land along river Kapingazi should take advantage of the positive correlations between the leaf litter decomposition and the physico-chemical properties of the river to boost the habitat quality of the river.

### 5.4.3 Proposed Further Research

a) Future studies at River Kapingazi should consider the role of in-fauna in the leaf litter decomposition.
b) Future studies at River Kapingazi should also consider assessing the effect of bio-chemical composition plant species on decomposition rates.

# REFERENCES

Ágoston-Szabó, E., Schöll, K., Kiss, A., & Dinka, M. (2016). Mesh size and site effects on leaf litter decomposition in a side arm of the River Danube on the Gemenc floodplain (Danube-Dráva National Park, Hungary). *Hydrobiologia*, *774(1),* 5368.

Alvim, E., A., C., C., de Oliveira, M., A., Rezende, R., S., & Gonçalves, J., F. (2015). Small leaf breakdown in a Savannah headwater stream. *Limnologica -Ecology, and Management of Inland Waters, 51*, 131-138.

Amani, M., Graça, M., A., & Ferreira, V. (2019). Effects of elevated atmospheric CO2 concentration and temperature on litter decomposition in streams: A meta-analysis. *International Review of Hydrobiology*, *104(1-4)*, 14-25.

Andrade, C. M., Neres-Lima, V., & Moulton, T. P. (2017). Differentiating the roles of shrimp and aquatic insects in leaf processing in a Neotropical stream. *Marine and Freshwater Research*, *68*(9), 1695-1703.

Apha, A. (2005). WEF, 2005. *Standard Methods for the Examination of Water and Wastewater*, *21*, 258-259.

Aristi, I., Arroita, M., Larrañaga, A., Ponsatí, L., Sabater, S., Von, S., D., Acuña, V. (2014). Flow regulation by dams affects ecosystem metabolism in mediterranean rivers. *Freshwater Biology*, *59*, 1816–1829.

Atkinson, C., L., & Vaughn, C., C. (2015). Biogeochemical hotspots: temporal and spatial scaling of the impact of freshwater mussels on ecosystem function. *Freshwater Biology*, *60*(3), 563-574.

Bailey, R., C., Norris, R., H., & Reynoldson, T., B. (2001). Taxonomic resolution of benthic macroinvertebrate communities in bioassessments. *Journal of the North American Benthological Society*, *20*(2), 280-286.

Balana, B., B., Yatich, T., & Mäkelä, M. (2011). A conjoint analysis of landholder preferences for reward-based land-management contracts in Kapingazi watershed, Eastern Mount Kenya. *Journal of Environmental Management*, *92*, 2634-2646.

Bärlocher, F. (2016). Aquatic fungal ecology. *Fungal Ecology*, *100*(19), 1-4.

Bastias, E., Bolivar, M., Ribot, M., Peipoch, M., Thomas, S., S., Sabater, F., & Martí ,E. (2019). Spatial heterogeneity in water velocity drives leaf litter dynamics in streams. *Freshwater Biology*.

Benfield, E., F., Fritz, K., M., & Tiegs, S., D. (2017). Leaf-Litter Breakdown. *Methods in Stream Ecology: Volume 2: Ecosystem Function*, 71.

Benítez-Mora, A., & Camargo, J., A. (2014). Ecological responses of aquatic macrophytes and benthic macroinvertebrates to dams in the Henares River Basin (Central Spain). *Hydrobiologia*, *728*(1), 167-178.

Biasi, C., Tonin, A. M., Restello, R. M., & Hepp, L. U. (2013). The colonisation of leaf litter by Chironomidae (Diptera): the influence of chemical quality and exposure duration in a subtropical stream. *Limnologica*, *43*(6), 427-433.

Bracken, M. E., Oates, J. M., Badten, A. J., & Bernatchez, G. (2018). Predicting rates of consumer-mediated nutrient cycling by a diverse herbivore assemblage. *Marine Biology*, *165*(10), 165.

Boyero, L., Pearson, R., G., Hui, C., Gessner, M., O., Pérez, J., Alexandrou, M., A., Graça, M., A., Cardinale, B., .J., Albariño, R., J., Arunachalam, M., & Barmuta, L., A. (2016). Biotic and abiotic variables influencing plant litter breakdown in streams: a global study. In *Proc. R. Soc. B* (Vol. 283, No. 1829, p. 20152664). The Royal Society.

Bruder, A., Salis, R., K., McHugh, N., J., & Matthaei, C., D. (2016). Multiple-stressor effects on leaf litter decomposition and fungal decomposers in agricultural streams contrast between litter species. *Functional Ecology*, *30*(7), 1257-1266.

Callisto, M., Graça, M., Ferreira, V., & Goncalves Jr, J. F. (2015). Microbial colonization and litter decomposition in a Cerrado stream is limited by low dissolved nutrient concentration. *Limnetica*, *34*(2), 283-292.

Camarena-Tello, J., C., Rocha-Guzmán, N., E., Gallegos-Infante, J., A., González-Laredo, R., F., Pedraza-Bucio, F., E., López-Albarrán, P., Herrera-Bucio, R., & Rutiaga-Quiñones, J., G. (2015). Chemical composition of biomass generated in the P.guajava tree pruning. *EXCLI journal, 14*, 204.

Campbell, A., J., Carvalheiro, L., G., Maués, M., M., Jaffé, R., Giannini, T., C., Freitas, M., A., B., Coelho, B., W., T., & Menezes, C. (2018). Anthropogenic disturbance of tropical forests threatens pollination services to açaí palm in the Amazon river delta. *Journal of applied ecology*, *55*(4), 1725-1736.

Canhoto, C., & Graça, M., A. (1996). Decomposition of Eucalyptus globulus leaves and three native leaf species (Alnus glutinosa, Castanea sativa and Quercus faginea) in a Portuguese low order stream. *Hydrobiologia*, *333*(2), 79-85.

Canhoto, C., Calapez, R., Gonçalves, A., L., & Moreira-Santos, M. (2013). Effects of Eucalyptus leachates and oxygen on leaf-litter processing by fungi and stream invertebrates. *Freshwater Science*, *32*(2), 411-424.

Carvalho, F., Pascoal, C., Cássio, F., & Sousa, R. (2016). Direct and indirect effects of an invasive omnivore crayfish on leaf litter decomposition. *Science of the Total Environment*, *541*, 714-720.

Casas, J., J., Gessner, M., O., Lopez, D., & Descals, E. (2011). Leaf litter colonisation and breakdown in relation to stream typology: insights from Mediterranean low-order streams. *Freshwater Biology*, *56*(12), 2594-2608.

Casotti, C. G., Kiffer, W. P., Costa, L. C., Barbosa, P., & Moretti, M. S. (2019). The longer the conditioning, the better the quality? The effects of leaf conditioning time on aquatic hyphomycetes and performance of shredders in a tropical stream. *Aquatic Ecology*, *53*(2), 163-178.

Chiba, W., A., C., Cunha-Santino, M., B., & Bianchini, J., I. (2015). Differential aerobic decomposition between a native and exotic macrophytes of tropical reservoirs. *Brazilian Journal of Biology*, *75*(2), 501-502.

Connolly, N., M., Crossland, M., R., & Pearson, R., G. (2004). Effect of low dissolved oxygen on survival, emergence, and drift of tropical stream macroinvertebrates. *Journal of the North American Benthological Society*, *23*(2), 251-270.

Contreras, L. M., Fierro-Cabo, A., & Cintra-Buenrostro, C. E. (2017). Early drivers of black mangrove (Avicennia germinans) leaf litter decomposition in the water column. *Hydrobiologia*, *803*(1), 147-157.

Cowan, O. S., & Anderson, P. M. L. (2019). Litter decomposition variation across a degradation gradient and two seasons in a critically endangered vegetation type within the Fynbos biome, South Africa. *South African Journal of Botany*, *121*, 200–209. https://doi.org/10.1016/j.sajb.2018.11.002

Crook, D., A., Lowe, W., H., Allendorf, F., W., Erős, T., Finn, D., S., Gillanders, B., M., Hadwen, W., L., Harrod, C., Hermoso, V., Jennings, S., & Kilada, R., W. (2015). Human effects on ecological connectivity in aquatic ecosystems: integrating scientific approaches to support management and mitigation. *Science of the Total Environment*, *534*, 52-64.

Dabrowski, J., Oberholster, P., J., & Dabrowski, J., M. (2014). Water quality of Flag Boshielo Dam, Olifants River, South Africa: *Historical Trends and the Impact of Drought. Water SA*, *40*(2), 345-358.

Dobson, M., Mathooko, J., M., Ndegwa, F., K., & M'Erimba, C. (2004). Leaf litter processing rates in a Kenyan highland stream, the Njoro River. *Hydrobiologia*, *519*(1-3), 207-210.

Dodds, W., K., Perkin, J., S., & Gerken, J., E. (2013). Human impact on freshwater ecosystem services: A global perspective. *Environmental Science and TechnologyDodds, W. K., Perkin, J. S., & Gerken, J. E. (2013). Human Impact on Freshwater Ecosystem Services: A Global Perspective. Environmental Science and Technology, 47(16), 9061–9068.*

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

Duarte, S., Pascoal, C., Alves, A., Correia, A., & Cássio, F. (2010). Assessing the dynamic of microbial communities during leaf decomposition in a low-order stream by microscopic and molecular techniques. *Microbiological research*, *165*(5), 351-362.

Dunck, B., Lima-Fernandes, E., Cássio, F., Cunha, A., Rodrigues, L., & Pascoal, C. (2015). Responses of primary production, leaf litter decomposition and associated communities to stream eutrophication. *Environmental Pollution*, *202*, 32-40.

Eckert, S., Kiteme, B., Njuguna, E., & Zaehringer, J., G. (2017). Agricultural expansion and intensification in the foothills of Mount Kenya: A landscape perspective. *Remote sensing*, *9*(8), 784.

Feio, M., J., Reynoldson, T., B., & Graça, M., A. (2006). The influence of taxonomic level on the performance of a predictive model for water quality assessment. *Canadian Journal of Fisheries and Aquatic Sciences*, *63*(2), 367-376.

Fenoy, E., Casas, J., J., Díaz-López, M., Rubio, J., Guil-Guerrero, J., L., & Moyano-López, F., J. (2016). Temperature and substrate chemistry as major drivers of interregional variability of leaf microbial decomposition and cellulolytic activity in headwater streams. *FEMS microbiology ecology*, *92*(11).

Fernández, D., Voss, K., Bundschuh, M., Zubrod, J., P., & Schäfer, R., B. (2015). Effects of fungicides on decomposer communities and litter decomposition in vineyard streams. *Science of the Total Environment*, *533*, 40-48.

Ferreira, V., Castagneyrol, B., Koricheva, J., Gulis, V., Chauvet, E., & Graça, M., A. (2015). A meta-analysis of the effects of nutrient enrichment on litter decomposition in streams. *Biological Reviews*, *90*(3), 669-688.

Ferreira, V., Larrañaga, A., Gulis, V., Basaguren, A., Elosegi, A., Graça, M., A., S., & Pozo, J. (2015). Forest Ecology and Management The effects of eucalypt plantations on plant litter decomposition and macroinvertebrate communities in Iberian streams, *335*, 129–138.

Ferreira, V., Castela, J., Rosa, P., Tonin, A., M., Boyero, L., & Graça, M., A. (2016). Aquatic hyphomycetes, benthic macroinvertebrates and leaf litter decomposition in streams naturally differing in riparian vegetation. *Aquatic ecology*, *50*(4), 711-725.

Ferreira, V., & Guérold, F. (2017). Leaf litter decomposition as a bioassessment tool of acidification effects in streams: Evidence from a field study and meta-analysis. *Ecological Indicators*, *79*, 382-390.

Fierro, P., Bertrán, C., Tapia, J., Hauenstein, E., Peña-Cortés, F., Vergara, C., Cerna, C., & Vargas-Chacoff, L. (2017). Effects of local land-use on riparian vegetation, water quality, and the functional organization of macroinvertebrate assemblages. *Science of the Total Environment*, *609*, 724-734.

Follstad, S., J., J., Kominoski, J., S., Ardón, M., Dodds, W., K., Gessner, M., O., Griffiths, N., A., Hawkins, C., P., Johnson, S., L., Lecerf, A., LeRoy, C., J., & Manning, D., W. (2017). Global synthesis of the temperature sensitivity of leaf litter breakdown in streams and rivers. *Global change biology*, *23*(8), 3064-3075.

Fu, L., Jiang, Y., Ding, J., Liu, Q., Peng, Q., Z., & Kang, M., Y. (2016). Impacts of land use and environmental factors on macroinvertebrate functional feeding groups in the Dongjiang River basin, southeast China. *Journal of Freshwater Ecology*, *31*(1), 21-35.

Gachengo, C., N., Vanlauwe, B., Palm, C., A., & Cadisch, G. (2004). Chemical characterisation of a standard set of organic materials.

Gachimbi, L., N. (2002). Technical Report of Soil Survey and Sampling Results : Embu – Mbeere Districts , Kenya by Technical Report of Soil Survey and Sampling Results : Embu – Mbeere Districts , Kenya, 1–200.

Gallardo, B., Clavero, M., Sánchez, M., I., & Vilà, M. (2016). Global ecological impacts of invasive species in aquatic ecosystems. *Global change biology*, *22*(1), 151-163.

García-Palacios, P., McKie, B., G., Handa, I., T., Frainer, A., & Hättenschwiler, S. (2016). The importance of litter traits and decomposers for litter decomposition: a comparison of aquatic and terrestrial ecosystems within and across biomes. *Functional Ecology*, *30*(5), 819-829.

Gilvear, D., J., Spray, C., J., & Casas-mulet, R. (2013). River rehabilitation for the delivery of multiple ecosystem services at the river network scale. *Journal of Environmental Management*, *126*, 30–43.

Given, E., K., Haas, W., D., Dyer, J., E., Rocha, O., J. 2019. In-stream decomposition and macroinvertebrate community dynamics: a comparison between invasive Glossy Buckthorn and native Silky *Dogwood leaf litter. In; Environmental Science & Research Design*, 24.

Gleick, P., H., & Heberger, M. (2014). Water and conflict. In *The world's water* (pp. 159-171). Island Press, Washington, DC.

Godoy, O., Castro-Díez, P., Van, L., R., S., Cornelissen, J., H., & Valladares, F. (2010). Leaf litter traits of invasive species slow down decomposition compared to Spanish natives: a broad phylogenetic comparison. *Oecologia, 162* (3), 781-790.

Gómez, C., F., Correa, A., F., Díaz, M., E., Olguín, M., Encina, M., F., & Figueroa, R. (2017). Leaf litter decomposition from native and non-native species in a freshwater forested wetland of Chile.

Gholizadeh, M., & Heydarzadeh, M. (2019). Functional feeding groups of macroinvertebrates and their relationship with environmental parameters (case study: in Zarin-Gol River). *Iranian Journal of Fisheries Sciences*.

Graça, M. A. (2001). The role of invertebrates on leaf litter decomposition in streams– a review. *International Review of Hydrobiology: A Journal Covering all Aspects of Limnology and Marine Biology*, *86*(4-5), 383-393.

Graça, M., A., Ferreira, V., Canhoto, C., Encalada, A., C., Guerrero-Bolaño, F., Wantzen, K., M., & Boyero, L. (2015). A conceptual model of litter breakdown in low order streams. *International Review of Hydrobiology*, *100*(1), 1-12.

Griffiths, N., A., & Tiegs, S., D. (2016). Organic-matter decomposition along a temperature gradient in a forested headwater stream. *Freshwater Science*, *35*(2), 518-533.

Gulis, V., Ferreira, V., & Graça, M., A., S. (2006). Stimulation of leaf litter decomposition and associated fungi and invertebrates by moderate eutrophication: implications for stream assessment. *Freshwater Biology, 51*(9), 1655-1669.

Haddeland, I., Heinke, J., Biemans, H., Eisner, S., Flörke, M., Hanasaki, N., Konzmann, M., Ludwig, F., Masaki, Y., Schewe, J., & Stacke, T. (2014). Global water resources affected by human interventions and climate change. *Proceedings of the National Academy of Sciences*, *111*(9), 3251-3256.

Harner, M., J, Crenshaw, C., L., Abelho, M., Stursova, M., Shah, J., J., F., & Sinsabaugh, R., L. (2009). Decomposition of leaf litter from a native tree and an actinorhizal invasive across riparian habitats. *Ecological Applications, 19*(5), 1135-1146.

Hauer, F., & Lamberti, G. (Eds.). (2011). *Methods in stream ecology*. Academic Press.

Hauer, F., R., & Resh, V., H. (2017). Macroinvertebrates. In *Methods in Stream Ecology, Volume 1 (Third Edition)* (pp. 297-319).

Hoellein, T. J., Zarnoch, C. B., Bruesewitz, D. A., & DeMartini, J. (2017). Contributions of freshwater mussels (Unionidae) to nutrient cycling in an urban river: filtration, recycling, storage, and removal. *Biogeochemistry*, *135*(3), 307-324.

Hopper, G. W., Gido, K. B., Vaughn, C. C., Parr, T. B., Popejoy, T. G., Atkinson, C. L., & Gates, K. K. (2018). Biomass distribution of fishes and mussels mediates spatial and temporal heterogeneity in nutrient cycling in streams. *Oecologia*, *188*(4), 1133-1144.

Hubai, K., E., Kucserka, T., Karádi-Kovács, K., Vass, M., Kósa, Á., Honti, M., & Padisák, J. (2017). Decoupling shredder activity and physical abrasion in leaf litter decomposition process: experiments in the Torna-stream (Hungary) affected by red sludge spill. *Hydrobiologia*, *785*(1), 233-248.

Ibrahima, A., Ntonga, J., C., & Ze, A., D., M. (2016). Litter Decomposition and Nutrient Dynamics in Tropical Rainforests of Ebom, Southwestern Cameroon: Effects of Logging-Disturbed. *Global Journal of Botanical Science, 4,* 24-36.

Jacobs, S., R., Breuer, L., Butterbach-Bahl, K., Pelster, D., E., & Rufino, M., C. (2017). Land use affects total dissolved nitrogen and nitrate concentrations in tropical montane streams in Kenya. *Science of The Total Environment*, *603*, 519-532.

Johnson, R., K., Angeler, D., G., Hallstan, S., Sandin, L., & McKie, B., G. (2017). Decomposing multiple pressure effects on invertebrate assemblages of boreal streams. *Ecological indicators*, *77*, 293-303.

Kaboré, I., Moog, O., Alp, M., Guenda, W., Koblinger, T, Mano, K., Ouéda, A., Ouédraogo, R., Trauner, D., & Melcher, A., H. (2016). Using macroinvertebrates for ecosystem health assessment in semi-arid streams of Burkina Faso. *Hydrobiologia*, *766*(1), 57-74.

Kaushal, S., S., Likens, G., E., Jaworski, N., A., Pace, M., L., Sides, A., M., Seekell, D., Belt, K., T., Secor, D., H., & Wingate, R., L. (2010). Rising stream and river temperatures in the United States. *Frontiers in Ecology and the Environment*, *8*(9), 461-466.

Kennedy, K., T., M. (2016). *Exotic vs. native: global and urban investigations of leaf litter decay in streams* (Doctoral dissertation).

Kibena, J., Nhapi, I., & Gumindoga, W. (2014). Assessing the relationship between water quality parameters and changes in landuse patterns in the Upper Manyame River, Zimbabwe. *Physics and Chemistry of the Earth, Parts A/B/C, 67,* 153-163.

Kong, X., Wu, W., Tian, K., Jia, Y., Siddiq, A., Lin, H., & Tian, X. (2019). Promotive performance of shrimp Neocaridina denticulata on Typha angustifolia leaf litter decomposition. *Hydrobiologia*, *827*(1), 75-87.

Kuglerová L, García L, Pardo I, Mottiar Y & Richardson JS. (2017). Does leaf litter from invasive plants contribute the same support of a stream ecosystem function as native vegetation? *Ecosphere, 8*(4).

Leite-Rossi, L., A., Saito, V., S., Cunha-Santino, M., B., & Trivinho-Strixino, S. (2016). How does leaf litter chemistry influence its decomposition and colonization by shredder Chironomidae (Diptera) larvae in a tropical stream? *Hydrobiologia*, *771*(1), 119-130.

Landeira-Dabarca, A., Pérez, J., Graça, M., A., & Boyero, L. (2019) "Joint effects of temperature and litter quality on detritivore-mediated breakdown in streams." *Aquatic Sciences* 81 (1), 1.

Liu, G., Sun, J., Tian, K., Xiao, D., & Yuan, X. (2017). Long-term responses of leaf litter decomposition to temperature, litter quality and litter mixing in plateau wetlands. *Freshwater Biology*, *62*(1), 178-190.

Makkonen, M., Berg, M., P., Handa, I., T., Hättenschwiler, S., Van Ruijven, J., Van Bodegom, P., M., & Aerts, R. (2012). Highly consistent effects of plant litter identity and functional traits on decomposition across a latitudinal gradient. *Ecology letters*, *15*(9), 1033-1041.

Mangadze, T., Wasserman, R., J., Froneman, P., W., & Dalu, T. (2019). Macroinvertebrate functional feeding group alterations in response to habitat degradation of headwater Austral streams. *Science of the Total Environment*, *695*, 133910.

Martínez, A., Larrañaga, A., Pérez, J., Descals, E., & Pozo, J. (2014). Temperature affects leaf litter decomposition in low-order forest streams: field and microcosm approaches. *FEMS Microbiology Ecology*, *87*(1), 257-267.

Martínez, A., Larranaga, A., Pérez, J., Casado, C., Casas, J., J., González, J., M., Menéndez, M., Mollá, S., & Pozo, J. (2017). Climate modulates the magnitude of the effects of flow regulation on leaf-litter decomposition. *Aquatic Sciences*, *79* (3), 507-514.

Martins, R., T., Melo, A., S., Gonçalves, J., F., & Hamada, N. (2015). Leaf-litter breakdown in urban streams of Central Amazonia: direct and indirect effects of physical, chemical, and biological factors. *Freshwater Science*, *34*(2), 716–726.

Marwick, T., R., Tamooh, F., Ogwoka, B., Teodoru, C., Borges, A,, Darchambeau, F., & Bouillon, S. (2014). Dynamic seasonal nitrogen cycling in response to anthropogenic N loading in a tropical catchment, Athi–Galana–Sabaki River, Kenya. *Biogeosciences*, *11*(2), 443-460.

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

Masese, F., O. (2015). *Dynamics in organic matter processing, ecosystem metabolism and trophic sources for consumers in the Mara River, Kenya*. CRC Press/Balkema.

Mbaka, J., G., & Schäfer, R., B. (2016). Effect of small impoundments on leaf litter decomposition in streams. *River research and applications, 32*(5), 907-913.

Medeiros Cavalcanti júnior, M., Hepp, L. U., Molozzi, J., & de Brito Melo TROVÃO, D. M. (2018). Leaf traits of Brazilian semiarid species as regulatory factors for associated aquatic invertebrates. *Journal of Limnology*, *77*(3).

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

Mlambo, M., C., Paavola, R., Fritze, H., Louhi, P., & Muotka, T. (2019). Leaf litter decomposition and decomposer communities in streams affected by intensive forest biomass removal. *Ecological indicators*, *101*, 364-372.

M'Erimba, C., M., Mathooko, J., M., Karanja, H., T., & Mbaka, J., G. (2014). Monitoring water and habitat quality in six rivers draining the Mt. Kenya and Aberdare Catchments using Macroinvertebrates and Qualitative Habitat Scoring. *Egerton Journal of Science & Technology*, *14*.

Molineri, C., Tejerina, E., G., Torrejón, S., E., Pero, E., J., & Hankel, G., E. (2020). Indicative value of different taxonomic levels of Chironomidae for assessing the water quality. *Ecological Indicators*, *108*, 105703.

Monroy, S., Menéndez, M., Basaguren, A., Pérez, J., Elosegi, A., & Pozo, J. (2016). Drought and detritivores determine leaf litter decomposition in calcareous streams of the Ebro catchment (Spain). *Science of the Total Environment*, *573*, 1450-1459.

Mormul, R. P., Ahlgren, J., & Brönmark, C. (2018). Snails have stronger indirect positive effects on submerged macrophyte growth attributes than zooplankton. *Hydrobiologia*, *807*(1), 165-173.

Moulton, T., P., Magalhães-Fraga, S., A., P., Brito, E., F., & Barbosa, F., A. (2010). Macroconsumers are more important than specialist macroinvertebrate shredders in leaf processing in urban forest streams of Rio de Janeiro, Brazil. *Hydrobiologia*, *638*(January), 55–66.

Muehlbauer, J., D., LeRoy, C., J, Lovett, J., M., Flaccus, K., K., Vlieg, J., K., & Marks, J., C. (2009). Short-term responses of decomposers to flow restoration in Fossil Creek, Arizona,USA. *Hydrobiologia*, *618*(1), 35-45.

Noel, L., Bärlocher, F., Culp, J., M., & Seena, S. (2016). Nutrient enrichment and flow regulation impair structure and function of a large river as revealed by aquatic hyphomycete species richness, biomass, and decomposition rates. *Freshwater Science*, *35*(November 2015), 1148–1163.

Numbere, A., O., & Camilo, G.,. R. (2017). Mangrove leaf litter decomposition under mangrove forest stands with different levels of pollution in the Niger River Delta, Nigeria. *African journal of ecology*, *55*(2), 162-167.

Nyairo, W., N., Owuor, P., O., & Kengara, F., O. (2015). Effect of anthropogenic activities on the water quality of Amala and Nyangores tributaries of River Mara in Kenya. *Environmental monitoring and assessment*, *187*(11), 691.

Ontoya, F., R., E., N., & Igueroa, R., I., F. (2017). Leaf litter decomposition from native and non-native species in a freshwater forested wetland of Chile, *81*(1), 1–8.

Pascoal, C., & Cássio, F. (2004). Contribution of fungi and bacteria to leaf litter decomposition in a polluted river. *Applied and environmental microbiology*, *70*(9), 5266-5273.

Paudel, E., Dossa, G.,G, de Blécourt, M., Beckschäfer, P., Xu, J., & Harrison, R., D. (2015). Quantifying the factors affecting leaf litter decomposition across a tropical forest disturbance gradient. *Ecosphere*, *6*(12), 1-20.

Pouyat, R. V., Setälä, H., Szlavecz, K., Yesilonis, I. D., Cilliers, S., Hornung, E., ... & Whitlow, T. H. (2017). Introducing GLUSEEN: a new open access and experimental network in urban soil ecology. *Journal of Urban Ecology*, *3*(1), jux002.

Prieto, I., Almagro, M., Bastida, F., & Querejeta, J., I. (2019). Altered leaf litter quality exacerbates the negative impact of climate change on decomposition. *Journal of Ecology*, *107*(5), 2364-2382.

Protano, C., Zinnà, L., Giampaoli, S., Spica, V., R., Chiavarini, S., & Vitali, M. (2014). Heavy metal pollution and potential ecological risks in rivers: a case study from southern Italy. *Bulletin of environmental contamination and toxicology*, *92*(1), 75-80.

Raposeiro, P., M., Ferreira, V., Gea, G., & Gonçalves, V. (2018). Contribution of aquatic shredders to leaf litter decomposition in Atlantic island streams depends on shredder density and litter quality. *Marine and Freshwater Research*, *69*(9), 1432-1439.

R Core Team. (2018). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Rincón, J., & Santelloco, R. (2009). Aquatic fungi associated with decomposing Ficus sp. leaf litter in a neotropical stream. *Journal of the North American Benthological Society*, *28*(2), 416-425.

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

Rueda-Delgado, G., Wantzen, K. M., & Tolosa, M. B. (2006). Leaf-litter decomposition in an Amazonian floodplain stream: Effects of seasonal hydrological changes. *Journal of the North American Benthological Society*, *25*(1), 233–249. https://doi.org/10.1899/0887-3593(2006)25[233:LDIA AF] 2.0.CO;2

Salomão, V., P, Tonin, A., M, de Souza, R., R., Leite, G., F., M, Alvim, E., A., C., C,

Sanpera, C., I., S., I., S., Lecerf, A., & Chauvet, E. (2009). Leaf diversity influences in-stream litter decomposition through effects on shredders. *Freshwater Biology, 54*(8), 1671-1682.

Santonja, M., Rodríguez-Pérez, H., Le Bris, N., & Piscart, C. (2019). Leaf Nutrients and Macroinvertebrates Control Litter Mixing Effects on Decomposition in Temperate Streams. *Ecosystems*, 1-17.

Shabani, I., E., Liu, M., H., Yu, H., X., Muhigwa, J., & Geng, F., F. (2019). Benthic macroinvertebrate diversity and functional feeding groups in relation to physicochemical factors in sanjiang plain wetlands, northeast china. *Applied ecology and environmental research*, *17*(2), 3387-3402.

Seeney, A., Pattison, Z., Willby, N., J., Boon, P., J., & Bull, C., D. (2019). Stream invertebrate diversity reduces with invasion of river banks by non-native plants. *Freshwater biology*, *64*(3), 485-496.

Serra, S., R., Graça, M., A., Dolédec, S., & Feio, M., J. (2017). Chironomidae traits and life history strategies as indicators of anthropogenic disturbance. *Environmental monitoring and assessment*, *189*(7), 326.

Silva-Junior, E., F., Moulton, T, P., Boëchat, I., G., & Gücker, B. (2014). Leaf decomposition and ecosystem metabolism as functional indicators of land use impacts on tropical streams. *Ecological Indicators, 36*, 195-204.

Stoker, D., Falkner, A., J., Murray, K., M., Lang, A., K., Barnum, T., R., Hepinstall-Cymerman, J., Conroy, M., J., Cooper, R., J., & Pringle, C., M. (2017). Decomposition of terrestrial resource subsidies in headwater streams: Does consumer diversity matter? *Ecosphere*, *8*(6), e01868.

Strickland, M., S., Keiser, A., D., & Bradford, M., A. (2015). Climate history shapes contemporary leaf litter decomposition. *Biogeochemistry*, *122*(2-3), 165-174.

Tabucanon, A., S., Xue, W., & Fujino, T. (2019). Assessing alteration of leaf litter breakdown rate influenced by dam operation in Nakatsugawa River and Arakawa River, Central Japan. *Watershed Ecology and the Environment*, *1*, 1-9

Tachet, H., Richoux, P., Bournaud, M., & Usseglio-Polatera, P. (2003). Freshwater invertebrates: taxonomy, biology, ecology: CNRS Editions.

Taylor, B., R., & Chauvet, E., E. (2014). Relative influence of shredders and fungi on leaf litter decompo sition along a river altitudinal gradient. *Hydrobiologia*, *721*(1), 239-250.

Tenkiano, N., S., D., & Chauvet, E. (2018). Leaf litter decomposition in Guinean savannah streams. *Inland Waters*, *8*(4), 413-421.

Thompson, P., L., & Bärlocher, F. (1989). Effect of pH on leaf breakdown in streams and in the laboratory. *Journal of the North American Benthological Society*, *8*(3), 203-210.

Tonin, A., M., Hepp, L., U., & Gonçalves, J., F. (2018). Spatial variability of plant litter decomposition in stream networks: from litter bags to watersheds. *Ecosystems*, *21*(3), 567-581.

Tsisiche, A., M'merimba, C., M., & Mbaka, J., G. (2019). Effect of Land Use on Leaf Litter Decomposition in Upper Mara Streams, Kenya. *Egerton Journal of Science and Technology*, *16*(1-139).

Tumusiime, J., Tolo, C., U., Dusabe, M., C., & Albrecht, C. (2019). Reliability of the Tanzania river scoring system (TARISS) macroinvertebrate index of water quality: a case study of the river Mpanga system, Uganda. *Journal of Freshwater Ecology*, *34*(1), 541-557.

Tumwesigye, C., Yusuf, S., K., & Makanga, B. (2000). Structure and composition of benthic macroinvertebrates of a tropical forest stream, River Nyamweru, western Uganda. *African Journal of Ecology*, *38*(1), 72-77.

Uieda, V., S., & Carvalho, E., M. (2015). Experimental manipulation of leaf litter colonization by aquatic invertebrates in a third order tropical stream. *Brazilian journal of biology*, *75*(2), 405-413.

Vaz, A. A., Pelizari, G. P., Butturi-Gomes, D., & Smith, W. S. (2019). Leaf colonization by aquatic invertebrates in a tropical stream: are there differences between native species and Eucalyptus grandis (Hill ex Maiden) in the rainy and dry seasons?. *Biotemas*, *32*(3), 51-64.

Vivanco, L., & Austin, A., T. (2019). The importance of macro-and micro-nutrients over climate for leaf litter decomposition and nutrient release in Patagonian temperate forests. *Forest ecology and management*, *441*, 144-154. Vivanco, L., & Austin, A. T. (2019). The importance of macro-and micro-nutrients over climate for leaf litter decomposition and nutrient release in Patagonian temperate forests. *Forest ecology and management*, *441*, 144-154.

Von, S., D., Acuña, V., Aristi, I., Arroita, M., Basaguren, A., Bellin, A., Boyero, L., Butturini, A., Ginebreda, A., Kalogianni, E., & Larrañaga, A. (2017). River ecosystem processes: A synthesis of approaches, criteria of use and sensitivity to environmental stressors. *Science of the Total Environment*, *596*, 465-480.

Waite, I., R., Herlihy, A., T., Larsen, D., P., Urquhart, N., S., & Klemm, D., J. (2004). The effects of macroinvertebrate taxonomic resolution in large landscape bioassessments: an example from the Mid-Atlantic Highlands, USA. *Freshwater Biology*, *49*(4), 474-489.

Wantzen, K., M., Yule, C., M., Mathooko, J., M., & Pringle, C., M. (2008). Organic matter processing in tropical streams. In *Tropical stream ecology* (pp. 43-64).

Wootton, A., Pearson, R., G., & Boyero, L. (2019). Patterns of flow, leaf litter and shredder abundance in a tropical stream. *Hydrobiologia*, *826*(1), 353-365.

Wymore, A., S., Salpas, E., Casaburi, G., Liu, C., M., Price, L., B., Hungate, B., A., McDowell, W., H., & Marks, J., C. (2018). Effects of plant species on stream bacterial communities via leachate from leaf litter. *Hydrobiologia*, *807*(1), 131-144.

Yule, C.,, M., Gan, J., Y., Jinggut., T & Lee, K., V. (2015). Urbanization affects food webs and leaf-litter decomposition in a tropical stream in Malaysia. *Freshwater Science*, *34*(2), 702-715.

Zhang, M., Cheng, X., Geng, Q., Shi, Z., Luo, Y., & Xu, X. (2019). Leaf litter traits predominantly control litter decomposition in streams worldwide. *Global Ecology and Biogeography*, *28*(10), 1469-1486